



Performance of alternative harvest control rules for two Pacific groundfish stocks with uncertain natural mortality: Bias, robustness and trade-offs



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ABSTRACT

Harvest control rules (HCRs) are used in fisheries management to reduce fishing mortality as the level of perceived risk to the fish stock increases. This is typically done by adjusting fishing mortality rates based on estimated stock status relative to operational control points (OCPs). OCPs represent the stock status level at which management responses are taken. OCPs differ from biological reference points (BRPs), which represent biomass targets to be achieved, or low biomass thresholds to avoid. Both BRPs and OCPs can be based on theoretical quantities such as Maximum Sustainable Yield (MSY), Spawning Potential Ratio (SPR), or unfished spawning biomass (B_0). However, they can also be based on quantities such as the estimated average spawning biomass and fishing mortality during a productive period. Formal evaluation of the performance of HCRs that account for potential biases in estimated model parameters and stock status relative to OCPs can help managers and stakeholders select HCRs expected to provide acceptable outcomes and trade-offs. We use closed-loop simulation to evaluate the performance of five HCRs for two British Columbian groundfish stocks for which there is considerable uncertainty in underlying productivity: Hecate Strait Pacific Cod (*Gadus macrocephalus*) and Hecate Strait Rock Sole (*Lepidopsetta* spp.). Performance metrics representing ecological and economic fishery objectives are reported for two alternative productivity scenarios for each stock, including depensatory mortality for Pacific Cod, and alternative levels of natural mortality (M) for Rock Sole. We present an algorithm for calculating equilibrium M in the presence of density-dependence, and show general effects of uncertainty in M on reference point calculations. Mechanisms for differences in performance among alternative HCRs are explored, and we show that even when model parameters or OCPs are very biased, some HCRs can still produce desirable management outcomes. We show that trade-off considerations are important because differential sources of stock assessment bias between the two species, and between scenarios within a given species, meant that no single HCR performed consistently. We suggest that prospective evaluation of alternative harvest policies using closed-loop simulation could be conducted routinely on a stock-specific basis, and can facilitate choice of HCRs, with a focus on outcomes rather than uncertainty *per se*.

1. Introduction

Sustainable management of fisheries typically relies upon the definition of biological reference points (BRPs). BRPs define biomass targets to be achieved, and low biomass thresholds to be avoided with high probability (Sainsbury, 2008). A key BRP is the limit reference point (LRP), which is defined under international and Canadian policy as a threshold of stock biomass below which serious harm can occur to the stock (e.g., UNFSA, 1995; DFO, 2006a, 2009). Definitions of serious harm include slowly reversible or irreversible states, and are often interpreted in terms of recruitment overfishing (Mace and Sissenwine, 1993; Myers et al., 1994; Shelton and Rice, 2002), but can also be estimated empirically (Kronlund et al., 2018). While serious harm and

appropriate limit thresholds are difficult to quantify in practice (Kronlund et al., 2018), these are generally assumed to be related to the life history of the species. Target reference points also take into account socio-economic objectives and therefore should be developed collaboratively with stakeholders and managers (Punt et al., 2016; Hilborn, 2007, 2010). BRPs represent outcomes for quantifiable fishery objectives that are ideally achieved through application of a harvest control rule (HCR) that adjusts fishing mortality to achieve the objectives (Punt et al., 2008). Stock status thresholds that trigger management actions (e.g., reduction in fishing mortality or cessation of fishing) are sometimes called operational control points (OCPs) (Cox et al., 2013). Application of OCPs should result in achievement of fishery objectives (e.g., the LRP is avoided over a specified time-frame with high

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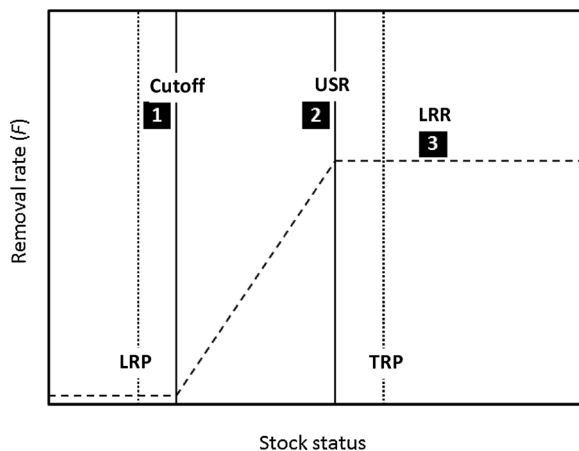


Fig. 1. Illustration of the PA Framework (DFO, 2006a, 2009) with solid vertical lines indicating two OCPs: (1) the “Cutoff”; and (2) the upper stock reference (USR). The removal rate (fishing mortality) in each of the zones is indicated by a dashed line, with its maximum defined as (3) the limit removal rate (LRR). The BRPs LRP and TRP are shown as vertical dotted lines, where the LRP is the threshold of “serious harm”. Note that in the PA Framework (DFO, 2006a, 2009), the Cutoff and LRP are not distinguished from each other.

probability). While BRPs are usually based on theoretical considerations, choice of OCPs may be driven by more practical considerations such as data-availability, understanding of stock assessment limitations, or specific fishery objectives (Cox et al., 2013). The progression from BRPs to OCPs in a HCR is illustrated in Cox et al. (2013, their Fig. 1).

Canada’s “Harvest Decision-Making Framework Incorporating the Precautionary Approach” policy (DFO, 2006a, 2009), hereafter called the PA Framework, requires that fishing mortality be adjusted in relation to two levels of stock status that delineate when fishing mortality is reduced or ceased (Fig. 1). As currently presented, the PA Framework does not distinguish between the lower OCP in the HCR, where fishing is ceased, and the LRP, a lower biomass threshold “below which productivity is sufficiently impaired to cause serious harm to the resource” (DFO, 2006a). As written, the PA Framework therefore implies that fishing can continue down to the LRP. In general, however, the lower OCP need not be coincident with the LRP. It may therefore be appropriate to re-name the HCR’s lower OCP as the “Cutoff” to distinguish it from the LRP. This is consistent with terminology used in Canada for management of Pacific Herring (*Clupea pallasii*), which distinguishes between the fishing cut-off and the LRP (Kronlund et al., 2018).

In the absence of stock-specific alternatives, the PA Framework recommends a provisional Cutoff (and LRP) at $0.4B_{MSY}$ and an upper stock reference (USR) at $0.8B_{MSY}$, where B_{MSY} is the expected equilibrium biomass when the stock is fished at F_{MSY} . The PA Framework also defines a limit removal rate (LRR) representing an upper limit to fishing mortality which should not be exceeded (Fig. 1), provisionally set at F_{MSY} . The provisional target reference point (TRP), set above the USR (Fig. 1), is B_{MSY} (DFO, 2009).

While limitations of MSY-based policies are well known in both single species (Larkin, 1977) and multispecies (Hilborn et al., 2004; Walters et al., 2005) contexts, the choice of B_{MSY} as a biomass target is based on sound biological theory, where F_{MSY} represents a precautionary limit to fishing mortality that can theoretically avert both growth and recruitment overfishing (Cook et al., 1997; Punt, 2000; Mace, 1994, 2001; Punt and Smith, 2001). Difficulties can arise with implementation of MSY-based harvest policies, however, since even small variation in data choices, prior assumptions, and the representation of structural dynamics in assessment models can produce substantial changes in estimates of F_{MSY} (Haltuch et al., 2008; Mangel et al., 2013; Punt and Szuwalski, 2012; Forrest et al., 2013). In the absence of reliable estimates of B_{MSY} and F_{MSY} , the use of proxies is

often recommended (Restrepo et al., 1998; Sainsbury, 2008). Recommended proxies may be based on the spawning potential ratio (SPR) (Clark, 1991, 2002; Mace, 1994), or on fractions of unfished biomass (B_0) (Restrepo et al., 1998; Smith et al., 2007; Sainsbury, 2008). Some jurisdictions also endorse “history-based” proxies based on estimated average spawning biomass and fishing mortality over a productive historical period (DFO, 2009).

SPR-based proxies for F_{MSY} are often recommended because, unlike F_{MSY} , they do not rely on estimates of stock-recruitment parameters (Clark, 1991). However, most proxies are still dependent on estimates of natural mortality (M), which can be subject to several sources of bias. For example, assumptions of time-invariant M can be violated due to external causes such as climate variation (reviewed by Drinkwater et al., 2010), predator-prey dynamics (e.g., Walters et al., 1986; Spencer and Collie, 1996; Mohn and Bowen, 1996; Swain and Benoît, 2015), or climate-mediated growth or predation effects (e.g., Benoît and Swain, 2008; Drinkwater et al., 2010). Adult mortality can also vary through time as a result of density-dependent processes, which may be linked with growth, predator-prey effects or both (e.g., Fournier, 1983; Deriso et al., 1986; Parma and Deriso, 1990; Gascoigne and Lipcius, 2004). Bias in estimates of M and other productivity parameters, such as the steepness of the stock-recruit relationship (h), can propagate through the entire fishery management cycle, affecting estimates of stock status and implementation of HCRs (Deroba and Bence, 2008; Haltuch et al., 2008; Punt et al., 2008; A’mar et al., 2009).

Pacific Cod (*Gadus macrocephalus*) and Rock Sole (*Lepidopsetta* spp.) in Hecate Strait, British Columbia (BC), are two commercially important fish stocks for which there is considerable uncertainty in productivity, particularly M (Forrest et al., 2015; Holt et al., 2016). Natural mortality for Pacific Cod has been estimated as high as $0.6–0.65\text{ y}^{-1}$ in some assessments (e.g., Fournier, 1983; Sinclair and Starr, 2005), although lower estimates in the range $0.35–0.4\text{ y}^{-1}$ have also been obtained (Sinclair et al., 2001; Forrest et al., 2015). Pacific Cod is a relatively short-lived, fast-growing species (maximum age < 12 years) that has shown particularly volatile dynamics in Hecate Strait over the last half-century (Haist and Fournier, 1997; Sinclair and Starr 2005; Forrest et al., 2015) (Fig. 2a). One hypothesis to explain the historical cyclic patterns in abundance (reviewed by Westrheim, 1996) is northward transport of larvae negatively impacting recruitment (Fournier, 1983; Tyler and Westrheim, 1986; Tyler and Crawford, 1991; Sinclair and Crawford, 2005), where positive sea level anomalies are considered to be an index of larval transport out of Hecate Strait, thus reducing recruitment strength. While this was the dominant hypothesis for many years (Westrheim, 1996), recent analyses have found the correlation between sea level anomalies and recruitment is no longer significant (R. Forrest, Pacific Biological Station, unpublished data). Other hypotheses include predator-prey cycles affecting M (Walters et al., 1986), and depensatory mortality of adult cod (Fournier, 1983). Mechanisms for depensation include increased predator effectiveness when prey schools become smaller, and reduced effects of predator feeding saturation at low prey-densities, i.e., predators take an increasing proportion of prey as the prey population size gets smaller (Liermann and Hilborn, 2001).

Depensatory adult mortality could explain the cycles seen in the historical catch time series for Pacific Cod (Fournier, 1983). Under a depensation hypothesis, M decreases as biomass increases, which could result in a faster rate of biomass growth at high biomass levels. If the fishing fleet responds to increased stock biomass by targeting it, this would reduce the biomass, which would lead to increased M , which would accelerate biomass decline. If the fleet switched to other species when biomass became low, this could result in some biomass recovery, reduced M , etc. As long as the stock did not fall below a biomass threshold from which it could recover (e.g., Gascoigne and Lipcius, 2004; Hutchings and Rangeley, 2011; Swain and Benoît, 2015), this fishing pattern could result in cycles. Depensatory population regulation is, however, difficult to detect (Liermann and Hilborn, 2001; but see Liermann and Hilborn, 1997; Thorson et al., 2015). Despite finding

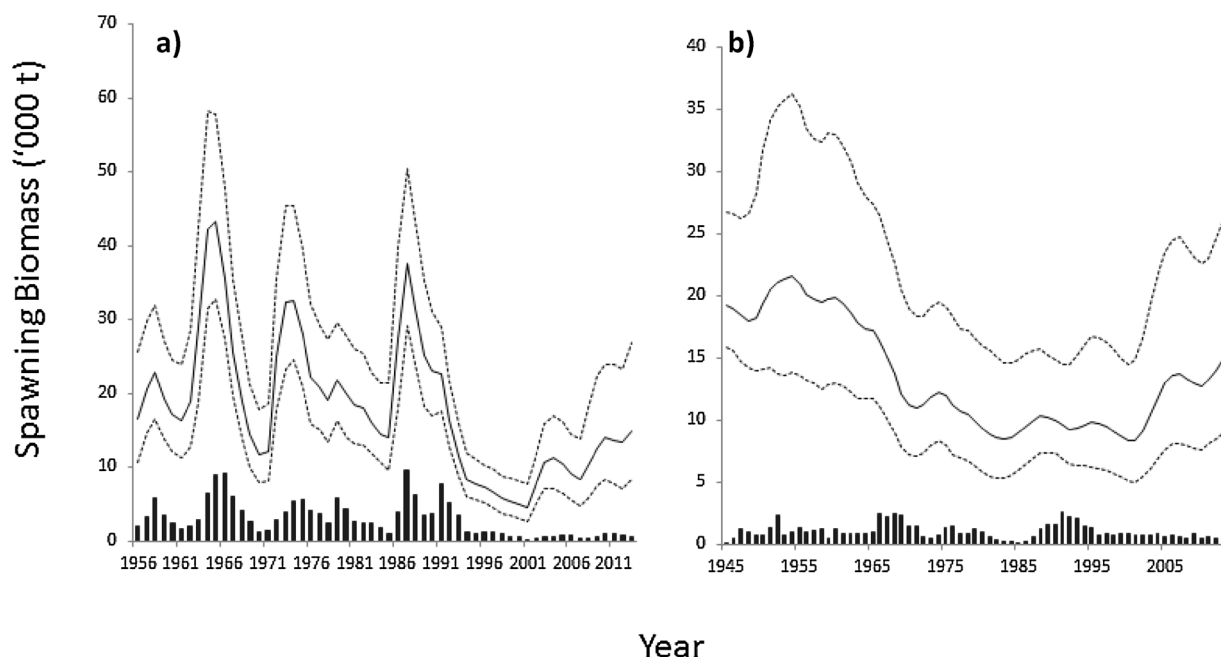


Fig. 2. Annual catches (bars) and posterior median spawning biomass estimates ('000 metric tons) (solid line) with 95% credibility interval (dashed lines) for: a) Hecate Strait Pacific Cod (Forrest et al., 2015); and b) Hecate Strait Rock Sole (Holt et al., 2016).

the depensatory mortality hypothesis to be consistent with observations of cyclic biomass in Pacific Cod, earlier studies (Fournier, 1983) and our own unpublished analyses have been unable to distinguish depensatory mortality from external environmental drivers as explanatory mechanisms for cyclic patterns in biomass. MSY-based OCPs are currently considered unreliable for this stock due to uncertainty in model parameters, particularly M (Forrest et al., 2015). Alternative history-based OCPs based on quantities calculated from reconstructions of biomass and fishing mortality trajectories (Sinclair and Starr, 2005) were used in the most recent assessment (Forrest et al., 2015), but these have never been tested in terms of their performance with respect to fishery objectives.

The Hecate Strait Rock Sole stock has shown relatively stable dynamics during its catch history, according to the most recent assessment (Holt et al., 2016) (Fig. 2b). MSY-based OCPs are also considered unreliable for this stock due to their strong dependence on informative prior probability distributions for two key productivity parameters, h and M (Holt et al., 2016). The 2016 BC stock assessment used a Bayesian statistical catch-age model that assumed a normal prior probability distribution for M with mean $= 0.2 \text{ y}^{-1}$ and $\text{CV} = 0.2$. The resulting maximum posterior density (MPD) estimate of M was close to 0.25 y^{-1} , which is higher than previously assumed or estimated for female Rock Sole. The 2012 stock assessment for Northern Rock Sole in the Bering Sea estimated $M = 0.19 \text{ y}^{-1}$ for females (Wilderbuer and Nichol, 2013), while the 2013 stock assessment for Rock Sole in the Gulf of Alaska and previous Rock Sole assessments in British Columbia have assumed a fixed $M = 0.2 \text{ y}^{-1}$ for females (A'mar et al., 2013; Fargo et al., 2000; DFO, 2006b). An alternative parameterization of the 2016 British Columbia stock assessment model in which the value of M was held fixed at 0.2 y^{-1} , resulted in lower estimates of productivity, including lower estimates of MSY and optimal harvest rate U_{MSY} , compared to when M was estimated (Holt et al., 2016). Concerns about the reliability of MSY-based OCPs for Rock Sole given uncertainty in key productivity parameters led to application of history-based OCPs (Holt et al., 2016), based on those from an earlier study (DFO, 2006b).

Authors of stock assessments for both Pacific Cod and Rock Sole recommended formal evaluation of the performance of history-based OCPs (Forrest et al., 2015; Holt et al., 2016). The Canadian PA Framework states that performance of HCRs and associated OCPs should be evaluated in terms of achieving fishery objectives (Kronlund et al., 2014). This should include demonstration of robustness against uncertainties that arise from process, observation and implementation errors, and competing structural assumptions about the dynamics of the fish population and fishery (FAO, 1995). Ideally, performance evaluation of HCRs should be done in a closed-loop simulation framework, but this is rarely done in practice (Punt, 2017). Instead, harvest advice is usually presented in decision tables, which are tabulated probabilities of exceeding stock status thresholds for a range of fixed catch levels. In this approach, the presence of various stock assessment uncertainties frequently results in the addition of several sensitivity cases with decision tables based on alternative “scenarios”, which are intended to meet obligations to communicate scientific uncertainty.

Decision tables are not however useful for selecting among alternative management procedures (e.g., choice of HCR) or for evaluating trade-offs associated with choice of management procedure, especially in the presence of structural uncertainty (Punt, 2017). Moreover, presentation of multiple versions of advice with no guidance for weighting alternatives can create a complex decision-making environment where decision-makers must integrate consideration of risk over dissimilar sensitivity cases (de la Mare, 1998; Cox et al., 2013). Closed-loop simulation provides a means for prospective evaluation of the performance of alternative management procedures, by testing them in a simulated environment, the “operating model”, where all underlying properties are known (e.g., de la Mare, 1996; Kirkwood, 1997; Butterworth and Punt, 1999; Smith et al., 1999; Punt and Hobday, 2009). Closed-loop simulation integrates performance-evaluation across alternative hypotheses for system dynamics within a common framework, thereby avoiding the situation where decision-makers are forced to choose a single preferred hypothesis (Smith, 1994; Walters, 1986; de la Mare, 1998). Closed-loop simulation has been particularly

useful for evaluation of the performance of alternative choices of OCPs in north Pacific fisheries (Punt, 2003; Cox and Kronlund, 2008; Cleary et al., 2010; Cox et al., 2013), notably in cases where underlying productivity is misspecified in the stock assessment model (Parma, 2002; Haltuch et al., 2008; Punt et al., 2008; Haltuch and Punt, 2011).

This study uses closed-loop simulation to evaluate robustness of alternative choices of OCPs for Hecate Strait Pacific Cod and Rock Sole. The recent stock assessments for these species highlight three problems in selection of OCPs for decision-making: (i) there can be a high likelihood of bias in the estimated OCPs due to uncertainty in productivity parameters; (ii) alternative choices of OCPs can result in different perceptions of stock status; and (iii) there is usually little to no evaluation of the performance of alternative OCPs with respect to achieving management objectives. We develop operating models for each species that mimic alternative hypotheses for population productivity, including compensatory mortality for Pacific Cod (based on Fournier, 1983); and a low M scenario for Rock Sole (based on Holt et al., 2016).

We explore five alternative HCRs that use combinations of OCPs based on MSY , SPR , B_0 , or estimated average historical biomass and fishing mortality. We evaluate performance of the HCRs in terms of five performance criteria that capture ecological and economic objectives common in fisheries. We explore mechanisms for differences in performance among alternative HCRs, particularly the propagation of estimation errors through the management cycle and their effect on performance. We discuss properties of alternative OCPs that make their performance more or less susceptible to bias and show that even when estimated parameters or OCPs are very biased, some HCRs can still produce desirable management outcomes, although consideration of trade-offs is important.

2. Methods

2.1. Species

Hecate Strait Pacific Cod and Rock Sole stocks are both currently assessed and managed as closed populations, with the majority of the BC catch of both species taken by bottom trawl vessels. Both stocks were recently assessed (Forrest et al., 2015; Holt et al., 2016) and we draw on data, parameter settings and maximum posterior density (MPD) estimates from these assessments for parameterizing the operating models in the present study. Both species are surveyed by fishery-independent bottom trawl surveys conducted by the Department of Fisheries and Oceans Canada (DFO). Biological data (length, sex and maturity) are routinely collected from the survey and commercial fishery for both species. Otoliths for subsequent age determination are also collected for Rock Sole. Very little age composition data exist for Pacific Cod due to well-known difficulties with ageing this species (Beamish, 1981; Chilton and Beamish, 1982; Johnston and Anderl, 2012). Commercial trawlers in BC are managed by an Individual Transferrable Quota (ITQ) program with 100% at-sea observer coverage and dockside monitoring of landings, resulting in highly accurate estimates of catch and discards (DFO, 2016).

2.2. Closed-loop simulation

The steps of closed-loop simulation have been described extensively elsewhere and readers are referred to Kell et al. (1999; their Fig. 2) and Punt and Hobday (2009; their Fig. 1) for schematic representations. Briefly, the approach is characterised by three components: (1) an operating model (OM) that represents hypotheses about the stock and fishery dynamics, and which is used to generate simulated observed

data and calculate “true” performance metrics (Tables A1–A3); (2) a management procedure (MP) that defines how data will be collected, analysed and translated into fishery catches using a HCR; and (3) an evaluation of the relative performance of MPs in (2) against objectives using the performance metrics in (1) (Cox et al., 2013). Two key components of the MP are the stock assessment model (AM), and the HCR.

We explored the performance of a set of five MPs (Section 2.2.4) under two alternative OM scenarios for Pacific Cod and Rock Sole (Fig. 3; Section 2.2.2). For each OM scenario-MP combination, the AM was a Bayesian statistical catch-age model (Table A4), which was fit to simulated survey, catch and age-composition data output from the OM (with observation errors) (Tables A.2.21–A.2.24) at biennial time steps during the projection period.

At each biennial time step, the AM produced maximum posterior density (MPD) estimates of: (i) model parameters (Eq. A.4.1); (ii) OCPs conditioned on the parameter estimates; and (iii) estimated stock status relative to OCPs in the HCR. Catch was removed from the OM population according to the HCR, assuming perfect implementation of the calculated catch from the HCR. In years between AM assessments, the catch was set to the same value as the previous year. At the end of the projection period, “true” performance metrics were calculated from the OM (Section 2.2.3). For each OM scenario-MP combination, the entire process was replicated $N = 100$ times, with random observation and process errors (Table 1).

Two time periods are defined: (1) the “historical period” t_1 to 2013, where t_1 is the first year of the historical period (1956 and 1945 for Pacific Cod and Rock Sole, respectively) and 2013 is the final year of available historical observations from the most recent stock assessments; and (2) the projected 50-year projection period 2014–2063, where 2014 is the first simulation year that the MP is applied and 2063 is the final simulation year of the projections (see Figs. 4 and 5). The historical period for each OM scenario was conditioned on MPD estimates of annual log recruitment deviations (ω_t) and fishing mortality rates (F_t) from a statistical catch-age model that had been specified to match the structural assumptions of each scenario and fit to real stock assessment data for each species (Forrest et al., 2015; Holt et al., 2016). In each case, this conditioning step produced a very close facsimile of the external stock assessment for the historical period (compare Fig. 2 with Figs. 4 and 5 for Pacific Cod and Rock Sole, respectively).

2.2.1. Implementation

Simulations were implemented in the *mseR* software framework (Kronlund et al., 2012; Cox et al. in press), modified for the present study. Details of the modelling approach are provided in Cox et al. (in press). Model equations specific to the present study are reported here in Appendix A (Tables A1–A5). The OM was implemented in the R programming environment (R Core Team, 2013). Key OM model equations are provided in Tables A1 and A2. Biological reference points were calculated in the OM using the equations in Table A3. The AM was a statistical catch-age model, conditioned on simulated catch data and fit to an index of abundance simulated in the OM with observation errors. The AM was implemented in AD Model Builder (Fournier et al., 2012) and is described in Table A4, with likelihoods described in Table A5.

2.2.2. OM scenarios

2.2.2.1. Pacific Cod. Two OM scenarios are defined for Pacific Cod: (i) Base; and (ii) Compensatory Mortality (DM). Key parameters for these scenarios are provided in Table 1. The Base OM scenario was conditioned on MPD time series estimates of fishing mortality and log-recruitment anomalies, estimated from a revised version of the

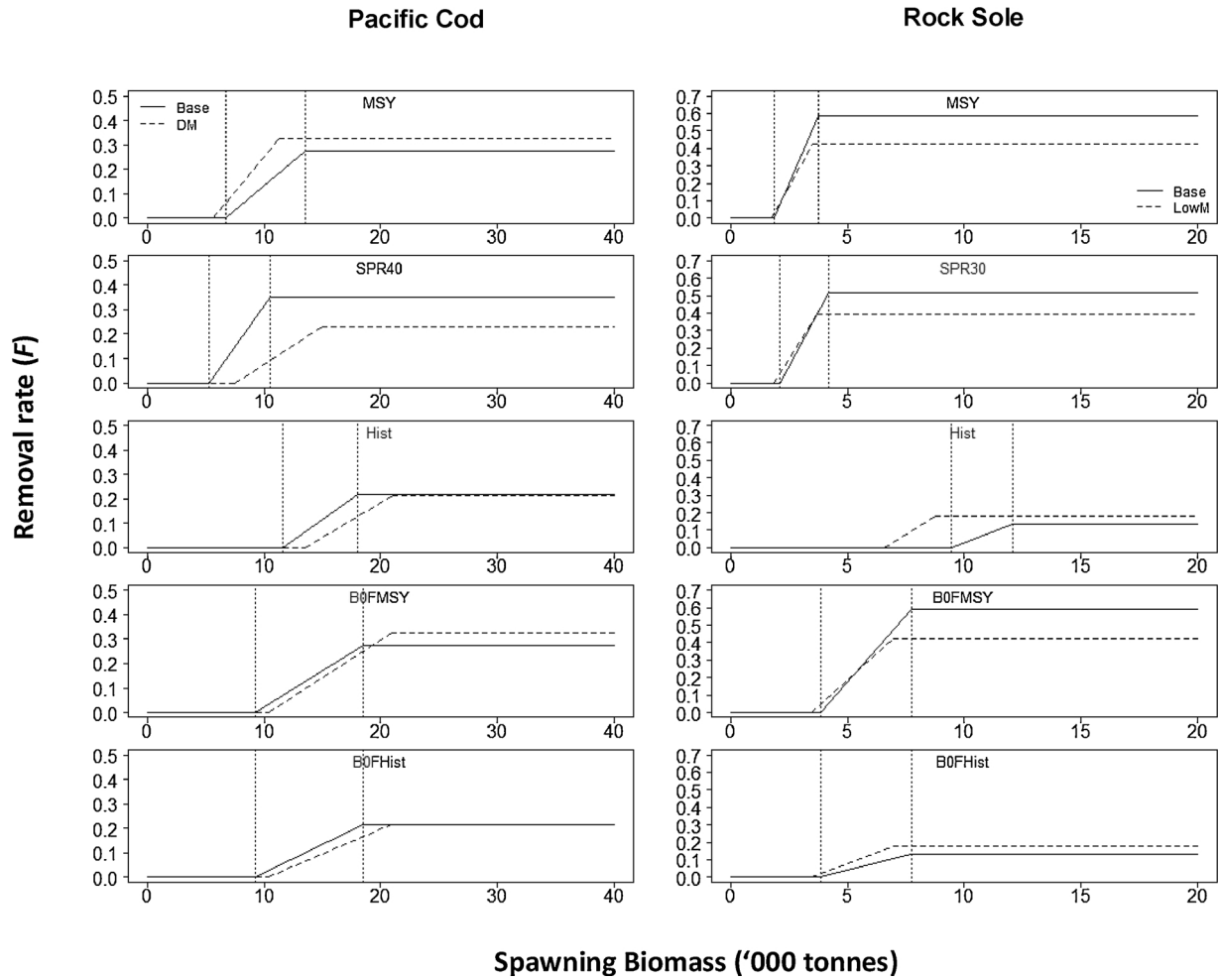


Fig. 3. HCRs for Pacific Cod and Rock Sole arising from two OM scenarios for each species. OM values of the Cutoff and USR for the Base OM scenario are shown as vertical dashed lines for the Base OM scenario. See Section 2.2.2 for description of OM scenarios. See Section 2.2.4 for description of HCRs.

Table 1

List of operating model (OM) scenarios and OM parameter values and settings. See Table A1 for key to symbols.

| Scenario | h | M | q | B_0 | σ_R | γ_R | a_{50}^{mat} | a_{95}^{mat} | $a_{50}^{sel,F}$ | $a_{95}^{sel,F}$ | $a_{50}^{sel,S}$ | $a_{95}^{sel,S}$ | τ_I | τ_P^F | τ_P^S | Other OM settings |
|-------------------|-------|---------|-----|-------|------------|------------|----------------|----------------|------------------|------------------|------------------|------------------|----------|------------|------------|-----------------------------|
| Pacific Cod: Base | 0.608 | 0.4 | 1 | 46.3 | 0.8 | 0.00 | 1.8 | 2.5 | 1.8 | 2.5 | 1.8 | 2.5 | 0.2 | 0.2 | 0.2 | NA |
| Pacific Cod: DM | 0.795 | Eq. (1) | 1 | 52.3 | 0.8 | 0.00 | 1.8 | 2.5 | 1.8 | 2.5 | 1.8 | 2.5 | 0.2 | 0.2 | 0.2 | $M_0 = 0.35$; $M_1 = 0.43$ |
| Rock Sole: Base | 0.86 | 0.25 | 1 | 19.33 | 0.22 | 0.41 | 5.6 | 8.0 | 5.7 | 7.7 | 4.8 | 7.0 | 0.2 | 0.2 | 0.2 | NA |
| Rock Sole: LowM | 0.85 | 0.20 | 1 | 17.40 | 0.23 | 0.46 | 5.6 | 8.0 | 5.7 | 7.6 | 4.6 | 6.8 | 0.2 | 0.2 | 0.2 | NA |

Integrated Statistical Catch Age Model (iSCAM) (Martell et al., 2011) that was configured to match the assumptions of the delay-difference model used in the 2013 stock assessment (Forrest et al., 2015), including fixed knife-edged maturity and selectivity schedules. Natural mortality was fixed in the Base OM at 0.4 y^{-1} , based on the MPD value from the most recent stock assessment (Forrest et al., 2015).

The DM scenario (Appendix B) represents the hypothesis that Pacific Cod productivity is driven by depensatory M in the adult population (Fournier, 1983), where M at time t (M_t) was modelled as a function of relative total biomass, i.e.,

$$M_t = \begin{cases} M_0 + (M_1 - M_0) \left(1 - \frac{B_t^{Tot}}{B_0^{Tot}}\right) & B_t^{Tot} \leq B_0^{Tot} \\ M_0 & B_t^{Tot} > B_0^{Tot} \end{cases} \quad (1)$$

where B_t^{Tot} is the current total biomass and B_0^{Tot} is the equilibrium

unfished total biomass (Logan et al., 2005). We use the symbol B_t^{Tot} to distinguish total biomass (Eq. A.2.19) from spawning biomass (Eq. A.2.17). M_0 and M_1 are constants, where M_t reduces to M_0 when the population is unfished ($B_t^{Tot} = B_0^{Tot}$), and approaches M_1 as B_t^{Tot} approaches zero (Eq. (1)). Therefore, as long as $0 < B_t^{Tot} \leq B_0^{Tot}$, M_t is constrained between M_1 and M_0 (Fig. B1a). To prevent unbounded linear increase or decrease in M_t when $B_t^{Tot} > B_0^{Tot}$, M_t was constrained to M_0 whenever B_t^{Tot} exceeded B_0^{Tot} in the simulations. A key property of this function is that when $M_1 < M_0$, mortality is compensatory (M_t decreases as B_t^{Tot} decreases), due to mechanisms such as reduced competition at lower biomass. Conversely, when $M_1 > M_0$, mortality is depensatory (M_t increases as B_t^{Tot} decreases), e.g., due to increased rates of predation at lower biomass. We selected OM values of M_0 and M_1 that resulted in depensatory mortality with mean $\bar{M}_t = 0.4 \text{ y}^{-1}$ across the whole time series (Table 1; Fig. B1b). We conditioned the historical period of the OM using MPD time series estimates of fishing mortality

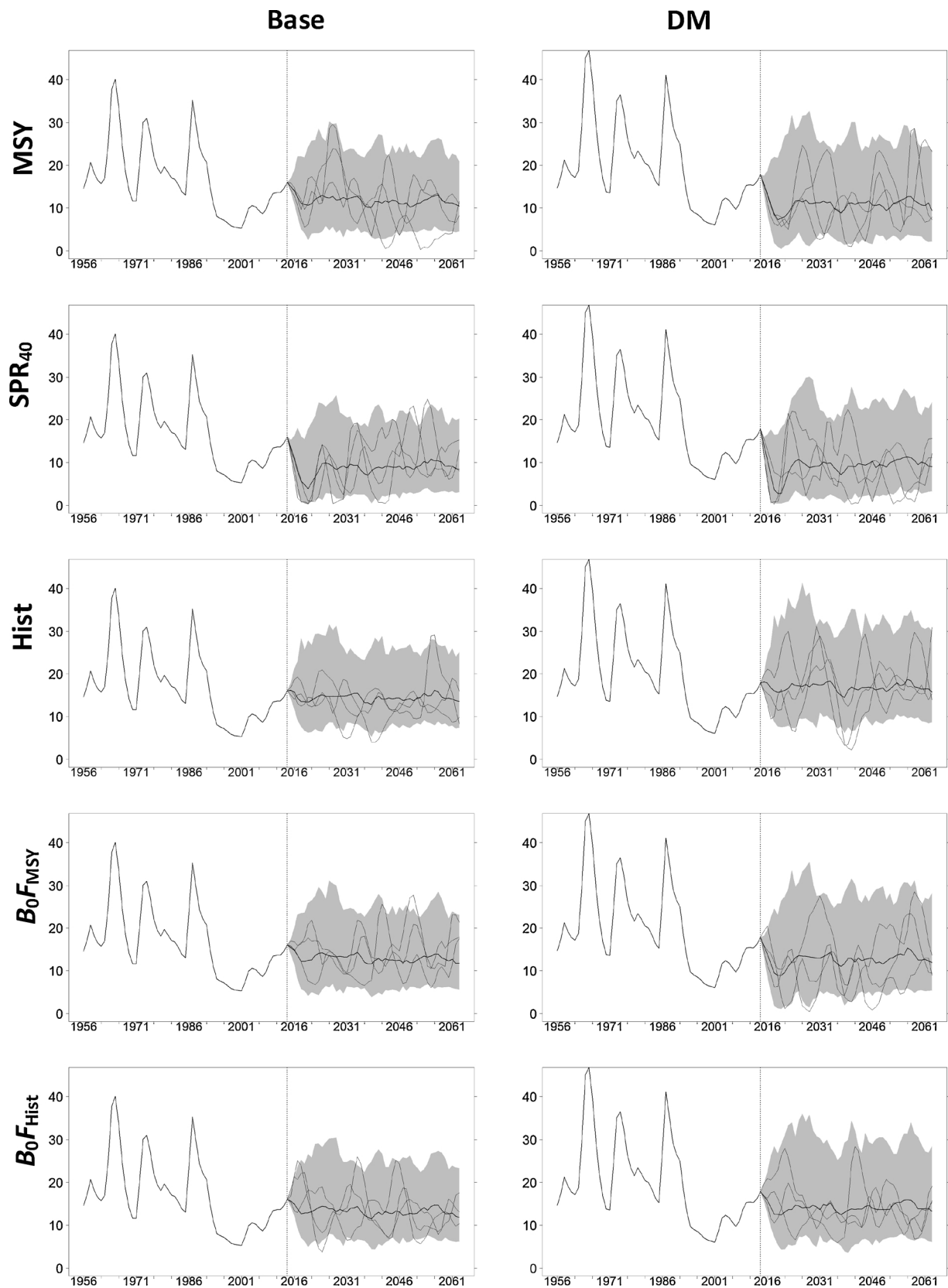


Fig. 4. OM values of spawning biomass ('000 metric tons) for Pacific Cod under the five HCRs for the Base (left panel) and DM (right panel) OM scenarios. The start year of the projection period (t_{MP}) is indicated with a vertical dashed line in each panel. Plot shows median (black line) of 100 replicates with shaded area indicating the 90% probability interval of median biomass among replicates in the projection period. Traces from three random replicates are provided to illustrate that individual replicates have varying trajectories.

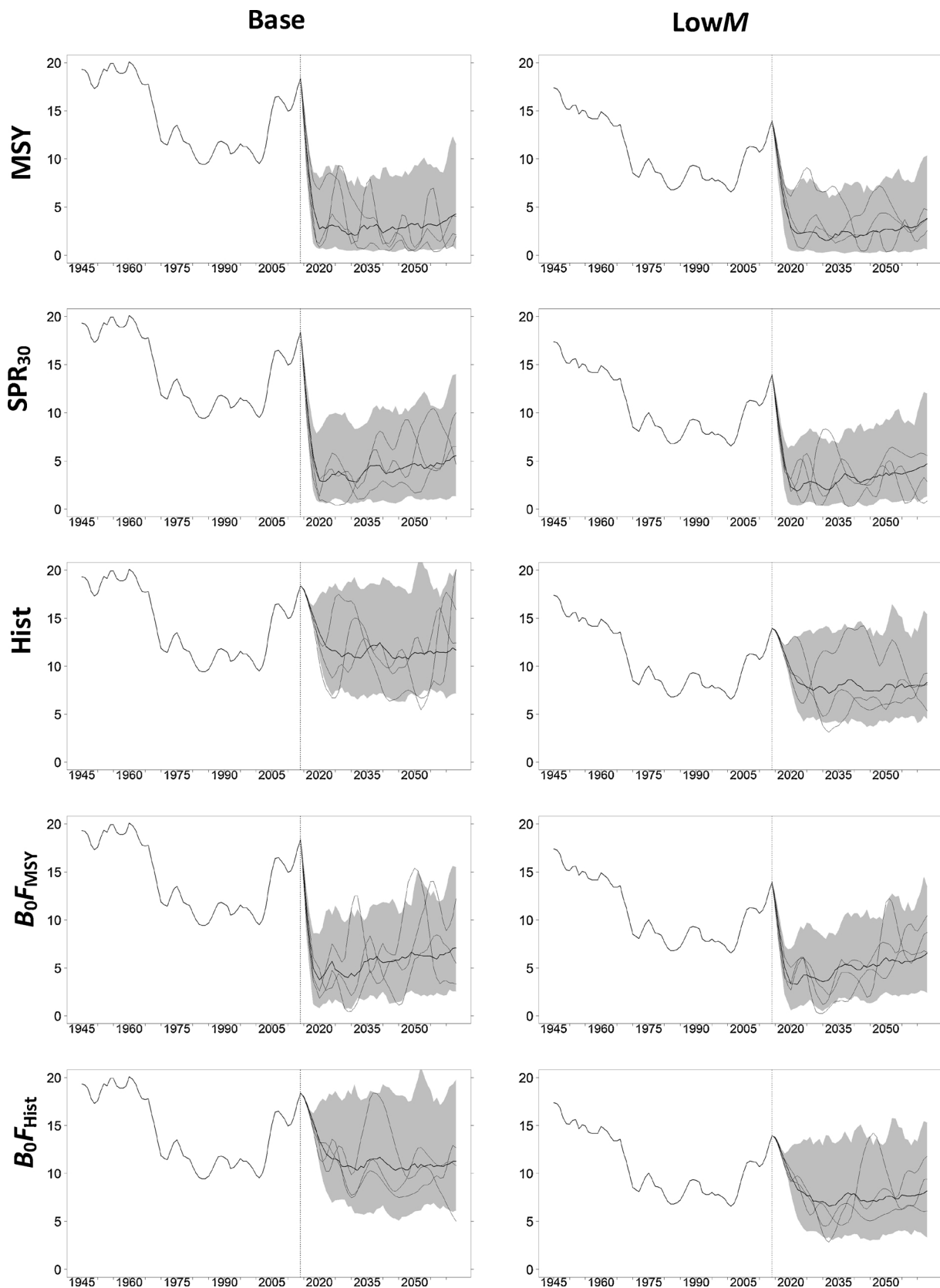


Fig. 5. OM values of spawning biomass ('000 metric tons) for Rock Sole under the five HCRs for the Base (left panel) and LowM (right panel) OM scenarios. The start year of the projection period (t_{MP}) is indicated with a vertical dashed line in each panel. Plot shows median (black line) of 100 replicates with shaded area indicating the 90% probability interval of median biomass among replicates in the projection period. Traces from three random replicates are provided as examples of individual replicates.

and log-recruitment anomalies, from a statistical catch age model configured with Eq. (1) and fixed values of M_0 and M_1 (Table 1). Note that the OM parameter values in Table 1 match the MPD or fixed parameter estimates from each external stock assessment model. Structural differences in these two models resulted in different estimates of other parameters such as steepness and B_0 .

Further properties of this function are described in Appendix B, which describes an algorithm for calculating equilibrium M for any given equilibrium value of fishing mortality. The equilibrium value of M is needed for calculation of reference points in the presence of density-dependent mortality. Appendix B shows that MSY-based reference points can differ considerably in the presence of density-dependent mortality.

2.2.2.2. Rock Sole. The two OM scenarios differed in terms of the constant value used for M . The Base scenario had $M = 0.25 \text{ y}^{-1}$ in the OM, and the Low M scenario had $M = 0.2 \text{ y}^{-1}$, based on the Base and “5CD_fixM” scenarios in the stock assessment (Holt et al., 2016). Other key OM parameter values are provided in Table 1. The OM Base scenario was conditioned on MPD time series estimates of fishing mortality and log-recruitment anomalies, from a Bayesian statistical catch-age model that assumed a normal prior probability distribution for M with mean $= 0.2 \text{ y}^{-1}$ and CV = 0.2, which produced an MPD estimate close to 0.25 y^{-1} (Holt et al., 2016). The Low M scenario was conditioned on MPD time series estimates of fishing mortality and log-recruitment anomalies from a sensitivity run in the stock assessment that had M fixed at 0.2 y^{-1} (Holt et al., 2016). As for Pacific Cod, there were differences in estimates of other key parameters between scenarios resulting from the different priors for M used in the stock assessments.

2.2.3. Performance metrics

We used five performance metrics to capture a range of ecological and economic factors of potential interest to decision-makers and stakeholders (Punt et al., 2008; Cox et al., 2013). Performance of each MP was evaluated over short-term (first 17 years) and long-term projection periods (last 16 years) of the projection period. For each period, the performance metrics in each replicate are:

- (1) Final relative spawning biomass $\left(\frac{B_{t2}}{B_0}\right)$;
- (2) Final spawning biomass relative to $B_{\text{MSY}} \left(\frac{B_{t2}}{B_{\text{MSY}}}\right)$;
- (3) Probability of $B_{t2} < 0.4B_{\text{MSY}}$ in year t_2 ($P(B_{t2} < 0.4B_{\text{MSY}})$);
- (4) Average catch ($\bar{C}_{t1:t2}$); and
- (5) Average annual variation catch (AAV), given by

$$\text{AAV} = \frac{\sum_{t=t_1}^{t_2} |C_t - C_{t-1}|}{\sum_{t=t_1}^{t_2} C_t} \quad (2)$$

where t_1 and t_2 define the first and last years of the short- or long-period, respectively. For clarity, the 1 and 2 subscripts are dropped from hereon.

Performance metrics 2 and 3 relate to the PA Framework’s target and limit spawning biomass objectives, where B_{MSY} and $0.4B_{\text{MSY}}$ are given as provisional values for the TRP and the LRP, respectively (DFO, 2009). We acknowledge that B_{MSY} and $0.4B_{\text{MSY}}$ are not fully-specified objectives, as they do not include a target probability or time frame over which to achieve the target probability (Kronlund et al., 2014). We present performance metrics in relative terms, e.g., a lower probability of breaching the LRP over each of the short and long time-horizons is considered to outperform a higher probability for that performance

metric. All performance metrics were calculated based on states in the OM to represent the “true” underlying performance rather than “perceived” AM performance.

The Cutoff in the HCR (Fig. 1) is usually labelled the LRP in the PA Framework (DFO, 2006a, 2009). In this study, the term LRP always refers to the low biomass threshold below which serious harm to the stock may occur ($0.4B_{\text{MSY}}$ as calculated in the OM).

2.2.4. Management procedures (MPs)

Five MPs were defined for each stock: MSY, SPR_x , Hist, B_0F_{MSY} , and B_0F_{Hist} . For each stock, the MPs differed only in terms of the OCPs that defined the HCR. Within each MP, OCPs and stock status were estimated biennially by the AM (Table A4). Fig. 3 presents the HCRs for each OM scenario. A perfectly-implemented HCR may be nominally more “precautionary” in two possible ways: (i) because the USR and Cutoff thresholds occur at higher biomasses (resulting in the stock falling below the OCPs at higher biomasses, triggering reduction or cessation in fishing); or (ii) because the LRR (i.e., fishing mortality rate) is lower. Typically, HCRs are considered to be more or less precautionary based on the nominal values of their OCPs. However, biased estimates of OCPs can result in implementation error.

2.2.4.1. MSY-based MP (MSY). The MSY-based MP is based on the provisional HCR defined under the PA Framework, which suggests $\text{USR} = 0.8B_{\text{MSY}}$ and $\text{Cutoff} = 0.4B_{\text{MSY}}$ (DFO, 2009). In this MP, we set the Cutoff at the LRP value of $0.4B_{\text{MSY}}$, which could result in harvests with substantial probability of breaching the LRP. For Pacific Cod, the values of the Cutoff and USR are slightly lower for the DM scenario than for the Base scenario (see Appendix B), while “true” F_{MSY} is higher (Fig. 3). While this may seem counter-intuitive, it reflects the higher steepness value for this scenario compared to the Base (Table 1). For Rock Sole, “true” F_{MSY} is high ($> 0.4 \text{ y}^{-1}$) under both scenarios, reflecting the relatively high values of steepness (Table 1, Fig. 3). F_{MSY} is especially high (0.59 y^{-1}) in the Base scenario relative to the Low M scenario (Fig. 3).

2.2.4.2. SPR-based MP (SRP_{30} , SPR_{40}). In the United States (US), proxies for F_{MSY} based on percentage of unfished spawning biomass per recruit are recommended (PFMC, 2014). By definition, F_x is the long-term constant fishing mortality rate that would reduce the reproductive per-recruit output to $x\%$ of unfished spawners-per-recruit, resulting in the long-term equilibrium spawning biomass B_x . Values for x are recommended between 35% and 60%, depending on the life history of the species (Clark, 1991, 2002; Mace, 1994), where the value of x that results in F_{MSY} is an inverse function of steepness (Williams, 2002; Punt et al., 2008; Forrest et al., 2010). However, while F_{MSY} is a function of both h and M , once x is selected, F_x is independent of h (although it can still be very sensitive to estimates of M). We elected to use values of $x = 40\%$ for Pacific Cod and $x = 30\%$ for Rock Sole, consistent with values applied in west coast US fisheries (PFMC, 2014).

For the Pacific Cod Base OM scenario F_{40} is considerably higher than F_{MSY} (Fig. 3). The LRR is much lower under the DM scenario than under the Base scenario, and both the Cutoff and USR are higher under the DM scenario than under the Base scenario, reflecting the need to maintain the stock at higher biomasses when mortality is depensatory. A more detailed presentation of the properties of these reference points with and without density-dependent adult mortality is provided in Appendix B.

For the Rock Sole Base OM scenario, F_{30} is lower than F_{MSY} (Fig. 3). While the Cutoff occurs at a similar value under both scenarios, the USR is slightly lower (nominally less precautionary) in the Low M scenario.

2.2.4.3. History-based MP (Hist). History-based OCPs for Pacific Cod (Fig. 3) are based on those used in the two previous stock assessments for this stock (Sinclair and Starr, 2005; Forrest et al., 2015). This set of OCPs consists of: (i) $USR =$ the estimated average biomass for the period 1956–2004 ($\bar{B}_{1956-2004}$); (ii) $Cutoff =$ the estimated minimum biomass from which the spawning biomass recovered to an above-average biomass level (B_{1971}); and (iii) $LRR = F_{Hist}$, the estimated average fishing mortality for the period 1956–2004 ($\bar{F}_{1956-2004}$). The history-based USR and $Cutoff$ in both OM scenarios are higher than their MSY -based counterparts, while the LRR is lower (Fig. 3).

History-based reference points for Rock Sole (Fig. 3) are based on those used in the most recent stock assessment (Holt et al., 2016), initially proposed in 2006 (DFO, 2006b). The set of OCPs consists of: (i) $USR =$ the estimated average biomass for the period 1971–1980 ($\bar{B}_{1971-1980}$), which was considered a period of average abundance (DFO, 2006b); (ii) $Cutoff =$ the estimated minimum biomass over the time period 1966 and 2005 ($B_{Min[1966-2005]}$); and (iii) $LRR = F_{Hist}$, the estimated average fishing mortality for the period 1966–2005 ($\bar{F}_{1966-2005}$). The history-based USR and $Cutoff$ are more than three times higher than their MSY -based counterparts and the LRR is more than three times lower (Fig. 3), nominally making this a very precautionary rule.

2.2.4.4. B_0 -based MPs (B_0F_{MSY} , B_0F_{Hist}). We test two B_0 -based MPs, each with USR and $Cutoff$ set to $0.4B_0$ and $0.2B_0$, respectively. The two MPs differ with respect to the LRR , which is set to either F_{MSY} or F_{Hist} (Section 2.2.3.3; Fig. 3). For both stocks, and all scenarios, the B_0 -based USR and $Cutoff$ are higher than their MSY or SPR -based counterparts (i.e., they are nominally more precautionary). They also tend to be further apart from each other, especially for Pacific Cod (Fig. 3), creating a wider ramp in the HCR over which the harvest rate is progressively reduced as biomass declines.

2.2.5. Bias

Bias in AM estimates of key model parameters and OCPs was summarized for each OM scenario-MP combination to evaluate the effects of parameter bias. We define bias as the difference between the estimated value (E) of a parameter or state variable and its “true” OM value (T). Bias is quantified using the Relative Error (RE) metric, given by

$$RE = 100 \left(\frac{E - T}{T} \right) \quad (3)$$

Taking the median RE across the $N = 100$ replicates gives the median relative error (MRE), which captures both bias and precision across replicates, where values approaching zero indicate unbiased estimates. Values greater than zero indicate positive bias and values less than zero indicate negative bias. Some studies calculate the median absolute relative error (e.g., Haltuch et al., 2008). In the present study, we were interested in the direction of the bias as well as its magnitude.

3. Results

3.1. Performance of alternative management procedures

3.1.1. Time series

Population dynamics for the historical period (i.e., prior to 2014) were similar between the two scenarios for Pacific Cod (Fig. 4). This is because both *iSCAM* models fit the commercial CPUE index very well (Forrest et al., 2015). Differences in model fit are, however, apparent in differences in parameter estimates from the two models, notably steepness and B_0 (Table 1). For the projection period, there was slightly greater variance in spawning biomass under the DM scenario, indicated

by broader envelopes for some MPs, especially for the Hist and B_0 MPs (Fig. 4). In both scenarios, the SPR_{40} MP resulted in the lowest median long-term relative spawning biomass (Table C1). The Historical MP generally resulted in higher relative spawning biomass under both scenarios (Fig. 4). Randomly-selected traces of individual replicates indicate large fluctuations in abundance from year to year, with overall magnitude varying among replicates and MPs (Fig. 4). In all cases, the relatively stable median relative spawning biomass did not reflect this large annual variation.

Large annual fluctuation in abundance within replicates also occurred for Rock Sole (Fig. 5). For this species, there was a more marked difference in relative spawning biomass among MPs. The lowest relative spawning biomass occurred under the MSY and SPR_{30} MPs, with the largest relative spawning biomass occurring under the Historical and B_0F_{Hist} MPs. The MSY MP resulted in frequent stock crash in both scenarios, as did the SPR_{30} MP (indicated by random traces and the bottom of the simulation envelope close to zero in Fig. 5). It is important to note that, for the entire historical OM period, Rock Sole was assessed to have been well above B_{MSY} (Holt et al., 2016). The MSY and SPR_{30} MPs are designed to maintain average biomass around B_{MSY} , so large initial depletion from the high historical state occurred under these MPs.

3.1.2. Long-term conservation metrics

3.1.2.1. Pacific Cod. The SPR_{40} MP resulted in the lowest long-term (years 2048–2063) median values of both final relative spawning biomass ($\frac{B_t}{B_0}$) and stock status ($\frac{B_t}{B_{MSY}}$) across both scenarios (Fig. 6a,b), and resulted in the greatest probability of spawning biomass falling below the OM LRP of $0.4B_{MSY}$ (Tables C1 and C2; Fig. 6b).

The Historical, B_0F_{MSY} and B_0F_{Hist} MPs performed similarly to each other, in terms of both long-term relative spawning biomass and $\frac{B_t}{B_0}$ and $\frac{B_t}{B_{MSY}}$, with the exception that the B_0F_{Hist} MP resulted in slightly higher relative spawning biomass under the DM scenario (Tables C1 and C2; Fig. 6a,b). The Historical MP was the only one that resulted in median long-term spawning biomass greater than the target B_{MSY} , which occurred under the DM scenario. Even though the AM's assumptions were mismatched with those of the DM OM (i.e., did not incorporate depensation in M), feedback in the system worked to maintain higher levels of biomass in this scenario. This is explained by the depensatory relationship between biomass and M , where M is reduced when biomass is high (Eq. (1); Appendix B). Therefore, more precautionary HCRs (Hist, B_0F_{MSY} and B_0F_{Hist}) that did not deplete biomass (Fig. 3) could prevent increases in M occurring with lower biomass.

Under the Base scenario, all MPs resulted in median spawning biomass less than the target B_{MSY} (Fig. 6b), indicating implementation bias even under the Base scenario, despite the fact that the AM's structural assumptions about M and other parameters matched those of the Base OM.

3.1.2.2. Rock Sole. There was noticeable contrast in final long-term (years 2048–2063) performance among MPs and scenarios for Rock Sole (Fig. 6c,d). The MSY MP resulted in the lowest long-term median values of both $\frac{B_t}{B_0}$ and $\frac{B_t}{B_{MSY}}$ for all scenarios, and the greatest probability of spawning biomass falling below the OM LRP (Table C1; Fig. 6c,d). Both the MSY and SPR_{30} MPs had non-zero probability of the final spawning biomass falling below $0.4B_{MSY}$ in the long-term (Table C1). The highest long-term median values of $\frac{B_t}{B_0}$ and $\frac{B_t}{B_{MSY}}$ were obtained under the Historical MP, where the entire probability interval exceeded B_{MSY} for all scenarios (Fig. 6d). The entire 95% probability interval for final spawning biomass was also maintained above B_{MSY} under the B_0F_{Hist} MP in the Base scenario. (Table C1; Fig. 6d).

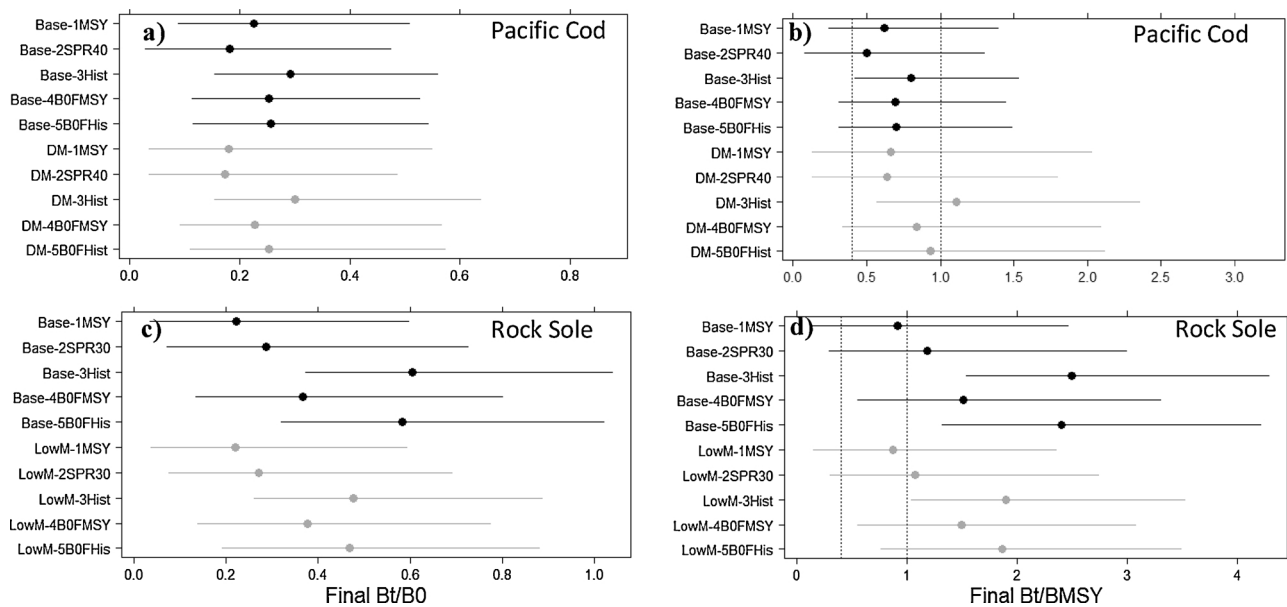


Fig. 6. Long-term conservation-based performance metrics under the five HCRs for: (a,b) Pacific Cod for the Base (black bars) and DM (grey bars) OM scenarios; and (c,d) Rock Sole for Base (black bars) and LowM (grey bars) OM scenarios. Figure shows final OM $\frac{B_t}{B_0}$ (left panel) and final OM $\frac{B_t}{B_{MSY}}$ (right panel). The median of 100 replicates is shown as a point, with bars representing the 95% probability interval. The provisional DFO BRPs of B_{MSY} and $0.4B_{MSY}$ (DFO, 2009) are shown as vertical lines on the right panel.

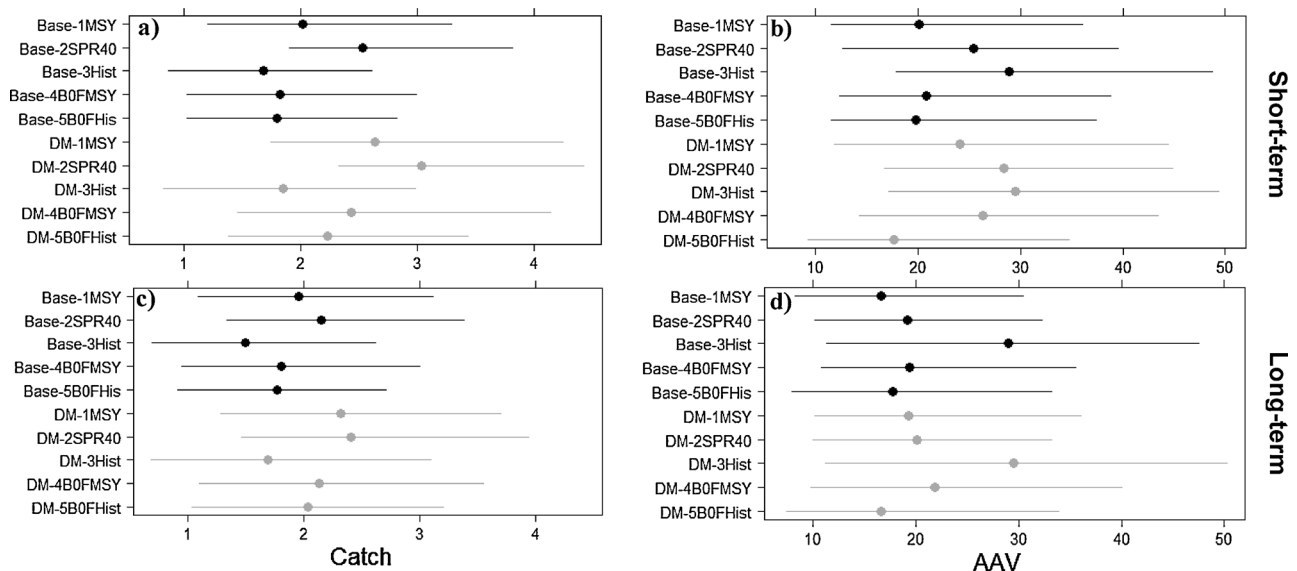


Fig. 7. (a,b) Short-term and (c,d) long-term catch-based performance metrics for Pacific Cod under the five HCRs for the Base (black bars) and DM (grey bars) OM scenarios. Figure shows final OM average catch (left panel) and OM AAV (right panel) for the short and long-term projection periods. The median of 100 replicates is shown as a point with bars representing the 95% probability interval.

3.1.3. Catch metrics

3.1.3.1. Pacific Cod. In the short-term (years 2014–2030), median average catch was notably higher under the SPR_{40} MP for both scenarios (Table C1; Fig. 7a). Within scenarios, the MSY MP resulted in the second highest median short-term catch (Table C1; Fig. 7a). Relative performance of the Historical and B_0 -based MPs differed between scenarios. Under the Base scenario, the Historical MP performed similarly to the B_0 -based MPs, especially in the short term. However, under the DM scenario, the B_0 -based MPs resulted in relatively higher short-term and long term median catches (Fig. 7a,b).

Short- and long term AAV was greatest under the Historical MP for both scenarios. In most replicates, the stock status relative to OCPs, estimated by the AM, frequently resulted in harvest rates being reduced and even in fishery closures. This occurred because the USR and Cutoff in the Historical HCR occurred at higher biomass than in the other MPs (Fig. 3), resulting in greater probability of harvest rate reduction (i.e., on the “ramp” of the HCR) and fishery closures. In addition, estimates of the Historical OCPs tended to be biased high, exacerbating this effect (see Section 3.2.1). Note that for the Historical MP, the “true” OM LRP of $0.4B_{MSY}$ is far below the Historical Cutoff (Fig. 3), resulting in the

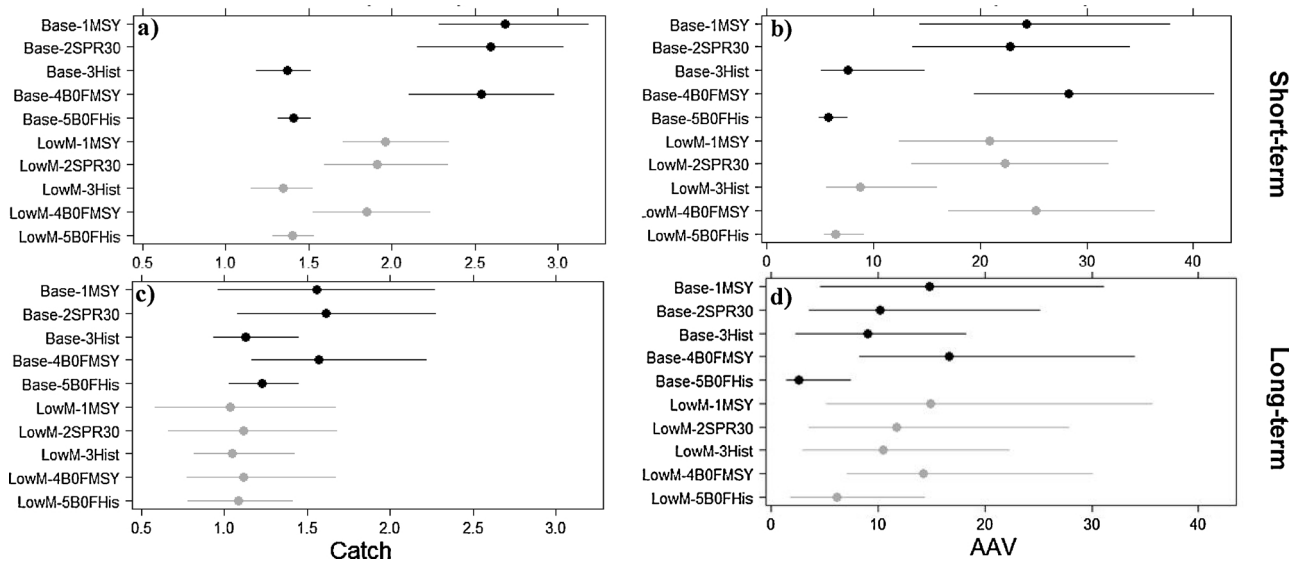


Fig. 8. (a,b) Short-term and (c,d) long-term catch-based performance metrics for Rock Sole under the five HCRs for the Base (black bars) and LowM (grey bars) OM scenarios. Figure shows final OM average catch (left panel) and OM AAV (right panel) for the short and long-term projection periods. The median of 100 replicates is shown as a point with bars representing the 95% probability interval.

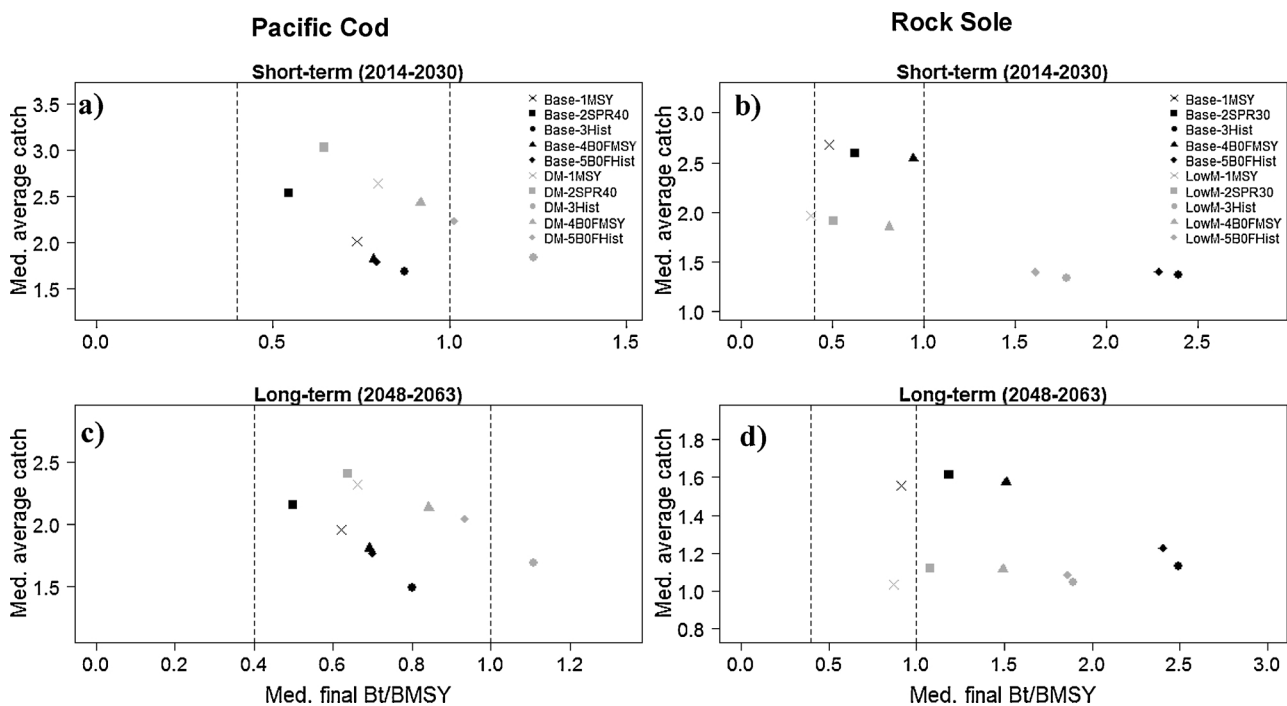


Fig. 9. Trade-off between $\frac{B_t}{B_{MSY}}$ and median average catch for: (a,c) Pacific Cod and (b,d) Rock Sole for the (a,b) short-term and (c,d) long-term projection periods, under the Base scenario (black symbols) and DM or LowM scenarios (grey symbols). The provisional B_{MSY} -based DFO BRPs of B_{MSY} and $0.4B_{MSY}$ (DFO, 2009) are shown as vertical lines. Note different scales.

stock being maintained well above the LRP under the Historical MP (Fig. 6b).

Short- and Long-term AAV were lowest under the B_0F_{Hist} MPs for both scenarios (Table C1; Fig. 7b). B_0 tended to be estimated without bias, while B_t tended to be overestimated (see Section 3.2.1). Therefore there was higher probability that the stock was estimated to be above the Cutoff and less frequent closures. In the long term, there was

slightly less contrast among MPs, although long-term catch was lowest and AAV was highest under the Historical MP for both scenarios.

3.1.3.2. Rock Sole. Differences between short and long-term catch metrics were large for Rock Sole (Table C1; Fig. 8). Median short-term average catches (years 2014–2030) were around twice as low under the Historical and B_0F_{Hist} MPs than for the other MPs, with no

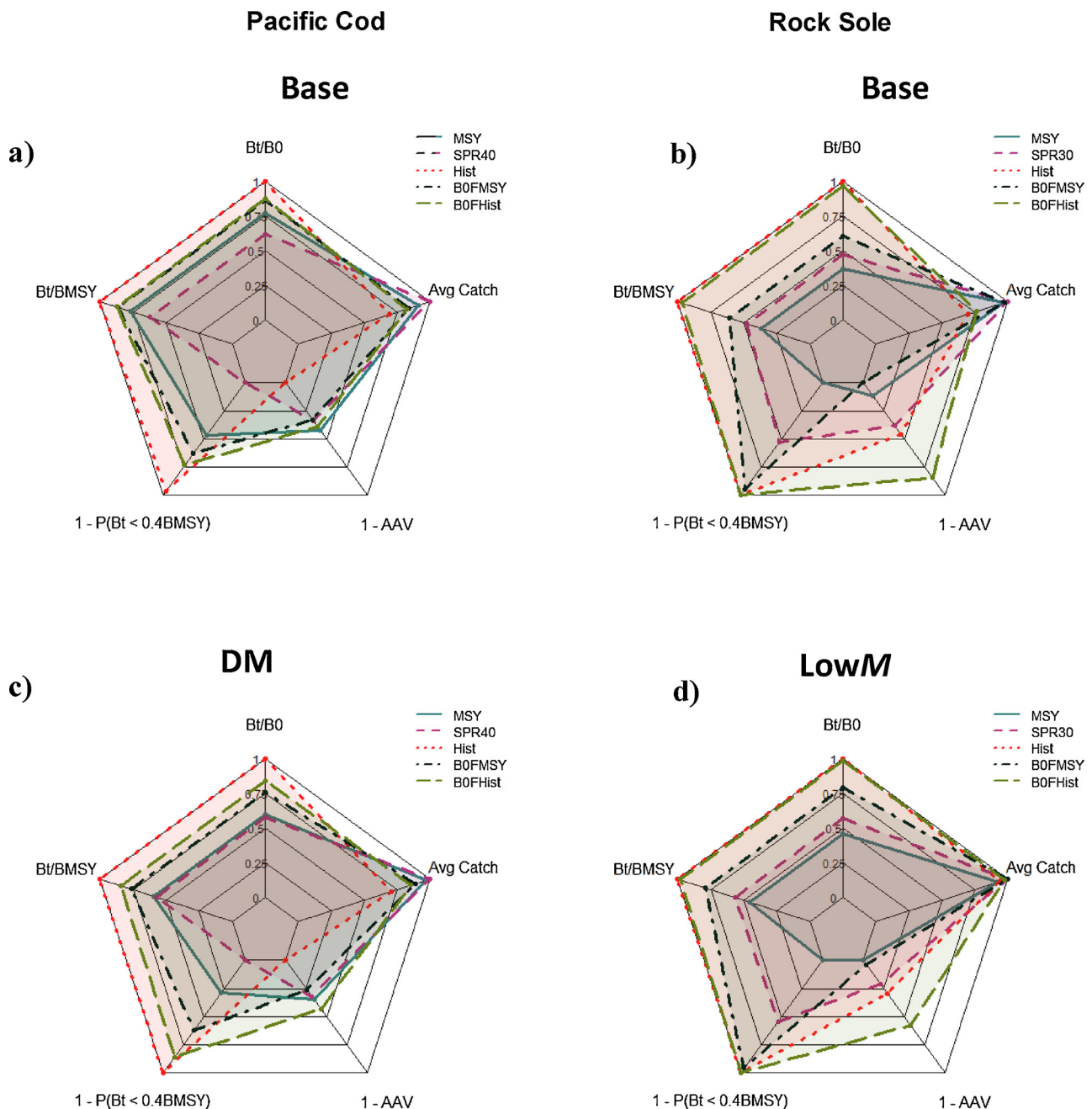


Fig. 10. Radar plots for: (a,c) Pacific Cod and (b,d) Rock Sole, for the Base, DM and LowM scenarios, showing performance of the five HCRs in terms of five performance metrics: $\frac{B_t}{B_0}$, $\frac{B_t}{B_{MSY}}$, Average Catch, $1 - AAV$ and $1 - P(B_t < B_{MSY})$. Larger polygons indicate better performance across all scenarios. (For interpretation of the references to colour in the text, the reader is referred to the web version of this article.)

overlap in probability intervals (Fig. 8a). Short term average catches were generally lower for the LowM scenario, especially under the MSY, SPR₃₀ and B_0F_{MSY} MPs. In contrast to Pacific Cod, short-term AAV was very low under the Historical MP for both OM scenarios. In this case, even though the OCPs were much higher than for other MPs (Fig. 3), the low fishing mortality rate resulted in low catch variance for many replicates. As for Pacific Cod, the lowest median AAV resulted from the B_0F_{Hist} MP under both scenarios (Fig. 8b).

The large differences in average catch and AAV between MPs were less pronounced in the long-term period (Fig. 8c,d). For the Base scenario only, long-term average catches under the Historical and B_0F_{Hist}

MPs were still lower than the other MPs (Fig. 8c). However, average catches under the other MPs had decreased significantly since the start of the projection period. This was, by design of the MP, the result of a short period of “fishing down” the large standing stock at the beginning of the projection period to achieve MSY, followed by stabilization of catch and biomass at lower levels. Given that the Historical MP is effectively designed to maintain the stock at historical average levels, this fishing down effect did not occur under the Historical MP, and catches remained low in both the short- and long-term. There was a small difference in long-term catch performance between scenarios (Fig. 8c), with lower overall catches under the LowM scenario.

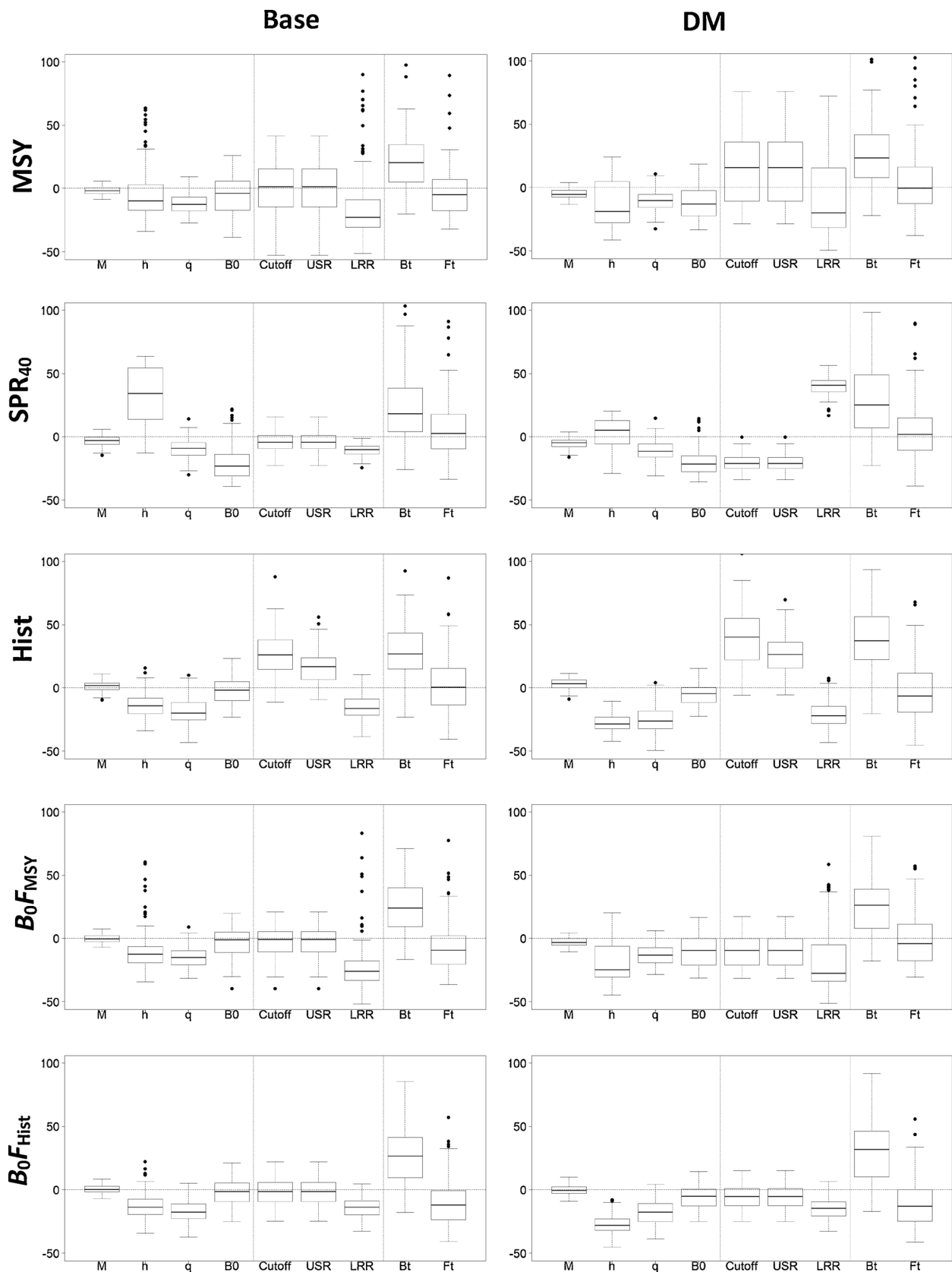


Fig. 11. Long-term Relative Errors (RE) for Pacific Cod for four estimated parameters (M , h , q , B_0), three OCPs (LRR, USR, LRR), and two state variables (B_t , F_t), for the two OM scenarios (columns) under the five HCRs (rows). RE measures the difference between the estimated value of a parameter in the MP and its “true” OM value (Eq. (3)). Boxes show the range of relative errors across the 100 replicates for each scenario-MP combination, with the median relative error (MRE) shown as a bar in the centre of each box (Table C2). Boxes represent the interquartile range (IQR). Whiskers represent 1.5 IQR, with outliers shown as dots.

3.1.4. Trade-offs

There was a notable negative relationship between median $\frac{B_t}{B_{MSY}}$ and median average catch for Pacific Cod, for both scenarios, in both the short and long term (Fig. 9a,c). In all cases, the SPR₄₀ MP resulted in the highest average catch and the lowest relative biomass, while the opposite was true for the Historical MP. The most striking difference for Rock Sole was between the short and long term, where average catches in the short term were almost twice as large as in the long term, creating a much larger trade-off (Fig. 9b,d). For the Base scenario, the long-term trade-off patterns were not as strong as for Pacific Cod, with three MPs (MSY, SPR₃₀ and B_0F_{MSY}) resulting in similar levels of average catch for the Base scenario (Fig. 9d). In the long-term, only the MSY MP resulted in median biomass below the proxy target of B_{MSY} and no MPs resulted in median biomass below $0.4B_{MSY}$ (Fig. 9d). Unlike for Pacific Cod, this suggests a less marked trade-off.

Overall median performance across all five performance metrics was captured using radar plots (or kite diagrams) (Fig. 10). Each metric was scaled to the maximum value within each MP and scenario, resulting in a maximum value of 1 for each metric. Then, because smaller values are “better” for the metrics $P(B_t < 0.4B_{MSY})$ and AAV, we took the complement of these metrics to ensure that larger polygons always represent better trade-off performance (Fig. 10). For example, for Pacific Cod, while the SPR₄₀ MP performed best in terms of long-term average catch, its performance was poorer for all of the other metrics as indicated by its small polygon (Fig. 10a,c, magenta dashed line). While the Historical MP had the best conservation performance, it performed worst in terms of AAV and average catch (Fig. 10a,c, red dotted line). These trade-offs were robust between the two scenarios. The B_0F_{Hist} MP resulted in similar long-term average catches to the MSY MP, especially under the DM scenario, but had better conservation performance, indicated by the larger polygon (Fig. 10a,c, green dashed line). While the B_0F_{Hist} MP performed similarly to the B_0F_{MSY} MP (blue dot-dashed line) under the base scenario (Fig. 10a), it had better trade-off performance than B_0F_{MSY} under the DM scenario (Fig. 10c).

For Rock Sole, while the MSY MP resulted in the largest long-term average catch, its small polygon indicates that it performed poorly for all other metrics under both scenarios (Fig. 10b,d, blue solid line). The B_0F_{Hist} MP had the largest polygon for both scenarios and, while there was a trade-off with average catch, especially for the Base scenario, this could arguably be offset by greater stability in catches indicated by the larger relative 1–AAV (Fig. 10b,d, green dashed line). This was in contrast to the B_0F_{Hist} MP for Pacific Cod, the same MP which performed poorly in terms of AAV. As noted above, in the long term, all but the MSY MP resulted in median biomass above B_{MSY} , suggesting that conservation performance could be less useful for selecting MPs in the long term.

For both species and all scenarios, the best long-term conservation performance was achieved with the Historical and B_0F_{Hist} MPs, with better trade-offs evident for the B_0F_{Hist} MP.

3.2. Effects of bias

3.2.1. Pacific Cod

Median relative errors (Eq. (3)) in model parameters for the short-term (years 2014–2030) and long-term (years 2048–2063) projection periods are provided in Table C2. Long-term MREs in model parameters, OCPs, final spawning biomass and final fishing mortality are shown in Fig. 11. MREs tended to be greater in magnitude for the short-term period, reflecting the shorter time-series of data available for parameter estimation (Table C2).

The AM tended to overestimate long-term spawning biomass under all scenarios. This was largely driven by underestimation of survey

catchability (q). Surprisingly, M was estimated with very little bias in both scenarios (Table C2; Fig. 11). The other model parameters in the DM scenario were, however, estimated with quite large bias in some cases, particularly steepness. Relative errors were not uniform across MPs within scenarios (Fig. 11), indicating feedback between the management procedure and the information in the data for estimating parameters.

To illustrate that performance can arise from different mechanisms, we point to results from the SPR₄₀ MP, which resulted in the lowest median relative spawning biomass in the final year (Fig. 6a). Relative errors for this MP indicate different mechanisms under the Base and DM scenarios. For the Base scenario, the greater depletion that occurred under the SPR₄₀ MP compared to other MPs was due to the nominally higher LRR and lower OCPs (Fig. 3), combined with overestimated biomass and slightly underestimated Cutoff and USR (Fig. 11). Even though steepness was mostly overestimated with the SPR₄₀MP (Fig. 11), the LRR F_{40} is unaffected by steepness, which is the reason F_{40} is often promoted as a robust alternative to F_{MSY} . Under the DM scenario, however, there was large overestimation of F_{40} due to failure to account for compensatory M in the equilibrium calculations of F_{40} (Fig. 3, Appendix B), coupled with underestimation of the Cutoff and USR (Fig. 11). Overestimates of F_{40} combined with large overestimates of B_t to result in lower values of $\frac{B_t}{B_0}$ and $\frac{B_t}{B_{MSY}}$ compared to the other MPs (Fig. 6a,b).

When the B_0F_{MSY} MP was applied under the DM scenario, bias patterns in OCPs and B_t were similar to the SPR₄₀ MP, but depletion was less severe because the B_0 -based Cutoff and USR were nominally more precautionary (Fig. 3), and were also estimated with less bias (Fig. 11).

The Historical MP was very precautionary in terms of long-term $\frac{B_t}{B_0}$ and $\frac{B_t}{B_{MSY}}$ (Fig. 6a,b). The Cutoff and USR for this HCR were calculated directly from estimated historical biomass. Therefore, as biomass was overestimated, so were these OCPs, and the LRR was correspondingly underestimated (Fig. 11). This effect was stronger under the DM scenario. The effects of overestimating biomass were, therefore, counteracted in the Historical MP by positive bias in estimates of the Cutoff and USR for both scenarios.

The B_0F_{Hist} MP produced a very stable catch time series, indicated by very low AAV (Fig. 7c,d). Estimates of B_0 and associated OCPs were relatively unbiased (Fig. 11), while the LRR (F_{Hist}) in this MP was similarly underestimated, as in the Historical MP. Therefore, the underestimated LRR countered the overestimates in B_t (as for the Historical MP), and the unbiased B_0 -based Cutoff resulted in fewer unnecessary closures, resulting in lower AAV.

3.2.2. Rock Sole

In general, there were less long-term AM biases for Rock Sole (Fig. 12). However, the MSY and SPR₃₀ MPs produced large positive bias in estimated biomass under both scenarios (Fig. 12). This was due to a positive retrospective pattern (Mohn, 1999) that was present in many of the replicates (not shown), during the period of rapidly declining biomass that followed implementation of F_{MSY} and F_{30} . Bias in estimated model parameters was considerably greater in the short-term, especially for M and B_0 (Table C2). MSY and SPR₃₀-based OCPs tended to be estimated with relatively little bias in the long-term (Fig. 12). However, the overestimated biomass resulted in implementation error that caused the stock to fall below the estimated Cutoff in some replicates.

The Historical MP for Rock Sole was much more precautionary than for Pacific Cod (Fig. 3). Average historical biomass, on which the Cutoff and USR were based, was well above B_{MSY} , and the Historical MP maintained biomass close to these historical levels (Fig. 5). Any positive assessment biases in estimated B_t were therefore mirrored in the

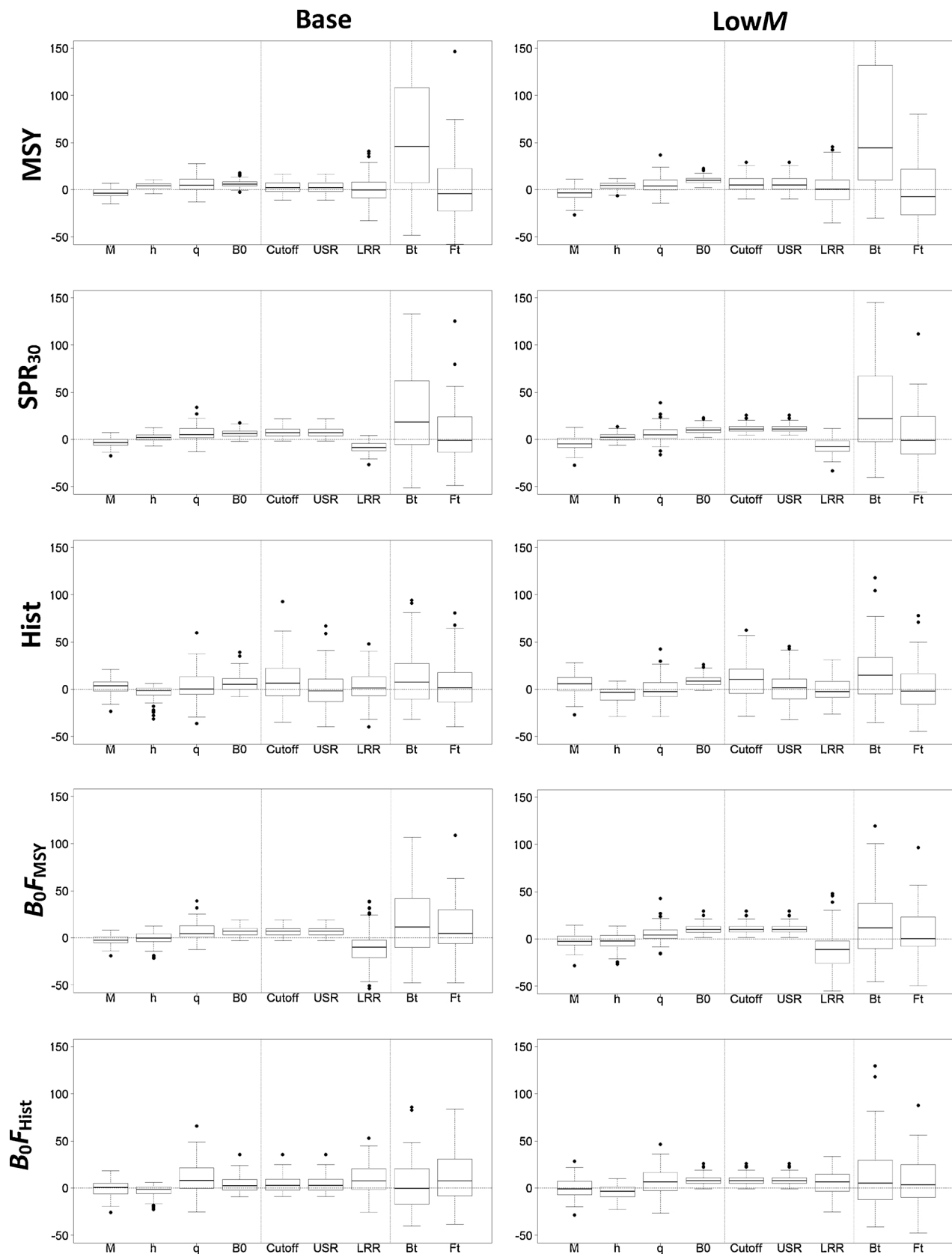


Fig. 12. Long-term Relative Errors (RE) for Rock Sole for four estimated parameters (M , h , q , B_0), three OCPs (LRR, USR, LRR), and two state variables (B_t , F_t), for the two OM scenarios (columns) under the five HCRs (rows). RE measures the difference between the estimated value of a parameter in the MP and its "true" OM value (Eq. (3)). Boxes show the range of relative errors across the 100 replicates for each scenario-MP combination, with the median relative error (MRE) shown as a bar in the centre of each box (Table C2). Boxes represent the interquartile range (IQR). Whiskers represent 1.5 IQR, with outliers shown as dots.

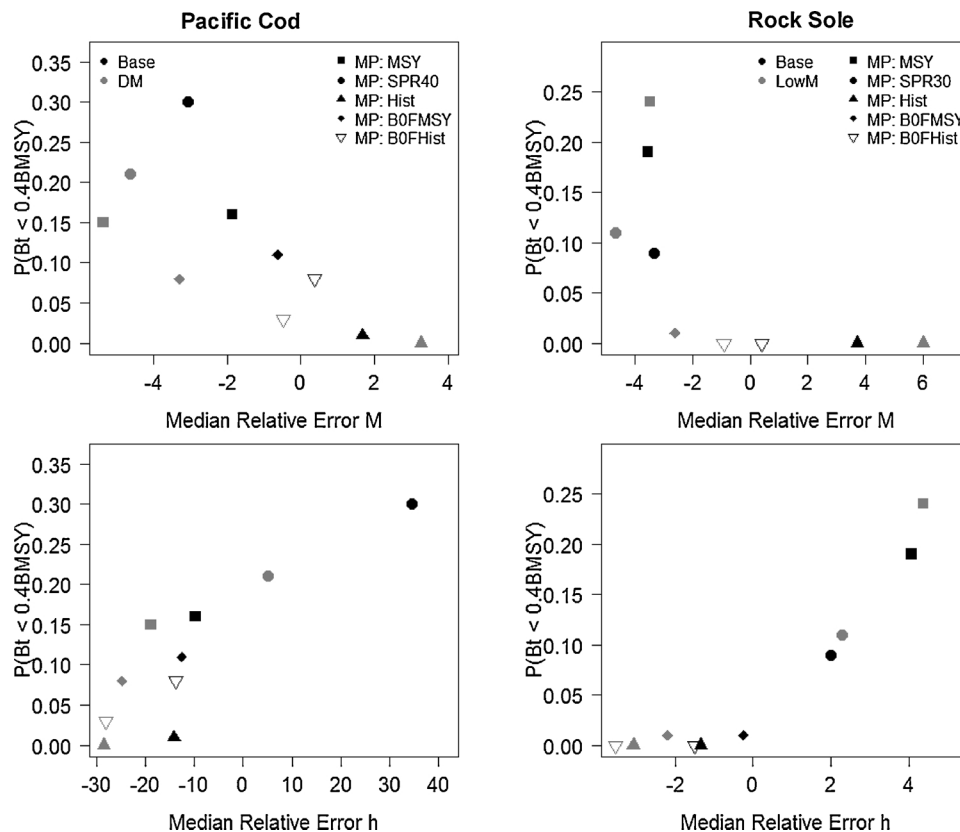


Fig. 13. Long-term MRE for M (top panel) and h (bottom panel) plotted against long-term $P(B_t < 0.4B_{MSY})$ for Pacific Cod (left panel) and Rock Sole (right panel). Note different scales.

Historical OCPs, making them more precautionary and resulting in higher average biomass.

There was very little bias in estimates of all OCPs under the Base scenario (Fig. 12). Overall performance was therefore better than under the LowM scenario (Fig. 6c,d). For both the Historical and B_0F_{Hist} MPs, feedback between the management procedure and the OM resulted in less biased estimates of B_t than for other MPs (Table C2; Fig. 12), which also contributed to their more stable behaviour.

3.2.3. Relationships between bias and performance

Mechanisms for performance of MPs differed among species, scenarios and HCRs, but could be understood relatively easily by looking at biases in AM estimates of key model parameters and OCPs. Common causes of poorer performance in terms of conservation metrics included: (i) overestimation of biomass; (ii) overestimation of the LRR; and (iii) underestimation of the Cutoff and USR. Many of these biases could be attributed to biases in estimated model parameters, arising from incorrect structural assumptions about M (Pacific Cod) or biased prior probability assumptions for M (Rock Sole).

Fig. 13 shows MREs for M and h plotted against the probability of the “true” biomass falling below the LRP in the long term ($P(B_t < 0.4B_{MSY})$). For both species, there was a weak negative relationship between MRE in M and long-term $P(B_t < 0.4B_{MSY})$; and a positive relationship between MRE in h and long-term $P(B_t < 0.4B_{MSY})$. Conservation performance of the MSY and SPR_x-based MPs were most affected by bias in M and h , while performance of the Historical and B_0 -based HCRs was little or unaffected (Fig. 13). Results for both species indicate that effects of bias in parameter estimates are dependent on the choice of MP.

4. Discussion

This study evaluated the performance of five MPs that differed in the choice of OCPs for two BC groundfish stocks under alternative assumptions about stock productivity. Both structural uncertainty (Pacific Cod) and parameter uncertainty (Rock Sole) were considered. Performance of the five MPs differed between OM scenarios, highlighting the importance of considering multiple hypotheses when designing HCRs. Trade-offs between conservation and yield objectives differed substantially among MPs and were large in some cases. While the Historical MPs were the most precautionary for both species across all productivity scenarios, they were associated with considerable losses in short-term average catch for both species, highlighting the need to consider performance over both short and long time-horizons. The two B_0 -based MPs tended to perform intermediately between the Historical MP and the SPR and MSY MPs in terms of yield metrics, and similarly to the Historical rule in terms of conservation metrics, resulting in less severe long-term trade-offs.

We presented an efficient numerical algorithm for calculating equilibrium density-dependent M for any constant equilibrium value of F , thus enabling the calculation of OCPs and BRPs with depensatory adult mortality for Pacific Cod. All other parameters equal, equilibrium B_{MSY} and F_{MSY} are lower when adult mortality is depensatory. Other studies have also shown that density-dependent dynamics can significantly influence calculation of reference points and can have large management implications (Liermann and Hilborn, 2001; Powers, 2005; Brooks and Powers, 2007; Forrest et al., 2013). However, in the context of a full stock assessment model, some estimated reference points will also be contingent on estimates of other model parameters, particularly

steepness. In our study, “true” F_{MSY} was slightly larger in the DM scenario compared to the Base, due to higher estimated steepness. Conversely, the “true” value of F_{40} , which does not depend on steepness, was considerably lower in the DM scenario compared to the Base. Under the DM scenario, misspecified M led to large overestimates of F_{40} by the AM and, coupled with overestimation of spawning biomass, overall worse conservation performance of the SPR_{40} MP. For both species, especially Pacific Cod, long-term performance patterns among MPs were similar between scenarios. Much greater differences were obtained among MPs within scenarios, suggesting that, within the context of our simulations, the choice of HCR was more important than correctly modelling system dynamics in the AM.

Overestimation of spawning biomass was present in all MPs under all scenarios, especially for Pacific Cod, which led to overharvesting under some MPs, notably those with SPR and MSY-based MPs. For this species, neither the MSY nor SPR-based MPs were successful in maintaining median biomass above “true” target B_{MSY} value determined by the OM. This emphasizes the risk of using more aggressive MPs in the presence of assessment errors since bias in estimates of OCPs coupled with positive assessment bias could lead to overharvesting. However, even when OCP estimates were unbiased, poor conservation performance could occur due to implementation error arising from overestimation of biomass in the stock assessment.

Our results agree with those of previous closed-loop simulation studies, which showed that high precision or accuracy in parameter estimates is not always sufficient to achieve desired outcomes, where effects of implementation error could outweigh the benefits of precise parameter estimation (Peterman et al., 2000; Dorner et al., 2009; Collie et al., 2012). While understanding relationships between reference points and their component parameters is necessary (Clark, 1991; Mace, 1994; Punt et al., 2008; Mangel et al., 2013), bias in these quantities needs to be evaluated in the context of the management cycle as a whole (de la Mare, 1998; Kronlund et al., 2014). In our study, correlations between $P(B_t < 0.4B_{MSY})$ and bias in steepness and M showed that there was only a relationship between conservation performance and bias for the MSY and SPR-based HCRs. Even though the magnitude of bias in steepness could be quite large under the Historical HCRs, there was little effect on conservation performance. This was because estimates of the Historical OCPs scaled in the same direction as the estimates of biomass. For the MSY and SPR-based HCRs for Pacific Cod, it was sometimes the case that estimates of biomass and OCPs were biased in opposite directions, resulting in overestimated biomass with underestimated OCPs.

Results were not necessarily consistent for both species. For the Pacific Cod DM scenario, very large overestimates of the Historical Cutoff and USR buffered against overestimation of biomass. However, AAV was high for Pacific Cod under the Historical MP, due to the more precautionary OCPs and frequent reductions in fishing. Conversely, for Rock Sole, the Historical MP was associated with low AAV, largely because the very low harvest rate resulted in more stable catches and less time on the “ramp” between the Cutoff and USR. For both species, management procedures that had the largest AAV were those that resulted in frequent closure of the fishery. For Pacific Cod, frequent closures occurred under the Historical MP because of large positive bias in the Cutoff and USR, creating assessments where the stock was assessed to be below the Cutoff when in fact it was well above the “true” OM LRP. Rock Sole had the highest AAV under the MSY MP. In this case, frequent closures occurred when assessed stock status was below the Cutoff, but in this case the stock actually was below the OM LRP, as a result of overharvesting due to large overestimates of spawning biomass. Stakeholders tend to give high priority to stable catches, reflecting the need to maintain a stable market share (Cox and Kronlund, 2008; Mapstone et al., 2008; Cox et al., 2013; Martell et al., 2015). The

large differences in performance of MPs with respect to AAV between the two species highlights that generic HCRs are not guaranteed to perform consistently in any specific context (Kronlund et al., 2014).

We have not attached absolute dollar values to the conservation-yield trade-offs we presented. Cox et al. (2013) evaluated the initial financial cost of implementing Canada’s provisional U80-40 HCR for the valuable Sablefish (*Anoplopoma fimbria*) fishery at CAN\$2.7–10 million y^{-1} , or 10–50% of current landed value. The trade-off was unacceptable for stakeholders and led to development of an alternative control rule that was able to maintain a high probability of keeping biomass above the LRP, but at a lower cost. Punt (2017) reviewed two strategies for evaluating trade-offs: (i) “trading-off”; and (ii) “satisficing”. Satisficing (Miller and Shelton, 2010) involves screening out MPs that do not meet some minimum, pre-specified standards for all or some of the performance metrics. Such pre-specified standards may be somewhat arbitrary (Punt, 2017). Alternatively, trading-off does not involve setting minimum standards, and instead relies on decision-makers and stakeholders attempting to find the best balance among performance metrics (Cox and Kronlund, 2008; Cox et al., 2013; Punt 2017). In either case, it is helpful if decision-makers and stakeholders can identify desirable ranges for all or some of the performance metrics. This is likely to be an iterative process as experience is gained (Cox et al., 2013; Martell et al., 2015).

The trade-off between short-term economic goals and long-term conservation objectives is ubiquitous in fisheries (Hilborn and Walters, 1992). One of the key characteristics of this trade-off is that, while short-term losses in yield can be predicted with relative certainty, long-term conservation and yield predictions are much less certain (Walters, 1998; Cox et al., 2013). For Rock Sole, very large conservation-yield trade-offs in the short-term were reduced in the longer term, reflecting stabilisation of catches at a lower level after a short period of fishing down the large standing biomass. The apparent large short-term catch losses under the Historical MP compared to the MSY- or SPR-based MPs were a result of the Historical MP not increasing fishing on a large standing stock. This is in contrast to the Sablefish example (Cox et al., 2013), in which short-term losses came from decreasing fishing from existing levels. Fishing down a large standing stock to the target B_{MSY} could be consistent with the guidelines in the PA Framework (DFO, 2009). However, our simulations showed that this strategy came with risk of implementation error resulting from large overestimates of biomass under the MSY and SPR_{30} MPs, especially for Rock Sole. Furthermore, while rapid fishing down of large standing stocks can certainly occur for valuable species (e.g., Branch, 2001), markets will dictate whether this is likely to occur in reality.

One of the most striking outcomes of our simulations was variation in MREs among MPs, especially for Pacific Cod, even within scenarios. For example, steepness was underestimated for all Pacific Cod MPs and scenarios except for the SPR_{40} MP, where it was overestimated. For Rock Sole, overestimation of biomass was greater under the MSY and SPR_{30} MPs than under the other MPs, with estimation errors most likely the result of lag effects in estimation of recruitment and stock size as spawning stock biomass decreased rapidly due to large increases in catch. Recruitment estimates can take many years to stabilize as more information about cohort strength becomes available in the age composition data (e.g., International Joint Technical Committee for Pacific Hake, 2013).

Productivity parameters are almost always confounded with each other and with parameters that determine the scale of the population (Walters, 1986). Steepness in particular is difficult to estimate without long, contrasting time series of catch and abundance data (Magnusson and Hilborn, 2007; Conn et al., 2010) or informative priors (Dorn, 2002; Forrest et al., 2010). Steepness is also difficult to disentangle from external drivers of recruitment (Walters and Collie, 1988; Walters,

1989). While many studies have found evidence for environmentally-driven productivity, correlations often break down over time (Myers et al., 1995; Myers, 1998). Indeed, studies that have considered multiple hypotheses for population productivity processes (e.g., density-dependent mortality vs environmentally-driven recruitment) have often been unable to distinguish among the alternative hypotheses (Skud, 1975; Fournier, 1983; Deriso et al., 1986; Walters et al., 1986; Walters and Parma, 1996; Tyler and Crawford, 1991). By fitting alternative assessment models to real data for Hecate Strait Pacific Cod, we were able to produce equally plausible stock assessment results with and without depensatory adult mortality, indeed producing very similar biomass trajectories. Reliable age-composition data are not available to resolve this issue for this stock.

The results of our study should be considered in the context of a number of caveats. Chiefly, we assumed perfect implementation of the estimated HCRs and did not account for deviations that could arise for a number of reasons. The BC groundfish trawl fishery has 100% at-sea and dockside monitoring (DFO, 2016), so under-reporting of catches is unlikely to occur. However, fishery managers can deviate from a HCR for economic or other reasons, especially if they are using a decision table to evaluate risk. Also, our assessment models were run every two years, with catch held constant between years. In reality, there are often several years between assessments. Catch decisions in BC are species-dependent, but quotas are often held at status quo between assessments. In reality, there may therefore be greater potential for implementation error as stock status estimates become out of date.

The simulated datasets provided to our MP assessment models were much better than those that are available in reality. Our MP assessment models were provided with biennial, fishery-independent survey observations from the beginning of the time series. In reality, fishery-independent surveys didn't begin in the Hecate Strait until 1983, and both stock assessments relied on commercial CPUE data (Forrest et al., 2015; Holt et al., 2016). While the AMs were conditioned on simulated catch data from the OM, we expect that providing them with shorter time series of survey data would have provided more variable results. We also made biennial commercial and survey age-composition data available for the whole time series, allowing for accurate estimation of fishery selectivity and recruitment strength. Because of the high quality simulated age-composition data, we have largely ignored the impact of errors in selectivity on OCPs and biomass estimates, even though fishery selectivity is a principle determinant of F_{MSY} and other key reference points (Forrest et al., 2008; Martell et al., 2008; Mangel et al., 2013). Future work on evaluating management procedures for Pacific Cod and Rock Sole would thus benefit from testing the data collection component of MPs.

Providing the assessment models with a long time series of age-composition data was perhaps more problematic for our Pacific Cod results. Pacific Cod is notoriously difficult to age (Beamish, 1981; Chilton and Beamish, 1982; Johnston and Anderl, 2012), and the last three Hecate Strait assessments have employed delay-difference models (Deriso, 1980) with the assumption of knife-edged selectivity and maturity at two years of age (Sinclair et al., 2001; Sinclair and Starr, 2005; Forrest et al., 2015). While we parameterised our age-structured assessment model identically to the delay-difference model used in the assessment, we have not investigated impacts of violating assumptions of knife-edged selectivity and maturity. An improvement to our analysis would therefore be to use a delay-difference model in the management procedure, under alternative assumptions about the selectivity and maturity ogives in the OM. Also, for both species, we conditioned our operating models on versions of the most recent assessments (Forrest et al., 2015; Holt et al., 2016). Our results are therefore strongly driven

by the current view of stock status.

We did not evaluate the performance of time-varying management procedures. These may be preferred if there is strong evidence of time-varying productivity, as they may be more responsive to high and low productivity cycles and prevent overharvesting (Peterman et al., 2000; Collie et al., 2012). Attempts to incorporate environmental correlates directly into stock assessments may be of little benefit (e.g., Walters, 1989; Basson, 1999; Kell et al., 2005) or even have a negative effect on management outcomes (e.g., A'mar et al., 2009; Haltuch and Punt, 2011). However, the use of dynamic OCPs such as average recruitment or "dynamic B_0 " may be beneficial (Haltuch et al., 2009; Punt et al., 2014).

5. Conclusions

Fisheries scientists have a role to play in helping managers and stakeholders select HCRs that will meet management needs and result in acceptable trade-offs between conservation and yield objectives (de la Mare, 1998). Our study and others indicate that the choice of OCPs in the HCR may be more important to achieving management objectives than attempting to capture complex biological and fishery dynamics (A'mar et al., 2009; Haltuch et al., 2009; Punt et al., 2008, 2014). Management strategy evaluation (MSE) is an effective approach that fosters engagement and can lead to more effective communication among scientists, managers and stakeholders. However, MSE in its full realisation can take several years and many resources to produce acceptable results, as scientists, managers and stakeholders go through an iterative process of developing objectives, reviewing results and refining procedures (Cochrane et al., 1998; Smith et al., 1999; Cox and Kronlund, 2008; Mapstone et al., 2008; Martell et al., 2015). While it has many benefits, full MSE is perhaps too resource-intensive for application in all fisheries. Closed-loop simulation is the technical component of MSE and has been used to great effect on its own to explore many aspects of the performance of fishery management procedures (e.g., Walters, 1989; Punt et al., 2008; Haltuch and Punt, 2011; Wetzel and Punt, 2011; Carruthers et al., 2014). We suggest that the tools are now available to develop simple closed-loop simulation evaluations of the type in this study that can be routinely presented along with stock assessment advice. We suggest that this would be a valuable allocation of resources that would improve on current practices of leaving managers on their own to choose from alternative forms of advice that were intended to demonstrate parameter sensitivity. Shifting the focus of discussions from uncertainty *per se* to desired objectives and performance with respect to objectives would be an added benefit.

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Appendix A. Model description

Table A1

Model parameters used in the Operating Model (OM). Parameters with the same fixed values across OM scenarios are provided below. Parameters with values that differ across OM scenarios are marked with an asterisk * (see Table 1). Modified from Cox et al. (in press).

| Symbol | Description | Fixed Value | |
|------------------------------------|---|-------------------------------|--------------------------------|
| | | Pacific Cod | Rock Sole |
| Fixed parameters | | | |
| t | Time step (y) | | |
| a | Age-class (y) | | |
| t_1 | First year of historical period | 1956 | 1945 |
| t_{MP} | Year in which the management procedure begins | 2014 | 2014 |
| t_T | Year in which the simulation ends | 2063 | 2063 |
| A | Number of age-classes | 10 | 12 |
| B_0 | Unfished spawning biomass ('000 metric tons) | * | * |
| h | Recruitment function steepness | * | * |
| M_t | Instantaneous natural mortality rate (/y) | * | * |
| L_∞ | Asymptotic length (cm) | 89.48 | 46.44 |
| L_1 | Mean length-at-age-1 (cm) | 23.20 | 7.62 |
| k | von Bertalanffy growth constant (/y) | 0.307 | 0.21 |
| g_1 | Scaling parameter of the length weight relationship | 7.4×10^{-6} | 6.6×10^{-6} |
| g_2 | Exponent of the length weight relationship | 3.10 | 3.18 |
| a_{50}^{mat} | Age-at-50% maturity (y) | 1.8 | * |
| a_{95}^{mat} | Age-at-95% maturity | 2.5 | * |
| $a_{50}^{sel,X}$ | Age-at-50% selectivity by survey ($X = S$) and fishery ($X = F$) | 1.8 | * |
| $a_{95}^{sel,X}$ | Age-at-95% selectivity by survey ($X = S$) and fishery ($X = F$) | 2.5 | * |
| q | Survey catchability coefficient | 1 | 0.2 |
| γ_R | Lag-1 autocorrelation in log-recruitment | 0 | * |
| $\tau_{t,t}$ | Survey coefficient of variation in year t | 0.2 | 0.2 |
| τ_P^X | Std. error of proportions-at-age in fishery catch ($X = F$) and surveys ($X = S$) | 0.2 | 0.2 |
| ε_t | Uncorrelated Normal (0,1) error in log-survey | | |
| $\eta_{a,t}^X$ | Uncorrelated Normal(0,1) error in logistic-transformed proportions-at-age | | |
| $x_{a,t}^X$ | Zero-centred log-residual of proportion-at-age | | |
| $p_{a,t}^X$ | Observed proportion-at-age a in year t catch | | |
| ρ | Observation error proportion of total error variance (errors-in-variables) | 0.05 | 0.25 |
| σ_C | Standard deviation in catch residuals | 0.05 | 0.05 |
| Derived quantities | | | |
| R_0 | Unfished recruitment | | |
| m_a | Proportion mature-at-age a | | |
| s_a^X | Proportion selected-at-age by survey ($X = S$) and fishery ($X = F$) | | |
| w_a | Individual weight-at-age a | | |
| l_a | Individual length-at-age a | | |
| $N_{a,t}$ | Number of age a fish in year t | | |
| $B_{a,t}$ | Biomass of age a fish in year t | | |
| B_t^{Sp} | Spawning biomass in year t (B_t in the main text) | | |
| B_t^{Ex} | Exploitable biomass in year t | | |
| ω_t^R | AR-correlated error in log-recruitment | | |
| $C_{a,t}$ | Number of age a fish in year t catch | | |
| C_t | Fishery catch numbers | | |
| $u_{a,t}$ | True proportion-at-age a in time t catch | | |
| I_t | Survey biomass estimate in time t | | |
| $x_{a,t}^X$ | Zero-centred log-residual of proportion-at-age a | | |
| $p_{a,t}^X$ | Observed proportion-at-age a in year t catch | | |
| ϕ_x | Equilibrium yield ($x = y$) or spawning biomass ($x = ssb$) per recruit | | |
| AM Prior Probability Distributions | | | |
| h | $\sim \text{Beta}(\mu, \sigma)$ | $\alpha = 3.54, \beta = 2.12$ | $\alpha = 43.05, \beta = 9.93$ |
| M | $\sim \text{Normal}(\mu, \sigma)$ | $\mu = 0.4, \sigma = 0.15$ | $\mu = 0.25, \sigma = 0.15$ |
| $\text{Log}(q)$ | $\sim \text{Normal}(\mu, \sigma)$ | $\mu = 0, \sigma = 1$ | $\mu = 0, \sigma = 1$ |
| F_t | $\sim \text{Normal}(\mu, \sigma)$ | $\mu = 0.2, \sigma = 5$ | $\mu = 0.2, \sigma = 5$ |

*See Table 1 for scenario-specific values.

Table A2

Equations defining the operating model (OM). Model equations modified from Cox et al. (2013). Note that only Rock Sole numbers-at-age were initialised at unfished equilibrium (A.2.7 and A.2.8). Pacific Cod numbers-at-age were initialised using initial numbers-at-age from the external stock assessment used to condition the OM. Model equations modified from Cox et al. (in press).

| Eq. | Formula |
|--|---|
| Parameters | |
| A.2.1 | $\Theta = (B_0, h, M_t, \gamma_R, \sigma_R, q, \tau_I, \tau_P^X, L_\infty, L_1, k, g_1, g_2, a_{50}^{mat}, a_{95}^{mat}, a_{50}^{sel,F}, a_{95}^{sel,F}, a_{50}^{sel,S}, a_{95}^{sel,S})$ |
| Fixed life history schedules | |
| A.2.2 | $m_a = \frac{1}{1 + \exp[-\log(19)(a - a_{50}^{mat}) / (a_{95}^{mat} - a_{50}^{mat})]}$ |
| A.2.3 | $s_a^{X=F,S} = \frac{1}{1 + \exp[-\log(19)(a - a_{50}^{sel,X}) / (a_{95}^{sel,X} - a_{50}^{sel,X})]}$ |
| Stock-recruitment parameters and equilibrium population | |
| A.2.4 | $R_0 = B_0 / \phi_{ssb}$ |
| A.2.5 | $a = \frac{4hR_0}{B_0(1-h)}$ |
| A.2.6 | $b = \frac{5h-1}{B_0(1-h)}$ |
| A.2.7 | $N_{a,1} = R_0 e^{-M_1(a-1)} \quad 1 \leq a \leq A-1$ |
| A.2.8 | $N_{A,1} = N_{A-1,1} / (1 - e^{-M_1})$ |
| A.2.9 | $B_{a,1} = N_{a,1} w_{a,1}$ |
| State dynamics | |
| A.2.10 | $\omega_t^R = \begin{cases} \frac{\sigma_R}{\sqrt{1-\gamma_R^2}} \delta_t^R & t = 1 \\ \gamma_R \omega_{t-1}^R + \sigma_R \delta_t^R & t > 1 \end{cases}$ |
| A.2.11 | $M_t = \begin{cases} M & \text{All OM scenarios except DM} \\ M_0 + (M_1 - M_0) \left(1 - \frac{B_t^{Tot}}{B_0^{Tot}}\right) & \text{DM scenario only (see Appendix B)} \end{cases}$ |
| A.2.12 | $l_{a,t} = L_{\infty,t-a} + (L_{1,t-a} - L_{\infty,t-a}) e^{(-k_l - a(a-1))}$ |
| A.2.13 | $w_{a,t} = g_1 l_{a,t}^{g_2}$ |
| A.2.14 | $N_{1,t} = \frac{a_{B_t^{Sp}}}{1 + b_{B_t^{Sp}}} \exp[\omega_t^R - 0.5\sigma_R^2 / (1 - \gamma_R^2)]$ |
| A.2.15 | $N_{a,t} = N_{a-1,t-1} e^{-M_{t-1} + s_a^F F_{t-1}} \quad 2 \leq a \leq A-1$ |
| A.2.16 | $N_{A,t} = N_{A-1,t-1} e^{-M_{t-1} + s_A^F F_{t-1}} + N_{A,t-1} e^{-M_{t-1} + s_A^F F_{t-1}}$ |
| A.2.17 | $B_t^{Sp} = \sum_{a=1}^A m_a w_{a,t} N_{a,t}$ |
| A.2.18 | $B_t^{Ex} = \sum_{a=1}^A s_a^S w_{a,t} N_{a,t}$ |
| A.2.19 | $B_t^{Tot} = \sum_{a=1}^A w_{a,t} N_{a,t}$ |
| A.2.20 | $C_{a,t} = \frac{s_a^F F_t}{M_t + s_a^F F_t} \left(1 - e^{-M_{t-1} - s_a^F F_t}\right) N_{a,t} w_{a,t}$ |
| Survey and proportion-at-age observations | |
| A.2.21 | $I_t = q B_t^{Ex} \exp[\tau_{I,t} \varepsilon_t - 0.5\tau_{I,t}^2]$ |
| A.2.22 | $u_{a,t}^X = s_a^X B_{a,t} / \sum_{j=1}^A s_j^X B_{j,t}$ |
| A.2.23 | $x_{a,t}^X = \log u_{a,t}^X + \tau_P^X \eta_{a,t}^X - \frac{1}{A} \sum_{j=1}^A \left[\log u_{j,t}^X + \tau_P^X \eta_{j,t}^X \right]$ |
| A.2.24 | $p_{a,t}^X = \exp[x_{a,t}^X] / \sum_{j=1}^A \exp[x_{j,t}^X]$ |

^aSee Table 1 for M_0 and M_1 values.

Table A3

Equilibrium solutions for spawning biomass, \bar{B}^{Sp} , exploitable biomass, \bar{B}^{Ex} , and yield, \bar{Q} , given an equilibrium fishing mortality rate, \bar{F} . Top set of parameters, Θ , is used to calculate OM reference points. Elements of the parameter set, $\hat{\Theta}_T$ are estimates updated to time T by the AM – these are substituted for their OM counterparts to compute estimated equilibrium quantities B_0 , F_{MSY} and F_X as required by the HCRs. Values for F_{MSY} are obtained by numerically maximizing \bar{Q} with respect to \bar{F} . Values for F_X are obtained by numerically solving for the value of \bar{F} that results in $\phi_{ssb} = X\%$ of ϕ_{ssb0} , where ϕ_{ssb0} is obtained from Eq. A.3.4 with \bar{F} set to zero in Eq. A.3.3. In this study, $X = 40$ for Pacific Cod and $X = 30$ for Rock Sole (Section 2.2.4.2). Model equations modified from Cox et al. (in press).

| Eq. | Formula |
|--------------------|--|
| A.3.1 ^a | $\Theta = (a, b, m_{1:A}, s_{1:A}^F, w_{1:A}, M)$ |
| A.3.2 | $\hat{\Theta}_T = (\hat{a}_T, \hat{b}_T, \hat{s}_{1:A,T}^F, \hat{M}_T)$ |
| A.3.3 | $\ell_a = \begin{cases} 1 & a = 1 \\ \ell_{a-1} e^{(-M - s_{a-1}^F \bar{F})} & 2 \leq a < A \\ \ell_{A-1} e^{(-M - s_{A-1}^F \bar{F})} / (1 - e^{(-M - s_A^F \bar{F})}) & a = A \end{cases}$ |
| A.3.4 | $\phi_{ssb} = \sum_{a=1}^A \ell_a m_a w_a$ |
| A.3.5 | $\phi_{tb} = \sum_{a=1}^A \ell_a w_a$ |
| A.3.6 | $\bar{R} = (a\phi_{ssb} - 1)/b\phi_{ssb}$ |
| A.3.7 | $\bar{B}^{Sp} = \bar{R}\phi_{ssb}$ |
| A.3.8 | $\bar{B}^{Tot} = \bar{R}\phi_{tb}$ |
| A.3.9 | $\phi_y = \sum_{a=1}^A \frac{s_a^F \bar{F}}{M + s_a^F \bar{F}} (1 - e^{(-M - s_{a-1}^F \bar{F})}) \ell_a w_a$ |
| A.3.10 | $\bar{Q} = \bar{R}\phi_y$ |

^a For the depensatory Pacific Cod OM scenario (DM), the fished and unfished equilibrium values of M_t are used in all reference point calculations in the OM. See Appendix B for methodology.

Table A4

Catch-at-age assessment model (AM) quantities that differ from operating model values, where the hat symbol indicates parameters estimated by the AM. The tilde symbol indicates derived parameters or variables in the AM. The generic superscript “X” is used for selectivity because fishery F and survey S selectivity functions only differ in the parameters given in Table 1. Model equations modified from Cox et al. (in press).

| Eq. | Formula |
|--------|--|
| A.4.1 | $\hat{H}_T = \left(\hat{B}_0, \hat{h}, \hat{M}, \hat{F}_{1:t_T}, \hat{R}_{Avg}, \hat{R}_{AvgInit}, \hat{a}_{50}^{sel,F}, \hat{a}_{95}^{sel,F}, \hat{a}_{50}^{sel,S}, \hat{a}_{95}^{sel,S}, \hat{F}_{1:t_T}, \hat{\omega}_{2:A}^{Rinit}, \hat{\omega}_{1:t_T-a}^{Rmat} \right)$ |
| A.4.2 | $\hat{s}_a^X = \frac{1}{1 + \exp[-\log(19)(a - \hat{a}_{50}^{sel,X}) / (\hat{a}_{95}^{sel,X} - \hat{a}_{50}^{sel,X})]}$ |
| A.4.3 | $\hat{C}_{a,t} = \frac{\hat{s}_a^F \hat{F}_t}{\hat{M}_t + \hat{s}_a^F \hat{F}_t} \left(1 - e^{-\hat{M}_t - \hat{s}_a^F \hat{F}_t} \right) \hat{N}_{a,t} w_{a,t}$ |
| A.4.4 | $\hat{R}_0 = \frac{\hat{B}_0}{\sum_{a=1}^A \hat{e}_a m_a w_a}$ |
| A.4.5 | $\hat{a} = \frac{4\hat{h}\hat{R}_0}{\hat{B}_0(1 - \hat{h})}$ |
| A.4.6 | $\hat{b} = \frac{5\hat{h} - 1}{\hat{B}_0(1 - \hat{h})}$ |
| A.4.7a | $\hat{N}_{a,1} = \begin{cases} \hat{R}_0 a = 1 \\ \hat{R}_0 e^{-\hat{M}(a-1)} 2 \leq a \leq A \end{cases} \quad \text{Rock Sole}$ |
| A.4.7b | $\hat{N}_{a,1} = \begin{cases} \hat{N}_{a,1} = \hat{R}_{Avg} e^{\hat{\omega}_1^R - \frac{\sigma_R^2}{2}} & a = 1 \\ \hat{N}_{a,1} = (\hat{R}_{AvgInit} e^{\hat{\omega}_a^{Rinit}}) e^{-\hat{M}(a-1)} & 2 \leq a < A \\ \hat{N}_{a,1} = \frac{(\hat{R}_{AvgInit} e^{\hat{\omega}_a^{Rinit}}) e^{-\hat{M}(a-1)}}{(1 - e^{-\hat{M}})} & a = A \end{cases} \quad \text{Pacific Cod}$ |
| A.4.8 | $\hat{N}_{a,t} = \begin{cases} \hat{R}_{Avg} e^{\hat{\omega}_t^R - \frac{\sigma_R^2}{2}} & a = 1 \\ \hat{N}_{a-1,t-1} e^{-(\hat{F}_{a-1,t-1} + \hat{M}_{a-1,t-1})} & 1 \leq a < A \\ \hat{N}_{a-1,t-1} e^{-(\hat{F}_{a-1,t-1} + \hat{M}_{a-1,t-1})} + \hat{N}_{a,t} e^{-(\hat{F}_{a,t} + \hat{M}_{a,t})} & a = A \end{cases}$ |
| A.4.9 | $\hat{B}_t^{Sp} = \sum_{a=1}^A m_a w_{a,t} \hat{N}_{a,t}$ |
| A.4.10 | $\hat{B}_t^{Ex} = \sum_{a=1}^A s_a^S w_{a,t} \hat{N}_{a,t}$ |
| A.4.11 | $\hat{B}_t^{Tot} = \sum_{a=1}^A w_{a,t} \hat{N}_{a,t}$ |
| A.4.12 | $\hat{R}_t = \frac{\hat{a} \hat{B}_t^{Sp} - 1}{1 + \hat{a} \hat{B}_t^{Sp} - 1}$ |

Table A5

Components of the total negative log-posterior density function (G) given data up to time T . Negative log-likelihood functions for biomass index and recruitment (nll_{IR}), age-proportion data (nll_p) and catch (nll_c). The AM uses an errors-in-variables maximum likelihood approach for modeling the combined biomass index and process error likelihood (nll_{IR}) in which the total error variance (κ^2) is assumed to be the sum of observation error (τ_t^2) and age-1 recruitment process error (σ_R^2) components (Table A1). Assuming that the observation error proportion of this total (ρ) is known, the individual variance estimates are $\hat{\tau}_t^2 = \rho \hat{\kappa}^2$ and $\hat{\sigma}_R^2 = (1 - \rho) \hat{\kappa}^2$, where the estimate of the total variance $\hat{\kappa}^2$ is given by A.5.5. A robust normal likelihood (Fournier et al., 1998) for the age-proportion data (A.5.7) assuming sample sizes are all equal to an effective size $n = 100$ (Cox et al., 2015). The total negative log-posterior distribution function (A.5.14) includes prior probability distributions for h , M and q (A.5.9–A.5.11), as well as two penalty functions that: (i) constrain the estimates of annual recruitment to conform to a Beverton-Holt stock-recruit function (A.5.12.); and (ii) weakly constrain estimates of log fishing mortality to a normal distribution (A.5.13).

| Eq. | Formula |
|--------|--|
| A.5.1 | $z_t = \ln(I_t) - \ln\left(\hat{\tilde{B}}_t^{Ex}\right)$ |
| A.5.2 | $\widehat{\log q} = \frac{1}{T} \sum_{t=1}^T z_t$ |
| A.5.3 | $Z_I = \sum_{t=1}^{IT} (z_t - \widehat{\log q})^2$ |
| A.5.4 | $Z_R = \sum_{t=2}^{T-a_0^{mat}} (\hat{\omega}_t^R)^2$ |
| A.5.5 | $\hat{\kappa}^2 = \frac{1}{2IT + a_0^{mat} - 1} \left(\frac{Z_I}{\rho} - \frac{Z_R}{1-\rho} \right)$ |
| A.5.6 | $nll_{IR} = \frac{2IT + a_0^{mat} - 1}{2} \log(\hat{\kappa}^2)$ |
| A.5.7 | $nll_p^X = \sum_{t=1}^T \sum_{a=1}^A \log \left[\exp \left(-\frac{n(p_{a,t}^X - \hat{u}_{a,t}^X)^2}{2p_{a,t}^X(1 - p_{a,t}^X)} + 0.1/A \right) + 0.01 \right]$ |
| A.5.8 | $nll_C = \ln(\sigma_C^2) + \frac{(\ln(C_t) - \ln(\hat{C}_t))^2}{2\sigma_C^2}$ |
| A.5.9 | $nll_h = -[(\beta_1^h - 1) \log \hat{h} + (\beta_2^h - 1) \log(1 - \hat{h})]$ |
| A.5.10 | $nll_M = \frac{1}{2\sigma_M^2} (\hat{M}_1 - \mu_M)^2$ |
| A.5.11 | $nll_q = \frac{1}{2\sigma_q^2} (\hat{q} - \mu_q)^2$ |
| A.5.12 | $nll_R = \ln(\sigma_R^2) + \frac{\left(\ln(\hat{\tilde{N}}_{1,t}) - \ln(\hat{\tilde{R}}_t) \right)^2}{2\sigma_R^2}$ |
| A.5.13 | $nll_F = \ln(\sigma_F^2) + \frac{\left(\ln(\hat{\tilde{F}}_t) - \ln(\mu_F) \right)^2}{2\sigma_F^2}$ |
| A.5.14 | $G = nll_{IR} + nll_p^F + nll_p^S + nll_C + nll_h + nll_M + nll_q + nll_R + nll_F$ |

Appendix B. Reference points with density-dependent adult mortality

Fournier (1983) reported evidence for depensatory adult mortality for Hecate Strait Pacific Cod. In this study, we use a simple equation to model adult density-dependent M at time t

$$M_t = M_0 + (M_1 - M_0) \left(1 - \frac{B_t^{Tot}}{B_0^{Tot}} \right) \quad (B.1)$$

(Logan et al., 2005), where M_t is modelled as a function of the current total biomass B_t^{Tot} (Eq. A.2.19) relative to equilibrium unfished total biomass B_0^{Tot} , M_0 represents the bound of M_t when $B_t^{Tot} = B_0^{Tot}$, and M_1 represents the bound approached by M_t as $B_t^{Tot} \rightarrow 0$. Eq. (B.1) can be used to model density-dependent mortality when the relationship is compensatory (M_t decreases as B_t^{Tot} decreases; $M_1 < M_0$) or when the relationship is depensatory (M_t increases as B_t^{Tot} decreases; $M_1 > M_0$). In both compensatory and depensatory cases, M_t converges on M_0 when $B_t^{Tot} = B_0^{Tot}$ (Fig. B1a). Fig. B1b illustrates the effect on M_t of incorporating Eq. (1) into an age-structured stock assessment model tuned to catch, survey and mean weight data for Hecate Strait Pacific Cod. In this case, we assumed depensatory M with fixed values of $M_0 = 0.35$ and $M_1 = 0.43$ in Eq. (B.1). This resulted in an average $M_t = 0.4 y^{-1}$ across the time series (Fig. B1b).

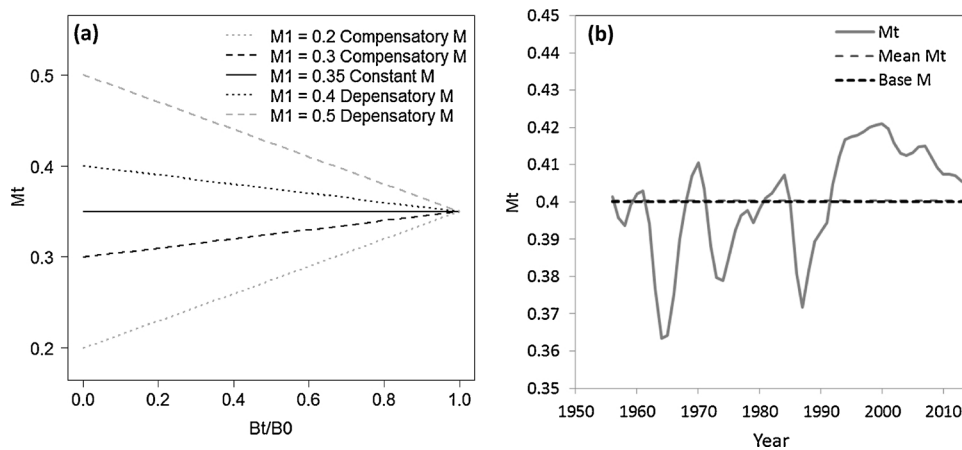


Fig. B1. a) Relationship between $\frac{B_t^{Tot}}{B_0^{Tot}}$ and M_t for the case where $M_0 = 0.35$ and two values of M_1 that result in a compensatory relationship ($M_1 = 0.2$ and 0.3), and two values of M_1 that result in a depensatory relationship ($M_1 = 0.4$ and 0.5). The case where $M_1 = M_0 = 0.35$ is also shown, resulting in constant M_t . b) Predicted M_t from the two age-structured models used to condition the Base (black line) and DM (grey line) OM scenarios for the closed loop simulation study (see Section 2.2.2.1). For the DM case, M_0 and M_1 values were fixed at 0.35 and 0.43 respectively in both the external age-structured model and in the *mseR* OM.

For calculation of reference points, Eq. (B.1) can be modified to represent density-dependent M when the stock is at equilibrium

$$\bar{M} = M_0 + (M_1 - M_0) \left(1 - \frac{\bar{B}_F^{Tot}}{\bar{B}_0^{Tot}} \right) \quad (B.2)$$

where \bar{M} is equilibrium natural mortality, \bar{B}_F^{Tot} is equilibrium total biomass under some level of constant long-term fishing mortality \bar{F} (Eq. A.3.8), where \bar{F} is applied in the survivorship function (Eq. A.3.3). Similarly, unfished equilibrium total biomass \bar{B}_0^{Tot} is obtained by setting $\bar{F} = 0$ (Eq. A.3.3). In the absence of density-dependent mortality, calculation of F_{MSY} is done straightforwardly by numerically maximizing \bar{Q} (Eq. A.3.10) with respect to \bar{F} .

Similarly, F_X is obtained in the OM by numerically solving for the value of \bar{F} that results in $\phi_{ssb} = X\%$ of ϕ_{ssb0} , where ϕ_{ssb0} is obtained from Eq. A.3.4 with $\bar{F} = 0$ in Eq. A.3.3. In the presence of density-dependent mortality (Eq. (B.2)), further steps are required to calculate reference points. Eq. A.3.3–A.3.10 indicate that an estimate of \bar{M} is required for the calculation of both F_{MSY} and F_X . Therefore, in the presence of density-dependent mortality, a solution must be obtained for \bar{M} at each step of the numerical search.

It should be apparent from Eq. (B.2) that there is no analytical solution for \bar{M} given a value of \bar{F} . The equilibrium depletion $\bar{D} = \frac{\bar{B}_F^{Tot}}{\bar{B}_0^{Tot}}$ is dependent on an estimate of \bar{M} , which is itself dependent on an estimate of \bar{D} , resulting in a circular problem. We therefore solved Eq. B.2 using a numerical algorithm that quickly converged on the solution for \bar{M} and \bar{D} , conditional on \bar{F} and the vector θ of life history and selectivity parameters. The pseudo-code is:

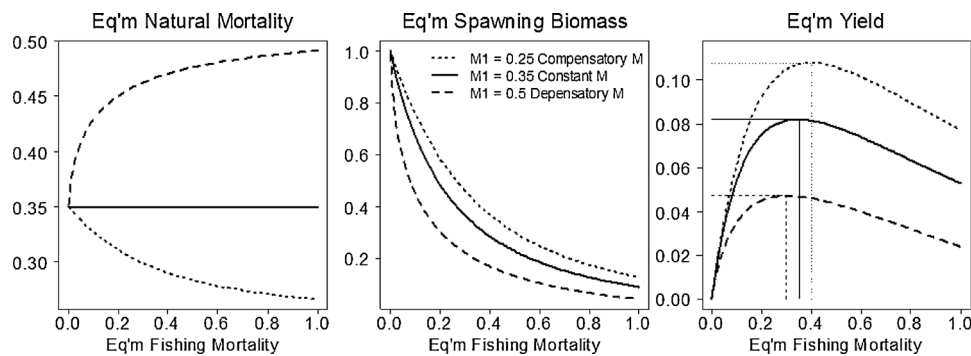


Fig. B2. Equilibrium values of \bar{M} , spawning biomass \bar{B}^{Sp} and yield \bar{Q} for the case where $M_0 = 0.35$ and one value of M_1 that results in a compensatory relationship ($M_1 = 0.25$; dotted lines), and one value of M_1 that results in a depensatory relationship ($M_1 = 0.5$; dashed lines). The case where $M_1 = M_0 = 0.35$ is also shown, resulting in constant \bar{M} (solid lines). Vertical and horizontal lines show F_{MSY} and MSY , respectively (right panel). Life history and selectivity parameters were fixed at values used for Pacific Cod in the OM (Table A1).

```

get_me <- function(  $\bar{F}$ ,  $\Theta$  ){
  #initial guess at equilibrium depletion
   $\bar{D} <- 0.4$ 
  for(i in 1:20)
  {
     $\bar{M} = M_0 + (M_1 - M_0)(1 - \bar{D})$ 
     $\bar{D}_1 = \frac{\bar{B}_0^{tot}}{\bar{B}_0^{tot}}$  # (Eq. A.3.3 - A.3.8 conditional on  $\bar{M}$  and  $\bar{F}$ )
    DiffDe =  $\bar{D} - \bar{D}_1$ 
     $\bar{D} = \bar{D}_1$ 
    if(abs(DiffDe) < 1.e-10) break;
  }
  #end for i
  return(  $\bar{M}$  )
}

```

We ran the above algorithm to find \bar{M} for a sequence of values of \bar{F} (0–1, increment 0.01). For each tested value of \bar{F} we were then able to calculate equilibrium spawning biomass per recruit (A.3.4) and equilibrium yield (A.3.10), and subsequently, F_X and F_{MSY} . Consequently, we calculated B_X and B_{MSY} using Eq. A.3.7 with $\bar{F} = F_X$ or $\bar{F} = F_{MSY}$ in Eq. A.3.3, respectively. We verified that the F_{MSY} solution indeed maximised long-term catch by comparing results with those obtained by running out an age-structured model with density-dependent mortality (Eq. (B.1)) for 100 years under a sequence of constant values of \bar{F} (0–1, increment 0.01) and obtained identical results for all equilibrium quantities.

Results are shown in Fig. B2 for Pacific Cod with M_0 set at 0.35 and three alternative values of M_1 resulting in compensatory (dotted lines), constant (solid lines) and depensatory (dashed lines) mortality. When mortality is compensatory, \bar{M} decreases as \bar{F} increases, whereas the opposite is true when the relationship is depensatory (dashed lines) (Fig. B2, left panel). While \bar{B}^{Sp} decreases with increased \bar{F} in all three scenarios, the rate of decrease is fastest when mortality is depensatory (Fig. B2, centre panel), a result of increased \bar{M} (left panel). Therefore, the effects of increasing fishing mortality are exacerbated by increased M in the presence of depensatory mortality, rather than mitigated when mortality is compensatory. All other parameters equal, this translates into lower F_{MSY} and MSY in the presence of depensatory mortality (Fig. B2, right panel). Note that the DM scenario used in the closed-loop simulations resulted in higher F_{MSY} than the base scenario (Fig. 3). This was because steepness in the DM scenario was considerably higher than the Base scenario (Table 1). The higher steepness resulted from the conditioning step, where a stock assessment model parameterised with depensatory mortality was fit to real catch and survey data for Hecate Strait Pacific Cod.

Appendix C

Table C1

Median long-term conservation performance statistics for Pacific Cod and Rock Sole (Section 2.2.3; see also Fig. 6); and short- and long-term catch statistics (Average Catch, AAV) (see also Figs. 7 and 8).

| Pacific Cod | Harvest Control Rule | B_t/B_0 (Long-term) | B_t/B_{MSY} (Long-term) | $P(B_t < 0.4B_{MSY})$ (Long-term) | Avg Catch (Short-term) | AAV (Short-term) | Avg Catch (Long-term) | AAV (Long-term) |
|-------------|----------------------|-----------------------|---------------------------|-----------------------------------|------------------------|------------------|-----------------------|-----------------|
| Base | MSY | 0.226 | 0.620 | 0.160 | 2.015 | 20.109 | 1.952 | 16.621 |
| Base | SPR ₄₀ | 0.182 | 0.498 | 0.300 | 2.531 | 25.384 | 2.150 | 19.202 |
| Base | Hist | 0.293 | 0.802 | 0.010 | 1.683 | 28.879 | 1.492 | 28.945 |
| Base | $B_0 F_{MSY}$ | 0.253 | 0.692 | 0.110 | 1.822 | 20.765 | 1.804 | 19.407 |
| Base | $B_0 F_{Hist}$ | 0.256 | 0.701 | 0.080 | 1.798 | 19.742 | 1.766 | 17.768 |
| DM | MSY | 0.179 | 0.662 | 0.150 | 2.638 | 24.030 | 2.321 | 19.256 |
| DM | SPR ₄₀ | 0.173 | 0.638 | 0.210 | 3.030 | 28.336 | 2.404 | 20.096 |

(continued on next page)

Table C1 (continued)

| Pacific Cod | Harvest Control Rule | B_t/B_0 (Long-term) | B_t/B_{MSY} (Long-term) | $P(B_t < 0.4B_{MSY})$ (Long-term) | Avg Catch (Short-term) | AAV (Short-term) | Avg Catch (Long-term) | AAV (Long-term) |
|------------------|----------------------|-----------------------|---------------------------|-----------------------------------|------------------------|------------------|-----------------------|-----------------|
| DM | Hist | 0.300 | 1.109 | 0.000 | 1.844 | 29.533 | 1.689 | 29.502 |
| DM | B_0F_{MSY} | 0.228 | 0.842 | 0.080 | 2.431 | 26.290 | 2.129 | 21.880 |
| DM | B_0F_{Hist} | 0.253 | 0.935 | 0.030 | 2.231 | 17.670 | 2.038 | 16.645 |
| Rock Sole | | | | | | | | |
| Base | MSY | 0.222 | 0.915 | 0.190 | 2.681 | 24.396 | 1.557 | 14.810 |
| Base | SPR ₃₀ | 0.287 | 1.185 | 0.090 | 2.596 | 22.826 | 1.612 | 10.252 |
| Base | Hist | 0.605 | 2.494 | 0.000 | 1.373 | 7.585 | 1.130 | 9.060 |
| Base | B_0F_{MSY} | 0.366 | 1.511 | 0.010 | 2.538 | 28.320 | 1.572 | 16.681 |
| Base | B_0F_{Hist} | 0.583 | 2.404 | 0.000 | 1.407 | 5.830 | 1.229 | 2.629 |
| LowM | MSY | 0.219 | 0.870 | 0.240 | 1.962 | 20.893 | 1.035 | 14.884 |
| LowM | SPR ₃₀ | 0.271 | 1.076 | 0.110 | 1.909 | 22.359 | 1.118 | 11.790 |
| LowM | Hist | 0.477 | 1.892 | 0.000 | 1.343 | 8.750 | 1.049 | 10.492 |
| LowM | B_0F_{MSY} | 0.376 | 1.493 | 0.010 | 1.850 | 25.241 | 1.115 | 14.298 |
| LowM | B_0F_{Hist} | 0.468 | 1.859 | 0.000 | 1.399 | 6.443 | 1.088 | 6.216 |

Table C2

Short and long-term Median Relative Errors (MRE) for Pacific Cod and Rock Sole for the estimated parameters M , h , q , B_0 for the five HCRs under each OM scenario. MRE measures the median percent difference between the estimated value of a parameter or state variable in the MP and its OM value for the time period (Eq. (33)).

| Scenario | Pacific Cod | Harvest control rule | Short-Term | | | | Long-Term | | | |
|------------------|-------------|----------------------|------------|--------|--------|-------|-----------|--------|--------|--------|
| | | | M | h | q | B_0 | M | h | q | B_0 |
| Base | | MSY | 3.52 | −21.02 | −24.20 | 20.62 | −1.86 | −9.74 | −12.61 | −3.86 |
| Base | | SPR ₄₀ | 2.67 | −5.26 | −21.34 | 7.00 | −3.06 | 34.47 | −9.09 | −23.29 |
| Base | | Hist | 4.27 | −21.77 | −25.46 | 20.80 | 1.68 | −14.13 | −19.86 | −1.78 |
| Base | | B_0F_{MSY} | 3.89 | −21.71 | −25.26 | 20.82 | −0.63 | −12.59 | −14.95 | −1.13 |
| Base | | B_0F_{Hist} | 4.39 | −21.88 | −25.65 | 21.33 | 0.38 | −13.77 | −17.55 | −1.43 |
| DM | | MSY | 2.32 | −29.36 | −27.43 | 7.13 | −5.36 | −18.93 | −10.15 | −13.01 |
| DM | | SPR ₄₀ | 1.72 | −15.79 | −26.40 | −1.79 | −4.64 | 5.11 | −11.39 | −21.51 |
| DM | | Hist | 7.04 | −31.77 | −33.26 | 10.87 | 3.26 | −28.6 | −26.23 | −4.56 |
| DM | | B_0F_{MSY} | 3.68 | −30.53 | −27.99 | 8.73 | −3.3 | −24.81 | −13.24 | −9.59 |
| DM | | B_0F_{Hist} | 5.70 | −32.05 | −32.14 | 11.34 | −0.49 | −28.12 | −17.57 | −5.25 |
| Rock Sole | | | | | | | | | | |
| Base | | MSY | 7.48 | 2.97 | −7.72 | 13.30 | −3.55 | 4.06 | 4.74 | 5.83 |
| Base | | SPR ₃₀ | 7.27 | 2.36 | −7.77 | 13.37 | −3.34 | 2.00 | 5.00 | 6.14 |
| Base | | Hist | 9.33 | −1.59 | −10.73 | 16.02 | 3.73 | −1.33 | 0.25 | 5.43 |
| Base | | B_0F_{MSY} | 7.57 | 1.29 | −7.65 | 13.28 | −2.60 | −0.24 | 4.52 | 7.07 |
| Base | | B_0F_{Hist} | 9.34 | −1.59 | −10.76 | 16.02 | 0.41 | −1.49 | 8.10 | 2.60 |
| LowM | | MSY | 13.13 | 1.67 | −12.35 | 12.44 | −3.48 | 4.37 | 3.93 | 9.73 |
| LowM | | SPR ₃₀ | 12.77 | 1.74 | −11.93 | 12.33 | −4.68 | 2.29 | 4.82 | 9.66 |
| LowM | | Hist | 17.38 | −3.72 | −17.89 | 16.41 | 5.99 | −3.07 | −2.43 | 8.53 |
| LowM | | B_0F_{MSY} | 13.38 | −0.46 | −12.02 | 13.14 | −2.60 | −2.18 | 4.11 | 10.06 |
| LowM | | B_0F_{Hist} | 17.38 | −3.69 | −17.91 | 16.21 | −0.92 | −3.53 | 6.45 | 7.78 |

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