



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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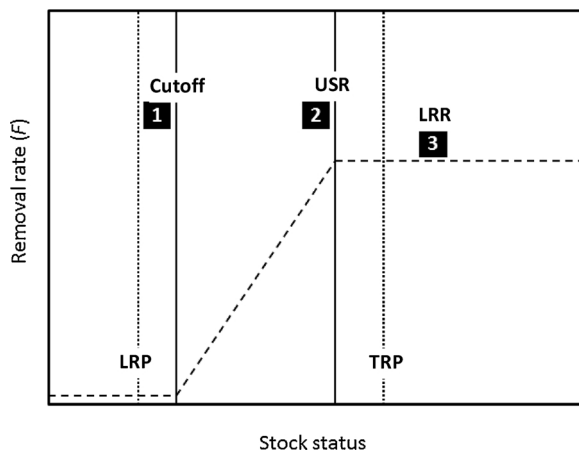


Fig. 1. Illustration of the PA Framework (DFO, 2006a, 2009) with solid vertical lines indicating two OCPs: (1) the “Cutoff”; and (2) the upper stock reference (USR). The removal rate (fishing mortality) in each of the zones is indicated by a dashed line, with its maximum defined as (3) the limit removal 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

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