

In what direction should the fishing mortality target change when natural mortality increases within an assessment?

Christopher M. Legault and Michael C. Palmer

Abstract: Traditionally, the natural mortality rate (M) in a stock assessment is assumed to be constant. When M increases within an assessment, the question arises how to change the fishing mortality rate target (F_{Target}). Per recruit considerations lead to an increase in F_{Target} , while limiting total mortality leads to a decrease in F_{Target} . Application of either approach can result in nonsensical results. Short-term gains in yield associated with high F_{Target} values should be considered in light of potential losses in future yield if the high total mortality rate leads to a decrease in recruitment. Examples using yellowtail flounder (*Limanda ferruginea*) and Atlantic cod (*Gadus morhua*) are used to demonstrate that F_{Target} can change when M increases within an assessment and to illustrate the consequences of different F_{Target} values. When a change in M within an assessment is contemplated, first consider the amount and strength of empirical evidence to support the change. When the empirical evidence is not strong, we recommend using a constant M . If strong empirical evidence exists, we recommend estimating F_{Target} for a range of stock-recruitment relationships and evaluating the trade-offs between risk of overfishing and forgone yield.

Résumé : Traditionnellement, le taux de mortalité naturelle (M) dans une évaluation de stock est présumé être constant. Quand M augmente dans une évaluation, cela soulève la question à savoir comment changer le taux cible de mortalité par pêche (F_{Target}). Des considérations relatives aux recrues individuelles mènent à une augmentation de F_{Target} , alors que le fait de limiter la mortalité totale mène à une réduction de F_{Target} . L'application d'une ou l'autre de ces approches peut se traduire par des résultats qui n'ont aucun sens. Les augmentations à court terme du rendement associées à des valeurs de F_{Target} élevées devraient être évaluées par rapport aux réductions potentielles du rendement futur si le taux de mortalité totale élevé se traduit par une baisse du recrutement. Des exemples basés sur la limande à queue jaune (*Limanda ferruginea*) et la morue franche (*Gadus morhua*) sont utilisés pour démontrer que F_{Target} peut changer quand M augmente dans une évaluation et pour illustrer les conséquences de différentes valeurs de F_{Target} . Quand la possibilité de changer M dans une évaluation est examinée, il faut d'abord prendre en compte la quantité et la force des preuves empiriques qui appuient ce changement. Si ces preuves ne sont pas fortes, nous recommandons d'utiliser un M constant. Si les preuves empiriques sont fortes, nous recommandons d'estimer F_{Target} pour une gamme de relations stock-recrutement et d'évaluer les compromis entre le risque de surpêche et le rendement non réalisé. [Traduit par la Rédaction]

Introduction

The natural mortality rate (M) in a stock assessment is a difficult parameter to estimate because it is not observed and is often confounded with other parameters within the model (Vetter 1988; Thompson 1994; He et al. 2011; although see Magnusson and Hilborn 2007; Lee et al. 2011). Because of these difficulties in estimation, it has traditionally been assumed to be constant over years and ages. This has always been recognized as a simplifying assumption to allow other parameters to be estimated more precisely (Brodziak et al. 2011; Deroba and Schueller 2013). The impacts of this assumption are small when M is small relative to the total mortality rate (Z), but become more important as M/Z increases.

More recently the assumption of constant M has been relaxed for a number of reasons. The impact of disease outbreaks has been considered through estimated or assumed changes in M (Patterson 1996; Marty et al. 2003; Wilberg et al. 2011). A number of Canadian Atlantic cod (*Gadus morhua*) stock assessments allow M to increase over time based on the lack of stock recovery under low fishing pressure (Chaput 2011). The Atlantic herring (*Clupea harengus*) stock in United States waters assumed a 50% increase in M to account for large increases in estimated consumption by a

dozen finfish species (NEFSC 2012b). More generally, multispecies models allow M to change as the abundance of predators and prey change (Curti et al. 2013). Advances in computing have allowed estimation of M as a random walk over time (Jiao et al. 2012; Swain and Mohn 2012; Swain and Benoît 2015). Additionally, changes in M have been used to address retrospective patterns (Legault 2009), for example in the eastern Georges Bank Atlantic cod assessment (Wang and O'Brien 2012), the 2012 Georges Bank yellowtail flounder (*Limanda ferruginea*) assessment (Legault et al. 2012), and the Gulf of Maine Atlantic cod assessment (NEFSC 2013).

Changes in M over time complicate the estimation of biological reference points because of the common assumption of equilibrium conditions. Biological reference points typically account for changes in the fishery selectivity pattern and biological parameters such as weights at age and maturity (NEFSC 2008). These changes in fishery selectivity and biological conditions lead to changes in both the fishing intensity and the target population size and have been well-understood since the time of Beverton and Holt (1957). The consequences for biological reference points when M changes are not as clear, as evidenced by the difficulties in estimating biological reference points in integrated ecosystem models (Möllmann et al. 2014). While multispecies modeling

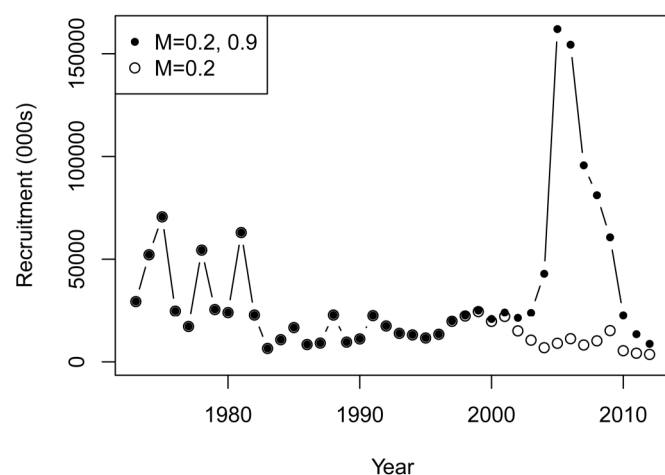
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C.M. Legault and M.C. Palmer. National Marine Fisheries Service, Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA 02543, USA.

Corresponding author: Christopher M. Legault (email: chris.legault@noaa.gov).

Fig. 1. Estimated recruitment for the Georges Bank yellowtail flounder example with a sudden increase in M from 0.2 to 0.9 year⁻¹ between 2004 and 2005 (solid circles) and M equal to 0.2 year⁻¹ for all years (open circles). Note the recruitment estimates are identical for years 1973 through 1995.



approaches have been used to estimate biological reference points that change with estimated changes in natural mortality (e.g., Gislason 1999; Collie and Gislason 2001; Overholtz et al. 2008), these models have not so far been widely used for management purposes.

A common set of biological reference points are those associated with maximum sustainable yield (MSY; Mace 2001). These typically are computed combining equilibrium per recruit analysis with a stock production function, the stock–recruitment (SR) relationship. When the SR relationship is poorly defined, often owing to a limited range of observations, proxies are used for the biological reference points. A common fishing intensity proxy is $F_{X\%}$, the fishing mortality rate that decreases per recruit spawning stock biomass (SSB) to $X\%$ of the unfished per recruit SSB, with X often set to 30, 35, or 40 (Clark 1991, 2002).

Conceptually there are two approaches to deal with an increase in M when estimating the target fishing mortality rate (F_{Target}). One is to increase F_{Target} based on per recruit considerations because a given cohort of fish will die off quicker owing to the increased M , and so fishing intensity should be increased to catch the fish before they die. An alternative approach is to decrease F_{Target} when M increases to maintain the total amount of mortality experienced by the stock because the stock can withstand only so many removals before productivity is impacted. These alternatives are explored in this paper through the use of two examples. The first example computes the MSY biological reference points for a range of M and SR curves when M is increased in the Georges Bank yellowtail flounder assessment to demonstrate how both increases and decreases in F_{Target} can occur. The second example examines the short- and long-term catch advice consequences of choosing to increase or decrease F_{Target} when M is increased in the Gulf of Maine Atlantic cod assessment. Based on these examples and our experience, we conclude with a set of considerations for stock assessments regarding changing M .

MSY reference points: Georges Bank yellowtail flounder example

Georges Bank yellowtail flounder stock assessment models have been plagued by retrospective problems since 2005 (see table 19 in Legault et al. 2014). Estimates of SSB in the terminal year of the assessment have decreased substantially as additional years of data have been added, while fishing mortality rates have increased substantially. The 2012 assessment conducted a thorough

Table 1. Georges Bank yellowtail flounder life history and fishery parameters used in calculations (mean of values for years 2008–2012).

Age	Catch weight (kg)	Stock weight (kg)	Maturity	Selectivity
1	0.138	0.099	0.000	0.003
2	0.326	0.203	0.462	0.078
3	0.439	0.372	0.967	0.489
4	0.540	0.482	1.000	1.000
5	0.670	0.600	1.000	1.000
6+	0.850	0.850	1.000	1.000

Note: Time of spawning for SSB calculations is 0.4167 of the year.

examination of the timing and magnitude of change in the amount of catch or the natural mortality rate (or both) needed to eliminate the retrospective pattern within the virtual population analysis model (Legault et al. 2012). The search found that a number of combinations could essentially eliminate the retrospective pattern. For this paper, the natural mortality rate change selected for demonstration purposes sets $M = 0.2$ year⁻¹ for years 1973–2004 and $M = 0.9$ year⁻¹ for years 2005–2012 using the data from the 2013 assessment (Legault et al. 2013). While this magnitude of change in M may be hard to believe, similar changes in magnitude for other stocks have been estimated (Swain and Benoit 2015), and it reflects the magnitude of the retrospective problem plaguing this assessment. This example is provided for demonstration purposes only and is not meant to replace the official assessment.

Increasing the natural mortality rate reduced the retrospective error, expressed as rho values (Mohn 1999), from 369% to -5% for SSB and from -81% to -9% for F . The large change in the natural mortality rate between 2004 and 2005 had a substantial impact on the recruitment estimates in recent years because the high M requires an increase in recruitment (Fig. 1). Plotting $F = 0$ replacement lines associated with M values ranging from 0.2 to 0.9 year⁻¹ derived from life history information for this stock (Table 1) shows most of the estimated SR pairs above the low M replacement line and below the high M replacement line (Fig. 2). The location of the SR pairs relative to the replacement lines cannot be used to determine an appropriate spawning potential ratio proxy for the fishing mortality rate associated with MSY (F_{MSY} ; Legault and Brooks 2013), but are important when used with SR curves to determine equilibrium conditions. A range of Beverton–Holt SR curves were estimated for the stock and recruitment pairs with nearly equal fits (Table 2; Fig. 2). The Beverton–Holt SR curve was defined as

$$(1) \quad R = \frac{\alpha S}{\beta + S}$$

where R denotes recruitment and S denotes spawning stock biomass. Combining the SR curves with the range of M values allows estimation of the steepness associated with each SR curve and M as well as direct estimates of F_{MSY} values. Note that a number of the SR curves do not intersect with replacement lines associated with high M values (e.g., dotted line Fig. 2A), thus resulting in no positive equilibrium solutions for MSY reference points.

The SR curves were defined by fixing $M = 0.2$ year⁻¹ and steepness at values from 0.65 to 0.90 in steps of 0.05 and finding the unfished recruitment that minimized the residual sum of squares. These curves were then converted to the alpha and beta version of the Beverton–Holt curve (eq. 1) and held fixed while M was allowed to change. Many other curves could have been fit to these SR pairs with nearly equal fits. This approach was chosen to allow a simple ordering of the SR curves. The steepness values associated with a

Fig. 2. Georges Bank yellowtail flounder SR relationship (points) with corresponding unfished replacement lines for a range of natural mortality rates (A) and fitted Beverton–Holt SR curves (B), based on steepness values of 0.65 to 0.90 in increments of 0.05 when $M = 0.2 \text{ year}^{-1}$. The parameters for the six SR curves are provided in Table 2. [Colour online.]

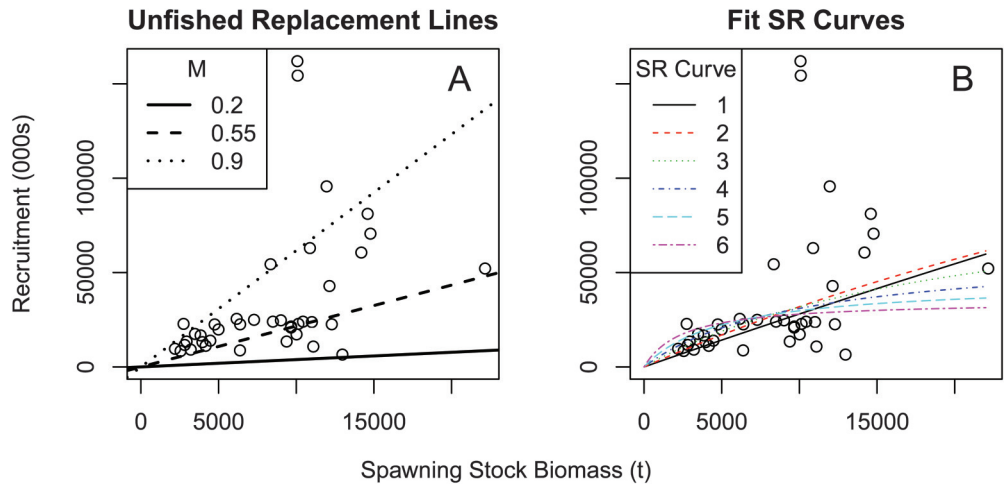


Table 2. Parameters of the Beverton–Holt SR relationship and residual sum of squares (RSS) for six different curves fit to the Georges Bank yellowtail flounder example.

SR curve	Alpha	Beta	RSS	Steepness ($M = 0.2 \text{ year}^{-1}$)
1	1 040 000	360 899	17.81	0.65
2	268 800	74 242	16.36	0.70
3	100 364	21 560	16.14	0.75
4	61 867	9 968	16.61	0.80
5	44 985	5 116	17.70	0.85
6	34 971	2 504	19.35	0.90

Note: These values for alpha and beta were derived from the assumption of $M = 0.2 \text{ year}^{-1}$ and steepness = 0.65 to 0.90 in increments of 0.05.

given SR curve decrease as M increases (Fig. 3A). This is a result of the definition for steepness, the proportion of unfished recruitment at one-fifth the unfished SSB, and the shape of Beverton–Holt SR curves. As M increases, the unfished point on the SR curve is reduced, but the reduction in the recruitment at one-fifth unfished SSB is greater than the associated reduction in unfished R , causing steepness to decrease. Given eq. 1, steepness (τ) is defined as the ratio of recruitment at one-fifth unfished SSB (denoted S_0) to unfished recruitment:

$$\tau = \frac{\frac{0.2\alpha S_0}{\beta + 0.2S_0}}{\frac{\alpha S_0}{\beta + S_0}} = \frac{\beta + S_0}{5\beta + S_0}$$

For a given value of β , steepness has the following properties:

$$\lim_{S_0 \rightarrow 0} \tau = 0.2 \text{ and } \lim_{S_0 \rightarrow \infty} \tau = 1$$

As S_0 decreases, steepness decreases. All other things held constant, when natural mortality increases, S_0 decreases, due to fewer individuals surviving to contribute to SSB. Thus, the use of the steepness formulation for the Beverton–Holt curve is problematic when M changes within an assessment because a given SR curve has different steepness values for differing M values (e.g., Forrester et al. 2010). Application of meta-analysis to estimate steepness from similar stocks will be difficult as well if the natural mortality rate is changing within the assessment time period, because the

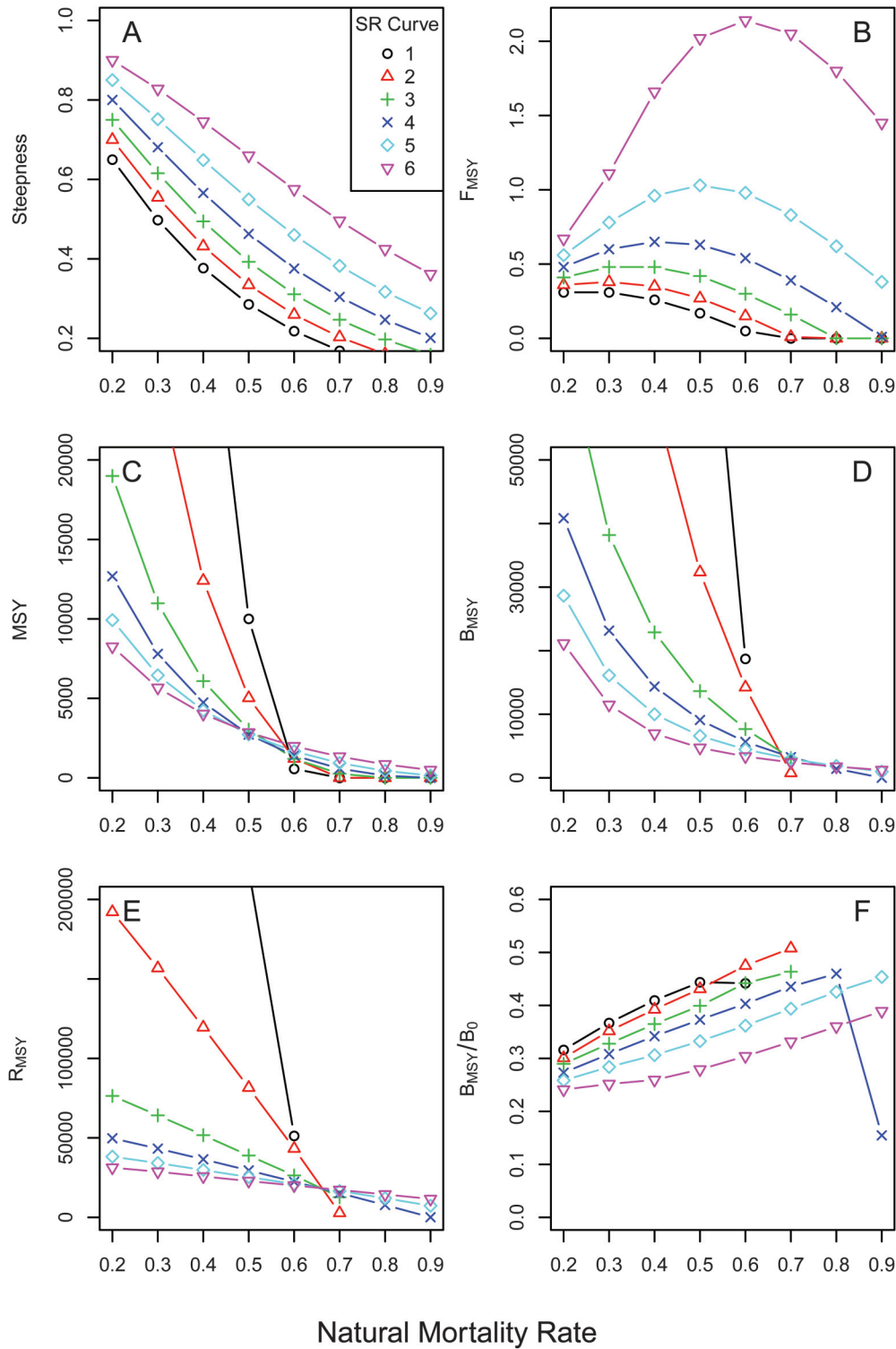
value of M to use in the meta-analysis will not be clearly defined, especially when the change has a trend over time. Conceptually, holding the SR curve fixed when M changes is similar to the standard fisheries approach of holding the SR curve fixed when fishing mortality varies throughout the time series and M is constant.

For each SR curve and natural mortality rate combination, there is one F_{MSY} that maximizes the equilibrium yield given the fixed selectivity, weight, and maturity vectors at age. This combination of per recruit and SR information to derive biological reference points is often referred to as the Sissenwine–Shepherd approach (Shepherd 1982; Sissenwine and Shepherd 1987). The F_{MSY} for each combination of SR curve and value of M was found through a search over values of F from zero to two in steps of 0.01. The equilibrium yield was computed from the yield per recruit and intersection of the replacement line with that total mortality rate (natural plus fishing) with the SR curve for each F value. For a given SR curve, the F_{MSY} values can initially increase as M increases, but eventually will decrease to zero with a high enough M (Fig. 3B) as the replacement line moves to the left of the SR curve. For a given M , higher steepness in the SR curve results in higher F_{MSY} , as expected. The associated MSY values decrease with increasing M , with SR curves 1 and 2 in combination with low M producing extremely high values (Fig. 3C). Similarly, the MSY biomass (B_{MSY}) and recruitment (R_{MSY}) values decrease with increasing M (Figs. 3D–3E). All of the B_{MSY} values for M above 0.6 year^{-1} are well below historical catch amounts for this stock. Thus, using the most recent M in the example assessment of 0.9 year^{-1} results in low B_{MSY} and MSY regardless of which SR curve is used (some of the SR curves do not have equilibrium values because the replacement line for $M = 0.9 \text{ year}^{-1}$ is to the left of the curves).

The ratio of B_{MSY} to unfished SSB (B_{MSY}/B_0) is consistent, increasing with increasing M , with the exception of a case where the replacement line just barely intersects the SR curve ($M = 0.9 \text{ year}^{-1}$ and SR curve 4; Fig. 3F). These are not direct estimates of spawning potential ratios from per recruit analysis because they include the changes in recruitment associated with the SR curves. However, they do indicate that F_{MSY} is generally associated with reductions in the SSB to between 25% and 45% of unfished conditions over a range of SR curves and M values for this example.

This example demonstrates the challenges of estimating MSY biological reference points for a stock with an increasing M during the assessment time period. The SR curves are not easily distinguished, but produce widely different estimates of reference points. For a given SR curve, the points used to fit the curve are derived under different conditions of M , and thus the equilibrium

Fig. 3. Equilibrium values for steepness, F_{MSY} , MSY , B_{MSY} , R_{MSY} , and B_{MSY}/B_0 (A–F, respectively) as a function of the natural mortality rate for six SR curves fit to the Georges Bank yellowtail flounder data (Fig. 2B). Note some of the SR curves did not allow solutions for the MSY reference points at high natural mortality rates (steepness < 0.2). [Colour online.]



consequences of fishing may no longer reflect the conditions that created the observations. However, this is the assumption made when it is the fishing mortality rate that changes the SR estimates and so, in theory, should apply to changes due to M as well. The challenge associated with changing M is due to changes in the unfished conditions, which implicitly change the steepness of

the SR curve and thus the productivity and biological reference points.

Catch advice: Gulf of Maine Atlantic cod example

Stock assessments for the Gulf of Maine Atlantic cod stock were conducted in 2011 (NEFSC 2012a) and again in 2012 (NEFSC 2013)

Table 3. Gulf of Maine Atlantic cod life history and fishery parameters used in calculations (mean of values for years 2009–2011).

Age	Catch weight (kg)	Stock weight (kg)	Maturity	Selectivity
1	0.310	0.119	0.092	0.004
2	1.015	0.520	0.287	0.027
3	2.068	1.256	0.613	0.165
4	3.068	2.194	0.862	0.588
5	3.786	3.123	0.961	0.911
6	4.551	3.819	0.990	0.987
7	5.795	4.767	0.997	0.998
8	7.561	6.546	0.999	1.000
9+	12.494	12.495	1.000	1.000

Note: Time of spawning for SSB calculations is 0.25 of the year.

using the statistical catch at age program ASAP (Legault and Restrepo 1999). The results of the two assessments were similar when M was assumed to be 0.2 year⁻¹ for all years and ages. However, in the 2012 assessment, there was an additional model introduced that assumed a linear increase in M over the assessment time series. The early years of the assessment (1982–1988) assumed M equaled 0.2 year⁻¹, then increased linearly (or “ramped”) to 0.4 year⁻¹ between 1989 and 2003, and remained at 0.4 year⁻¹ for the remaining years (2004–2011). This model, denoted the Mramp model, will be used as the basis for comparing the two approaches for estimating the target fishing mortality rate when the natural mortality rate changes within an assessment. The linear ramp was used because it seemed more biologically plausible than an abrupt change, though it is important to note that there was no direct evidence of a change in natural mortality. The primary support for the Mramp model was that it exhibited a lower retrospective pattern and an improved fit to the data compared with the $M = 0.2$ model. The merits of the two models are not considered in this paper.

The maturity and fishery selectivity at age vectors from the recent 3 years (2009–2011) indicate maturity occurs before entry to the fishery (Table 3). The SR relationship is poorly defined, so $F_{40\%}$ was selected as a proxy for F_{MSY} . For simplicity, in this paper the assessment stochastic projections to estimate MSY and B_{MSY} proxies are replaced by simply multiplying the yield and SSB per recruit by the average recruitment from 1982 to 2009 (10.2 million fish).

The use of $F_{40\%}$ as a proxy for F_{MSY} is simple for the case when $M = 0.2$ year⁻¹ for all years and ages, but creates issues for the Mramp model. Calculation of the biological reference points associated with the $F_{40\%}$ proxy in the Mramp model with $M = 0.4$ year⁻¹ instead of $M = 0.2$ year⁻¹ leads to an increase in the F_{Target} from 0.183 to 0.452 year⁻¹, a decrease in the B_{MSY} proxy from 83 119 to 16 702 t, and a decrease in the MSY proxy from 14 299 to 5820 t (Table 4). The higher M allows higher fishing intensity but results in lower catch and population size. This is a concern for a stock that is already at low abundance and assumes that recruitment will be produced no matter how low the stock is driven.

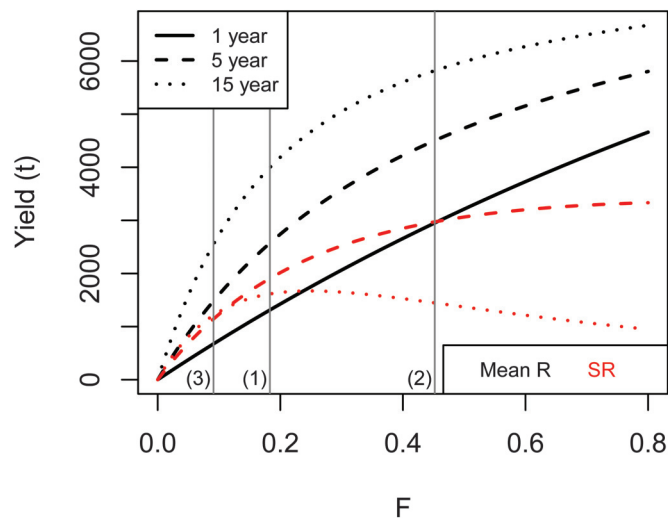
An alternative approach to setting the biological reference points is to consider the total mortality as a limiting factor. Since M doubled within the assessment, one approach (of many possible ones) would be to halve the fishing mortality rate in response. In this case, it would mean setting the F_{MSY} proxy at 0.09 year⁻¹. Applying this fishing mortality rate in equilibrium results in the B_{MSY} proxy of 31 659 t and MSY proxy of 2585 t. In this case, the higher M is associated with a lower F and results in lower catch and higher stock size than when F increased, as expected. In this example, trying to maintain the total mortality by reducing F directly when M increased is not possible because the original $F_{40\%}$ is less than the amount that M increased (meaning the modified F would be 0.18–0.20 = –0.02 year⁻¹).

Table 4. Biological reference points for Gulf of Maine cod for three cases of natural mortality (M) and fishing mortality (F) based on constant recruitment assumption.

	Case 1	Case 2	Case 3
M (year ⁻¹)	0.2	0.4	0.4
SSBPR ($F = 0$) (kg)	20.344	4.088	4.088
F (year ⁻¹)	0.183	0.452	0.092
Basis for F	$F_{40\%} (M = 0.2)$	$F_{40\%} (M = 0.4)$	$(F_{40\%} (M = 0.2))/2$
SSBPR (kg)	8.138	1.635	3.100
YPR (kg)	1.400	0.570	0.253
R (millions)	10.214	10.214	10.214
SSB (t)	83 119	16 702	31 659
Yield (t)	14 299	5 820	2 585

Note: SSBPR denotes spawning stock biomass per recruit, YPR denotes yield per recruit, R denotes recruitment of age-1 fish, SSB denotes spawning stock biomass, and $F_{40\%}$ denotes the fishing mortality rate that decreases SSBPR to 40% of the unexploited SSBPR, which is denoted SSBPR ($F = 0$).

Fig. 4. Yield as a function of the F_{Target} when M remains at 0.4 year⁻¹ in the future for the Gulf of Maine Atlantic cod example. Three specific values of F_{Target} are highlighted using vertical lines corresponding to the three cases in Table 4 (case numbers shown in parentheses along x axis). The solid line denotes the yield next year, the dashed lines denote the yield in 5 years, and the dotted lines denote the yield in 15 years assuming recruitment is either determined as a constant value of 10.214 million fish per year (black lines) or deterministically from the SR relationship described in the text (red lines). Note the next year projections are the same for the two recruitment scenarios. [Colour online.]



These equilibrium SSB and yield calculations make the strong assumption that recruitment is independent of stock size. If instead an SR curve is fit to the data and used in projections, the consequences of increasing or decreasing F_{Target} can change. The use of an SR curve does not matter for 1-year projections because the recruited fish have not entered the exploitable population. In the 1-year projection, increased F leads directly to increased yield (Fig. 4). However, as the fishing mortality is held constant for a number of years, the differences between constant recruitment and the SR curve become larger. At 5 years, the gains in yield due to fishing at the higher F of case 2 (highest F_{Target}) are reduced substantially when using the SR curve relative to the constant recruitment. These reductions are also seen for cases 1 (middle F_{Target}) and 3 (lowest F_{Target}), but are not as large in magnitude. After 15 years of fishing and deterministic recruitment, the gains due to fishing at the higher F of case 2 are all gone when the SR curve is used. The yields from fishing at the lower F of cases 1 or 3

now produce as much or more yield than that from the higher F of case 2. This is because the stock size has been driven to a much lower size in case 2, and so recruitment is much lower using the SR curve.

The short-term gains associated with high F_{Target} values should be considered in light of potential losses in future yield if the high total mortality rate leads to lower recruitment in response to reductions in SSB and reduced age structure (e.g., Scott et al. 1999; Morgan et al. 2014). A recent update of the Gulf of Maine Atlantic cod assessment indicates that recruitment over the past 5 years has declined to time series lows as both SSB and mean population age have declined (Palmer 2014). A decision analysis approach could be considered to evaluate the potential losses and gains due to a higher F_{Target} in light of the associated potential decreases in stock size. Such an analysis could be either qualitative or quantitative, the latter if management objectives were clearly specified.

Discussion

Both the per recruit approach, which increases the F_{Target} as M increases, and the total mortality approach, which decreases the F_{Target} as M increases, can result in extreme F_{Target} values when M increases substantially. While there are conceptual reasons supporting both, blind application of either can cause nonsensical results. For example, when $M = 0.9 \text{ year}^{-1}$, the Georges Bank yellowtail flounder per recruit F_{Target} of $F_{40\%}$ is 9.33 year^{-1} and one total mortality approach produces $F_{\text{Target}} = (Z_{40\%} \text{ when } M = 0.2 - \text{increase in } M) = (0.52 - 0.70) = -0.18 \text{ year}^{-1}$. Neither approach produces a reasonable F_{Target} . However, the F_{MSY} values for a range of SR curves are also problematic, indicating the difficulty of estimating biological reference points when M increases greatly within an assessment.

It is hard to understand how F_{Target} values based on stock-specific life history characteristics in per recruit or total mortality calculations could change so dramatically when M increases within an assessment. This is because there is not enough time for evolution to occur and the species to adopt a new life history to account for the change in M (Jørgensen and Holt 2013). However, there would not be a reason for the species to adapt if the change in M was strictly to address a different problem in the assessment causing a retrospective pattern. Despite the ability of an increase in M to address a retrospective problem (Legault 2009), this approach leads to the difficulties noted above for setting reference points. Thus, we do not recommend using an increase in M as a fix to a retrospective problem.

In some parts of the world, natural mortality is held constant over time in stock assessments despite estimates of changes due to predation by other species. For example, stock assessments for Gulf of Alaska and Bering Sea – Aleutian Islands groundfish hold M constant over time in all their single-species stock assessments (SAFE 2014a, 2014b). These single-species stock assessments are used to set the biological reference points that serve as the starting point for catch advice. Multispecies stock assessments have been conducted that provide time and age-varying estimates of M for some species (e.g., Livingston and Methot 1998; Hollowed et al. 2000; Jurado-Molina et al. 2005). These models generally provide some ecosystem considerations that have functionally served to evaluate the level of precaution built into the catch advice arising from the single-species stock assessments. However, the multispecies stock assessment models are not used to set the biological reference points, thus avoiding the difficulty of determining biological reference points when natural mortality changes over time within an assessment. The potential bias in the stock and F estimates due to using a constant M in the assessment when it is changing in reality should be considered in the context of the amount of change expected in M and any directional trends, with small nondirectional changes in M expected to have the least impact on management advice.

There are a number of “rules of thumb” in fisheries that are based on meta-analyses across species but are sometimes incorrectly invoked for changes within a stock assessment. One such rule of thumb is that a target fishing mortality rate of $F_{\text{Target}} = C \times M$, where C is some constant, is appropriate (Francis 1974; Thompson 1992). This is generally true over a wide range of life histories (Zhou et al. 2012). However, it encounters difficulty when M is changing within an assessment because the ability to estimate M and the nonstationarity of M mean that predicting the correct F_{Target} will be challenging. There are also other rules of thumb related to M (Williams and Shertzer 2003). For example, M/K is sometimes considered to be a constant value for a species. Under this rule, an increasing M within an assessment should be associated with an increased growth rate of the species. Similarly, a species with a given M should have a fecundity strategy that allows each adult to replace itself. So when M increases within an assessment, the fish must increase either the fecundity of younger fish or the survival of the eggs produced by these younger fish for this rule to hold (assuming F is constant). Finally, M is often related to the maximum observed age in a population (Then et al. 2014). An increase in M will reduce the maximum observed age in the population (again, assuming F is constant). The absence of life history trait changes can confound the interpretation of assumed changes in M within a stock assessment. Thus, rules of thumb are useful guides but must be kept in context of their origin as comparisons across species instead of being applied to the situation of an increasing M within an assessment.

Estimation of the steepness of an SR curve is difficult even when natural mortality is constant within an assessment (Conn et al. 2010). Often, parameters of the SR relationship require priors from meta-analysis or other information to allow estimation (Punt and Dorn 2014). Using tight prior distributions for SR parameters is similar to specifying common proxies for MSY reference points due to the one-to-one relationship between steepness and F_{MSY} in the calculations when all other parameters are fixed (Brooks et al. 2010). This means that changes in biological parameters, for example due to changes in environmental conditions, will translate directly into changes in the biological reference points. This is often considered for changes in maturity or weights at age or changes in the fishery selectivity patterns by using the most recent estimates, although longer-term considerations can also be made (e.g., Haltuch et al. 2009). However, changes in the natural mortality rate generally have a much larger impact on biological reference points (Haltuch et al. 2008). As demonstrated above in the Georges Bank yellowtail flounder example, steepness is a function of M when the SR curve is fixed. The challenge of estimating equilibrium biological reference points when biological conditions are changing remains a topic for future research.

The difficulty of management in a changing environment has been recognized for many years, especially as related to changes in recruitment (Walters and Parma 1996; King and McFarlane 2006). Climate change brings additional difficulty to setting appropriate fishing targets (Brander 2010; Hollowed et al. 2013). Delays in responding to changes brought about by climate change can result in increased probability of stock collapse, and the long-term cost of delay may offset any benefit due to short-term increases in fishing (Brown et al. 2012) and may require adaptive management (Plaganyi et al. 2011). If climate change impacts the natural mortality rate within the time period of an assessment, it is unlikely that the species will be able to respond quickly enough from an evolutionary perspective, and so care must be taken when setting target fishing mortality rates. Horbowy and Luzeńczyk (2012) and Morgan et al. (2014) suggest that using yield per recruit or spawners per recruit reference points during periods of low recruitment may result in catch advice that is dangerously high. These authors recommend using recent productivity, including at least a hockey-stick type SR relationship, to estimate F_{MSY} to reduce the chances of recruitment overfishing. This

approach is particularly important for stocks with strong maternal effects and highly exploited stocks that are likely to lead to reduced recruitment, such as Atlantic cod (e.g., Scott et al. 1999).

Many stocks of Atlantic cod have been driven to low abundance through a combination of overfishing and poor environmental conditions (Lambert 2011). Increases in natural mortality rate due to seals have been hypothesized to prevent recovery of some Atlantic cod stocks (Chassot et al. 2009), while high fishing pressure has been implicated in other stocks (Cardinale and Svedang 2004). The latter can occur even when the catch amounts are low (Shelton et al. 2006). The potential long-term consequences of setting F_{Target} too high need to be considered when natural mortality changes within an assessment both in terms of the empirical evidence to support a change in M and the difficulty in setting F_{Target} if sufficient evidence exists for such a change.

Selecting a reference point for a stock is always a trade-off that tries to balance the risk of overfishing (and possibly stock collapse) if the F_{Target} is set too high with the risk of forgone yield if the F_{Target} is set too low (Hart 2013). In the unrealistic situation of complete knowledge about the relationships between stock and recruitment and ability to forecast stock productivity, this trade-off would be a simple calculation. However, these functional relationships are never known exactly and, as shown above, can lead to different F_{MSY} values based on which particular relationship is selected. This has also been found in dynamic food web models, where uncertainty in the functional response of predator-prey interactions leads to different consequences of fishing (Gaichas et al. 2012). Balancing the risks of overfishing and forgone yield given these types of uncertainty can be addressed through management strategy evaluations (e.g., A'mar et al. 2010; Sissenwine et al. 2014). Care must be taken to ensure that the operating models employed during such evaluations provide a sufficient range to encompass both directional and cyclic change in parameters. The trade-offs also have consequences for short- and long-term yield. As shown above in the Gulf of Maine Atlantic cod example, a high F may result in higher short-term yield but lower long-term yield compared with a lower F .

Allowing M to change within an assessment can reduce retrospective problems in some stocks relative to holding M constant. This additional flexibility in the model has a cost though. Predictions about the value of M in the future are needed to set reference points. While estimating a single M is difficult (Francis 2012), estimating time-varying M values is even more problematic owing to the lack of direct observations to inform the estimates. This suggests that strong empirical evidence of changes in natural mortality from sources such as long-term tagging programs, comprehensive food habits studies, or disease monitoring are needed before using this approach in a stock assessment.

Considerations

When a change in M within an assessment is contemplated, the first consideration should be the amount and strength of empirical evidence to support such a change. Given the difficulties in estimating even a single M value for a species, strong evidence for a change in M will be difficult to collect. Potential sources include large-scale, well-designed tagging studies during different time periods but analyzed in a common framework and estimates of total mortality in situations where F can be considered negligible (e.g., fisheries closed for many years or large spatial closures for species that do not move much; Hart et al. 2013). When the empirical evidence is not strong for a change in M , we recommend using a natural mortality rate that does not change over time in the stock assessments. This avoids the difficulty of estimating or assuming how much, and when, this parameter changes over time. It also avoids the difficulty of setting biological reference points that vary depending on the approach taken to address the change in natural mortality.

If strong empirical evidence supports the use of a changing natural mortality rate over time within an assessment, we recommend estimating F_{MSY} for a range of SR relationships and evaluating the trade-offs between risk of overfishing and forgone yield (both short- and long-term) through the use of a management strategy evaluation or other means. This approach avoids defining one specific set of biological reference points and instead focuses on the trade-offs between short-term gains in yield and risk to the stock. It can be conducted either qualitatively or quantitatively, although the latter requires specification of management objectives.

Finally, we do not recommend using changes in M as a "fix" for retrospective patterns or the use of simple yield per recruit, spawner per recruit, or rules of thumb such as $F = M$ to set F_{Target} in stock assessments with a changing M . As noted above, the common "rules of thumb" in fishery science are based on comparisons across species and are generally not appropriate to apply to changes within a stock assessment for a single species. Retrospective patterns can arise from many sources, such as missing catch, changes in survey catchability, or unaccounted for changes in survey or fishery selectivity. The lack of direct observations associated with M means that it is easy to change M to account for these other problems without incurring an obvious penalty. This means that changes in M will often be found as more likely solutions than other changes that have direct observations in the model. Using changes in M simply as a fix for a retrospective pattern causes unnecessary difficulty for estimation of biological reference points that cannot be justified and could be dangerously high or nonsensical.

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