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Moving towards Ecosystem-Based Fisheries Management: options for parameterizing multi-species biological reference points

Elizabeth A. Moffitt [a,1], André E. Punt [a], Kirstin Holsman [b], Kerim Y. Aydin [c], James N. Ianelli [c], Ivonne Ortiz [b]

[a] School of Aquatic and Fishery Sciences, Box 355020, University of Washington, Seattle, WA 98115, USA
[b] Joint Institute for the Study of the Atmospheric and Ocean, University of Washington, Alaska Fisheries Science Center, National Oceanic and Atmospheric Administration, 7600 Sand Point Way N.E. Building 4, Seattle, WA 98115, USA
[c] Alaska Fisheries Science Center, National Oceanic and Atmospheric Administration, 7600 Sand Point Way N.E. Building 4, Seattle, WA 98115, USA
[1] Present address: Sea Star Scientific Editing, seastarscientificediting.com, Seattle WA, 98115

Corresponding author: E.A. Moffitt
Email: elizabeth@seastarscientificediting.com
Postal address: 7326 24th Ave NE, Seattle WA, 98115
Phone: 1-206-877-2945

aepunt@uw.edu
Kirstin.Holsman@noaa.gov
kerim.aydin@noaa.gov
Jim.Ianelli@noaa.gov
ivonne@u.washington.edu

Abstract
Multi-species models can improve our understanding of the effects of fishing so that it is possible to make informed and transparent decisions regarding fishery impacts. Broad application of multi-species assessment models to support ecosystem-based fisheries management (EBFM) requires the development and testing of multi-species biological reference points (MBRPs) for use in harvest-control rules. We outline and contrast several possible MBRPs that range from those that can be readily used in current frameworks to those belonging to a broader EBFM context. We demonstrate each of the possible MBRPs using a simple two species model, motivated by walleye pollock (*Gadus chalcogrammus*) and Pacific cod (*Gadus macrocephalus*) in the eastern Bering Sea, to illustrate differences among methods. The MBRPs we outline each differ in how they approach the multiple, potentially conflicting management objectives and trade-offs of EBFM. These options for MBRPs allow multi-species models to be readily adapted for EBFM across a diversity of management mandates and approaches.

1. Introduction

1.1 Background

Support for ecosystem-based fishery management (EBFM) has grown in the past few decades due to: increased knowledge of marine ecosystems; increased realism of models incorporating multiple species, climate, and habitat effects; the exponential growth in computing ability required for complex models; and increased interest in quantitative methods for trade-off analysis (Fowler and McCluskey, 2011; Link 2010). Here, we adopt the classification of approaches as described by Link and Browman (2014) where single-species fisheries management (SSFM) and Ecosystem-Based Management (EBM) represent bounding philosophies along a management spectrum (Link 2005; Link 2010; Link and Browman 2014). At one end of the spectrum is SSFM, which focuses on a single species or stock. The Ecosystem Approach to Fisheries Management (EAFM) incorporates ecosystem considerations such as habitat, environmental drivers, and predator-prey dynamics into the management of a single stock. EBFM builds upon EAFM; multi-species interactions and environmental drivers are included, and it remains solely fisheries focused, but the focus is on sustainability of the entire community or ecosystem instead of a single stock. At the other end of the spectrum, EBM represents a holistic approach to management that goes beyond fisheries to include goals and trade-offs across multiple sectors (e.g., other marine species, tourism, extraction, shipping, fisheries, land use, and conservation).

EBFM is expected to lead to more holistic management recommendations by explicitly considering species interactions (including non-target and protected) and ecosystem-wide processes (Link, 2010). Specifically, it quantifies the value of marine ecosystems beyond fishery harvest, and allows trade-offs between fishery fleets, sectors, and competing interests (e.g., harvest maximization, economic performance, biological diversity) to be explicitly confronted (Fogarty, 2014; Link, 2010). Additionally, with the ecosystem approach it is possible to manage for both fishery and ecosystem-level goals or ecosystem "health" (Fogarty, 2014; Large et al., 2013; Link, 2010; Samhouri et al., 2010).

Some form of EAFM or EBFM is now suggested or required by international policy, such as the U.N. agreement on Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks, and the International Council for the Exploration of the Sea (Anonymous, 2004 in Marasco et al., 2007). National policies are also requiring an ecosystem approach to management. In the United States, the Sustainable Fisheries Act, a 1996 amendment to the Magnuson-Stevens Fisheries Conservation and Management Act (Magnuson-Stevens Act), specified a framework to apply ecosystem principles, including defining total allowable catches for all species categorized as being in the fishery. Following reports from the Pew Ocean Commission (2003) and the US Commission on Ocean Policy (2004) expressing the need for EBFM/EBM, the reauthorized (2008) Magnuson-Stevens Act called for an evaluation and incorporation of ecosystem science in marine resource management. US Executive Order 13547 which established a National Ocean Policy also identified EBFM as one of its core elements. Canada's Ocean Act (1996) calls for an ecosystem approach to the management of marine species and the European Commission requires a multi-species management plan to manage Baltic cod (*Gadus morhua*; Voss et al., 2014). Australia's Oceans Policy requires regional ocean planning, which must include all uses and users of the marine

environment, and several policies call for EBFM (Smith et al., 2007).

While the incorporation of ecosystem processes in fisheries management is increasing, SSFM remains the current paradigm for providing fisheries management advice worldwide. Due to lack of data or resources, many stocks are not managed using stock assessment models. In the stocks for which a model is used, single-species population models are often used to determine the stock status (i.e. whether the stock is undergoing overfishing and whether it is an overfished state), and harvest control rules are applied to compute what in the United States (US) are called overfishing levels (OFLs) and acceptable biological catches (ABCs) (Link 2010; European Commission 2010; Gislason, 1999; Ministry of Fisheries, 2008; Restrepo and Powers, 1999; Smith et al., 2009). Broadly speaking, the quantities used to define stock status and apply harvest control rules are referred to as biological reference points. In the US, for example, the OFL may be calculated by applying the fishing mortality corresponding to maximum sustainable yield (MSY, F_{MSY}) to current biomass for stocks estimated to be above the biomass level at which MSY is achieved (B_{MSY}), and by applying a lower fishing mortality rate for stocks below B_{MSY} (i.e., sloping control rule; Fig. 1) (NPFMC, 2012; PFMC, 2011; Restrepo and Powers, 1999). In the US (and elsewhere with similar processes), if a stock is below its minimum biomass threshold (Minimum Stock Size Threshold or MSST in the US), it is declared overfished, and a rebuilding plan is required (A'mar and Punt, 2005; Annala, 1993).

Some current US management practices are beyond SSFM into the spectrum of EBFM (most at the level of EAFM). For example, the US North Pacific Fisheries Management Council has specifically identified various ecosystem goals for the region such as conserving food-web resources, maintaining biodiversity, and reducing bycatch of non-target species (Livingston et al., 2011; NPFMC, 2012). These goals have led to implementation of limits such as maximum annual harvested groundfish caps (i.e., 2 and 0.7-1.4 million t for optimum yield for the eastern Bering Sea and Gulf of Alaska ecosystems, respectively) and minimum biomass thresholds (e.g., 20% of unfished biomass for walleye pollock (*Gadus chalcogrammus*) in the Gulf of Alaska; A'mar et al., 2009). The Pacific Fishery Management Council limits harvest of forage species in the California Current below their SSFM-calculated MSY levels to account for the consumptive needs of their predators (PFMC, 2006). The Commission for the Conservation of Antarctic Marine Living Resources requires the spawning stock biomass of krill (*Euphausia superba*) in the Southern Ocean to be maintained at 75% of the unexploited level to ensure their many predators will be supported (Constable, 2011). In the Barents Sea, a multi-species model has been used to estimate parameters in the single-species stock assessment model for capelin (*Mallotus villosus*; Gjøsæter et al., 2002), and in the Baltic Sea, natural mortality of clupeid stocks is calculated dependent on the size of the cod stock (Voss et al., 2014).

Several multi-species assessment models have been developed to estimate stock biomass and trends (Gaichas et al., 2012a; Gislason, 1999; Gjøsæter et al., 2002; Holsman et al., this issue; Jurado-Molina and Livingston, 2002; Kinzey and Punt, 2009; Uchiyama et al., this issue). Several whole-of-ecosystem models have also been developed (e.g. Fulton et al., 2011; Kaplan et al., 2013; Smith et al. in press), and are often used to derive system-wide MSY. However, methods to translate ecosystem-level output of these models into quantitative fisheries management decisions within an EBFM

context need additional development. One specific challenge in the US is that overfishing and being in an overfished state is based on MSY, but there are multiple ways to compute MSY in an ecosystem context. Specifically, the "excess production" (often called "surplus production") concept of MSY is clear in SSFM; there is a surplus of individuals that can be harvested when the population is above its threshold for collapse and below its unfished level. However, an ecosystem context deals explicitly with the complexity of the concept of "excess production"; it varies with annual mortality, which depends on the numbers of predators, prey, and competitors (amongst other factors). From an ecosystem perspective, "there is no fixed single-species MSY – it rests on a multidimensional surface that is constantly changing" (Fogarty, 2014). In fact, MSY may not be achievable for multiple species at the same time (Collie and Gislason, 2001; Fogarty et al., 2012; Mueter and Megrey, 2006; Walters et al., 2005). Some of the "excess production" calculated in SSFM must remain unfished if humans want to preserve the current ecosystem state (Kaplan et al., 2013; Walters et al., 2005). Multi-species objectives further confound management reference points as management decisions may also include minimum biomass thresholds needed to avoid depletion of predators and prey while "achieving MSY" (e.g. Gaichas et al., 2012a; Gislason, 1999).

MSY in single-species assessment models is well established, but depends on the form of the stock-recruitment relationship, which can be difficult to accurately estimate, especially if stocks lack contrasting data at low or high biomasses, or when data exhibit high variability. Consequently, "proxy" values are used for F_{MSY} and B_{MSY} in several regions of the US and in Australia and New Zealand (Punt et al., 2014a). Common proxies for F_{MSY} are F_{MAX} (the fishing mortality rate which maximizes yield-per-recruit), and $F_{x\%}$ (the fishing mortality rate which reduces spawning biomass-per-recruit to x% of its unfished level). These levels of fishing lead to corresponding spawning biomass levels ($B_{x\%}$) in which the spawning biomass is x% of the unfished biomass (B_0) under the assumption that recruitment is independent of spawning biomass. For example, $F_{35\%}$ is used as the default proxy for F_{MSY} for federally-managed red (*Paralithodes camtschaticus*), blue (*Paralithodes platypus*), and golden (*Lithodes aequispinus*) king crab stocks in the Bering Sea and Aleutian Islands region of Alaska (NPFMC, 2008), while $F_{50\%}$ is the default proxy for rockfishes off the US west coast (PFMC, 2011). Common proxies for B_{MSY} include a pre-specified fraction of the unfished biomass (e.g. Punt et al., 2014a), and the average biomass over a period of years when the stock was considered to be close to B_{MSY} (NPFMC, 2008).

If EBFM is to be broadly implemented, multi-species biological reference points (MBRPs) to be used in a variety of harvest control rules must be constructed. In contrast to single species biological reference points, multi-species biological reference points must reflect multiple, potentially conflicting management objectives (Link, 2010; Gislason, 1999). MBRPs for EBFM can be developed in several ways (see review in Link, 2010), but using multi-species assessment models to define ABCs and OFLs for each species individually is closest to the current management paradigm. An alternative would be to define ABCs and OFLs for species in aggregate, either for several groups or for the entire ecosystem, and could include goals such as optimizing total biomass yield, optimizing total revenue, or optimizing biomass yield or revenue for a particular set of species.

1.2 Previous considerations

Several authors have explored the use of MBRPs for EAFM/EBFM/EBM. Collie and Gislason (2001) examined the sensitivity of commonly used biological reference points to trophic interactions in a simple, two-species, predator-prey fish community. Their results suggest that biological reference points for prey species should be defined according to the level of predation mortality. In fact, their model indicated that surplus production was zero for a fishery on a prey species when predation mortality was high. This sensitivity is expected given the outputs from multi-species production models (Larkin, 1966; Walter and Hogman, 1971). Similarly, Gamble and Link (2009) incorporated Lotka-Volterra terms for predation and competition into a surplus production model to explore whether all stocks can simultaneously reach and maintain B_{MSY} in the face of multi-species interactions and harvesting. They found that biomass estimates differed when both exploitation and species interactions were present, compared to cases where such interactions were ignored. Tyrrell et al. (2011) argued results similar to these reinforces the need to take multi-species interactions into account when calculating reference points.

Mueter and Megrey (2006) fit surplus production models to the time series of aggregated catch and biomass of groundfish in Alaska. They found that their estimates of system-wide MSY were lower than the sum of the single-species MSYs from recent stock assessments, and argued that including multi-species interactions may lead to different, perhaps more realistic, results. Moustahfid et al. (2009a) used a multi-species surplus production model to estimate B_{MSY} for longfin inshore squid (*Loligo pealeii*), and found that B_{MSY} was higher when multi-species interactions were taken into account. Uchiyama et al. (this issue) fit a multi-species biomass dynamics model and a multi-species delay difference model to seven groundfish species in the eastern Bering Sea and found that estimated B_0, system-wide MSY, and F_{MSY} were lower than the sum of these estimates from single-species assessment models. Moustahfid et al. (2009b) found that accounting for predation mortality in an age-structured assessment model led to an estimate of B_{MSY} that was more than twice as high for Atlantic mackerel (*Scomber scombrus*) in the Northwest Atlantic, compared to the same single-species model without predation mortality. Similarly, multiple comparative analyses of single- and multi-species production models applied to 13 northern hemisphere ecosystems confirmed that system-wide MSY is almost always less than the sum of single-species MSY (Gaichas et al., 2012a; Holsman et al., 2012; Link et al., 2012). More recently, Kaplan et al. (2013) conducted projections for the California Current using two ecosystem models. They calculated system-wide unfished biomass by projecting the system forward in the absence of exploitation. They also projected the system forward under varying levels of exploitation rate, and plotted equilibrium biomass and yield versus fishing mortality from these simulations. From these plots they estimated reference points for individual stocks (F_{MSY} and $F_{x\%}$). Smith et al. (in press) used a similar approach for the Atlantis ecosystem model applied to species in the southern Benguela ecosystem, and found that multi-species MSY was higher than single-species MSY from the same model and data, due in part to reduction of competition and predation mortality through fishing.

Here we outline and contrast several possible MBRPs used in harvest control rules, that range from those that can be readily used in current SSFM and EAFM frameworks to broader EBFM limits that may not be as easily implemented. We demonstrate each of the

possible MBRPs within a simple, two species model to illustrate the differences among the methods. The discussion is focused on the framework for EBFM in the US, specifically how MBRPs are applied off the US West Coast and Alaska. However, the general principles underlying this work have broader national and international applications in jurisdictions where harvest control rules are used extensively, and biomass reference points are generally based on some fraction of unfished biomass, B_0 such as Australia, New Zealand and South Africa.

2. Material and Methods

2.1 Candidate MBRPs to be used in multi-species harvest control rules

Status quo harvest control rules are likely to be of the forms in Figure 1, which require values for F_{MSY} and B_{MSY}. MBRPs to be used in EBFM which aim to achieve MSY-related goals fall within two separate categories. They can either be based on each stock's status in relation to its MSY individually, or a system-wide MSY (or proxy) can be used. MBRPs based on individual-stock reference points are more closely tied to the current philosophy of SSFM and EAFM (and therefore potentially easier to implement within current management structures), while system-wide MBRPs are more consistent with the broader interpretation of EBFM.

The two classes of harvest control rules in Figure 1 differ in terms of whether there are breakpoints that depend on biomass relative to some fraction of the unfished biomass, B_0. We develop candidate MBRPs (Table 1) for a case in which there is uncertainty about the values for the parameters of the stock-recruitment relationship, but there is information on predation relationships, as is the case for the stocks in the North Pacific region of the US. Thus, we base the candidate MBRPs on analogies to a spawning biomass-per-recruit proxy for F_{MSY}, i.e. $F_{MSY} \sim F_{x\%}$ and a biomass proxy for $B_{MSY} = x*B_0$. The value of x assumed when applying MBRPs differs among jurisdictions (e.g. 0.25 for flatfishes of the US West Coast; 0.4 for groundfishes off the US West Coast except flatfish, 0.35 for groundfishes and crab off Alaska).

Defining unfished biomass, B_0, can be challenging, particularly in a multi-species context. In principle, the process of using a multi-species framework to derive estimates for a single target species is relatively straightforward; unfished biomass is determined by projecting the model forward under average (or zero) fishing mortality rate for other species in the model (or some other defined fishing mortality rate), but without fishing on the target species. However, if the multi-species model is being used to derive biological reference points for multiple species that are linked through predation, unfished biomass may be lower when fishing is simultaneously set to zero for predators and their prey than if fishing is set to zero for only a subset of predators (Holsman et al., this issue). For the purposes of this paper we generally (the exception being one variant of Option A below) define multi-species B_0 as the set of biomasses when fishing mortality is set to zero for all species simultaneously (*sensu* Kaplan et al., 2013). As such, B_0 for each species is computed by projecting the multi-species model several years forward with no fishing until the biomass values of all the species come to some relative equilibrium. B_0 is then set for each species to the resulting long-term average biomass level, or the biomass level at the end of the projection period.

Estimation of B_0 is subject to considerable uncertainty irrespective of whether it is based on single- and multi-species models. For example, it is generally impossible to

estimate annual historical recruitment strengths since the start of exploitation; biological parameters such as growth and functional forms may have changed over time, but the data are insufficient to recognize this, and historical removals may be in error. The sizes of the errors in estimating B_0 in a single-species context are lowest for stocks, such as those in Australia and New Zealand, where fisheries started relatively recently and data on relative abundance are available from near the start of the fishery. However, extrapolation of current biomass to B_0 can be subject to substantial error when fisheries have existed for centuries and/or when ecosystem shifts and changes have occurred. Methods for defining B_0 in a changing environment exist (e.g. MacCall et al. 1985; Punt et al., 2014b) and the values for B_0 in this paper should be considered to be the "current" values.

2.1.1 Calculate $F_{x\%}$ using model simulation

Conceptually, this approach involves conducting projections using a multi-species model[1] to find the fishing mortality rates so that each species equilibrates at $x*B_0$ (Options A-B2 in Table 1). In principle, this can be accomplished by analytically or numerically solving a system of non-linear equations relating biomass of all species to fishing mortality. The approach that is closest to single-species management, as it presently exists, is to solve $F_{x\%}$ for each species so that the biomass equilibrates at $x*B_0$. Fishing mortality rates for all other species are set to current average values obtained by fitting the multi-species model to data for the ecosystem (Option A in Table 1). Alternatively, the fishing mortality for all other species could be set to zero instead of the current average values (Option B1 in Table 1). If management involves a constraint that the biomass of every species must remain at or above $y*B_0$ (or if it is required by law such as in the US), Option B2 (Table 1) could be used. In this case, the fishing mortality of all other species could be set either at current average values (Option B2a) or zero (Option B2b). If no simultaneous solution for all species can be found using Options A-B2, all of these options could be iterated several times, i.e. the fishing mortality rate would be calculated for a species, setting the fishing mortality rates for the remaining species based on a previous iteration and this process repeated for each species until convergence occurs (Option B3 in Table 1). However, this iterative process is not guaranteed to converge to a single solution, depending on its starting values.

2.1.2 Calculate $F_{x\%}$ using a single-species model and multi-species model-generated M and B_0

Options A-B could be computationally very intensive, and a unique solution may not exist. A simpler, alternative approach would to be replace the multi-species system with a set of single-species models in which the natural mortality rates by species are set to those derived from the multi-species model when fishing mortality is zero for all species (i.e., multi-species B_0). This approach is simple because $F_{x\%}$ can then be calculated for each species separately. However, it ignores the feedback between natural mortality by age for the species of interest and fishing mortality rates for the other species in the

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[1] Projections of multi-species models should include the technical interactions that arise because of mixed fisheries in which fishing for a target species leads to by-catch of other species. This is standard practice for the single-species projections, which are used to compute biological reference points (Turnock and Rugolo, 2013).

ecosystem and that the rates of natural mortality in fished ecosystems may differ markedly from those in an unfished ecosystem. This approach is similar to what has been done for Barents Sea capelin (Gjøsæter et al., 2002). Each single-species model would then be used to compute the $F_{x\%}$ (Option C in Table 1).

2.1.3 Sum of all stocks are at x% of sum of individual B_0's

The biological reference points can also be defined at a system-wide level. For Option D (Table 1), system-wide unfished biomass ($B_{0,all}$) would be defined as the sum of the unfished biomass over all species (i.e., $B_{0,all} = \sum_i B_{0,i}$). The fishing mortality rate for each species would then be a scalar multiplied by a relative fishing mortality rate (e.g. $F_i = \tilde{F} \, F_{R,i}$ where $F_{R,i}$ is the relative fishing mortality rate for species i and $\tilde{F}$ is the scalar, the value of which is selected to maximize total yield). This approach could be straightforwardly extended to situations in which there are multiple fisheries for each species by defining $F_{R,i}$ as the relative fishing mortality by fleet and species, as is common in single-species calculations of reference points (e.g., Methot and Wetzell, 2013). An intuitive way to set the relative fishing mortality rates would be to use the natural mortality rates at B_0. In many jurisdictions, some species are protected – the relative fishing mortality rates for such species would be set to zero.

2.1.4 Solve for system-wide MMSY for key species

The other major way to define system-wide biological reference points involves solving for the system-wide multi-species maximum sustainable yield (MMSY; Option E in Table 1). This is similar to an option in Ecopath-with-Ecosim that involves estimating fishing mortality patterns to maximize a utility function (Christensen and Walters, 2004).

There are multiple ways to calculate MMSY. The simplest is to find the unconstrained theoretical mathematical solution (Option E1 in Table 1). A second way, which may be more consistent with existing law, would be to constrain the optimization so that no stocks are predicted to equilibrate below $y*B_0$ (Option E2 in Table 1). In this case, the sustainability of each individual stock is equally important even if that means reducing system-wide MMSY. A third option is analogous to Option D where the relative fishing mortalities are pre-specified to split the system-wide F_{MSY} among component stocks (Option E3 in Table 1). System-wide MMSY weights species by their mass rather than their economic values. This approach could be generalized to maximize other measures such as system-wide profit. Such an approach would need to be adopted for jurisdictions such as Australia for which management objectives are expressed in terms of maximizing economic benefits (Kompas et al., 2010; Rayns, 2007).

2.1.5 Accounting for uncertainty

Biological reference points are generally based on deterministic model projections where the values for the parameters are set to the best estimates. This need not be the case. For example, the projections could allow for process error in recruitment, growth and natural mortality so that concepts such as equilibrating at some fraction of B_0 would need to be replaced by a 50% probability of being above or below that fraction of B_0. Similarly, the constraint of not dropping below some fraction of B_0 would need to be evaluated probabilistically. Parameter uncertainty could be accounted for by conducting projections

in which the parameters of the model are sampled from appropriately developed distributions.

2.2 Example using predator-prey model
It is likely that any multi-species and ecosystem models used for management purposes will represent major species using age-structured models. However, a simple, two-species, predator-prey population dynamics (surplus production) model is used here to illustrate the properties of the various options for developing biological reference points in a multi-species context. The purpose of this example is to contrast the options in a situation in which it is easy to understand their behavior. This understanding is likely to be obscured in systems with more than two species as well as when the models include age-structure. The dynamics are governed by a system in which the feeding functional relationship has the Type II form (Holling, 1959) for both prey and predator density dependence, there is a constant level of background natural mortality, and there is a fishery on both species.

$$\frac{dP}{dt} = \frac{\alpha_P P}{1 + \beta_P P} - \frac{\lambda PQ}{1 + \mu Q} - M_P P - F_P P \qquad (1a)$$

$$\frac{dQ}{dt} = \frac{\alpha_Q Q}{1 + \beta_Q Q} - g\frac{\lambda PQ}{1 + \mu Q} - M_Q Q - F_Q Q \qquad (1b)$$

In the above equations, P is the biomass of the prey species, Q is biomass of the predator species, α_P and α_Q are the intrinsic growth rates of the prey and predator species, with the density-dependent terms β_P and β_Q determining the rate of increase of each species in the absence of the other species (due to other species in the ecosystem), λ is the maximum per capita predation rate, μ is a term governing the density-dependence of the predation rate, g is consumption efficiency parameter, F_P, and F_Q are the fishing mortality rates for the prey and predators, and M_P, and M_Q are the natural mortality rates due to predator species not in the modeled system.

For this example, the functional response forms and values chosen for the parameters were based on walleye pollock (*Gadus chalcogrammus*) and Pacific cod (*Gadus macrocephalus*) from the Ecosim model of the eastern Bering Sea (Aydin et al., 2007; Aydin and Mueter, 2007). It is important to note that the set of parameters used for this example were not fit to time-series data, and the original simulation model used age-structure for these species rather than aggregated biomass. So while "pollock" and "cod" are used for illustration of a predator/prey pair, the species as modeled represent a hypothetical set of parameters – actual results may differ substantially as the parameters are based on fitting Equation 1 to data on abundance. In addition, because our goal was a generic model, cannibalism was not included.

To illustrate the MBRPs for this hypothetical predator/prey pair, the model was run using a full range of annual fishing mortality rates for each species from 0 to 0.8yr^{-1} in increments of 0.01yr^{-1}. The equilibrium biomass and catch levels for each species were determined for each pair of fishing rates in the form of a two-dimensional set of biomass and catch curves (Fig. 2). The results from the projections were then used to determine the target fishing mortality rates under each potential MBRP, where the target biomass

($x*B_0$) was taken to be 40% of B_0 and the level at which species should equilibrate ($y*B_0$) was taken to be at or above 35% of B_0, consistent with current policy for groundfish in Alaskan waters.

3. Results

Biomass and catch for pollock and cod as a function of fishing rates are shown in Figure 2. Pollock biomass decreased with increasing pollock fishing rate, and increased with cod fishing rate; the latter due to lessened predation (Fig. 2a). Cod biomass decreased with increasing fishing rates for both cod and pollock; the latter due to fewer available prey (Fig 2b). In terms of catch, both pollock and cod showed a "traditional" surplus production catch/fishing rate relationship, peaking at a maximum yield, but that maximum level depended on the level of the fishing mortality rate on pollock (Fig 2c) and cod (Fig 2d). The black contour lines show the "single-species" overfishing limit (fishing rate > $F_{35\%}$ in Option A).

The MBRP options considered in this paper (Table 1) can be illustrated as a function of fishing rates (Fig. 3), which can be overlain on each of the images in Fig. 2 to determine the biomass and catch of each species as a result of the given strategy. For Option A (Fig. 3), each red line represents the $F_{40\%}$ contour (solid for pollock, dashed for cod), where the reference value for unfished biomass B_0 is calculated dependent on the biomass of the other species. The fishing rates converge at the red point on the figure. The target fishing mortality rates under Option A are shown by the red dot; the value represents an analogy to single-species management in that single-species policy involves setting natural mortality in assessment models to the prevailing values given mortality rates for all other species in the system. As such, Option A is used in Fig. 3 as a proxy for overfishing limits (OFLs); the gray area in Fig. 3 shows the area in which one or both stocks are below $B_{35\%}$ as defined in Option A, and is analogous to the black lines in Fig. 2.

For Option B1 (dark blue lines and point in Fig. 3), each line represents the $F_{40\%}$ contour for each species when referenced against a B_0 for each species when there is no fishing in the ecosystem. The point represents the set of fishing mortality rates when both species are at $B_{40\%}$; this option leads to the most conservative fishing rate for cod. The target fishing mortality rates under Options A and B1 in this example are below those resulting from when the two species equilibrate at $0.35B_0$; in this case they also represent Options B2a and B2b. Because our example contains only two species, the additional process of iterating by species (Option B3) was not needed and is therefore not shown.

Option C could also be integrated straightforwardly into the current management paradigm (light blue lines and point in Fig. 3). M-at-age values for each species are obtained from the multi-species model, then used in a single-species context to determine the values at B_0. This option leads to the most conservative fishing rate for pollock.

Option D is shown in green; the solid line represents the locus of fishing mortality rates corresponding to $B_{40\%}$ for the combined (summed) cod and pollock biomasses, where B_0 is the sum of cod and pollock biomass with no fishing (Fig. 3). The green dashed line represents the line on which $F_{cod} = rF_{pollock}$, where r is the ratio of M_{cod} to $M_{pollock}$ in the unfished state (in this case ~ 0.74).

Option E1 uses unconstrained optimization to solve for the system-wide B_{MSY} (MMSY) (Fig. 3). Option E2 solves for system-wide B_{MSY} with the constraint that no

stocks fall below 35% of B_0 (Fig. 3). Option E3 solves for system-wide B_{MSY}, but splits the relative fishing mortality among species in a pre-specified way – in this case proportional to natural mortality (Fig. 3).

The resulting equilibrium biomass, catch, and fishing mortality rates resulting from each MBRP option (points in Fig. 3) are shown in Figure 4. The management implications resulting from the different MBRP options are less variable for pollock than for cod (Fig. 4). Option A is closest to current single-species management in the US West Coast region. If Option B is used instead, the fishing mortality rates are reduced and biomass is increased, but the catches are only slightly reduced for pollock. Option C, which takes the M-at-age vectors for all species from the multi-species model to use in single-species B_0 calculations, results in the lowest fishing mortality rates, and large pollock and cod biomass. Pollock catch is slightly decreased under Option C, and it leads to the largest cod catch (Fig. 4). Option D, which defines system-wide B_0 as the sum over all species, imposes a higher fishing rate on cod, resulting in a lower equilibrium cod biomass. Results for pollock are similar to Option A. Option E1 fishes cod at the highest rate to remove the predation on pollock, which results in a higher pollock (and hence total) catch (note the differences in scale between cod in pollock in Figure 4). The fishing rate for pollock is similar to Option A, but with higher resulting pollock catch. Option E2 does not allow cod to be depleted below $0.35B_0$ and results in similar outcomes to Option A. Option E3 has the highest fishing rate on pollock, resulting in the smallest biomass and a small increase in pollock catch; cod is also fished strongly with lower resulting biomass and catch (Fig. 4).

4. Discussion

These multi-species biological reference points give many options for using multi-species assessment model output in the current management context of harvest control rules. These MBRPs each differ in how they approach the competing interests and trade-offs of EBFM. Option A, which solves $F_{x\%}$ for each species with fishing rates for all other species set to current average rates, is closest to how reference points are computed within single-species management frameworks that are currently used in many jurisdictions such as the west coast of North America, Alaska, and Australia. As such, Option A would be the easiest to integrate into the current management processes in these jurisdictions. While MBRPs from Option A are set for each species when all other species are set at current fishing mortality levels, MBRPs from Option B1 are set for each species when B_0 is defined in the case where all other species are unfished. Option B1 led to similar catches compared to Option A, with lower fishing mortality rates (Fig. 4). If effort costs are important, Option B1 may be an overall improvement over Option A, at least for the example system. Options B2a and B2b are the same as Options A and B1, but with the constraint that no stock is allowed to drop below $y*B_0$ (Uchiyama et al., this issue; Voss et al., 2014). This adds conservative limits in keeping with current policy in many regions, by ensuring that all stock sizes exceed conventional choices for MSSTs. Short-term yield may be sacrificed for sustainability and conservation objectives. Options A, B1, and B2 all implicitly involve solving the system of equations defining the multi-species model. However, an analytical or numerical solution may not exist that allows all species to be at $x*B_0$ simultaneously, or there may be multiple solutions if the multi-species model has strong ecological interactions among species (Holsman et al., this issue). In these cases, Option B3 can be used, and prioritization of species order in the

iteration becomes an important policy decision (Voss et al., 2014). Options A, B1, B2, and B3 all fall within the context of EBFM.

Option C uses the B_0 and M-at-age parameter values that resulted from the multi-species model in the single-species assessment. This method falls within EAFM, and is similar to methods of stock assessment in which values for M-at-age taken from a multi-species stock assessment are used to conduct a single-species stock assessment (e.g. Dorn et al., 2014; Gjøsæter et al., 2002; ICES, 2011), and stays within current SSFM conservation limits. However, in Option C, the M-at-age values used in the single-species analysis are taken from the multi-species model when there is no fishing. In reality, M-at-age would be expected to change as species are selectively harvested due to trophic linkages. It is common to use yield-per-recruit methods to compute target fishing mortalities (NPFMC, 2008; PFMC, 2011). These methods ignore time-variation in weight-at-age as well as M-at-age, whereas weight-at-age varies due to temperature, ecological interactions, and/or population sizes in some multi-species models such as Atlantis and CEATTLE (Fulton et al., 2011; Holsman et al., this issue). An alternative to this approach would be to set M-at-age when computing $F_{x\%}$ based on current estimates of fishing mortality rates.

Option D finds the system-wide unfished biomass (the sum of the unfished biomass over all species). This option allows flexibility in trade-offs - in how the total fished biomass is split among species. Consistent with conservation objectives, fishing mortality rates can be set to zero for protected species. Fishing policies based on summed biomass may be dominated by the most abundant species. In our example, the green solid line (summed $B_{40\%}$) is nearly identical to the red solid line (single-species) for pollock $B_{40\%}$ (Fig. 3).

Options E1-3, which use MMSY calculated using a multi-species assessment model, is the control rule we evaluated that is furthest along the spectrum of SSFM-EBM. Multi-species maximum sustainable yield (MMSY) provides a metric of optimal ecosystem productivity. Option E1 uses the unconstrained mathematical solution found using the multi-species assessment model. This method favors maximum yield at the expense of other interests, and may allow multiple solutions, including unrealistic scenarios (e.g., harvest removals exceeding fishery capacity). MMSY based on catch biomass alone often produces the answer "eliminate the predator to harvest the prey" (e.g. Gislason, 1999; Uchiyama et al., this issue) since a prey species tends to be considerably more productive than its predators (black point in Fig. 3). Yet in these cases, socioeconomic and by-catch constraints on fishing rates should also be considered; high fishing rates may not be practical if there is no market or fishery for the predator species or if by-catch limits harvest. Option E1 also allows depletion of some species below their nominal MSSTs, which in many regions would conflict with regulations (Annala, 1993; Gamble and Link, 2009; Hilborn et al., 2004; Link, 2002) and conservation objectives. While perhaps not directly practical for management, such scenarios allow managers to evaluate multi-species dynamics of a modeled system and serve as a comparative reference for alternative options. These problems can be fixed by splitting the estimated MMSY among stocks in more practical ways.

Option E2 includes conservative constraints; the target stock size is never allowed to be less than $y*B_0$. This is a way to use the ecosystem knowledge in terms of the maximum amount of yield to be taken from the ecosystem, but still comply with current

regulations that prevent stocks becoming overfished (Fig. 3). Implementation of constraints when selecting a F_{MSY} proxy (Options B2 and E2) would add conservative limits, better support biodiversity objectives, and ensure that the target stock size exceeds conventional choices for MSSTs (e.g., Holsman et al., this issue; Voss et al., 2014). Gaichas et al. (2012a) found that biomass thresholds were needed to balance yield and biodiversity objectives for species assemblages with disparate productivities. Specifically, a small reduction in overall yield prevented multiple species extinctions. However, these MBRPs may sacrifice potential yield from the ecosystem, similar to other methods that prevent depleting stocks below nominal target levels (Hilborn et al., 2004; Hilborn, 2011).

System-wide MMSY weights species by their mass rather than their economic values. An alternative is Option E3, in which the relative fishing mortalities are pre-specified (similar to Option D) to split the MMSY among component stocks. Weighting based on profit or relative societal importance supports the objective of economic value, reflects current realities of different stocks, and could be used to balance joint maximization (Gislason, 1999; Voss et al., 2014). Such an approach would need to be adopted for jurisdictions such as Australia for which management objectives are expressed in terms of maximizing economic benefits (Kompas et al., 2010; Rayns, 2007). However, for both Options D and E3, stakeholder consensus on how fishing mortality should be allocated among stocks may prove challenging.

The MBRPs described here are likely to be sensitive to the type of multi-species model used as the basis for forecasts, whether and how relationships between spawning biomass and recruitment are modeled, and which species or aggregate groups are included (Gislason, 1999; Kaplan et al., 2013; Link et al., 2012). We used a very simple two species model for demonstration purposes. These two species exist in a more complicated Bering Sea ecosystem. Therefore a more extensive multi-species model would be more appropriate for this region (see Holsman et al., this issue). However, in simpler ecosystems, simple, multi-species models can be useful (Gislason, 1999).

Existing multi-species models range from simple to incredibly complex, with varying data and processor requirements (Plagányi 2007; Plagányi et al., 2014). While the costs of both single- and multi-species assessment need to be considered, the costs of the latter do not need to be prohibitive. Multi-species models can use data that is already being collected for SSFM and EAFM (e.g. Holsman et al. this issue). Food habits data is available in regions such as the Northwest Atlantic (Moustahfid et al., 2009b), the Barents Sea (Gjøsæter et al., 2002), the Baltic Sea (Gislason, 1999), and Alaska (Aydin et al., 2007). Moustahfid et al. (2009b) incorporated predation mortality of Atlantic mackerel (*Scomber scombrus*) simply as a fishing fleet in a single-species assessment model using existing diet data. In more data-poor regions, multi-species assessments could be performed on aggregated groups (Link et al., 2012), particularly for species that do not currently have assessments. For those areas in which age data are not available, surplus production models have been found to provide similar results (Fogarty et al., 2012).

Although some methods for quantifying the extent of uncertainty associated with the outputs of multi-species models have been developed (e.g. Gaichas et al., 2012b, Ianelli et al. this issue), model validation and sensitivity of biological reference points to model assumptions is an important area of research. A management strategy evaluation (MSE)

is a powerful tool towards this end. MSEs involve assessing the performance of alternative candidate management strategies relative to performance measures, which quantify the management and legal goals for the managed ecosystem (Butterworth, 2007; Goodman et al., 2002; Sainsbury et al., 2000; Smith, 1994). MSEs that focus on multi-species and ecosystem objectives have been accomplished (e.g. Dichmont et al., 2008, 2013; Fulton et al., 2007; Plaganyi et al., 2013), but an MSE evaluating the range of MBRPs outlined in this paper would be a powerful tool for evaluating their performance. Such testing would allow one to fully explore the implications of the simplifications that we describe.

Multi-species models have evolved over the past 40 years (Gislason 1999; Gjøsæter et al., 2002; Holsman et al., this issue; Jurado-Molina and Livingston, 2002; Kinzey and Punt, 2009; Uchiyama et al., this issue; Voss et al., 2014), and are no longer simply a heuristic tool (Link, 2010). Here we have described several options for MBRPs that allow multi-species models to be readily adapted for EBFM across a diversity of council mandates and approaches.

Acknowledgements

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Figure Legends

Figure 1. **Current single-species harvest control rules used in the Pacific States of the US.** A) North Pacific Fishery Management Council Tier 3 harvest control rules. F_{ABC} defines the fishing rate corresponding to the maximum Acceptable Biological Catch (ABC). F_{OFL} is the fishing mortality rate that would produce the Overfishing Limit (OFL) and is defined by F_{MSY}. B) Pacific Fishery Management Council flatfish harvest control rules. The Overfishing Limit (OFL) is defined by F_{MSY} (or a proxy) and estimated current biomass (B). The maximum Acceptable Biological Catch (ABC) is defined by F_{MSY} (or a proxy), estimated current biomass (B), and an uncertainty buffer (P).

Figure 2. **Predator-prey model dynamics.** Equilibrium biomass (A,B) and catch (C,D) for a hypothetical predator ("cod") and prey ("pollock") pair under all combinations of annual fishing rates from 0 to 0.8 yr^{-1} in increments of $0.01 yr^{-1}$. The black contour lines represent the "single-species" overfishing limit (fishing rate > $F_{35\%}$ in Option A). Units for biomass and catch are t/km^2, based on the Ecopath model by Aydin et al. (2007).

Figure 3. **Candidate multi-species biological reference points modeled in this paper as a function of pollock and cod fishing mortality rates.** The gray area represents the "single-species" overfishing limit (fishing rate > $F_{35\%}$ in Option A), similar to the black lines in Fig. 2. Options are defined in Table 1.

Figure 4. **Comparative multi-species biological reference points from harvest scenarios.** Biomass, catch, and fishing mortality rate for the unfished two-species model (first column), and for harvest scenarios A-E. Units for biomass and catch are t/km^2, based on the Ecopath model by Aydin et al. (2007).

Table 1. The candidate multi-species biological reference points encompass both individual stock and system-wide biological reference points.
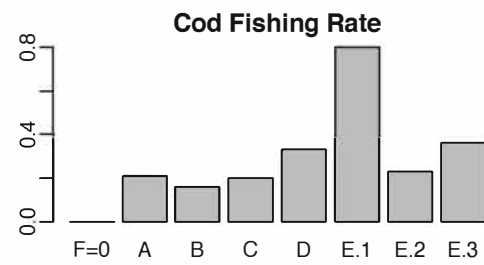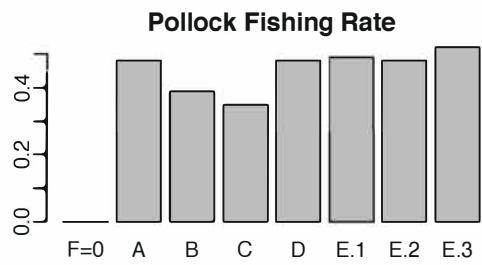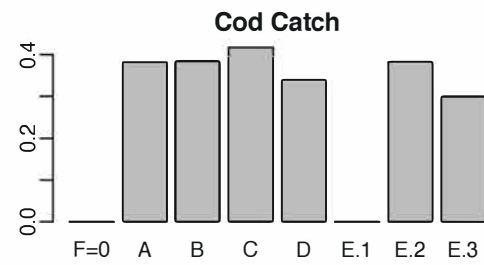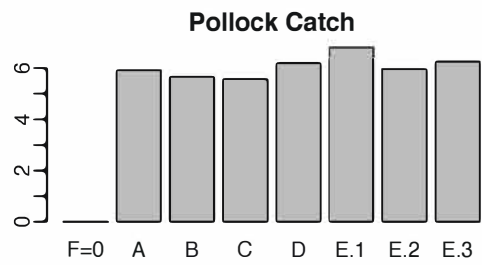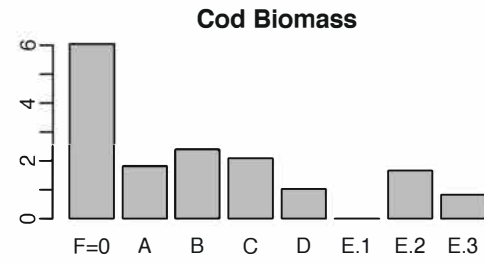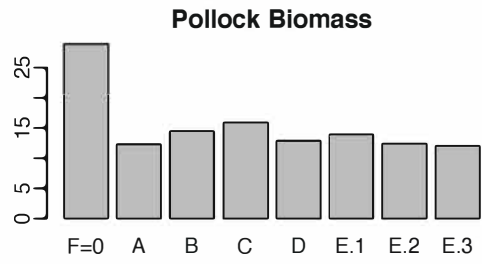
Individual stocks:

Option A. Solve for $F_{x\%}$ by species when fishing mortality for all other species is set to current average values.

Option B1. Solve for $F_{x\%}$ by species when fishing mortality for all other species is set to zero.

Option B2a/b. Solve for $F_{x\%}$ by species when fishing mortality for all other species is set to current average values (a) or zero (b) with the additional requirement that no stock is allowed to drop below $y*B_0$.

Option B3a-d. Approaches A and B2a iterated by species to find a solution.

Option C. Calculate $F_{x\%}$ when M-at-age for each species is set to the values at B_0

System-wide:

Option D. $x*B_0$ would apply over all species combined. F_{MSY} for each species would be a scalar multiplied by M.

Option E. Solve for system-wide B_{MSY}

    E1. Unconstrained optimization

    E2. Constrained so that no stocks fall below $y*B_0$.

    E3. Unconstrained with relative fishing mortality pre-specified

Control rules

(A) Pollock Biomass

(B) Cod Biomass

(C) Pollock Catch

(D) Cod Catch

Cod fishing rate

Pollock fishing rate

(A)

$F_{OFL} = F_{MSY}$

F_{ABC}

Fishing mortality rate

Percent unfished spawning biomass (B_0)

(B)

Catch levels (OFL, ABC)

- - - OFL = $F_{MSY} \times B$
—— ABC = $F_{MSY} \times B \times P$

Percent unfished spawning biomass (B_0)