

BALANCING EFFICIENCY AND RISK WITH FLEXIBLE CATCH BALANCING POLICIES

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ABSTRACT

In multispecies fisheries managed with individual fishery quotas (IFQs), fishers targeting certain species often have insufficient quota to cover other jointly caught species. New Zealand employs a unique dual price-quantity system to address this problem. In lieu of acquiring quota, fishers can opt to pay a fee per unit of catch known as deemed value. Although designed primarily to create flexibility in catch balancing for individuals, this system can allow aggregate catches to exceed total allowable catches (TACs). The deemed value system reduces the likelihood that target species catches are constrained by TACs of bycatch species, but also increases the risk of overexploitation of bycatch species. Using a bioeconomic model, we evaluate the risk and efficiency of alternative deemed value policies in fisheries with one target and one bycatch species. Our simulations suggest that increasing deemed values above ex-vessel price in response to TACs being exceeded can control risk of overexploitation without reducing overall efficiency but does shift rents from owners of target species quota to owners of bycatch species quota.

Keywords: ITQs, IFQs, multispecies fisheries, bycatch, deemed values

INTRODUCTION

Total allowable catches (TACs) in multispecies fisheries are typically set independently, with little or no consideration of relative catch rates and productivity. Consequently, catches are often out of balance with TACs. This can be particularly problematic when some stocks are substantially less productive than others. If it is not possible to reduce relative catch rates of less productive stocks, harvest of stronger stocks may have to be constrained below maximum sustainable yields (MSY) to protect weaker stocks (Ricker 1958, Paulik et al. 1967). This can dramatically reduce total yields and revenues from a multispecies complex, particularly if TACs are set to maintain all stocks at or above levels that support single-species MSY (Hilborn et al. 2004).

If relative catch rates can be altered through fishing behavior or gear choice, it may be possible to maintain higher TACs on productive stocks without putting weaker stocks at risk. However, in fisheries managed with competitively fished TACs (i.e., simple aggregate constraints on harvest), individual fishermen have no incentive to alter their catch composition if doing so increases their costs or decreases overall catch. Consequently it is often necessary to close fisheries before some TACs are taken, so as to avoid overharvest and discarding of jointly caught species (OECD 1997).

Individual fishery quota (IFQ) systems provide economic incentives to reduce the imbalance of relative catch rates and TACs: Scarcity of quota for a species increases the quota price and thereby reduces the net value (i.e., ex-vessel price less cost of quota) of harvesting the species. Fishers should therefore reduce harvest of species with the most constraining TACs (Squires et al. 1998), which mitigates the constraints weaker stocks impose on stronger ones. Branch et al. (2005, 2006) demonstrate that these incentives can significantly alter behavior and the resulting catch composition.

Reducing the catch rates of one species may significantly increase the cost of taking jointly caught species, for example by necessitating the use of less efficient gear or operation in areas where catch rates of the target species are lower. These requirements can potentially reduce the value of the overall multispecies fishery, particularly if the species with the constraining TAC provides only a small portion of the overall value of the fishery. Costly avoidance of bycatch, or some reduction in target species harvest, may be desirable if higher catches of the bycatch stock put its viability at risk. In some cases, however, the TAC of the bycatch species is set unnecessarily low due to an underestimate of stock size, in which case exceeding the TAC may not necessarily threaten the viability of the stock. Such errors are likely for stocks that fluctuate substantially, and for which stock assessments are inaccurate and TAC adjustments are not timely. In New Zealand, many bycatch stocks do not have formal assessments on which to base TACs, which are changed only rarely. In such cases, the implicit costs of a rigid TAC might be high.

When the assignment of TACs is inflexible due to information flows or logistics, and adjusting catch composition is costly, the overall value of a multispecies system may be increased by allowing fishers to exceed their quota holdings under certain conditions. This flexibility can also deter discarding, which is wasteful and can undermine the quality of data used in stock assessments. A number of mechanisms have been employed to add

flexibility to catch balancing in IFQ systems: allowing exchange of quota across species; allowing fishers to carry back (borrow) or carry forward (bank) quota between years; charging a fee for catch in excess of quota holdings; and allowing fishers to surrender or discard catch they can not match with quota (Sanchirico et al. 2006). While these mechanisms are designed primarily to add flexibility to individual catch balancing, they can also lead aggregate catches in excess of the TAC. This may increase the risk that some species will be depleted to the point where their viability is threatened. Ideally, a policy would add flexibility to a multispecies IFQ fishery without substantially increasing the risk of collapse of any component fish stock.

Although a variety of policies can facilitate catch balancing, our analysis focuses on the use of a “deemed value,” a fee paid on each unit of catch in excess of quota holdings. This system is the primary means of enhancing catch balancing flexibility in the New Zealand Quota Management System (QMS). Deemed values are set each year by the government, generally at a level lower than the ex-vessel price. This gives fishers an incentive to land (rather than discard) “bycatch” while at the same time reducing incentives to target species for which they have insufficient quota. Deemed values lower than ex-vessel prices are not, however, always sufficient to deter catches in excess of TACs. In fact, TACs for a number of species have been exceeded even with deemed values set above ex-vessel prices.

A deemed value system effectively creates a dual quantity-price regime for managing catches at both the individual and aggregate level. In the presence of uncertainty about the size and productivity of fish stocks, price mechanisms have been shown to yield superior economic performance to quantity controls (Weitzman 2002). However, this result presumes a strong positive correlation between stock size and catch per unit effort (CPUE), and can be reversed when prices or harvest costs are uncertain (Hannesson and Kennedy 2005). Most previous analyses comparing price and quantity instruments consider single species fisheries; our analysis focuses on managing a bycatch species taken while targeting a more economically important species.

We use a model to explain how the biological and economic characteristics of multispecies fisheries affect the performance of different deemed value policies, both in terms of biological risk and economic efficiency. We also explore the impacts of these policies on both the distribution of rents among owners of quota for different stocks and the flow of deemed value payments to the government. Data constraints prevented us from developing predictive models of deemed value systems in specific multispecies fisheries. Rather, we investigate stylized two-species cases, each with one clear target and one clear bycatch species.

We explore two different cases which are based on actual New Zealand multispecies fisheries but abstract from some of their complexities. We examine fisheries whose biological characteristics suggest that flexible catch balancing policies are particularly likely to pose risks. The first case includes two very slow growing species with highly variable recruitment. The second contains a target species that is considerably more productive and variable than the bycatch species. The results of these two-species cases should provide insights relevant to more complex multispecies fisheries. However, because fisheries with three or more species may respond differently, the generality of our results may be limited.

METHODS

We develop a bioeconomic model of an age-structured fishery consisting of a target species (1) and an incidentally caught, or bycatch, species (2). We use the model to evaluate the economic performance and risk associated with different policies for setting total allowable catch (TAC) for each species and the deemed value (DV_2) charged by the government for catches of the bycatch species in excess of its TAC. In theory, fishers also have the option of violating TAC_1 for the target species, and paying DV_1 for each additional unit of harvest; we make the simplifying assumption that the deemed value for the target is sufficiently high to discourage catch in excess of quota. This removes one choice variable on the part of the regulator, and allows us to focus on the impact of the DV system on harvest of the bycatch species.

We use age structured biological models with a variable number of cohorts, depending on natural mortality of the species. We employ stochastic Beverton and Holt stock-recruitment functions parameterized by virgin biomass B_0 and steepness h (mean recruitment as a fraction of virgin recruitment when spawning stock biomass, SSB, is 20% of B_0). Our model therefore allows us to explore the effects on the risk of overexploitation of both stock-recruitment steepness and differences between selectivity and maturity ogives, both of which have been shown to be key risk factors (Francis 1992, 1993).

Multispecies fishery models are scarce (Butterworth and Punt 1999), and most assume exogenous relative catchability of species (e.g., Clark 1990; Overholtz et al. 1995; Holland and Maguire 2003). Deemed value systems, however, presumably provide an economic incentive for fishers to adjust their catch mix, so it seems more realistic to model catch composition as endogenous. The present model depicts a harvest sector that maximizes expected

profits by choosing from a set of discrete harvest opportunities in each time period, each opportunity yielding a different composition of harvest of the two species. Our model also endogenizes total catch, in particular that of the bycatch species. Both species are managed with TACs. However, if fishers find it profitable to do so, they may choose to exceed the TAC for the bycatch species and make deemed value payments to the government. Total effort and cost vary endogenously as a function of the collective location choices of the fleet.

Once overall species abundance has been determined, the government chooses three regulatory parameters: TAC_j for each species $j=1,2$, and the deemed value, DV_2 , for the bycatch species. The fleet then responds to these regulatory parameters by choosing where to fish. Note that we do not explicitly present an optimization problem on the part of the regulator. Instead, we compare the performance, along various dimensions, of different rules-of-thumb for choosing the regulatory parameters.

A stochastic set of discrete production possibilities is randomly generated each period. Depending on expected profits, the harvest sector (presumably a collection of profit-maximizing agents, though not modeled explicitly as such) makes a binary decision whether or not to exploit each discrete possibility. Profits depend upon expected catches in each location, ex-vessel prices, equilibrium quota rental prices emerging in each time period as described below, and on DV_2 .

Expected catches are a function of total recruited biomass of the stocks and an underlying covariance matrix, COV , which is a function of the spatial distribution of the stocks and perhaps other factors affecting relative catch rates. In our simulations, COV is hypothetical, but it could be calculated if there were adequate survey data covering the viable fishing grounds. A covariance matrix estimated from commercial catch would differ from COV , because harvesters' choices are biased toward more profitable locations. The populations of the two stocks are randomly distributed across 1000 locations such that the distribution and codistribution of the stocks across the 1000 locations reflects the covariance matrix COV .

We employ a reduced-form representation of the behavior of the fleet in aggregate during each time period. While this prevents us from drawing conclusions about decision making at the trip or vessel level in response to the regulatory framework, it simplifies our analysis and computations and allows us to focus on system-wide effects of different management approaches. The fleet chooses to exploit a subset of the available production choices, based on expected catches associated with each choice and the net prices received for each species. Locations are ranked in order of descending expected profits, π_i , calculated as

$$(1) \quad \pi_i = (P_1 - R_1)q_1Y_{i,1} \exp(\sigma_1^c \epsilon_{i,1}^c) + (P_2 - R_2)q_2Y_{i,2} \exp(\sigma_2^c \epsilon_{i,2}^c) - C; \quad \epsilon_{i,j}^c \sim N(0,1)$$

P_1 and P_2 are exogenous ex-vessel prices (paid to fishermen at the dock) for target and bycatch respectively; the R_j are the rental prices for species j quota, determined as described below; $Y_{i,j}$ are biomasses of species j at location i ; and q is the catchability coefficient, assumed equal across species and locations. The cost, C , associated with harvesting a location is assumed to be constant across the options. Relative profitability includes the cost of renting quota, so profits depend upon net prices ($P_j - R_j$) received for each species rather than just ex-vessel prices.

The ϵ^c are location-specific random variables with zero mean and unit variance. The σ^c are multipliers on errors in fishers' expectations of profits in each location – due to uncertainty about either stock abundances or the equilibrium quota prices. Actual catch is deterministic once the location is chosen. We initially assume perfect information then later explore the implications of relaxing this assumption.

Since 2001 in New Zealand, quota shares have generated annual catch entitlements (ACE): catch rights, denominated in kilograms, that must be used during a given fishing year. Catches are now balanced with ACE rather than quota per se. ACE replaced quota leasing and other forms of short-term contracting between owners and users of quota. The R_j in our model are effectively quota rental prices for species $j = 1,2$, but we refer to them henceforth as ACE prices. The R_j are key determinants of relative profitability of different locations, and therefore of the location choices and realized aggregate catches. These prices are endogenously determined as functions of the regulatory parameters and stock distributions.

The ACE price for the bycatch species, R_2 , equilibrates at the highest level that does not constrain catches of the bycatch species below the TAC, with the additional stipulation that $R_2 \leq DV_2$. Paying DV_2 is a perfect substitute for purchasing ACE for Species 2, so fishers will not lease quota for a price higher than DV_2 .

The algorithm used to determine the ACE price R_2 for the bycatch species therefore starts by setting $R_2 = DV_2$. If the aggregate catch of Species 2, H_2 , exceeds TAC_2 at this price, then the equilibrium remains $R_2 = DV_2$; catch in excess of the TAC_2 results in DV payments of $DV_2 * (H_2 - TAC_2)$ to the government. If $H_2 < TAC_2$ when $R_2 = DV_2$, R_2 is progressively lowered, and the new aggregate catches computed. The process is repeated as long as $H_2 < TAC_2$, although R_2 also has a lower bound of 5% of the bycatch ex-vessel value. That is, $R_2 \geq 0.05P_2$, even if $H_2 < TAC_2$ at

that price. This lower bound was chosen arbitrarily, based on the presumption it would be necessary to offset cost recovery levies and other transactions costs.

In theory R_1 should equilibrate on a price that drives marginal profit to zero (i.e., $\pi_i=0$ in the last location fished) given the R_2 determined as described above. Our method approximates this “market clearing” level of R_1 , while avoiding a marginal price overly influenced by the catch ratio of the last single catch event. R_1 is calculated such that the average residual profit per kilogram of target catch in the least profitable 5% of areas fished is zero.

In short, ACE prices for the two species depend upon the exogenous and perfectly inelastic supply of ACE (the TACs), the derived demand for ACE, and DV_2 , which serves as a backstop. Determination of the endogenous ACE prices R_i by the algorithm described above is dual to the problem of determining the optimal production levels of two jointly produced goods, given a set of exogenous prices and a production possibility frontier, as depicted in Turner (1997). Rather than finding the point on the production possibility frontier that is tangent to an exogenous price ratio, our model finds the ACE prices R_1 and R_2 such that a line with slope $-(P_1 - R_1)/(P_2 - R_2)$ is tangent to the point on the production possibility frontier exogenously determined by the TACs. Effectively it determines a set of net prices that leads to a combination of aggregate catch levels that satisfies the TACs and also maximizes profits. Our scenario is complicated slightly by the fact that the TACs do not constitute an inviolable constraint; fishers can (in aggregate) choose to expand production along the Species 2 dimension beyond that dictated by TAC_2 , at a price (i.e., by paying DV_2 on catches above TAC_2).

Another feature of our analysis is that, contrary to the situation pictured in Turner (1997), our simulations presume a discrete set of harvest options. Aggregate catch is determined by summing the catches that result from the chosen locations. These choices do not produce a smooth aggregate production possibility frontier.

Although the covariance matrix that determines the relative catch rates across location choices is constant, actual catch rates vary stochastically from one year to the next. At the beginning of each year, a new matrix of random numbers is generated and the fish populations are remixed. We do not attempt to model how the spatial distribution would develop over time in response to fishing and fish movement, and we assume that harvest costs are homogeneous across the production choice set. To do otherwise would greatly complicate the model with little analytical or conceptual benefit.

Policy evaluations

Our analysis focuses on cases where one species is targeted and the other taken primarily as incidental catch (bycatch). These are the most problematic cases, because even deemed values set above the ex-vessel price of bycatch may not always provide sufficient incentives to discourage catching it – in particular if reducing bycatch results in a significant drop in target species catch rates. This problem is exacerbated when the ratio of bycatch to target species catch is very small, in which case very large penalties per unit of bycatch may be needed to change targeting behavior.

Trade-offs in economic efficiency and biological risk associated with alternative deemed value and harvest strategies depend on a number of biological and economic factors. These include, but are not limited to: relative potential size of target and bycatch stocks (e.g., B_0); productivity factors (e.g., steepness and variability of recruitment, individual growth rates and natural mortality); spatial correlation of the two species, which affects the ability of the fleet to alter relative catch rates through fishing choices; and ex-vessel prices and harvest costs. A complete factorial analysis of all possible combinations of these characteristics is beyond the scope of this analysis. Instead we focus on a few cases with characteristics that exacerbate the risk that the bycatch stock will be overexploited. We do not attempt to represent the full complexity of multispecies fisheries, but rather model pairs of species that are consistent with known characteristics of actual New Zealand multispecies fisheries.

The first case has orange roughy (*Hoplostethus atlanticus*) as the target species and smooth oreo (*Pseudocyttus maculatus*) as the bycatch. Biological parameters are based on the Mideast Coast stock of orange roughy (ORH3A) and the Chatham Rise smooth oreo stock (OEO3A). Although the two stocks are similar in size and productivity, the price of orange roughy (\$3.70) is nearly five times the price of smooth oreo (\$0.80), so targeting decisions are largely driven by catch rates for orange roughy. This case is illustrative of the risk-efficiency trade-offs in a fishery with two jointly caught species that are both very long-lived and slow growing, with highly variable recruitment. The target exploitation rate for orange roughy is also higher than for smooth oreo, increasing the risk that smooth oreo will be overexploited.

The second case has red cod (*Pseudophycis bachus*) as the target species and elephantfish (*Callorhynchus milii*) as the bycatch. This case is representative of the red cod fishery off the East coast of the South Island in New Zealand in QMA 3, which includes the red cod stock RCO3 and the elephantfish stock ELE3. The biological parameters of the model are based on these stocks. Although the price of red cod (\$0.54) is only about one-third that of elephantfish (\$1.58), the targeting decisions for these simulations are dominated by red cod catch rates because of the high ratio of red cod to elephantfish in catches. The MSY for RCO3 is more than an order of magnitude higher than that for ELE3.

This case is of particular interest because the bycatch species, elephantfish, is considerably less productive than the target species, red cod, so its viability might be threatened if exploited at rates appropriate for the target stock.

In the base cases for both scenarios, we assume a relatively high spatial covariance of species (0.5). High covariance increases both the opportunity costs of avoiding bycatch and the potential risk of overexploiting the bycatch stock. Results from models with high covariance are expected to be more conservative; risk and efficiency losses are likely to be lower if covariance is lower. As a sensitivity analysis, we also run simulations where the covariance is reduced to 0.25 and 0.0.

Consistent with prescriptions of the New Zealand Fishery Act 1996 (Part 3. No. 13. s. 2(a)), TACs in the base cases are set to maintain SSB at or above levels that support MSY in the case of deterministic recruitment. For Case 1, the base case simulation adjusts TACs of both orange roughy and smooth oreo annually to achieve a constant fishing mortality equal to the deterministic F_{msy} . The high variability of red cod makes it impractical to take this approach. In practice, the TAC for red cod has been constant at 12 389 tonnes since 1992, and between 12 000 and 13 000 tonnes since introduction to the QMS in 1986. Therefore, for Case 2, the base case simulation assumes the TAC for red cod is fixed at the deterministic MSY (12 529) while the TAC for elephantfish is adjusted annually to maintain constant fishing mortality F_{msy} . As a sensitivity analysis, we also simulate alternative TAC setting policies for both cases, as discussed in the results.

Historically, deemed values have often been set at some fraction of ex-vessel price, and are mostly kept below ex-vessel prices to encourage fishers to land catches rather than discard them (a policy consideration, despite the illegality of discarding). This is not entirely consistent with the mandate of the New Zealand Fishery Act 1996, which states that the primary consideration when setting deemed values should be the incentive to balance catch with ACE. Because deemed value is a perfect substitute for ACE, deemed values must be maintained above ACE prices to create incentives for individuals to purchase ACE rather than simply paying deemed values. The simulations set the deemed value for the bycatch species at 150% of the prior year's ACE price. However, the one-year lag between the deemed value and the ACE price means that the deemed value in a given year of the simulation can be higher or lower than 150% of that year's ACE price. If demand for ACE to cover bycatch decreases, the ACE price falls and the deemed value is reduced the next year and the opposite occurs when ACE demand rises. In all simulations, the deemed value for the bycatch species is adjusted each year to be equal to 150% of the prior year's ACE price, subject to constraints on minimum and maximum levels.

Deemed values are constrained between a minimum of 50% of the ex-vessel price and a maximum equal to either the ex-vessel price or twice the ex-vessel price. Note that even a deemed value equal to twice the ex-vessel price does not necessarily create sufficient incentive to constrain bycatch within the TAC; the opportunity cost of avoiding a unit of bycatch (e.g., forgone profits from harvesting the target species), rather than the value of the bycatch itself, is what determines whether it is worth paying deemed value. This assumes sufficient enforcement to deter illegal discarding.

The model does not explicitly set a deemed value for the target species. In reality, fishers can balance catch of target species with deemed value just as they can bycatch and thus catch of the target species could exceed the TAC. Our implicit assumption is that deemed values are utilized only as a tool to allow flexibility in catch balancing for the bycatch species. Thus the deemed for the target species is always kept high enough to remove any profits from targeting the species without ACE, so the TAC for the target is never exceeded.

Since we assume that recruitment and other parameters vary stochastically, we evaluate average results for each simulation from 500 replications, each of which runs for 100 years starting with randomly generated starting stocks consistent with the target fishing mortality rates. We compare the performance of alternative maximum deemed values in terms of risk, economic efficiency and the distribution of fishery rents between target and bycatch quota owners.

The base case simulations assume fishers have perfect information about the relative profitability of different locations, thus the uncertainty multipliers in equation (1) are set to zero. Under this assumption, the most profitable locations are fished. We then explore the impacts of introducing uncertainty regarding relative profitabilities of locations by setting both the uncertainty multipliers to 0.25. This leads to a lognormal distribution of perceived net revenues for each species. This distribution has a mode equal to the true value but is skewed, with the perceived value falling between 50% and 200% of the true value approximately 95% of the time.

We also evaluate the impacts of introducing uncertainty in the stock assessment used in the setting of TACs. This uncertainty is introduced as a random multiplicative observation error in the estimate of recruited stock biomass. The random error is autocorrelated, so error in the stock assessment error is likely to persist for a few time periods, but then dissipate.

The model tracks a variety of different performance indicators: average catches, total fishery rents and their distribution, and risks of overexploitation. Several possible measures of risk might be used to evaluate policies. Hilborn

(1997) argues that measures of risk beyond the average level and variability of catches and rents may be unnecessary if models include the mechanisms that create variability in productivity (e.g., variation in recruitment, natural mortality and fishing mortality). However some potential biological risks associated with low stock levels are not modeled in these simulations (e.g., depensation). Therefore we track and report proxies for biological risk based on biomass reference points.

Reference points based on SSB are commonly used by fishery managers to evaluate the riskiness of alternative harvest strategies. Target or limit reference points for SSB set at some percentage of B_0 are often used (Butterworth and Punt 1999). However, the risk of collapse when stocks fall below these levels depends greatly on the productivity of the stocks, and B_0 is notoriously difficult to estimate (Myers et al. 1994). Myers et al. recommend recruitment-related biomass reference point ($50\%R_{max}$) equal to the SSB that produces 50% of the maximum recruitment from a fitted stock recruitment relationship. Rather than choose a single measure of risk, we report the percentage of years that SSB falls below a variety of SSB reference points including 10% of B_0 and $50\%R_{max}$. We also monitor the frequency with which SSB falls below 20% B_0 , but do not use this as a risk threshold since several of the stocks naturally fluctuate around this level under an F_{MSY} policy without any TAC violations. Note that the SSB of either species can fall below the given biological reference points without catches exceeding TACs. Our primary interest here is to determine if the frequency with which these reference points are breached is decreased as a result of stricter deemed value policies (i.e., higher maximum deemed values) that reduce the frequency or magnitude of TAC overruns.

Overall economic efficiency is represented by the average annual revenues across both species minus harvest costs. The simulations also track the distribution of fishery rents. This includes the share of profits accruing as ACE revenues to quota owners of the target and bycatch species, deemed value payments accruing to the government and residual inframarginal rents accruing to fishers. The simulations determine ACE prices for both the target and bycatch species endogenously. For the target species, ACE revenues for quota owners are equal to the total catch of the target species multiplied by the ACE price. The same is true for the bycatch species, except that deemed values are paid on any catch exceeding the TAC and those revenues are assumed to accrue to the government. Because catch rates at different fishing locations vary but ACE prices are constant, some locations generate inframarginal profits once harvest costs and ACE costs are deducted from revenues. These latter profits presumably accrue to fishers who may or may not be quota owners.

RESULTS

Case 1: Biological risk and stock productivity

The incidentally caught species (smooth oreo) for this case is vulnerable to overexploitation because the fish recruit to the fishery ten years before maturity. The SSB of smooth oreo falls below 10% of B_0 and below $50\%R_{max}$ a significant proportion of the time for some simulations. This is generally not the case for simulations where both TACs are adjusted to maintain F_{MSY} , unless covariance of the stock distributions is high. For the base case scenario, with TACs for both species set at F_{MSY} and a high covariance (0.5) of species distribution, the SSB of smooth oreo falls below 10% of B_0 29% of the time and below $50\%R_{max}$ 4% of the time when the maximum deemed value is \$0.80. However the frequency of breaching either reference point falls to less than 1% with the higher maximum deemed value of \$1.60. For covariance of zero or 0.25, SSB of smooth oreo never falls below 10% of B_0 or $50\%R_{max}$, even with the lower maximum deemed value.

We also simulate a policy where the TAC for orange roughy is set to maintain fishing mortality at 65% of F_{msy} . This TAC policy, which we refer to as the maximum economic yield (MEY), yields the highest total annual rents from the orange roughy–oreo fishery when recruitment is deterministic. It is not a true MEY policy since it does not account for the dynamic nature of the fishery or variability, focusing solely on equilibrium rents in the deterministic case. A 35% reduction in the target fishing mortality for orange roughy completely prevents smooth oreo from ever falling below the biological reference points, even with the highest level of covariance and the lowest maximum deemed value. This is achieved with less than a 5% reduction in average catch of either orange roughy or smooth oreo.

Adding uncertainty in the relative profitability of locations to the base case scenario slightly increases the frequency that SSB falls below reference points. Relative to the base case scenario with a maximum deemed value of \$0.80, the frequency of SSB falling below 10% of B_0 increases from 29% to 32%, and the frequency of SSB falling below $50\%R_{max}$ increases from 4% to 5%. But results with higher deemed values are unchanged from the base case with no uncertainty in relative profitability.

Introducing uncertainty in the stock assessment for orange roughy has variable impacts on the percentage of years that smooth oreo SSB falls below reference points. For the base case with an F_{MSY} policy for both stocks, a high covariance, and a deemed value of \$0.80, the frequency of SSB falling below 10% of B_0 decreases from 29% to 24% of

years when stock assessment error is added. With a maximum deemed value of \$1.60, SSB falls below 10% of B_0 less than 1% of the time.

In practice, the TAC for oreo is not adjusted annually to maintain a constant fishing mortality, though it has been reduced over time as the stock has been fished down. As another sensitivity analysis we consider scenarios where the smooth oreo TAC is fixed at the deterministic MSY while the orange roughy TAC is adjusted to maintain F_{MSY} . For this TAC setting scenario smooth oreo SSB falls below reference points considerably more frequently than when both TACs are adjusted to F_{MSY} . With covariance of 0.5, SSB falls below 10% of B_0 46% of years with a maximum deemed value of \$0.80, and 15% of years with a maximum deemed value of \$1.60. SSB falls below 50% R_{max} 25% of years with a maximum deemed value of \$0.80, and 7% of years when the maximum deemed value is \$1.60.

Although our focus is on the effect deemed value policies have on biological risk for the bycatch species, we also track SSB relative to reference points for the target species. SSB stays above all three reference points more than 99% of years for all scenarios modeled. Orange roughy catches decline below the TAC when the stock declines because taking the full TAC becomes unprofitable as catch rates fall. This reflects our implicit assumption that the relative distribution of the stock remains constant as the stock contracts so CPUE falls as the stock declines.

Case 1: Economic efficiency and distribution of resource rents

For the base case model (TACs set at F_{MSY} for both stocks and the highest covariance of 0.5), total annual average fishery rents are \$8.23 million and increase slightly to \$8.34 million when the maximum deemed value is \$1.60 (Figure 1). Although total rents are not greatly affected by the maximum deemed value, the distribution of rents is. The share of total fishery rents accruing to orange roughy ACE owners is 31% with a maximum deemed value of \$0.80, but falls to 10% when the maximum deemed value is raised to \$1.60. In contrast, total ACE revenues for smooth oreo rise from 15% to 37% of total fishery rents when the maximum deemed value is raised. Deemed value payments actually decline from 9% of total rents to 4% with the higher maximum deemed value, reflecting the fact that doubling the maximum deemed value reduces catches above the TAC by over 50% on average. Inframarginal profits increase from 46% of total rents to 49% when the deemed value is raised to \$1.60. In sum, there are actually small gains in total fishery rents with the stricter deemed value policy for this scenario, but it results in substantial shifts in fishery rents from orange roughy ACE owners to oreo ACE owners.

If covariance is 0.25 and the maximum deemed value is set at \$0.80, there is no difference in the total fishery rents or in the share of fishery rents accruing to orange roughy ACE owners, relative to the base case with higher covariance. However, smooth oreo ACE revenues increase while deemed value revenues fall. As with the base case, increasing the maximum deemed value reduces orange roughy ACE revenues and deemed value payments, while increasing smooth oreo ACE revenues and inframarginal profits. With zero covariance, total fishery rents are slightly lower than in the base case (\$8.14 million). The share of rents earned by orange roughy ACE owners is similar to the base case, but a higher percentage of rents accrue to smooth oreo ACE and a smaller share to deemed value payments. With zero covariance, increasing the maximum deemed value has little impact on either the total or the distribution of rents, because the lower deemed value is sufficient to keep smooth oreo catches within the TAC.

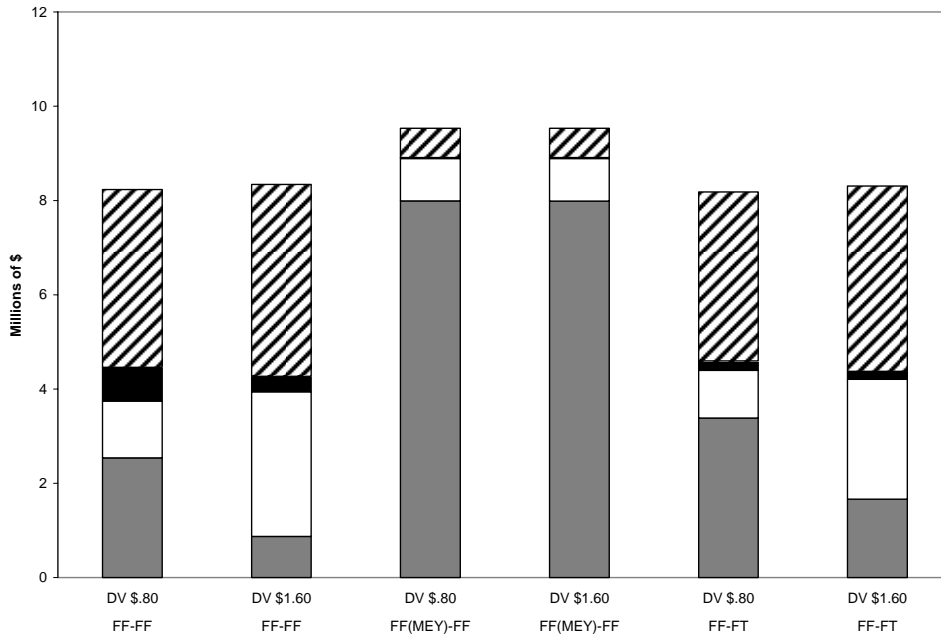
Reducing the target fishing mortality rate for orange roughy by 35% (to the MEY level, assuming high covariance) increases total fishery rents by over 15%, to \$9.53 million despite a drop in average catches and revenues for both species, (Figure 1). These gains reflect lower total harvest costs. At the lower exploitation rate for orange roughy, it is no longer costly to stay within the smooth oreo TAC, and total deemed value payments fall to zero. Orange roughy ACE revenues increase dramatically to 84% of the total, while both inframarginal profits and smooth oreo ACE revenues decline substantially (in absolute terms as well as the share of total rents). Raising the maximum deemed value has no impact in this scenario.

Uncertainty in relative profitability across locations leads to a small overall decline in total fishery rents relative to the base case. This reflects the fact that it generally takes more effort and therefore higher harvest costs to take the orange roughy TAC. Neither smooth oreo ACE revenues nor total deemed value payments change substantially from the base case.

Relative to the base case, uncertainty in the stock assessment results in almost no change in total fishery rents when the maximum deemed value is \$0.80, and a slight increase in total fishery rents when the maximum deemed value is \$1.60. It does result in a decrease in the share of rents going to smooth oreo ACE owners, particularly with higher maximum deemed values.

The scenario with a fixed TAC for smooth oreo set equal to deterministic MSY results in slightly lower total fishery rents than the base case model where both TACs are maintained at F_{MSY} . However, this TAC setting strategy generally increases orange roughy ACE revenues while reducing deemed value payments and smooth oreo ACE revenues.

Figure 1: Average orange roughy (*Hoplostethus atlanticus*) annual catch entitlement (ACE) revenues (grey), smooth oreo (*Pseudocyttus maculatus*) ACE revenues (white), deemed value payments (black) and inframarginal profits (crosshatched) for Case 1 with covariance of 0.5 under three total allowable catch (TAC) setting scenarios each simulated with maximum deemed value on smooth oreo at \$0.80 and \$1.60. Columns 1-2 have a variable TAC with fishing mortality at the deterministic maximum sustainable yield (F_{msy}) for both species. Columns 3-4 have a variable TAC for orange roughy with fishing mortality set at $0.65 F_{msy}$ and TAC for smooth oreo set at F_{msy} . Columns 5-6 have a variable F_{MSY} TAC for orange roughy and a fixed MSY TAC for smooth oreo.



Case 2: biological risk and stock productivity

No significant biological risk to the elephantfish stock is evident in any of the scenarios modeled for this case. Neither the SSB of red cod nor elephantfish ever falls below 10% of B_0 or $50\%R_{max}$, and both stay above 20% of B_0 more than 95% of the time. These results hold for several sensitivity analyses where: the TAC for red cod is set at three times the deterministic MSY or where there is uncertainty in relative profitability across locations or in stock assessments.

Catches of elephantfish generally remain within the TAC even with the lower deemed value mainly because catches of red cod are well below the TAC in most years. These simulated results are consistent with reality. The simulations were calibrated by adjusting harvest costs and catchability to achieve an average catch and exploitation rate consistent with observed values in the fishery. In most years the fishery is constrained by economics rather than regulations. Fishing is constrained to locations with high catch rates because it is unprofitable to fish locations where catch rates are low.

Case 2: economic efficiency and distribution of resource rents

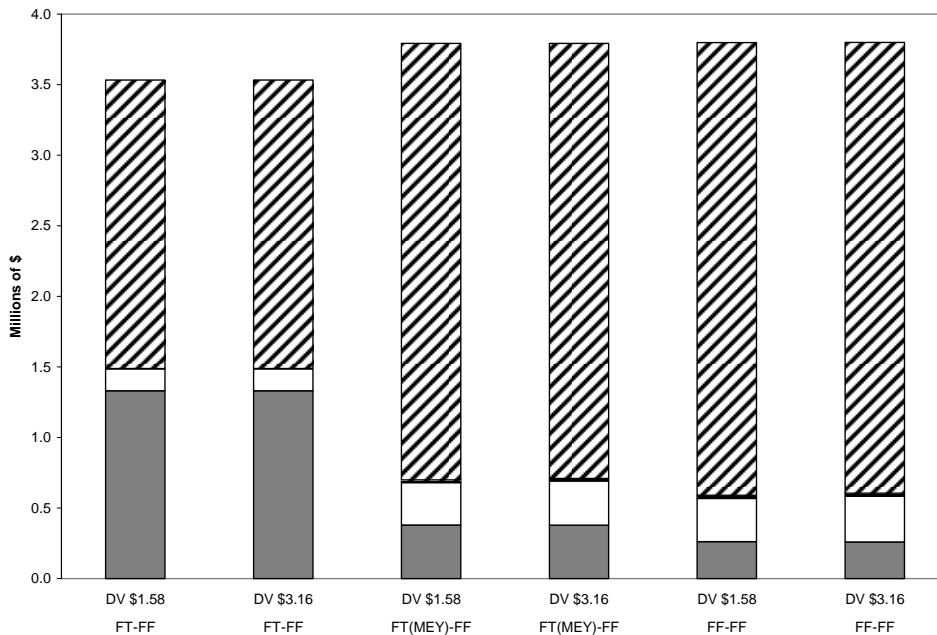
For the base case scenario, with a covariance of 0.5 and the red cod TAC fixed at MSY while the elephantfish TAC varies to maintain F_{MSY} , average annual total fishery rents are \$3.53 million. Total rents and their distribution are unchanged by doubling the maximum deemed value on elephantfish from \$1.58 to \$3.16. Although elephantfish accounts for about 20% of average gross revenues, elephantfish ACE revenues are only 4% of the total. Red cod ACE revenues are 38% of the total, and inframarginal profits account for 58% (Figure 2). No deemed value is paid. Results are changed only slightly with a lower covariance of stock distributions (e.g., 0.25 or 0). Total fishery rents are slightly lower, and the share of rents accruing to red cod ACE owners is a percentage point higher.

Unlike the previous case, the fixed red cod TAC that yields the highest total annual fishery rents is above, rather than below, the deterministic MSY. In fact, total fishery rents continue to increase up to the point where the TAC for red cod is increased by a factor of three. The higher TAC removes constraints on the catch of red cod when its biomass is very high and it is profitable to catch. Harvests costs are sufficient to ensure the stock is never overexploited. Total fishery rents with a fixed TAC for red cod set at three times the deterministic MSY and an F_{MSY} policy for elephantfish are \$3.8 million, roughly a 7% increase from the base case. The distribution of rents is substantially different from the base case. Red cod ACE revenues decrease from 38% to 10% of total rents, while elephantfish ACE revenues increase from 4% to 8%. Inframarginal profits increase to 82%.

Adding uncertainty in relative profitability across locations results in a small decrease in total rents as a result of higher harvest costs. The reduction is in inframarginal profits; ACE revenues for both species are fairly stable. For the base case, stock assessment uncertainty for elephantfish actually increases total fishery rents, and shifts some rents away from red cod ACE revenues into deemed value payments and elephantfish ACE revenues.

Although adjusting the TAC of red cod to maintain it at F_{MSY} is unrealistic, we consider this scenario for comparative purposes. The results are similar to the scenario with the red cod TAC fixed at three times MSY. Total fishery rents increase to \$3.8 million, and rents shift from red cod ACE revenues to inframarginal profits and elephantfish ACE revenues.

Figure 2: Average red cod (*Pseudophycis bachus*) annual catch entitlement (ACE) revenues (grey), elephantfish (*Callorhinchus milii*) ACE revenues (white), deemed value payments (black), and inframarginal profits (crosshatched) for Case 2 with covariance of 0.5 under three total allowable catch (TAC) setting scenarios each simulated with maximum deemed value on elephantfish at \$1.58 and \$3.16. Columns 1-2 have a fixed TAC for red cod set at maximum sustainable yield (MSY) and a variable TAC for elephantfish with fishing mortality set at the deterministic MSY (F_{MSY}). Columns 3-4 have a fixed TAC for red cod set at 3 times MSY and a variable F_{MSY} TAC for elephantfish. Columns 5-6 have a variable F_{MSY} TAC for both species.



DISCUSSION

The deemed value system provides an economic incentive to keep catches of bycatch stocks within TACs, but allows those TACs to be exceeded when the cost of reducing bycatch is particularly, or unexpectedly, high. This flexibility can increase the risk of overexploiting a bycatch stock and threatening its viability. A key determinant of this risk, as illustrated in both cases we examine, is whether the target exploitation rate of the bycatch is lower than that of the target stock. When the exploitation rate of the target species is higher than that of the bycatch species, and the distribution of the two species is highly correlated, it is possible to overexploit the bycatch species even without

targeting it. Raising the deemed value to the ex-vessel price of the bycatch species in response to overcatch theoretically eliminates incentives to target the bycatch species, but does not create incentives to actively avoid it. The bycatch species may be overexploited simply as a byproduct of harvesting the target species. The risk of overexploitation is aggravated if the fish become vulnerable to the fishery before they mature, as in the case of smooth oreo (our first case). If, however, there is a low spatial correlation between the target and bycatch species, eliminating incentives to target the bycatch should be sufficient to avoid overexploitation.

For the cases modeled, allowing deemed values to rise to twice the ex-vessel price in response to TAC violations eliminates the risk of overexploitation, as long as both TACs are adjusted annually to maintain fishing mortality at or below F_{msy} . Higher deemed values provide sufficient incentives for fishers to actively avoid bycatch and generally keep catches within the TACs, with only temporary exceptions. This is true even when the distribution of stocks is highly correlated, so that bycatch avoidance is costly. An exception to this may be if the catchability of the bycatch stock is higher than the target stock (i.e., in terms of the percentage of the fish biomass in a given area taken with a unit of fishing effort). The reverse relationship is actually more likely, since harvest gear has presumably been designed specifically to maximize catch of the target species. In that case, the risk of bycatch overexploitation would be reduced relative to our simulations.

Ideally, deemed values can be maintained below ex-vessel price so that they discourage intentional catches above the TAC, but maintain the incentive to land (rather than discard) unintended catch. For most stocks, deemed values have been kept below ex-vessel price for this reason. However, our simulations demonstrate it is sometimes necessary to raise deemed values well above ex-vessel prices to constrain bycatch. This can create strong incentives for illegal discarding. If not deterred, discarding could reduce fishery rents and increase risk of overexploitation. Furthermore, data quality may degenerate, making stock assessments less precise. When high deemed values create incentives for illegal discarding, it may be advisable to use on-board observers or other mechanisms to increase compliance with discarding prohibitions.

Fixed TACs for the bycatch stock generally increase biological risk, particularly if the bycatch species is vulnerable to fishing before maturity or if it is less productive than the target stock. Higher maximum deemed values reduce, but do not eliminate, this risk because depletion can occur even when catches are kept with the TAC. For the bycatch stock, total catch is likely to decline as stock declines, but under a fixed TAC there may actually be a decreasing incentive to avoid it. As biomass falls it becomes easier to keep catch within the TAC and the ACE price for the bycatch species will fall, reducing incentives to avoid it. Although it did not occur with any of the cases we modeled, the bycatch stock in theory could be driven to extinction if the TAC is not reduced when the stock declines.

Neither of our cases suggests that increasing deemed values to well above ex-vessel price in response to TACs violations reduces overall fishery rents. This is true even for cases where abundance of the target or bycatch stocks is highly variable. The deemed value policy has surprisingly little impact on combined fishery rents even when deemed value rates on the bycatch species are increased to twice the ex-vessel price. In some cases, combined fishery rents actually increase with higher deemed values because fishing intensity is reduced, bringing the fishery closer to MEY.

A more conservative harvest strategy for the target species, such as the MEY policy in Case 1, can increase total fishery rents as well as the share of rents accruing to ACE owners of the target species. A lower exploitation rate allows biomass of the target stock to grow, reducing the nominal effort required to take the TAC. Even if effort were randomly distributed vis-à-vis the bycatch stock, this would be expected to reduce total bycatch unless the bycatch stock had grown proportionately. The reduction in bycatch and potentially other environmental impacts of fishing, not to mention lower risk of overexploiting the target stock, provide an additional impetus to reduce exploitation rates below MSY levels. The simulations also suggest that quota owners of the target stock may substantially increase their share of fishery rents by supporting a more conservative harvest strategy. As Case 1 shows, an MEY policy not only increases overall fishery rents, but also results in a substantial shift away from inframarginal profits and bycatch ACE revenues and into ACE revenues for the target species.

If a stricter deemed value policy is imposed on a fishery where bycatch regularly exceeds the TAC, total fishery rents are likely to decline initially and rebound only as the target stock's biomass increases and harvest costs fall. Our simulations did not look at the present value of imposing a stricter deemed value policy on a fishery that had been operating under a more lenient one. For a slow growing target species, the benefits of a lower exploitation rate may not be realized for some time and the present value of the policy change could be negative despite an increase in long-run average rents. Higher maximum deemed values can also increase the variability of catches and rents over time. This is generally undesirable though the economic impact is hard to quantify.

The distribution of rents can be quite sensitive to the TAC setting policy and deemed value policy, even if total rents are not. This may be less important if all fishers have balanced quota portfolios, but generally there is heterogeneity in quota portfolios so there will be winners and losers. Stricter deemed value policies (i.e., with higher maximums) generally shift rents from owners of target species quota to owners of bycatch species quota.

New Zealand has recently introduced a number of species taken primarily as bycatch and is expected to introduce more in the future. In the past, bycatch of these species was essentially unregulated. Once they are in the quota system, fishers must balance their landings with ACE. Strict deemed value policies that create strong penalties for exceeding TACs can be expected to shift value out of quota for existing target species into quotas of these new species, even if they have little commercial value, simply because they constrain the ability to utilize target species quota. Since much of the quota for these new quota species is being auctioned rather than given to existing quota owners based on recent catch histories, the introduction of these new species to the quota management system may shift value from existing quota assets to the government.

With few exceptions, the simulations showed that higher deemed values rates actually reduce the total deemed value payments. Quota violations are elastic with respect to the deemed value price; the magnitude of the TAC violation decreases by a larger percentage than the deemed value rate increases. This might not hold if harvesters cannot easily control bycatch rates, but even then they might choose to reduce target catch to avoid punitive deemed values. Deemed value revenues currently accrue to the government in New Zealand. This creates a moral hazard problem for the government, which can increase its deemed value revenues by lowering the deemed value rate. By doing so, they effectively undercut the ACE price that would prevail if catch were strictly limited to the TAC.

As a benchmark, we assumed perfect information on the part of harvesters about the catches of both target and bycatch species across all locations, and the ability to correctly choose the most profitable options. Our simulations suggest that greater uncertainty in relative profitability of locations that could be caused by uncertainty in catch rates may increase the risks of breaching reference points. Uncertainty in relative catch rates across locations also reduces total fishery rents slightly, as one would expect, but this impact is small and the distribution of rents is virtually unchanged. In some cases, the actual uncertainty in catch rates, and consequently relative profitability, might be considerably higher than the levels we include in our sensitivity analysis. If uncertainty were sufficiently high, it might become necessary to reduce bycatch by forgoing use of target quota. Ironically, this might increase fishery profits if the target fishery TAC is higher than MEY. The impacts of uncertainty in stock assessments on biological risk and efficiency are ambiguous, and provide little insight into how this factor might influence either the risk or efficiency associated with various deemed value policies.

Our model implicitly assumes competitive markets cause bycatch ACE prices to converge on the marginal value of an increase in the TAC. This causes all fishers to consider the true opportunity costs of bycatch ACE when choosing between fishing alternatives, and leads catches of both species that maximize total fishery rents in a time period, given the TAC and deemed value constraints. In reality, the initial ACE price at the beginning of the season might be too high or too low but would adjust toward the market clearing level as the year went on. Too high an initial price would leave surplus quota, causing downward pressure on prices; a price set too low would provide insufficient incentives to reduce bycatch and would lead to catch rates that threaten to “bust” the TAC. This would create upward pressure on the ACE price by those who would otherwise have to pay deemed values (as long as the ACE price does not exceed the deemed value). As long as ACE prices are not systematically biased up or down from those arising under perfect information and competitive markets, the effect of this evolution of ACE prices during a season should be somewhat similar to the effect of uncertainty in relative catch rates; it effectively changes the relative profitability of fishing alternatives in the same way. This uncertainty tends to decrease fishery rents, and may increase the risks of overexploiting the bycatch stock.

In summary, we find that allowing TACs to be exceeded through a deemed value system will generally not create substantial risks as long deemed values are increased in response to TACs being exceeded and TACs are reduced in response to stock declines. However, it may be necessary to raise deemed values to well above ex-vessel prices in some cases which is likely to necessitate a strong compliance program to deter illegal discarding. Allowing deemed values to rise to well above ex-vessel prices is unlikely to reduce overall efficiency but will cause rents to shift from owners of target species quota to owners of bycatch species quota. This can be avoided by setting more conservative TACs for the target species.

These simulations modeled only cases with one target species and one bycatch species. Some of the conclusions drawn from this analysis may not apply to more complex multispecies fisheries. For example, if there are multiple bycatch species that vary independently, this may increase the costliness of keeping bycatch below TACs since avoiding bycatch of one species could increase bycatch of another. Modeling deemed value policies for fisheries with more than 2 species would be a useful extension of this analysis.

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