Economic Effects of Multispecies Catch Share Management

Andrew M. Scheld

A dissertation
submitted in partial fulfillment of the
requirements for the degree of

Doctor of Philosophy

University of Washington
2014

Reading Committee:
Christopher M. Anderson, Chair
David F Layton
Trevor Branch
Hirotsuga Uchida

Program Authorized to Offer Degree:
School of Aquatic and Fishery Sciences

University of Washington

Abstract

Economic Effects of Multispecies Catch Share Management

Andrew M. Scheld

Chair of the Supervisory Committee:
Associate Professor Christopher M. Anderson
School of Aquatic and Fishery Sciences

Catch share management is a common and increasingly relied upon form of fisheries management in which, frequently, shares of a hard total allowable catch are allocated to individuals or groups of harvesters. A style of rights-based-management, catch shares are often thought to promote efficient resource use and long-term stewardship, improving both economic and ecological conditions within the fishery. Their success in multispecies fisheries, where non-selective gear captures several, separately managed stocks, has been the subject of ongoing debate however. Limited flexibility in production might dampen or entirely remove any and all of catch shares' potential benefits as harvesters are unable to effectively target or avoid individual stocks. In this dissertation, I explore the economic effects of multispecies catch shares following a 2010 application to New England groundfish, a diverse and overexploited multispecies fishery. In the first chapter, I combine market models of ex-vessel inverse demand and counterfactual models of individual harvesting behavior to estimate the market timing benefits of catch share management. I find that fleet revenues were improved by over US $30 million and that individual benefits were heterogeneously distributed, with large and more diverse operations better able to take advantage of market externalities. In the second chapter, I theoretically develop and empirically explore a model of costly avoidance wherein production of target stocks is given up to reduce that of the avoided. An error in the management of pollock, initially setting a low and constraining allocation that was later relaxed, is used to identify behavioral response to multispecies production constraint, finding harvesters engaged in costly avoidance strategy, which had the low pollock allocation persisted, would have cost the fleet US $3 million. I then develop a neoclassical multispecies production technology in the third chapter which is used to test technological restrictions of strong disposability on pairs of demersal species and also estimate the costs, in terms of forgone production, of pollock avoidance. For catch share regulated species, strong disposability is rejected more than half the time, suggesting output controls may frequently lead to choked production. Additionally, pollock avoidance costs are estimated at US $6 million.

Table of Contents

Page

List of Figures

Page

List of Tables

Page

Acknowledgments

I am deeply appreciative of all the support provided by my advisor, Christopher Anderson, whom I consider a mentor as well as a friend. Over the last several years I have been extremely fortunate to have been surrounded at all times by incredibly bright, talented, and thoughtful individuals who have contributed in many ways, large and small, to this dissertation. I would like to thank Trevor Branch, David Layton, and Hirotsugu Uchida for their constructive criticism, insightfulness, energy and time spent as part of my dissertation committee, Zhi Li for innumerable thought-provoking conversations, and Patricia Berkman for her continued love, support, and selflessness.

To Anne Patricia Munz

Chapter 1

Market effects of catch share management

Abstract

In 2010, management of New England multispecies groundfish transitioned from input restrictions on harvester effort to collective rights-based management. Faced with a large reduction in harvesting days, 432 active vessels, representing 98% of historical landings, joined one of the 17 sectors allocated catch shares. The incentives presented under sector management, combined with regulations of several separately managed, revenue-important species, led to changes in harvest strategies and the timing of landings for both multispecies groundfish and many other species targeted by the sector vessels. Temporally modified landings altered the ex-vessel market mix of different species throughout the fishing year, significantly impacting prices received as well as annual harvester revenues. Two counterfactual individual harvester landings' timing scenarios for 25 species are combined with independent fixed effects models of inverse dealer demand in estimating the revenue effects of catch shares during their first year. Aggregate gains of over $30,000,000 were found to result from advantageous market timing changes brought on by more flexible catch share management.

1.1 Introduction

Catch share management is a broadly defined fishery regulatory mechanism in which individuals or groups are provided annual harvesting privileges to a portion of the total allowable catch (TAC). The use of catch shares globally – typically in the form of individual transferable quotas (ITQs) – has been steadily increasing in recent decades, though their impact on fishery resources, fishing communities, and individual fishermen has been a topic of ongoing empirical and theoretical debate (for a brief overview of this discussion as it pertains to the use of ITQs see Sumaila 2010). Applications in multispecies fisheries can raise additional challenges as low allocations for depressed stocks may restrict harvest of abundant species and incentivize discard of limiting species (Copes 1986; Squires et al. 1998; Turner 1997). Nevertheless, the National Oceanic and Atmospheric Administration (NOAA) declared catch shares a crucial tool in revitalization of biologically and economically underperforming United States (US) fisheries, initiating a recent wave of catch share programs (NOAA 2010). Empirical evaluation of these programs is vital in connecting circumstance with biological, economic, and social results so that existing programs can be refined while new programs may be designed more effectively.

The Northeast (NE) multispecies groundfish fishery, a historically prominent New England resource, was recently transitioned from days-at-sea (DAS) to catch share management under Amendment 16 to its Fishery Management Plan (FMP). Overlapping habitat and the use of non-selective fishing gear had motivated individual effort control with DAS management, but lack of avoidance incentives exacerbated overfishing on depressed stocks and resulted in a perpetual shrinking of allocated days, which were calibrated by allowable catch of the weakest stocks. Faced with strict stock rebuilding time tables and required by law to implement hard TACs, managers provided multispecies harvesters with two options: join a harvesting cooperative (“sector”) and receive group allocations for individual multispecies stocks, or face severe cuts in individual DAS while harvesting under common pool TAC. Nearly all active permit holders with substantial historical landings joined sectors.

Evaluating the economic performance of the NE multispecies catch share program requires understanding the induced changes in harvest and landings strategy, as well as the concomitant effects on revenues and costs. An initial performance report (Kitts et al. 2011) indicated NE groundfish stocks yielded more nominal value per pound at lower effort levels during the first year of catch shares than in any of the previous three years under DAS. This finding comports with observations that tradable quota-based programs increase profitability (e.g., Arnason 1993; Casey et al. 1995; Dupont and Grafton 2000; Gauvin et al. 1994): more ex-vessel revenue can be drawn from a similar or smaller quantity of fish harvested at potentially lower cost. As of 2011, ex-vessel price improvements had been documented for species managed under all but two of the 15 US catch share programs (NOAA 2011).

In these fisheries, management-based revenue improvements have been driven by two mechanisms: enhanced quality made possible by mitigating the race-to-fish (Homans and Wilen 2005), and price increases due to improved market timing (Scheld et al. 2012). Competitive harvest pressure producing the race-to-fish results from common pool TAC management and is rarely seen in fisheries managed with input restrictions such as DAS. Under multispecies DAS a large volume of high quality fish regularly supplied the premium market, limiting opportunity for revenue gain through quality improvements. Additionally, market flooding, producing low average prices, was relatively infrequent for multispecies stocks under DAS given the lack of race incentives. From whence, then, did the observed price increases following the introduction catch shares come?

Historically reactive and piecemeal, US fisheries management has given rise to many instances of separate, very different, and disjoint regulation for various species exploited by the same group of fishermen. New England groundfish harvesters target and land over 20 different species which are regulated by nearly a dozen different management plans. Many of these species are harvested jointly with non-selective trawl gear. Technical interaction among separately regulated stocks forces harvesters to make trip-level targeting and/or avoidance decisions, evaluating tradeoffs of participation in one fishery versus another. Multispecies catch shares introduced new management to a subset of target species; given the presence of numerous separately regulated, jointly harvested stocks, it is expected the transition from DAS altered exploitation patterns for the broader complex harvested by the fleet.[1]

Prior to multispecies catch shares, harvesters were allocated a limited number of DAS during which to land multispecies stocks. Scarcity of this harvest privilege promoted strategic use: if a DAS were to be used, it would be in the harvester's best interest to be sure to land a large quantity of multispecies fish. Trips landing primarily non-multispecies stocks along with small amounts (but exceeding incidental catch limits) of multispecies fish carried a large opportunity cost in foregone multispecies landings. This trip-level constraint on harvest and landings of non-multispecies fish was relaxed under catch shares, which allow harvesters to continuously vary the ratio of multispecies to non-multispecies fish according to market prices, harvest costs, and biological availability. The introduction of catch share management therefore enhanced flexibility in landings, increasing the range of profitable multispecies/non-multispecies landing combinations.

The resulting changes in multispecies and non-multispecies landings altered the mix of species being processed and sold down the supply chain at each point in time, ultimately

---

[1] Spillover effects (Asche et al. 2007; Branch 2009) are a related, though distinct, phenomenon.

affecting ex-vessel prices as dealers reacted to modified market conditions. The extent to which adjusted landings by one individual responding to new regulatory incentives affect other harvesters will depend on price response to quantity changes, or price flexibility. By determining the change in timing of individual harvests and identifying market relationships between quantities and prices, controlling for exogenous price determinants, a robust prediction of the revenue effects resulting from such a policy change may be estimated.

The analysis presented in this article investigates the impact of multispecies catch shares on individual permit revenues, across all harvested species.[2] Two counterfactual sets of individual daily landings for 25 revenue important species were constructed to predict what harvesters would have landed each day, absent multispecies catch share management.[3] Species-level models of ex-vessel dealer demand were used to predict prices given actual or counterfactual landings. The difference between actual and constructed counterfactual predicted revenues was used as an estimate of multispecies catch shares' ex-vessel revenue effect for each individual harvester.

1.2 New England fisheries management

The NE multispecies FMP manages 15 species: Atlantic cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), pollock (*Pollachius virens*), Acadian redfish (*Sebastes fasciatus*), yellowtail flounder (*Limanda ferruginea*), American plaice (*Hippoglossoides platessoides*),

---

[2] Limited access multispecies harvesting permits are attached to fishing vessels. In what follows the terms vessel and harvester will be considered synonymous with permit.

[3] In addition to the nine sector allocated and two small mesh multispecies species, this analysis modeled multispecies harvester landings for: monkfish (*Lophius americanus*), black sea bass (*Centropristis striata*), butterfish (*Peprilus triacanthus*), Atlantic herring (*Clupea harengus*), longfin squid (*Loligo pealeii*), American lobster (*Homarus americanus),* Atlantic mackerel (*Scomber scombrus*), Atlantic Sea scallop (*Placopecten magellanicus*), scup (*Stenotomus chrysops*), northern shrimp (*Pandalus borealis*), little skate (*Leucoraja erinacea*), winter skate (*Leucoraja ocellata*), summer flounder (*Paralichthys dentatus*), and spiny dogfish (*Squalus acanthias*). Little and winter skate landings were modeled as generic wings and rounds to align with management delineation. The top 25 revenue important species made up on average 97% of annual fleet revenues.

witch flounder (*Glyptocephalus cynoglossus*), winter flounder (*Pseudopleuronectes americanus*), white hake (*Urophycis tenuis*), Atlantic halibut (*Hippoglossus hippoglossus*), windowpane flounder (*Scophthalmus aquosus*), ocean pout (*Zoarces americanus*), Atlantic wolffish (*Anarhichas lupus*), whiting (*Merluccius bilinearis*), and red hake (*Urophycis chuss*).

Since the New England Fishery Management Council's establishment in 1976, multispecies management has been a succession of relative failures (Apollonio and Dykstra 2008). The NE multispecies FMP has at this time 18 amendments and 50 framework adjustments, representing major and minor changes respectively. Intense bureaucracy and diverse stakeholder interests have produced regulations that are often too little too late, cultivating a sentiment of inept management among industry and local communities while simultaneously squandering multispecies populations and the profits they could produce (Acheson and Gardner 2011). Increasingly restrictive management has forced many harvesters originally solely dependent on multispecies groundfish to diversify, creating a complex web of harvesting alternatives. The remaining fleet is a heterogeneous group of harvesters who target a variety of stocks. Vessels range from small hook-and-line operations that fish near shore on single day trips to moderate-scale (20-30 meter) trawlers that fish grounds over a hundred miles offshore and may be at sea for over a week.

Beginning in 1994, multispecies groundfish was regulated with limited entry joint stock DAS. Under this management regime, harvesters were charged with using one of their allocated DAS if landed multispecies volume was above incidental catch allowances.[4] The collapse of many multispecies stocks coincided with an emergence of new export and domestic markets for demersal species like spiny dogfish, monkfish, and skate, which commix with, and are often caught when targeting, multispecies groundfish. The newfound marketability of these stocks,

---

[4] In general, a per stock incidental catch of 5% total landed weight was allowed without expending a DAS.

which had previously been discarded to free up hold space for valuable species like cod and yellowtail flounder, presented harvesters with an important trip level decision regarding targeting, retained catch, and DAS use.[5]

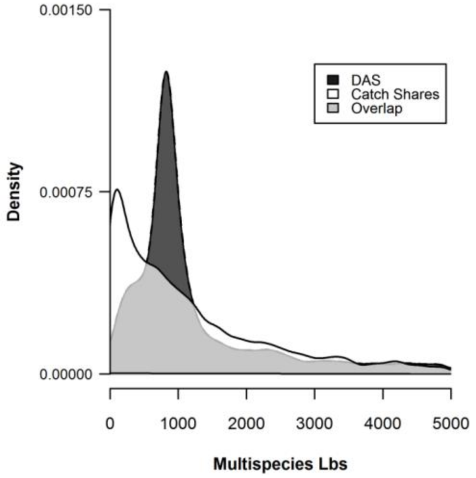


Figure 1.1 Kernel density plots of individual vessel multispecies landings made when also landing commixing non-multispecies stocks under DAS and catch shares. Catch shares density (translucent white) overlays DAS (dark grey). Only landings < 5,000 lbs are shown. Incidental catch under DAS has been removed.

On May 1st 2010 – the first day of fishing year (FY) 2010 – NOAA allocated catch shares for 14 stocks of nine groundfish species, collectively referred to as “multispecies” throughout this article.[6] The 17 sectors, or self-identifying groups of harvesters, were allocated collective quota based on members’ landing histories. The 432 participating vessels represented 55% of the eligible limited access multispecies fishing permits and 98% of historical multispecies landings.

---

[5] It should be noted that regulatory noncompliance throughout this period was commonplace, calling into question the degree to which regulation incentivized behavior (King and Sutinen 2010). Still, since 2004 an active DAS lease market has ensured harvesters operate on the margin with respect to DAS use.

[6] Vessels were allowed incidental bycatch of one halibut per trip, while there was a prohibition on landings of windowpane flounder, ocean pout, and Atlantic wolffish. Whiting and red hake were small-mesh multispecies regulated in 2010 with gear restrictions and trip limits

Though afforded considerable management autonomy, all approved harvesting sectors operated with a loose form of ITQs, where individual vessel allocations were determined by contributions to the group aggregate. Quota was tradable both within and between sectors. Figure 1.1 captures an important behavioral consequence of multispecies DAS management not present following the transition to catch shares: under DAS, small landings of multispecies stocks made jointly with commixing non-multispecies fish were avoided or illegally discarded. Catch share management facilitated proportionately more low-quantity multispecies landings when targeting non-multispecies stocks, an indication of increased flexibility in landings that may lead to greater profitability.

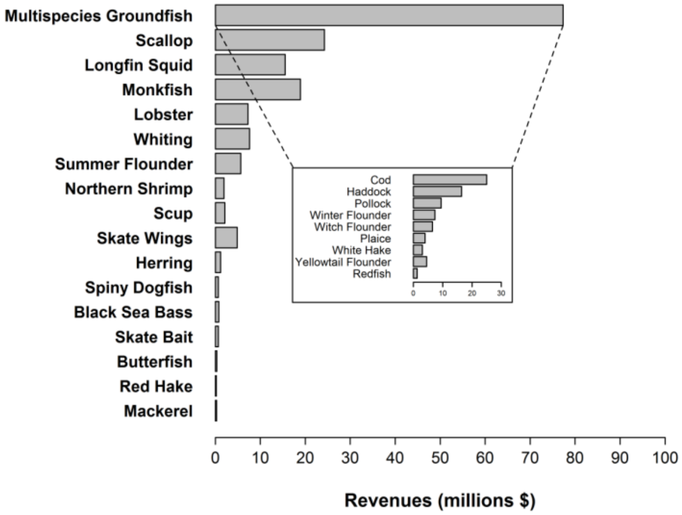


Figure 1.2 Average annual total revenues by species for multispecies sector vessels before catch shares (FY's 2007 – 2009). Bars are ordered according to species revenue ranking after catch shares (FY 2010).

Under DAS, limited access multispecies permit holders developed harvesting patterns that varied in their dependence on the multispecies complex; in aggregate, revenues from multispecies groundfish made up slightly less than half of the average annual sector fleet total during the three years preceding catch share management (Figure 1.2). Throughout this period, lobster and scallop made up majority revenue shares for 15% of permit holders. These vessels landed multispecies groundfish as bycatch or secondary targets, relying on the species complex for less than 10% of annual revenues. The financial importance of multispecies groundfish varied regionally for the remaining 85% of the fleet. The majority of multispecies permits (60-70%) were held by harvesters in northern New England (Maine, New Hampshire, and Massachusetts) who relied heavily on multispecies stocks (70% of annual revenues) and to a lesser extent, monkfish (10-20%) and skate (5%). Permit holders south of Massachusetts were more diverse, having annual revenues made up of longfin squid (30-40%), whiting (15-20%), multispecies groundfish (10-20%), summer flounder (10%), monkfish (5%), and scup (5%).

Following the introduction of catch share management, the sector fleet did not dramatically modify annual harvest strategies. Figure 1.2 indicates that revenue rankings during the first year of catch share management were similar to their prior ordering under DAS. As evidenced by a mismatch in descending order and bar length, longfin squid supplanted monkfish to become the third most important revenue source for sector harvesters after catch shares. Other shifts in revenue importance were more subtle, and an overwhelming majority of species saw no considerable change. At the root of this consistency were fairly stable individual vessel annual landings before and after multispecies catch shares. T-tests indicate that, on average, vessels landed significantly more lobster and summer flounder – species which saw increases in abundance – while less skate. Additionally, three of nine multispecies fish experienced statistically significant change in average annual ex-vessel volume. For all modeled species however, shifts in inter-annual sector vessel landings were not statistically different from those experienced by limited access multispecies vessels that did not join sectors. Though it appears that in their first year multispecies catch shares did not meaningfully affect the amount of fish landed by individual harvesters, as previously noted, incentives dictating how and when particular species were pursued and landed did importantly change.

The behavioral effects of multispecies catch shares were driven by landings of non-multispecies stocks under disjoint management regimes. For example, early in the season there was a noticeable increase in harvest of possession-limit regulated groundfish species jointly landed with small amounts of multispecies groundfish, a targeting choice that is less costly under the new catch share program. Under DAS, these low-multispecies-landing trips were either forgone in favor of less profitable trips, supplemented with additional multispecies tows, or multispecies catch in excess of the incidental threshold was discarded. Similarly, there was an increased pace in exploitation of spiny dogfish, summer flounder, and scup – three separately managed race-to-fish demersal stocks. In FY 2010, during each respective derby, these three commixing species were landed by sector vessels in higher daily volumes than was observed under DAS. Here again, the ability to profitably land a small quantity of multispecies groundfish appears to have modified harvest and landings strategies, facilitating greater participation in the race-to-fish for these commixing non-multispecies stocks. Finally, there were changes in sector landings of stocks which do not commix, or are harvested with different technology, likely motivated by exogenous market and non-market forces such as non-sector harvests, weather, and regulatory structure.

1.3 Analysis

To estimate the individual harvester revenue impacts of the NE multispecies catch share program, we took a three-phase approach following Scheld et al. (2012). First, a 25 equation price model was fit using ex-vessel landings data. Second, two counterfactual environments were constructed to predict the daily landings of each sector harvester. Third, the price model was used to predict ex-vessel revenues under actual and counterfactual landings. The difference in revenue predictions is a measure of catch shares' ex-vessel revenue effect.

1.3.1 Data

In the NE US all business entities buying federally managed fish directly from vessels are required to electronically report the species, weight, market category, grade, and transaction value of each ex-vessel sale. The data used in fitting price models consisted of 1.6 million dealer reported ex-vessel species landing observations from January 1st 2007 through April 30th 2010. There were approximately 375,000 individual species landing observations by multispecies sector vessels during FYs 2009 and 2010 which were used in assessing the effects of catch share management.

1.3.2 Price model

The price model is designed to predict ex-vessel prices at given landed quantities, facilitating comparison of actual and counterfactual revenues. That price and quantity are jointly determined in competitive market equilibrium is perhaps the most fundamental concept of microeconomic theory. Empirical studies of ex-vessel markets have historically relied on models of inverse demand, where price is the dependent variable and quantities the independent variables (Barten and Bettendorf 1989; Bell 1968). Under this framework the modeler must identify a market's spatial scope as well as substitute and complementary products (for consumers or processors) for which markets are integrated. There is evidence for statistically significant market integration of certain seafood products at national (Asche et al. 1997; Gordon and Hannesson 1996) and international scales (Asche et al. 1999; Asche et al. 2002), as well as long-run substitutability between different species in wholesale markets (Bose and McIlgorm 1996; Gordon et al. 1993; Shabbar et al. 1999).

In New England, many secondary fish dealers, processors, restaurateurs, and grocers look to maintain a steady supply of fish, often purchasing available products at centralized markets supplied by ex-vessel dealers or having them shipped from regional ports (for a detailed description of this market which, aside from a degree of consolidation, has not dramatically changed in structure for several decades, see Wilson 1980). This suggests that ex-vessel prices are determined by landings at the individual dealer, and also influenced by total same- and different-species quantities in the port, across the state, and within the region. The broad spatial and product scope of this model is needed for several reasons. First, though specialization exists to a degree, New England ex-vessel wholesale dealers are product generalists: in FY 2009 the top 100 revenue grossing dealers bought (and sold) 17 species on average. Processing and distribution require use of scarce inputs (e.g., skilled labor, refrigeration, fuel, etc.) creating production tradeoffs, or possibly complementarities, between species. Second, different seafood products are often similar in protein content, taste, texture, and availability and may be substituted for one another by end consumers. Third, a sizable volume of fish is transacted in two large daily auctions which distribute price reports to all dealers, providing reliable market signals throughout the region. Note however that transportation costs, time and perishability, as well as local availability and cost of production inputs, limit coast-wide market integration, preventing the law of one price from holding ubiquitously (Linnemann 1966).

Scheld et al. (2012) identified statistically significant cross- and own-species price flexibilities for daily aggregate statewide landings in New England, indicating extensive market relationships among a variety of ex-vessel products. For each of the 25 species pursued by sector vessels, our price model estimates each landing's ex-vessel price to be a function of regional, port, dealer, and individual harvester landed quantity aggregations across all modeled species (see Appendix A for further discussion and a technical description).

1.3.3 Counterfactuals

Because multiple states of nature cannot be simultaneously observed (Holland 1986), policy analysts are presented with two options in estimating the relationship between cause and effect: randomized controlled experiments, where the investigator is able to impose variation of conditions and use statistical inference to draw conclusions, or quasi-experimental procedures, where statistical methods use existing data, before and after the treatment, to extract information about the effect (Greenstone and Gayer 2008). As a result of complex context and often ex-post analysis, evaluation of environmental policy and management actions tend to rely on methods of the latter (Bennear and Coglianese 2005; Ferraro 2009).

Until recently, evaluation of fishery management decisions using robust counterfactuals has largely been absent from the literature (Smith et al. 2006). Early work investigating the economic impacts of rights-based management used associational evidence (Batstone and Sharp 1999; Casey et al. 1995; Dupont and Grafton 2000), analyzed quota market performance (Newell et al. 2005), or extrapolated aggregate behavior (Herrmann 2005; Homans and Wilen 2005). Not controlling for exogenous factors (e.g., climate, ecology, and macroeconomy) and/or muting fleet heterogeneity, such strategies may fail to isolate a policy's true treatment effect and instead produce results driven by spurious correlations or weakened through aggregation.

The analysis presented here follows Scheld et al. (2012) in estimating $Effect_i$ from equation (1.1). Rev_i^A are the estimated actual (*A*) revenues for harvester *i*, generated from price predictions using actual landings, while Rev_i^{CF} are the estimated counterfactual (*CF*) revenues for harvester *i*, generated from price predictions using counterfactual estimates of what landings would have been had DAS remained in effect during FY 2010. The difference between Rev_i^A and Rev_i^{CF} is an estimate of multispecies catch shares' effect on harvester *i*'s revenues.

$$Effect_i = Rev_i^A - Rev_i^{CF} \tag{1.1}$$

Two methods of counterfactual construction relying on individual pace of landings in FY 2009 were used in predicting Rev_i^{CF}:

$$\sum_{\tau=1}^{t} q_{ij\tau}^{10CF} / \sum_{\tau=1}^{365} q_{ij\tau}^{10} = \sum_{\tau=1}^{t} q_{ij\tau}^{09} / \sum_{\tau=1}^{365} q_{ij\tau}^{09} \tag{1.2}$$

$$\sum_{\tau=1}^{t} q_{ij\tau}^{10CF} / \sum_{\tau=1}^{365} q_{ij\tau}^{10} = \left[\sum_{\tau=1}^{t} q_{ij\tau}^{09} / \sum_{\tau=1}^{365} q_{ij\tau}^{09}\right] \frac{\sum_{\tau=1}^{t} Q_{j\tau}^{10CP} / \sum_{\tau=1}^{365} Q_{j\tau}^{10CP}}{\sum_{\tau=1}^{t} Q_{j\tau}^{09CP} / \sum_{\tau=1}^{365} Q_{j\tau}^{09CP}}. \tag{1.3}$$

These relationships offer alternative ways to identify q_{ijt}^{10CF}, harvester *i*'s landings of species *j* at time *t* in FY 2010 under the counterfactual DAS policy. Equation (1.2) models counterfactual landings to be equivalent in pace with those during FY 2009, under DAS management. Specifically, on each day *t* of FY 2010, the sum of counterfactual daily landings (*q*) for harvester *i* of species *j* in proportion to total FY 2010 landings by harvester *i* of species *j* was set equal to a corresponding ratio of FY 2009 landings. Rearranging (1.2) and solving for $\sum_{\tau=1}^{t} q_{ij\tau}^{10CF}$, one obtains harvester *i*'s counterfactual cumulative landings of species *j* at each day *t*, from which counterfactual daily landings are calculated.[7] For robustness, a second method was also used. Equation (1.3) provided counterfactual estimates equivalent to those derived from (1.2) weighted by the pace of aggregate daily landings (Q) by multispecies permit holders who did not join sectors and thus fished in the common pool (CP) under DAS.[8] The second right hand ratio term in (1.3) measures the relative change in timing of aggregate common pool DAS landings. If species *j* was being landed relatively faster in FY 2010 by these vessels this term would be greater than one, increasing the pace of species *j* counterfactual landings for each sector harvester. The addition of this term was used to control for exogenous climatic, ecological, or regulatory shocks affecting all harvesters in FY 2010 but not FY 2009.[9]

1.4 Results

1.4.1 Price and counterfactual models

Equation (1.4) defines the price flexibility (F_{jm}) of species *j* with respect to landings of species *m* as the ratio of a percentage change in species *j* price to a percent change in species *m* quantity.

$$F_{jm} := \frac{\partial P_j / P_j}{\partial Q_m / Q_m} \tag{1.4}$$

An F_{jm} value between zero and negative one indicates the price received for species *j* decreases when species *m* quantity increases, though the change in price is less than proportionate to that in quantity.

---

[7] If harvester *i* did not make any landings on the same day *t* in both years, counterfactual landings were assigned to the next landing day observed. Harvesters landed six fewer days on average during FY 2010.

[8] While this group of ~350 harvesters had somewhat insignificant multispecies groundfish landings, they made up significant components of other revenue important fisheries.

[9] Note that both methods control for general macroeconomic effects.

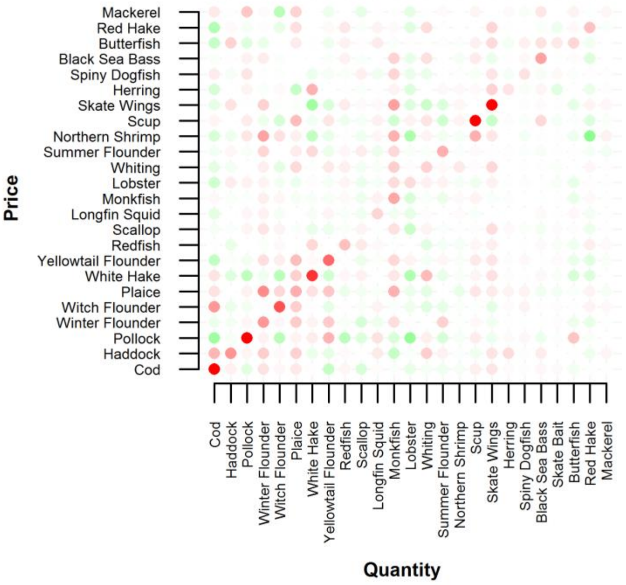


Figure 1.3 Total price flexibilities are calculated as a weighted sum of statistically significant ($P \leq 0.05$) individual price flexibilities. Red (green) indicates negative (positive) total price flexibility; color is scaled to the maximum absolute total flexibility value ($F_{pollock,pollock} = -0.4860$). Species are ordered left-to-right/bottom-to-top by FY 2010 revenue importance among multispecies groundfish (southwest corner) and other species.

Total price flexibilities were constructed from the estimated inverse demand model by summing individual price flexibilities across quantity aggregations.[10] Own total price flexibilities were predominantly large and negative, while cross total price flexibilities were comparatively smaller and a mixture of negative and positive (Figure 1.3).[11] The former implies downward sloping dealer demands (i.e., when more of a product is landed, ex-vessel prices decrease) while

---

[10] During summation, port, dealer, and individual harvester quantity aggregations were multiplied by$(1 - CV_j^X)$, or one minus the coefficient of variation in normalized spatial or individual landings distributions for species j at quantity aggregation X. This weighting accounts for variation in ex-vessel sales across space, dealers, and harvesters. Total price flexibility therefore indicates the average percent change in species j ex-vessel price from a 1% increase in same day landings, inventory levels, and expectations of species m quantity. Total price flexibilities were constructed for illustrative purposes only – price predictions relied on disaggregate values.

[11] Note that for graphical interpretative purposes, skate bait price flexibilities are absent from figure 3. Skate bait is an extremely low-value product; small price changes are therefore quite large in percentage. To better gauge color scale these price flexibilities were excluded.

the latter captures the intricacies and complexities of the ex-vessel market structure. All estimated price flexibilities were between positive and negative one ($\bar{F}_{jm} = -.0008$), indicating ex-vessel prices are relatively inelastic. Statistically significant cross-species price flexibilities may arise from competing or complementary use of scarce processing resources or end consumer substitution.

Counterfactual landings were modeled to reflect how individual sector harvesters would have landed each included species throughout FY 2010 had catch share management not been introduced. Methods used altered only the individual pace of actual FY 2010 landings and therefore do not consider changes in harvest scale, which were found to be minimal. Divergence in timing of actual and counterfactual landings by the fleet varied species to species, though spiny dogfish and whiting, two high volume non-multispecies stocks, were landed at a substantially faster initial pace than predicted by counterfactuals. Additionally, haddock – the healthiest multispecies stock for which ample allocation was provided – was landed earlier in the season than predicted, while yellowtail flounder and pollock were landed later (see Appendix A for further discussion of price and counterfactual model results).

1.4.2 Individual average and aggregate estimated impacts

To estimate the effect of temporal shifts in landings on individual harvester revenues, inverse demand and counterfactual models were combined to produce three sets of revenues: Rev_i^A, harvester i's predicted revenues with actual landings; Rev_i^{CRCF}, harvester i's predicted revenues with counterfactual landings using equation (1.2), referred to as catch rate counterfactual (CRCF); and Rev_i^{CPCF}, harvester i's predicted revenues with counterfactual landings using equation (1.3), referred to as common pool counterfactual (CPCF). The difference between Rev_i^A

and each Rev_i^{CF} is a measure of the impact multispecies catch shares had on harvester i's annual revenues. The summation of this difference over i provides an estimate of total program effects.

Table 1.1 Average and total estimated revenue effects of multispecies catch shares for sector vessels from CRCF and CPCF. Standard errors were derived from 1,000 draws of $\beta \sim N(\beta,\sigma^2)$ and are in parentheses beneath estimates. All estimates are significant at a 99% confidence level.

Effect ($)	Multispecies (CRCF)	Other (CRCF)	Multispecies (CPCF)	Other (CPCF)
Average	+ 37,020 (2,585)	+ 42,182 (6,179)	+ 54,581 (2,742)	+ 41,429 (7,977)
Total	+ 1.55 x 10^7 (1.08 x 10^6)	+ 1.77 x 10^7 (2.59 x 10^6)	+ 2.29 x 10^7 (1.15 x 10^6)	+ 1.74 x 10^7 (3.34 x 10^6)

Following the introduction of catch shares, change on the intensive margins of production was found to be enormously beneficial for the multispecies sector fleet. Statistically significant estimated total gains of $33,200,000 and $40,200,000, with individual harvester averages of $79,202 and $96,010, were predicted by CRCF and CPCF, respectively (Table 1.1). During FY 2010, total volume landed by multispecies sector harvesters was only 1% above the prior three year average, yet actual ex-vessel revenues rose by approximately 18%, from roughly 170 to 200 million US dollars. The models presented here predict FY 2010 ex-vessel revenues would have fallen had DAS remained in place, without even accounting for the potentially large loss in total landed volume brought on through necessary DAS reductions. This analysis suggests observed price and revenue increases for the multispecies sector fleet were the direct result of landings' timing and compositional changes made subsequent to the introduction of catch share management for a subset of target species.

1.4.3 Distribution of estimated impacts

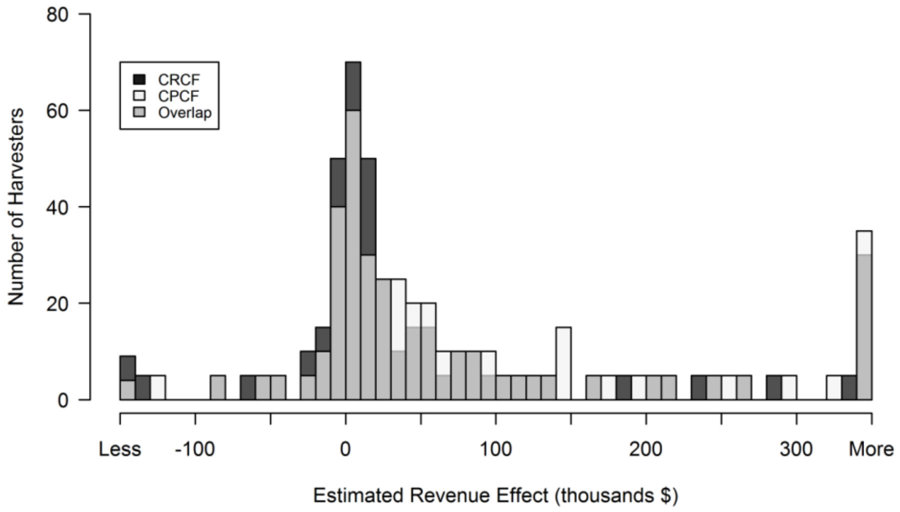


Figure 1.4 Histograms of individual harvester program effects from CRCF (dark grey) and CPCF (translucent white). Bin width is set to $10,000 and predicted effects < -$150,000 and > $350,000 are collapse into “Less” and “More” respectively.

Positive impacts were spread throughout the multispecies sector fleet, with gains estimated for 75-80% of individual vessels (Figure 1.4). Both counterfactual methods predict similar distributional results: a high concentration of slightly positive effects with a long positive tail. The 20-25% of sector harvesters predicted to have suffered losses were vessels from Maine and Massachusetts who relied heavily on monkfish and lobster in addition to multispecies groundfish. Conversely, the roughly 10% predicted to have gains in excess of $350,000 were large southern New England operations harvesting primarily scallop, squid, whiting, cod, and haddock.

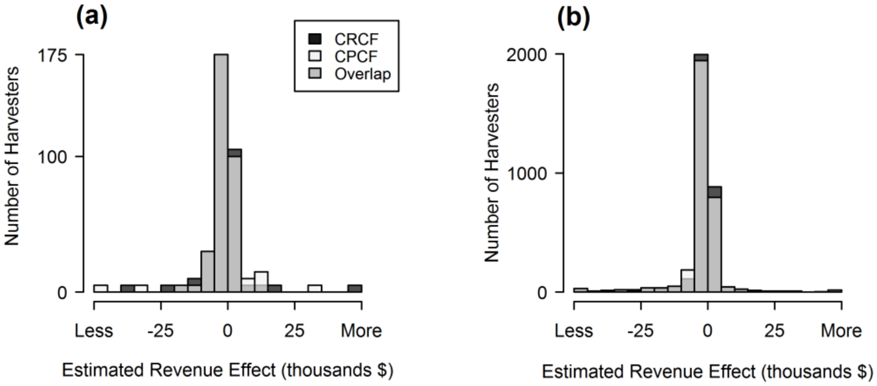


Figure 1.5 Histograms of individual harvester program effects for multispecies DAS common pool (a) and non-multispecies (b) vessels from CRCF (dark grey) and CPCF (translucent white). Bin width is set to $5,000 and effects < -$50,000 and > $50,000 are collapse into "Less" and "More" respectively.

Though this analysis focused on estimating program impacts for harvesters transitioned from DAS to catch share management, program effects for non catch share vessels were also estimated (Figure 1.5). Vessels landing in the NE US during FY 2010 not operating under multispecies catch shares were one of two types: multispecies common pool DAS and non-multispecies harvesters. The former group was ~350 vessels comparable with sector harvesters in scale and geographic distribution, while the latter group was comprised of over 3,200 generally smaller operations. Both groups relied primarily on monkfish, lobster, scallop, summer flounder, whiting, and herring for ex-vessel revenues. On average and in total, both groups were predicted to have been slightly worse off.

1.4.4 Effect of diversity in species landed

Table 1.2 Select output from OLS regressions of $Effect_i$ (Equation 1.1) on harvester FY 2010 total revenue, a measure of species diversity in annual harvest, and dummy variables indicating state of permit registration. Species diversity is calculated as $\hbar^i = \sum_{j=1}^{m} \theta_{ij}^2$ where θ_{ij} is the share of harvester i's annual landings made up by species j. Regressions were run for predicted effects from both CRCF and CPCF. Standard errors are beneath coefficient estimates.

Variable	Coefficient (CRCF)	P-value	Coefficient (CPCF)	P-value
Intercept	68,321 (21,132)	0.001	71,381 (24,407)	0.004
Total Revenue	.1528 (.0177)	0.000	.2232 (.0205)	0.000
Herf. Index	-106,109 (29,164)	0.000	-134,316 (33,684)	0.000

One factor influencing the extent of individual benefit from catch share management was the vessel's diversification in landings from different fisheries. A Herfindahl-Hirschman index, also called the Simpson index, is a bounded sum of squares measure that increases with decreasing diversity (Hirschman 1964; Simpson 1949). Only recently has the index been applied in fisheries economics, estimating the relationship between harvester income diversity, with respect to participation in multiple fisheries, and risk, as measured by the coefficient of variation in annual revenues (Kasperski and Holland 2013). To investigate the relationship between annual landings species diversity and predicted program impacts, individual sector harvester estimated effects were regressed on FY 2010 vessel revenues, a Herfindahl-Hirschman index of individual annual landings species diversity, and dummy variables indicating state of permit registration (Table 1.2). Results of this regression reveal that, controlling for vessel scale and home state, increased species diversity in annual landings was significantly correlated with predicted gains, suggesting harvesters more flexible in their targeting strategy had greater success in timing ex-vessel markets.

1.4.5 Interpreting the effects of price flexibilities

Estimated revenue effects can be interpreted as the difference in cumulative impacts of price externalities arising under each management regime and the associated distribution of landings during the season. Since each price equation included total daily coast-wide quantities of all species modeled, a shift in landings by any given harvester affected ex-vessel prices received by all other harvesters landing that day. The degree to which price externalities were realized by an individual was a function of market level overlap, with harvesters landing at the same dealer on the same day imposing the greatest externality on one another. Large predicted revenue gains imply that after catch shares there were either fewer spatially and temporally coincident landings of products with negative price externalities, more of products with positive price externalities, or both.

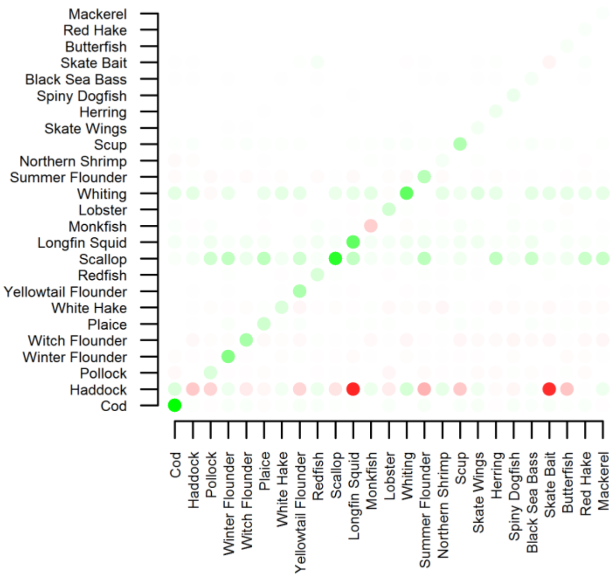


Figure 1.6 Graphical representation of counterfactual sensitivity results. Each point depicts the effect of including column species counterfactual landings on row species FY 2010 aggregate sector revenues. Color is scaled to the maximum absolute effect. Darker green (red) indicates larger predicted gains (losses) resulting from species' counterfactual inclusion. Species are ordered left-to-right/bottom-to-top by FY 2010 revenue importance among multispecies groundfish (southwest corner) and other species.

Sensitivity of the overall results to counterfactual predictions can be assessed by predicting prices after replacing counterfactual with actual landings, one species at a time. Differencing predicted revenue effects produced by this set of one actual and 24 counterfactual landings from those predicted by a full counterfactual provides a measure of gains by individual species timing change.[12] Figure 1.6 indicates that predicted program impacts result from a combination of own- and cross-species market interactions. A primarily green diagonal signifies timing changes were generally own-species beneficial, while numerous green off diagonal points highlight the benefits derived from cross-species price externalities. Interestingly, timing changes

---

[12] The effects of counterfactual inclusion captured by this method are only partial; predicted revenue effects were the result of multiple simultaneous timing changes.

for a number of species were detrimental to haddock revenues. This result may be a product of haddock's non-binding allocation and lack of quota value providing limited incentive to efficiently time the market and take advantage of haddock price externalities.

Following the introduction of multispecies catch shares, harvesters seasonally intensified targeting of many stocks. Multispecies sector vessels typically land around 90% of total white hake, pollock, yellowtail and winter flounder ex-vessel volume. All four species have large and negative total own price flexibilities indicating an increase in daily volume decreases contemporaneous prices. During FY 2010, white hake, pollock, yellowtail and winter flounder all experienced periods of intensified targeting and, counter-intuitively, higher prices. This result implies cross-species price flexibilities played a dominant role in revenue outcomes for these particular multispecies stocks. For example, decreased mid- and late-season landings of winter flounder resulted in lower levels of market mix with price impairing summer flounder (-0.0436), plaice (-0.0504), witch flounder (-0.0064), and yellowtail flounder (-0.0492). Despite the price depressing effect of higher daily winter flounder landings (-0.1004), harvesters gained by concentrating those landings away from species the market considers substitutes.[13]

Together the nine multispecies groundfish accounted for 45-52% of aggregate sector fleet gains. Cod ex-vessel price (15% of gains) benefited from a slightly more constant supply throughout the season and a large negative total own price flexibility (-0.3176), as well as reduced spiny dogfish (-0.0040), winter flounder (-0.0292), and skate (-0.0244) landings during fall and winter. Though spiny dogfish, winter flounder, and skate would not generally be thought of as substitutable for cod in end consumer markets, processing, shipping, and inventory tradeoffs likely exist as large quantities of spiny dogfish, skate, and winter flounder are frozen and may be exported, while cod is processed fresh and supplies local markets. When cod

---

[13] All price flexibilities included in text are total price flexibilities calculated as previously described.

landings are homogenous dealers can focus on cleaning it well and bringing it to the best markets, but if landed with large amounts of dogfish, skate, or winter flounder it must be unloaded quickly so efforts can focus instead on the freezing line. Flatfish species (25% of gains) are highly substitutable to consumers (average cross price flexibility of -0.0493), and when individually landed more homogenously throughout the season they yielded higher ex-vessel prices.[14]

Scallop, longfin squid, and whiting (31-37% of gains) – three economically important non-multispecies stocks – saw revenues increase from timing changes of many different species, highlighting the value of targeting flexibility afforded through catch share management. Despite some lack of temporal flexibility in these fisheries arising from opening times, weather, or derby incentives, the fleet used catch shares to advantageously alter concurrent landings of other stocks. Increases in concomitant landings of scallop with haddock (0.0034), lobster (0.0499), and plaice (0.0031), in addition to decreases with pollock (-0.0094), enhanced scallop prices.[15] Scallop, lobster, haddock, and plaice all generally serve high-end restaurant and grocery markets whose proprietors likely place a positive value on product diversity. Pollock on the other hand is typically considered low-grade groundfish, potentially competing with scallop for processing and shipping resources. Whiting and longfin squid are high volume species which are commonly frozen when landed due to the high perishability of the former and minimal quality reduction of

---

[14] Yellowtail and summer flounder gains were partially derived from increased market mix and a positive cross-price flexibility in landings of the former on prices of the latter (0.0196) which outweigh negative effects of the opposite relation (-0.0156). This interesting and statistically significant market relationship perhaps results from a combination of supply chain functioning and unidirectional substitutability.

[15] Ex-vessel scallop prices were found to depend substantially more on landings of other species than own market supply. In FY 2010, both sector and non-sector vessels saw their average price/lb for scallop increase by over 25%. Our models suggest that this dramatic price improvement arose from shifts in sector vessel landings of non-scallop species, contributing to 13% of total estimated gains. This result may be slightly confounded by a simultaneous introduction of the scallop general category IFQ program.

the latter. Increases in concurrent landings between these two species, which have positive cross-price flexibilities suggesting processing complementarity, improved ex-vessel prices.

1.5 Discussion

Improved ex-vessel revenues following the introduction of multispecies catch shares arose from advantageous market timing changes by sector vessels. This conclusion suggests DAS created inefficiencies, restricting harvesters' ability to exploit price externalities and respond to market incentives. The observed dramatic shift in sector vessel joint landing behavior of multispecies groundfish with commixing, separately regulated stocks additionally suggests DAS may severely constrain catch and landing opportunities when vessels participate in overlapping fisheries with different management systems. This analysis combined counterfactual predictions of individual harvester landings timing, reflective of prior management, with models of ex-vessel inverse demand in estimating total revenue gains of over $30,000,000.

Though this analysis was only concerned with ex-vessel revenues, it appears likely that fleet profits were also enhanced as a result of catch share management. With variable operational costs at 38% of gross revenues – the rate used by sector managers when generating initial projections – a no profit increase would require fishing costs to have grown by over 12 million dollars. This seems unlikely however as vessels made 10% fewer trips and average trip costs were found to fall well within the range observed under DAS (Kitts et al. 2011). Other recent empirical findings have shown tradable access rights are often associated with decreased fishing expenditures, increasing profits through cost reduction (Andersen et al. 2010; Nielsen et al. 2012). The results of this study indicate that in fisheries with flexible prices, constraining management regimes may severely limit potential profits by weakening ex-vessel revenues.

Significant revenue increases resulting solely from a temporal redistribution of supply further question the utility of optimum yield, which has thus far remained an elusive management goal in New England groundfish (Rothschild et al. 2013).

The newfound production flexibility introduced through catch share management may appear inflated if discard of multispecies stocks was frequent. In FY 2010, only 30% of trips were observed, suggesting vessels may have had ample opportunity to discard constraining catch share stocks. This level of observer coverage did represent a dramatic increase from that under DAS management however, and landings from unobserved trips were monitored, penalizing vessels when discard was suspected. Additionally, it is conceivable that harvesters may have been reluctant to discard during the first year of new management, where each sector member was for the first time jointly liable in fishing activities. Unaccounted discard does not affect the landings revenue results presented here, though it is expected that increased control of illegal discard might reduce revenue gains by curbing flexibility in landings.

Had catch shares not been introduced multispecies harvesters would likely have experienced drastic cuts in DAS and multispecies fishing income as regulators responded to stock rebuilding requirements specified in the Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006. The two counterfactual methods used relied on information from the most recent prior management year; they are not wholly descriptive of the fishery which would have existed had regulators significantly cut DAS without providing the option to join catch share sectors. Additionally, counterfactual methods altered landings’ pace within the set of actual trips taken, which, if endogenous, may confound results. Significant room exists for the development of harvester behavioral models; the work presented here methodologically supplements the current literature, extending methods used in Scheld et al. (2012).

The introduction of catch share management for multispecies stocks significantly affected seasonal exploitation patterns of several separately regulated species. For example, increased participation in the race-to-fish for spiny dogfish, summer flounder, and scup occurred perhaps as a direct result of multispecies catch share management, which made low-quantity multispecies landings profitable by reducing opportunity cost. This implies a mismatch in the scope of regulations and individual economic decisions, which may exacerbate management difficulties already complicated by joint harvest relationships (Kirkley and Strand 1988; Squires 1987). Understanding behavioral drivers and the tradeoffs of fishery participation, as well as identifying user groups that overlap regulatory regimes, will ultimately yield superior ex-ante program analysis, reducing management implementation uncertainty and unexpected outcomes (Fulton 2010).

Including complex micro-market structure in the model of inverse dealer demand allowed for robust estimation of ex-vessel price and revenue effects resulting from changes in the daily market mix of landed species. Numerous different individual strategies, modifying landings in response to expected price externalities, likely produced the aggregate fleet level behavior and outcomes observed and discussed. Though analyzing each of potentially millions of individual harvester-species daily timing changes is an unrealistic pursuit, conclusions drawn from their combined effect suggest multispecies catch shares afforded individual harvesters increased opportunity and flexibility in supplying market demand. Large ex-vessel revenue advantages owing to market timing changes have been previously documented after the introduction of rights-based management (e.g., Herrmann 2005; Scheld et al. 2012), though gains are generally seen to result from a downward sloping demand and own-species effects. Following the introduction of multispecies catch shares, cross-species price flexibilities were found to play an important role. A counterfactual sensitivity analysis demonstrated that individual species revenue gains often arose from timing changes of several complementary or substitute species, or species groups. Intermediary processing and supply chain structure has often been of limited concern in fisheries policy analysis, though here it was a major driver of policy outcomes.

As a management institution, catch shares have been argued to prevent fishery collapse (Costello et al. 2008) and align individual and collective incentives (Grafton et al. 2006). The transition from DAS to catch shares ensures each multispecies TAC, if adequately enforced, places an upper bound on possible harvest – a necessary, though not sufficient, condition for ecological benefit (Branch 2009). Multispecies catch shares may wield biological consequence if seasonal changes in targeting intensify harvesting pressure during periods of amplified stock vulnerability, or result in increased mortality for separately managed stocks lacking overall catch limits, shutdown provisions, and/or avoidance incentives. Additionally, multispecies catch shares may impose cultural and societal costs if intended reductions in overcapitalization or consolidation of catch share holdings dramatically alter working waterfronts, community structure, or other factors affecting quality of life for New England residents.

Chapter 2

Costly avoidance in a multispecies catch share fishery

Abstract

Joint harvest technologies in multispecies fisheries are often characterized by imperfect selectivity, allowing only limited control of catch composition. This type of technology can be modeled by considering outputs weakly disposable, suggesting targeting and avoidance of individual stocks may be costly. In this paper, a simple individual optimization model is considered with a joint production transformation function which relaxes the assumption of free output disposal to analyze decisions under imperfect output selectivity. Costly avoidance, where production of joint outputs is given up to reduce that of the avoided species, is found to result when the marginal reward for landing an avoided stock is negative, a possible consequence of intense regulatory constraint. This model is then applied to New England groundfish, a multispecies fishery recently transitioned to catch share management, to test for costly avoidance. A hierarchical Bayesian estimation procedure is used to uncover marginal rates of product transformation between a constraining stock and the aggregate mix, finding generally positive values heterogeneously distributed throughout the fleet. We reject the null of costless avoidance, indicating constraining output controls for certain stocks decreased joint output of quota-abundant species. Furthermore, a unique management setting in which a constraining species quota increased 600% mid-season allows us to identify ex-post costs of quota constraint. We estimate that had the low allocation remained, its cost to harvesters would have been US $3 million.

2.1 Introduction

Over the last twenty years, revelations of substantial bycatch and subsequent discard in many fisheries (Alverson 1994; Harrington et al. 2005; Kelleher 2005) have led to serious concern and dissatisfaction among scientists, policy makers, and the general public. Recent US estimates indicate an overall bycatch rate of 0.17, with several fisheries discarding more than 30% of total catch (NMFS 2011). The widely recognized cause of such squander is a misalignment in the scope of production with that of regulatory and post-harvest economic structures.

In wild-capture fisheries, harvesters often attempt to exploit particular species within diverse ecosystems using nondiscriminatory nets, hooks, or traps, catching both target and non-target stocks. Exploitation patterns have historically been driven by external market forces (Sethi et al. 2010), though more recently, an impetus to align incentives shaping harvest with conservation objectives, by way of management, has emerged (Hilborn et al. 2005; Grafton et al. 2006). Managing multispecies fisheries through intentional modification of harvesters' choice criteria may improve ecological outcomes by reducing non-target catch, and in fact several researchers have observed targeting and avoidance behavior following management action (Grafton et al. 2004; Graham et al. 2007; Branch and Hilborn 2008), though this ability appears to be bounded.

Non-selective or imperfectly selective output technologies can be modeled in a neoclassical framework through modification of standard disposal assumptions. Strong, or free, output disposability is a ubiquitous assumption in joint production theory (e.g., Lau 1972, Diewert 1973, Sakai 1974), though when applied to a multispecies fishing technology implies perfect and costless targeting and avoidance. Specifically, this assumption requires individual output decreases never command increased inputs or decreased joint output, i.e., marginal production costs must be monotonically increasing in each output. Free disposal technology does not correspond to anecdotal descriptions of production in multispecies fisheries, where avoiding one stock may decrease joint harvest of other species. Indeed, a few researchers have argued production in multispecies fisheries likely violates this assumption and instead only satisfies conditions of weak disposability (Turner 1995, 1997; Singh and Weninger 2009). This looser technological restriction requires proportional reduction in joint outputs be free but accommodates costly disposal otherwise (Shephard 1970).

To explore the implications of weak disposability on producer behavior, we develop a notion of costly avoidance, where, due to a weakly disposable multispecies production technology, avoiding a particular stock may decrease joint output of other stocks. Costly avoidance is shown to result in response to constraining management conditions such as a low allowable catch, high quota price, or a restrictive bycatch limit. This theory is timely as the proportion of species and areas under the purview of management continues to increase, global populations, their wealth, and demands for marine protein similarly grow, though several basic questions surrounding behavioral responses to regulatory incentives remain unresolved.

Catch shares are perhaps the most widespread and increasingly utilized form of fisheries management. Typically, catch shares are implemented by first setting an annual upper bound on harvest and then allocating use rights to individuals or groups of harvesters. In the US, recent legislation lends itself to catch share management as hard output caps are now required for all federally managed fisheries (MSRA 2007), while a 2010 report issued by the National Oceanic and Atmospheric Administration (NOAA) proclaimed broad support for catch share systems, encouraging regional management councils to consider their use “wherever appropriate” (NOAA

2010). Currently, of the 15 US catch share programs, six manage multispecies complexes where harvest is best characterized as a joint production process.

Not surprisingly, management of multispecies fisheries with catch share programs has been acknowledged as acutely difficult and potentially problematic due to imperfectly selective technologies (Copes 1986; Squires et al. 1998). Restrictions on harvest of individual stocks in a multispecies fishery may incentivize illegal discard or result in significant under-fishing of allowable catch. Both are symptoms of a mismatch in catch share holdings to the intercepted mix, though their relative severities are likely inversely related and depend on the degree of enforcement. Discard incentives brought on through weak output disposability have been explored by Turner (1995, 1997) and Singh and Weninger (2009), who find individual multispecies quota systems, if not enforced, incentivize discard when the least cost harvest bundle diverges from relative quota holdings. Their results rely on an assumption of costless discard however, and say nothing about under-fishing or costly avoidance – relevant subjects which are addressed here.

Understanding how and when costly avoidance manifests is important in quantifying short and long run efficiency of policy measures in multispecies fisheries, which are increasingly catch share regulated or subject to output controls of some kind. In what follows, a model of costly avoidance is developed and discussed in the context of exogenous technological, economic, and management conditions. Then, a hierarchical Bayesian estimation procedure is used to uncover marginal rates of product transformation and statistically test for costly avoidance in New England multispecies groundfish during the first year of catch share management and also estimate the costs imposed by a restrictive allocation. Our findings suggest costly avoidance is common, potentially exacerbated by poorly functioning quota exchange mechanisms, and may manifest indirectly, through forgone production, as well as directly, through costly harvest strategy.

2.2 Theoretical model

Weak output disposability is a technological frontier concept which permits costly output reductions (see Färe et al. 1994 for a comprehensive mathematical treatment). The distinction between strong and weak disposability can be seen in Figure 2.1.

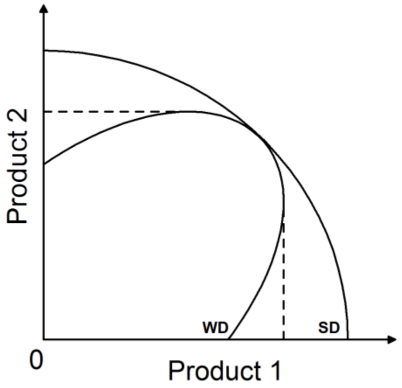


Figure 2.1 Technological production frontiers exhibiting strong disposability (SD) and weak disposability (WD). Dashed line defines strong disposal criterion for WD frontier.

In Figure 2.1, the inner WD frontier exhibits weak disposability of both products. Reductions in product one (product two) beyond where the dashed line meets this frontier require reduced output of product two (product one). This technological relationship stands in contrast to that of the outer SD frontier, where reductions of product one (product two) can always be met with increases in product two (product one). Technologies which exhibit weak disposability can be said to display costly avoidance, the consequences of which will be explored with the following general model.

2.2.1 Production Technology

Let $\boldsymbol{Q} \in \mathbb{R}_{+}^{m}$ denote a vector of *m* outputs, or species-specific landings, and let $Z \in \mathbb{R}_{+}^{1}$ denote a composite measure of fishing effort. $F(Z, \boldsymbol{Q})$ defines the set of possible fishing trip input-output combinations and is assumed to be closed, bounded, non-empty, and strictly convex for $Z \geq 0$. It is also assumed that $F(Z, \boldsymbol{Q}) \subseteq F(Z', \boldsymbol{Q})$ for $Z' \geq Z$ and $F(Z, \theta\boldsymbol{Q}) \subseteq F(Z, \boldsymbol{Q})$ for $0 \leq \theta \leq 1$, reflecting that input is freely disposable, though disposal of outputs may be costly. Given this production technology the following transformation function can be defined:

$$f(\boldsymbol{Q}) = \begin{cases} \min_Z \{Z \mid (Z, \boldsymbol{Q}) \in F\} \\ -\infty \quad \mathit{otherwise} \end{cases}. \tag{2.1}$$

The transformation function (2.1) describes the minimum amount of effort required to land output bundle $\boldsymbol{Q}$. If $\boldsymbol{Q}$ is not producible, the function returns minus infinity – a convention of the literature. f is assumed to be continuous and twice differentiable over producible quantities, non-decreasing in proportional output increases, strictly non-negative, and equal to zero when zero output is produced.[16]

---

[16] Since landings are the result of harvest and discard behavior, allowing landings technology to exhibit costly disposal implies discard is a potentially costly activity. Previous research has argued discarding may impose costs due to sorting and disposing of unwanted catch (Anderson 1994; Arnason 1994; Vestergaard 1996), expected penalties if illegal (Arnason 1994; Kristofersson and Rickersten 2009), and/or related social implications (Arnason 1994). At the extremes of infinite and zero discarding costs, the technology described in (2.1) would be identical to the underlying harvest technology or free disposal production. The harvest technologies of Turner (1995, 1997) and Singh and Weninger (2009) can therefore be considered special cases of (2.1).

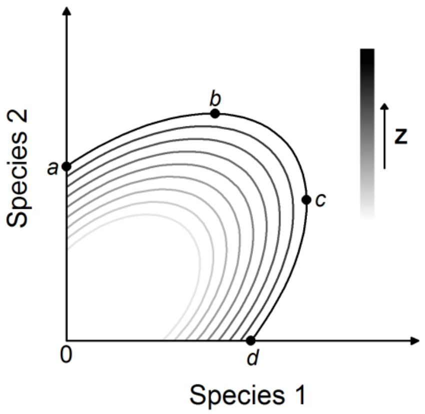


Figure 2.2 Costly disposal production isoquants for a two species fishery

Notice that, aside from restrictions implied through weak output disposability, there are no conditions placed on first derivatives of (2.1). That is, $\frac{\partial f}{\partial Q_i}$, the marginal production cost of species *i*, is not assumed to be strictly non-negative as would be the case if f exhibited strong disposability. In wild-capture multispecies fisheries, where harvest often exhibits imperfect selectivity, intense avoidance of a particular stock may reduce output of joint targets, suggesting a higher level avoided-stock output may actually be feasible with a lower level of input, without altering joint production levels. This can be seen in Figure 2.2, which plots a homothetic representation of production technology exhibiting costly avoidance. Between points *a* and *b* on the outermost frontier the marginal production cost of increasing species one is negative – more could be produced using less effort without reducing species two production (i.e., a horizontal shift to an inner frontier). Similarly, between points *c* and *d* the marginal production cost of species two is negative. In the models described here, joint production points for which marginal production costs are negative, or equivalently, marginal rates of product transformation are positive, will be considered costly avoidance. Notice that assuming weak disposability ensures that everything cannot be simultaneously costly avoided. To see why this is so, remember weak disposability requires that proportional output increases must always increase costs. A situation where all marginal production costs are negative violates weak disposability since a proportional output increase would decrease costs.

2.2.2 Optimization model

The trip-level optimization problem assumed to be facing each harvester in a multispecies fishery is:

$$\max_{all\,\{Q_i\}} \sum_{i=1}^{m} r_i(Q_i) \tag{2.2}$$
$$s.t.\ \ f(Q_1, \dots, Q_m) \leq Z,$$

where, for each of *m* species, r_i represents the reward from landing quantity Q_i. A reward function is used in place of revenue or profit to maintain generality. For our purposes, r_i can be thought of as ex-vessel revenue net of all regulatory costs imposed either directly (e.g., tax, subsidy, or quota price) or indirectly (e.g., transaction costs in quota market, shadow cost from output constraint, expected penalty if illegal, etc.). Every trip, the harvester must choose an *m*-species landings bundle which maximizes the sum of species-specific rewards subject to constraint on production inputs and landings technology, where $f(\cdot)$ is defined by the transformation function (2.1). That is, the optimization problem (2.2) is constrained by inputs and technology, a reasonable formulation for a fishery located on distant grounds, which once reached, targeting and avoiding individual species requires negligible inputs (e.g., small modifications in tow/soak depth, duration, timing, or location).

To facilitate investigation of costly disposal landings technologies and their consequence for individual harvesters, our optimization problem (2.2) is modeled as static and considers only one input. Though dynamics are often the focal subject of modeling exercises in fisheries economics, here the questions of interest are related to imperfectly selective technology. Modeling (2.2) in a static framework eliminates the possibility that our conclusions are confounded by these processes. Considering only a single production input similarly focuses analysis. Note however that (2.2) can accommodate any non-infinite number of inputs, provided they are freely disposable.[17]

Solution of (2.2) proceeds by first specifying a Lagrangian:

$$L = \sum_{i=1}^{m} r_i(Q_i) + \mu\big(Z - f(Q_1, \dots, Q_m)\big), \tag{2.3}$$

and then determining Kuhn-Tucker first order conditions:

$$\forall \;\; i \quad \frac{\partial L}{\partial Q_i} = \frac{\partial r_i}{\partial Q_i} - \mu \frac{\partial f}{\partial Q_i} \leq 0 \;\;,\;\; Q_i \geq 0 \;\;,\;\; \frac{\partial L}{\partial Q_i} Q_i = 0 \tag{2.3a}$$

$$\frac{\partial L}{\partial \mu} = Z - f(Q_1, \dots, Q_m) \leq 0 \;\;,\;\; \mu \geq 0 \;\;,\;\; \mu\big(Z - f(Q_1, \dots, Q_m)\big) = 0, \tag{2.3b}$$

from which we arrive at the general result:

$$\forall \;\; i,j \;\mid\; Q_i, Q_j > 0 \;\;,\;\; \frac{\partial r_i/\partial Q_i}{\partial r_j/\partial Q_j} = \frac{\partial f/\partial Q_i}{\partial f/\partial Q_j}. \tag{2.3c}$$

---

[17] In general, a one-to-one correspondence between a transformation function and the production possibilities set requires some form of free disposability (Diewert 1973; Lau 1976).

Result (2.3c) indicates that, for any two positively landed species *i* and *j*, it is optimal to equate the ratio of species-specific marginal rewards to marginal costs. An increase in the marginal reward of landing species *i* must be met with an increase in marginal costs *ceteris paribus* (i.e., targeting). Likewise, avoidance of species *i* occurs if marginal rewards decrease. Costly avoidance is only optimal when marginal rewards are negative. Furthermore, it can be shown:

PROPOSITION: If $\frac{\partial r_j}{\partial Q_j} \geq 0 \; \forall \, j \in \, -\boldsymbol{i},$ then $\frac{dQ_i}{dQ_j} > 0 \mid dQ_k = 0 \;\; \forall \;\; k \neq i, j$ and $\frac{dQ_i}{\sum_j dQ_j} > 0$ if and only if $\frac{\partial r_i}{\partial Q_i} < 0.$

PROOF: See Appendix B

The proposition above states that if marginal rewards for landings of all species –*i* are non-negative, a positive marginal rate of product transformation between any two outputs *i* and *j*, fixing all other outputs at constant levels, or between an output *i* and the aggregate output mix -*i*, results if and only if the marginal reward for landing species *i* is negative. The logic of this argument can be easily extended to all species combinations, indicating a positive technological tradeoff between landings of different species, or costly avoidance, occurs in response to negative rewards.

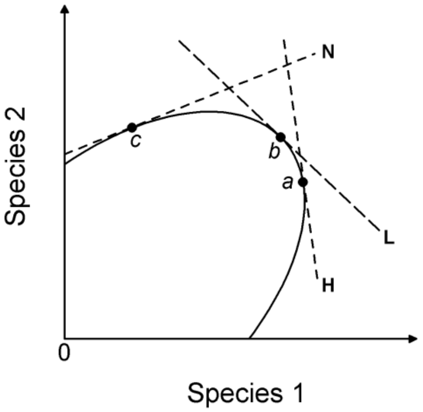


Figure 2.3 Optimal joint production bundles. Dashed lines are linear iso-reward curves whose slope is equal to $-\frac{\partial r_1/\partial Q_1}{\partial r_2/\partial Q_2}$ for high (H), low (L), and negative (N) $\frac{\partial r_1}{\partial Q_1}$ (marginal rewards are not restricted to be constant across Q, it is imposed here to ease interpretation)

Costly avoidance of economically undesirable output must balance the losses incurred from negative marginal rewards against those from lost joint production. This situation can be seen for a two species fishery in Figure 2.3. At a high level of marginal rewards for landings of species one, relative to those for species two, it is optimal to target and land a large amount of species one at a point like *a*. If this marginal reward were lower, giving up some species one to land more species two, at a point like *b,* would instead be optimal. Costly avoidance occurs only when marginal rewards are negative, implying a positively sloped iso-reward curve. At point *c*, the marginal benefit of avoidance – reduced loss in species one rewards – is equal to the marginal cost of avoidance – reduced landings of species two. At this point of balance, where the marginal rate of product transformation equals the negative marginal reward ratio, the individual harvester can do no better.

But, what might cause or create a negative marginal reward? Here we propose three possible mechanisms, all the result of regulatory action.[18] First: large taxes in excess of competitive market prices. While this may not be particularly germane to marine fisheries, which generally rely on output management rather than Pigovian controls, industries emitting taxed effluent for which no market exists certainly realize negative marginal rewards and the results of this general model may be extended in that direction. Second: strict output quantity controls. If added to (2.2), output controls would enter the model as constraints in (2.3), which if low enough, could bind at a point of costly avoidance, creating a shadow value for this Lagranian constraint larger than market price. This mechanism may be applicable to multispecies fisheries where strict bycatch limits for marine mammals or protected species exist. Third: quota price in excess of ex-vessel market value. While we do not speculate here on quota price formation, assuming it exogenous, it is easy to think of cases where, due to joint production expectations, quota price exceeds saleable value. We see this last mechanism as particularly relevant to global fisheries which are increasingly moving toward tradable quota management systems.

2.3 New England multispecies catch shares

On May 1st, 2010 – the first day of the fishing year (FY) – catch share management was introduced to New England multispecies groundfish, a collection of 20 jointly managed stocks earning over $80 million in ex-vessel revenue annually. Prior to this, the species complex had been managed with individual effort controls, where each limited entry vessel received a tradable allocation of days-at-sea. The Magnuson-Stevens Reauthorization Act (MSRA 2007) requires hard annual catch limits for all federally managed stocks however, forcing regional managers to

---

[18] Other costs, such as those arrived at through transaction or social interaction, may produce negative marginal rewards. For brevity, the list discussed here is non-exhaustive, though we hope it captures those mechanisms most important to environmental and resource management problems.

control output directly. To meet this legal mandate while incentivizing targeting of healthy stocks and avoidance of weak – behavior absent under joint-stock days-at-sea – New England managers developed Amendment 16 to the Northeast Multispecies Fishery Management Plan which established hard caps on harvest and allowed permit holders to enter a catch share program (NEFMC 2009). Under Amendment 16, self-identifying groups of harvesters, called sectors, would receive aggregate allocation to manage semi-autonomously. While the collective allocation was intended in-part to insure against shutdown resulting from limited compositional control, during the program's first year, all 17 sectors chose to manage their allocations for 14 stocks of nine groundfish species as individual transferable quota (ITQ).[19,20]

---

[19] Transfer between individuals in different sectors required facilitation by sector managers. Intra-sector and bundled inter-sector trades were most common.

[20] The Northeast Multispecies Fisheries Management Plan manages 20 stocks composed of 15 species: Atlantic cod (Gadus morhua), haddock (Melanogrammus aeglefinus), pollock (***Pollachius virens*)**, Acadian redfish (*Sebastes fasciatus*), yellowtail flounder (*Limanda ferruginea*), American plaice (*Hippoglossoides platessoides*), witch flounder (*Glyptocephalus cynoglossus*), winter flounder (*Pseudopleuronectes americanus*), white hake (*Urophycis tenuis*), Atlantic halibut (*Hippoglossus hippoglossus*), windowpane flounder (*Scophthalmus aquosus*), ocean pout (*Zoarces americanus*), Atlantic wolffish (*Anarhichas lupus*), whiting (*Merluccius bilinearis*), and red hake (*Urophycis chuss*). In FY 2010, commercial fisheries existed for the first nine and last two species listed. Catch shares were allocated for the former while the latter were regulated separately under small mesh provisions.

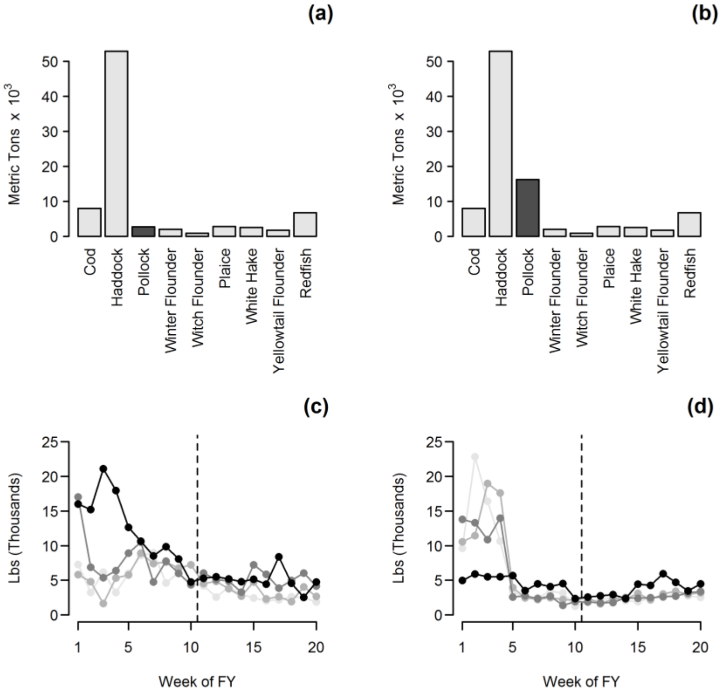


Figure 2.4 Top row: FY 2010 initial (a) and subsequent (b) total quota allocations by species. Left-to-right ordering reflects FY 2009 gross revenues. Bottom row: Trip average non-pollock multispecies groundfish landings from southern (c) and northern (d) New England vessels. Location type was identified by 2010 vessel registration, with vessels docked south of Cape Cod labeled “southern”. FY 2010 observations are in black; lighter grey signifies earlier FYs. Dashed line indicates pollock allocation increase.

During this first year, fleet-wide total allocations ranged from over 50 thousand combined metric tons for the healthy and highly abundant stocks of haddock, to under one thousand metric tons for depressed stocks of cod and certain flatfish species. At just a third of recent annual landing levels, the initial allocation for pollock was recognized as exceedingly low before the season even began (Hemmerdinger 2010). Indeed, many agreed the allowable pollock catch was “certain to strangle the groundfishery” (Gaines 2010). Pollock is one of five New England multispecies groundfish stocks whose management area spans all three of the major fishing grounds. Under new catch share provisions, when an allocation for a particular stock was reached sector vessels were required to halt fishing in the associated stock area for the remainder of the season. A drastically low pollock allocation could spell disaster for the fleet if they were unable to avoid it. Shortly after the season began however, a new pollock stock assessment was completed and revealed the species was fairly healthy and highly abundant. Two and a half months into the season, on July 15th, 2010, managers revised the allocation for pollock upwards, from 2,686 to 16,178 metric tons (Figure 2.4, panels a and b), providing individual sectors a 600% increase in allocation.

Observations of early season landing behavior suggest harvesters' initial production strategy differed from that in previous years. Possibly reflecting a modified approach in response to an intensely constraining pollock allocation, the third of the fleet from southern New England ports landed significantly more non-pollock multispecies groundfish, focusing on stocks of cod and haddock, while vessels from the north decreased formerly high early season production levels (Figure 2.4, panels c and d).[21] Then, several weeks before the pollock allocation increased, landings returned to more typical amounts. Were these changes in response to pollock constraint? Both groups of harvesters did have lower than average pollock landings during this time. Perhaps those vessels south of Cape Cod were fishing different locations or using special harvest methods to avoid pollock, simultaneously increasing landings of other less constraining stocks, while northern boats avoided pollock at the cost of decreased non-pollock production. To explore the effect of pollock quota constraint on non-pollock production we develop an empirical model in the next section which will be used to construct technological parameters as well as estimate trip level effects and counterfactual season-long constraining pollock costs.

---

[21] Note that this was not due to switching port of registration by vessels. Only one vessel switched regions following the catch share implementation and this vessel was relatively small scale, with total daily landings never exceeding 2,000 lbs.

2.4 Empirical analysis

2.4.1 Model specification

Results of the theoretical model indicate a positive marginal rate of product transformation will be observed only under costly avoidance. During the first few months of FY 2010, pollock was a severely constraining stock and, as such, was intensely avoided. Anecdotal evidence and an unmatched disparity in previous year landings to allocated quota leave little reason to suspect other stocks were also actively avoided during this time. A number of issues inhibit use of quota transaction data and we therefore rely on observed production behavior to analyze costly avoidance.[22]

Empirical investigations of multi-output production problems typically specify technology through either frontier or dual methods. Multi-output frontier models construct efficient production sets from observed data using linear programming routines, assuming observations inside the frontier are the result of technical inefficiency (Farrell 1957; Charnes et al. 1978).[23] While potentially appropriate here, this approach is not undertaken to allow explicit incorporation and analysis of latent individual production heterogeneity. Additionally, by relying on a parametric approach rather than DEA estimators, we obtain information from the mean rather than the tails of the data and can avoid complicated and, for problems of weak disposability, often intractable bootstrap procedures when quantifying statistical uncertainty (see Simar and Wilson 1998, Kneip et al. 2008, and Simar and Wilson 2011 for methods to bootstrap DEA measures). Dual principles, theoretically extended to production by Shephard (1953),

---

[22] Unfortunately, individual quota transaction data was not available. Additionally, the majority of trades in this fishery are pounds-for-pounds for which identification of implicit prices has been shown to be difficult (Holland 2013). Finally, observed/derived transaction prices do not incorporate a number of factors acting on effective marginal rewards and influencing behavioral outcomes (e.g., transaction costs, social capital, risk/time preferences, etc.).

[23] Note that stochastic frontier methods (Aigner et al. 1977) are not in general valid under weak disposability.

provide a workable basis in which estimation of complicated joint production processes becomes fairly simple. Over the years, fisheries economists and practitioners have made great use of duality theory (see Jensen 2002 for an overview), though here it is inappropriate. Once the common assumption of strong disposability is replaced with that of weak, neoclassical profit, revenue, and cost functions transform into multi-valued set correspondences and Hotelling's lemma cannot be applied.[24]

To investigate costly avoidance in New England multispecies groundfish during the first year of catch share management, we estimate the following model:

$$ln\left(Q_{MG_{i,t}}\right) = \beta_0 + \beta_1\delta_t + \beta_2 ln\left(Q_{P_{i,t}}\right) + \beta_3 ln\left(Q_{P_{i,t}}\right) \times \delta_t + \beta_4 ln\left(CI_{i,t}\right) + \beta_5\omega_{i,t} + \varepsilon_{i,t}. \quad (2.4)$$

Equation (2.4) models ex-vessel landings of non-pollock multispecies groundfish (Q_{MG}) by harvester i on day t to be a function of pollock landings (Q_P), a composite input (CI), and a normally distributed error term (ε). The intercept and elasticity of non-pollock landings with respect to pollock were allowed to shift by allocation regime ($\delta_t = 0{,}1$ for high and low pollock allocations, respectively). Additionally, we included a zero-pollock intercept shifter to capture differences in trips not landing any pollock ($\omega_{i,t} = 0{,}1$ for non-zero- and zero-pollock landings respectively, zero-pollock landings made up ~36% of the sample). Before taking natural logs, one unit was added to each logged variable. Estimation of (2.4) returned a measure of covariance between non-pollock and pollock landings, controlling for inputs and allocation, which was used uncover multispecies technological parameters.

---

[24] Shephard (1970) examines weakly disposable production in a dual framework, though parametric extensions of this work considering multiple outputs are not immediately obvious.

Many aspects of (2.4) are deserving of further discussion and justification. First, the dependent variable is an aggregate mix of non-pollock multispecies groundfish landings. Empirical production models commonly aggregate across product types, grades, and classes due to data resolution, to enhance sample size, or to remain complete while focusing analysis on a subset of output. In multispecies fisheries, the use of "other flounder" (Squires 1987; Kirkley and Strand 1988), "other salmon" (Larson et al. 1996; Larson et al. 1998), or just "other" categories (Gunatilake and Leung 2003; Pascoe et al. 2007; Asche 2009; Felthoven et al. 2009) is not uncommon. Here, aggregation of non-pollock landings affords a simple and parsimonious specification capable of testing hypotheses derived from our theoretical model while simultaneously estimating pollock avoidance costs in terms of non-pollock production. Second, pollock is included as a right-hand-side independent variable. While pollock landings are not independent *per se*, the effects of an exogenous pollock management shift on non-pollock production are captured through this variable. Prior to analysis, endogeneity of pollock in (2.4) was strongly rejected after residuals from an OLS specification were regressed on pollock landings and indicated no contemporaneous error correlation ($R^2 < .01$).[25] Third, only one production input is considered in (2.4). Unfortunately, data on fine-scale (tow-level) variable inputs were unavailable and so a composite measure was constructed by first taking the leading principle component of vessel length, gross tons, and horsepower – highly collinear fixed factors (average $R^2 = .79$) – and then multiplying this by trip length. Our composite input therefore scales the observed variable input (trip length) by fixed capital. The assumption of asymmetric separability is fairly common in multispecies production studies which often multiply a single

---

[25] Treatment of "undesirable" output as a production input has a theoretical basis (see Cropper and Oates 1992), though the validity of this approach has been contested in more recent non-parametric, Shephard (1970) distance function applications (Färe and Grosskopf 2003; Kuosmanen 2005). We divorce ourselves from this discussion however as (2.4) should not be considered a production function in the usual sense.

fixed input (e.g., gross tons) by days fished to construct the composite measure (see e.g., Kirkley and Strand 1988, Squires and Kirkley 1991, Campbell and Nicholl 1994, and Thunberg et al. 1995).

Multilevel models consider regression coefficients to be drawn from an underlying population probability model which is defined by a set of hyperparameters, often the mean and variance (Gelman and Hill 2007; Jackman 2009). With advances in computing power and canned statistical packages, multilevel models offer large gains in fit and prediction for only small increases in computational burden. These models are additionally very flexible. For example, in a standard two-level specification, the extremes of infinite and zero hyper-variance correspond to group-level fixed effects or identical coefficients across groups. To incorporate and test for individual vessel production heterogeneity, the following multilevel structure was introduced:

$$\beta_{ki} \sim N\left(\boldsymbol{M_{\beta}}, \boldsymbol{\Sigma_{\beta}}\right) \; for \; k = 0,1,2,3. \tag{2.5}$$

For each harvester *i*, the intercept, output elasticity of pollock, as well as their low allocation shifters, were assumed to be drawn from an underlying multivariate-normal population model with mean $\boldsymbol{M_{\beta}}$ and covariance matrix $\boldsymbol{\Sigma_{\beta}}$.

2.4.2 Model estimation

Four different specifications of (2.4) were estimated: hierarchical Bayesian (HB), mixed effects (ME), non-hierarchical Bayesian (NH), and fixed effects (FE). HB included and explicitly modeled the multilevel structure of (2.5), ME allowed for individual-vessel random deviations from fleet level fixed effects for the parameters considered in (2.5), and NH and FE modeled all parameters constant across individuals. Comparisons between these models serve two purposes. First, evaluation of parameter estimates and model selection criteria between HB and NH and also ME and FE test for, and identify the effects of, significant multispecies groundfish production heterogeneity at the individual level. The New England groundfish fleet includes vessels from Maine through the mid-Atlantic who visit three distinct fishing grounds, each comprised of numerous specific harvest locations. Site heterogeneity and fidelity (Holland and Sutinen 2000), in addition to the often identified "skipper effect" (Hilborn 1985; Hilborn and Ledbetter 1985; Squires and Kirkley 1999), suggest multispecies groundfish production may vary considerably vessel-to-vessel. Second, side-by-side evaluation of models estimated using Bayesian and non-Bayesian maximum likelihood methods allow advantages of the former to be discussed while maintaining transparency of model results and conclusions.

The data used to estimate (2.4) and (2.5) consisted of 4,304 dealer-reported, individual-vessel landing transaction observations from 248 fishermen from May 1st, 2010 through September 30th, 2010. A restricted temporal range (i.e., five months of the FY) was used balance the panel of observations before and after the allocation increase and to control for possible seasonal effects. Summary statistics and anecdotal evidence suggested the fleet correctly anticipated the large allocation increase. The method of Bai and Perron (1998, 2003) was used to test for breakpoints in a regression of daily fleet aggregate non-pollock on daily fleet aggregate pollock.[26] A single significant breakpoint was found on June 24th, 2010, three weeks prior to the allocation increase, though just a short time after industry representatives and media sources widely reported the increase was a "done deal" (Gaines 2010). This breakpoint was used to partition the sample into low and high allocation regimes. Trip length was calculated by

---

[26] Note that a specification endogenously determining the breakpoint(s) in (2.4) and (2.5) was computationally infeasible.

differencing the dates of successive landing observations, implying a minimum trip length of one day. A maximum trip length of seven days was imposed to control for periods where a vessel may docked and inactive.

All models were estimated in the statistical program R (R Core Team 2013). Maximum likelihood models were fit using the package lme4 (Bates et al. 2014), which for the ME model relied on an iterative restricted maximum likelihood process. Bayesian models were fit using the package rjags (Plummer 2013) which integrates R with JAGS (Plummer 2003), a stand-alone program that efficiently performs Markov chain Monte Carlo Gibbs sampling procedures. Diffuse normal priors were used for hyper-means and single-level parameters, while the prior on the hyper-covariance matrix was specified as a diffuse inverse Wishart, all fairly standard specifications (Gelman and Hill 2007). Bayesian posterior parameter vectors were drawn after burn-ins of 1,000,000 (HB) and 10,000 (NH), using thinning rates of 500 (HB) and 100 (NH) to keep 1,000 draws from each of three parallel chains. See Appendix B for convergence diagnostics of the more complicated HB model.

2.4.3 Calculating costly avoidance

To investigate costly pollock avoidance we used parameter draws from the HB specification. Whereas maximum likelihood methods identify the set of parameters which best explain the data, Bayesian procedures sample across the posterior distribution and provide a range of credible parametric combinations. Analytically, this makes construction of a variety of marginal distributions quite simple, which in the present setting, enables robust investigation of marginal rates of product transformation. There are philosophical grounds for choosing a Bayesian course as well. Acknowledgment and incorporation of multilevel parametric structure suggests the modeler believes the data generating process involves random deviations from underlying population parameters. Though quite at home in a Bayesian world, this belief is a bit at odds with frequentists' notion of true parameter values (Jackman 2009). Additionally, Bayesian concepts of probability and inference, which look to confirm or refute a hypothesis in light of the observed data, are more in line with ex-post objectives than their frequentist analogs.

For each joint landing observation in our dataset ($n = 2{,}736$), posterior parameter draws were used to construct 3,000 estimates of the marginal rate of pollock transformation (MRPT), or the additional/fewer pounds of non-pollock which would have been landed had pollock landings increased by one pound. Log-transformed estimation of (2.4) produced unit-less pollock elasticity and elasticity shifter coefficients, identifying the percentage change in landings of non-pollock multispecies groundfish resulting from a one-percent increase in pollock. To extract MRPTs, the following equation was used:

$$MRPT_{i,t} = \frac{(1-\delta_t)[0.01\times\beta_{2i}]Q_{MG_{i,t}}+\delta_t[0.01\times(\beta_{2i}+\beta_{3i})]Q_{MG_{i,t}}}{0.01\times Q_{P_{i,t}}}. \tag{2.6}$$

Equation (2.6) indicates the MRPT for harvester *i* at time *t* is equal to the allocation-dependent non-pollock multispecies groundfish increase/decrease resulting from a one-percent increase in pollock landings, divided by a one-percent increase in pollock landings. The null hypothesis of costless avoidance was tested by inspecting the distribution of median fleet MRPT across posterior parameter draws.

To estimate the cost of a counterfactual, season-long low pollock allocation, the following steps were taken. First, we identified 109 harvesters who jointly landed pollock and non-pollock at least twice during both low and high allocations. Then, for each of these harvesters, 1,000 random draws were taken from $ln\ N(\mu^L_{P_i}, \sigma^L_{P_i})$, $ln\ N(\mu^L_{CI_i}, \sigma^L_{CI_i})$, $ln\ N(\mu^H_{P_i}, \sigma^H_{P_i})$, and $ln\ N(\mu^H_{CI_i}, \sigma^H_{CI_i})$, where $\mu^L_{P_i}$, $\mu^L_{CI_i}$, $\mu^H_{P_i}$, and $\mu^H_{CI_i}$ are the means of pollock landings and our composite input variable, on the natural log scale, for harvester *i* under low and high pollock allocations, with corresponding standard deviations σ. Using these random draws of individual covariates, 1,000 predictions of low and high allocation $\hat{Q}_{MG_i}$ were made for each of the 3,000 posterior parameter vectors. The following equation was then calculated for each harvester and posterior parameter vector to obtain an average trip-level estimate of forgone production (FP):

$$FP_i = \left(\bar{\hat{Q}}^H_{MG_i} - \bar{\hat{Q}}^L_{MG_i}\right) \times 1.63 + \left(\bar{\hat{Q}}^H_{P_i} - \bar{\hat{Q}}^L_{P_i}\right) \times 0.97. \tag{2.7}$$

Equation (2.7) estimates the difference in individual median non-pollock and pollock production, scaled by their respective average prices, corresponding to the allocation increase.[27] Note that in (7), the $\hat{Q}_{MG_i}$'s will vary according to posterior parameter estimates, though the $\hat{Q}_{P_i}$'s will remain fixed for a given harvester. Estimates of (2.7) could take any number of values, both positive and negative. For example, a positive value would indicate pollock quota constraint decreased joint production of non-pollock, production of pollock, or some mixture of the two. A negative value may result if the increase in pollock allocation decreased output of other species. While somewhat counterintuitive, this may also be viewed as a cost given that relaxing pollock constraint only serves to expand the set of possible trips. That more productive trips were forgone suggests they were sub-optimal, and therefore a negative estimate of (2.7) may be interpreted as a lower bound of opportunity cost. Fleet-level, season-long measures were constructed by averaging (2.7) across harvesters, for each posterior parameter vector, and then

---

[27] We use log-normal distributions and medians here due to the fat-right-tail common in fishery production.

multiplying by the expected number of joint landing trips, which was calculated as a simple extrapolation from the number of low allocation joint landing observations.

2.5 Results

2.5.1 Model comparison

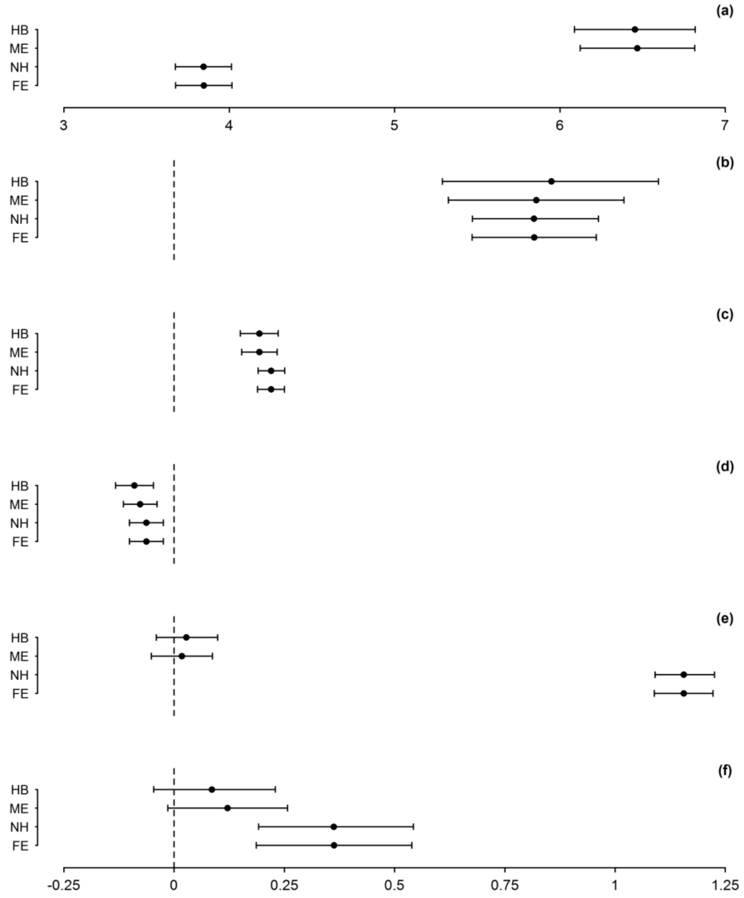


Figure 2.5 95% credible/confidence intervals of intercept (a), low allocation intercept shifter (b), pollock elasticity of output (c), low allocation pollock elasticity shifter (d), composite input (e), and zero-pollock intercept shifter (f) for hierarchical Bayesian (HB), mixed-effects (ME), non-hierarchical Bayesian (NH), and fixed effects (FE) specifications of (2.4). Points indicate median parameter estimates. Hyperparameter means (HB) and fixed effects (ME) are plotted in (a), (b), (c), and (d).

Introduction of individual production heterogeneity through the HB and ME specifications resulted in changes to several parameter estimates (Figure 2.5). Specifically, in the HB and ME models, the zero-pollock intercept shifter and composite input parameter became statistically no different from zero while the hyper-mean (HB) and fixed-effect (ME) for individual intercepts nearly doubled those values estimated for fleet-wide parameters. A shift in the variation explained by our composite input to individual intercepts is not completely surprising as the former is scaled by fixed capital, which is identical across observations for a single vessel. However, it is somewhat unexpected that intra-vessel variation in days fished ($\overline{\mathrm{cv}} = 0.55$), as well as trips landing zero pollock, had no significant explanatory power when including multilevel structure.

In addition to modifying parameter estimates, the HB and ME specifications were found to improve fits significantly. Allowing random deviations from fleet fixed effects in our ME model decreased AIC by 3,565 while incorporating hierarchical structure in HB lowered DIC by 4,611.[28] Burnham and Anderson (2002) assert an AIC difference greater than ten implies there is essentially no support for the larger AIC model. Here, when compared to multi-level formulations, both single-level models had information criterions several orders of magnitude greater than this generally accepted threshold. This result, together with the insignificance of β_4 and β_5, indicates $ln\left(Q_{MG_{i,t}}\right) = \beta_{0i} + \beta_{1i}\delta_t + \beta_{2i}ln\left(Q_{P_{i,t}}\right) + \beta_{3i}ln\left(Q_{P_{i,t}}\right) \times \delta_t + \varepsilon_{i,t}$ is an improved specification to (2.4), explaining landings of non-pollock multispecies groundfish solely by the amount of pollock jointly landed under a specific allocation regime. This finding

---

[28] Statisticians often exercise caution when using DIC as selection criteria in hierarchical settings due to the inexact meaning of "free parameters" which can lead to locally unstable estimates of the effective number of parameters in highly complex models. The DIC difference reported here is very much in excess of the maximum number of potentially free parameters however. Additionally, posterior predictive distributions indicate HB more closely resembles the data generating process than does NH.

suggests a degree of non-jointness in non-pollock/pollock landings which was a function of pollock quota constraint.

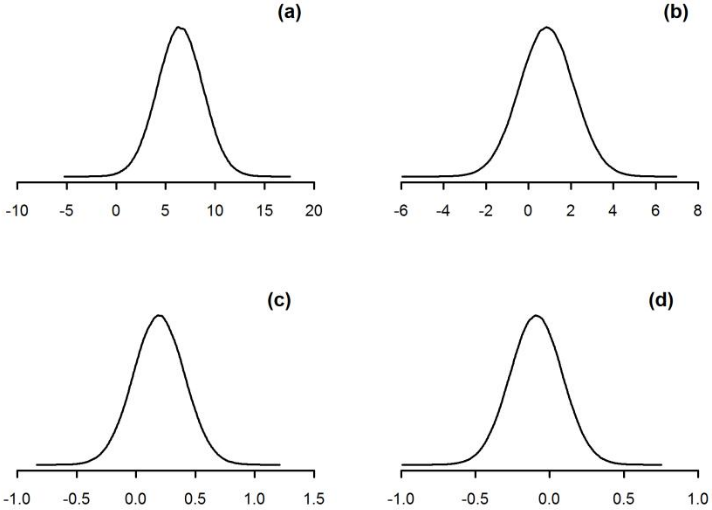


Figure 2.6 Densities of individual intercepts (a), low allocation intercept shifters (b), pollock elasticities (c), and low allocation pollock elasticity shifters (d) resulting from 1,000,000 random multivariate draws of $N(\bar{M}_\beta, \bar{\Sigma}_\beta)$, where $\bar{M}_\beta$ and $\bar{\Sigma}_\beta$ are median posterior hyperparameter estimates from the HB specification.

Taking multivariate random draws from our underlying population probability model, we find that, save for the intercept, all individual parameter distributions show significant densities on both sides of zero, highlighting the diversity of production within the New England groundfish fleet (Figure 2.6). The majority of low allocation intercept shifter draws were positive, indicating baseline production decreased for many vessels following the pollock allocation increase. This finding agrees with our observations of southern New England vessels, who, once pollock became less constraining, produced less non-pollock per trip. These vessels may have been avoiding pollock via costly harvest strategy. Such coarse targeting ability does not appear to have been ubiquitous however as significant probability mass exists at points less than zero for our pollock output elasticity measure, implying several harvesters were operating on the negatively sloped portions of their production isoquants.

2.5.2 Costly pollock avoidance

Table 2.1 Average landings of non-pollock multispecies groundfish (MG), pollock, and estimated MRPTs, under low and high allocations. Standard deviations (standard errors) are in parentheses beneath means (medians).

Allocation	Mean MG (lbs)	Mean Pollock (lbs)	Median MRPT
Low	6,547	370	0.687
	(13,831)	(1,885)	(0.252)
High	3,463	862	0.602
	(8,191)	(3,169)	(0.090)

Early in FY 2010, southern New England vessels made trips landing unusually large amounts of non-pollock together with small amounts of pollock while their northern counterparts decreased production from historical levels. We cannot accept the null of costless avoidance during this time, finding a statistically significant positive MRPT for the fleet (Table 2.1). While under the low allocation, a pound increase in pollock landings would have increased non-pollock production by 0.687 pounds on average. Once the pollock allocation increased, or rather once harvesters began to operate under the assumption that it would increase, average landings of non-pollock decreased while those of pollock increased, returning to regular intensities. The estimated median MRPT also decreased slightly to 0.602.

Using median MRPT estimates for each individual observation to look within and across harvesters, it was found that technological variation between harvesters decreased, while that within observations by a single harvester increased, following the pollock allocation increase. At all times, variation between harvesters was more than five times greater than that within however. These observations are suggestive of a poorly functioning quota market. Had transparent and frictionless mechanisms been in place, the 12% of harvesters found to have negative average MRPTs under the low allocation would likely have sold quota to those 15% whose MRPTs exceeded 10. Interestingly, the only factor found to be significantly predictive of MRPT was sector affiliation, with three sectors exhibiting average MRPTs four times larger than the rest of the fleet. At year's end, two of these sectors had leased more pollock than any other species while the third was a net lessor whose vessels reduced total production by 15%.

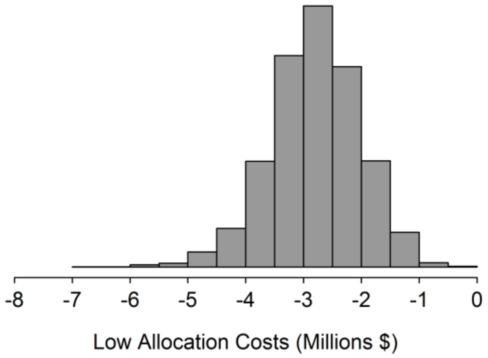


Figure 2.7 Posterior density of counterfactual low allocation costs

Predicting the costs associated with a low pollock allocation reveals that decreased pollock constraint decreased ex-vessel revenues. On average, ex-vessel revenues were predicted to have been $763/trip (s.e. $191) lower under the high allocation. Extrapolating this out for the season suggests that, had the low allocation remained, harvesters would have avoided pollock and simultaneously produced more non-pollock, increasing total revenues by US $3 million (Figure 2.7). The absence of these more productive and revenue enhancing trips following a relaxation of harvest constraint suggests they were more costly, and we can use this measure then as a lower bound on opportunity cost of the constraining allocation. During the first several weeks of FY 2010, harvesters were landing ~500 lbs less in pollock though earning $763 more in total revenues. This suggests that every pound of pollock they were avoiding was worth roughly $1.50, or about $0.50 more than pollock's ex-vessel price.

2.6 Discussion

The topic of costly avoidance in joint production of multispecies fisheries was explored here using two models. First, an individual input-constrained optimization model was analyzed after relaxing the common assumption of free output disposability. Results of this model suggest costly avoidance occurs in response to negative marginal rewards, potentially created through intense regulatory constraint, and is characterized by positive marginal rates of product transformation. An empirical model was then developed to identify costly pollock avoidance during the first year of catch share management in New England multispecies groundfish. We cannot accept the null of costless avoidance, finding statistically significant positive technological tradeoffs under low and high pollock allocations, however, paradoxically, for many vessels joint production was found to decrease after relaxation of harvest constraint. Further investigation suggests pollock avoidance likely occurred via costly location selection or harvest method, which, had the low allocation persisted, would have resulted in total costs of US $3 million.

Positive marginal rates of product transformation between pollock and non-pollock were found under both the low and high pollock allocations, indicating costly avoidance in both instances. We see three possible mechanisms for this result. First, though early on pollock was widely considered the primary choke species, following the allocation increase, shares of certain cod and flounder stocks were recognized as limiting. Early season costly pollock avoidance may have given way to costly avoidance of other stocks. Second, the production process is necessarily discrete and stochastic, over- or undershooting optimal targets is entirely possible. Harvesters may very well respond to economic and regulatory incentives, bound by their technology as modeled in (2.1) and (2.2), however, a high degree of tow-by-tow and soak-by-soak variation make this response problematic. Additionally, the negatively sloped portion of an underlying production isoquant may only cover a small range, exacerbating difficulties in achieving optimal production levels. Third, our model may not adequately capture variable inputs, confounding conclusions of costly avoidance with the effects of production expansion and contraction. While this latter mechanism cannot be ruled out, it is worth reiterating that our composite input measure is quite similar to that used in pervious multispecies production models, where free output disposal was an implicit assumption (e.g., Kirkley and Strand 1988, Squires and Kirkley 1991, Campbell and Nicholl 1994, and Thunberg et al. 1995).

The models developed in this work look to explore when, why, and how we might observe costly avoidance in a multispecies fishery. Similar questions have been empirically addressed by Scheld and Anderson (2014), who apply a non-parametric DEA estimator to uncover avoidance costs, in terms of forgone production, for the efficient harvester. The empirical work and findings discussed here are both complementary to Scheld and Anderson (2014) as well as novel. Utilizing a subset of the data we analyze, Scheld and Anderson (2014) address an identical policy question, though find significant costs to have resulted from forgone production of non-pollock species. There are several reasons for this difference in conclusion. First, landings by one-third of the vessels included in analysis here were excluded from their analysis to restrict spatial heterogeneity which may confound DEA measures. Scheld and Anderson (2014) estimate avoidance costs for vessels registered north of Cape Cod who do not frequent more diverse southern New England waters, and who, in this study, were commonly found to have been less productive under the constraining pollock allocation. Second, the methods used by Scheld and Anderson (2014) estimate avoidance costs by differencing the costs of disposal, or the difference in strong and weak disposability frontiers, before and after the allocation increase. Such methods do not incorporate costs related to shifts in individual production isoquants, which were found here to be substantial as many harvesters were more productive under the low allocation, suggesting they were operating on an outer, more costly isoquant. Third, output from DEA estimators reflects behavior of the efficient harvesters. For the segment of the fleet north of Cape Cod, efficient vessels avoided pollock by reducing landings of haddock, plaice, witch flounder, and monkfish and were generally less productive under the restrictive pollock allocation.

Our empirical findings suggest production heterogeneity may be a key factor in welfare impacts and policy outcomes of New England groundfish management. Statistical model selection techniques indicate production occurs along fixed individual isoquants, while analysis of parameter estimates reveal compositional control may largely be modified through costly harvest methods. Additionally, MRPTs were found to be highly variable, with large values concentrated among 38 members in three of 17 sectors. In heterogeneous, individually-inflexible multispecies fleets like the New England groundfish fleet, well-functioning quota markets may alleviate a degree of constraint by allowing harvesters to match multispecies catch to quota (Sanchirico et al. 2006). Unfortunately, protracted periods of price discovery have been commonly observed in lab (Anderson and Sutinen 2005) and field quota markets (Newell et al. 2005; Larkin and Milon 2000). Early in FY 2010, harvesters were reluctant to sell quota (Hemmerdinger 2010), and what trades did take place were identified as outliers during subsequent quota market analysis (Kitts *et al.*, 2011). A well-functioning quota market may have dramatically improved efficiency as those vessels easily able to avoid pollock would likely have been willing to lease quota to less-selective harvesters had quota value been transparent and transaction costs minimal.

The theoretical model of costly avoidance contained in this work represents a unique contribution to the literature, adding a layer of realism to the standard model which is often in conflict with our understanding of production processes in multispecies fisheries. Explicitly modeling landings technology focused analysis on the foregone production resulting from imperfect selectivity and constraining output controls – a practical, relevant, and, until now, unaddressed concern in multispecies catch share systems. Costly disposal joint production is of course not unique to the multispecies fishery and has been empirically investigated for firms emitting regulated pollutants (Färe et al. 1993; Chung et al. 1997; Färe et al. 2007), in agricultural systems (Zhengfei and Lansink 2003; Piot-Lepetit and Le Moing 2007), and for congestion problems in transportation settings (Dervaux et al. 1998). Utilizing a non-monotonic transformation function, our theoretical model provides a general structure that could be applied in a variety of costly disposal optimization problems.

As of 2010, over 250 catch share systems were in place in 35 countries (Bonzon et al. 2010). With the recent support of large and influential governmental and non-governmental agencies (e.g., NOAA 2010, EDF 2007), applications of this style of fisheries management are likely to increase. A unifying principle of catch share systems is their use of hard output caps, which may create conditions for costly avoidance behavior. In order to understand and quantify the economic and ecological tradeoffs in regulated multispecies fisheries, it is critical that applied and theoretical resource economists develop models capable of handling costly disposal joint production.

Chapter 3

Joint production of New England groundfish: weak pollock disposability

Abstract

In multispecies fisheries, targeting of individual stocks within the species complex is often difficult or impossible. This type of joint production technology can be modeled by considering outputs weakly disposable. In this study, we develop a multispecies production technology which accommodates imperfect output selectivity and apply it to joint production of New England groundfish. Over the last 20 years, managers have struggled to rebuild depleted New England groundfish species as non-selective gear together with a lack of avoidance incentives have stymied simultaneous exploitation of healthy stocks and conservation of weak. Recently, the multispecies fishery transitioned from effort restrictions to individual stock catch share management in an attempt to promote targeting and avoidance incentives while also meeting legal mandates which require hard total catch caps. Using data envelope analysis to construct multispecies output sets, we test the technological restriction of strong disposability on 156 individual species joint production pairs and also estimate the costs of avoiding a constraining low allocation species. Strong disposability is rejected in joint production nearly half the time, suggesting targeting and avoidance of individual species may be problematic and questioning the future success of catch share management in a fishery with stocks of mixed health. Additionally, a mid-season upward revision in allocation for a single species allows us to identify avoidance costs, in terms of forgone production, for jointly produced species. We find marginal and trip-level avoidance costs to be statistically and economically significant for certain target stocks and counterfactual season-long low allocation costs are estimated at US $6 million.

3.1 Introduction

Production processes in multispecies fisheries are often characterized by imperfect selectivity. Though ecological overlap often adheres to spatial, temporal, and oceanographic patterns and conditions (Dunn et al. 2011), compositional control by individual vessels has been found to be limited (Jannot and Holland 2013). When incentivized, harvesters have been observed to exercise targeting control through location choice (Branch and Hilborn 2008), gear depth, deployment duration, and timing (Grafton et al. 2004), and by adopting more selective harvesting equipment (Graham et al. 2007). Regulations in many managed multispecies fisheries set stock-specific guidelines and policies which, if harvest is not easily manipulated, may result in failed management targets or resource underutilization.

Applications of individual stock catch share management have been increasing globally for several decades. This style of management typically allocates annual harvest privileges to individuals or groups and is generally thought to incentivize rent maximization and resource stewardship (Hilborn et al. 2005; Grafton et al. 2006). Recent figures indicate over 200 programs are currently in place in several dozen countries (Bonzon et al. 2010), many of which regulate diverse multispecies fisheries. In 2010, the United States' National Ocean and Atmospheric Administration (NOAA) issued their Catch Share Policy (NOAA 2010) which urged regional fisheries managers to consider catch shares a mechanism to improve ecological and economic outcomes. Nearly in stride with this declaration, a number of new catch share programs were implemented in US fisheries. Currently, there are 15 US catch share programs, six of which manage multispecies complexes.

Following decades of failed management, declining population abundances, and bitter stakeholder resentment (Acheson and Gardner 2011), Northeast US fishery regulators transitioned the New England multispecies groundfish fleet from days-at-sea to catch share management. Once the premier US fishery (Ackerman 1941; Innis 1954), sustained overexploitation greatly depleted many commercially important stocks, potentially altering ecosystem structure and function (Fogarty and Murawski 1998). Days-at-sea management was introduced in 1994 in an attempt to curb harvest of the aggregate species complex by controlling individual effort. However, the absence of stock-specific targeting and avoidance incentives, combined with misguided management allowing excessive annual mortality limits (Acheson and Gardner 2011), resulted in continually decreasing groundfish populations and, subsequently, allocated days. Faced with an ecologically and economically failing fishery, and required by law to implement hard annual catch limits (MSRA 2007), on May 1st, 2010, New England managers allocated tradable quota for 14 stocks of nine groundfish species to 17 self-identifying harvest sectors (NEFMC 2009), all of whom chose to fish their allocations as individual transferable quota. During the first year, a fleet of over 400 vessels landed 65 million lbs of catch share regulated multispecies groundfish, receiving $90 million in ex-vessel revenues.

Empirical analyses of New England multispecies groundfish production have frequently rejected nonjointness and separability, indicating technological relationships exist between production factors and that management of individual stocks may be problematic (Squires 1987; Kirkley and Strand 1988). Otter trawl, the most common gear used, is largely non-selective, catching a variety of demersal stocks which reside on or just above the seafloor. In the three years preceding catch share management, vessels on average landed four of the nine allocated catch share species on each multispecies groundfish trip. Implementation of hard annual catch limits for New England multispecies groundfish through the 2010 catch share program would be challenging if allocations and catch composition were not aligned.

Ex-post, it appears that an imbalance did exist, or perhaps, quota exchange mechanisms, which work to balance fleet-level catch with quota (Sanchirico et al. 2006), had largely failed. In 2010, only 38% of the total allocation was harvested. Quota utilization was of course heterogenous across stocks. For example, harvesters caught 94% of the Georges Bank yellowtail flounder allocation, but only 16% of Georges Bank haddock. Significant levels of uncaught quota suggest imperfect selectivity among groundfish stocks: low allocation species could not be avoided, leaving large amounts of healthy stock quota unfished. Prior to implementation, a number of stocks were identified by harvesters as probable “choke species”, whose low allocations were expected to stifle production (Hemmerdinger 2010). Due to abundance estimates which indicated the stock was considerably depleted, pollock, a low-valued staple groundfish species, received an allocation of just 35% recent annual landing levels. Once the season began however, harvesters and industry members claimed that pollock was healthy, abundant, and nearly impossible to avoid (Gaines 2010). On July 15th, two and a half months into the season, following a more thorough stock assessment, the pollock allocation was increased six-fold. In this study we look to address two primary questions: 1) Is joint production in New England demersal fisheries characterized by imperfect selectivity?; and 2) Was early season pollock avoidance costly?

Multi-output production processes are commonly investigated using either frontier or dual methods. The former strategy relies on technological frontier estimation from joint production observations, while the latter employs market data and Hotelling’s lemma to estimate output-supply equations. Both methods have been applied to fishery production problems (see Jensen 2002 for an overview of dual applications). Here, we utilize data envelope analysis (DEA), a nonparametric linear programming method which constructs the production frontier directly from observed production data (Farrell 1957; Charnes et al. 1978). Imperfect selectivity among stocks in a multispecies fishery suggests reducing catch of one stock may only be possible through simultaneous reductions in catch of other stocks (Turner 1995, 1997; Singh and Weninger 2009). This form of multispecies technology, characterized by costly targeting and avoidance, can be modeled by treating outputs as weakly disposable – a looser technological restriction than strong, or free, disposability (Shephard 1970; Färe et al. 1994). Weakly disposable production technologies exhibit non-monotonic marginal costs, set-valued production correspondences, and cannot be modeled in a dual framework. DEA is a flexible, nonparametric tool, based on minimal assumptions which can handle multiple inputs and outputs and accommodate weakly disposable technologies. Additionally, contrary to dual methods, DEA makes no assumptions regarding technical or allocative efficiency, a desirable property when modeling fishery production where resource and environmental conditions, as well as individual skill or luck, may significantly affect production outcomes (Hilborn 1985; Hilborn and Ledbetter 1985; Squires and Kirkley 1999).[29] Our specification of an imperfectly selective multispecies production technology is similar to environmental technologies which model joint production of good and bad outputs, typically to investigate efficiency within industries emitting regulated pollutants (Färe et al. 1989; Chung et al. 1997; Färe et al. 2007).

The empirical model and results presented here add to the current literature in three significant ways. First, the technological restriction of strong disposability in joint production of New England demersal species is statistically tested and rejected for a number of species pairs, indicating a high degree of imperfect compositional control. Previous DEA models of joint production in multispecies fisheries have considered output weakly disposable (e.g., Färe et al. 2006, Kjærsgaard et al. 2009), however, testing this restriction on individual species pairs is a

---

[29] Note that stochastic frontier methods (Aigner et al. 1977) cannot be applied when considering weak disposability.

unique contribution. Second, we estimate and analyze pollock avoidance costs resulting from a severely constraining allocation. A mid-season pollock allocation increase allows us to identify the economic costs of the low allocation by comparing disposal costs before and after the allocation shift, providing a unique ex-post perspective of the costs and difficulties inherent in output-managed multispecies fisheries. Third, when estimating pollock avoidance costs we apply a propensity score weighted m-bootstrap. It is well known that the standard n-bootstrap (Efron 1979), where a statistic is computed after drawing n samples with replacement from the original n-observation data set, is inconsistent for bounded support problems (see Bickel and Freedman 1981 and Mammen 1992 for a general discussion and Simar and Wilson 1999a, b for discussion with respect to DEA estimators). Consistency has recently been proved for sub-sampling bootstrap procedures in DEA estimators however (Kneip et al. 2008). The m-bootstrap, where $m < n$, is computationally simple and useful when calculating statistics derived from DEA efficiency estimates or when testing technological restrictions (Simar and Wilson 2011). Our propensity score weighted m-bootstrap allows us to evaluate the sampling distribution of our test statistic while also accounting for uncertainty in the assumption of homogenous decision making units.

The remainder of this paper is organized as follows. First, we specify a multispecies joint production technology that allows for both strong and weak disposability of outputs, from which we construct two reference technologies, one allowing only perfect targeting among stocks and another which permits imperfect control. Then, our DEA estimators, avoidance cost measures, and the m-bootstrap are described, followed by a section detailing data and methods. Results from strong disposability tests and the m-bootstrapped low allocation costs are then presented. A short discussion concludes the paper.

3.2 Production technology

The technology defined here is similar to that often considered when modeling joint production of desirable and undesirable outputs (see e.g., Färe et al. 1989, Chung et al. 1997, Seiford and Zhu 2002, and Färe et al. 2007). In these models, undesirable output is treated as weakly disposable with respect to desirable output and production of desirable and undesirable outputs is considered null-joint. That is, reducing undesirable output requires a reduction in desirable output (weak disposability) and to produce any desirable output requires some production of undesirable output (null-joint). In the multispecies fishery, joint production is such that avoiding one stock may reduce output of other jointly caught stocks (weak disposability) and to produce any of one stock may require some production of another jointly caught stock (null-joint). For our purposes, undesirable outputs may be considered imperfectly targeted species.

When considering a multispecies production technology, we denote inputs by $x = (x_1, \ldots, x_Q) \in \mathbb{R}_+^Q$, perfectly targeted outputs by $y = (y_1, \ldots, y_R) \in \mathbb{R}_+^R$, and imperfectly targeted outputs by $u = (u_1, \ldots, u_S) \in \mathbb{R}_+^S$. Defining the production technology as:

$$T = \{(x, y, u) : x\ can\ produce\ (y, u)\}, \tag{3.1}$$

we may introduce the following restrictions, where the inequalities of (3.1a) are element-by-element:

$$(x, y, u) \in T,\ x' \geq x\ ,\ y' \leq y\ \ imply\ (x', y', u) \in T, \tag{3.1a}$$

$$(x, y, u) \in T \; and \; 0 \leq \theta \leq 1 \;\; imply \; (x, \theta y, \theta u) \in T, \tag{3.1b}$$

$$if \; (x, y, u) \in T \; and \; u = 0 \; then \; y = 0. \tag{3.1c}$$

Equation (3.1a) indicates that perfectly targeted outputs and inputs are strongly disposable. Producing less of perfectly targeted outputs or using more inputs is always feasible, *ceteris paribus*. Equation (3.1b) indicates that imperfectly targeted outputs are weakly disposable with respect to perfectly targeted outputs. Proportional reductions in weakly disposable factors are feasible (Shephard 1970). Equation (3.1c) imposes the null-joint condition on production of perfectly and imperfectly targeted outputs.

If we consider *k* observations of production inputs and outputs, we may define two output oriented reference technologies, one considering all outputs as perfectly targeted (*PT*) and another allowing for imperfect targets (*IT*):

$$\begin{aligned} P(x)^{PT} = \{y : \; & \textstyle\sum_{k=1}^{K} \lambda_k x_{q,k} \leq x_q \quad q = 1, \dots, Q \\ & \textstyle\sum_{k=1}^{K} \lambda_k y_{r,k} \geq y_r \quad r = 1, \dots, R \\ & \textstyle\sum_{k=1}^{K} \lambda_k = 1 \\ & \lambda_k \geq 0 \quad k = 1, \dots, K \}, \end{aligned} \tag{3.2}$$

$$\begin{aligned} P(x)^{IT} = \{(y, u) : \; & \textstyle\sum_{k=1}^{K} \lambda_k x_{q,k} \leq x_q \quad q = 1, \dots, Q \\ & \theta \textstyle\sum_{k=1}^{K} \lambda_k y_{r,k} \geq y_r \quad r = 1, \dots, R' \\ & \theta \textstyle\sum_{k=1}^{K} \lambda_k u_{s,k} = u_s \quad s = 1, \dots, S \\ & \textstyle\sum_{k=1}^{K} \lambda_k = 1 \end{aligned} \tag{3.3}$$

$$0 \leq \theta \leq 1\,,\ \lambda_k \geq 0 \quad k = 1, \dots, K\,\}.$$

The inequality and equality signs for inputs and outputs in (3.2) and (3.3) correspond to strong and weak disposability, respectively. The k λ's are intensity variables which are used to construct the feasible technology polytope from convex combinations of observed inputs and outputs. Variable returns to scale is imposed by requiring the summation of intensity variables equal one (Banker et al. 1984). The scalar factor θ allows simultaneous contraction of perfectly and imperfectly targeted outputs and is necessary when constructing a weakly disposable, variable returns to scale technology (Färe and Grosskopf 2003; Zhou et al. 2008; Färe and Grosskopf 2009). Note additionally that (3.2) and (3.3) are compact and allow for inactivity – common restrictions placed on output sets (Shephard 1970; Färe et al. 1994).

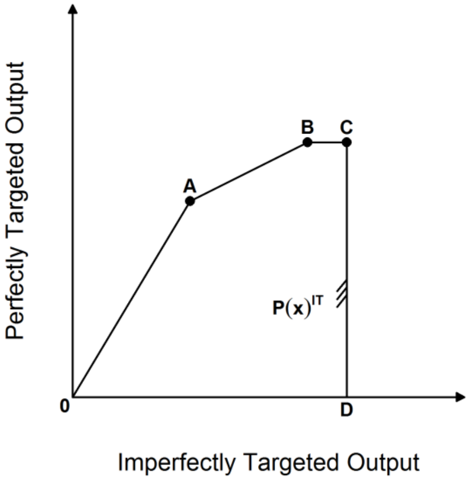


Figure 3.1 Multispecies output set allowing for weak disposability of an imperfectly targeted output.

The difference between strong and weak output disposability can be seen in Figure 3.1, which depicts $P(x)^{IT}$ constructed from three joint production observations at a single input level. The piecewise linear production frontier is formed by the segments connecting 0ABCD. Weak disposability of imperfectly and perfectly targeted outputs can be seen in the segment 0A, which is the ray of proportional output decreases contracting from observation A. Strong disposability of the perfectly targeted output can be seen in the segment CD, which shows that at this largest amount on imperfectly targeted output, reductions in the perfectly targeted output never reduce output of the imperfect target.

3.3 DEA specification

DEA is used here to address two questions: 1) Is joint production in New England demersal fisheries characterized by imperfect selectivity?; and 2) Was early season pollock avoidance costly? The following directional distance functions (Chambers et al. 1996; Chung et al. 1997) will enable investigation of both questions:

$$\vec{D}_O^{PT}(x, y, u) = max\{\beta : y + \beta g_y^{PT} \in P(x)^{PT}\}; \tag{3.4}$$

$$\vec{D}_O^{IT}(x, y, u) = max\{\beta : (y + \beta g_y^{IT}, u) \in P(x)^{IT}\}. \tag{3.5}$$

In (3.4) and (3.5), g_y^{PT} and g_y^{IT} are $\mathbb{R}_+^R$ and $\mathbb{R}_+^{R'}$ dimensional vectors that define the direction inefficient, perfectly targeted outputs are shifted to reach the efficient boundaries of our reference technologies. Directional distance functions are often applied to environmental technologies, though generally these functions are constructed to allow for simultaneous increases in desirable outputs and decreases in undesirable outputs (see Zhang and Choi 2014 for a recent review of environmental directional distance functions). Here, it is advantageous to define our directional vectors as increasing in the direction of a single output, considered a perfect target under both reference technologies.

For an observation k_0, solution of the directional distance functions (3.4) and (3.5) is achieved via the following two linear programming problems:

$$\begin{aligned}
&\max_{\beta_{k_0}^{PT},\boldsymbol{\lambda}} \ \beta_{k_0}^{PT} \\
&s.t. \ \textstyle\sum_{k=1}^{K} \lambda_k x_{q,k} \leq x_{q,k_0} \quad q = 1, \dots, Q \\
&\qquad \textstyle\sum_{k=1}^{K} \lambda_k y_{r,k} \geq y_{r,k_0} + \beta_{k_0}^{PT} g_{y_r}^{PT} \quad r = 1, \dots, R \\
&\qquad \textstyle\sum_{k=1}^{K} \lambda_k = 1 \\
&\qquad \lambda_k \geq 0 \quad k = 1, \dots, K;
\end{aligned} \tag{3.6}$$

$$\begin{aligned}
&\max_{\beta_{k_0}^{IT},\boldsymbol{\mu},\theta} \ \beta_{k_0}^{IT} \\
&s.t. \ \textstyle\sum_{k=1}^{K} \mu_k x_{q,k} \leq \theta x_{q,k_0} \quad q = 1, \dots, Q \\
&\qquad \textstyle\sum_{k=1}^{K} \mu_k y_{r,k} \geq y_{r,k_0} + \beta_{k_0}^{IT} g_{y_r}^{IT} \quad r = 1, \dots, R' \\
&\qquad \textstyle\sum_{k=1}^{K} \mu_k u_{s,k} = u_{s,k_0} \quad s = 1, \dots, S \\
&\qquad \textstyle\sum_{k=1}^{K} \mu_k = \theta \\
&\qquad \mu_k \geq 0 \quad k = 1, \dots, K \\
&\qquad 0 \leq \theta \leq 1,
\end{aligned} \tag{3.7}$$

where $g_{y_r}^{PT}$ and $g_{y_r}^{IT}$ are individual output directions, each of which is set equal to zero except for a single perfectly targeted output *r*, whose direction is set equal to one. A change of intensity variables in (3.7), i.e. $\mu_k = \theta\lambda_k$, affords a linearization of (3.3) and allows us to avoid the more complicated non-linear programming problem (Zhou et al. 2008). Solutions to (3.6) and (3.7) find the additive factor which places k_0 along the efficient boundary of each respective reference technology constructed from all *k* observations; if $\beta_{k_0} = 0$, k_0 would be considered efficient.

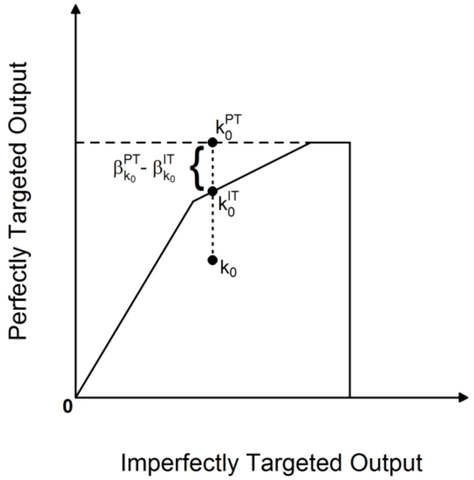


Figure 3.2 Estimating disposal costs as the difference between strong (dashed) and weak (solid) disposability frontiers for an imperfectly targeted multispecies output

Assuming output *i* is a perfect target in both (3.6) and (3.7), $g_{y_i}^{PT} = g_{y_i}^{IT} = 1$, and $g_{y_j}^{PT} = g_{y_j}^{IT} = 0 \; \forall \; j \neq i$, the two measures returned for an observation *k*, β_k^{PT} and β_k^{IT}, identify *i*-directional distances to the efficient frontiers of $P(x)^{PT}$ and $P(x)^{IT}$, respectively. Differencing these estimates yields a measure of disposal cost, in terms of output *i*, for imperfect targets *u* at observation *k*:

$$Disposal\ Cost_{u,k}^{i} = \beta_k^{PT} - \beta_k^{IT} : \ \ g_y^{PT} = \hat{e}_i^{PT}, g_y^{IT} = \hat{e}_i^{IT}, \tag{3.8}$$

where $\hat{e}_i^{PT}$ and $\hat{e}_i^{IT}$ are $\mathbb{R}_+^R$ and $\mathbb{R}_+^{R'}$ dimensional unit vectors in the *i* direction. For environmental technologies, this measure has been shown to be equivalent to pollution abatement costs as estimated by environmental production functions and can be understood as the opportunity cost of a regulation which restricts free disposal (see Färe et al. 2007). In a multispecies production setting, this measure captures the degree of imperfect selectivity among stocks. A large disposal cost indicates producing less of the imperfect target significantly reduces joint production of the perfect target. Figure 3.2 depicts disposal costs for an imperfectly targeted output at observation k_0, seen as the difference between strong and weak disposability frontiers. Clearly, the magnitude of disposal costs will be driven by observations at low levels of the imperfectly targeted output. If small amounts of the imperfectly targeted output are observed together with high levels of the perfectly targeted output, the two frontiers will largely coincide and disposal costs will be zero. To answer the research questions posed earlier, (3.8) will be estimated for individual species pairs, measuring disposal costs of each species with respect to every other species jointly produced.

3.4 *m*-Bootstrap

Bootstrapping bounded support problems by drawing *n* samples from the empirical distribution of *n* observations leads to inconsistent results as boundary points are drawn too frequently. To illustrate this issue, consider the probability of drawing a single observation from the empirical distribution to be 1/*n*. The probability of not drawing this observation over the course of *n* draws, with replacement, is then $(1 - n^{-1})^n$. Now if a boundary test statistic, say the sample maximum, is computed at each n-bootstrap iteration, the bootstrap estimate will equal the original estimate with probability $1 - e^{-1} \approx 0.632$ as $n \rightarrow \infty$, not zero as would be required for a consistent bootstrap estimator (Bickel and Freedman 1981; Efron and Tibshirani 1993).

In DEA, efficiency scores or distance metrics are defined with respect to the empirical production frontier and are therefore, boundary measures. Failure of the standard bootstrap in DEA applications has been remedied through two different approaches. The first, an n-bootstrap, initially recommended sampling from a smoothed empirical distribution (Simar and Wilson 1998; Simar and Wilson 2000), though for consistency a double smoothing procedure – smoothing both the empirical distribution as well as frontier estimates – was later shown to be necessary (Kniep et al. 2008). The second approach, drawing bootstrap samples of $m < n$, has been considered in general bounded support problems for some time (see e.g., Bickel and Freedman 1981 and Politis and Romano 1994), though only recently has this method been extended to DEA estimators (Kniep et al. 2008; Simar and Wilson 2011). Including and explicitly testing for weak disposability limits our bootstrap options to the latter, which can accommodate this type of technology and also test technological restrictions (Simar and Wilson 2011)

Consistency of the *m*-bootstrap is generally shown to hold if *m* is chosen such that $m \rightarrow \infty$ and $m/n \rightarrow 0$ as $n \rightarrow \infty$. In application, the choice of *m* is non-trivial. Simulation results of Kniep et al. (2008) indicate statistic coverages can be sensitive to the choice of *m*, where an *m* which is too low or too high results in over- or under-coverage, respectively. Selecting the best *m* through a data driven approach first proposed by Politis et al. (2001), Simar and Wilson (2011) choose *m* by minimizing volatility in the confidence bounds of their estimated statistic, calculated at each of fifty different bootstrap sample sizes. Replicating the strategy of Simar and

Wilson (2011) here would require solving billions of linear programming problems, each with thousands of constraints. We choose to take a more modest approach and apply an *m*-bootstap at four different sample sizes.

3.5 Data and methods

We consider production of 13 species traditionally harvested by otter trawl: Atlantic cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), pollock (*Pollachius virens*), Acadian redfish (*Sebastes fasciatus*), yellowtail flounder (*Limanda ferruginea*), American plaice (*Hippoglossoides platessoides*), witch flounder (*Glyptocephalus cynoglossus*), winter flounder (*Pseudopleuronectes americanus*), white hake (*Urophycis tenuis*), monkfish (*Lophius americanus*), little skate (*Leucoraja erinacea*), winter skate (*Leucoraja ocellata*), and spiny dogfish (*Squalus acanthias*). Catch shares were implemented for the first nine species listed. Monkfish was regulated in 2010 with days-at-sea, while skate and spiny dogfish were trip-limit regulated.

To investigate weak disposability among these 13 species, we solved linear programming problems (3.6) and (3.7) for individual species pairs using 2010 ex-vessel landings transaction data from all New England sector vessel trips catching federally managed species, mandatorily reported by all permitted dealers. Spatial and temporal restrictions were imposed on this dataset to limit the influence of exogenous factors affecting production, leading to the exclusion of landings by 103 vessels who regularly fish southern New England waters and all landings which occurred during the latter seven months of the fishing year.[30] The final sample contained 4,102

---

[30] An additional 119 vessels were excluded because they either did not land any of the 13 species considered during the specified dates, had a hook gear permit, or could be positively identified as using fixed gear. Our dataset included only landings from the first five months of the fishing year (May – September) to control for seasonal effects and also to temporally balance the set of observations between low and high pollock allocations.

trips made by 173 vessels, all docked north of Cape Cod, from May 1st through September 30th, 2010. Total landings from these trips amounted to 19 million pounds, 97% of which was from included species. Over 80% of trips landed more than one species while roughly 50% landed over five.

Addressing questions of production disposability using landings data frames the production decision at the trip-level. While this is a departure from the ecological and fisheries literature, which has generally investigated imperfect selectivity using tow-by-tow data (e.g., Dunn et al. 2013, Jannot and Holland 2013), fisheries economists have tended to model multispecies production at the trip- (Squires and Kirkley 1991; Campbell and Nicholl 1994) or annual-level (Squires 1987; Kirkley and Strand 1988), as this is where the majority of economic decision making takes place. In New England, regulations primarily affect trip-level behavior, inputs (e.g., crew, fuel, ice) are largely decided on a trip-by-trip basis, and inter-tow decisions are often made to adjust trip output.[31] Testing for, and evaluating the consequences of, weak output disposability at the trip-level broadens the scope of production technology to that which occurs after several decision making opportunities, possibly at many different locations and over the course of multiple days. A similar analysis using tow-by-tow data might fail to account for important inter-tow decisions which serve to actively manipulate catch composition, biasing disposal and avoidance cost measures. Weak disposability between trip outputs i and j indicates existence of jointly productive areas (in horizontal or vertical space), fishing times or other harvest methods, and/or stochastic environmental or oceanographic factors which result in sporadically high spatial co-occurrence. Costly avoidance in response to a constraining allocation

---

[31] Note that although catch shares apply to tow-by-tow harvest, regulatory observation, accounting, and quota management all generally take place at the trip level.

suggests advantageous production strategies were forgone, giving up catch of the target, or desirable, species to reduce catch of the avoided, or undesirable, stock.

In New England demersal fisheries, joint production possibilities are determined in part by gear used. The majority of vessels employ otter trawl, though there are several fixed gear and hook and line operations. Otter trawl vessels tend to be larger and have access to more distant fishing grounds – their joint production decisions are in many ways not comparable to those of smaller vessels using different gear. Unfortunately, publicly available permit data only allows separation of hook from trawl and fixed gear vessels. There was however one homogenous group of ~40 Cape Cod gillnetters, called the "Fixed Gear Sector", who could be positively identified and removed from our data. To control for gear when testing strong disposability restrictions on individual species pairs, we removed all vessels less than 40ft in length, leaving a sample of 131 vessels and 2,857 trips. When estimating the costs of a low pollock allocation, we applied an *m*-bootstrap and weighted sampling draws by otter trawl propensity scores which were calculated as one minus the predicted probability of being a Fixed Gear Sector vessel. Predicted probabilities were estimated with a probit regression (pseudo $R^2 = 0.684$) which used vessel length and individual deviations from annual state-level quantity averages in landings of cod, haddock, winter flounder, and winter skate to identify vessels similar to members of the Fixed Gear Sector.[32]

Linear programming problems (3.6) and (3.7) were solved in R (R Core Team 2013) using the lpSolveAPI package (Konis 2013). We considered two inputs: days spent fishing and a composite measure of fixed inputs equal to the first principle component of vessel length, gross tons, and horse power; and three outputs: landings of the two individual species for which

---

[32] Individual deviations from state averages were used in place of actual landings because the latter are confounded with Fixed Gear Sector probabilities since all Fixed Gear Sector vessels were from Massachusetts.

disposal costs were being calculated and the sum of all other outputs landed that trip. To test the technological restriction of strong disposability on individual species pairs, we computed 96,544 disposal costs corresponding to 48,272 joint landing observations (i.e., if *i* and *j* were landed together, disposal costs were calculated for *i* in terms of *j* and *j* in terms of *i*). Median disposal costs for each of the 156 joint landing combinations were then calculated and tested against a null of zero using nonparametric sign tests contained in the BSDA package (Arnholt 2012).[33] To examine pollock avoidance costs, we differenced median pollock disposal costs under the high allocation from those under the low allocation for each of the 12 jointly landed species. A positive difference, indicating disposal costs were higher under the low pollock allocation, implied pollock avoidance decreased joint production. Medians were used in general due to the fat-right-tail prevalent in fishery production data, however, relying on the median, rather than mean, also served to remove distributional effects from our measure of avoidance costs, which may be interpreted as the expected trip-level reduction in joint production for a randomly chosen vessel within the fleet.

The following steps were taken to bootstrap pollock avoidance costs for each jointly landed species: 1) Draw equiproportional samples of joint landing observations from May 1st – July 15th (low allocation) and July 16th – September 30th (high allocation), where each individual draw is done with replacement and using propensity score weights; 2) Compute median pollock disposal costs in each period; 3) Difference median pollock disposal costs and store this estimate; 4) Repeat 1000 times for each of m = 25, 50, 75, and 99%. We marginalized the resulting trip-level measure by dividing by the difference in trip median pollock landings from joint production observations under the low and high allocations. Multiplying this by the average ex-vessel price

---

[33] There were no cases where two individual species were never landed together. The fewest joint landings (4) occurred between little skate and redfish while the most (1,755) were between cod and monkfish.

of the target species produced a statistic which captured the forgone target-species dollars per pound of pollock avoided. Two methods were used to select an optimal bootstrap sample size. First, we applied the method of Politis et al. (2001), choosing *m* by minimizing the sum of running standard deviations on the 95% confidence bounds at increasing sample sizes.[34] We also constructed and minimized a volatility index which was equal to the sum of absolute deviations from the mean of 95% confidence bounds. Both measures select *m* by evaluating stability in confidence bounds.

---

[34] Note this implies an *m* of 25% could never be chosen with this method.

3.6 Results

3.6.1 Strong disposability tests

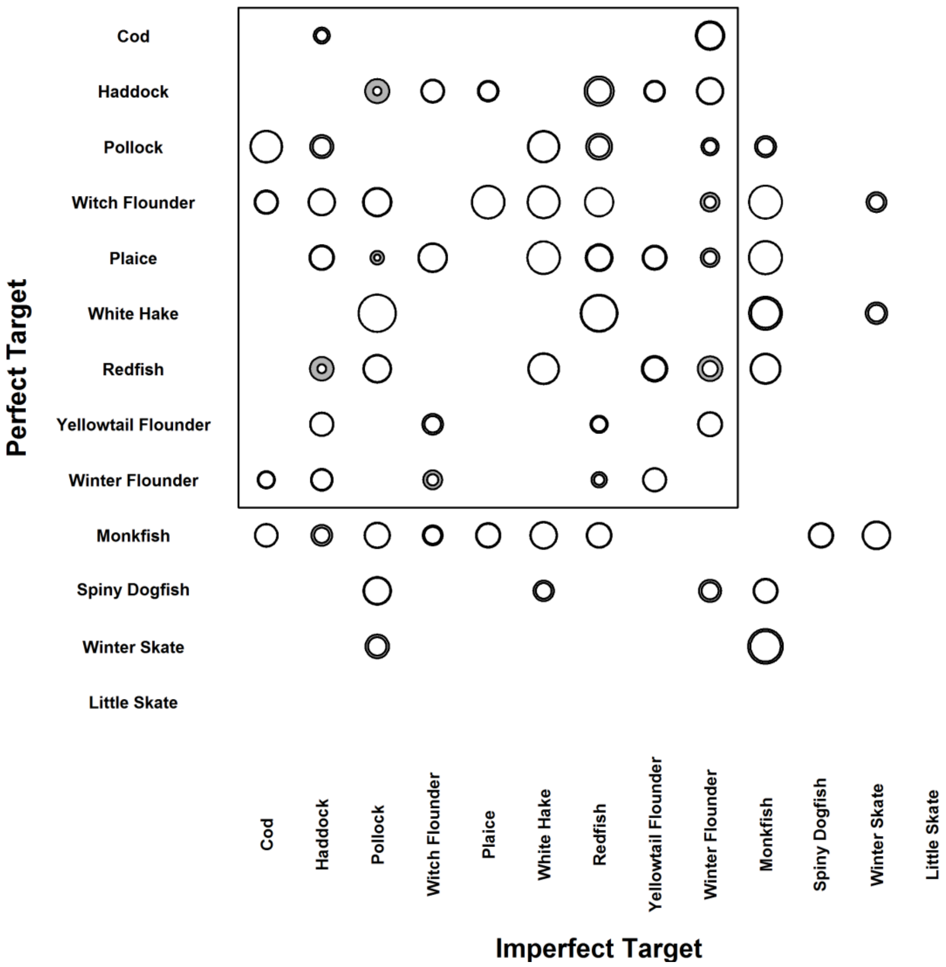


Figure 3.3 Median disposal cost 95% confidence rings for column species when jointly producing row species. Radaii of outer (inner) rings correspond to the natural log of linearly interpolated upper (lower) median confidence bounds. Larger rings indicate greater median disposal costs while grey area between outer and inner rings indicates uncertainty in median estimates. Catch share species are in northwest box. Species are ordered top-to-bottom/left-to-right by revenue importance within catch share and non catch share groupings.

Of the 156 joint landing combinations, the null hypothesis of strong disposability was rejected in 69 instances (Figure 3.3). Weak disposability was found to be more common among catch share species (63%) than between those regulated with days-at-sea and trip limits (8%). This result aligns with ex-ante expectations as discard of catch share species was strictly prohibited and enforced through onboard observers (~30% of trips) and retrospective analysis of reported catch and landings. The greatest median disposal costs were for pollock when jointly landing white hake (5088 lbs), suggesting certain trip-level harvest strategies or stochastic variables produce large simultaneous increases in pollock and white hake production. Haddock, pollock, redfish, winter flounder, and monkfish showed significant weak disposability with more than half of all other included species. In 2010, haddock, pollock, and redfish received the largest allocations and were considered to be the healthiest and most abundant multispecies groundfish stocks; widespread weak disposability among these species agrees with this general perception.[35] That highly abundant species could not be avoided without simultaneous decreases in landings of joint targets is not suprising, nor is it overly concerning. The presence of several mismatches in allocations, paired with statistically significant imperfect selectivity, is however worrisome and predictive of seasonally choked production. For example, avoiding cod when targeted pollock, white hake when targeting redfish, or any of the flounder species when targeting haddock, is potentially only feasible through reductions in landings of these quota-abundant species.

Disposal costs were found to be highly variable both within ($\overline{cv} = 2.2$) and across species ($cv = 3.2$). Nearly half of all measures were less than 1 lb, though the interquartile range was greater 850 lbs and several disposal costs were many thousands of pounds. Such a high

---

[35] This statement refers to managers' attitudes following the pollock allocation increase.

degree of variation among these measures suggests fleet heterogeneity in production decisions and/or a significant amount of environmental variability.[36]

Table 3.1 Select output from OLS regression of the natural log of disposal costs on sector, state, and month fixed effects, a low pollock allocation dummy variable, perfect and imperfect target species lbs, total trip lbs less those of perfect and imperfect targets, trip length, vessel size, and the share of landing vessel's total 2009 revenues attributed to the perfect target (PT) and imperfect target (IT). All disposal costs were increased by one unit before the natural log was taken. $N = 96{,}544$, $R^2 = 0.073$.

Variable	Coefficient	P-value
Maine	0.7595	0.000
New Hampshire	0.6985	0.000
May	- 0.2166	0.000
June	- 0.1113	0.044
July	- 0.0176	0.658
August	- 0.0097	0.788
Revenue Share PT	- 1.7330	0.000
Revenue Share IT	- 1.1480	0.000

Regressing disposal costs on observable covariates explains only a small amount of total variation (Table 3.1). This is not suprising as the vast majority of variables influencing joint production decisions (e.g., individual quota constraints, production costs, expectations) as well as realizations (e.g., environmental and oceanographic conditions) were unavailable or

---

[36] While, typically, DEA estimators are used in strictly homogenous production environments (or environments which are assumed to be so), and methods to control for noise and exogenous factors do exist (see e.g., Fried et al. 2002 for a three-stage approach and Kjærsgaard et al. 2009 for an application to a multispecies fishing technology), here we make no such effort to define efficiency scores apart from the effects of exogenous environmental or regulatory parameters. This is defensible because: 1) We are interested in fleet level disposal cost averages, not individual efficiency scores; 2) Controlling for such factors would introduce estimation error via staged equations, potentially biasing disposal cost measures; and 3) Changes in disposal costs may largely be driven *by* exogenous variables, eliminating these forces would be counterproductive to our research agenda.

unobservable.[37] Sector, home state, and month fixed effects were all found to be statistically significant however, indicating joint production outcomes varied predictably at spatial, temporal, and organizational levels.[38] Disposal costs were found to be increasing throughout the time period considered, possibly due to increased quota constraint for both catch share and days-at-sea species producing increased avoidance behavior. Interestingly, increases in individuals' 2009 perfect and imperfect target species revenue shares, proxies for a vessel's experience, were found to significantly decrease disposal costs, implying those more familiar with and dependent on joint production of particular species pairs were able to consistently land them together in large quantities.

3.6.2 Pollock avoidance costs

A constraining initial pollock allocation forced many harvesters to forgo landings of other species in order to decrease joint production of pollock. Comparing median disposal costs before and after the change in pollock allocation provides us with a trip-level measure of this loss.

---

[37] In several regressions we included lagged and concurrent data on wind speed and wave height, recorded by NOAA's Georges Bank buoy. Interestingly, these variables never showed a significant effect on disposal costs, suggesting they are poor proxies of ecological processes and/or production costs.

[38] Sector fixed effects were excluded from Table 3.1 for confidentiality

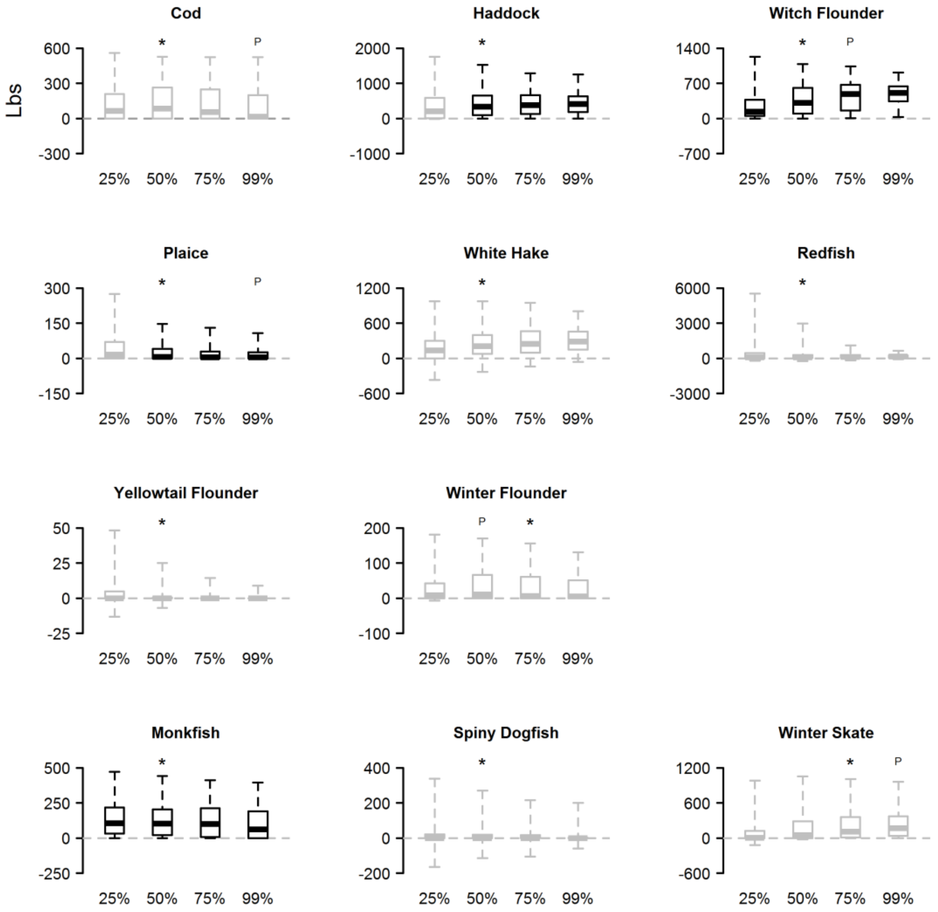


Figure 3.4 Box-and-whisker plots of *m*-bootstraped pollock avoidance costs. Box indicates interquartile range and whiskers cover 95% confindence interval. Light grey box-and-whiskers indicate median is not statistically different from zero at a 95% confindence level. Sample size specified on x-axis. Optimal sample size as chosen by the Politis et al. (2001) method denoted by "P", our method denoted by "*". If both methods select the same sample size, only "*" is shown. Note difference in scale of y-axis across species. Species are ordered left-to-right/top-to-bottom by revenue importance within catch share (top three rows) and non catch share groupings.

For all jointly landed dermersal species and *m*-bootstrap sample sizes, median pollock avoidance costs were greater than zero (Figure 3.4).[39] This suggests that pollock is integral to multispecies joint production in New England and that the low allocation constrained landings,

---

[39] Pollock avoidance costs for little skate were undefined as these two species were not jointly landed following the pollock allocation increased.

revenues, and profits during the first few months of the 2010 fishing year, costing harvesters $758/trip (s.e. $348) in forgone production. Once the allocation increased, vessels were able to increase landings of pollock and, simultaneously, landings of other jointly produced demersal stocks. Avoidance costs were statistically significant at a 95% confidence level for most bootstrap sample sizes of haddock, witch flounder, plaice, and monkfish. These species appear to be the most vulnerable to production constraint imposed by a low pollock allocation. Constructing a simple counterfactual, wherein we assume the fleet would have landed haddock, witch flounder, plaice, and monkfish, together with the avoided pollock, at frequencies equal to those observed under the low pollock allocation, a season-long constraining pollock quota would have cost the fleet US $6.8 million (s.e. $3.1 million) in forgone production. Increasing the allocation mid-season resulted in losses of just over a million dollars instead.

Optimal bootstap sample sizes were selected using two methods which minimized volatility in 95% confindence bounds. The methods agreed in six out of 11 instances, each time choosing an *m* of 50%. This suggests that boundary points were oversampled when bootstrapping with a sample size greater than half of *n*. The relative presence of outliers can be inferred through changes in coverage for a single species across *m*. Significant shrinking of 95% confidence bounds occurs as a result of decreasing leverage for individual outliers when calculating median disposal costs. Plaice, redfish, yellowtail flounder, and spiny dogfish all exhibited large relative decreases in their 95% confidence range. Outliers in production of these species jointly with pollock may be the result of seasonal migratory patterns (spiny dogfish migrate up the coast throughout the summer), fleet heterogeneity (a small group of highliners occassionally lands very large quantities of redfish), and/or a high level of stochastic mixing with pollock.

Across species, the scale of avoidance costs differed markedly. For example, avoiding pollock when also producing haddock or witch flounder led to large decreases in landings of these species as harvesters abstained from jointly productive areas, times, or methods, giving up haddock and witch flounder production in favor of lower pollock landings. For other species, like cod, winter flounder, yellowtail flounder, and spiny dogfish, pollock avoidance was fairly costless, suggesting an ability in joint production of these species to refocus landings away from pollock and toward less constraining stocks. Interestingly, white hake and redfish, species which exhibited large and significant weak disposability with pollock, did not display statistically significant avoidance costs. The reason for this appears to be complete avoidance however, as these species saw 50% fewer joint landings under the low allocation.

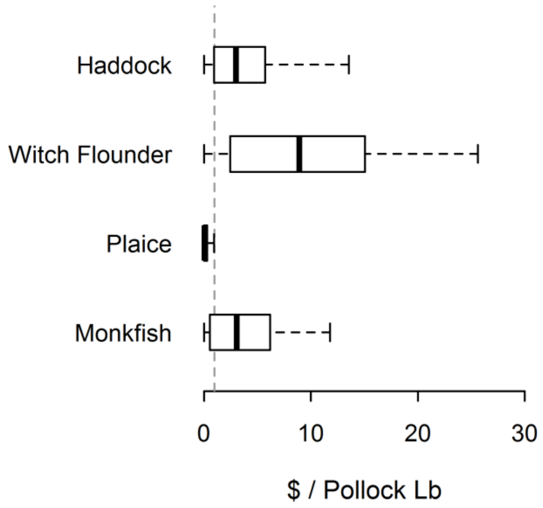


Figure 3.5 Box-and-whisker plot of avoidance costs in dollars of target species per pound of pollock avoided. Box indicates interquartile range and whiskers cover 95% confindence interval. Only estimates significantly greater than zero at a 95% confidence level for optimal bootstrap sample sizes as selected by our method are shown. Grey dashed line indicates average pollock

ex-vessel price per pound. Top-to-bottom ordering reflects revenue importance among catch share species (first three).

Haddock, witch flounder, plaice, and monkfish were found to have statistically significant median pollock avoidance costs of \$2.94, \$8.11, \$0.05, and \$3.05 per pound, respectively (Figure 3.5). For haddock, witch flounder, and monkfish, every pound of pollock avoided cost more, on average, than a pound of pollock sold ex-vessel. That is, the direct costs of the low pollock allocation were far outweighed by the indirect costs when pursuing these species. This finding is a product of disparity in ex-vessel prices between avoided and target species (witch flounder and monkfish regulary fetch well above \$3/lb while pollock generally sells for less than \$1/lb) as well as significantly decreased joint production under the low pollock allocation.

3.7 Discussion

Imperfect selectivity in the joint production of multispecies fisheries is a technical barrier to sucessful resource management. Incentivizing avoidance and targeting behavior by allocating tradable quota which reflects management targets, but not necessarily relative abundances or encounter probabilities, may lead to significant under-fishing and less-than-desirable economic returns given only limited compositional control of catch. In New England, a course correction after decades of mismanagement produced catch share regulations for a number of multispecies groundfish stocks, introducing explicit targeting and avoidance incentives. Testing the structure of multispecies production technology for the New England fleet, we find that nearly half of all species pairs exhibit significant weak disposability, a result which indicates species avoidance may often be costly. To examine avoidance costs directly, we calculate the difference in pollock disposal costs between constraining and lax allocations, finding pollock avoidance signficantly decreased joint production of several species, and had the low allocation remained it would have cost the fleet US $6 million.

That catch share regulations will fail to simultaneously achieve management targets and full resource utilization in multispecies fisheries is not a new argument (see e.g., Copes 1986, Squires et al. 1998). Still, adherence to legal mandates regarding the conservation and rebuilding of US marine resources, requiring direct control of fishery outputs (MSRA 2007), makes relevant the tradeoff between ecological and economic well-being. Rejecting strong disposability in landings for over 60% of catch share regulated species combinations suggests that New England will face, or continue to face, this tradeoff head-on. Indeed, the magnitude of under-fishing recently experienced in New England has led to calls for management reform and a movement away from single species models (Rothschild et al. 2013). Here, making use of an error in the initial assessment of the pollock stock and subsequent management correction, we are able to estimate forgone revenues resulting from pollock avoidance, finding this behavior often cost harvesters several times more than ex-vessel value of the avoided species. To be sure, this situation represents an extreme. Pollock was, and is, one of the healthiest and most abundant New England groundfish stocks. However, it is not unreasonable to expect avoidance might often generate excessive costs and lead to low quota utilization for healthy stocks; by year's end the fleet had caught five million pounds more than their initial pollock allocation. Interestingly, in a parallel analysis addressing identical policy questions, Scheld and Anderson (2014) attribute pollock avoidance costs largely to changes in harvest strategy. They find that southern New England vessels, which were removed from analysis here, became less productive after the pollock quota increased. That quota relaxation, which could only increase the set of possible trip strategies, would decrease production, suggests that under the low pollock allocation, these harvesters were operating on an outer and more costly production isoquant.

Costly disposal manifests here as a consequence of the limited ability to target among individual multispecies stocks. This feature of production technology is not unique to non-selective fisheries however. Much of the original work incorporating weak disposability in production technologies was motivated by pollution problems (Färe and Grosskopf 1983; Färe et al. 1986; Färe et al. 1989), while more recently, researchers have modeled agricultural systems (Zhengfei and Lansink 2003; Piot-Lepetit and Le Moing 2007) and transportation networks (Dervaux et al. 1998) using weakly disposable technologies. This work adds to a growing literature recognizing disconnect between the commonly applied, albeit restrictive, theoretical assumption of free disposal and technological specifications which incorporate more realistic production phenomena such as imperfect compositional control, null-joint production, and congestion.

Bibliography

Acheson, James M, and Roy Gardner. 2011. “Modeling Disaster: The Failure of the Management of the New England Groundfish Industry.” *North American Journal of Fisheries Management* 31 (6): 1005–18.

Ackerman, Edward Augustus. 1941. *New England’s Fishing Industry.[With Illustrations and Maps.]*. University of Chicago Press.

Aigner, Dennis, C A& Lovell, and Peter Schmidt. 1977. “Formulation and Estimation of Stochastic Frontier Production Function Models.” *Journal of Econometrics* 6 (1): 21–37.

Alverson, Dayton L. 1994. *A Global Assessment of Fisheries Bycatch and Discards*. Food & Agriculture Organization.

Andersen, Peder, Jesper Levring Andersen, and Hans Frost. 2010. “ITQs in Denmark and Resource Rent Gains.” *Marine Resource Economics* 25 (1): 11–22.

Anderson, Christopher M, and Jon G Sutinen. 2005. “A Laboratory Assessment of Tradable Fishing Allowances.” *Marine Resource Economics* 20 (1): 1–23.

Anderson, Lee G. 1994. “An Economic Analysis of Highgrading in ITQ Fisheries Regulation Programs.” *Marine Resource Economics* 9 (3): 209–26.

Apollonio, Spencer, and Jacob J Dykstra. 2008. *“ An Enormous, Immensely Complicated Intervention”: Groundfish, the New England Fishery Management Council, and the World Fisheries Crisis*. E-book Time.

Arnason, Ragnar. 1993. “The Icelandic Individual Transferable Quota System: A Descriptive Account.” *Marine Resource Economics* 8 (3): 201–18.

———. 1994. “On Catch Discarding in Fisheries.” *Marine Resource Economics* 9 (3): 189–207.

Arnholt, Alan T. 2012. *BSDA: Basic Statistics and Data Analysis*. R package version 1.01. http://http://cran.r-project.org/package=BSDA.

Asche, Frank. 2009. “Adjustment Cost and Supply Response in a Fishery: A Dynamic Revenue Function.” *Land Economics* 85 (1): 201–15.

Asche, Frank, Helge Bremnes, and Cathy R Wessells. 1999. “Product Aggregation, Market Integration, and Relationships between Prices: An Application to World Salmon Markets.” *American Journal of Agricultural Economics* 81 (3): 568–81.

Asche, Frank, Daniel V Gordon, and Rögnvaldur Hannesson. 2002. “Searching for Price Parity in the European Whitefish Market." *Applied Economics* 34 (8): 1017–24.

Asche, Frank, Daniel V Gordon, and Carsten L Jensen. 2007. "Individual Vessel Quotas and Increased Fishing Pressure on Unregulated Species." *Land Economics* 83 (1): 41–49.

Asche, Frank, Frode Steen, and Kjell G Salvanes. 1997. "Market Delineation and Demand Structure." *American Journal of Agricultural Economics* 79 (1): 139–50.

Bai, Jushan, and Pierre Perron. 1998. "Estimating and Testing Linear Models with Multiple Structural Changes." *Econometrica*, 47–78.

———. 2003. "Computation and Analysis of Multiple Structural Change Models." *Journal of Applied Econometrics* 18 (1): 1–22.

Banker, Rajiv D, Abraham Charnes, and William Wager Cooper. 1984. "Some Models for Estimating Technical and Scale Inefficiencies in Data Envelopment Analysis." *Management Science* 30 (9): 1078–92.

Barten, Anton P, and Leon J Bettendorf. 1989. "Price Formation of Fish: An Application of an Inverse Demand System." *European Economic Review* 33 (8): 1509–25.

Bates, Douglas, Martin Maechler, Ben Bolker, and Steven Walker. 2013. "lme4: Linear Mixed-Effects Models Using Eigen and S4." *R Package Version 1.0-5*. http://CRAN.R-project.org/package=lme4.

Batstone, Chris J, and Basil MH Sharp. 1999. "New Zealand's Quota Management System: The First Ten Years." *Marine Policy* 23 (2): 177–90.

Bell, Frederick W. 1968. "The Pope and the Price of Fish." *The American Economic Review*, 1346–50.

Bennear, Lori Snyder, and Cary Coglianese. 2005. "Measuring Progress: Program Evaluation of Environmental Policies." *Environment: Science and Policy for Sustainable Development* 47 (2): 22–39.

Bickel, Peter J, and David A Freedman. 1981. "Some Asymptotic Theory for the Bootstrap." *The Annals of Statistics* 9 (6): 1196–1217.

Bonzon, Kate, Karly McIlwain, C Kent Strauss, and Tonya Van Leuvan. 2010. "Catch Share Design Manual: A Guide for Managers and Fishermen." *Environmental Defense Fund*.

Bose, Shekar, and Alistair McIlgorm. 1996. "Substitutability Among Species in the Japanese Tuna Market: A Cointegration Analysis." *Marine Resource Economics* 11 (3): 146–56.

Branch, Trevor A. 2009. "How Do Individual Transferable Quotas Affect Marine Ecosystems?" *Fish and Fisheries* 10 (1): 39–57.

Branch, Trevor A, and Ray Hilborn. 2008. “Matching Catches to Quotas in a Multispecies Trawl Fishery: Targeting and Avoidance Behavior under Individual Transferable Quotas.” *Canadian Journal of Fisheries and Aquatic Sciences* 65 (7): 1435–46.

Burnham, Kenneth P, and David R Anderson. 2002. *Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach*. Springer.

Campbell, Harry Fleming, and RB Nicholl. 1994. “Can Purse Seiners Target Yellowfin Tuna?” *Land Economics* 70 (3): 345–54.

Casey, Keith E, Christopher M Dewees, Bruce R Turris, and James E Wilen. 1995. “The Effects of Individual Vessel Quotas in the British Columbia Halibut Fishery.” *Marine Resource Economics* 10 (3): 211–30.

Chambers, Robert G, Yangho Chung, and Rolf Färe. 1996. “Benefit and Distance Functions.” *Journal of Economic Theory* 70 (2): 407–19.

Charnes, Abraham, William W Cooper, and Edwardo Rhodes. 1978. “Measuring the Efficiency of Decision Making Units.” *European Journal of Operational Research* 2 (6): 429–44.

Chung, Yangho H, Rolf Färe, and Shawna Grosskopf. 1997. “Productivity and Undesirable Outputs: A Directional Distance Function Approach.” *Journal of Environmental Management* 51 (3): 229–40.

Copes, Parzival. 1986. “A Critical Review of the Individual Quota as a Device in Fisheries Management.” *Land Economics* 62 (3): 278–91.

Costello, Christopher, Steven D Gaines, and John Lynham. 2008. “Can Catch Shares Prevent Fisheries Collapse?” *Science* 321 (5896): 1678–81.

Cropper, Maureen L, and Wallace E Oates. 1992. “Environmental Economics: A Survey.” *Journal of Economic Literature* 30 (2): 675–740.

Dervaux, Benoît, Kristiaan Kerstens, and Philippe Vanden Eeckaut. 1998. “Radial and Nonradial Static Efficiency Decompositions: A Focus on Congestion Measurement.” *Transportation Research Part B: Methodological* 32 (5): 299–312.

Diewert, W Erwin. 1973. “Functional Forms for Profit and Transformation Functions.” *Journal of Economic Theory* 6 (3): 284–316.

Dunn, Daniel C, Andre M Boustany, and Patrick N Halpin. 2011. “Spatio-temporal Management of Fisheries to Reduce By-catch and Increase Fishing Selectivity.” *Fish and Fisheries* 12 (1): 110–19.

Dunn, Daniel C, Andre M Boustany, Jason J Roberts, Eric Brazer, Melissa Sanderson, Beth

Gardner, and Patrick N Halpin. 2013. “Empirical Move-on Rules to Inform Fishing Strategies: A New England Case Study.” *Fish and Fisheries*.

Dupont, Diane P, and R Quentin Grafton. 2000. “Multi-Species Individual Transferable Quotas: The Scotia-Fundy Mobile Gear Groundfishery.” *Marine Resource Economics* 15 (3): 205–20.

Efron, Bradley. 1979. “Bootstrap Methods: Another Look at the Jackknife.” *The Annals of Statistics* 7 (1): 1–26.

Efron, Bradley, and Robert J Tibshirani. 1994. *An Introduction to the Bootstrap*. Vol. 57. CRC press.

Environmental Defense Fund (EDF). 2007. “Sustaining America’s Fisheries and Fishing Communities: An Evaluation of Incentive-Based Management.”

Färe, Rolf, and Shawna Grosskopf. 1983. “Measuring Output Efficiency.” *European Journal of Operational Research* 13 (2): 173–79.

———. 2003. “Nonparametric Productivity Analysis with Undesirable Outputs: Comment.” *American Journal of Agricultural Economics* 85 (4): 1070–74.

———. 2009. “A Comment on Weak Disposability in Nonparametric Production Analysis.” *American Journal of Agricultural Economics* 91 (2): 535–38.

Fare, Rolf, Shawna Grosskopf, and CA Knox Lovell. 1994. *Production Frontiers*. Cambridge University Press.

Färe, Rolf, Shawna Grosskopf, CA Knox Lovell, and Carl Pasurka. 1989. “Multilateral Productivity Comparisons When Some Outputs Are Undesirable: A Nonparametric Approach.” *The Review of Economics and Statistics* 71 (1): 90–98.

Färe, Rolf, Shawna Grosskopf, CA Knox Lovell, and Suthathip Yaisawarng. 1993. “Derivation of Shadow Prices for Undesirable Outputs: A Distance Function Approach.” *The Review of Economics and Statistics* 75 (2): 374–80.

Färe, Rolf, Shawna Grosskopf, and Carl Pasurka. 1986. “Effects on Relative Efficiency in Electric Power Generation due to Environmental Controls.” *Resources and Energy* 8 (2): 167–84.

Färe, Rolf, Shawna Grosskopf, and Carl A Pasurka Jr. 2007. “Environmental Production Functions and Environmental Directional Distance Functions.” *Energy* 32 (7): 1055–66.

Färe, Rolf, James E Kirkley, and John B Walden. 2006. “Adjusting Technical Efficiency to Reflect Discarding: The Case of the US Georges Bank Multi-Species Otter Trawl

Fishery." *Fisheries Research* 78 (2): 257–65.

Farrell, Michael J. 1957. "The Measurement of Productive Efficiency." *Journal of the Royal Statistical Society. Series A (General)* 120 (3): 253–90.

Felthoven, Ronald G, William C Horrace, and Kurt E Schnier. 2009. "Estimating Heterogeneous Capacity and Capacity Utilization in a Multi-Species Fishery." *Journal of Productivity Analysis* 32 (3): 173–89.

Ferraro, Paul J. 2009. "Counterfactual Thinking and Impact Evaluation in Environmental Policy." *New Directions for Evaluation* 2009 (122): 75–84.

Fogarty, Michael J, and Steven A Murawski. 1998. "Large-Scale Disturbance and the Structure of Marine Systems: Fishery Impacts on Georges Bank." *Ecological Applications* 8 (S1): S6–S22.

Fried, Harold O, CA Knox Lovell, Shelton S Schmidt, and Suthathip Yaisawarng. 2002. "Accounting for Environmental Effects and Statistical Noise in Data Envelopment Analysis." *Journal of Productivity Analysis* 17 (1-2): 157–74.

Fulton, Elizabeth A, Anthony DM Smith, David C Smith, and Ingrid E van Putten. 2011. "Human Behaviour: The Key Source of Uncertainty in Fisheries Management." *Fish and Fisheries* 12 (1): 2–17.

Gaines, Richard. 2010. "Pollock Relief in the Works; Pew, Cape Fishermen Report Done Deal." *Gloucester Times*, June 16. http://www.gloucestertimes.com/fishing/x1617552895/Pollock-relief-in-the-works-Pew-Cape-fishermen-report-done-deal/print.

Gauvin, John R, John M Ward, and Edward E Burgess. 1994. "Description and Evaluation of the Wreckfish (Polyprion Americanus) Fishery under Individual Transferable Quotas." *Marine Resource Economics* 9 (2): 99–118.

Gelman, Andrew, and Jennifer Hill. 2007. *Data Analysis Using Regression and Multilevel/hierarchical Models*. Cambridge University Press.

Gordon, Daniel V, and Rögnvaldur Hannesson. 1996. "On Prices of Fresh and Frozen Cod Fish in European and US Markets." *Marine Resource Economics* 11 (4): 223–38.

Gordon, Daniel V, Kjell G Salvanes, and Frank Atkins. 1993. "A Fish Is a Fish Is a Fish? Testing for Market Linkages on the Paris Fish Market." *Marine Resource Economics* 8 (4).

Grafton, R Quentin, Ragnar Arnason, Trond Bjørndal, David Campbell, Harry F Campbell, Colin W Clark, Robin Connor, Diane P Dupont, Rögnvaldur Hannesson, and Ray

Hilborn. 2006. “Incentive-Based Approaches to Sustainable Fisheries.” *Canadian Journal of Fisheries and Aquatic Sciences* 63 (3): 699–710.

Grafton, R Quentin, Harry W Nelson, and Bruce Turris. 2004. “How to Resolve the Class II Common Property Problem? The Case of British Columbia’s Multi-Species Groundfish Trawl Fishery.” http://een.anu.edu.au/download_files/een0506.pdf.

Graham, Norman, Richard ST Ferro, William A Karp, and Philip MacMullen. 2007. “Fishing Practice, Gear Design, and the Ecosystem Approach—three Case Studies Demonstrating the Effect of Management Strategy on Gear Selectivity and Discards.” *ICES Journal of Marine Science: Journal Du Conseil* 64 (4): 744–50.

Greenstone, Michael, and Ted Gayer. 2009. “Quasi-Experimental and Experimental Approaches to Environmental Economics.” *Journal of Environmental Economics and Management* 57 (1): 21–44.

Gunatilake, HM, and Pin Sun Leung. 2003. “Technology and Management of Bottomfish Fisheries in Northwestern Hawaiian Islands.” *Marine Policy* 27 (1): 59–67.

Harrington, Jennie M, Ransom A Myers, and Andrew A Rosenberg. 2005. “Wasted Fishery Resources: Discarded By-catch in the USA.” *Fish and Fisheries* 6 (4): 350–61.

Hemmerdinger, Jonathan. 2010. “Sector Management Promises Benefits, Faces Obstacles, and Raises Concerns.” *Saving Seafood*, February 17. http://www.savingseafood.org/management-regulation/sector-management-promises-benefits-faces-obstacles-and-raises-conc-3.html.

Herrmann, Mark. 2000. “Individual Vessel Quota Price-induced Effects for Canadian Pacific Halibut: Before and After Alaska IFQs.” *Canadian Journal of Agricultural Economics/Revue Canadienne D’agroeconomie* 48 (2): 195–210.

Hilborn, Ray. 1985. “Fleet Dynamics and Individual Variation: Why Some People Catch More Fish than Others.” *Canadian Journal of Fisheries and Aquatic Sciences* 42 (1): 2–13.

Hilborn, Ray, and Max Ledbetter. 1985. “Determinants of Catching Power in the British Columbia Salmon Purse Seine Fleet.” *Canadian Journal of Fisheries and Aquatic Sciences* 42 (1): 51–56.

Hilborn, Ray, JM Lobo Orensanz, and Ana M Parma. 2005. “Institutions, Incentives and the Future of Fisheries.” *Philosophical Transactions of the Royal Society B: Biological Sciences* 360 (1453): 47–57.

Hirschman, Albert O. 1964. “The Paternity of an Index.” *The American Economic Review*, 761–62.

Holland, Daniel S. 2013. “Making Cents Out of Barter Data from the British Columbia Groundfish ITQ Market.” *Marine Resource Economics* 28 (4): 311–30.

Holland, Daniel S, and Jon G Sutinen. 2000. “Location Choice in New England Trawl Fisheries: Old Habits Die Hard.” *Land Economics* 76 (1): 133–49.

Holland, Paul W. 1986. “Statistics and Causal Inference.” *Journal of the American Statistical Association* 81 (396): 945–60.

Homans, Frances R, and James E Wilen. 2005. “Markets and Rent Dissipation in Regulated Open Access Fisheries.” *Journal of Environmental Economics and Management* 49 (2): 381–404.

Innes, Harold A. 1954. *The Cod Fisheries: The History of an International Economy*. Toronto: University of Toronto Press.

Jackman, Simon. 2009. *Bayesian Analysis for the Social Sciences*. Vol. 846. John Wiley & Sons.

Jaffry, Shabbar A, Sean Pascoe, and Catherine Robinson. 1999. “Long Run Price Flexibilities for High Valued UK Fish Species: A Cointegration Systems Approach.” *Applied Economics* 31 (4): 473–81.

Jannot, Jason E, and Daniel S Holland. 2013. “Identifying Ecological and Fishing Drivers of Bycatch in a US Groundfish Fishery.” *Ecological Applications* 23 (7): 1645–58.

Jensen, Carsten Lynge. 2002. “Applications of Dual Theory in Fisheries: A Survey.” *Marine Resource Economics* 17 (4): 309–34.

Kasperski, Stephen, and Daniel S Holland. 2013. “Income Diversification and Risk for Fishermen.” *Proceedings of the National Academy of Sciences* 110 (6): 2076–81.

Kelleher, Kieran. 2005. *Discards in the World’s Marine Fisheries: An Update*. Food & Agriculture Organization.

King, Dennis M, and Jon G Sutinen. 2010. “Rational Noncompliance and the Liquidation of Northeast Groundfish Resources.” *Marine Policy* 34 (1): 7–21.

Kirkley, James E, and Ivar E Strand. 1988. “The Technology and Management of Multi-Species Fisheries.” *Applied Economics* 20 (10): 1279–92.

Kitts, Andrew W, Evan Bing-Sawyer, John Walden, Chad K Demarest, Matthew McPherson, Peter Christman, Scott R Steinback, Patricia M Clay, and Julia Ann Olson. 2011. *2010 Final Report on the Performance of the Northeast Multispecies (Groundfish) Fishery (May 2010-April 2011)*. US Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries

Science Center.

Kjærsgaard, Jens, Niels Vestergaard, and Kristiaan Kerstens. 2009. “Ecological Benchmarking to Explore Alternative Fishing Schemes to Protect Endangered Species by Substitution: The Danish Demersal Fishery in the North Sea.” *Environmental and Resource Economics* 43 (4): 573–90.

Kneip, Alois, Léopold Simar, and Paul W Wilson. 2008. “Asymptotics and Consistent Bootstraps for DEA Estimators in Nonparametric Frontier Models.” *Econometric Theory* 24 (6): 1663–97.

Kristofersson, Dadi, and Kyrre Rickertsen. 2009. “Highgrading in Quota-Regulated Fisheries: Evidence from the Icelandic Cod Fishery.” *American Journal of Agricultural Economics* 91 (2): 335–46.

Kuosmanen, Timo. 2005. “Weak Disposability in Nonparametric Production Analysis with Undesirable Outputs.” *American Journal of Agricultural Economics* 87 (4): 1077–82.

Larkin, Sherry L, and J Walter Milon. 2000. “Tradable Effort Permits: A Case Study of the Florida Spiny Lobster Trap Certificate Program.” In *Proceedings of the 2000 International Institute of Fisheries of Economics and Trade*. Corvallis, Oregon. www.oregonstate.edu/dept/IIFET/2000/papers/larkin.pdf.

Larson, Douglas M, Brett W House, and Joseph M Terry. 1996. “Toward Efficient Bycatch Management in Multispecies Fisheries: A Nonparametric Approach.” *Marine Resource Economics* 11 (3): 181–201.

———. 1998. “Bycatch Control in Multispecies Fisheries: A Quasi-Rent Share Approach to the Bering Sea/Aleutian Islands Midwater Trawl Pollock Fishery.” *American Journal of Agricultural Economics* 80 (4): 778–92.

Lau, Lawrence J. 1972. “Profit Functions of Technologies with Multiple Inputs and Outputs.” *The Review of Economics and Statistics* 54 (3): 281–89.

———. 1976. “A Characterization of the Normalized Restricted Profit Function.” *Journal of Economic Theory* 12 (1): 131–63.

Linnemann, Hans. 1966. *An Econometric Study of International Trade Flows*. North-Holland Publishing Company Amsterdam.

Magnuson-Stevens Fishery Conservation (MSRA). 2007. “Management Reauthorization Act of 2006.” *US Public Law 109* 479.

Mammen, Enno. 1992. *When Does Bootstrap Work?* Springer.

National Marine Fisheries Service (NMFS). 2011. “National Bycatch Report”. NOAA Technical Memorandum. http://www.nmfs.noaa.gov/by_catch/National_Bycatch_Report/2011/2011_National_Bycatch_Report.pdf.

National Oceanic and Atmospheric Administration (NOAA). 2010. “NOAA Catch Share Policy.” http://www.nmfs.noaa.gov/sfa/domes_fish/catchshare/docs/noaa_cs_policy.pdf.

———. 2012. “Fisheries Economics of the United States 2011”. NOAA Technical Memorandum NMFS-F/SPO-128.

New England Fishery Management Council (NEFMC). 2009. “Amendment 16 to the Northeast Multispecies Fishery Management Plan.” http://federalregister.gov/r/0648-AW72.

Newell, Richard G, James N Sanchirico, and Suzi Kerr. 2005. “Fishing Quota Markets.” *Journal of Environmental Economics and Management* 49 (3): 437–62.

Nielsen, Max, Ola Flaaten, and Stafan Waldo. 2012. “Management of and Economic Returns from Selected Fisheries in the Nordic Countries.” *Marine Resource Economics* 27 (1): 65–88.

Pascoe, Sean, Phoebe Koundouri, and Trond Bjørndal. 2007. “Estimating Targeting Ability in Multi-Species Fisheries: A Primal Multi-Output Distance Function Approach.” *Land Economics* 83 (3): 382–97.

Piot-Lepetit, Isabelle, and Monique Le Moing. 2007. “Productivity and Environmental Regulation: The Effect of the Nitrates Directive in the French Pig Sector.” *Environmental and Resource Economics* 38 (4): 433–46.

Plummer, Martyn. 2003. “JAGS: A Program for Analysis of Bayesian Graphical Models Using Gibbs Sampling.” In *Proceedings of the 3rd International Workshop on Distributed Statistical Computing (DSC 2003). March,* 20–22.

Politis, Dimitris N., and Joseph P. Romano. 1994. “Large Sample Confidence Regions Based on Subsamples under Minimal Assumptions.” *The Annals of Statistics* 22 (4): 2031–50.

Politis, Dimitris, Joseph Romano, and Michael Wolf. 2001. “On the Asymptotic Theory of Subsampling.” *Statistica Sinica* 11 (4): 1105–24.

R Core Team. 2013. “R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.” http://www.R-project.org/.

Rothschild, Brian J, Emily F Keiley, and Yue Jiao. 2014. “Failure to Eliminate Overfishing and Attain Optimum Yield in the New England Groundfish Fishery.” *ICES Journal of Marine Science: Journal Du Conseil* 71 (2): 226–33.

Sakai, Yasuhiro. 1974. “Substitution and Expansion Effects in Production Theory: The Case of Joint Production.” *Journal of Economic Theory* 9 (3): 255–74.

Sanchirico, James N, Daniel Holland, Kathryn Quigley, and Mark Fina. 2006. “Catch-Quota Balancing in Multispecies Individual Fishing Quotas.” *Marine Policy* 30 (6): 767–85.

Scheld, Andrew M, Christopher M Anderson, and Hirotsugu Uchida. 2012. “The Economic Effects of Catch Share Management: The Rhode Island Fluke Sector Pilot Program.” *Marine Resource Economics* 27 (3): 203–28.

Scheld, Andrew M., and Christopher M. Anderson. 2014. “Costly Avoidance in a Multispecies Catch Share Fishery”. Working Paper. University of Washington.

Scheld, Andrew M., and Christopher M. Anderson,. 2014. “Joint Production in New England Groundfish: Weak Pollock Disposability”. Working Paper. University of Washington.

Seiford, Lawrence M., and Joe Zhu. 2002. “Modeling Undesirable Factors in Efficiency Evaluation.” *European Journal of Operational Research* 142 (1): 16–20.

Sethi, Suresh A, Trevor A Branch, and Reg Watson. 2010. “Global Fishery Development Patterns Are Driven by Profit but Not Trophic Level.” *Proceedings of the National Academy of Sciences* 107 (27): 12163–67.

Shephard, Ronald W. 1953. “Cost and Production Functions”. DTIC Document.

Shephard, Ronald William, David Gale, and Harold W Kuhn. 1970. *Theory of Cost and Production Functions*. Princeton University Press Princeton.

Simar, Leopold, and Paul W Wilson. 1998. “Sensitivity Analysis of Efficiency Scores: How to Bootstrap in Nonparametric Frontier Models.” *Management Science* 44 (1): 49–61.

Simar, Léopold, and Paul W Wilson. 2011. “Inference by the M out of N Bootstrap in Nonparametric Frontier Models.” *Journal of Productivity Analysis* 36 (1): 33–53.

Simar, Léopold, and Paul W. Wilson. 2000. “A General Methodology for Bootstrapping in Non-Parametric Frontier Models.” *Journal of Applied Statistics* 27 (6): 779–802.

Simar, Léopold, and PaulW. Wilson. 1999a. “Of Course We Can Bootstrap DEA Scores! But Does It Mean Anything? Logic Trumps Wishful Thinking.” *Journal of Productivity Analysis* 11 (1): 93–97.

———. 1999b. “Some Problems with the Ferrier/Hirschberg Bootstrap Idea.” *Journal of Productivity Analysis* 11 (1): 67–80.

Simpson, Edward H. 1949. “Measurement of Diversity.” *Nature* 163: 688.

Singh, Rajesh, and Quinn Weninger. 2009. "Bioeconomies of Scope and the Discard Problem in Multiple-Species Fisheries." *Journal of Environmental Economics and Management* 58 (1): 72–92.

Smith, Martin D, Junjie Zhang, and Felicia C Coleman. 2006. "Effectiveness of Marine Reserves for Large-Scale Fisheries Management." *Canadian Journal of Fisheries and Aquatic Sciences* 63 (1): 153–64.

Squires, Dale. 1987. "Public Regulation and the Structure of Production in Multiproduct Industries: An Application to the New England Otter Trawl Industry." *The Rand Journal of Economics*, 232–47.

Squires, Dale, Harry Campbell, Stephen Cunningham, Christopher Dewees, R Quentin Grafton, Samuel F Herrick Jr, James Kirkley, Sean Pascoe, Kjell Salvanes, and Bruce Shallard. 1998. "Individual Transferable Quotas in Multispecies Fisheries." *Marine Policy* 22 (2): 135–59.

Squires, Dale, and James Kirkley. 1991. "Production Quota in Multiproduct Pacific Fisheries." *Journal of Environmental Economics and Management* 21 (2): 109–26.

———. 1999. "Skipper Skill and Panel Data in Fishing Industries." *Canadian Journal of Fisheries and Aquatic Sciences* 56 (11): 2011–18.

Sumaila, U Rashid. 2010. "A Cautionary Note on Individual Transferable Quotas." *Ecology & Society* 15 (3).

Thunberg, Eric M, Edward W Bresnyan, and Charles M Adams. 1995. "Economic Analysis of Technical Interdependencies and the Value of Effort in a Multi-Species Fishery." *Marine Resource Economics* 10 (1): 59–76.

Turner, Matthew A. 1995. "Economics without Free-Disposal: Quota-Induced Discarding in Heterogenous Fisheries". University of Toronto. http://ideas.repec.org/p/tor/tecipa/mturner-95-02.html.

———. 1997. "Quota-Induced Discarding in Heterogeneous Fisheries." *Journal of Environmental Economics and Management* 33 (2): 186–95.

Verstergaard, Niels. 1996. "Discard Behavior, Highgrading and Regulation: The Case of the Greenland Shrimp Fishery." *Marine Resource Economics* 11 (4): 247–66.

Wilson, James A. 1980. "Adaptation to Uncertainty and Small Numbers Exchange: The New England Fresh Fish Market." *Bell Journal of Economics* 11 (2): 491–504.

Zhang, Ning, and Yongrok Choi. 2014. "A Note on the Evolution of Directional Distance Function and Its Development in Energy and Environmental Studies 1997–2013."

Renewable and Sustainable Energy Reviews 33 (May): 50–59. doi:10.1016/j.rser.2014.01.064.

Zhengfei, Guan, and Alfons Oude Lansink. 2003. “Input Disposability and Efficiency in Dutch Arable Farming.” *Journal of Agricultural Economics* 54 (3): 467–78.

Zhou, P., B.W. Ang, and K.L. Poh. 2008. “Measuring Environmental Performance under Different Environmental DEA Technologies.” *Energy Economics* 30 (1): 1–14.

Appendix A (Chapter 1)

A.1 Price model

Modeling inverse demand at the individual dealer level represents a significant reconceptualization of ex-vessel market structure which has largely been assumed, often implicitly, to have market clearing weekly, monthly, or annual prices and quantities. When estimating the impacts of management and policy action, researchers have traditionally formulated NE groundfish markets as systems of demand (e.g., Crutchfield 1985; Felixson et al. 1987) necessitating some degree of aggregation over time and species due to the requirement of non-missing price observations. This research is concerned with the ability of individual harvesters to alter landings' timing when provided more flexible management; a systems approach requiring aggregation of observations would fail to capture the ex-vessel price externalities governing individual revenue outcomes.[40] Consequently, the model developed measures responsiveness of an individual dealer to directly observable variables (landing weight and quality, individual inventory and expectations, relationship with harvester, and season) as well as current market conditions (region and port total landings).[41]

$$p_{\mathrm{ijklt}}=\beta_0+\beta_1 f_{\mathrm{j}}(\mathrm{q}_{\mathrm{ijklt}})+\boldsymbol{\beta}_{\mathbf{2}}\boldsymbol{f}_{\mathbf{j}}(\mathbf{Q}_{\boldsymbol{it}}^{\boldsymbol{H}})+\boldsymbol{\beta}_{\mathbf{3}}\boldsymbol{f}_{\mathbf{j}}(\mathbf{Q}_{\boldsymbol{lt}}^{\boldsymbol{P}})+\boldsymbol{\beta}_{\mathbf{4}}\boldsymbol{f}_{\mathbf{j}}(\mathbf{Q}_{\boldsymbol{t}}^{\boldsymbol{R}})+\boldsymbol{\beta}_{\mathbf{5}}\boldsymbol{f}_{\mathbf{j}}(\mathbf{I}_{\boldsymbol{kt}}^{\boldsymbol{D}})$$

$$+\boldsymbol{\beta}_{\mathbf{6}}\boldsymbol{f}_{\mathbf{j}}(\mathbf{I}_{\boldsymbol{lt}}^{\boldsymbol{P}})+\boldsymbol{\beta}_{\mathbf{7}}\boldsymbol{f}_{\mathbf{j}}(\mathbf{I}_{\boldsymbol{t}}^{\boldsymbol{R}})+\boldsymbol{\beta}_{\mathbf{8}}\boldsymbol{f}_{\mathbf{j}}(\mathbf{E}_{\boldsymbol{kt}}^{\boldsymbol{D}})+\boldsymbol{\beta}_{\mathbf{9}}\boldsymbol{\psi}_{\boldsymbol{ijklt}}+\boldsymbol{\beta}_{\mathbf{10}}\mathbf{D}_{\mathbf{t}}+\eta_{ijk}+\varepsilon_{\mathrm{ijklt}} \qquad \text{(A1)}$$

---

[40] The effects of a disaggregate approach on coefficient estimates is uncertain given that aggregation may both introduce bias (Thiel 1957) as well as limit it (Grunfeld and Griliches 1960).

[41] For a more detailed description of a similar model estimated on a subset of the data used here, see Scheld et al. (2012). In their analysis a small amount of contemporaneous error correlation was detected and argued to be the result of unobtainable omitted variables rather than simultaneously determined prices and quantities. Given the circumstances of exchange and lags in information flows characteristic of this ex-vessel market, such an argument is considered reasonable for this analysis as well.

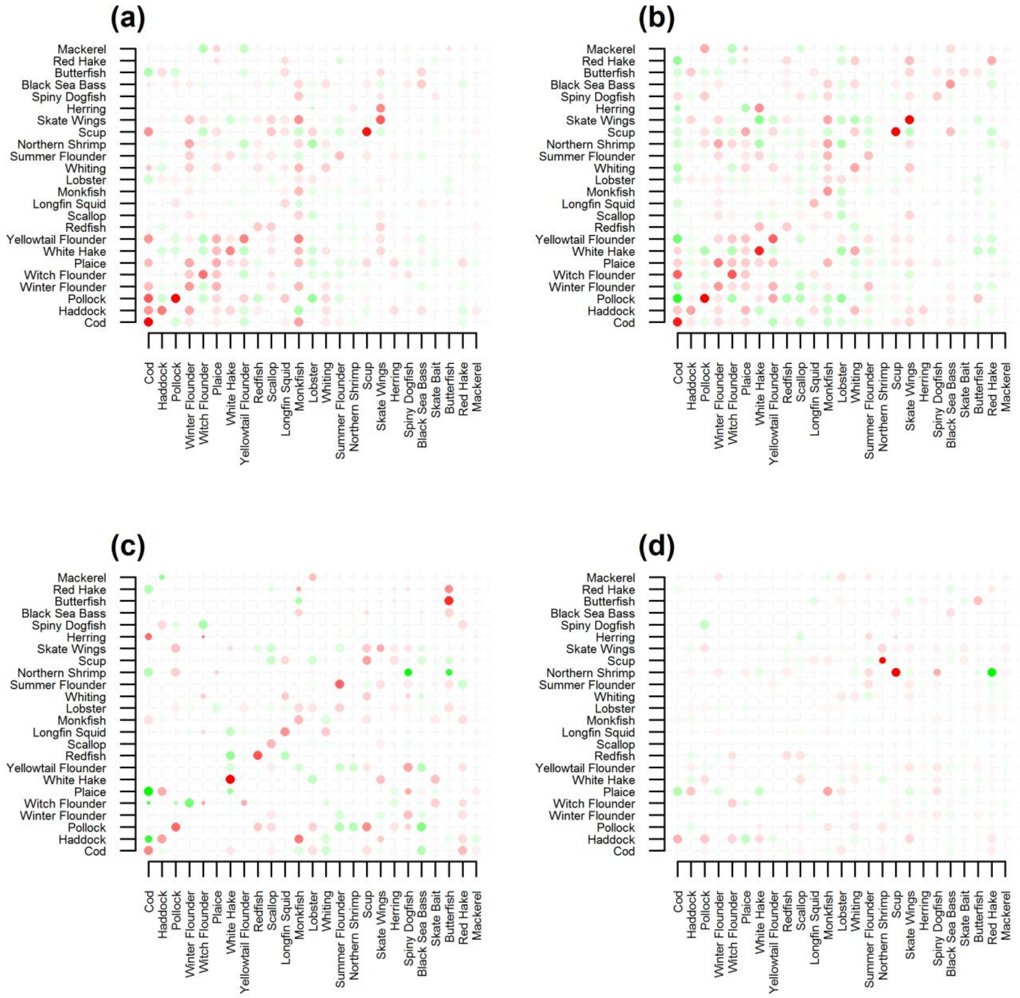
Dealer set, landing specific ex-vessel prices for the top 25 revenue important species were estimated following equation A1. Each independent fixed-effects regression contained approximately 260 independent variables with species-specific log transformations of quantities based on root mean squared error tests.[42] The price of a landing by harvester *i* of species *j* sold to dealer *k* in port *l* on day *t* was modeled as a function of own landing quantity, q_{ijklt}; a 25x1 quantity vector of individual species landed by harvester *i* on day *t*, $\mathbf{Q}_{it}^{H}$; a 25x1 quantity vector of individual species landed in port *l* on day *t*, $\mathbf{Q}_{lt}^{P}$; a 25x1 quantity vector of individual species landed in the NE region on day *t*, $\mathbf{Q}_{t}^{R}$; a 25x1 inventory vector of individual species landed at dealer *k* from four to two days prior to day *t*, $\mathbf{I}_{kt}^{D}$; a 25x1 inventory vector of individual species landed in port *l* from four to two days prior to day *t*, $\mathbf{I}_{lt}^{P}$; a 25x1 inventory vector of individual species landed in the NE region from four to two days prior to day *t*, $\mathbf{I}_{t}^{R}$; a 25x1 rational expectations vector of individual species landed at dealer *k* from one day prior to one day ahead of day t, $\mathbf{E}_{kt}^{D}$; quality indicator variables, $\boldsymbol{\psi}_{ijklt}$; time indicator variables, $\mathbf{D}_{t}$; vessel-dealer fixed effects, η_{ijk}; and a normally distributed error term, ε_{ijklt}. Overall fits were modest and variable between species with an R^2 mean and standard deviation of 0.22 and 0.19. Estimation was poorest for high-end products (e.g., cod, lobster, and scallop) whose prices may be significantly influenced by factors external to regional ex-vessel markets.

Use of a variety of quantity aggregations, devoid of market equilibrium assumptions, importantly alters the interpretation of estimated price flexibilities. At each level of quantity (i.e., individual, dealer, port, and region) point partial price flexibilities measure the average response by dealers to variation in quantity at that level, independent of variation at all other

---

[42] Logged dependent variable specifications were also tested.

levels.[43] For example, an estimated point partial price flexibility of -0.0325 for the price of haddock in response to a 1% increase in same day region-wide cod landings indicates that regardless of local market, dealer, and harvester quantity amounts, dealers on average pay slightly less for haddock when more cod is landed at the regional level.



---

[43] Due to model form marginal response varies with covariate values; estimated price flexibilities are point measures taken at the mean of all variables. Additionally, estimates are noted as partial given multiple quantity levels.

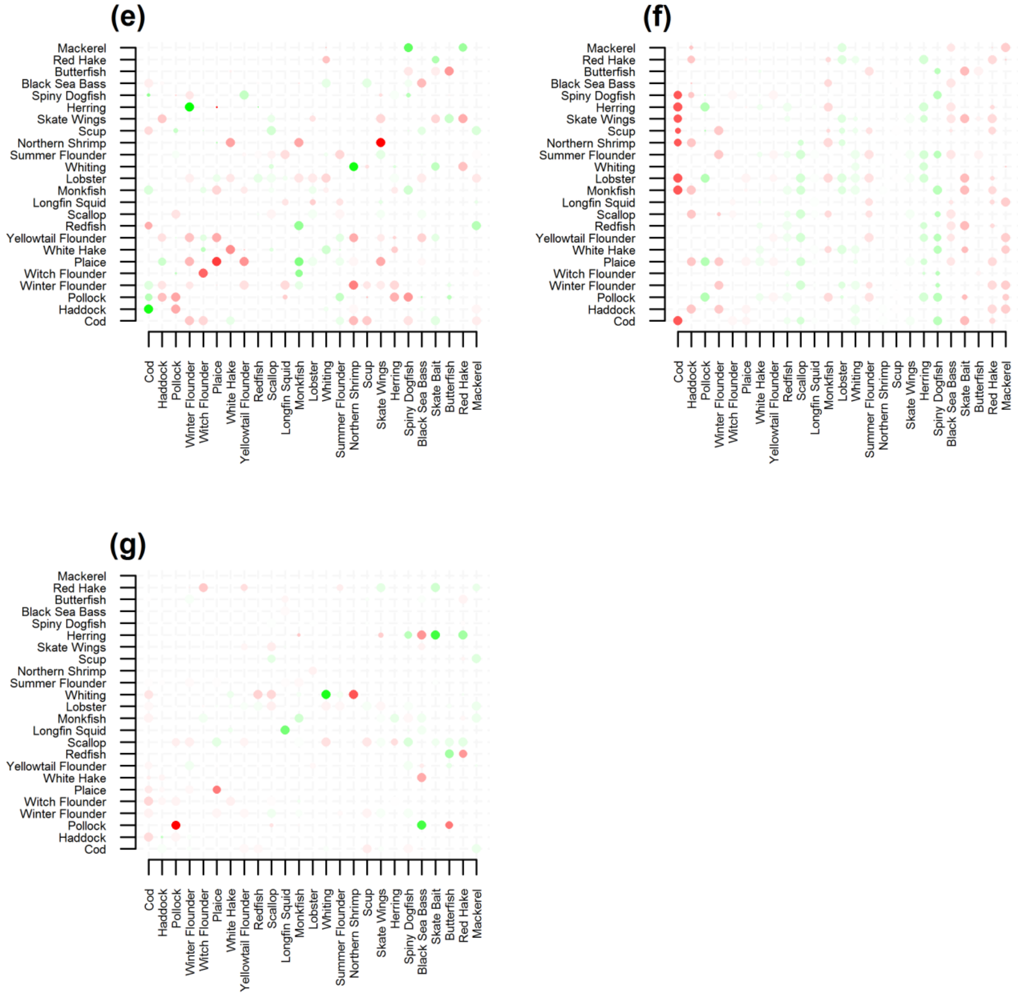


Figure A.1 Statistically significant $(P \leq 0.05)$ point partial price flexibilities for same day regional landings (a), total regional landings from four to two days prior (b), same day port landings (c), total port landings from four to two days prior (d), dealer inventory (e) and rational expectations (f), and harvester same day landings (g). Price flexibilities are calculated at the mean of all variables. In all panels, y axis corresponds to ex-vessel price and x-axis to quantity. Red (green) indicates negative (positive) price flexibility; color is scaled to panel maximum absolute flexibility value. Point size is scaled to significance level. Species are ordered left-to-right/bottom-to-top by FY 2010 revenue importance among multispecies groundfish (southwest corner) and other species.

Table A.1 Descriptive statistics for statistically significant ($P \leq 0.05$) same day (S), inventory (I), and expectation (E) point partial price flexibilities.

Level	Minimum	Maximum	Mean	Standard Dev.	% $P \leq .05$
Region S	-0.1604	0.0315	-0.0028	0.0142	64.9
Region I	-0.3294	0.1394	-0.0036	0.0326	74.4
Port S	-0.0161	0.0296	-0.0001	0.0028	39.5
Port I	-0.0922	0.0174	-0.0006	0.0061	50.6
Dealer I	-0.0312	0.0175	-0.0001	0.0031	43.1
Dealer E	-0.0505	0.0284	-0.0005	0.0052	45.8
Harvester S	-0.0995	0.0443	-0.0005	0.0071	34.5

Estimated price flexibilities were on average negative, statistically significant, and much less than one, indicating general substitutability between products as well as inelastic prices (Table A.1). Own-price flexibilities were predominantly negative, however a number of significant positive cross-price flexibilities were found (Figure A.1).[44] Additionally, results indicate that two species may be substitutes at one quantity level while complements at another. For example, while substitutes at the regional market level cod is a complement with haddock at port (0.0115) and dealer inventory (0.0143) levels. These species are highly substitutable groundfish products for end consumers, regularly treated as a composite when modeling consumer demand (e.g., Crutchfield 1985; Felixson et al. 1987). Complementarity at the port and dealer level may result from an expectation of fixed proportions in the supply chain or perceived relative scarcity. Shipping, secondary processing, and distribution entities may also place a positive value on diversity of marketed products. It is interesting to note that at the dealer rational expectations level (Figure A.1, panel f) individual species are generally either universal substitutes or complements, suggesting a somewhat remarkable similarity across dealers with respect to tradeoffs among species in first processing.

---

[44] Note that for graphical interpretative purposes skate bait price flexibilities are absent from Figure A1.

A.2 Counterfactual model

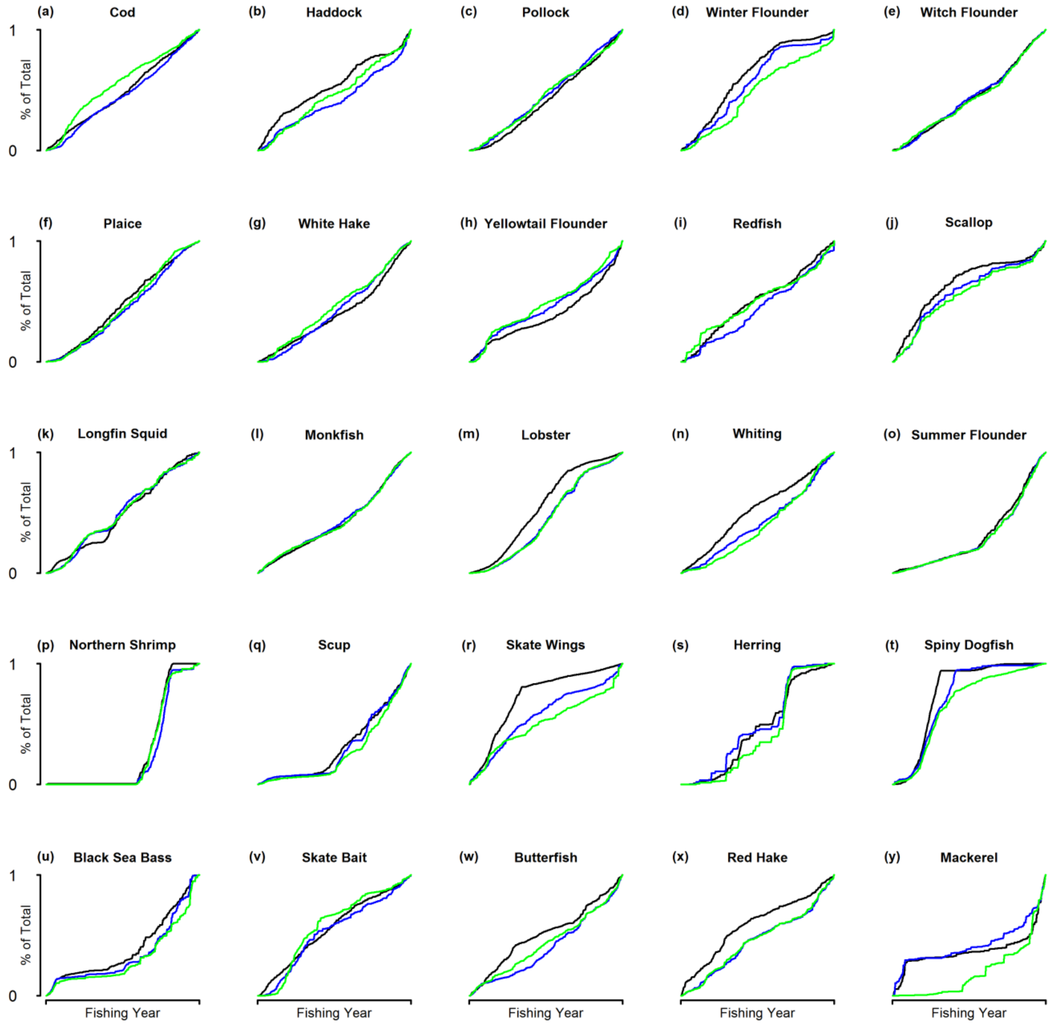


Figure A.2 Harvest pace for multispecies groundfish (a) – (i) and all other species modeled. Each plot depicts the cumulative percent of total caught using actual (black), CRCF (blue), and CPCF (green) landings. Plots are ordered by FY 2010 total fleet revenue among multispecies and other stocks.

In addition to spiny dogfish, whiting, and the aforementioned multispecies stocks, the pace of actual landings for scallop, lobster, butterfish, red hake, and skate wings was noticeably divergent from counterfactual predictions (Figure A.2). In FY 2010, fewer scallop and lobster landings by multispecies sector vessels during winter months, possibly due to bad weather avoidance, was offset by increased early and mid season exploitation. Butterfish and red hake are often caught when targeting a number of groundfish species. Increased exploitation of whiting, spiny dogfish, winter flounder, and haddock early on led to more butterfish and red hake landings. During FY 2010, skate wing trip limits were drastically reduced decreasing total multispecies sector skate wing landings by approximately 50%. Though actual early season pace did not significantly differ from prior years, a reduction in late season landings together with an overall substantial reduction in landed volume resulted in counterfactuals with low daily quantities at a fairly constant pace.

In a small number of instances the common pool counterfactual method, which weighted multispecies sector vessel landings by the pace of aggregate multispecies DAS common pool landings, produced problematic counterfactual predictions. Early season counterfactual estimates for cod landings using this method show a fast pace initially, resulting from heavy exploitation by multispecies DAS vessels before the common pool TAC was reached (i.e., a race-to-fish). This behavior is a direct result of FY 2010 multispecies management and therefore, in this instance, may confound results. Counterfactual mackerel predictions suffer from the idiosyncratic behavior of a small group of common pool harvesters who land mackerel late season. Occasionally, extremely low levels of common pool landings combined with significant inter-annual multispecies sector vessel timing change produced seasonally decreasing counterfactual catch rates. Common pool counterfactual landing rates were thus temporally smoothed by interpolating points where the rate was predicted to have decreased. Additional information about price and counterfactual models is available from the authors upon request.

A.3 References

Crutchfield, Stephen R. 1985. "An Econometric Model of the Market for New England Groundfish." *Northeast Journal of Agricultural and Resource Economics* 14: 128–43.

Felixson, Tryggvi, P Geoffrey Allen, and David A Storey. 1987. "An Econometric Model of the Market for Fresh New England Groundfish with Emphasis on the Role of Canadian Imports." *Northeast Journal of Agricultural and Resource Economics* 16 (1): 24–34.

Grunfeld, Yehuda, and Zvi Griliches. 1960. "Is Aggregation Necessarily Bad?" *The Review of Economics and Statistics* 42: 1–13.

Theil, H. 1957. "Specification Errors and the Estimation of Economic Relationships." *Revue de l'Institut International de Statistique*, 41–51.

Appendix B (Chapter 2)

B.1 Proof of proposition

First, it must be shown that the constraint in (2.2) always binds at non-zero production, implying it is always optimal to operate along the efficient frontier. From (2.3a) and (2.3b) it can be seen that if the marginal reward is greater than zero for any output i, $\mu > 0$ and therefore $Z = f(Q_1, \dots, Q_m)$. If, however, marginal rewards are less than or equal to zero for all *i* then it is optimal to produce nothing. This can be seen by contradiction between conditions (2.3a) and (2.3b), which indicate marginal reward and marginal cost are of the same sign for all positive landing amounts, and the weak disposability assumption, which requires proportional output increases increase costs, ruling out across-the-board non-positive marginal costs.

Now it will be shown that if marginal rewards for all species –*i* are non-negative, a positive marginal rate of product transformation between any two outputs *i* and *j*, fixing all other outputs at constant levels, or between an output *i* and the aggregate output mix -*i*, results if and only if the marginal reward of *i* is negative. Taking the total derivative of $f(Q_1, \dots, Q_m)$, setting it equal to zero, and rearranging we get:

$$\frac{dQ_i}{dQ_j} = -\frac{\partial f/\partial Q_j}{\partial f/\partial Q_i} - \frac{\sum_{k \neq i,j} dQ_k \times \partial f/\partial Q_k}{\partial f/\partial Q_i * dQ_j}. \qquad \text{(B.1)}$$

Equation (B.1) indicates that a positive marginal rate of product transformation between outputs *i* and *j*, fixing all other output at constant levels (i.e., $dQ_k = 0 \ \forall \ k \neq i, j$), occurs only when the marginal costs of *i* and *j* are opposite in sign. From (B.1) and (2.3a), if the marginal reward of *j* is strictly positive, a positive marginal rate of product transformation indicates the marginal reward of *i* must be negative.

Rearranging the total derivative of $f(\cdot)$ to $dQ_i = -\frac{\sum_{j\neq i} dQ_j \times \partial f/\partial Q_j}{\partial f/\partial Q_i}$, dividing both sides by $\sum_{j\neq i} dQ_j$, and inverting, we arrive at a useful representation of the marginal rate of product transformation between *i* and the aggregate output mix –*i*:

$$\frac{\sum_{j\neq i} dQ_j}{dQ_i} = -\frac{\sum_{j\neq i} dQ_j \times \partial f/\partial Q_i}{\sum_{j\neq i} dQ_j \times \partial f/\partial Q_j}. \qquad \text{(B.2)}$$

Note that the rhs denominator in (B.2) can be rewritten as:

$$\sum_{j\neq i} dQ_j \times \partial f/\partial Q_j = \left(\sum_{j\neq i} dQ_j\right)\left(\sum_{j\neq i} \partial f/\partial Q_j\right) - \sum_{j\neq i}\left(dQ_j \times \sum_{k\neq i,j} \partial f/\partial Q_k\right). \qquad \text{(B.3)}$$

Using (B.3), it will now be shown that $sign\left(\sum_{j\neq i} dQ_j\right) = sign\left(\sum_{j\neq i} dQ_j \times \partial f/\partial Q_j\right)$ whenever marginal rewards are non-negative for all outputs –*i*, indicating a positive value for (B.2) requires costly avoidance of species *i*. First, assume $\sum_{j\neq i} dQ_j \times \partial f/\partial Q_j > 0$, then from (B.3):

$$\sum_{j\neq i} dQ_j > \frac{\sum_{j\neq i}\left(dQ_j \times \sum_{k\neq i,j} \partial f/\partial Q_k\right)}{\sum_{j\neq i} \partial f/\partial Q_j}. \qquad \text{(B.4)}$$

If all marginal rewards -*i* are non-negative, then from (2.3a) all marginal costs are also non-negative and:

$$0 \leq \frac{\sum_{k\neq i,j} \partial f/\partial Q_k}{\sum_{j\neq i} \partial f/\partial Q_j} \leq 1. \qquad \text{(B.5)}$$

Given (B.5), the inequality in (B.4) will hold only if $\sum_{j\neq i} dQ_j > 0$. A symmetric argument can easily be shown to hold when $\sum_{j\neq i} dQ_j \times \partial f/\partial Q_j < 0$. If $\sum_{j\neq i} dQ_j \times \partial f/\partial Q_j = 0$ then the marginal rate of product transformation is undefined. Therefore, $sign\left(\sum_{j\neq i} dQ_j\right) = sign\left(\sum_{j\neq i} dQ_j \times \partial f/\partial Q_j\right)$ when all marginal rewards –*i* are non-negative, and if the marginal reward of *i* is also positive then (B.2) is strictly negative. This completes the proof.

■

B.2 Convergence diagnostics

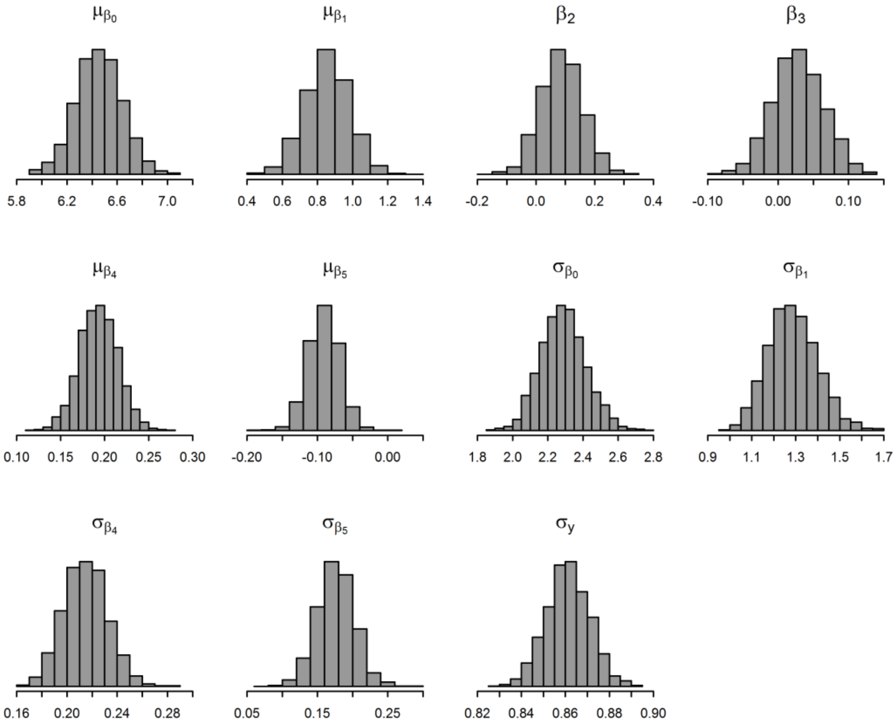


Figure B.1 Posterior histograms for important hyperparameters.

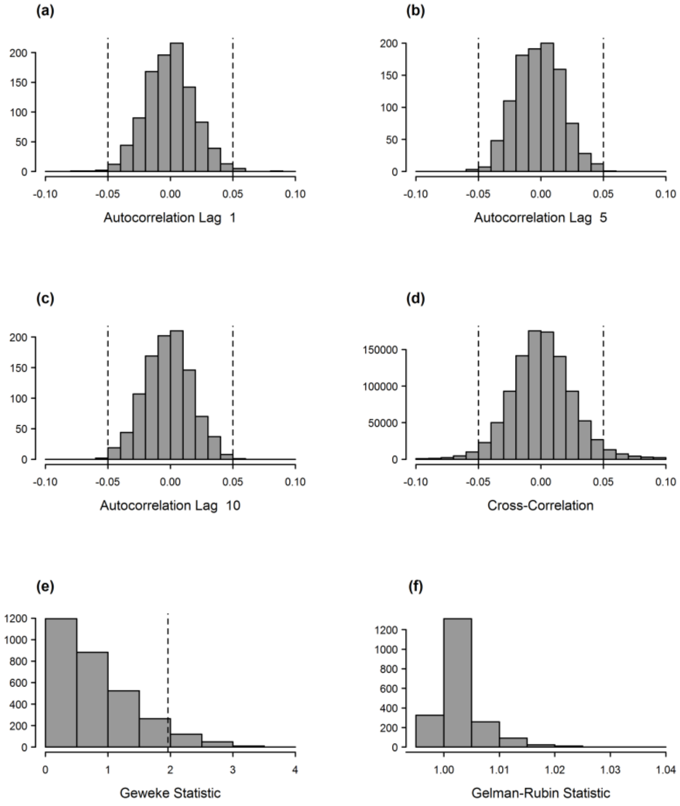


Figure B.2 Frequency histograms of aggregate convergence diagnostics for the HB model. Commonly used critical values indicated by dashed lines.

After drawing posterior parameter samples using Markov chain Monte Carlo methods, a number of diagnostic statistics were computed and evaluated to assess convergence. Given the large number of estimated parameters (>1000), individual parameter posteriors and/or diagnostics are not show. Figure B.2 shows six histograms which aggregate important diagnostic statistics across parameters. Specifically, measures which quantify parametric autocorrelation (panels a-c), cross-correlation (d), mean stability (e), and within/between chain variance (f) are depicted. Additionally diagnostics are available from the author upon request.