APPLYING PORTFOLIO MANAGEMENT TO IMPLEMENT ECOSYSTEM-BASED FISHERY MANAGEMENT

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Abstract

Portfolio management has been suggested as a tool to help implement ecosystem-based fisheries management (EBFM). The portfolio approach involves the application of financial portfolio theory to multispecies fishery management to account for species interdependencies, uncertainty, and sustainability constraints. By considering covariance among species, this approach allows economic risks and returns to be calculated across varying combinations of stock sizes. Tradeoffs between expected aggregate returns and portfolio risk can thus be assessed. We develop a procedure for constructing portfolio models to help implement EBFM in the northeastern United States, using harvest data from the National Marine Fisheries Service. Extending the work of Sanchirico *et al*. (2008), we propose a measure of excessive risk taking, which may be used by managers to monitor signals of non-optimal harvests. In addition, we conduct portfolio assessments of historical commercial fishing performance at different accounting stances: the large marine ecosystem, the New England region, and the community (fishing ports). We show that portfolio analysis could inform management at each level. Results of the study suggest that excessive risk taking is associated with overfishing, and risk management is therefore important for ensuring sustainability.

Introduction

With the evolution of increasingly more sophisticated tools for simulating the dynamic features of marine ecosystems, such as end-to-end models, among others (Link *et al*. 2011), fisheries scientists and managers now see concrete possibilities for the conservation and management of large-scale aggregate systems. This management would comprise multiple commercial stocks and other ecological components valued for their linkages to commercial targets or for their own non-market attributes. This broader approach to fisheries management has been characterized as "ecosystem-based," and while many difficulties remain in putting ecosystem-based fisheries management (EBFM) into practice, it is now viewed widely as the future of fisheries management (Link 2002; Brodziak and Link 2002; Hall and Mainprize 2004; Pikitch *et al*. 2004; Rosenberg and McLeod 2005; Leslie and McLeod 2007; Fogarty 2013; GOC 2014; Jacques 2015).

Apart from its many motivations, the use of EBFM as a decision framework necessitates consideration of the tradeoffs that arise when allocations or other policy alternatives are proposed or implemented. In particular, issues may arise when fishing quotas are set based solely on biological information for species that are valued differentially in seafood markets. At the core, human preferences for seafood can lead to targeting of species (or species groups) that differs fundamentally from those seen as appropriate from the perspective of ecological science. Further, different segments of society may disagree on desired ecological outcomes, leading to a collective inability to implement the most effective management measures (Arkema *et al*. 2006; Pitcher *et al*. 2009; Levin and Möllman 2015). Even so, for any given return from the harvest of a "portfolio" of fish from an ecosystem, society ought to choose a goal that minimizes the risks involved in realizing that return.

Financial "portfolio management" has been suggested as an archetype for implementing EBFM (Hanna 1998; Hilborn *et al*. 2001; Sanchiricho and Smith 2003; Edwards *et al*. 2004; Sanchirico *et al*. 2008). Modern portfolio theory (MPT) has been used widely in managing financial investment accounts (*e.g*., retirement accounts). In MPT, assets (*e.g*., bonds and stocks) in an investment portfolio are selected jointly to minimize the overall risk associated with a specific target for the return on investment. The construction of a portfolio should consider how each asset price might change relative to the changes in the prices of other assets in the portfolio in order to maximize the probability of actually achieving a target aggregate return (Markowitz 1952, Bordley and LiCalzi 2000). By selecting assets that have either negative or low correlation in their price fluctuations, the overall risk to the portfolio can be reduced.

The concept of financial portfolio management is useful for EBFM for several reasons. First, fish stocks are biological assets that have the potential to generate a flow of financial returns indefinitely (Edwards *et al*. 2004). Next, multispecies fishery management must account for species interdependencies, uncertainty, and sustainability constraints concurrently across all stocks under management. The portfolio approach provides a tractable manner to account for the time-varying interdependencies between harvested stocks stemming from the economic market, species biology, harvesting technology, and management regulations. The analytical framework captures a captain's choice of the fisheries in which to participate in a year, or a manager's choice of how to set preseason quotas allowing for return maximizing for the fleet, subject to uncertainty about catchability and markets during the season. Finally, through the explicit consideration of covariance among species, the portfolio approach allows economic risks to be traded off with the value of seafood supply. It should be emphasized, however, that the effective implementation of the approach ultimately requires accurate and timely assessments of targeted species biomass. To assist in real-world management decisions, the portfolio model should be coupled with models that capture the structures and dynamics of relevant ecosystems (Sanchirico *et al*. 2008).

In the context of the portfolio framework, society's objectives and constraints would be fully defined and evaluated according to normative criteria, specifying optimal policies that should be pursued to achieve desired social objectives. Within the context of EBFM, however, the full suite of social objectives can be difficult to define and model. Most applications of the portfolio approach rely on assumptions about either the form of the objective function (*e.g*., a quadratic function exhibiting the law of diminishing returns, implying that, as the revenue from fishing rises, the incremental growth in social benefits declines) or the distribution of returns (*i.e*., the fluctuation of fishing revenue follows a normal distribution). In this study, the latter is satisfied (*cf*., Meyer 1987).

Notwithstanding computational difficulties, it is the role of fishery managers (*e.g*., on the US Northeast Shelf Large Marine Ecosystem, these include the New England and Mid-Atlantic Fisheries Management Councils (NEFMC, MAFMC) and the Atlantic States Marine Fisheries Commission (ASMFC)) to identify management goals consistent with the Magnuson-Stevens Fisheries Conservation and Management Act (FCMA) [P.L. 94-265; P.L. 109-479] and the weights to be assigned to those goals. The portfolio analysis can be used to assess the additional risk induced into the system in attaining a broader suite of (often latent) objectives, by comparing the difference between the minimal possible risk level and the risk level associated with the revenue mix generated by management alternatives; and explicitly defining one facet of the trade-off made.

A small number of studies have begun to apply the concepts of portfolio theory to fishery management. Schindler *et al*. (2010) studied variance dampening across runs of heavily exploited Sockeye Salmon *Oncorhynchus nerka* in Bristol Bay, Alaska. The authors found that the decreased variability associated with multiple runs leads to an order-of-magnitude fewer required fisheries closures. Perruso *et al.* (2005) developed a static portfolio model to examine the behavior of fishermen faced with multiple targeting options in a random harvest fishery, applying the model to the pelagic longline fleet in the US Atlantic, Caribbean, and Gulf regions. The authors found that the model could be used to improve the spatial distribution of fisheries closures to reduce the mortality of juvenile Swordfish *Xiphias gladius*. Yang (2011) focused on the decisions of individual fishermen operating within the context of a transferable quota fishery in New Zealand. The author found that it could be rational for fishermen to purchase additional quota to establish a mix of yields that reduced risk. Halpern *et al*. (2011) applied a portfolio framework in the spatial dimension to examine the increased risks associated with policies that enhance equitable allocations of access to fish stocks in Southern California.

Focusing on harvest levels, Sanchirico and Smith (2003) assessed the historical pattern of fisheries exploitation in the northwest Atlantic during 1950 to 2001 to show how food web interactions influenced sustainable harvest frontiers (which depict maximum catches possible across different risk levels). Sanchirico *et al*. (2008) developed a dynamic portfolio model with biological constraints, constructing mean-variance frontiers from 1975 to 2003 using 1962-2003 data from the Chesapeake Bay. When setting species total allowable catches, the authors demonstrated gains from considering the variances and covariances of gross fishing revenues. Over the period from 1962 to 2003, they found that managers could have increased revenues from fishing and reduced revenue variances by employing EBFM frontiers in setting catch levels.

Our main objectives are to develop a procedure for constructing portfolio models to help implement EBFM in the US Northeast Region, using empirical harvest data from NMFS, and to demonstrate the feasibility and usefulness of the procedure through case studies. Our study builds on the framework explored by Sanchirico *et al*. (2008) with two extensions. First, we propose a method that managers can use to identify excessive risk taking and non-optimal harvest levels. In addition, we develop portfolio assessments of the historical performance of commercial fishing at different geographic scales (accounting stances): the Northeast Shelf LME (from Maine to North Carolina); the New England region (Maine, Massachusetts, and Rhode Island); and the community (selected fishing ports).

Analyses at different accounting stances are important. One of the national standards (National Standard 8) in federal fisheries law mandates that conservation and management measures should be adopted to minimize, to the extent practicable, adverse impacts on fishing communities. We undertake portfolio analyses for selected fishing ports to help fishery managers, municipal officials, and commercial fishermen who are concerned with managing risks at local levels. The community-level analysis identifies the sub-regional geographic distribution of risk, and comparing a community risk profile with profiles from geographically more aggregate models may show whether local risk is amplified or moderated at a broader regional scale.

We argue that the portfolio approach could contribute to improved management at each level. Results of our research may advance our understanding of the potential for portfolio management as a practical approach to help achieve EBFM in the US northeast.

Method

Extending the classical financial portfolio model of Markowitz (1952), Sanchirico *et al.* (2008) presented a dynamic portfolio framework with biological constraints. The revenues from the fisheries are stochastic due to random variability in catches and fish prices, and there is a tradeoff between the mean and the variance of total revenue. In their model, a risk averse regional manager minimizes the variance (or risk) associated with generating an expected total revenue from the harvest of n different species that is at least as large as a target revenue R. By varying the target revenue, an efficient mean-variance frontier can be mapped out. In this framework, efficiency means identifying the mix of species' harvest levels that generates the smallest possible risk of failure in achieving the target revenue due to random variability. At time t, this efficient frontier can be estimated using quadratic programming by solving the minimization problem:

$$min_{\mathbf{w_t}} \mathbf{w}_t^{\prime} \mathbf{\Sigma}_t \mathbf{w}_t \text{ , } s.t. \ \mathbf{w}_t{}' \boldsymbol{\mu}_t \ \geq \ R_t, \ \ w_{i,t} \leq \ W_{i,t} \ \forall \ i, \tag{1}$$

where i (= 1, ... , n) is the species index; $\boldsymbol{\mu}_t$ is the n×1 vector of expected revenues; and $\mathbf{\Sigma}_t$ is the n×n revenue covariance matrix at t. $\mathbf{w}_t$ is the n×1 vector of revenue weights to be calculated for time t. The revenue weights are control variables which enable a manager to choose harvest levels for individual species in the portfolio so that the overall risk is minimized. For example, $w_{i,t}$ is an element of $\mathbf{w}_t$, the revenue weight for species i in time t, which the fishery manager chooses to identify the harvest level for the species, so that the revenues from the species may be above or below its historical mean (an element in the vector $\boldsymbol{\mu}_t$). $W_{i,t}$ is the maximum weight that can be placed on any single species in time period t.

An element of the covariance matrix $\mathbf{\Sigma}_t$ is the covariance of revenues of species i and j or the variance of species i (when $j = i$) at t, calculated as a weighted average over time with a decay factor λ:

$$\Sigma_{i,j,t} = \frac{\sum_{k=1}^{t} \lambda^{t-k+1} (r_{i,k} - \mu_{i,t})(r_{j,k} - \mu_{j,t})}{\sum_{k=1}^{t} \lambda^{t-k+1}} \tag{2}$$

with $$\mu_{i,t} = \frac{\sum_{k=1}^{t} \lambda^{t-k+1} r_{i,k}}{\sum_{k=1}^{t} \lambda^{t-k+1}} \tag{3}$$

where $r_{i,k}$ is the revenue of species i at time k. $\mu_{i,t}$ is an element in the vector $\boldsymbol{\mu}_t$ in (1). Multiple drivers affecting the covariance matrix include ecological (food web trophic interactions), biological (fish stocks), and economic (market prices) effects, fishing operations and technologies (bycatch), and management (input and output controls, area management, etc.).

Biological constraints enter the problem (1) as the maximum weight for species i at t ($W_{i,t}$):

$$W_{i,t} = \frac{\gamma_{i,t} B_{i,t}}{\Omega_{i,t}} \tag{4}$$

with $$\Omega_{i,t} = \frac{\sum_{k=1}^{t} \lambda^{t-k+1} p_{i,k}\, y_{i,k}}{\sum_{k=1}^{t} \lambda^{t-k+1} p_{i,k}} \tag{5}$$

where $\gamma_{i,t}$ is the sustainability parameter for species i at t, used by the manager to control harvest levels. $\gamma_{i,t}$ can be understood as a conduit to bring information external to the model to bear on the sustainable exploitation rate. $B_{i,t}$ is the maximum sustainable catch; $\Omega_{i,t}$ is the weighted average of catches over time with decay; p is the fish price; and y is the catch quantity. Because the revenue from fishing is determined by the price and quantity of fish landed, the efficient frontier is affected by both γ and B. An increase in γ or B would lead to a rise in return, and the frontier would shift up from F to F' (Figure 1).

In our simulations, the decay factor is set at λ = 0.549 (meaning that 5% of the weight remains after 5 years), and the sustainability parameter is kept constant at γ = 1. Unfortunately, considerable uncertainties exist about the stock and flow relationships within the dynamic marine ecosystems. Actual values for B_i's in the ecosystem context are unknown. Thus, our evaluation of risk-return tradeoffs is valid only with an intertemporal comparison for a given reference value of B. Although there are different ways to specify B_i, we examine two specifications in the study. One is to set B_i constant over time as the maximum catch in the entire study period (1964-2012) for each species i. The other is to set B_{it} equal to the maximum catch up to year t for each species i, reflecting the fact that fishermen and managers are learning about the maximum catch levels over time. The second specification is used for all simulations unless noted otherwise.

The revenue weights calculated from eqs. (1) through (5) can be used by fishery managers to design harvest strategies for the next period (t + 1). Note that the framework also can be used to examine fisheries performance *ex post*:

$$g_t = \frac{\sqrt{\widetilde{\mathbf{w}}_t{}'\boldsymbol{\Sigma}_t\widetilde{\mathbf{w}}_t} - \sqrt{\widehat{\mathbf{w}}'_t\boldsymbol{\Sigma}_t\widehat{\mathbf{w}}_t}}{\widetilde{\mathbf{w}}_t{}'\boldsymbol{\mu}_t} \qquad (6)$$

where $\widetilde{\mathbf{w}}_t$ is the n×1 vector of implicit weights that the fisheries manager would have chosen to realize the actual revenues at t, $\widetilde{w}_{i,t} = r_{i,t}/\mu_{i,t}$; and $\widehat{\mathbf{w}}_t$ is the vector of optimal revenue weights estimated at the actual total revenue $R_t = \widetilde{\mathbf{w}}'\boldsymbol{\mu}_t$. The two terms in the numerator of (6) are represented as points b and a, respectively in Figure 1. The gap between the actual risk level borne by society and the optimal (minimized) risk level is the horizontal distance from a to b.

Thus, g_t is a performance indicator measuring inefficient levels of risk in the fisheries or the normalized "risk gap" at t. This measure can be expressed as the risk-gap per dollar of revenue.

The portfolio approach to marine resource management should be distinguished from a financial portfolio model. One of the key differences is that the weights do not sum to one. Essentially, an optimal harvest strategy is different from an investment strategy. Due to ecosystem constraints, fish harvests are feasible only within the available ranges of the corresponding fish stocks. Weights here represent only harvest levels, and there is no ability to "short" a fish species (*i.e.,* to bet that returns from a fish stock will decline in the future), which would most realistically necessitate a futures market for fishing quota, which does not exist in the US Northeast Region.

Note that fisheries management has not been integrated explicitly into the portfolio approach presented here. Although eqs. (2) and (3) could accommodate the effects of management changes, such effects necessarily are entwined with other biological and technological effects. A clear understanding of management changes would require a structural model incorporating management variables. Further, fishing technologies have not been incorporated explicitly into the model constraints. Although selective harvesting is feasible across some fisheries regarded as distinct, such as for American Lobster *Homarus americanus* or Atlantic Sea Scallop *Placopecten magellenicus*, fisheries for other species, such as groundfish, involve the joint production of an array of species. Both the nonseperability of the production technology and the nonmalleability of capital could constrain portfolio selections. Consequently, in terms of its practical applications, the portfolio approach is more useful in identifying significant shifts in the linked nature-human system that would require closer investigation, not in setting specific harvest strategies for individual fisheries.

Northeast Fisheries and Data

Data are from the National Marine Fisheries Service (NMFS) federal dealer purchase records for the Mid-Atlantic and New England regions. The data set comprises catches of all fish and shellfish landed in three New England states (Maine, Massachusetts, and Rhode Island) during 1964-2012, and most other states in the Northeastern United States from 1990 to 2012. For 1990-2012, landing data are available from New Hampshire, New York, New Jersey, Delaware, Maryland, and Virginia. Data for Connecticut and North Carolina are from 1996-2012. Over 300 species are assembled into 26 species groups as specified in the ATLANTIS model (Link *et al*. 2010) (see Table 1). We used live weights for the portfolio analysis, given that the biological constraints are based upon on the in-situ biomass of the species—not on processed landings. All values are in 2012 dollars.

The data were corrected for landings of Silver Hake (Whiting) *Merluccius bilinearis* and Atlantic Herring *Clupea harengus*. Other species of hake, including Offshore Hake *Merluccius albidus*, White Hake *Urophycis tenuis*, Red Hake *Urophycis chuss*, and Spotted Hake *Urophycis regia*, are not always differentiated by dealers. We use percentages from NMFS biological stock assessment surveys to decompose Silver Hake from other hakes. Atlantic herring is a high volume fishery, with landings sold by volume instead of weight. The dealer purchase records are, however, recorded in pounds, and these records are known to under-report the true landed weight. Further, some herring landings from state waters never enter the federal dealer purchase records. We therefore follow the stock assessments for herring by using data provided by the Maine Department of Marine Resources and federal Vessel Trip Report logbooks instead of data from the dealer database to address the issues of under-reporting in the time series.

For the past five decades, biomass and yields from fisheries found in the Northeast Shelf Large Marine Ecosystem (NSLME) have been affected significantly by unsustainably high levels of harvests on many species, ecological shifts, and changes to management regimes. Prior to 1977, during a period when the northeast fisheries were essentially unregulated and significant harvests were taken by foreign fleets, total revenues accruing to US fleets in Maine, Massachusetts, and Rhode Island averaged $700 million (2012 dollars, Figure 2). In 1976, the US Fishery Conservation and Management Act (FCMA) established US jurisdiction over fisheries within a 200 nmi fishery conservation zone. With the concomitant exclusion of foreign fishing, fishing revenues increased to around $1 billion (for the same three states) by the early 1980s.

In recent decades, total commercial fishing revenues in New England have fluctuated around $1.1 billion, and, when the Mid-Atlantic region is also included, NSLME revenues rose to $1.7 billion (Figure 2). New England has accounted for 65-70% of the total NSLME revenue in recent years. Maine, Massachusetts, and Rhode Island accounted for over 90% of the total revenue from New England. The fishery portfolio of the three New England states became increasingly more concentrated over the 49 years (Figures 3(a) and (b)). In 1964, the shares for Atlantic Sea Scallop, American Lobster, and the three major groundfish species (Atlantic Cod *Gadus morhua*, Haddock *Melanogrammus aeglefinus*, and Yellowtail Flounder *Pleuronectes ferruginea*) accounted for 13, 25, and 33%, respectively. In contrast, shares for the same species were 31, 34, and 3% in 2012. That year, a commercial fishery "disaster" was declared for the northeast groundfish fishery.

Key historical events help to explain the shifts in revenue trends and shares (Figures 2 and 3). With a post-FCMA fishing fleet expansion, management became more challenging. From

1977 to 1982, the groundfish fishery was managed under output quotas for the three most important species: Atlantic Cod, Haddock, and Yellowtail Flounder. Under quota management, investment and fishing decisions were distorted by incentives to take quotas as quickly as possible. Dissatisfaction with quota management led to its abandonment in 1982 in favor of indirect effort controls such as minimum fish sizes and fishing gear restrictions (Jin *et al*. 2002). The Atlantic Sea Scallop fishery grew unregulated until 1982, when the New England Fishery Management Council (NEFMC) implemented a minimum meat size standard with the Atlantic Sea Scallop FMP (Table 2). The number of full-time Atlantic Sea Scallop vessels increased eight-fold between 1977 and 1993, including construction of 152 new vessels during 1977-82 alone. Aggregate fishing effort increased 500%. To survive financially, the Atlantic Sea Scallop sector depended upon the harvest of small recruits (Edwards 2001).

In the late 1980s and early 1990s, sharp declines in the catches of the traditionally valuable groundfish species led to the introduction of the Northeast Multispecies Fishery Management Plan (FMP) in 1986. In 1994, FMP Amendment 5 led to significantly more stringent effort control measures, comprising a moratorium on new entrants and a days-at-sea program, in conjunction with increased mesh size requirements and the expansion of closed areas. In 1995, a fishery resource disaster was declared for the northeast groundfish fishery. The Multispecies Sector Program was introduced in 2010, establishing transferable output controls on the groundfish fishery (Table 2).

Amendment 4 to the Atlantic Sea Scallop FMP created a limited-access permit system in 1994 in the Georges Bank and Mid-Atlantic fisheries. The limited-access vessels were restricted by a seven-man crew limit and allocated nontransferable days-at-sea effort quotas depending on full-time, part-time, or occasional permit categories. The Atlantic Sea Scallop fishery struggled with low landings for several years after Amendment 4 was implemented. Successive years of historically heavy sets of Atlantic Sea Scallop cohorts were protected by effort controls and by three large areas of the continental shelf that were closed in December 1994 in an attempt to rebuild important stocks of groundfish. The growth of these cohorts allowed the Atlantic Sea Scallop biomass to increase significantly during 2001 and 2005, thereby supporting unprecedented landings (Edwards 2005).

American Lobster landings have risen continuously since the early years of the series, increasing noticeably at the time of the establishment of the US fishery conservation zone in 1977. At first, this increase comprised mostly otter trawl landings from deeper waters, but these yields were replaced by trap landings as deepwater trap technologies were refined. Increases in the last decade of the series were spurred by growing seafood demand and a concomitant expansion of nearshore effort. During this last decade, annual landings and revenues were increasingly variable but averaged 100 million pounds and $420 million respectively, making this fishery commercially the most lucrative in the US Northeast Region. Most of the landings occurred in the Gulf of Maine, which constitutes one of three distinct ecological stock areas. Stocks in the Inshore Southern New England area experienced recent severe declines that were likely tied to increased water temperatures and disease. The Gulf of Maine stock has not been biologically overfished, and, even with high levels of fishing effort, technically overfishing has not occurred there. Management has relied heavily upon industry self-governance, focusing on restrictions on carapace size and gear and bans on the taking of gravid females. Only marginal changes in regulation have occurred over time and across the seven conservation management areas established by the ASMFC.

Results

Due to data limitations, we focus on the long-term evolution of fishery portfolios using data from only three New England states (Maine, Massachusetts, and Rhode Island). The efficient frontier is affected by biological constraints, namely the historical maximum sustainable catch for individual species (B_i). Annual efficient frontiers and actual risk-returns under the two different stock reference levels in 1965-2012 are illustrated in Figures 4(a) and (b). Because B_i is larger for all species under the first specification, efficient frontiers in panel (a) are above those in panel (b), especially during the first two decades. In panel (a), the actual returns in the 1960s were significantly below the efficient levels, for the same risk levels, due to the presence of foreign fleets. In contrast, the second specification captures only the stocks accessible to the US fleet, and the actual returns in those years were very close to or above the frontiers in panel (b). Note, however, that model estimates for the 1960s are based on limited historical data. For $\lambda = 0.549$, the model's burn-in period is 1964-1968 (5 years), see equations (2) and (3). An actual return occurring above the frontier is a violation of portfolio theory, but recall that the marine resource portfolio differs from a financial portfolio. The "violation" is a result of the specification of the stock constraints B (Figure 1). Note that the actual risk and return are unaffected by B, but the frontier is affected by it.

The optimal revenue weights ($w_{i,t}$) for the 26 species groups underlying the efficient frontiers in Figure 4(b) are depicted in Figure 5(a) as shares of the corresponding maximum revenue weights ($W_{i,t}$). As indicated in Figure 6(a) (subplots for Groups 3 and 4 in Figure 5(a)), the optimal strategies for 2005 and 2012 called for significantly lower harvests than the biological constraints (the shares were significantly below one) for both Atlantic Sea Scallop and American Lobster. For comparison, panel (b) of Figure 5 shows the ratio of implicit revenue weights (representing actual harvest) to the maximum weight ($\widetilde{w}_{i,t}/W_{i,t}$). The implicit weights were above one for Atlantic Sea Scallop and close to one for American Lobster, indicating that both species were harvested at or above the maximum levels (Figure 6(b)). The model results suggest that overreliance on Atlantic Sea Scallop and American Lobster contributed to elevated risk-taking in 2005 and 2012.

The level of inefficiencies (*i.e*., excessive risk taking) in the commercial fishing industry in the three New England states, measured by the risk gap g, from 1964 to 2012 is plotted in Figure 7(a). Four relatively large risk-gap "excursions" occurred during 1978-1981, 1991, 2005, and 2012, with one smaller excursion in 2000. Overall, excursions from risk-minimizing portfolios seem short-lived, as fleets appear to adjust to new constraints within at most a few years. This feature is robust with respect to the decay factor (λ). The inefficiencies are likely the results of (i) non-optimal harvests (as reflected by non-optimal revenue weights) of species with large revenue shares (Figure 4), resulting in elevated risks (the numerator in equation (6)), or (ii) reductions in total revenues (the denominator in equation (6)) due to the fact that the biological constraints have not been explicitly incorporated.

The ratio of implicit revenue weight to the maximum weight was greater than one in some years for some species (*e.g*., Atlantic Sea Scallop in 2005), implying that the biological constraints were violated ($\widetilde{w}_{i,t} > W_{i,t}$) (Figure 5(b), and in more detail in Figure 6(b)). The overall level of overfishing can be estimated by the difference between the actual total revenue and adjusted total revenue. The adjusted total revenue at t is calculated as the sum of $\widetilde{w}_{i,t}\mu_{i,t}$ over all species with the adjustment $\widetilde{w}_{i,t} = W_{i,t}$ if $\widetilde{w}_{i,t} > W_{i,t}$. Thus, the adjusted total revenue is within the biological constraints. Results of the calculation suggest that significant overfishing occurred in 1978, 1980, 1981, 1991, 1992, and 2005 (Figure 7(b)). Excessive harvests led to sharp increases in both total revenues (Figure 2) and risk levels (Figure 4(b)) in those years. Note that the actual risk at *t* is calculated in the same covariance matrix (eq. (2)) and the *implicit* weights, reflecting the actual harvest revenue, in eq. (6). Figures 7(a) and (b) depict the coincidence of the elevated risk gap with overfishing.

The risk gap shown in Figure 7(a) is based on an assumption that the fish stock constraint B_{it} is equal to the maximum catch up to year t. To examine the robustness of the results, we conducted a sensitivity analysis of the risk gap with respect to the fish stock constraints (B). The results suggest that elevated risk gaps occur in the same years, and the fish stock constraints affect only the magnitude of the gaps (Figure 7(c)). The actual risk-return ratio, calculated using the same data that are depicted in Figure 4(b), is also correlated with the risk gap.

Results of analyses at community levels demonstrate the usefulness of the portfolio framework for alternative geographic scopes. New Bedford, Massachusetts is an important fishing port in the northeast with total fishing revenues reaching $409 million in 2012. The primary species landed in New Bedford include Atlantic Sea Scallop, Yellowtail Flounder, Winter Flounder *Pseudopleuronectes americanus*, and Atlantic Cod, with average annual revenue shares over the entire study period (1964-2012) of 51.1, 14.2, 7.3 and 6.9%, respectively. The revenue share of Atlantic Sea Scallop rose to more than 80% in 2011-2012. As the share continued to grow from 1998 to 2012 (Figure 8), the optimal weights (shown in red) for Atlantic Sea Scallop were significantly lower than the implicit weights (blue) between 2000 and 2012, apparently calling for the diversification of harvests into other stocks, such as groundfish. The potential for diversification was constrained, however, by the depletion of Atlantic Cod and other groundfish stocks.

Another important fishing port is Gloucester, Massachusetts, where primary landings have been groundfish. Over the 5 decades, the annual average revenue shares for Atlantic Cod, Haddock, Atlantic Herring, American Lobster, Atlantic Pollock *Pollachius pollachius*, and Silver Hake were 19.6, 13.4, 7.6, 6.7, 6.3 and 6.2%. With the depletion of groundfish, total annual revenues declined from over $120 million in the early 1980s to below $60 million in recent years. As shown in Figure 9, the optimal weights for Atlantic Cod (shown in red) switched to below the implicit weights (actual harvests) in 1987, preceding Amendment 5 in 1994. In the late 1980s and 1990s, the optimal revenue weights stayed low relative to the implicit weights, implying that Atlantic Cod landings should be reduced. Declining stocks led eventually to the adoption of Amendment 13 and limits on days at sea in 2004. Although the portfolio model called for increased yields of Atlantic Cod in 2003 (see Figure 9), the Atlantic Cod stock was depleted and unavailable for harvest. As noted in the introduction and method sections, the biological constraints in the current portfolio model are based on historical catches and do not reflect the actual stock available for harvest in each period. If the portfolio model could be coupled with an ecosystem model, then stock availabilities could be updated in each period.

A closer look at the risk gaps for the fishing ports (Figure 10) reveals that the regional risk-gap excursions (Figure 7(a)), the three New England states) can be explained by inefficiencies at the community level. The risk-gap excursions in 1978-1981 were the result of inefficiencies in Gloucester and New Bedford. The elevated gap in 2005 was driven by the large risk gaps in New Bedford. Note also that the magnitudes of the risk gaps at the community level are greater than those at the regional level due to a compensating effect across ports at the higher level of aggregation. For example, in 2005, the risk gap was close to 1.5 in New Bedford (Figure 10) but only 0.123 for the three New England states as a whole (Figure 7(a)). In the same year, the risk gap was 0.115 at the LME level (Figure 11). A similar attenuation can be seen in the spike in the risk gap for Gloucester in 2000 (0.67 in Figure 10), which registers in the three state model at a much lower level of inefficiency (0.04 in Figure 7(a)). Overall, these inefficiencies were short lived, and the industry adjusted quickly for ecological and subsequent regulatory changes.

Discussion

As noted above, significant increases in the risk gap coincided with revenue growth and overfishing (Figures 2, 7(a) and (b)). In 1979, in the three New England states, Atlantic Sea Scallop revenues grew 49% over the previous year. Significant increases in Atlantic Cod landings (8-55% per year) and Haddock landings (7-19% per year) occurred between 1977 and 1981. Stock declines in subsequent years led to major management actions in 1982.

The revenues from groundfish landings declined during most of the 1980s. This trend reversed itself in 1990. In the three New England states, revenues from Yellowtail Flounder, Haddock, and Atlantic Cod rose respectively by 115, 31, and 25% over previous years. Greater than optimal harvests resulted in further declines in the groundfish stocks, leading to more stringent control measures in 1994 (Amendment 5 and the establishment of emergency area closures).

The next overfishing event occurred in 2005, when annual growth in Atlantic Sea Scallop revenues was 55% in the three New England states. In 2006, excessive Atlantic Sea Scallop harvests led to a depletion of the stocks and the imposition of control measures, including area closures.

Our analysis reveals the following dynamic cycle: without effective effort control, rent-seeking behavior leads to excessive harvests of certain valuable species and short-term increases in revenues. Overfishing causes stock depletion and the subsequent adoption of regulation. Because the optimal portfolio is dynamic and based on historical data, which incorporates changing ecological, economic, and regulatory factors, the model appears capable of detecting excessive harvests of certain species as excursions from the risk minimizing portfolio, *i.e.*, excessive risk taking. Thus, the portfolio framework foreshadows imminent stock depletion, providing a motivation for implementing management measures to levelize returns and to reduce excessive risk (Figure 12).

Our analysis highlights a need for improvements in understanding ecological structures and processes (Sanchirico *et al*. 2008). We do not know the true maximum sustainable catch for individual species (B_i), necessitating a reliance on historical harvest levels as a proxy. Consequently, it is unclear whether the overharvesting identified in the model is unsustainable exploitation or merely sustainable harvesting outside the bounds of the historical time series. Indeed, without knowing the true biological constraints, any unprecedented catch level above historical landings could be viewed as risky, even if the biomass would easily permit such catches.

An example is the development of a new fishery, such as that for Monkfish *Lophius americanus* in the 1990s, where, even if it was being exploited sustainably, the new fishery by definition would be harvesting above historical landing levels in the first few years. On the other hand, without accurate and timely assessments of targeted species biomass, catch levels below historical series also could be misinterpreted as non-optimal. In 2012, the fleets exhibited apparently risky concentration in harvests of Atlantic Sea Scallops and American Lobster. This specialization was not the result of an economic decision to target shellfish but rather the result of a lack of groundfish to catch, precluding risk reduction through diversification into the latter.

Nevertheless, this study highlights portfolio theory's robust ability to identify imbalances in management strategies and to quantify objectively the historical extent of these imbalances. At its center, EBFM is concerned with managing the trade-offs within an aggregate fishery, and the portfolio approach equips managers with a tool for assessing those trade-offs strategically. The portfolio approach is an important addition to the suite of management tools now employed, which mostly ignore biological, technological, and market interactions.

A range of biological multispecies models now are under development for the Northeast Shelf LME (Gamble and Link 2009, Link *et al.* 2009, Link *et al*. 2010, Gaichas *et al.* 2012, Curti *et al.* 2013, Fogarty 2013). Future research should focus on linking these multispecies models to portfolio models to characterize the biological constraints necessary for more realistic management evaluations. When coupled with multispecies Monte Carlo projections, the portfolio approach ultimately could allow the risks and returns of alternative management options to be assessed *ex ante*, thereby generating a better understanding of how risks are distributed across a range of geographic scales.

The portfolio approach is a risk management tool that allows the explicit analysis of tradeoffs among risks and returns. We argue that fish stocks are biological assets that are comparable to financial assets, where the revenues from fish harvests comprise stochastic returns to the assets. Our model presents a risk-return analysis from an ecosystem perspective by including all major species (or species groups) in the US Northeast Region (*e.g.*, groundfish, Atlantic Sea Scallop, and American Lobster). The financially optimal management of fishery resources attempts to minimize the variance (the risk) associated with a target level of returns from fishing.

There are several advantages to the portfolio approach as a tool for implementing EBFM. Excessive risk taking often is associated with overfishing, and risk management is therefore important for moving toward sustainability. Typically, a substantial increase in the risk gap is a signal that an unsustainable level of harvesting may be occurring. A closer investigation is needed to ascertain the cause of the increase in the risk gap, because, absent timely assessments of relevant biological constraints, catch levels below historical series also could lead to a rise in the risk gap. The portfolio approach identifies those species (or species groups) that are being overfished through a comparison of optimal harvests with actual harvests (optimal revenue weights with implicit weights). The set of optimal weights provides useful information that could help fishery managers implement EBFM at a range of geographic levels. A critical insight is that EBFM could be enhanced with a better understanding of the underlying ecological structures and processes that could limit adjustments to minimize risks.

A key concept in the management of risk is the diversification across assets in a portfolio to take full advantage of negative correlations in returns. With respect to the portfolio comprising the commercial fisheries of the US Northeast Region, we show that excessive risk typically is associated with lack of variety in the mix of species landed. Regions or communities may exhibit high levels of risk, resulting from constraints on the abilities of the relevant fishermen to diversify their catches, due to depleted stocks, regulations meant to conserve depleted stocks, or both. In such cases, the difficulties faced by fishermen of switching among available target stocks, because of nonselective technologies or human capital constraints, may exacerbate risk taking. Importantly, increased riskiness at local levels may be moderated at broader geographic levels, and, in the US Northeast Region, riskiness tends to be fleeting. The analysis of the scale and duration of risk gaps could help characterize the capacity for diversification to mitigate risk and thereby help improve sustainability in fisheries management.

Acknowledgements

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Table 1. Species Groups

Group No.	Group Code	Species
1	BFF1	Hard Clam *Mercenaria mercenaria*, Soft Shell Clam *Mya arenaria*, Ocean Quahog *Arctica islandica*, Unclassified Clam Species
2	BFF3	Blue Mussel *Mytilus edulis*, Eastern Oyster *Crassostrea virginica*, Bay Scallop *Argopecten irradians*
3	BFS	Atlantic Sea Scallop *Placopecten magellanicus*
4	BML	American Lobster *Homarus americanus*
5	BMS	Blue Crab *Callinectes sapidus*, Lady Crab *Ovalipes ocellatus*, Green Crab *Carcinus maenas*, Red Crab *Chacean quinquedens*, Jonah Crab *Cancer borealis*, Rock Crab *Cancer irroratus*, Cancer Crab *Cancer pagurus*, Spider Crab *Libinia emarginata*, Snow Crab *Chionoecetes opilio*, Horseshoe Crab *Limulus polyphemus*, Knobbed Whelk *Busycon carica*
6	FBP	Bay Anchovy *Anchoa mitchilli*, Butterfish *Peprilus triacanthus*, Atlantic Chub Mackerel *Scomber japonicus*, Atlantic Silverside *Menidia menidia*, Spanish Mackerel *Scomberomorus maculatus*
7	FDB	Silver Hake *Merluccius bilinearis*
8	FDC	Atlantic Croaker *Micropogonias undulatus*, Cusk *Brosme brosme*, Black Drum *Pogonias cromis*, Red Drum *Sciaenops ocellatus*, American Eel *Anguilla rostrata*, Grenadiers *Macrouridae spp*., Offshore Hake *Merluccius albidus*, Red Hake *Urophycis chuss*, John Dory *Zeus faber*, Opah *Lampris guttatus*, Ocean Pout *Zoarces americanus*, Scup *Stenotomus chrysops*, Black Sea Bass *Centropristis striata*, Weakfish *Cynoscion regalis*, Spotted Sea Trout *Cynoscion nebulosus*, Spot *Leiostomus xanthurus*, Striped Bass *Morone saxatilis*, Atlantic Sturgeon *Acipenser oxyrhynchus oxyrhynchus*, Tautog *Tautoga onitis*, Blueline Tilefish *Caulolatilus microps*, Sand Tilefish *Malacanthus plumieri*, Golden Tilefish *Lopholatilus chamaeleonticeps*, Unclassified Tilefish, White Perch *Morone americana*, Offshore Hake Unclassified *Merluccius spp.* or *Urophycis spp.*
9	FDC2	Atlantic Pollock *Pollachius pollachius*
10	FDC7	Acadian Redfish *Sebastes fasciatus*
11	FDD	Monkfish *Lophius americanus*
12	FDE1	Atlantic Menhaden *Brevoortia tyrannus**
13	FDF	Yellowtail Flounder *Pleuronectes ferruginea*
14	FDO	Haddock *Melanogrammus aeglefinus*
15	FDS	Atlantic Cod *Gadus morhua*
16	FPL	Atlantic Mackerel *Scomber scombrus*
17	FPS	Atlantic Herring *Clupea harengus*
18	FVB2	Summer Flounder *Paralichthys dentatus*
19	FVB3	Winter Flounder *Pseudopleuronectes americanus*
20	FVB4	Witch Flounder *Glyptocephalus cynoglossus*
21	FVB6	American Plaice *Hippoglossoides platessoides*
22	FVD	Atlantic White Hake *Urophycis tenuis*
23	FVT3	White Marlin *Kajikia albidus*, Atlantic Blue Marlin *Makaira nigricans*, Swordfish *Xiphias gladius*
24	PWN2	Brown Shrimp *Crangon crangon*, Atlantic Shrimp *Litopenaeus setiferus*, Gulf of Maine Shrimp *Pandalus borealis*, Crangon *Crangon Septemspinosa*, Unclassified Shrimp
25	SSK1	Rosette Skate *Leucoraja garmani*, Little Skate *Leucoraja erinacea*, Winter Skate *Leucoraja ocellata*, Barndoor Skate *Dipturus laevis*, Smooth Skate *Malacoraja senta*, Thorny Skate *Amblyraja radiata*, Clearnose Skate *Raja eglanteria*
26		Others**

* Database includes partial data for menhaden.

** Including Groups BFF2 (Atlantic Surf Clam *Spisula solidissima*), CEP (Longfin Squid *Loligo pealei*, Northern Shortfin Squid *Illex illecebrosus*), FVT1 (Bluefin Tuna *Thunnus thynnus*), FVT2 (Other Tuna Species), SHB (Spiny Dogfish *Squalus acanthias*), and other species/groups. These groups cannot be examined separately due to incomplete data series or relatively low economic values.

Table 2. Management Timeline.

Year	Groundfish	Atlantic Sea Scallop
1977	TACs established for Atlantic Cod, Haddock, and Yellowtail Flounder	
1982	Interim Groundfish Plan replaced TACs with input controls	Created long-term management program for Atlantic Sea Scallop fishery (FMP)
1986	Northeast Multispecies FMP approved	FMP Amend 1, established minimum size meat count standard
1994	FMP Amend 5 established mesh size requirements, expanded closed areas and imposed moratorium on new entrants	FMP Amend 4, implemented an effort control system
1995	Northeast groundfish fishery declared a fishery resource disaster	Framework Adjustments 4, 5 and 6, reduction in crew size limit, gear restrictions, and vessel tracking system (VTS)
1996	FMP Amend 7 accelerated DAS reduction program	Framework Adjustment 7, permanent reduction in crew size
1997		Amend 6, implemented a gear conflict management program
1998	Northeast groundfish vessel buyback program expanded	Framework Adjustment 10, area closure
2001	Northeast groundfish permit buyback program implemented	Extension of closed areas
2004	FMP Amend 13 established a DAS transfer program and created a process for establishing sectors	Amend 10, introduced an area rotation management program
2005		Framework Adjustment 17, vessel monitoring system (VMS) requirements
2007	Framework 42 of FMP Amend 13 reduced DAS and implemented differential DAS	Amend 13, industry-funded observer program

	counting areas	
2010	FMP Amend 16 established Northeast Multispecies Sector Program and ACLs	
2011		Amend 15, implemented annual catch limits (ACLs) and accountability measures (AMs)
2012	Fishing year 2013 of Northeast groundfish declared a commercial fishery failure	Amend 17, enforcement of collection-of-information requirements

Figure Captions and Notes

Figure 1. Efficient Frontier and the Risk Gap. *R* represents a given level of total revenue; *F* and *F'* are two efficient frontiers; *b* denotes the actual portfolio; *a* and *a'* denote the optimal portfolios on *F* and *F'*.

Figure 2. Total Revenue of Fish Landings in the Northeast Region, 1964-2012.

Figure 3 (a). Revenue Shares by Species, ME, MA and RI, 1964-2012.

Figure 3 (b). Herfindahl–Hirschman Index, 1964-2012.

Figure 4 (a). Efficient Frontiers and Actual Portfolios, ME, MA and RI, 1965-2012. *B* = maximum landings in the entire study period (1964-2012). Vertical axis depicts the expected return ($100m, 2012); Horizontal axis depicts the risk level (s.d. of revenue); Green circle denotes the actual portfolio in that year.

Figure 4 (b). Efficient Frontiers and Actual Portfolios, ME, MA and RI, 1965-2012. B_t = maximum landings up to year *t*. Vertical axis depicts the expected return ($100m, 2012); Horizontal axis depicts the risk level (s.d. of revenue); Green circle denotes the actual portfolio in that year.

Figure 5(a). Optimal Revenue Weight Shares by Species Groups, ME, MA and RI, 1965-2012**.** The revenue weight share = weight ($w_{i,t}$) /maximum weight ($W_{i,t}$) for each species group *i*, calculated at expected return = half of the max return. Species groups: 1-BFF1/Clams, 2-BFF3/Blue Mussel, etc., 3-BFS/ Atlantic Sea Scallop, 4-BML/American Lobster, 5-BMS/crabs, 6-FBP/Bay Anchovy, etc., 7-FDB/Silver Hake, 8-FDC/Atlantic Croaker, etc., 9-FDC2/Pollock, 10-FDC7/Redfish, 11-FDD/Goosefish, 12-FDE1/Menhaden, 13-FDF/Yellowtail Flounder, 14-FDO/Haddock, 15-FDS/Atlantic Cod, 16-FPL/Atlantic Mackerel, 17-FPS/Atlantic Herring, 18-FVB2/Summer Flounder, 19-FVB3/Winter Flounder, 20-FVB4/Witch Flounder, 21-FVB6/American Plaice, 22-FVD/White Hake, 23-FVT3/Marlin, etc., 24-PWN2/Shrimp, 25-SSK1/Skates, 26-others (see Table 1).

Figure 5(b). Implicit Revenue Weights by Species Groups, ME, MA and RI, 1965-2012**.** Species group definition is the same as in Figure 5(a).

Figure 6(a) and (b). Atlantic Sea Scallop and American Lobster: Optimal Revenue Weight Shares (a) and Implicit Revenue Weights (b) by Species Groups, ME, MA and RI, 1965-2012. A closer look of subplots for Groups 3 and 4 in Figure 5(a) and (b).

Figure 7(a) and (b). Inefficiency in the Commercial Fishing Industry (a) and Overfishing (b), ME, MA, RI, 1964-2012. Vertical axis in (a) shows the risk gap (risk level per dollar of revenue). Vertical axis in (b) shows the difference between the actual total revenue and the adjusted total revenue under stock constraints ($100m, 2012).

Figure 7(c). Sensitivity of Risk Gap with Respect to Fish Stock Constraints (*B*).

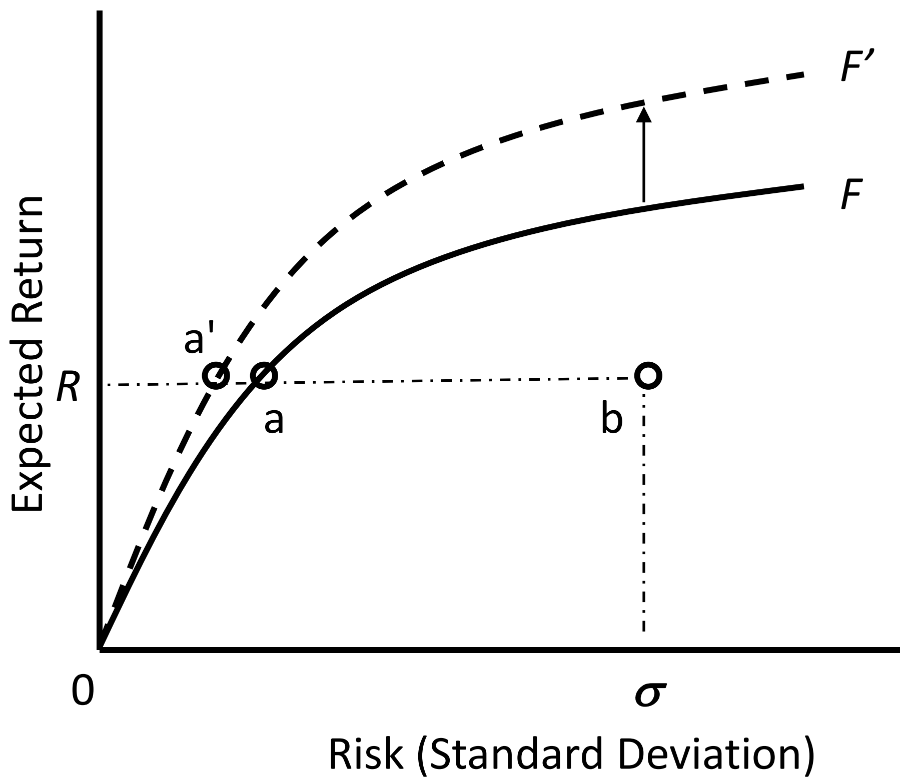
Figure 8. Optimal and Implicit Weights and Revenue Share for Atlantic Sea Scallop, New Bedford, 1964-2012.

Figure 9. Optimal and Implicit Weights for Atlantic Cod, Gloucester, 1964-2012.

Figure 10. Risk Gaps at Port Level, 1964-2012. Gloucester (a); New Bedford (b). Vertical axis shows the risk gap (risk level per dollar of revenue).

Figure 11. NELME Efficient Frontiers and Actual Portfolios under Stock Assumption I, 1990-2012. Vertical axis shows the risk gap (risk level per dollar of revenue).

Figure 12. Portfolio Model as an Ecosystem Management Tool.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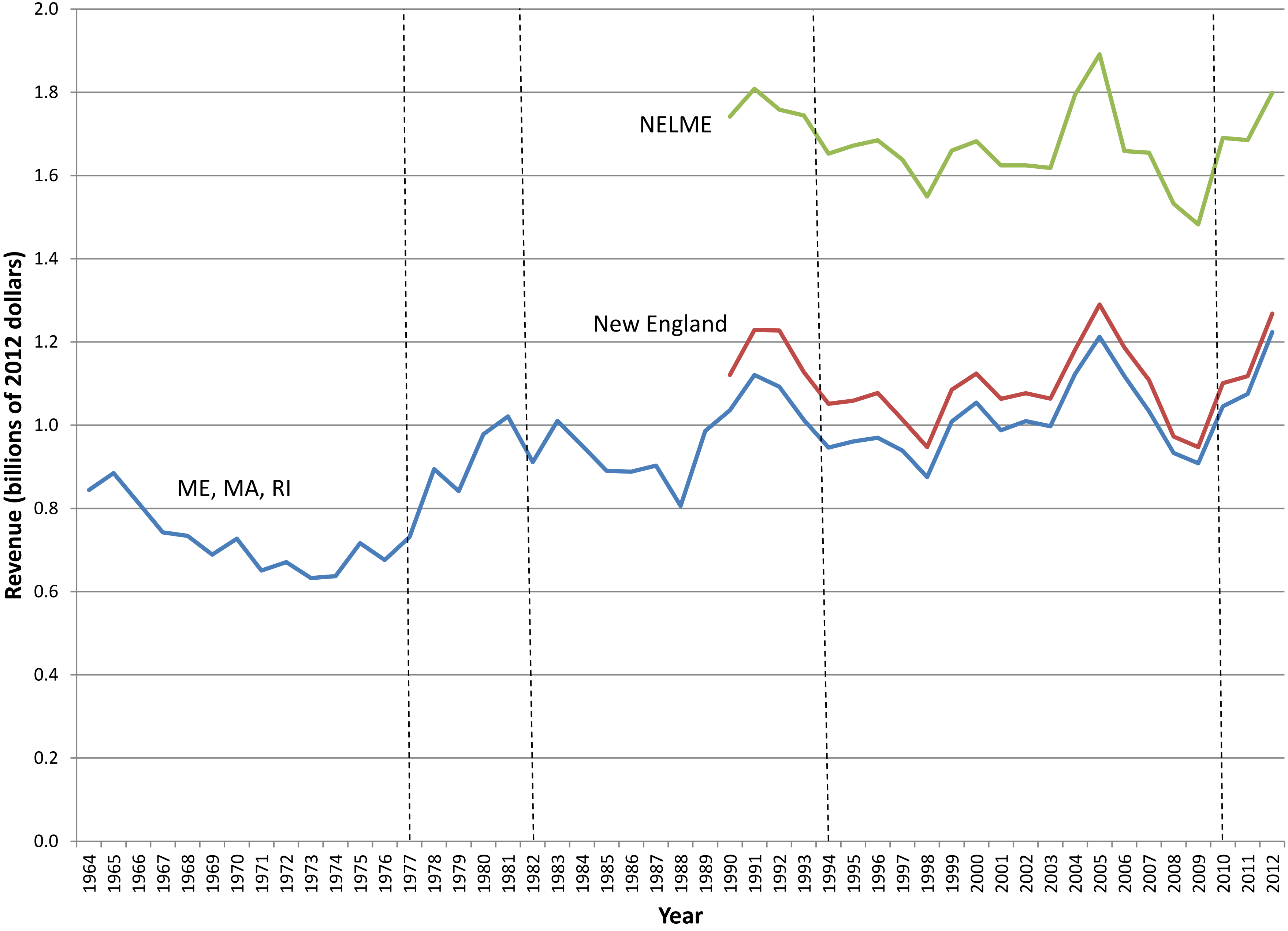
Risk (Standard Deviation)



Revenue (billions of 2012 dollars)
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1.2
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0.6
0.4
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NELME
New England
ME, MA, RI
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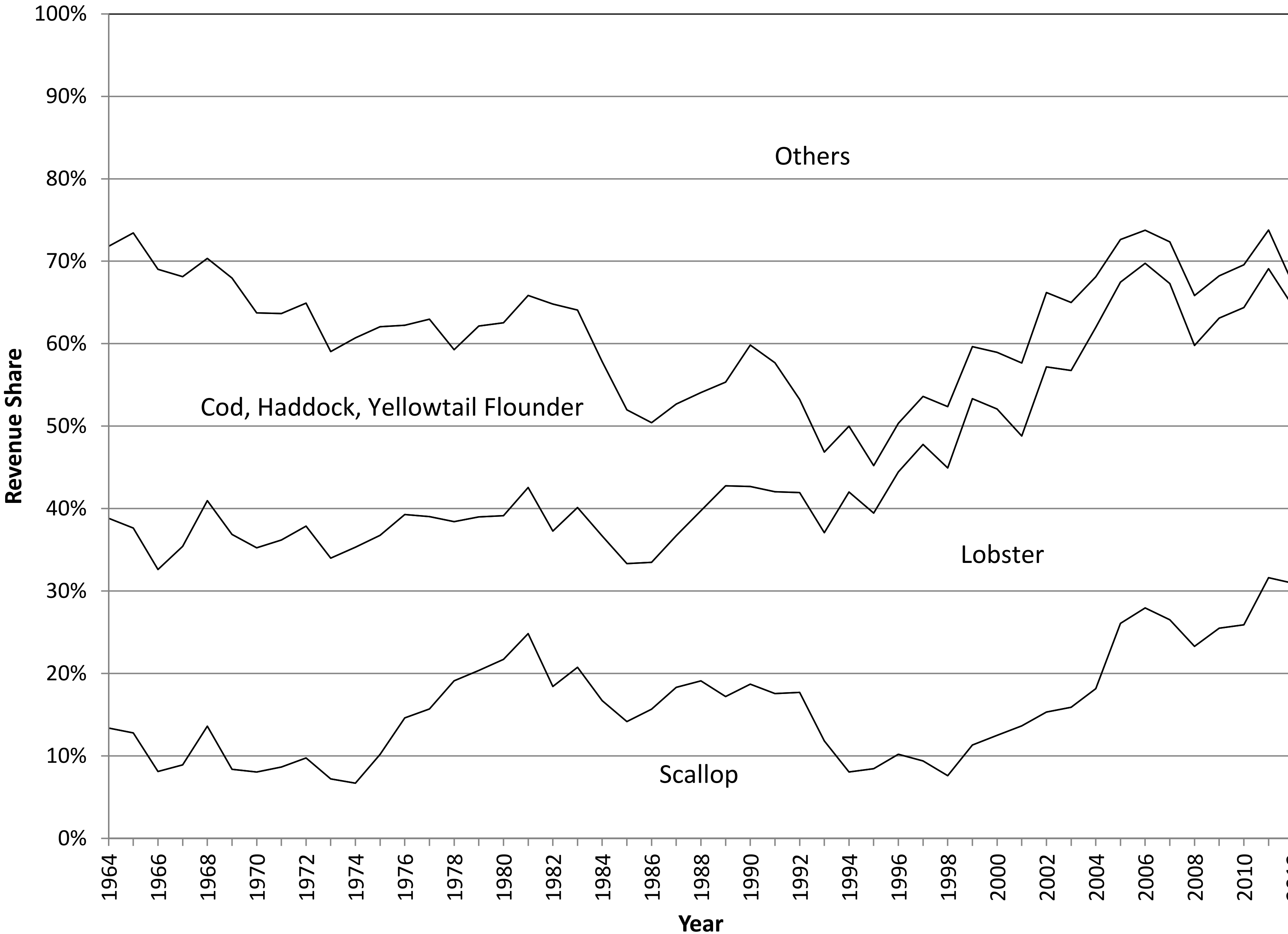
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80%
70%
60%
50%
40%
30%
20%
10%
0%
Revenue Share
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Lobster
Scallop
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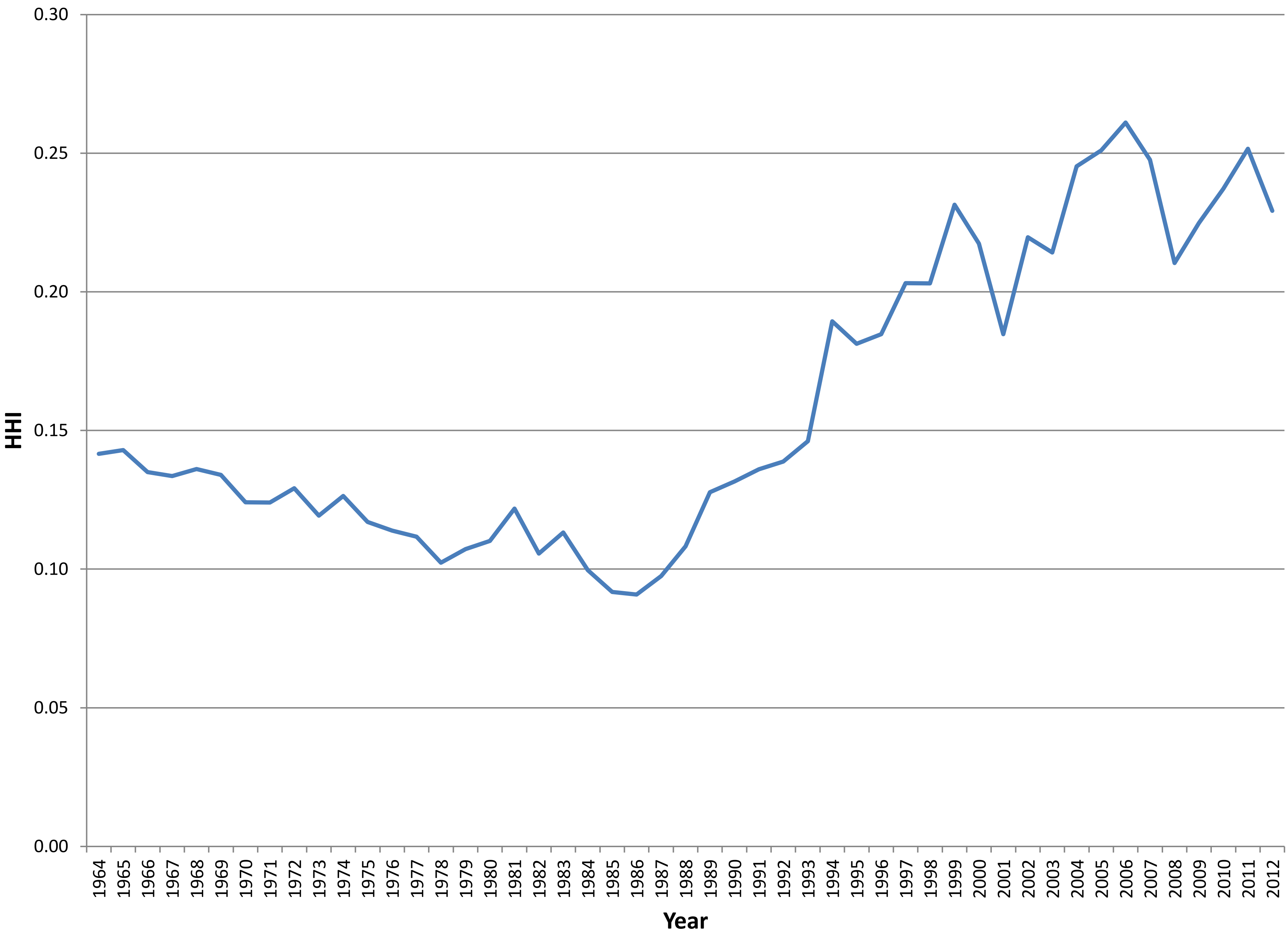
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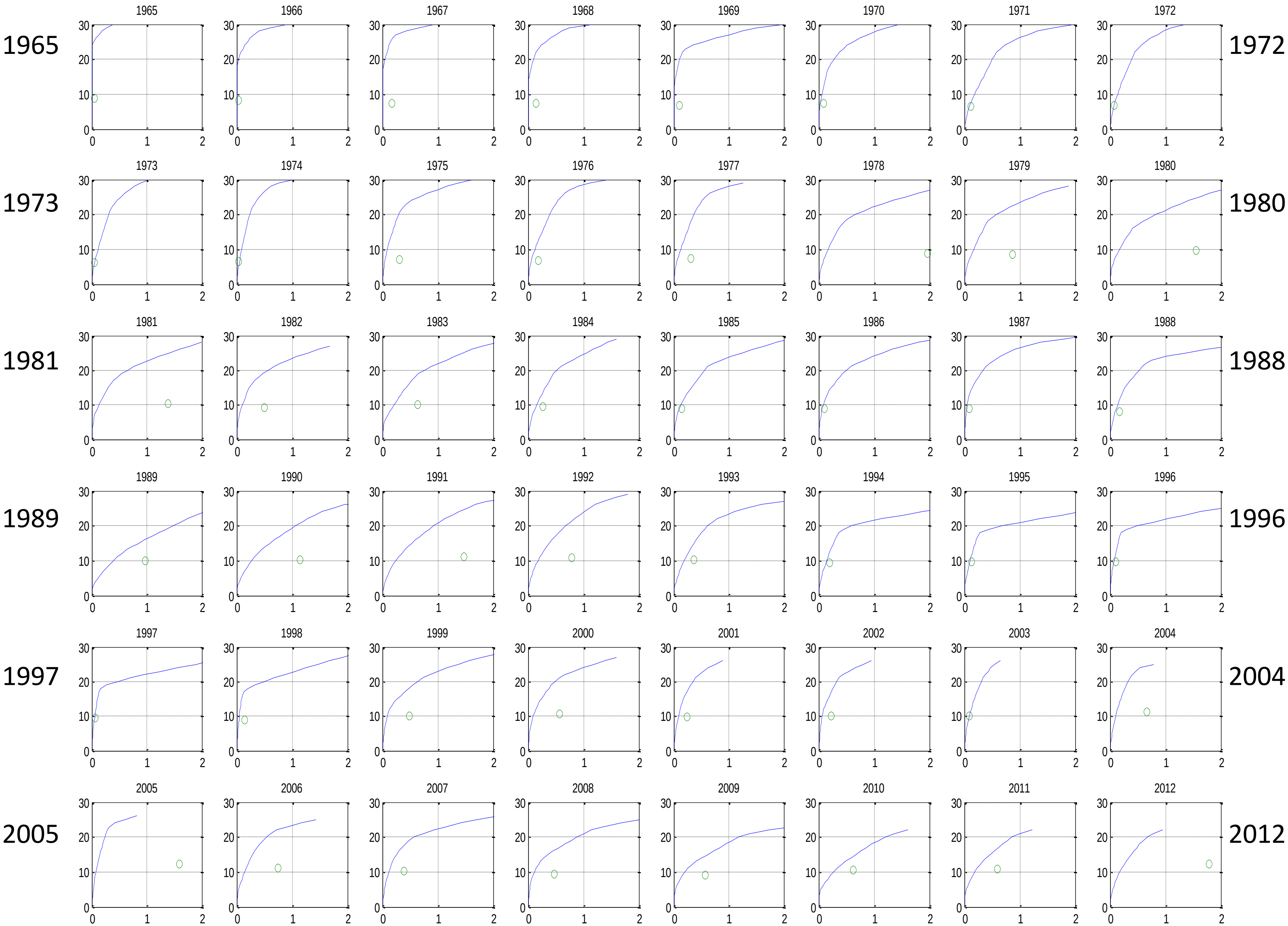
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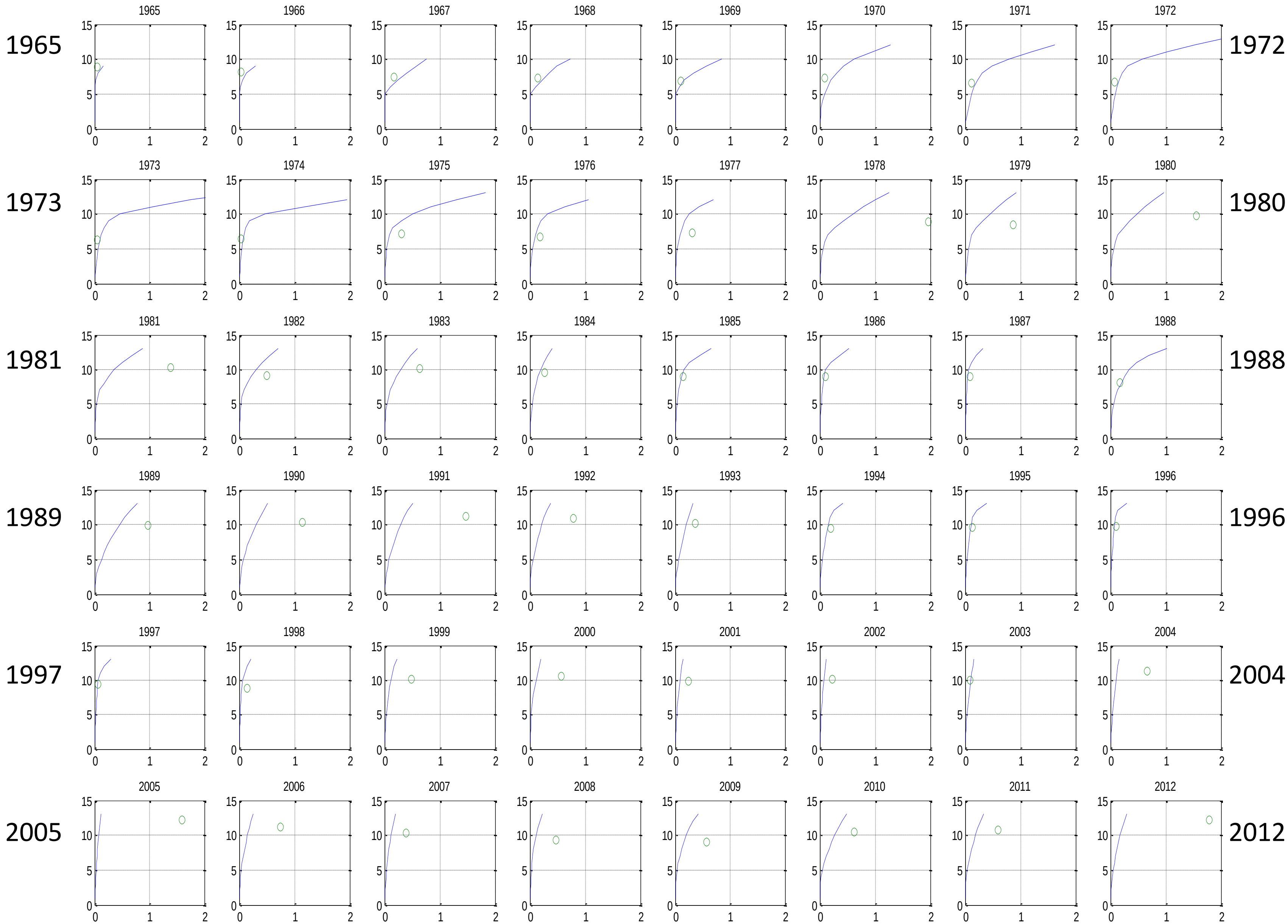
4(b)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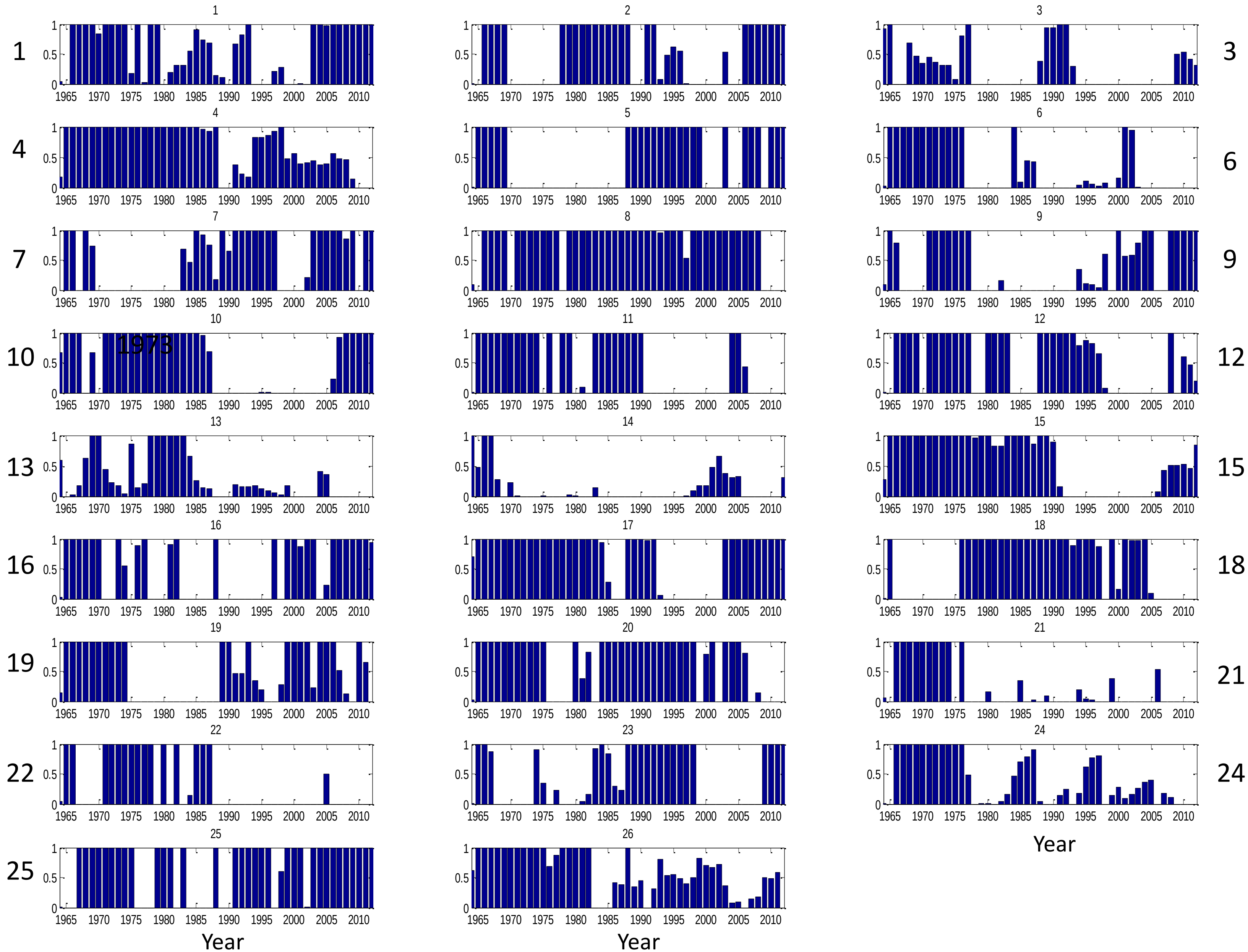
5(a)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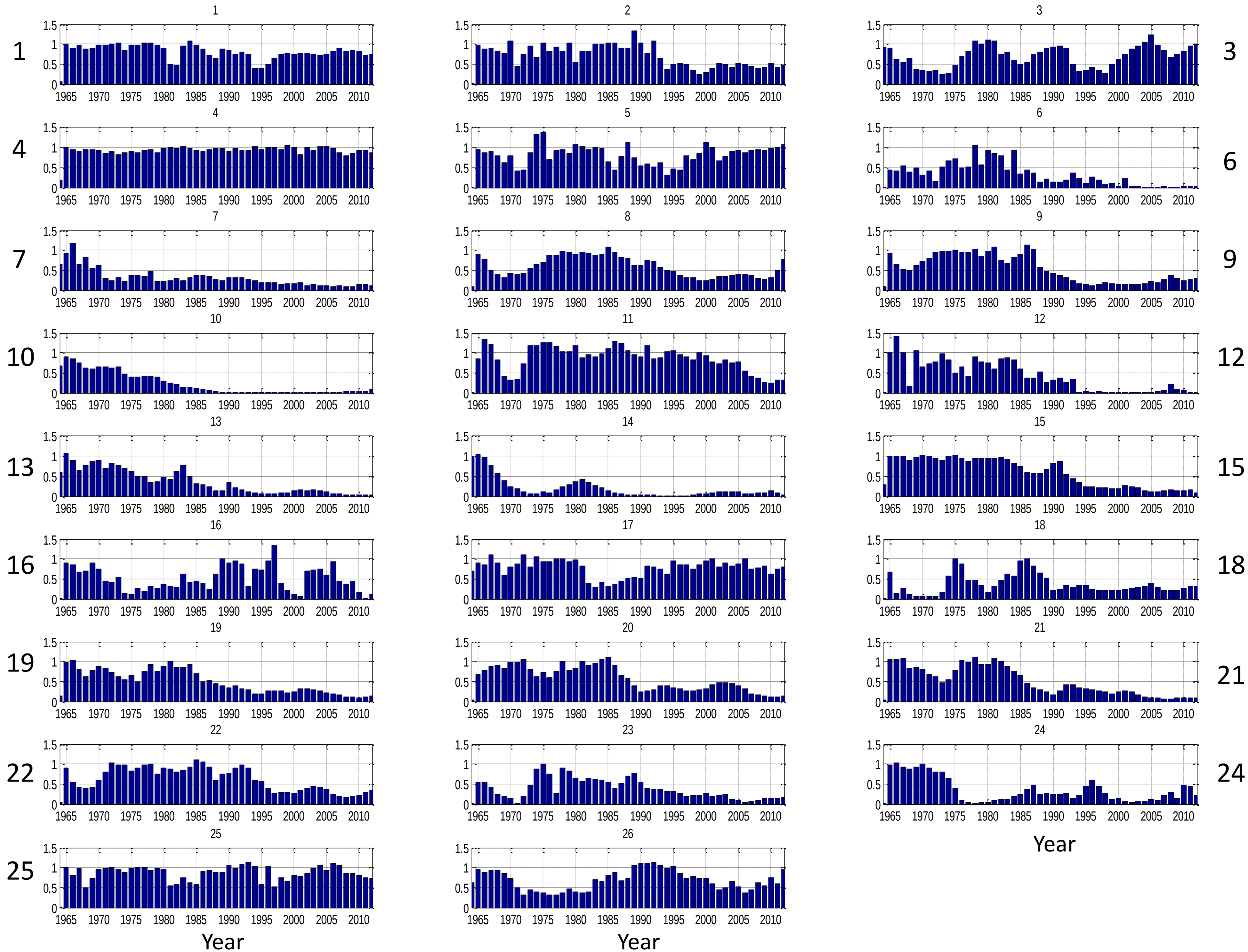
5(b)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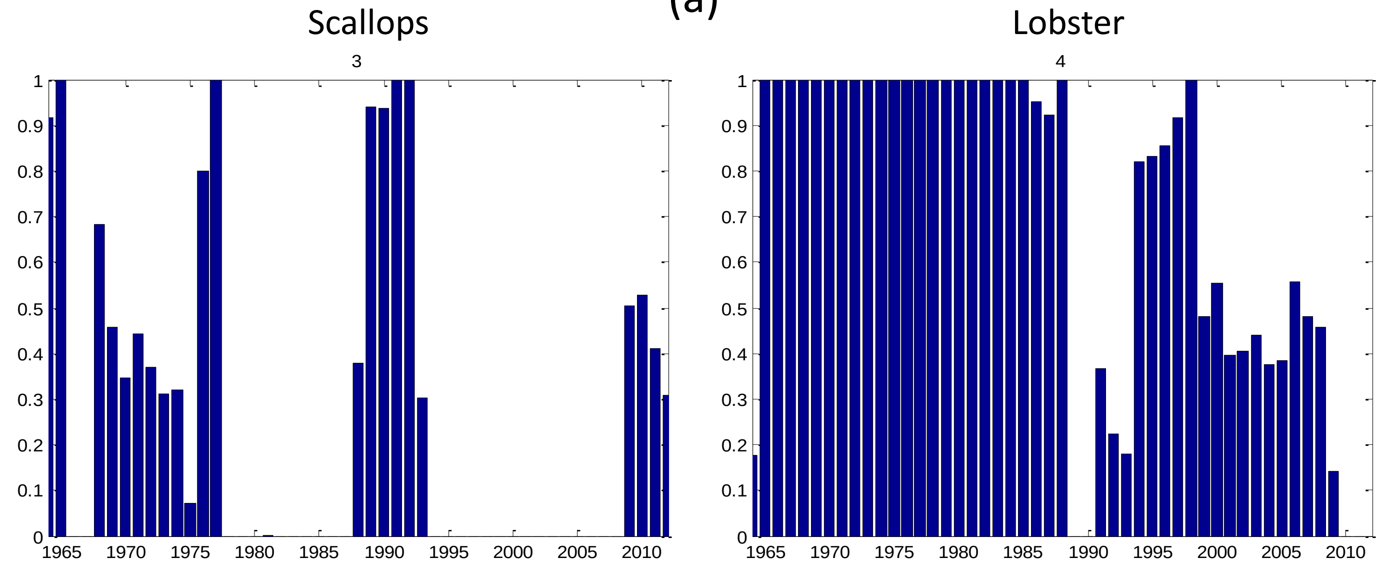
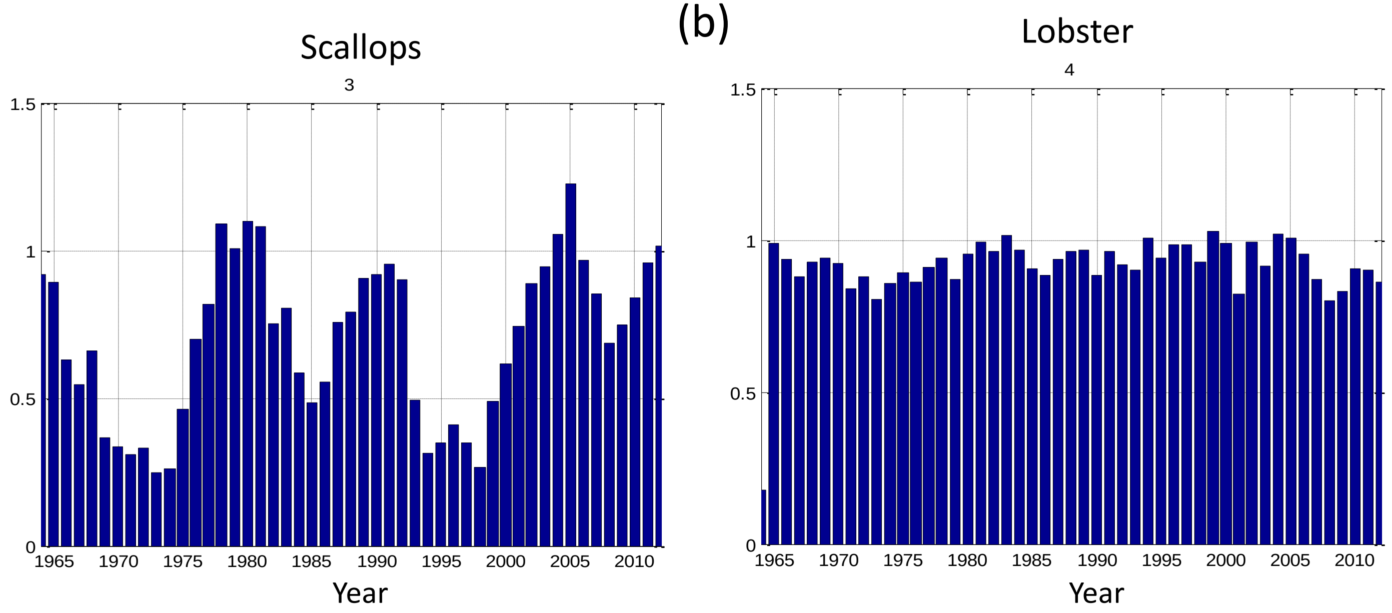
6(a) and (b)

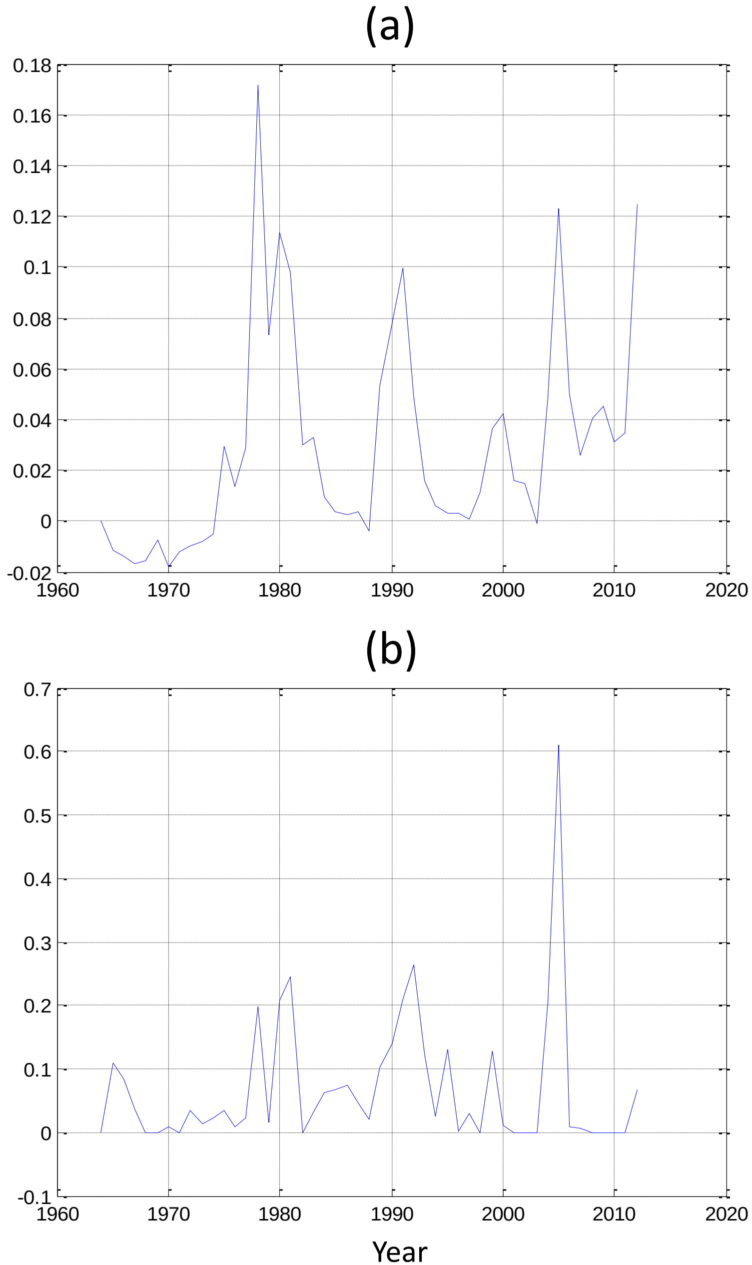
(a)



(b)



7(a) and (b)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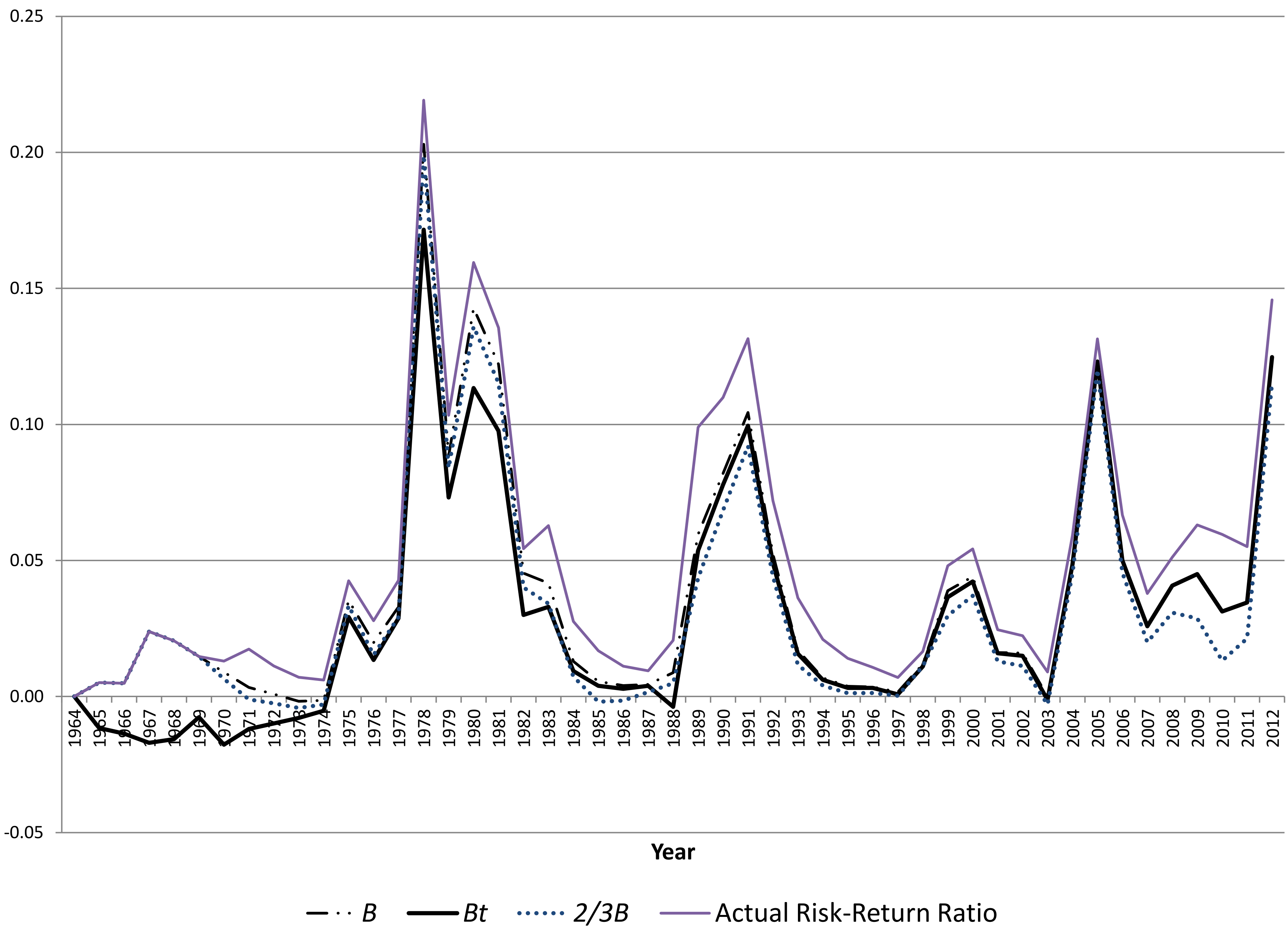
1960
1970
1980
1990
2000
2010
2020
Year

7(c)



0.25
0.20
0.15
0.10
0.05
0.00
-0.05
1964
1965
1966
1967
1968
1969
1970
1971
1972
1973
1974
1975
1976
1977
1978
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
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1992
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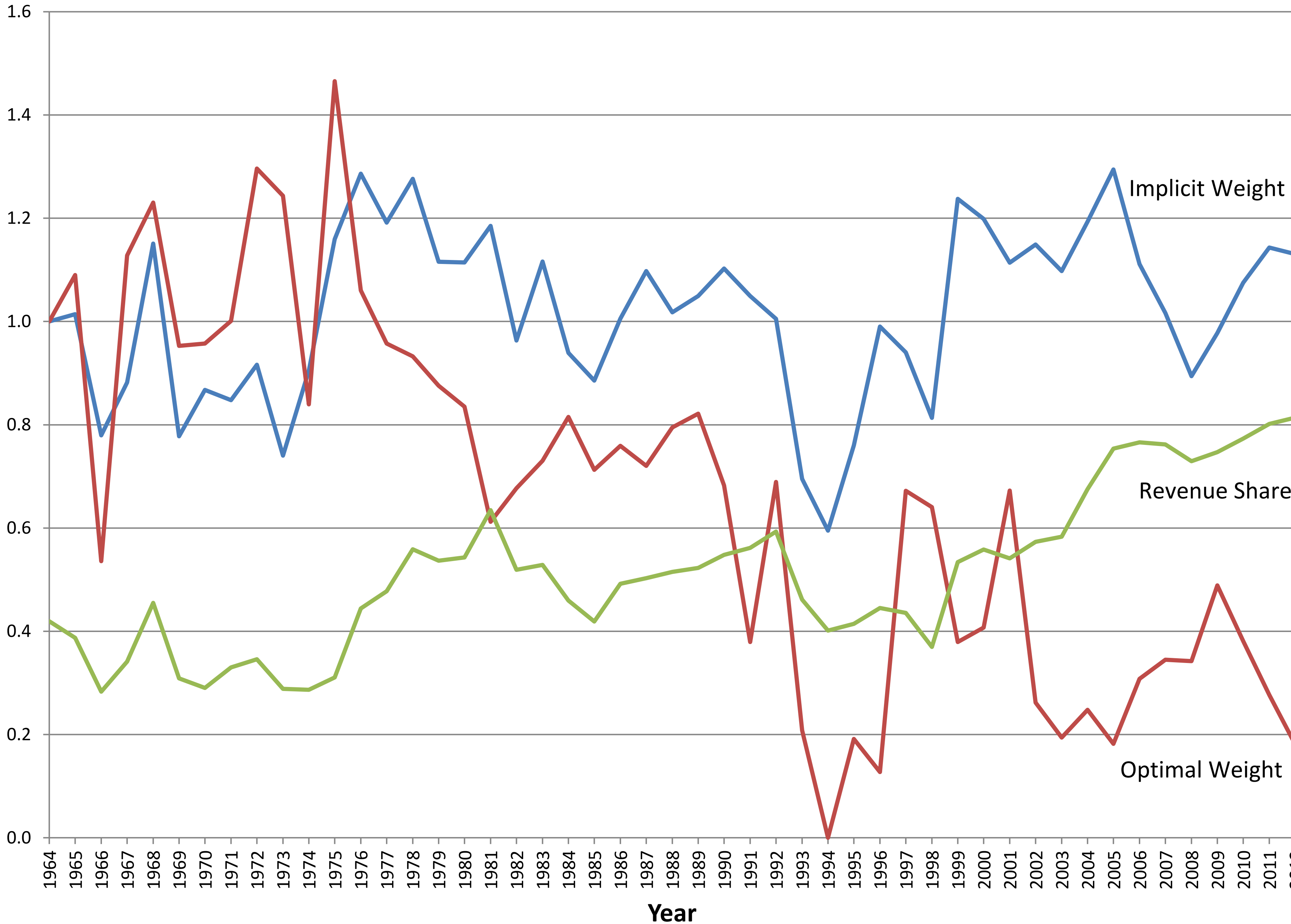
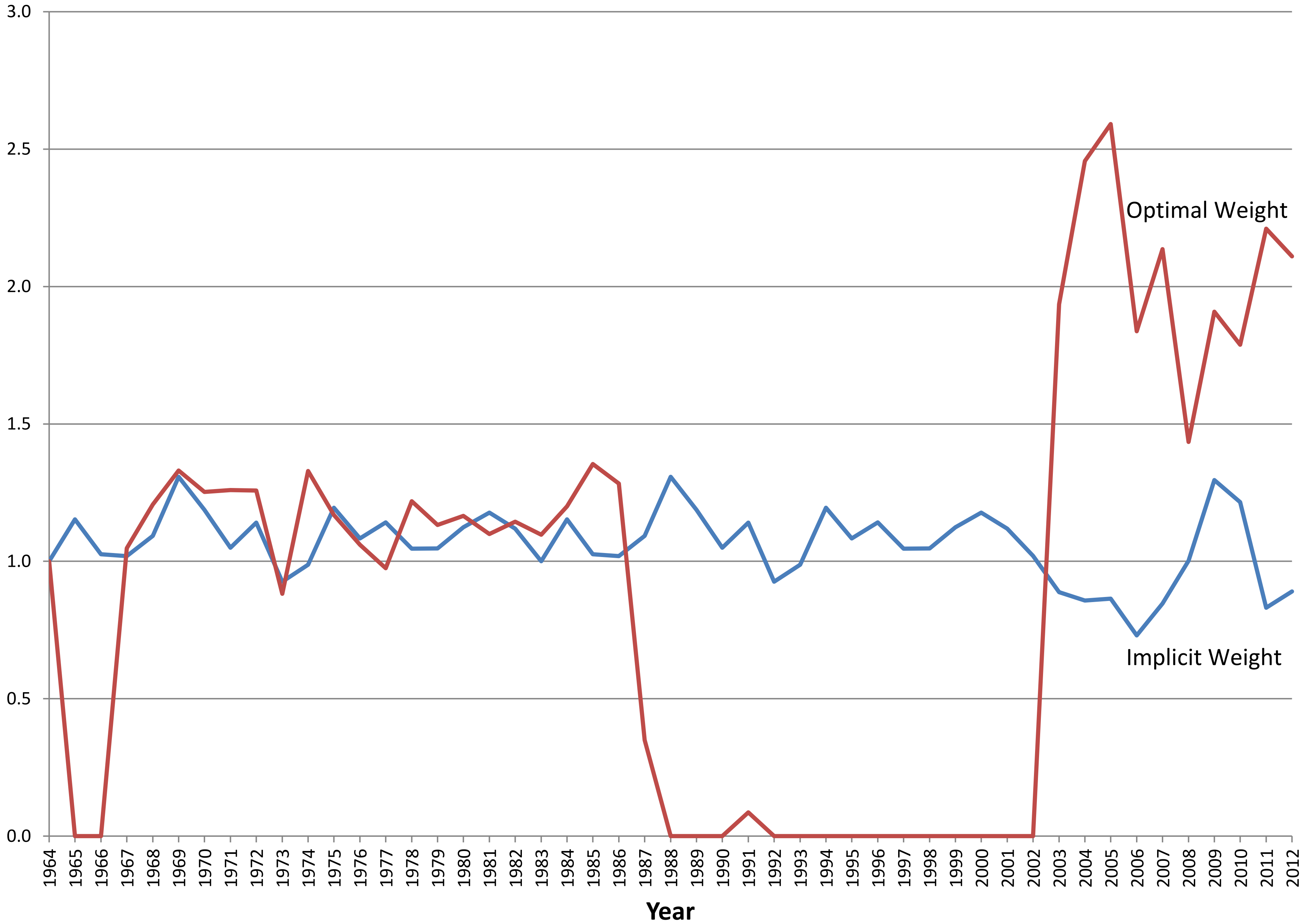
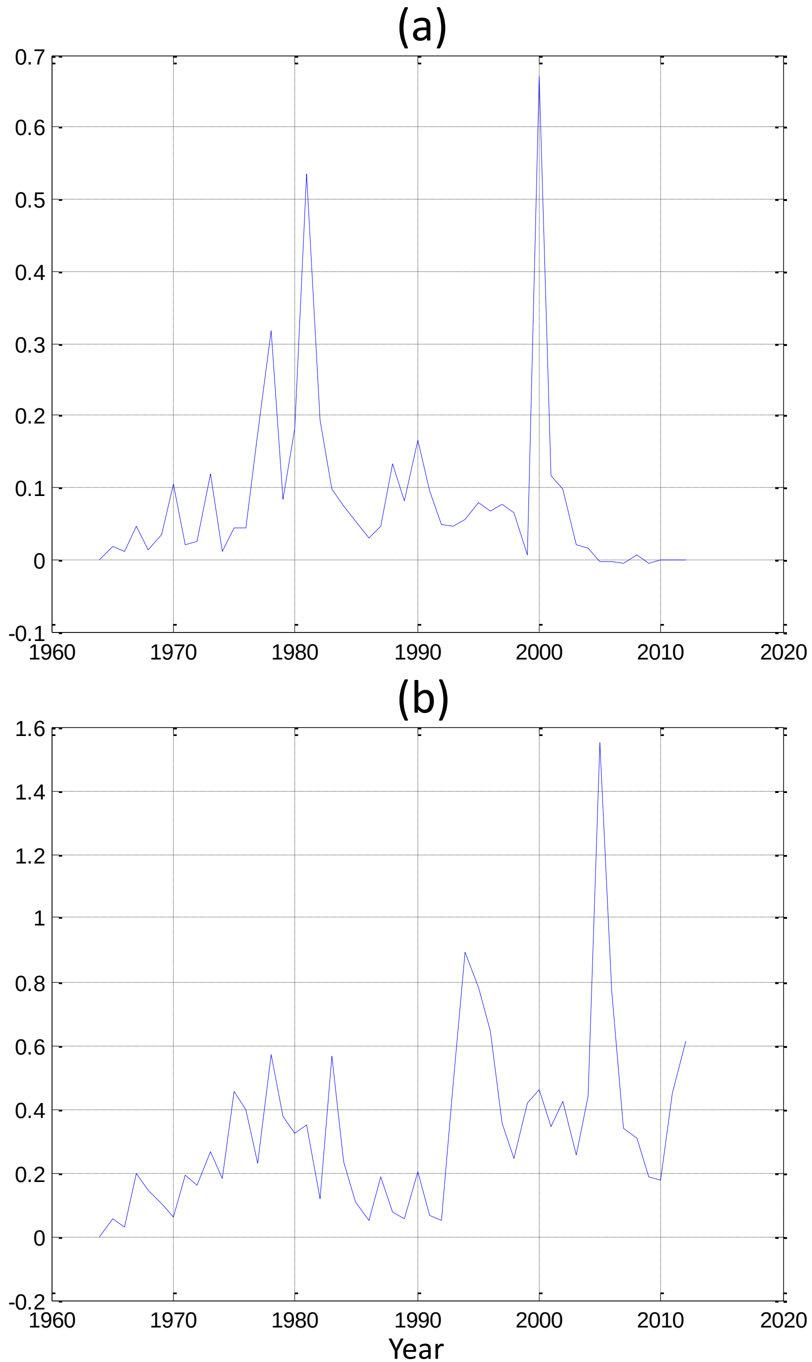
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
2010
2011
2012
Year
B
Bt
2/3B
Actual Risk-Return Ratio

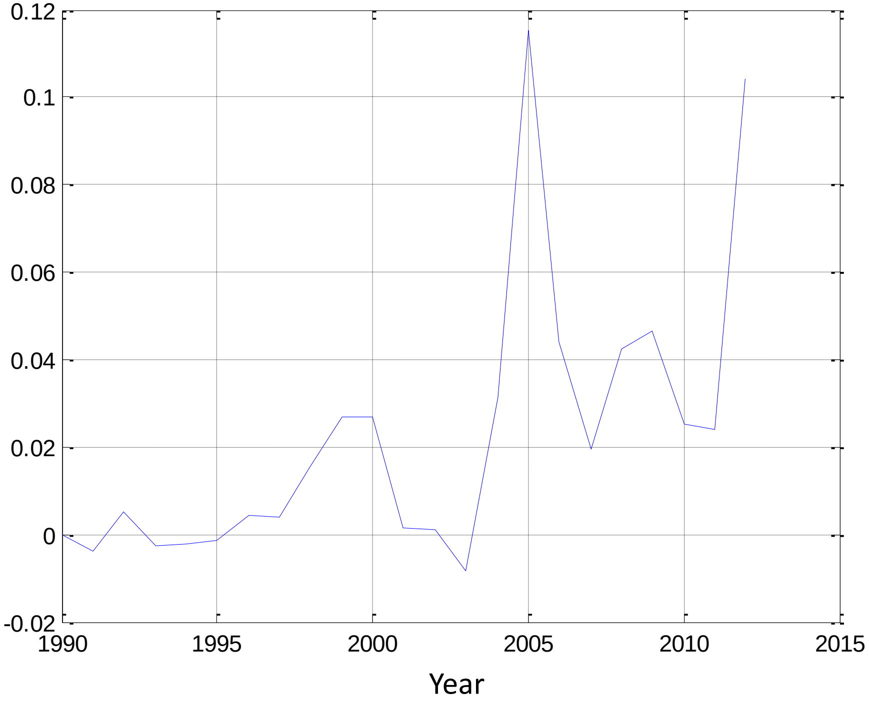

1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
Implicit Weight
Revenue Share
Optimal Weight
1964 1965 1966 1967 1968 1969 1970 1971 1972 1973 1974 1975 1976 1977 1978 1979 1980 1981 1982 1983 1984 1985 1986 1987 1988 1989 1990 1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001 2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012
Year


3.0
2.5
2.0
1.5
1.0
0.5
0.0
Optimal Weight
Implicit Weight
1964 1965 1966 1967 1968 1969 1970 1971 1972 1973 1974 1975 1976 1977 1978 1979 1980 1981 1982 1983 1984 1985 1986 1987 1988 1989 1990 1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001 2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012
Year

10(a) and (b)



(a)
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
-0.1
1960
1970
1980
1990
2000
2010
2020
(b)
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
-0.2
1960
1970
1980
1990
2000
2010
2020
Year


0.12
0.1
0.08
0.06
0.04
0.02
0
-0.02
1990
1995
2000
2005
2010
2015
Year


EXCESSIVE HARVEST
YIELD (REVENUE) INCREASE
EXCURSION FROM RISK MINIMIZING PORTFOLIO
STOCK DECLINE
CONSERVATION & MANAGEMENT MEASURES