Ecosystem Portfolios: A finance-based approach to ecosystem management

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Paper to be presented at the
AERE Workshop 2005: Natural Resources at Risk
Jackson, WY
June 2005

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* Sanchirico and Smith share first authorship. The authors thank the National Oceanic and Atmospheric Administration Chesapeake Bay Program for financial support of this research, NOAA Grant#NA04NMF4570356.

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I. Introduction

The recent collapse of some fish stocks along with the uncertainty involved in managing marine systems have prompted fisheries scientists to suggest a precautionary approach (Garcia, 1994; Lauck et al., 1998; Myers and Mertz, 1998; Darcy and Matlock, 1999; Hilborn et al., 2001; Charles, 2002; Gerrodette et al., 2002; Ludwig, 2002; McAllister and Kirchner, 2002; and Rosenberg, 2002; Weeks and Parker, 2002). In the short-term, many argue that managers should address the inherent risks in complex ecosystems by taking out an insurance policy for each stock, where the event to insure against is a stock collapse. Mechanisms often cited as means to accomplish this are setting or lowering harvest quotas, closing areas and seasons off to fishing, and changing mesh size. At the same time, there is momentum to shift the policy focus from managing species independently to one that takes an ecosystem-based perspective (Botsford et al. 1997, Pew Oceans Commission 2003, U.S. Commission on Ocean Policy 2004, Pikitch et al 2004). In this paper, we develop an approach to ecosystem-based management of fisheries by drawing on financial portfolio theory, and we illustrate the method using historical catch data for the Chesapeake Bay.

Financial portfolio analysis provides a useful framework for conceptualizing and implementing ecosystem management in fisheries. First, managing risk and returns in marine ecosystems is similar to daily decisions in financial markets, where financial managers balance relative risks and returns across a set of correlated assets. Species interdependencies effectively mean that risks associated with each species are correlated—whether positively or negatively—and because of this correlation, there are potential benefits from considering multiple fish stocks jointly. Second, a broader view of how to manage risk beyond insurance is necessary for fishery management, since it is neither clear what the payout nor the premium should be. Finally, a portfolio approach provides an empirical basis for assessing the tradeoffs of harvesting multiple interacting species without the complications of a fully structural model of the ecosystem that would involve substantial nonlinearities and numerous quantities that are difficult to measure.[1] Ecosystem portfolio analysis adopts a more reduced-form description of species interactions by determining the joint probability distribution across species, where the interactions stem from predator-prey relationships, environmental conditions and shocks, and human exploitation.

Drawing the analogy between managing risky assets and managing multispecies fisheries is a relatively new idea, even though the foundation for this idea is neither new to ecology nor economics. In ecology, Walters (1975) derives a mean-variance frontier for single-species management, while Real (1991) uses portfolio theory to describe animal behavior. In economics, the capital-theoretic underpinnings of managing natural resources has a long history (Clark and Munro 1975). Portfolio management of fisheries can be a means of allocating catch across life history stages (Baldursson and Magnusson 1997). Arnason (1998) alludes to multispecies portfolio management in a deterministic bioeconomic model by suggesting that managers choose a vector of Total Allowable Catches (TACs), while Hanna (1998) explicitly discusses the idea of selecting “species portfolios” as a means to match management objectives with ecosystem structure. Hilborn et al. (2001) provide a justification for portfolio management at the regional level by pointing out that total productivity aggregated across species is subject to less variability

[1] See, for example, the structural ecosystem models being developed at the University of British Columbia Fisheries Centre or the general equilibrium ecosystem approach of Finnoff and Tschirhart (2003).

than the productivity of individual species. Edwards et al.(2004) formally develop the analogy within the context of standard bioeconomic models (Clark 1990) and provide a stylized simulation of a three-species system. More recently, Perruso et al. (2005) apply portfolio theory to individual fishermen targeting decisions in the U.S. Atlantic and Gulf of Mexico pelagic longline fishery.

Our contribution is to investigate and empirically derive ecosystem portfolios to inform multispecies fisheries management. In particular, we derive mean-variance revenue frontiers for Chesapeake Bay fisheries using historical data on catches and prices. A point on the frontier represents a portfolio (combinations of species) that achieves a level of expected ecosystem revenues at minimum variance. After deriving the efficient frontiers, we compare the observed regional portfolios to the frontier to evaluate potential risk reductions from taking a portfolio approach. We also measure the risk reductions from taking an ecosystem versus a single species perspective. Finally, we incorporate information on the species trophic levels in the Chesapeake Bay ecosystem to derive efficient trophic-aggregation portfolios.[2]

The Chesapeake provides a useful application for several reasons. First, there is a long time series of detailed catch data. Second, some important species in the Chesapeake have experienced dramatic declines (e.g., oysters, blue crabs), while others have declined and rebounded (e.g., striped bass). The importance of these multispecies interactions was the topic of a recent fisheries management workshop (Houde, et al. 1998). In the latest update to the Chesapeake Bay Agreement[3] there is a goal to develop ecosystem-based fisheries management plans for target species by 2005. To guide the development of these plans a Chesapeake Bay

[2] In the future, we plan to investigate using trophic information whether “fishing down the food web” (Pauly et al. 1998) is an efficient or an inefficient way to manage risks and returns.
[3] Chesapeake 2000 is the latest agreement between the State and Federal entities that share management authority for Chesapeake restoration. See: http://www.chesapeakebay.net/agreement.htm

Fisheries Ecosystem Plan has been developed that calls for examination of patterns of removals (i.e., harvests) as well as characterizing and incorporating uncertainty into fisheries management decisions.[4] Furthermore, there is parallel work underway in marine ecology to inform ecosystem-based management in the Chesapeake using a specific structural ecological model called ECOPATH with ECOSIM (Christensen and Walters 2004).

Along with providing information on how to reallocate the shares of exploitable species in an ecosystem, our analysis provides managers with an *ecosystem-based indicator* that describes the state of the system by measuring the distance between the current state of affairs and the frontier. The importance of developing metrics for ecosystem-based fisheries management was stressed by Brodziak and Link (2002). Depending on how the assets are defined, our indicator can summarize portfolios along ecological dimensions (e.g., trophic levels), socioeconomic dimensions (e.g., employment or profits), or both simultaneously. Because of this we believe that the measure is a useful complement to the standard biological measure, such as overfished, fully exploited, and underfished.

In the next section we develop the conceptual framework and adapt financial portfolios to ecosystem management. The following section provides background on Chesapeake Bay fisheries and describes the data set that we use to estimate the portfolios. We present results in the next section and conclude with a discussion of future research on the portfolio approach to ecosystem management.

II. Conceptual Framework

Markowitz (1959) developed basic portfolio theory as a systematic means to minimize risk for a given level of expected return. The necessary data to construct an efficient portfolio

[4] See http://noaa.chesapeakebay.net/fepworkshop/netfep.htm.

include measures of expected returns for each investment, variance of returns for each investment, and all of the pair-wise covariances. The efficient mean-variance frontier for a portfolio of assets is derived by solving a sequence of quadratic programming problems. Based on an investor's wealth and risk preferences, a money manager can choose a particular point on the frontier and purchase the assets that comprise the optimal portfolio.

The financial manager constructs portfolios by choosing shares of each asset in the portfolio (c_i) such that the variance of the portfolio is minimized and the expected returns are at least as great as the target level of returns. Formally, the quadratic programming problem is:

$$\begin{aligned} &\min_{c_i} \mathbf{c'\Sigma c} \\ &s.t. \quad \mathbf{c'\mu} \geq \tilde{\mathrm{M}} \\ &0 \leq c_i \leq 1 \quad and \quad \sum_{i=1}^{n} c_i = 1 \quad \forall i = 1,\ldots,n \end{aligned} \tag{1}$$

where $\boldsymbol{\mu}$ is the (n x 1) vector of expected returns with individual expected returns denoted by μ_i, the covariance matrix of asset returns is $\mathbf{\Sigma}$ with individual elements denoted by σ_{ij}, $\tilde{\mathrm{M}}$ is the target level of expected returns, and n is the number of assets. The total expected return of the portfolio is $\mathbf{c'\mu}$ and the total variance of the portfolio is $\mathbf{c'\Sigma c}$. Since c_i is the share of the asset in the portfolio, each must be between zero and one, and together they must sum to one. Note that to derive points on the frontier for a given level of returns, there are a total of *(2n + 2)* constraints.

In this paper, we derive fishery revenue portfolios (not asset returns), but as we mentioned earlier, the method can be used to consider other objectives. Under this objective, $\boldsymbol{\mu}$ is an *(n x 1)* vector of mean revenue of the *n* harvestable species in the Chesapeake Bay ecosystem. Contrary to structural bioeconomic models, we lump together many factors that together can lead to negative or positive correlations among species—portfolio managers take a similar approach with assets. In fisheries, these factors include trophic interactions, environmental fluctuations, and fishing gear choices that could induce covariance amongst catch rates, and output market interactions that could induce covariances in prices. All of these conditions lead to covariance in revenues, which we denote by the *(n x n)* matrix $\mathbf{\Sigma}$. For a given portfolio of shares $\mathbf{c}$, we express total mean revenue as $\mathbf{c}'\boldsymbol{\mu}$, and the variance of revenue as $\mathbf{c}'\mathbf{\Sigma}\,\mathbf{c}$.

While the construction of ecosystem portfolios uses the same architecture as in finance, a couple of issues arise when applying the quadratic programming problem to ecosystems. For example, in financial analysis, the ability to borrow money can help investors purchase the quantity of the assets implied by the optimal shares. In an ecological system, however, the optimal shares of the portfolio might correspond to a level of extraction that is not sustainable. There is no ecological mechanism for borrowing to "purchase" the asset at the level implied by the efficient frontier. Therefore, we need to modify the financial architecture to ensure that shares along with the allocation of absolute quantities to each species represent sustainable solutions.[5]

One biological point of reference for sustainability is maximum sustainable yield (MSY), which corresponds to a fish stock size equal to the half the carrying capacity in the logistic biological growth function. The need to ensure that optimal shares correspond to sustainable catch levels could be met, therefore, if we constrain the catches to be less than or equal to MSY. Another possibility is a system-wide productivity measure that could allow the MSY of one species to be violated in a given period as long as the total catch from the ecosystem was sustainable. Sanchirico and Smith (2003) discuss and illustrate both types of sustainability constraints using FAO fisheries data for the Northwest Atlantic.

[5] A financial analogy would be adding a budget constraint in the quadratic programming problem. In this setting, the quantity not just the shares of the assets would be constrained to satisfy the investor's budget.

We focus on the constraint that the catch of each species is less than or equal to its MSY. Ideally, we would have fishery-independent estimates of the MSY for each species. Unfortunately, within fishing regions like the Chesapeake Bay, such estimates rarely exist and if they do, it is usually only for several of the top revenue-producing species.

In what follows, we customize the portfolio quadratic program (1) to an ecosystem application. Let, x_{it} be the catch of species *i* in period *t*, and $\max\left\{x_{i1,}x_{i2,}...x_{iT,}\right\}$ is the maximum observed catch for species i over the time series. In addition, suppose that maximum sustainable yield for species *i* (χ_i) can be approximated as the maximum observed catch times a scaling coefficient (γ_i) such that $\chi_i = \gamma_i \cdot \max\left\{x_{i1,}x_{i2,}...x_{iT,}\right\}$.[6] For an underexploited species, $\gamma_i > 1$, and for an overexploited species, $0 < \gamma_i < 1$.

Using the sample mean ($\overline{x}_i = T^{-1}\sum_{i=1}^{T} x_{it}$) as an estimate of expected catch for species i, expected catch from species *i* in the portfolio is $c_i\overline{x}_i$. The sustainability constraint ensures that the expected catches must be less than or equal to the species estimated MSY, or $c_i\overline{x}_i \le \chi_i = \gamma_i \cdot \max\left\{x_{i1,}x_{i2,}...x_{iT,}\right\}$. Rearranging the constraint, we solve for the upper bound on the shares (c_i^{max}), $c_i^{\max} = \gamma_i \cdot \max\left\{x_{i1,}x_{i2,}...x_{iT,}\right\} \ / \ \overline{x}_i$.

We are now ready to write down the ecosystem portfolio problem with the objective of fishery revenues. Formally, the optimal shares in the ecosystem portfolio are the solution of

$$\begin{aligned} &\min_{c_i} \mathbf{c'\Sigma c} \\ &s.t. \quad \mathbf{c'\mu} \ge \tilde{\mathrm{M}} \\ &0 \le c_i \le c_i^{\max}, \quad \sum_{i=1}^{n} c_i \le \sum_{i=1}^{n} c_i^{\max} \quad \forall \mathrm{i=1,...,n}, \end{aligned} \qquad (2)$$

[6] One could, of course, also limit the sample over which you are approximating the MSY, such as the maximum catch over the last five years. Other more sophisticated approaches could be to specify or estimate a moving average over time.

where $\mu_i = c_i \bar{r}_i,\ \bar{r}_i = T^{-1}\sum_{i=1}^{T} r_{it}$, r_{it} is revenue from species i at time t, and $\tilde{M}$ is the target level of ecosystem expected revenues.

It is important to point out that the portfolio problem as presented assumes time invariance in the means and covariances. This is an overly restrictive assumption, and there are many reasons why this might not hold in practice. For example, means and covariances would evolve over time if the ecosystem has crossed a threshold and there is no returning to previous conditions (Jackson et al. 2001). Whether such a structural change has occurred in either the ecological or economic domain is not immediately clear for the Chesapeake Bay, even though there have been some major shifts in catches. We plan to investigate non-stationarity in the means and covariances in future work. Having said that, our assumption is a reasonable starting place given that empirical multispecies fisheries portfolios have not been derived previously.

III. Chesapeake Bay Background and Data

Chesapeake Bay fishermen are known locally as watermen, reflecting their ability to earn a living off the water from a variety of activities (Paolisso 2002). The fishing activities themselves are varied, employing different gears and relying on a variety of species. Hildebrand and Schroeder's (1928) description of the fisheries in 1920 remains a fairly accurate representation of current species fished and gears used for finfish. They describe the predominant finfishing gear as pound nets followed by seines and gillnets. Bottom trawls are generally not allowed in Chesapeake Bay. Temperature and migration patterns determine the seasonality of the catch with activity beginning earlier in the season in the Virginia portion of the bay when anadromous river herrings and shad return to spawn in the early spring. Similarly, blue crabs emerge from their winter hibernation and begin being caught in the late spring as the

Chesapeake Bay water's warm in a south to north pattern (Lipcius et al. 2002). For blue crab fishing, crab pots are the predominant gear type, but scrapes and dredges may also be used. Oysters are caught predominantly by tongs that are either completely operated by hand or with an hydraulic assist. Limited dredging for oysters is also allowed. The oyster fishery operates in the fall and continues through winter, weather and ice conditions permitting.

The pattern of harvests from Chesapeake Bay fisheries has changed over time, even though there has been no significant change in the total harvest volume. Some of these changes might be due to biological shifts while others relate directly to management actions that may have been adopted in response to changes in the health of fish stocks. The most dramatic changes concern three of the most valuable species: oysters, blue crab, and striped bass. Lipton et al. (1992) describe the events leading to and resulting from a decline in Chesapeake Bay oyster harvests post-1950. A significant factor was the outbreak of the disease multi-nucleated sphere unknown (MSX)[7] mostly in Virginia oysters around 1960. MSX did not affect Maryland's production greatly until 1981, and Maryland conducted an oyster repletion program that planted oyster shells from shucking houses and mined from deposits in Chesapeake Bay to maintain production at around 2-3 million bushels per year. While MSX has waxed and waned in subsequent years, the current situation is that oyster production, which was the most valuable product harvested from the Chesapeake Bay, is virtually non-existent.

Striped bass landings have also exhibited changes since 1950. Catches and reproductive success were severely limited so that by 1985, Maryland imposed a moratorium on possession of striped bass and Virginia imposed a moratorium in 1989. After three years of successful recruitment, the fishery reopened in 1990 and the stock is considered fully recovered.

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[7] MSX is a parasitic disease that is now classified as *Haplosporidium nelsoni*.

Blue crab was not a major fishery and income producer for watermen in Chesapeake Bay until the 1960's. Blue crab harvests peaked in 1981, remained at fairly high levels until about 1998 and have declined to near record low harvests in the last four years. A spawning stock rebuilding plan was implemented in 2001 and remains in place for the Maryland, Virginia and Potomac River Fisheries (Chesapeake Bay Commission 2003).

Management of Chesapeake Bay fisheries is complex because of multiple jurisdictions and the migratory nature of many key species. Species harvested may be under individual state management authority (Virginia and Maryland) and the Potomac River Fisheries Commission. Some stocks (e.g., striped bass) are managed under the auspices of the Atlantic States Marine Fisheries Commission, while others that are principally caught in Federal waters are managed by the Mid-Atlantic Fisheries Management Council. As mentioned earlier, ecosystem-based fisheries management plans are being developed for key species that will serve as input to these multiple management entities when adopting fisheries management actions and regulations.

In this paper, we use 42 years of Chesapeake Bay landings data (1962-2003). The data on Chesapeake landings are readily available from the National Marine Fisheries Service and combine all Maryland and Virginia harvests, including offshore landings. For this analysis, Chesapeake Bay landings were extracted from the raw data files using the recorded water body. Menhaden, the largest catch by volume from the Chesapeake Bay, presents a particular data problem because of the need to protect confidentiality of landings when there are only a few firms reporting harvest. Chesapeake Bay menhaden landings for 1985-1996 were obtained from Smith (1999). Menhaden landings from 1997- 2003 and pre-1985 data were obtained from Smith (personal communication)[8].

[8] Joseph W. Smith. NOAA Fisheries, Beaufort, NC.

For the portfolio analysis, we focus on species or species groups that generate at least $500,000 dollars in real dockside landings revenues (measured in March 2005 dollars) in at least one year.[9] This leaves us with 22 distinct species or species groups that are listed in Table 1. These groupings represent a range of aggregation levels, where species aggregations represent a compromise between economic and ecological taxonomy. For instance, blue crab (*Callinectes sapidus*) is a single species but aggregates across several market categories based on sex, size and stage of molting (i.e., hard, soft or peeler). In contrast, there are several species of catfish caught in the Chesapeake (e.g., *Ictalurus punctatus* and *Ictalurus nebulosus*), but landings data do not differentiate among them, and they are necessarily lumped together. Some species have separate market categories but are typically caught together and catch often is reported in one species category only, e.g. gizzard shad, alewife, and blueback herring. Unclassified finfishes is the greatest taxonomic compromise. It typically contains small fishes, and though there is no species-specific information for this product category, the category is large enough to meet our economic threshold for inclusion in the analysis. Our analysis seeks portfolio allocations across these 22 distinct fishery assets.

Table 1 contains descriptive statistics for our 22 categories. We group fishes by trophic levels extracted from FishBase (Froese and Pauly, 2005). When more than one trophic level is available for a species or when the category contains multiple species, we take a simple average across available estimates. This means that some species could reasonably be classified as high (>3.5) or low (<3.5). However, for the most valuable fish species (striped bass and menhaden), average trophic level is far from the 3.5 cutoff. Table 1 shows that a handful of species have dominated catch and revenues in the Chesapeake, and all of these species have experienced

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[9] We convert nominal revenues to real using the Bureau of Labor Statistics Consumer Price Index (CPI) for the U.S. South (all urban consumers) for 1967-2003 and CPI U.S. (all urban consumers) prior to 1967.

substantial variation as shown by high standard deviations and absolute spreads between minima and maxima. Catches and revenues of less economically dominant species have also shown substantial variability. These descriptive statistics reinforce the importance of systematically considering mean-variance tradeoffs in setting TACs.

IV. Results

A. Portfolios with 22 Distinct Fishery Assets

Deriving portfolios involves solving a series of quadratic programming problems that require a level of expected returns and an estimate of the variance-covariance matrix. We find a point on the frontier for each $2 million interval, starting at zero. Then we interpolate between the points to trace out the continuous frontier. For this analysis, we assume that $\gamma_i=\gamma$ for all species.[10] We examine frontiers for three different values of γ: 0.5, 0.75, and 1.0. These values span the range of high to low stringency of the sustainability constraint. For example, a value of 1.0 is low stringency, because by construction none of the actual portfolio allocations in the sample period would be in violation of the sustainability constraints. Thus, the efficient frontier bows outward as γ increases (Figure 1). If the system is fully exploited, but not overexploited, 1.0 is a reasonable value for γ. The maximum expected revenue from this system is just under $250 million, whereas the maximum expected revenue is just over $130 million when γ=0.5.

If we have measured expected revenues and covariances correctly, these frontiers in Figure 1 represent the best that managers can do. That is, setting a vector of TACs that is on the

[10] This assumption means that for species that are consistently underutilized throughout the sample period, the sustainability constraint is overly restrictive and our analysis precludes the possibility of recommending an increase in harvest over the historic maximum. We argue that this is not a limitation of our method but rather a limitation of the available data. If an estimate of MSY is available for an underutilized species k, then we could use it directly for $c_k^{\max}$.

efficient frontier minimizes the *ex ante* variability in the system for a given level of *ex ante* expected return. As in finance, these frontiers demonstrate the inherent tradeoff between risk and return. Fisheries managers, by operating on the frontiers ensure that they are not accepting more risk than necessary for a given level of return for the ecosystem.

A.1. Comparing Frontiers

The frontiers in Figure 1 are derived using the full covariance matrix. But what if managers choose the shares based only on the variance of the assets? We explore this question by deriving the frontier separately for the full covariance matrix (the Ecosystem case) and for the diagonal-only covariance matrix (the Species case). The Species case would correspond to a multispecies management strategy that accounts for individual species variability but ignores cross species interactions.

Comparing the two portfolios provides insight into the importance of species correlations in the Chesapeake Bay ecosystem. Theoretically, it is not clear which portfolio will have lower risk (variance). For example, consider a two-asset portfolio. It can be shown that if the assets are negatively correlated, then the variance of the portfolio will be lower when taking the asset covariance into account than when the shares are derived only with asset variances. If the assets are positively correlated, however, the Ecosystem portfolio could have higher variance than the Species case. In an *n* asset portfolio, the relative portfolio variance is an empirical question that depends on the actual covariances of species in the portfolio and we do not have any *ex ante* reason for either outcome.

Along with the Ecosystem and Species portfolio, we compute the variance of a portfolio where the shares are derived only taking into account asset variances, but the portfolio variance is calculated using the full covariance matrix. We denote this case as Species in Ecosystem.

This portfolio represents the case where managers ignore species covariances but the covariances are present and therefore affect the variance of the portfolio. It is important to point out that this counterfactual case illustrates the likely variance of the Species portfolio, but the manager decision process is still based on species-level risks only, not ecosystems.

Figure 2 illustrates the three sets of portfolios for two values of γ. We find that if managers ignore covariance—ecological and economic interdependence are left out of portfolio decisions—then they accept more risk for a given level of return (compare Species to Ecosystem). Empirically, this means that there are opportunities to exploit negative covariances across species in the Chesapeake Bay ecosystem in optimal portfolio design.

For the Species in Ecosystem case, the portfolio shares are based on an incorrect assumption, but the corresponding frontier is based on the true covariance. This case can lead to non-monotonicity in the frontier. We find that the positive covariance penalties at low to medium expected revenues lead to high portfolio variance. But eventually, the Species in Ecosystem frontier must converge to the Ecosystem case. When constraints bind, a naive strategy coincides with an informed strategy because all species are included in the maximum amounts possible in the portfolio, and both frontiers are computed with the true covariance matrix. This phenomenon illustrates that the benefits of considering species interdependencies—in terms of lower variability— are smaller when managers push the system to the limits, and the benefits appear greatest at moderate levels of system-wide exploitation. In the limit, there are absolute limits to exploitation that even the most carefully crafted ecosystem-based management cannot exceed.

We can systematically look at how the Species portfolio construction differs from the Ecosystem portfolio construction at different target expected revenue levels and sustainability stringency. To this end, for a given expected revenue and γ, we compute the difference in the coefficient of variation of the Species and Ecosystem portfolios. This provides a standardized measure of how risk per unit return changes over the parameter space. Figure 3 depicts the result for four levels of γ.

In Figure 3, at low revenue levels, only a small number of species need to be in the portfolio to meet the revenue target, and which ones get chosen are simply different when covariances are considered. This creates a big difference between the variability of the two portfolio types. As the revenue target increases to medium levels, it is necessary to bring more species into the portfolio. In other words, there is a forced diversification in the portfolio, and this causes the two portfolios to diverge less and thus the large drop-off. In essence, there are a lot of ways to meet the revenue target with similar variance levels. Still, at low levels there is a non-monotonicity in the variability decline. This reflects the discreteness of portfolio selection. Forced diversification with low negative covariance (or high positive covariance) will actually increase the variability of the portfolio. As the revenue targets get higher, implied portfolios gradually become more similar and the variability difference decreases smoothly. Eventually, sustainability constraints start to bind for some species, there are fewer ways to meet the revenue target, and the two portfolios must converge. This leads to a sharp drop in the variability difference. For each γ, there is a maximum expected revenue that is feasible, and that is the point where the difference in coefficient of variation hits zero.

Comparing across the γ parameters, at high levels there is more flexibility to substitute across different species because sustainability constraints are less binding. That increases the ability to capitalize on negative covariances, and the difference in variability across portfolio types is thus large. This relationship holds at moderate to high levels of expected revenues. However, at low expected revenues, a lower γ *ceteris paribus* can lead to higher differences in variability. This result again reflects the discreteness of different species entering the portfolio at low revenue levels.

To compare the Species and Ecosystem portfolios further, we look at prescriptions from the two models on a species-by-species basis (Figure 4). Here we depict the share of $c_i^{\max}$ for each species when γ=1 and the revenue target is $126 million. Figure 4 shows that even at moderately high revenue targets when portfolios select the maximum possible harvest for some species, considering covariance leads to different policy implications. Considering covariance in this case, the manager would allocate more catch to oysters, blue crabs, and snails, and less to a wide range of species including menhaden, soft clams, striped bass, Atlantic croaker, sea trout, and horseshoe crabs. To provide further intuition, Table A.1 in the Appendix contains the full correlation matrix for all species. Focusing on pairwise correlations can be misleading in portfolio analysis, but some pairwise results do coincide with portfolio prescriptions. For instance, blue crab is negatively correlated with oysters and positively correlated with menhaden. Because these are the top three revenue-producing species, this implies that the Ecosystem portfolio relative to the Species portfolio blends larger allocations of blue crab and oysters and a smaller allocation to menhaden. It is important to remember that these are comparisons across portfolios; they are not prescriptions relative to actual harvests. We address this issue in the next section.

A.2. Comparison to the Actual

In this section, we ask how much less variance managers could have achieved for the same level of mean revenues by reallocating catch across species. To assess how actual catches compare to efficient mean-variance frontiers, it is first necessary to compute the implicit choice variable for each species in each year, $\hat{c}_{it}$. This is simply the actual catch divided by mean catch,

$\hat{c}_{it} = \bar{x}_i^{-1} x_{it}$. However, for some species-year combinations, the implied $\hat{c}_{it}$ may not have been sustainable given a level of γ. That is, catch for species *k* may have violated the sustainability constraint in year τ and thus $\hat{c}_{k\tau} > c_i^{\max}$. In these cases, it is necessary to compute an adjusted $\hat{c}_{it}$ such that $\tilde{c}_{it} = \begin{cases} \hat{c}_{it}, if\ \hat{c}_{it} \le c_i^{\max} \\ c_i^{\max}, otherwise \end{cases}$. This adjustment ensures that gains associated with moving from the actual to the optimal portfolio are truly attributable to the portfolio approach and not to corrections of unsustainable harvests. The vector of adjusted shares at time *t* is $\tilde{\mathbf{c}}_\mathbf{t}$. We can substitute $\tilde{\mathbf{c}}_\mathbf{t}$ for **c** into (1) and (2) to compute actual portfolio expected revenue and variance.

The results are plotted in Figure 5 along with the three approaches to deriving the efficient frontier and assuming γ=1. We also fit a quadratic regression line to the actual data using Ordinary Least Squares. By construction, the actual choices cannot lie beyond the efficient frontier based on the full covariance matrix.

Since γ=1, the case in Figure 5 allows managers the most flexibility in terms of sustainability constraints. We still find, however, that the actual mean-variance combinations fall significantly short of the frontier. Management with portfolios that consider species interdependencies could have achieved the same revenue levels with much lower risk. This effect is particularly pronounced in the 1960s through early 1980s when relative lack of diversification led to high variability. At lower expected revenues in the 1990s, actual portfolios imply substantially lower variability. This could reflect explicit or implicit attempts to move towards greater diversification. The data points in later years are clustered around the Species frontier that is consistent with management decisions that reduce variability on a species-by-species basis. However, the results fall considerably above the Ecosystem frontier and thus, there appears to be potential benefits from taking into account species interdependencies.

Table 2 examines actual species allocations in four different years and compares them to the optimal portfolio that corresponds to the same level of expected revenue with γ=1. In other words, this table shows how managers could have achieved the same expected revenues with lower risk. For some species, the actual allocation in a year is close to the optimum. For instance, catches of striped bass, perch, and puffers in 1970 are all close to what the corresponding optimal portfolio prescribes. However, other species catches are far from the optimum. Continuing to look at 1970, oyster catch was twice the expected revenue share that the optimal portfolio prescribed and soft clam was more than three times the prescribed revenue share, whereas blue crab catch was just over half of the revenue share that it should have been. In later years, the oyster relationship reverses. In 2000, the prescribed share for oysters was more than twice the actual share of expected revenues.

A different reversal for menhaden occurs over time. In 1970, actual share fell short of the prescribed share, but by 1980 the actual share was more than twice the prescribed share. In 1990 and 2000, the actual share for menhaden was more than three times the prescribed share. The optimal share of blue crab does not vary much over the implied revenue targets in these years and constitutes roughly 40% of the optimal portfolio. The actual share falls short of this level in 1970 but is around 40% in later years. At this juncture, it is important to recall that all historic catches by definition are sustainable when γ=1. This means that mean-variance tradeoffs alone support a lower oyster catch in the early years of the sample and lower menhaden catches later in the sample period.

B. Trophic Aggregation

We can also analyze higher levels of economic or ecological aggregation with the portfolio approach. For example, trophic aggregation may be suitable for exploring the implications of concentrating harvest at particular trophic levels. We illustrate trophic portfolios by aggregating our 22 species into the four categories listed in Table 1. Figure 6 shows optimal revenue shares for two distinct revenue targets under the Species and Ecosystem approaches. In absolute terms, the prescribed share of crabs is similar across approaches, just as the prescribed share of high trophic fishes is similar across approaches. However, for both revenue targets, the prescribed share of oysters, clams, and snails is higher under the Ecosystem approach, whereas the prescribed share of low trophic fishes is lower under the Ecosystem approach. Thus, in relative terms, the ratio of high trophic fishes to low trophic fishes and the ratio of oysters, clams, and snails to crabs are both higher under the Ecosystem approach. The implication is that systematically considering species interdependence through covariances even at high levels of species aggregation leads to different policy implications than ignoring species interdependence.

V. Discussion

As fisheries management moves towards an ecosystem-based approach a systematic means to balance tradeoffs across harvests of different species is needed. In traditional stock assessment models, accounting for ecological and economic interdependencies across species is difficult at best and virtually impossible at worst. Portfolio management provides an alternative to structural modeling that we believe is much simpler to implement and more amenable to considering different objectives. Rather than trying to model ecological, economic, and environmental interactions, portfolio theory treats these processes as generating a multivariate stochastic process. Implementing the portfolio approach thus requires as inputs only a vector of means, a covariance matrix, and a vector of species-level constraints. Perhaps the greatest challenge of ecosystem management for fisheries is to ensure a high flow of value from harvested resources that is sustainable but is not highly variable. The portfolio approach is designed to address these three features of the management problem. A limitation of the portfolio approach, however, is that the policy prescriptions are only as good as the estimates of the means and covariances that characterize the multivariate stochastic process. In the current analysis, we assumed time-invariance of means, variances, and covariances because our goal was to develop and illustrate the portfolio method for ecosystem management. Naturally, the next step is to explore the time series properties of our data. This analysis will allow the vector of means and the covariance matrix to change over time, and thus for a given revenue target, the implied portfolio will change over time as well. A time series of portfolios will also allow us to explore whether there is an economic rationale that supports or argues against "fishing down marine food webs" (Pauly et al. 1998).

Financial portfolio theory provides an alternative and complementary way of viewing a multispecies system. When policy prescriptions from structural models of the ecosystem are consistent with those from the portfolio approach, we can have more confidence in them. When they differ, we have a starting place for asking whether differences are due to true structural breaks or whether they emerge from unmodeled structural features that our simpler approach is able to handle in a reduced-form manner.

References

Arnason, R., “Ecological Fisheries Management Using Individual Transferable Share Quotas,” *Ecological Applications* 8, S151-S159, 1998.

Baldursson, F.M. and G. Magnusson (1997), “Portfolio Fishing,” *Scandinavian Journal of Economics* 99(3), 389-403.

Botsford, Louis W., Castilla, Juan Carlos, Peterson, Charles H. “The Management of Fisheries and Marine Ecosystems” *Science* 1997 277: 509-515

Brodziak, J. and J. Link, “Ecosystem-based Fishery Management: What It Is and How Can We Do It?” *Bulletin of Marine Science* 70, 589-612, 2002.

Charles, A.T., “The Precautionary Approach and ‘Burden of Proof’ Challenges in Fishery Management,” *Bulletin of Marine Science* 70, 683-694, 2002.

Chesapeake Bay Fisheries Ecosystem Plan, (http://noaa.chesapeakebay.net/fepworkshop/netfep.htm).

Chesapeake Bay Commission. 2003. The Blue Crab 2003 Status of the Chesapeake Population and its Fisheries. Chesapeake Bay Commission Blue Crab Technical Workgroup. 16pp.

Christensen, V., and Walters, C. J. Ecopath with Ecosim: methods, capabilities and limitations. *Ecological Modelling, 172:*109-139, 2004.

Clark, C.W., *Mathematical Bioeconomics: The Optimal Management of Renewable Resources,* 2nd Edition, John Wiley & Sons, Inc., New York, 1990.

Clark, C.W. and G.R. Munro, “The economics of fishing and modern capital theory: a simplified approach,” *Journal of Environmental Economics and Management* 2:92-106, 1975.

Darcy, G.H. and G.C. Matlock, “Application of the Precautionary Approach in the National Standard Guidelines for Conservation and Management of Fisheries in the United States,” *ICES Journal of Marine Science 56*, 853-859, 1999.

Edwards, S.F., J.S. Link, and B.P. Rountree, “Portfolio Management of Wild Fish Stocks,” *Ecological Economics* 49:317-329, 2004.

Finnoff, D. and J. Tshchirhart, "Harvesting in an eight-species ecosystem," *Journal of Environmental Economics and Management* 45(3): 589-611, 2003.

Froese, R. and D. Pauly, Editors, FishBase, World Wide Web electronic publication, www.fishbase.org, version (02/2005), 2005.

Garcia, S.M., “The Precautionary Principle: Its Implications in Capture Fisheries Management,” *Ocean and Coastal Management* 22, 99-125, 1994.

Gerrodette, T., P.K. Dayton, S. Macinko, and M.J. Fogarty, “Precautionary Management of Marine Fisheries: Moving Beyond Burden of Proof,” *Bulletin of Marine Science* 70, 657-668, 2002.

Hanna, S., “Institutions for Marine Ecosystems: Economic Incentives and Fishery Management,” *Ecological Applications* 8, S170-S174, 1998.

Hildebrand S.F. and W.C. Schroeder. 1928. *Fishes of the Chesapeake Bay*. United States Bureau of Fisheries Bulletin vol. 53, pt. 1. (reprinted 1972). 388pp.

Hilborn, R., J-J. Maguire, A.M. Parma, A.A. Rosenberg, “The Precautionary Approach and Risk Management: Can They Increase the Probability of Successes in Fishery Management?” *Canadian Journal of Fisheries and Aquatic Sciences* 58, 99-107, 2001. on 98-002.

Houde, E.D, M.J. Fogarty, and T.J. Miller. 1998. Prospects for Multispecies Fisheries Management in Chesapeake Bay: A Workshop. STAC Publication 98-002.

Jackson, Jeremy B. C., Kirby, Michael X., Berger, Wolfgang H., Bjorndal, Karen A., Botsford, Louis W., Bourque, Bruce J., Bradbury, Roger H., Cooke, Richard, Erlandson, Jon, Estes, James A., Hughes, Terence P., Kidwell, Susan, Lange, Carina B., Lenihan, Hunter S., Pandolfi, John M., Peterson, Charles H., Steneck, Robert S., Tegner, Mia J., Warner, Robert R. "Historical Overfishing and the Recent Collapse of Coastal Ecosystems" *Science* 293: 629-637, 2001.

Lauck, T.C., C.W. Clark, M. Mangel, and G.R. Munro (1998), "Implementing the Precautionary Principle in Fisheries Management Through Marine Reserves," *Ecological Applications* 8(1), S72-S78.

Lipcius, R.N., R.D. Seitz, W.J. Goldsborough, M.M. Montane and W.T. Stockhausen. 2001. A deep-water dispersal corridor for adult female blue crabs in Chesapeake Bay. in: *Spatial Processes and Management of Marine Populations* (Eds. Gordon H. Kruse, Nicolas Bez, Anthony Booth, Martin W. Dorn, Sue Hills, Romuald N. Lipcius, Dominique Pelletier, Claude Roy, Stephen J. Smith, and David Witherell), pp. 643-666. University of Alaska Sea Grant, AK-SG-01-02, Fairbanks.

Lipton, D.W. E.F. Lavan and I.E. Strand. 1992. Economics of molluscan introductions and transfers: The Chesapeake Bay dilemma. *Journal of Shellfish Research* 11(2):511-519.

Ludwig, D., "A Quantitative Precautionary Approach," *Bulletin of Marine Science* 70, 485-497, 2002.

Markowitz, Harry (1959), *Portfolio Selection*, New York: John Wiley and Sons.

McAllister, M. and C. Kirchner, "Accounting for Structural Uncertainty to Facilitate Precautionary Fishery Management: Illustration with Namibian Orange Roughy," *Bulletin of Marine Science* 70, 499-540, 2002.

Myers, R.A. and G. Mertz, "The Limits of Exploitation: A Precautionary Approach," *Ecological Applications* 8, S165-S169, 1998.

Paolisso, M. 2002. Blue crabs and controversy on the Chesapeake Bay: A cultural model for understanding watermen's reasoning about blue crab management. *Human Organization* 61(3):2236-239.

Pauly, D., V. Christensen, J. Dalsgaard, R. Froese, and F. Torres, Jr. (1998), "Fishing Down Marine Food Webs," *Science* 279, 860-863.

Peruzzo, L., R.N. Weldon and S.L. Larkin. "Predicting optimal targeting strategies in multispecies fisheries: A portfolio approach," *Marine Resource Economics* 20(1):25-45, 2005.

Pew Oceans Commission. *America's Living Oceans: Charting a Course for Sea Change*. A Report to the Nation. Pew Oceans Commission, Arlington, Virginia. May 2003.

Pikitch, E. K., Santora, C., Babcock, E. A., Bakun, A., Bonfil, R., Conover, D. O., Dayton, P., Doukakis, P., Fluharty, D., Heneman, B., Houde, E. D., Link, J., Livingston, P. A., Mangel, M., McAllister, M. K., Pope, J., Sainsbury, K. J. "ECOLOGY: Ecosystem-Based Fishery Management" *Science* 2004 305: 346-347

Real, L.A., "Animal Choice Behavior and the Evolution of Cognitive Architecture," *Science* 253, 980-986, 1991.

Rosenberg, A.A., "The Precautionary Approach in Application from a Manager's Perspective," *Bulletin of Marine Science* 70, 577-588, 2002.

Sanchirico, J.N. and M.D. Smith, "Trophic Portfolios in Marine Fisheries: A Step Towards Ecosystem Management," Selected Paper, American Agricultural Economics Association

Annual Meetings, Montreal, Canada, July 2003, working paper available at http://agecon.lib.umn.edu/.
Smith J.W. Distribution of Atlantic menhaden, *Brevoortia tyrannus*, purse-seine sets and catches from southern New England to North Carolina, 1985-96. NOAA Technical Report NMFS 144. 22pp. 1999.
U.S. Commission on Ocean Policy. *An Ocean Blueprint for the 21st Century***.** Final Report of the U.S. Commission on Ocean Policy (www.oceancommission.gov), 2004.
Walters, C. (1975), "Optimal Harvest Strategies for Salmon in Relation to Environmental Variability and Uncertain Production Parameters," *Journal of the Fisheries Research Board of Canada* 32, 1777-1784.
Weeks, H and Parker, S, "Scientific and Management Uncertainty Create Competing Precautionary Needs for Fishery Managers," *Fisheries*. 27, no. 3, pp. 25-27. Mar 2002

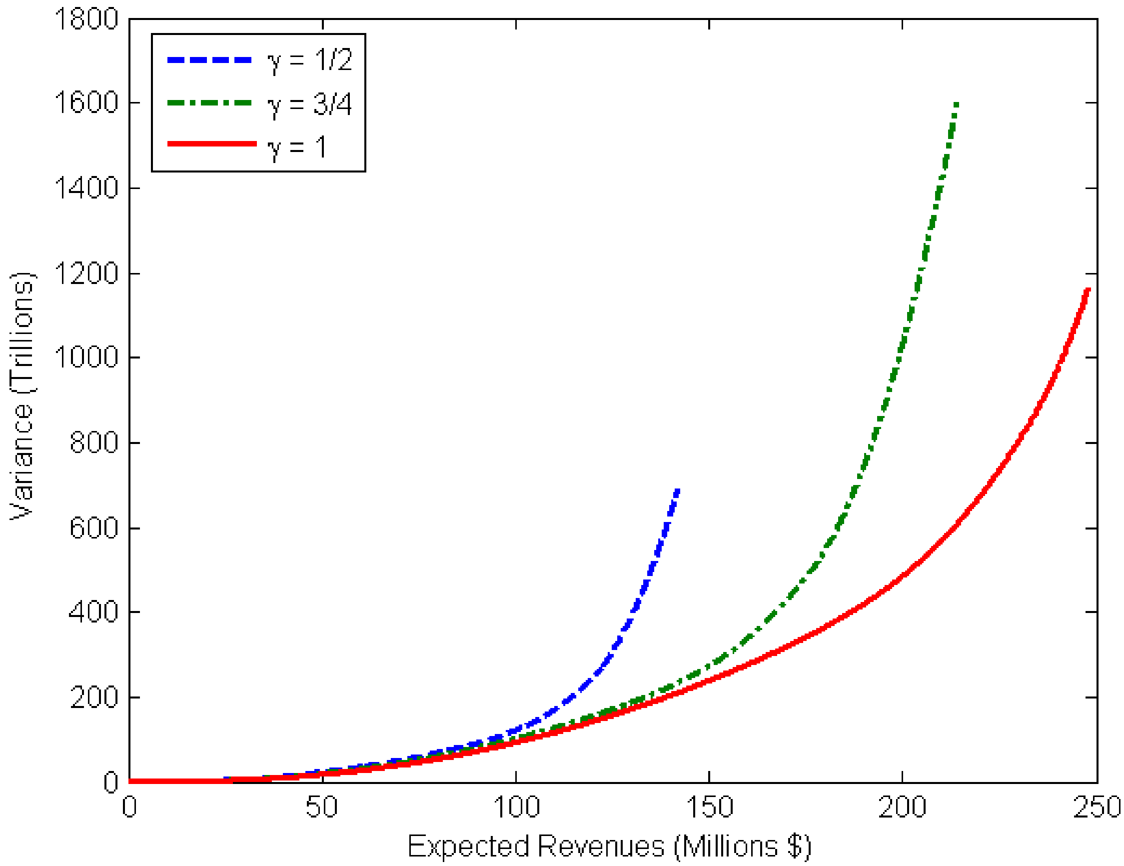

Figure 1: Efficient Frontiers Using the Full Covariance Matrix for Three Levels of Gamma
Note: The parameter γ controls the flexibility that the manager has to substitute across species. When flexibility is high, the manager can achieve the same expected revenue at a lower variance.

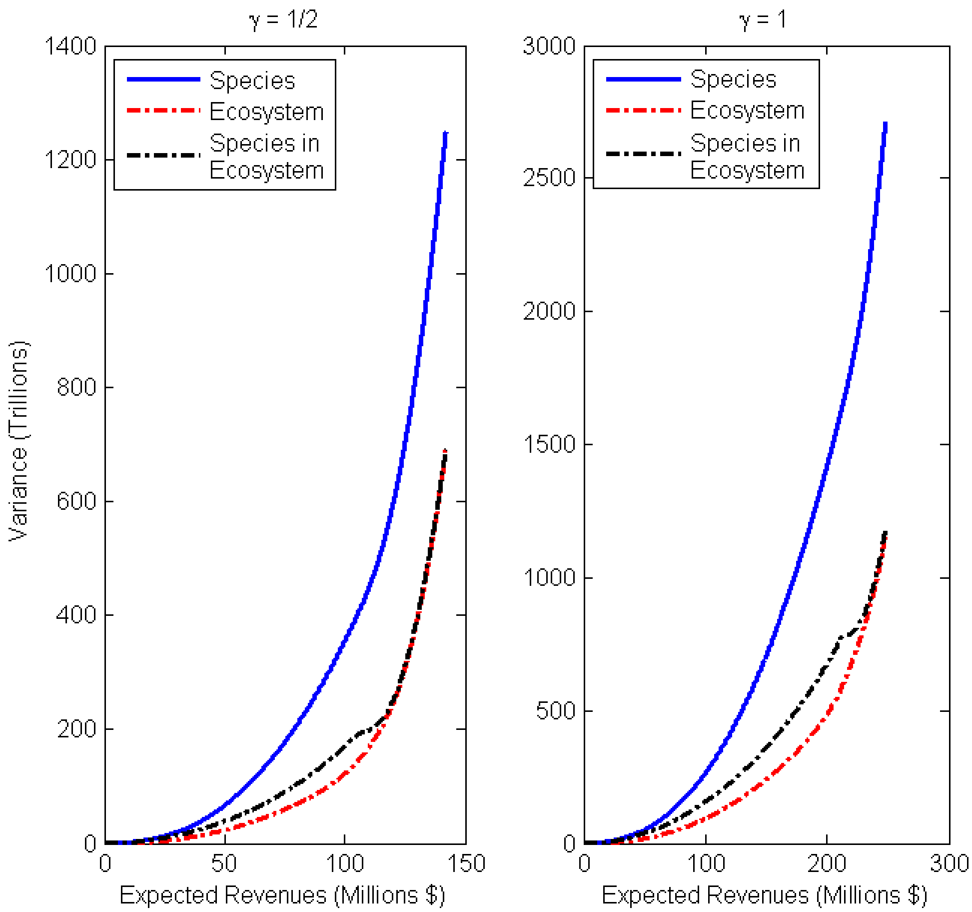

Figure 2: Comparison of Frontiers with and without Species Interdependence
Note: The Species frontier is derived with a diagonal covariance matrix. The Ecosystem frontier is derived with the full covariance matrix that includes non-zero off-diagonal elements. The Species in Ecosystem frontier is derived with the diagonal covariance matrix to generate shares in the portfolio but uses the full covariance matrix to compute variance of the portfolio.

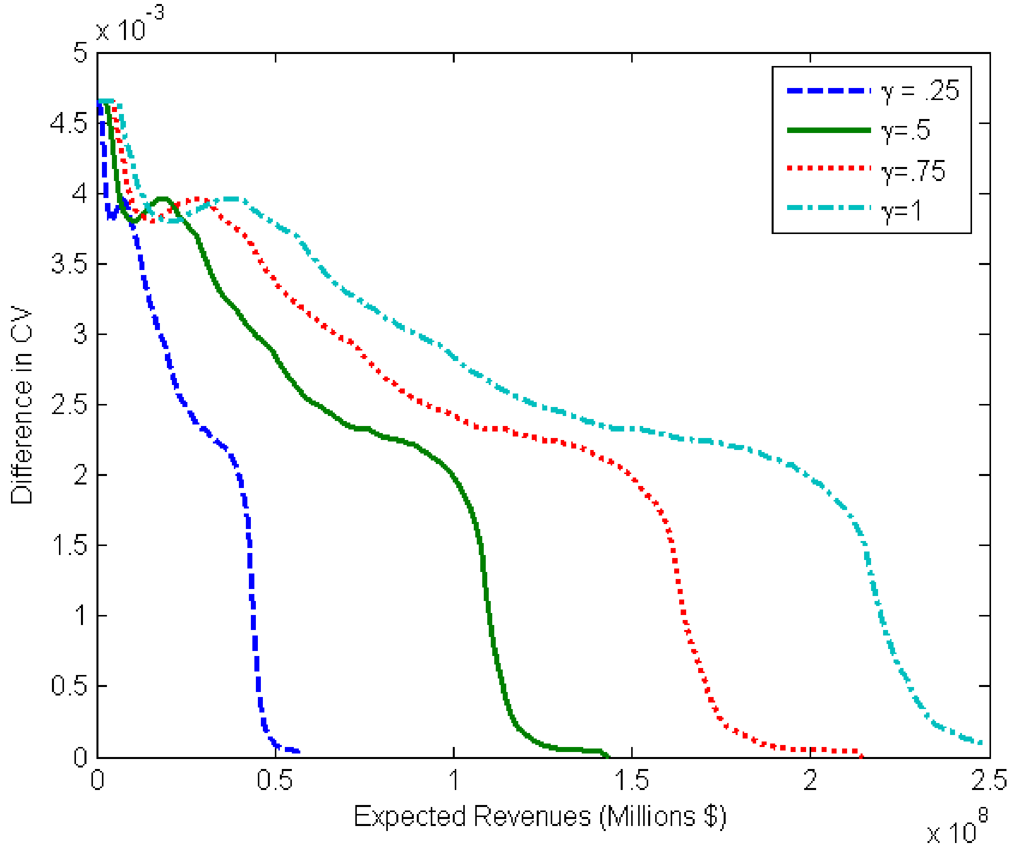

Figure 3: Comparison of Species and Ecosystem Frontiers
Note: The four contours correspond to four levels γ and, for each value of expected revenue, compare differences in coefficient of variation between the Species and Ecosystem frontiers.

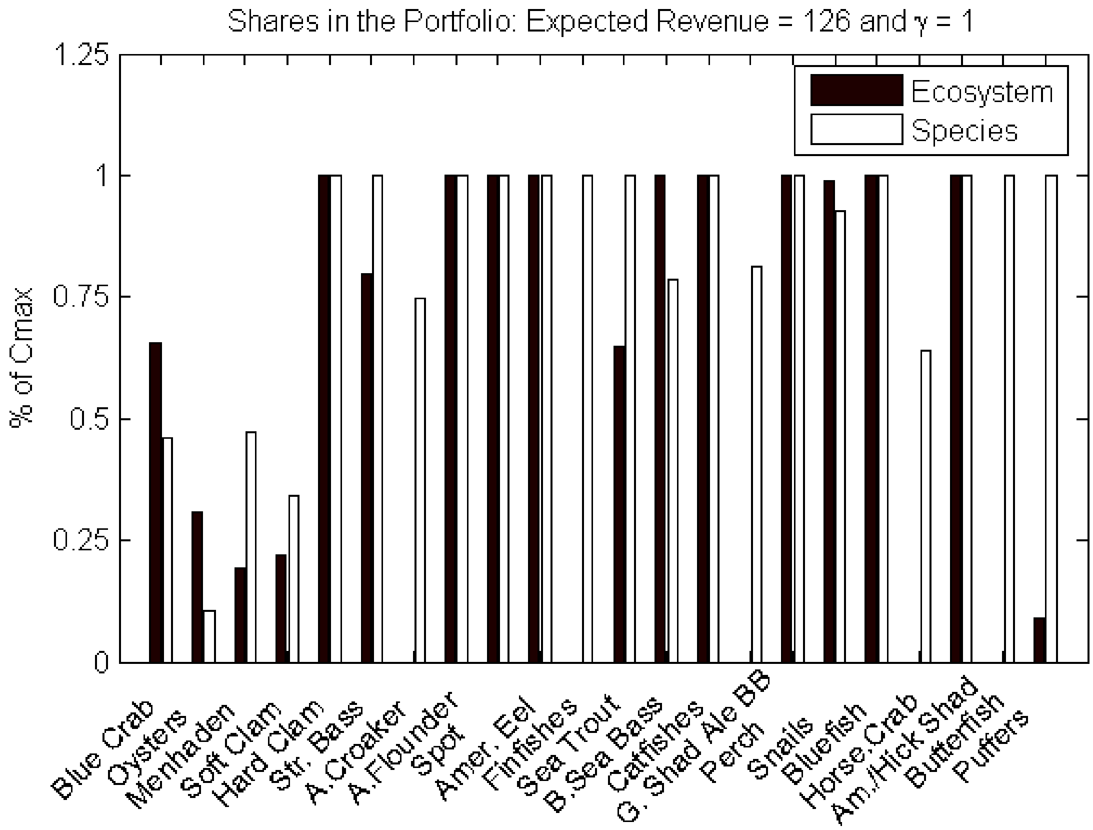

Figure 4: Comparing Ecosystem and Species Portfolios by Species
Note: The Y-axis indicates the share of the maximum allowable allocation for each species (c_i^{max}) that the optimal portfolios prescribe. The black bars are based on the Ecosystem portfolio, while the white bars are based on the Species portfolio.

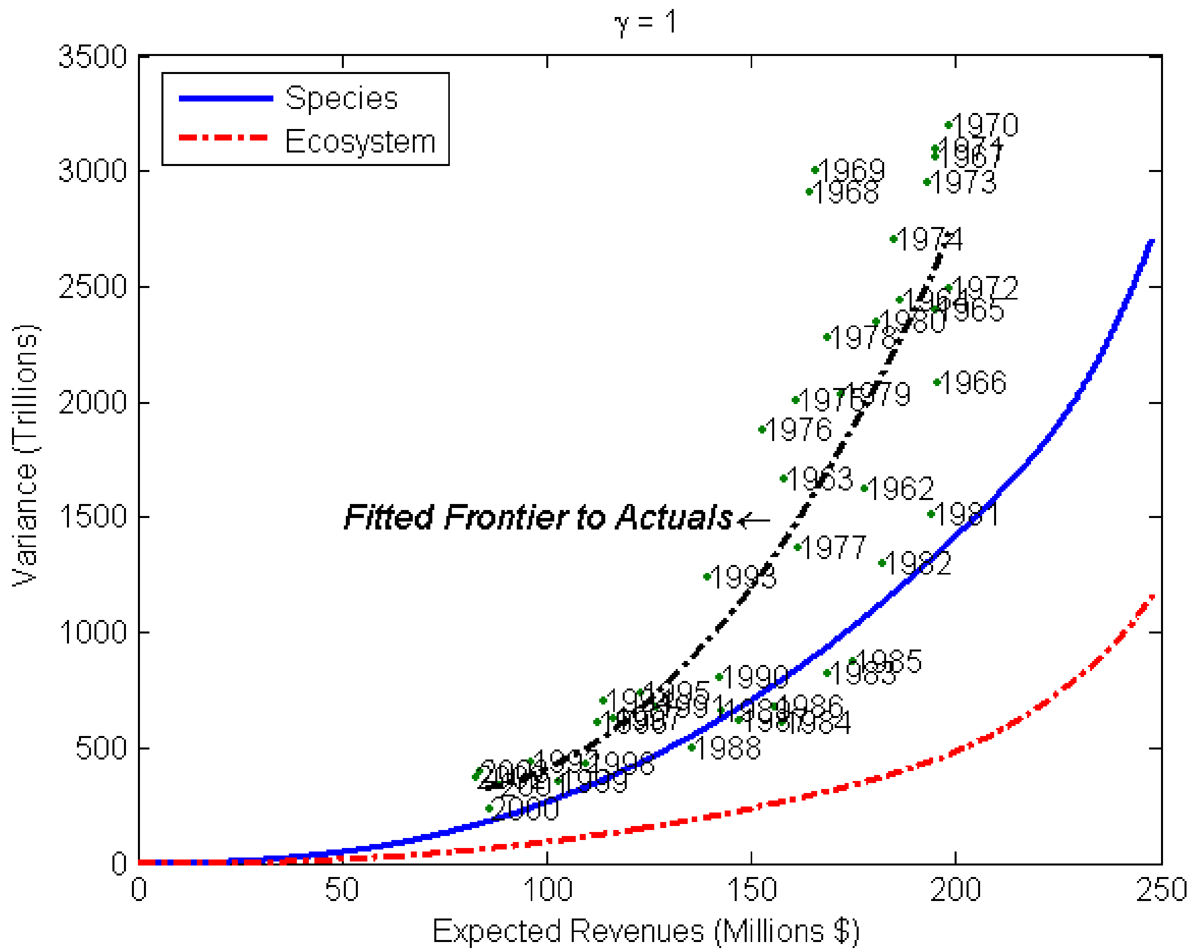

Figure 5: Comparison of Actual Management to Efficient Frontiers
Note: The red line (Ecosystem) is the efficient frontier based on the full covariance matrix that is the true covariance matrix. The blue line (Species) is the efficient frontier based on the diagonal covariance matrix. The black line (Fitted Frontier to Actuals) traces out predicted variances based on a linear regression of actual variance on a quadratic function of expected revenues.

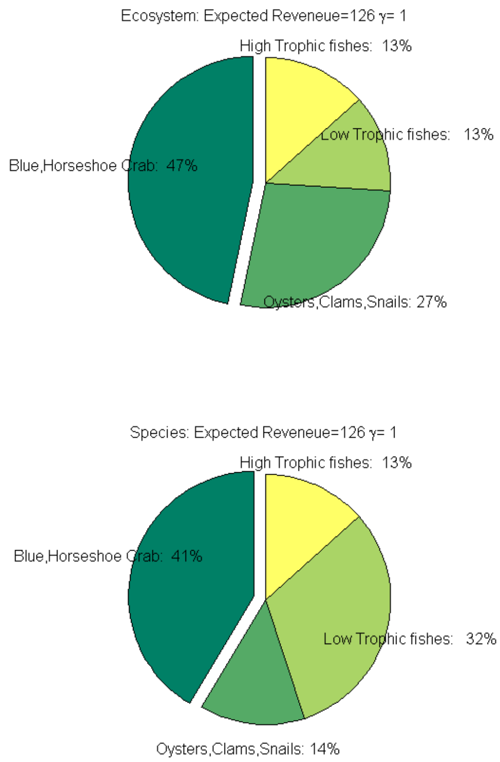

Figure 6: Optimal Aggregated Portfolios under Species and Ecosystem Approaches
Note: Depict shares of expected revenues for each species aggregation prescribed by the optimal portfolio. Aggregations of species categories follow the groupings in Tables 1 and 2. The relative share of high trophic fish to low trophic fish and the relative share of oysters, clams, and snails to crabs are both higher using the Ecosystem approach.

Table 1: Descriptive Statistics for Chesapeake Bay Landings (1962-2003)

	Catch (Pounds)				Revenues (2005 Dollars)			
	Min	**Max**	**Mean**	**St. Dev.**	**Min**	**Max**	**Mean**	**St. Dev.**
Crabs								
Blue Crab	43,971,200	113,111,523	72,120,115	17,611,635	26,181,396	93,439,140	49,274,458	17,199,070
Horseshoe Crab	0	1,039,407	67,160	208,847	0	691,798	32,430	119,460
Oysters, Clams, and Snails								
Oysters	236,504	24,909,400	12,721,553	9,163,801	1,016,199	95,260,469	43,460,285	30,229,311
Soft Clam	0	8,164,300	2,549,562	2,702,696	0	15,746,313	6,320,823	5,015,672
Hard Clam	267,500	1,241,500	717,702	277,121	1,484,380	8,010,947	3,581,591	1,745,468
Snails	3,500	2,970,988	351,003	500,658	5,400	1,623,722	359,083	446,276
Low Trophic Fishes (trophic level < 3.5)								
Menhaden	131,431,900	607,503,000	403,434,890	119,041,386	10,813,757	88,491,182	34,757,703	14,584,282
Perch	543,718	2,804,300	1,376,998	664,364	364,881	2,268,858	1,014,704	445,763
Gizzard Shad, Alewife, Blueback Herring	546,589	38,625,700	9,369,250	12,034,369	89,788	3,581,153	1,069,583	1,178,334
Bluefish	127,100	3,941,300	1,198,068	1,111,131	75,661	1,037,716	363,709	270,176
Finfishes (unc)	48,600	15,411,700	4,592,079	3,976,933	21,348	1,713,745	452,438	431,529
High Trophic Fishes (trophic level > 3.5)								
Striped Bass	0	7,322,700	3,036,487	2,149,974	0	9,198,027	4,281,871	2,310,382
Atlantic Croaker	4,000	12,540,503	3,561,213	4,308,528	4,565	6,162,573	1,510,127	1,581,167
Atlantic Flounder	73,743	608,800	286,144	136,866	112,655	604,469	374,431	124,756
Spot	466,600	5,842,300	2,322,028	1,128,185	260,698	3,095,457	1,316,707	599,599
American Eel	320,600	1,578,200	829,194	333,579	230,834	2,745,016	1,116,068	617,940
Sea Trout	379,812	5,113,500	1,788,303	1,083,012	344,725	2,857,374	1,239,215	620,388
Black Sea Bass	0	530,046	73,709	144,696	0	1,371,924	169,386	361,156
Catfishes and Bullheads	1,307,000	3,890,565	2,208,491	709,644	586,646	1,998,844	1,094,547	304,380
American and Hickory Shad	0	127,100	21,175	28,308	0	45,002	7,830	10,249
Butterfish	12,116	2,101,200	243,103	386,685	8,775	1,054,136	142,554	191,653
Northern Puffer	0	12,118,600	1,099,406	2,552,843	0	1,008,127	153,196	241,942

Table 2: Comparison of Actual Portfolios to Optimal Portfolios

	1970					1980				
	Actual Revenue	Expected Revenue	Share of ER	Share of CMAX	Optimal Port. Share	Actual Revenue	Expected Revenue	Share of ER	Share of CMAX	Optimal Port. Share
Crabs										
Blue Crab	34,403,000	40,135,000	24.186%	97%	44.47%	34,656,000	42,434,000	24.67%	100%	44.41%
Horseshoe Crab	0	0	0.000%	0%	0.00%	49	97	0.00%	0%	0.00%
Oysters, Clams, and Snails										
Oysters	67,295,000	72,163,000	43.486%	43%	21.84%	65,708,000	70,758,000	41.13%	45%	21.95%
Soft Clam	14,461,000	19,609,000	11.817%	28%	3.43%	13,594,000	7,147,200	4.15%	29%	3.43%
Hard Clam	1,924,600	2,915,400	1.757%	100%	3.69%	1,881,700	1,639,800	0.95%	100%	3.56%
Snails	87,631	191,000	0.115%	100%	1.81%	216,250	111,820	0.07%	100%	1.75%
Low Trophic Fishes (trophic level < 3.5)										
Menhaden	10,814,000	11,323,000	6.823%	27%	8.42%	54,323,000	36,772,000	21.38%	29%	8.77%
Perch	2,268,900	2,066,500	1.245%	100%	1.23%	604,760	634,760	0.37%	100%	1.19%
Gizzard Shad, Alewife, Blueback Herring	3,503,500	3,878,800	2.337%	0%	0.00%	242,700	210,270	0.12%	0%	0.00%
Bluefish	148,980	76,836	0.046%	100%	0.71%	692,720	789,640	0.46%	100%	0.69%
Finfishes (unc)	124,570	139,040	0.084%	0%	0.00%	1,191,800	1,226,900	0.71%	0%	0.00%
High Trophic Fishes (trophic level > 3.5)										
Striped Bass	6,154,200	9,214,300	5.553%	100%	6.15%	3,758,000	1,860,800	1.08%	100%	5.93%
Atlantic Croaker	33,849	18,489	0.011%	18%	0.57%	996,730	750,440	0.44%	27%	0.84%
Atlantic Flounder	112,660	216,560	0.131%	100%	0.47%	504,620	566,340	0.33%	100%	0.46%
Spot	649,680	542,610	0.327%	100%	1.97%	1,185,200	1,310,600	0.76%	100%	1.90%
American Eel	859,740	1,445,600	0.871%	100%	1.26%	2,745,000	1,937,200	1.13%	100%	1.22%
Sea Trout	437,950	558,590	0.337%	84%	1.77%	2,758,800	2,992,500	1.74%	84%	1.70%
Black Sea Bass	837	3,677	0.002%	100%	0.72%	154	230	0.00%	100%	0.70%
Catfishes and Bullheads	1,046,300	783,610	0.472%	100%	1.15%	778,970	756,700	0.44%	100%	1.11%
American and Hickory Shad	23,919	47,000	0.028%	100%	0.03%	770	666	0.00%	100%	0.03%
Butterfish	88,106	91,419	0.055%	0%	0.00%	173,710	124,200	0.07%	0%	0.00%
Puffers	615,100	526,350	0.317%	30%	0.30%	0	0	0.00%	38%	0.37%
Total	**145,053,522**	**165,945,781**	**100.00%**		**100.00%**	**186,012,933**	**172,024,162**	**100.00%**		**100.00%**

	1990					2000				
	Actual Revenue	Expected Revenue	Share of ER	Share of CMAX	Optimal Port. Share	Actual Revenue	Expected Revenue	Share of ER	Share of CMAX	Optimal Port. Share
Crabs										
Blue Crab	61,720,000	59,656,000	41.81%	78%	42.48%	72,290,000	43,760,000	42.56%	52%	39.37%
Horseshoe Crab	6,206	10,720	0.01%	0%	0.00%	26,826	21,544	0.02%	0%	0.00%
Oysters, Clams, and Snails										
Oysters	19,918,000	13,703,000	9.60%	35%	20.86%	9,234,200	9,494,100	9.23%	26%	21.34%
Soft Clam	15,746,000	9,988,100	7.00%	26%	3.74%	0	0	0.00%	16%	3.27%
Hard Clam	8,010,900	5,444,500	3.82%	100%	4.36%	2,813,800	1,788,200	1.74%	100%	6.07%
Snails	234,030	298,110	0.21%	100%	2.14%	988,630	568,950	0.55%	76%	2.27%
Low Trophic Fishes (trophic level < 3.5)										
Menhaden	39,996,000	47,377,000	33.20%	22%	8.19%	30,029,000	28,209,000	27.43%	15%	7.54%
Perch	688,510	569,690	0.40%	100%	1.46%	1,389,900	1,362,500	1.33%	100%	2.03%
Gizzard Shad, Alewife, Blueback Herring	254,150	178,760	0.13%	0%	0.00%	1,098,700	286,410	0.28%	0%	0.00%
Bluefish	190,900	166,700	0.12%	100%	0.84%	135,380	122,560	0.12%	100%	1.17%
Finfishes (unc)	298,030	235,410	0.16%	0%	0.00%	333,190	434,490	0.42%	0%	0.00%
High Trophic Fishes (trophic level > 3.5)										
Striped Bass	0	0	0.00%	98%	7.16%	6,763,000	5,103,300	4.96%	63%	6.41%
Atlantic Croaker	724,500	353,490	0.25%	0%	0.00%	4,003,400	5,109,900	4.97%	0%	0.00%
Atlantic Flounder	260,050	148,260	0.10%	100%	0.56%	455,750	257,260	0.25%	100%	0.78%
Spot	1,644,400	1,380,000	0.97%	100%	2.33%	1,163,300	1,621,300	1.58%	77%	2.50%
American Eel	1,723,100	1,071,700	0.75%	100%	1.50%	610,170	803,060	0.78%	100%	2.08%
Sea Trout	1,157,600	602,460	0.42%	77%	1.92%	1,081,100	1,056,200	1.03%	56%	1.96%
Black Sea Bass	8,821	22,521	0.02%	100%	0.86%	791,970	905,070	0.88%	100%	1.19%
Catfishes and Bullheads	1,221,500	1,410,900	0.99%	100%	1.36%	1,303,100	1,846,000	1.80%	100%	1.89%
American and Hickory Shad	153	74	0.00%	100%	0.03%	901	1,123	0.00%	100%	0.05%
Butterfish	65,962	57,115	0.04%	0%	0.00%	81,281	65,644	0.06%	0%	0.00%
Puffers	160,920	24,636	0.02%	18%	0.21%	23,833	5,882	0.01%	4%	0.06%
Total	**154,029,732**	**142,699,146**	**100.00%**		**100.00%**	**134,617,431**	**102,822,493**	**100.00%**		**100.00%**

Note: Actual Revenue is the revenue from landings in the corresponding year-species combination reported in our data set. Expected revenue uses actual catch to infer an implied portfolio share ($\tilde{c}_{it}$) and then multiplies this share by average revenue for the species μ_i. Share of ER is the species-level expected revenue divided by total expected revenue summed across all 22 species. Share of CMAX indicates the share of the maximum allowable allocation for each species (c_i^{max}) that the optimal Ecosystem portfolio prescribes given that year's total expected revenue. Optimal Port. Share for each species is the percentage of total expected revenue in the optimal portfolio generated by that species.

Table A.1: Species-level Revenue Correlations

	Blue Crab	Eastern Oyster	Menhaden	Soft Clam	Hard Clam	Striped Bass	Atlantic Croaker	Atlantic Flounder	Spot	American Eel	Unclassified Finfishes	Sea Trout	Black Sea Bass	Catfishes and Bullheads	Gizzard Shad, Alewife, & Blueback Herring	Perch	Snails	Bluefish	Horseshoe Crab	American & Hickory Shad	Butterfish	Puffers
Blue Crab	1.00																					
Eastern Oyster	-0.82	1.00																				
Menhaden	0.06	-0.01	1.00																			
Soft Clam	-0.56	0.69	-0.05	1.00																		
Hard Clam	0.56	-0.50	0.13	-0.20	1.00																	
Striped Bass	-0.25	0.23	-0.20	-0.14	-0.63	1.00																
Atlantic Croaker	0.54	-0.62	-0.20	-0.50	-0.05	0.34	1.00															
Atlantic Flounder	0.20	-0.18	0.21	-0.18	-0.11	0.24	0.46	1.00														
Spot	0.11	-0.09	-0.14	-0.12	0.13	0.13	0.30	0.13	1.00													
American Eel	-0.04	-0.06	0.14	0.21	0.34	-0.48	-0.27	-0.01	-0.18	1.00												
Unclassified Finfishes	0.24	-0.18	0.29	-0.08	-0.07	0.17	0.38	0.34	-0.02	0.13	1.00											
Sea Trout	-0.19	0.24	0.74	0.27	-0.16	-0.13	-0.22	0.26	-0.23	0.25	0.37	1.00										
Black Sea Bass	0.31	-0.53	-0.29	-0.52	-0.17	0.40	0.75	0.26	0.29	-0.34	-0.08	-0.34	1.00									
Catfishes and Bullheads	0.39	-0.48	-0.06	-0.41	0.19	0.16	0.52	0.09	0.02	-0.20	0.37	-0.22	0.40	1.00								
Gizzard Shad, Alewife, & Blueback Herring	-0.54	0.70	-0.44	0.40	-0.32	0.44	-0.37	-0.23	0.13	-0.33	-0.34	-0.38	-0.23	-0.17	1.00							
Perch	-0.02	0.01	-0.45	-0.03	-0.13	0.44	0.12	-0.20	0.06	-0.20	-0.10	-0.57	0.16	0.21	0.50	1.00						
Snails	0.58	-0.67	-0.29	-0.58	0.15	0.16	0.73	0.20	0.33	-0.16	0.00	-0.37	0.70	0.28	-0.34	0.13	1.00					
Bluefish	-0.45	0.43	0.53	0.30	-0.28	0.02	-0.31	0.09	-0.31	0.20	0.21	0.71	-0.37	-0.23	-0.11	-0.56	-0.47	1.00				
Horseshoe Crab	0.36	-0.31	-0.03	-0.25	-0.01	0.21	0.47	0.35	0.15	-0.09	0.13	-0.20	0.37	0.29	-0.12	0.17	0.10	-0.21	1.00			
American & Hickory Shad	-0.51	0.48	0.03	0.09	-0.29	0.52	-0.27	-0.06	0.13	-0.19	-0.08	-0.05	-0.10	0.02	0.59	0.20	-0.28	0.29	-0.16	1.00		
Butterfish	-0.28	0.52	-0.23	0.25	-0.13	0.30	-0.17	0.02	0.16	-0.24	-0.19	-0.18	-0.22	-0.21	0.70	0.33	-0.17	-0.16	-0.11	0.32	1.00	
Puffers	-0.33	0.48	-0.43	0.34	-0.04	0.10	-0.35	-0.19	0.17	-0.13	-0.42	-0.45	-0.21	-0.30	0.76	0.48	-0.21	-0.40	-0.15	0.26	0.69	1.00