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ANALYSIS

An empirical approach to ecosystem-based fishery management

James N. Sanchirico[a,*], Martin D. Smith[b,1], Douglas W. Lipton[c]

[a]Quality of Environment Division, Resources for the Future, 1616 P Street NW, Washington DC 20036, United States
[b]Nicholas School of the Environment and Earth Sciences, Duke University, Box 90328, Durham, NC 27708, United States
[c]Department of Agricultural and Resource Economics, University of Maryland, College Park, MD 20742, United States

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ABSTRACT

Marine scientists and policymakers are encouraging ecosystem-based fishery management (EBFM), but there is limited guidance on how to operationalize the concept. We adapt financial portfolio theory as a method for EBFM that accounts for species interdependencies, uncertainty, and sustainability constraints. Illustrating our method with routinely collected data available from the Chesapeake Bay, we demonstrate the gains from taking into account variances and covariances of gross fishing revenues in setting species total allowable catches. We find over the period from 1962–2003 that managers could have increased the revenues from fishing and reduced the variance by employing EBFM frontiers in setting catch levels.

1. 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; Hilborn et al., 2001; Charles, 2002; Ludwig, 2002; Weeks and Parker, 2002). In the short-term, many argue that managers should address the inherent risks in complex ecosystems by forming large-scale protected areas as a form of insurance, where the event to insure against is a stock collapse. 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). Ecosystem-based fishery management (EBFM) requires recognition of system component interactions in determining management targets. Some argue that EBFM has the potential to account for risks inherent in managing interacting populations in uncertain and changing environments (Hofmann and Powell, 1998), while others directly equate EBFM with taking a precautionary approach (Essington, 2001; Gerrodette et al., 2002).

Although there is considerable discussion about precautionary management and EBFM, there is little guidance on how to operationalize these two ideas together in fisheries management. Considering precaution and EBFM together charges fishery managers to develop quantitative management tools with three features: 1) the objective function must account for risk preferences, 2) constraints (or state equations) must represent system interactions and uncertainty, and 3) decisions must rely on existing or easily collected data.

We argue that financial portfolio theory provides a foundation for considering all of these features. We build on a

* Corresponding author.
E-mail addresses: sanchirico@rff.org (J.N. Sanchirico), marsmith@duke.edu (M.D. Smith), dlipton@arec.umd.edu (D.W. Lipton).
[1] Tel.: +1 919 613 8028.

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conceptual work that proposes portfolio management for multispecies fisheries (Edwards et al., 2004), and we illustrate the method empirically for the Chesapeake Bay ecosystem. Portfolio methodology does not replace the standard approach for dealing with uncertainty in renewable resource management or analyses for addressing species interactions. Innovations in conventional deterministic and stochastic bioeconomic modeling are also likely to contribute to tools for EBFM. Instead, our approach is a complement to existing models and management of renewable resource systems.

The standard approach to incorporating uncertainty in renewable resource management explicitly relies on risk neutrality of the social planner. The Reed (1979) model maximizes expected rents from a stochastically evolving renewable resource stock. The solution for the single species case is constant escapement, and deriving this solution relies on knowledge of the stock dynamics as well as state separability. The constant escapement policy implies periods for which the fishery is shut down entirely. This policy, in turn, induces intertemporal variability in fishing returns that may be costly to a risk averse harvest sector. However, incorporating risk aversion into the objective function undermines the state separability assumption that is required to derive the constant escapement solution.

The standard approach to dealing with system interactions is to build structural models of the ecosystem within which total allowable catch (TAC) for each harvested species can be determined. Traditional bioeconomic models propose optimal control of interacting species from predator–prey foundations (Quirk and Smith, 1970; Hannesson, 1983; Wilen and Brown, 1986). In the ecology literature, ecological network analysis models such as Ecopath with Ecosim are used to develop structural ecosystem models that that can optimize various objectives and set TACs across multiple species (Dame and Christian, 2006; Christensen and Walters, 2004). Other advances in food web models for marine systems make structural modeling more plausible for guiding EBFM (Bascompte et al., 2005; Finnoff and Tshchirhart, 2003). However, these models are still costly to develop, data intensive, and fraught with uncertainties regarding species interactions, effects of fishing, and environmental factors (Botsford et al., 1997; Pikitch et al., 2004; Fulton et al., 2003). Moreover, even in single-species bioeconomic models, the optimal policy can be sensitive to small differences in the biological or economic parameters (Clark, 1990). Adding more species to models simply increases the difficulty of making qualitatively robust policy prescriptions.

As a complement to constant escapement and structural ecosystem models, we propose a flexible empirical approach to EBFM that minimizes the variability in the system due to fishing subject to meeting a target level of fishing return. Because less stable systems can lead to lower diversity and maintaining diversity is desirable (Worm et al., 2005), our objective is consistent with the idea that managing ecosystems to improve stability (or reduce variance) can be ecologically and economically beneficial (Roughgarden and Smith, 1996; Armsworth and Roughgarden, 2003).

Our approach is loosely based on techniques employed in financial asset management. Prior to the development of portfolio theory (Markowitz, 1952), investors focused on the risk (variance) and rewards (expected returns) of *individual* securities in developing a portfolio—similar to how fisheries are managed today (Pikitch et al., 2004). Portfolio theory changed the perspective from choosing individual stocks to portfolios, where taking into account the correlations across securities could reduce risks (reduce variability) for a given level of return. Similarly, species interdependencies mean that risks from harvesting each species are correlated—whether positively or negatively—and because of this correlation, there are potential benefits from considering multiple fish stocks jointly (Essington et al., 2006).

Using historical fishing data from the Chesapeake Bay (hereafter "Bay"), we derive an EBFM frontier that is an explicit representation of the tradeoffs between risk and return. A point on the frontier maps into a set of total allowable catches (TACs) for each species in the portfolio. Setting TAC levels simultaneously for a set of species taking into account the interrelationships is consistent with one of the stated objectives of EBFM (Arnason, 1998; Hanna, 1998; Pikitch et al., 2004).

Though the foundation for this idea is not new to fisheries biology (Walters, 1975; Hilborn et al., 2001), ecology (Real, 1991) or fishery economics (Baldursson and Magnusson, 1997; Arnason, 1998; Hanna, 1998; Edwards et al., 2004; Perusso et al., 2005), our contribution is taking these theoretical ideas and applying them in a particular ecosystem, which is not a trivial exercise. Walters (1975) derives a mean-variance catch frontier for a single species. The first economics papers on portfolio fishing develop conceptual models to allocate fishing across young and old cohorts of the same species (Baldursson and Magnusson, 1997) and, similar to our study, to allocate catches across multiple species (Edwards et al., 2004). Previous empirical work on fishing portfolios includes work on the northwest Atlantic (Sanchirico and Smith, 2003) and Perusso et al. (2005), who use a portfolio framework in a positive analysis to predict individual fishing vessel decisions. Furthermore, our analysis is timely and relevant for policy makers in the Bay, as they recently adopted an EBFM plan that calls for an examination of patterns of removals (*i.e.*, catches) as well as characterizing and incorporating uncertainty into fisheries management decisions. There is also parallel work underway in marine ecology to develop a Bay ECOPATH with ECOSIM model (Christensen and Walters, 2004).

Along with presenting a potential tool for EBFM, our analysis illustrates the importance of taking a broader perspective than just one species at a time for fisheries management. In particular, we show the advantages of utilizing information on species interactions (as embodied in covariances) by comparing an EBFM frontier to a species frontier, which is analogous to a single-species management approach.[2] We also compare the actual catches for each species in the Bay for each year from 1976–2003 to the implied catches from the EBFM frontier.

[2] In our analysis, we focus on species in the Bay where commercial catch and price information are available. In one way, this limits our analysis in that we are not considering explicitly the non-commercially harvested species that might be an important part of EBFM in practice. In theory, however, this methodology is not limited and if information on stock sizes and/or recreational values were available for this broader set of species they could be incorporated into the analysis.

2. Methods

Any analysis of management options at a scale greater than a species confronts the following questions: How to define the boundaries of the analysis and which components to include? Should the scale of analysis be at trophic level or functional group? Should the objective of management be based on ecological criteria (Pikitch et al., 2004), such as species contribution to ecosystem biomass or productivity or on socioeconomic criteria, such as species contribution to fishing profits or social welfare? Because our approach is empirical, resolution of these issues depends on the availability of data.

We illustrate our approach to EBFM at the species-level with an objective based on fishing revenues. While admittedly not the best measure of the value of an ecosystem, revenues do provide a common metric to trade off fishery value across species, and variability in revenues does have economic and ecological costs. Unfortunately, the time series data for ecosystem stability (species-level population estimates) or social welfare (species-level contributions to total economic value), both of which are more appropriate objectives, are not available. Gross fishery revenues, in contrast, are routinely collected by managers, and mean revenues signal determinants of management objectives, such as fish stock size, employment, and returns to fishing. Similarly, variance in revenues (volatility) is costly to individual fishermen—who may have boat and home mortgage payments but limited income outside of fishing. By extension, volatility may be harmful to fishing-dependent communities. Revenue volatility also potentially harms the processing sector by increasing the riskiness of capital investments, and revenue volatility can indicate variability in fish populations.[3]

If the data were available, fishery rents or changes in the asset values of fish stocks would be a more appropriate economic measure of returns than gross revenues. In a fishery managed with individually transferable quotas (ITQ), one could calculate expected returns and variances from quota prices. This approach would essentially be an extension of Arnason (1998) to account for volatility and not just the mean returns. Short of having an ITQ fishery, one could collect data on cost per unit of catch by species and value of fishing capital. One would then combine this information with gross revenues to construct a measure of rents. The drawback of using gross revenues in isolation is that the correlation with rents (or asset values) will depend on other fishery management institutions beyond TACs, such as whether there is limited entry. Nevertheless, using gross revenues is empirically tractable and provides an illustration of our approach with routinely collected data in fisheries. Collecting data to implement portfolio management for different metrics of returns is important for future work.

[3] Because both ecological and economic variability have costs and humans are part of EBFM plans, we believe that using a measure of value rather than just a biological measure for determining the species interactions is appropriate. Empirically, this means that our covariances are just as likely to stem from economic or management actions than ecological interactions. This is not a problem, because the structural factors leading to the covariances are not being modeled, as is the case in financial portfolio analysis.

Let $\boldsymbol{\mu}(t)$ be an $(n\times1)$ vector of expected revenue of the n harvestable species in the ecosystem in period t and $\boldsymbol{\Sigma}(t)$ be the $(n\times n)$ matrix of covariances in revenues in period t. Because substitute protein sources and world seafood markets exist, we assume fish prices are exogenous, *i.e.* unresponsive to changes in ecosystem-wide catch levels, though prices do change over time. In other words, we assume that the outcomes of management with EBFM frontiers do not to generate own- or cross-price feedbacks within the system. The appropriateness of this assumption, of course, will vary from ecosystem to ecosystem. Correlations between species revenues can be negative or positive depending on the relative strength of trophic interactions, environmental fluctuations, and fishing intensity and gear choices that determine fish stocks and corresponding catch rates, as well as output market interactions that affect prices. Our measures of the expected revenues and covariances change over time in accordance with recent research that discusses how fishing, pollution, and environmental forces have led to structural changes in marine ecosystems (Jackson et al., 2001; Pauly et al., 1998).

We apply the value-at-risk methodology popularized by J.P. Morgan in their RiskMetrics VaR model (J.P. Morgan/Reuters, 1996). The technique uses exponential smoothing where the influence of years far in the past on the current calculations diminishes at a rate equal to the decay factor (λ). In particular, the i, j^{th} element of the variance/covariance matrix of revenues (r) in period t is equal to

$$\Sigma_{i,j}(t)=\frac{\sum_{k=0}^{t-1}\lambda^{t-k}(r_{j,t-k}-\bar{r}_i(t))^2}{\sum_{k=0}^{t-1}\lambda^{t-k}}\quad\text{where }\bar{r}_i(t)=\frac{\sum_{k=0}^{t-1}\lambda^{t-k}r_{i,t-k}}{\sum_{k=0}^{t-1}\lambda^{t-k}}.\tag{1}$$

When the decay factor is equal to .741, five percent of the weight remains after ten years, and if the factor is .549, five percent of the weight remains after five years. With $\lambda=1$, each period receives equal weight.

Application of portfolio methods for making fishery management decisions would require more careful modeling of the time series properties of the data. Depending on the availability of sufficiently long time series, one could model expected revenues using vector autoregresion (Sims, 1980), changing variances using conditional heteroskedasticity (Bollerslev, 1986), and changing sustainability constraints using fishery-independent data or cointegration (Engle and Granger, 1987). For expositional reasons, the value-at-risk methodology is sufficient.

Let $c_i(t)$ be the revenue weights chosen by the manager for species i in period t. For a set of weights $\mathbf{c}(t)$ (an $n\times1$ vector), total expected ecosystem revenue is $\mathbf{c}(t)'\boldsymbol{\mu}(t)$, and the ecosystem variance of revenue is $\mathbf{c}(t)'\boldsymbol{\Sigma}(t)\mathbf{c}(t)$ in period t. Formally, we derive a mean–variance frontier in period t by solving the quadratic programming problem:

$$\min_{\mathbf{c}_i(t)}\mathbf{c}(t)'\boldsymbol{\Sigma}(t)\mathbf{c}(t)\ \ s\cdot t\cdot\ \mathbf{c}(t)'\boldsymbol{\mu}(t)\geq M(t),\tag{2}$$

where $M(t)$ is a target level of ecosystem revenues in period t. The formulation in Eq. (2) follows the approach in Sanchirico and Smith (2003) and differs from Edwards et al. (2004) in that it is cast solely in terms of observables. For any feasible $M(t)$, we can find the revenue weights that minimize the total variance from the ecosystem. Solving the quadratic program in Eq. (1) is consistent with a quadratic utility function that balances mean returns with variance. The manager's risk tolerance will determine which point on the frontier is chosen and hence the corresponding $M(t)$. For any risk-averse manager, it is always optimal to minimize risk for a given level of expected return. A corner solution is possible if the expected return is the maximum possible in the system.

While the construction of EBFM frontiers uses the same architecture as in finance, a couple of issues arise when applying the quadratic programming problem (Eq. (2)) 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. 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.

To ensure that each weight is feasible, we append sustainability constraints to Eq. (2) that impose upper bounds on the $c_i(t)$'s. Because the revenue weights are non-negative, the full programming problem is Eq. (2) with $0 \le c_i(t) \le c_i^{\max}(t), \forall i,t$. These upper limits ensure that the implied revenues are within the physical limits of the system; that is, the weights do not imply catches that exceed the current standing stock (or some allowable fraction thereof) at current prices.[4] Formally, for each species i, the upper portion of the constraint ($c_i(t) \le c_i^{\max}(t)$) is

$$c_i(t)\Omega_i(t) \le \gamma_i(t) * B_i(t), \tag{3}$$

where $B_i(t)$is the stock level in period t, $\gamma_i(t)$ is the sustainability parameter in period t (or the fraction of the standing stock susceptible to catch in period t), and $\Omega_i(t)$ is a weighted average of catches (see Appendix for derivation). $\gamma_i(t) * B_i(t)$ is the *ex ante* maximum sustainable catch for species in period t, which can be thought of as just offsetting the biological growth in the period such that the stock size remains constant. Of course, the *ex post* ecosystem catch can differ from this amount, because we are choosing the share (c_i's) that then determines the yield in the period. By assumption, we build in a precautionary buffer explicitly into the formulation by ensuring that the ex post catch level is always less than or equal to the *ex ante* levels. Rearranging this constraint, we have

$$c_i(t) \le c_i^{\max}(t) \equiv \gamma_i(t) * B_i(t)/\Omega_i(t). \tag{4}$$

Another formulation is to replace $B_i(t)$ with the maximum sustainable yield, where $\gamma_i(t)$ could exceed one for an under-exploited species and be less than one for a period of time to allow recovery for an overexploited species. In many fisheries, managers set total allowable catches without the availability of stock assessments (Annala, 1996; NOAA Fisheries, 2006). In this case, an approximation to MSY can be used that is based on a time-series of observed catches.

In our analysis, we treat the sustainability parameter as an exogenous choice to the ecosystem manager. In each period, the ecosystem manager is choosing the portfolio of catches subject to meeting the period's revenue target and sustainability parameter. A more general approach would allow the sustainability parameters to be chosen endogenously over time with forward-looking dynamic feedbacks incorporated into the analysis. Changes over time are incorporated, however, into the analysis, as the constraint is recalibrated over time based on historic catch levels (exponential smoothing with same weights as those used in the variance–covariance estimation).

3. Empirical application and data

A portfolio approach to Bay fisheries management is appropriate to the nature of commercial fishing in the region. 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. A description of the fisheries in 1920 remains a fairly accurate representation of current species fished and gears used for finfish (Hildebrand and Schroeder, 1928). The predominant finfishing gears are pound nets, seines, and gillnets. Bottom trawls are generally not allowed in the Bay. Water temperature and species migration patterns determine the seasonality of the catch with fishing activity beginning earlier in the season in the Virginia portion of the Bay when anadromous river herrings and shad return to spawn in early spring. Similarly, blue crabs emerge from their winter hibernation and begin being caught in the late spring as the Bay water's warm in a south to north pattern (Lipcius et al., 2001). 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 a 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 species composition of harvests from 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,

[4] A referee raised the point that because of our data limitations on stock sizes, it is possible that the c_i in our demonstration of the methodology could be based on historical catches that were not sustainable. The constraint, however, permits the c_i to be adjusted to build in any degree of precaution. For example, $\gamma_i(t)$ could be equal to .25, which would imply that the manager is restricting this period's catch to be less than or equal to 25% of the maximum catch rate observed.

blue crab, and striped bass. Several events led to a decline. A significant factor in the decline of Bay oyster harvest volume post-1950 was the outbreak of the MSX parasite (*Haplosporidium nesloni*), mostly in Virginia oysters, around 1960 (Lipton et al., 1992). 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 the 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 Bay, is virtually non-existent.

Striped bass catches 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.

Blue crab was not a major fishery and income producer for watermen in the 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, 2006).

Management of 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. The Mid-Atlantic Fisheries Management Council manages species that are principally caught in Federal waters.

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. For example, the latest update to the Chesapeake Bay Agreement (http://dnrweb.dnr.state.md.us/bay/res_protect/c2k/index.asp) includes a goal to develop EBFM plans for target species by the end of 2005 and the current Chesapeake Bay Fisheries Ecosystem Plan (2006) specifically calls for examining patterns of harvests as well as incorporating uncertainty into fisheries management decisions. Preliminary work has begun on developing an ecosystem based fisheries management plan for striped bass, with future plans to include blue crab and menhaden.

The data on commercial Chesapeake catches from 1962–2003 are readily available from the National Marine Fisheries Service and combine all Maryland and Virginia harvests, including offshore landings. For this analysis, Bay catches were extracted from the raw data files based on the indicator of where the catches occurred. Menhaden are by far the largest catch by volume from the Bay. Menhaden catches for 1985–1996 were obtained from Smith (1999). Estimates of menhaden catches from 1997–2003 and pre-1985 data were obtained from Joseph W. Smith. (NOAA Fisheries, Beaufort, NC, personal communication). We select species to include in the analysis based on the criteria that the species generates at least $500,000 dollars in real dockside revenues (measured in 2005 dollars) in at least one year. One limitation of our analysis is that we do not have historical data on recreational catches.

Table 1 – Descriptive statistics for Chesapeake Bay catches (1962–2003)

	Catch (Pounds)				Revenues (2005 Dollars)			
	Min	Max	Mean	St. Dev.	Min	Max	Mean	St. Dev.
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
Oysters	236,504	24,909,400	12,721,553	9,163,801	1,016,199	95,260,469	43,460,285	30,229,311
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
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
Striped Bass	0	7,322,700	3,036,487	2,149,974	0	9,198,027	4,281,871	2,310,382
Atlantic Croaker	4000	12,540,503	3,561,213	4,308,528	4565	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
Finfishes (unc)	48,600	15,411,700	4,592,079	3,976,933	21,348	1,713,745	452,438	431,529
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
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
Perch	543,718	2,804,300	1,376,998	664,364	364,881	2,268,858	1,014,704	445,763
Snails	3500	2,970,988	351,003	500,658	5400	1,623,722	359,083	446,276
Bluefish	127,100	3,941,300	1,198,068	1,111,131	75,661	1,037,716	363,709	270,176
Horseshoe Crab	0	1,039,407	67,160	208,847	0	691,798	32,430	119,460
American and Hickory Shad	6753	5,196,100	1,312,028	1,534,553	5865	3,181,377	831,235	943,407
Butterfish	12,116	2,101,200	243,103	386,685	8775	1,054,136	142,554	191,653
Puffers	0	12,118,600	1,099,406	2,552,843	0	1,008,127	153,196	241,942

We convert nominal revenues to real using the Bureau of Labor Statistics Consumer Price Index (CPI) for all urban consumers in the U.S. South, which includes all of the relevant states for the Bay, for 1967–2003 and we use CPI U.S. (all urban consumers) prior to 1967.

Our species 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 catch data often do not differentiate among them, and they are necessarily lumped together in our analysis. Some species have separate market categories but are typically caught together or are difficult to distinguish so that catch often is reported in only one species category, *e.g.* alewife, and blueback herring. Unclassified finfishes are the greatest taxonomic compromise. The category typically contains a variety of 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. Table 1 contains descriptive statistics for the 22 species or species groups that meet this criterion. The average actual dockside revenues from these 22 species were $152 million (2005 dollars).

Because estimates of stock levels or maximum sustainable yield are not available for all 22 fish stocks, we approximate maximum sustainable yield with the observed maximum catch over the period (denoted χ_i). This assumption implies that for consistently underutilized species group, the sustainability constraint is overly restrictive, and our analysis precludes recommending an increase in harvest over the historic maximum. This is not a limitation of our method but rather a limitation of the data. If an estimate of MSY is available for an underutilized species k, then we could use it directly for $c_k^{\max}$. Nevertheless, we acknowledge that this assumption is imperfect, and we invoke it here as a starting place for illustrating portfolio management for a real system. As such, managers should interpret our results with caution.

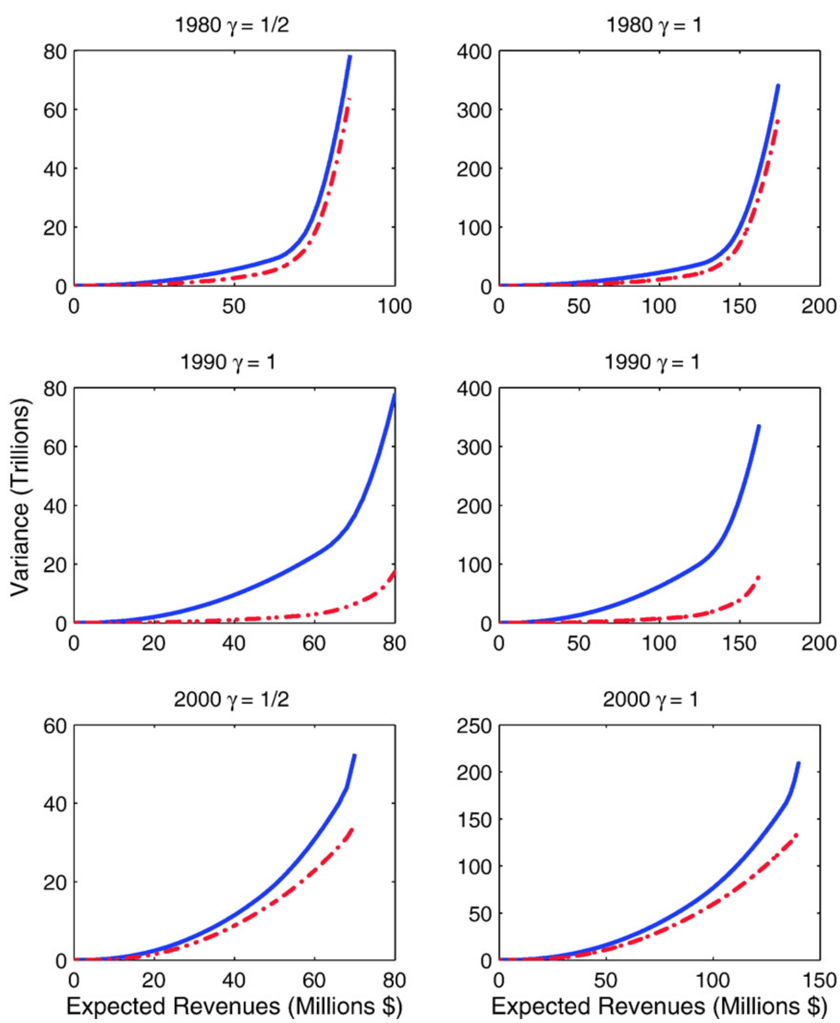

Fig. 1 – Efficient frontiers for two levels of γ. γ controls the flexibility that the manager has to substitute across species. When γ is high, the manager can achieve the same expected revenue at a lower variance. The species frontier (solid line) is derived with a diagonal covariance matrix. The EBFM frontier (dotted line) is derived with the full covariance matrix that includes non-zero off-diagonal elements. The panels have different scales, where the possible revenues are higher in the less constrained system (γ=1), as is the variance. The scales also change over time based on the ability (or lack thereof) of the Bay to sustain certain productivity levels.

Under these assumptions, the upper bound for each species group in the Bay is

$$c_i^{\max}(t) = \gamma_i(t)\chi_i(t)/\Omega_i(t). \tag{5}$$

For comparison, we will use two different values of the sustainability parameter (1/2 and 1) for all fish stocks for all periods. That is, $\gamma_i = \gamma = 1/2$ indicates a maximum catch equal to one-half the observed maximum catch. The window length (how far back in time observations on revenues are considered) and weighting scheme (the value of the geometric decay parameter) over which the maximum catch is calculated varies depending on the specification.

4. Results

We derive annual EBFM mean–variance frontiers from 1975–2003 (solve Eq. (2) for different levels of M in each year) conditional on three retrospective weighting schemes. We present the EBFM frontiers for 1980, 1990, and 2000 in Fig. 1 (dashed line) calculated with an exponential smoothing weighting parameter where only 5% of the weight is remaining after 10 years. Qualitatively similar results emerge from equal weighting and where only 5% of the weight remains after five years. A vector of TACs corresponding to a point on the frontier minimizes the *ex ante* variability for a given level of *ex ante* expected revenues. By operating on the frontier, fishery managers ensure that they are not accepting more risk than necessary for a given level of return from the ecosystem.

For low expected revenues, managers can diversify catches by limiting catches for certain high risk, low return fish stocks to zero. Moving up the curve, however, managers must maximize the catch of more fish stocks in order to reach the revenue target. The extreme occurs at the maximum possible revenues where the managers are at the boundary of what is permitted ($c_i(t) = c_i^{\max}(t)$ for all i). A similar result holds when we impose an upper bound on the feasible sustainable catch levels (imposed by the sustainability constraint); for lower target revenue levels the constraint binds (fish stocks reach $c_i^{\max}$) sooner. The maximum possible revenue is also more limited as the sustainability constraint is tightened.

The EBFM frontiers are derived using the full covariance matrix. What if, however, managers choose the portfolio based only on the variances (the diagonal-only covariance matrix)? The resulting species frontier is akin to a multispecies management strategy that accounts for individual species variability but ignores species interactions. The solid lines in Fig. 1 represent the species frontier.

Comparing the two frontiers provides insight into the importance of species correlations in the Bay ecosystem.

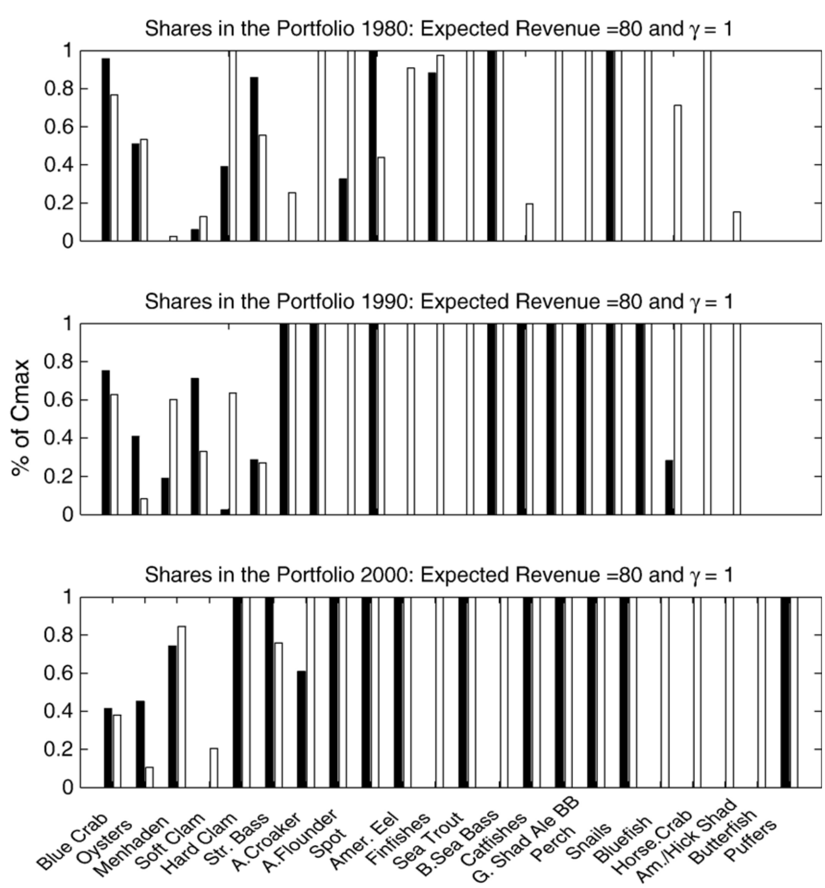

Fig. 2 – Comparing catch allocations by species for the ecosystem and species frontiers. The *Y*-axis indicates the share of the maximum allowable allocation for each species ($c_i^{\max}$) that the optimal portfolios prescribe with $\gamma = 1$ (for expected revenue of \$80 million in 2005 dollars). Black bars correspond to the EBFM frontier and white to the species frontier.

Theoretically, either frontier could have lower risk (variance) for a given level of revenues, as it depends on the signs and magnitudes of covariances across fish stocks. For example, consider a two fish stock portfolio. With fish stocks that are negatively correlated, then the variance of the portfolio will be lower when taking the covariance into account than when the catch shares are derived only with variances. If the fish stocks are positively correlated, however, the EBFM portfolio could have higher variance than the species case. In an *n* asset portfolio, the relative portfolio variance depends on the actual covariances of the fish stocks in the portfolio and we do not have any *ex ante* reason for either outcome. We find that if managers ignore covariance—ecological and economic interdependence are left out of decisions—then they accept more risk for a given level of revenue. Empirically, this means that there are opportunities to exploit negative covariances across species in the Bay ecosystem.

We decompose prescriptions for each fish stock in Fig. 2 with a revenue target equal to $80 million. In this example, the EBFM frontier prescribes *less diversity* in catch, which implies *more diversity* in the standing stocks. We also find that to achieve the same revenue level, the EBFM frontier concentrates more harvest on benthic invertebrates and less on predator fish—a result consistent with concerns over losses in predator diversity (Worm et al., 2005). It is also apparent that to achieve the same level of revenue the number of species in the catch portfolio increases over time, which is due to the recent decline in the productivity of the Bay ecosystem and the resulting lower catches (Jackson, 2001).

A retrospective analysis reveals how much less variance managers could have achieved by operating on the EBFM frontier by reallocating catch across species. The results are plotted in Fig. 3, where the inset describes how the percentage reductions were calculated. It measures the distance off of the frontier in percentage terms and illustrates how the gains vary over the years with more recent years experiencing some of the lowest gains, which is consistent with management decisions that reduce variability on a species-by-species basis in a system that is already fully or overexploited.

Since the actual observed data points fall to the left of the EBFM frontier, there are potential benefits from incorporating species interdependencies. Model predictions both within-sample and out-of-sample corroborate these results, implying that the managers could improve on current management by taking into account the data that existed up to the previous year. The out-of-sample predictions were done by using the

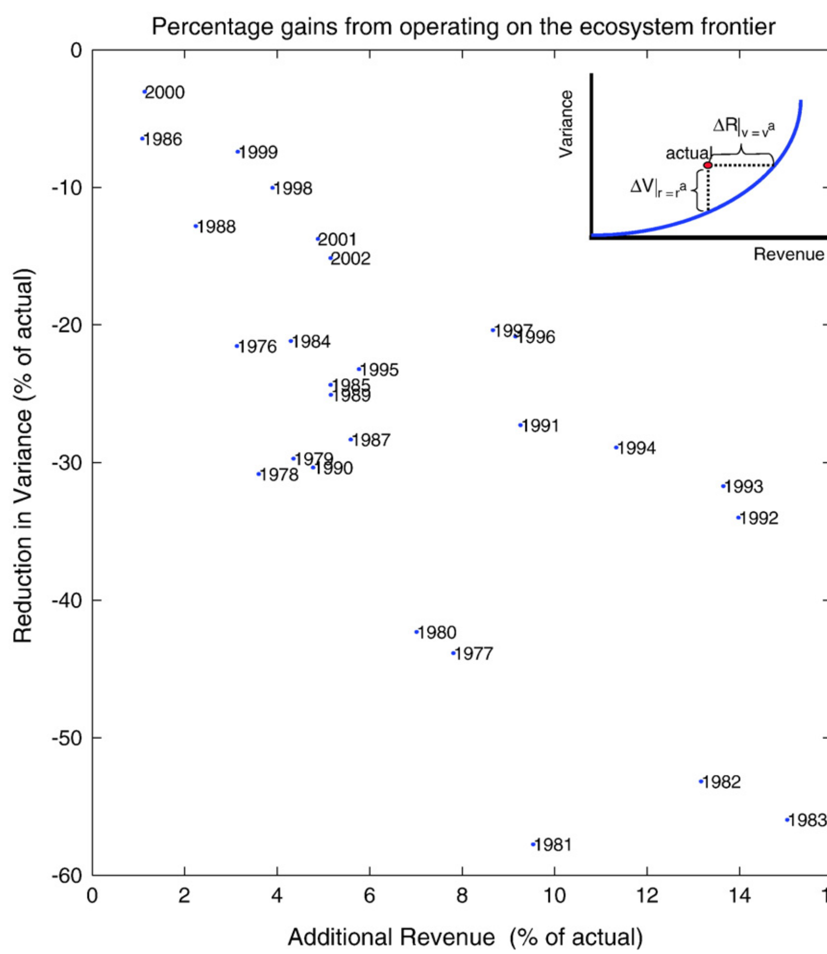

Fig. 3 – Comparison of actual management to efficient frontiers. (a) illustrates the orthogonal distance in percentage terms between the actual point and the frontier in that year (see inset). (b) measures the distance to the ecosystem and species frontier in percentage terms and plots the time series of these differences.

data up to year t to develop the variances, covariances, *etc.* and comparing the predicted outcome for year $t+1$ against the actual allocation in $t+1$ holding the level of total revenues fixed at the actual level across the cases. Table 2 shows for four different years how managers could have achieved the same expected revenues with lower risk at the species level.

Table A1 in the Appendix contains the full correlation matrix for all species assuming equal weighting over the entire sample. Though focusing on pairwise correlations can be misleading in portfolio analysis, some pairwise results do coincide with portfolio prescriptions and can account for differences across species and EBFM frontiers. 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 EBFM 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.

Also in the Appendix, we break the sample pre- and post-1981, which is a point at which catches of oysters and striped bass began to decrease, and compare the actual data to the frontier (assuming equal weighting in each interval). We find that the frontier for pre-1981 is to the right of the post-1981 frontier, implying that potential revenues (variances) were greater (lower) from 1962–1981 than from 1982–2003. This result appears consistent with the recent decline in the Bay and represents the potential mean-variance gains from restoring the ecosystem.

Table 2 – Comparison of actual to optimal allocations for the same total expected revenue

	1970	1980	1990	2000
Blue Crab	27.43%	27.11%	−2.36%	−10.55%
Oysters	−23.11%	−14.11%	4.82%	13.15%
Menhaden	−12.42%	−23.43%	−21.23%	−12.52%
Soft Clam	−3.67%	−1.73%	3.70%	3.45%
Hard Clam	3.09%	1.98%	−0.67%	3.52%
Striped Bass	4.64%	4.04%	7.15%	1.43%
Atlantic Croaker	0.03%	0.59%	−0.11%	−5.42%
Atlantic Flounder	0.44%	0.19%	0.52%	0.41%
Spot	0.49%	1.20%	1.64%	0.74%
American Eel	0.39%	0.44%	0.49%	1.26%
Finfishes (unc)	−0.17%	−0.23%	−0.11%	−0.28%
Sea Trout	1.26%	0.20%	0.92%	1.15%
Black Sea Bass	0.46%	0.45%	0.57%	−0.17%
Catfishes and Bullheads	0.67%	0.55%	0.64%	0.81%
Gizzard Shad, Alewife, Blueback Herring	−1.35%	−0.12%	−0.13%	−0.08%
Perch	0.41%	0.73%	1.10%	0.76%
Snails	1.98%	1.69%	2.22%	1.14%
Bluefish	0.75%	0.49%	0.85%	1.20%
Horseshoe Crab	0.00%	0.00%	0.00%	−0.02%
American and Hickory Shad	−1.30%	−0.41%	−0.08%	−0.01%
Butterfish	−0.13%	−0.04%	−0.01%	−0.06%
Puffers	0.12%	0.43%	0.09%	0.09%

For each year, we find the vector of optimal c_i's where M is the actual total revenue for that year's allocation of catches. We report the share difference of the total value of all catches. In general, the largest percentage changes correspond to species that contribute the largest shares of total landed value.

5. Discussion

One difficulty in implementing the marine ecosystem based fishery management concept is that trade-offs are inevitable and while there is a scientific consensus that ecosystem perspectives are the right approach (Scientific Consensus Statement, 2005), there is very little discussion on how these trade-offs are to be made (Sanchirico and Hanna, 2004). A portfolio approach provides an empirical means *via* a common risk-return metric for assessing unavoidable tradeoffs (Pikitch et al., 2004) from harvesting multiple interacting species. Another significant advantage is that our approach employs data routinely collected by fishery managers. This along with making trade-offs explicit and transparent, and allowing management to be adaptive are important features of an EBFM policy (Essington, 2001).

At the same time, addressing precautionary management requires some means of incorporating risk preferences into the manager's objective. Compared to structural models of the ecosystem, deriving EBFM frontiers provides a complementary view that is simple to implement and flexible to accommodate different ecological, economic and social objectives by including additional constraints or objective functions.

A limitation of EBFM frontiers is that the policy prescriptions are only as good as the estimates of the means and covariances that characterize the multivariate stochastic process (this holds for financial securities). The logical next step is to explore the time series properties of the data. But even with more data analysis, it is important to remember that the approach in this paper is non-structural. As such, the results do not yet incorporate dynamic feedbacks from policy recommendations. If marginal policy changes are implemented, we might expect time series of revenues to continue to convey information signals about the true structural bioeconomic system. But with major policy changes, the time series may have little relevance after a single period of management.

Naturally, before using EBFM frontiers to guide real-world management decisions, we would like to know how dynamically robust policy prescriptions are. One can only speak to this issue by knowing the true structural model of a system. This suggests two important areas for research. First, in ongoing work, we are exploring the performance of EBFM frontiers using simulated stochastic bioeconomic systems in which, by construction, we know the true stock dynamics. Second, since a structural ecosystem model of the Chesapeake is under construction using Ecopath with Ecosim, research comparing policy prescriptions from the structural model with those of the non-structural EBFM frontiers seems worthwhile.

Another extension is to develop *EBFM indicators* (Brodziak and Link, 2002) by measuring the distance between the current state of affairs and the frontier. A similar analysis can be done at the species level. For example, a species may be overfished according to EBFM but not from the traditional single-species perspective (Pikitch et al., 2004). Knowledge of whether the system is overinvested, fully invested, or underinvested in a species could complement the standard biological measure of overfished, fully exploited, or underfished. An underinvested and overfished resource might be a priority for a recovery plan, as there are gains to increasing its allocation in the portfolio, but current population levels limit such an action. In this sense, our approach is a step towards a definition of ecological overfishing that hinges on whether the extractive flows from an ecosystem are optimally diversified.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecolecon.2007.04.006.

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