HERRING
ALLIANCE

September 21, 2010
Mr. Paul Howard, Executive Director
New England Fishery Management Council
50 Water Street
Newburyport, Massachusetts 01950

RE: **Amendment 5 to the Atlantic Herring Fisheries Management Plan**

Dear Mr. Howard:

The Herring Alliance is committed to reforming the management of the Atlantic herring and mackerel fisheries and mitigating the impact that these small-mesh fisheries are having on the target stocks, the animal populations that rely on these stocks for food and the incidental catch of weak stocks, including river herrings, shad and groundfish. On behalf of our members we have supported comprehensive amendment of the Atlantic herring fisheries management plan (FMP) since the inception of the original Amendment 4 over two years ago. Unfortunately, the critically important issues of catch monitoring, specific measures for protecting non-target species (e.g., river herrings, shad, and groundfish), and protecting spawning Atlantic herring were split off to Amendment 5 in June of 2009. Alternatives for addressing these Council objectives remain to be fully developed in a coherent Amendment 5 with accompanying Draft Environmental Impact Statement (DESI). The Herring Alliance has been particularly troubled by the wholesale elimination of alternatives from the amendment draft document by the Herring Oversight Committee, without full analysis, discussion by the Council, or public comment. We write today about Amendment 5 as the Council will make crucial decisions on this amendment at its next meeting.

In the remainder of this letter we detail the following specific concerns the Herring Alliance has about the continued development of the Amendment 5 document and DEIS:

- It is critical for the Council to ensure that a comprehensive range of management alternatives is made available for public comment and analysis
- Amendment 5 must include at least two alternatives for annual catch caps for river herring
- All catch in the Atlantic herring fishery must be subject to high levels of at-sea sampling allowing accurate estimates for the catch of key species including river herrings, shad, and groundfish
- River herring incidental catch hotspots must be protected; the Amendment must include alternatives based on time/area fishery exclusions and a system of move-along rules supported by a catch cap, reliable at-sea sampling, and administered by NMFS
- Amendment 5 must meet its objective of protecting adult spawning herring; alternatives must be included based on a threshold for spawning herring that triggers move-along rules and possible time/area closures to minimize catch of spawning fish on Nantucket Shoals
- All groundfish closed areas must be protected from midwater herring vessels (categories A and B)

The Herring Alliance has grown to include 19 member organizations representing approximately 1.5 million individuals. Concerns about the demise of river herring and shad populations up and down the Eastern seaboard are growing. As the in-river fisheries are closing down in state after state, the fishery for these important fish has rapidly transformed into one that is dominated by catches at sea in federally managed-small mesh fisheries. This bycatch fishery is a complicated mixed-stock fishery that includes many extremely weak stocks. The individual river-specific stocks cannot yet be recognized in the at-sea catch. With abundance so low, these stocks are no longer commercially significant, but they could become so again and could also contribute to the recovery of inshore stocks of groundfish and other commercially important animals.

The catch of river herrings and shad is poorly monitored and not regulated in federal waters. Since the Council chose not to recognize these fish as stocks in the fishery for Amendment 4, no catch limit has yet been established for fish that are clearly impacted by this fishery. This makes the actions taken through Amendment 5 exceedingly important. In our estimation, the Council will not be in compliance with the annual catch limit, bycatch and overfishing requirements of the Magnuson-Stevens Act if it fails to set an annual catch cap and establish strong measures for monitoring and to reduce at-sea catch of these fish.

We have repeatedly drawn attention to the National Environmental Policy Act (NEPA) requirement that the Council develop a wide range of alternatives for Amendment 5 in preparation for analysis and public comment. The National Marine Fisheries Service (NMFS) has also expressed concern about including a robust set of alternatives for this amendment.[1] Despite these considerations, the record shows that the Herring Oversight Committee has developed a substantial number of alternatives and subsequently eliminated them prior to Council consideration, full analysis, or public comment. It is imperative that Council approach this Amendment with an open mind about strengthening the good alternatives that are in the current document and adding additional alternatives where it is deficient.

A comprehensive range of management alternatives is a legal requirement. The central purpose of the National Environmental Policy Act (NEPA) is to ensure that both decision-makers and the public are well-informed about the potential environmental effects of proposed actions.[2] This is accomplished through the Environmental Impact Statement (EIS). The requirement under NEPA to analyze a comprehensive range of the reasonable alternatives is "the heart of the [EIS]."[3] The NEFMC and NMFS must "study, develop, and describe appropriate alternatives to recommended courses of action in any proposal which involves unresolved conflicts concerning alternative uses of available resources."[4] The environmental impacts of the proposed action and any alternatives must be presented in comparative form, "sharply defining the issues and providing a clear basis for choice among options by the decision-maker and the public."[5]

---

[1] Letter from Regional Administrator Patricia Kurkul to NEFMC Chairman John Pappalardo, dated 25 August 2010.

[2] *See Robertson v. Methow Valley Citizens Council*, 490 U.S. 332, 349 (1989)(NEPA ensures that the agency will "carefully consider detailed information concerning significant environmental impacts" and that such information is available to the public); *accord*, *Baltimore Gas & Electric Co. v. NRDC*, 462 U.S. 87, 97 (1983). NOAA Fisheries' regulations emphasize their duty to prepare an EIS that adequately informs the public of the environmental impacts of the proposed action: "An EIS must provide a full and fair discussion of significant environmental impacts." (National Oceanic and Atmospheric Administration Administrative Order 216-6, hereafter "AO 216-6") AO216-6 § 5.04.a.1.

[3] 40 C.F.R. § 1502.14; 42 U.S.C. § 4332(2)(c)(iii)..

[4] 42 U.S.C. § 4332(2)(E).

[5] 40 C.F.R. § 1502.14.

NOAA Fisheries' own NEPA regulations underscore the importance of an adequate analysis of alternatives: "An EIS must provide a full and fair discussion of significant environmental impacts and inform decision makers and the public of the reasonable alternatives which would avoid or minimize adverse impacts or enhance the quality of the human environment."[6] While technical and economic factors must be considered when identifying alternatives for consideration in an EIS, NEPA contemplates that agencies even consider alternatives beyond current funding levels and current law because the EIS may serve as the basis for a change in funding or law.[7]

Thus, it is critical at this stage in the development of Amendment 5, when deciding which alternatives to include as part of the DEIS, that a full range of the reasonable alternatives be identified in order to meet the goals and specific requirements of NEPA. Such alternatives should allow decision-makers to make clear choices about the range of potential uses for the resources impacted by the Atlantic herring fishery, ranging from the harvest of Atlantic herring for commercial use, to its role in the ocean ecosystem, to the impacts on species caught as bycatch such as river herring, shad, and groundfish.

We have been alarmed by the Herring Oversight Committee's aggressive elimination of alternatives from the draft amendment in advance of analysis and the opportunity for public comment and fear that the document will not contain the range of alternatives and environmental impact analysis required by NEPA. Examples include elimination of the proposed accountability measures to prevent abuse of the exceptions to a slippage prohibition (i.e. trip termination, slippage caps), elimination of alternatives for a maximized retention system, and the decision to strip out consideration of modern electronic monitoring technologies.

An annual catch cap for river herring and shad must be established. The draft Amendment 5 document currently includes good background information on *Measures To Address River Herring Bycatch* (section 3), but the specific management alternatives remain weak because they are not backed up with an annual catch cap for river herring and shad. This is a fundamental flaw in the draft Amendment 5 document. The at-sea incidental catch (i.e., bycatch) of river herring and shad is not being addressed by the Atlantic States Marine Fisheries Commission (ASMFC), or any other regulatory body, and must be addressed by the New England Council for those fisheries under its jurisdiction.

Without a specified annual cap and monitoring sufficient to track progress toward the cap, the fleet will lack a strong incentive to minimize incidental catch. The bycatch avoidance program for Chinook salmon in the Bering Sea pollock fishery, for example, functions because the fishery is closed when the annual catch cap is reached.[8] This provides a powerful incentive to avoid bycatch through any mechanism possible. As a related matter, we feel the discussion of the Bering Sea program in the Amendment 5 Discussion Document is incomplete at this time in that it does not mention or discuss the underlying salmon cap. A similar system is used to control the bycatch of yellow tail flounder in the New England scallop fishery. The move-along rules and many of the other alternatives outlined in the current Amendment 5 document will not work effectively absent a strong incentive to hold catch of river herring and shad below an annual cap.

---

[6] AO 216-6 at § 5.04.a.1.
[7] See, e.g., NEPA's Forty Most Asked Questions 1-2; 40 C.F.R. Sections 1500.1(a), 1502.14, 1506.2(d)).
[8] Federal Register / Vol. 75, No. 55 / Tuesday, March 23, 2010 / Proposed Rules on Chinook Salmon Bycatch Management in the Bering Sea Pollock Fishery.

The Council should add at least two Alternatives to Amendment 5 for establishing annual catch caps. One of these should be based upon recent catch history as reported by the industry in Vessel Trip Reports (VTR) and a second should be based upon a scientific approach based on the population status of these stocks. Clearly caps should ultimately be based upon the best available science. However, to date the Council does not have an appropriate scientific analysis for setting catch caps. The Amendment should include a mechanism for setting catch caps based on recent catch history and a plan for replacing these caps with science-based caps when the scientific analysis is completed. The New England SSC has essentially taken this approach with its advice on Acceptable Biological Catch (ABC) for Atlantic herring and the scientific panel conducting the recent (2010) benchmark TRAC assessment for Atlantic mackerel provided similar guidance: in the absence of better scientific information, catch levels should provisionally be based on recent catch history. [9] In both cases, the average catch of the three most recent years was identified as a suitable option. The Council should avoid alternatives based on bycatch tolerances (i.e., cap set as percent of target catch) because this approach will not provide adequate protection for river herrings and shad and cannot be defended on scientific grounds.

Catch caps based on recent catch history. The NMFS maintains a VTR database that includes records furnished by vessel captains. The catch history for river herring and shad should be assembled to determine the average annual catch over the past three years to produce a provisional basis for the annual catch cap. NMFS currently uses captain’s reports in databases for the observer and other programs.

Science-based catch caps based on population status. River herring and shad exist in the coastal oceans as mixed-stocks and are harvested as such with no present method for identifying the river-specific stocks from which the fish caught at sea originate. This means that management must be based on an appropriate mixed-stock assessments and management that recognizes the presence of weak stocks mixed with stronger stocks. The available river- and state-specific stock assessments do not provide a suitable basis from which to develop appropriate science-based catch caps. However, this is a problem that can be solved through coast-wide stock assessment methods and which has been investigated by a number of authors.[10]

In a recent analysis Miller (manuscript in preparation) reviewed approaches taken to similar management challenges from around the world, and specifically examined the case of river herrings and shad in the at-sea bycatch fisheries of the US eastern shore. [11] Miller presents a method for developing coast-wide bycatch targets for American shad and river herring using *stochastic stock reduction analyses*. The model structure uses estimates of vital rates of growth, maturity and recruitment to the fishery to parameterize a population dynamic model. Forecasts from this model are statistically compared to observed time series of catch and an index of abundance, and maximum likelihood estimates of the maximum sustainable yield (MSY), the fraction of the population caught at MSY, UMSY and the compensation ratio of the population at low stock

---

[9] Memo from Dr. Steve Cadrin, Chairman, Scientific and Statistical Committee, November 17, 2009, to Mr. Paul Howard, Executive Director, NEFMC; TRAC. 2010. Atlantic Mackerel in the Northwest Atlantic. TRAC Status Report 2010/01. Fisheries and Oceans Canada and NOAA Fisheries / NMFS US.

[10] Walters, C. J., S. J. D. Martell, and J. Korman. 2006. A stochastic approach to stock reduction analysis. Canadian Journal of Fisheries and Aquatic Sciences 63(1):212-223; Berkson, J., and coauthors. 2010. Guidance on setting ABCs when only average catch is known. American Fisheries Society, Pittsburg, PA.; Forrest, R. E., S. J. D. Martell, M. C. Melnychuk, and C. J. Walters. 2008. An age-structured model with leading management parameters, incorporating age-specific selectivity and maturity. Canadian Journal of Fisheries and Aquatic Sciences 65(2):286-296;

[11] Miller, TJ (2010). Estimating Bycatch Limits for American Shad and River Herrings in the Northwest Atlantic. Report prepared for MRAG Americas (report appended).

sizes (i.e., a measure of steepness of the stock recruitment curve compared to the replacement line). This approach can be used to estimate MSY. MSY may then be reduced by estimated in-river fishing mortalities thereby arriving at an appropriate coast-wide bycatch target for all at sea fisheries. This limit would then need to be apportioned to the various fisheries by managers.

Amendment 5 should include an alternative that identifies the stock reduction analytic approach as a method for deriving biologically appropriate caps. It should include some analysis of this and other alternative approaches, and establish a process for replacing catch history-based caps with caps developed from a coast-wide assessment of the stock complex.

All catch must be subject to robust at-sea sampling. Reliable catch estimation is a critical objective for Amendment 5. The target catch (Atlantic herring) and the incidental catch must be reliably estimated from a robust sampling program. As the Herring PDT recently wrote:

> *…sea sampling will remain the best method for estimating at-sea discards, an important piece of information that cannot be generated at all from a portside sampling program…*[12]

Sampling must be sufficient to allow fleet-wide extrapolation so that total catch can be reliably estimated (i.e., landed and discarded catch). An analysis recently present to the Herring Oversight Committee indicated that at-sea coverage levels must be substantially increased over those used in the recent years before the standards (i.e., Coefficient of Variation for estimated catch, CV) that the Council has identified for river herring are attained.[13] This analysis indicated that at-sea observers may need to be present on at least 70% of trips to achieve Council goals. The Herring Alliance argued strongly for higher levels of at sea coverage for a variety of reasons, and strongly supports 100% observer coverage for the Atlantic herring fishery, at least for the vessels with the most fishing power (permit categories A and B).

Without reliable estimates of the target and incidental catch, the Atlantic herring fishery cannot be managed within the law.[14] Amendment 5 must include a range of alternatives for achieving the catch estimation objectives, including strong plans for at-sea sampling, maximized sampling and dockside monitoring including catch alternatives that can provide for independent verification of reported catch weights.

Obviously discarding of catch without sampling (e.g., release or slipping of trawl nets) is not compatible with the Council's monitoring objectives. If current fishing practices do not allow for sampling, the fishing practices must be modified to allow sampling for catch reliable estimation.

River herring incidental catch hotspots must be protected. The Herring PDT has done an outstanding job of utilizing the best available science [15] to identify times and areas where the incidental catch of river herring and shad is expected to be high. Amendment 5 must include well-developed alternatives for protecting these depleted stocks within these hotspots.

---

[12] Final Herring PDT Report, June 15, 2010, Holiday Inn, Mansfield, MA, pg 2.

[13] Presentation by M. Cieri to the Herring Oversight Committee 2 September 2010, Portsmouth, NH.

[14] National Standards 1 and 9: 16 U.S.C. 1851-1852 MSA §§ 301-302., 98-623 & 104-297.

[15] National Standard 2: 16 U.S.C. 1851-1852 MSA §§ 301-302., 98-623

The Committee motion recommending development of an alternative prohibiting directed fishing for Atlantic herring within hotspots should be fully developed as one or more alternatives in Amendment 5 (McGee/Libby, 28 July 2010).

Alternatives should also be developed that would establish an incidental catch threshold that, when exceeded, would exclude herring vessels from hotspot areas. For example, a catch rate threshold (e.g., an amount per haul) and a total catch level per trip would be established for river herring and shad to regulate access of Atlantic herring vessels to each hotspot area. If either threshold were exceeded, access to the hotspot area would be suspended. An additional alternative could be modeled after the system used to regulate bycatch of groundfish within Groundfish Closed Areas. In this case, a 1% tolerance is used for bycatch of regulated multispecies such as haddock, where the tolerance is a percentage of the target catch of Atlantic herring and mackerel. [16] Although this type of tolerance approach is problematic for conservation of river herring and shad, it is an approach that has been used with some success and might serve as an interim measure. The new rules currently being developed by NMFS for access to CA I will be ineffectual if applied to river herring hotspots without some form of limiting catch cap.

Spawning Atlantic herring must be protected. The Council wisely identified protection of Atlantic herring while spawning as a priority for Amendment 4/5. Although little has been done by the PDT to analyze approaches to achieve this objective, we urge the Council to include alternatives in Amendment 5 for the protection of spawning grounds. Two approaches should be included: (1) time/area closures and (2) a move-along rule triggered by a spawning fish threshold.

Specifically, the best available data should be used to identify times and areas where protections could effective for the Georges Bank and Nantucket Shoals sub-populations. Modeling studies of Atlantic herring stock complex, or *metapopulation*, indicate the critical importance of the Nantucket Shoals component to the whole complex.[17] According to Copper et al., the Nantucket Shoals sub-population serves as a source of recruits for the entire metapopulation and thus protects them against depletion. This analysis also suggests that the Georges Bank sub-population is the least resilient of all the spawning components and thus relatively vulnerable to collapse.

The draft Amendment already includes alternatives for move along rules for bycatch of river herring. This kind of system requires good observers who can report the data needed to determine when fishing a given area possess a risk and then the data would be used to trigger the requirement that the relevant fleet move out of a specified area for an appropriate period of time (e.g., 2 weeks). Adding a trigger based upon a threshold for catch of ripe and running Atlantic herring is thus well within reach. NMFS personnel who participate in the seasonal bottom trawl survey routinely classify herring and other species according to spawning stage. NMFS personnel or contractors serving as observers for NEFOP could expand their repertoire of tasks to classify Atlantic herring. We strongly urge the Council to include alternatives for protecting spawning Atlantic herring.

---

[16] Regulations specified in CFR Section 648.81 (a)(2)(iii)

[17] Cooper AB, Wakeford RC, AA Rosenberg (ms in preparation) How sensitive is the success of current management measures for Northwest Atlantic herring to assumptions about stock structure and productivity? This analysis was presented to the SSC by Dr. Andrew Rosenberg (UNH) 1 May 2009, and again to the Herring PDT on Wednesday, July 01, 2009, by Dr. Robert Wakeford (MRAG Americas, Inc).

Groundfish closed areas must be protected from midwater herring vessels. Observer data and other forms of information show that midwater herring vessels catch significant amounts of haddock and other groundfish.[18] When midwater trawlers were granted access to groundfish closed areas it was assumed that midwater trawl gear did not contact the bottom or interact with groundfish. To the best of our knowledge, this assumption was not based on any substantive scientific analysis. It has become abundantly clear that this assumption was in error.

Midwater trawlers should not be allowed in areas specifically managed for the protection of groundfish unless it can be demonstrated that these vessels can modify their fishing practices so that gear remains well clear of the bottom (e.g., not closer than 100 feet) and such that groundfish bycatch is reduced to a negligible amount (i.e., near zero). As outlined in the Draft Amendment 5 Discussion Document (July 27-28, 2010 Herring Oversight Meeting), Amendment 5 should include alternatives that prohibit midwater trawl vessels from all of the groundfish closed areas, while recognizing that with appropriate gear modifications and/or practices midwater trawl vessels might regain access through a tightly regulated Exempted Fishing Permit (EFP). If successful in reducing groundfish bycatch, the measures evaluated through the EFP could form the basis for measures regulating future access.

Atlantic herring is a vital resource for many different groups. Concern about the issues related to this fishery is extremely high, extending well beyond those directly involved in the business of catching and marketing Atlantic herring. We urge Council members to recall the enormous outpouring of interest in monitoring and bycatch registered during the scoping process for what has become Amendment 5. When the Council invited public comment during scoping in 2008, over ten thousand written suggestions were received and many of them focused on the very problems this amendment can address.

This widespread interest is understandable since, as food, all herrings (Atlantic, blueback, alewife and shad) together form a vital link in the ecosystem – they are *keystone* species whose status has profound impacts throughout the ecosystem, including effects on the human environment. The management decisions made for this fishery impact groundfishermen due to the bycatch of groundfish by midwater trawl ships and due to the depletion of herring as a food source for groundfish. Those whose commercial and recreational interests depend on tuna and other large pelagic fish are also impacted, again by the removal of food from the ecosystem and through bycatch. The success of inshore spawning in codfish is also thought to be linked to the availability of herring as forage, including river herring.[19] The direct harvest of river herring and shad is being shut down in state after state, striking at the heart of coastal communities and the region's fishing heritage.

The Council has a responsibility that extends beyond the short-term interests of those few in the herring business who have most conspicuously fought against accountability in the herring fishery. The herring belong to all manner of user groups – those mentioned above, those who value healthy ocean ecosystems

---

[18] *Observed Haddock Bycatch in the Closed Areas in the Midwater Trawl Herring Fishery* - audited NMFS observer data examined for the period May 2004 through October 2008. Presented by NMFS, at New England Fishery Management Council meeting, April 8, 2009, Mystic, CT. Available as *#4 Information Re. Haddock Bycatch in Closed Area I* at www.nefmc.org/press/council_discussion_docs/list_of_april2009_discussion_docs.html.

[19] Jordaan A, Hall C, Frisk M (2008) Is the recovery of cod (*Gadus morhua*) along the Maine coast limited by reduced anadromous river herring populations? Mia J. Tegner Memorial Research Grant in Marine Historical Ecology and Environmental History Final Report, October 2008; Ames T (2010) Multispecies Coastal Shelf Recovery Plan: A Collaborative, Ecosystem-Based Approach. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 2:217–231, 2010.

from land-locked and coastal states alike, and those whose businesses depend upon enriching the lives of visitors who come to New England for ecotourism. The herring are a public trust resource and the Council's stewardship must take this into account in addition to the near-term interests of a few in the herring business. The committee's job includes safeguarding the interests of those who are unable to do so, including future generations.

Industry representatives on the Council and attending public meetings have argued that it is too dangerous to sample the portion of the midwater trawl catch that is not pumped into the ship's hold. At the same time they have attempted to assure the public that the residual catch is only a few hundred pounds – nothing we should worry about. All catch must be sampled, and if current fishing practices are not amenable to catch sampling then these fishing practices need to be changed.

The industry and some Council members have worked hard to defeat programs that could establish effective disincentives to the practice of releasing (i.e., *slipping*) large quantities of fish to the sea without sampling. Managers, fishermen and the public need to be able to determine the amount and composition of the catch through reliable sampling which estimates the total catch and includes the catch released to the sea. Herring that interact with trawl gear have a very high mortality rate, much higher than that for seines, even when released directly to the sea.[20] Thus, mortality is expected to be underestimated if catch from trawl nets is released without sampling. Additionally, catch remaining in nets at the time of release cannot be assumed to be representative of portions of the catch that may have been pumped aboard, due to stratification and mechanical sorting at the intake.

Closing comment. We urge the Council not to lose sight of the importance of including a full range of management alternatives in the Amendment 5 DEIS document for public review. This will enable the Council to fully realize the benefits of a robust public comment opportunity and PDT analyses that clearly define the impacts of this amendment and the choices to be made. We urge you to be aware that the Committee may well have eliminated of a number of alternatives that should be further analyzed, and put out for public comment. This is a critical juncture for the Council. This document has been long in the making yet still is at risk of not meeting its essential monitoring and bycatch reduction objectives. The Council can substantially reduce that risk through proactive decision making this month.

Sincerely,

Science and Policy Manager
Pew Environment Group

cc: Mr. John Pappalardo, Chairman, New England Fisheries Management Council
Ms. Lori Steele, Fishery Analyst, NEFMC Staff – Herring FMP

Miller (2010) Report Appended

---

[20] Suuronen et al,.(1996) Mortality of herring escaping from pelagic trawl codends. Fisheries Research **25**: 305-321

Ref: [UMCES]CBL 10-089

Estimating Bycatch Limits for American Shad and River Herrings in the Northwest Atlantic

Thomas J. Miller

Chesapeake Biological Laboratory

University of Maryland Center for Environmental Science

P. O. Box 38

Solomons, MD 20688

Prepared For:

MRAG Americas

New England Office

65 Eastern Avenue, Unit B2C

Essex, MS 01929

Executive Summary

American shad and river herrings have declined dramatically since the period of European colonization of North America as a result of over-exploitation and the damming of major and minor rivers alike that has prevented these anadromous species from reaching spawning habitats. Concern over the perilous state of stocks of these species in many coastal rivers has lead to US management jurisdictions along the east coast to close targeted fisheries for these species. However, these actions have not lead to sustained recovery. As a result attention has focused to bycatches of these species in diverse coastal fisheries including those for Atlantic herring, Atlantic mackerel and squids.

Here I review approaches to setting bycatch limits for shad and river herrings. Four principal approaches are identified: tolerance-based management, per recruit modeling, delay-difference modeling and fully-age structured modeling. The first two of these approaches are not to be recommended. The first does not limit bycatch as the amount of bycatch varies in direct proportion to the weight of target species landed. Per recruit modeling offers a way of establishing bycatch targets, but no way of telling how current levels of bycatch relate to the target. The last two approaches listed can provide the foundation for establishing management reference points and an approach for determining how the current levels of bycatch compare to the reference points established. For American shad and river herring, data gaps currently prevent implementation of fully age-structured models.

Additionally, because the bycatch occurs in the coastal ocean where the stocks from the different management jurisdictions and regions mix, I recommend abandoning the river-specific stock assessments that have been completed to date, in favor of a coastwide assessment. This approach, while ignoring the plasticity in vital rates observed along the east coast in these species precludes the need to estimate the faction of each stock that is caught as bycatch in these coastal fisheries.

Here, as a proof of concept, I develop and recommend coastwide bycatch limits for American shad using stock reduction analyses. The model structure uses estimates of vital rates of growth, maturity and recruitment to the fishery to parameterize a population dynamic model. Forecasts from this model are statistically fit to observed time series of catch and an index of abundance. The model produces maximum likelihood estimates of the maximum sustainable yield (MSY), the fraction of the population caught at MSY, U_{MSY} and the compensation ratio κ of the population at low stock sizes (a measure of steepness of the stock recruitment curve compared to the replacement line). Catch limits for American shad were developed using coastwide catch data from 1980-2005, a fishery-dependent index of abundance from the haul seine fishery in Lewes, Delaware and vital rates estimated from fish sampled from the Hudson River, NY. The recommended coastwide catch limit for American shad is 575 metric tonnes (mt) per year. In 2005, the last year of landings used in the model, the targeted landings of American shad were 370 mt. If management wishes to support this level of targeted landings, the implication is that an appropriate bycatch limit is 205mt (catch limit – targeted landings). Fishery-specific bycatch limits could be developed based on a status quo proportion or alternatively management jurisdictions could alloctate bycatch allocations to individual fisheries.

TABLE OF CONTENTS

1. INTRODUCTION

The eastern seaboard of North America supports a diverse fish fauna of more than 600 species (Scott and Scott 1998). Diadromous species such as the catadromous American eel (*Anguilla rostrata*) and the anadromous striped bass (*Morone saxatilis*) are an important component of this fauna transferring energy and materials between the marine and freshwater ecosystems. In particular several species of anadromous, herring like fishes from the Alosidae family have been historically abundant. This complex of species that includes the American shad (*Alosa sapidissima* (Wilson, 1811)), the blueback herring (*Alosa aestivalis* (Mitchell, 1815)), the alewife (*Alosa pseudoharengus* (Wilson, 1811)) and the hickory shad (*Alosa mediocris* (Mitchell, 1814)) is often referred to as shads and river herrings. Their biology and ecology has been extensively studied. In general, species return to natal rivers along the eastern seaboard in spring and summer, spawning in fresh or low salinity water, and then returning to the sea. However, there is some straying between natal rivers (Messieh 1977), suggesting a complex metapopulation structure in these species (Jones 2006). In the ocean, individuals in all of these species undertake long migrations up and down the coast following favorable thermal regimes (Leggett and Whitney 1972). In seminal work on American shad, Leggett and Carscadden (1978) demonstrated a strong latitudinal gradient in the number of times an individual female spawns with single spawning (semelparity) favored in more northern latitudes and multiyear spawning (iteroparity) favored toward the south. Plasiticity in this and other life history traits is common throughout the complex.

Shad and river herrings were once abundant in rivers, estuaries and the coastal regions along the eastern seaboard of North America. Their former abundance placed these species at the heart of colonial society. It is not for nothing that John McPhee (2002) called the American shad (*Alosa sapidissima*) the "founding fish." McPhee documents the importance of seasonally abundant shad to seventeenth and eighteenth century society. Harvest records show that more than 49,000 metric tonnes (mt – 1 mt = 2,204.6 Lbs) of shad were harvested from the Potomac River in the early nineteenth century. However, continued harvest pressure and habitat loss resulting from development and damming of watersheds throughout the east coast of North America has lead to dramatic declines in the species that form this complex. A century on, the harvest of American shad in the Potomac River had declined by two orders of magnitude and today, almost two centuries later, commercial harvests of American shad are effectively banned with only incidental take permitted in mixed species fisheries such as pound nets (capped at < 2% of total). This pattern of decline in American shad is not unique. Limburg and Waldman (2009) document that abundance time series for 13 of 35 diadromous species examined in the Atlantic basin show declines to less than 98% of historic levels. Despite closures of most of the in-river fisheries and efforts to remove dams to open spawning habitat, convincing signs of recovery of these species is lacking (Atlantic States Marine Fisheries Commission 2007c; Atlantic States Marine Fisheries Commission 2008).

Shad and river herrings are managed by the Atlantic States Marine Fisheries Commission (ASMFC) under authority granted to it by the Atlantic Coastal Fisheries Cooperative Management Act (1993). In the ASMFC process, management is based on advice from stock assessments. Shad and river herring should be considered "data poor" species for assessment purposes both because of data limitations and because of their complex metapopulation structure (Jones 2006). Previous assessments of shad and river herring have focused on in-river analyses. These assessments have compiled data on historical in-river catches, mark-recapture studies, fishery-dependent and fishery-independent surveys, on the length and age composition in those surveys and on estimating vital rates from individual rivers and geographic regions. These data have been analyzed, when possible, to develop estimates of the total instantaneous rate of mortality (Z) experienced by the stock in each river (e.g., Atlantic States Marine Fisheries Commission 2007c). These estimates of recent rates of mortality have then been compared to biological reference points based on proxies for the rate of mortality at maximum sustainable yield developed from spawner per recruit analyses. This framework provides an indication of current stock status, but does not adequately account for population dynamic processes. For example, the estimates of Z from year-specific estimates do not permit either forecasts or hindcasts of population structure and abundance. Such analyses are critical to determine the extent of density-dependent processes.

Concerns over declining abundances noted above and the loss of access to spawning habitat by damming of coastal rivers have led most management jurisdictions to close targeted fisheries for these species. However, concern remains over the bycatch of shad and river herrings in other fisheries, notably in pelagic trawl fisheries for Atlantic herring (*Clupea harengus*), Atlantic mackerel (*Scomber scombrus*) and squids. When taken in these fisheries, shad and river herrings are often sold as bait. In Amendments 2 and 3 to the Interstate Fishery Management Plan for shad and river herrings, ASMFC specifically identifies concerns over quantifying and effectively managing the bycatch mortality of these species as a restoration tool. By these amendments, ASFMC member states are required to monitor and annually submit reports of the bycatch and discard of shad and river herrings in fisheries that operate in their jurisdictions. They commit also the Commission to working with NOAA Fisheries and the regional Fishery Management Councils to monitor bycatch in federal waters. To date, efforts to manage bycatch of shad and river herrings have focused on documenting the extent of spatial and temporal overlap of these alosids with pelagic fisheries (e.g., Cournane 2010). This knowledge may permit management jurisdictions to implement seasonal and spatial closures to reduce bycatch of shad and river herring in these pelagic fisheries.

Managing bycatch in fisheries is often controversial because it involves balancing demands from different stakeholder groups. On occasions the principal bycatch concern is the catch of endangered or threatened large marine species including seals, sharks, turtles and seabirds (Moore et al. 2009). Even when such species are not involved, bycatch and discards of fish can be substantial. In a recent review Harrington et al (2005) estimated that an amount equivalent to 28% of the commercial fish and shellfish landings is discarded as bycatch each year in US fisheries. This figure is similar to the global figure of 26% estimated by Alverson et al. (1994). These broadscale estimates mask considerable regional variation. For example, in the northeast region of the US, discards represent an additional removal from the ecosystem equivalent to 49% of the commercial catch (Harrington et al. 2005). This magnitude of bycatch indicates that assessment of bycatch must be a routine aspect of management (Harrington et al. 2005). When marine mammals and other large species are involved demographic modeling is often used to estimate the impacts of bycatch and to estimate safe levels of harvest of the non-target species (Lewison et al. 2004). For fish bycatch, quantitative stock assessments are the preferred tool. For example, in the northwest Atlantic butterfish (*Peprillus triacantus*) is harvested almost exclusively as bycatch in the squid fishery (Northeast Fisheries Science Center 2010). The impact of this bycatch on the population dynamics of butterfish is estimated using an age-structured, delay-difference model. Absent such quantitative models, management jurisdictions have no way of knowing which levels of bycatch are injurious to non-target species and which are injurious.

Management of many fisheries around the world is achieved through establishment of management reference points – either limits which establish threshold values of minimum abundance and maximum exploitation rate that should not be exceeded, or targets which indicate desirable and sustainable levels of exploitation and abundance (Restrepo et al. 1998). In the US, the Magnusson-Stevens Fishery Conservation and Management Act (FCMA - 2006) requires management jurisdictions to establish limit reference points. In this framework the maximum sustainable exploitation rate is termed the overfishing limit (OFL). The Act also requires management jurisdictions to set allowable biological catches (ABCs) that are lower than or no greater than that associated with OFL to reflect scientific uncertainty. Although not bound by the FCMA, the ASMFC endeavors to follow its guidelines. This would require establishing an OFL and ABC for shad and river herring.

The current focus of shad and river herring assessments on in-river summaries challenges the ability of management to establish coastwide bycatch management reference points such as an OFL. When the majority of removals occurred in the rivers themselves, each in-river assessment could attribute removals unequivocally to that sub population. Accordingly, management agencies could regulate catch within their jurisdiction to ensure that the population stayed within limit reference points. However, now a large fraction of the removals from shad and river herring populations occur as bycatch in coastal fisheries. Thus, for in river assessment to be effective, the removals that occur in the coastal ocean must be distributed and appropriately allocated back to the individual river stocks. Currently, data to do this are lacking. Genetic techniques do not appear sufficiently discriminatory to provide such resolution because of the dynamic and variable nature of the composition of the catch (Brown et al. 1999). Recently, research has indicated that otolith microchemical techniques may provide the fine spatial resolution that is required (Jones 2006), but the cost of such a sampling program suggests its widespread application is unlikely in the near future. In the absence of these data, three approaches to estimating the composition of the mixed catch are outlined here. The first approach would be to weight the contribution to the ocean catches of the different stocks by the estimates of their inherent productivities – as measured by their population growth rates or slopes of their stock-recruit functions. This approach makes the assumptions that productive populations will contribute more the overall bycatch than less productive stocks. A second approach might be to weight the contribution of each population to the total bycatch based on the “through the water distance” (Jensen et al. 2006) from the natal river to the area of maximum bycatch. This approach makes the assumption that closer stocks contribute more than more distance stocks. However, absolute determination of the correct approach would require monitoring of the contribution of each stock to the overall level of bycatch by analyzing the natal sources of fish in the bycatch. Thus at present there is no simple way of accurately estimating sustainable bycatch levels from multiple individual in-river assessments.

A more direct approach to setting management reference points for shad and river herring would be to use a coastwide assessment model for each species that does not recognize individual river sub-populations. The approach of using a coastwide assessment is not without precedence within the ASMFC framework. For example, tautog (*Tautoga onitis*) is managed by a coastwide limit even though there is evidence of local subpopulations (Atlantic States Marine Fisheries Commission 2005). Adopting a coastwide approach to shad and river herrings would provide a single estimate of the limit reference points for each species associated with MSY. The catch associated with MSY could be calculated, the sustainable coastwide yield could then be divided among the fisheries that target shad and river herrings and the bycatch. The final step required would then be to allocate the coastwide bycatch limit to the individual fisheries that catch shad and river herrings as bycatch.

Here I review the approaches to establishing management reference points for fisheries, and for bycatch in particular, that have been used in other fisheries. In a forthcoming report, Berkson et al. (2010) recommend a hierarchy of assessment approaches for data poor species such as shad and river herrings. Whenever possible, Berksen et al. recommend a detailed quantitative assessment be conducted. When this is not possible, Berkson et al. recommend modified stock reduction approaches (Dick and MacCall 2010; Walters et al. 2006). Finally, Berkson et al. suggest that in particularly data poor assessment a productivity-susceptibility analysis (Patrick et al. 2009) may be helpful in guiding management. Accordingly, I present the results of productivity-susceptibility analysis for the species complex, and the results of stochastic stock reduction analysis with leading management parameters (Forrest et al. 2008) for American shad as an example of an approach to estimating a coastwide MSY and thus establishing a bycatch limit.

2. A REVIEW OF METHODS TO ESTABLISH BYCATCH REFERENCE POINTS IN OTHER FISHERIES

The Magnusson-Stevens Fishery Conservation and Management Act (2006), which regulates fisheries management in federal waters, defines bycatch as “fish which are harvested in a fishery, but which are not sold or kept for personal use.” Technically, this definition would not include shad and river herring as bycatch, because they are often sold for bait when caught in other fisheries. However, Crowder and Murawski (1998) offer a broader definition. They recognize three categories of bycatch: those fish not directly targeted, but kept and landed by the fishery (sometimes referred to as kept bycatch), animals returned either dead or alive to the ocean after capture (discards) and those inadvertently killed by the gear but not retained during deployment (unobserved mortality). Bycatch may also include catches of sub-legal sizes of targeted species in addition to its impact on non-target species. All three categories defined by Crowder and Murawski (op. cit.) impact the population dynamics of non-target species and should ideally be incorporated into any assessment of the impact of the total removals. Yet obtaining estimates of the mortality rates associated with discarding and unobserved mortality is very challenging. Accordingly, conclusions regarding the importance of bycatch in any specific fishery have substantial associated uncertainties. Additionally, assessing the bycatch from any of these sources requires accurate catch monitoring systems – many of which can be easily subverted (Crowder and Murawski 1998). All of these factors combine to make establishing bycatch targets difficult.

Three general approaches can be recognized to establishing bycatch limits and targets. The first is based on practical considerations of culling non-target species from the catch. These limits are usually given as tolerances. For example, the tolerance for American shad in commercial catches in the Potomac River is 2% - that is no more than 2% of any day's harvest can be American shad. Catch tolerances based on minimum sizes are also common. The blue crab fisheries in the Chesapeake Bay have a 10% tolerance for undersize crabs in each bushel harvested (pers. obs.). Tolerances can be effective when a few large animals are caught (e.g., sharks and turtles) that can be isolated and removed from the catch quickly and effectively. In such cases a 0% tolerance may be set. In other cases such as the shrimp fisheries in the Gulf of Mexico for which the total weight of bycatch is 4.5x that of the target shrimp (Harrington et al. 2005), management by a tolerance is not practical. Management by tolerance is often used to provide an incentive to avoid bycatch in commercial fisheries while avoiding excessive regulatory penalties when bycatch is retained (Crowder and Murawski 1998). However, because tolerances are established as a percentage of the catch of the target species, increases in catches of the target species inevitably imply increases in the catch of non-target species. Thus for tolerance approaches to be effective in establishing a limit on bycatch, there has also to be an upper limit in the catch of the target species.

There are few examples where quantitative analyses exist as a foundation for the tolerance adopted. In most cases the tolerances are compromises between the management jurisdictions desire to achieve an end point and the fishermen's ability to meet the end point without undue burden. However, although rare, quantitative analyses are possible. For example, the International Commission for the Conservation of Atlantic Tuna (ICCAT) includes a tolerance for undersize bluefin tuna of 15% of the catch. The size regulation is set to protect juvenile fish. The impacts of the combination of size limits and tolerances have been investigated in this fishery (Porch and Turner 2007). Porch and Turner demonstrated that for the Mediterranean population of bluefin tuna, tolerances were completely unnecessary provided that a sufficiently high minimum size had been established. This was not the case for the Atlantic population for which tight tolerances were needed to avoid negative impacts on early maturing females.

Tolerance-based approaches for setting bycatch targets for the shad and river herring fisheries would effectively be status-quo conditions. Estimating the current level of tolerance for these species would involve gear- and region-specific analyses of the Northeast Fisheries Science Center's observer program and the standardized bycatch reporting program (e.g., Cournane 2010). The nature of the data from these programs would make the results of such analyses highly uncertain. More importantly, in the absence of a quantitative population model, determining the impacts of the empirically determined tolerances in the range of fisheries thought to catch shad and river herring is impossible. Thus at the bare minimum, this approach would require development of a population dynamic model of shad and river herring to connect the empirical tolerance to any recommended bycatch limit.

The two other general approaches to establishing bycatch targets use the same general approach. They both represent removals taken as bycatch as they would any source of targeted removals. In these approaches, a sustainable rate of exploitation or a sustainable yield is calculated. If there is a single fishery responsible for the removals, all of the sustainable yield can be taken in that fishery. This would be the case whether the single source of removals was bycatch or a targeted fishery. In this simple case, the sustainable yield is the limit reference point for either bycatch or for the fishery. If there are multiple fisheries responsible for the removals, the total sustainable yield is allocated among them based on historic patterns or management allocations and each allocation becomes the target for that fishery. The two techniques differ only in how the level of sustainable yield is calculated

The first general approach is demographic modeling which has been used effectively to estimate sustainable yields and bycatch targets for marine mammals, sharks and seabirds. Crouse et al. (1987) were one of the first to apply this approach. Crouse and colleagues used a matrix projection model to estimate the population growth rate (r) of the loggerhead sea turtle (*Carretta caretta*). This analysis indicated that under then-existing conditions the population abundance would continue to decline. The analyses further indicated that “head starting” – that is protecting nestling turtles on the natal beaches - was not capable of producing an increasing population ($r > 0$), but that reducing bycatch of pre-mature adult turtles in shrimp trawls could be effective. Model results suggested a reduction of juvenile mortality of 16-18.5% was necessary for positive r values. More recently Dans et al. (2003) have used a similar approach to estimate the impacts of bycatch on dusky dolphins (*Lagenorhynchus obscures*) off of Patagonia, Argentina. Dans et al. (op. cit) used a stochastic projection modeling approach that accounted for uncertainties in model inputs. In their work, they defined a maximum population growth rate and the associated proportion of the population caught (U) at this rate. In this framework U represents the upper limit of the rate of bycatch that the dolphin population can sustain. Using a standard developed by the International Whaling Commission, Dans et al. then defined U/2 as a safe bycatch target. Dans et al. concluded that bycatch levels were too high by comparing observed catch levels to the U/2 level.

The demographic approach is not restricted solely to large marine species. Diamond and colleagues (Diamond et al. 2000; Diamond et al. 1999) have used a stage within age projection approach to quantify the impacts of bycatch of Atlantic croaker (*Micropogonias undulatus*) in shrimp trawls. These authors concluded that croaker populations in the Gulf of Mexico and along the Atlantic seaboard were declining between 1970-1995 as a result of excessive bycatch (Diamond et al. 2000). Although Diamond et al.’s analyses concluded that the juvenile stage, which was most subject to bycatch, was not the most sensitive life history stage affecting r, they did indicate that a reduction in juvenile mortality of only 5% would be necessary to achieve stabile population abundances. Diamond and colleagues recommended adoption of bycatch reduction devices without providing a numerical bycatch target.

The demographic approach to estimating bycatch targets is attractive in that it places less emphasis on determining the current stock status. Rather, as exemplified by the Dans et al. (2003) approach, it focuses on estimating demographic rates and then projecting the consequences of these rates into the future. As such it is possible to define a maximum potential population growth rate and from this estimate a catch target based on some fraction of the catch at the maximum growth rate (U). However, the application of the demographic approach is not without problems for shad and river herrings. Most obviously, the key vital rates in many of these species varies across their latitudinal range (Leggett and Carscadden 1978). There are several solutions to incorporating this variability in vital rates. One solution would be to conduct analyses specific to each region, but this is the situation we are trying to avoid. A second approach would be to use a single value for each parameter, either taken from a subpopulation in the middle of the range or by using the average of all observed parameter values (e.g., Hewett et al. 2008). A third alternative is to view the different parameter values as representing a statistical distribution and developing stochastic projections such that each individual projection uses parameter values sampled randomly from each distribution.

The final general approach to establishing bycatch targets relies on information from a quantitative stock assessment of the non-target species. Such models can produce estimates of MSY itself or the exploitation rate at MSY, expressed as an annual rate (U_{MSY}) or as an instantaneous rate (F_{MSY}). Stock assessments requires estimates of the total removals from the population. There is no constraint that removals have to be targeted in a fishery – bycatch is simply an additional source of removals. A range of models have been used for this purpose (Quinn and Deriso 1999). Surplus production models are perhaps the most simple. These model population biomass in the aggregate. To fit such models one needs a time series of total removals and a relative abundance time series. An advantage of these models is that catch reference points based on MSY or F_{MSY} are calculated directly in the model. However, surplus production models require substantial contrast in the data for accurate fits. They do not perform well under one-way trips (Hilborn and Walters 1992) – a condition often found in data poor species with high bycatches.

Delay-difference models, such as the Collie-Sissenwine model decompose the aggregate biomass into two or more stages and include a population dynamic relationship between a number of stages (Collie and Sissenwine 1983). For example, the original Collie-Sissenwine model represents the population in two stages: fish that have yet to recruit and fish that have recruited to the fishery. A delay-difference approach using more than two stages is used to estimate stock status and reference points for butterfish in the northwest Atlantic, a species for which bycatch is the dominant source of removals. (Northeast Fisheries Science Center 2010). A delay-difference approach is a feasible one for shad and river herrings as indices for both young of year (pre recruit) and adult (post recruit) are available. However, one drawback with such approaches is that often they do not estimate reference points internally. Instead a separate model, often a per recruit model (Quinn and Deriso 1999), has to be used to determine reference points, against which the results of the delay-difference model can be compared.

Recently more flexible models have been developed which have several advantages over the simpler delay difference models. One such age-structured model is stock reduction analysis (SRA), originally developed by Kimura and Tagart (1982). The original goal of SRA was to use the historical catch record and a simple index of relative abundance to reconstruct possible population trajectories. The main question was “How large must the stock have been, and how large must recruitment rates have been over time, in order that historical catches have caused an assumed or observed relative change in stock size?” (Walters et al. 2006). In SRA the observed harvests are used to estimate an exploitation fraction that drives the population dynamics. In Kimura and Tagart’s original model only one population trajectory was produced conditional on specific estimates of the original population size B_0, and the instantaneous rate of natural mortality, M. This approach was developed further by Walters et al. (2006) using a fully age-structured population dynamics model. Similar to the stochastic demographic models, the Walter et al. technique samples from distributions of key parameters to production many possible population trajectories. Walter's et al.'s approach used a leading parameters scheme involving the parameters of the stock-recruit relationship at equilibrium (Walters et al. 2006). In this revised approach, estimates of the recruitment compensation ratio, κ (Goodyear 1980), and the average recruitment (R_0) and biomass (B_0) of an unharvested population are used to parameterize the initial conditions. The compensation ratio parameter is the ratio of the lifetime reproductive output under fished and unfished conditions. For each modeled population trajectory has an associated suite of estimates for κ, B_0 and R_0. The ensemble of estimates from each trajectory for each parameter represents a probability distribution of likely values for that parameter. Subsequently, Forrest et al. (2008) demonstrated that this model could be reparameterized once more to have leading parameters of the maximum sustainable yield (MSY) and the exploitation rate at MSY (U_{MSY}) – parameters that are of more direct use to managers. Dick and MacCall (2010) have suggested a parallel approach, termed depletion-based stock reduction analysis (DBSRA), that uses a production model rather than a fully age-structured model. The data are available to fit this model for shad and river herrings and will be the foundation of estimated developed in Section 4.

For species not constrained by available data, more elaborate age-structured models can be developed including virtual population analyses and forward-projecting statistical catch at age models. For example, Ehrhardt and Legault (1997) used a virtual population analysis (VPA) framework to explore the impact of uncertainties in bycatch on management reference points in Spanish mackerel (*Scombermorus maculates*) in the Gulf of Mexico. In this analysis, the stock status derived from the VPA and the allowable biological catch (ABC) recommendation was based on a non-equilibrium spawner per recruit analysis that used recruitments sampled from those predicted in the VPA. Uncertainty in the level of bycatch of Spanish mackerel in the Gulf of Mexico shrimp fishery was simulated by drawing 250 samples from the predictions of a general linear model that accounted for yearly, seasonal and area effects in predicted bycatch CPUE. Ehrhardt and Legault reported that introducing bycatch uncertainty led to unique challenges in setting the ABC as there was no overlap between the ABCs estimated among the 250 bootstrap samples. Thus, the precautionary principle would dictate setting the ABC associated with the highest bycatch levels. Ehrhardt and Legault commented that in most assessments there are multiple sources of uncertainty – and that uncertainty in one estimate may be balanced by uncertainty in another data input, such that a more nuanced ABC could be developed.

The available data for shad and river herring do not lend themselves currently to a highly parameterized age-structured assessment model. Efforts to develop such a model would require detailed estimates of the contribution of each stock to the bycatch. Although potential approaches to estimating this are outlined above, the demands of highly parameterized models mean that such estimates are unlikely to be of sufficient reliability. Thus, I would suggest that highly parameterized age-structured models are currently unsuitable as a foundation for estimating bycatch targets in shad and river herrings.

2.1 EMPIRICAL BYCATCH REFERENCE POINTS

Although many of the approaches discussed above provide estimates of the current level of bycatch and the status of the stock, they do not always by themselves help to identify appropriate bycatch targets. Thus, quantitative assessments need to be paired, either explicitly within their structure, or *post hoc* with a separate model to determine biological reference points (Jacobson et al. 2002). However, for many data poor stocks quantitative assessments of any form are a challenge. For these cases empirically derived reference points have been suggested. A range of approaches involving different life history parameters have been suggested (e.g., Hewitt et al. 2007).

For data poor stocks, $F_{MSY} = M$ has been suggested as a reasonable surrogate for a limit reference point (Quinn and Deriso 1999). This approach assumes that each species has evolved to be able to withstand a level of mortality at least as high as the natural rate of mortality it experiences. It makes no assumptions about variation in the rate of natural mortality – that in some circumstance and exploitation rate equivalent to average M might cause a population to decline. For these reasons Thompson (1993) and others have suggested that $F_{MSY}=M$ is too risky for some stocks and that $F_{MSY}=0.8 \cdot M$ is more conservative limit reference point. Even more precautionary, Walters and Martell (2002) have suggested that $F_{MSY}=0.5 \cdot M$ is appropriate for data poor stocks. This precautionary recommendation is similar to that adopted by the International Whaling Commission as described in Dans et al. (2003).

A broader range of empirical approaches to estimating M have been suggested (e.g., Hewitt et al. 2007). Hewitt and colleagues identify 9 different approaches to estimating M, involving estimates of maximum age (t_{max}), age at maturity (t_m), the von Bertalanffy parameters (K, L_∞ and W_∞) and the average water temperature (T). I used all of these approaches to account for scientific uncertainty in M for shad and river herring.

Considerable data were available for American shad with which to estimate empirical reference points (Table 1). Based on the assumed relationship between M and the limit reference point F_{MSY}, I generated distributions of F_{MSY}, $0.8 \cdot F_{MSY}$, and $0.5 \cdot F_{MSY}$ for each species overall and for several key regions based on estimates of life history parameters estimated for each region (Figs. 1-7). A summary of bycatch reference points based on the analysis of the 9 nine different approaches to the 6 different management jurisdictions or river systems is provided in Table 2. Because these values are not really a sample from a statistical viewpoint, the median value is likely the most reliable measure of central tendency. Using Walters and Martell precautionary $0.5 * F_{MSY}$ approach for data poor stocks, appropriate bycatch reference points for American shad are $0.18<F<0.25$ across all systems and $F=0.224$ across its entire range (Table 2).

In contrast there were few data available on which to base similar calculations for alewife and blueback herring (Table 3). The ASMFC river herring stock status document (Atlantic States Marine Fisheries Commission 2008) provides only one estimate of maximum age and only four systems (Maine, Hudson River, and the Cooper and Santee Rivers in South Carolina) provide growth estimates. I could find no other growth estimates in the literature. I applied the maximum age estimate for blueback herring in Maine to all regions. Distributions of F_{MSY}, $0.8\ F_{MSY}$, and $0.5F_{MSY}$ for each species overall and for four management jurisdictions or rivers are provided in Figs. 8-12. In general, these distributions are not as well defined as are the similar distributions for American shad. However, as with American shad, I used Walters and Martell precautionary 0.5*F_{MSY} approach for data poor stocks. The appropriate bycatch reference points for blue back herring is F=0.25 across its entire range (Table 4).

The situation is even worse for alewife for which growth curves from South Carolina are lacking (Table 3). As there were only two regions reporting, I provide only the summary of reference points in Table 4. The appropriate bycatch reference point for alewife is F= 0.29 across its entire range.

3. PRODUCTIVITY-SUSCEPTIBILITY ANALYSIS

Numerous authors have applied multivariate statistical techniques to evaluate the inherent variability in the population dynamics of different fishes. For example, Winemiller and Rose (1992) collated ecological data on life history traits from 216 species of North American fishes (including shad and river herrings) to examine factors likely important in population regulation. These authors recognized three broad categories of life history strategies: periodic, opportunistic and equilibrium that seemed to explain the diversity of life histories observed in North American fishes. Importantly species that were grouped into each strategy shared common factors likely to regulate their population dynamics.

More recently several authors have combined ecological and fisheries data in multivariate analyses in attempts to identify species that might be more vulnerable to exploitation. Recently Ihde et al. (submitted) combined ecological information and fisheries data including information on stock status, the extent of overlap between the distributions of the species and their fisheries and range of sectors exploiting the species. Ihde et al conclude that such ordinations may be of limited use in predicting species vulnerabilities in specific cases as society seems capable of exploiting species regardless of their life history strategy if they either taste good or are perceived to “fight” well.

One multivariate approach that has gained a great deal of attention is productivity-susceptibility analysis or PSA (Patrick et al. 2009). This multivariate was originally developed to address concerns over the impact of bycatch of elasmobranchs in shrimp fisheries in Australia (Stobutzki et al. 2001a; Stobutzki et al. 2001b). These Australian shrimp fisheries take 411 other species as bycatch, with elasmobranchs representing 82% of the entire bycatch (Stobutzki et al. 2001a). Stobutzki and colleagues recognized that species could be arranged against two axes – one being the inherent productivity of the stock (originally termed recovery by Stobutzki) and the second being susceptibility. The productivity axis reflects the rate at which species could recover from periods of low abundance as well as the level of surplus production available for harvest. The second axis reflects the power of the fisheries to catch each species. Importantly, unlike the analyses of Winemiller and Rose (1992) and Ihde et al (submitted) discussed above, the PSA analysis is not a statistical ordination – rather it is a multivariate description of data into predefined categories that are displayed in two dimensions. The multivariate data are summarized and plotted against two axes such that values closer to the origin are more productive and least susceptible. Thus the Euclidean distance from the origin to the point at which a species is plotted is a measure of its overall vulnerability. The categories used in PSA have varied among different applications of the approach. The approach has been adopted in the US as one approach to providing information regarding scientific uncertainty to management as required by the revised Magnuson

Stevens Act (Patrick et al. 2010). As implemented in the NOAA Fisheries Toolbox (http://nft.nefsc.noaa.gov/PSA.html), PSA requires information on 10 different productivity attributes and 12 different susceptibility attributes (Table 5).

I used PSA analysis to determine whether shad and river herrings are unique within the context of the northwest Atlantic coastal shelf fish assemblage. If so, this might imply that reference poitns for these species could not be developed using existing approaches. Expanding on data presented in Patrick et al. (2009), I compiled data for 36 different fish species and stocks that are common throughout the Northwest Atlantic fishery ecosystem (Table 6). Each species was scored with regard to four grouping variables: F-reference point (available or not), overfished status (yes/no), overfishing status (yes/no), and life history mode (anadromous, pelagic, groundfish, reef associated and other). Subsequently and to the extent possible, I scored each species for the 10 productivity traits and the 12 susceptibility traits. All traits were given equal weight in the analysis. Data were entered into the NFT PSA software and plots produced for each of the four groups. Separate analyses were run that represented the shad and river herrings at the species level and also at the subpopulation or river level. There was little additional resolution provided in the PSA stocks by recognizing individual shad and river herring stocks and so only species level groups are discussed further here.

Figure 13 shows the productivity susceptibility plot for 36 species and stocks of fishes in the northwest Atlantic coastal shelf fishery ecosystem identified according to life history mode (anadromous, pelagic, groundfish, structure-associated and other). PSA resulted in a considerable range of values for the targeted species. Figure 13 shows contours of equal vulnerability (blue, green and red lines on Figure 13). Within this PSA, Gulf of Maine cod and spiny dogfish were the most vulnerable species. American shad, alewife and blueback herring were the 22nd, 23rd and 26th most vulnerable species out of the 36 considered. This strongly suggests that there is nothing unique about the species that needs to be considered that would prevent normal assessment methods being applied.

4. DEVELOPMENT OF BYCATCH REFERENCE POINTS FOR AMERICAN SHAD

As a proof of concept of how a coastwide assessment framework could be used to develop management reference points and bycatch limits for American shad and river herrings I used a stochastic stock reduction approach (SRA) to estimate management reference points for American shad. I chose American shad for the proof of concept because of the greater amount of critical data available to me in the current American shad assessments from ASMFC ((Atlantic States Marine Fisheries Commission 2007c). Model development followed Forrest et al. (2008). In this form, the SRA yields estimates of the virgin biomass (B_0), the compensation ratio (κ), the maximum sustainable yield (MSY), and the exploitation rate at MSY (U_{MSY}). The exploitation rate at MSY will be interpreted as the foundation of the overfishing limit, and MSY itself as the catch at the overfishing limit. Table 7 provides the model parameters. A glossary of terms in provided in Appendix A. Tables 8 and 9 provide pseudocode for the initialization scheme and the population dynamics equations respectively. Table 10 provides the likelihood equations used to statistically fit the model to the observed data. The model was implemented in AD model builder (http://admb-project.org), an open-source programming language incorporating a sophisticated non-linear estimation algorithm that is widely used in modern stock assessments. Full model code and the structure of the input data file are given in Appendices B and C. Here, I summarize the general approach and refer to the relevant equations by their numbers. For example, equation T8.2 refers to the survivorship function which is formally defined in Table 8, Equation 2.

At the heart of the stochastic reduction analysis is a fully age-structured population model (Equations T9.7 – T9.10). The model is not fit to age-structured catch or survey data. Rather the model is used to generate aggregate catches and survey indices which are then used to fit to the observed data (Equation T7.13). The population model operates at on an annual time step, t. At any time t, the population is represented by a vector of abundances at age ($N_{a,t}$) from a=0 – A, the maximum age. There was no "plus" group in the model. For each age, I calculated a length at age (L_a - Equation T8.3) from published von Bertalanffy growth parameters (Equation T7.1) and an associated weight at age (W_a - Equation T8.4) using a simple allometric model with previously defined parameter values (Equation T7.2). Values for these and subsequent parameters were taken from estimates in the 2007 ASMFC assessment for the Hudson River (ASFMC 2007b). Age-schedules of maturity (Equation T8.5) and vulnerability to the fishery (Equation T8.6) were also defined. These two schedules were fit to logistic functions using parameter estimates derived from published estimates of the age distribution of spawning fish (Equation T7.3) and the age distribution of fish in surveys (Equation T7.4) outside of the model. Recruitment in the model was determined by a Beverton and Holt (1957) stock recruitment function (Equation T9.9), but alternative stock recruitment models could have been applied. At each time annual time step t, the population egg production E_0 is calculated (Equation T9.8). Numbers at age for older age classes were based on a difference equation (Equation T9.10). The equation projected the population forward at each time step based on the time-independent survivorship function and the calculated year-specific exploitation rate. The exploitation rate was given as the ratio of the observed aggregate catch and the model-estimated vulnerable biomass (EquationT9.7).

The model was initialized using leading parameters of direct relevance to management: MSY and U_{MSY}. Forrest et al (2008) define incidence functions that define the scaled net reproductive rate (Equation T8.7) and the biomass per recruit (Equation T8.8) and then derive numerical methods to estimate these per recruit functions at MSY. Forrest et al. show that the leading parameters can be used to calculate the Beverton and Holt (1957) stock recruitment parameters, s_0 and b (Equations T9.3 – T9.6). As noted by Walters et al. (2006) stochastic stock reduction analysis generates a number of possible population trajectories that could have produced the observed catches and level of reduction in abundance. The approach relies on providing single estimates of the recruitment compensation ratio, κ (Goodyear 1980), and the virgin recruitment R_0, for each population trajectory. To accomplish this I used an observation error model to generate distributions of the leading parameters (Equations T9.13 – T9.15). There was no process error in the model, implying that the only source of error in the estimation is in the observations and not in the population dynamics. Specifically, it assumes no error in the estimated vital rates.

The estimation step involves minimizing the overall negative log-likelihood of a function with three components. The full likelihood equations are provided in Table 10. The first component is the standard errors of the observations. These were assumed to be deviations from a lognormal distribution with mean $\bar{z}$ and standard error σ_t (Equation T10.1). The second component of the likelihood function involved the compensation ratio, κ (Equation T10.2). These were assumed to deviations from a normal distribution with mean 35 and standard deviation 0.1 (Equation T10.4). From experience with trying to fit the model to the observed data, I also included an additional penalty in the likelihood component to limit the calculated annual exploitation fractions, U_t (equation 7), from exceeding 1.0. In cases where the estimation procedure estimated $U_t > 0.75$, U_t was set to 0.75 (Equation T10.3). To calculate the distribution of model parameters I used a Markov Chain, Monte Carlo (MCMC) algorithm. This algorithm samples from the overall distribution of all possible parameter values using a random walk. To ensure that a valid final parameter distribution is achieved, I used 1×10^6 samples, each with a 10,000 sequence burn-in and thinned every 50 samples. This algorithm resulted in a sample of 20,000 values of key leading parameters, from which estimates of the distribution of each parameter could be determined.

Catch records were assembled for American shad from the most recent ASMFC assessment (Atlantic States Marine Fisheries Commission 2007a; Atlantic States Marine Fisheries Commission 2007b). Records from individual states were combined to provide a total coastwide harvest. Each state contributed different amounts to the total harvest of American shad (Figure 14). Extraordinarily high catches were reported early (before 1850) in the time series from the Potomac River. However, the majority of jurisdictions did not consistently report landings until 1960 and even then, the more southern states did not report until the mid 1980s. As indicated in Figure 14, these southern states may account for approximately 25% of the landings in the period 1960-2005. Accordingly, analyses were conducted using the twenty-five year period 1980-2005. I used the Lewes (DE) haul seine CPUE as an index of abundance. This fishery has been active since the 1830’s and effort has been documented since 1925 (Atlantic States Marine Fisheries Commission 2007a). The previous ASMFC shad assessment indicates that it still provides a good index of spawning run strength. Time series for both the total catch and the CPUE index are provided in Figure 15.

Model fits were obtained using a range of values of M, the instantaneous rate of natural mortality, from $0.15<M<0.6$. Model results were highly sensitive to the value of M. Solutions to the model could be determined for $0.15<M<0.4$. Model runs with values outside of this range did not produce the full variance-covariance matrix of parameter values and are considered to be invalid. Results for each valid run of the model are provided in Table 11 and Figures 16-20. With M=0.15, the estimated distribution of B_0 and MSY appeared bimodal (Fig. 16). MSY exhibited peaks at MSY=1.2×10^3 mt and 2.0×10^3 mt. There was a strong positive and apparently linear covariation between estimates of B_0 and MSY. However, there was no bimodality apparent in estimates of κ or UMSY. But, there was a strong covariation between κ and U_{MSY} – exhibiting a concave relationship. Other parameter pairs did not exhibit a strong pattern of covariation. The bimodality in B_0 and MSY remained apparent when M was increased to M=0.2 (Fig. 17). Parameter estimates of B_0 and MSY for M=0.2 were broadly similar to the estimates for M=0.15. The compensation ratio κ and U_{MSY} were well estimated, with unimodal distributions in both parameters at this level of M. All patterns of covariation between parameter pairs were similar. At M=0.25, the bimodality in B_0 and MSY disappeared (Fig. 18). Both parameters appeared well estimated with unimodal parameter distributions. For this level of M, the most likely value of B_0 was 9.5 x 10^3 mt. The most likely value of MSY was 1.15 x 10^3 mt. Model performance, in terms of distributions of parameters, was reduced for values of M > 0.25 (Figs. 19 and 20). In addition for values of M> 0.25, those parameter estimates that were generated were often close to the limits for those parameters set in the model. Accordingly, parameter estimates for models with M>0.25 were considered unreliable.

Based on the pattern of model results described above, I selected results for the model with M=0.25 as being the most reliable foundation for development of bycatch targets in American shad. Specific parameter estimates for this run are given in Table 11. This model yielded the most likely estimate of virgin biomass of ~ 9 x 10^3 mt. This level is considerably lower than known harvests from the late 19th century. The virgin biomass estimate reflects the temporal range of data used in the assessment. Were early catch data to be available from early years from a wider range of systems it might be possible to generate estimates of virgin biomass more reflective of this earlier period. The most likely estimated value of the recruitment compensation ratio, κ (Goodyear 1980), was 25. This level is not unreasonable given reported values in other species (Myers et al. 1999). The most likely estimate of U_{MSY} was 0.51. Using Walters and Martell's (2002) caveat regarding the sustainability of MSY estimates, a more precautionary exploitation rate limit would be U_{MSY}~ 0.25.

The OFL definition of U_{MSY}~ 0.25 can be used to set an ABC – a total allowable biological catch of 575 mt of American shad. This value represents a coastwide limit of total catch of American shad in all fisheries, and does not include any specific buffer for scientific uncertainty. In 2005, the reported targeted landings of American shad coastwide was 370 mt. This figures can be used to calculate a bycatch limit for 2005. The bycatch limit would be the difference between the MSY catch and the reported targeted landings. Thus for 2005, the bycatch limit was 205 mt.

5. DISCUSSION

I have argued here that in-river assessments are an inappropriate foundation for estimating sustainable harvest levels for shad and river herrings now that the principal sources of removals of these species are as bycatch in several ocean fisheries. In contrast, I argue that coastwide assessments of these species are needed if sustainable catch limits are to be developed. As an example of one possible I approach, I used a stochastic stock reduction analysis of American shad to estimate an annual sustainable harvest limit. I combined the output of the stock reduction analysis with guidance from Walters and Martell (2002) that a precautionary approach to fishery management would set overfishing definitions as 0.5 x U_{MSY} in the absence of compelling evidence to the contrary. This approach produced an annual sustainable harvest limit of 575 mt of American shad. In 2005, it was estimated that 370 mt of American shad were harvested in targeted fisheries. Thus, my analysis would suggest that a suitable bycatch limit for this species is 205 mt. A further step would be required to establish specific bycatch limits for individual fisheries. One simple approach to setting bycatch limits for specific at-sea fisheries would be to allocate the overall bycatch limit (e.g., 205 mt) according to the current fraction of the total shad bycatch taken by each sea fishery. However, it might also be possible to negotiate bycatch allocations for each fishery that may deviate from the current distribution. I have not completed these calculations here.

If this approach were adopted as the foundation for developing an annual bycatch limit, the management jurisdictions would be required to establish limits for the targeted fisheries coastwide. This landings would be subtracted from the MSY catch to generate a bycatch limit. Management jurisdictions would have the flexibility to adjust the bycatch limit by adjusting the level of targeted landings allowed. If all targeted landings were banned, the bycatch limit could increase to the level of the MSY catch. If this option is not selected and targeted landings continue to be allowed, then the bycatch limit will simply be the difference between the constant MSY catch and the annual allowance for the targeted fishery.

In the framework adopted here, I used a coastwide stochastic stock reduction analysis to estimate a coastwide MSY catch. Stock reduction analysis is a data-poor stock assessment method that has been applied in a range of fisheries as diverse as hake (*Merluccius spp.*) off the southwest coast of Africa (Forrest et al. 2008) and blue crab (*Callinectes sapidus*) in Florida (Murphy et al. 2007). Most recently Dick and McCall (2010) applied it to develop reference points for 50 species of groundfish on the Pacific coast of the US. Berkson et al. (2010) have recommended the approach for data-poor species in general. In the form implement here, stock reduction analysis is a Bayesian approach to estimating the likely distribution of key management parameters, including MSY and U_{MSY}, that best explain an observed time series of catch and survey abundance. Like any stock assessment model, stock reduction analysis is sensitive to the assumed model structure, the input data and starting conditions. Thus, evaluating the reliability of the reference points developed from these methods requires careful evaluation.

Arguably, the biggest single assumption made in modeling shad and river herrings was to take a coastwide approach. All previous assessments for these species have taken a river or state specific approach (e.g., Atlantic States Marine Fisheries Commission 2007c; Atlantic States Marine Fisheries Commission 2008). It could be argued that such an approach is necessary given the high variable life history traits observed in these species (Leggett and Carscadden 1978). Further, historically the principal fisheries for these species have been local, “in-river” fisheries. Under such circumstances, in-river assessments are certainly justified. However, the situation now is fundamentally different. Certainly considerable life history variation remains. But, most major in-river fisheries have now been closed and the principal sources of removals from populations come from bycatch in coastal fisheries. There are two ways of meeting this new challenge. One would be to continue the state-by-state assessments. This would require an approach to prorating the ocean removals back to each natal system. Although such an approach might be possible, no extant data are available on which to base such a proration. Thus, state-by-state approaches must make strong inferences regarding the proportion of bycatch attributable to each natal river. As an alternative, I chose to conduct a single coastwide assessment that ignores the variability in life histories along the coast and considers shad and river herrings as representing single well mixed populations. Using this approach, there is no requirement to allocate bycatch. This approach also results in a single estimated bycatch limit. The bycatch target represents an aggregate level of bycatch believed to be sustainable regardless of the source of bycatch.

The accuracy of any model results depends critically on the quality of the input data. There are three sources of input data for the stochastic stock reduction analysis: catch data, survey data and vital rates. I used catch data for American shad from 1980-2005 reported by coastal states from Maine to Florida in the most recent ASMFC assessment (Atlantic States Marine Fisheries Commission 2007a; Atlantic States Marine Fisheries Commission 2007b; Atlantic States Marine Fisheries Commission 2007c). I have assumed that these reports are complete and include all sources of removals from the American shad stock coastwide. Data from earlier periods were available for some states, but the majority of southern states did not consistently report landings until this period. The extent to which the reported landings include bycatch was unclear. No effort was made to include recreational landings in the catch time series because of concerns over the reliability of the estimates. Moreover, efforts to include recreational landings would have further shortened the time series. It is important that future analyses critically examine the catch time series to fully evaluate all sources of information and, when possible, to develop the longest, most reliable time series of catches possible.

The model also used the Lewes, DE haul seine index as an estimate of abundance. Several potential concerns may arise from the use of this index. The haul seine is not a fishery-independent survey. Some fishery independent surveys were available from many of the individual states. In many cases however, these surveys were not consistently conducted each year. In contrast, the haul seine fishery has been consistently conducted since 1925. An additional concern that might be raised over the use of the Lewes survey is that it is highly localized and may not reflect coastwide abundance. The same concern applies to many of the individual state surveys. The sole coastwide surveys are conducted by the Northeast Fisheries Science Center. These surveys target groundfish and use a large bottom trawl as the survey gear and are widely recognized as highly inefficient for pelagic species such as American shad. Thus on balance, I suggest that the Lewes survey is no worse than any of the other alternatives. Furthermore the decision to use the Lewes survey was made prior to any model fitting – it was an *a priori* rather than *post hoc* choice based on considerations of consistency and coverage. However, additional work on the impact of the use of alternative surveys in the model would be beneficial.

Input parameters in stock assessments can affect the reliability of model results. The stochastic stock reduction requires estimates of the von Bertalanffy growth parameters, weight at age, and the schedules of maturity and recruitment to the fishery. For American shad I chose to parameterize the model using values reflective of studies conducted in the Hudson River, NY. It is clear that the growth, maturity and survival schedules for this species vary from location to location. Selecting one single parameter set cannot fully reflect the diversity of vital rates known for this species. Several state agencies have conducted extensive sampling of American shad in their jurisdictions. Of those that provided all required variables (NH, RI, NY, VA and NC), I selected the Hudson River, NY as approximately the centre of this distribution. I chose to use an internally consistent set of estimates from the same system rather than trying to average all values which might not reflect the inherent pattern of covariation among parameters. I would recommend that exploring the impact of alternative parameter sets on model results is a high priority for future work. This could be done by simply using parameter sets from different states and regions, or by resampling from all published parameters regardless of their geographic source.

There was insufficient information in the data for American shad to estimate all four parameters: MSY, U_{MSY}, M and κ. After a period of evaluation, I determined that M could not be effectively estimated in the model and thus all runs of the assessment model estimated only MSY, U_{MSY} and κ. As with almost all assessment models, results from the stochastic stock reduction analyses were extremely sensitive to the specific value of M. Indeed for American shad there was only a limited range of M for which valid solutions to the estimation set could be achieved. It could be that a fuller exploration of all possible input data sets and suites of parameter estimates could reveal some combinations for which M could have been estimated. Such a fuller exploration was not possible within the time constraints of this project but should be conducted in the future.

The assessment model also requires Bayesian priors on the distribution of MSY, U_{MSY} and κ. For each parameter, I used fairly strong priors to limit the ability of the estimation procedure to "walk" to invalid regions of the parameter space. I also included a penalty on estimated exploitation rates, such that a penalty was imposed if the estimation procedure produced estimates > 0.75. This did not restrict the models to exploitation estimates < 0.75, rather it simply imposed a penalty on such estimates. However, additional work on whether the model fits reflect information provided to model in the priors or in the data is required.

In an effort to assess the reliability of model estimates, I compared the empirically derived limit reference point for American shad (F=0.224, Table 2), derived from life history parameters with that generated from the model. Using the same value of natural mortality used in the model (M=0.25), the empirical F-based limit reference point translates to an annual exploitation rate of U=0.62. If we use the Walters and Martell (2002) precautionary approach, this estimate becomes U=0.31, which compares favorably with the U=0.25 calculated in the model. It is true that the two estimates are not entirely independent as the estimates of vital rates used in the SRA were used in the empirical estimates. However, the model was fit to the catch and survey time series rather than the life history parameters. The congruence of the two exploitation rate estimates does suggest that the two approaches are at least producing compatible estimates.

The several next steps required to validate and improve on the analyses presented here. The model implemented here was a specific form of stock reduction analysis that utilized an age structured population model. Alternative formulations are also available. Recently Dick and MacCall (2010) used a stock reduction approach to estimating reference points in 50 west coast groundfishes, term "depletion-biased stock reduction analysis" or DB-SRA. Their modeling approach uses an aggregate biomass dynamic model at the heart of the assessment and does not require use of an independent index of abundance. If such an approach were used here, the assessment would be able to use the longer catch time series that are already available because the survey indices currently limit the use of these earlier data.

Additional time and effort in assembling the available landings database will pay dividends in terms of improving the reliability of bycatch targets. For example, Hall (2009) has reconstructed likely historical run sizes for river herring in coastal Maine. In her analysis indicates that there was a near total blockage of all rivers by 1860, resulting in a state-wide loss of 6.5 billion alewife from 1600-1900. The historical approach to fisheries may enable catch time series to be reconstructed in earlier periods. Even beyond these more demanding tasks, a simple task of ensuring that data from all sources of removals of shad and river herring are assembled. I did not have time here to verify and ensure that all data reported in previously conducted studies and assessments were accurate and comprehensive. It is possible that important sources of removals were overlooked. Effort should also be expended to evaluate how best to include data on sources of removals that have only been collected for short periods of time – e.g., recreational landings, and removals by foreign fleets prior to the extension of jurisdiction to 200 nautical miles in 1986. Failure to include important sources of mortality will likely substantially alter calculated bycatch limits.

The Fishery Management and Conservation Act (2006) requires that management jurisdictions account for scientific uncertainty when establishing the allowable biological catch, whether from targeted fisheries or from bycatch. In developing guidelines to achieve this goal, the National Marine Fisheries Service requires that the catch associated with OFL is selected such that there is no more than a 50% chance of the OFL being exceeded. Regional management jurisdictions have further interpreted this to mean that incorporation of scientific uncertainty will mean that catch levels lower than those forecast to have a 50% chance of exceeding the OFL are used to set the ABC. No consideration of scientific uncertainty has been included here. However, it could be argued that the adoption of the Walters and Martell (2002) standard of using $0.5 \cdot U_{MSY}$ acts as a buffer against scientific uncertainty. In the framework of FMCA, selection of the Walters and Martell standard would implicitly be setting the ABC as a catch level that had a 25% chance of exceeding the OFL. Buffers of such magnitude are under consideration by the South Atlantic, the Mid Atlantic and the New England Fishery Management Councils.

Considerable further work must be invested in this model framework before the catch limits and their associated bycatch limits are used in management. Not only, as discussed above must considerable analytical work be conducted to verify the reliability and sensitivity of model results, but management agencies must begin discussion over the allocation of the catch limits so calculated. The bycatch limit for 2005 of 205 mt of American shad that I have presented here presumes that the existing targeted fisheries should continue. Reductions in the levels of removals by targeted fisheries will increase the bycatch limit. Management agencies should begin a broad and comprehensive discussion with stakeholders of what the best allocation of shad and river herrings among the diverse users (Miller et al. 2010). However, in principle, the approach presented here represents one possible path forward in estimating bycatch limits for shad and river herrings.

LITERATURE CITED

1993. Atlantic Coastal Fisheries Cooperative Management Act.
2006. Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006. 16 U.S.C. 1801.

Alverson, D. L., and M. J. Carney. 1975. Graphic Review of Growth and Decay of Population Cohorts. Journal Du Conseil 36(2):133-143.

Alverson, D. L., M. H. Freeberg, J. G. Pope, and S. A. Murawski. 1994. A global assessment of fisheries bycatch and discards. United Nations Food and Agriculture Organization, 339, Rome.

Atlantic States Marine Fisheries Commission. 2005. Tautog Stock Assessment. Atlantic States Marine Fisheries Commission, Washington DC.

Atlantic States Marine Fisheries Commission. 2007a. American shad stock assessment report for peer review: Volume II. Atlantic States Marine Fisheries Commission, 07-01 (Supplement), Washington, D.C.

Atlantic States Marine Fisheries Commission. 2007b. American shad stock assessment report for peer review: Volume III. Atlantic States Marine Fisheries Commission, Washington, D. C.

Atlantic States Marine Fisheries Commission. 2007c. American Shad Stock Assessment Report, Volume 1. Atlantic States Marine Fisheries Commission, Washington, DC.

Atlantic States Marine Fisheries Commission. 2008. 2008 river herring stock status report. Atlantic States Marine Fisheries Commisssion, Washington DC.

Berkson, J., and coauthors. 2010. Guidance on setting ABCs when only average catch is known. American Fisheries Society, Pittsburg, PA.

Beverton, R. J. H., and S. J. Holt. 1957. On the Dynamics of Exploited Fish Populations, volume 19. Her Majesty's Stationary Office, London, UK.

Brown, B. L., P. E. Smouse, J. M. Epifanio, and C. J. Kobak. 1999. Mitochondrial DNA mixed-stock analysis of American shad: coastal harvests are dynamic and variable. Transactions of the American Fisheries Society 128(6):977-994.

Charnov, E. L. 1993. Life History Invariants: Some Explorations of Symmetry in Evolutionary Ecology. Oxford University Press, Oxford.

Collie, J. S., and M. P. Sissenwine. 1983. Estimating population size from relative abundance data measured with error. Canadian Journal of Fisheries and Aquatic Science 40:1871-1879.

Cournane, J. M. 2010. Developing alternatives to mitigate river herring bycatch at sea. Pages 24 *in*. Institute for the Study of Earth, Ocean and Space, Universtiy of New Hampshire, Durham, NH.

Crouse, D. T., L. B. Crowder, and H. Caswell. 1987. A Stage-Based Population-Model for Loggerhead Sea-Turtles and Implications for Conservation. Ecology 68(5):1412-1423.

Crowder, L. B., and S. A. Murawski. 1998. Fisheries bycatch: Implications for management. Fisheries 23(6):8-17.

Dans, S. L., M. K. Alonso, S. N. Pedraza, and E. A. Crespo. 2003. Incidental catch of dolphins in trawling fisheries off Patagonia, Argentina: Can populations persist? Ecological Applications 13(3):754-762.

Diamond, S. L., L. G. Cowell, and L. B. Crowder. 2000. Population effects of shrimp trawl bycatch on Atlantic croaker. Canadian Journal of Fisheries and Aquatic Sciences 57(10):2010-2021.

Diamond, S. L., L. B. Crowder, and L. G. Cowell. 1999. Catch and bycatch: The qualitative effects of fisheries on population vital rates of Atlantic croaker. Transactions of the American Fisheries Society 128(6):1085-1105.

Dick, E. J., and A. D. MacCall. 2010. Estimates of sustainable yield for 50 data-poort stocks in the Pacific coast groundfish fishery management plan. Department of Commerce, NOAA-TM-NMFS-SWFSC-460, Washington, DC.

Ehrhardt, N. M., and C. M. Legault. 1997. The role of uncertainty in fish stock assessment and management: A case study of the Spanish mackerel, Scomberomorus maculatus, in the US Gulf of Mexico. Fisheries Research 29(2):145-158.

Forrest, R. E., S. J. D. Martell, M. C. Melnychuk, and C. J. Walters. 2008. An age-structured model with leading management parameters, incorporating age-specific selectivity and maturity. Canadian Journal of Fisheries and Aquatic Sciences 65(2):286-296.

Goodyear, C. P. 1980. Compensation in fish populations. Pages 253-280 *in* C. H. Hocutt, and J. R. Stauffer, editors. Biological Monitoring of Fish. Lexington Books, Lexington, MA.

Hall, C. J. 2009. Damming of Maine Watersheds and the Consequences for Coastal Ecosystems with a Focus on the Anadromous River Herring (Alosa pseudoharengus and Alosa aestivalis): A Four Century Analysis. Masters. Stony Brook University, Stony Brook.

Harrington, J. M., R. A. Myers, and A. A. Rosenberg. 2005. Wasted fishery resources: discarded by-catch in the USA. Fish and Fisheries 6(4):350-361.

Hewitt, D. A., and coauthors. 2007. Direct and indirect estimates of natural mortality for blue crab. Transactions of the American Fisheries Society 136:1030-1040.

Hilborn, R., and C. J. Walters. 1992. Quantitative Fisheries Stock Assessment: Choices, Dynamics, & Uncertainty. Chapman and Hall, New York.

Hoenig, J. M. 1983. Empirical use of longevity data to estimate mortality rates. Fishery Bulletin 82:898-902.

Ihde, T. F., M. J. Wilberg, D. L. Loewensteiner, D. H. Secor, and T. J. Miller. submitted. Increasing importance of marine recreational fishing: challenges for management. Fish and Fisheries.

Jacobson, L. D., S. X. Cadrin, and J. R. Weinberg. 2002. Tools for estimating surplus production and F-MSY in any stock assessment model. North American Journal of Fisheries Management 22(1):326-338.

Jensen, O. P., M. C. Christman, and T. J. Miller. 2006. Landscape-based geostatistics: a case study of the distribution of blue cab in Chesapeake Bay. Environmetrics 16:1-17.

Jones, C. M. 2006. Estuarine and diadromous fish metapopulations. Pages 119-154 *in* J. P. Krtizer, and P. F. Sale, editors. Marine Metapopulations. Academic Press, Burlington, MA.

Kimura, D. K., and J. V. Tagart. 1982. Stock Reduction Analysis, Another Solution to the Catch Equations. Canadian Journal of Fisheries and Aquatic Sciences 39(11):1467-1472.

Leggett, W. C., and J. E. Carscadden. 1978. Latitudinal variation in reproductive characteristics of American shad (*Alosa sapidissima*): Evidence for population specific life history strategies in fish. Can J. Fish. Aquat. Sci. Spec. Publ. 35:1469-1478.

Leggett, W. C., and R. R. Whitney. 1972. Water temperature and the migrations of American shad. Fishery Bulletin 70:659-670.

Lewison, R. L., L. B. Crowder, A. J. Read, and S. A. Freeman. 2004. Understanding impacts of fisheries bycatch on marine megafauna. Trends in Ecology & Evolution 19(11):598-604.

Limburg, K. E., and J. R. Waldman. 2009. Dramatic declines in North Atlantic diadromous fishes. BioScience 59(11):955-965.

Lorenzen, K. 1996. The relationship between body weight and natural mortality in juvenile and adult fish: A comparison of natural ecosystems and aquaculture. Journal of Fish Biology 49(4):627-647.

McPhee, J. 2002. The Founding Fish. Farrar, Straus and Giroux, New York, NY.

Miller, T. J., and coauthors. 2010. FishSmart: An innovative role for science in stakeholder-centered approaches to fisheries management. Fisheries 35(9):424-433.

Moore, J. E., and coauthors. 2009. A review of marine mammal, sea turtle and seabird bycatch in USA fisheries and the role of policy in shaping management. Marine Policy 33(3):435-451.

Murphy, M. D., A. L. McMillen-Jackson, and B. Mahmoudi. 2007. A stock assessment for blue crab, *Callinectes sapidus*, in Florida waters. Florida Fish and Wildlife Conservation Commission, IHR 2007-006, St. Petersburg, FL.

Myers, R. A., K. G. Bowen, and N. J. Barrowman. 1999. Maximum reproductive rate of fish at low population sizes. Canadian Journal of Fisheries and Aquatic Sciences 56(12):2404-2419.

Northeast Fisheries Science Center. 2010. 49th Northeast regional stock assessment workshop: Assessment report. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, 10-03, Woods Hole, MA.

Patrick, W. S., and coauthors. 2010. Using productivity and susceptibility indices to assess the vulnerability of United States fish stocks to overfishing. Fishery Bulletin 108(3):305-322.

Patrick, W. S., and coauthors. 2009. Use of Productivity and Susceptibility Indices to Determine Stock Vulnerability, with Example Applications to Six U.S. Fisheries. U. S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NMFS-F/SPO-101, Washington, DC.

Pauly, D. 1980. On the interrelationship between natural mortality, growth parameters and mean environmental temperature in 175 fish stocks. Journal Du Conseil 39(2):175-192.

Porch, C. E., and S. C. Turner. 2007. Preliminary analyses of the possible impacts of alternative minimum size limits for Atlantic bluefin tuna. Pages 1171-1178 *in* Collected Volumes of Scientific Papers of ICCAT.

Quinn, T. J., II, and R. B. Deriso. 1999. Quantitative Fish Dynamics. Oxford University Press, Oxford, UK.

Restrepo, V. R., and coauthors. 1998. Technical Guidance On the Use of Precautionary Approaches to Implementing National Standard 1 of the Magnuson-Stevens Fishery Conservation and Management Act. Pages 56 *in* U. S. D. o. Commerce, editor, Washington DC.

Roff, D. A. 1984. The evolution of life history parameters in teleosts. Canadian Journal of Fisheries and Aquatic Science 41:989-1000.

Scott, W. B., and M. G. Scott. 1998. Atlantic Fishes of Canada. Department of Fisheries and Oceans Canada, 219, Ottawa, Ontario, Canada.

Stobutzki, I., M. Miller, and D. Brewer. 2001a. Sustainability of fishery bycatch: a process for assessing highly diverse and numerous bycatch. Environmental Conservation 28(2):167-181.

Stobutzki, I. C., M. J. Miller, P. Jones, and J. P. Salini. 2001b. Bycatch diversity and variation in a tropical Australian penaeid fishery; the implications for monitoring. Fisheries Research 53(3):283-301.

Thompson, G. G. 1993. A proposal for threshold stock size and maximum fishing mortality rate. Pages 303-320 *in* J. J. Hunt, S. J. Smith, and D. Rivard, editors. Risk Evaluation and Biological Reference Points for Fisheries Management, volume 120. National Research Council Canada, Ottawa, Ontario.

Walters, C., and S. J. D. Martell. 2002. Stock assessment needs for sustainable fisheries management. Bulletin of Marine Science 70(2):629-638.

Walters, C. J., S. J. D. Martell, and J. Korman. 2006. A stochastic approach to stock reduction analysis. Canadian Journal of Fisheries and Aquatic Sciences 63(1):212-223.

Winemiller, K. O., and K. A. Rose. 1992. Patterns of life-history diversification in North-American fishes - Implications for population regulation. Canadian Journal of Fisheries and Aquatic Sciences 49(10):2196-2218.

Table 1. Parameter definitions and values used to estimate empirical bycatch reference points for American shad.

Region	Equation	Parameters	F_{MSY}	$0.8*F_{MSY}$	$05*F_{MSY}$	Reference
Maine	M=g/tmax	g=5.5-6.5 tmax=11	0.5 -0.59	0.4-0.47	0.25-0.29	(Charnov 1993)
	M=exp(1.44-0.928ln(tmax)	tmax=11	0.46	0.36	0.23	(Hoenig 1983)
New Hampshire	M=g/tmax	g=5.5-6.5 tmax=11	0.5 -0.59	0.4-0.47	0.25-0.29	(Charnov 1993)
	M=exp(1.44-0.928ln(tmax)	tmax=11	0.46	0.36	0.23	(Hoenig 1983)
	M=X*K	X=0.47-2 K=0.243	0.11-0.48	0.09-0.38	0.05-0.24	(Charnov 1993)
	$M=3K/(e^{0.38Ktmax}-1)$	K=0.243 tmax=11	0.41	0.33	0.21	(Alverson and Carney 1975)
	$M=3K/(e^{Ktm}-1)$	K=0.243 tm=5.7	0.24	0.19	0.12	(Roff 1984)
	Log M=-0.0066-0.279*log(L∞)+0.6543Log(K)+0.4634Log(T)	L∞=64.1 K=0.243 T=11	0.44	0.35	0.22	(Pauly 1980)
	$M=3*W^{-0.288}$	W=3.51	0.28	0.23	0.14	(Lorenzen 1996)
	Average					
Rhode Island	M=g/tmax	g=5.5-6.5 tmax=11	0.5 -0.59	0.4-0.47	0.25-0.29	(Charnov 1993)
	M=exp(1.44-0.928ln(tmax)	tmax=11	0.46	0.36	0.23	(Hoenig 1983)
	M=X*K	X=0.47-2 K=0.28	0.13-0.56	0.10-0.45	0.06-0.28	(Charnov 1993)
	$M=3K/(e^{0.38Ktmax}-1)$	K=0.28 tmax=11	0.381	0.30	0.19	(Alverson and Carney 1975)

	$M=3K/(e^{Ktm}-1)$	K=0.28 tm=5.7	0.21	0.17	0.10	(Roff 1984)
	$\text{Log } M=-0.0066-0.279*\log(L\infty)+0.6543\text{Log}(K)+0.4634\text{Log}(T)$	L∞=57.1 K=0.28 T=11	0.50	0.40	0.25	(Pauly 1980)
	$M=3*W^{-0.288}$	W=2.434	0.32	0.25	0.16	(Lorenzen 1996)
Hudson River, NY	$M=g/tmax$	g=5.5-6.5 tmax=14	0.39 -0.46	0.31 -0.37	0.19-0.23	(Charnov 1993)
	$M=\exp(1.44-0.928\ln(tmax))$	tmax=14	0.36	0.29	0.18	(Hoenig 1983)
	$M=X*K$	X=0.47-2 K=0.4	0.18-0.8	0.15-0.64	0.09-0.4	(Charnov 1993)
	$M=3K/(e^{0.38Ktmax}-1)$	K=0. 4 tmax=14	0.16	0.13	0.08	(Alverson and Carney 1975)
	$M=3K/(e^{Ktm}-1)$	K=0. 4 tmax=4.7	0.21	0.17	0.10	(Roff 1984)
	$\text{Log } M=-0.0066-0.279*\log(L\infty)+0.6543\text{Log}(K)+0.4634\text{Log}(T)$	L∞=58.7 K=0. 4 T=12	0.65	0.52	0.32	(Pauly 1980)
	$M=3*W^{-0.288}$	W=3.51	0.31	0.25	0.15	(Lorenzen 1996)
York River, VA	$M=g/tmax$	g=5.5-6.5 tmax=12	0.45 -0.54	0.36 -0.43	0.22-0.27	(Charnov 1993)
	$M=\exp(1.44-0.928\ln(tmax))$	tmax=12	0.42	0.33	0.21	(Hoenig 1983)
	$M=X*K$	X=0.47-2 K=0.36	0.17-0.72	0.13-0.57	0.08-0.36	(Charnov 1993)
	$M=3K/(e^{0.38Ktmax}-1)$	K=0. 36 tmax=12	0.26	0.21	0.13	(Alverson and Carney 1975)
	$M=3K/(e^{Ktm}-1)$	K=0. 36 tmax=4.6	0.26	0.21	0.13	(Roff 1984)
	$\text{Log } M=-0.0066-0.279*\log(L\infty)+0.6543\text{Log}(K)+0.4634\text{Log}(T)$	L∞=56.5 K=0. 36 T=19	0.76	0.61	0.38	(Pauly 1980)

	$M=3*W^{-0.288}$	W=2.36	0.32	0.26	0.16	(Lorenzen 1996)
Albermarle River, NC	M=g/tmax	g=5.5-6.5 tmax=10	0.55 -0.65	0.44 -0.52	0.28-0.32	(Charnov 1993)
	M=exp(1.44-0.928ln(tmax)	tmax=10	0.50	0.40	0.51	(Hoenig 1983)
	M=X*K	X=0.47-2 K=0.38	0.18-0.75	0.14-0.60	0.09-0.38	(Charnov 1993)
	$M=3K/(e^{0.38Ktmax}-1)$	K=0. 38 tmax=10	0.36	0.29	0.18	(Alverson and Carney 1975)
	$M=3K/(e^{Ktm}-1)$	K=0. 38 tmax=4.6	0.23	0.19	0.12	(Roff 1984)
	Log M=-0.0066-0.279*log(L∞)+0.6543Log(K)+0.4634Log(T)	L∞=60.8 K=0. 38 T=21	0.80	0.64	0.40	(Pauly 1980)
	$M=3*W^{-0.288}$	W=2.97	0.29	0.24	0.15	(Lorenzen 1996)

Table 2. Summary of bycatch reference points for American shad derived from 9 different empirical estimates of M for 6 different systems.

	F MSY		0.8 FMSY		0.5 FMSY	
	Mean	Median	Mean	Median	Mean	Median
Overall	0.427	0.448	0.341	0.358	0.213	0.224
Maine	0.516	0.500	0.413	0.400	0.258	0.250
NH	0.392	0.439	0.313	0.351	0.196	0.220
RI	0.405	0.456	0.324	0.365	0.202	0.228
NY	0.393	0.365	0.315	0.292	0.197	0.182
York	0.434	0.421	0.347	0.337	0.217	0.210
Albermarle	0.480	0.498	0.384	0.399	0.240	0.249

Table 3. Parameter definitions and values used to estimate empirical bycatch reference points for river herrings.

Region	Equation	Parameters	F_{MSY}	$0.8*F_{MSY}$	$05*F_{MSY}$	Reference
		Blueback Herring				
Maine	M=g/tmax	g=5.5-6.5 tmax=11	0.5 -0.59	0.4-0.47	0.25-0.29	(Charnov 1993)
	M=exp(1.44-0.928ln(tmax)	tmax=11	0.46	0.36	0.23	(Hoenig 1983)
	M=X*K	X=0.47-2 K=0.41	0.19-0.82	0.15-0.65	0.09-0.41	(Charnov 1993)
	$M=3K/(e^{0.38Ktmax}-1)$	K=0.41 tmax=11	0.27	0.22	0.13	(Alverson and Carney 1975)
	$M=3K/(e^{Ktm}-1)$	K=0.41 tm=4	0.30	0.24	0.15	(Roff 1984)
	Log M=-0.0066-0.279*log(L∞)+0.6543Log(K)+0.4634Log(T)	L∞=33 K=0.41 T=11	0.74	0.59	0.37	(Pauly 1980)
	$M=3*W^{-0.288}$	W=0.416	0.53	0.42	0.26	(Lorenzen 1996)
Hudson River, NY	M=g/tmax	g=5.5-6.5 tmax=11	0.5 -0.59	0.4-0.47	0.25-0.29	(Charnov 1993)
	M=exp(1.44-0.928ln(tmax)	tmax=11	0.46	0.36	0.23	(Hoenig 1983)
	M=X*K	X=0.47-2 K=0.7	0.33-1.4	0.26-1.1	0.16-0.7	(Charnov 1993)
	$M=3K/(e^{0.38Ktmax}-1)$	K=0.7 tmax=11	0.12	0.09	0.06	(Alverson and Carney 1975)
	$M=3K/(e^{Ktm}-1)$	K=0.7 tm=4	0.14	0.11	0.07	(Roff 1984)
	Log M=-0.0066-0.279*log(L∞)+0.6543Log(K)+0.4634Log(T)	L∞=27.6 K=0.7 T=11	1.11	0.88	0.55	(Pauly 1980)
	$M=3*W^{-0.288}$	W=0.233	0.62	0.50	0.31	(Lorenzen 1996)

Cooper River, SC	M=g/tmax	g=5.5-6.5 tmax=11	0.5 -0.59	0.4-0.47	0.25-0.29	(Charnov 1993)
	M=exp(1.44-0.928ln(tmax)	tmax=11	0.46	0.36	0.23	(Hoenig 1983)
	M=X*K	X=0.47-2 K=0.63	0.29-1.26	0.23-1.01	0.15-0.63	(Charnov 1993)
	$M=3K/(e^{0.38Ktmax}-1)$	K=0.63 tmax=11	0.15	0.12	0.07	(Alverson and Carney 1975)
	$M=3K/(e^{Ktm}-1)$	K=0.63 tm=5.7	0.16	0.12	0.08	(Roff 1984)
	Log M=-0.0066-0.279*log(L∞)+0.6543Log(K)+0.4634Log(T)	L∞=28.65 K=0.63 T=11	1.02	0.82	0.51	(Pauly 1980)
	$M=3*W^{-0.288}$	W=0.261	0.60	0.48	0.30	(Lorenzen 1996)
Santee River, SC	M=g/tmax	g=5.5-6.5 tmax=11	0.5-0.59	0.4-0.47	0.25-0.29	(Charnov 1993)
	M=exp(1.44-0.928ln(tmax)	tmax=11	0.46	0.36	0.23	(Hoenig 1983)
	M=X*K	X=0.47-2 K=0.61	0.29-1.22	0.29-0.97	0.14-0.61	(Charnov 1993)
	$M=3K/(e^{0.38Ktmax}-1)$	K=0. 61 tmax=11	0.15	0.12	0.08	(Alverson and Carney 1975)
	$M=3K/(e^{Ktm}-1)$	K=0. 61 tmax=4	0.17	0.14	0.09	(Roff 1984)
	Log M=-0.0066-0.279*log(L∞)+0.6543Log(K)+0.4634Log(T)	L∞=28.1 K=0. 61 T=11	1.05	0.84	0.52	(Pauly 1980)
	$M=3*W^{-0.288}$	W=3.51	0.61	0.49	0.31	(Lorenzen 1996)
			Alewife			
Maine	M=g/tmax	g=5.5-6.5 tmax=9	0.61-0.72	0.49-0.58	0.30-0.36	(Charnov 1993)
	M=exp(1.44-0.928ln(tmax)	tmax=9	0.55	0.44	0.27	(Hoenig 1983)

	M=X*K	X=0.47-2 K=0.41	0.19-0.82	0.15-0.66	0.09-0.41	(Charnov 1993)
	$M=3K/(e^{0.38Ktmax}-1)$	K=0. 41 tmax=9	0.40	0.32	0.20	(Alverson and Carney 1975)
	$M=3K/(e^{Ktm}-1)$	K=0. 41 tmax=3	0.51	0.41	0.25	(Roff 1984)
	Log M=-0.0066-0.279*log(L∞)+0.6543Log(K)+0.4634Log(T)	L∞=35.3 K=0. 41 T=11	0.73	0.58	0.36	(Pauly 1980)
	$M=3*W^{-0.288}$	W=0.754	0.44	0.36	0.22	(Lorenzen 1996)
Hudson River, NY	M=g/tmax	g=5.5-6.5 tmax=9	0.61-0.72	0.19-0.58	0.30-0.36	(Charnov 1993)
	M=exp(1.44-0.928ln(tmax)	tmax=9	0.55	0.44	0.27	(Hoenig 1983)
	M=X*K	X=0.47-2 K=0.79	0.37-1.58	0.29-1.26	0.18-0.79	(Charnov 1993)
	$M=3K/(e^{0.38Ktmax}-1)$	K=0. 79 tmax=9	0.17	0.14	0.09	(Alverson and Carney 1975)
	$M=3K/(e^{Ktm}-1)$	K=0. 79 tmax=3	0.24	0.19	0.12	(Roff 1984)
	Log M=-0.0066-0.279*log(L∞)+0.6543Log(K)+0.4634Log(T)	L∞=60.8 K=0. 79 T=11	1.23	0.98	0.61	(Pauly 1980)
	$M=3*W^{-0.288}$	W=0.219	0.63	0.51	0.32	(Lorenzen 1996)

Table 4. Summary of bycatch reference points for river herrings derived from 9 different empirical estimates of M for up to 4 different systems.

	F MSY		0.8 FMSY		0.5 FMSY	
	Mean	Median	Mean	Median	Mean	Median
			Blueback herring			
Overall	0.549	0.500	0.439	0.400	0.274	0.250
Maine	0.489	0.500	0.391	0.400	0.244	0.250
Hudson River	0.585	0.500	0.468	0.400	0.292	0.250
Cooper River, SC	0.560	0.500	0.448	0.400	0.280	0.250
Santee River, SC	0.561	0.500	0.449	0.400	0.280	0.250
			Alewife			
Overall	0.616	0.580	0.493	0.464	0.308	0.290
Maine	0.553	0.549	0.443	0.439	0.277	0.275
Hudson River, NY	0.679	0.611	0.543	0.489	0.340	0.306

Table 5. The attributes and rankings used in the PSA analysis for the northwest Atlantic coastal shelf ecosystem.

Productivity Attributes	**High (3)**	**Moderate (2)**	**Low (1)**
R	>0.5	0.5-0.16 (mid-pint 0.10)	<0.16
Maximum Age	< 10 years	10 - 30 years (mid-point 20)	> 30 years
Maximum Size	< 60 cm	60-150 cm (mid-point 105)	> 150 cm
von Bertalanffy Growth Coefficient (k)	> 0.25	0.15-0.25 (mid-point 0.20)	< 0.15
Estimated Natural Mortality	> 0.40	0.20-0.40 (mid-point 0.30)	< 0.20
Measured Fecundity	> 10e4	10e2-10e3	< 10e2
Breeding Strategy	0	between 1 and 3	≥4
Recruitment Pattern	highly frequent recruitment success (> 75% of year classes are successful)	moderately frequent recruitment success (between 10% and 75% of year classes are successful)	infrequent recruitment success (< 10% of year classes are successful)
Age at Maturity	< 2 years	2-4 years (mid-point 3.0)	> 4 years
Mean Trophic Level	<2.5	2.5-3.5 (mid-point 3)	>3.5
Susceptibility Attributes	**Low (1)**	**Moderate (2)**	**High (3)**
Management Strategy	Targeted stocks have catch limits and proactive accountability measures; Non-target stocks are closely monitored.	Targeted stocks have catch limits and reactive accountability measures	Targeted stocks do not have catch limits or accountability measures; Non-target stocks are not closely monitored.
Areal Overlap	< 25% of stock occurs in the area fished	Between 25% and 50% of the stock occurs in the area fished	> 50% of stock occurs in the area fished
Geographic Concentration	stock is distributed in > 50% of its total range	stock is distributed in 25% to 50% of its total range	stock is distributed in < 25% of its total range

Vertical Overlap	< 25% of stock occurs in the depths fished	Between 25% and 50% of the stock occurs in the depths fished	> 50% of stock occurs in the depths fished
Fishing rate relative to M	<0.5	0.5 - 1.0	>1
Biomass of Spawners (SSB) or other proxies	B is > 40% of B0 (or maximum observed from time series of biomass estimates)	B is between 25% and 40% of B0 (or maximum observed from time series of biomass estimates)	B is < 25% of B0 (or maximum observed from time series of biomass estimates)
Seasonal Migrations	Seasonal migrations decrease overlap with the fishery	Seasonal migrations do not substantially affect the overlap with the fishery	Seasonal migrations increase overlap with the fishery
Schooling/Aggregation and Other Behavioral Responses	Behavioral responses decrease the catchability of the gear	Behavioral responses do not substantially affect the catchability of the gear	Behavioral responses increase the catchability of the gear [i.e., hyperstability of CPUE with schooling behavior]
Morphology Affecting Capture	Species shows low selectivity to the fishing gear.	Species shows moderate selectivity to the fishing gear.	Species shows high selectivity to the fishing gear.
Survival After Capture and Release	Probability of survival > 67%	33% < probability of survival < 67%	Probability of survival < 33%
Desirability/Value of the Fishery	stock is not highly valued or desired by the fishery	stock is moderately valued or desired by the fishery	stock is highly valued or desired by the fishery
Fishery Impact to EFH or Habitat in General for Non-targets	Adverse effects absent, minimal or temporary	Adverse effects more than minimal or temporary but are mitigated	Adverse effects more than minimal or temporary and are not mitigated

Table 6. Species and stocks used in the PSA together with estimated productivity, susceptibility and vulnerability scores.

Number	Common name and stock	Species Name	Family	Productivity	Susceptibilty	Vulnerability	Vulnerability ranking
1	American shad	*Alosa sapidissima*	Clupeidae	2.300	2.333	1.55	22
2	Alewife	*Alosa pseudoharengus*	Clupeidae	2.300	2.333	1.51	23
3	Blueback herring	*Alosa aestivalis*	Clupeidae	2.209	2.333	1.48	26
4	black seabass	*Centropristis striata*	Pleuronectidae	2.209	1.667	1.03	35
5	Striped bass	*Morone saxatilis*	Moronidae	1.197	2.583	1.8	9
6	GOM haddock	*Melanogrammus aegelfinus*	Gadidae	2.000	2.583	1.78	13
7	GB haddock	*Melanogrammus aegelfinus*	Gadidae	1.830	2.583	1.68	17
8	GOM cod	*Gadus morhua*	Gadidae	2.000	1.667	1.82	4
9	GB cod	*Gadus morhua*	Gadidae	1.830	1.667	1.98	1
10	Pollock	*Pollachius virens*	Pleuronectidae	2.600	2.583	1.51	24
11	white hake	*Urophycis tenuis*	Malacanthidae	1.197	1.833	1.5	25
12	CCGOM yellowtail	*Limanda ferruginea*	Clupediae	2.209	2.583	1.82	5
13	George Bank Yellow Tail	*Limanda ferruginea*	Clupediae	2.000	2.583	1.75	15
14	SNE Yellow tail	*Limanda ferruginea*	Clupediae	2.450	2.583	1.82	6
15	Plaice	*Hippoglossoides platessoides*	Clupediae	2.600	2.583	1.48	27
16	Witch flounder	*Gyptochephalus cynoglossus*	Pleuronectidae	2.222	2.500	1.7	16
17	GB winter flounder	*Pseudopleuronectes americanus*	Gadidae	2.000	2.583	1.8	10
18	GB winter flounder	*Pseudopleuronectes americanus*	Gadidae	2.000	2.583	1.8	11
19	SNE winter flounder	*Pseudopleuronectes americanus*	Gadidae	2.450	2.583	1.8	12
20	northern windowpane flounder	*Scopthalmus aquosus*	Pleuronectidae	2.600	2.250	1.6	20
21	southern windowpane flounder	*Scopthalmus aquosus*	Pleuronectidae	2.450	2.250	1.6	21
22	Halibut	*Hippogloissoides hippoglossus*	Clupediae	2.600	2.583	1.63	19
23	Summer flounder	*Paralichthys dentatus*	Arcticidae	1.197	2.583	1.77	14
24	Redfish	*Sebastes marinus*	Sparidae	2.500	2.250	1.31	31
25	Ocean pout	*Macrozoarcidae americanus*		2.600	2.583	1.37	29

26	spiny dogfish	*Squalus acanthias*	Squalidae	1.197	1.833	1.98	2
27	Winter skate	*Raja ocelata*	Gadidae	1.600	2.583	1.82	7
28	Butterfish	*Peprilis tricanthus*	Arcticidae	2.209	2.583	1.17	34
29	Scup	*Stenotomus chrysops*	Squalidae	2.450	1.833	0.94	36
30	Bluefish	*Pomatomus saltatrix*	Gadidae	2.209	2.583	1.22	33
31	Tilefish	*Lopholatilus chamaeleonticeps*		1.197	2.583	1.81	8
32	ocean quahog	*Arctica islandica*	Clupeidae	2.600	2.333	1.88	3
33	Surfclam	*Spisula solidissima*	Squalidae	1.197	1.833	1.27	32
34	Weakfish	*Cyonscion regalis*	Pleuronectidae	1.197	1.667	1.41	28
35	northern monkfish	*Lophius americanus*		2.600	2.583	1.67	18
36	Herring	*Clupea harengus*	Pleuronectidae	2.600	1.667	1.33	30

Table 7. Definition of subscripts, input data and input parameters for stochastic stock reduction analysis.

Equation N^{o}	Details	Definition
Indices		
	A	Age
	A	Maximum age
	T	Index for time
Life history parameters		
T7.1	Growth parameters	$L_{\infty} = 58.7$ $K = 0.4$ $t0 = -0.1$
T7.2	Length-weight relationship	$\alpha = 5.35 \times 10^{-6}$ $\beta = 3.2207$
T7.3	Age at 50% maturity	$A_h = 4.779$
	Standard deviation of age at 50% maturity	$\gamma_h = 0.461$
T7.4	Age at 50% recruitment	$A_v = 5.30$
	Standard deviation of age at 50% recruitment	$\gamma_v = 0.896$
T7.5	Natural mortality rate	$M = 0.15 – 0.4$
Time series data (1980-2005)		
T7.6	Catch, Survey index	C_t, I_t

Table 8. Notation for estimated parameters, age-schedule calculations and initial state calculations

Equation No	Details
T8.1	$\Theta = (U_{MSY}, F, \kappa\ B_0, \sigma)$
Age schedules	
T8.2	$l_a = (e^{-M})^a$
T8.3	$L_a = L_\infty \cdot \left(1 - e^{-K\cdot(a-t_0)}\right)$
T8.4	$W_a = \alpha \cdot L_a{}^{\beta}$
T8.5	$m_a = \dfrac{1}{\left(1 + e^{\frac{-(a-A_h)}{\gamma_h}}\right)}$
T8.6	$v_a = \dfrac{1}{\left(1 + e^{\frac{-(a-A_v)}{\gamma_v}}\right)}$
Incidence functions and initial conditions	
T8.7	$\varphi_E = \sum_a l_a \cdot m_a$ $\qquad \varphi_e = \sum_a \cdot \widehat{l_a} m_a$
T8.8	$\varphi_B = \sum_a l_a \cdot W_a \cdot v_a$ $\qquad \varphi_b = \sum_a \widehat{l_a} \cdot W_a \cdot v_a$

Table 9. Dynamic model and observation residuals for stochastic stock reduction analysis.

Equation No	Details
Unobserved states	
T9.1	n_{at}, N_t, B_t, U_t
Derived variables	
T9.2	$k = \dfrac{\left(\varphi_E \cdot \left(\varphi_B \cdot \varphi_F + U_{MSY} \dfrac{\delta\varphi_b}{\delta U} \cdot \varphi_F - U_{MSY} \dfrac{\delta\varphi_f}{\delta U} \cdot \varphi_B\right)\right)}{\left(\varphi_f \cdot \left(\varphi_B + U_{MSY} \cdot \dfrac{\delta\varphi_b}{\delta U}\right)\right)^2}$
T9.3	$R_0 = \dfrac{MSY \cdot (k-1)}{\varphi_B \cdot U_{MSY} \cdot \left(k - \dfrac{\varphi_E}{\varphi_F}\right)}$
T9.4	$B_0 = R_0 \cdot \varphi_B$
T9.5	$s_0 = \dfrac{k}{\theta_e}$
T9.6	$b = \dfrac{k-1}{R_0 \cdot \theta_e}$
State dynamics (1980 – 2005)	
T9.7	$U_t = \dfrac{C_t}{\sum_a n_{a,t} \cdot v_a \cdot W_a}$
T9.8	$E_{0,t} = n_{a,t} \cdot m_a \cdot W_a$
T9.9	$n_{0,t+1} = \dfrac{s_0 \cdot E_{0,t}}{\left(1 + b \cdot E_{0,t}\right)}$
T.9.10	$n_{a+1,t+1} = n_{a,t} \cdot e^{-M} \cdot (1 - v_a \cdot U_t)$
T9.11	$N_t = \sum_a n_{a,t}$
T9.12	$B_t = \sum_a W_a \cdot v_a \cdot n_{a,t}$
Observed states	
T9.13	$Z_t = \log I_t - \log B_t$
T9.14	$\sigma_t = Z_t - \bar{Z}_t$
T9.15	$\Delta_n = k - \hat{k}$

Table 10. Likelihood functions for the stochastic stock reduction analysis.

Equation	Detail
Likelihoods	
T10.1	$l_1 = 0.5 \cdot (n-1) \cdot \ln\left(\sum_t {\sigma_t}^2\right)$
T10.2	$l_2 = 0.5 \cdot (n-1) \cdot ln\left(\sum \Delta\right)$
Penalties and Priors	
T10.3	If U_t >0.75 → $U_{pen} = 100 \cdot (U_t - 0.75)^2$
T10.4	$P(k) \propto normal(\mu = 35, \sigma = 0.1)$

Table 11. Parameter estimates for stochastic stock reduction analysis with M=0.25.

Moment	B_0 x 10^3 MT	kappa	Umsy	MSY x 10^3 MT
Median	9.563	28.310	0.506	1.215
Mean	9.580	30.379	0.509	1.217

Figure 1. Empirical estimates of a) FMSY, b) 0.8 FMSY and c) 0.5 FMSY for American shad coastwide. The distribution reflects the range of values calculated by up to 9 different empirical relationships between M and FMSY and for 6 different systems (See Table 1) for details.

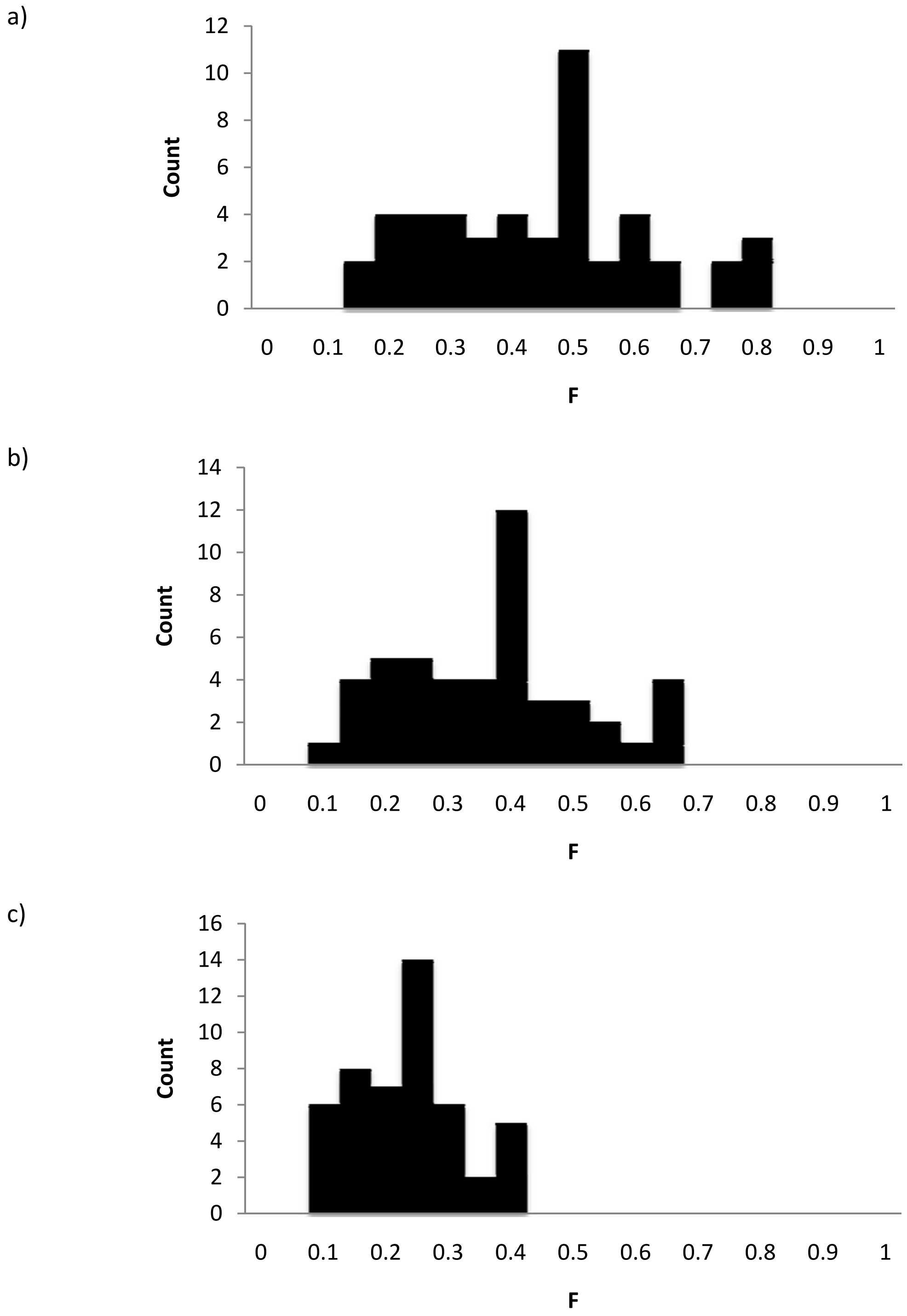


Figure 2. Empirical estimates of a) FMSY, b) 0.8 FMSY and c) 0.5 FMSY for American shad in Maine. The distribution reflects the range of values calculated by up to 9 different empirical relationships between M and FMSY and for 6 different systems (See Table 1) for details.

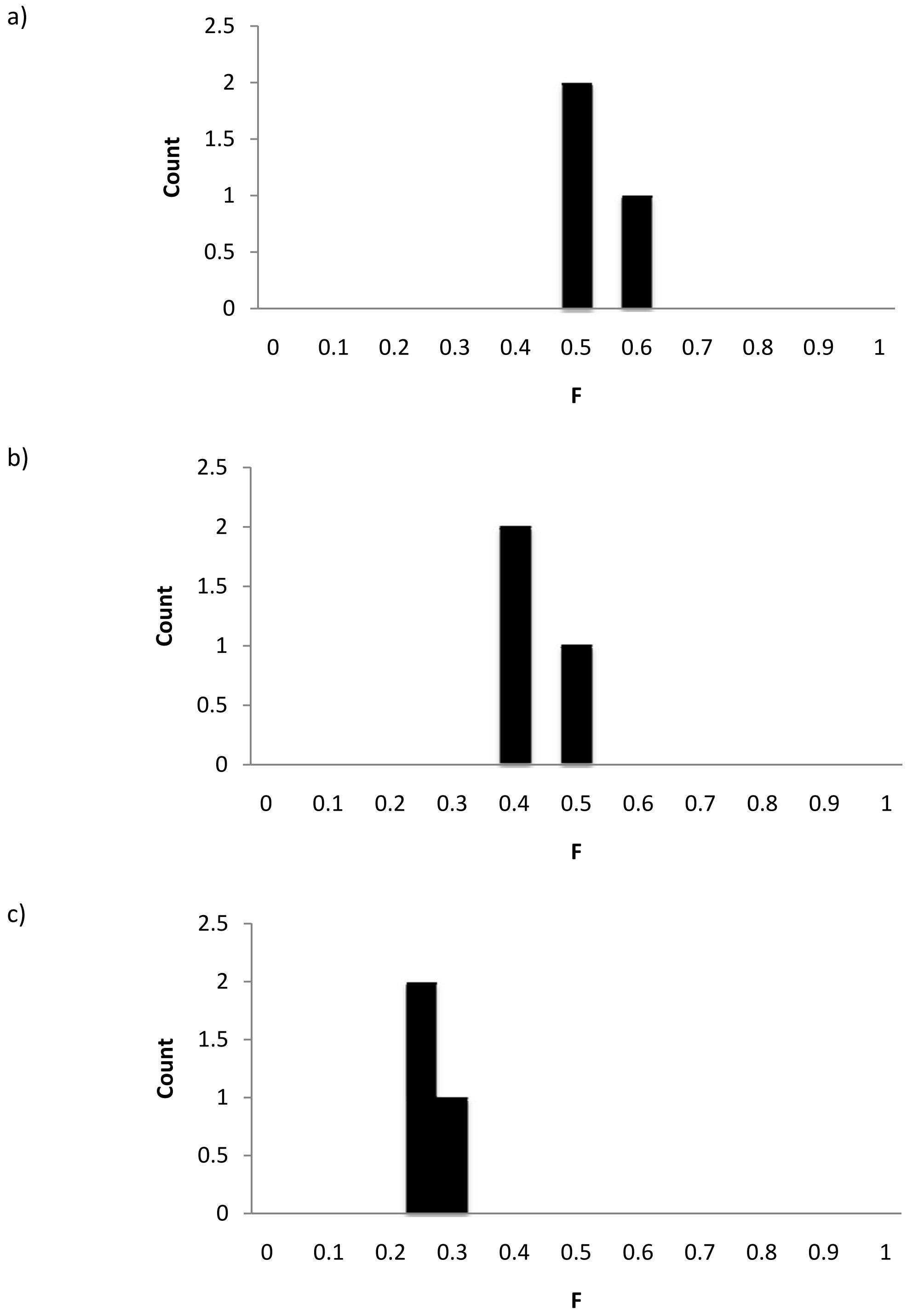


Figure 3. Empirical estimates of a) FMSY, b) 0.8 FMSY and c) 0.5 FMSY for American shad in New Hampshire. The distribution reflects the range of values calculated by up to 9 different empirical relationships between M and FMSY and for 6 different systems (See Table 1) for details.

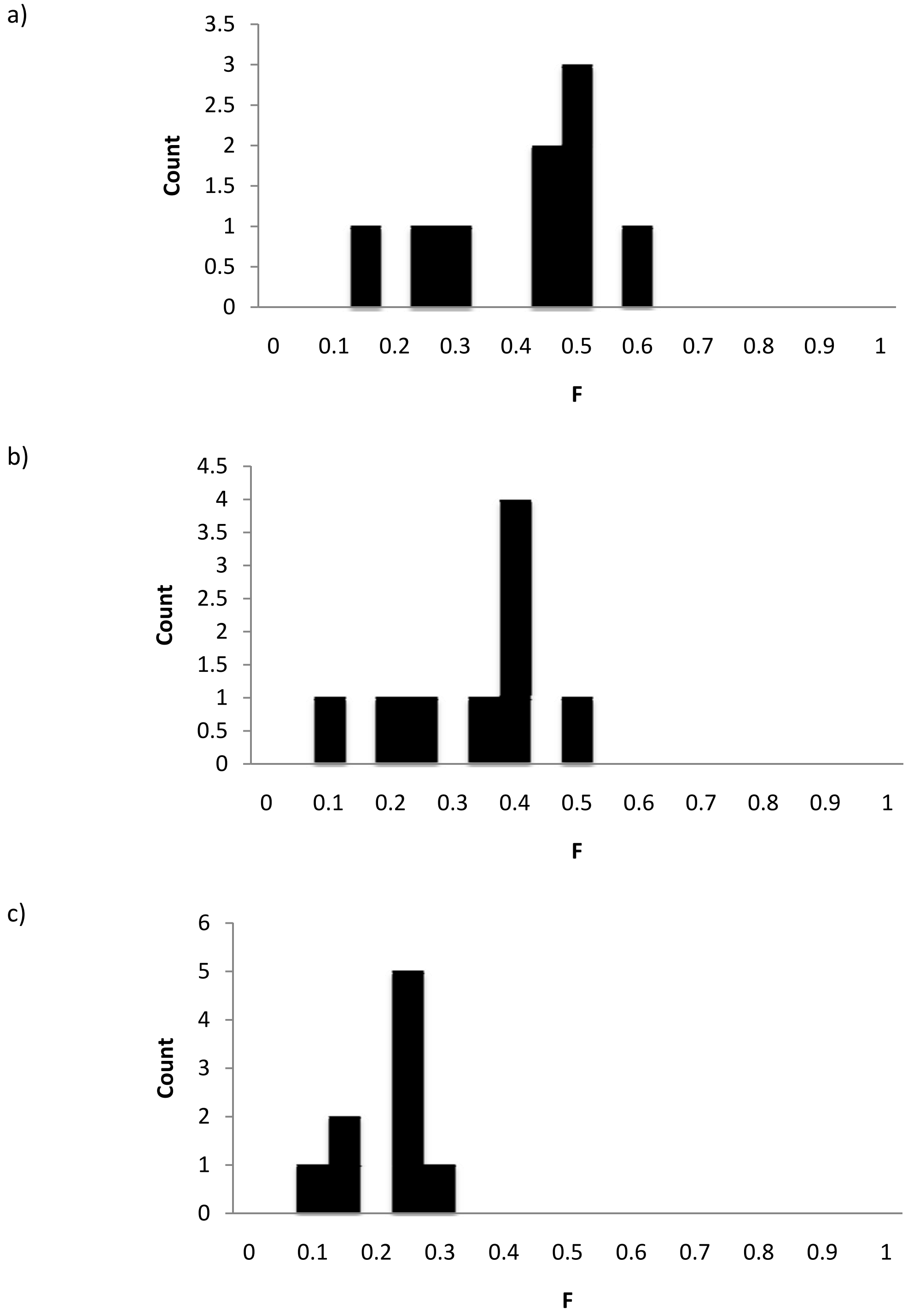


Figure 4. Empirical estimates of a) FMSY, b) 0.8 FMSY and c) 0.5 FMSY for American shad in Rhode Island. The distribution reflects the range of values calculated by up to 9 different empirical relationships between M and FMSY and for 6 different systems (See Table 1) for details.

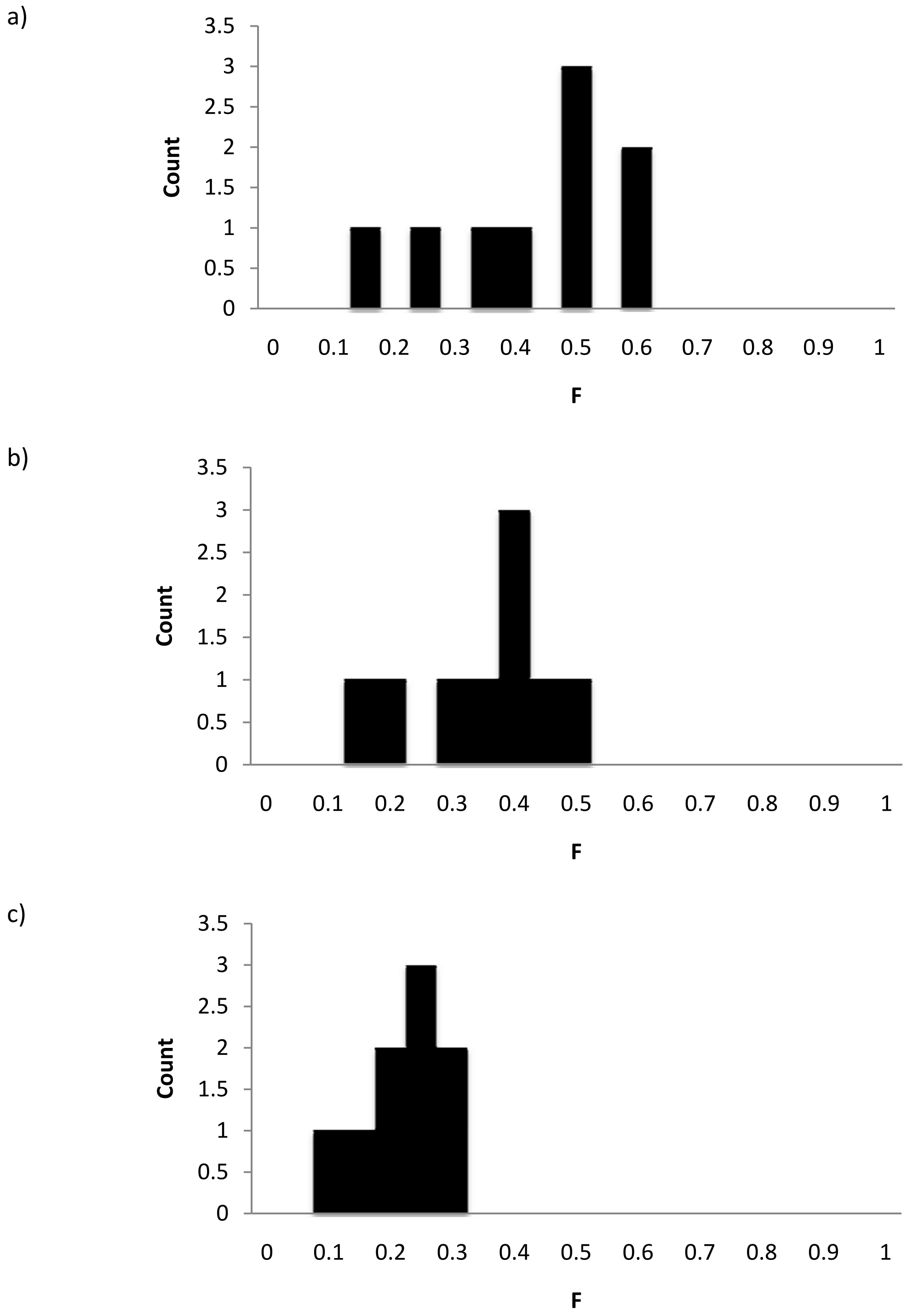


Figure 5. Empirical estimates of a) FMSY, b) 0.8 FMSY and c) 0.5 FMSY for American shad in the Hudson River, New York. The distribution reflects the range of values calculated by up to 9 different empirical relationships between M and FMSY and for 6 different systems (See Table 1) for details.

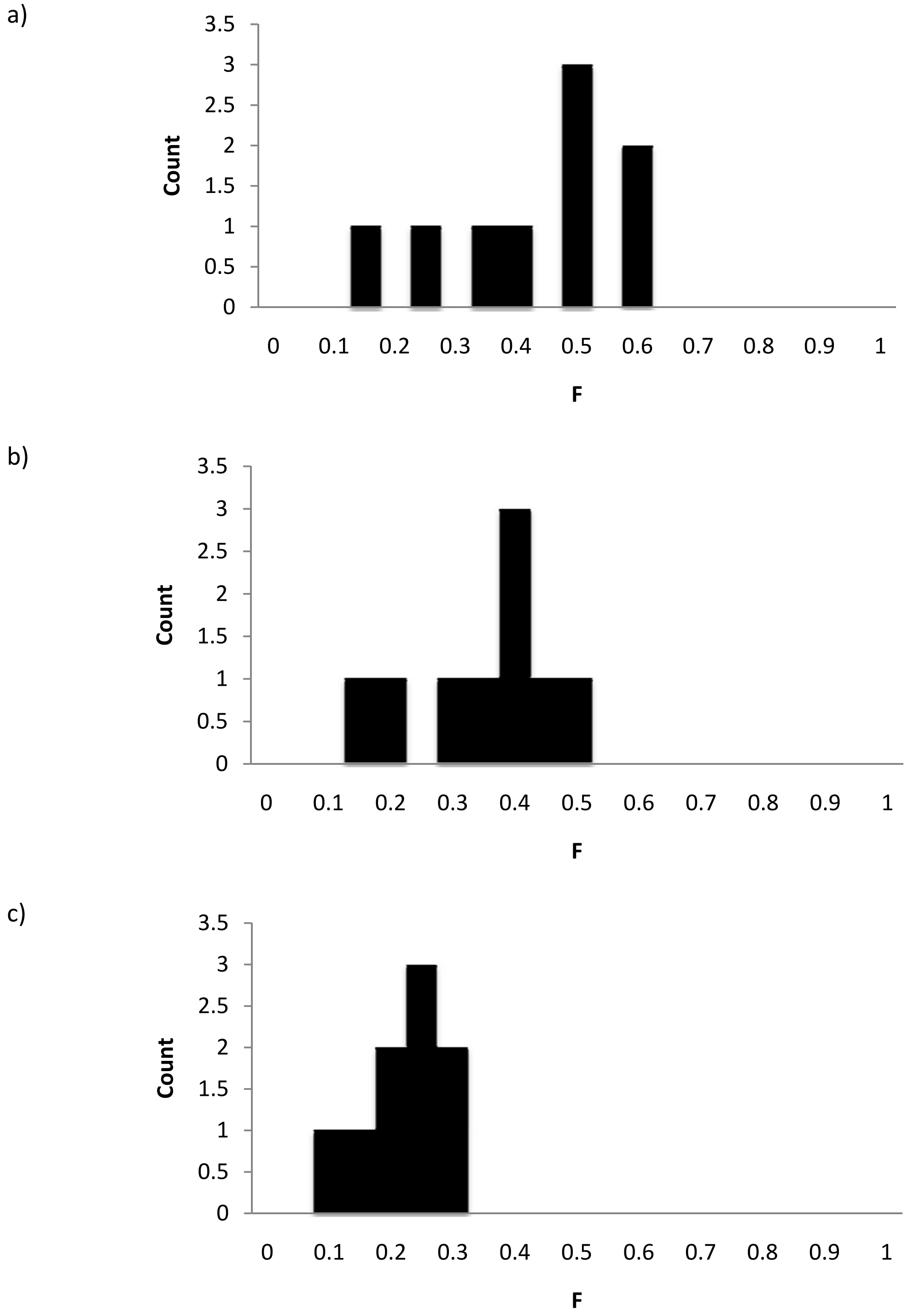


Figure 6. Empirical estimates of a) FMSY, b) 0.8 FMSY and c) 0.5 FMSY for American shad in the York River, Virginia. The distribution reflects the range of values calculated by up to 9 different empirical relationships between M and FMSY and for 6 different systems (See Table 1) for details.

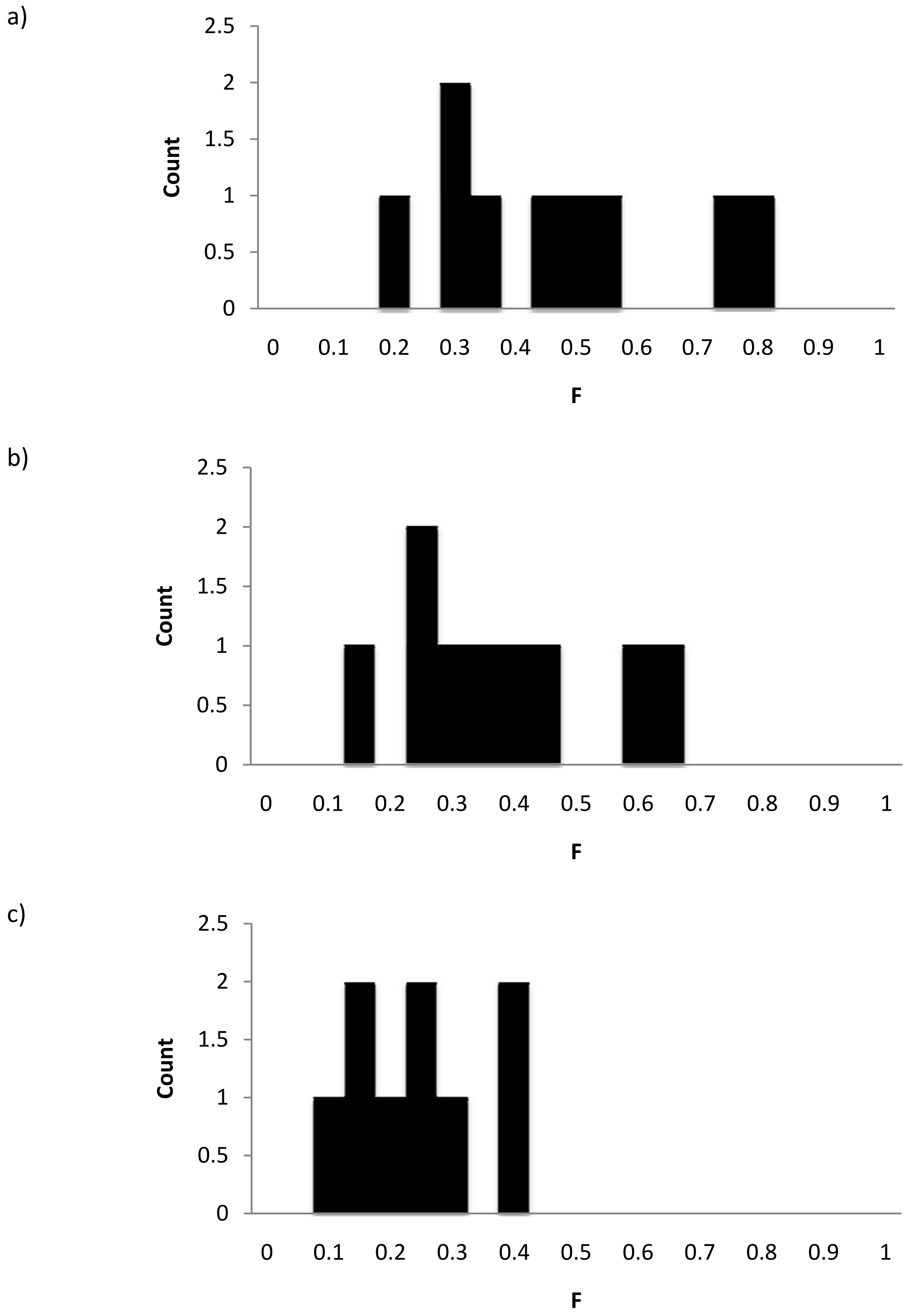


Figure 7. Empirical estimates of a) FMSY, b) 0.8 FMSY and c) 0.5 FMSY for American shad in the Albermarle Sound, N.C.. The distribution reflects the range of values calculated by up to 9 different empirical relationships between M and FMSY and for 6 different systems (See Table 1) for details.

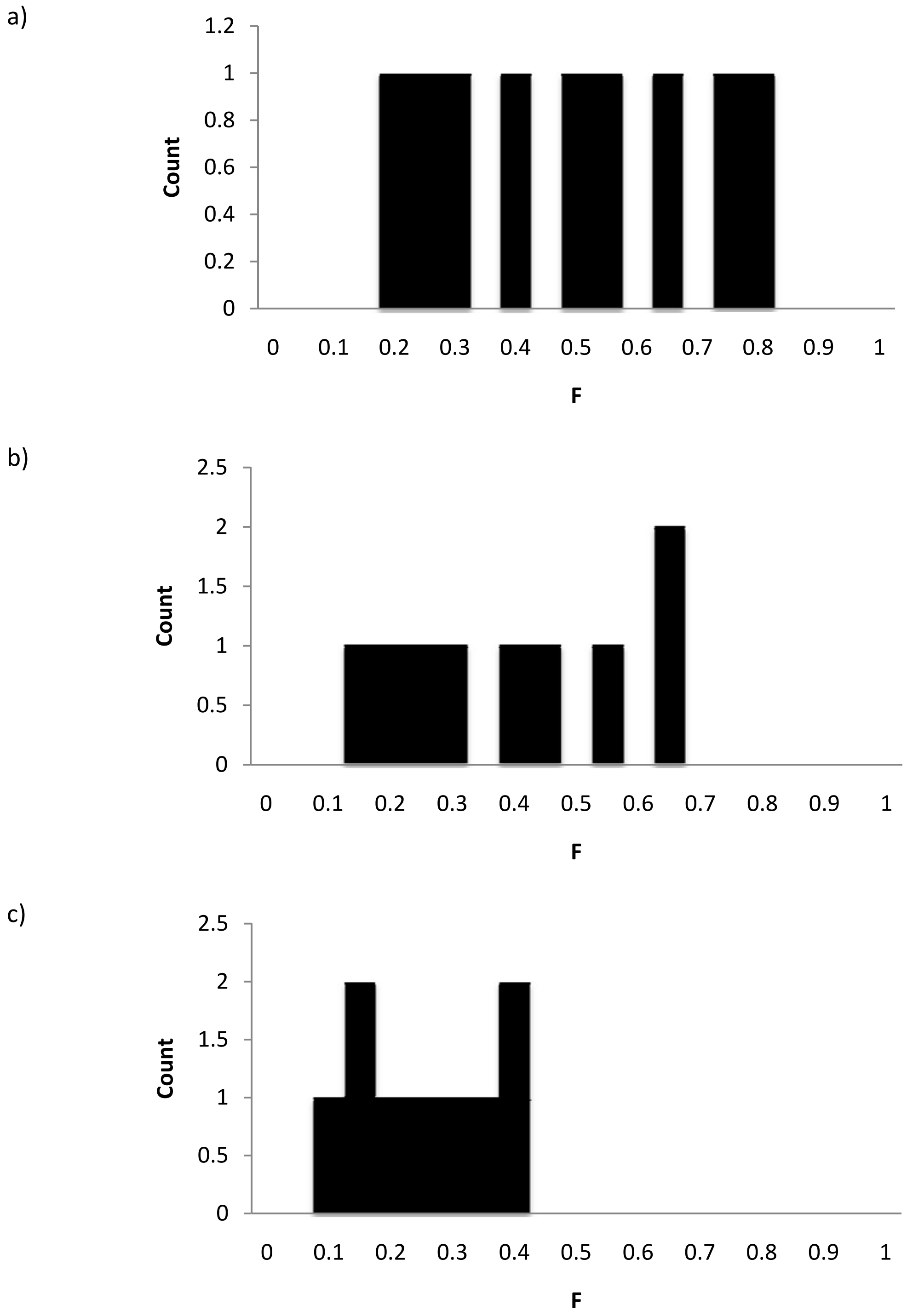


Figure 8. Empirical estimates of a) FMSY, b) 0.8 FMSY and c) 0.5 FMSY for blueback herring coastwide. The distribution reflects the range of values calculated by up to 9 different empirical relationships between M and FMSY and for 4 different systems (See Table 3) for details.

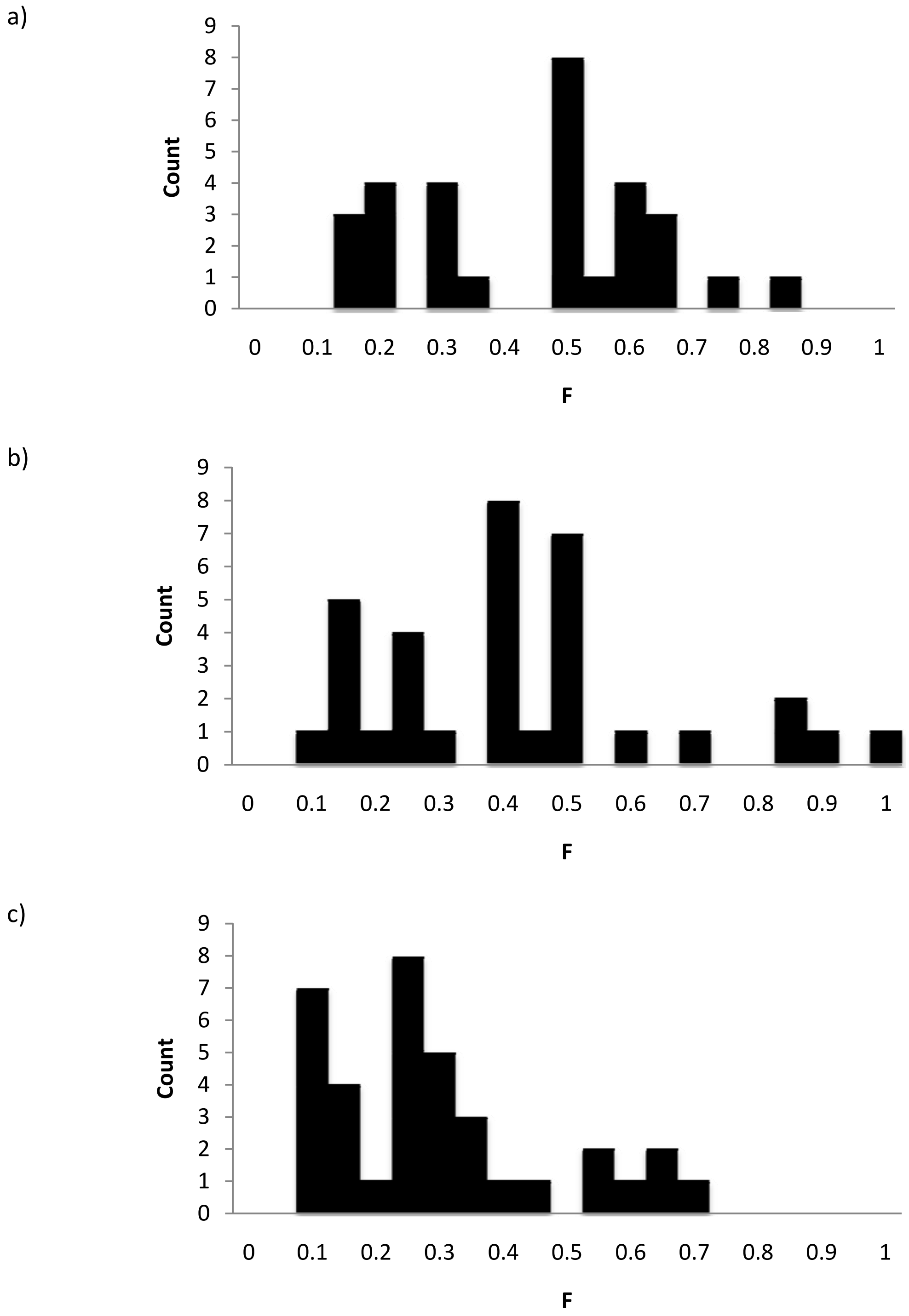


Figure 9. Empirical estimates of a) FMSY, b) 0.8 FMSY and c) 0.5 FMSY for blueback herring in coastal Maine. The distribution reflects the range of values calculated by up to 9 different empirical relationships between M and FMSY and for 4 different systems (See Table 3) for details.

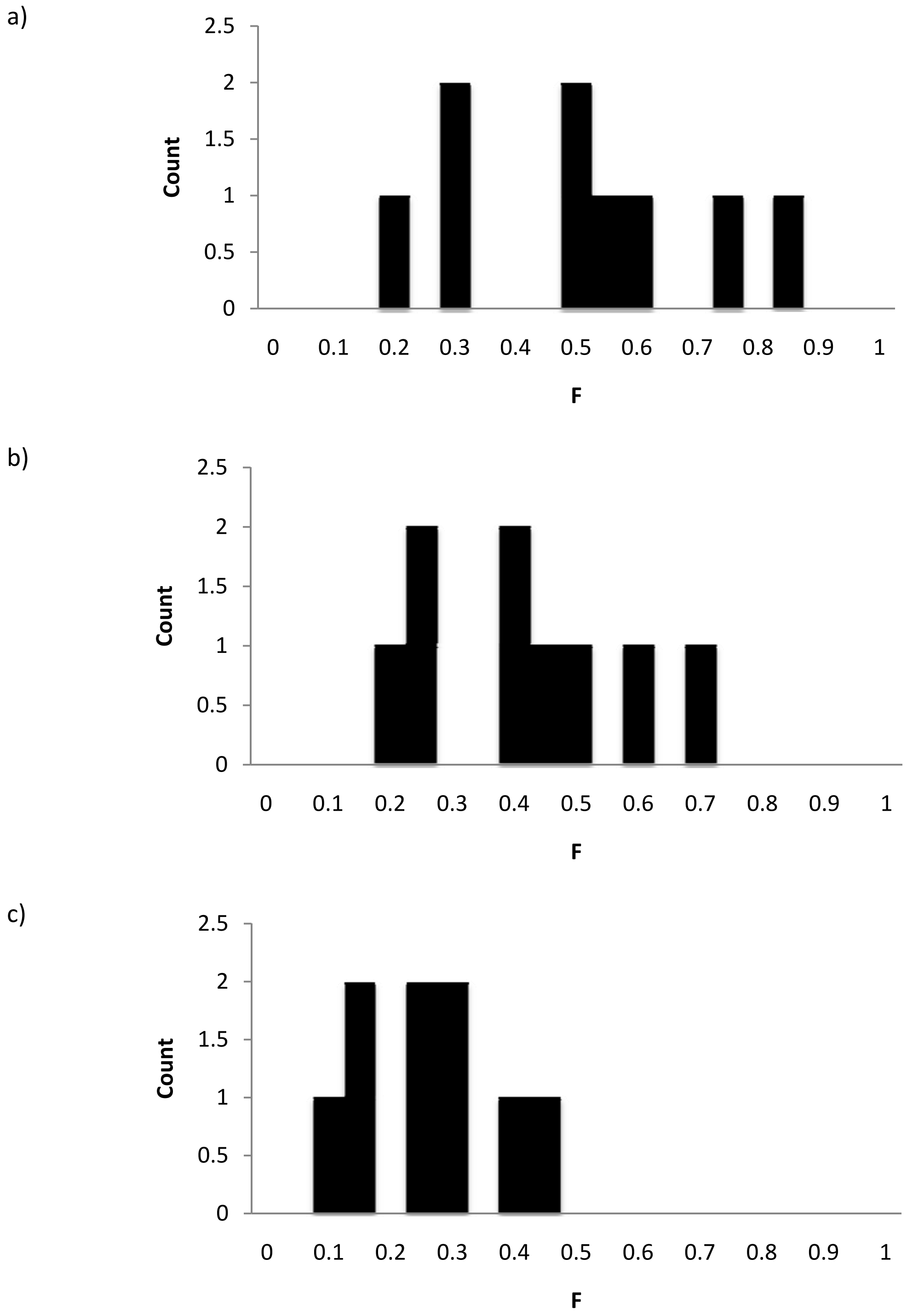


Figure 10. Empirical estimates of a) FMSY, b) 0.8 FMSY and c) 0.5 FMSY for blueback herring in the Hudson River, NY. The distribution reflects the range of values calculated by up to 9 different empirical relationships between M and FMSY and for 4 different systems (See Table 3) for details.

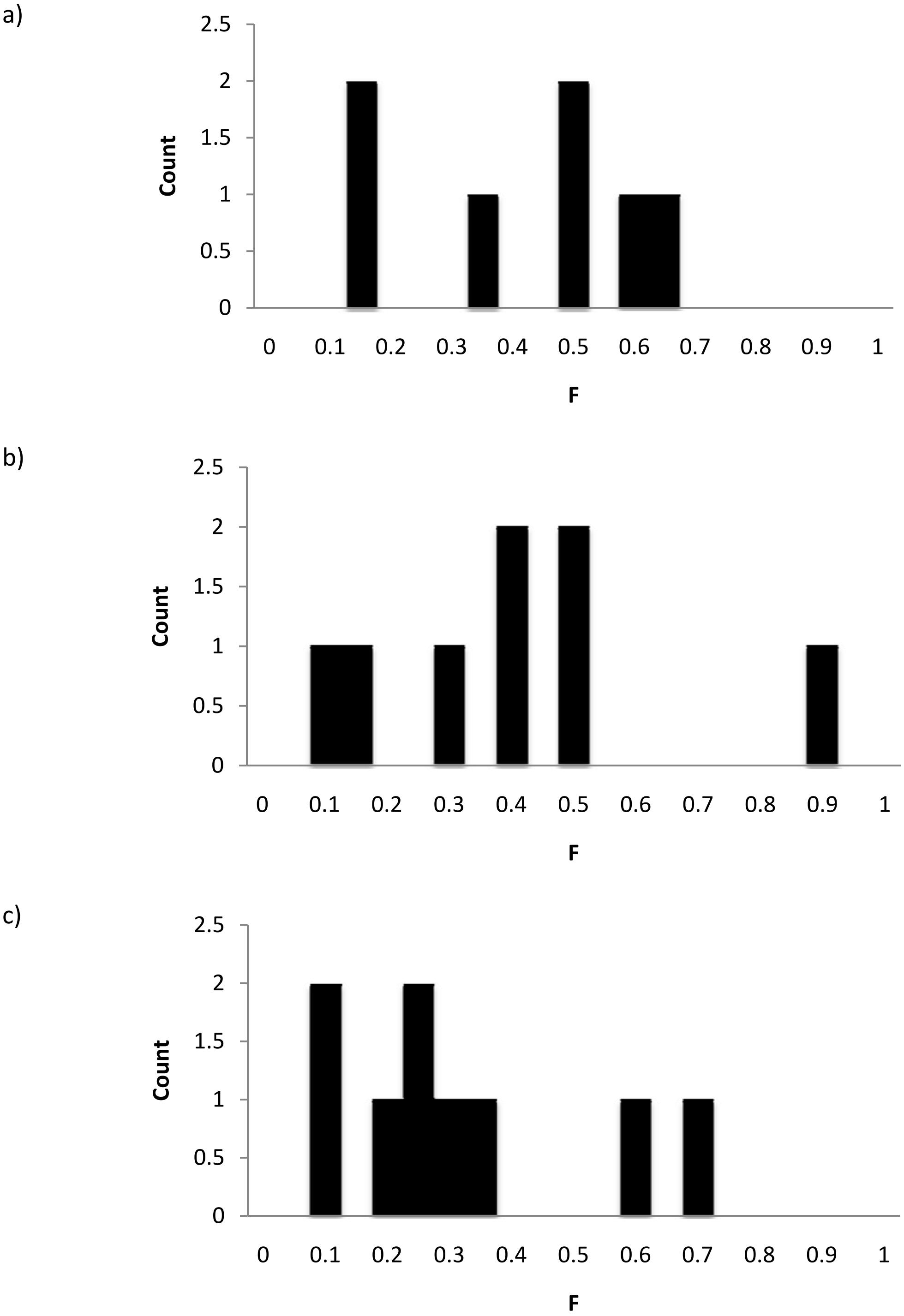
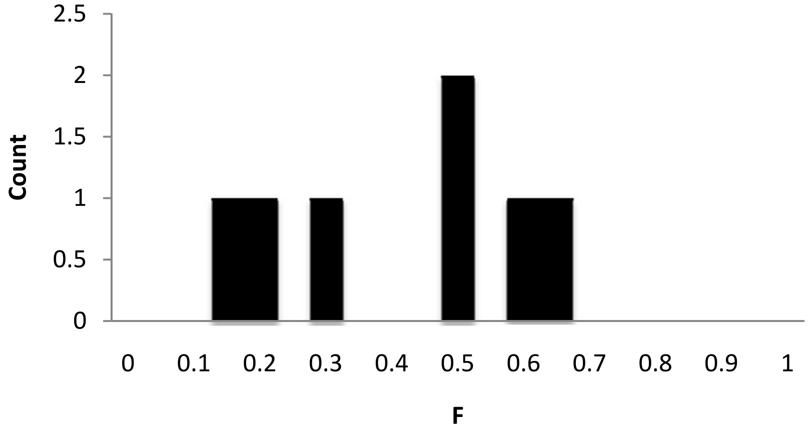
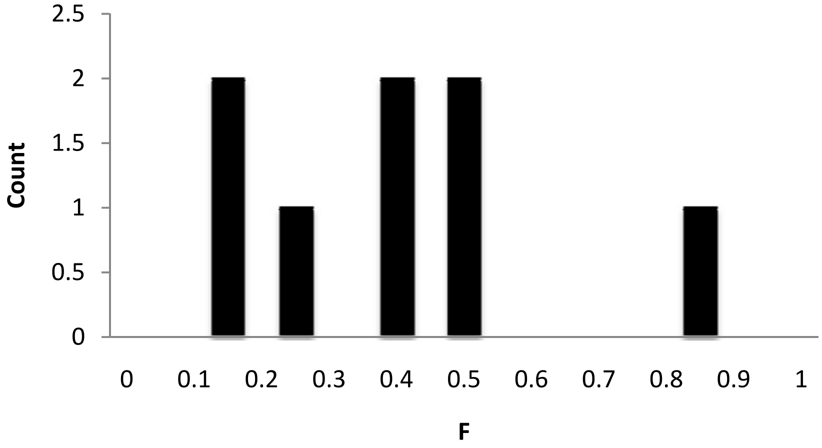


Figure 11. Empirical estimates of a) FMSY, b) 0.8 FMSY and c) 0.5 FMSY for blueback herring in the Cooper River, S.C.. The distribution reflects the range of values calculated by up to 9 different empirical relationships between M and FMSY and for 4 different systems (See Table 3) for details.

a)



b)



c)

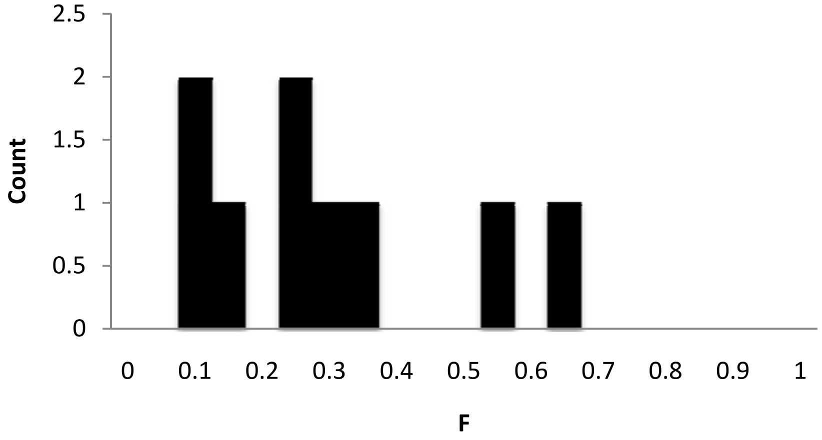


Figure 12. Empirical estimates of a) FMSY, b) 0.8 FMSY and c) 0.5 FMSY for blueback herring in the Santee River, S.C.. The distribution reflects the range of values calculated by up to 9 different empirical relationships between M and FMSY and for 4 different systems (See Table 3) for details.

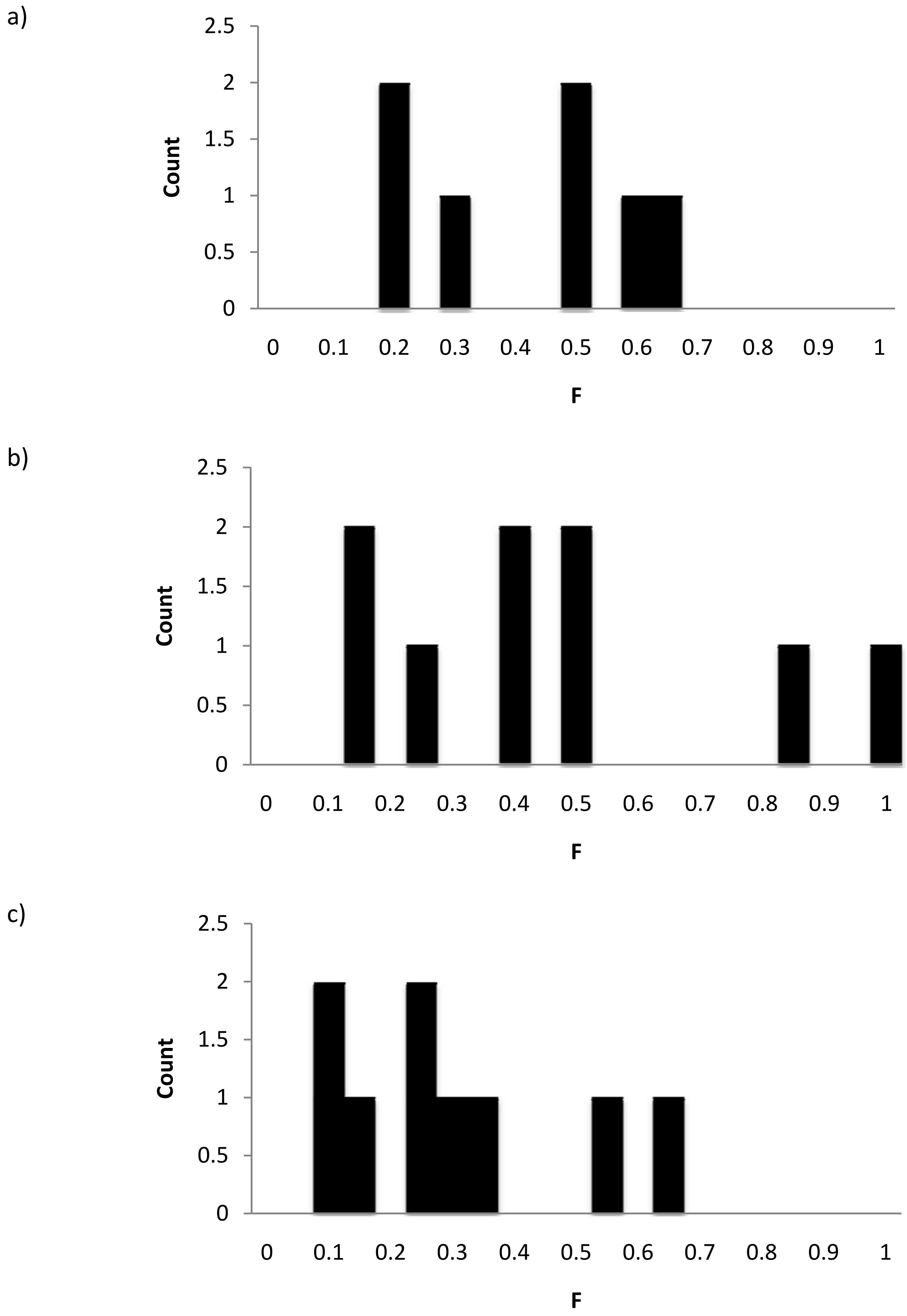


Figure 13. Results of a PSA analysis for 36 species and stocks of fishes in the northwest Atlantic coastal shelf fishery ecosystem. The plot depicts the arrangement of species and stocks against two axes: a productivity axis (X) and a susceptibility axis (Y). The axes are defined such that values near the origin of the plot are less vulnerable than values toward the extremes. Thus, one can define contours of equal species vulnerability. Three such contours are shown in blue, green and red, with the red contour indicated the highest vulnerability. Each point on the figure represents a separate species/stock. Each point is identified with a number representing species and or stocks defined in Table 4. For reference American shad, alewife and blueback herrings are species 1-3. The

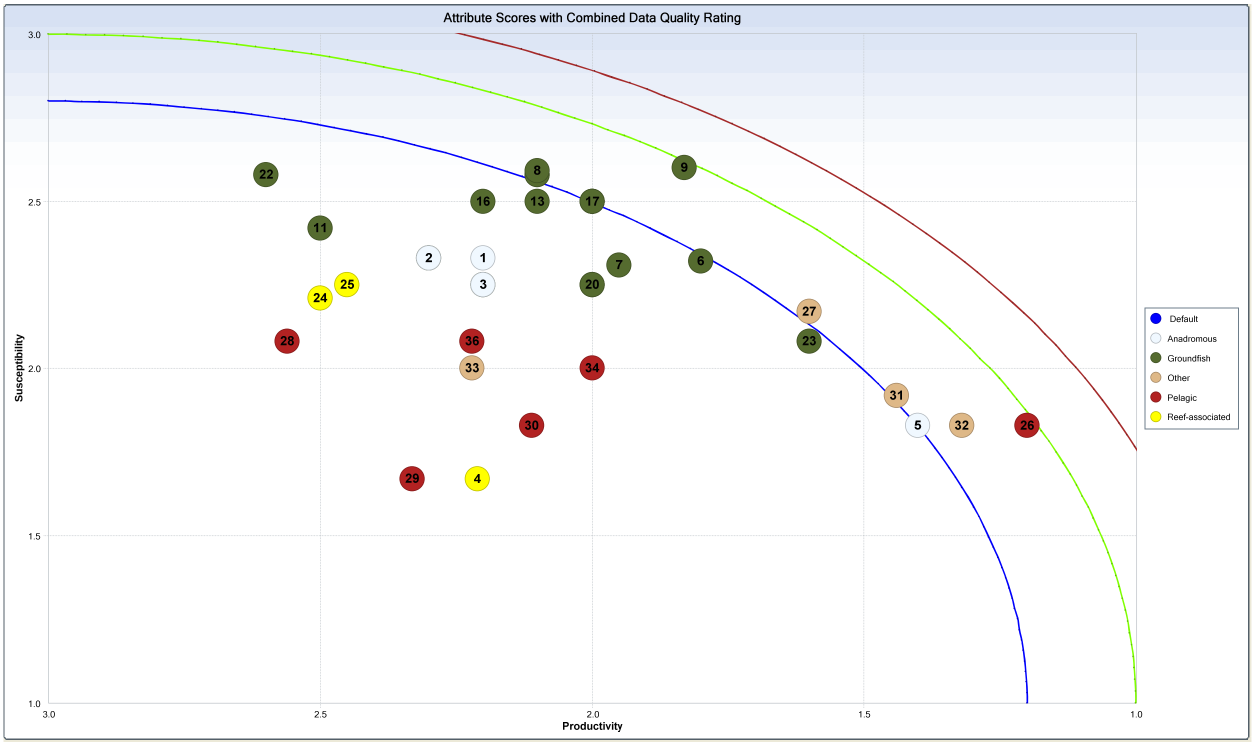


Figure 14. Contributions to total coastwide catch of American shad by individual jurisdictions.

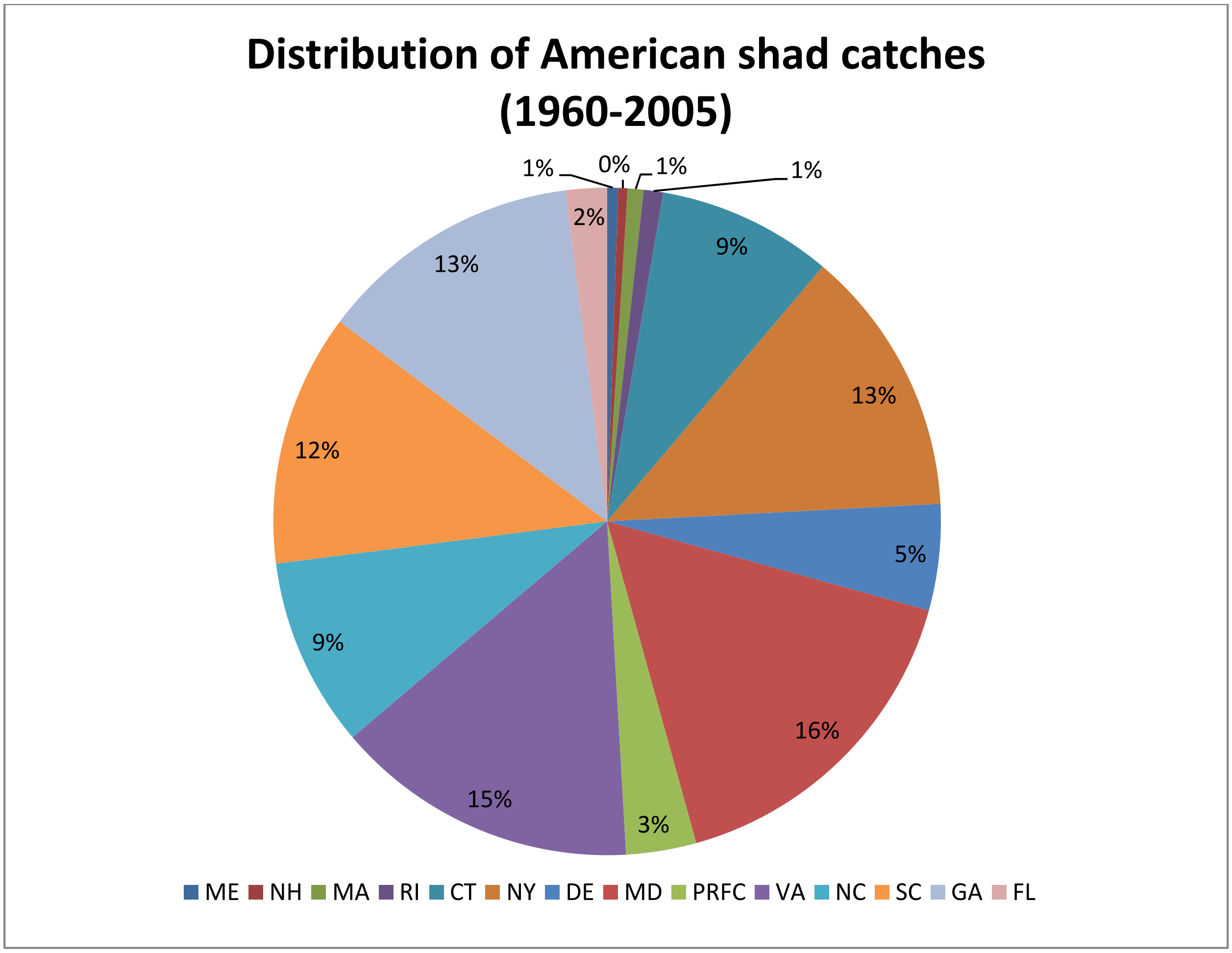


Figure 15. Time series of coastwide commercial catch (MT x 10^3) and the Lewes, DE haul seine CPUE index.

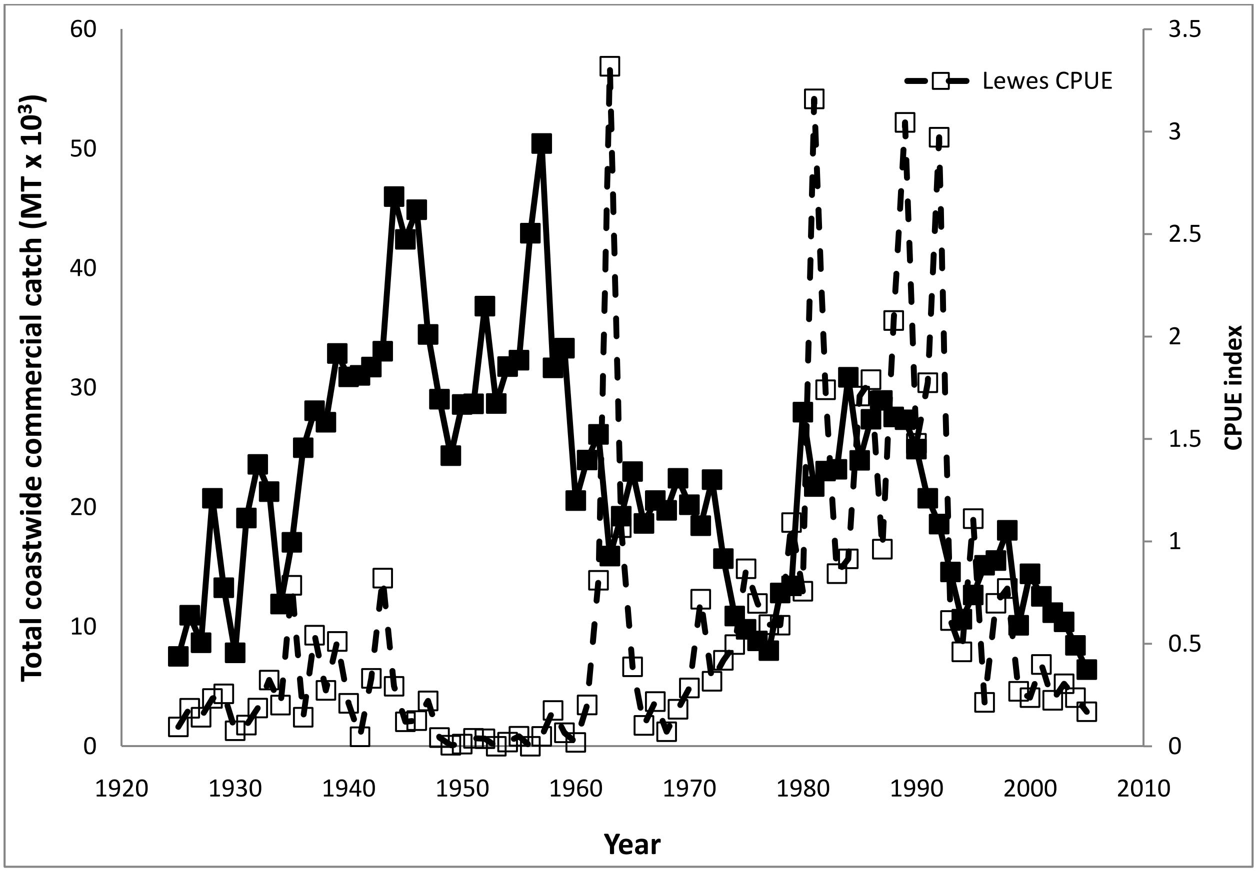


Figure 16. Distribution of leading parameters (B0, κ, UMSY and MSY) from a stochastic stock reduction analysis of American shad for a natural mortality rate, M=0.15. The diagonal panels show frequency histograms of each parameter. The axis scale forB_0 and MSY are in MT x 10^3. Scales for k and UMSY are unitless ratios. The upper triangular show distributions of pairs of parameters given by the appropriate row and column combination. Each upper triangular panel reflects 20,000 samples from the MCMC algorithm. The lower triangular panels show confidence intervals derived from the MCMC data plotted in the upper triangular panels. The inner most yellow zone represents the 95% confidence interval. The outer levels are the 80th, 10th and 1st percentile confidence intervals. The “fried egg plots” are courtesy of Dr. Steve Martell, University of British Columbia.

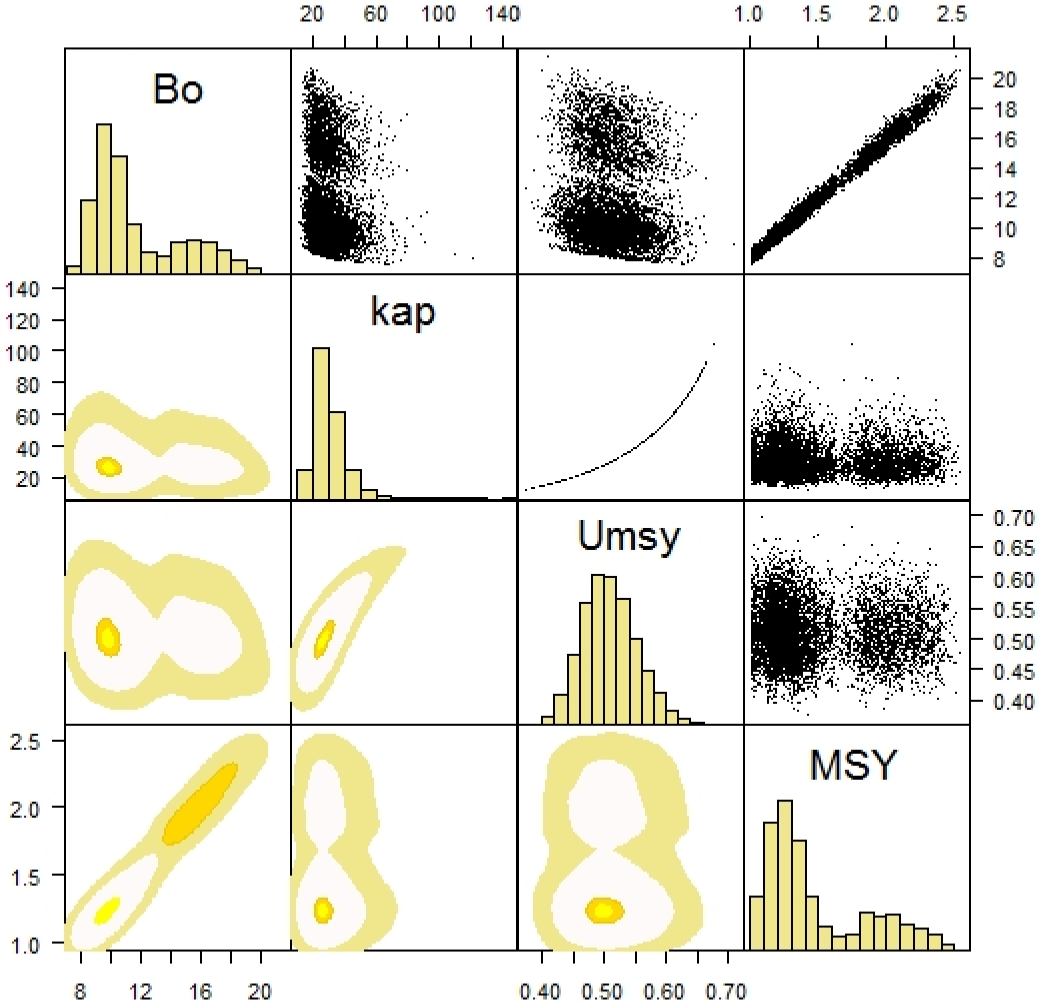


Figure 17. Distribution of leading parameters (B0, κ, UMSY and MSY) from a stochastic stock reduction analysis of American shad for a natural mortality rate, M=0.20. Plots are as described in Figure 11.

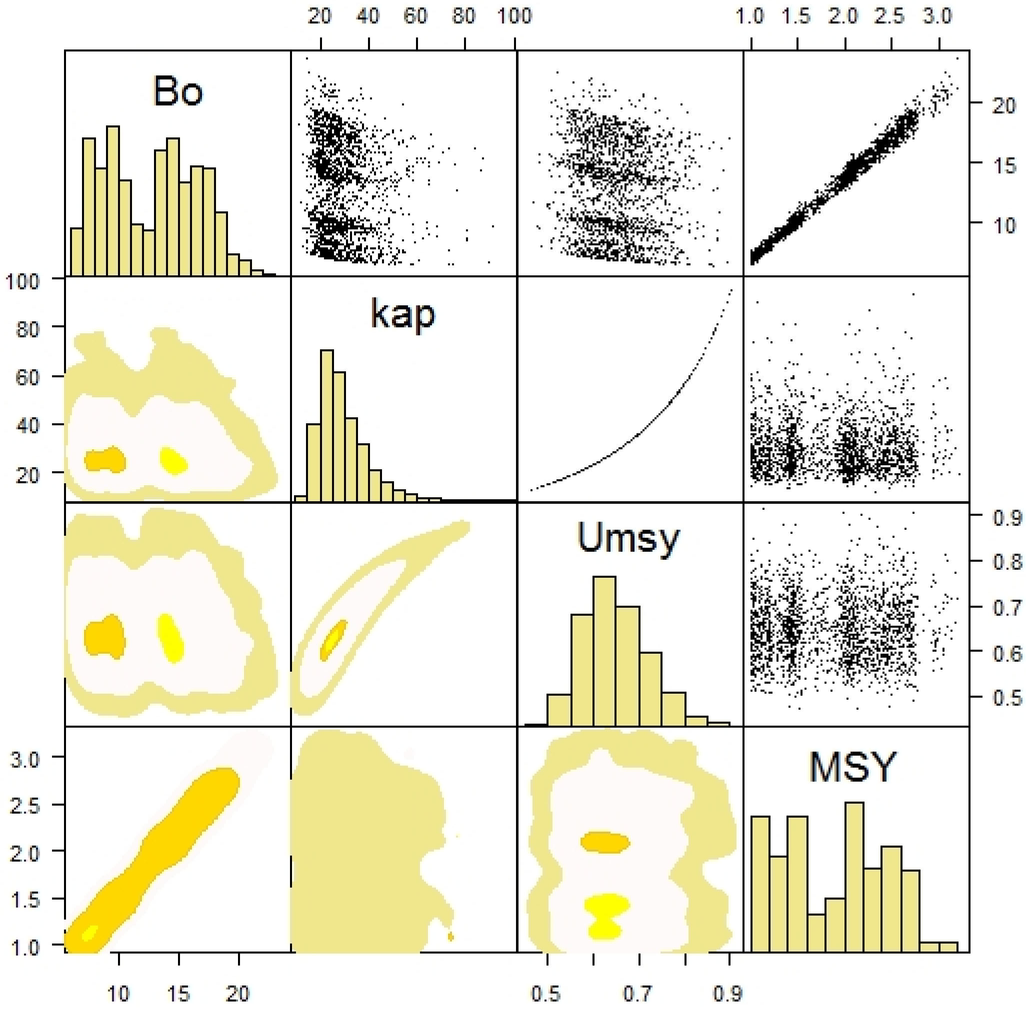


Figure 18. Distribution of leading parameters (B0, κ, UMSY and MSY) from a stochastic stock reduction analysis of American shad for a natural mortality rate, M=0.25. Plots are as described in Figure 11.

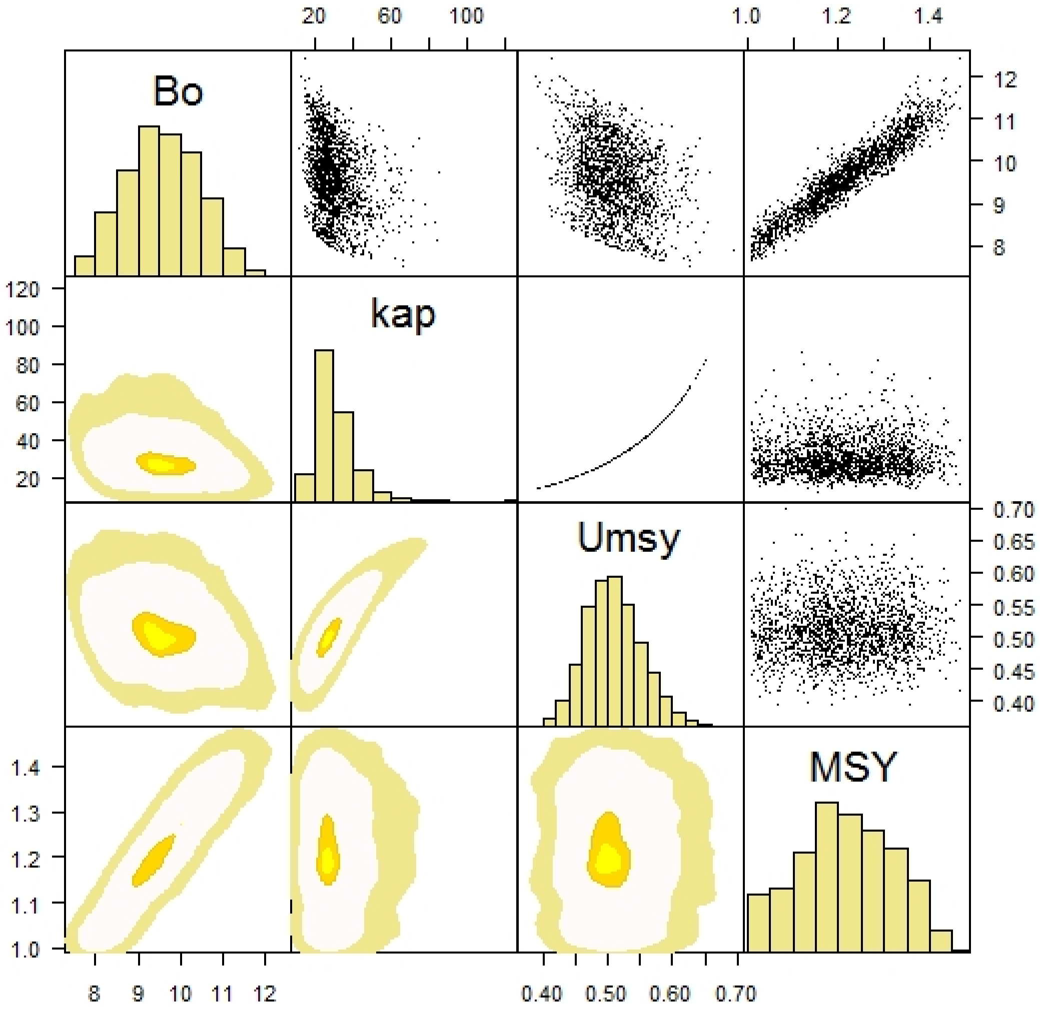


Figure 19. Distribution of leading parameters (B0, κ, UMSY and MSY) from a stochastic stock reduction analysis of American shad for a natural mortality rate, M=0.30. Plots are as described in Figure 11.

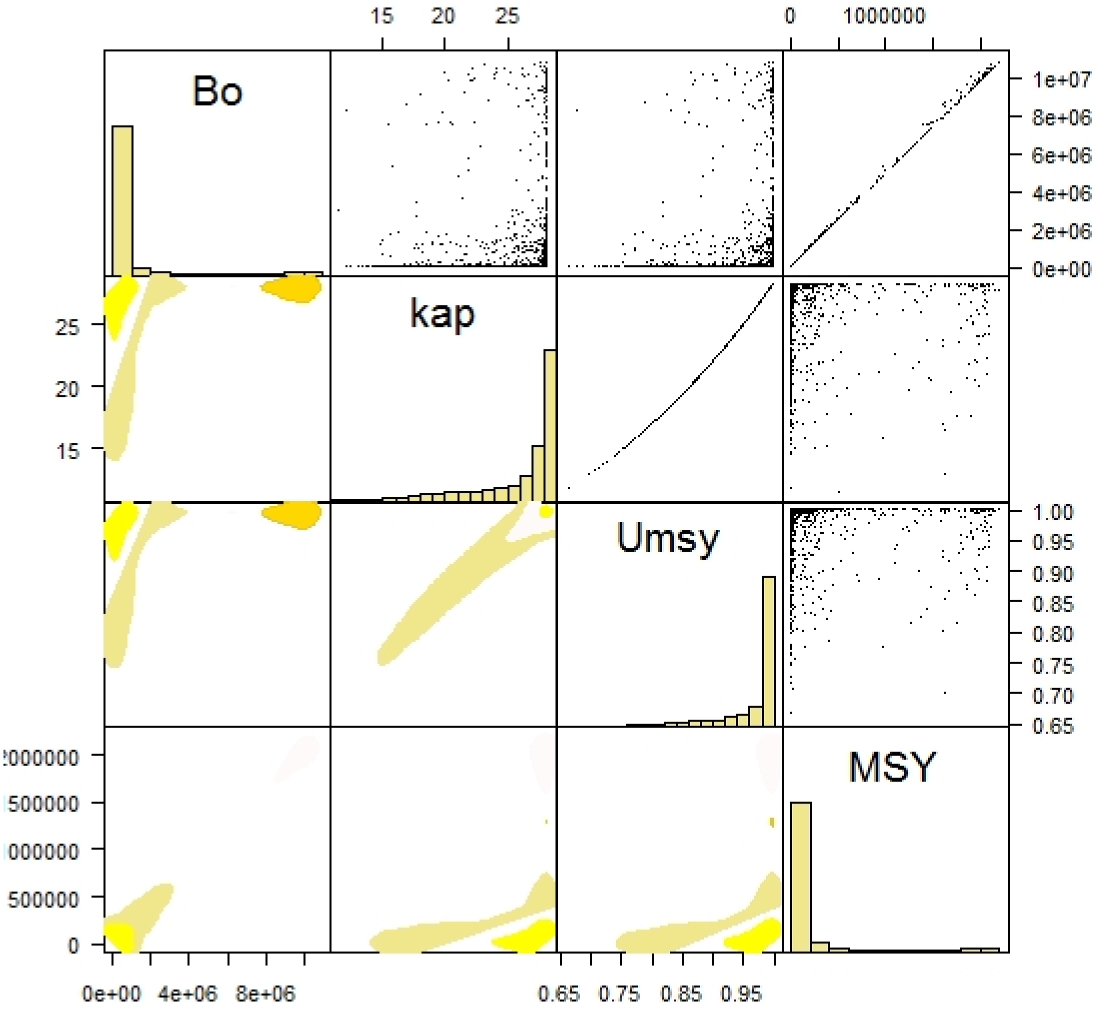
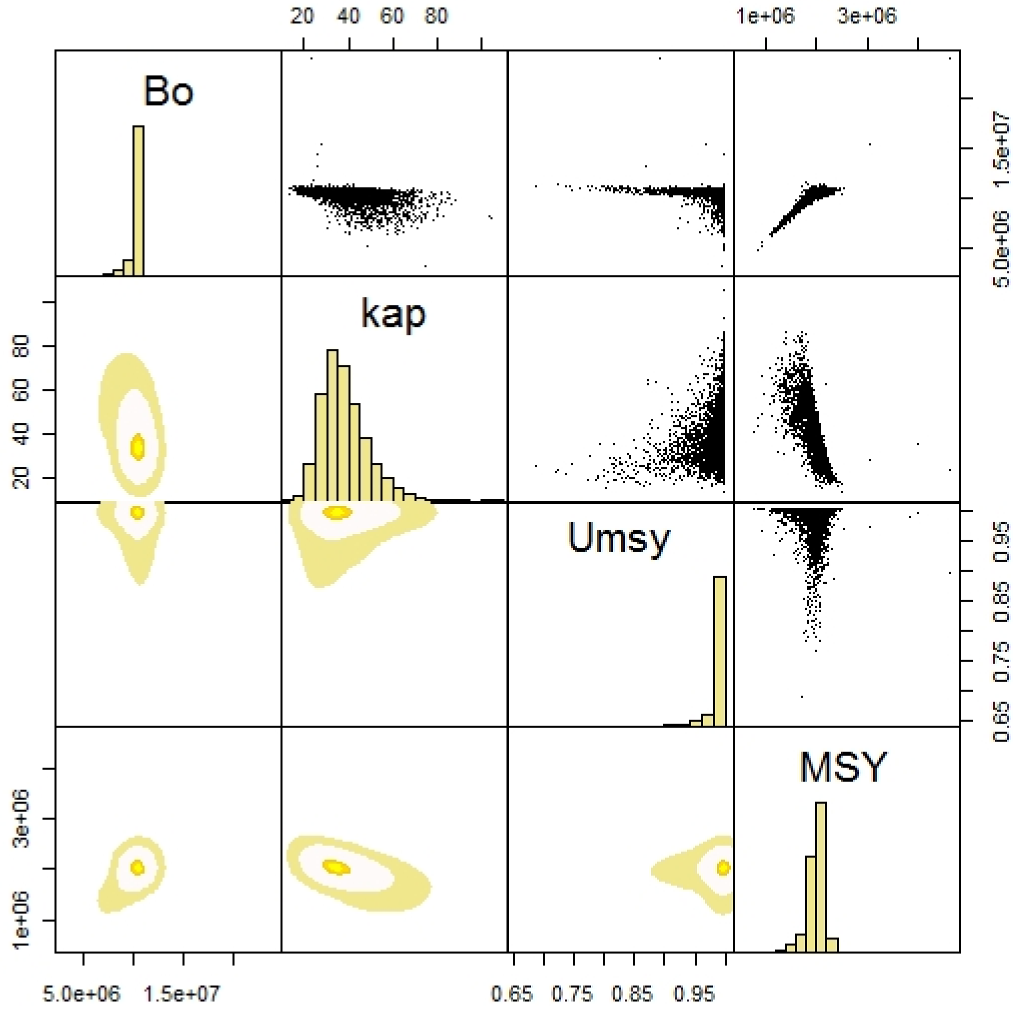


Figure 20. Distribution of leading parameters (B0, κ, UMSY and MSY) from a stochastic stock reduction analysis of American shad for a natural mortality rate, M=0.40. Plots are as described in Figure 11.



6. APPENDIX A. GLOSSARY OF TERMS

Note: Definitions for those terms shown underlined are taken from the glossary provided for assessments produced by NOAA's Northeast Fishery Science Center.

α (alpha): The intercept of an allometric relationship between length and weight.

β (beta): The exponent of an allometric relationship between length and weight.

γ_h (gamma h): The variance in age at maturity – it is a measure of the variability in the proportion of the population that is mature at a given age.

γ_v (gamma v): The variance in age at vulnerability to the fishery – it is a measure of the variability in the proportion of the population that is vulnerable to the fishery at a given age.

Δ: The difference between the prior value of the compensation ratio (κ) and the estimated value of k estimated in the model

σ: **The observation error in the model. It is the difference between the observed difference and average difference between the observed survey abundance and the predicted biomass.**

⍰$_B$, ⍰$_b$: The expected biomass distribution in the population under virgin (B) and exploited (b) conductions.

Φ_E, ⍰$_e$: The expected number of eggs produced in a biomass distribution in the population under virgin (E) and exploited (e) conductions.

A_h: The average age at maturity of fish in a population. Sometimes referred to as the age at 50% maturity.

A_v: The average age of vulnerability of fish in a population to a fishery. Sometimes referred to as the age at 50% vulnerability.

B: The total stock biomass

B_0. Virgin stock biomass, i.e., the long-term average biomass value expected in the absence of fishing mortality.

Biological Reference Points. Specific values for the variables that describe the state of a fishery system which are used to evaluate its status. Reference points are most often specified in terms of fishing mortality rate and/or spawning stock biomass. The reference points may indicate 1) a desired state of the fishery, such as a fishing mortality rate that will achieve a high level of sustainable yield, or 2) a state of the fishery that should be avoided, such as a high fishing mortality rate which risks a stock collapse and long-term loss of potential yield. The former type of reference points are referred to as “target reference points” and the latter are referred to as “limit reference points” or “thresholds”. Some common examples of reference points are F0.1, FMAX, and FMSY, which are defined later in this glossary.

Biomass Dynamics Model. A simple stock assessment model that tracks changes in stock using assumptions about growth and can be tuned to abundance data such as commercial catch rates, research survey trends or biomass estimates

Catch, C: The weight of fish removed from a population by fishing. The fate of the catch can vary. Some portion of the catch is discarded as bycatch or reported as landings.

E_0: The total number of eggs produced by the spawning population.

F_{MSY}. The fishing mortality rate that produces the maximum sustainable yield.

K: The somatic growth rate of fish in a population. This is a parameter of the von Bertalanffy growth model and is sometimes known as the Brody growth coefficient.

I: A annual index of abundance from a fishery-dependent or a fishery-independent survey.

κ (kappa): The compensation ratio, defined by Goodyear (1980). This parameter is a ratio of the slope of the stock recruitment relationship at low stock sizes to the slope of the line on the stock recruitment relationship that defines replacement. Estimates of k are bounded $1 < k < \infty$.

l_a: The expected survival probability of a fish to its a^{th} birthday.

L_a: The expected length of a fish of age a in the population.

L_∞: The expected maximum length of fish in a population. This is a parameter of the von Bertalanffy growth model.

M: The instantaneous rate of natural mortality.

m_a: The fraction of the population that is expected to be mature at age a.

Maximum Sustainable Yield (MSY). The largest average catch that can be taken from a stock under existing environmental conditions.

N: The total population abundance (in numbers).

r: The intrinsic rate of natural increase in a population. The value of r can theoretically range from - ∞ to ∞. A value of r=0 indicates no population growth.

R_0. Virgin recruitment, i.e., the long-term average recruitment value expected in the absence of fishing mortality.

Reference Points. Values of parameters (e.g. B_{MSY}, and F_{MSY}) that are useful benchmarks for guiding management decisions. Biological reference points are typically limits that should not be exceeded with significant probability or targets for management.

t_0: The hypothetical age of fish in a population with zero length. This is a parameter of the von Bertalanffy growth model.

U: The exploitation fraction – the proportion of the population available at the beginning of the fishing year that is caught by the end of the year.

U_{MSY}: The exploitation fraction taken when the population is harvested at MSY.

v_a: The fraction of the population that is expected to be vulnerable to the fishery at age a.

W_a: The expected weight of a fish in the population that is age a.

Z: The difference between the survey index of abundance and the predicted stock biomass (B) in a given year

7. APPENDIX B: ADMB CODE USED TO IMPLEMENT THE STOCHASTIC STOCK REDUCTION ANALYSIS FOR SHAD AND RIVER HERRINGS.

```
//****************************************************
//        Programmer: Tom MIller
//        Project Name: American shad stock reduction analysis
//        Date:
//        Version:  1
//        Comments:  Based on Forest et al. 2008
//
//
//****************************************************/

DATA_SECTION

//Get life history parms
 init_int Amax;
 init_number linf;
 init_number vbk;
 init_number t0;

//Get life history schedule parms
 init_number lna;
 init_number b;
 init_number ah;
 init_number scale_ah;
 init_number av;
 init_number scale_av;

//Initial parameter guesses & phaze
 init_number lUmsy;
 init_int phz_Umsy;
 init_number lMSY;
 init_int phz_MSY;
 init_number im;
 init_number sigmam;
 init_int phz_m;
 init_number isigma;
 init_int phz_sigma;
 init_number ireck;
 init_number sigmareck;

//Get cpue and catch time series
 init_int syr;
 init_int eyr;
 init_vector Ct(syr,eyr);
 init_vector It(syr,eyr);
```

```
 int nyr;
 !!nyr=(eyr-syr)+1;
 !!Ct=1000*1000*Ct;

PARAMETER_SECTION
 // Leading parameters
 init_bounded_number logitUmsy(-10,10,phz_Umsy);
 init_bounded_number logMSY(0.01,1000,phz_MSY);
 init_bounded_number m(0,3,phz_m);
 init_bounded_number sigma(0,50,phz_sigma);
 number Umsy;
 number MSY;

 //establish calculated numbers
 sdreport_number So;
 number beta;
 number Ro;
 number reck;
 number Bo;
 number phiE;
 number phiB;
 number phiF;
 number utpen;

 //!!So=1.361186;
 //!!beta=0.008612995;

 //Set up vector arrays to hold all the life history estimates
 vector age(1,Amax);
 vector lx(1,Amax);
 vector la(1,Amax);
 vector wa(1,Amax);
 vector ma(1,Amax);
 vector va(1,Amax);
 vector fa(1,Amax);

 //Set up vectors and matrices to hold the population dynamics
 matrix nt(syr,eyr+1,1,Amax);    //rows 1960 - 2005 and columns 0 to 14
 vector ut(syr,eyr);
 vector Bt(syr,eyr+1);
 vector Nt(syr,eyr+1);
 vector Zt(syr,eyr);
 vector epsilon(syr,eyr);

 objective_function_value f;

 //Set up leading parameters
 LOCAL_CALCS
```

```
   logitUmsy=lUmsy;
   logMSY=lMSY;
   m = im;
   sigma = isigma;
 END_CALCS

PROCEDURE_SECTION
 //*********MAIN FUNCTION CALLS*************
    initial_calculations();
    //cout<<"  Initial calculations completed "<<endl;

    get_leading_parms();
    //cout<<"  Got leading parameters!"<<endl;

    age_Model();
    cout<<"   Age structured population simulated  "<<endl;

    calc_objfunc();
    //cout<<"  Objective function minimized   "<<endl;

    //code to output MCMC results
    if (mceval_phase()) MCMC_report();

 //*********************************************

FUNCTION initial_calculations
 // set up life history schedules
 dvar_vector age(1,Amax);
 age.fill_seqadd(1,1);

 lx=pow(exp(-m),(age-1));
 la=linf*(1.-exp(-vbk*(age-t0)));
 wa=exp(lna)*pow(la,b);
 ma=1./(1.+exp(-1*(age-ah)/scale_ah));
 va=1./(1.+exp(-1*(age-av)/scale_av));
 fa=elem_prod(ma,wa);

 phiE = sum(elem_prod(lx,fa));
 phiB= sum(elem_prod(elem_prod(lx,wa),va));

 nt.initialize();

 //++++++++++++++++++++++++++++++++++++++++++++

FUNCTION get_leading_parms

 int i;
 dvariable lz;
```

```
 dvariable lzz;
 dvariable DlDu;
 dvariable DphifDu;
 dvariable DphibDu;

 //set initial values
 lz = 1.0;
 lzz = 1.0;
 phiF = 0.0;
 phiB = 0.0;
 DlDu = 0.0;
 DphifDu = 0.0;
 DphibDu = 0.0;

 Umsy=mfexp(logitUmsy)/(1+mfexp(logitUmsy));
 MSY=mfexp(logMSY);

 for(i=1; i<=Amax; i++)
 {
		phiF = phiF + lz*fa(i);
		phiB = phiB + lz*wa(i)*va(i);
		if (i>1) DlDu=DlDu*exp(-m)*(1.-Umsy*va(i-1))-lzz*exp(-m)*va(i-1);
		DphifDu = DphifDu + fa(i)*DlDu;
		DphibDu = DphibDu + va(i)*wa(i)*DlDu;
		lzz = lz;
		lz = lz*exp(-m)*(1.-Umsy*va(i));
 }

 reck=(phiE*(phiB*phiF+Umsy*DphibDu*phiF-
Umsy*phiB*DphifDu))/(square(phiF)*(phiB+Umsy*DphibDu));
 Ro=MSY*(reck-1)/(phiB*Umsy*(reck-phiE/phiF));
 Bo=Ro*phiB;

 //+++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++

FUNCTION age_Model
 int i;

 // preliminary calcs
 So = reck / phiE;
 beta = (reck - 1)/(Ro*phiE);
 nt(syr)=lx*Ro;    //should be lx*ro
 utpen=0.;

 for(i = syr; i <= eyr; i++)
  {
	ut(i)=Ct(i)/sum(elem_prod(elem_prod(nt(i),va),wa));
	if(ut(i)>1)
```

```
        { ut(i)=0.75;
             utpen+=square(ut(i)-0.75)*100;
        }
        dvariable E0=sum(elem_prod(nt(i),fa));
        nt(i+1,0)=So*E0/(1.+beta*E0);
        if(nt(i+1,0)<0)
       {
        nt(i+1,0)=0;
        }
        nt(i+1)(1,Amax) =++ elem_prod(nt(i)(0,Amax-1),exp(-m)*(1.-ut(i)*va(0,Amax-1)));
        Nt(i)=sum(nt(i));
  }

 Bt=elem_prod(wa,va)*trans(nt);   // vulnerable biomass vector

//++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++

FUNCTION calc_objfunc
  dvar_vector T(1,4);
  T.initialize();

 // Observation model
   Zt=log(It)-log(Bt(syr,eyr));
   epsilon=Zt-mean(Zt);

  T[1] = (double(nyr)/2.0)*log(norm2(epsilon));
  T[2] = 0.5*(square(m-im)/sigmam);
  T[3] = 0.5*(square(reck-ireck)/sigmareck);;
  T[4] = utpen;
  f=sum(T);

  cout<< "likelihood"<<f<<endl;

//++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++

FUNCTION MCMC_report

  ofstream ofest("mcmc_results.dat", ios::app);
  {
    ofest << f << " " << ut<< " " << Nt << " " << Bt<< endl;
  }
  ofstream ofpar("mcmc_par.dat", ios::app);
  {
    ofpar << logitUmsy << " " << logMSY << " " << m << " " << sigma<< endl;
  }

//+++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++++
REPORT_SECTION
```

```
report<<"Observed catches"<<endl<<Ct<<endl;
report<<"Population abundance"<<endl<<Nt<<endl;
report<<"Vulnerable biomass"<<endl<<Bt<<endl;
```

8. APPENDIX C: SAMPLE INPUT DATA FILE

```
#Am_shad_SRA.dat

#Maximum age
14

# Von B parms
#  Linf (cm) , k, t0
58.7  0.4  -0.1

# L-W relationship lna  b
-12.1377   3.2207

# age at 50% maturity and shape
4.77989  0.46114

# age at 50% vulnerability and shape
5.3005  0.896782


#Initial parameter guesses and phz (NB use -1 for phz if not estimating)
#LUmsy
0.25      1
#lMSY
7          1
#m, its variance and phase
0.225    0.01   -1
# sigma
0.1149  2
# reck and its variance
35        0.1

# period for catch and cpue data
1980   2005


# catches  (Total catch in MT x 103)

1.631099441
1.267616543
1.343296956
1.351788299
1.800836997
1.395488914
1.596125111
1.688247584
1.607671914
```

```
1.593865948
1.450545729
1.210606984
1.084627918
0.850670043
0.61989239
0.738481845
0.883493061
0.906539197
1.054038364
0.590485958
0.841700875
0.731890006
0.652075696
0.607131844
0.492424569
0.37443044


# CPUE  (Lewes haul seine CPUE)

12.97
54.17
29.83
14.44
15.68
29.3
30.67
16.49
35.62
52.2
25.35
30.42
50.96
10.52
7.9
19.05
3.67
11.96
13.2
4.6
4.07
6.84
3.85
5.23
4.07
2.89
```