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RE: Scoping Comments on Amendment 15 to the Atlantic Mackerel, Squid, and Butterfish Fishery Management Plan (RIN 0648–XC329)

Dear Dr. Moore:

We are writing on behalf of the Herring Alliance[1] to provide scoping comments on proposed Amendment 15 to the Atlantic Mackerel, Squid, and Butterfish Fishery Management Plan ("Scoping Document" for Amendment 15 to the MSB FMP). The Mid-Atlantic Fishery Management Council is legally obligated under the Magnuson-Stevens Fisheries Conservation and Management Act, to conserve and manage depleted river herring and shad.[2] Monitoring program improvements and promised river herring catch caps in Amendment 14 to the MSB FMP and Amendment 5 to the Atlantic Herring FMP will not provide sufficient management of river herring and shad, either under the law or on the water. These fish need science-based catch limits, rebuilding plans, and other required measures to recover and reclaim their role as keystone species in the ocean and coastal ecosystem. Specifically, the Herring Alliance urges development of the following conservation and management alternatives:

- Science-based annual catch limits that prevent overfishing including:
  - Options for forage-based ABC control rules;
  - Options that allow for establishing overfishing levels and corresponding catch limits under a range of information scenarios;
  - Additional options for criteria for determining if overfishing is occurring and for when a stock has reached an overfished condition;
- Alternatives to establish rebuilding plans to restore populations to sustainable levels;

[1] The Herring Alliance includes 52 organizations representing over 2 million individuals concerned about the Atlantic coast's forage fish that play a critical role in the food web as prey to a large number of predators, many of which support valuable recreational and commercial fisheries. A current list of members is attached to this letter. The Herring Alliance is also resubmitting its June 4, 2012 comments on Amendment 14 because they are directly relevant to many questions raised in the Amendment 15 Scoping Document.

[2] Amendment 15 seeks to conserve and manage four species involved in the MSB fishery including blueback herring and alewife (collectively "river herring"), and American shad and hickory shad (collectively "shad").

- Alternatives that address the forage role that river herring and shad play in the ecosystem and how this role will be accounted for in establishing an Optimum Yield (OY), consistent with National Standard 1 Guidelines;
- Measures necessary to collect the coast-wide data needed to improve scientific assessments and to monitor the health of these fish populations;
- Accountability measures that (1) close the fishery before catch limits are exceeded (at 85% of the catch limit), and (2) deduct any overages from the next year;
- Identification of essential fish habitat and closures to protect critical habitat areas;
- Additional measures to avoid and minimize the incidental catch of river herring and shad, including further development of bycatch "hotspots" avoidance and closure alternatives initially developed in Amendment 14;
- An option to manage these stocks as part of the MSB FMP that is coordinated with the ASMFC Shad and River Herring and NEMFC Atlantic Herring plans with a clear delineation of responsibilities on necessary issues like catch limits, and additional options for joint management of river herring and shad through the appropriate FMPs;
- Alternatives for protecting additional forage species;

The analysis of these and other proposed alternatives must carefully evaluate the full range of benefits from the actions proposed to protect and restore river herring and shad populations, including the ecological benefits to the species that depend on them for food and the economic benefits to coastal communities from related fisheries, tourism, and other businesses.

Introduction

The Mid-Atlantic Council is legally obligated under the Magnuson-Stevens Fishery Conservation Management Act ("Magnuson-Stevens Act") to prepare a plan for all stocks in a fishery under its management that require conservation and management. 16 U.S.C. §§ 1802(13), 1852(h)(1). Neither the current management regime under the ASMFC's Shad and River Herring Fishery Management Plan nor pending regulatory measures to be implemented under Amendment 14 to the MSB FMP will adequately conserve and manage these species. Conservation and management of river herring and shad by the Mid-Atlantic Council would bring the MSB FMP into compliance with the Magnuson-Stevens Act, and has widespread public support.[3]

Current management under the Atlantic States Marine Fisheries Commission's ("ASMFC") Shad and River Herring Interstate Fishery Management Plan ("ISFMP") is insufficient, among other

[3] Over 46,000 individuals' submitted comments into the Amendment 14 record in support of designating river herring and shad as stocks in the fishery under alternative set 9. *See* June Meeting Documents, available at: http://mafmc.org/fmp/msb_files/msbAm14current.htm. In addition, the Council received comments supporting inclusion of these species from hundreds of organizations and dozens of Congressmen and State legislators from all over the Mid-Atlantic region, who called upon the Council to add river herrings and shad as managed stocks. *See* http://mafmc.org/fmp/msb_files/Am14/Comment-Supplement.pdf. These comments expressed the belief that the Council could and would act on the question of whether to add these stocks in Amendment 14. We formally request that you review these comments and include them in the Amendment 15 record.

reasons, because it fails to address mortality of river herring and shad in federal waters[4] and does not establish a plan for the successful recovery of these species. Communities across the Atlantic seaboard have spent countless hours working to restore habitat, monitor water quality, and clean waterways to help restore river herring and shad populations. State and local governments have devoted millions of dollars towards restoring our coastal estuaries and rivers by regulating pollution, restoring habitat, and stocking rivers. However, despite these efforts and longstanding bans on fishing both in-river and in coastal state waters in a number of states, most river herring and shad runs have not recovered to sustainable levels and runs have continued to decline in many places. In 2012, all but five (5) states[5] on the East Coast placed a moratorium on river herring in state waters for both commercial and recreational fishing. Even in the states without a moratorium, fishing for river herring is extremely restricted. In 2013, many states will add new restrictions to the catch of American shad within state waters, or implement moratoriums as well. Without a federal management plan that complements the rebuilding efforts within state waters, river herring and shad fisheries in state waters are unlikely to reopen in the future.

Although the Mid-Atlantic Council's decisions in Amendment 14 to improve monitoring and develop a mortality cap for river herring and shad were positive steps, Amendment 14 is also insufficient.[6] River herring and shad are in desperate need of conservation and management, yet Amendment 14 contains no effective annual catch limits ("ACLs") or accountability measures ("AMs") that limit how many river herring and shad can be taken from federal waters, where these fish spend most of their lives. The mortality cap measure passed by the Council, even if implemented by the National Marine Fisheries Service ("NMFS" or "NOAA Fisheries"), does not require that the cap be set at a level that ensures at-sea mortality will be limited to levels that will rebuild and maintain river herring and shad populations. The New England Council's decisions in Amendment 5 to the Atlantic Herring FMP also improved catch monitoring, however, the mortality cap contemplated in that action was merely a promise to consider establishing a similar cap in in a future action.

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[4] At-sea catch of river herring and shad has continually been identified as a significant threat to the restoration of these anadromous species, and a significant factor in the decline of river herring and shad populations over the last 50 years. *See* ASMFC. May 2012. Stock Assessment Report No. 12-02 of the Atlantic States Marine Fisheries Commission: River Herring Benchmark Stock Assessment, Volume 1; *see also* ASMFC American Shad Stock Assessment Peer Review Panel. Stock Assessment Report No. 07-01 of the Atlantic States Marine Fisheries Commission, Terms of Reference & Advisory Report to the American Shad Stock Assessment Peer Review. Conducted on July 16-20, 2007, Alexandria, Virginia; *see also* June 24, 2009 Letter from MAFMC Chairman Richard B. Robins, Jr. to Secretary of Commerce Gary Locke.

[5] Maine, New Hampshire, New York, North and South Carolina submitted Sustainable Fishing Plans under ASMFC regulations and received approval from ASMFC for limited in-river and state waters fisheries.

[6] Amendment 14 provides a list of the foregone benefits of a decision <u>not</u> to list river herring and shad as stocks in the fishery: 1) no additional federal support of management or additional coordination among partners including more federal involvement in assessments; 2) no explicit consideration of river herring and shad observer coverage needs; 3) no direct controls (ACLs/AMs) on federal catch of river herring and shad; 4) inability to address the catch and/or discarding in other fisheries; 5) Essential fish habitat would not be designated which could mean less habitat improvements for RH/S. Amendment 14 DEIS (April 2012), p. 440.

I. The Magnuson-Stevens Act Requires Depleted River Herring and Shad be Added as Stocks in the MSB FMP

The Mid-Atlantic Council must include river herring and shad as non-target stocks within the MSB-FMP as required by the Magnuson-Stevens Act and provided for by the revised National Standard 1 (NS1) Guidelines.[7] Under the Magnuson-Stevens Act, NOAA Fisheries must, through the regional councils, prepare an FMP or amendments for ***all*** fish stocks that are in need of conservation and management.[8] This requirement was recently affirmed in *Flaherty v. Bryson*, which reiterated the Act's directive that, under Section 302 of the Magnuson-Stevens Act, Councils must prepare an FMP or amendment for any stock of fish that "requires conservation and management."[9] The Council must then set ACLs, AMs, and other conservation and management measures for all of the stocks in the fishery.[10]

As shown below, river herring and shad are in desperate need of conservation and management at the federal level. Management could take place directly through federal FMPs created by regional councils and implemented by NMFS, a Secretarial FMP created and implemented by NMFS alone, or through NMFS implementation of regulations consistent with an ISFMP and the Magnuson-Stevens Act's National Standards.[11] In this case, given the absence of independent action by NMFS, the Mid-Atlantic Council's failure to including river herring and shad in the MSB FMP would violate the Magnuson-Stevens Act, and be inconsistent with the NS1 Guidelines.

As an initial step, the Magnuson-Stevens Act requires that federal FMPs describe the fish stocks involved in a fishery.[12] To comply with the Act's mandate to prevent overfishing, the revised NS1 Guidelines require relevant councils to identify the stocks in the fishery, including the non-targeted stocks that are caught incidentally and retained or discarded at sea. Regulations define 'non-target stocks' as fish that are "caught incidentally during the pursuit of target stocks in a fishery, including 'regulatory discards' as defined under section 3(38) of the Magnuson-Stevens Act. They may or may not be retained for sale or personal use."[13] In *Flaherty*, the Court made clear that the Magnuson-Stevens Act requires management of populations in need of conservation and management, such as depleted river herring and shad, stating:

> . . .the MSRA [Revised Magnuson-Stevens Act] requires ACLs and AMs for <u>all</u> stocks in need of conservation and management, not just those stocks which were part of the

[7] 50 CFR § 600.310(d)(3-4).
[8] *See* 16 U.S.C. § 1852(h)(1); § 1854(c)(1); 16 U.S.C. § 5103(b)(1). (Emphasis added). *See also Flaherty v. Bryson*, 850 F.Supp.2d 38, 53-54 (D.D.C. Mar. 8, 2012).
[9] *See* 850 F.Supp. 2d at 53-54. ("[16 U.S.C. § 1852(h)(1)] requires FMPs and necessary amendments for all 'stocks of fish which can be treated as a unit for purposes of conservation and management' and which are in need of conservation and management. *Id.* §§ 1802(13)(a), 1852(h)(1).").
[10] *See* 850 F.Supp. 2d at 51.
[11] *See* 16 U.S.C. § 5103 (b)(1). This provision of the Atlantic Coastal Fishery Cooperative Management Act provides that in the absence of an approved and implemented federal FMP, after consulting the appropriate council(s) NMFS can implement regulation for federal waters that are both compatible with the ISFMP and consistent with the national standards. Regulations implementing an approved federal FMP that was prepared by the appropriate council would supersede regulations issued by the Secretary alone.
[12] 16 U.S.C. § 1853(a)(2).
[13] 50 C.F.R. § 600.310(d)(4).

> fishery prior to the passage of the MSRA…. The setting of ACLs and AMs necessarily entails a decision as to which stocks require conservation and management.

850 F. Supp. 2d at 52. Further, the Court held that NOAA Fisheries' passive approval of the New England Fishery Management Council's (New England Council's) failure to include river herring as a non-target stock in the Atlantic Herring FMP, without ensuring that it was consistent with the Magnuson-Stevens Act's "conservation and management requirement," was unlawful. *Id.* at 55. In its final review of Amendment 15, NOAA Fisheries must ensure that the Council decision regarding the stock in the fishery question complies with the requirements of the Magnuson-Stevens Act, and must disapprove the decision if it does not.[14]

Contrary to the suggestion in the Scoping Document that the benefits of management must outweigh the costs,[15] the decision about which stocks to include in the fishery under the Magnuson-Stevens Act is not discretionary.[16] Courts are clear that the Magnuson-Stevens Act prioritizes conservation over economic factors.[17] The question here is whether river herring and shad need conservation and management, not whether the costs to the agency are justified by the benefits of conservation. For the purposes of Amendment 15, the Council must determine which species of fish are capable of management as a unit, and therefore should be included in the fishery and managed together in the plan, and whether river herring and shad are in need of conservation and management.[18] The fact that there are conservation and management measures in state waters is irrelevant to whether or not river herrings and shad are in need of conservation and management measures in *federal* waters. Further, the fact that NOAA Fisheries and the ASMFC have failed to officially designate these stocks as overfished or subject to overfishing is equally irrelevant.[19]

A. <u>River Herring and Shad Need Conservation and Management in Federal Waters</u>

The Mid-Atlantic Council should look to the definition of "conservation and management"[20] found in the Magnuson-Stevens Act when making its decision to add these species to the FMP. This definition addresses stocks where action is necessary to rebuild, restore, or maintain "any fishery resource and the marine environment," to ensure a constant food supply and recreational benefits, and to avoid irreversible or long-term adverse effects on the fishery resources and the marine environment. National Standard 7 and its guidelines include additional criteria that can provide limited additional guidance.[21]

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[14] 16 U.S.C. § 1854(a)(1)(A).
[15] *See*: http://www.mafmc.org/fmp/msb_files/Am15/SCOPING_DOC_MSB15_FINAL.pdf, p. 7.
[16] *See 850 F.Supp.2d at 56.* ("Simply put, 50 C.F.R. § 600.310(d)(1) cannot be understood to permit NMFS to ignore its duty to ensure compliance with the MSA. The councils do not have unlimited and unreviewable discretion to determine the makeup of their fisheries.").
[17] *See e.g., NRDC v. NMFS*, 421 F.3d 872, 879 (9th Cir. 2005); *NRDC v. Daley*, 209 F.3d 747, 753 (D.C. Cir. 2000)
[18] *See* 16 U.S.C. §§ 1802(13), 1852(h)(1); *see also* 850 F.Supp.2d at 51.
[19] *See* 850 F.Supp. 2d at 54-55.
[20] 16 U.S.C. § 1802(5).
[21] *See* 50 C.F.R. 600.340(b). Although the criteria note that adequate management by an entity like the ASMFC could be one factor in determining whether a stock should be added to a fishery, in this case, the ASMFC plan does not address the catch of river herring and shad in federal waters. The Court in *Flaherty v. Bryson* did not address this in its opinion because even NMFS recognized that the ASMFC plan does not address federal waters.

River herring and shad populations are at historic and dramatic lows.[22] Currently river herring and shad are managed by the Atlantic States Marine Fisheries Commission (ASMFC) under Amendments 2 and 3 to the Interstate Fishery Management Plan for Shad and River Herring. The ASMFC's recently released stock assessment for river herring found that alewife and blueback herring along East Coast are "depleted," with many populations in a dangerously diminished state.[23] Their disappearance from traditional fishing grounds in rivers and estuaries is alarming, not only for the communities and fishermen that depend on them, but for the coastal ecosystem as a whole. Restoration of these anadromous species depends on a comprehensive management plan that protects them throughout their lifecycle, including while at sea where most of the life cycle is carried out.

Total catch (bycatch and incidental) in federal waters is impeding shad and river herring rebuilding efforts. According to the ASMFC's 2012 stock assessment, at-sea fisheries are a significant factor in the decline of river herring populations over the last 50 years.[24] In some years, more than 2 million pounds of adult and juvenile river herring are killed incidentally in federal waters by at-sea fisheries, of which the Mid-Atlantic mackerel and squid fisheries contribute to approximately half of the total at-sea catch.[25] Of the roughly 5 million river herring taken at sea every year, many are immature. The majority of the 600,000 American shad taken are also juveniles.[26] High fishing mortality on immature fish has a significant negative effect on stock status and reduces the effectiveness of rebuilding efforts,[27] an issue of concern highlighted by the Peer Review Panel in the recent river herring stock assessment.[28] The Peer Review Panel also found that total mortality levels in all runs, for which a determination could be made, surpassed the recommended mortality benchmark and called for all sources of mortality to be addressed, including ocean bycatch.[29] NMFS observer records show that at-sea fishing vessels may take as much as 20,000 pounds of blueback herring in a *single net haul*.[30] To put this in perspective, consider that the 2008 commercial blueback herring landings from the states of New

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[22] The ASMFC lists the status of American shad, alewife and blueback herring as "depleted" based on the most recent stock assessments for these species. *See* American Shad: ASMFC. August 2007. Stock Assessment Report No. 07-01 (Supplement) of the Atlantic States Marine Fisheries Commission: American Shad Stock Assessment for Peer Review, Volume 1. River Herring: ASMFC. May 2012. Stock Assessment Report No. 12-02 of the Atlantic States Marine Fisheries Commission: River Herring Benchmark Stock Assessment, Volume 1. *See also:* 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' Thesis, Marine and Atmospheric Science, Stony Brook University; Limburg KE, Waldman JR (2009) Dramatic Declines in North Atlantic Diadromous Fishes. BioScience 59(11): 955-965. The ASMFC does not have a stock assessment for hickory shad, and many states list their status as "unknown." However, some states, such as Pennsylvania, list hickory shad on its states endangered species list. *See* http://www.pacode.com/secure/data/058/chapter75/chap75toc.html.

[23] *See* ASMFC, River Herring Benchmark Stock Assessment Report, Executive Summary.

[24] *See* River Herring Benchmark Stock Assessment, Peer review report, p. 8.

[25] *See* Amendment 14 DEIS, Incidental Catch Analysis, p. 571

[26] *See* Amendment 14 DEIS, p. 111

[27] *See* Vasilakopoulos, P., O'Neill, F. G., and Marshall, C. T. 2011. Misspent youth: does catching immature fish affect fisheries sustainability? – ICES Journal of Marine Science, 68: 1525–1534.

[28] *See* ASMFC. May 2012. Stock Assessment Report No. 12-02 of the Atlantic States Marine Fisheries Commission: River Herring Benchmark Stock Assessment, Volume 1. pp. 15-16.

[29] *Id*, at p. 29.

[30] Haul data from North East Fisheries Observer Program, NMFS; Landings data from NOAA's Annual Commercial Landing Statistics: www.st.nmfs.noaa.gov/st1/commercial/landings/annual_landings.html

York, Delaware, and Virginia *combined* totaled just 26,000 pounds. If the fish are aggregated while at sea, a single haul could obliterate an entire river's population of blueback herring.

B. River Herring and Shad are Involved in the MSB Fishery and Capable of Being Managed as a Unit

The Scoping Document for Amendment 15 and the Amendment 14 DEIS illustrate that significant numbers of river herring and shad are caught in federal waters in the MSB fishery, and are therefore these species are involved in this fishery. As indicated above, the relevant legal standard under the Magnuson-Stevens Act requires that Councils determine which stocks of fish are capable of management as a unit, and therefore should be included in the fishery and managed together in the plan. To do this, Councils "must decide (1) which stocks "can be treated as a unit for purposes of conservation and management" and therefore should be considered a "fishery" and (2) which fisheries "require conservation and management." 16 U.S.C. §§ 1802(13), 1852(h)(1)."[31] The Councils must then set ACLs and AMs and other required management measures for those stocks.[32]

There is no question that restoration of these species will depend on management actions that protect them throughout their range and lifecycle. To accomplish this while these fish are at sea – where they spend the majority of their lives – the Mid-Atlantic Council's MSB FMP must include measures not only to monitor their catch, such as those recommended to NMFS by the Council through Amendment 14, but also to ensure their conservation and management. The Scoping Document itself provides multiple examples of successful management of fish that cross state and federal management boundaries.[33] River herring and shad must receive similar conservation management, including addition to the MSB FMP where they are currently caught and already monitored as part of the management unit.

With respect to the question of the appropriate management unit for these fish, raised in the Scoping Document, river herring and shad should be managed in federal waters as a biological unit consistent with the other stocks in the MSB fishery (i.e. all northwest Atlantic mackerel, longfin squid, and butterfish under U.S. jurisdiction[34]), with allocations (sub-ACLs) assigned to the fisheries (economic unit) that have a significant impact on shad and river herring populations. The mid-water trawl fishery for Atlantic herring and Atlantic mackerel (managed by two councils under two separate federal FMPs) accounts for 71% of combined river herring and shad incidental catch. Likewise, fleet overlap between the New England and the Mid-Atlantic small-mesh bottom trawl fisheries, are responsible for an estimated 24% of the combined incidental catch.[35] The Draft Action Plan for Amendment 15 acknowledges that the Atlantic mackerel fishery has a significant impact and notes that other small-mesh fisheries under its management may be affected.[36] River run-by-run variability should not impede management in federal waters

[31] *See* 850 F.Supp.2d at 51.
[32] *Id.*
[33] *See* Scoping Document, pp. 10-11.
[34] *See* Amendment 14 DEIS, p. 116.
[35] *See* Amendment 14 DEIS, Appendix 2, Table 3, p. 581.
[36] *See* October 16, 2012 Draft Action Plan, available at: http://www.mafmc.org/.

while scientists and managers collect the data necessary to fully understand the variability of river runs and the mixing of river herring and shad populations while at sea.

II. Goals and Objectives of Amendment 15

As it develops Amendment 15, the Council should update the overarching goals and objectives of the MSB FMP. The following recommendations, specific to Amendment 15, are consistent with the goals and objectives of the MSB FMP, Amendment 14 to the MSB FMP, and Amendment 5 to the Atlantic Herring FMP. These recommendations should be considered as alternatives to the goals and objectives of Amendment 15 and should also be incorporated as appropriate into the updated goals and objectives of the MSB FMP.

The Goal of Amendment 15 should be to manage river herring and shad at long-term sustainable levels consistent with the National Standards of the Magnuson- Stevens Act.

Objectives of Amendment 15:

1. Implement management measures to ensure compliance with the Magnuson-Stevens Act.
2. Promote the health of these four species consistent with their important role as forage fish throughout their ranges.
3. Enhance the probability of successful (i.e., the historical average) recruitment to the fisheries.
4. Increase understanding of the conditions of river herring and shad.
5. Increase marine recreational fishing opportunities, recognizing the contribution of recreational fishing to the national economy.
6. Minimize harvesting conflicts among commercial and recreational fishermen.
7. Protect and preserve the biological, recreational and commercial integrity of all stocks caught in the fishery.

III. Amendment 15 and its DEIS Should Include the Following Conservation and Management Alternatives and Analyses:

1. <u>Science Based Annual Catch Limits That End and Prevent Overfishing</u>

Alternatives in Amendment 15 should include science-based ACLs that prevent overfishing based on a forage-based ABC control rule and options that allow for establishing overfishing levels and corresponding catch limits under a range of information scenarios. Analysis should include:

- Options for developing harvest controls (ABC control rules) that are appropriate for forage fish. Recent scientific studies have pointed to the importance of (1) managing forage stocks to relatively high biomass targets; and (2) employing a conservative harvest control rule that systematically reduce catch rates as stock biomass falls below the target, curtailing fishing when a threshold has been reached; *hockey-stick* harvest control

strategies are recommended.[37] The amendment should develop alternative control rules that employ current scientific recommendations and are tailored to river herring and shad.

- Options that allow for establishing overfishing levels and corresponding catch limits under a range of information scenarios including:
    - Catch history only for coast-wide stock complexes, for example the ORCS working group approach.[38]
    - Depletion-Based Stock Reduction Analysis (DB-SRA).
    - Population dynamic assessment modeling approaches that may become appropriate in the future when additional data are available.
- Additional options for determining if overfishing is occurring and for when an overfished state has been reached. To the extent the above methods are determined to be insufficient for establishing one or more of the required status determination criteria, alternatives should be developed for the use of proxies consistent with current Council policy.

2. Rebuilding plans to restore populations to sustainable levels

As described above, river herring and shad are severely depleted and their populations need to be rebuilt to higher levels. Although rebuilding plans are typically established after NMFS officially designates a stock as overfished, the Magnuson-Stevens Act requires that FMPs contain conservation and management measures that are necessary to "rebuild overfished stocks, and to protect, restore, and promote the long-term health and stability of the fishery." 16 U.S.C. § 1853(1). Because river herring and shad are overfished coastwide by any objective measure, the Mid-Atlantic Council should include options for establishing rebuilding plans consistent with 16 U.S.C. 1854(e)(4).

3. Alternatives that Address the Role of River herring and Shad as Forage in the Ecosystem and How this Role will be Accounted for in Establishing Optimum Yield Consistent with NS 1 Guidelines

Amendment 15 must explicitly analyze the impacts of forage fish catch decisions on all the other fisheries dependent on forage fish as prey in the water, as opposed to just considering the impacts on forage fish harvesters, processors, and secondary markets such as bait users. Further, such analysis should also apply to economic and social OY adjustments, where these "derivative impacts" (*i.e.*, impacts on other fisheries dependent on healthy forage populations as prey) are typically overlooked. Based on the best available science that shows river herring and shad are depleted, there should be an explicit prohibition on directed fishing for these species in federal waters until such time as they are rebuilt. In addition, when analyzing alternatives that address the role of river herring and shad as forage in the ecosystem and establishing optimum yield ("OY"), the following issues should be analyzed:

[37] Pikitch E et al (2012) Little Fish, Big Impact: Managing a Crucial Link in Ocean Food Webs. Lenfest Ocean Program. Washington, DC. 108 pp.; Smith ADM et al (2011). Impacts of Fishing Low–Trophic Level Species on Marine Ecosystems. *Science* 333 (6046): 1147-50.

[38] *See* NOAA Technical Memorandum NMFS-SEFSC-616 (2011), III the ORCS working group approach in Calculating Acceptable Biological Catch for stocks that have reliable catch data only, pp. 19-26.

- Optimum yield should take into account ecosystem considerations. Ecosystem considerations for river herring and shad are particularly important because they are forage fish. As such, the dependence of other components of the ecosystem such as predatory fish, birds, and protected marine mammals is unusually high. Plans for ascertaining OY for these and other forage fish should include an evaluation of predatory dependencies under current stock conditions (i.e., depleted) and under biomass levels predicted with successful rebuilding. Options for AM 15 should include:
    - Evaluating river herring and shad in mid-Atlantic ecosystems as forage, including developing expectations about ecosystem needs for these fishes based upon new ecosystem modeling studies and meta-analyses from other ecosystems.
    - Developing a formal ecosystem modeling framework in which the role of river herring and shad as forage fish for the ecosystem can be predicted under a range of conditions, including scenarios in which depleted predator populations grow. For example, federal programs are directed at increasing the biomass of Atlantic cod and other groundfish, whales and some endangered seabirds (e.g., roseate tern). An ecosystem model could be used to examine how predator demands for river herring and shad may increase if these other programs are successful. This type of analysis must be considered in establishing harvest levels that provide optimal benefit for the nation.
- Options for establishing OY should explicitly state OY, outline the derivation of OY, and provide detailed justifications for OY. Among the values specifically listed in the NS1 Guidelines for serious attention in the application of OY adjustments is "maintaining adequate forage for all components of the ecosystem."[39] The NS1 Guidelines go further, directing that in FMPs "consideration should be given to managing forage stocks for higher biomass than B_{MSY} to enhance and protect the marine ecosystem."[40]

4. <u>Measures Necessary to Collect Coast-wide Data and Improve Scientific Assessments</u>

Amendment 15 should include measures that require the collection of coast-wide data and improve the scientific assessments in order the monitor the health of these fish populations. Specifically, alternatives should be analyzed which improve the sampling and analysis of at-sea catch of river herring and shad, even in high-volume fisheries where the catch of these species is a small percentage of very large catches of other fish, such as in the SMB fishery. Although significant improvements in fishery monitoring, and thus in the data on river herring and shad catch in the SMB fishery, will result from the implementation of Amendment 14, Amendment 15 should include additional options for improving data collection and scientific understanding of the impacts of the SMB fishery on river herring and shad, including:

- Options for deployment of federal resources to census sentinel rivers and streams within coastal watersheds.
- Monitoring of individual river runs should be augmented by development of electronic tagging (PIT tags) and through genetic tagging methods that are now used for similar purposes to monitor salmon in some regions of the Pacific. These genetic methods should be developed so that they can be used to ascertain the geographic origin of river

[39] 50 C.F.R. § 600.310(e)(3)(iii)(C).
[40] 50 C.F.R. § 600.310(e)(3)(iv)(C).

herring and shad caught at sea. As migratory fish, river herring and shad cover hundreds of miles while at sea. However, they are accessible as they ascend spawning rivers and streams which offers an opportunity for improved monitoring of individual components of the meta-population when caught at sea.

5. Accountability Measures To Close the Directed Fishery Before Catch Limits are Exceeded and Promptly Deduct Overages

Amendment 15 should include a wide range of accountability measures that close the directed fishery before catch limits are exceeded (pro-active AMs) and promptly deduct overages (reactive AMs) when they occur including the following options:

- Options for an in-season closure of the fishery at precautionary thresholds with a high likelihood of ensuring that catch limits for river herring and shad are not exceeded (i.e. 85% of the ACL or ACT). Although the Atlantic Herring FMP aims to close directed herring fishing at 95% of the sub-ACL, this level has proven inadequate to prevent overages, with the sub-ACL exceeded almost 25% of the time between 2003 and 2012.[41]
- Options for pound-for-pound paybacks of catch limit overages, to be applied in the fishing year immediately following the year in which the overage took place.
- Options to modify the definition of directed MSB fishing and thus revise the universe of fishing activity that ceases after a river herring or shad catch limit is exceeded. This would more effectively ensure that directed MSB fishing stops, that discarding does not continue or increase, and that river herring and shad removals cease if a catch limit is reached. The options should lower the incidental trip allowance of mackerel that can be fished for, possessed or retained upon closure. The current mackerel incidental allowance of 20,000 pounds may not sufficiently deter directed fishing. This alternative set should be modified to be consistent with the Atlantic Herring FMP, which uses a 2,000 pound incidental Atlantic herring limit to define, deter, and close directed herring fishing, including for the purposes of enforcing herring ACL's and sub-ACL's.[42] This incidental limit has proven effective in Atlantic herring management[43] and would provide for more consistent regulation of the mixed herring and mackerel fisheries, including for the purposes of river herring and shad catch limits. Further, the implementing language for that incidental limit should be consistent with the language in the Herring FMP such that the 2,000 pound limit would apply to vessels "fishing for, catching, possessing,

[41] *See* NEFMC, Draft Tri-Annual Specifications Document for Atlantic Herring 2013-2015, available at: http://www.nefmc.org/herring/council_mtg_docs/Nov%202012/1_DRAFT%202013-2015%20Specs%20Document%20For%20Nov%202012%20Herring%20AP%20OS.pdf, pp. 58-60

[42] The 2,000 pound limit used to close the directed fishery was approved in Amendment 4 to the Herring FMP as part of the sole proactive Accountability Measure for preventing ACL overages. It is described at http://www.nefmc.org/herring/planamen/final_a4/AM%204%20DOCUMENT%20FORMAL%20SUBMISSION_100423.pdf, p. 29.

[43] While there have been numerous sub-ACL overages in the herring fishery that demonstrate the inadequacies of the Amendment 4 ACL/AM regime, these have typically been the result of a failure to close the directed fishery by implementing the 2,000 pound limit in a timely fashion, rather than as a result of any failure of the 2,000 pound limit to adequately end directed fishing.

transferring, or landing more than 2,000 lb (907.2 kg) of herring per calendar day in or from the specified management area for the remainder of the closure period."[44]

6. **Identification of Essential Fish Habitat and Closures to Protect Important Habitat Areas**

Alternatives in Amendment 15 should include:

- Options for designating critical habitat.
- Options for criteria to be used for identifying any necessary EFH protections, including closures, in future actions.
- Analysis of whether the bycatch hotspots identified and considered in Amendment 14 should be designated as EFH and evaluated as possible EFH closures based on the presence of river herring and shad in the water column.

7. **Additional Measures to Avoid and Minimize Incidental Catch of River Herring / Shad**

Amendment 15 should include additional measures to avoid and minimize the incidental catch of river herring and shad, including further development of bycatch "hotspots" avoidance and closure alternatives initially developed in Amendment 14 and the following options:

- Options to refine and further consider and analyze the closure to directed mackerel and squid fishing of temporal and spatial protection areas identified in AM 14 as having high rates of river herring or shad bycatch ("hot-spots").
- Because the hot-spots considered in AM 14 were small, and the MAFMC's Fishery Management Action Team (FMAT) analyses indicated that protection of small areas alone may not be adequate to effectively reduce incidental catch, consideration should be given to affording protection to larger areas, including those identified as "River Herring Monitoring/Avoidance Areas"[45] in Amendment 5 to the Atlantic Herring FMP.

8. **Management Coordination/Unit alternatives**

River herring and shad must be added to every FMP where they are caught in significant numbers. Options include the following:

- An option for separate Mid-Atlantic Council Management as part of the MSB FMP. Under this option, river herring and shad would be added to the MSB FMP, as required under the Magnuson-Stevens Act, and the Mid-Atlantic Council would manage river herring and shad independently within the plan. Coordination with ASMFC and New England Council would be required for certain measures to ensure complementary management, such as establishing the overall ACLs for river herring and shad, and the

[44] *See e.g.*, http://www.nero.noaa.gov/nero/regs/frdoc/12/12HerAear2ClosureTR.pdf.
[45] *See* Amendment 14 DEIS, pp. 72-77.

appropriate sub-ACLs for all fisheries where river herring and shad are caught. Areas of additional coordination may be established at the discretion of the management bodies.

- Additional options for joint management of river herring and shad through the appropriate FMPs should also be considered. Because river herring and shad must also be managed by the New England Council due to their involvement with (at least) the Atlantic herring fishery, and are already managed in state waters by the ASMFC, the Mid-Atlantic Council could work jointly with the New England Council and/or the ASMFC to manage river herring and shad through the appropriate plan or plans. Consideration of options for joint management must not slow down the timeline for AM 15 due to the severely depleted conditions of river herring and shad populations.
- All options evaluated should clearly delineate the relative responsibilities between the three management bodies and NMFS. For example, in addition to establishing the overall ACLs for river herring and shad, there should be robust data reporting between the management bodies.

9. Additional issues that must be analyzed in the EIS

- The analysis of these and other proposed alternatives should carefully evaluate the full range of benefits from actions proposed to protect and restore river herring and shad populations, including the ecological benefit to those species that depend on them for food, and consider the economic benefits to coastal communities from related fisheries, tourism, and other businesses.[46]
- River herring and shad have been an integral part of Atlantic coastal community life for centuries, and the MSB fishery is adversely affecting these economically, biologically, and culturally important resources. In previous decades, when abundance was substantially higher, these fish also played a key role as forage for a great number of predators including larger, commercially important fish such as Atlantic cod and striped bass – alosines were once a vital link between the sea and coastal estuaries, streams and lakes. These ecological and cultural functions must be restored. Further, because they are forage fish critically important to the diets of dozens of other marine and terrestrial species, these adverse impacts ripple through the ecosystem and coastal economies.

10. Alternatives for Additional Forage Species

- The costly depletion of important forage fish species like river herring and shad demonstrates the need for the Mid-Atlantic Council to proactively address other forage fish that may not be currently targeted in commercial fisheries in federal waters or are not

[46] *See* Lenfest Forage Fish Task Force Report, pp. 5-13, *available at*: http://www.oceanconservationscience.org/foragefish/files/Little%20Fish,%20Big%20Impact.pdf

currently part of an FMP. Current policy for such forage species allows for unregulated and therefore unsustainable directed fishing.[47]

- Thus, Amendment 15 should include management alternatives to protect forage species other than those currently in the MSB FMP or proposed for inclusion in the FMP in the Amendment 15.
- The best available science strongly supports the precautionary approach outlined here for currently unfished forage species. In April 2012, the Lenfest Forage Fish Task Force, a group of 13 preeminent scientists from around the globe, released a report providing practical, science-based recommendations for the management of forage species, given their critical role in marine ecosystems and the need to transition toward an ecosystem-based approach to fisheries management. The Task Force recommends that no new forage fisheries be initiated until sufficient information is available to manage an ecologically sustainable fishery.[48] Information needed to manage a sustainable forage fishery includes population status and trends, environmental drivers, identification of dependent predators and their status, and foraging patterns.
- Specifically, the Council should develop and consider alternatives to add all other forage species commonly found in the waters under its jurisdiction that are not currently the subject of directed fisheries to the FMP as Ecosystem Component Species (ECS).
- Management measures for these species should prohibit new directed fisheries unless and until sufficient scientific knowledge is available to manage an ecologically sustainable fishery. The severe depletion of river herring and shad, i.e. severe depletion in part due in significant part to unregulated EEZ fishing shows how quickly forage species, with their unique vulnerabilities and poorly understood population dynamics, can be negatively affected by increased mortality in EEZ fisheries.
- Preliminary species that should be included under this alternative include, but are not limited to, the following: Atlantic Rainbow Smelt, American Sandlance, Northern Sandlance, Northern Krill, Bristled Longbeak, and Calanoid copepods. Precluding new fisheries on groups of animals based on higher taxonomic orders than Genus and species is a proven approach and may be appropriate. This list should be analyzed and additional forage fish could be added now, or through future actions.

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[47] However, according to the 1998 NOAA Fisheries Ecosystem Principles Advisory Panel (EPAP) report and others, "[t]he modus operandi for fisheries management should change from the traditional mode of restricting fishing activity only after it has demonstrated an unacceptable impact, to a future mode of only allowing fishing activity that can be reasonably expected to operate without unacceptable impacts." EPAP Report, Mangel, M. et al. 1996. Principles for the conservation of wild living resources. Ecological Applications 6(2):338-362.; *see also* Sissenwine, M. P. 1987. Councils, NMFS, and the Law. Pages 203-204 in: R. Stroud (ed.) *Recreational Fisheries* (11). Sport Fishing Institute. Washington, D. C.; Dayton, P. K. 1998. Reversals of the burden of proof in fisheries management. Science 279:821–822.

[48] Pikitch, E., et al. 2012. Little Fish, Big Impact: Managing a Crucial Link in Ocean Food Webs. Lenfest Ocean Program. Washington, DC. 108 pp.

Thank you for considering these comments.

Sincerely yours,
/s/ Roger Fleming
Roger Fleming, Attorney
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On behalf of the Herring Alliance

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Chesapeake Bay Foundation
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Choptank Riverkeeper
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Greenpeace
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Island Institute
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Jones River Watershed Association
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Lowell Parks & Conservation Trust
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Mystic River Watershed Association
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National Coalition for Marine Conservation
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Natural Resources Defense Council
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www.nrdc.org

Neponset River Watershed Association
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Neuse Riverkeeper Foundation
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New England Coastal Wildlife Alliance
Middleboro, Massachusetts
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North and South River Watershed Association
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NY/NJ Baykeeper
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Oceana
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Ocean River Institute
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Parker River Clean Water Association
Byfield, Massachusetts
www.businessevision.info/parker_river

Peconic Baykeeper
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PennEnvironment
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Pennsylvania Organization for Watersheds and Rivers
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South River Federation
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West and Rhode Riverkeeper
Shady Side, MD
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ECOSYSTEM-BASED FISHERY MANAGEMENT

A Report to Congress

by the

Ecosystem Principles Advisory Panel

As mandated by the Sustainable Fisheries Act amendments to the

Magnuson-Stevens Fishery Conservation and Management Act 1996

NATIONAL MARINE FISHERIES SERVICE
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PREFACE

Seeking solutions to reverse the decline of New England's fisheries in 1871, Congress created the U.S. Commission of Fish and Fisheries (Hobart 1995). The first appointed Commissioner, Spencer Baird, initiated marine ecological studies as one of his first priorities. According to Baird, our understanding of fish "... would not be complete without a thorough knowledge of their associates in the sea, especially of such as prey upon them or constitute their food...." He understood that the presence or absence of fish was related not only to removal by fishing, but also to the dynamics of physical and chemical oceanography.

Despite this historical, fundamental understanding of fisheries as part of ecosystems, we have continued to struggle to manage fish harvests while simultaneously sustaining the ecosystem. Recognizing the need for a more holistic management approach, Congress charged the National Marine Fisheries Service (a direct descendant of the U.S. Commission of Fish and Fisheries) with establishing an Ecosystem Principles Advisory Panel to assess the extent that ecosystem principles are used in fisheries management and research, and to recommend how such principles can be further implemented to improve our Nation's management of living marine resources. The resulting Panel was composed of members of industry, academia, conservation organizations and fishery management agencies. The Panel's diversity played a substantial role in the development of a pragmatic approach to expand ecosystem-based fishery management within the context of the existing fishery management system.

The Panel attempted to build on the progress of past efforts, namely the 1996 Sustainable Fisheries Act's (SFA) amendments to the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA) (NMFS 1996). The provisions of the SFA require the Regional Fishery Management Councils to set harvest rates at or below maximum sustained yield levels; develop rebuilding plans for those species that are currently below the long-term sustainable yield; better account for and minimize bycatch and discard of fish; identify essential fish habitat and take measures to protect it; and determine the effects of fishing on the environment. These actions are being implemented and are vital to achieving ecosystem-based management. Still, it will take years to decades before the results are fully realized.

The Panel forged a consensus on how to expand the use of ecosystem principles in fishery management. We do not have a magic formula, but we offer a practical combination of principles and actions that we believe will propel management onto ecologically sustainable pathways. By asking more encompassing questions about fisheries management such as, "What are the effects of fishing on other ecosystem components?" and "What are acceptable standards for fisheries removals from ecosystems?" we are broadening the scope of management and ultimately making fisheries sustainable.

Ecosystem-based fishery management is likely to contribute to increased abundance of those species that have been overfished. It may, however, require reduced harvest of species of critical importance to the ecosystem. We expect that ecosystem-based fishery management will contribute to the stability of employment and economic activity in the fishing industry and to the protection of marine biodiversity on which fisheries depend. As a society, we are recognizing the limits of the sea to provide resources and of our abilities to stay within those limits. What are acceptable levels of change in marine environments due to fishing? This Report does not answer that question for society, but it does set a framework for beginning to take actions based on the insight of Baird 125 years ago.

David Fluharty
Chair, Ecosystem Principles Advisory Panel
Seattle, Washington
November 15, 1998

EXECUTIVE SUMMARY

Ecosystem-based management can be an important complement to existing fisheries management approaches. When fishery managers understand the complex ecological and socioeconomic environments in which fish and fisheries exist, they may be able to anticipate the effects that fishery management will have on the ecosystem and the effects that ecosystem change will have on fisheries. However ecosystem-based management cannot resolve all of the underlying problems of the existing fisheries management regimes. Absent the political will to stop overfishing, protect habitat, and support expanded research and monitoring programs, an ecosystem-based approach cannot be effective.

A comprehensive ecosystem-based fisheries management approach would require managers to consider all interactions that a target fish stock has with predators, competitors, and prey species; the effects of weather and climate on fisheries biology and ecology; the complex interactions between fishes and their habitat; and the effects of fishing on fish stocks and their habitat. However, the approach need not be endlessly complicated. An initial step may require only that managers consider how the harvesting of one species might impact other species in the ecosystem. Fishery management decisions made at this level of understanding can prevent significant and potentially irreversible changes in marine ecosystems caused by fishing.

Recognizing the potential of an ecosystem-based management approach to improve fisheries management, Congress requested that the National Marine Fisheries Service (NMFS) convene a panel of experts to: 1) assess the extent to which ecosystem principles are currently applied in fisheries research and management; and 2) recommend how best to integrate ecosystem principles into future fisheries management and research. In response, NMFS created the National Marine Fisheries Service Ecosystem Principles Advisory Panel (Panel).

WHAT BASIC ECOSYSTEM PRINCIPLES, GOALS AND POLICIES CAN BE APPLIED TO FISHERIES MANAGEMENT AND RESEARCH?

To guide our deliberations, we developed a set of eight ecosystem operating principles (Principles) with societal goals for ecosystems (Goals), and a set of six management policies (Policies). These Principles, Goals and Policies were used to evaluate the current application of ecosystem-based fisheries management and to develop recommendations for further implementation of such approaches.

BASIC ECOSYSTEM PRINCIPLES, GOALS AND POLICIES

Based on the Panel's experience and review of the fisheries ecosystem literature, we suggest that the following Principles, Goals and Policies embody key elements for ecosystem-based management of fisheries.

Principles

- The ability to predict ecosystem behavior is limited.
- Ecosystems have real thresholds and limits which, when exceeded, can effect major system restructuring.
- Once thresholds and limits have been exceeded, changes can be irreversible.
- Diversity is important to ecosystem functioning.
- Multiple scales interact within and among ecosystems.
- Components of ecosystems are linked.
- Ecosystem boundaries are open.
- Ecosystems change with time.

Goals

- Maintain ecosystem health and sustainability.

Policies

- Change the burden of proof.
- Apply the precautionary approach.
- Purchase "insurance" against unforeseen, adverse ecosystem impacts.
- Learn from management experiences.
- Make local incentives compatible with global goals.
- Promote participation, fairness and equity in policy and management.

TO WHAT EXTENT ARE ECOSYSTEM PRINCIPLES, GOALS AND POLICIES CURRENTLY APPLIED IN RESEARCH AND MANAGEMENT?

The Panel considered a management system based on the ecosystem Principles, Goals and Policies, as a framework with which to evaluate the current application in U.S. marine fisheries management and research. This model was then compared to the current state of research and management.

We conclude that NMFS and the Regional Fishery Management Councils (Councils) already consider and apply some of the Principles, Goals and Policies outlined above, but they are not applied comprehensively or evenly across Council jurisdictions, NMFS Regions, or ecosystems. The fact that the Principles are not applied consistently in U.S. fisheries management and research should not be interpreted as reluctance or intransigence on the part of these entities to adopt ecosystem approaches. Rather, these agencies lack both a clear mandate and resources from Congress to carry out this more comprehensive, but ultimately more sustainable approach. Furthermore, the ecosystem-based management of fisheries is a relatively new concept and there are considerable gaps in knowledge and practice.

HOW CAN WE EXPAND THE APPLICATION OF ECOSYSTEM PRINCIPLES, GOALS AND POLICIES TO FISHERIES RESEARCH AND MANAGEMENT?

Several practical measures can be implemented immediately to make U.S. fisheries management and research more consistent with the ecosystem Principles (see **Summary of Recommendations**). These measures comprise an incremental strategy for moving toward ecosystem-based fisheries research and management.

Councils should continue to use existing Fishery Management Plans (FMP) for single species or species complexes, but these should be amended to incorporate ecosystem approaches consistent with an overall Fisheries Ecosystem Plan (FEP). The FEP, to be developed for each major ecosystem under Council jurisdiction, is a mechanism for incorporating the Principles, Goals and Policies into the present regulatory structure. The objectives of FEPs are to:

- Provide Council members with a clear description and understanding of the fundamental physical, biological, and human/institutional context of ecosystems within which fisheries are managed;
- Direct how that information should be used in the context of FMPs; and
- Set policies by which management options would be developed and implemented.

Fisheries management based on the ecosystem Principles, Goals and Policies must be supported by comprehensive research. Significant ecosystem research is now conducted by the National Oceanic and Atmospheric Administration (NOAA) and other agencies, as well as the academic community. This research is critical and must continue, but must expand into several key areas. First, we must better understand the long-term dynamics of marine ecosystems and how they respond to human-induced change, particularly changes brought about by fishing. Second, we must develop governance systems which have ecosystem health and sustainability, rather than short-term economic gain, as their primary goals.

THE FUTURE OF ECOSYSTEM APPROACHES IN U.S. FISHERIES MANAGEMENT

Fisheries scientists and managers are beginning to grasp the potential of ecosystem-based fishery management to improve the sustainability of fisheries resources. Given the depressed state of many U.S. fisheries, this awareness must be expanded and actions taken to implement this approach. Our management recommendations and research actions provide a pragmatic framework within which to apply the ecosystem Principles, Goals and Policies. The success of this approach depends on full implementation of measures already underway as a result of the passage of the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA) (NMFS 1996), particularly the essential fish habitat (EFH) requirements and strengthened national standards. The recommendations contained in this report provide the required next steps.

While some of the recommended actions can start

immediately, we believe that legislation is required to implement measures like the FEP. Given that legislative processes may require three to five years to enact the proposed regulations, we recommend interim actions by the Secretary of Commerce to develop demonstration FEPs and to encourage voluntary adoption by management Councils of the Principles, Goals and Policies proposed herein. We also are aware that these new tasks will require additional human and financial resources for full implementation.

The benefits of adopting ecosystem-based fishery management and research are more sustainable fisheries and marine ecosystems, as well as more economically-healthy coastal communities. We have identified the actions required to realize these benefits. We urge the Secretary and Congress to make those resources available.

SUMMARY OF RECOMMENDATIONS

Fisheries management and policy recommendations are directed toward Congress for implementation by NMFS and the Councils. Interim measures and research recommendations are directed toward the Secretary of Commerce for implementation by NMFS and other appropriate agencies.

Develop a Fisheries Ecosystem Plan (FEP)

Require each Council to develop an FEP for the ecosystem(s) under its jurisdiction. The FEP is an umbrella document containing information on the structure and function of the ecosystem in which fishing activities occur, so that managers can be aware of the effects their decisions have on the ecosystem, and the effects other components of the ecosystem may have on fisheries.

Each FEP should require the Councils to take, at least, the following eight actions:

1. **Delineate the geographic extent of the ecosystem(s) that occur(s) within Council authority, including characterization of the biological, chemical and physical dynamics of those ecosystems, and "zone" the area for alternative uses.**

The first step in using an ecosystem approach to management must be to identify and bound the ecosystem. Hydrography, bathymetry, productivity and trophic structure must be considered; as well as how climate influences the physical, chemical and biological oceanography of the ecosystem; and how, in turn, the food web structure and dynamics are affected. Transfers across ecosystem boundaries should be noted.

Within each identified ecosystem, Councils should use a zone-based management approach to designate geographic areas for prescribed uses. Such zones could include marine protected areas, areas particularly sensitive to gear impacts and areas where fishing is known to negatively affect the trophic food web.

2. **Develop a conceptual model of the food web.**

For each targeted species, there should be a corresponding description of both predator and prey species at each life history stage over time. FEPs can then address the anticipated effects of the allowed harvest on predator-prey dynamics.

3. **Describe the habitat needs of different life history stages for all plants and animals that represent the "significant food web" and how they are considered in conservation and management measures.**

Essential fish habitat (EFH) for target and non-target species at different life stages should be identified and described. Using habitat and other ecosystem information, Councils should develop zone-based management regimes, whereby geographic areas within an ecosystem would be reserved for prescribed uses. FEPs should identify existing and potential gear alternatives that would alleviate gear-induced damage to EFH, as well as restrict gears which have adverse affects. Further, FEPs should evaluate the use of harvest refugia as a management tool to satisfy habitat needs.

4. **Calculate total removals—including incidental mortality—and show how they relate to standing biomass, production, optimum yields, natural mortality and trophic structure.**

Total removals (i.e., reported landings, unreported landings, discards, and mortality to fish that come into contact with fishing gear but are not captured) should be incorporated into qualitative

food web and quantitative stock assessment models. These models will allow managers to reduce uncertainty, monitor ecosystem health and better predict relative abundance of species affected by the harvest of target species.

5. Assess how uncertainty is characterized and what kind of buffers against uncertainty are included in conservation and management actions.

Given the variability associated with ecosystems, managers should be cognizant of the high likelihood for unanticipated outcomes. Management should acknowledge and account for this uncertainty by developing risk-averse management strategies that are flexible and adaptive.

6. Develop indices of ecosystem health as targets for management.

Ecosystem health refers to a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organization that has evolved naturally. Provided that a healthy state can be determined or inferred, management should strive to generate and maintain such a state in a given ecosystem. Inherent in this management strategy would be specific goals for the ecosystem, including a description of "unhealthy" states to be avoided.

7. Describe available long-term monitoring data and how they are used.

Changes to the ecosystem cannot be determined without long-term monitoring of biological indices and climate. Long-term monitoring of chemical, physical and biological characteristics will provide a better understanding of oceanic variability and how climate changes affect the abundance of commercially important species and their corresponding food webs.

8. Assess the ecological, human, and institutional elements of the ecosystem which most significantly affect fisheries, and are outside Council/Department of Commerce (DOC) authority. Included should be a strategy to address those influences in order to achieve both FMP and FEP objectives.

Councils and DOC have authority over a limited range of the human, institutional and natural components of a marine ecosystem. It is important to recognize those components of the ecosystem over which fisheries managers have no direct control, and to develop strategies to address them in concert with appropriate international, Federal, State, Tribes and local entities.

Measures to Implement FEPs

The following are general recommendations to ensure effective development and implementation of FEPs:

1. Encourage the Councils to apply ecosystem Principles, Goals and Policies to ongoing activities.

In preparation for FEP implementation, Councils should begin to apply the ecosystem Principles, Goals and Policies to the conservation and management measures of existing and future FMPs. Three actions are particularly important; specifically, each FMP's conservation and management measures should:

- Consider predator-prey interactions affected by fishing allowed under the FMP.
- Consider bycatch taken during allowed fishing operations and the impacts such removals have on the affected species and the ecosystem as a whole, in terms of food web interactions and community structure.
- Minimize impacts of fisheries operations on EFH identified within the FEP.

2. Provide training to Council members and staff.

To facilitate an ecosystem approach and to aid the development and implementation of FEPs, NMFS should provide all Council members with basic instruction in ecological principles. Further, training materials should be made available to the fishing industry, environmental organizations and other interested parties.

3. Prepare guidelines for FEPs.

The Secretary of Commerce should charge NMFS and the Councils with establishing guidelines

for FEP development, including an amendment process. NMFS and the Councils should conduct a deliberative process—similar to the process of developing National Standards Guidelines—to ensure that FEPs are realistic and adaptive.

4. Develop demonstration FEPs.

While encouraging all Councils to develop framework FEPs, the Secretary of Commerce should designate a Council or Councils to develop a demonstration FEP, as a model to facilitate rapid implementation of the full FEP when required in MSFCMA reauthorization.

5. Provide oversight to ensure development of and compliance with FEPs.

To ensure compliance with the development of FEPs, the Secretary of Commerce should establish a review panel for FEP implementation oversight. Implicit in this action is the establishment of a timetable for development of a draft FEP, its review by the panel, and any necessary revisions before the draft FEP becomes a basis for policy.

6. Enact legislation requiring FEPs.

To provide NMFS and the Councils with the mandated responsibility of designing and implementing FEPs, Congress should require full FEP implementation in the next reauthorization of the MSFCMA.

Research Required to Support Management

Require, and provide support for NMFS and other appropriate agencies to initiate or continue research on three critical research themes which will provide the information necessary to support ecosystem-based fisheries management. These themes are:

1. Determine the ecosystem effects of fishing.

Fishing affects target species, non-target species, habitat and potentially marine ecosystems as a whole. A directed program must be initiated to determine all effects of fishing on marine ecosystems.

2. Monitor trends and dynamics in marine ecosystems (ECOWATCH).

In order to detect, understand and react appropriately to ecosystem changes, a broad-scale ecosystem research and monitoring program must be undertaken based on the best available technology. We refer to this program as "ECOWATCH" because it will enable scientists and managers to observe ecosystem changes in a comprehensive manner.

3. Explore ecosystem-based approaches to governance.

Many of today's fisheries problems stem from governance systems which create incentives that are incompatible with, or inimical to, ecosystem-level Goals (e.g., health and sustainability). Alternate governance systems must be identified which provide fishermen and others with incentives to consider the health and sustainability of the ecosystem as primary goals.

ECOSYSTEM-BASED FISHERY MANAGEMENT

SECTION ONE: INTRODUCTION

The National Marine Fisheries Service (NMFS) was charged by Congress to establish an Ecosystem Principles Advisory Panel (Panel) to identify ecosystem principles, evaluate how those principles are currently used in fishery management and research, and then to recommend measures that would expand their use in fishery management and research. Our Charter (Appendix A) describes the rationale for our effort and provides the charge to this Panel. Here we outline our views of the historical developments and current issues leading to this charge. We lay out a conceptual framework that includes management actions and research on marine resources and fisheries in an ecosystem context.

THE PROBLEM

The world's oceans are at or near maximum sustainable fishery yields. The number of overexploited stocks increased by 2.5 times between 1980 and 1990 (Alverson and Larkin 1994). Much of the global sustained yield is being accomplished by increased fishing for species at progressively lower trophic levels (Pauly et al. 1998). The prospect of increasing total sustained yield is unlikely (Pauly and Christensen 1995). Although fisheries provide direct or indirect employment to about 200 million people (Garcia and Newton 1997), overfishing is the most commonly observed result of fishery development. The consequences of overharvesting are expressed in social, economic, cultural and ecological changes. The ecological consequences of overfishing often are undocumented and may be poorly known or overlooked.

Since 1990, annual harvests by U.S. fleets have been slightly in excess of 4.5 million metric tons, with nearly half of that coming from two fisheries—menhaden and Alaska pollock. In its annual report to Congress on the status of the fisheries of the U. S., NMFS states that of the 727 managed stocks in the United States, 86 are overfished, 10 are approaching overfished status, and 183 are not overfished (NMFS 1997). This leaves 448 stocks, for which the status is virtually unknown. NMFS (1997) also indicates that "additional stocks will likely be identified as overfished" under the new definition of overfishing in the Magnuson-Stevens Fisheries Conservation and Management Act (MSFCMA).

While there are some encouraging recoveries (e.g., striped bass in the Atlantic and Pacific sardine), record-setting yields (e.g., Alaska salmon), and management successes (e.g., Pacific halibut), those cases are the exceptions rather than the rule. As in the global case, we should be concerned that overfishing will be a common consequence for most fisheries (Ludwig et al. 1993, Mooney 1998), although this need not be the case (Rosenberg et al. 1993).

This issue is urgent because the current harvest levels are high and because new fisheries will rise, be fully capitalized and reach unsustainable levels of catch levels before the management process can establish effective constraints. That, unfortunately, is the too-common lesson of history (Ludwig et al. 1993). In many cases, the ecological correlates of changing fish populations could have served as evidence of intensified exploitation effects. Frequently, the advent of a fishery and implementation of catch restrictions have unknown ecological consequences. Too often, we learn about ecological consequences after the fact, because we do not consider them in our decision-making, nor do we monitor ecosystem changes due to increased exploitation. Those lessons are not unique to fisheries. Many Federal, regional and State resource management agencies are now moving toward or considering an ecosystem approach in their attempt to provide a holistic framework for resource management. Fisheries must do so as well (Langton and Haedrich 1997).

FISHERIES IN AN ECOSYSTEM CONTEXT

Much of the foundation of fisheries science provides a basis for determining maximum yields so that fishing can safely remove surplus production (Hilborn and Walters 1992). However, when fishing is examined in an ecosystem context, the rationale for harvesting surplus production is unclear. Marine ecosystems are effective at capturing energy, cycling nutrients and producing biomass. Very little, if any

of this biomass, is truly "surplus" to an ecosystem; before the advent of fisheries, it was recycled within the ecosystem. Consequently, our societal decision to harvest fish, induces ecological changes among competitors, prey and predators as the system responds to fishing and the trophically-induced changes fishing causes in ecosystems. These changes affect future levels of surplus production of the harvested population, including the possibility that there may be none.

We understand that fisheries must continue, because they provide food and desirable social and economic benefits and because the cultural traditions of fishing are highly valued. However, we also understand that overutilized fisheries are a serious threat to those traditions and benefits (National Research Council 1999). Conflict thus develops when management agencies (e.g., NMFS, Regional Fishery Management Councils, etc.) seek to implement sustainable yield policies for open-access resources, when fishery effects extend to animals protected by our Endangered Species Act or Marine Mammal Protection Act, and, most recently, when conservation and management interests assert that the burden of proof should be placed on the fishing industry (i.e., to demonstrate that exploitation does not produce large-scale and long-term ecological changes) (Dayton 1998). Finding the balance between competing interests is a difficult challenge, and each fishery will have its unique solutions. On the Federal level, NMFS will be expected to provide the ecological insights that are essential for long-term protection of fish stocks and their ecosystems.

Nature has limits

If nature is a shifting mosaic or in essentially continuous flux, then it may be wrong to conclude that whatever societies choose to do in or to the natural world is fine. The question can be stated as, "If the state of nature is flux, then is any human-generated change okay?" ... The answer to this question is a resounding "No!" ... Human-generated changes must be constrained because nature has functional, historical, and evolutionary limits. Nature has a range of ways to be, but there is a limit to those ways, and therefore, human changes must be within those limits. (Pickett et al. 1992).

Decisions regarding fishing practices derive from our social, economic, political and cultural context, and only secondarily from the ecological context that supports fisheries (Mooney 1998). A holistic view requires that we recognize fishery management and exploitation as a real and integral part of the marine ecosystem (Langton and Haedrich 1997). Because fishing actively removes a percentage of one or several species, it can affect the predators and prey of those species, their physical habitat, and it can change the growth and mortality rates of target and non-target species alike. In short, fishing can and is likely to alter the structure and function of marine ecosystems (Dayton 1998, Pauly et al. 1998). Humans are at the top of the global marine food chain. We thus have the obligation and opportunity to make choices to affect the marine environment positively.

While fishing has a long history, it is a relatively new force in the scales of evolutionary time. Fishing is typically a species-selective and size-selective agent of mortality and, therefore, is unlike the natural causes of mortality. Most of the fish removed by fishing activities are in the middle or near the tops of food webs in their habitats. Fishing can be viewed as a keystone predator; the ecological effects of fishing are therefore substantially greater and more complex than simply the biomass removed. Thus, we should expect that substantial changes have or could occur in those ecosystems due to fishing. We have witnessed changes in the landscape around us with the advent of technology evolved from the axe and the plow. We should expect equally profound ecological changes from modern, large-scale uses of the hook and net.

MANAGING FISHERIES IN AN ECOSYSTEM CONTEXT

Ecosystem-based fisheries management does not require that we understand all things about all components of the ecosystem. We know that the traditional single-species approach of fisheries management is tractable, but we also know that it may not be sufficient. We know that an ecosystem perspective is desirable, but it is complex and unpredictable. There simply is not enough money, time or talent to develop a synthetic and completely

informed view of how fisheries operate in an ecosystem context. There will always be unmeasured entities, random effects, and substantial uncertainties, but these are not acceptable excuses to delay implementing an ecosystem-based management strategy.

Each fishery and each ecosystem is unique and yet, in all cases, we are confronted with four fundamental problems:

- We do not have a complete understanding of the ecological system that produces and supports fishes.
- We cannot forecast weather or climate and their effects on ecosystems.
- Systems evolve over time and knowing how the system works does not necessarily mean that an ecosystem would respond predictably to future changes in weather, climate or fisheries.
- Our institutions are not configured to manage at the ecosystem scale. Fish and the fisheries that pursue them are not easily aligned with our political and jurisdictional boundaries.

These constraints are not unique to fisheries, they confront all attempts to manage natural resources in an ecosystem context. We know that the removal of one species can and does affect others, but rarely have we developed management plans that adequately account for those direct and indirect effects. We know that ecosystems have a limited carrying capacity that results in bounds on fish yields. We know that habitat loss contributes to declines in species abundance, but too often we only regulate catch, gear or effort for one target species as a way to compensate for habitat loss and its effects on other species. We know that major, unexpected events (e.g., El Niño) can alter ecosystem processes, thus affecting species targeted by fisheries, but we have no method for integrating these events into our assessments of target species population trends (Mantua et al. 1997, Francis et al. 1998).

What are the potential gains of implementing an ecosystem approach to management, and how do we develop a holistic view that is both sufficient and tractable? In this report, we develop a strategy for implementing ecosystem-based management.

First, we develop a conceptual model that sets fisheries in the context of what we know about ecosystem theory (which is provided in the section on **Ecosystem Principles, Goals and Policies**). Second, we provide a brief assessment of the extent to which ecosystem principles, goals and policies are applied in U.S. fisheries research and management (**Current Applications of the Principles, Goals and Policies**). Third, we offer a series of specific recommendations for applying these principles to the operational context of NMFS, the Regional Fishery Management Councils (Councils), their administrative structure and their management activities (**Recommendations for Implementing the Ecosystem Principles, Goals and Policies in U.S. Fisheries Conservation, Management and Research**). Finally, we recommend a comprehensive research program to provide the ecological and governance underpinnings for ecosystem-based fishery management.

Taken as a whole, the report presents our best

Legal Authorities for Ecosystem Management of Fisheries

The Magnuson-Stevens Fishery Conservation and Management Act allows fishery managers to consider ecosystems in setting management objectives. National Standard 1 requires conservation and management measures to "prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery" (Sec. 301(a)(1)). The "optimum" yield is defined as providing "the greatest overall benefit to the Nation, particularly with respect to food production and recreational opportunities, and taking into account the protection of marine ecosystems" (Sec. 3(28)(A)). Moreover, the optimum yield is prescribed as "the maximum sustainable yield from each fishery, as reduced by any relevant economic, social or ecological factor" (Sec. 3(28)(B)). In addition, the Act states as one of its purposes "to promote the protection of essential fish habitat" (Sec. 2(b)(7)). To the extent that ecosystems are not being adequately considered in FMPs, it is not because of a lack of statutory authority so much as it is a lack of direction about what information is required and how it should be put into operation.

advice about innovative approaches that can help set fisheries in an ecosystem context. Ecosystem-based management is an important new challenge. We expect that NMFS and Council managers and scientists will develop creative ways to help meet that challenge. But these new approaches cannot substitute for compliance with existing mandates. Ecosystem-based management will require re-evaluation of the institutional structure necessary for effective management. It will also demand a strong political will expressed through Congress, NMFS and the Councils—one based on a broader appreciation of the ecosystem context within which we prosecute our fisheries (Hutchings et al. 1997).

SECTION TWO: ECOSYSTEM PRINCIPLES, GOALS AND POLICIES

There are two requirements for managing human interactions with marine ecosystems. One is to develop an understanding of the basic characteristics and principles of these ecosystems—what patterns they exhibit and how they function in space and time. The second is to develop an ability to manage activities that impact marine ecosystems, consistent with both their basic principles and with societal goals concerning the kinds of behavior we would like ecosystems to exhibit (i.e., health and sustainability).

This section lists eight basic ecosystem principles (Principles) and their parallels in human systems that are part of marine ecosystems. A discussion of societal goals (Goals) for ecosystem-based management follows. Finally, a list of general management policies (Policies) to achieve the Goals is provided.

BASIC ECOSYSTEM PRINCIPLES

Marine ecosystems are complex, adaptive systems composed of interconnected groups of living organisms and their habitats. Living organisms are constantly adapting and evolving to their environment (both to the physical environment, which varies on multiple scales, and to other living organisms with which they co-exist); this evolution leads to complex, sometimes chaotic dynamics.

Marine ecosystems are generally extensive and open. Their fluid environments are subject to variability in both local and remote inputs of energy (a consequence of physics operating on many spatial and temporal scales) which may dominate such systems. Highly variable and chaotic dynamics of living resources are often observed as well.

Today, humans are a major component in most ecosystems. The human component of the ecosystem includes the humans themselves, their artifacts and manufactured goods (economies), and their institutions and cultures. The human imposition of fishing mortality, at rates often higher than natural mortality, can have major impacts not only on targeted species but on the ecosystem itself.

The following eight Principles have analogs in both the human and nonhuman aspect of ecosystems:

1. **The ability to predict ecosystem behavior is limited.**

   Uncertainty and indeterminacy are fundamental characteristics of the dynamics of complex adaptive systems. Predicting the behaviors of these systems cannot be done with absolute certainty, regardless of the amount of scientific effort invested. We can, however, learn the boundaries of expected behavior and improve our understanding of the underlying dynamics. Thus, while ecosystems are neither totally predictable nor totally unpredictable, they can be managed within the limits of their predictability.

Properties characterizing marine ecosystems may vary within wide bounds on decadal and longer time scales (Fig.1). For example, El Niño events and decadal climate changes may displace species, restructure communities and alter overall productivity in broad oceanic areas. Other phenomena, sometimes operating on smaller time scales, may precipitate regime shifts characterized by major fluctuations in constituent species (Steele 1996), but our ability to predict such events is only now evolving (Langton et al. 1996) and will always be shrouded in a degree of uncertainty. Nevertheless, management policies can be guided by the broad understanding we possess of marine ecosystem boundaries and production potential limits.

The ability to predict human behavior in fishery systems is also limited, but evolving. Many fishermen pass through rounds of fishing in regular annual patterns, markets respond in predictable ways to price changes, and fishermen often have predictable responses to policy proposals or regulatory changes. Fisheries systems respond to

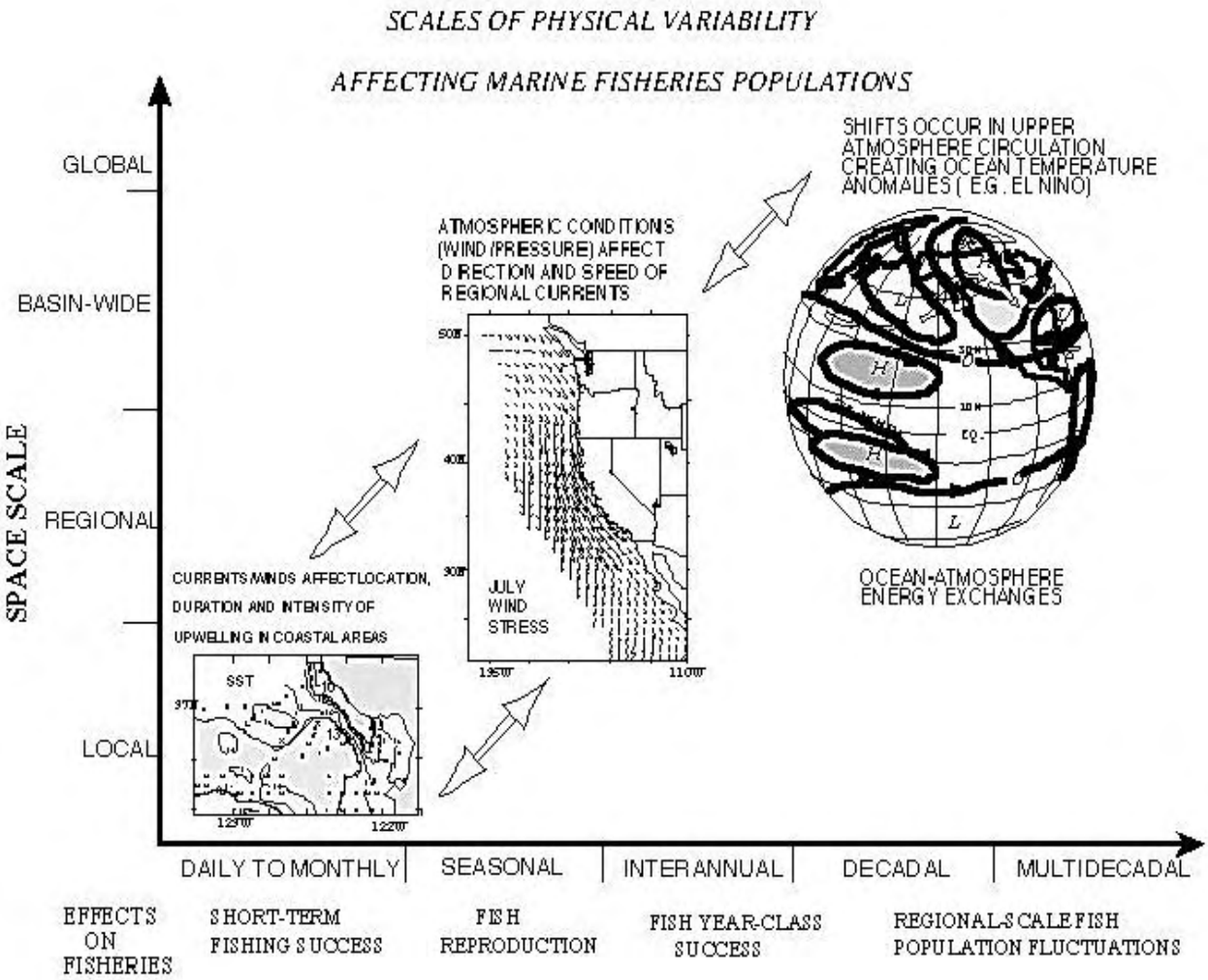

Figure 1. Scales of physical variability affecting marine resources. Variability in marine ecosystems is linked to variability in the physical environment on a continuum of time and space scales. We are often constrained to work on scales at which data are available, and long term monitoring must be carefully designed to address appropriate scales. Figure courtesy of NMFS Pacific Fisheries Environmental Laboratory.

global market trends and economic changes, social preferences and philosophies. The ability to describe, explain and predict these human behaviors, although the behaviors vary according to circumstance, is increasing with the growing body of social scientific data and information on fishery systems.

2. **Ecosystems have real thresholds and limits which, when exceeded, can effect major system restructuring (Holling and Meffe 1996).**

Ecosystems are finite and exhaustible, but they usually have a high buffering capacity and are fairly resilient to stress. Often, as stress is applied to an ecosystem, its structure and behavior may at first not change noticeably. Only after a critical threshold is passed does the system begin to deteriorate rapidly. Because there is little initial change in behavior with increasing stress, these thresholds are very difficult to predict. The nonlinear dynamics which cause this kind of behavior are a basic characteristic of ecosystems.

The concepts of limits and thresholds have been misused in single-species fishery management in the sense that they have been viewed as targets for fish catches rather than levels to be avoided. Because single-species management has prevailed, limits and thresholds rarely have been applied in a broader ecosystem context. Limits in fisheries management often have been biological reference points such as prescribed fishing mortality rates or yields, that are set without concern for other components in the ecosystem. Many limits are in fact thresholds that, when exceeded, challenge the resilience of the managed stock and associated species. Experience has shown that some past target levels used by managers, for example maximum sustainable yield, because they are too close to critical thresholds (Caddy and Mahon 1995), ultimately lead to stock declines or damage to ecological communities. Thresholds are to be avoided to maintain resilience at the species and community levels. Fishery targets should be set conservatively, well below the limits and critical thresholds that compromise the productive potential and stability of the ecosystem. Limits and thresholds of non-targeted organisms have only recently been considered through mandates of the Marine Mammal Protection Act, the Endangered Species Act, and in the new overfishing level definitions, bycatch and essential fish habitat (EFH) provisions of the MSFCMA.

Human systems (fishermen, their communities and fishery management systems) are both resilient and generally resistant to change. Thresholds of profitability, tolerance of regulatory conditions, and risk or uncertainty-induced stress on fishery-dependent human communities are real. Thresholds must be determined through both constituent advice and independent research on individual and group responses to stress. Identification of reference points for the limits of human resilience may be possible.

3. Once thresholds and limits have been exceeded, changes can be irreversible.

When an ecosystem is radically altered, it may never return to its original condition, even after the stress is removed. This phenomenon is common in many complex, adaptive systems.

It is probable that some estuaries, coral reefs (Hughes 1994), and mangrove ecosystems have been irreversibly altered by fishing, aquaculture, and other habitat-destructive activities. Farther offshore, effects of fishing itself on abundances of target and non-target organisms may radically alter communities and ecosystems. It is too soon to know whether heavily fished systems, such as Georges Bank, will return to their previous states when fishing effort is relaxed (Fogarty and Murawski 1998). Fisheries scientists and managers have demonstrated an abiding faith in the ability of fish stocks to compensate for fishing effects by increasing their level of productivity. Implicitly, that faith is extended to ecosystems which support exploited stocks. Up to a point, recoveries are possible. In some coastal ecosystems, however, resilience and limits have been exceeded, often by the combined effects of habitat destruction and fishing, and it is doubtful if they will return to their original condition.

Changes in ecosystems may permanently alter human behaviors. When a fisherman goes out of business, when an annual season of fishing is disturbed, or when market flow is interrupted, it is often not possible to reestablish the former business, pattern or market. Some aspects of human systems and behavior can be reestablished given enough time and attention, whereas changes in natural components of ecosystems are typically more enduring. In contrast, policy and management systems are continually subject to change and reversal.

4. Diversity is important to ecosystem functioning.

The diversity of components at the individual, species, and landscapes scales strongly affects ecosystem behavior. Although the overall productivity of ecosystems may not change significantly when particular species are added or removed, their stability and resilience may be affected.

Long-term consequences of diversity losses due to overfishing or poor fishing practices in marine systems are largely unknown. It is clear, however, that the economic value of specific components of catch change dramatically as some stocks are overfished, to be replaced in the ecosystem by lower-valued species (Deimling and Liss 1994, Fogarty and Murawski 1998). At the ecosystem level, drastic alterations of diversity certainly have occurred, and biological productivity has been redirected to alternative species, but it is not clear that these

ecosystems are less productive or less efficient. However, such ecosystems are often valued less; witness the loss of tourist revenue in areas that have suffered damage to coral reef systems. It is prudent to presume that changes in biodiversity will decrease resiliency of species, communities and ecosystems, especially with perturbations that occur over long time scales (Boehlert 1996).

This principle also applies to the human element. An economy with more than one sector, a community with more than one industry, a fishing family with more than one income from different sources, or an industry large enough to foster technological innovation, are all aspects of the strength in diversity found in human society. Communities which lose such diversity are more susceptible to stress and unexpected sources of change.

5. Multiple scales interact within and among ecosystems.

Ecosystems cannot be understood from the perspective of a single time, space, or complexity scale. At minimum, both the next larger scale and the next lower scale of interest must be considered when effects of perturbations are analyzed.

Consequences of perturbations at one scale in marine systems may be magnified at larger and smaller scales (Langton et al. 1995). For example, destruction of a species' spawning habitat—typically a small fraction of its range—may translate into major impacts on species associations and trophic interactions in the broader feeding areas of recruited fish. Likewise, effects of fishing on a broad ecosystem scale may have profound impacts on components of ecosystems far removed in space and time—scientists are investigating the relationship between pollock fishing and the general decline of Steller sea lion populations in the eastern Bering Sea and Gulf of Alaska. Seemingly small human perturbations, applied at a point in time or in one part of a marine ecosystem, may have unforeseen impacts because of the open nature and fluid environment that characterize marine ecosystems. These features elevate the probability that a stress applied at one scale will be transmitted and may have unforeseen effects at other scales in the ecosystem.

Human impacts on ecosystems cannot be understood from the perspective of a single time, space, or complexity scale. A fishing community is subject to perturbations both from its own members and from outside forces. Fishery systems in one location are subject to environmental, social, economic and regulatory forces far removed in time and space, especially with respect to markets.

6. Components of ecosystems are linked.

The components within ecosystems are linked by flows of material, energy, and information in complex patterns.

Critical linkages in marine ecosystems are sustained by key predator-prey relationships. Large, long-lived predators and small, short-lived prey (e.g., forage fishes) both contribute in major ways to marine fish catches. Heavy fishing may precipitate species replacements, both at lower trophic levels (e.g., sand lance replacing herring and vice-versa) and at upper trophic levels (e.g., sharks and rays replacing Atlantic cod) (Fogarty and Murawski 1998). Loss from ecosystems of large and long-lived predators is of particular concern because they potentially exercise top-down control of processes at lower trophic levels. Global data sets have indicated that the mean trophic level of fish caught declined significantly from 1950-1994 (Pauly et al. 1998). Fishing down food webs (i.e., fishing at lower trophic levels) disrupts natural predator-prey relationships and may lead first to increasing catches, but then to stagnating or declining yields.

Disruption of ecosystem linkages clearly may have resounding impacts on human economies and, in the worst cases, ecosystem stability and productivity are compromised. Components of human systems are linked by flows of material, energy and information. The collapse of a market may drastically change fishing behavior. A technological innovation or entry of a new segment of a fishing fleet may cause far-reaching changes in dependent human communities.

7. Ecosystem boundaries are open.

Ecosystems are far from equilibrium and cannot be adequately understood without knowledge of their boundary conditions, energy flows, and internal cycling of nutrients and other materials. Environmental variability can alter spatial boundaries and energy

inputs to ecosystems.

Productive potential of marine ecosystems is especially sensitive to environmental variability over a spectrum of temporal and spatial scales. The unbounded structure of marine communities provides the backdrop for the high (relative to terrestrial) variability that is observed (Steele 1991). Boundaries of ecosystems, or productive regions, shift with weather and longer-term climate change. Species abundances and distributions vary in accord with annual to decadal shifts in ocean features (e.g., Pearcy and Schoener 1987, Polovina et al. 1995, Roemmich and McGowan 1995, Francis et al. 1998, McGowan et al. 1998). In open systems, local heavy fishing in combination with major changes in ocean conditions (e.g., El Niño), can lead to fishery collapses and associated shifts in the partitioning of energy or biomass among trophic levels (e.g., Walsh 1981, Barber and Chavez 1983).

Human behavioral systems are also subject to variability over a spectrum of temporal and spatial scales, and cannot be understood without knowledge of their boundary conditions. Certain components of human systems (people) are closely related and interact regularly over time; others are only sporadically in contact and interact in cyclical or irregular patterns. The more intermittent or sporadic the contact or interaction, the less stable the human system (Axelrod 1984).

8. Ecosystems change with time.

Ecosystems change with time in response to natural and anthropogenic influences. Different components of ecosystems change at different rates and can influence the overall structure of the ecosystem itself and affect the services provided to society in the form of fish catch, income and employment.

Marine ecosystems experience directional changes. Shifts in climate are responsible for many such changes, but the role of biological interactions in the absence of human influence are largely unknown. Dramatic changes in coastal and estuarine ecosystems, attributable to long-term geological and erosional processes are easily observed (e.g., Chesapeake Bay, see Mountford 1996). Anthropogenic changes are all too common, especially in neritic and estuarine ecosystems, or enclosed seas (e.g., San Francisco Bay (Nichols et al. 1986), Great Lakes, Black Sea, Aral Sea, Chesapeake Bay). Species introductions, excess nutrient loading, damming of tributaries, poor stewardship of bordering forests, bad agricultural practices, and poorly-managed fisheries are examples of factors that cause change. Rapid advances in fishing technologies (e.g., vessel power, navigation, sensing-locating and harvest efficiency), the propensity for fisheries to selectively remove species, failure to control bycatch, and unintended damage to the physical structure of ecosystems, have changed the character of heavily fished ecosystems (e.g., Georges Bank) (Fogarty and Murawski 1998). Selective fishing, that often targets long-lived predators, can have cascading effects on community structure (Marten 1979, Laws 1977), while heavy industrial fishing on forage species may have unintended impacts on top predators, especially those (e.g., marine mammals) unable to adapt quickly to changes in the forage base. Removal of large whales through past whaling practices, likewise, may have lingering effects on the nature of ecosystem structures today (National Research Council 1996). Deterioration of coastal ecosystems may also generate active attempts at remediation or enhancement through aquaculture and other means (Morikawa 1994), which can also generate pollution and wastes (Wu 1995).

Human activities dependent on ecosystems may change in response to environmental change and changes induced by fishing and other activities. In the short run, these impacts may be considered the normal consequences of a highly variable activity. However, humans adapt to long-term changes in composition of fisheries by stopping fishing or shifting effort to other species; changes which may produce adverse impacts. In addition, changes in perception, values, preferences, patterns of use, and accumulation of knowledge or expertise may cause changes over time in the ways humans interact within ecosystems. Human components of ecosystems (especially technology and institutions) can change rapidly in ways that outstrip the capacity for change of other ecosystem components. Communities may continue to grow and consumption rates increase, for example, yet the capacity of the seas to increase yields of living marine resources is limited. Thus, fishery management policies must be prepared to take into account these factors.

BROADENING SOCIETAL GOALS FOR ECOSYSTEMS

Traditionally, societal goals have emphasized benefits to humans resulting from extractive uses of ecosystem components. For example, fishery management has typically had revenues, employment, recreational fishing opportunities, and/or maintenance of traditional lifestyles as explicit or implicit goals. From an ecosystem perspective, these goals need to be broadened to include concepts of health and sustainability (Lubchenco et al. 1991, National Research Council 1999). Ecosystem health is the capability of an ecosystem to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organization comparable to that of the natural habitat of the region (Sparks 1995). This concept is also referred to as biotic integrity, which is defined as a system's wholeness, including the presence of all appropriate elements and occurrence of all processes at appropriate rates (Angermeier and Karr 1994, Angermeier 1997). While the concept of health applied to marine ecosystems is relatively new and untested, it has become a guiding framework in several areas, including forest ecosystems (Kolb et al. 1994), agroecosystems (Gallopin 1995), desert ecosystems (Whitford 1995) and others (Rapport et al. 1995).

A healthy ecosystem provides certain ecosystem goods and services, such as food, fiber, the capacity for assimilating and recycling wastes, potable water, clean air, etc. (International Society for Ecosystem Health, 1998). How do we extract from, and otherwise utilize ecosystems, while maintaining their health and the array of non-use services that they also provide (Costanza et al. 1997) into the indefinite future?

The challenge to scientists and managers is to develop useful, quantitative measures of ecosystem health which can guide management. What level of fishing, for example, can a "healthy" ecosystem sustain? How can vigor and resilience be expressed quantitatively so that managers can maintain them within healthy limits? These are difficult questions which will not be answered in their entirety in the foreseeable future, but incremental implementation of ecosystem-based fisheries management will begin to identify ecosystem variables (or indicators) that are unacceptable. These could be used to guide management away from unhealthy ecosystem states.

GENERAL ECOSYSTEM-BASED MANAGEMENT POLICIES

Ecosystem Principles to achieve societal Goals must be implemented through ecosystem-based management Policies. There are three overriding aspects of the Principles that are taken into account in the six Policies discussed below. These are the exhaustibility of ecosystems (reflected in Principles 2 and 3), uncertainty about ecosystems (reflected in Principles 1, 2, 4, and 8), and the role of humans within ecosystems (reflected in all of the Principles). The exhaustibility of the ecosystem requires a policy to change the burden of proof (Policy 1). Both the exhaustibility of ecosystems and uncertainty about ecosystems require policies to manage by a precautionary approach (Policy 2) and to "purchase insurance" (Policy 3) against adverse ecosystem impacts. Uncertainty about ecosystems also dictates that there is learning from management experiences (Policy 4). The role of humans within ecosystems requires policies to make incentives for human behavior consistent with societal goals for ecosystems (Policy 5). Acceptance and effective implementation of the policies and management is served by promoting participation, fairness and equity (Policy 6). Each of the Policies is discussed below.

1. Change the burden of proof.

We live in a world where humans are an important component of almost all ecosystems. Thus, it is reasonable to assume that human activities will impact ecosystems. The *modus operandi* for fisheries management should change from the traditional mode of restricting fishing activity only after it has demonstrated an unacceptable impact, to a future mode of only allowing fishing activity that can be reasonably expected to operate without unacceptable impacts.

To date, almost any type of fishing activity has been allowed until problems arise and regulations are established to solve them. Decision makers have to be convinced that management restrictions are needed. As W. F. Thompson (1919) wrote ". . . proof that seeks to change the way of commerce and sport must be overwhelming." Several authors have argued that a change is needed in this "burden of proof" (Sissenwine 1987, Mangel et al. 1996, Dayton 1998).

The key elements of the change are: 1) that future fishing activity should be allowed, if and only if it is explicitly provided for by fishing regulations which take into account risk and uncertainty and are promulgated to protect all elements of the ecosystem, and 2) that to a substantial degree the responsibility for providing the information and other support (e.g., the cost of management) necessary to manage fisheries in a sustainable manner, lies with participants in the fishery.

The first part of the change is analogous to changing the "null" hypothesis from "marine fisheries are inexhaustible" (Huxley 1883), to today's reality that marine fisheries will usually evolve to a state of overfishing unless they are carefully managed (Garcia and Newton 1997). The second element of the change makes clear that the direct beneficiaries from fishing should accept a greater share of the burden (i.e., costs) of fishery management. The standard of proof associated with the change (i.e., how much certainty is needed before a fishing activity is allowed) should be commensurate with the severity of the risk of a mistake. Applying the proper standard of proof is implicitly an element of the precautionary approach (see Policy 2).

In practice, changing the burden of proof will mean that, when the effects of fishing on either the target fish population, associated species, or the ecosystem are poorly known (relative to the severity of the potential outcome), fishery managers should not expand existing fisheries by increasing allowable catch levels or permitting the introduction of new effort and should not promote or develop new fisheries for so-called "underutilized species."

2. Apply the precautionary approach.

The precautionary approach is a key element of the United Nations Agreement for Straddling Stocks and Highly Migratory Species (United Nations 1996) and the Food and Agriculture Organization of the United Nations (FAO) Code of Conduct for Responsible Fisheries (FAO 1995). The U.S. is a signatory of both.

All ecosystems are complex and uncertainty is unavoidable. Within uncertainty, there is always a risk of undesirable consequences on fishery resources (e.g., overfishing) and/or on ecosystems. The precautionary approach was motivated by the widely accepted conclusion of scientists and fishery managers that many of the current problems of fisheries (i.e., a large number of overfished stocks) have been caused by the practice of making risk-prone fishery management decisions (i.e., to err toward overfishing) in the face of uncertainty (Garcia and Newton 1994). One approach to coping with uncertainty, which is widely applied to other human endeavors, is to encourage behaviors (often by enacting regulations) that reduce risk. Thus, the precautionary approach calls for risk averse decisions (i.e., to err toward conservation). FAO (1995) provides guidelines on the application of the precautionary approach.

3. Purchase "insurance" against unforeseen, adverse ecosystem impacts.

Even under the precautionary approach, there is a risk of unforeseen, adverse impacts on ecosystems. Insurance can be used to mitigate these impacts if and when they occur.

Insurance is a common method for guarding against the risks of unforeseen, adverse impacts of many human endeavors, and it has been proposed to guard against adverse ecosystem impacts (Costanza and Cornwell 1992). A requirement to purchase insurance provides an incentive to avoid risk-prone behavior (to reduce the cost of insurance). Thus, this management policy supports the precautionary approach.

Insurance can take many forms in addition to the traditional form of insurance policies or environmental bonds. Marine protected areas, for example, are a form of insurance. Protecting parts of the ecosystem from exploitation can insure future productivity and sustainability (Carr and Reed 1993, Dugan and Davis 1993, Agardy 1994, Bohnsack and Ault 1996, Roberts 1997, Lauck et al. 1998). Reserves also serve as baseline areas to evaluate natural variation in animal and plant populations that are free from fishing impacts.

Another form of insurance is a system to detect adverse impacts at an early stage so that actions can be taken to prevent further damage and/or to repair damage. This form of insurance is more effective if corrective actions have already been planned and adopted, such that there is minimal delay when a

problem is detected.

Environmental bonding, marine protected areas and a system to detect and respond to adverse impacts can serve as both insurance and elements of a precautionary approach.

4. Learn from management experiences.

Management actions and policies can be considered as experiments and should be based upon hypotheses about the ecosystem response. This requires close monitoring of results to determine to what extent the hypotheses are supported.

Sustainable management of complex, adaptive ecosystems must itself be adaptive (Holling 1978). Management policies are experiments from which we can learn and improve, rather than absolute "solutions." Adaptive management in an "active" context would demand that hypotheses be put forward for testing and that alternative models be considered. Active, adaptive management often presumes that changes in fishing mortality rates will be imposed purposefully to induce a response in the fished stock or in the ecosystem under investigation (Walters 1986, Hilborn and Walters 1992). This "active" experimental approach to management is scientifically sound, but may have limited applicability in extensive marine ecosystems, at least within the time scales in which managers must act and in which fisheries operate. Walters (1997), while arguing eloquently about potential advantages of active adaptive management, recognizes the many arguments that detract from its adoption. For instance, modeling exercises and experiments required for the implementation of adaptive management have often been seen as excessively expensive or ecologically risky. A less aggressive form of the adaptive approach, however, is more generally acceptable and applicable. In this form, managers learn from actions to the greatest extent possible and respond expeditiously with alternative management actions. The willingness and institutional capability to respond are critical for this form of management to succeed.

5. Make local incentives compatible with global goals.

Changing human behavior is most easily accomplished by changing the local incentives to be consistent with broader social goals. The lack of consistency between local incentives and global goals is the root cause of many "social traps," including those in fisheries management (Costanza 1987). Changing incentives is complex and must be accomplished in culturally appropriate ways.

Global goals, such as long-term sustainability of a fish population or ecosystem health, are generally beyond the control of people at a local scale. Their incentive for conservation is diminished if they have no assurance that others will conserve or if they will not share in future benefits from conservation. This phenomenon is illustrated by the well known "race for the fish" which can lead to overfishing and wasteful overcapitalization (Graham 1935, Gordon 1954, Sissenwine and Rosenberg 1993).

A key element of making local incentives consistent with global goals is to allocate shares of the fishery such that people at local scales (down to the scale of individuals) have the incentive to use their shares efficiently (i.e., not wasting resources by racing for a share) and to conserve the entire resource to enhance the value of their shares in the future. Shares can take many forms such as a fraction of the total allowable catch (known as an individual quota), units of fishing effort, or exclusive rights to fish specific areas. Share-based allocation schemes might be broadened to take account of indirect impacts on ecosystems. There are several options for the local scale to which shares are allocated, such as to individuals or to communities. The most effective configuration of a share-based allocation scheme depends on the specific fishery and ecosystem that is being managed, but some form of share-based allocation will usually be necessary to fulfill this management policy.

6. Promote participation, fairness and equity in policy and management.

Ecosystem approaches to management rely on the participation, understanding and support of multiple constituencies. Policies that are developed and implemented with the full participation and consideration of all stakeholders, including the interests of future generations, are more likely to be fair and equitable, and to be perceived as such.

The level and quality of stakeholder participation

in fishery management varies widely, as does the definition of "stakeholder." Participation varies from passive consultation to shared decision making authority (Sen and Nielsen 1996). Systems organized to promote the maximum involvement of stakeholders, including the interests of future generations, and to emphasize the maximum appropriate delegation of responsibility and authority to the lowest possible levels of the management system (e.g., the local or regional level), tend to have the highest credibility among fishery constituents (Pinkerton 1989). This often leads to such effects as better data sharing and lower enforcement costs.

SECTION THREE: CURRENT APPLICATION OF THE ECOSYSTEM PRINCIPLES, GOALS AND POLICIES

We reviewed how the Councils and NMFS currently apply the ecosystem Principles, Goals, and Policies in order to help shape strategies for greater application in the future. We could not undertake a comprehensive fishery-by-fishery assessment of the application of the ecosystem Principles in current research and management activities. Such a task was beyond our scope given the limited time and resources available, and was certain to be incomplete. In addition, we saw little to be gained by evaluating the past performance of agencies relative to a set of ecosystem Principles, Goals, and Policies that were not known to the organizations whose performance might be judged. Most importantly, the 1996 amendments to the MSFCMA substantially changed the guidelines for certain management actions so that past practices are no longer relevant.

Information for the assessment was solicited from a number of sources, including NMFS Regional Offices and Fishery Science Centers. NMFS was asked to consult with Councils and other appropriate organizations to prepare this information. At our first meeting, representatives from each NMFS Fishery Science Center briefed us on the application of general ecosystem principles. Relying on that input and on our own knowledge and experience we then prepared regional overviews which served as the basis for this assessment.

To organize the assessment, we posed a series of questions that reflect the application of the Principles. These questions and our answers to each are given below.

Q: Have science-based ecosystem boundaries been identified, and are they used to specify resource management units?

A: Marine ecosystem boundaries are generally open, but bathymetric and other oceanographic features create biological discontinuities or shape gradients that allow marine ecosystems to be defined. On a regional scale, the Council jurisdictions reasonably correspond to such bathymetric and oceanographic features. Within these jurisdictions, management unit boundaries generally parallel the scientific information about the distribution of exploited fish stocks. Because fish distributions are also affected by the topographic and oceanographic features that are important to other biological components of ecosystems, it is often the case that management units corresponding to stock distributions also correspond to ecosystem boundaries. For example, this occurs with cod in the Gulf of Maine ecosystem, which are managed as a single stock by the New England Fishery Management Council. There are many situations where this is not the case, and many cases where the scientific basis for defining stock boundaries is minimal. Exchange rates across boundaries are seldom known or explicitly considered in management. This is particularly true for highly migratory species such as tunas, swordfish and billfishes. Exchange rates are important within ecosystems for some forms of management, such as area closures (including marine protected areas) that are used to conserve exploited stocks of fish, or more broadly, to conserve marine ecosystems.

The issue of ecosystem boundaries also has connections with human institutions. In some cases, the jurisdiction of management institutions does not match ecosystem boundaries or stock boundaries of some resources. This has led to various arrangements for interjurisdictional management of fisheries, such as international commissions, interstate fishery management commissions, and joint Fishery Management Plans (FMP) of two or more Councils. While some useful steps have been taken to deal with interjurisdictional issues, little consideration has been given to mobility of the fishing industry (both recreational and commercial) between jurisdictions, or to the diversity of people within the jurisdictions.

Another factor related to the definition of ecosystem boundaries is the impact that nonfishing sectors of society have on marine ecosystems. Management of coastal resources, agriculture and forestry, in addition to fisheries, is also required to effectively apply the ecosystem Principles, Goals and

Policies. If it is impractical to include these activities within ecosystem boundaries, exchanges across boundaries caused by these activities must be considered. In addition, institutional arrangements are needed to address cross-sectorial effects on ecosystems. Generally, such arrangements are lacking, although the recent MSFCMA amendment that calls for the identification of EFH should be an impetus for making such arrangements.

We conclude that ecosystem boundaries are generally defined and are reflected in management, but these definitions will have to be amended in order to integrate our recommendations for an ecosystem approach to management.

Q: Is scientific uncertainty in stock assessments and knowledge about marine ecosystems described to managers, and is this uncertainty considered in FMPs (such as by including buffers)?

A: Many sources of uncertainty affect stock assessments: 1) imperfections in catch statistics (sometimes from misreporting), 2) imprecise estimates of biological parameters, 3) variability in fishery independent resource surveys, and 4) natural variability in biological processes, particularly in recruitment. All these sources of uncertainty should be considered when determining the variance associated with estimates of current and future stock size. But, the uncertainty in stock assessment estimates is not always characterized, and even when it is, the true uncertainty is probably greater since it is difficult to account for all sources of uncertainty. Nevertheless, managers are usually made aware of at least some degree of uncertainty; their reaction to uncertainty varies among regions. For example, the North Pacific Fishery Management Council is noted for generally acting conservatively in the face of uncertainty (i.e., applying the precautionary approach), whereas some other Councils have consistently done the opposite (i.e., making risk-prone decisions) in the past. Recent changes in the MSFCMA and international agreements requiring the application of the precautionary approach should encourage risk-averse decisions by all Councils in the future.

Stock assessment uncertainty is only one of several areas of imprecision that should concern fishery managers. Uncertainty about fishery effects on ecosystems is high and generally is not characterized. There are some cases where fishery managers have attempted to account for ecological relationships in spite of uncertainty, such as prohibiting pollock trawling within 10-20 miles of islands that are occupied by endangered Steller sea lions, to minimize the risk that near-shore fishing will deplete their prey, however, these cases are rare.

Scientific uncertainty in stock assessments and ecosystems is an inherent reflection of highly complex systems that extend over vast areas and depths. We conclude that uncertainty is characterized to some degree. In the future, fishery managers need to consistently apply the precautionary approach in the face of uncertainty.

Q: Is there routine monitoring of ecosystems and are the results used to support management?

A: The fish component of marine ecosystems is monitored routinely for many stocks and in most U.S. regions. Standardized trawl surveys of the northeastern U.S., initiated in 1963 and now conducted three times per year, are the most extensive example of monitoring of the fish component, yet, some fish stocks are virtually unsampled by the current survey program. In other regions, fish stocks are only surveyed every third year. In addition, fishery-dependent monitoring is conducted.

Monitoring of fish is far more extensive than is the monitoring of other marine ecosystem components. Some systems such as San Francisco Bay, Chesapeake Bay and the Northeast U.S. have long-standing ecosystem monitoring programs which measure ecosystem components other than fish, but the use of such programs is not widespread for ecosystems and fisheries under the jurisdiction of NMFS and the Councils.

Other ecosystem components that might be monitored are human demographics, marine mammals, birds, benthos, zooplankton, phytoplankton, and physical and chemical factors. While there is a significant amount of human census data and other information about people, changes in the demographics and cultural aspects of participants in fisheries are not routinely monitored, nor are there studies of economics. As a result of the Marine Mammal Protection Act, many populations of marine mammals are monitored, although this monitoring is limited in extent. Coastal sea birds are monitored

in some regions. There are long-term time-series of plankton data, such as California Cooperative Oceanic Fisheries Investigations data off of California, and Marine Resources Monitoring Assessment and Prediction and Continuous Plankton Recorder data in New England waters. With advances in satellite remote sensing, it is now possible to monitor primary production and some physical variables synoptically over vast regions. There has been very little monitoring of benthos, except for a few sites and generally for only a few years. Lack of time-series data on the benthos is an impediment to understanding the effects of mobile fishing gear on benthic habitats.

Monitoring data are used in a variety of ways in the management process. Fish monitoring results constitute a critical input to stock assessments, which are used to support fisheries management. Limited socioeconomic data are used for various impact analyses that accompany fishery management decisions. Information on other ecosystem components is sometimes considered to help explain variability in fishery resources, but such relationships are usually uncertain or speculative and therefore are seldom used by managers.

Q: Have the food webs of target species been identified and is this information used in FMPs?

A: There are extensive databases on the stomach content of fishes in some regions, such as the Northeast and Alaska where hundreds of thousands of fish of many species have been sampled over several decades. Some multispecies predator/prey models have been developed, but generally these models are better at explaining the effects that trophic relationships might have had, rather than predicting future patterns and variations.

To date, use of food web information in fisheries management has been limited. This reflects the limited predictive power of existing multispecies predator/prey models. Knowledge of food webs is considered qualitatively in some management decisions, such as the Pacific Fishery Management Council's FMP for anchovies which sets aside some of the population as forage.

Q: Are total removals, including discards, taken into account in stock assessments and management?

A: Total removals are made up of the reported landings, unreported landings, discards, and mortality to fish that come in contact with fishing gear but are not captured. Stock assessments are routinely based on reported landings and discard estimates, if available. Discard estimates are derived from fishing vessel logbook reports and/or from at-sea observers on fishing vessels. Larger groundfish vessels operating in the northeast Pacific are required to have 100% observer coverage, and this improves the quality of discard data for these fisheries. Observers in the Gulf of Mexico shrimp fishery estimate that discards of finfish are over four times larger than the catch of shrimp. For at least one important Gulf species, red snapper, discards are the largest component of mortality. But there are many species where there are virtually no discard data (although discarding exists). Estimates of unreported landings and/or mortality of fish that come in contact with fishing gear, but are not captured, are very rare. Stock assessments are robust to under estimates of total removals so long as the proportion not included in removal estimates is constant, which is a reasonable assumption under some circumstances.

There are alternative ways for fisheries management to account for total removals. When discards are estimated, they are usually included in the stock assessments which support fisheries management. For example, discards of juvenile swordfish are factored into the swordfish stock assessments conducted by the member countries of the International Commission for the Conservation of Atlantic Tunas. The discards may be taken into account by reducing the allowable catch based on the expected level of discards, or by counting estimates of discards against the allowable catch. Alternatively, management might use measures that are less dependent on knowing total removals, such as gear restrictions, effort controls or area closures.

We conclude that total removals are probably underestimated, and significantly so in some cases. Therefore, more effort is needed to estimate total removals and to apply management strategies that are robust in the face of uncertainty about total removals.

Q: Have the effects of fishing on the ecosystem been studied?

A: This is a relatively new research endeavor. There

is clear evidence that fishing alters species composition (e.g., fishing on Georges Bank appears to have shifted the community from predominately Atlantic cod to sharks and skates (Fogarty and Murawski 1998)). Pauly et al. (1998) recently showed that there has been a significant worldwide reduction in mean trophic level of species fished. Several studies that have demonstrated that mobile fishing gear alters benthic habitat (Auster and Langton 1999), but little is known about the implications of these changes. Further, there has been even less research conducted on other fishing gears.

Q: Are the habitat needs of different life history stages of target and nontarget species known and are they considered in FMPs?

A: The habitats that are used by some or all of the life-history stages of many species of fish are known. But habitat utilization does not mean that the habitat is obligatory (i.e., the species must have that habitat to successfully complete its life-cycle). The mechanistic relationship between a fish species at a particular life history stage, and the type of habitat it occupies, is unknown for most species and life-history stages. It is most critical to understand the essential habitat needs of fish near shore, where anthropogenic effects on habitat are likely to be most significant.

The relationships between fish and habitat are summarized as a basis of EFH determinations to be included in FMP amendments, as required by the MSFCMA. These amendments require that the habitat needs of fish populations be given serious consideration in the future when government agencies make decisions that are likely to adversely affect EFH. Fishing itself is an activity that has the potential to affect EFH. Taking account of these potential effects is a major challenge facing Councils.

SECTION FOUR: RECOMMENDATIONS FOR IMPLEMENTING THE ECOSYSTEM PRINCIPLES, GOALS AND POLICIES IN U.S. FISHERIES CONSERVATION, MANAGEMENT AND RESEARCH

In this section, we describe approaches for incorporating the Principles, Goals and Policies established in Section II into the fisheries management and research processes of the current Council system. We strongly believe that the key to an effective ecosystem approach is to fish more conservatively. The depressed condition of many U.S. stocks is related primarily to unsustainable levels of fishing effort, rather than ecosystem effects. With few exceptions, scientists understand the levels of fishing effort required to produce sustainable yields, but fishery managers are challenged by a highly politicized process to exceed those levels for short-term gains. Setting maximum sustainable yield and optimum yield conservatively, and respecting these conservative goals in the face of political and economic pressure is essential in any ecosystem approach.

Many current U.S. fishery management problems such as overfishing, bycatch and protection of EFH are addressed in the Sustainable Fisheries Act (SFA) of 1996. Each of these SFA provisions is an important step toward the use of ecosystem principles in fishery management. However, these measures do not add up to an ecosystem approach.

FMPs for single species or species complexes should continue to be the basic tool of fisheries management for the foreseeable future. However, managements actions under FMPs alone are not sufficient to implement an ecosystem approach. A mechanism is required to integrate FMPs and include the ecosystem Principles, Goals, and Policies in a way that will be meaningful. That mechanism is the Fisheries Ecosystem Plan (FEP).

THE FISHERIES ECOSYSTEM PLAN (FEP)

Our primary recommendation is that each Council (including NMFS in the case of Atlantic highly migratory species) develop the FEP as a mechanism for incorporating ecosystem Principles, Goals and Policies into the present fisheries management structure. The objectives of FEPs are to:

- Provide Council members with a clear description and understanding of the fundamental physical, biological, and human/institutional context of ecosystems within which fisheries are managed;
- Direct how that information should be used in the context of FMPs; and
- Set policies by which management options would be developed and implemented.

Councils would develop FEPs for each major ecosystem under their jurisdiction. For example, the North Pacific Fishery Management Council might have two FEPs—one for the Bering Sea/Aleutian Islands and one for the Gulf of Alaska. Councils with overlapping ecosystems, or with significant species migration across ecosystem boundaries would work together on a joint FEP. In the event of transnational ecosystems, appropriate international arrangements would be sought to implement an ecosystem approach.

The FEP should be used as a metric against which all fishery-specific FMPs are measured to determine whether or not management effectively incorporates the ecosystem Principles, Goals and Policies. The FEP should also contain regulations or management measures which extend across individual FMPs. The FEP should serve as a nexus for existing FMPs and provide a context for considering Council management actions with respect to all living marine resources, whether managed or not.

FEPs must contain the information about ecosystem that allows managers to make informed decisions, but the primary purpose of the plans is to prescribe how fisheries will be managed from an ecosystem perspective. Careful consideration must be given to the structure and required content of an FEP to balance the needs for plans to be both substantive and realistic. It is appropriate that NMFS lead a deliberative and inclusive (of a broad range of interests and expertise) process to prepare guidelines for FEPs (analogous to the processes that have been used to prepare guidelines for implementing National Standards). Preparation of such specific guidelines was beyond the scope of our Panel Charter, but we did identify Council actions that must be taken when guidelines are prepared, to be consistent with the Panel's recommendations:

1. Delineate the geographic extent of the ecosystem(s) that occur(s) within Council authority, including characterization of the biological, chemical and physical dynamics of those ecosystems, and "zone" the area for alternative uses.

2. Develop a conceptual model of the food web.

3. Describe the habitat needs of different life history stages for all plants and animals that represent the "significant food web" and how they are considered in conservation and management measures.

4. Calculate total removals—including incidental mortality—and show how they relate to standing biomass, production, optimum yields, natural mortality and trophic structure.

5. Assess how uncertainty is characterized and what kind of buffers against uncertainty are included in conservation and management actions.

6. Develop indices of ecosystem health as targets for management.

7. Describe available long-term monitoring data and how they are used.

8. Assess the ecological, human, and institutional elements of the ecosystem which most significantly affect fisheries, and are outside Council/Department of Commerce (DOC) authority. Included should be a strategy to address those influences in order to achieve both FMP and FEP objectives.

The eight FEP actions are elaborated below:

1. Delineate the geographic extent of the ecosystem(s) that occur(s) within Council authority, including characterization of the biological, chemical, and physical dynamics of those ecosystems, and "zone" the area for alternative uses.

The ecosystems supporting fisheries in the United States vary markedly (Apollonio 1994), and the way in which fisheries are managed within them will vary according to their individual characteristics. Managers must be able to geographically delineate the systems under their authority, and have a scientific understanding of the structure, function, and processes that occur within their respective ecosystems, and between their systems and others. This delineation should include both ecological and human/institutional components and their interactions. This includes the extent of our knowledge of climate, how climate affects the physical and biological oceanography of the system, and how, in turn, these affect food web structure and dynamics.

Councils should use information from FEPs to develop zone-based management regimes. In a zoning approach, geographic areas within an ecosystem would be reserved for prescribed uses. For example, use of gears which are demonstrated to have an adverse effect on EFH could be limited to prescribed areas. Currently, FMPs are required to describe and mitigate gear effects on EFH, but FEPs should go further, not only identifying where habitat impacts occur, but also identifying specific zones where certain gears should be restricted. A zone-based approach could also limit fishing activities in areas where potential negative trophic impacts could occur. The North Pacific Fisheries Management Council's establishment of no-trawl zones in red king crab habitat is an example of such a measure. Zoning can also be used to limit bycatch, by restricting fishing activities in areas where high levels of bycatch are likely to occur.

A zoning approach should also include the establishment of marine protected areas. A species-specific approach to habitat protection, as currently practiced, may result in many small protected areas

with occasionally conflicting regulations that are difficult to understand and often difficult to enforce. Complete protection of relatively large portions of marine ecosystems, in the form of harvest refugia, may provide the best way to characterize habitat needs and also serve as management tools (Bohnsack and Ault 1996, Roberts 1997). Each FEP should consider and evaluate the potential benefits of harvest refugia and support research to evaluate their use.

Marine Protected Areas

Marine Protected Areas (MPAs) offer promise as a means to implement the precautionary approach and mitigate the effects of fishing in an ecosystem (Yoklavich 1998). However, the utility of the approach depends on the way MPAs are defined and established. The concept of MPAs represents a continuum, from marine wilderness areas to areas in which only a few specific activities might be restricted. We use the term to mean the entire spectrum of usage, and suggest that managers carefully define their conservation and management objectives before determining the characteristics of a given MPA.

MPAs should be representative of the larger ecosystem and, as such, would serve as experimental sites for investigating processes and mechanisms that would be operable throughout the region. MPAs must be established with the understanding that ecosystems change over time and that research results have to be evaluated relative to this natural variability as distinct from variability resulting from human exploitation of a resource. MPAs represent a form of insurance against excessive exploitation. Although we aspire to a level of understanding that would allow for strategic management of our nation's fisheries, uncertainty and indeterminacy are fundamental ecosystem characteristics. Hence, research is needed on the optimal size of MPAs, sources and sinks for new recruits, and the social and management issues required for successful implementation.

2. Develop a conceptual model of the food web.

Fisheries managers cannot control the weather or long-term physical changes in the ecosystems that produce the managed resources. They can, however, control what species are fished and the total numbers and individual sizes of resources removed. Thus, managers should have a conceptual understanding of the food web, and should use that information in making decisions about harvest. For each species for which there is an FMP, there should be a description of both the prey species and the predators at each stage in the life cycle. Where information on certain species is not available for all life stages, managers should refer to species inhabiting similar ecological niches or their functional equivalents as the basis for defining trophic links. Following this, the FEP should contain an analysis of the anticipated impacts of the allowed harvest on predator-prey dynamics, even if data gaps force such a statement to be largely qualitative.

Ecosystem Modeling

Modeling is an essential scientific tool in developing ecosystem approaches for fishery management. Simple descriptions of prey and predator species and models of how they interrelate are good starting points but they are inadequate. What is required is a food-web based mathematical model. Such a model could examine factors that affect primary productivity and how changes in it affect the relationships that exist among all components of the ecosystem. Such a model could assist in assessing the trade-offs among harvests of fish species in different parts of the food web, how abundance of marine mammals relates to populations of its prey species, and how much of the total primary production is required to sustain ecosystem harvest. Recent models such as ECOPATH (Polovina 1984, Christensen and Pauly 1995, Pauly and Christensen 1995) have been applied and have provided insight into some fundamental ecosystem questions. ECOPATH provides a framework for summarizing natural rates of growth and consumption of marine populations. This allows small-scale studies or models (such as fish bioenergetics models or diet composition data) to be viewed in a common currency, in the context of the ecosystem as a whole.

Presently, dynamic mathematical models (e.g., ECOSIM (Walters et al. 1997)) are being developed but they have been applied only experimentally in actual fishery management situations. Using them as active parts of the FEP could facilitate model development and testing. Most importantly, models have the potential to provide managers with information about how ecosystems are likely to respond to changes in fishery management practices (Botsford et al. 1997). Like FEPs, these models will be unique to each system and its important attributes.

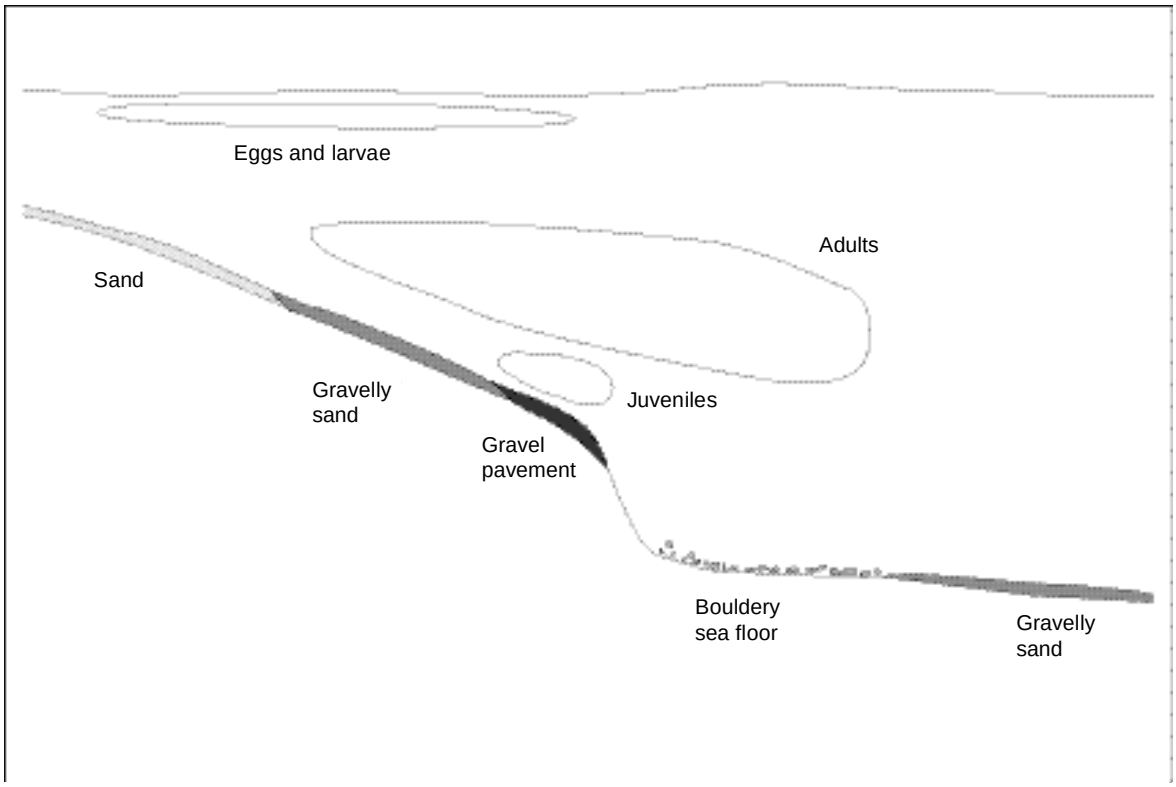

Figure 2. Life history stages of Atlantic cod versus habitat requirements as characterized for Georges Bank in the Northwest Atlantic (artwork by Dave Stanton, adapted from Lough 1989).

3. Describe the habitat needs of different life history stages for all plants and animals that represent the "significant food web" and how they are considered in conservation and management measures.

Marine organisms generally have different dietary and habitat requirements for each life cycle stage (e.g., Atlantic cod on Georges Bank; Fig. 2). Traditional management practices often limit fishing effort in an attempt to protect spawning stock while ignoring management strategies that would prevent negative effects on survivorship at each life cycle stage. In an effort to address this issue, FMPs are now required to include a description of EFH. This is probably best considered in a multiple-species context, including overlapping habitats of suites of species with similar life cycles that occupy similar habitats as well as their prey. Thus, each Council should include EFH considerations within the FEP, using the ecosystem approach to describe such habitat based on the EFH descriptions from existing FMPs.

4. Calculate total removals—including incidental mortality—and show how they relate to standing biomass, production, optimum yield, natural mortality and trophic structure.

Ecosystem overfishing occurs when fishing directly or indirectly results in a reduction of ecosystem health. Direct impacts on target species include changes in the total population status, age structure, and sex ratio within the population. Indirect impacts can occur on component species or on ecosystem health. Pauly et al. (1998) describe trophic effects of fishing which yield apparently nonlinear, unanticipated results with potential negative effects on sustainability. Thus, a measure of total removals of a target species should include fish landed and fish caught and released (with some determination of mortality rates of released fish), predation at each life history stage, and loss through incidental capture.

Mortality associated with bycatch can produce significant biological losses and ecological shifts in community structure within ecosystems (Alverson et al. 1994). To address bycatch issues, FEPs should: 1) identify potential shifts in community structure and their consequences, and indicate how they should be mitigated; 2) identify bycatch associated with particular gear types, not just by providing a list of species, but also by identifying how bycatch in a given species changes both spatially and temporally; and 3) identify existing or potential alternative gear types which would reduce bycatch.

5. Assess how uncertainty is characterized and what kind of buffers against uncertainty are included in conservation and management actions.

The more complex an ecosystem, the greater the unpredictability. The ultimate uncertainty and risk is associated with those practices that affect ecosystem equilibrium, such as significant changes in climate or hydrology that have potentially significant global effects. Therefore management actions that aim for specific outcomes should be accompanied by the anticipated probabilities associated with achieving those outcomes. Given the variability associated with ecosystem states and the general low precision, high variance, and unknown potential for bias in fisheries data—and thus in the models used to predict outcomes—managers must recognize the high likelihood for unanticipated results. Hence, decision-makers should account for this uncertainty with the development of flexible, adaptive, and risk-averse management strategies.

FEP should identify those factors or issues which are likely to bear the greatest degree of uncertainty within that ecosystem. Stock assessment reports, prepared for each new or continuing FMP, should characterize uncertainty and indicate how that uncertainty is incorporated into the assessment. The characterization of uncertainty in stock assessments is an example of how the policy of the precautionary approach should be incorporated into the FEP, and one of the best example of insurance against unknowable ecosystem dynamics.

Although uncertainty may render management strategies that are effective in one system ineffective in another, the application of the precautionary approach is a policy which can be implemented in any ecosystem. Because each ecosystem will have different levels of uncertainty and risk associated with it, managers must develop specific risk criteria for application of the precautionary approach within each system.

6. Develop indices of ecosystem health as targets for management.

The use of a goal such as ecosystem health to guide fishery management forces resource scientists and managers to define desired ecosystem states, typically based on historical information reflecting ecosystem structure and yield. Once this has been accomplished, management strategies can be developed to generate and maintain these healthy states. Defining a healthy ecosystem is problematic in practice, so we also recommend that managers identify "unhealthy" ecosystem states which should be avoided. For example, FEP goals could be to prevent the extinction of any ecosystem component, to maintain a specific, high mean trophic level in the ecosystem, or to maintain benthic biomass within the range of natural variability. Each Council should be charged to develop its own FEP goals and metrics based on unique ecosystem characteristics.

7. Describe available long-term monitoring data and how they are used.

Although most physical and biological databases represent relatively short periods of time and therefore do not characterize long-term variability, the amount and quality of physical data available relevant to fisheries have improved markedly in recent years (Boehlert and Schumacher 1997). These data are essential for the development of models to predict changes in oceanographic conditions. Biological baseline data often are difficult to evaluate, given the current impacts of fisheries on marine ecosystems and the largely unpredictable outcomes of these impacts. However, reasonable estimates of preexploitation conditions can be made in some cases (Pauly 1995).

Each FEP should include a prioritized long-term monitoring plan, designed to allow the assessment of the changing states of ecosystem health relative to established baseline conditions. This will be facilitated through the implementation of the research recommendations. As discussed by Christensen et al. (1996), monitoring programs should include ways to determine whether

Aquaculture and Stock Enhancement: Are Cautions being Heeded?

With declining fish stocks, there is growing pressure to artificially boost harvests, either through aquaculture in coastal waters or through stock enhancement. The potential benefits of aquaculture include: increased production of cultured fish which can contribute to food and economic security without placing additional pressure on wild stocks. In addition, stock enhancement may help rebuild or sustain depleted wild stocks.

However, many existing aquaculture programs have developed without attention to their impacts on marine ecosystems (Naylor et al. 1998). Salmon culture and ocean ranching provide good examples. Hatcheries have led to manifold problems, including interbreeding between native and non-native stocks (Lannan et al. 1989), decreases in genetic biodiversity (Ryman et al. 1995), introduced species problems, and threats to carrying capacity, even in the open ocean (Ogura and Ito 1994). Early calls to genetically "upgrade the wild stocks" (Moav et al. 1978) to improve production have given way to attention to the "usually negative" genetic impacts of aquaculture (Beveridge et al. 1994). Wilcove et al. (1992) captured this sentiment, stating "Introduced genes can be as harmful as introduced species, especially when hatchery-bred fish compete with wild populations."

Dramatic examples of human manipulation of coastal ecosystems are provided in Japan, where coastal fisheries have been maintained at a near constant level by increasing mariculture production and stock enhancement while natural production has declined (Morikawa 1994). Aside from potential genetic effects as noted above, high intensity coastal aquaculture decreases public access to the coastal ocean for recreation and other pursuits. Marine fish culture can also lead to additional pollution and wastes. Excess feed, feces and other organic matter from fish farms can accumulate in the benthos and result in a substantial alteration of the benthic community. (Wu 1995, Henderson and Ross 1995, Hansen 1994). In addition, some prophylactic chemicals and drugs used in fish culture have unknown impacts on marine ecosystems. Clearly, both stock enhancement and marine aquaculture must be approached carefully to maximize their benefits while ensuring the health of natural ecosystems and the continued production of wild stocks (Travis et al. 1998).

management actions effectively protect ecosystem function. Thus, these programs must be empirically sound and supported by rigorous statistical sampling that avoids bias. While the probability of accomplishing this is low—because replication is often unrealistic and sample sizes are, of necessity, quite small—it does not justify avoidance of establishing long-term monitoring programs (Walters 1986). In particular, the issue of cumulative impacts cannot be addressed without baseline data. Monitoring programs are essential to the success of fisheries management, particularly if we are to discern effects due to fishery policies from those due to other factors.

8. Assess the ecological, human, and institutional elements of the ecosystem which most significantly affect fisheries, and are outside Council/DOC authority. Included should be a strategy to address those influences in order to achieve both FMP and FEP objectives.

In many cases the preponderance of the ecosystem relevant to a particular fishery is under the jurisdiction of the Councils and DOC, but in many cases significant portions of the ecosystem will be outside of that jurisdiction. Examples include salmon, where inland water and habitat issues are paramount and under the jurisdiction of other Federal, State, local and tribal authorities; highly migratory species, where significant parts of the ecosystem are under the jurisdiction of different nations; or ecosystems as extensive as the Gulf of Mexico, where general water quality is critically affected by inflow from ecosystems as broad as the Mississippi River drainage area. Some elements of the ecosystem may be outside of Council/DOC jurisdiction; human constituents may move in and out of Council/DOC jurisdiction and many institutions other than the Councils/DOC may share authority over parts of the ecosystem.

Accounting for the effects of these external influences in the FEP is a two-stage process. First, Councils must identify the most significant elements which are outside Council/DOC authority. This list should include the most significant external effects on ecosystem health. Second, Councils should develop a strategic approach to mitigate each of the major impacts. This approach could include the development of agreements with other agencies to address significant ecosystem impacts, or increased research on ecosystem functions or processes which

are affected by outside influences, and which may require mitigation.

Institutional and Human Ecologies—
The Case of Pacific Coast Salmon

The ecology of a Pacific coast salmon fishery includes not only the ocean environment but the rivers in which the fish spawn and the terrestrial habitat related to those rivers. The human ecology of that salmon fishery includes not only the commercial, tribal and recreational fishermen, but also their ancillary businesses and industries. There are also the businesses and industries which have direct effects on the ocean and the coastal riverine habitats (oil and gas, logging, hydroelectric power, development and construction, agriculture and other water diverters) and the citizens who are concerned about the salmon and their habitat even though they do not directly interact with the fish.

The institutional ecology of this salmon fishery includes NMFS, other Federal and State fishery agencies, Native American tribes, and all those institutions which govern the behavior of all of the constituent groups of the human ecology. In fact, 37 Federal agencies, in 9 executive level departments, have some authority over activities affecting marine fisheries and their habitat (Hinman and Safina 1992). Not only is it important to recognize the critical role of this broader set of institutions, but also the role of information, education, and involvement of all of the individuals and groups within the broader set of human constituents whose behaviors are governed by those institutions.

MEASURES TO IMPLEMENT FEPS

The following are general recommendations to ensure effective development and implementation of FEPs:

1. Encourage the Councils to apply ecosystem Principles, Goals and Policies to ongoing activities.

In preparation for FEP implementation, Councils should begin to apply the ecosystem Principles, Goals and Policies to the conservation and management measures of existing and future FMPs. Three actions are particularly important; specifically, each FMP's conservation and management measures should:

- **Consider predator-prey interactions affected by fishing allowed under the FMP.**

  Optimum yields should be set considering ecological factors and the integrity of the ecosystem, and total allowable catches should be justified with respect to total ecosystem biomass, production and interspecies relationships.

- **Consider bycatch taken during allowed fishing operations and the impacts such removals have on the affected species and the ecosystem as a whole, in terms of food web interactions and community structure.**

  FMPs should identify bycatch taken by gear types and should not just provide a list of species, but describe how bycatch changes temporally and spatially in a given fishery. Management actions should consider the implications of such removals and their consequences. FMPs should identify and consider existing or potential alternative gear types or fishing practices which could reduce such bycatch.

- **Minimize impacts of fisheries operations on EFH identified within the FEP.**

  Gear effects on habitat can be considerable. Gear used to harvest a particular species may directly or indirectly affect other species—managed or unmanaged—within the ecosystem. FMPs should not only identify such impacts but should also identify existing or potential alternative gear types or fishing patterns, such as area closures, which could alleviate these impacts.

2. Provide training to Council members and staff.

To facilitate an ecosystem approach and to aid the development and implementation of FEPs, NMFS should provide all Council members with basic instruction in ecological principles. Further, training materials should be made available to the fishing industry, environmental organizations and other interested parties.

3. Prepare guidelines for FEPs.

The Secretary of Commerce should charge NMFS and the Councils with establishing guidelines

for FEP development, including an amendment process. NMFS and the Councils should conduct a deliberative process—similar to the process of developing National Standards Guidelines—to ensure that FEPs are realistic and adaptive.

4. Develop demonstration FEPs.

Choose one or more of the Councils to develop a demonstration FEP. Convene a workshop involving all Councils and other relevant participants which would help develop useful demonstration FEPs.

Encourage all Councils to develop framework FEPs, consisting of such information as can be collected with little additional effort, to facilitate rapid implementation of the full FEP when required by the next MSFCMA reauthorization.

5. Provide oversight to ensure development of and compliance with FEPs.

To ensure compliance with the development of FEPs, the Secretary of Commerce should establish a review panel for FEP implementation oversight. Implicit in this action is the establishment of a timetable for development of a draft FEP, its review by the panel, and any necessary revisions before the draft FEP becomes a basis for policy.

6. Enact legislation requiring FEPs.

To provide NMFS and the Councils with the mandated responsibility of designing and implementing FEPs, Congress should require full FEP implementation in the next reauthorization of the MSFCMA.

RESEARCH REQUIRED TO SUPPORT MANAGEMENT

Our identification of the Principles and associated management Policies reflects a vast amount of scientific knowledge about marine ecosystems and their relationship to humankind. This knowledge is the result of more than 125 years of scientific investment. Yet, the current state of scientific knowledge is not sufficient to fully implement the Principles and Policies. To more fully benefit from the application of the Principles and Policies, there is an urgent need for a better understanding of ecosystem processes in general, and about the state and dynamics of specific ecosystems.

The Panel did not attempt to develop an exhaustive set of research recommendations. That is better left to more specialized groups of scientists. Instead, we highlighted three research themes based on several criteria. First, we selected themes that were clearly related to the Principles and the Policies that form the basis of an ecosystem approach to fisheries management. Second, we placed a priority on identification of new research directions, compared to current research programs that support fisheries management. These new research directions are not recommended as alternatives to the current research programs, rather they are an additional requirement. Third, we highlighted themes for which NMFS has a unique responsibility.

The three recommended research themes are: 1) determine the ecosystem effects of fishing, 2) monitor trends and dynamics of marine ecosystems, and 3) explore ecosystem-based approaches to governance. Each of the themes is briefly described and discussed below.

1. Determine the ecosystem effects of fishing.

The effects of fishing on the species that are landed are generally understood, although the data that are necessary to assess specific stocks of fish are sometimes minimal. It is well known that the effect of fishing on a “target species” can be severe, with abundance reduced by a factor of 10 or more. Fishing is a form of directional selection on fished species that may alter not only population characteristics (i.e., age structure), but also the genetic makeup of the population. Research on genetic changes from fishing is appropriate. It is also known that fishing can have significant effects on nontarget species and, potentially, on marine ecosystems as a whole. These effects occur as a result of bycatch and discarding of non-target species (including marine mammals, reptiles and birds), trophic linkages between target and non-target species, and alteration of habitat caused by fishing gear. All three of these effects need to be studied. The research should consider how fishing changes ecosystems (i.e., abundance and diversity of species, food web dynamics, amount of various habitat types, and the functional significance of changes). An important element of this research will be to explore the utility of quantitative ecosystem health indices as a tool for managers. The research should also

include consideration of strategies for applying the precautionary approach in light of uncertainty about ecosystem effects of fishing, and mitigation of undesirable effects. One particularly promising approach for risk-averse management is the establishment of marine protected areas and through traditional fisheries management techniques like time/area closures.

2. Monitor trends and dynamics in marine ecosystems (ECOWATCH).

We recommend the initiation of a significant new ecosystem monitoring program. We refer to the program as "ECOWATCH" because it will enable scientists and policy makers to observe natural and human-caused changes in marine ecosystems in a comprehensive manner. Target fish species are routinely monitored using landings data and resource surveys that apply standardized sampling methods. But even for some important exploited species, landings data and/or resource survey data are limited. Data on other components of marine ecosystems are even more limited, although there are some valuable time series of plankton data for a few ecosystems and for some marine mammal populations. For these reasons, ECOWATCH should be scientifically designed to provide data to improve existing models (i.e., stock assessments), but also for input for future ecosystem models. Research on ecosystem models based on current concepts of important ecosystem linkages is a useful application of ECOWATCH monitoring data.

We recommend substantial expansion of existing programs that collect data on trends and dynamics of marine ecosystems and which characterize the biological and physical relationships pertinent to ecosystem-based management. This expansion is needed to fill gaps in current data collection programs for some target species where data are limited, and systematically observe how other components of ecosystems vary. There are several reasons to observe marine ecosystems holistically. Such observations are needed to determine and understand indirect effects of fishing within marine ecosystems. In a sense, these observations are a form of ecosystem insurance. Because we cannot currently predict all of the ecosystem effects of fishing, we should be watching for evidence of such changes so that it is possible to react if the changes are adverse or positive. Ecosystem observations are also needed to distinguish human caused changes from natural changes. Large spatial and temporal scale (over ocean basins and decades) changes in ecosystems, called regime shifts, are known to occur. Routine monitoring and analysis of key ecosystem variables are needed in order to detect regime shifts and, if possible, to forecast them.

We envision that ECOWATCH will assess the productive capacity of marine ecosystems, including data on fish, shellfish, primary production, plankton, benthic communities (impacts on fishing sites versus control sites), marine mammals, birds, and physical and chemical factors. It will be necessary to make a major investment in new technology to make ECOWATCH feasible. It will be necessary to employ several different sampling "vehicles" including research vessels; dockside and sea sampling of fisheries; remote sensing from satellites, aircraft, and buoys; submersibles and autonomous underwater vehicles. It will be essential to develop modern data management systems so that variables can be related to each other and so that information is accessible. Models need to be developed to assimilate data and produce information products that enhance our ability to evaluate and make conscious decisions regarding marine ecosystems.

3. Explore ecosystem-based approaches to governance.

Many of today's fishery problems result from failed governance systems. One of the major shortcomings of past and most present governance systems is that they do not create incentives for humans to be prudent predators (i.e., efficient in the uses of natural resources and concerned about long-term conservation). A related problem is that members of the fishing industry and the concerned public often feel alienated from the institutions that govern fisheries. The challenge of achieving effective governance from an ecosystem perspective is even greater. From such a perspective, incentives for efficiency and conservation must apply to indirect effects of fishing on segments of society that are not directly concerned with fisheries, and to other industry sectors that indirectly affect fisheries. A broad array of stakeholders should have the opportunity to participate in the system of governance.

We envision a multifaceted research program including: 1) research on the social and economic importance of fisheries, and of other ecosystem uses

that affect fisheries, to better understand social objectives, motivations for behavior, and options for creating effective incentive systems; 2) case studies and comparative studies (with other industry sectors) to identify factors that determine success or failure of governance systems; and 3) management experiments to test approaches for involving stakeholders in governance systems and for making decisions when faced with multiple objectives (i.e., from different societal perspectives and across sectors).

While NMFS clearly has lead responsibility for these themes, the research strategies should be developed and implemented as National, interagency programs, involving academic as well as government scientists. Because the ecosystem Principles apply globally, the U.S. should participate in, and initiate when necessary, international programs that further fisheries management objectives. A significant enhancement in resources (e.g., funding, staff, fishery research vessels) will be required if these research recommendations are to be fulfilled.

SECTION FIVE: SUMMARY AND CONCLUSIONS

Recognition of major problems in U.S. fisheries prompted Congress to legislate the Sustainable Fisheries Act (SFA) in 1996. This amendment strengthened the MSFCMA and gave new direction to NMFS and the Councils to halt overfishing, develop recovery plans for overfished fisheries, avoid and reduce bycatch mortality, identify and protect EFH, investigate ways to reduce fishing capacity, and implement numerous other conservation measures. These represent the beginnings of an ecosystem approach to fishery management. Rapid response and hard work by NMFS, the Councils, fishing industries, environmental groups and other interested parties will produce change that eventually will result in marked improvements in the status and management of our fisheries resources. Still, there is more to be done.

The appointment of the NMFS Ecosystem Principles Advisory Panel is a key provision of the SFA. Congress called for an assessment of the extent to which ecosystem principles are being applied in fishery conservation, management and research and for recommendations on how to use them further to improve management. Our review of the use of ecosystem principles finds some positive indications, but much room for further application. The fisheries ecosystem science being conducted is of high quality, but the types of research and assessments, and the geographic coverage are extremely limited and inadequate to inform fishery management. Where scientific information on fisheries ecosystems is produced, it is often used in the management process. However, it is inadequate relative to the scope of the problems and the geographic scale of our Nation's marine fisheries.

At present, NMFS and the Councils often are using the best available science to manage stocks on a single species or species-complex basis. If fishery management is to further incorporate ecosystem principles, Congress must provide a specific mandate to NMFS and the Councils to do so and must fund the scientific infrastructure required to support the decision-making process. Requiring Councils to prepare FEPs provides a mechanism to focus and inform fishery management, to measure progress toward implementation of ecosystem-based fishery management, to identify research needs and ultimately to insure healthy and productive ecosystems.

U.S. fisheries under an ecosystem-based management system are likely to be quite different than today's fisheries. New management tools will be employed including share-based systems. Fisheries and gear types that have significant adverse impacts on other ecosystem components may be modified or phased out and other types of fisheries and gears may replace them. In some cases, fish stocks may have to be exploited at lower harvest levels than presently indicated in order to sustain other ecosystem components. Some areas that are now fished may become fisheries reserves where harvests are restricted to protect a spawning stock or other sensitive life-history stages; this may result in changes to traditional fishing practices. The short-term consequences of such changes, which may be painful, must be balanced against future benefits in the form of sustainable fisheries and fishing communities.

The next ten years are critical for the future of U.S. fisheries. Already, important changes are underway as a result of the SFA, and the next round of legislation/reauthorization of the MSFCMA should provide additional impetus for reform. Implementation of an ecosystem-based approach will take time and there will be trials and errors. A great deal of education about this new approach will be required, and all involved must be prepared to learn. The two hardest lessons are likely to be shifting the burden of proof to the fishery to demonstrate that the ecosystem will not be damaged by fishing, and to develop a truly precautionary approach to fishery management. The learning curve will be steep for all involved; society as a whole, will be increasingly challenged to help define ecosystem health and the limits of acceptable change in marine ecosystems, while still allowing sustainable fishing practices.

GLOSSARY

Allowable Biological Catch—Catch that can be taken in a specific year that achieves the biological objectives, or avoids the biological constraints, of fishery management. Such objectives and constraints are usually set in terms of stock sizes that must be maintained and/or fishing mortality rates that shall not be exceeded. Estimates of allowable biological catch should be based on the best scientific advice available.

Burden Of Proof—The responsibility to demonstrate that a fishing activity will or will not lead to overfishing or negative effects on the ecosystem.

Bycatch—Unintentional catch; i.e., catch that occurs incidentally in a fishery that intends to catch fish with other characteristics (e.g., size, species).

Carrying Capacity—The numbers or biomass of resources that can be supported by an ecosystem.

Conservation and Management—The rules, regulations, conditions, methods, and other measures (A) which are required and useful to rebuild, restore, or maintain, any fishery resource and the marine environment; and (B) which are designed to ensure that: (i) a supply of food and other products may be taken, and that recreational benefits may be obtained, on a continuing basis; (ii) irreversible or long-term adverse effects on fishery resources and the marine environment are avoided; and (iii) there will be a multiplicity of options available with respect to future uses of these resources (NMFS 1996).

Discards—A portion of what is caught and returned to the sea unused. Discards may be either alive or dead. There are many types of discards, such as economic discards (when a portion of the catch that it is not economically rational to land is discarded), regulatory discards (when discarding occurs because of a prohibition on retaining some of the catch), highgrade discards (discarding of the portion of the catch with a lower value than the portion retained in order to comply with regulations that limit how much catch can be retained). Highgrading is a form of regulatory discarding.

Ecosystem-based Fishery Management—Fishery management actions aimed at conserving the structure and function of marine ecosystems, in addition to conserving the fishery resource.

Essential Fish Habitat—Those waters and substrate necessary for fish to spawn, breed, feed and grow to maturity (NMFS 1996).

Fish—Defined herein as finfish, mollusks, crustaceans, and all other forms of marine animal and plant life other than marine mammals and birds (NMFS 1996).

Fishery—(A) One or more stocks of fish which can be treated as a unit for purposes of conservation and management and which are identified on the basis of geographical, scientific, technical, recreational and economics characteristics; and (B) any fishing for such stocks (NMFS 1996).

Fishing—Any activity which can reasonably be expected to result in the catching, taking or harvesting of fish; or any operations at sea in support of, or in preparation for such activities.

Fishing Mortality—A measurement of the rate of mortality of fish in a population caused by fishing.

Fish Stock—A species, subspecies, geographical grouping, or other grouping of fish that is managed as a unit (NMFS 1996).

Maximum Sustainable Yield—A management goal specifying the largest long-term average catch or yield (in terms of weight of fish) that can be taken, continuously (sustained) from a stock or stock complex under prevailing ecological and environmental conditions, without reducing the size of the population.

Optimum Yield—(A) The amount of fish which will provide the greatest overall benefit to the Nation, particularly with respect to food production and recreational opportunities, and taking into account the protection of marine ecosystems; (B) is prescribed as such on the basis of the maximum sustainable yield from the fishery, as reduced by any

relevant economic, social, or ecological factor; and (C) in the case of an overfished fishery, provides for rebuilding to a level consistent with producing the maximum sustainable yield in such fishery (NMFS 1996).

OVERFISHING—Fishing at a rate or level that jeopardizes the capacity of a stock or stock complex to produce maximum sustainable yield on a continuing basis (NMFS 1996).

PRIMARY PRODUCTION—Creation of organic matter by plants through photosynthesis (using inorganic carbon, nutrients and an external energy source) to form the base of the food chain.

RECRUITMENT—A measure of the weight or number of fish which enter a defined portion of the stock such as fishable stock (those fish above the minimum legal size) or spawning stock (those fish which are sexually mature).

REGIME SHIFT—Major changes in levels of productivity and reorganization of ecological relationships over vast oceanic regions which could be caused by various sources including climate variability or overfishing.

RESILIENCE—The ability of a population or ecosystem to withstand change and to recover from stress (natural or anthropogenic).

SIGNIFICANT FOOD WEB—A predator/prey interaction that is important to either the predator or prey population.

STOCK ASSESSMENT—An evaluation of a stock in terms of abundance and fishing mortality levels and trends, and relative to fishery management objectives and constraints if they have been specified.

SURPLUS PRODUCTION—Total weight of fish that can be removed by fishing without changing the size of the population. It is calculated as the sum of the growth in weight of individuals in a population, plus the addition of biomass from new recruits, minus the biomass of mortality of animals lost to natural mortality, during a defined period (usually one year).

TARGET SPECIES—Those fish explicitly sought by fishermen to meet social and economic needs. Their catch are the direct consequence of targeted fishing effort. **NON-TARGET SPECIES** include all others.

TOTAL ALLOWABLE CATCH—The annual catch from a stock that is allowed according to fishery management regulations.

TROPHIC WEB—The network that represents the predator/prey interactions of an ecosystem.

LITERATURE CITED

Agardy, M. T. 1994. Advances in marine conservation: The role of protected areas. Trends in Ecology and Evolution 9(7):267–270.

Alverson, D. L., M. H. Freeburg, S. A. Murawski and J. G. Pope. 1994. *A Global Assessment of Fisheries Bycatch and Discards*. FAO Fisheries Technical Paper 339, Food and Agriculture Organization, Rome.

Alverson, D. L. and P. A. Larkin. 1994. Fisheries: Fisheries Science and Management. Pages 150-167 in: C. D. Voigtlander (ed.) *The state of the world's fishery resources: Proceedings of the World Fishery Congress*, Plenary Session, Oxford and IBH Publishing, New Delhi.

Angermeier, P. L. 1997. Conceptual roles of biological integrity and diversity. Pages 49-65 in: J. E. Williams, C. A. Wood and W. P. Dombeck (eds.) *Watershed Restoration: Principles and Practices*. American Fisheries Society, Bethesda, Maryland.

Angermeier, P. L. and J. R. Karr. 1994. Biological integrity versus biological diversity as policy directives - Protecting biotic resources. Bioscience 44(10):690-697.

Apollonio, S. 1994. The use of ecosystem characteristics in fisheries management. Reviews in Fisheries Science 2(2):157–180.

Auster, P. J. and R. W. Langton. 1999. The effects of fishing on fish habitat. Pages 150-187 in: L. Benaka (ed.) *Fish Habitat: Essential Fish Habitat and Rehabilitation*. American Fisheries Society, Bethesda, Maryland.

Axelrod, R. 1984. *Evolution of Cooperation*. Basic Books, New York.

Barber, R. T. and F. P. Chavez. 1983. Biological consequences of El Niño. Science 222:1203–1210.

Beveridge, M. C. M., L. G. Ross and L. A. Kelly. 1994. Aquaculture and biodiversity. Ambio 23(8):497–502.

Boehlert, G. W. 1996. Biodiversity and the sustainability of marine fisheries. Oceanography 9(1):28–35.

Boehlert, G. W. and J. D. Schumacher (eds.), 1997. *Changing oceans and changing fisheries: environmental data for fisheries research and management*. United States Department of Commerce, National Oceanic and Atmospheric, National Marine Fisheries Service, Southwest Fisheries Science Center, Technical Memorandum NMFS-SWFSC-239.

Bohnsack, J. A. and J. S. Ault. 1996. Management strategies to conserve marine biodiversity. Oceanography 9:73–82.

Botsford, L. W., J. C. Castilla and C. H. Peterson. 1997. The management of fisheries and marine ecosystems. Science 277:509-515.

Caddy, J. F. and R. Mahon. 1995. *Reference Points for Fisheries Management*. FAO Fisheries Technical Paper 347, Food and Agriculture Organization, Rome.

Carr, M. H. and D. C. Reed. 1993. Conceptual issues relevant to marine harvest refuges: Examples from temperate reef fishes. Canadian Journal of Fisheries and Aquatic Science 50:2019–2028.

Christensen, N. L. and 12 others. 1996. The report of the Ecological Society of America committee on the scientific basis for ecosystem management. Ecological Applications 6(3):665–691.

Christensen, V. and D. Pauly. 1995. Fish production, catches and the carrying capacity of the world oceans. ICLARM Quarterly 18(3):34-40.

Costanza, R. 1987. Social traps and environmental policy. BioScience 37:407–412.

Costanza, R. and L. Cornwell. 1992. The 4P approach to dealing with scientific uncertainty. Environment 34(9):12–20, 42.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, S. Naeem, K. Limburg, J. Paruelo, R. V. O'Neill, R. Raskin, P. Sutton and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387:253–260.

Dayton, P. K. 1998. Reversals of the burden of proof in fisheries management. Science 279:821–822.

Deimling, E. A. and W. J. Liss. 1994. Fishery development in the eastern North Pacific: A natural-cultural system perspective, 1888-1976. Fisheries Oceanography 3:60–77.

Dugan, J. E. and G. E. Davis. 1993. Applications of marine refugia to coastal fisheries management. Canadian Journal of Fisheries and Aquatic Science 50:2029–2042.

Fogarty, M. J. and S. A. Murawski. 1998. Large-scale disturbance and the structure of marine systems: Fishery impacts on Georges Bank. Ecological Applications Supplement 8(1):S6–S22.

Food and Agriculture Organization. 1995. *Code of Conduct for Responsible Fisheries*. Food and Agriculture Organization of the United Nations, Rome.

Francis, R. C., S. R. Hare, A. B. Hollowed and W. S. Wooster. 1998. Effects of interdecadal climate variability on the oceanic ecosystems of the NE Pacific. Fisheries Oceanography 7:1–21.

Gallopin, G. C. 1995. The potential of agroecosystem health as a guiding concept for agricultural research. Ecosystem Health 1(3):129–141.

Garcia, S. M. and C. H. Newton. 1994. Responsible fisheries: An overview of Food and Agricultural Organization policy developments (1945-1994). Marine Pollution Bulletin 29: 528–536.

Garcia, S. M. and C. H. Newton. 1997. Current situation, trends and prospects in world capture fisheries. Pages 3-27 in: E. K. Pikitch, D. D. Huppert and M. P. Sissenwine (eds.) *Global Trends: Fisheries Management*. American Fisheries Society Symposium 20. American Fisheries Society, Bethesda, Maryland.

Gordon, H. S. 1954. The economic theory of common property resources: The fishery. Journal of Political Economy 62:124–142.

Graham, M. 1935. Modern theory of exploiting a fishery, and its application to North Sea trawling. Journal du Conseil 13:264–274.

Hansen, P. K. 1994. Benthic impact of marine fish farming. Pages 77-81 in: A. Ervik, P. K. Hansen, and V. Wennevik (eds.) *Proceedings of Canada-Norway Workshop on Environmental Impacts of Aquaculture*. Fisken-Havet, Bergen, Norway. Havforskningsintituttet. No. 13.

Henderson, A. R. and D. J. Ross. 1995. Use of macrobenthic infaunal communities in the monitoring and control of the impact of marine cage fish farming. Aquaculture Research 26:659-678.

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

Hinman, K., and C. Safina. 1992. Summary and recommendation. Pages 245-249 in: R. H. Stroud (ed.) *Stemming the Tide of Coastal Fish Habitat Loss: Proceedings of a Symposium on Conservation of Coastal Fish Habitat*. Baltimore, Maryland, March 7-9, 1991, Marine Recreational Fisheries Symposium 14, National Coalition for Marine Conservation, Inc., Savannah, Georgia.

Hobart, W. L. (ed.) 1995. *Baird's Legacy: The History and Accomplishments of NOAA's National Marine Fisheries Service, 1871–1996*. United States Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Technical Memorandum NMFS-F/SPO-18.

Holling, C. S. (ed.) 1978. *Adaptive Environmental Assessment and Management.* John Wiley and Sons, London.

Holling, C. S. and G. K. Meffe. 1996. Command and control, and the pathology of natural resource management. Conservation Biology 10(2):328-337.

Hughes, T. P. 1994. Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. Science 265:1547-1551.

Hutchings, J. A., C. Walter and R. L. Haedrich. 1997. Is scientific inquiry incompatible with the government information control? Canadian Journal of Fisheries and Aquatic Science 54(5):1198.

Huxley, T. H. 1883. Inaugural Address. *The Fisheries Exhibition Literature, International Fisheries Exhibition*, London 4:1–22.

International Society for Ecosystem Health. 1998. What is ecosystem health? URL: http://www.uoguelph.ca/~rmoll/whatiseh.html.

Kolb, T. E., M. R. Wagner and W. W. Covington. 1994. Concepts of forest health: Utilitarian and ecosystem perspectives. Journal of Forestry 92: 10–15.

Langton, R. W., P. J. Auster and D.C. Schneider. 1995. A spatial and temporal perspective on research and management of groundfish in the northwest Atlantic. Reviews in Fisheries Science 3(3):201-229.

Langton, R. W. and R. L. Haedrich. 1997. Ecosystem-based management. Pages 153-158 in: J. Boreman, B. S. Nakashima, J. A. Wilson and R. L. Kendall (eds.) *Northwest Atlantic Groundfish: Perspectives on a Fishery Collapse*. American Fisheries Society, Bethesda, Maryland.

Langton, R. W., R. S. Steneck, V. Gotcetas, F. Juanes and P. Lawton. 1996. The interface between fisheries research and habitat management. North American Journal of Fisheries Management 16:1–7.

Lannan, J. E., G. A. E. Gall, J. E. Thorpe, C. E. Nash and B. E. Ballachey. 1989. Genetic resource management of fish. Genome 31:798–804.

Lauck, T., C. W. Clark, M. Mangel and G. R. Munro. 1998. Implementing the precautionary principle in fisheries management through marine reserves. Ecological Applications Supplement 8(1):S72–S78.

Laws, R. M. 1977. Seals and whales of the Southern Ocean. Philosophical Transactions of the Royal Society, London B279:81–96.

Lough, R. G., P. C. Valentine, D. C. Potter, P. J. Auditore, G. R. Bolz, J. S. Neilson and R. I. Perry. 1989. Ecology and distribution of juvenile cod and haddock in relation to sediment type and bottom currents on eastern Georges Bank. Marine Ecology Progress 56:1-12.

Lubchenco, J., A. M. Olson, L. B. Brubaker, S. R. Carpenter, M. M. Holland, S. P. Hubbell, S. A. Levin, J. A. MacMahon, P. A. Matson, J. M. Melillo, H. A. Mooney, C. H. Peterson, H. R. Pulliam, L. A. Real, P. J. Regal and P. G. Risser. 1991. The sustainable biosphere initiative: An ecological research agenda. A report of the Ecological Society of America. Ecology 72(2):371–412.

Ludwig, D., R. Hilborn and C. J. Walters. 1993. Uncertainty, resource exploitation, and conservation: Lessons from history. Science 260:17, 36.

Mangel, M. and 41 co-authors. 1996. Principles for the conservation of wild living resources. Ecological Applications 6(2):338-362.

Mantua, N. J., S. R. Hare, Y. Zhang, J. M. Wallace and R. C. Francis. 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin of the American Meteorological Society 78:1069–1079.

Marten, G. G. 1979. Predator removal: Effect on fisheries yields in Lake Victoria (East Africa). Science 203:646–648.

McGowan, J. A., D. R. Cayan and L. M. Dorman. 1998. Climate-ocean variability and ecosystem response in the Northeast Pacific. Science 281:210-217.

Moav, R., T. Brody and G. Hulata. 1978. Genetic improvement of wild fish populations. Science 201:1090–1094.

Mooney, H. A. (ed.) 1998. Ecosystem management for sustainable fisheries. Ecological Applications Supplement 8(1): S1.

Morikawa, T. 1994. Maintenance of the fisheries environment and efforts to increase the resources in the coastal waters of Japan. Marine Pollution Bulletin 29:537–549.

Mountford, K. 1996. A capsule history of the Chesapeake Bay. Chesapeake Bay Program. URL:http://www.epa.gov/r3chespk/cbp_home/bay_eco/history/histcont.htm.

National Marine Fisheries Service (NMFS). 1996. *Magnuson-Stevens Fishery Conservation and Management Act*. United States Department of Commerce, National Oceanic and Atmospheric Administration, NMFS, Technical Memorandum NMFS-F/SPO-23.

National Marine Fisheries Service (NMFS). 1997. *Report to Congress: Status of fisheries of the United States*. United States Department of Commerce, National Oceanic and Atmospheric Administration, NMFS, Silver Spring, Maryland.

National Research Council. 1996. *The Bering Sea Ecosystem*. National Academy Press, Washington, D. C.

National Research Council. 1999. *Sustaining Marine Fisheries.* National Academy Press, Washington, D. C.

Naylor, R. L., R. J. Goldburg, H. Mooney, M. Beveridge, J. Clay, C. Folke, N. Kautsky, J. Lubchenco, J. Primavera and M. Williams. 1998. Nature's subsidies to shrimp and salmon farming. Science 282:883-884.

Nichols, F. H., J. E. Cloern, S. N. Luoma and D. H. Peterson. 1986. The modification of an estuary. Science 231:567-573.

Ogura, M. and S. O. Ito. 1994. Change in the known ocean distribution of Japanese chum salmon, *Oncorhynchus keta*, in relation to the progress of stock enhancement. Canadian Journal of Fisheries and Aquatic Science 51:501–505.

Pauly, D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. Trends in Ecology and Evolution 10:430.

Pauly, D. and V. Christensen. 1995. Primary production required to sustain global fisheries. Nature 374:255–257.

Pauly, D., V. Christensen, J. Dalsgaard, R. Froese and F. Torres Jr. 1998. Fishing down marine food webs. Science 279:860–863.

Pearcy, W. G. and A. Schoener. 1987. Changes in the marine biota coincident with the 1982-83 El Niño in the Northeastern Subarctic Pacific Ocean. Journal of Geophysical Research 92:14417–14420.

Pickett, S. T. A., V. T. Parker and P. L. Fiedler. 1992. The new paradigm in ecology: Implications for conservation biology above the species level. Pages 65-88 in: P. L. and S. K. Jain (eds.) *Conservation Biology: The Theory and Practice of Nature Conservation, Preservation, and Management*. Chapman and Hall, New York.

Pinkerton, E. (ed.) 1989. *Co-operative Management of Local Fisheries: New Directions for Improved Management and Community Development*. University of British Columbia Press, Vancouver, British Columbia.

Polovina, J. J. 1984. Model of a coral reef ecosystem I. The ECOPATH model and its application to French Frigate Shoals. Coral Reefs 3:1-10.

Polovina, J. J., G. T. Mitchum and G. T. Evans. 1995. Decadal and basin-scale variation in mixed layer depth and the impact on biological production in the Central and North Pacific, 1960–88. Deep-Sea Research I 42:1701–1716.

Rapport, D. J., C. Gaudet and P. Calow (eds.) 1995. *Evaluating and Monitoring the Health of Large-scale Ecosystems*. Springer-Verlag, New York.

Roberts, C. M. 1997. Ecological advice for the global fisheries crisis. Trends in Ecology and Evolution 12:35–38.

Roemmich, D. and J. McGowan. 1995. Climatic warming and the decline of zooplankton in the California Current. Science 267:1324–1326.

Rosenberg, A. A., M. J. Fogarty, M. P. Sissenwine, J. R. Beddington and J. G. Shepherd. 1993. Achieving sustainable use of renewable resources. Science 262:828–829.

Ryman, N., F. Utter and L. Laikre. 1995. Protection of intraspecific biodiversity of exploited fishes. Reviews in Fish Biology and Fisheries 5:417–446.

Sen, S. and J. R. Nielson. 1996. Fisheries co-management: A comparative analysis. Marine Policy 20(5):405–418.

Sissenwine, M. P. and A. Rosenberg. 1993. Marine fisheries at a critical juncture. Fisheries 18(10):6–14.

Sissenwine, M. P. 1987. Councils, NMFS, and the Law. Pages 203-204 in: R. Stroud (ed.) *Recreational Fisheries* (11). Sport Fishing Institute. Washington, D. C.

Sparks, R. E. 1995. Need for ecosystem management of large rivers and flood plains. Bioscience 45(3):168-182.

Steele, J. H. 1991. Marine ecosystem dynamics: comparison of scales. Ecological Research 6:175–183.

Steele, J. H. 1996. Regime shifts in fisheries management. Fisheries Research 25:19–23.

Thompson, W. F. 1919. The scientific investigation of marine fisheries, as related to the work of the Fish and Game Commission in Southern California. Fisheries Bulletin (Canada) 2:3–27.

Travis, J., F. Coleman, C. B. Grimes, D. Conover, T. M. Bert and M. Tringali. 1998. Critically assessing stock enhancement: An introduction to the Mote Symposium. Bulletin of Marine Science 62(2):305-311.

United Nations. 1996. *Agreement for the Implementation of the Provisions of the United Nations Convention on the Law of the Sea of 10 December 1982 Relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks*. United Nations General Assembly, New York.

Walsh, J. J. 1981. A carbon budget for overfishing off Peru. Nature 290:300–304.

Walters, C. J. 1986. *Adaptive Management of Renewable Resources*. Macmillan, New York.

Walters, C. J. 1997. Challenges in adaptive management of riparian and coastal ecosystems. Conservervation Ecology (online) 1(2): 1. URL: http://www.consecol.org/vol1/iss2/art1

Walters, C., V. Christensen and D. Pauly. 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Reviews in Fish Biology and Fisheries 7(2):139-172.

Whitford, W. H. 1995. Desertification: Implications and limitations of the ecosystem health metaphor. Pages 273-293 in: D. J. Rapport, C. Gaudet, and P. Calow (eds.) *Evaluating and Monitoring the Health of Large-scale Ecosystem*s. Springer-Verlag, New York.

Wilcove, D., M. Bean and P. C. Lee. 1992. Fisheries management and biological diversity: Problems and opportunities. Transactions of the North American Wildlife and Natural Resources Conference 5:373–383.

Wu, R. S. S. 1995. The environmental impact of marine fish culture: Towards a sustainable future. Marine Pollution Bulletin 31(4-12):159-166.

Yoklavich, M. M. 1998. Marine harvest refugia for west coast rockfish: An Introduction to the Workshop. Pages 1-5 in: M. M. Yoklavich (ed.) *Marine Harvest Refugia for West Coast Rockfish: A Workshop.* United States Department of Commerce, National Oceanic and Atmospheric Administration, NMFS, Technical Memorandum NMFS-SWFSC-255.

APPENDIX A: CHARTER—NATIONAL MARINE FISHERIES SERVICE ECOSYSTEM PRINCIPLES ADVISORY PANEL

The Charter was provided to the Panel as initial guidance from NMFS. It was subsequently modified after Panel review.

INTRODUCTION

Section 406 of the Magnuson-Stevens Fisheries Conservation and Management Act (MSFCMA) as amended through 1996 (Appendix B) requires the Secretary of Commerce to establish a Panel to provide advice to the Secretary and Congress on ways to incorporate ecosystem principles in fisheries conservation and management activities. The need for such a Panel has arisen from the perceived failure of traditional management approaches to ensure sustainable fisheries. Yields of many marine fisheries worldwide have declined in recent years; in the U.S., 42% of fish stocks are considered overutilized. The causes of these declines have been complex, and include overharvesting of target and non-target species, habitat alteration and loss, pollution and natural environmental change. Stocks in this condition are not able to provide the same sustained economic and social benefits as those in healthy fisheries.

A basic premise of ecosystem-based management is that the relationship between living marine resources and the ecosystem within which they exist must be well understood. This requires a more comprehensive approach to fisheries research than is necessary for traditional single-species management approaches, although single-species stock assessments have become increasingly sophisticated and some now incorporate environmental parameters. Successful implementation of ecosystem-based management will require consideration of, inter alia, essential habitat requirements, hydrography, trophic relationships and physical and biological processes. An important element of the Panel's duties will be to determine what information is essential to the task of ecosystem-based fisheries conservation and management, and how that information should be collected.

Managers must also understand the complex linkages between natural ecosystems and the economic, social and political dynamics of human systems. Humans are integral components of ecosystems and their interests, values and motivations must be understood and factored into resource management decisions. Information on human systems is as important as that from natural systems and must be included in any ecosystem research and management efforts.

Efficient use of existing information and information flow to management are important topics for Panel consideration. In developing an ecosystem approach to research and management, it is important to recognize that a great deal is already known about marine ecosystems, but that this information is not consistently applied in current management efforts. This is, in large part, because there is no agreed upon method or process for applying it. Therefore, emphasis must be placed not only on what new information is required, but also on how to apply existing information effectively. In addition, it must be recognized that both science and management are ongoing processes, and that mechanisms are required to incorporate new scientific, social, cultural, economic and institutional information into the management process as it becomes available. This may require managers to be trained in ecosystem approaches, so that valuable new information will be recognized and utilized where appropriate.

The complicated legislative and institutional framework that currently regulates resource management decision making poses a significant challenge to the implementation of ecosystem-based fisheries conservation and management. Although the MSFCMA is the principal legislation governing U.S. marine fisheries, other Federal legislation including the Marine Mammal Protection Act and

the Endangered Species Act, as well as State laws and international agreements, provide for the conservation and management of marine resources. This geographic, legislative and institutional fragmentation of conservation and management responsibilities is not consistent with ecosystem principles, which ignore human boundaries and jurisdictions. It also indicates the need for an 'institutional ecology' and a 'legislative ecology' which parallel more closely the natural ecosystem. Coordination of these legislative and institutional responsibilities across jurisdictional lines, as well as the appropriate involvement of all stakeholders in the decision making process, will be a significant task in implementing ecosystem-based management.

The U.S. lacks a single and unifying legislative mandate or policy governing the use of resources from marine ecosystems. Consequently, decisions on resource management within marine ecosystems often are in conflict with one another. For example, it is axiomatic that fishery yields cannot be maximized for all species simultaneously. Likewise, the goal of protecting all marine mammals within an ecosystem may not be consistent with the goal of sustaining maximum fisheries yields, and vice versa. From the outset, resource managers must determine what values are placed on a marine ecosystem and its components, and which goods and services are expected to be produced from each ecosystem. The recommendations of this Panel regarding the development of such policies will be an important step towards improved fisheries conservation and management.

Numerous panels, committees and task forces have been constituted in the past to consider how ecosystem approaches should be applied to natural resource management issues. Many solid recommendations have emerged from these efforts, however few appear to be implemented in fisheries management, as evidenced by Congress' mandate for this Panel. While the reasons for this failure are probably multiple, an underlying cause may be that many of the recommendations have been more theoretical than practical, and have provided the practicing manager with little in the way of implementable management tools. Unlike these previous efforts, it is fully intended that the NMFS Ecosystem Principles Advisory Panel will develop specific, practical and implementable recommendations for the research, conservation and management of living marine resources, along with longer term goals and directions.

PURPOSE

The Panel's purpose is to advise NMFS and Congress on the application of ecosystem principles in fisheries conservation and management and research activities.

TERMS OF REFERENCE

The Panel will:

1. **Conduct an analysis of the extent to which ecosystem principles are being applied in fishery[1] conservation and management[2] activities, including research activities. The analysis should include the following:**

Conservation and management issues

A review of the extent to which ecosystem principles are being applied in: 1) the development of fishery management plans by the Councils; 2) the development of advice by NMFS to the Councils; and 3) other regulatory and rule-making activities of NMFS.

An identification and analysis of cases in which ecosystem principles have been successfully applied in fisheries conservation and management activities.

Research issues

A review of the status of ecosystem science

[1]The term "fishery" means — (A) one or more stocks of fish which can be treated as a unit for purposes of conservation and management and which are identified on the basis of geographical, scientific, technical, recreational and economics characteristics; and (B) any fishing for such stocks.

[2]The term "conservation and management" refers to all the rules, regulations, conditions, methods, and other measures (A) which are required and useful to rebuild, restore, or maintain, any fishery resource and the marine environment; and (B) which are designed to ensure that:

(i) a supply of food and other products may be taken, and that recreational benefits may be obtained, on a continuing basis;
(ii) irreversible or long-term adverse effects on fishery resources and the marine environment are avoided; and
(iii) there will be a multiplicity of options available with respect to future uses of these resources.

within NOAA and other entities involved with research in the marine environment (e.g., academic institutions, other Federal and State agencies).

An analysis of whether current research efforts within these agencies and institutions are adequate to support fisheries ecosystem conservation and management.

2. Propose a specific, prioritized course of actions that the Secretary of Commerce, Congress and NMFS should undertake to expand the application of ecosystem principles in fishery conservation and management. For example, the following issues might be considered:

Conservation and management issues

What specific, practical actions can be taken to apply ecosystem principles in fisheries conservation and management activities in the near term, before more complete information is available on ecosystem structure and function?

What barriers (scientific, social, institutional, economic, administrative, legislative) exist to the application of ecosystem principles in U.S. fisheries conservation and management activities? What solutions can be proposed?

Should changes be made to the Council structure or mission to better apply ecosystem principles in conservation and management activities? If so, what should the changes be?

Does the U.S. need additional legislation, or changes to current legislation, to improve the scientific and regulatory infrastructure to support ecosystem-based conservation and management?

Research issues

Which research topics should be priorities for the development of a long-term information base to support marine ecosystem management?

How can agencies and institutions involved in marine and fisheries science collaborate more effectively to take advantage of complementary research efforts, and synergize results from a broader ecosystem perspective?

What are the most meaningful time and space scales for marine ecosystem research which will directly support conservation and management efforts?

Is sufficient information available to determine the value of harvest refugia in fisheries ecosystem management? If not, what additional information is required?

3. Produce a report to Congress by October 1998 which includes the above information, plus any other information as may be appropriate.

The principal focus of the analyses in Section 1 above should be on conservation and management and research activities conducted within the U.S., including those marine ecosystems and their resources which are shared by the U.S. and other countries (e.g., transboundary stocks). However, the Panel should consider pertinent examples from other areas of the world where ecosystem approaches have been used. The Panel should focus on research, conservation and management activities which pertain to ecosystems or species under the jurisdiction of the MSFCMA.

Panel Membership

According to MSFCMA Section 406, the Advisory Panel shall consist of not more than 20 individuals and include:

Individuals with expertise in the structures, functions and physical and biological characteristics of ecosystems; and

Representatives from the Regional Fishery Management Councils, States, fishing industry, conservation organizations or others with expertise in the management of marine resources.

Nominations for panelists were solicited from the National Academy of Sciences, Councils, States, fishing industry and conservation organizations, as well as other appropriate regional and national stakeholders. The Panel membership is balanced geographically, so that regional issues can be addressed.

Travel Costs

Travel expenses for the panelists to attend panel

meetings will be paid by the government at prevailing government rates.

Format and Panel Duration

The Panel will convene three two-day meetings in September 1997, November-December 1997, and February-March 1998. Additional meetings or conference calls may be held as required. The Panel may be requested to continue to advise NMFS on ecosystem issues after October 1998 if such advice is required.

All meetings will be open to the public, and each meeting will include a specific opportunity for public input. Members of the public wishing to make presentations or statements at the meetings must notify the NMFS Office of Science and Technology at least two weeks in advance of the meeting date, which will be published in the Register.

APPENDIX B: MSFCMA SECTION 406 FISHERIES SYSTEMS RESEARCH

(a) ESTABLISHMENT OF PANEL.—Not later than 180 days after the enactment of the Sustainable Fisheries Act, the Secretary shall establish an advisory panel under this Act to develop recommendations to expand the application of ecosystem principles in fishery conservation and management activities.

(b) PANEL MEMBERSHIP.—The advisory panel shall consist of not more than 20 individuals and include—

(1) individuals with expertise in the structures, functions, and physical and biological characteristics of ecosystems; and

(2) representatives from the Councils, States, fishing industry, conservation organizations, or others with expertise in the management of marine resources.

(c) RECOMMENDATIONS.—Prior to selecting advisory panel members, the Secretary shall, with respect to panel members described in subsection (b)(1), solicit recommendations from the National Academy of Sciences.

(d) ECOSYSTEM REPORT.—Within two years of the date of enactment of this Act, the Secretary shall submit to the Congress a completed report of the panel established under this section, which shall include—

(1) an analysis of the extent to which ecosystem principles are being applied in fishery conservation and management activities, including research activities;

(2) proposed actions by the Secretary and by the Congress that should be undertaken to expand the application of ecosystem principles in fishery conservation and management; and

(3) such other information as may be appropriate.

(e) PROCEDURAL MATTER.—The procedural matters under section 302(j) with respect to advisory panels shall apply to the Fisheries Ecosystem Management advisory panel..

APPENDIX C: MEETING PARTICIPANTS

First Meeting—September 9 & 10, 1997
Washington, DC

Presenters:

Dave Allison
Allison Associates

Larry Buckley
NMFS, Northeast Fisheries Science Center

David Evans
NMFS, Deputy Assistant Administrator

Karen Garrison
Natural Resources Defense Council

Craig Harrison
Pacific Seabird Group

Don Leedy
NMFS, Office of Sustainable Fisheries

Pat Livingston
NMFS, Alaska Fisheries Science Center

Jeff Polovina
NMFS, Southwest Fisheries Science Center

Mike Schiewe
NMFS, Northwest Fisheries Science Center

Jim Thomas
NMFS, Office of Habitat Protection

Nancy Thompson
NMFS, Southeast Fisheries Science Center

Guests:

Roger Griffis
NOAA, Office of Policy and Strategic Planning

Kate Wing
Staff, Senate Commerce Committee

Tom Eagle
NMFS, Office of Protected Resources

Second Meeting—December 15 & 16, 1997
Seattle, Washington

Presenters:

John Gauvin
Executive Director, Groundfish Forum

Chuck Fowler
NMFS, National Marine Mammal Lab

Lowell Fritz
NMFS, Alaska Fisheries Science Center

Peter Fricke
NMFS, Office of Sustainable Fisheries

Rod Fujita
Environmental Defense Fund

Tom Okey
Center for Marine Conservation

Ken Stump

Dave Witherell
North Pacific Fishery Management Council

Guests:

Kerim Aydin
University of Washington

Jim Balsiger
Director, NMFS Alaska Fisheries Science Center

Ed Casillas
NMFS, Northwest Fisheries Science Center

Tracy Collier
NMFS Alaska Fisheries Science Center

John Fell
University of Washington

Bill Hines
NMFS, Alaska Region

Loh-Lee Low
NMFS, Alaska Fisheries Science Center

Clarence Pautzke
Executive Director, North Pacific Fisheries Management Council

Mike Schiewe
NMFS, Northwest Fisheries Science Center

John Stein
NMFS, Northwest Fisheries Science Center

Usha Varanasi
Director, NMFS Northwest Fisheries Science Center

Kate Wing
Senate Commerce Committee

Third Meeting—February 26 & 27, 1998
Key Largo, Florida

Presenters:

Kimberly Davis
Center for Marine Conservation

Graeme Parks
Marine Resources Assessment Group Americas

Alexander Stone
Reefkeeper International

Guests:

Tom Eagle
NMFS, Office of Protected Resources

Chuck Fowler
NMFS, National Marine Mammal Lab

William Fox, Jr.
Director, NMFS Office of Science and Technology

Eduardo Martinez
NMFS, Southeast Fisheries Science Center

EARTHJUSTICE

ALASKA CALIFORNIA FLORIDA MID-PACIFIC NORTHEAST NORTHERN ROCKIES
NORTHWEST ROCKY MOUNTAIN WASHINGTON, DC INTERNATIONAL

June 4, 2012

Dr. Chris Moore, Executive Director
Mid-Atlantic Fishery Management Council
Suite 201, 800 N. State St
Dover, DE 19901
MSBAmendment14@noaa.gov

RE: Comments on Amendment 14 to the Atlantic Mackerel, Squid, and Butterfish Fishery Management Plan and Draft EIS (No. 20120106)

Dear Dr. Moore:

We are writing on behalf of the Herring Alliance[1] to urge the Mid-Atlantic Council to approve final management measures for inclusion in Amendment 14 to the Atlantic Mackerel, Squid, and Butterfish Fishery Management Plan (MSB FMP) that immediately begin to recover and rebuild river herring and shad populations. The existing fragmented management approach for these species has left river herring and shad with no meaningful regulation in federal waters where they are caught in the MSB fishery, with inadequate catch monitoring, no measures to minimize incidental catch, and no catch limits. This has contributed to the severely depleted status of these keystone species and left them in dire need of conservation and management.[2] The National Marine Fisheries Service is legally obligated to conserve and manage these depleted stocks in federal waters, and the Mid-Atlantic Council should assume leadership in shaping this management by selecting approving the following management measures:

- Add river herring and shad as non-target stocks in the fishery. **Alternative Set 9b-9e.**
- Implement immediate interim measures to reduce and limit incidental catch of river herring and shads until the full suite of conservation and management measures to integrate them as non-target stocks in the fishery is developed and implemented:
    - Implement mortality caps for river herring and shads (alosines) in the mackerel fishery. Modify the proposed caps to reduce the incidental mackerel catch allowable

[1] The Herring Alliance includes 52 organizations representing nearly 2 million individuals. The Herring Alliance is concerned about the status of the Atlantic coast's forage fish (e.g., Atlantic herring, menhaden, and mackerel, river herring and shads, butterfish, and squids), that play a critical role in the food web as prey to a large number of predators, many of which support valuable recreational and commercial fisheries. A current list of Herring Alliance members is attached to this letter. *See* membership at: www.herringalliance.org/alliance-members.

[2] For example, the 2012 river herring stock assessment and peer review conclude that river herring are depleted, that ocean catch is an issue, and that they require fisheries management. Stock Assessment Report No. 12-02 of the ASMFC *Terms of Reference & Advisory Report of the River Herring Stock Assessment Peer Review,* at 8, available at: http://www.asmfc.org/meetings/2012SpringMtg/ShadandRiverHerringManagementBoard_2.pdf. Similarly, the 2007 the American Shad stock assessment and peer review concluded that shad populations have been declining in abundance for years, are not recovering, and are in need of management actions addressing fishing impacts to the species. *See* Stock Assessment Report No. 07-01 of the ASMFC *Terms of Reference and Advisory Report to the American Shad Stock Assessment Peer Review (2007)* at 19, available at: http://www.asmfc.org/.

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to 2000 pounds per trip once the cap is exceeded and directed fishing for mackerel stops. **Combined and Modified Alternatives 6b and 6c, and** Add mortality caps to the list of frameworkable measures. **Alternative 6f.**

- Close river herring hotspots to directed squid and mackerel fishing. Close the "River Herring Protection Areas" identified by the NEFMC in Amendment 5 to the Atlantic Herring FMP. **Modified Alternative 8eMack and Alternative 8eLong**. Also create a mechanism under which the larger "River Herring Monitoring/Avoidance Areas" identified in Amendment 5 could be closed through a future Framework Adjustment. **Modified Alternative 8b.**

- Improve vessel reporting and catch monitoring program for all MSB permits, including 100% observer coverage for Mid-water trawl vessels in the Mackerel fishery, in order to improve precision and accuracy in incidental catch estimates. **Alternatives 1c, Modified Alternative 1d48, Alternatives 1eMack & 1eLong, Modified Alternative1fMack, Alternatives 1gMack & 1gLong; Modified Alternatives 2b, 2c, &2d, Alternatives 2e &2f; Alternatives 3b & 3c, Modified Alternative 3d, Modified Alternative 3j, Alternatives 3l, 3n, & 3o; Modified Alternative 5b4, Modified Alternatives 5c1 & 5c4, Modified Alternative 5d2, Modified Alternatives 5f, Alternative 5g, and Modified Alternative 5h.**
- Include flexible management options, either through the specifications process or through a framework option, to easily adapt management in the future.

Detailed comments are provided below. Where alternatives have been modified, the modification is noted.

Thank you for considering these comments.

Sincerely yours,

/s/ Roger Fleming
Roger Fleming, Attorney
Erica Fuller, Attorney
Earthjustice

Detailed Herring Alliance Comments

1. The Council Must Add River Herring and Shad to the MSB FMP

2.1.9 Alternative Set 9 – Addition of RH/S as "Stocks in the Fishery" in the MSB FMP

The Herring Alliance strongly supports the suite of options in Alternative Set 9b-9e that add all four (4) species of river herring and shad (RH/S) to the MSB FMP and launch an amendment process to add the additional measures necessary to fully integrate blueback herring, alewife, American shad, and hickory shad as stocks in the fishery in the MSB FMP.

The Magnuson-Stevens Act requires federal FMPs to describe the fish stocks involved in a fishery, and NMFS and the councils to manage those stocks in need of conservation and management.[3] FMPs must contain conservation and management measures consistent with the National Standards, including National Standards One and Nine which requires management measures that prevent overfishing and minimize bycatch.[4] The Act also requires annual catch limits (ACLs) and accountability measures (AMs) for *all* stocks in the fishery.[5] The National Standard 1 Guidelines require councils to identify the stocks in the fishery, including the non-targeted stocks that are caught incidentally and retained or discarded at sea.[6] Identification as a stock in the fishery triggers federal annual catch limit (ACL) requirements and the standard

[3] The Magnuson-Stevens Act requires an FMP or an amendment for those fisheries requiring "conservation and management." *See* 16 U.S.C. §§ 1852(h)(1); 1853(a)(2). For purposes of determining which target and non-target stocks require conservation and management, the Act provides a definition of "conservation and management" at 16 U.S.C. § 1802(5). *Id.* at *1, fn 3. This definition should be looked to for guidance in making decision about what stocks belong in a FMP, and refers to rebuilding, restoring, or maintaining "any fishery resource and the marine environment," assuring among other things, a food supply, recreational benefits, and avoiding long-term adverse effects on fishery resources and the marine environment. National Standard 7 guidelines include a set of criteria for determining whether a fishery needs management that tracks this statutory definition and other MSA objectives and requirements, including examination of the condition of the stock or stocks of fish. The criteria also note that "adequate" management by an entity like the ASMFC could be one factor in determining whether a stock should be added to a fishery. In this case, although ASMFC management was noted by NMFS during briefing the *Flaherty v. Bryson* case, the Court did not address it in its opinion because (as even NMFS recognized) the ASMFC plan does not address federal waters.

[4] 16 U.S.C. § 1851(a)(1) & (9).

[5] *Id.* § 1853(a)(15).

[6] 50 CFR § 600.310(d)(3), (4). A "fishery" is defined as "one or more stocks of fish which can be treated as a unit for purposes of conservation and management and which are identified on the basis of geographical, scientific, technical, recreational, and economic characteristics." *Id.* § 1802(13). A "stock of fish" is defined as a "species, subspecies, geographical grouping, or other category of fish capable of management as a unit." *Id.* § 1802(42). The National Standard One Guidelines provide additional guidance on the classification of the stocks in an FMP. *See* 50 C.F.R. § 600.310(d)(1) ("Magnuson-Stevens Act section 303(a)(2) requires that an FMP contain, among other things, a description of the species of fish involved in the fishery. The relevant Council [in the first instance] determines which specific target stocks and/or non-target stocks to include in a fishery" consistent with the Act's requirements. *See Flaherty v. Bryson*, 2012 WL 752323, *14. The regulations define "target stocks" as "stocks that fishers seek to catch for sale or personal use, including 'economic discards' as defined under Magnuson-Stevens Act section 3(9)," and "non-target species" and "non-target stocks" as "fish caught incidentally during the pursuit of target stocks in a fishery, including 'regulatory discards' as defined under the Magnuson-Stevens Act section 3(38). They may or may not be retained for sale or personal use." Non-target species included in a fishery should be identified at the stock level. *Id.* § 600.310(d)(2)-(4).

approach to setting ACLs contained in the National Standard 1 Guidelines.[7] NMFS must review council decisions to ensure that they comply with the Act, and disapprove those that do not. 16 U.S.C. § 1854(a).

The question of which stocks must be included in a fishery management plan was recently addressed in federal court. *See Flaherty v. Bryson*, 2012 WL 752323 at *13 (D.D.C. Mar. 8, 2012) (holding that the Magnuson-Stevens Act requires Councils to prepare an FMP or amendment for any stock of fish that "requires conservation and management."). Councils must make two determinations: (1) which stocks can be treated as a unit for purposes of management, and therefore should be considered a fishery, and (2) which of these fisheries then "require conservation and management." *Id.* at *9. The law does not allow managers to unreasonably delay decision-making regarding the appropriate composition of a fishery given their statutory obligations to ensure that overfishing does not occur. *Id.* at *12. The court also rejected any interpretation of the National Standard One Guideline found at 50 C.F.R. § 600.310(d)(1), as providing the Council with unreviewable discretion to determine what stocks belong in an FMP. *Id.* The Court held that while the Council's role is to name the species to be managed "in the first instance," NMFS has a duty "in the second instance" to ensure an FMP, including the composition of its fishery, satisfies MSA requirements. *Id.* at **13, 14. Moreover, Councils and NMFS cannot limit the stocks they include in a fishery to just those stocks that already happen to be part of an FMP, or those they have officially designated as overfished (or where overfishing is occurring). *Id.* at **12-14.

There is no question that river herring and shad are involved in the mackerel and herring fisheries and are capable of being managed as part of the MSB FMP. *See Flaherty*, 2012 WL at * 12 ("Defendants' conclusory statement that river herring would simply have to wait until a future amendment does not suffice."). First, it is undisputed that river herring and shads are in the Atlantic mackerel fishery because they are caught, kept, landed, and sold in it as well as discarded as bycatch.[8] It is **e**stimated that the mid-water trawl fishery for Atlantic herring and Atlantic mackerel accounts for 71% of the combined incidental catch of river herring and shads; fleet overlap between the small-mesh bottom trawl fisheries account for another approximately 24% of the combined incidental catch.[9] NMFS Observer records show that at-sea fishing vessels may take as much as 20,000 pounds of blueback herring in a single net haul.[10] Because they are involved in this fishery, considering and implementing solutions to the problems of incidental catch in ocean intercept fisheries was the <u>purpose</u> of Amendment 4 and cannot be ignored.[11]

River herring are in dire need of conservation and management. In addition to the science identified in the DEIS that shows that river herring are in need of conservation and management,[12] new information makes conservation and management of these species even more critical. The new benchmark stock assessment for river herring, approved for management

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[7] *See* 50 C.F.R. § 600.310(a), (b)(ii).
[8] *See* Amendment 14 DEIS, Incidental Catch Analysis, at pp. 569-582.
[9] *See* Amendment 14 DEIS, Appendix 2, p. 581.
[10] Haul data from North East Fisheries Observer Program, NMFS; Landings data from NOAA's Annual Commercial Landing Statistics, available at: www.st.nmfs.noaa.gov/st1/commercial/landings/annual_landings.html.
[11] *See* Amendment 14 DEIS, Executive Summary at 9.
[12] *See* Amendment 14 DEIS, § 2.1.9 Addition of RH as "Stocks in the Fishery" in the MSB FMP at pp. 82-89, § 6.2.5 River Herring Stock Status at p. 210, §6.5.6 Description of Candidate Species for Listing Under the ESA, at p. 240.

use by the ASMFC on May 1, 2012, confirms that river herring along the entire Atlantic seaboard are depleted, with many of the river runs barely persisting and near historic lows.[13] Of 24 river stocks for which the stock assessment team was able to characterize current condition, 92% were described as depleted.[14] According to the assessment report "severe declines in [fishery] landings began coastwide in the early 1970s and domestic landings are now a fraction of what they were at their peak having remained at persistently low levels since the mid-1990s."[15] U.S. commercial landings today are down 93% from the 1970's.[16] The peer review panel similarly observed that "[s]tocks of river herring are greatly depleted compared to the early 17th century baseline, as well as compared to that of the late 19th century."[17] It "…concurs with the SASC [stock assessment sub-committee] conclusions that river herring stocks are depleted, that ocean bycatch is an issue, and that recovery will require management on multiple fronts…"[18] For the first time ocean bycatch of river herring was examined in a stock assessment and it concluded that at-sea fisheries are a significant factor in the decline of the species' populations over the last 50 years. [19]

In addition to the benchmark stock assessment, NMFS recently determined that a listing of river herring (blueback herring and alewife) as "threatened" under the Endangered Species Act may be warranted.[20] Finding that NRDC's petition presented "substantial scientific information indicating the petitioned action may be warranted" NMFS initiated a year-long status review. As described in the petition, existing state and federal regulatory mechanisms have proven to be insufficient for river herring. [21] This is due in large part because of the federal/state/regional management framework with shared responsibilities for these migratory fish that has avoided the type of coordinated management necessary to conserve and manage the species. For example, in state waters river herring are managed by the Atlantic States Marine Fisheries Commission ("ASMFC") under Amendment 2 to the Interstate Fishery Management Plan for Shad and River Herring ("Amendment 2"). Regulatory measures drafted by the ASMFC, and implemented through state laws, have proven insufficient in significant part because this interstate compact agency and Amendment 2 have confined the reach of their management plan to state waters only. Although Amendment 2 was drafted in response to dramatic declines in the abundance of river herring, it contains no measures necessary to adequately monitor, limit, and reduce the incidental catch of river herring in federal waters where millions are caught every year by mid-water trawlers targeting Atlantic herring and Atlantic mackerel. Currently, states without an approved Sustainable Fisheries Plan must close their commercial and recreational fisheries; however, state moratoriums on directed fisheries for river herring have been in place for several years in a

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[13] *See* The ASMFC's River Herring Stock Assessment Report No. 12-02, Volume I – Coastwide (May 2012) ("Stock Assessment Report").
[14] Stock Assessment Report - Executive Summary.
[15] Stock Assessment Report - Executive Summary.
[16] Stock Assessment Report - Executive Summary.
[17] *Terms of Reference & Advisory Report of the River Herring Stock Assessment Peer Review* (March 2012)("Peer Review Report"), Introduction at p. 8.
[18] *Id.* at p. 8.
[19] *Id.*
[20] In response to a petition filed by the in response to a petition filed by the National Resources Defense Council (NRDC), NMFS made a 90 day finding that a listing may be warranted. 76 Fed. Reg. 67652 (Nov. 2, 2011). Listing determinations are made solely on the basis of the best scientific and commercial data available, after a full status review, and taking into account all efforts to protect and manage the species. 16 U.S.C. § 1533(b)(1)(A).
[21] NRDC Petition at 78-79.

number of critical states without sufficiently beneficial results.[22] Although the ASMFC is required to coordinate its management measures with NMFS, acting through NMFS to promote the conservation of stocks throughout their ranges, this authority has not been exercised.

Shads are involved in the fishery and in need of conservation and management. Figures used the Mid-Atlantic Council to develop Amendment 14 indicate that on average 120,000 pounds of shad were caught in ocean intercept fisheries from 2006-2010.[23] Of the approximately 600,000 fish, many of them were juveniles that had not spawned.[24] Shads are managed under Amendment 3 to the Interstate Fishery Management Plan for Shad and River herring and, similar to river herring, the ASMFC lists the status of American shad as depleted in accordance with its most recent stock assessment.[25] Despite efforts in state waters, the 2007 stock assessment found that "stocks were at all-time lows and did not appear to be recovering to acceptable levels."[26] The stock assessment also noted that coast-wide declining trends raised flags that ocean mortality was having a serious impact, and the peer review team listed bycatch monitoring as a high priority for future action.[27] No assessments are available for Hickory Shad but as noted in the DEIS, "many runs are likely below historical levels for reasons similar to those discussed below for Atlantic Shad."[28]

With a "stock in the fishery" designation under <u>Alternative Set 9b-9e</u>, the Mid-Atlantic Council would take immediate action to implement incidental catch limits for river herring and shad in the directed fishery for Atlantic mackerel (and Atlantic herring) based on the best available science about what catch level is sustainable and in line with restoration goals. At a minimum, the species meet the definition of non-target stocks because they are caught incidentally in the pursuit of Atlantic mackerel (and Atlantic herring). The trailing amendment, triggered by the choice of Alternative Set 9b-9e, would further develop the required ACLs and other management measures required by law. As the DEIS notes, the law provides for some flexibility in meeting the National Standard 1 requirements and could allow the Mid-Atlantic Council to seek assistance in meeting its legal obligations from the Atlantic States Marine Fisheries Commission.[29] However, as the DEIS makes clear (*see* §§ 6.2.5 and 6.2.6 summary of stock status), the existing federal/state/regional management framework is insufficient to adequately conserve RH/S stocks - the no action Alternative 9a is unacceptable. The designation of these four species as stocks in the fishery is the foundational decision triggering determination of status determination criteria, establishment of ACLs, and

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[22] Connecticut, Massachusetts, Rhode Island, and North Carolina have prohibited harvest for several years without recovery of species' populations. *See* Species Profile: River Herring States and Jurisdictions Work to Develop Sustainable Fisheries Plans for River Herring Management, p.2 available at: http://www.asmfc.org/shadRiverHerring.htm. Sustainable Fishery Plans have been approved for 5 states (Maine, New Hampshire, North Carolina, South Carolina and New York) that met the July 1, 2011 deadline.
[23] *See* Amendment 14 DEIS, §4.1.B at p. 111.
[24] *Id.*
[25] ASMFC. August 2007. Stock Assessment Report No. 07-01 (Supplement) of the Atlantic States Marine Fisheries Commission: American Shad Stock Assessment for Peer Review, Volume 1.
[26] *See* Amendment 14 DEIS, §6.2.6 at p. 213.
[27] *See* ASMFC American Shad Stock Assessment Peer Review Panel, Stock Assessment Report No. 07-01 of the ASMFC, Terms of Reference & Advisory Report to the American Shad Stock Assessment Peer Review. Conducted July 16-20, 2007, Alexandria, Virginia.
[28] Amendment 14 DEIS, s. 6.2.6 at p. 213.
[29] *See* March 18, 2011 Letter submitted by Earthjustice on behalf of the Herring Alliance to Dr. Malcolm Rhodes, Chairman of the ASMFC Shad and River Herring Board.

identification of essential fish habitat, in addition to development of the other required measures necessary to make this FMP comply with the law.[30]

2. Industry's Argument Regarding Stock in the Fishery Designation Is Incorrect

Industry, in their comments on Alternative Set 9, attempts to inject an entirely new (and misguided) legal theory into the discussion of whether river herring and shad should be added as stocks in the fishery of the MSB FMP. *See* June 4, 2012, Letter from Lund's Fisheries Incorporated to Executive Director MAFMC re Amendment 14, at p. 8. In its letter, industry claims that "stock determination criteria" are a "necessary condition for a Council to establish a species as a 'stock in the fishery'" under the National Standard One guidelines, and that the ASMFC stock assessment is fraught with disclaimers preventing its use to assess status. *Id.* This interpretation of the final rule is incorrect for a number of reasons.

As outlined above, the relevant inquiry into *what* species should be added to an FMP is found in the Magnuson-Stevens Act. The Magnuson-Stevens Act requires Councils to first determine the species involved in their fisheries and then prepare an FMP for those that require conservation and management.[31] The Act also requires any plan to specify "objective and measurable criteria for identifying when the fishery to which the plan applies is overfished . . .."[32] The National Standard One Guidelines reinforce this analysis and require stocks involved in the fishery be identified, so they can be added to an FMP, and status determination criteria can be used to prevent overfishing. *See* 50 C.F.R. §§ 600.310 (d)(1) (an FMP must contain a "description of species of fish involved in the fishery"), (d)(4) ("Non-target species may be included in a fishery and, if so, they should be identified at the stock level"), and (e)(2)("status determination criteria (SDC) mean the quantifiable factors, MFMT, OFL, and MSST, or their proxies, that are used to determine if overfishing has occurred, or if the stock complex is overfished."). In that order, stocks are identified as needing conservation and management, added to a plan, and criteria are established (if not already available) to ensure that the plan prevents overfishing.

Alternative Set 9b-9e identifies a two-step process that will make the designation of river herring and shad consistent with the requirements of the Magnuson-Stevens Act stating:

> The Council could add none, one, or any combination of these species as "stocks" in the fishery. Selecting any of the action alternatives would result in the Council immediately beginning another amendment to add all of the provisions 1-15 above to the FMP for any species that is added. Such a process would likely take another 1-2 years to complete, with the development of ACLs/AMs (or ACL alternatives) and essential fish habitat designations taking the most time and being the most substantive of those provisions.

Amendment 14 DEIS, § 5.9.3 at 194. Prior to publication of these alternatives, NMFS determined that an EIS was the appropriate level of NEPA review for this proposed federal

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[30] In addition, the Council should consider any other measures necessary to reduce bycatch, as required by National Standard 9. *See* 16 U.S.C. § 1851(a)(9).
[31] *See* 16 U.S.C. §§ 1852(h)(1), 1853(a)(2); *see also Flaherty v. Bryson*, 2012 WL 752323 at * 11, 12.
[32] *Id.* at § 1853(a)(10).

action, 74 Fed. Reg. 68577 (Dec. 28, 2009), and approved this DEIS, NOAA Award No. NA10NMF4410009.

"Need" does not equate to a prerequisite. Although no citation was provided, the preamble to the final rule the industry refers to simply states that "'Stocks in the fishery' need status determination criteria, other reference points, ACL mechanisms and AMs." *Final Rule*, 74 Fed. Reg. 3178 at 3179 (Jan. 16, 2009). No one disputes that stocks ultimately "need" these criteria to prevent overfishing; however, nothing in the Act, the Final Rule, or the regulations interpreting National Standard One contemplates the necessity of status determination criteria prior to adding a stock in the fishery. On the contrary, the regulations contemplate the order discussed above and even the use of proxies (if necessary), based on the best scientific information available, for reference points not yet identified including proxies for MSY, F_{MSY} and B_{MSY}.[33] A plain reading of the regulation does not support industry's distorted view.

Finally, the ASMFC's river herring stock assessment has now been peer-reviewed and approved by the ASMFC Shad & River Herring Management Board for management use and it cannot be used as an excuse not to manage these species. Under the scenario outlined in Alternative set 9b-9e, the Council has 1-2 years to complete the trailing amendment and identify the SDC. Moreover, if necessary, the Mid-Atlantic Council could use proxies for those values as it has used in other managed species.[34] This stock assessment report represents best available science and while it did not provide reference points for the coastwide river herring complex, it provided ample evidence that river herring and shad are in need of conservation and management, and thus should be added to a plan.

3. <u>Until River Herring and Shad are Fully Integrated into the FMP the Council Must Establish a River Herring and Shad Catch Cap</u>

2.1.6 Alternative Set 6: Mortality Caps

The Herring Alliance strongly supports interim measures that:

- **<u>Combine and modify Alternatives 6b and 6c</u> to implement a <u>single mortality cap</u> for all river herring and shad species (alosines) in the mackerel fishery (closing the mackerel fishery when the cap is exceeded), and modifying the incidental catch allowance of Atlantic mackerel after the fishery is closed to <u>2,000</u> pounds.**
- **Implement <u>Alternative 6f</u> to allow mortality caps to be added to the list of measures that can be frameworked.**

The addition of river herring and shad as stocks in the MSB fishery through Amendment 14 will not sufficiently reduce incidental catch while the Mid-Atlantic Council develops a trailing amendment that fully integrates them into the MSB FMP; therefore, the Herring Alliance supports a mortality cap that immediately reduces and limits the at-sea mortality of these depleted species. This interim catch cap should be effective in 2013, and remain in effect until

[33] *See* 50 C.F.R. § 600.310(e)(1)(iv) ("Where this uncertainty cannot be directly calculated, such as when proxies are used, then a proxy for the uncertainty itself should be established based on the best scientific information, ...").
[34] *See* May 3, 2011 Staff Memorandum regarding 2012 Atlantic Mackerel, *Illex*, *Loligo*, and Butterfish OFL/ABC Recommendations; *see also* May 23, 2012, SSC recommendations setting OFL proxy for butterfish.

replaced by ACLs or similar conservation measures under the MSB FMP once the river herring and shads are fully integrated in the FMP.

A combination and modification of Alternatives 6b and 6c could function as a single mortality cap in the mackerel fishery as more precise estimates of incidental catch are obtained with increased observer coverage and more robust sampling. Currently, the overlap of the Atlantic mackerel and Atlantic herring mid-water trawl fishery complicates the implementation of a mortality cap on the mackerel fishery alone because if the mortality cap operated to shut the Atlantic mackerel fishery down, Atlantic herring fishing could continue in the same Quarter and same Area allowing incidental catch of river herring and shad to continue. Further, the current mackerel incidental allowance of 20,000 pounds proposed under 6b and 6c is far too liberal to deter directed fishing and minimize fishing effort should a mortality cap on RH/S be reached. **This alternative set, and others below, should be modified to be consistent with the Atlantic Herring FMP which uses a 2,000 pound incidental Atlantic herring limit to define, deter, and close directed herring fishing, including for the purposes of enforcing herring ACL's and sub-ACLs.**[35] The 2,000 pound incidental Atlantic herring limit, implemented after a herring management area closes to enforce sub-ACLs, has proven effective. For example, when the Atlantic herring Area 2 closed on February 20th of this year, mackerel fishing taking place in the same area leveled off.[36] Thus, a combined and modified cap would improve the effectiveness of the cap and ensure that vessels cannot circumvent a cap by simply declaring into another fishery. The modification from the current incidental allowance of 20,000 pounds of mackerel to 2,000 pounds would more effectively ensure that once the cap is reached that directed mackerel fishing stops, that mackerel discarding does not continue or increase, and that river herring and shad removals cease if a cap is reached by lowering the incidental trip allowance of mackerel that can be fished for, possessed or retained. **Further, the implementing language for that incidental limit should be consistent with the language in the Atlantic Herring FMP such that the 2,000 pound incidental limit would apply to vessels "fishing for, catching, possessing, transferring, or landing more than 2,000 lb."**[37]

Alternative 6f adds mortality caps to the list of frameworkable measures and is appropriate in order to allow for a catch cap on the squid or butterfish fisheries (should a directed butterfish fishery become higher than the current level) as data improves through catch monitoring and sampling and as the need arises. Currently the MSB FMP does not list incidental catch caps as frameworkable measures, and this alternative would facilitate implementation should new data reveal a more significant alosine catch in any of the target fisheries.

Note: The Council Can Not Rely on a Voluntary Bycatch Avoidance Program such as the SFC/SMAST/DMF Project described in Alternative 4f to Satisfy its National Standard 9

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[35] The 2,000 lb limit used to close the directed herring fishery was approved in Amendment 4 to the Atlantic Herring FMP as the sole proactive accountability measure for preventing ACL overages and is described in Amendment 4 at: http://www.nefmc.org/herring/planamen/final_a4/AM%204%20DOCUMENT%20FORMAL%20SUBMISSION_100423.pdf, at p.29.

[36] *See* NERO. Weekly Quota and Landing Report available at: http://www.nero.noaa.gov/ro/fso/reports/reports_frame.htm.

[37] Those regulations state that upon closure of the directed Atlantic Herring fishery, NMFS shall "prohibit herring vessel permit holders from fishing for, catching, possessing, transferring, or landing more than 2,000 lb (907.2 kg) of herring per calendar day in or from the specified management area for the remainder of the closure period." *See* 77 Fed. Reg. 10668 (Feb. 23, 2012).

Obligation to Minimize Bycatch. This University based voluntary program is inappropriate as a regulatory measure and would be ineffective without a mortality cap. This alternative contemplates a "stand-alone approach for river herring bycatch" and should be removed from consideration. There are simply no meaningful incentives to avoid bycatch through the program without a cap. Any positive results from the program to date are the result of the incentive to avoid meaningful regulation, and will likely disappear as soon as Amendments 14 and Amendment 5 to the Atlantic Herring FMP pass. The bycatch avoidance program for the Atlantic scallop fishery is successful at reducing bycatch only because there is a yellowtail flounder cap that scallop fishermen must avoid to continue fishing.

4. Until River Herring are Fully Integrated into the FMP the Council Must Implement Hotspot Closures

2.1.8 Alternative Set 8: Hotspot Restrictions

As interim measures the Herring Alliance supports the following alternatives:

- **Modified Alternative 8b: Make implementing the hotspot requirements of NEFMC's Amendment 5 to the Atlantic Herring Plan for Mackerel/longfin squid vessels frameworkable. Modified to provide a mechanism through which the Mid-Atlantic Council could expand the hotspots identified in Amendment 5 to encompass the larger River Herring Monitoring/Avoidance Areas, or adjust hotspot requirements to achieve consistency with the Atlantic Herring FMP through a Framework Adjustment.**
- **Modified Alternative 8eMack: Vessels possessing a federal mackerel permit would not be able to fish for, catch, possess, retain, transfer, or land[38] more than an incidental level of fish (2,000 pounds of mackerel) while in a River Herring Protection Area unless no mesh smaller than 5.5 inches is onboard the vessel.**
- **Alternative 8eLong: Vessels possessing a federal moratorium longfin squid permit would not be able to fish for, catch, possess, retain, transfer, or land[39] more than an incidental level of fish (2,500 pounds longfin squid) while in a River Herring Protection Area unless no mesh smaller than 5.5 inches is onboard the vessel.**

As an interim measure, the Herring Alliance supports a Modified Alternative 8b, that closes the temporal and spatial protection areas identified as having high rates of river herring or shad bycatch ("hot-spots") to directed mackerel and squid fishing as an additional tool that should be deployed to reduce catch of river herring and shad as an interim measure (i.e., until these stocks are fully integrated as stocks in the fishery), in addition to the immediate implementation of a mortality cap. The protection areas identified by the New England Council's Herring Plan Development Team (PDT) are small, and the Mid-Atlantic Council's Fishery Management Action Team (FMAT) analyses indicate that protection of small areas alone may not be adequate to effectively reduce catch, or, may result in a fishing effort shift that could increase river herring and shad morality. However, coupled with a mortality cap, and based on the PDT's analysis of

[38] Proposed revisions make this measure more consistent with the incidental catch allowance regulations in the Atlantic Herring FMP as previously described.
[39] *Id.*

the same provisions in Amendment 5, the river herring protection areas will provide a positive conservation benefit until management measures for river herrings and shads are fully integrated into the FMP (as stocks in the fishery). As more data becomes available, through increased monitoring and reporting, the Council should expand the protection areas as necessary through a framework action and give consideration to the larger areas identified in Amendment 5 and described in Amendment 14 as "River Herring Monitoring/Avoidance Areas."[40]

For all of the reasons described in the sections on mortality caps and observer coverage, the Herring Alliance also supports a Modified 8eMack which reduces the incidental level of mackerel a federal permit would be able to fish for, catch, possess, retain, transfer or land in a River Herring Protection Area from 20,000 pounds to 2,000 pounds, and Alternative 8eLong, as modified to reduce the total catch of river herring and shad at sea. The Council should carefully monitor the effectiveness of the hotspot regime for squid vessels to determine if any similar adjustments are warranted.

5. Require 100% observer coverage for Mid-water trawl vessels in the Mackerel fishery

2.1.5 Alternative Set 5: At-Sea Observer Coverage Requirements

To achieve the stated goals of Amendment 14 and ensure the effectiveness of the other alternatives the Herring Alliance supports the following alternatives, while opposing a sunset clause for increased observer coverage levels implemented through Amendment 14 and the waiver associated with Alternative 5f and discussed on page 160:

- **Modified Alternative 5b4: Require 100% of MWT mackerel trips by federal vessels intending to fish for, catch, possess, retain, transfer, or land over 2,000 pounds of mackerel to carry observers. The NEFSC would assign coverage based on pre-trip notifications. This alternative would be modified such that vessels would not be able to fish for, catch, possess, retain, transfer, or land more than 2,000 pounds of mackerel unless they had notified their intent to retain more than 2,000 pounds of mackerel.**
- **Modified Alternative 5c1: Require 25% of SMBT trips by Tier 3 limited access mackerel vessels intending to fish for, catch, possess, retain, transfer, or land over 2,000 pounds of mackerel to carry observers. The NEFSC would assign coverage based on pre-trip notifications. Vessels would not be able to fish for, catch, possess, retain, transfer, or land more than 2,000 pounds of mackerel unless they had notified their intent to retain more than 2,000 pounds of mackerel.**
- **Modified Alternative 5c4: Require 100% of SMBT (<3.5 in) mackerel trips by Tier 1 and Tier 2 limited access mackerel vessels intending to fish for, catch, possess, retain, transfer, or land over 2,000 pounds of mackerel to carry observers. The NEFSC would assign coverage based on pre-trip notifications. Vessels would not be able to fish for, catch, possess, retain, transfer, or land more than 2,000 pounds of mackerel unless they had notified their intent to retain more than 2,000 pounds of mackerel.**

[40] *See* Amendment 14 DEIS, § 2.1.8 at pp. 72-77.

- **<u>Modified Alternative 5d2</u>: Require 50% of SMBT (<3.5 in) longfin squid trips by <u>major longfin squid moratorium permitted</u> vessels intending to retain over 2,500 pounds of longfin squid to carry observers. The NEFSC would assign coverage based on pre-trip notifications. Vessels would not be able to retain more than 2,500 pounds of longfin squid unless they had notified their intent to retain more than 2,500 pounds of longfin squid.**
- **<u>Modified Alternative 5f</u>: Vessels would have to pay for observers to meet any observer coverage goals adopted by the Council that are greater than existing sea day allocations assigned through the sea day allocation process (already implemented in other fisheries). NEFSC would accredit observers and vessels would have to contract and pay observers. <u>Modified</u> to prohibit waivers, especially without explicit limits and accountability measures to ensure that waivers do not undermine the target coverage level. <u>Modified</u> to require States receive full provider certification in order to be providers.**
- **<u>Modified Alternative 5h</u>: Requires reevaluation of coverage requirements after 2 years to determine if incidental catch rates <u>should be adjusted - up or down based on circumstances</u>.**

Monitoring an industrial fishery is a mandatory precondition of access to millions of pounds of public resources. Congress intended that there be both "limits" and "accountability" in fisheries, with the ultimate goal of "protect[ing], restor[ing], and promot[ing] the long-term health and stability of the fishery." 16 U.S.C. § 1853(a)(1)(A). In order to achieve accountability, the Magnuson-Stevens Act requires that FMPs include monitoring and reporting measures necessary to track retained catch and discarded bycatch, including a standardized reporting methodology to assess the amount and type of bycatch occurring in the fishery). *See* 16 U.S.C. §§ 1853(a)(5), (a)(11). Adequate accountability measures are also vital to fulfilling National Standard One's mandate to prevent overfishing, *id*., and National Standard Nine's requirement that "[c]onservation and management measures shall, to the extent practicable, (A) minimize bycatch and (B) to the extent bycatch cannot be avoided, minimize the mortality of such bycatch," *id.* § 1851(a)(9). These directives are critical to the effective implementation of Amendment 14 which depends upon the accurate measurement of the amounts of river herring and shad caught and discarded in this fishery[41] and if this fishery cannot be monitored adequately, it should not have access to this national public resource.

In order to achieve the stated goals of Amendment 14, and ensure the effectiveness of the recommended alternatives, it is imperative that the Mid-Atlantic Council increase observer coverage and ensure that observers have access to all catch.[42] Adequate monitoring and bycatch measures are vital to ensuring that overfishing is prevented. *See e.g.*, *Flaherty*, 2012 WL at *16 ("to meet their responsibility to ensure compliance with the National Standards, Defendants must demonstrate that they have evaluated whether the FMP or amendment minimized bycatch to the extent practicable."). The at-sea observer program, which obtains data for both kept and discarded catch, is critical to understanding total catch of river herring and shads, and must be

[41] By themselves vessel catch reports have been found unreliable. *See Conservation Law Foundation v. Evans*, 209 F. Supp. 2d 1 at 13, n. 25 (D.D.C. 2001) (noting unlawful underreporting of bycatch) (Kessler, J.). In addition, dealers have no possible way of knowing the amount of river herring and other species discarded at sea as bycatch because they only see and buy what is brought to their facility.
[42] *See* http://www.mafmc.org/fmp/current/SMB/River_Herring_Letters.pdf.

prioritized by the Council. In contrast to at-sea observers, portside sampling only obtains information for the catch that is retained, and therefore misses an important part of the equation. Only those alternatives which increase the accuracy and timeliness of vessel and dealer reporting (discussed below), coupled with management measures that greatly improve the accuracy and precision of third-party (i.e. observer) incidental catch estimates should be selected in Amendment 14. In addition, these alternatives should be consistent with Amendment 5 in order to avoid discrepancies that would cause significant difficulties in implementation or allow for fishing effort to avoid more robust monitoring in one of the FMP's by selectively declaring into the other.

In order to properly cover mid-water trawl mackerel trips, 100% observer coverage is necessary and the Herring Alliance supports a Modified Alternative 5b4. Mid-water trawl vessels account for 75.7% of river herring incidental catch and 41.8% of shad incidental catch,[43] and are responsible for the majority of mackerel landings, accounting for 62% of landings in 2010.[44] According to information presented in Amendment 11 to the MSB FMP, there are 15 mid-water trawl vessels that are eligible for the mackerel limited access program (13 in Tier 1 and 2 in Tier 2).[45] Given the high volume nature of these vessels, and the fact that significant shad and river herring catch events may be infrequent (but events can be large when they occur), 100% coverage is necessary for an accurate accounting of incidental catch. In addition, mid-water trawl vessels are in the top permit tiers of the Atlantic herring limited access fishery, for which the New England Council is considering 100% observer coverage. Given the overlap in the mid-water trawl fisheries for Atlantic herring and Atlantic mackerel, observer coverage levels should be consistent between the FMPs.[46] Further, for essentially the same reasons stated above in our explanation for the need to adjust the mackerel incidental limit downward from 20,000 pounds to 2,000 pounds to ensure the integrity of a mortality cap, the same adjustment should be made for this alternative. Absent this modification, it is possible that a significant amount of directed mackerel fishing could take place outside the scope of a 100% observer coverage requirement if the vessels simply declared an intent to fish in the herring fishery (and if the Herring FMP did not have a similar coverage requirement). Limiting vessels to 20,000 pounds of mackerel will not sufficiently deter directed fishing by the large MWT vessels which comprise the most significant component of the herring-mackerel fishery overlap.

In order to properly cover small-mesh bottom trawl mackerel trips, the Herring Alliance supports a hybridization of Modified Alternatives 5c1 and 5c4 with a tier approach to assigning different coverage levels to small mesh bottom trawl vessels (SMBT). Small-mesh bottom trawls are believed to contribute to 23.7% and 25.6% of river herring and shad incidental respectively; therefore, it is important to improve observer coverage in this fleet to achieve accuracy and precision in incidental catch estimates. Because industry funding will be necessary to achieve coverage levels above the status quo, it is important to distribute the observer cost burden equitably among fishery participants. For the mackerel limited access program, 10 SMBT vessels are eligible for Tier 1, and 19 are eligible for Tier 2.[47] Neither Tier 1 nor Tier 2 vessels

[43] *See* Amendment 14 DEIS, Appendix 2 at p. 581.
[44] *See* Amendment 14 DEIS, Table 29 at p. 247.
[45] *See* MAFMC. Amendment 11 to the Atlantic Mackerel, Squid, and Butterfish Fishery Management Plan (May 2011), Tables 94-96 at pp. 447-448.
[46] *See* Amendment 14 DEIS, Appendix 2 at p. 574.
[47] *Id.*

are capped by a percentage of the quota, with no trip limits for Tier 1 vessels. For Tier 3, however, 138 vessels qualify,[48] and this tier is capped at 3% of the annual quota. Additionally, the average length of a Tier 3 vessel is 65 ft., compared to 78 ft. for Tier 2 and 110 ft. for Tier 1,[49] likely making the observer costs significantly more burdensome for vessels in Tier 3 relative to their daily operating costs. 100% coverage on Tier 1 and Tier 2 SMBT vessels engaging in directed mackerel fishing represents a manageable objective that will cover the majority of the catch by this gear type without undue burden on small boats or the observer infrastructure.

The Herring Alliance supports a Modified Alternative 5d2, which calls for 50% observer coverage on the major longfin squid vessels. Currently only 3.5% of longfin squid catches by weight have been observed (2006-2010),[50] contributing to great uncertainty in the shad and river herring incidental catch estimates for this fishery. As described above, small-mesh bottom trawls (SMBT) contribute significantly to shad and river herring incidental catch, and higher levels of at-sea observer coverage will be needed for the Northeast's SMBT fleet in order to obtain reasonably accurate and precise estimates of this catch. Coverage must be equitably distributed among vessels according to their activity in the fishery. While there are approximately 400 vessels that hold moratorium permits, an average of only 103 vessels have been major vessels in this fishery in the last 5 years, and these major vessels account for around 95% of the annual landings.[51] The Mid-Atlantic Council should identify these approximately 100 most active longfin squid vessels (or outline procedures whereby they would self-identify) in advance of the fishing year so that they are clearly and explicitly assigned to the 50% observer coverage bin for that year. Criteria that could be utilized to sort and assign the fleet in this manner include an analysis of recent catch to identify whether these vessels vary significantly from year to year and/or whether there is a logical annual landings threshold where the line can be drawn. Alternatively, the Council could identify a reasonable and typical annual threshold for landings that makes it likely they will capture the most active vessels (i.e. those which collectively catch 95% of the longfin squid) and require that vessels wishing to land over that number for the year must declare into the higher observer coverage program. While herring-mackerel fishery overlap and consistency concerns are likely not as acute for squid vessels, if the Council's intent is to ensure observer coverage on a target percentage of directed squid fishing trips, it may want to consider revising this alternative to reflect the previously noted language used in the Atlantic herring FMP to define directed fishing ("fishing for, catching, possessing, transferring, or landing"), and which has been proven effective.

With respect to Modified Alternative 5f, the Herring Alliance opposes the addition of a sunset clause for any increased observer coverage levels implemented through Amendment 14. The alternatives already contemplate a future review of the observer requirements by the Council in Alternative 5h and the Service has indicated that it may take time for an expanded observer program to be designed once these fisheries are fully established on the water. It would be unfortunate for a sunset clause to kick in prior to a full observer program, and prior to gaining the necessary data that the coverage was intended to obtain. The Herring Alliance also opposes the issuance of waivers (as discussed under Alternative 5f on page 160), under which a vessel or

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[48] *Id.*
[49] *See* MAFMC Amendment 11 DEIS, Table 82 at p. 435.
[50] *See* Amendment 14 DEIS at p. 147.
[51] *See* MAFMC April 2012 Staff Loligo AP Informational Document, at Tables 4 and 6, available at: http://www.mafmc.org/meeting_materials/SSC/2012-05/Loligo%20APInfo-2012.pdf.

trip assigned an observer would be allowed to sail without an observer. A robust at-sea monitoring program on vessels of this size, gear type and fishing power, with a known potential for infrequent but destructive bycatch events, must have 100% coverage. 100% coverage must mean just that: 100%. A blanket provision allowing the unlimited issuance of waivers with no backstops or other accountability measures is likely to seriously undermine any 100% coverage requirement or other target coverage level.

On the issue of review, the Herring Alliance supports a Modified Alternative 5h that requires reevaluation in 2 years to determine whether observer coverage rates should be adjusted; however, as written Alternative 5h is too restrictive and hints at foregone conclusions. The Herring Alliance supports a comprehensive analysis of whether coverage levels should be adjusted in general, including whether they need to be increased.

Note: Without maximized retention (not considered in Amendment 14) the Herring Alliance cannot support portside sampling (Alternative Set 4) for deriving estimates on river herring and shad incidental catch. Taken alone, it would miss at-sea discards and provide insufficient data.

6. Additional Measures to Improve At-Sea Sampling

2.1.3 Alternative Set 3: Additional At-Sea Observation Optimization Measures

In order to successfully reduce total catch of river herring and shad and achieve the goals of Amendment 14, it must have reliable total catch estimates. Estimates of the amount of catch are dependent upon accurate estimates because total catch is used to scale up from the amounts observed in samples. **To ensure reliable total catch estimates of river herring and shad, and achieve the goals of Amendment 14 the Herring Alliance supports the alternatives listed below:**

- **Alternative 3b: Require the following reasonable assistance measures: provision of a safe sampling station; help with measuring decks, codends, and holding bins; help with bycatch collection; and help with basket sample collection by crew on vessels with mackerel limited access and/or longfin squid/Butterfish moratorium permits.**
- **Alternative 3c: Require vessel operators to provide observers notice when pumping/haul-back occurs on vessels with mackerel limited access and/or longfin squid moratorium permits.**
- **Modified Alternative 3d: When observers are deployed on trips involving more than one vessel, observers would be required on any vessel taking on fish wherever/whenever possible on vessels with mackerel limited access and/or longfin squid moratorium permits. Modified to remove the whenever/wherever language.**
- **Modified Alternative 3j: Modified to apply "Closed Area I" (CA1) requirements to all mackerel limited access and longfin squid moratorium permitted vessels intending to fish for, catch, possess, retain, transfer, or land over 2,000 pounds of mackerel. These requirements are currently in force in the Atlantic herring fishery for mid-water trawl vessels intending to fish in Groundfish Closed Area 1. This alternative would require that all fish be brought aboard for observer sampling with exceptions made for safety, mechanical failure, or spiny dog fish clogging the pump.**

- **<u>Alternative 3l</u>*: Related to 3j, for mackerel limited access permitted vessels, NMFS would track the number of slippage events. Once a cap of 10 slippage events (adjustable via specifications) occur in any given year for notified and observed mackerel trips then subsequent slippage events on any notified and observed mackerel trip would result in trip termination for the rest of that year. The goal is to minimize slippage events.**
- **<u>Alternative 3n</u>: For longfin squid moratorium permitted vessels, NMFS would track the number of slippage events. Once a cap of 10 slippage events (adjustable via specifications) occur in any given trimester for notified and observed longfin squid trips then subsequent slippage events on any notified and observed longfin squid trip would result in trip termination for the rest of that trimester. The goal is to maximize sampling of catch on observed trips and to discourage and minimize slippage events.**
- **<u>Alternative 3o</u>: For mackerel and/or longfin squid permitted vessels, if a trip is terminated within 24 hours because of any of the anti-slippage provisions (3g, 3h, 3k-3n), then the relevant vessel would have to take an observer on its next trip.**

In <u>Alternative 3d</u>, the language "wherever/whenever possible" should be removed. Should the Council choose 100% monitoring, this language would provide a loophole to such a requirement and frustrate the goal of more accurate observer data. The majority of "Fish NK" (or fish unknown) records are associated with fish that are pumped to the paired trawl vessel not carrying the observer.[52] Between July 2009 and June 2010 over 5.7 million pounds of catch was recorded as Fish NK in the observer database.[53] The Council should be clear and explicit that any pair trawl trip assigned observer coverage will require an observer on each platform, and should prohibit the taking of fish on a vessel without an observer.

<u>Modified Alternative 3j</u> should clarify that consistent with the current CA1 sampling regulations, operational discards must be brought aboard for sampling, and may only be dumped under one of the other three allowable exceptions, and therefore if dumped would be subject to the accountability requirements outlined in 31, 3n and 3o. **Vessels would be permitted to discard (release) un-sampled catch under those limited exceptions, and only those circumstances. Further, consistent with these CA1 rules, and in order to prevent any abuse of those limited exceptions, an accountability framework should be layered over the three exceptions as outlined below (Alternatives 3l, 3n and 3o).** NMFS has acknowledged that accurate catch composition records cannot be obtained for dumped catch (including operational discards), that there are safe and operationally feasible ways to get all catch aboard for sampling (including operational discards), and that issues such as stratification of catch in the cod-end or the strainer-like effect of the pump-intake grate raise serious questions about the composition of operational discards.[54] The implementing language should also be revised so that the measures apply to trips "fishing for, catching, possessing, transferring, or landing" the specified amount of the target species to be consistent with the Atlantic Herring FMP.

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[52] *See* Amendment 14 DEIS, Appendix 5 at p.662.
[53] *See* NEFSC. Standardized Bycatch Reporting Methodology Annual Discard Report 201: Section 2 at p. 189, available at: http://www.nefsc.noaa.gov/fsb/SBRM/2011/SBRM_Annual_Discard_Rpt_2011_Section2.pdf.
[54] *See* Final Rule entitled Fisheries of the Northeastern United States; Discard Provision for Herring Midwater Trawl Vessels Fishing in Groundfish Closed Area I, 75 Fed. Reg. 73979 (Nov. 30, 2010).

To minimize slippage events, the Herring Alliance supports Alternative 3l to cap the number of slippage events per year in the mackerel fishery at 10. From 2006-2010, 26% of hauls on observed mackerel trips had some unobserved catch.[55] In contrast, vessels fishing under Closed Area I (CA1) regulations in the Atlantic herring fishery had no observed slippage events recorded in 2010.[56] However, prior to the implementation of the CA1 rules, the Atlantic herring fishery had an observed slippage rate of 35%.[57] **This reduction in dumping in the herring fishery clearly demonstrates the effectiveness of the CAI rules.** It is important to note, however, that the effectiveness of the CAI regulations is due to the accountability measures tied to the dumping exceptions, which requires a vessel to stop fishing and exit CA1 if it releases an un-sampled net. The Mid-Atlantic Council should select final measures in Amendment 14 that replicate the CA1 regulations. Given the three exceptions provided for under Alternative 3j, permitting 10 slippage events before slippage results in trip termination provides a reasonable balance that will deter slippage without unduly penalizing those involved.

To minimize slippage events, the number of slippage events in the longfin squid fishery should also be capped at 10 events consistent with Alternative 3n and implemented in conjunction with Alternative 3j. On observed longfin squid trips, an average of 9% to 14 % of hauls are not seen and sampled by observers.[58] As discussed above, an accountability measure is an important component to the CA1 sampling requirements to ensure compliance, and an allowance of 10 slippage events per trimester before trip termination is implemented is appropriate for deterring slippage.

Alternative 3o, which requires a vessel which has had its trip terminated within 24 hours because of an of the anti-slippage provisions to take an observer on its next trip, is necessary if observer coverage levels are not high enough to effectively deter vessels from dumping unwanted catch or catch they suspect contains bycatch on the rare occasions they are observed. If there is a high likelihood the next trip will not be observed, vessels may not be sufficiently discouraged from dumping early in a trip by the trip termination requirement.

Note: The Herring Alliance is opposed to the Released Catch Affidavits as discussed in Alternative 3e because the Mid-Atlantic Council does not track the cause of the slippage and in and this alternative will not ensure results.

7. Additional Measures to Improve At-Sea Sampling

2.1.1 Alternative Set 1: Additional Vessel Reporting Measures

[55] *See* Amendment 14 DEIS at p.130.
[56] *See* Amendment 14 DEIS, Appendix 5 at p. 658.
[57] *See* MEFMC Herring Committee Meeting, July 27, 2010. In 2009, 35% of observed Atlantic herring hauls were completely or partially released during 2009, with over a thousand metric tons released. With only 1/5 of the trips (in 2009) observed, the total quantity of fish released in much high than actually observed.
[58] *See* Amendment 14 DEIS at p.130 ("From 2006-2010 approximately 9% of hauls on observed longfin squid trips [] and 26% of hauls on observed mackerel trips [] had some unobserved catch."). *See also* SSC materials from Mary 2012 which suggests that slippage has increased from previous estimates under the Butterfish Bycatch Program to 14%, available at: http://www.mafmc.org/meeting_materials/SSC/2012-05/3-2011-Butterfish-Cap-Report(May%202012).pdf.

To improve quota monitoring and enforcement the Herring Alliance strongly supports the following Alternatives:

- **<u>Alternative 1c</u>: Institute weekly vessel trip reporting (VTR) for all MSB permits (Mackerel, longfin quid//Butterfish, *Illex*) so as to facilitate quota monitoring (directed landings and/or incidental mortality cap if applicable) and cross checking with other data sources.**
- **<u>Modified Alternative 1d 48</u>: Require <u>all</u> mackerel trips <u>give 48 hour pre-trip notification to NMFS to fish for, catch, possess, retain, transfer, or land</u> more than <u>2,000</u> pounds of mackerel so as to facilitate observer placement.**
- **<u>Alternatives 1eMack & 1eLong</u>: require VMS for limited access mackerel vessels and longfin squid/butterfish moratorium vessels;**
- **<u>Modified Alternative 1fMack</u> – requires daily VMS reporting of catch by limited access mackerel vessels. <u>Modified</u> to make this frameworkable in the event that a mortality cap becomes necessary in the squid fishery.**
- **<u>Modified Alternative 1f Long</u>: Require daily VMS reporting of catch by longfin squid moratorium permits so as to facilitate monitoring (directed and/or incidental catch) and cross checking with other data sources. Requiring VMX and trip declarations would be a prerequisite for this alternative. <u>Modified</u> to make frameworkable in the event that a mortality cap becomes necessary in the squid fishery.**
- **<u>Modified Alternatives 1gMack and 1g Long:</u> require 6 hour pre-landing notification via VMS to land more than <u>2,000</u> pounds of mackerel or more than 2,500 pounds of longfin squid, which would facilitate quota monitoring, enforcement, and/or portside monitoring.**

Weekly VTR for all MSB permits (mackerel, longfin squid/butterfish, Illex) will facilitate quota monitoring (directed landings and/or incidental mortality cap if applicable) and cross checking with other data sources. 48 hour pre-trip notification to NMFS for those mid-water trawl and Tier 1 and 2 SMBT vessels intending to retain, possess, or transfer more than 2,000 pounds of mackerel is critical for observer placement and consistent with the recommended alternatives for observer coverage above. Because the VMS on limited access mackerel vessels and longfin squid/butterfish moratorium vessels, as well as daily reporting of catch will also facilitate monitoring (directed and/or incidental) and cross checking of other data sources. As noted in the DEIS a great majority of these limited access mackerel and squid/butterfish moratorium permitted vessels are already equipped with VMS.[59] Six (6) hour pre-landing notification via VMS to land more than 20,000 pounds of mackerel or more than 2,500 pounds of longfin squid, will also facilitate quota monitoring, enforcement, and/or portside monitoring. In addition, the Atlantic herring FMP already mandates reporting measures identical to or very similar to each of the alternatives listed above, making these proposed measures necessary for improved consistency between the two plans.

2.1.2 Alternative Set 2: Additional Dealer Reporting Measures

[59] *See* Amendment 14 DEIS at pp. 292, 294.

To precisely estimate incidental catch of RH/S in these fisheries the Herring Alliance supports the following alternatives:

- **<u>Modified Alternative 2b</u>: Require federally permitted MSB dealers to obtain vessel representative confirmation of SAFIS transaction records for mackerel landings over <u>2,000</u> pounds, Illex landings over 10,000 pounds, and longfin squid landings over 2,500 pounds to catch data errors at first point of entry; and**
- **<u>Modified Alternatives 2c and 2d</u>: Require that federally permitted SMB dealers weigh all landings related to mackerel transactions over <u>2,000</u> pounds. If dealers do not sort by species they would need to document in dealer applications <u>or</u> with each transaction so long as the proper methodology was documented.**
- **<u>Alternative 2e</u>: Require that federally permitted SMB dealers weigh all landings related to longfin squid transactions over 2,500. If dealers do not sort by species, they would need to document in dealer applications how they estimate relative compositions of a mixed catch.**
- **<u>Alternative 2f</u>: Require that federally permitted SMB dealers weigh all landings related to longfin squid transactions over 2,500 pounds. If dealers do not sort by species, they would need to document with each transaction how they estimate relative compositions of a mixed catch.**

Standardizing the methods by which dealers are required to weigh all catch and confirm the amount of fish landed will aid in better overall estimates of catch, in addition to being essential for ensuring that directed quotas are not exceeded. More accurate data on landings will also aid in the monitoring of a mortality cap or in achieving the objective of better catch and bycatch estimates of river herring and shad. As the DEIS points out, "accurate monitoring of the target species can be as important as determining the encounter rates of [river herring and shad]" in the determination of river herring and shad catch estimates, due to the use of discard to kept ratios or other bycatch/incidental catch extrapolations.[60]

Dealer or vessel self-reporting of unverifiable, unstandardized "hail" weights or visually-based volumetric estimates are inadequate and unacceptable. They present far too much opportunity for deliberate or accidental misreporting, and offer no opportunity for third-party observers, port samplers, or law enforcement personnel to verify that accurate, complete and honest catch weights are being reported. The Herring Alliance views this suite of alternatives (<u>Modified Alternatives 2b- 2d, Alternative 2e, and Alternative 2f</u>) as working together to minimize dealer reporting while providing for increased efficiency and flexibility. Dealers that do not sort by species could document their method for estimating the composition of a mixed catch. Consistent with previous alternative chosen, the modification of Alternative 2b, 2c, and 2d will decrease the incidental landings limit of mackerel from 20,000 pounds to 2,000 as the trigger for dealers to weigh all landings and to obtain vessel representative confirmation of SAFIS transactions.

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[60] *See* Amendment 14 DEIS at p. 279.

BioOne RESEARCH EVOLVED

Centuries of Anadromous Forage Fish Loss: Consequences for Ecosystem Connectivity and Productivity

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Centuries of Anadromous Forage Fish Loss: Consequences for Ecosystem Connectivity and Productivity

CAROLYN J. HALL, ADRIAN JORDAAN, AND MICHAEL G. FRISK

Lost biomass of anadromous forage species resulting from the seventeenth to nineteenth century damming of waterways and from overharvest in the northeastern United States contributed to significant changes in coastal marine–terrestrial ecosystems. Historic alewife populations in Maine for the years 1600–1900 were assessed using analyses of nineteenth and twentieth century harvest records and waterway obstruction records dating to the 1600s. Obstructed spawning access in nine watersheds reduced the annual alewife productivity per watershed to 0%–16% of virgin estimates, equaling a cumulative lost fisheries production of 11 billion fish from 1750 to 1900. Including preharvest production, our estimates suggest a lost flux of anadromous forage fish increasing from 10 million fish per year in 1700 to 1.4 billion annually by 1850. Our results suggest a realignment of current restoration goals is needed to recognize oceanic and freshwater ecosystem interdependence and the gap between current targets and potential productivity.

Keywords: applied ecology, history, coastal ecosystems, fisheries, dams

Recognition of a shifting baseline in natural resource policy and science (Pauly 1995) has resulted in increasing application of historical data to understand pre-exploitation conditions (Swetnam et al. 1999, Jackson et al. 2001, Alexander et al. 2009). Anadromous fish species require migration from marine to freshwater habitat to spawn and to complete their life cycle. This predictable annual appearance of spawning individuals has resulted in those species' long history as important resources for coastal communities. A paucity of baseline productivity estimates of northeastern US rivers has obscured the role that anadromous species played in precolonial coastal ecosystems. Many North Atlantic anadromous populations have presently been afforded *endangered* and *threatened* status, and several appear to have declined 90%–99% since the early twentieth century (Limburg and Waldman 2009). In systems that have large anadromous fish populations, the marine–terrestrial connection is important for ecological functioning (Schindler et al. 2003, Walters et al. 2009), which suggests that lost connections between land and sea result in impaired ecosystems. Furthermore, a number of the declining anadromous species are important forage fish, whose loss may leave only a small portion of the original prey base and may limit nutrient exchange between freshwater and marine habitats.

Limburg and Waldman (2009) analyzed 35 long-term anadromous species harvest and abundance data sets for the North Atlantic, but only 8 included data from the nineteenth century, and only 1 contained data collected prior to 1880. The pre-1880 data series, for American shad (*Alosa sapidissima*), indicated that catches from the early 1800s were around an order of magnitude higher than those of a century later. The loss of anadromous fish resulted from anthropogenic impacts and large-scale changes to coastal ecosystems throughout the Industrial Revolution, during which rapid increases in natural-resource use occurred (Bolster 2008, Alexander et al. 2009, Hall et al. 2011).

Ecosystem services

Alewives (*Alosa pseudoharengus*), one of two river herring species, undergo substantial coastal ocean migrations (Neves 1981), returning annually to spawn in lakes and slow-flow sections of rivers (Bigelow and Schroeder 1953). During spawning runs—especially at sites of constricted passage—alewives are easy and predictable targets for human, fish, and bird predators (Lindholdt 1988, Bolster 2008). European colonists described an abundance of spawning alewives in most northeastern coastal waterways "in such multitudes as is almost incredible, pressing up such shallow waters as will scarce permit them to swime [sic]" (Wood 1977, p. 56).

The harvest of alewives created economic opportunities throughout the northeastern United States as a principal bait in the current American lobster (*Homarus americanus*) and

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historical Atlantic cod (*Gadus morhua*) fisheries (Baird 1874, ASMFC 2009), for local consumption (Baird 1874), and as an export to the West Indies to feed slaves and laborers (Perley 1852). But beginning in the seventeenth century, dams constructed to power saw- and gristmills began blocking access to anadromous freshwater spawning sites. Efforts to prevent the continuing reduction of Atlantic salmon (*Salmo salar*), alewife, and shad populations resulted in 161 legislative acts passed in the state of Maine from 1800 to 1880 (Atkins 1887).

Alewife current status: A past problem?

Despite repeated efforts to restore alewives throughout the twentieth century, directed and nondirected fishing pressure, dam obstruction, pollution, and poor fishway construction and maintenance contributed to the ongoing population depletion (Rounsefell and Stringer 1945, Decker 1967). In a 1990 river herring (alewife and blueback herring) stock analysis including 11 New England rivers, Crecco and Gibson (1990) found that all populations were at least partially exploited and that 1 significant alewife river in Maine—the Damariscotta—was severely overfished. River herring populations continued to decline after the 1990 stock analysis, with Atlantic coast commercial landings decreasing from 6.2×10^6 kilograms (kg) in 1985 to less than 5.0×10^5 kg in 2007 (ASMFC 2009). River herring were federally listed as a species of concern in 2006 (NOAA 2006) and are currently undergoing a formal review of population status and trends for potential listing under the US Endangered Species Act.

The population decline of river herring is not a recent phenomenon. In 1868, impassable dams were identified in the first Maine Commissioner of Fisheries Report as the most damaging of anthropogenic impacts on anadromous species (Atkins and Foster 1868). Therefore, although the widespread decline of anadromous fish populations can be attributed to a multitude of anthropogenic effects, including pollution, overfishing, and changes to native watersheds (Köster et al. 2007, Limburg and Waldman 2009), the damming of waterways predates all other significant impacts (Perley 1852, Atkins and Foster 1868, Hall et al. 2011). In 1887, Fish Commissioner Atkins estimated that the productive capacity of Maine rivers had been reduced by 90% because of dam construction (Atkins 1887), suggesting that the 1887 alewife catch of 1.15×10^6 kg, or over 5 million fish, may have been only 10% of the potential fisheries production. Establishing goals and targets for restoration depends on the current distribution and abundance of the species but also on the historical capacity of populations. Here, we present two analyses to estimate the lost capacity of dammed rivers in the northwest Atlantic. First, we present a reconstruction of changes in the relative contribution of individual watersheds to alewife harvest in Maine. Second, we include estimates of historical alewife production per square kilometer based on twentieth century harvest records applied to accessible spawning area prior to the construction of dams to show lost watershed productivity during the period of 1600–1900.

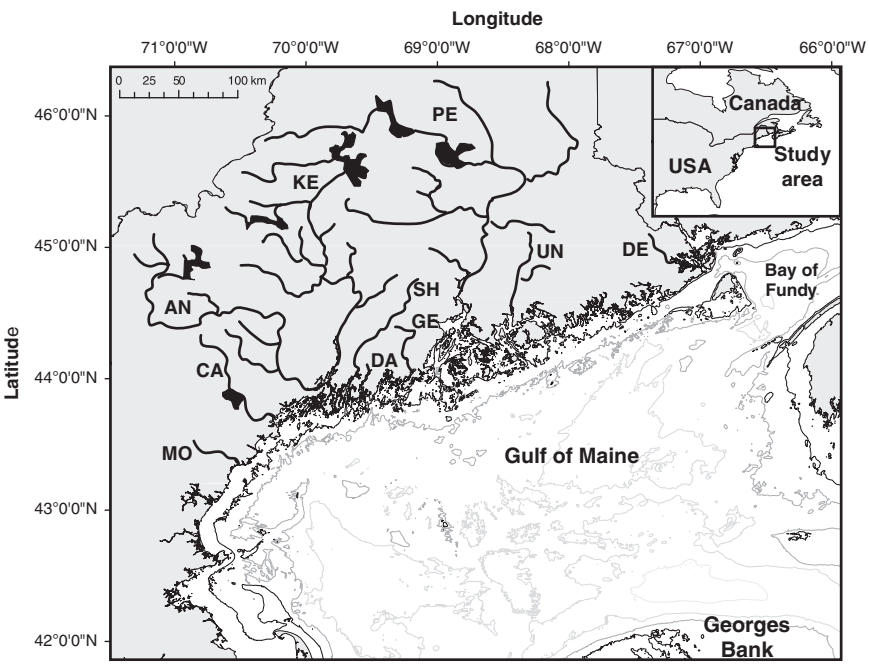

Figure 1. State and Gulf of Maine with historical river herring watersheds assessed for lost habitat due to damming: the Mousam River (MO), the Presumpscot River and Casco Bay (CA), the Androscoggin River (AN), the Kennebec River (KE), the Sheepscot River (SH), the Damariscotta River (DA), the St. George River (GE), the Penobscot River (PE), the Union River (UN), and the Dennys River (DE). Depth contours for the Gulf of Maine at 100, 200, 300, and 400 meters are also shown. The inset map displays the study location. Abbreviation: km, kilometers.

Historical fluctuations in Maine alewife harvest

Alewives were historically harvested from large and small watersheds along the entire Maine coast (figure 1). The earliest recorded alewife landings are in the Commonwealth of Massachusetts and State of Maine Fish Inspector reports, beginning in 1804 (Maine Secretary of State 1804–1893). The reports are from fishery and shipping towns along the length of the Maine coast and contain records of pickled-fish barrels and smoked-fish boxes intended for export, listed by town and species. The fish inspector records contain less information for entire river systems than later Fish Commissioner and Maine Department of Marine

Resources records because reporting was not enforced for each harvester in each town, nor was it replicated year to year, and it also did not account for local consumption. Although these records are inconsistent, they are the only regular Maine harvest records for the early nineteenth century and provide an estimate of the geographic range of harvest.

To compare barrel and box quantities to contemporary units, all quantities were converted to number of fish on the basis of values found in individual fish inspector reports. With an average weight of 90.72 kg of alewives per barrel and 0.227 kg per alewife (Rounsefell and Stringer 1945, Bigelow and Schroeder 1953), the number of alewives was determined to be 400 fish per barrel. For the boxes, 220,000 smoked alewives were packed into 3200 boxes, equaling 69 fish per box. All of these conversions were calculated on the basis of the average size of alewife captured in Maine during the late nineteenth and early twentieth centuries. Later nineteenth century landings reports included the first Maine Commissioner of Fisheries (MCOF) report (Atkins and Foster 1868), subsequent MCOF reports (MCOFG 1888, 1890), and special reports on river and alewife fisheries (Atkins 1887, Smith HM 1899). Mid-twentieth century annual landings were found in Rounsefell and Stringer's (1945) alewife fishery report. Town reports from 1943 to 2007 provided recent alewife catches per watershed and represented 90% of all of Maine's harvest (Gail Wippelhauser, Marine Resources Scientist, Maine Department of Marine Resources, personal communication, 28 August 2008).

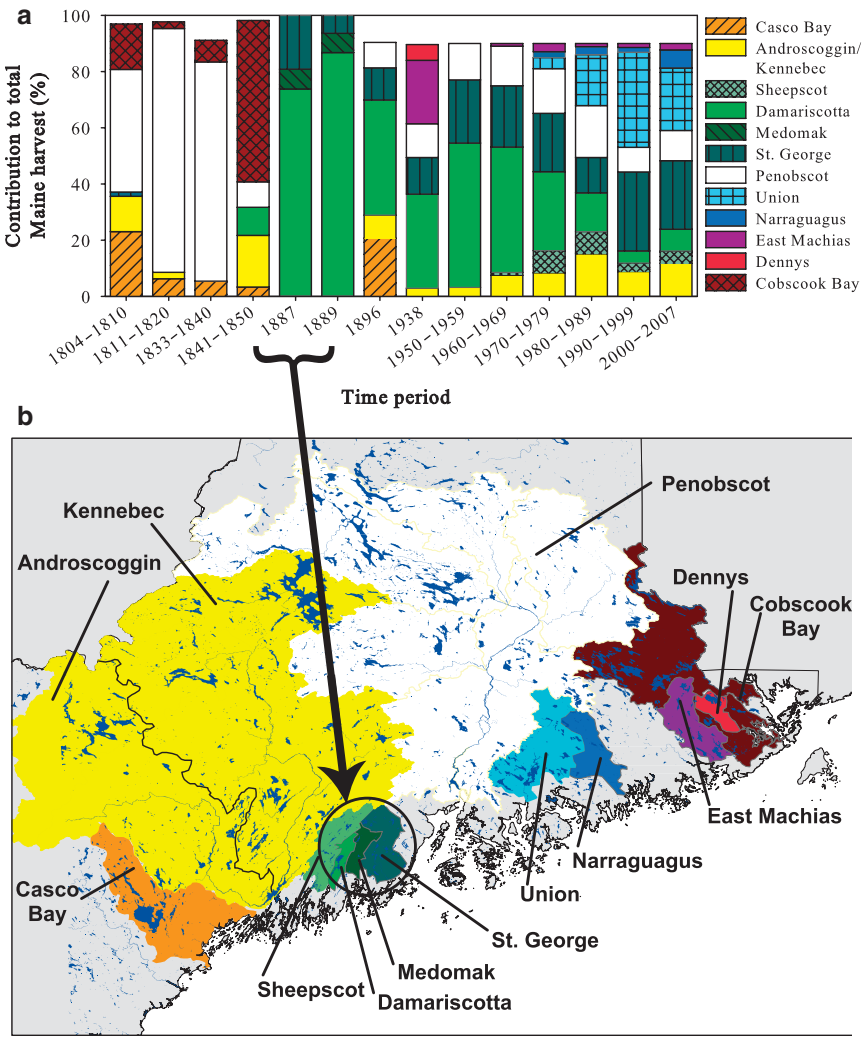

Figure 2. (a) Watershed harvest contribution: changes in individual watershed percentage contribution to state alewife landings reported between 1800 and 2007. Watersheds are displayed in the legend from west (top) to east (bottom). The harvest periods were determined from available watershed landings data. (b) Geographic shift of watersheds contributing to 1800s alewife harvest. The Maine watersheds are color coded to the legend in panel (a). The harvest watersheds of the 1880s are circled to illustrate when the landings were restricted to sites within the midcoast region, in contrast to the harvests of 1800–1840, when reported landings were distributed coastwide.

The historical and current fishery landings reports were evaluated to detect changes in watershed contribution to statewide landings. Spanning 1804–2007, these evaluations provide snapshots representing watershed alewife productivity and geographic shifts in historical habitat over time. A watershed fisheries productivity index was calculated as the percentage contribution of an individual watershed to the total Maine landings for each time interval. Although using landings data to calculate productivity shifts can result in bias because of other factors, such as unaccounted natural fluctuations, the behavior of harvesters, and changes in demand, they have been used to document declines in species productivity over long time periods (Myers and Worm 2003, Limburg and Waldman 2009). The landings were standardized across time by presenting each watershed as a percentage contribution rather than in weight or in the number of fish. A comparison of total alewife landings over the years is not provided here because, as a result of the inconsistencies stated above, the fish inspector data do not permit a reliable calculation of the total landings prior to 1880.

Harvest intervals were determined from single-year comprehensive records or multiyear periods representative of all watersheds harvested during a focused time frame and resulted in 14 time intervals from 1804 to 2007 (figure 2a). During the first part of the nineteenth century (1804–1840), five watersheds spanning the coast from Casco Bay in the west to easternmost

Cobscook Bay contributed over 90% of the state harvest (figure 2a). These five watersheds include the greatest-area watersheds in the study, with the Penobscot—the largest in Maine and second largest in New England—contributing the greatest portion (figure 2). By the late 1880s, only three watersheds—the Damariscotta, the Medomak, and the St. George—recorded yields. All three are located in the center of the coast and are significantly smaller watersheds than the Penobscot or Kennebec and Androscoggin Rivers (figure 2b). Harvests reexpanded along the coast in the twentieth century, including in Casco Bay and on the Dennys River in 1896 and 1938, respectively, but became more centrally focused again by the 1950s (figure 2). In addition, town records from the 1950s through the 2000s specified that all harvests were taken below the *head of tide*, or the most upstream reach of marine tidal waters (Gail Wippelhauser, Marine Resources Scientist, Maine Department of Marine Resources, personal communication, 28 August 2008). Therefore, the alewife harvest moved from large watersheds and the use of inland locations to an entirely coastal fishery that was focused on rivers with vastly lower potential capacities (Hall et al. 2011).

The Damariscotta and the Penobscot watersheds contributed the most to state landings in 11 and 12 time intervals, respectively, and constituted 86% of the state landings for one interval each prior to 1900. A replacement occurred in the mid-nineteenth century, with the Penobscot having the highest contribution through the 1840s, supplanted by the Damariscotta by the 1880s (figure 2). The Penobscot River system has 327.8 square kilometers (km^2) of potential lake habitat, compared with only 18.9 km^2 in the Damariscotta. In the 1880s, the St. George (24.3 km^2) also began to replace the contribution of the Casco (136.1 km^2) and Cobscook Bay systems (figure 2). The Damariscotta and Penobscot watershed contributions decreased to 7.7% and 10.7%, respectively, in the 2000s, and the St. George provided 24.4%. Notable among the percentage harvest values is the continuous shift from large to small river systems, often spanning orders-of-magnitude reductions in potential capacity.

Harvest-based estimates of productivity per unit area

Potential alewife production was calculated as the estimated adult (spawner escapement), recruit (first-time spawners), and harvest (annual fishery landings) median weight per area (in kilograms per square kilometer [kg/km^2]) for the Damariscotta River. Harvest data were available for 1949–2007. Adult and recruit estimates were available for 1949–1989 and 1949–1983, respectively (Crecco and Gibson 1990). Rather than focus on the span of overlapping years for the three population components (1949–1983), we analyzed all of the data in such a way that we fully utilized all of the available time series.

The estimated number of recruits was based on age-structure and abundance data and, in this case, is the sum of those returning to the river (the fishery) for the first time (Crecco and Gibson 1990). The number of adults is the estimated escapement (those allowed to pass to spawning habitat), which is based on the number of days without harvest (Crecco and Gibson 1990). For example, if 1 day of passage were permitted, the parents would be estimated as 16.7% of the harvested biomass. These data do not permit any sophisticated population modeling, so we provide the median and 25th and 75th percentiles as estimates for each population component and use box plots to display these with their extreme values. Landings data were also available from four other systems that were associated with a defined spawning region for which the area could be calculated. These are the Orland River (1943, 1949–1955, 1960–1969, 1971–1994, 1998–2007), Nequasset Lake (1943, 1958–1969, 1971–1989, 1992–1994, 1996–2004, 2006), Winnegance Lake (1943, 1969, 1971–1989, 1991, 1993–1994, 1998–2007), and the Dyer River (1973–1980, 1982–1986, 1988, 1993, 2000–2007). We applied the same technique and estimated the median value of harvest production per square kilometer for each of these systems. A statewide average median harvest production per square kilometer was calculated using the five rivers.

The estimated productivity of recruits and harvest per unit area using data available for the Damariscotta are 3.0×10^4 kg/km^2 and 2.5×10^4 kg/km^2, respectively (figure 3). The harvest estimate generated from the Damariscotta data is almost identical to the five-river state average harvest median value of 2.5×10^4 kg/km^2 (figure 4). The harvest estimates ranged from 6.6×10^3 kg/km^2 using Dyer River data to 5.0×10^4 kg/km^2 for Winnegance Lake (figure 4). These are not much different from two estimates for adult alewife returns currently used—2.9×10^4

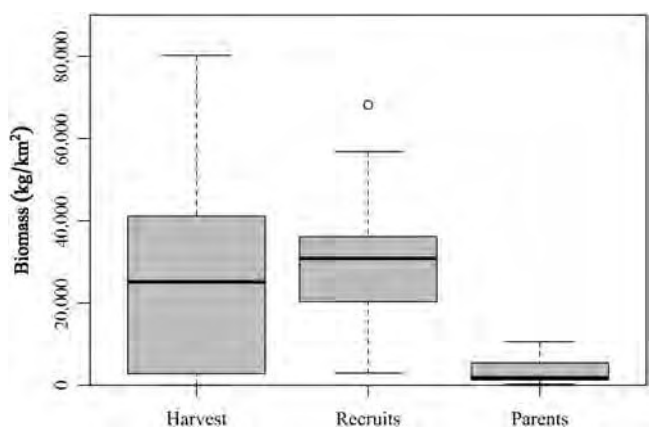

Figure 3. Damariscotta River production estimates are shown for the biological metrics of recruits, parents, and fisheries productivity per area (in kilograms per square kilometer [kg/km²]). The box plots show the median (the dark line); 25th and 75th percentiles (the ends of each box); the maximum value or 1.5 times the interquartile range, whichever is smaller (the error bars); and outliers (the open circles). Time series for the recruits represented the period from 1949 to 1983, that for the parents was from 1949 to 1989, and that for the harvest was from 1949 to 2007.

and 5.8×10^4 adults per area of lake—which are also based on annual harvest yields (Flagg 2007).

Historical alewife habitat availability and productivity

Estimating historical productivity is made difficult by a lack of abundance data and a paucity of landings data between 1600 and the present. We calculated lost historical alewife production (in kg) and the number of fish using available alewife habitat (km^2) altered by damming from 1600 to 1900 (Hall et al. 2011) and the Damariscotta data estimate of fisheries productivity (kg/km^2) detailed above. Our analysis is based on twentieth century data of watershed productivity and does not take into account anthropogenic and ecological changes over time to individual watersheds, such as eutrophication. However, it does provide the first baseline estimates of anadromous forage populations and illustrates how one key ecosystem change greatly affects productivity.

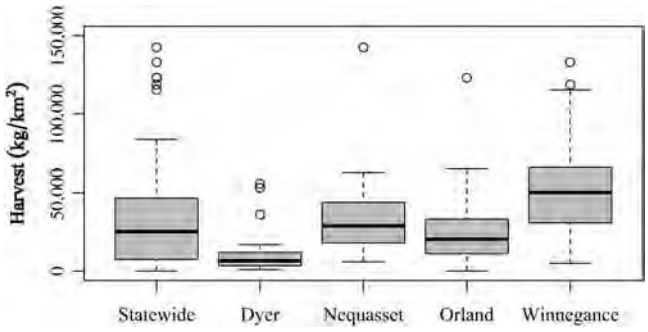

Figure 4. Harvest production estimates for a statewide average and for the Dyer River, Nequasset Lake, the Orland River, and Winnegance Lake. The time series represented the Orland River for 1943, 1949–1955, 1960–1969, 1971–1994, and 1998–2007; that for Nequasset Lake was for 1943, 1958–1969, 1971–1989, 1992–1994, 1996–2004, and 2006; that for Winnegance Lake was for 1943, 1969, 1971–1989, 1991, 1993–1994, and 1998–2007; and that for the Dyer River was for 1973–1980, 1982–1986, 1988, 1993, and 2000–2007. The statewide values are an average of the five watersheds with harvest-data time series. The open circles represent outliers. Abbreviation: kg/km², kilograms per square kilometers.

Historical alewife recruit, adult, and harvest productivity were calculated by applying the per-area productivity estimates from the Damariscotta to historically available alewife spawning habitat. The results demonstrate the near-complete loss of potential production well before 1900, with the majority of the loss occurring between 1750 and 1850 (figure 5a). As a comparison with contemporary population productivity, the total annual US river herring harvest during the 1950–2000 time period is also included (figure 5b), which demonstrates that the virgin alewife harvest productive capacity of nine Maine watersheds is equivalent to the average US river herring (alewife and blueback herring) harvest between 1950 and 1970, when stocks were considered healthy. Many of those "healthy" populations were maintained through assistance by trucking and stocking. In our estimates, we have not accounted for lost annual production of the other river herring species, blueback herring, which coexists in much of the same habitat as the alewife and is also a major contributor to the US Atlantic Coast river herring fisheries (Limburg and Waldman 2009). Therefore, the pre-1900 loss of alewives being comparable to the total US river herring landings from 1950 to 1970 is alarming, because the production of a single species in a handful of watersheds equaled that of the coastwide harvests of both species.

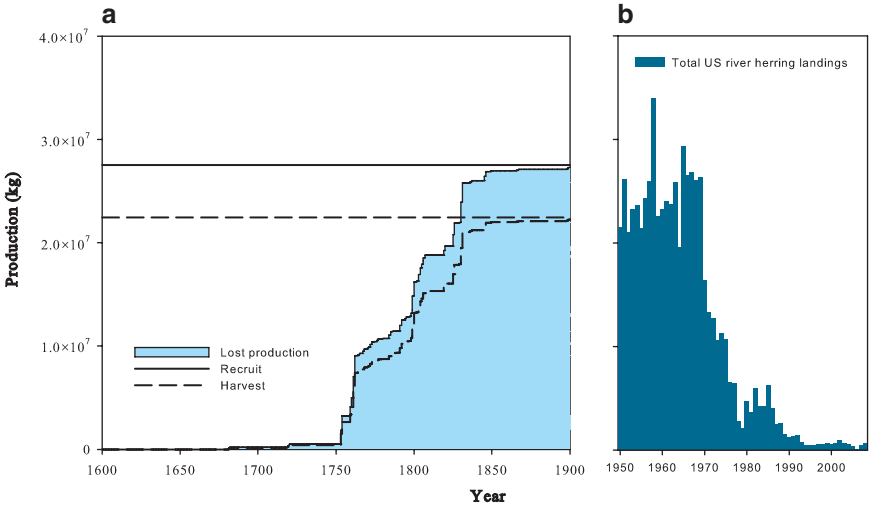

Figure 5. (a) Cumulative lost annual production of alewives from nine Maine watersheds for the time period of 1600–1900. The Damariscotta average production estimates (figure 3) were used to generate values for the recruits and harvest. (b) Total US river herring landings, which ranged between 1.8×10^7 and 2.9×10^7 kilograms (kg) per year from 1950 to 1970, are included to add perspective to the estimated lost production. The data were generated using the Atlantic Coastal Cooperative Statistics Program (www.accsp.org).

The annual productive capacity of the nine Maine rivers declined from 1600 to 1900 at least by 5.8×10^6 kg (using Dyer River data), at most by 4.4×10^7 kg (using Winnegance Lake data), and by 2.4×10^7 kg using the state average (figure 6). Therefore, regardless of the productivity estimates, dammed northeastern rivers appear to have lost at least six orders of magnitude in production capacity compared to their virgin, or undammed, potential. Assuming the average weight of adult fish applies, the result is the aggregate loss of 11.8 billion fish from harvest. As was stated earlier, Atkins estimated that the 1887 harvest of 5 million alewives was 10% of the undammed-waterway productivity potential. By his calculations, the annual

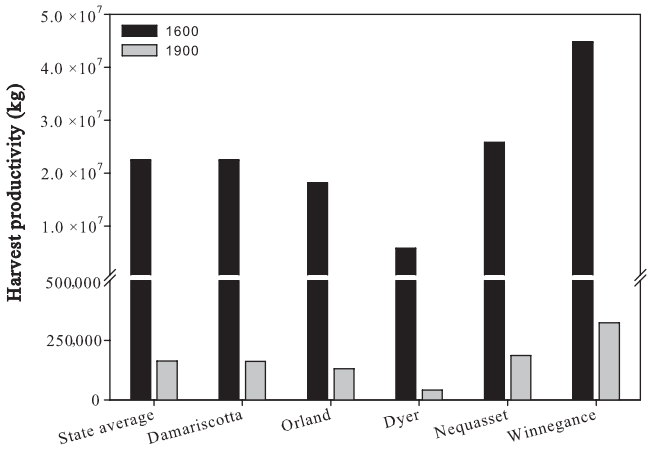

Figure 6. Lost alewife production: A comparison of 1600 to 1900. Lost production estimates using the statewide harvest data productivity estimate (figure 4) are combined with lost habitat for the nine study rivers (figure 5).

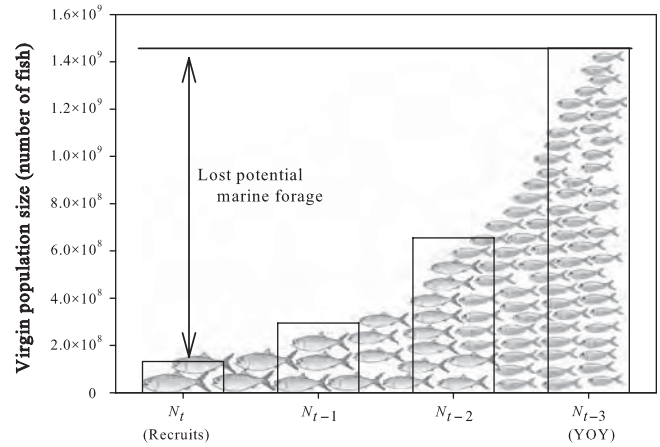

*Figure 7. After estimating the number of 4-year-old fish in 1880, assuming an average adult weight of 0.204 kilograms, the theoretical population sizes were back calculated to year 1 using an instantaneous mortality rate of 0.8. This provides an indication of the lost marine forage base supplied into the Gulf of Maine from the nine study rivers. Abbreviation: YOY, young of the year. The fish images are used courtesy of the Integration and Application Network, University of Maryland Center for Environmental Science (***http://ian.umces.edu/symbols***).*

harvest could have been 50 million fish. Our retrospective analysis indicates that in 1880, according to harvest productivity estimates and lost habitat, the nine river systems were 109 million fish short of their unobstructed potential—twice what Atkins surmised was the statewide condition.

Although the above estimates provide a measure of potential fisheries productivity, harvest-based metrics (including the number of recruits) vastly underestimate the true number of fish generated in each area, because the data were collected after a 4–6-year marine growth phase. To back calculate an estimated number of fish at age 1–3 years, we used the projection form of the exponential equation (Gotelli 1998) to account for natural mortality losses. It was assumed that all recruiting fish were 4 years old, weighed 0.204 kg, and experienced a natural instantaneous mortality rate of 0.8, or 55% mortality in one year (Crecco and Gibson 1990). Using the cumulative weight of recruits (2.93×10^9 kg) that were lost between 1600 and 1900 based on Damariscotta productivity estimates, 14 billion returning 4-year-old fish were absent from returning spawning runs. Annually, this translates into a missing 203,081 kg in 1700, which increased to 27,327,452 kg in 1900. Assuming a modest natural instantaneous mortality rate of 0.8, we estimate that from 1700 to 1800, the lost forage base from the Gulf of Maine increased from 10 million to 795 million juvenile fish per year. By 1850, the annual loss totaled 1.4 billion juvenile fish, as is displayed in the schematic demonstrating the back calculation of juvenile numbers in figure 7. The changes reflect the period of lost production concentrated from 1750 to 1850.

Setting a baseline

River herring stock status is frequently judged on the basis of performance relative to the annual US landings of the 1950s to the 1970s. However, our pre-1900 estimate of alewives from nine Maine rivers indicates that this mid-twentieth century reference represents a population that is at a level considerably reduced from baseline conditions and demonstrates the need for the application of historical data. In historical ecology studies, commercial harvest data of northwest Atlantic fish populations, ships' logs, and naturalist publications have been used to document dramatic declines in both body size (Jackson et al. 2001) and abundance (Lotze and Milewski 2004, Rosenberg et al. 2005) of the primary historical predator Atlantic cod (*Gadus morhua*) and 99% reductions of anadromous fish (Lotze and Milewski 2004). Although these estimates seem extreme, such studies are often conservative, are restricted to data from the nineteenth century, and do not incorporate the first 200 years of northwest Atlantic colonial exploitation. Our historical alewife population baseline determined from landings records, productivity estimates, and a time series of available habitat from 1600 to 1900 illustrates another such population in decline—in this case, a highly migratory anadromous forage fish.

Lost forage and ecosystem processes

The most significant implication of the lost forage base is the impact on coastal trophic relationships—in particular, as it relates to restoration goals and ecosystem-based management. After being heavily overfished until federal law required restrictive management and moratoriums on commercial fishing in 1984, the Atlantic coastal stocks of striped bass (*Morone saxatilis*) are at or near former twentieth century population highs (Hartman and Margraf 2003). The case of the striped bass is a single-species restoration success story, but it points to a larger issue for ecosystem approaches.

Because striped bass are generalist predators, enabling recovery without a concomitant increase in prey resources is now having negative impacts on salmon and other anadromous fish through predation (Grout 2006). Hudson River estimates of striped bass predation on alosines, which are considered a critical prey item, exceeded biomass in recent years, indicating that prey resources are limiting (Hartman 2003). Not only have prey populations decreased, but striped bass themselves have exhibited declining condition and decreased growth rates (Griffin and Margraf 2003, Hartman 2003, Uphoff 2003, Walter et al. 2003). Therefore, the restoration of single species cannot be viewed independently, and ecosystem-based management requires that forage fish be included in predator restoration planning.

Lost production from alosine species also intersects with the management of other forage species, such as the Atlantic herring (*Clupea harengus*), the largest Gulf of Maine commercial fishery, which shares habitat with river herring in coastal waters (Jordaan et al. 2010). It is not possible to reasonably conjecture on the historical interplay among forage species in the Gulf of Maine; however, it is clear that current management must consider the migrations and habitat overlap of forage species, particularly if active fisheries are involved. For example, bycatch of alewives in the Atlantic herring fisheries can have a large impact on populations (Cieri et al. 2008). The lost production of alewives because of obstructed rivers has left reduced population sizes susceptible to overharvest even if only as bycatch. As a result, harvest thresholds for maintaining sustainable commercial fisheries will have to be reduced to alleviate additional river herring mortality. Therefore, restoring forage species biodiversity requires not only harvest controls but also consideration of the interdependence of species, many of which are federally managed, in a comprehensive management plan.

Management implications

Management reaction to the dramatic decline of river herring stocks in the late twentieth century resulted in the closure of numerous Atlantic state commercial river herring fisheries or in much-reduced harvests (ASMFC 2009). The legacy of removing watersheds from harvest followed the temporal and spatial pattern of obstructions to New England's rivers. In Maine, as alewife habitat was dammed (Hall et al. 2011), harvest intensity was focused on fewer and fewer rivers. In 1835, the completion of the Veazie Dam on the main stem of the Penobscot River at the head of tide (Hall et al. 2011) essentially eliminated the historically significant level of harvest in the state of Maine. Within one decade of the dam's construction, the Penobscot's contribution to state landings decreased nearly 70%. In contrast to the diminishing Penobscot contribution, the Damariscotta began to register as a regular contributor to alewife harvest. Midcoast Maine became the principal alewife fishing region during the 1880s and fishing was focused on watersheds of significantly less spawning habitat than the Penobscot, with most harvest occurring at head of tide dams and downstream estuarine weirs (Atkins 1887). By 1846, all watersheds were obstructed at the head of tide, except the artificially accessible Damariscotta Lake (Hall et al. 2011), and annual statewide fishery potential productivity dropped 98%. The result of landscape changes is a dramatic shift in ecosystems long forgotten by society, with former connected and productive systems lost to impounded recreational ponds and hydropower dams.

Restoring access to geographically diverse historical spawning sites would result in increased population biocomplexity and improved species resilience in the face of environmental changes (Hilborn et al. 2003). Restored access would also reestablish missing marine–terrestrial nutrient exchange and a forage base for a vast number of predators. Alewife are an ideal candidate for restoration efforts, with high fecundity and straying rates that allow for rapid colonization of reopened spawning habitat within 3–5 years (Atkins and Foster 1868, Bigelow and Schroeder 1953, Pardue 1983). This has been demonstrated where dams have been removed or successful fish passages installed (Lichter et al. 2006). In addition to improving the resilience of alewife populations and nutrient exchange, the restoration of alewives will have profound impacts on other species. The alewife floater (*Anodonta implicata*), a freshwater mussel dependent on alewives for life cycle completion (Davenport and Warmuth 1965), has undergone a range expansion because of dam removal and the restoration of alewife populations (Smith DG 1985). Atlantic salmon smolt could benefit from increased alewife spawning populations that may provide critical prey protection from aquatic and terrestrial predators (Fay 2003). Numerous coastal predators, many of which are also at low population levels, would benefit from the additional influx of forage. For example, the restoration of striped bass demonstrated that any increase in coastal predator populations such as Atlantic cod, which have lost numerous spawning populations (Ames 2004), would require the reestablishment of a large and varied forage base. The anadromous river herrings link the management of local-scale habitat restoration, freshwater and oceanic predators, and open-ocean fisheries. In the case of alewife, removing obstructions would result in a natural reintroduction from nearby stocks that would benefit the greater Gulf of Maine ecosystem.

Native American cultures, ecosystems of the past, and alewives

The significance of the impacts of precolonial river herring harvest and river obstruction in Maine is uncertain. Although river herring were certainly used by Native American cultures, there is a lack of any evidence of the restriction of herring access to spawning habitat. River herring remains have been found in midden sites in Maine around Damariscotta, but they appeared not to be a major component of the diet. Instead, larger fish, including cod, flounder, sculpin, swordfish, striped bass, and sturgeon, represented the majority of the bones recovered (Spiess and Lewis 2001). It is possible that native cultures of North America used alewives for agricultural fertilizer (Goode 1880). However, the reality of

this practice is debated (Clifton 2007), and at the very least, the quantities of alewife used were not likely to be significant compared with the species' abundance in precolonial times.

As in all historical studies, there are several caveats to the methods employed here to determine abundances. Out of necessity and for the reasons described above, production estimates were not recorded during precolonial conditions; instead, we utilized values from the twentieth century derived from harvested watersheds. However, because the time period used encompasses a significant population decline, the likelihood that our production estimates are conservative is high. It would be ideal to have pre-twentieth century data or data representing out-emigrating young alewives to understand watershed productivity without depending on returning-age adults (4 years old or older). However, even without more refined calculations, the estimates of historic alewife productivity presented here allow for a more robust and realistic evaluation of current ecosystem recovery initiatives and can help guide more effective management programs in the future.

Restoration and dam construction: A cautionary note for ecologists and managers

In the United States, a change in coastal river and anadromous species restoration is under way with the removal of numerous dams and a noted increase in fishway construction (Opperman et al. 2011). In Pacific systems, fish passage success has been examined (Ferguson et al. 2011) and has been subject to judicial review (Service 2011). After over 100 years of intensive fishway efforts, little progress has been made to integrate anadromous fishes into a holistic view of ecosystem restoration (Katopodis and Williams 2011). As the United States struggles to balance current hydropower and fish-restoration policy, China, the Lao People's Democratic Republic, and Cambodia are on the verge of constructing 200, including 11 mainstem, dams in the Mekong River (Ferguson et al. 2011, Grumbine and Xu 2011). The Mekong River is the second most biodiverse freshwater system, currently hosts the world's most productive inland fisheries (Ferguson et al. 2011), and is relied on by 2.1 million people (Grumbine and Xu 2011). The present study offers a cautionary tale for the future, suggesting that losing the connections and ecosystem services provided by anadromous species will have lasting ecosystem-wide impacts that may not be compensated through fish passage technology. Furthermore, these changes are rarely accounted for by contemporary ecologists evaluating structure and processes in North America and Europe.

It is impossible to fully appreciate how far present-day ecosystems have shifted structurally and productively without pre-exploitation baseline estimates. The challenge lies in finding historical data with which to calculate and estimate those baseline values. In our approach, we combined an extensive analysis of historical records spanning 300 years to estimate available river habitat with productivity estimates to determine the loss of historical anadromous fish populations from Maine's ecosystems. Although the project was limited to alewives, the findings indicate that billions of forage fish are missing annually from coastal ecosystems and potential harvests. If similar methods were applied globally, with many regions having much longer timelines of human river obstruction than the northeastern United States, it is hard to imagine the magnitude of lost productivity and exchange between marine and freshwater ecosystems. The implications of this magnitude of change for our current understanding of system connectivity and wildlife management, developed primarily over the past 50 years, will require integration into policy decisions in order for restoration actions to be made with an ecosystem-based perspective.

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References cited

Alexander KE, et al. 2009. Gulf of Maine cod in 1861: Historical analysis of fishery logbooks, with ecosystem implications. Fish and Fisheries 10: 428–449.

Ames EP. 2004. Atlantic Cod stock structure in the Gulf of Maine. Fisheries 29: 10–28.

[ASMFC] Atlantic States Marine Fisheries Commission. 2009. Amendment 2 to the Interstate Fishery Management Plan for Shad and River Herring. ASMFC. (14 May 2012; *www.jcaa.org/news/amendment2_River Herring.pdf)*

Atkins CG. 1887. The river fisheries of Maine. Pages 673–728 in Goode BG, ed. The Fisheries and Fishery Industries of the United States, vol. 1, sect. V. US Government Printing Office.

Atkins CG, Foster N. 1868. First Report of the Commissioners of Fisheries of the State of Maine, 1867. Owen and Nash.

Baird ST. 1874. U.S. Commission of Fish and Fisheries, part II. US Government Printing Office.

Bigelow HB, Schroeder WC. 1953. Fishes of the Gulf of Maine. Fisheries Bulletin no. 74, vol. 53. US Government Printing Office.

Bolster WJ. 2008. Putting the ocean in Atlantic history: Maritime communities and marine ecology in the northwest Atlantic, 1500–1800. The American Historical Review 113: 19–47.

Cieri M, Nelson G, Armstrong M. 2008. Estimates of River Herring Bycatch in the Directed Atlantic Herring Fishery. Maine Department of Marine Resources and Massachusetts Division of Marine Fisheries.

Clifton JA, ed. 2007. The Invented Indian: Cultural Fictions and Government Policies, 5th ed. Transaction.

Crecco VA, Gibson MR. 1990. Stock Assessment of River Herring from Selected Atlantic Coast Rivers. Atlantic States Marine Fisheries Commission. Special Report no. 19.

Davenport D, Warmuth M. 1965. Notes on the relationship between the freshwater mussel *Anodonta implicata* Say and the alewife *Pomolobus pseudoharengus* (Wilson). Limnology and Oceanography 10 (suppl.): 74–78.

Decker LF. 1967. Fishways in Maine. Maine Department of Inland Fisheries and Game.

Dudgeon D. 2011. Asian river fishes in the Anthropocene: Threats and conservation challenges in an era of rapid environmental change. Journal of Fish Biology 79: 1487–1524.

Fay C. 2003. Biological and ecological role of alewives. Paper presented at the Maine Rivers Annual Conference; 27–28 September 2003, Indian Island, Maine.

Ferguson JW, Healey M, Dugan P, Barlow C. 2011. Potential effects of dams on migratory fish in the Mekong River: Lessons from salmon in the Fraser and Columbia Rivers. Environmental Management 47: 141–159.

Flagg LN. 2007. Historical and current distribution and abundance of the anadromous alewife (*Alosa pseudoharengus*) in the St Croix River. Atlantic Salmon Commission. (14 May 2012; *www.maine.gov/dmr/searunfish/reports/stcroixalewifeflagg07.pdf*)

Goode GB. 1880. The use of agricultural fertilizers by the American Indians and the early English colonists. American Naturalist 14: 473–479.

Gotelli NJ. 1998. A Primer of Ecology, 2nd ed. Sinauer.

Griffin JC, Margraf FJ. 2003. The diet of Chesapeake Bay striped bass in the late 1950s. Fisheries Management and Ecology 10: 323–328.

Grout DE. 2006. Interactions between striped bass (*Morone saxatilis*) rebuilding programmes and the conservation of Atlantic salmon (*Salmo salar*) and other anadromous fish species in the USA. ICES Journal of Marine Science 63: 1346–1352.

Grumbine RE, Xu J. 2011. Mekong hydropower development. Science 332: 178–179.

Hall CJ, Jordaan A, Frisk MG. 2011. The historical influence of dams on diadromous fish habitat with a focus on river herring and hydrologic longitudinal connectivity. Landscape Ecology 26: 95–107.

Hartman KJ. 2003. Population-level consumption by Atlantic coastal striped bass and the influence of population recovery upon prey communities. Fisheries Management and Ecology 10: 281–288.

Hartman KJ, Margraf FJ. 2003. US Atlantic coast striped bass: Issues with a recovered population. Fisheries Management and Ecology 10: 309–312.

Hilborn R, Quinn TP, Schindler DE, Rogers DE. 2003. Biocomplexity and fisheries sustainability. Proceedings of the National Academy of Sciences 100: 6564–6568.

Jackson JBC, et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293: 629–637.

Jordaan A, Chen Y, Townsend DW, Sherman S. 2010. Identification of ecological structure and species relationships along an oceanographic gradient in the Gulf of Maine using multivariate analysis with bootstrapping. Canadian Journal of Fisheries and Aquatic Sciences 67: 701–719.

Katopodis C, Williams JG. 2011. The development of fish passage research in a historical context. Ecological Engineering. doi:10.1016/j.ecoleng.2011.07.004

Köster D, Lichter J, Lea PD, Nurse A. 2007. Historical eutrophication in a river-estuary complex in mid-coast Maine. Ecological Applications 17: 765–778.

Lichter J, Caron H, Pasakarnis TS, Rodgers SL, Squiers TS Jr, Todd CS. 2006. The ecological collapse and partial recovery of a freshwater tidal ecosystem. Northeastern Naturalist 13: 153–178.

Limburg KE, Waldman JR. 2009. Dramatic declines in North Atlantic diadromous fishes. BioScience 59: 955–965.

Lindholdt PJ, ed. 1988. John Josselyn, Colonial Traveler: A Critical Edition of Two Voyages to New-England. University Press of New England.

Lotze HK, Milewski I. 2004. Two centuries of multiple human impacts and successive changes in a North Atlantic food web. Ecological Applications 14: 1428–1447.

Maine Secretary of State. 1804–1893. Fish Inspector Records, boxes 1–9. Maine State Archives.

[MCOFG] Maine Commissioners of Fisheries and Game. 1888. Report of the Commissioners of Fisheries and Game of the State of Maine for the Year 1888. Burleigh and Flint.

———. 1890. Report of the Commissioners of Fisheries and Game of the State of Maine for the Years 1889–1890. Burleigh and Flint.

Myers RA, Worm B. 2003. Rapid worldwide depletion of predatory fish communities. Nature 423: 280–283.

Neves RJ. 1981. Offshore distribution of alewife, *Alosa pseudoharengus*, and blueback herring, *Alosa aestivalis*, along the Atlantic Coast. US Fisheries Bulletin 79: 473–485.

[NOAA] National Oceanic and Atmospheric Administration. 2006. Endangered and threatened species: Revision of species of concern list, candidate species definition, and candidate species list. Federal Register 71: 61022–61025.

Opperman JJ, Royte J, Banks J, Day LR, Apse C. 2011. The Penobscot River, Maine, USA: A basin-scale approach to balancing power generation and ecosystem restoration. Ecology and Society 16 (art. 7). (14 May 2012; *www.ecologyandsociety.org/vol16/iss3/art7*)

Pardue GB. 1983. Habitat Suitability Index Models: Alewife and Blueback Herring. US Fish and Wildlife Service. Report no. FWS/OBS-82/10.58.

Pauly D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. Trends in Ecology and Evolution 10: 430.

Perley MH. 1852. Reports on the Sea and River Fisheries of New Brunswick, 2nd ed. Simpson.

Rosenberg AA, Bolster WJ, Alexander KE, Leavenworth WB, Cooper AB, McKenzie MG. 2005. The history of ocean resources: Modeling cod biomass using historical records. Frontiers in Ecology and the Environment 3: 84–90.

Rounsefell GA, Stringer LD. 1945. Restoration and management of the New England alewife fisheries with special reference to Maine. Transactions of the American Fisheries Society 73: 394–424.

Schindler DE, Scheuerell MD, Moore JW, Gende SM, Francis TB, Palen WJ. 2003. Pacific salmon and the ecology of coastal ecosystems. Frontiers in Ecology and the Environment 1: 31–37.

Service RF. 2011. Rejected salmon plan could bring changes to U.S. dams. Science 333: 811.

Smith DG. 1985. Recent range expansion of the freshwater mussel *Anodonta implicata* and its relationship to clupeid fish restoration in the Connecticut River system. Freshwater Invertebrate Biology 4: 105–108.

Smith HM. 1899. Notes on the extent and condition of the alewife fisheries of the United States in 1896. Pages 31–43 in US Commissioner of Fish and Fisheries Report Part XXIV for the Year Ending June 30, 1898. Government Printing Office.

Spiess AE, Lewis RA. 2001. The Turner Farm Fauna: 5000 Years of Hunting and Fishing in Penobscot Bay, Maine. Maine State Museum, Maine Historic Preservation Commission, and Maine Archaeological Society. Occasional Publications in Maine Archaeology no. 11.

Swetnam TW, Allen CD, Betancourt JL. 1999. Applied historical ecology: Using the past to manage for the future. Ecological Applications 9: 1189–1206.

Uphoff JH. 2003. Predator–prey analysis of striped bass and Atlantic menhaden in upper Chesapeake Bay. Fisheries Management and Ecology 10: 313–322.

Walter JF III, Overton AS, Ferry KH, Mather ME. 2003. Atlantic coast feeding habits of striped bass: A synthesis supporting a coast-wide understanding of trophic biology. Fisheries Management and Ecology 10: 349–360.

Walters AW, Barnes RT, Post DM. 2009. Anadromous alewives (*Alosa pseudoharengus*) contribute marine-derived nutrients to coastal stream food webs. Canadian Journal of Fisheries and Aquatic Sciences 66: 439–448.

Wood W. 1977. New England's Prospect, ed. Vaughan AT. University of Massachusetts Press. (Originally published 1634.)

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Dramatic Declines in North Atlantic Diadromous Fishes

KARIN E. LIMBURG AND JOHN R. WALDMAN

We examined the status of diadromous (migratory between saltwater and freshwater) fishes within the North Atlantic basin, a region of pronounced declines in fisheries for many obligate marine species. Data on these 24 diadromous (22 anadromous, 2 catadromous) species are sparse, except for a few high-value forms. For 35 time series, relative abundances had dropped to less than 98% of historic levels in 13, and to less than 90% in an additional 11. Most reached their lowest levels near the end of the observation period. Many populations persist at sharply reduced levels, but all species had suffered population extirpations, and many species are now classified as threatened or endangered. Habitat loss (especially damming), overfishing, pollution, and, increasingly, climate change, nonnative species, and aquaculture contributed to declines in this group. For those diadromous fishes for which data exist, we show that populations have declined dramatically from original baselines. We also discuss the consequences of these changes in terms of lost ecosystem services.

Keywords: diadromous fishes, overfishing, dams and other threats, habitat loss, shifting baselines

We examined the status of North Atlantic diadromous fishes, that is, those species that migrate between marine waters and continental watersheds to complete their life cycles. The North Atlantic basin receives the drainage of major rivers such as the St. Lawrence, the Mississippi, and the Rhine, and hundreds of smaller rivers, all of which host diadromous fishes. Diadromy occurs in two primary forms: anadromy, in which spawning takes place in freshwater, and catadromy, in which reproduction occurs at sea. Diadromous fishes comprise less than 1% of world fish fauna, but their value to humans far exceeds this portion. Many diadromous fishes such as salmons, sturgeons, and shads are not only economically important, but they also serve as crucial links for energy flow between fresh and marine environments (Helfman 2007).

Recent analyses have shown major declines in many North Atlantic obligate marine fishes (Christensen et al. 2003). For these species, declines generally take the form of population reductions to the level of commercial extinction, but not extirpation (Casey and Myers 1998). Unlike many marine fishes that have few but large, geographically widespread populations, most anadromous fishes have numerous but smaller river-specific populations (Powles et al. 2000). This renders them more susceptible to population-level extirpations, and, if these extirpations occur serially, species extinction may occur.

Diadromy as a life-history strategy has evolved in phylogenetically diverse fish groups (McDowall 1997). It appears to offer the benefits of lessened predation in early life stages, access to increased food resources in marine environments for individuals, and the potential for demographic and morphological sculpting to the particulars of each population's migratory circuit (McDowall 2001). These habitat-switching life histories may have evolved in response to geographic differentials in marine and freshwater productivity, with anadromous species dominating the higher latitudes where marine productivity far exceeds that of inland waters (Gross et al. 1988). But these more complicated life histories come with costs, including osmoregulatory and energetic demands for movement between two distinctly different environments. Moreover, occurrence both in freshwater and in the sea exposes populations to the uncertainties of environmental conditions in two realms.

Recent work has shown that migratory movements of diadromous fishes are far more complex than originally thought (e.g., Secor and Rooker 2000, Limburg et al. 2001). Many display spectacular long-distance migrations not only at sea but also as they traverse thousands of kilometers inland and ascend hundreds of meters in elevation. Because the spawning aggregations of diadromous fishes often place them within easy reach of humans, these runs have been particularly important sources of protein.

"Ecosystem goods and services" is a recently derived paradigm (Daily 1997, Ruffo and Kareiva 2009) used to demonstrate the value and benefits to humans of the natural world. Ecosystem services are defined as natural ecological functions

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or properties that support human well-being either directly or indirectly. In this paradigm, diadromous fishes have four special roles, although we will show that their importance in these functions has diminished greatly as a result of their population declines. First, provisioning of protein and other products is a primary ecosystem service of diadromous fishes because of their (historic) vast abundances, the high predictability of these runs, and the ease of their capture as they aggregate near or on their spawning grounds (Bolster 2008). Second, these fishes link continental and marine ecosystems, transporting embodied productivity from one to the other. Semelparous anadromous fishes (those that spawn once and then die) may act as keystone species (Willson and Halupka 1995): They have a major impact in their ecological communities because their carcasses are consumed directly by wildlife or stream infauna, or they decompose and release their nutrients to the water or riparian zones. Garman (1992) estimated that the nontidal James River, in Virginia, may have received annual biomass input from anadromous alosines of 1.55 kilograms (kg) per hectare (ha) (representing 3.6 million individuals in the run, with 70% mortality) before dams blocked their movements. Garman (1992) determined mean decomposition rates on the order of 10 days. These subsidies of "marine-derived nutrients" often serve as critical additions of energy and nutrients that fuel food webs well beyond the streams in which they died (Gende et al. 2002).

A third ecosystem service generated by diadromous species is the support of marine food chains through the addition of fish that emigrate from natal rivers to the sea, again transporting energy and nutrients, but in the reverse direction. At northern temperate latitudes, these fluxes are composed mainly of young fishes emigrating seaward. Nineteenth-century reports noted that the voluminous outpourings of young anadromous fishes provided important forage for marine species such as cod, *Gadus morhua,* tightly coupling inland production to coastal food webs (Stevenson 1899); today, such continental-marine linkages are broken to a large extent in the North Atlantic basin. This coupling also enabled fishers to harvest marine predators closer to shore without having to venture onto the high seas (Stevenson 1899).

Finally, diadromous species have played important roles for both indigenous and nonindigenous peoples. Because these fishes could supply great amounts of food after long periods with little to eat, they enjoyed high cultural status. For many coastal Native American communities, Atlantic sturgeon (*Acipenser oxyrinchus*), American eel (*Anguilla rostrata*), and other diadromous fishes had enormous practical and totemic importance (Bolster 2008). In modern American society, coastal communities still celebrate the return of American shad (*Alosa sapidissima*), hickory shad (*Alosa mediocris*), river herring (alewife, *Alosa pseudoharengus*, and blueback herring, *Alosa aestivalis*) (Waldman 2003), although these runs, and celebrations thereof, have diminished greatly.

Metrics of change

We synthesized information on the current status of North Atlantic diadromous fishes using these metrics: the number of original populations versus extant populations (table 1), temporal changes in population abundances or harvests (table 2, figure 1), and official conservation status (table 1). We identified 24 diadromous fishes in the North Atlantic. Of these, 12 are restricted to North America, 9 to Europe and Africa, and 3 are common to both shores. Each coast has only one strongly catadromous species, American eel and European eel (*Anguilla anguilla*). Information about the survival status of populations of diadromous fishes was obtained from the broadest and most recent sources available. The conservation status listed also was from the broadest possible listing identified.

Time-series data sets were collected mostly from published literature; two sets (European eel recruitment in Swedish rivers, and Atlantic salmon [*Salmo salar*] catches in the River Dee) were obtained from scientists in their respective fields of expertise (see the acknowledgments). Because few species have long time series of fisheries-independent data, catch statistics were the most commonly found time series. While fishery data are often subject to biases due to factors such as markets, fads, and misreporting (Ocean Studies Board 2000), in general, the species in our survey were in demand throughout most of the periods of observation.

We analyzed the time series in two ways. First, because of the variety of response variables (abundances, tons, catches per unit effort, recruitment indices), as well as the differences in absolute magnitudes of the variables, we normalized the time series so that the maximum value equals one and the minimum equals zero. These transformed data were then plotted (figure 1) for visual comparisons of trends. Second, because of the uncertainty about the meaning of individual data points (i.e., a peak in a time series in a particular year probably does not correspond to a peak in abundance or even to peak catch per unit effort expended), the untransformed data were smoothed by running averages corresponding to a particular species' generation time, thereby lessening the importance of individual points and emphasizing the trends over the time frame of the data. The slopes of the log transformation of these smoothed time series were computed and used to calculate the percentage change in relative abundance over the period of observation (table 2).

We had an especially rich and long set of American shad landings from the Atlantic States Marine Fisheries Commission (ASMFC 2007) that could be examined for evidence of multiple shifting baselines. These were normalized to the number of river kilometers available for spawning within each river system along the eastern US coast (ASMFC 2007).

Numbers of populations

For many species, data on historical and present numbers of populations are deficient; the availability of information appears positively associated with their commercial importance. Of the 14 anadromous species for which comparisons

Table 1. The original reproductive range of North Atlantic diadromous fish species, numbers of original and extant populations, and current highest institutional-level species conservation status.

Common name	Latin name	Original reproductive range	Number of original populations	Number of extant populations	Conservation status
Western Atlantic					
Sea lamprey	*Petromyzon marinus*	Florida to New Brunswick	116 (Beamish 1980)	DD	LC (IUCN 2008)
Shortnose sturgeon	*Acipenser brevirostrum*	Florida to New Brunswick	> 20 (NMFS 1988)	About 20 (NMFS 1988)	VU (IUCN 2008)
Atlantic sturgeon	*Acipenser oxyrinchus*	Mississippi to Quebec	> 35 (Waldman and Wirgin 1998)	About 35 (Waldman and Wirgin 1998)	NT (IUCN 2008)
Alewife	*Alosa pseuodharengus*	South Carolina to Newfoundland	DD	DD	SC (NMFS 2009)
Blueback herring	*Alosa aestivalis*	Florida to Nova Scotia	DD	DD	SC (NMFS 2009)
Hickory shad	*Alosa mediocris*	Florida to Maine	DD	DD	Status unknown[a]
Skipjack herring	*Alosa chrysochloris*	Texas to Florida	DD	DD	Stable (Warren et al. 2000)
American shad	*Alosa sapidissima*	Florida to Quebec	138 (Limburg et al. 2003)	68 (Limburg et al. 2003)	Lowest in history (ASMFC 2007)
Alabama shad	*Alosa alabamae*	Louisiana to Florida	DD	7 (Mettee and O'Neil 2003)	EN (IUCN 2008)
Atlantic whitefish	*Coregonus huntsmani*	Nova Scotia	2	1	VU (IUCN 2008)
Arctic char	*Salvelinus alpinus*	Newfoundland to the Arctic Ocean	DD	DD	LC (IUCN 2008)
Atlantic salmon	*Salmo salar*	Connecticut to Quebec	600 (of which 398 are DD; WWF 2001)	135 of 202 (WWF 2001)	LR/lc (IUCN 2008); needs updating
Rainbow smelt	*Osmerus mordax*	Delaware to Labrador	DD	DD	SC[a]
American eel	*Anguilla rostrata*	Brazil to Greenland	1 (panmictic)	1 (panmictic)	Highly depleted in Great Lakes drainage
Striped bass	*Morone saxatilis*	Louisiana to Quebec	About 50 (Fruge et al. 2006)	< 50 (Fruge et al. 2006)	Not overfished[a]
Eastern Atlantic					
Sea lamprey	*Petromyzon marinus*	Greenland/Norway to the western Mediterranean	DD	DD	Declining regionally
River lamprey	*Lampetra fluviatilis*	Finland to the western Mediterranean	DD	DD	DD (IUCN 2008)
European sea sturgeon	*Acipenser sturio*	Baltic Sea to the Black Sea	> 18 (Elvira et al. 2000)	1 (Elvira et al. 2000)	CR (IUCN 2008)
Allis shad	*Alosa alosa*	Spain to Germany	29 (Bagliniere et al. 2003)	16 (Bagliniere et al. 2003)	LC (IUCN 2008)
Twaite shad	*Alosa fallax*	Morocco to Lithuania	About 35 (Aprahamian et al. 2003)	About 30 (Apraha-mian et al. 2003)	LC (IUCN 2008)
European eel	*Anguilla anguilla*	Morocco to Scandinavia	1 (panmictic)	1 (panmictic)	CR (IUCN 2008)
European whitefish	*Coregonus lavaretus*	Arctic Ocean to Denmark	DD	DD	VU (IUCN 2008)
Houting	*Coregonus oxyrinchus*	England to Germany	About 4 (Freyhof and Schöter 2005)	0 (Freyhof and Schöter 2005)	EX (IUCN 2008)
Arctic char	*Salvelinus alpinus*	Arctic Ocean to Sweden	DD	DD	See above
Atlantic salmon	*Salmo salar*	Portugal to Greenland	2015 (of which 206 are DD; WWF 2001)	1809 (of which 1572 are DD; WWF 2001)	See above
Sea trout	*Salmo trutta*	Russia to Portugal	DD	DD	LC (IUCN 2008)
European smelt	*Osmerus eperlanus*	France to Russia	DD (21 England) (Maitland 2003)	DD (14 England) (Maitland 2003)	LC (IUCN 2008)

CR, critically endangered; DD, data deficient; EN, endangered; EX, extinct; LC, least concern; LR, lower risk; LR/Ic, lower risk taxa that do not qualify for conservation-dependent or near-threatened status; LR/nt, lower risk taxa close to qualifying as vulnerable; NT, near threatened; SC, species of concern; VU, vulnerable.

a. Agency designations by the National Marine Fisheries Service and the Atlantic States Marine Fisheries Commission.

Note: Populations are assumed to be reproducing; multiple tributary populations in a single drainage are considered part of one population.

could be made, all have reduced numbers of populations (table 1). Strongly managed North American fishes such as Atlantic sturgeon, shortnose sturgeon (*Acipenser brevirostrum*), and striped bass (*Morone saxatilis*) had lost few populations. Where data allow cross-continental comparisons, Atlantic salmon in Europe have suffered relatively fewer population extirpations (13%) than in North America (33%). Alosine herrings have lost moderate numbers of populations on both sides of the Atlantic, but as much as nearly half for American shad and allis shad (*Alosa alosa*). Anadromous whitefishes

Table 2. Characteristics of time series data for selected diadromous fishes.

Species	Unit of measurement	Maximum value	Year of maximum	Minimum value	Year of minimum	Period of record	Location	Slope	R^2 of slope	Percentage increase or decrease or (fitted)	Long–term increase or decline	Reference
Eastern Atlantic												
Alosa alosa	Abundance	277,637	1886	0	1933	1880–1934	Rhine River, Netherlands	–0.1519	0.87	–99.94	D (E)	Bagliniere et al. 2003
Alosa alosa	Abundance	115,974	1925	120	1988	1914–1990	Minho River, Portugal	–0.0710	0.82	–99.48	D	Bagliniere et al. 2003
Alosa alosa	Metric tons	860.7	1967	0	1992	1961–1993	Oued Sebou, Morocco	–0.1326	0.92	–98.13	D (E)	Bagliniere et al. 2003
Alosa alosa	Abundance	106,706	1996	2979	2007	1985–2007	Garonne River, France	–0.2195	0.93	–95.37	D	Migado (*www.migado.fr*)
Alosa fallax	Abundance	1,174,137	1938	283	1947	1893–1950	Rhine River, Netherlands	–0.5669 (*)	0.85	–99.80	D	de Groot 2002
Anguilla anguilla	Abundance	48,615	1976	375	2004	1975–2005	Imse River, Norway	–0.1139	0.93	–91.84	D	EIFAC/ICES 2006
Anguilla anguilla	Kilograms	8011	1953	30	1998	1950–2005	Swedish eel rivers	–0.0554	0.97	–92.60	D	EIFAC/ICES 2006
Anguilla anguilla	Kilograms	6215	1960	5	1997	1951–2005	Ems and Vidå River, Denmark	–0.0673	0.72	–95.48	D	EIFAC/ICES 2006
Anguilla anguilla	Metric tons	49.37	1979	0.88	2005	1960–2005	British Isles	–0.0588 (*)	0.96	–65.30	D	EIFAC/ICES 2006
Anguilla anguilla	Number per haul	138	1963	0.58	2001	1950–2005	Den Oever River, Netherlands	–0.0625	0.79	–94.70	D	EIFAC/ICES 2006
Anguilla anguilla	Kilograms	946	1974	0.831	2004	1964–2005	Ijzer River, Belgium	–0.1612	0.93	–99.51	D	EIFAC/ICES 2006
Anguilla anguilla	Metric tons	1137	1979	10.86	2005	1950–2005	French rivers	–0.0902 (*)	0.96	–88.52	D	EIFAC/ICES 2006
Anguilla anguilla	Metric tons	88.89	1981	0.51	2004	1953–2005	Iberian Peninsula	–0.1085 (*)	0.98	–90.81	D	EIFAC/ICES 2006
Anguilla anguilla	Metric tons	11	1975	0.02	2002	1975–2005	Tiber River, Italy	–0.2121	0.82	–99.06	D	EIFAC/ICES 2006
Acipenser sturio	Metric tons	58	1950	0.11	1966	1891–1980	Eider, Gironde, and Guadalquivir Rivers, Europe	–0.2372 (*)	0.93	–99.31	D	Williot et al. 2002
Acipenser spp.	Metric tons	765.3	1927	0.5	1991	1920–1999	Danube River	–0.0416	0.78	–93.58	D	Williot et al. 2002
Acipenser spp.	Metric tons	32,000	1977	2	2002	1913–2002	Ponto–Caspian	–0.077 (*)	0.92	–72.99	D	Williot et al. 2002, Pikitch et al. 2005
Lampetra fluviatilis	Metric tons	44	1890–1899	0.6	1980–1989	1887–1999	Southern Baltic Sea	–0.0343	0.45	–96.29	D	Thiel et al. 2005
Petromyzon marinus	Metric tons	130,252	1897	84	1979	1887–1999	Southern Baltic *Sea*	–0.0375	0.50	–97.98	D	Thiel et al. 2005
Lamprey	Scaled relative abundance	2.2	2004	–0.95	1994	1986–2005	Garonne and Adour Rivers, France	0.0758	0.73	+230	I	Beaulaton et al. 2008
Salmo salar	Abundance	5707	1928	552	2000	1928–2004	River Dee, Wales	–0.0206	0.69	–77.31	D	Aprahamian et al. 2008
Salmo salar	Abundance	104,000	1885	0	1957	1863–1957	Rhine River, Netherlands	–0.0526	0.70	–98.97	D	de Groot 2002
Salmo salar	Metric tons	3032	1967	912	1997	1960–2005	North Europe	–0.0217	0.79	–62.34	D	WGNAS 2006
Salmo salar	Metric tons	4604	1973	778	2005	1960–2005	South Europe	–0.0397	0.86	–83.25	D	WGNAS 2006
Salmo salar	Metric tons	160	1971	9	2002	1960–2005	Faroes and Greenland	–0.1736 (*)	0.89	–99.81	D	WGNAS 2006
Salmo trutta	Abundance	25,244	2004	5096	1987	1987–2007	Iceland	0.0439	0.93	+220	I	Gudbergsson 2007

Table 2. (continued)

Species	Unit of measurement	Maximum value	Year of maximum	Minimum value	Year of minimum	Period of record	Location	Slope	R^2 of slope	Percentage increase or decrease or (fitted)	Long–term increase or decline	Reference
Western Atlantic												
Acipenser oxyrhinchus	Metric tons	3294	1888	6	1924	1880–1994	North America	–0.0239	0.28	–91.05	D	Kahnle et al. 2007
Alosa sapidissima	Metric tons	22,408	1814	18	1892	1814–2005	North America	–0.0189	0.67	–97.14	D	ASMFC 2007
Alosa pseudo-harengus	Metric tons	16,148	1958	7.5	2006	1950–2006	North America	–0.0829	0.86	–98.76	D	NOAA Fisheries statistics
Alosa aestivalis	Metric tons	23,800	1969	109.9	2006	1950–2006	North America	–0.0963	0.90	–99.39	D	NOAA Fisheries statistics
Alosa mediocris	Metric tons	303.8	1952	5.6	1990	1950–2006	North America	–0.0323	0.36	–81.95	D	NOAA Fisheries statistics
Anguilla rostrata	Metric tons	1792.6	1979	290.9	2002	1950–2006	North America	–0.0533 (*)	0.99	–72.20	D	NOAA Fisheries statistics
Osmerus mordax	Metric tons	163	1966	0.1	1997	1950–2004	North America	–0.0852	0.67	–99.08	D	NOAA Fisheries statistics
Salmo salar	Metric tons	2864	1967	132	2005	1960–2005	North America	–0.0736	0.82	–96.36	D	WGNAS 2006
Morone saxatilis	Metric tons	6704	1973	100	1989	1950–2006	North America	0.1635	0.85	+1.368	I	NOAA Fisheries statistics

D, decline; E, extirpated; I, increase.

Note: Slopes were calculated from normalized data that had been smoothed with running averages corresponding to generation times, and then log-transferred. Generation times: alosines, salmons, and brown trout, 4 years; smelt, 2 years; striped bass, 6 years; lampreys, 9 years; eels, 10 years; sturgeons, 15 years. Slopes with an asterisk (*) indicate that they were calculated after a clear peak or nadir (e.g., after a "fishing up" period or following a collapse and subsequent recovery). Percentage increase or decrease is calculated with the fitted slope, and include the most recent years in the time series. Type of record was catch for all species except *Alosa alosa* (fish passage), *Anguilla anguilla* (recruitment index), and lamprey (fishery).

(*Coregonus* spp.) are in perilous condition: Only a small and now purposely landlocked population of *Coregonus huntsmani* persists in Nova Scotia and a European species, *Coregonus oxyrinchus*, has become extinct (Freyhof and Schöter 2005). Also, sea sturgeon (*Acipenser sturio*), once found in as many as 18 major rivers over much of Europe, now verges on extinction, remaining only as a small, struggling population in France's Gironde River (Williot et al. 2002).

Abundances of populations

Although some anadromous species have suffered numerous extirpations, the predominant pattern among these fishes has been for continuation of runs, but at drastically reduced levels that may be trending to inviability, as low as about 100 individuals for shortnose sturgeon in two populations (Kynard 1997). These declines have also been manifested—often profoundly so, especially with many long-exploited fish populations—in reduced biomass, age distributions, age at maturity, and maximum size and growth (Law 2007).

The length and quality of time series data sets vary, but the trend is nearly always the same: Diadromous fishes have declined, often to historic lows (figure 1, table 2). Of the 35 species or stocks for which we were able to obtain time series data sets, 32 had declined and only 3 had increased (table 2). Where long-term records exist, losses from baseline levels are often dramatic. American shad offers a good example, as data on the Potomac River date back to 1814, but the baseline for restoration efforts is derived from US Fisheries Commission records, which began in 1887 (figure 2a). The highest catch (51,136,364 kg) occurred in 1832 (figure 2b; Massman 1961). The Potomac could produce more than 22 million shad (3 kg in weight and 0.9 meters [m] in length, on average, versus approximately 1.8 kg and 0.5 m today) "in a good year" (Tilp 1978); today, only a minor recreational fishery persists there. Time series of American shad landings (normalized to kilometers of available river or estuary) for 10 major producing areas show a long-term exponential decline with a slope of –0.035 per year with all the data ($R^2 = 0.33$, $p < 10^{-5}$), or –0.033 per year if the early Potomac landings are excluded ($R^2 = 0.26$, $p < 10^{-5}$).

Comparisons of landings between diadromous fish taxon pairs from both sides of the Atlantic often show similar patterns (figure 1). These include moderate to sharp declines in the 1900s (some with occasional short-term recoveries), followed by low harvests or a mandated cessation of fishing, that continue to the present. Not only do most diadromous species exhibit precipitous declines over time, but the differences between maximum levels and recent

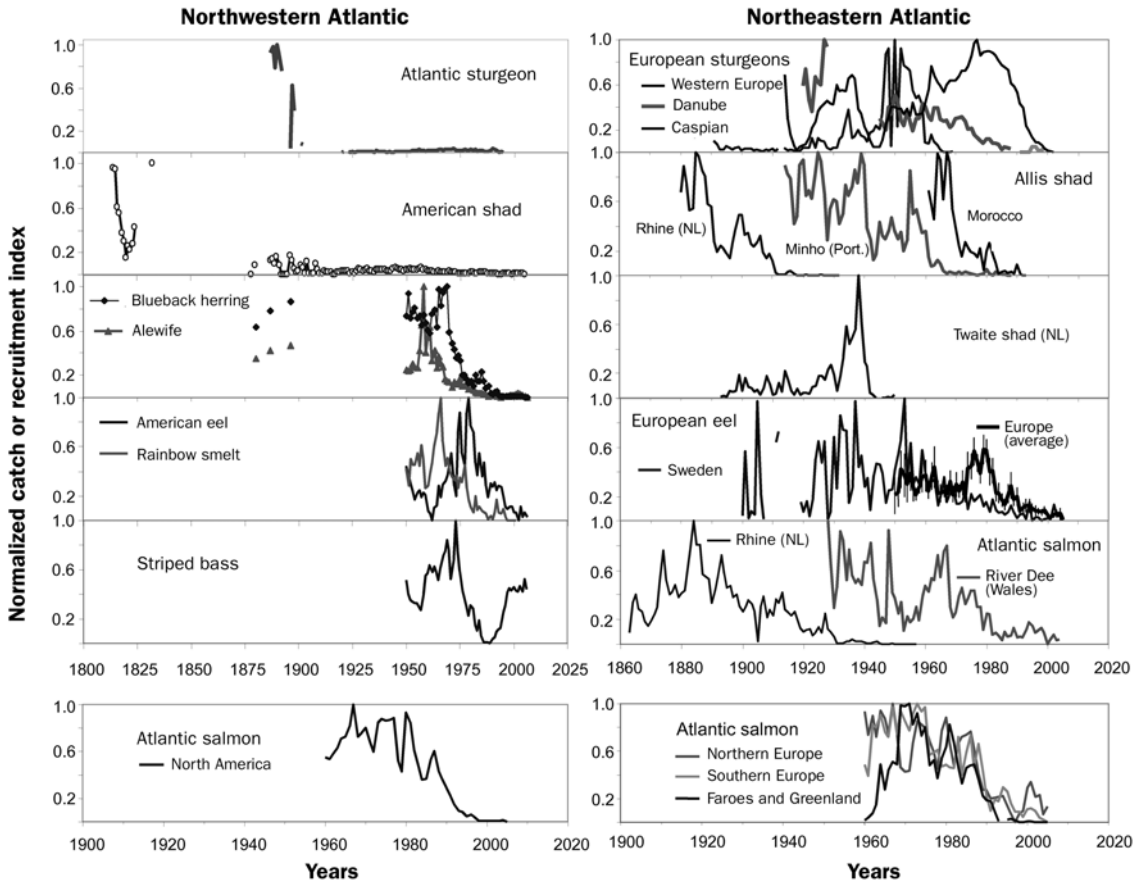

Figure 1. Normalized time series of indices of abundance of selected north Atlantic diadromous species. European eel includes standard errors of means for nine regions. The lower two panels compare Atlantic salmon. For type of index, maxima, minima, percentage change, and data sources, see table 2. Unless otherwise stated, northwestern Atlantic data are US summary statistics.

ones are even greater than what has been observed in many obligate marine species. Thirteen of the 35 time series in table 2 had declined by more than 98%; another 11 had declined by more than 90%. The few exceptions include the coastal migratory stock of striped bass, northern European populations of Atlantic salmon, and Icelandic populations of sea-run brown trout (*Salmo trutta*). This last example shows a marked increase in records over the smoothed observation period (1991–2007), and may be attributable to a true increase in population or an increase in sport fishing, or both (Gudbergsson 2007).

Conservation status

We believe the conservation status of anadromous fishes integrates knowledge of population persistence, abundance, and threats. Of the 12 exclusively North American species, the International Union for the Conservation of Nature (IUCN) Red List classifies 1 as endangered and 2 as vulnerable; the National Marine Fisheries Service lists 3 others as species of concern; and the ASMFC rates 1 more as having its lowest abundance in history, and is in the process of assessing 2 more species that are also likely at historic lows. Of the 9 eastern Atlantic species, 1 has gone extinct, 2 are now critically endangered (including the once abundant European eel), 1 is vulnerable, and 2 are listed by the IUCN as data deficient (table 1). At least one (*A. alosa*) appears to be in serious decline, although noted as "least concern" by the IUCN. Of the pan-Atlantic salmonids (Atlantic salmon and arctic char, *Salvelinus alpinus*), wild *S. salar* is at historic lows in North America, and overall, its status is in need of updating (IUCN 2008).

Threats

North Atlantic diadromous fishes must navigate a gauntlet of threats. The primary triad that affects most taxa is damming of rivers, overfishing, and pollution. However, there are now a host of threats beyond the three that have long been considered primary.

Dams and other habitat losses. Industrialization depended on rivers for water power, and many waterways became multiply dissected with dams. Dams often block access to historical spawning reaches, causing population reductions and extirpations. Few larger rivers remain undammed: It is estimated

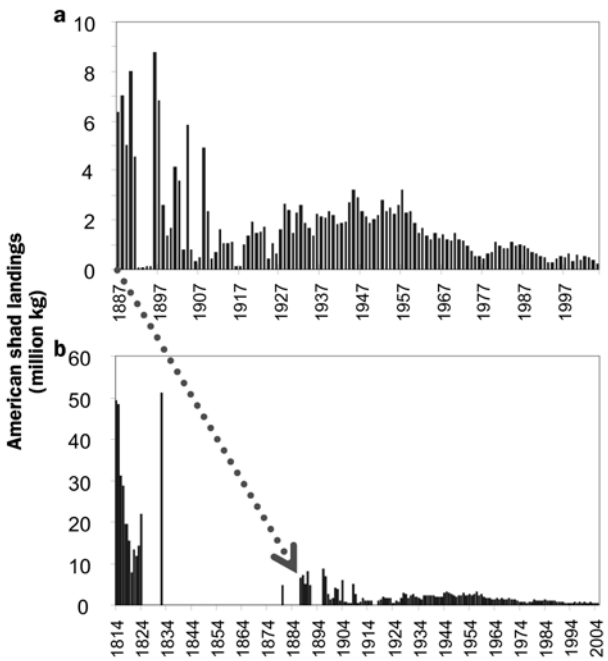

Figure 2. Example of how baselines shift. (a) Baseline for American shad restoration is typically referenced to 1887, when the US Fishery Commission began to collect statistics. (b) Earlier data show that levels for the 1887 baseline are considerably lower than they were in the past. Source: ASMFC (2007).

that in the United States alone, there are more than 80,000 dams of 6 feet in height or more, and perhaps as many as 2,000,000 of all sizes (Graf 2003). For example, within the Hudson River watershed there are 797 registered dams (Swaney et al. 2006); that figure does not include small dams (< 0.6 m tall), which also can hinder migration. In Spain, some dams have blocked fish movements continuously since the 2nd century, and the nations of Europe together have about 7000 large (more than 15 m) dams, most of which are situated on Atlantic drainages. Engineered solutions to fish passage in the form of ladders and lifts have been fitted to some dams, but generally passage is species specific, and the number of fish traveling through them is far fewer than it would be in the absence of dams; these dams also inhibit downstream migration of young. One useful metric of the effect of dams is the number of kilometers of river they occlude to migrants. For American shad, approximately 4000 of an original 11,200 km of spawning habitat have been lost to dams (Limburg et al. 2003); these dams have similar effects on other anadromous species.

Dams also have numerous other ecological effects on rivers, many of which may affect diadromous fishes directly or indirectly. Among these are the blocking of normal movements and changes in the community composition of resident fishes that interact with diadromous fishes; microevolution of populations isolated by barriers; pronounced alterations of water temperatures upriver and downriver; retention of nutrients and sediments; and, even where fish passage is successful, the imposition of the need to cross sometimes large, unnatural stillwater habitats (Helfman 2007). Dams that are operated for hydropower also cause direct mortality (death by turbines) and may radically alter water discharges (Helfman 2007)—and hence, habitat availability (water or no water)—on daily or even hourly timescales.

In addition to the large habitat changes wrought by dams, dredging and channelization may cause short-term stresses while these activities occur and, more important, long-term diminution of habitat quality through the changes they create. Culverts impede fish movements by species such as river herring in smaller systems. Gravel and water removals reduce habitat in many waterways. Because many anadromous fishes use rivers as nurseries, reductions in the extent and quality of marshes and other shallow water habitats may lessen productivity and, therefore, recruitment.

Overfishing. Harvest has strongly compromised diadromous fish populations. Atlantic sturgeon were taken at an extraordinary rate during the international caviar craze of the 1890s (Secor and Waldman 1999); with continued fishing and their low intrinsic rate of increase, many populations have shown little subsequent recovery, despite greater protection. In the Delaware River, the chief US fishery for Atlantic sturgeon, landings in 1901 were only 6% of their 1889 peak of more than 2000 metric tons (Secor and Waldman 1999). Atlantic sturgeon remain so scarce in the Delaware that it is not known whether any reproduction still occurs there.

Overfishing is a major factor in the nearly complete demise of the once-widespread European sea sturgeon (Williot et al. 2002). Extirpations led to a range contraction to just the Gironde estuary in France, and even when fishing was halted there in 1982, the population continued to decline. Despite regulatory protection, accidental bycatch threatens sturgeons on both the American and European coasts.

Alewives were once so numerous in northeastern US rivers that they were likened to "passenger pigeons of the sea" (Bolster 2006); their numbers have since plummeted, and several states have banned any takings. Runs in several large rivers from Maine to the Chesapeake Bay have declined by 99.9%; for example, at the Holyoke Dam on the Connecticut River, counts went from approximately 630,000 in 1985 to 21 in 2006. Bycatch at sea is one likely contributor, as subadults are taken along with the targeted Atlantic herring (*Clupea harengus*) fisheries. Another alosine that appears to be undergoing a similar collapse because of recruitment overfishing is the allis shad; juvenile recruitment in the Gironde, the center of its range, has been negligible for the past few years.

Extensive analysis of decadal trends in eel fisheries suggests that exploitation is a major factor in European eel decline (Dekker 2004), with many fisheries collapsed. Eels are targeted not only as immature (yellow phase, in lakes and running waters) or adolescent (silver phase, migrating toward the Sargasso Sea to spawn) but also as postlarval glass eels entering continental waters. The highly lucrative glass eel fishery is

driven by demand in Southeast Asia, where imported American and European glass eels are pond-reared to market size. Glass eel fisheries sometimes harvest all available individuals at a particular locale, but in general the harvest has been 80% to 95% (Dekker 2004), which is still an alarming statistic.

Pollution. Water pollution also has reduced runs of diadromous fishes. Some river systems received so much raw or lightly treated human sewage—which induced low oxygen levels—that they became equivalent to "chemical dams" blocking spawning migrations. Examples include the Thames in the United Kingdom and the Delaware River in the United States (Chittenden 1971); however, both rivers have shown dramatic improvements as a result of new laws and management actions. Over the past few decades, shortnose sturgeon has made an unusually robust recovery in the Hudson River not only because of its placement on the US endangered species list but also because the population's original spawning location near the head of tidewater was reoxygenated through measures to control sewage, which stemmed from the Clean Water Act of 1972 (Waldman 2006). However, late 20th-century exurbanization (sprawl development) has led to more impervious surface cover in many drainage basins, further altering water quantity and quality.

Contaminants such as polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons, and heavy metals may induce difficult-to-quantify sublethal effects in fishes in riverine environments. Highly biomagnified levels of PCBs in boreal regions are causing concerns for Artic char. Laboratory experiments with Arctic char have shown that these compounds impair hypo-osmoregulatory ability and reduce growth rate and survival upon transfer to seawater (Jørgensen et al. 2004). Also, European and American eel reproduction may be compromised by fat-soluble, teratogenic organic compounds (Palstra et al. 2006), which are translocated into developing embryos from maternal lipid stores.

Acidification from atmospheric deposition of contaminants has been devastating for some Atlantic salmon stocks. In Norway, 18 populations are extirpated and 8 more are threatened, with others sustained only by liming rivers to raise pH (Sandøy and Langåker 2001).

Climate change. Climate change is altering species distributions. The boreal rainbow smelt *Osmerus mordax*, which in the 1880s ran in US rivers as far south as the Delaware, was extirpated from the Hudson in the 1990s (Waldman 2006) and is becoming scarce everywhere south of Maine. Meanwhile, gizzard shad (*Dorosoma cepedianum*), a euryhaline clupeid of no commercial value and uncertain ecological effects, has been colonizing rivers northward, establishing in large numbers in the Hudson in the 1970s and recently reaching as far as Maine (Waldman 2006).

Warming also appears to be shifting the phenologies of anadromous fishes towards earlier spawning runs. Monitoring in Maine revealed that the median capture date for Atlantic salmon in the Penobscot River advanced by 1.3 days per year between 1986 and 2001, and by 1.2 days per year between 1983 and 2001 for alewife in the Androscoggin River (Huntington et al. 2003). The consequences of such acceleration are unknown, but the rapidity of the change has the potential to disrupt these fishes' established ecological relationships at various life history stages.

In the future, warming may intensify the severity of floods and droughts, lessening the frequency of successful annual reproduction for anadromous fishes. In Europe, models predict that, collectively, 22 species will lose 336 suitable catchments and gain only 113 as a result of the most likely climate change scenario (Lassalle and Rochard 2009). The Gulf sturgeon (*Acipenser oxyrinchus desotoi*) depends on limited numbers of cool thermal springs to survive hot summer temperatures in Gulf of Mexico rivers (Carr et al. 1996); warming may impose even greater stresses on this scarce and federally threatened subspecies.

Warming will also impose complex and difficult-to-forecast shifts in the relationships between freshwater and saltwater habitats. Both American and European eels have evolved to capitalize on the transport and trophic resources of the Gulf Stream. However, the recent effects of climate change on this current may be contributing to the declines seen in both eel species in freshwaters (Wirth and Bernatchez 2003). In Arctic regions, warming may increase the productivity of inshore marine habitats used by anadromous fishes, but this may be counterbalanced by decreased flows in spawning rivers. Increased productivity of inland waters may also reduce facultative anadromy for plastic species such as Arctic char, with higher proportions of populations opting for freshwater residency (Reist et al. 2006).

Other threats

Electric generating plants and other facilities that withdraw water from rivers may kill high numbers of early life stages of diadromous fishes through entrainment and by impinging larger individuals against intake screens; power plants may also alter local temperature regimes though discharges of warm water (Barnthouse et al. 1988). Disease, competition, and genetic introgression with escapees from aquacultured Atlantic salmon threaten wild stocks in northeastern North America and Scandinavia (Naylor et al. 2005). Progeny of Atlantic sturgeon used in experimental culture have been opportunistically stocked in the wild (St. Pierre 1999) while ignoring protocols for the maintenance of appropriate effective population sizes. Similarly, research-culture escapees of a nonnative sturgeon species now compete in the Gironde with the few remaining sea sturgeon (Maury-Brachet and Rochard 2008). Many invasive and nonnative species also disrupt lotic ecology. Introduction of black bass (*Micropterus* spp.) and other piscivores increased the predation regime for juvenile alosines and other young diadromous fishes in US rivers. Invasive zebra mussels (*Dreissena polymorpha*) have altered the Hudson River's spring production cycle, to the detriment of its alosines (Strayer et al. 2004).

Conclusions

Few of the North Atlantic's diadromous fishes face any of the abovementioned threats in isolation; rather, it is likely that reasons for the losses we have outlined are multifactorial, and possibly synergistic. Many of these declines have been steady and insidious, fitting well into the "shifting baselines" paradigm, whereby new generations of managers accept that recent environmental conditions and levels of species reflect historical conditions and levels, and set restoration goals accordingly (Humphries and Winemiller 2008, Waldman 2008). Loss of historical baselines contributes to marginalization of the species, as social customs relating to bygone (collapsed) fisheries also perish, and ecosystems unravel at rates that go unnoticed.

Especially troublesome is the outright loss of many populations and their genetic legacies in the face of changing environments. The high phylogenetic diversity of these 24 species and the differences in life histories, geographic ranges, and commercial values conspire to make generalized solutions impossible. There is a strong need for better information on the population-specific status of many species of low commercial interest. Harvests of some species have been reduced and moratoria have even been applied, but usually not until abundances had become dangerously low. Atlantic coast populations of migratory striped bass are one of the few successful recoveries for an anadromous species, but the severe measures needed to generate this recovery were not taken until the stock fell to crisis levels (Richards and Rago 1999). Even with moratoria, populations may fail to recover (e.g., *A. sturio* in the Gironde, *A. sapidissima* in Chesapeake Bay), suggesting changes occurring systemwide are collectively hindering recovery.

Fishermen and other stakeholders need to elevate their long-term interests in a species' welfare over their own short-term economic interests, with the understanding that the more the populations are fished, the less the likelihood of recovery (and the lengthier the period of recovery), and hence the more damage to the future sustainability of the fishery. A laudatory example of an early intervention is the moratorium imposed in late 1997 on Atlantic sturgeon fishing in US waters in response to indications that some populations were rapidly declining because of suddenly increased fishing pressure (Waldman 2006). Almost exactly a century after the international caviar craze left many US stocks sharply reduced or decimated, the few remaining commercial Atlantic sturgeon fishermen acquiesced to an ambitious protection plan that prohibits their take for up to 40 years—two generations for this slowly maturing species.

The environmental movement has resulted in a reduction of new sources of pollution in the United States and Europe, but many rivers still have a legacy of contaminants produced from the Industrial Revolution through the mid-1900s. Although cleanup actions have been helpful for some species in some places, the single broadest and most useful recovery action has been to remove dams wherever possible. This is especially true for large mainstem dams. For example, when the Edwards Dam on Maine's Kennebec River was removed in 1999, the benefits to the full suite of this river's diadromous fishes were almost immediately visible as the fishes reoccupied their historical spawning grounds. Where dams cannot be removed, it is far preferable to install fish passage devices, despite their flaws, than to impede the movements of all diadromous fishes in a river. Research to enable passage of anadromous species that shun conventional fish ladders, such as sturgeons, should also be encouraged.

Viewed collectively, North Atlantic diadromous species underwent similar sequences of events that led to their declines (figure 3). Although quantitative data are largely lacking, anecdotal evidence from diaries, journals, and other historical accounts suggests that pristine populations of diadromous fishes were staggering in their plenitude (Waldman 2008), and formed the basis of important fisheries. Gradually, some populations became extirpated, but the pace of extirpations through the mid-20th century was slow enough to forestall great alarm (but note that overfishing of American shad in the 19th century spurred concerted management efforts).

The cumulative impacts resulted in declines, but these declines in themselves have had another unintended consequence: namely, a loss of standing or "saliency" among issues considered important by society at large. As species became scarce, fisheries declined, and often demand dropped off. Other watershed uses gained prominence. As demand dwindles and constituencies are lost, it becomes increasingly difficult to motivate and secure funding for adequate management and restoration measures. This downward spiral of

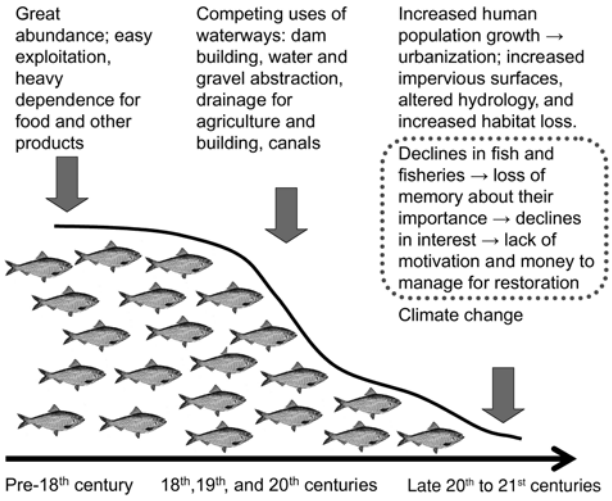

Figure 3. Conceptual diagram of the general history and factors leading to declines in North Atlantic diadromous species. Most species were heavily exploited before industrialization and physical alteration of waterways; further watershed alterations due to human population expansion and climate change increased habitat loss. Gradually, the declines also led to the loss of institutional and societal memory about past abundance and importance (outlined for emphasis).

events lacks a term, but we suggest that it is a kind of *ecosocial anomie,* a breakdown both of expectations of what species should be present in healthy populations, and societal loss of interest. The result is not only the loss of populations and species but also the loss of services the species provided when their inland ecosystems were more intact.

The stories of individual stocks that perished or are commercially extinct are numerous, but it is clear that the diminishment of diadromous fishes, taken as a group, represents one of the greatest corruptions of the ecological connections between North American and European watersheds and the North Atlantic ecosystem. Although management needs to consider the specifics of each species and population, the causes of decline we have outlined appear to be general and widespread. If there is to be a future for this group, societies must make difficult decisions concerning the trade-offs between maintaining healthy populations within healthy ecosystems and taking actions that degrade and imperil those systems. The emerging field of ecosystem service quantification may provide a means to enhance restoration, since it highlights those services that depend on ecosystem function as well as provisioning services. If ecosystem service quantification becomes mainstreamed (Cowling et al. 2008), local and regional decisionmaking would have an alternative to conventional cost-benefit schemes. These alternatives would support ecosystem and habitat restoration. It may take decades to bring back diadromous species, but restoring the watersheds and their connectivity with coastal marine ecosystems is a critical first step in that direction.

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References cited

Aprahamian MW, Bagliniere J-L, Sabatie MR, Alexandrino P, Thiel R, Aprahamian CD. 2003. Biology, status, and conservation of the anadromous Atlantic twaite shad *Alosa fallax fallax*. American Fisheries Society Symposium 35: 103–124.

Aprahamian MW, Davidson IC, Cove RJ. 2008. Life history changes in Atlantic salmon from the River Dee, Wales. Hydrobiologia 602: 61–78.

[ASMFC] Atlantic States Marine Fisheries Commission. 2007. Terms of Reference and Advisory Report of the American Shad Stock Assessment Peer Review. ASMFC. Stock Assessment Report no. 07-01.

Bagliniere J-L, Sabatie MR, Rochard E, Alexandrino P, Aprahamian MW. 2003. The allis shad *Alosa alosa*: Biology, ecology, range, and status of populations. American Fisheries Society Symposium 35: 85–102.

Barnthouse LW, Klauda RJ, Vaughn DS, Kendall RL, eds. 1988. Science, Law, and Hudson River Power Plants. American Fisheries Society. Monograph 4.

Beamish FWH. 1980. Biology of the North American anadromous sea lamprey, *Petromyzon marinus*. Canadian Journal of Fisheries and Aquatic Sciences 37: 1924–1943.

Beaulaton L, Tavery C, Castelnaud G. 2008. Fishing, abundance and life history traits of the anadromous sea lamprey (*Petromyzon marinus*) in Europe. Fisheries Research 92: 90–191.

Bolster WJ. 2006. Opportunities in marine environmental history. Environmental History 11: 567–597.

———. 2008. Putting the ocean in Atlantic history: Maritime communities and marine ecology in the Northwest Atlantic: 1500–1800. American Historical Review 113: 19–47.

Carr SH, Tatman F, Chapman FA. 1996. Observations on the natural history of the Gulf of Mexico sturgeon (*Acipenser oxyrinchus desotoi* Vladykov 1955) in the Suwannee River, southeastern United States. Ecology of Freshwater Fish 5: 169–174.

Casey JM, Myers RM. 1998. Near extinction of a large, widely distributed fish. Science 281: 690–692.

Chittenden ME Jr. 1971. Status of the striped bass, *Morone saxatilis,* in the Delaware River. Chesapeake Science 12: 131–136.

Christensen V, Guénette S, Heymans JJ, Walters CJ, Watson R, Zeller D, Pauly D. 2003. Hundred-year decline of North Atlantic predatory fishes. Fish and Fisheries 4: 1–24.

Cowling RM, Egoh B, Knight AT, O'Farrell PJ, Reyers B, Rouget M, Roux DJ, Welz A, Wilhelm-Rechman A. 2008. An operational model for mainstreaming ecosystem services. Proceedings of the National Academy of Sciences 105: 9483–9488.

Daily GC, ed. 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press.

de Groot SJ. 2002. A review of the past and present status of anadromous fish species in the Netherlands: Is restocking the Rhine feasible? Hydrobiologia 478: 205–218.

Dekker W. 2004. Slipping through our hands: Population dynamics of the European eel. PhD dissertation, University of Amsterdam.

[EIFAC/ICES] European Inland Fisheries Advisory Commission/International Council for Exploration of the Sea. 2006. Report of the 2006 session of the joint EIFAC/ICES working group on eels. 23–27 January 2006, Rome; European Inland Fisheries Advisory Commission, EIFAC Occasional Paper no. 38. FAO/ICES. (28 October 2009; *www.ices.dk/reports/ACFM/2006/WGEEL/WGEEL06.pdf*)

Elvira B, Amodóvar A, Birstein VJ, Gessner J, Holčík J, Lepage M, Rochard E, eds. 2000. Symposium on Conservation of the Atlantic Sturgeon *Acipenser sturio* L., 1758 in Europe. Boletín Instituto Espanol de Oceanografia 16.

Freyhof J, Schöter C. 2005. The houting *Coregonus oxyrinchus* (L.) (Salmoniformes: Coregonidae), a globally extinct species from the North Sea basin. Journal of Fish Biology 67: 713–729.

Fruge DJ, et al. 2006. The Striped Bass Fishery of the Gulf of Mexico. United States: A Regional Management Plan. Gulf States Marine Fisheries Commission. Report no. 137.

Garman GC. 1992. Fate and potential significance of postspawning anadromous fish carcasses in an Atlantic coastal river. Transactions of the American Fisheries Society 121: 390–394.

Gende SM, Edwards RT, Willson MF, Wipfli MS. 2002. Pacific salmon in aquatic and terrestrial ecosystems. BioScience 52: 917–928.

Graf WL, ed. 2003. Dam Removal Research: Status and Prospects. Heinz Center.

Gross MR, Coleman RM, McDowell RM. 1988. Aquatic productivity and the evolution of diadromous fish migration. Science 239: 2392–2393.

Gudbergsson G. 2007. Icelandic Salmon, Trout and Charr Catch Statistics 2007. Institute of Freshwater Fisheries. VMST/08024.

Helfman GS. 2007. Fish Conservation: A Guide to Understanding and Restoring Global Aquatic Biodiversity and Fishery Resources. Island Press.

Humphries P, Winemiller KO. 2009. Historical impacts on river fauna, shifting baselines, and challenges for restoration. BioScience 59: 673–684.

Huntington TG, Hodgkins GA, Dudley RW. 2003. Historical trend in river ice thickness and coherence in hydroclimatological trends in Maine. Climatic Change 61: 217–236.

[IUCN] International Union for the Conservation of Nature. 2008. IUCN Red List of Threatened Species, 2008. (29 October 2009; *www.iucnredlist.org/*)

Jørgensen EH, Aas-Hansen Ø, Maule AG, Espen Tau Strand JE, Vijayan MM. 2004. PCB impairs smoltification and seawater performance in anadromous Arctic char (*Salvelinus alpinus*). Comparative Biochemistry and Physiology C: Toxicology and Pharmacology 138: 203–212.

Kahnle AW, Hattala KA, McKown K. 2007. Status of Atlantic sturgeon (*Acipenser oxyrinchus*) of the Hudson River Estuary, New York, USA. Pages 347–364 in Munro J, Hatin D, Hightower JE, McKown K, Sulak KL, Kahnle AW, Caron F, eds. Anadromous Sturgeons: Habitats, Threats, and Management. American Fisheries Society. Symposium 56.

Kynard B. 1997. Life history, latitudinal patterns, and status of the shortnose sturgeon. Environmental Biology of Fishes 48: 319–334.

Lassalle G, Rochard E. 2009. Impact of 21st century climate change on diadromous fish spread over Europe, North Africa and the Middle East. Global Change Biology 15: 1072–1089.

Law R. 2007. Fisheries-induced evolution: Present status and future directions. Marine Ecology Progress Series 335: 271–277.

Limburg KE, Landergren P, Westin L, Elfman M, Kristiansson P. 2001. Flexible modes of anadromy in Baltic sea trout (*Salmo trutta*): Making the most of marginal spawning streams. Journal of Fish Biology 59: 682–695.

Limburg KE, Hattala KA, Kahnle AW. 2003. American shad in its native range. American Fisheries Society Symposium 35: 125–140.

Maitland PS. 2003. The Status of Smelt *Osmerus eperlanus* in England. English Nature. English Nature Research Reports no. 516.

Massmann WH. 1961. A Potomac River shad fishery, 1814–1824. Chesapeake Science 2: 76–81.

Maury-Brachet R, Rochard E. 2008. The 'Storm of the Century' (December 1999) and the accidental escape of Siberian sturgeons (*Acipenser baerii*) into the Gironde estuary (southwest France). Environmental Science and Pollution Research 15: 89–94.

McDowall RM. 1997. The evolution of diadromy (revisited) and its place in phylogenetic analysis. Reviews in Fish Biology and Fisheries 7: 443–462.

———. 2001. Anadromy and homing: Two life-history traits with adaptive synergies in salmonid fishes? Fish and Fisheries 2: 78–85.

Mettee MF, O'Neil PE. 2003. Status of Alabama shad and skipjack herring in Gulf of Mexico drainages. American Fisheries Society Symposium 35: 157–170.

[NMFS] National Marine Fisheries Service. 1998. Recovery Plan for Shortnose Sturgeon (*Acipenser brevirostrum*). Prepared by the Shortnose Sturgeon Recovery Team (NMFS 2008).

———. 2009. Species of Concern: River Herring (Alewife and Blueback Herring). NMFS. (17 November 2009; *www.nmfs.noaa.gov/pr/pdfs/species/riverherring_detailed.pdf*)

Naylor R, Hindar K, Fleming IA, Goldburg R, Williams S, Volpe J, Whoriskey F, Eagle J, Kelso D, Mangel M. 2005. Fugitive salmon: Assessing the risks of escaped fish from net-pen aquaculture. BioScience 55: 427–437.

Ocean Studies Board, National Research Council. 2000. Improving the Collection, Management, and Use of Marine Fisheries Data. National Academy Press.

Palstra AP, van Ginneken VJT, Murk AJ, van den Thillart GEEJM. 2006. Are dioxin-like contaminants responsible for the eel (*Anguilla anguilla)* drama? Naturwissenshaften 93: 145–148.

Pikitch EK, Doukakis P, Lauck L, Chakrabarty P, Erickson DL. 2005. Status, trends, and management of sturgeon and paddlefish fisheries. Fish and Fisheries 6: 233–265.

Powles H, Bradford MJ, Bradford RG, Doubleday WG, Innes S, Levings CD. 2000. Assessing and protecting endangered marine species. ICES Journal of Marine Science 57: 669–676.

Reist JD, Wrona FJ, Prowse TD, Dempson JB, Power M, Köck G, Carmichael TJ, Sawatzky CD, Lehtonen H, Tallman RF. 2006. Climate change impacts on Arctic freshwater ecosystems and fisheries. Ambio 35: 402–410.

Richards RA, Rago PJ. 1999. A case history of effective fishery management: Chesapeake Bay striped bass. North American Journal of Fisheries Management 19: 356–375.

Ruffo S, Kareiva PM. 2009. Using science to assign value to nature. Frontiers in Ecology and the Environment 7: 3.

Sandøy S, Langåker RM. 2001. Atlantic salmon and acidification in southern Norway: A disaster in the 20th century, but a hope for the future? Water, Air, and Soil Pollution 130: 1343–1348.

Secor DH, Rooker JR. 2000. Is otolith strontium a useful scalar of life cycles in estuarine fishes? Fisheries Research 46: 359–371.

Secor DH, Waldman JR. 1999. Historical abundance of Delaware Bay Atlantic sturgeon and potential rate of recovery. American Fisheries Society Symposium 23: 203–216.

Stevenson CH. 1899. The shad fisheries of the Atlantic coast of the United States. Report of the Commissioner of the US Commission for Fish and Fisheries, pt. 24. US Government Printing Office.

St. Pierre RA. 1999. Restoration of Atlantic sturgeon in the northeastern USA with special emphasis on culture and restocking. Journal of Applied Ichthyology 15: 180–182.

Strayer DL, Hattala KA, Kahnle AW. 2004. Effects of an invasive bivalve (*Dreissena polymorpha*) on fish populations in the Hudson River estuary. Canadian Journal of Fisheries and Aquatic Sciences 61: 924–941.

Swaney DP, Limburg KE, Stainbrook K. 2006. Some historical changes in the patterns of population and land use in the Hudson River watershed. American Fisheries Society Symposium 51: 75–112.

Thiel R, Winkler HM, Riel P, Neumann R. 2005. Survey of river and sea lampreys in German waters of the Baltic Sea—basis of successful rebuilding programmes. ICES CM 2005/W: 06. International Council for the Exploration of the Seas.

Tilp F. 1978. This Was Potomac River. Willich.

Waldman JR. 2003. Introduction to the shads. American Fisheries Society Symposium 35: 3–9.

———. 2006. The diadromous fish fauna of the Hudson River: Life histories, conservation concerns, and research avenues. Pages 171–188 in Levinton JS, Waldman JR, eds. The Hudson River Estuary. Cambridge University Press.

———. 2008. The world's diadromous fishes: Resetting shifted baselines. Pages 151–163 in Fearn E, ed. State of the Wild 2008–2009: A Global Portrait of Wildlife, Wildlands, and Oceans. Island Press, for the Wildlife Conservation Society.

Waldman JR, Wirgin II. 1998. Status and restoration options for Atlantic sturgeon in North America. Conservation Biology 12: 631–638.

Warren LW Jr., et al. 2000. Diversity, distribution, and conservation status of the native freshwater fishes of the southern United States. Fisheries 25: 7–29.

[WGNAS] Working Group on North Atlantic Salmon. 2006. Report of the working group on North Atlantic salmon (WGNAS). ICES WGNAS Report 2006. ICES Advisory Committee on Fisheries Management. ICES CM. 2006/ACFM: 23. International Council for the Exploration of the Sea.

Williot P, et al. 2002. Status and management of Eurasian sturgeon: An overview. International Review of Hydrobiology 87: 483–506.

Willson MF, Halupka K.C. 1995. Anadromous fish as keystone species in vertebrate communities. Conservation Biology 9: 489–497.

Wirth T, Bernatchez L. 2003. Decline of North Atlantic eels: A fatal synergy? Proceedings of the Royal Society B 270: 681–688.

[WWF] World Wildlife Fund. 2001. The Status of Wild Atlantic Salmon: A River by River Assessment. (28 October 2009; *http://assets.panda.org/downloads/salmon2.pdf*)

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CALCULATING ACCEPTABLE BIOLOGICAL CATCH FOR STOCKS THAT HAVE RELIABLE CATCH DATA ONLY (Only Reliable Catch Stocks – ORCS)

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U.S. DEPARTMENT OF COMMERCE
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May 2011

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EXECUTIVE SUMMARY

In response to the 2006 reauthorization of the Magnuson-Stevens Act (MSA), the National Marine Fisheries Service established National Standard 1 (NS1) Guidelines, which included a requirement to set an acceptable biological catch (ABC) that accounts for scientific uncertainty in the estimate of a stock's overfishing limit (OFL). This is an exceedingly difficult task for the large number of stocks for which reliable catch data are the only information available, as these stocks cannot be assessed with traditional stock assessment methods. For the purpose of this document, these stocks will be called "only reliable catch stocks" (ORCS). Despite the inherent problem of setting ABCs for ORCS, the MSA requirement remains.

At the second National SSC meeting November 10-13, 2009 in St. Thomas, USVI, an ad-hoc Working Group was established to identify, suggest, and evaluate alternative approaches for the setting of ABCs for ORCS. Working Group members represent seven of the eight SSCs, five of the six NMFS Science Centers, NMFS Headquarters, as well as a regional fishery management council, academic institutions, a state agency, and an NGO. The goal of the Working Group was to develop an approach for addressing ABCs in ORCS that could potentially be applied in all jurisdictions under a flexible framework.

This report reviews existing methods for setting catch limits for ORCS. Each approach is briefly summarized followed by a description of the required data, the major assumptions and consequent cautionary advice in utilizing the particular approach, its potential for use in a risk-based decision-making framework, the status of the approach along with examples of its implementation, and the pros and cons of using the approach as viewed by the Working Group.

The Working Group also presents its own approach, designed to build on existing approaches, while strengthening the biological and population dynamics underpinnings. The method provides additional flexibility and allows policymakers to set risk levels, as required under the NS1 guidelines.

Ultimately, the Working Group recommends that the following tiered approach be used when setting ABCs for ORCS:

- Apply depletion-based stock reduction analysis (DB-SRA) to a stock, if possible. The main limitation here is the requirement for a complete time series of historical catches, which is often not available.

- If it is not possible to apply DB-SRA, apply depletion-corrected average catch (DCAC) to a stock. DCAC's main limitation is that it is only appropriate for stocks with moderate to low natural mortality rates ($\leq 0.20\ yr^{-1}$).

- If DB-SRA and DCAC are not appropriate, apply the ORCS Working Group's Approach. The main limitation with this approach is that a number of critical decisions are required before it can be made operational. Some would also view this as an advantage, as it provides flexibility in its establishment.

- Finally, in some cases none of the above methods are practical for setting ABCs for an individual stock, as specific ORCS may not be capable of being effectively managed or monitored. In these cases, it may be best to use a stock complex approach. There are many limitations of applying a stock complex approach as described in this report, and the ORCS Working Group cautions against overusing or misusing this approach, as it may result in the converse of precautionary management, exactly what MSA was designed to avoid.

It is important to note that the methods for setting ABCs for ORCS are in various stages of development and will be better understood and improved upon over time. For that reason, a list of research recommendations is included in the report that highlights the most important activities that must be supported to make substantive progress in the future.

The Working Group emphasizes that none of the methods discussed in this report are a substitute for additional data and monitoring. Therefore, all of the methods impose a certain risk and imprecision that fisheries managers must acknowledge when using the results of these methods.

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Symbols and Abbreviations used in this Document

Δ	The expected proportional change in stock biomass from the first to the last year of the catch series in DCAC; the proportional reduction in biomass relative to K in DB-SRA
ABC	Acceptable biological catch
ACL	Annual catch limit
ACT	Annual catch target
AM	Accountability measure
B_*	Equilibrium biomass at some level of fishing mortality F_* in the Pella-Tomlinson production model
B_0	Virgin biomass
$B_{20\%}$	Biomass level that corresponds to 20% of the unfished biomass
B_{MSY}	Biomass that would produce MSY
B_{peak}	The ratio of B_{MSY} / K as used in DB-SRA
c	Natural variability factor used in natural mortality-based approach
CCAMLR	Convention on the Conservation of Antarctic Marine Living Resources
CFMC	Caribbean Fishery Management Council
CPUE	Catch per unit effort
DB-SRA	Depletion-based stock reduction analysis
DCAC	Depletion-corrected average catch
EC	Ecosystem Component
F_{MSY}	The fishing mortality that produces MSY
GMFMC	Gulf of Mexico Fishery Management Council
h	Steepness of the Beverton-Holt stock recruitment relationship
ICCAT	International Commission for the Conservation of Atlantic Tunas
K	Carrying capacity
M	Natural mortality
m	Maximum productivity (MSY) in the Pella-Tomlinson production model
MAY	Maximum average yield
MCY	Maximum constant yield
MFMT	Maximum fishing mortality threshold
MSST	Minimum stock size threshold
MSA	Magnuson-Stevens Act
MSE	Management strategy evaluation
MSY	Maximum sustainable yield
NEFMC	New England Fishery Management Council
NGO	Non-governmental organization
NMFS	National Marine Fisheries Service
NOAA	National Oceanographic and Atmospheric Administration

NPFMC	North Pacific Fishery Management Council
NS1	National Standard 1
OFL	Overfishing limit
ORCS	Only reliable catch stocks
OY	Optimum yield
P*	Risk level; Probability of overfishing
PFMC	Pacific Fishery Management Council
PGY	Pretty good yield
PSA	Productivity Susceptibility Analysis
r	Intrinsic rate of increase
R_0	Virgin recruitment
SAFMC	South Atlantic Fishery Management Council
SD	Standard deviation
SSC	Scientific and Statistical Committee
TAC	Total allowable catch
Y_{AV}	Average catch used in natural mortality-based approach
Y_*	Annual equilibrium yield for the Pella-Tomlinson production model

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I. BACKGROUND

A. Requirement for ABC specifications and ACLs

The Magnuson-Stevens Fishery Conservation and Management Act (MSA) of 1996 required Regional Fishery Management Councils (Councils) to end overfishing and rebuild overfished stocks. It strengthened US fisheries law by putting in place firm timelines for rebuilding and specified requirements for rebuilding plans. In 2006, however, the majority of overfished stocks were still not rebuilt and overfishing continued to be a widespread problem because fishery management plans failed to sufficiently reduce exploitation rates (Rosenberg et al. 2006). As a result, Congress amended the MSA during the 2006 reauthorization with requirements for annual catch limits (ACLs) and accountability measures (AMs) for each managed fishery by fishing year 2010 for all stocks experiencing overfishing and by fishing year 2011 for all other stocks in the fishery (DOC, 2007). The reauthorized MSA further strengthened the role of science in the fishery management process by requiring that ACLs set by Councils may not exceed the fishing level recommendations of the Councils' Scientific and Statistical Committees (SSCs).

In the 2009 National Standard 1 (NS1) guidelines, the National Marine Fisheries Service (NMFS) provided specific guidance on how to comply with the new requirements of the MSA, including limit and target reference points for fisheries (NMFS, 2009) (Figure 1). The OFL is the annual estimate of the catch that would be obtained if a stock were fished at a rate producing the long-term maximum sustainable yield (MSY); overfishing occurs when catch exceeds the OFL. The ABC is the upper limit at which Councils can set the ACL. The SSCs were designated with the responsibility to set the acceptable biological catch (ABC), which is the catch level that accounts for scientific uncertainty in the estimate of the overfishing limit (OFL) and other sources of scientific uncertainty. The NS1 guidelines further require each Council, in conjunction with its SSC, to establish an ABC control rule that specifies how ABC is calculated based on the scientific uncertainty in the OFL estimate and the Council's risk policy. These requirements apply to data-rich stocks that can be assessed through quantitative stock assessment models, as well as data-poor stocks that cannot be assessed with traditional stock assessment methods. This report focuses on the ABC requirements for stocks that have only catch history data available for estimating harvest limits. We refer to these stocks here as "Only Reliable Catch Stocks" (ORCS).

B. History of dealing with ORCS

The 1998 NS1 technical guidelines (Restrepo et al. 1998) recommended that Councils "*adopt a precautionary approach to specification of [optimum yield] OY,*" stemming from the 1996 MSA requirement to end overfishing and rebuild depleted fishery resources. The precautionary approach was implemented to reduce the risk of overfishing in circumstances where scientific evidence of overfishing was not available (Restrepo et al. 1998). As it was recognized that all

regions possessed data of varying states of quality for stock assessment and management purposes, subsequent guidance provided an array of precautionary control rules that could be used to set exploitation targets below the risk-neutral limits based on MSY-related benchmarks, such as the maximum fishing mortality threshold (MFMT) and minimum stock size threshold (MSST) or reasonable proxies for one or both of these status determination criteria (Restrepo et al., 1998; Restrepo and Powers, 1999).

In the absence of biomass and fishing mortality reference points, the 1998 Technical Guidance (Restrepo et al., 1998) for implementing the NS1 guidelines suggested using the historical average catch from a period during which there was no evidence of declining abundance as a proxy for MSY. This recent catch would be multiplied by a scalar (ranging from 25% to 75%) based on "informed judgment" regarding the qualitative estimate of stock size relative to B_{MSY} (stock biomass at maximum sustainable yield) and MSST to obtain the limit catch, but the performance of this recommendation was never investigated (Restrepo and Powers, 1999). From discussions among members of this Working Group, however, it appears that many Councils have used a constant scalar (e.g., 50%, 75%) as their precautionary approach regardless of the stock's size relative to B_{MSY} and MSST.

C. Unique problem for ORCS

The 2009 NS1 guidelines state that the ABC should be based, when possible, on the probability of overfishing, which cannot exceed and should be less than 50 percent. The guidelines further require that "*the ABC control rule must articulate how ABC will be set compared to the OFL based on the scientific knowledge about the stock or stock complex and the scientific uncertainty in the estimate of OFL and any other scientific uncertainty. The ABC control rule should consider uncertainty in factors such as stock assessment results, time lags in updating assessments, the degree of retrospective revision of assessment results, and projections*". Thus, the NS1 guidance for setting ABCs is clearly directed towards stocks that can be assessed through traditional stock assessment methods. Many stocks under US federal management, however, lack current stock assessments and are not likely to be assessed in the near future, due to substantial data limitations. For example, the 2009 Report to Congress on the Status of U.S. Fisheries reported that "*272 stocks or stock complexes have overfishing thresholds not defined or applicable, or are unknown with respect to their overfishing status*".[1] In these data-limited situations, the guidelines are vague with respect to factors that could be considered for setting ABCs and simply suggest the use of reasonable proxies.

Many of the ABC control rules that are currently being developed in the regions follow a tiered approach in which the size of the buffer between the OFL and ABC increases as the level of

[1] NMFS 2009 Report to Congress on U.S. Fisheries, May 2010. Available online at http://www.nmfs.noaa.gov/sfa/statusoffisheries/sos_full28_press.pdf

scientific uncertainty increases (Witherell, 2010). Since uncertainty is expected to increase with decreasing availability of reliable data, it follows that data-poor stocks should generally have larger buffers than data-rich stocks for the same desired risk of overfishing. Without a system in place that monitors key fishery indicators and responds to changes in these indicators, scientists and managers have no means of evaluating whether any newly established catch limits for ORCS are too conservative or too liberal.

D. Catch vs. landings

These two terms are not synonymous, since catch is considered the landed catch plus the total amount of dead discard (i.e., bycatch). Too often an evaluation of historical catch becomes an examination of historical landings. Bycatch levels in other fisheries, as well as discard rates and discard mortality levels, should be discussed and factored into the evaluation of historical catch. Anecdotal information, fishermen's knowledge, and local expertise should be considered in such cases.

E. Formation of the ORCS working group

At the second National SSC meeting November 10-13, 2009 in St. Thomas, USVI, an ad-hoc Working Group was established to address the question of how to develop ABCs for data-poor fisheries under the jurisdiction of Regional Councils, where traditional stock assessment techniques cannot be applied due to data deficiencies.

The Working Group was established to identify, suggest, and evaluate alternative approaches for the setting of ABCs for ORCS. Working Group members represent seven of the eight SSCs, five of the six NMFS Science Centers, NMFS Headquarters, as well as a regional fishery management council, academic institutions, a state agency, and an NGO. The Working Group has communicated general process-related comments, as well as stock assessment and management ideas through email and teleconference.

The overriding goal of the Working Group was to develop an approach for addressing ABCs in ORCS that could potentially be applied in all jurisdictions under a flexible framework process. To this end, the Working Group reviewed existing methods that have been used both nationally and internationally to address data-deficient fisheries, and developed a hierarchy of recommended models or techniques for use in a particular fishery, given only that the fishery possesses a time-series of reliable catch data.

F. Scientific and management uncertainty in ORCS

Unlike the 1998 NS1 guidelines, the 2009 guidelines make the distinction between two types of uncertainty that are to be considered in the catch-setting process: management and scientific. Management uncertainty arises from uncertainty in quantifying the true catch amount and uncertainty in the ability of managers to limit actual catches to the ACL. Councils have the

flexibility to account for management uncertainty by setting an annual catch target (ACT) at or below ACL. Scientific uncertainty has been discussed earlier, and deals with the estimate of the OFL and ABC.

While the two types of uncertainty are distinct, they are not independent because the realized catch affects abundance and consequently, future OFLs, which then feed back into ACLs (Shertzer et al. 2008). It is not always possible to distinguish between scientific and management uncertainty, especially in the case of ORCS, where total catches may be highly uncertain because of missing information regarding bycatch and discard mortality, affecting both scientific and management uncertainty. The NS1 guidelines allow for both scientific and management uncertainty to be incorporated into a single control rule, but ABCs by definition address only scientific uncertainty, which is the scope of this report.

G. Incorporating risk

It is the responsibility of stock assessment scientists and the SSCs to determine the level of scientific uncertainty that exists in an assessment or estimated level of sustainable yield, but it is the role of the Councils to determine the acceptable risk of overfishing given the scientific uncertainty. When the probability distribution around the OFL estimate can be computed and characterized, the median estimate of the OFL implies a risk level of 50 percent, which is the level of risk the NS1 guidelines state is not to be exceeded in setting ABC. When the OFL and its statistical distribution can be estimated, probability-based methods can be used to compute the ABC that corresponds to the Council-desired risk of overfishing (e.g., Prager and Shertzer 2010). In the case of most ORCS, quantitative estimates of reference points from assessment models are unavailable, and formal risk statements cannot be made because the uncertainty is often not quantifiable. In those cases, an adaptive approach to developing ABCs that involves monitoring key fishery indicators may need to be adopted.

H. Report outline

The report is divided into seven primary sections:

- Section I, which you are currently reading, provides background on ORCS, the need to set ABCs, and the difficulties specific to ORCS.

- Section II reviews existing national and international methods that are currently in use or in the process of being further developed. Each approach is briefly summarized followed by a description of the required data, the major assumptions and consequent cautionary advice in utilizing the particular approach, its potential for use in a risk-based decision-making framework, the status of the approach along with examples of its implementation, and finally, the pros and cons of using the approach as viewed by the Working Group.

- Section III introduces a new approach for setting ABCs for ORCS developed by the authors of this paper.

- Section IV examines the suitability of the previously described methods for setting ABCs for stock complexes and presents any necessary modifications, additional assumptions, or important caveats that need be considered prior to applying each approach to stock complexes.

- Section V provides a discussion of the topics raised in this paper.

- Section VI provides research recommendations to further our ability to understand, set ABCs for, and manage ORCS.

- The final section, Section VII, puts forth a set of recommendations to Councils and SSCs for moving forward in addressing the 2006 MSA mandate, under the 2009 NS1 guidelines, for ORCS.

II. REVIEW OF METHODS

A. Scalar approaches

1. Summary of approach

Scalar approaches involve specification of future catch by using simple scalar multipliers applied to current or historical catch patterns. The primary reference for this approach is Restrepo et al. (1998) who formalized the concept in their Technical Guidance document for the 1998 National Standard 1. Scalar approaches were presented in the sections of the document specifying catch targets and catch limits in data-poor situations (this is henceforth termed the Restrepo approach). Although Restrepo et al. (1998) is the primary citation for this particular set of scalar tiers, it is quite likely that the concept was widely used historically in fishery management. The Restrepo approach proposed scalar multipliers for catch targets ranging from 0.25 to 0.75, depending on the estimated stock status at the time. For example, if the stock was overfished and hence below the MSST, then the catch multiplier for the Restrepo approach was 0.25 with the intent to reduce fishing effort and allow the stock to rebuild. If the stock was above B_{MSY}, the multiplier was 0.75, which reflected the precautionary buffer between the catch target and catch limit, with the catch limit being status-quo catch levels in a presumed healthy fishery. For intermediate stock conditions the multiplier was 0.5.

2. Data needs

The Restrepo approach uses an average catch. In the original document this was defined as the average catch during a time period, not necessarily the most recent, for which there is evidence

of stable abundance. Ideally, there should be no quantitative or qualitative evidence of declining or increasing abundance trends in the selected time period. We note that approaches for deriving catch recommendations for stocks with decreasing trends are developed in sections II.E and F of this report. In an optimal situation there is an adequate catch data stream to objectively identify such a time period, and may vary temporally in location and span for particular stocks and fisheries. Since it was realized that stock status information is not available in many data-poor cases, it was suggested to explore several definitions of recent catch such as the mean or median catch during the last 5, 10, or 15 years. In minimal data situations, the Restrepo approach could be applied to a single year of fishery catch data, but this is obviously a tenuous application unless the single year of data was highly significant for some reason. A logical extension of the variable scalar multiplier would be to similarly reduce the value for shorter catch data streams owing to likely greater uncertainty.

3. Informed judgment

Some type of expert or otherwise informed judgment is required for the Restrepo approach if stock status information is lacking, which would likely be the case for any potential application of the approach. This judgment is critical because an overfished determination can result in catch limits that are adjusted downward to a third of what could conceivably be taken if stock status was not judged to be in an overfished condition. Such a declaration of stock status is generally difficult even with strong quantitative support. Scientific judgments should be supported with as much objective analysis as possible. Careful examination of all available biological and fishery indicators is warranted. Even if a formal stock assessment is lacking, a diverse assemblage of data (including qualitative and anecdotal information) can be evaluated in a meta-framework to infer stock status (e.g., Porch et al., 2006). The Restrepo et al. document mentions a variety of similar alternative approaches such as informed judgments, Delphi approaches, qualitative approaches, expert opinions, and consensus-building methods. In addition, Bayesian statistical methodology is an appropriate tool for heterogeneous data and variable prior knowledge.

4. Caveats

The primary assumptions of the Restrepo approach are that the fishery is at or near a sustainable equilibrium, the stock is stable, and some qualitative determination of stock status is possible. However, without adequate information, it can be difficult to judge stock status, and, likewise, without a protracted period of near-constant and/or sustainable fishing effort and catches, it can be difficult to verify stability. If fishing effort is highly variable or if a fishery is in development or experiencing overfishing, then the catch data stream will be problematic for the Restrepo approach.

5. Risk assignment

Restrepo et al. (1998) conducted simulation modeling to explore what an appropriate default target catch control rule for data-adequate stocks might look like and found that fishing at 75% F_{MSY} resulted in equilibrium yields of 94% MSY or higher and equilibrium biomass levels between 125% and 131% of B_{MSY} while reducing the probability that the stock would decline to ½ B_{MSY}. Based on these results, the recommended default target control rule became fishing at 75% F_{MSY}. The data-poor proxy of this default rule for stocks judged to be above B_{MSY} thus became 75% of recent catch. Additional risk can be built into the approach by simply reducing the scalar multipliers. This is analogous to the catch limit and catch target differential multipliers in the 1998 technical guidance document. Biological and/or fishery information can be incorporated into the approach by using natural mortality rate or risk assessments like the PSA (productivity susceptibility analysis; Patrick et al., 2009; 2010) inputs to the scalar specification. These potential improvements will be discussed in forthcoming sections of text.

6. Status of approach

The Restrepo approach and variants thereof are used in the management of many fisheries across the nation. Scalar multipliers range from 0.25 to 0.75 consistent with the original guidance. There is considerable variability in the time window of recent catch ranging from 1 year to 18 years. The location of this recent catch time window also varies considerably from recent years to over 30 years into the past. As pointed out earlier, these parameters for the recent catch specification will have to be tailored to individual stocks and fisheries on a case-by-case basis.

Some examples of current use for ORCS:

- *The Pacific Fishery Management Council (PFMC) coastal pelagics ABC is specified using a scalar multiplier of 0.25 applied to average catch and scaled by proportion of stock available in U.S. waters.*
- *The International Commission for the Conservation of Atlantic Tunas (ICCAT) specifies total allowable catch (TAC) to be no more than the product of scalars of 0.33 for white marlin and 0.50 for blue marlin applied to 1996 or 1999 landings, whichever is greater. These reference years were chosen because they were thought to be particularly reliable. The scalars reflect the understanding of the recent level of overfishing, particularly for white marlin.*
- *OY for some PFMC groundfish stocks is specified using a scalar multiplier of 0.50 applied to average catch.*
- *The North Pacific Fishery Management Council (NPFMC) specifies ABC using a scalar multiplier of 0.75 applied to average catch from 1978-1995.*
- *The Caribbean Fishery Management Council (CFMC) specifies ABC and ACL using a scalar multiplier of 0.85 applied to average catch from 1999-2005 or 2000-2005 depending on the management area.*

- *The New England Fishery Management Council (NEFMC) used a scalar of 1.0 for ABC of Atlantic herring because a provisional analysis suggested that the stock was not overfished and overfishing was not occurring.*
- *The NEFMC also used a scalar of 1.0 for ABC of red crab because there was no evidence of depletion since the beginning of the fishery.*

7. Pluses/minuses of approach as viewed by Working Group

Some advantages of the Restrepo approach are that it a) is straightforward and therefore easily explained and understood by scientists, policymakers and stakeholders, b) can easily be applied even by those not specifically trained in stock assessment procedures, and c) is broadly applicable across species with different biological characteristics. Some of the disadvantages are that a) the appropriateness and performance of the recommended multipliers has not been evaluated, b) the assumptions of a stable stock which is at or near sustainable equilibrium can often not be verified, c) it is not suitable for application to an ORCS stock that is very lightly exploited since it does not allow for a catch limit larger than recent average catch, d) it does not explicitly account for species differences in productivity, and e) continued application could ratchet catch downwards as the recent average catch was forced to decline. The method was intended to be used as a short-term fix, until either additional data could be collected or an improved method could be developed.

B. Scalar multiplied by the ABC of the target species, when ORCS are bycatch species

1. Summary of approach

In one international arena, ORCS species believed to be exploited well below MSY levels and caught incidentally in directed fisheries are regulated in concert with the targeted species, based on a proportion (or harvest cap) associated with the targeted stock's quota. In those cases, the targeted stock's ABC is multiplied by a scalar, for example 5%, to obtain the ABC of the bycatch species. Management for these bycatch species focuses on collecting additional information to elevate these fisheries to a formal assessment status as soon as possible and thereby allow what may have started as an exploratory fishery to safely expand to a targeted fishery.

2. Data needs

The only data required to carry out this management approach is a catch limit for the targeted species of the exploited assemblage.

3. Informed judgment

Judgment is needed for deciding what the proportion of the targeted species' catch limit should be that serves as the scalar for determining the ABC of the bycatch species which requires expert

opinion regarding the abundance of the bycatch stock relative to the target stock. The choice of scalars should be guided by the precautionary principle to avoid overfishing but should also allow for data collection and potential fishery expansion. Where this method has been applied, quotas have ranged from 5-16% of the targeted species' catch limit.

4. Caveats

The appropriateness of the chosen scalar cannot be known initially, and therefore, ongoing monitoring programs are imperative to the application of this general approach. Precise estimates of species composition from the landings, as well as observer data and fishery-independent survey data are necessary to ensure current proportional allocations for bycatch species are representative of recent resource/fishery dynamics and are ultimately sustainable.

5. Risk assignment

Although formal risk cannot be explicitly assigned in this straightforward method, the risk of overfishing bycatch species is considered to be relatively low by the management body implementing the approach. Higher landings caps imply higher risk of overfishing.

6. Status of approach

This approach is being implemented by the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR). In most cases, all bycatch species associated with the directed fisheries have recommended harvest levels that are defined in accordance with the CCAMLR.

7. Pluses/minuses of approach as viewed by Working Group

The approach is very simple to apply, as it involves multiplying a scalar by the quota of a targeted species. Given species are selected because they are believed to be underutilized, it is assumed that there is a relatively low risk of overfishing using this method, but ultimately, there is little information to inform the initial choice of any particular scalar. Since this method sets an ABC for a group of species rather than an individual stock, it is a special case of a stock complex approach, which is discussed in Section IV of this report. If implemented correctly, the method allows for fishery expansion to occur slowly and in a coordinated fashion.

C. Natural mortality-based approach

1. Summary of approach

The natural mortality-based approach (Anon 2009) is another variant of a scalar approach. It is based on the formula:

$$MCY = c\ Y_{AV}$$

Where MCY is the maximum constant yield, c is the natural variability factor (defined below) and Y_{AV} is the average catch over an appropriate period.

If the catch data are from a period when the stock was fully exploited (i.e., fishing mortality near the level that would produce MAY [= Maximum Average Yield]), then the method should provide a good estimate of MCY. In this case, Y_{AV} = MAY. If the population was under-exploited, the method gives a conservative estimate of MCY.

The natural variability factor is defined as in Table 2. It is assumed that because a stock with a higher mortality rate will have fewer age-classes, it will also suffer greater fluctuations in biomass. The deviations from values of c in the table are for stocks where there is evidence that recruitment variability is unusually high or low.

2. Data needs

Familiarity with stock demographics and the history of the fishery is necessary for the determination of an appropriate period on which to base estimates of Y_{AV}. The period chosen to perform the averaging will depend on the behavior of the fishing mortality or fishing effort time series, the prevailing management regime, the behavior of the catch time series, and the lifespan of the species.

The period should be selected so that it contains no systematic changes in fishing mortality (or fishing effort, if this can be assumed to be proportional to fishing mortality). The period chosen should also contain no systematic changes in catch. If the period shows a systematic upward (or downward) trend in catches then the MCY will be under-estimated (or over-estimated). It is desirable that the period be equal to at least half the exploited life span of the fish.

An estimate of natural morality is required to obtain the value of c, the natural variability factor. Knowledge of recruitment variability levels is needed to modify the natural variability factor, if necessary.

3. Informed judgment

In many cases informed judgment will be needed to select the period chosen to perform the averaging, as all of the information required to adequately select the period may not be available.

4. Caveats

The primary assumptions of the natural mortality-based approach are that the fishery is at or near a sustainable equilibrium and the stock is stable. However, it can be difficult to estimate stability without a protracted period of near-constant and/or sustainable fishing effort and catches. If fishing effort is highly variable or if a fishery is in development or experiencing overfishing, then the catch data stream will be problematic for this approach.

5. Risk assignment

Risk is incorporated through the use of the natural variability factor, which takes into account the natural mortality of the stock. It is assumed that because a stock with a higher mortality rate will have fewer age-classes it will also suffer greater fluctuations in biomass. In addition this can be modified where there is evidence that recruitment variability is unusually high or low.

6. Status of approach

The approach is currently being implemented for ORCS in New Zealand.

7. Pluses/minuses of approach as viewed by Working Group

The natural mortality-based approach has limited potential for application in the U.S. It is not designed for stocks that are currently in an overfished state. It is designed for stocks that have either been fully exploited or under exploited. It does not take into account cases where stocks have been over-exploited (overfished). Further, it requires a time period of stable catch, which may not be available for all stocks. Shorter life spans are viewed as inherently more risk prone and difficult to manage effectively, given they exhibit greater population fluctuations, requiring a smaller scalar to account for the increased risk. Other factors affecting risk are not incorporated into the method. The method has not been evaluated

D. Depletion-Corrected Average Catch (DCAC)

1. Summary of Approach

Restrepo et al. (1998) provide guidance on estimating sustainable catch in situations where only a catch time series is available, suggesting that a sequence of relatively constant catches is evidence that the average annual harvest is sustainable. Although this approach can be useful for providing catch advice for data-poor stocks, the inference of sustainability is only true if both fishing mortality and stock abundance are stable during the period in question. A constant catch could be produced during a period of increasing fishing mortality and decreasing stock abundance, in which case the catch may not be sustainable. Nonetheless, Restrepo et al. (1998) argued that an average catch taken from a time period of stable harvest is a useful proxy estimate of sustainable yield.

MacCall (2009) developed an approach that allows for changing population abundance during the period when catches are obtained. He described the method as “depletion-corrected average catch” (DCAC) because it accounts for the windfall augmentation of catch that occurs due to a one-time reduction in standing stock, also known as “fishing up.” Conveniently, the method works just as well if a stock is increasing in abundance during the time interval. Fundamentally, DCAC is based on the premise that knowledge of natural mortality (M) is informative of F_{MSY}, a reasonable prior for relative B_{MSY} (B_{MSY} / B_0; B_0 = virgin biomass) is available, and some view of relative stock depletion can be obtained.

2. Data needs

The basic DCAC calculation requires: a) an average catch calculated over some period of years, b) an estimate of natural mortality, which may be obtained from the relationship between longevity and M developed by Hoenig (1983) or other indirect methods, c) an estimate of the ratio of F_{MSY} to M, which MacCall (2009) argues is typically in the range of 0.6–1.0, and d) an idea of how much stock abundance may have changed during the time period when catch statistics are summarized. This last input value is termed Δ and represents the relative decline (or increase) in stock size, with a larger value representing a greater decrement to the stock. In addition, the method has recently been generalized to include a prior distribution for relative B_{MSY} (Stock Assessment Toolbox; http://nft.nefsc.noaa.gov/index.html).

3. Informed judgment

The DCAC method is a generalization of the average catch approach because an adjustment is made for changes in stock size (Δ). This is, however, a quantity that is difficult to obtain, and expert opinion must be used to decide on relative stock status. Likewise, informed judgment may be helpful in deciding on the ratio of F_{MSY} to M and B_{MSY} to B_0. Prior distributions for Δ, ratio of F_{MSY} to M and B_{MSY} to B_0 could be based on meta-analysis for related stocks, rather than expert opinion.

4. Caveats

DCAC assumes that the catch statistics used in the calculation are unbiased. Also, the method is only appropriate for stocks with moderate to low natural mortality rates ($\leq 0.20\ yr^{-1}$) because the depletion correction becomes negligible at higher values of M. Moreover, in its initial implementation the calculation assumed that relative B_{MSY} occurs at 0.40. While this is a robust proxy supported by the simulations conducted by Clark (1991), the newest version of the calculation (i.e., the NOAA Fisheries Stock Assessment Toolbox) allows the user to specify a prior distribution for this quantity.

5. Risk assignment

Propagation of uncertainty is a strong point of the DCAC method, which is accomplished by Monte Carlo simulation based on draws from distributions of the key input quantities. In particular, the principal inputs (M, F_{MSY}/ M, B_{MSY}/ B_0, and Δ) are specified as distributions. Importantly, MacCall (2009) provides a variance statistic for M based on reanalysis of the data summarized in Hoenig (1983). The result of the algorithm is an output distribution of catch that would have been sustainable over the specified timeframe, conditional on the input distributions, which can be used as a basis for risk assessment (Figure 2).

6. Status of the approach

The NEFMC and its SSC evaluated an application of DCAC to deep-sea red crab and concluded that because it provides an estimate of a sustainable yield and not MSY, it was inappropriate to

use in calculating OFLs. Moreover, because survey information did not indicate that the population abundance of red crab had changed between 1974 and 2005, no depletion correction was required and an ABC was set based simply on average landings during that time period.

Because the DCAC calculation utilizes a sum of catches calculated over a period of years, the PFMC endorsed its use in developing OFLs for seven groundfish stocks that are characterized by erratic and/or incomplete catch histories. Those stocks included six rockfishes (*Sebastes* spp.) and one elasmobranch (i.e., blue rockfish, blackgill rockfish, gopher rockfish, honeycomb rockfish, Mexican rockfish, squarespot rockfish, and soupfin shark).

7. Pluses/minuses of approach as viewed by Working Group

There are a number of appealing features of the DCAC method, including: a) it is based principally on catch statistics and basic life history information, b) the catch time series need not be comprehensive, c) stock abundance is explicitly allowed to vary, d) the method's inputs are approximate and are specified as distributions as opposed to point estimates, and e) uncertainty is propagated to produce a distribution of sustainable yield. Some of the disadvantages of the approach are: a) the estimated yield is typically sustainable, but not maximal, b) expert opinion is required to characterize stock depletion, and c) the estimated yield may not be sustainable if the stock at the end of the time series is not representative of the production that occurred during the time series (i.e., it is severely depleted).

E. Depletion-Based Stock Reduction Analysis (DB-SRA)

1. Summary of Approach

Depletion-Based Stock Reduction Analysis (DB-SRA) is an extension of DCAC that incorporates full stock dynamics (Dick and MacCall, In press). At a basic level stock production is the product of *per capita* production (= productivity) scaled by the total size of the population. For example, under Beverton-Holt spawner-recruit dynamics these quantities are represented by steepness (h) and virgin recruitment (R_0), respectively. Likewise, under a Schaefer surplus production model they are equal to the intrinsic rate of increase (r) and the carrying capacity (K). The DB-SRA method relies on specifying a plausible range of "scaled" production parameters and depletion levels in the form of prior distributions. Then, given the availability of a comprehensive catch history to scale the problem, virgin biomass can be uniquely calculated, conditional on each draw from the input distributions.

2. Data needs

Because the DB-SRA method is fully dynamic, a complete history of removals is required, i.e., annual catches from the beginning of the fishery are needed. Moreover, the method, at least in its current form, has been implemented as a delay-difference production model (Quinn and Deriso, 1999) and age-at-maturity is used to lag recruitment relative to production. Beyond

those fixed inputs, the technique depends on the same four "prior" input distributions as DCAC, including: a) natural mortality (M), b) the ratio of F_{MSY} to M, c) the ratio of B_{MSY} to B_0, and d) stock depletion (Δ). The DB-SRA method is also formulated in a way that provides considerable independence between F_{MSY} and B_{MSY} by implementation of a generalized production function (Fletcher 1978; McAllister et al. 2000; Dick and MacCall, In press).

As with DCAC, F_{MSY} is scaled relative to the natural mortality rate, and the product of the scalar and M provides an estimate of F_{MSY}. By also drawing an estimate of relative B_{MSY} from its input distribution, production is then completely specified on a relative biomass basis. Next, the time series of catches and a random draw from the depletion distribution (Δ) are used to scale biomass and solve for the unique value of B_0 and current biomass that satisfy all conditions (Figure 3). Of course in some instances the time series of catches is impossible with the random draws from the input distributions and the population trajectory goes negative. Those realizations are considered biologically implausible and are dropped from the collection of feasible outcomes (see also Walters et al. 2006). The process is repeated numerous times and posterior distributions of B_0, $B_{current}$, MSY ($F_{MSY} \times B_{MSY}$) and OFL ($F_{MSY} \times B_{current}$) are summarized from the individual results.

3. Informed judgment

The DB-SRA method further generalizes DCAC and, like that method, requires expert opinion to provide a general idea of stock depletion at some point in the catch time series. However, the depletion distribution can be somewhat vague and/or uninformative without great loss in performance. Likewise, informed biological judgment is needed to provide the initial input distributions for the ratios of F_{MSY} to M and B_{MSY} to B_0. However uncertainty in those distributions can be captured explicitly in their variances and they are biological characteristics that can reasonably be informed by conventional scientific wisdom.

4. Caveats

Other than assumptions associated with generating the four key input distributions, the DB-SRA method is very general. In particular, the implementation of a generalized production function that uncouples F_{MSY} from B_{peak} allows a broad range of models to be explored. Also, the method is robust to stochastic variation in stock recruitment, as long as recruitment is not highly episodic or strongly autocorrelated. Nonetheless, the method requires the user to provide four distributional inputs, which can be difficult to specify. No doubt the most troubling of these is the depletion (Δ) distribution, which is perhaps the main output statistic obtained in a data-rich stock assessment; requiring it as an input would seem to undermine the utility of the DB-SRA approach. In practice, however, the same type of inference is required of all ORCS methods (see above), but with DB-SRA it is expressed quantitatively as a distribution. The obvious benefit of this is that the prior distribution of Δ can be vaguely specified, which is to say the variance about the mean of the distribution implies that not much is actually known about depletion. Also,

given that the approach incorporates depletion as an input, it is not an appropriate method for determining relative stock status; rather its strength is in yield estimation (MSY and OFL). Finally, the method requires a complete time series of total catch (landings + discards) to implement. To the extent that discards are underreported or not accounted for the method will produce biased results.

5. Risk assignment

Expression and depiction of uncertainty is a major goal of the DB-SRA method and is accomplished by Monte Carlo simulation of four input distributions through to the output distributions of management concern, i.e., current biomass, F_{MSY}, unfished biomass, and OFL. An example of how uncertainty and risk are characterized within the DB-SRA framework is given in Figure 4, which shows the probability of overfishing for brown rockfish (*Sebastes auriculatus*) from 1920 to the present, as well as the posterior distribution of OFL values for 2011 (Dick and MacCall, 2010). Given a distribution of OFL, it is possible to develop a control rule that maps ABC onto the probability of overfishing, a direct expression of scientific uncertainty.

6. Status of the approach

In 2010, the PFMC SSC endorsed the use of DB-SRA to estimate OFLs for 42 groundfish stocks, including 34 rockfishes (*Sebastes* sp.), four flatfishes (Pacific sanddab, rex sole, rock sole, and sand sole), one roundfish (kelp greenling), two elasmobranches (leopard shark and dogfish), and one complex (grenadiers). All data-poor rockfish stocks are managed within assemblages that are defined based on: a) distribution north or south of Cape Mendocino (lat. 40°10'N), and b) cross-shelf distribution (nearshore, shelf, or slope). These 42 stocks include approximately half of the species listed in the PFMC Groundfish Fishery Management Plan and the development of OFL estimates for these species represents a significant improvement in the scientific information used to manage these stocks.

The medians of the bias-corrected posterior distributions of OFL were used by the PFMC as stock-specific point estimates of OFL. These were aggregated into single OFLs for each assemblage, and a semi-quantitative estimate of scientific uncertainty was endorsed for the DCAC and DB-SRA methods by the Council's SSC (i.e., quadrupling the uncertainty of Tier 1, data-rich assessments). The Council also established a policy on buffering all groundfish ABCs below their OFLs by limiting the probability of overfishing (P*) to ≤ 0.45. Harvest specifications for the 2011-2012 biennial fishing cycle are being developed under this general paradigm.

The DB-SRA method has been coded in R (R Development Core Team, 2009) and is documented in two manuscripts. The first of these describes application of the method to 31 different Tier 1 stocks and compares estimates of OFL, MSY, and B_0 from DB-SRA to those

obtained from a full data-rich stock assessment, which have typically been conducted using the maximum likelihood based, integrated Stock Synthesis model (Dick and MacCall, In press). The second publication describes and documents the application of the DB-SRA method to the 42 data-poor, Tier 3 groundfish stocks listed above, including development of bias-corrections based on PSA and performance relative to data-rich stock assessments (Dick and MacCall, 2010).

7. Pluses/minuses of approach as viewed by Working Group

Like DCAC, DB-SRA is based principally on catch statistics and basic life history information, uses inputs that are approximate and are specified as distributions as opposed to point estimates, and allows for the propagation of uncertainty to produce a distribution of sustainable yield. The method was evaluated by comparison of OFL estimates from DB-SRA to those from 31 data-rich stock assessments. Results of that comparison showed that DB-SRA sometimes underestimates and sometimes overestimates OFL for individual stocks. As might be expected, the bias in OFL depended on PSA scores associated with each of the stocks. In particular, DB-SRA applied to flatfish generally underestimated OFL by a factor of 0.55. For high vulnerability non-flatfish stocks DB-SRA was largely unbiased, whereas for low vulnerability non-flatfish stocks the method underestimated OFL by a factor of 0.83. These biases were quantified and applied as an adjustment in estimating OFL for the 42 data-poor stocks by the PFMC. A primary disadvantage is that this method is rather time and resource-extensive and requires application by trained stock assessment scientists. In addition, for many ORCS species, it may not be possible to fully reconstruct catch history.

F. The Methot Table Conceptual Framework

1. Summary of approach

During the second National SSC meeting, NMFS's Rick Methot gave a presentation on the 2009 NS1 guidance regarding ACLs and the treatment of scientific uncertainty. In that presentation, a table was provided that showed an example of how ABCs might be set in catch-only situations. The original purpose of this conceptual framework was to generate discussion and inspire thought. The structured approach that it offered has since been used in discussions in different parts of the country to base ABC recommendations on, and the working group therefore deemed it appropriate to review the method here. The table, which we refer to here as "the Methot table conceptual framework", is based on the same basic concept as the Restrepo approach and requires an expert evaluation of fishery impact. The Methot table generates four fishery impact categories of historic catch: trivial, small, moderate, and moderately high and proposes a possible action for ABC determination for each (Table 1). The first impact category highlights the fact that trivial catches of non-targeted species are unlikely to affect the species population status and under these circumstances the Council should consider listing these species as "Ecosystem Components" (EC species) within their fishery management plan. EC species are not required to

specify OFL, ABC, or ACL thresholds; however, their catch levels should be monitored to ensure they are not targeted by the fishery in the future. If historic catches are judged to be small (the second impact category), it is assumed that the stock is not overfished and that the target catch could be set at the historic level while setting ABC and ACL above that. If historic catches are moderate, the fishery should be capped because any increase in catches might mean the stock could become overfished. If historic catches are moderately high, the stock might be overfished and recent catches should be considered as the limit. In that case, ABC would be set below recent catch levels to allow the stock to rebuild. The approach is fairly qualitative in that it does not provide specific methods for calculating the degree to which catch should be set above or below historic levels. Methot does suggest that a stock's vulnerability should be a consideration.

2. Data needs

The data needs for this method are similar to the Restrepo approach. Catch history is required along with any information that may help to determine stock status from the catch data. In addition, vulnerability information is recommended. Vulnerability can be determined through risk assessments such as the PSA analysis that evaluates a stock's productivity and susceptibility to the fishery (Patrick et al. 2009, 2010). The final vulnerability score could be a factor in the setting of a scalar used to multiply recent catch; the scalar would be lowest for the most vulnerable species and highest for the least vulnerable species, scaled to fit within some predetermined range. This relates the level of allowable catch directly to the biology of the species.

3. Informed judgments

At the onset, expert judgment is needed in order to assign species to one of the four impact categories, analogous to the judgment call needed for the Restrepo method for determining stock status with respect to MSST and B_{MSY}. In addition, informed judgment is needed to determine how much the ABC should be set above or below historic catch levels, and a judgment call is also needed to determine what the appropriate period of recent or historic catch is relative to which ABCs should be set.

4. Caveats

Although the Methot method does not make any explicit assumptions about stock stability or fishery equilibrium, the period of historic or recent catch used to determine future ABCs could have potentially large impacts on the final ABC that is calculated. Moreover, this method in its current state of development provides only qualitative statements about relative catch. Establishing absolute values or formulas for how much to increase or decrease OFL from historic catch in the case of low or moderately high impact, respectively, and how vulnerability is used as a relative scalar would still need to be fleshed out.

5. Risk assignment

Risk for this method could be assigned by setting boundaries on how much ABC can be increased or decreased from historic catch. For example, in the case of low historic catch, it would be less risk-prone to specify that ABC can be maximally 50% higher than historic catch instead of 100% higher. Similarly, in the case of moderately high historic catch, higher reductions in ABC translate into a higher probability that the stock will rebuild quickly than low reductions. Risk could also be assigned by deciding how much weight should be given to vulnerability. The PSA risk categories of low, medium and high could be converted into discrete scalars, and how much these scalars differ is a reflection of how much more risk one is willing to take for less vulnerable species.

6. Status of approach

A variation of this approach is currently being developed by the SSC of the Gulf of Mexico Fishery Management Council (GMFMC). The current Gulf ABC control rule consists of three tiers, the lowest of which contains the ORCS. The Gulf SSC is considering only two of the four dimensions from the Methot Table Conceptual Framework: small and moderately high impact (tier 3a and 3b, respectively). In the case of small impact, recent average catch over a stable period would be designated as the target catch, ABC would be set at either 0.5, 1, or 1.5 standard deviations (SDs) above the target, and OFL will be set at 2 SDs above that target. The rationale for setting OFL at 2 SDs above the mean is that this will result in only a 2.5% probability of catches in any given year exceeding and OFL so defined. The choice of SD level for ABC reflects a choice of risk because even though the SSC would recommend that target catch be set at the mean of recent average catch, the ultimate setting of ACT and ACL rests with the Council, and the Council could choose to set both equal to ABC, in which case an ABC of 0.5 SDs above the mean would constitute a less risk-prone upper limit than an ABC set at 1.5 SDs above the mean. In the case of moderately high impact, the GMFMC SSC approach would set OFL equal to the recent average catch and ABC would be set at 65%, 75%, 85%, or 100% of the OFL. Neither GMFMC tier 3a nor 3b currently use species vulnerability as part of their ABC considerations.

7. Pluses/minuses of approach as viewed by Working Group

The Methot Table Conceptual Framework represents a general approach for addressing ORCS and offers only qualitative advice for adjusting the magnitude of future catch limits with respect to recent catches. This can be advantageous in that it allows flexibility in regional application but it is also a drawback in its lack of specificity because it could result in potentially inappropriate application of the concept. As the GMFMC SSC has found out, the expression "the devil is in the details" seems to hold true, in taking an intuitive concept and making it operational. Like the Restrepo approach, the Methot Table Conceptual Framework is intuitive and easy to explain and therefore extremely useful for scientists, policymakers and stakeholders. Another advantage is that it takes into account species vulnerability, thereby acknowledging the

differences in resource response to exploitation. It can also be applied to stocks for which there is evidence that exploitation levels can be increased safely, and time and resources needed to apply this method are minimal because data needs are small and it does not require application by highly trained stock assessment scientists. The method's performance has not yet been tested in either simulations or application.

III. The ORCS Working Group Approach

A. Introduction

While this report has already summarized several control rules based on average catch scalars, the Working Group felt that the existing scalar approaches lacked a solid technical basis and that inadequate guidance had been provided for their application, leading to widespread misuse. Therefore, the Working Group developed a new control rule for the managers and scientists to address these issues. The proposed control rule for catch-only stocks builds on methods in Restrepo et al. (1998) and the Methot Table Conceptual Framework (summarized in Witherell 2010 and reviewed in section II F of this report). The Restrepo et al. (1998) approach assigns stocks to one of three status categories (less than MSST, between MSST and B_{MSY}, and above B_{MSY}) and uses a different average catch scalar for stocks in each category. The scalars are intended to be precautionary, so it would be difficult to use the Restrepo et al. (1998) approach in the new OFL/ABC framework where scientific uncertainty is explicitly taken into account. The new approach presented here also uses different scalars for three stock status categories, but defines the categories differently, and develops a scoring procedure for assigning stocks to these categories. Alternative buffers are proposed to account for scientific uncertainty in setting ABCs, since this is regarded as a policy decision.

The Working Group is fully aware that these methods rely heavily on assumptions and expert judgment, and are not intended to be a substitute for quantitative information on stock status and trend. Nevertheless there is a need for robust methods that provide useful scientific advice in less than ideal situations. Our goal is to improve on existing methods and provide a structured and transparent approach, but we recognize that further improvements are probably needed. With these caveats in mind, the basic approach is the following:

1. Assign stocks to one of three exploitation categories using an evidence-based scoring procedure;
2. Obtain an OFL by multiplying a statistical measure of historical catch (e.g., mean, median, maximum, minimum, percentile, etc.) by a scalar that depends on the exploitation category; and
3. Obtain an ABC as a proportion (< 1) of the OFL to reflect a policy decision on acceptable risk, which may depend on productivity of the stock (see Patrick et al., 2009; 2010).

B. Assigning stocks to exploitation categories

Stocks can be grouped into three broad exploitation categories for which different management objectives apply (Table 3): 1) lightly exploited; 2) moderately exploited; and 3) heavily exploited. For stocks that are considered lightly exploited, catches could generally be increased without harm to the stock. For stocks that are considered moderately exploited, management objectives will focus on maintaining status quo catch levels, and preventing non-sustainable increases. For stocks that are considered heavily exploited and possibly overfished, the management objective is to end overfishing and rebuild the stock to B_{MSY} levels as mandated by the MSA.

1. Background

The concept of 'pretty good' yield (PGY) provides a theoretical basis for broadly classifying stocks into exploitation categories. This concept, proposed by Alec MacCall and developed further by Hilborn (2010), is based on the observation that a large percentage of maximum sustainable yield (>80%) can be produced on a long-term basis over a broad range of stock sizes. This concept is particularly meaningful in data-limited situations, since it implies that successful management outcomes are possible even if stock status is not known precisely. To illustrate the PGY concept and to develop a technical basis for catch multipliers, a Pella-Tomlinson production model was used. The Pella-Tomlinson model duplicates the results of the more complex age-structured model used by Hilborn (2010), but allows equilibrium yield to be calculated directly for any percentage of unfished stock size.

Annual equilibrium yield (Y_*) for the Pella-Tomlinson production model is:

$$Y_* = \frac{\gamma\ m}{B_0} B_* - \frac{\gamma\ m}{B_0^n} B_*^n,$$

where:

$$\gamma = \frac{n^{n/(n-1)}}{n-1},$$

m is maximum productivity (MSY), B_0 is unfished biomass, and B_* is equilibrium biomass at some level of fishing mortality F_*, with $Y_* = F_*\ B_*$ (Quinn and Deriso 1999). Setting n = 1.2 results in a B_{MSY} that occurs at 40% of the unfished stock size, which is often considered a reasonable default value (Clark, 1991). For these assumptions, equilibrium stock abundance in a range from $B_{19\%}$ to $B_{65\%}$ of the unfished biomass provides at least 80% of the MSY yield on a

sustainable basis (Figure 5). Stocks above this range would be considered lightly exploited, while stocks below this range would be considered heavily exploited (i.e., overfished). These results are comparable to those obtained by Hilborn (2010) for an age-structured population. Special cases of the Pella-Tomlinson model are n=2, which becomes the Graham-Schaefer production model where B_{MSY} is 50% of unfished biomass, and $n \rightarrow 1$, which translates to the Fox production model where B_{MSY} is approximately 37% of unfished biomass.

2. Guidelines for assigning stock status

Status assignments based on historical catches will not have the benefit of a stock assessment, but will instead need to rely on 'expert' judgment. Experts in this context are those with experience conducting research, working on management issues, or participating in a fishery, and may include scientists, fishery managers, fishermen, and other involved parties. It will be important to be as comprehensive as possible when making status assignments and evaluate multiple lines of evidence. Given the absence of definitive information, the effort to generate these assignments may not be straightforward. It is important to note that that the overriding goal here is simply to assign stocks to very broad status categories with acceptable accuracy (e.g., say greater than a 70% success rate), recognizing that some inappropriate assignments will be inevitable.

An evidence-based scoring procedure (Table 4) has been developed to help assign stocks to the different status categories. This table incorporates some of the susceptibility elements in a PSA analysis (Patrick et al., 2010), as well as several new elements. The susceptibility scores in PSA evaluate the likelihood that a stock is captured in a fishery and the probable levels of fishing mortality, but PSA also includes productivity scores as a second dimension that takes into account the consequences of stock becoming overfished. In the framework we develop, productivity is considered separately when setting a buffer between OFL and ABC. While scoring procedures are a relatively recent development in fishery management, multi-attribute scoring algorithms have been used to evaluate the risk of species extinction (see Musick, 1999 and Dulvy et al., 2003). Multi-attribute scoring algorithms are also used in the medical field for making diagnoses and deciding treatment plans (Ebell, 2001). Elements of the evidence-based scoring procedure are described below.

Overall fishery exploitation based on assessed stocks. In general, the characteristics of the fishery in which the stock is caught are the most important factor to consider when assigning stocks to exploitation categories. If there are assessed stocks in the fishery, are they mostly overfished, moderately exploited, or are most lightly exploited? Unless there are reasons to think that the stock is more or less vulnerable than assessed stocks, it may be reasonable to assign it the same status as an associated stock that has been assessed. Certain habitats may have an overall level of exploitation that can be used to infer the status of unassessed stocks that live in that habitat.

Presence of natural or managed refugia. A stock that is fished throughout its range is more likely to be impacted by fishing than a stock that is fished only in a portion of its range. Species with extensive natural or managed refugia are unlikely to become severely depleted. This consideration would only apply to species that are not highly mobile as adults in relation to size of the refugia.

Schooling, aggregation, or other behavior responses affecting capture. This element encompasses both the behavioral response of individual fish to fishing gear and group behaviors that affect capture such as schooling or aggregating for spawning in known locations. Individual responses may include, for example, herding or gear avoidance behavior that would affect catchability.

Morphological characteristics affecting capture. This element pertains to the ability of the fishing gear to capture fish based on their morphological characteristics. For example, are there aspects of morphological characteristics affecting capture (i.e., large spines) that could make the fish more or less susceptible to capture? Because gear selectivity varies with size and age, this measure should be based on the age or size classes most representative of the entire stock.

Targeted species or Bycatch; and rarity. Targeting behavior by the fishery may help inform stock status assignments. Targeting may be inferred if a species has high commercial value or is considered highly desirable in a recreational fishery. Stocks that are caught primarily as bycatch in fisheries that target other stocks are likely to be lightly exploited relative to the targeted stock. However a non-targeted stock may still become overfished if it is much less productive than the targeted stock. Some stocks are simply too rare to be targeted, and would tend have low fishing impacts.

Natural mortality compared to targeted species in the fishery. This element provides a relative gauge of the stock's productivity compared to the dominant or targeted species in the fishery. Generally, for stocks subject to similar fishing mortality rates, those with low natural mortality have a higher likelihood of becoming overfished than those with higher natural mortality.

Value or desirability. Highly valued fish stocks are more susceptible to overfishing or becoming overfished by the recreational or commercial fishery due to targeting behavior with the goal of maximizing profits or non-market value. To identify the value of the fish, we suggest using the approach of Patrick et al. (2010) who used price per pound, or retention rates for recreational fisheries.

Trend in catches and effort. Finally, trends in historical catches may also be informative under some circumstances. If fishing effort is stable, a declining trend in catches may be an indicator

of stock depletion. Again, if effort is not increasing, stable or increasing catches are an indication that the stock is exhibiting resiliency and not likely being severely impacted by fishing, but caution is warranted when interpreting catch patterns in the absence of other indicators and sources of data. Qualitative measures of effort, such as the number of active vessels or employment in the fishery, are likely to be all that are available for data-poor stocks, but may be misleading if there are technological advancements in the fishery. Increasing catches could also be an indication of fishery expansion, i.e., a stock that is transitioning from lightly exploited to moderately or heavily exploited status.

The evidence-based scoring procedure provided (Table 4) includes default-scoring thresholds; however, we realize that revisions to the scoring procedure will likely be needed in different regional ecosystems and recommend that the scoring table be used flexibly. A starting point would be to assign status using the arithmetic mean of all attributes that can be scored, but weighting factors could be considered, or taking the geometric mean rather than the arithmetic mean. Careful consideration should be given to the logistics of scoring stocks. One possibility would be to assemble a core group of scientific experts that draws on information from formally appointed advisors that may include fishery managers, fishermen, and other knowledgeable individuals. Through trial and error techniques, it may also be useful to separate the scoring process into two steps by first ranking stocks along a continuum from lightly exploited to heavily exploited, and then identifying the break points between the lightly exploited, moderately exploited, and heavily exploited categories. Given management implications of identifying the break points, a higher-level science advisory body, such as the Regional Council's SSC, may be more appropriate for this task.

C. Determining an appropriate catch statistic for an OFL calculation

Calculating the OFL using the ORC methodology is based on two terms: a scalar (or multiplier) that is based on the stock status category (described above), and a catch statistic derived from a time series of historical catches. Ideally, historical catches should represent a period with a stable harvest rate, i.e., a harvest rate where fishing removals are balanced by stock production and the stock can be assumed to be in a steady state condition or at its long term equilibrium. Stability in catches should be considered relative to the longevity of the stock. Catches of a long-lived species can be stable over a long period even though the stock is declining during this period. Although historical catches can be very stable with low variability, more often they are highly variable, sometimes with large outliers, or could be characterized by alternate periods of stability and periods of high variability or strong trends. Catches of relatively uncommon stocks can vary for a number of reasons unrelated to increases or decreases in abundance. These stocks may be incidental catches in fisheries that target other stocks or are minor members in a multispecies complex. The greater or lesser occurrence of a stock in the catch could be a chance event, caused by changes in the spatial or ecological overlap between that stock and other stocks that are more actively targeted in the fishery. Furthermore, fishery sampling programs can

produce imprecise estimates of catches of stocks that are relatively uncommon. Evaluation of historical catch should include discussion of data quality and potential bias of catch estimates. If landings are highly variable, an attempt should be made to identify the reason for the variation and evaluate implications on the sustainability of historical catches. Other potential reasons for high fluctuations or outliers could be species misidentification, underreporting, effort variability, gear changes, or changes to the regulations for targeted species.

Although in many cases taking the arithmetic mean of historical catches is appropriate for an OFL calculation, the use of an alternative catch statistic may be needed in some situations to provide useful results. Several issues are described below, and suggestions presented for dealing with them are provided.

1. Outliers

In some cases, catch time series include extreme outliers that cannot be fully supported or rejected with available information. Several approaches to handling outliers are possible. First, a trimmed mean can be used (i.e., the inter-quartile mean) when the extreme values are considered unreliable. A similar approach would be to use the Winsorized mean, which is obtained by replacing all the values greater than or less than some quantile of catches by the largest (or smallest) of the remaining values. Usually 10 or 25 percent of the tails of the distribution are replaced. This approach would be appropriate when the extreme values are thought to carry some information about the catch quantity, but their actual values are considered unreliable.

2. Avoiding a ratchet effect

If catches are highly variable, the use of average catch as an OFL may be more constraining than is necessary, particularly when stocks are considered lightly or moderately exploited. When the management objective is to maintain current catch levels, setting the OFL equal to average catch could have the negative effect of depressing the mean level of the catch in the future, since presumably the management measures will need to prevent catches from exceeding the OFL, thereby truncating half of the distribution that was used to calculate the historical average. One possibility is to define the OFL to be some upper percentile of the historical catch, e.g., the 75% percentile of historical catch, with the rationale being that such a value would be exceeded on average one year in four if the fishery was prosecuted similar to historical patterns. Using the maximum catch is another alternative to average catch, but this should only be considered for non-target species with compelling evidence that they are lightly exploited. A similar approach has been proposed by the GMFMC SSC to, in some situations, base OFL on average catch plus two standard deviations (97.5 percentile), but it is unclear whether this approach provides sufficient constraint to prevent stocks from becoming depleted.

3. Recent trends

The theoretical development of average catch multipliers assumes that stocks are in equilibrium at some level of biomass, but this is necessarily an approximation to the real world, and in some cases it may be an inappropriate assumption from which to proceed. When there are downward trends in the landings, the safest approach (i.e., the most precautionary approach) would be to use an average based on the more recent lower values. However, if the downward trend in catches can be clearly linked to a reduction in effort, as when management restrictions are implemented for other species in a multi-species fishery, average catches from an earlier period may be more appropriate. If catches are trending upwards, using an average over all years may be the most reasonable approach.

D. Obtaining OFL scalars for different exploitation categories

When catch trends are stable and the stock is considered to be moderately exploited, setting the OFL to current catch levels is an appropriate action. For these stocks, a multiplier of 1.0 is recommended for the OFL.

For stocks that are considered to be heavily exploited, fishing mortality will need to be reduced to at least F_{MSY} to end overfishing and begin rebuilding the stock to levels closer to B_{MSY}. Since catch is proportional to fishing mortality for the Pella-Tomlinson model, a proportional reduction in catch will result in the same proportional reduction in fishing mortality for a given stock size. There is a time-dependency implicit in this recommendation, since a stock will immediately start to increase when fishing mortality is reduced to F_{MSY}. The Pella-Tomlinson model suggests that multipliers on average catch that reduce fishing mortality to F_{MSY} range from 0.17 when the stock is close to zero to 0.61 when the stock is at $B_{20\%}$ (Figure 6) The average of multipliers from $B_{5\%}$ to $B_{20\%}$ is 0.48. Stock levels below $B_{5\%}$ were excluded because it is unlikely that fishing mortality could be high enough to reduce stock size to such low levels. These results suggest that a multiplier of 0.5 is appropriate for the OFL when the stock is considered to be heavily exploited. Since increased yields should be possible once the stock rebuilds, use of a 0.5 multiplier for the OFL should be considered a temporary measure that will be re-evaluated periodically.

When the stock is considered lightly exploited, fishing mortality is lower than F_{MSY} and thus could potentially be increased. However a multiplier on catch would result in an immediate decrease in biomass so that that F_{MSY} would quickly be exceeded. An alternative multiplier when the stock is lightly exploited is a multiplier that would increase yield to *MSY*, so that annual catches of this amount would move the stock into the moderately exploited category without overfishing. The average of yield multipliers from $B_{66\%}$ to $B_{90\%}$ is 1.98 (Figure 7). Stock levels above $B_{90\%}$ were excluded because these stocks would likely be classified as ecosystem component species. These results indicate that a multiplier of 2.0 is appropriate for the OFL when the stock is lightly exploited. Comparisons between the Pella-Tomlinson model

with n = 1.2, the Graham-Schaefer model, and Fox model indicate that the recommended multipliers are reasonably robust to the shape of the production function. Due to the simple modeling approach used to derive these multipliers, we suggest using Table 5 as a starting point in discussions regarding appropriate OFLs.

Although three categories have been broadly defined in the above analysis, distinguishing between lightly exploited and moderately exploited stocks may be difficult in some circumstances (e.g., widely varying catch data). Under such circumstances, it may be more practical to combine these two categories and use a 1.0 scalar for both; however this would imply a decision to constrain the catch of stocks that may be lightly exploited.

E. Obtain an ABC as a proportion of the OFL

The last step in the control rule is determining the appropriate buffer between OFL and the ABC, which is based on the scientific knowledge about the stock and the uncertainty in the estimate of OFL (i.e., historical catch analysis). Since both risk policy and scientific uncertainty are involved in the choice of an ABC multiplier, input will be required from managers (i.e., Regional Fishery Management Councils) and science advisors (i.e., SSCs). Technical approaches to characterizing uncertainty are not yet possible for data-poor stocks, but it is clear that uncertainty is greater for these stocks than for data-rich assessed stocks. The size of the ABC multipliers derived from data-rich stocks provides a starting point for considering ABC multipliers for data-poor stocks. In developing ABCs, managers should consider distinguishing between high productivity stocks and low productivity stocks, the latter of which can be considered higher risk because they are more prone to becoming overfished and have long recovery times if they do become overfished. Assigning stocks to productivity categories is largely a scientific task, and can be done using productivity scores from a PSA analysis (Patrick et al., 2010) or other approaches. The degree to which different ABC multipliers are used for the productivity categories is more of a policy issue that should be decided by managers.

Table 6 lists some ABC options we developed as examples, but these are not meant to preclude managers from developing their own alternatives based on their risk preference. The alternatives in Table 6 have a greater or lesser degree of risk aversion, and contrast policy decisions to be more risk averse for low productivity stocks with those that do not. The most productive stocks tend to be coastal pelagic species such as anchovy and sardine, which have characteristics other than productivity that may be taken into account in setting the ABCs (or ACLs), such as decadal variability or importance as forage species. Other ways of grouping stocks into risk categories by productivity scores or some other characteristic are possible and should be considered.

IV. STOCK COMPLEXES

The National Standard One Guidelines (NMFS, 2009) describe the concept of a stock complex management, which is defined as a group of stocks that are managed as a single unit. Stock

complexes are considered an approach to deal with stocks that are harvested together and cannot be assessed separately because of insufficient data or resources. Stock complexes can include similar species (e.g., southeastern U.S. reef fishes) or distinct populations of the same species that support mixed-stock fisheries (e.g., the Georges Bank-Gulf of Maine stock complex of Atlantic herring). In all fishery management systems, priority is given to assessing and monitoring stocks with the highest economic value or ecological importance. Nevertheless, marine ecosystems are diverse, and become increasingly so at lower latitudes. Although there is a general need for additional stock assessments, the cost of monitoring and assessing some stocks could potentially exceed the value of landings, suggesting that there is a limit to how many stocks should be individually assessed and managed. Management of stock complexes is an approach to addressing complexity by managing stocks at a higher level than an individual stock. Whether management by stock complexes is considered successful depends on how well the approach achieves management objectives, which can be evaluated like any other management strategy. Stock complexes are likely to be useful in the same data-poor situations as average catch assessments. This section discusses the issues that should be considered when these two approaches are used together.

The formation of stock complexes should take into account life history, geographic distribution, depth distribution, and vulnerability to the fishery (NMFS, 2009). When stock complexes are formed using these criteria, it is assumed that 1) a single catch limit will be sustainable for all members of the stock complex, and 2) fishery impacts are relatively uniform across the members of stock complex (i.e., there is no targeting of individual stocks in the complex). NMFS (2009) also recommends the use of indicator stocks, which is a stock selected as being representative of the complex, and is assessed periodically as a proxy for the other members of the complex. Indicator stocks have been used in various fisheries (e.g., Hawaii Seamount and Bottomfish Fishery, Alaska Salmon Fishery, North Pacific Groundfish Fishery, etc.) and have shown various levels of success. Shertzer and Williams (2008) evaluated the utility of stock complexes and indicator stocks as a proxy of status for reef fisheries off the southeast United States coast. Two difficulties were encountered: 1) species did not group naturally into well-defined complexes based on a cluster analysis of catch data, and, 2) fishery CPUE trends of member stocks within complexes showed little synchrony, suggesting that a single stock could not be used as an indicator for the complex. This study did not distinguish between the utility of using stock complexes and indicator stocks to prevent overfishing, as opposed to being simply used for status determination. At this point, it is not possible to conclude that Shertzer and Williams (2008) results generally apply to other stock complexes, and the indicator stock approach warrants further evaluation (see Branton and Richardson, 2011). Preliminary work with Pacific Coast groundfish using the results of a PSA as well as geographic distribution in a clustering algorithm to define stock complexes shows promising results, but is not expected to be implemented until the next management cycle (Cope et al., In press).

A stock complex can be managed in-season by monitoring the aggregate landings of the complex relative to an annual catch limit as a way to control the fishing mortality experienced by the stock complex in its entirety. Determination of stock status relative to target or limit stock size could be done for the complex as whole, or for an indicator stock that is a member of the group. Determining stock status may be difficult or impossible for data-poor stocks, but a management system that successfully limits catch to sustainable levels would be expected to prevent any stock from becoming overfished. While an inability to determine whether stocks are below a critical threshold is a weakness of average catch assessments, a management system that is designed to be precautionary should accommodate this uncertainty with an appropriate response.

It is difficult to find examples where stock complexes have been implemented following the principles in NMFS (2009), most likely because the guidance is relatively new (earlier versions of the NS1 guidelines did not provide guidance on the formation of stock complexes). Stock complexes have often been established based on broad taxonomic groupings. For example, in the North Pacific, stock complexes have been established for squids and sculpins, while in New England, skates are managed as a complex despite large differences in productivity and susceptibility for members of the complex. In other cases, stock complexes are treated as a kind of warehouse for stocks that have not been dealt with using other assessment and management approaches. For an example, the "Other fish" complex used by PFMC includes several skate, shark, deepwater (e.g., finescale codling and Pacific rattail), and nearshore species (e.g., cabezon and kelp greenling). A more appropriate use of stock complexes is the PFMC management of minor rockfish species, which are grouped into complexes based on geographic distribution (north and south of 40°10" lat. N.), and depth distribution (nearshore, shelf, and slope). Another example is the "Shelf Demersal Rockfish" stock complex in the Gulf of Alaska, consisting of an assessed stock, yelloweye rockfish, and a number of other rockfish stocks occupying similar habitats that are not assessed. ABCs and OFLs are based on the assessed stock with an adjustment to account for the percent of the total catch of the stock complex consisting of other members of the complex.

Reef fishes in the Gulf of Mexico and U.S. Southeastern Atlantic Ocean were grouped into assemblages for management purposes based on multivariate statistical analyses conducted by the NMFS Southeast Regional Office. The analysis was based on landings associations, life history, and PSA. In the Gulf of Mexico, depth was the most important factor influencing assemblage composition. In the U.S. Southeastern Atlantic Ocean, depth and latitude were both important factors. Each identified assemblage contained at least one targeted, assessed species.

OFLs and ABCs for stock complexes can be specified for indicator stock(s) of the complex or set for the complex as a whole. When indicator species is not a feasible option, and OFLs and ABCs need to be set for the complex as a whole, average catches can be compiled for the complex and the OFL and ABC calculations can be done for the entire complex. This is because

the average catch of a complex is simply the sum of the average catches of the individual members of the complex. This approach would also be useful for stock complexes when estimates of the catch by species are unavailable, however some level of catch sampling is necessary to track the relative landings of stocks in a complex. Although the OFL and ABC of a stock complex can be the sum of the OFLs and ABCs for its individual stocks, the best scientific information available may not support the definition of stock-specific reference points. In the most data-poor situations, OFL and ABC may need to be based on the time series of aggregate stock catch.

The ABCs established for the indicator stocks for a complex as a whole should reflect the risk policy adopted by the Council. It is recommended by NMFS (2009) that indicator stocks be representative of the stocks within the complex with respect to their vulnerability to the fishery; otherwise the indicator stock should be chosen to represent the more vulnerable stocks in the complex. Similar rationale should be used when setting ABCs for the complex as a whole, which should take into account more vulnerable stocks within the complex. An important consideration in the use of stock complexes for management of data-poor species is that the catch of individual species within the complex is not monitored or controlled in-season. Consequently there is additional uncertainty associated with management by stock complexes that is not present when stocks are managed independently. If the objective is precautionary management, it may be necessary to build some additional conservatism into the system to account for the additional uncertainty associated with management using stock complexes. One approach would be to set an ACT for the stock complex that is less than ACL to account for management uncertainty.

V. DISCUSSION

This review of methods covers a wide range of scientific approaches to confront the challenges associated with recommending appropriate catch recommendations for data-poor stocks. Unlike previous guidance on data-poor stocks, we view the range of methods as a hierarchy, from the most informative to the most data-limited approaches, with the scalar approach recommended by Restrepo et al. (1998) for the bottom tier. A hierarchical approach to catch advice can be used for determining the most appropriate method for each stock in the short-term, depending on stock properties and data availability, as well as a broader perspective on how fishery and resource monitoring information can be improved to advance the catch advice to a more informative tier of methodology (e.g., Cadrin et al., 2004). The ORCS Working Group recognized these method-based tiers and developed an adaptive approach in which the appropriate method is hierarchical with the goal to eventually improve the scientific basis of catch limits.

The adaptive approach to determining appropriate methods for setting ABC accepts that lower-level approaches for the most data-poor stocks do not meet all of the needs of the mandated

management system or the desires of fishery stakeholders. Although it is beyond the scope of this report, the top-tier of scientific support is a stock assessment that incorporates and fully accounts for key sources of uncertainty to yield an estimate of the distribution of the OFL. Given this information on OFL and its statistical distribution, Fishery Management Councils can develop ABC control rules in which ABC is derived from an evaluation of scientific uncertainty and their acceptable probability of overfishing (see for example Ralston et al., In press). Several intermediate-tier methods (e.g., DB-SRA) support such a probabilistic approach to ABC and fully comply with NS1 guidelines. By contrast, lower tier methods (e.g., scalars of average catch) are not explicitly based on the Council's desired risk tolerance.

Lower tier methods are designed to provide catch advice so that the fishery will be sustainable, but the optimality of the derived catch and the probability of overfishing are not known. These deficiencies of the lower tier approaches can impose substantial costs in the form of larger uncertainty buffers and substantial foregone yield. The hierarchical and adaptive approach to data-poor methods for determining ABC provides incentives for improving the scientific information.

Ideally, the performance of each method in the tiered system should be evaluated for avoidance of overfishing and maintaining optimum yield (and any other potential benefits identified as management objectives) through simulation of an operating model that is tailored to the stock of interest. Furthermore, the entire tier system could be evaluated through management strategy evaluation if a decision rule is simulated for improving data and moving from lower to higher tiers.

While it is important to improve methods used to set ABCs for ORCS, even improved methods will never take the place of data and monitoring. Informed judgment plays a critical part in every ORCS approach. It cannot be avoided or assumed away. Data collection through research and monitoring are needed to eliminate the need for informed judgment.

VI. RESEARCH RECOMMENDATIONS

Due to the new requirements of the reauthorized Magnuson-Stevens Act (2006), development of methods to evaluate the status of data-poor stocks, including ORCS, is an active area of research. In particular, status determination and characterization of uncertainty are two focal study areas where significant advances are being achieved. In this regard, we believe that continued progress could be accomplished if additional research is conducted along the following lines:

- Develop and accept formal methods to elicit expert opinion from scientists, stakeholders, and managers.

- Conduct Management Strategy Evaluations (MSEs) to evaluate the robustness of methods used to characterize data-poor stocks and control rules for their management.
- Collect basic life history information on data-poor stocks, especially maximum age, to better inform estimation of natural mortality.
- Conduct stock delineation for fish species that occur over extensive ranges and/or overlapping jurisdictions.
- Improve the coverage and accuracy of catch sampling programs.
- As a basis for risk assessment, complete Productivity-Susceptibility Analyses (Patrick et al. 2009) for all stocks that are currently under fishery management plans.
- Increase the study of data-rich stocks within a meta-analytic framework to develop priors and proxies for application to data-poor stocks.
- Coordinate efforts to assemble regional landings statistics into databases in a comprehensive, thorough way.
- Monitor fishery indicators to provide additional information on sustainability of data-poor catch limits.

VII. CONCLUSIONS

The problem of setting appropriate catch levels (now called ABCs) for ORCS is not new, is not going away, and doesn't have an ideal solution. As discussed earlier, methods to deal with ORCS go back to the Restrepo et al. (1998) technical guidance. It is not realistic to assume that all, the majority of, or even many of these "data-limited" ORCS stocks will become "data-rich," allowing for comprehensive stock assessments. Past, present, and proposed methods all require the incorporation of "informed judgment" and major assumptions in critical steps of the process.

Given this situation and all of the information presented in this report, the ORCS Working Group recommends the following tiered approach to setting ABCs for ORCS:

- Apply DB-SRA to a stock, if possible. The main limitation here is the availability of a complete time series of historical catch, which is often not available.

- If it is not possible to apply DB-SRA, apply DCAC to a stock. DCAC's main limitation is that it is only appropriate for stocks with moderate to low natural mortality rates ($\leq 0.20\ yr^{-1}$).

- If DB-SRA and DCAC are not possible, apply the ORCS Working Group's Approach. The main limitation with this approach is that a number of critical decisions are required before it can be made operational. Some would also view this as an advantage, as it provides flexibility in its establishment.

- Finally, in some cases none of the above methods are practical for setting ABCs for an individual stock, as specific ORCS stocks may not have the capability to be effectively managed or monitored. In these cases, it may be best to use a stock complex approach. There are many limitations of applying a stock complex approach as described above, and the ORCS Working Group cautions against overusing or misusing this approach, as it may result in converse of precautionary management, exactly what MSA was designed to avoid.

Finally, we recommend moving forward with the research recommendations listed above, given the methods for setting ABCs for ORCS are in various stages of development and necessarily depend on adequate attention and funding in the future.

VIII. LITERATURE CITED

Anon. 2009. May 2009 Plenary Report Complete. Ministry of Fisheries, Government of New Zealand, 1,040 p. [http://fs.fish.govt.nz/Page.aspx?pk=113&dk=22080]

Branton, M., and J.S. Richardson. 2011. Assessing the value of the umbrella-species concept for conservation planning with meta-analysis. Cons. Biol. 25: 9-20.

Cadrin, S.X., J.A. Boutillier, and J.S. Idoine. 2004. A hierarchical approach to determining reference points for Pandalid shrimp. Can. J. Fish. Aquat. Sci. 61:1373-1391.

Clark, W.G. 1991. Groundfish exploitation rates based on life history parameters. Can. J. Fish. Aquat. Sci. 48:734-750.

Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR). 2009a. Schedule of conservation measures in force (2009/10 season) *in* Article IX of the Convention on the Conservation of Antarctic Marine Living Resources. 232 p.

Cope, J. M., J. DeVore, E. J. Dick, J. Budrick, D. Erickson, J. Grebel, G. Hanshew, R. Jones, L. Mattes, C. Niles, and S. Williams. In press. An approach to defining species complexes for U.S. west coast groundfishes using vulnerabilities and ecological distributions. N. Am. J. Fish. Manage.

Department of Commerce (DOC). 2007. Magnuson-Stevens Fishery Conservation and Management Act. U.S.A. Public Law 94-265.

Dick, E. J., and A. D. MacCall. 2010. Estimates of sustainable yield for 50 data-poor stocks in the Pacific Coast groundfish fishery management plan. NOAA Tech. Memo. NMFS-SWFSC-460, 201 p.

Dick, E. J., and A. D. MacCall. In press. Depletion-Based Stock Reduction Analysis: a catch-based method for determining sustainable yields for data-poor fish stocks. Fish. Res..

Dulvy, N. K., Y. Sadovy, and J. D. Reynolds. 2003. Extinction vulnerability in marine populations. Fish. Fish. 4:25-64.

Ebell, M. 2001. Evidence-based diagnosis: A handbook of clinical prediction rules. Springer, New York. 416 p.

Fletcher, R.I. 1978. On the restructuring of the Pella-Tomlinson system. Fish. Bull. 76:515-521.

Hilborn, R. 2010. Pretty good yield and exploited fishes. Mar. Pol. 34:193-196.

Hoenig, J. M. 1983. Empirical use of longevity data to estimate mortality rates. Fish. Bull. 82:898-903.

MacCall, A. D. 2009. Depletion-corrected average catch: a simple formula for estimating sustainable yields in data-poor situations. ICES J. Mar. Sci. 66:2267-2271.

McAllister, M. K., E. A. Babcock, E. K. Pikitch, and M. H. Prager. 2000. Application of a non-equilibrium generalized production model to South and North Atlantic swordfish: combining Bayesian and demographic methods for parameter estimation. Col. Vol. Sci. Pap. ICCAT, 51:1523-1550.

Musick, J. A. 1999. Criteria to define extinction risk in marine fishes. Fisheries 24:6–14.

National Marine Fisheries Service (NMFS). 2009. Final rule national standard guidelines. 74 Federal Register (January 16, 2009). 3178-3212.

Patrick, W. S., P. Spencer, J. Link, J. Cope, J. Field, D. Kobayashi, P. Lawson, T. Gedamke, E. Cortés, O. Ormseth, K. Bigelow, and W. Overholtz. 2010. Using productivity and susceptibility indices to assess the vulnerability of United States fish stocks to overfishing. Fish. Bull. 108:305-322.

Patrick, W. S., P. Spencer, O. Ormseth, J. Cope, J. Field, D. Kobayashi, T. Gedamke, E. Cortés, K. Bigelow, W. Overholtz, J. Link, and P. Lawson. 2009. Use of productivity and susceptibly indices to determine stock vulnerability, with example applications to six U.S. fisheries. NOAA Tech. Memo. NMFS-F/SPO-101, 90 p.

Porch, C. E., A. M. Eklund, and G. P. Scott. 2006. A catch-free stock assessment model with application to goliath grouper (*Epinephelus itajara*) off southern Florida. Fish. Bull. 104:89-101.

Prager, M., and K. Shertzer. 2010. Deriving Acceptable Biological Catch from the Overfishing Limit: Implications for Assessment Models. N. Am. J. Fish. Manage 30:289-294.

Quinn, T. J., II, and R. B. Deriso. 1999. Quantitative Fish Dynamics. Oxford University Press, New York, NY. 542 p.

R Development Core Team. 2009. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Ralston, S., A. E. Punt, O. S. Hamel, J. D. DeVore, and R. J. Conser. In press. A meta-analytic approach to quantifying scientific uncertainty in stock assessments. Fish. Bull..

Restrepo, V. R., G. G. Thompson, P. M. Mace, W. L. Gabriel, L. L. Low, A. D. MacCall, R. D. Methot, J. E. Powers, B. L. Taylor, P. R. Wade, and J. F. Witzig. 1998. Technical guidance on the use of precautionary approaches to implementing National Standard 1 of the Magnuson–Stevens Fishery Conservation and Management Act. NOAA Tech. Memo. NMFS-F/SPO-31, 54 p.

Restrepo, V. R., and J. E. Powers. 1999. Precautionary control rules in U.S. fisheries management: specifications and performance. ICES J. Mar. Sci. 56:846-852.

Rosenberg, A. A., J. H. Swasey, and M. Bowman. 2006. Rebuilding US fisheries: progress and problems. Front. Ecol. Environ. 4:303-308.

Shertzer, K., M. Prager, and E. Williams. 2008. A probability-based approach to setting annual catch levels. Fish. Bull. 106:225-232.

Shertzer, K. W. and E. H. Williams. 2008. Fish assemblages and indicator species: reef fishes off the southeastern United States. Fish. Bull. 106:257-269.

Walters, C. J., S. J. D. Martell, and J. Korman. 2006. A stochastic approach to stock reduction analysis. Can. J. Fish. Aquat. Sci. 63:212-223.

Witherell, D. (editor). 2010. Second National Meeting of the Regional Fishery Management Councils' Scientific and Statistical Committees. Report of a National SSC Workshop on Establishing a Scientific Basis for Annual Catch Limits. Caribbean Fishery Management Council, St. Thomas, November 10-13, 2009, 71 p.

Table 1. The Methot table showing possible actions for determining ABC based on different fishery impact categories and expert opinion. Taken from the workshop report of the 2nd National SSC meeting.

Historical Catch	**Expert Judgment**	**Possible Action**
Nil, not targeted	Inconceivable that catch could be affecting stock	Not in fishery; Ecosystem Component; SDC not required
Small	Catch is enough to warrant including stock in the fishery and tracking, but not enough to be of concern	Set ABC and ACL above historical catch; Set ACT at historical catch level. Allow increase in ACT if accompanied by cooperative research and close monitoring.
Moderate	Possible that any increase in catch could be overfishing	ABC/ACL = f(catch, vulnerability) So caps current fishery
Moderately high	Overfishing or overfished may already be occurring, but no assessment to quantify	Set provisional OFL = f(catch, vulnerability); Set ABC/ACL below OFL to begin stock rebuilding

Table 2. The natural variability factor, c, used in the New Zealand approach, as determined by the value of the natural mortality rate, M.

M	**c**
< 0.05	1.0
0.05-0.15	0.9
0.16-0.25	0.8
0.26-0.35	0.7
> 0.35	0.6

Table 3. Potential management objectives depending on stock status for ORCS Working Group Approach.

Stock status	**Potential management objectives**
Lightly exploited	Maintain current catch levels or allow for limited increases in catch
Moderately exploited	Maintain current catch levels
Heavily exploited, possibly overfished	Reduce catches to end overfishing

Table 4. Table of attributes for assigning stock status for historical catch-only assessments.

Overall scores are obtained by an unweighted average of the attributes for which scoring is possible, although alternative weighting schemes could also be considered. An initial assignment to a stock status category is: mean scores>2.5—heavily exploited; stocks with mean scores 1.5-2.5--moderately exploited; and stocks with mean scores<1.5--lightly exploited. When the attribute does not apply or is unknown it can be left unscored.

Attribute	**Stock status**		
	Lightly exploited (1)	**Moderately exploited (2)**	**Heavily exploited (3)**
Overall fishery exploitation based on assessed stocks	All known stocks are either moderately or lightly exploited. No overfished stocks	Most stocks are moderately exploited. No more than a few overfished stocks	Many stocks are overfished
Presence of natural or managed refugia	Less than 50% of habitat is accessible to fishing	50%-75% of habitat is accessible to fishing	>75% of habitat is accessible to fishing
Schooling, aggregation, or other behavior responses affecting capture	Low susceptibility to capture (specific behaviors depend on gear type)	Average susceptibility to capture (specific behaviors depend on gear type)	High susceptibility to capture (specific behaviors depend on gear type)
Morphological characteristics affecting capture	Low susceptibility to capture (specific characteristics depend on gear type)	Average susceptibility to capture (specific characteristics depend on gear type)	High susceptibility to capture (specific characteristics depend on gear type)
Bycatch or actively targeted by the fishery	No targeted fishery	Occasionally targeted, but occurs in a mix with other species in catches	Actively targeted
Natural mortality compared to dominant species in the fishery	Natural mortality higher or approximately equal to dominant species ($M \geq \overline{M}$)	Natural mortality equal to dominant species ($M \approx \overline{M}$)	Natural mortality less than dominant species ($M < \overline{M}$)
Rarity	Sporadic occurrence in catch	Not uncommon, mostly pure catches are possible with targeting	Frequent occurrence in catch
Value or desirability	Low value (< $1.00/lb, often not retained (< 33% of the time)	Moderate value ($1.00 - $2.25), usually retained (34-66% of the time)	Very valuable or desirable (e.g., > $2.25/lb), almost always retained (>66% of the time).
Trend in catches (use only when effort is stable)	Catch trend increasing or stable (assign score of 1.5)	Catch trend increasing or stable (assign score of 1.5)	Decreasing catches

Table 5. Recommended OFLs using ORCS Working Group Approach.

Stock category		
Lightly exploited ($B > B_{65\%}$)	Moderately exploited ($B \sim B_{MSY}$)	Heavily exploited ($B < B_{20\%}$)
2.0 x catch statistic	1.0 x catch statistic	0.50 x catch statistic

Table 6. Example ABC options for catch-only stocks using the ORCS Working Group Approach.

Risk level	**Alternative A**	**Alternative B**	**Alternative C**	**Alternative D**
Low risk (high productivity)	0.75 x OFL	0.75 x OFL	0.90 x OFL	0.90 x OFL
Moderate risk (moderate productivity)	0.75 x OFL	0.75 x OFL	0.75 x OFL	0.80 x OFL
High risk (low productivity)	0.75 x OFL	0.50 x OFL	0.50 x OFL	0.70 x OFL

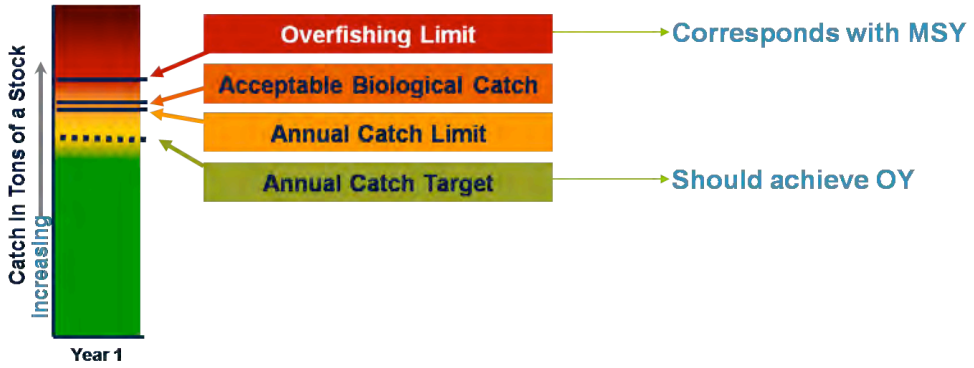

Figure 1. The relationship of catch reference points under National Standard 1.

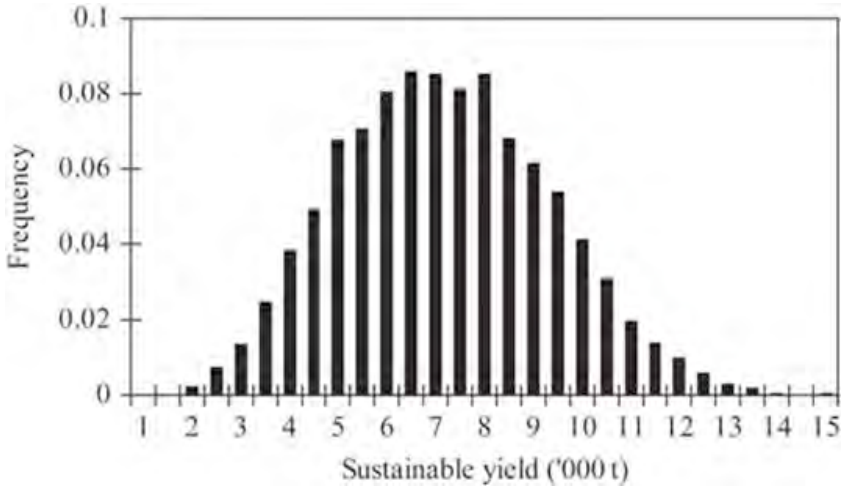

Figure 2. Distribution of 1989 widow rockfish yields from DCAC analysis (taken from MacCall 2009). The median of the sustainable yield distribution is 6,849 mt, which compares with MSY that was estimated to be 8,300 mt.

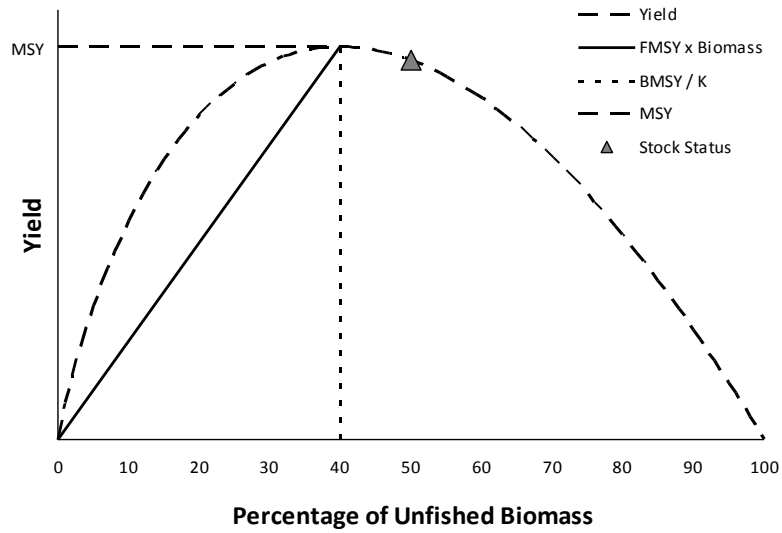

Figure 3. Graphical representation of one iteration of the DB-SRA method, shown on rescaled biomass (B_0 = 1.0).

The slope of the diagonal solid line is determined by the current value of F_{MSY}, which is the product of draws from the M and $F_{MSY} \div M$ distributions. The relative biomass that generates maximum sustainable yield (B_{MSY}/B_0) is also drawn from its distribution (value shown = 0.4). Lastly, stock status relative to unfished biomass is determined by a draw from the distribution of relative biomass depletion (Δ, value shown = 0.5). For each set of draws from the four input distributions, the catch time series determines the unique value of unfished biomass (B_0) that satisfies the current estimate of stock status. Figure courtesy of E.J. Dick.

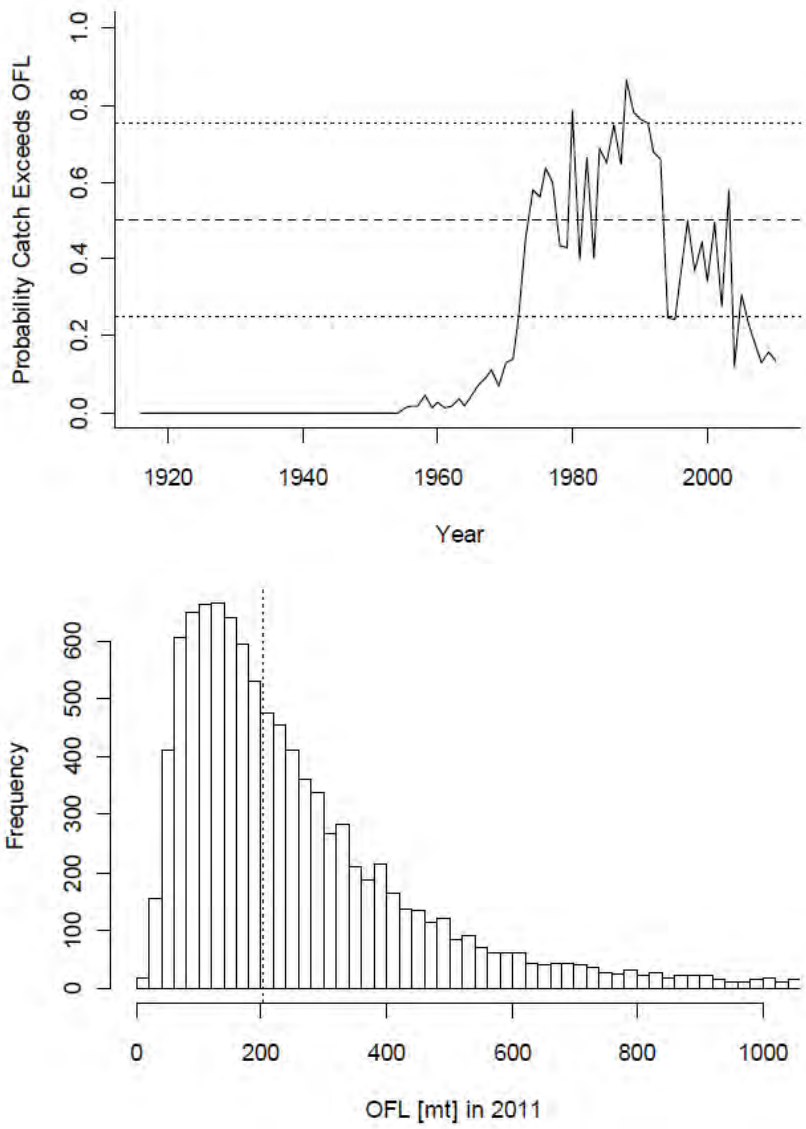

Figure 4. DB-SRA output for brown rockfish. The upper panel shows a time series of the probability that overfishing occurred in any particular year. The lower panel provides the posterior distribution of OFL in 2011 (vertical dotted line = median of distribution).

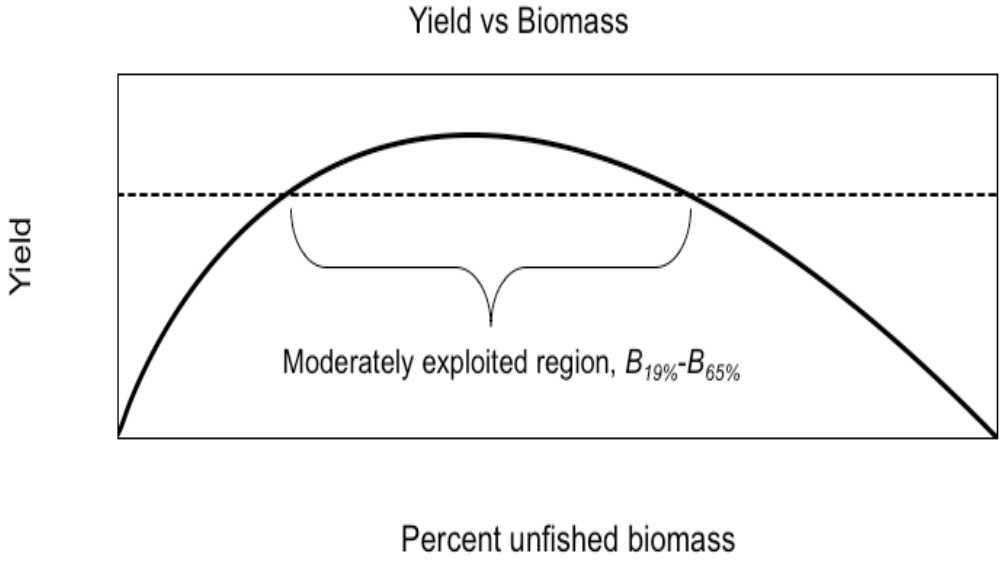

Figure 5. Equilibrium yield as a function of biomass for the Pella-Tomlinson model with n = 1.2.

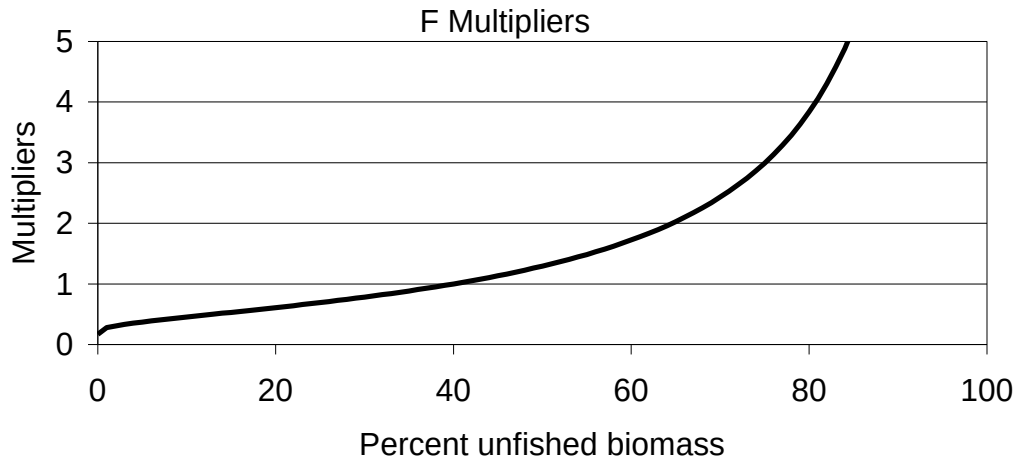

Figure 6. Multiplier for fishing mortality to reduce or increase fishing mortality to F_{MSY} for the Pella-Tomlinson model with n = 1.2.

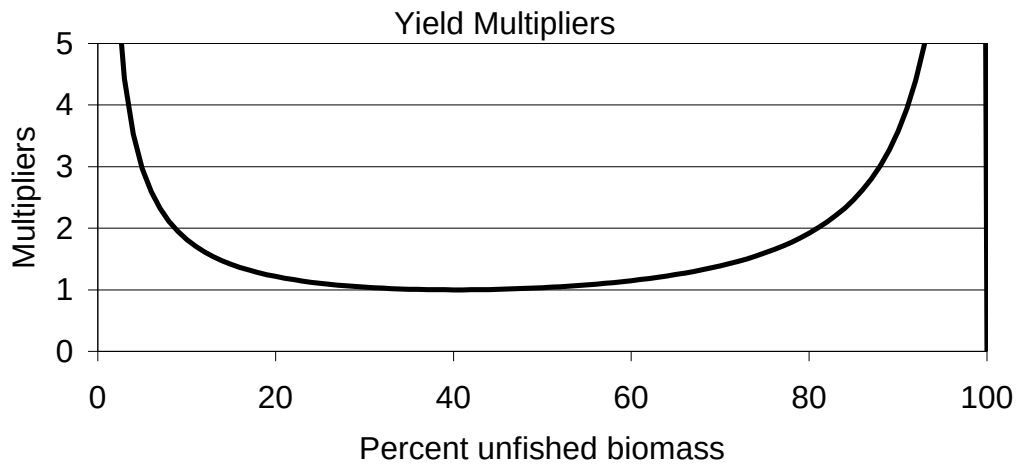

Figure 7. Multiplier for yield to reduce or increase yield to MSY for the Pella-Tomlinson model with n = 1.2.

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The global contribution of forage fish to marine fisheries and ecosystems

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Abstract

Forage fish play a pivotal role in marine ecosystems and economies worldwide by sustaining many predators and fisheries directly and indirectly. We estimate global forage fish contributions to marine ecosystems through a synthesis of 72 published Ecopath models from around the world. Three distinct contributions of forage fish were examined: (i) the ecological support service of forage fish to predators in marine ecosystems, (ii) the total catch and value of forage fisheries and (iii) the support service of forage fish to the catch and value of other commercially targeted predators. Forage fish use and value varied and exhibited patterns across latitudes and ecosystem types. Forage fish supported many kinds of predators, including fish, seabirds, marine mammals and squid. Overall, forage fish contribute a total of about $16.9 billion USD to global fisheries values annually, i.e. 20% of the global ex-vessel catch values of all marine fisheries combined. While the global catch value of forage fisheries was $5.6 billion, fisheries supported by forage fish were more than twice as valuable ($11.3 billion). These estimates provide important information for evaluating the trade-offs of various uses of forage fish across ecosystem types, latitudes and globally. We did not estimate a monetary value for supportive contributions of forage fish to recreational fisheries or to uses unrelated to fisheries, and thus the estimates of economic value reported herein understate the global value of forage fishes.

Keywords Ecosystem service, ecosystem-based management, fisheries value, forage fish, supportive values, trade-offs

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Introduction

'Forage fish' species are small or intermediate-sized pelagic species (e.g. sardine, anchovy, sprat, herring, capelin, krill) that are the primary food source for many marine predators, including mammals (Thompson *et al.* 1996; Pauly *et al.* 1998; Weise and Harvey 2008), seabirds (Crawford and Dyer 1995; Jahncke *et al.* 2004; Furness 2007; Daunt *et al.* 2008) and larger fish (Walter and Austin 2003; Butler *et al.* 2010; Logan *et al.* 2011; Magnussen 2011). Forage fish feed on zooplankton and phytoplankton and are important conduits of energy transfer in food webs for many marine ecosystems, from the tropics to the Earth's poles (Cury *et al.* 2000, 2003; Fréon *et al.* 2005; Bakun *et al.* 2010).

Fisheries for forage fish occur across broad latitudinal ranges (FAO 2010) and constitute a large and growing fraction of the global wild marine fish catch (Alder *et al.* 2008). In addition, five of the top ten fish species caught (by weight) in 2008 were forage fish species. Notably, the Peruvian anchoveta (*Engraulis ringens*, Engraulidae) supports the largest fishery in the world (FAO 2010). Nearly 90% of the global forage fish catch is used by reduction industries, which produce fish meal and fish oil (Alder *et al.* 2008). While economic studies of forage fish have focused primarily on their role as a directly harvested commodity (Herrick *et al.* 2009; Mullon *et al.* 2009; Tacon and Metian 2009), few have attempted to quantify the indirect economic contributions that these species provide (Hannesson *et al.* 2009; Herrick *et al.* 2009; Hannesson and Herrick 2010). Accounting for the indirect or support service values that prey species provide to other fisheries is inherently more difficult (Hannesson *et al.* 2009; Hannesson and Herrick 2010; Hunsicker *et al.* 2010), but doing so can provide important information to assess the trade-offs between exploiting forage fish and other species in the same marine ecosystem.

There has been growing scientific consensus for the application of ecosystem-based management approaches (Pikitch *et al.* 2004; McLeod *et al.* 2005; McLeod and Leslie 2009) in contrast to traditionally applied single-species approaches (Beddington *et al.* 2007; FAO 2010). Single-species management generally seeks to maintain populations of a target species yet ignores most ecosystem factors. Even in cases where forage fish are well managed from a single-species perspective (i.e. overfishing is not occurring), a form of 'ecosystem overfishing' *sensu* Murawski (2000) can occur, whereby depleted abundance of forage fish may negatively affect the ecosystem (Gislason 2003; Coll *et al.* 2008). Implementing an ecosystem-based approach to the management of forage fisheries seems especially warranted (Pikitch *et al.* 2004; Richerson *et al.* 2010; Smith *et al.* 2011), as these

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species exhibit strong trophic linkages and fluctuate in abundance along with seasonal, annual and inter-decadal variations in oceanographic forces (Barber and Chavez 1983; Francis *et al.* 1998; Polovina *et al.* 2001; Chavez *et al.* 2003).

Human decision-making is often influenced by comparisons of monetary values or trade-offs between different products or services (Polasky and Segerson 2009). By quantifying the value of these ecosystem products and services, such trade-offs, and the impacts of degrading ecosystems, are made more explicit (Costanza *et al.* 1997; Balmford *et al.* 2002; Barbier *et al.* 2011). The majority of economic analyses conducted for forage fish fisheries have been one dimensional (Herrick *et al.* 2009), focusing on factors or management strategies affecting the direct value of these species as a landed commodity. Only a handful of studies have enumerated the indirect values that species targeted by fisheries provide (Hannesson *et al.* 2009; Hannesson and Herrick 2010; Hunsicker *et al.* 2010; Kamimura *et al.* 2011). Because of their key position in marine food webs, the overall global importance of forage fish to fisheries and ecosystems has likely been significantly understated.

This study provides the first global estimate of forage fish value to commercially important marine fisheries and enumerates the contributions of forage fish to ecosystem predator production. We synthesized data obtained from Ecopath models representing marine ecosystems around the world. This approach allowed for broad relationships to be detected and described by summarizing data from multiple independent studies (Gurevitch and Hedges 1999), including information on feeding habits, production and catch rates. We estimated the contribution that forage fish species make to: (i) the diets and production of all forage fish predators within each modelled ecosystem, (ii) forage fish fisheries, in terms of catch and catch value and (iii) the catch and value of other commercially targeted predator species (e.g. tunas, cod, striped bass), based on their diet dependence on forage fish. We compared and contrasted these contributions and values, and investigated the effects of model structure, ecosystem type and latitude (Table 1). Finally, we use the relationships and properties revealed by these models, together with estimates of catch values at the scale of economic exclusive zones (EEZ) and high seas areas (HSA), to estimate the total value that forage fish contribute to global marine fisheries.

Methods

Compilation and synthesis of Ecopath models

Of the more than 200 Ecopath models that have been published (Fulton 2010), 72 were obtained and selected for this synthesis. The requirements for inclusion in our analysis were that the Ecopath models had to represent a marine or estuarine ecosystem in a relatively recent state (within the last 40 years), include at least one forage fish model group, and have all the necessary data and parameters openly available. The majority of Ecopath models used (90%, 65 out of 72) represented ecosystems within the past 30 years. We obtained Ecopath models from peer-reviewed publications (n = 33), technical reports (n = 36) and theses/dissertations (n = 3) (Table 1). Ecopath models that were not included failed to have at least one forage fish model group, did not have data openly available, represented older time periods (>40 years old), or a combination of all three. Collected models spanned a wide geographical range and provided relatively good global coverage of most coastal ocean areas and marine ecosystem types, with the exception of the Indian Ocean, which is poorly studied compared with other ocean areas (De Young 2006) (Fig. 1). When available, we also obtained Ecopath pedigree index information (Christensen and Walters 2004; Christensen *et al.* 2005) to assess data quality of the models.

To examine the patterns in forage fish contributions and values, we grouped Ecopath models by latitude and by ecosystem type. Latitude groupings consisted of three categories: Tropical-Subtropical (less than 30° N – less than 30° S), Temperate (greater than or equal to 30° N – 58° N and greater than or equal to 30° S – 58° S) and High latitude (greater than 58° N and greater than 58° S). We separated upwelling ecosystem models from the latitude groupings due to the dominant roles forage fish catches play in these ecosystems. Ecosystem types included: upwelling ecosystems, semi-enclosed ecosystems, non-upwelling coastal ecosystems, tropical lagoon ecosystems, open ocean ecosystems, Arctic high latitude ecosystems, and Antarctic ecosystems. All models were categorized into only one ecosystem type and latitude group (Table 1).

In this analysis we define 'forage fish' as species that occupy an important intermediary trophic position and that retain that ecological role throughout their life. We thus excluded from our

Table 1 List of the 72 Ecopath models used in this synthesis. Full model references can be found in Appendix S1 (available in the online version of this article).

Model No.	Model name	Model year(s)	Latitude group	Ecosystem type	Model area (km^2)	Pedigree index	No. of MG	No. of PMG	No. of FFMG	Value data	Krill MG	Citation
1	Western Bering Sea[1]	1980s–1990s	High latitude	Arctic high latitude	254 000	–	48	22	3	Yes	Yes	Aydin *et al.* (2002)
2	Eastern Bering Sea (1)	1980s	High latitude	Arctic high latitude	484 508	–	25	14	1	Yes	No	Trites *et al.* (1999)
3	Eastern Bering Sea (2)	1980s–1990s	High latitude	Arctic high latitude	485 000	–	38	19	2	Yes	No	Aydin *et al.* (2002)
4	Prince William Sound, Alaska (1)[2]	1980–89	High latitude	Arctic high latitude	8800	0.351	19	6	2	Yes	No	Dalsgaard and Pauly (1997)
5	Prince William Sound, Alaska (2)[3]	1994–96	High latitude	Arctic high latitude	9000	0.675	48	20	5	Yes	No	Okey and Pauly (1999)
6	Hecate Strait, Northern British Columbia	2000	High latitude	Non-upwelling coastal	70 000	–	50	34	5	Yes	Yes	Ainsworth *et al.* (2002)
7	Northern California Current	1990	Upwelling	Upwelling	69 176	–	63	38	3	Yes	Yes	Field *et al.* (2006)
8	Gulf of California	1978–79	Tropical-Subtropical	Semi-enclosed	27 900	–	25	8	1	Yes	No	Arreguín-Sánchez *et al.* (2002)
9	Huizachi-Caimanero lagoon complex, Mexico	1970–2000	Tropical-Subtropical	Tropical lagoon	175	0.750	26	6	1	Yes	No	Zetina-Rejón *et al.* (2003)
10	Golfo de Nicoya, Costa Rica	1980s–1990s	Tropical-Subtropical	Tropical lagoon	1530	–	20	10	1	Yes	No	Wolff *et al.* (1998)
11	Golfo Dulce, Costa Rica	1960–90s	Tropical-Subtropical	Tropical lagoon	750	–	20	9	1	No	No	Wolff *et al.* (1996)
12	Eastern Subtropical Pacific Ocean[4]	1993–97	Tropical-Subtropical	Open ocean	32 800 000	–	40	31	2	Yes	No	Olson and Watters (2003)
13	Northern Humboldt Current[5]	1995–96	Upwelling	Upwelling	165 000	0.638	32	15	3	Yes	No	Tam *et al.* (2008)
14	Northern Humboldt Current[6]	1997–98	Upwelling	Upwelling	165 000	0.638	32	16	3	Yes	No	Tam *et al.* (2008)
15	Sechura Bay, Peru	1996	Upwelling	Upwelling	400	0.462	22	5	1	Yes	No	Taylor *et al.* (2008)
16	Central Chile	1998	Upwelling	Upwelling	50 042	–	21	8	5	Yes	Yes	Neira *et al.* (2004)
17	Tongoy Bay, Chile	1980s–1990s	Upwelling	Upwelling	60	–	17	5	1	No	No	Wolff (1994)
18	Falkland Islands	1990s	Temperate	Non-upwelling coastal	527 000	–	44	32	2	Yes	Yes	Cheung and Pitcher (2005)
19	South Brazil Bight	1998–99	Tropical-Subtropical	Non-upwelling coastal	97 000	–	25	6	2	Yes	No	Gasalla and Rossi-Wongtschowski (2004)

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Table 1 Continued.

Model No.	Model name	Model year(s)	Latitude group	Ecosystem type	Model area (km^2)	Pedigree index	No. of MG	No. of PMG	No. of FFMG	Value data	Krill MG	Citation
20	Caeté Estuary, Brazil	1999	Tropical-Subtropical	Non-upwelling coastal	220	–	18	4	1	Yes	No	Wolff *et al.* (2000)
21	Gulf of Paria	1980s–1990s	Tropical-Subtropical	Tropical lagoon	7600	–	23	11	1	No	No	Manickchand-Heileman *et al.* (2004)
22	Northeastern Venezuela shelf	1970s–1980s	Tropical-Subtropical	Non-upwelling coastal	30 000	–	16	10	1	Yes	No	Mendoza (1993)
23	Gulf of Salamanca	1997	Tropical-Subtropical	Tropical lagoon	955	0.743	18	6	1	Yes	No	Duarte and García (2004)
24	Celestun lagoon, Mexico	2001	Tropical-Subtropical	Tropical lagoon	28	0.362	19	1	2	Yes	No	Vega-Cendejas and Arreguín-Sánchez (2001)
25	Terminos lagoon, Mexico	1980s–1990s	Tropical-Subtropical	Tropical lagoon	2500	–	20	5	1	No	No	Manickchand-Heileman *et al.* (1998a)
26	Southwestern Gulf of Mexico	1980s–1990s	Tropical-Subtropical	Tropical lagoon	65 000	–	19	9	1	No	No	Manickchand-Heileman *et al.* (1998b)
27	Laguna Alvarado, Mexico	1991–94	Tropical-Subtropical	Tropical lagoon	62	0.500	30	9	2	Yes	No	Cruz-Escalona *et al.* (2007)
28	Tampamachoco lagoon, Mexico	1980s–1990s	Tropical-Subtropical	Tropical lagoon	15	–	23	6	1	No	No	Rosado-Solórzano and Guzmán del Próo (1998)
29	Gulf of Mexico	1950–2004	Tropical-Subtropical	Non-upwelling coastal	1 530 387	–	61	23	6	No	No	Walters *et al.* (2008)
30	West Florida shelf	1980s–1990s	Tropical-Subtropical	Non-upwelling coastal	170 000	–	59	18	2	No	No	Okey *et al.* (2004)
31	Chesapeake Bay	2000	Temperate	Non-upwelling coastal	10 000	0.450	45	17	5	Yes	No	Christensen *et al.* (2009)
32	Gulf of Maine	1977–86	Temperate	Non-upwelling coastal	90 700	–	30	12	2	Yes	No	Heymans (2001)
33	Northern Gulf of St. Lawrence	1985–87	Temperate	Non-upwelling coastal	103 812	0.651	31	19	3	Yes	No	Morissette *et al.* (2003)
34	Newfoundland	1995	Temperate	Non-upwelling coastal	495 000	0.396	50	30	4	Yes	No	Heymans and Pitcher (2002)
35	Lancaster Sound region, Canada	1980s	High latitude	Arctic high latitude	97 698	–	32	2	1	No	No	Mohammed (2001)
36	West Greenland	1991–92	High latitude	Arctic high latitude	63 500	0.439	12	4	1	Yes	No	Pedersen (1994)

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Table 1 Continued.

Model No.	Model name	Model year(s)	Latitude group	Ecosystem type	Model area (km^2)	Pedigree index	No. of MG	No. of PMG	No. of FFMG	Value data	Krill MG	Citation
37	Icelandic shelf	1997	High latitude	Arctic high latitude	115 000	0.295	21	10	2	No	No	Mendy (1999)
38	Barents Sea (1)	1990	High latitude	Arctic high latitude	1 400 000	–	41	18	5	Yes	No	Blanchard *et al.* (2002)
39	Barents Sea (2)	1995	High latitude	Arctic high latitude	1 400 000	–	41	18	5	Yes	No	Blanchard *et al.* (2002)
40	Baltic Sea	1974–2000	Temperate	Semi-enclosed	396 838	–	16	5	4	Yes	No	Harvey *et al.* (2003)
41	North Sea	1981	Temperate	Non-upwelling coastal	570 000	–	25	8	4	Yes	Yes	Christensen (1995)
42	English Channel	1995	Temperate	Non-upwelling coastal	89 607	–	48	15	4	Yes	No	Stanford and Pitcher (2004)
43	Western English Channel	1994	Temperate	Non-upwelling coastal	56 452	–	52	20	4	Yes	No	Araújo *et al.* (2005)
44	Bay of Mont. St. Michel, France	2003	Temperate	Non-upwelling coastal	250	–	19	1	1	Yes	No	Arbach Leloup *et al.* (2008)
45	Cantabrian Sea shelf	1994	Temperate	Non-upwelling coastal	16 000	0.669	28	9	2	Yes	No	Sánchez and Olaso (2004)
46	Azores Archipelago	1997	Temperate	Non-upwelling coastal	584 000	0.409	44	15	1	Yes	No	Guénette and Morato (2001)
47	Northwestern Mediterranean Sea	1994	Temperate	Semi-enclosed	4500	–	23	10	3	Yes	No	Coll *et al.* (2006)
48	Orbetello lagoon, Italy	1996	Temperate	Non-upwelling coastal	27	–	9	4	1	Yes	No	Brando *et al.* (2004)
49	Northern & Central Adriatic Sea	1990s	Temperate	Semi-enclosed	55 500	0.657	40	16	3	Yes	No	Coll *et al.* (2007)
50	Black Sea	1989–91	Temperate	Semi-enclosed	423 000	–	11	4	1	Yes	No	Örek (2000)
51	Atlantic coast of Morroco	1984	Upwelling	Upwelling	586 900	0.382	38	19	2	Yes	No	Stanford *et al.* (2004)
52	Banc d'Arguin, Mauritanie	1988–98	Tropical-Subtropical	Non-upwelling coastal	10 000	0.537	22	7	1	Yes	No	Sidi and Diop (2004)
53	Cape Verde Archipelago	1981–85	Tropical-Subtropical	Non-upwelling coastal	5394	–	31	9	1	Yes	No	Stobberup *et al.* (2004)
54	Central Atlantic Ocean	1997–98	Temperate	Open ocean	18 419 191	–	39	14	1	Yes	No	Vasconcellos and Watson (2004)
55	Gambian continental shelf	1995	Tropical-Subtropical	Non-upwelling coastal	4000	–	23	7	2	No	No	Mendy (2004)

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Table 1 Continued.

Model No.	Model name	Model year(s)	Latitude group	Ecosystem type	Model area (km^2)	Pedigree index	No. of MG	No. of PMG	No. of FFMG	Value data	Krill MG	Citation
56	Guinea-Bissau continental shelf	1990–92	Tropical-Subtropical	Non-upwelling coastal	40 816	–	32	12	2	Yes	No	Amorim *et al.* (2004)
57	Senegambia	1990	Tropical-Subtropical	Non-upwelling coastal	27 600	–	18	7	2	Yes	No	Samb and Mendy (2004)
58	Guinean continental shelf	2005	Tropical-Subtropical	Non-upwelling coastal	42 969	–	35	21	2	Yes	No	Gascuel *et al.* (2009)
59	Southern Benguela Current	1990	Upwelling	Upwelling	220 000	–	32	15	4	Yes	No	Shannon *et al.* (2003)
60	South Orkneys/South Georgia	1990s	High latitude	Antarctic	1 880 000	–	30	22	2	Yes	Yes	Bredesen (2004)
61	Antarctic Peninsula	1991–2001	High latitude	Antarctic	3400	–	39	20	2	Yes	Yes	Erfan and Pitcher (2005)
62	Kerguelen Archipelago EEZ	1987–88	Temperate	Non-upwelling coastal	575 100	–	23	15	2	Yes	Yes	Pruvost *et al.* (2005)
63	Maputo Bay, Mozambique	1980s–1990s	Tropical-Subtropical	Non-upwelling coastal	1100	–	10	4	1	Yes	No	Paula e Silva *et al.* (1993)
64	Great Barrier Reef, Australia	2000	Tropical-Subtropical	Tropical lagoon	325 848	–	30	12	2	No	No	Gribble (2005)
65	Darwin Harbour, Australia	1990–2000	Tropical-Subtropical	Non-upwelling coastal	250	0.375	21	5	1	No	No	Martin (2005)
66	Brunei Darussalam	1989–90	Tropical-Subtropical	Tropical lagoon	7396	–	13	4	1	Yes	No	Silvestre *et al.* (1993)
67	Terengganu, Malaysia	1980s	Tropical-Subtropical	Tropical lagoon	1050	–	13	2	2	Yes	No	Liew and Chan (1987)
68	Hong Kong, China	1990s	Tropical-Subtropical	Non-upwelling coastal	1680	–	37	12	1	No	No	Buchary *et al.* (2002)
69	Tapong Bay, Taiwan	1999–2001	Tropical-Subtropical	Tropical lagoon	4	0.820	18	1	2	No	No	Lin *et al.* (2006)
70	East China Sea	1997–2000	Tropical-Subtropical	Open ocean	770 000	0.636	45	19	6	Yes	No	Jiang *et al.* (2008)
71	Bohai Sea	1982–83	Temperate	Non-upwelling coastal	77 000	–	13	5	1	Yes	No	Tong *et al.* (2000)
72	Central North Pacific	1990s	Tropical-Subtropical	Open ocean	9 888 350	–	25	20	2	No	No	Cox *et al.* (2002)

[1]48 group model, [2]pre-oil spill model, [3]post-oil spill model, [4]ETP7 model, [5]La Niña model, [6]El Niño model. MG = Model groups, PMG = Predator model groups, FFMG = Forage fish model groups.

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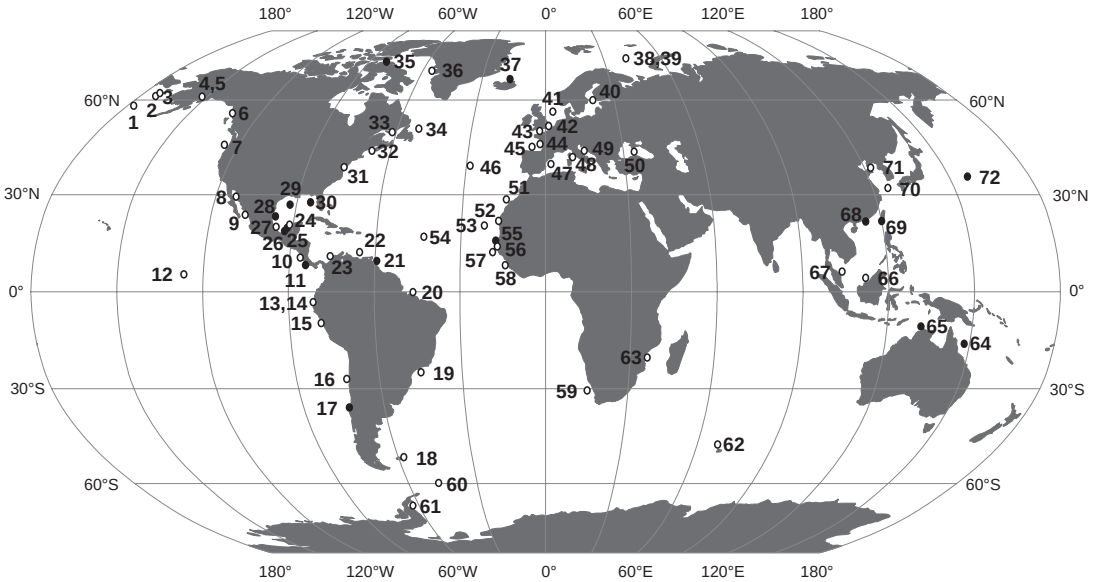

Figure 1 Approximate locations of the 72 Ecopath models used in this synthesis. Ecopath models where monetary value could (white circles) and could not (black circles) be calculated. Model numbers correspond to Table 1.

definition species that assume this role early in life but later move into higher trophic categories as they age (e.g. North Pacific hake, Blue whiting, Alaska pollock).

Data extraction

We extracted model groups, catch data, diet composition matrices, biomass data, production-to-biomass ratios and model area (km^2) from tables in Ecopath model publications and transferred them into separate Microsoft©Excel spreadsheets. When necessary, we converted all Ecopath catch and biomass data not conforming to the standard Ecopath units for catch (tonne km^{-2} $year^{-1}$) and biomass (tonne km^{-2}).

The majority (83%) of Ecopath models in this analysis had data on total catch (landings plus discards). The remaining 17% (12 out of 72) of the models only published landings data with no estimates of discards. For these 12 models we assumed that discards were zero in our analysis. Discards represent approximately 8% of the marine fisheries catch by weight globally but vary greatly among species and ecosystems (Kelleher 2005).

Ecopath models contain interactive 'groups' which can be composed of either single or multiple species that share similar life histories or ecological functions (Polovina 1984). We used the Ecopath models assembled with the original model groups as specified by the model authors. The published models generally included a list of species or taxa constituting each model group. When such taxonomic information was provided, we used this information to create an inventory of all species. In this study, we classified a model group as a forage fish group whenever at least one forage fish species was included. For instance, if an anchovy species was a component of a larger model group called 'Small Pelagics', along with gobies and juvenile mackerels, then we considered this group as a forage fish group, even though other species in that group may not necessarily meet our definition of forage fish. The majority (65% or 105 out of 161) of forage fish model groups consisted entirely of forage fish species. Of the remaining 56 forage fish model groups, 30 were discerned to be dominated by forage fish species, while information on the preponderance of forage fish species was lacking for the other 26 model groups. The one exception to our classification of forage fish model groups applied to krill (Order: Euphausiaea), which were only represented as separate model groups in 9 of the 72 Ecopath models in this analysis (Table 1). In the few remaining Ecopath models where krill were present in the ecosystem but not as a separate model group, they were grouped into various 'Zooplankton' groups. We chose to exclude these 'Zooplankton' model groups as forage fish groups in this analysis and only included contributions of krill from models

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with defined krill model groups. We acknowledge that this modelling approach may cause differences between ecosystems in terms of forage fish contributions (i.e. those that have a separate krill group and those that do not) but assumed in this analysis that if model authors grouped krill separately it was due to their perceived importance in the ecosystem. We considered it was more appropriate to include krill groups as forage fish in this analysis when present than to completely exclude them.

Importance of forage fish to ecosystem predators

We identified forage fish predators in all models and their dependence on forage fish (percent of forage fish in diet) from the respective model diet matrix. We defined forage fish predators as model groups whose diets contained any fraction of one or more forage fish model groups (i.e. diet of >0% forage fish). This definition allowed for forage fish species to be included as forage fish predators, if their diet consisted of forage fish. This rarely occurred, with only 3.9% (35 out of 895) of forage fish predators also included as forage fish. Forage fish predators were then categorized into the following dependence groups: (i) low dependence on forage fish (>0 to <25%), (ii) moderate dependence (≥ 25 to <50%), (iii) high dependence (≥ 50 to <75%) and (iv) extreme dependence on forage fish (≥ 75%).

We estimated the portion of each forage fish predator's production supported by forage fish across all ecosystem models using equations modified from Hunsicker *et al.* (2010). First, we calculated the total annual production (P_j, units: tonne km^{-2} year^{-1}) of each forage fish predator group j in each Ecopath model using Equation (1), in which predator group j's biomass (B_j, units: tonne km^{-2}) was multiplied by that respective predator group's production-to-biomass ratio ($P\ B^{-1}$, units: year^{-1}).

$$P_j = B_j \left(\frac{P}{B}\right)_j \tag{1}$$

Second, we found the portion of each predator group's total annual production ($P_{i,j}$) supported by forage fish prey groups (i), by multiplying predator group j's respective diet dependence on forage fish ($D_{i,j}$) by P_j using Equation (2).

$$P_{i,j} = D_{i,j} P_j \tag{2}$$

The total support service contribution of forage fish to ecosystem predator production (S_z) therefore can be found using Equation (3), as the product of ($D_{i,j}$) and (P_j) summed over all forage fish groups (i) and predator groups (j) in an ecosystem.

$$S_z = \sum_j \sum_i D_{i,j} P_j \tag{3}$$

Hunsicker *et al.* (2010) showed that $D_{i,j}$ is equivalent to the contribution of prey group i to predator group j's production ($P_{i,j}$) when assimilation and energy content of prey items are roughly equivalent. In the absence of detailed data on these variables, we assumed they were equal to one another but note that our analysis underestimates $P_{i,j}$ because of the generally high energy content of forage fish species (Van Pelt *et al.* 1997; Anthony *et al.* 2000) compared to most predators. Thus, our estimates for the support service contribution of forage fish to ecosystem predator production can be considered conservative in this regard.

Direct and support service contributions of forage fish to commercial fisheries

We calculated the contributions of forage fish to fisheries in terms of catch (tonne km^{-2} year^{-1}) for all 72 Ecopath models and catch value (2006 USD km^{-2} year^{-1}) for a subset of models that had adequate taxonomic information (n = 56). Ecopath models were grouped into categories based on ecosystem type and latitude of the model (Table 1). We used a global ex-vessel price database, developed by Sumaila *et al.* (2007) to obtain ex- vessel 'real' price data for all fished species in our Ecopath models. Ex-vessel 'real' price is defined as the actual prices that fishermen receive for their products before processing and is hereafter simply referred to as price. In this analysis, we use 'value' to refer to ex-vessel fish price times quantity (gross returns) and not economic profit (net returns).

We obtained total catch data for every country participating in fisheries in a respective Large Marine Ecosystem (LME) in year 2006 from the Sea Around Us project LME database (Watson *et al.* 2004; www.seaaroundus.org), and used the ex-vessel price database to compile country specific ex-vessel price data for every species in the 56 models. Information on every fishing country in each LME and their respective total catch can be accessed on the Sea Around Us project LME database website (www.seaaroundus.org). To account for differences in prices between countries operating in a given LME, we calculated a weighted average based on the total catch in 2006 of all participating countries within that LME. When

model groups consisted of two or more species, the ex-vessel price for the model group was found by averaging the ex-vessel prices for all respective species within, which were each weighted by the catches of participating countries. We used these averaged ex-vessel model group prices to calculate fisheries value (2006 USD km^{-2} $year^{-1}$) for each respective model group in all 56 Ecopath models.

For small geographic areas (e.g. estuaries, lagoons, and small coastal areas), we assumed that only the country surrounding these waters fished them. We made this assumption because detailed information about which specific countries fish within an Ecopath model area is not usually published. For the few Ecopath models that were located outside a defined LME area (e.g. Central North Pacific Ocean, Central Atlantic Ocean and Eastern Subtropical Pacific Ocean), we assumed participating fishing countries to be those nearest to, and surrounding, the model locations. Ecopath models of island countries and territories that fell outside of LME boundaries (e.g. the Azores Archipelago) were assumed to be fished only by that country, or the country of which it is a territory.

We estimated forage fish catch by summing the catch of all forage fish model groups in each respective ecosystem model. Catch value (2006 USD km^{-2} $year^{-1}$) was estimated for each respective forage fish model group by multiplying the catch (tonne km^{-2} $year^{-1}$) by the respective ex-vessel price (2006 USD tonne $^{-1}$) (Sumaila *et al.* 2007). Similarly, we summed catch values for all forage fish model groups to find the total forage fish catch value (2006 USD km^{-2} $year^{-1}$) for each Ecopath model. We estimated the support service contributions of forage fish to the catch (S_C) and catch value (S_V) of other commercially targeted model groups by using Equation (3), except that the predator group's total annual production (P_j) was replaced by the catch (C_j, Equation 4) and catch value (V_j, Equation 5) of each predator group j.

$$S_c = \sum_j \sum_i D_{i,j} C_j \tag{4}$$

$$S_V = \sum_j \sum_i D_{i,j} V_j \tag{5}$$

Forage fish contribution to global fisheries value

Forage fish species contribute to the value of global fisheries in two important ways: (i) by their direct catch value and (ii) by their support service as prey to the value of other commercially targeted species. Using forage fish value estimates for these contributions from each Ecopath model, we extrapolated to Exclusive Economic Zone (EEZ) or High Seas Area (HSA) regions to derive global estimates. We worked at the scale of EEZs and HSAs because independent estimates of forage fish catch values were available at this scale (Sumaila *et al.* 2007) to complement the values we estimated in Ecopath models. We assumed that a single Ecopath model representing an area within an EEZ or HSA region provided a reasonable depiction of the relationship between the support service value of forage fish and the total fisheries value for the entire region. A breakdown of the actual area covered by our Ecopath models as a percentage of the total EEZ/HSA area or the total Inshore fishing area (IFA) can be found in Table S1 (see Appendix S2). The IFA is defined by the Sea Around Us Project database (www.seaaroundusproject.org) as the area between the shoreline and whichever comes first, either the 200 m bathycline or a distance of 50 km from the shoreline. The majority of the global marine fisheries catch value (78%) and forage fish catch value (97%) is derived from IFAs (Sumaila *et al.* 2007)(www.seaaroundusproject.org). A summary of Ecopath model coverage in terms of EEZ/HSA or IFA area and fisheries value is provided in Table S2 (see Appendix S2). When multiple Ecopath models were available for a given EEZ or HSA region, we used average values weighted by the geographic area covered by each ecosystem model. We quantified global forage fisheries value by summing the value of forage fish across all EEZs and HSAs in the Sea Around Us project database. The majority of forage fish species in these databases were separated into two commercial groups, 'Herring-likes' and 'Anchovies'. We assumed that the total direct forage fish catch value for each respective EEZ and HSA was the sum of these two commercial groups. When data on 'Herring-likes' and 'Anchovies' were missing from this database, we used data available for forage fish categorized by species group. This method may slightly underestimate forage fisheries value, as it did not include some forage fish species that were grouped into other non-forage fish commercial groups.

To estimate the global support service value of forage fish to other commercially targeted species, we extrapolated the values estimated for each

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Ecopath model to each corresponding EEZ and HSA region. To do this, we used Ecopath models with value data available and calculated an Ecopath value ratio (EVR) using Equation (6). In Equation (6), the catch value of forage fish predators supported by forage fish (Sv) was divided by the total fishery catch value (y) of the Ecopath model, excluding non-cephalopod, non-krill invertebrates (e.g. other decapods, bivalves). By assuming that EVRs found in our Ecopath models are representative of the larger EEZs or HSAs in which they are located, we calculated the total support service value ($Supportive) of forage fish in each EEZ and HSA. Using Equation (7) we multiplied the respective EVR for an EEZ or HSA by the total fishery catch value (excluding non-cephalopod, non-krill invertebrates) for that area calculated from the Sea Around Us database ($SAUP).

$$\mathrm{EVR} = \frac{Sv}{y} \tag{6}$$

$$\$\mathrm{Supportive} = \mathrm{EVR} \times \$\mathrm{SAUP} \tag{7}$$

Ecopath models were available for 25% (64 out of 257) of the world's EEZs and HSAs, which represents 33% of the total EEZ/HSA area (Table S2, Appendix S2). In the majority (36 out of 64) of these EEZ/HSA areas, Ecopath model coverage was >50% of the respective EEZ/HSA area (see Appendix S2, Tables S1 and S2). These EEZ/HSAs constitute 39% of the global marine catch value (2006 $USD) excluding non-cephalopod and non-krill invertebrates (i.e. other decapods, bivalves) and 53% of the global forage fish catch value (2006 $USD) (Table S2, Appendix S2). Ecopath model coverage of IFAs was even greater, representing 47% of the total area (km^2) (Table S2, Appendix S2). An additional 86 EEZs and HSAs (see Table S1, Appendix S2), which did not have Ecopath models, were included under the assumption that the Ecopath model in the EEZ or HSA immediately adjacent was representative of that neighbouring EEZ or HSA. These EEZs and HSAs represented an additional 28% of the global forage fish catch value to fisheries. The remaining 107 EEZs or HSAs did not have Ecopath models or an adjacent neighbour with an Ecopath model (e.g. isolated islands) and represented only 19% of the global forage fish value to fisheries. In these EEZ/HSA areas, we applied an EVR based on the average of EVRs from other Ecopath models in the same latitudinal group. We calculated all values as ex-vessel price values in 2006 $USD and summed all support service values and forage fisheries catch values across all EEZs and HSAs. This produced our estimate of forage fish contribution to global fisheries value.

Results

Quality of Ecopath models

Ecopath pedigree indices (Christensen and Walters 2004) were available for 22 models (Table 1). The Ecopath pedigree index varies with the quality of data within Ecopath models, and values can range from 0 (not reliable) to 1 (highly reliable) (Christensen and Walters 2004; Christensen *et al.* 2005). Ecopath pedigree indices in this analysis ranged from 0.295 to 0.820 with the majority (55%, 12 out of 22) exceeding 0.5 (Table 1). Differences were observed in pedigree indices of models published in peer-reviewed journals (Ecopath pedigree mean = 0.625, median = 0.638, n = 11) and technical reports (Ecopath pedigree mean = 0.450, median = 0.408, n = 11). None of our indices were in the poorest quality level grouping, wherein data are considered to be no better than guesses (<0.2; Christensen and Walters 2004; Christensen *et al.* 2005). Moreover, the average and median pedigree indices observed in this study (0.518 and 0.537, respectively) were substantially higher than those for other studies (0.441 and 0.439, respectively) (Morissette *et al.* 2006; Morissette 2007).

Extent of predator dependence on forage fish

Seventy-five percent (54 out of 72) of the Ecopath models used in this analysis had at least one model group that was highly ($\geq$ 50% but <75% of diet) or extremely dependent ($\geq$ 75% of diet) on forage fish. Twenty-nine percent (21 out of 72) of the models included at least one extremely dependent predator group. We found extremely dependent predators present across all latitude groups and ecosystem types, with the exception of open ocean ecosystems. Extremely dependent predators accounted for only 5.8% (52 out of 895) of all forage fish predators and consisted of fishes (n = 30), seabirds (n = 12), marine mammals (n = 9) and one species of squid (*Loligo gahi*, Loliginidae). Amongst conspecific predator groups, however, seabirds had the highest percentage of extremely dependent predators, with

19% (12 out of 62) of all seabird predators having diets $\geq 75\%$ forage fish. Extremely dependent predators groups were most commonly found in upwelling and Antarctic ecosystem types, with an average of two and five extremely dependent predators per model, respectively. Many of these extremely dependent predator species were also listed on the IUCN Red List (Table 2).

We evaluated the relative frequency of various levels of forage fish dependencies and how they varied across ecosystem types by combining data from all models. Pooled data across all ecosystem models indicated that on average, 49% of all predator groups in our models relied on forage fish for at least 10% of their dietary requirements (Fig. 2). Forage fish predators that are highly or extremely dependent on forage fish account for 16% of all predator groups in marine ecosystem models on average. Predators with diets consisting of more than 90% forage fish were also found but represented fewer than 5% of all predator groups in this analysis.

When comparing across ecosystem types, Antarctic ecosystem models generally had the greatest

Table 2 Extremely dependent forage fish predators ($\geq 75\%$ forage fish in their diets) found in this synthesis that have taxonomic information and are evaluated by the International Union for Conservation of Nature (IUCN) Red List. Model numbers correspond to model names in Table 1.

Common name	Scientific name	Family	IUCN Status[1]	Population trend	Model No(s)
Marine Mammals					
Sei Whale	*Balaenoptera borealis*	BALAENOPTERIDAE	Endangered	Unknown	(1, 60)
Blue Whale	*Balaenoptera musculus*	BALAENOPTERIDAE	Endangered	Increasing	(1, 60)
Fin Whale	*Balaenoptera physalus*	BALAENOPTERIDAE	Endangered	Unknown	(1, 60)
Common Minke Whale	*Balaenoptera acutorostrata*	BALAENOPTERIDAE	Least Concern	Stable	(1, 60)
Southern Right Whale	*Eubalaena australis*	BALAENIDAE	Least Concern	Increasing	60
Grey Seal	*Halichoerus grypus*	PHOCIDAE	Least Concern	Increasing	40
Crabeater Seal	*Lobodon carcinophagus*	PHOCIDAE	Least Concern	Unknown	60
Humpback Whale	*Megaptera novaeangliae*	BALAENOPTERIDAE	Least Concern	Increasing	(1, 60)
Ringed Seal	*Phoca hispida*	PHOCIDAE	Least Concern	Unknown	40
Seabirds					
Black-browed Albatross	*Thalassarche melanophrys*	DIOMEDEIDAE	Endangered	Decreasing	18
Macaroni Penguin	*Eudyptes chrysolophus*	SPHENISCIDAE	Vulnerable	Decreasing	(60, 62)
Humboldt Penguin	*Speriscus humboldtii*	SPHENISCIDAE	Vulnerable	Decreasing	17
Peruvian Pelican	*Pelecanus thagus*	PELECANIDAE	Near Threatened	Decreasing	(13–14, 15, 17)
Guanay Cormorant	*Phalacrocorax bougainvillii*	PHALACROCORACIDAE	Near Threatened	Decreasing	(13–14, 15)
Sooty Shearwater	*Puffinus griseus*	PROCELLARIIDAE	Near Threatened	Decreasing	1
Gentoo Penguin	*Pygoscelis papua*	SPHENISCIDAE	Near Threatened	Decreasing	(60, 62)
King Penguin	*Aptenodytes patagonicus*	SPHENISCIDAE	Least Concern	–	62
Rhinoceros Auklet	*Cerorhinca monocerata*	ALCIDAE	Least Concern	–	1
Southern Rockhopper Penguin	*Eudypte schrysocome*	SPHENISCIDAE	Least Concern	Decreasing	62
Tufted Puffin	*Fratercula cirrhata*	ALCIDAE	Least Concern	–	1
Southern Giant-petrel	*Macronectes giganteus*	PROCELLARIIDAE	Least Concern	Decreasing	18
Cassin's Auklet	*Ptychoramphus aleuticus*	ALCIDAE	Least Concern	–	1
Peruvian Booby	*Sula variegate*	SULIDAE	Least Concern	–	(13–14, 15, 17)
Common Guillemot	*Uria aalge*	ALCIDAE	Least Concern	–	1
Fish					
Yellowfin Tuna	*Thunnus albacares*	SCOMBRIDAE	Near Threatened	Decreasing	(13–14, 56)
Common Dolphinfish	*Coryphaena hippurus*	CORYPHAENIDAE	Least Concern	Stable	(13–14)
West African Ladyfish	*Elops lacerta*	ELOPIDAE	Least Concern	Unknown	56
Skipjack Tuna	*Katsuwonus pelamis*	SCOMBRIDAE	Least Concern	Stable	56
North Pacific Hake	*Merluccius productus*	MERLUCCIIDAE	Least Concern	Unknown	7
Sockeye Salmon	*Oncorhynchus nerka*	SALMONIDAE	Least Concern	Stable	(4–5)
Pacific Bonito	*Sarda chiliensis*	SCOMBRIDAE	Least Concern	Decreasing	(13–14)

[1]IUCN (2011) IUCN Red List of Threatened Species. Version 2011.2 http://www.iucnredlist.org Downloaded on 2 December 2011.

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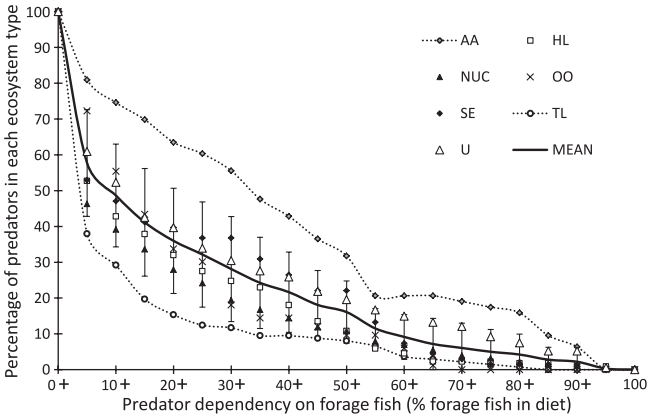

Figure 2 Percentage of forage fish predators in analysed ecosystems (n = 72) and their dependence on forage fish (% forage fish in diet). Solid line represents the Mean ± SD for all predators in this analysis. Ecosystem types: AA, Antarctic; OO, open ocean; U, upwelling current; HL, Arctic high latitude; SE, semi-enclosed; NUC, non-upwelling coastal; TL, tropical lagoon.

proportion of forage fish predators in their models for any level of forage fish dependence compared to other ecosystem model types (Fig. 2). Upwelling ecosystems had the second highest percentage of predators with 90% forage fish dependence levels. Tropical lagoon ecosystem types had the lowest proportion of predators for a given forage fish dependence level (Fig. 2).

Support service contribution to ecosystem predator production

The total predator production (tonne km^{-2} year^{-1}) supported by forage fish varied greatly among the 72 models in this analysis (Fig. 3). Supported predator production was the largest for two upwelling ecosystem models, the northern California Current model and central Chile model, and one non-upwelling coastal ecosystem (Falkland Islands model). Forage fish contributed 52 and 17 tonne km^{-2} year^{-1} to predator production in northern California Current and central Chile models respectively, and the contribution in the Falkland Islands model was 18.9 tonne km^{-2} year^{-1}. When the contribution of krill to the production of other forage fish (e.g. krill, sardines, anchovies) was removed in the northern California Current and Falkland Islands models, the support service to predators dropped to 32 and 3.3 tonne km^{-2} year^{-1} respectively.

Across ecosystem types, the greatest support service contribution of forage fish to predator production was seen in upwelling and Antarctic ecosystems (Fig. 4a). The support service contribution to predator production in both these ecosystem types exceeded 9 tonne km^{-2} year^{-1}, and were more than three times greater than values seen for Arctic ecosystems and non-upwelling coastal ecosystems and more than an order of magnitude greater than open-ocean, tropical lagoon and semi-enclosed ecosystem types (Fig. 4a). In terms of latitude groupings (with upwelling ecosystems excluded), we found the greatest support service contributions to predator production in high latitude regions (3.79 tonne km^{-2} year^{-1} ± 1.23 SE), followed by temperate latitudes (1.81 tonne km^{-2} year^{-1} ± 0.59 SE) and finally tropical-subtropical latitudes (1.18 tonne km^{-2} year^{-1} ± 0.17 SE; Fig. 4b).

Importance of forage fish to commercial fisheries

Forage fish catch varied greatly among models examined, both in tonnage and ex-vessel price value. In some models, we found no forage fish catch reported (e.g. Central Atlantic Ocean), while others had extremely large forage fish catches (e.g. Sechura Bay, Peru). The highest forage fish catches were found in the Humboldt Current models where the Peruvian anchoveta fishery operates. Of the three Humboldt Current models, the Sechura Bay (Peru) model had an extraordinarily high level of forage fish catch (81 tonne km^{-2} year^{-1}) valued at \$35 497 (USD km^{-2} year^{-1}), whereas in the northern Humboldt Current models for El Niño and La Niña periods, forage fish catches

Figure 3 Support service of forage fish to ecosystem predator production across all Ecopath models in this analysis (n = 72).

were 20 tonne km^{-2} year^{-1} (\$934 USD km^{-2} year^{-1}) and 39 tonne km^{-2} year^{-1} (\$2020 USD km^{-2} year^{-1}), respectively.

Forage fish contributed important support to other commercial fisheries in all models that contained such fisheries. Of the ecosystems we examined, forage fish were most important as prey, in terms of tonnage, to commercial fisheries in central Chile (3.82 tonne km^{-2} year^{-1}), Prince William Sound (pre-oil spill model; 3.58 tonne km^{-2} year^{-1}) and the northern California Current (3.13 tonne km^{-2} year^{-1}; Fig. 5). In terms of value, forage fish provided the greatest support service to fisheries in the Prince William Sound model (pre-oil spill model) at a value of \$5942 USD km^{-2} year^{-1}, followed by the Chesapeake Bay at a value of \$3095 USD km^{-2} year^{-1}. The high support service values in these ecosystems are due to the large contribution of forage fish to the diets of salmon (*Oncorhynchus* spp., Salmonidae) in Prince William Sound and striped bass (*Morone saxatilis*, Percichthyidae) in Chesapeake Bay, both of which have relatively high ex-vessel price values.

In 13 out of 56 models, 100% of the total forage fish value was derived from support to other fisheries (i.e. there were no forage fish fisheries reported in these 13 ecosystems). In more than half the models (30 out of 56), the value of the fisheries

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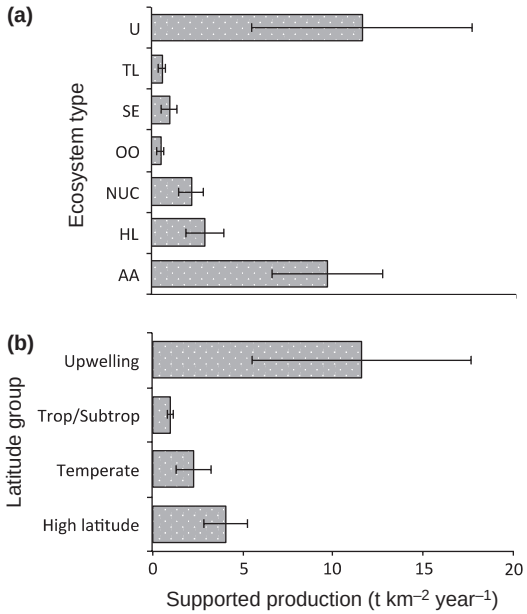

Figure 4 Mean forage fish contribution to (non-commercial) ecosystem predator production by ecosystem type (a) and latitude grouping (b) with standard error plotted. Ecosystem types: U, upwelling current; TL, tropical lagoon; SE, semi-enclosed; OO, open ocean; NUC, non-upwelling coastal; HL, Arctic high latitude; and AA, Antarctic.

supported by forage fish was greater than the value of forage fish catch (Fig. 6).

Comparisons across latitude groups and ecosystem types

The largest forage fish catches were found in the tropical-subtropical latitude group (4.95 tonne km^{-2} year^{-1} ± 2.5 SE) and decreased monotonically as polar regions were approached. In contrast, the level of other commercial catch supported by forage fish was the lowest in the tropical-subtropical latitude group (0.23 tonne km^{-2} year^{-1} ± 0.05 SE) but greater in temperate (0.63 tonne km^{-2} year^{-1} ± 0.2 SE) and high latitude ecosystems (0.35 tonne km^{-2} year^{-1} ± 0.29 SE). We separated upwelling ecosystem models from these latitude groupings, as forage fish catches play a dominant role in these ecosystems. We found that temperate models had the highest forage fish fisheries catch when compared with the remaining two latitude groups (Fig. 7a). Forage fish catch value (excluding upwelling ecosystems) was the greatest in the tropical-subtropical latitude group and diminished poleward (Fig. 7b). The support service provided by forage fish for other commercial fisheries, in both catch and catch value, increased poleward so that it was equivalent (in catch) or exceeded (in catch value) the forage fish catch or catch value in high latitudes (Fig. 7a,b).

Forage fish catch (tonne km^{-2} year^{-1}) was the highest in upwelling ecosystems (Fig. 8a), exceeding that of all other ecosystem types combined by a factor of four. Forage fish catch exceeded the catch of other model groups that preyed on forage fish for all ecosystem types (Fig. 8a). Similarly, forage fish had the highest catch value in upwelling ecosystems at \$5660 USD km^{-2} year^{-1} ± \$4980 SE (Fig. 8b). Other ecosystem types had substantially lower forage fish catch values, each contributing < \$830 USD km^{-2} year^{-1}. The value of forage fish catches was the smallest in high latitude Arctic and Antarctic ecosystems (\$184 USD km^{-2} year^{-1} and \$149 USD km^{-2} year^{-1}, respectively). In contrast, the support service value of forage fish was the greatest in the Arctic ecosystems (HL, mean = \$706 USD km^{-2} year^{-1}) – over 3.5 times greater than the forage fish value for that ecosystem type (Fig. 8b).

Global estimate of forage fish value to fisheries

The estimated total ex-vessel price value of forage fish to global commercial fisheries was \$16.9 billion (\$USD). This estimate combines global forage fish fishery value of \$5.6 billion (33%, USD) with a support service value to other fisheries of \$11.3 billion (67%, USD). This value represents nearly 20% (\$16.9b/\$85b) of the ex-vessel catch values of all world fisheries, estimated at between \$80 and 85 billion USD year^{-1} (Sumaila *et al.* 2007; FAO 2010). Importantly, we found that the value of commercial fisheries supported by forage fish (e.g. cod, striped bass, salmon) was twice the value of forage fish fisheries at a global scale.

Discussion

We recognize that using Ecopath models, like any mathematical representation of an ecosystem, has certain limitations. However, our approach was built around the idea that, within the constraints of the model assumptions, averaging across many models will at least reduce the effects of stochastic uncertainty. Ecopath models provide only a single spatial and temporal representation of an ecosystem and they contain numerous assumptions

Figure 5 Support service contributions of forage fish to other fisheries catch across all Ecopath models (n = 72).

whose consequences are often impossible to assess and could be important. This means, at the very least, that they do not capture changes in ecosystem dynamics and fisheries effort over space and time. Models are constructed based on data availability and the author's understanding of the ecosystem and research objectives, allowing for a gradient in model complexity and quality. The models contain simplified diet information of predators included in the models, which needs to be considered when interpreting or using the results of this study. For example, some Ecopath models lacked predators that are known to prey on forage fish, and in other cases, investigators pooled individual predator species together into a single trophic group. Nearly 30% (21 out of 72) of the models in our study did not have any seabird model groups, while 33% (24 out of 72) did not have a marine mammal group. Our estimates for predator production therefore are likely conserva-

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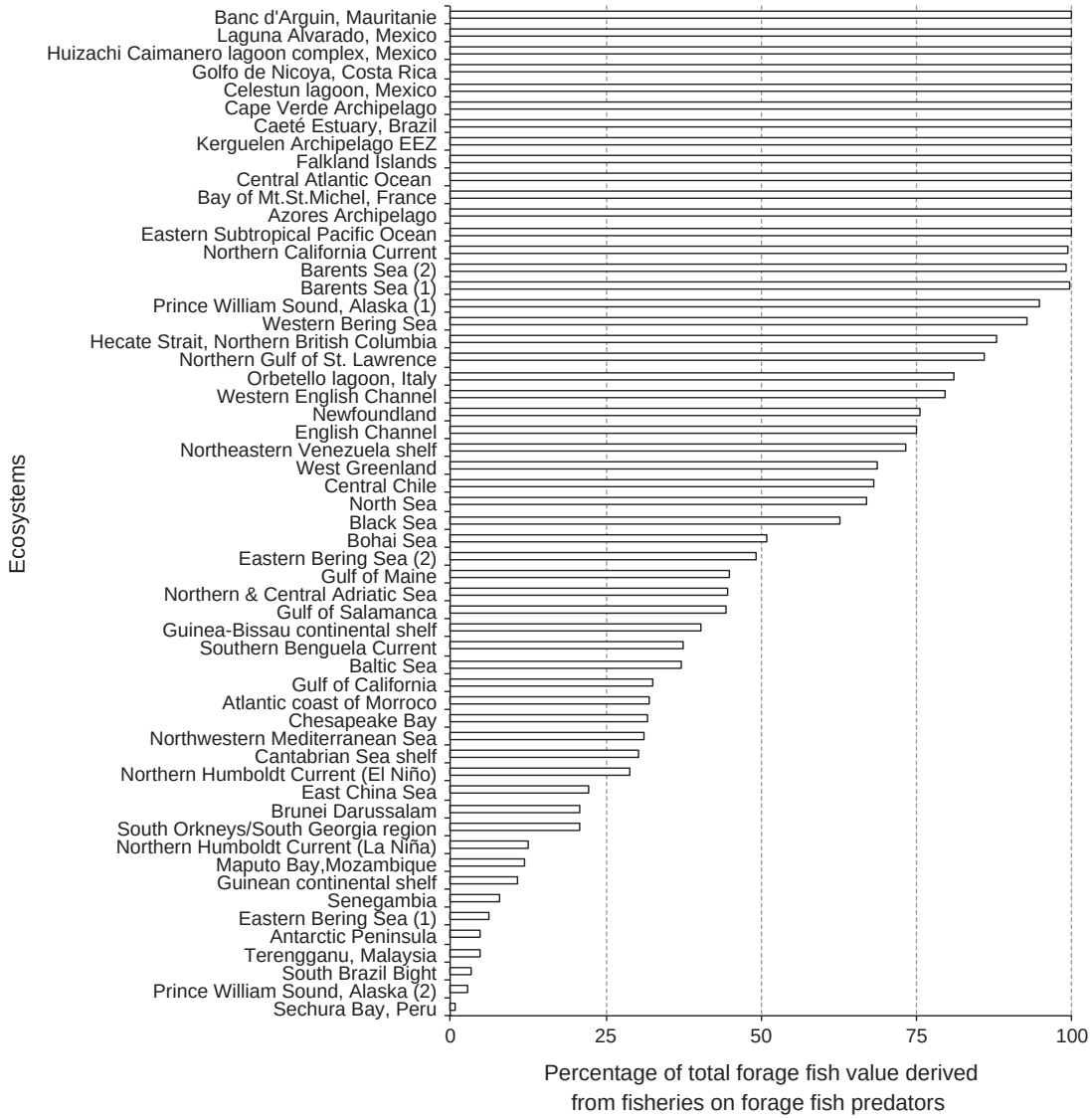

Figure 6 Percentage of total forage fish values (forage fish fisheries value + support service value to other fisheries) across Ecopath models (n = 56) derived from forage fish support service to other commercial fisheries. Ecosystems with 100% support service to other commercial fisheries do not have active forage fish fisheries in their respective ecosystem model.

tive, as we were not able to capture the importance of forage fish to these predators not included in the models. Likewise, aggregating predator species into model groups results in an averaged diet dependence on forage fish for the model group, which may mask high diet dependence for one or more individual species in that group. Averaging diet dependence for a single species over a large geographic area may also mask high diet dependencies that occur on smaller spatial or temporal scales. Validating every model to determine how well it represents its respective ecosystem and biological components was beyond the scope of this analysis, but Ecopath pedigree index information for a subset of models shows that the majority used in this analysis are of acceptable quality (Table 1). Using published models provided us with a large number of models covering the widest range of ecosystems and latitudes possible.

Here we used information on catches, catch values and food web connections to estimate the global contribution of forage fish to fisheries and

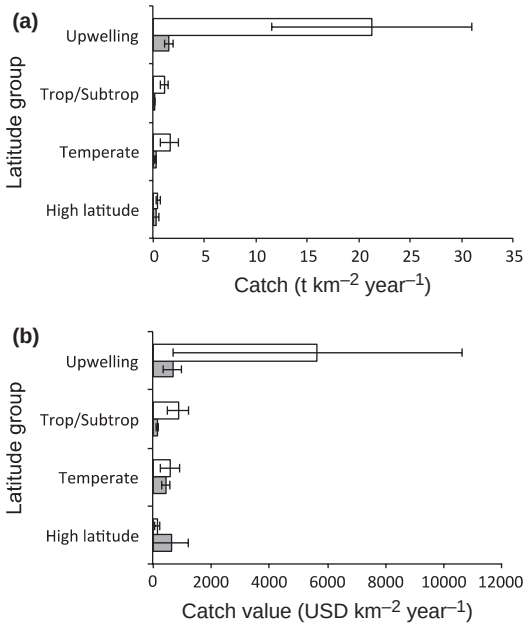

Figure 7 Mean catch (a) and mean catch value in 2006 USD (b) of forage fish (white bars) and mean supportive contribution of forage fish to other species' catch and catch value (grey bars), by latitude group. Bars indicate standard error. Upwelling ecosystem models were separated out to more clearly demonstrate latitudinal patterns.

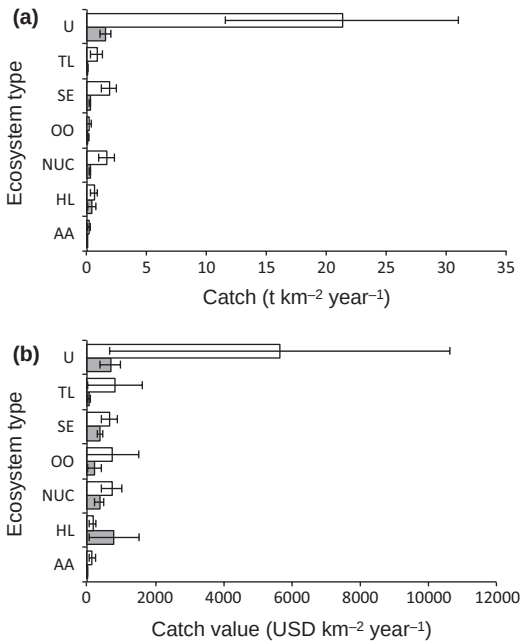

Figure 8 Mean catch (a) and catch value in 2006 USD (b) of forage fish (white bars) and mean supportive contribution of forage fish to other species' catch and catch value (grey bars). Bars indicate standard error. Ecosystem types: U, upwelling current; TL, tropical lagoon; SE, semi-enclosed; OO, Open ocean; NUC, non-upwelling coastal; HL, Arctic high latitude; and AA, Antarctic.

ecosystems. While we find that the importance of forage fish varies geographically, it is clear that these species are of critical importance to many predators, including humans. We consider our approach as a reliable and relatively quick way of assessing the importance of forage fish in marine ecosystems and fisheries around the world. Ecopath models in this analysis covered 33% of the total EEZs and HSAs and covered 47% of the IFA (Table S2, Appendix S2), which is where 97% of the global forage fisheries catch value is derived (Sumaila *et al.* 2007). We acknowledge that geographic coverage is limited in the Indian Ocean. Although EEZ and HSA areas in the Indian Ocean account for 20% of the total EEZ and HSA area, they represent <15% of the total fisheries catch value (excluding non-cephalopod or non-krill invertebrates) and <12% of the total forage fish catch value. Furthermore, Indian Ocean EEZ and HSA areas accounted for <10% of the total global supportive value of forage fish. More robust fisheries information from this data-poor region (De Young 2006) would benefit future analyses.

At the global scale the supportive value of forage fish to fisheries greatly exceeds their direct commodity value. We note that the estimated total ex-vessel value ($16.9 billion USD annually) is likely an underestimate, because it does not take into account the contribution of forage species to early life history stages of predators that are not yet of commercial catch size (e.g. juvenile cod, juvenile striped bass). We also have not included in our analysis the contributions of species that are considered forage fish only during juvenile life stages (e.g. Alaska pollock). Accounting for these types of forage species would increase our estimates of support to ecosystem predator production and marine fisheries in certain ecosystems. More importantly, the ex-vessel value of commercial fisheries is only one of many other indicators of the economic contributions of forage fish, and thus is clearly an underestimate of total economic worth. We have not accounted for the potential economic value of forage fish to recreational fisheries, to ecotourism [e.g. the whale watching industry is estimated at $2.5 billion 2009 USD

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annually (Cisneros-Montemayor *et al.* 2010)], as bait for fisheries, and to the provision of other ecosystem services such as water filtration.

Forage fish are integral to marine food webs as prey for a wide variety of higher trophic-level species. For many predators, forage fish constitute a substantial percentage of their diet, possibly making them vulnerable to reductions or fluctuations in forage fish biomass. We found that many extremely dependent predators were species listed on the IUCN Red List as 'Near Threatened', 'Vulnerable' or 'Endangered' (Table 2). These predators were commonly found in upwelling ecosystems, where empirical evidence shows that changes in forage fish abundance – caused by fishing, the environment, or a combination of both – negatively impact predator reproduction (Sunada *et al.* 1981; Becker and Beissinger 2006), breeding (Crawford and Dyer 1995; Cury *et al.* 2011), abundance (Crawford and Jahncke 1999; Jahncke *et al.* 2004), and carrying capacity (Crawford *et al.* 2007). This analysis has identified ecosystems that are likely to have highly to extremely dependent forage fish predators and may assist in ecosystem-based management efforts that consider both commercial fisheries and effects on threatened or endangered species.

We provide the first global estimates of the importance of forage fish as support for predators in marine ecosystems. Quantifying forage fish catch, support service to other commercially targeted predators, and support to all other ecosystem predators allows for identification of potential trade-offs that may occur among uses (Fig. 9). Competition for the use of forage fish biomass among ecological and fisheries interests can result in trade-offs, which can lead to conflicts in the management of forage fish. This is especially important, as forage fish are an increasingly valued commodity (Naylor *et al.* 2009; Tacon and Metian 2009) and provide fundamental ecological support to many other species. Taking a holistic viewpoint of their value is a step towards quantification of the overall contributions forage fish make to marine ecosystems and to the global economy. A challenge that remains for fisheries managers and policy makers is determining acceptable levels of catch that account for the roles forage fish play in the larger marine environment.

The management of trade-offs in marine ecosystems can often be challenging (Okey and Wright 2004; Cheung and Sumaila 2008; Salomon *et al.* 2011), but accounting for trade-offs is important and can lead to more sustainable levels of exploitation without compromising ecosystem integrity (Okey and Wright 2004). Ultimately, accounting for trade-offs between forage fish fisheries and conservation goals will require knowledge and understanding of the sensitivity to which commercially targeted and non-commercial predator species respond to fisheries induced changes in forage fish abundance. A combination of modelling (Okey and Wright 2004; Cheung and Sumaila 2008; Smith *et al.* 2011) and empirical (Read and Brownstein 2003; Brodziak *et al.* 2004) methods will likely be required to fully understand trade-offs in forage fishery management.

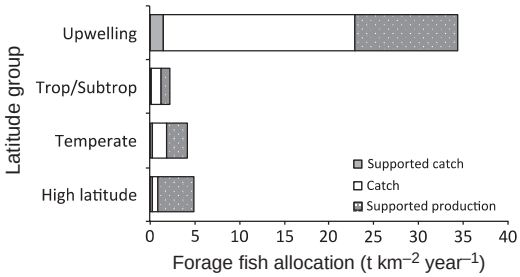

Figure 9 Forage fish allocation across latitude groups in terms of support service to fisheries (grey bars), forage fish catch (white bars) and support service to ecosystem predator production (dotted grey bars).

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References

Alder, J., Campbell, B., Karpouzi, V., Kaschner, K. and Pauly, D. (2008) Forage fish: from ecosystems to markets. *Annual Review of Environment and Resources* **33**, 153–166.

Anthony, J.A., Roby, D.D. and Turco, K.R. (2000) Lipid content and energy density of forage fishes from the northern Gulf of Alaska. *Journal of Experimental Marine Biology and Ecology* **248**, 53–78.

Bakun, A., Babcock, E.A., Lluch-Cota, S.E., Santora, C. and Salvadeo, C.J. (2010) Issues of ecosystem-based management of forage fisheries in "open" non-stationary ecosystems: the example of the sardine fishery in the Gulf of California. *Reviews in Fish Biology and Fisheries* **20**, 9–29.

Balmford, A., Bruner, A., Cooper, P. *et al.* (2002) Economic reasons for conserving wild nature. *Science* **297**, 950–953.

Barber, R.T. and Chavez, F.P. (1983) Biological consequences of El Niño. *Science* **222**, 1203–1210.

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C. and Silliman, B.R. (2011) The value of estuarine and coastal ecosystem services. *Ecological Monographs* **81**, 169–193.

Becker, B.H. and Beissinger, S.R. (2006) Centennial decline in the trophic level of an endangered seabird after fisheries decline. *Conservation Biology* **20**, 470–479.

Beddington, J.R., Agnew, D.J. and Clark, C.W. (2007) Current problems in the management of marine fisheries. *Science* **316**, 1713–1716.

Brodziak, J.K.T., Mace, P.M., Overholtz, W.J. and Rago, P.J. (2004) Ecosystem trade-offs in managing New England fisheries. *Bulletin of Marine Science* **74**, 529–548.

Butler, C.M., Rudershausen, P.J. and Buckel, J.A. (2010) Feeding ecology of Atlantic bluefin tuna (*Thunnus thynnus*) in North Carolina: diet, daily ration, and consumption of Atlantic menhaden (*Brevoortia tyrannus*). *Fishery Bulletin* **108**, 56–69.

Chavez, F.P., Ryan, J., Lluch-Cota, S.E. and Ñiquen, C.M. (2003) From anchovies to sardines and back: multidecadal change in the Pacific Ocean. *Science* **299**, 217–221.

Cheung, W.W.L. and Sumaila, U.R. (2008) Trade-offs between conservation and socio-economic objectives in managing a tropical marine ecosystem. *Ecological Economics* **66**, 193–210.

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

Christensen, V., Walters, C.J. and Pauly, D. (2005) *Ecopath with Ecosim: A User's Guide*. Fisheries Centre, University of British Columbia, Vancouver. November 2005 edition, 154 p. (available online at http://www.ecopath.org).

Cisneros-Montemayor, A.M., Sumaila, U.R., Kaschner, K. and Pauly, D. (2010) The global potential for whale watching. *Marine Policy* **34**, 1273–1278.

Coll, M., Libralato, S., Tudela, S., Palomera, I. and Pranovi, F. (2008) Ecosystem overfishing in the Ocean. *PLoS One* **3**, 1–10.

Costanza, R., d'Arge, R., de Groot, R. *et al.* (1997) The value of the world's ecosystem services and natural capital. *Nature* **387**, 253–260.

Crawford, R.J.M. and Dyer, B.M. (1995) Responses by 4 seabird species to a fluctuating availability of cape anchovy *Engraulis capensis* off South Africa. *Ibis* **137**, 329–339.

Crawford, R.J.M. and Jahncke, J. (1999) Comparison of trends in abundance of guano-producing seabirds in Peru and southern Africa. *South African Journal of Marine Science* **21**, 145–156.

Crawford, R.J.M., Underhill, L.G., Upfold, L. and Dyer, B. M. (2007) An altered carrying capacity of the Benguela upwelling ecosystem for African penguins (*Spheniscus demersus*). *ICES Journal of Marine Science* **64**, 570–576.

Cury, P., Bakun, A., Crawford, R.J.M. *et al.* (2000) Small pelagics in upwelling systems: patterns of interaction and structural changes in "wasp-waist" ecosystems. *ICES Journal of Marine Science* **57**, 603–618.

Cury, P., Shannon, L.J. and Shin, Y.-J. (2003) The functioning of marine ecosystems: a fisheries perspective. In: *Responsible Fisheries in the Marine Ecosystem* (eds M. Sinclair and G. Valdimarsson). CAB International, Wallingford, pp. 103–123.

Cury, P.M., Boyd, I.L., Bonhommeau, S. *et al.* (2011) Global seabird response to forage fish depletion-One-third for the birds. *Science* **334**, 1703–1706.

Daunt, F., Wanless, S., Greenstreet, S.P.R., Jensen, H., Hamer, K.C. and Harris, M.P. (2008) The impact of the sandeel fishery closure on seabird food consumption, distribution, and productivity in the northwestern North Sea. *Canadian Journal of Fisheries and Aquatic Sciences* **65**, 362–381.

De Young, C. (2006) *Review of the State of World Marine Capture Fisheries Management: Indian Ocean*. FAO Fisheries Technical Paper, Food and Agriculture Organization, Rome, p. 458.

FAO (2010) *The State of World Fisheries and Aquaculture 2010*. Food and Agriculture Organization, Rome, p. 197.

Francis, R.C., Hare, S.R., Hollowed, A.B. and Wooster, W.S. (1998) Effects of interdecadal climate variability on the oceanic ecosystems of the NE Pacific. *Fisheries Oceanography* **7**, 1–21.

Fréon, P., Cury, P., Shannon, L. and Roy, C. (2005) Sustainable exploitation of small pelagic fish stocks challenged by environmental and ecosystem changes: a review. *Bulletin of Marine Science* **76**, 385–462.

Fulton, E.A. (2010) Approaches to end-to-end ecosystem models. *Journal of Marine Systems* **81**, 171–183.

Furness, R. (2007) Responses of seabirds to depletion of food fish stocks. *Journal of Ornithology* **148**, 247–252.

Gislason, H. (2003) The effects of fishing on non-target species and ecosystem structure and function. In:

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Responsible Fisheries in the Marine Ecosystem (eds M. Sinclair and G. Valdimarsson). CAB International, Wallingford, pp. 255–275.

Gurevitch, J. and Hedges, L.V. (1999) Statistical issues in ecological meta-analyses. *Ecology* **80**, 1142–1149.

Hannesson, R. and Herrick Jr, S.F. (2010) The value of Pacific sardine as forage fish. *Marine Policy* **34**, 935–942.

Hannesson, R., Herrick, S. and Field, J. (2009) Ecological and economic considerations in the conservation and management of the Pacific sardine (*Sardinops sagax*). *Canadian Journal of Fisheries and Aquatic Sciences* **66**, 859–868.

Herrick, S.F., Norton, J.G., Hannesson, R., Sumaila, U.R., Ahmed, M. and Pena-Torres, J. (2009) Global production and economics. In: *Climate Change and Small Pelagic Fish* (eds D.M. Checkley, J. Alheit, Y. Oozeki and C. Roy). Cambridge University Press, Cambridge, pp. 256–274.

Hunsicker, M.E., Essington, T.E., Watson, R. and Sumaila, U.R. (2010) The contribution of cephalopods to global marine fisheries: can we have our squid and eat them too? *Fish and Fisheries* **11**, 421–438.

Jahncke, J., Checkley, D.M. and Hunt, G.L. (2004) Trends in carbon flux to seabirds in the Peruvian upwelling system: effects of wind and fisheries on population regulation. *Fisheries Oceanography* **13**, 208–223.

Kamimura, Y., Kasai, A. and Shoji, J. (2011) Production and prey source of juvenile black rockfish *Sebastes cheni* in a seagrass and macroalgal bed in the Seto Inland Sea, Japan: estimation of the economic value of a nursery. *Aquatic Ecology* **45**, 367–376.

Kelleher, K. (2005) *Discards in the World's Marine Fisheries*. An update. FAO Fisheries Technical Paper. No. 470. Food and Agriculture Organization of the United Nations (FAO), Rome, p. 131.

Logan, J.M., Rodriguez-Marin, E., Goni, N. *et al.* (2011) Diet of young Atlantic bluefin tuna (*Thunnus thynnus*) in eastern and western Atlantic foraging grounds. *Marine Biology* **158**, 73–85.

Magnussen, E. (2011) Food and feeding habits of cod (*Gadus morhua*) on the Faroe Bank. *ICES Journal of Marine Science* **68**, 1909–1917.

McLeod, K.L. and Leslie, H. (2009) *Ecosystem-based Management for the Oceans*. Island Press, Washington, DC, USA, p. 368.

McLeod, K.L., Lubchenco, J., Palumbi, S.R. and Rosenberg, A.A. (2005) Scientific Consensus Statement on Marine Ecosystem-Based Managment. Signed by 217 academic scientists and policy experts with relevant expertise and published by the Communication Partnership for Science and the Sea at http://compassonline.org/?q=EBM (last accessed on 13 November, 2011).

Morissette, L. (2007) *Complexity, cost and quality of ecosystem models and their impact on resilience: a comparitive analysis, with emphasis on marine mammals and the Gulf of St. Lawrence*. PhD thesis, University of British Columbia, Vancouver, BC, 260 pp.

Morissette, L., Hammill, M.O. and Savenkoff, C. (2006) The trophic role of marine mammals in the northern gulf of St. Lawrence. *Marine Mammal Science* **22**, 74–103.

Mullon, C., Mittaine, J.F., Thebaud, O., Peron, G., Merino, G. and Barange, M. (2009) Modeling the global fishmeal and fish oil markets. *Natural Resource Modeling* **22**, 564–609.

Murawski, S.A. (2000) Definitions of overfishing from an ecosystem perspective. *ICES Journal of Marine Science* **57**, 649–658.

Naylor, R.L., Hardy, R.W., Bureau, D.P. *et al.* (2009) Feeding aquaculture in an era of finite resources. *Proceedings of the National Academy of Sciences of the United States of America* **106**, 15103–15110.

Okey, T.A. and Wright, B.A. (2004) Toward ecosystem-based extraction policies for Prince William Sound, Alaska: integrating conflicting objectives and rebuilding pinnipeds. *Bulletin of Marine Science* **74**, 727–747.

Pauly, D., Trites, A.W., Capuli, E. and Christensen, V. (1998) Diet composition and trophic levels of marine mammals. *ICES Journal of Marine Science* **55**, 467–481.

Pikitch, E.K., Santora, C., Babcock, E.A. *et al.* (2004) Ecosystem-based fishery management. *Science* **305**, 346–347.

Polasky, S. and Segerson, K. (2009) Integrating ecology and economics in the study of ecosystem services: some lessons learned. *Annual Review of Resource Economics* **1**, 409–434.

Polovina, J.J. (1984) Model of a coral reef ecosystem: 1. The Ecopath model and its application to French Frigate Shoals. *Coral Reefs* **3**, 1–11.

Polovina, J.J., Howell, E., Kobayashi, D.R. and Seki, M.P. (2001) The transition zone chlorophyll front, a dynamic global feature defining migration and forage habitat for marine resources. *Progress in Oceanography* **49**, 469–483.

Read, A.J. and Brownstein, C.R. (2003) Considering other consumers: fisheries, predators, and Atlantic herring in the Gulf of Maine. *Conservation Ecology* **7**, 1–12.

Richerson, K., Levin, P.S. and Mangel, M. (2010) Accounting for indirect effects and non-commensurate values in ecosystem based fishery management (EBFM). *Marine Policy* **34**, 114–119.

Salomon, A.K., Gaichas, S.K., Jensen, O.P. *et al.* (2011) Bridging the divide between fisheries and marine conservation science. *Bulletin of Marine Science* **87**, 251–274.

Smith, A.D.M., Brown, C.J., Bulman, C.M. *et al.* (2011) Impacts of fishing low-trophic level species on marine ecosystems. *Science* **333**, 1147–1150.

Sumaila, U., Marsden, A., Watson, R. and Pauly, D. (2007) A global ex-vessel fish price database: construction and applications. *Journal of Bioeconomics* **9**, 39–51.

Sunada, J.S., Kelly, P.R., Yamashita, I.S. and Gress, F. (1981) The brown pelican as a sampling instrument of age group structure in the northern anchovy population. *California Cooperative Oceanic Fisheries Investigations Reports* **22**, 65–68.

Tacon, A.G.J. and Metian, M. (2009) Fishing for feed or fishing for food: increasing global competition for small pelagic forage fish. *Ambio* **38**, 294–302.

Thompson, P.M., McConnell, B.J., Tollit, D.J., Mackay, A., Hunter, C. and Racey, P.A. (1996) Comparative distribution, movements and diet of harbour and grey seals from the Moray Firth, NE Scotland. *Journal of Applied Ecology* **33**, 1572–1584.

Van Pelt, T.I., Piatt, J.F., Lance, B.K. and Roby, D.D. (1997) Proximate composition and energy density of some north pacific forage fishes. *Comparative Biochemistry and Physiology Part A: Physiology* **118**, 1393–1398.

Walter, J.F. and Austin, H.M. (2003) Diet composition of large striped bass (*Morone saxatilis*) in Chesapeake Bay. *Fishery Bulletin* **101**, 414–423.

Watson, R., Kitchingman, A., Gelchu, A. and Pauly, D. (2004) Mapping global fisheries: sharpening our focus. *Fish and Fisheries* **5**, 168–177.

Weise, M.J. and Harvey, J.T. (2008) Temporal variability in ocean climate and California sea lion diet and biomass consumption: implications for fisheries management. *Marine Ecology-Progress Series* **373**, 157–172.

Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Full references for Ecopath models used in this analysis in alphabetical order.

Appendix S2. Supplementary Tables (Tables S1 and S2).

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-3, into which the fluorescent proteins enhanced green fluorescent protein (EGFP) or mCherry were introduced to distinguish between the two cell lines. These cells were sparsely cultured to allow the formation of independent colonies. When their colony edges came into contact with one another, their boundaries were examined. Cells expressing identical nectin types did not intermingle at the border, whereas those expressing nectin-1 and -3 mutually invaded the counter colony, resulting in the formation of a mosaic pattern (Fig. 4, A to E, and fig. S6). We also performed time-lapse video microscopy using a coculture of MDCK cells expressing nectin-1 or -3 (N1- and N3-MDCK cells). In the supporting movie (movie S1 and Fig. 4E), one N1-MDCK cell (arrowhead) initially adhered to one of a pair of N3-MDCK cells (asterisks); subsequently, the former cell invaded the space between the two N3-MDCK cells. As a result, N1- and N3-MDCK cells were rearranged into a mosaic pattern. Similar behavior of cells was repeatedly observed in multiple experiments.

Thus, we propose that the heterophilic interactions between nectin-1 and -3 are critical for establishing the checkerboard-like pattern of hair cells and supporting cells. The molecular interaction between nectin-1 and -3 is the strongest of all possible combinations of the three nectins, which is likely to be responsible for the checkerboard-like assembly of these cells (Fig. 4F), as predicted by the mathematical model (*8*). The loss of nectin-3 removed such biased cell-cell adhesion, leading to cell rearrangement, including attachments between hair cells (Fig. 2D), as explained by the differential adhesiveness hypothesis (*18*). Nectin-1 KO mice displayed milder phenotypes. In these mice, the relatively strong interaction between nectin-3 and -2 probably retained the adhesion between hair cells and supporting cells; on the other hand, the adhesion between supporting cells should have been enhanced as a result of the redistribution of nectin-3 to these sites. These combinatory situations probably suppressed adhesion between hair cells (Fig. 4F). In nectin-2 KO mice, the heterophilic interactions between nectin-1 and -3 persisted; this explains the absence of a phenotype in these mice. In the absence of nectins, the cell junctions were not disrupted. This is most likely due to the coexpression of classic cadherins in the auditory epithelia. Hair cells and supporting cells are thought to be segregated through the process of lateral inhibition mediated by Notch-Delta signaling (*4*, *19*), and such processes themselves might contribute to the spatial separation of these cells (*20*–*22*). However, genetic inactivation of Notch signaling does not impair the checkerboard-like pattern, although it does result in an increase in the number of hair cells (*4*). This suggests that lateral inhibition is insufficient to create the checkerboard-like cellular pattern, stressing the importance of nectins in this patterning process. It is of note that heterophilic interactions between Hibris and Roughest, other members of the immunoglobulin superfamily, also contribute to the cell arrangement in the *Drosophila* eye (*23*, *24*), suggesting that similar mechanisms are conserved for cellular patterning across species.

References and Notes

1. H. I. Yamanaka, H. Honda, *Int. J. Dev. Biol.* **34**, 377 (1990).
2. R. L. Gulley, T. S. Reese, *J. Neurocytol.* **5**, 479 (1976).
3. E. McKenzie, A. Krupin, M. W. Kelley, *Dev. Dyn.* **229**, 802 (2004).
4. P. J. Lanford *et al.*, *Nat. Genet.* **21**, 289 (1999).
5. P. Chen, J. E. Johnson, H. Y. Zoghbi, N. Segil, *Development* **129**, 2495 (2002).
6. M. Montcouquiol *et al.*, *Nature* **423**, 173 (2003).
7. M. Montcouquiol, M. W. Kelley, *J. Neurosci.* **23**, 9469 (2003).
8. H. Honda, H. Yamanaka, G. Eguchi, *J. Embryol. Exp. Morphol.* **98**, 1 (1986).
9. M. Takeichi, *Science* **251**, 1451 (1991).
10. B. M. Gumbiner, *Nat. Rev. Mol. Cell Biol.* **6**, 622 (2005).
11. Y. Takai, H. Nakanishi, *J. Cell Sci.* **116**, 17 (2003).
12. Y. Takai, W. Ikeda, H. Ogita, Y. Rikitake, *Annu. Rev. Cell Dev. Biol.* **24**, 309 (2008).
13. Y. Takai, J. Miyoshi, W. Ikeda, H. Ogita, *Nat. Rev. Mol. Cell Biol.* **9**, 603 (2008).
14. K. Satoh-Horikawa *et al.*, *J. Biol. Chem.* **275**, 10291 (2000).
15. H. Togashi *et al.*, *J. Cell Biol.* **174**, 141 (2006).
16. N. Okabe, K. Ozaki-Kuroda, H. Nakanishi, K. Shimizu, Y. Takai, *Dev. Dyn.* **230**, 174 (2004).
17. F. D. Nunes *et al.*, *J. Cell Sci.* **119**, 4819 (2006).
18. M. S. Steinberg, *Science* **141**, 401 (1963).
19. M. W. Kelley, D. R. Talreja, J. T. Corwin, *J. Neurosci.* **15**, 3013 (1995).
20. J. Kimble, P. Simpson, *Annu. Rev. Cell Dev. Biol.* **13**, 333 (1997).
21. R. Kopan, R. Cagan, *Trends Genet.* **13**, 465 (1997).
22. J. Lewis, *Curr. Opin. Neurobiol.* **6**, 3 (1996).
23. S. Bao, R. Cagan, *Dev. Cell* **8**, 925 (2005).
24. S. Bao, K. F. Fischbach, V. Corbin, R. L. Cagan, *Dev. Biol.* **344**, 948 (2010).

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Supporting Online Material
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Materials and Methods
Figs. S1 to S6
References (*25*–*27*)
Movie S1

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Impacts of Fishing Low–Trophic Level Species on Marine Ecosystems

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Low–trophic level species account for more than 30% of global fisheries production and contribute substantially to global food security. We used a range of ecosystem models to explore the effects of fishing low–trophic level species on marine ecosystems, including marine mammals and seabirds, and on other commercially important species. In five well-studied ecosystems, we found that fishing these species at conventional maximum sustainable yield (MSY) levels can have large impacts on other parts of the ecosystem, particularly when they constitute a high proportion of the biomass in the ecosystem or are highly connected in the food web. Halving exploitation rates would result in much lower impacts on marine ecosystems while still achieving 80% of MSY.

Concerns about the trophic impact of harvesting marine species were recognized more than three decades ago (*1*). Despite recent successes in reducing exploitation rates in some marine ecosystems (*2*), concerns remain over the effects of fishing on the structure and function of marine ecosystems (*3*, *4*).

Low–trophic level (LTL) species in marine ecosystems comprise species that are generally plankton feeders for the larger part of their life cycle. They are often present in high abundance and tend to form dense schools or aggregations. They include small pelagic "forage" fish such as anchovy, sardine, herring, mackerel, and capelin but also invertebrate species such as krill. Humans harvest across the trophic levels in marine food webs, and landings of LTL species have been increasing generally in proportion with global catches (*5*). Forage fish account for over 30% of global fish landings, most of which is

now used for fishmeal production as feed for livestock industries and aquaculture rather than being consumed directly (*6*). However, LTL species also contribute directly to food security in many developing countries, and between 10 and 20% of global landings are consumed directly by humans (*7*). One species alone, Peruvian anchovy, contributes up to 50% of global landings used for fishmeal production. Driven by global markets for fertilizer, animal feed, and increases in the production of seafood from aquaculture, demand for fishmeal continues to increase (*8*).

LTL species play an important role in marine food webs because they are the principal means of transferring production from plankton to larger predatory fish and to marine mammals and seabirds. Several studies have raised concerns about the impacts on seabirds of local depletion of forage fish [anchovy in Perú (*9*), sand eels in the North Sea (*10*), and anchovy and sardines in South Africa (*11*)]. Similar concerns have been raised about the prospects of a large increase in catch of krill in the Southern Ocean and its potential impact on recovery of depleted marine mammals such as whales (*12*). Of particular concern are "wasp waist" systems, where a large part of the plankton production is funnelled through a small number of LTL species to higher trophic levels (*13*, *14*).

Although studies in individual ecosystems have raised concerns about the ecological effects of fishing LTL species, there has been no systematic attempt to examine and summarize what these broader effects might be or under what circumstances various effects might be expected to arise. In this study, we used ecosystem models in five well-studied regions to examine systemic effects of fishing LTL species. The regions include three eastern boundary current ecosystems—the northern Humboldt, the southern Benguela, and the California current—and two systems less dominated by upwelling, including the North Sea and the southeast Australian shelf and continental slope (Fig. 1). To avoid conclusions being dominated by structural assumptions in particular types of model, we used three different ecosystem models to explore the responses: Ecopath with EcoSim (EwE) (*15*, *16*), OSMOSE (*17*, *18*), and Atlantis (*19*, *20*). For each ecosystem and model, we selected up to five LTL species or groups and subjected them one by one to a range of fishing pressures, resulting in depletion levels relative to unfished biomass from zero (no fishing) to 100% (extirpated). The LTL species selected included some that are currently fished (such as anchovy) and others that are not currently exploited in those ecosystems (such as krill and mesopelagic fishes). We did not include harvested shellfish such as scallops and prawns, notwithstanding their commercial importance (*21*), because most of the models did not resolve these species well. Impacts on other ecological groups in the ecosystem were measured relative to biomass levels of those groups produced by simulations in which the focal LTL species was unfished, and all other groups were fished at current levels. Details of the ecosystems, models, groups, and experiments are provided in (*22*).

We found widespread impacts of harvesting LTL species across the ecosystems and LTL species selected (Fig. 2). The percent of ecological groups exhibiting effects greater than 40% increased with the level of depletion of the LTL species, but the extent of impact also varied across LTL species. Impacts on other ecological groups were both positive and negative (fig. S1), ranging up to very severe impacts for some groups (>60% change in biomass) even at relatively low levels of depletion (25% below unfished levels—that is, biomass reduced to 75% of unfished levels) of the LTL species. Negative impacts (reductions in abundance) tended to predominate for marine mammals and seabirds, although the majority of impacts on such groups were small. Some commercial species could also be negatively affected, although again impacts on most commercial species were small. Results

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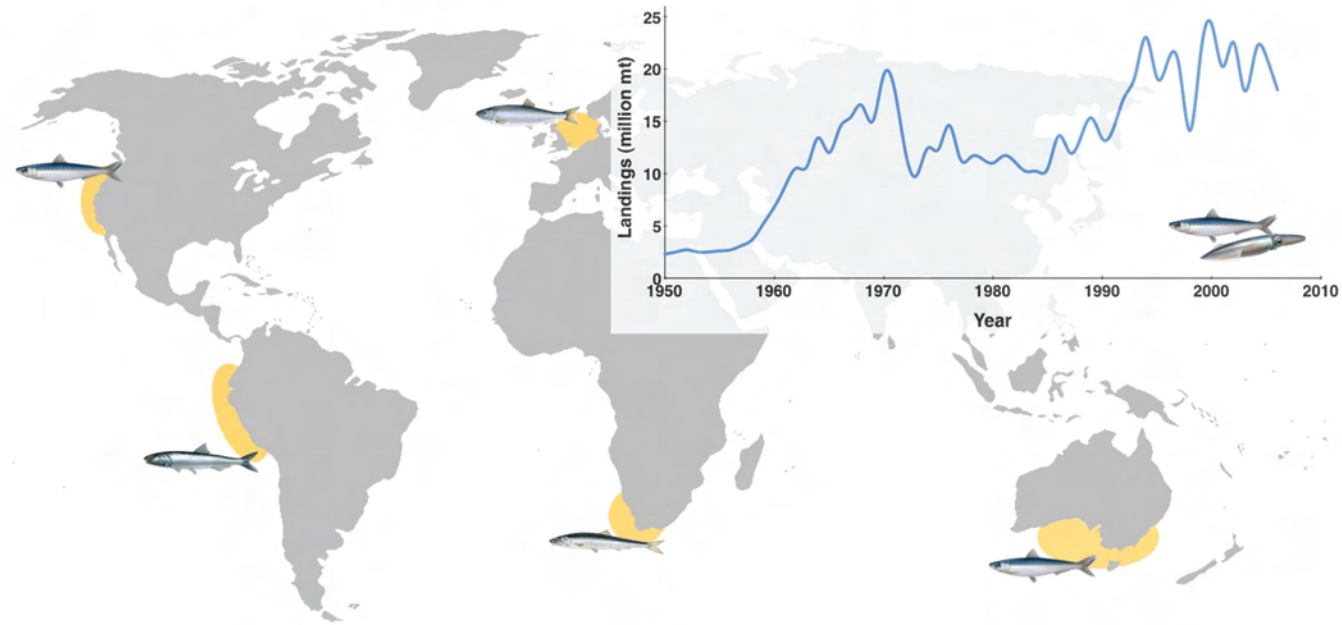

Fig. 1. Global map showing location of study ecosystems. From left to right are the California current, northern Humboldt, North Sea, southern Benguela, and southeast Australia. Graph shows trend in landings of forage species from 1950 to 2009. [Source: Sea Around Us Project, www.seaaroundus.org/global/1/3.aspx. Images of forage fish are copyright Casson Trenor, 2010, at www.sustainablesushi.net]

were generally robust to the three types of model used in the analysis (fig. S2).

The variation in impact of harvesting different LTL species has potentially important management implications; large impacts may require a change in harvest levels, whereas LTL species with small impacts could be harvested at conventional single-species levels. In each ecosystem, harvesting several of the LTL species was found to have high impacts, although the species with high impacts were not always consistent across ecosystems (Fig. 2). For example, in the northern Humboldt ecosystem, harvesting anchovy had high impacts, and harvesting sardine had low impacts, but in the southern Benguela ecosystem, harvesting sardines had the larger impact, whereas the impacts of fishing both species were low in the southeast Australian and California current ecosystems. Impacts of harvesting mesopelagic fishes (a group not currently targeted in any of these ecosystems or generally in global fisheries) were consistently high across ecosystems, and impacts of harvesting krill (large zooplankton), also not currently exploited in these ecosystems, also tended to be medium to high. Fishing sand eels had the highest impact in the North Sea.

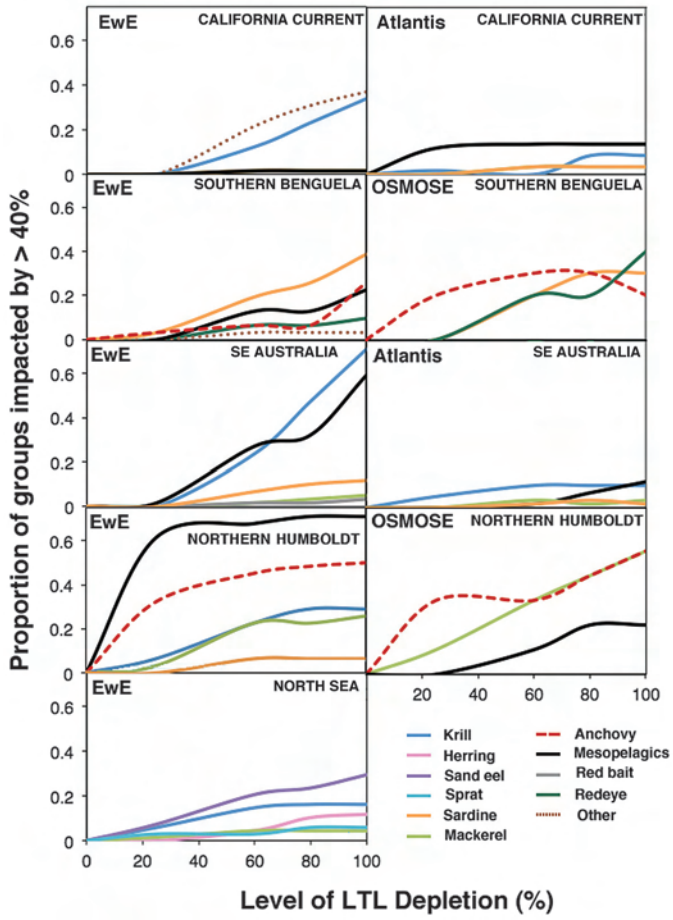

Fig. 2. Effects of level of depletion of LTL species on the proportion of other trophic groups whose biomass varied by more than 40% relative to their level where the LTL species was not fished. Results are shown for a variety of LTL species fished in each modeled ecosystem.

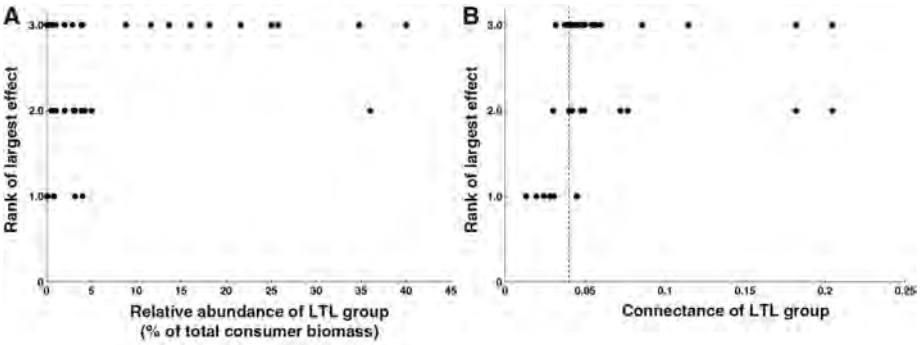

Fig. 3. Relationships between attributes of depleted LTL species and their ecosystem impact. Impacts are scored as the rank of the largest effect: rank 1, no change greater than 20% in any other ecological group; rank 2, no change greater than 60% in any other ecological group; and rank 3, change greater than 60% in at least one other ecological group. Each point corresponds to one ecosystem, model, and LTL species. All LTL species are depleted by 60%. (**A**) Impact of relative biomass of LTL species (biomass as a percent of total consumer biomass in the ecosystem) on rank of largest effect. (**B**) Impact of connectance (proportion of all ecosystem trophic links involving the LTL species) on rank of largest effect.

To explain this range of impacts across LTL species, we looked for more generic properties of these groups (other than taxonomy) that might explain and predict the variation. Three potential predictors were the relative abundance of the group in the ecosystem (for example, Peruvian anchovy accounts for up to 35% of the consumer biomass in the northern Humboldt ecosystem), the trophic level of the group, and the connectivity of the group in the food web. Trophic level was not a good predictor of impact, but the other two factors appear to be important. Abundant groups have consistently large impacts, whereas smaller groups can have either small or large impacts (Fig. 3A). There appears to be a threshold effect for connectance (the proportion of total trophic connections in the food web for each LTL species), with species that have a connectance value greater than ~0.04 having larger impacts (Fig. 3B). However, factors other than total connectance are likely to be important, including the presence of groups with trophic niches similar to those of the exploited species that can dampen the ecosystem effects of depleting the targeted species.

There are important tradeoffs to examine in considering the wider implications of these results for exploitation of LTL species. In particular, impacts on other parts of the ecosystem will be smaller at lower exploitation rates, but yields also will be lower (Fig. 4). There is a tension here between achieving broader goals of protecting and maintaining biodiversity (including ecosystem structure and function) and global food security. LTL species support the latter both through direct human consumption and through providing feed for livestock and aquaculture production. Considerable reductions in impact can be achieved by moving from exploitation at MSY levels (achieved at close to 60% depletion levels) to a target of 75% of unexploited biomass (25% depletion) for an LTL species, as shown in Fig. 4. The cost of such a change would be slightly less than 20% of long-term yield. This target could be achieved at significantly lower exploitation rates (mostly less than half MSY rates) (fig. S3), which would imply much lower fishing effort and may be closer to long-term economic optimum levels. There could also be some benefit of a reduction in harvest rate of LTL species to yields for other

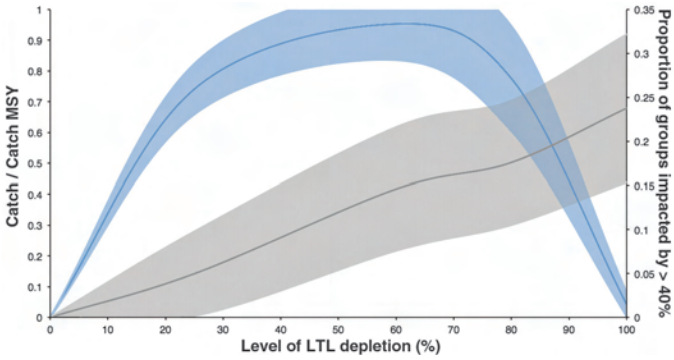

Fig. 4. Tradeoff between yield and ecological impact as level of LTL depletion varies. Yield (blue) is shown as a proportion of MSY. Ecological impact (gray) is measured as the proportion of other ecological groups whose biomass varied by more than 40%. Shaded zones show ±1.96 times SE. Results are for all ecosystems, models, and LTL species.

commercially targeted species (fig. S1). Although we did not explicitly examine multi-species harvest strategies, exploitation rates well below MSY levels are consistent with previous findings that lower exploitation rates should be adopted for most species (*2*).

These results are based on model predictions. Each of the models has been validated against time-series data from well-studied systems, and additional empirical validation for impacts on seabirds and marine mammals is provided in (*22*). Clearly, the details of which groups respond to depletion of LTL species is sensitive to both model parameterization and to choice of model structure (*22*). For this reason, we do not consider that these models should be used to determine tactical management decisions. However, the overall findings reported here are robust to details of model choice.

The conclusion that lower exploitation rates are needed for forage species also finds support from a wider set of model types (*23*). Spatial structure in marine ecosystems is an important factor in species interactions, and local prey depletion may be particularly important for land-based predators such as penguins and seals (*24*). Two of the models used in this study (OSMOSE and Atlantis) incorporate spatial structure, but not always at the resolution needed to address such issues. All of the models incorporate environmental forcing and variability, which is also an important feature driving the dynamics of many LTL species (*25*, *26*). The finding that connectance influences which species are likely to have larger impacts is potentially important, but although the measure of connectance is easily derived in models, it may be more difficult to determine empirically (and the empirical validity of the indicator would need verification). Previous studies have shown that the ways in which species are connected in the food web can influence system properties (*27*, *28*). Previous studies have also emphasized that additional protection may be needed for forage species (*29*).

The exploitation patterns examined in this study have involved constant fishing mortality rates. Initial explorations of other forms of exploitation, including use of biomass thresholds or "set asides" (biomass levels below which no exploitation will occur), suggest that lower ecological impacts could be achieved for similar long-term average yields, but at the cost of higher year-to-year variation in catches. Use of such set asides is already a feature of some LTL fisheries, including a 5-million-ton-minimum spawning stock biomass level for Peruvian anchovy (*30*) and 150,000 tons for California sardine (*31*). Closed areas are also used in some fisheries so as to reduce impacts on predators, such as closures for sand eels in some parts of the North Sea to improve the breeding success of sea birds (*32*).

Although harvest strategies for LTL species vary widely, many stocks are currently fished at levels below the biomass that achieves MSY (*22*). The results of this study combined with set asides and targeted spatial closures should help inform harvest strategies that achieve ecological objectives while ensuring ongoing substantial yields from LTL groups in support of the ongoing challenge of feeding the global human population (*33*).

References and Notes

1. R. M. May, J. R. Beddington, C. W. Clark, S. J. Holt, R. M. Laws, *Science* **205**, 267 (1979).
2. B. Worm *et al.*, *Science* **325**, 578 (2009).
3. D. G. Ainley, L. K. Blight, *Fish Fish.* **10**, 13 (2009).
4. M. Llope *et al.*, *Glob. Change Biol.*, published online 14 October 2010 (10.1111/j.1365-2486.2010.02331.x).
5. T. A. Branch *et al.*, *Nature* **468**, 431 (2010).
6. J. Alder, B. Campbell, V. Karpouzi, K. Kaschner, D. Pauly, *Annu. Rev. Environ. Resour.* **33**, 153 (2008).
7. A. G. J. Tacon, M. Metian, *Ambio* **38**, 294 (2009).
8. G. Merino, M. Barange, C. Mullon, *J. Mar. Syst.* **81**, 196 (2010).
9. J. Jahncke, D. M. Checkley, G. L. Hunt, *Fish. Oceanogr.* **13**, 208 (2004).
10. M. Frederiksen *et al.*, *Mar. Ecol. Prog. Ser.* **300**, 201 (2005).
11. R. J. M. Crawford, L. G. Underhill, L. Upfold, B. M. Dyer, *ICES J. Mar. Sci.* **64**, 570 (2007).
12. A. J. Constable *et al.*, *ICES J. Mar. Sci.* **57**, 778 (2000).
13. P. Cury *et al.*, *ICES J. Mar. Sci.* **57**, 603 (2000).
14. L. J. Shannon, P. M. Cury, A. Jarre, *ICES J. Mar. Sci.* **57**, 720 (2000).
15. C. Walters, D. Pauly, V. Christensen, J. F. Kitchell, *Ecosystems (N. Y.)* **3**, 70 (2000).
16. V. Christensen, C. J. Walters, *Ecol. Modell.* **172**, 109 (2004).
17. Y.-J. Shin, P. Cury, *Can. J. Fish. Aquat. Sci.* **61**, 414 (2004).
18. M. Travers *et al.*, *Ecol. Modell.* **220**, 3089 (2009).
19. E. A. Fulton *et al.*, *Ecol. Modell.* **173**, 371 (2004).
20. E. A. Fulton, A. D. M. Smith, A. E. Punt, *ICES J. Mar. Sci.* **62**, 540 (2005).
21. S. C. Anderson, J. M. Flemming, R. Watson, H. K. Lotze, *PLoS ONE* **6**, e14735 (2011).
22. Materials and methods are available as supporting material on *Science* Online.
23. M. C. Tyrrell, J. S. Link, H. Moustahfid, *Fish. Res.* **108**, 1 (2011).
24. J. A. Santora, C. S. Reiss, A. M. Cossio, R. R. Veit, *Fish. Oceanogr.* **18**, 20 (2009).
25. R. T. Barber, F. P. Chavez, *Science* **222**, 1203 (1983).
26. F. P. Chavez, J. Ryan, S. E. Lluch-Cota, M. Niquen C, *Science* **299**, 217 (2003).
27. T. Gross, L. Rudolf, S. A. Levin, U. Dieckmann, *Science* **325**, 747 (2009).
28. J. M. Montoya, G. Woodward, M. C. Emmerson, R. V. Solé, *Ecology* **90**, 2426 (2009).
29. C. J. Walters, V. Christensen, S. J. Martell, J. F. Kitchell, *ICES J. Mar. Sci.* **62**, 558 (2005).
30. P. Freon, M. Bouchon, C. Mullon, C. García, M. Ñiquen, *Prog. Oceanogr.* **79**, 401 (2008).
31. K. T. Hill *et al.*, Assessment of the Pacific Sardine Resource in 2008, for U.S. Management in 2009 (2009); available at www.pcouncil.org/wp-content/uploads/2009_CPS_SAFE_APP1_Sardine.pdf.
32. S. P. R. Greenstreet *et al.*, *ICES J. Mar. Sci.* **63**, 1530 (2006).
33. H. C. J. Godfray *et al.*, *Science* **327**, 812 (2010).

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Supporting Online Material

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Misspent youth: does catching immature fish affect fisheries sustainability?

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The "spawn-at-least-once" principle suggests that sustainability is secured if fish become vulnerable to commercial gears only after they have spawned. However, some studies suggest that protecting immature fish is not essential to sustainability because extrinsic factors determine both recruitment and stock status. A meta-analysis was conducted to quantify the independent effects of exploitation pattern and exploitation rate on current stock status. The analysis used empirical data for 38 fish stocks of 13 species in the NE Atlantic. Two metrics of exploitation pattern were used and their sensitivity was compared. As expected, exploitation rate had a significant negative effect on current stock status. Exploitation patterns associated with high proportional fishing mortality of immature fish also had a significant negative effect on current stock status, providing empirical support for the "spawn-at-least-once" principle. When the fishing mortality of immature fish exceeds half that of mature fish, stock status falls below precautionary limits. Our results suggest that a sensitive metric of exploitation pattern could provide useful information about an aspect of exploitation that is currently overlooked by fisheries management regimes that focus primarily on exploitation rate.

Keywords: exploitation pattern, exploitation rate, fisheries management, ICES, meta-analysis, spawn-at-least-once, sustainability.

Introduction

Fisheries science distinguishes between two distinct aspects of exploitation of commercial species: exploitation rate (fishing intensity) and exploitation pattern (fishing selection). Exploitation rate is defined as the proportion of the population or biomass of a fish stock that is removed each year (FAO, 2010) and is usually quantified by fishing mortality (F; ICES, 2008). The precautionary approach to fisheries in Europe and elsewhere has led to the adoption of reference points for F and spawning-stock biomass (SSB; Wakeford *et al.*, 2007; ICES, 2008), and the main focus of fisheries management has been regulating F (Froese *et al.*, 2008; ICES, 2008). Exploitation pattern is defined as the distribution of fishing mortality over the age or length composition of the fish population (FAO, 2010), and it depends on the selectivity of the gears used in a fishery and on the extent to which particular age/size classes are targeted. Both factors are associated with design specifications of the fishing gear, fish behaviour, the temporal and spatial distribution of fish, and skipper expertise (Wileman *et al.*, 1996). Exploitation pattern also represents the proportion of immature fish caught, because size in fish reflects their maturity stage. This relationship between size and maturity stage can be impacted by directional shifts in the maturation rate, for instance if, over time, fish tend to mature smaller (de Roos *et al.*, 2006).

The need to protect immature fish to enhance fishery yields is an intuitive concept pre-dating fisheries science itself (Halliday and Pinhorn, 2002; Kennelly and Broadhurst, 2002). This traditional approach assumes that exploitation pattern plays an important role in fisheries management and that a key element of good management is the use of fishing gears that retain large fish in the catch while allowing juveniles to escape (Armstrong *et al.*, 1990). The "spawn-at-least-once" principle presumes a significant correlation between the size of the spawning stock and subsequent recruitment (Myers and Barrowman, 1996; Myers and Mertz, 1998), so allowing fish to survive long enough to spawn is assumed to prevent recruitment-overfishing (Myers and Mertz, 1998; Halliday and Pinhorn, 2002). There are many studies focusing on specific stocks of pelagic and demersal species throughout the world that suggest that implementation of the "spawn-at-least-once" principle through the adoption of suitable exploitation patterns promotes sustainability and profitability of fisheries (e.g. Mori *et al.*, 2001; Enberg, 2005; Klanjšček and Legović, 2007; Ishida *et al.*, 2009). Additionally, the protection of immature fish, which are typically small, could indirectly reduce the potential for growth-overfishing. The yield-per-recruit theory (Beverton and Holt, 1957) indicates that for a given stock there is an optimal size at first capture (L_{opt}) for a given level of F that can maximize fisheries yield, this optimal size being usually larger than the size at which 50% of the individuals have reached maturity (L_{50}; Froese *et al.*, 2008).

The validity of the "spawn-at-least-once" principle is, however, open to challenge. The very existence of a connection between SSB and subsequent recruitment has been disputed (Gilbert, 1997), and other factors such as environmental variability (Jarre-Teichmann *et al.*, 2000), physiological condition (Marshall

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et al., 1999), and the age range of the spawning population (Marteinsdottir and Thorarinsson, 1998; Cardinale and Arrhenius, 2000) have been shown to play an important role in determining levels of recruitment. In addition, protecting older and more experienced females (megaspawners) that tend to produce more, larger, and qualitatively superior eggs (Marteinsdottir and Steinarsson, 1998; Cardinale and Arrhenius, 2000; Trippel and Neil, 2004) has been suggested as a more effective alternative management strategy for promoting sustainability and profitability (Caddy and Seijo, 2002; Palakovich Carr and Kaufman, 2009). Moreover, doubts have been raised about the likelihood of obtaining the predicted increased yields by increasing the size at first capture, because of uncertainty in the calculation of the fish population parameters implemented in yield-per-recruit analyses (Haddon, 2001; Halliday and Pinhorn, 2002). Finally, fishing populations more evenly instead of selectively catching large fish has also been proposed as a means of reducing the impact on stocks and ecosystems (Zhou *et al.*, 2010). All these perspectives support the perception that exploitation patterns leading to the capture of fewer immature fish have no effect on stock status and that it is exploitation rate and/or protection of other demographic components of the fish population that matter.

These two competing hypotheses, i.e. whether or not immature fish should be protected, essentially describe two different approaches to fisheries management. Froese (2004) synthesized elements of both approaches to suggest that fisheries sustainability can be achieved by fishing at L_{opt} and that the "spawn-at-least-once" principle should be implemented in conjunction with the protection of megaspawners. Therefore, he proposed that both the percentage of mature fish and the percentage of megaspawners in the catch should be used as simple indicators of sustainability. That approach requires fishing mortality to vary with age/size class and hence inherently implies that exploitation pattern does indeed play a role in sustainability.

To date, therefore, the debate about the potential benefits from protecting immature fish remains unresolved. To test empirical support for the "spawn-at-least-once" principle, a metric of exploitation pattern that characterizes the relative exploitation of immature to mature fish is needed. Only a few such metrics have been suggested in the literature. Froese (2004) proposed the percentage of mature fish in the catch as a straightforward and simple proxy for exploitation pattern that could be also used as an indicator of fisheries sustainability. Petrakis and Stergiou (1997) used the divergence between the length where 50% of the fish are retained (L_{r50}) and L_{50} as a proxy for exploitation pattern and to assess the stock sustainability associated with different fishing gears. This divergence can be calculated easily for a single type of gear applied in a specific fishery, but its calculation would be more complex in multinational, multigear fisheries, as for NE Atlantic stocks. In this study, exploitation pattern is quantified as the proportion of fishing mortality of immature fish to that of mature fish. This metric has the advantage of taking into account the whole fish population and being easy to calculate for many assessed stocks.

The effect of historical values of exploitation pattern and exploitation rate on current stock status through a meta-analysis of 38 fish stocks in the NE Atlantic is investigated. The data used are widely available empirical data from ICES stock assessment reports. The meta-analysis approach was taken because general trends are more easily observed when all data are combined (Myers, 2001). Meta-analyses have already demonstrated the overall negative effect of exploitation rate on stock status (Sparholt *et al.*, 2007; Worm *et al.*, 2009). However, the overall effect of historical values of exploitation pattern on current stock status has not yet been assessed using empirical data for a wide range of species and stocks. As outlined above, an *F*-based metric of exploitation pattern is used, but the adequacy of the simpler catch-based proxy proposed by Froese (2004) is also explored. The ability of these metrics to capture the effect of exploitation pattern on current stock status and their potential to act as indicators of fisheries sustainability are also discussed.

Material and methods

Data from 38 stock assessment reports (13 fish species) conducted by nine ICES Working Groups were used in the analysis (Table 1). For each stock, the information available was year-specific values of catch-at-age ($C_{a,y}$), SSB (B_y), fishing mortality-at-age ($F_{a,y}$), abundance-at-age ($N_{a,y}$), maturity-at-age ($M_{a,y}$), and the stock-specific precautionary spawning biomass limits (B_{pa}) set by ICES.

As the available time-series were not of equal length for all stocks and different stocks exhibited different maturity rates, the interval used for the analysis was standardized according to the approximate number of generations represented rather than using a fixed, arbitrary number of years. For each stock, a time interval $T = 4 \times A_{50}$ years rounded to the nearest integer was used to calculate the average levels of exploitation rates and patterns, where A_{50} is the age at which 50% of the fish have matured (Table 1). The value of A_{50} was calculated for each stock by fitting a logistic model to maturity-at-age data provided by ICES. For the stocks where different maturity-at-age data were given for each year, the average values of the most recent 3 years were used. The time interval T was regarded as long enough to capture the effect of recent trends in exploitation pattern and exploitation rate on current stock status. The effect of using a different time interval was also investigated by repeating the analysis using a fixed 20-year interval for all stocks (except for the Rockall haddock stock, because just 18 years were available).

Model variables

The variables used in the models included current stock status as a response variable, and exploitation rate, exploitation pattern, and assessment effect (*AE*) as explanatory variables. Each of these is described in detail below.

The status of each stock in year y is defined by ICES as the value of B_y in relation to the precautionary level, B_{pa}, set by the Working Group assessing the stock. In this analysis, the average value of B_y/B_{pa} over the final 3 years of each assessment was used to describe current stock status (*CSS*):

$$CSS = \frac{1}{3}\sum_{y=fy-2}^{fy}\frac{B_y}{B_{pa}}, \quad (1)$$

where *fy* is the final year of the assessment (2009 usually). In this context, $CSS > 1$ indicates a stock within its precautionary biomass limit, and $CSS < 1$ indicates a stock below that level.

In ICES assessment reports, the exploitation rate in year y is expressed by $\bar{F}_y$ (year^{-1}), i.e. the average fishing mortality of the age classes primarily targeted by the fisheries. The age classes used to estimate $\bar{F}$ vary across stocks (Table 1). In this analysis, mean exploitation rate (*ER*) was defined accordingly as the

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average value of $\bar{F}$ in the last T years of each stock assessment:

$$ER = \frac{1}{T}\sum_{y=fy-T+1}^{fy} \bar{F}_y. \quad (2)$$

To express exploitation pattern, two alternative metrics were used. The more complicated metric (EP_1) was an F-based quantity directly quantifying exploitation pattern as the proportion of fishing mortality of immature (F_{imm}) to that of mature fish (F_{mat}). In year y, $F_{\text{imm},y}$ and $F_{\text{mat},y}$ were calculated for each stock as

$$F_{\text{imm},y} = \frac{\sum_{a=1}^{n} N_{a,y} I_{a,y} F_{a,y}}{\sum_{a=1}^{n} N_{a,y} I_{a,y}} \quad (3)$$

$$F_{\text{mat},y} = \frac{\sum_{a=1}^{n} N_{a,y} M_{a,y} F_{a,y}}{\sum_{a=1}^{n} N_{a,y} M_{a,y}}. \quad (4)$$

where $a = 1, 2, 3, \ldots n$ are the age classes, and $I_{a,y} = 1 - M_{a,y}$ is the proportion of immature fish. EP_1 was then calculated for every stock using Equations (3) and (4):

$$EP_1 = \frac{1}{T}\sum_{y=fy-T+1}^{fy} \frac{F_{\text{imm},y}}{F_{\text{mat},y}}. \quad (5)$$

Fish belonging to age class 0 were excluded from the estimations of $F_{\text{imm},y}$ and $F_{\text{mat},y}$ because relevant data were not available for most stocks and fishing mortality in that age group was usually negligible (Table 1). If there were missing age classes in a stock's dataset (Table 1), they were reconstructed using the formula

$$N_{a,y} = N_{a+1,y+1} e^{M'_{a,y}} \quad (6)$$

where $M'_{a,y}$ is the natural mortality at age a in year y (Haddon, 2001). This is essentially a back-calculation, so missing abundance values ($N_{a,fy}$) in the last year of the time-series (terminal N values) were calculated as the average of the numbers-at-age of the three preceding years (Haddon, 2001).

The second metric used to approximate exploitation pattern (EP_2) was the simpler catch-based proxy proposed by Froese (2004), and it was expressed as the proportion of immature fish in the total catch:

$$EP_2 = \frac{1}{T}\sum_{y=fy-T+1}^{fy} \left(\frac{\sum_{a=1}^{n} C_{a,y} I_{a,y}}{\sum_{a=1}^{n} C_{a,y}}\right). \quad (7)$$

This metric is easy to calculate, but it takes into consideration only a subgroup of the total fish population, i.e. those caught. Its potential to act as an indicator of sustainability, as Froese (2004) proposed, has not been fully explored. When the analysis was conducted based on a 20-year time interval, T was replaced by 20 in Equations (2), (5), and (7) to obtain the mean values of the variables over a 20-year period.

The categorical variable *AE* was included to express the potential degree of underestimation of immature fish exploitation (F_{imm}) attributable to unaccounted discarding in every stock. Discard estimates were explicitly included in the respective ICES assessments for just 10 of the 38 stocks, whereas for 16 other stocks, discards were not included but reported to be negligible, so not affecting the estimates of the assessments (Table 1). Those 26 stock assessments were classified at level A (low bias owing to the effect of discarding on the assessment estimates). For the other 12 stocks, discards were not included in the assessments but were reported to be potentially high, causing underestimation of the levels of fishing mortality, especially at younger ages. Those 12 stock assessments were classified at level B (possible high bias owing to the unaccounted effect of discarding).

Statistical analysis

The relationship between EP_1 and EP_2 was investigated by plotting one against the other and examining their correlation and the outliers. The reasons that five stocks appeared to be outliers in this relationship were investigated in relation to the different characteristics of the two metrics.

The dependence of *CSS* on the explanatory variables *ER*, EP_1, EP_2, and *AE* was investigated using generalized linear models (GLMs). Data exploration before model fitting was carried out as described by Zuur *et al.* (2007); it involved checking for collinearities and investigating the properties of the data to set the type of GLMs to be used. All correlation coefficients and values of variance inflation factor (VIF) were low (VIF < 1.5) in terms of explanatory variables used within the same models, indicating the absence of collinearities. The response variable (CSS) comprised positive values with a positively skewed distribution, so a gamma distribution was chosen for the GLMs (Zuur *et al.*, 2009). *CSS* was also non-linearly related to the continuous explanatory variables, so a log-link function was implemented.

Candidate models (GLMs) were constructed including all possible combinations of explanatory variables (*ER*, EP_1 or EP_2, *AE*) plus the interaction term *ER:EP*. The interaction term was added as a result of some patterns being observed during the data-exploration process, as well as the fact that the possible existence of such an interaction has been suggested in the literature (Myers and Quinn, 2002).

Model selection and the estimation of model selection uncertainty were based on the information theoretic approach (Burnham and Anderson, 2002). In this context, the best models were selected based on Akaike's information criterion (AIC): AIC $= -2$(log-likelihood) $+ 2$(number of parameters) (Crawley, 2007). AIC is known as a "penalized log-likelihood", so when comparing multiple models, the smaller the AIC value, the better the model (Burnham and Anderson, 2002; Crawley, 2007). The differences in AIC between the model with the lowest AIC value ($\text{AIC}_{\min}$) and the rest of the candidate models, $\Delta i = \text{AIC}_i - \text{AIC}_{\min}$, were also computed (Burnham and Anderson, 2002; Katsanevakis, 2006). The model with the lowest AIC value and the model with the fewest parameters (i.e. the most parsimonious) within two units of the lowest AIC ($\Delta i < 2$) were identified and parameter estimates obtained (Burnham and Anderson, 2002; Katsanevakis, 2006). Model validation was carried out by checking for normality and patterns in the residuals and by plotting the residuals against the explanatory variables to check for patterns and homoscedasticity (Zuur *et al.*, 2007).

Finally, effect plots were constructed for the optimal model to visualize the separate effects of *ER* and EP_1 on *CSS* on a meta-scale when the other explanatory variables are fixed at typical (mean) values (Fox, 2003). This allows controlling for *ER* when exploring the impact of EP_1, and vice versa. All statistical analyses were carried out using R statistical software version 2.12.1 (R Development Core Team, 2010).

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Results

A positive correlation ($r = 0.57$, $p < 0.01$) was found between EP_2 and EP_1, as expected. However, there was a distinct group of stocks with high values of EP_2 for low values of EP_1 (Figure 1). These stocks (NE Arctic cod, NE Arctic haddock, and three stocks of saithe) displayed large proportions of immature fish in the catch but also large numbers of immature fish that were not captured by the fisheries. Moreover, all five stocks were within the precautionary limits ($CSS > 1$) despite 45–73% of the fish caught being immature (Table 1).

EP_1 and EP_2 were alternatively used to quantify exploitation pattern in the GLMs. The model with the lowest AIC value (m_1) included ER, EP_1, and AE, whereas the most parsimonious within two units of the lowest AIC (m_2) included just EP_1 and ER (Table 2). This suggests that both exploitation rate and exploitation pattern as described by the historical values of proportional fishing mortality of immature fish affect current stock status. All other models were either less parsimonious or had $\Delta i > 2$, so they were not investigated further. In particular, all models including EP_2 had $\Delta i > 4.9$ (Table 2), showing that EP_1 is better at explaining the variation in CSS and that it is a more sensitive metric of exploitation pattern than EP_2. The high AIC values of models including EP_2 suggest that different proportions of immature fish in the catch have no effect on CSS because they do not correspond to the proportion of immature fish removed from the population.

The comparison of models m_1 and m_2 demonstrates that inclusion of the factor AE in the model increased the deviance explained by ~4% (Table 2). Moreover, the model including only ER (m_5) explained 18.4% of the deviance more than the model including only EP_1 (m_{12}), showing the higher explanatory power of ER than EP_1.

Both ER and EP_1 had a negative effect on CSS in both m_1 and m_2 (Tables 3 and 4). In other words, higher historical values of exploitation rate and of relative fishing mortality of immature fish decreased current stock status. Level B of factor AE also had a negative effect on CSS in m_1, indicating overestimation of CSS values when there is high bias in the estimation of F_{imm} because of the unaccounted effect of discarding, but at a non-significant level (Table 3). The residuals of both models (m_1, m_2) were normally distributed, and no patterns were observed when the residuals were plotted against the fitted values or any of the explanatory variables. The effect plots for m_1 showed that the CSS fell below the precautionary limit ($CSS < 1$) when ER exceeded 0.63 (for mean values of EP_1 and AE; Figure 2a) and when EP_1 exceeded 0.50 (for mean values of ER and AE), i.e. when F_{imm} exceeded half of F_{mat} (Figure 2b).

Conducting the same analysis using a time interval of 20 rather than T years produced qualitatively similar results, suggesting that the effect of ER, EP_1, and EP_2 on CSS is not sensitive to the number of years over which the explanatory variables are averaged.

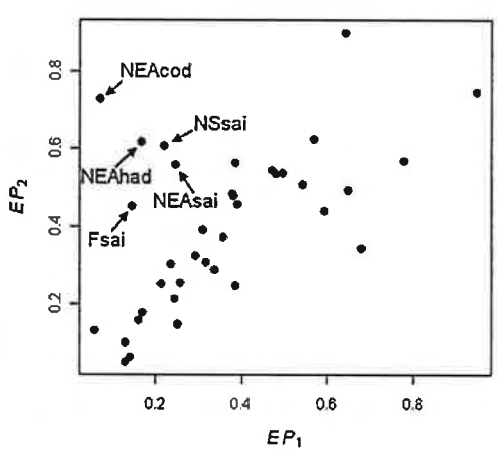

Figure 1. Plot of EP_2 against EP_1. The arrows indicate stocks with large proportions of immature fish in the catch (high values of EP_2) and small proportional fishing mortality of immature fish (low values of EP_1). NEAcod, NE Arctic cod stock; NEAhad, NE Arctic haddock stock; NEAsai, NE Arctic saithe stock; NSsai, North Sea saithe stock; Fsai, Faroe saithe stock.

Discussion

The meta-analysis presented here of empirical dataseries covering 38 stocks and 13 species of the NE Atlantic shows that stock status is negatively affected by (i) a high fishing mortality of immature fish compared with the fishing mortality of mature fish, and (ii) a high overall exploitation rate.

The first conclusion demonstrates that historical exploitation patterns that result in large proportional fishing mortality of immature fish lead to lower current stock status. This supports the "spawn-at-least-once" principle which suggests that recruitment-overfishing is avoided when relatively more immature fish are given the chance to contribute to recruitment (Myers and Mertz, 1998). The result could be attributed to some extent too to the reduced growth-overfishing attributable to the capture of relatively fewer small fish (Froese *et al.*, 2008). Values of $EP_1 > 0.50$ (F_{imm} more than half F_{mat}) appear to be related to the stock status falling below precautionary limits ($CSS < 1$; Figure 2b), and it is also worth noting that the nine stocks having F_{imm} values <20% of F_{mat} ($EP_1 < 0.2$) were well within precautionary limits ($CSS > 1$; Table 1). These results suggest that fishing gears with relatively sharp selection ranges and with L_{r50} values close to L_{50} seem to be preferable to those with flat selection ranges and $L_{r50} < L_{50}$.

The second conclusion reached here illustrates the unsurprisingly negative effect of exploitation rate on stock status that has already been suggested by earlier meta-analytical studies (Sparholt *et al.*, 2007; Worm *et al.*, 2009). The effect of exploitation rate on current stock status appears to be stronger than that of exploitation pattern; the model containing ER alone (m_5) has a lower AIC value and explains a greater percentage of the deviance than the one containing just EP_1 (m_{12}; Table 2). High exploitation rates can result in both growth- and recruitment-overfishing (Myers and Quinn, 2002; Worm *et al.*, 2009), so pushing stock status below precautionary limits.

The assessment effect attributable to unaccounted discarding does not appear to be very strong. Model m_1 was only slightly better than m_2 (Table 2), and the negative effect from the non-inclusion of high discards in the assessment was not statistically significant (Table 3). Nevertheless, the outcome of stock assessments has been reported to be highly sensitive to the inclusion of discards and to the raising procedures (ICES, 2007). It is therefore expected that assessments producing less biased estimates of F as a result of the inclusion of discards would lead to more accurate results for meta-analyses based on ICES data.

The F-based metric of exploitation pattern (EP_1) constructed within this study appears to be more sensitive and efficient than the catch-based proxy (EP_2) proposed by Froese (2004).

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Table 1. The dataset used in the analysis.

ICES Working Group	Species	Area	Report time-series	Time interval for *ER*, *EP*	Age classes	Age classes for *ER*	A_{50} (years)	*ER* (year^{-1})	EP_1	EP_2	*CSS*	*AE*	Comments on discarding	Other comments
NWWG	Cod	Faroe (Vb)	1961–2008	1997–2008	2–10	3–7	3.12	0.599	0.234	0.303	0.481	A	No discards mentioned	Age 1 reconstructed
WGBFAS	Cod	West Baltic (22–24)	1970–2008	1996–2008	1–7	3–6	3.14	1.069	0.495	0.538	1.152	A	Discards included in the assessment	–
WGNSSK	Cod	North Sea (IV, IIIaN, VIId)	1963–2008	1994–2008	1–7	2–4	3.73	0.866	0.642	0.898	0.298	A	Discards included in the assessment	–
AFWG	Cod	NE Arctic (I, II)	1946–2008	1983–2008	3–13	5–10	6.61	0.697	0.070	0.729	1.505	A	Low discard rates (less than 10%)	Ages 1, 2 reconstructed
WGCSE	Cod	West of Scotland (VIa)	1978–2008	2001–2008	1–7	2–5	2.11	0.962	0.778	0.570	0.243	A	Discards included in the assessment	$F = Z - 0.2$
WGCSE	Cod	Irish Sea (VIIa)	1968–2008	2001–2008	1–5	2–4	2.02	1.387	0.381	0.479	0.204	B	High. Underestimated *F* at age 1	–
WGSSDS	Cod	Celtic Sea (VIIe–k)	1971–2007	1999–2007	1–7	2–5	2.30	0.847	0.472	0.546	0.534	B	High. Up to 65% after 2003	No assessment conducted in 2009
WGNSSK	Haddock	North Sea (IV, IIIa)	1963–2008	1999–2008	0–8	2–4	2.62	0.411	0.542	0.508	1.706	A	Discards included in the assessment	Age 0 excluded ($F < 0.04$)
AFWG	Haddock	NE Arctic (I, II)	1950–2008	1986–2008	3–11	4–7	5.79	0.381	0.166	0.618	2.534	B	No data. Discarding might present a serious problem	Ages 1, 2 reconstructed
NWWG	Haddock	Faroe (Vb)	1957–2008	1997–2008	1–10	3–7	3.04	0.358	0.160	0.157	1.286	A	Minimal discarding	–
WGCSE	Haddock	West of Scotland (Via)	1978–2008	2001–2008	1–8	2–6	1.99	0.556	0.592	0.441	1.196	A	Discards included in the assessment	–
WGCSE	Haddock	Rockall (VIb)	1991–2008	1999–2008	1–7	2–5	2.50	0.518	0.390	0.459	2.070	A	Discards included in the assessment	–
NWWG	Saithe	Faroe (Vb)	1961–2008	1987–2008	3–12	4–8	5.54	0.458	0.145	0.452	1.109	A	Little discarding	Ages 1, 2 reconstructed
WGNSSK	Saithe	North Sea (IV, IIIaN, VI)	1963–2008	1990–2008	3–10	3–6	4.75	0.376	0.219	0.609	1.337	A	Assumed to be a small problem	Ages 1, 2 reconstructed
AFWG	Saithe	NE Arctic (I, II)	1960–2008	1988–2008	3–11	4–7	5.16	0.285	0.246	0.560	3.925	A	Not considered a problem	Ages 1, 2 reconstructed
WGCSE	Whiting	Celtic Sea (VIIe–k)	1982–2008	2004–2008	1–8	2–5	1.16	0.793	0.139	0.062	1.264	B	Discarding a major feature of the fishery	–
WGNSSK	Norway pout	North Sea (IV, IIIaN)	1983–2008	2005–2008	0–4	1–2	1.08	0.100	0.479	0.536	0.831	A	Not mentioned	Age 0 excluded ($F < 0.03$), quarters transformed to years
WGHMM	Hake	Southern stock (VIIIc, IXa)	1982–2008	1997–2008	1–8	2–5	3.09	0.498	0.569	0.624	0.574	B	Discard rate considered to be high at young ages	–
WGHMM	Hake	Northern stock (IIIa, IV, VI, VII, VIIIa,b)	1978–2008	1994–2008	1–8	2–6	3.80	0.299	0.385	0.565	0.936	B	Underestimated *F* at younger ages	–

Continued

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Table 1. *Continued*

ICES Working Group	Species	Area	Report time-series	Time interval for *ER*, *EP*	Age classes	Age classes for *ER*	A_{50} (years)	*ER* (year^{-1})	EP_1	EP_2	*CSS*	*AE*	Comments on discarding	Other comments
WGNSSK	Sandeel	North Sea (IV)	1983–2008	2003–2008	0–4	1–2	1.51	0.558	0.948	0.747	0.641	A	Not mentioned	Age 0 excluded ($F < 0.07$)
WGCSE	Plaice	Irish Sea (VIIa)	1964–2008	1997–2008	2–9	3–6	3.00	0.263	0.257	0.255	2.377	B	High levels of discarding	Age 1 reconstructed
WGCSE	Plaice	Celtic Sea (VIIf,g)	1977–2008	1998–2008	1–9	3–6	2.85	0.556	0.292	0.326	0.561	B	Heavy discarding of youngest age classes	–
WGNSSK	Plaice	North Sea (IV)	1957–2008	1999–2008	1–10	2–6	2.49	0.501	0.647	0.495	1.226	A	Discards included in the assessment	–
WGNSSK	Plaice	Eastern Channel (VIId)	1980–2008	1998–2008	1–10	3–6	2.86	0.743	0.379	0.485	0.524	B	Substantial discarding, up to 79%	–
WGCSE	Plaice	Western Channel (VIIe)	1976–2008	1998–2008	1–10	3–7	2.85	0.596	0.357	0.374	0.620	B	Variable discarding	–
WGCSE	Sole	Celtic Sea (VIIf,g)	1971–2008	1997–2008	1–10	4–8	3.05	0.428	0.316	0.307	1.446	A	Low discard rates (up to 5%)	–
WGCSE	Sole	Irish Sea (VIIa)	1970–2008	1999–2008	2–8	4–7	2.45	0.415	0.250	0.146	0.478	A	Low discard rates (0–8%)	Age 1 reconstructed
WGNSSK	Sole	North Sea (IV)	1957–2008	1999–2008	1–10	2–6	2.50	0.508	0.213	0.252	0.819	B	High discard rates (6–29%)	–
WGNSSK	Sole	Eastern Channel (VIId)	1982–2008	1999–2008	1–11	3–8	2.50	0.436	0.336	0.289	1.390	A	Discards not substantial	–
WGBFAS	Sole	Skagerrak and Kattegat (IIIa)	1984–2008	1999–2008	2–9	4–8	2.50	0.301	0.130	0.099	2.427	A	Negligible discarding	Age 1 reconstructed
WGHMM	Sole	Bay of Biscay (VIIa,b)	1984–2008	1999–2008	2–8	3–6	2.38	0.504	0.244	0.214	1.008	B	Discards may be important at age 1	Age 1 reconstructed
NWWG	Herring	Iceland (Va)	1986–2008	1995–2008	3–12	5–10	3.45	0.292	0.169	0.177	2.219	A	No discarding	Ages 1, 2 reconstructed
HAWG	Herring	Celtic Sea (VIIa(S),g,h,j,k)	1958–2008	2005–2008	1–6	2–5	1.00	0.281	0.129	0.050	1.132	A	Discarding not a feature of the fishery	–
HAWG	Herring	North Sea (IV, IIIa, VIId)	1960–2008	2002–2008	0–9	2–6	1.86	0.297	0.311	0.392	0.817	A	Discards included in the assessment	Age 0 excluded ($F < 0.03$)
WGWIDE	Herring	NE Atlantic (IIa,b, Va,b, XIVa)	1988–2008	1992–2008	0–15	5–14	4.29	0.143	0.057	0.132	2.390	A	Very low discarding	Age 0 excluded ($F < 0.001$)
WGWIDE	Mackerel	NE Atlantic (IIa, IIIa,b,d, IV, Va, Vb, VI, VII, VIIIa,b,d,e, XII, VIIIc, IXa, XIV)	1972–2008	2001–2008	0–12	4–8	2.00	0.327	0.250	0.147	1.083	A	Discards included in the assessment	Age 0 excluded ($F < 0.02$)
WGWIDE	Horse Mackerel	Western Stock (IIa, IIIa (W), IVa, Vb, VIa, VIIa–c, VIIe–k, VIIIa–e)	1982–2008	1995–2008	0–11	4–8	3.55	0.085	0.679	0.345	1.518	A	Discards included in the assessment	Age 0 excluded ($F < 0.01$)
WGWIDE	Blue Whiting	NE Atlantic (I–IX, XII, XIV)	1981–2008	2000–2008	1–10	3–7	2.35	0.424	0.385	0.247	2.648	A	Minor discarding	–

Comments on discarding are as expressed in respective ICES reports. A_{50}, age at 50% maturity; *ER*, mean exploitation rate ($\bar{F}$), Equation (2); EP_1, mean F_{imm}/F_{mat}, Equation (5); EP_2, mean proportion of immature fish in the catch, Equation (7); *CSS*, average B/B_{pa} in the last 3 years of every assessment, Equation (1), and *AE*, assessment effect. A indicates low bias and B higher bias.

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Table 2. Evaluation of candidate models for CSS using either EP_1 or EP_2 to quantify exploitation pattern.

Model	Formula	Deviance explained (%)	AIC	Δi
m_1	$c + ER + EP_1$ + factor (*AE*)	44.3	66.30	0.0
m_2	$c + ER + EP_1$	40.4	67.00	0.70
m_3	$c + ER{:}EP_1 + ER + EP_1$ + factor (*AE*)	44.8	67.98	1.68
m_4	$c + ER{:}EP_1 + ER + EP_1$	40.5	68.97	2.67
m_5	$c + ER$	33.6	69.27	2.97
m_6	$c + ER$ + factor (*AE*)	35.4	70.22	3.92
m_7	$c + ER + EP_2$	33.7	71.20	4.90
m_8	$c + ER + EP_2$ + factor (*AE*)	35.6	72.06	5.76
m_9	$c + ER{:}EP_2 + ER + EP_2$ + factor (*AE*)	38.1	72.49	6.19
m_{10}	$c + ER{:}EP_2 + ER + EP_2$	34.6	72.67	6.37
m_{11}	$c + EP_1$ + factor (*AE*)	24.6	76.37	10.07
m_{12}	$c + EP_1$	15.2	79.08	12.78
m_{13}	c + factor (*AE*)	6.1	83.24	16.95
m_{14}	$c + EP_2$ + factor (*AE*)	9.8	83.56	17.27
m_{15}	c	0	83.72	17.42
m_{16}	$c + EP_2$	4.2	83.97	17.68

AIC, Akaike's information criterion; Δi, Akaike's differences.

Table 3. Summary of the optimal model selected for CSS (m_1).

Coefficient	Estimate	s.e.	*t*-value	*p*-value
Intercept	1.240	0.210	5.913	1.12e−06
ER (year^{-1})	−1.232	0.344	−3.582	0.001
EP_1	−1.058	0.432	−2.450	0.020
Factor(*AE*) B	−0.294	0.190	−1.547	0.131

s.e., standard error.

Table 4. Summary of the second best model selected for CSS (m_2).

Coefficients	Estimate	s.e.	*t*-value	*p*-value
Intercept	1.157	0.218	5.320	6.09e−06
ER (year^{-1})	−1.360	0.342	−3.982	3.29e−04
EP_1	−0.883	0.444	−1.986	0.055

s.e., standard error.

Presumably, this is because EP_1 is a direct metric of exploitation pattern that takes into account the whole fish population, whereas EP_2 is a proxy that ignores the immature fish that are not captured and will potentially be allowed to spawn in future. Plotting EP_2 against EP_1 (Figure 1) reveals that the stocks appearing as outliers are those with a large proportion of immature fish in the catch in conjunction with low F_{imm} values relative to F_{mat}. The fact that all five outlier stocks are within precautionary limits ($CSS > 1$) despite the large proportions of immature fish in the catch also illustrates the greater sensitivity of EP_1.

The quantity F_{imm}/F_{mat} could be used potentially as an indicator of fisheries sustainability for data-rich stocks where values of *F*-at-age are available. It has the main desirable characteristics for this purpose (Liu and Ou, 2007): (i) scientific validity; (ii) availability over broad scales of time and space; (iii) acceptability, simplicity, and relevance to fisheries; and (iv) ease of monitoring. F_{imm}/F_{mat} could be implemented in fisheries management to describe exploitation pattern in the same way that $\overline{F}$ is used today to represent exploitation rate. Further investigation on a stock-by-stock basis would be needed though to suggest stock-specific limits of exploitation pattern analogous to those used

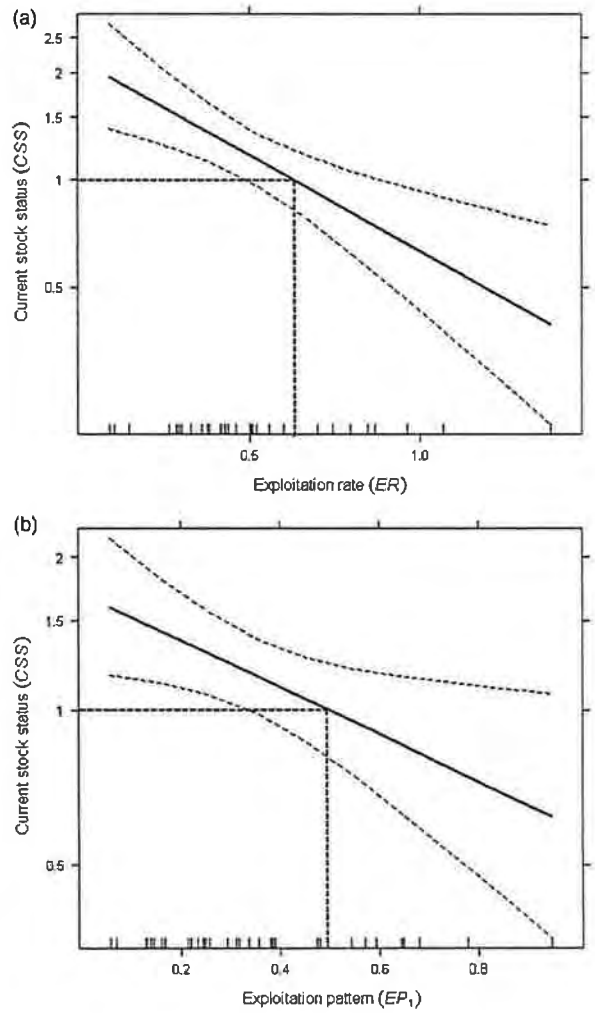

Figure 2. Effect displays for the continuous explanatory variables of model m_1. Curved dashed lines indicate the 95% confidence intervals. Straight dashed lines indicate *ER* and EP_1 values for $CSS = 1$. (a) Effect display of *ER*; (b) effect display of EP_1.

today for exploitation rate, so that exploitation pattern could become part of fisheries advice. For fish stocks where *F*-at-age is not available, EP_2 could be used as an indicator of sustainability only if it can be confirmed that it is representative of the pressure on the immature fish in the population, i.e. if there are not great numbers of immature fish that remain unfished.

Another conclusion that can be derived from this type of analysis is the existence of a trade-off between exploitation rate and exploitation pattern; the lower the exploitation rate, the less the importance of EP_1 and vice versa. Consequently, either a combination of a relatively high exploitation rate with a low relative exploitation of immature fish, or a combination of a high relative exploitation of immature fish with a low exploitation rate, could improve stock status. For example, the NE Arctic cod stock and the Celtic Sea whiting stock are in good condition despite being exploited at quite high rates owing to a very low relative exploitation of immature fish (Table 1). On the other hand, the saithe fisheries of the NE Arctic and North Sea exploit immature fish more

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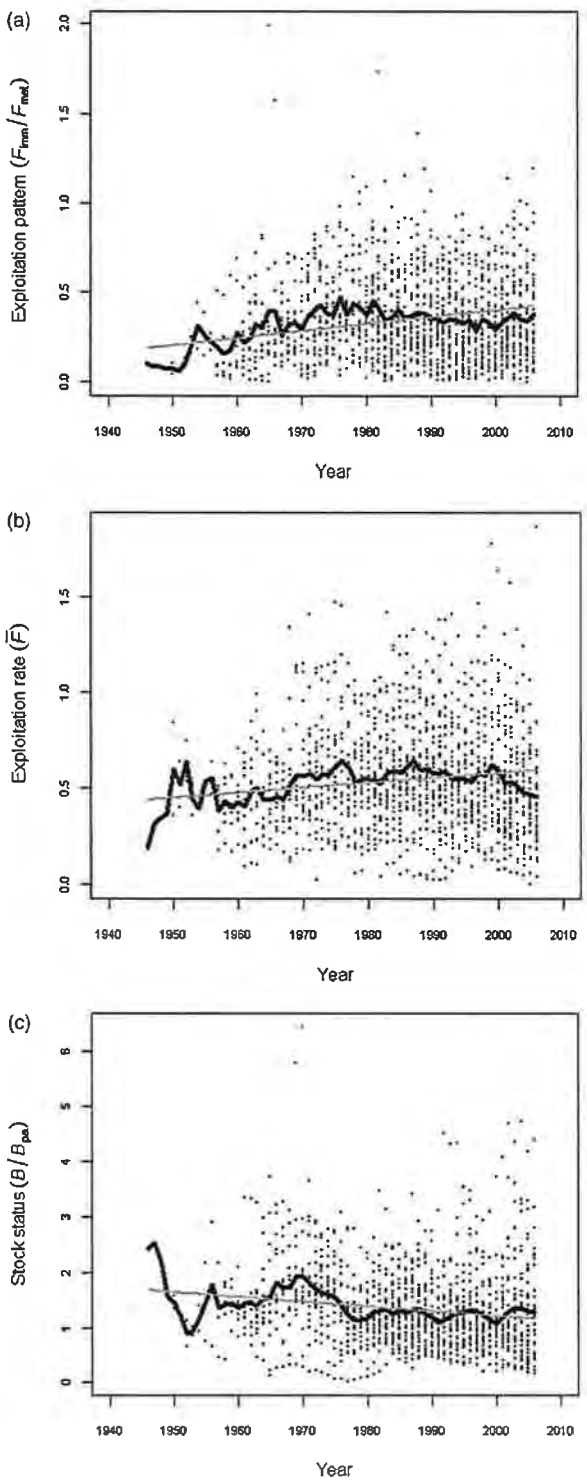

Figure 3. (a) Exploitation pattern, (b) exploitation rate, and (c) stock status for all 38 stocks over time. The black line reflects the mean values by year and the grey line the linear regression. One data point ($F_{imm}/F_{mat} = 4.6$ in 1978 for the North Sea herring stock) was omitted from (a) as an outlier.

heavily but good stock status is sustained by lower exploitation rates (Table 1). Moreover, the independent effects of exploitation rate and exploitation pattern on stock status should be taken into consideration in management plans where fishers are compensated for using more selective gears by being given more days at sea (e.g. the Scottish conservation credits scheme; Edwards, 2009). Protection of immature fish is not a panacea; if the cost of improving exploitation pattern is to increase exploitation rate, then the management plans might not fulfil their goals. This agrees with the perception that fisheries managers should not rely on measures regulating either exploitation rate or exploitation pattern, but both (Halliday and Pinhorn, 2002; Caddy and Agnew, 2004).

To evaluate the general status of the 38 fish stocks used in this study, the trends of F_{imm}/F_{mat}, $\bar{F}$, and B/B_{pa} of all stocks for the period 1946–2006 were also investigated. Estimates for the most recent 2 years were excluded because of their low precision (Pope, 1972). This analysis is directly comparable with the one conducted by Sparholt *et al.* (2007), who studied the trends of exploitation rate and stock status for the period 1946–2003 using a similar dataset of 38 stocks assessed by ICES. Froese and Proelss (2010) also conducted a similar meta-analysis for 54 fish stocks in the NE Atlantic. F_{imm}/F_{mat} has overall an increasing trend, but it appears to have been more or less stable in the past 20 years (Figure 3a). Meanwhile, there is a clear decreasing trend of exploitation rate in the last 8 years of the time-series (Figure 3b). The fact that this decreasing trend of exploitation rate has not resulted in a noticeable improvement in stock status (Figure 3c) possibly indicates that a twofold management approach aiming at reducing both exploitation rate and proportional exploitation of immature fish might be more effective. The trends of exploitation rate and stock status presented here (Figure 3b and c) are similar to those observed by Sparholt *et al.* (2007) and Froese and Proelss (2010).

The results of this study must, however, be interpreted with caution because they are a meta-analysis of data ($N_{a,y}$, $F_{a,y}$, B_y), which are outputs of other models (mostly VPA type) used by ICES to assess fish stocks and which accordingly include a certain amount of uncertainty. Moreover, the primary data ($C_{a,y}$, maturity ogives) of the stock assessment reports are also prone to misreporting and/or bias (Pinhorn and Halliday, 2001); for instance, there are stocks where the maturity ogive is updated every year (e.g. NE Arctic cod) and others where it has not been updated for decades (e.g. blue whiting). The exploitation pattern metrics used here also assume the same vulnerability to fishing for immature and mature fish within the same age classes, which might not be the case for species such as herring or blue whiting that are caught in spawning areas or when undergoing spawning migrations (Horbowy, 2005). Therefore, there is a need for stock assessment reports to provide more accurate information on the maturity ogives of exploited stocks and on the maturity stage of the fish caught in the future. Finally, defining current stock status based on B_{pa} is in accordance with ICES practice, but B_{pa} is not well defined and can be formulated differently between stocks. Nevertheless, this type of meta-analysis makes general trends and patterns clearer and can help to synthesize the results of individual stock assessments into good management strategies (Sparholt *et al.*, 2007).

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The results of this study describe the independent effects of exploitation rate and pattern on fisheries sustainability. They do not consider important stock-specific features such as different levels of resilience to the same levels of exploitation rate and exploitation pattern and/or the role of environmental variability (e.g. Fréon *et al.*, 2005). Such effects are expected to account for the large amount of residual deviance that our models did not explain (ca. 56% for m_1). The results suggest that immature fish should be protected from fishing (e.g. through the implementation of more selective fishing gears) and that fisheries management could benefit by placing more emphasis on exploitation pattern, which is currently overlooked in favour of exploitation-rate-based approaches.

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References

Armstrong, D. W., Ferro, R. S. T., MacLennan, D. N., and Reeves, S. A. 1990. Gear selectivity and the conservation of fish. Journal of Fish Biology, 37: 261–262.

Beverton, R. J. H., and Holt, S. J. 1957. On the dynamics of exploited fish populations. Fishery Investigations, London, Series II, XIX. 533 pp.

Burnham, K. P., and Anderson, D. R. 2002. Model Selection and Multimodel Inference: a Practical Information-Theoretic Approach, 2nd edn. Springer, New York.

Caddy, J. F., and Agnew, D. J. 2004. An overview of recent global experience with recovery plans for depleted marine resources and suggested guidelines for recovery planning. Reviews in Fish Biology and Fisheries, 14: 43–112.

Caddy, J. F., and Seijo, J. C. 2002. Reproductive contributions foregone with harvesting: a conceptual framework. Fisheries Research, 59: 17–30.

Cardinale, M., and Arrhenius, F. 2000. The relationship between stock and recruitment: are the assumptions valid? Marine Ecology Progress Series, 196: 305–309.

Crawley, M. J. 2007. The R book. John Wiley and Sons Ltd, West Sussex, UK. 942 pp.

de Roos, A. M., Boukal, D. S., and Persson, L. 2006. Evolutionary regime shifts in age and size at maturation of exploited fish stocks. Proceedings of the Royal Society of London, Series B: Biological Sciences, 273: 1873–1880.

Edwards, T. 2009. Sea Fishing: Quotas for 2010. Scottish Parliament Information Centre Briefing, 23 October 2009, 09/74, p. 16. The Scottish Parliament, Edinburgh. 32 pp.

Enberg, K. 2005. Benefits of threshold strategies and age-selective harvesting in a fluctuating fish stock of Norwegian spring spawning herring *Clupea harengus*. Marine Ecology Progress Series, 298: 277–286.

FAO. 2010. Fisheries Glossary. http://www.fao.org/fi/glossary/

Fox, J. 2003. Effect displays in R for generalised linear models. Journal of Statistical Software, 8: 1–27.

Fréon, P., Cury, P., Shannon, L., and Roy, C. 2005. Sustainable exploitation of small pelagic fish stocks challenged by environmental and ecosystem changes: a review. Bulletin of Marine Science, 76: 385–462.

Froese, R. 2004. Keep it simple: three indicators to deal with overfishing. Fish and Fisheries, 5: 86–91.

Froese, R., and Proelss, A. 2010. Rebuilding fish stocks no later than 2015: will Europe meet the deadline? Fish and Fisheries, 11: 194–202.

Froese, R., Stern-Pirlot, A., Winker, H., and Gascuel, D. 2008. Size matters: how single-species management can contribute to ecosystem-based fisheries management. Fisheries Research, 92: 231–241.

Gilbert, D. J. 1997. Towards a new recruitment paradigm for fish stocks. Canadian Journal of Fisheries and Aquatic Sciences, 54: 969–977.

Haddon, M. 2001. Modelling and Quantitative Methods in Fisheries. Chapman and Hall/CRC, Boca Raton, FL.

Halliday, R. G., and Pinhorn, A. T. 2002. A review of the scientific and technical bases for policies on the capture of small fish in North Atlantic groundfish fisheries. Fisheries Research, 57: 211–222.

Horbowy, J. 2005. Assessing Baltic herring stocks with a model that incorporates migration. Fisheries Research, 76: 266–276.

ICES. 2007. Report of the Workshop on Discard Raising Procedures, 6–9 February 2007, San Sebastian, Spain. ICES Document CM 2007ACFM: 06. 57 pp.

ICES. 2008. Report of the ICES Advisory Committee 2008. ICES Advice, 1–5.

Ishida, Y., Funamoto, T., Honda, S., Yabuki, K., Nishida, H., and Watanabe, C. 2009. Management of declining Japanese sardine, chub mackerel and walleye pollock fisheries in Japan. Fisheries Research, 100: 68–77.

Jarre-Teichmann, A., Wieland, K., MacKenzie, B. R., Hinrichsen, H., Plikshs, M., and Aro, E. 2000. Stock–recruitment relationships for cod (*Gadus morhua* L.) in the central Baltic Sea incorporating environmental variability. Archive of Fishery and Marine Research, 48: 97–123.

Katsanevakis, S. 2006. Modelling fish growth: model selection, multimodel inference and model selection uncertainty. Fisheries Research, 81: 229–235.

Kennelly, S. J., and Broadhurst, M. K. 2002. By-catch begone: changes in the philosophy of fishing technology. Fish and Fisheries, 3: 340–355.

Klanjšček, J., and Legović, T. 2007. Is anchovy (*Engraulis encrasicolus*, L.) overfished in the Adriatic Sea? Ecological Modelling, 201: 312–316.

Liu, W. H., and Ou, C. H. 2007. A comparative analysis of sustainable fishery development indicator systems in Australia and Canada. Sustainable Development, 15: 28–40.

Marshall, C. T., Yaragina, N. A., Lambert, Y., and Kjesbu, O. S. 1999. Total lipid energy as a proxy for total egg production by fish stocks. Nature, 402: 288–290.

Marteinsdottir, G., and Steinarsson, A. 1998. Maternal influence on the size and viability of Iceland cod *Gadus morhua* eggs and larvae. Journal of Fish Biology, 52: 1241–1258.

Marteinsdottir, G., and Thorarinsson, K. 1998. Improving the stock–recruitment relationship in Icelandic cod (*Gadus morhua*) by including age diversity of spawners. Canadian Journal of Fisheries and Aquatic Sciences, 55: 1372–1377.

Mori, M., Katsukawa, T., and Matsuda, H. 2001. Recovery plan for an exploited species, southern bluefin tuna. Population Ecology, 43: 125–132.

Myers, R. A. 2001. Stock and recruitment: generalizations about maximum reproductive rate, density dependence, and variability using meta-analytic approaches. ICES Journal of Marine Science, 58: 937–951.

Myers, R. A., and Barrowman, N. J. 1996. Is fish recruitment related to spawner abundance? Fishery Bulletin US, 94: 707–724.

Myers, R. A., and Mertz, G. 1998. The limits of exploitation: a precautionary approach. Ecological Applications, 8: S165–S169.

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Myers, R. A., and Quinn, T. J. 2002. Estimating and testing non-additivity in fishing mortality: implications for detecting a fisheries collapse. Canadian Journal of Fisheries and Aquatic Sciences, 59: 597–601.

Palakovich Carr, J., and Kaufman, L. 2009. Estimating the importance of maternal age, size, and spawning experience to recruitment of Atlantic cod (*Gadus morhua*). Biological Conservation, 142: 477–487.

Petrakis, G., and Stergiou, K. I. 1997. Size selectivity of diamond and square mesh codends for four commercial Mediterranean fish species. ICES Journal of Marine Science, 54: 13–23.

Pinhorn, A. T., and Halliday, R. G. 2001. The regulation of exploitation pattern in North Atlantic groundfish stocks. Fisheries Research, 53: 25–37.

Pope, J. G. 1972. An investigation of the accuracy of virtual population analysis using cohort analysis. ICNAF Research Bulletin, 9: 65–74.

R Development Core Team 2010. R: a Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, http://www.R-project.org/.

Sparholt, H., Bertelsen, M., and Lassen, H. 2007. A meta-analysis of the status of ICES fish stocks during the past half century. ICES Journal of Marine Science, 64: 707–713.

Trippel, E. A., and Neil, S. R. E. 2004. Maternal and seasonal differences in egg sizes and spawning activity of Northwest Atlantic haddock (*Melanogrammus aeglefinus*) in relation to body size and condition. Canadian Journal of Fisheries and Aquatic Sciences, 61: 2097–2110.

Wakeford, R. C., Agnew, , D. J., and Mees, C. C. 2007. Review of institutional arrangements and evaluation of factors associated with successful stock recovery plans. CEC 6th Framework Programme No. 022717 UNCOVER. MRAG Report, March 2007. 58 pp.

Wileman, D. A., Ferro, R. S. T., Fonteyne, R., and Millar, R. B. (Eds). 1996. Manual of methods of measuring the selectivity of towed fishing gears. ICES Cooperative Research Report, 215. 126 pp.

Worm, B., Hilborn, R., Baum, J. K., Branch, T. A., Collie, J. S., Costello, C., Fogarty, M. J., *et al.* 2009. Rebuilding global fisheries. Science, 325: 578–585.

Zhou, S., Smith, A. D. M., Punt, A. E., Richardson, A. J., Gibbs, M., Fulton, E. A., Pascoe, S., *et al.* 2010. Ecosystem-based fisheries management requires a change to the selective fishing philosophy. Proceedings of the National Academy of Sciences of the USA, 107: 9485–9489.

Zuur, A. F., Ieno, E. N., and Smith, G. M. 2007. Analysing Ecological Data. Springer Science + Business Media, LLC, New York.

Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A., and Smith, G. M. 2009. Mixed Effects Models and Extensions in Ecology with R. Springer Science + Business Media, LLC, New York.

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