AGENDA C-4
APRIL 2003

MEMORANDUM

TO: Council, SSC and AP Members

FROM: Chris Oliver
Executive Director

ESTIMATED TIME
4 HOURS

DATE: March 25, 2003

SUBJECT: Essential Fish Habitat

ACTION REQUIRED

(a) Receive progress report on EIS development.
(b) Review mitigation objectives and research plan.

BACKGROUND

Staff will provide a progress report on the development of the EIS for the EFH amendments. A packet of materials was mailed out to you last week. These materials included a draft table of contents for the EIS and RIR, drafts of Chapters 1 & 2, a discussion paper on research and monitoring approaches, and an unpublished draft manuscript on living substrates. Staff will provide a brief overview of these documents, and also review how the SSC concerns (regarding conceptual approach, goals and objectives, research plan, and analytical components) have been addressed to date.

In February, the Council provided additional guidance and definition regarding the alternatives to minimize the effects of fishing (motion attached as Item C-4(a)). Included in the suite of mitigation alternatives is Alternative 5B, which contains a number of measures designed to reduce the effects of fisheries on corals and sponges in the Aleutian Islands area. Measures contained in Alternative 5B include additional monitoring requirements, coral/bryozoan and sponge bycatch limits, designated 'open areas' where bottom trawling is allowed, closure of areas with high coral and sponge bycatch rates and low target species CPUE, and a reduction of groundfish TAC by amount that historically came from the closure areas.

Since the February meeting, NMFS staff has applied the methodology used for the Aleutian Islands component of Alternative 5B to the Bering Sea and Gulf of Alaska, to determine whether or not the approach merits evaluation as an additional alternative. A letter from Dr. Balsiger regarding this effort, along with the resulting closure area maps, is attached Item C-4(b). NMFS staff will be on hand to report their findings.

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AGENDA C-4(a)
APRIL 2003

Essential Fish Habitat
Final Mitigation Alternatives
Based on February 3, 2002 Council Motion

Alternative 1: Status quo. No additional measures would be taken at this time to minimize the effects of fishing on EFH.

Alternative 2: Gulf Slope Bottom Trawl Closures: Prohibit the use of bottom trawls for rockfish in 11 designated areas of the GOA slope (200m-1000m), but allow vessels endorsed for trawl gear to fish for rockfish in these areas with fixed gear or pelagic trawl gear.

Alternative 3: Bottom Trawl Gear Prohibition for GOA Slope Rockfish on upper slope area (200-1,000m). Prohibit the use of bottom trawl gear for targeting GOA slope rockfish species on upper slope area (200-1000m), but allow vessels endorsed for trawl gear to fish for slope rockfish with fixed gear or pelagic trawl gear.

Alternative 4: Bottom Trawl Closures in All Management Areas: Prohibit the use of bottom trawl gear in designated areas of the Bering Sea, Aleutian Islands, and Gulf of Alaska. In the Bering Sea only, bottom trawl gear used in the remaining open areas would be required to have disks/bobbins on trawl sweeps and footropes.

> Bering Sea: Prohibit the use of bottom trawl gear for all groundfish fisheries in the Bering Sea except within a designated "open" area. The open area is designated based on historic bottom trawl effort. Within the open area, there would be a rotating closure to bottom trawl gear in 5 areas to the west, north and northwest of the Pribilof Islands. Closure areas would be designated in Blocks 1, 2, 3, 4, and 6 as identified by the EFH Committee, with ten-year closed periods for 25% of each block. After ten years, the closed portion of each block would re-open and a different 25% of each block would close for ten years, and so on thereafter. After 40 years, all areas within each block will have been subjected to a ten-year closure. This assures that 20% of the habitat has matured to an 'unaffected by fishing' status, assuming a two-year recovery interval.
>
> Aleutian Islands: Prohibit the use of bottom trawl gear for all groundfish fisheries in designated areas of the Aleutian Islands. Closure areas would be designated in the areas of Stalemate Bank, Bowers Ridge, Seguam Foraging Area, and Semisopochnoi Island.
>
> Gulf of Alaska: Prohibit the use of bottom trawl gear for rockfish fisheries on 11 designated sites of the GOA slope (200m-1000m). Allow vessels endorsed for trawl gear to fish for rockfish in these areas with fixed gear or pelagic trawl gear.

Alternative 5: Expanded Bottom Trawl Closures in All Management Areas: Prohibit the use of bottom trawl gear in designated areas of the Bering Sea, Aleutian Islands, and Gulf of Alaska. In the Bering Sea only, bottom trawl gear used in the remaining open areas would be required to have disks/bobbins on trawl sweeps and footropes.

Bering Sea: Prohibit the use of bottom trawl gear for all groundfish fisheries in the Bering Sea except within a designated "open" area. The open area is designated based on historic bottom trawl effort. Within the open area, there would be a rotating closure to bottom trawl gear in 5 areas to the west, north and northwest of the Pribilof Islands. Closure areas would be designated in Blocks 1, 2, 3, 4, and 6 as identified by the EFH Committee, with five-year closed periods for 33 1/3% of each block. After five years, the closed portion of each block would re-open and a different 33 1/3% of each block would close for five years, and so on thereafter. After 15 years, all areas within each block will have been subjected to a five-year closure. This assures that 20% of the habitat has matured to an 'unaffected by fishing' status, assuming a two-year recovery interval.

Aleutian Islands: Prohibit the use of bottom trawl gear for all groundfish fisheries in designated areas of the Aleutian Islands. Closure areas would be designated in the areas of Stalemate Bank, Bowers Ridge, Seguam Foraging Area, Yunaska Island , and Semisopochnoi Island. These closure areas extend to the northern and southern boundaries of the AI management unit.

Suboption for Aleutian Islands: Oceana's Aleutian Seafloor Habitat Protection Alternative dated Dec. 6, 2002. Close areas to bottom trawling that have high coral and sponge bycatch rates and low target species CPUE and reduce TAC by amount that historically came from that those. No expansion of bottom trawl fisheries to new areas. Pelagic trawls may be used in the closed areas, but only in the off-bottom mode. Institute area-specific coral/ sponge bycatch limits that close specific areas if exceeded. If implemented it would include the following actions: Expand observer coverage to 100%, utilize the CADRES program, and require each vessel to have VMS.

Additionally the proposal requests a comprehensive plan for research and monitoring that would include: Seafloor mapping, benthic research, and habitat impacts of all bottom tending gears, annual habitat assessment reports, experimental fishing permits to identify additional open areas.

Gulf of Alaska: Prohibit the use of bottom trawl gear for all groundfish fisheries on 10 designated sites of the GOA slope (200m-1000m). Additionally, prohibit the use of bottom trawls for targeting GOA slope rockfish on the GOA slope (200-1000 meters), but allow vessels endorsed for trawl gear to fish for rockfish in these areas with fixed gear or pelagic trawl gear.

Alternative 6: Closures to All Bottom Tending Gear

Prohibit the use of all bottom tending gear (dredges, bottom trawls, pelagic trawls that contact the bottom, longlines, and pots) within approximately 20% of the fishable waters (i.e., 20% of the waters shallower than 1,000m) in each of the regions described below.

Gulf of Alaska: The Gulf of Alaska would be subdivided into 3 regions: Western (corresponding to regulatory area 610), Central (areas (620 and 630), and Eastern (areas 640 and 650).

Aleutian Islands: The Aleutian Islands would be subdivided into 4 regions: Western (corresponding to regulatory area 543), Central (area 542), Eastern (area 541), and two smaller Bering Sea regulatory areas adjacent tot he Aleutians (combination of areas 518 and 519).

Bering Sea: The Bering Sea would be subdivided into 3 regions south of St. Lawrence Island denoting each of the predominant substrate types (sand, sand/mud, and mud) and taking into consideration the varying depth distribution of each substrate.

The closed areas would be identified based on the presence of habitat such as high relief coral, sponges, and Boltenia, with emphasis on areas with notable benthic structure and / or high concentrations of benthic invertebrates that provide shelter for managed species. The closed areas would include a mix of relatively undisturbed habitats and habitats that currently are fished. Within a given region, existing area closures could comprise all or a portion of the closed areas for this alternative.

DRAFT MOTION February 3, 2003 4:40pm

1. Alternatives 1 through 6 shall be included in the analysis.

2. The EIS package shall move forward with no new or modified alternatives added, with the exception of those changes listed below.

3. The following changes to Alternatives 4 and 5 for the Bering Sea shall be made to reflect the SSC comments on rotational closures:

Alternatives 4 and 5 for the Bering Sea would be modified as follows:

Bering Sea Alternative 4, starting in the third sentence:

"Closure areas would be designated in Blocks 1, 2, 3, 4, and 6 as identified by the EFH Committee, with ten-year closed periods for 25% of each block. After ten years, the closed portion of each block would re-open and a different 25% of each block would close for ten years, and so on thereafter. After 40 years, all areas within each block will have been subjected to a ten-year closure. This assures that 20% of the habitat has matured to an 'unaffected by fishing' status, assuming a two-year recovery interval."

Bering Sea Alternative 5, starting in the third sentence:

"Closure areas would be designated in Blocks 1, 2, 3, 4, and 6 as identified by the EFH Committee, with five-year closed periods for 33 1/3% of each block. After five years, the closed portion of each block would re-open and a different 33 1/3% of each block would close for five years, and so on thereafter. After 15 years, all areas within each block will have been subjected to a five-year closure. This assures that 20% of the habitat has matured to an 'unaffected by fishing' status, assuming a two-year recovery interval."

4. In Alternatives 4 and 5, the map of the open area should be corrected to reflect the Committee's intent that the open area at the southernmost boundary be extended to include Cod Alley.

5. In Alternatives 4 and 5, the language in the second sentence should be modified by adding at the beginning: "In the Bering Sea only, bottom trawl gear . . . "

6. In Alternative 5b, the sub-option for the Aleutians, add language clarifying that pelagic trawls may be used in the closed areas, but only in the off-bottom mode. Discuss in the analysis a method for identifying bottom contact acceptable to enforcement.

7. In Alternative 5b, Area Definition, open areas shall be modified by staff per suggestions from USCG in the following manner: use Lat/Lon grid based on 3 minutes of latitude by 6 minutes of longitude. This will align with and subdivide existing ½ by 1 degree ADF&G statistical areas in the geo-reference system familiar to the fishing fleets, and is roughly equivalent to a 5 by 5 km block.

8. Under all alternatives, evaluate how VMS and/or a secure on-board tracking systems may or may not improve enforcement.

9. Each mitigation alternative shall include a research and monitoring component to help determine the efficacy of that alternative, should it be implemented, and to determine to the extent practical the effects of fishing on habitat. Each alternative shall contain specific language as to the intent and objectives of its research component, linked with the goals of the alternative. The final hypothesis driven research design shall be developed when the preferred alternative is selected, in a subsequent process that includes public and stakeholder input.

All alternatives should contain benthic mapping to improve future management and meet research goals.

In the proposed research components – <u>attempt</u> to have all research closure/open blocks square rather than irregular shapes.

The Council supports full funding of the essential fish habitat research described in 9 above.

National Oceanic and Atmospheric Administration
National Marine Fisheries Service
P.O. Box 21668
Juneau, Alaska 99802-1668

AGENDA C-4(b)
APRIL 2003

March 25, 2003

RECEIVED
MAR 25 2003
N.P.F.M.C

David Benton, Chairman
North Pacific Fishery Management Council
605 W 4th Street
Anchorage, Alaska 99501-2252

Dear Mr. Benton:

Since the last Council meeting, National Marine Fisheries Service (NMFS) staff have completed additional tasks related to Alternative 5b for the Essential Fish Habitat Environmental Impact Statement (EIS). In particular, staff have completed preliminary work on applying the methodology from Alternative 5b for the Aleutian Islands to the Bering Sea and Gulf of Alaska to determine whether that approach merits evaluation as an additional alternative. We undertook this work in response to the request from Oceana that NMFS and the Council evaluate this option. NMFS staff will present their work, including the enclosed figures and tables, during the April Council meeting.

The staff work associated with this task was necessary regardless of whether the EIS ultimately includes a 5b-type alternative for the Bering Sea and Gulf of Alaska. If the Council and NMFS decide to add such an alternative, these work products will help the Council select specific open and closed areas. If the Council and NMFS decide not to pursue such an alternative, the EIS will need to discuss why it was considered yet not carried forward, and these work products will help to document many of the relevant considerations. We look forward to discussing through the Council process the pros and cons of adding another alternative.

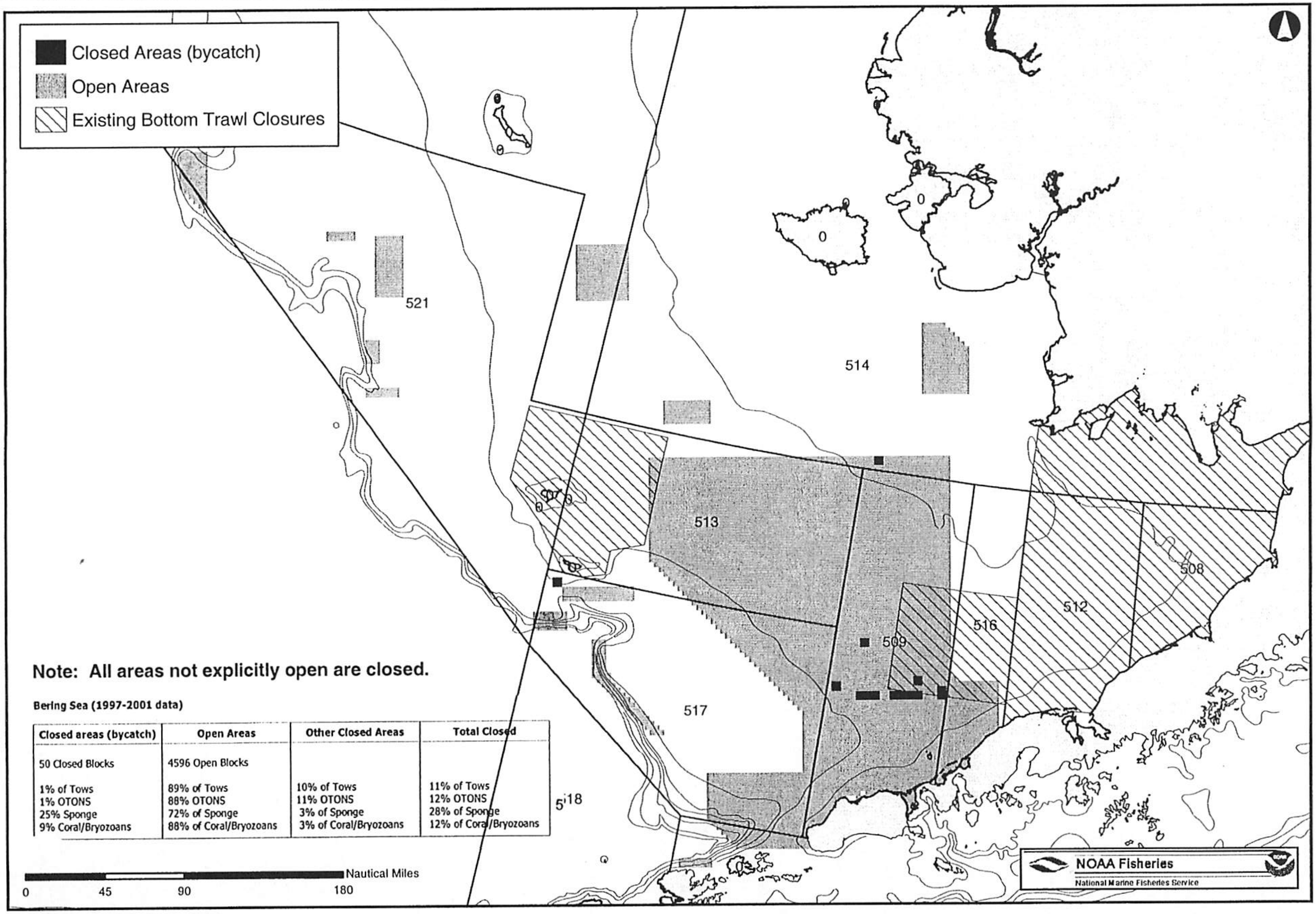
Sincerely,

James W. Balsiger
Administrator, Alaska Region

Enclosures

ALASKA REGION - www.fakr.noaa.gov

Alternative 5b - Bering Sea

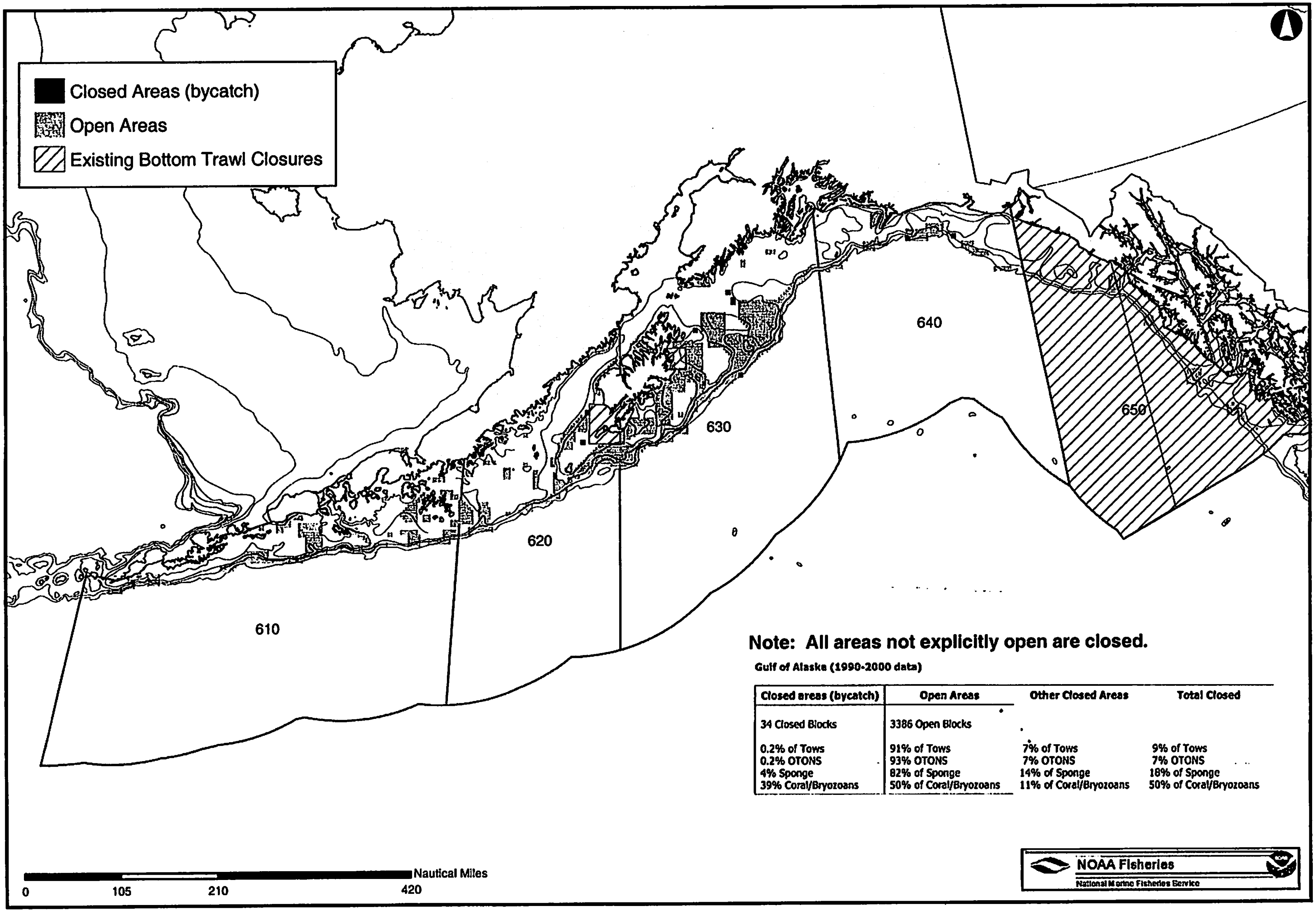


Note: All areas not explicitly open are closed.

Bering Sea (1997-2001 data)

Closed areas (bycatch)	Open Areas	Other Closed Areas	Total Closed
50 Closed Blocks	4596 Open Blocks		
1% of Tows 1% OTONS 25% Sponge 9% Coral/Bryozoans	89% of Tows 88% OTONS 72% of Sponge 88% of Coral/Bryozoans	10% of Tows 11% OTONS 3% of Sponge 3% of Coral/Bryozoans	11% of Tows 12% OTONS 28% of Sponge 12% of Coral/Bryozoans

Alternative 5b - Gulf of Alaska



Note: All areas not explicitly open are closed.

Gulf of Alaska (1990-2000 data)

Closed areas (bycatch)	Open Areas	Other Closed Areas	Total Closed
34 Closed Blocks	3386 Open Blocks		
0.2% of Tows	91% of Tows	7% of Tows	9% of Tows
0.2% OTONS	93% OTONS	7% OTONS	7% OTONS
4% Sponge	82% of Sponge	14% of Sponge	18% of Sponge
39% Coral/Bryozoans	50% of Coral/Bryozoans	11% of Coral/Bryozoans	50% of Coral/Bryozoans

AGENDA C-4
Supplemental
APRIL 2003

OCEANA

The Ocean Conservancy

March 7, 2003

Delivered via First Class Mail and Facsimile (907) 586-7249

RECEIVED
MAR 10 2003
N.P.F.M.C

Dr. James Balsiger
Regional Administrator
Alaska Region
National Marine Fisheries Service
709 W. 9th St.
Juneau, AK 99802-1668

RE: North Pacific EFH EIS

Dear Dr. Balsiger:

Thank you for the recent opportunity to meet and discuss the progress that the agency and the North Pacific Fishery Management Council (NPFMC) have made on the Essential Fish Habitat Environmental Impact Statement (EFH EIS). During our meeting, we noted that the inclusion of alternatives 5B and 6 is viewed as a positive step forward and we commit to continue to work with you to maximize habitat protection in Alaska while maintaining vibrant fisheries. You noted that the agency is requesting an additional 12 months to complete a quality EIS. You asked for our concerns and "ideas."

As we discussed, National Marine Fisheries Service (NMFS) information and data is imperative to a quality document and must be brought forward and used to develop a scientifically based reasonable range of differentiated alternatives. It is our understanding that the NMFS Headquarters has given this explicit direction. We all agree that additional time alone will not produce a viable EIS. Since it is in everyone's best interest to achieve a successful EIS and the best habitat protection possible, we offer this summary of our discussions. We hope and request to meet with you again before the next NPFMC meeting to discuss the information and data and the applications for the formation of alternatives.

Science-Based Approach and the Formulation of Alternatives

Throughout the stakeholder negotiations at the EFH Committee, we have attempted to convince the agency and the NPFMC to undertake the formulation of alternatives with a science-based approach. While we appreciate the work undertaken on the Fujioka/Rose model, we are concerned that scientific input ceased before the formulation of the alternatives. As we discussed in February, the agency has data from which to structure mitigation alternatives that maximize habitat protection at any chosen level of economic and social practicability, but has not yet used those data to develop reasonable alternatives.

We would also like to commend the agency for developing research closures to be included in the EFH EIS. These will provide critical information on recovery time, effects of fishing and expected other crucial information to the future management of EFH. As we have agreed, all mitigation measures should include a comprehensive research, monitoring, and mapping program to provide better information about benthic habitats and their importance to commercial fish species.

At the December and February meetings of the NPFMC, the Science and Statistical Committee (SCC) raised many questions concerning the NPFMC's formulation of the alternatives. At the December meeting, the SCC stated that they found the alternatives difficult to evaluate because there was no statement of goals or objectives of the mitigation effort.[1]

This was repeated in the February minutes, where the SSC also commented on the piecemeal nature of the alternatives and questioned the combinations of tools that were under consideration, the scientific validity of the Bering Sea rotational closure approach, and the use of closures in areas of low to zero fishing intensity.[2] The SSC concluded that the proper approach to EFH would be to develop clear criteria for the designation of EFH and then apply a combination of protective measures recommended by the National Research Council (NRC) in its report on the effects of bottom trawling.[3]. These protective measures should have clear goals and objectives and specific management tools to meet these goals and objectives.

The SSC concerns speak to the very heart of the mitigation alternatives on the table and the need to take a second look at the management options that the NPFMC has directed the agency to evaluate. The SSC has indicated, and we agree, that the structure, composition and charge of the EFH Committee have led to alternatives that do not represent a series of rational and distinct approaches to minimizing the adverse effects of fishing on essential fish habitat. The agency must augment the current alternatives with a thorough science review during the EFH process in order to meet its statutory charge.

The Bering Sea

As we discussed with you, from a geographical perspective, the biggest hole in the range of mitigation alternatives is the number and efficacy of alternatives that apply to the Bering Sea. The EFH Committee was only able to come up with one approach to protecting the Bering Sea. This approach is repeated in Alternatives 4 and 5. Mitigation in these alternatives (1) is premised on the questionable approach of rotational closures in backdoor fishing grounds, (2) does not mitigate the pelagic pollock fishery, which the Rose/Fujioka model presented by the agency identified as having the highest impact on benthic habitat in the Bering Sea, and (3) does not focus on protecting the slope, the "greenbelt" of the Bering Sea with the greatest diversity, highest habitat complexity, and longest recovery time.

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[1] Draft Feb 2003 SSC Minutes, pg. 4.

[2] Id. at 5-6.

[3] National Research Council (2002). *Effects of Trawling & Dredging on Seafloor Habitat.* National Academy Press, Washington, D.C.

The SSC stated that the EFH Committee approach, "[w]hile disruptive to fishing, will provide little to no meaningful benefit to [Fishery Management Plan] FMP species and their habitats".[4] While the SSC ultimately recommended 'much longer' rotational closures for that approach, we share their initial concerns about the efficacy of rotational closures, which may actually cause more damage than they prevent.

The only other approach to habitat protection in the Bering Sea is Alternative 6, which is premised upon the scientifically defensible approach of mitigating impacts to habitat by closing 20 percent of the fishable area to fishing gear that causes impacts to habitat. As we all agreed at our recent meeting, much of the value of this approach was lost in the transition from the closure concept to specific closed areas. We all noted that the closures in Alternative 6, as presented to the NPFMC by the agency, were presented without any public input and are not effective in its current form because it fails to incorporate the criteria proffered by The Ocean Conservancy and thus disproportionately impacts gear types and communities that do not cause the majority of impacts.

Given the fundamental defects with the two approaches to mitigation in the Bering Sea, we reiterated specific suggestions for a methodology (generally ideas of NMFS and other scientists) that could be used to develop a reasonable mitigation alternative for the Bering Sea. The objective of this approach is to mitigate the adverse impacts on EFH for FMP species associated with complex benthic structure, vulnerable habitat, and/or high productivity by enacting trawl closures based on bathymetric features and concentrations of benthic invertebrate indicators while minimizing impacts to total catch.

Based on NMFS professional presentations and abilities, we believe you could develop a reasonable alternative with the criteria we discussed:

- No expansion of trawl fisheries to new areas.
- Close the Bering Sea slope (>200m depth) to trawling except in locations of high target species catch per unit effort (CPUE) and low bycatch of benthic invertebrates.
- Close areas to trawling on the Bering Sea shelf with high concentration of benthic invertebrates including those discussed in Malecha et al (2003) and low relative historical trawl effort.[5]
- Close areas to trawling where there is a high ratio of coral and sponge bycatch to CPUE (Alt. 5B methodology) and areas of highest total benthic habitat bycatch.
- Maintain subsistence and local community fisheries.
- Allow conversion in closed areas from trawl to fixed gear.
- Implement the research, mapping and monitoring components of Alternative 5B.

This approach would be based upon the rich data sets that exist for the Bering Sea, including the recent paper by Malecha et al. (2003).

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[4] Id., pg. 6.

[5] Malecha, P., Stone, R., and J. Heifetz. (2003). *Living Substrate in Alaska: Distribution, Abundance, and Species Associations*. Submitted for publication in Proceedings of the Benthic Habitat Symposium, Tampa Florida

In addition, it is our understanding that you have already developed a similar habitat protection alternative for the Bering Sea and other regions in the January 2001 Draft Programmatic Supplemental EIS under Alternative 5, entitled "Policy to Increase Protection to Habitat". We encourage you to bring this forward so we can discuss it at our next meeting. With these different and more scientifically-defensible mitigation options available, the agency and the NPFMC could formulate additional alternatives, driven by the available data.

At this time, we also reiterate our request for the available data so that we may continue to be constructive in our participation. Specifically, we discussed and request again the following:

- Trawl survey data detailing the distribution of invertebrates on the Bering Sea slope and shelf.
- Trawl CPUE of target species for the Bering Sea since 1990 queried by 5 x 5 km blocks.
- The ratio of coral and sponge bycatch to total catch for Alaska's Exclusive Economic Zone (EEZ), queried by 5 x 5 km blocks.

Gulf of Alaska

We are similarly concerned with the range and efficacy of approaches in the Gulf of Alaska. Mitigation seems to be focused on rockfish bottom trawling in areas where there is little to no fishing intensity. It is difficult to understand how closing areas without impacts will meet the Magnuson-Stevens Fishery Conservation and Management Act's mandate to mitigate the impacts of fishing on essential fish habitat. Furthermore, all alternatives would allow for pelagic trawling in any closed areas. As we discussed, pelagic nets are in contact with the bottom at least 44-70 percent of the time. WE could all agree pelagic trawling is therefore an unacceptable alternative and provides little mitigation, especially in areas of high coral or sponge abundance. A combination of gear definitions, performance criteria (e.g., limits on benthos in catch), and/or gear monitoring would be needed to ensure that all practices allowed as pelagic trawling are indeed not contacting the seafloor.

Habitat Areas of Particular Concern

We are very concerned that the identification and protection of Habitat Areas of Particular Concern (HAPCs) are not being considered in the EFH EIS. HAPCs are the most important EFH to protect. It is our understanding that the agency and the NPFMC are presently designing a process by which to designate HAPCs. In the June 2002 EFH Motion, the NPFMC indicated that this process to be completed prior to the April 2003 meeting. We look forward to participating in this process and attempting to protect extremely rare and important habitats from the effects of fishing. We understand that the HAPC process will be on a parallel track with the EFH process and will implement Dr. Hogarth's indication that HAPCs such as sponges and corals will receive heightened protection.

Conclusion

Since all stakeholders in the EFH process want to see the protection of habitat and the maintenance of vibrant Alaskan fisheries, we look forward to continuing to work with the agency and the NPFMC to maximize habitat protection and minimize the economic cost of such protection. The key to this approach is ensuring that there are clear goals and objectives and scientifically valid strategies for meeting these goals and objectives in the most effective manner. As noted by the Science and Statistical Committee, the EFH process has not operated from this premise and suffers from the above enumerated problems.

We are assuming, however, that this is an iterative process and we will see continued scientific and stakeholder input that will lead to more comprehensive science-based alternatives. By utilizing all the available information, science data and working together, we can develop higher quality EIS. We look forward to meeting with you again before the April 2003 NPFMC meeting and working with you to meet our data requests.

Sincerely,

Jim Ayers
Oceana

Kris Balliet
The Ocean Conservancy

cc: David Benton, Chairman
North Pacific Fishery Management Council
605 W. 4th Ave.
Anchorage, AK 99501
Fax: 907-586-4675

ARCTIC STORM, INC.
400 North 34th Street, Suite 306
Seattle, Washington 98103 U.S.A.

March 25, 2003

David Benton, Chairman
North Pacific Fishery Management Council
605 West 4th Ave. Ste. 306
Anchorage, AK
FAX: 907-271-2817

Dear Mr. Chairman:

Arctic Storm has read the letter from Oceana and the Ocean Conservancy addressed to Dr. Balsiger and is writing to express its serious concerns regarding the EFH EIS process. The above referenced letter, dated March 7, allegedly recounts agreements between the agency and the plaintiffs regarding several issues in the development of the EFH EIS. Additionally, the plaintiffs continue to press for additional EFH alternatives beyond those they raised during the public process, knowing full well that this will force the agency into non-compliance with the court ordered EIS completion deadline.

Since consideration of the plaintiff's latest round of new alternatives is scheduled for the April Council meeting, Arctic Storm would like to comment on the faulty rationale employed by the plaintiffs in 1) expanding the 5b approach to the Bering Sea and GOA, 2) requesting modification of the MPA alternative, and 3) addition of the habitat protection alternative in the old PSEIS. Arctic Storm also wishes to provide constructive comment on why the range of alternatives is adequate and why the agency should be asked to comply with the court order.

A review of how the existing range of EFH alternatives was developed:

As you know, the North Pacific Council, at your direction, embarked on an aggressive and thorough process to develop EFH mitigation alternatives. The EFH Committee, which included a number of representatives from the environmental community, met more than a dozen times to craft detailed alternatives. During the course of its meetings, the EFH Committee reviewed a model developed by Dr. Craig Rose and Dr. Jeff Fujioka of the Alaska Fisheries Science Center (The Rose/Fujioka model). This model is a state-of-the- art, quantitative model used to estimate impact to habitat caused by various fisheries. It is based on the best scientific information available. In developing this model, Drs. Rose and Fujioka reviewed existing published literature on impacts caused to benthic habitat by various fishing gears, known and estimated types of substrate and recovery times. It incorporated this information into a model that used observed catch and distribution data to quantify possible fishery impacts on habitat.

The Rose/Fujioka model was specifically developed to be used by the EFH Committee and Council to craft alternatives that might mitigate possible impacts caused by the highest impact fisheries. Based on this concept, the Committee developed alternatives that sought to mitigate the rockfish fishery in the GOA which was estimated to have an impact of about 9%, and the Bering Sea bottom-trawl fisheries, thought to have a combined impact of about 5%. Alternatives were developed for the AI primarily to protect areas of known coral and sponge abundance, a substrate type in the Aleutian Islands for which some EFH committee members felt additional mitigation alternatives might be needed.

The plaintiffs in EFH litigation, Oceana and the Ocean Conservancy, participated in those meetings and put forward alternatives that were fully considered by the Committee by not adopted because they were not as responsive to the Rose/Fujioka impact model as other alternatives. Nevertheless, in response to concerns raised by the plaintiffs, the Council added an additional alternative for the BSAI and GOA that sought to protect areas of high relief bioshelter where gorgonian coral, sponges and sea onions were abundant.

When the Council took final action in October 2002 in its selection of EFH alternatives the scheduled completion of the EFH EIS was April 2003, in time to comply with the court order. However, after meeting with the plaintiffs, the agency advised the Council at the December meeting that unless it adopted an additional, MPA alternative, it might not have an adequate range of alternatives. During the meeting, the agency revealed its MPA alternative and the Council adopted it as Alternative 6 in place of the high relief bioshelter alternative it had crafted during the October meeting. The Council further included an additional alternative, 5b, put forward by Oceana that would close all areas in the Aleutian Islands except those identified as "open areas." The rationale provided by Oceana in support of this alternative was the "patchy," unknown distribution of fragile coral areas in the AI that could be disturbed if fishing occurred in new areas or areas where bycatch of coral was high. During their presentation, the plaintiffs made no mention of a 5b, "open area" approach to be expanded to the Bering Sea and the Gulf of Alaska.

At the time, the agency supported including the new MPA approach supported by the Ocean Conservancy (Alt. 6) and Oceana's AI Alternative (5b). The Council did not learn until the February Council meeting that their inclusion of the plaintiff's alternatives would delay completion of the EIS and jeopardize compliance with the court-ordered deadline. Now we understand that the agency and the plaintiffs are engaged in discussions which may further undermine the public process of the Council by again looking at yet more alternatives, while at the same time setting the stage for failure to meet the Court's deadline.

Now that the agency is engaged in negotiations with the plaintiffs regarding the 12-month extension caused by the inclusion of their alternatives, we are disappointed to learn that the plaintiffs may not be acting in good faith. In addition to their two new alternatives added to the EIS package in December, they want three more alternatives, modification of an existing alternative as well as an outside review of the alternatives. Specifically, they want the 5b, "open area" approach taken in the Aleutian Islands expanded to the Bering Sea and the Gulf of Alaska, the inclusion of the habitat protection alternative from the old PSEIS and overhaul of their own MPA alternative.

Arctic Storm wonders why the Council is again being asked to reconsider its range of alternatives outside the process established for other stakeholders, especially when it further jeopardizes compliance with the court order. We are also surprised and dismayed to learn that, according to the plaintiff's letter, the agency has agreed to bring forward new information and data to develop additional alternatives "that maximize habitat protection at any chosen level of social and economic practicability." We wonder when this process will end and fear that unless a preliminary draft EIS is completed by the August deadline, the plaintiffs will continue to manipulate this process at the expense of the public and those dependent on the resource.

<u>Expansion of 5b approach to BSAI and GOA is inappropriate and unjustified.</u>

When Oceana argued for an "open area" approach in the AI, it based the weight of its justification on the "patchy," unknown distribution of fragile corals in that region and the high degree to which fishing occurs in the same areas each year in the Aleutian Islands. They argued that by restricting the fleets to areas where they had historically fished, no new coral areas would be disturbed. This was a precautionary approach based on the unknown distribution of coral in the AI and the predictable location of target species concentrations in that region.

None of these conditions are present in the Bering Sea and GOA where the bottom is largely mud and sand with a high degree of annual fluctuation in temperature and food abundance caused by changes in currents. The fish are highly migratory causing fleet dispersion to dramatically change on a seasonal and annual basis. The stationary concept of the "open area" approach is therefore, entirely inappropriate and contrary to the ecosystems of the Bering Sea and Gulf of Alaska. Additionally, long-lived and fragile invertebrates such as coral are not abundant in the sand-wave dominated substrates of the relatively shallow and flat Bering Sea shelf.

The first problem here is that in spite of the scientific advice of NMFS scientists, the plaintiffs repeatedly attempt to characterize the Bering Sea and GOA shelf as area where corals are abundant. As NMFS has explained, observers have only minimal training in classification of invertebrates and therefore observer data does not distinguish between short-lived bryozoans and true corals. Secondly, closure of a multitude of these small areas sprinkled throughout the limited, remaining Bering Sea and GOA "open areas" seems unreasonable, unenforceable and impracticable.

Finally, the plaintiffs' new proposals fail to take into account the extensive closures already in place in the BSAI and GOA that encompass large areas of the type of habitat they profess to want to protect. Specifically, the claims made by the plaintiffs that the BS and GOA have only one mitigating alternative is simply untrue. In addition to the suboptions and proposed gear modifications, both regions are subject to the extensive mitigating measures under the current management regime as described in Alternative 1 which most conspicuously include:

- Trawl closures totaling 30,000 sq. nm. in the BS and 72,000 sq. nm. in the GOA or about 25% of the continental shelf.
- The OY cap of 2 million mt lessens potential impacts on habitat by reducing potential TAC amounts well below the overall ABC. In 2003, the cap reduced the amount of potential harvest by approximately one-third.
- The SSL seasonal and area restrictions imposed in the pollock, cod and Atka mackerel fisheries reduces impacts to habitat through the establishment of more than sixty no trawl zones for those fisheries and dozens of no entry zones for all fishing vessels out to three miles.

A goal of NMFS and the Council is to provide sound conservation of living marine resources, while also providing socially and economically sustainable fisheries. Because this alternative would run counter to this goal, it does not seem a reasonable or "practicable" alternative to minimize any adverse effects of fishing and so should not be included in the range of alternative.

Modification of Alternative 6, MPA approach

Arctic Storm was disappointed to learn from the Oceana letter that both the plaintiffs and the agency agreed in a recent meeting "that much if the value of this approach was lost in the transition from the closure concept to specific closed areas." The letter further notes "that the closures in Alternative 6, as presented to the NPFMC by the agency, were presented without any public input and are not effective in its current form because it fails to incorporate the criteria proffered by the Ocean Conservancy and thus disproportionately impacts gear types and communities that do not cause the majority of impacts."

We are surprised that the weakness of this alternative has belatedly been recognized by the plaintiffs. Arctic Storm was among many stakeholder participants who voiced concern over the lack of public input in the development of this radical alternative, but we reluctantly supported its inclusion because the agency advised the Council that the plaintiffs strongly supported it and the agency believed its inclusion would "bullet proof" NEPA requirements for an adequate range of alternatives.

The plaintiffs suggested in public testimony at the time that adoption of such an alternative might be difficult because of the lack of public input but that, nonetheless, the presence of a marine reserve alternative that would close areas to all commercial fishing would provide a vehicle for the analysis to "compare and contrast" alternatives and better enable the Council to take a "hard look" at the impacts of varying alternatives.

Now the plaintiffs are revising their rationale for inclusion of this alternative and requesting that it be overhauled and limited to trawl closures to meet their new specifications which include:

- No expansion of trawl fishing to new areas
- Closure of most of the Bering Sea slope to trawling
- Closure of areas to trawling where bycatch of coral and sponges may occur.
- Convert trawl gear to fixed gear

An earlier analysis of the DPSEIS 2001 Alt.5, which is very similar to the one now put forward by the plaintiffs as their new MPA alternative, showed re-allocation of TAC from fixed gear to trawl gear may not provide the benefits envisioned. The analysis showed that the bycatch rate of some emergent epifauna (e.g. coral, anemones, sea whips, sponges) in fixed gear was similar to trawl gear for some fisheries, so reallocation of TAC would not result in large reductions of habitat impacts.

Arctic Storm is a strong supporter of conservation measures that protect the sustainability of our fishery resource, including the possible use of MPAs when scientifically justified. However, we are dismayed that the plaintiffs have chosen to manipulate this EFH process to include an unvetted MPA alternative that will not measure-up to the National Standards and has likely discredited to the public use of MPAs as potential, future management measures. This alternative should remain unchanged or, alternatively, it should be eliminated based on the rationale the plaintiffs provided in their letter which describes it as having lost its value. However, elimination of this alternative would require that a record be built that this alternative has been considered but rejected based on the lack of support by the plaintiffs and that the remaining range of alternatives is adequate

Pelagic gear targeted for additional closures

In their letter the plaintiffs complain that pelagic gear is insufficiently targeted in the existing alternatives. This is simply untrue. Pelagic gear is targeted in both Alternative 6 and in Alternative 1, the current management regime which prohibits all trawling in large areas of the Bering Sea. Unlike many environmental organizations, fishery managers and scientific literature, the plaintiffs fail to recognize pelagic gear as a mitigation of bottom trawl gear and the Bering Sea pollock fishery as one of the true success stories in fisheries management.

Based on their remarks on the need to further mitigate pelagic gear, it appears the plaintiffs do not understand how or where pelagic gear is used nor the implications of the pelagic pollock footprint as quantified in the Rose/Fujioka model. Such gear cannot be fished in contact with high relief structures such as coral and rocky bottom without sustaining severe and costly damage. Its occasional contact with the sand and mud bottom is fundamentally different than bottom trawl gear. Pelagic gear is designed with mid-water doors that never touch bottom and without protective bobbins, cookies or roller gear attached to the footrope because its intent is to stay beneath diving fish rather than scoop fish from the bottom.

The Bering Sea pollock fishery, with a bycatch rate of less than 1% of all species in the ocean, has been documented as one of the cleanest fisheries in the world. The Rose/Fujioka model estimated that its "foot print" may impact 2.3% of Bering Sea habitat. That means that nearly 98% of the Bering Sea habitat is unimpacted by a fishery that harvests 75% of the entire Bering Sea quota. For purposes of comparison, the estimated 2.3% impact of the pollock fishery is only slightly more than the estimated impact of the flathead sole fishery (2.1%) which is 1/100th the size of the pollock fishery. Importantly, it should not be forgotten that the SQ alternative provides significant mitigation to both pelagic and bottom trawl impacts with existing closures and the OY cap.

For the above described reasons, the proposal to expand alternatives to increase pelagic closures above those already included is unreasonable in that it is counter to the conservation, economic and social goals of the Council and provides no discernable net benefits to habitat.

Inclusion of PSEIS Habitat Protection Alternative is Unreasonable

Arctic Storm is struck by the plaintiff's audacious request to include the old PSEIS Alternative 5 in the EFH suite of alternatives even though they admit in their letter they don't know what elements it contains. That alternative, as described in the 2001 Draft PSEIS, includes the following elements:

- Make P.cod and Greenland turbot fisheries fixed-gear only
- Require that rockfish be taken only with pelagic and fixed gear
- Reduce grounds and TACs for flatfish and Atka mackerel fisheries
- Close HAPC gorgonian coral areas to all fishing

Arctic Storm reviewed this alternative two years ago and supported comments by the Marine Conservation Alliance that found this alternative unsatisfactory because, 1) the general conclusions about bottom trawling impacts were not supported by scientific evidence, 2) it was highly allocative, 3) it may not result in any demonstrable net benefits to habitat because conversion to fixed gear does not reduce the rate of epifauna bycatch in some fisheries and, 4) the current management regime better accomplishes the objectives of the stated policy to protect habitat while providing for sustainable groundfish fisheries.

No new alternatives are required

When the North Pacific Council drafted its first round of EFH mitigation alternatives in 1998, it's important to recall that the range of alternatives included only Status Quo based on the scientific information available at the time. The judge found fault with the agency for development of an EA instead of an EIS in its determination of a Finding of No Significant Impact (FONSI). In the development of the Rose/Fujioka model, the agency dramatically increased the scientific information available on impacts to habitat and, in crafting several alternatives that go far beyond Status Quo in proposing additional closures, the Council seems assured of moving forward with an adequate range of alternatives without including the latest round of alternatives proposed by the plaintiffs.

In contrast to the large closed areas developed for the North Pacific region, the New England Council developed nine EFH alternatives that, for the most part, are variations on Status Quo. Most of the alternatives are based on roller gear modification or redesignation of existing closures far smaller than those that exist under the current management regime in the North Pacific. Only one alternative proposed additional closure in areas currently fished. The New England Council recently chose its preferred alternative and its EIS is scheduled for public review next week.

Compliance with the Court Order:

As demonstrated in their letter, the plaintiffs are trying to mimic here what they did when threatening further litigation on the PSEIS. We appreciate that the agency has committed to support Council authority in the selection of alternatives. Nonetheless, the agency and the Council have been put in a difficult position because the plaintiffs have failed to act in good faith. Having secured the inclusion of their additional alternatives at the December meeting, they now refuse to support extension of the deadline. Based on their letter to Dr. Balsiger, we can only surmise plaintiffs will agree to the extension only if the Council agrees to add three more alternatives, fulfill their data requests, secure outside review of the alternatives and revise the Rose/Fujioka model to their specifications.

There are three ways to get out from underneath this untenable position:

1) Agree to the plaintiff's demands,
2) Secure a deadline extension from the judge by explaining that the need for the extension is to accommodate additional analysis required by the inclusion of two alternatives proposed by the plaintiffs, and or
3) Comply with the court order.

Arctic Storm supports options two and three. Specifically, the agency should be asked to give highest priority to completion of a *preliminary* Draft EFH with a target completion date that will comply with the court order. At the same time the agency should file papers with the court seeking an extension for completion of the final EIS.

The agency has already worked on the EFH EIS for several months and, based on its strong performance record in the development of court-ordered EISs, should be able to complete a preliminary analysis by August. The agency has voiced some concern about being able to complete a sufficiently quantitative analysis in that time. However, a *preliminary* draft could be developed using a balance of qualitative and quantitative data to be supplemented later when the final EIS is drafted.

Because most of the fisheries in the North Pacific are monitored with observer coverage, the preliminary draft will have more quantitative catch, bycatch and distribution data than any other region developing an EFH EIS. That important data has been incorporated into two state-of-the-art models to be used in this analysis: 1) the Rose/Fujioka model used to determine the impact of fisheries on habitat and 2) the Multi-Species Model used to determine the anticipated impact of each alternative on habitat, catch and bycatch as well as the human environment which includes socio-economic impacts.

The development of a preliminary draft EIS will help the public and the Council better understand these complex alternatives. And finally, the judge can be provided with a preliminary EIS in August that demonstrates the high priority the agency and Council have given this task in a good faith effort to meet the deadline.

In closing Arctic Storm recognizes and supports the role of groups like Oceana and the Ocean Conservancy in the Council process. Their participation has raised important issues for public debate and analysis. We urge the Council to request these groups to demonstrate their respect for this public process by supporting extension of the EIS deadline and the current range of alternatives.

In sum, Arctic Storm urges the Council to ask NMFS to comply with the court-ordered deadline and to support the current range of alternatives.

Sincerely,

Donna Parker

Donna Parker

AGENDA C-4
APRIL 2003
Supplemental

March 27, 2003

Approximately 700 of these comments were received by fax - originals are on file at Council office

Dr. James Balsiger
NMFS Regional Administrator
fax number: (907) 586-7557

Mr. David Benton
Chairman of North Pacific Fishery Management Council
fax number: (907) 271-2817

Dear Chairman Benton,

I am concerned...

I am concerned about the destruction of the fragile coral and sponge in the Bering Sea and North Pacific. These living coral and sponge habitats are essential nursery areas for fish. Only recently documented by science to be important to commercial fish and shellfish, these beautiful cold water corals and sponges are being demolished by destructive fishing practices. Bottom trawling is destroying these rainforests of the sea. Corals, which live for hundreds and even thousands of years, have incredibly slow growth rates and are particularly sensitive to disturbance. As these underwater forests disappear, we are losing an international treasure that won't come back for centuries. We need to keep the world the way God intended.

I am writing to voice my support for cold water coral protections in Alaska. It is important to protect these coral habitats while maintaining vibrant fisheries. I support the approach taken in Alternative 5B for the Aleutian Islands and request that the Council and NMFS apply a similar approach to the Bering Sea and Gulf of Alaska. For the Bering Sea, this should include substantial protections for corals, sponges, and other important seafloor invertebrates as well as productive areas of the continental slope.

Thank you.

Sincerely,

Kathy Long
2215 n park rd
spokane, WA 99212

UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
P.O. Box 21668
Juneau, Alaska 99802-1668

AGENDA C-4
April 2003
Supplemental

March 26, 2003

Jim Ayers
Director, North Pacific
Oceana
175 South Franklin Street, Suite 418
Juneau, AK 99801

Kris Balliet
Director, Alaska Regional Office
The Ocean Conservancy
425 G Street, Suite 400
Anchorage, AK 99501

Dear Mr. Ayers and Ms. Balliet:

Thank you for your recent letter regarding the Essential Fish Habitat (EFH) Environmental Impact Statement (EIS), and for taking the time to meet again with me and my staff as well as the North Pacific Council Executive Director to discuss the EIS alternatives and time line. I share your frustration with the process of developing a comprehensive suite of alternatives under difficult time constraints.

The National Marine Fisheries Service (NMFS) and the Council are continuing to work toward addressing the concerns you have raised. At the upcoming Council meeting, NMFS and Council staff will present the current draft of Chapter 2 of the EIS, including a discussion of the objectives and rationale behind each of the alternatives and a summary of the alternatives considered but not carried forward for detailed analysis. Staff will be seeking input from the Scientific and Statistical Committee, Advisory Panel, Council, and stakeholders to ensure that they have captured the basis for the alternatives and the reasons for not pursuing other options. Staff will also present preliminary work on applying the methodology from Alternative 5b for the Aleutian Islands to the Bering Sea and Gulf of Alaska to determine whether that approach merits evaluation as an additional alternative. We have completed this preliminary staff work in response to your request that we evaluate this potential alternative, and we look forward to discussing through the Council process the pros and cons of adding another alternative.

Regarding the various data requests you have made over the past year, we are reviewing our records to determine whether we have provided you with all of the readily available information we ca release under existing confidentiality rules. We will contact you shortly to verify whether NMFS can make available additional data sets to fulfill your requests.

I appreciate your continued involvement in the EFH EIS process.

Sincerely,

James W. Balsiger
Administrator, Alaska Region

cc: David Benton, North Pacific Fishery Management Council
Records.fakr
Jon Kurland and Cindy Hartmann, HCD

AGENDA C-4
APRIL 2003
SUPPLEMENTAL

Essential Fish Habitat

Progress Report April 2003

Review of Materials Distributed
- EIS Table of Contents
- RIR Table of Contents
- Chapter 1 (Purpose and Need)
- Chapter 2 (Alternatives)
- Research and Monitoring Approaches
- Unpublished manuscript on living substrates (Melacha et al.)

Review of SSC Concerns from February 2002
- Conceptual approach to minimizing effects of fishing on EFH.
- Goals, origin, justification, tools, and objectives of minimization alternatives.
- Research plan.
- Analysis components.

Overview of Alternatives to Minimize the Effects of Fishing on EFH

Notes on Chapter 2

- Includes overview of previous actions (section 2.2.2), a review of development EIS alternatives (section 2.2.3 and Table 1), examples for EFH & HAPC designation alternatives, objectives of minimization alternatives, updated maps and text, and description of alternatives considered but not analyzed (section 2.4).
- Details added to Alternative 5B. Still need data to determine: 1) TAC reduction for each species, and 2) coral/bryozoan and sponge bycatch limits. (see p. 2-44)
- Alternative 6 clarified to include the longline halibut fishery. [Note that jigs, dinglebar, troll, gillnet, and all other legal gear types would be allowed within the reserve areas, and that subsistence and recreational fisheries would not be affected.]

SSC Concern #1: Conceptual Approach
pages 2-37 through 2-39

- A spreadsheet analysis showed that the scallop, salmon, and crab fisheries had much smaller footprints and habitat impacts than groundfish fisheries, so focus turned to groundfish fisheries.
- Alternatives 1-5 are based primarily on Rose model incorporating groundfish fishery spatial fishing intensity, sensitivity, and habitat recovery. The spreadsheet and Rose model comprise the evaluation required by the EFH final rule.
- No quantitative threshold has been established to decide what fishing activity adversely effects EFH in a manner that is more than minimal and not temporary.
- The Rose model ranked fisheries based on relative impacts to habitat. Fisheries with highest impacts addressed in all alternatives, lower impacts addressed in higher # alternatives. Measures for AI added in Alternatives 4 and 5 to address limitations of the draft model.
- Alternatives 5B and 6 were not directly based on the model results.

SSC Concern #2: Objectives of Alternatives
pages 2-39 through 2-46, table on 2-64

- Goal of all alternatives is to minimize adverse effects of fishing to the extent practicable (EFH regulations in section 1.5.6)
- Objectives and rationale have been listed for each alternative.
- Alternatives 2-5, developed by the EFH Committee, include area closures and gear modifications (Alts 4-5) designed to reduce the effects of specific fisheries on specific habitat features, and to allow some portion of the bottom to recover. The size of the closures and the number of fisheries included, and the relative amount of EFH conservation, increase with alternative #.
- Alternative 5B, proposed by Oceana, aims to reduce the effects on sessile epifauna in the Aleutian Islands.
- Alternative 6, proposed by The Ocean Conservancy, aims to eliminate all effects of all bottom fishing gear (dredges, trawls, longlines, and pots) on 20% of the seafloor.

SSC Concern #2: Tools used in Alternatives pages 2-38 through 2-46, table on 2-64-67

The NRC report "Effects of Trawling and Dredging on Seafloor Habitat" recommended 3 tools for managing effects of trawling on habitat: effort reduction, gear modifications, and closed areas.

- Effort reduction is not directly considered in most of the alternatives because trawl effort is relatively low off Alaska, and fishing effort is already directly controlled (through IFQs, CDQs, LLPs, rationalization programs) and indirectly controlled through OY cap, bycatch limits, and conservative TACs. Alternative 5B reduces effort via TAC reduction.
- Gear modifications are included in Alternatives 2-5: voluntary change from bottom trawl to fixed or pelagic trawl gear for GOA slope rockfish fisheries, and minimum roller size for footrope and sweeps on bottom trawls used in the Bering Sea.
- Closed areas are included in all alternatives. There are bottom trawl closures in Alternatives 2-5, and closures to all bottom tending gear in Alternative 6.

SSC Concern #3: Research Plan

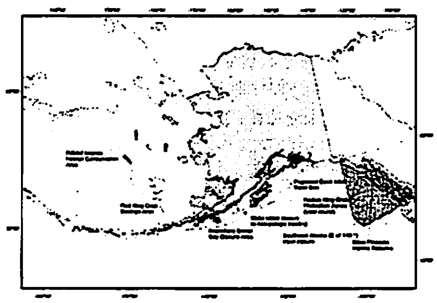
- A draft discussion outline of research and monitoring approaches was prepared by NMFS.
- The EIS will describe the overall goals and objective for research and monitoring for each alternative
- Once the Council selects a preferred alternative, staff will develop the necessary analysis to implement research and monitoring in a subsequent process (EA/RIR).

SSC Concern #4: Analytical Components

- The draft table of contents for the EIS and RIR list components of the analysis.
- SSC recommended that analysis should include: 1) ability of alternative to meet the stated objectives; 2) biological consequences of recolonization of invertebrates and fish; 3) economic and social costs by sector and community; and 4) enforceability.

Overview of Minimization Alternative 1

page 2-39



Origin: National and Council policy, FMP amendments, regulations

Objectives: Conserve, restore, and maintain habitat for fish productivity

Measures: gear restrictions, MPAs harvest limits, effort limits, rationalization programs, other regulations (reviewed on pages 2-2 through 2-8)

Overview of Minimization Alternative 2

page 2-39

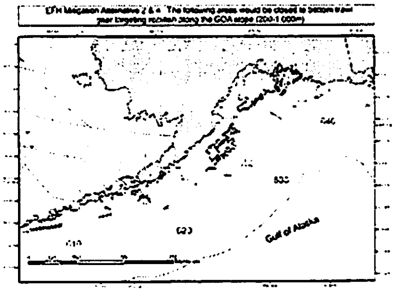
Origin: EFH Committee

Rationale:

- Addresses fishery with highest score from Rose model

Objectives:

- Allow some recovery of GOA slope
- Provide incentive for gear conversion
- Limit restrictions to reasonable
- measures
- Measures:
- Prohibit bottom trawling for rockfish in designated areas of GOA slope,
- Allow conversion from bottom trawl to pelagic trawl or fixed gear to fish for rockfish within these areas



Overview of Minimization Alternative 3

page 2-40

Origin: EFH Committee

Rationale:

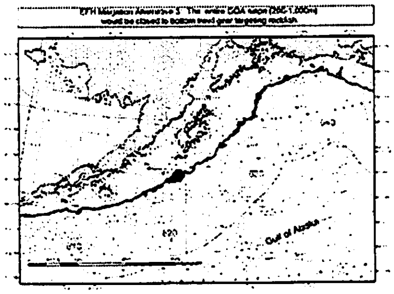
- Addresses fishery with highest score from Rose model; more productive

Objectives:

- Allow more recovery of GOA slope
- Provide incentive for gear conversion
- Limit restrictions to reasonable measures

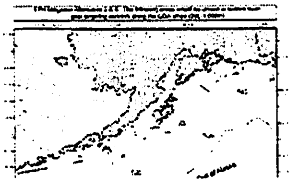
Measures:

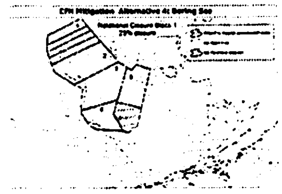
- Prohibit bottom trawling for rockfish on ALL areas of GOA slope,
- Allow conversion from bottom trawl to pelagic trawl or fixed gear to fish for rockfish on the slope

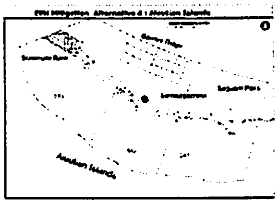


Overview of Minimization Alternative 4

page 2-40







Origin: EFH Committee

Rationale:

- Addresses fisheries with highest score from Rose model, plus protection for all areas

Objectives:

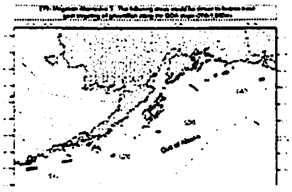
- Allow some recovery in areas of BS and AI shelf/slope, and GOA slope
- Reduce contact of gear on bottom (Bering Sea trawl disc requirement)
- Provide incentive for gear conversion (GOA rockfish fisheries)
- Limit restrictions to reasonable measures

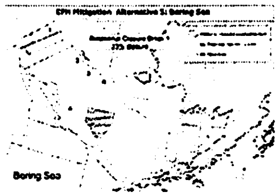
Measures:

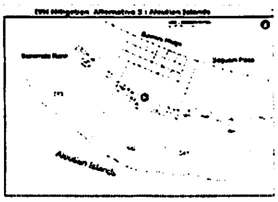
- 'Open' areas in for BS bottom trawl
- Bottom trawl closures in all areas
- Gear regulations for BS trawl
- Voluntary gear conversion (GOA slope)

Overview of Minimization Alternative 5A

page 2-42







Origin: EFH Committee

Rationale:

- Similar to Alternative 4 but larger areas, more protective

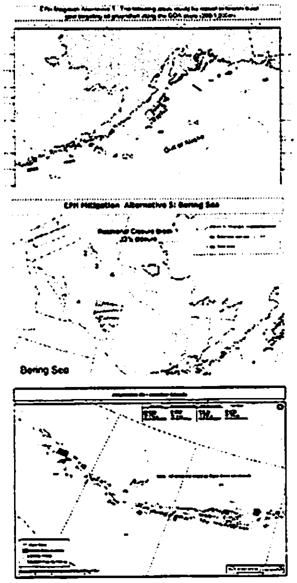
Objectives:

- Prevent expansion of trawl effort (BS)
- Allow more recovery in areas of BS and AI shelf/slope, and GOA slope
- Reduce contact of gear on bottom (Bering Sea trawl disc requirement)
- Provide incentive for gear conversion (GOA rockfish fisheries)
- Limit restrictions to reasonable measures

Measures:

- 'Open' areas for BS bottom trawl
- Bottom trawl closures in all areas (more extensive in BS and AI; more restrictive in GOA - all slope closed to rockfish bottom trawl & designated areas closed to all bottom trawl)
- Gear regulations for BS trawl
- Voluntary gear conversion (GOA slope)

Overview of Minimization Alternative 5B
page 2-43

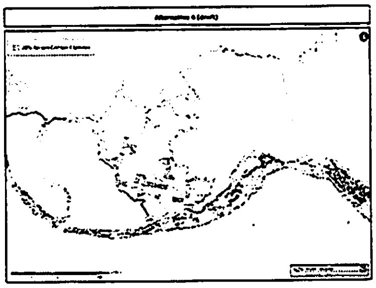


Origin: Oceana (AI portion); EFH Committee (BS, GOA)

Objectives:
- Prevent expansion of trawl effort (BS,AI)
- Allow more recovery in areas of BS and AI shelf/slope, and GOA slope
- Reduce contact of gear on bottom (Bering Sea trawl disc requirement)
- Provide incentive for gear conversion (GOA rockfish fisheries)
- Indirectly control effort in AI (via TAC reduction)
- Control/reduce bycatch of sessile invertebrates (AI)

Measures:
- 'Open' areas in BS and AI
- Bottom trawl closures in all areas
- AI only: TAC reduction; bycatch limits for sponges, corals, and bryozoans; VMS; mandatory research plan
- Gear regulations for BS trawl
- Voluntary gear conversion (GOA slope rockfish)

Overview of Minimization Alternative 6
page 2-45



Origin: The Ocean Conservancy/NMFS

Rationale:
- Addresses impacts from all fisheries

Objectives:
- Allow 20% of all shelf and slope areas to fully recover from any and all impacts due to fisheries

Measures:
- Prohibit commercial fisheries from using bottom tending gear in designated areas.
- Includes all status quo measures as well.

EFH EIS – Mitigation Alternative 5b

The Aleutians suboption has four components:

- No expansion of bottom trawl fisheries
- Close areas with high rates of bycatch and low rates of catch
- Area-specific bycatch limits
- Comprehensive research and monitoring plan

Also assumes 100% VMS and observer coverage

EFH EIS – Mitigation Alternative 5b

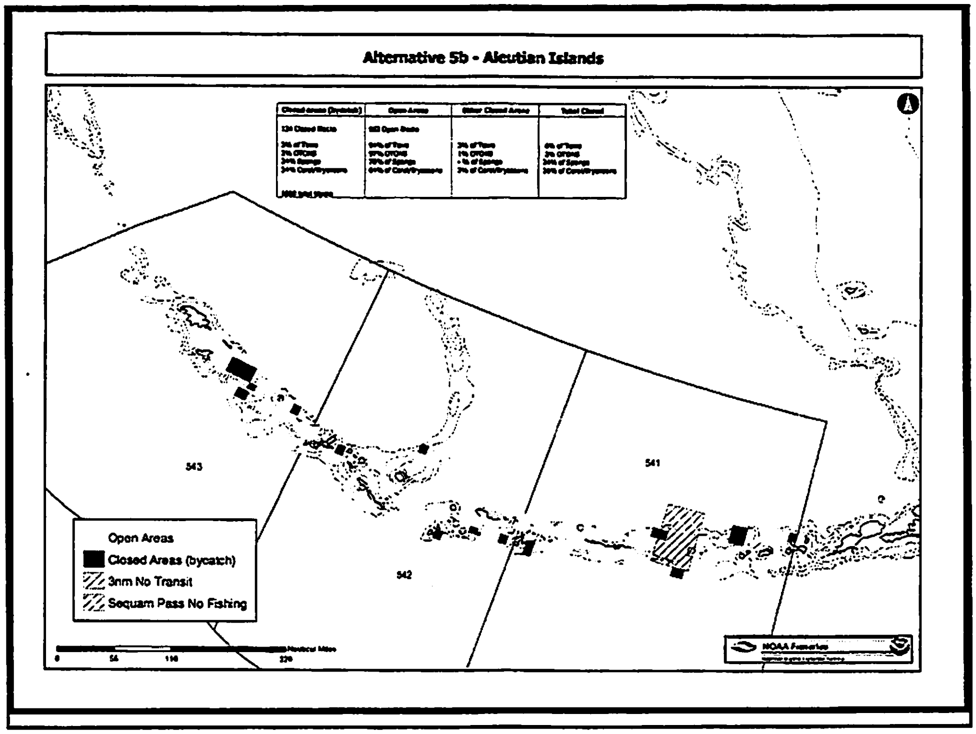
No expansion of bottom trawl fisheries (Open Areas)

- Based on effort during 1990-2001 (NORPAC)
- Summed points to 5k grids
- Three categories based on distribution
- Included all grids in the highest category
- Open areas include grids of high effort as well as low and none due to an attempt to square areas off.

EFH EIS – Mitigation Alternative 5b

Close areas with high rates of bycatch and low rates of catch (areas closed due to bycatch)

- Summed points to 5k grids
- Created ratio of bycatch CPUE to catch CPUE
- Displayed 5 categories
- Included all grids in highest two categories, adjacent blocks in third category
- Minimum size was 4 blocks. Closed areas include grids of high ratio as well as low and none due to an attempt to square areas off.



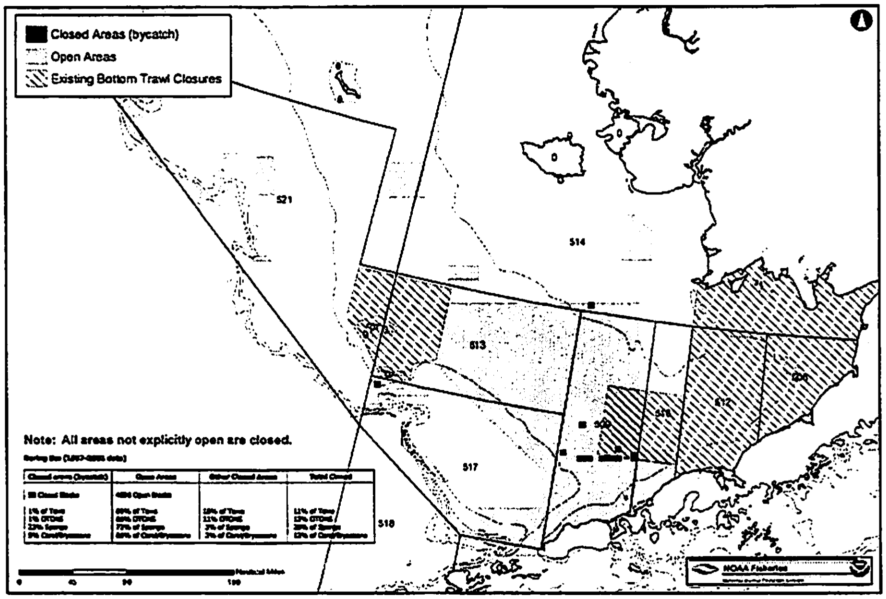
Alternative 5b - Aleutian Islands

EFH EIS – Mitigation Alternative 5b

Aleutians (1990-2001 data)

Open Areas	Closed areas (bycatch)	Other Closed Areas	Total Closed
94% of Tows 97% OTONS 76% of Sponge 64% of Coral/Bryozoans	3% of Tows 2% OTONS 24% Sponge 34% Coral/Bryozoans	3% of Tows 1% OTONS - % of Sponge 2% of Coral/Bryozoans	6% of Tows 3% OTONS 24% of Sponge 36% of Coral/Bryozoans

Alternative 5b - Bering Sea

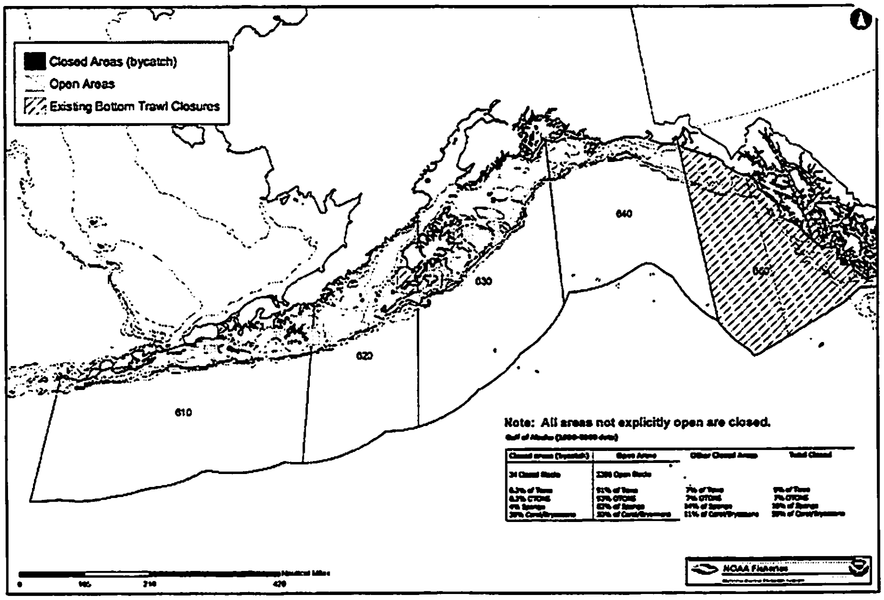


EFH EIS – Mitigation Alternative 5b

Bering Sea (1997-2001 data)

Closed areas (bycatch)	Open Areas	Other Closed Areas	Total Closed
1% of Tows 1% OTONS 25% Sponge 9% Coral/Bryozoans	89% of Tows 88% OTONS 72% of Sponge 88% of Coral/Bryozoans	10% of Tows 11% OTONS 3% of Sponge 3% of Coral/Bryozoans	11% of Tows 12% OTONS 28% of Sponge 12% of Coral/Bryozoans

Alternative 5b - Gulf of Alaska



EFH EIS – Mitigation Alternative 5b

Gulf of Alaska (1990-2000 data)

Closed areas (bycatch)	Open Areas	Other Closed Areas	Total Closed
0.2% of Tows 0.2% OTONS 4% Sponge 39% Coral/Bryozoans	91% of Tows 93% OTONS 82% of Sponge 50% of Coral/Bryozoans	7% of Tows 7% OTONS 14% of Sponge 11% of Coral/Bryozoans	9% of Tows 7% OTONS 18% of Sponge 50% of Coral/Bryozoans

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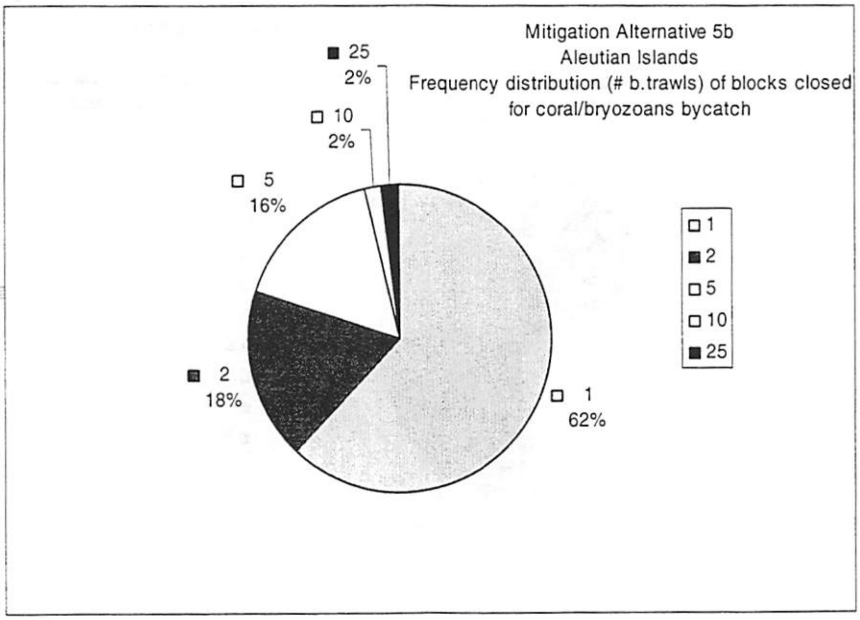
EFH EIS – Mitigation Alternative 5b

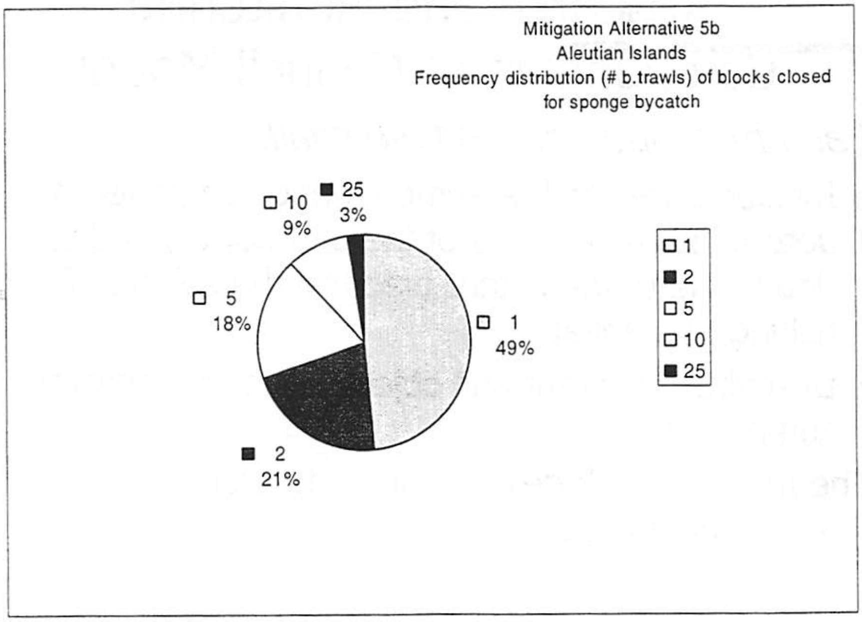
During analysis, trend was noticed that closed blocks with the highest CPUE tended to have a lower number to total hauls in the grid.

Initial frequency analysis of sponge and coral/bryozoan catch.

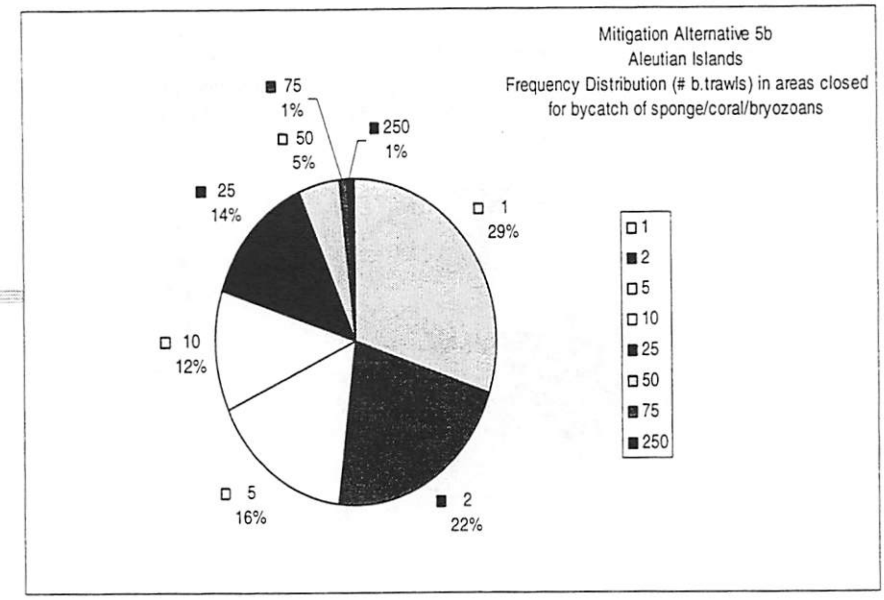
EFH EIS – Mitigation Alternative 5b



EFH EIS – Mitigation Alternative 5b



EFH EIS – Mitigation Alternative 5b



Research and Monitoring Components of Council Motion

Each mitigation alternative shall:

- Include a research and monitoring component to determine the efficacy of the alternative, and to determine to the extent practical the effects of fishing on habitat.
- Describe the intent and objectives of its research component

The final research design will be developed in a subsequent process

Discussion of Research and Monitoring in the EFH EIS

The EFH EIS WILL describe the overall goals and objectives for research and monitoring for each mitigation alternative.

The EFH EIS will NOT discuss different research areas (specific research closures) or specific experimental designs for each alternative.

Analyses to implement research and monitoring will occur in a subsequent process.

Research and Monitoring Components of Council Motion

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- Include a research and monitoring component to determine the efficacy of the alternative, and to determine to the extent practical the effects of fishing on habitat.
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Discussion of Research and Monitoring in the EFH EIS

The EFH EIS WILL describe the overall goals and objectives for research and monitoring for each mitigation alternative.

The EFH EIS will NOT discuss different research areas (specific research closures) or specific experimental designs for each alternative.

Analyses to implement research and monitoring will occur in a subsequent process.

This Subsequent Process Will:

- Develop a hypothesis driven research design for the preferred alternative
- Include public and Council input to help select research areas
- Evaluate options through an Environmental Assessment
- Have a Regulatory Impact Review and Regulatory Flexibility Act analysis of the socioeconomic impacts
- However, implementation will be contingent upon availability of funds

Preliminary Approaches
Discussion Outline of DRAFT Research and Monitoring Approaches

For Each Minimization Alternative This Outline Contains:

- Objective(s)
- General Research Question(s)
- Research Activities
- Research Time Frame

Alternative 1: Status Quo
General Research Questions

- Consideration of ecosystem health and the effect of fishing on EFH with focus on whether adverse impacts alter structure, function, and/or rates of ecosystem processes.
- Scientific assessments should address whether fishing activities reduce habitat suitability for marine resources and thus affect sustainable harvest levels.
- In particular, habitat-mediated effects on spawning, breeding, feeding, growth and shelter of FMP species should be examined.
- A two stage process that requires identification of specific effects attributable to fishing activities and interpretation of these effects to determine the positive/negative ecological implications.

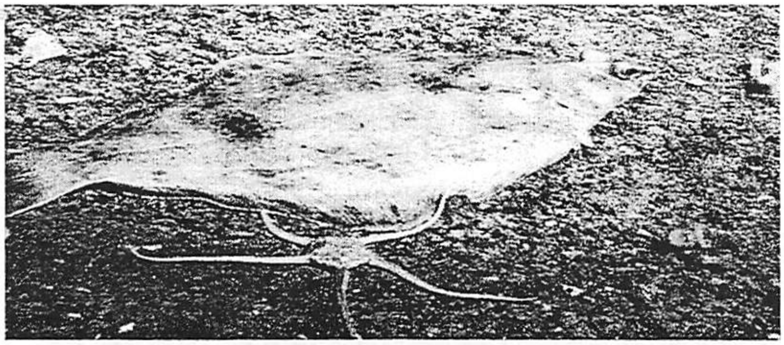
Alternative 1: Status Quo
Research Activities

Three experimental approaches are applicable:

- Compare conditions in heavily fished and lightly fished/unfished areas that are in close proximity and otherwise similar.
- Compare conditions before and after experimental fishing to identify short-term (acute) effects on the benthos.
- Determine rates of disturbance with repetitive fishing of specific grounds.

Alternative 1: Status Quo
Research Time Frame

Long Term: Until such time as more systematic methods are developed and implemented, and the overall level of research effort increases.



Alternatives 2 – 6
Objectives, Research Questions, and Research Activities

Two Components to Test the Efficacy of the Alternatives:

1. Are Impacts Reduced ?
2. Is Benthic Habitat Recovered ?

EXAMPLE

Alternative 2: Gulf Slope Bottom Trawl Closures: Prohibit the use of bottom trawls for rockfish in 11 designated areas of the GOA slope, but allow vessels endorsed for trawl gear to fish for rockfish in these areas with fixed gear or pelagic trawl gear.

Objectives to Reduce Impacts:

Restrict the higher impact trawl fisheries from a portion of the slope, thus encouraging a switch to fixed gear and pelagic trawls.

Objectives for Benthic Habitat Recovery:

Allow benthic habitat within these areas to recover or remain relatively undisturbed

Alternative 2 – Research Questions for Reduce Impacts

- Does the closure effectively restrict higher impact trawl fisheries from a portion of the GOA slope?
- Is there increased use of alternative gears in the closed areas?
- Does total bottom trawl effort in adjacent open areas increase as a result of effort displaced from closed areas?
- Do bottom trawls affect these benthic habitats more than the alternative gears?

Alternative 2 – Research Activities To Determine if Impacts are Reduced

- Use effort data to establish a baseline for comparison of fishing gear activity in the closed and open areas.
- Investigate experimentally, in a comparable and relatively undisturbed area, the relative effects of bottom trawl and alternative gears.
- Compare changes in the structure and function of benthic communities and populations, as well as important physical features of the seabed after comparable harvests.

Alternative 2 – Research Questions to Determine Benthic Habitat Recovery

- Did the habitat within these areas recover or remain unfished because of these closures?
- Do recovered habitats support more/healthier FMP fish?

Alternative 2 – Research Activities

To Determine Recovery of Benthic Habitat

- Monitor the structure and function of benthic communities and populations, as well as physical features of the seabed.
- Replicated biological sampling with grabs, trawls, and underwater ROV or submersible observations.
- Use acoustical surveys with multibeam, side scan, or single beam devices, coupled with grab and video groundtruthing to compare physical features.
- Assess the impacts of alternative gears while also monitoring recovery in areas that are unfished.

For Detailed information
See DRAFT "Discussion
Outline of Research and
Monitoring Approaches
For Evaluation of EFH
Mitigation Alternatives"
NMFS, March 20, 2003

PUBLIC TESTIMONY SIGN-UP SHEET FOR AGENDA ITEM [redacted] [redacted] C-4 EFH

PLEASE SIGN ON THE NEXT BLANK LINE.
LINES LEFT BLANK WILL BE DELETED.

	NAME	AFFILIATION
1.	~~JOHN GAUVIN~~	~~GROUNDFISH FORUM~~
2.	Josh Nowlis	Alaska Ocean Network
3.	Ben Enticknap	AMCC
4.	Geoff Shester	Oceana
5.	Whit Sheard	The Ocean Conservancy
6.	Jim Ayers	Individual
7.	[illegible]	Russian Federal
8.	Donna Parker	ACB
9.	Dave Fraser	Muir Milach
10.	Heather McCarty / ~~Frank Kelty~~ Ron Clarke	Marine Conservation Alliance
11.	Julie Bonney	AGDB
12.	GERRY MERRIGAN	PETERSBURG VESSEL OWNERS ASSOC.
13.	JANIS SEARLES	Earthjustice
14.	Paul MacGregor	At-Sea Processors
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BULLETIN OF MARINE SCIENCE, 66(3): 651–662, 2000

MOTE SYMPOSIUM INVITED PAPER

SHORT- AND LONG-TERM EFFECTS OF THREE FISHERY-MANAGEMENT TOOLS ON DEPLETED FISHERIES

Joshua Sladek Nowlis

ABSTRACT

Marine reserves have come under criticism because of the short-term fishery losses likely to be associated with them. At the same time, marine reserves have been touted as a tool to rehabilitate depleted populations, at least for species with relatively limited adult movement. I used models to compare the short- and long-term fisheries consequences of three fishery-management tools for depleted populations. These management tools included temporary closure of the entire fishery, minimum size limits, and marine reserves. I compared them using three short-term indices: magnitude of initial drop in catches relative to those prior to new management, years until catches reached prior levels, and cumulative loss during those years. I examined a single long-term index: long-term sustainable yields. Results highlighted the potential of reserves as an efficient and effective fishery-management tool for species that will remain within reserve boundaries. Reserves created few short-term losses beyond those associated with other management measures, yet produced the highest stable catch levels. Moreover, peak catches with reserves occurred with less restriction than peak catches with other management measures. These results were consistent across two species that matured before entering the fishery but did not apply to one species that was fished while immature. In that latter case, minimum size limits produced more substantial benefits than reserves could. Nevertheless, these analyses suggest that a wide range of circumstances exist where reserves, if properly designed to minimize adult spillover while allowing abundant larval transport, can maximize fisheries harvests with a minimum of total restrictions.

Fisheries on wild stocks are in decline worldwide. According to a United Nations report, two-thirds of all commercial fisheries are fished beyond capacity or in danger of becoming so (Food and Agriculture Organization, 1995). In the United States, one-third of assessed species are classified as overfished or approaching overfished. More disturbingly, we do not know the status of over 60% of all managed stocks in the United States (National Marine Fisheries Service, 1998). It is particularly disturbing that these failures have occurred in industrialized countries like the United States with resources for science, management, and enforcement. In countries lacking these resources, many traditional fishery-management tools are simply unavailable.

In response to this growing problem, fisheries managers are showing a willingness to consider a broader range of tools. These nontraditional tools include no-take marine reserves, areas where fishing is prohibited indefinitely. Closed-area management is not a new idea; theory and examples stretch back decades (e.g., Beverton and Holt, 1957). Field studies have shown repeatedly that the number and average size of fished species increase within these closed areas, as does the total number of species (Roberts and Polunin, 1991; Dugan and Davis, 1993; Rowley, 1994; Bohnsack, 1996; and references therein). These changes alone offer fishery benefits by protecting vulnerable species and serving as insurance against management failure. In addition, theory suggests that reserves, if properly designed, can increase overall fish catches despite the loss of fishing area and can stabilize annual fluctuations in catches (Sladek Nowlis and Roberts, 1999, and references therein). These theoretical benefits remain unproven in the field, although existing studies are suggestive (Alcala and Russ, 1990; McClanahan and Kaunda-Arara, 1996; Russ and Alcala, 1996).

Recently, reserves have received a great deal of attention, more than other forms of fishery management. The Plan Development Team (1990) qualitatively compared reserves to other management tools. The present study is an attempt to do so quantitatively. I assessed three management tools: marine reserves (protection as a function of space), minimum size limits (protection as a function of size), and temporary closures (protection as a function of time). Specifically, I used computer models to assess the ability of these tools to promote recovery of depleted populations. Because models are necessarily simplifications of reality, the results here would benefit from field testing.

METHODS

I built three population models, one for each of three fishery management tools: temporary closure of the entire fishery, minimum size limits that reduced fishing mortality for the smallest fish, and no-take marine reserves. These models were mathematically based computer simulations and were general enough to adapt to a variety of fishery species. Using life-history parameters and management strategies, each model generated a time series of predicted catches, as biomass caught per year. These series were first run to stability without new management. Then the new management tool was enacted and the series continued until catches had restabilized.

All models shared fundamental biological assumptions (see Appendix for a detailed mathematical formulation), including a stock-recruitment relationship in which the rate of recruitment was evenly distributed and a function of the adult population biomass averaged over the entire management area. This relationship was density dependent (negative exponential) and assumed that forces such as competition or predation limited recruitment at high densities. All models were deterministic and included no built-in environmental variability. I also assumed that, before new management, each population had been reduced by fishing to population densities lower than those that would produce maximum sustainable yields, and I will refer to these conditions as depleted.

Each model had additional assumptions based on the specific management tool being tested. In the model examining temporary closures of the entire fishery, I assumed that, upon reopening, fishing rates were equal to those before closure. This assumption may not be met. If the closure is successful and produces a build-up in fish biomass, effort is likely to increase dramatically upon reopening, but the closure may also drive some fishermen and women out of business or into other fisheries, reducing effort after reopening. Because of these uncertainties, I chose an assumption of no overall effect.

In the model examining minimum size limits, I assumed that enactment of the limits eliminated fishing mortality for the smallest fish in the fishery and that this loss did not displace fishing effort to the still-available larger fish. These assumptions imply that the limits reduced fishing effort. It would in fact be sensible, in light of the depleted status of the populations I examined, for managers to enact regulations designed to reduce fishing effort in this way, probably by reducing quotas when size limits are raised.

In the model examining no-take marine reserves, I assumed that adult fish did not cross reserve boundaries, remaining for life in the areas where they settled. Conversely, I assumed that larvae dispersed widely across the reserve boundary, creating equal rates of settlement in all areas. These movement assumptions will not fit every species in every circumstance, but they are reasonable for the many species with restricted adult movement and long-lived larvae, including invertebrates and bottom-associated fish, especially if the system of reserves is designed with adult and larval movement patterns in mind (Sladek Nowlis and Yoklavich, 1998; Sladek Nowlis and Roberts, 1999). For this model, I further assumed that fishing mortality in the remaining fishing grounds did not change as a result of the closure. As in the previous model, I assumed that any tendency toward a compensatory increase in effort outside the reserves would be counteracted by additional regulations, such as reductions in quotas. Some examples suggest, however, that effort per unit fishing area need not change with the creation of even a very large reserve, even in the absence of additional management measures (McClanahan and Kaunda-Arara, 1996).

The assumptions were designed to create comparable levels of restriction across management tools. Specifically, a 20% reserve would protect that proportion of the fish population and thus lead to a 20% reduction in catch during the first year. Similarly, a 20% size limit would allow the smallest 20% of the catch biomass to escape, producing a 20% reduction in catch during the first year. I could not create a comparable level of restriction for temporary closures, and instead equated a 10-yr closure with 100% restriction. Future extensions of this work will consider additional management tools, including general effort reductions, which affect all size classes equally, and seasonal/temporal closures in which the fishery alternately opens and closes.

I monitored the models' performances in both short-term and long-term catches (biomass caught per year). Each model run began at an arbitrary point and ran until catches stabilized, at which point the new management tool was enacted. The models continued to monitor catches until they restabilized (Fig. 1). To compare strategies, I had to simplify these catch histories into measures of short- and long-term performance. I used the standard long-term sustainable yield as a measure of long-term performance, a technique used by several other modeling studies of marine reserves (Sladek Nowlis and Roberts, 1999, and references therein). Because no such standard index exists to measure short-term performance, I examined three: magnitude of initial drop in catches relative to those prior to new management, years until catches reached prior levels, and cumulative loss during those years. This approach is consistent with but not identical to those taken previously (e.g., by Hightower and Grossman, 1987; Sladek Nowlis and Roberts, 1997). Together, these measures provided a good indication of, respectively, the magnitude of initial catch losses, the duration of losses, and the total loss.

To compare these results generally, I produced grids of graphs that allowed visual comparison of the effects of four factors simultaneously. Within each grid, dependent variables included:

- Species, including *Panulirus penicillatus*, the Red Sea spiny lobster; *Haemulon plumieri*, the white grunt; and *Epinephelus guttatus*, the red hind. The species varied in two key respects: population growth rate (λ = 1.08, 1.16, and 1.31, respectively) and whether they entered the fishery before (the lobster) or after (the red hind and white grunt) sexual maturity. Parameter values for all of these species are listed by Sladek Nowlis and Roberts (1999);
- Fishing rates, at three levels from lightly to heavily depleted;
- Management tool, including no-take marine reserves, minimum size limits, and temporary closures; and
- Degree of restriction, ranging from 0 to 1 (0 represented no new restriction and 1 represented complete, permanent closure). Temporary closures where handled slightly differently, as explained earlier in the methods.

Each graph shows results for a different measure of performance, including long-term sustainable catch levels, initial losses (as a proportion of the catch prior to new management), number of years until catches exceeded those prior to new management, and the cumulative amount of catch loss during those years (expressed in terms of annual catch rates prior to new management).

RESULTS

Each run of the models generated a catch history, starting with the last year before new restrictions were enacted (year 0) and running until catches had reached stability. These catch histories had a characteristic shape for most runs; catches initially dropped as a result of the new restrictions and eventually rose to levels higher than those prior to new management (Fig. 1). The magnitude and duration of the initial drop, as well as level of

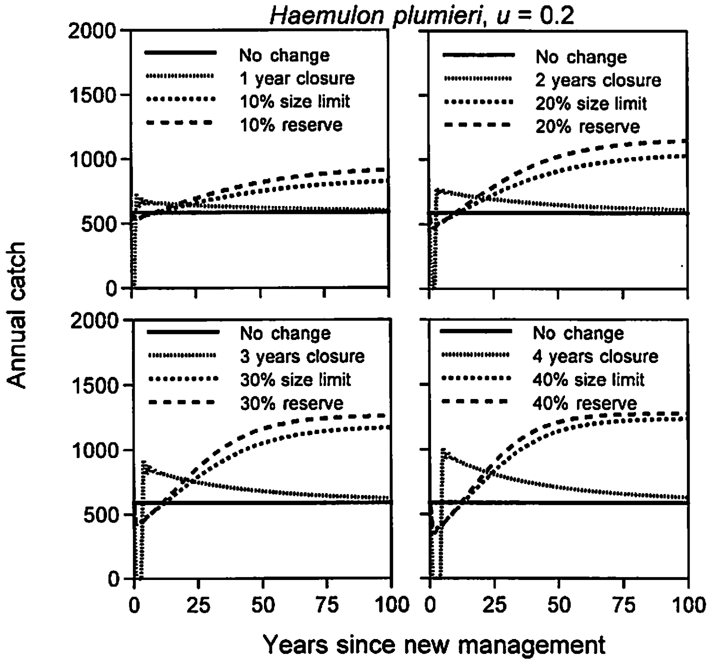

Figure 1. Examples of catch histories. These were for the white grunt, *Haemulon plumieri*, with adult fish facing 20% annual fishing mortality rate ($u = 0.2$). Each graph represents a different degree of restriction and shows annual catches graphed over 100 yrs from when new management measures were enacted. The model actually ran longer before catches stabilized. All histories show initial declines after enactment of new management, followed by some kind of recovery. The magnitude and duration of initial losses and the level of ultimate recovery depended on the species, fishing mortality rate, and the type and degree of restriction.

stable catch, varied from run to run depending on several key variables, including the species, fishing rate, and degree and type of restriction (Fig. 1). When I examined the effect of these factors on of the catch profile characteristics, some interesting patterns emerged.

Stable catch rates did not change with the duration of temporary closures. Instead, catches always returned to their pre-enactment levels (Fig. 2), as would be expected because effort after reopening matched levels prior to the closure. In contrast, when size limits or no-take reserves were used, catches increased unless restrictions were excessive. For example, when reserves were used for the white grunt at a fishing mortality rate of $u = 0.2$, catches peaked at a reserve size encompassing approximately 30% of the management area (Fig. 2). If reserves encompassed more than 80%, catches never rebuilt to pre-enactment levels. Size limits had a similar effect, but peak catches occurred under greater restrictions (e.g., 45% for the white grunt at a fishing mortality rate of $u = 0.2$), and peak yield was slightly lower than that for reserves (Fig. 2). These patterns were consistent across all fishing rates for both the white grunt and the red hind (Fig. 2), but the spiny lobster, which was fished before reaching maturity, showed greater catch increases under size limits than through no-take reserves (Fig. 2).

Initial catch losses were directly related to the definition of restriction, so the results were very straightforward. These initial losses showed the same pattern for all species

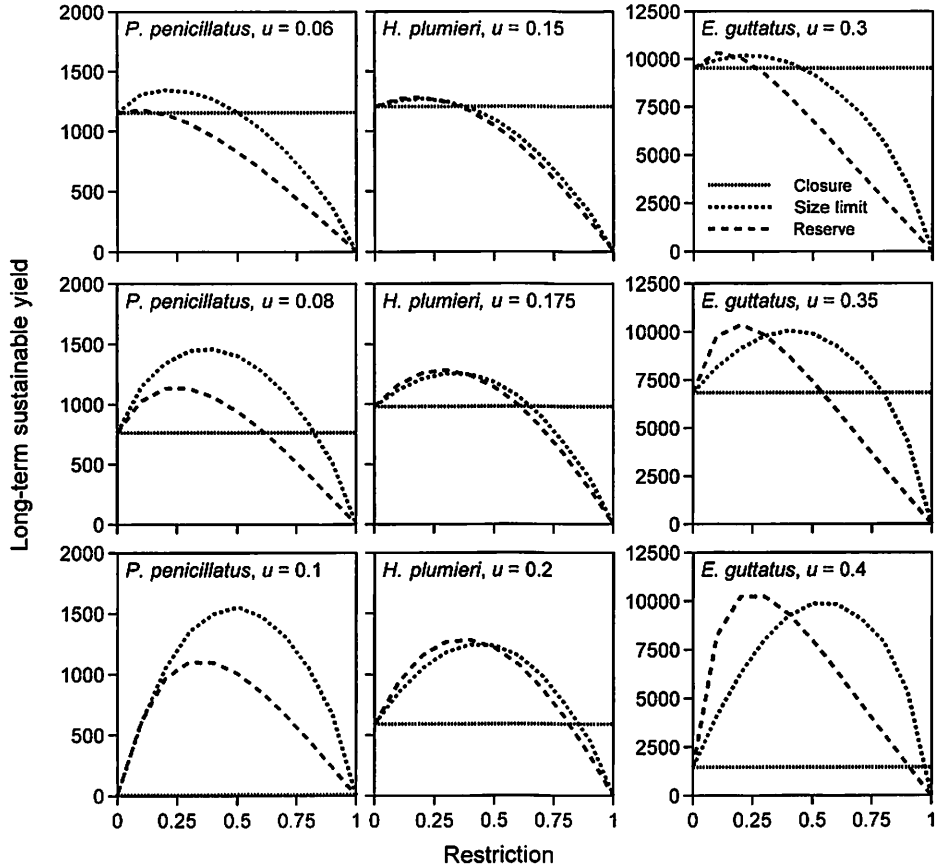

Figure 2. Long-term sustainable yields. Enhancement, or the degree to which the stable yields exceeded those prior to new management, depended on the species, fishing mortality rate, and the type and degree of restriction. Enhancements were greatest at high fishing rates but required greater restrictions under these conditions. For the two species in which individuals were not vulnerable to fishing until they had reached sexual maturity (the white grunt, *Haemulon plumieri*, and the red hind, *Epinephelus guttatus*), reserves produced greater enhancements than size limits and did so under lighter restrictions. For the spiny lobster, *Panulirus penicillatus*, reserve enhancements peaked under lighter restrictions, but this peak was lower than that resulting from size limits. Temporary closures did not produce any long-term catch benefits because of the lack of new restrictions at the reopening of the fishery.

and all fishing rates. Under temporary closures, the entire first-year catch was lost, whereas under size limits or no-take reserves, an amount of catch equal to the degree of restriction was lost (Fig. 3).

The duration of loss was more complex. In all cases, losses due to temporary closures lasted as long as the closure. When the fishery was reopened, fish biomass was higher than before new management, so catches exceeded those prior to new management in the first year of fishing (Fig. 4). Reserves and size limits took longer to show overall enhancements (Fig. 4). For the spiny lobster, losses lasted longest with no-take reserves (Fig. 4). For the other two species, duration of loss was roughly equal for reserves and size limits (Fig. 4).

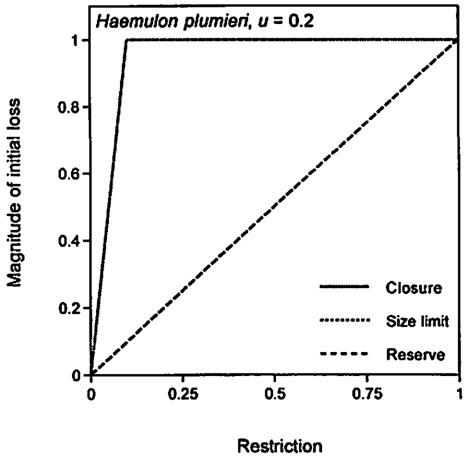

Figure 3. Magnitude of initial loss. The magnitude of initial loss depended on the type and degree of restriction but was consistent across all species and fishing mortality rates. For temporary closures, the entire catch was lost during the time of closure. For size limits and reserves, losses were equal to the degree of restriction because restriction was defined in terms of initial loss (the lines for these two measures overlapped completely).

Cumulative losses synthesize the magnitude and duration of losses, and thus might be the best of the short-term measures. For all three species, cumulative losses were greater for temporary closures than for size limits or no-take reserves at equivalent degrees of restriction. For the white grunt and red hind, size limits and marine reserves showed equivalent cumulative losses (Fig. 5). For the spiny lobster, size limits produced fewer cumulative losses than no-take reserves (Fig. 5).

DISCUSSION

Except where juveniles were vulnerable to fishing, marine reserves provided greater long-term catches and did so with less restriction than other management tools. Reserves allowed a subgroup of fish to grow much larger and achieve high reproductive output. In contrast, minimum size limits freed small fish from fishing mortality for a relatively short period of time. The increased reproductive output from these small fish did not compare with the output from larger individuals within reserves. Temporary closures did not provide any long-term catch enhancements.

In the short term, reserves also fared well. Initial losses were less severe with reserves and size limits than with temporary closures, although recovery times were longer. Cumulatively, reserves fared as well, per unit restriction, as any other tool for species where only adults were fished. For species where juveniles were susceptible to fishing, reserves did have higher cumulative losses than size limits at similar levels of restriction. However, reserves produced maximal benefits with less restriction than size limits. Because reserves required fewer restrictions and because losses increased with degree of restriction, reserves are likely to entail more modest short-term losses than other management tools, at least for species where juveniles are not fished. For species where juveniles are caught, these results suggest that size limits might be a better first choice as a tool for

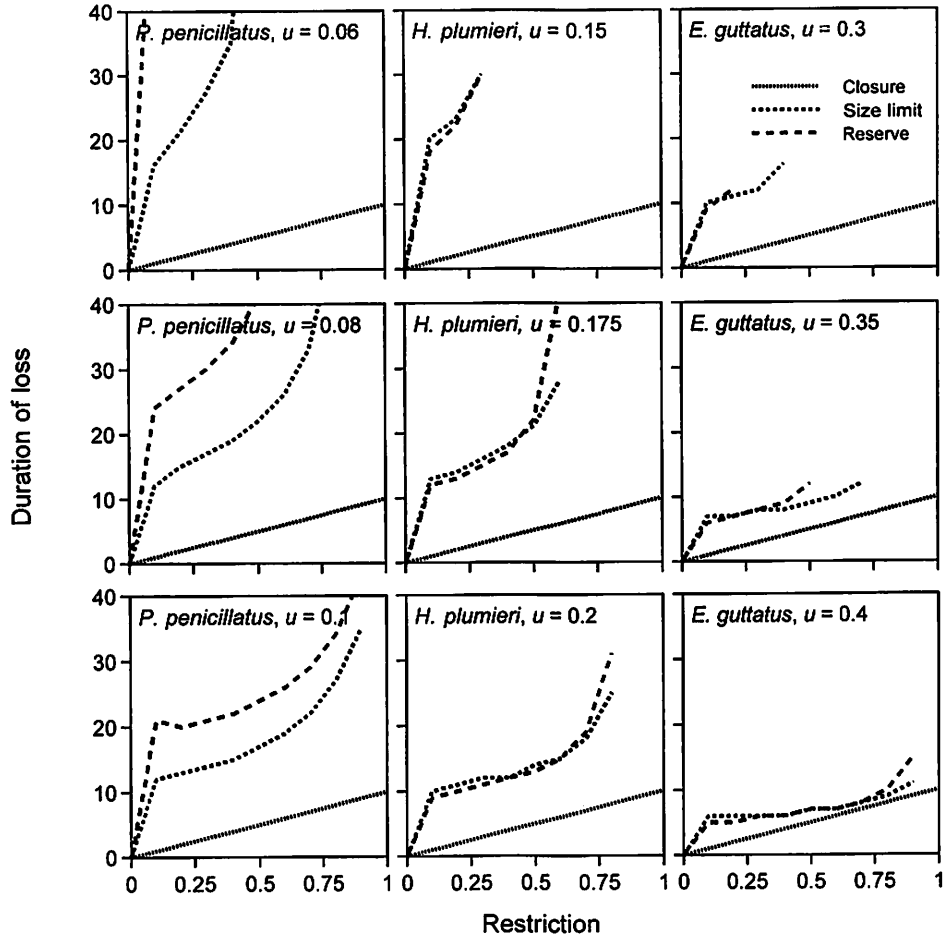

Figure 4. Duration of loss. The duration of initial loss depended on the species, fishing mortality rate, and the type and degree of restriction. Losses were generally greatest for the slowest-growing species and at the lowest fishing mortality rates. For the two species in which individuals were not vulnerable to fishing until they had reached sexual maturity (the white grunt, *Haemulon plumieri*, and the red hind, *Epinephelus guttatus*), cumulative losses were similar with reserves and size limits; both were somewhat higher than losses due to temporary closure of the entire fishery. For the spiny lobster, *Panulirus penicillatus*, losses were greatest with reserves and least for temporary closure of the entire fishery.

rehabilitating depleted populations. For species where juveniles are not caught, reserves may provide the greatest long-term benefits with the fewest short-term losses.

These results are fundamentally affected by the assumptions of the models, which are in turn central to the questions I was interested in asking and to the results I present. Had I assumed no relationship between adult density and recruitment, there would have been little incentive in the model environment to conserve adult fish. Consequently, the best strategies would have involved more substantial fishing pressure and less conservation. An assumption that the environment was variable would also have had significant, albeit less predictable, effects on the models' catches. We know from previous studies that reserves can stabilize annual catches (Lauck et al., 1998; Sladek Nowlis and Roberts, 1999)

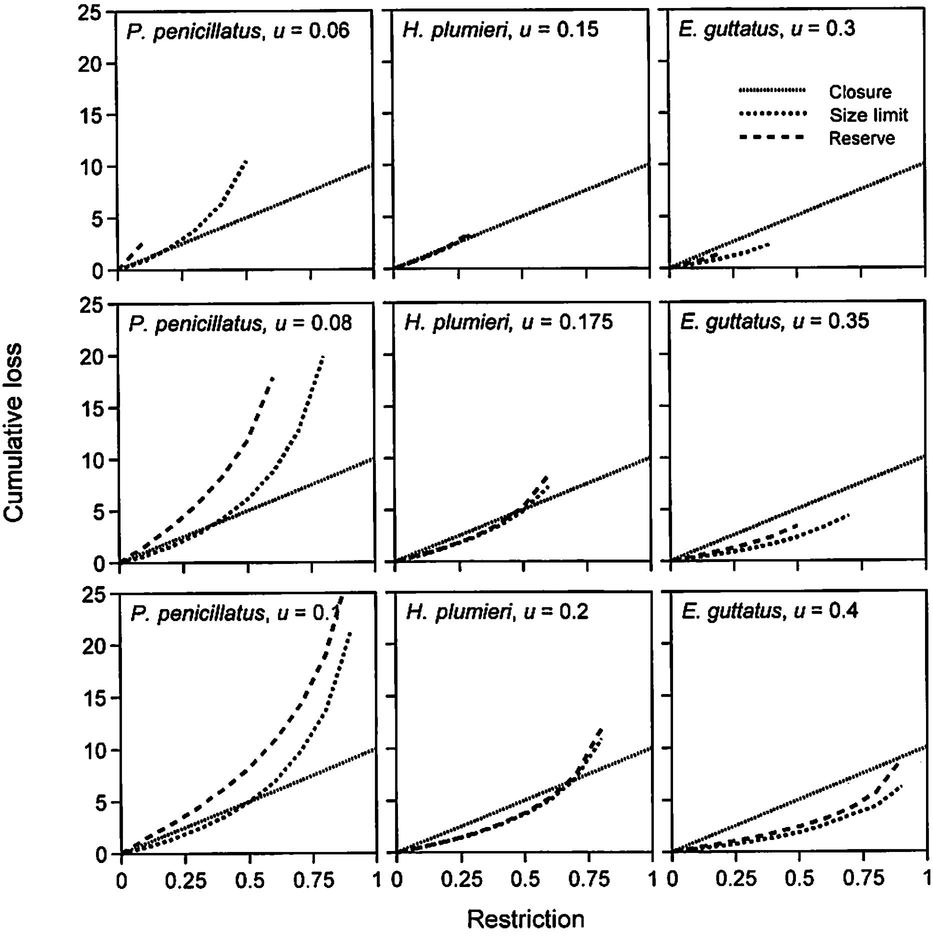

Figure 5. Cumulative loss. The cumulative loss depended on the species, fishing mortality rate, and the type and degree of restriction. Losses were generally greatest for the slowest-growing species and at the lowest fishing mortality rates. For the two species in which individuals were not vulnerable to fishing until they had reached sexual maturity (the white grunt, *Haemulon plumieri*, and the red hind, *Epinephelus guttatus*), cumulative losses were similar for reserves and size limits; both were somewhat lower than losses due to temporary closure of the entire fishery. For the spiny lobster, *Panulirus penicillatus*, losses were greatest with reserves and least with size limits unless restrictions were severe.

in a fluctuating environment, in part because they maintain a richer age structure than other tools. Thus, the assumption of a deterministic environment probably underestimated the benefits from reserves. Had I assumed that populations were underfished or fished to maximum capacity, no additional fishery restrictions would have been necessary to increase catches, and none of the new management tools would have fared well.

The assumptions specific to each model also affected the results. In the model examining temporary closures, I assumed that fishing rates upon reopening were equal to those prior to closure. Had I assumed that fishing rates increased, as might be expected because of the incentive of the increased biomass, the long-term results from this management strategy would have been even more dismal. Alternatively, if the closure drove some fishermen and women out of business, and effort upon reopening was therefore lower than before closure, the long-term results would have looked more favorable.

In the model examining size limits, my assumption that the smallest fish in the fishery escaped fishing mortality biased results favorably toward this management tool. In reality, size limits via prohibitions against landings or even gear restrictions may not allow all smaller individuals to escape, if smaller individuals are simply caught and thrown back dead. The further assumption that effort did not increase on the larger fish biased results favorably toward this management tool in the long run but against it in the short run. If larger fish did become more vulnerable to fishing, some of the initial costs would be offset, but long-term benefits would be lost.

In the model examining reserves, the assumption that larvae crossed reserve boundaries freely but that adults never crossed them may bias results in favor of this management strategy in the long run but against it in the short run. If a reserve leaks adults, it will be less effective at protecting them; catches will remain higher in the short run, but reserves will not provide as many benefits in the long run. However, growing evidence regarding fish movement patterns suggests that these assumptions may be widely applicable, even for fish that are not thought of as bottom associated (Holland et al., 1996). As with the size-limit model, I assumed that fishermen and women did not increase their efforts on the remaining available fish. In reality, effort may increase outside the reserve because of displacement, or it may decrease if the establishment of reserves provides other economic incentives. Real-life examples suggest that effort can remain relatively constant outside reserves (e.g., McClanahan and Kaunda-Arara, 1996). If effort were to become more concentrated, reserves would provide more favorable short-term results but poorer long-term results.

These results provide several important lessons for managers. First, they demonstrate that, when a population is depleted and juveniles are vulnerable to fishing, management that eliminates fishing mortality for juveniles is highly effective. This lesson is common sense: fisheries should not be based on immature individuals.

Second, reserves show excellent promise as a management tool for rebuilding depleted populations. According to the results of this study, reserves can provide higher catches with less restriction and lower initial losses than other management tools. In order to reap these benefits, managers must design reserves to satisfy the basic assumptions of the model presented here, namely that larval fish cross reserve boundaries, that adult fish do not, and that reserve networks are designed on a scale that corresponds to the larval-dispersal capacity of the species of interest. This lesson may gain importance over the next several years. Many fisheries are depleted in the United States (National Marine Fisheries Service, 1998) and worldwide (Food and Agriculture Organization, 1995). Our growing awareness of overfishing is likely to steer us toward rebuilding tools, and reserves show great promise.

Finally, reserves are not a panacea. The positive results demonstrated here were based on the assumption that fishing mortality stayed constant in the remaining fishing grounds after the creation of the reserves. To meet this assumption, managers would offer incentives not only to reduce overall effort temporarily but also to limit future increases. Reserves do show promise as a tool but are only one of many in the toolbox of a good manager. This study also demonstrated the utility of size limits, and additional tools may be useful for maintaining and rebuilding productive fisheries.

ACKNOWLEDGMENTS

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LITERATURE CITED

Alcala, A. C. and G. R. Russ. 1990. A direct test of the effects of protective management on abundance and yield of tropical marine resources. J. Cons. Cons. Int. Explor. Mer 47: 40–47.

Beverton, R. J. H. and S. J. Holt. 1957. On the dynamics of exploited fish populations. Chapman & Hall, New York. 533 p.

Bohnsack, J. A. 1996. Maintenance and recovery of fishery productivity. Pages 283–313 *in* N. V. C. Polunin and C. M. Roberts, eds. Reef fisheries. Chapman & Hall, London.

Dugan, J. E. and G. E. Davis. 1993. Applications of marine refugia to coastal fisheries management. Can. J. Fish. Aquat. Sci. 50: 2029–2042.

Food and Agriculture Organization. 1995. The state of the world fisheries and aquaculture. FAO Fisheries Department, FAO Fish. Circ. 884. 105 p.

Hightower, J. E. and G. D. Grossman. 1987. Optimal policies for rehabilitation of overexploited fish stocks using a deterministic model. Can. J. Fish. Aquat. Sci. 44: 803–810.

Holland, K. N., C. G. Lowe and B. M. Wetherbee. 1996. Movements and dispersal patterns of blue trevally (*Caranx melampygus*) in a fisheries conservation zone. Fish. Res. (Amst.) 25: 279–292.

Lauck, T., C. W. Clark, M. Mangel and G. R. Munro. 1998. Implementing the precautionary principle in fisheries management through marine reserves. Ecol. Appl. 8: S72–S78.

McClanahan, T. R. and B. Kaunda-Arara. 1996. Fishery recovery in a coral-reef marine park and its effects on the adjacent fishery. Conserv. Biol. 10: 1187–1199.

National Marine Fisheries Service. 1998. Report to Congress: status of the fisheries of the United States. U.S. Dept. Comm., Silver Spring, Maryland. 97 p.

Plan Development Team. 1990. The potential of marine fishery reserves for reef fish management in the U.S. southern Atlantic. Snapper-Grouper Plan Development Team Report for the South Atlantic Fisheries Management Council. NOAA Tech. Memo. NMFS-SEFC-261. NOAA National Marine Fisheries Service, Southeast Fisheries Center, Miami, Florida. 45 p.

Roberts, C. M. and N. V. C. Polunin. 1991. Are marine reserves effective in management of reef fisheries? Rev. Fish Biol. Fish. 1: 65–91.

Rowley, R. J. 1994. Marine reserves in fisheries management. Aquat. Conserv. Mar. Freshw. Ecosyst. 4: 233–254.

Russ, G. R. and A. C. Alcala. 1996. Do marine reserves export adult fish biomass? Evidence from Apo Island, central Philippines. Mar. Ecol. Prog. Ser. 132: 1–9.

Sladek Nowlis, J. and C. M. Roberts. 1997. You can have your fish and eat it, too: theoretical approaches to marine reserve design. Proc. 8th Int'l. Coral Reef Symp. 2: 1907–1910.

______________ and ______________. 1999. Fisheries benefits and optimal design of marine reserves. Fish. Bull., U.S. 97: 604–616.

______________ and M. M. Yoklavich. 1998. Design criteria for rockfish harvest refugia from models of fish transport. Pages 32–40 *in* M. M. Yoklavich, ed. Marine harvest refugia for west coast rockfish: a workshop. NOAA Tech. Memo. NOAA-TM-NMFS-SWFSC-255. U.S. Dept. Comm., Pacific Grove, California.

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APPENDIX

SIZE STRUCTURE.—Populations were divided into size categories by the same technique used by Sladek Nowlis and Roberts (1999). The concept is to select the size range encompassed by each size class so that year 0 fish have exactly a 100% chance of growing one size class during their first year. This categorization avoids the potential difficulty caused by fish that grow more than one size class in any year.

DENSITY DEPENDENCE.—Density dependence was incorporated into survivorship of size/year 0 fish during their first year. I used the negative exponential function

$$n_{1,t+1} = v_0 n_0 e^{-n_{0,t}/K} \qquad \text{Eq. 1}$$

where $n_{x,t}$ is the density of size x fish in year t, v_0 is the density-independent survival rate for size 0 fish, and K is a measure of the carrying capacity for the population. Other factors could also affect the density of size 1 fish, including fishing mortality and contributions from size 1 fish that survived the year but did not grow to size 2. Equation 1 merely illustrates the form of density dependence used in this modeling exercise.

GENERAL POPULATION MODEL.—This discrete-time model was projected yearly according to the equation:

$$n_{t+1} = F\left(Xn_t\right) \qquad \text{Eq. 2}$$

where n_t is a vector representing the density of the population, by size class, at time t; X is a matrix representing nonfishing life history factors as defined below; and F is a diagonal matrix whose elements represent the probability that members of each size class survive fishing for a year. X has three sets of elements: top row, diagonal, and below-diagonal. The top-row elements represent fecundities (as defined in Sladek Nowlis and Roberts, 1999) multiplied by the probability that eggs survive to become new recruits (see Sladek Nowlis and Roberts, 1999). The below-diagonal elements represent the probability for each size class that individuals survive natural mortality and grow to the next size class. The diagonal elements represent the probability for each size class that individuals survive natural mortality yet grow insufficiently to move to the next size class.

TEMPORARY CLOSURES.—The fishing matrix, F, was replaced by an identity matrix, I, for the years of the closure. Otherwise this model functioned like the general model.

SIZE LIMITS.—The fishing matrix, F, was modified in this model. Starting with the smallest size class vulnerable to fishing, x, the fishing mortality rate f_{xx} was reduced either until it became zero or such that the catch from the stable population size prior to new management had been reduced to the desired restriction level. If reducing f_{xx} to zero was insufficient to achieve the desired catch reduction, the program iteratively ran the same process with the next larger size class, x+1, and so on until the desired reductions were achieved. This new fishing matrix G was substituted for F in Equation 2 and used for the remainder of the model run.

MARINE RESERVES.—This model used the same assumptions and techniques as the model presented by Sladek Nowlis and Roberts (1999). The population was partitioned proportionately between the reserve and the remaining fishing grounds. The remaining fishing population followed Equation 2, as did the reserve population but with an identity matrix, I, substituted for the fishing mortality matrix, F. The two populations interacted solely through larval dispersal. Size 0 population densities were calculated from a weighted average of the population densities in the reserve and fishing area. For a more detailed description, see Sladek Nowlis and Roberts (1999).

BULLETIN OF MARINE SCIENCE, 70(2): 715–731, 2002 MOTE SYMPOSIUM INVITED PAPER

METHODS FOR INCREASING THE LIKELIHOOD OF RESTORING AND MAINTAINING PRODUCTIVE FISHERIES

Joshua Sladek Nowlis and Bruce Bollermann

ABSTRACT

Years of scientific inquiry have developed sophisticated methods for setting fishing quotas. Unfortunately, these methods tend to be information intensive and can lead to population crashes if information is wrong. Previous work has illustrated that highly responsive quota systems, which curb fishing decisively when stocks drop below target abundance levels, promote optimum average yields in varying environments and when parameters are uncertain. These policies have generally been rejected, though, because they make fishing yields uncertain and create the potential for temporary closures. They have also been criticized because the managed population can crash if abundance is overestimated. We performed analyses to reexamine the performance of highly responsive management systems. Our analyses show that these systems outperform less-responsive alternatives at maintaining healthy stocks and productive fish catches when managers misestimate parameters. Although these systems can cause populations to crash under the circumstances previously identified, we were able to show that they are less prone to do so under all circumstances than less-responsive constant-fishing-mortality systems. We discuss the implications of this work for fisheries management and highlight methods for achieving highly responsive management systems that are both precautionary and ecosystem-oriented.

> *"Science is being asked to deliver far more than it can produce"*
> (Peter Leipzig, Fisherman's Marketing Group,
> Environmental News Network, 2000).

> *"We need a better way of... managing with gaps in scientific information"*
> (Lisa Speer, Natural Resources Defense Council,
> H. John Heinz III Center for Science, 2000).

Scientists and managers have struggled for decades with how to set appropriate fishing quotas—the amount of fish that can be caught without jeopardizing the future productivity of the population. By the 1950s, sophisticated mathematical models were developed to aid managers with setting quotas. Despite these models, fisheries have not fared well. Nearly half of the known fish stocks in the U.S. are considered overfished, experiencing overfishing, or both (Fig. 1), and nearly 70% of fish stocks worldwide are estimated to be fished to or beyond their maximum capacity (FAO, 1998; NMFS, 2001).

One of the biggest problems with using the sophisticated models as tools has been that they require a substantial amount of information. Applying these models to reality breaks down because most fish stocks have never been studied to the extent required for accurate use. In the U.S. for example, which has better resources for studying fish populations than most other countries, scientists have fully assessed the status of less than one-quarter of the fish stocks under federal management (NMFS, 2001; Fig. 1).

These problems with quota setting raise three important areas of concern. First, what do we know about fish stocks, what do we need to know under conventional management, and can we do better? Second, are there less information-intensive quota systems,

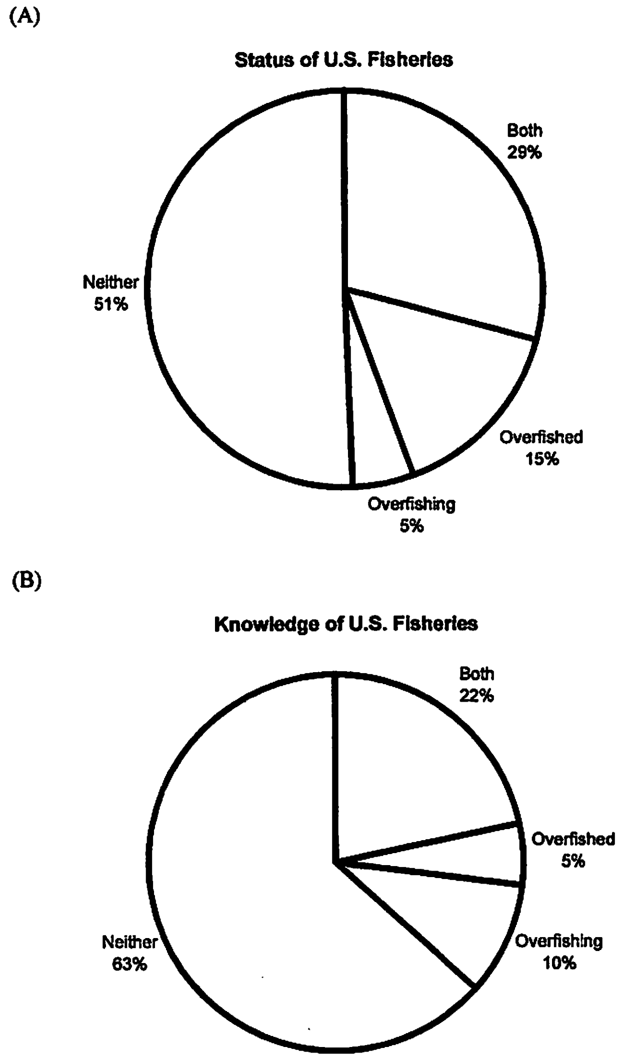


Figure 1. The status of federally managed fish stocks in the United States. (A) Known fish stocks were categorized as overfished, experiencing overfishing, both, or neither on the basis of criteria that vary somewhat across stocks. Generally speaking, a stock was declared overfished only if it had dropped below one-half of the target MSY level and experiencing overfishing only if it was experiencing fishing mortality rates in excess of the best available estimate or proxy for the rates associated with MSY. (B) Most stocks are categorized as unknown with respect to one or both of these characteristics because they have not been assessed. Data from National Marine Fisheries Service (NMFS, 2001).

and how do these perform relative to conventional methods? Finally, how do we manage stocks for which we lack the most basic information?

INFORMATION NEEDS AND STATUS UNDER CONVENTIONAL MANAGEMENT.—Conventional management is information intensive. Historically, fisheries mangers set quotas using a constant fishing mortality rate—a fixed proportion of the population removed by fishing each year or season. Proper use of a constant-fishing-mortality system requires a thor-

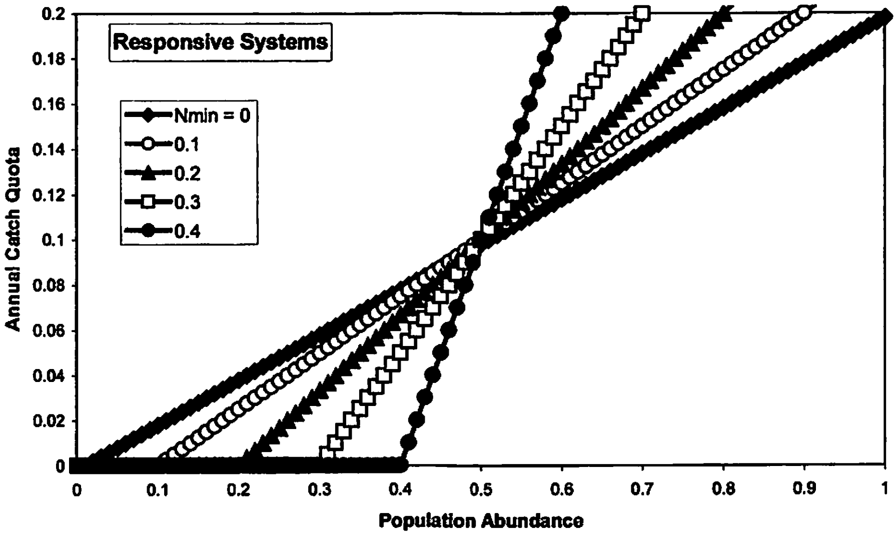


Figure 2. Quota policies. This graph shows a series of lines representing different management systems. The line nearest horizontal ($N_{min} = 0$) represents a constant-fishing-mortality-rate policy; steeper lines are more responsive. All policies were engineered to pass through one common point—the maximum sustainable yield of the fishery. As a result, if managers made no mistakes, each of these policies would achieve the maximum sustainable yield.

ough understanding of the biology and ecology of the stock in the form of productivity values, current abundance, and actual catches (Fig. 2). Each of these information needs has its pitfalls, which is one of the reasons so many stocks remain categorized as unknown with respect to their overfished status.

Productivity estimates are generally created in the context of maximum sustainable yields (MSY). The MSY is defined as the maximum rate of exploitation a population can sustain and is associated with a fishing mortality rate F_{MSY} and an abundance level N_{MSY}, also sometimes called B_{MSY}. It is determined through a stock assessment, in which one or more scientists develop a complex computer simulation using detailed information about the history of abundance and fishing pressure experienced by the stock in question. This assessment also incorporates basic biological characteristics, such as the growth rate of individuals. Because these biological characteristics are generally poorly understood, most stocks remain unassessed. Even the best stock assessments must make educated guesses when it comes to certain factors, in particular the natural mortality rate and the stock-recruitment relationship. Both of these factors help to define the ecology of the stock in question and are typically unknown or at best poorly understood. The uncertainty surrounding them adds a great deal of uncertainty for management of stocks that are assessed.

Stock abundance also plays a key role in conventional management. The assessment and determination of MSY define the appropriate fishing mortality rate F_{MSY}—the fraction of the stock that, if removed on a continuing basis, will ultimately produce the MSY. An estimate of absolute stock abundance is required to calculate quotas because the total allowable catch will be the fraction F_{MSY} of the total. Counting fish is not easy. The ocean is large and complex, and some of the many difficulties in estimating stock abundance include the limits of making direct underwater observations at depth due to human limitations (SCUBA) or expense (submersible), the impracticality of mechanically sampling many habitats due to their structural complexity, changes in fish behavior in response to counting efforts, and many other challenges. Consequently, fish abundance is sampled infrequently, if at all. Even the data for sampled stocks may not be analyzed. Only recently, the barndoor skate (*Raja laevis*), a large elasmobranch caught incidentally in northwest Atlantic bottom-trawl fisheries, was found to be severely depleted when decades of information were finally examined (Casey and Myers, 1998).

Although catches seem easy to measure by comparison, they are typically poorly tracked at best. Commercial catches brought into port are fairly easy to count, but it is much more difficult to account for fish discarded at sea. They may be discarded because they are too small, exceed the legal quota, or are of an unmarketable or prohibited species. These fish are typically only counted if the government or industry pays an independent observer to do so, although fishing log books can also be of use. Recreational catches are equally challenging to document because recreationally fishing men and women are so spread out in space and time.

THE QUOTA DEBATE—Fishery managers arrived at the current convention for setting quotas after a robust scientific debate spanning several decades (see Thompson, 1999, for review). Scientists examined constant catch, under which fishers are permitted to catch an invariant number of fish each year (Russell, 1931; Hjort et al., 1933); constant-escapement policies, where maximum fishing is allowed if a population is above a threshold abundance but prohibited if it is below that abundance (Ricker, 1958); and constant-fishing-mortality-rate policies, under which fishers are permitted to catch an invariant fraction of the population each year (Thompson and Bell, 1934; Graham, 1935).

These techniques have been compared and contrasted extensively (e.g., by Reed, 1978), and many additional complexities have been examined, including policies defined by multiple parameters allowing changes in policy in relation to changes in stock size (e.g., by Ricker, 1958). Ricker's work helped to establish some basic principles in comparing the performance of policies in a varying environment. He showed that more responsive policies provided higher average catches but introduced higher rates of variability from year to year in catch levels and could lead to temporary fishery closures. These findings have been confirmed in a number of studies (see Thompson, 1999, for review). Scientists have also used highly complex dynamic programming exercises to identify detailed quota-setting rules (e.g., Walters, 1975; Hilborn, 1976). These complex models allow managers to fine tune an optimal policy but at the expense of accessibility (Thompson, 1999). They also often derive policies similar to those derived by simple rules (see, e.g., Walters, 1975; Hilborn, 1976). A third alternative was proposed by Walters and Hilborn (1978), that of the fixed form optimization. In this technique, one conjures up an appropriate functional form for the quota-setting process based on intuition and experience and then optimizes its parameters on the basis of the balance a manager wants between high average catches and low catch variance (see, e.g., Quinn et al., 1990).

Scientists and managers have generally not adopted highly responsive management systems because of high variability in catches, frequent fishery closures, and potential for population crashes if abundance is overestimated (Engen et al., 1997). Instead, they tend to rely on constant fishing-mortality rates but with a few promising signs of changing to policies with constant rates at higher abundance that are tapered down if abundance drops to low levels (Thompson, 1999).

We set up a framework to revisit responsive systems over a broad range of management error and environmental variability. The most responsive system resembled constant-escapement policies, and the least responsive used constant-fishing-mortality-rate policies. We compared the performance of these systems across a wide range of errors in estimation of parameters related to production, abundance, and catches in a stable environment and compared their performance with specified errors in two different variable environments.

DERIVATION OF A FRAMEWORK

We built and analyzed a series of mathematical models, starting with general forms and becoming more specific as necessary to address particular questions of interest. Generally, we wanted to examine a population of size N governed by the dynamic.

$$\frac{dN}{dt} = p(N)N - h(N) \qquad \text{Eq. 1}$$

where N is some measure of population abundance, $p(N)$ is the per capita growth function, and $h(N)$ is the catch function. For simplicity we assumed a linear-control feedback loop for the catch function,

$$h(N) = f(N - N_{\min}) \qquad \text{Eq. 2}$$

based on intuition gained from designing rocket guidance systems. We defined per capita production deterministically with steady state $r_{ss} = p(N_{ss})$. Note that this catch function sets aside a reserve population N_{min} and exploits only the population abundance above the reserve at a rate f, similar to models examined previously by Ricker (1958), Engen et al. (1997), and others (reviewed by Thompson, 1999).

DETERMINISTIC LOGISTIC EXAMPLE

To illustrate this framework, we assumed a simple logistic production function

$$p(N)N = r(1 - N)N \qquad \text{Eq. 3}$$

Note that this production function peaks at $N_{MSY} = 1/2$ with a productivity of r/4. Any quota-setting policy will hit this peak if

$$f_{MSY} = \frac{r}{2 - 4N_{\min}} \qquad \text{Eq. 4}$$

and N_{min} falls in the range $N_{MSY} > N_{min} \geq 0$. To illustrate the effects of management errors, we can introduce error terms representing the three types of information necessary under conventional fisheries management: production-estimation error, ε_r; abundance-estimation error, ε_n; and catch-estimation error, ε_h. A positive production-estimate error would occur if a manager overestimated production and consequently set quotas higher than intended. A positive abundance-estimation error would indicate that managers overestimated the actual abundance, and therefore probably set higher quotas than intended. A positive catch-estimation error would occur if catches exceeded quotas. All three errors range from −1 to infinity; negative values represent lower catches than intended.

With errors, catches indicated in equation Eq. 2 differed from quotas by a factor of $(1 + \varepsilon_h)$.

$$h(N) = (1+\varepsilon_h) f_{est}(N_{est} - N_{\min}) \qquad \text{Eq. 5}$$

where f_{est} represents the estimated optimum fishing mortality rate and N_{est} represents the estimated population abundance. We can obtain an MSY-intended policy by combining Eqs. 4 and 5. Adding errors in the estimation of the MSY fishing mortality rate and of population abundance yields

$$h(N) = \frac{(1+\varepsilon_h)(1+\varepsilon_r) r\left[(1+\varepsilon_n)N - N_{\min}\right]}{2-4N_{\min}} \qquad \text{Eq. 6}$$

With this formulation, errors in the estimation of production, ε_r and actual catches, ε_h, are functionally equivalent (i.e., $\varepsilon_{h/r}$) and were so treated in analyses of the system equation:

$$\frac{dN}{dt} = r(1-N)N - \frac{(1+\varepsilon_{h/r}) r\left[(1+\varepsilon_n)N - N_{\min}\right]}{2-4N_{\min}} \qquad \text{Eq. 7}$$

Note that errors are not introduced into the production function $r\,(1-N)N$ because it is beyond the control of managers. We consider the effects of a variable environment below by making the production parameter r stochastic.

We found stable abundances and catches by solving this equation for conditions where change in abundance $dN/dt = 0$ (see Appendix for derivation). In general, the stable population abundance and catch levels in this system were given by:

$$N_{ss} = \frac{2-4N_{\min} - (1+\varepsilon_{h/r})(1+\varepsilon_n) + \sqrt{(1-\varepsilon_{h/r} - \varepsilon_n - \varepsilon_{h/r}\varepsilon_n)^2 + 16\varepsilon_{h/r}N_{\min}(1-N_{\min}) + 8\varepsilon_n N_{\min}(1+\varepsilon_{h/r})}}{4-8N_{\min}} \qquad \text{Eq. 8}$$

$$h_{ss} = \frac{(1+\varepsilon_{h/r}) r\left[(1+\varepsilon_n)N_{ss} - N_{\min}\right]}{2-4N_{\min}} \qquad \text{Eq. 9}$$

but only when N_{SS}, as defined by Eq. 8, multiplied by $(1 + \varepsilon_n)$ is greater than N_{min}. Otherwise, no fishing would be allowed, and the population would remain at its unfished abundance of 1. Although the quadratic solution provides two potential solutions, one adding the square root and the other subtracting, only negative abundances are achieved if the square root is subtracted.

If we set the abundance estimation error $\varepsilon_n = 0$, Eqs. 8 and 9 become

$$N_{ss} = \frac{2 - 4N_{\min} - \left(1 + \varepsilon_{h/r}\right) + \sqrt{\left(1 - \varepsilon_{h/r}\right)^2 + 16\varepsilon_{h/r} N_{\min}\left(1 - N_{\min}\right)}}{4 - 8N_{\min}} \qquad \text{Eq. 10}$$

$$h_{ss} = \frac{\left(1 + \varepsilon_{h/r}\right) r \left(N_{ss} - N_{\min}\right)}{2 - 4N_{\min}} \qquad \text{Eq. 11}$$

Or, if we set the production and catch estimation error $e_{h/r} = 0$, Eqs. 8 and 9 become

$$N_{ss} = \frac{2 - 4N_{\min} - \left(1 + \varepsilon_n\right) + \sqrt{\left(1 - \varepsilon_n\right)^2 + 8\varepsilon_n N_{\min}}}{4 - 8N_{\min}} \qquad \text{Eq. 12}$$

$$h_{ss} = \frac{r\left[\left(1 + \varepsilon_n\right)\right] N_{ss} - N_{\min}}{2 - 4N_{\min}} \qquad \text{Eq. 13}$$

or $h_{ss} = 0$ and $N_{ss} = 1$ if N_{SS}, as defined by Eq. 12, multiplied by $(1 + \varepsilon_n)$ is less than N_{min}.

These results provide several important insights (Fig. 3). First, catches were equivalent under all policies if management estimates were correct. Consequently, no long-term costs were associated with using responsive management systems when information was good, at least in the stable environments represented by these equations. Second, catches were most affected by management error when management systems were least responsive. Under a constant-fishing-mortality system (i.e., $N_{min} = 0$), the fishery was reduced to commercial extinction when any combination of production, catch, or abundance estimates was off by a factor of two. More responsive management systems fared better in the face of abundance overestimates, but catches still did decline as a result of these errors. Responsive management systems fared exceptionally well in the face of production or catch overestimates. Highly responsive systems (e.g., $N_{min} = 0.4$) maintained almost optimum catch levels even when production or catch estimates were off by a factor of four. Only when abundance was underestimated did responsive systems yield smaller catches than less-responsive systems. In some of these cases, fishing was not allowed even though fish populations were at peak abundance because abundance was estimated to be below N_{min}, the threshold at which fishing was prohibited. These cases present only a minor management challenge because fish are still plentiful and capable of providing fishing opportunities once the management error is rectified, without any long-term loss of fish productivity or ecosystem function.

Bigger problems arose when responsive systems were examined in a fluctuating environment. We simulated a discrete version of Eq. 7 in two different environments over a period of 500 yrs (Fig. 4). In both cases, we sampled from distributions of production constants (r) with a mean of 0.4 and used the same production history for each run in a given environment. Our moderately variable environment was normally distributed and

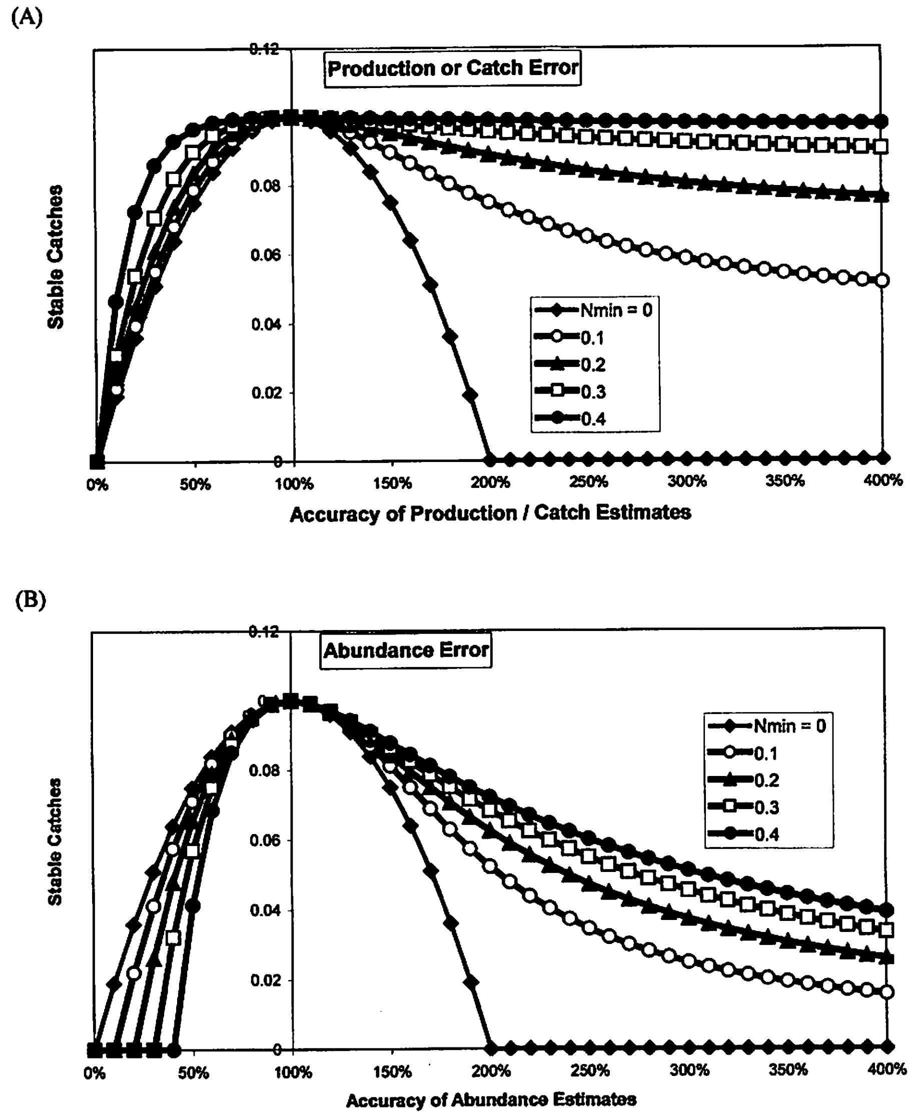
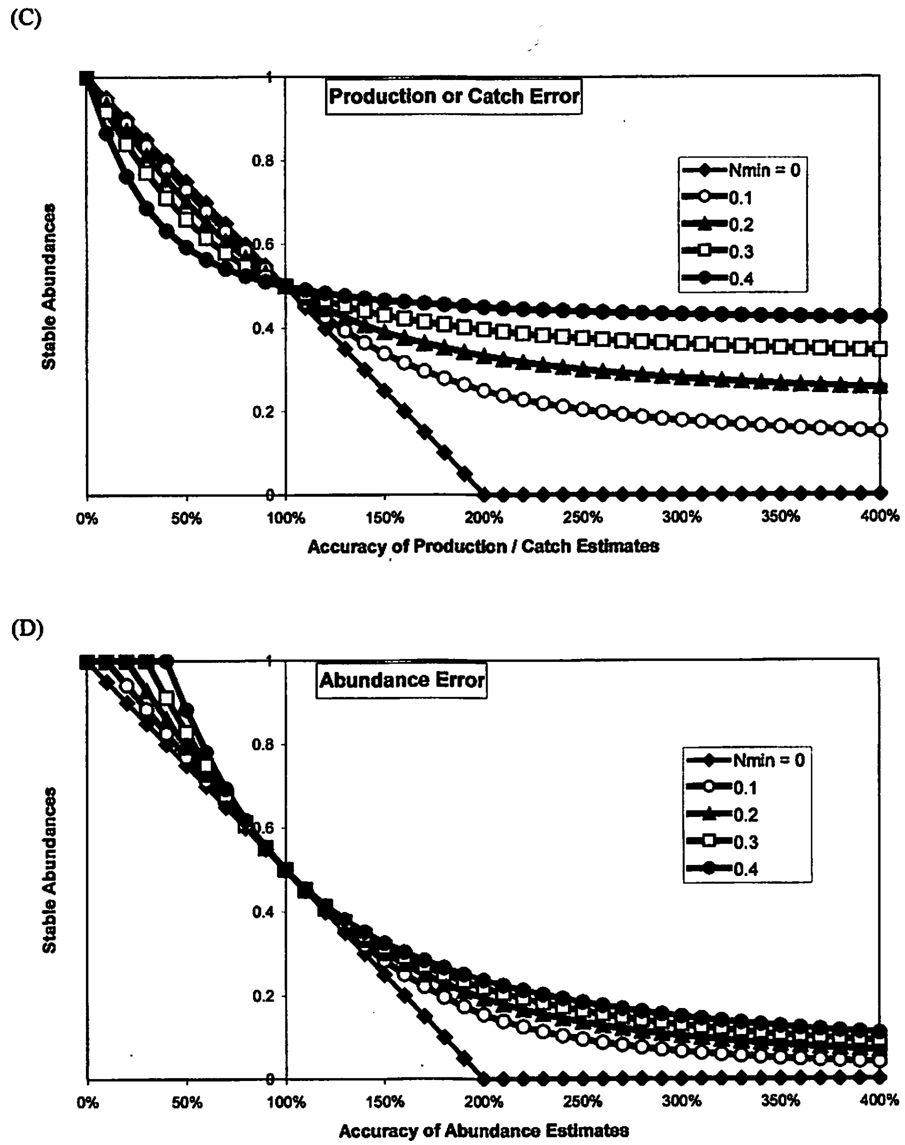


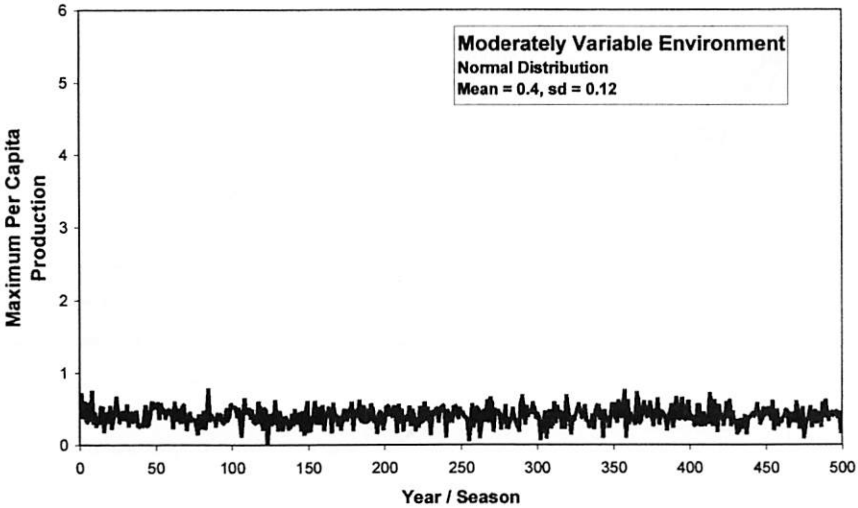
Figure 3. Resiliency in a stable environment. Stable catches (A, B) and abundance levels (C, D) *(opposite page)* were influenced by errors in estimates of production or actual catch (A, C) and of abundance (B, D). Results were obtained by means of a mathematical analysis of a logistic production fishery model and quota-setting systems as defined in Fig. 2. In all cases r = 0.4, so MSY = 0.1.

had a standard deviation of 0.12, or 30% of the mean value. Our highly variable environment was chi-square distributed, providing a greater frequency of bad years and a greater magnitude of good years. This distribution had a standard deviation of 0.6, or 150% of the mean value. Each simulation started with the population at 50% of its unfished abundance, and we assumed positive errors of 50% in production/catch estimates and 50% in abundance, leading to inadvertent overfishing. These values were intended to reflect realistic challenges managers regularly face. We should therefore expect acceptable performance from potential management systems under these circumstances.



In the moderately variable environment, average catches and abundance did in fact increase with responsiveness (Fig. 5A). The population under least-responsive management crashed to levels that were barely able to maintain fish or fishery, whereas the most highly responsive system maintained a population at 60% of desired (MSY) levels and catches near the MSY value of 0.1 despite the management errors. With increased responsiveness, catches did become more variable, and fisheries were closed more frequently. These results were more dramatic in the highly variable environment (Fig. 5B). Highly responsive systems were characterized by highly variable catches and frequent closures. Moreover, the population under most highly responsive management crashed, contradicting the pattern of higher abundance and catches with greater responsiveness. In

(A)



(B)

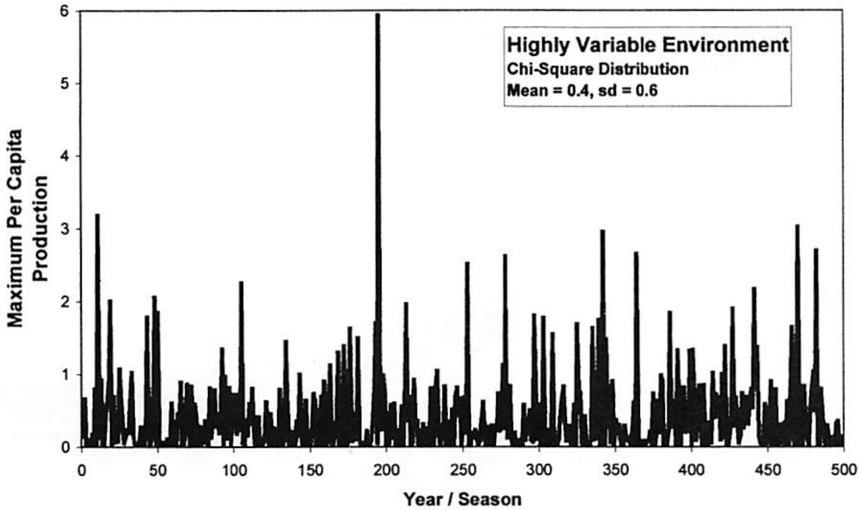
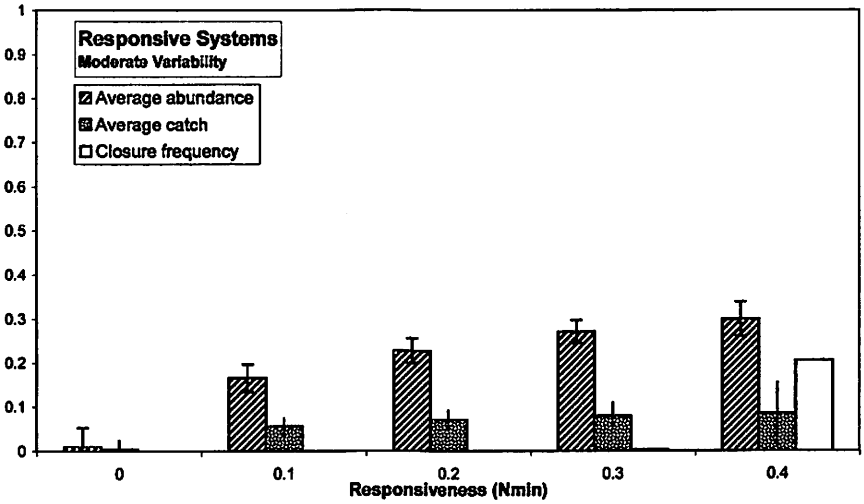


Figure 4. Simulated environments. This figure shows the two simulation environments we used. (A) The moderately variable environment was characterized by a normally distributed production constant distribution with an average of 0.4 and a standard deviation of 0.12. (B) The highly variable environment was characterized by a chi-square-distributed production constant distribution with an average of 0.4 and a standard deviation of 0.6.

this case, the overestimate of abundance after a really productive year led to a quota so high as to provoke extinction. Note, though, that this level of error caused population crashes under the least responsive system in both environments, but did so only under the most responsive management system in a highly variable environment.

(A)



(B)

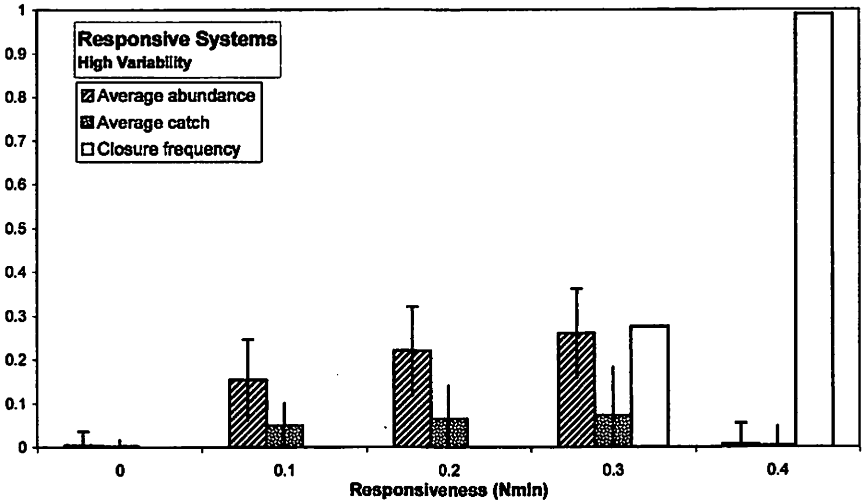


Figure 5. Resiliency in a variable environment. Performances of various management systems were compared under two variable environments and with positive management errors of 50% in catch/production and abundance estimates. Averages over 500-yr model runs are presented, and error bars represent one standard deviation. Closure frequencies are the proportion of years of the 500 during which the fishery was closed either for rebuilding or because the target species went extinct. Results are presented for the moderately variable (A) and highly variable (B) environments pictured in Fig. 4.

CONCLUSIONS

We examined the performance of management systems that ranged from constant fishing mortality to constant escapement policies. Our results showed that the more responsive policies provided higher catches in varying environments and in the face of even very large management errors. In contrast, constant-fishing-mortality policies led to population crashes with modest errors even in stable environments. Our results provide perspective on the sensitivity of responsive systems to errors in abundance estimation. Although these errors can reduce the effectiveness of highly responsive systems and even lead to population crashes, they cause even greater problems under less-responsive systems. These results contradict prior concern about the potential for crashes under highly responsive management systems and leave one major problem associated with responsive systems—variability in catches and resulting temporary fishery closures.

We are currently studying systems that are highly responsive when stocks drop below target levels but cap quotas at constant catch levels above. Preliminary results suggest these policies virtually eliminate the chance of fishery crashes and provide managers with a range of options from near-maximum catches with high variability to lower catches with virtually no variance.

Given the apparent success of these systems, it is surprising they are not used more widely or aggressively. In the United States, a few fish stocks are managed with responsive policies. The best-studied North Pacific groundfish stocks are managed according to a policy that reduces fishing mortality rates if stocks are below target levels. These reductions are modest, though, and prohibit fishing only when stocks drop to 2% of their historic abundance (NMFS, 2001). Pacific groundfish are managed according to a similar policy. Their fishing mortality rates are reduced more slowly initially but prohibit fishing when stocks drop to 10% of their historic abundance. This policy is optional, however, and fishing has not been eliminated even on stocks that have dropped to 2 to 4% of their historic abundance (Pacific Fishery Management Council, 1998). The National Marine Fisheries Service has recommended a departure from constant-fishing-mortality systems as well, but their recommendation does not begin to reduce fishing mortality rates until stocks have dropped to a threshold below the target abundance and then only recommends reducing fishing mortality to zero when a stock is extinct (Restrepo et al., 1998). Consequently, these policies are generally less responsive than the $N_{min} = 0.1$ system considered above. Many less-well-studied fish stocks are fished under constant-catch policies (Pacific Fishery Management Council, 1998; NMFS, 2001) even though these policies are substantially less responsive than constant-fishing-mortality policies and thus highly likely to lead to crashes.

Although highly responsive systems carry a cost in the form of unpredictable catches and occasional temporary closures, our results raise the question of whether these costs are not but a small price to pay for the long-term stability these systems provide to fish stocks and the ecosystems and fishing communities they support.

NEW INSIGHT INTO SCIENCE NEEDS.—Using these buffered policies would reduce our reliance on good information about productivity and actual catches, as long as catches are really reduced to zero when populations fall below the threshold population size (N_{min}). To make these buffered policies work, we would rely more heavily on two types of data—the target abundance and the current abundance. The target abundance should ideally be based on historical information about the population before fishing or a clear understanding of the biology and ecology of the stock projected through modeling, but the target can be estimated without this information. If it is the sense of managers, fishers, and others that fish are fairly abundant and catches reasonable, current abundance may serve as a reasonable target. In many cases, some rebuilding may be required, because historical analyses suggest that many stocks have been reduced dramatically over hundreds or thousands of years (Jackson et al., 2001). When we have no information about historical abundance, marine reserves can help (see Implementation, below).

Measuring current abundance also poses challenges, but there are encouraging developments on this front. Cooperative efforts between the fishing industry and scientists are providing much more timely and geographically representative data. Costs of running smaller boats part-time are lower than those of maintaining and running a large government research vessel. These costs may also be offset at least in part if boat and crew are paid for with a cut of the quota. With lower costs, much more frequent sampling is possible. Assessments are also time consuming and a bottleneck in the management process. Assessments are used primarily to estimate productivity and appropriate fishing rates. Therefore, they would become less crucial under highly responsive management systems where these pieces of information are less critical. Although assessments also provide estimates of abundance over time, abundance can also be estimated directly from fishery-independent surveys and fishery-dependent data without the long process of a full stock assessment.

PRECAUTIONARY AND ADAPTIVE MANAGEMENT.—Highly responsive systems, where a reserve population is protected from fishing, are highly effective ways to manage in a precautionary manner. Alternatively, overall fishing mortality rates could be reduced to provide adequate precaution, but this technique has some major disadvantages. If we have little information about the fish population, even a reduced fishing mortality rate could be excessive. Moreover, if we do reduce the fishing mortality rate below the optimal level, we do so at the expense of overall catch levels. In contrast, responsive systems provide the opportunity to maintain optimal catch levels over the long term while providing substantial protection against fishery collapses.

Responsive systems also provide an opportunity to learn. With experience, responsive systems allow us to discover appropriate fishing rates. Doing so requires some tracking of catches, not just abundance, although this information can be estimated with existing techniques. With this information, we can gain a thorough understanding of the productivity of a stock at and near its target level and therefore a fishing mortality rate that is appropriate. It does not provide us with an understanding of how productivity varies with stock abundance but does provide perhaps more valuable information about how much fishing is appropriate.

IMPLEMENTING RESPONSIVE SYSTEMS.—Management systems with the characteristics of the highly responsive systems we tested can be designed in several ways. The responsiveness can be engineered into the quota-setting process directly. This approach may be the best for some species but faces two challenges. First, many fisheries catch multiple species simultaneously, intentionally or otherwise. The species will almost certainly differ in their productivity, and the least productive species are likely to reach no-fishing levels before more productive species. Ricker (1958) showed that the best catch rates in a multispecies fishery were obtained when each stock was caught independently. He recommended that managers explore ways to increase selectivity of multispecies fisheries. Highly responsive quota systems face a second challenge in that they still require a fair amount of information, albeit less than conventional systems. This information includes target abundance, actual abundance, and a rough estimate of productivity, all of which may not be available for many species.

Size limits are an alternative to quotas and can contribute to a highly responsive management system. If regulations prohibit fishers from killing fish until the fish are large enough to have reproduced one or more times, the fishery is much less prone to collapse (Myers et al., 1997; Myers and Mertz, 1998; Sladek Nowlis, 2000). This approach at minimum requires information on the size at maturity of fish, and more detailed growth and fecundity information can help to ensure that a reasonable total fraction of reproductive output is assured through the size limit. This information may not be available for some species, and the approach has two additional problems. First, many types of fishing gear, from trawls to individually held hook and line in deep water, often kill fish before they are brought on board, so size limits do not prevent their death. Second, size limits are likely to cause conflict in a multispecies fishery. Gear that allows large-maturing fish to escape is likely to substantially reduce catches of smaller species.

Marine reserves, areas closed to fishing and protected from other major human impacts, are another method for creating highly responsive management policies. Reserves maintain a proportion of a species' range off-limits to fishing in much the same way that a highly responsive quota system protects a proportion of the population (N_{min}). Reserves may be a particularly useful technique when even basic information on target and current abundance are lacking. Ideally, to use reserves as a buffer against management mistakes, we should know the habitat requirements of the fish population as well as its movement tendencies. Habitat requirements may become less important when large-scale marine reserve networks are created. Under this scenario, a representative proportion of each marine habitat is included in the reserve network, so many species are likely to find suitable refuge inside the reserve network. Movement tendencies can be more troublesome. Mobile species are more likely to cross reserve boundaries and become vulnerable to fishing, so smaller effective population sizes of these species will be protected in reserves. Consequently, for these species either individual reserves will need to be large enough to incorporate the home ranges of widely migrating fish, or smaller reserves will have to be supplemented with quotas, size limits, and possibly gear restrictions.

Reserves also provide the opportunity to minimize the impacts of by-catch (Ricker, 1958; Sladek Nowlis, 2000). Even our approach used without reserves could malfunction if fisheries with multiple target species cannot avoid depleted populations while harvesting abundant ones. By protecting a core number of all fish species, reserves can lessen the chance that fishing outside the reserve will drive less-productive species to problematically low levels. Mobile species will still be of concern but to a lesser extent than under conventional management.

ECOSYSTEM-BASED MANAGEMENT.—Highly responsive management policies have several useful implications for ecosystem-based management. By virtually guaranteeing that no population will drop to low levels, responsive policies ensure that all species will perform their ecosystem functions, at least to some degree. The use of marine reserves to protect relatively unknown species can also protect habitat features and functioning ecosystems within their borders. Finally, our framework can be modified to make it even more ecosystem-based through setting of ecosystem-informed targets.

So far, we have only discussed cases where targets are set at MSY, and where N_{min} thresholds are used to help obtain the target, but abundance targets can be set with consideration of roles that species play in their ecosystems. Hard corals, for example, provide hiding places and substrate for numerous species in the coral reef ecosystem, and squid are a key prey species for numerous species in their pelagic ecosystem. In both of these cases, abundance targets should be set higher than those associated with maximum yields. In fact, for many species lower on the food web, fishing to achieve maximum yields can have widespread negative ecosystem-level effects (May et al., 1979).

Through ecosystem-based target setting, appropriately responsive quota systems, and a network of marine reserves, fisheries managers can achieve precautionary and ecosystem-based management. Given the poor state of fishery resources in the U.S. and elsewhere, though, these changes will require substantial economic costs in the short run. We propose that managers commit themselves to adopting better management systems immediately, using a phased approach to minimize the costs during the time of transition. The end result will be more productive ocean ecosystems and more stable fisheries.

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LITERATURE CITED

Casey, J. M. and R. A. Myers. 1998. Near extinction of a large, widely distributed fish. Science 281: 690–692.

Engen, S., R. Lande and B.-E. Saether. 1997. Harvesting strategies for fluctuating populations based on uncertain population estimates. J. Theor. Biol. 186: 201–212.

Environmental News Network. 2000. Bottom dropping out of the groundfish industry. Environmental News Network Website, http://www.enn.com. 17 October 2000.

FAO (Food and Agriculture Organization of the United Nations). 1998. The state of the world fisheries and aquaculture. Food and Agriculture Organization Fisheries Department, FAO Fisheries Circular.

Graham, M. 1935. Modern theory of exploiting a fishery, and application to North Sea trawling. J. Cons. Cons. Inter. Explor. Mer 10: 264–274.

H. John Heinz III Center for Science, Economics and the Environment. 2000. Fishing grounds: defining a new era for American fisheries management. Island Press, Washington, D.C. 253 p.

Hilborn, R. 1976. Optimal exploitation of multiple stocks by a common fishery: a new methodology. J. Fish. Res. Bd. Can. 33: 1–5.

Hjort, J., G. Jahn and P. Ottestad. 1933. The optimum catch. Hvalradets Skr. 7: 92–127.

Jackson, J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, J. Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell, C. B. Lange, H. S. Lenihan, J. M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J. Tegner and R. R. Warner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293: 629–638.

May, R. M., J. R. Beddington, C. W. Clark, S. J. Holt and R. M. Laws. 1979. Management of multispecies fisheries. Science 205: 267–277.

Myers, R. A. and G. Mertz. 1998. The limits of exploitation: a precautionary approach. Ecol. Appl. 8: S165–S169.

__________, J. A. Hutchings and N. J. Barrowman. 1997. Why do fish stocks collapse? The example of cod in Atlantic Canada. Ecol. Appl. 7: 91–106.

NMFS (National Marine Fisheries Service). 2001. Report to Congress: Status of the fisheries of the United States. U.S. Dept. Commerce, Silver Spring, Maryland.

Pacific Fishery Management Council. 1998. Final environmental assessment/regulatory impact review for amendment 11 to the Pacific Coast Groundfish Fishery Management Plan. Pacific Fishery Management Council, Portland, Oregon.

Quinn, T. J., II, R. Fagen and J. Zheng. 1990. Threshold management policies for exploited populations. Can. J. Fish. Aquat. Sci. 47: 2016–2029.

Reed, W. J. 1978. The steady state of a stochastic harvesting model. Math. Biosci. 41: 273–307.

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. U.S. Dept. Commerce, Washington, D.C. 54 p.

Ricker, W. E. 1958. Maximum sustained yields from fluctuating environments and mixed stocks. J. Fish. Res. Bd. Can. 15: 991–1006.

Russell, E. S. 1931. Some theoretical considerations on the 'overfishing' problem. J. Cons. Cons. Inter. Explor. Mer 6: 3–20.

Sladek Nowlis, J. 2000. Short- and long-term effects of three fishery-management tools on depleted fisheries. Bull. Mar. Sci. 66: 651–662.

Thompson, G. G. 1999. Optimizing harvest control rules in the presence of natural variability and parameter uncertainty. Pages 124–145 *in* V. R. Restrepo, ed. Proc. 5th Nat'l. NMFS Stock Assessment Workshop: Providing Scientific Advice to Implement the Precautionary Approach Under the Magnuson-Stevens Fishery Conservation and Management Act. NOAA Tech. Memo. NMFS-F/SPO-40. U.S. Dept. Commerce, Washington, D.C.

Thompson, W. F. and F. H. Bell. 1934. Biological statistics of the Pacific halibut fishery: 2. Effects of changes in intensity upon total yield and yield per unit of gear. Report of the International Fisheries Commission 8.

Walters, C. J. 1975. Optimal harvest strategies for salmon in relation to environmental variability and uncertain production parameters. J. Fish. Res. Bd. Can. 32: 1777–1784.

__________ and R. Hilborn. 1978. Ecological optimization and adaptive management. Ann. Rev. Ecol. Syst. 8: 157–188.

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APPENDIX

Equation 8 was derived as follows. We set the change in abundance *dN/dt* = 0 so that

$$0 = r\left(1 - N_{ss}\right)N_{ss} - \frac{\left(1+\varepsilon_{h/r}\right)r\left[\left(1+\varepsilon_n\right)N_{ss} - N_{\min}\right.}{2 - 4N_{\min}}$$

$$= \left(2-4N_{\min}\right)\left(1-N_{ss}\right)N_{ss} - \left(1+\varepsilon_{h/r}\right)\left[\left(1+\varepsilon_n\right)N_{ss} - N_{\min}\right]$$

$$= \left(2-4N_{\min}\right)\left(N_{ss}-1\right)N_{ss} + \left(1+\varepsilon_{h/r}\right)\left[\left(1+\varepsilon_n\right)N_{ss} - N_{\min}\right]$$

$$= \left(2-4N_{\min}\right)N_{ss}^2 - \left(2-4N_{\min}\right)N_{ss} + \left(1+\varepsilon_{h/r}\right)\left(1+\varepsilon_n\right)N_{ss} - \left(1+\varepsilon_{h/r}\right)N_{\min}$$

$$= \left(2-4N_{\min}\right)N_{ss}^2 - \left[2-4N_{\min} - \left(1+\varepsilon_{h/r}\right)\left(1+\varepsilon_n\right)\right]N_{ss} - \left(1+\varepsilon_{h/r}\right)N_{\min}$$

Solving according to the quadratic formula yields

$$N_{ss} = \frac{2-4N_{\min} - \left(1+\varepsilon_{h/r}\right)\left(1+\varepsilon_n\right) \pm \sqrt{\left[2-4N_{\min} - \left(1+\varepsilon_{h/r}\right)\left(1+\varepsilon_n\right)\right]^2 + 4\left(2-4N_{\min}\right)\left(1+\varepsilon_{h/r}\right)N_{\min}}}{4-8N_{\min}}$$

Separating the square root and solving yields

$$\sqrt{\ } = \left(2-4N_{\min} - 1 - \varepsilon_{h/r} - \varepsilon_n - \varepsilon_{h/r}\varepsilon_n\right)^2 + \left(8-16N_{\min}\right)\left(1+\varepsilon_{h/r}\right)N_{\min}$$

$$= \left(1 - \varepsilon_{h/r} - \varepsilon_n - \varepsilon_{h/r}\varepsilon_n - 4N_{\min}\right)^2 + \left(8N_{\min} - 16{N_{\min}}^2\right)\left(1+\varepsilon_{h/r}\right)$$

$$= \left(1 - \varepsilon_{h/r} - \varepsilon_n - \varepsilon_{h/r}\varepsilon_n\right)^2 - 8N_{\min}\left(1 - \varepsilon_{h/r} - \varepsilon_n - \varepsilon_{h/r}\varepsilon_n\right) + 16{N_{\min}}^2 + \left(8N_{\min} - 16{N_{\min}}^2\right)$$
$$\left(1+\varepsilon_{h/r}\right)$$

$$= \left(1 - \varepsilon_{h/r} - \varepsilon_n - \varepsilon_{h/r}\varepsilon_n\right)^2 - 8N_{\min} + 8\varepsilon_{h/r}N_{\min} + 8\varepsilon_n N_{\min} + 8\varepsilon_{h/r}\varepsilon_n N_{\min} + 16{N_{\min}}^2$$

$$= \left(1 - \varepsilon_{h/r} - \varepsilon_n - \varepsilon_{h/r}\varepsilon_n\right)^2 + 16\varepsilon_{h/r}N_{\min} + 8\varepsilon_n N_{\min} + 8\varepsilon_{h/r}\varepsilon_n N_{\min} - 16\varepsilon_{h/r}{N_{\min}}^2 + 8N_{\min}$$
$$-16{N_{\min}}^2 + 8\varepsilon_{h/r}N_{\min} - 16\varepsilon_{h/r}{N_{\min}}^2$$

$$\sqrt{\ } = \left(1 - \varepsilon_{h/r} - \varepsilon_n - \varepsilon_{h/r}\varepsilon_n\right)^2 + 16\varepsilon_{h/r}N_{\min}\left(1 - N_{\min}\right) + 8\varepsilon_n N_{\min}\left(1+\varepsilon_{h/r}\right)$$

Substituting back into the full equation yields

$$N_{ss} = \frac{2-4N_{\min} - \left(1+\varepsilon_{h/r}\right)\left(1+\varepsilon_n\right) \pm \sqrt{\left(1 - \varepsilon_{h/r} - \varepsilon_n - \varepsilon_{h/r}\varepsilon_n\right)^2 + 16\varepsilon_{h/r}N_{\min}\left(1 - N_{\min}\right) + 8\varepsilon_n N_{\min}\left(1+\varepsilon_{h/r}\right)}}{4-8N_{\min}}$$

Sladek Nowlis, J. and M.M. Yoklavich (1998) Design criteria for rockfish harvest refugia from models of fish transport. Pp. 32-40 *in* M.M. Yoklavich (ed.) *Marine Harvest Refugia for West Coast Rockfish: A Workshop*. NOAA Technical Memorandum **NOAA-TM-NMFS-SWFSC-255**. U.S. Department of Commerce, Pacific Grove, CA. 161 pp.

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MARINE HARVEST REFUGIA FOR WEST COAST ROCKFISH: A WORKSHOP

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Design Criteria for Rockfish Harvest Refugia from Models Of Fish Transport

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Abstract

We used an existing model in our initial examination of the effects of marine harvest refugia, i.e. protected areas where fishing is prohibited, on the bocaccio rockfish (*Sebastes paucispinis*) population off central and northern California. We incorporated size-specific life history information into the model, including growth, survival, and fecundity, and examined the long-term fisheries consequences of refugia. The key assumptions of the model were that larvae dispersed widely from their areas of origin and adults remained in the areas where they settled. Using the model based on these assumptions, we predict moderate-to-great potential for enhancement of bocaccio catch if refugia are established, depending on the magnitude of fishing mortality outside the protected area. From this model, we also suggest that refugia could decrease variability in annual catches. In the future, we will expand this study to include rockfish species with different life history characteristics and to examine in greater detail the effects of adult movements on the model output.

Introduction

Growing theoretical and empirical evidence supports the use of marine harvest refugia, i.e. protected areas that are closed to fishing, as a supplemental management technique for both fisheries and conservation. Potential fisheries benefits arise from the export of adult and larval fishes from the refuge to surrounding fishing areas, which theoretically can increase catches if the augmentation exceeds lost catches from reduced fishing area. Potential conservation benefits occur on population and ecosystem levels. If designed properly, a refuge can protect self-sustaining populations of harvested species (Russ 1985, Plan Development Team 1990, Roberts and Polunin 1991, 1993, Dugan and Davis 1993, Rowley 1994, Roberts et al. 1995, Bohnsack 1996, Sladek Nowlis and Roberts, unpubl. MS). Additionally, the lack of fishing in an area can prevent physical damage to the ecosystem from fishing gear (e.g. McAllister 1988) and can minimize ecosystem shifts due to selective fishing (e.g. Hay 1984, Castilla and Durán 1985, McClanahan and Shafir 1990, Roberts 1995, McClanahan et al. 1996, Pauly et al. 1998).

Most studies of marine harvest refugia have focused on tropical systems, where the majority of these protected areas exist. There is strong empirical evidence that some tropical reef fish species increase in abundance within refugia (Roberts and Polunin 1991, Dugan and Davis 1993, Rowley 1994, Bohnsack 1996, among others). Weaker empirical evidence suggests that refugia can enhance the populations in surrounding unprotected waters (McClanahan and Kaunda-Arara 1996, Russ and Alcala 1996). Two mechanisms have been proposed for this augmentation: spillover, where adults move from the refuge to fishing areas (Polacheck 1990), and larval transport, where adults within the refuge maintain outside populations through the dispersal of their offspring (Sladek Nowlis and Roberts, unpubl. MS).

Modeling the contributions of these two mechanisms -- adult spillover and larval transport -- to the enhancement of unprotected populations offers some useful insight. Polacheck (1990) modified a model by Beverton and Holt (1957) to examine fisheries enhancements via adult spillover from refugia. This cohort model included explicit parameters for fishing effort and for the propensity of the species to move across the refuge boundary. Because of the cohort approach, the supply of new recruits was not affected by the size of the adult population. Thus, Polacheck could only examine the effect of larval transport indirectly, through the spawning stock biomass -- a measure of the productive potential of the population. This model predicted fisheries enhancements from the refuge (i.e. greater fishery yields despite a smaller fishing area) only under limited circumstances. Enhancements were more likely at high fishing pressures outside the refuge and with intermediate rates of movement by adult fishes. Even under these conditions, the catch augmentation was always modest (maximum 8-20%, depending on the amount of fish movement). From these analyses, we can conclude that adult spillover from refugia results in, at best, only moderate fisheries enhancements.

Sladek Nowlis and Roberts (1997) used cyclical population models to examine fisheries augmentation by larval transport from harvest refugia. In these models, recruitment in the entire management area was affected by the population densities in both the protected and unprotected areas. The models assumed no adult spillover so as to focus on the effects of larval transport. This assumption also seems to fit well with the growing evidence regarding the limited movements of coral reef fish (Holland et al. 1993, Holland et al. 1996). In contrast to models of adult spillover, Sladek Nowlis and Roberts predicted fisheries enhancements from harvest refugia under a wide range of conditions, but specifically any time the resources are overfished (i.e. fished beyond the maximum sustainable yield). Moreover, their models predicted that refugia provide enormous catch enhancements via larval transport, particularly when fishing mortality is high. Polacheck's (1990) findings complement the result that larval transport is a more effective mechanism for providing fisheries enhancements from refugia. Spawning stock biomass, and thus potential reproductive output, was greatest in his models at the lowest rates of adult movement.

Sladek Nowlis and Roberts' (1997 unpubl. MS) study of larval transport focused specifically on coral reef fishes, whose mobility is limited and whose populations receive minimal management. Here we present preliminary results for the temperate bocaccio rockfish (*Sebastes paucispinis*) using the same model structure. This species has been economically valuable in both the commercial and recreational fisheries off central and northern California for at least the past two decades. The latest assessment of the bocaccio rockfish population in this area indicates a significant decline in biomass, and current abundance now is less than 10% of that estimated in 1970 (Ralston et al. 1996). Like most rockfish species, bocaccio have highly variable annual recruitment, and the last strong year class occurred in 1977. Initial fisheries in the early 1970's took advantage of accumulated biomass of this moderately long-lived species (e.g. maximum age is at least 50 yr [Ralston et al. 1996]), and subsequently on the survivors of the 1977 year class. Bocaccio are now at the lowest level of abundance of all federally managed rockfish species, relative to initial surveys in 1969. Consequently, the Pacific Fisheries Management Council has reduced the Acceptable Biological Catch of bocaccio in 1996 and 1997 (Ralston 1998).

For this species, we predict optimal refuge proportions, the proportion of the total managed area closed to fishing, and corresponding sustainable yields as functions of fishing mortality. We also investigate how refugia might impact yearly catch variations. Finally, we compare our results to historical records of fishing pressure on bocaccio to determine the likelihood that refugia might provide benefits to this fishery. This study is part of an ongoing investigation with several goals: (1) to determine the potential effectiveness of harvest refugia for rockfish; (2) to assess the effect of adult mobility on potential refuge benefits; and (3) to examine the influence of minimum size limits, particularly above or below the size at first reproduction, on potential refuge benefits.

Methods

We applied a model developed by Sladek Nowlis and Roberts (unpubl. MS) to a size-structured bocaccio rockfish population in central and northern California. This model examines long-term fishery yields based on various values for fishing mortality and refuge size. We represented fishing mortality by the parameter *u* (the rate of exploitation), which is the proportion of the fish population caught per year and is related to instantaneous rate of fishing mortality (F) as $u = 1\text{-}e^{-F}$. Only fish in the fishing areas were subjected to this mortality. We represented refuge size by the parameter s, the proportion of the management area closed to fishing. We also used species-specific life history information, including larval and juvenile survival rate, adult natural survival rates, von Bertalanffy growth parameters, and size-specific fecundities (see Table 1 for all estimated parameters and their references). Fish had to reach threshold sizes before they became reproductive and before they were vulnerable to the fishery.

The key assumption of the model involves transport of fishes from the closed area to nearby fishing areas. We assumed that adults did not enter or leave the refuge, whereas the larvae were dispersed widely across the refuge boundaries, resulting in an even distribution of newly settled juveniles. From previous studies (Sladek Nowlis and Roberts 1997, unpubl. MS), we know that the quantitative results of these models depend on the accuracy of all parameter values, as well as the functional relationship between stock and recruitment. In contrast, the qualitative results largely depend on the movement assumptions and the existence of some form of a density-dependent relationship between stock and recruitment.

Results

Results of the model using parameters from the bocaccio population off central and

Table 1. Model parameter values, and their sources, for the bocaccio rockfish (*Sebastes paucispinis*) population in central and northern California. [1]Modified from Yoklavich et al. (1996) and Ralston and Howard (1995); [2] best guess; [3] Rogers and Pikitch (1989); [4] Wyllie Echeverria (1987); [5] Ralston, et al. (1996); [6] Annual survival probability = e^{-M}; [7] Thomas and Bence (1992); [8] Wilkins (1980); [9] Phillips (1964).

[1]Larval Survival			Fecundity	Adult Survival	von Bertalanffy for Females
Period	Instantaneous Mortality	Survival Through Period	No. annual spawns = 1	[5]M = 0.2	[8]L_∞ = 87.76 cm [8]k = 0.11
0-20 d	0.14	0.06081	[3]Fecundity = 0.001878 x $(\text{Fork length})^{4.878193}$	[6]Annual survival probability = 0.8187	[8]t_0 = -1.73 yr
21-60 d	0.08	0.04076			[9]c = 0.0079 kg/cm^3
61-180 d	0.04	0.00823	[4]Fork length at 1st maturity = 26 cm	[7]Total length at recruitment to fishery = 40 cm	
180-365 d	0.01124	[2]0.125			[9]x = 3.1067

northern California qualitatively match those from coral reef fishery species (Sladek Nowlis and Roberts unpubl. MS). In a deterministic environment where the conditions remained constant, sustainable yields without a harvest refuge increased with annual fishing mortality until they peaked at the maximum sustainable yield (Fig. 1). They then fell as rapidly as they rose. In this latter region of the curve, where catches fell with increasing fishing mortality, the fishery can be classified as overfished. The optimal refuge proportion was non-zero, indicating that a reduction in fishing area resulted in higher catches than if the entire management area had been fished, whenever the fishery was overfished. Additionally, the optimal refuge proportion increased with fishing mortality. Yields with an optimally-sized refuge remained similar across a wide range of conditions, from u = 0.10 and no refuge to a heavy fishing mortality of u = 0.6 or more and a refuge encompassing approximately 25% of the managed area. In sum, refugia enhanced catches whenever the fishery was overfished, and the optimal refuge size increased with fishing pressure while the yields remained similar to the maximum sustainable yields.

In a stochastic environment, where larval survival varied from year to year, we found that catch variability generally decreased with refuge size (Fig. 2). This pattern was particularly common when fishing mortality was high and with initial increments in refuge size. We used the ratio of the standard deviation to the average annual catch as our measure of variability. This ratio represents the catch variability in terms of the mean. Thus, a ratio of 1 indicated that the standard deviation in annual catches is equal to the mean -- an extremely high degree of variation. Variability in annual catch increased at the highest refuge proportions for all levels of fishing because, as the population declines

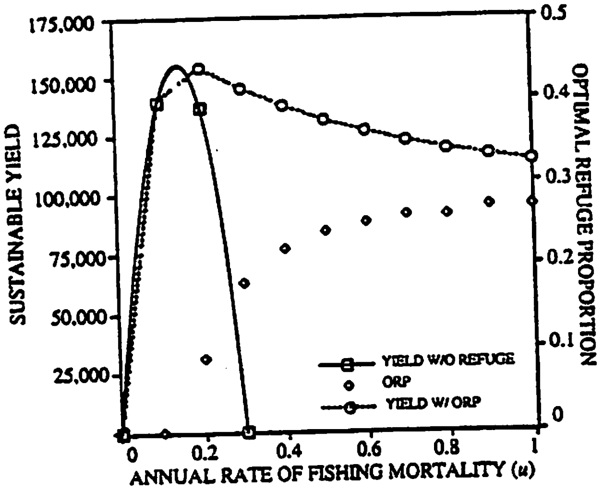

Figure 1. Central and northern California bocaccio rockfishes yield enhancements from refugia over a range of annual fishing mortalities. Long-term sustainable yields (in kg per year) without harvest refugia (solid lines and square points), optimal refuge proportions (ORP) that maximize long-term sustainable yields (dotted line and diamonds), and yields with optimally-sized refugia (dashed line and circles) are graphed against annual fishing mortality (u).

toward extinction, the catches approach zero more quickly than the variation in catch. This is not a trivial result because, if this phenomenon is realistic, we would expect to see wildly variable fisheries when they are on the verge of disaster. Specifically, there might be a few moderately productive years in an otherwise collapsing fishery.

Finally, we compared our estimate of the fishing mortality at which the bocaccio population would begin to benefit from a harvest refuge to the history of fishing mortality for this species, as determined in the most recent stock assessment (Ralston et al. 1996). Any fishing mortality above $u = 0.15$ suggests that a refuge would have been useful for augmenting catches (Fig. 3). That is, refugia might have enhanced catches consistently in the central and northern California population since the late 1970s.

Discussion

The general conclusions of this study regarding the benefits produced by harvest refugia to a fishery, and the magnitude of those benefits, are consistent with those from coral reef species (Sladek Nowlis and Roberts unpubl. MS). Specific predictions, such as how much area to close, are more species-specific. According to the best available parameters, the bocaccio has a higher population growth rate (i.e. $\lambda = 1.21$) than most coral reef fishery species previously

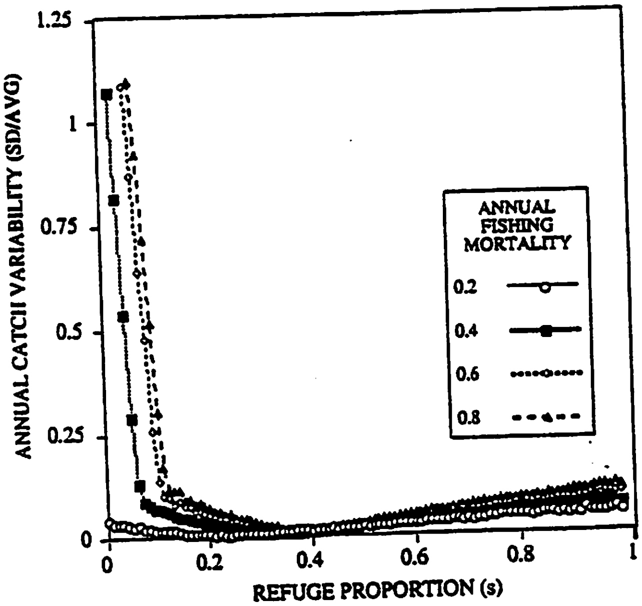

Figure 2. Effect of increasing refuge proportion (the size of the area closed to fishing, relative to the entire management area) on catch variability for the bocaccio rockfish population off central and northern California. Each line represents a different annual fishing mortality, varying from 0.2 (solid line, open circles) to 0.8 (dashed line, filled triangles). For each possible combination of fishing mortality and refuge proportion, the model was run ten times. The mean and standard deviation of the catches were for the next 100 years.

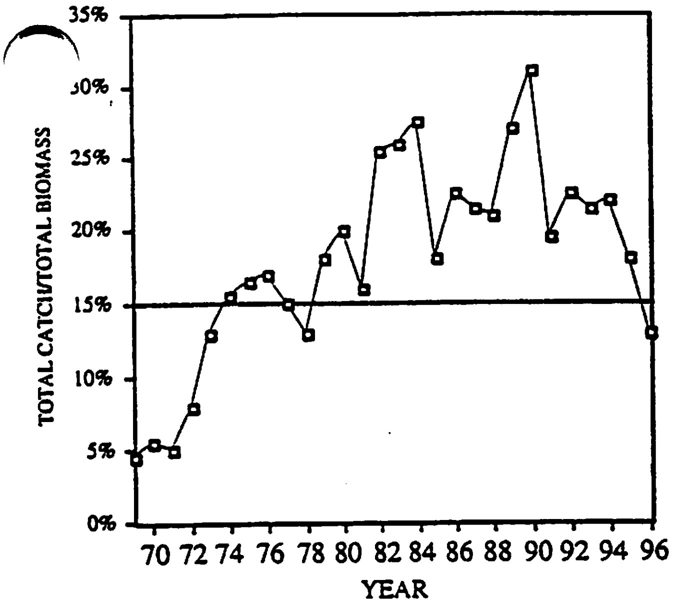

Figure 3. Potential for bocaccio rockfish off central and northern California to benefit from refugia. The solid line is the proportion of the bocaccio population's biomass caught each year from 1969 to 1996 (from Ralston, et al. 1996), a quantity approximately equivalent to fishing mortality (u) as used in the model. The dotted line is the threshold fishing mortality above which the population is overfished and would yield higher catches with refugia.

examined (mostly $\lambda < 1.16$). As a result, the model predicts that bocaccio populations can maintain a more productive fishery in the absence of refugia than those coral reef fishery species. The population growth potential is affected by virtually every parameter included in the model. Consequently, when making these comparisons across species, particularly tropical species whose life history is poorly studied, it is critical to be aware of the model's sensitivity to small errors in parameter values.

There clearly is room to improve our parameter estimates for bocaccio rockfish. In particular, we assumed that stochastic larval survivorship is normally distributed, when in reality it is characterized by a few good years interspersed among many bad ones. Future modeling efforts will more accurately incorporate variability in recruitment.

More generally, we can improve the model by including a transfer rate to represent the probability that adults move across the refuge boundary. This transfer rate was not necessary in earlier models for several reasons, including analytical simplicity, direct comparison to Polacheck's (1990) model of adult transport, and the relatively high site fidelity of many coral reef fish species and some rockfish species.

For bocaccio rockfish, however, it is unrealistic to consider only refugia that have been designed to minimize adult spillover. Because bocaccio potentially can move 150 km or more (Hartmann 1987), our current model's results will only apply to large management areas (e.g. a significant portion of the central California coast and Monterey Bay National Marine Sanctuary). Interestingly, older bocaccio rockfish seem to be relatively sedentary and inactive (M. Yoklavich pers. observation). A model with stage-specific transfer rates will allow us to design refugia for managing this species on a more accurate spatial scale.

At this point in the development of our model, we also do not account for the influence of regional oceanography and associated physical transport on the dispersal and retention of larval rockfishes and subsequent distribution of newly settled juveniles. Patterns in ocean circulation likely have significant consequences to the survival of young stages of rockfishes (Ralston and Howard 1995, Yoklavich et al. 1996), and therefore to the placement of effective harvest refugia along the coast. Additionally, because the distribution of adult bocaccio can be habitat-specific (Yoklavich unpubl. MS), the amount and quality of benthic habitat need to be accounted for in updated versions of the model in order to determine the value of the refuge.

With a revised model, we also plan to compare our results and conclusions using parameters from bocaccio with those from other rockfish species. We will examine an inshore species having relatively high site fidelity, such as the grass rockfish *(Sebastes rastrelliger)*, as well as an offshore, deepwater species with relatively high site fidelity, such as the yelloweye rockfish (*S. ruber*) or greenspotted rockfish (*S. chlorostictus*). These comparisons should give us additional insight into the relative effects of movement propensities, population growth potential, and other life history traits on refuge benefits.

From our initial findings, we suggest that harvest refugia can enhance total catches of bocaccio while dampening annual fluctuations in catches. The benefits are more likely and of greater magnitude when refugia are designed to facilitate larval transport rather than adult spillover. Thus, a harvest refugia system might best comprise individual units large enough to contain a sufficient spawning population of the target species.

Harvest refugia may provide other benefits as well, including multi-species assemblage management, enhanced persistence of heavily targeted species, reduced ecosystem damage from fishing, the maintenance of fishery-favorable genetic complexes, and increased economic potential from tourism. This formidable combination of possible benefits make refugia a management option that cannot be overlooked.

Acknowledgements

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References

Beverton, R.J.H. and S.J. Holt. 1957. On the Dynamics of Exploited Fish Populations. U.K. Ministry of Agriculture, Fisheries, and Food. Fishery Investigations, Ser. 2, Vol. 29, 533 p.

Bohnsack, J.A. 1996. Maintenance and recovery of fishery productivity. *In* N.V.C. Polunin and C.M.Roberts (eds.). Reef Fisheries. Chapman and Hall, London. Pp. 283-313.

Castilla, J.C. and L.R. Durán. 1985. Human exclusion from the rocky intertidal zone of central Chile: the effects on *Concholepas concholepas* (Gastropoda). Oikos 45:391-399.

Dugan, J.E. and G.E. Davis. 1993. Applications of marine refugia to coastal fisheries management. Can. J. Fish. Aquat. Sci. 50:2029-2042.

Hartmann, A.R. 1987. Movements of scorpionfishes (Scorpaenidae: *Sebastes* and *Scorpaena*) in the southern California Bight. CA Dept. Fish and Game, Fish Bull. 73:68-79.

Hay, M.E. 1984. Patterns of fish and urchin grazing on Caribbean coral reefs: are previous results typical? Ecol. 65:446-454.

Holland, K.N., C.G.Lowe, and B.M. Wetherbee. 1996. Movements and dispersal patterns of blue trevally (*Caranx melampygus*) in a fisheries conservation zone. Fish. Res. 25:279-292.

Holland, K.N., J.D. Peterson, C.G. Lowe, and M. Wetherbee. 1993. Movements, distribution and growth rates of the white goatfish *Mulloides flavolineatus* in a fisheries conservation zone. Bull. Mar. Sci. 52:982-992.

McAllister DE. 1988. Environmental, economic and social costs of coral reef destruction in the Philippines. Galaxea 7:161-178.

McClanahan, T.R., A.T.Kamukuru, N.A. Muthiga, Y.M. Gilagabher, and D. Obura. 1996. Effect of sea urchin reductions on algae, coral, and fish populations. Cons. Bio. 10:136-154.

McClanahan, T.R. and B. Kaunda-Arara. 1996. Fishery recovery in a coral-reef marine park and its effect on the adjacent fishery. Cons. Bio. 10:1187-1199.

McClanahan, T.R. and S.H. Shafir. 1990. Causes and consequences of sea urchin abundance and diversity in Kenyan coral reef lagoons. Oecol. 83:362-370.

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

Phillips, J.B. 1964. Life history studies on ten species of rockfish (genus *Sebastodes*). CA Dept. Fish and Game, Fish Bull. 126, 70p.

Plan Development Team. 1990. The potential of marine fishery reserves for reef fish management in the U.S. Southern Atlantic. U.S. Dep. Comm. NOAA Tech. Memo. NMFS-SEFC-261, 40p.

Polacheck, T. 1990. Year around closed areas as a management tool. Nat. Res. Model. 4:327-354.

Ralston, S. 1998. The status of federally managed rockfish on the U.S. West Coast. *In* Yoklavich, M.M. (ed.) Proceedings of Marine Harvest Refugia for West Coast Rockfishes: A Workshop. US Dep. Comm. NOAA-NMFS Tech. Memo. (This volume).

Ralston, S., J.N. Ianelli, R.A. Miller, D.E. Pearson, D. Thomas, and M.E. Wilkins. 1996. Status of bocaccio in the Conception/ Monterey/Eureka INPFC areas in 1996 and recommendations for management in 1997. *In* Pacific Fishery Management Council. 1996. Appendix Volume I: To the Status of the Pacific Coast Groundfish Fishery Through 1996 and Recommended Acceptable Biological Catches for 1997: Stock Assessment and Fishery Evaluation. Appendix B. B1-B48.

Ralston, S. and D.F. Howard. 1995. On the development of year-class strength and cohort variability in two northern California rockfishes. Fish. Bull. 93:710-720.

Roberts, C.M. 1995. Effects of fishing on the ecosystem structure of coral reefs. Cons. Bio. 9:988-995.

Roberts, C.M., W.J. Ballantine, C.D. Buxton, P. Dayton, L.B. Crowder, W. Milon, M.K. Orbach, D. Pauly, J. Trexler, and C.J. Walters. 1995. Review of the use of marine fishery reserves in the U.S. southeastern Atlantic. U.S. Dep. Comm. NOAA Tech. Memo. NMFS-SEFC-376, 31p.

Roberts, C.M. and N.V.C. Polunin. 1991. Are marine reserves effective in management of reef fisheries? Rev. Fish Bio. Fish. 1:65-91.

Roberts, C.M. and N.V.C. Polunin. 1993. Marine reserves: simple solutions to managing complex fisheries? Ambio 22:363-368.

Rogers, J.B. and E.K. Pikitch. 1989. Life history characteristics of commercially important groundfish species off California, Oregon, and Washington. U.S.Dep. Comm. NOAA Tech. Memo. NMFS F/NWC-166:1-39.

Rowley, R.J. 1994. Marine reserves in fisheries management. Aquat. Cons.: Mar. Fresh. Eco. 4:233-254.

Russ, G. 1985. Effects of protective management on coral reef fishes in the central Philippines. Proceedings of the 5th International Coral Reef Congress 4:219-224.

Russ, G.R.and A.C. Alcala. 1996. Do marine reserves export adult fish biomass? Evidence from Apo Island, central Philippines. Mar. Ecol. Prog. Ser. 132:1-9.

Sladek Nowlis, J. and C.M. Roberts. 1997. You can have your fish and eat it, too: theoretical approaches to marine reserve design. Proceedings of the 8th International Coral Reef Symposium 2:1907-1910.

Thomas, D.H. and J.R. Bence. 1992. Bocaccio. *In* Leet, W.S., C.M. Dewees, and C.W. Haugen CW (eds.) California's Living Marine Resources and Their Utilization. California Sea Grant Ext. Pub. UCSGEP-92-12:120-122.

Wilkins, M.E. 1980. Size composition, age composition, and growth of chilipepper, *Sebastes goodei*, and bocaccio, *S. paucispinis*, from the 1977 rockfish survey. Mar. Fish. Rev. 42:48-53.

Wyllie Echeverria, T. 1987. Thirty-four species of California rockfishes: maturity and seasonality of reproduction. Fish. Bull. 85:229-250.

Yoklavich, M.M., V.J. Loeb, M. Nishimoto, and B. Daly. 1996. Nearshore assemblages of larval rockfishes and their physical environment off central California during an extended El Nino event, 1991-1993. Fish. Bull. 94:766-782.

Proc 8th Int Coral Reef Sym 2:1907–1910. 1997

YOU CAN HAVE YOUR FISH AND EAT IT, TOO: THEORETICAL APPROACHES TO MARINE RESERVE DESIGN

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ABSTRACT

Increasing fishing activity on coral reefs threatens both their fisheries and biodiversity. Marine fishery reserves, areas in which fishing is permanently prohibited, offer potential economic and conservation benefits. Despite their strong potential, we lack a fundamental understanding of the design of marine fishery reserves. Computer models -- in which fish life history, movement dynamics, and fishing pressure are included -- predict that reserves can maintain productive fisheries even if they encompass large proportions of management areas. Moreover, the models suggest that reserves will increase the persistence of easily over-harvested species and will also decrease year-to-year variations in catches. The reserve design that will maximize long-term fish catches depends on the life history, larval dispersal, and adult movement dynamics of the target species, as well as the fishing effort in the management area. Nevertheless, these computer models predict that the use of marine fishery reserves is an effective general fishery management technique, useful in particular for multi-species fisheries, and which also provides significant conservation benefits.

INTRODUCTION

World-wide, marine fisheries are in distress. In recent years figures show that global fishery yields have dropped for the first time despite expanding fleets, more efficient equipment, and efforts directed at previously unexploited stocks (Food and Agriculture Organization 1995). Many fish stocks declined in areas where resources are lacking for adequate management. However, even intensively managed fisheries have crashed in recent years (Norse 1993).

Marine fishery reserves, areas permanently closed to fishing, have the potential to maintain productive fisheries even in areas where management resources are lacking. As a fishery management strategy, reserves offer simple enforcement, conservation benefits, and fishery enhancements (e.g., Roberts and Polunin 1991; Rowley 1994; Bohnsack 1996). Reserves are simpler to enforce than traditional management techniques because they do not require inspection of catches or gear. Moreover, once established, the reserves become a common resource for fishers and encourages them to police themselves. Reserves also provide conservation benefits by protecting fish populations from over-fishing (Sladek Nowlis and Roberts, in preparation) and ecosystems from damaging fishing practices (Roberts 1995). Finally, management areas with reserves can provide catches that meet or exceed the maximum catches if a reserve were not used, while also reducing uncertainty in annual fish catches caused by environmental fluctuations (Sladek Nowlis and Roberts, in preparation).

To attain the long-term benefits associated with marine fishery reserves, fishers and managers must accept short-term losses when reserves are established because fishers have less area to fish. Using existing models of marine fishery reserve function (Sladek Nowlis and Roberts, in preparation), we asked several questions about these short-term losses.

1) How does fishing intensity, assumed to remain constant before and after reserve establishment, affect the speed at which reserve benefits offset losses from reduced fishing grounds?

2) How does reserve size affect the speed at which reserve benefits offset losses from reduced fishing grounds?

3) Is fishing intensity or reserve size more important in determining the rate at which benefits are achieved?

4) Can losses from reduced fishing grounds be minimized by phasing in reserves over several years and does a phasing in approach result in higher or lower overall catches in the long run?

METHODS

To investigate these questions, we developed size-classified life history models, described by Sladek Nowlis and Roberts (in preparation). In the models, fish eggs develop into newly settled fish and these fish grow through several size classes. Fish face natural and fishing mortality (the latter only outside reserves) and can contribute to the future population through reproduction. We varied reserve size (proportion of the total management area in which fishing is prohibited) and fishing intensity (proportion of the naturally-surviving fishery-recruited fish that are caught each year) and examined their influence on long-term fishery yields.

The key assumptions of our models addressed movement of fish and eggs or larvae between reserve and fishing areas. We assumed that adults stay in the area where they settled while eggs and larvae disperse widely across reserve boundaries. The majority of coral reef fish and invertebrate species disperse narrowly as adults and widely as larvae (Boehlert 1996). Even in cases where adults move widely or eggs and larvae move short distances, the assumptions of the model can be met and valid conclusions drawn if the reserve area is broken into appropriately-sized units (Sladek Nowlis and Roberts, in preparation).

Using the best available life history data for two exploited species of Caribbean reef fish, the queen trigger fish *Balistes vetula* (Aiken 1975; Houde 1989) and the white grunt *Haemulon plumieri* (Darcy 1983; Houde 1989), we adapted these models to examine short-term losses in fisheries associated with reserve establishment. In all cases, we determined the stable size-distribution and density of fish under a fixed fishing intensity and no reserve. We then took this stable population, created a reserve within its geographic range, and tracked annual catches for up to 100 years, making particular note of the time it took until fishery yields exceeded their productivity prior to reserve establishment.

We determined the effect of fishing intensity by establishing 20% reserves in queen trigger and white grunt fisheries over a range of fishing intensities. We studied the effect of reserve size by establishing a variety of reserve proportions for each species at a fixed fishing intensity of 0.2. We then studied the interaction of reserve size and fishing intensity by comparing the responses of fisheries that varied in their fishing intensities and in their reserve proportion. To compare equivalent pairs of fishing intensity and reserve proportion, we chose the optimal reserve proportion -- the proportion that produced the highest long-term yields in our models -- at various fishing intensities. For both of these species, sustainable yields are similar across a wide array of fishing intensities if optimal reserve proportions are used (Sladek Nowlis and Roberts, in preparation).

Finally, we determined how a phasing in of reserves might affect both short-term and long-term fishery yields. To do so, we used a white grunt population experiencing a fishing intensity of 0.2. We established an optimally-sized reserve of 37% and compared the short-term, long-term, and cumulative fish catches when the reserve was phased in over 1, 5, and 10 years. We phased in reserves by adding equal portions to the reserve each year for the time specified. Consequently, we established the whole 37% reserve at once for the 1 year case while for the 10 year case, we closed 3.7% of the management area each year for 10 years.

RESULTS

Reserves were only effective at increasing fish yields when the fisheries were over-fished in the absence of a reserve (Sladek Nowlis and Roberts, in preparation). In the absence of a reserve, the queen trigger had its maximum sustainable yield at a fishing intensity (FI) of 0.094, while the white grunt had its maximum sustainable yield at a fishing intensity of 0.117. Both were over-fished at higher fishing intensities and we only examined these cases (i.e., those where reserves would lead to a long-term increase in catches).

Heavily over-fished fisheries recovered fastest from losses associated with reserve establishment (Fig. 1). Both queen trigger and white grunt populations rebounded above pre-reserve productivity faster the more heavily they were over-fished. We established a 20% reserve for the queen trigger, the optimal size for this species under a fishing intensity of 0.125. While the lightly over-fished population (FI = 0.125) took nearly 30 years to recover from lost fishing grounds, the heavily over-fished population (FI = 0.2) recovered in half the time. Similarly, the heavily over-fished white grunt population (FI = 0.225) recovered more quickly than less-heavily over-fished populations when a 17% reserve was established (the optimal size for the white grunt under 0.15 fishing intensity). Both species recovered more quickly from reserve establishment losses if they were in bad shape to begin with.

Results were less clear when we held fishing intensity constant and varied only reserve size. When we established reserves for a heavily over-fished queen trigger population (FI = 0.2), larger reserves led to faster recovery than small reserves (Fig. 2). In contrast, smaller reserves made up losses faster when we established them for a less-heavily over-fished white grunt population (FI = 0.2). In both cases, the larger reserves (optimal size in both cases) lead to higher catches than smaller reserves within 30 years. However, recovery time could increase or decrease with increasing reserve size, depending on the fish species and fishing intensity.

When we varied both fishing intensity and reserve size we found that fishing intensity was generally more important in determining the speed to recovery from fishery reserve establishment losses. To compare equivalent pairs of fishing intensity and reserve proportion, we chose the optimal reserve proportion for any given fishing intensity. We found, as we did when just varying fishing intensity, that reserves led to more rapid population recovery when the populations were heavily over-fished prior to reserve establishment (Fig. 3). As was true when just fishing intensity was varied, heavily over-fished queen trigger populations (FI = 0.2) reached pre-reserve productivity levels in half the time of lightly over-fished populations (FI = 0.125). White grunts showed an even more dramatic pattern, with the most heavily over-fished populations recovering in one-third the time of the least over-fished. Despite their quicker recovery to pre-reserve productivity, note that the heavily over-fished populations are still the least productive after 30 years.

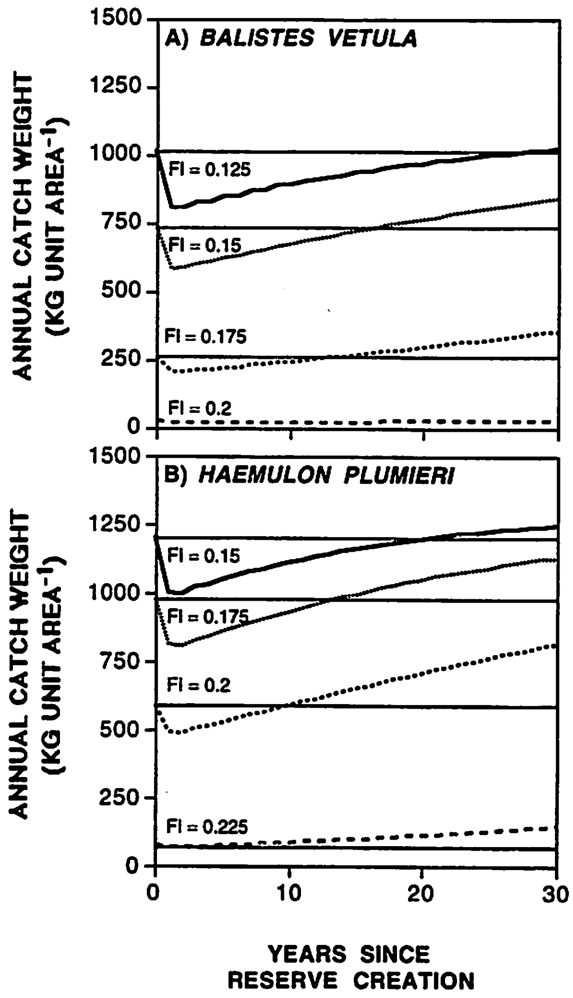

Fig. 1: Fishing intensity and fishery recovery following reserve establishment. (A) *Balistes vetula*, the queen trigger fish, with 20% of the management area closed to fishing. (B) *Haemulon plumieri*, the white grunt, with 17% of the management area closed to fishing. Each line represents the annual fish catches (in kg unit area^{-1}) following reserve creation. Catches in year 0 were the stable catches at the specified fishing intensity (FI) with no reserve. Fisheries recovered to pre-reserve productivity (indicated by thin solid lines) faster the heavier the fishing intensity.

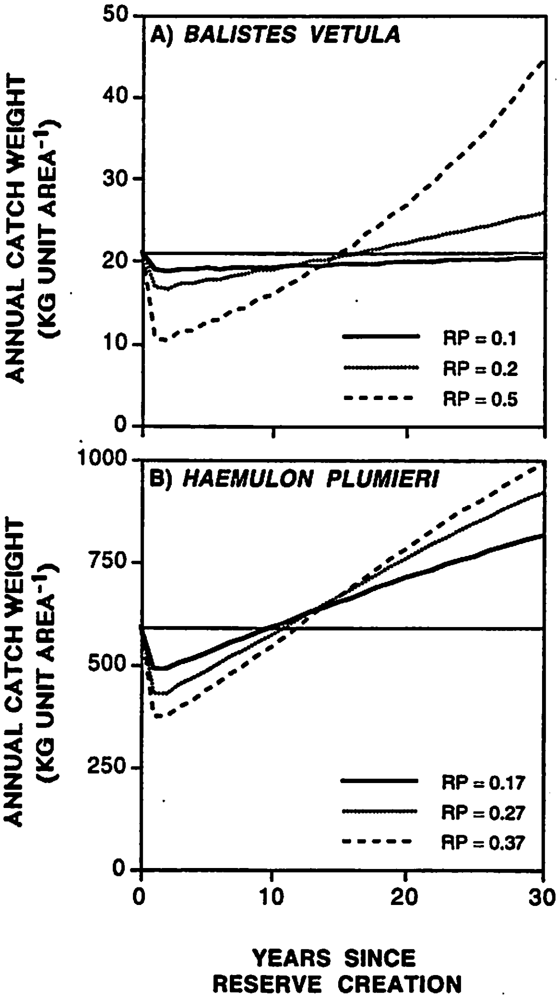

Fig. 2: Reserve size and fishery recovery following reserve establishment. (A) *Balistes vetula*, the queen trigger fish, with a fishing intensity of 0.2. (B) *Haemulon plumieri*, the white grunt, with a fishing intensity of 0.2. Reserves varied in size (RP = reserve proportion) from small reserves to the optimal reserve proportion for each species at this fishing intensity. Speed of recovery to pre-reserve productivity (indicated by the thin solid lines) did not vary consistently with reserve size. Smaller reserves caused smaller initial losses in fishery productivity but also reaped fewer rewards in the long run.

When we phased in reserves over periods of up to 10 years, we found that we could reduce short-term reserve losses but at the expense of long-term reserve gains (Fig. 4a). We established an optimally-sized 37% reserve for a white grunt population experiencing a fishing intensity of 0.2. The longer we took to phase in the reserve, the shallower the initial dip in annual fish catches. Therefore, fishers and managers would have relatively better early years if reserves are phased in over a long period of time. However, phased in reserves took longer for catches to exceed pre-reserve levels and had lower annual catches for over 50 years. We used cumulative catches -- the sum of all catches since reserve establishment -- to examine whether short-term gains or long-term losses from a phasing in approach were greater. To aid in comparison, we subtracted the cumulative catches from a reserve created in a single year from all cumulative catches (Fig. 4b). These results show that while the total catch over the first 10 to 15 years is higher when reserves are phased in, the total catch over the first 20 or more years is higher when reserves are created in a single year (Fig. 4b). In the long run, total catches since reserve creation from management areas where reserves are created in a single year exceed total catches since reserve creation from management areas where reserves are phased in.

DISCUSSION

Reserves do cause short-term fishery losses because they remove fishing grounds. However, these losses may be minor in many cases and can be addressed through other management measures. In all of our runs, short-term losses were worst within the first year or two, and catches dropped from previous years by a proportion similar to the reserve proportion. When 50% of an area was set aside as a reserve, this could mean a 50% reduction of catch in year 1 (Fig. 2a). However, very large reserves were only appropriate for increasing yields of extremely heavily over-fished populations (Sladek Nowlis and Roberts, in preparation). In these cases, fishers have so much to gain from reserve establishment and it may be more tractable to make up their relatively small (in magnitude, not proportion) short-term losses through other programs, including subsidies or alternate business opportunities. We found that these heavily over-fished fisheries also recovered most quickly, in as little as 7 years (Fig. 2b). Note that at 'recovery' these previously heavily over-fished fisheries still have years before they reach their peak potential productivity. Less heavily over-fished fisheries took longer to recover but also experienced smaller losses and higher overall yields.

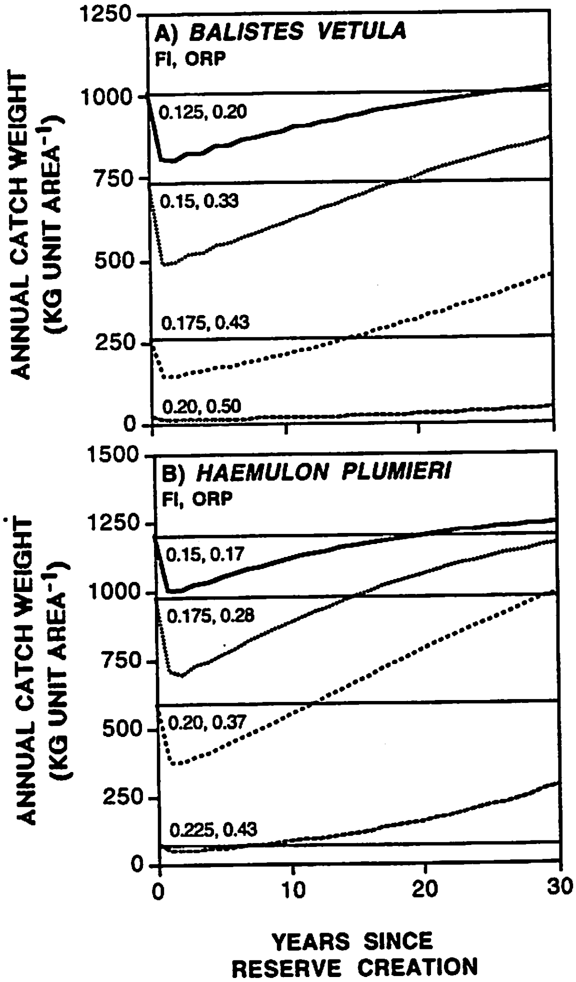

Fig. 3: Combined effects of fishing intensity and reserve size on fishery recovery following reserve establishment. A) *Balistes vetula*, the queen trigger fish. (B) *Haemulon plumieri*, the white grunt. For equivalency, we used optimal reserve proportions for each fishing intensity. As with patterns based on fishing intensity, fisheries recovered faster the more heavily fished they were prior to reserve establishment. Reserve sizes primarily affected the magnitude of initial fishery losses but did not significantly influence the speed of recovery (compare to Fig. 1).

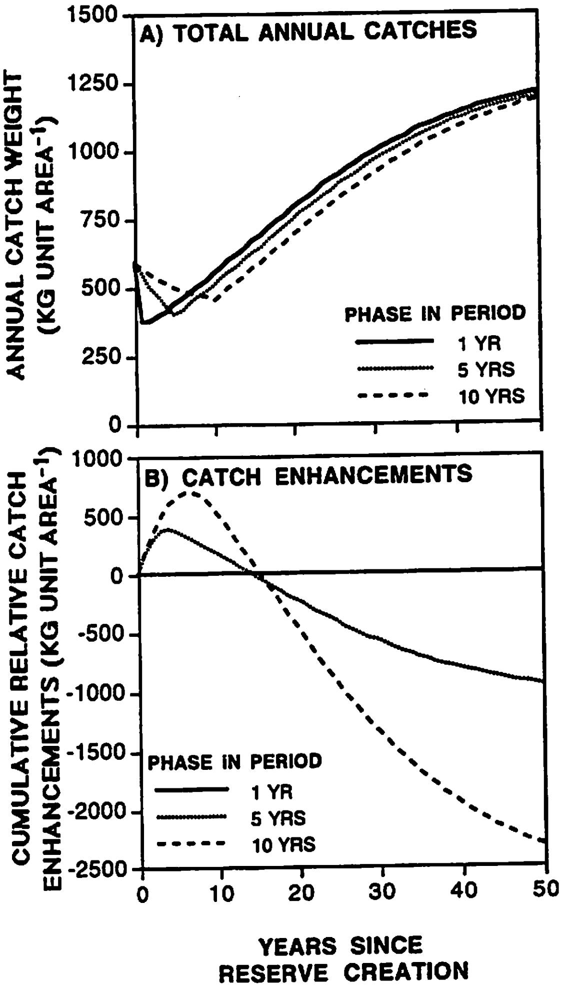

Fig. 4: Phasing in reserves, fishery recovery, and long-term effects. (A) Total annual catches. (B) Cumulative relative catches, calculated by adding all the fish catches since reserve creation and subtracting the cumulative catches for the same period of a reserve created all at once. Reserves phased in over a series of years had better initial catches because more fishing grounds were available in early years. However, the phasing in of reserves delayed reserve benefits, condemning phased in reserves to lag behind a reserve created all at once for 50 more years. In the long run, cumulative catches are lower for reserves phased in over several years, as the reduced initial losses from phasing in are outweighed by the delay in reserve benefits.

We were encouraged that reserve size had less of an effect on recovery time than fishing intensity. It is not surprising because reserve size influences both the amount of short-term loss and the amount of long-term benefit. For example, we might expect a small reserve to recover more quickly because it will create smaller short-term losses in fishing grounds. However, we might also expect it to recover more slowly because a small reserve will provide fewer long-term benefits. These short- and long-term forces balance out in a way that is not particularly dependent on reserve size. This result implies that managers can design reserves to effectively enhance fisheries in the long run without strongly influencing how long it takes for the fishery to recover to pre-reserve productivity. The time to recovery is affected more by the state of the fishery prior to reserve establishment. If it was in fairly good shape, the recovery will be slow but the losses mild (in proportion, not magnitude). If it was in poor shape, the recovery will be more rapid but the initial losses more extreme.

If reserve establishment will cause unacceptable hardships, they can be phased in to reduce short-term losses (Fig. 4a). The worst year when a reserve was phased in over 10 years resulted in approximately half the productivity loss of the same reserve created all at once. The disadvantage of the phasing in approach is that it slows the speed at which reserve benefits accumulate. For example, the reserve that was phased in over 10 years took several years longer before productivity had returned to pre-reserve levels. Moreover, the phased in reserve continued to under-perform the reserve established all at once for over 50 years. Consequently, the cumulative catches -- the total weight of fish caught since the reserve was created -- was lower when reserves were phased in. The phasing in approach can be useful if managers need to minimize initial losses in productivity, but it actually reduces the speed of recovery and long-term catches.

Marine fishery reserves have many important benefits, including ease of enforcement, increased long-term fish yields, and enhanced conservation of marine species and ecosystems (Bohnsack 1996). However, we cannot forget that reserve establishment will cause short-term losses to fishers. Fortunately, these losses can be mild or quickly recovered, depending on the status of the fishery prior to reserve establishment. If these short-term losses are likely to over-burden fishers, managers can opt for a phasing in approach, or they could compensate fishers for losses in the early years following reserve establishment. Phasing in offers higher short-term catches, but larger losses in time to recovery and future productivity.

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REFERENCES

Aiken KA (1975) The biology, ecology and bionomics of the triggerfishes, Balistidae. In: Munro JL (ed) Caribbean coral reef fishery resources. Institute for Caribbean and Latin American Regional Management pp 191-276.

Boehlert GW (1996) Larval dispersal and survival in tropical reef fishes. In: Polunin NVC and Roberts CM (eds) Reef fisheries. Chapman and Hall, London pp 61-84.

Bohnsack JE (1996) Maintenance and recovery of fishery productivity. In: Polunin NVC and Roberts CM (eds) Reef fisheries. Chapman and Hall, London pp 283-313.

Darcy GH (1983) Synopsis of biological data on the grunts *Haemulon aurolineatum* and *H. plumieri* (Pisces: Haemulidae). National Oceanic and Atmospheric Administration Technical Report NMFS Circular 448.

Food and Agriculture Organization (1995) The state of the world fisheries and aquaculture. Food and Agriculture Organization of the United Nations, Rome.

Houde ED (1989) Comparative growth, mortality and energetics of marine fish larvae: temperature and implied latitudinal effects. Fish Bull, US 87:471-495.

Norse EA (1993) Global marine biological diversity. Island Press, Washington, DC.

Roberts CM (1995) Effects of fishing on the ecosystem structure of coral reefs. Cons Biol 9:988-995.

Roberts CM and Polunin NVC (1991) Are marine reserves effective in management of reef fisheries? Rev Fish Biol Fish 1:65-91.

Rowley RJ (1994) Marine reserves in fisheries management. Aquat Conserv 4:233-254.

Fisheries benefits and optimal design of marine reserves

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Abstract.–We used fishery population models to assess the potential for marine fishery reserves, areas permanently closed to fishing, to enhance long-term fishery yields. Our models included detailed life history data. They also included the key assumptions that adults did not cross reserve boundaries and that larvae mixed thoroughly across the boundary but were retained sufficiently to produce a stock-recruitment relationship for the management area. We analyzed the results of these models to determine how reserve size, fishing mortality, and life history traits, particularly population growth potential, affected the fisheries benefits from reserves. We predict that reserves will enhance catches from any overfished population that meets our assumptions, particularly heavily overfished populations with low population growth potential. We further predict that reserves can enhance catches when they make up 40% or more of fisheries management areas, significantly higher proportions than are typical of existing reserve systems. Finally, we predict that reserves in systems that meet our assumptions will reduce annual catch variation in surrounding fishing grounds. The fisheries benefits and optimal design of marine reserves in any situation depended on the life history of the species of interest as well as its rate of fishing mortality. However, the generality of our results across a range of species suggest that marine reserves are a viable fisheries management alternative.

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Fish. Bull. 97:604–616 (1999).

Fishing activity impacts the marine environment in several ways. Individual species risk severe declines from overexploitation. Over two-thirds of all fisheries world-wide are classified as fished beyond capacity or in danger of becoming so (FAO, 1995), and higher trophic levels are particularly affected (Pauly et al., 1998). Along with target species, fishing can reduce the populations of nontarget species that are caught and discarded. The ecosystems that support the fisheries also face risks. Fishing can cause biological damage to ecosystems when the reduction of key species or trophic levels causes ecological shifts (e.g. Hay, 1984; Castilla and Durán, 1985; Hay and Taylor, 1985; Durán and Castilla, 1989; McClanahan and Shafir, 1990; Roberts, 1995; McClanahan et al., 1996; Pauly et al., 1998). Fishing can also cause physical damage to ecosystems, particularly when dwindling fish catches promote the incentive to use damaging fishing practices (McAllister, 1988).

Reserves can protect the ecosystems within them from damaging fishing practices and have the potential to reestablish a natural ecosystem balance (Russ, 1985; Plan Development Team, 1990; Roberts and Polunin, 1991; Dugan and Davis, 1993; Roberts and Polunin, 1993; Rowley, 1994; Roberts et al., 1995; Bohnsack, 1996). Field studies have generally demonstrated that fish stocks build up within a protected area (Roberts and Polunin, 1991; Dugan and Davis, 1993; Rowley, 1994; Bohnsack, 1996, and references within) but much less information exists on fishery enhancements.

In theory, reserves can maintain productive fisheries by protecting a critical stock within their borders. These stocks may enhance catches through adults that grow larger in the reserve and then migrate to fishing areas (adult spillover), or through enhanced recruitment in fishing areas due to increased population fecundity from the reserve (larval transport). In practice, fisheries benefits from reserves have rarely been demonstrated or even measured. This lack of field evidence reflects the difficulty of performing controlled and replicated experiments in unpredictable political and biological systems.

The few existing field studies addressing fisheries benefits from reserves show promise. A marine fishery reserve encompassing over 60% of the former fishing grounds north of Mombasa, Kenya, showed a 110% increase in catch per unit of effort after only two years (McClanahan and Kaunda-Arara, 1996). Total catches had not yet met those prior to reserve establishment, but trends looked favorable. On Apo Island, Philippines, total fish density and species richness had increased by over 400% in both the reserve and the fishing grounds after 11 years of reserve protection (Russ and Alcala, 1996). Large fish were particularly abundant in the fishing grounds near the border of the reserve, possible evidence of adult spillover. These same authors showed previously that overall catches dropped more than 50% two years after the re-opening of a closed area on Sumilon Island, Philippines, despite the increased fishing area (Alcala and Russ, 1990), suggesting that the reserve had provided enhancements to surrounding fishing grounds.

These field studies show that under certain circumstances, reserves are likely to produce fisheries enhancements. However, models are also necessary because they allow more general analyses of the conditions under which reserves are likely to produce benefits and of the design attributes that will maximize these benefits. By making use of controlled replicates and large-scale manipulations, models can provide a theoretical background on which to interpret field results.

Several authors have built and analyzed models of marine fishery reserves. These models can be classified as those examining adult spillover (Beverton and Holt, 1957; Polacheck, 1990; DeMartini, 1993) and those examining larval transport (Quinn et al., 1993; Man et al., 1995; Holland and Brazee, 1996; Sladek Nowlis and Roberts, 1997; Holland et al.[1]). All of these models predict fisheries enhancements from reserves in at least some situations, particularly under heavy exploitation. However, the predicted enhancements were small and uncommon for the adult spillover models. Previous models have examined a variety of factors that influence potential reserve benefits, including adult movement tendencies (Polacheck, 1990: DeMartini, 1993), individual growth rate (DeMartini, 1993), Allee effects (Quinn et al., 1993), metapopulation patch dynamics (Man et al., 1995), and socioeconomic factors (Holland and Brazee, 1996: Holland et al.[1]). None of these examined the effect of population growth potential on reserve benefits.

In order to fill this gap, we built a set of models looking at reproductive enhancement and larval transport as mechanisms for providing reserve benefits. We analyzed these models with particular emphasis on how reserve size, fishing mortality, and life history traits, particularly population growth potential, affect long-term fishery yields. We also analyzed the short-term consequences of reserve establishment and these results are presented elsewhere (Sladek Nowlis and Roberts, 1997).

We used our models to achieve several goals. First, we wanted to identify conditions that favored the success of reserves at enhancing fisheries. Second, we wanted to establish design criteria to help maximize the benefits that could accrue from a closed fishing area. Third, we wanted to assess whether reserves can decrease year-to-year variation in catches. Finally, we wanted to provide guidelines for future field research through the identification of important but poorly understood biological processes and through the generation of testable predictions about the design and function of marine fishery reserves.

Methods

Our basic model followed yearly changes in a population separated into size categories. Although categorization by age is more common than by size, we felt size better represented size-dependent processes such as reproduction and fishing mortality (Polunin and Roberts, 1996). Each size category contributed to future populations through some simple rules (Fig. 1).

We used the best-available estimates of size-based fecundity and larval survivorship for various species (see Table 1). Little is known about larval survivorship in fish, especially for coral reef species (Boehlert, 1996). The best estimates we could find came from an analysis of larval performance across a global array of ambient temperatures (Houde, 1989). Houde used linear regression on data from various studies to relate ambient temperature to fish larval duration and daily survivorship. This process produced statistically significant and predictive, but crude, relationships that could then be combined to estimate total larval survivorship. At a temperature of 26°C, Houde's estimate of survivorship for larvae through the entire larval stage was 5×10^{-5} (see Appendix for equations). Whenever we had additional information about larval stage duration, we used it along with Houde's temperature-based estimate for daily survivorship to produce our estimate of total larval survivorship.

Natural mortality estimates were also taken from the literature (see Table 1). Those adults that survived had the additional possibilities of either growing to the next size class or staying in the same one. We used von Bertalanffy growth parameters (Ricker, 1975) to determine the chance that a fish of one size class grew to the next in a given year (Fig. 2). Von Bertalanffy parameters describe the growth of individual fish and are widely estimated in the literature (see Table 1 for the estimates that we used and

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[1] Holland, D. S., J. B. Braden, and R. J. Brazee. 1995. Managing artisanal fisheries with marine fishery reserves: an alternative to managing catch or effort. Environmental and Natural Resources Policy and Training/Midwest Universities Consortium for International Activities Supplementary Paper 3. 36 p.

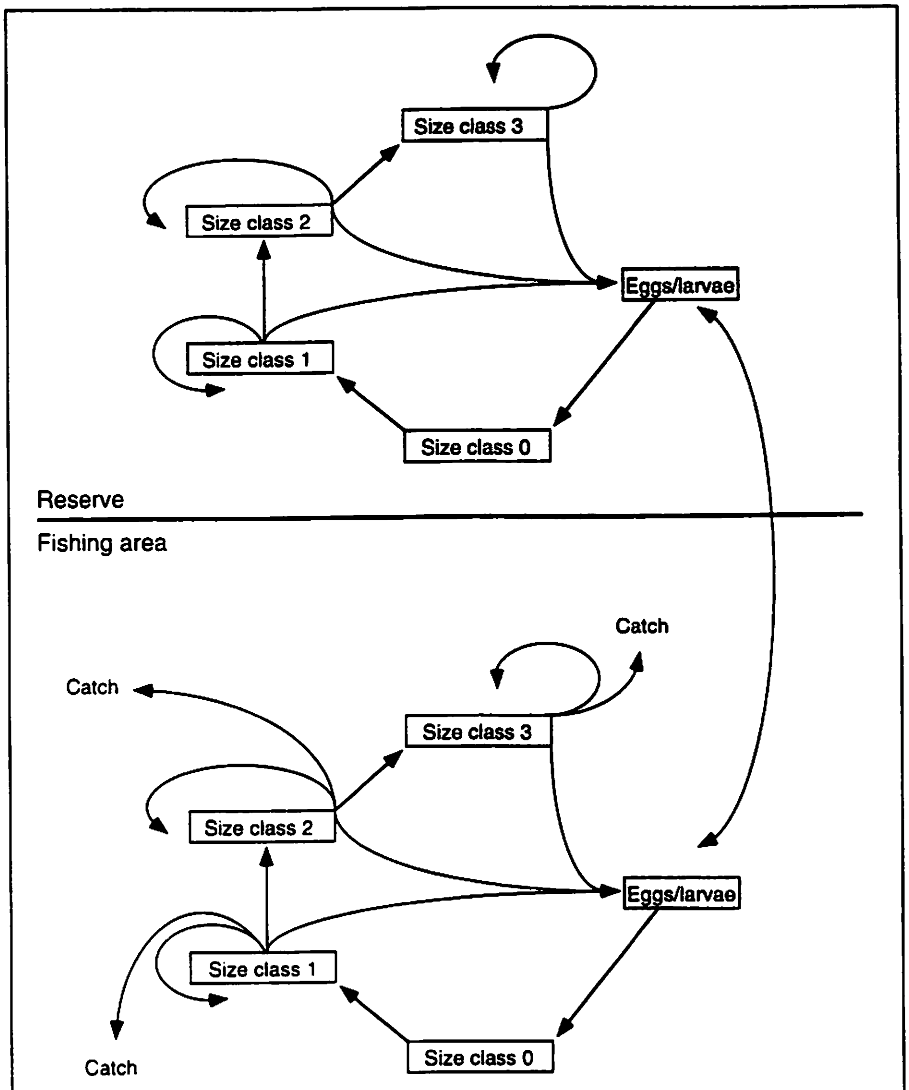

Figure 1

Flow diagram for the model. Lines represent contributions from one size class to itself and others. Populations in the fishing area and reserve interacted through larval dispersal between them, whereas adults remained in the area where they settled. Populations in the fishing area experienced fishing mortality ("catch"), whereas reserve populations did not. Otherwise, populations in reserve and fishing areas experienced the same population phenomena. This figure is an example of the model construction for white grunt, *Haemulon plumieri*—the number of size classes, size class at reproductive maturity, and size class at recruitment to the fishery differed for other species.

Appendix for relevant equations). To convert these continuous measures into probabilities, we used the standard von Bertalanffy parameters to estimate the lengths of the smallest and largest individual in each size category for the following year. We calculated the proportion of this size range that fell into the next size class and used this value to represent the probability that a fish of this size class grew to the next size class. The remaining individuals stayed the same size over the next year, with a probability determined by subtracting the probability of growing from 1. We chose the size-class interval for each species such that newly settled fish had exactly 100% chance of growing to the next size class during the first year (see Appendix for formula). Consequently, fish were never able to grow more than one size class in a year. As with all von Bertalanffy growth relationships, growth slowed with age—in our case from

Table 1

Parameter values for four fishery species. See Appendix for detailed explanation of parameters. Parameter values were taken from the literature where available. Where we could not find values in the literature, we made our best educated guess by looking at a related species, by running several values until we got realistic population growth (i.e. growth in the absence of fishing), or by measuring some other consequence, for example length at recruitment, to check for realism. References: (1) Plaut (1993); (2) Houde (1989); (3) Plaut and Fishelson (1991); (4) Aiken (1983); (5) Darcy (1983); (6) Thompson and Munro (1983); (7) best guess.

Species	Fecundity	Larval survival	Adult survival	Growth
Panulirus penicillatus, Red Sea spiny lobster	r = 4 spawns per year × 2.715 × $L^{2.581}$ eggs per spawn Mature at L = 50 mm (1)	D = 35.5 days (1) Z = 0.3454 (2) $N = 5 \times 10^{-6}$ (2)	v = 0.8 (M=0.223) Fishery recruitment = 25.7 mm (7)	k = 0.1066 L_∞ = 84.7 mm t_0 = 0 $c = 6.43 \times 10^{-4}$ y = 2.89 (3)
Balistes vetula, queen triggerfish	r = 3 spawns per year × 73 per g body weight Mature at size 23.5 cm (4)	D = 28.7 days (2) Z = 0.3454 (2) $N = 5 \times 10^{-5}$ (2)	v = 0.07427 (M=2.6) (4) Fishery recruitment = 17 cm (7)	k = 0.57 (4) L_∞ = 45 cm (4) t_0 = −0.5 (7) c = 0.05164 (4) y = 2.875 (4)
Haemulon plumieri, white grunt	r = 1 spawn per year × (626 per g body weight − 93000) Mature at size = 22 cm (5)	D = 28.7 days (2) Z = 0.3454 (2) $N = 5 \times 10^{-5}$ (2)	v = 0.17 (M=1.77) Fishery recruitment = 14.9 (5)	k = 0.34 (5) L_∞ = 42 (5) t_0 = −1 (7) c = 0.0238 (5) y = 2.93 (5)
Epinephelus guttatus the red hind	r = 1 spawn per year × 873.454 per g body weight − 194086) Mature at size < 25 cm (6)	D = 28.7 days (2) Z = 0.3454 (2) $N = 5 \times 10^{-5}$ (2)	v = 0.5066 (M=0.68) Fishery recruitment = 31 cm (6)	k = 0.24 (6) L_∞ = 52 (6) t_0 = −0.5 (7) c = 0.0107 (6) y = 2.96 (6)

100% growth for new settlers to 0% for fish in the largest size class.

Adults in the fishing area that grew larger than the minimum catch size experienced fishing mortality. We represented fishing mortality using the parameter u, equal to the proportion of fishery-recruited individuals caught per year, and related to the more common F by the equation:

$$u = 1 - e^{-F}.$$

We independently varied the two key parameters in our models: fishing mortality (u) and reserve proportion (s).

We made settlement a density-dependent process by incorporating a negative exponential function into survivorship for new settlers during their first year (see Appendix for equation). There is evidence to suggest that shelter is limiting for coral reef fish, especially new settlers (Hixon, 1991; Hixon and Beets, 1993; Hixon and Carr, 1997)—a process that would fit well with our density dependence assumptions. We also performed runs in which larval rather than new settler survivorship was density-dependent and

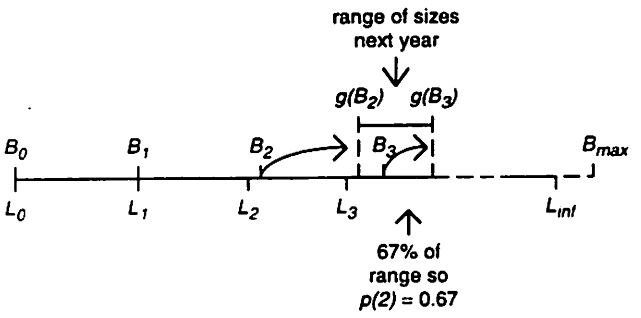

Figure 2

Determination of growth probabilities. The probability $p(x)$ that a fish of size class x grows to size class x+1 is the proportion of the range of sizes that size class x individuals will span one year later. Using von Bertalanffy growth parameters (see Appendix), we determined one year in the future the size of the smallest and largest fish in size class x. All other fish in x would fall between these two values. We then determined what proportion of these sizes fell into size class x+1, 0.67 in the case illustrated. In this hypothetical example, we would assume that $p(x)$ = 0.67, or that 67% of the fish from size class x grew to class x+1 by the next year. L_t represents the sizes attained at age t and B_x represents the lower bounds of each size class. We begin counting age at the moment of settlement (so $L_0 = B_0$).

obtained qualitatively identical results. Little information exists on density-dependent relationships for tropical fish; therefore we were forced to use a standard theoretical logistic equation (see Appendix). For simplicity in analysis and in recognition of this knowledge gap, we constructed the models with a fixed carrying capacity of 1000 one-year-old individuals per unit area. We modeled population processes on the basis of density measures in reserve and fishing areas and used relative proportions of each to calculate catches and population fecundity. Thus, yields are expressed as kg per year from the whole management area.

In order to ask the general questions we intended, we used two simple movement assumptions that emphasized the benefits accruing from larval transport rather than from adult spillover. Larvae dispersed widely across reserve boundaries, resulting in an even density of new fish settlement in reserve and nonreserve areas. This assumption does not negate the possibility that larvae drift to the open ocean and become lost—these can be accounted for in larval mortality. Rather, the assumption implies that settlement in the reserve and the management area are equally affected by the stock averaged over both areas. We also assumed that adults did not move across reserve boundaries; that is to say fish spent their entire lifetime in the area in which they settled. These assumptions specifically addressed the case where enhanced fecundity within the reserve was exported to fishing areas through larval transport. Thus they complemented previous models that focused on enhancements from adult spillover (Polacheck, 1990; DeMartini, 1993). These models also examined potential increases in reproductive output by means of increased spawning stock biomass per recruit. However, they were unable to examine the equilibrium consequences with their particular model construction.

Our movement assumptions apply to the majority of fishery species on at least some spatial scales. Most aquatic species disperse more widely as larvae than as adults (Boehlert, 1996). Consequently, larvae are more likely to cross boundaries than are adults. As long as individual reserve units stretch beyond the dispersal distance of adults but remain well within the dispersal distance of larvae for a given species, the model assumptions will approximate reality. For large reserve proportions, our assumptions could still be met if the reserve area were partitioned into several smaller units. Some areas of concern here include ontogenetic and reproductive migrations. To fit the assumptions of this model, reserves must be designed with these movements in mind so that fish are likely to remain in the reserve during the phase of their life in which they are vulnerable to fishing.

We ran the models over reserve proportions varying from 0 to 99% of the management area (s=0 to 0.99), and fishing mortalities varying from 1% to 100% mortality of fishery-recruited individuals per year (u=0.01 to 1.00). For each combination of fishing mortality and reserve proportion, the model ran until the fish catch—calculated for the whole management area rather than per km^2 of available fishing area—had stabilized at the long-term sustainable yield. The model stored the yield, fishing mortality, and reserve proportion. It sometimes took hundreds of years to reach stability, and those interested in our model's predictions about the short-term dynamics of reserve creation should refer to Sladek Nowlis and Roberts (1997).

We used these results to determine the optimal reserve proportions and fishing mortalities for individual fishery species. For each fishing mortality, we found the reserve proportion that maximized sustainable yields and stored it and the yield. We compared these yields when an optimally-sized reserve was used with the yields without a reserve (s=0) to establish fisheries benefits. We plotted this information using fishing mortality as an independent variate.

We also examined the effects of marine fishery reserves on year-to-year catch variability. Bohnsack (1996) suggested that marine fishery reserves could dampen natural fluctuations in catches, thus making fisheries more stable and easier to manage. We tested this hypothesis by adding a stochastic component to larval survivorship in our model. These new models drew larval survivorship randomly from a normal distribution around the mean larval survivorship whose standard deviation we could define. We examined all species over a range of fishing mortalities and present the results from u = 0.2, 0.4, 0.6, and 0.8 to illustrate the pattern. We also examined some of these conditions at three levels of environmental variation, with standard deviations of 5, 10, and 20% of the mean larval survivorship, to gain insight into whether reserve benefits are influenced by the degree of environmental variability.

For each possible combination of fishing mortality and reserve proportion, we performed 10 replicate runs of our stochastic models. In each run, we ran the models for 500 years to allow the fisheries to stabilize to the maximum extent possible and thus minimize the influence of our arbitrarily chosen initial state. The mean and standard deviation of the catches were measured over the next 100 years. We examined the ratio of the standard deviation to the average catch over this period because this measure gave us an estimate of the likelihood of percentage fluctuations in catches rather than absolute changes.

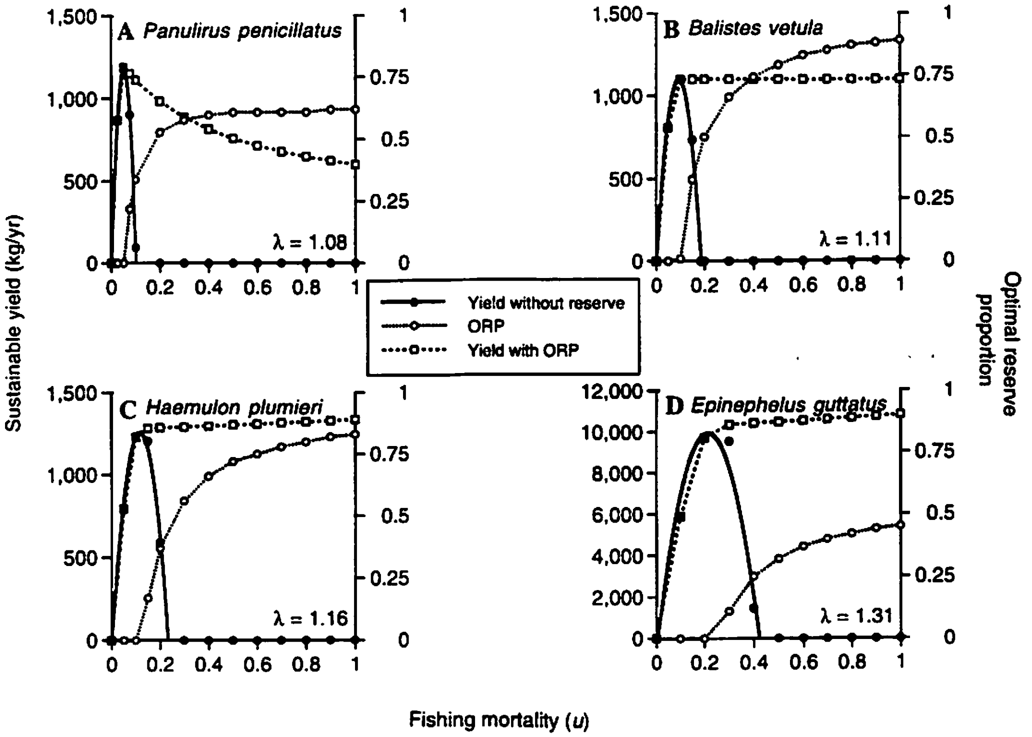

Figure 3

Optimal reserve proportions and corresponding yields. (A) *Panulirus penicillatus*, Red Sea spiny lobster. (B) *Balistes vetula*, queen triggerfish. (C) *Haemulon plumieri*, white grunt. (D) *Epinephelus guttatus*, red hind. In all graphs, the solid circles and line represent the sustainable yield (kg of catch per year from the whole management area) that occurs in the absence of a reserve, the open circles and dotted line represent the optimal reserve proportion (that which produced maximum sustainable yields for each fishing mortality), and the dashed line and squares represent the sustainable yield when the optimal reserve proportion was used. Intrinsic population growth rates (λ) determine the robustness of the populations to fishing, high growth rates sustaining heavy fishing and low rates requiring reserves at low fishing mortalities.

This measure is better than standard deviation alone which would treat a 10-kg fluctuation equally, regardless of whether it occurred in a 100 kg or 1,000,000 kg per year fishery. We graphed these results with reserve proportion as the independent variate and examined the graphs for trends.

We performed these analyses on four coral reef fishery species for which we obtained relatively complete parameter sets. These included *Balistes vetula*, queen triggerfish; *Epinephelus guttatus*, red hind; *Haemulon plumieri*, white grunt; and *Panulirus penicillatus*, Red Sea spiny lobster (see Table 1 for parameter estimates).

Results

When we ran the models without a reserve ($s=0$), they produced standard yield-effort curves (Fig. 3). These curves are characterized by steep initial gains in long-term sustainable yields with increases in fishing mortality (and thus effort), followed by equally steep declines (Clark, 1990). The curves peaked at the maximum sustainable yield, one of several goals a manager might try to achieve with a fishery (Clark, 1990), and we will refer to the corresponding fishing mortality as the MSY mortality for the rest of this paper. Above the MSY mortality, the fishery can be defined as overfished because it is less productive than it would be with less fishing activity.

When a reserve was present, the yield-mortality curves were still parabolas passing through the origin but spread farther to the right, and the larger the reserve, the more pronounced were these shifts. Consequently, larger reserves required higher fishing mortalities to maximize long-term sustainable yields (remember that this mortality only affected fish in fishing areas), whereas the sustainable yields decreased more slowly as fishing mortality increased past the MSY mortality.

Our analyses of optimal reserve proportions produced several key results. First, reserves produced fisheries enhancements, meaning that the overall catches with a reserve exceeded those without one, whenever the fisheries were overfished (Fig. 3), here defined as fished above the MSY mortality level. When fisheries were overfished, they produced higher yields with a reserve even though the reserve decreased the amount of fishing area. The optimal reserve proportion increased with increasing fishing mortality, and heavily exploited fisheries required particularly large reserves to remain productive. The fishery benefit attributable to reserves, calculated by subtracting the yield without a reserve from that with an optimally sized reserve, increased with increasing fishing mortality up to a near-maximum yield in most cases (Fig. 4). Consequently, a wide span of reserve sizes (up to 80% of the management area for some species) produced similarly high yields for most species as long as fishing mortalities were chosen accordingly.

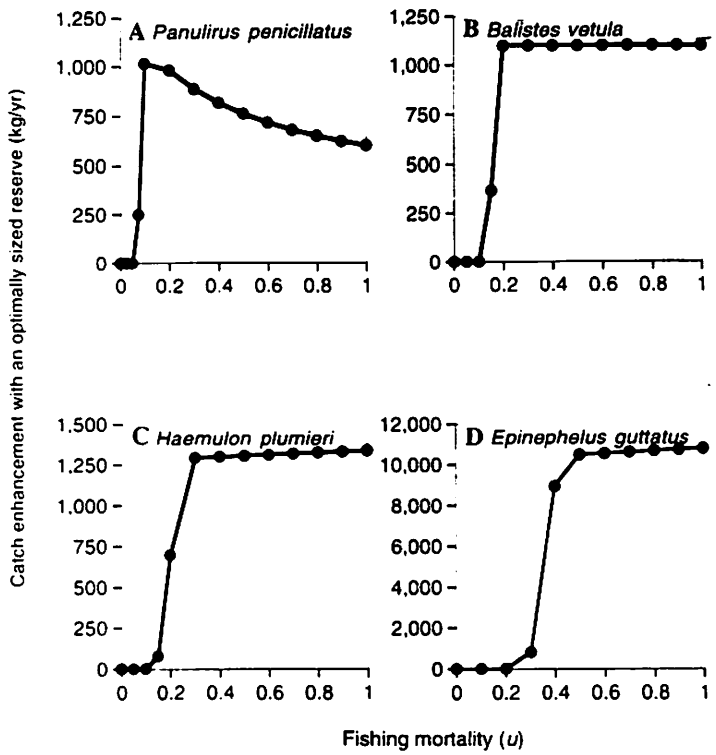

Figure 4

Catch enhancements with the use of an optimally proportioned reserve (OPR). (**A**) *Panulirus penicillatus*, Red Sea spiny lobster. (**B**) *Balistes vetula*, queen triggerfish. (**C**) *Haemulon plumieri*, white grunt. (**D**) *Epinephelus guttatus*, red hind. Values represent the increase in yield, in kg of catch per year from the whole management area, one could expect if an optimally sized reserve system were established in a management area that lacked reserves initially.

Using this information (Fig. 3), we predicted optimal reserve proportions under real-life fishing mortalities. For queen triggerfish, the fishing mortality estimate of $u = 0.45$ from Puerto Rico and the Virgin Islands (Aiken, 1983) corresponded to an optimal reserve proportion of approximately $s = 0.8$. For white grunt, a reported heavy fishing mortality of $u = 0.99$ from Jamaica (Darcy, 1983) corresponded to an optimal reserve proportion of just over $s = 0.75$. Thus, for these species in these locations, our models predicted that 75–80% of the fishing grounds should be made off-limits to fishing in order to maximize long-term sustainable yields. These numbers may seem unrealistically high, especially since most models predict maximum yields when approximately 50% of the population density at carrying capacity is protected from fishing (see Clark, 1990, for an overview). In the case of our models, populations within the reserve did not reach carrying capacity when fishing was heavy outside, and the conditions of peak production corresponded to those that protected approximately 50% of the population density at carrying capacity.

The qualitative conclusions outlined above were consistent across all the species we examined. However, the model's quantitative predictions of the long-term fishery yields and optimal reserve proportion varied from species to species for any given fishing mortality (Fig. 3). The key differences between species were the speeds at which the yield and optimal reserve proportion changed with increasing fishing mortality (Fig. 3). These differences reflected differences in intrinsic population growth rates (λ)—the maximum growth rate of a population with no density-dependent constraints or fishing mortality. This summary parameter integrates most of the life history data that we used. It does not include the growth rate of individuals in the population and consequently does not adequately predict yields. However, it is a useful summary of the ability of a population to sustain harvesting. For example, life history parameters from the literature suggested that the Red Sea spiny lobster had a relatively low $\lambda = 1.08$, just above the $\lambda = 1$ necessary for a population to sustain itself with no fishing pressure. This species had a low MSY fishing mortality because its slow population growth could only sustain modest harvesting effort (Fig. 3). In contrast, life history parameters from the literature suggested that red hind had a relatively high $\lambda = 1.31$. Consequently, its maximum sustainable yield occurred at the highest fishing mortality of any species we tested (Fig. 3). The two other species we examined had intermediate intrinsic rates of population growth rates and responses to reserves.

The sensitivity of our models' quantitative predictions was also clear within a species when we varied larval survivorship. For all species, optimal reserve proportion and yield without a reserve varied greatly (senior author's unpubl. data) because we varied larval survivorship from 10^{-5} to 10^{-4}. This sensitivity to poorly understood parameter values renders any quantitative estimates of optimal reserve proportion unreliable, whether the inaccuracy is in larval survivorship, the relationship, or parameters for density dependence, or any other life history parameter.

Finally, we examined how reserves might influence unpredictable catches resulting from environmental variation. Our stochastic models predicted that catches will be more stable with larger reserve proportions. In these models, we saw general decreases in catch variability with increasing reserve proportion (Fig. 5). The results presented here showed drops in variation that were more pronounced at higher fishing mortalities for all four species. We also tested these results at three levels of environmental variation. Our results showed that the drop in catch variability was most extreme when the environment was most variable, suggesting that the stability offered by reserves will be most valuable in highly variable fisheries.

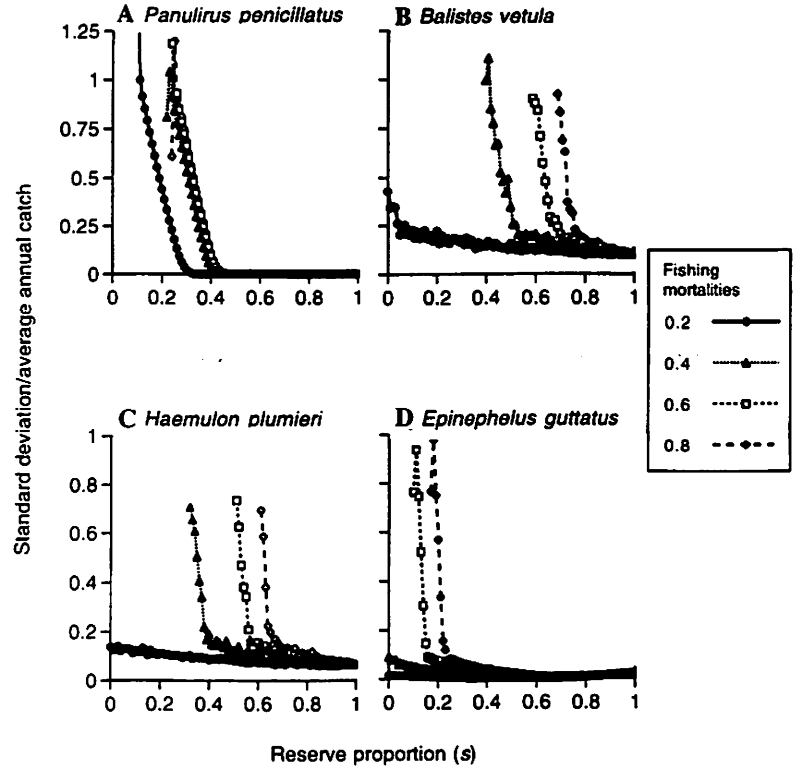

Figure 5

Catch variability and reserve size. (A) *Panulirus penicillatus*, Red Sea spiny lobster. (B) *Balistes vetula*, the queen triggerfish. (C) *Haemulon plumieri*, white grunt. (D) *Epinephelus guttatus*, red hind. Each graph shows decreasing catch variability with increasing reserve proportion at four levels of fishing mortality.

Discussion

Effects of life history and fishing mortality on reserve benefits

Our models predicted that marine fishery reserves will provide catch enhancements to any overfished fishery that meets our basic assumptions regarding the movement of adults and larvae. The results from previous modeling efforts by Man and colleagues (1995) and Holland and co-workers (Holland and Brazee, 1996; Holland et al.[1]) support these findings if one compares their results in specific cases to the patterns we found for a variety of species. Two key variables help determine whether a population is overfished: intrinsic population growth rate (λ) and fishing mortality. Managers can control fishing mortality to varying extents. Apparently, this control is inadequate in many industrial fisheries (FAO, 1995) and is probably even less effective in subsistence fisheries (Roberts and Polunin, 1993). Managers have no control over population growth potential but can take into account that species with low population growth have a greater tendency to be overfished and consequently show greater promise for fisheries enhancements from reserves.

Even in a well-managed fishery, it may be helpful to close large areas. This strategy could allow the relaxation of some fishing restrictions in remaining waters. Consequently, recreational and commercial fishermen may feel greater equity with fewer restrictions on the number of participants or their catches. Moreover, reserves have the potential to reduce variability in catches from year to year and to enhance conservation of species and ecosystems. Fishing is not the only threat to marine ecosystems, though, and fisheries regulations are not sufficient to protect these systems (Allison et al., 1998).

To our knowledge, no field study has yet examined the effects of population growth potential or fishing mortality on reserve benefits. In part, such studies are made difficult by the uncontrolled nature in which reserves are established.

Relation of fishery benefits to reserve size

Real-world fisheries span a range from lightly fished to heavily overfished, and the optimal reserve size will depend on the fishing mortality as well as the population growth potential of the target species. Because many fisheries involve multiple species with widely divergent population growth potentials, choosing a single best reserve size may be difficult. Moreover, key aspects of the life history of marine fish, the larval phase in particular, remain a mystery. Because of these gaps in knowledge, it would be difficult to make an accurate prediction of the optimal reserve size even in a well-studied single-species fishery.

Although our research sheds doubt on the use of a universal reserve proportion, it does lend support for the use of large reserve systems under certain circumstances. In the two real-world cases where the necessary information existed, our models predicted that reserves should encompass 75–80% of the management area. These proportions are enormous and may be unrealistic for several reasons. First, the short-term economic losses from closing 80% of a management area would be large, although our models predict that the recovery time for such heavily overfished fisheries would be rapid (Sladek Nowlis and Roberts, 1997). Second, the political challenges of establishing such large reserves would be a formidable barrier. Finally, we do not stand firmly behind these predictions because their accuracy is dependent on parameter values that are poorly understood. Nevertheless, consistent results across several species suggest that reserves encompassing 40% or more of a heavily fished management area could produce substantial fisheries benefits.

Though rare, at least one large reserve system does exist. The Mombasa Marine National Park closed over 60% of local fishing grounds (McClanahan and Kaunda-Arara, 1996). This example fits nicely with our model's assumptions because levels of fishing effort remained similar in the fishing grounds before and after the closure. After two years, total yields had not surpassed those prior to reserve establishment (McClanahan and Kaunda-Arara, 1996). However, catch per unit of effort had increased dramatically and total yields showed potential for future increases. In this case and others involving extensive use of marine reserves, our research encourages an adaptive approach that reflects the lack of knowledge about fish life histories and the high degree of uncertainty in these complex biological systems.

Relation of reserve size to catch variability

Our model supported Bohnsack's (1996) hypothesis that catch variability will decrease with increasing reserve size. Our models predicted decreases in catch variability across a variety of levels of environmental variability and fishing mortalities. Our results also complement other studies that showed that reserves could reduce catch variability,[2] decrease the likelihood of bad years (Lauck et al., 1998), and increase the persistence of fisheries vulnerable to overfishing (senior author's unpubl. data).

To our knowledge, no field study has yet examined the effects of reserves on catch variability. Although they may be confounded by variability in fishing effort, the necessary data should be practical to collect before and after reserve creation.

Assumptions revisited

As with all models, one must be careful in interpreting the results of this one. It is based on parameter values that in some incidences—larval survivorship in particular—are poorly understood. However, the model's predictions are qualitatively robust to parameter errors, meaning that its general predictions hold true across a wide range of values and a wide variety of species. Our assumptions regarding the movement of adults and larvae were far more critical in influencing the conclusions we have drawn here.

Our assumptions regarding adult movement have wide applicability. Many fisheries target sessile organisms such as harvested kelp (Bustamente and Castilla, 1990), slow-moving organisms including many invertebrates (Davis and Dodrill, 1980; Davis and Dodrill, 1989), and organisms with high site-specificity such as many reef fish (Polunin and Roberts, 1996). All of these systems are likely to approximate our assumptions of no adult movement. This model is not universally applicable, as highly mobile and migratory species, including many pelagic fisher-

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[2] Mangel, M 1998. Environmental Studies Board, University of California, Santa Cruz, CA 95064. Unpubl. data.

ies (Safina, 1993), will only fit our adult movement assumptions if large reserves are established. Recent studies, though, have shown high site fidelity by fish species previously thought to range widely (Holland et al., 1993; Holland et al., 1996), demonstrating the need for more field data on adult movement patterns. Moreover, recent modeling efforts by Holland and colleagues[1] and others[3] suggest that reserves can benefit highly mobile species through enhanced population fecundity gained from temporary protection.

If adults do cross reserve boundaries, our predictions regarding fisheries benefits from reserves will be influenced in opposing ways. Under heavy fishing pressure and intermediate movement tendencies, minor yield enhancements may be possible from this adult spillover (Polacheck, 1990; DeMartini, 1993). However, this same movement would dilute the ability of reserves to enhance larval transport to fishing areas. As Polacheck showed (1990), spawning stock biomass, or the potential for fisheries enhancement through larval transport, is highest at lowest levels of adult movement. Because the potential benefits from larval transport presented here far outweigh those predicted from adult spillover (Polacheck, 1990; DeMartini, 1993), it is likely that adult movement across boundaries will decrease the predicted yields from reserves. Consequently, reserves will have the highest potential for enhancing surrounding fisheries if they are designed as a collection of units large enough to contain populations of adults with relatively little movement across boundaries.

Our assumptions regarding larval transport have less supporting evidence. Most aquatic species disperse more widely as larvae than as adults (Boehlert, 1996), and the potential for long-distance dispersal across reserve boundaries is great for species with long-lived larvae (Roberts, 1997), including most food fish. Consequently, larvae are likely to move from reserves to fishing areas as long as oceanographic conditions and larval behavior permit. Without larval transport, the potential for fisheries benefits from reserves is more limited, although Holland and colleagues[1] did show that a reserve system in which larvae stayed in place but adults moved widely across boundaries could produce some benefits. Reserves, especially in heavily overfished or large management areas, may need to be partitioned into several subunits that maintain adult populations within them but allow larvae to disperse to remaining fishing areas.

We also assumed a stock-recruitment relationship, implying that a significant portion of the population fecundity from reserves stays in or returns to the management area. The degree to which marine populations are locally sustained remains an active area of debate in marine ecology. Larvae of most tropical food fish are often found in greatest quantities offshore (Boehlert, 1996), suggesting the possibility of long-distance dispersal. However, studies that show this result may be biased because sampling within the complex structure of the reef itself is difficult (Boehlert, 1996). Therefore, reefs may harbor greater concentrations of larvae than are measured above the reef. This complexity (Wolanski and Sarsenski, 1997), along with potential for larval behavior to influence their distribution (e.g. Breitburg et al., 1995), suggests that larvae may be retained at higher concentrations than predicted by simple oceanographic models (e.g. Roberts, 1997). If recruitment dynamics are influenced on a much larger spatial scale than encompassed by the management area, such that the stock in the management area has a minimal impact on recruitment back to it, reserve benefits to the management area are likely to be much more limited. Cohort models, including those by Polacheck (1990) and DeMartini (1993), can be interpreted as situations in which larval supply is constant and not influenced by local stock. As has been discussed, these models show limited potential for fisheries benefits from reserves. It is necessary to think of reserve systems at a scale that fits stock-recruitment relationships. Yet our knowledge of these relationships remains poor. Even if larvae have the potential to disperse over large distances, stock-recruitment relationships could still exist on a local level if a significant portion of larval production is retained. The safest approach to this uncertainty is to design reserve systems at large scales. However, there is still the potential for reserves to produce fisheries benefits on small scales if larvae have the capacity to be retained. Further research on stock-recruitment relationships in marine populations will be invaluable for resolving this pressing issue along with many others in fisheries management.

Field needs and testable predictions

Our results identify areas in need of additional field work and make testable predictions. The needs in regard to field work differ for our quantitative and qualitative predictions. The quantitative predictions were highly sensitive to all parameters that affected intrinsic population growth potential. The most important and least understood of these parameters is larval survivorship. We need significantly better information about the duration of the egg and larval stages of coral reef fishes and their daily mortality

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[3] Guenette, S. 1998. Fisheries Centre, University of British Columbia, 2204 Main Mall, Vancouver, BC V6T 1Z4, Canada. Unpublished data.

risk. Until we understand these life history stages better, it will be impossible to make quantitatively accurate predictions of the optimal design of any fishery management strategy.

We also need better insight into how fecundity changes with size. Fecundity-size relationships should be fairly easy to measure and can be incorporated into any standard fishery study where adequate numbers of adults are sampled. We would further benefit from estimates of size-specific natural mortality. Few natural mortality estimates for coral reef fish species exist in the literature, and most that do are based on highly indirect methods of association. Marine fishery reserves actually offer the potential to generate more accurate predictions of natural mortality because fishing mortality does not confound the attempt in unfished areas. Moreover, despite numerous studies, we still have a poor understanding of population regulation and density dependence in coral reef fishes. This understanding is also necessary before we can generate accurate quantitative predictions of reserve benefits.

In contrast to the long list necessary to generate quantitative predictions, our qualitative predictions require additional knowledge in only one key area: fish movement. Because the qualitative predictions were robust across life history patterns, the key to knowing whether a fish species fits our assumptions is the movement of this species as eggs, larvae, and as adults. To some extent, we can skirt this issue because in our model, reserve size was based on proportion of coastline rather than actual size. Consequently, if we choose the management area to match the scale of fish movement, our model can fit most species. For example, a 20% reserve divided into areas of tens of hectares might ensure that adults of the species we examined here will stay in the area in which they settled while their larvae disperse widely among the reserve and nonreserve areas. In contrast, the management area might have to encompass whole ocean basins for the movement assumptions to fit bluefin tuna (Safina, 1993). Thus, we need to understand the movement dynamics of larvae and adults of a species to know the scales at which it will fit the assumptions of our model.

From the species that we ran and the resulting qualitative predictions of our model, we can generate a list of testable predictions. We predict that

1) Reserves will be beneficial for any over-fished population. Populations with low intrinsic growth rates and high fishing mortality stand to benefit the most, as is the case for the majority of reef fisheries in many regions of the world, such as the Caribbean. The location and size of the reserve will also affect reserve benefits. For a fair test of this prediction, reserve should be representative of typical fish habitat and large enough to contain a viable population of adults.
2) Although no universal best reserve proportion exists, we predict reserves will enhance fishery productivity even when they encompass areas much larger than those of current reserve systems.
3) Reserves will reduce variation in catches resulting from unpredictability in fishing mortality as well as recruitment strength and larval survivorship. Such an effect will simplify fishery management and increase the ability of fishermen to predict future income.

Acknowledgments

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Literature cited

Aiken, K. A.
1983. The biology, ecology and bionomics of the triggerfishes, Balistidae. *In* J. L. Munro (ed.), Caribbean coral reef fishery resources, p. 191–276. International Council for Living Aquatic Resources Management, Manila.

Alcala, A. C., and G. R. Russ.
1990. A direct test of the effects of protective management on abundance and yield of tropical marine resources. J. Cons. Int. Explor. Mer 47:40–47.

Allison, G. W., J. Lubchenco, and M. H. Carr.
1998. Marine reserves are necessary but not sufficient for marine conservation. Ecol. Appl. 8:S79–92.

Beverton, R. J. H., and S. J. Holt.
1957. On the dynamics of exploited fish populations. Chapman and Hall, New York, NY, 533 p.

Boehlert, G. W.
1996. Larval dispersal and survival in tropical reef fishes. *In* N. V. C. Polunin and C. M. Roberts (eds.), Reef fisheries, p. 61–84. Chapman and Hall, London.

Bohnsack, J. A.
1996. Maintenance and recovery of fishery productivity. *In* N. V. C. Polunin and C. M. Roberts (eds.), Reef fisheries, p. 283–313. Chapman and Hall, London.

Breitburg, D. L., M. A. Palmer, and T. Loher.
1995. Larval distributions and the spatial patterns of settlement of an oyster reef fish: responses to flow and structure. Mar. Ecol. Prog. Ser. 125:45–60.

Bustamente, R. H., and J. C. Castilla.
1990. Impact of human exploitation of populations of the intertidal bull-kelp *Durvillaea antarctica* (Phaeophyta, Durvilleales) in central Chile. Biol. Cons. 52:205–220.

Castilla, J. C., and L. R. Durán.
1985. Human exclusion from the rocky intertidal zone of central Chile: the effects on *Concholepas concholepas* (Gastropoda). Oikos 45:391–399.

Clark, C. W.
1990. Mathematical bioeconomics: the optimal management of renewable resources. John Wiley & Sons, New York, NY, 399 p.

Darcy, G. H.
1983. Synopsis of biological data on the grunts *Haemulon aurolineatum* and *H. plumieri* (Pisces: Haemulidae). U.S. Dep. Commer., NOAA Technical Report NMFS Circular 448, 41 p.

Davis, G. E., and J. W. Dodrill.
1980. Marine parks and sanctuaries for spiny lobster fisheries management. Fish. Bull. 78:979–984.
1989. Recreational fishery and population dynamics of spiny lobster, *Panulirus argus*, in Florida Bay, Everglades National Park, 1977–1980. Bull. Mar. Sci. 44:78–88.

DeMartini, E. D.
1993. Modeling the potential of fishery reserves for managing Pacific coral reef fishes. Fish. Bull. 91:414–427.

Dugan, J. E., and G. E. Davis.
1993. Applications of marine refugia to coastal fisheries management. Can. J. Fish. Aquat. Sci. 50:2029–2042.

Durán, L. R., and J. C. Castilla.
1989. Variation and persistence of the middle rocky intertidal community of central Chile with and without human harvesting. Mar. Biol. 103:555–562.

FAO (Food and Agriculture Organization).
1995. The state of the world fisheries and aquaculture. Food and Agriculture Organization Fisheries Department FAO Fisheries Circular 884, 105 p.

Hay, M. E.
1984. Patterns of fish and urchin grazing on Caribbean coral reefs: are previous results typical? Ecology 65:446–454.

Hay, M. E., and P. R. Taylor.
1985. Competition between herbivorous fishes and urchins on Caribbean reefs. Oecologia 65:591–598.

Hixon, M. A.
1991. Predation as a process structuring coral reef fish communities. *In* P. F. Sale (ed.), The ecology of fishes on coral reefs, p. 475–508. Academic Press, San Diego, CA.

Hixon, M. A., and J. P. Beets.
1993. Predation, prey refuges, and the structure of coral-reef fish assemblages. Ecol. Mono. 63:77–101.

Hixon, M. A., and M. H. Carr.
1997. Synergistic predation, density dependence, and population regulation in marine fish. Science (Wash. D. C.) 277:946–949.

Holland, D. S., and R. J. Brazee.
1996. Marine reserves for fisheries management. Mar. Resource Econ. 11:157–171.

Holland, K. N., C. G. Lowe, and B. M. Wetherbee
1996. Movements and dispersal patterns of blue trevally (*Caranx melampygus*) in a fisheries conservation zone. Fish. Res. 25:279–292.

Holland, K. N., J. D. Peterson, C. G. Lowe, and B. M. Wetherbee.
1993. Movements, distribution and growth rates of the white goatfish *Mulloides flavolineatus* in a fisheries conservation zone. Bull. Mar. Sci. 52:982–992.

Houde, E. D.
1989. Comparative growth, mortality, and energetics of marine fish larvae: temperature and implied latitudinal effects. Fish. Bull. 87:471–495.

Lauck, T., C. W. Clark, M. Mangel, and G. R. Munro.
1998. Implementing the precautionary principle in fisheries management through marine reserves. Ecol. App. 8:S72–78.

Man, A., R. Law, and N. V. C. Polunin.
1995. Role of marine reserves in recruitment to reef fisheries: a metapopulation model. Biol. Cons. 71:197–204.

McAllister, D. E.
1988. Environmental, economic and social costs of coral reef destruction in the Philippines. Galaxea 7:161–178.

McClanahan, T. R., A. T. Kamukuru, N. A. Muthiga, M. Gilagabher Yebio, and D. Obura.
1996. Effect of sea urchin reductions on algae, coral, and fish populations. Cons. Biol. 10:136–154.

McClanahan, T. R., and B. Kaunda-Arara.
1996. Fishery recovery in a coral-reef marine park and its effect on the adjacent fishery. Cons. Biol. 10:1187–1199.

McClanahan, T. R., and S. H. Shafir.
1990. Causes and consequences of sea urchin abundance and diversity in Kenyan coral reef lagoons. Oecologia 83:362–370.

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

Plan Development Team.
1990. The potential of marine fishery reserves for reef fish management in the U.S. Southern Atlantic. U.S. Dep. Commer., Technical Memorandum NMFS-SEFC-261, 48 p.

Plaut, I.
1993. Sexual maturity, reproductive season and fecundity of the spiny lobster *Panulirus penicillatus* from the Gulf of Eilat (Aqaba), Red Sea. Aust. J. Mar. Freshwater Res. 44:527–535.

Plaut, I., and L. Fishelson.
1991. Population structure and growth in captivity of the spiny lobster *Panulirus penicillatus* from Dahab, Gulf of Aqaba, Red Sea. Mar. Biol. 111:467–472.

Polacheck, T.
1990. Year around closed areas as a management tool. Nat. Resource Model. 4:327–354.

Polunin, N. V. C., and C. M. Roberts (eds.).
1996. Reef fisheries. Chapman and Hall, New York, NY, 495 p.

Quinn, J. F., S. R. Wing, and L. W. Botsford.
1993. Harvest refugia in marine invertebrate fisheries: models and applications to the red sea urchin *Strongylocentrotus franciscanus*. Am. Zool. 33:537–550.

Ricker, W. E.
1975. Computation and interpretation of biological statistics of fish populations. Fisheries and Marine Service, Ottowa, 400 p.

Roberts, C. M.
1995. Effects of fishing on the ecosystem structure of coral reefs. Cons. Biol. 9:988–995.
1997. Connectivity and management of Caribbean coral reefs. Science (Wash. D.C.) 278:1454–1457.

Roberts, C. M., W. J. Ballantine, C. D. Buxton, P. Dayton, L. B. Crowder, W. Milon, M. K. Orbach, D. Pauly, and J. Trexler.
1995. Review of the use of marine fishery reserves in the U.S. southeastern Atlantic. U.S. Dep. Commer., NOAA Technical Memorandum NMFS-SEFC-376, 31 p.

Roberts, C. M., and N. V. C. Polunin.

1991. Are marine reserves effective in management of reef fisheries? Rev. Fish Biol. Fish. 1:65–91.

1993. Marine reserves: simple solutions to managing complex fisheries? Ambio 22:363–368.

Rowley, R. J.

1994. Marine reserves in fisheries management. Aquat. Cons. Mar. Freshwater Ecosyst. 4:233–254.

Russ, G.

1985. Effects of protective management on coral reef fishes in the central Philippines. Proceedings of the 5th International Coral Reef Congress 4:219–224,

Russ, G. R., and A. C. Alcala.

1996. Do marine reserves export adult fish biomass? Evidence from Apo Island, central Philippines. Mar. Ecol. Prog. Ser. 132:1–9.

Safina, C.

1993. Bluefin tuna in the West Atlantic: negligent management and the making of an endangered species. Cons. Biol. 7:229–234.

Sladek Nowlis, J., C. and M. Roberts.

1997. You can have your fish and eat it, too: theoretical approaches to marine reserve design. Proceedings of the 8th International Coral Reef Symposium 2:1907-1910, Smithsonian Tropical Research Institute, Panama City, Panama.

Thompson, R., and J. L. Munro.

1983. The biology, ecology and bionomics of the hinds and groupers, Serranidae. *In* J. L. Munro (ed.), Caribbean coral reef fishery resources, p. 59–81. International Council for Living Aquatic Resources Management, Manila.

Wolanski, E., and J. Sarsenski.

1997. Larvae dispersion in coral reefs and mangroves. Am. Sci. 85:236–243.

Appendix

Fecundities

Fecundities were size-specific, but the general form of the equation relating size to fecundity varied from species to species. The specific relationships are listed in Table 1 as *r*. These *r* values were set to zero for all classes smaller than the size at maturity.

Larval survival

We used equations developed by Houde (1989) that relate ambient temperature during development to duration of larval stage, daily mortality risk, and probability of surviving through the entire larval stage.

$$D = 952.5\, T^{-1.0752} \qquad (1)$$

$$Z = 0.0003149\, T \qquad (2)$$

$$N = e^{-ZD}, \qquad (3)$$

where T = ambient temperature during development, in degrees Celsius;
D = duration of larval stage, in days;
Z = probability of mortality, per day; and
N = probability of surviving through the entire larval stage.

Adult survival

We assumed that newly settled fish experienced density dependence. Thus, instead of surviving at a rate v_i like individuals in other size classes, their survival was weighted by a density-dependent function of the form $e^{-\rho/K}$ where ρ = the population density and K = a measure of carrying capacity arbitrarily set at 1000 due to a lack of information on carrying capacities for the fish we studied. Note that size-class-1 individuals included new recruits that survived and grew as well as old size-class-1 individuals that survived but did not grow to size class 2. Thus, at time t, the densities of size-class-1 individuals in the reserve ($S_{1,t}$) and the fishing area ($F_{1,t}$) are

$$S_{1,t} = v_0 p(0) S_{0,t-1} e^{-S_{0,t-1}/K} + v_1(1 - p(1)) S_{1,t-1} \qquad (4)$$

$$F_{1,t} = v_0 p(0) F_{0,t-1} e^{-F_{0,t-1}/K} + v_1(1 - p(1)) F_{1,t-1}, \qquad (5)$$

where v_x = the density-independent survival rate for individuals in size class x.

Note that the density in the fishing area is decreased later in the program to account for fishing mortality but only for size classes larger than the size at fishery recruitment. Also note that other size classes experience the density-independent survival rate v_x.

Growth

We began with standard von Bertalanffy equations (Ricker, 1975), relating length to age and weight to length (Fig. 2) and categorized them as described by Figure 2. Through algebraic manipulation, we established a formula for $g(B_x)$, the size of an individual projected one year in the future:

$$g(B_x) = e^{-k} B_x + (1 - e^{-k}) L_{inf}. \qquad (6)$$

We used this formula to establish the following calculation for $p(x)$, the probability that an individual in size class x grows to size class x+1 by next year.

$$p(x) = \frac{g(B_{x+1}) - B_{x+1}}{g(B_{x+1}) - g(B_x)} = \frac{L_\infty - B_{x+1}}{L_\infty - st - L_0} = \frac{L_\infty - B_{x+1}}{L_\infty - L_1}. \qquad (7)$$

TAB I
EFH/MPA
April 2003

Suggested Working Definitions for the Joint Protocol Committee

Marine: all areas seaward of the mean higher high water line, out to the 200 mile limit of the territorial sea.

Marine Protected Area (MPA): Graphically defined areas designated for special protection to enhance the management of marine resources with year round protection (NRC 2001). This definition includes areas where extraction of certain specific fishery resources is prohibited, and/or areas where specific gear types are prohibited to protect marine habitats. NMFS recognizes the definition of a Marine Protected Area as defined by Executive Order 13158: "Any area of the marine environment reserved by Federal, State, territorial, tribal, or local laws or regulations to provide lasting protection for part or all of the natural and cultural resources therein."

Marine Reserve(MRV): A type of MPA where removal or disturbance of resources is prohibited. Marine reserves are also known as "no-take zones". Marine reserves are a restrictive class of MPAs.

Marine managed Area (MMA) – a geographically defined area designated with special protections, including seasonal protections, of marine resources. This is similar to a marine protected area (MPA) but without the requirement of year-round protection; hence, an MPA is a restrictive class of MMA.

Marine Research Reserve (MRR) A MRR is an area where all marine resources are protected from any disturbance or removal activity, except as necessary for monitoring or research.

Other managed areas: This includes areas that already have a legislative designation and include: state parks, national wildlife refuges, and estuarine reserves.

Definitions
of EFH Terms
by EFH Committee

Essential fish habitat (EFH) means those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity. For the purpose of interpreting the definition of essential fish habitat: "Waters" include aquatic areas and their associated physical, chemical, and biological properties that are used by fish and may include aquatic areas historically used by fish where appropriate; "substrate" includes sediment, hard bottom, structures underlying the waters, and associated biological communities; "necessary" means the habitat required to support a sustainable fishery and the managed species' contribution to a healthy ecosystem; and "spawning, breeding, feeding, or growth to maturity" covers a species' full life cycle. (EFH Final Rule 600.10)

Habitat Areas of particular concern (HAPC) : , Subsets of EFH that are identified by a Council under 50 CFR 600.815(a)(8) Councils should identify specific types or areas of habitat within EFH as habitat areas of particular concern-based on one or more of the following considerations:

(i) The importance of the ecological function provided by the habitat.

(ii) The extent to which the habitat is sensitive to human-induced environmental degradation.

(iii) Whether, and to what extent, development activities are, or will be, stressing the habitat type.

(iv) The rarity of the habitat type.

Sources:

Essential Fish Habitat Final Rule 50 CFR 600

Marine Protected Areas in Alaska: Recommendations for a public process. Regional Information Report 5J02-08, Alaska Department of Fish and Game, Juneau. July, 2002;

NRC, 2001. Marine protected areas: tools for sustaining ocean ecosystems. Washington, D.C., National Academy Press. 272p.

C-4
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April 4, 2003

David Benton, Chairman
North Pacific Fishery Management Council
605 W. 4th Avenue, Suite 306
Anchorage, AK 99501-2252

RE: Agenda Item C-4: Essential Fish Habitat

"The designation of HAPCs is a valuable way to highlight priority areas within EFH for conservation and management."

- Federal Register/ VOL. 67, NO 12/ January 17, 2002/ Rules and Regulations. pg 2357.

Dear Chairman Benton:

The North Pacific Fishery Management Council has made the commitment to include in the EFH SEIS, a process for public engagement with the Council in developing and designating Habitat Areas of Particular Concern (HAPC). Most recently, at the October 2002 meeting, the Council stated that HAPC sites and mitigation measures would be analyzed in a trailing amendment to the EFH SEIS.

> "The Council would not be designating any specific HAPC sites or types in the EFH SEIS. Instead, HAPC sites, together with any HAPC mitigation actions, would be analyzed in a trailing amendment. The Council further requested that the EFH Committee develop, for inclusion into the SEIS, a process for the public to interact with the Council in developing and amending HAPC designations in the future." NPFMC, October 2002.

AMCC recommends the NPFMC reconvene the EFH Committee to draft the process that will facilitate future stakeholder participation for developing and amending HAPCs. After the Council approves a stakeholder process, it should commence and run parallel to the development of the EFH SEIS. Specific HAPC sites and any appropriate mitigation measures can then be implemented on the same timeline as the EFH designation and mitigation measures adopted by the Council. The HAPC analysis should be relatively simple because stakeholders, scientists and the Council will have done the necessary groundwork during the HAPC development phase.

The HAPC stakeholder process should be made a high priority to address the existing holes in the EFH mitigation alternatives. There remain ecologically important, sensitive, vulnerable and rare habitats in the Gulf of Alaska, Bering Sea and Aleutian Islands that will not be addressed by adoption of any of the six EFH mitigation alternatives.[1] For example, there are known coral gardens in the Aleutians that should be considered for HAPC designation with mitigation alternatives that address all potential impacts. The Council has recognized that additional areas and gear types should be considered for mitigation through a HAPC process. The December 2002 motion on EFH states, "Scallop and Aleutian Islands longline crab fisheries will be considered more thoroughly under HAPC." **Since HAPCs are a subset of EFH and they are the most important areas of EFH, the NPFMC and NMFS must move ahead with the HAPC process.**

We would like to see a HAPC process established that incorporates the full involvement of coastal residents, fishermen, scientists and other stakeholders. To facilitate a sound process and product based on ecological criteria, we urge the Council to ensure substantial participation by federal, state and independent scientists. One way to do this is to have a habitat scientific committee that works separately to, but in cooperation with the stakeholder participants. The scientific committee would advise the stakeholder group on HAPC proposals before the group makes recommendations to the Council.

A shortcoming of the EFH Committee process was the lack of engagement by federal and state scientists in the actual design of mitigation measures. Although the EFH Committee had the benefit of the draft Rose/ Fujioka model to assess habitat impacts, the Committee was essentially creating mitigation alternatives without full access to the necessary data and scientific expertise. The result is that some of the Committee's recommendations are flawed in design. For example, the rotating closures in the Bering Sea components of Alternatives 4 and 5 have generated doubt and concern by scientists who question the efficacy of rotating closures in meeting habitat conservation objectives.

Summary:

- Known sensitive habitats are not being addressed in the EFH SEIS mitigation alternatives.
- A HAPC process can focus on discrete sites, informed by engaged stakeholder participants and scientific expertise.
- The groundwork for designing this process should begin now.
- Soon after, the HAPC process should commence and run parallel to the development of the EFH SEIS. Amendments and implementation of the product of this process should mirror the EFH SEIS timeline.

Sincerely,

Ben Enticknap
Fishery Project Coordinator

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[1] "FMPs should identify specific types or areas of habitat within EFH as habitat areas of particular concern based on one or more of the following considerations:

i. The importance of the ecological function provided by the habitat
ii. The extent to which the habitat is sensitive to human-induced environmental degradation.
iii. Whether, and to what extent, development activities are, or will be, stressing the habitat type.
iv. The rarity of the habitat type."

- EFH Final Rule. §600.815(a)(8)

MARINE
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ALYESKA SEAFOODS
ALASKA DRAGGERS ASSOCIATION
ALASKA GROUNDFISH DATA BANK
ALASKA PACIFIC SEAFOODS
ALEUTIAN PRIBILOF ISLAND COMMUNITY DEVELOPMENT ASSOC.
AKUTAN, ATKA, FALSE PASS, NELSON LAGOON, NIKOLSKI, ST. GEORGE
MID-WATER TRAWLERS COOPERATIVE
AT-SEA PROCESSORS ASSOCIATION
BRISTOL BAY ECONOMIC DEVELOPMENT CORP.
ALEKNAGIK, CLARK'S POINT, DILLINGHAM, EGEGIK, EKUK, EKWOK, KING SALMON, LEVELOCK, MANOKOTAK, NAKNEK, PILOT POINT, PORT HEIDEN, PORTAGE CREEK, SOUTH NAKNEK, TOGIAK, TWIN HILLS, UGASHIK
CENTRAL BERING SEA FISHERMEN'S ASSOC.
SAINT PAUL
CITY OF UNALASKA
COASTAL VILLAGES REGION FUND
CHEFORNAK, CHEVAK, EEK, GOODNEWS BAY, HOOPER BAY, KIPNUK, KONGIGANAK, KWIGILLINGOK, MEKORYUK, NAPAKIAK, NAPASKIAK, NEWTOK, NIGHTMUTE, OSCARVILLE, PLATINUM, QUINHAGAK, SCAMMON BAY, TOKSOOK BAY, TUNTUTULIAK, TUNUNAK
GROUNDFISH FORUM
HIGH SEAS CATCHERS
PERATIVE
SEAFOODS
KAN LEADER FISHERIES
NORTH PACIFIC FISHERIES RESEARCH FOUNDATION
NORTH PACIFIC LONGLINE ASSOCIATION
NORTH PACIFIC SCALLOP COOPERATIVE
NORTON SOUND ECONOMIC DEVELOPMENT CORP.
BREVIG MISSION, DIOMEDE, ELIM, GAMBELL, GOLOVIN, KOYUK, NOME, SAINT MICHAEL, SAVOONGA, SHAKTOOLIK, STEBBINS, TELLER, UNALAKLEET, WALES, WHITE MOUNTAIN
ALEUTIAN ISLANDS BROWN CRAB COALITION
PROWLER FISHERIES
TRIDENT SEAFOODS CORPORATION
SEAFOOD COLD STORAGE ASSOC.
SOUTHWEST ALASKA MUNICIPAL CONFERENCE
UNITED CATCHER BOATS
AKUTAN CATCHER VESSEL ASSOC.
ARCTIC ENTERPRISE ASSOC.
NORTHERN VICTOR FLEET
PETER PAN FLEET COOPERATIVE
UNALASKA CO-OP
UNISEA FLEET COOPERATIVE
WESTWARD FLEET COOPERATIVE
MOTHERSHIP GROUP
CELLENCE
EAN PHOENIX
LDEN ALASKA
TERN ALASKA ISHERIES, INC.
YUKON DELTA FISHERIES DEVELOPMENT ASSOC.
ALAKANUK, EMMONAK, GRAYLING, KOTLIK, MOUNTAIN VILLAGE, NUNAM IQUA

C-4
Ron Clarke
+ Heather McCarty

April 4, 2003

Mr. David Benton, Chairman
North Pacific Fishery Management Council
605 West Fourth Street, Suite 306
Anchorage, AK 99501-2252

AGENDA C-4, Essential Fish Habitat

Dear Chairman Benton:

The Marine Conservation Alliance has serious concerns about the Essential Fish Habitat (EFH) process and the lack of progress toward completing the Environmental Impact Statement (EIS) or securing an extension of the court-ordered deadline for its completion. We are dismayed that the plaintiffs have developed yet another new set of alternatives for you to consider at your present meeting; their adoption would further jeopardize timely compliance with the settlement agreement. The plaintiffs appear determined to drive the discussion outside of the Council process and eliminate public participation in deciding this important matter. We remain convinced this course would be counterproductive and urge you to take every action necessary to avoid that eventuality.

As you know, we have consistently maintained a position in favor of the Council and the National Marine Fisheries Service ("the Agency") completing the EFH EIS by the court-ordered deadline. We appreciate and support the aggressive schedule the Council and its EFH Committee has pursued in developing a complex and comprehensive range of alternatives. We are, however, concerned by the presentation of additional alternatives in the face of an impending deadline. Until the Agency applies for and is granted an extension, it is unreasonable to impose additional work on the analysts, resulting in failure to meet the deadline.

We are especially concerned by continuing efforts by the plaintiffs to create new EFH mitigation alternatives which would extend Alternative 5 (b)'s "open area" approach, developed by the plaintiffs for the Aleutian Islands, into the Bering Sea (BS) and the Gulf of Alaska (GOA). As was acknowledged by the EFH committee when they reviewed an open area alternative for the BS, differences in physical and biological regimes between the BS and the AI merit completely different approaches to each area, not simple export or extension of approaches used in the AI mitigation alternatives. The plaintiffs' interest in including their version of an "open area" approach for the BS and GOA is based on the erroneous assumption that fish concentrations have occurred consistently in the same locations in those regions over time. The Council's EFH Committee rejected this ill-informed assumption after a thorough discussion based on sound scientific information.

We believe the present range of Alternatives is scientifically sound and satisfies procedural demands for an adequate range of available options, and encourage the Council to move forward in completing the EIS by the August 1 deadline.

The MCA appreciates your continuing defense of the public process by which the Council's approach to this important issue has so far been shaped. We trust the Council's impending actions will be informed by this open public process as well. Thank you for consistently driving the discussion back into the public arena where it belongs.

Thank you.

Sincerely yours,

Ronald G. Clarke
Executive Director
Marine Conservation Alliance

HAPC Invertebrates of the Bering Sea

Sea onions

Hydroids

Sea potato

Sea anemones

Sponge

Bryozoan

Horny coral

Sea pen

Sea peach

Sea raspberry

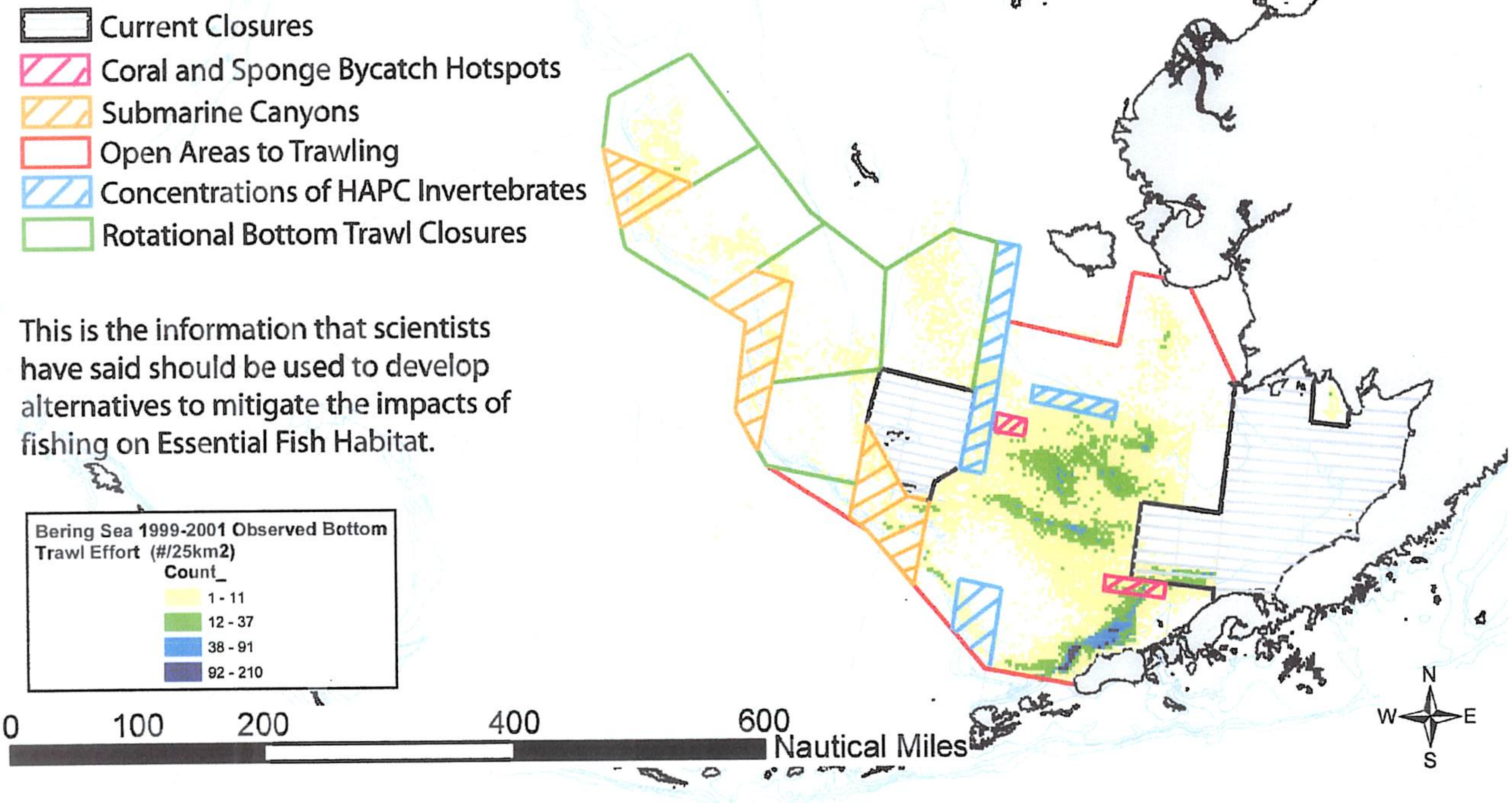
Images Courtesy NMFS-AFSC

Alternative for the Bering Sea

OCEANA

- ▭ Current Closures
- ▨ Coral and Sponge Bycatch Hotspots
- ▨ Submarine Canyons
- ▭ Open Areas to Trawling
- ▨ Concentrations of HAPC Invertebrates
- ▭ Rotational Bottom Trawl Closures

This is the information that scientists have said should be used to develop alternatives to mitigate the impacts of fishing on Essential Fish Habitat.

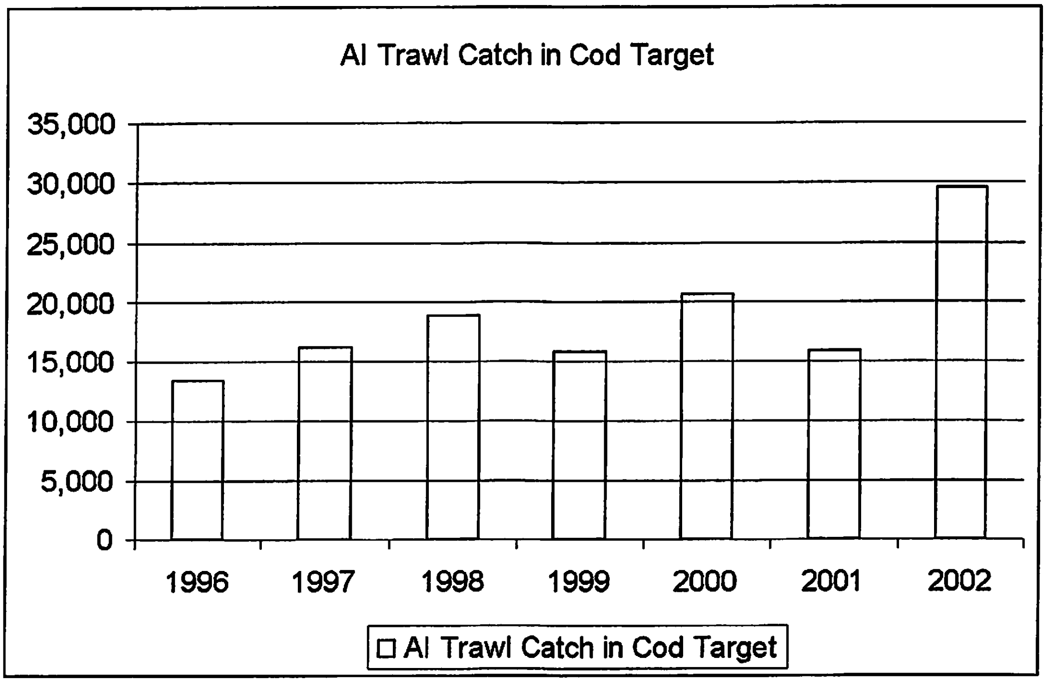


C-4
Dave Fraser

Habitat impacts on EFH are in part a function of effort levels. The Aleutian Island cod fishery has been experiencing an explosive growth of trawl effort in the last couple years.

The Council needs to task the Plan Team with considering a TAC split between the AI and BS for the protection of Essential Fish if not for the protection of Essential Fish Habitat.

Alternative 5b calls for TAC reductions in proportion to the catch in closed areas. However, there is one TAC for cod covering the BSAI. Thus any adjustment in the TAC is not constraining on AI harvests but could simply reduce the harvest from the BS portion of the TAC.



Year	BSAI Cod TAC	AI Groundfish Catch in Cod Target	Catch As % of TAC	AI Trawl Catch in Cod Target	Trawl % of Catch	Catch % in 541
1996	270,000	24,540	9%	13,386	55%	85%
1997	270,000	24,735	9%	16,083	65%	84%
1998	210,000	35,015	17%	18,846	54%	70%
1999	177,000	28,640	16%	15,731	55%	76%
2000	193,000	36,656	19%	20,597	56%	60%
2001	188,000	37,415	20%	15,882	42%	42%
2002	200,000	32,067	16%	**29,508**	**92%**	72%

C-4
Thorn Smith

Thorn Smith

From: "FIS" <fis@gci.net>
To: "Thorn Smith"
Sent: Wednesday, March 26, 2003 10:44 AM
Subject: Re: EFH update

Ballpark estimate for the fall cod fishery in 2002
Under EFH 6
we would have lost

18% of catch and 17% of fishable habitat.

For winter cod fishery in 2002

15% of catch and 14 % of fishable habitat.

These are very rough since I had to guesstimate which stat areas. Cathy Coon is now saying she sent me the files I needed in January, I keep everything that comes in to my email and there is no sign of them. Anyway she just sent them "again" so I should be able to do a better estimate some time later.

JS

> ----- Original Message -----
> **From:** Thorn Smith
> **To:** FIS
> **Sent:** Tuesday, March 25, 2003 12:22 PM
> **Subject:** Re: EFH update
>
> Ballpark!
>
> > ----- Original Message -----
> > **From:** FIS
> > **To:** Thorn Smith
> > **Sent:** Tuesday, March 25, 2003 1:27 PM
> > **Subject:** Re: EFH update
> >
> > Will give it a try, tomorrow AM.
> >
> > JS
> >
> > > ----- Original Message -----
> > > **From:** Thorn Smith
> > > **To:** FIS
> > > **Sent:** Tuesday, March 25, 2003 11:13 AM
> > > **Subject:** Re: EFH update
> > >
> > > Janet, I was hoping you could sort of eyeball it and say that it denies us about X % of our effective fishing grounds, Y% of our catch - something like that. Like, a very loose guess. If not, if it would take a lot of time, I guess we'll wait and comment on the analysis.
> > >
> > > > ----- Original Message -----
> > > > **From:** FIS
> > > > **To:** Thorn Smith

C-4
Thorn Smith

Coral in the Aleutians

At the January meeting of the North Pacific Management council, additional results of the draft analysis of the effects of fishing on Alaska EFH were presented. These included results representing the effects of using recovery and sensitivity values that better represented hard corals. That table did not include values for the effects of longlines and pots. Responding to a request from the North Pacific Longline Association, the following table includes results for all gear types.—Craig Rose

Table 1

Aleutian coral with sensitivity = 27% and recovery = 200 years						
	(bioshelter 30%, 5 years and SubstShelter = 2%, 100 years)					
Coral		0-200m			200-500m	
	Bioshelter	SubsShelter	Coral	Bioshelter	SubsShelter	Coral
CodTr	0.9	1.7	9.9	0.6	1.1	4.5
NrckTr	0.4	0.6	5.1	0.8	1.5	6.5
AtkTr	0.4	0.7	4.4	0.6	1.0	6.4
PopTr	0.2	0.3	2.9	0.5	0.8	4.1
CodLL	0.00	0.03	0.02	0.00	0.01	0.01
STLL	0.00	0.01	0.01	0.00	0.01	0.00
CodPot	0.00	0.00	0.00	0.00	0.00	0.00
Total	1.9	3.3	22.3	2.5	4.4	21.6

495 450