



Integrated Ecosystem Assessment of the California Current

Phase II Report 2012

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TABLE OF CONTENTS

| | |
|---|------------|
| Table of Contents..... | iii |
| List of Tables and Figures..... | i |
| Acknowledgments..... | ii |
| Preface..... | iv |
| What is an Integrated Ecosystem Assessment? | iv |
| Scope of this Report..... | v |
| EBM Components, Drivers, and Pressures in the California Current Ecosystem..... | v |
| Next Steps for the California Current IEA | vi |
| Developing the capacity to assess priorities - Engagement with stakeholders and Managers | 1 |
| Introduction | 2 |
| 2011-2012 Management Scenario interviews..... | 4 |
| 2011-2012 interview Synthesis | 9 |
| 2013 and beyond: engagement with stakeholders and management processes..... | 10 |
| References cited | 12 |
| Oceanographic and climatic drivers and pressures..... | 13 |
| Executive Summary | 16 |
| Detailed Report..... | 18 |
| References cited | 64 |
| Anthropogenic drivers and pressures..... | 71 |
| Executive Summary | 78 |
| References cited | 164 |
| Ecological Integrity | 179 |
| Executive Summary | 184 |
| Detailed Report..... | 186 |
| References cited | 230 |
| Chinook and coho salmon | 244 |

| | |
|--|------------|
| Overview | 249 |
| Executive summary | 249 |
| Detailed report..... | 251 |
| References Cited..... | 293 |
| Seabirds | 295 |
| Executive summary | 298 |
| Detailed Report..... | 301 |
| References cited | 331 |
| Marine Mammals – Indicators and Status | 339 |
| Executive summary | 342 |
| Detailed report..... | 344 |
| Literature Cited..... | 368 |
| Coastal pelagics and forage fishes..... | 374 |
| Executive Summary | 378 |
| Detailed Report..... | 379 |
| References cited | 397 |
| Groundfish | 400 |
| Executive Summary | 407 |
| Detailed Report..... | 410 |
| References cited | 468 |
| Resilient and economically viable coastal communities | 473 |
| Executive summary | 476 |
| Detailed report..... | 479 |
| References cited | 497 |
| Ecological Integrity Risk Assessment | 500 |
| Executive Summary | 505 |
| Detailed Report..... | 507 |
| References cited | 571 |

| | |
|---|------------|
| Marine Mammals Risk Assessment | 585 |
| Executive Summary | 591 |
| Detailed Report..... | 594 |
| References cited | 626 |
| Groundfish Risk Assessment..... | 629 |
| Executive Summary | 637 |
| Detailed Report..... | 639 |
| Non-fisheries threats data | 707 |
| Climate Change threats | 722 |
| References cited | 726 |
| Management Testing and Scenarios in the California Current..... | 730 |
| Summary of conceptual framework..... | 736 |
| Full description of scenario rationale..... | 739 |
| Full description of methods | 748 |
| Detailed results..... | 752 |
| Synthesis: Lessons Learned | 772 |
| References Cited..... | 777 |
| Appendix MS1. Assessing potential conflicts with wave energy generation along the Oregon coast | 779 |
| Appendix MS2: Forecasting the response of Klamath Basin Chinook populations to dam removal and restoration of anadromy versus no action | 809 |
| Appendix MS3. Impacts of depleting forage species in the California Current | 860 |
| Appendix MS4. Variable impacts of future fisheries development in the California current on ecosystem stability and spatially explicit biomass patterns..... | 861 |
| Appendix MS5. Biological and Economic Effects of Catch Changes Due to the Pacific Coast Groundfish individual quota system..... | 886 |
| appendix MS6. Finding the accelerator and brake in an individual quota fishery: Linking ecology, economics, and fleet dynamics of US West Coast trawl fisheries | 901 |
| Appendix MS7. Commercial Fishing Economics Technical Report For the Secretarial Determination on Whether to Remove Four Dams on the Klamath River in California and Oregon | 902 |

LIST OF TABLES AND FIGURES

Refer to the front of each section within the report for a more detailed table of contents and complete list of figures and tables. Figures, tables, and page number prefixes are based on a section name abbreviation, as listed below.

| | |
|--|-----|
| List of Tables and Figures (OC) | 14 |
| List of Tables and Figures (AP) | 74 |
| List of Tables and Figures (EN) | 181 |
| List of Tables and Figures (S) | 246 |
| List of Tables and Figures (SB) | 297 |
| List of Tables and Figures (MM) | 341 |
| List of Tables and Figures (C) | 376 |
| List of Tables and Figures (GF) | 402 |
| List of Tables and Figures (HD) | 475 |
| List of Table and Figures (ENR) | 502 |
| List of Tables and Figures (MMR) | 588 |
| List of Tables and Figures (GFR) | 632 |
| List of Tables and Figures* (MS) | 734 |
| List of Tables and Figures (MS7) | 905 |

*Does not include appendix tables and figures

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PREFACE

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WHAT IS AN INTEGRATED ECOSYSTEM ASSESSMENT?

NOAA defines an ecosystem as a “geographically specified system of organisms (including humans), the environment, and the processes that control its dynamics.” NOAA further defines the environment as “the biological, chemical, physical, and social conditions that surround organisms. When appropriate, the term environment should be qualified as biological, chemical, and/or social.”

An ecosystem management approach is one that provides a comprehensive framework for marine, coastal, and Great Lakes resource decision-making. Integrated ecosystem assessments (IEAs) are a critical science support element enabling ecosystem-based management (EBM) strategies. An IEA is a formal synthesis and quantitative analysis of information on relevant natural and socioeconomic factors in relation to specified ecosystem management goals. It involves and informs citizens, industry representatives, scientists, resource managers, and policy makers through formal processes to contribute to attaining the goals of EBM.

An IEA uses approaches that determine the probability that ecological or socioeconomic properties of systems will move beyond or return to within acceptable limits as defined by management objectives. An IEA must provide an efficient, transparent means of summarizing the status of ecosystem components, screening and prioritizing potential risks, and evaluating alternative management strategies against a backdrop of environmental conditions. To this end, IEAs follow the following steps:

- Scoping: Identify management objectives, articulate the ecosystem to be assessed, identify ecosystem attributes of concerns, and identify stressors relevant to the ecosystem being examined.
- Indicator development: Researchers must develop and test indicators that reflect the ecosystem attributes and stressors specified in the scoping process. Specific indicators are dictated by the problem at hand and must be linked objectively to decision criteria.
- Risk Analysis: The goal of risk analysis is to fully explore the susceptibility of an indicator to natural or human threats as well as the ability of the indicator to return to its previous state after being perturbed.
- Evaluation: Evaluate the potential different management strategies to influence the status of ecosystem components of management concern or the drivers and pressures that affect these ecosystem components.

Further description of IEAs can be found in Levin et al. (2008, 2009).

SCOPE OF THIS REPORT

The primary goal of the California Current IEA is to inform the implementation of EBM by melding diverse ecosystem components into a single, dynamic fabric that allows for coordinated evaluations of the status of the California Current ecosystem. We also aim to involve and inform a wide variety of stakeholders and agencies that rely on science support for EBM, and to integrate information collected by NOAA and other federal agencies, states, non-governmental organizations, and academic institutions. The essence of IEAs is to inform the management of diverse, potentially conflicting ocean-use sectors. As such, a successful California Current IEA must encompass a variety of management objectives, consider a wide-range of natural drivers and human activities, and forecast the delivery of ecosystem goods and services under a multiplicity of scenarios.

A full IEA of the California Current is thus a massive undertaking. Our approach to the daunting task of completing this IEA was to systematically decompose the California Current into a series of ecosystem components and ecosystem pressures that are of keen interest to resource managers, policy makers, and the public. Working with regional managers, we then selected a limited set of EBM components and pressures that we could address in the initial phase of the IEA (Levin and Schwing 2011)). This dialogue is ongoing (Scoping and Engagement), and thus we expect this framework to evolve over time.

EBM COMPONENTS, DRIVERS, AND PRESSURES IN THE CALIFORNIA CURRENT ECOSYSTEM

We define EBM components as the biological, physical, or human dimension entities that policy makers, managers, or citizens are trying to manage or conserve. Defined this way, the list of management concern targets is quite long; however, the IEA Action Team grouped these into six bins:

- Habitat—including biogenic and abiotic habitats both on the seafloor and in the water column.
- Wild fisheries—this EBM component is centered on the condition of fishery stocks included in the coastal pelagic species, highly migratory species, groundfish, and salmon fishery management plans.
- Ecosystem integrity—refers to the structure and function of marine and coastal ecosystems and ecological communities.
- Vibrant coastal communities—including social, economic, and cultural well-being and human health as it is tied to the marine environment.
- Protected resources—species legally designated as protected (e.g., Marine Mammal Protection Act, Migratory Bird Treaty Act, Endangered Species Act).

The ultimate aim of the California Current IEA is to fully understand the web of interactions that links drivers and pressures to EBM components and to forecast how changing environmental conditions and management actions affect the status of EBM components. In this, the second year of our IEA work, we focused on four EBM components:

- Ecosystem Integrity
- Fisheries (groundfish and coastal pelagic species)
- Protected species (marine mammals, seabirds, Pacific salmon)
- Vibrant coastal communities

Similarly, a lengthy list of drivers and pressures was created. Here, we define drivers as factors that result in pressures that in turn cause changes in the ecosystem. For the purposes of an IEA, both natural and anthropogenic forcing factors are considered; an example of the former is climate variability while the latter include factors such as human population size in the coastal zone and associated coastal development, demand for seafood, etc. In principle, human driving forces can be assessed and controlled. Natural environmental changes cannot be controlled but must be accounted for in management. Pressures include factors such as coastal pollution, habitat loss and degradation, and fishing effort that can be mapped to specific drivers. For example, coastal development results in increased coastal armoring and the loss of associated intertidal habitat.

As we did for EBM components, we binned drivers and pressures into a series of broad categories (Levin and Schwing 2011). These are:

- Shipping
- Freshwater habitat loss or degradation
- Coastal zone development
- Fishing
- Invasive species
- Naval exercises
- Aquaculture
- Energy development
- Marine habitat disturbance
- Oil spills
- Climate change

Status, trends and impacts of oceanographic / climatic drivers and anthropogenic pressures are addressed throughout the IEA. Most prominently, we discuss status and trends of drivers and pressures in Chapters 2 and 3. Additionally, for some ecosystem components we examine the risk to the component from specific drivers or pressures. Finally, in Chapter 10 we articulate a series of scenarios that link large-scale drivers to pressures in the California Current, and then use a variety of techniques to estimate how the status of ecosystem components might change under different scenarios.

NEXT STEPS FOR THE CALIFORNIA CURRENT IEA

This report is the second in a series of efforts to complete a full IEA of the California Current. In addition to improving analytical techniques, models, and filling data gaps, the third iteration of

the IEA will expand to include more ecosystem components and pressures. Specifically, in FY2013 the California Current IEA will add “habitat” as an EBM component. In addition, existing EBM components will be expanded in several ways: “highly migratory species” (e.g., albacore, sharks, etc.) will be added to the fisheries component; risk assessments will be added to or improved for all the components; and, where appropriate, additional indicators will be evaluated and included in analyses.

In this document, we develop a semi-quantitative approach to conduct an ecosystem risk assessment and apply this approach to a limited set of human activities and ecosystem components in the Monterey Bay National Marine Sanctuary. In subsequent years, this approach will be extended to include additional regions, human activities and ecosystem components throughout the California Current. In addition, we envision more quantitative risk analyses will be developed for some ecosystem components. Chapter 10 lays out several detailed scenarios that underlie the management strategy evaluations we conducted. FY2013 will see the development of additional quantitative approaches that will allow us to more fully evaluate these scenarios.

Levin, P. S., M. J. Fogarty, S. A. Murawski, and D. Fluharty. 2009. Integrated ecosystem assessments: developing the scientific basis for ecosystem-based management of the ocean. *PLoS Biology* 7:e1000014 doi:1000010.1001371/journal.pbio.1000014.

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DEVELOPING THE CAPACITY TO ASSESS POLICY PRIORITIES - ENGAGEMENT WITH STAKEHOLDERS AND MANAGERS

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OVERVIEW

Engagement with resource managers along the west coast began in 2010. Significant progress has been made with the Pacific Fishery Management Council and Monterey Bay National Marine Sanctuary identifying ways management considers ecosystem science and translates it into management actions and decisions.

INTRODUCTION

NOAA is primarily a science agency, although its research priorities are shaped by a host of laws that govern management of or consider human interactions with the natural world. A common theme in those laws is that, whether in predicting paths of hurricanes or protecting endangered species, the United States must strive toward, develop, and use the “best available” science tools and analyses. Integrated Ecosystem Assessments (IEAs) are, in part, an attempt by NOAA and its partners to develop next-generation science tools that expand how the best available science characterizes the relationships between species (including humans) within food webs and between those species and the physical world and its dynamic processes. In Levin et al. 2008, 2009, NOAA staff and colleagues first outlined the approach the agency hoped to take to IEAs, emphasizing roots in international efforts to frame the science-policy dialogue needed to implement ecosystem-based management (Caddy 1999, Sainsbury et al. 2000, Smith et al. 2007).

As envisioned in Levin et al. 2008, 2009, a first step in IEAs should be scoping, a process intended to identify management goals within a given ecosystem and the potential challenges or threats to achieving those goals. In de Reynier et al. (2010), NOAA staff explored the IEA scoping process in more detail, discussing the potential challenges of conducting a formal scoping process with a public and with managers unfamiliar with IEAs. To familiarize stakeholders and the public with the IEA concept, and to better engage scientists in discussions with other stakeholders, deReynier and colleagues recommended a basic first step of educating potential IEA users about the possibilities and limits of IEA science for a given ecosystem.

In 2011, NOAA released its first California Current IEA science products in a NOAA Technical Memorandum (Levin and Schwing 2011), largely intended to showcase the kinds of scientific analyses possible given available data, models, and technology. With this Technical Memorandum and with other agency staff publications on the California Current emerging, we were developing the scientific base to begin educating managers and the public about the state of knowledge on drivers, pressures, and interactions within the California Current Ecosystem. Over 2011-2012, NOAA science staff have met with a host of entities to begin the California Current IEA (CCIEA) education process, engaging in forums including: the Pacific Fishery Management Council, the California Cooperative Oceanic Fisheries Investigations partnership, the NOAA Science Advisory Board, the Monterey Bay National Marine Sanctuary, Ecosystem Based Management Tools Network, and West Coast Governors Alliance on Ocean Health.

For 2012, CCIEA scientists developing management strategies for the California Current worked with 16 resource managers and stakeholders to identify:

- Drivers and pressures in the California Current
- Management options for coping with these drivers and pressures
- Metrics for success in addressing drivers and pressures.

In this context, pressures are human activities or natural processes that cause some impact on the condition of the ecosystem; drivers are forcing factors that result in pressures that in turn cause changes in the system; and metrics of success are the socially-determined measures by which policy makers or stakeholders judge whether they have reached their goals. We investigated drivers, pressures, management options, and performance metrics specifically to inform the Management Strategy Evaluation portion of the CCIEA, where a team of modelers applied a variety of models to evaluate possible futures for the California Current. These discussions between CCIEA scientists, managers, and other stakeholders do not constitute scoping. Instead, the discussions served as a test for how scientists might bring the perspectives of other stakeholders into the CCIEA science process to test different resource management strategies.

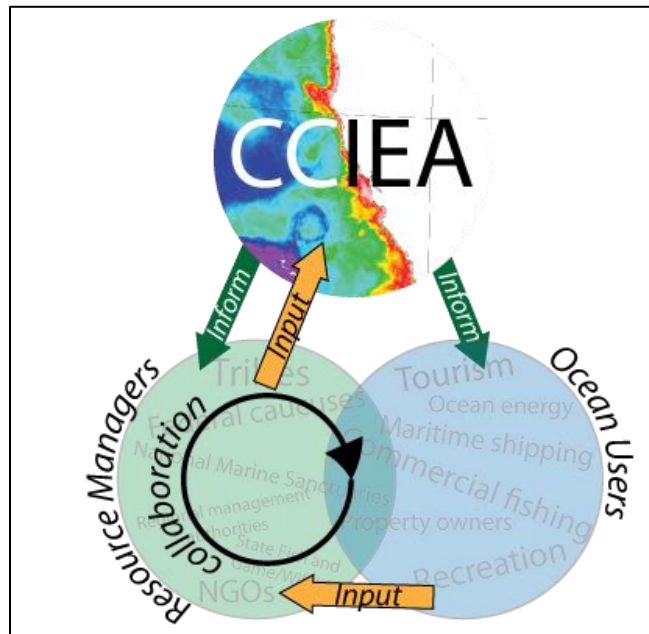


Figure EG1. CCIEA communication cycle

Although NOAA believes that scoping is essential to conducting a complete IEA, to developing tools for ecosystem-based management, and to assessing whether scientific products are bending toward “best available” science, the agency has no immediate plans to conduct broad public scoping in connection with the CCIEA. During 2012 discussions with managers and stakeholders, CCIEA scientists became aware that there is a host West Coast coastal and marine resource management processes already underway, many of which conduct regular scoping with their stakeholders. Rather than initiating a new scoping process, the agency plans to deepen its education and engagement efforts in 2013 so that its science products can become more responsive to existing resource conservation and management processes and mandates. As illustrated in Figure EG1, NOAA anticipates that developing the CCIEA will require ongoing communication efforts. Below, we discuss the 2011-2012 manager/stakeholder interview process used to develop and test the management strategy evaluations, and, we identify some of the regional science and management partnerships that could be useful in further assessing regional priorities for conserving and managing the California Current Ecosystem.

Box 1: Affiliations of CCIEA

Manager/Stakeholder Interviewees

Northwest Indian Fisheries Commission

Monterey Bay Aquarium

California Fish and Game Commission

West Coast Seafood Processors Association

Washington Department of Natural Resources

Pacific Fishery Management Council

NOAA Fisheries’ Northwest Region

NOAA Fisheries’ Southwest Region

Natural Resources Consultants, Inc.

National Marine Sanctuaries

The Nature Conservancy

University of Washington

2011-2012 MANAGEMENT SCENARIO INTERVIEWS

We intended this initial set of interviews with managers and other stakeholders to inform ongoing science in 2012, rather than as a comprehensive overview of West Coast marine policy priorities. We identified interviewees based on their expertise with respect to the attributes of interest to 2012 CCIEA scientists: protected species, ecosystem integrity, fisheries, human communities, and habitat. We did not attempt a broad or representative survey, nor did we attempt to get a balanced amount of input for each of the attributes. In Box 1, we identify the affiliations of interviewees. These experts participated in interviews as individuals, not as representatives of opinions or policy stances of the organizations with which they were affiliated. In general, conversations focused on issues related to groundfish, salmon, marine mammals, and forage fish, with less focus on other protected species, habitat and human communities. This set of issues likely reflects both the bias in our selection of experts, and the pressing management questions in 2012. Interviews were conducted by telephone in March 2012, were typically 30-60 minutes long, and involved only one expert at a time (one exception involved two people from a single non-governmental organization). Experts discussed topics that matched their areas of expertise and declined to discuss other topics. They identified main drivers and pressures, management options, and metrics of success. Drivers and pressures were discussed in the context of the next 10-30 years, except for issues related to climate change, which typically involved longer time frames. Experts listed drivers and pressures even in cases for which they were not aware of any related management options. Generally they were asked not to focus narrowly on particular quantitative methods (in the context of the CCIEA) that might eventually be applied to themes elicited here. The interviewees identified a broad set of drivers and pressures on the California Current ecosystem, including aspects not included as targets of management concern in the CCIEA (Fisheries, Protected Species, Habitat, Human Communities, and Ecological Integrity).

Below, the main themes from the 16 interviews are organized by category. The diagrams and related themes can be used in the context of the CCIEA to ask:

- What drivers and pressures may affect the California Current?
- What are impending management needs or mandates, related to current issues or future drivers and pressures?
- What existing or potential scientific and resource management tools can address these needs, drivers, and pressures?
- How can we test and judge new management strategies that could address these needs, drivers, and pressures (via either virtual testing or in the real world)?

In many cases these themes clearly involve drivers and pressures in the IEA terminology, but there is some blurring of language because the themes are taken from informal interviews. Note also that many of these drivers are not independent: for instance population growth is related to demands for energy and water, but since several experts discussed these topics separately, we have treated them separately here. We summarized conversation themes from the interviews into a set of five narrative (and graphical) scenarios, described in more detail below, focused on key drivers of the California Current:

- human population growth,
- climate change,
- conservation demands,
- energy crunch
- status quo

POPULATION GROWTH ISSUES

Human West Coast population growth was mentioned by experts primarily as a driver for freshwater and nearshore habitats, particularly for salmon – see Figure EG2. Experts directly involved in salmon management mentioned conflicts among water availability for salmon, agriculture, and urban populations. Summer was identified as a critical period, when water supplies were lowest and agricultural demand greatest. Three experts discussed the synergism between this water demand and climate change, which is predicted to cause decreased snowpack and more acute water shortages in the summer. Management actions

that might mitigate these effects included decreased salmon harvest and potential changes in dam water management. One such change in water management would be to reduce intentional spills of water during the winter, which are typically conducted to leave capacity in reservoirs for flood control. Reduced winter spills would lead to increased reservoir retention of water into drier periods of the year, but with the risk of potential winter flood damage. Other potential impacts of West Coast population growth included additional ship-based and terrestrial pollution, and increased need for energy infrastructure, but details of the scope and severity of these were generally outside the expertise of the selected experts.

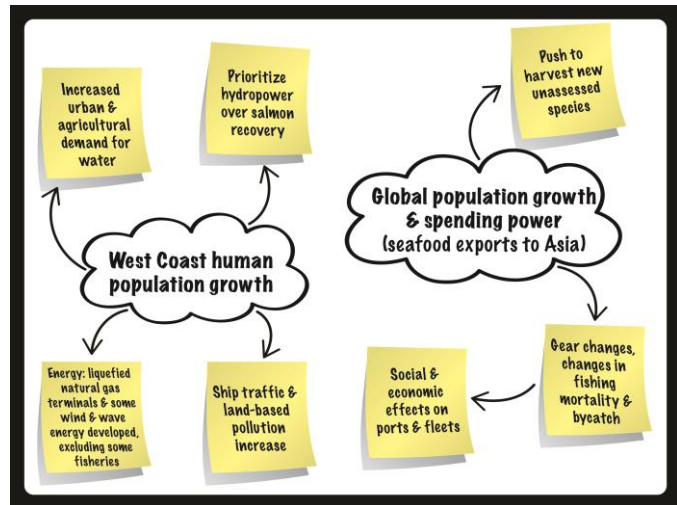


Figure EG2: Population growth scenario issues

State and federal managers discussed increased seafood demand due to global population growth and rising affluence of global consumers, particularly in Asia. Managers particularly cited export of Dungeness crab to China as one recent development, as well as strong markets for octopus, geoducks (*Panopea generosa*), live rockfish (*Sebastes* spp.), and hagfish (*Eptatretus* spp.). Harvest of species such as geoducks requires gears that are potentially damaging to habitat; harvest of live rockfish focuses on different size and age classes than trawl gears. Countering the trend for increased demand for wild-caught West Coast fish, increased global aquaculture and imports to the US reduce demand for low-value whitefish such as Dover sole (*Microstomus pacificus*). Comparison of the environmental cost of imported aquaculture-raised seafood versus locally caught seafood was raised as a potential research topic. Federal managers were generally confident that safeguards were in place to prevent rapid development and overexploitation of new species; however several of the new nearshore target species are managed by state agencies.

CLIMATE CHANGE ISSUES

The majority of experts (13 of 16) discussed threats to the California Current from climate change and ocean acidification, typically focusing on timescales of several decades or more – see Figure EG3. Salmon (*Oncorhynchus* spp.) were a common focus due to potential changes in streamflow (warmer winters and less snowpack), which could impact stream-type fish such as spring Chinook. Potential northward shifts of the southern extent of salmon ranges were listed as one threat to California and Oregon salmon populations. Additionally, the vulnerability of salmon prey, such as pteropods,

to ocean acidification was listed as one potential effect that could lead to declines in salmon abundance. Potential climate change effects for other marine species included increased frequency of shifts between sardine (*Sardinops sagax*) and anchovy (*Engraulis mordax*) abundance, and northward shifts in ranges for sardine and hake. Two experts mentioned that the effects on species distribution and productivity caused by warming, acidification, and hypoxia were likely to be spatially patchy, as well as varying by latitude, and therefore the economic impacts would differ between ports.

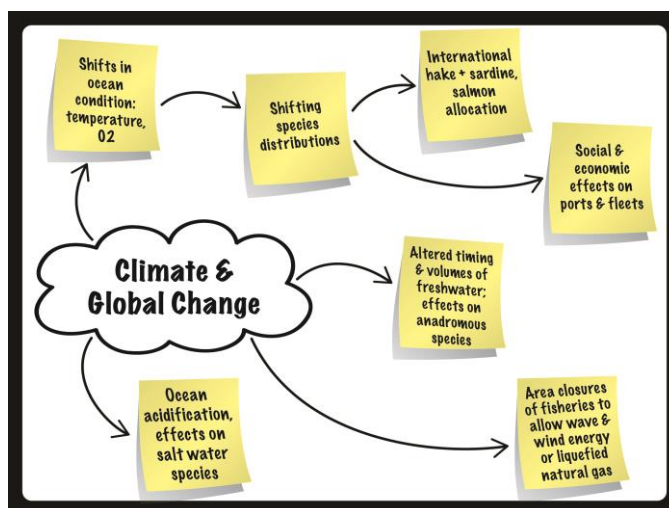


Figure EG3: Climate change scenario issues

Very few specific policy actions were mentioned by experts in relation to climate change and ocean acidification. The primary sentiment from experts was that they would assess climate change impacts through existing monitoring programs; reductions in harvest were often mentioned as the policy response. Two salmon managers identified habitat restoration in streams as a method to mitigate climate change. Several experts pointed out the high degree of uncertainty regarding the exact long-term implications of climate change and acidification. One expert felt that overall the link between climate phenomena (such as El Niño-Southern Oscillation and the Pacific Decadal Oscillation) and marine communities was poorly understood, and improved understanding of these phenomena was a necessary step to scientifically addressing trends in global change. Two experts identified increased community-based management, monitoring, and allocation, as methods to identify and manage for spatially patchy effects of climate change. These two experts suggested that community-based management at the scale of ports or clusters of nearby ports could respond to localized changes in ocean conditions. Though most of the 16 interviews focused on local management actions related to fisheries, respondents also mentioned potential shifts in national energy policy, such as development of alternative energy and liquefied natural gas facilities.

CONSERVATION DEMAND ISSUES

Twelve of 16 experts discussed potential management actions and consumer choices that aim to protect or recover particular species or ecosystem components – see Figure EG4. Potential conservation management actions included increased “set-asides” for forage fish (thresholds of abundance below which harvest is prohibited). Increasing these thresholds might increase the availability of forage for marine mammals, birds, and other predators. Adoption of catch shares (individual transferable quotas) for additional fisheries was discussed as one means to reduce bycatch and prevent catches from exceeding quotas, as well as to increase profitability. Development of regional community-based management was

stated as one method to improve data collection and flexible management responses aimed at conserving marine stocks. Conservation actions to increase abundance of salmon included harvest reductions and time/area closures, as well as additional and ongoing dam removal (for example, in the Elwha River). Ship strikes of marine mammals and entanglement of marine mammals, birds, and turtles in fishing gear were mentioned by several experts as motivation for potential spatial management actions. Five experts stated that there major scientific gaps in understanding forage needs for killer whales, and the impact of forage species harvest on the rest of the food web. They noted the need to identify key forage species, concerns regarding local depletion of forage species (sardine or squid) by fisheries near seabird or seal rookeries, and a need to quantify the economic value of forage species consumed by harvested predators.

Policy developments likely to lead to further conservation actions included implementation of Ocean Commission recommendations and the National Ocean Policy, regional governance efforts such as the West Coast Governors Alliance, and spatial planning within state waters. One manager felt that improved coastal and marine spatial planning (CMSP) was likely to resolve many spatial conflicts between fishing, shipping, and conservation needs; others felt that whether CMSP was likely to develop in each state depended on the local management and political climate.

Experts expressed mixed views on the impact of seafood eco-labeling (e.g. certification or rankings of sustainability) and the preference for local seafood. Salmon fisheries with strong exports to Europe were cited as a case where eco-labeling was likely to alter both prices and fishery practices, since-European markets were said to generally respond positively to eco-labeled products. One expert pointed out that there are many eco-labeling schemes available to the industry, with a variety of standards. A second expert pointed out that consumers were demanding higher quality seafood, but not necessarily eco-labeled or local fish. A contrasting observation was that there is increased demand for locally caught, high-value fish such as swordfish and albacore. NOAA FishWatch, a consumer seafood education website, was identified as one alternative to eco-labeling schemes.

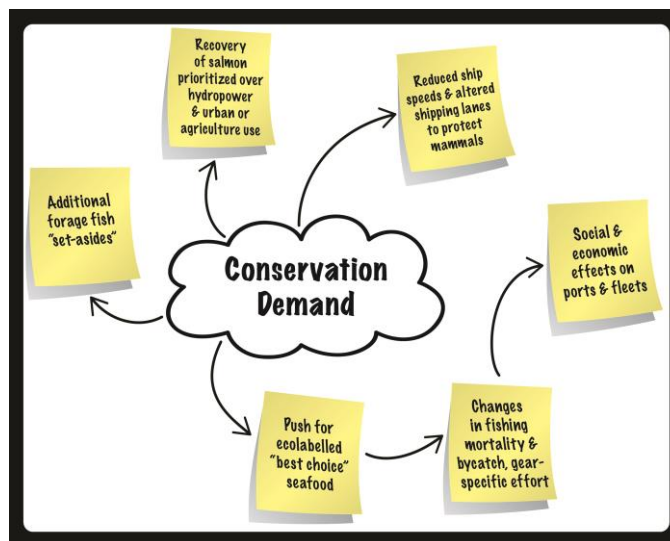


Figure EG4: Increased demand for conservation scenario issues

ENERGY CRUNCH ISSUES

Future increases in price of diesel fuel were generally predicted to lead to changes in fishing fleet operations, and increased establishment of energy facilities (wind, wave, or liquefied natural gas) were stated as likely to lead to reductions in fishing areas near such facilities – see Figure EG5. Most experts assumed some future increase in fuel price for fishing vessels. Two stakeholders also mentioned the high fuel demand involved in processing and transporting fish, narrowing profit margins, and the negative impact that gasoline prices have on consumer demand for seafood. Trawl, albacore troll, mackerel purse seine, and recreational fleets were identified as being fuel intensive or sensitive to fuel price. Salmon experts mentioned ongoing tradeoffs between hydropower and salmon, but also did not foresee reduced protections for salmon under the Endangered Species Act.

Wave energy facilities were identified as a policy response to the energy crunch, and experts cited new pilot projects near Reedsport and Newport, Oregon. Potential impacts from these could include acoustic (sound) impacts on marine mammals. Fisheries could be directly affected if they were excluded from operating near wave energy facilities. Wave, wind, and LNG facilities were mentioned by three respondents as potentially having ecological effects similar to marine protected areas, including local increases in abundance of fish and demersal species.

Finally, experts considered potential changes to shipping traffic in relation to increased energy prices. This was discussed in terms of increased shipping as industries push for low-cost methods (freighters, tankers) to move goods. Additional increases in West Coast marine traffic could stem from tanker transport of Canadian crude oil from Pacific Northwest ports. Expansion of the Panama Canal was also discussed as likely to restructure West Coast shipping patterns, allowing more direct shipping from Asia to the U.S. East Coast rather than to West Coast

shipping terminals. Potential management actions to mitigate the effects of shipping on marine mammals included reduced ship speeds and altered shipping lanes.

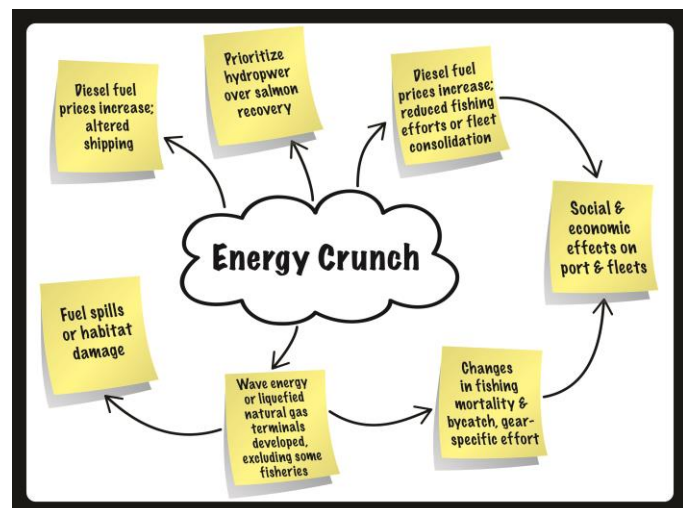


Figure EG5: Energy crunch scenario issues

STATUS QUO MANAGEMENT ISSUES

Comments about Status Quo management primarily addressed challenges within the existing management process, and issues relate to the groundfish catch shares program that was implemented in January 2011 – see Figure EG6. Lengthy multi-year review processes and lags between data collection and fishery management actions were identified as one impediment to rapid, flexible responses to shifting stock abundances. Two respondents also pointed out that many fishery restrictions on groundfish gear specifications and areas fished (e.g. Rockfish Conservation Areas) may now be counterproductive under a catch share program that aims to foster innovation and incentives for individual vessels to reduce bycatch.

The high costs of management, monitoring, and assessment were identified as one factor that may limit the continued operation of some less economically viable fisheries in the future.

The implications of the groundfish catch share program for fisheries and marine species were discussed by eight of the 16 experts. These managers and stakeholders were aware of quota that was not being harvested, primarily for flatfish, due to constraining bycatch of rockfish. Opinion differed on whether those flatfish quotas might be met via the catch shares system. One expert stated that there was unlikely to ever be sufficient demand; another said that deeper water species might be targeted more effectively, but that nearshore species could not be targeted without exceeding rockfish bycatch quotas. One alternate opinion was that new midwater rockfish fisheries, targeting widow (*S. entomelas*) and yellowtail rockfish (*S.*

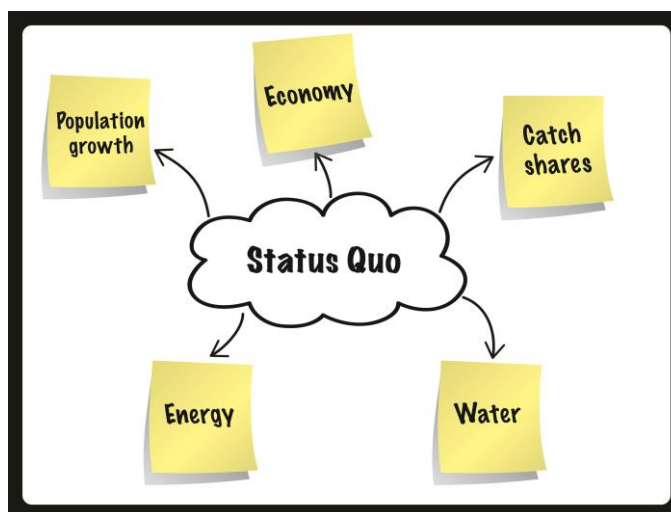


Figure EG6: Status quo scenario issues

flavidus), might evolve as fishers improve their targeting precision. This would shift these two species from being avoided bycatch to being actively targeted. Two experts discussed the formation of risk pools being formed between fishermen, to pool the limited quotas of bycatch species such as rockfish. Such risk pools could reduce the likelihood that any single fisherman would be forced out of the fishery by unexpectedly high catches of bycatch species. Economic implications of the groundfish catch share program were mentioned by three experts, including potential fleet consolidation (reductions in number of vessels) and subsequent changes in fishing location and port utilization, changes in infrastructure, or potential movement of some vessels into open access fisheries. The potential for quota shares to be used as collateral was mentioned as one potential factor contributing to reinvestment in the fishery. Such reinvestment might lead to more fuel efficient vessels, since many groundfish vessels are over 20 years old and might be replaced. The main management action invoked to address challenges with the catch share fishery was flexibility in gears and areas fished, consistent with the individual incentives offered to fishers under the catch share program.

2011-2012 INTERVIEW SYNTHESIS

The interviewees identified both formal and informal measures of success for future management programs, but tended to focus on management of fish and marine mammals. For instance, metrics of success for marine fisheries included landed value, profitability, and rebuilding progress of overfished species. Practical goals included “no one going broke”, “no unhappy political constituents”, having a “responsive and coherent fishery management plan that is producing the expected results”, and “profitable fisheries that allow fishermen to invest in the industry and engage in management and sustainability.” Metrics of success for salmon were primarily those defined in salmon recovery plans (Viable Salmonid Population parameters), related to abundance, population growth rate, population spatial structure, and diversity. Access to the fishery was also stated as a metric of success, in terms of number of recreational angler trips and active commercial licenses. For marine mammal management, the number of strandings was mentioned as one metric, as were population growth rates, mortality relative to potential biological removals, and the economic value of whale watching. More comprehensive metrics of economic and ecological success included the

number of jobs in fishing sectors, the health of seafood consumed locally for subsistence, and the ability to keep pollution concentrations below allowable levels to ensure subsistence consumption. Perhaps reflecting the expertise of this set of managers and stakeholders, most of the metrics of success focused on wild fisheries and marine mammals (protected species). Other ecosystem components such as habitat, ecosystem integrity, human communities (aside from economics) and protected species (beyond mammals) were not emphasized.

The responses and themes from these interviews are not novel. In fact, in many cases experts suggested that they were simply communicating “common knowledge”. However, the breadth of experience from these 16 experts allows a somewhat synoptic view of current drivers, pressures, and management concerns in the region, and this is likely greater than that of any individual. Additionally, specific concerns raised in the interviews involve key details that can guide future research, which necessarily must move beyond broad-brush trends. For instance, climate change was suggested to have potentially strong effects specifically for spring-run Chinook salmon; pelagic mackerel and tuna fleets were identified as likely to be most sensitive to fuel prices; markets for Dungeness crab were linked specifically to rising Chinese import demand. Such details are essential for identifying and prioritizing future scientific analyses and ecological and economic monitoring.

These interviews provided motivation for the Management Strategy Evaluation portion of the 2012 and 2013 California Current Integrated Ecosystem Assessment. In the management strategy evaluation portion of the IEA we incorporate these themes into scenario narratives on population change, climate change, conservation demands, energy crunch, and evolution of status quo management. We then apply quantitative tools that allow us to investigate how some pressures affect attributes of interest for the IEA. Future outreach efforts will also continue to guide research related to risk assessment, status and trends of ecosystem components, and ecosystem drivers and pressures.

2013 AND BEYOND: ENGAGEMENT WITH CALIFORNIA CURRENT STAKEHOLDERS AND MANAGEMENT PROCESSES

In support of the CCIEA, NOAA will be continuing to engage with California Current Ecosystem stakeholders and management processes in 2013 and beyond. Our intent is to: 1) educate a larger audience on the capabilities and potential value of an IEA approach, 2) expand the range of stakeholder input incorporated in the CCIEA, and 3) build on the success of preliminary manager engagement by maintaining an ongoing dialogue between IEA scientists and other stakeholders. We plan to collaborate with multiple organizations, including the West Coast Governors Alliance on Ocean Health (WCGA) and West Coast EBM Network, to share resources and strengthen partnerships across and within governmental and non-governmental agencies.

To broaden our education efforts, we have launched a website on IEA work to date, and are developing webinars and other presentations for IEA outreach. Webinars began in fall 2012 and address: why IEAs can be useful to understanding ecosystem interactions, what science products are emerging from the California Current IEA, and the data and methods used to generate IEA science; how to engage with the IEA process. We are conducting webinars both to open new and to advance existing relationships with groups interested in the California Current and natural resource management. In 2013 and beyond, we plan to use webinars to share IEA science products and engage with:

- Internal NOAA staff

- WCGA IEA Action Coordination Team
- West Coast EBM Network
- West Coast Sanctuary Advisory Councils
- West Coast NGOs
- Tribal Groups
- Federal Caucuses (Columbia River, Bay Delta, Puget Sound)

Beyond webinars and other presentations to interested managers, stakeholder groups and the public, NOAA also plans to build on 2011-2012 manager/stakeholder interviews with a more broad-based questionnaire and targeted interviews intended to elicit public values for the California Current. The questionnaire is intended to help IEA scientists better sort through drivers and pressures within the California Current by soliciting more information and opinions on those drivers and pressures, and on potentially useful indicators of ecosystem status and ecosystem-based management strategies. Issues to be addressed with the questionnaire include:

- What are current regional resource management priorities?
- Are there geographic regions within the larger California Current are of particular interest and relevance to managers and other stakeholders?
- What management strategies are available under current legal authorities and funding constraints (what is on or off the table)?
- What are hurdles to achieving management goals (data limitations, bureaucratic, procedural, e.g.)?
- How do stakeholders measure management success (indicators - why a specific indicator? can you make a decision based on it? are there threshold values?)
- What indicators do resource managers use to make decisions on a monthly/annual basis?
- Current drivers and pressures (aquaculture, ocean energy, fishing, e.g.), and potential for interacting or cumulative impacts
- What are the best strategies for facilitating cooperative management between and among sectors?

We plan to make the questionnaire available online and distribute it through networks within NOAA and through NOAA partners. Information gathered through this process will be available to IEA scientists as reference for future IEA work.

Undertaking an ecosystem assessment for such a large region, encompassing thousands of jurisdictional boundaries and priorities, is an iterative and lengthy process, whether in development of defensible science products and processes, or in building relationships to allow policy expertise to enhance and inform the science process. Regional natural resource management and marine policy efforts have already established networks with stakeholders across multiple sectors, and have expertise and a mandated forum for stakeholder engagement. Members of NOAA's CCIEA group, who are primarily of biologists and ecologists, do not often interact with the cross-sectoral stakeholder community, which is why building relationships with other stakeholders is essential if CCIEA scientists are to conduct work useful to management processes. For these reasons, NOAA's CCIEA process will eschew scoping solely in support of the IEA, and instead focus more on using information already scoped through public policy processes, or on

tuning scientific products so that they more directly address questions from or issues under consideration by existing public policy processes.

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OCEANOGRAPHIC AND CLIMATIC DRIVERS AND PRESSURES

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LIST OF TABLES AND FIGURES (OC)

| | |
|---|----|
| Figure E1. Broad scale indices and status from 2008-2012 with anomaly and trend values. Anomalies are the difference between the mean of the past 5 years and the long-term mean while trends are calculated over the past 5 years. A negative MEI for both summer and winter indicates slight La Niña conditions. The NPGO is positive although decreasing, indicating a more productive CCLME. The negative PDO values indicate a cool, more productive phase of the CCLME. | 16 |
| Figure E2. The a. Spring Transition (STI), b) Length of Upwelling Season (LUSI), and c) Total Upwelling Magnitude Indices (TUMI) at 45°N. The three indices together give a metric of the upwelling season in the CCLME. | 17 |
| Figure OC1. Coastal sea level heights from 1906-2012 for both a. summer and b. winter. San Diego coastal sea level was chosen to illustrate patterns in the southern portion of the CCLME. | 21 |
| Figure OC2. Coastal sea level heights from 1898-2012 for both a. summer and b. winter. San Francisco coastal sea level was chosen to illustrate patterns in the central portion of the CCLME. | 22 |
| Figure OC4. Sea surface temperature (SST) buoy data from early 1990 -2012 during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. Buoy 46050 was chosen to illustrate patterns in the northern portion of the CCLME. | 24 |
| Figure OC5. Sea surface temperature (SST) buoy data from early 1990 -2012 and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. Buoy 46014 was chosen to illustrate patterns in the central portion of the CCLME. | 25 |
| Figure OC6. Sea surface temperature (SST) buoy data from early 1990 -2012 and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. Buoy 46025 was chosen to illustrate patterns in the southern portion of the CCLME. | 26 |
| Figure OC7. Pacific Decadal Oscillation (PDO) index values from 1900 -2012 during both a. summer and b. winter. | 27 |
| Figure OC8. Northern Oscillation Index (NOI) values from 1948 -2012 during both a. summer and b. winter. | 28 |
| Figure OC11. Pycnocline depth data from 1998 -2012 and during both a. summer and b. winter from Newport line station NH25, chosen to illustrate patterns in the northern portion of the CCLME. | 31 |
| Figure OC12. Pycnocline strength data from 1998 -2012 and during both a. summer and b. winter from station 67.55, chosen to illustrate patterns in the central portion of the CCLME. | 32 |
| Figure OC14. Pycnocline strength data from 1998 -2012 and during both a. summer and b. winter from Newport line station NH25, chosen to illustrate patterns in the northern portion of the CCLME. | 34 |
| Figure OC15. Eddy Kinetic Energy satellite data from 1992 -2012 at 33°N and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. 33°N was chosen to illustrate patterns in the southern portion of the CCLME. | 36 |

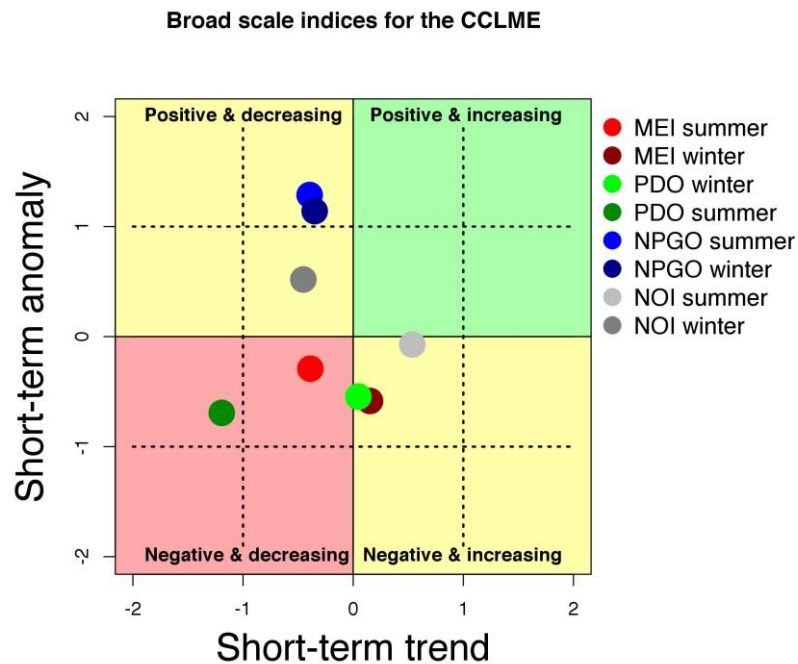
| | |
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| Figure OC16. Eddy Kinetic Energy satellite data from 1992 -2012 at 39°N and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. 39°N was chosen to illustrate patterns in the central portion of the CCLME..... | 37 |
| Figure OC17. Eddy Kinetic Energy satellite data from 1992 -2012 at 45°N and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. 45°N was chosen to illustrate patterns in the northern portion of the CCLME. | 38 |
| Figure OC18. The Upwelling Index calculated from 1967 -2012 at 33°N and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. 33°N was chosen to illustrate patterns in the southern portion of the CCLME. | 40 |
| Figure OC19. The Upwelling Index calculated from 1967 -2012 at 39°N and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. 39°N was chosen to illustrate patterns in the central portion of the CCLME..... | 41 |
| Figure OC21. The Spring Transition Index (STI) calculated yearly from 1967 -2012 at a. 39°N, and b. 45°N. 33° is not included because there is not a reliable downwelling phase each season. | 43 |
| Figure OC28. North Pacific Gyre Oscillation values (NPGO) from 1950 -2012 during both a. summer and b. winter..... | 51 |
| Figure OC29. Summer nutrient data (nitrate + nitrite) at 150m a. 1997 -2012 at station NH25 from the Newport line in the northern CCLME, b. 1999-2011 at CALCOFI station 67.55 for the central CCLME, and c. 1984-2011 at CALCOFI station 93.30 for the southern CCLME. | 52 |
| Table OC1. Top indicators for ocean and climatic pressures. Three stations were chosen when possible for northern, central, and southern portions of the California current. Time series availability often differed across the three locations..... | 61 |

OVERVIEW

At the end of 2011 and beginning of 2012, the California Current Large Marine Ecosystem continued to experience periods of strong upwelling and cooler sea surface temperatures associated with a cool phase of the Pacific Decadal Oscillation and a generally more productive ecosystem.

EXECUTIVE SUMMARY

The California Current Large Marine Ecosystem (CCLME) is primarily driven by bottom-up physical oceanographic processes, thus understanding trends in the physical state can inform our knowledge of ecosystem processes and management of ecosystem services. The Pacific Decadal Oscillation (PDO) and the North Pacific Gyre Oscillation (NPGO), indicators of sea surface temperature change and changes in ocean circulation respectively, continue to describe a cool phase that has been observed since 1999 in the CCLME (Figure EX1). Both of these indices change on decadal time scales so the past five years is only an indication of the trend. From late 2009 to early 2010, a short duration El Niño with stronger than average downwelling-favorable winds was observed. The El Niño was quickly followed by increased offshore transport with La Niña conditions in the summer of 2010 and increased upwelling and productivity persisted through early 2012 from Baja through central California. Highest anomalies in the northern copepod biomass were observed in March 2011 through the beginning of 2012, coincident with strongly negative PDO values. The



upwelling season has started later since 2007 resulting in a shorter upwelling season, particularly in the northern CCLME (Figure E2). Over the past five years, all of the broad-scale indices have remained within one standard deviation of the long-term mean save the NPGO. The positive anomaly of the NPGO is not a steric trend but instead is a result of decadal variability, as similar peaks were observed in 1976-8 and 1998-2003.

Figure E1. Broad scale indices and status from 2008-2012 with anomaly and trend values. Anomalies are the difference between the mean of the past 5 years and the long-term mean while

trends are calculated over the past 5 years. A negative MEI for both summer and winter indicates slight La Niña conditions. The NOI in winter is positive and decreasing while summer is neutral and increasing indicating neutral ENSO conditions for the North Pacific. The NPGO is positive although decreasing, indicating a more productive CCLME. The negative PDO values indicate a cool, more productive phase of the CCLME.

We have seen increases in sea surface temperatures (SST) and coastal sea level across the CCLME over the past 50 years, although SSTs have been cooler in recent years due to broad scale atmospheric forcing. The trend of decreasing dissolved oxygen (DO) continues to suggest increased habitat compression for pelagic species and more severe hypoxic events on the shelf that can lead to physiological stress or large scale die-offs. Since 1983, the CCLME has been characterized by periods of delayed upwelling approximately every 10 years. Since 2007, the spring transition has occurred later at 45° N, the length of the upwelling season has decreased, and the total upwelled magnitude has remained below the mean but within one standard deviation. The CCLME continues to exhibit natural interannual and multi-decadal variability, while time series of temperatures, DO, and nutrients are consistent with projections from climate models driven by global warming scenarios.

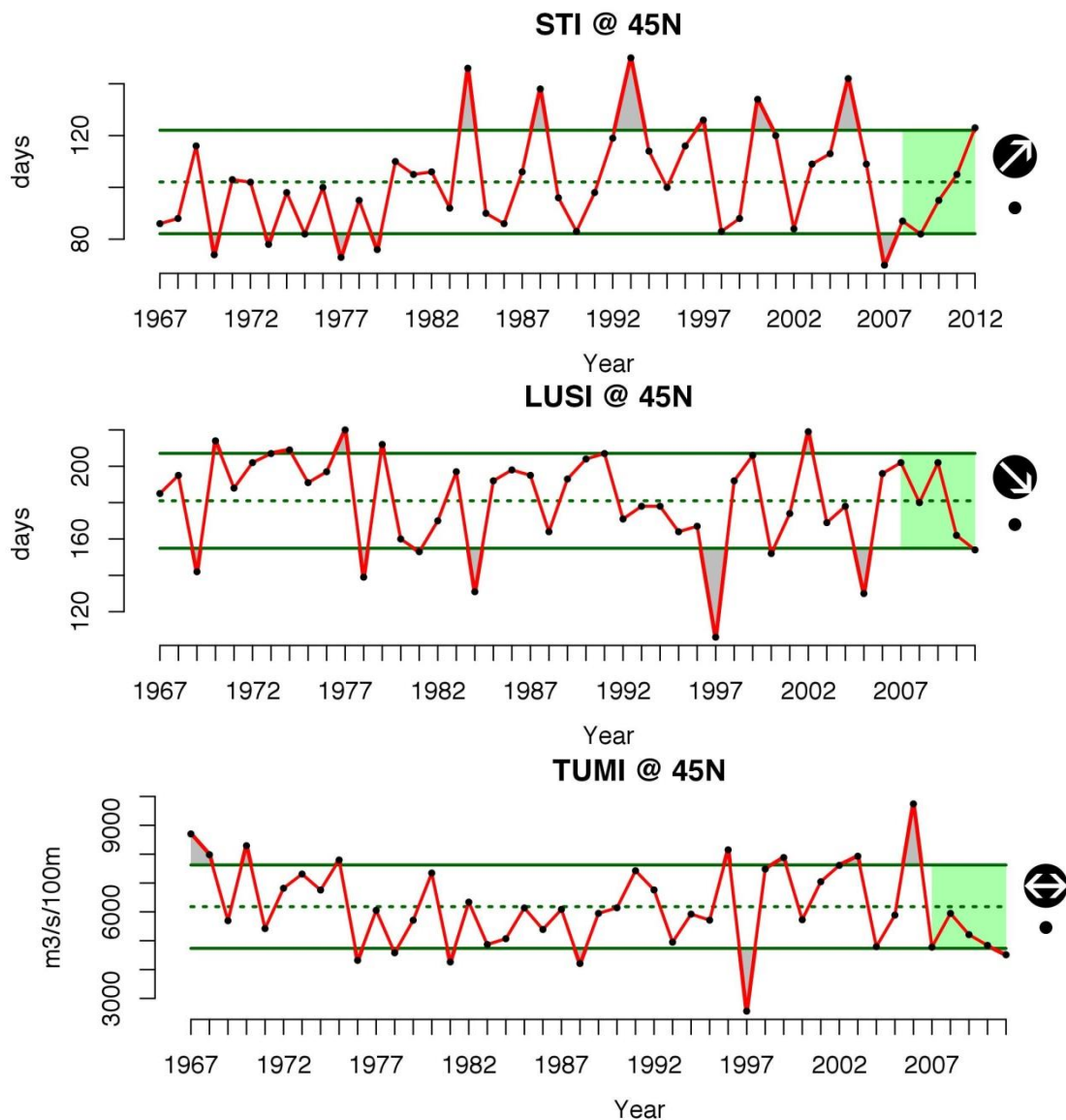


Figure E2. The a. Spring Transition (STI), b) Length of Upwelling Season (LUSI), and c) Total Upwelling Magnitude Indices (TUMI) at 45°N. The three indices together give a metric of the upwelling season in the CCLME.

DETAILED REPORT

The ultimate aim of the California Current Integrated Ecosystem Assessment (IEA) is to quantify the web of interactions that links drivers and pressures to ecosystem-based management (EBM) components and to forecast how changing environmental conditions and management actions affect the status of EBM components. In order to capture the breadth of pressures acting on the California Current Large Marine Ecosystem (CCLME), a lengthy list of drivers and pressures was developed and consolidated. Here we define drivers as forcing factors that result in pressures that in turn cause changes in the ecosystem. Anthropogenic pressures include factors such as coastal pollution, habitat loss and degradation, and fishing effort that can be mapped to specific drivers. For example, coastal development is a driver that results in increased coastal armoring and the loss of associated intertidal habitat. Indicators are chosen from time series data that best serve as proxies or measures of either drivers or pressures. For the purposes of the CCIEA, both natural and anthropogenic forcing drivers are considered. An example of the former is climate variability and the latter include human population size in the coastal zone and associated coastal development, and demand for seafood. In principle, human driving forces can be assessed and controlled. Natural environmental fluctuation cannot be controlled but must be incorporated and accounted for in management efforts.

The first step in developing indicators was to identify a suite of drivers and pressures that were most closely associated with impacts and changes to the different EBM components in the California Current IEA. We used several publications (Halpern et al. 2008, Sydeman and Elliott 2008, Halpern et al. 2009, Sydeman and Thompson 2010, Teck et al. 2010, Peterson et al. 2012) to develop an initial list of potential pressures on the CCLME and then supplemented this list with additional identified pressures. During reviews of the literature, we identified 32 primary groups of pressures on the CCLME, and these were categorized as “oceanographic and climatic” or “anthropogenic”. Indicators for each of these pressures were then evaluated using the indicator selection framework developed by Levin et al. (2011) and Kershner et al. (2011) and used in the previous version of NOAA’s Integrated Ecosystem Assessment for the California Current (Levin and Schwing 2011).

The second step was to develop time-series of data for each of the top indicators for each pressure. These time-series were used to determine the current status, short-term trends, and five-year anomalies for each pressure in the CCLME. Each section then ends with examples of the linkages between certain drivers and pressures and specific key EBM components of the CCLME.

Three broad pressures were described by Teck et al. (2010) as physical state variables: climate change and ocean acidification, climate change and sea level rise, climate change and changes in sea surface temperature. Climate change includes long-term natural variability, short-term, event driven variability, and an anthropogenic global warming signal, but separating these processes is difficult in the California Current. The CCLME is an eastern boundary current system largely driven by upwelling, so we have included a few additional pressures (9 total) presented below and summarized in Table OC1. It is important to mention that this document is not aimed to provide extensive reviews of the state of the California Current, but instead cataloguing and presenting existing information in a Driving Forces-Pressures-State-Impacts-Responses framework (e.g. Levin et al. 2009) as a foundation for forthcoming IEA sections. There are a number of high quality status reports for the California Current including the state of the California Current (Bjorkstedt et al. 2011), PICES Ecosystem status report (Sydeman and Bograd 2010), ocean ecosystem indicators (<http://www.nwfsc.noaa.gov/research/divisions/fed/oeip/a-ecinhome.cfm>), and PaCOOS Quarterly Update of Climatic and Ecological Conditions (<http://www.pacoos.org/QuarterlyClimaticEcol.htm>) among others.

We have used long term running means of the whole dataset, and highlight deviations from the mean and trends over the past 5 years for conformity across IEA figures, however many of the state variables fluctuate at decadal to multi-decadal scales. The 5 year window is used to show short term trends and anomalies in the environmental indicator. On each figure, the dotted line represents the long-term mean of the time series with the green lines representing 1 standard deviation above and below. The arrows represent positive (↗), negative (↘) or lack of (↔) trend over the past 5 years while a +, -, or • indicate that the mean of the past 5 years is greater than, less than, or within 1 standard deviation from the long-term mean respectively.

There is a close mechanistic and correlative link between coastal upwelling and ecosystem productivity on seasonal, annual, and interannual scales (Chavez et al. 2003). Also, upwelling in the central-northern CCLME occurs in two distinct seasonal modes (winter and summer), with certain biological processes being more sensitive to one or the other (Black et al. 2011, Thompson et al. 2012). Thus in this section we present indicators when there is monthly data as winter and summer means. Summer means were calculated from June 1st - August 31st and winter means were calculated from December 1st - March 31st. Indicator selection followed the IEA framework and identified datasets with the most relevance to the pressure, and had the longest and most complete time series. Indicator evaluation, data indices and sources are summarized in Table OC1.

IMPLICATIONS OF CLIMATE DRIVERS FOR COASTAL AND MARINE SPATIAL PLANNING

There are regional differences within the CCLME in climate forcing (Mendelssohn et al. 2003) and ecosystem response (García-Reyes and Largier 2012). Therefore, an assessment of the southern California Current region may vary from that for the northern California Current. When considering an overall IEA for the CCLME, it may prove most useful to evaluate each ecoregion/subecosystem separately initially. But in no single region are all the physical and especially biological attributes available for comprehensive analyses. Therefore, to understand ecosystem form, function, and control, we must combine information between regions with the goal for a uniform CCIEA. The IEA is spatially and temporally targeted for specific management foci; thus IEA evaluations will be scenario driven as a function of the management strategies being evaluated. When possible, we have examined three locations in the California current using cruise data such as CALCOFI (California Cooperative Oceanic and Fisheries Investigations) and the Newport line, buoy data (National Buoy Data Center), and satellite products.

The northern CCLME is dominated by strong seasonal variability in winds, temperature, upwelling, and plankton production (Huyer 1983). In addition to weak, delayed, or otherwise ineffectual upwelling, warm-water conditions in this region could result from either onshore transport of offshore subtropical water or northward transport of subtropical coastal waters (King et al. 2011). Low copepod species richness and high abundance of northern boreal copepods is associated with cold, subarctic water masses transported to the northern CCLME from the Gulf of Alaska (Peterson and Schwing 2003, Hooff and Peterson 2006, Peterson 2009, Bi et al. 2011, Keister et al. 2011). Therefore, copepod community composition may be used as an indicator of this physical oceanographic process.

Preliminary evidence suggests covariation between ecoregions. As an example, when fatty, subarctic northern boreal copepods are present in the northern CCLME during cool-water conditions, the productivity of the planktivorous Cassin's auklet in the central subregion increases. Conversely, when the less fatty subtropical copepods dominate the system in warm-water years (i.e., a higher southern copepod anomaly index), Cassin's auklet breeding success is reduced (Sydeman et al. 2011). Because patterns in northern copepods affect central bird species, it is important to perform analyses across boundaries and ecoregions.

As noted previously, there are regional differences in oceanography and biology. Moreover, within each region, there are differences in habitats that may be related to bathymetry and geology. Understanding the relationships between topography, oceanography, species distributions, and interactions will promote better management of CCLME resources spatially as well as temporally. The relationships between bottom topography and ecosystem productivity are not well known, but so-called benthic-pelagic coupling is likely to be an important driver for top predators. Identification and assessment of predictable locations of high species diversity and increased trophic interactions can serve as an important science basis for coastal and marine spatial planning and a common currency to assess trade-offs across sectoral uses of CCLME regions.

CHANGE IN SEA LEVEL

BACKGROUND

Sea level rise from climate change is expected to accelerate in the next century. The International Panel on Climate Change (IPCC) estimates that the global average sea level will rise further between 0.6 and 2 feet (0.18 to 0.59 meters) in the next century (IPCC 2007) as a result of natural processes and anthropogenic global warming. At its simplest, sea level rise is due to the thermal expansion of seawater (Domingues et al. 2008) and increased freshwater inputs from melting polar and glacier ice from the continents (Radić and Hock 2011). To best estimate the rate of sea level rise vertical movements of the land such as post-glacial rebound need to be considered to get an adequate rate (Douglas 1991). Multiple time scales are associated with sea level rise, on multidecadal timescales steric changes in the density field are often attributed to climate variability, while seasonal to interannual time scales variations are due to atmospheric and oceanic effects that can result in geostrophic readjustments.

EVALUATION AND SELECTION OF INDICATORS

Records of sea level rise must be multiple decades in length to distinguish changes over naturally occurring low-frequency signals that derive from atmospheric and oceanic forcing (Parker 1991). Three tidal gauge locations along the CCS achieve the criteria of being exceptionally long in length thus good indicators of change in sea level. They are: San Diego, CA (1906-present), San Francisco, CA (1897-present), and South Beach, OR (1967-present). Combining coastal tide gauges with satellite altimetry (Saraceno et al. 2008) can provide a more direct measure of stratification and circulation however time series are limited by satellite altimetry availability.

STATUS AND TRENDS

Coastal sea level is used as a proxy for nearshore surface current strength and direction. In the winter, sea levels are high due to the poleward flowing counter current (Davidson Current). With the onset of upwelling winds in the spring, sea levels lower and the current is directed equatorward; the equatorward flow is dominant in the spring and summer (King et al. 2011). Since 1950, there has been an increasing trend particularly until 1977 with more numerous and extreme positive anomalies (Figures OC1 – OC3). Over the past five summers, the San Diego station values have been greater than one standard deviation from the mean although there were no significant short-term trends throughout the California Current. Coastal sea level trends have been somewhat muted since 1980 due to wind changes and PDO masking any upper-ocean temperature steric effect (Bromirski et al. 2012).

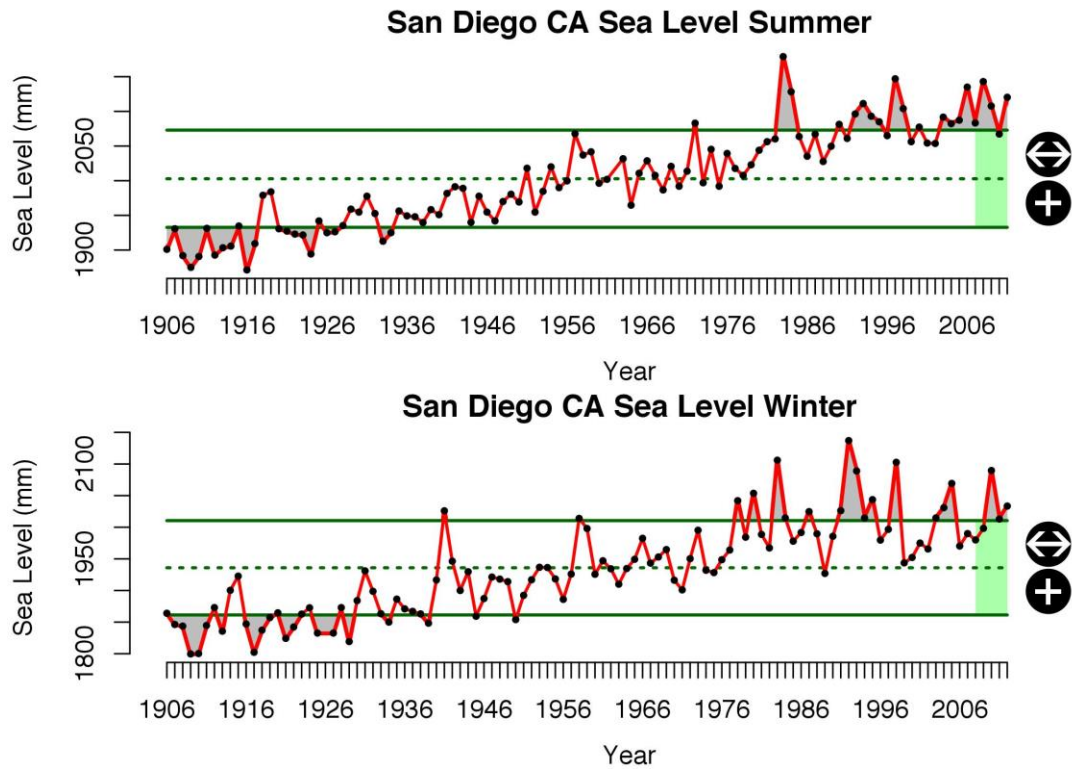


Figure OC1. Coastal sea level heights from 1906-2012 for both a. summer and b. winter. San Diego coastal sea level was chosen to illustrate patterns in the southern portion of the CCLME.

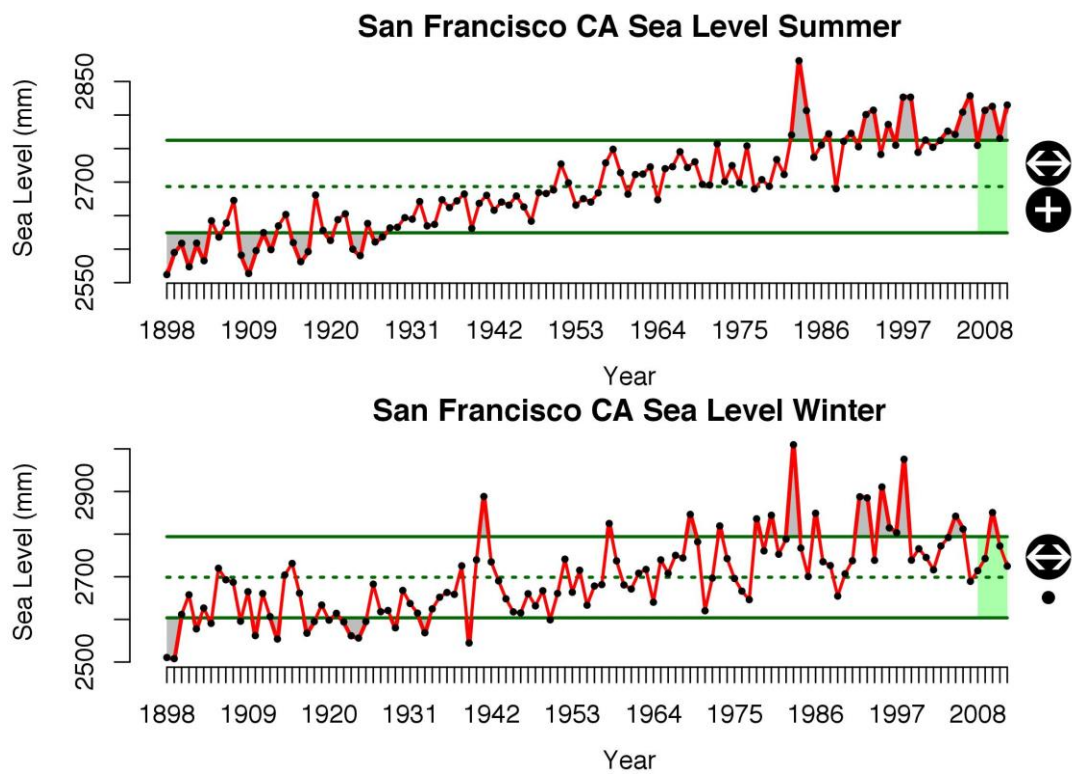


Figure OC2. Coastal sea level heights from 1898-2012 for both a. summer and b. winter. San Francisco coastal sea level was chosen to illustrate patterns in the central portion of the CCLME.

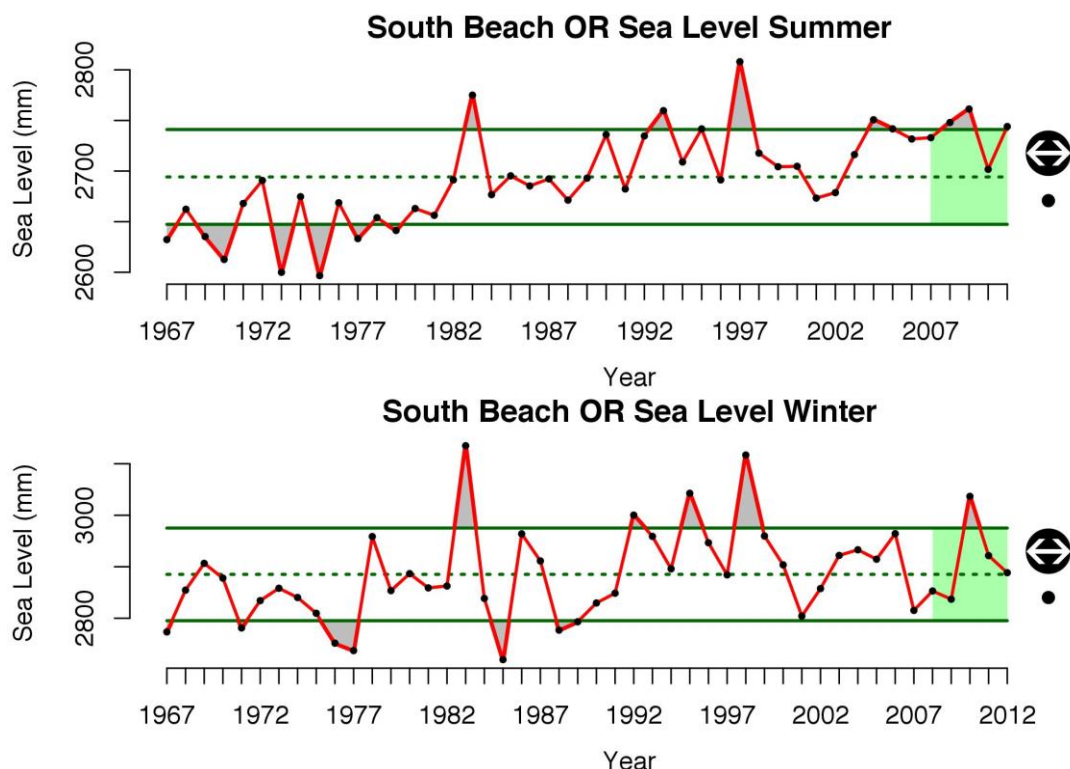


Figure OC3. Coastal sea level heights from 1967-2012 for both a. summer and b. winter. South Beach, Oregon coastal sea level was chosen to illustrate patterns in the northern portion of the CCLME.

CHANGE IN SEA SURFACE TEMPERATURE

BACKGROUND

Temperatures in the California current vary at multiple time scales: seasonally due in large part to upwelling, inter-annually due to broad scale forcing, and at the broadest scales due to anthropogenic climate change. Upwelling timing and strength greatly influences the California ecosystem through productivity and temperature changes (see section below), and many species in the CCLME are thermally limited directly (Song et al. 2012) or indirectly through trophic interactions (Wells et al. 2008). ENSO events and climatic forcing has the greatest influence on interannual temperatures resulting in changes in species composition and biodiversity in the CCLME. At the broadest scales, temperatures in the world's oceans are predicted to warm up to 6 degrees Celsius by 2100 (IPCC 2007). The effects of ocean warming on marine ecosystems are being examined more in recent years, and multiple studies have observed or predicted range shifts in marine over the next century (Hazen et al. 2012, Sunday et al. 2012), spatial changes in productivity and diversity (Rijnsdorp et al. 2009), and changes in timing of migration for oceanic and riverine fish (Spence and Hall 2010). Long term warming in the California current may be buffered by upwelling, but changes in source waters and stratification may limit any buffering effect.

EVALUATION AND SELECTION OF INDICATORS

There are numerous indicators of sea surface temperature at various spatial and temporal scales in the CCS. The Pacific Decadal Oscillation (PDO) index is used to show low frequency changes in sea surface temperature (SST) over the north Pacific (Mantua et al. 1997). When the PDO is positive, SST within the CCLME (especially the northern region) is warmer. The PDO does an inadequate job of describing SST variability in the coastal zone. The Multivariate ENSO index (MEI) represents patterns in six main observed variables over the tropical Pacific, to identify status of the El Niño southern oscillation, but the impact of ENSO on the CCLME varies. The Northern Oscillation Index (NOI) indexes the interannual changes of atmospheric forcing relevant to the CCLME, still a broad index. Thus, coastal zone water temperature change indicators are chosen via SST measured by NDBC buoys. SST winter/summer means are taken from three NDBC buoys in the CCLME. The three buoys are located in the California Bight, Central California and Oregon.

STATUS AND TRENDS

SST

Cold upwelled water often results in high productivity but nutrient content depends upon remotely forced state of the ocean, which can be indicated by large-scale climate indices (NPGO, PDO, MEI, and NOI). Negative NPGO, positive PDO, and positive MEI would act in concert to create an extremely warm, low-productivity regime in the CCLME. According to many long-term data sets from the open ocean, SSTs have increased by 0.5°C to 1.0°C over the past 50 years (IPCC 2007, Levitus et al. 2009). SST from three NOAA National Data Buoy Center (NDBC) buoys showed highs in 1983 and 1998 corresponding with increased MEI values (Figures OC4 – OC6). Most SST values in the past 5 years were lower than the 20-year mean average at all stations and for both seasons. The exceptions were warmer than average SSTs in winter 2010 at all three stations which had high SSTs due to the short duration El Niño, and the summer of 2011 in central California as well.

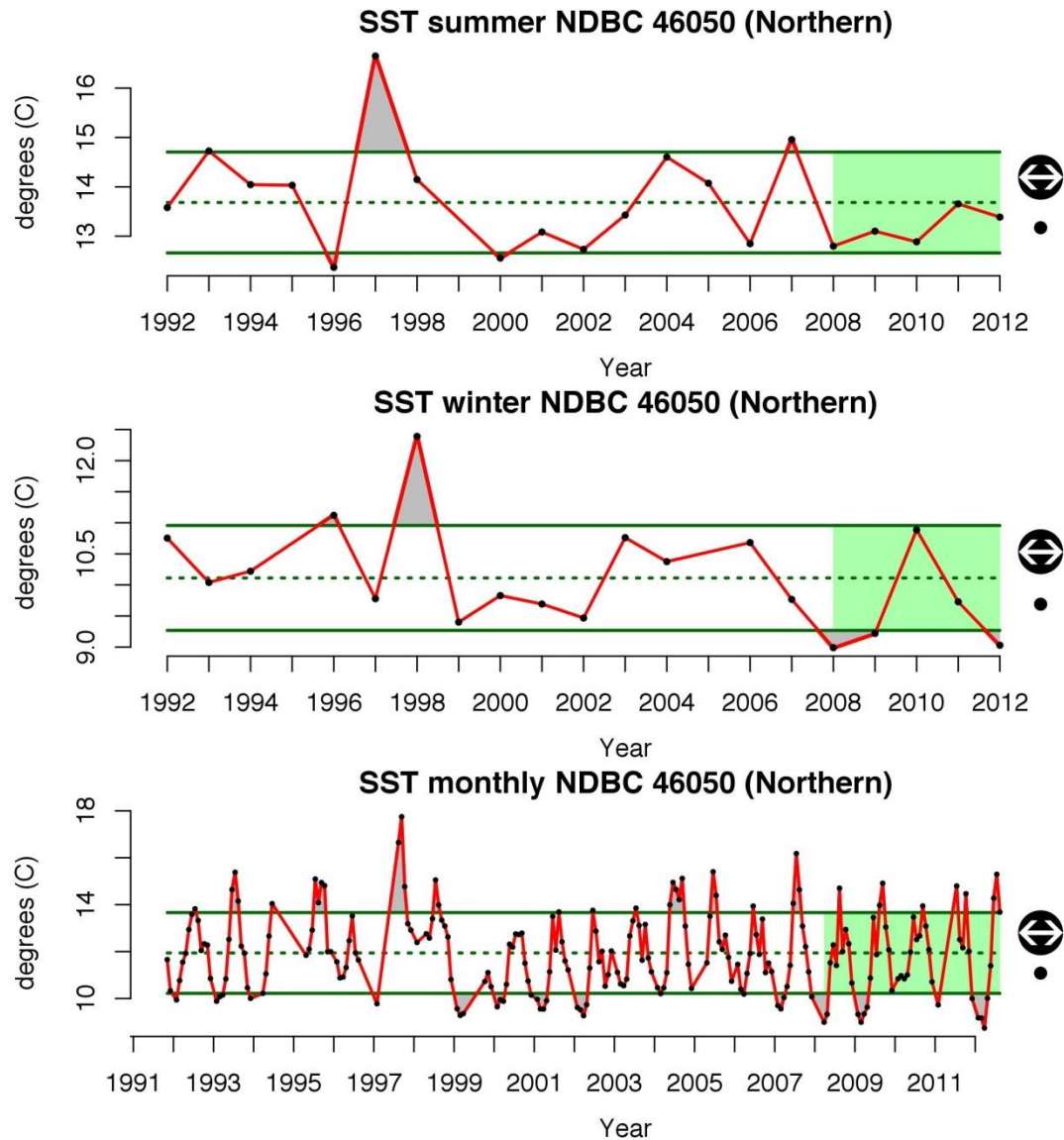


Figure OC4. Sea surface temperature (SST) buoy data from early 1990 -2012 during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. Buoy 46050 was chosen to illustrate patterns in the northern portion of the CCLME.

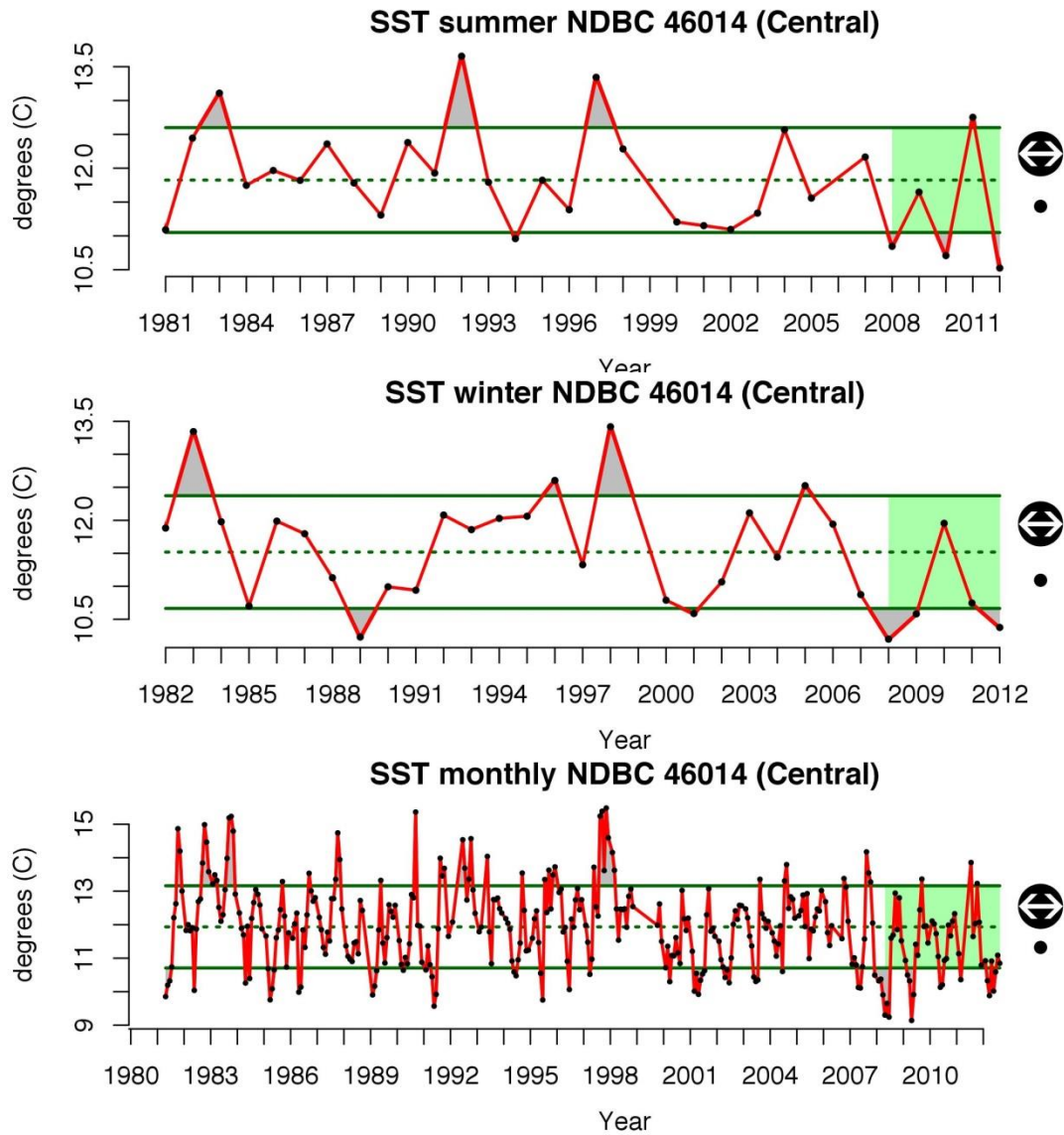


Figure OC5. Sea surface temperature (SST) buoy data from early 1990 -2012 and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. Buoy 46014 was chosen to illustrate patterns in the central portion of the CCLME.

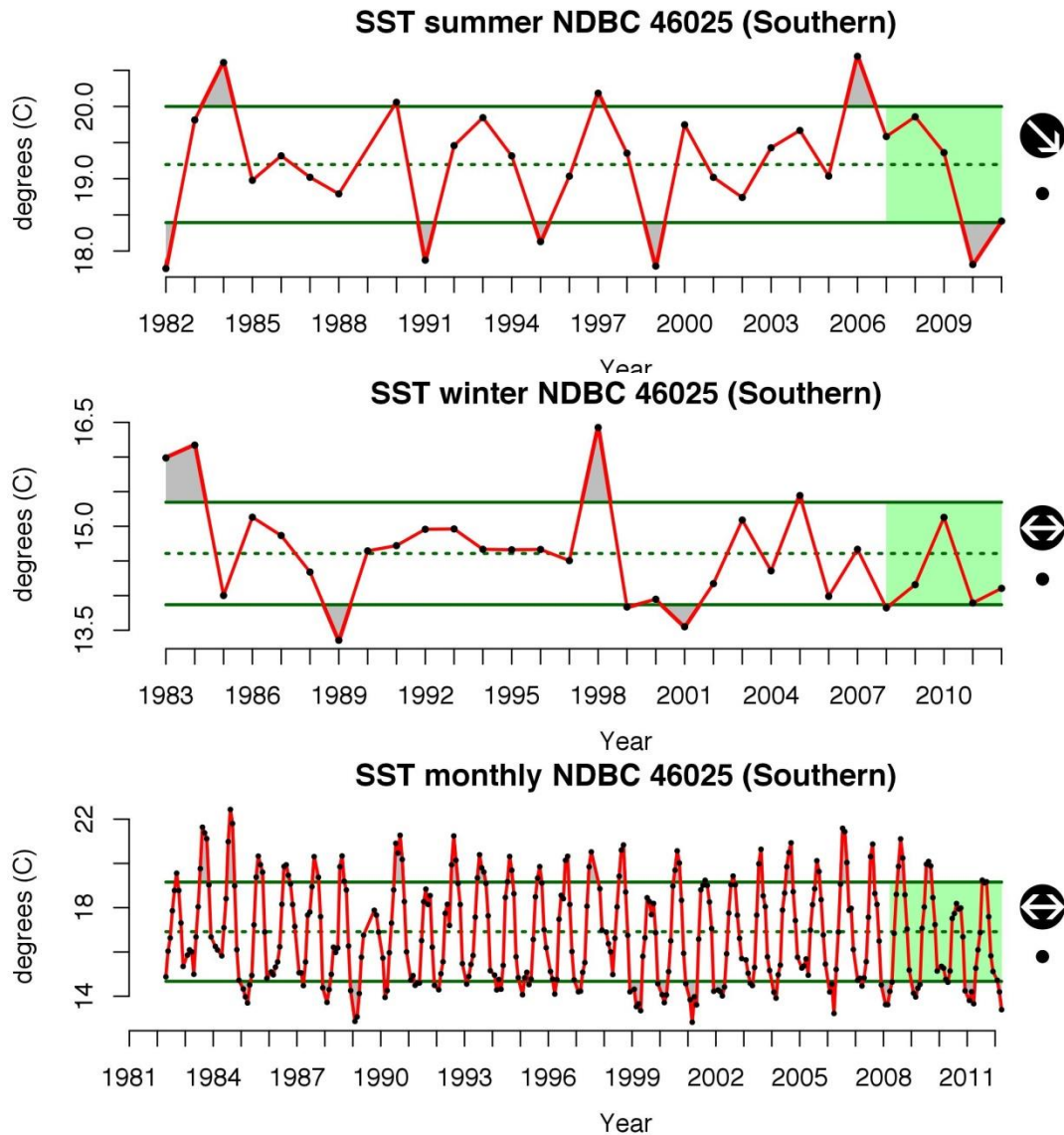


Figure OC6. Sea surface temperature (SST) buoy data from early 1990 -2012 and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. Buoy 46025 was chosen to illustrate patterns in the southern portion of the CCLME.

PDO

PDO is a low frequency signal in North Pacific sea surface temperatures that affects biological productivity in the Northeast Pacific. (Mantua et al. 1997) Cold (negative values of the PDO) eras are associated with enhanced productivity in the CCLME and vice versa (King et al. 2011). The PDO index has been largely in a positive (i.e., warm California Current and Northeast Pacific) state since late 1977, resulting in warmer waters along the coast of the CCLME with a negative phase since with 1998 with occasional warm episodes from El Niños (Figure OC7). Over the past 5 years, the winter index declined from 2005 to 2009 with a sharp increase in 2010. Both summer and winter PDO values have continued to decline in 2011 and 2012.

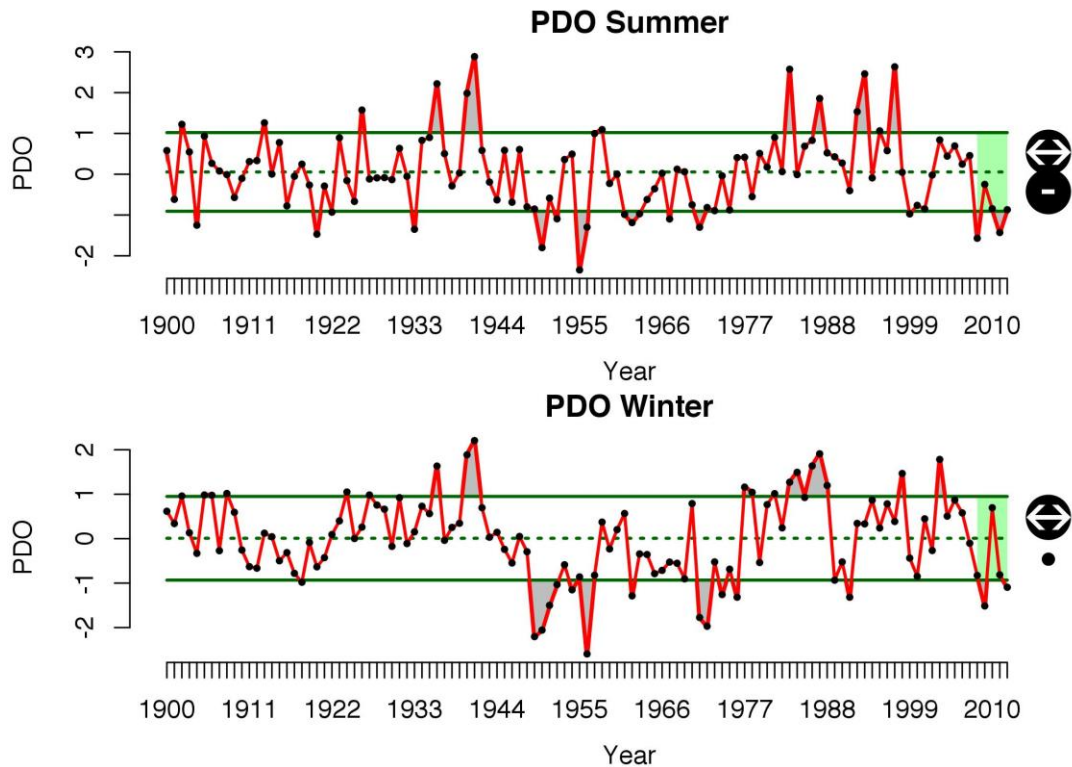


Figure OC7. Pacific Decadal Oscillation (PDO) index values from 1900 -2012 during both a. summer and b. winter.

NOI

NOI is the sea level pressure difference between the climatological mean position of the North Pacific High and Darwin, Australia (Schwing et al. 2002). NOI describes the strength of atmospheric forcing between the equatorial Pacific and the North Pacific, particularly in terms with ENSO. Positive values of the NOI are related to a more intense North Pacific High and stronger north winds over the CCS, and stronger northeasterly trade winds in the subtropics resulting in cooler waters. NOI was largely positive from 1950 to 1977, but switched to more negative values until 1998 (Figure OC8). In the winter, NOI values were positive from 2006 to 2009 with a drop and overall negative trend in 2010 representing the brief El Niño event. In summer 2010, NOI values became strongly positive which should result in increased coastal upwelling in the California Current, and have since returned to near neutral values.

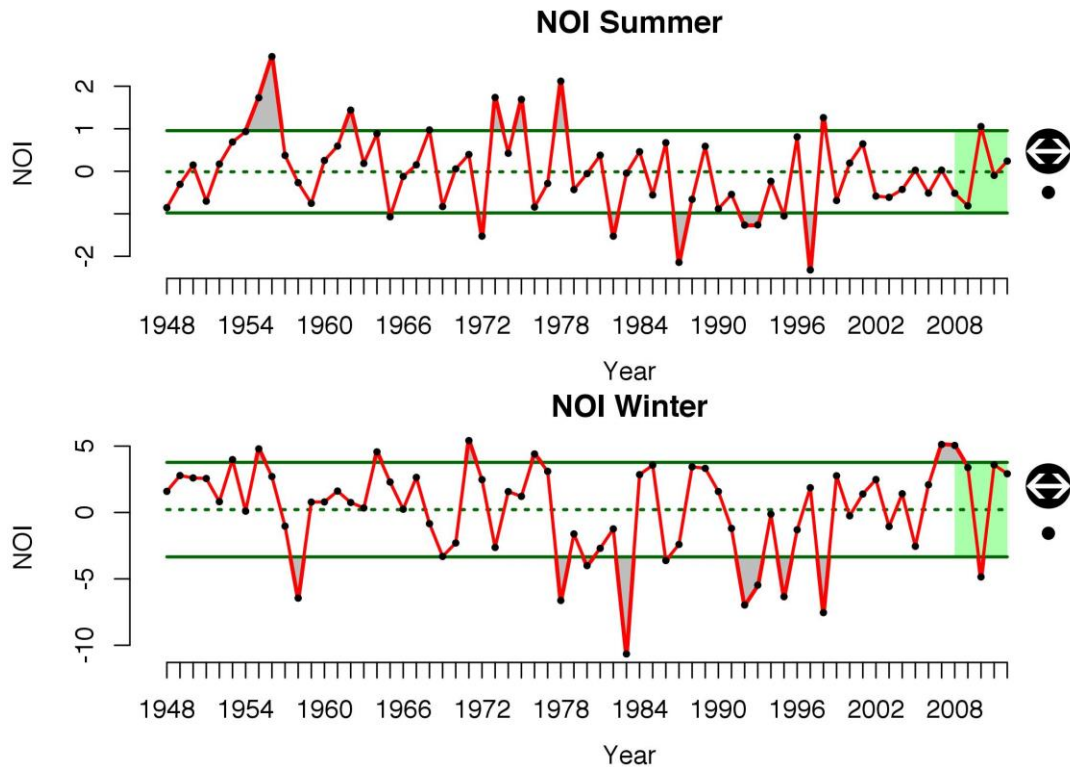


Figure OC8. Northern Oscillation Index (NOI) values from 1948 -2012 during both a. summer and b. winter.

MEI

See Timing and Frequency of El Niño events

WATER COLUMN STRUCTURE

BACKGROUND

The water column of the ocean is stratified into layers of water masses of different properties, such as nutrients, oxygen, temperature, salinity and density. For the water column structure attribute, we will focus on stratification due to density differences. Layers of more dense water lie below less dense water and the boundary between the layers acts as a barrier to mixing. Mixing between layers is easier when the density difference between the layers is small. The formation of the layers is due to several different geophysical processes, which act on different spatial and temporal timescales. For example, any physical processes that can change the water density, such as wind mixing, fresh water inputs and atmospheric thermal heating/cooling, will affect water column stratification. The effectiveness of upwelling winds in the CCLME can be reduced if the water column is highly stratified thus limiting the injection of nutrients from deep water into the surface euphotic zone (Palacios et al. 2004, Behrenfeld et al. 2006). In this report we will characterize the water column structure by quantifying information of the upper surface water mass. Two variables of interest are the mixed layer depth (pycnocline depth) and the strength of the stratification (the gradient between the density of the surface layer and the adjacent lower layer). Buoyancy frequency, or Brunt-Väisälä frequency, can be used to determine how stratified the water column is. The buoyancy frequency is proportional to vertical changes in density; the largest buoyancy frequency will mark the

pycnocline (Pond and Pickard 1983). Upwelling can be constrained if the pycnocline depth is deep and the strength of stratification is strong.

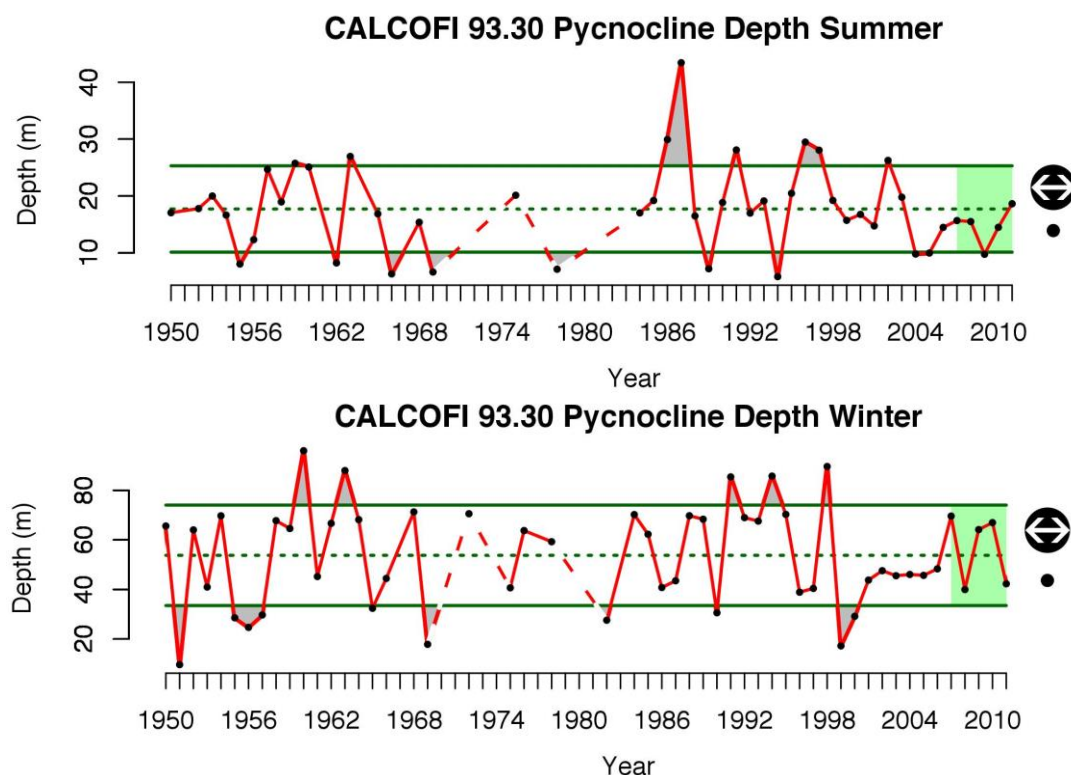
EVALUATION AND SELECTION OF INDICATORS

Long time series of strength and depth have been compiled at three stations for this report, but broader spatial coverage would be ideal for future IEAs. Additionally, El Niño events result in a deepening of the pycnocline due to the propagation of Kelvin waves and atmospheric teleconnections that favor an intensified Aleutian Low pressure cell that is also displaced to the south and east of its climatological position. This pressure pattern favors intense south-southwesterly winds that cause intense coastal onshore Ekman transports and downwelling, and reduced heat fluxes from the ocean to the atmosphere, that together lead to a warmer than average upper ocean over the continental shelf. Thus the MEI can give information on pycnocline depth on interannual time scales.

STATUS AND TRENDS

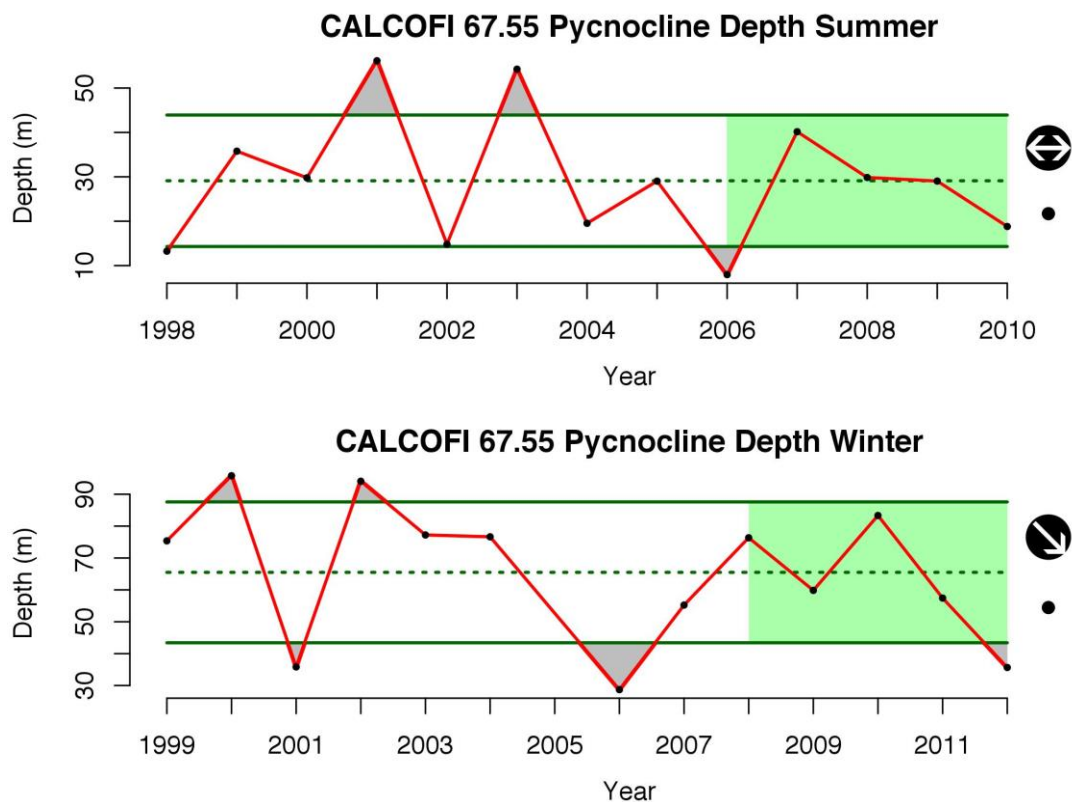
PYCNOCLINE DEPTH

Pycnocline depth, the greatest change in density in the vertical water column, represents the separation between warmer nutrient poor surface waters and cooler nutrient rich deep waters. The shallower the pycnocline, the more nutrients are available to the photic zone. Over the past five years (2007-2011), pycnocline depth has decreased steadily at station 67.55 in central California for both summer and winter (Figure OC9). In southern California (station 93.30), thermocline depth is highly variable with no clear trend over the past 5 years (Figure OC10). In the northern California current (station NH25), the pycnocline has become deeper in the winter but has no clear trend in the summer (Figure OC11).



Figure

OC9. Pycnocline depth data from 1950 -2012 and during both a. summer and b. winter from station 93.30, chosen to illustrate patterns in the southern portion of the CCLME. Dashed lines show data gaps of greater than 2 years.



Figure

OC10. Pycnocline depth data from 1998 -2012 and during both a. summer and b. winter from station 67.55, chosen to illustrate patterns in the central portion of the CCLME.

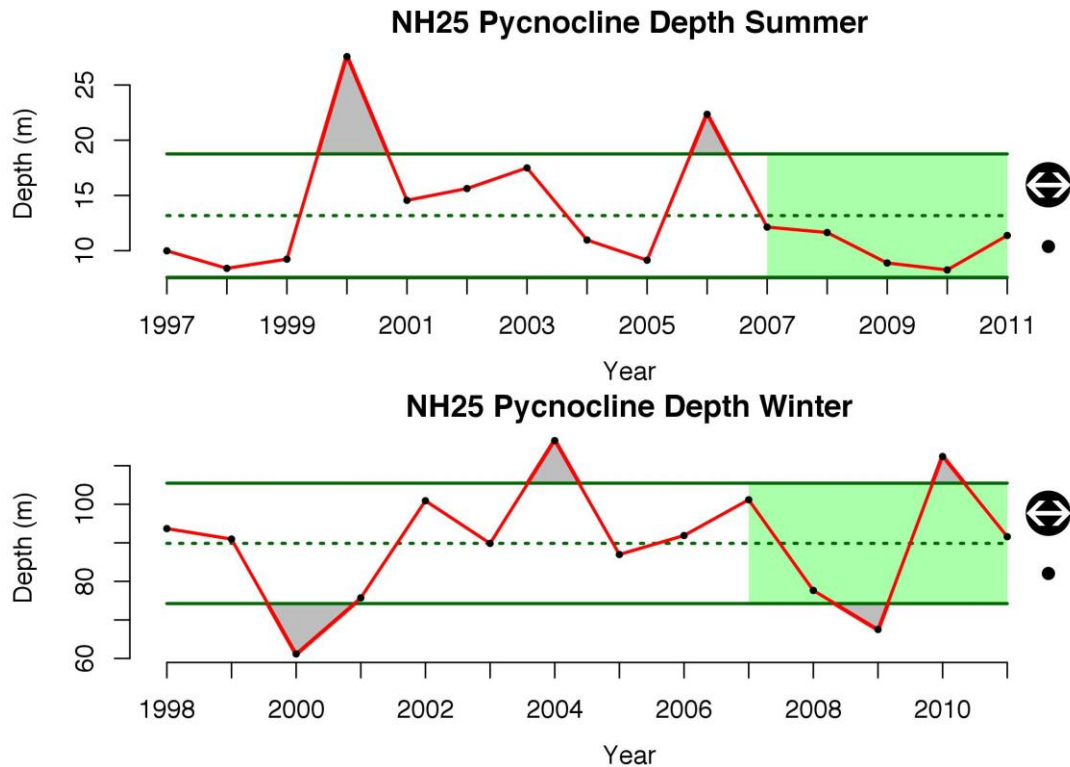


Figure OC11. Pycnocline depth data from 1998 -2012 and during both a. summer and b. winter from Newport line station NH25, chosen to illustrate patterns in the northern portion of the CCLME.

PYCNOCLINE STRENGTH

The BVF (Brunt-Väisälä frequency) value indicates the strength of density gradient in the vertical water column. The stronger the pycnocline, the less mixing of nutrients occurs across the pycnocline. Over the past five years (2007-2011), pycnocline strength has increased steadily at station 67.55 in central California for both summer and winter (Fig. OC12). In southern California (station 93.30), thermocline strength has been highly variable with no clear trend over the past 5 years (Fig. OC13). In the northern California current (station NH25), the pycnocline has strengthened (Fig. OC14).

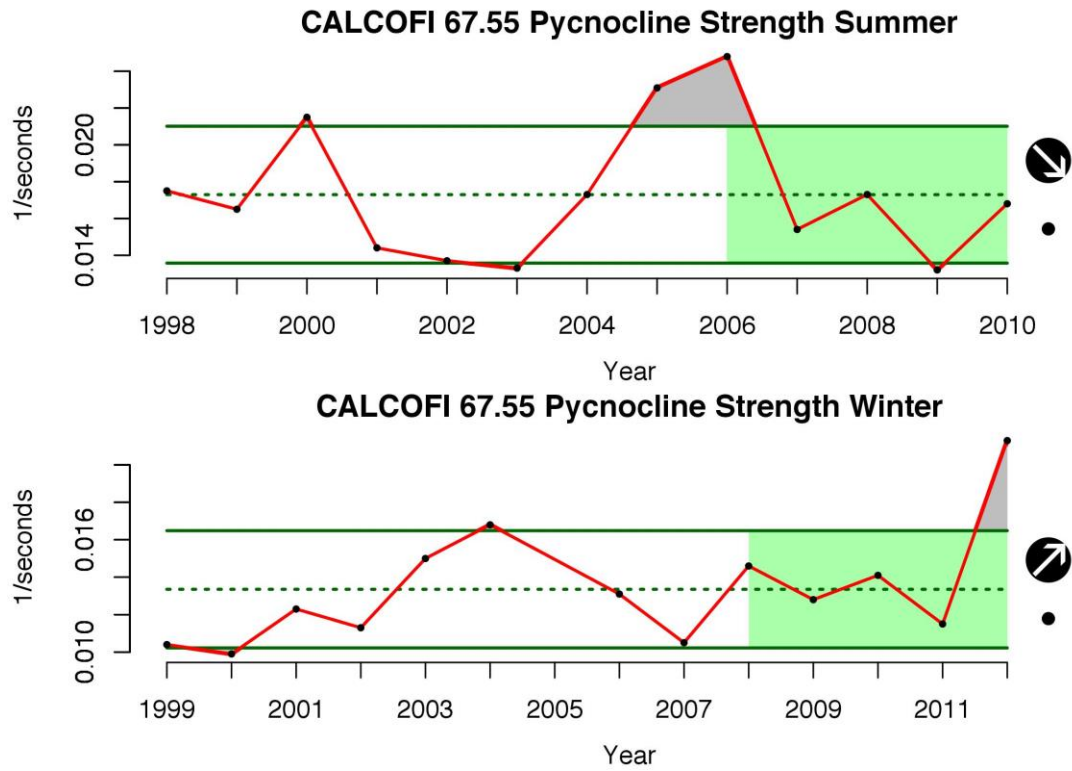
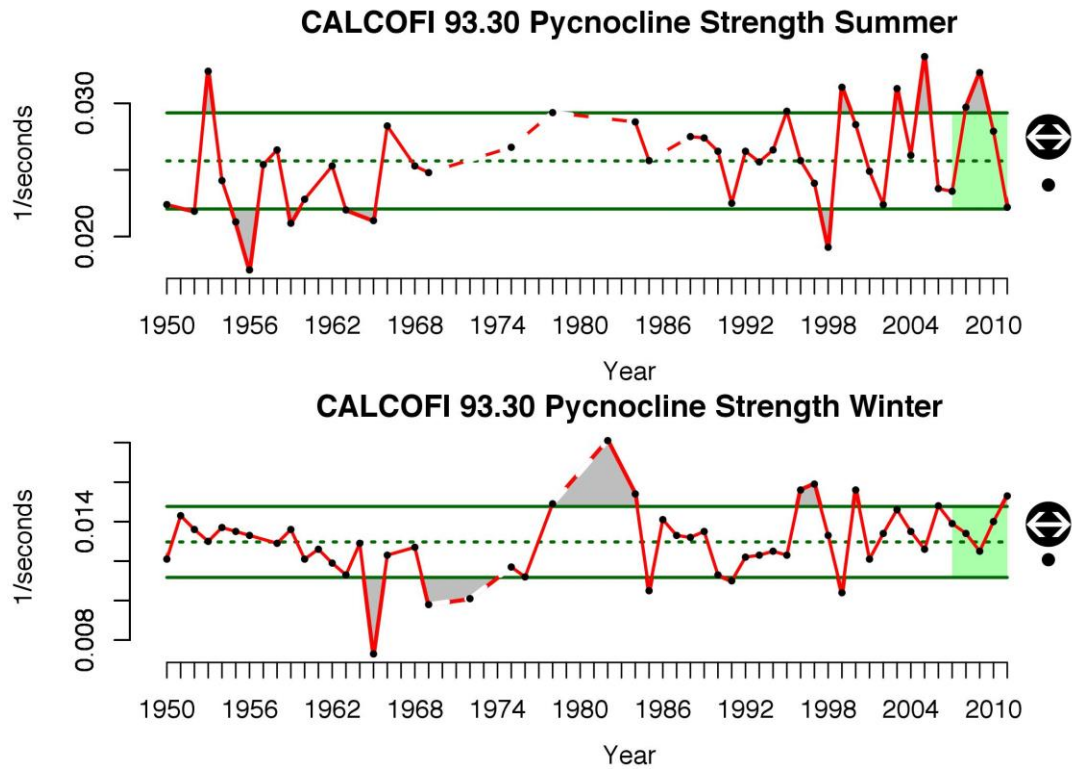


Figure OC12. Pycnocline strength data from 1998 -2012 and during both a. summer and b. winter from station 67.55, chosen to illustrate patterns in the central portion of the CCLME.



Figure

OC13. Pycnocline strength data from 1950 -2012 and during both a. summer and b. winter from station 93.30, chosen to illustrate patterns in the southern portion of the CCLME. Dashed lines identify data gaps of greater than 2 years.

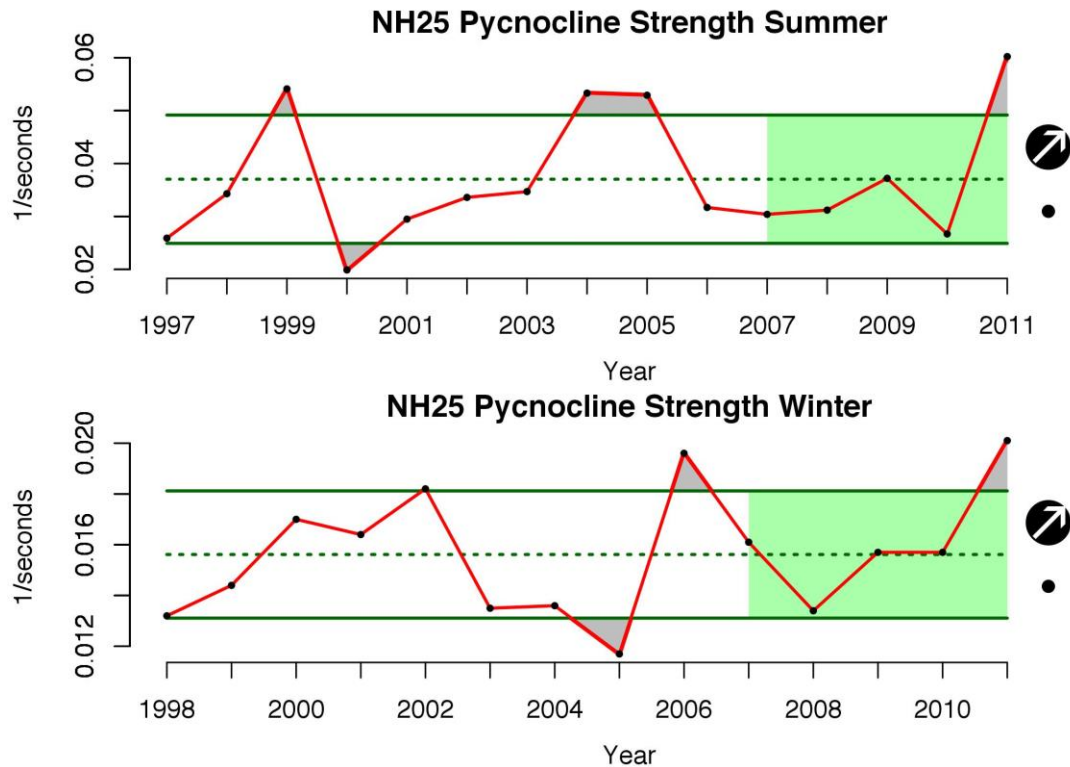


Figure OC14. Pycnocline strength data from 1998 -2012 and during both a. summer and b. winter from Newport line station NH25, chosen to illustrate patterns in the northern portion of the CCLME.

CHANGES IN CALIFORNIA CURRENT TRANSPORT AND MESOSCALE ACTIVITY

BACKGROUND

The major currents of the CCLME are the equatorward flowing California Current and coastal jet, the poleward flowing Undercurrent and Davidson Current, and the Southern California Eddy (Checkley and Barth 2009, King et al. 2011). These currents strengthen at particular times during the year due to local and remote forcing. Embedded in the slow flowing (<5 cm/s) California Current are mesoscale eddies, upwelling filaments and jets (Checkley and Barth 2009). The geostrophically balanced California Current is present throughout the year, and is surface intensified. In winter a broad northward flowing current, called the Davidson Current, forms when upwelling inducing winds diminish in strength. The source waters of the California Current and Undercurrent are different, with the California Current being fed by the low-salinity, high-oxygen and high-nitrate water from the North Pacific Current and the Undercurrent's source waters originating from the eastern tropical North Pacific, which are high-salinity, low-oxygen and low-nitrate. Changes in the volume transport of the California Current can result from changes to the North Pacific Current, which is affected by variations in the sea level height over the Northeast Pacific (Cummins and Freeland 2007). The North Pacific Gyre Oscillation (NPGO) index is linked to variations in the eastern and central regions of the North Pacific Gyre circulation and is significantly correlated with nutrients and chlorophyll a in the southern CCLME providing a rough index of California Current transport (Di Lorenzo et al. 2008). Eddies and fronts provide important habitat for top predators in the California Current through prey aggregation (Wells et al. 2008, Kappes et al. 2010). We have indexed mesoscale activity using remotely-

sensed measures of eddy kinetic energy (EKE) calculated from altimetry data (Strub and James 2000, Haney et al. 2001).

EVALUATION AND SELECTION OF INDICATORS

Winter and summer means of eddy kinetic energy from three locations in the CCLME are used as indicators of mesoscale activity in the CCLME (Strub and James 2000, Marchesiello et al. 2003). The three regions are the mean EKE over 6 degrees centered at latitudes 33, 39 and 45°N, with each region extending zonally from the shore to 300 km. Winter/summer means of the NPGO show low frequency variations of circulation in the CCLME. Positive (negative) values of the NPGO are linked to increased (decreased) upwelling, nitrate and chl-a, especially in the southern CCLME (Chenillat et al. 2012).

STATUS AND TRENDS

Eddy Kinetic Energy (EKE) is a measure of mesoscale activity calculated from the square of the zonal and meridional geostrophic flow. High EKE values indicate more mesoscale activity (front, eddies, jets) with much of the eastern Pacific having low EKE values ($<300 \text{ cm}^2/\text{s}^2$). EKE has not shown a long-term trend at any of the three locations (33°N, 39°N, and 45°N) in winter or summer (Figures OC15 – OC17). Since 2007, winter EKE has increased at 33°N and summer EKE has decreased at 39° and 45°N.

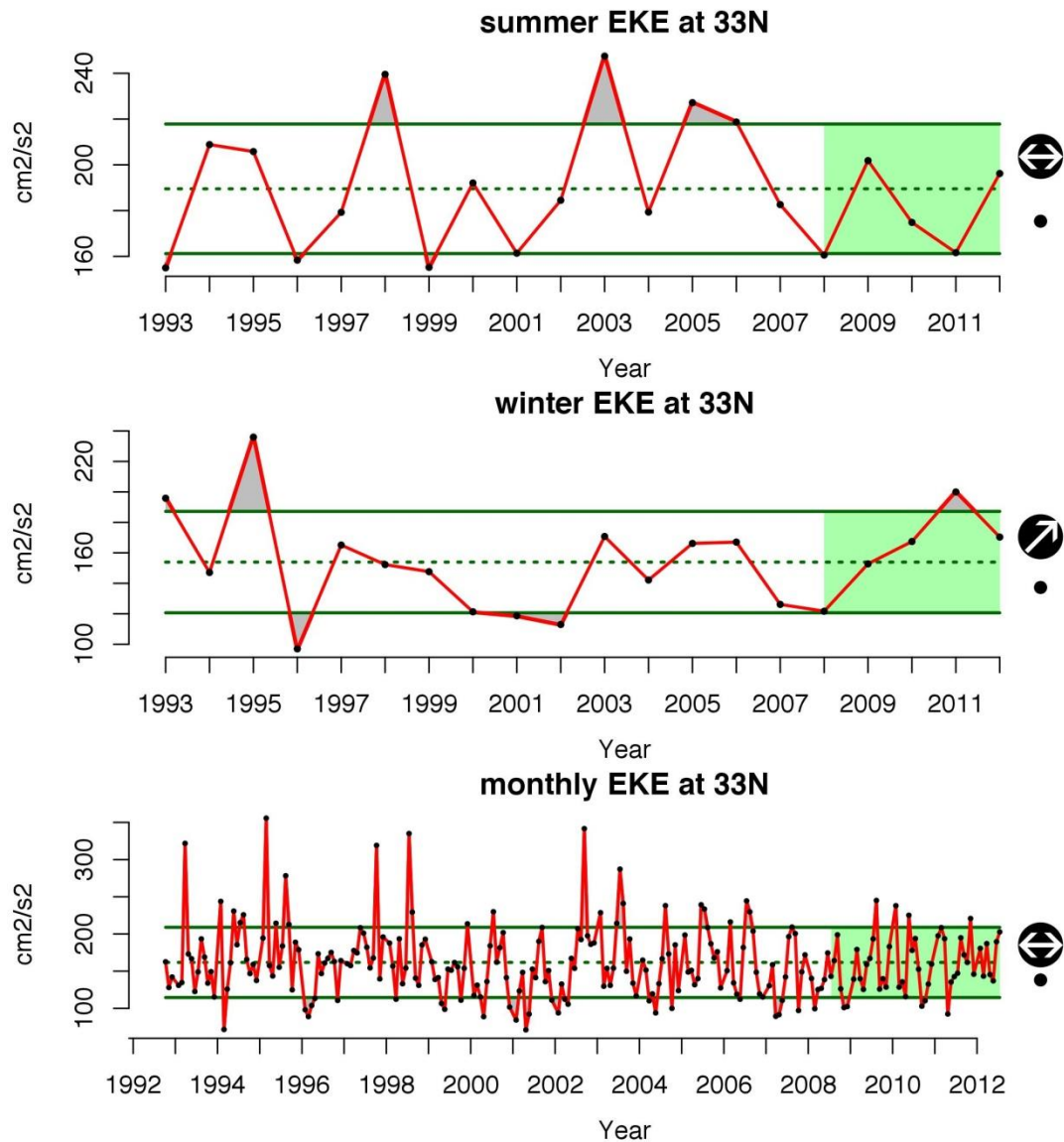


Figure OC15. Eddy Kinetic Energy satellite data from 1992 -2012 at 33°N and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. 33°N was chosen to illustrate patterns in the southern portion of the CCLME.

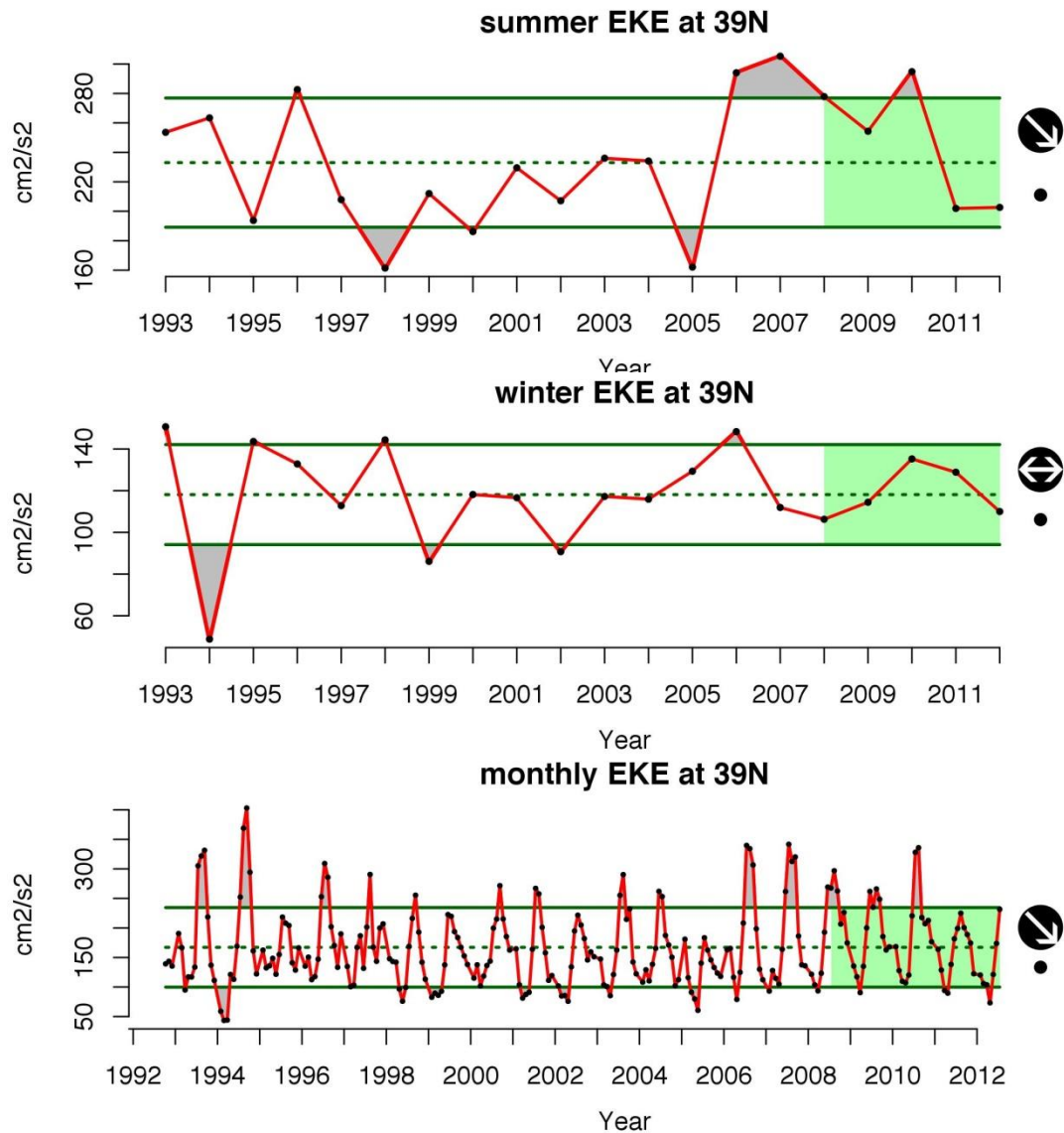


Figure OC16. Eddy Kinetic Energy satellite data from 1992 -2012 at 39°N and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. 39°N was chosen to illustrate patterns in the central portion of the CCLME.

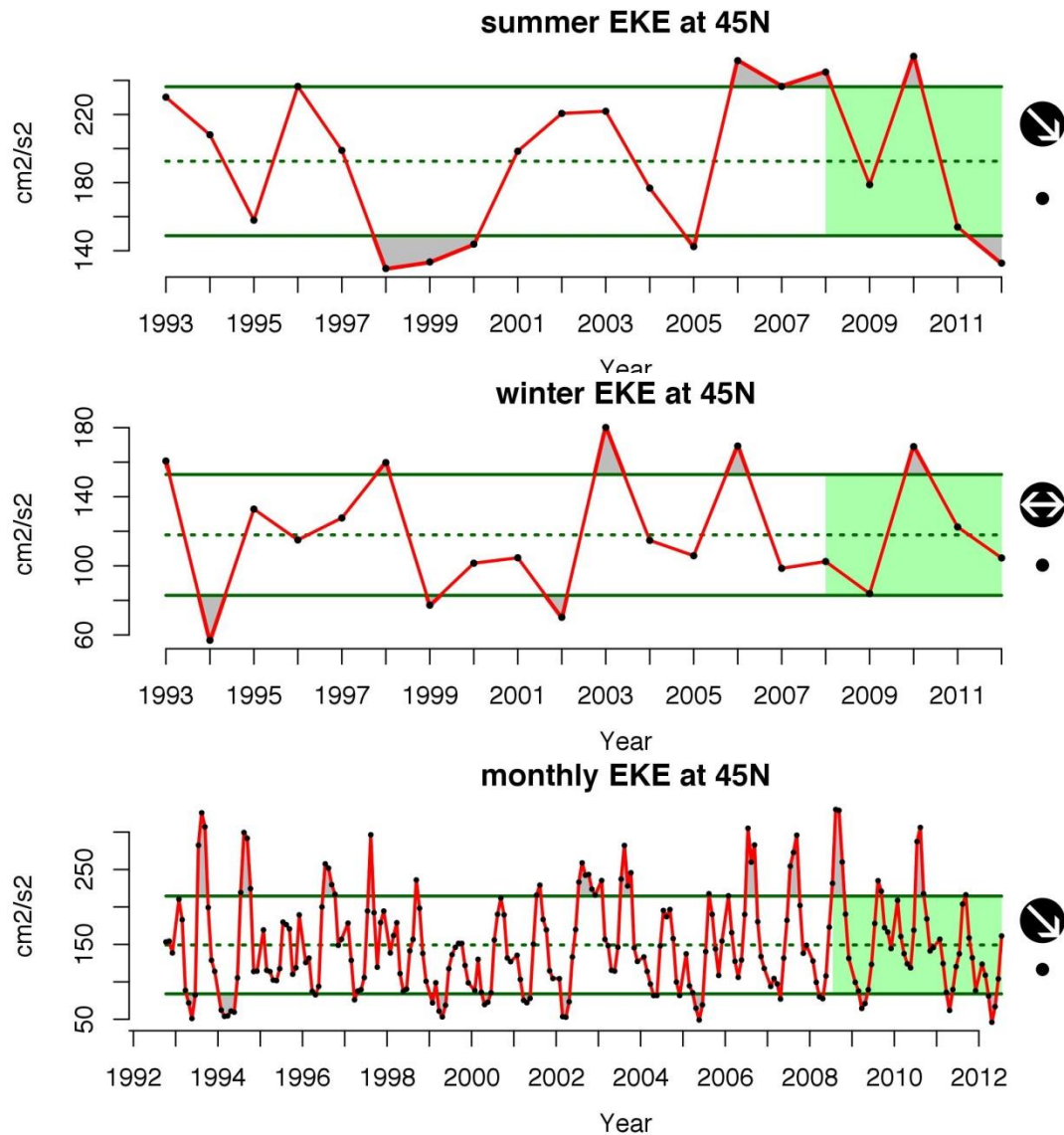


Figure OC17. Eddy Kinetic Energy satellite data from 1992 -2012 at 45°N and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. 45°N was chosen to illustrate patterns in the northern portion of the CCLME.

TIMING AND STRENGTH OF UPWELLING

BACKGROUND

Upwelling is critically important to productivity and ecosystem health in the CCLME (Huyer 1983). The strength and duration of upwelling in the CCLME is highly variable, and is forced by large-scale atmospheric pressure systems. More specifically, the pressure gradient between the oceanic North Pacific High and continental Low situated over the southwestern United States drives upwelling-favorable northerly winds. The interaction (friction and Coriolis force) of the northerly winds and the water surface moves water offshore in the surface layer, and this water is replaced by water upwelled from depths of greater than 50 - 100 m. The upwelled water is cooler, saltier and higher in nutrient concentrations than the surface water it

replaces. The onset and duration of the upwelling season varies latitudinally, starting earlier and lasting longer in the southern CCLME (Bograd et al. 2009).

Because of the close mechanistic and correlative link between coastal upwelling and ecosystem productivity on seasonal, annual, and interannual scales (Chavez et al. 2003), scientists have a strong need for operational products that quantify and forecast upwelling within marine ecosystems. However, it is extremely difficult to quantify upwelling directly, and measurements of coastal upwelling are scarce.

EVALUATION AND SELECTION OF INDICATORS

Timing and strength of upwelling were indexed using two sources: meridional winds from NDBC buoys and the Upwelling Index (UI) (Bakun 1975). We have included both to provide both the raw data and derived product often used for measuring upwelling in the CCLME given its importance to the ecosystem. The NOI can also serve as a broad-scale proxy for winds as positive values mean that winds from the north are typically more intense. The meridional winds from buoys are winter/summer means from three locations along the CCLME. Three derived products (STI, TUMI and LUSI) using the UI identify the timing and strength and duration of upwelling in the CCLME (Bograd et al. 2009). The spring transition index (STI) identifies the time when upwelling starts and varies with latitude in the CCLME. The units for STI are days and a year with a small STI value will have an earlier start to the onset of upwelling winds. The length of upwelling season index (LUSI) will provide information on the duration of upwelling during a particular year. The units for LUSI are days and a larger LUSI value indicates that the upwelling season for the given year is long. The total upwelling magnitude (TUMI) measures the ultimate amount of upwelling. There may be years of short but intense periods of upwelling, or longer but weaker upwelling seasons. Time series of STI and LUSI will be at three locations in the CCLME.

STATUS AND TRENDS

UI

The 2005 upwelling season was unusual in terms of its initiation, duration, and intensity. In 2005 upwelling was delayed or interrupted and SSTs were approximately 2-6°C warmer than normal (Barth et al. 2007). The situation in the southern ecoregion was different in both 2005 and 2006, as average upwelling and SST prevailed (Peterson et al. 2006). Other than a brief period of weaker than normal upwelling in the summer of 2008, west coast upwelling has been increasing since the late summer of 2006 (Figures OC18 – OC20). Wind patterns in early 2009 reflect anomalously strong high pressure over the Northeast Pacific and very high upwelling while early to mid 2010 appears to be a below average upwelling year at lat 35–45°N.

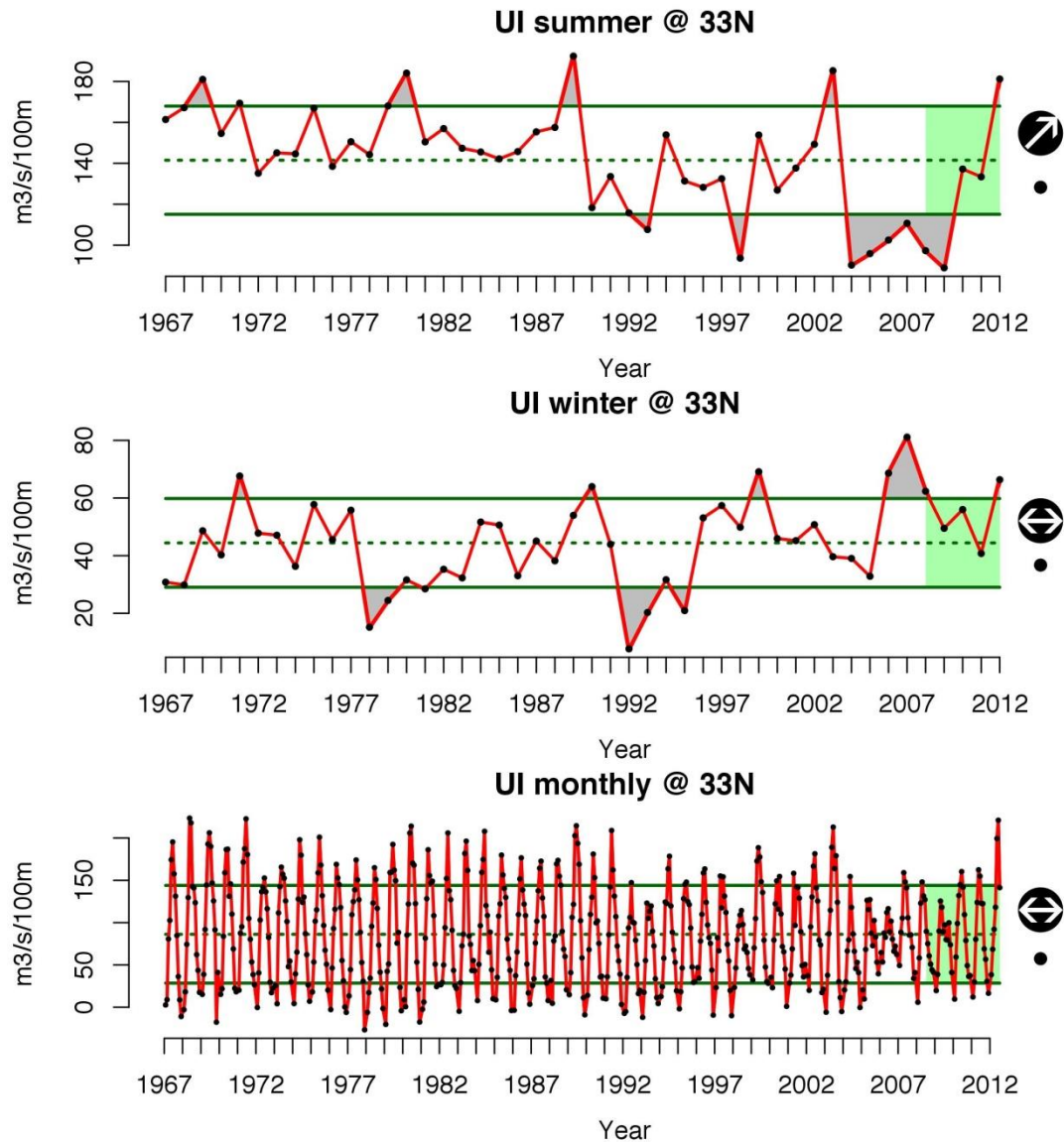


Figure OC18. The Upwelling Index calculated from 1967 -2012 at 33°N and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. 33°N was chosen to illustrate patterns in the southern portion of the CCLME.

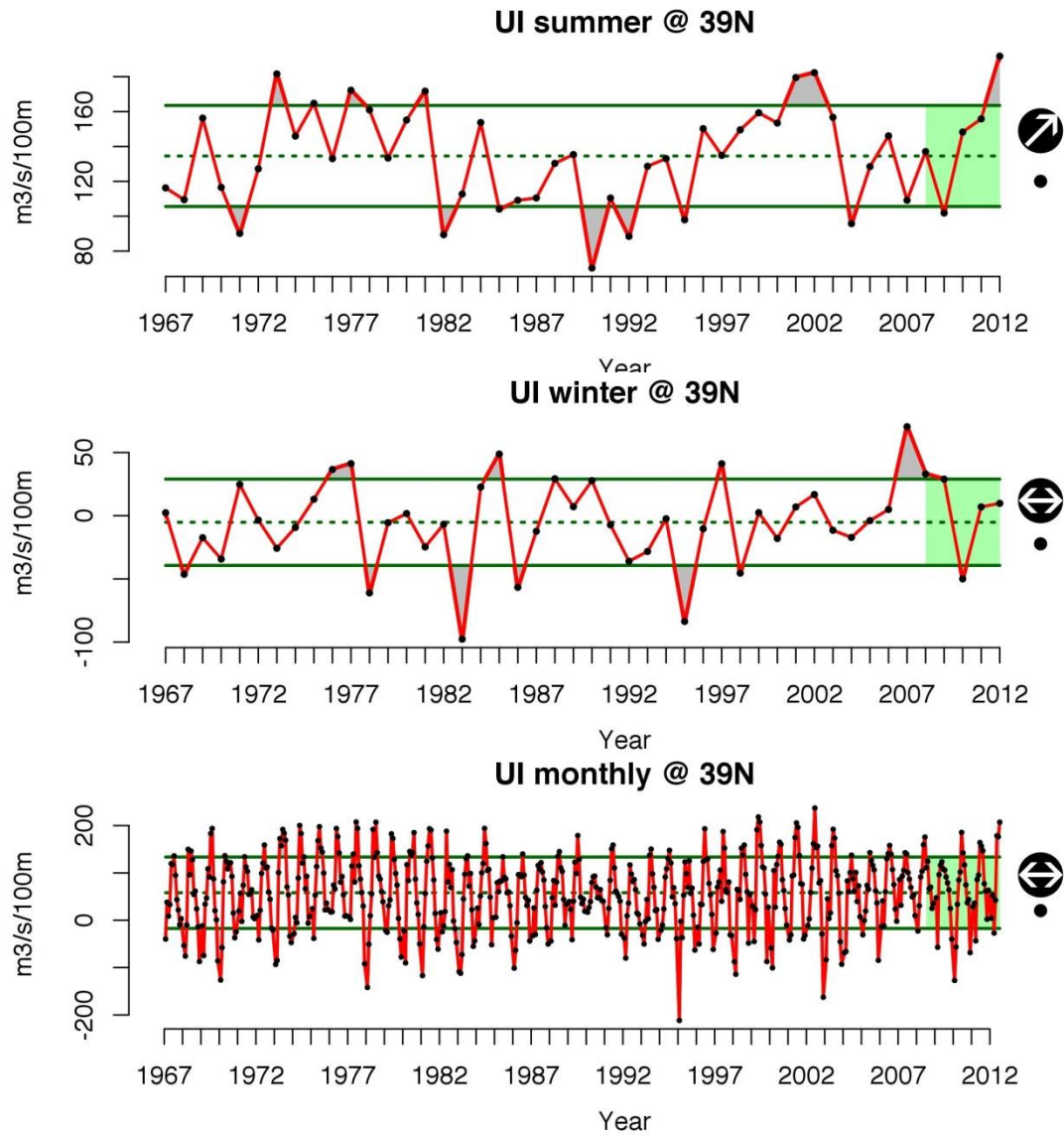
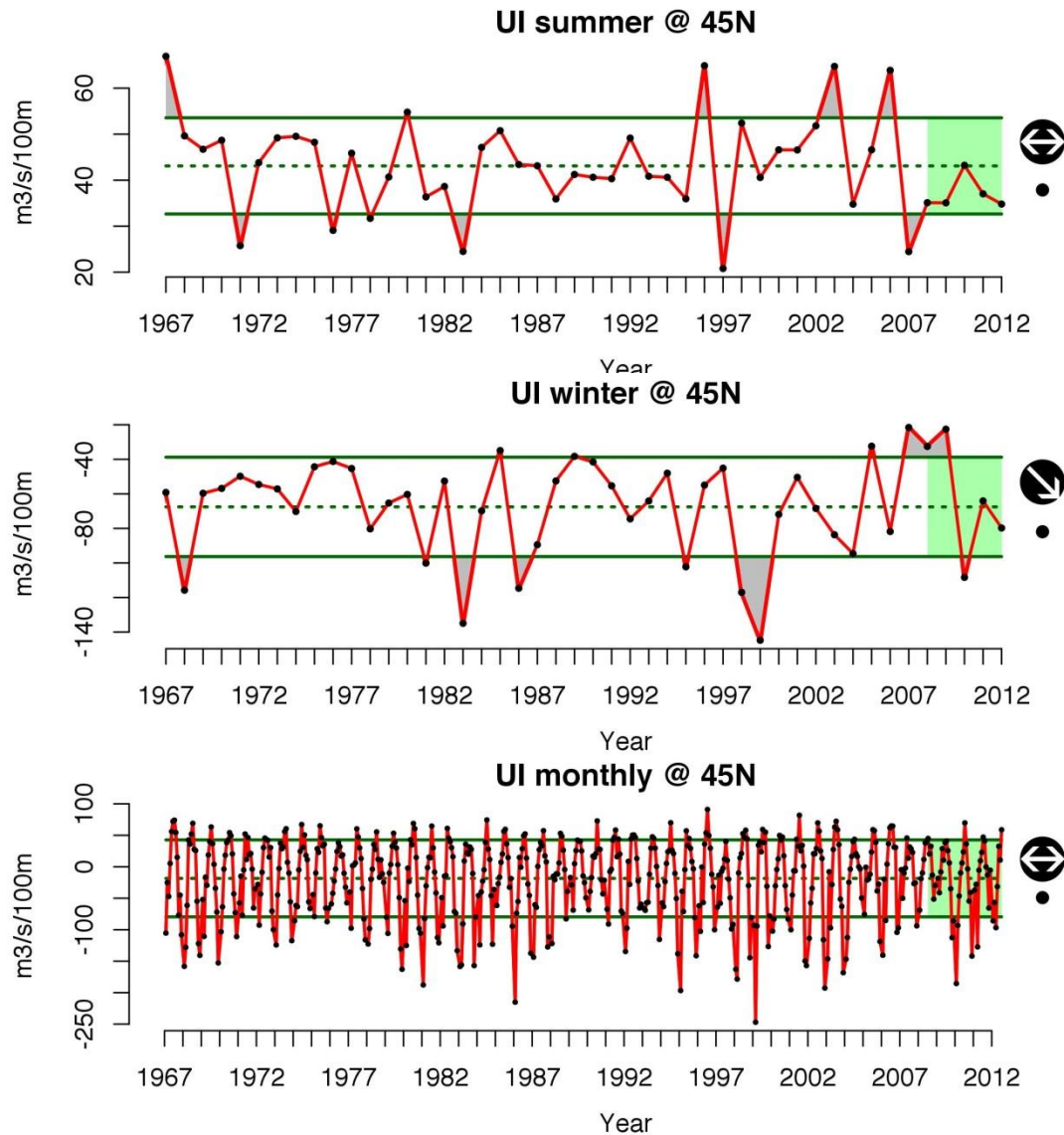


Figure OC19. The Upwelling Index calculated from 1967 -2012 at 39°N and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. 39°N was chosen to illustrate patterns in the central portion of the CCLME.



Figure

OC20. The Upwelling Index calculated from 1967 -2012 at 45°N and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. 45°N was chosen to illustrate patterns in the northern portion of the CCLME.

STI

The spring transition index (STI) indicates roughly the start of the upwelling season. It is defined by the date the CUI reaches its minimum value (Bograd et al. 2009). The STI fluctuates around 10 days past March 1st with a few extremely early or late years. In the early 90's and in 2005 anomalously late upwelling occurred with a severe effect on many biological time series. The past 5 years other than 2008 have had relatively average day of spring transition indicating the start of the upwelling season (Figure OC21). Given the UI often remains positive at 33°, we have excluded this time series from STI consideration.

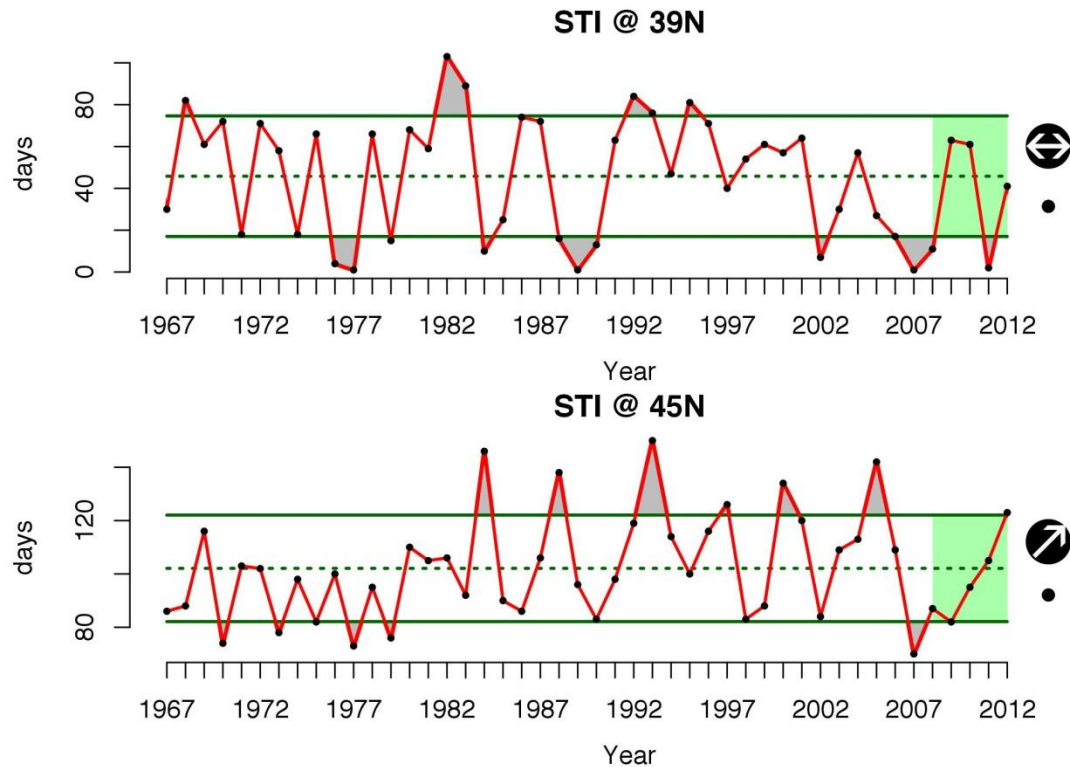


Figure OC21. The Spring Transition Index (STI) calculated yearly from 1967 -2012 at a. 39°N, and b. 45°N. 33° is not included because there is not a reliable downwelling phase each season.

LUSI

The length of the upwelling season (LUSI) is determined by the date of the STI until the date of the CUI maximum. This length of upwelling season indicates how long the upwelling favorable conditions persisted over the year. Over the past 5 years, LUSI showed a neutral trend at 39° while showing a declining trend at 45° (Figure OC22).

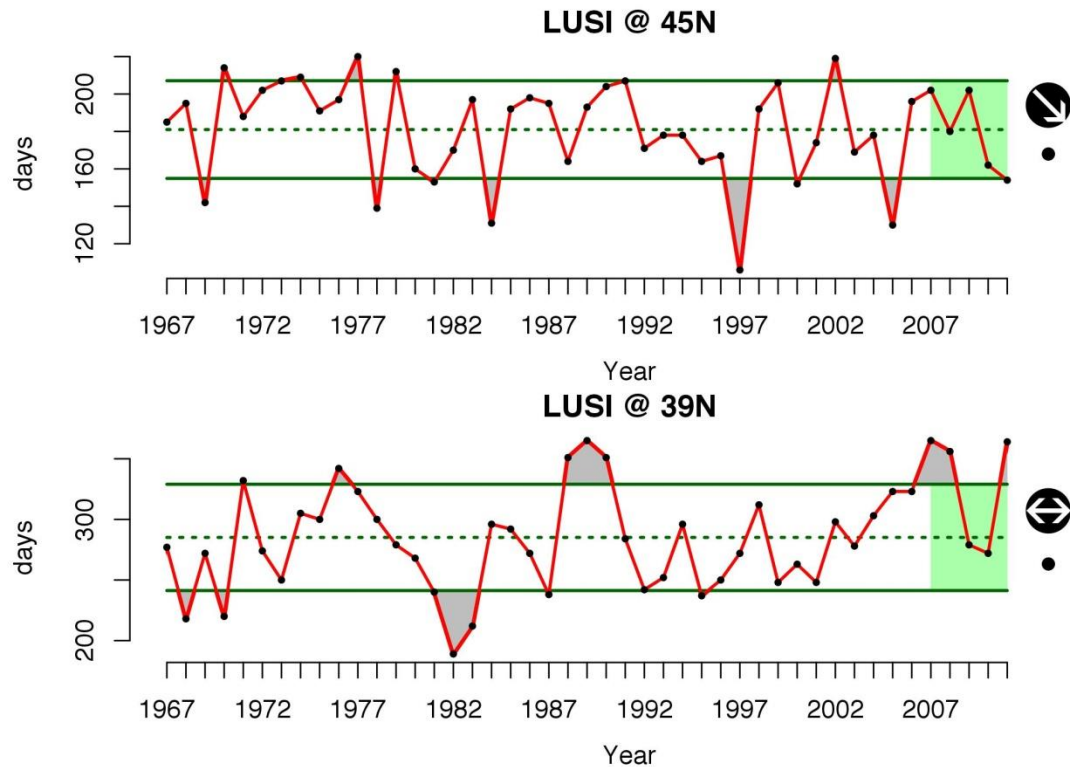


Figure OC22. The Length of the Upwelling Season Index (LUSI) calculated yearly from 1967 -2012 at a. 39°N and b. 45°N.

TUMI

The total upwelled magnitude index (TUMI) is the sum of the UI over the duration of the upwelling season (e.g. LUSI). This index represents the total amount of upwelled water as an indicator of total upwelled nutrient availability to the photic zone for the year. At the southernmost station, TUMI has been variable with minima in 1992-1993 and 2004-2005 although no clear trend since 2007 (Figure OC23). TUMI at 39° N shows a decadal pattern with high values in the 1970s, low values in the 1980s-1998 and high values since 1999 with the exception of 2003-2004. At 45° N, TUMI had a minima in 1997 and a maxima in 2006. Since 2006, values have been below the mean but not extremely so.

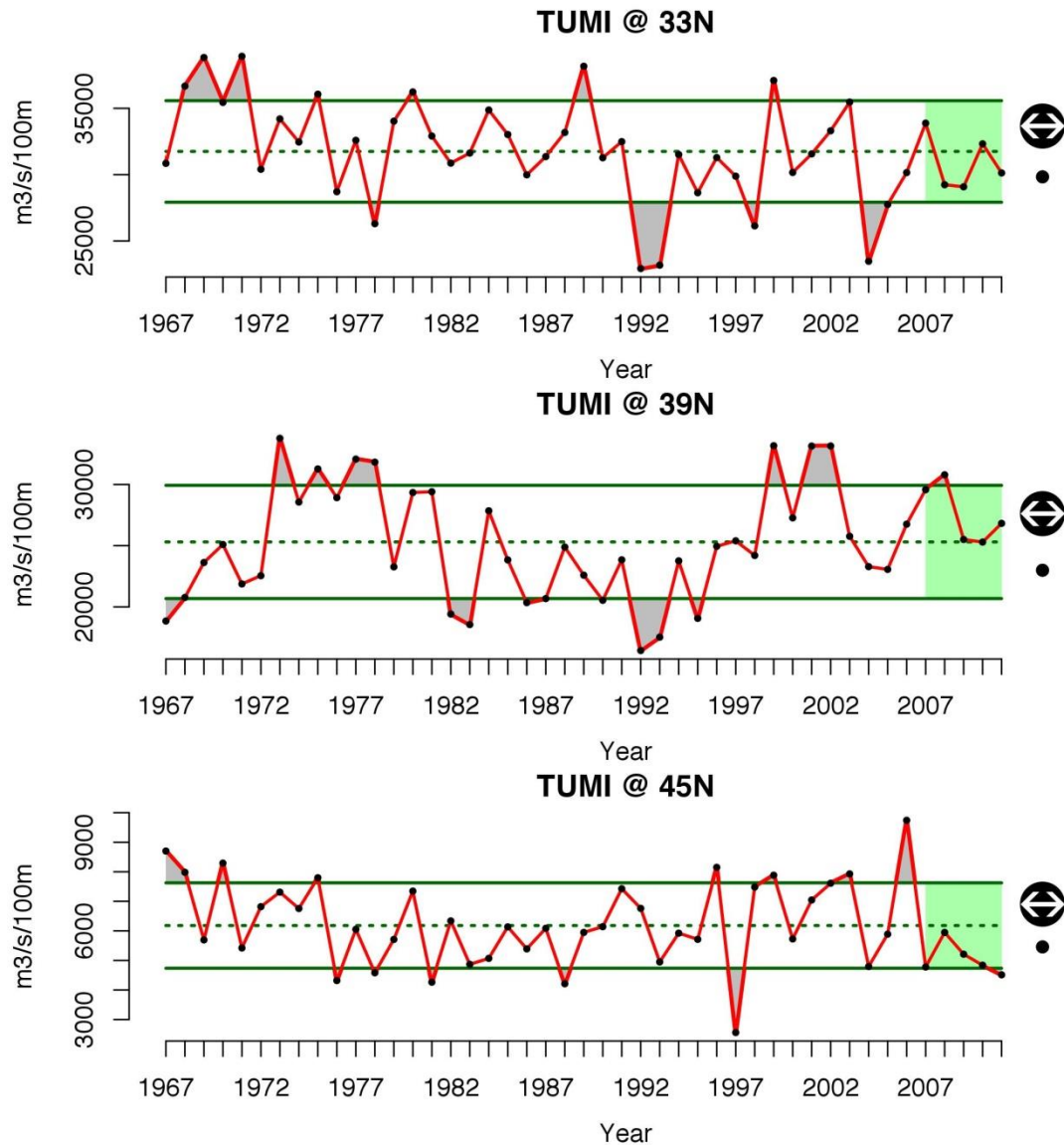


Figure OC23. The total upwelled magnitude index (TUMI) calculated yearly from 1967 -2012 at a. 33°N, b. 39°N, and c. 45°N.

WINDS

Northerly winds in the CCLME result in offshore transport and upwelling of cold, nutrient rich water into the photic zone. In the winter, meridional (north/south) winds were consistently northward in 1998 and 2010, indicative of downwelling favorable conditions (positive MEI and NOI; Figures OC24 – OC26). In winter 2006, winds were also indicative of downwelling although less extreme than 1998 and 2010. In summer 2006 and winter 2007, there were highly favorable upwelling winds at the northern buoys (A and B). In summer 2010, upwelling favorable winds dominated all three buoys, although they declined at 39° N in 2011.

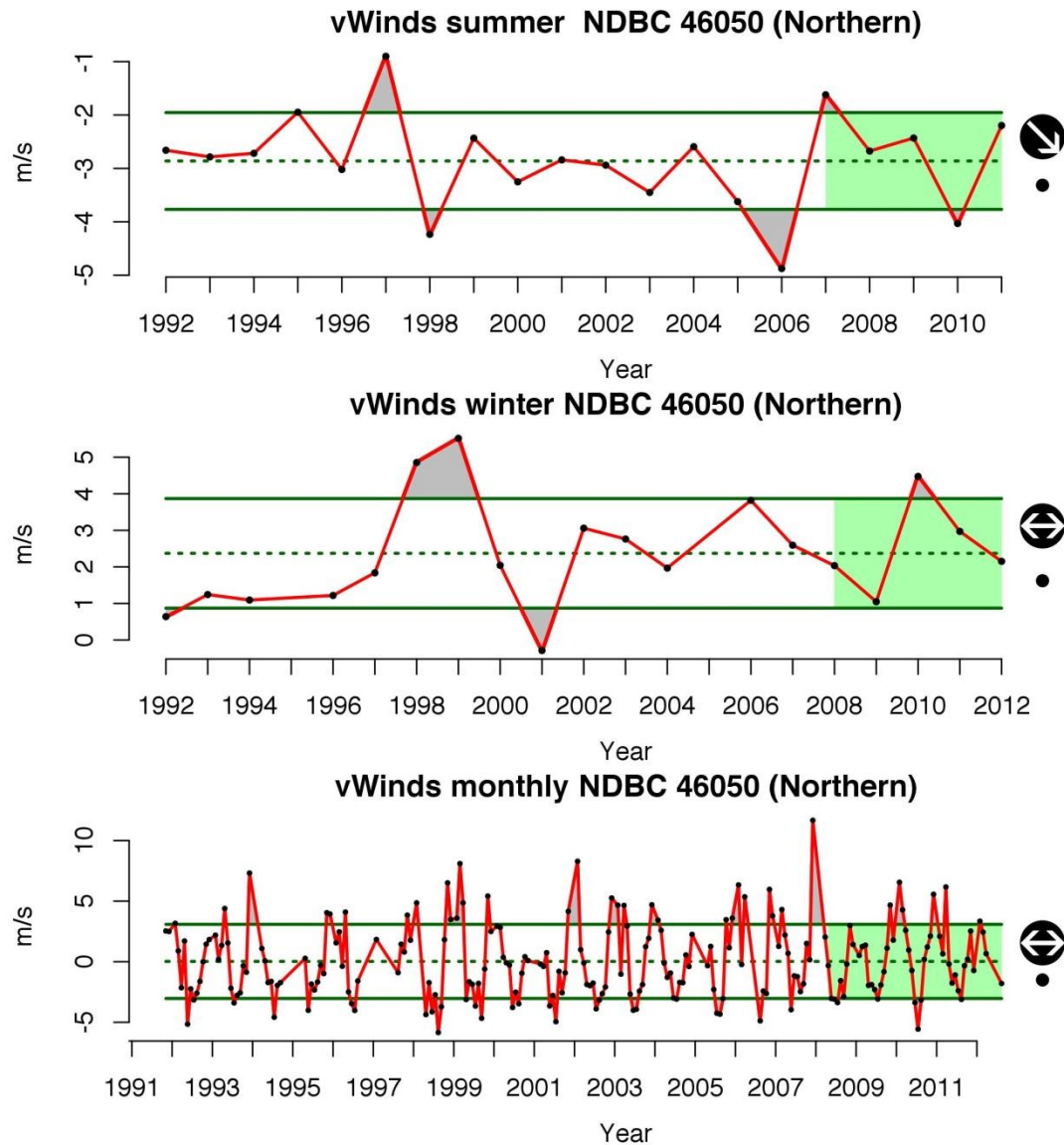


Figure
OC24. Alongshore, meridional winds (vWinds) buoy data from early 1990 -2012 and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. Buoy 46050 was chosen to illustrate patterns in the northern portion of the CCLME.

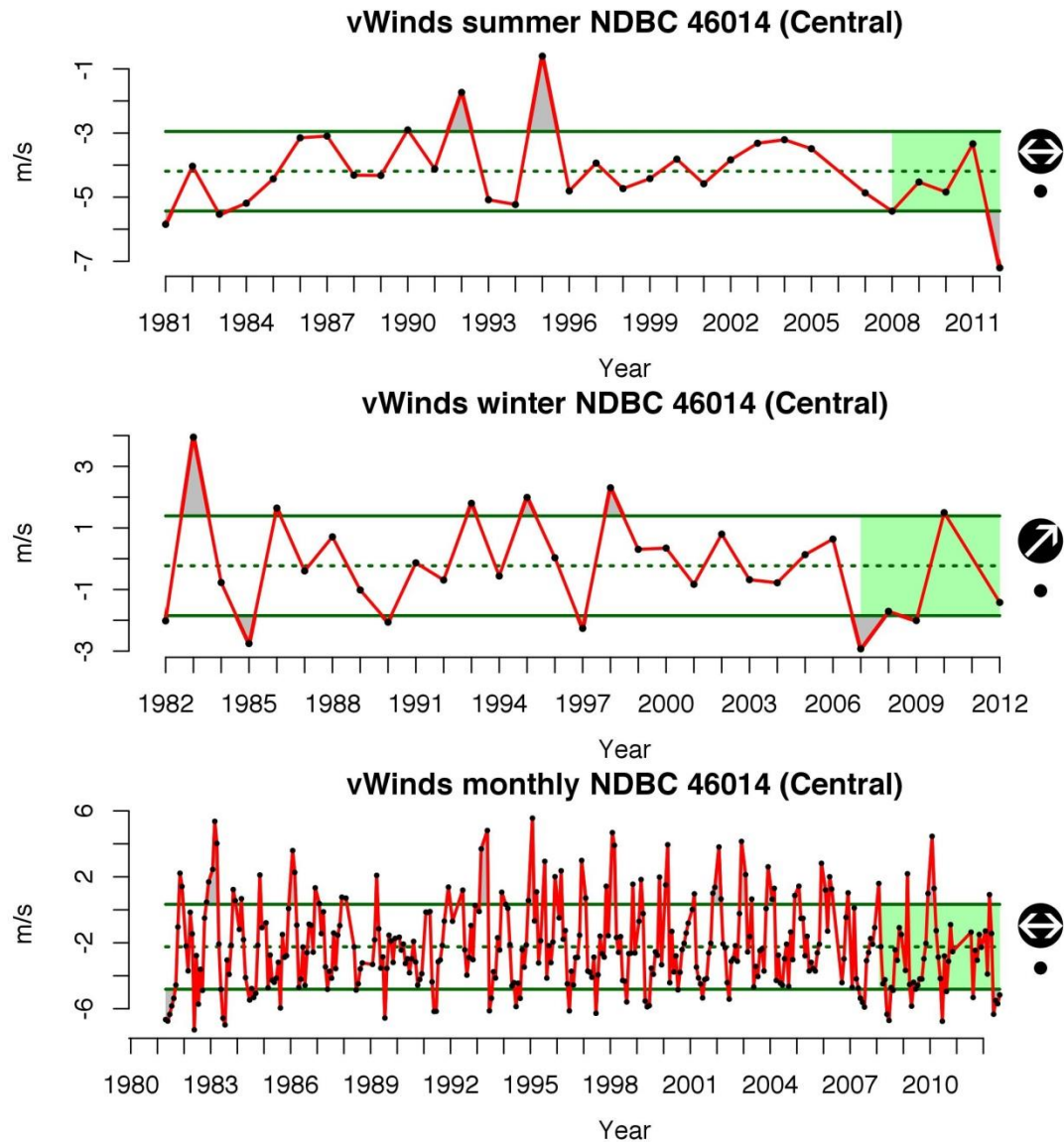


Figure
OC25. Alongshore, meridional winds (vWinds) buoy data from early 1990 -2012 and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. Buoy 46014 was chosen to illustrate patterns in the central portion of the CCLME.

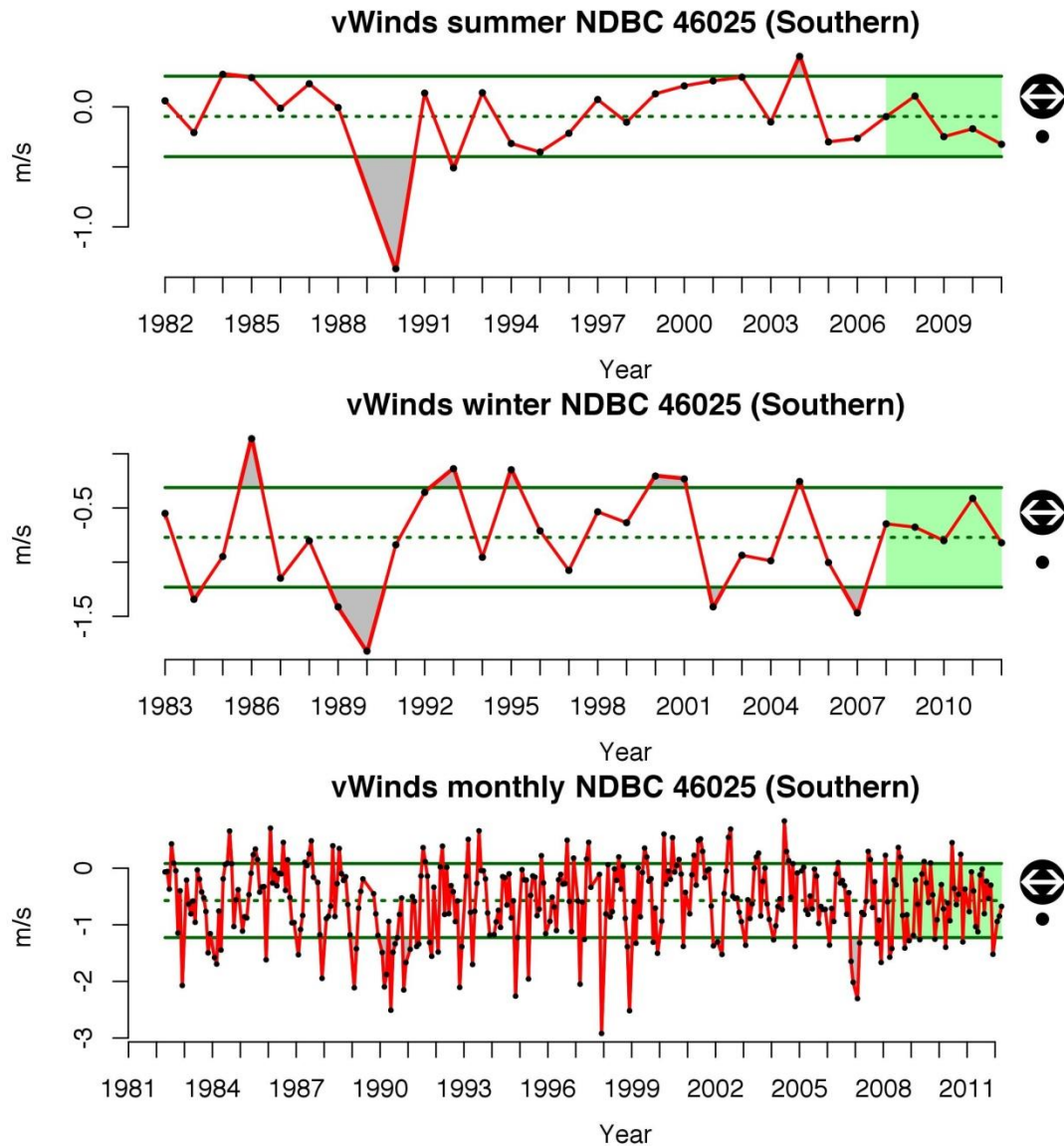


Figure OC26. Alongshore, meridional winds (vWinds) buoy data from early 1990 -2012 and during both a. summer and b. winter. c. Monthly values are included to show seasonal cycles and a continuous time series. Buoy 46025 was chosen to illustrate patterns in the southern portion of the CCLME.

TIMING AND FREQUENCY OF EL NIÑO EVENTS

BACKGROUND

El Niño Southern Oscillation (ENSO) events result from variations in sea level pressure, winds and sea surface temperatures between the eastern and western tropical Pacific. The resulting changes in the tropics have wide reaching consequences on the physical attributes in the CCLME. ENSO events can affect the CCLME through atmospheric teleconnections between the western equatorial Pacific and the North Pacific and by the propagation of Kelvin waves from the equatorial Pacific. El Niño events result in ecosystem-wide effects from changes in species composition to lack of prey availability and breeding failure in top predators, while La Niña events can increase productivity in the system (Chavez et al. 2002).

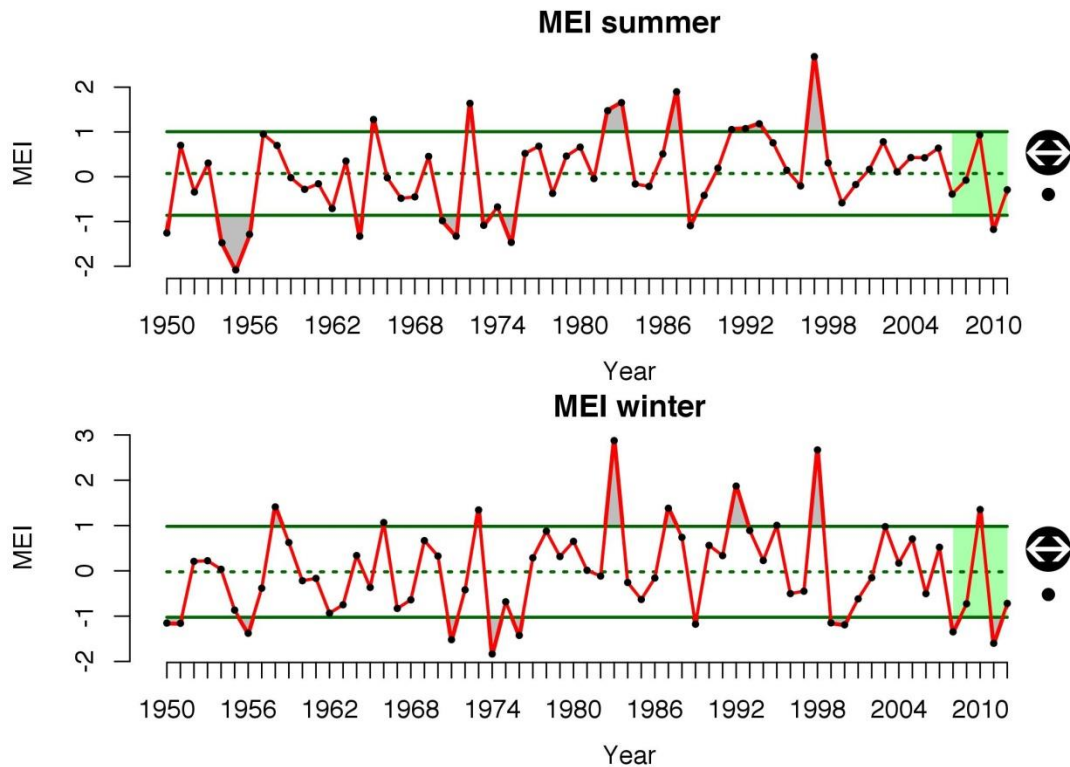
EVALUATION AND SELECTION OF INDICATORS

Winter/summer means of the Northern Oscillation Index (NOI) and the Multivariate ENSO Index (MEI) are used for the timing and strength of El Niño and La Niña events. The NOI measures the teleconnection between the western equatorial Pacific and the north Pacific and is the difference between sea level pressure at the climatological location of the North Pacific High and sea level pressure at Darwin Australia. Large positive (negative) values correspond to a strong (weak) NPH that will result in more (less) coastal upwelling. During an El Niño the influence of the NPH is diminished and the NOI has large negative values. The MEI is derived from several physical indicators and it does not have units. The MEI is one of many potential ENSO indicators (Wolter and Timlin 2011). Large positive values represent El Niño conditions while large negative values represent La Niña conditions. Local SST anomalies from satellite or buoy data also can serve as important local indicators of El Niño effects on the CCLME (Messié and Chavez 2011).

STATUS AND TRENDS

MEI

The Multivariate ENSO Index (MEI) describes ocean-atmosphere coupling in the equatorial Pacific. Positive values of the MEI represent El Niño conditions while negative values represent La Niña conditions. El Niño conditions in the CCLME are associated with warmer surface water temperatures and weaker upwelling winds. The MEI also had an increasing trend, with more positive values since 1977 (Figure OC27). Most recently, the MEI had a relatively strong negative value in the winter of 2008 indicating La Niña conditions that typically favor ocean/atmospheric teleconnections and high productivity coupled with subarctic conditions in the CCS. The MEI switched to positive indicating El Niño conditions in the beginning of 2010, which switched to a negative value in the summer of 2010. La Niña conditions continued through mid-2011 and have begun to return to neutral in late 2011.



Figure

OC27. Multivariate ENSO Index values (MEI) from 1950 -2012 during both a. summer and b. winter.

NOI

See sea surface temperature change above.

CHANGES IN SOURCE WATERS

BACKGROUND

Subarctic and tropical waters are important contributors of source waters to the CCLME at the upstream end and through local upwelling cells. Variations in the volume of subarctic waters occur both at the origination through ventilation (Bograd et al. 2008), transport eastward (Di Lorenzo et al. 2008) in the North Pacific Current (NPC) and as a function of where the NPC approaches the continental shelf and bifurcates into the southward-flowing California Current and the northward-flowing Alaska Current (Bi et al. 2011, Sydeman et al. 2011). Broad scale changes in nutrients and hypoxia in the California current are a function of source water changes and we have observed increased nutrients and decreased oxygen in the Southern California Bight over the past 25 years (Bograd et al. 2008). Earth system models have predicted further decreases in nutrients and oxygen over the next century (2001-2100) in part due to changes in offshore stratification and ventilation (Rykaczewski and Dunne 2011). Broad scale forcing (e.g. indexed by the Pacific Decadal Oscillation, PDO and North Pacific Gyre Oscillation, NPGO) can influence both the strength of transport and the location of bifurcation in the CCLME with downstream ecosystem consequences (King et al. 2011). Increases in subarctic source waters can result in changes in the food-web as cooler arctic waters carry larger, lipid-rich copepods and other plankton, compared to the smaller, often lipid-poor warm water copepods found offshore and to the south. Differences in copepod species composition can serve as ecological corroboration of changes in source water (Peterson and Keister 2003). The result is different trophic

structure near the bifurcation (Bi et al. 2011). Dissolved Oxygen (discussed below) can also be used as an indicator of changes in source water (Bograd et al. 2008, Pierce et al. 2012)

EVALUATION AND SELECTION OF INDICATORS

There are a number of indicators that can assess the status of source waters flowing into the CC including temperature:salinity:oxygen relationships at depth (e.g. spiciness), bifurcation latitude of the NPC, nutrient content of source waters, dissolved oxygen (DO) of source waters, phases of the PDO and NPGO, and volume transport. We have narrowed the list to nutrient content, DO of source waters, copepod biomass anomaly and community structure data, and broad scale indices of the PDO and NPGO. As with previous indicators, the suite offers longevity with time, interpretability, but also measurements relevant to multiple spatial scales.

STATUS AND TRENDS

NPGO

NPGO is a low frequency signal in sea surface heights over the Northeast Pacific. Positive (negative) values of the NPGO are linked with increased (decreased) surface salinities, nutrients, and Chlorophyll-a values in the CCLME (Di Lorenzo et al. 2008). Many NPGO events since 1975 seem to have been more extreme or had a longer duration than those earlier in the time series (Figure OC28). Winter and summer trends were very similar with a broad low from 1991 to 1997 and a peak from 1998 to 2004. Since 2006, values have been increasing with the past 5 years falling above 1 standard deviation from the mean.

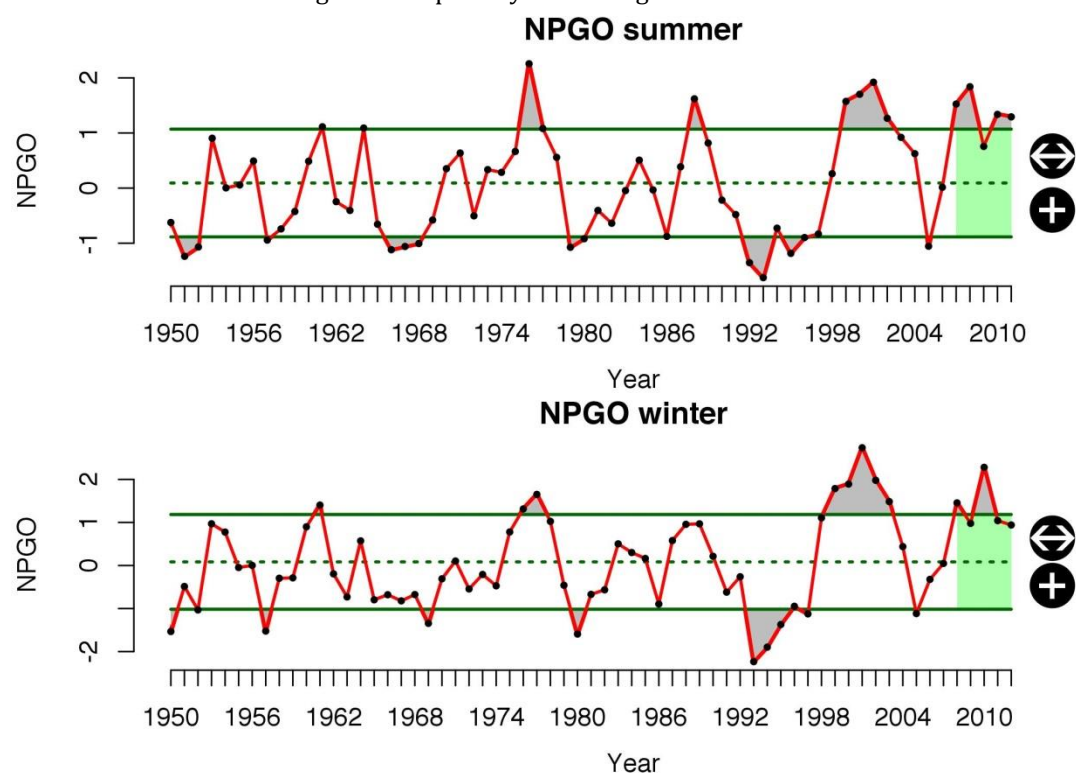


Figure OC28. North Pacific Gyre Oscillation values (NPGO) from 1950 -2012 during both a. summer and b. winter.

NUTRIENT CONTENT

Nutrient content (e.g. NO_2 and NO_3) is a function of upwelling intensity, stratification, but also depends on the source waters that are upwelled. Deep casts at stations 93.30 in CALCOFI and NH25 (150 m) reflect the status of the source waters while the nearshore surface values have more influence from upwelling and stratification (NH05). There was a spike in deep nutrients from NH25 in 2008 but 2009-2011 have shown below average nutrient values at both Newport stations. CALCOFI nutrients in central (station 67.55) and southern (station 93.30) California at 150 m depth show no long-term trend from the data available. Central California nutrients peaked in 2009 and have declined since, while southern California nitrate and nitrite values had a large drop in 2008 and have increased through 2011 (Figure OC29).

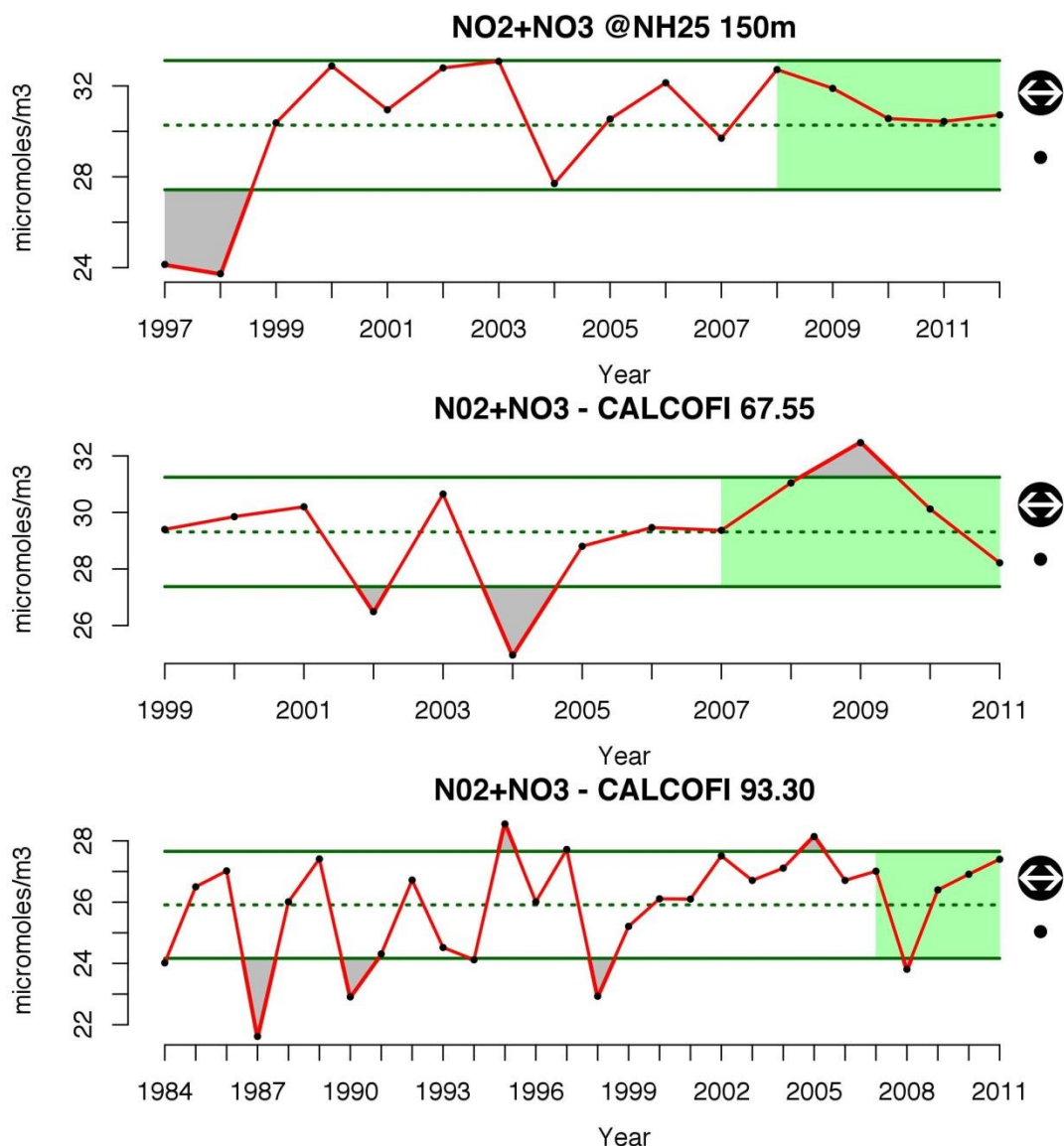
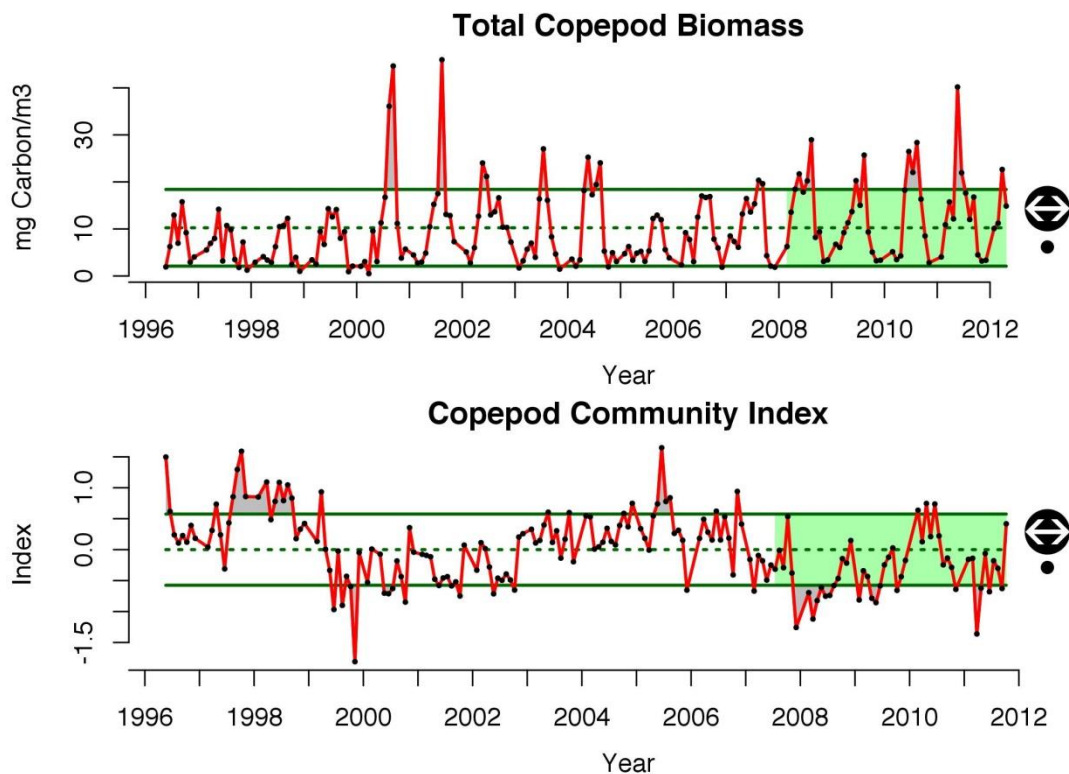


Figure OC29. Summer nutrient data (nitrate + nitrite) at 150m a. 1997 -2012 at station NH25 from the Newport line in the northern CCLME, b. 1999-2011 at CALCOFI station 67.55 for the central CCLME, and c. 1984-2011 at CALCOFI station 93.30 for the southern CCLME.

TOTAL COPEPOD BIOMASS AND SPECIES COMPOSITION

Copepod biomass and species composition vary seasonally with the highest biomass occurring in the summer months, when food is most plentiful, and the lowest biomass in the winter months (Figures OC30). Copepods are transported to the Oregon coast, either from the north/northwest or from the west/south. Copepods that arrive from the north are cold-water species with higher lipid stores and result in greater productivity of downstream predators; these are referred to as the northern copepods. Copepods that arrive from the west or south are referred to as the southern copepods and are less rich in lipids. The cold-water group, the northern copepods, usually dominates the Washington/Oregon coastal zooplankton community in summer, whereas the warm-water southern copepods group usually dominates during winter (Peterson and Miller 1977, Peterson and Keister 2003, Peterson and Schwing 2003). However, the northern and southern copepod anomalies track the PDO and MEI fairly closely, thus this seasonal pattern in species composition can be altered during El Niño events or during periods when the PDO is consistently positive or negative. The copepod community index tracks the shift in species with more northern copepods indexed by positive values and more southern species indexed by negative values. In general, higher abundances of the northern copepods (positive anomalies) are indicative of favorable conditions for many upper trophic-level species, including salmon and seabirds.



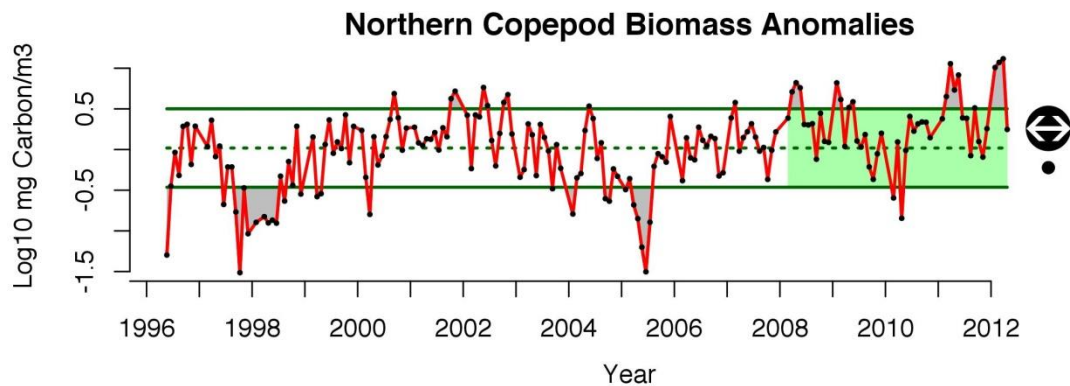
Figure

OC30. a. Total copepod biomass and b. copepod community index monthly from 1996-2012 in the northern California current.

NORTHERN COPEPOD ANOMALY

During the 1997-98 El Niño event, the biomass anomalies of northern copepods was one order of magnitude lower than normal (Figure OC31). With the change in sign of the PDO from positive to negative in mid 1999, the northern copepods responded by showing consistently positive anomalies that prevailed

through 2002 (Peterson et al. 2002, Peterson and Keister 2003). In late 2002, the PDO and MEI turned positive (indicating El Niño conditions) and the northern copepods showed negative anomalies. The anomalies were strongly negative during the summer of 2005, a summer characterized by a two-month delay to the start of upwelling (Kosro et al. 2006) and anomalous species composition among the zooplankton (Mackas et al. 2006). Over the past few years, the northern species have predominated with increases in biomass beginning in late 2006. High biomass values were observed for northern species both in 2008 and 2009 with a brief period of negative anomalies during the small El Niño from May 2009 through May 2010. The highest anomalies in the northern copepod biomass time series (since 1996) were observed in March and April 2011 and also the beginning of 2012, coincident with strongly negative PDO values.

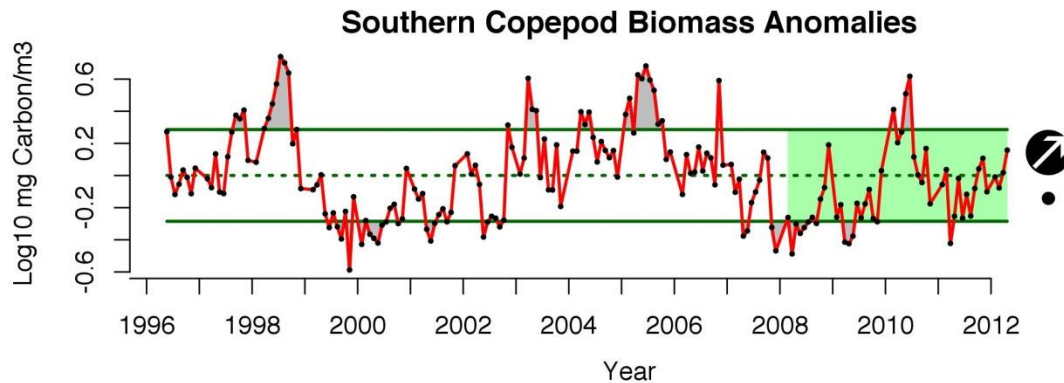


Figure

OC31. Northern Copepod Biomass Anomaly index monthly from 1996-2012 in the northern California current.

SOUTHERN COPEPOD ANOMALY

The highest positive anomalies of the southern species were observed during the 1997-1998 El Niño (Figure OC32). Consistently high positive anomalies of southern species were also observed from 2003 through 2006 coinciding with a period of positive PDO and mostly positive MEI. Over the past few years (since mid-to-late 2009) ocean conditions have been unsettled in that recently there was another small El Niño at the equator. MEI values were positive from May 2009 through May 2010 and the southern copepods responded quite strongly, having anomalies that were similar to those observed during the 1998 and extended (2003-2006) El Niño events. Both the MEI and PDO returned to negative values (signaling a cold ocean) in June 2010 and the southern copepod biomass anomaly became negative in early 2011 following the PDO and MEI sign change by about six months.



Figure

OC32. Southern Copepod Biomass Anomaly index monthly from 1996-2012 in the northern California current.

OCEAN ACIDIFICATION

BACKGROUND

For seawater, an increase in CO_2 leads to a decrease in pH (increased acidification) and carbonate concentration $[\text{CO}_3^{2-}]$. Lower pH and reduced availability of carbonate negatively impacts organisms that rely on calcium carbonate (CaCO_3) for structural and protective shells (Barton et al. 2012). Several planktonic species, such as coccolithophores and pteropods, also rely on calcium carbonate for structural components. Pteropods are important prey for several salmon species in the California Current ecosystem. Aragonite and calcite are the most common forms of CaCO_3 used by organisms. The 'saturation-state' of these minerals changes with pH, temperature and pressure. As ocean waters become more acidic they tend towards undersaturation of CaCO_3 and protective shells and structural parts more readily dissolve. Non-calcifying organisms may also be susceptible to a reduction in pH. Physiological stress through acid-base regulation and cellular ion exchange varies greatly among organisms. There are very limited data available on how different species compensate for a lower pH environment, but the data that are available indicate that invertebrate species are likely the most susceptible, and in general, fish tend to be much less sensitive due to a better capacity for acid-base regulation (Pörtner 2008). However, there is potential for increased vulnerability during reproduction and early life history development, both of which are the focus of recent research.

EVALUATION AND SELECTION OF INDICATORS

The saturation state of aragonite and calcite, the pH, and the DO of waters in the California current all can serve as indicators of ocean acidification. It is likely that synergistic responses among these indicators will be quite difficult to isolate. Although some time series of calcium-carbonate chemistry (e.g. pCO_2 , pH, alkalinity) have been started, we do not have enough data yet to say anything about status and trends. Because increases in CO_2 occur along with decreases in dissolved oxygen, we can use DO as a proxy for acidification in the California current. DO serves as an indicator of multiple pressures and also has a longer time series available than the other indicators of ocean acidification. The trends in DO are discussed below.

Data are obtained from Newport, Oregon's NH Line 25 nautical miles (46 km) offshore, from the central California CALCOFI station 67.55, and from the southern California Bight via CalCOFI's station 93.30 at 22 km offshore.

DISSOLVED OXYGEN AND HYPOXIC EVENTS

BACKGROUND

Low dissolved oxygen concentrations in coastal and shelf waters of the California Current ecosystem is a relatively recent issue (Grantham et al. 2004, Bograd et al. 2008). When dissolved oxygen concentrations fall below 1.4 ml L⁻¹, the waters are considered to be 'hypoxic'. Dissolved oxygen (DO) concentrations in the ocean are dependent on a number of physical and biological processes, including circulation, ventilation, air-sea exchange, production and respiration. In Oregon, upwelling transports hypoxic waters onto productive continental shelves, where respiration can reduce water-column DO and thus subject coastal ecosystems to hypoxic or anoxic conditions. Off southern California, the boundary between oxygenated and hypoxic waters has shoaled in recent years. Some nutrients are supplied from rivers and surface runoff, but these sources are minor inputs to the coastal and shelf ecosystem when compared to upwelling in contrast to the Gulf of Mexico and Chesapeake Bay. For the northern California Current, upwelling primarily occurs during the summer months (May – Sept.) when the seasonal winds blow from the north. Towards the south, upwelling occurs throughout the year (Bograd et al. 2009). The deep, nutrient-rich waters that are brought up onto the shelf are often low in oxygen, but are rarely 'hypoxic' (Hales et al. 2006). Biochemical respiration in the water column and within the sediments draws the oxygen level down further, sometimes to hypoxic or anoxic levels (Connolly et al. 2010). The areas most vulnerable to hypoxia tend to be banks and wider shelf regions where water may be retained for extended periods of time with minimal ventilation from horizontal and vertical mixing (Grantham et al. 2004). There is evidence that the frequency, duration and spatial coverage of hypoxic events has been increasing over the last 20 years (Diaz and Rosenberg 2008), potentially due to increased stratification (reduced vertical mixing) and a decrease in the oxygen concentration of upwelled waters. In the southern portions of the California Current, the shoaling of the permanent Oxygen Minimum Zone is a contributing factor (Helly and Levin 2004, Bograd et al. 2008).

The impact of hypoxia on organisms in the California Current is poorly understood (Keller et al. 2010). Severe events have been shown to kill sessile and slow-moving benthic invertebrates and displace demersal fish species (Grantham et al. 2004, McClatchie et al. 2010). Studies from coastal regions of the Gulf of Mexico and Eastern United States indicate that a range of trophic levels, from plankton to fish, show behavioral changes, may be displaced or killed, or have negative impacts on early life history growth when exposed to low oxygen for extended periods (Rabalais and Turner 2001, Kidwell et al. 2009).

EVALUATION AND SELECTION OF INDICATORS

The indicators for DO are water column profiles of oxygen in % saturation or ml/L. We have chosen DO data from Newport, Oregon's Newport Line at 25 nm (46 km) offshore, from central California's 67.55, and from the southern California Bight via CalCOFI's station 93.30 at 22 km offshore because of their long history and good spatial representation of two portions of the CCS. The data are from 150 meters as this depth as this common depth is targeted to sample source waters.

STATUS AND TRENDS

The northern CCLME has had increased continental shelf hypoxia and shoaling of the hypoxic boundary resulting from enhanced upwelling, primary production, and respiration over the past 15 years (Pierce et al. 2012). Severe and persistent anoxic events have had downstream effects on both demersal fish and benthic invertebrate communities off Oregon (Keller et al. 2010). For example, during a severe anoxic

event in August 2006, surveys found an absence of rockfish on rocky reefs and a large mortality event of macroscopic benthic invertebrates (Chan et al. 2008). Seasonality in oxygen concentrations show oxygenated summer waters along the Newport Hydrographic Line since September 2005 (Figure OC33). In 2007, low oxygen concentrations were observed in the summer although the mean was above 1.4 ml/L. The 2011 data point had lower than average oxygen at NH25. Despite higher than average upwelling in 2008, boundary waters remained well oxygenated save two occasions.

In the southern CCLME (Bograd et al. 2008), deepening of the thermocline and decreased oxygen in deep source waters have resulted in decreased subsurface oxygen through 2007 (Figures OC34-OC35). Contrary to the past five years, large-scale wind forcing models predict hypoxia will continue to expand under Intergovernmental Panel on Climate Change warming scenarios (Rykaczewski and Checkley 2008).

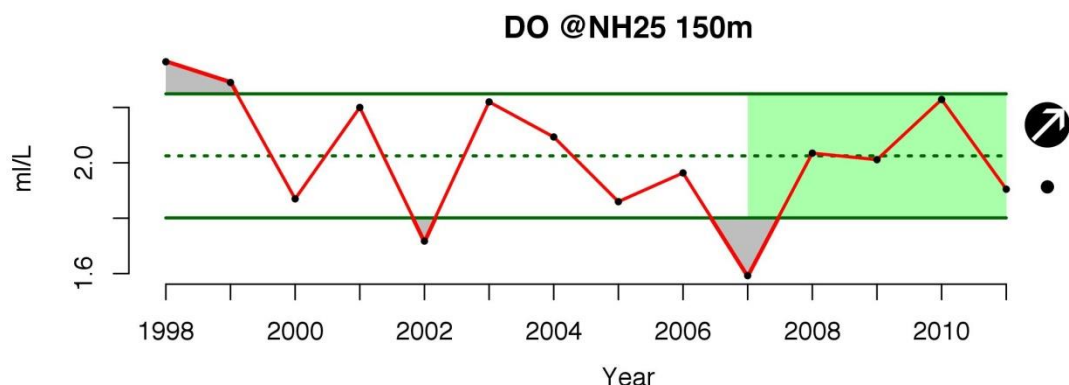


Figure OC33. Summer values of Dissolved Oxygen at 150 meters from the Newport line (Newport, Oregon NH25).

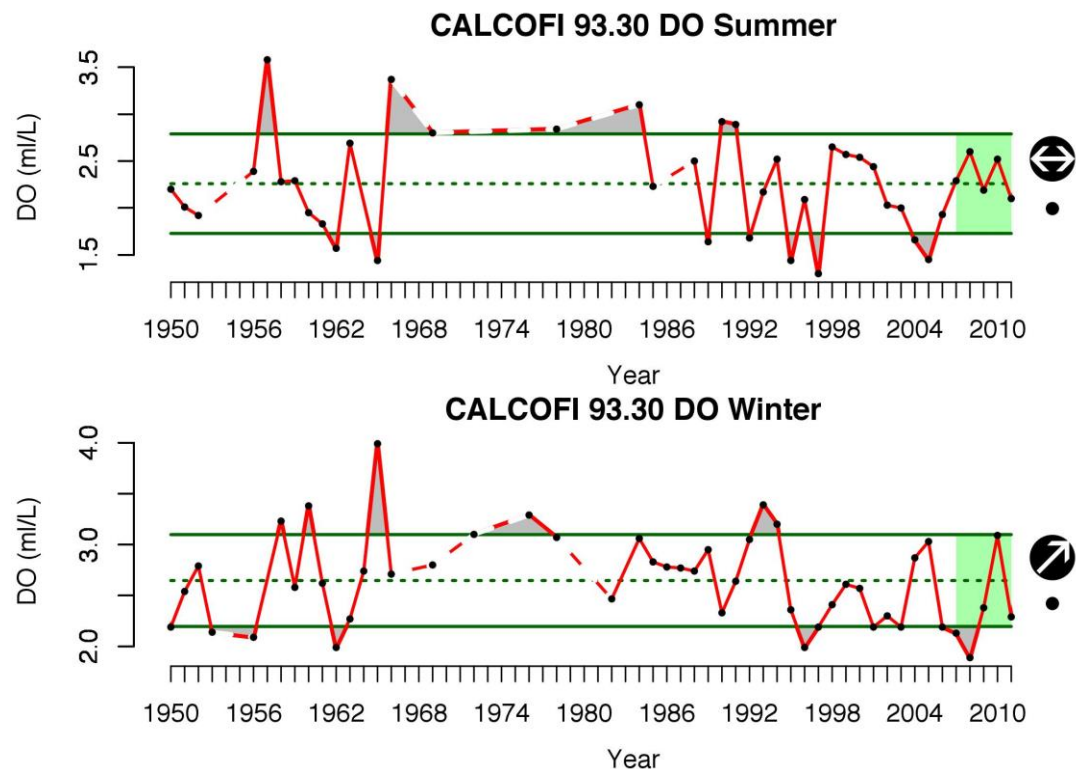
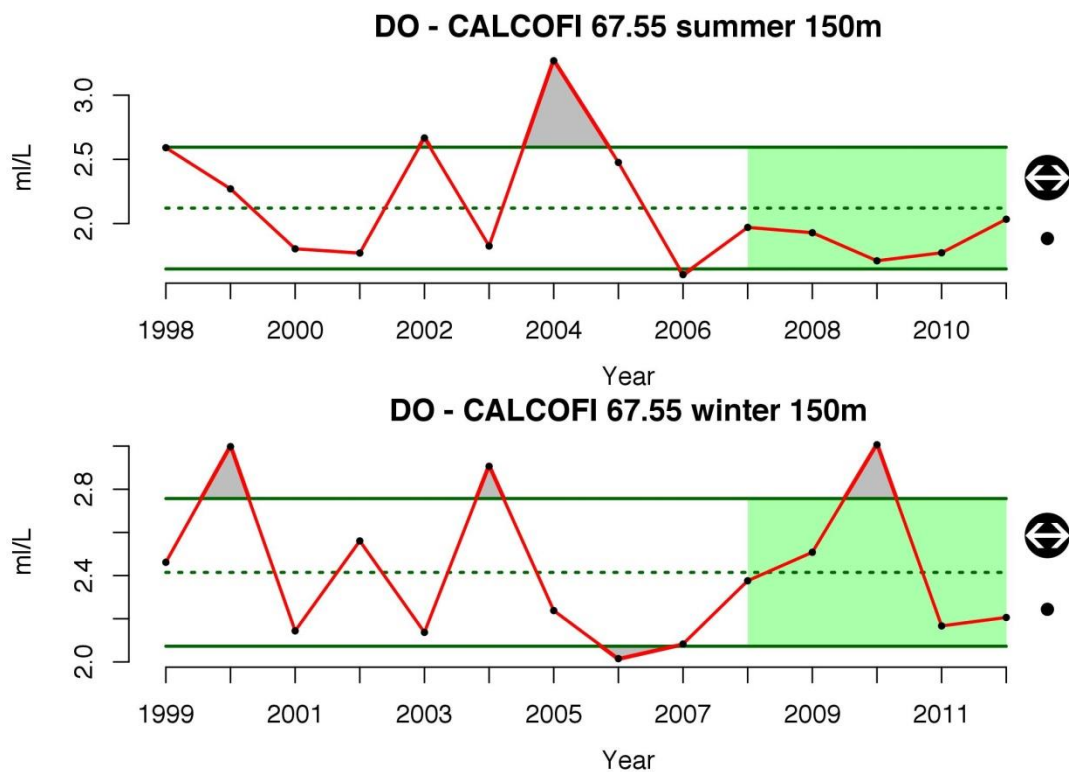


Figure OC34 a. Summer values of Dissolved Oxygen at 150 meters from the southern California bight dataset

(CALCOFI 93.30). b. Winter values of Dissolved Oxygen at 150 meters from the southern California bight dataset (CALCOFI 93.30).



Figure

OC35 a. Summer values of Dissolved Oxygen at 150 meters from the central California dataset (CALCOFI 67.55). b. Winter values of Dissolved Oxygen at 150 meters from the central California dataset (CALCOFI 67.55).

EFFECTS OF ANTHROPOGENIC CLIMATE CHANGE

Ocean temperatures have increased, and are likely to continue to increase for the foreseeable future. Land is expected to heat faster than the ocean and these contrasts in temperatures may result in higher wind speeds (Bakun 1990, Snyder et al. 2003). Warmer waters are also increasing stratification (Roemmich and McGowan 1995, McGowan et al. 2003). The effects of stronger winds and increased stratification on upwelling, temperature, and primary productivity in the CCLME are not well known (Schwing and Mendelssohn 1997, Mendelssohn and Schwing 2002), but clearly will have ecosystem consequences beyond warming surface temperatures. It is important to note that dynamics in the CCS are often dominated by changing wind patterns at local, regional, and basin scales which have masked long-term thermodynamic-forced trends apparent in other ocean ecosystems.

The timing of the seasonal cycle of productivity is changing (GRL 2006, Bograd et al. 2009). Just as terrestrial biological systems are experiencing earlier phenology (IPCC 2007), we may observe an earlier start to the upwelling season in the CCLME, and these patterns may vary by ecoregion. If upwelling occurs earlier, this could result in an earlier seasonal cycle, from earlier phytoplankton blooms to earlier peaks in zooplankton abundance. In contrast, as noted previously, if the efficacy of upwelling is weakened or delayed by increased water stratification, the seasonal cycle of different organisms may be offset, leading to mismatches among trophic levels in both abundance and availability of prey (Sydeman and Bograd 2010).

We are already seeing changes in nutrient values and shoaling of hypoxic zones in many parts of the California Current (Bograd et al. 2008, Chan et al. 2008). These trends are predicted to increase as decreased ventilation of the North Pacific will lead to greater nutrient concentrations in CCLME source waters (Rykaczewski and Dunne 2010). Predicted increases in nitrate are accompanied by decreased DO and increased ocean acidification leading to the potential for multiple stressors on the California Current ecosystem (Doney 2010, Halpern et al. 2010, Keeling et al. 2010).

With these varied scenarios in mind, there is the potential for increased interannual variability in the CCLME upwelling (Bograd et al. 2009, Sydeman and Bograd 2010). A more volatile climate with more extreme events will impact biological systems of the CCLME (Francis and Mantua 2003). Increased upwelling has been hypothesized and predicted in some global climate models (Bakun et al. 2010, Rykaczewski and Dunne 2010, Doney et al. 2012), but there is still much debate as to the ultimate effects of global climate change on upwelling intensity. In addition, evidence of variability and declines in biological systems in the CCLME since about 1990 has already been observed (Sydeman and Bograd 2009). Such changes and others (e.g., range shifts in species' distributions) are likely to continue.

LINKAGES BETWEEN CLIMATE DRIVERS AND SOME EBM COMPONENTS

We examined the hypothesis of covarying trends in physical and biological attributes of the CCLME which is discussed here. The response of biology to anomalous years (e.g. 2005) indicates there has been substantial ecological change in the CCLME, spanning multiple trophic levels (Sydeman and Bograd 2010). Moreover, many of the biological changes are related to physical conditions of the ecosystem in a manner consistent with expectations under global warming. For the biological components investigated, with few exceptions, this generally meant a decline in abundance or productivity. Future reports should examine variance as increased variance results in higher standard error on management targets, potentially requiring more precautionary management of stocks and resources. In summary, multiple dimensions of the CCS are highly dynamic yet many key dimensions are correlated with each other (winds, SST, salinity, nutrients, DO, transport, coastal sea level, etc.) and it is important to remember that the CCLME responds to a combination of remote and local atmospheric forcing that may or may not be sensitive to AGW.

Of note is the recent substantial decline of coho salmon survival off Oregon and the dramatic plunge of Chinook salmon escapement in California in 2007 and 2008 after a peak in 2002. Related to this observation is the reproductive failure of Farallon Island Cassin's auklets in 2005 and 2006 after gradually improving reproductive success throughout the 1990s and early 2000s to a peak in 2002. Previously, changes in seabirds and salmon in central California have been related to one another (Roth et al. 2007), although observations of salmonid declines lag changes in other fish and birds by at least one year. Sydeman et al. (2006) and Jahncke et al. (2008) suggested that the decline in auklet breeding success in 2005 was tied to a reduction of prey abundance (euphausiid crustaceans) due to atmospheric blocking and weak upwelling, but the results in these papers were not conclusive due to limited information on the prey. Chinook salmon are known to feed directly upon euphausiids (Brodeur 1990), particularly during their initial time at sea, as well as forage fish such as Pacific herring (Brodeur and Pearcy 1992), which are known to prey on euphausiids (Foy and Norcross 1999). The abundance and availability of euphausiids to these predators is undoubtedly related to oceanographic processes, such as upwelling and possibly currents, but to date the environmental forcing of these important zooplankton remains largely unknown.

These top predator species appear sensitive to variation in the abundance of prey, which are highly dependent on climatic and oceanic conditions, but linkages have been difficult to establish and may have more to do with spatial availability of prey rather than prey abundance. The past 15 years have seen a mix of

boom and bust years for west coast salmon, and better mechanisms from oceanography to prey to predator will improve our ecological knowledge for these highly variable species. However, declines in the relative abundance of forage fish (juvenile rockfish, herring, and juvenile hake) were recorded and related to changes in salmon and seabird populations and productivity. Thus it is clear that oceanographically mediated predator-prey relationships are key to understanding recent failures in these species and that marine climate variability is playing a role in driving predator-prey interactions.

LINKS TO DATA, AS APPROPRIATE

Table OC1. Top indicators for ocean and climatic pressures. Three stations were chosen when possible for northern, central, and southern portions of the California current. Time series availability often differed across the three locations.

| Pressure | Indicator | Definition and source of data | Time series | Sampling frequency |
|---------------------|-------------------------------|--|-------------|--------------------|
| Ocean acidification | DO | Newport line station NH25 and at 150 meters deep as representative of the northern CCLME. | 1998 - 2011 | monthly |
| | | CALCOFI station 93.30 at 150 meters deep as representative of the southern CCLME. | 1984 - 2012 | quarterly |
| | | CALCOFI station 67.55 at 150 meters deep as representative of the central CCLME. | 1998 - 2011 | quarterly |
| Decreasing oxygen | DO | Newport line station NH25 and at 150 meters deep as representative of the northern CCLME. | 1998 - 2011 | monthly |
| | | CALCOFI station 93.30 at 150 meters deep as representative of the southern CCLME. | 1984 - 2012 | quarterly |
| | | CALCOFI station 67.55 at 150 meters deep as representative of the central CCLME. | 1998 - 2011 | quarterly |
| Sea level rise | Coastal Sea Level | Sea Level measured by tide gauges at South Beach, OR. | 1967 - 2012 | daily |
| | | Sea Level measured by tide gauges at San Francisco, CA. | 1897 - 2012 | daily |
| | | Sea Level measured by tide gauges at San Diego, CA. Data were obtained from UHawaii Sea Level Center: http://uhslc.soest.hawaii.edu/home | 1906 - 2012 | daily |
| Temperature change | Sea surface buoy temperatures | Sea surface temperatures measured by NDBC buoy 46050 (44.639° N 124.534° W; 37 km from land). | 1991 - 2012 | hourly |
| | | Sea surface temperatures measured by NDBC buoy 46014 (39.235° N 123.974° W; 17 km from land). | 1981 - 2012 | hourly |
| | | Sea surface temperatures measured by NDBC buoy 46025 (22.749° N 119.053° W; 40 km from land). Data are available at National Data Buoy Center: http://ndbc.noaa.gov | 1982 - 2012 | hourly |
| | PDO | Pacific Decadal Oscillation (PDO) is the dominate pattern of North Pacific SST anomalies. Data are available at: http://jisao.washington.edu/pdo/ | 1900 - 2012 | monthly |
| | NOI | Northern Oscillation Index (NOI) measures atmospheric teleconnections between North Pacific High and northeast Pacific. Data are available at: http://coastwatch.pfeg.noaa.gov/erddap/index.html | 1967 - 2012 | monthly |
| | MEI | Multivariate ENSO Index (MEI) reports on the status of the coupled ocean-atmosphere ENSO events. Data are available at: http://www.esrl.noaa.gov/psd/enso/mei/mei.ht | 1950 - 2012 | monthly |

ml

| | | | | |
|---|--|---|-------------|-----------|
| Water column structure | Pycnocline depth | Three stations, Newport NH25, CALCOFI 93.30, and 67.55 were used for water column structure. | 1984 - 2012 | quarterly |
| | Pycnocline strength | Three stations, Newport NH25, CALCOFI 93.30, and 67.55 were used for water column structure. | 1984 - 2012 | quarterly |
| Changes in source waters | Nutrient content (NO ₂ +NO ₃) | Three stations, Newport NH25, CALCOFI 93.30, and 67.55 were used for water column structure. Nitrate+nitrate concentrations at 150 m shows variations in source water. | 1984 - 2012 | quarterly |
| | Zooplankton community structure | Newport line data are compiled into four indices (total biomass, northern anomaly, southern anomaly, and copepod index). | 1998 - 2011 | monthly |
| | NPGO | North Pacific Gyre Oscillation (NPGO) explains variations in the circulation of the North Pacific Gyre. The NPGO describes nutrient concentrations in the CCS. Data are available at: http://www.o3d.org/npgo/ | 1950 - 2012 | monthly |
| Changes in CC transport & mesoscale variability | EKE | Eddy Kintectic Energy (EKE) was calculated over three spatial locations (6 degree mean), at 33°, 39° and 45° N. Meridional and zonal geostrophic velocities used in the EKE calculations are distributed by Aviso at: http://www.aviso.oceanobs.com/duacs/ | 1992 - 2012 | daily |
| Timing and strength of upwelling | vWinds | North winds are drive much of coastal upwelling and are measured by NDBC buoy 46050 (44.639° N 124.534° W; 37 km from land) | 1991 - 2012 | hourly |
| | | North winds are measured by NDBC buoy 46014 (39.235° N 123.974° W; 17 km from land) | 1981 - 2012 | hourly |
| | | North winds are measured by NDBC buoy 46025 (22.749° N 119.053° W; 40 km from land). Data are available at National Data Buoy Center: http://ndbc.noaa.gov | 1982 - 2012 | hourly |
| | UI | Upwelling Index (UI) denote the strength of coastal upwelling and downwelling; data are presented at 33°, 39° and 45° N. | 1967- 2012 | daily |
| | STI | Spring Transition Index (STI) denotes the start of the upwelling season. It is derived from the daily UI and data are presented at 33°, 39° and 45° N. | 1967- 2012 | yearly |
| | TUMI | Total Upwelling Magnitude Index (TUMI) is the amount of upwelling between the spring and fall transition dates. It is derived from the daily UI at 33°, 39° and 45° N. | 1967- 2012 | yearly |
| | LUSI | Length of Upwelling Index (LUSI) is the number of days during the upwelling season. It is derived from the daily UI at 33°, 39° and 45° N. Data are available at: http://www.pfeg.noaa.gov/products/las.html | 1967- 2012 | yearly |

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| Timing and frequency of El Niño events | MEI | Multivariate ENSO Index (MEI) measures the magnitude and duration of El Niño and La Niña events. Data are available at: http://www.esrl.noaa.gov/psd/enso/mei/ | 1950 - 2012 | monthly |
| | NOI | Northern Oscillation Index (NOI) measures atmospheric teleconnections between North Pacific High and northeast Pacific. Large negative values usually occur during an El Niño event. Data are available at http://coastwatch.pfeg.noaa.gov/erddap/index.html | 1950 - 2012 | monthly |

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ANTHROPOGENIC DRIVERS AND PRESSURES

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TABLE OF CONTENTS (AP)

| | |
|--|-----|
| Executive Summary | 78 |
| Detailed Report..... | 81 |
| Fisheries pressures..... | 86 |
| Fishery removals..... | 86 |
| Habitat destruction..... | 97 |
| Non-fisheries pressures..... | 105 |
| Summary of non-fisheries pressures | 105 |
| Aquaculture..... | 118 |
| Atmospheric pollution..... | 120 |
| Benthic structures | 121 |
| Coastal engineering | 123 |
| Commercial shipping activity | 126 |
| Disease/pathogens..... | 129 |
| Dredging..... | 130 |
| Freshwater retention | 131 |
| Inorganic pollution..... | 133 |
| Invasive species..... | 137 |
| Light pollution..... | 139 |
| Marine debris | 141 |
| Nutrient input | 144 |
| Ocean-based pollution | 147 |
| Ocean mining..... | 149 |
| Offshore oil and gas activity | 149 |
| Organic pollution | 151 |

| | |
|--|-----|
| Power plants..... | 153 |
| Recreational use..... | 156 |
| Seafood demand..... | 157 |
| Sediment input..... | 158 |
| Tourism..... | 162 |
| Linkages between drivers and EBM components..... | 162 |
| References cited | 164 |

LIST OF TABLES AND FIGURES (AP)

| | |
|---|----|
| Figure AP.S.1. Short- and long-term status of annual landings (1981 – 2011) by species groups in the CCLME. The short-term trend indicates whether landings increased, decreased, or remained the same over the last five years. The long-term trend represents the difference between the mean of the last five years and the mean of the full time series..... | 79 |
| Figure AP.S.2. Short- and long-term status of non-fisheries pressures in the CCLME. See Fig. AP.S1 for description of axes and interpretation of data points. Numbers in parentheses in the legend are the number of years in the time series for each pressure. | 80 |
| Table AP1. General ecosystem impacts, types and identified anthropogenic pressures in the CCLME. | 82 |
| Figure AP1. Annual landings of groundfish in the CCLME from 1981 – 2011 (Pacific hake <i>Merluccius productus</i> excluded)..... | 89 |
| Figure AP2. Annual landings of Pacific hake <i>Merluccius productus</i> in the CCLME from 1981 – 2011..... | 89 |
| Figure AP3. Annual landings of coastal pelagic species (CPS) in the CCLME from 1981 – 2011. CPS include Pacific sardine <i>Sardinops sagax</i> , Pacific mackerel <i>Scomber japonicus</i> , northern anchovy <i>Engraulis mordax</i> , jack mackerel <i>Trachurus symmetricus</i> , and market squid <i>Loligo opalescens</i>). | 90 |
| Figure AP4. Annual landings of highly migratory species (HMS) in the CCLME from 1981 – 2011. HMS include tunas, sharks, billfish/swordfish and dorado <i>Coryphaena hippurus</i> | 90 |
| Figure AP5. Annual landings of salmon in the CCLME from 1981 – 2011..... | 90 |
| Figure AP6. Annual landings of crab in the CCLME from 1981 – 2011. | 91 |
| Figure AP7. Annual landings of shrimp in the CCLME from 1981 – 2011..... | 91 |
| Figure AP8. Annual landings of shellfish in the CCLME from 1981 – 2011..... | 91 |
| Figure AP9. Annual landings of all other species in the CCLME from 1981 – 2011..... | 92 |
| Figure AP10. Short- and long-term status of annual landings (1981 – 2011) by species groups in the CCLME. The short-term trend indicates whether landings increased, decreased or remained the same over the last five years. The long-term trend represents the difference between the mean of the last 5 years and the mean of the full time series..... | 92 |
| Figure AP11. Annual commercial trawl landings in the CCLME from 1981 – 2011. | 93 |
| Figure AP12. Annual commercial shrimp trawl landings in the CCLME from 1981 – 2011. | 93 |
| Figure AP13. Annual hook-and-line landings in the CCLME from 1981 – 2011..... | 93 |
| Figure AP14. Annual net-gear landings in the CCLME from 1981 – 2011. | 94 |

| | |
|--|-----|
| Figure AP15. Annual pot and trap landings in the CCLME from 1981 – 2011. | 94 |
| Figure AP16. Annual troll-caught landings in the CCLME from 1981 – 2011. | 94 |
| Figure AP17. Annual landings of all other miscellaneous gear in the CCLME from 1981 – 2011. | 95 |
| Figures AP18. Short- and long-term status of annual landings (1981 – 2011) by gear type in the CCLME. The short-term trend indicates whether landings increased, decreased or remained the same over the last five years. The long-term trend represents the difference between the mean of the last 5 years and the mean of the full time series. | 95 |
| Figure AP19. Total mortality estimates of groundfish (Pacific hake <i>Merluccius productus</i> excluded) in the CCLME from 2005 - 2010. | 96 |
| Figure AP20. Total mortality estimates of Pacific hake <i>Merluccius productus</i> in the CCLME from 2005 - 2010. | 96 |
| Figure AP21. Short- and long-term status of annual total fishing mortality (2005 – 2010) by species groups in the CCLME. The short-term trend indicates whether total fishing mortality increased, decreased or remained the same over the last five years. The long-term trend represents the difference between the mean of the last 5 years and the mean of the full time series. | 97 |
| Figure AP22. Total distance trawled (km) along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels. | 99 |
| Figure AP23. Distance trawled (km) within slope sedimentary habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels. | 99 |
| Figure AP24. Distance trawled (km) within shelf sedimentary habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels. | 99 |
| Figure AP25. Distance trawled (km) within slope rocky habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels. | 100 |
| Figure AP26. Distance trawled (km) within shelf rocky habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels. | 100 |
| Figure AP27. Distance trawled (km) within basin sedimentary habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels. | 100 |
| Figure AP28. Distance trawled (km) within basin rocky habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels. | 101 |
| Figure AP29. Distance trawled (km) within ridge sedimentary habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels. | 101 |
| Figure AP30. Distance trawled (km) within ridge rocky habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels. | 101 |

| | |
|--|-----|
| Figure AP31. Short- and long-term status of total distance trawled (1999 – 2004) by habitat type in the CCLME. The short-term trend indicates whether distance trawled increased, decreased or remained the same over the last five years. The long-term trend represents the difference between the mean of the last 5 years and the mean of the full time series. | 102 |
| Table AP2. Summary of fisheries indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, three out of five primary considerations criteria support “landings” as an indicator of fishery removals. | 103 |
| Table AP3. Top indicators for fisheries pressures. | 104 |
| Figure AP32. Short- and long-term status of non-fisheries pressures in the CCLME. The short-term trend indicates whether the indicator increased, decreased or remained the same over the last five years. The long-term trend represents the difference between the mean of the last 5 years and the mean of the full time series.. | 106 |
| Table AP4. Summary of non-fisheries indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, finfish production as an indicator of finfish aquaculture has peer-reviewed literature supporting two out of five primary considerations criteria. | 107 |
| Table AP5. Top indicators for non-fisheries related anthropogenic pressures. | 114 |
| Figure AP33. Production of finfish aquaculture occurring in marine waters of the CCLME. | 119 |
| Figure AP34. U.S. production of shellfish (clams, mussels and oysters) aquaculture. | 120 |
| Figure AP35. Precipitation-weighted mean concentration (mg/L) of sulfates deposited out of the atmosphere in CA, OR, and WA. | 121 |
| Figure AP36. The number of offshore oil and gas wells in production or shut-in in the CCLME. | 123 |
| Figure AP37. U.S. population in coastline counties of WA, OR and CA. | 126 |
| Figure AP38. Volume (trillions m ³) of water disturbed during transit of commercial shipping vessels along the coast of the CCLME. | 128 |
| Figure AP39. Volume (millions m ³) of dredged sediments from projects originating in WA, OR and CA. | 131 |
| Figure AP40. Volume (millions m ³) of freshwater stored behind dams in WA, OR and CA. | 133 |
| Figure AP41. Polygon of the watershed that drains into the CCLME and used to clip impervious surface area data layers (based on Halpern et al. 2009). | 135 |
| Figure AP42. Normalized index of ISA-toxicity-weighted chemical releases in WA, OR and CA industrial facilities. | 137 |
| Figure AP43. Metric tons (millions) of cargo moved through ports in WA, OR and CA. | 139 |

| | |
|---|-----|
| Figure AP44. Polygon of the CCLME used to clip all nighttime lights data layers | 140 |
| Figure AP45. Normalized index of the sum of average nighttime lights in waters of the CCLME. | 141 |
| Figure AP46. Predicted counts of debris along the a) northern and b) southern coasts of the CCLME (Point Conception separates the regions). Data provided by Christine Ribic (Ribic et al. 2012). | 143 |
| Figure AP47. Normalized index of the sum of nitrogen and phosphorus applied as fertilizers in WA, OR and CA. | 147 |
| Figure AP48. Normalized index that combines the volume (millions m ³) of water disturbed by vessels during transit in port and along the coast and the volume of cargo moving through U.S. ports. | 149 |
| Figure AP49. Normalized index of the sum of oil and gas production from offshore wells in CA. | 151 |
| Figure AP50. Normalized index of toxicity-weighted concentrations of 16 pesticides measured in WA, OR and CA. | 153 |
| Figure AP51. Daily saline water withdrawals (millions m ³) from thermoelectric power plants in CA, OR and WA. | 155 |
| Figure AP52. Annual beach attendance (millions of persons) at state parks and beaches with access points to a beach in WA, OR and CA. | 157 |
| Figure AP53. Total consumption of edible and non-edible fisheries products across the United States. | 158 |
| Figure AP54. Volume (millions m ³) of freshwater impoundments in WA, OR and CA (increasing freshwater storage is a proxy for decreasing sediment input). | 161 |

OVERVIEW

Fisheries landings of crab and shrimp have increased in recent years, while landings of salmon and groundfish remain at historically low levels. Many non-fisheries pressures (e.g., shipping activity, industrial pollution, recreational use) have decreased over the short term, possibly reflecting slowing economic conditions; while, nutrient input, dredging, and shellfish aquaculture may be increasing at historically high levels if short-term trends persist over the next few years.

EXECUTIVE SUMMARY

As human population size and demand for seafood increases globally and within the California Current Large Marine Ecosystem (CCLME), numerous human activities in the ocean (e.g., fishing and shipping activity) and on land (e.g., pollutants and runoff from agricultural activities) need to be recognized and incorporated into management of aquatic resources. However, information about the status and trends of these human-related pressures is often buried in state agency reports, described at small spatial scales, or measured inconsistently among local, state and federal entities. Here, we gathered and produced the best available time series data on anthropogenic pressures across the entire CCLME. We used these data sets to quantify relative changes in anthropogenic pressures, which in turn can provide the foundation for subsequent integrative analyses, such as risk analyses and management strategy evaluations, of cumulative effects on multiple components of the California Current ecosystem (e.g., fisheries, protected species, ecological integrity, and human dimensions).

We developed indicators for 23 anthropogenic pressures on the CCLME. These pressures were divided into fisheries and non-fisheries related pressures and ranged in scope from land-based pressures such as inorganic pollution and nutrient input to at-sea pressures such as fisheries removals, commercial shipping, and offshore oil and gas activities. Ultimately, we evaluated 44 different indicators and selected the best indicator(s) to describe the status and trends of each pressure. Indicators were evaluated using the indicator selection framework developed by Levin et al. (2011) and Kershner et al. (2011) and used in the previous version of NOAA's Integrated Ecosystem Assessment for the California Current (Levin & Schwing 2011). We gathered data for each of the chosen indicators from numerous sources to develop time series and describe the status and trends for each pressure across the entire CCLME.

The status of each indicator was evaluated against two criteria: short-term trend (over the last five years) and status relative to the long-term historic mean. The historical status of each indicator should be placed in context with the temporal range of data available for each time series. For example, data available for some indicators was limited to <10 years while other indicators had data spanning >50 years; thus, the short-term mean will not likely be different from the long-term mean for time series of shorter duration simply because of data availability. However, most indicators were chosen specifically because they were the most fundamentally sound datasets and will continue to be measured over time, providing meaningful comparisons in future iterations of the IEA.

Fisheries provide important services to society, including production of food, employment, livelihood, and recreation, but can also affect the ecosystem by directly removing individual fish and by disturbing habitat from the use of bottom trawls and other bottom-tended gear. Total mortality estimates are the best indicator of fisheries removals, but data are limited to very few years and are only calculated for groundfish species. Thus, we evaluated landings of catch as the best indicator of fisheries removals across the entire CCLME (Fig. AP.S.1). Landings of coastal pelagic species were higher than historic levels over the last five years; crab and shrimp landings increased over the short term but were still within historic levels; and landings of salmon and groundfish species (excluding hake) were at historically low levels for the last five years. All other species groups were consistently within historic landing levels. In addition, trawling effort showed a shift among habitat types, which corresponded, in part, to depth-related spatial closures implemented by the Pacific Fishery Management Council to reduce fisheries' impact on depleted species.

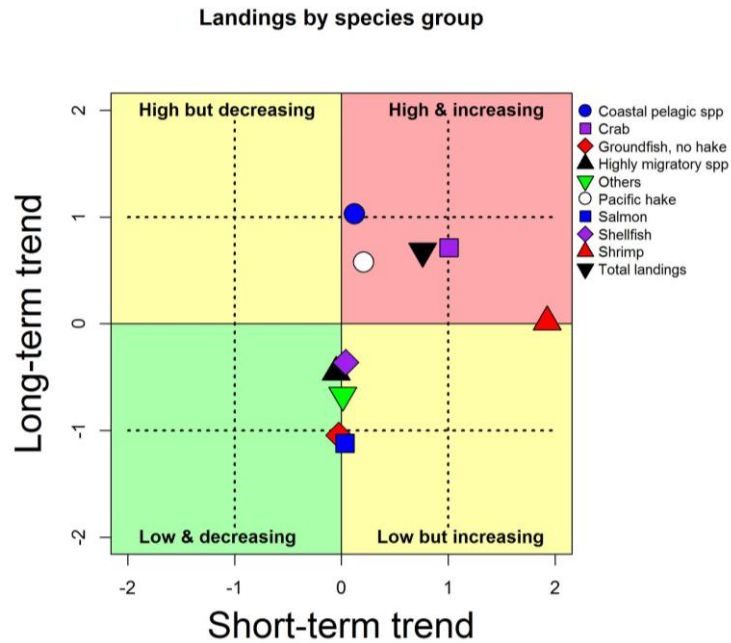


Figure AP.S.1. Short- and long-term status of annual landings (1981 – 2011) by species groups in the CCLME. Prior to plotting, time series data were normalized to place them on the same scale. The short-term trend indicates whether landings increased, decreased, or remained the same over the last five years. The long-term trend represents the difference between the mean of the last five years and the mean of the full time series. Data points outside the dotted lines (1.0 standard deviation) are considered to be increasing or decreasing in the short term or show that the current status is lower or higher than the long-term mean of the time series.

Most indicators of non-fisheries related pressures showed either significant short-term trends or their current status was at historically high or low levels (Fig. AP.S.2). Indicators of inorganic, organic and ocean-based pollution, commercial shipping activity, recreational use, and invasive species have all decreased over the short-term, while indicators of dredging, shellfish aquaculture, and marine debris (in the northern CCLME) increased; all of these pressures, though, remained within historic levels. In contrast, indicators of seafood demand, sediment and freshwater retention, nutrient input, power plant activity, and coastal engineering remained relatively constant over the short-term, but were above historic levels, while indicators of offshore oil and gas activity and related benthic structures were at historically low levels.

Taken together, these results support two primary conclusions: 1) decreasing trends of several non-fisheries pressures (e.g., shipping related indicators, industrial pollution, and recreational activity) potentially reflect slowing economic conditions over the last few years and 2) non-fisheries pressures at historically high levels have leveled off and are not continuing to increase, although nutrient input has increased over the last decade and dredging and shellfish aquaculture

are increasing and may increase to historically high levels if short-term trends persist over the next few years (see specific time series data for each pressure in the *detailed report*).

The interpretation of the status and trends of these pressures may differ depending on the EBM component of interest. For example, a decreasing trend in fisheries removals may be “good” for rebuilding stocks of protected resources or it could be “bad” for the economies of vibrant coastal communities. In addition, none of these pressures act upon the ecosystem independently (i.e. many pressures are acting simultaneously on

populations), and we have little understanding about whether the overall effects of multiple pressures will be additive, synergistic, or antagonistic on

populations of interest. Subsequent sections of the IEA begin to integrate the cumulative effects of multiple pressures on multiple EBM components (see “risk” sections for each EBM component in *Section 3* and the various management strategy evaluations in *Section 4*). Moreover, these anthropogenic pressures will interact with the underlying effects of climatic and oceanographic pressures (detailed in *Section 2: Oceanographic and Climatic Drivers and Pressures*). The integration of anthropogenic, oceanographic, and climatic pressures on multiple EBM components can now be modeled using various “end-to-end” ecosystem models (e.g., Atlantis; Fulton et al. 2011), but marine ecologists and fisheries scientists need to develop creative methods in the field to test the validity of these models’ hypotheses and increase managers’ confidence in decision making.

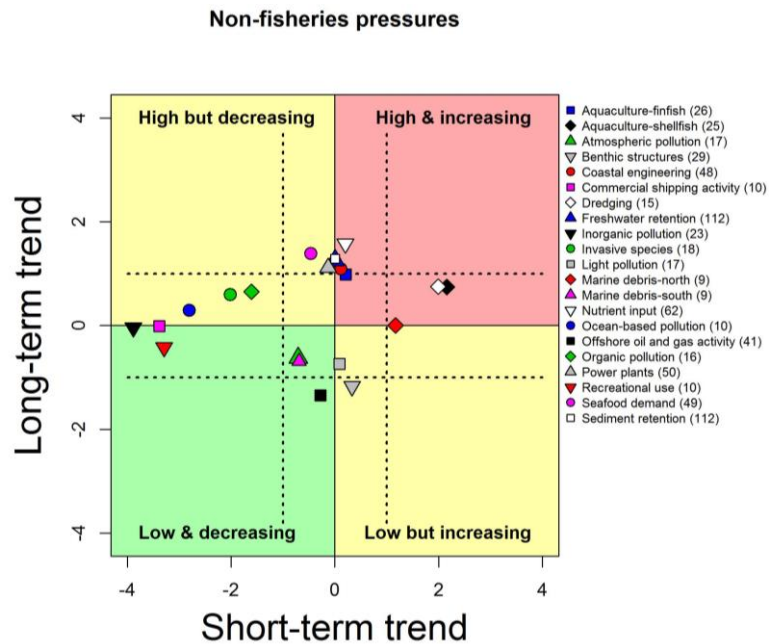


Figure AP.S.2. Short- and long-term status of non-fisheries pressures in the CCLME. See Fig. AP.S1 for description of axes and interpretation of data points. Numbers in parentheses in the legend are the number of years in the time series for each pressure.

DETAILED REPORT

The ultimate aim of the California Current Integrated Ecosystem Assessment (CCIEA) is to fully understand the web of interactions that links drivers and pressures to ecosystem-based management (EBM) components (see *Preface* for description of EBM components addressed in the IEA) and to forecast how changing environmental conditions and management actions affect the status of EBM components. In order to capture the breadth of pressures acting upon the California Current Large Marine Ecosystem (CCLME), a lengthy list of drivers and pressures was compiled. Here we define drivers as factors that result in pressures that in turn cause changes in the ecosystem. For the purposes of an IEA, both natural and anthropogenic forcing factors are considered. Natural forces, such as climate variability, generally cannot be controlled but must be accounted for in management. In contrast, pressures related to anthropogenic factors can be controlled or managed, at least in principle. For example, human population size in the coastal zone can be directly related to anthropogenic pressures such as coastal development, habitat loss and degradation, and fishing effort – all activities that are currently managed by various regulatory agencies and jurisdictions.

The first step was to identify a suite of drivers/pressures that were most closely associated with impacts and changes to the different EBM components in the CCIEA. We used several publications (Halpern et al. 2008, Sydeman and Elliott 2008, Halpern et al. 2009, Sydeman and Thompson 2010, Teck et al. 2010, Peterson et al. 2012) to develop an initial list of potential pressures on the CCLME and then supplemented this list with other identified pressures. During reviews of the literature, we identified 32 primary groups of pressures on the CCLME, and these were categorized as “oceanographic and climatic” or “anthropogenic”. Each category of pressures is discussed in separate sections of the CCIEA. Indicators for each of these pressures were then evaluated using the indicator selection framework developed by Levin et al. (2011) and Kershner et al. (2011) and used in the previous version of NOAA’s Integrated Ecosystem Assessment for the California Current (Levin and Schwing 2011). Briefly, each indicator was scored against 18 different criteria in three categories: Primary considerations (e.g., is the indicator theoretically sound?), data considerations (e.g., does data exist across time and space?), and other considerations (e.g., is the indicator easily understood by managers and the public?). Scoring was based on whether each indicator had good support (score of 1), mixed support (score of 0.5) or no support (score of 0) in the scientific literature for each criteria. These scores were added up and compared across indicators for the same pressure. Highly-ranked indicators were used in further analyses.

The second step was to compile or develop time-series of data for each of the top indicators for each pressure. These time-series were analyzed to determine the current status of each pressure in the CCLME based on short-term and long-term trends of the dataset. We end the “Oceanographic and Climatic” and “Anthropogenic” pressures sections with examples of the linkages between certain drivers and pressures and specific EBM components of the CCLME.

ANTHROPOGENIC DRIVERS AND PRESSURES

As human population size and demand for seafood increases globally and within the CCLME, numerous human activities in the ocean (e.g., fishing and shipping activity) and on land (e.g., pollutants and runoff from agricultural activities) need to be recognized and incorporated into management of marine resources. However, data on the status and trends of these human-related

pressures are often buried in state agency reports, described at small spatial scales and measured inconsistently among local, state and federal entities. Here, we attempted to gather and produce the best available time series data on anthropogenic pressures across the entire CCLME. These data sets are intended to quantify relative changes in anthropogenic pressures and provide the foundation for subsequent integrative analyses of cumulative effects on multiple EBM components (e.g., risk analysis and management strategy evaluations).

We identified 23 anthropogenic pressures on the CCLME, primarily relying on previous work by Halpern et al. (2008, 2009) and Teck et al. (2010). Oceanographic and climatic pressures are addressed in a separate section of the CCIEA (*Section 2: Oceanographic and Climatic Drivers and Pressures*). Anthropogenic pressures ranged in scope from land-based pressures such as inorganic pollution and nutrient input to at-sea pressures such as fisheries removals, commercial shipping and offshore oil and gas activities. The general impacts of pressures on the marine environment have been broadly categorized by Eastwood et al. (2007) and we have summarized anthropogenic pressures for the CCLME into this modified framework (Table AP1). Because these pressures originate from human activities, we should be able to assess current and historic levels, as well as predict future levels of the pressure. Here, we describe how fisheries and non-fisheries related human pressures affect various components of the CCLME, evaluate which indicators are best suited to capture the trends and variability of these pressures and then gather time series data which describe the status and trends of each pressure based on chosen indicators. Indicator evaluation, data indices and sources are summarized in Tables AP2-5.

Table AP1. General ecosystem impacts, types and identified anthropogenic pressures in the CCLME.

| General ecosystem impact | Type | Identified pressures |
|--------------------------|-------------|------------------------------|
| Habitat loss | Smothering | Benthic structures |
| | | Dredging |
| | | Sediment input |
| | Obstruction | Benthic structures |
| | | Coastal engineering |
| | | Ocean mining |
| Habitat modification | Siltation | Freshwater retention |
| | | Sediment input |
| | | Dredging |
| | | Coastal engineering |
| | | Ocean mining |
| | Abrasion | Commercial shipping activity |

| General ecosystem impact | Type | Identified pressures |
|--------------------------|---|-------------------------------|
| Non-physical disturbance | Conversion | Habitat destruction |
| | | Dredging |
| | | Aquaculture |
| | Noise | Commercial shipping activity |
| | | Tourism |
| | Visual | Recreational use |
| | | Light pollution |
| | | Coastal engineering |
| | | Tourism |
| Toxic contamination | Introduction of synthetic compounds | Inorganic pollution |
| | | Atmospheric pollution |
| | | Marine debris |
| | | Ocean-based pollution |
| | Introduction of non-synthetic compounds | Offshore oil and gas activity |
| | | Ocean-based pollution |
| Non-toxic contamination | Nutrient enrichment | Nutrient input |
| | Organic enrichment | Organic pollution |
| | Changes in thermal regime | Power plants |
| | Changes in turbidity | Freshwater retention |
| | | Power plants |
| | | Sediment input |
| | | Dredging |
| | Changes in salinity | Freshwater retention |
| | | Power plants |

| General ecosystem impact | Type | Identified pressures |
|--------------------------|--|-----------------------------|
| Biological disturbance | Introduction of microbial pathogens | Aquaculture |
| | Introduction of non-native species | Invasive species |
| | Translocations or aggregation of individuals | Coastal engineering |
| | | Benthic structures |
| | | Offshore oil & gas activity |
| | | Marine debris |
| | | Ocean mining |
| | Extraction of species | Fisheries removals |
| | | Seafood demand |

*General ecosystem impacts and types based on pressure categories identified in Eastwood (2007).

The ‘status’ of each pressure (see *Data Analysis and Presentation* box) was measured on a short-term basis (increasing, decreasing or the same over the last five years) and measured relative to the historic average of the dataset (higher than, lower than or the same as historic levels). The historical status of each indicator should be placed in context with the amount of data available for each time series. For example, the entire time series for some indicators was only six years while the time series for other indicators was > 50 years. For shorter time series, the mean of the last five years (short-term) was not likely different from the mean of the entire time series; thus, the relative status for indicators with short time series was more related to the availability of data and not actual historic trends. However, many of these indicators were chosen because they were the most fundamentally sound datasets and will continue to be measured over time, providing meaningful historic comparisons in future iterations of the IEA.

In this section of the CCIEA, we do not provide interpretation of the status and trends of each pressure because this may vary depending on the EBM component of interest. For example, a decreasing trend in fisheries removals may be “good” for rebuilding stocks of Protected Resources or it could be “bad” for Vibrant Coastal Communities. The interpretation of pressures’ effects on various EBM components will be presented in analyses in the “risk” sections for each EBM component (*Section 3: Status, trends and risk of key ecosystem components in the CCLME*) and in the management strategy evaluations (*Section 4: Management Testing and Scenarios for the California Current*). The pressures identified in this section were selected primarily for their relevance to the non-human components of the CCLME (i.e. Protected Resources, Wild Fisheries, Ecosystem Integrity and Habitat), but some also contain relevant information for Vibrant Coastal Communities. Specific socio-economic indicators for Vibrant Coastal Communities have begun to be developed and can be found in *Section 3: Resilient and Economically Viable Coastal Communities*.

Two goals of future iterations of the CCIEA will be to (1) evaluate the ‘status’ of a pressure relative to specific target levels for each indicator, and (2) develop a temporal index of the cumulative effects of all anthropogenic pressures. Establishing specific target levels of a pressure (e.g., fisheries landings quotas or concentration of nitrogen in coastal waters) is a critical step in the management and policy planning process (Samhoury et al. 2012). Placing the current status of an indicator into context with historic levels or with management goals allows managers to determine whether the current status and trend of a specific pressure is moving in the right direction or whether alternative management strategies are necessary. Target levels have been established for many of these pressures in general terms (Halpern et al. 2012), and we will refine these values specifically for the CCLME.

Importantly, the pressures identified below do not act upon the ecosystem individually, but collectively. Pressures from terrestrial-based pollution, shipping, offshore energy development, fisheries and coastal development exert cumulative effects on the ecosystem and should be managed in a holistic way (Vinebrooke et al. 2004, Crain et al. 2008, Halpern et al. 2008, Curtin and Prellezo 2010, Stelzenmüller et al.

DATA ANALYSIS AND PRESENTATION

The status of each indicator was evaluated against two criteria: recent short-term trend and status relative to the long-term mean—reported as short-term status and long-term status, respectively.

Short-term trend. An indicator was considered to have changed in the short-term if the trend over the last five years of the time series showed an increase or decrease of more than 1.0 standard deviation (SD) of the mean of the entire time series.

Status relative to the long-term mean. An indicator was considered to be above or below historical norms if the mean of the last five years of the time series differs from the mean of the full time series by more than 1.0 SD of the full time series.

Time series figures. Time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 SD (solid line) of the full time series. The shaded green area is the last five years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the modeled trend over the last 5 years increased (\nearrow) or decreased (\searrow) by more than 1.0 SD, or was within 1.0 SD (\leftrightarrow) of the long-term trend. The lower symbol indicates whether the mean of the last five years was greater than (+), less (-), or within (\bullet) 1.0 SD of the long-term mean.

2010). However, quantifying the cumulative effects of these pressures is a difficult task primarily because our understanding of whether effects are additive, synergistic or antagonistic is relatively poor (Darling and Côté 2008, Hoegh-Guldberg and Bruno 2010). There are three general frameworks for quantifying the cumulative effects of pressures on ecosystem structure and function that we will rely on to develop a temporal index of cumulative impacts for the CCLME (Link et al. 2002, Halpern et al. 2009, Stelzenmüller et al. 2010). Two of the primary issues we will consider are how to apply weightings to the relative importance of each pressure to the CCLME and how to incorporate time series data that vary in duration and time period. Weightings for most of the pressures have been developed from expert opinion surveys (Teck et al. 2010, Halpern et al. 2012) and should only require limited refinement for our purposes, while incorporating time series of different durations and time periods may result in sub-sampling many of the time series.

FISHERIES PRESSURES

Fishing provides important services to society, including production of food, employment, livelihood and recreation. At the same time, fisheries have potential to adversely affect the ecosystem that supports them. Impacts of fisheries on ecosystems have been extensively discussed in the literature (Dayton et al. 1995, Kaiser and Spencer 1996, Goni 1998, Agardy 2000, Garcia et al. 2003, Gislason 2003, Pauly and Watson 2009) with major effects associated with fishery removals and destruction of habitats in which fishing occurs. Below, we discuss these two major pressures (fishery removals and habitat destruction) and illustrate their potential impacts to various components of the CCLME.

FISHERY REMOVALS

BACKGROUND

Fishery removals directly impact target resources by reducing their abundance. When poorly managed, fisheries can develop excessive pressure on fishery stocks, leading to overfishing, and causing major ecological, economic and social consequences. Fisheries for the Pacific ocean perch and widow rockfish are among the most notable examples of overexploitation in the CCLME. Fishery targeting Pacific ocean perch developed in the Northern California Current Ecosystem in the 1950s, and catches quickly grew from just over 1000 metric tons in 1951 to almost 19,000 metric tons in 1966, reducing the stock below the overfished threshold of 25% of unfished biomass, established by the Pacific Fishery Management Council, in 1980 (Hamel and Ono 2011). Fisheries targeting widow rockfish developed in the late 1970s, after it was discovered that the species forms aggregations in the pelagic waters at night. Widow rockfish catches sharply increased from 1,107 tons in 1978 to 28,419 tons in 1981 and started to drop, indicating reduction in the resource, so that severe catch limits were imposed in 1982 (Love et al. 2002).

Fisheries are rarely selective enough to remove only the desired targets (Garcia et al. 2003), and they often take other species incidentally, along with targets. Even though incidentally taken fish (often referred to as bycatch) are routinely discarded, discard mortality can be quite high, especially for deep-water species. Therefore, fisheries can significantly reduce abundance of bycatch species associated with removals of targeted resources as well. Unintended removals can be also be facilitated by lost (or dumped) fishing gear, particularly pots, traps and gillnets, which may cause entanglement of fish, marine mammals, turtles and sea birds. The extent of such “ghost” fishing in the

CCLME is unknown, but studies conducted elsewhere suggest that the impact might be non-trivial (Fowler 1987, Goni 1998, Garcia et al. 2003).

Fisheries typically target larger individuals. By removing particular size groups from a population, fisheries can alter size and age structure of targeted and bycatch stocks, their sex ratios (especially when organisms in a population exhibit sexual dimorphism in growth or distribution), spawning potential, and life history parameters related to growth, sexual maturity and other traits.

Extensive fishery removals may also affect large scale ecosystem processes and cause changes in species composition and biodiversity. These can occur with gradual decrease in the average trophic level of the food web, caused by reduction in larger, high trophic level (and high value) fish and increase in harvest of smaller, lower trophic level species, a process described as “fishing down the food chain” (Pauly et al. 1998, Pauly and Watson 2009). The extensive removal of forage fish species, mid trophic level components, can also modify interactions within a trophic web, alter the flows of biomass and energy through the ecosystem, and make systems less resilient to environmental fluctuations through a reduction of the number of prey species available to top predators (Garcia et al. 2003, Pauly and Watson 2009).

EVALUATION AND SELECTION OF INDICATORS

Fishery removals consist of two components: retained catch that is subsequently landed to ports (landings) and discarded catch that is thrown overboard. When discarded, fish either survive or die depending upon the characteristics of species and fishing and handling practices employed by the fishery. Thus, the total removals are the sum of landings and dead discard.

The best source for information on stock-specific fishery removals is typically stock assessments that report landings, estimate amount of discard, and evaluate discard mortality. Stock assessments also provide the longest time series of removals, commonly dating back to the beginning of exploitation. Stock assessments conducted for CCLME species are available via Pacific Fishery Management Council website (<http://www.pcouncil.org>) by species and year of assessment. However, only some species from each fishery have been assessed. For non-assessed stocks, information on fishery removals can be obtained from a variety of state and federal sources. The most detailed and reliable CCLME fishery landing data are summarized in the Pacific Fisheries Information Network (PacFIN) (<http://pacfin.psmfc.org>), a regional fisheries database that manages fishery-dependent information in cooperation with the National Marine Fisheries Service (NMFS) and West Coast state agencies. The data in PacFIN go back to 1981. NMFS and its predecessor agencies, the U.S. Fish Commission and Bureau of Commercial Fisheries, has also been reporting fishery landing statistics collected via comprehensive surveys of all U.S. coastal states conducted since 1951. These data are available via NMFS Science and Technology website at (<http://www.st.nmfs.noaa.gov/st1/commercial/index.html>). Recreational catches since the late 1970's can be found in the Recreation Fisheries Information Network (RecFIN) (<http://www.recfin.org>), a project of the Pacific States Marine Fisheries Commission.

There have been a few historical studies conducted to evaluate discard in CCLME fisheries (Pikitch et al. 1988, Sampson 2002), but those studies focused on specific areas and/or species groups, so that thorough analysis would be needed to extrapolate those estimates to other areas, species and years. Currently there are two observer programs operated by the NMFS NWFSC on the U.S. West Coast. These programs include the At-Sea Hake Observer Program (A-SHOP), which monitors the at-sea hake processing vessels, and the West Coast Groundfish Observer Program

(WCGOP), which monitors catcher vessels that deliver their catch to a shore-based processor or a mothership. The A-SHOP dates back to the 1970s, while WCGOP was implemented in 2001. The WCGOP began with gathering data for the limited entry trawl and fixed gear fleets. Observer coverage has expanded to include the California halibut trawl fishery, the nearshore fixed gear and pink shrimp trawl fishery. Since 2011, the U.S. West Coast groundfish trawl fishery has been managed under a new groundfish catch share program. The WCGOP provides 100% at-sea observer monitoring of catch for the new, catch share based Individual Fishing Quota (IFQ) fishery, including both retained and discarded catch.

Since 2005, the WCGOP has been generating estimates of the groundfish total mortality from commercial, recreational and research sources including incidental catch from non-groundfish fisheries. For groundfish, WCGOP total fishing mortality estimates were selected as an indicator of fishery removal recognizing that the data to inform this indicator is only available for the most recent years. For other species groups, the PacFIN landings were selected as the best long-term fishery removal indicator, since they represent the bulk of removals for most species and have been routinely reported. However, if available, total mortality estimates would be the preferred indicator for all species groups, due to its higher evaluation in the “Primary considerations” criteria (Table AP2).

STATUS AND TRENDS

The status of total removals was measured using: 1) commercial landings, by species group (groundfish, coastal pelagic species, highly migratory species, salmon, crab, shrimp, shellfish and others) and gear (trawl, shrimp trawl, hook and line, net gear, pot and trap, troll, and other miscellaneous gear), as reported by the Pacific Fisheries Information Network (PacFIN) at <http://pacfin.psmfc.org> for Washington, Oregon and California, and 2) for groundfish, total mortality estimates generated and provided by the West Coast Groundfish Observer Program (WCGOP; Table AP3).

Commercial landings – This indicator represents commercial landings from shoreside commercial fisheries. It also includes tribal removals and catches from exempted fishing permit studies. Commercial landings represent the bulk of fishery removals for highly priced, high retention rate species, but not for bycatch species that are often discarded when caught. Status and trends of this indicator, therefore, may not thoroughly represent changes in fishery removals, and will also reflect changes in markets or/and management measures employed by the Pacific Fishery Management Council and NMFS to prevent overfishing.

Figures AP1-AP9 and AP11-AP17 show the time-series of commercial landings by different species groups in the CCLME and by gear types. Figures AP10 and AP18 represent short- versus long-term trends in landings by species groups and gear, respectively. Landings of Pacific hake are reported separately from other groundfish species, since the Pacific hake fishery is the largest (in weight) on the U.S. West Coast, and when combined with other species, hake overwhelms the landings of the entire group, and obscures interannual changes in catch of other groundfish species.

Since 1981, commercial landings of groundfish species (other than Pacific hake) and salmon decreased, in part due to management measures (Figs. AP1, AP5). Pacific hake, coastal pelagic species and crab exhibited a positive long-term trend in landings (Figs. AP2, AP3, AP6), although for Pacific hake and crab, the mean of the last five years was greater than one standard deviation of the mean for the entire time series (Figs. AP1, AP6). Highly migratory species, shrimp and shellfish landings

did not change significantly over the last 40 years, apart from the peak reported for both groups in the early 1980s (Figs. AP4, AP7, AP8). Relative to the mean of the entire time series, landings of coastal pelagic species have been higher over the last five years but not increasing, crab and shrimp landings have been increasing but are still within historic levels, and landings for salmon and groundfish excluding hake have been at constant low levels over the last five years. All other species groups have been relatively constant within historic landing levels (Fig. AP10).

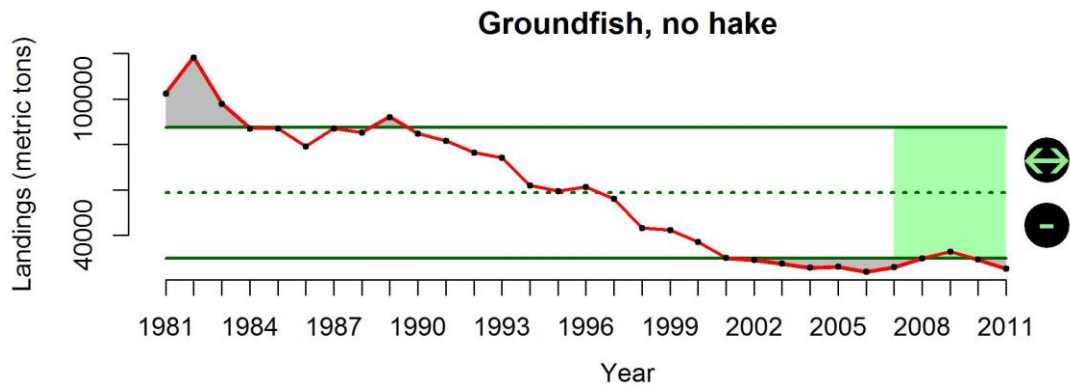


Figure AP1. Annual landings of groundfish in the CCLME from 1981 – 2011 (Pacific hake *Merluccius productus* excluded).

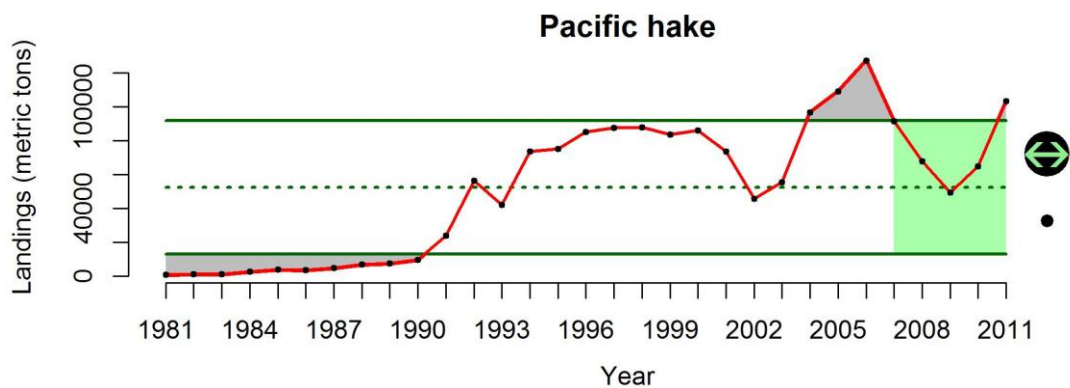


Figure AP2. Annual landings of Pacific hake *Merluccius productus* in the CCLME from 1981 – 2011.

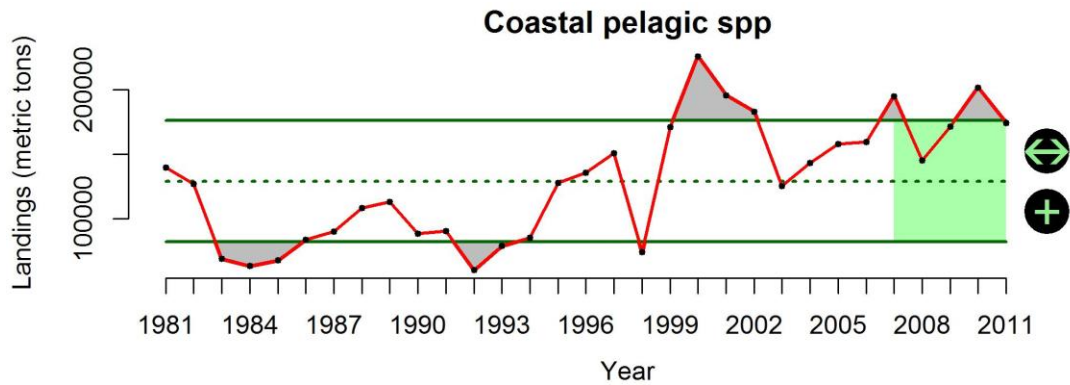


Figure AP3. Annual landings of coastal pelagic species (CPS) in the CCLME from 1981 – 2011. CPS include Pacific sardine *Sardinops sagax*, Pacific mackerel *Scomber japonicus*, northern anchovy *Engraulis mordax*, jack mackerel *Trachurus symmetricus*, and market squid *Loligo opalescens*).

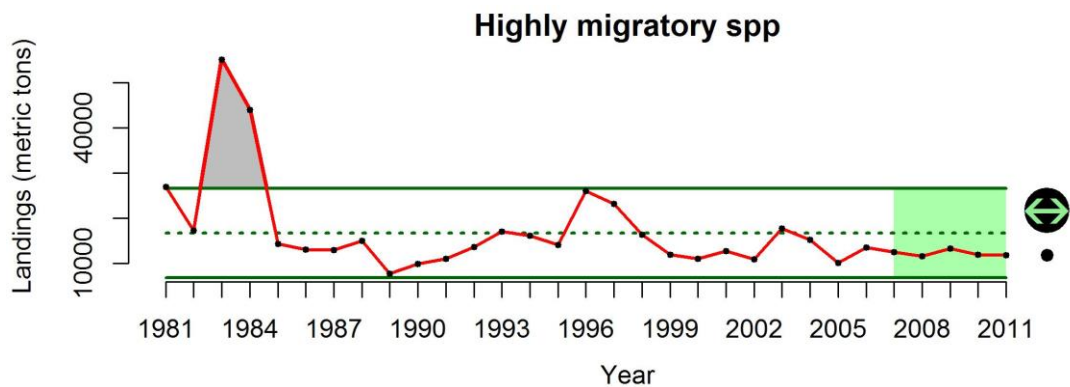


Figure AP4. Annual landings of highly migratory species (HMS) in the CCLME from 1981 – 2011. HMS include tunas, sharks, billfish/swordfish and dorado *Coryphaena hippurus*.

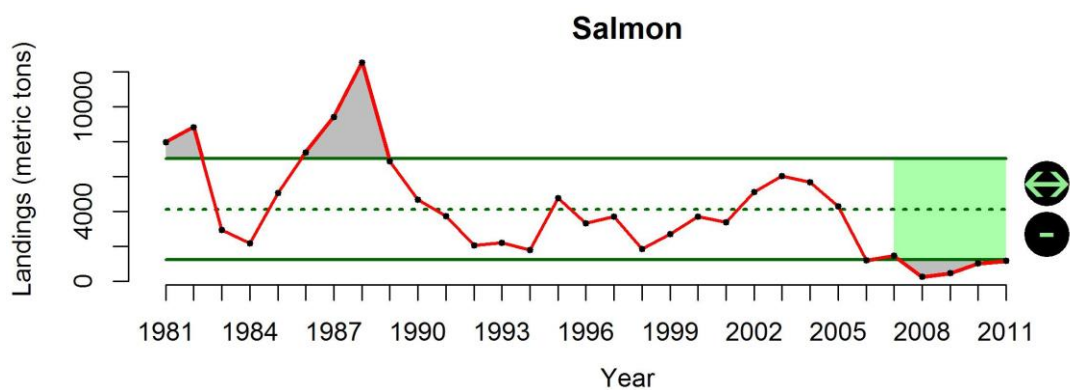


Figure AP5. Annual landings of salmon in the CCLME from 1981 – 2011.

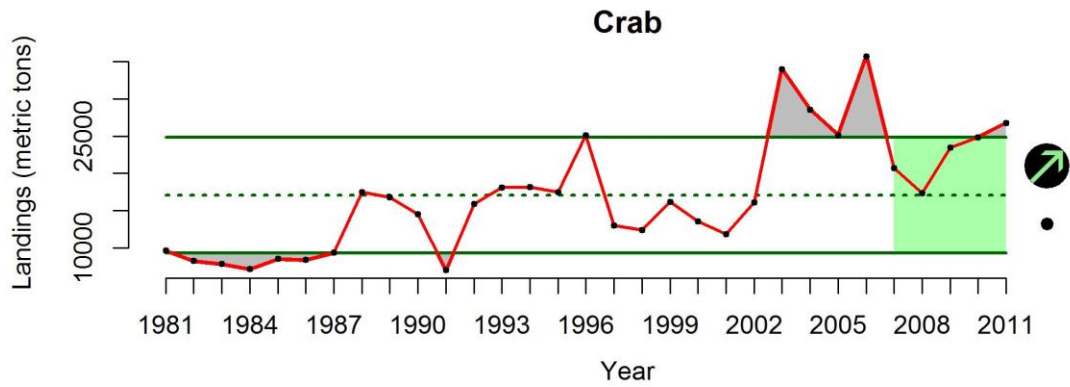


Figure AP6. Annual landings of crab in the CCLME from 1981 – 2011.

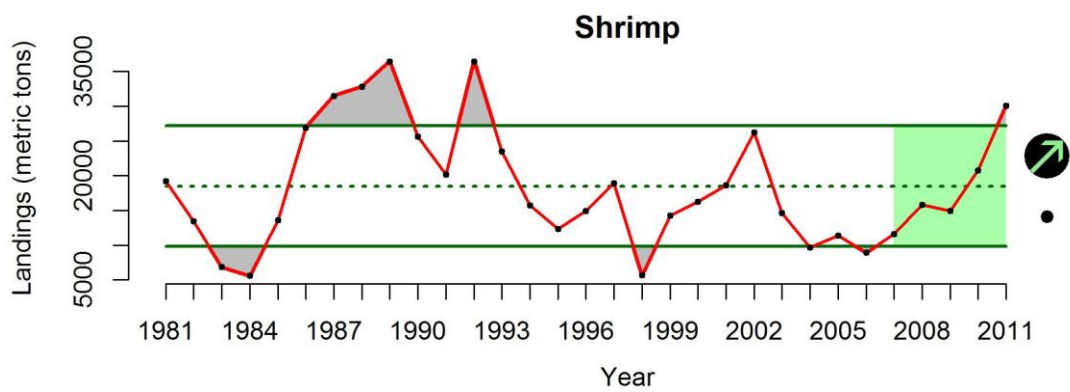


Figure AP7. Annual landings of shrimp in the CCLME from 1981 – 2011.

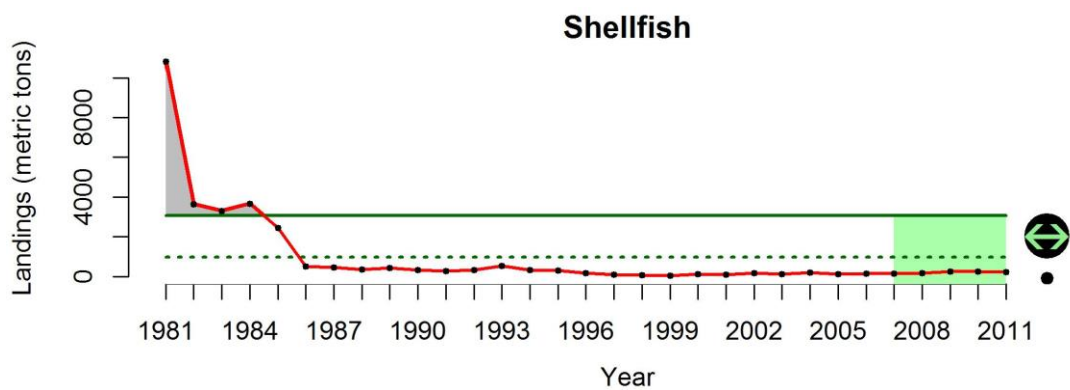


Figure AP8. Annual landings of shellfish in the CCLME from 1981 – 2011.

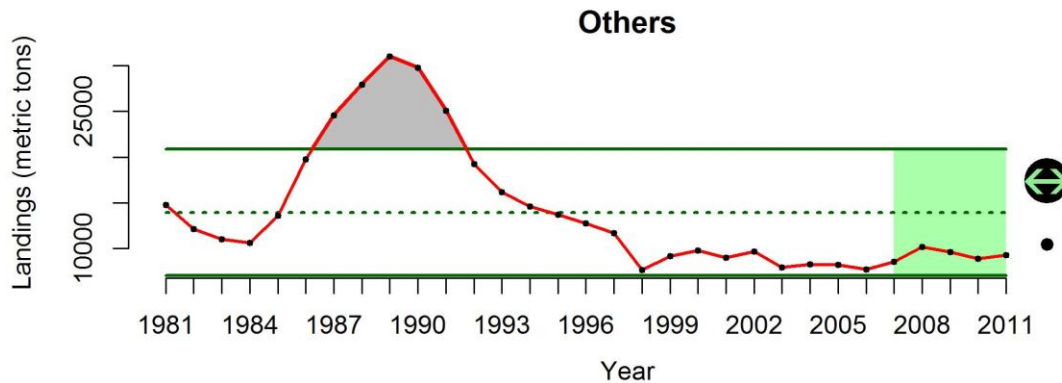


Figure AP9. Annual landings of all other species in the CCLME from 1981 – 2011.

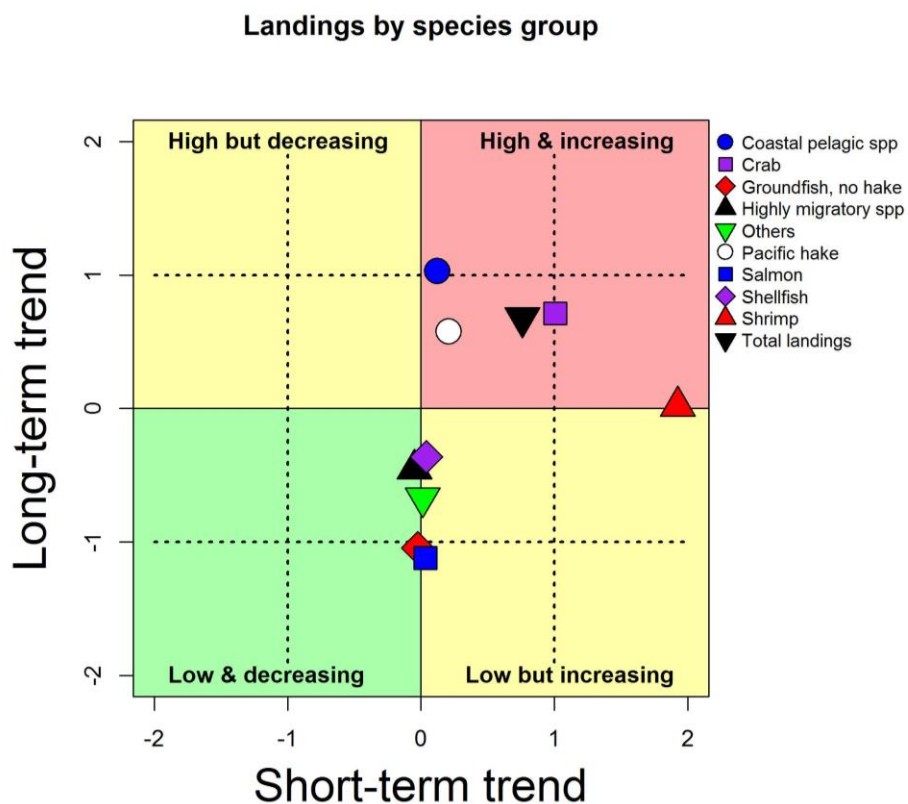


Figure AP10. Short- and long-term status of annual landings (1981 – 2011) by species groups in the CCLME. Prior to plotting, time series data for each indicator were normalized to place them on the same scale. The short-term trend indicates whether landings increased, decreased or remained the same over the last five years. The long-term trend represents the difference between the mean of the last 5 years and the mean of the full time series. The dotted lines represent ± 1.0 SD; thus, data points outside the dotted lines are considered to be increasing or decreasing over the short term or the current status is lower or higher than the long-term mean of the time series.

Landings made by all gear types varied considerably over the last 40 years (Figs. AP11 – AP17), but only hook-and-line landings (Fig. AP13) exhibited a decreasing trend since 1981. Over the

last five years, shrimp trawl, pot and trap landings increased (Figs. AP12, AP15), while landings made by other gear types did not exhibit clear trends. If compared to the mean of the entire time series, shrimp trawls and pot and trap landings have been increasing over the last five years but within historic levels, while hook-and-line landings have been at consistently lower than average levels over the last five years. All other gear types have been relatively constant within historic landing levels (Fig. AP18).

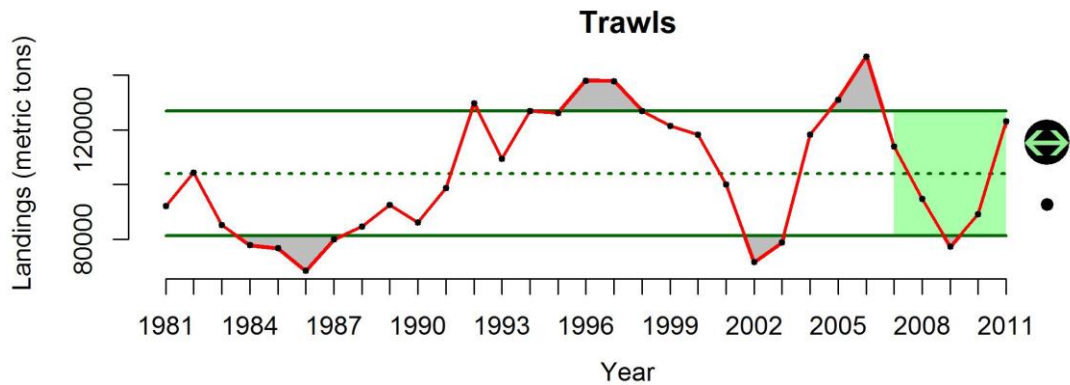


Figure AP11. Annual commercial trawl landings in the CCLME from 1981 – 2011.

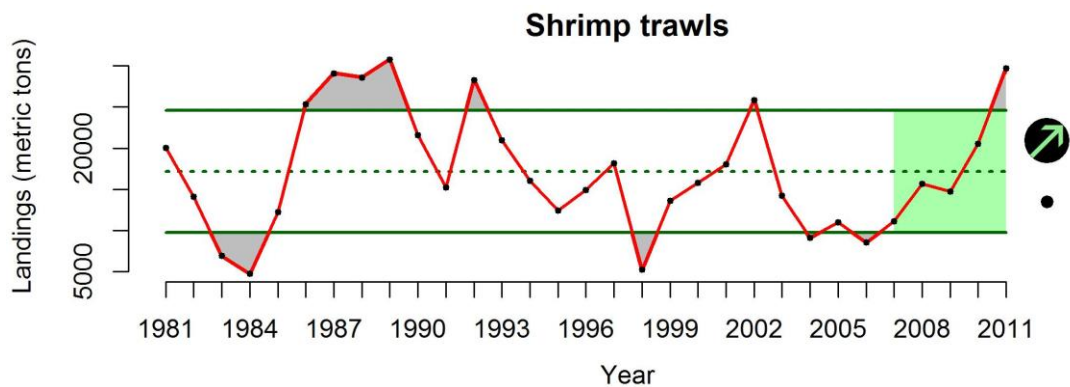


Figure AP12. Annual commercial shrimp trawl landings in the CCLME from 1981 – 2011.

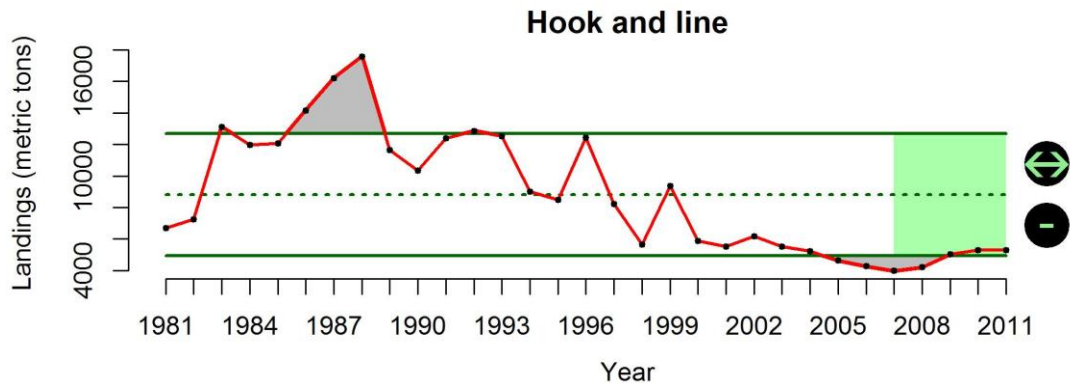


Figure AP13. Annual hook-and-line landings in the CCLME from 1981 – 2011.

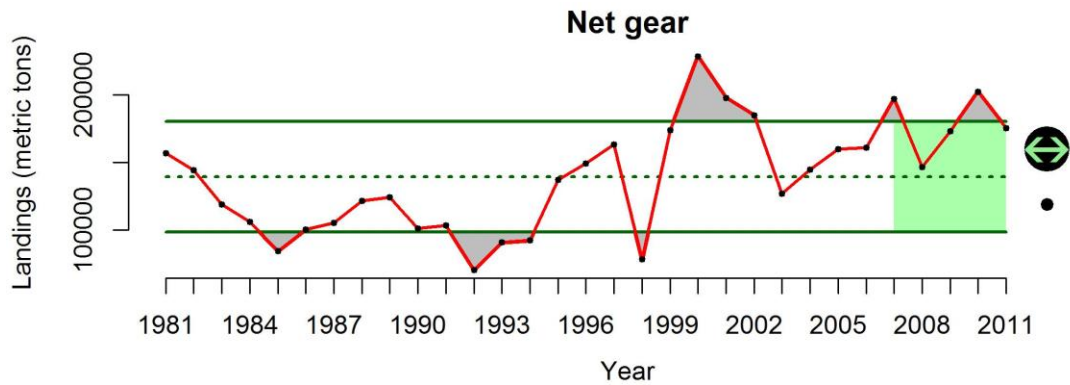


Figure AP14. Annual net-gear landings in the CCLME from 1981 – 2011.

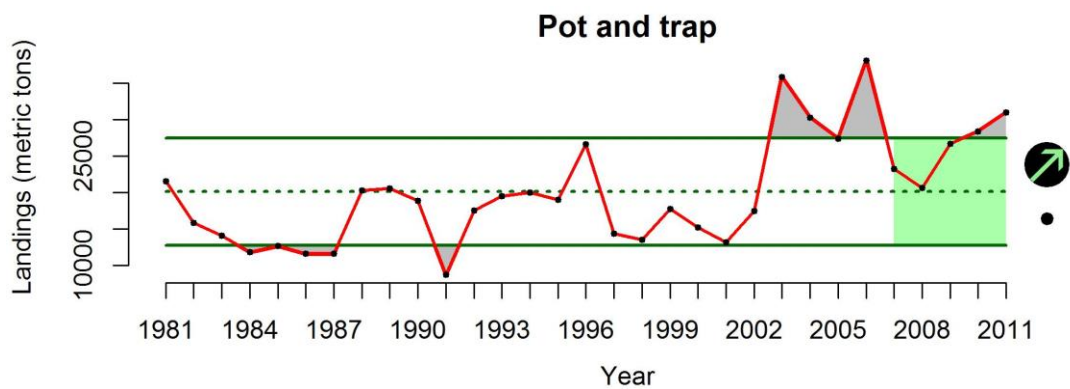


Figure AP15. Annual pot and trap landings in the CCLME from 1981 – 2011.

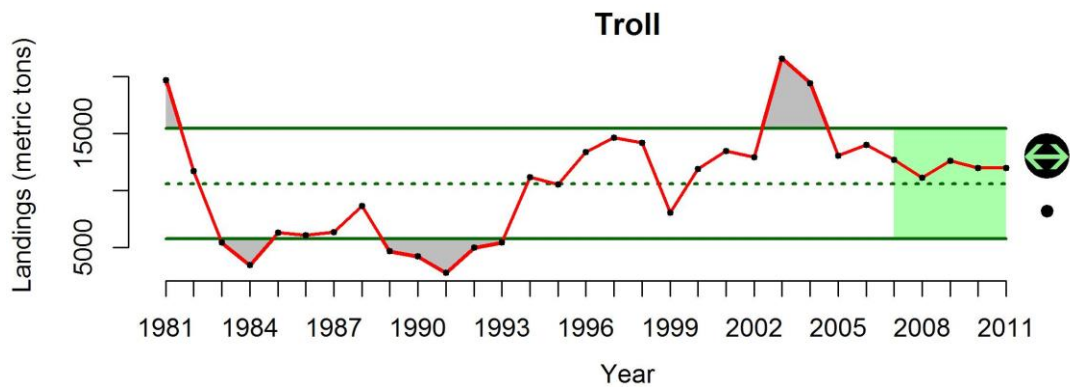


Figure AP16. Annual troll-caught landings in the CCLME from 1981 – 2011.

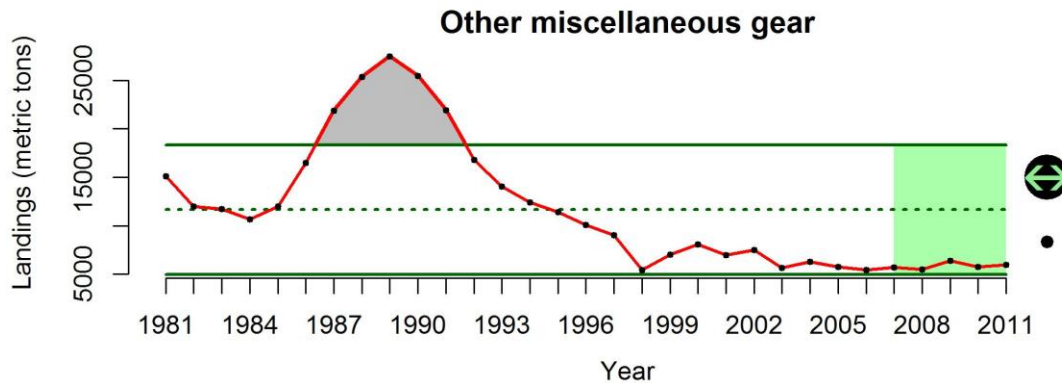
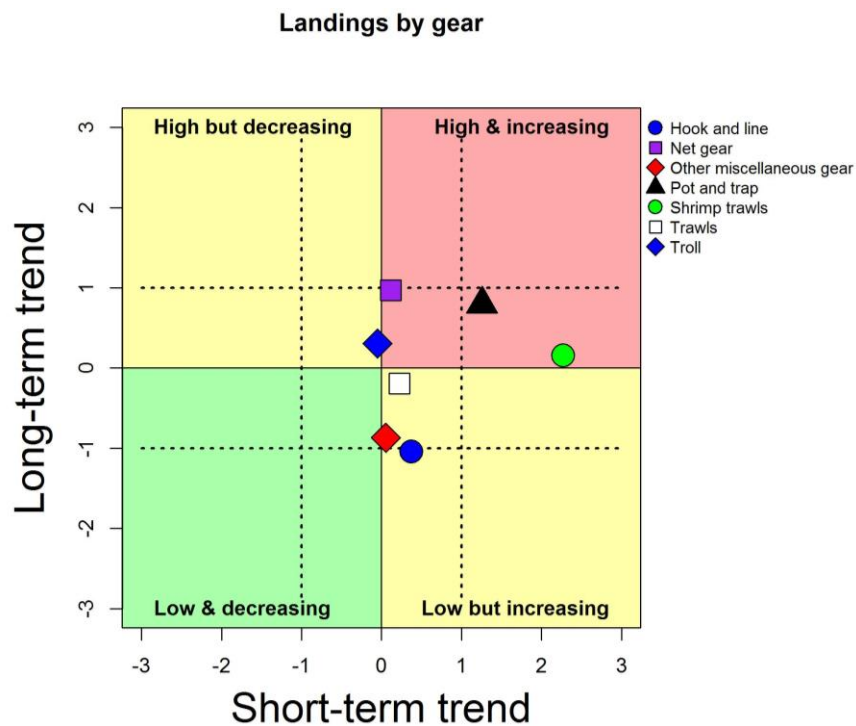


Figure AP17. Annual landings of all other miscellaneous gear in the CCLME from 1981 – 2011.



Figures AP18. Short- and long-term status of annual landings (1981 – 2011) by gear type in the CCLME. Prior to plotting, time series data for each indicator were normalized to place them on the same scale. The short-term trend indicates whether landings increased, decreased or remained the same over the last five years. The long-term trend represents the difference between the mean of the last 5 years and the mean of the full time series. The dotted lines represent ± 1.0 SD; thus, data points outside the dotted lines are considered to be increasing or decreasing over the short term or the current status is lower or higher than the long-term mean of the time series.

Total fishing mortality estimates (groundfish only)– This indicator represents the total removals of groundfish species from a suite of fishery-dependent and fishery-independent sources, including shoreside commercial fisheries and at-sea hake removals, tribal and recreational catches, as well as incidental catch of groundfish in non-groundfish fisheries. It also includes removals from

the research surveys conducted within the CCLME. As in the case of groundfish landings, total mortality estimates of Pacific hake are reported separately. The Pacific hake fishery is the largest (in weight) on the U.S. West Coast, and, when combined with other species, total mortality of the Pacific hake overwhelms the total mortality for the entire group, and obscures changes in catch of other groundfish species. Over the last 5 years, groundfish total mortality estimates for groundfish species increased (Fig. AP19), while those of Pacific hake decreased (Fig. AP20). This is also evident in Fig. AP21, which compares short- versus long-term trends in total mortality estimates for Pacific hake and other groundfish species.

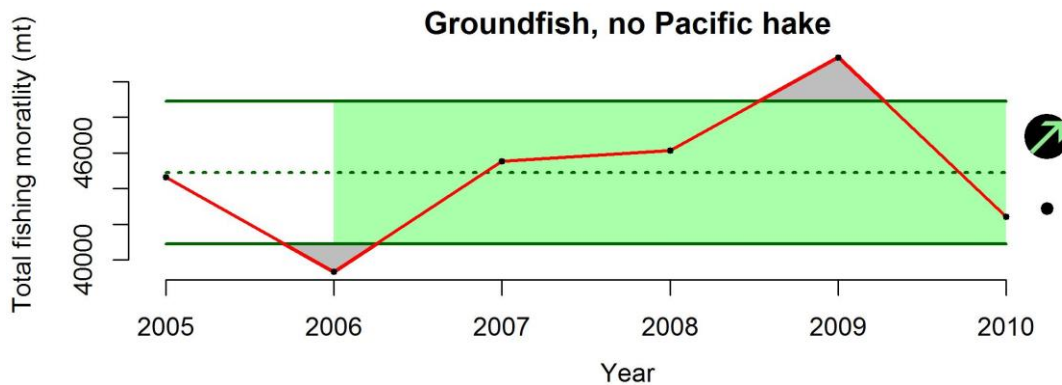


Figure AP19. Total mortality estimates of groundfish (Pacific hake *Merluccius productus* excluded) in the CCLME from 2005 - 2010.

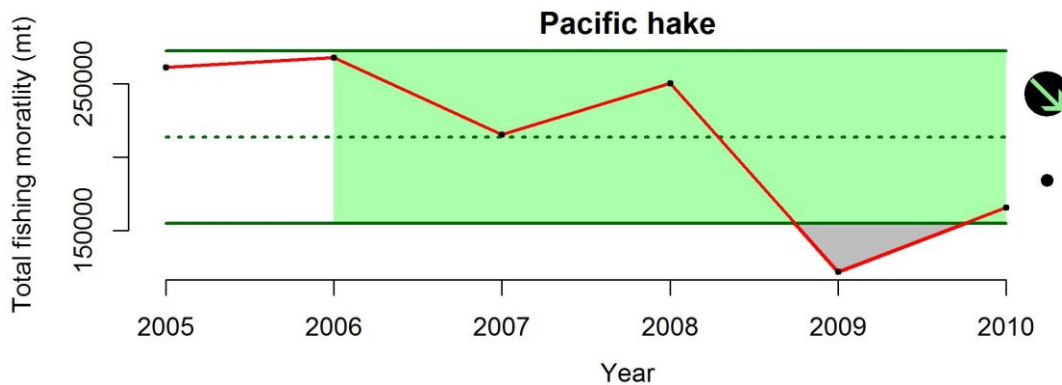


Figure AP20. Total mortality estimates of Pacific hake *Merluccius productus* in the CCLME from 2005 - 2010.

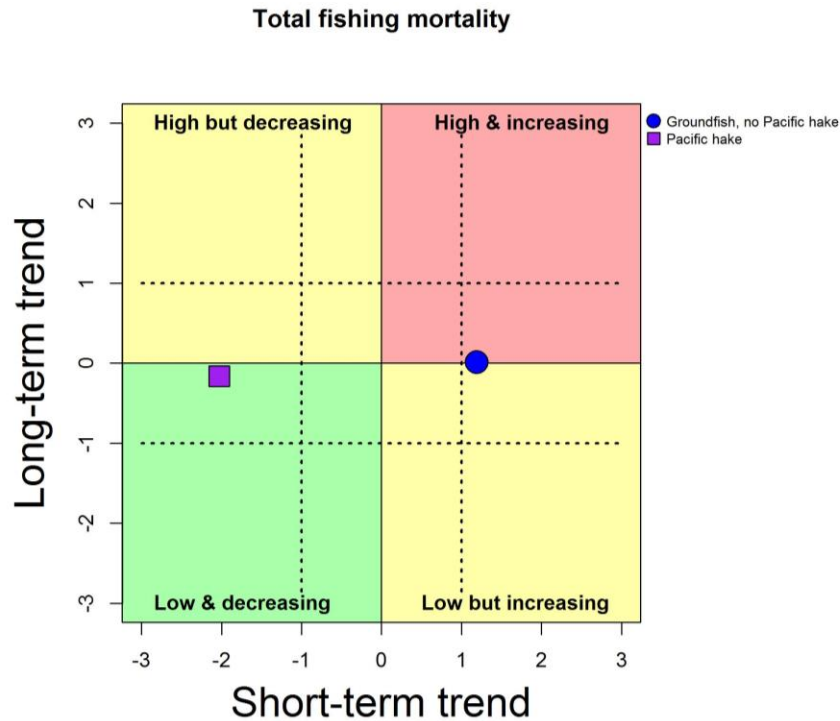


Figure AP21. Short- and long-term status of annual total fishing mortality (2005 – 2010) by species groups in the CCLME. Prior to plotting, time series data for each indicator were normalized to place them on the same scale. The short-term trend indicates whether total fishing mortality increased, decreased or remained the same over the last five years. The long-term trend represents the difference between the mean of the last 5 years and the mean of the full time series. The dotted lines represent ± 1.0 SD; thus, data points outside the dotted lines are considered to be increasing or decreasing over the short term or the current status is lower or higher than the long-term mean of the time series.

HABITAT DESTRUCTION

BACKGROUND

Fishing can alter benthic habitats by disturbing and destroying bottom topography and associated communities, from the intense use of trawls and other bottom gear (Kaiser and Spencer 1996, Hiddink et al. 2006). Habitat destruction, in turn, can lead to extirpation of vulnerable benthic species and disruption of food web processes (Hall 1999, Hiddink et al. 2006). The effect is particularly dramatic when those gears are used in sensitive environments with sea grass, algal beds, and coral reefs, and is less evident on soft bottoms (Garcia et al. 2003). However, fisheries often tend to operate within certain areas more than others (Kaiser et al. 1998), and long-term impacts of trawling may cause negative changes in biomass and the production of benthic communities in any habitat type, to various degrees (Hiddink et al. 2006).

In the CCLME, implementation of Essential Fish Habitats (EFH), areas necessary for fish spawning, breeding, feeding, or growth to maturity, and Marine Protected Areas (MPA), in

combination with gear regulation measures, have been used to reduce adverse impact of fisheries on vulnerable habitats. Also, the introduction of the Cowcod Conservation Area (CCA) and Rockfish Conservation Areas (RCAs) as management measures to prevent overfishing makes additional areas along the coast inaccessible to fishing during some or all of the year.

EVALUATION AND SELECTION OF INDICATORS

Habitat destruction could be expressed using a metric such as distance trawled by certain gear types, in certain habitat types. Development of such a metric, however, is non-trivial and requires a thorough analysis, since the destructive capacity of different trawl gear varies according to habitat/bottom type in which it is used. Such an analysis would also require very detailed habitat data that are currently unavailable.

Bellman and Heppell (2007) estimated distance trawled within the limited entry groundfish trawl fishery in the U.S. West Coast by habitat type, defined based on type of bottom substrate. The habitat types considered were of four basic categories, including shelf, slope, basin and ridge, and two subcategories, rocky and sedimentary. Logbook data was used to obtain information on vessel, date, time and location of each individual tow as well as gear used (Bellman and Heppell 2007). These data were then overlaid with GIS seafloor habitat maps off Washington, Oregon and California compiled by Goldfinger et al. (2003), Romsos (2004) and Green & Bizzarro (2003).

We used estimates of coast-wide distance trawled (1999 – 2004) as an indicator for habitat destruction (Table AP3; Bellman and Heppell 2007). Currently, NOAA's Northwest Fisheries Science Center is in the process of producing improved GIS seafloor habitat maps of the CCLME to better define and describe Essential Fish Habitats (EFH). These GIS maps along with logbook, observer and trawl tracks from vessel monitoring system data will be used to improve and further extend time series of the estimated distance trawled.

STATUS AND TRENDS

The status and trends of habitat destruction was measured using distance trawled by habitat type, made by the limited entry groundfish trawl fishery, as estimated by Bellman and Heppell (2007). Overall, distance trawled declined coast-wide between 1999 and 2004 (Fig. AP22). During this period, the majority of trawling occurred in sedimentary continental slope habitat (Fig. AP23), followed by the sedimentary continental shelf (Figure AP24). A shift in trawling effort between habitat types was observed during this period (Figs. AP23-AP29), which in part corresponded to depth-related spatial closures implemented by the Pacific Fishery Management Council to reduce fisheries' impact on depleted species (Bellman and Heppell 2007). If compared to the mean for the entire time series, the distance trawled for all sedimentary habitats, except on the shelf, has been decreasing over the last five years but still within historic levels, while distance trawled in rocky basin habitat has been increasing over the last five years but still within historic levels (Fig. AP31).

The time series of this indicator will soon be extended as the work on Essential Fish Habitat (EFH) progresses; this work involves simultaneous analysis of the most recent GIS sea floor habitat maps, logbook, observer and trawl tracks from vessel monitoring system data.

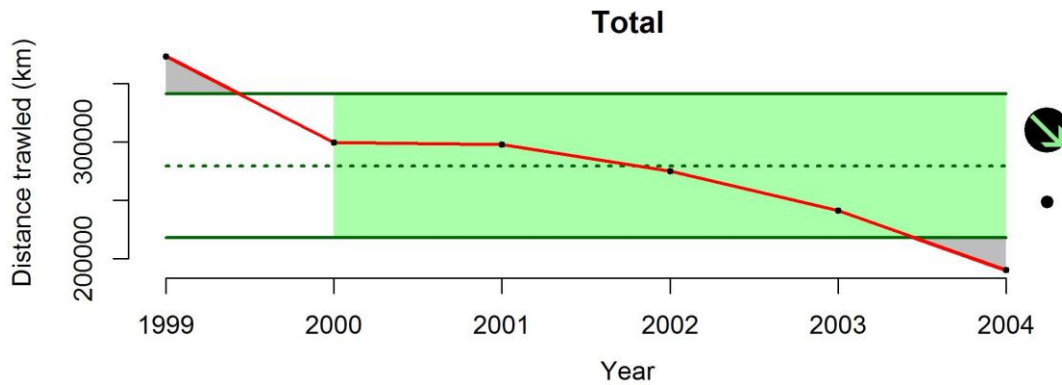


Figure AP22. Total distance trawled (km) along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels.

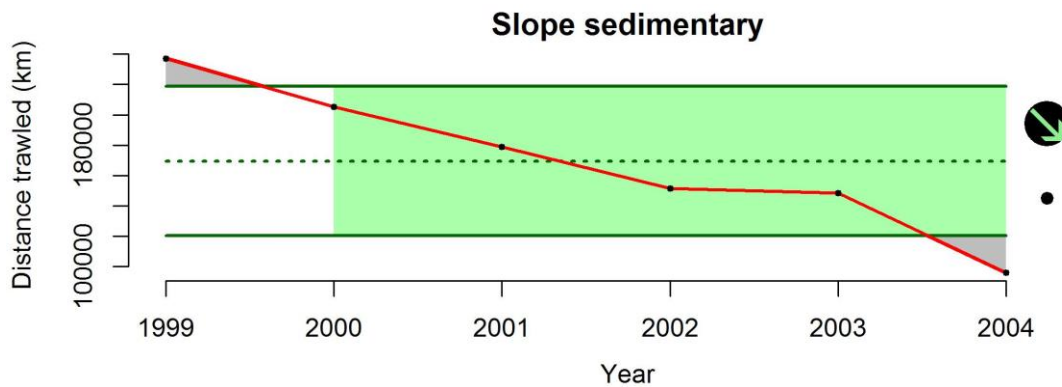


Figure AP23. Distance trawled (km) within slope sedimentary habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels.

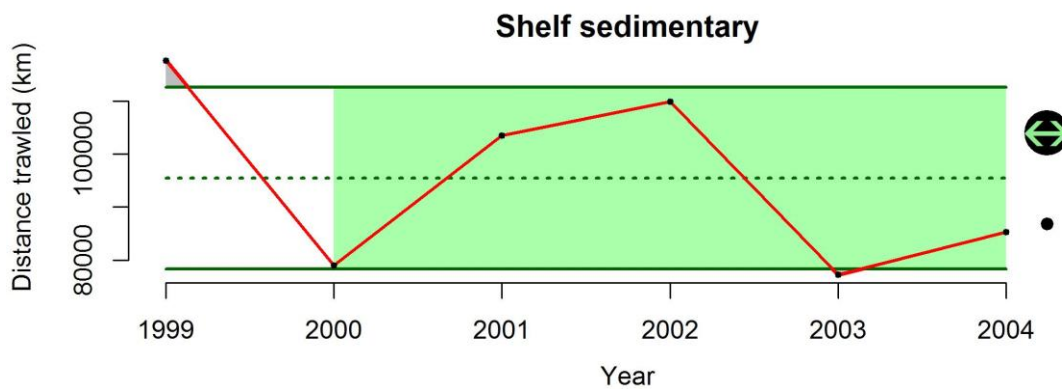


Figure AP24. Distance trawled (km) within shelf sedimentary habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels.

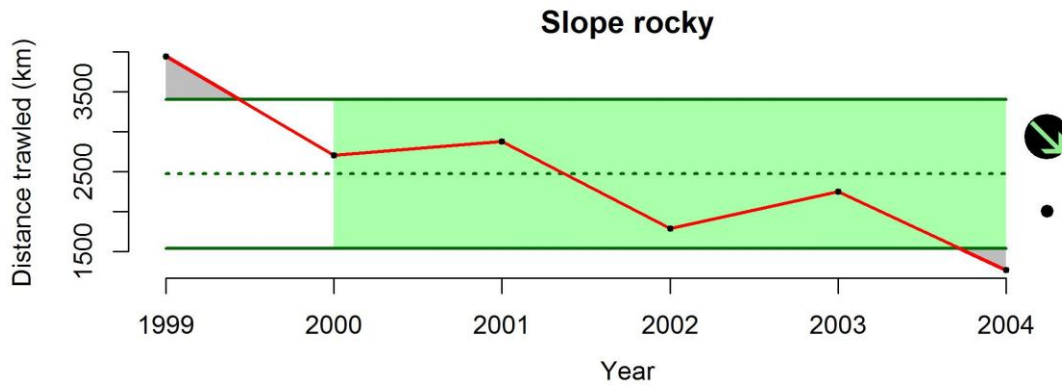


Figure AP25. Distance trawled (km) within slope rocky habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels.

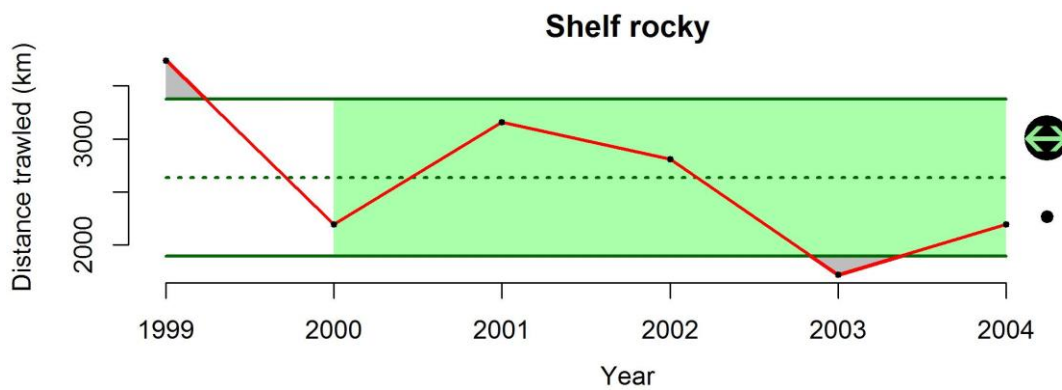


Figure AP26. Distance trawled (km) within shelf rocky habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels.

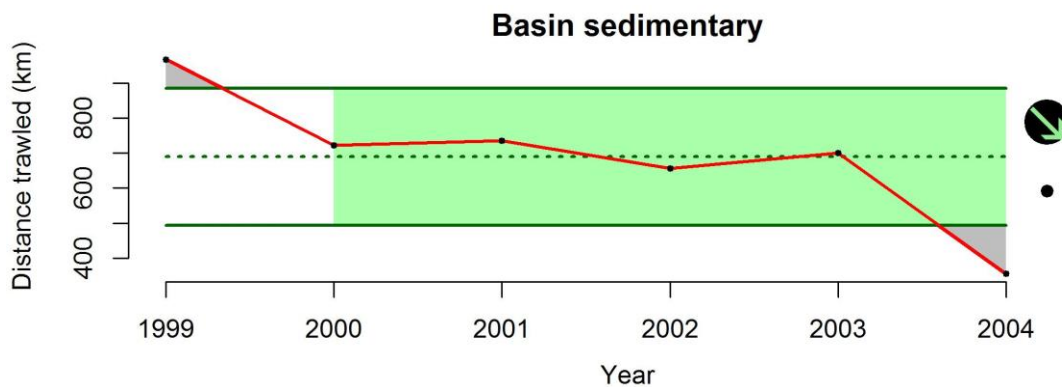


Figure AP27. Distance trawled (km) within basin sedimentary habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels.

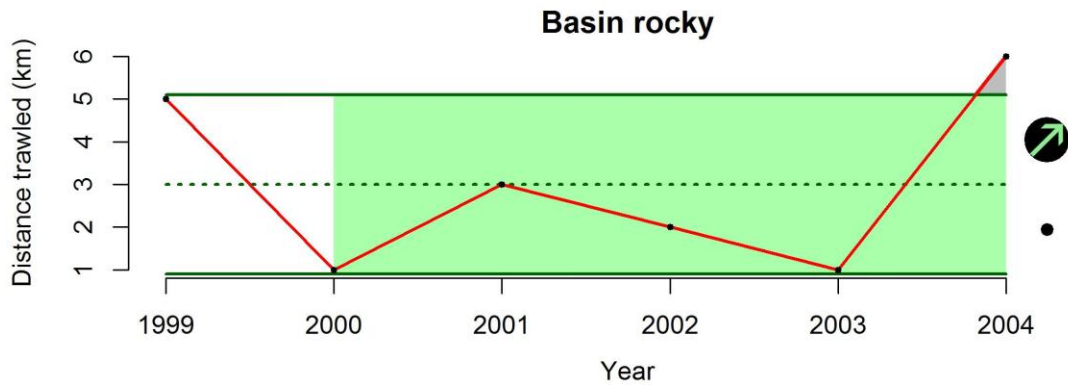


Figure AP28. Distance trawled (km) within basin rocky habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels.

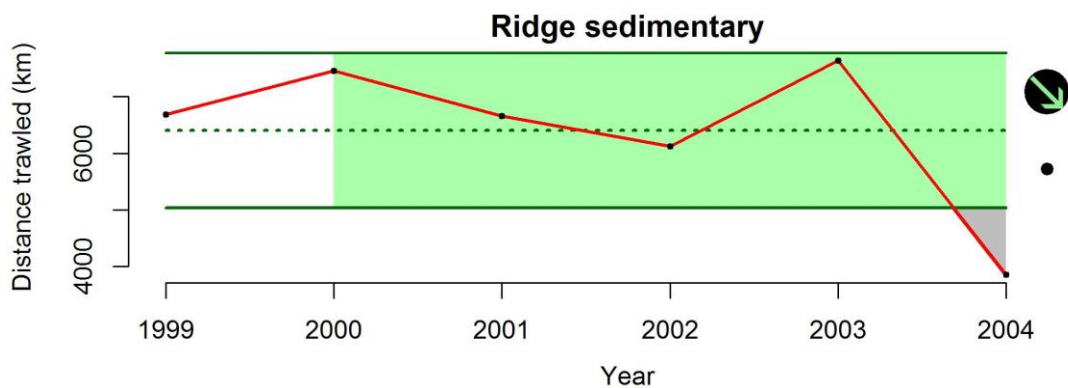


Figure AP29. Distance trawled (km) within ridge sedimentary habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels.

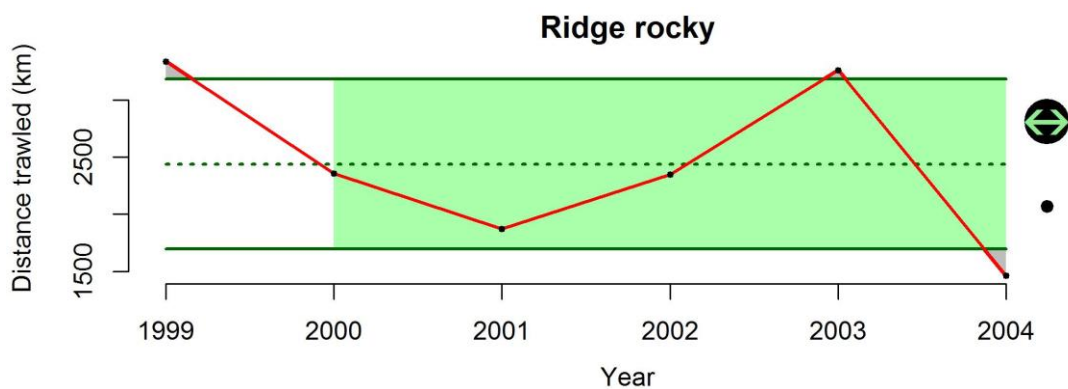


Figure AP30. Distance trawled (km) within ridge rocky habitats along the coast of Washington, Oregon and California by limited-entry groundfish trawl fishery vessels.

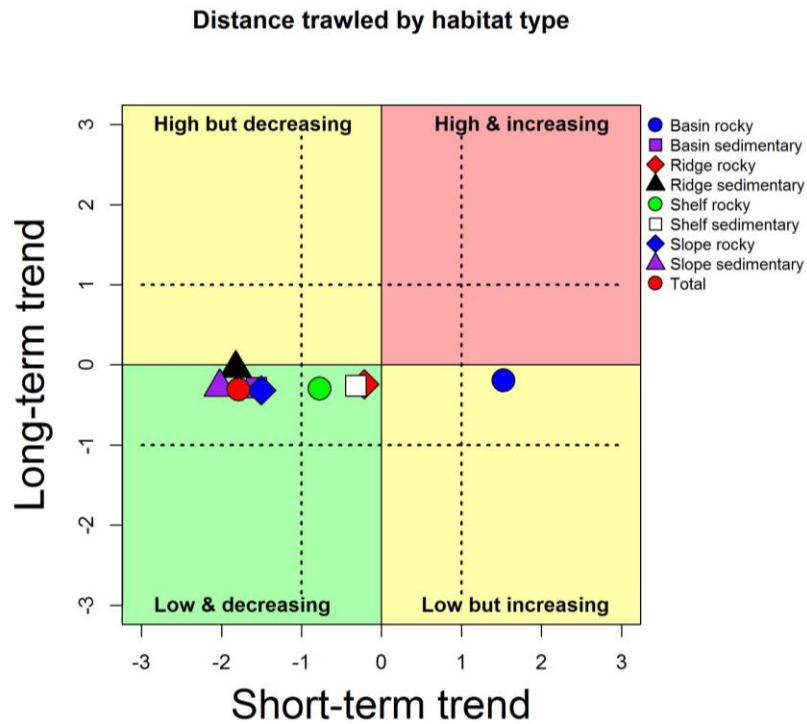


Figure AP31. Short- and long-term status of total distance trawled (1999 – 2004) by habitat type in the CCLME. Prior to plotting, time series data for each indicator were normalized to place them on the same scale. The short-term trend indicates whether distance trawled increased, decreased or remained the same over the last five years. The long-term trend represents the difference between the mean of the last 5 years and the mean of the full time series. The dotted lines represent ± 1.0 SD; thus, data points outside the dotted lines are considered to be increasing or decreasing over the short term or the current status is lower or higher than the long-term mean of the time series.

Table AP2. Summary of fisheries indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, three out of five primary considerations criteria support “landings” as an indicator of fishery removals.

| Pressure | Indicator | Primary considerations (5) | Data considerations (7) | Other considerations (6) | Summary comments |
|---------------------|--|----------------------------|-------------------------|--------------------------|---|
| Fisheries removals | Landings | 3 | 7 | 4 | Commercial landings represent the majority of removals for most species. This metric does not include discarded catch. Landings records from 1981 forward are available via http://pacfin.psmfc.org . |
| Fisheries removals | Groundfish total fishing mortality estimates | 5 | 4 | 4 | Groundfish total fishing mortality estimates are generated by the West Coast Groundfish Observer Program. These estimates are for groundfish only. The data are available from 2005 forward. |
| Habitat destruction | Distance trawled | 2 | 2 | 1 | Coast-wide estimates of distance trawled by habitat type were generated by Bellman and Heppell (2007) based on logbook data on each individual tow and GIS seafloor habitat maps. These estimates are available between 1999 and 2004. Work is currently under way to improve seafloor maps and further extend time series of estimated distance trawled. |

Table AP3. Top indicators for fisheries pressures.

| Pressure | Indicator | Definition and source of data | Time series | Sampling frequency |
|---------------------|---------------------------|---|----------------|--------------------|
| Fishery removals | Landings | Metric tons and pounds of the species landed by commercial fisheries in CA, OR and WA. Data are available from the Pacific Fisheries Information Network at http://pacfin.psmfc.org . | 1981 – Present | yearly |
| Fishery removals | Total mortality estimates | Metric tons and pounds of the groundfish species removed by commercial, recreational and research sources as well as incidental catch from non-groundfish fisheries in CA, OR and WA. Data are available from the West Coast Groundfish Observer Program in the FRAM division of NOAA's Northwest Fisheries Science Center. | 2005 – Present | yearly |
| Habitat destruction | Distance trawled | Kilometers (km) trawled by the limited-trawl groundfish fishery in CA, OR and WA by habitat type. Data are available from Bellman and Heppell (2007). | 1999-2004 | yearly |

NON-FISHERIES PRESSURES

For non-fisheries related anthropogenic pressures in the CCLME, we primarily focused on pressures identified by Halpern et al. (2008, 2009) and Teck et al. (2010). The range of identified pressures affects all habitats in the CCLME, from beaches to canyon outfalls and from estuarine to offshore pelagic waters. We describe below the definition, potential impacts and the selection and evaluation of indicators for each identified pressure in alphabetical order. For many non-fisheries related pressures, human population growth (particularly along the coast) is the ultimate driver and can be used as an indicator of the status and trends of numerous pressures. In most instances, however, we have found or developed more specific indicators that capture the spatiotemporal variability in the pressure more closely than human population growth and present the individual time series below.

SUMMARY OF NON-FISHERIES PRESSURES

We developed indicators for 21 non-fisheries pressures on the California Current Large Marine Ecosystem (CCLME). These pressures ranged in scope from land-based pressures such as inorganic pollution and nutrient input to at-sea pressures such as commercial shipping and offshore oil and gas activities. Ultimately, we evaluated 41 different indicators and selected the best indicator(s) to describe the status and trends of each pressure. Indicators were evaluated (Table AP4) using the indicator selection framework developed by Levin et al. (2011) and Kershner et al. (2011) and used in the previous version of NOAA's Integrated Ecosystem Assessment for the California Current (Levin and Schwing 2011). Data for each of the chosen indicators were compiled to develop time series and determine the status and trends for each pressure (Table AP5).

Most indicators of non-fisheries related pressures showed either significant short-term trends or their current status was at historically high or low levels (Fig. AP32). Indicators of inorganic, organic and ocean-based pollution, commercial shipping activity, recreational use and invasive species have all decreased over the short-term, while indicators of dredging, shellfish aquaculture, and marine debris (in the northern CCLME) increased; all of these pressures, though, remained within historic levels. In contrast, indicators of seafood demand, sediment and freshwater retention, nutrient input, power plant activity and coastal engineering remained relatively constant over the short-term, but were above historic levels, while indicators of offshore oil and gas activity and related benthic structures were at historically low levels. Taken together, these results support two primary conclusions: 1) decreasing trends of several non-fisheries pressures (e.g., shipping related indicators, industrial pollution and recreational activity) potentially reflect slowing economic conditions over the last few years and 2) non-fisheries pressures at historically high levels have leveled off and are not continuing to increase, although shellfish aquaculture, dredging and nutrient input are near the point where they will show increasing values at historically high levels if current trends continue over the next few years.

The interpretation of the status and trends of these pressures may differ depending on the EBM component of interest. For example, a decreasing trend in fisheries removals may be “good” for rebuilding stocks of Protected Resources or it could be “bad” for the economies of Vibrant Coastal Communities. In addition, none of these pressures act upon the ecosystem independently (i.e. many pressures are acting simultaneously on populations), and we have little understanding about whether the cumulative effects of multiple pressures will be additive, synergistic or antagonistic on populations of interest. Subsequent sections of the IEA begin to integrate the cumulative effects of multiple pressures on multiple EBM components (see “risk” sections for each EBM component in *Section 3* and the various management strategy evaluations in *Section 4*). Moreover, these anthropogenic pressures will interact with the underlying effects of

climatic and oceanographic pressures (detailed in *Section 2: Oceanographic and Climatic Drivers and Pressures*). The integration of anthropogenic, oceanographic and climatic pressures on multiple EBM components can now be modeled using various “end-to-end” ecosystem models (e.g., Atlantis; Fulton et al. 2011), but marine ecologists and fisheries scientists need to develop creative methods in the field to test the validity of these models’ hypotheses and increase managers’ confidence in decision making.

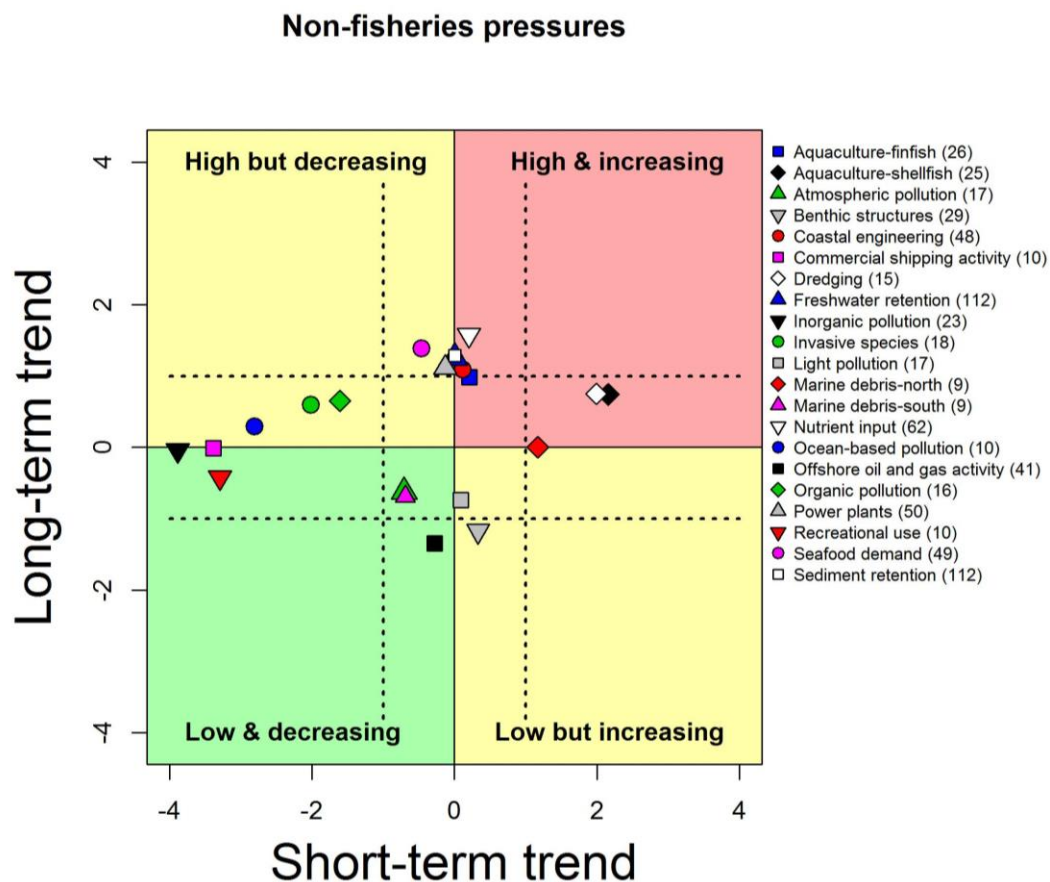


Figure AP32. Short- and long-term status of non-fisheries pressures in the CCLME. Prior to plotting, time series data for each indicator were normalized to place them on the same scale. The short-term trend indicates whether the indicator increased, decreased or remained the same over the last five years. The long-term trend represents the difference between the mean of the last 5 years and the mean of the full time series. The dotted lines represent ± 1.0 SD; thus, data points outside the dotted lines are considered to be increasing or decreasing over the short term or the current status is lower or higher than the long-term mean of the time series. Numbers in parentheses in the legend are the number of years in the time series for each pressure. Some symbols are smaller or larger than others to help distinguish them from overlapping symbols.

Table AP4. Summary of non-fisheries indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, finfish production as an indicator of finfish aquaculture has peer-reviewed literature supporting two out of five primary considerations criteria.

| Pressure | Indicator | Primary considerations (5) | Data considerations (7) | Other considerations (6) | Summary comments |
|-------------------------|------------------------------------|-------------------------------|----------------------------|-----------------------------|---|
| Aquaculture (finfish) | Finfish production | 2 | 7 | 4 | Finfish production is limited to the state of WA. Production will correlate with certain aspects of the pressures (e.g., escapement, disease, nutrient input, waste, fishmeal) on the ecosystem, but specific impacts may not increase/decrease with production as new technology is used to mitigate impacts on water quality or interactions with wild stocks. |
| Aquaculture (finfish) | Acres of habitat used | 1 | 2 | 3 | The amount of habitat used is relevant to determine impacts on the ecosystem. However, this metric may not account for advances in technology or growing capabilities. Data are limited to netpen dimensions of the current year's permit, so there is little temporal data. |
| Aquaculture (finfish) | Wild fish used to feed aquaculture | 1 | 0 | 0 | Increases in feed will impact wild-caught fisheries as well as contribute to effluent and waste effects on the local environment. Fishmeal increases with increased production of carnivorous species, but that may change with new sources of protein. Data are not available due to proprietary information. |
| Aquaculture (shellfish) | U.S. Shellfish production | 2 | 7 | 4 | Shellfish production has positive (e.g., filtering, removal of nutrients) and negative effects (e.g. habitat modification, invasive species) but the cumulative effects are unknown and these effects may change over time with advances in technology or growing capabilities. Washington state produces the greatest quantity of shellfish in the US, so total US shellfish production should reflect the current status and trends of shellfish production on the West Coast |
| Aquaculture (shellfish) | CCLME Shellfish production | 2 | 5 | 4 | Shellfish production has positive (e.g., filtering, removal of nutrients) and negative effects (e.g. habitat modification, invasive species) but the cumulative effects are unknown and these effects may change over time with advances in technology or growing capabilities. Estimates of production are available for CA and OR, but WA (which produces the most) does not have reliable estimates. |
| Aquaculture (shellfish) | Acres of habitat used | 1 | 5 | 3 | The amount of habitat used for aquaculture is relevant to determining the effects of aquaculture activities on various elements of the ecosystem. However, this metric may not account for advances in technology that allow more production per acre. Data are available from 1971 for CA, 1996 for OR and 2005 for WA. |

| Pressure | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|------------------------------|------------------------------------|---------------------------------------|------------------------------------|-------------------------------------|---|
| Atmospheric pollution | Concentration of deposited sulfate | 5 | 7 | 4 | The concentration of sulfate deposition measured by the National Atmospheric Deposition Program is a proxy for all chemicals deposited across the landscape. This dataset has been used in multiple publications as an indicator for atmospheric pollution. |
| Benthic structures | # oil & gas wells | 1 | 7 | 3 | Potential negative impacts of offshore oil and gas wells may be balanced out by the possible enhanced productivity brought about by colonization of novel habitats by associated fishes and invertebrates. Annual reports of the California State Department of Conservation's Division of oil, gas, and geothermal resources contain information on the total number of offshore oil and gas wells in production on an annual basis from 1970 to 2010. |
| Coastal engineering | % modified shoreline | 2 | 2 | 1 | Detailed inventories of coastal engineering have been carried out throughout the Pacific Coast of North America by a variety of federal, state, and local agencies under a number of programs. Most, however, provide a baseline indication of current or recent conditions and are generally unavailable coastwide or over time. |
| Coastal engineering | Coastal population | 2 | 6 | 2 | The rate of shoreline armoring has been shown to correspond with the rate of population growth in coastal areas, and in the absence of good time-series of geospatial data for hardened shorelines, coastal population data (US Census) for the west coast of the United States provide a good proxy for this stressor. |
| Commercial shipping activity | Tons of cargo moved | 0 | 7 | 5 | The size of vessels plays an important role in determining how well "activity" compares to cargo moved. As this pressure is used to describe the probability of striking marine organisms, ground strikes, etc., this metric is not as good as an indicator including "number of trips" or "volume of water disturbed during transit". |
| Commercial shipping activity | # of trips | 3 | 6 | 5 | Correlated with shipping activity; perhaps this indicator could be improved if size of vessel and transit mileage was added to quantify the vessel's footprint and pathway. Otherwise, the number of trips doesn't tell us anything about the extent of the CCLME affected by these trips. |
| Commercial shipping activity | Volume of water disturbed | 4 | 7 | 4 | This indicator has not been used before, but it is similar to indicators that measure habitat modification caused by bottom-trawl fishing gear. Using the actual draft and breadth of each vessel times the distance travelled each trip provides a better estimate of the risk associated with the movement of shipping vessels through the CCLME. |

| Pressure | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|-----------------------|----------------------------|---------------------------------------|------------------------------------|-------------------------------------|---|
| Direct human impact | Beach attendance | 4 | 6 | 4 | Beach attendance has been used as a proxy for direct human impacts to the intertidal and nearshore ecosystems. |
| Disease/ pathogens | % of scientific articles | 0 | 5 | 2 | The percentage of scientific articles reporting disease in marine taxa is a worldwide measure, so there may be significant differences in this trend and what is occurring in the CCLME. This indicator also does not account for the severity of the disease outbreak, a very large outbreak counts the same as a relatively small outbreak. |
| Dredging | Dredge volumes | 3 | 7 | 5 | The amount of material (in cubic yards - CY) dredged from all US waterways off the US West coast is a concrete, spatially explicit indicator that concisely tracks the magnitude of this human activity throughout the California Current region. |
| Dredging | Dredge dump volumes | 2 | 5 | 3 | Annual offshore dump volumes are not summarized and reported separately, but can be determined with some data manipulation. Most dredging-associated material disposal on the US West coast occurs in open water or is integrated into beach nourishment programs. |
| Freshwater retention | Runoff magnitude | 3 | 4 | 4 | Discharge trends for many rivers mostly reflect changes in precipitation, primarily in response to short- and longer-term atmospheric-oceanic signals, and it is difficult to distinguish signal from noise in rivers with widely variable interannual discharge. Stream discharge data are accessible from a variety of gauged streams; incomplete gauging records or unmonitored streamflow can be simulated by a comprehensive land surface model. |
| Freshwater retention | Impoundment volume | 2 | 6 | 2 | Data series associated with parameters of consumption and storage likely provide some of the best indicators of human impacts to freshwater input. For most normal rivers, reservoirs can affect the timing of discharge, but appear to have little effect on annual discharge. Freshwater storage data are available from state agency databases, which include information on construction date and impoundment area/volume for all dams. |
| Inorganic pollution | Total inorganic pollutants | 3 | 7 | 4 | Measures of total inorganic pollutants disposed or released on site or in water will provide a relative measure over time of what gets into the CCLME. However, variation in other variables (e.g., precipitation and specific pollutants released) will de-couple these measurements from observations in the CCLME as well as the impact on organisms. |

| Pressure | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|---------------------|---|---------------------------------------|------------------------------------|-------------------------------------|--|
| Inorganic pollution | Total inorganic pollutants * toxicity | 3 | 7 | 4 | Adding a measure of toxicity to the amount of pollutants released will provide better context to the severity and potential impacts of pollutants released. However, variation in other variables will still limit the correlation between these land-based pollutants and observations in the CCLME. |
| Inorganic pollution | Total inorganic pollutants * toxicity* impervious surface areas | 5 | 1 | 1 | Including ISA helps to account for other variables and more closely links how much land-based pollutants reach the CCLME; however, the data are only available for 2000-2001 and 2010. We assumed a linear relationship between years to provide weightings for each year. New analyses of archived data could produce yearly measures of ISA with appropriate levels of funding. |
| Invasive species | # of invasive species | 5 | 2 | 4 | A quantitative global assessment scored and ranked invasive species impacts based on the severity of the impact on the viability and integrity of native species and natural biodiversity (http://conserveonline.org/workspaces/global.invasive.assessment/). This database is pooled by go-region, serves as a baseline for invasion, and has not been updated since its creation. |
| Invasive species | # of shipping ports | 2 | 5 | 4 | Shipping is considered the key invasion pathway for habitats in northern California and the southern California Bight; 'number of shipping ports' was significantly correlated with harmful species introductions in most regions globally. Simple indicator, but perhaps less informative due to lack of time-series data. |
| Invasive species | Shipping cargo volume | 2 | 4 | 4 | Shipping is considered the key invasion pathway for habitats in northern California and the southern California Bight; 'shipping cargo volume' was significantly correlated with harmful species introductions in most regions globally. Port volume data (in metric tons) were available from the US Army Corps of Engineers Navigation Data Center (http://www.ndc.iwr.usace.army.mil/data/datawcus.htm) during 1993-2009. |
| Light pollution | Nighttime stable lights | 4 | 7 | 5 | Light pollution has considerable effects on some organisms' nocturnal behaviors, predator/prey relationships, bioenergetics, nesting and migratory patterns. Data using average nighttime lights from the National Geophysical Data Center from 1994-2010 were used (http://www.ngdc.noaa.gov/dmsp/downloadV4composites.html#AXP) |
| Marine debris | National Marine Debris Program coastal trash | 2 | 4 | 4 | Standardized sampling programs of measuring marine debris will be better than community groups, but it is unknown whether coastal measurements correlate with ocean measurements. |

| Pressure | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|-------------------------------|-----------------------------------|---------------------------------------|------------------------------------|-------------------------------------|--|
| Marine debris | Coastal trash cleanup programs | 2 | 1 | 5 | Community group clean-ups are great, but they are not standardized and data will vary with sampling effort, not necessarily with abundance of marine debris. Coastal measurements may not correlate with ocean measurements. Beach trash is cleaned up by volunteers during the annual California Coastal Cleanup Day along California beaches, bays, rivers, and streams. Data are recorded by volunteers and summarized by the California Coastal Commission's Public Education Program: www.coastal.ca.gov/publiced/ccd/data.xls . 1989 – 2011. |
| Marine debris | Ocean-based measurement | 2 | 1 | 2 | Ocean-based surveys have not used consistent methods and have been performed sporadically at small spatial scales. Estimates are likely lagging indicators of debris currently going into the ecosystem. |
| Nutrient input | Nutrient loading | 3 | 5 | 4 | Nutrient loading from surface waters can be estimated using publicly available data on nutrient concentrations and flow rates from various US watersheds sampled by the USGS and various state and local agencies. Flow adjusted trends in concentration can be complex, as there often are multiple and possibly counteracting anthropogenic factors influencing nutrient source and transport in a watershed. |
| Nutrient input | Fertilizer loading | 3 | 5 | 5 | Models can predict the probability of nitrate contamination in ground waters of the United States based on fertilizer loading and other factors; it is unclear how this relates to coastal systems, however. County-level estimates are available of nutrient inputs (kg/km ²) to the land surface of the US from 1982-2001 (data to 2010 pending) based on fertilizer use, livestock manure, and atmospheric deposition. Nationwide fertilizer application data are available from 1945-1986. |
| Ocean-based pollution | Shipping activity and port volume | 4 | 7 | 4 | Ocean-based pollution was assumed to be primarily driven by vessel activities and port volume. This indicator evaluated well in most criteria and is a combination of the indicators for commercial shipping activity and invasive species. See these indicators for location of data. |
| Ocean mining | Unknown | . | . | . | This pressure has not been evaluated to date. |
| Offshore oil & gas activities | Annual production | 4 | 7 | 3 | The environmental risks posed by offshore oil and gas exploration and production are well known. Annual reports of the California State Department of Conservation's Division of oil, gas, and geothermal resources contain information on the number of barrels of oil/ cubic feet of gas produced on an annual basis from 1970 to 2010. |

| Pressure | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|-------------------------------|---|---------------------------------------|------------------------------------|-------------------------------------|--|
| Offshore oil & gas activities | # oil & gas wells | 1 | 7 | 3 | The environmental risks posed by offshore oil and gas exploration and production are well known. Annual reports of the California State Department of Conservation's Division of oil, gas, and geothermal resources contain information on the total number of offshore oil and gas wells in production an annual basis from 1970 to 2010. |
| Organic pollution | Toxicity-weighted pesticide concentration | 5 | 6 | 4 | This indicator is well supported for use as a measure of organic pollution. Data are collected as part of the U.S. Geological Survey's National Water-Quality Assessment Program, so data will continue to be collected using standardized methods that will be useful for temporal and spatial analyses in the future. |
| Power, desalination plants | Water withdrawal volumes | 2 | 5 | 2 | Coastal power plants draw in huge amounts of marine water for cooling purposes, creating an area around the intake pipes where larvae and small plants are entrained. The USGS has conducted water-use compilations in the US by state every 5 years since 1950, and thermoelectric power has represented the largest total category of water withdrawals in every compilation since 1960. |
| Power, desalination plants | Entrainment mortality | 3 | 3 | 3 | Models for estimating organism entrainment mortality relies on estimates of power plant entrainment and source water larval populations; however, a variety of other considerations may play a more important role in determining entrainment impacts. In California, calculation of daily entrainment mortality has been limited to a few power plants; historical data are limited and time series information is generally lacking. |
| Seafood demand | Total consumption | 5 | 7 | 5 | Total consumption of edible and non-edible fisheries products is well supported as an indicator of seafood demand. Data are available at national levels, which is likely the right scale as products are used all over the nation as well as internationally, and over long temporal scales. |
| Seafood demand | Per capita consumption | 3 | 7 | 5 | Per capita consumption of edible and non-edible fisheries products may not be the best indicator if thinking about total impact to the CCLME, but it is important because if this indicator rises, as recommended by US Dept. of Agriculture (DGAC 2010), then increases in total consumption may increase dramatically. |
| Sediment input | Impoundment volume | 4 | 6 | 3 | Decreases in sediment input are largely the result of river damming or diversions, which directly influence the rate of coastal retreat. Dam impoundment area volume data are available from state agency databases, which include information on construction date and impoundment area/volume for all dams. |

| Pressure | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|-------------------|----------------------------------|---------------------------------------|------------------------------------|-------------------------------------|--|
| Sediment input | Suspended sediment loading | 4 | 2 | 3 | Sediment loading from surface waters can be estimated using publicly available data on suspended sediment concentrations and flow rates from various US watersheds sampled by the USGS and various state and local agencies. Flow adjusted trends in concentration can be complex, as there often are multiple and possibly counteracting anthropogenic factors influencing sediment source and transport in a particular watershed. |
| Tourism | Unknown | . | . | . | This pressure has not been evaluated to date. |

Table AP5. Top indicators for non-fisheries related anthropogenic pressures.

| Pressure | Indicator | Definition and source of data | Time series | Sampling frequency |
|-------------------------------------|-----------------------------------|--|-------------|--------------------|
| Aquaculture: finfish | Finfish production | Washington state estimates (from WDFW) of Atlantic salmon aquaculture production (kg). | 1986 – 2011 | yearly |
| Aquaculture: shellfish | U.S. Shellfish production | Total U.S. shellfish production: Fisheries of the United States 2010: http://www.st.nmfs.noaa.gov/st1/publications.html . Using only “clams”, “mussels” & “oysters” estimates. | 1985 – 2009 | yearly |
| Atmospheric pollution | Atmospheric deposition of sulfate | Annual precipitation-weighted mean concentrations of sulfate measured at sites in CA, OR, and WA from the National Atmospheric Deposition Program (http://nadp.sws.uiuc.edu/data/ntndata.aspx) | 1994 – 2010 | yearly |
| Benthic structures | # offshore oil & gas wells | Total number of offshore oil and gas wells in production: Annual reports of the California State Department of Conservation’s Division of oil, gas, and geothermal resources (ftp://ftp.consrv.ca.gov/..pub/oil/annual_reports/). | 1981 - 2009 | yearly |
| Coastal engineering | Human coastal population | Population size of coastline counties in CA, OR, WA; US Census Bureau (http://www.census.gov/prod/2010pubs/p25-1139/p25-1139st1.csv) | 1960 – 2008 | decadal |
| Commercial shipping activity | Volume of water disturbed | Calculated using draft, breadth and distance traveled within CCLME while in transit between shipping and receiving ports for domestic (data from USACE Navigation Data Center, New Orleans, LA) and foreign (http://www.ndc.iwr.usace.army.mil/data/dataclen.htm) vessels. | 2001 – 2010 | yearly |

| Pressure | Indicator | Definition and source of data | Time series | Sampling frequency |
|-------------------------------|--|---|-------------|--------------------|
| Recreational use | Beach attendance | Summed beach attendance from CA, OR, and WA based on data from California State Park System Annual Statistical Reports, Oregon Parks and Recreation Dept., and Annual Attendance Reports from the Washington State Parks and Recreation Commission. | 2002 – 2011 | yearly |
| Disease/ pathogens | No appropriate indicator data available. | | | |
| Dredging | Dredge volumes | U.S. Army Corps of Engineers navigation data center dredging information system: http://www.ndc.iwr.usace.army.mil/data/datadrgsel.htm ; data includes dredge volumes for individual private contracts and Corps operated dredge projects from 1997 through 2011 in WA, CA, and OR. | 1997 – 2011 | yearly |
| Freshwater retention | Impoundment volume | Total reservoir storage area in CA and Pacific Northwest water resource regions; data from state agency databases, which include information on construction date and impoundment area/volume for all dams (California: http://cdec.water.ca.gov/misc/resinfo.html , Idaho: http://www.usbr.gov/projects/FacilitiesByState.jsp?StateID=ID , Oregon: http://www.usbr.gov/projects/FacilitiesByState.jsp?StateID=OR , Washington: https://fortress.wa.gov/ecy/publications/summarypages/94016.html). | 1900 – 2011 | yearly |
| Inorganic pollution | ISA-toxicity-weighted chemical releases | Total pounds of inorganic pollutants disposed of or otherwise released on site to the ground or water for '1988 core chemicals'; Environmental Protection Agency, Toxics Release Inventory (http://www.epa.gov/tri/). These release values were weighted by toxicity scores (Indiana Relative Chemical Hazard Score) and impervious surface area in the drainage watersheds of the CCLME (http://www.ngdc.noaa.gov/dmsp/download_global_isa.html). | 1988 – 2010 | yearly |

| Pressure | Indicator | Definition and source of data | Time series | Sampling frequency |
|------------------------------|--|---|-------------|--------------------|
| Invasive species | Tons of cargo | Total tons of cargo moved through ports in CA, OR and WA; Data from US Army Corps of Engineers Navigation Data Center (http://www.ndc.iwr.usace.army.mil/data/datawcus.htm) | 1993 – 2010 | yearly |
| Light pollution | Average nighttime visible light | Data are cloud-free composites of average visible nighttime lights made using all the available archived DMSP-OLS smooth resolution data for each calendar year. Data grid cell size is 1 km ² at the equator ; NOAA’s National Geophysical Data Center’s Version 4 DMSP-OLS Nighttime Lights Time Series Average Lights X Pct (http://www.ngdc.noaa.gov/dmsp/downloadV4composites.html) | 1994 – 2010 | yearly |
| Marine debris | Predicted counts of debris | The National Marine Debris Monitoring Program established standardized sampling of coastal trash along the Pacific coast. Ribic et al. (2012) modeled the predicted counts of debris in the northern and southern CCLME. This provides a standardized method that is not biased by number of volunteers or by type of debris collected. | 1999 – 2007 | yearly |
| Nutrient input | Nitrogen and phosphorus input from fertilizers | Total nitrogen and phosphorus input from fertilizer use by county has been summarized from 1987 – 2006 by the USGS (Ruddy et al. 2006, Gronberg and Spahr 2012). We use these data along with nationwide data (1945 – 2001) to develop an index for the CCLME across the longer time series. County-level data are available at: http://water.usgs.gov/lookup/getspatial?sr2012-5207_county_fertilizer . Nationwide data are from Ruddy et al. (2006) | 1945 – 2006 | yearly |
| Ocean-based pollution | Commercial shipping activity combined with tons of cargo | This indicator combines two previously used indicators. See “Commercial shipping activity” and “Invasive species” for details of data. | 2001 – 2010 | yearly |

| Pressure | Indicator | Definition and source of data | Time series | Sampling frequency |
|--------------------------------|----------------------------------|--|-------------|--------------------|
| Offshore oil activities | Offshore oil & gas production | Number of barrels of oil/ft ³ of gas produced: Annual reports of the California State Department of Conservation's Division of oil, gas, and geothermal resources (ftp://ftp.consrv.ca.gov/./pub/oil/annual_reports/); verified by National Ocean Economics Program at the Monterey Institute of International Studies (http://www.oceaneconomics.org/Minerals/oil_gas.asp). | 1970 – 2010 | yearly |
| Organic pollution | Toxicity-weighted concentrations | Data are toxicity-weighted concentrations of 16 pesticides measured in water samples from stream-water sites in WA, OR and CA; U.S. Geological Survey Scientific Investigations Report 2010-5139 (http://pubs.usgs.gov/sir/2010/5139/). | 1993 – 2008 | yearly |
| Power plants | Saline water withdrawal volumes | Average daily withdrawal volumes (millions of metric tons per day) of saline water from all thermoelectric power plants on the west coast of North America (Pacific Northwest and California regions, from Kenny et al. (2009) and other previous USGS water use reports (http://water.usgs.gov/watuse/50years.html)). | 1955 – 2005 | Every 5 years |
| Seafood demand | Total consumption | Total consumption or utilization of edible and non-edible fisheries products as reported by annual NOAA Fisheries of the United States reports: (http://www.st.nmfs.noaa.gov/st1/publications.html) | 1962 – 2010 | yearly |
| Sediment input | Impoundment area | Same as “Freshwater input” | 1900 – 2011 | yearly |

AQUACULTURE

BACKGROUND

The increased demand for seafood products in conjunction with declines in capture fisheries has led to worldwide increases in commercial aquaculture (Naylor et al. 2000, Sequeira et al. 2008). Aquaculture provides several socio-economic benefits including improved nutrition and health and the generation of income and employment (Barg 1992). Environmental benefits of aquaculture include the prevention and control of aquatic pollution because of the inherent need for good water quality, the removal of excess nutrients and organic matter in eutrophic waters from the filtering action of molluscs and seaweeds, and the removal of incorporated nitrogen by shellfish when individuals are harvested (Barg 1992, Shumway et al. 2003). However, environmental impacts resulting from aquaculture production include: (1) impacts to the water quality from the discharge of organic wastes and contaminants; (2) seafloor impacts; (3) introductions of exotic invasive species; (4) food web impacts; (5) gene pool alterations; (6) changes in species diversity; (7) sediment deposition; (8) introduction of diseases; (9) habitat replacement or exclusion; and (10) habitat conversion (Johnson et al. 2008).

The impacts of aquaculture operations on various components of the CCLME vary according to the species cultured (finfish or shellfish), the type and size of the operation, and the environmental characteristics of the site (Johnson et al. 2008). Finfish aquaculture generally occurs in large cage and floating net-pen systems that release excess food and waste directly into the environment, whereas shellfish aquaculture is generally associated with benefits to water quality aspects (Shumway et al. 2003). The relative impact of finfish and shellfish aquaculture also differs depending on the foraging behavior of the cultured species. Finfish require the addition of a large amount of feed into the ecosystem, which can result in environmental impacts from the introduction of the feed, but also from the depletion of species harvested to provide the feed. Bivalves are filter feeders and typically do not require food additives; however, fecal deposition can result in benthic and pelagic habitat impacts, changes in trophic structure and nutrient and phytoplankton depletion (Dumbauld et al. 2009). Aquaculture activities can affect fisheries at both a habitat and species-level. Planting of culture species, harvesting practices and structure placement can alter the habitat as well as the community composition of the seafloor (Goldburg and Triplett 1997, Ruesink et al. 2005, Bendell-Young 2006, Dumbauld et al. 2009).

Growing U.S. and worldwide demand for seafood is likely to continue as a result of increases in population and consumer awareness of seafood's health benefits. The most recent federal *Dietary Guidelines for Americans (DGAC 2010)* recommend Americans more than double their current seafood consumption. Because wild stocks are not projected to meet increased demand even with rebuilding efforts, future increases in supply are likely to come either from foreign aquaculture or increased domestic aquaculture production, or some combination of both (NOAA Aquaculture Draft Policy).

EVALUATION AND SELECTION OF INDICATORS

Based on differences in the suite of impacts caused by different types of aquaculture, we have separated finfish and shellfish aquaculture and selected indicators for each. For finfish aquaculture, we evaluated 3 indicators (Table AP4): finfish production, acres of area used, and the amount of wild fish needed to feed aquaculture fish. For shellfish aquaculture, we evaluated 3

indicators (Table AP4): Total U.S. shellfish production, CCLME shellfish production and acres of land leased by shellfish growers.

For both types of aquaculture, production estimates evaluated as the best indicator for measuring the status and trends of aquaculture activities in the CCLME primarily because production values are a direct measure of the intensity of aquaculture operations, whereas indicators such as acres of land will not reflect advances in technology and growing capabilities over time. For finfish, the only marine netpen operations in the CCLME occur in Washington State. Data are available from the Washington Department of Fish & Wildlife (WDFW) for the years 1986-present. For shellfish production, “Total U.S. shellfish production” evaluated higher than “CCLME shellfish production” for two reasons: (1) Washington State produces the most shellfish aquaculture in the United States and produces ~86% of shellfish on the West Coast; thus, total U.S. estimates should reflect the primary status and trend of shellfish aquaculture production in the CCLME, and (2) Shellfish production data are collected by the California Department of Fish and Game and the Oregon Department of Agriculture, but these data are not collected by any state agency in Washington; thus, values from CA and OR may not reflect the actual status and trends of shellfish aquaculture in the CCLME since WA represents 86% of production on the West Coast. Two years of data (2000 (PSAT 2003) & 2009 (PCSGA 2011)) was found for Washington State, but this lack of historical data and a continuous time series causes “CCLME shellfish production” to score lower than “Total U.S. shellfish production” as the best indicator.

STATUS AND TRENDS

The status and trends of aquaculture were divided into an indicator for finfish aquaculture and an indicator for shellfish aquaculture. The status and trends of finfish aquaculture were measured using estimates of Atlantic salmon aquaculture production in the state of Washington (Table AP5) because there are no other commercial marine netpen aquaculture operations along the U.S. West Coast. Using this dataset, finfish aquaculture over the last five years has been constant and at the upper limits of the long-term average (Fig. AP33). With an increase in finfish aquaculture production over the next few years, the short-term average (last five years) will likely be greater than 1 standard deviation (SD) above the long-term average.

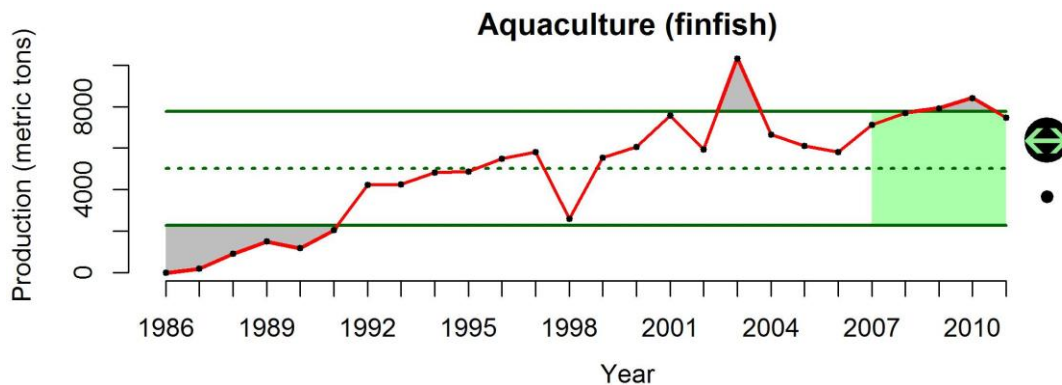


Figure AP33. Production of finfish aquaculture occurring in marine waters of the CCLME.

The status and trends of shellfish aquaculture was measured using estimates of U.S. shellfish production (Table AP5) because estimates of shellfish production in Washington State are not readily available and because Washington produces the most shellfish in the entire U.S. Using this dataset, shellfish aquaculture has increased significantly over the last five years, but is still within 1 SD of the long-term average (Fig. AP34). Similar to finfish aquaculture production, any increases in shellfish aquaculture production over the next couple of years will likely cause the short-term average to be greater than 1 SD above the long-term average.

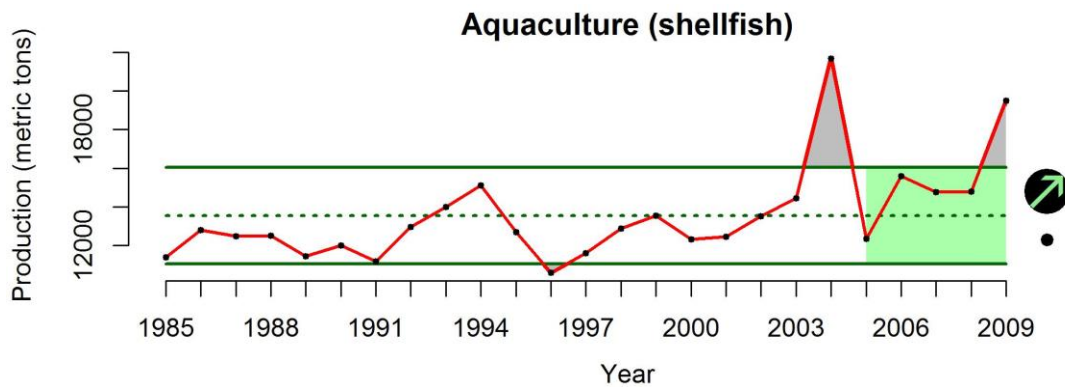


Figure AP34. U.S. production of shellfish (clams, mussels and oysters) aquaculture.

ATMOSPHERIC POLLUTION

BACKGROUND

The impact of pollutants deposited from the atmosphere on marine populations is largely unstudied; however, many nutrient, chemical and heavy-metal pollutants are introduced to marine ecosystems from sources that are geographically far away via this process (Ramanathan and Feng 2009). Substances such as sulfur dioxide, nitrogen oxide, carbon monoxide, lead, volatile organic compounds, particulate matter, and other pollutants are returned to the earth through either wet or dry atmospheric deposition (Johnson et al. 2008). Atmospheric nitrogen input is rapidly approaching global oceanic estimates for N_2 fixation and is predicted to increase further due to emissions from combustion of fossil fuels and production and use of fertilizers (Paerl et al. 2002, Duce et al. 2008). Atmospheric deposition is one of the most rapidly increasing means of nutrient loading to both freshwater systems and the coastal zone, as well as one of the most important anthropogenic sources of mercury pollution in aquatic systems (Johnson et al. 2008). Industrial activities have increased atmospheric mercury levels, with modern deposition flux estimated to be 3-24 times higher than preindustrial flux (Swain et al. 1992, Hermanson 1998, Bindler 2003). In the southwestern U.S., atmospheric deposition rates have been calculated at the upper end of this range, 24 times higher than pre-industrial deposition rates (Heyvaert et al. 2000). We assume these pollutants represent similar pressures on marine populations as pollutants introduced through other mechanisms (e.g., urban runoff and dumping).

EVALUATION AND SELECTION OF INDICATORS

We evaluated only one indicator for atmospheric deposition: the mean concentration of sulfates monitored by the National Trend Network (NTN) of the National Atmospheric Deposition Program (Table AP4). The NTN provides a long-term record of precipitation chemistry for sites located throughout the U.S. Data have been consistently collected weekly using the same protocols since 1994. Specific ions that are measured include calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^+), potassium (K^+), sulfate (SO_4^{2-}), nitrate (NO_3^-), chloride (Cl^-), and ammonium (NH_4^+) ions. These data are easily accessible via the NADP website: <http://nadp.isws.illinois.edu/ntn/>. This indicator of atmospheric deposition evaluated very high under all criteria categories (Table AP4).

STATUS AND TRENDS

The status and trends of atmospheric pollution were measured using the National Atmospheric Deposition Program's National Trends Network database (Table AP5). Annual precipitation-weighted means (mg/L) from all sites in CA, OR, and WA were used to calculate annual means for sulfate deposition in the CCLME. This monitoring network has data that goes back to 1985, but there was a major protocol shift in 1994, so we have limited the dataset to years from 1994 to the present. Using this dataset, atmospheric pollution has been constant over the last five years in the CCLME and is within 1SD of the long-term average (Fig. AP35).

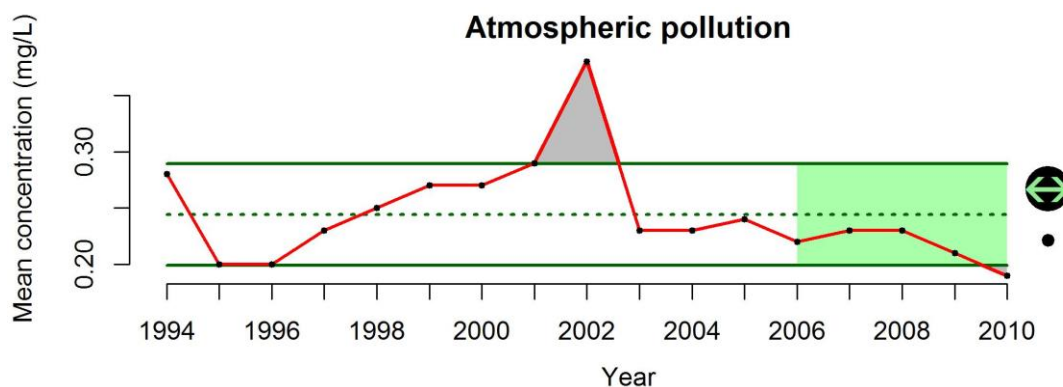


Figure AP35. Precipitation-weighted mean concentration (mg/L) of sulfates deposited out of the atmosphere in CA, OR, and WA.

BENTHIC STRUCTURES

BACKGROUND

The effects of benthic structures, such as oil rigs, wells and associated anchorings, on fish and other organisms will be initially destructive with the loss or modification of habitat, but these risks may dissipate in the long term by potential enhanced productivity brought about by colonization of novel habitats by structure-associated fishes and invertebrates (e.g., rockfish, encrusting organisms, etc.) (Love et al. 2006). Decommissioned rigs could also enhance biological productivity, improve ecological connectivity, and facilitate conservation/restoration of deep-sea benthos (e.g. cold-water corals) by restricting access to fishing trawlers.

Petroleum extraction and transportation can lead to a conversion and loss of habitat in a number of other ways. Activities such as vessel anchoring, platform or artificial island construction, pipeline laying, dredging, and pipeline burial can alter bottom habitat by altering substrates used for feeding or shelter. Disturbances to the associated epifaunal communities, which may provide feeding or shelter habitat, can also result. The installation of pipelines associated with petroleum transportation can have direct and indirect impacts on offshore, nearshore, estuarine, wetland, beach, and rocky shore coastal zone habitats. The destruction of benthic organisms and habitat can occur through the installation of pipelines on the sea. Benthic organisms, especially prey species, may recolonize disturbed areas, but this may not occur if the composition of the substrate is drastically changed or if facilities are left in place after production ends (Johnson et al. 2008).

Increasing pressure to find energy resources, such as oil and gas on continental shelves will likely increase exploration and the addition of various structures on the seafloor in the North Pacific: Canada, the U.S.A., Republic of Korea and Japan have all indicated that they intend either to begin or to expand exploration on the continental shelves of the Pacific, and drilling already occurs off Alaska and California and in the East China Sea (Macdonald et al. 2002).

EVALUATION AND SELECTION OF INDICATORS

We evaluated only one indicator of benthic structures in the CCLME: the number of oil and gas wells within the CCLME (Table AP4). In the future, the inclusion of other large-scale benthic structures with emerging uses, such as tidal- and offshore wind energy, large ocean net-pen aquaculture operations and ocean mining projects should be done to account for the increasing activity of these industrial sectors. The number of oil and gas wells only provides estimates of structures off California waters, as this is the only state along the coast of the CCLME that has offshore wells. Data are available from 1981 – 2009 on a yearly basis. The number of wells is easily understood and communicated to the public and policymakers.

STATUS AND TRENDS

The status and trends of benthic structures were measured using the number of oil and gas wells in offshore waters of the CCLME (Table AP5). These data are available in annual reports from the California Department of Conservation's Oil, Gas and Geothermal Resources Division for the years 1981 – 2009 (ftp://ftp.consrv.ca.gov/pub/oil/annual_reports/). We summed the number of state and federal offshore wells "producing" and "shut-in" (i.e. temporarily sealed up). The number of benthic structures in the CCLME has been constant over the short term (2005 – 2009), but has been greater than 1SD below the long-term average of the entire time series for the last decade (Fig. AP36).

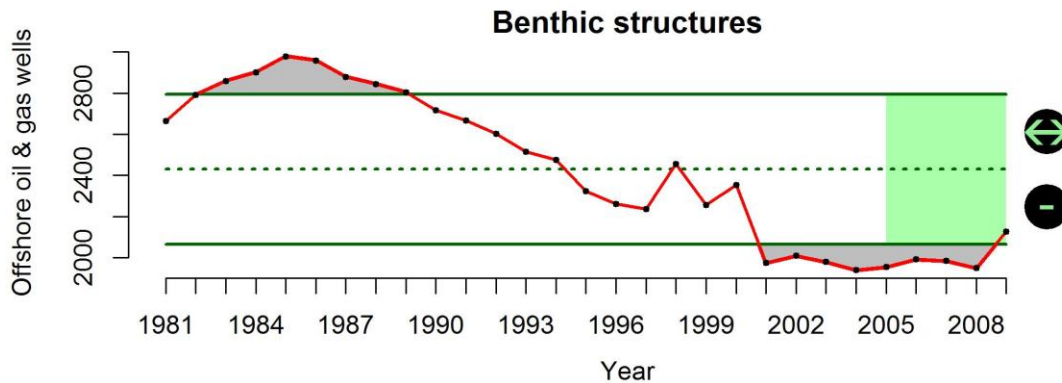


Figure AP36. The number of offshore oil and gas wells in production or shut-in in the CCLME.

COASTAL ENGINEERING

BACKGROUND

Many of the largest cities in the world are located in the coastal zone, and more than 75% of people worldwide are expected to live within 100 km of a coast by 2025 (Bulleri and Chapman 2010). In 2003, 53% of the population of the United States lived in the 673 coastal counties and this is expected to increase (Crossett et al. 2005). Transformation of coastal landscapes in response to urbanization also affects the intertidal zone and nearshore estuarine and marine waters, which are also increasingly altered by the loss and fragmentation of natural habitats and by the proliferation of a variety of built structures, such as breakwaters, seawalls, jetties and pilings.

Coastal engineering structures destroy the habitat directly under them and can significantly modify surrounding ecosystems through changes in circulation patterns and sediment transport (National Research Council 2007, Halpern et al. 2009, Shipman et al. 2010). Any structural modification of the shoreline will alter several important physical processes and can therefore be considered an impact (Williams and Thom 2001, Shipman et al. 2010). For the most part, impact potential can be related to the size and location of the structure and the types of physical processes it alters. Impacts may be considered direct or indirect. Direct impacts are generally associated with construction activities, including excavation, burial, and various types of pollution. Indirect impacts occur following physical disturbance, and are chronic in nature due to permanent alteration of physical processes such as sediment transport and wave energy. “Cumulative impacts” are associated with increasing number or size of indirect or direct impacts, which can have either linear or non-linear cumulative responses. Various engineering approaches have been adopted to minimize these effects, however (Thom et al. 2005, Bulleri and Chapman 2010).

Many shoreline “hardening” structures, such as seawalls and jetties, tend to reduce the complexity of habitats and the amount of intertidal habitats (Williams and Thom 2001, Bulleri and Chapman 2010). Because shorelines are highly diverse in their geologic nature and wave climate, acceptable ranges of armoring likely differ significantly from one location to another (Shipman et al. 2010). The definition of acceptable also will vary depending on the ecosystem response variable of interest. Differences in fish behavior and usage between modified and unmodified shorelines are caused by physical and biological effects of the modifications, such as changes in water depth, slope, substrate, and shoreline vegetation (Toft et al. 2007, Morley et al. 2012). Urban infrastructure

supports different epibiota and associated assemblages and does not function as a surrogate of natural rocky habitats (Bulleri and Chapman 2010). Its introduction in the intertidal zone or in nearshore waters can cause fragmentation and loss of natural habitats. Furthermore, the novel hard substrata along sedimentary shores can alter local and regional biodiversity by modifying natural patterns of dispersal of species, or by facilitating the establishment and spread of exotic species.

Almost all coastal engineering activities are subject to environmental reviews associated with the Coastal Zone Management Act, Endangered Species Act, and the US Army Corps of Engineers to assess potential impacts to natural resources and navigation. As coastal populations build, artificial structures are becoming ubiquitous features of coastal waters in urbanized centers, where they can form the dominant intertidal and shallow subtidal habitat. Ecological issues related to the introduction of coastal engineering structures into shallow coastal waters are only now beginning to receive more attention, with several recent reviews being published (e.g., Bulleri and Chapman 2010).

EVALUATION AND SELECTION OF INDICATORS

We evaluated two indicators of coastal engineering: proportion of modified (e.g., armoring, overwater structures) shoreline and coastal population estimates. Although both scored equally well with regard to theoretical considerations, the coastal population indicator scored significantly better for data considerations (Table AP4).

Inventories of coastal engineering have been carried out throughout the Pacific Coast of North America by a variety of federal, state, and local agencies under a number of programs, including Washington State's shoreline management act (http://www.ecy.wa.gov/programs/sea/sma/st_guide/intro.html), the USGS national assessment of shoreline change (<http://coastal.er.usgs.gov/shoreline-change/>), and NOAA's environmental assessment program (<http://response.restoration.noaa.gov/maps-and-spatial-data/environmental-sensitivity-index-esi-maps.html>), and the California Coastal Conservancy. However, time-series data of coastal engineering do not exist coastwide and therefore cannot be used to conduct change analysis. Most of these inventories only provide a baseline indication of current or recent conditions (e.g., Halpern et al. 2009) and if they represent data over multiple time periods, are generally only available over smaller spatial scales (e.g., county- or region-wide; personal communication, Lesley Ewing, California Coastal Commission). Coastal engineering structures are classified in a variety of ways, but primarily account for the percent of modified shoreline along a particular reach. The NOAA Environmental Sensitivity Index (ESI) maps provide a concise summary of coastal resources that are at risk if an oil spill occurs nearby. Anthropogenic structures are classified as follows: Exposed, solid man-made structures (1B), Riprap (class 6B), sheltered, solid man-made structures (8B), and sheltered riprap (8C). Inventories exist primarily for central and southern California (<http://www.coastal.ca.gov/recap/rcpubs.html>) and parts of Puget Sound; GIS ESI atlases have been completed for all of California, Puget Sound, the lower Columbia River; ESI atlases (no GIS) have been completed for the outer coasts of WA and OR. Inventories of shoreline classification and modifications maps (baselines) exist for the following years: southern CA: 1980, 1995, 2010; San Francisco Bay: 1986, 1998; central CA: 1995, 2006; northern CA: 1995, 2008 (M. Sheer, NOAA *pers. comm.*); OR and WA coast: 1985; and Puget Sound: 2000 (<http://response.restoration.noaa.gov/maps-and-spatial-data/shoreline-rankings.html>). To classify each shoreline unit, ESI map developers use information and observations from a combination of sources, including: overflights, aerial photography, remotely sensed data, ground-truthing (visits to

individual shorelines to validate aerial observations), and existing maps and data. Future assessments will attempt a change analysis as more recent classification actions are completed. This analysis will correlate the changes observed in shoreline armoring of specific counties in southern California with corresponding changes in coastal population growth.

The rate of shoreline armoring has been shown to correspond with the rate of population growth in coastal areas (Douglass and Pickel 1999), and in the absence of good time-series of geospatial data for hardened shorelines, coastal population data for the coastline counties of the West Coast of the United States provides a good proxy for this stressor. Population density has a long history of reporting and is known to affect coastal regions disproportionately (Crossett et al. 2005). Coastal population density data have been summarized by Crossett et al. (2005), who found that in 2003 the coastal population density (not including Alaska) of the Pacific Region was 303 persons per square mile, up from 207 in 1980, and expected to increase to 320 in 2008. From 2003 to 2008, the Pacific region is expected to increase by 2.2 million people or 6 percent in coastal population (Crossett et al. 2005). Population density is becoming increasingly understood in some regions as an agent of shoreline change (e.g. Puget Sound Partnership; http://www.psp.wa.gov/vitalsigns/shoreline_armoring.php). Coastline counties of the United States, located along the country's saltwater edges, account for just 254 of the nation's 3,142 counties yet contain 29 percent of its population, 5 of its 10 most populous cities, and 7 of its 10 most populous counties (Wilson and Fischetti 2010). To qualify as coastline, a county has to be adjacent to water classified as either coastal water or territorial sea. Transformation of coastal landscapes in response to urbanization also affects the intertidal zone and nearshore estuarine and marine waters, which are also increasingly altered by the loss and fragmentation of natural habitats and by the proliferation of a variety of built structures, such as breakwaters, seawalls, jetties and pilings. Unclear however, at this time, is the explicit relationship between coastal population levels and the relative amount of shoreline affected by coastal engineering structures; this data gap is likely driven by the lack of good time-series data on the latter.

STATUS AND TRENDS

The status and trends of coastal engineering were measured using estimates of human population in counties classified as "coastline" in WA, OR and CA (Table AP5). Data for coastline population estimates were retrieved from the U.S. Census Bureau's analysis of coastline population trends (Wilson and Fischetti 2010; <http://www.census.gov/prod/2010pubs/p25-1139/p25-1139st1.csv>). Because data were sampled decadal, we interpolated the last five years assuming a linear relationship in order to keep the short-term status (most recent five years) consistent with the other pressure indicators. The mean and standard deviation of the time series was calculated using only the original dataset. Using this indicator, coastal engineering has been increasing steadily over the entire time series. Over the last five years of this dataset, however, there was no change, but the current status is >1SD above the long-term average (Fig. AP37). Populations along the coast continue to increase, but perhaps the rate of increase is slowing. Nonetheless, the ultimate driver of many non-fisheries related pressures will continue to increase for the foreseeable future.

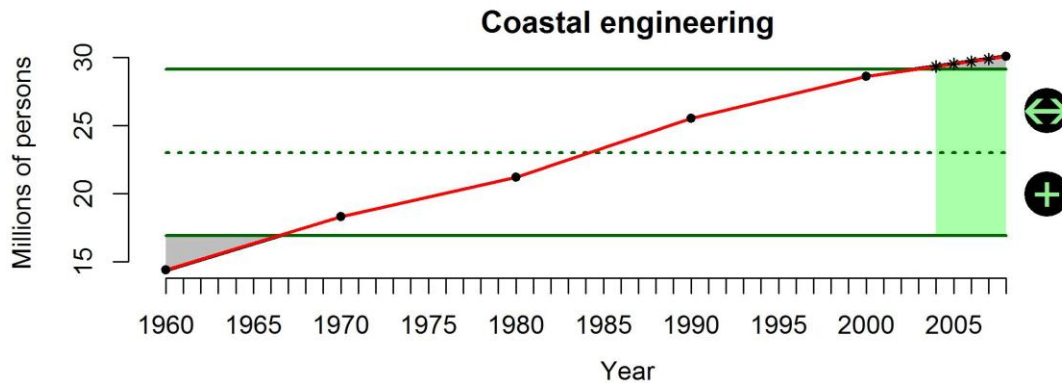


Figure AP37. U.S. population in coastline counties of WA, OR and CA.

COMMERCIAL SHIPPING ACTIVITY

BACKGROUND

Approximately 90% of world trade is carried by the international shipping industry and the volume of cargo moved through U.S. ports is expected to double (as compared to 2001 volume) by 2020 (AAPA 2012) due to the economic efficiencies of transporting goods via ocean waterways. The impacts of commercial shipping activity on the CCLME are numerous, but we used commercial shipping activity as a proxy for the potential risk of ship strikes of large animals, noise pollution and the risk of habitat modification due to propeller scouring, sediment resuspension, shoreline erosion, and ship groundings or sinkings (similar definition as Halpern et al. (2008)). Vessel activity in coastal waters is generally proportional to the degree of urbanization and port and harbor development within a particular area (Johnson et al. 2008). Benthic, shoreline, and pelagic habitats may be disturbed or altered by vessel use, resulting in a cascade of cumulative impacts in heavy traffic areas. The severity of boating-induced impacts on coastal habitats may depend on the geomorphology of the impacted area (e.g., water depth, width of channel or tidal creek), the current velocity, the sediment composition, the vegetation type and extent of vegetative cover, as well as the type, intensity, and timing of boat traffic (Johnson et al. 2008).

Ship strikes have been identified as a threat to endangered blue, humpback and fin whales (NMFS 1991, 1998, 2006), and this is of particular concern along the California coastline (Abramson et al. 2009, Berman-Kowalewski et al. 2010, Davidson et al. 2012). In addition to direct mortality from ship strikes, shipping vessels increase noise levels in the ocean which could interfere with normal communication and echolocation practices of marine mammals. When background noise levels increase, many marine mammals amplify or modify their vocalizations which may increase energetic costs or alter activity budgets when communication is disrupted among individuals (Holt et al. 2009, Dunlop et al. 2010). Underwater noise levels associated with commercial shipping activity increased by approximately 3.3 dB/decade between 1950 and 2007 (Frisk 2012).

The effects of commercial shipping activity on fish populations is not very well understood, but some data suggest responses will be behavioral in nature (e.g. Rostad et al. 2006) and related to loss of habitat (Uhrin and Holmquist 2003, Eriksson et al. 2004) or noise pollution (Slabbekoorn et al. 2010). Some fish species may be attracted to vessels, rather than being repelled by them and are not

bothered by noisy, passing ships (Rostad et al. 2006). However, frequently traveled routes such as those traveled by ferries and other transportation vessels may impact fish spawning, migration, communicative, and recruitment behaviors through noise and direct disturbance of the water column (Barr 1993, Codarin et al. 2009).

EVALUATION AND SELECTION OF INDICATORS

We evaluated three indicators of commercial shipping activity in the CCLME: port volume of cargo, number of vessel trips, and the volume of disturbed water during transit. Each of these indicators scored high in nearly all of the “Data Considerations” criteria (Table AP4) because most data are available from the U.S. Army Corps of Engineers (USACE) Navigation Data Center (<http://www.ndc.iwr.usace.army.mil/index.htm>). Each of these indicators is certainly correlated with some aspect of commercial shipping activity. The port volume of cargo moved through ports along the West Coast of the U.S. describes the total volume moving between ports, but this value does not give us any indication of how far shipping vessels are transporting these goods throughout the CCLME. This indicator is also probably not a relevant measure that management could use to “turn the dial” up or down. Increases or decreases to port volume may not have anything to do with the risk associated with ships striking marine mammals or increases to noise pollution off the coast (Table AP4).

Using the number of vessel trips within the CCLME as an indicator of commercial shipping activity provides a better link between the amount of risk shipping vessels have on various components of the CCLME; however, this indicator does not distinguish between vessels of different sizes or between trips that occur within a single port (exposure is low) and trips that span the entire length of the U.S. West Coast (exposure is high).

The final indicator evaluated was the volume of disturbed water during transit. We have not found this metric used specifically in other literature sources, but it is similar to metrics used as an indicator of habitat modification caused by the disturbance of bottom-trawl fishing gear (Bellman and Heppell 2007). We calculated the distance traveled within the CCLME by each vessel during transit from their shipping port to their receiving port and multiplied this value by the vessel’s draft and the vessel’s breadth. These values were then summed across domestic and foreign fleet vessels for the years 2001 – 2010. This indicator provided a more accurate estimate of the absolute exposure of the CCLME to commercial shipping vessels. There are not any likely reference points or target values for this indicator on a coastwide basis, but this indicator could be used in a spatially-explicit way (create GIS data layers) to monitor trends in shipping activity in specific corridors or during specific times of year that are frequently used by marine mammals (Table AP4).

In order to develop this indicator, we received port-to-port coastwise trip data with shipping and receiving drafts and names of all domestic shipping vessels for years 2001 – 2010 from the USACE Waterborne Commerce Statistics Center, New Orleans, LA. From the USACE Navigation Data Center database ([http://www.ndc.iwr.usace.army.mil/data/dataclen.htm#Foreign Traffic Vessel Entrances and Clearances](http://www.ndc.iwr.usace.army.mil/data/dataclen.htm#Foreign%20Traffic%20Vessel%20Entrances%20and%20Clearances)), we downloaded foreign traffic vessel entrances and clearances data to get all foreign port-to-port trips with draft and vessel names of each vessel for years 2001 – 2010. We then looked up the breadth of individual vessels from the USACE “Vessel Characteristics” database (<http://www.ndc.iwr.usace.army.mil/data/datavess.htm>). For vessels that were not contained within this database, we used the mean breadth of vessels within the same “Vessel type” for domestic vessels or within the same “Rig type” for foreign vessels.

We categorized trips into two categories. If the shipping and receiving port was the same (i.e. the vessel was moving from one dock to another or moving a barge within the port), this was categorized as “port” traffic, while all other trips were categorized as “coastal” traffic. For this analysis, we removed all “port” traffic because this pressure is defined as a measure of the risk of vessels striking marine mammals, causing noise pollution, and modifying coastal habitat. We include “port” traffic in the indicator for ocean-based pollution below. In order to calculate the distance traveled within the CCLME for each vessel, we used distances between ports as measured by NOAA’s Office of Coast Survey and documented in USDOC (2012). For trips that traveled outside of the CCLME, we used the distance from the port within the CCLME to the boundary of the CCLME following the major shipping lane pathways. For example, if a vessel traveled from San Diego, CA to Houston, TX, we calculated the distance from San Diego to the southern boundary of the CCLME on the vessel’s way toward the Panama Canal (estimated at 602 nm (1115 km)). These distances were then multiplied by the vessel’s shipping draft (m) and breadth (m) to give a volume (m³) of water directly disturbed by the vessel during transit through the CCLME. Obviously the wake of a vessel will disturb more than our calculated volume, so this is a conservative estimate of absolute volume, but the trends over time will be relative.

STATUS AND TRENDS

The status and trends of commercial shipping activity were measured using the volume of water disturbed within the CCLME (Table AP5). Using this dataset, we found that commercial shipping activity in the CCLME has decreased over the last five years, but the short-term mean is within 1SD of the long-term mean of the entire dataset (Fig. AP38). The decreasing trend in this dataset likely reflects current economic conditions over the last five years; thus, this indicator is likely to increase as economic conditions improve. The predominant contributor to this trend is foreign vessel traffic and these data are available back to 1997, while the domestic data may be available back to 1994 if funding were available to the USACE to perform this data inquiry.

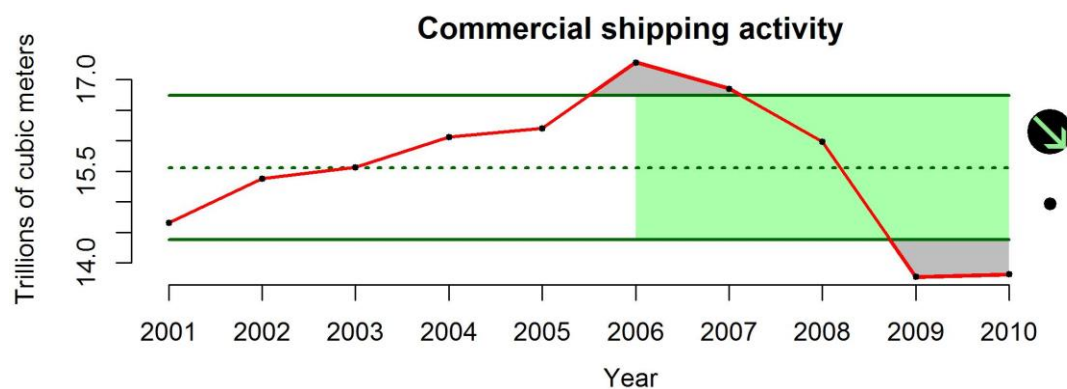


Figure AP38. Volume (trillions m³) of water disturbed during transit of commercial shipping vessels along the coast of the CCLME.

DISEASE/PATHOGENS

BACKGROUND

The last few decades have seen a worldwide increase in the reports of disease in the marine environment (Harvell et al. 1999), though these increases appear to be taxa related (Ward and Lafferty 2004). Diseases are thought to be fostered by increases in climate variability and human activity as many outbreaks are favored by changing environmental conditions which increase pathogen transmission or undermine host resistance (Anderson 1998). Marine flora and fauna serve as hosts for numerous parasites and pathogens that may affect the host populations as well as have cascading effects throughout the ecosystem. For example, the near elimination of seagrass (*Zostera marina*) beds from many North Atlantic U.S. coastlines in the 1930's due to wasting disease (thought to be caused by a pathogenic strain of *Libinia*, which has since been confirmed and identified in eelgrass beds in the 1980's on both coasts of the United States (Short et al. 1987)) was responsible for numerous alterations to coastal habitats (Rasmussen 1977) and fauna, including a reduction or loss of migratory waterfowl populations (Addy and Aylward 1944) and the loss of the scallop fishery in the mid-Atlantic coast of the U.S. (Thayer et al. 1984).

The population dynamics of many pathogens are sensitive to changes in their physical environment (e.g., temperature) which could modify pathogen development and survival, disease transmission and host susceptibility (Harvell et al. 1999, Harvell et al. 2002, Selig et al. 2006). Thus, understanding how climate variability affects disease transmission in the marine environment is necessary for successful management efforts. These efforts, however, are hindered by the absence of baseline and epidemiological data on the normal disease levels in the ocean (Harvell et al. 1999).

EVALUATION AND SELECTION OF INDICATORS

The only indicator we evaluated for marine disease/pathogens was the percentage of scientific articles published each year that reported disease among marine taxa (Ward and Lafferty 2004). Overall, this indicator did not evaluate well in Primary Considerations criteria (Table AP4). The percentage of scientific articles reporting disease in marine taxa is a very broad proxy for testing whether diseases in the marine environment are increasing or decreasing - though it is the first quantitative baseline created to measure this. This measure may or may not respond predictably to actual measurements of disease in the ocean. There are many other factors - such as funding and the number of investigators interested in studying this topic - which will heavily influence this indicator each year. However, data are available from Ward & Lafferty (2004) for several marine taxa from 1970-2001 and the methods seem to be reproducible such that the time series could be updated in the future with yearly literature searches. Ward & Lafferty's (2004) data are a worldwide estimate, so spatial variation is not understood and is not specific to the CCLME. It is easily understood by the public and policymakers, but there has been no history of reporting the trend of disease in the marine environment with this indicator.

The overall trend of the Ward & Lafferty (2004) data suggests that disease may be increasing in marine ecosystems globally, but there are no time series data available to evaluate disease incidence in the CCLME; thus, we have concluded that there are no appropriate indicators of disease to include at this time. The methods of Ward & Lafferty (2004) could be applied to studies of disease in the CCLME and used as a baseline, but determining whether the trends are due to actual increases in disease or simply increases in the investigation and reporting of disease will be difficult to

separate. The California Cooperative Oceanic Fisheries Investigations (CalCOFI) and NOAA's Southwest Fisheries Science Center's ecosystem surveys have been collecting and archiving plankton samples since 1951. If pathogens are preserved in these samples, perhaps this could be a line of research that could produce a baseline of disease incidence in the CCLME given necessary funding.

DREDGING

BACKGROUND

Dredging is the removal or displacement of any material from the bottom of an aquatic area (USACE 1983). It is required in many ports of the world to deepen and maintain navigation channels and harbor entrances. Elsewhere, commercial sand mining and extraction of sand and gravel from borrowing areas is conducted to meet demand for sand for construction and land reclamation. Excavation, transportation, and disposal of soft-bottom material can have various adverse impacts on marine or estuarine environments (Johnston 1981). These effects may be due to physical or chemical changes in the environment at or near the dredging site, and may include: reduced light penetration by increased turbidity; altered tidal exchange, mixing, and circulation; reduced nutrient outflow; increased saltwater intrusion; alteration, disruption, or destruction of areas in which fish live, feed and reproduce; re-suspension of contaminants affecting water quality; and creation of an environment highly susceptible to recurrent low dissolved oxygen levels.

EVALUATION AND SELECTION OF INDICATORS

We evaluated two indicators of dredging impacts: dredging volumes and dredge dump volumes (Table AP4). Dredge volumes scored better than the latter, primarily due to reporting omissions related to spatial coverage.

Most of the dredging activities conducted on the US West coast involve maintenance dredging of harbor or port areas and associated navigation channels, with associated material disposal in open water or integrated into beach nourishment programs. The amount of material (in cubic yards - CY) dredged from all US waterways off the US West coast is a concrete, spatially explicit indicator that concisely tracks the magnitude of this human activity throughout the California Current region.

These data are accessible through the U.S. Army Corps of Engineers navigation data center dredging information system: <http://www.ndc.iwr.usace.army.mil/data/datadrgsel.htm>; data include dredge volumes, locations, and costs for individual private contracts and Corps operated dredge projects from 1997 through 2011 nationwide. We summarized annual dredge volumes (converted to cubic meters) for all projects conducted in California, Oregon, and Washington. Annual offshore dump volumes are not summarized and reported separately, but can be determined with some data manipulation from this database. In some locations, dredge dump volumes are also reported to give an indication of the extent of, and trends in dredging activities (e.g., Annual OSPAR Reports on the Dumping of Wastes at Sea).

STATUS AND TRENDS

The status and trends of dredging in the CCLME were measured using dredged volume (millions of m³) of sediments from projects originating in WA, OR and CA waters (Table AP5). Using

this indicator, dredging has increased over the last five years, but the short-term average is still within 1SD of the long-term average of the entire time series (Fig. AP39).

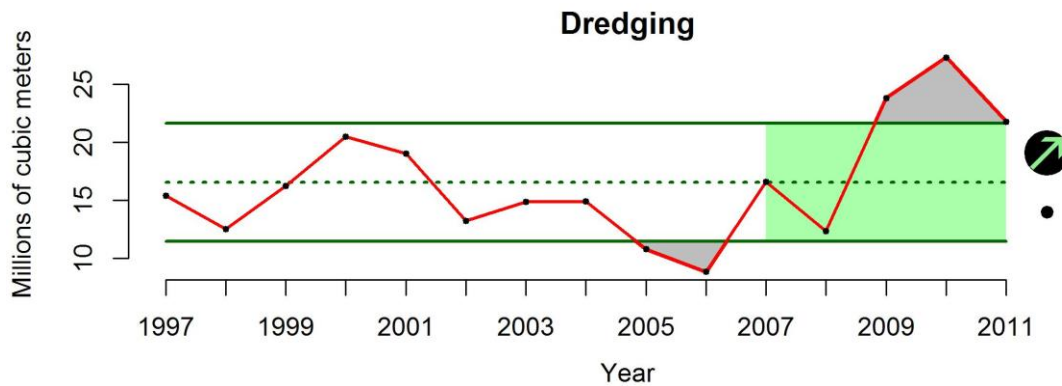


Figure AP39. Volume (millions m³) of dredged sediments from projects originating in WA, OR and CA.

FRESHWATER RETENTION

BACKGROUND

As the world's population grows along with increasing demands for freshwater, interannual variability and long-term changes in continental runoff are of great concern to water managers (Dai et al. 2009). Freshwater flow also affects fisheries and ESA-listed species. River discharge into many estuaries and coastal marine areas has been substantially altered by diversion for human use (Vorosmarty et al. 2000). Water withdrawals for public-supply and domestic uses have increased steadily since estimates began, with freshwater withdrawals of almost 350 Bgal/d (billion gallons per day) in 2005. Thermoelectric-power generation (see Power Plants, below) and irrigation withdrawals have generally been the two largest human use categories since these estimates were made. Hydropower is considered an "in-stream use" of freshwater, but associated dams and dam operations also alter flow patterns, volume, and depth of water within and below impoundments. Dam projects operating as "store and release" facilities drastically affect the magnitude, timing, and duration of downstream water flow and depth, resulting in dramatic deviations to natural fluctuations in habitat accessibility, acute temperature changes, and overall water quality.

Modified freshwater flow regimes change the salinity gradient and pattern in salinity variation within estuaries and coastal systems, and can induce large shifts in community composition and ecosystem function (Gillanders and Kingsford 2002). These ecosystems often respond most strongly on an interannual timescale to variability in freshwater flow. Several mechanisms for positive or negative flow effects on biological populations in estuaries have been proposed (Kimmerer 2002), with positive effects appearing to operate mainly through stimulation of primary production, with effects propagating up the food web. Overall impacts on the biota are generally considered negative, however, with documented changes to migration patterns, spawning habitat, species diversity, water quality, and distribution and production of lower trophic levels (Drinkwater and Frank 1994). For freshwater systems, a framework has been developed for assessing environmental flow needs for many streams and rivers to foster implementation of environmental flow standards at the regional scale (Poff et al. 2010). Studies focused on reductions in freshwater

flow have generally shown detrimental ecosystem effects and altered community composition (Gillanders and Kingsford 2002). However, freshwater subsidies to estuaries or hypersaline lagoons have also been shown to cause major shifts in vegetation, fish, and macroinvertebrate assemblages (Nordby and Zedler 1991, Strydom et al. 2002, Rutger and Wing 2006).

Discharge trends for many rivers reflect mostly changes in precipitation, primarily in response to short- and longer-term atmospheric-oceanic signals; notably, the cumulative discharge from many rivers globally decreased by 60% during the last half of the 20th century, reflecting in large part impacts due to damming, irrigation and interbasin water transfers (Dai et al. 2009). However, a comprehensive analysis of worldwide river gauging data suggests that direct human influence on annual streamflow is likely small compared with climatic forcing during 1948–2004 for most of the world’s major rivers (Dai et al. 2009). The immediate effect of dams on freshwater impact is also seemingly mixed. Reservoirs can affect the timing of discharge as well as the amount of discharged sediment and dissolved constituents, but for most normal rivers, reservoirs appear to have little effect on annual discharge (Milliman et al. 2008). However, most deficit rivers have flow regulation and irrigation indices, underscoring the importance of reservoirs and irrigation in facilitating water loss by increased consumption and (ultimately) increased evapotranspiration (Milliman et al. 2008).

EVALUATION AND SELECTION OF INDICATORS

We evaluated two potential indicators of freshwater input: river runoff or stream discharge and impoundment area behind dams (Table AP4). Other potential indicators of consumption and flow regulation (Milliman et al. 2008) were identified but not comprehensively evaluated at this time. Stream discharge data are accessible from a variety of gauged streams (<http://water.usgs.gov/nsip/>) from 1948-2004, although one of the major obstacles in estimating continental discharge is incomplete gauging records or unmonitored streamflow. Dai et al. (2009) have updated streamflow records for the world’s major rivers with streamflow data simulated by a comprehensive land surface model. However, it has been shown that it is very difficult to distinguish signal from noise in rivers with widely variable interannual discharge (Milliman et al. 2008). The effects of human activities on annual stream flow are likely small compared with those of climate variations during 1948–2004 (Dai et al. 2009) and ENSO-induced precipitation anomalies are a major cause for the variations in continental discharge (Dai et al. 2009). Furthermore, regional analyses of trends in US streamflow (generally characterized by increases in streamflow across all water-resource regions of the conterminous U.S. between 1940 and 1999) have been designed specifically to detect climate signals and minimize anthropogenic effects (Lins and Slack 2005)

River runoff (R) can also be expressed as the difference between precipitation (P) and the sum of evapo-transpiration (ET), storage (S) (e.g., groundwater), and consumption (C) (e.g., irrigation) (Milliman et al. 2008). Therefore, data series associated with the anthropogenically-derived parameters, C and S, likely provide some of the best indicators of human impacts to freshwater input. Freshwater storage (S) data are accessible and can be obtained on an annual basis from state agency databases, which include information on construction date and impoundment area/volume for all dams (California: <http://cdec.water.ca.gov/misc/resinfo.html>; Idaho: <http://www.usbr.gov/projects/FacilitiesByState.jsp?StateID=ID>; Oregon: <http://www.usbr.gov/projects/FacilitiesByState.jsp?StateID=OR>; Washington: <https://fortress.wa.gov/ecy/publications/summarypages/94016.html>). Furthermore, large-scale

hydrological alteration are known to cause a variety of downstream habitat changes, such as deterioration and loss of river deltas and ocean estuaries (Rosenberg et al. 2000).

We selected impoundment volume as our indicator of changing freshwater flow, primarily based on the long-term availability of annual impoundment data and the additional known effects of these large-scale hydrological alterations to downstream habitats (Table AP4).

STATUS AND TRENDS

The status and trends of freshwater retention in the CCLME were measured using the total impoundment volume (millions m³) of freshwater stored behind dams in CA, OR and WA (Table AP5). Using this dataset, the storage of freshwater has been relatively constant for the last 40 years, but the short-term average was greater than 1SD above the long-term average of the entire time series (Fig. AP40). This time series reflects the large increases in reservoir impoundment during the period of major dam building from the 1940's to the early 1970's with relatively little change since then.

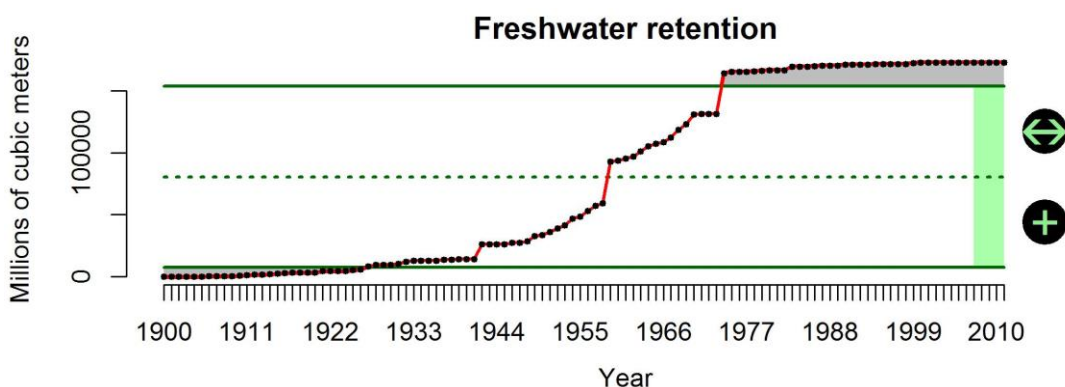


Figure AP40. Volume (millions m³) of freshwater stored behind dams in WA, OR and CA.

INORGANIC POLLUTION

BACKGROUND

Tens of thousands of chemicals are used by industries and businesses in the United States for the production of goods which our society depends. Many of the chemicals used in the manufacturing and production of these goods are toxic at some level to humans and other organisms and some are inevitably released into the environment. The production, use and release of various toxic chemicals have changed over time depending on economic indices, management methods (recycling and treatment of chemicals), and environmental regulations (USEPA 2010). The pathway of these chemicals to estuarine and marine environments can be direct (e.g., wastewater discharge into coastal waters or rivers) or diffuse (e.g., atmospheric deposition or urban runoff). Over the past 40 years, direct discharges have been greatly reduced; however, the input of pollutants to the marine

environment from more diffuse pathways such as runoff from land-based activities is still a major concern (Boesch et al. 2001).

While all pollutants can become toxic at high enough levels, there are a number of compounds that are toxic even at relatively low levels (Johnson et al. 2008). The US Environmental Protection Agency (USEPA) has identified and designated more than 126 analytes as “priority pollutants.” According to the USEPA, “priority pollutants” of particular concern for aquatic systems include: (1) dichlorodiphenyl trichloroethane (DDT) and its metabolites; (2) chlorinated pesticides other than DDT (e.g., chlordane and dieldrin); (3) polychlorinated biphenyl (PCB) congeners; (4) metals (e.g., cadmium, copper, chromium, lead, mercury); (5) polycyclic aromatic hydrocarbons (PAHs); (6) dissolved gases (e.g., chlorine and ammonium); (7) anions (e.g., cyanides, fluorides, and sulfides); and (8) acids and alkalis (Kennish 1998, USEPA 2003). While acute exposure to these substances produce adverse effects on aquatic biota and habitats, chronic exposure to low concentrations probably is a more significant issue for fish population structure and may result in multiple substances acting in “an additive, synergistic or antagonistic manner” that may render impacts relatively difficult to discern (Thurberg and Gould 2005).

Coastal and estuarine pollution can affect all life stages of fish, but fish can be particularly sensitive to toxic contaminants during the first year of life (Rosenthal and Alderdice 1976). Over time, organisms will accumulate contaminants from water, sediments or food in their tissues, which then transfers to offspring through reproduction and throughout the food web via trophic interactions. One of the most widely recognized effects of inorganic pollution was the decline of bald eagles and brown pelicans during the 1960’s and 1970’s. These birds accumulated DDT in their tissues which changed their ability to metabolize calcium, which resulted in birds producing abnormally thin eggshells which led to reproductive failure (Hickey and Anderson 1968, Blus et al. 1971). Negative impacts of pollution on commercial fish stocks have generally not been demonstrated, largely due to the fact that only drastic changes in marine ecosystems are detectable and the difficulty in distinguishing pollution-induced changes from those due to other causes (Sindermann 1994). Normally, chronic and sublethal changes take place very slowly and it is impossible to separate natural fluctuations from anthropogenic causes. Furthermore, fish populations themselves are estimated only imprecisely, so the ability to detect and partition contaminant effects is made even more difficult. However, measurements of marine biodiversity have shown that species richness and evenness are reduced in areas of anthropogenic pollution (Johnston and Roberts 2009).

EVALUATION AND SELECTION OF INDICATORS

We used inorganic pollution to describe the status and trends of inorganic pollution at locations that likely drain into the CCLME. We excluded releases of inorganic pollution into the air, as this pressure is covered by “atmospheric pollution” above. We evaluated three different indicators of inorganic pollution in the CCLME: total inorganic pollutants, toxicity-weighted inorganic pollutants, and ISA-(Impervious Surface Area) toxicity-weighted inorganic pollutants (Table AP4). Each of these indicators relies on data contained within the USEPA’s Toxic Release Inventory (TRI; <http://www.epa.gov/tri/>) database. Thousands of facilities from all across the United States have been required to report detailed information on the disposal (onsite and offsite) and releases to air, water, land or underground wells of over 650 chemicals since 1988. This provides a long-term, continuous time series of data across watersheds that drain directly into the CCLME.

Two of the three indicators scored high in our evaluation based on the amount of data available and the historical use of this type of data to communicate trends to the public. However, users of TRI information should be aware that TRI data reflect releases and other waste management activities of chemicals, not whether (or to what degree) the public has been exposed to those chemicals. Release estimates alone are not sufficient to determine exposure or to calculate potential adverse effects on human health and the environment. TRI data, in conjunction with other information, can be used as a starting point in evaluating exposures that may result from releases and other waste management activities which involve toxic chemicals. The determination of potential risk depends upon many factors, including the toxicity of the chemical, the fate of the chemical, and the amount and duration of human or other exposure to the chemical after it is released. Thus, simply using “total inorganic pollutants” data from the database scored lower than the other two indicators because it doesn’t take any other factors into account.

Toxicity-weighted pollutants provide more context to the types and risk of pollutants being released by industrial facilities; however, most studies trying to account for and quantify runoff of pollutants into streams and watersheds or the contamination of groundwater sources use impervious surface area (ISA) as an indicator or a leading contributing factor (Arnold and Gibbons 1996, Gergel et al. 2002, Halpern et al. 2008, Halpern et al. 2009). Impervious surface area generally allows greater concentrations of excess nutrients and pollutants to run into nearby streams and rivers. This can lead to stream communities with fewer fish species and lower indices of biotic integrity (Wang et al. 2001). Other researchers have documented increased erosion, channel destabilization and widening, loss of pool habitat, excessive sedimentation and scour, and reduction in large woody debris and other types of cover as a consequence of urbanization (Lenat and Crawford 1994, Schueler 1994, Arnold and Gibbons 1996, Booth and Jackson 1997).

The difficulty of incorporating ISA into this indicator was that there were only two years of data which quantify the amount of ISA within all of the watersheds that drain into the CCLME. Because these data were lacking, its evaluation is much lower in the data consideration criteria than the other two potential indicators. However, spatially-explicit ISA data for all the watersheds of the CCLME could be quantified from archived satellite data by the U.S. National Geophysical Data Center if it became a higher priority; thus we have chosen this as the best indicator in hopes that future processing of satellite data will increase the precision of ISA estimates at the scale of the CCLME.

In order to calculate this indicator, we downloaded data from 1988 – 2010 from the TRI Explorer’s database under ‘Chemical Release’ reports (http://iaspub.epa.gov/triexplorer/tri_release.chemical) using ‘All Industries’ and ‘1988 Core chemicals’ as data selection criterion for California, Oregon and Washington states. In some years,

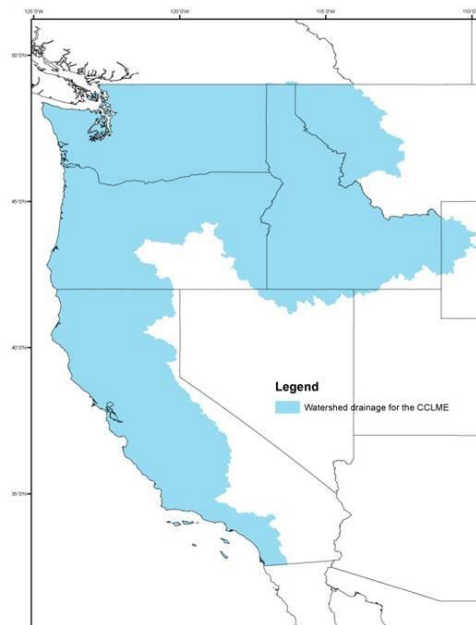


Figure AP41. Polygon of the watershed that drains into the CCLME and used to clip impervious surface area data layers (based on Halpern et al. 2009).

data were reported in different disposal categories, but we used data from all categories that were related to “surface water discharges” or included in the “total on-site releases to land” category. Data (lbs of releases) for each chemical were converted to kg and summed across each release category. In order to weight each chemical by its relative toxicity, we multiplied the amount of releases for each chemical by its score in the Indiana Relative Chemical Hazard Ranking Score (IRCHS; <http://cobweb.ecn.purdue.edu/CMTI/IRCHS/>) divided by 100:

$$\text{Toxicity-weighted releases} = \text{chemical releases} * (\text{IRCHR}/100)$$

For chemicals not listed in the IRCHR, we used the most closely-related substance on the list. These relative toxicity scores can range from 0 -100, but within our dataset, the highest scoring chemical was methyl hydrazine (IRCHR = 58.3). Toxicity-weighted releases were then summed across all chemicals for each year.

In order to provide weightings of ISA for each year, we used the ISA GIS data layers developed by the U.S. National Geophysical Data Center for the years 2000-2001 (global estimates) and January – June 2010 (estimates for the United States only). These data are available at http://www.ngdc.noaa.gov/dmsp/download_global_isa.html). We used the watershed drainage boundary for the CCLME developed by Halpern et al. (2009) to delineate the watersheds in which ISA values would be summed across (Fig. AP41). The 2000 – 2001 and 2010 ISA data layers were clipped to the watershed boundary polygon and then ISA values were summed across all cells. Because there were only two years of ISA data, we assumed a linear relationship between 2001 and 2010 and simply extrapolated summed ISA values to the remaining years between 1988 and 2010 based on this linear assumption. Summed ISA values were then standardized as a proportion of the maximum value (i.e., summed ISA value each year/maximum summed ISA value) such that the year with the highest summed ISA value had a weighting of 1 and all others were a proportion. Toxicity-weighted releases were then multiplied by the corresponding ISA weighting for each year. Finally, the ISA-Toxicity-weighted releases were normalized.

STATUS AND TRENDS

The status and trends of inorganic pollution in the CCLME were measured using ISA-Toxicity-weighted chemical releases from data collected by the Environmental Protection Agency and reported by the Toxics Release Inventory (TRI) Program (Table AP5). This indicator incorporates the amount and toxicity of chemicals released into water and onto land by industrial facilities as well as the amount of impervious surface area in the CCLME’s drainage basin. Using this indicator, inorganic pollution has decreased over the last five years, but is still within 1SD of the long-term average of the entire time series (Fig. AP42). A couple more years of low levels of chemical releases should bring the short-term average below historic levels.

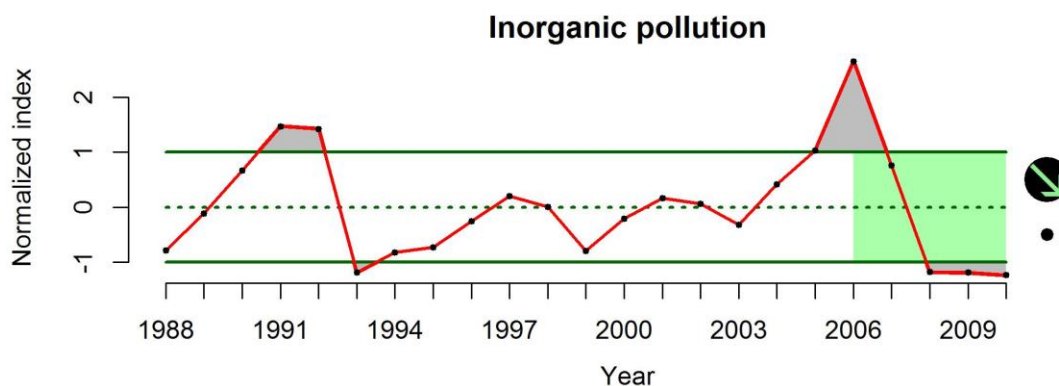


Figure AP42. Normalized index of ISA-toxicity-weighted chemical releases in WA, OR and CA industrial facilities.

INVASIVE SPECIES

BACKGROUND

Introductions of nonnative invasive species into marine and estuarine waters are considered a significant threat to the structure and function of natural communities and to living marine resources in the United States (Carlton 2001, Johnson et al. 2008). The estimated damage from invasive species in the United States alone totals almost \$120 billion per year (Pimentel et al. 2005). The mechanisms behind biological invasions are numerous, but generally include the rapid transport of invaders across natural barriers (e.g. plankton entrained in ship ballast water, organisms contained in packing material (Japanese eelgrass *Zostera japonica*) or fouling on aquaculture shipments, aquarium trade with subsequent release to natural environments) (Molnar et al. 2008). Nonnative species can be released intentionally (i.e., fish stocking and pest control programs) or unintentionally during industrial shipping activities (e.g., ballast water releases), aquaculture operations, recreational boating, biotechnology, or from aquarium discharge.

EVALUATION AND SELECTION OF INDICATORS

We evaluated three indicators of invasive species from the literature: number of alien species from regional records, number of shipping ports, and shipping cargo volume (Table AP4).

The rate of biological species introductions has increased exponentially over the past 200 years, and it does not appear that this rate will level off in the near future (Carlton 2001). In a recent paper, Molnar et al. (2008) provided a quantitative global assessment of invasive species impacts, scored and ranked based on the severity of the impact on the viability and integrity of native species and natural biodiversity (<http://conserveonline.org/workspaces/global.invasive.assessment/>). This database serves as a regional baseline for invasion worldwide; unfortunately, it has not been updated since its creation and therefore lacks time series information, limiting its utility as an indicator.

Molnar et al. (2008) also examined potential pathways for invasion, using generalized linear models to quantify the correlation between the number of harmful species reported and various pathways of introduction (e.g., shipping, aquaculture, canals). Shipping was considered the most likely pathway of harmful species introductions in most regions, with statistically significant correlations found between the shipping indicators number of ports and shipping cargo volume. In

the California Current, shipping was the key invasion pathway for northern California and the southern California Bight, whereas aquaculture was considered the more important invasion pathway in the Puget Trough/Georgia Basin and Oregon, Washington, Vancouver region. Empirical evidence increasingly indicates that the number of released individuals and number of released species are key determinants of the species that successfully invade new habitats (Lockwood et al. 2009). However, recent studies suggest this relationship may be taxa-specific, with invertebrates and diatoms appearing to be more sensitive to selective pressures during transportation that cause greater fluctuations in the number of released species than for other taxa, like dinoflagellates (Briski et al. 2012).

Invasive species were modeled as a function of ballast water release in ports by Halpern et al. (2009) when mapping cumulative human impacts to the CCLME. In this case, port volume data (in metric tons) were available for 618 global ports from several sources: the 2002 World Port Ranking (N=36) and 2003 U.S. Port Ranking (N=102) compiled by the American Association of Port Authorities (<http://www.aapa-ports.org>), Australia ports database (N=30; <http://www.aapma.org.au/tradestats>; access date 3/19/05), and Lloyds List database [N=450; Ref (S17)]. Thus, data are available and comparable at many different scales around the globe. It should be noted, however, that changes in ballast water regulations and treatment technologies may have or will likely in the future influence the risk of invasive species introduction (Waite et al. 2003).

The U.S. Department of Transportation projects that, compared to 2001, total freight moved through U.S. ports will increase by more than 50 percent by 2020 and the volume of international container traffic will more than double (American Association of Port Authorities Fact Sheet 2011: <http://www.aapa-ports.org/files/PDFs/facts.pdf>). In order to estimate the potential for species invasions, we used data on the total amount of shipping cargo (thousands of short tons converted to millions of metric tons) that moved through each port along the Pacific coast of the United States. These data were available from the US Army Corps of Engineers Navigation Data Center (<http://www.ndc.iwr.usace.army.mil/data/datawcus.htm>). CSV files were available for years 1993 – 2010. These data included port tonnage data from Alaska, so we used data from 2001 – 2010 from the “State Summary Tonnage Data” (<http://www.ndc.iwr.usace.army.mil/data/datastat.htm>) to calculate the proportion of tonnage along the Pacific Coast that was attributable to Alaska. We then used this proportion to subtract Alaska tonnage from the original dataset. For years in which we did not have an estimate of Alaska’s proportion (1993 – 2000), we used the average proportion from 2001 – 2010. This provided a dataset that was of the greatest temporal duration, but also removed the effects of Alaska’s port volume.

In addition to port volume, aquaculture has been associated with historic increases in invasive species, so an index that combines port activity and aquaculture (perhaps imports) should be added to this list of indicators and evaluated in the future. There are examples of combining these two metrics into a single spatial snapshot (Halpern et al. 2008, Halpern et al. 2009), but we need to modify this method into a temporal time series.

STATUS AND TRENDS

The status and trends of invasive species in the CCLME were measured using the amount of cargo moving through coastal ports of the CCLME (Table AP5). Using this indicator, the number of potentially invasive species entering ports along the CCLME has decreased over the last five years, but the short-term average is still within 1SD of the long-term average of the entire time series (Fig.

AP43). In addition to using this indicator, it would be good to develop an index that combines port volume and aquaculture as a more thorough indicator of the status and trends of invasive species.

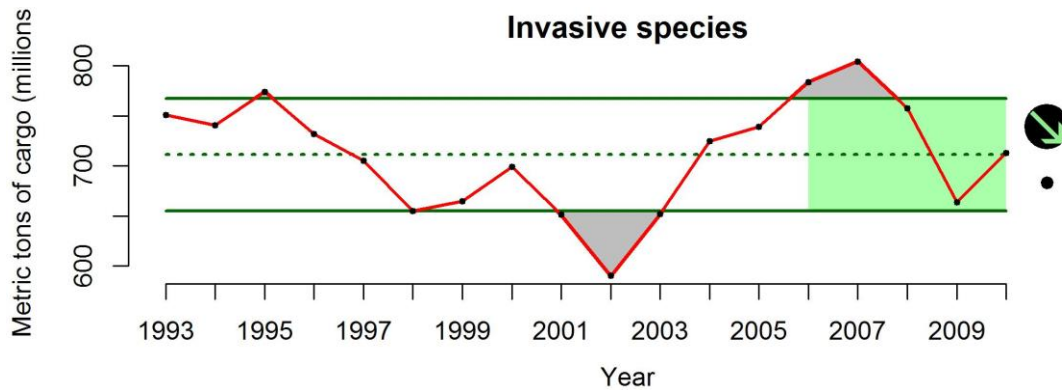


Figure AP43. Metric tons (millions) of cargo moved through ports in WA, OR and CA.

LIGHT POLLUTION

BACKGROUND

Ecological light pollution has demonstrable effects on the behavioral and population ecology of organisms in natural settings (Rich and Longcore 2006). As a whole, these effects derive from changes in orientation, disorientation, or misorientation, and attraction or repulsion from the altered light environment, which in turn may affect foraging, reproduction, migration, and communication (Longcore and Rich 2004). Many nocturnally migrating birds die or lose a large amount of their energy reserves during migration as a result of encountering artificial light sources (Poot et al. 2008). Juvenile sablefish exposed to a horizontal light gradient exhibited an avoidance of bright light (Sogard and Olla 1998). While juvenile sablefish were primarily surface-oriented, they nonetheless displayed clear day/night differences in vertical distribution. Proximity to the surface and low activity at night contrasted with higher activity and the greater range of vertical movement that typified daytime behavior. Movement throughout the water column during the day and the negative phototaxis observed in a horizontal gradient suggests that juveniles in nature, at least during the day, may not be restricted to the neuston.

For some species that nest on beaches, such as sea turtles, excess amounts of light along the coast cause considerable disruptions to their innate behaviors. Light pollution on nesting beaches alters critical nocturnal behaviors such as, how to choose a nesting site, how to return to the sea after nesting, and how hatchlings find the sea after emerging from their nests (Witherington and Martin 2000). Changes in the amount of polarized light also affect predator-prey relationships. As many marine species are visual predators, they use changes in the surrounding water's polarization signature to identify the presence of prey (Horváth et al. 2009). Planktivores are well-adapted at using changes in the polarization of the water to detect zooplankton in the water column that would otherwise be transparent (Flamarique and Browman 2001). Cephalopods also use polarized light as a hunting cue (Shashar et al. 1998) while other aquatic predators use light to detect camouflaged or distant prey resources (Shashar et al. 1998, Marshall et al. 1999). These alterations to the natural light dark cycles may allow for increased predation rates and subsequent changes to the community structure of areas with high levels of light pollution (Longcore and Rich 2004).

EVALUATION AND SELECTION OF INDICATORS

We evaluated only one indicator of light pollution in the CCLME: a normalized index of nighttime light pixels present in waters of the CCLME (Table AP4). This indicator is based on data collected by the US Air Force Weather Agency and processed by NOAA's National Geophysical Data Center (NGDC). This dataset is available from 1992 – 2010 on the NGDC's website:

<http://www.ngdc.noaa.gov/dmsp/downloadV4composites.html>. Specifically, we used the “Average Lights x Pct” (average nighttime lights, hereafter) data layers for satellites F12-18 and years 1994 – 2010 (we deleted data from satellite F10 based on recommendations from Elvidge et al. (2009)). These data layers were derived from the average visible band digital number (DN) of cloud-free light detections multiplied by the percent frequency of light detection. The inclusion of the percent frequency of detection term normalized the resulting digital values for variations in the persistence of lighting. For instance, the value for a light only detected half the time is discounted by 50%. Note that this product contains detections from fires and a variable amount of background noise. We first clipped each



Figure AP44. Polygon of the CCLME used to clip all nighttime lights data layers

We then clipped each data layer to the area of the CCLME. This polygon was created from the California Current LME data layer provided on NOAA's Large Marine Ecosystems of the World website (<http://www.lme.noaa.gov/>). However, we extended the northern boundary to the northern tip of Vancouver island, British Columbia as defined by the previous California Current Integrated Ecosystem Assessment (Fig. AP44; Levin and Schwing 2011).

Data layers were collected by different satellites with no internal calibration instruments, so data values are not directly comparable among years without a calibration method. Because data were collected by overlapping satellites we were able to calibrate among years using calibration equations provided by Chris Elvidge of the NGDC. We used the coefficients in the calibration equations to standardize the underlying data values in each pixel cell of each data layer. After calibration, we summed the value of all average nighttime lights for each cell in each data layer. For years in which multiple satellites collected data, we averaged the summed values for that year. These sums-of-average-nighttime-lights values were then normalized across years for the final metric.

STATUS AND TRENDS

The status and trends of light pollution in the CCLME were measured using a normalized index of the sum of average nighttime lights (Table AP5). These data was processed and made available by the U.S. Geophysical Data Center. Using this indicator, light pollution has been constant over the last five years and is within 1SD of the long-term average of the time series (Fig. AP45). This result is a little unexpected due to the contrasting increases observed in coastline populations. The

overall time series showed that light pollution steadily decreased from 1995 – 2004 within the CCLME and has been at these relatively low levels ever since.

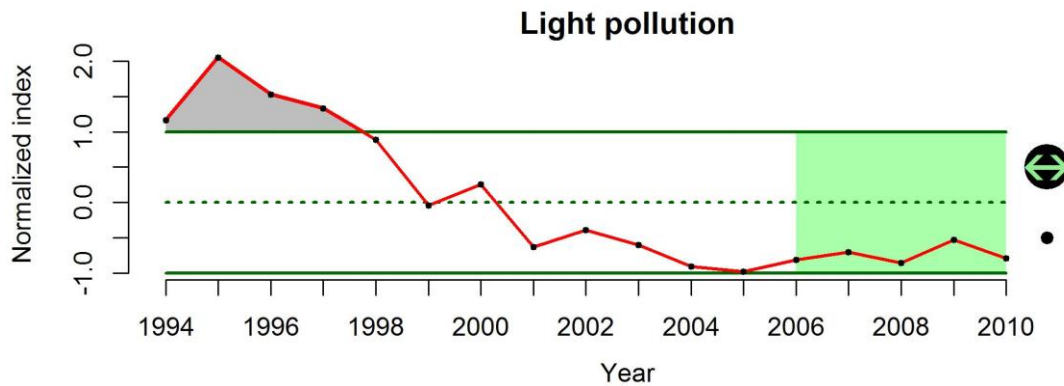


Figure AP45. Normalized index of the sum of average nighttime lights in waters of the CCLME.

MARINE DEBRIS

BACKGROUND

Marine debris is ubiquitous to all habitats of the ocean, whether it's in the form of metal cans or plastic bags on the beach (Ribic et al. 2012), derelict fishing gear caught on rocky bottom habitats (Good et al. 2010), household garbage in deep-water canyons (Watters et al. 2010) or micro-plastics in offshore surface waters (Doyle et al. 2011). The presence of marine debris along the coast extends from the shoreline to the greatest depths of the California Current, while 80% of this debris has been estimated to be from terrestrial runoff (Faris and Hart 1994). Data collected by Watters et al. (2010) using submersibles showed increases in marine debris on the ocean bottom in deep submarine canyons and continental shelf locations off California from the 1990's – 2007. Bauer et al (2008) found significantly higher densities of recreational fishing and other debris on rock ledges compared to other bottom types due to concentrated fishing effort where recreationally important fishes associate and the likelihood of gear becoming snagged on complex habitat.

While in some areas of the world the quantities of marine debris apparently show a decreasing trend during the past two decades (Ribic et al. 1997), other authors have reported increases (Coe and Rogers 1997). In general, the National Academy of Sciences (Criddle et al. 2008) has concluded that there is little quantitative information on amounts, sources, and trends of marine debris. However, recent programs such as the National Marine Debris Monitoring Program has developed standardized methods to quantify coastal debris and other surveys have begun to systematically quantify debris in meaningful ways (Gilfillan et al. 2009, Keller et al. 2010, Doyle et al. 2011). There are many coastal clean-up programs quantifying "marine debris" from beach cleanup surveys but these are not effective for quantifying temporal trends as the amount of debris collected is most likely related to the number of volunteers instead of the amount of debris. In addition, beach surveys are assumed to be an index of conditions in the surrounding waters, but there are no corresponding estimates of actual debris in the water to validate this assumption. Standardized programs with standardized metrics of measuring marine debris along the coast have been funded

by the Environmental Protection Agency in the past (NMDMP) and these methods could be adopted by other community groups which could make these data more effective.

Numerous researchers have documented the magnitude of marine debris and the threat that its ingestion or entangling poses to marine biota (Fowler 1987, Ryan 1990, Bjorndal et al. 1994, Moore et al. 2001, Moore et al. 2002). Marine debris, especially plastics, produces fragments that can be ingested by many marine organisms, resulting in mortality (Derraik 2002, Thompson et al. 2004, Browne et al. 2008). Marine debris in the form of lost fishing gear continues to “fish” by trapping fish, invertebrates, seabirds and marine mammals (Kaiser et al. 1996, Good et al. 2010). Marine debris may also impact populations behaviorally by concentrating individuals both at the water’s surface (FAD – floating aggregation devices; Aliani and Molcard 2003)) and on the bottom (artificial reefs; Stolk et al. 2007).

EVALUATION AND SELECTION OF INDICATORS

We evaluated three indicators for marine debris in the CCLME. The first is marine debris measured by the National Marine Debris Monitoring Program (NMDMP). This program developed standardized methods using volunteers to record specific types of marine debris among 18 sites in the northern and southern CCLME with Point Conception as the boundary between the two regions. Semi-permanent transects (500 m in length) were sampled at sites every 28 days from 1999 – 2007. This standardized sampling protocol allows for a temporal analysis of the data. Marine debris estimates from beach clean-ups or standardized sampling methods are still suspect as indicators of what debris is actually in the ocean waters or on the seafloor, so this indicator scores poorly in many criteria because of this. However, the data are sound and provide nearly a decade of broad-scale spatiotemporal information that has been lacking.

The second indicator evaluated was beach trash collected during the annual California Coastal Cleanup Day which is organized by the California Coastal Commission’s Public Education Program and occurs on the same day as the International Coastal Cleanup day organized by the [Ocean Conservancy](#). Volunteers show up and remove trash from beaches, lakes and other waterways. This trash is recorded by the volunteers and reported to the Education Program where the data are summarized and available for download: www.coastal.ca.gov/publiced/ccd/data.xls. Sampling is not standardized by material or number of volunteers, so the amount of trash collected is most likely an indicator of the number of people who volunteer each year, rather than the actual amount of trash and debris on the coast; thus this indicator scored low in comparison with the NMDMP program.

The final indicator evaluated was ocean-based measurements. This would be an actual measurement of debris in the oceans rather than measurements of trash on the beach that may or not make its way into the ocean. There are some surveys that record marine debris including the Northwest Fisheries Science Center’s annual groundfish bottom trawl survey (Keller et al. 2010) which has collected and recorded marine debris since 2007. There are also examples of plankton surveys (e.g., California Cooperative Oceanic Fisheries Investigations (CalCOFI) and NOAA’s Southwest Fisheries Science Center’s ecosystem surveys) that also collect and quantify micro-plastics present in samples (Moore et al. 2002, Gilfillan et al. 2009, Doyle et al. 2011). However, these studies are usually short-term studies (1-2 years). The CalCOFI plankton samples (1951 to present) are archived at Scripps Pelagic Invertebrates Collection, so there is opportunity to retroactively quantify plastics in these samples, but funding for this work is not presently available. Lack of data for ocean-based measurements of marine debris eliminates it from being useful.

Thus, we used estimates of marine debris from the NMDMP as the indicator for marine debris in the CCLME. Christine Ribic (U.S. Geologic Survey) provided predicted counts of marine debris data from the model developed by Ribic et al. (2012). These data were separated into northern and southern CCLME regions and into three different debris categories: land, ocean and general. We summed the predicted counts for all three debris categories to provide a single estimate for each region.

STATUS AND TRENDS

The status and trends of marine debris in the CCLME were measured using data from the Nation Marine Debris Monitoring Program (Ribic et al. 2012). These data were derived from a generalized additive model that used standardized surveys of debris along the coast of the CCLME. Using this indicator, marine debris in the northern CCLME (north of Point Conception, CA) was increasing between 2003 and 2007, but the short-term average was within historic levels (Fig. AP46a). In the southern CCLME, marine debris was relatively constant across the last five years of this time series and within historic levels (Fig. AP46b). This program no longer collects data, so an extension of this dataset will not occur unless funding for the program is revisited.

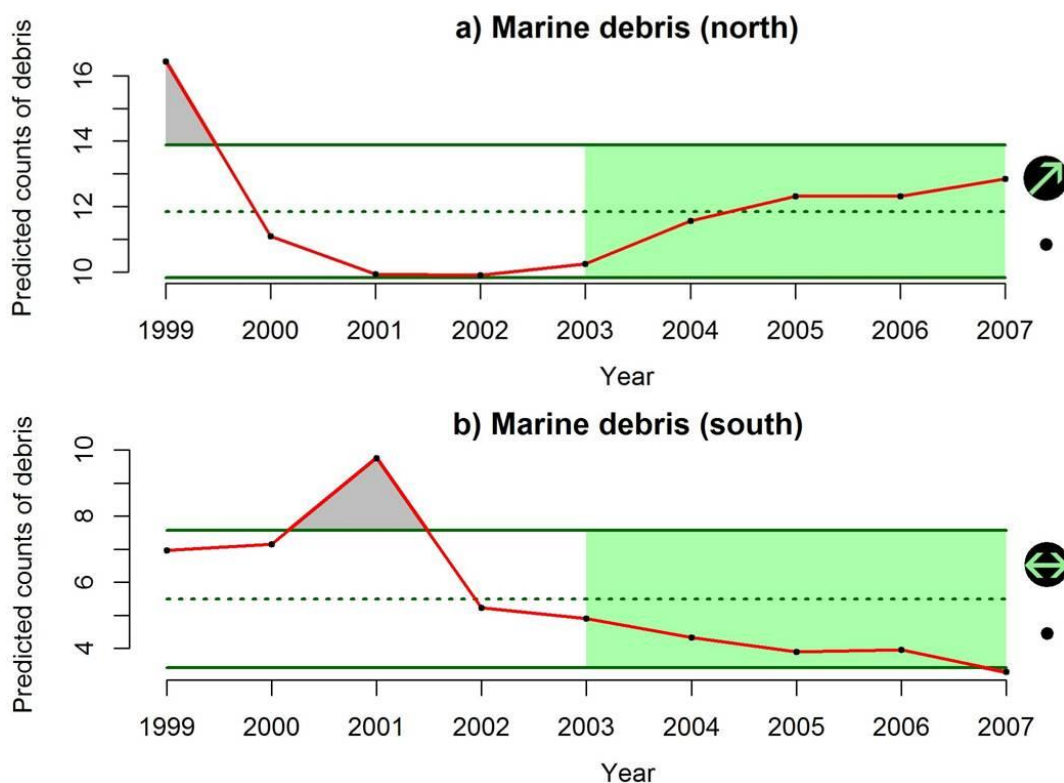


Figure AP46. Predicted counts of debris along the a) northern and b) southern coasts of the CCLME (Point Conception separates the regions). Data provided by Christine Ribic (Ribic et al. 2012).

NUTRIENT INPUT

BACKGROUND

Elevated nutrient concentrations are a leading cause of contamination in streams, lakes, wetlands, estuaries, and ground water of the United States (USEPA 2002). Nutrients (primarily nitrogen and phosphorus) are chemical elements that are essential to plant and animal nutrition; in marine waters, either phosphorus or nitrogen can limit plant growth. However, in high concentrations they can be considered water contaminants (USEPA 1999a).

Excess nutrients in a body of water can have many detrimental effects on drinking water supplies, recreational use, aquatic life use, and fisheries, and there are multiple indirect effects of nutrient enrichment of surface waters on human health. However, excessive nutrients are more often a cause of concern because of their role in accelerating eutrophication, which produces a wide range of other impacts on aquatic ecosystems and fisheries. Severely eutrophic conditions may adversely affect aquatic systems in a number of ways, including: algae blooms; declines in submerged aquatic vegetation (SAV) populations through reduced light transmittance, epiphytic growth, and increased disease susceptibility; mass mortality of fish and invertebrates through poor water quality (e.g., via oxygen depletion and elevated ammonia levels); and alterations in long-term natural community dynamics (Dubrovsky et al. 2010). Algal toxins harmful to animal and human health can be produced from blooms of some cyanobacteria species. High algal biomass also is associated with hypoxia (low dissolved-oxygen concentrations), which can contribute to the release of toxic metals from bed sediments, increased availability of toxic substances like ammonia and hydrogen sulfide, and fish kills. In recent years, nitrate and other nutrients discharged from the Mississippi River Basin have been linked to a large zone of hypoxia in the Gulf of Mexico along the Louisiana-Texas coast (Sprague et al. 2009).

Nonpoint sources of nutrients which affect stream and groundwater concentrations include fertilizer use, livestock manure, and atmospheric deposition (Ruddy et al. 2006). Within some coastal regions of the U.S. (e.g., mid-Atlantic states), much of the excess nutrients originates from point sources, such as sewage treatment plants, whereas failing septic systems often contribute to non-point source pollution and are a negative consequence of urban development (Johnson et al. 2008). However, nutrient loading can be a complex indicator to interpret, as a variety of hydro-geomorphic features (basin slope, basin area, mean annual precipitation, stream flow, and soil type) may also interact with possible nutrient sources to complicate estimates of nutrient concentration and loading. As well, there often are multiple and possibly counteracting anthropogenic factors influencing nutrient source and transport in a watershed, and without detailed knowledge of all important factors in each watershed, it may be difficult to discern the specific cause(s) of a trend in concentration (Sprague et al. 2009). Best land-use practices are known to reduce nutrient loading. Protocols for establishing total maximum daily load (TMDL) values of nutrients have been developed for specific bodies of water throughout the country (USEPA 1999a); however, we uncovered few examples in the literature of TMDLs for marine systems on the Pacific coast of the US.

Despite some of the previous cautions, nutrient loading in freshwater systems is generally a well understood indicator with a long history of reporting, as evidenced by requirements under the Clean Water Act, intensive nationwide monitoring programs at the federal, state, and local level, and a variety of national and regional trend reports by USGS (Ruddy et al. 2006, Wise et al. 2007, Sprague et al. 2009, Dubrovsky et al. 2010, Kratzer et al. 2011).

EVALUATION AND SELECTION OF INDICATORS

Nutrient input to coastal areas can be estimated in multiple ways. For this analysis, we evaluated only two types of nutrient input indicators: county-level inputs of nitrogen and phosphorus via fertilizers and nutrient loading (TN, TP) from stream monitoring records.

Halpern et al. (2009) used time series data from Nolan and Hitt (2006) on county-level fertilizer application data from 1992-2001 (kgs/hectare) and confined manure (primarily from dairy farms) from 1992-1997. These files (<http://water.usgs.gov/GIS/dsdl/gwava-s/index.html>) (Nolan and Hitt 2006) have a relatively limited temporal range (between 1992 – 2001). A comparable alternative would be to compile county-level estimates of nutrient inputs (kg/km²) to the land surface of the conterminous United States, presented from 1982-2006 based on fertilizer use, livestock manure, and atmospheric deposition (Ruddy et al. 2006, Gronberg and Spahr 2012)). An older time series (1945-1986) of nationwide fertilizer application data (Ruddy et al. 2006, Dubrovsky et al. 2010) could expand the time series further by assuming that watersheds bordering the Pacific coast follow the same historic trends in fertilizer applications. More recent data (2007 – 2010) is expected to be analyzed and summarized in 2013 (N. Dubrovsky, USGS, *pers comm*). Models have been used to predict the probability of nitrate contamination in ground waters of the United States based on fertilizer loading and other factors (Nolan and Hitt 2006). It is unclear how this relates to coastal systems, however.

A more data-intensive approach would be to estimate nutrient loading from surface waters using publicly available data on nutrient concentrations and flow rates from various US watersheds sampled by the USGS and various state and local agencies. Changes in stream flow are an important influence on nutrient concentrations in streams: depending on the particular nutrient sources in a watershed and how these nutrients are transported to the stream, increases or decreases in stream flow can lead to increases or decreases in concentrations (Sprague et al. 2009). Nutrient data are publicly accessible through the online USGS National Water Information System (NWIS) database at (<http://nwis.waterdata.usgs.gov/usa/nwis/qwdata>). The majority of data contained in the NWIS database are from water samples collected using standard methods described in U.S. Geological Survey (variously dated). USGS flow data can be accessed from (http://nwis.waterdata.usgs.gov/nwis/dv/?referred_module=sw). Nutrient (TN and TP) loading can be estimated at various time increments (e.g., daily, annual) using LOADEST, a USGS program that finds a best fit data model for flux as a function of discharge. The Yale University interface LOADRUNNER (<http://environment.yale.edu/loadrunner/>) calculates daily, monthly, and annual element fluxes, from these USGS water quality sample and stream flow data sources.

Nutrient trends in west coast rivers (1993-2003) have been summarized using similar methods in a recent report by Sprague et al (2009), which showed that flow adjusted trends in total phosphorus concentrations were generally upward or non-significant at sites in the Southwestern U.S. and non-significant in the Northwestern U.S. Trends in total nitrogen concentrations generally were downward or non-significant at sites in the Northwestern U.S., but mixed in all other regions. Regional reports include an analysis of trends (1993 – 2003) in the Columbia River and Puget Sound basins (Wise et al. 2007) and the Sacramento, San Joaquin, and Santa Ana Basins, California (Kratzer et al. 2011). In the Pacific Northwest study, point-source nutrient loads generally were a small percentage of the total catchment nutrient loads compared to nonpoint sources, with most of the monitoring sites showing decreasing trends in TN and TP, indicating that inputs from nonpoint sources of nutrients probably have decreased over time in many of the catchments (Wise et al. 2007). In the California study, most trends in flow-adjusted concentrations of nutrients in the Sacramento

Basin and Santa Anna River were downward, whereas nitrogen trends in the San Joaquin Basin were upward, especially over the 1975–2004 time period (Kratzer et al. 2011). As all of these studies note, fertilizer use, livestock manure, land-cover, atmospheric deposition, population growth, source loading (e.g., wastewater treatment plants) are all known nutrient sources that can contribute to increasing nutrient stream loads. However, basin slope, basin area, mean annual precipitation, and soil type may also interact with these sources, and flow-adjusted trends in concentration can also be complex, as there often are multiple and possibly counteracting anthropogenic factors influencing nutrient source and transport in a watershed. Without detailed knowledge of all important factors in each watershed, it may be difficult to discern the specific cause(s) of a trend.

Each of these indicators scored relatively well and there were no glaring differences (Table AP4) to discern which to use. One of the goals of the indicator selection process is to develop operationally simple indicators, so we have chosen to use the simple alternative: county-level inputs of nitrogen and phosphorus via fertilizers. We extracted data from Ruddy et al. (2006) and Gronberg & Spahr (2012) for counties in WA, OR, CA, ID, MT and WY that drain into the California Current. We only used counties that had at least 50% of its area within a CC watershed. We then summed ‘farm’ and ‘nonfarm’ input of nitrogen and phosphorus from fertilizer use across relevant counties for the years 1987 – 2006 (data available at: http://water.usgs.gov/GIS/metadata/usgswrd/XML/sir2012-5207_county_fertilizer.xml). We then extracted nationwide data for 1945 – 2001 from figure 7 in Ruddy et al. (2006). We calculated the proportion of nitrogen and phosphorus that these counties accounted for in the nationwide data for the years 1987 – 2001. We then used the average proportion and multiplied that by the nationwide data for the years 1945 – 1986 to get estimates of nitrogen and phosphorus input across an extended temporal scale. We then normalized the time series data for nitrogen and phosphorus separately, summed the normalized values for each year, and then re-normalized these sums across all years to get a single normalized index of the sum of nitrogen and phosphorus input from fertilizers across counties that drain into the California Current.

STATUS AND TRENDS

The status and trends of nutrient input into the CCLME were measured using a normalized index of the sum of nitrogen and phosphorus applied to lands as fertilizers in counties that drain into the California Current (Table AP5). Using this dataset, nutrient input was constant over the last five years of the dataset (2002 – 2006) but the short-term average was > 1SD of the long-term average of the time series (Fig. AP47). Overall, the application of nitrogen and phosphorus increased steeply since the beginning of this time series until the early 1980’s. Input of these nutrients seemed to plateau through the 1980’s and 1990’s until this most recent increase occurred. Data from 2007 – 2010 are being compiled by the USGS and should be available for updating this indicator in 2013.

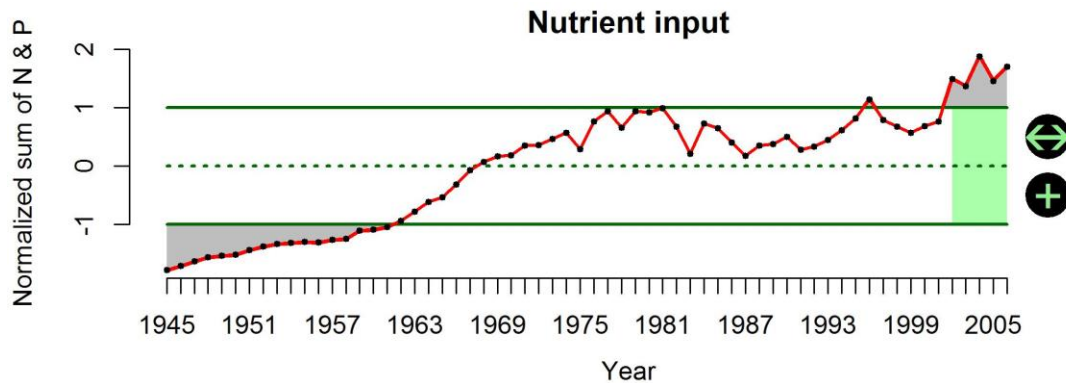


Figure AP47. Normalized index of the sum of nitrogen and phosphorus applied as fertilizers in WA, OR and CA.

OCEAN-BASED POLLUTION

BACKGROUND

The impact of ocean-based pollution is wide-spread as we include pollution from sea-going vessels and activity within ports throughout the California Current. Marine ports in the United States are major industrial centers providing jobs and steady revenue streams yet contributing significantly to pollution. Ships with huge engines running on bunker fuel without emission controls, thousands of diesel trucks per day, diesel locomotives, and other polluting equipment and activities at modern seaports cause an array of environmental impacts that can seriously affect local communities and marine and land-based ecosystems throughout a region (Bailey and Solomon 2004). As vessels transit within ports, along the coast, and along international shipping lanes, there are inevitable discharges of waste, leaks of oil and gas, loss of cargo during rough seas, and increased risk of oil spills from oil shipping vessels. Beaches close in proximity to oil shipping lanes have been observed to have high tar content related to the degree of oil pollution in the sea (Golik 1982).

The effects of oil pollution on components of the CCLME are both direct and indirect. Because seabirds and marine mammals require direct contact with the sea surface, these taxa experience high risk from floating oil (Loughlin 1994). Oiled seabirds and marine mammals lose the insulating capacity of their feathers and fur which can lead to death from hypothermia (Peterson et al. 2003). Chronic exposure to partially weathered oil is toxic to eggs of pink salmon *Oncorhynchus gorbuscha* and herring *Clupea pallasii* (Marty et al. 1997, Heintz et al. 2000). Many effects of exposure to oil and the associated polycyclic aromatic hydrocarbons (PAHs) are sublethal and have lasting effects on individual survival which may scale up to population-level responses. For example, embryos of zebrafish *Danio rerio* exposed to PAHs showed delayed changes in heart shape and reduced cardiac output (Hicken et al. 2011). Strandings of oiled seabirds have been used as an indicator of chronic oil pollution along heavily used shipping lanes in the North Sea and recent studies show declining oil-rates reflecting reduced oil spills (Camphuysen 1998, Camphuysen 2010).

In addition to the potential for pollution, other common impacts of vessel activities include vessel wake generation, anchor chain and propeller scour, vessel groundings, the introduction of invasive or nonnative species, and the discharge of contaminants and debris.

EVALUATION AND SELECTION OF INDICATORS

Ocean-based pollution was used as a measure of the risk associated with pollution that occurs and originates from ocean-use sectors. This pollution was assumed to derive from two primary sources (Halpern et al. 2009): the movement of commercial vessels (oil and gas leaks, loss of cargo, waste dumping, discharges, etc.) and activity within ports (oil and gas leaks, loss of cargo, discharges, etc.). We evaluated only one indicator for ocean-based pollution which combined data from commercial shipping activity and port volume in the CCLME (Table AP4). This indicator is well supported in the literature as a proxy for ocean-based pollution and there are long-term continuous time series of data collected by the U.S. Army Corps of Engineers.

This indicator combined the use of two previously described indicators for commercial shipping activity (volume of water disturbed during transit of vessels) and invasive species (port volume). The only difference is that for volume of water disturbed, we summed all vessel movements within ports and along the coast. Commercial shipping activity was a measure of the risk associated with ship strikes on large animals, groundings, and habitat modification, so movement within ports was not relative to that pressure. The addition of the volume of water disturbed within ports was relatively undetectable and did not alter the trends of the original data. In order to combine these two datasets into one indicator, we normalized each time series separately, summed the normalized values, and then re-normalized these sums to produce the final normalized index.

STATUS AND TRENDS

The status and trends of ocean-based pollution were measured in the CCLME using a normalized index which combined 1) the volume of water disturbed by vessels in the CCLME during transit between or within ports and 2) the annual port volume of ports in the CCLME (Table AP5). Using this indicator, ocean-based pollution has decreased over the last five years, but the short-term average is within 1SD of the long-term average (Fig. AP48). The decreasing trend in this dataset reflects current economic conditions of the shipping and port industries over the last five years; thus, this indicator is likely to increase as economic conditions improve. The predominant contributor to the trend with the “Commercial shipping activity” data is foreign vessel traffic and these data are available back to 1997, while the domestic data may be available back to 1994 if funding were available to the USACE to perform this data inquiry. These data could be integrated with the port volume data, which is available back to 1993, to increase the duration of this indicator’s time series.

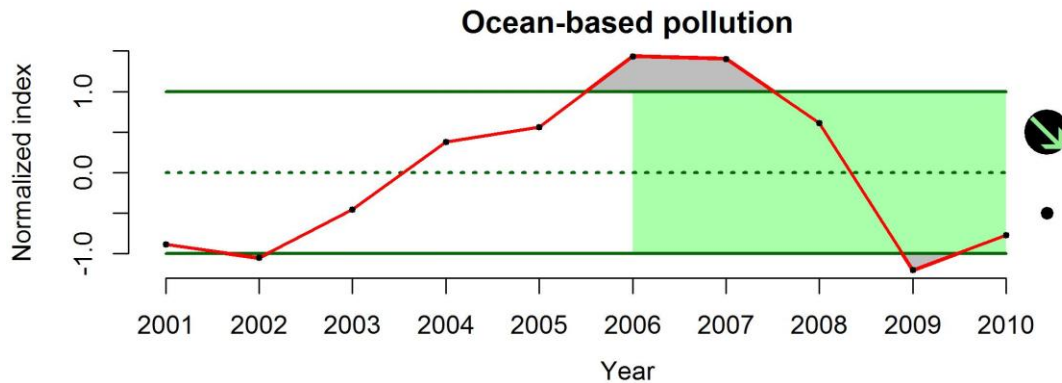


Figure AP48. Normalized index that combines the volume (millions m³) of water disturbed by vessels during transit in port and along the coast and the volume of cargo moving through U.S. ports.

OCEAN MINING

BACKGROUND

This pressure has not been evaluated to date.

EVALUATION AND SELECTION OF INDICATORS

This pressure has not been evaluated to date.

STATUS AND TRENDS

Indicators have not been evaluated in order to determine the status and trends of this pressure.

OFFSHORE OIL AND GAS ACTIVITY

BACKGROUND

The environmental risks posed by offshore exploration and production of oil and gas are well known. They include the loss of hydrocarbons to the environment, smothering of benthos, sediment anoxia, destruction of benthic habitat, and the use of explosives (Macdonald et al. 2002). Petroleum exploration involves seismic testing, drilling sediment cores, and test wells in order to locate potential oil and gas deposits (Johnson et al. 2008). Petroleum production includes the drilling and extraction of oil and gas from known reserves. Oil and gas rigs are placed on the seabed and as oil is extracted from the reservoirs, it is transported directly into pipelines. While rare, in cases where the distance to shore is too great for transport via pipelines, oil is transferred to underwater storage tanks. From these storage tanks, oil is transported to shore via tanker. According to the

Minerals & Management Service, there are 21,000 miles of pipeline on the United States outer coastal shelf (OCS). According to the National Research Council (NRC), pipeline spills account for approximately 1,900 tons per year of petroleum into US OCS waters, primarily in the central and western Gulf of Mexico. Other potential negative impacts include: physical damage to existing benthic habitats within the “drop zone”, undesired changes in marine food webs, facilitation of the spread of invasive species, and release of contaminants as rigs corrode (Macreadie et al. 2011).

However, the effects of oil rigs on fish stocks is less conclusive, with these risks balanced out by the possible enhanced productivity brought about by colonization of novel habitats by structure-associated fishes and invertebrates (e.g., rockfish, encrusting organisms, etc.) (Love et al. 2006). Decommissioned rigs could enhance biological productivity, improve ecological connectivity, and facilitate conservation/restoration of deep-sea benthos (e.g. cold-water corals) by restricting access to fishing trawlers.

Petroleum extraction and transportation can lead to a conversion and loss of habitat in a number of other ways. Activities such as vessel anchoring, platform or artificial island construction, pipeline laying, dredging, and pipeline burial can alter bottom habitat by altering substrates used for feeding or shelter. Disturbances to the associated epifaunal communities, which may provide feeding or shelter habitat, can also result. The installation of pipelines associated with petroleum transportation can have direct and indirect impacts on offshore, nearshore, estuarine, wetland, beach, and rocky shore coastal zone habitats. The destruction of benthic organisms and habitat can occur through the installation of pipelines on the sea. Benthic organisms, especially prey species, may recolonize disturbed areas, but this may not occur if the composition of the substrate is drastically changed or if facilities are left in place after production ends (Johnson et al. 2008).

Offshore oil rigs in the California Current are exclusively found in southern California. Increasing pressure to find oil on continental shelves will probably increase the risk of hydrocarbon pollution to the North Pacific: Canada (British Columbia), the U.S.A. (California), Republic of Korea and Japan have all indicated that they intend either to begin or to expand exploration on the continental shelves of the Pacific, and drilling already occurs off Alaska and California and in the East China Sea (Macdonald et al. 2002).

EVALUATION AND SELECTION OF INDICATORS

In order to estimate the temporal trend in activities related to offshore oil and gas activities off California, we evaluated two indicators: oil and gas production and the number of oil and gas wells in the CCLME (Table AP4). Both indicators have long time-series of data available and are easily used to communicate status and trends to the public and policymakers. However, the number of oil and gas wells may not likely reflect how much continuous activity surrounds each oil platform or well and thus, may not capture the variability associated with impact to the seafloor. Production of oil and gas from producing wells will capture the potential effects of continued activities (e.g., new anchorings, drilling, or maintenance of wells) on the seafloor habitat. In addition, available data for production values have a broader temporal extent (1970 – 2010) than number of wells (1981 – 2009), thus this indicator evaluated higher and will be used to measure the status and trends of this pressure.

We retrieved state and federal offshore oil and gas production data from annual reports of the California State Department of Conservation’s Division of Oil, Gas, and Geothermal Resources (ftp://ftp.consrv.ca.gov/./pub/oil/annual_reports/) for the years 1981 – 2009. A second on-line

data resource, the National Ocean Economics Program at the Monterey Institute of International Studies (http://www.oceaneconomics.org/Minerals/oil_gas.asp), was used to verify these numbers and expand the temporal extent of the production rate data series from 1970 to 2010.

STATUS AND TRENDS

The status and trends of offshore oil and gas activity in the CCLME were measured using a normalized index of the sum of oil and gas production from offshore wells in California waters (Table AP5). In order to combine oil (millions of barrels) and gas (1000's of cubic feet) production values for each year, we normalized each dataset between 0 – 1. We then summed the normalized values for each year and renormalized these sums between 0 – 1. This provided equal weight to oil and gas production to the final index. Using this dataset, offshore oil and gas activity in the CCLME has been constant over the last five years, but the short-term average was greater than 1SD below the long-term average (Fig. AP49). A rather steady decrease in oil and gas production has occurred over the last 15 years.

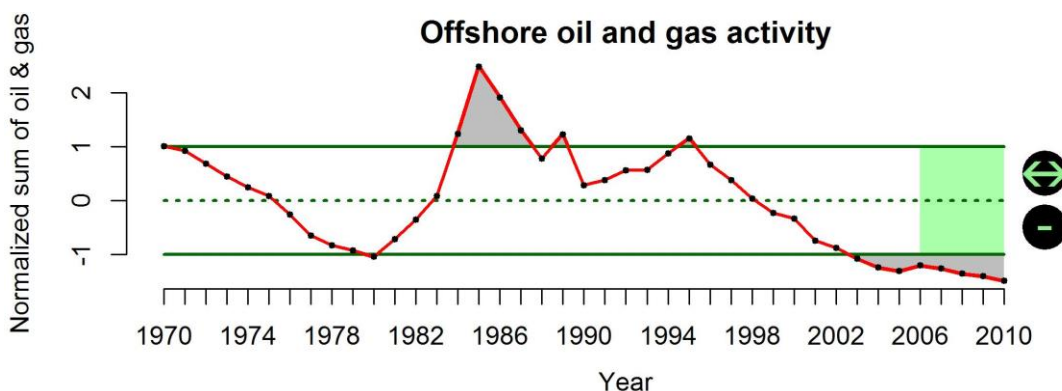


Figure AP49. Normalized index of the sum of oil and gas production from offshore wells in CA.

ORGANIC POLLUTION

BACKGROUND

Organic pollution encompass numerous classes of chemicals including pesticides, polycyclic aromatic hydrocarbons (PAHs) and other persistent organic pollutants (POPs) and is introduced to the marine environment via runoff to rivers, streams and groundwater, poor-disposal practices and the discharge of industrial wastewater. Pesticides can affect the health and productivity of biological populations in three basic ways: (1) direct toxicological impact on the health or performance of exposed individuals; (2) indirect impairment of the productivity of the ecosystem; and (3) loss or degradation of vegetation that provides physical structure for fish and invertebrates (Hanson et al. 2003, Johnson et al. 2008). For many marine organisms, the majority of effects from pesticide exposures are sublethal, meaning that the exposure does not directly lead to the mortality of individuals. Sublethal effects can be of concern, as they impair the physiological or behavioral performance of individual animals in ways that decrease their growth or survival, alter migratory

behavior, or reduce reproductive success (Hanson et al. 2003, Johnson et al. 2008), but in general the sublethal impacts of pesticides on fish health are poorly understood. Early development and growth of organisms involve important physiological processes and include the endocrine, immune, nervous, and reproductive systems. Many pesticides have been shown to impair one or more of these physiological processes in fish (Gould et al. 1994, Moore and Waring 2001). The direct and indirect effects that pesticides have on fish and other aquatic organisms can be a key factor in determining the impacts on the structure and function of ecosystems (Preston 2002).

Petroleum products, including PAHs, consist of thousands of chemical compounds which can be particularly damaging to marine biota because of their extreme toxicity, rapid uptake, and persistence in the environment (Johnson et al. 2008). PAHs have been found to be significantly higher in urbanized watersheds when compared to non-urbanized watersheds. Low-level chronic exposure to petroleum components and byproducts (i.e., polycyclic aromatic hydrocarbons [PAH]) have been shown in Atlantic salmon *Salmo salar* to increase embryo mortality, reduce growth (Heintz et al. 2000), and lower the return rates of adults returning to natal streams (Wertheimer et al. 2000). Effects of exposure to PAH in benthic species of fish include liver lesions, inhibited gonadal growth, inhibited spawning, reduced egg viability and reduced growth (Johnson et al. 2002). In general, the early life history stages of most species are most sensitive, juveniles are less sensitive, and adults least so.

Municipal wastewater treatment facilities have made great advances in treatment practices to eliminate pollutants prior to discharge, but any discharges will undoubtedly affect the quality of habitat in estuarine environments (Diaz and Rosenberg 1995, Kam et al. 2004). Several studies have shown that many benthic species increase in abundance and biomass in response to increased organic loading (Weston 1990, Savage et al. 2002, Alves et al. 2012). However, excessive nutrient enrichment can lead to hypoxia and potentially anoxic conditions, consequently leading to declines or shifts in biomass and diversity in the benthic community (Ysebaert et al. 1998, Essington and Paulsen 2010). Species richness among benthic communities has been shown to increase in relation to both temporal and spatial distance from organic loading sources (Savage et al. 2002, Wear and Tanner 2007). In addition to municipal wastewater treatment facilities, widely-distributed poorly-maintained septic systems contaminate shorelines in many places (Macdonald et al. 2002).

EVALUATION AND SELECTION OF INDICATORS

We evaluated a single indicator for organic pollution in the CCLME: toxicity-weighted concentrations of pesticides (Table AP4). The toxicity of a chemical is an important factor when trying to understand the potential effects of pollution on biological components and is widely used to weight the relative importance of specific chemicals (Toffel and Marshall 2004); thus, we did not evaluate concentrations alone as an indicator.

Concentrations of 16 pesticides in streams were assessed by the U.S. Geological Survey using data from standardized sites all across the United States (Ryberg et al. 2010). These data are easily accessible from the U.S. Geological Survey (<http://pubs.usgs.gov/sir/2010/5139/downloads/appendix6.txt>). We used data from the five sites located in WA, OR and CA and summed the recovery-adjusted concentrations across all five sites for each pesticide in each year (1993 – 2008). Because three of the pesticides (fipronil, desulfinylfipronil, and fipronil sulfide) did not have data prior to 2002, we eliminated them. We then multiplied the recovery-adjusted concentrations by their toxicity score and summed these values across all pesticides for each year. The toxicity score was calculated by dividing the pesticides Indiana Relative

Chemical Hazard Score (<https://engineering.purdue.edu/CMTI/IRCHS/>) by 100 (maximum value of the scoring system). For pesticides that were not in the IRCHS list, we used the average value of the other pesticides in our dataset. The toxicity-weighted sums were then normalized to provide the final indicator.

STATUS AND TRENDS

The status and trends of organic pollution in the CCLME were measured using a normalized index of the toxicity-weighted concentrations of 16 pesticides measured in streams in WA, OR and CA (Table AP5). Using this indicator, organic pollution has decreased over the last five years, but the short-term average remains within 1SD of the long-term average of the time series (Fig. AP50). Prior to this most recent trend, organic pollution had been increasing since the early 1990's.

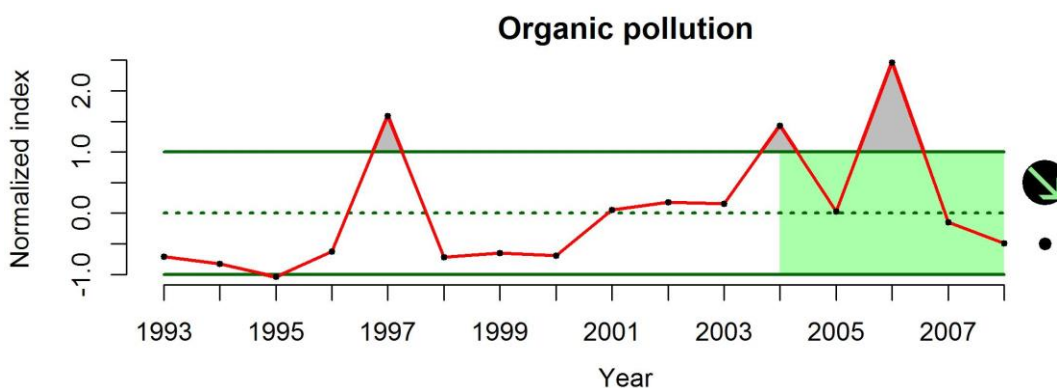


Figure AP50. Normalized index of toxicity-weighted concentrations of 16 pesticides measured in WA, OR and CA.

POWER PLANTS

BACKGROUND

Water for thermoelectric power is used in generating electricity with steam-driven turbine generators. Coastal power plants draw in huge amounts of marine water for cooling purposes, creating an area around the intake pipes where larvae and small plants are entrained. These entrainment 'plumes' will vary in size and shape depending on ocean currents and the size of the power plant. The construction and operation of water intake and discharge facilities can have a wide range of physical effects on the aquatic environment including changes in the substrate and sediments, water quality and quantity, habitat quality, and hydrology. Most facilities that use water depend upon freshwater or water with very low salinity for their needs (Johnson et al. 2008).

The entrainment and impingement of fish and invertebrates in power plant and other water intake structures have immediate as well as future impacts to estuarine and marine ecosystems (Johnson et al. 2008). Not only is fish and invertebrate biomass removed from the aquatic system, but the biomass that would have been produced in the future would not become available to the

ecosystem. Water intake structures, such as power plants and industrial facilities, are a source of mortality for managed-fishery species and play a role as one of the factors driving changes in species abundance over time. Organisms that are too large to pass through in-plant screening devices become stuck or impinged against the screening device or remain in the forebay sections of the system until they are removed by other means.

Determining the relative importance of these impacts, however, is more controversial, and may be equally dependent on year-class strength, recruitment, fishery mortality, predation, and a variety of other human facilities (dams, etc.) (Barntouse 2000). The primary approach for assessing adult-equivalent population losses at coastal power plants in California has used the “Empirical Transport Model” (ETM), which relies on estimates of power plant entrainment and source water larval populations (Steinbeck et al. 2006). Although Steinbeck et al. (2006) conclude that the ETM may be the best current approach for these impact assessments, a variety of other considerations may play a more important role in determining entrainment impacts, including effectively sampling organisms potentially affected by entrainment (often determined by life history, including spawning location and timing), sampling frequency, determining source water areas potentially affected, and design, location, and hydrodynamics of the power plant intake structure. Helvey and Dorn (1987) examined the selective removal of reef fish associated with an offshore cooling-water intake structure, and found that removal was a selective process governed by species’ behavioral characteristics associated with the intake currents and visibility (fish may not be capable of rheotropic responses when illumination falls below a critical threshold. Diurnally active species seeking benthic cover at night were least susceptible to intake removal. Diurnally active species that hover in the water column at night and predators that periodically feed at twilight and evening hours (e.g., *Sebastes paucispinis*) were more susceptible to removal. Nocturnally active transient species, such as *Seriphus politus* and *Engraulis mordax*, were most susceptible to removal (Helvey and Dorn 1987).

EVALUATION AND SELECTION OF INDICATORS

We evaluated two indicators of power plant activity in the CCLME: 1) average daily saline water withdrawal volumes and 2) daily entrainment mortality (Table AP4). The largest total thermoelectric withdrawals on the west coast are in California, where nearly all of the water was withdrawn from marine surface waters for use in once-through cooling systems (Kenny et al. 2009). Washington and Oregon thermoelectric power withdrawals rely almost exclusively on fresh surface waters. In 2005, the total daily water withdrawals for thermoelectric power generation from all west coast states combined (WA, OR, CA) equaled over 13 Bgal/d (billion gallons per day), with the vast majority (96%; 12.6 Bgal/d) attributed to CA marine surface water withdrawals. Over the course of record-keeping, marine surface water withdrawals from California have consistently represented more than 80% of thermoelectric water withdrawals. The USGS has conducted water-use compilations in the United States every 5 years since 1950 (<http://water.usgs.gov/watuse/50years.html>), and thermoelectric power has represented the largest total category of water withdrawals in every compilation since 1960 (Hutson et al. 2005, Kenny et al. 2009). Withdrawals by thermoelectric-power plants have ranged from a low of 40 Bgal/d during 1950 to a high of 210 Bgal/d in 1980. In 2005, thermoelectric water withdrawals totaled 201 Bgal/d and comprised 49 percent of total water use in the country. Declines in thermoelectric-power water withdrawals from 1980 to present are primarily a result of Federal legislation requiring stricter water-quality standards for return flow and by limited water supplies in some areas of the United States. Consequently, power plants have increasingly been built with or

converted to closed-loop cooling systems or air-cooled systems instead of using once-through cooling systems. By 2000, an alternative to once-through cooling was used in about 60 percent of the installed steam-generation capacity in the power plants (Hutson et al. 2005).

There is a long history of studying and reporting impacts of cooling systems on fish populations, especially the Hudson River and other coastal estuaries along the mid-Atlantic (Barnthouse 2000). In California, calculations of daily entrainment mortality have been limited to a few power plants; historical data are limited and time series information is generally lacking. Furthermore, the uncertainties associated with estimating larval durations and hydrodynamics used in estimating the size of the source water populations make estimating variance for ETM problematic (Steinbeck et al. 2006).

Primarily due to data considerations (Table AP4), we selected average daily water withdrawals to estimate the potential entrainment impact of coastal power plants. We extracted the average daily withdrawal volumes (millions of gallons per day converted to millions of m^3 per day) of saline water over time from all thermoelectric power plants on the west coast of North America (Pacific Northwest and California regions, from Kenny et al. (2009) and other previous USGS water use reports (<http://water.usgs.gov/watuse/50years.html>). The temporal extent of these data ranges from 1955 to 2005 and the reporting interval is every five years.

STATUS AND TRENDS

The status and trends of power plants in the CCLME were measured using the volume (millions of m^3) of saline water withdrawn daily by thermoelectric power plants in WA, OR and CA (Table AP5). Because these data were sampled every 5 years, we interpolated the annual value over the last five years (asterisks in Fig. AP51) assuming a linear relationship between the last two data points in order to keep the short-term status (most recent five years) consistent with the other pressure indicators. The mean and standard deviation of the dataset were calculated using the original dataset. Power plant activity was constant over the last five years of the dataset (2000 – 2005), but the short-term average was >1SD above the long-term average (Fig. AP51). Trends of water withdrawals by thermoelectric power plants have been constant or decreasing across the U.S. since the 1980's (Kenny et al. 2009), so the CCLME may have slightly elevated its power plant activity compared to the rest of the U.S. in the early 2000's.

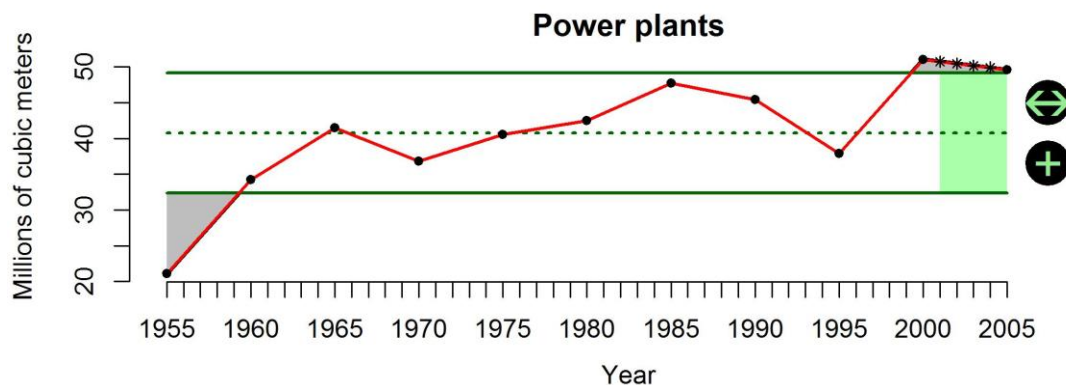


Figure AP51. Daily saline water withdrawals (millions m^3) from thermoelectric power plants in CA, OR and WA.

RECREATIONAL USE

BACKGROUND

People visiting beaches and coastal areas can impact intertidal and nearshore ecosystems through direct trampling or by disturbing or displacing species that would normally use those locations (Halpern et al. 2009). This may be particularly important to species which inhabit intertidal zones their entire lives or for species that reproduce or rest on populated beaches (Moffett et al. 1998, McClenachan et al. 2006, Defeo et al. 2009). Species which represent some value as a source of food (e.g., shellfish) or collections (e.g. seashells) will also be impacted with increases in beach visitations.

EVALUATION AND SELECTION OF INDICATORS

We evaluated only one indicator of recreational use: beach attendance. This indicator evaluated highly in most criteria (Table AP4) because it was used in previous studies as an indicator of direct human impact to intertidal and nearshore ecosystems (Halpern et al. 2008, Halpern et al. 2009). However, the use of state beaches and parks may not necessarily reflect how many people are actually spending time walking around on the beach or in the intertidal zones, but rather may reflect time spent at upland areas or simply sitting in their vehicles. There is also recent evidence that the methodologies used to calculate beach attendance by state agencies overestimate actual attendance in a non-random fashion (King and McGregor 2012).

For California, we extracted total visitor attendance at 48 California state parks identified as “State Beach” from the California State Park System Annual Statistical Reports: 2001 -2010 (http://www.parks.ca.gov/?page_id=23308). For Oregon, the only measure of annual beach attendance is collected by the Oregon Parks and Recreation Department’s Stewardship Division for the years 2002 – 2011. This estimate is measured using automated car counters in the parking lots of coastal state parks. These estimates are based on the assumption that there are on average four occupants per vehicle (based on results of a statewide visitor survey). These measures are likely an overestimate of actual pressure on the associated beaches as some people use the parking lots and do not go to the beach. For Washington, the Washington State Parks and Recreation Commission collects attendance data at parks with ocean beach access and these data are available in annual “Attendance Reports”. We limited these datasets to years in which data were available for all three states (2002 – 2010) and to parks/beaches that were open and censused in all years (i.e. if a state park was closed at some point during the time series, this park was excluded from the analysis).

STATUS AND TRENDS

The status and trends of recreational use were measured using annual estimates of beach attendance at state parks and beaches in WA, OR and CA (Table AP5). Using this dataset, we found that direct human impact has decreased significantly over the last five years, but the short-term mean is still within 1 SD of the long-term mean of the dataset (Fig. AP52).

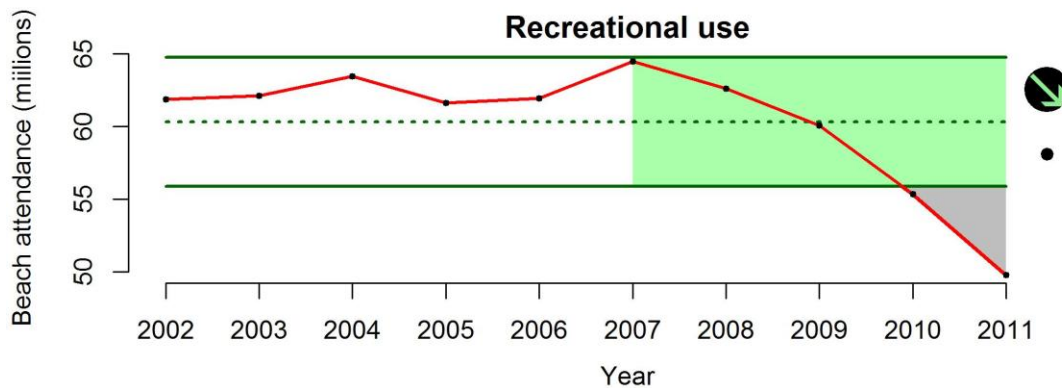


Figure AP52. Annual beach attendance (millions of persons) at state parks and beaches with access points to a beach in WA, OR and CA.

SEAFOOD DEMAND

BACKGROUND

The global population continues to increase and seafood is one of the most important sources of protein for humans all over the world, so demand for edible fisheries products will continue to be a strong pressure on the world's oceans (Garcia and Rosenberg 2010). In addition to the underlying driver of population growth, the most recent report of the Dietary Guidelines for Americans has recommended Americans more than double their intake of seafood due to a variety of health benefits (DGAC 2010). Depending on the response and potential change in dietary behaviors by humans, pressure could increase greatly for the production of high-quality seafood. However, the production of world capture fisheries has been relatively constant since the 1980's (NRC 2006), and there is little room for increase. The world's demand for seafood has thus, become more dependent on aquaculture production, which has been growing at about 8% annually, making it the fastest growing form of food production in the world. However, much of the feed for the aquaculture (and pig and poultry) industry is derived from forage fish species such as anchovy and capelin (Hannesson 2003). This pressure to catch fish in order to grow fish may not necessarily result in a net increase in the production of edible fish. Another common use of fisheries products is for use as fertilizers.

This pressure has obvious effects on the biological components of the CCLME through direct removals of individuals from the benthic and pelagic communities. Direct fishery removals, however, also have a whole host of indirect effects that have been discussed under the Fisheries Pressures.

EVALUATION AND SELECTION OF INDICATORS

We identified two primary indicators of seafood demand: total consumption and per capita consumption (Table AP4). Both indicators are published in NOAA's "Fisheries of the United States" annual reports to describe the utilization of fisheries products (<http://www.st.nmfs.noaa.gov/st1/publications.html>). Total edible and non-edible seafood demand evaluates higher (Table AP4) because fundamentally total consumption provides a concrete estimate

of what is being used, whereas per capita consumption is simply based on the total consumption estimates divided by the population of the U.S.

We retrieved total consumption estimates (billion pounds) of total (imports and commercial landings) edible and non-edible seafood from each of the Fisheries of the United States annual reports which provided data from 1962 – 2010.

STATUS AND TRENDS

The status and trends of seafood demand in the CCLME were measured using total consumption of edible and non-edible fisheries products (Table AP5). Using this dataset, seafood demand has been relatively constant over the last five years (Figs. AP53), but the short-term average was greater than 1SD of the long-term average. With total demand already at historic levels, increasing populations, and recommendations by the U.S. Dietary Guidelines to increase our intake of seafood, this indicator will likely increase over the next few years. If per capita consumption increases, as recommended, total consumption could increase dramatically as human populations continue to increase globally as well as in the CCLME. In many ways, seafood demand in states or countries outside of the CCLME will have a large impact on the trends of this indicator and may limit the ability of regional or national managers to alter the effects of this pressure.

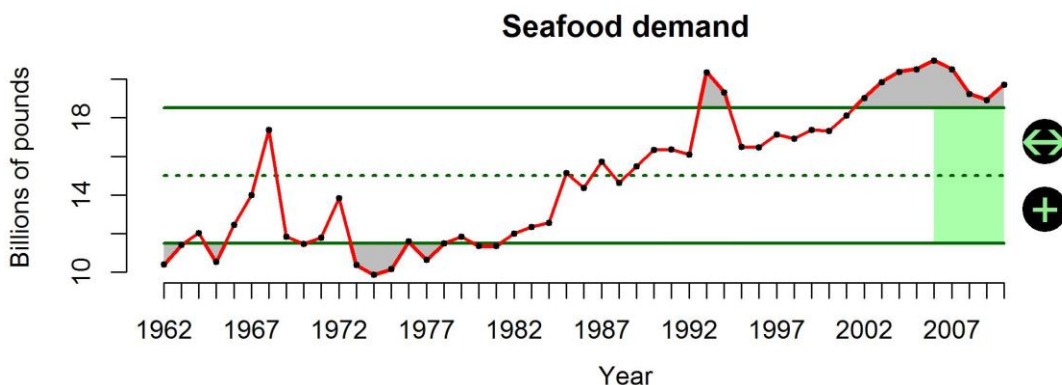


Figure AP53. Total consumption of edible and non-edible fisheries products across the United States.

SEDIMENT INPUT

BACKGROUND

Sediment is a natural component in water bodies and the uses they support, but can also impair them in many ways (USEPA 1999b). Excessive sediments in waterways can cause direct physical harm to organisms (e.g. clogged gills), as well as impairment of aquatic feeding, rearing, spawning, and refuge habitats. As well, sediment deficits can result in stream channel scour and destruction of other habitat features. As a result, the federal Clean Water Act requires states, territories, and authorized tribes to identify and list impaired waters every two years and to develop total maximum daily loads (TMDLs) for sediment in these waters, with oversight from the U.S. Environmental Protection Agency. TMDLs establish the allowable pollutant loadings, thereby providing the basis for establishing water quality-based controls (USEPA 1999b).

Rivers are important conduits of large amounts of particulate and dissolved minerals and nutrients to the oceans, and play a key role in the global biogeochemical cycle (Dai et al. 2009). Humans are simultaneously increasing the river transport of sediment and dissolved constituents through soil erosion activities, and decreasing this flux to the coastal zone through sediment retention in reservoirs (Syvitski et al. 2005, Milliman et al. 2008). The net result is a global reduction in sediment flux by about 1.4 BT/year over pre-human loads. Rivers are globally getting dirtier and would otherwise move more sediment to the coast if not for the impact of reservoirs. The seasonal delivery of sediment to the coast affects the dynamics of nutrient fluxes to the coast and has serious implications to coastal fisheries, coral reefs, and seagrass communities (Syvitski et al. 2005). One example includes a reduction in natural dissolved silicate loads, which translates into silicon limitation in the coastal zone that discourages diatom blooms and favors nuisance and toxic phytoplankton, thereby compromising the integrity of coastal food webs (Vorosmarty and Sahagian 2000). Coastal retreat, which is directly influenced by the reduction of river-supplied sediment, has major implications for human habitat because >37% (2.1 billion people in 1994) of the world's population live within 100 km of a coastline (Syvitski et al. 2005). Dam removal restores the natural sediment transport regime and has become an increasingly adopted strategy to manage the environmental costs of these structures (Graf 1999, The Heinz Center 2002).

Changes in sediment supply can greatly influence the benthic environment of coastal estuaries, coral reefs, and seagrass communities, and are intimately tied to nutrient fluxes in these systems (Syvitski et al. 2005). Sediment delivery rates also affect harbor maintenance and pollutant burial or resuspension. Decreases in sediment input are largely the result of river damming or diversions, which directly influence the rate of coastal retreat. Dams affect the physical integrity of watersheds by fragmenting the lengths of rivers, changing their hydrologic characteristics, and altering their sediment regimes by trapping most of the sediment entering the reservoirs and disrupting the sediment budget of the downstream landscape (The Heinz Center 2002, Johnson et al. 2008). Because water released from dams is relatively free of sediment, downstream reaches of rivers may be altered by increased particle size, erosion, channel shrinkage, and deactivation of floodplains (The Heinz Center 2002). The consequence of reduced sediment also extends to long stretches of coastline where the erosive effect of waves is no longer sustained by sediment inputs from rivers (World Commission on Dams 2000). The effects to fishes of a reduced sediment regime would be indirect and primarily experienced through the long-term loss of soft-bottom habitat features and coastal landforms and/or changes to benthic habitat composition.

Increases in sediment input are largely due to land use practices that increase erosion rates (e.g., deforestation, wetland drainage, mining) or human activities in or near aquatic habitats (e.g., dredging) that re-suspend bottom sediments and create turbid conditions (Syvitski et al. 2005). Suspended sediments can elicit a variety of responses from aquatic biota; these responses may range from an active preference for turbid conditions, presumably to facilitate feeding and avoidance behaviors, to detrimental physical impacts that may result in egg abrasion, reduced bivalve pumping rates, and direct mortality (Wilber and Clarke 2001). Much of the available data on biological effects on organisms come from bioassays that measure acute responses and require high concentrations of suspended sediments to induce the measured response, usually mortality (Wilber and Clarke 2001). Although anadromous salmonids have received much attention, little is known of behavioral responses of many estuarine fishes to suspended sediment plumes. There is a high degree of species variability in response to sedimentation; reports of “no effect” were made at concentrations as great as 14,000 mg/L for durations of 3 d and more (oyster toadfish and spot) and mortality was observed at a concentration/duration combination of 580 mg/L for 1 d (Atlantic silversides). For both

salmonid and estuarine fishes, the egg and larval stages are more sensitive to suspended sediment impacts than are the older life history stages.

EVALUATION AND SELECTION OF INDICATORS

Two indicators of sediment input were evaluated: dam/reservoir storage area and suspended sediment loading (Table AP4). To estimate the temporal change in sediment decrease, we focused on dams as the key feature affecting this change, per Halpern et al. (2008). Construction of large dams peaked in the 1970's in Europe and North America (World Commission on Dams 2000). Today most activity in these regions is focused on the management of existing dams, including rehabilitation, renovation, and optimizing the operation of dams for multiple functions. The history of total reservoir storage area by US water resource region was summarized from the early 1900's to the early 1990's by Graf (1999), based on data from the U.S. Army Corps of Engineers (1996). Since these data are no longer available electronically from the USACE, we compiled total reservoir storage in 10^9 cubic m over time (year of construction) for the California and PNW water resource regions. Freshwater storage was obtained from state agency databases, which include information on construction date and impoundment area/volume for all dams (California: <http://cdec.water.ca.gov/misc/resinfo.html>; Idaho: <http://www.usbr.gov/projects/FacilitiesByState.jsp?StateID=ID>; Oregon: <http://www.usbr.gov/projects/FacilitiesByState.jsp?StateID=OR>; Washington: <https://fortress.wa.gov/ecy/publications/summarypages/94016.html>). Note that the data compiled using this summary does not precisely replicate the Graf (1999) data, but the temporal trends are comparable.

Another more data-intensive approach would involve estimating sediment loading from surface waters using publicly available data on sediment concentrations and flow rates from various US watersheds sampled by the USGS and various state and local agencies. Sediment data are publicly accessible through the online USGS National Water Information System (NWIS) database at (<http://nwis.waterdata.usgs.gov/usa/nwis/qwdata>). The majority of data contained in the NWIS database are from water samples collected using standard methods described in U.S. Geological Survey (variously dated). USGS flow data can be accessed from (http://nwis.waterdata.usgs.gov/nwis/dv/?referred_module=sw). Suspended sediment loading can be estimated at various time increments (e.g., daily, annual) using LOADEST, a USGS program that finds a best fit data model for flux as a function of discharge. The Yale University interface LOADRUNNER (<http://environment.yale.edu/loadrunner/>) calculates daily, monthly, and annual fluxes, from these USGS water quality sample and streamflow data sources. Currently, we have queried data from the USGS surface water database (<http://infotrek.er.usgs.gov/apex/f?p=NAWQA:HOME:5572182579967972>) for suspended sediment (SS) levels [mg/L] from sampled Pacific coastal basins from 1991-2011. Flow adjusted trends in concentration can be complex, as there often are multiple and possibly counteracting anthropogenic factors influencing sediment source and transport in a particular watershed.

A recent report from USGS summarizes the annual mean loads for SS in the Puget Sound and Columbia River basins using the USGS computer program Load Estimator (LOADEST), which uses a linear regression model that incorporates flow, time, and seasonal terms to estimate loads of mass over specified time periods (for this study, annual loads) (Wise et al. 2007). During water year 2000, considered an average streamflow year in the Pacific Northwest, the Columbia River discharged about 14,000 tons per day of SS to the Pacific Ocean. For most catchments between water years

1993-2003, the net change in non-hydrologic characteristics (land use and other human activities) was not great enough to cause any significant ($p \leq 0.05$) flow-adjusted trend in concentration (FATC) for suspended sediment (SS). Nineteen of the 48 sites available for SS trend analysis had significant FATC for SS (4 increasing, 15 decreasing), seven sites showed significant trend in load for SS (1 increasing, 6 decreasing), and more than 65 percent of the sites had decreasing (but not necessarily significant) FATC and trend in load for SS. There is currently no comparable analysis yet available for California basins.

We selected dam/reservoir storage area as our proxy for sediment input, primarily based on data considerations (Table AP4); furthermore, the net global reduction in sediment flux to coastal areas is primarily due to reservoir construction (Syvitski et al. 2005).

STATUS AND TRENDS

The status and trends of sediment input in the CCLME were measured using the total reservoir impoundment volume (millions m^3) of dams along rivers in WA, OR and CA (Table AP5). Using this dataset, sediment input has been constant over the last five years and the short-term average was greater than 1SD of the long-term average of the time series (Fig. AP54). Increases in reservoir impoundment volume lead to less sediment making its way to the deltas of the dammed rivers; thus, increases in this indicator represent decreases in sediment input to estuarine and marine habitats. This is one of the longest datasets for non-fisheries pressures, so changes in the long-term trend will only occur in the future if large changes occur over the next few decades. In contrast, many of the other indicators have short time series, so relatively smaller changes over just a few years will impact the short- and long-term trends and thus our interpretation of the current status of these indicators.

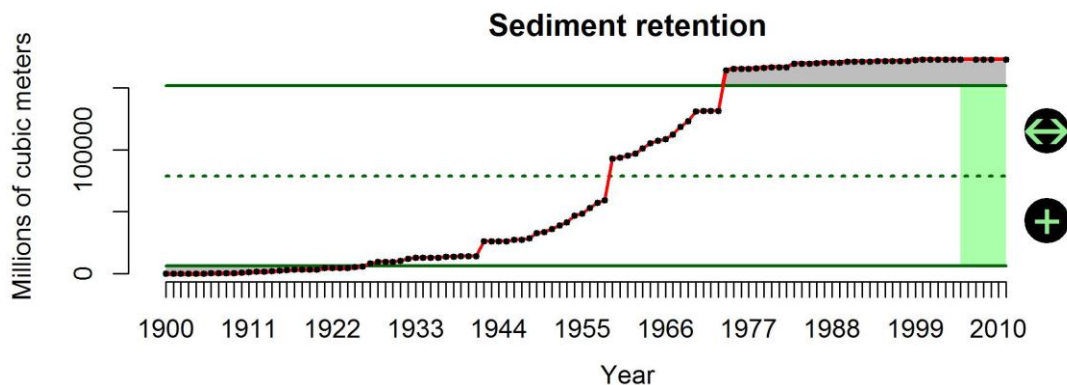


Figure AP54. Volume (millions m^3) of freshwater impoundments in WA, OR and CA (increasing freshwater storage is a proxy for decreasing sediment input).

TOURISM

BACKGROUND

This pressure has not been evaluated to date.

EVALUATION AND SELECTION OF INDICATORS

This pressure has not been evaluated to date.

STATUS AND TRENDS

Indicators have not been evaluated in order to determine the status and trends of this pressure.

LINKAGES BETWEEN DRIVERS AND EBM COMPONENTS

By definition, anthropogenic pressures on the ecosystem are based on human activities and thus the ultimate driver behind most of these pressures is human population growth. The status and trends of individual pressures are then modified by technological advances, management practices and regulatory actions. For the CCLME, the demand for edible and non-edible fisheries products and increased interest in harnessing natural resources (e.g., oil and gas, tidal energy, aquaculture, ocean mining) has been and is predicted to continue increasing into the foreseeable future. These drivers will ultimately affect the biological components of the CCLME in ways we do not fully understand. Some linkages are direct, such as fisheries removals, habitat destruction and mortality caused by oil spills, while others may be indirect such as light pollution which increases the efficiency of visual predators along the coast, which subsequently changes predator/prey dynamics and ultimately affect community structure (Longcore and Rich 2004).

The linkage between fisheries and several IEA EBM components is direct: fishery removals decrease abundance of targeted fisheries as well as some protected species via directed removals and bycatch. The Pacific Fishery Management Council uses biological reference points to determine whether a stock is in an overfished state, and whether overfishing is occurring. For groundfish, for instance, the former is determined using an estimated depletion level, which is the ratio of spawning stock output (number of eggs or embryos) in the fished condition, to the spawning output in the unfished condition. The latter is determined by a fishing mortality rate (F), expressed based on spawning potential ratio (SPR). This ratio is the number of eggs produced by an average recruit over its lifetime when the stock is fished, divided by the same metric when the stock is unfished. The SPR is based on the principle that certain proportions of fish have to survive in order to spawn and replenish the stock at a sustainable level. When removals or fishing mortality exceed established reference points, management measures are implemented to correct the issue. There had been significant declines in a number of groundfish species managed by the Pacific Fishery Management Council. Since implementing the Magnuson-Stevens Act (MSA) of 1976, the Sustainable Fisheries Act (SFA) of 1996, and the reauthorization of MSA in 2006, many species have increased their abundance toward levels where they are not considered overfished, and overfishing of these species is not occurring (Miller et al. 2009). For example, lingcod, which dropped below 10% of its unfished

biomass in 1986, was fully rebuilt in 2005, four years earlier than the target year established in the species rebuilding plan (Hamel et al. 2009). Based on the most recent rebuilding analyses, all groundfish species that are still considered overfished exhibit upward trends, with three species (yelloweye rockfish, bocaccio and darkblotched rockfish) being ahead of their rebuilding plan schedules (Field 2011, Stephens 2011, Taylor 2011).

For most of the non-fisheries related pressures, there are few direct mechanistic linkages between pressures and effects on population growth of specific populations (with the notable exception of studies showing population-level effects from oil exposure). This is undoubtedly a function of natural fluctuations in most populations, imprecise estimates of populations across time and space, and a mismatch in the scale at which specific pressures act upon specific populations. Thus, our ability to detect and partition effects of specific contaminants is made even more difficult. In addition, none of these pressures act upon the ecosystem in a vacuum (i.e. many pressures are acting simultaneously on populations), and we have little understanding about whether the cumulative effects of multiple pressures will be additive, synergistic or antagonistic on populations of interest. This makes detecting direct links even more difficult. Moreover, these anthropogenic pressures will interact with the underlying effects of climatic and oceanographic pressures. These types of interactions can be modeled with “end-to-end” ecosystem models (e.g., Atlantis; Fulton et al. 2011) that have been developed over the last decade and we need to develop creative methods in the field to test the validity of these models’ hypotheses and increase managers’ confidence in decision making.

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ECOLOGICAL INTEGRITY

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TABLE OF CONTENTS (EN)

| | |
|---|-----|
| Executive Summary | 184 |
| Detailed Report..... | 186 |
| Background - ecological integrity | 186 |
| Indicator selection process | 186 |
| Top ranked indicators..... | 193 |
| Final suite of indicators | 202 |
| Status and Trends: Ecological Integrity | 205 |
| Major findings..... | 205 |
| Data analysis and presentation..... | 205 |
| Mean trophic level (groundfishes and coastal pelagic fishes)..... | 207 |
| Interpretation of time series figures..... | 207 |
| Scavenger biomass (groundfishes and crabs) | 208 |
| Simpson diversity (groundfishes, coastal pelagic fishes, seabirds & copepods)..... | 209 |
| Species richness and density (groundfishes, coastal pelagic fishes, seabirds, copepods).. | 214 |
| Northern copepod biomass anomaly..... | 220 |
| Data sources and Methodology | 222 |
| Groundfishes..... | 222 |
| Coastal Pelagic Fishes..... | 227 |
| Seabirds..... | 228 |
| Copepods..... | 228 |
| Northern copepod biomass anomaly..... | 229 |
| References cited..... | 230 |

LIST OF TABLES AND FIGURES (EN)

| | |
|---|-----|
| Short and long-term status of indicators of Ecological Integrity for the California Current Large Marine Ecosystem. Prior to plotting, time series were normalized to place them on the same scale. The short-term trend indicates whether the indicator increased or decreased over the last 5 years of the time series relative to the long-term mean. The long-term axis compares the mean of the last 5 years to the mean of the full time series. Changes or differences of more than 1.0 standard deviation of the full time series (dotted lines) are considered to show an effect. Cop = copepod, Cpel = coastal pelagics, GF = groundfishes, SB = seabirds, anom = anomaly, Rich = species richness, Simp = Simpson diversity, Den = species density, Scav bio = scavenger biomass (GF and crabs), NS = nearshore (0-2 km), shelf=shelf waters (2070 km), s = summer, w = winter. | 184 |
| Table EN1. Summary of ecological integrity indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, jellyfish biomass status and trends has peer-reviewed literature supporting four out of five primary considerations criteria..... | 187 |
| Table EN2. Top-ranked indicators for Ecosystem Integrity. | 204 |
| Figure EN.S.1. Short and long-term status of indicators of Ecological Integrity for the California Current Large Marine Ecosystem. Prior to plotting, time series were normalized to place them on the same scale. The short-term trend indicates whether the indicator increased or decreased over the last 5 years of the time series relative to the long-term mean. The long-term axis compares the mean of the last 5 years to the mean of the full time series. Changes or differences of more than 1.0 s.d. of the full time series (dotted lines) are considered to show an effect. Cop = copepod, Cpel = coastal pelagics, GF = groundfishes, SB = seabirds, anom = anomaly, Rich = species richness, Simp = Simpson diversity, Den = species density, Scav bio = scavenger biomass (GF and crabs), NS = nearshore (0-2 km), shelf=shelf waters (2070 km), s = summer, w = winter. | 206 |
| Figure EN.S.3. Estimated biomass (CPUE) of groundfish and crab scavengers for the west coast shelf and slope from 2003-2011. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov). | 209 |
| Figure EN.S.4. Simpson diversity ($1-\lambda$) for west coast groundfishes from 2003 to 2011. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness | 210 |
| Figure EN.S.5. Simpson diversity ($1-\lambda$) for coastal pelagic fishes in the northern California current from 1998 to 2011..... | 210 |
| Figure EN.S.6. Simpson diversity for nearshore (within 2 km of shore) seabirds in the northern California current (NCC). Data are those birds observed and quantified while resting on water. Data courtesy of Jen Zamon (jen.zamon@noaa.gov). | 211 |
| Figure EN.S.7. Simpson diversity for nearshore (within 2 km of shore) seabirds in the northern California current (NCC). Data are those birds observed and quantified while actively flying. Data courtesy of Jen Zamon (jen.zamon@noaa.gov). | 212 |

| | |
|---|-----|
| Figure EN.S.8. Simpson diversity in May for seabirds on the shelf and shelf break (2 – 70 km off shore) in the northern California current (NCC). Data courtesy of Jen Zamon (jen.zamon@noaa.gov)..... | 212 |
| Figure EN.S.9. Simpson diversity in June for seabirds on the shelf and shelf break (2 – 70 km off shore) in the northern California current (NCC). Data courtesy of Jen Zamon (jen.zamon@noaa.gov)..... | 213 |
| Figure EN.S.10. Time series of Simpson diversity ($1-\lambda$) from 1997 – 2012 for the winter (Oct– April) for west coast copepods in the northern California Current (NCC). Data courtesy of Bill Peterson (bill.peterson@noaa.gov). | 213 |
| Figure EN.S.11. Time series of Simpson diversity ($1-\lambda$) from 1996 – 2011 for the summer (May – Sept) for west coast copepods in the northern California Current (NCC). Data courtesy of Bill Peterson (bill.peterson@noaa.gov). | 214 |
| Figure EN.S.12. Species richness for west coast groundfishes from 2003 -2011. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov). | 215 |
| Figure EN.S.13. Area-weighted mean number of groundfish species per 12 trawls for 2003-2011 from the from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth | 215 |
| Figure EN.S.14. Mean number of groundfish individuals per trawl 2003-2011 from the from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness | 216 |
| Figure EN.S.15. Median number of groundfish individuals per trawl 2003-2011 from the from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov). | 216 |
| Figure EN.S.16. Mean number of species per sample for coastal pelagic fishes in the northern California current from 1998 to 2011..... | 217 |
| Figure EN.S.17. Taxon richness for seabirds observed while resting on water in the nearshore (< 2km from shore) for 2004-2012 for the northern California current (NCC). Data rarefied to 442 individuals (minimum seen in the time series) using individual-based rarefaction (Hurlbert 1971). Data courtesy of Jen Zamon (jen.zamon@noaa.gov)..... | 218 |
| Figure EN.S.18. Taxon richness for seabirds observed while flying in the nearshore (< 2km from shore) for 2005-2012 for the northern California current (NCC). Data rarefied to 442 individuals (minimum seen in the time series) using individual-based rarefaction (Hurlbert 1971). Data courtesy of Jen Zamon (jen.zamon@noaa.gov). | 218 |
| Figure EN.S.19. Taxon richness for seabirds May on the shelf and shelf break (2-70km from shore) for 2003-2012 for the northern California current (NCC). Data rarefied to 442 individuals (minimum seen in the time series) using individual-based rarefaction (Hurlbert 1971). Data courtesy of Jen Zamon (jen.zamon@noaa.gov)..... | 219 |
| Figure EN.S.20. Taxon richness for seabirds May on the shelf and shelfbreak (2-70km from shore) for 2003-2012 for the northern California current (NCC). Data rarefied to 2386 individuals (minimum seen in the time series) using individual-based rarefaction (Hurlbert 1971). Data courtesy of Jen Zamon (jen.zamon@noaa.gov). | 219 |
| Figure EN.S.21. Copepod species richness for the winter in waters off Oregon in the northern California Current (NCC) from 1997-2012. Data courtesy of Bill Peterson (bill.peterson@noaa.gov)..... | 220 |

| | |
|--|-----|
| Figure EN.S.22. Copepod species richness for the summer in waters off Oregon in the northern California Current (NCC) from 1997-2012. Data courtesy of Bill Peterson (bill.peterson@noaa.gov)..... | 220 |
| Figure EN.S.23. Northern copepod biomass anomaly for 1996-2012 in the waters off of Oregon during the winter (Oct-April). Data courtesy of Bill Peterson. | 221 |
| Figure EN.S.24. Northern copepod biomass anomaly for 1996-2012 in the waters off of Oregon during the summer (May - Sept). Data courtesy of Bill Peterson. | 221 |
| Table EN.S.1. Depth and latitude bins uses in the area-weighted analyses showing total area of the bins, corresponding weights and total number of trawls from 2003 – 2011. See text for depth and latitude borders. | 223 |
| Figure EN.S.25. Mean trophic level for west coast groundfishes from 2003 – 2011. MTL was calculated for survey data without adjusting for sampling effort in different depth x latitude strata. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov). | 224 |
| Table EN.S.2. Groundfish and decapod taxa included in the quantification of scavenger biomass 2003-2011. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov). | 224 |
| Figure EN.S.26. Mean number of groundfish species per trawl for 2003-2011 from the West Coast Groundfish Bottom Trawl Survey. Species number values were not subjected to rarefaction. Data courtesy of Beth Horness (Beth.Horness@noaa.gov). | 226 |
| Figure EN.S.27. (A) Number of species identified in any given year by the trawl survey, (B) cumulative number of species identified by the trawl survey. | 227 |

OVERVIEW

Indicators of ecological integrity suggest generally neutral to good conditions in the California Current; however, changes in the loss of some predator fishes and low seabird diversity warrant close monitoring.

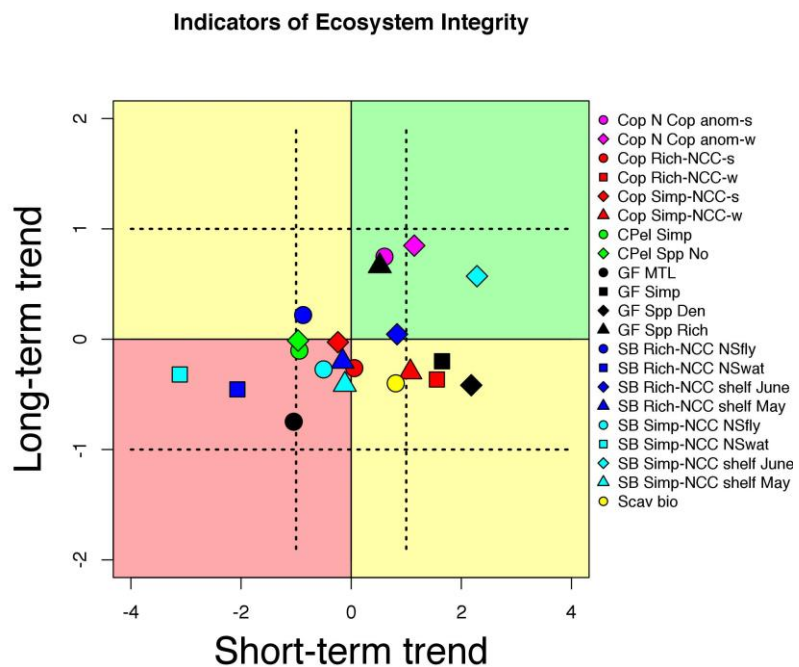
EXECUTIVE SUMMARY

Ecological integrity refers to indicators that describe individual components within an ecosystem and the relative extent of their potential interactions. We identified and evaluated 40 potential indicators of ecological integrity across a variety of species and foraging guilds, using the ecological literature as a basis for their rankings. We selected indicators that ranked in the top quartile to track two aspects of the California Current Large Marine Ecosystem (CCLME):

- trophic structure: (mean trophic level, scavenger biomass, and the northern copepod biomass anomaly) and
- biodiversity (Simpson's diversity, species richness).

In addition to the indicators reported in this section, indicators on oceanographic features, coastal pelagic- and ground-fishes, and protected species (marine mammals, seabirds, and Pacific salmon) provide information that can influence ecological integrity.

The spatial extent of CCLME data coverage varies among taxa. The groundfish data span the US west coast (~ 32-48 °N, ~50-1200 m) and conclusions related to this data set (mean trophic level, scavenger biomass, species richness, species density, and Simpson diversity) are applicable to the full CCLME. For coastal pelagic fishes, seabirds, and copepods, however, the data analyzed here are currently limited to the northern California Current; future versions of the assessment will integrate data spanning the entire coast.



Short and long-term status of indicators of Ecological Integrity for the California Current Large Marine Ecosystem.

Prior to plotting, time series were normalized to place them on the same scale. The short-term trend indicates whether the indicator increased or decreased over the last 5 years of the time series relative to the long-term mean. The long-term axis compares the mean of the last 5 years to the mean of the full time series. Changes or differences of more than 1.0 standard deviation of the full time series (dotted lines) are considered to show an effect. Cop = copepod, Cpel = coastal pelagics, GF = groundfishes, SB = seabirds, anom = anomaly, Rich = species richness, Simp = Simpson diversity, Den = species density, Scav bio = scavenger biomass (GF and crabs), NS = nearshore (0-2 km), shelf=shelf waters (2070 km) s = summer w = winter

Status and Trends: Indicators of ecological integrity showed no notable change from their long-term state, but 9 of the 21 indicators showed substantive change over the short-term.

Trophic structure: Trophic structure indicators showed some changes in the CCLME food web. The northern copepod anomaly (winter) increased in the short term, indicating increased abundance of larger copepods and generally good food supply for planktivores in the system. Mean trophic level of groundfishes decreased in the short term indicating a shift in the food web. The decline in mean trophic level was related to decreases in the abundance of Pacific hake and spiny dogfish—both high biomass and high trophic level species. Since many of the higher trophic level groundfishes consume krill and forage fish, lower abundance of these species may make these prey available to other animals in the CCLME. Scavenger biomass and the northern copepod anomaly (spring) showed no appreciable change.

Diversity: Four diversity indicators increased in the short-term: groundfish Simpson diversity, copepod species richness and Simpson diversity in the winter, and Simpson diversity for seabirds on the continental shelf in June. Two indicators decreased in the short-term: seabird species richness and Simpson diversity in near shore waters for birds observed resting on-water.

The increase in Simpson diversity for groundfishes indicates a more even distribution of species within the system and a change in the assemblage structure. While high diversity is generally considered good, the structural changes in the groundfish assemblage require more detailed examination before that conclusion can be reached. The decreases in Pacific hake and spiny dogfish noted above are likely to be the main drivers of increasing Simpson diversity for the groundfishes. Thus while the assemblage is more diverse as species are more evenly represented, the change is due in a large part to a decline in several highly abundant species.

The causes of the decline in seabird diversity on the shelf are not known, but may indicate more recent changes in offshore conditions that have not been detected in other indicators. Increases in diversity of the winter copepod assemblage likely reflect changes in the predominant types of water masses present in the northern California Current region during the winter; planktonic fauna with subtropical neritic and warm-water offshore affinities generally are more species rich.

Future: We propose continuing the development of a fifth indicator in future assessments based on the strong scores of two phytoplankton-related indicators in our evaluation and the important role of phytoplankton in driving biomass production in the upwelling-driven CCLME. We will continue to accumulate, synthesize, and evaluate phytoplankton indicator data-series, including possibly several related to phenology, into 2013. Habitat-based indicators, such as kelp forest coverage, which ranked highly in this evaluation process, will likely form a separate component of the IEA in 2013. Several other indicators warrant more examination in the future, including the biomasses of jellyfish and euphausiid. The emergence of several long-term datasets from multiple geographic regions along the coast may strengthen the scores of these indicators in 2013 and provide insight into where we might want to emphasize future data collection efforts. Additionally, we will seek to develop composite indicators for those cases where multiple time series are available (e.g., birds) to simplify the presentation and interpretation of output. Finally, most of the ecological integrity indicators do not have reference points or thresholds, and our evaluation relies upon the internal statistics of the time series. Future work should seek to establish reference points for these indicators.

DETAILED REPORT

BACKGROUND - ECOLOGICAL INTEGRITY

Ecological integrity has been defined as “the ability of an ecological system to support and maintain a community of organisms that has a species composition, diversity, and functional organization comparable to those of natural habitats within a region” (Parrish et al. 2003). Implicit in this definition is the concept that an ecological system has integrity when its dominant ecological characteristics (e.g., elements of composition, structure, function, and ecological processes) occur within their natural ranges of variation and can withstand and recover from most perturbations imposed by natural environmental dynamics or human disruptions. As it is applied in this report, ecological integrity (previously termed ecosystem health in Levin et al. 2011) is defined by indicators of community structure which describe individual components within an ecosystem and the relative extent of their potential interactions. These include community level metrics such as taxonomic diversity, trophic structure and ratios between different foraging guilds, and functional group redundancy and relative biomass. Community composition indicators also include population level trends and conditions across some lower-level trophic levels not typically subject to fisheries, such as zooplankton.

There are numerous publications that cite indicators of ecosystem health or ecological integrity in marine systems. For this report, we generally relied on several core references from the literature to develop an initial list of potential indicators (Jennings and Kaiser 1998, Link et al. 2002, Rochet and Trenkel 2003, Fulton et al. 2005, Jennings 2005, Jennings and Dulvy 2005, Link 2005, Shin et al. 2005, Samhoury et al. 2009, Sydeman and Thompson 2010). In many cases, indicators identified in the literature were chosen by the authors based on expert opinion or the context of the researchers’ expertise. For example, many reviews of marine ecosystem indicators are put into the context of fisheries (e.g., Fulton et al. 2005, Link 2005) and ask the question: Which indicators reflect changes in the population as a result of fishing pressure?

INDICATOR SELECTION PROCESS

EVALUATION OF POTENTIAL INDICATORS

We identified and evaluated 40 potential indicators of overall ecological integrity across a variety of taxa and foraging guilds (Table EN1) using the ecological literature as a basis for their rankings (see Levin and Schwing 2011 for detailed methods). Many of the indicators evaluated under the ecosystem health goal in 2011 (Levin and Schwing 2011) were covered under individual fishery or protected species goals in this report (e.g., guild-specific goals such as groundfishes, coastal pelagic species, or marine mammals), and therefore are not addressed in this section to limit report duplication.

Table EN1. Summary of ecological integrity indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, jellyfish biomass status and trends has peer-reviewed literature supporting four out of five primary considerations criteria.

| Guild | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|---------------|--|---|--|---|--|
| Invertebrates | Jellyfish biomass, status and trends | 4 | 3 | 2 | Indicator of trophic energy transfer and pelagic community composition, abundance can be linked to human activities, no existing reference condition. Historical data in CCLME are limited; however, this indicator appears worthy of re-evaluation in 2013 pending review of emerging datasets that may strengthen data considerations. |
| | Squid, Humboldt | 1 | 2 | 2 | Range expansion correlated with reduction in top predators; possibly indicates shifts in climate regimes, ocean circulation, and ecosystem-wide food webs; data minimal and of limited spatial and temporal scale. |
| | Crustaceans: catch and survey trends | 4 | 5 | 4 | Indicative of community regime shift: high trophic level groundfish to low trophic level crustaceans; often attributed to climate induced changes in water column temperature and fishing; multiple data sources available, including larval abundance |
| | Benthic invertebrate biomass | 5 | 3 | 0 | Correlates well with ecosystem health and responds to fishing pressure; some databases available, although depth strata and sampling design not readily apparent; gradual change should show major community reorganization |
| Zooplankton | Zooplankton abundance and biomass | 4 | 7 | 5 | Base of food web, fundamental component of CCLME, correlated with regime shift and climate change, can be used to estimate thresholds, several ongoing long-term data sets. |
| | Copepod biomass anomalies | 5 | 7 | 5 | Reflect modifications in water masses, currents, or atmospheric forcing; respond rapidly to climate variability; some taxa reflect influence of different water types on ecosystem structure; data availability as above. |

| Guild | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|------------------------|--|---|--|---|--|
| Zooplankton (cont.) | Euphausiid biomass and richness | 5 | 2 | 3 | Indicator of plankton biomass changes, critical link in marine food web, low counts and high patchiness in samples may increase variability, data availability appears to be improving; expect improved evaluation in 2013. |
| Diversity indices | Species richness | 2 | 6 | 3 | Can provide a useful measure of diversity if the study area can be successfully delimited in space and time; simple to interpret; may not be highly sensitive to change or management actions; data available from groundfish, zooplankton, benthic invertebrate surveys. |
| | Hurlbert's delta (evenness) | 4 | 7 | 2 | Reflects taxonomic evenness; calculated from abundance estimates; change detectable with latitude and depth at large scales; natural and baseline levels of evenness may vary; significance of certain types of change not known; data available from groundfish, zooplankton, benthic invertebrate surveys. |
| | Simpson Diversity Index | 4 | 7 | 3 | Theoretically sound, calculated from abundance estimates; difficulty linking diversity indices to targets or reference points; numerical; does not show bias in mean value in relation to number of individuals in a sample; for data availability see Hurlbert's biodiversity index above. |
| | Shannon Diversity | 4 | 5 | 2 | Measures taxonomic richness and evenness, community stability related to higher diversity, difficulty linking diversity indices to targets or reference points, for data availability see Hurlbert's biodiversity index above. |
| | Taxonomic distinctness (average and variation in) | 3 | 6 | 1 | Uses species lists, not abundance data; minimal data requirements allows integration of data sets, use of historical data, and data of varying quality; for data availability see Hurlbert's biodiversity index above. |
| | Number of threatened species (IUCN A1 criteria) | 4 | 5 | 1 | Composite indicator based on weighted average of species threat, criteria somewhat arbitrary, linking index to targets or reference points is difficult, data available and numerical. |

| Guild | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|-------------------|---|---|--|---|---|
| Trophic structure | Mean trophic index / mean trophic level | 3 | 6 | 4 | Much recent discussion of this indicator in the literature. Ecosystem MTL, which is calculated from available fisheries independent data, will correctly track changes in the trophic structure of the ecosystem. Internationally recognized; provides both leading and lagging guidance. |
| Functional groups | Top predator biomass (trophic level > 4.0) | 5 | 2 | 4 | Top predator removal typically results in trophic cascades. Data available for many groundfish and seabird top predators, but data for sharks and marine mammals are less reliable. |
| | Invertivore biomass | 2 | 7 | 2 | Correlated with several measures of diversity and total biomass in modeling exercises, but variation in community composition may not be detected by variation in this functional group alone. |
| | Detritivore biomass | 3 | 7 | 2 | Similar comment as above. |
| | Herbivore biomass | 3 | 7 | 2 | Similar comment as above. |
| | Scavenger biomass | 4 | 7 | 2 | Some evidence that disturbances, such as fishing activities, induce chronic increases in scavenger populations, but changes in this one functional group may (or may not) be indicative of the entire community. |
| Biomass ratios | Forage fish and jellyfish biomass ratio | 3 | 2 | 1 | Highly correlated with diversity measures and mean trophic level in modeling exercises. Data limited for both groups; ratios of functional groups are not easily understood indicators. |
| | Piscivorous and Zooplanktivorous fish biomass ratio | 2 | 0 | 2 | Highly correlated with diversity measures in modeling exercises, but how many species have data available is unknown. |

| Guild | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|---------------------------|--|---|--|---|---|
| Biomass ratios (cont.) | Pelagic and demersal fish biomass ratio | 2 | 1 | 2 | Appears to be a proxy for differential impact of nutrients on the pelagic and benthic food webs based on modeling exercises. |
| | Zooplankton and phytoplankton biomass ratio | 3 | 1 | 1 | Highly correlated with measures of diversity and mean trophic level in modeling exercises, but data are particularly limited for phytoplankton, although proxies such as chl-a have been used. |
| | Rockfish and flatfish biomass ratio | 3 | 7 | 1 | Highly correlated with measures of diversity and total biomass in modeling exercises. |
| | Invertivore and herbivore biomass ratio | 1 | 7 | 1 | Similar to comment above. |
| | Finfish and crustacean biomass ratio | 3 | 7 | 1 | Indicative of community regime shift in several systems from high trophic level groundfish to a low trophic level, crustacean-dominated system; see comments above under crustacean and groundfish biomass and survey trends for data availability. |
| Fishery catch | Trophic level of catch (mean biomass) | 2 | 1 | 0 | Shortcomings associated with typical catch-based data; size-based indicators are better because they do not require diet data, are less error prone, and more easily collected. |
| | Proportion noncommercial species (unfished groups) | 3 | 4 | 2 | Modeling results show response to variation in fishing pressure and correlation with ecosystem attributes, one of the more sensitive indicators of changes in species composition. |
| | Total catch and landings of target species | 2 | 4 | 1 | Considered good indicator of fishing effects but poor indicator of marine ecosystem performance, primarily a function of fishing effort and a poor approximation of production, landings can be misleading in assessments ecosystems. |

| Guild | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|--------------------------|---|---|--|---|--|
| Fishery catch (cont.) | Total fishery removals of all species (including bycatch) | 3 | 3 | 1 | See above, bycatch data often not recorded. |
| | Total fishery removals of all species | 2 | 6 | 2 | See above. |
| | Mean length, all species | 3 | 1 | 4 | Useful and simple indicator to evaluate effects of fishery removals, but may not be observable over short-term monitoring data sets. |
| | Slope size spectrum, all species | 2 | 1 | 1 | Good indicator of fishing effects, models show change is predictable and consistent, unclear what attributes it would act as an indicator for besides general ecosystem health, thresholds unclear, size data sparse for some species. |
| Habitat species | Kelp forest coverage | 4 | 5 | 5 | Kelp forests occur at small scales compared to the entire California Current, so overall ecosystem structure may not be tied to kelp coverage, but these are important habitats for recruitment of important species. Likely habitat-based indicator for future assessments. |
| | Area of live, hard coral | 5 | 2 | 2 | Similar comment as above. Data on spatial extent of coral cover are limited. |
| Phytoplankton | PPC:PSC ratio (carotenoid ratio) | 2 | 3 | 0 | Ratio of photosynthetic pigments (PSCs), such as chlorophyll and some carotenoids, to photoprotective carotenoids (PPCs); ratios can be used to detect areas of upwelling. Limited history of reporting; not well-understood by public or policymakers. |

| Guild | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|--------------------------|---|---|--|---|---|
| Phytoplankton (cont.) | Diatom: dinoflagellate ratio | 1 | 4 | 0 | Ratio of diatoms to dinoflagellates, which can be related to copepod production and hatching success/survival. Time-consuming method based on direct microscope counts. Phytoplankton cell counts have been taken on all CalCOFI cruises since 2002. Limited history of reporting. |
| | Community structure - pigments | 3 | 4 | 0 | Robust method to identify pigment compounds specific to different microalgae and thereby estimate composition of phytoplankton community; implications for energy cycling. Some limitations in understanding of community controls remain. Phytoplankton pigments have been surveyed on all CalCOFI cruises since 2002. |
| | Community structure - cell counts | 3 | 5 | 2 | Taxonomic structure has implications for ecosystem function and energy cycling; time-consuming method based on direct microscope counts, phytoplankton cell counts have been taken on all CalCOFI cruises since 2002. |
| | Biomass – Chl- <i>a</i> | 4 | 5 | 3 | Good indicator of phytoplankton biomass and amount of energy fueling the ecosystem, satellite remotely sensed chlorophyll concentration data available system wide; spatially and temporally variable; rarely reported as a single indicator value |

TOP RANKED INDICATORS

What follows is a general description of the indicators ranked in the top quartile (10 indicators), as well as several others making a strong showing and with future potential. Weighted numeric ranks are noted in parentheses after each indicator name, with rank ties indicated by the letter (T). General categories of indicators are grouped by the taxa or processes they represent, and include: Biodiversity, Trophic structure, Zooplankton, Phytoplankton, and Other. A summary of all 40 of the indicator evaluations is provided in Table EN1, including the number of evaluation criteria supported by the peer-reviewed literature and some brief comments.

Zooplankton were represented by two of the top-ranked indicators, with four indicators of biodiversity and two of trophic structure also represented in the top quartile. In general, the initial ranking process retained the same pool of indicators as after weighting the evaluation criteria based on expert opinion. Exceptions include *scavenger biomass* and some of the diversity indicators (i.e., *Shannon diversity* and *number of IUCN threatened species*) which jumped into the top quartile after weighting, and *species richness* and two phytoplankton indicators (i.e., *phytoplankton community structure* and *Chl-a / phytoplankton biomass*) which fell out of the top quartile. Shortcomings of the two phytoplankton indicators were primarily due to weak links to scientifically-defined reference points or progress targets, poor understanding of temporal and spatial variation, and lack of understanding by the public and policymakers.

Indicators that scored well under primary considerations generally included foraging guild trends and biomass. Many functional group ratios have been identified by modeling exercises as good indicators of diversity and total biomass in the system. However, a common theme for many of these indicators was that they performed poorly for criteria related to their responsiveness and sensitivity to changes in community composition. This is because changes in species' or foraging guilds' trends and abundance will influence community composition and ecosystem structure, but changes in community composition may not be reflected in any one species or foraging guild. Moreover, it is conceivable that many of the foraging guild ratio indicators (e.g., piscivorous to zooplanktivorous fish ratio) could have scientifically defined reference points and progress targets, but these ratios may not be easily understood by the public and policy makers. Furthermore, changes in many of these community-level metrics cannot be observed in short-term monitoring sets and may be more useful at longer management time scales (Nicholson and Jennings 2004).

Most of the indicators related to fishery catch scored well with regard to data considerations, but rarely met primary considerations associated with being theoretically sound or predictably responsive. This could be attributed to the often misleading nature of landings in assessments of fisheries ecosystems (de Mutsert et al. 2008) and weak showing as indicators of marine ecosystem performance in food web model simulations (Samhuri et al. 2009). In contrast, several indicators showed promise by meeting most theoretical considerations (e.g., *jellyfish* and *euphausid biomass*) but often lacked adequate historic data over a broad geographic scale.

ZOOPLANKTON

Two of the three zooplankton-related community indicators ranked among the highest of the 40 indicators that were evaluated.

ZOOPLANKTON SPECIES BIOMASS ANOMALY (NORTHERN COPEPOD BIOMASS) (1)

Zooplankton time series provide some of the best opportunities to understand marine ecosystem responses to climate change because zooplankton are the foundation of the ocean food web, linking oceanographic conditions and primary production to upper trophic levels and fueling the delivery of ocean ecosystem services. Zooplankton life cycles are short (on the order of weeks to a year) and populations have the potential to respond to and reflect event-scale and seasonal changes in environmental conditions (Hooff and Peterson 2006). Moreover, many zooplankton taxa are known to be indicator species whose presence or absence may represent the relative influence of different water types on ecosystem structure. Thus zooplankton may serve as sentinel taxa that reflect changes in marine ecosystems by providing early indications of a biological response to climate variability and are often used as an indicator to detect climate change or regime shifts (Hooff and Peterson 2006, Mackas et al. 2006, Peterson 2009). Finally, zooplankton are abundant and can be quantified by relatively simple and comparable sampling methods and, because few are fished, most population changes can be attributed to environmental causes (Mackas and Beaugrand 2010). As such, they may prove useful as a leading indicator of what may happen to regional commercial fish stocks several years later (Mackas et al. 2007, Peterson et al. unpubl. manusc.).

All along the California Current, anomalies in zooplankton species composition shifts have been correlated with regional climate patterns (Mackas et al. 2006). For example, off the Oregon coast zooplankton indices have been developed based on the affinities of copepods for different water types: those with cold water and those with warm water affinities (Peterson et al. unpubl. manusc.). The cold water group usually dominates the coastal zooplankton community during the summer (typically May through September) upwelling season, whereas the warm water group usually dominates during winter, although this pattern is altered during summers with El Niño events or when the Pacific Decadal Oscillation (PDO) is in a positive (warm) phase. Perhaps the most significant aspect of the copepod index is that two of the cold water species, *Calanus marshallae* and *Pseudocalanus mimus*, are lipid-rich species. Therefore, an estimate of northern copepod biomass may also index the amount of wax esters and fatty acids being fixed in the food chain, compounds which appear to be essential for many pelagic fishes if they are to grow and survive through the winter successfully.

Several long-term zooplankton monitoring programs, representing seven subregions spanning the entire CCLME from Baja California to Vancouver Island, now provide zooplankton time series of various lengths from 1969 to the present. Although differences in processing and sampling zooplankton time series introduce a variety of biases that often prevent comparisons between data sets, many major questions can still be answered because an individual data set can be presented and analyzed as a time series of log-scale anomalies relative to the local long-term-average seasonal climatology. Anomalies are primarily used to separate interannual variability from the often large annual seasonal cycle of zooplankton stock size (Mackas and Beaugrand 2010). The specific species associated with these anomalies vary regionally, but can generally be classified as resident versus nonresident species. Regional anomalies can be combined into a single index using multivariate techniques (e.g., principal component analysis) in similar fashion to the calculation of regional climate indices, such as the Multivariate El Niño Southern Oscillation (ENSO) Index (Wolter and Timlin 1993). This index can then be tested for use as a leading indicator of regional climate signals, such as ENSO or PDO, using existing time series from the last 20 years, during which time the California Current saw at least two major climate regime shifts.

ZOOPLANKTON ABUNDANCE AND BIOMASS (2)

As noted above, zooplankton time series provide some of the best opportunities to understand marine ecosystem responses to climate change. As an important link at the base of the pelagic food web, they are considered a fundamental component in the CCLME (Brand et al. 2007, Horne et al. 2010, Sydeman and Thompson 2010). Because the biomass of planktivorous fish is inversely related to zooplankton biomass, which in turn is inversely related to phytoplankton biomass, zooplankton may prove useful as a leading indicator of what may happen to regional commercial fish stocks several years later (Sherman 1994, Mackas et al. 2007, Mackas and Beaugrand 2010, Peterson et al. unpubl. manusc.). Zooplankton biomass declines have been correlated with warming of surface waters (Roemmich and McGowan 1995, Sydeman and Thompson 2010) and used to detect regime shifts (Hare and Mantua 2000). However, for time series observations of ecosystem state variables such as biomass or chemical concentrations, standard deviations may increase, variance may shift to lower frequencies in the variance spectrum, and return rates in response to disturbance may decrease prior to a change (Carpenter et al. 2008).

The feeding effect of pink salmon (*Oncorhynchus gorbuscha*) has been shown to control summer macrozooplankton and phytoplankton biomass in the subarctic North Pacific (Shiomoto et al. 1997). Trophic cascade theory holds that reductions in harvest of zooplanktivorous fish would ultimately result in lower biomass of zooplankton, but it is unclear whether this has been demonstrated in the field for large marine systems (Pace et al. 1999). There are a number of (up to seven) long-term zooplankton biomass time series that have been maintained throughout various regions of the CCLME (Hooff and Peterson 2006, Mackas and Beaugrand 2010); one of the oldest of these data sets is the California Cooperative Oceanic Fisheries Investigative (CalCOFI) reports time series, which has been collected since 1956 (McClatchie et al. 2009). In freshwater systems, zooplankton biomass has been used as a leading indicator of trophic cascades (Carpenter et al. 2008).

BIODIVERSITY

Species diversity is an integrative measure that encompasses species richness - the number of species in the ecosystem, and species evenness - how individuals or biomass are distributed among species within the ecosystem (Pimm 1984). Diversity has remained a central theme in ecology and is frequently seen as an indicator of the wellbeing of ecological systems (Magurran 1988). It is therefore not surprising that four diversity-related indicators ranked in the top quartile of our evaluation.

Theoretical modeling results have been used to show that some ecosystem structural (e.g., diversity) attributes can be related to thresholds in the level of human-induced pressure. The correlation between diversity and ecosystem function (productivity and stability) has been reviewed recently for terrestrial and marine systems, suggesting that the relationship is complex but communities are more stable at higher richness (Hooper et al. 2005, Stachowicz et al. 2007). In general, populations can be more variable but community level processes are more stable at higher diversity (i.e., the biomass of species A and species B may fluctuate, but $A + B$ tends to be stable). Linking diversity indices to targets or reference points is difficult, and the significance of certain types of change is not known for biodiversity indices (Link 2005, Dulvy et al. 2006). However, some authors have provided a rationale to manage for biodiversity as an approach to EBM (Palumbi et al. 2009). The general public tends to have a basic understanding and positive impression toward biodiversity as it relates to ecosystem health (Thompson and Starzomski 2007). Species richness has been shown to decrease with fishing, although these results appear largely related to trawling and dredging on benthic invertebrates (Gaspar et al. 2009, Reiss et al. 2009).

All of the following diversity indices can be used with a variety of existing survey data, including: groundfish trawl surveys (Weinberg et al. 2002, Keller et al. 2010), pelagic or midwater trawl surveys (Brodeur et al. 2003, Sakuma et al. 2006), reef fish surveys conducted by trained divers (REEF 2008), zooplankton surveys (e.g., NWFSC Newport Line, CalCOFI), invertebrates from benthic grabs conducted by the EPA (<http://www.epa.gov/emap/index.html>), and a variety of seabird and marine mammal surveys (Ainley et al. 1995, Barlow and Forney 2007, Carretta et al. 2007, McClatchie et al. 2009). Other possible data sources include intertidal invertebrate surveys from 2002 to 2010 (PISCO at <http://www.piscoweb.org/>) and datasets available at smaller spatial and temporal scales (e.g., National Park Service kelp forest monitoring program in the Channel Islands). Many of these data would need to be combined to investigate trends over time across the entire scale of the CCLME.

SIMPSON'S INDEX OF DIVERSITY (3)

Simpson's index is a dominance measure that estimates the probability that any two individuals drawn at random from an infinitely large community would belong to different species (Magurran 1988). It is similar to Hurlbert's (1971) concept of the probability of an interspecific encounter when individuals are drawn with replacement, and is relevant to predator-prey and food-web analyses. It is a numerical measure and does not show bias in mean value in relation to the number of individuals in a sample (Clarke and Warwick 2001). Model simulations, used to evaluate the ability of candidate indicators to track ecosystem attributes of interest, have shown that Simpson's diversity was strongly correlated to the biomass of marine mammals in a system. Samhouri et al. (2009) notes that the indicator-attribute relationship can switch depending upon the type of fishing pressure used in the model. This result might make the indicator-attribute relationship unpredictable in the real world.

EVENNESS (HURLBERT'S DELTA) (4)

Hurlbert's delta is a measure of evenness that, when applied to abundance estimates from a particular ecological community, estimates the probability of an interspecific encounter: that two individuals in a sample are different species (Hurlbert 1971). It has a clear, concise ecological interpretation and is relevant to predator-prey and food-web analyses and has been applied as an indicator for detecting the impact of fishing on a fish community (Trenkel and Rochet 2003). Hurlbert's delta measure has been applied in measuring detectable spatial variation with depth and latitude at large scales and, although temporal patterns may be unknown, could be calculated from historical data (Tolimieri 2007). It can also be used to detect changes in community composition after change has occurred, although natural and baseline levels of taxonomic evenness may vary so much that absolute values may not be comparable in terms of thresholds.

SHANNON DIVERSITY (9-T)

Shannon diversity is a measure that incorporates both richness (the number of different species within a system) and evenness (the number of individuals of each species within a system). A marine ecosystem model for British Columbia showed that adult sablefish biomass was positively correlated with Shannon diversity, suggesting that changing levels of fishing on a particular species may produce substantial improvements toward protecting ecosystem goals based on this structural attribute (Samhouri et al. 2010). The model also describes how to incorporate uncertainty into the estimation of utility thresholds and their value in the context of understanding EBM trade-offs. These modeling results may be equally applicable to the CCLME because of many similarities between these ecosystems. The value of this indicator is predicated not only on the correlation between sablefish biomass and ecosystem diversity, but also on how well each of these independent indicators meet individual evaluation considerations.

NUMBER OF THREATENED SPECIES (IUCN A1 CRITERIA, AS MODIFIED BY DULVY ET AL. (2006) (9-T)

This is a composite indicator based on a weighted average of species threat, as determined by the International Union for the Conservation of Nature (IUCN 2008), which may be different from those considered threatened under the U.S. Endangered Species or the Marine Mammal Protection acts. The weighting criteria for this indicator are somewhat arbitrary and linking the index to targets or reference points is difficult; however, data are readily available and numerical. The same approach used by the IUCN could be applied to a variety of existing survey data.

SPECIES RICHNESS (15)

Species richness, which is a count of the number of species present, can provide an extremely useful measure of diversity if the study area can be successfully delimited in space and time and the constituent species enumerated and identified (Magurran 1988). Though ranked low in the evaluation, we included a description here because species richness can be used to help inform other standard measures of diversity. Species richness may not be highly sensitive to change and may not respond unambiguously to variations in ecological integrity or management action; furthermore, the species--sampling intensity relationships will require rarefaction to standardize for sampling effort. Studies have shown that species richness tends to decline with fishing, primarily based on trawling/dredging effects on benthic invertebrate communities (Gaspar et al. 2009, Reiss et al. 2009).

Richness can influence stability and productivity in two ways: 1) sampling/selection effect or 2) compensatory effect (Stachowicz et al. 2007). Under the sampling effect, higher richness leads to a greater chance of highly productive species being present. This type of relationship is not considered a real richness effect by some, but more of a compositional or keystone species effect. Under the compensatory effect, higher production or stability occurs in two ways: via resource complementarity, where more species occupy more niches and better utilize all resources (e.g., different type of nitrogen), and facilitation, where some species combinations do better.

TROPHIC STRUCTURE

Organisms within an ecosystem can be classified according to their trophic level, or position within the food web (e.g., functional groups include herbivores, carnivores or predators, detritivores, and scavengers), and indicators of trophic structure attempt to measure their relative abundance, biomass, and interactions. Indicators related to the biomass of specific trophic levels within the ecosystem ranked highly in the evaluation, especially within the context of theoretical or practical considerations.

SCAVENGER BIOMASS (5)

Scavengers play significant roles in the ecosystem by recycling dead and decomposing organic matter back into the food web. However, human interference in the marine ecosystem has likely increased the abundance and number of species that forage on carrion (Britton and Morton 1994). For example, many fishing operations discard dead bycatch to the ocean floor or damage organisms on the seabed during bottom fishing operations (Ramsay et al. 1998). Scavenger population increases may be related to these types of fishing activities (Britton and Morton 1994, Ramsay et al. 1998, Demestre et al. 2000).

When evaluating this indicator, we use the definition of scavenger used in the Atlantis ecosystem models for the California Current (Brand et al. 2007, Horne et al. 2010). In these models, scavengers include all large crabs, large demersal sharks, grenadiers, deposit feeders (i.e., isopods and amphipods), and carnivorous infauna such as polychaetes. Detectable changes in the community composition may be a result of changes in various foraging guilds, but a change (or no change) in a single guild may not be indicative of the ecosystem as a whole. Fisheries-based reference points include B40 (target level where production is predicted to be greatest) and B25 (overfished). These single-species reference points could be adapted and used for foraging guilds such as scavengers. Alternatively, Link (2005) describes a framework of reference points that could be applied to most any indicator.

Fishery-independent data are available since 1977 for all scavenger species susceptible to bottom trawling across the U.S. portion of the CCLME. There are also data available at smaller spatial scales and at various temporal scales in untrawlable habitats from submersible, ROV, and the NWFSC hook-and-line surveys. Fishery-dependent data for crab species are available in the PacFIN database (<http://pacfin.psmfc.org/>). Some species of the scavenger guild, such as isopods, amphipods, and polychaetes, will need new surveys to quantify these components. Benthic grab samples are commonly used to quantify benthic infauna, but it may be difficult to perform this type of survey at the scale of the CCLME at necessary temporal scales. Moreover, quantifying a value for many foraging guilds will require quantitative analyses to combine data sets which collect data using very different methods. For example, bottom trawl surveys, longline surveys, and benthic grab samples will need to be combined at various spatial and temporal sampling scales to quantify the biomass of grenadiers, crabs, large demersal sharks, and deposit feeders.

The public can easily understand whether a foraging guild, such as scavengers, is trending up or down, but this particular indicator may be less attractive to the public than more charismatic groups (i.e., marine mammals or sharks). Detecting changes in the biomass of scavengers would likely be measured against long-term averages, so unless dramatic changes are observed, scavenger biomass will be a lagging indicator of changes in community composition. Monitoring foraging guilds such as scavengers has been performed in other regions of the United States (Link and Almeida 2002) and in other nations (Demestre et al. 2000, Greenstreet and Rogers 2000).

MEAN TROPHIC LEVEL (6)

Mean trophic level (MTL) is the biomass-weighted average trophic level of all species in an ecosystem. Mean trophic level provides a synoptic view of the organization of trophic structure in marine ecosystems, and is a pervasive and heavily discussed indicator used to measure marine ecosystem status, especially in communities dominated by exploited species (Pauly and Watson 2005, Essington et al. 2006, Branch et al. 2010). Conceptually, MTL is linked to top-down control and trophic cascades; a decline in MTL represents a decrease in the ability of predators to 'control' prey populations and may have far-reaching consequences to ecological communities (Daskalov 2002, Estes et al. 2004, Pauly and Watson 2005, Baum and Worm 2009). Theoretical modeling results have been used to show that mean trophic level can be a good univariate indicator of fishing effects on an ecosystem, although it may be sensitive to data quality (e.g., landings v. survey data) (Fulton et al. 2005, Samhuri et al. 2009). Trends in 'catch' MTL, estimated from fisheries landings and other fishery dependent data sources may not provide a good indicator of actual changes in the ecosystem. Instead, 'ecosystem' MTL, estimated from data sources like fisheries-independent surveys, is indicative of current ecosystem status especially when coupled with an exploration of the processes responsible for such patterns (Branch et al. 2010).

Mean trophic level is an operationally simple, concrete, numerical indicator, calculated each year using the simple mean of biomass weighted trophic levels within an ecosystem. Trophic levels can be estimated for species worldwide from an online data base (<http://www.fishbase.org/search.php>); species biomass can be obtained from historical, annual estimates derived from standardized surveys throughout the California Current (various groundfish, zooplankton, marine mammal, and seabird survey, etc.). These surveys are generally continuous, have broad spatial coverage, and are designed with appropriate power to have a high signal-to-noise ratio (for most species). The spatial temporal variation in these time-series is becoming increasingly understood as more data are collected each year.

Mean trophic level is included in the list of provisional indicators for assessing progress toward the 2010 biodiversity target, proposed by the Convention of Biological Diversity (<http://www.cbd.int/>). As such, it is understood by the public and policymakers, considered internationally compatible, and demonstrates a relatively recent history of reporting (Pauly and Watson 2005, Stergiou and Tsikliras 2011). Mean trophic level can be estimated in a cost-effective manner using existing survey data (various groundfish Stock Assessments, REEF.org, etc.). Mean trophic level may fill a role as a leading indicator because the concept of MTL is closely related to top-down forcing and trophic cascades. As such, changes in the trophic structure revealed by MTL may predictably preface changes to other taxa (unpublished data, N. Tolimieri).

PHYTOPLANKTON

Two phytoplankton indicators fell just outside of the top quartile rankings in our evaluation. Phytoplankton indicators in general tended to meet most theoretical considerations, based on taxonomic structure and biomass, the latter of which performed particularly well under simulations involving seven marine food web models (Samhouri et al. 2009). Because phytoplankton forms the base of the food web for most pelagic communities and drives biomass production in the upwelling-driven California Current marine ecosystem, we strongly suggest continuing the search or development of a more appropriate phytoplankton indicator in future evaluations, perhaps by inclusion of indicators related to phytoplankton phenology.

PHYTOPLANKTON BIOMASS – CHLOROPHYLL-A (12)

Chl-*a* can be used as an indicator of phytoplankton biomass, which itself is a good indicator of the amount of energy fueling the ecosystem (Falkowski and Kiefer 1985, Cole and Cloern 1987, Polovina et al. 2001, Edwards and Richardson 2004, Fulton et al. 2005). The amount of primary productivity, measured as total chlorophyll per unit area (mg m^{-3}), has been recognized as an important aspect of the marine food web, and Chl-*a* values are used to estimate phytoplankton biomass for mass-balance models of the CCLME (Falkowski and Kiefer 1985, Brand et al. 2007, Horne et al. 2010). Chl-*a* has been shown to respond predictably to reductions or increases in nutrient inputs (eutrophication). It should be possible to identify time-specific and location-specific reference points for upwelling or transition fronts.

Chl-*a* has been used to provide basic data for CCLME ecosystem model building and calibration based on values from GLOBEC sampling cruises between 1997 and 2004 and CalCOFI cruises from 2000 to 2004 (Brand et al. 2007). Satellite remotely sensed Chl-*a* concentration (mg m^{-3}) data can be obtained at minimal cost from the Sea-viewing Wide Field-of-View Sensor (SeaWiFS at <http://oceancolor.gsfc.nasa.gov/SeaWiFS/>) to derive broad-scale coverage of values over the CCLME (Polovina and Howell 2005) or at smaller regional scales (Sydeman and Thompson 2010). Phytoplankton color, a visual index of chlorophyll derived from continuous plankton recorder (CPR) surveys (<http://www.sahfos.ac.uk/about-us/cpr-survey/the-cpr-survey.aspx>), can also be used to show intensity and seasonal extent of Chl-*a* (Edwards and Richardson 2004). However, the relationship between reflectance and phytoplankton biomass must be derived and

requires ground-truthing studies. Some species or subsets of species of phytoplankton that affect Chl-*a* concentration can serve as an indicator of change in phytoplankton biomass, but physical measurements of upwelling intensity may provide a better leading indicator. The approach for compressing the temporal and spatial variability of this information into a single datum presents challenges, however.

PHYTOPLANKTON COMMUNITY STRUCTURE – CELL COUNTS (13)

The taxonomic structure of phytoplankton communities is an important determinant of ecosystem function, with far-reaching implications for the cycling of energy and matter in the marine environment (Goericke 2011). In the California Current, the phytoplankton community changes predictably as total chlorophyll *a* (TChl-*a*) increases in the system, a pattern driven by increasing nutrient loading or nutrient content. Time series of specific phytoplankton taxa reflect the short term variability of phytoplankton biomass, which is likely controlled by varying rates of nutrient-autotroph and autotroph- heterotroph interactions (Mackey et al. 1996, Goericke 2011). Reference limits have been established that provide a general relationship of Chl-*a* to phytoplankton community structure and as an early indicator that can differentiate and identify bloom assemblages.

Total phytoplankton biomass is generally limited by the availability of a critical nutrient (i.e., by bottom-up forces), but the biomass of some taxa, particularly picoautotrophs, is controlled by grazers (i.e., top-down forces) under mesotrophic to eutrophic conditions. The distribution of cyanobacteria suggests that their population dynamics are not tightly linked to the dynamics of their grazers, likely because the latter are grazing concurrently on heterotrophic bacteria (Goericke 2011). However, there are some noted limitations in our understanding of the controls of phytoplankton community: data sets from southern California suggest that nearshore communities may differ fundamentally from coastal oceanic communities, and temperature may exert important secondary effects on phytoplankton community structure based on cyanobacteria distributions (Thomas and Strub 2001). Variability of autotroph biomass in the California Current is primarily due to blooms of dinoflagellates and to some extent diatoms, i.e., the larger autotrophs that contributed 81% to the variability of TChl *a* over time (Goericke 2011).

Phytoplankton samples, complemented by complete hydrographic data, have been routinely taken on CalCOFI cruises from at least 2002-present. The California Current Ecosystem – Long Term Ecological Research (CCE-LTER) program collects samples for nano and microplankton identification and sizing using microscopy, and for picoplankton using flow cytometry, providing additional information on community. Two permanent sampling lines (#90 and 80) are sampled extensively for micro/nano/pico plankton, plus mesozooplankton; line 80 is off Point Conception and line 90 is near Catalina and San Clemente Islands. Each line has 8 stations, with 4 cardinal stations that are always sampled at multiple depths. Additional data is also available from smaller, geographically limited projects such as WEST (May-June 2000-2002), RISE (May-August 2004-2006), and EcoHAB (July-Sept 2003-2006). These projects have detailed phytoplankton community composition data which are compatible with the CalCOFI pigment groupings. Continuous historical data is limited to the CalCOFI sampling area, which extends from southern California to central California. Data from CalCOFI samples is available from at least 2002 to the present. Data from central CA is available from 2000-2002 and data from WA/OR coast is available from 2003-2006. Seasonal succession and patterns of bloom/post-bloom communities is fairly well understood. Spatially, features like the Columbia River Plume are known to influence communities on the WA/OR coasts, as are upwelling areas, such as Pt. Conception on the CA coast (Thomas and Strub 2001)

This indicator would be considered cost-effective because data is already collected as part of CalCOFI and CCE-LTER. Phytoplankton community structure, would be considered an anticipatory indicator of

conditions in the California Current (Gallegos 1992, Macedo et al. 2001, Nuccio et al. 2003). These techniques have been applied successfully in many locations from the Antarctic to the Equator (Mackey et al. 1996).

OTHER INDICATORS

Other indicators in or near the top quartile included those related to fishery catch, invertebrate surveys, and habitat distributions.

CRUSTACEAN SURVEY TRENDS (7-T)

Crustaceans are a prominent component of the CCLME and contribute to the delivery of several important ecosystem services in the region through commercially and recreationally important fisheries (Fogarty and Botsford 2006). They also comprise several important predatory and scavenger groups in existing CCLME models (Brand et al. 2007). They are highly responsive to top-down effects in the food web, and predatory finfish abundance may be a negative indicator for invertebrate fishery productivity (Caddy 2004). For instance, shrimp biomass has been strongly negatively related to cod biomass in the North Atlantic Ocean, showing that changes in predator populations can have strong effects on prey populations in oceanic food webs (Worm and Myers 2003). Fishing effects may exacerbate these patterns: the Gulf of Maine shifted from a high trophic level, groundfish-dominated, system to a low trophic level, crustacean-dominated system during the 1980s to 1990s (Zhang and Chen 2007).

As a group, crustaceans are often found low in the food web, are highly fecund, and may be sensitive to bottom-up effects; therefore, indicators measuring plankton productivity, turbidity, oxygen levels, and eutrophication should be useful in predicting the typically large variations in recruitment success that drive these fisheries (Caddy 2004). Climate change manifested in water column temperature also has an effect on lower trophic levels of boreal marine ecosystems, and changes in crustacean recruitment patterns may be one of the first indicators of community regime shift (Zheng and Kruse 2000). For instance, declines in several species of pandalid shrimp and other community effects in the Gulf of Alaska have been attributed to climate induced changes in water column temperature (Anderson 2000). Pandalid shrimp surveys are also used as indicators of Pacific Ocean conditions off British Columbia (DFO 2009). The abundance of decapod larvae in the plankton also appears to be positively correlated to changes in North Sea sea surface temperature (SST) (Kirby et al. 2009).

For the most part, data availability for this group is relatively good. Zooplankton time series are spatially and temporally extensive (Mackas et al. 2007, McClatchie et al. 2009) and crustacean larval surveys represent a long established means of estimating the spawning stocks of decapods (Kirby et al. 2009). Harvest data records are fairly extensive through PacFIN (though biased by typical catch issues) and some aspects of the ongoing West Coast groundfish surveys may be useful in deciphering abundance/biomass patterns (Keller et al. 2008).

KELP FOREST COVERAGE (7-T)

Kelp forests are ecologically and economically important, as they are the foundational structure for diverse communities in most coastal waters of the CCLME (Dayton 1985, Graham 2004). The persistence of many biologically and commercially important species of algae, invertebrates, fish, and marine mammals are directly coupled to the production of energy from kelp (Foster and Schiel 1985, Steneck et al. 2002). Kelp forests may also serve functional roles in cycling carbon between coastal marine, littoral (Polis and Hurd 1996, Dugan et al. 2003), and continental shelf (Harrold et al. 1998, Vetter and Dayton 1999) ecosystems.

Most kelp forests exist in waters less than 60 m deep, so at the scale of the CCLME community composition may not be tied to the abundance of kelp, but because of its importance as essential fish habitat for many species of concern, including young-of-year fishes (Carr 1991), understanding the temporal variation and spatial heterogeneity (Jones 1992, Bustamante and Branch 1996) of kelp forest coverage in the CCLME may be a useful indicator of ecosystem structure. Following the framework of Link (2005), reference points related to percent change in aerial coverage of kelp could be established.

The density and distribution of kelp forests has been measured historically in numerous ways. Many historical data sets include scuba diving surveys (e.g., Partnership for Interdisciplinary Studies of Coastal Oceans [PISCO] at <http://www.piscoweb.org/>, U.S. National Park Service at <http://www.nps.gov/chis/contacts.htm>), but these are generally over small spatial and short temporal scales. Recent advances in satellite and infrared photography have allowed researchers to measure areal canopy cover and biomass of kelp along much of the U.S. West Coast (Deysher 1993, Cavanaugh et al. 2010).

Kelp forest coverage is easily understood by the public and has been used by policy makers to develop guidelines related to provisions of the marine statistical area on the identification of essential fish habitat (16 U. S. C. §1855b). Changes in kelp forest coverage affect recruitment of invertebrates and other species (e.g., Carr 1991), such that kelp forest coverage could anticipate recruitment of older life stages into the bottom trawl surveys or into the fishery. Kelp forest coverage could be a leading indicator for the community composition of the CCLME and will be considered under a separate habitat goal category in forthcoming IEAs.

PROPORTION OF NONCOMMERCIAL SPECIES (9-T)

The proportion of noncommercial species in fishery catch data has been shown to be strongly related to 12 attributes of ecosystem health, based on modeling results from numerous systems (Samhuri et al. 2009). It has been used as one of the more sensitive indicators for detecting the impacts of fishing on fish communities, with a coefficient of variation around 20% for either biomass or abundance (Trenkel and Rochet 2003). Modeling results show the proportion of noncommercial species responds to variation in fishing pressure and correlates to ecosystem attributes (Samhuri et al. 2009). If this indicator is monitored, gradual change should be detected prior to major community reorganization (i.e., leading indicator). Data for this indicator include a limited number of time series with good spatial coverage: Marine Recreational Fisheries Statistics Survey (MRFSS 1980–2003), data for nontrawl species (<http://www.recfin.org/>), and data from the observer program (bycatch species) (Bellman et al. 2009). However, in the limited entry bottom trawl fishery that operates in the California Current, only a relatively small percentage of the catch (approximately 5% by weight) is composed of noncommercial spp.

http://www.nwfsc.noaa.gov/research/divisions/fram/observer/sector_products.cfm.

FINAL SUITE OF INDICATORS

Recent analyses have shown that a single indicator is not sufficient to provide a complete picture of ecosystem state (Fulton et al. 2005); conversely, too many indicators can lead to too many conflicting signals that lead to indecision. Therefore, we ranked the evaluation scores of all 40 indicators for the ecological integrity goal and selected four of those ranked in the top quartile. Below we list the full suite of indicators chosen and discuss the final selection process.

From the 40 indicators ranked in the top quartile, we propose using these four as indicators in the CCLME during 2012:

- Zooplankton species biomass anomalies (specifically, *Northern copepod biomass anomaly*)
- Biodiversity (*Simpson's index of diversity*, with comparison to *Species richness*)
- *Mean trophic level*
- *Scavenger biomass*

The data sources we propose for these indicators, including extent of time-series and sampling frequency, are documented in Table EN2. The indicators we selected related to biodiversity and mean trophic level integrate a variety of time-series from among several components of the ecosystem (i.e., pelagic and demersal communities). The zooplankton indicator focuses on a single, critical component known to form the foundation of the ocean food web, linking oceanographic conditions and primary production to upper trophic levels. Scavenger biomass is another indicator of trophic structure which has been shown to respond to various fishing activities; it also serves to integrate data on crustacean populations (see the highly ranked, *crustacean survey trends* indicator), which can be highly responsive to top-down effects in the food web and predatory finfish populations.

We propose continuing the development of a fifth indicator in future assessments based on the strong scores of two phytoplankton-related indicators in our evaluation and the important role of phytoplankton in driving biomass production in the upwelling-driven CCLME. We will continue to accumulate, synthesize, and evaluate phytoplankton indicator data-series, including possibly several related to phenology, into 2013.

Habitat-based indicators, such as kelp forest coverage which ranked highly in this evaluation process, will likely form a separate component of the IEA in 2013.

Finally, several other indicators warrant more examination in the future, including *jellyfish* and *euphausiid biomass*. Both showed promise by meeting most theoretical considerations in our evaluation, but lacked adequate historic data over a broad geographic scale. Notably, lower trophic level, high productivity functional groups like jellyfish and phytoplankton biomass showed relatively strong correlations with at least half of the ecosystem attributes in a food-web modeling exercise that evaluated the performance of candidate indicators of ecosystem structure and function (Samhuri et al. 2009). The emergence of several long-term datasets from multiple geographic regions along the coast may strengthen the scores of these indicators in 2013 and provide insight into where we might want to emphasize future data collection efforts.

Table EN2. Top-ranked indicators for Ecosystem Integrity.

| Attribute / Guild | Indicator | Definition and source of data | Time series | Sampling frequency |
|--|------------------------------------|--|-------------|--------------------|
| Biodiversity | Simpson's index & Species Richness | Index of zooplankton community composition; measures dominance & number of species present in study area (Peterson et al., NOAA) | 1996 – 2012 | Biweekly |
| | | Index of coastal pelagic species community composition, northern California Current (Brodeur et al., NOAA) | 1998 – 2011 | June, Sept; Annual |
| | | Index of groundfish community composition (Keller et al. NWFSC) | 2003 - 2011 | Summers, Annual |
| | | Index of seabird community composition; northern California Current (Zamon et al. NWFSC) | 2004 - 2012 | Summers, Annual |
| Trophic structure | Mean trophic level | Trophic structure of groundfish community (Keller et al. NWFSC) | 2003 - 2011 | Summers, Annual |
| | | Trophic structure of coastal pelagic fish community (<i>currently in development</i>) (Brodeur et al., NOAA) | 1998 – 2011 | June, Sept; Annual |
| | | Trophic structure of seabird community (<i>currently in development</i>) | | |
| | | Trophic structure of marine mammal community (<i>currently in development</i>) | | |
| Trophic structure | Scavenger biomass | Relative biomass of scavengers, as defined by Brand et al. (2007), from fishery independent surveys (Keller et al. NWFSC) | 2003 - 2011 | Summers, Annual |
| Zooplankton community structure | Northern copepod anomaly | Monthly anomalies in the relative biomass of copepods with cold-water affinities off Newport, OR (Peterson et al., NOAA); | 1996 – 2012 | Biweekly |

STATUS AND TRENDS: ECOLOGICAL INTEGRITY

MAJOR FINDINGS

Indicators for Ecological Integrity are ecosystem and community level indices that were chosen to track two community level aspects of the CCLME: trophic structure (mean trophic level, scavenger biomass, and the northern copepod anomaly) and diversity (Simpson diversity, species richness). The extent to which the data for these indicators cover the California Current Large Marine Ecosystem (CCLME) varies among taxa. The groundfish data span the US west coast (~ 32-48 °N, ~50-1200 m). Thus, conclusions for indicators based on the groundfish data set (MTL, scavenger biomass, species richness, species density and Simpson diversity) are applicable to the full extent of the CCLME. For coastal pelagic fishes, seabirds and copepods the data analyzed here are currently limited to the northern California current (NCC). See the Ecological Indicators: Data Sources and Methodology for a more complete discussion of the data sets.

Indicators of Ecological Integrity showed generally neutral or positive results, although there are some potentially important negative trends (summarized in Fig. EN.S., EN1). All indicators showed little in the way of long-term trends (quantified by comparing the mean of the last five years of the data set to the mean of the full time series). Given that most data sets are relatively short time series, this particular threshold is probably not highly informative since the last five years of many data sets contribute 50% or more of the data the time series.

DATA ANALYSIS AND PRESENTATION

The status of each indicator is evaluated against two criteria: recent short-term trend, and status relative to the long-term mean—reported as short-term status and long-term status, respectively. This approach holds for those indicators for which thresholds have not currently been set. For those indicators with established thresholds, those specific thresholds are used to evaluate the indicators.

Short-term trend. An indicator is considered to have changed in the short-term if the trend over the last five years of the time series showed an increase or decrease of more than 1.0 standard deviations (s.d.) of the mean of the entire time series.

Status relative to the long-term mean. An indicator is considered to be above or below historical norms if the mean of the last five years of the time series differs from the mean of the full time series by more than 1.0 s.d. of the full time series.

Most indicators (12 of 21) showed no short-term trend (trend over the last five years changed by less than 1.0 s.d. of the full time series) suggesting fairly stable status.

Six indicators increased in the short-term: groundfish Simpson diversity, groundfish species density, northern copepod anomaly for the winter, copepod species richness and Simpson diversity in the winter, and Simpson diversity for seabirds on the continental shelf in June. Additionally, some indicators like the northern copepod anomaly in the summer, while not triggering threshold values, suggest generally good ocean conditions.

Three indicators decreased in the short-term (trend over the last five years of the data decreased by more than 1.0 s.d. of the full time series). These include MTL of groundfishes, and seabird species richness and Simpson diversity in near shore waters for birds observed resting on-water. The causes of the decline in seabird diversity on the shelf are not known, but may indicate more recent changes in offshore conditions that have not been detected in other indicators. Seabird data are the most recent (June 2012), while many other indicator time series are current only to 2011 or the winter of 2011-12.

The decline in MTL represents approximately a 25% decrease in the primary production required to support the assemblage (Pauly and Christensen 1995, Essington et al. 2006) and may represent an important shift in the trophic structure of the groundfish community (but a temporary one within normal fluctuation, see

Branch et al. 2010). Previous work (Keller et al. 2012) suggests that proximal cause of the decrease in groundfish MTL was due to a decline in abundance of Pacific hake *Merluccius productus* and spiny dogfish *Squalus acanthias*. Hake in particular consume large amounts of forage fish and krill, and their lower abundance may mean an increase in food resources for other species that utilize these prey.

The decline in the biomass of Pacific hake and spiny dogfish is also likely the cause of the increase in Simpson diversity for groundfishes. Both fish are high biomass species. Because Simpson diversity is a measure of equitability of species abundance, a decline in the abundance of dominant species like hake and dogfish would lead to greater equitability (evenness) in the community.

Looking to the future

In the current IEA, the data for groundfishes span the US west coast. However, time series for the coastal pelagic fishes, seabirds and copepods were largely limited to the northern portion of the California current. Data exist for the central and southern California current for some of these taxa, but they were not available in the appropriate format in time to include in the present IEA. In the future, these additional data should be included so that the indicators span the CCLME and provide a more complete picture of the status of the CCLME.

Future work should also look to integrate or select data for those taxa with multiple data sources. In some cases, multiple data sets exist that were used to calculate the selected indicators. For example, three data sets (one with two time periods) were available for seabirds representing nearshore and shelf environments. Using multiple data sets produces a plethora of indicators, which can be difficult to evaluate. Combining multiple data sources to produce a single or at least fewer indicators for each taxon without losing important regional information (i.e., NCC vs. SCC) should be a goal of future work.

Indicators of trophic structure are currently limited to only several taxa – primarily groundfishes (and three crabs) and copepods. MTL time series for coastal pelagic fishes, seabirds and mammals will require some development. Many of the available data sets for these taxa are counts, while MTL is a biomass weighted average. Count data will, therefore, need to be converted to biomass using length-weight relationships or average adult biomass as appropriate for the taxon in question. Ultimately, efforts should focus on using these time series to produce a composite MTL or top predator biomass index that spans the geographic extent of the CCLME.

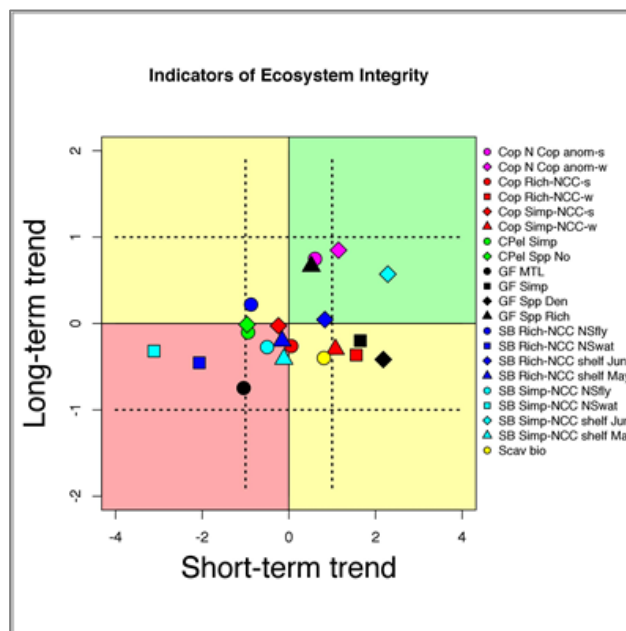


Figure EN.S.1. Short and long-term status of indicators of Ecological Integrity for the California Current Large Marine Ecosystem. Prior to plotting, time series were normalized to place them on the same scale. The short-term trend indicates whether the indicator increased or decreased over the last 5 years of the time series relative to the long-term mean. The long-term axis compares the mean of the last 5 years to the mean of the full time series. Changes or differences of more than 1.0 s.d. of the full time series (dotted lines) are considered to show an effect. Cop = copepod, Cpel = coastal pelagics, GF = groundfishes, SB = seabirds, anom = anomaly, Rich = species richness, Simp = Simpson diversity, Den = species density, Scav bio = scavenger biomass (GF and crabs), NS = nearshore

MEAN TROPHIC LEVEL (GROUNDFISHES AND COASTAL PELAGIC FISHES)

Mean trophic level (MTL) is the biomass weighted average of the trophic levels of the species in a sample (Pauly et al. 1998). It is widely used as an indicator of change in trophic structure (Pauly and Watson 2005). Conceptually, a decrease in the abundance of higher trophic level predators (whether absolute or relative to lower trophic level taxa) influences the strength of trophic cascades and top-down forcing in the ecosystem. MTL comes in two forms (Branch et al. 2010). 'Catch MTL' is calculated from fisheries dependent data and reflects changing fishing practices and availability of target species. 'Ecosystem MTL' is calculated from fisheries independent data and represents changes in the ecosystem. MTL for groundfishes only is reported here. In the future, for an extended range of taxa (coastal pelagics, seabirds, mammals, highly migratory fishes etc) should be incorporated into the IEA either as individual indicators or as a composite MTL indicator.

MTL - GROUNDFISHES

Groundfish MTL was calculated from the West Coast Groundfish Bottom Trawl Survey, and therefore, produced an estimate of Ecosystem MTL for groundfishes.

MTL for groundfishes declined from 2003 until 2010 increasing marginally in 2011 (Fig. EN.S.2). The fluctuation over the entire time series was approximately 0.077 points from a high of 3.72 in 2004 to a low of 3.64 in 2010, which represents a ~25% decrease in the primary productivity required to support a given amount of catch (Pauly and Christensen 1995, Essington et al. 2006). While threshold values for MTL have not been set, future work could set thresholds based on changes in necessary primary production.

Over the last five years of the time series, groundfish MTL declined by more than 1.0 s.d. of the long-term mean. The mean of the last five years of the time series is within 1.0 s.d. of the full time series. However, given the short length of the time series (nine years), comparing the mean of the last five years with the mean of the full time series is of limited value, and the reader should focus on the short-term trend. Comparisons with other long-term data sets suggests that fluctuations in MTL are not uncommon (Branch et al. 2010).

INTERPRETATION OF TIME SERIES FIGURES

Time series figures. Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last five years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the modeled trend over the last 5-years increased (\nearrow), or decreased (\searrow) by more than 1.0 s.d., or was within one 1.0 s.d. (\leftrightarrow) of the long-term trend. The lower symbol indicates whether the mean of the last five years was greater than (+), less than (-), or within (\cdot) one s.d. of the long-term mean.

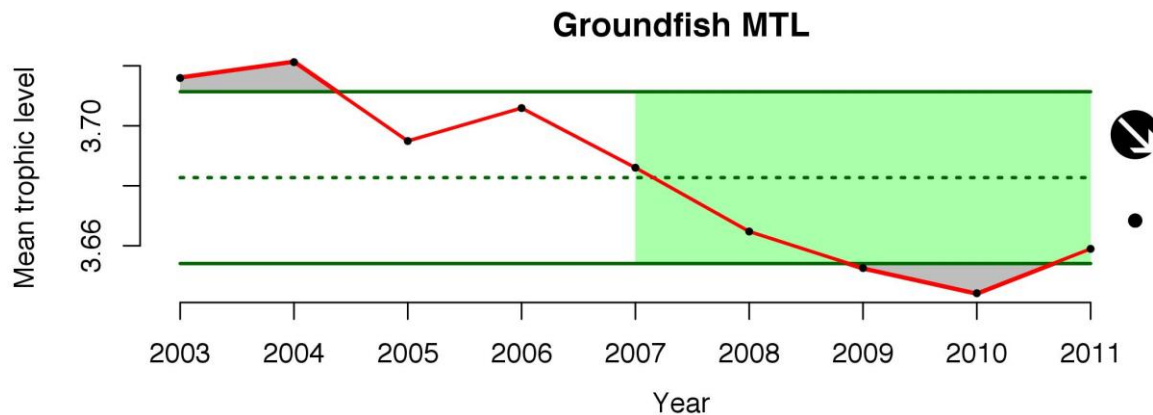


Figure EN.S.2. Area-weighted mean trophic level (MLT) for west coast groundfishes from 2003 – 2011. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness

MTL - COASTAL PELAGIC FISHES, SEABIRDS, MARINE MAMMALS

In development

SCAVENGER BIOMASS (GROUNDFISHES & CRABS)

Scavengers are generally defined as active foragers of carrion (Britton and Morton 1994). Changes in the biomass of scavengers are related to responses to fisheries discards and disturbance of bottom habitat due to trawling. The indicator presented here includes groundfishes and three species of crab quantified in the West Coast Groundfish Bottom Trawl Survey (see Data Sources and Methodology).

After an initially steep decline from 2003 to 2005, scavenger biomass has been fairly stable (Fig. EN.S.3). The trend over the last five years showed an increase of less than 1.0 s.d. of the full time series. The mean of the last five years was within 1.0 s.d. of the long-term mean. However, given the short length of the time series (nine years), comparing the mean of the last five years with the mean of the full time series is of limited value, and the reader should focus on the short-term trend.

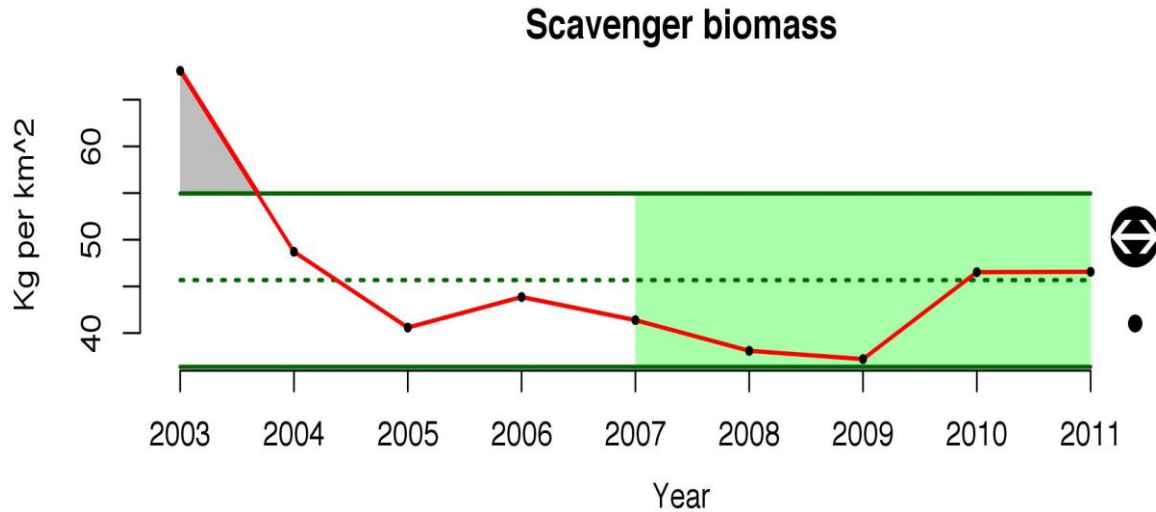


Figure EN.S.3. Estimated biomass (CPUE) of groundfish and crab scavengers for the west coast shelf and slope from 2003-2011. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

Figure EN.S.3. Estimated biomass (CPUE) of groundfish and crab scavengers for the west coast shelf and slope from 2003-2011. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

SIMPSON DIVERSITY (GROUNDFISHES, COASTAL PELAGIC FISHES, SEABIRDS & COPEPODS)

Along with species richness, evenness is one of the two components of diversity. Simpson diversity (in the $1-\lambda$ form, a.k.a. Gini-Simpson index) is a measure of the equitability of species in a sample (Tuomisto 2012). When individuals are well distributed among species, Simpson diversity is high. For large samples, it approximates the probability of an interspecific encounter and is relevant to predator-prey relationships and food-web analyses. Diversity is related to community stability, production and ecosystem function.

SIMPSON DIVERSITY - GROUNDFISHES

Simpsons index ($1-\lambda$) for west coast groundfishes decreased between 2003 and 2008 (Fig. EN.S.4). Over the last five years (2007-2011) Simpson's index increased by more than one standard deviation (s.d.) of the complete time series to levels similar to 2003 and 2004. The mean of the last five years is within one s.d. of the long-term mean. However, given the short length of the time series (nine years), comparing the mean of the last five years with the mean of the full time series is of limited value, and the reader should focus on the short-term trend.

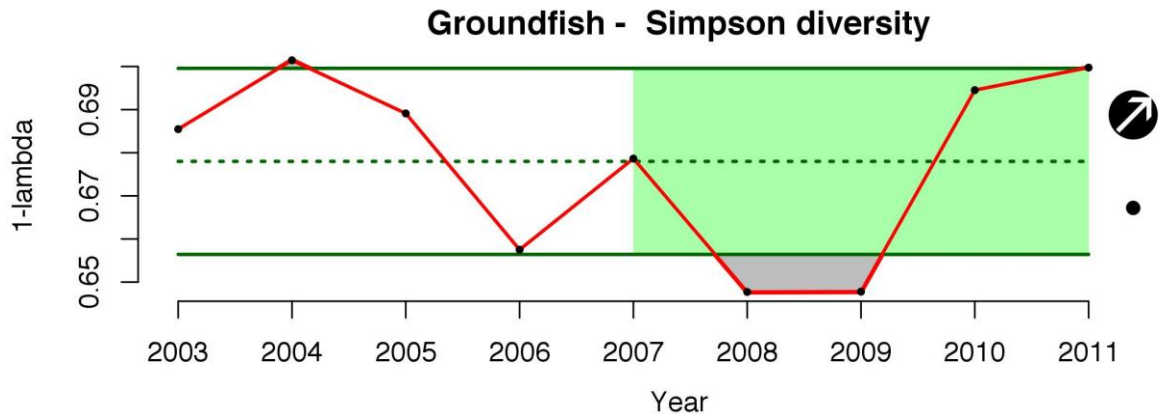


Figure EN.S.4. Simpson diversity ($1-\lambda$) for west coast groundfishes from 2003 to 2011. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

SIMPSON DIVERSITY - COASTAL PELAGIC SPECIES (FISH)

Simpson diversity for coastal pelagic species increased from the beginning of the time series in 1998 and reached a high in 2004 (Fig. EN.S.5). While it has declined since 2004 the trend from 2007 – 2011 (last five years of data) was stable with a minor decrease less than one s.d. of the full time series. The mean of the last five years of the time series was within one s.d. of the long-term mean.

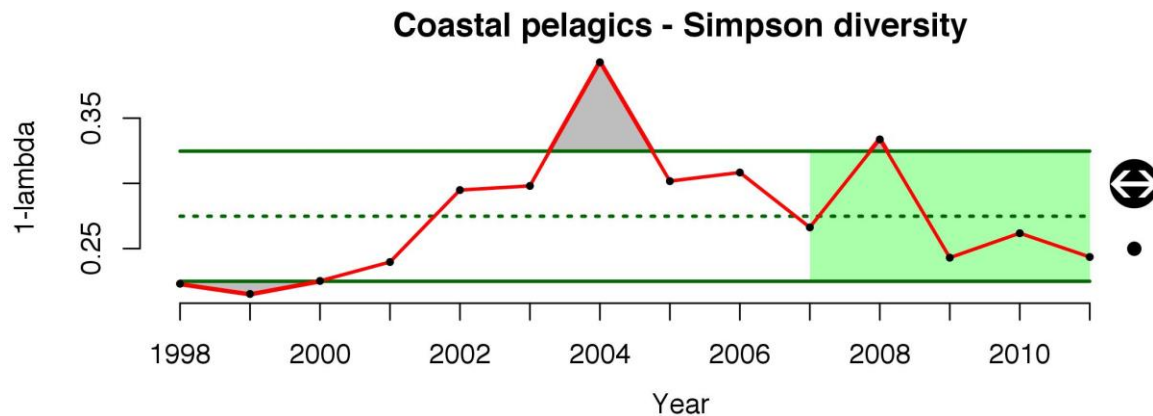


Figure EN.S.5. Simpson diversity ($1-\lambda$) for coastal pelagic fishes in the northern California current from 1998 to 2011.

SIMPSON DIVERSITY – SEABIRDS

Simpson diversity is presented for three surveys (one with two time periods) in the northern California current. Data for the southern California current exist but were not available at the time of publication in proper format for diversity analyses and should be added to future IEAs.

Trends in the nearshore and those on shelf/shelf break differ. For birds in the nearshore observed resting on the water (Fig. EN.S.6), Simpson diversity was fairly stable from 2004 to around 2008, after which it declined steadily to a low in 2012. The trend declined by more than 1.0 s.d. of the long-term mean over the last five years of the data set (2008-2012). While the mean of the last five years was within 1.0 s.d. of the mean of the full time series, the final two years were below the 1.0 s.d. line. Given the short duration of the time series, more emphasis should probably be placed on the trend than on the mean of the last five years. While variable Simpson diversity for nearshore seabirds observed while flying did not show any trends over the last five years of the time series largely because of a low in 2008 (Fig. EN.S.7). Since 2009 there has been a steady decrease in Simpson diversity (over 2.0 s.d.) suggesting that the seabird community in the nearshore is coming to be dominated by fewer taxa. While the threshold (1.0 decline in the last five years) was not triggered in the current IEA year, the nearshore seabird community bears watching.

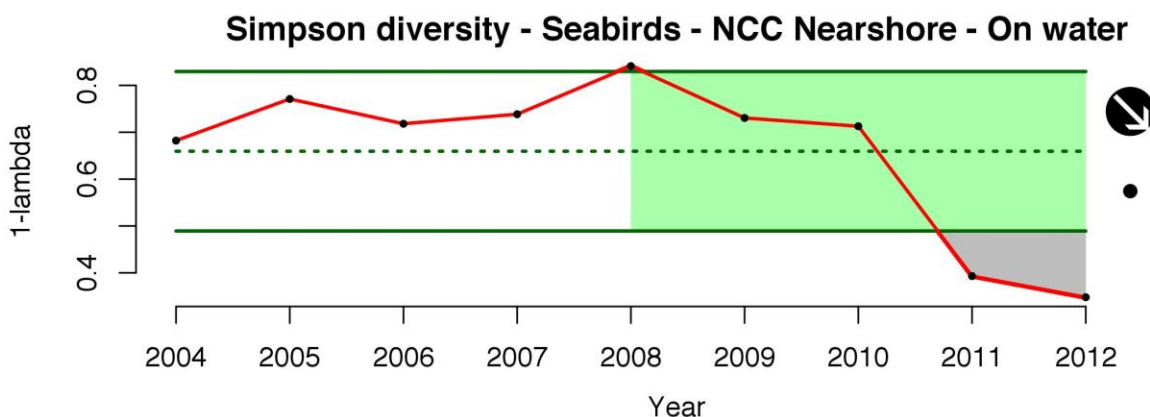


Figure EN.S.6. Simpson diversity for nearshore (within 2 km of shore) seabirds in the northern California current (NCC). Data are those birds observed and quantified while resting on water. Data courtesy of Jen Zamon (jen.zamon@noaa.gov).

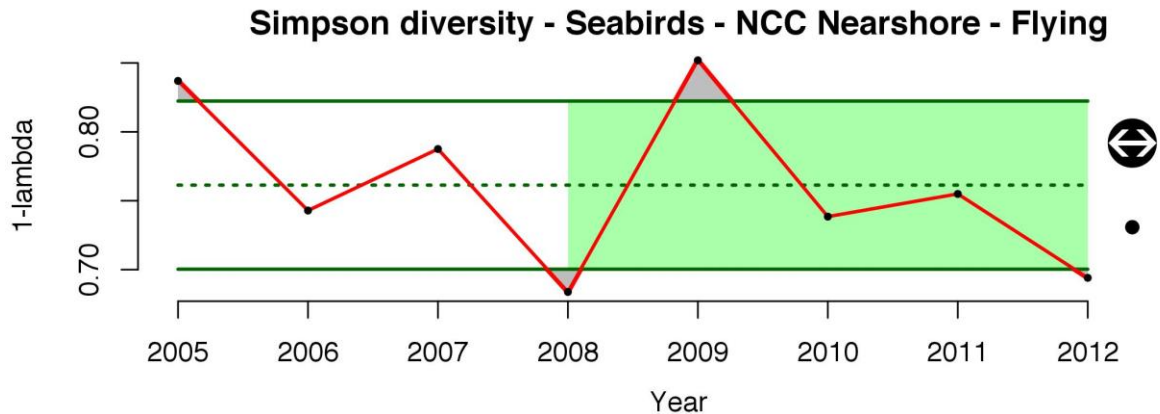


Figure EN.S.7. Simpson diversity for nearshore (within 2 km of shore) seabirds in the northern California current (NCC). Data are those birds observed and quantified while actively flying. Data courtesy of Jen Zamon (jen.zamon@noaa.gov).

In contrast to the nearshore, seabird diversity on the shelf (2-70 km offshore) increased in recent years. This increase was not seen in the data for May (Fig. EN.S.8), but in the June data (Fig. EN.S.9) Simpson diversity for seabirds increased of the last five years of the data series by more than 1.0 s.d. of the full data set. While the mean of the last five years of the data was within 1.0 s.d. of the mean of the full time series, Simpson diversity in June of 2011 and 2012 was above 1.0 s.d. of the mean of the full data set indicating high diversity in the most recent years.

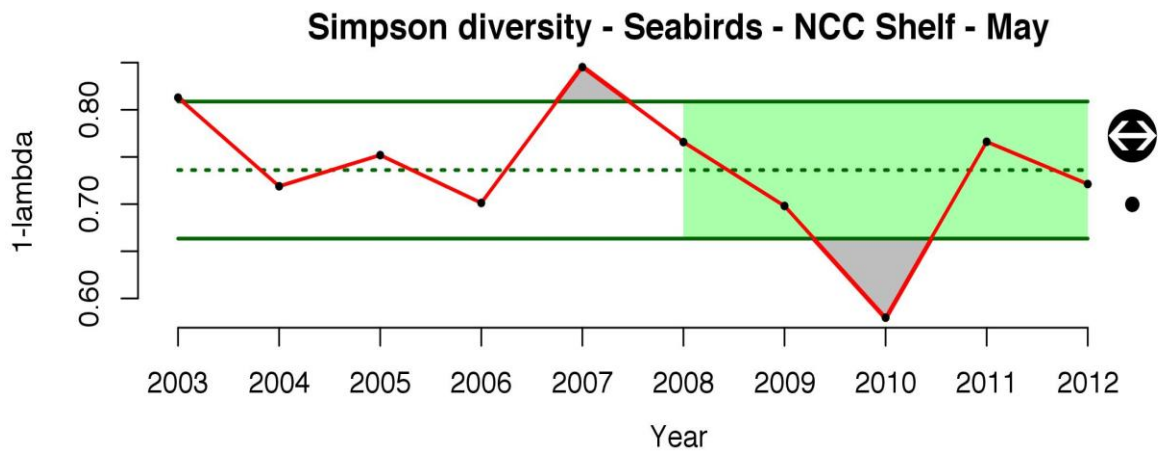


Figure EN.S.8. Simpson diversity in May for seabirds on the shelf and shelf break (2 – 70 km off shore) in the northern California current (NCC). Data courtesy of Jen Zamon (jen.zamon@noaa.gov).

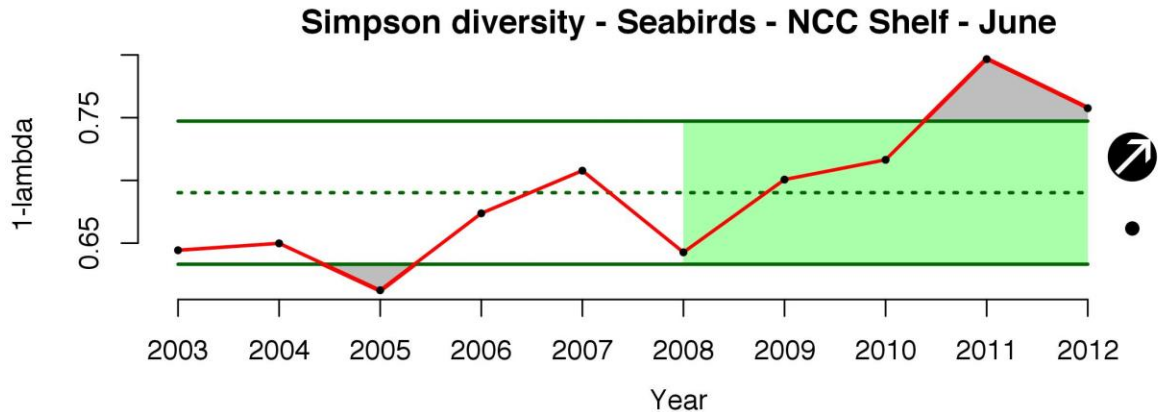


Figure EN.S.9. Simpson diversity in June for seabirds on the shelf and shelf break (2 – 70 km off shore) in the northern California current (NCC). Data courtesy of Jen Zamon (jen.zamon@noaa.gov).

SIMPSON DIVERSITY - COPEPODS

Simpson diversity for copepods was calculated by season using the same seasons as Peterson (2009) (Figs. EN.S.10-11). For all seasons Simpson diversity was variable through time. Simpson diversity for winter (Oct – April) assemblages showed a short-term increase (5-year trend trend showed an increase of greater than 1.0 s.d. of the full time series), but this increase is largely due to a rise from a historic low value in 2008. Visual inspection suggests that the trend has shifted to declining over the last two years. The mean of the last five years was within historical norms. Although the five-year trend showed an increase, closer inspection suggests that Simpson diversity is cycling within typical levels and has actually declined over the last two years. Simpson diversity for summer (May – Sept) assemblages showed no trend and the mean of the last five years was within 1.0 s.d. of the mean of the full time series.

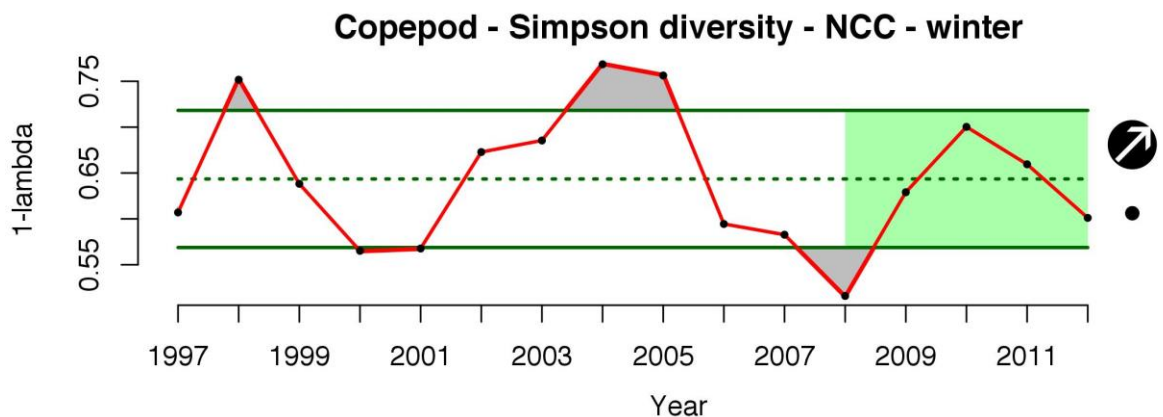


Figure EN.S.10. Time series of Simpson diversity ($1-\lambda$) from 1997 – 2012 for the winter (Oct–April) for west coast copepods in the northern California Current (NCC). Data courtesy of Bill Peterson (bill.peterson@noaa.gov).

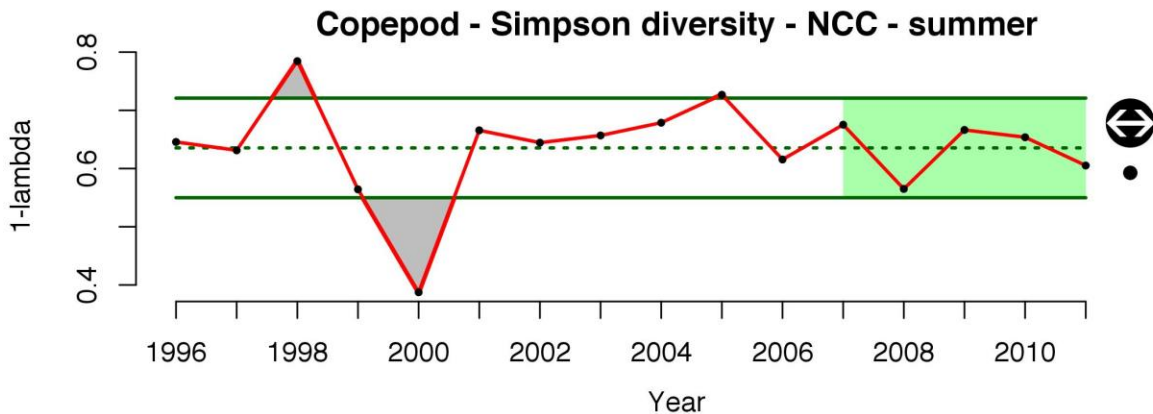


Figure EN.S.11. Time series of Simpson diversity ($1-\lambda$) from 1996 – 2011 for the summer (May – Sept) for west coast copepods in the northern California Current (NCC). Data courtesy of Bill Peterson (bill.peterson@noaa.gov).

SPECIES RICHNESS & DENSITY (GROUNDFISHES, COASTAL PELAGIC FISHES, SEABIRDS & , COPEPODS)

Along with evenness, richness is one of the two components of diversity and is easily understood as the number of species in a community. Richness is important for many ecological models, and there is a substantial literature on the complex relationship between biodiversity and ecosystem function (Hooper et al. 2005, Stachowicz et al. 2007), with some consensus that community-level processes are more stable at higher richness.

Because sample effort, whether the number of individuals collected or area surveyed or both, has strong, non-linear effects on the number of species encountered, estimates of richness need to be scaled to a common effort level through rarefaction (Gotelli and Colwell 2001). The number of species in a community can then be presented in two forms: species richness and species density. Species richness in the rarefaction sense is the number of species observed for some number of individuals collected. Species density is the number of species per some unit area. Both are relevant to different questions and purposes. Since most theoretical models in ecology are based on *per capita* interactions, species richness is relevant to these models. At the same time, species density is important to conservation and applied purposes since it measures the number of species in a given area. Here, both species richness and species density are reported where possible based on the attributes of the particular data set. See Gotelli and Colwell (2001) further discussion of rarefaction and species richness and species density.

SPECIES RICHNESS & DENSITY - GROUNDFISHES

Species richness for west coast groundfishes increased steadily from 2003-2007 after which it has remained more or less stable (Fig. EN.S.12). The mean of the last five years was within 1.0 s.d. of the mean of the full time series and there was no short-term trend. Given the fairly recent implementation of the West Coast Groundfish Bottom Trawl Survey (< 10 years), the increase in richness in the early years may be related in part to better species identification over the development of the survey.

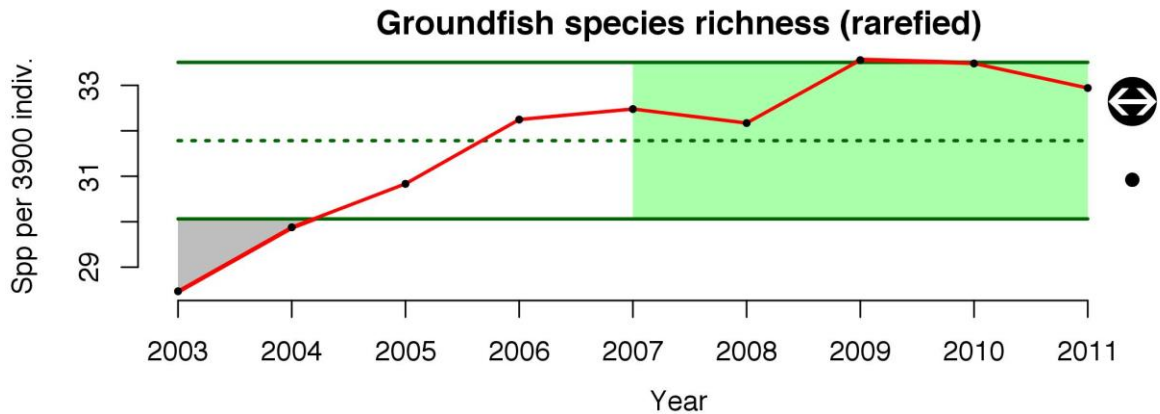


Figure EN.S.12. Species richness for west coast groundfishes from 2003 -2011. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

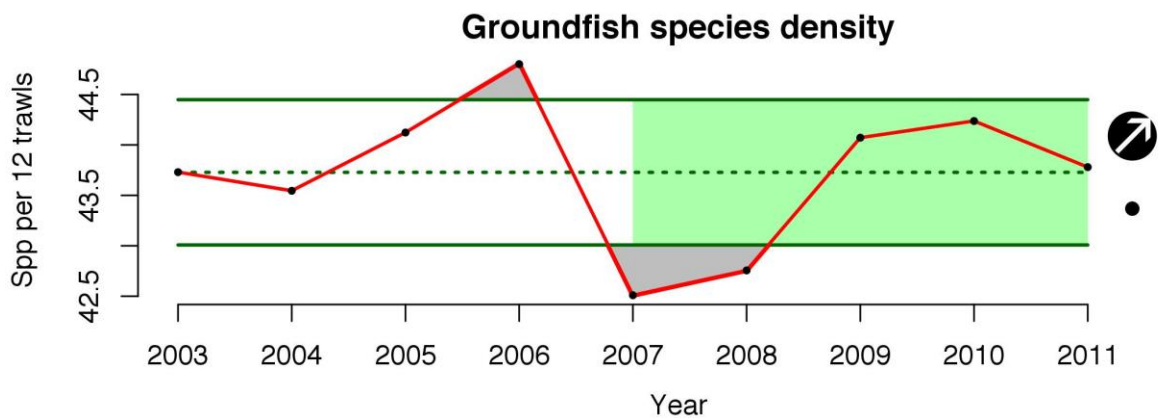


Figure EN.S.13. Area-weighted mean number of groundfish species per 12 trawls for 2003-2011 from the from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

Species density initially increased from 2003 – 2006 then decreased sharply in 2007-2008 (Fig EN.S.13). Over the last five years species density has increased by more than 1.0 s.d. of the full time series. Given the short length of the time series (nine years), comparing the mean of the last five years with the mean of the full time series is of limited value, and the reader should focus on the short-term trend.

The differences between species richness and species density trends seen in the first half of the groundfish data are likely driven by changing number of fishes in the trawl survey (Fig. EN.S.14-15). Both the mean number fishes per trawl and the median number of fishes per trawl declined from 2003 to 2007, after which they remained stable. From 2007 – 2011 species density increased. During this period the number of individuals per haul remained stable suggesting the increase was due to other processes.

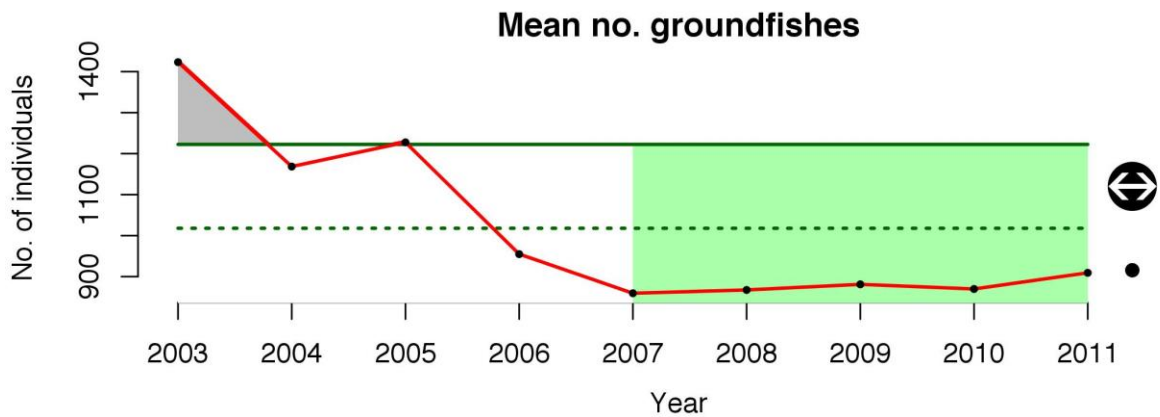


Figure EN.S.14. Mean number of groundfish individuals per trawl 2003-2011 from the from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

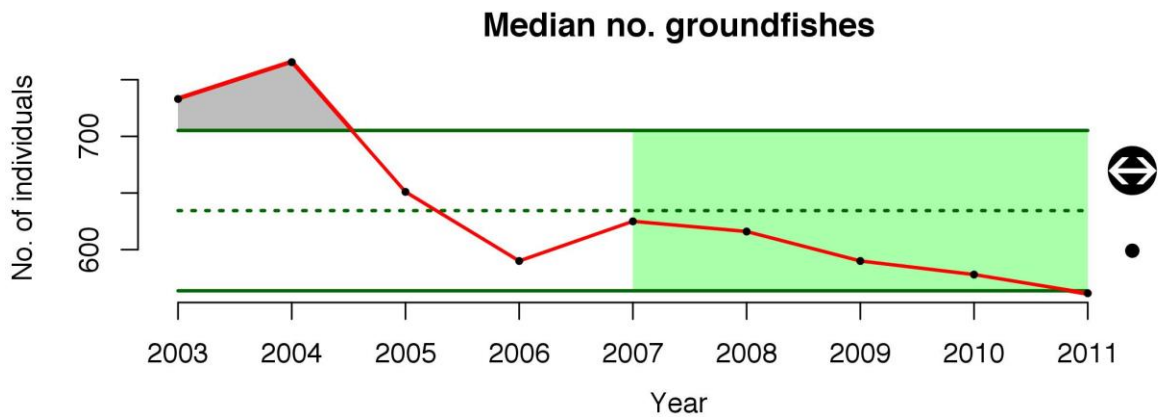


Figure EN.S.15. Median number of groundfish individuals per trawl 2003-2011 from the from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

Note that the trend in species richness seen here differs from that reported by Keller et al. (2012) who reported an initial decrease in the number of species per haul followed by an increase. The two trends differ because Keller et al. (2012) report raw species number per haul, while the data presented here were subject to rarefaction (Gotelli and Colwell 2001, Colwell et al. 2004). Additionally, the data in Keller et al. (2012) are better thought of as species density, since they are species per trawl for trawls with a relatively consistent area. The difference between Keller et al. (2012) and the richness values seen here is likely due to a decrease in the number of individuals per haul through time. Sampling effort (whether number of samples, area sampled or number of individuals collected affects estimates of richness with the number of species increasing with sampling effort non-linearly. Here data were subjected to sample-based rarefaction (since fish school individuals are not sampled at random) and rescaled to 3900 individuals (Colwell et al. 2004).

SPECIES NUMBER - COASTAL PELAGIC FISHES

Data for coastal pelagic fishes were not rarefied due to the data format. Mean number of species per sample for coastal pelagic fishes was variable through time with lows in 1999 and 2000 and 2010 and highs in 2003 and 2004 (Figure EN.S.16). Overall, however, current values are within the long-term norms (the mean of the last five years is within 1.0 s.d. of the long-term mean) with little trend over the last five years (change within 1.0 s.d. of the mean of the full time series).

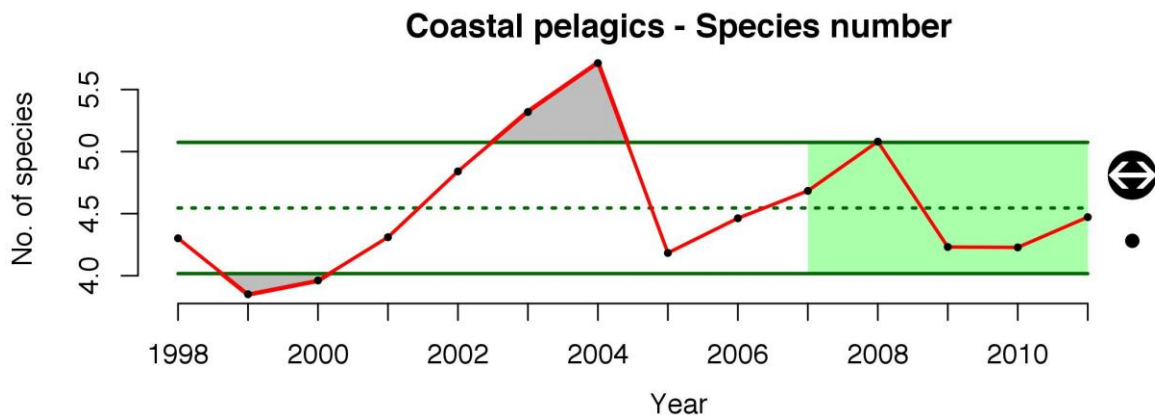


Figure EN.S.16. Mean number of species per sample for coastal pelagic fishes in the northern California current from 1998 to 2011.

TAXON RICHNESS—SEABIRDS

Richness for seabirds is presented as taxon richness since some taxa were pooled at levels above that of species. As seabird density increases it becomes increasingly hard to quantify and identify all individuals to the species level. For example all gulls were included in the taxon 'gulls' and all shearwaters in the taxon 'shearwaters'. Taxon richness is presented for three surveys in the northern California current (METHODS for data details). Data for the southern California current exist but were not available at the time of publication in proper format for diversity analyses and should be added to future IEAs. Data were subjected to individual-based rarefaction (Hurlbert 1971, Gotelli et al. 2009) to standardize for abundance.

Richness for seabirds in the nearshore has declined over the last five years for birds observed resting on the water (Fig. EN.S.17). While the mean over the last five years was within 1.0 s.d. of the mean of the full time series, the data for 2012 show the lowest levels of richness of the whole time series. For the birds observed flying in the nearshore (Fig. EN.S.18), richness declined substantially over the last three years of the data, but this decline was to levels only slightly lower than that seen earlier in the time series.

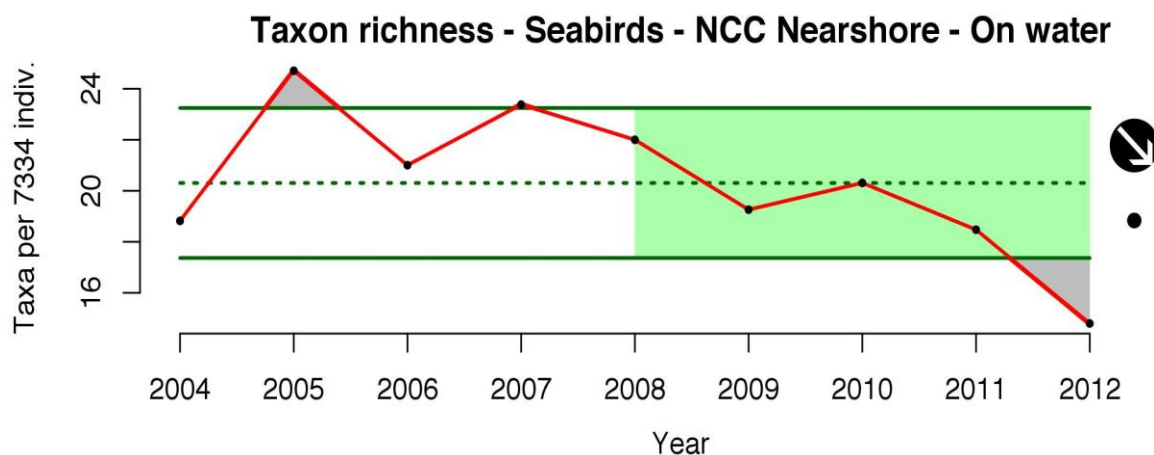


Figure EN.S.17. Taxon richness for seabirds observed while resting on water in the nearshore (< 2km from shore) for 2004-2012 for the northern California current (NCC). Data rarefied to 442 individuals (minimum seen in the time series) using individual-based rarefaction (Hurlbert 1971). Data courtesy of Jen Zamon (jen.zamon@noaa.gov).

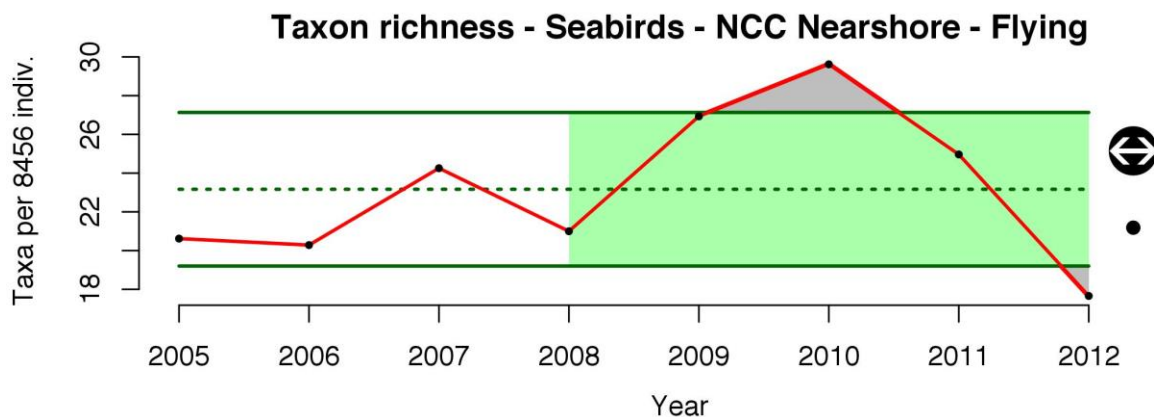


Figure EN.S.18. Taxon richness for seabirds observed while flying in the nearshore (< 2km from shore) for 2005-2012 for the northern California current (NCC). Data rarefied to 442 individuals (minimum seen in the time series) using individual-based rarefaction (Hurlbert 1971). Data courtesy of Jen Zamon (jen.zamon@noaa.gov).

Taxon richness for seabirds on the continental shelf showed no trends over the last five years of data, and the mean of the final five years was within 1.0 s.d. of the long-term mean in both May and June (Fig. EN.S.

19-20). For June data, the trend over the last five years appears to be increasing but the change was less than 1.0 s.d. of the full time series.

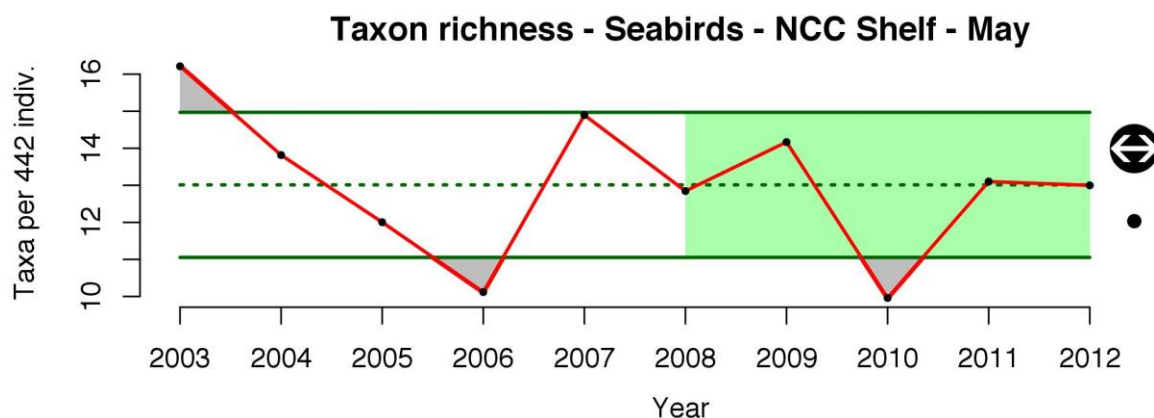


Figure EN.S.19. Taxon richness for seabirds May on the shelf and shelf break (2-70km from shore) for 2003-2012 for the northern California current (NCC). Data rarefied to 442 individuals (minimum seen in the time series) using individual-based rarefaction (Hurlbert 1971). Data courtesy of Jen Zamon (jen.zamon@noaa.gov).

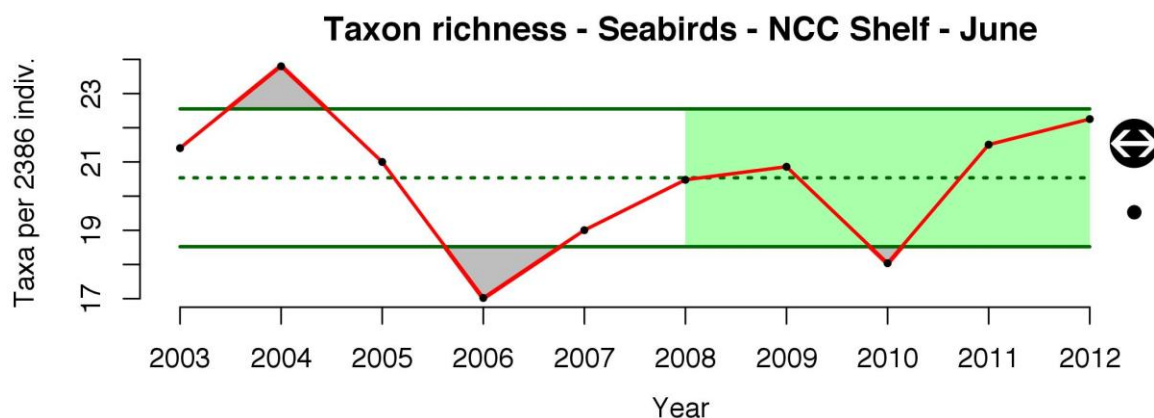


Figure EN.S.20. Taxon richness for seabirds May on the shelf and shelfbreak (2-70km from shore) for 2003-2012 for the northern California current (NCC). Data rarefied to 2386 individuals (minimum seen in the time series) using individual-based rarefaction (Hurlbert 1971). Data courtesy of Jen Zamon (jen.zamon@noaa.gov).

SPECIES RICHNESS - COPEPODS

Copepod species richness has been tied to food chain structure and survival of coho salmon *Oncorhynchus kisutch* in the California current (Peterson 2009). Low species richness is correlated with the southern transport of northern waters, high abundance of lipid-rich northern copepods and increased growth and survival of some species (Peterson 2009). Species density was not calculated for copepods.

Species richness for copepods (quantified as the number of species per sample of approximately 200-400 individuals) was highly variable over time (Figs. EN.S.21-22). Species richness for the winter assemblage showed an increasing trend in the short-term (an increase of more than 1.0 s.d. over the last five years) indicating worsening conditions in the short term. The mean of the last five years was within 1.0 s.d. of the long-term mean. While highly variable, species richness for the summer did not show any recent trends beyond typical cycling seen in the earlier parts of the time series.

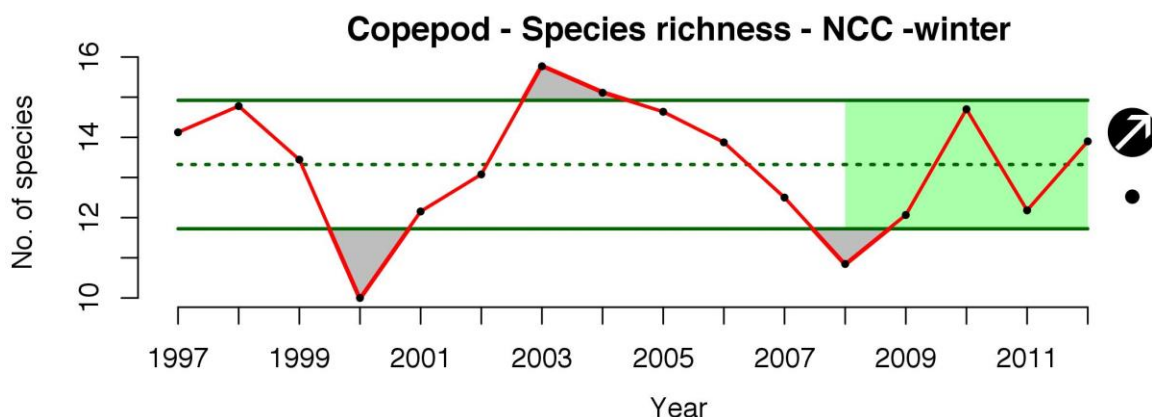


Figure EN.S.21. Copepod species richness for the winter in waters off Oregon in the northern California Current (NCC) from 1997-2012. Data courtesy of Bill Peterson (bill.peterson@noaa.gov).

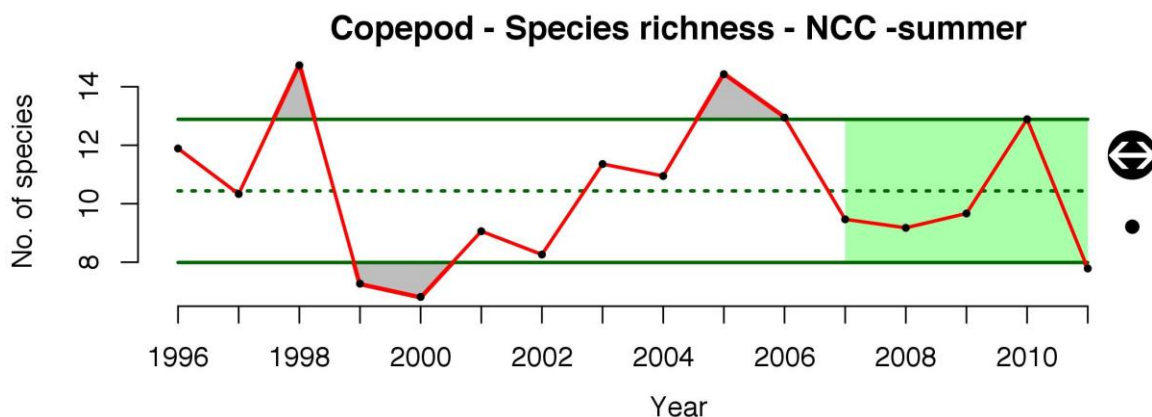


Figure EN.S.22. Copepod species richness for the summer in waters off Oregon in the northern California Current (NCC) from 1997-2012. Data courtesy of Bill Peterson (bill.peterson@noaa.gov).

NORTHERN COPEPOD BIOMASS ANOMALY

The northern copepod biomass anomaly represents the ratio of northern and southern copepod species off of the Oregon coast. Two of the cold-water species, *Calanus marshallae* and *Pseudocalanus mimus*, are lipid-rich, and the index may represent the amount of lipid (wax-esters and fatty acids) available to pelagic fishes for whom these fatty compounds appear to be essential. Beamish and Mahnken (2001)

provide an example of this for coho salmon. See

<http://www.nwfsc.noaa.gov/research/divisions/fed/oeip/eb-copepod-anomalies.cfm> for further detail.

The northern copepod anomaly has fluctuated between 1996 – 2012. In the last five years, the anomaly increased in the winter (Fig. EN.S.23), but there was no trend in the summer (Fig. EN.S.24). For both seasons, the mean of the last five years was within 1.0 s.d. of the long term mean of the full time series, although in the winter values for the last two years were above 1.0 s.d. of the long-term mean, as is the last data point for the summer time series. Threshold values for the anomaly have not been set. However, positive values in the summer period are correlated with stronger returns of fall and spring ocean-type Chinook to Bonneville dam, and values greater than 0.2 are associated with better survival of coho. Overall the high anomalies in recent years, especially for the summer data, suggest that ocean conditions are in a generally good state.

See <http://www.nwfsc.noaa.gov/research/divisions/fed/oeip/eb-copepod-anomalies.cfm> for further detail.

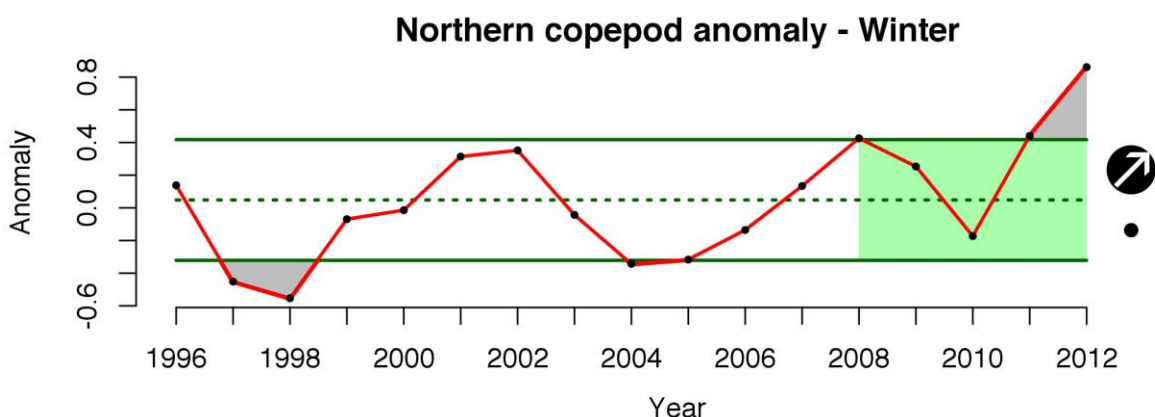


Figure EN.S.23. Northern copepod biomass anomaly for 1996-2012 in the waters off of Oregon during the winter (Oct-April). Data courtesy of Bill Peterson.

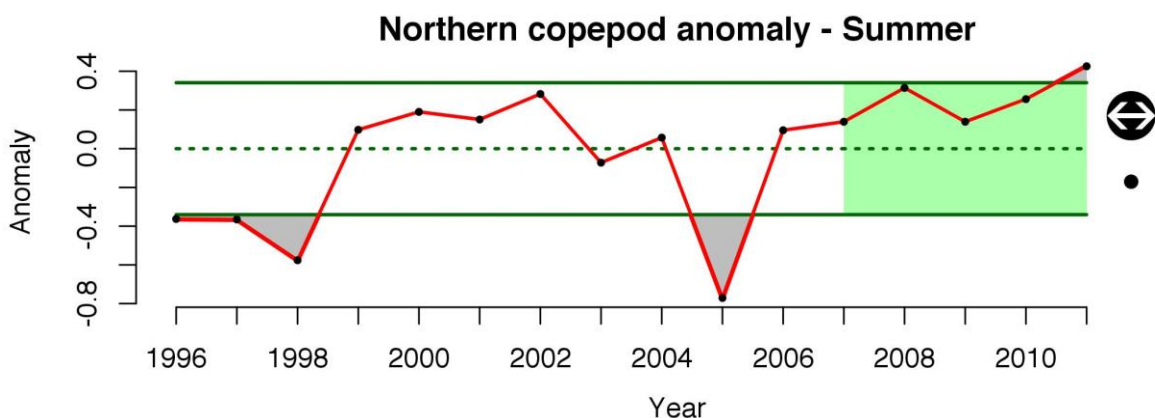


Figure EN.S.24. Northern copepod biomass anomaly for 1996-2012 in the waters off of Oregon during the summer (May - Sept). Data courtesy of Bill Peterson.

DATA SOURCES AND METHODOLOGY

GROUNDFISHES

Data for the groundfish time series come from the Northwest Fisheries Science Center's annual West Coast Bottom Trawl Survey (WCBTS, data courtesy of Beth Horness, beth.horness@noaa.gov) (Keller et al. 2008). The survey is a depth stratified, random sample that spans approximately 32 - 48.5° N and 55-1200 m depth for 2003-2011. Data were limited to those hauls deemed acceptable for stock assessment. Hauls from areas subsequently closed to sampling were not included in analyses. The data include 5743 trawls/hauls from 2003 - 2011 and 313 taxa identified to species. Of these two pairs of rockfishes were combined because of difficulty in discriminating between each species pair in the field. Sunset rockfish *Sebastes crocotulus* and vermillion rockfish *S. miniatus* were combined into one taxa. Blackspotted rockfish *S. melanostictus* and roughey rockfish *S. aleutianus* were also combined into a single taxa. Both combined taxa were included in species level analyses.

AREA-WEIGHTED MEANS (GROUNDFISHES)

Area-weighted means were calculated for mean trophic level, scavenger biomass, Simpson diversity, and species richness because some areas of the shelf and slope are more heavily sampled than others and because the total bottom area of the shelf and slope for any given depth range varies with latitude (Table EN.S.1).

Data (for both groundfishes and bottom area) were binned into five depth zones (<200, 201-600, 600-1200 m depth) and four latitude regions (south of Point Conception [32 - 34.5° N], Point Conception to Cape Mendocino 40.4° N], Cape Mendocino to Cape Blanco [42.5° N], and Cape Blanco to Cape Flattery (actual 48.38 but here to 48.4511° N—the extent of the groundfish data) based on previous analyses on groundfish assemblage structure (Tolimieri and Levin 2006, Tolimieri 2007). The areal extent of each depth x region bin was calculated from the U.S. Coastal Relief Model (<http://www.ngdc.noaa.gov/mgg/coastal/crm.html>). The native coordinate system of these bathymetry data does not conserve area throughout the study region (e.g., a 1 X 1 degree area in the south is larger than a 1 X 1 degree area to the north). To correct this problem, we created a regular 1/10 degree grid over the study area and then re-projected this grid to a Cylindrical Equal-Area projection (units = meters, projection type = 3, longitude of the centre of projection = -122 0' 0.00", latitude of the centre of projection = 56 30' 0.00", Azimuth = 120.95, and Scale factor = 1). The new data layer had the correct area for each 1/10 degree latitude/longitude grid cell. The total area of a given depth x region bin was calculated by summing the area of the relevant grid cells. Each depth x region bin was then assigned a weight equal to its proportion of the total area of all depth x region bins. These weights were then used to calculate the area-weighted mean for each groundfish-based indicator of ecological integrity.

Table EN.S.1. Depth and latitude bins uses in the area-weighted analyses showing total area of the bins, corresponding weights and total number of trawls from 2003 – 2011. See text for depth and latitude borders.

| Depth zone | Latitude zone | Total area (km ²) | Weight | Trawls |
|---------------|---------------|-------------------------------|--------|--------|
| Shelf | Flattery | 36,394 | 0.231 | 1289 |
| Shallow slope | Flattery | 11,020 | 0.070 | 707 |
| Deep slope | Flattery | 10,916 | 0.069 | 439 |
| Shelf | Blanco | 5,407 | 0.034 | 232 |
| Shallow slope | Blanco | 2,182 | 0.014 | 180 |
| Deep slope | Blanco | 5,258 | 0.033 | 220 |
| Shelf | Mendocino | 16,689 | 0.106 | 672 |
| Shallow slope | Mendocino | 8,326 | 0.053 | 511 |
| Deep slope | Mendocino | 12,518 | 0.079 | 420 |
| Shelf | Conception | 10,176 | 0.064 | 314 |
| Shallow slope | Conception | 11,702 | 0.074 | 431 |
| Deep slope | Conception | 27,243 | 0.173 | 270 |

INDICATORS

Mean trophic level (MTL, Pauly and Watson 2005, Branch et al. 2010) was calculated as the biomass-weighted mean trophic level for each haul, which was then used to calculate an area-weighted mean for the west coast shelf and slope (see Area-weighted means, below). Information on trophic level was taken from Fishbase.org. Taxa included in the analyses were all fishes identified to the species level. Data were CPUE biomass (kg per km²) by species per haul.

Previous analyses of MTL (Pauly et al. 2001, Essington et al. 2006, Branch et al. 2010) have generally not corrected for survey area. In part this is because many workers have focused on catch-MTL, which is derived from fisheries catch data. The data used here are from a fishery-independent trawl survey, and therefore, represent ecosystem-MTL. To correctly evaluate the change in the trophic structure of the groundfish assemblage it is important to correct the data for survey area since the total area of various depth x latitude bins is not constant.

Area-adjust MTL for groundfishes is presented in the results section. For comparison, the raw MTL trend is shown below (Fig. EN.S.25). While the over all trend is similar (a decline from 2003), there are important differences. Adjusting for area. Most importantly the absolute level of decline is quite different: 0.077 when adjusting for area versus 0.19 not. A decrease in MTL of ~0.15 represents a decrease of 50% in the primary production required to support the assemblage. Therefore, failing to account for sample area, overestimates the change in the trophic structure and energy requirements for the assemblage in question.

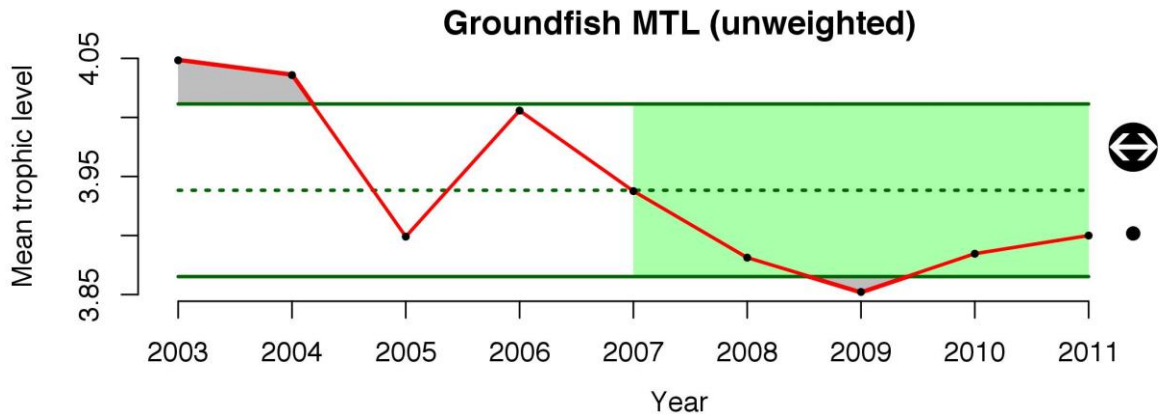


Figure EN.S.25. Mean trophic level for west coast groundfishes from 2003 – 2011. MTL was calculated for survey data without adjusting for sampling effort in different depth x latitude strata. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

Scavenger biomass was calculated from the West Coast Bottom Trawl Survey (WCBTS), which provides quantitative biomass data for groundfishes, as well as for several crab species for 2003-2011. We followed Brand et al. (2007) and Horne et al. (2010) in defining large crabs, large demersal sharks and grenadiers as scavengers (Table EN.S.2). Further detail was taken from Yeh and Drazen (2011) who used baited-cameras to evaluate scavenger ecology on the California slope, and from Buckley et al. (1999) who examine food habits of several groundfishes. Scavengers are generally defined as active foragers of carrion (Britton and Morton 1994). Many of the species on the list are predators that responded strongly to baited cameras (grenadiers) or had large amounts of fisheries offal in their diet (thornyheads and sablefish). While carrion may not normally make up a substantial portion of the diets of these animals in the absence of anthropogenic influences, part of the objective of monitoring scavenger biomass is to track the effects of fisheries on the ecosystem. Therefore, it is relevant to include taxa that respond strongly to these activities. Scavenger biomass was calculated for each haul by summing the CPUE biomass of all taxa. Biomass (CPUE) per haul was then used to calculate the area-weighted mean CPUE per trawl (kg per km², see Area-weighted means, below). Data were biomass by species per km² (CPUE, kg/km²).

Table EN.S.2. Groundfish and decapod taxa included in the quantification of scavenger biomass 2003-2011. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

| <u>Species</u> | <u>Family</u> | <u>Order</u> | <u>Class</u> | <u>Total catch (kg)</u> |
|---------------------------------|---------------|--------------|----------------|-------------------------|
| <i>Albatrossia pectoralis</i> | Macrouridae | Gadiiformes | Actinopterygii | 17,421.2 |
| <i>Coelorinchus scaphopsis</i> | Macrouridae | Gadiiformes | Actinopterygii | 0.4 |
| <i>Coryphaenoides acrolepis</i> | Macrouridae | Gadiiformes | Actinopterygii | 30,221.2 |
| <i>Coryphaenoides cinereus</i> | Macrouridae | Gadiiformes | Actinopterygii | 40.0 |
| <i>Coryphaenoides filifer</i> | Macrouridae | Gadiiformes | Actinopterygii | 0.5 |
| Macrouridae | Macrouridae | Gadiiformes | Actinopterygii | 0.0 |
| <i>Malacocephalus laevis</i> | Macrouridae | Gadiiformes | Actinopterygii | 0.2 |
| <i>Nezumia liolepis</i> | Macrouridae | Gadiiformes | Actinopterygii | 132.6 |
| <i>Nezumia stelgidolepis</i> | Macrouridae | Gadiiformes | Actinopterygii | 257.4 |
| <i>Antimora microlepis</i> | Moridae | Gadiiformes | Actinopterygii | 2,793.2 |

| <u>Species</u> | <u>Family</u> | <u>Order</u> | <u>Class</u> | <u>Total catch (kg)</u> |
|------------------------------------|----------------------|---------------------|---------------------|--------------------------------|
| <i>Anoplopoma fimbria</i> | Anoplopomatidae | Scorpaeniformes | Actinopterygii | 81,309.6 |
| <i>Bolinia euryptera</i> | Cottidae | Scorpaeniformes | Actinopterygii | 4.6 |
| <i>Chitonotus pugetensis</i> | Cottidae | Scorpaeniformes | Actinopterygii | 3.1 |
| <i>Clinocottus acuticeps</i> | Cottidae | Scorpaeniformes | Actinopterygii | 1.5 |
| <i>Enophrys bison</i> | Cottidae | Scorpaeniformes | Actinopterygii | 2.9 |
| <i>Enophrys taurina</i> | Cottidae | Scorpaeniformes | Actinopterygii | 9.3 |
| <i>Gymnocanthus tricuspid</i> | Cottidae | Scorpaeniformes | Actinopterygii | 2.4 |
| <i>Hemilepidotus hemilepidotus</i> | Cottidae | Scorpaeniformes | Actinopterygii | 3.0 |
| <i>Hemilepidotus spinosus</i> | Cottidae | Scorpaeniformes | Actinopterygii | 3.5 |
| <i>Icelinus borealis</i> | Cottidae | Scorpaeniformes | Actinopterygii | 0.0 |
| <i>Icelinus burchami</i> | Cottidae | Scorpaeniformes | Actinopterygii | 1.7 |
| <i>Icelinus filamentosus</i> | Cottidae | Scorpaeniformes | Actinopterygii | 480.6 |
| <i>Icelinus fimbriatus</i> | Cottidae | Scorpaeniformes | Actinopterygii | 4.2 |
| <i>Icelinus tenuis</i> | Cottidae | Scorpaeniformes | Actinopterygii | 0.6 |
| <i>Jordania zonope</i> | Cottidae | Scorpaeniformes | Actinopterygii | 1.6 |
| <i>Leptocottus armatus</i> | Cottidae | Scorpaeniformes | Actinopterygii | 67.5 |
| <i>Paricelinus hopliticus</i> | Cottidae | Scorpaeniformes | Actinopterygii | 0.0 |
| <i>Radulinus asprellus</i> | Cottidae | Scorpaeniformes | Actinopterygii | 2.8 |
| <i>Radulinus taylori</i> | Cottidae | Scorpaeniformes | Actinopterygii | 0.1 |
| <i>Scorpaenichthys marmoratus</i> | Cottidae | Scorpaeniformes | Actinopterygii | 5.0 |
| <i>Triglops macellus</i> | Cottidae | Scorpaeniformes | Actinopterygii | 0.2 |
| <i>Sebastolobus alascanus</i> | Scorpaenidae | Scorpaeniformes | Actinopterygii | 35,692.5 |
| <i>Sebastolobus altivelis</i> | Scorpaenidae | Scorpaeniformes | Actinopterygii | 101,664.0 |
| <i>Sebastolobus sp.</i> | Scorpaenidae | Scorpaeniformes | Actinopterygii | 0.0 |
| <i>Hexanchus griseus</i> | Hexanchidae | Hexanchiformes | Chondrichthyes | 102.5 |
| <i>Eptatretus deani</i> | Myxinidae | Myxiniformes | Myxini | 219.8 |
| <i>Eptatretus sp.</i> | Myxinidae | Myxiniformes | Myxini | 2.0 |
| <i>Eptatretus stouti</i> | Myxinidae | Myxiniformes | Myxini | 89.7 |
| <i>Cancer magister</i> | Canceridae | Decapoda | Malacostraca | 42,227.1 |
| <i>Chionoecetes bairdi</i> | Oregoniidae | Decapoda | Malacostraca | 14.6 |
| <i>Chionoecetes tanneri</i> | Oregoniidae | Decapoda | Malacostraca | 27,762.5 |

Simpson Diversity (1- λ , Gini-Simpson index) was calculated (using the vegan package in R 2.15 (R Development Core Team 2012)) for each haul and then used to calculate an area-weighted mean for the west coast shelf and slope (see Area-weighted means, below). Taxa included in the analyses were all fishes identified to the species level (319 species). Data were number of individuals by species per haul.

Species Richness and Species Density were calculated for each depth x latitude bin (see Area-weighted means, below) and then used to derive the area-weighted mean for the combined west coast shelf and slope. Because the number of species per sample will increase in relation to sampling effort non-linearly (either area or number of individuals), the groundfish data required rarefaction to standardize annual estimates of species richness and species density (Gotelli and Colwell 2001). Individual based rarefaction assumes that individuals are randomly distributed in space or time (Gotelli and Colwell 2001), and is, therefore, not the

best approach for fishes since many species school. Individual based rarefaction will tend to overestimate species richness when individuals are patchily distributed (Colwell et al. 2004). Instead, sample-based rarefaction was used to calculate species density curves by pooling samples (trawls) in depth x latitude bins. Rarefaction was conducted using the 'specaccum' function in the 'vegan' package for R 2.15 (Mao Tau, Colwell et al. 2004, R Development Core Team 2012).

Sample-based rarefaction curves initially produce estimates of species density (species per area) and not species richness (species *per capita*), and data need to be re-scaled to the number of species per some number of individuals based on the mean number of individuals per trawl (Gotelli and Colwell 2001). Since the purpose was not to estimate mechanism, a high-order polynomial was chosen to maximize fit (minimum $r^2 = 0.9988$ for all regressions) and to achieve the best fits at the lower end of the data range where more mechanistic models overestimate the number of species. Rarefaction by samples produced an estimated number of species per cumulative number of samples from one to the total number of samples for each depth x latitude bin. For each depth x latitude bin, the mean number of individuals per haul was calculated and multiplied by the number of samples (1 to total) to give the cumulative expected number of individuals per total samples. This number of individuals per total samples was then used in the regression analysis to predict richness. Rarefied data were rescaled to the number of species per 3900 individuals. The reference value of 3900 individuals was chosen based on two criteria. First, it was larger than the smallest mean number of individuals per sample for all depth x latitude bins so that all predicted values were from within the range of the data. Second it was just smaller than the number of individuals (3991) from the depth x latitude bin with the lowest total number of individuals per 12 samples (the lowest number of samples from a depth x latitude bin by year). Species richness for a given year was then calculated as the area-weighted mean richness for that year.

For background the raw number of species per trawl is presented here (Figure EN.S.26) and should be considered analogous to species density since the data are species per area (trawls of more or less consistent swept area). Raw species density declined around 2007-2008 but then subsequently increased.

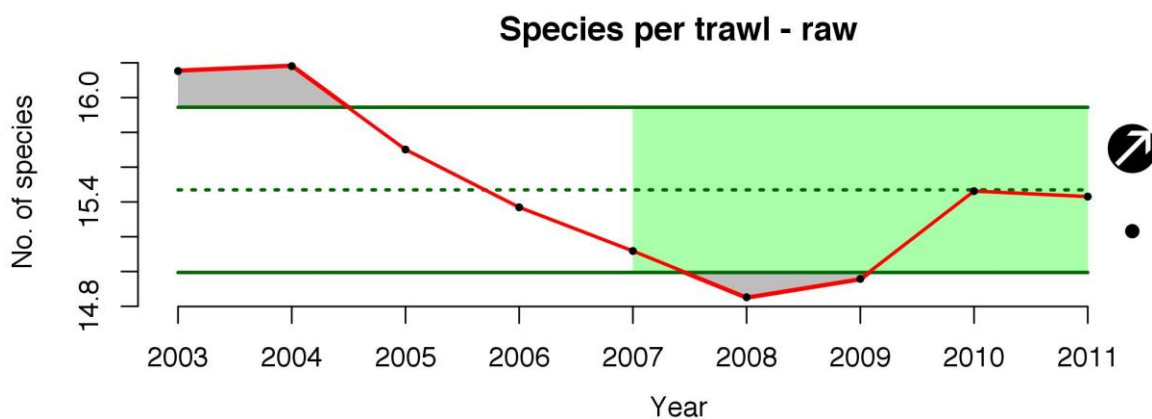


Figure EN.S.26. Mean number of groundfish species per trawl for 2003-2011 from the West Coast Groundfish Bottom Trawl Survey. Species number values were not subjected to rarefaction. Data courtesy of Beth Horness (Beth.Horness@noaa.gov).

The initial decrease in species per trawl is probably related to a decline in the number of individuals per trawl (see Species Richness & Species Density - Groundfishes). The increase in species richness later in the time series may be attributable to better species identification. The number of species identified in any given year has increased over the time series by about 25-30 species from 2003 to 2011 (Fig. EN.S.27a). The total number of species recorded by the trawl survey has steadily increased as expected by species-area relationships (Fig. EN.S.27b).

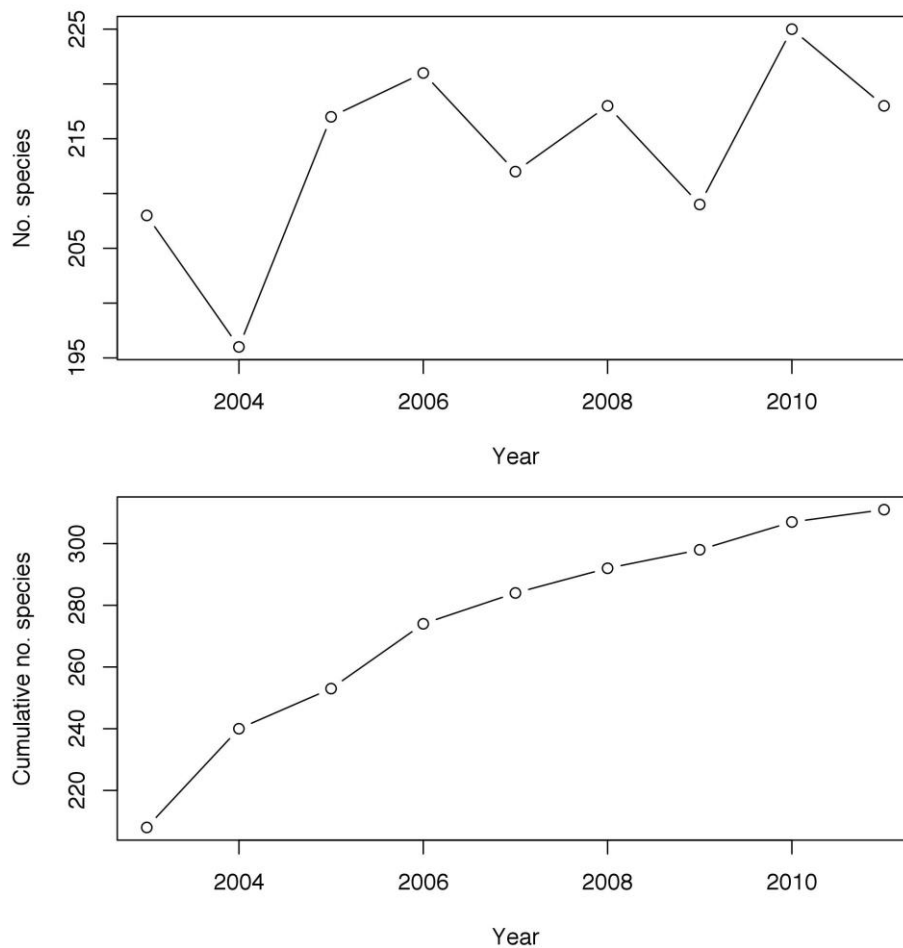


Figure EN.S.27. (A) Number of species identified in any given year by the trawl survey, (B) cumulative number of species identified by the trawl survey.

COASTAL PELAGIC FISHES

Data courtesy of Rick Brodeur (Richard.brodeur@noaa.gov). See the 'Data Sources and Methodology' for the Coastal Pelagic Fishes section for details on the data collection and processing.

Mean Trophic Level for coastal pelagic fishes (MTL, Pauly and Watson 2005, Branch et al. 2010) was calculated as the biomass-weighted mean trophic level for each haul and then averaged for each year. Information on trophic level was taken from Fishbase.org and updated based on current research (R. Brodeur, unpublished data). Taxa included in the analyses were all fishes identified to the species level. Counts of individuals per haul were transformed to biomass per haul using length-weight relationships and size class data from the survey..

Simpson Diversity (1- λ , Gini-Simpson index) was calculated for each sample and then averaged for each year. Samples from June and September were pooled to create a single annual value.

Species richness data for coastal pelagic fishes were not subject to rarefaction as they were not count data), and are raw estimates of species per sample. The number of species was calculated for each sample and then averaged for each year. Samples from June and September were pooled to create a single annual value.

SEABIRDS

Data are courtesy of Jen Zamon (jen.zamon@noaa.gov). For specifics on seabird data sets and collection see 'Data Sources and Methodology' in the Seabirds section.

Data analyzed here were total counts for the BPA and Lighthouse (On-water and Flying) data sets. The Lighthouse data are for the nearshore (0-2 km from shore). The BPA data set is for the shelf and shelfbreak (2-70 km from shore). Both are for the northern California current. The data are total counts by species or taxa for the sampling period (month or year). As seabird density increases, it become increasing difficult to quantify all individuals to species and some taxa are pooled into broader groups during the survey. Therefore, for the analyses some species were collapsed to higher-level taxa for analysis. For the BPA data all gulls, shearwaters and phalarope species were pooled into broader taxa (i.e., gulls, shearwaters and phalaropes) for analysis. For the Lighthouse data seabirds were pooled into cormorants, gulls, phalaropes, scaups, scoters, geese, ducks, loons, and shorebirds. Individual species not belonging to one of these groups remained categorized as individual species.

Simpson Diversity (1- λ , Gini-Simpson index) was calculated for each time period based on total counts within that sampling period.

Species richness was calculated using individual-based rarefaction on the total counts of individuals within a sampling period using the 'rarefy' function in the 'vegan' package for R. (Hurlbert 1971, R Development Core Team 2012).

COPEPODS

Data courtesy of Bill Peterson (bill.peterson@noaa.gov). See Peterson (2009) for details on the data collection and processing.

Note that the data are for the 'Newport Line' near Newport OR and do not span the full coast. Future IEA efforts should work to incorporate available data sets to produce better coastwide estimates zooplankton dynamics. Work has shown that copepod diversity calculated from this data source is a good predictor of system characteristics and correlates with population dynamics of some salmon species (Peterson 2009).

Simpson Diversity (1- λ , Gini-Simpson index) was calculated for each sample/tow then averaged for each year by season following XXX: winter (Oct - April) and summer (May - Sept). Data were number of individuals by taxa per m³. Winter means included data from the previous calendar year. That is, winter 2000 was the average of data from Oct - Dec 1999 and Jan - April 2000.

Species richness estimates of species per sample were not subject to rarefaction as in the case of groundfish. Enumeration of zooplankton data uses subsamples of a generally consistent number of individuals (200-400

individuals per sample for copepods, Peterson 2009), and therefore, does not require rarefaction to account for differences in sampling effort.

NORTHERN COPEPOD BIOMASS ANOMALY

Data courtesy of Bill Peterson (bill.peterson@noaa.gov).

See <http://www.nwfsc.noaa.gov/research/divisions/fed/oeip/eb-copepod-anomalies.cfm> for a discussion of the mechanisms behind the northern copepod biomass anomaly and for methodology in calculating the index. Seasonal estimates of the anomaly were calculated winter (Oct - April) and summer (May - Sept). Winter means included data from the previous calendar year. That is, winter 2000 was the average of data from Oct - Dec 1999 and Jan - April 2000.

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CHINOOK AND COHO SALMON

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TABLE OF CONTENTS (S)

| | |
|-----------------------------------|-----|
| Executive summary..... | 249 |
| Detailed report..... | 251 |
| Indicator selection process | 251 |
| Indicator evaluation..... | 251 |
| Status and trends..... | 272 |
| Major findings..... | 272 |
| Summary and status of trends..... | 272 |
| Risk..... | 293 |
| References Cited | 293 |

LIST OF TABLES AND FIGURES (S)

| | |
|---|-----|
| Salmon abundance. Quadplot summarizes information from multiple time series of coho and Chinook salmon abundances. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. | 250 |
| Table S1. Key indicators for salmon, identified during the ESA listing and recovery planning processes. Indicators categories chosen for this analysis are in <i>bold italic</i> font. | 252 |
| Table S2. California ESUs/Stocks and Data available for Abundance Estimates. Those series indicated by <i>bold italics</i> were used for analyses. Period is the period of availability for the longest series for that population. | 257 |
| Table S3. Data series that met the criteria for inclusion in the condition analyses of California ESUs. Period is the period of availability for the longest series for that population. | 261 |
| Table S4. Oregon-Washington ESUs/stocks and data available for abundance estimates. Each of these series met the criteria for inclusion in the analyses and was used. | 262 |
| Table S5. Oregon-Washington ESUs/stocks and data available for condition estimates. These data series met the criteria for inclusion in the condition analyses. Data types available are: HC – hatchery contribution to natural spawning; PGR – population growth rate; Age – spawning age structure. Period is the period of availability for the longest series for that population. | 267 |
| Figure S1. <i>California Chinook salmon abundance.</i> Quadplot summarizes information from multiple time series figures. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. | 273 |
| Figure S2. <i>California Chinook salmon abundance.</i> Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. Subpopulations listed include: California Coastal (CC), Central Valley (CV) fall, late-fall, and spring, Sacramento River (SR) winter runs, Klamath River fall run, and Sothern Oregon-Northern California (SONCC). | 275 |
| Figure S3. <i>California Chinook salmon condition.</i> Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. Subpopulations listed include: Central Valley (CV) fall run, Klamath River fall-run, and Sothern Oregon-Northern California (SONCC). | 276 |
| Figure S4. <i>California Chinook salmon condition.</i> Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. Subpopulations listed include: Central Valley (CV) fall run, Klamath River fall-run, and Sothern Oregon-Northern California (SONCC). | 278 |

| | |
|--|-----|
| Figure S5. <i>California coho salmon abundance.</i> Quadplot summarizes information from multiple time series figures. Subpopulations listed include: California coastal (CaCoastal) and Sothern Oregon-Northern California (SONCC)..... | 279 |
| Figure S6. <i>California Chinook salmon abundance.</i> Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. Subpopulations listed include: California coastal (CaCoastal) and Sothern Oregon-Northern California (SONCC). | 280 |
| Figure S7. <i>Oregon-Washington Chinook salmon abundance.</i> Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. Subpopulations listed include: lower Columbia River (LowerCR), Snake fall, Snake spring-summer (SnakeSpSu), upper Columbia River summer-fall (UpCRSuFa), and Willamette..... | 281 |
| Figure S8. <i>Oregon-Washington Chinook salmon abundance.</i> Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. Subpopulations listed include: lower Columbia River (LowerCR), Snake fall, Snake spring-summer (SnakeSpSu), upper Columbia River summer-fall (UpCRSuFa), and Willamette. | 283 |
| Figure S9. <i>Oregon-Washington Chinook salmon condition.</i> Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. Subpopulations listed include: lower Columbia River (LowerCR), Snake fall, Snake spring-summer (SnakeSpSu), upper Columbia River summer-fall (UpCRSuFa), and Willamette..... | 284 |
| Figure S10 a,b,c. <i>Oregon-Washington Chinook salmon condition.</i> Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. Subpopulations listed include: lower Columbia River (LowerCR), Snake fall, Snake spring-summer (SnakeSpSu), upper Columbia River summer-fall (UpCRSuFa), and Willamette. | 288 |
| Figure S11. <i>Oregon-Washington coho salmon abundance.</i> Quadplot summarizes information from multiple time series figures. Subpopulations listed include: lower Columbia River (LowerCR) and Oregon coastal (ORCoast)..... | 289 |
| Figure S13. <i>Oregon-Washington coho salmon condition.</i> Quadplot summarizes information from multiple time series figures. We evaluated percent natural spawners (PctNat) and population growth rate (PopGR). Subpopulations listed include: lower Columbia River (LowerCR) and Oregon coastal (ORCoast)..... | 291 |

Figure S14. *Oregon-Washington coho salmon condition.* Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. Subpopulations listed include: lower Columbia River (LowerCR) and Oregon coastal (ORCoast).....292

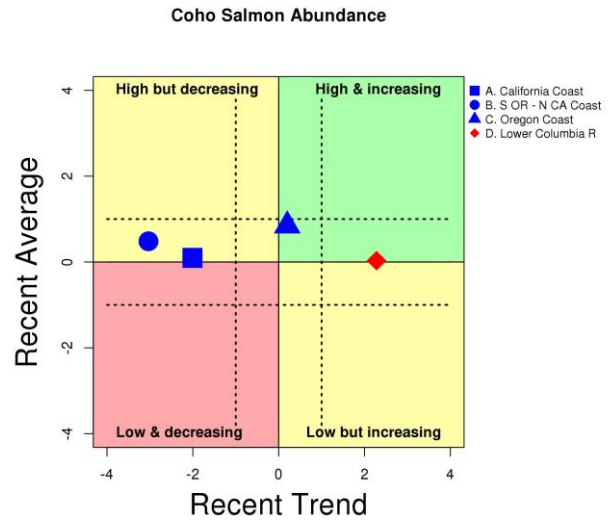
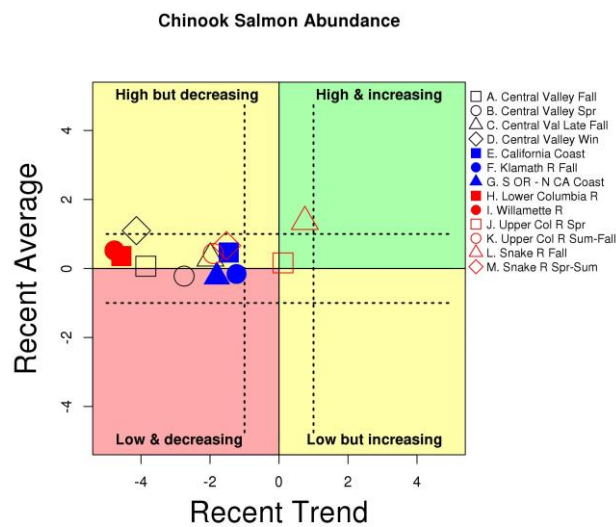
OVERVIEW

Generally, California Chinook and coho salmon populations are below their historical abundance levels and have continued to decline over the last decade. Most of the Chinook salmon populations from Columbia River Basin (including the Snake and Willamette Rivers) have experienced declines in abundance over the last ten years, with only Snake River Fall-run Chinook salmon populations exhibiting increased abundance. Abundances of coho salmon populations are relatively stable along the Oregon Coast and increasing in the lower Columbia River.

EXECUTIVE SUMMARY

Over the last ten years, there has been a significant decline in abundance of California populations of Chinook and coho salmon. While river Winter-run Chinook salmon had recent increases in abundance in 2002, 2003, and 2006, this population still remains only a fraction of its historical abundance even when compared with abundance levels just 30 years ago. Central Valley Fall and Late Fall-run abundance levels are projected to increase in 2012 following their collapse in 2007-2010, but the high proportion of hatchery-origin fish is a concern. In contrast, the growth rate and proportion of natural fall-run Chinook salmon in the Klamath River (part of the Southern Oregon and Northern California Coast Chinook salmon ESU) are relatively stable and the age structure is becoming more complex. With the exception of the Snake River Fall-run, Chinook salmon populations from the Columbia River Basin have experienced declines in abundance over the last ten years following high abundance levels in the early 2000s. Chinook salmon populations from the Snake River had increases in abundance for the last few years of available data, although the 10-year trends were negative for Snake River Spring/Summer-run Chinook salmon and unchanged for Snake River Fall-run Chinook salmon. With the exception of the Chinook salmon in the Willamette River, Chinook salmon populations in the Columbia River Basin exhibited increases in the proportion of hatchery-origin fish.

California populations of coho salmon have experienced declines in abundance over the past ten years. Coho salmon abundance from the lower Columbia River was variable but increasing over the past 10 years. The abundance of Oregon Coast coho salmon was variable with no significant trend over the past 10 years, although recent abundance levels were greater than that observed during the late-1990s.



Salmon abundance. Quadplot summarizes information from multiple time series of coho and Chinook salmon abundances. Prior to plotting, time series were normalized to place them on the same scale. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. Dotted lines show ± 1.0 s.d.

DETAILED REPORT

Pacific salmon (*Oncorhynchus* spp.) are iconic members of North Pacific rim ecosystems, historically ranging from Baja California to Korea (Groot and Margolis 1991). Historically, salmon supported extensive native estuarine and freshwater fisheries along the U.S. West Coast, followed more recently by large commercial marine and recreational marine and freshwater harvest. Because they are anadromous with extensive migrations, salmon connect marine and freshwater ecosystems.

The purpose of this chapter of the CCIEA is to examine trends in available indicators relevant to salmon along the California Current. This is the first step in finding valuable data series that can be used to describe various aspects of the CCE and its salmon community. The analysis is largely qualitative at this early stage of the CCIEA. It is important to recognize that we refer to “status” quite differently than that reported by Pacific Fisheries Management Council (PFMC) and in current Endangered Species Act status reports, therefore, any difference between our status statements and those should not be considered a conflict. We are not using similar models nor benchmarks as those traditionally used. Our purpose is to set the framework for evaluating the salmon community from an ecosystem perspective. This approach starts with a simple selection of indicators and evaluation of the trends. However, in following reports we will use these biological indicators in combination with indicators of environmental and anthropogenic pressures to evaluate potential risk to the salmon community and develop additional assessment tools useful for ecosystem based management. Indicators for various pressures can be found in other chapters of the full CCIEA (e.g., Anthropogenic Drivers and Pressures, Oceanographic and Climatic Drivers and Pressures).

Due to a variety of factors, CCLME salmon populations have experienced substantial declines in abundance (Nehlsen et al. 1991), to the extent that a number of stocks have been listed under the U.S. Endangered Species Act. This has resulted in extensive reviews of salmon population status and recovery efforts (Good et al. 2005, (Ford 2011, Williams et al. 2011). Rather than attempting to summarize the extensive data and literature that has been accumulated regarding West Coast salmon status, we focus on a few key stocks and indicators that relate to the overall condition of the CCLME.

The two most abundant salmon species in the CCLME are Chinook salmon (*O. tshawytscha*) and coho salmon (*O. kisutch*), and these two species have supported large fisheries (PFMC 2012a). For this reason, we focus on these two species, and selected stocks within the species that provide a range of geographic and life-history variation. There are a variety of ways to define 'stock' (for example, (Cushing 1981, Dizon et al. 1992) and Pacific salmon species have complex population structures. Here, we have chosen to use the Evolutionarily Significant Unit (ESU) defined by NOAA for use in Pacific salmon conservation management (Waples 1991). ESUs are defined on the basis of reproductive isolation and their contribution to the evolutionary legacy of the species as a whole, and are often composed of a number of geographically contiguous populations. They do not correspond exactly to the stock delineations that are used for harvest management, in most cases several stocks/populations make up an ESU.

INDICATOR SELECTION PROCESS

INDICATOR EVALUATION

Two underpinning elements of an IEA are data management infrastructure and the ecosystem-modeling infrastructure. The development of the ecosystem-modeling infrastructure requires the development of standard indicators, in our case, indicators useful for assessing the status and trends of Chinook salmon and coho salmon in the CCLME.

Rather than develop a unique suite of indicators for this report, we have relied on the extensive previous work in evaluating the status of salmon populations and ESUs on the Pacific coast (Allendorf et al. 1997, Wainwright and Kope 1999, McElhany et al. 2000, Good et al. 2005, Lindley et al. 2007). In particular, we selected indicators that were not inconsistent with these previous efforts and also the Viable Salmon Population (VSP) characteristics (McElhany et al. 2000) that are the foundation of current conservation and recovery planning efforts for Pacific salmonids; in addition, they are the bases for on-going evaluation of status updates of Pacific salmonid populations. McElhany et al. (2000) described four characteristics of populations that should be considered when assessing viability: abundance, productivity, diversity, and spatial structure. Since a high priority of the IEA effort is to develop frameworks that can expand to include new data and address multiple issues (e.g., protected species, fisheries, and ecosystem health), we felt it most appropriate to use indicators that are used in status reviews and ESA recovery planning documents (Table 1). From this list of potential indicators, we selected those with the most widespread data availability (to allow for comparisons across species and regions) and with most relevance to the state of the marine ecosystem. The following sections describe the indicators we considered as measures of stock abundance and condition.

Table S1. Key indicators for salmon, identified during the ESA listing and recovery planning processes. Indicators categories chosen for this analysis are in ***bold italic*** font.

| Indicator | Selection/Deselection Reasoning |
|---|--|
| Abundance | |
| <i>Spawning escapement</i> | Widely measured; key measure of reproductive population |
| Ocean abundance (recruitment) | Requires stock-specific harvest rate estimates; not widely available |
| Juvenile abundance | Not widely available, but key indicator of reproduction for some ESUs |
| Population Condition | |
| <i>Population growth rate (λ)</i> | Widely available, standard measure of population trend |
| Natural return ratio (NRR) | A measure of sustainability of the natural component of mixed hatchery-natural stocks: requires both age-structure and natural |

| | |
|--|---|
| Intrinsic rate of increase | Widely available, but depends on a specific formulation of density dependence. |
| <i>Proportion of natural spawners</i> | Widely available; Indicator of stock genetic integrity and effectiveness of natural production |
| Genetic diversity | |
| <i>Age structure diversity</i> | Available for most Chinook salmon stocks; a quantifiable measure of phenotypic diversity; indicator of harvest-related risk |
| Population spatial structure | Available for few stocks. |

POTENTIAL INDICATORS FOR ASSESSING ABUNDANCE (POPULATION SIZE)

Monitoring population size provides information of use both for protected species conservation and for harvest management. We considered three primary indicators of abundance, and chose to focus on one (spawning escapement) as the most widely available and relevant.

1. Spawning escapement—Estimates of spawning escapement are extremely important to salmon management as an indication of the actual reproductive population size. The number of reproducing adults is important in defining population viability, as a measure of both demographic and genetic risks. It is equally important to harvest management, which typically aims at meeting escapement goals such that the population remains viable (for ESA-listed populations) or near the biomass that produces maximum recruitment (for stocks covered by a fisheries management plan). Spawning escapement is the most widely available measure of abundance for West Coast salmon, although these data are often limited to the most commercially important stocks and often stock/population estimates only make up a portion of an ESU.

2. Recruitment—An estimate of the number of adults in the ocean that would be expected to return to spawn in freshwater if not harvested. This is typically estimated as the number of adults that return to spawn divided by the total fishery escapement rate (one minus the total harvest rate). Recruitment is the primary indicator of importance for harvest management, as it determines how much harvest can be tolerated while still meeting escapement goals. It is also the best indicator of overall system capacity for the stock. However, because estimation depends on stock-specific harvest rates, recruitment estimates are not always available.

3. Juvenile abundance—The abundance of juveniles in freshwater or early marine environments is a good measure of reproductive success for a stock. This is monitored for many West Coast salmon stocks, but data series are typically short, and often are made for only a small proportion of an ESU, so are difficult to interpret and compare on a regional basis.

POTENTIAL INDICATORS FOR ASSESSING POPULATION CONDITION

There are a number of potential metrics for assessing the condition of a managed salmon population. These fall into the broad categories of population growth/productivity, diversity, and spatial structure (McElhany et al. 2000). We considered the seven commonly-used metrics, and based on data availability and relevance, chose three of those metrics (population growth rate, hatchery contribution, and age-structure diversity) to reflect a range of assumptions about the effects of various stressors on the populations.

1. Population growth rate—Calculated as the proportional change in abundance between successive generations, population growth rate is an indication of the population's resilience. In addition, growth rate can act as a warning of critical abundance trends that can be used for determining future directions in management. Also, the viability of a population is dependent in part on maintaining life-history diversity in the population. Because of limited information on hatchery fish and natural return ratio (see below) this value includes hatchery origin fish.

2. Natural return ratio (NRR)—NRR is the ratio N/T , where N is naturally produced (i.e., natural-origin) spawning escapement and T is total (hatchery-origin plus natural-origin) spawning escapement in the previous generation. It is a measure of the sustainability of the natural component of mixed hatchery-natural stocks and is an important conservation-oriented measure of stock productivity. However, the calculation requires both age-structure and natural proportion data, and depends on assumptions regarding the relative fitness of hatchery-origin fish in natural environments. This makes it problematic as an ecosystem status indicator.

3. Intrinsic rate of increase—The intrinsic rate of increase is estimated from the statistical fitting of stock-recruit models and is a measure of the rate of population increase when abundance is very low. It is an important parameter in harvest management theory, used in the estimation of optimum yield from a fishery. However, computations require long-term data on both harvest rate and age-structure data, and an assumed theoretical form for the stock-recruit function; therefore it is not easy to use as a status indicator.

4. Hatchery contribution—Defined as the proportion of hatchery-origin fish in naturally-spawning populations. Hatchery fish are relatively homogeneous genetically in comparison to naturally produced populations, typically are not well-adapted to survival in natural habitats, and their presence may reduce the fitness of natural populations (Bisson et al. 2002, Lindley et al. 2007). Thus, this is an important measure of the health of natural populations. Data are available for most West Coast salmon ESUs.

5. Genetic diversity—Genetic diversity is an important conservation consideration for several reasons, particularly in providing adaptive capacity that makes populations resilient to changes in their environment (Waples et al. 2010). Genetic monitoring of salmon populations has become common, and is being used for genetic stock identification as part of harvest management (Beacham et al. 2008). However, there are as yet no time series of genetic data that would allow detection of trends in diversity nor is there an understanding of historical population-specific patterns of genetic diversity to provide context when evaluating contemporary patterns, so this is not a useful status indicator at this time.

6. Age structure diversity—A diverse age structure is important to improve population resilience. Larger, older Chinook salmon produce more and larger eggs (Healey and Heard 1984). Therefore, they produce a brood that may contribute proportionally more to the later spawning population than broods from younger, smaller fish. However, the diversity of ages including younger fish is important to accommodate variability in the environment. If mortality on any given cohort is great, there is benefit to having younger spawners. An individual that produces off spring that return at different adult ages (i.e., overlapping generations) may

increase the likelihood of contributing to future generations when environmental conditions are less than favorable one year to the next. This bet hedging is a critical aspect of Chinook salmon that allow it to naturally mitigate year-to-year environmental variability (Heath et al. 1999). Adult age structure is not an issue for coho salmon, which in our region spawn predominantly at age three (with the exception of a small proportion of younger male 'jacks'). While coho salmon in our region spawn predominantly at a single age, Chinook salmon typically spawn over an age range of 3 or 4 years, and exhibit differences in spawning age both among years and among populations. Data are available for most Chinook salmon populations of commercial importance or of ESA interest ESUs (e.g., Sacramento River Winter-run), although data are typically stock/population specific and might not be representative of an ESU.

7. Spatial structure—The spatial structure of a stock, both among- and within- subpopulations, is important to the long-term stability and adaptation of the stock/population/ESU. A number of methods have been proposed for indexing the structure of both spawning and juvenile salmon (McElhany et al. 2000, Wainwright et al. 2008, Peacock and Holt 2012). Unfortunately, there are not widespread data nor a consistent method used for evaluating spatial structure of West Coast salmon ESUs.

SELECTING APPROPRIATE STOCKS/POPULATIONS FOR EVALUATION OF ABUNDANCE AND CONDITION

Stock selection was based on economic and ecosystem importance, geographic and life-history diversity, and data availability. This resulted in selections consistent with current ESU delineations. Because of regional differences in the availability of data, we considered stocks and data series separately within two regions: California (including southern Oregon south of Cape Blanco) and Oregon-Washington coasts (Cape Blanco to the mouth of the Strait of Juan de Fuca). For each ESU, a variety of data series are available; each series has been used in management documents, status reports, and/or the scientific literature. Any data series that was less than 15 years long was removed; within each ESU, all data series were truncated to match the shortest series. Available data series meeting these criteria for given ESUs are listed in Tables 2-5. It should be noted that in many cases we used data that were not used for recent ESA status updates. Many of the time series available are at the stock or population scale and may not be representative of the whole ESU (the listing unit for ESA efforts) and therefore not appropriate for evaluating the status of an ESU. For our purposes we determined that development of the indicators and ecosystem models using stock/population scale measures was appropriate at this initial stage of development of IEA and we should be able to accommodate ESU representative data as rigorous monitoring programs are established.

For California ESUs (Tables 2 & 3), the data series were compiled from a variety of sources and are presented in Williams et al. (2011), PFMC (2012c), and Spence and Williams (2011). Because of the diversity of data types available, indicators for each stock were selected based on their availability, time series lengths, and scientific support. Data series that were used are highlighted in the tables.

For Oregon and Washington ESUs, data were obtained from the NWFSC's "Salmon Population Summary" database (<https://www.webapps.nwfsc.noaa.gov/apex/f?p=238:home:0>), with additional data for Oregon Coast coho salmon (Oregon Department of Fish and Wildlife, <http://oregonstate.edu/dept/ODFW/spawn/data.htm>), and from PFMC (2012c) for the Upper Columbia Summer/Fall-run Chinook Salmon.

When data were only available for a portion of an ESU (e.g., single stream or tributary, but not necessarily representative of the whole ESU) and no ESU-wide estimates were available, we used these data as a proxy for the ESU unless it was not recent enough or was incomplete (Table 2). If data restrictions or reporting required multiple series be used for a given indicator within a single ESU, we computed an ESU-

wide average (e.g., Table 2, Central Valley Spring-run). To do this, series were standardized and then averaged across populations within ESUs. These standard scores represent the index for abundance or conditions for that ESU. Data series that represented similar values (e.g., escapements) were weighted by absolute spawning abundance.

APPROPRIATE INDICATORS

We evaluated abundance using the metric of escapement of natural-origin spawners. Selection rationale for assessing only escapement and no other abundance metrics is listed in Table 1. The populations/ESUs that had sufficiently met the criteria for inclusion in the analyses are listed in Tables 2 and 4. When ESU-wide estimates were available and sufficient they were used. If data were only available at the sub-ESU level, escapement values from the component subpopulations were used. As well, we only used data beginning in 1985 so that, when possible, the longer time series could be compared equivalently between populations. Data series for multiple subpopulations were standardized by subtracting the series mean and dividing by the series standard deviation. If a consolidated index for the stock was needed we computed an annual weighted average of the standardized series, with weights proportional to the average abundance for each subpopulation.

To evaluate condition we restricted our analyses to examination of population growth rate, proportion of natural-origin spawners, and age-structure diversity. Selection rationale for assessing only these metrics of condition and no other condition metrics is listed in Table 1. The populations/ESUs that had sufficiently met the criteria for estimation of condition are listed in Tables 3 and 5.

Population growth rate for each subpopulation was estimated as the ratio of the 4-year running mean of spawning escapement in one year to the 4-year running mean for the previous year (Good et al. 2005). Proportion of natural-origin spawners was calculated for those populations where spawning abundance estimates are broken down into hatchery-origin and natural-origin components; the proportion was computed for a single population as the fraction N_N/N_T , where N_N is the number of naturally-origin spawners, and N_T is the total number of spawners. Population fractions were then averaged across the populations within the ESU, weighted by total spawner abundance. Age-structure diversity for Chinook salmon was computed as Shannon's diversity index of spawner age for each population within each year. The indices were then averaged across populations, weighted by total spawner abundance.

Table S2. California ESUs/Stocks and Data available for Abundance Estimates. Those series indicated by ***bold italics*** were used for analyses. Period is the period of availability for the longest series for that population.

| Population | Data Available: Escapement | Period |
|------------------------------|---|-----------------------------|
| Chinook Salmon | | |
| Central Valley Fall Run | <i>Escapement to system</i> | <i>1983-Present</i> |
| | Coleman | 1970-Present |
| | Feather | 1970-present |
| | Nimbus | 1970-present |
| | Mokelumne | 1970-present |
| | Merced | 1970-present |
| Central Valley Late Fall Run | <i>Escapement to system</i> | <i>1971-Present</i> |
| Central Valley Winter Run | <i>Escapement to system</i> | <i>1970-2008</i> |
| Central Valley Spring Run | Escapement to Sacramento R. | 1970-2008 |
| | <i>Escapement Antelope Cr.</i> | <i>~1982-Present</i> |
| | <i>Escapement Battle Cr.</i> | <i>1989-Present</i> |
| | <i>Escapement Big Chico Cr.</i> | <i>1970-Present</i> |
| | <i>Escapement Butte Cr.</i> | <i>1970-Present</i> |
| | <i>Escapement Clear Cr.</i> | <i>1992-Present</i> |
| | <i>Escapement Cottonwood Cr.</i> | <i>~1973-Present</i> |

| Population | Data Available: Escapement | Period |
|---------------------|---|----------------------|
| Klamath R. Fall Run | <i>Escapement Deer Cr.</i> | <i>1970-Present</i> |
| | <i>Escapement FRH</i> | <i>1970-Present</i> |
| | <i>Escapement Mill Cr.</i> | <i>1970-Present</i> |
| | <i>Escapement to system (Klamath+Trinity)</i> | <i>1978-Present</i> |
| | Shasta | 1930-present |
| SONCC Chinook Fall | Scott | 1978-present |
| | Salmon | 1978-present |
| | <i>Umpqua Escapement</i> | <i>1946 Present</i> |
| Cal Coastal Chinook | <i>Rogues Escapement N+H (Gold Ray Dam)</i> | |
| | Prairie Cr. AUC | 1998-Present |
| | <i>Freshwater Cr. Weir Count</i> | <i>1994-Present</i> |
| | <i>Tomki Cr. (Live/Dead Counts)</i> | <i>1979-Present</i> |
| | Mattole R. Redd Index | 1994-Present |
| | <i>Cannon Cr. (live/Dead Counts)</i> | <i>1981-Present</i> |
| | <i>Sprowl Cr. (Live/Dead Counts)</i> | <i>1974-Present</i> |
| | <i>Eel R. Dam Counts</i> | <i>~1950-Present</i> |
| | Russian R. Video Counts | 2000-Present |

| Population | Data Available: Escapement | Period |
|--|---|----------------------------|
| Coho salmon | | |
| Coho SONCC | Wild adult abundance | 2002-2004, 2006-2008 |
| | Adult density on spawning grounds | 2004-2008 |
| | Adult weir counts in Shasta | 2001-Present |
| | Spawning numbers Prairie Cr. | 1998-Present |
| | Spawning numbers | 2002-Present |
| <i>Abundance of wild coho in Rogue R.</i> | | |
| | Wild adult coho from Gold Ray Dam, OR | |
| | Spawning numbers Mattole R. | 1994-Present |
| | Freshwater Wier Count | 2002-2009 |
| | WB Mill Cr. count | 1998-present |
| | EB Mill Cr. Count | 1998-present |
| | Cannon Count (Mad R.) | 1981-present |
| | Illinois R. Counts | 2002-2008 varies |
| | California Coastal Coho | |
| | Scott Cr. Weir | 2002-present |
| | Redwood Cr. counts | 1997-present |
| | <i>Lagunitas/Olema coho reddcounts</i> | <i>1995-present</i> |

| Population | Data Available: Escapement | Period |
|------------|-------------------------------------|----------------------------|
| | Caspar Cr. Redd Counts | 1999-present |
| | Little Rvier Redd Counts | 1999-present |
| | Noyo R. Redd countes | 2000-present |
| | Noyo redd Upstream | 1999-present |
| | SF Noyo Weir Count | 1998-present |
| | Pudding Cr. Counts | 2000-present |
| | <i>Sprowl Cr. Escapement</i> | <i>1978-present</i> |

Table S3. Data series that met the criteria for inclusion in the condition analyses of California ESUs. Period is the period of availability for the longest series for that population.

| Population | Series on Condition | Period |
|--------------------------------|--------------------------|----------------|
| Chinook Salmon | | |
| CV Fall Sacramento R. Fall Run | Hatchery contribution | 1983 - Present |
| | Population Growth Rate | 1983-present |
| Klamath R. Fall Run | Klam Age diversity (S-W) | 1981-present |
| | Hatchery contribution | 1978 - Present |
| | Population Growth Rate | 1981-present |
| SONCC Chinook Fall | Rogue Age Diversity | 1980-present |
| | Hatchery Contribution | 1972-present |

Table S4. Oregon-Washington ESUs/stocks and data available for abundance estimates. Each of these series met the criteria for inclusion in the analyses and was used.

| Stock/ESU | Data Available: Escapement | Period |
|-----------------------|----------------------------|-----------|
| Chinook Salmon | | |
| Lower Columbia R. ESU | Clatskanie R. Fall | 1974-2006 |
| | Coweeman R. Fall | 1977-2009 |
| | Elochoman R. Fall | 1975-2009 |
| | Grays R. Fall | 1964-2009 |
| | Kalama R. Fall | 1964-2009 |
| | Kalama R. Spring | 1980-2008 |
| | Lewis R. | 1964-2009 |
| | Lewis R. Fall | 1977-2009 |
| | Lower Cowlitz R. Fall | 1977-2009 |
| | Mill Cr. Fall | 1980-2009 |
| | North Fork Lewis R. Spring | 1980-2008 |
| | Sandy R. Fall (Bright) | 1981-2006 |
| | Sandy R. Spring | 1981-2008 |
| | Toutle R. Fall | 1964-2009 |
| | Upper Cowlitz R. Spring | 1980-2008 |

| Stock/ESU | Data Available: Escapement | Period |
|--------------------------------|--------------------------------|-----------|
| | Upper Gorge Tributaries Fall | 1964-2008 |
| | Washougal R. Fall | 1977-2009 |
| | White Salmon R. Fall | 1976-2009 |
| Snake R. Fall-run ESU | Snake R. Lower Mainstem Fall | 1975-2008 |
| Snake R. Spring/Summer-run ESU | Bear Valley Cr. | 1960-2008 |
| | Big Cr. | 1957-2008 |
| | Camas Cr. | 1963-2006 |
| | Catherine Cr. Spring | 1955-2009 |
| | Chamberlain Cr. | 1985-2008 |
| | East Fork Salmon R. | 1960-2008 |
| | East Fork South Fork Salmon R. | 1958-2008 |
| | Grande Ronde R. Upper Mainstem | 1955-2009 |
| | Imnaha R. Mainstem | 1949-2009 |
| | Lemhi R. | 1957-2008 |
| | Loon Cr. | 1957-2008 |
| | Lostine R. Spring | 1959-2009 |
| | Marsh Cr. | 1957-2008 |

| Stock/ESU | Data Available: Escapement | Period |
|------------------------------------|------------------------------------|-----------|
| | Minam R. | 1954-2009 |
| | Pahsimeroi R. | 1986-2008 |
| | Salmon R. Lower Mainstem | 1957-2008 |
| | Salmon R. Upper Mainstem | 1962-2008 |
| | Secesh R. | 1957-2008 |
| | South Fork Salmon R. Mainstem | 1958-2008 |
| | Sulphur Cr. | 1957-2008 |
| | Tucannon R. | 1979-2009 |
| | Valley Cr. | 1957-2008 |
| | Wenaha R. | 1964-2009 |
| Upper Columbia R. Spring-run ESU | Yankee Fork | 1961-2008 |
| | Entiat R. | 1960-2008 |
| | Methow R. | 1960-2008 |
| | Wenatchee R. | 1960-2008 |
| Upper Columbia Summer-Fall-run ESU | Escapement estimated at Bonneville | 1996-2010 |
| Upper Willamette R. ESU | Clackamas R. Spring | 1974-2008 |
| | McKenzie R. Spring | 1970-2005 |

| Stock/ESU | Data Available: Escapement | Period |
|-----------------------|----------------------------|-----------|
| Coho Salmon | | |
| Lower Columbia R. ESU | Clackamas R. | 1974-2010 |
| | Sandy R. | 1974-2010 |
| Oregon Coast ESU | Alsea R. | 1990-2010 |
| | Beaver Cr. | 1990-2010 |
| | Coos R. | 1990-2010 |
| | Coquille R. | 1990-2010 |
| | Floras/New R. | 1990-2010 |
| | Lower Umpqua R. | 1990-2010 |
| | Middle Umpqua R. | 1990-2010 |
| | Necanicum R. | 1990-2010 |
| | Nehalem R. | 1990-2010 |
| | Nestucca R. | 1990-2010 |
| | North Umpqua R. | 1990-2010 |
| | Salmon R. | 1990-2010 |
| | Siletz R. | 1990-2010 |
| | Siltcoos Lk. | 1990-2010 |

| Stock/ESU | Data Available: Escapement | Period |
|-----------|----------------------------|-----------|
| | Siuslaw R. | 1990-2010 |
| | Sixes R. | 1990-2010 |
| | South Umpqua R. | 1990-2010 |
| | Tahkenitch Lk. | 1990-2010 |
| | Tenmile Lk. | 1990-2010 |
| | Tillamook Bay | 1990-2010 |
| | Yaquina R. | 1990-2010 |

Table S5. Oregon-Washington ESUs/stocks and data available for condition estimates. These data series met the criteria for inclusion in the condition analyses. Data types available are: HC – hatchery contribution to natural spawning; PGR – population growth rate; Age – spawning age structure. Period is the period of availability for the longest series for that population.

| Stock/ESU | Population | Data Types | Period |
|-----------------------|----------------------------|--------------|-----------|
| Chinook Salmon | | | |
| Lower Columbia R. ESU | Clatskanie R. Fall | HC, PGR, Age | 1974-2006 |
| | Coweeman R. Fall | HC, PGR | 1980-2009 |
| | Elochoman R. Fall | HC, PGR | 1975-2009 |
| | Grays R. Fall | HC, PGR | 1964-2009 |
| | Kalama R. Fall | HC, PGR | 1964-2009 |
| | Kalama R. Spring | PGR | 1980-2008 |
| | Lewis R. | HC, PGR | 1978-2009 |
| | Lewis R. Fall | PGR | 1964-2009 |
| | Lower Cowlitz R. Fall | HC, PGR | 1977-2009 |
| | Mill Cr. Fall | HC, PGR | 1980-2009 |
| | North Fork Lewis R. Spring | PGR | 1980-2008 |
| | Sandy R. Fall (Bright) | HC, PGR, Age | 1981-2006 |
| | Sandy R. Spring | HC, PGR, Age | 1981-2008 |
| | Toutle R. Fall | PGR | 1964-2009 |
| | Upper Cowlitz R. Spring | PGR | 1980-2008 |

| Stock/ESU | Population | Data Types | Period |
|--------------------------------|------------------------------|--------------|-----------|
| | Upper Gorge Tributaries Fall | HC, PGR | 1964-2008 |
| | Washougal R. Fall | HC, PGR | 1977-2009 |
| | White Salmon R. Fall | HC, PGR, Age | 1976-2009 |
| Snake R. Fall-run ESU | Snake R. Lower Main. Fall | HC, PGR, Age | 1975-2008 |
| Snake R. Spring/Summer-run ESU | Bear Valley Cr. | HC, PGR, Age | 1960-2008 |
| | Big Cr. | HC, PGR, Age | 1957-2008 |
| | Camas Cr. | HC, PGR, Age | 1963-2006 |
| | Catherine Cr. Spring | HC, PGR, Age | 1955-2009 |
| | Chamberlain Cr. | HC, PGR, Age | 1985-2008 |
| | East Fork Salmon R. | HC, PGR, Age | 1960-2008 |
| | E. Fork S. Fork Salmon R. | HC, PGR, Age | 1958-2008 |
| | Grande Ronde R. Upper Main. | HC, PGR, Age | 1955-2009 |
| | Imnaha R. Mainstem | HC, PGR, Age | 1949-2009 |
| | Lemhi R. | HC, PGR, Age | 1957-2008 |
| | Loon Cr. | HC, PGR, Age | 1957-2008 |
| | Lostine R. Spring | HC, PGR, Age | 1959-2009 |
| | Marsh Cr. | HC, PGR, Age | 1957-2008 |

| Stock/ESU | Population | Data Types | Period |
|------------------------------------|------------------------------------|--------------|-----------|
| Upper Columbia R. Spring-run ESU | Minam R. | HC, PGR, Age | 1954-2009 |
| | Pahsimeroi R. | HC, PGR, Age | 1986-2008 |
| | Salmon R. Lower Mainstem | HC, PGR, Age | 1957-2008 |
| | Salmon R. Upper Mainstem | HC, PGR, Age | 1962-2008 |
| | Secesh R. | HC, PGR, Age | 1957-2008 |
| | South Fork Salmon R. Mainstem | HC, PGR, Age | 1958-2008 |
| | Sulphur Cr. | HC, PGR, Age | 1957-2008 |
| | Tucannon R. | HC, PGR, Age | 1979-2009 |
| | Valley Cr. | HC, PGR, Age | 1957-2008 |
| | Wenaha R. | HC, PGR, Age | 1964-2009 |
| | Yankee Fork | HC, PGR, Age | 1961-2008 |
| Upper Columbia R. Spring-run ESU | Entiat R. | HC, PGR, Age | 1960-2008 |
| | Methow R. | HC, PGR, Age | 1960-2008 |
| | Wenatchee R. | HC, PGR, Age | 1960-2008 |
| Upper Columbia Summer-Fall-run ESU | Escapement estimated at Bonneville | HC, PGR, Age | 1996-2010 |
| Upper Willamette R. ESU | Clackamas R. Spring | HC, PGR, Age | 1974-2008 |
| | McKenzie R. Spring | HC, PGR, Age | 1970-2005 |

| Stock/ESU | Population | Data Types | Period |
|-----------------------|------------------|------------|-----------|
| Coho Salmon | | | |
| Lower Columbia R. ESU | Clackamas R. | HC, PGR | 1974-2010 |
| | Sandy R. | HC, PGR | 1974-2010 |
| Oregon Coast ESU | Alsea R. | HC, PGR | 1990-2010 |
| | Beaver Cr. | HC, PGR | 1990-2010 |
| | Coos R. | HC, PGR | 1990-2010 |
| | Coquille R. | HC, PGR | 1990-2010 |
| | Floras/New R. | HC, PGR | 1990-2010 |
| | Lower Umpqua R. | HC, PGR | 1990-2010 |
| | Middle Umpqua R. | HC, PGR | 1990-2010 |
| | Necanicum R. | HC, PGR | 1990-2010 |
| | Nehalem R. | HC, PGR | 1990-2010 |
| | Nestucca R. | HC, PGR | 1990-2010 |
| | North Umpqua R. | HC, PGR | 1990-2010 |
| | Salmon R. | HC, PGR | 1990-2010 |
| | Siletz R. | HC, PGR | 1990-2010 |
| | Siltcoos Lk. | HC, PGR | 1990-2010 |

| Stock/ESU | Population | Data Types | Period |
|-----------|-----------------|------------|-----------|
| | Siuslaw R. | HC, PGR | 1990-2010 |
| | Sixes R. | HC, PGR | 1990-2010 |
| | South Umpqua R. | HC, PGR | 1990-2010 |
| | Tahkenitch Lk. | HC, PGR | 1990-2010 |
| | Tenmile Lk. | HC, PGR | 1990-2010 |
| | Tillamook Bay | HC, PGR | 1990-2010 |
| | Yaquina R. | HC, PGR | 1990-2010 |

STATUS AND TRENDS

MAJOR FINDINGS

Central Valley Fall and Late Fall-run Chinook salmon and Central Valley Spring-run Chinook salmon escapement has demonstrated declines over the last ten years. Central Valley Fall and Late Fall-run Chinook salmon were near their long-term average of abundance over the past ten years whereas Central Valley Spring-run Chinook salmon were below their long-term average of abundance (although Spring-run data are only available from 1995 to present). Sacramento River Winter-run Chinook salmon had recent increases in abundance in 2002, 2003, and 2006 but still remain only a fraction of their historical abundances of even just 30 years ago. Central Valley Fall and Late Fall-run Chinook salmon population abundances have increased following the collapse of 2007-2010 and 2012 estimates of adult abundance are similar to the long-term average, but the proportion of hatchery-origin fish is a concern. In contrast, Chinook salmon in the Klamath River (part of the Southern Oregon and Northern California Coast Chinook salmon ESU) natural production and growth rate are relatively stable as measured by the indices used and the age structure is becoming more complex. With the exception of the Snake River Fall-run, Chinook salmon populations from the Columbia River Basin have experienced declines in abundance over the last ten years. Chinook salmon populations from the Snake River had increases in abundance for the last few years of available data, although the 10-year trends were negative for Snake River Spring/Summer-run Chinook salmon and unchanged for Snake River Fall-run Chinook salmon. With the exception of the Chinook salmon in the Willamette River, Chinook salmon populations in the Columbia River Basin exhibited increases in the proportion of hatchery-origin fish.

California populations of coho salmon have had declines in abundance over the past ten years with the populations in the California portion of the Southern Oregon/Northern California Coast (SONCC) Coho Salmon ESU having significant declines in the past five years. Coho salmon abundance from lower Columbia River was variable but increasing over the past 10 years whereas Oregon Coast coho salmon abundance was variable with no significant trend over the past 10 years although recent abundances were greater than that observed during the late-1990's.

SUMMARY AND STATUS OF TRENDS

Both short- and long-term trends are reported in this summary. An indicator is considered to have changed over the short-term if the trend over the last 10 years (2002-2011) the series showed a significant increasing or decreasing slope. An indicator is considered to be above or below long-term norms if the mean of the last 10 years of the time series differs from the mean of the full time series by more than 1.0 s.d. of the full time series. A major motivation of presenting long- and short-term trends is to distinguish between stocks/populations that were once very large and suffered historical declines but have stabilized at lower abundances from populations with ongoing declines. This was a particular issue for populations with very long time series of abundance (e.g., certain Columbia River Chinook salmon populations). Such very long time series aren't available for most California populations. In addition, one should be cautious using pre-1980 data from Columbia River stocks/populations (and perhaps other locations) since data collection and methods have significantly improved since the early 1980s. Therefore it should be noted that when references are made to "long-term" abundances, conditions, etc. that this is in the context of the time period going back to 1985. Uncertainty about data prior to 1985 led us to limit data used to this time period. In addition, information on historical values of abundance indicate that for many if not most of these populations current values are now at levels far below historical values – so caution should be used when considering the term "long-term".

CALIFORNIA CHINOOK SALMON: ABUNDANCE

Generally all California stocks, minus Sacramento River Winter-run Chinook salmon were within 1 s.d. of their long term average however, during the last ten years there has been a significant decline in abundance of all the California populations examined (Figs. S1 & S2). Largely, though, this relates to a reduction from series highs during 2002 and a return to, generally, average values (Sacramento River Winter-run Chinook salmon time series, which was above average, stopped in 2008).

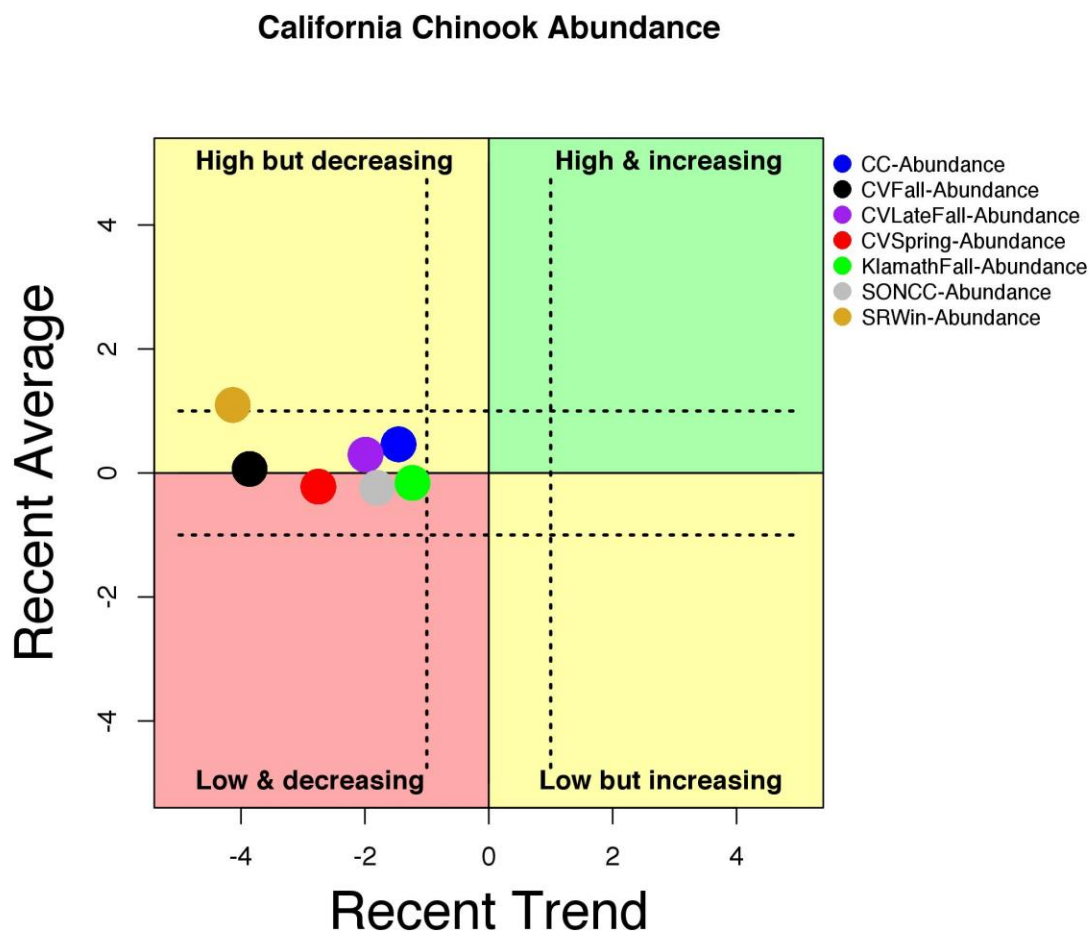
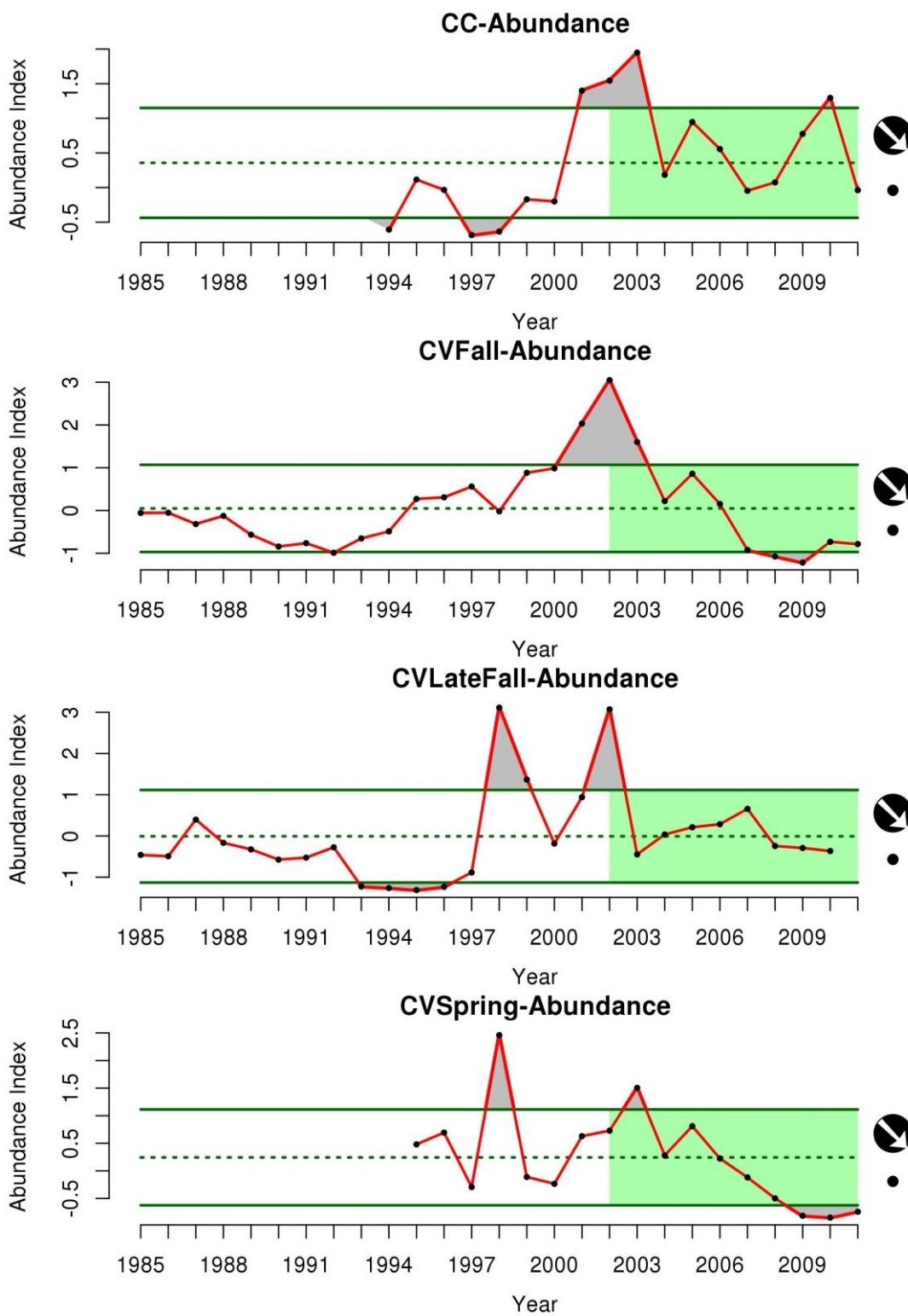


Figure S1. *California Chinook salmon abundance.* Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. Dotted lines show ± 1.0 s.d. Subpopulations listed include: California Coastal (CC), Central Valley (CV) fall, late-fall, and spring, Sacramento River (SR) winter runs, Klamath River fall run, and Sothern Oregon-Northern California (SONCC).



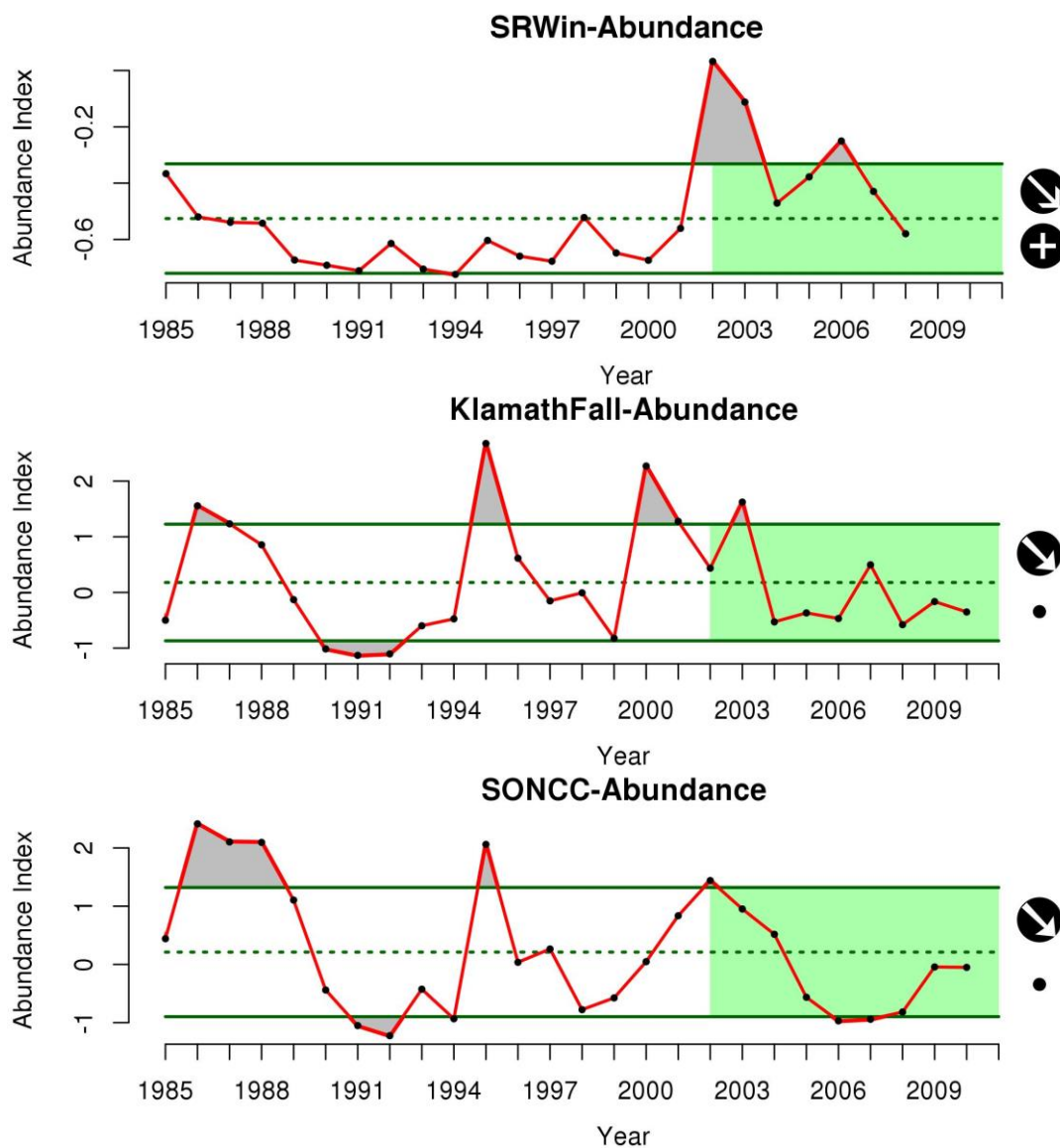


Figure S2. *California Chinook salmon abundance.* Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend was significant over the last 10-years. The lower symbol indicates whether the mean during the last 10 years was greater or less than or within one s.d. of the long-term mean. Subpopulations listed include: California Coastal (CC), Central Valley (CV) fall, late-fall, and spring, Sacramento River (SR) winter runs, Klamath River fall run, and Sothorn Oregon-Northern California (SONCC).

CALIFORNIA CHINOOK SALMON: CONDITION

While there is a recent (last two years) increase in the population growth rate (recovery rate) of the Central Valley Fall and Late Fall-run Chinook salmon, over the last 10 years there has been a decline. In addition, the proportion of the stock that is natural is below the long term average and decreasing. Chinook salmon in the Klamath River (below the confluence of the Klamath and Trinity rivers, part of the SONCC ESU) have, in recent years, had an increase in the diversity of ages and the proportion of wild fish spawning was increasing (Fig. S3, S4).

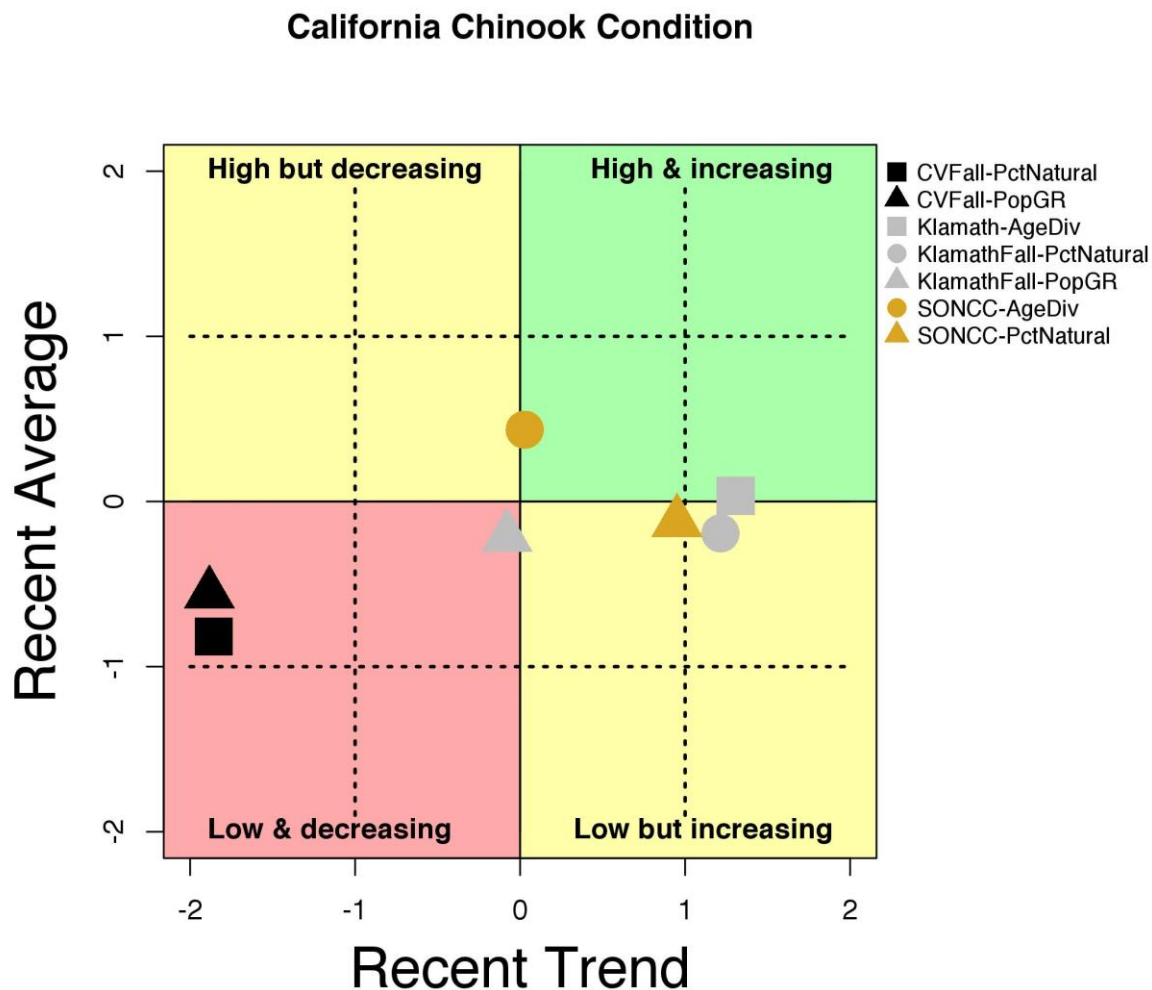
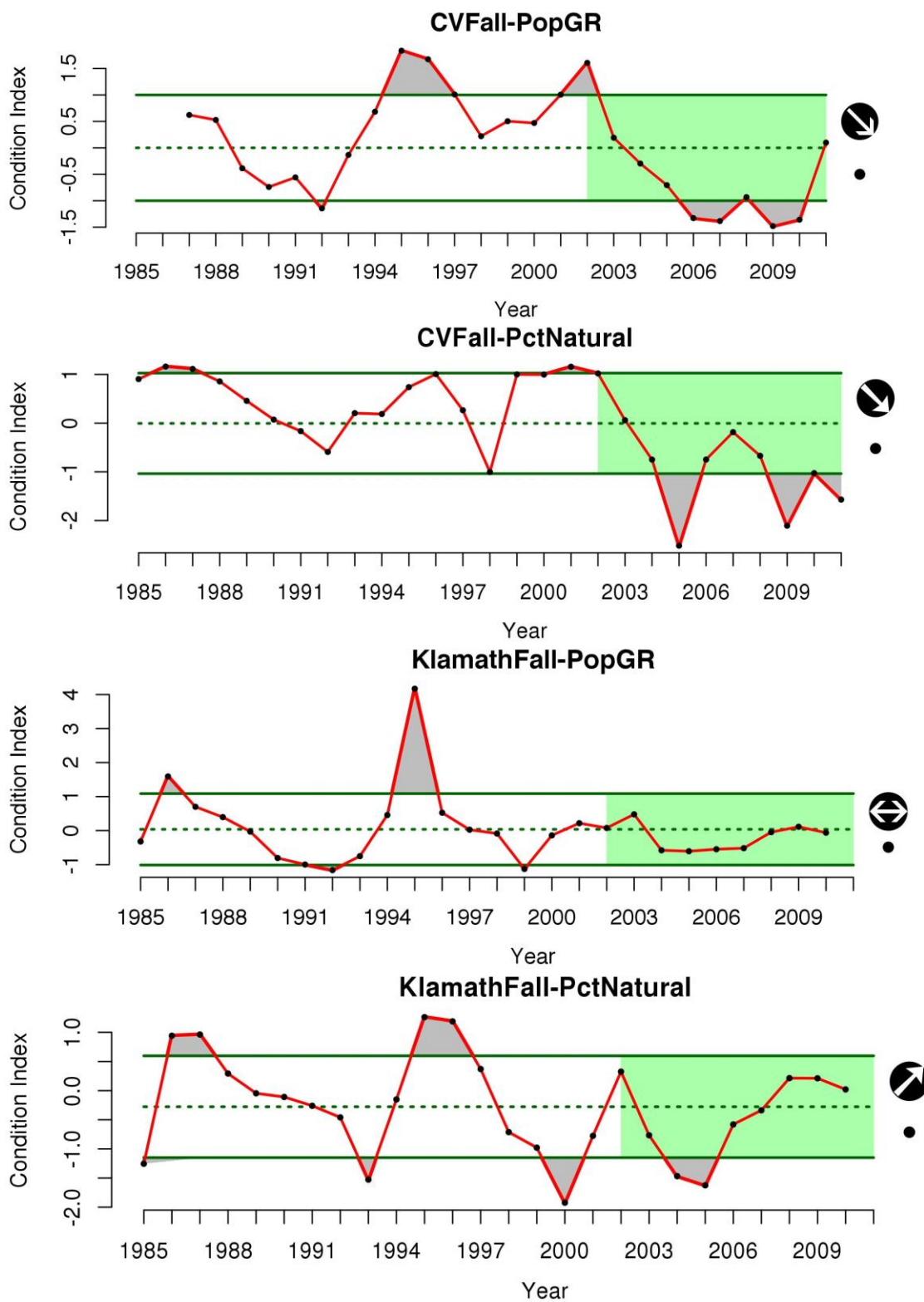


Figure S3. *California Chinook salmon condition.* Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. Dotted lines show ± 1.0 s.d. When possible we evaluated percent natural spawners (PctNatural), age-structure diversity (AgeDiv), and population growth rate (PopGR). Subpopulations listed include: Central Valley (CV) fall run, Klamath River fall-run, and Sothern Oregon-Northern California (SONCC).



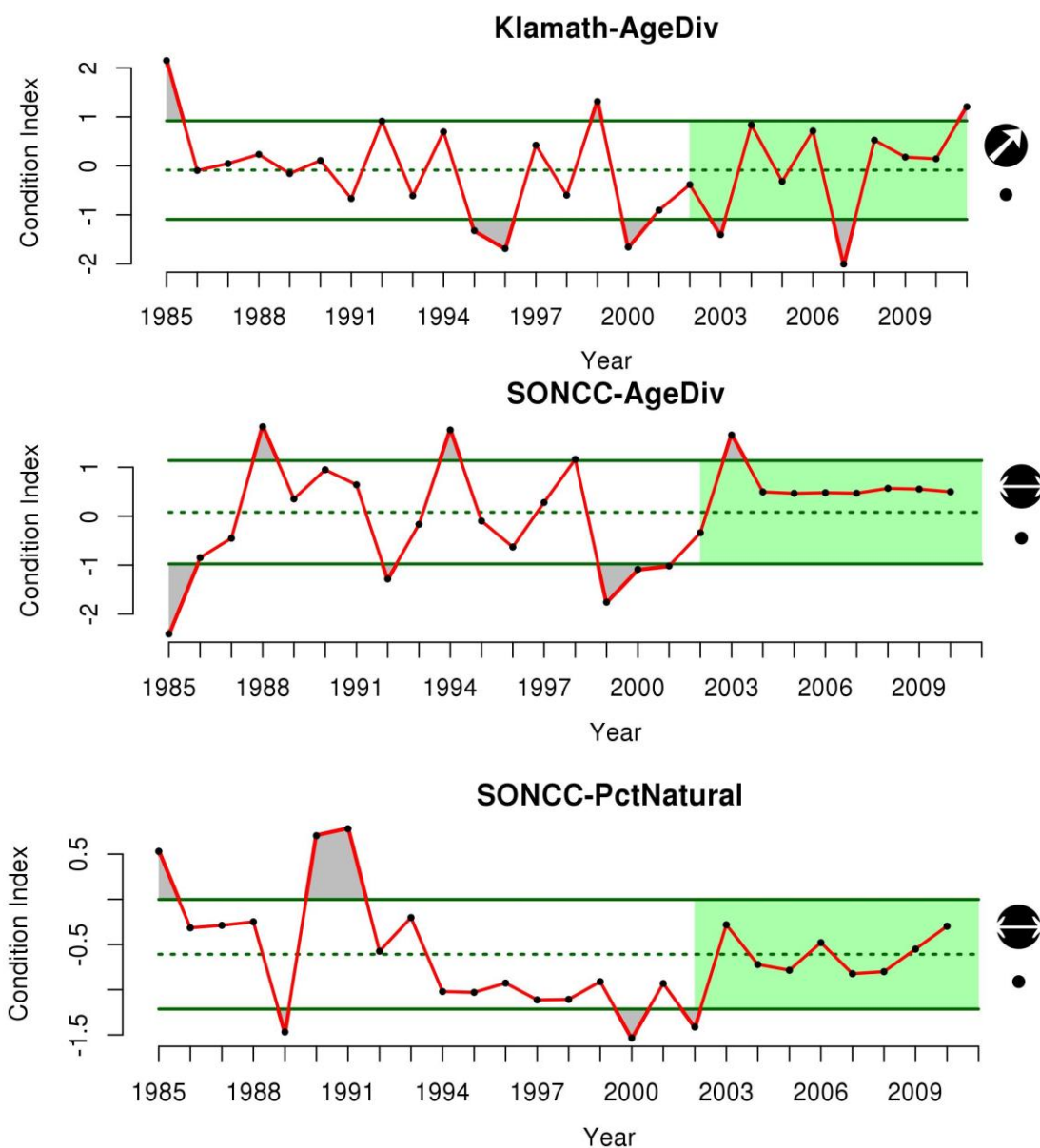


Figure S4. *California Chinook salmon condition.* Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend was significant over the last 10-years. The lower symbol indicates whether the mean during the last 10 years was greater or less than or within one s.d. of the long-term mean. When possible we evaluated percent natural spawners (PctNatural), age-structure diversity (AgeDiv), and population growth rate (PopGR). Subpopulations listed include: Central Valley (CV) fall run, Klamath River fall-run, and Sothern Oregon-Northern California (SONCC).

CALIFORNIA COHO SALMON: ABUNDANCE

Central California Coast coho salmon abundance has not been within 1 s.d. of the long- and short-term average for only two of the 17 years of data available. From those two high abundance years of 2003 and 2004 the abundance declined over the past ten years (Fig. S6). Abundance of California populations of Southern Oregon/Northern California Coast coho salmon have declined over the past 10 years from high abundance during 2004 (Figs. S5, S6).

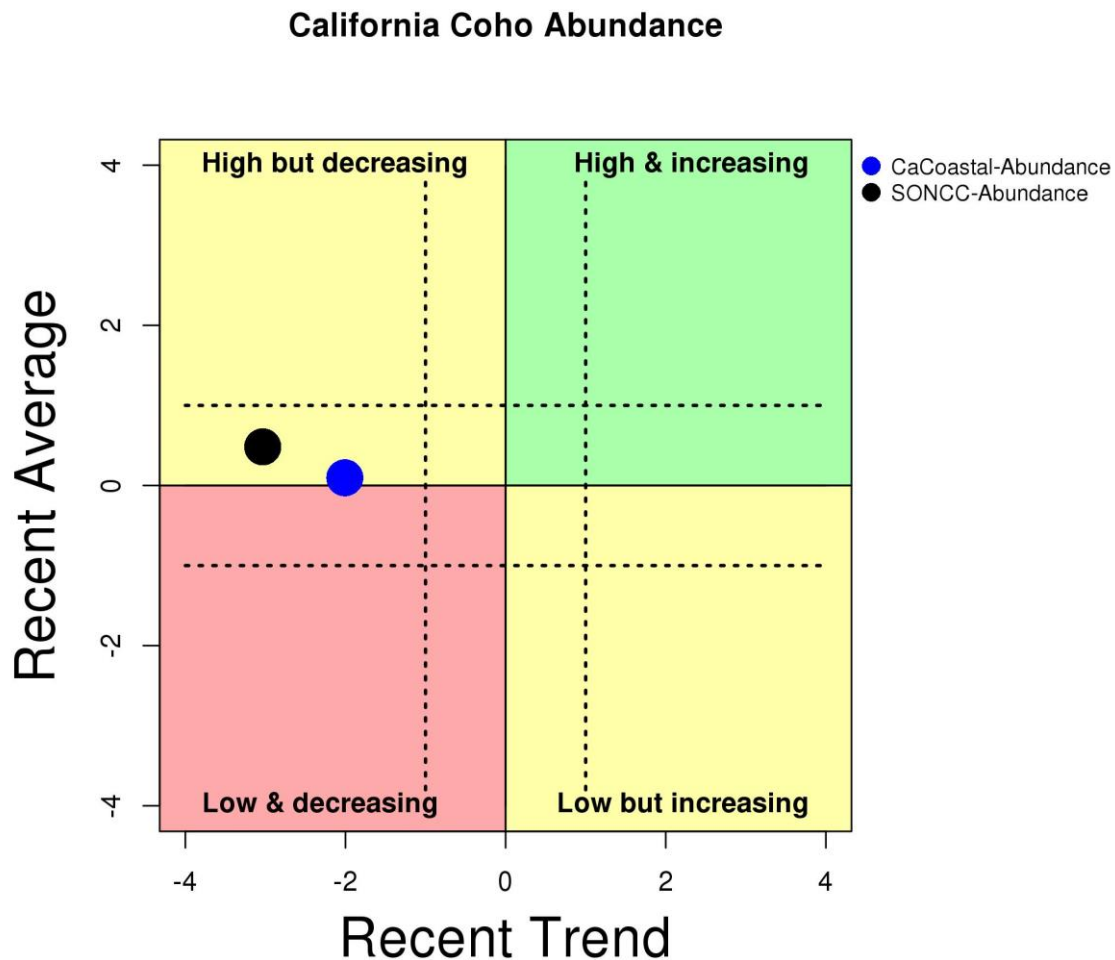


Figure S5. *California coho salmon abundance.* Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. Dotted lines show ± 1.0 s.d. Subpopulations listed include: California coastal (CaCoastal) and Sothern Oregon-Northern California (SONCC).

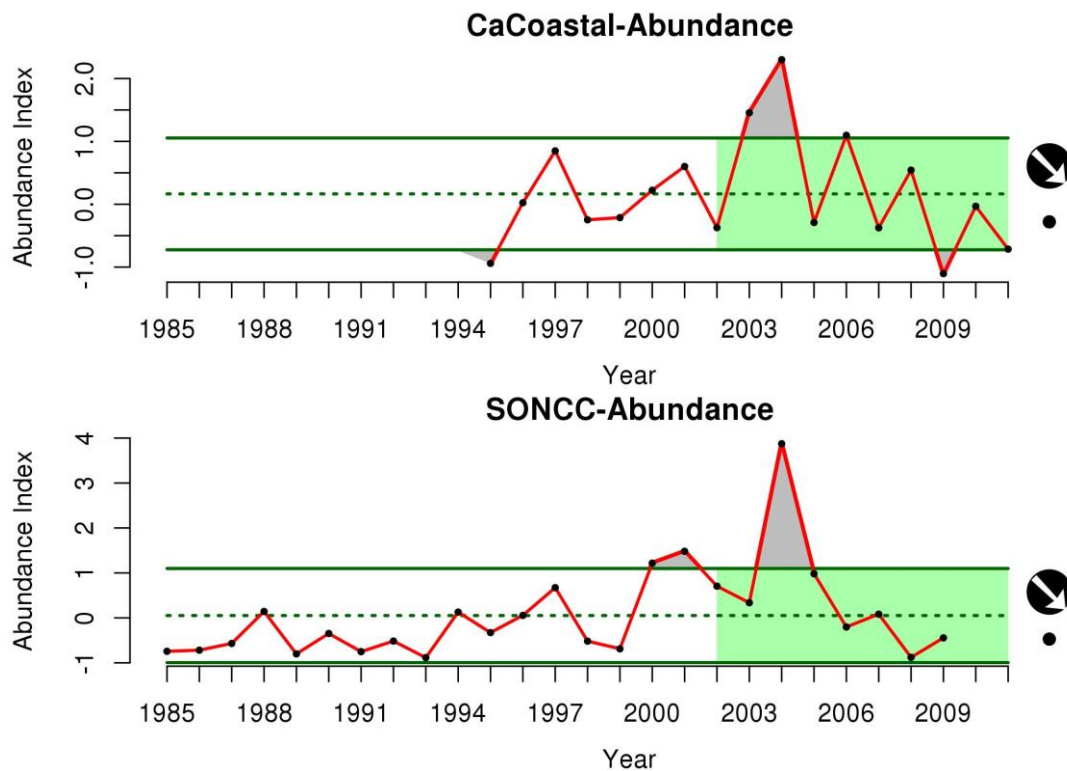


Figure S6. *California Chinook salmon abundance.* Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend was significant over the last 10-years. The lower symbol indicates whether the mean during the last 10 years was greater or less than or within one s.d. of the long-term mean. Subpopulations listed include: California coastal (CaCoastal) and Southern Oregon-Northern California (SONCC).

CALIFORNIA COHO SALMON: CONDITION

No data available.

OREGON-WASHINGTON CHINOOK SALMON: ABUNDANCE

Over the long-term, Oregon and Washington Chinook salmon abundances have exhibited substantial variation (Fig. S7) with all but Snake River Fall-run Chinook salmon and Upper Columbia River Spring-run Chinook salmon declining over the past 10 years (Fig. S8). While there has not been a significant trend the Snake River Fall-run Chinook salmon has been above its long term average in the last ten years.

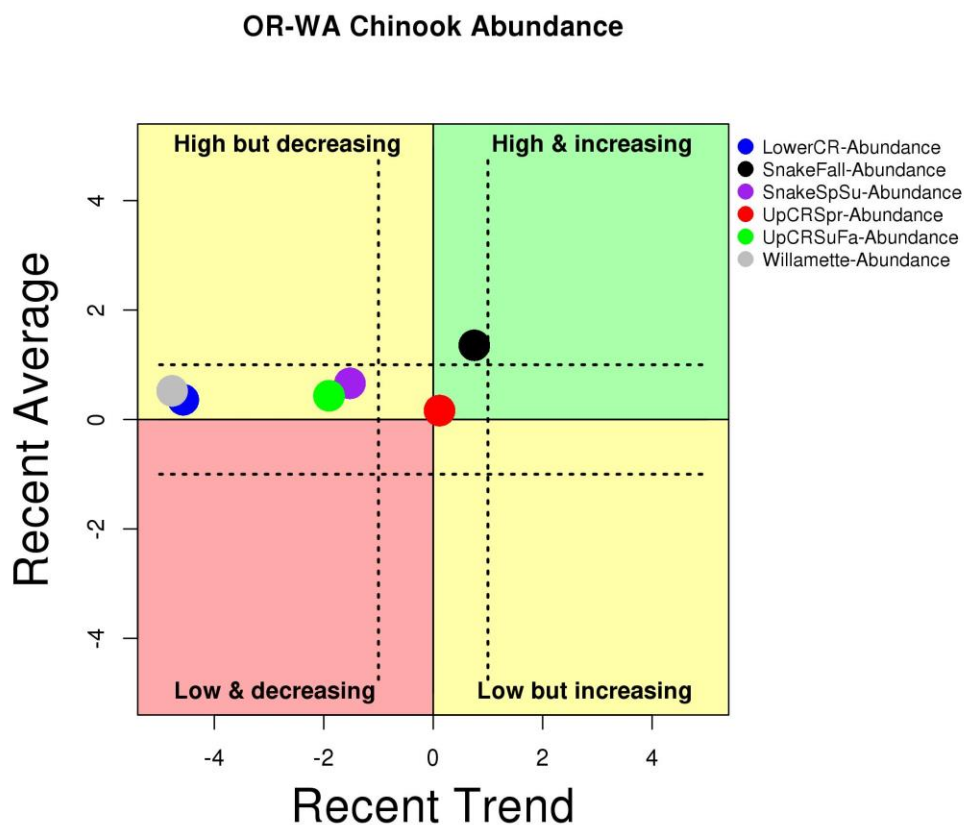
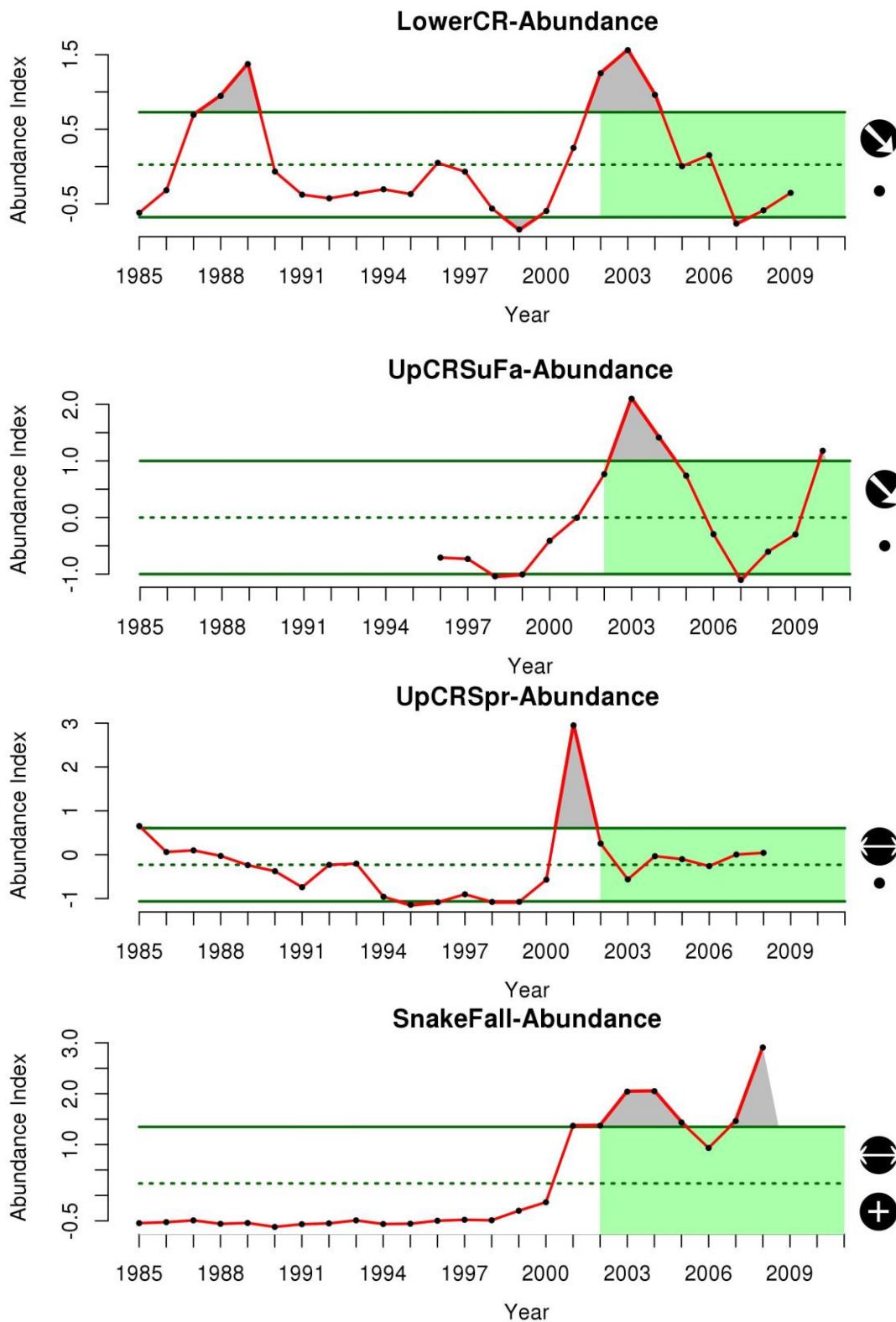


Figure S7. Oregon-Washington Chinook salmon abundance. Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. Dotted lines show ± 1.0 s.d. Subpopulations listed include: lower Columbia River (LowerCR), Snake fall, Snake spring-summer (SnakeSpSu), upper Columbia River summer-fall (UpCRSuFa), and Willamette.



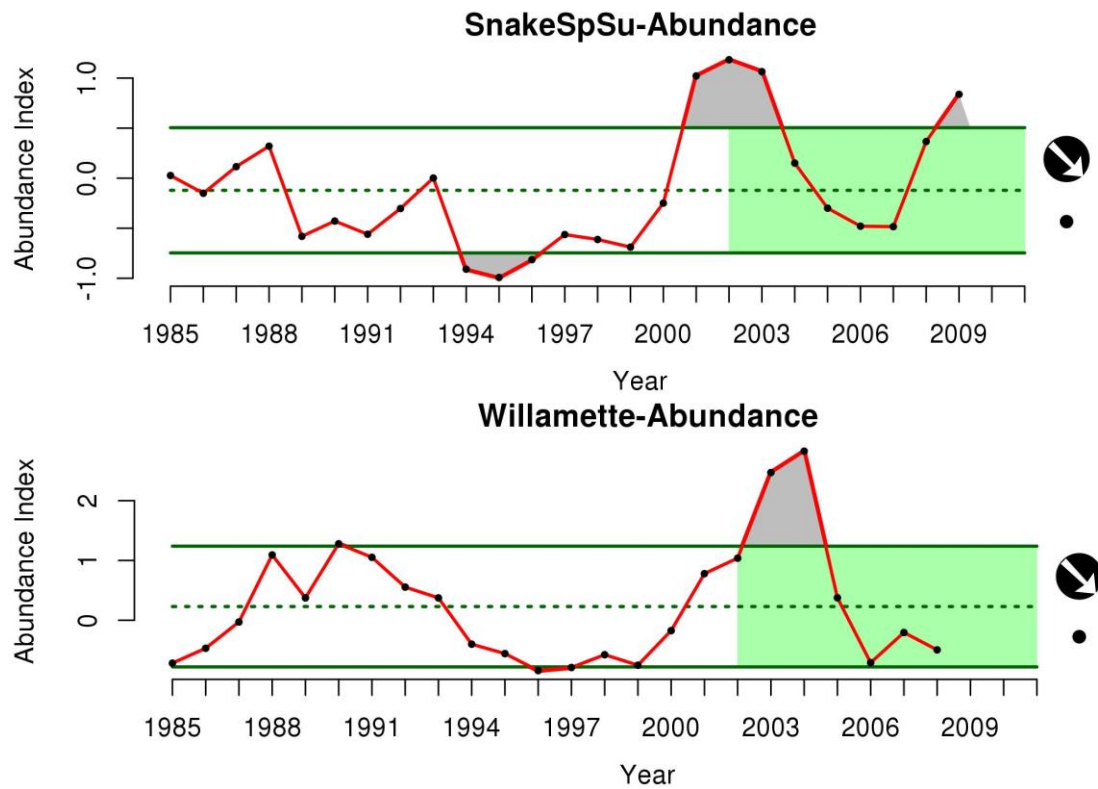


Figure S8. Oregon-Washington Chinook salmon abundance. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend was significant over the last 10-years. The lower symbol indicates whether the mean during the last 10 years was greater or less than or within one s.d. of the long-term mean. Subpopulations listed include: lower Columbia River (LowerCR), Snake fall, Snake spring-summer (SnakeSpSu), upper Columbia River summer-fall (UpCRSuFa), and Willamette.

OREGON-WASHINGTON CHINOOK SALMON: CONDITION

There are few obvious patterns in the condition indicators for Oregon and Washington Chinook salmon, with a wide mix of positive and negative trends at both time scales (Fig. S9, S10). One apparent pattern is the concentration of points in the “low and decreasing” quadrant for the proportion of natural spawners (“PctNat”), suggesting an increasing overall influence of hatchery production for these stocks. This is likely due to increases in Columbia Basin hatchery production during the 1970s as mitigation for dam construction (long-term trends) and starting in the late 1990s as supplementation for stock rebuilding (short-term trends).

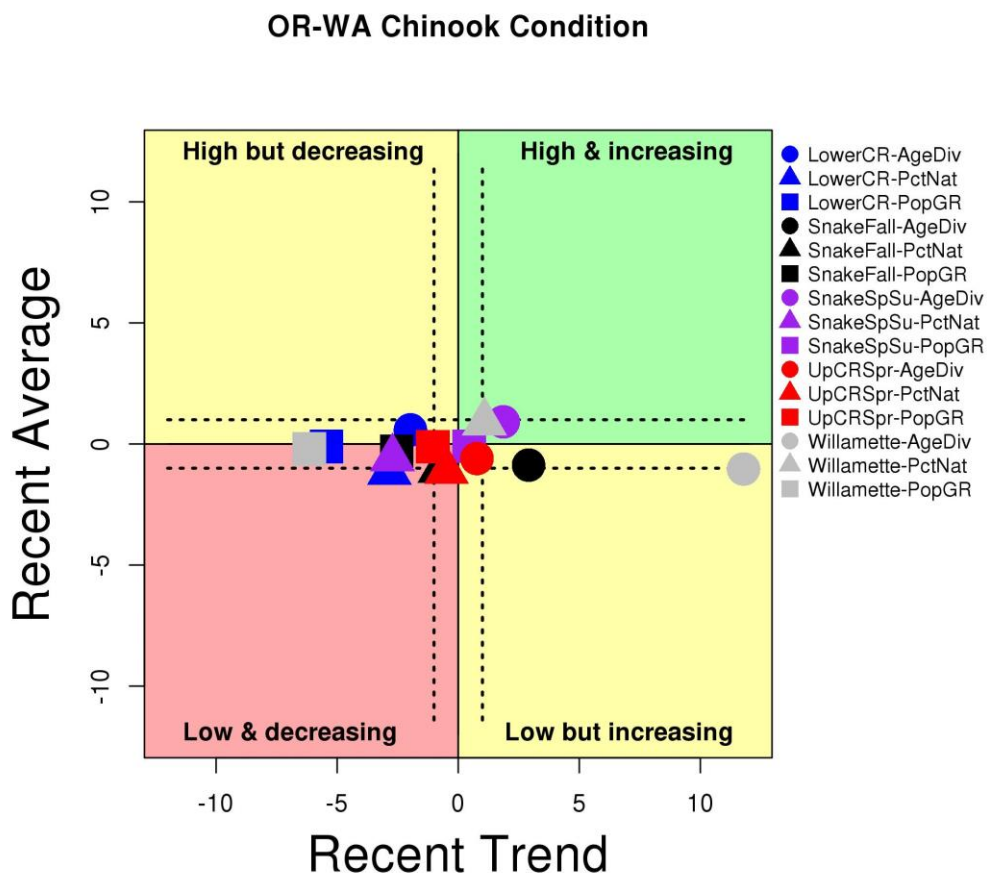
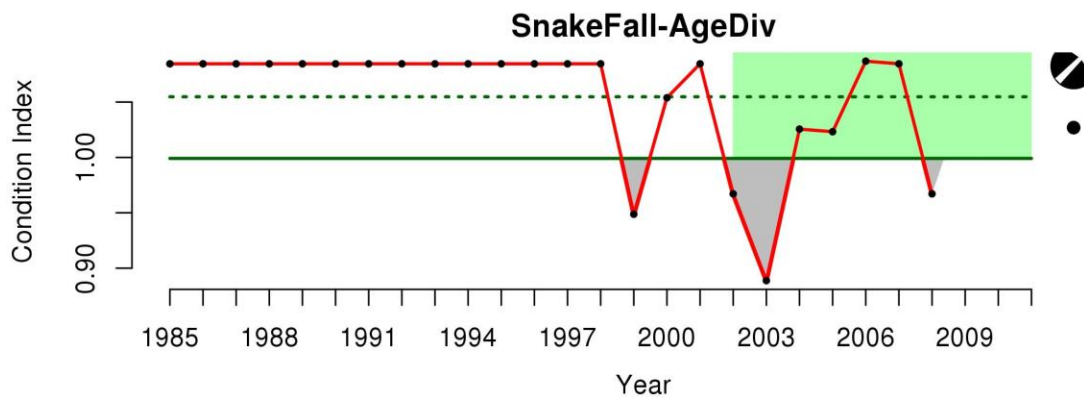
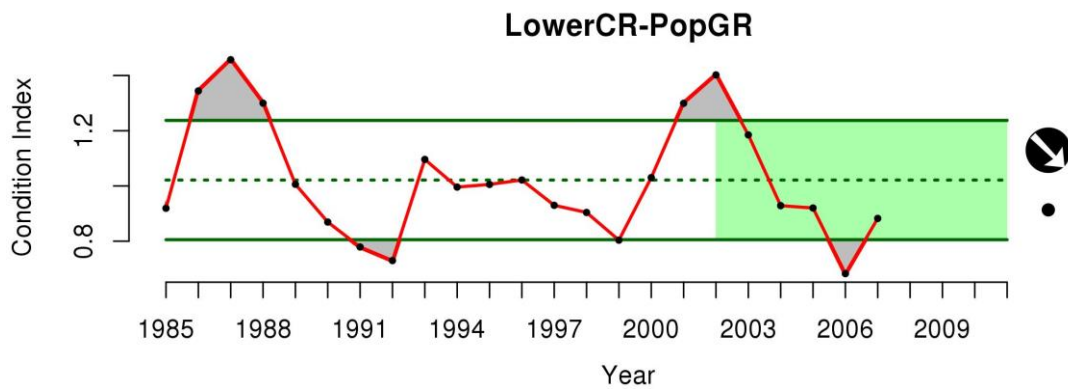
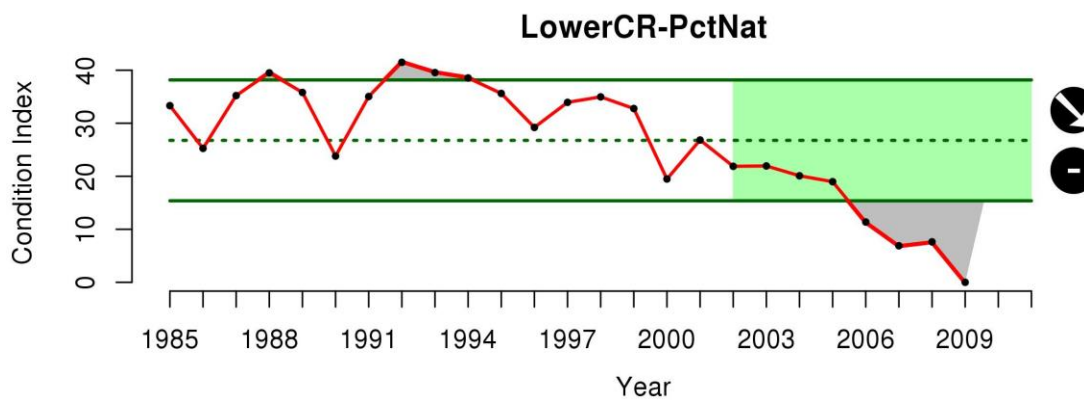
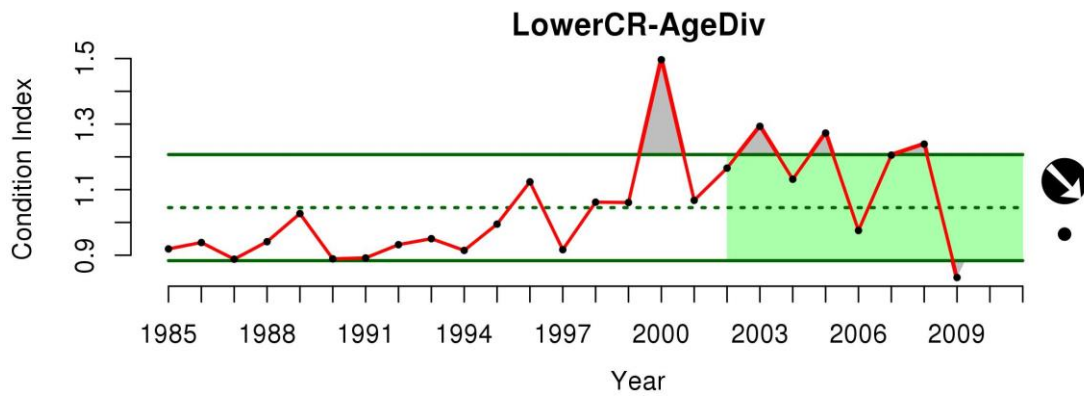
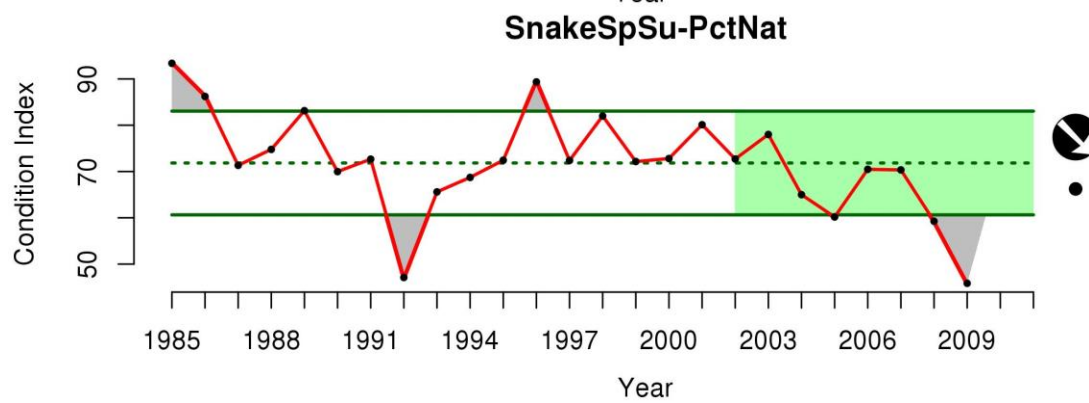
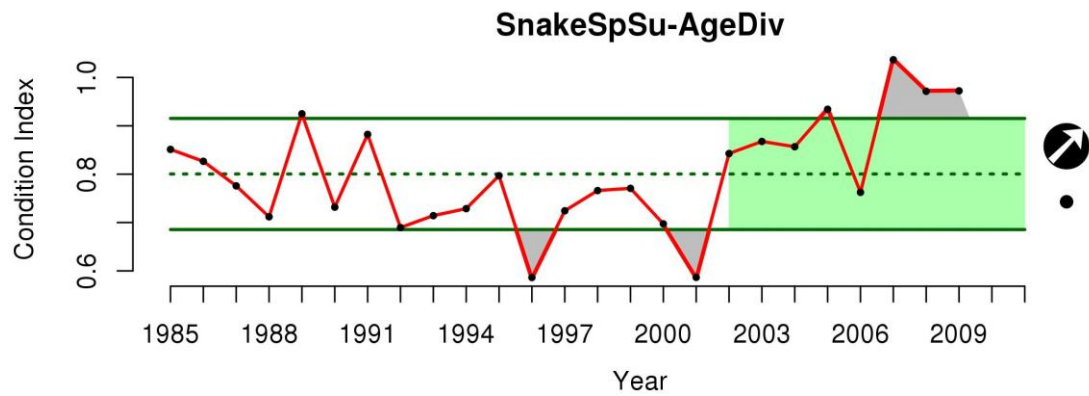
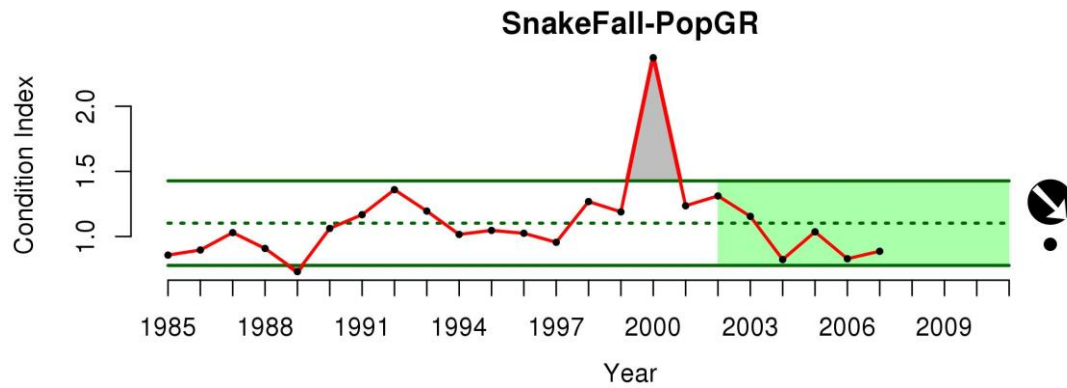
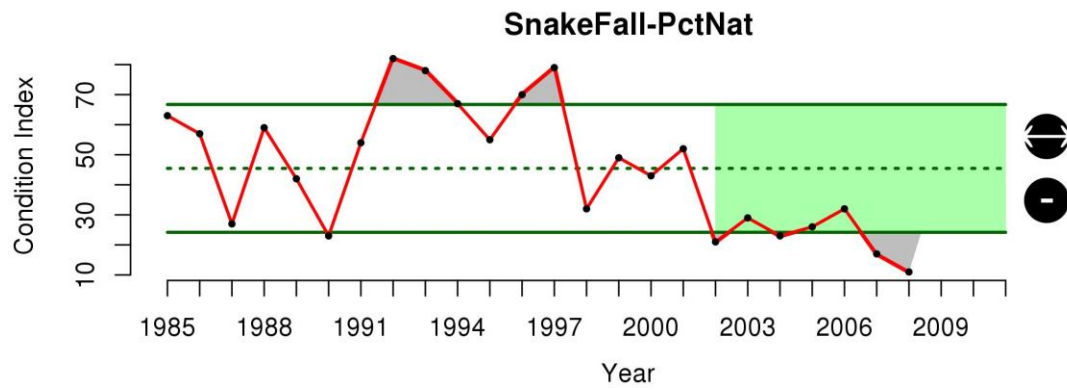
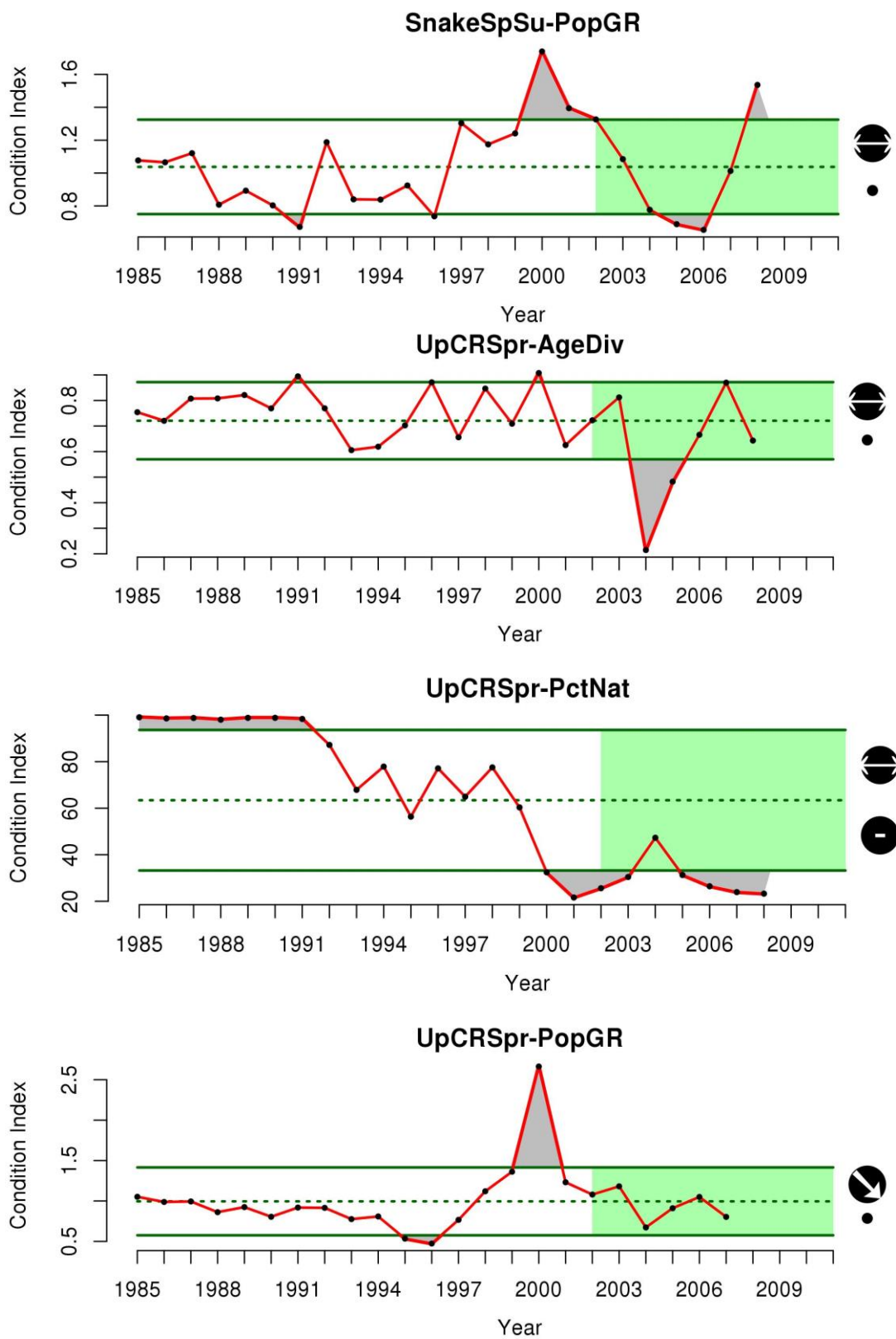


Figure S9. Oregon-Washington Chinook salmon condition. Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. Dotted lines show ± 1.0 s.d. When possible we evaluated percent natural spawners (PctNatural), age-structure diversity (AgeDiv), and population growth rate (PopGR). Subpopulations listed include: lower Columbia River (LowerCR), Snake fall, Snake spring-summer (SnakeSpSu), upper Columbia River summer-fall (UpCRSuFa), and Willamette.







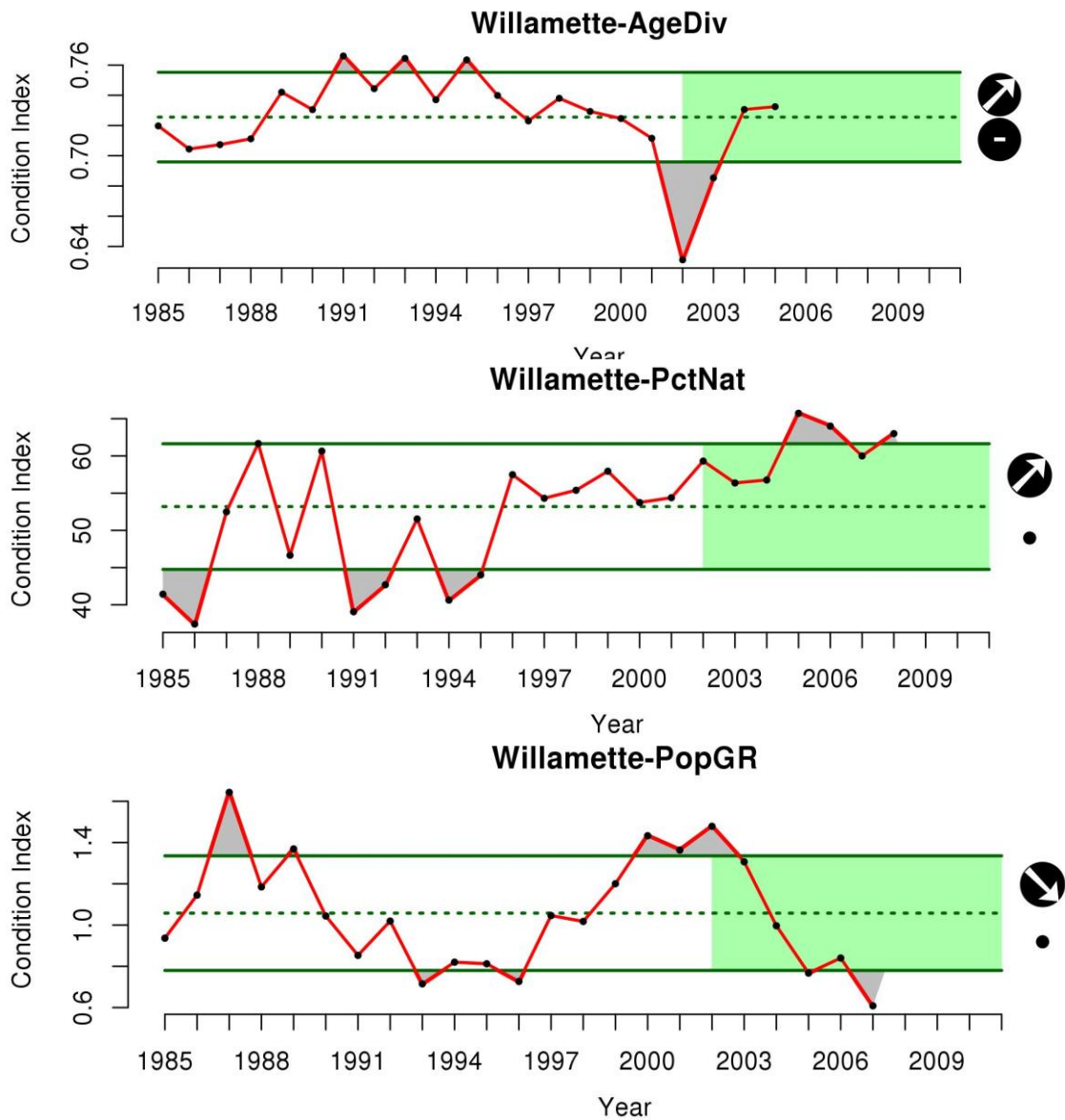


Figure S10 a,b,c. Oregon-Washington Chinook salmon condition. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend was significant over the last 10-years. The lower symbol indicates whether the mean during the last 10 years was greater or less than or within one s.d. of the long-term mean. When possible we evaluated percent natural spawners (PctNatural), age-structure diversity (AgeDiv), and population growth rate (PopGR). Subpopulations listed include: lower Columbia River (LowerCR), Snake fall, Snake spring-summer (SnakeSpSu), upper Columbia River summer-fall (UpCRSuFa), and Willamette.

OREGON-WASHINGTON COHO SALMON: ABUNDANCE

Coho salmon abundance from lower Columbia River was variable but increasing over the past 10 years whereas Oregon Coast abundance was variable with no significant trend over the past 10 years although recent abundances were greater than that observed during the late-1990's. (Fig. S11, S12).

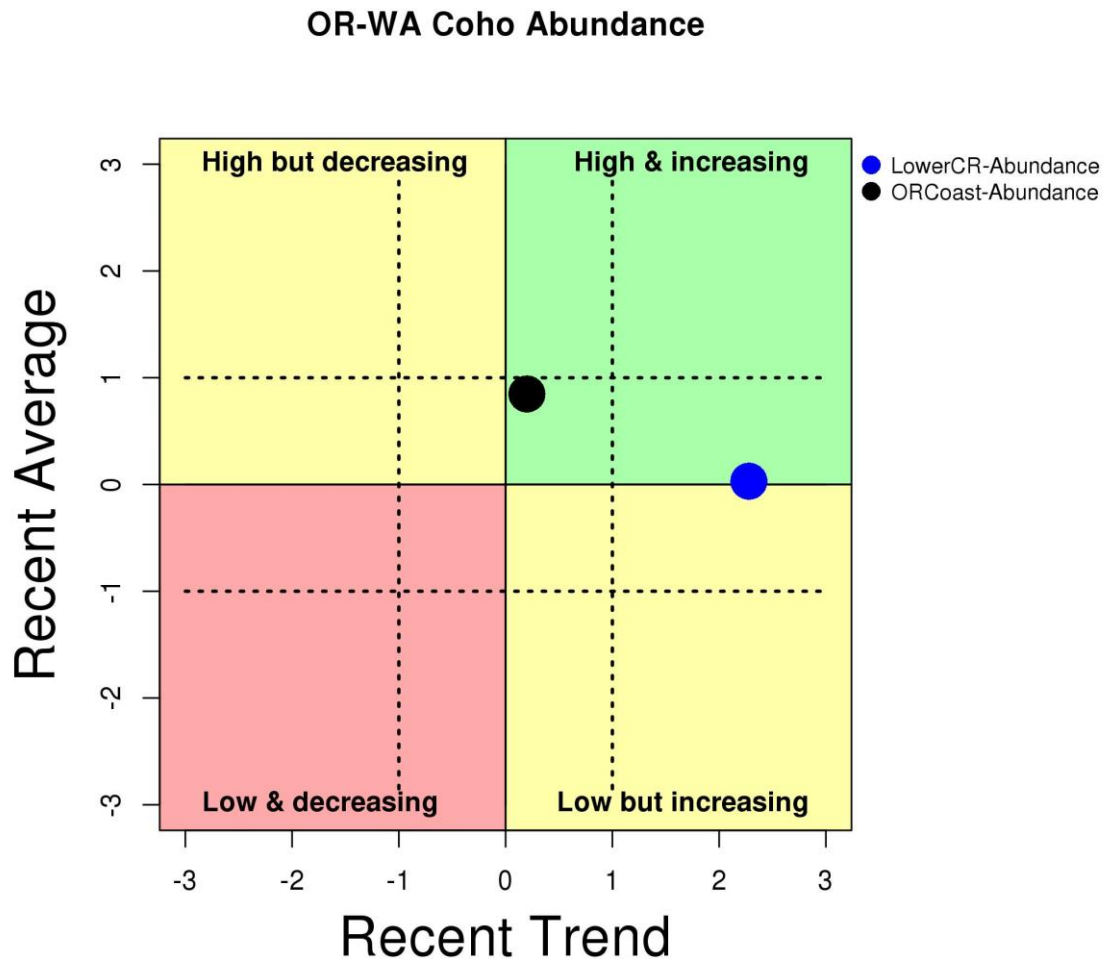


Figure S11. *Oregon-Washington coho salmon abundance.* Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. Dotted lines show ± 1.0 s.d. Subpopulations listed include: lower Columbia River (LowerCR) and Oregon coastal (ORCoast).

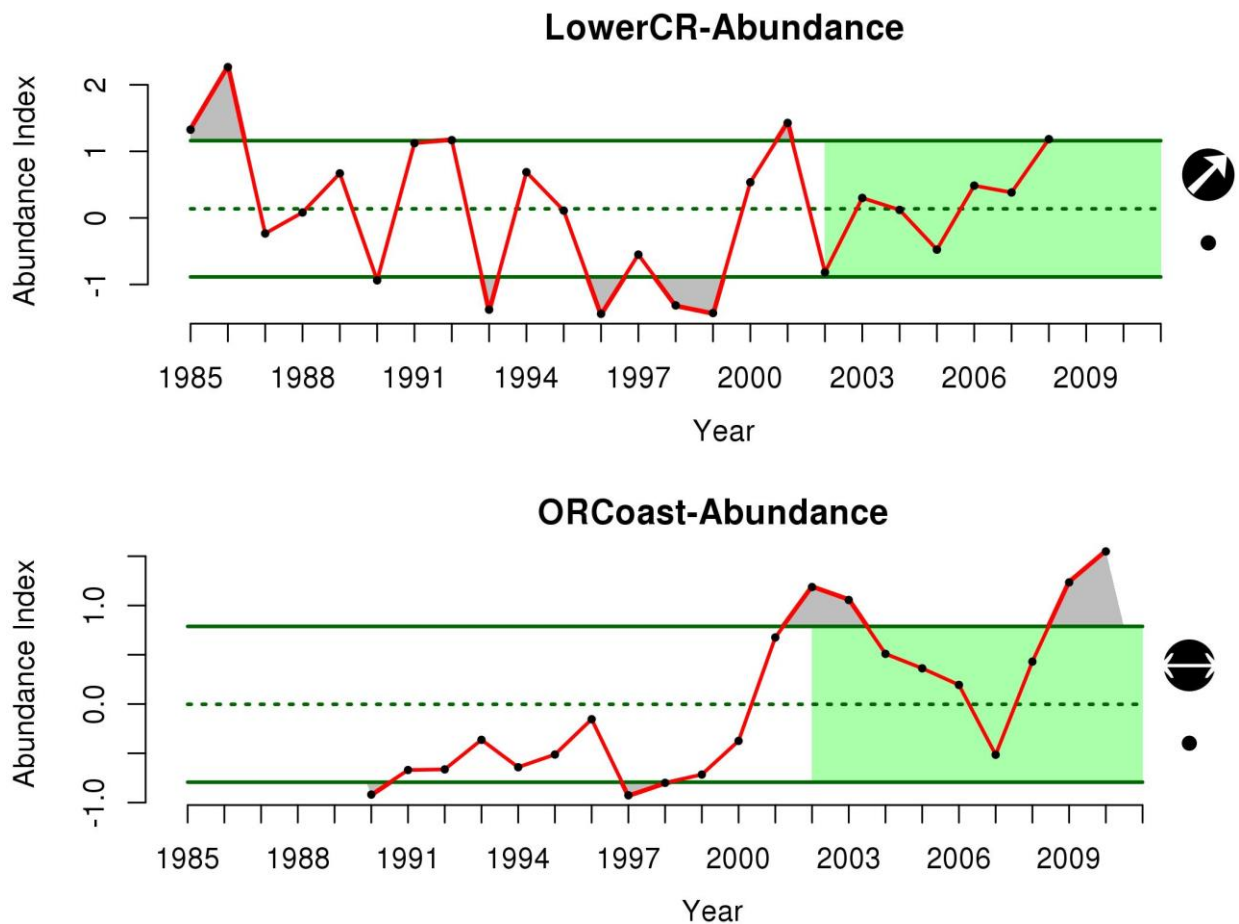


Figure S12. *Oregon-Washington coho salmon abundance.* Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend was significant over the last 10-years. The lower symbol indicates whether the mean during the last 10 years was greater or less than or within one s.d. of the long-term mean. Subpopulations listed include: lower Columbia River (LowerCR) and Oregon coastal (ORCoast).

OREGON-WASHINGTON COHO SALMON: CONDITION

Trends in proportion of natural spawners (“PctNat”) and population growth rate (“PopGrowth”) for these ESUs are neutral or positive at both time scales (Fig. S13, S14). The long term increase of PctNat for Oregon Coast coho salmon is encouraging.

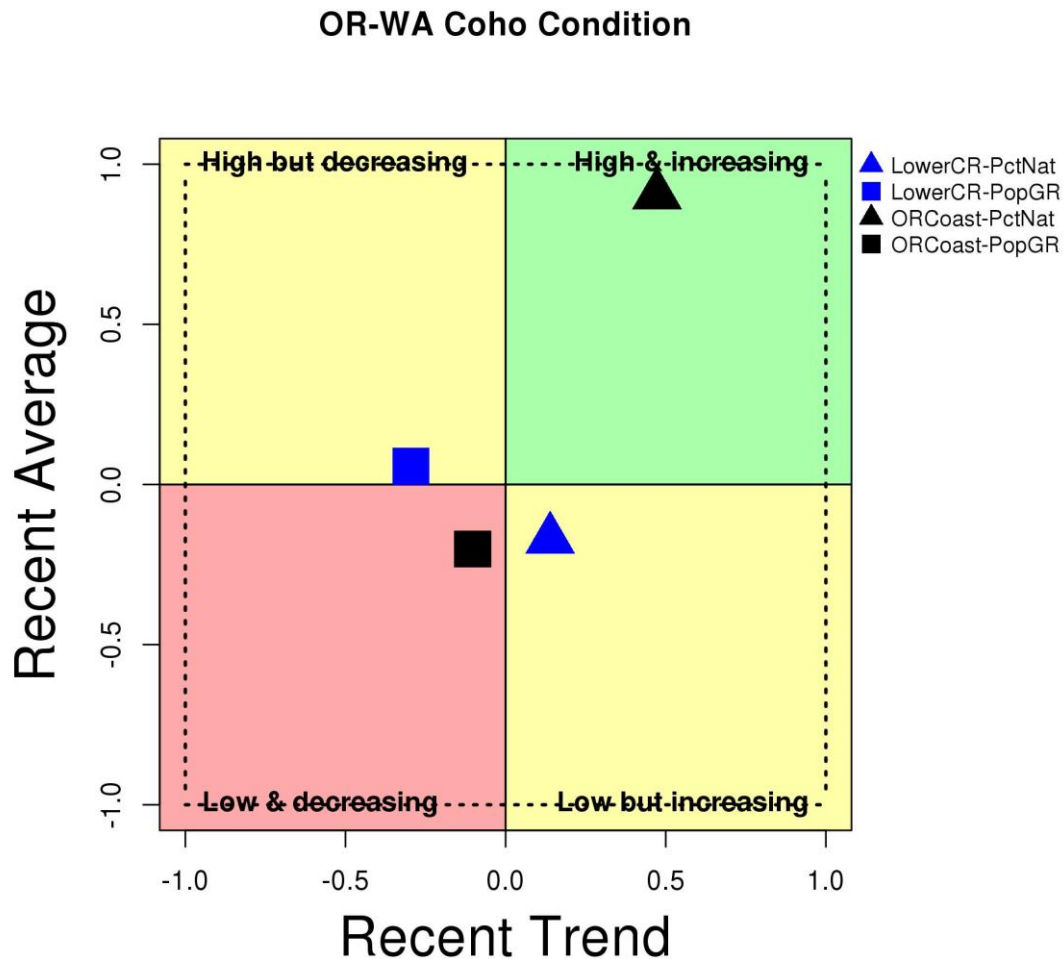


Figure S13. *Oregon-Washington coho salmon condition.* Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. Dotted lines show ± 1.0 s.d. We evaluated percent natural spawners (PctNat) and population growth rate (PopGR). Subpopulations listed include: lower Columbia River (LowerCR) and Oregon coastal (ORCoast).

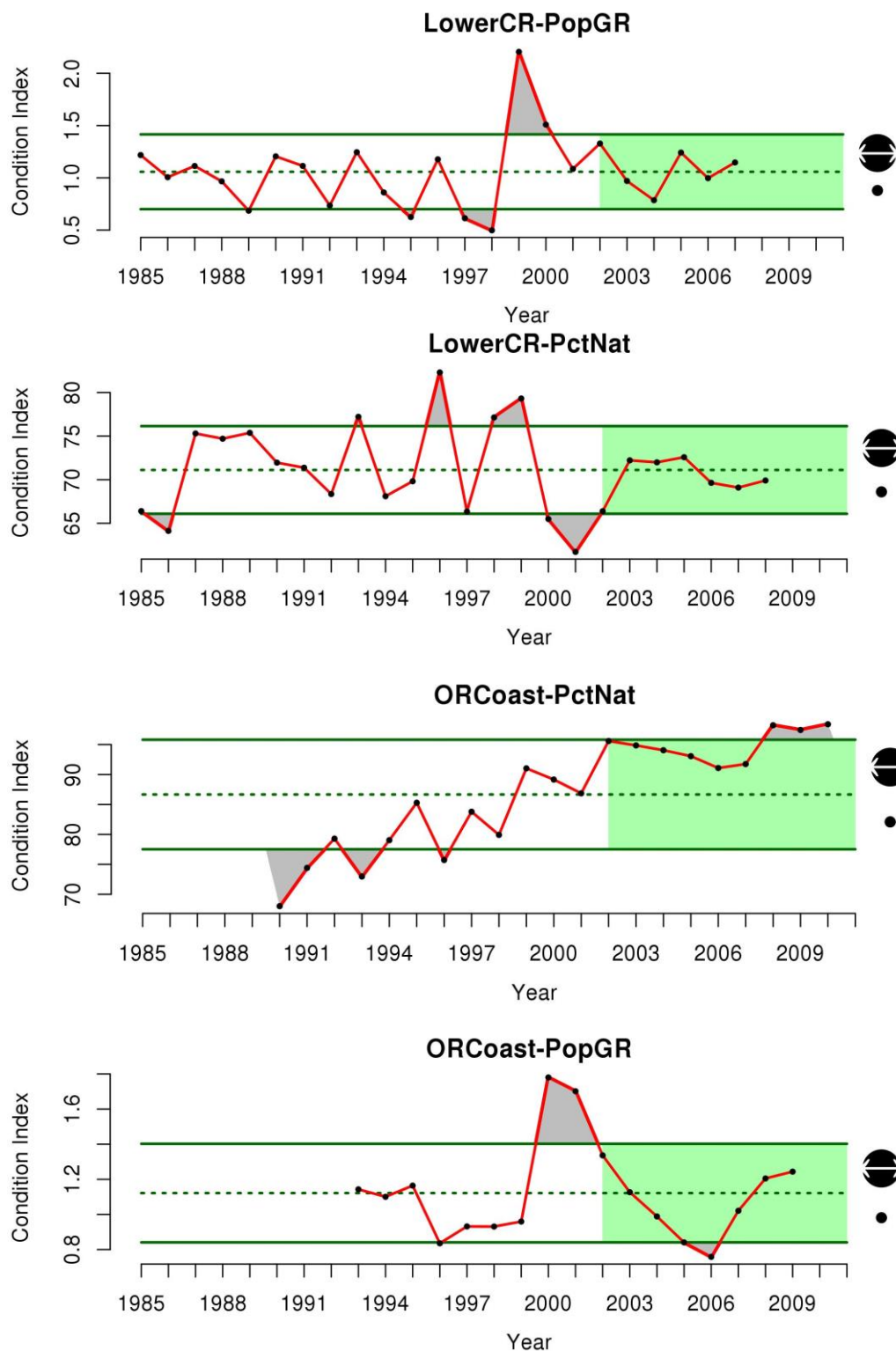


Figure S14. Oregon-Washington coho salmon condition. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend was significant over the last 10-years. The lower symbol indicates whether the mean during the last 10 years was greater or less than or within one s.d. of the long-term mean. We evaluated percent natural spawners (PctNat) and population growth rate (PopGR). Subpopulations listed include: lower Columbia River (LowerCR) and Oregon coastal (ORCoast).

RISK

We do not evaluate risk in this chapter but are working toward developing metrics of risk that could be helpful for evaluating harvest control rules on the populations. Risk evaluation and forecast will be further developed in subsequent reports.

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SEABIRDS

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TABLE OF CONTENTS

| | |
|--|-----|
| Executive summary | 298 |
| Detailed Report | 301 |
| Indicator Evaluation..... | 301 |
| Initial selection and evaluation of candidate indicators. | 301 |
| Top Ranked Indicators | 302 |
| Final Suite of Indicators | 309 |
| Status and Trends..... | 323 |
| Risk assessment – Seabirds (future work)..... | 329 |
| Data Links | 330 |
| References cited | 331 |

LIST OF TABLES AND FIGURES (SB)

| | |
|---|-----|
| Figure SBX. Quadratic plot of trends in abundance at sea for the two most common piscivores in the CCLME (common murre, sooty shearwater) and one of the common planktivores (Cassin's auklet) | 300 |
| Table SB1. Summary of seabird indicator evaluations. The numerical value that appears under each of the considerations represents the score from evaluation criteria supported by peer-reviewed literature. Initials of the individual evaluating each indicator are provided..... | 313 |
| Table SB2. Potential data sets available for contemporary seabird indicator data. | 318 |
| Table SB3. Indicators used to examine Status and Trends data..... | 322 |
| Figure SB1. Trends in seabird density (birds km ²) over time for common murres in the northern (NCC) and southern (SCC) domains of the California Current. | 326 |
| Figure SB2. Trends in seabird density (birds km ²) over time for sooty shearwaters in the northern (NCC) and southern (SCC) domains of the California Current. | 326 |
| Figure SB3. Trends in seabird density (birds km ²) over time for Cassin's auklet in the northern (NCC) and southern (SCC) domains of the California Current..... | 328 |

OVERVIEW

Resident and migratory populations of over 75 legally-protected seabird species depend upon habitats and food webs in the California Current Large Marine Ecosystem. Seabirds provide one of the most publically-visible indicators of ecosystem productivity and health.

EXECUTIVE SUMMARY

The Integrated Ecosystem Assessment (IEA) for the California Current Large Marine Ecosystem (CCLME) explicitly includes seabirds as an ecosystem component because seabirds require CCLME habitats and food webs to maintain healthy populations. The CCLME supports more than 75 species of seabirds, including breeding, non-breeding, and migratory populations. NOAA Fisheries has legal, management, and conservation mandates to understand and protect seabird populations. All seabird species are legally protected by the Migratory Bird Treaty Act (1918). Executive Order 13186 (2001) requires NOAA Fisheries to incorporate migratory birds into Agency planning, address migratory bird concerns, and cooperate with other agencies that have responsibilities for managing or protecting migratory birds. A Memorandum of Understanding between NOAA Fisheries and the US Fish and Wildlife Service addresses areas of joint concern (NMFS 2012). The Endangered Species Act (1973), the Magnuson-Stevens Act (1976), and the US National Plan of Action for Reducing Incidental Catch of Seabirds in Longline Fisheries (2001) also require NOAA to protect threatened seabirds, conserve seabird habitat, address seabird mortality caused by bycatch in fisheries, and evaluate seabird impact on ESA-listed fishes. Although the United States is not currently a signatory on the Agreement on the Conservation of Albatrosses and Petrels treaty (2004), NOAA regularly sends delegates to these international meetings. In response to these mandates and responsibilities, NOAA Fisheries implements a National Seabird Program that specifically calls for the use of seabird indicators to improve ecosystem-based science and management. Inclusion of seabirds in the 2012 IEA is not only necessary for advancement of the IEA process, but also supports several other national-level priorities for science and stewardship of marine resources.

INDICATORS

Through a rigorous selection process, we chose four key seabird indicators from an initial list of 12 indicators. The final indicators are:

- habitat use at sea
- annual reproductive performance
- mortality rates and agents
- diet composition

Information gaps identified include (1) winter data for density/habitat use at sea and diet in all domains of the CCLME; (2) very short, and potential loss of, time series data for habitat use at sea, annual reproductive performance, and diet in Oregon/Washington (northern domain of CCLME); and (3) diet information for non-breeding birds and adult birds in all domains.

STATUS

Seabird indicator data are collected independently by different institutions or individuals, making data synthesis challenging. The 2012 IEA process identified at least 19 sources of contemporary data potentially available from all three CCLME biogeographic domains.

Although it is beyond the scope of 2012 IEA to synthesize all potential 19 seabird indicator data sets, we examined trends in sample data sets from the northern and southern domains of the CCLME. Comparisons of preliminary trends from two common piscivores and one common planktivore showed an increasing trend for one piscivore (common murre, *Uria aalge*); a stable trend for another piscivore (sooty shearwater, *Puffinus griseus*) and a slight decreasing trend for the planktivore (Cassin's auklet, *Ptychoramphus aleuticus*).

Seabird data sets are funded, collected by, and maintained by many different entities (NOAA and non-NOAA); cooperation between these groups and compilation of seabird data is time-intensive. We recommend that future work support (1) a comprehensive synthesis of representative indicator data sets for each biogeographic region of the CCLME and (2) a risk analysis for seabirds based on that synthesis as we were unable to accomplish this task with available resources presently

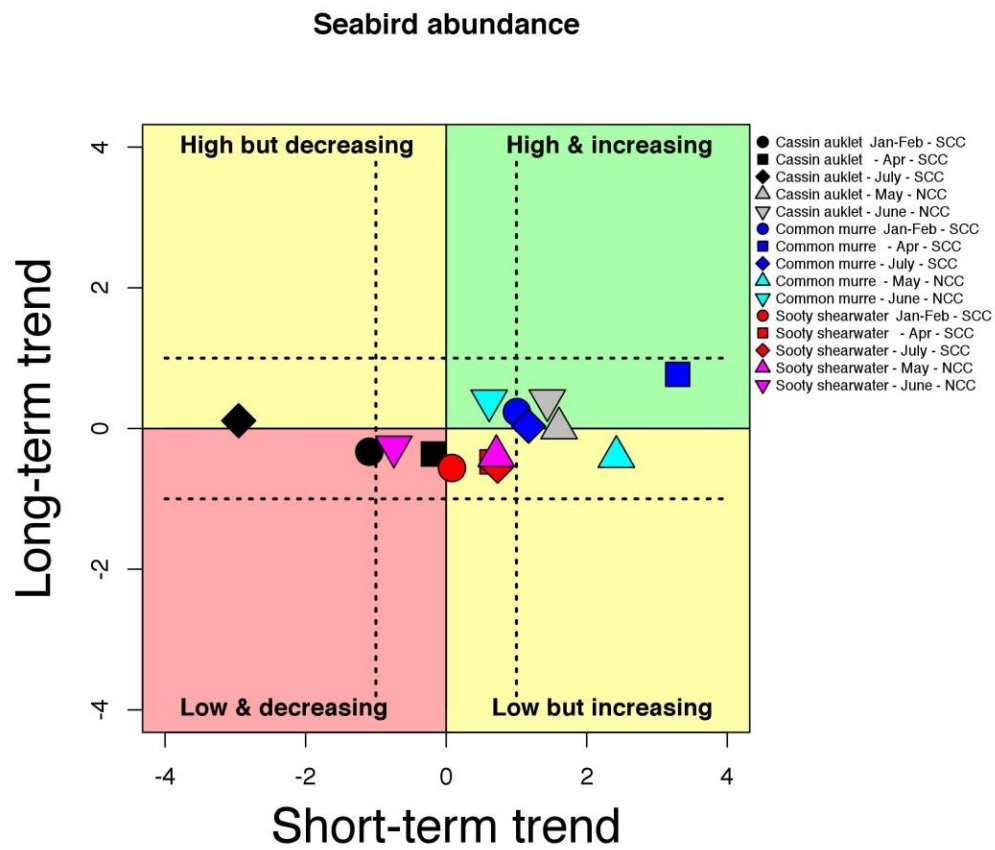


Figure SBX. Quadratic plot of trends in abundance at sea for the two most common piscivores in the CCLME (common murre, sooty shearwater) and one of the common planktivores (Cassin's auklet).

DETAILED REPORT

INDICATOR EVALUATION

BACKGROUND – JUSTIFICATION FOR INCLUSION OF SEABIRD INDICATORS

The Integrated Ecosystem Assessment (IEA) for the California Current Large Marine Ecosystem (CCLME) explicitly includes seabirds as an ecosystem component because seabird species require CCLME habitats and food webs to maintain healthy populations. The CCLME supports more than 75 species of seabirds, including breeding, non-breeding, and migratory populations. NOAA Fisheries has legal, management, and conservation mandates to understand and protect seabird populations. All seabird species are legally protected by the Migratory Bird Treaty Act (1918). Executive Order 13186 (2001) requires NOAA Fisheries to incorporate migratory birds into Agency planning, address migratory bird concerns, and cooperate with other agencies that have responsibilities for managing or protecting migratory birds. A Memorandum of Understanding between the National Marine Fisheries Service (NMFS) and the US Fish and Wildlife Service addresses these areas of joint concern (NMFS and USFWS 2012). The Endangered Species Act (1973), the Magnuson-Stevens Act (1976), and the US National Plan of Action for Reducing Incidental Catch of Seabirds in Longline Fisheries (2001) also require NOAA to protect threatened seabirds, conserve seabird habitat, address seabird mortality caused by bycatch in fisheries, and evaluate seabird impact on ESA-listed fishes. Although the US is not currently a signatory on the Agreement on the Conservation of Albatrosses and Petrels treaty (2004), NOAA regularly sends delegates to these international meetings. In response to all these mandates and responsibilities, NOAA Fisheries implements a National Seabird Program that specifically calls for use of seabird indicators to improve ecosystem-based science and management (<http://www.fakr.noaa.gov/protectedresources/seabirds/national.htm>). That plan specifically calls for the use of seabird indicators to improve ecosystem-based science and management. Therefore, the inclusion of seabirds in the 2012 IEA is not only necessary for advancement of the IEA process, but also supports several other national-level priorities for science and stewardship of marine resources.

INITIAL SELECTION AND EVALUATION OF CANDIDATE INDICATORS.

To evaluate the ecosystem attributes of seabird population size and condition, we required quantitative metrics. Inclusion of measurements of population size and condition for breeding, non-breeding, and migratory birds was considered a requirement.

The Seabird Indicator Team conducted two “brainstorming” sessions with each other, where ideas for all possible quantitative indicators of these seabird-ecosystem attributes were shared. Because we were searching for the best possible indicators, the brainstorming process considered all data types, regardless of whether they had been or are currently being measured in the CCLME. If resources were not limiting and full knowledge of seabird ecosystem attributes were possible, then all 12 indicator types should be measured. However, because resources are limiting, it was necessary to rank the importance indicators as to how well they might represent population attributes. The matrix evaluation process we used to perform this evaluation and ranking is described by Levin and Schwing (2011). This scheme explicitly includes evaluation criteria for data availability, such as extent of geographic coverage or the existence of time series in the evaluation procedure.

Candidate indicators were nominated because it was agreed each one would be an important element to developing accurate, complete, science-based knowledge of seabird populations in this or any other ecosystem. Group members agreed on 12 possible indicators to evaluate (Table SB1).

Each team member was assigned 4 candidate indicators to evaluate and rank. We used literature reviews (primarily from publications in the last 10 years, including gray literature and reports, using citation databases such as Web of Science and Aquatic Sciences and Fisheries Abstracts) to evaluate each indicator. In cases where a team member was aware of very recent work from our own professional experience or contacts (e.g. new diet studies initiated by state biologists), we contacted the principal investigator to point us to any available but unpublished reports. Because there were 12 candidate indicators, this led to a 204-element evaluation matrix for the entire process.

Every matrix element was assigned one of three color codes and scores:

- **green (score 1.0)** = strong literature support;
- **yellow (score 0.5)** = moderate or limited literature support;
- **red (score 0.0)** = weak or no support, or no data/information available

To ensure team members assigned ranks using very similar evaluation criteria, an initial independent run-through and scoring of one indicator by each team member was conducted. We then discussed the thought process each individual used during their literature search, and how each person assigned scoring of the different consideration factors. Once satisfied that everyone was using similar criteria to assign ranks, individual team members then took responsibility for their assigned literature reviews and remaining matrix evaluations.

TOP RANKED INDICATORS

After individuals completed assigned matrix evaluations, the team shared and discussed matrix results. All indicators were ranked according to the sum of scores in the 17 matrix elements across Primary Considerations (n=5 elements), Data Considerations (n=7 elements), and Other Considerations (n=6 elements). Score assignment was reviewed briefly for each element, so that any new literature information provided by the two members not assigned to score a given matrix element could be considered. Only 12 of 204 cases had matrix element scores that were changed due to newly provided information.

Final rank score sums were sorted in descending rank order.

The Top Three seabird indicators selected, with their cumulative score out of 17 possible, were as follows:

- (1) Indicator: habitat use at sea (Attribute: population size and condition, 15/17)
- (2) Indicator: annual reproductive performance (Attribute: population size and condition, 14.5/17)
- (3) Indicator: mortality rates and agents (Attribute: population condition, 14.5/17)

SEABIRD POPULATION SIZE AND CONDITION – HABITAT USE AT SEA

- (1) **Habitat use at sea.** For purposes of this evaluation, the metric “habitat use at sea” includes the use of direct observation of seabirds from ships, land, or aircraft to characterize distribution and abundance at sea; telemetry deployed on individual birds to characterize species ranges, habitat use,

and foraging ecology; and individual marks such as leg or wing bands and dyes to quantify individual use of habitat.

THEORETICAL CONSIDERATIONS

An understanding of spatially-explicit habitat use and requirements is an essential component of applying ecosystem-based management to marine spatial planning (Burger and Shaffer 2008, Crowder et al. 2008, Nur et al. 2011). The use of ships, land, and aircraft to collect these data and estimate population size has a long history in peer-reviewed literature (Spear et al. 1992, Clarke et al. 2003), and studies of seabirds at sea in the CCLME are many (Wiens and Scott 1975, Briggs et al. 1985a, b, Briggs et al. 1987, Briggs et al. 1992, Veit et al. 1996, Veit et al. 1997, Mason et al. 2007, Sydeman et al. 2009, Ainley and Hyrenbach 2010). Use of satellite telemetry began in the 1980s, and is becoming common, affordable, and sophisticated with technological improvements over the last two decades (Burger and Shaffer 2008, Hart and Hyrenbach 2009). At-sea information has been used to evaluate and define habitat for managed species, especially for species of conservation concern (Croxall et al. 2012), ESA-listed species requiring critical habitat designations (Piatt et al. 2006, Suryan et al. 2006, Burger and Shaffer 2008), and for sooty shearwaters which are actively managed in New Zealand as a traditional Maori food source (Lyver et al. 1999, Hunter and Caswell 2005, Nevins et al. 2009).

Data from this indicator type have been used to detect and track population declines as they relate to ecosystem change (Veit et al. 1996, Veit et al. 1997, USFWS 2009, Piatt et al. 2011). Although establishing habitat use is relatively straightforward, understanding the ecological mechanisms driving those patterns may be more difficult to accomplish, as it requires understanding the variance and persistence of underlying marine processes over time (Weimerskirch 2007, Nur et al. 2011, Suryan et al. 2012). In the absence of long-term data sets, data from this indicator can be combined other ecosystem indicators such as reproductive output and diet to make strong inferences and predictions about ecosystem change (Piatt et al. 2007, Field et al. 2010, Cury et al. 2011).

Clear indicator response to management actions, reference points, and targets is possible in systems where actions include introduction, re-introduction, or exclusion of birds from nesting or foraging habitat or significant changes in fisheries practices where bycatch is management concern (Roby et al. 2002, Suryan et al. 2004, Lyons et al. 2005). In other cases, isolating the response to specific management actions or other pressures can be difficult because many factors affect habitat use.

DATA CONSIDERATIONS.

Quantitative, operationally-straightforward methods to examine habitat use in space and time are well-established for direct observation from ships (Tasker et al. 1984, Spear et al. 1992), land (Zamon 2003, Zamon et al. 2007), and air (Briggs et al. 1985a, Mason et al. 2007). Telemetry methods are more complex but also well-established (Burger and Shaffer 2008, Hart and Hyrenbach 2009). While telemetry can provide geographical coverage of an entire ecosystem (Adams et al. 2012), there are substantial spatial and temporal data gaps for direct observations of seabirds at sea (see “Data gaps” section). California is relatively well-sampled because long-term observations are maintained in both southern (CalCOFI/CCE-LTER sites: www.calcofi.org, cce.lternet.edu) and central California (www.sanctuarysimon.org). Northern California and southern Oregon coasts have no regular sampling programs, and the only annual ocean ecosystem sampling program for northern Oregon and Washington is presently in immediate jeopardy of ending (<http://www.nwfsc.noaa.gov/research/divisions/fed/oeip/a-ecinhome.cfm>). For the central and northern domains of the CCLME, winter data on seabird distributions and abundance at sea are rare.

Although data gaps exist in spatial and temporal coverage of the CCLME, there is a well-developed world-wide literature on understanding spatial and temporal variation in seabird habitat use at sea, including the seminal paper by Hunt and Schneider (1987) and other more recent syntheses (Fauchald 2009, Gonzalez-Solis and Shaffer 2009), as well as an extensive literature for those parts of the CCLME where data exist (Ainley et al. 2005, Ainley et al. 2009, Sydeman et al. 2009, Ainley and Hyrenbach 2010, Adams et al. 2012, Suryan et al. 2012, Zamon et al. 2013).

OTHER CONSIDERATIONS.

The use of direct observations, counting, and telemetry to understand how animals are using habitat is something that is intuitively communicated to and understood by both the public and managers. Maps of habitat use are one of the most intuitive tools for communication, and are commonly used in guiding management actions, policy, regulatory processes, and educational or outreach materials. Students can be utilized for some types of data collection. Pairing bird observations with at-sea physical or biological oceanographic surveys and other platforms of opportunity can make this type of data very cost-effective and can provide significant value-added information for ecosystem management. The National Seabird Program, for example, has recently put seed funding into capital equipment purchases for the Northwest Fisheries Science Center so observers can be deployed on ships-of-opportunity, and the Oregon Wave Energy Trust has provided funding to pay trained observers for two surveys. Additional cost-savings are possible by increasing the use of NOAA's Small Boat Program (< 65 ft.) to conduct nearshore surveys, a cost-effective strategy employed by other federal and state agencies in California, Oregon, and Washington (Strong 2009, Pearson et al. 2011). Quantitative use of these data to generate leading indicators of change is not very common, although there are intriguing possibilities suggested by some investigators for birds which migrate north to the CCLME from the southern hemisphere (Lyver et al. 1999, Hyrenbach and Veit 2003). Indicator data are already being used to predict future habitat opportunity, habitat degradation, or potential conflict with human uses (Burger and Shaffer 2008, Nur et al. 2011, Suryan et al. 2012). All of the data types for this indicator are used for and compatible with regional, national, and international work, especially work to identify marine habitats of international conservation concern (e.g. <http://web4.audubon.org/bird/iba/>, see also (Burger and Shaffer 2008, Hart and Hyrenbach 2009, Croxall et al. 2012).

DATA GAPS.

Temporal and spatial coverage is generally of higher resolution in the southern and central domains of the CCLME due to maintenance of the CalCOFI and NOAA Fisheries rockfish surveys over several decades, and due to the location of several National Marine Sanctuaries in California actively involved in at-sea research. Some historical information for the Olympic Coast National Marine Sanctuary exists from ship-based work (http://olympiccoast.noaa.gov/science/surveyscruises/2011/seabird_density.html), and a new small boat ocean survey began in 2011 (<http://olympiccoast.noaa.gov/science/surveyscruises/2011/marinebirds.html>). There are no National Marine Sanctuaries in Oregon.

Annual Ocean Ecosystem Surveys by NOAA Fisheries NWFSC on the Oregon and Washington coasts began in 1998. In addition to collecting data characterizing physical ocean conditions, chlorophyll-*a* distribution, zooplankton communities, juvenile salmon distributions, and epipelagic fish communities, these also include shipboard surveys of seabird distribution and abundance after 2003 (<http://www.nwfsc.noaa.gov/research/divisions/fed/oceanecology.cfm>). The Ocean Ecosystem Surveys filled a data gap for habitat-at-sea information in the northern CCLME domain; however, this entire program is in immediate jeopardy of losing funding for ocean surveys.

Similarly, the use of telemetry to examine bird use of habitat at sea is more frequently used in California than in either Oregon or Washington. Therefore, less is understood about seabird habitat use in the northern domain of the CCLME, although there are notable exceptions such as Hamel et al. (2008) and Adams et al. (2012).

Information on fall and winter habitat use at sea from shipboard or aerial surveys is very rare due to two primary limiting factors. First, there are simply fewer research surveys take place during fall and winter than during spring and summer. More importantly, however, there is no consistent funding source to support placing trained observers on the survey platforms that do go to sea in these seasons. Some winter surveys have been funded as part of collecting baseline ecological data for ocean energy development (Zamon, unpublished data). With a modest amount of support for trained observers, the use of ships-of-opportunity could be better developed for all domains of the CCLME.

SEABIRD POPULATION SIZE AND CONDITION – ANNUAL REPRODUCTIVE PERFORMANCE

- (2) **Annual reproductive performance.** For purposes of this evaluation, the metric “annual reproductive performance” includes quantifying metrics such as the number of breeding pairs (direct observation, plot counts, nest counts, or aerial photographs), timing of egg-laying, egg production, timing of hatching, hatching success, chick growth, timing of fledging, fledging success, fledgling mass, and juvenile-to-adult ratios.

THEORETICAL CONSIDERATIONS.

The use of annual reproductive performance to track population trends and responses in seabirds is well-accepted and a required part of seabird population ecology and conservation, as it allows one to measure responses to both ecosystem change and management actions (Cairns 1987, Furness and Camphuysen 1997, Nur and Sydeman 1999, Caswell 2006, Piatt et al. 2007, Gaston et al. 2009, Field et al. 2010, Cury et al. 2011). Counts and identification of breeding pairs from colonies is necessary to include because state and federal management agencies require information on population sizes for management and conservation actions. Information on annual reproductive performance has been used to set and monitor defined reference points and targets for population recovery, as well as document range expansions or contractions of breeding birds (USFWS 2009, Wolf et al. 2009, Cury et al. 2011). Attribution of population responses to specific ecosystem changes or management actions requires the synthesis of several variables to make strong inferences regarding mechanisms driving population change (Frederiksen et al. 2007).

Tracking abundance trends of multiple coexisting species on breeding colonies is accepted standard operating procedure for quantifying seabird population size (Ainley et al. 1994, Walsh et al. 1995). Having quantitative population estimates is essential for agency agreement on managing species protected by the Migratory Bird Treaty Act (1918) and the Endangered Species Act (1973) (Warzybok and Bradley 2010). Because seabirds are long-lived species with low variation in reproductive rates, it can be difficult to attribute population responses to specific causes such as ecosystem-wide change in ocean climate, regional changes in the forage base, or local effects on a particular colony (Manuwal et al. 2001, Thibault et al. 2010). Land-based management actions such as predator removal, invasive species control, and limiting human disturbance are often easier to link to population responses than marine-based management actions (USFWS 2008, Dunlevy et al. 2011, Towns et al. 2011). However, reference points and targets for populations are often set in terms of population size thresholds for protected species such as marbled murrelets (*Brachyramphus marmoratus*) or short-tailed albatross (*Phoebastria albatrus*), so this metric is important to practical stewardship in management agencies (USFWS 1997, 2006, 2008).

It is important to note, however, that CCLME habitat is important to non-breeding individuals and migratory populations as well. In those cases, indicator data of this type would necessarily come from research and monitoring external to the CCLME ecosystem.

DATA CONSIDERATIONS.

Data of this type are quantitative and operationally simple to measure (Sydeman et al. 2001). Historical data records are available from at least one source in all three domains of the CCLME (Sydeman et al. 2001, Saenz et al. 2006, Thayer and Sydeman 2007, Millus and Stapp 2008, Gaston et al. 2009). Sites with the most complete and consistent temporal coverage are in southern and central CCLME, whereas coverage in the northern domain is typically too sparse for time series analysis except in one or two cases (Lee et al. 2007, Sydeman et al. 2009).

Quantitative methods for surveying seabird colonies are well-established (Ainley et al. 1994, Walsh et al. 1995), and historical data do exist for some well-studied species in the CCLME dating back to the 1960s (Anderson and Gress 1983, Ainley et al. 1994). Very few species are monitored with broad geographic coverage throughout the CCLME (although the common murre *Uria aalge* is an exception to this, (Manuwal et al. 2001)). The Farallon Islands in the central domain of the CCLME has the most complete multispecies, time series data set (Warzybok and Bradley 2010). Spatial and temporal variation in breeding numbers is influenced by a number of factors which can make it difficult to separate cause and effect. Local predator disturbance at colonies is becoming an increasingly important confounding factor when attempting to attribute population responses to terrestrial vs. marine causation (e.g. Hipfner et al. (2012). There is often a low signal-to-noise ratio inherent in seabird population counts due to their longevity and low reproductive rates (Ainley and Boekelheide 1990, Ainley et al. 1994).

Recognition of the importance of annual reproductive performance data to ecosystem management is growing. Investigators are establishing new monitoring programs or resurrecting discontinued programs in key areas to fill data gaps (see “Data Gaps”, this section).

OTHER CONSIDERATIONS.

Population counts are readily understood by the public and by managers because this is the most commonly used metric of population size for all organisms. In areas where there is conflict caused by seabird predation on ESA-listed species (e.g. Pacific salmon in estuaries and coastal areas), population size is of both public and management concern (e.g. Good et al. (2007), Anderson et al. (2004)). Counting birds on colonies is generally accepted as a reliable and meaningful method to track seabird populations (Ainley et al. 1994, Walsh et al. 1995). Although it can be expensive to maintain long-term colony monitoring for areas where access to remote sites is required, cost-sharing by multiple agencies and organizations can make such programs affordable. Pairing this work with a larger research effort is also a way to keep data collection cost-effective (Mallory et al. 2010). Population size is typically used for retrospective analyses (e.g. Piatt et al. (2007). However, present population size and past variation in population size can be used to project extinction probabilities into the future, and in some cases these are being explored as ways to manage ESA-listed bird species such as the marbled murrelet and short-tailed albatross (USFWS 2008, 2009). Seabird colony counts are found elsewhere in the region, nation, and world. For reviews of global information, see Anker-Nilssen et al. (1996), Walsh et al. (1995), and Hatch (2003).

The concept of successfully replacing adults with offspring to maintain population viability is an intuitive one that is readily communicated to and understood by the public and resource managers. Popular culture maintains an interest in seabirds through production of nature shows for television and film. Some

recent work is beginning to explore predictive applications for annual reproductive performance (Kitaysky et al. 2010), but in general these data provide retrospective and real time measures of population condition, not predictive measures. This data type is collected regionally, nationally, and internationally, and has been used to make global inferences and recommendations for fisheries practices (Cury et al. 2011).

DATA GAPS.

Consistently-maintained time series with durations greater than 10 years are missing from the northern domain of the CCLME. New information is becoming available on the Washington coast for a few locations (Destruction Island, Tatoosh Island, and Protection Island), via collaborations among Washington Department of Fish and Wildlife, the University of Puget Sound, the University of Washington, and NOAA Fisheries NWFSC (S. Pearson, P. Hodum, and T. Good, pers. comm; see also <http://wdfw.wa.gov/conservation/research/projects/seabird/>). Coverage of annual reproductive performance on the Oregon coast is particularly lacking, in part due to the logistical difficulty of accessing colonies and in part due to lack of historical programs. Robert Suryan (Oregon State University) is establishing a program at Yaquina Head, OR. Unfortunately, one monitoring program which had been maintaining historical data sets on Leach's storm-petrel (*Oceanodroma leucorhoa*) at Saddle Rock, OR, recently ended because that storm-petrel population was wiped out by raccoon and river otter predation on the colony (Janet Hodder, University of Oregon, pers. comm.).

Colony counts may or may not be made on an annual basis, depending on resources available to the agencies responsible for conducting surveys (e.g. Naughton et al. (2007)). Colony sizes can be quite dynamic, especially in recent years when predator disturbance has become an issue (Hipfner et al. 2012) In general, California, Oregon, and Washington do maintain inventories of seabird colony locations and sizes through federal and state wildlife programs.

With the exception of the ESA-listed marbled murrelet (*Brachyramphus marmoratus*), variation in the reproductive performance of less common seabird species, species that do not nest in dense colonies, and species that do not breed in the CCLME, is not as well-documented. Many species use the CCLME but do not breed in the CCLME, so measures of reproductive performance need to come from other ecosystems. Cross-ecosystem integration for conservation purposes is recognized as important but is not commonly practiced (Nevins et al. 2009). For example, the most common bird in the CCLME during summertime, the sooty shearwater (*Puffinus griseus*) breeds in New Zealand and Chile; albatross species of conservation concern (Diomedidae) breed on tropical or subtropical offshore Pacific islands; and pelagic seabirds such as northern fulmars (*Fulmarus glacialis*) and black-legged kittiwakes (*Rissa tridactyla*) breed in Alaska.

SEABIRD POPULATION CONDITION – COUNTS AND IDENTIFICATION OF MORTALITY

- (3) ***Mortality rates and agents.*** For purposes of this evaluation, the metric “mortality rates and agents” includes metrics such as number and species of mortalities reported from various sources including but not limited to mass strandings, beach-cast birds, bycatch in fisheries, harmful algal blooms, disease/pathogens/parasites, predation, collisions, and pollution/spills. It is also intended to include necropsy data where cause of death can be established.

THEORETICAL CONSIDERATIONS.

The ability to quantify mortality effects and mortality sources is a key element of population ecology. There is good support for examining mortality in seabirds as a way to understand what mortality factors are

affecting bird populations (Camphuysen and Heubeck 2001, Roletto et al. 2003, Moore et al. 2009, Materna et al. 2011, Phillips et al. 2011). Bycatch impacts from fisheries is of management concern (Fitzgerald et al. 2008). Mass strandings, beached birds, and oil spill mortality are of concern to scientists, management, and the public. Attributing cause to these mortality events may be straightforward in some cases (Phillips et al. 2011) but not in others (Parrish et al. 2007). Fisheries bycatch is one mortality agent for which a direct response to management action is measurable (Melvin et al. 2001, Fitzgerald et al. 2008). Mortality measures are actively used as management reference points and targets for populations of ESA-listed species (USFWS 1997, 2006, 2008) and for bycatch thresholds which trigger fisheries closures (USFWS 2008).

DATA CONSIDERATIONS.

Standard protocols for monitoring, reporting, and quantifying mortality are available and commonly used (Roletto et al. 2003, Hamel et al. 2009, Moore et al. 2009, Phillips et al. 2010, Phillips et al. 2011). Historical data are with relatively broad spatial coverage and good time series are available for beached bird surveys and inferred gill net mortality in the CCLME (Parrish et al. 2007, Moore et al. 2009), but information from direct observations of bycatch in West Coast fisheries is only now starting to be investigated (Jannot et al. 2011). There is a coordinated, multi-agency network in place to collect specimens from oil spills, but data are often subject to severe access restrictions because they are considered evidence for legal proceedings. Compiling recent or historical data on oil spill mortality can therefore be logistically complicated. There is no existing coordinated multi-agency effort to integrate coverage of episodic mortality events which are not caused by oil spills, but it is recommended that one be established to assist with documenting these types of mortality events, especially given ocean climate change may impact the frequency and intensity of harmful algal blooms (Phillips et al. 2011).

Understanding of spatial and temporal variation in seabird mortality is not well-developed, although the factors affecting mortality are theoretically understood (Camphuysen and Heubeck 2001, Parrish et al. 2007). The episodic nature of mortality events that humans can observe is almost always confined to events on or near shore, although there are rare occasions when mortality at sea is documented (Baduini et al. 2001). High variability in the probability of detecting mortality also makes it difficult to track trends in overall mortality over time. Partitioning mortality from one source (e.g., derelict fishing gear) relative to other sources (e.g., bycatch from active fishing) can be difficult due to a lack of comparable data from multiple mortality sources (Good et al. 2009). Except for certain cases where population sizes of ESA-listed species are known, it is most often unknown what portion of overall mortality each specific type of seabird mortality represents.

OTHER CONSIDERATIONS

Death and causes of death in wildlife are concepts which are commonly understood by citizens and managers. Mortality events are often highly visible to the public and almost always result in public inquiries as to the cause of such mortality events. In some cases, the public is the first to report an event that triggers agency responses (Phillips et al. 2011). Images of dead birds entangled in fishing gear, oiled by pollution, or emaciated due to starvation are powerful tools for communicating messages about ecosystem risk and health to the public and to managers. Recovery of specimens for necropsy can often be cost-effective, making use of volunteers and staff from multiple agencies in a response situation. Beached bird surveys are volunteer-driven and provide good examples of citizen-science in action which are used not only in the CCLME (for example, www.sanctuarysimon.org/monterey/sections/beachCombers/index.php, (Parrish et al. 2007)), but in other regions and countries as well (Powell Island and Imber 1988, Camphuysen and Heubeck 2001, Wiese and Ryan 2003, Zydelski et al. 2006).

DATA GAPS.

Beached bird programs have good coverage over much of the west coast where public beaches are available. Seabird bycatch in West Coast fisheries is not as quantitatively or broadly monitored as it is in Alaska, but that is starting to change as fishery observer data sets become available to seabird biologists due to concern about recent ESA-listed species interactions with some fishery sectors. For example, a new, collaborative program to quantify bycatch from commercial fisheries in the CCLME has begun with scientists from Washington Sea Grant (E. Melvin and T. Guy, <http://wsg.washington.edu/mas/resources/seabird.html>), NOAA Fisheries NWFSC West Coast groundfish observer program (J. Majewski, <http://www.nwfsc.noaa.gov/research/divisions/fram/observer/index.cfm>), and NOAA Fisheries Alaska Regional Office (K. Rivera and S. Fitzgerald, <https://www.fakr.noaa.gov/protectedresources/seabirds.htm>). Specimen recovery from bycatch for necropsy analysis is not yet established; we recommend necropsy analysis be supported.

Growing concern over the potentially increasing frequency, intensity, and duration of harmful algal blooms, and their impacts on seabirds, points to a need for a rapid, interdisciplinary response to understanding these ecosystem events as mortality sources for seabirds as well as other living marine resources (Jessup et al. 2009, Phillips et al. 2011).

The biggest data gap is in understanding natural, non-anthropogenic mortality at sea. This gap may be very difficult to fill because observations of natural mortality at sea are very rare. The best that can be done at this time is to take full advantage of opportunities to study at-sea mortality events whenever possible (e.g. Baduini et al. (2001)).

FINAL SUITE OF INDICATORS

Selection of a final, complete suite of indicators was discussed, and final indicator choice was based on the following criteria:

- rank score of 17 possible,
- the desire to include representative indicators for population size and condition for both breeding and non-breeding/migrant birds,
- the need to avoid redundant information, and
- the need to include complementary data types.

The Final Four seabird indicators we selected, with their cumulative score out of 17 possible, were:

- (1) Indicator: habitat use at sea (Attribute: population size and condition, 15/17)
- (2) Indicator: annual reproductive performance (Attribute: population size and condition, 14.5/17)
- (3) Indicator: counts and identification of mortality and mortality agents (Attribute: population condition, 14.5/17)
- (4) Indicator: diet composition (Attribute: population condition, 13.5/17)

The Top Three indicators were included for reasons discussed in the previous section.

Because we are explicitly interested in ecosystem-based management, we also decided it was critical to seabird diet as an indicator. Diet information is necessary to determining what food resources are being used by seabirds. Without it, one cannot link lower trophic level production in the ecosystem to birds, and one cannot make inferences or predict how ecosystem or fisheries management changes at lower trophic levels will affect seabirds.

Specific information supporting inclusion of diet as an indicator is discussed below.

SEABIRD POPULATION CONDITION – DIET COMPOSITION

- (4) ***Diet composition.*** For purposes of this evaluation, the metric “*diet composition*” includes methods such as traditional gut content and prey identification analysis, scat analysis, observations of bill loads, direct observation of predation events, stable isotope analysis, fatty acid analysis, and molecular analysis of prey remains. Diet composition also includes not only prey items fed to chicks (the most commonly collected diet information), but also prey taken by non-breeding birds and adults for self-feeding.

THEORETICAL CONSIDERATIONS.

Diet information is required by managing agencies to determine what prey species are supporting seabird populations in ecosystem-based management (Schrimpf et al. 2012). The use of diet data to track changes in prey use and prey resources as well as foraging and breeding success is widely accepted and has been used in the CCLME (Cairns 1987, Barrett et al. 2007, Piatt et al. 2007, Sydeman et al. 2009). The influence of fisheries discards on diet composition needs to be considered for some species, as has been seen in other systems (Navarro et al. 2009, Bugoni et al. 2010, Vaske 2011). Inferences from stable isotopes and fatty acids are more complicated to interpret because factors other than prey type can influence chemical composition (Iverson et al. 2007, Sears et al. 2009, Williams and Buck 2010). The most powerful approaches use two or more tools to examine diet composition (Sydeman et al. 1997, Karnovsky et al. 2008). In addition to informing management about seabirds, diet information can evaluate the direct effects of birds on prey species of conservation concern such as Pacific salmon (Roby et al. 2003) and can track ecosystem or prey community changes in the marine environment (Thayer et al. 2008). Diet information has also been used to evaluate the effectiveness of specific fisheries management actions (e.g. reducing avian predation on Pacific salmon), the potential effects of ocean energy development on prey species consumed by seabirds, and the ability to meet management targets for bird population sizes. However, partitioning the response component between management action and natural variation can be difficult (Pichegru et al. 2010, Perrow et al. 2011). In some cases, stable isotope and fatty acid sampling have been used to make inferences about resources supporting protected species and non-breeding species when it is not possible to sample diet directly (Kakela et al. 2010, Ronconi et al. 2010).

DATA CONSIDERATIONS.

Diet data are quantitative and well-accepted by seabird ecologists as indicators of what species in an ecosystem are necessary to support seabird populations (see comprehensive methodological review by Barrett et al. (2007)). Historical data from the 1970s, 1980s, and 1990s exist in locations within the CCLME (Gaston et al. 2009, Sydeman et al. 2009). Examination of museum specimens with stable isotope methods is possible (Newsome et al. 2010). Fatty acid data, however, were not commonly archived in historical data sets, as the frozen storage necessary to preserve specimens has only recently become available. Geographical coverage of diet in any form is limited to a handful of sites sparsely distributed in the three domains of the

CCLME (Channel Islands, CA; Farallon Islands, CA; San Francisco Bay, CA; Yaquina Head, OR; East Sand Island, OR; Destruction and Tatoosh Islands, WA – see Warzybok and Bradley (2011), Suryan et al. (2011), and Roby et al. (2003) for a few examples). Relatively continuous time series of diets for any species are rare; the most complete data set is from the Farallon Islands in the central CCLME domain, and this information is often used to make inferences for the entire CCLME. These inferences may not be appropriate for the northern and southern biogeographic domains of the CCLME because of the different physical and biological processes driving prey community dynamics in those locations.

OTHER CONSIDERATIONS.

It is well-understood by both the public and resource managers that an ecosystem must sustain the right kind and right amount of prey species to maintain healthy seabird populations. Diet is therefore perceived as being a reliable and meaningful indicator of what ecosystem resources are necessary to support seabird populations. There has been a great deal of recent public and legal attention given to the ecological importance of coastal pelagic species (forage fishes) in supporting many components of the CCLME (Enticknap et al. 2011). Field collections and processing of diet data can be labor-intensive. However, multi-agency partnerships (Suryan et al. 2011, Warzybok and Bradley 2011) and opportunistic sampling (Lance and Pearson 2012) can significantly increase cost-effectiveness. Although not typically used to forecast population trends in birds, some studies have shown that diet quality can predict subsequent reproductive success or survival (Sorensen et al. 2009, Kitaysky et al. 2010). Diet information from seabirds is collected worldwide, and comparisons can be made among sites when similar methodology is used (Barrett et al. 2007). Examples of recent global and regional reviews, including the CCLME, can be found in Cury et al. (2011), Gaston et al. (2009), and Sydeman et al. (2009).

DATA GAPS.

Almost all diet studies report what prey types breeding birds are feeding to chicks. Diets of adult birds, non-breeding birds, and migrants have seldom been examined in the CCLME (for an exception, see Varoujean and Matthews (1983)), and no time series for these diet types are available to our knowledge. Virtually nothing is known of winter diets for non-breeding birds in the CCLME. Because ocean climate shifts have significant effects on the species composition of fish and zooplankton, historical diets do not necessarily provide an accurate representation of present-day diet composition.

INDICATORS NOT INCLUDED IN THE FINAL SUITE

Two relatively high-ranking indicators had tied scores with indicators in the final suite of selected indicators, but were not selected as part of that final suite of five indicators (Counts and identification of birds at sea, score 13.5/17; and contaminant loads, score 13/17). Because determining habitat use at sea requires survey data from counts at sea, we considered it appropriate to drop counts and identification of birds at sea as a separate indicator from our final list due to the redundancy of information in these related indicators. Similarly, because it would be possible to include screening for tissue contaminant load in other sampling, and because historical records of contaminant loads are not common, we considered it appropriate to drop tissue contaminant load as a separate indicator. Instead we urge investigators to sample contaminants whenever possible opportunities for tissue samples arise (e.g. when salvaging dead specimens, tagging live specimens, or handling birds on the colony).

It was noted that multivariate seabird indices (score 11.5/17) could be derived from data types in the final list we selected. Therefore, this indicator type would be implicitly included seabird indicator data.

The other four candidate indicators ranked lower in cumulative scores (<11.5) and were therefore not included in the final list. The reasons for this typically included difficulty in detecting or attributing trends in these indicators to specific CCLME changes or management actions, a lack of historical information with good temporal or geographic coverage, and fewer papers available in peer-reviewed literature applying those indicators to ecosystem questions. We did not think that any of these lower-ranking indicators would miss a critical or essential component of ecosystem information at this time.

Table SB1. Summary of seabird indicator evaluations. The numerical value that appears under each of the considerations represents the score from evaluation criteria supported by peer-reviewed literature. Initials of the individual evaluating each indicator are provided.

| CONSIDERATIONS | | | | | |
|-----------------------------|-------------------------------------|-------------|----------|-----------|--|
| Attribute | Indicator | Primary (5) | Data (7) | Other (6) | Summary comments |
| Population size & condition | (1) Habitat use at sea | 4 | 7 | 4 | Essential indicator, demonstrated literature support for utility. Necessary to obtain information on non-breeding residents and migratory species. Primary methods include ship, land, or aircraft-based surveys, but individually-based marking and telemetry also provide complementary data. Oregon, Washington less well-studied than California. Winter conditions poorly understood in most locations. (JEZ) |
| Population condition | (2) Annual reproductive performance | 4.5 | 6.5 | 3.5 | Essential indicator with strong literature support for data utility. Long-term data sets exist in all domains of California Current, but need to fill gaps in Oregon and some areas of Washington. Applies to breeding residents; data for not non-breeding residents or migratory species must come from studies external to the California Current. (WJS) |

| CONSIDERATIONS | | | | | |
|----------------------|--|-------------|----------|-----------|---|
| Attribute | Indicator | Primary (5) | Data (7) | Other (6) | Summary comments |
| Population condition | (3) Counts, identification of mortality, morality agents | 4 | 6 | 4.5 | Required to assess population risk and suggest management actions for population recovery. Includes predators, disease, pathogens, parasites, contaminants/pollution, starvation, collisions, senescence. Mortality often highly visible to public. Long-term data sets exist for certain types of mortality: beach-cast birds, fisheries bycatch for all geographic domains. (TPG) |
| Population condition | (4) Diet | 4 | 5.5 | 4 | Necessary to link seabirds to food web components supporting seabird populations. Strong literature support for data utility, but most data sets examine chick diet, not adult or non-breeder diet. Geographic gaps in diet information for Oregon, Washington. (JEZ) |
| Population size | (5) Counts, identification of birds at sea | 4.5 | 5.5 | 3.5 | Essential indicator, demonstrated literature support for utility. Necessary to obtain information on non-breeding residents and migratory species. Also provides information on habitat use at sea. (JEZ) |

| CONSIDERATIONS | | | | | |
|----------------------|---|-------------|----------|-----------|---|
| Attribute | Indicator | Primary (5) | Data (7) | Other (6) | Summary comments |
| Population condition | (6) Contaminant loads | 4 | 5.5 | 3.5 | Useful to understand health of individuals, populations, and food web. However, not often collected, historical sampling across many species is missing, requires additional diet information to interpret. (TPG) |
| Population size | (7) Counts, identification of birds at colonies (breeding populations only) | 4 | 5.5 | 3.5 | Essential indicator, demonstrated literature support for utility. Can include on-colony counts as well as aerial surveys. Historical data from most areas available. However, considered redundant with (2) because that data type includes counts of breeding pairs as a component of annual reproductive performance. (TPG) |
| Population condition | (8) Multivariate seabird index | 4 | 5 | 2.5 | Requires other indicator data to be collected for meta-analysis. Not widely applied, but successful when applied. Non-intuitive for public application. (WJS) |

| CONSIDERATIONS | | | | | |
|-----------------------------|---|-------------|----------|-----------|---|
| Attribute | Indicator | Primary (5) | Data (7) | Other (6) | Summary comments |
| Population size | (9) Counts, identification of shorebird species in coastal habitats | 3.5 | 6 | 2.5 | Responses in shorebirds may be primarily due to local land-use practices rather than changes in the California Current large marine ecosystem <i>per se</i> . However, ocean climate change could affect food resources and available habitat for shorebirds. (TPG) |
| Population condition | (10) Survival rates, other demographic variables | 3.5 | 4 | 3.5 | Trends and responses in this indicator alone difficult to attribute to specific ecosystem change or management without context of data from other indicators. (WJS) |
| Population size & condition | (11) Metapopulation structure/dynamics | 3 | 4 | 3.5 | Includes both mark-recapture techniques and newer molecular techniques to examine population size, mixing, and migration. Few species have comprehensive information on metapopulation structure in California Current, but can be important for conservation applications. (JEZ) |

| CONSIDERATIONS | | | | | |
|----------------------|----------------------|-------------|----------|-----------|--|
| Attribute | Indicator | Primary (5) | Data (7) | Other (6) | Summary comments |
| Population condition | (12) Stress hormones | 3 | 1 | 2 | Relatively new in application to seabirds. Powerful for looking at individual responses to starvation or disturbance, but has not been scaled up to examine population level responses. Stress responses may occur on too short of a time scale to be useful as ecosystem indicator. (WJS) |

Table SB2. Potential data sets available for contemporary seabird indicator data.

| | Program title | Institution responsible for contemporary data collection | Indicator data type | Area of coverage | Data contact |
|---|---|--|--|--|--|
| 1 | California Current Cetacean and Ecosystem Assessment Surveys | NOAA - Southwest Fisheries Science Center – Protected Resources Division | Habitat use at sea | California, Oregon, Washington | Lisa Ballance – lisa.ballance@noaa.gov |
| 2 | California Cooperative Oceanic Fisheries Investigations (CalCOFI) | Farallon Institute & PRBO Conservation Science | Habitat use at sea | Southern California | William Sydeman – wsydeman@faralloninstitute.rc |
| 3 | Mediterranean Coast Network | Channel Islands National Park | Reproductive performance Breeding colony counts Diet | Channel Islands, California | Russell Galipeau – 1-805-658-5700 |
| 4 | Applied California Current Ecosystem Studies (ACCESS) | PRBO Conservation Science | Habitat use at sea | Cordell Bank, Gulf of the Farallones, Monterey Bay National Marine Sanctuaries | Jaime Jahncke – jjahncke@prbo.org |
| 5 | PRBO Seabird Monitoring on the Farallon Islands | PRBO Conservation Science | Reproductive performance Breeding colony counts | Farallon Islands, California | Jaime Jahncke – jjahncke@prbo.org |

| | Program title | Institution responsible for contemporary data collection | Indicator data type | Area of coverage | Data contact |
|----|---|---|--|--|--|
| | | | Diet | | |
| 6 | NOAA Fisheries Rockfish Surveys | Farallon Institute & PRBO Conservation Science | Habitat use at sea | Central California | William Sydeman – wsydeman@faralloninstitute.or |
| 7 | At Sea Marbled Murrelet Population Monitoring | Crescent Coastal Research | Habitat use at sea | Northern California to Northern Oregon | Craig Strong – cstrong.ccr@charter.net |
| 8 | Ocean Salmon Ecosystem Survey | NOAA –Southwest Fisheries Science Center – Fisheries Ecology Division | Habitat use at sea | Newport, OR to San Francisco, CA | Sean Hayes – sean.hayes@noaa.gov |
| 9 | Beach COMBERS | Moss Landing Marine Laboratories | Mortality – beaches | Central California | Hannah Nevins – hnevins@mlml.calstate.edu |
| 10 | Seabird Bycatch | NOAA – Northwest Fisheries Science Center – Fishery Resource Analysis and Monitoring Division | Mortality –bycatch in commercial fisheries | California, Oregon, Washington | Janell Majewski – janell.majewski@noaa.gov |
| 11 | Pacific Continental Shelf Environmental Assessment (PaCSEA) | USGS – Western Ecological Research Center | Habitat use at sea | Fort Bragg, CA to Grays Harbor, WA | Josh Adams – josh_adams@usgs.gov |

| | Program title | Institution responsible for contemporary data collection | Indicator data type | Area of coverage | Data contact |
|----|---|---|--|----------------------------------|---|
| 12 | Catalog of Oregon Seabird Colonies | USFWS – Newport Office | Breeding colony counts | Oregon | Roy Lowe – roy_lowe@fws.gov |
| 13 | Yaquina Head Seabird Study | Oregon State University | Habitat use at sea Reproductive performance Diet | Central Oregon | Robert Suryan – rob.suryan@oregonstate.edu |
| 14 | Ocean Salmon Ecosystem Survey | NOAA – Northwest Fisheries Science Center – Fish Ecology Division | Habitat use at sea Diet | Newport, OR to Cape Flattery, WA | Jeannette Zamon – jen.zamon@noaa.gov |
| 15 | Columbia River Avian Predation Project | Oregon State University | Reproductive performance Diet Breeding colony counts | East Sand Island, OR | Daniel Roby – daniel.robby@oregonstate.edu |
| 16 | Marine Bird and Mammal Surveys | NOAA – Northwest Fisheries Science Center – Fish Ecology Division | Habitat use at sea | North Head, WA | Jeannette Zamon – jen.zamon@noaa.gov |
| 17 | At Sea Marbled Murrelet Population Monitoring | Washington Department of Fish and Wildlife | Habitat use at sea Reproductive performance | Washington | Scott Pearson – scott.pearson@dfw.wa.gov |

| | Program title | Institution responsible for contemporary data collection | Indicator data type | Area of coverage | Data contact |
|----|---|---|----------------------------|---------------------------------------|--|
| | | | Diet | | |
| 18 | Coastal Observation and Seabird Survey Team | University of Washington | Mortality – beaches | Washington, Oregon, California | Julia Parrish – jparrish@u.washington.edu |
| 19 | Pelagic Seabird Surveys | NOAA – Olympic Coast National Marine Sanctuary | Habitat use at sea | Grays Harbor, WA to Cape Flattery, WA | Liam Antrim – liam.antrim@noaa.gov |

Table SB3. Indicators used to examine Status and Trends data.

| Attribute | Indicator | Definition and source of data | Time series | Sampling frequency |
|-----------------------------|--------------------------------------|---|--------------------|------------------------------------|
| Population size & condition | Habitat use at sea – northern domain | Northwest Fisheries Science Center Ocean Salmon Ecosystem surveys, from Newport, OR (44°40'N) to the Washington-British Columbia border (48°13'N). Strip-transect surveys of seabird distribution and abundance from NOAA-chartered research vessels. | 2003 – 2012 | May, Jun surveys annually |
| | Habitat use at sea – southern domain | California Cooperative Oceanic Fisheries Investigations surveys, transects from San Diego, CA (30°N) to Point Conception, CA (35°N). | 1987 – present | Jan/Feb, Apr, Jul surveys annually |

STATUS AND TRENDS

MAJOR FINDINGS

The CCLME seabird community consists of over 75 species of seabirds, the composition of which changes seasonally and includes breeding residents, nonbreeding residents that reside in the CCLME habitat for several months during their nonbreeding season, and migratory species which transit relatively rapidly through CCLME habitat during spring and fall migrations (e.g. Ainley and Hyrenbach (2010), Ford et al. (2004)). To measure status and trends in seabird populations, it is necessary to have time series which measure seabird indicator data for the last five years (2007-2011) as well as for earlier years from which a long-term mean can be calculated. Measures such as bird density (birds per km²) should track population trends over time. Measures such as diet would track whether or not the food resources supporting seabird populations are changing over time.

Unlike fish or mammal data sets required by NOAA Fisheries for annual stock assessments, most seabird indicator data sets are collected by many different institutions or individuals. Seabird programs frequently depend on funding from a variety of sources to support research or maintain time series because few data sets have long-term funding necessary to maintain relatively unbroken time series of seabird indicators. These circumstances make it challenging to maintain, integrate, and synthesize data sets required to track ecosystem trends and responses (e.g. Ford et al. 2004).

Literature searches and communications with professional contacts for the 2012 IEA process revealed that at least 19 sources of historical seabird indicator data, with accompanying contemporary data, are potentially available from all three biogeographic domains within the CCLME (i.e. southern, central, and northern; Table SB2). Reviews of status and trends for time series of habitat use at sea, annual reproductive performance, and diet exist for pre-2010 data sets, with at least one indicator reviewed in each CCLME domain (Hyrenbach and Veit 2003, Gaston et al. 2009, Sydeman et al. 2009, Ainley and Hyrenbach 2010, Cury et al. 2011). However, recent data necessary for the 2012 IEA process (2007-2011) were often not publicly or readily available in the format required for data processing to examine trends for the last five years.

Given the situation with contemporary data for seabird indicator variables, it was beyond the scope of 2012 IEA resources to secure access to, examine, and synthesize all potential indicator data sets. We recommend that support for various institutions to contribute to synthesis of as many indicator data sets as possible be made available in the future.

We were, however, able to examine sample data sets for “At sea habitat use” which were immediately available for the IEA process through two of our seabird subgroup members (JEZ – Ocean Salmon Ecology data, WJS – CalCOFI data). These data sources are summarized briefly in Table SB3.

SUMMARY OF STATUS AND TRENDS FOR SAMPLE INDICATOR TIME SERIES

SAMPLE INDICATOR TIME SERIES

Data from two long term studies of seabird habitat use at sea were available for inclusion in the 2012 IEA. These data are being collected as part of ecosystem studies in the northern CCLME (Ocean Ecosystem Surveys - <http://www.nwfsc.noaa.gov/research/divisions/fed/oeip/a-ecinhome.cfm>); and southern CCLME (California Cooperative Oceanic Fisheries Investigations (CalCOFI) - <http://www.calcofi.org/>). These data provide a first look at the most recent abundance trends for representative seabird species at sea which have been examined in peer-reviewed literature.

Data collected during both studies were derived from counts of all birds seen within a 300-m wide strip while the research vessel was underway (for detailed methodology, see Tasker et al. (1984) and Heinemann (1981); therefore, data include breeding residents, nonbreeding residents, and migratory populations. These counts were converted to mean densities of birds per km⁻² for each annual cruise. Data were log(x+1) transformed to normalize the data distribution and assist in visualization of short- and long-term trends.

SEABIRD ABUNDANCE TRENDS IN NORTHERN AND SOUTHERN CCLME.

Even within this sample data set, dozens of seabird species were available for examination. We chose to present data from three seabird species common to all three CCLME domains, and where population density at sea has been already examined in peer-reviewed literature. Those species are common murres (*Uria aalge*), sooty shearwaters (*Puffinus griseus*), and Cassin's auklet (*Ptychoramphus aleuticus*).

During spring and summer, the two numerically dominant seabirds on the continental shelf in all domains of the CCLME are common murres and sooty shearwaters (southern domain: Hyrenbach and Veit (2003); central domain: Ainley and Hyrenbach (2010); northern domain: Zamon et al. (2013), Ainley et al. (2009)). Murres are breeding residents in the CCLME, whereas the shearwaters migrate from the southern hemisphere to the CCLME during their austral winter before returning south in October to breed in Chile and New Zealand. Murres and shearwaters are piscivorous divers, feeding on coastal pelagic species such as anchovy (*Engraulis mordax*), smelt (Osmeridae), sandlance (*Ammodytes hexapterus*), herring (*Clupea pallasii*) and sardine (*Sardinops sagax*), but they will also occasionally consume krill (Euphausiidae) or other invertebrates (e.g. gammarid amphipods). Cassin's auklet is a breeding resident commonly found in all domains of the CCLME, but Cassin's auklets are planktivorous shallow-diving birds, and therefore depend on a different trophic level (krill and plankton) than murres and shearwaters (coastal pelagic fishes). Time series plots of at-sea densities for these three species are shown in Figures SB1, SB2, and SB3.

The long-term average density of murres was greater in the northern domain of the CCLME, which is what one would expect given that murres are considered to be associated with colder water masses (Figure SB1, c.f. Hyrenbach and Veit (2003)). Both domains showed increasing or stable densities. This pattern of stable or increasing densities in murres is similar to that seen in the central domain for a less recent time period (Ainley and Hyrenbach 2010).

The long-term average density of shearwaters was similar in both northern and southern domains, and showed neither an upward nor downward trend in this data set (Figure SB2). This contrasts with results from several prior studies of less recent data, which all showed downward trends in shearwater abundance from the central and southern CCLME domains (Veit et al. 1996, Hyrenbach and Veit 2003, Ainley and Hyrenbach 2010). The change in trends for sooty shearwaters could be due to changes in shearwater distribution within the CCLME (e.g. a shift to the northern domain), changes in productivity of coastal pelagic fishes (e.g. McClatchie et al. this report; Brodeur et al. (2005)), changes in shearwater reproductive productivity in the southern hemisphere (Lyver et al. 1999), or a combination of all three factors. However, it is clear there has been change in the previous trend of decline in the CCLME.

Both densities and trends for Cassin's auklet density were highly variable (Figure SB3). There appears to be an increasing trend in auklet abundance for the northern CCLME, but decreasing or stable trends for auklets in the southern CCLME. A lack of increasing trends in the southern CCLME is consistent with observations and inferences from earlier years attributing declines in this species to shifts in ocean plankton production associated with ocean warming in central and southern CCLME domains (Hyrenbach and Veit 2003, Ainley and Hyrenbach 2010).

Although we did not have access to data from time series for the central CCLME, Ainley and Hyrenbach (2010) recently published an analysis of data from that region. They observed somewhat similar patterns in the older data: declines in murres and shearwaters followed by an apparent return to higher densities in 2005 and 2006, but historically low densities of Cassin's auklets.

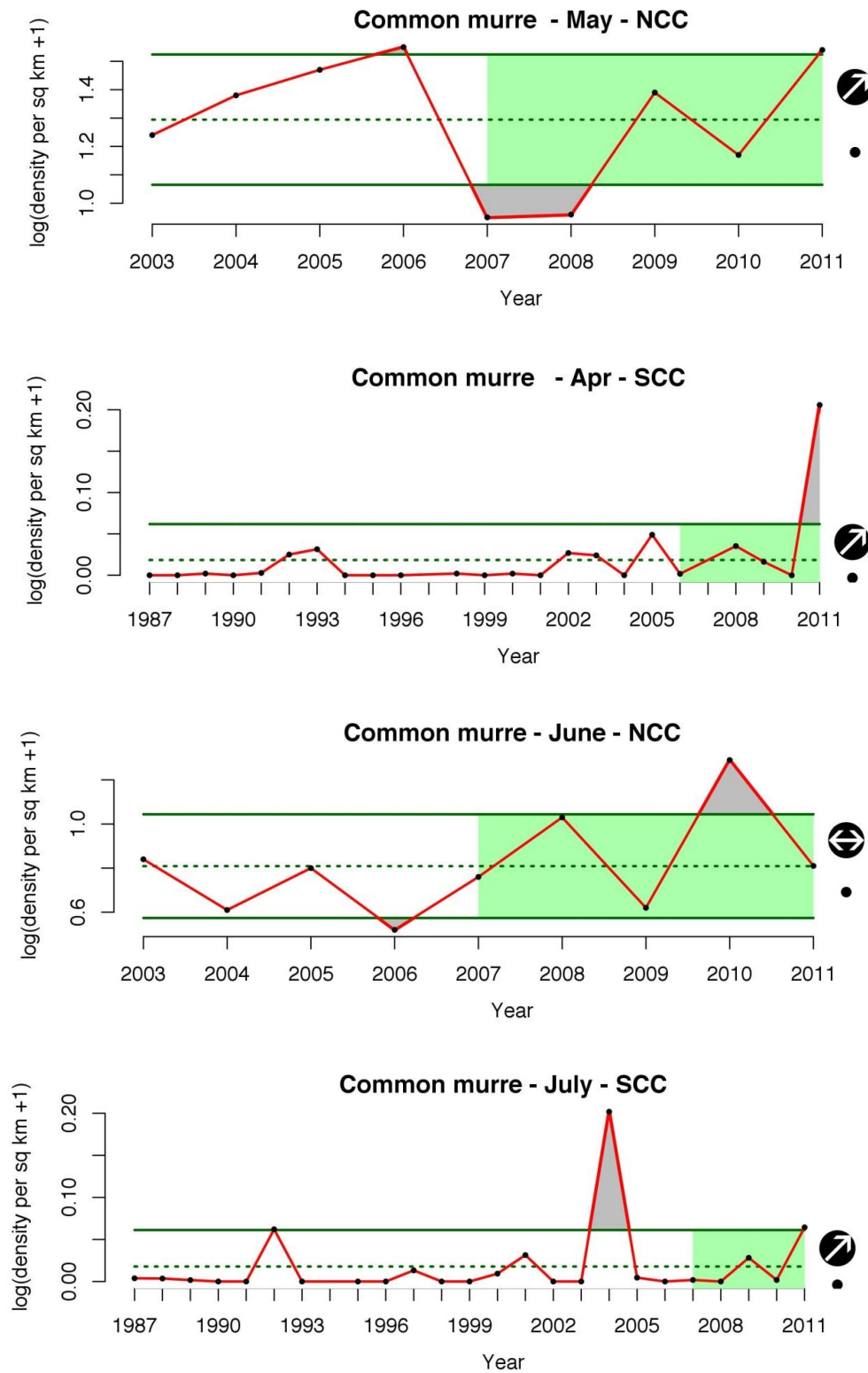


Figure SB1. Trends in seabird density (birds km²) over time for common murres in the northern (NCC) and southern (SCC) domains of the California Current

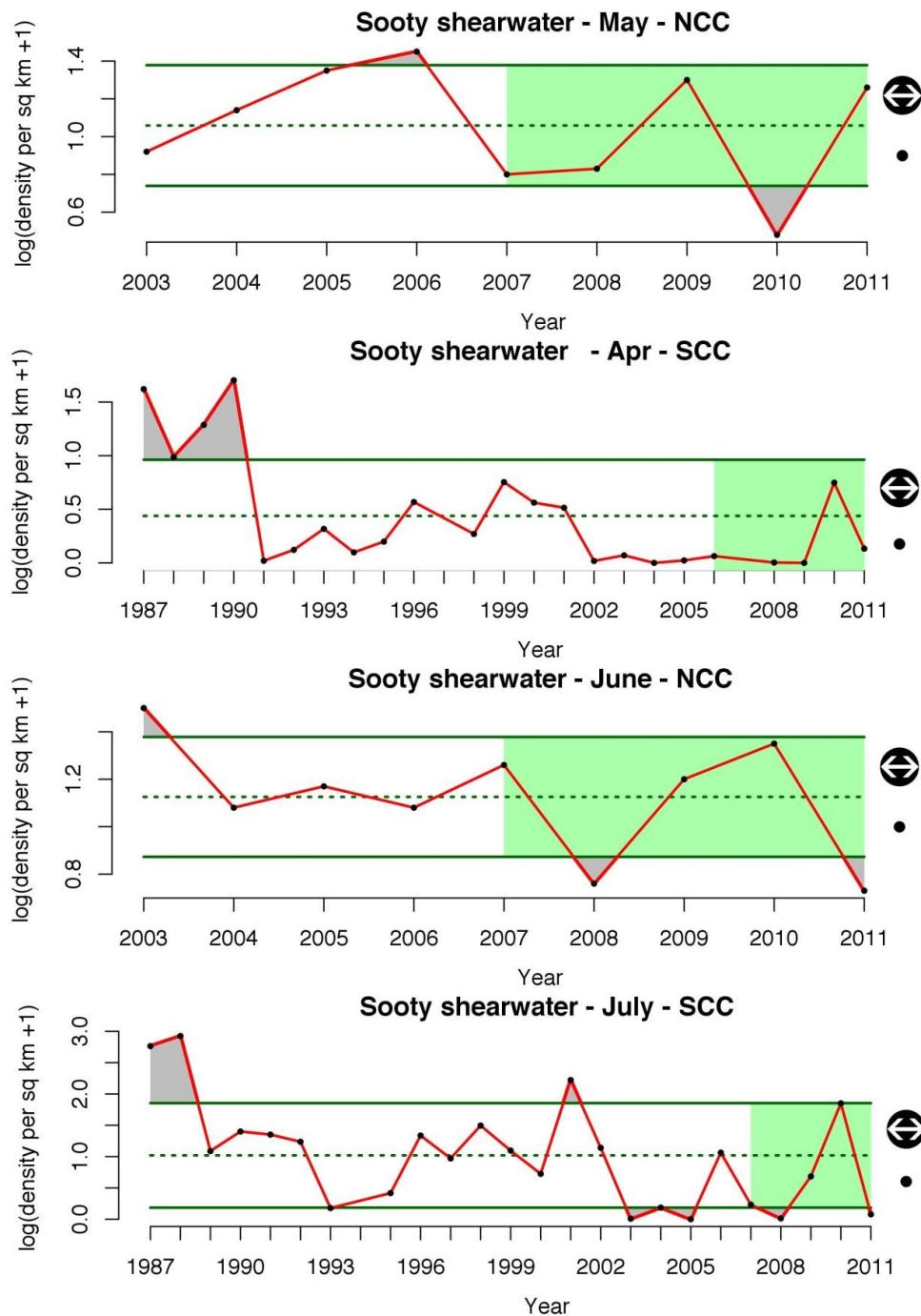


Figure SB2. Trends in seabird density (birds km²) overtime for sooty shearwaters in the northern (NCC) and southern (SCC) domains of the California Current.

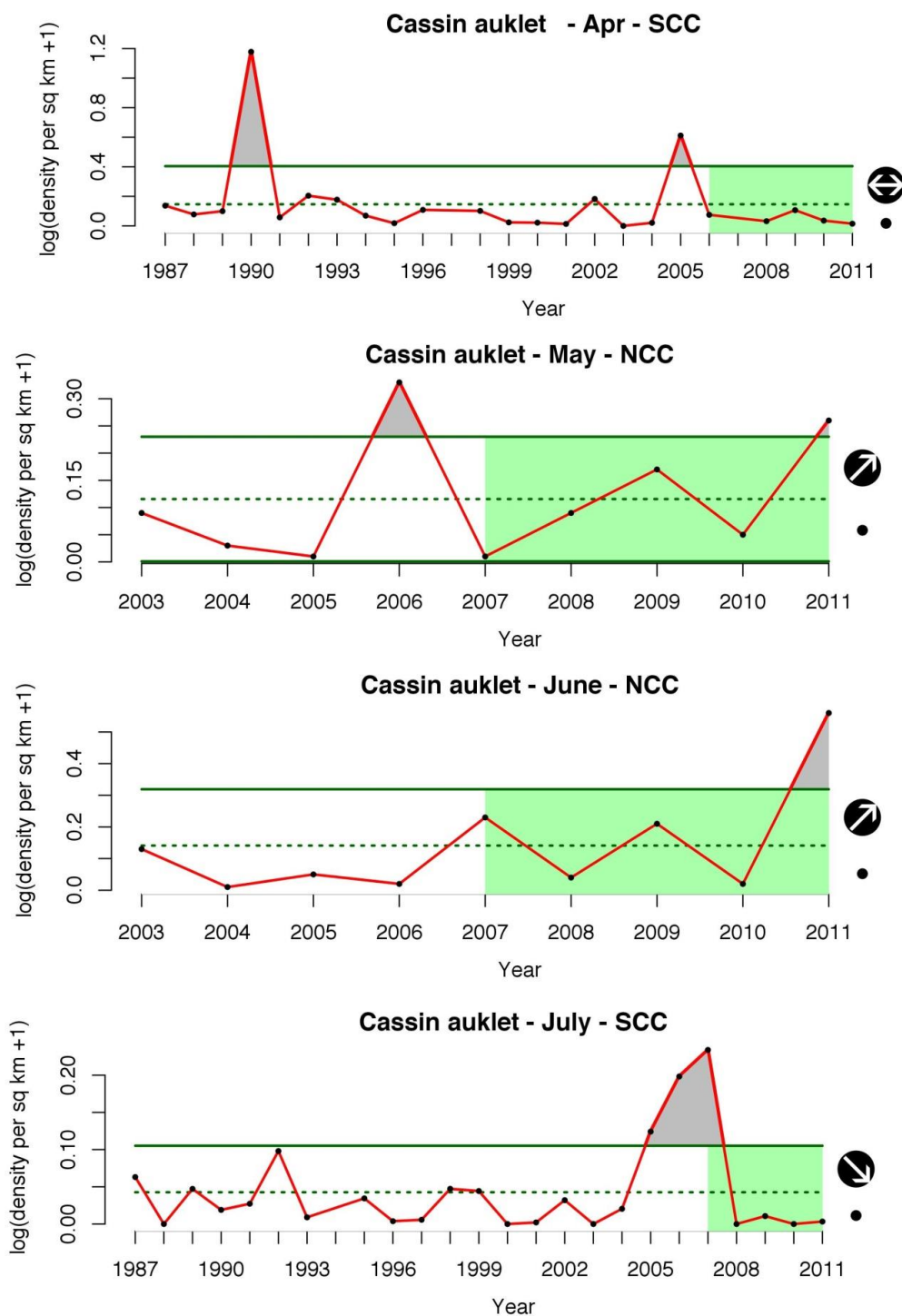


Figure SB3. Trends in seabird density (birds km²) over time for Cassin's auklet in the northern (NCC) and southern (SCC) domains of the California Current

RISK ASSESSMENT – SEABIRDS (FUTURE WORK)

Risk factors known to include at least the following:

- habitat reduction or disturbance on land (breeding birds)
- predation at colonies (breeding birds)
- commercial fishing
 - direct impacts of mortality as bycatch
 - indirect impacts to recruitment
- climate change effects on food web
 - prey species composition
 - timing, duration of prey species productivity
 - harmful algal blooms
 - direct effects of injury/mortality due to removal of waterproofing from plumage
 - indirect effects of bioaccumulation of toxins in prey
- pollution/contaminants/oil spills
- ocean energy development
 - direct effects of collisions, entanglement
 - indirect effects on prey distribution or food web structure

DATA LINKS

SEABIRD INDICATOR DATA USED IN FIGURES

Data credits for this document are as follows:

CCLME northern domain

- Ocean Ecosystem Survey, 2003-present
- NOAA Fisheries - Northwest Fisheries Science Center, Fish Ecology Division.
- This is a multi-investigator ecosystem survey for which seabird data were added as an ecosystem component in 2003. The original time series began in 1998 in response to collapse of Pacific salmon populations in the Columbia River system. The historical purpose of these surveys has been to understand how variation in physical and biological ecosystem components affects early marine growth and survival of juvenile salmon.
- Links to ocean program pages: <http://www.nwfsc.noaa.gov/research/divisions/fed/estuarine.cfm>
- Seabird data contact: Jeannette E. Zamon, jen.zamon@noaa.gov, 503-861-1818 x19

CCLME southern domain

- California Cooperative Oceanic Fisheries Investigations (CalCOFI), 1987-present
- This is a multi-agency, cooperative effort among NOAA Fisheries – Southwest Fisheries Science Center, Scripps Institution for Oceanography, and the California Department of Fish and Game. Seabird data were added as an ecosystem component in 1987. The original time series began in 1949 in response to the collapse of the sardine fishery in California. The historical purpose of these surveys has been to understand how variation in physical and biological ecosystem components affect recruitment processes for sardine and anchovy.
- Link to CalCOFI home page: <http://www.calcofi.org>
- Link to underway observation data pages: <http://www.calcofi.org/field-program/field-under.html>
- Seabird data contact: William J. Sydeman, wsydeman@faralloninstitute.org, 707-478-1381

CONTEMPORARY SEABIRD INDICATOR DATA SOURCES

Existing data sets which can contribute to the IEA are independently maintained by multiple agencies, institutions, and individuals. It was beyond the scope of the 2012 IEA effort to request, screen, format, and integrate data from all of these data sets. However, to facilitate future integration and synthesis of seabird indicators for the CCLME IEA, we provide a list of data programs and contacts presently collecting multi-year indicator data in the CCLME in Table SB2.

The majority of these programs do not have the resources to maintain continuous, long-term time series, but many have information from the past five years or access to discontinuous historical data which might be used for evaluating changes in seabird indicators.

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MARINE MAMMALS – INDICATORS AND STATUS

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TABLE OF CONTENTS

| | |
|-----------------------------------|-----|
| Executive summary..... | 342 |
| Detailed report..... | 344 |
| Indicator selection process | 344 |
| Status and trends..... | 362 |
| Major findings..... | 362 |
| Summary and status of trends..... | 363 |
| Literature Cited | 368 |

LIST OF TABLES AND FIGURES (MM)

| | |
|--|-----|
| Figure MM0. Spatially explicit risk assessments for marine mammals are an active area of research in the Marine Mammal and Turtle Division of the Southwest Fisheries Science Center. For example, Redfern et al. (2013) assessed the risk of ships striking large whales in the Southern California Bight. Whale densities predicted by habitat models are shown with the alternative shipping routes considered in the analyses. | 343 |
| Figure MM1. Ship survey estimates of abundance for short-beaked common dolphins, Dall’s porpoise, blue whales, humpback whales, and fin whales. | 365 |
| Figure MM2. Gray whale abundance estimates are shown with 95% log-normal confidence intervals. | 365 |
| Figure MM3. California sea lion pup abundance estimates. | 366 |

OVERVIEW

New quantitative tools are being used to improve population assessments of marine mammals. Recent assessments have shown strong evidence for increasing fin whale abundance in the California Current (Moore and Barlow 2011), an increase in the Eastern North Pacific Gray whale population (Punt and Wade 2010), and increases in California sea lion pup production (Carretta et al. 2011). In the California Current ecosystem, large-scale marine mammal assessment surveys have only been conducted 5 times in the last 20 years. To understand trends in marine mammal abundance, it is critical to continue conducting assessment surveys and monitoring as the populations respond to changes in their ecosystem.

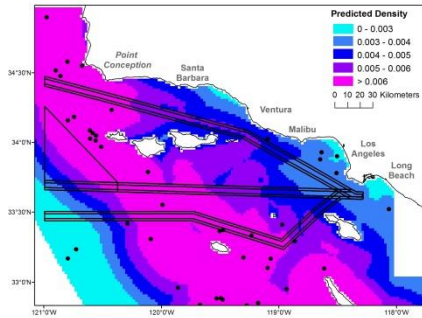
EXECUTIVE SUMMARY

The California Current supports a rich and diverse marine mammal community, including at least 30 cetacean species, 6 pinniped species, and one species of marine otter. For the integrated ecosystem assessment, we selected the following focal species: 1. Four species of baleen whales, 2. Dall's porpoise, 3. Short-beaked common dolphins, 4. Coastal bottlenose dolphins, 5. Resident and transient killer whales, 6. California sea lions. We selected these focal species to ensure representation of different trophic levels, representation of species with high estimated annual consumption levels, representation of species that are known to respond to changes in environmental conditions, and representation of species with restricted distributions that make them vulnerable to human activities.

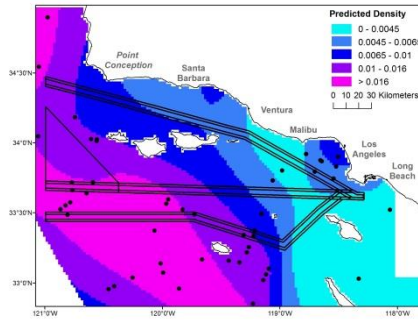
The trend assessment approach used in other sections of this report is inappropriate for assessing trends in marine mammal abundance for several reasons. First, the approach is only useful for identifying short-term trends that depart from a long-term stationary process (i.e., no long-term increase or decline in abundance). Marine mammal surveys in recent decades coincide with a period of expected long-term increases in many pinniped and large whale populations, as they recover from severe depletion following historical hunting. Second, the approach assumes that the abundance estimates are independently and identically distributed (*iid*) with constant variance. For marine mammals, the *iid* assumption is not met; consequently, an individual abundance point estimate falling outside of the long-term standard deviation cannot be interpreted as being anomalously high or low (i.e., does not constitute a basis for inferring trend). Finally, five years is not a reasonable timescale over which to evaluate trends in marine mammal abundance in the CCLME. Marine mammal abundance estimates are generally imprecise and these species generally have low population growth rates. Hence, even a relatively rapid change in total population abundance (e.g., $> \pm 5\%$ annually or 50% within 15 years) is extremely unlikely to be detected within a five-year time frame, especially when based on so few surveys (Taylor et al. 2007).

Assessment of marine mammal trends is best done over longer (e.g., decadal) time periods and using methods more appropriate for handling the issues described above. These issues highlight the importance of regularly conducting marine mammal assessment surveys in the California Current Ecosystem. In this report, we highlight research that has been conducted to explore variability in the abundance estimates and assess long-term trends abundance.

a) Blue whales



b) Fin whales



c) Humpback whales

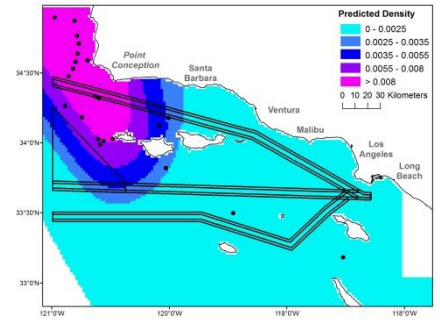


Figure MM0. Spatially explicit risk assessments for marine mammals are an active area of research in the Marine Mammal and Turtle Division of the Southwest Fisheries Science Center. For example, Redfern et al. (2013) assessed the risk of ships striking large whales in the Southern California Bight. Whale densities predicted by habitat models are shown with the alternative shipping routes considered in the analyses.

DETAILED REPORT

We evaluate the current, potential, and needed indicators representing variability in the mammal populations of the CCLME. Largely, our purpose is to identify currently available data, discuss current efforts that will be useful for future analyses, and identify gaps in marine mammal monitoring.

INDICATOR SELECTION PROCESS

INDICATOR EVALUATION

FOCAL SPECIES

The California Current supports a rich and diverse marine mammal community, including at least 30 cetacean species, 6 pinniped species, and one species of marine otter. For the integrated ecosystem assessment, we have selected the following focal species:

- Four species of baleen whales: blue (*Balaenoptera musculus*), humpback (*Megaptera novaeangliae*), fin (*B. physalus*), and gray whales (*Eschrichtius robustus*)
- Dall's porpoise (*Phocoenoides dalli*)
- Short-beaked common dolphins (*Delphinus delphis*)
- Coastal bottlenose dolphins (*Tursiops truncatus*)
- Resident and Transient killer whales (*Orcinus orca*)
- California sea lions (*Zalophus californianus*)

Indicators of population abundance and condition (see the sections below) have been selected for each focal species according the protocols outlined in (Kershner et al. 2011).

Criteria for selecting these focal species included ensuring representation of different trophic levels, representation of species having high estimated annual consumption levels, representation of species that are known to respond to changes in environmental conditions, and representation of species that have restricted distributions that make them vulnerable to human activities. Transient eco-type killer whales were selected because they prey on other marine mammals, including pinnipeds and cetaceans (e.g., large whale calves). Southern Resident killer whales are thought to be at risk from multiple human activities (Krahn et al. 2004). Where possible, we treat Resident and Transient killer whales separately because a recent workshop suggested that these ecotypes are likely to be at least a separate subspecies (Reeves et al. 2004); others have suggested that Residents and Transients are full species (Morin et al. 2010).

Barlow et al. (2008) estimated prey consumption for cetaceans in the California Current. Species having the highest annual consumption estimates were short-beaked common dolphins, fin whales, blue whales, sperm whales, humpback whales, and Dall's porpoise. All species except sperm whales are included as focal species. Sperm whales were excluded because the time series of abundance for this species has a relatively high coefficient of variance (Barlow and Forney 2007); therefore there is in a low signal-to-noise ratio for determining trends.

Although Dall's porpoise also have a high coefficient of variation (Barlow and Forney 2007), they are a cold-temperate species, and provide an interesting contrast to short-beaked common dolphins, considered

a tropical and warm-temperate species. Both species are distributed widely throughout the eastern north Pacific. Forney (2000) showed that changes in their abundance in the California Current reflected patterns in sea surface temperature.

The California Current is an important, seasonal feeding area for humpback and blue whales (Calambokidis et al. 2001, Calambokidis et al. 2009). Fin whales are present in the California Current throughout the year, but had higher abundances during the summer (Forney et al. 1995). Gray whales do not commonly feed in the California Current. A small number (100s) of whales called the “Pacific Coast Feeding Group” feed along the Pacific coast, however, between southern British Columbia and northern California during the summer feeding period (Calambokidis et al. 2010). Although gray whale abundance and condition are largely influenced by environmental variability on the Arctic feeding grounds (Moore 2008), coastal waters of the California Current serve as the migration corridor for gray whales in the eastern North Pacific. During their high (Arctic) to low (Baja California, Mexico) latitude round-trip migrations, gray whales are at risk from both ship strikes and fisheries entanglements (International Whaling Commission 2011). Transient killer whales have also been observed to prey on gray whale calves during this migration (Barrett-Lennard et al. 2011).

California sea lions of all age/sex classes are accessible on land, making them a cost-effective group of marine mammal species to include in the IEA. There is a long history demonstrating linkages between population parameters for California sea lions and ENSO events, including pup and yearling survival (DeLong et al. *In prep.*), natality (Melin et al. 2012a), and pup production (Lowry and Maravilla-Chavez 2005). Melin et al. (2010) also demonstrated linkages between upwelling and pup mortality during the 2009 oceanographic event in Central California. Studies have also explored the diets of California sea lions and linked diet to abundances of their prey (Lowry 1999), which include several commercial species: Pacific hake, market squid, Pacific sardine, northern anchovy, shortbelly rockfish, Pacific mackerel, and jack mackerel. Finally, studies have also shown a relationship between *Leptospirosis* disease and male survival (DeLong et al. *In prep.*) and impacts of man-made pollution on populations (Ylitalo et al. 2005).

The bottlenose dolphin is the most common cetacean in nearshore waters off California. This population, estimated to contain less than 500 individuals, has been under nearly continuous scientific study by researchers at San Diego State University (Defran and Weller 1999, Defran et al. 1999, Dudzik et al. 2006) and the National Marine Fisheries Service (Carretta et al. 1998) since the early 1980s. Members of this population are nomadic, regularly traveling within a range that extends from Ensenada, Baja California, Mexico in the south to Monterey Bay, California in the north. Despite their pronounced coastal movements, these dolphins typically occur no further than 1 km offshore, and are most commonly found just outside of the breaking surf (Carretta et al. 1998, Defran and Weller 1999). The California coastal bottlenose dolphin stock is relatively small and in combination with its coastal distribution places it at risk from a variety of potential human-related threats including fisheries interactions and exposure to chemical contaminants and disease (Carretta et al. 2011).

INDICATORS OF POPULATION ABUNDANCE: SHIP SURVEY ABUNDANCE ESTIMATES FOR KILLER WHALES, BLUE WHALES, HUMPBACK WHALES, FIN WHALES, SHORT-BEAKED COMMON DOLPHINS, AND DALL’S PORPOISE

The primary indicators of the abundance of offshore killer whales, fin whales, short-beaked common dolphins, and Dall’s porpoise in the California Current are the cetacean and ecosystem assessment surveys conducted by the Southwest Fisheries Science Center. The abundance estimate for killer whales includes all three types (i.e., Resident, Transient, and Offshore) because the types cannot be reliably differentiated at sea.

In the U.S. Pacific Marine Mammal stock assessment reports, the abundance of offshore killer whales is obtained by multiplying the killer whale abundance estimate by the proportion of offshore individuals observed in photo-identification studies (see Carretta et al. 2011). Abundances of blue and humpback whales are also obtained during these surveys, but mark-recapture techniques provide more precise estimates of their abundance (see below). However, all habitat models and spatially-explicit risk assessments for blue and humpback whales have been conducted with the survey data.

These surveys occur in U.S. West Coast waters out to a distance of approximately 300 nautical miles. The primary objectives of the surveys are to estimate the abundance and to understand the distribution of dolphins and whales which are commonly found off of the west coast. A secondary objective is to characterize the pelagic ecosystem within the study area, through the collection of underway and station-based physical and biological oceanographic sampling, studies of mid-trophic level organisms (using net sampling and acoustic backscatter methods) and research on non-protected apex predators (seabirds). A final objective is to conduct biopsy sampling and photo-identification studies of cetacean species of special interest.

Peer-reviewed literature or usage in a specific management application is available for each of the indicator selection categories, as listed below.

- Theoretically-sound
 - All stock assessments (e.g., Carretta et al. 2011)
 - (Barlow and Forney 2007)
- Relevant to management concerns
 - All stock assessments (e.g., Carretta et al. 2011)
- Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute
 - All stock assessments (e.g., Carretta et al. 2011)
- Responds predictably & is sufficiently sensitive to changes in a specific management action or pressure
 - All stock assessments (e.g., Carretta et al. 2011)
- Linkable to scientifically-defined reference points & progress targets
 - All stock assessments (e.g., Carretta et al. 2011)
- Concrete & Numerical
 - Visual counts, at-sea surveys (Barlow and Forney 2007, Moore and Barlow 2011)
- Historical data or information available
 - California (1991–2005) and in Oregon–Washington (1996–2005) (Barlow and Forney 2007, Moore and Barlow 2011)
- Operationally simple
 - Barlow and Forney (2007)
- Broad spatial coverage
 - Barlow and Forney (2007)
- Continuous time series
 - Conducted every 3-5 years (Barlow and Forney 2007)
- Spatial & temporal variation understood
 - Seasonal changes cannot be assessed because surveys are always conducted in the summer/fall. Interannual and longer-term variability can be assessed (Barlow and Forney 2007).
- High signal-to-noise ratio (Barlow and Forney 2007, Moore and Barlow 2011)
- Understood by the public & policymakers
 - All stock assessments (e.g., Carretta et al. 2011)

- History of reporting
 - All stock assessments (e.g., Carretta et al. 2011)
- Cost-effective
 - Surveys are expensive, but measure multiple components of the ecosystem and a majority of the California Current. All stock assessments (e.g., Carretta et al. 2011).
- Anticipatory or leading indicator
 - Can be used to forecast distribution and density (Becker et al. 2012, Forney et al. 2012)
- Regionally/nationally/internationally compatible
 - All stock assessments (e.g., Carretta et al. 2011).
- Other
 - Barlow et al (2008) suggest that the primary production requirement of cetaceans in the California Current is on the order of 12% of the net primary production

INDICATORS OF POPULATION ABUNDANCE: CENSUS OF SOUTHERN RESIDENT KILLER WHALES

An annual census for southern resident killer whales is conducted by the Center for Whale Research using photo-id documentation of all whales in the three pods, (Carretta et al. 2011). The census is conducted from small boats in the protected inland waters of Washington and British Columbia, which is the spring and early summer range of the whales.

Peer-reviewed literature or usage in a specific management application is available for each of the indicator selection categories, as listed below.

- Theoretically-sound
 - Stock assessment report for southern resident killer whales (Carretta et al. 2011)
- Relevant to management concerns
 - Stock assessment report for southern resident killer whales (Carretta et al. 2011)
 - NOAA ESA status review for southern resident killer whales
- Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute
 - Stock assessment report for southern resident killer whales (Carretta et al. 2011)
- Responds predictably & is sufficiently sensitive to changes in a specific management action or pressure
 - Stock assessment report for southern resident killer whales (Carretta et al. 2011)
- Linkable to scientifically-defined reference points & progress targets
 - Stock assessment report for southern resident killer whales (Carretta et al. 2011)
- Concrete & Numerical
 - Annual photo-id census by the Center for Whale Research (Carretta et al. 2011)
- Historical data or information available
 - 1974-2011 – entire range
- Operationally simple
 - Stock assessment report for southern resident killer whales (Carretta et al. 2011)
- Broad spatial coverage
 - Census conducted only in summer range
- Continuous time series
 - Census conducted annually

- Spatial & temporal variation understood
 - Census conducted only in summer range, limited information available for the occurrence within the winter range
- High signal-to-noise ratio
 - Stock assessment report for southern resident killer whales (Carretta et al. 2011)
- Understood by the public & policymakers
 - Stock assessment report for southern resident killer whales (Carretta et al. 2011)
- History of reporting
 - Stock assessment report for southern resident killer whales (Carretta et al. 2011)
- Cost-effective
 - Stock assessment report for southern resident killer whales (Carretta et al. 2011)
- Anticipatory or leading indicator
 - Can be used for population viability analyses. Stock assessment report for southern resident killer whales (Carretta et al. 2011)
- Regionally/nationally/internationally compatible
 - Stock assessment report for southern resident killer whales (Carretta et al. 2011)

INDICATORS OF POPULATION ABUNDANCE: MARK-RECAPTURE ABUNDANCE ESTIMATES FOR TRANSIENT KILLER WHALES

The population of the West Coast transient killer whale stock is periodically estimated by mark recapture using data from photo-id surveys (Allen and Angliss 2011).

Peer-reviewed literature or usage in a specific management application is available for each of the indicator selection categories, as listed below.

- Theoretically-sound
 - Stock assessment report for West Coast Transient killer whales (Carretta et al. 2011)
- Relevant to management concerns
 - Stock assessment report for West Coast Transient killer whales (Carretta et al. 2011)
- Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute
 - Stock assessment report for West Coast Transient killer whales (Carretta et al. 2011)
- Responds predictably & is sufficiently sensitive to changes in a specific management action or pressure
 - Stock assessment report for West Coast Transient killer whales (Carretta et al. 2011)
- Linkable to scientifically-defined reference points & progress targets
 - Stock assessment report for West Coast Transient killer whales (Carretta et al. 2011)
- Concrete & Numerical
 - Stock assessment report for West Coast Transient killer whales (Carretta et al. 2011)
- Historical data or information available
 - Southeast Alaska approximately 1991-1996, British Columbia approximately 1980-1995, California approximately 1986-1996, (Dahlheim et al. 1997, Ford et al. 2007)
- Operationally simple
 - Dahlheim et al. (1997)
- Broad spatial coverage
 - Dahlheim et al. (1997)

- Continuous time series
 - Surveys conducted opportunistically
 - Surveys are not range-wide
- Spatial & temporal variation understood
 - Seasonal changes cannot be assessed
- High signal-to-noise ratio
 - Ford et al. (2007)
- Understood by the public & policymakers
 - Stock assessment report for West Coast Transient killer whales (Carretta et al. 2011)
- History of reporting
 - Stock assessment report for West Coast Transient killer whales (Carretta et al. 2011)
- Cost-effective
 - Opportunistic surveys are relatively inexpensive
- Anticipatory or leading indicator
 - Annual changes in occurrence patterns in the Salish Sea can be used as an index of ecosystem change (Houghten et al. in prep)
- Regionally/nationally/internationally compatible.
 - Stock assessment report for West Coast Transient killer whales (Carretta et al. 2011)

INDICATORS OF POPULATION ABUNDANCE: MARK-RECAPTURE ABUNDANCE ESTIMATES OF BLUE AND HUMPBACK WHALES

Another indicator of abundance for blue and humpback whales is mark-recapture estimates based on photo-identification studies (Calambokidis and Barlow 2004). These studies are conducted primarily using inexpensive small boats operated from shore on day trips. Researchers concentrate sampling effort in areas of high reported whale density. Reports on aggregations of whales come from whale-watch vessels and a network of researchers and fishermen. Photographs have been taken for approximately 25 years of features that can be used to identify individuals of each species; information on movement patterns and life-history parameters is obtained from a time series of resightings of the identified individuals. The abundance of these species can be accurately estimated from photo-identification studies using mark-recapture analysis methods. Prior studies show that such estimates are more precise than line-transect estimates from large-ship surveys. For blue whales, however, the offshore component of the population cannot be sampled using day trips from shore and the identification photographs taken on large-ship surveys are a vital component of the mark-recapture abundance estimates for this species.

Peer-reviewed literature or usage in a specific management application is available for each of the indicator selection categories, as listed below.

- Theoretically-sound (peer-reviewed findings suggest it is a surrogate for ecosystem attributes)
 - Stock assessment reports for blue & humpback whales (e.g., Carretta et al. 2011)
 - NOAA ESA status review for humpback whales
 - (Calambokidis and Barlow 2004, Calambokidis et al. 2009, Barlow et al. 2011)
- Relevant to management concerns
 - Stock assessment reports for blue & humpback whales (e.g., Carretta et al. 2011)
 - NOAA ESA status review for humpback whales
- Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute
 - Stock assessment reports for blue & humpback whales (e.g., Carretta et al. 2011)

- Responds predictably & is sufficiently sensitive to changes in a specific management action or pressure
 - Stock assessment reports for blue & humpback whales (e.g., Carretta et al. 2011)
- Linkable to scientifically-defined reference points & progress targets
 - Stock assessment reports for blue & humpback whales (e.g., Carretta et al. 2011)
- Concrete & Numerical
 - Most precise estimates of abundance for these species
 - (Calambokidis and Barlow 2004, Calambokidis et al. 2009, Barlow et al. 2011)
- Historical data or information available
 - California (1988–2011) and in Oregon–Washington (1996–2011).
 - (Calambokidis and Barlow 2004, Calambokidis et al. 2009, Barlow et al. 2011)
- Operationally simple
 - (Calambokidis and Barlow 2004)
- Broad spatial coverage
 - (Calambokidis and Barlow 2004)
- Continuous time series
 - Conducted every year. Not entirely funded by NOAA.
- Spatial & temporal variation understood
 - Seasonal changes cannot be assessed. Interannual and longer-term variability can be assessed (Calambokidis and Barlow 2004).
- High signal-to-noise ratio
 - (Calambokidis and Barlow 2004)
- Understood by the public & policymakers
 - Stock assessment reports for blue & humpback whales (e.g., Carretta et al. 2011)
- History of reporting
 - Stock assessment reports for blue & humpback whales (e.g., Carretta et al. 2011)
- Cost-effective
 - Surveys are relatively inexpensive.
- Anticipatory or leading indicator
 - Annual changes in feeding preferences of humpback whales (as revealed by isotopes) can be used as index of ecosystem change (Fleming, *In prep.*).
- Regionally/nationally/internationally compatible
 - Stock assessment reports for blue & humpback whales (e.g., Carretta et al. 2011)
 - NOAA ESA status review for humpback whales

INDICATORS OF POPULATION ABUNDANCE: GRAY WHALES

The primary indicators of abundance for gray whales in the California Current are shore-based counts conducted by the National Marine Mammal Laboratory and the Southwest Fisheries Science Center. Abundance estimates for gray whales have been made for 23 years, between 1967 and 2007, from shore-based count data collected during the southbound migration past Granite Canyon, California (Laake et al. 2009). Mark-recapture estimators using photo-identification data are the primary indicators of abundance for Pacific Coast Feeding Group gray whales (Calambokidis et al. 2010).

Peer-reviewed literature or usage in a specific management application is available for the shore-based counts in each of the indicator selection categories, as listed below.

- Theoretically-sound
 - All stock assessments (e.g., Allen and Angliss 2011)
 - (Laake et al. 2009)
- Relevant to management concerns
 - All stock assessments (e.g., Allen and Angliss 2011)
- Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute
 - All stock assessments (e.g., Allen and Angliss 2011)
- Responds predictably & is sufficiently sensitive to changes in a specific management action or pressure
 - All stock assessments (e.g., Allen and Angliss 2011)
- Linkable to scientifically-defined reference points & progress targets
 - All stock assessments (e.g., Allen and Angliss 2011)
- Concrete & Numerical
 - Visual counts, shore-based (Laake et al. 2009, Punt and Wade 2010)
- Historical data or information available
 - Eastern North Pacific population (1967–2006) (Laake et al. 2009, Punt and Wade 2010)
- Operationally simple
 - Laake et al. (2009)
- Broad spatial coverage
 - Counts are conducted from a single location that occurs on the migration corridor used by nearly all individuals (Laake et al. 2009)
- Continuous time series
 - Counts have been conducted for 23 years at variable intervals (Laake et al. 2009)
- Spatial & temporal variation understood
 - Interannual and longer-term variability can be assessed (Laake et al. 2009)
- High signal-to-noise ratio
 - Laake et al. (2009), (Punt and Wade 2010)
- Understood by the public & policymakers
 - All stock assessments (e.g., Allen and Angliss 2011)
- History of reporting
 - All stock assessments (e.g., Allen and Angliss 2011)
- Cost-effective
 - Relatively inexpensive, shore-based surveys. All stock assessments (e.g., Allen and Angliss 2011)
- Anticipatory or leading indicator
 - Moore (2008)
- Regionally/nationally/internationally compatible
 - All stock assessments (e.g., Allen and Angliss 2011)
- Other
 - Gray whales are prey for transient killer whales (Barrett-Lennard et al. 2011)
 - Gray whales are at risk from ship strikes and fisheries entanglements when they migrate through the CCE (International Whaling Commission 2011)

INDICATORS OF POPULATION ABUNDANCE: CALIFORNIA SEA LIONS

The primary indicators of the abundance of California sea lions in the California Current are aerial surveys and ground counts of live pups conducted in July by the Southwest Fisheries Science Center and the Alaska Fisheries Science Center. Aerial surveys provide counts for all age/sex classes simultaneously; pups and other age/sex classes are counted from color photographs taken at rookeries and haulouts during aerial surveys of islands and the mainland coast of California. Ground counts of pups only provide an index of the population trend.

Peer-reviewed literature or usage in a specific management application is available for each of the indicator selection categories, as listed below.

- Theoretically-sound
 - All stock assessments (e.g., Carretta et al. 2011)
 - Lowry and Maravilla-Chavez (2005)
- Relevant to management concerns
 - All stock assessments (e.g., Carretta et al. 2011)
- Responds predictably & is sufficiently sensitive to changes in a specific marine ecosystem attribute
 - All stock assessments (e.g., Carretta et al. 2011)
- Responds predictably & is sufficiently sensitive to changes in a specific management action or pressure
 - All stock assessments (e.g., Carretta et al. 2011)
- Linkable to scientifically-defined reference points & progress targets
 - All stock assessments (e.g., Carretta et al. 2011)
- Concrete & Numerical
 - Counts from photographs taken during aerial surveys (Lowry and Maravilla-Chavez 2005)
- Historical data or information available
 - Count data are available from the late 1920's (Lowry and Maravilla-Chavez 2005)
- Operationally simple
 - Lowry and Maravilla-Chavez (2005)
- Broad spatial coverage
 - Covers all rookeries within the Channel Islands in southern California as well as rookeries in central California (Lowry and Maravilla-Chavez 2005)
- Continuous time series
 - Surveys conducted every third year or more frequently, if funding is available (Lowry and Maravilla-Chavez 2005)
- Spatial & temporal variation understood
 - Interannual and long-term changes in abundance can be assessed (Lowry and Maravilla-Chavez 2005)
- High signal to noise ratio
 - Lowry and Maravilla-Chavez (2005)
- Understood by the public & policymakers
 - California sea lions are well known to the public and policy makers due to their presence in aquaria and their frequent visits to land. They are a high profile species and a visitor attraction at places like San Francisco pier 39, sea lion caves in Oregon, and Ballard locks in Seattle.
 - Policymakers will be familiar with them due to concerns with fisheries interactions in the sports and commercial fishing industries along the California, Oregon, and Washington

coasts, interactions with endangered salmonids at the Ballard locks in Washington and more recently Bonneville Dam in Oregon where there is a limited removal of adult males that are consuming endangered salmon, and interactions with humans on piers and at boat marinas along the California coast.

- All stock assessments (e.g., Carretta et al. 2011)
- History of reporting
 - All stock assessments (e.g., Carretta et al. 2011)
- Cost-effective
 - While pinniped abundance surveys are conducted with aerial photography, the cost is quite small because large geographical areas can be sampled in a short time (land surveys only sample a small number of locations). All stock assessments (e.g., Carretta et al. 2011).
- Anticipatory or leading indicator
 - The documented linkages between California sea lions and environmental conditions are outlined in the section on focal species and are explored under population condition indicators. Numerous citations exist for these linkages.
- Regionally/nationally/internationally compatible
 - Pinnipeds have been used as indicator species in other ecosystems. A large and long-running program uses Antarctic fur seals at South Georgia, Antarctica as an indicator of ecosystem processes (Croxall et al. 1988, Boyd et al. 1994, Reid and Croxall 2001).
 - All stock assessments (e.g., Carretta et al. 2011).

INDICATORS OF POPULATION ABUNDANCE: COASTAL BOTTLENOSE DOLPHIN

The primary indicators of abundance for coastal bottlenose dolphins in the California Current are from photo-identification mark-recapture estimates derived from photo-identification data collected by San Diego State University and the Southwest Fisheries Science Center. The abundance of coastal bottlenose dolphins has been estimated for four multi-year periods between 1984 and 2005 (Defran and Weller 1999, Dudzik et al. 2006). Abundance has also been estimated by aerial surveys (Carretta et al. 1998).

Peer-reviewed literature or usage in a specific management application is available for each of the indicator selection categories, as listed below.

- Theoretically-sound
 - All stock assessments (e.g., Carretta et al. 2011)
 - Dudzik et al. (2006)
- Relevant to management concerns
 - All stock assessments (e.g., Carretta et al. 2011)
- Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute
 - All stock assessments (e.g., Carretta et al. 2011)
- Responds predictably & is sufficiently sensitive to changes in a specific management action or pressure
 - All stock assessments (e.g., Carretta et al. 2011)
- Linkable to scientifically-defined reference points & progress targets
 - All stock assessments (e.g., Carretta et al. 2011)
- Concrete & Numerical
 - Photo identification surveys conducted from small boats (Dudzik et al. 2006)
- Historical data or information available

- Abundance of coastal bottlenose dolphins has been estimated for four multi-year periods between 1984 and 2005 (Dudzik et al. 2006)
- Operationally simple
 - Dudzik et al. (2006)
- Broad spatial coverage
 - Dudzik et al. (2006)
- Continuous time series
 - Abundance of coastal bottlenose dolphins has been estimated for four multi-year periods between 1984 and 2005 (Dudzik et al. 2006). If possible, surveys are conducted annually.
- Spatial & temporal variation understood
 - Seasonal, interannual, and longer-term variability can be assessed (Dudzik et al. 2006)
- High signal-to-noise ratio
 - Dudzik et al. (2006)
- Understood by the public & policymakers
 - All stock assessments (e.g., Carretta et al. 2011)
- History of reporting
 - All stock assessments (e.g., Carretta et al. 2011)
- Cost-effective
 - Surveys are relatively inexpensive because they are conducted from a small boat. All stock assessments (e.g., Carretta et al. 2011).
- Anticipatory or leading indicator
 - Coastal bottlenose dolphins are a sentinel species (Levels of persistent organic pollutants in blubber of free-ranging bottlenose dolphins (*Tursiops truncatus*) off Southern California (In prep). David W. Weller, Gina M. Ylitalo, Nate Dodder, Nicholas Kellar, Gregory S. Campbell, Fionna Mattison, John Hyde, Aimee R. Lang, John A. Hildebrand and Wayne Perryman)
- Regionally/nationally/internationally compatible
 - All stock assessments (e.g., Carretta et al. 2011).
- Other
 - Coastal bottlenose dolphins are exposed to several human-related threats, including fisheries interactions and exposure to chemical contaminants and disease (Carretta et al. 2011)

INDICATORS OF POPULATION CONDITION: POPULATION STRUCTURE OF ALL FOCAL SPECIES

The Marine Mammal Genetics Group at the Southwest Fisheries Science identifies population structure using primarily genetic data. Population structure is identified at two levels: the evolutionary level, which is integral to implementing the Endangered Species Act, and the demographic level, which forms the basis for conservation under the Marine Mammal Protection Act. Effective conservation and management efforts of marine mammals rely on accurate identification of population structure at both of these levels. Research is supported by a state-of-the-art conservation genetics laboratory and genetics tissue archive. The latter houses a continually growing world-wide sample collection enriched by international scientific collaborations. The genetics archive, the genetics laboratory, and the database operated by the group provide the infrastructure necessary to drive the management science and ensure high quality data for current and future needs. The group also develops new molecular techniques and innovative analytical approaches

designed specifically to improve management decisions and provide expert advice on population structure at regional, national and international management and academic meetings.

Peer-reviewed literature or usage in a specific management application is available for each of the indicator selection categories for the work done by this group to identify population structure. In the list below, references are given for two specific case studies: 1) coastal bottlenose dolphin stock identification and 2) the Pacific coast feeding group of gray whales.

- Theoretically-sound (peer-reviewed findings suggest it is a surrogate for ecosystem attributes)
 - New stock assessments are based on stock identification
 - 1) Lowther (2006), Perrin et al. (2011)
 - 2) Lang et al. (2011)
- Relevant to management concerns
 - New stock assessments based on stock identification
- Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute
 - New stock assessments based on stock identification
- Responds predictably & is sufficiently sensitive to changes in a specific management action or pressure
 - New stock assessments based on stock identification
- Linkable to scientifically-defined reference points & progress targets
 - New stock assessments based on stock identification
- Concrete & Numerical
 - Stock identification based on hypothesis testing (Lowther 2006, Lang et al. 2011), identification of stranded individuals to stock (where appropriate) based on genetic assignment probability (Perrin et al. 2011)
- Historical data or information available
 - NA
- Operationally simple
 - Genetics are standardly used in stock identification of marine mammals (Taylor et al. 2010)
- Broad spatial coverage
 - Genetics are standardly used in stock identification of marine mammals (Taylor et al. 2010)
- Continuous time series
 - Biopsies are collected for genetics samples routinely on line-transect surveys (every 3-5 years, Barlow and Forney (2007)) and on other field operations, many of which are annual
- Spatial & temporal variation understood
 - Many types of analyses are available to understand spatial patterns in genetic signals. Genetic signals change on a generation time scale, which for marine mammals is on the order of 10 to 30 years. For most marine mammals sampling has occurred for at most one generation. Variation is estimated with computer simulations (Archer et al. 2010)
- High signal-to-noise ratio
 - Fine-scale structure detected in gray whales (Lang et al. 2011)
- Understood by the public & policymakers
 - All stock assessments (e.g., Carretta et al. 2011)
- History of reporting
 - All stock assessments (e.g., Carretta et al. 2011)
- Cost-effective
 - Surveys are expensive, but genetic sampling is done as part of the research. All stock assessments (e.g., Carretta et al. 2011).

- Anticipatory or leading indicator
 - Necessary to interpret abundance of indicator species (Taylor 1997).
- Regionally/nationally/internationally compatible
 - All stock assessments (e.g., Carretta et al. 2011)

INDICATORS OF POPULATION CONDITION: SOUTHERN RESIDENT KILLER WHALES

Ten indicators of population condition were identified for southern resident killer whales. These indicators include: age structure; size structure; genetic diversity (population/stock structure, effective population size); growth rate and age and size at maturity; fecundity, reproductive output, and life expectancy; diet and prey energy requirements; contaminant loads and health effects; stress hormones related to prey limitation and disturbance; behavioral and acoustic responses to vessels and acoustic disturbance; and behavioral and social responses to prey limitations.

INDICATORS OF POPULATION CONDITION: TRANSIENT KILLER WHALES

Six indicators of population condition were identified for transient killer whales. These indicators include: age structure of populations; genetic diversity of populations (population/stock structure, effective population size); fecundity, reproductive output, and life expectancy; diet and prey energy requirements; contaminant loads and health effects; and behavioral and acoustic responses to vessels and acoustic disturbance. References can be provided for each indicator.

INDICATORS OF POPULATION CONDITION: SOUTHERN CALIFORNIA BIGHT HEALTH AND CONDITION ASSESSMENTS

Long-term data sets of biological data exist for three of the focal species: the gray whale, short-beaked common dolphin, and coastal bottlenose dolphin. The data include 20 year time series of photogrammetric measurement data for gray whales and biological specimen data for short-beaked common dolphins incidentally killed in fisheries. They also include 40 year time series of biological specimen data collected from stranded common dolphins and coastal bottlenose dolphins. These time series provide valuable data to assess population health and condition. Additionally, biopsy samples have been collected opportunistically during the past two decades from short-beaked common dolphins and coastal bottlenose dolphins. These samples augment population condition assessments by estimating reproductive rates and contaminant loading using molecular techniques.

Peer-reviewed literature supporting use of the techniques for monitoring population condition and incorporating their use in management plans are provided for each of the indicator selection categories below. In the list below, references are provided for the focal species: 1) gray whale, 2) short-beaked common dolphin, and 3) coastal common bottlenose dolphin.

- Theoretically-sound
 - Photogrammetric techniques (Perryman and Lynn 1993, 2002, Cramer et al. 2008)
 - Standard life history analyses (Perrin and Reilly 1984)
 - New technique development (Kellar et al. 2006)
- Relevant to management concerns
 - Provides quality assessment of populations to accompany quantitative metrics
 - Demographic stochasticity, especially reproductive variability, reflects responses to density dependence and environmental change

- Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute
 - Variability in reproduction linked to
 - environmental signals and gray whale maternal condition (Perryman et al. 2002)
 - exploitation (Cramer et al. 2008)
- Responds predictably & is sufficiently sensitive to changes in a specific management action or pressure
 - NA
- Linkable to scientifically-defined reference points & progress targets
 - Provides qualitative assessment of populations to accompany assessment of trends in abundance
- Concrete & Numerical
 - Condition metrics are quantitative and suitable for numerical analyses (Perryman and Lynn 2002, Cramer et al. 2008)
- Historical data or information available
 - The data include 20 year time series of photogrammetric measurement data for gray whales and biological specimen data for short-beaked common dolphins incidentally killed in fisheries. They also include 40 year time series of biological specimen data collected from stranded common dolphins and coastal bottlenose dolphins.
- Operationally simple
 - Life history studies are based on standard, long-standing techniques
- Broad spatial coverage
 - Population condition metrics are available for many species providing opportunity for comparative analyses of cetacean species within an ecosystem
- Continuous time series
 - Photogrammetric data are collected during dedicated field projects.
 - Data from fishery takes during the fishing season, which is currently annually from August through January, and year-round from stranded specimens. Biopsy samples are routinely collected as part of the line-transect abundance survey cruises (every 3-5 years, Barlow & Forney 2007) and during other field projects. For example, biopsy samples are routinely collected during monthly small boat surveys of coastal bottlenose dolphins.
- Spatial & temporal variation understood
 - Length of time series and ability to incorporate data collected by other researchers working throughout the region provide adequate temporal and spatial coverage for tracking and monitoring changes in population condition (Perryman et al. 2002, Danil et al. 2010)
- High signal-to-noise ratio
 - Variability reflects population responses to changing environmental conditions (Perryman et al. 2002)
- Understood by the public & policymakers
 - Incorporated in stock assessments (e.g., Carretta et al. 2011)
- History of reporting
 - Annual IWC and US stock assessment reports
- Cost-effective
 - Fishery observer programs and surveys are typically expensive, but the collection of specimen data does not add significantly to the cost. Aerial surveys are cost-effective ways to collect large data sets (e.g., Chivers et al. 2010). Small boat surveys are used to study coastal common bottlenose dolphins and are a cost effective means to monitor the population and to collect biopsy samples.

- Anticipatory or leading indicator
 - Necessary to interpret variability in recruitment indices of indicator species populations (Perryman et al. 2002, Cramer et al. 2008).
- Regionally/nationally/internationally compatible
 - Standard methodology (e.g. IWC reports and US stock assessments: (Carretta et al. 2011))

INDICATORS OF POPULATION CONDITION: DIET, HEALTH, AND DEMOGRAPHY OF CALIFORNIA SEA LIONS

Long-term data (up to 40 years) exist for diet, demography, foraging behavior and health for California sea lions. Field work has been conducted on San Miguel Island for over forty years and includes time series on pup births and mortality, pup weights, female diet and foraging behavior, and 25 permanently marked cohorts of pups and resighting effort that has provided estimates of age-species birth rates (Melin et al. 2011) and age-specific survivorship (DeLong et al. *In prep.*). The diet of California sea lions has been studied at San Clemente Island, San Nicolas Island, and San Miguel Island. Exploratory analyses have linked population parameters and diet to environmental conditions, particularly ENSO events.

- Theoretically-sound
 - There is a long history demonstrating relationships between oceanographic conditions, particularly ENSO, and population parameters and diets of California sea lions.
 - DeLong et al. (*In prep.*) shows links between California sea lion pup survival and pup weight, ENSO events and pup and yearling survival, and the impact of *Leptospirosis* disease on male survival.
 - Melin et al. (2012a) provides measures of age-specific recruitment and natality and impacts of ENSO events on natality.
 - Melin et al. (2010) demonstrates links between upwelling and pup mortality during the 2009 oceanographic event in Central California.
 - Melin et al. (2012b) shows relationships between pup weights and female diets and oceanographic measures and pup production and early pup mortality.
 - Melin et al. (2012b) shows links between SST in female foraging region and pup growth and long term trends in pup weights.
 - Lowry et al. (1999) shows a correlation between the annual occurrence of market squid in the diet of California sea lions, oceanographic conditions, and commercial landings of market squid.
 - Lowry et al. (Lowry et al. 1990, Lowry et al. 1991) shows seasonal and annual variability in the diet of California sea lions at San Clemente Island and San Nicolas Island.
 - Lowry and Maravilla-Chavez (2003) show effects of ENSO on California sea lion pup production and estimate abundance and growth of California sea lion populations in Baja California, Mexico and the U.S. in 2000.
 - Lowry et al. (2008) demonstrate that diet of sea lions correlates with abundance of their prey.
 - Lowry and Holland (2006) demonstrates that diet of California sea lions reflects ENSO effects and shows how diet can be used to predict when California sea lions reach carrying capacity.
 - Lowry (*In prep.*) describes abundance, distribution, and growth of the U.S. California sea lion population during the breeding season through 2011.

- Impacts of man-made pollution on California sea lions have also been demonstrated (Delong et al. 1973, Gilmartin et al. 1976).
- Relevant to management concerns
 - While considered a nuisance species amongst fishermen, California sea lions are a key component of the California Current ecosystem that will be useful to measure impacts of global climate change on the marine environment, impacts of management decisions that influence their primary prey (Pacific hake, market squid, Pacific sardine, northern anchovy, shortbelly rockfish, Pacific mackerel, jack mackerel, etc.) and their predators (sharks and potentially transient killer whales), and impacts of changing relationships between California sea lions and other ecosystem components due to range expansions or contractions, population declines or increases, or marine habitat degradation or enhancement.
- Responds predictably & is sufficiently sensitive to changes in a specific marine ecosystem attribute
 - As little as 1° C shift in average sea surface temperature in the foraging range of adult females has been shown to impact pup production, growth, and survival.
 - Relationships between other oceanographic variables such as upwelling are less well understood but could be improved with linkages in the IEA with other available information.
 - Diet of California sea lions responds to abundance of their prey.
- Responds predictably & is sufficiently sensitive to changes in a specific management action or pressure
 - Scarring of sea lions by sharks has increased dramatically in the past three years and may indicate increased predation rates as the shark populations recover from decades of exploitation.
 - Diet data demonstrate shifts in prey that reflect availability due to the environment, the productivity and abundance of their primary prey, and possibly management actions about levels of fishing harvest of their primary prey
- Linkable to scientifically-defined reference points & progress targets
 - Current population assessment is showing recovery of a population that was once depleted by commercial harvests, bounty killing, and pollution.
 - Precise estimates can be measured cost effectively for any identified reference point or target and they can be interpreted in the context of a 40 year historical time series.
- Concrete & Numerical
 - All of the measures of demography, health (e.g., contaminant levels, disease prevalence), and diet are numerical with estimates of precision.
- Historical data or information available
 - Four decades of data are available for pup counts and pup weights.
 - Three plus decades of seasonal diet data are available.
 - Two plus decades of survival and natality data are available.
- Operationally simple
 - California sea lions are one of the easiest marine mammal species to study because of the limited number of rookeries where females can be found hauled out year round and most males and some juveniles are located during the pupping/breeding season.
 - The primary rookeries that account for 90% of the U.S. sea lion production are at San Miguel where there is a permanent research station in cooperation with the Channel Islands National Park and at San Nicolas Island where there is a permanent Naval facility. Transportation to San Nicolas Island is provided at no cost but there are lodging costs. There are transportation costs to San Miguel Island, but no lodging costs. Re-sighting

surveys of primarily juvenile sea lions have also been conducted for more than a decade at Año Nuevo Island with a contract to the University of California, Santa Cruz.

- Pups and juveniles are easily handled with physical restraint; large numbers of pups can be easily and safely handled by herding them on land. Typically 300-500 pups have been permanently marked from each birth cohort since 1987 at San Miguel Island. Adult females can and have been captured fairly easily on land at San Miguel, San Nicolas and San Clemente Islands. Adult males are harder to handle but have been caught and handled on float traps in Astoria OR, Seattle WA, and Monterey, CA. Capture methods are being developed for adult males on the rookery. Biopsy sampling of territorial adult males has been successfully accomplished at San Miguel Island.
- In addition access to stranded sea lions in California, Oregon, and Washington, and collaboration with The Marine Mammal Center in Sausalito, California has proved invaluable to our understanding of the impacts of diseases (*e.g.*, domoic acid, *Leptospirosis*) and cancer associated with pollution on reproduction and survival.
- Broad spatial coverage
 - Females and juveniles operate on relatively large spatial scales from southern to northern California and in coastal and offshore habitats. Outside of the breeding season sub-adult and adult males range from northern California to British Columbia. Thus, the species covers the entire extent of the California Current throughout the year except for the short 2 month summer pupping and breeding season when most are in southern and central California.
- Continuous time series
 - Most of the time series data have been collected uninterrupted for 20 to 40 years, but count data from the Channel Islands exist back to 1927.
- Spatial & temporal variation understood
 - That is a bold claim for any species but we are closer for California sea lions than most marine mammal species.
 - Seasonal migrations of males has been studied and is influenced by environmental conditions (Weise et al. 2006, Gearin et al. *In prep.*)
 - Temporal, spatial, sex, and age-specific segregation of the population is well understood, but variable depending on environmental conditions (DeLong et al. *In prep.*)
 - Juvenile and female seasonal foraging behavior is well understood and responsive to ENSO and smaller localized changes in environment (Kuhn 2006, Melin et al. 2008).
 - Temporal signal in population growth as reflected by pup counts shows continual recovery from depletion and ENSO signal in production.
- High signal-to-noise ratio
 - Large signals are created by small changes in SST and very precise measures can be obtained easily and cheaply.
- Understood by the public & policymakers
 - California sea lions are well known to the public and policy makers due to their presence in aquaria and their frequent visits to land. They are a high profile species and visitor attraction at places like San Francisco pier 39, sea lion caves in Oregon, and Ballard locks in Seattle.
 - Policymakers will be familiar with California sea lions due to concerns with fisheries interactions in the sport and commercial fishing industries along the California, Oregon, and Washington coasts, interactions with endangered salmonids at the Ballard locks in Washington and more recently Bonneville Dam in Oregon where there is a limited removal

- of adult males that are consuming endangered salmon, and interactions with humans on piers and at boat marinas along the California coast.
 - Collaboration with the states of Washington and Oregon, The Marine Mammal Center, the U.S. Navy, and the National Park Service has increased awareness of this species to both the public and policymakers.
- History of reporting
 - The impacts of the environment are evident to the public when there are mass stranding of pups and juveniles during ENSO events or natural toxic blooms that result in domoic acid toxicity of adult females and juveniles.
 - Abundance of sea lions and the trend in the population is of interest to sports, commercial, and tribal fishermen in regards to interactions with the fishing industries.
 - All of the indicator measures currently and historically are important to the species, relevant to the public, and essential to a California Current IEA.
- Cost-effective
 - Pinnipeds are low cost indicator species relative to cetaceans which require expensive ship time. While pinniped abundance surveys are conducted with aerial photography, the cost is quite small because large geographical areas can be sampled in a short time (land surveys only sample a small number of locations). There is a long track record of maintaining a long time series of data for California sea lions with a fairly small budget. While larger budgets would certainly improve the quality and quantity of information, there is no other marine mammal species that can be sampled as cheaply with as large a scale of influence in the California Current.
- Anticipatory or leading indicator
 - Most marine species will lag in their response to an environmental change, however in the case of California sea lions, this can occur in as little as two months and it is not unusual for California sea lions to reflect changes in the oceanographic environment before oceanographers recognize what is occurring. The oceanographic event of 2009 in central California is a perfect example where routine monitoring of stranding on the coast and pup production and mortality at San Miguel and San Nicolas Islands demonstrated a dramatic level of pup mortality and starving juveniles due to severely depressed upwelling during May and June in the pupping season (Melin et al. 2010). Initially oceanographers said that it was not an ENSO event because the signal was just starting to appear at the equator but they subsequently decided that it was an ENSO event that was not transported via the typical Kelvin wave. Clearly California sea lions are sensitive indicators of their marine environment that will also possibly help understand the oceanographic processes of ENSO events, local or regional anomalous events, and impacts of climate change in the California Current.
 - Most of the data collected for California sea lions are easily assembled and analyzed in a timely fashion for any relevant management decisions. The one exception is survival of the current pup cohort which can only be predicted after 2 years of resight data although that could be reduced by a more intense within year sampling effort that has only been conducted a few times.
- Regionally/nationally/internationally compatible
 - Pinnipeds have been used as indicator species in other ecosystems. A large and long-running program uses Antarctic fur seals at South Georgia, Antarctica as an indicator of ecosystem processes (Croxall et al. 1988, Boyd et al. 1994, Reid and Croxall 2001).
 - Although not yet started, comparative studies between California sea lions and the closely related Galapagos sea lion (*Zalophus wollebaeki*) in the Humbolt Current Large Marine

Ecosystem could reveal differences in how the two ecosystems respond to climate change and how the two species respond to the changes.

- Burton and Koch (1999) showed that stable isotope studies of California sea lions, northern fur seals and harbor seals throughout the northeastern Pacific can help understand differences in how different species of pinnipeds use their environment and track changes in the environment over time by changing stable isotope ratios.
- Comparison of California sea lion population parameters among the California Channel Islands have shown that the different island populations respond to regional environmental changes differently (DeLong et al. 1991).

STATUS AND TRENDS

MAJOR FINDINGS

The trend assessment approach used in other sections of this report is inappropriate for assessing trends in marine mammal abundance for several reasons. First, the approach is only useful for identifying short-term trends that depart from a long-term stationary process (i.e., no long-term increase or decline in abundance). Long-term growth or decline in the abundance of the population will inflate the standard deviation (compared to if the mean were constant), thus obscuring the ability to identify short-term patterns. Marine mammal surveys in recent decades coincide with a period of expected long-term increases in many pinniped and large whale populations, as they recover from severe depletion following historical hunting. Other populations may be experiencing long-term declines in CCLME.

Second, the approach assumes that the abundance estimates are independently and identically distributed (*iid*) with constant variance. For marine mammals, the *iid* assumption is not met for multiple reasons. First, marine mammals are highly mobile and their distributions have been shown to vary with oceanographic conditions (Forney 2000, Becker et al. 2010). For many species surveyed in the CCLME, these movements occur across survey area boundaries. Consequently, there is inter-annual variability in the proportion of the population occurring within the survey area, which results in varying levels of sampling error across surveys. Second, inter-annual variation in survey conditions results in variation in survey effort and detectability that translates into non-constant variances of the abundance estimates. Analyses by Moore and Barlow (2011) provide an example of this non-constant variance; across surveys from 1991-2008, annual coefficients of variation for fin whale abundance estimates ranged from 0.16 to 0.28. Finally, the abundance estimates are not independent because they are obtained using information pooled across years to estimate the detection function; the lack of independence biases the estimate of the standard deviation if not taken into account. The implication of these factors is that an individual abundance point estimate falling outside of the long-term standard deviation cannot be interpreted as being anomalously high or low (i.e., does not constitute a basis for inferring trend); sampling error may have just been really high in this year.

Finally, even if the above issues were not important, five years is not a reasonable timescale over which to evaluate trends in marine mammal abundance in the CCLME, for statistical and biological reasons. Marine mammal abundance estimates are generally imprecise and a five-year time window only includes two abundance estimates for many species. Marine mammals generally have low theoretical maximum net growth rates (e.g., 12% for pinnipeds and 4% for cetaceans, Wade and Angliss 1997). Hence, even a relatively rapid change in total population abundance (e.g., $> \pm 5\%$ annually or 50% within 15 years) is extremely unlikely to be detected within a five-year time frame, especially when based on so few surveys (Taylor et al. 2007). In short, substantial differences in the abundance estimates from the line-transect surveys conducted in 2005 and 2008 are not very informative, and the approach used in other sections for assessing trends

carries a fairly high risk of interpreting as short-term trend what may be better explained as a differences due to sampling error or in the proportion of the population in the study area at time of survey. Assessment of marine mammal trends is best done over longer (e.g., decadal) time periods and using methods more appropriate for handling the issues described above.

Assessment of trends in marine mammal abundance is an active area of research in the Marine Mammal and Turtle Division of the Southwest Fisheries Science Center. Moore and Barlow (2011) developed a Bayesian hierarchical model to estimate both abundance and population trends for fin whales in the California Current. Use of this model allowed them to implicitly address sampling covariance, accommodate random effects and covariates, compare trend models of different functional forms, and partition sampling and process error. These analyses were able to provide strong evidence for increasing fin whale abundance and resulted in more precise abundance estimates. Similar analyses are underway for additional cetacean species that occur in the waters off the U.S. West Coast.

In this section, we plot abundance estimates for many of the selected indicators. We also discuss the caveats associated with using the different types of abundance estimates to assess trends. Where possible, we describe research that has been conducted to explore variability in the abundance estimates and assess long-term trends abundance.

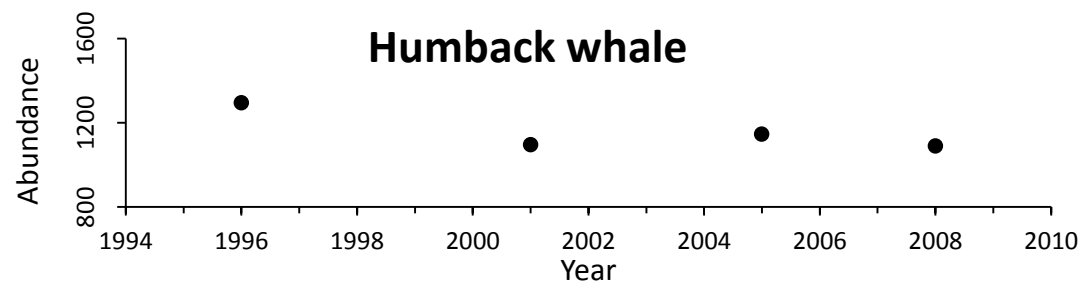
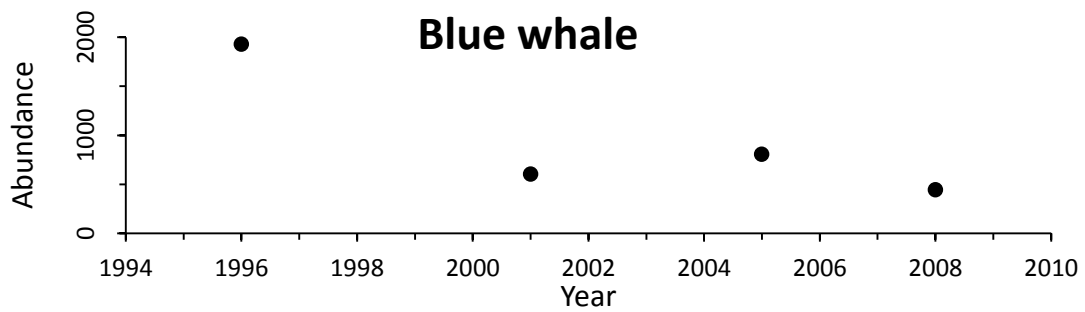
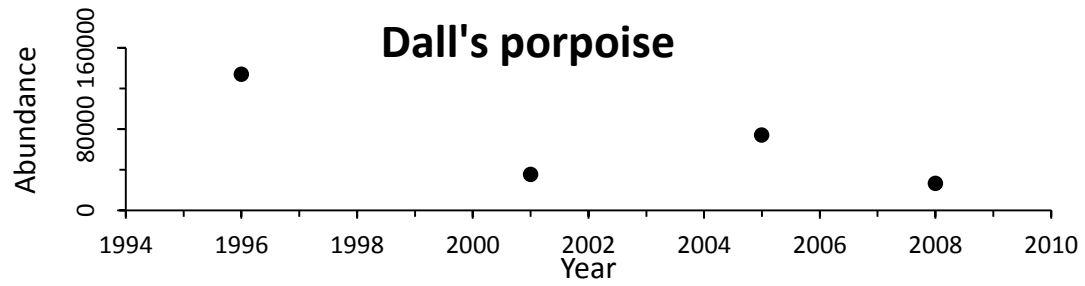
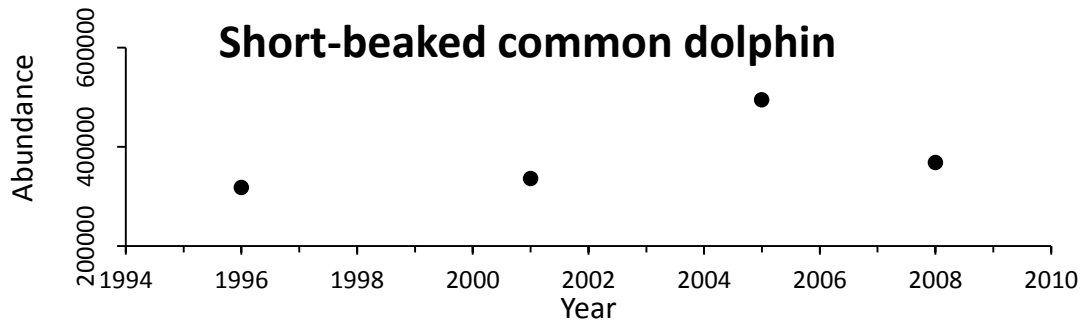
SUMMARY AND STATUS OF TRENDS

INDICATORS OF POPULATION ABUNDANCE: SHORT-BEAKED COMMON DOLPHINS, DALL'S PORPOISE, BLUE WHALES, HUMPBACK WHALES, AND FIN WHALES

Line-transect estimates of abundance for these species from 1991 to 2005 can be found in Barlow and Forney (2007). Line-transect abundance estimates are also available for a survey conducted in 2008 (Barlow 2010). The estimates from these papers for fin whales, humpback whales, blue whales, short-beaked common dolphins, and Dall's porpoise are shown in Figure MM1. To date, abundance has been estimated for a single stock of fin whales, humpback whales, short-beaked common dolphins, and Dall's porpoise; however, these species may have multiple stocks in the California Current. Determination of stock structure is an active area of research in the Marine Mammal and Turtle Division of the Southwest Fisheries Science Center.

Waters off the U.S. West Coast represent only a portion of the range of short-beaked common dolphins and Dall's porpoise (Forney 2000). Habitat models for these species (Forney 2000) and comparisons of seasonal abundance estimates (Forney and Barlow 1998) suggest that their distribution and abundance change with varying oceanographic conditions. Consequently, the abundance estimates in Figure MM1 are likely influenced by trans-boundary movements associated with oceanographic conditions.

More precise estimates of blue whale abundance and independent estimates of humpback whale abundance for west-coast populations are available from mark-recapture methods using photo-identification (Calambokidis and Barlow 2004). Subsequent studies have shown that the apparent decline in line-transect estimates of blue whale abundance are likely due to a shift of the population outside of the west-coast study area (Calambokidis et al. 2009)(Fig. MM2). The population of humpback whales along the U.S. West Coast is growing at approximately 7.5% per year (Carretta et al. 2011); occasional declines in abundance are likely associated with changes in oceanographic conditions (Carretta et al. 2011).



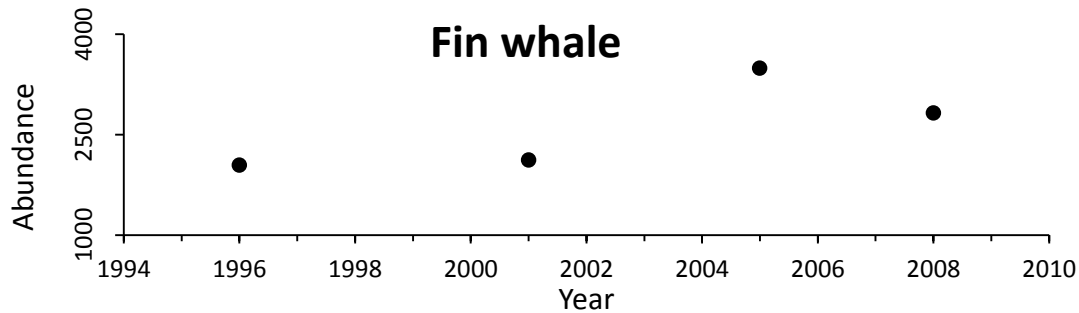


Figure MM1. Ship survey estimates of abundance for short-beaked common dolphins, Dall’s porpoise, blue whales, humpback whales, and fin whales.

Moore and Barlow (2011) found evidence of increasing fin whale abundance using a Bayesian hierarchical model developed from the line-transect survey data. Their approach allowed them to overcome many of the caveats associated with using these data. Some of these caveats include lack of independence among the abundance estimates because data from all years were used to estimate the effective strip width. Additionally, detection probabilities on the track line many vary by condition (e.g., glare or Beaufort sea state) for some species. Current estimates of track line detection probabilities are based on independent observer data or models that incorporate the diving characteristics of species. The model-based trackline detection probabilities are based on “average conditions”; consequently, the existing abundance estimates are not corrected for annual changes in survey conditions.

INDICATORS OF POPULATION ABUNDANCE: GRAY WHALES

Estimates of gray whale abundance can be found in Laake et al. (2009) and Caretta et al (2011); these estimates are plotted in Figure MM2. These abundance estimates assume that all individuals migrate and are available to be counted. The Eastern North Pacific population of gray whales has been increasing for the past several decades (Punt and Wade 2010), although an unusual mortality event occurred in 1999 and 2000.

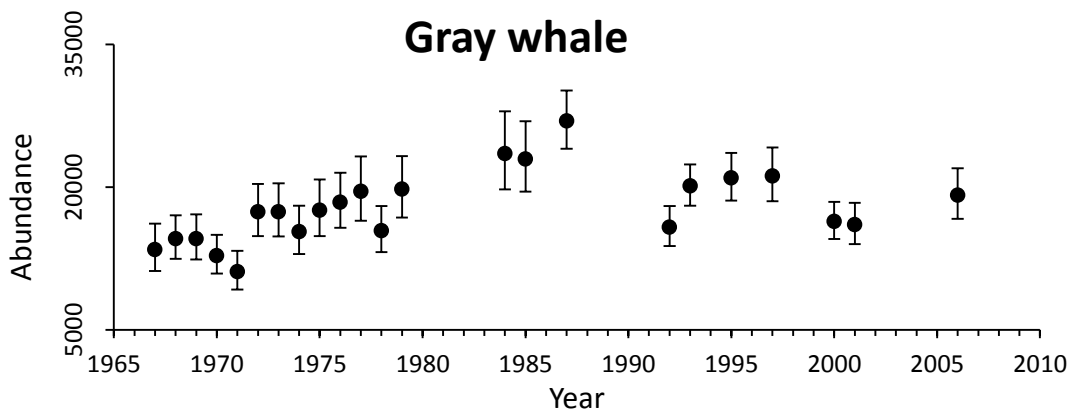


Figure MM2. Gray whale abundance estimates are shown with 95% log-normal confidence intervals.

INDICATORS OF POPULATION ABUNDANCE: CALIFORNIA SEA LIONS

Estimates of California sea lion abundance can be found in Lowry & Maravilla-Chavez (2005) and Caretta et al. (2011). Since 1975 sea lion pup numbers have increased at an average annual growth rate of 5.4% (Caretta et al. 2011) (Fig. MM3).

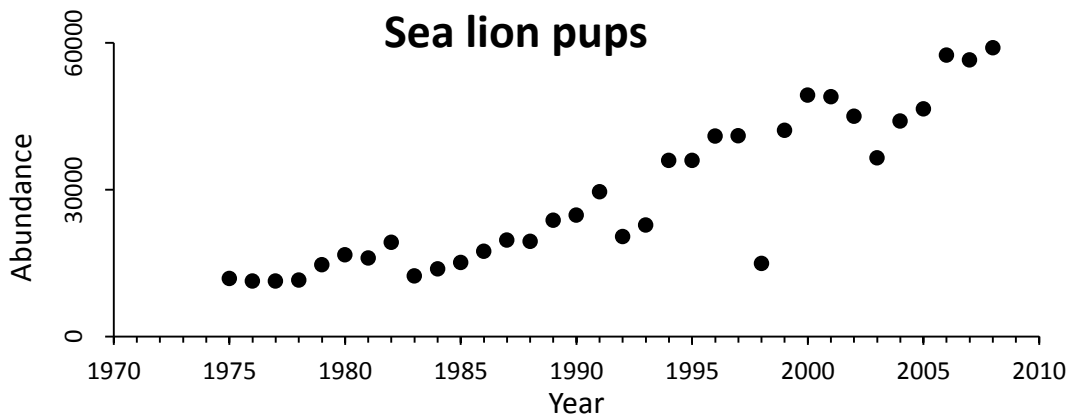


Figure MM3. California sea lion pup abundance estimates.

INDICATORS OF POPULATION ABUNDANCE: COASTAL BOTTLENOSE DOLPHIN

Estimates of coastal bottlenose dolphin abundance can be found in (Dudzik et al. 2006). These estimates are derived from surveys conducted in the waters off San Diego, California. The estimates are derived assuming panmixia.

INDICATORS OF POPULATION ABUNDANCE: SOUTHERN RESIDENT KILLER WHALES

The Center for Whale Research conducts annual photo-id surveys of this population (<http://www.whaleresearch.com>) and provides population counts on 1 July and 31 December each year. NOAA NWFSC includes the 1 July data in annual updates of the Pacific SAR (Caretta et al. 2011). Because the data are actual population counts (not estimates), they include the age and sex of each individual. The time series runs from 1974 through present.

INDICATORS OF POPULATION ABUNDANCE: TRANSIENT KILLER WHALES

Estimates of the abundance of the west coast stock of transient killers are periodically made from photo-id data obtained by researchers in different regions (British Columbia, southeast Alaska, and California). A time series of abundance for the entire stock is not available because of the difficulties associated with surveying such a wide ranging stock (see Allen and Angliss 2011). Population growth rates have been estimated for the British Columbia/southeast Alaska portion of the stock (DFO 2009)

INDICATORS OF POPULATION CONDITION: POPULATION STRUCTURE FOR ALL FOCAL SPECIES

Determination of population structure provides the foundation for assessing the status of species or stocks (e.g., management units). Although there is a single abundance estimate for fin whales, humpback whales, short-beaked common dolphins, and Dall's porpoise, these species may have multiple stocks in the California Current. Stock structure in the California Current is an active area of research for the Marine Mammal Genetics Group within the Marine Mammal and Turtle Division at the Southwest Fisheries Science Center.

INDICATORS OF POPULATION CONDITION: POPULATION STRUCTURE, DEMOGRAPHY, AND CONTAMINANT LEVELS OF SOUTHERN RESIDENT KILLER WHALES

The Center for Whale Research conducts annual photo-id surveys of this population (<http://www.whaleresearch.com>) and provides population counts on 1 July and 31 December each year. NOAA NWFSC includes the 1 July data in annual updates of the Pacific SAR (Caretta et al. 2011). Because the data are actual population counts (not estimates), they include the age and sex of each individual. The time series runs from 1974 through present. These data can and have been used to estimate fecundity, reproductive output, and life expectancy over time (e.g. (Olesiuk et al. 1990, Olesiuk et al. 2005, Ford et al. 2009, Ward et al. 2009a, Ward et al. 2009b, Ward et al. 2011)).

Several publications provide data on contaminant levels (including, PCBs, PBDEs, DDTs, CHLDs, HCHs, HCB, PCDDs, PCDFs, and PCNs) in individual Southern Resident killer whales. Ross *et al.* (2000), Rayne *et al.* (2004) including supplemental information on individual levels), and Ross (2006) provide contaminant levels in SRKWs from 1993-1996. Krahn et al. (2007) provides SRKW contaminant levels in 2004 and 2006; Krahn et al. (2009) reports SRKW contaminant levels in 2007. NWFSC has continued to collect samples since 2007 and can contribute data to a longer time series. When using these data to assess trends in contaminant levels, several caveats must be taken into account. Samples are only taken from select individuals and the same individuals were not sampled across all studies. However, the data from these publications provides general trends for each pod.

Mongillo et al. (2012) present a model that reconstructs historical PBDE and PCB profiles. The model is also forecasts individual contaminant levels under different scenarios. Model predictions were tested with actual contaminant levels from biopsied SRKW individuals. Results of the comparison between model output and measured values were used to refine model assumptions. The final model is exceptional at predicting contaminant levels in individual SRKWs from all three pods.

INDICATORS OF POPULATION CONDITION: HEALTH AND CONDITION ASSESSMENTS

Estimates of gray whale calf production from 1994 to 2000 can be found in Perryman et al. (2002). Calf production is estimated for each year in the time series. While the estimates use annual count data collected for the entire cow-calf migration period, they also incorporate correction factors for diel migration rates and observer performance. These correction factors were estimated from pooled data collected during a limited portion of the study. Perryman et al. (2002) also present evidence that calf production is correlated with ice cover extent on the gray whales' arctic feeding grounds.

CALIFORNIA SEA LIONS: DIET, HEALTH, AND DEMOGRAPHY

Melin et al. (2010) and Melin et al. (2012b) provide time series data of pup counts, pup weights and pup mortality and diet of adult females for an index area of San Miguel Island between 1997 and 2012. Melin et al. (2012a) and DeLong et al. (*In prep.*) provide age-specific natality estimates and survival estimates, respectively. Lyons et al. (1997, 2001, 2005), Acevedo-Whitehouse et al. (2006), and Spraker et al. (2007) describe the effects of hookworm disease on survival of California sea lion pups. Ylitalo et al. (2003, 2005), Brodie et al. (2006), de la Riva et al. (2009) describe other diseases and the effects of contaminants and natural toxins on the health of California sea lions.

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COASTAL PELAGICS AND FORAGE FISHES

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TABLE OF CONTENTS (C)

| | |
|--|-----|
| Executive Summary | 378 |
| Detailed Report | 379 |
| Indicator selection: Southern California Current, CalCOFI | 379 |
| Status and trends: Southern California Current, CalCOFI | 384 |
| Indicator selection: Central California Current, midwater trawl survey | 384 |
| Status and trends: Central California Current, midwater trawl survey | 386 |
| Indicator selection: Northern California Current, predator survey | 388 |
| Status and trends: Northern California Current, predator survey | 389 |
| Indicator selection: Northern California Current, Columbia River plume | 391 |
| Status and trends: Northern California Current, Columbia River plume | 392 |
| Indicator selection: Assessments | 394 |
| Status and trends: Assessments | 394 |
| Risk | 397 |
| References cited | 397 |

LIST OF TABLES AND FIGURES (C)

Abundance time series for anchovy and sardine from three regions of the California Current system. Absence of red line indicates years of no survey results , green area indicates the last five years of the data series, dashed green line indicates mean and solid green lines indicate ± 1 s.d..... 378

Table C1: List of mesopelagic and coastal pelagic species from CalCOFI surveys used in this report. Subcategory lists mesopelagic species associated with warm or cool water conditions in the Southern California Bight. All species were captured as larvae and enumerated in units of mean larvae/10m² captured in the CalCOFI core area within three month periods (i.e., quarters) and summed over all four quarters for a year..... 380

Figure C1. *CalCOFI Sampling Pattern for Oblique Net Tows.* Grid pattern of 3.3-line by 10-station cells in the core CalCOFI sampling area (lines 76.7-93.3) used for analysis of Southern California forage. Color key indicates actual number of samples collected within each cell for the period 1951-2010. Black dots indicate actual sample locations..... 381

Figure C2. *Southern California Forage, CalCOFI.* Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d. , or was within one 1.0 s.d of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-)or within (·) one s.d. of the long-term mean 383

Table C2. Species collected and enumerated in the mid-water trawl survey along Central California..... 385

Figure C3. *Central California Forage, mid-water trawl.* Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d. , or was within one 1.0 s.d of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-)or within (·) one s.d. of the long-term mean. 388

Table C3. Species collected in the surface trawl of the northern California Current predator survey..... 389

Figure C4. *Northern California Forage, Predator survey.* Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d. , or was within one 1.0 s.d of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-)or within (·) one s.d. of the long-term mean. 390

| | |
|--|-----|
| Table C4. Species collected din the surface trawl of the Northern California Current Columbia River plume survey. | 391 |
|--|-----|

| | |
|--|-----|
| Figure C5. Northern California Forage, Columbia River plume. Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d. , or was within one 1.0 s.d of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-)or within (.) one s.d. of the long- term mean | 393 |
|--|-----|

| | |
|--|-----|
| Figure C6. <i>Assessment biomasses.</i> Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d. , or was within one 1.0 s.d of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-)or within (.) one s.d. of the long-term mean..... | 396 |
|--|-----|

| | |
|--|-----|
| Figure C7. <i>Assessed age structure.</i> Shown are the biomass of sardines and the proportions of various ages (y-axis) in the population of sardine from 1993 to 2011..... | 397 |
|--|-----|

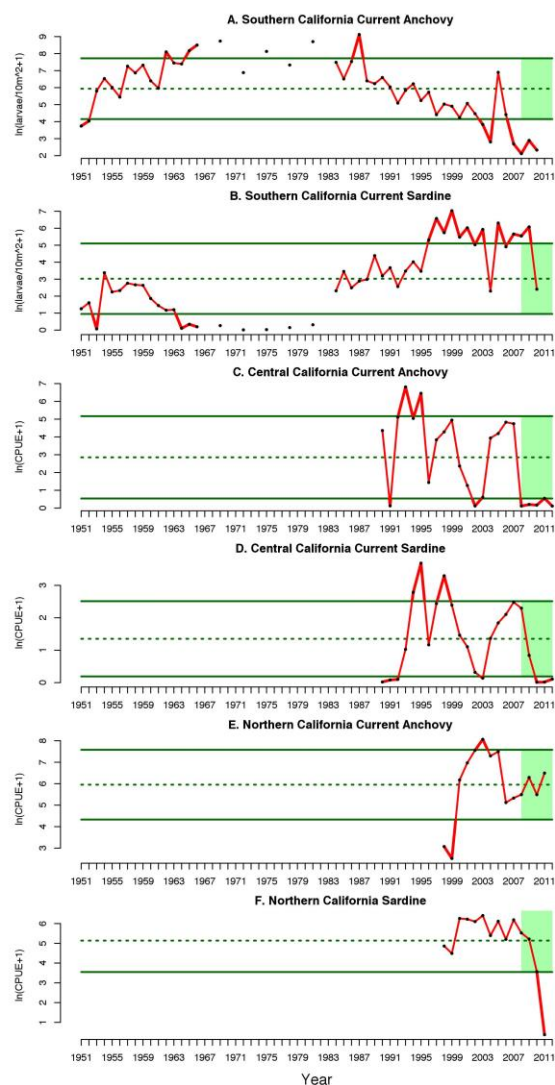
OVERVIEW

Although most assessed forage fish in the central and northern California Current regions are becoming more abundant or remained stable in recent years, sardine appear to be declining off the northern California coast. In the southern California Current region, there is an indication of recent reduction in forage production driven by the trends in anchovy, hake and the mesopelagic coolwater forage assemblage.

EXECUTIVE SUMMARY

Here, we examine trends in abundance and condition of coastal pelagic species and additional forage species throughout the California Current Large Marine Ecosystem (CCLME). Primarily we rely on the data collected from fishery independent surveys in southern California (1951-2010), central California (1990-2012), and Washington and Oregon (1998-2011). Given the differences in methods, catchability, and timing, these surveys are not directly comparable; however the intra-survey trends can be used to inform relative differences in abundance between the regions. We also utilize assessment reports of the Pacific Fisheries Management Council (1929-2011; (Crone et al. 2011, Hill et al. 2011) to estimate trends in biomass and age structure of assessed coastal pelagic species.

There is substantial regional variability in the forage base dynamics in the California Current system. Generally, in the central and northern California Current regions, the forage community became more abundant or remained stable, although sardine abundance declined in recent years. Anchovy is below average in Central California, yet stable in Northern California. Off Southern California, anchovy abundance appears to be on the same trajectory as for Central California, while sardine remain stable. However, in the context of the longer time frame (multiple decades), abundance of sardines is about average. The accompanying figure can be used to demonstrate these points.



Abundance time series for anchovy and sardine from three regions of the California Current system. Absence of red line indicates years of no survey results, green area indicates the last five years of the data series, dashed green line indicates mean and solid green lines indicate ± 1 s.d.

DETAILED REPORT

The purpose of this chapter of the CCIEA is to examine trends in available indicators relevant to coastal pelagic species and additional forage fishes along the California Current. This is the first step in finding valuable data series that can be used to describe various aspects of the CCE and its forage community. The analysis is largely qualitative at this early stage of the CCIEA. It is important to recognize that we refer to “status” here quite differently than the Pacific Fisheries Management Council (PFMC), and any difference between our status statements and those should not be considered a conflict. We are not using similar models nor benchmarks as those traditionally used. Our purpose is to set the framework for evaluating the forage community from an ecosystem perspective. This approach starts with a simple selection of indicators and evaluation of the trends. However, in subsequent reports we will use these biological indicators in combination with indicators of environmental and anthropogenic pressures to evaluate potential risk to the forage community and develop additional assessment tools useful for ecosystem based management. Indicators for various pressures can be found in other chapters of the full CCIEA (e.g., Anthropogenic Drivers and Pressures, Oceanographic and Climatic Drivers and Pressures).

Coastal pelagic species (CPS) and forage species support important commercial fisheries as well as a number of higher trophic level species including those that are commercially exploited (e.g., rockfish, salmon) and/or legally protected (e.g., salmon, marine mammals, seabirds). In the context of this report, we consider species to be a CPS and/or forage if they are often present in high abundance, feed on plankton for a portion of their life cycle and form dense schools or aggregations (e.g., anchovy, sardine, herring, mackerel, as well as invertebrate species such as squid and krill). Such species are often the principal means of transferring production from primary and secondary trophic levels (typically phytoplankton and zooplankton) to larger predatory fish, marine mammals and seabirds. Although the potential dynamics between the forage base and ecosystem integrity is not the primary aim of this section, we note that recent work Smith et al. (2011) demonstrates the likely negative effects on the ecosystem caused by reductions in abundance of lower trophic level species.

Here, we define coastal pelagic species as recognized by the PFMC: northern anchovy, Pacific sardine, jack mackerel, Pacific mackerel, market squid, and krill. However, when data are available, we also include trends in other fishes that make up the forage complex including juvenile groundfish, herring, whitebait smelt, sanddabs, and selected mesopelagic assemblages.

INDICATOR SELECTION: SOUTHERN CALIFORNIA CURRENT, CALCOFI

INDICATOR EVALUATION: SOUTHERN CALIFORNIA CURRENT, CALCOFI

We considered a number of indicators to represent the coastal pelagic larval and forage assemblage in southern California. Our choice of indicators was based on relative abundances, time series length and availability. As well, the literature indicates that unexploited oceanic assemblages are more sensitive to climatic effects than coastal and/or exploited species (Hsieh and Ohman 2006). Data sources potentially included: 1. estimates of small pelagic fish biomass from acoustics (MacLennan and Simmonds 1992, Zwolinski and Demer 2012, Zwolinski et al. 2012), 2. sardine biomass from aerial surveys (Jolly and Watson 1979, Lo et al. 1992), and 3. Daily Egg Production Method (DEPM) surveys for sardine (Lasker 1985, Lo et al. 1996). Although these series are valuable and both the acoustic surveys and the DEPM surveys produce

biomass or spawning biomass estimates, results from these surveys are integrated in the sardine stock assessment (Crone et al 2011, Hill et al. 2011) , and we therefore do not use them individually in this report.

An additional data source, the California Cooperative Oceanic Fisheries Investigations (CalCOFI) provides the longest and most complete estimates of abundance of over 400 combined fish and cephalopod species (Table C1). Here we utilized CalCOFI ichthyoplankton data from 1951 to 2010 collected through oblique vertical plankton tows as described by Kramer et al (1972) and Smith and Richardson (1977).

Table C1: List of mesopelagic and coastal pelagic species from CalCOFI surveys used in this report. Subcategory lists mesopelagic species associated with warm or cool water conditions in the Southern California Bight. All species were captured as larvae and enumerated in units of mean larvae/10m² captured in the CalCOFI core area within three month periods (i.e., quarters) and summed over all four quarters for a year.

| Genus species | Common name | Subcategory |
|-------------------------------------|---------------------------|-------------|
| <i>Bathylagus pacificus</i> | slender blacksmelt | cool-water |
| <i>Bathylagus wesethi</i> | snubnose blacksmelt | warm-water |
| <i>Ceratoscopelus townsend</i> | fangtooth lanternfish | warm-water |
| <i>Citharichthys sordidus</i> | Pacific sanddab | |
| <i>Diogenichthys atlanticus</i> | longfin lanternfish | warm-water |
| <i>Diogenichthys laternatus</i> | diogenes laternfish | warm-water |
| <i>Engraulis mordax</i> | northern anchovy | |
| <i>Leuroglossus stilbius</i> | California smoothtongue | cool-water |
| <i>Lipolagus ochotensis</i> | eared blacksmelt | cool-water |
| <i>Merluccius productus</i> | hake | |
| <i>Protomyctophum crockeri</i> | California flashlightfish | cool-water |
| <i>Sardinops sagax</i> | Pacific sardine | |
| <i>Sebastes jordani</i> | shortbelly rockfish | |
| <i>Stenobrachius leucopsarus</i> | northern lampfish | cool-water |
| <i>Symbolophorus californiensis</i> | bigfin laternfish | warm-water |
| <i>Tarletonbeania crenularis</i> | blue laternfish | cool-water |
| <i>Triphoturus mexicanus</i> | Mexican lampfish | warm-water |
| <i>Vinciguerria spp.</i> | lightfishes | warm-water |

We have restricted our analysis to the most abundant and potentially influential CPS and forage species for which we have data. To provide an integrated measure of large-scale responses to environmental variability, we aggregated the mesopelagic fishes into cool- and warm-water groups following Hsieh et al. (2005). These groups are likely to reflect general trends in the ecosystem better than time series for individual species, some of which are relatively data poor. The species and groups analyzed were Pacific sardine, northern anchovy, hake, jack mackerel, Pacific sanddab, shortbelly rockfish, cool-water mesopelagics, and warm-water mesopelagics (Figure C2.)

Summary of indicators: Southern California Current, CalCOFI

1. All data are from the core CalCOFI sampling area (lines 76.7-93.3, stations 28.0 – 120.0; Figure C1) for years when the core area was sampled during each quarter of the year. Mean larval abundances (larvae/10 m²) were estimated for each 3.3-line by 10-station cell in the core area for each quarter, and then cells were summed over the year. Means across the entire time series were then calculated using the delta-lognormal distribution (Pennington 1983). This procedure standardized the data given unequal sampling effort during some cruises, many zero catches, and seasonal but variable patterns of spawning for the fishes analyzed.
2. Individual species analyzed were Pacific sardine, northern anchovy, Pacific hake, jack mackerel, Pacific sand dab, and shortbelly rockfish.
3. The cold- and warm-water associated mesopelagic species were summed for each net tow and then analyzed as groups following the same method described above for individual species.
4. Summed forage is simply the sum of the larval abundances listed in Table C1 and expressed on a log scale.

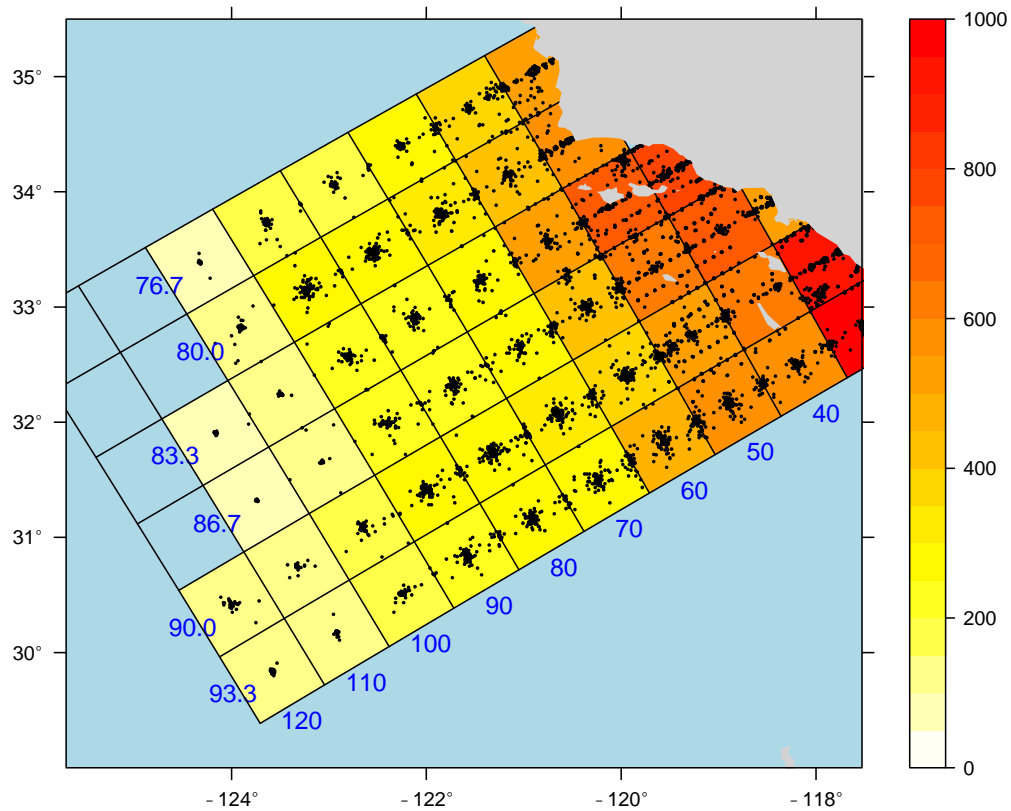
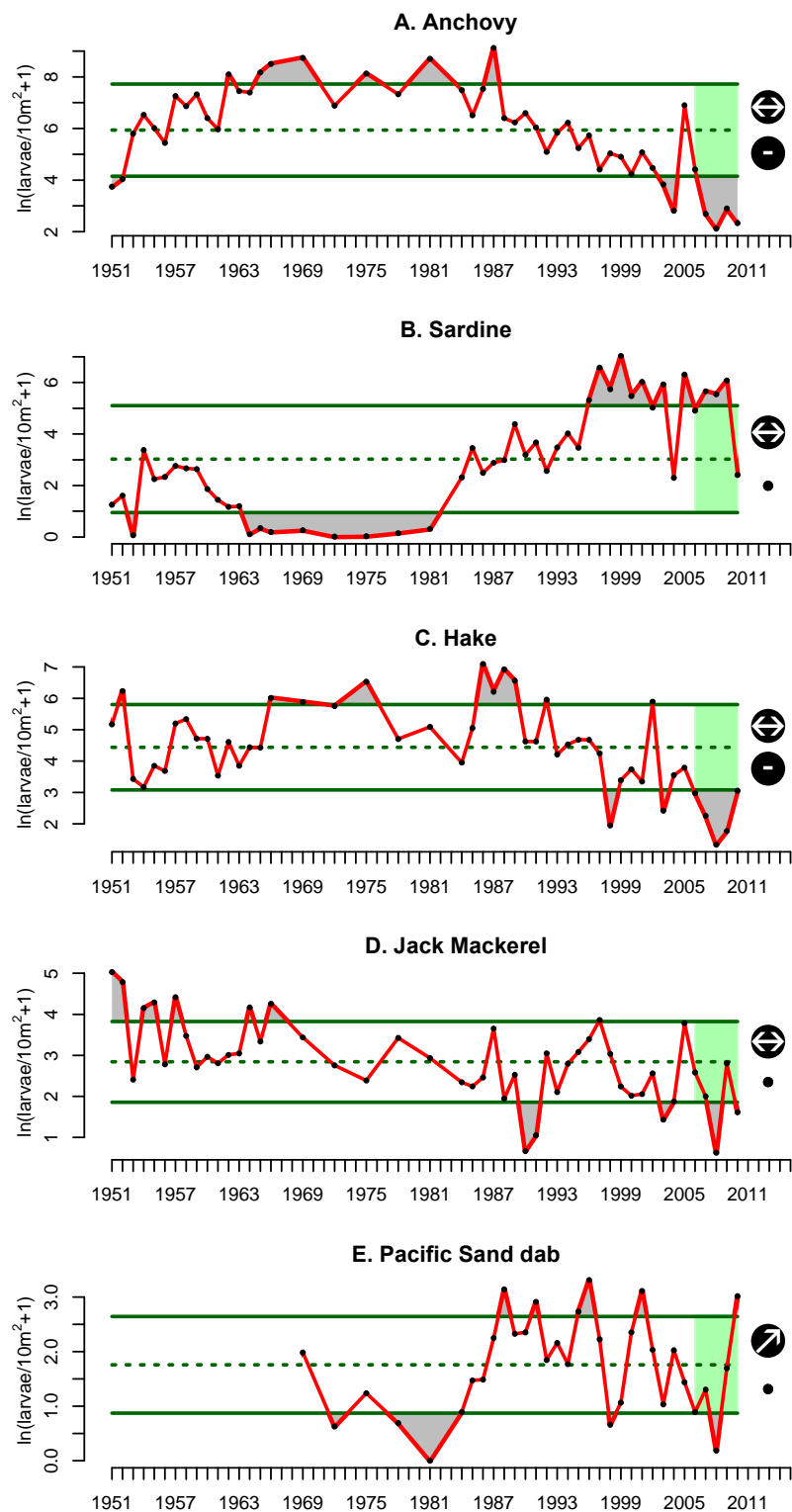


Figure C1. *CalCOFI Sampling Pattern for Oblique Net Tows.* Grid pattern of 3.3-line by 10-station cells in the core CalCOFI sampling area (lines 76.7-93.3) used for analysis of Southern California forage. Color key indicates actual number of samples collected within each cell for the period 1951-2010. Black dots indicate actual sample locations.



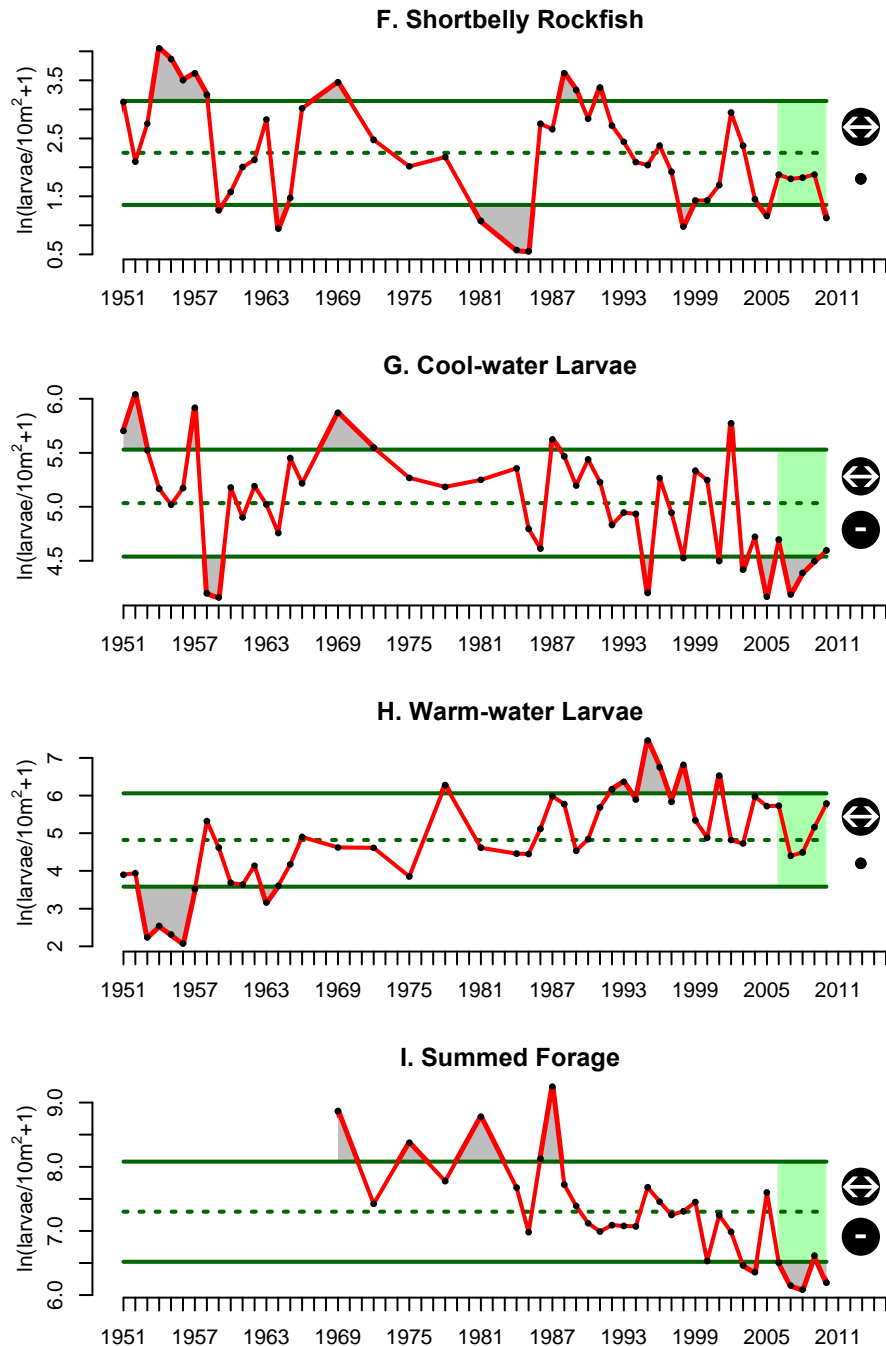


Figure C2. *Southern California Forage, CalCOFI.* Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d., or was within one 1.0 s.d. of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-) or within (.) one s.d. of the long-term mean

STATUS AND TRENDS: SOUTHERN CALIFORNIA CURRENT, CALCOFI

MAJOR FINDINGS: SOUTHERN CALIFORNIA CURRENT, CALCOFI

Since 1951 the 6 species indicators and 2 species group indicators have shown high variability and limited covariation. The trends presented here are a simple, qualitative examination of major patterns, but we are testing additional ways to examine these data series including: 1) separation of the secular from the shorter time scale variability 2) examining trends in variance over time since there are indications in these series of large changes in variance over the time series, and 3) elucidating the autocorrelation structure of the time series to determine if sudden shifts in abundance occurred.

SUMMARY AND STATUS OF TRENDS: SOUTHERN CALIFORNIA CURRENT, CALCOFI

We report both long-term means and recent trends in this status review. Under the current framework, an indicator is considered to have changed in the short-term if there are significant increasing or decreasing trends over the last five years. An indicator is considered to be above or below long-term norms if the mean of the last five years of the time series differs from the mean of the full time series by more than 1.0 standard deviation.

Anchovy, hake, and cool-water mesopelagics have generally decreased over the last 30 years (Figure C2). The current decline followed a long-term increase in anchovy larvae in the previous 30 years (1950s-70s) (Fig. C2). Such long-term changes hint at a multidecadal fluctuations in abundance, but this cannot be definitively assessed with the relatively short 65-year CalCOFI time series (McClatchie 2012). Summed forage species show the same pattern as anchovy, but that is because anchovy drive the forage fish time series (Fig. C2). Sardine larvae show an entirely different trend to anchovy, and it has been postulated (Chavez et al. 2003) that abundance peaks of these species alternate at decadal time scales (although the CalCOFI time series is too short to evaluate this hypothesis). Sardine larvae in the 1980s and 90s increased from the collapse of the stock in the 1950s. Although there has been a minor decline in sardine larval abundance since 2000, sardine abundance has generally remained above the mean of the last 60 years (Fig. C2). The warm water and cool water-associated oceanic mesopelagic larval assemblages show episodic, strong fluctuations about a mean value that has been relatively stable over the past 60 years.

INDICATOR SELECTION: CENTRAL CALIFORNIA CURRENT, MIDWATER TRAWL SURVEY

INDICATOR EVALUATION: CENTRAL CALIFORNIA CURRENT, MIDWATER TRAWL SURVEY

General description: Central California, midwater trawl survey

We evaluated a number of indicators to represent the coastal pelagic larval abundance in central California. Data are based on mid-water trawl collections. CPS and forage species data series that could potentially be included are krill (Euphausiids), market squid, anchovy, and sardine. In addition, numerous other members of the forage community are available including juvenile salmon, juvenile sanddabs, octopus, juvenile hake, mesopelagics, and juvenile winter-spawned rockfishes (Santora et al. 2012). Each of these data series have been collected since 1990 and some as early as 1983. However, we focus here on the most abundant, continuously present, and available species: anchovy, sardine, market squid, krill, juvenile rockfishes, juvenile sand dabs, and juvenile hake (Table C2). As well, we sum these (minus krill) to represent an index of total forage abundance. Four of these, anchovy, sardine, market squid and krill represent the CPS.

Importantly, the abundance of anchovy and sardine from this survey in central California is not likely to represent overall population strengths as much as it represents variability in the distribution throughout the CCE (Bjorkstedt et al. 2012, Song et al. 2012). As a consequence, unlike the overall trend in CCE, anchovy and sardine are positively correlated for the majority of the time series. We did not include juvenile salmon because the net is inefficient at collecting salmon. Future reports will include additional results for mesopelagic species.

Table C2. Species collected and enumerated in the mid-water trawl survey along Central California.

| Genus species | Common name | Stage | Units |
|-------------------------------|------------------|-----------------|-------------------|
| <i>Citharichthys sordidus</i> | Pacific sanddab | juvenile | CPUE individuals* |
| <i>Engraulis mordax</i> | northern anchovy | adult | CPUE individuals |
| Euphausiids | krill | adult | CPUE individuals |
| <i>Loligo opalescens</i> | market squid | juvenile, adult | CPUE individuals |
| <i>Merluccius productus</i> | Pacific hake | juvenile | CPUE individuals |
| <i>Sardinops sagax</i> | Pacific sardine | adult | CPUE individuals |
| <i>Sebastes spp.</i> | rockfishes | juvenile | CPUE individuals |

*CPUE is catch per unit of effort

This data is based on midwater trawl surveys that have operated annually during May-June from 1983-present. However, all the data used here has been taken only since 1990 as methodology has been consistent since. Samples were collected using a modified Cobb midwater trawl, with a head rope depth of 30 m (the average depth of the thermocline in the region) at a speed of ~2 knots for 15 minutes at depth, with the exception of stations that were too shallow (< ~60m) such as those in the Gulf of the Farallones for which the headrope depth was 10 m (Sakuma et al. 2006). In all cases, samples represent catch per standard 15 minute trawl (CPUE). The data was log-transformed data because it was log-normally distributed.

Appropriate indicators: Central California Current, midwater trawl survey

We examined trends in anchovy and Pacific sardine. Along the CCE northern anchovy abundance variability tends to be positively related to warmer, less productive conditions. In central California, temporal dynamics of northern anchovy abundance likely reflect abundance likely a change in the distribution relative to CCE as a whole (for the May-June period in which the survey is conducted) rather than overall changes in the stock. However, while the anchovy abundance variability is poorly correlated to ocean temperatures in central California, during times of low productivity across the CCE northern anchovy make up a greater proportion of the diets of seabirds locally and, therefore, their relative abundance in the forage community can indicate overall productivity conditions. Along the CCE Pacific sardine abundance variability is positively related to cooler, more productive conditions. In central California, Pacific sardine abundance likely represents a change in the average distribution.

As well, we examined trends in a number of additional fishes that during a period of their life cycle are important contributors to the forage community, including: juvenile Pacific hake, juvenile rockfish, and juvenile Pacific sanddabs. Currently the factors that drive variability in Pacific hake abundance in this survey are not entirely clear as high numbers may represent a strong year class or a shift in the distribution of young-of-the-year. Juvenile and sub-adult hake are an important prey for many other higher trophic level predators. Juvenile rockfish captured in this data series represent juveniles spawned in the current winter (e.g., age 0 individuals). While pelagic, they represent a critical prey resource for predators such as Common murre, rhinoceros auklets and Chinook salmon, and there is a significant relationship between juvenile rockfish abundance and breeding success of seabirds ((Wells et al. 2008a, Field et al. 2010). Pacific sanddabs,

when juveniles, are pelagic and represent a moderately important prey resource for many predators in the region.

Krill is a reasonable indicator of local environmental quality. Krill abundance is known to increase during productive conditions with optimal winds (Cury and Roy 1989). Central California represents a krill hot spot (Santora et al. 2011) where seabirds, mammals, salmon (adult and juveniles), juvenile rockfishes and a number of other species feed on krill. Wells et al. (2008b) and Wells et al (2012) demonstrate the critical role of krill on seabirds, rockfish and salmon. Here, we do not separate the two dominant species of krill in central California (*Euphausia pacifica* and *Thysanoessa spinifera*) because they were not identified to the species level until 2002. However, the two species generally occupy different habitats (inner-shelf vs outer-shelf, (Santora et al. 2012)) and have different life-histories. Predators tend to rely on one of the two species more than the other. For example, juvenile rockfish prey primarily on *E. pacifica* (Reilly et al. 1992) and juvenile salmon on *T. spinifera* (Wells et al. 2012). In the future, as these series continue, we will analyze each species separately.

Monterey Bay is a spawning ground for market squid, and this species forms one of the largest and most lucrative California fisheries. Both juvenile and adult squid make up a significant proportion of the diets of many predators. High market squid abundance is generally positively associated with cool, productive conditions. Data series is log-normally distributed so in these analyses we log-transformed the data.

Summed forage is simply the sum of the fish abundances listed above in Table CPS2. There are clear problems with this data series: not all fish are the same size, the data have not been weighted for seasonality of spawning, and responses to environmental variability may affect the relative abundance of species. We envision this indicator as a first step toward truer measures of forage biomass and suggest this be a gap to be filled in future IEAs.

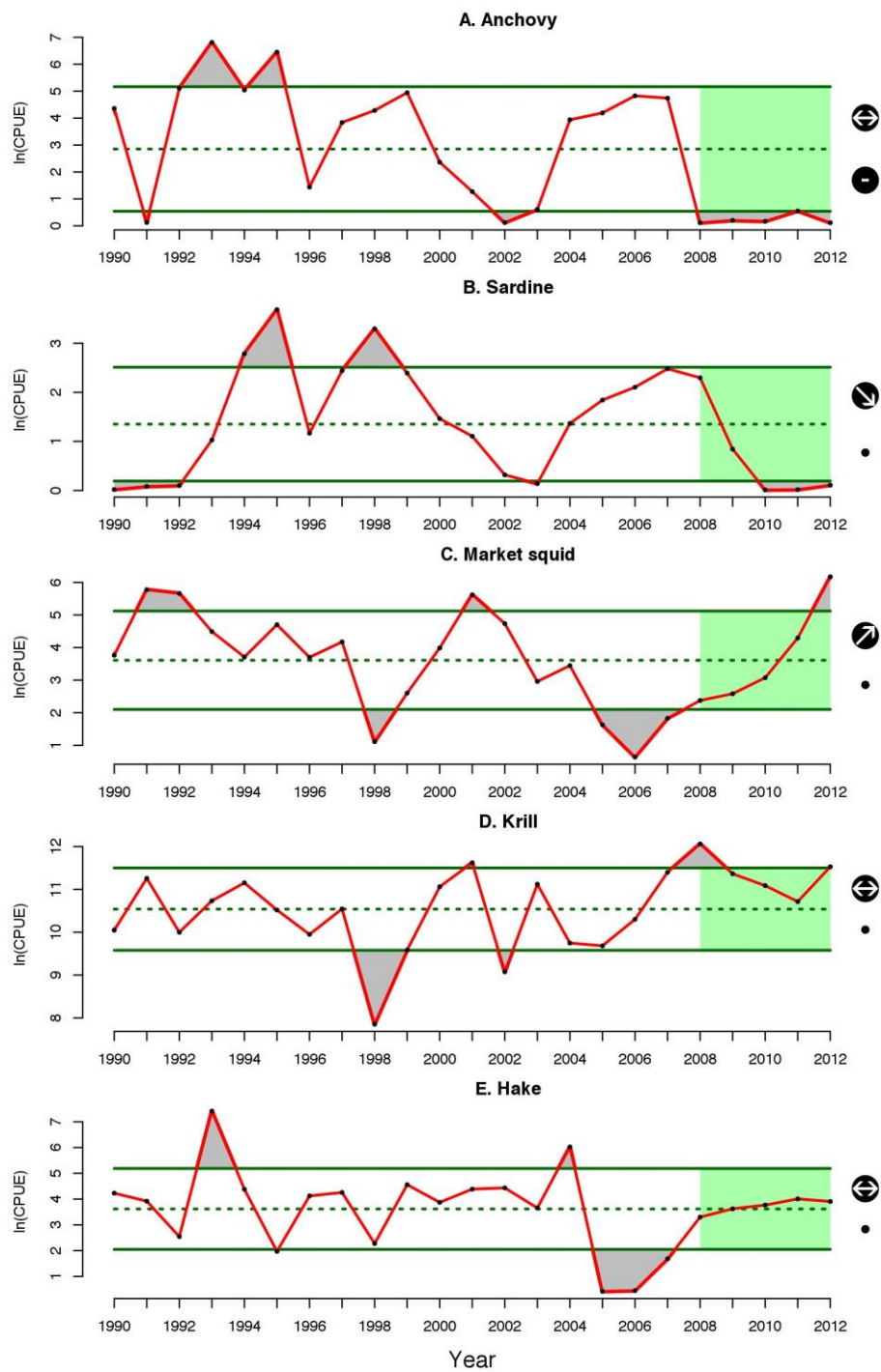
STATUS AND TRENDS: CENTRAL CALIFORNIA CURRENT, MIDWATER TRAWL SURVEY

MAJOR FINDINGS: CENTRAL CALIFORNIA CURRENT, MIDWATER TRAWL SURVEY

2005 and 2006 were poor production years for rockfish, sanddabs, squid and hake; consistent with observations of anomalous ocean conditions and poor reproductive success of higher trophic levels reported elsewhere. In the last five years all but anchovy and sardine abundance were within 1 s.d. of long-term mean (Figure C3). Notably, high abundance of krill over the past five years, and the greatest abundance on record of market squid was found in the last year (Figure C3). Overall, these data series suggest that recent years have been conducive to more production and stable or improved forage abundance, in agreement with Bjorkstedt et al. (2012)

SUMMARY AND STATUS OF TRENDS: CENTRAL CALIFORNIA CURRENT, MID-WATER TRAWL SURVEY

Anchovy are currently below their long-term average and there is no obvious recent trend (Figure C3). Similarly, sardine recently fell below average abundance following a negative trend since 2007. However, the remaining indicators are within 1 s.d. of their long-term means. Rockfish and sanddabs have recovered from the poor years of 2005 and 2006 (Figure C3). Likewise, krill abundance achieved record levels in 2008 and has maintained relatively high abundance in recent years (Figure C3). Finally, market squid is presently experiencing the greatest of the three boom periods of the last 20 years (Figure C3).



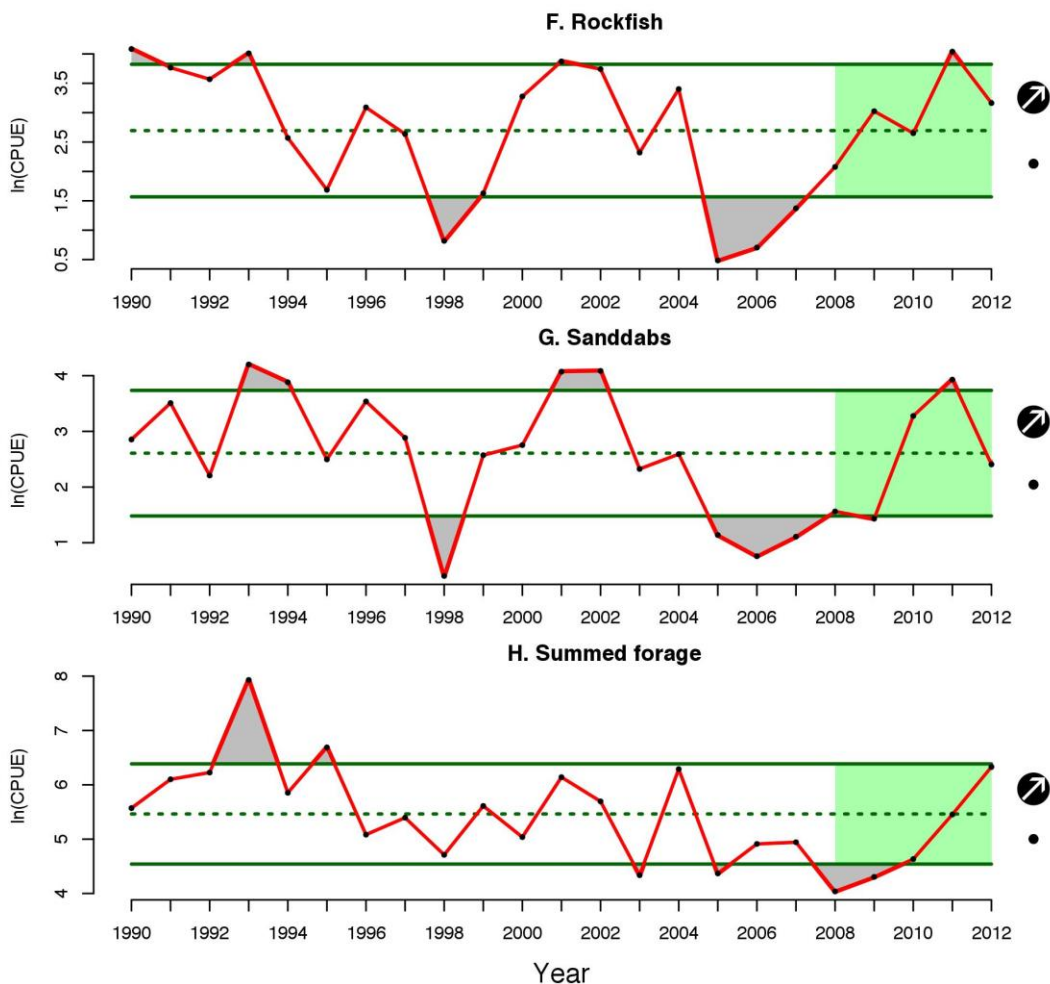


Figure C3. *Central California Forage, mid-water trawl.* Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d. , or was within one 1.0 s.d. of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-) or within (.) one s.d. of the long-term mean.

INDICATOR SELECTION: NORTHERN CALIFORNIA CURRENT, PREDATOR SURVEY

INDICATOR EVALUATION: NORTHERN CALIFORNIA CURRENT, PREDATOR SURVEY

General description: Northern California Current, predator survey

Forage fish data were collected by the NWFSC-NOAA predator surveys along two transects off the Columbia River and Willapa Bay, WA every two weeks from May through August (8-10 cruises a year starting in 1998 (Emmett et al. 2005). However, because the survey was scaled back to just the Willapa Line in 2011

we analyze only samples from this line. All tows were made at the surface to the upper 20 m of the water column during the night. Numbers of individuals were recorded for each species caught in each haul and standardized by the horizontal distance traveled by the towed net. Yearly abundance data was obtained by combining (summing) the standardized count data of each species captured during all cruises for each year.

Appropriate indicators: Northern California Current, predator survey

Table C3. Species collected in the surface trawl of the northern California Current predator survey.

| Genus species | Common name | Stage | Units |
|-----------------------------|------------------|-----------------|------------------|
| <i>Allosmerus elongatus</i> | whitebait smelt | juvenile, adult | CPUE individuals |
| <i>Clupea pallasii</i> | Pacific herring | juvenile, adult | CPUE individuals |
| <i>Engraulis mordax</i> | northern anchovy | juvenile, adult | CPUE individuals |
| <i>Sardinops sagax</i> | Pacific sardine | juvenile, adult | CPUE individuals |

Time series plots of standardized yearly abundance data are presented for each of the four most dominant and consistently collected forage species (Pacific sardine, northern anchovy, Pacific herring and whitebait smelt; Table C3). Although other forage species are caught in these surveys, these four species represent the bulk of the forage fish catch in surface waters at night. They include migratory (sardines and some anchovies) species that may spawn off the Pacific Northwest or migrate from California (Emmett et al. 2005, Litz et al. 2008). Herring and whitebait smelt are likely spawned locally. These species may have seasonal trends in abundance (Emmett et al. 2005) so may have different trends than taken twice a year but over a broader geographical area. The data are log-normally distributed therefore was log-transformed for this analysis.

Summed forage is simply the sum of the fish abundances listed above in Table C3. There are clear problems with this data series: not all fish are the same size, the data have not been weighted for seasonality of spawning, and responses to environmental variability may affect the relative abundance of species. We envision this indicator as a first step toward truer measures of forage biomass and suggest this be a gap to be filled in future IEAs Data series is log-normally distributed so in these analyses we log-transformed the data. Data series are log-normally distributed so in these analyses we log-transformed the data.

STATUS AND TRENDS: NORTHERN CALIFORNIA CURRENT, PREDATOR SURVEY

MAJOR FINDINGS: NORTHERN CALIFORNIA CURRENT, PREDATOR SURVEY

Sardines appear to be undergoing a decline in the northern California Current since their peak levels in 2000-2003 (Fig. C4). Whitebait smelt and summed forage fish are increasing in recent years (Fig. C4).

SUMMARY AND STATUS OF TRENDS: NORTHERN CALIFORNIA CURRENT, PREDATOR SURVEY

Over the last decade, anchovy showed a broad peak in above average abundance lasting 2-3 years, followed by a decline to much lower abundance that has remained relatively stable over the last 5 years. Herring showed a similar pattern to anchovy with higher abundances developing two years earlier than anchovy (Fig. C4). Sardine showed a similar pattern to herring, but declined more slowly and with greater variation. Notably, the sardine decline continued in the last 5 years, and did not stabilize as did anchovy and herring (Fig. C4). Cooler ocean conditions since the El Niño of 2010 may have resulted in decreased

abundance or survival of sardines but may be conducive to higher survival in whitebait smelt which displayed a positive recent trend. The summed forage also demonstrated a modest but significant upward trend.

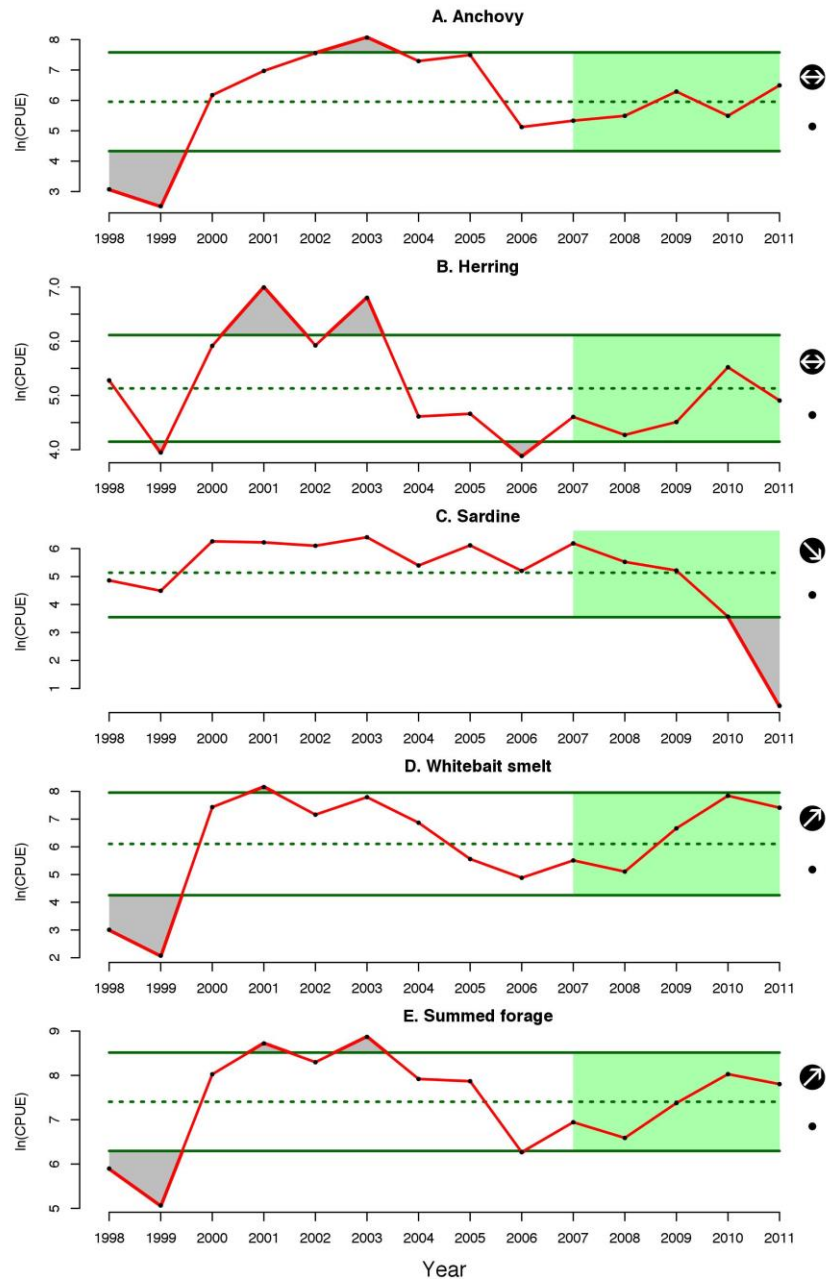


Figure C4. *Northern California Forage, Predator survey.* Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d. , or was within one 1.0 s.d of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-) or within (.) one s.d. of the long-term mean.

INDICATOR SELECTION: NORTHERN CALIFORNIA CURRENT, COLUMBIA RIVER PLUME

INDICATOR EVALUATION: NORTHERN CALIFORNIA CURRENT, COLUMBIA RIVER PLUME

General description: Northern California Current, Columbia River plume

Pelagic nekton catch data were collected by the NWFSC-NOAA Bonneville Power Administration survey surface trawls on standard transects and stations between Tatoosh Island, WA and Cape Perpetua, OR in June and September from 1998 to 2011. All tows were made during the day at predetermined locations along transects extending off the coast to the shelf break (Brodeur et al. 2005). Numbers of individuals were recorded for each species caught in each haul and were standardized by the horizontal distance sampled by the towed net as CPUE (no. km⁻¹ towed). Yearly abundance data were obtained by combining (summing) the standardized count data of each species captured during June and September for each year.

Appropriate indicators: Northern California Current, Columbia River plume

Table C4. Species collected in the surface trawl of the Northern California Current Columbia River plume survey.

| Genus species | Common name | Stage | Units |
|------------------------------|------------------|-----------------|------------------|
| <i>Allosmerus elongatus</i> | whitebait smelt | juvenile, adult | CPUE individuals |
| <i>Clupea pallasii</i> | Pacific herring | juvenile, adult | CPUE individuals |
| <i>Engraulis mordax</i> | northern anchovy | juvenile, adult | CPUE individuals |
| <i>Sardinops sagax</i> | Pacific sardine | juvenile, adult | CPUE individuals |
| <i>Trachurus symmetricus</i> | Jack mackerel | juvenile, adult | CPUE individuals |

Time series plots of standardized yearly abundance data are presented for each of the five most dominant and consistently collected forage species measured (jack mackerel, Pacific sardine, northern anchovy, Pacific herring and whitebait smelt; Table C4). Although other forage species are caught in these surveys, these five species represent the bulk of the forage fish catch in surface waters at night. They include migratory species (sardines and some anchovies) that may spawn off the Pacific Northwest or migrate from California (Emmett et al. 2005, Litz et al. 2008). Jack mackerel can be a forage fish at younger ages but off Oregon and Washington are too large to be fed upon by a number of predators such as seabirds or adult rockfishes. They spawn off southern California and arrive during summer to feed off Oregon and Washington. Herring and whitebait smelt are likely spawned locally. A number of these species may have seasonal trends in abundance (Emmett et al. 2005) so may have different trends than taken twice a year but over a broader geographical area. Because the data are log-normally distributed they were log-transformed for this analysis.

Summed forage is simply the sum of the fish abundances listed above in Table C4. There are clear problems with this data series: not all fish are the same size, the data have not been weighted for seasonality of spawning, and responses to environmental variability may affect the relative abundance of species. We envision this indicator as a first step toward truer measures of forage biomass and suggest this be a gap to be filled in future IEAs.

STATUS AND TRENDS: NORTHERN CALIFORNIA CURRENT, COLUMBIA RIVER PLUME

MAJOR FINDINGS: NORTHERN CALIFORNIA CURRENT, COLUMBIA RIVER PLUME

The environment has fluctuated during the period since 1998 between relatively cool years (2008, 2011) to warm El Niño years (2010) (Bjorkstedt et al. 2012), likely leading to great variability in jack mackerel, Pacific herring, and sardine.

SUMMARY AND STATUS OF TRENDS: NORTHERN CALIFORNIA CURRENT, COLUMBIA RIVER PLUME

Jack mackerel also show episodic outbreaks early in the decade, followed by a decline in the last 7 years to recent below average abundance (Figure C5). Herring shows a consistent, if non-monotonic declining trend over the last decade (Figure C5). The smelt population appears to be stable recently following two periods in 1999/2000 and 2006 of below average values (Figure C5). Anchovy has remained near above average abundance for much of the last ten years following well below average values in the late 90's and early 2000's (Figure C5). By contrast, sardine abundance was below average in 2008 and 2010 but there is apparent significant trend in recent years.

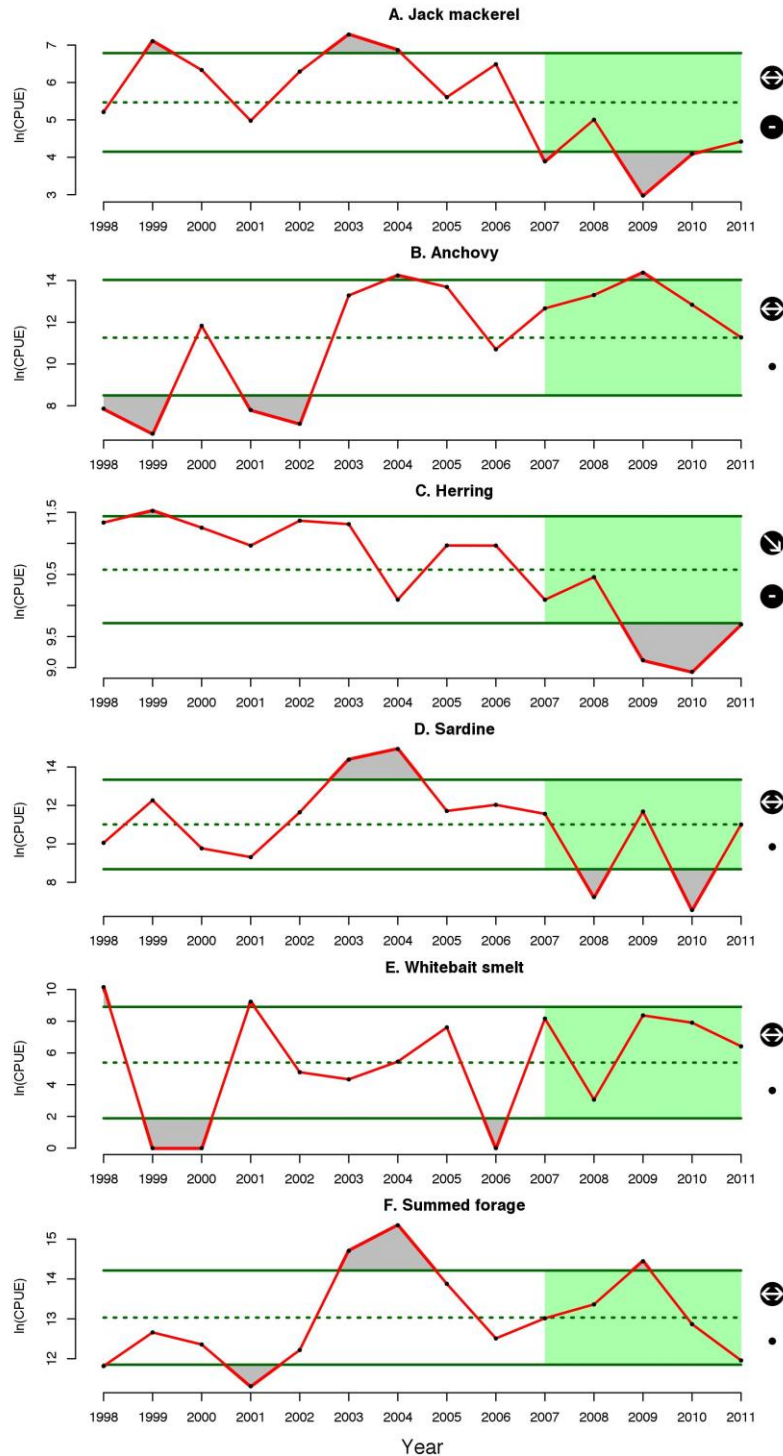


Figure C5. Northern California Forage, Columbia River plume. Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d., or was within one 1.0 s.d. of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-) or within (.) one s.d. of the long-term mean

INDICATOR SELECTION: ASSESSMENTS

INDICATOR EVALUATION: ASSESSMENTS

General description: Assessments

Pacific mackerel (Crone et al. 2011) and sardine (Hill et al. 2011) assessments are prepared for the PFMC annually to be used for developing harvest rules. These assessments incorporate data from a number of sources and determine the biomass and age distribution of the populations along the coast. They represent the most complete analysis of the abundance trends across the CCE. Therefore, we use these assessments to guide our estimation of population trends of abundance and condition for these two species.

The Pacific mackerel assessment is an age-structured model incorporating information on catch, length and age distributions, and recreational fishery surveys (Crone et al. 2011). Full model details, problems and uncertainties are disclosed at <http://www.pcouncil.org/coastal-pelagic-species/stock-assessment-and-fishery-evaluation-safe-documents/> and in Crone et al. (2011).

The sardine assessment includes fishery and survey data, egg production, aerial survey estimates of biomass, and acoustic estimates of biomass. Full model details, problems, uncertainties are disclosed at <http://www.pcouncil.org/coastal-pelagic-species/stock-assessment-and-fishery-evaluation-safe-documents/> and in Hill et al. (2011).

Appropriate indicators: Assessments

We focus on three indicators representing abundance and condition of Pacific mackerel and sardine. To estimate abundance trends we evaluate the biomasses of the two species.

1. Biomass of sardine is the most recent estimate provided by a member of the stock assessment team (Hill et al. 2011) in June 2012. Units are log-transformed metric tons.
2. Biomass of Pacific mackerel is the most recent estimate provided by a member of the stock assessment team (Crone et al. 2011) in June 2012. Units are log-transformed metric tons.
3. We evaluate the condition of sardine by examining their age distribution. A full and stable age distribution indicates that a population has had recruitment successfully for a number of years, older large fish are present in the population, and the fishery may be less prone to collapse. However following a few years of reduced recruitment (and typically lower biomass) the age structures can be weighted more toward older fish. Likewise, when there is a boom year the vast majority of the biomass will shift to younger fish that could destabilize the age structure for a number of years following. A gap in our analysis is to develop a more appropriate and interpretable index of condition.

STATUS AND TRENDS: ASSESSMENTS

MAJOR FINDINGS: ASSESSMENTS

In recent years the biomasses of Pacific mackerel and sardine have been average relative to the long-term mean yet, for sardine, the recent values are greater than the period following the population crash between 1950 and the early 1990s.

SUMMARY AND STATUS OF TRENDS: ASSESSMENTS

In the first half of the 20th century both Pacific mackerel and sardine were relatively abundant. In the late 1970s and 1980s Pacific mackerel demonstrated above average production but production has declined in the past two decades. In the last five years population estimates of biomass are within 1 s.d. of the long-term mean and there is no apparent trend (Figure C6). Similarly, sardine experienced near-average production in the past 10-20 years yet the estimates of biomass are with 1 s.d. of the long-term mean suggesting that, while the abundance is greater in recent years, it is still only a portion of that observed in the earlier part of the 20th century (Figure C7).

Sardine can live to greater than 10 years (we grouped 10+ into the 10 age category). The age structure is heavily weighted by the youngest ages, as is typical, but following major recruitment events (such as 2004) the age range can be quite large (Figure C7). During 1999-2001 recruitment was somewhat low relative to the other years shown as can be observed by the age structure being more heavily weighted by three and four year olds. Following a number of poor recruitment years as the older fish died out the biomass dropped to a low in 2003. In 2004 a larger recruitment event occurred and the biomass rebounded by 2006.

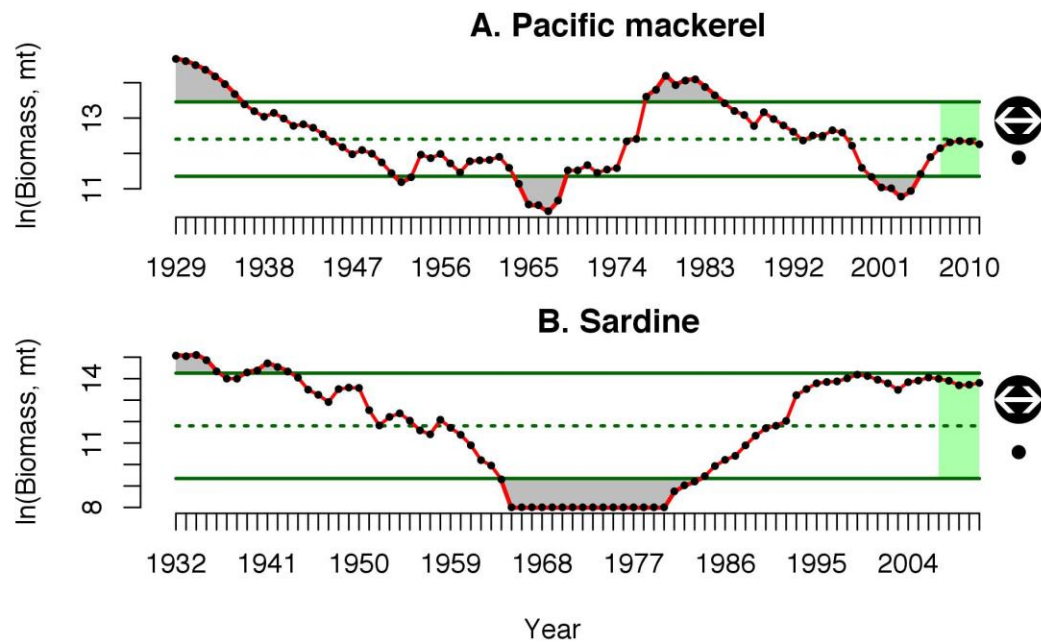


Figure C6. *Assessment biomasses.* Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d. , or was within one 1.0 s.d of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-) or within (.) one s.d. of the long-term mean.

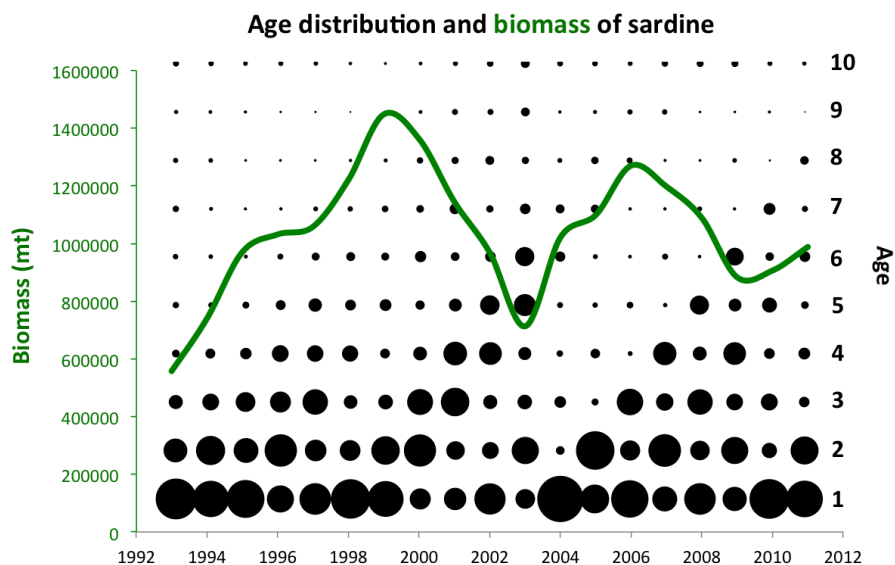


Figure C7. Assessed age structure. Shown are the biomass of sardines and the proportions of various ages (y-axis) in the population of sardine from 1993 to 2011.

RISK

We are not directly evaluating risk in this IEA. In the future, we will put efforts toward quantifying the relationships between environmental conditions and CPS responses (e.g., Brodeur et al. 2005, Song et al. 2012) and exposure to fishing and other anthropogenic impacts in such a way as to represent the potential risk posed to the CPS and forage community.

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GROUND FISH

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TABLE OF CONTENTS (GF)

| | |
|--------------------------------|-----|
| Executive Summary | 407 |
| Detailed Report..... | 410 |
| Summary: Indicators..... | 410 |
| Indicator Selection..... | 410 |
| Top Ranked Indicators | 411 |
| Final Suite of Indicators..... | 427 |
| Status and Trends..... | 431 |
| References cited | 468 |

LIST OF TABLES AND FIGURES (GF)

| | |
|--|-----|
| Figure GF1. Stock status plot relative to being overfished (x-axis) and overfishing (y-axis) for all species assessed since 2007. Vertical broken line indicates the target biomass reference point. Vertical solid line indicates the limit reference point indicating an overfished status (red for elasmobranchs, rockfishes, and roundfishes; purple for flatfishes)..... | 408 |
| Table GF1. Results for each stock evaluated for each of two status indicators: 1) Biomass and 2) Population structure. Two sources of information were used: 1) Stock assessments and 2) Northwest Fisheries Science Center (NWFSC) shelf-slope trawl survey, when assessments were not available, or older than 2007. | 409 |
| Table GF1. Selected key attributes for each goal. Relevant measures describe what each attribute means (e.g., population size is represented by the number of individuals in a population or the total biomass)..... | 411 |
| Table GF2. Summary of groundfish population size indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, CalCOFI egg/larvae abundance has peer-reviewed literature supporting two out of five primary considerations criteria. | 413 |
| Table GF3. Summary of groundfish population condition indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Cortisol/vitellogenin has peer-reviewed literature supporting two out of five primary considerations criteria. | 417 |
| Figure GF1. Example of the number and spatial extent of locations (triangles) surveyed by the West Coast groundfish trawl survey each year during 2003–2010. (Reprinted from Keller et al. 2008.)..... | 426 |
| Table GF5. List of groundfish for which the aforementioned indicators will be calculated. This list is composed of species in assemblages identified in Cope and Haltuch (2012), species with quantitative stock assessments completed from 2007-2011, and species that are well surveyed by the NWFSC annual trawl survey. | 429 |
| Table GF6. Top indicators for Attributes 1 and 2..... | 431 |
| Figure GF2. Stock status plot relative to being overfished (x-axis) and overfishing (y-axis) for all species assessed since 2007. Vertical broken line indicates the target biomass reference point. | 433 |
| Table GF7. Results for each stock evaluated for each of two status indicators: 1) Biomass and 2) Population structure. Two sources of information were used: 1) Stock assessments and 2) Northwest Fisheries Science Center (NWFSC) shelf-slope trawl survey, when assessments were not available, or older than 2007..... | 434 |
| Figure GF3. Relative abundance trajectory 1916-2007 for longnose skate..... | 435 |
| Figure GF4. Trawl survey design-based estimates of longnose skate biomass for years 2003-2011..... | 436 |
| Figure GF5. Relative abundance trajectory 1916-2011 for spiny dogfish. | 436 |
| Figure GF6. Trawl survey design-based estimates of spotted ratfish biomass for years 2003-2011..... | 436 |

| | |
|--|-----|
| Figure GF7. Relative abundance trajectory 1916-2007 for arrowtooth flounder..... | 437 |
| Figure GF8. Trawl survey design-based estimates of arrowtooth flounder biomass for years 2003-2011. | 437 |
| Figure GF9. Relative abundance trajectory 1876-2007 for English sole. | 438 |
| Figure GF10. Trawl survey design-based estimates of English sole biomass for years 2003-2011. | 438 |
| Figure GF11. Trawl survey design-based estimates of English sole biomass for years 2003-2011. | 439 |
| Figure GF12. Relative abundance trajectory 1876-2011 for Petrale sole..... | 439 |
| Figure GF13. Relative abundance trajectory 1910-2011 for Dover sole..... | 439 |
| Figure GF14. Trawl survey design-based estimates of flathead sole biomass for years 2003-2011..... | 440 |
| Figure GF15. Trawl survey design-based estimates of English sole biomass for years 2003-2011. | 440 |
| Figure GF16. Relative abundance trajectory 1916-2009 for black rockfish..... | 441 |
| Figure GF17. Relative abundance trajectory 1890-2011 for bocaccio..... | 441 |
| Figure GF18. Relative abundance trajectory 1916-2011 for canary rockfish..... | 442 |
| Figure GF19. Relative abundance trajectory 1892-2011 for chilipepper..... | 442 |
| Figure GF20. Trawl survey design-based estimates of chilipepper biomass for years 2003-2011. | 443 |
| Figure GF21. Relative abundance trajectory 1900-2011 for cowcod..... | 443 |
| Figure GF22. Relative abundance trajectory 1910-2011 for darkblotched rockfish..... | 443 |
| Figure GF23. Relative abundance trajectory 1916-2011 for greenspotted rockfish..... | 444 |
| Figure GF24. Relative abundance trajectory 1910-2011 for greenstriped rockfish..... | 444 |
| Figure GF25. Relative abundance trajectory 1940-2011 for Pacific Ocean perch..... | 445 |
| Figure GF26. Trawl survey design-based estimates of restripe rockfish biomass for years 2003-2011..... | 445 |
| Figure GF27. Trawl survey design-based estimates of shortbelly rockfish biomass for years 2003-2011. | 445 |
| Figure GF28. Trawl survey design-based estimates of stripetail rockfish biomass for years 2003-2011. | 446 |
| Figure GF29. Relative abundance trajectory 1916-2011 for widow rockfish..... | 446 |
| Figure GF30. Trawl survey design-based estimates of yellowtail rockfish biomass for years 2003-2011. | 447 |
| Figure GF31. Trawl survey design-based estimates of aurora rockfish biomass for years 2003-2011..... | 447 |
| Figure GF32. Relative abundance trajectory 1950-2011 for Dover sole..... | 447 |
| Figure GF33. Relative abundance trajectory 1910-2011 for Dover sole..... | 448 |
| Figure GF34. Relative abundance trajectory 1916-2011 for yelloweye rockfish..... | 448 |

| | |
|---|-----|
| Figure GF35. Relative abundance trajectory 1910-2011 for cabezon. | 449 |
| Figure GF36. Relative abundance trajectory 1910-2011 for lincod. | 449 |
| Figure GF37. Relative abundance trajectory 1910-2011 for Pacific hake. | 450 |
| Figure GF38. Relative abundance trajectory 1910-2011 for sablefish. | 450 |
| Figure GF39. Proportion of the longnose skate population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series. | 451 |
| Figure GF40. Proportion of the longnose skate population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series. | 452 |
| Figure GF41. Proportion of the spiny dogfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series. | 452 |
| Figure GF42. Proportion of the spotted ratfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2005) of the trawl survey time series. | 453 |
| Figure GF43. Proportion of the arrowtooth flounder population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series. | 453 |
| Figure GF44. Proportion of the arrowtooth flounder population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series. | 454 |
| Figure GF45. Proportion of the English sole population mature (red) and at the 95% cumulative age (blue) relative to the first year (1876) of the time series. | 454 |
| Figure GF46. Proportion of the English sole population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series. | 455 |
| Figure GF47. Proportion of the Pacific sanddab population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series. | 455 |
| Figure GF48. Proportion of the Petrale sole population mature (red) and at the 95% cumulative age (blue) relative to the first year (1876) of the time series. | 455 |
| Figure GF49. Proportion of the Dover sole population mature (red) and at the 95% cumulative age (blue) relative to the first year (1910) of the time series. | 456 |
| Figure GF50. Proportion of the flathead sole population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series. | 456 |
| Figure GF51. Proportion of the rex sole population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series. | 457 |
| Figure GF52. Proportion of the black rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series. | 457 |
| Figure GF53. Proportion of the bocaccio population mature (red) and at the 95% cumulative age (blue) relative to the first year (1895) of the time series. | 458 |
| Figure GF54. Proportion of the canary rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series. | 458 |

| | |
|---|-----|
| Figure GF55. Proportion of the chilipepper rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1892) of the time series (1892-2007)..... | 459 |
| Figure GF56. Proportion of the chilipepper rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series (2003-2007)..... | 459 |
| Figure GF57. Proportion of the cowcod population mature (red) and at the 95% cumulative age (blue) relative to the first year (1900) of the time series. | 460 |
| Figure GF58. Proportion of the darkblotched rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1895) of the time series..... | 460 |
| Figure GF59. Proportion of the greenspotted rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series..... | 460 |
| Figure GF60. Proportion of the greenstriped rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series..... | 461 |
| Figure GF61. Proportion of the Pacific Ocean perch population mature (red) and at the 95% cumulative age (blue) relative to the first year (1940) of the time series..... | 461 |
| Figure GF62. Proportion of the redstripe rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series. | 462 |
| Figure GF63. Proportion of the shortbelly rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series. | 462 |
| Figure GF64. Proportion of the striptail rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series. | 462 |
| Figure GF65. Proportion of the widow rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series..... | 463 |
| Figure GF66. Proportion of the yellowtail rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series. | 463 |
| Figure GF67. Proportion of the aurora rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series. | 464 |
| Figure GF68. Proportion of the blackgill rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1950) of the time series..... | 464 |
| Figure GF69. Proportion of the splitnose rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1900) of the time series..... | 465 |

| | |
|---|-----|
| Figure GF70. Proportion of the yelloweye rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series..... | 465 |
| Figure GF71. Proportion of the cabezon population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series. | 466 |
| Figure GF72. Proportion of the lingcod population mature (red) and at the 95% cumulative age (blue) relative to the first year (1930) of the time series. | 466 |
| Figure GF73. Proportion of the sablefish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1900) of the time series. | 467 |

OVERVIEW

The majority of West Coast groundfish populations have a stable abundance and population structure. However, 10% of groundfish stocks are declining in abundance and nearly 20% of stocks show declines in the percentage of mature individuals.

EXECUTIVE SUMMARY

Groundfish are an important component of the California Current. We identified population size and population condition as key attributes of groundfish stocks, and we identified and evaluated 46 indicators of these two attributes for use on the 90+ groundfish stocks of the California Current. We selected two indicators as indicators of population size: 1) biomass of groundfish relative to either the estimate of unfished biomass (when a stock assessment is available) or trends in the survey time series, and 2) the number of assessed species below management thresholds. Additionally, two indicators of population condition were selected: 1) the proportion of the population mature (using ages or size in the absence of ages), and 2) the 95% cumulative age or length of the population. In general, data sources that relied on fishery-independent data performed better during the indicator selection process than fishery-dependent data sources (e.g., commercial landings numbers, total catch). In addition to these indicators, groundfish data were used in construction of several indicators in the Ecological Integrity chapter of this report.

We summarized the status of stocks (based on biomass trends) and population demographic condition (as measured by the percentage of mature individuals and of maximum age or size) for 30 groundfishes. The remaining species did not have sufficient data to determine their status at this time. We found that most assessed groundfishes are above the biomass limit reference point, and thus are not overfished (Figure GF_i). The four assessed stocks currently in an overfished state are all rockfishes. All assessed groundfishes are below their target catch, thus overfishing is not occurring in these stocks. With respect to population condition measures, we discovered that age or length structure tended to show more changes (usually declines) over time than the proportion that are sexually mature. We also found that non-elasmobranch groundfishes tended to exhibit the most changes over time in both measures, with rockfishes being most sensitive to demographic changes. The development of additional data-limited methods may allow more species to be included in future iterations of the IEA.

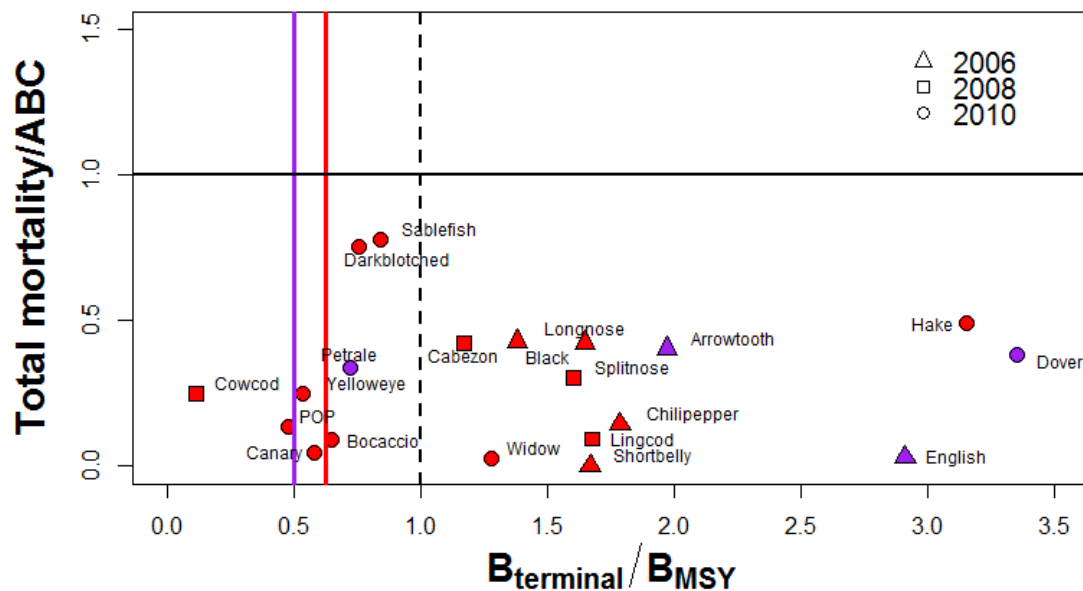


Figure GFi. Stock status plot relative to being overfished (x-axis) and overfishing (y-axis) for all species assessed since 2007. Vertical broken line indicates the target biomass reference point. Vertical solid line indicates the limit reference point indicating an overfished status (red for elasmobranchs, rockfishes, and roundfishes; purple for flatfishes). Horizontal line indicates overfishing wherein total mortality exceeds the allowable biological catch (ABC). For example, sablefish is below the target (black vertical broken line), but above the limit (red vertical solid line) biomass target, and below the overfishing limit (horizontal solid line). Symbols indicate the terminal year of the assessment in which the reference points are determined.

Table GF*i*. Results for each stock evaluated for each of two status indicators: 1) Biomass and 2) Population structure. Two sources of information were used: 1) Stock assessments and 2) Northwest Fisheries Science Center (NWFSC) shelf-slope trawl survey, when assessments were not available, or older than 2007.

“Depletion” refers to the relative change in spawning biomass; “5-year trend” is the trend in the last 5 years of the time series (details found in the text). “B final year” is the biomass value in the final year compared to the 5-year average. “Prop. mature” is proportion of the population mature relative to the beginning of the time series; “95% cum.” refers to the 95% cumulative age or length of the population relative to the beginning of the time series. +: above target limit or increasing; ●: between target and limit or stable; -: below limit or decreasing. Blank spaces indicate no information reported.

| Taxa | Stock | Biomass | | | | Population structure | | | |
|--------------|---------------------|------------|------------|--------------|------------|----------------------|--------------|--------------|--------------|
| | | Assessment | | NWFSC Survey | | Assessment | | NWFSC Survey | |
| | | Depletion | 5-yr trend | B final year | 5-yr trend | Prop. mature | 95% cum. age | Prop. mature | 95% cum. lt. |
| Elasmobranch | Longnose skate | + | ● | ● | ● | ● | - | ● | ● |
| | Spiny dogfish | + | ● | | | ● | - | | |
| | Spotted ratfish | | | ● | ● | | | + | ● |
| Flatfishes | Arrowtooth flounder | + | + | ● | + | - | - | - | - |
| | English sole | + | + | ● | ● | + | - | - | ● |
| | Pacific sanddab | | | ● | + | | | ● | ● |
| | Petrable sole | ● | ● | | | - | - | | |
| | Dover sole | + | ● | | | ● | ● | | |
| | Flathead sole | | | ● | ● | | | ● | ● |
| | Rex sole | | | ● | - | | | ● | ● |
| | Black | + | + | | | - | - | | |
| | Bocaccio | ● | ● | | | - | - | | |
| Rockfishes | Canary | - | ● | | | - | - | | |
| | Chilipepper | + | ● | ● | ● | ● | - | - | + |
| | Cowcod | - | ● | | | - | - | | |
| | Darkblotched | ● | + | | | - | - | | |
| | Greenspotted | ● | + | | | - | - | | |
| | Greenstriped | + | + | | | ● | - | | |
| | Pacific Ocean perch | - | ● | | | - | - | | |
| | Redstriped | | | + | ● | | | - | ● |
| | Shortbelly | | | ● | ● | | | ● | ● |
| | Stripetail | | | ● | ● | | | ● | ● |
| | Widow | + | + | | | ● | - | | |
| | Yellowtail | | | ● | ● | | | ● | ● |
| | Aurora | | | - | - | | | - | ● |
| | Blackgill | ● | ● | | | - | - | | |
| | Splitnose | + | + | | | - | - | | |
| | Yelloweye | - | ● | | | - | - | | |
| Roundfishes | Cabazon | + | + | | | - | - | | |
| | Lingcod | + | + | | | - | - | | |
| | Pacific Hake | + | + | | | | | | |
| | Sablefish | ● | - | | | ● | ● | | |

DETAILED REPORT

SUMMARY: INDICATORS

Forty six indicators of two attributes, population size and population condition, were evaluated. Two indicators were chosen for population size: 1) biomass of groundfish relative to either the estimate of unfished biomass (when a stock assessment is available) or trends in the survey time series, and 2) the number of assessed species below management thresholds. Two indicators were chosen for population condition, the proportion of the population mature (using ages or size in the absence of ages) and the 95% cumulative age or length of the population. Thirty groundfish species were identified for which these indicators would be calculated.

INDICATOR SELECTION

BACKGROUND - GROUND FISH

Groundfish are generally defined as a community of fishes that are closely associated with the ocean bottom. In the CCLME, some of the better known species include the rockfishes (Scorpaenidae), flatfishes (Pleuronectidae and Bothidae), sculpins (Cottidae), Pacific hake, sablefish (*Anoplopoma fimbria*), greenlings and lingcod (Hexagrammidae), skates (Rajidae), and benthic sharks (PFMC 2008a). Similar to most fishes, many groundfish species have a planktonic larval and young-of-year life history stage in which young fish inhabit surface waters and feed on a diet of zooplankton. After a few months in the plankton, most species settle to the bottom, generally moving to deeper waters and they age/grow. Groundfish vary across a wide range of trophic levels and inhabit all types of habitats (e.g., rocky, sandy, muddy, kelp) from the intertidal zone to the abyss and have generally variable recruitment, often mature late, and are long lived.

This community of fishes constitutes a large biomass in the CCLME and provides the economic engine for coastal communities in Washington, Oregon, and California. The Pacific Fishery Management Council (PFMC) manages a subset of groundfish species that are typically captured during fishing operations along the U.S. West Coast. Those species caught in the Pacific groundfish trawl fishery were worth approximately \$40 million in 2009 (NOAA Press Release 2010). Thus, understanding how groundfish populations fare over time is of great interest to ecosystem managers and the coastal communities that derive much of their wealth from this assemblage of fishes.

SELECTION PROCESS

Forty six potential indicators of groundfish population size and population condition were evaluated using the ecological literature as a basis for their rankings (see Levin and Schwing 2011 for detailed methods). Two indicators were chosen for population size: 1) biomass of groundfish relative to either the estimate of unfished biomass (when a stock assessment is available) or trends in the survey time series, and 2) the number of assessed species below management thresholds. Two indicators were chosen for population condition: 1) the proportion of the population mature (using ages or size in the absence of ages) and 2) the 95% cumulative age or length of the population. Thirty groundfish species were identified for which these indicators could be calculated.

POPULATION SIZE

Monitoring population size in terms of total number or total biomass is important for management and societal interests. For example, abundance estimates are used to track the status of threatened and endangered species and help determine whether a species is recovering or declining. Accurate population biomass estimates of targeted fisheries species are used to assess stock viability and determine the number of fish that can be sustainably harvested from a region. While population size can be used to assess population viability, more accurate predictions of viability can be obtained by including the mechanisms responsible for the dynamics of the population. Population dynamics thus provide a predictive framework to evaluate the combined effect of multiple mechanisms of population regulation (e.g., recruitment, mortality, immigration, and emigration) to evaluate changes in abundance through time.

POPULATION CONDITION

Whereas the preceding attribute is concerned with measures of population size, there are instances when the health of the population may be of interest. For example, monitoring changes in population condition may presage an effect on population size or provide insight into long-term population viability. The dynamics of many populations are better understood through knowledge of population conditions such as organism condition, age structure, genetic diversity, phenotypic diversity, and population structure. Impaired condition of any or all of these subcategories indicates biological resources at risk. In addition, monitoring changes in population condition could be used to infer changes in environmental conditions.

Table GF1. Selected key attributes for each goal. Relevant measures describe what each attribute means (e.g., population size is represented by the number of individuals in a population or the total biomass).

| Goal | Key attribute | Relevant measures |
|-------------|----------------------|--|
| Groundfish, | Population size | Number of individuals or total biomass, population dynamics, population size relative to unfished conditions (depletion) |
| | Population condition | Measures of population or organism condition including: age structure, population structure, phenotypic diversity, genetic diversity, organism condition |

TOP RANKED INDICATORS

A total of 46 indicators were evaluated for the two key attributes: population size and population condition. In general, the indicators that were evaluated scored well against the primary considerations criteria; however, when indicators performed poorly, it was generally because data were not available at large spatial scales or across long time series.

ATTRIBUTE 1 - POPULATION SIZE

First three primary indicators that are obvious and well established were evaluated: numbers of individuals, total biomass of the population (when model estimates were available), relative biomass (when survey indices were available), and population growth rate. These indicators performed well across all three evaluation criteria categories and are supported as indicators of population size by primary literature sources (e.g., Fulton et al. 2005, Link 2005). However, the ability of scientists and managers to estimate the abundance or growth rate of any population of groundfish over time relies on survey indices of relative abundance. Thus, data sets that measure the relative abundance or biomass of groundfish populations over time (fishery dependent and fishery independent) are evaluated, providing an evaluation of the strengths and weaknesses of various data sources that estimate groundfish population size. A total of 29 potential indicators of population size in the CCLME are identified and evaluated (Table GF2).

In general, data sources that relied on fishery-dependent data (e.g., commercial landings numbers, total harvest biomass) did not perform well against the primary considerations evaluation criteria. For example, recreational landings data are generally collected at docks and only include individuals and species that are kept by fishers. Thus these data are highly biased by fisher behavior in what species are targeted and what species or individuals they retain. When fishery-independent indicators did not perform well, it was generally because these data sources the surveys did not occur at large spatial scales or over long time scales (e.g., NWFSC's hook-and-line surveys, scuba surveys). Interestingly, "local ecological knowledge" scored well in the primary considerations categories, but these interviews of people's memories simply do not exist for most of the CCLME. One attempt in Puget Sound by Beaudreau et al. (2011) has shown a correlation between relative abundance trends of marine species derived from interviews with fishers and divers and scientifically collected relative abundance survey data.

ATTRIBUTE 2 - POPULATION CONDITION

Seventeen potential groundfish indicators (Table GF3) were identified and evaluated. Indicators related to age structure, fecundity, or spatial structure of populations generally scored well in the primary considerations categories. Many condition indicators did not score well in the data considerations categories because there is simply little data available across the entire CCLME or data do not exist at multiple periods through time. For example, age at maturity and genetic diversity score high in primary considerations, but there are few examples from a limited number of species in which these data have been collected or processed. Collecting the data (e.g., gonads or fin clips) is relatively easy to do during bottom trawl surveys, but processing the samples can be expensive and taxing for current staff levels.

Table GF2. Summary of groundfish population size indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, CalCOFI egg/larvae abundance has peer-reviewed literature supporting two out of five primary considerations criteria.

| Attribute | Indicator | Primary considerations (5) | Data considerations (7) | Other considerations (6) | Summary comments |
|-----------------|--|----------------------------|-------------------------|--------------------------|--|
| Population Size | Biomass | 5 | 7 | 4 | While biomass for each species is an obvious indicator for individual species, aggregate groundfish biomass is not necessarily indicative of the state of the entire groundfish community due to changes in a few large components of the community. |
| Population Size | Numbers | 5 | 7 | 4 | Similar comment as for biomass above. |
| Population Size | Population growth rate | 4 | 5 | 5 | Theoretically sound and can be calculated at numerous spatial and temporal scales as datasets can be integrated. |
| Population Size | Number of groups below management thresholds | 3 | 5 | 5 | Good snapshot of species trends over time, but only 30 of 90 managed groundfish species are assessed. |
| Population Size | Stock assessment estimated biomass | 5 | 7 | 5 | Stock assessments perform well for data-rich species. Similar to above, only 30 of 90 groundfish species are assessed. |
| Population Size | Bottom trawl survey relative biomass | 5 | 7 | 3 | Multiple surveys have occurred; surveys generally provide two large scale independent time series, one from 1980 to 2004 and the second from 2003 to 2010. |

| | | | | | |
|-----------------|---------------------------------------|---|---|---|--|
| Population Size | Bottom trawl survey relative numbers | 5 | 7 | 3 | Multiple surveys have occurred; surveys generally provide two large scale independent time series, one from 1980 to 2004 and the second from 2003 to 2010. |
| Population Size | Hake acoustic survey relative biomass | 4 | 5 | 3 | Effective indicator for the most abundant groundfish species in the CCLME, but may not reflect trends of other species. Survey has greater uncertainty when Humboldt squid are present. |
| Population Size | Hake acoustic survey relative numbers | 4 | 0 | 0 | Acoustic surveys generally calculate biomass, not numbers. |
| Population Size | Prerecruit survey relative biomass | 3 | 3 | 3 | The survey provides data on a limited number of species, is temporally limited, and has been historically centered on San Francisco (30 year time series). Since 2001 the survey has covered most of the U.S. West Coast between Cape Flattery and the U.S./Mexico border. |
| Population Size | Prerecruit survey relative numbers | 3 | 3 | 3 | Similar comment as above. |
| Population Size | Hook-and-line survey relative biomass | 5 | 3 | 3 | Survey is limited in spatial scale, but provides relative biomass trends in untrawlable habitats in the Channel Islands, California. |
| Population Size | Hook-and-line survey relative numbers | 5 | 3 | 3 | Similar comment as above. |
| Population Size | PISCO scuba surveys relative biomass | 5 | 0 | 0 | Scuba surveys are limited in spatial scale and highly variable for cryptic species. |

| | | | | | |
|-----------------|---|---|---|---|---|
| Population Size | PISCO scuba surveys relative numbers | 5 | 4 | 3 | Similar comment as for PISCO scuba surveys biomass above. |
| Population Size | National Park Service kelp monitoring relative survey biomass | 5 | 0 | 0 | Similar comment as for PISCO scuba surveys biomass above. |
| Population Size | National Park Service kelp monitoring relative survey numbers | 5 | 4 | 3 | Similar comment as for PISCO scuba surveys numbers above. |
| Population Size | IPHC longline survey relative biomass | 4 | 2 | 3 | Longline surveys are useful for a small number of species. |
| Population Size | IPHC longline survey relative numbers | 4 | 2 | 3 | Similar comment as above. |
| Population Size | CalCOFI egg/larvae relative abundance | 2 | 3 | 3 | Survey most effective for limited pelagic species, limited information for groundfish. DNA methods needed to ID most larval rockfish. |
| Population Size | Pot surveys relative biomass | 1 | 1 | 3 | Variation in behavior of fish biases these passive survey methods. Survey no longer occurs. |
| Population Size | Pot surveys relative numbers | 1 | 1 | 3 | Similar comment as above. |
| Population Size | Commercial landings biomass | 1 | 3 | 1 | Fishery-dependent data biased toward fisher behavior, fleet dynamics and management restrictions. Only economically valuable species. |

| | | | | | |
|-----------------|--|---|---|---|--|
| Population Size | Commercial landings numbers | 1 | 2 | 1 | Similar comment as above. |
| Population Size | Recreational landings biomass | 1 | 3 | 1 | Similar comment as above. |
| Population Size | Recreational landings numbers | 1 | 3 | 1 | Similar comment as above. |
| Population Size | Total harvest biomass, catch per unit effort | 1 | 4 | 1 | Similar comment as above. |
| Population Size | Bycatch abundance | 0 | 5 | 4 | Levels of bycatch are heavily influenced by fisher behavior and management restrictions. |
| Population Size | Local ecological knowledge | 4 | 1 | 4 | Theoretically sound, but limited data throughout the CCLME. |

Table GF3. Summary of groundfish population condition indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Cortisol/vitellogenin has peer-reviewed literature supporting two out of five primary considerations criteria.

| Attribute | Indicator | Primary consider- ations (5) | Data consider- ations (7) | Other consider- ations (6) | Summary comments |
|-------------------------|--|---|--|---|---|
| Population Condition | Age structure of populations | 5 | 7 | 4 | Strongly supported by the literature. Data are often limited to assessed species or those likely to be assessed in the near future due to the difficulty and expense of aging otoliths. |
| Population Condition | Size structure of populations | 0 | 5 | 4 | Size structure from catch data generally biased by gear selectivity and catchability. |
| Population Condition | Center of distribution (latitudinal or depth) | 2 | 5 | 5 | Distributional shifts tend to suggest a pressure is acting on the population (i.e., fishing or climate). |
| Population Condition | Genetic diversity of populations | 5 | 2 | 2 | Scores well in primary considerations, but there is an overall lack of data for most groundfish species at multiple points in time. |
| Population Condition | Age at maturity | 5 | 1 | 3 | Similar comment as above. |
| Population Condition | Size at maturity | 3 | 2 | 2 | Similar comment as above. |

| | | | | | |
|-------------------------|----------------------------------|---|---|---|---|
| Population Condition | Diet of groundfish | 0 | 1 | 1 | Prey is highly variable and there are few species with enough data over time and space to understand differences. |
| Population Condition | Larval abundance | 2 | 3 | 2 | Abundance of larvae most likely driven by oceanographic conditions and not be reflective of the condition of specific populations. |
| Population Condition | Parasitic load | 3 | 1 | 0 | Theoretically sound but little data for most species. |
| Population Condition | Condition factor (K) | 3 | 5 | 2 | Theoretically sound. condition of fish is related to growth and fecundity, but generally not described. Data limited to species with both individual length and weight measurements during surveys. |
| Population Condition | Cortisol/vitellogenin | 2 | 1 | 1 | May be related to condition, but changes in the attribute are not likely to vary with this indicator at any scale but the very smallest. |
| Population Condition | Disease (liver and gall bladder) | 2 | 1 | 1 | Similar comment as above. |
| Population Condition | Fecundity | 5 | 1 | 2 | Scores well in primary considerations, but there is an overall lack of data available for most species across time and space. |
| Population Condition | Body growth | 2 | 5 | 5 | Typically, age is calculated from otoliths collected during bottom trawl surveys, but growth could also be measured with these samples. |

| | | | | | |
|-------------------------|------------------------------------|---|---|---|--|
| Population Condition | Spatial structure of population | 5 | 5 | 4 | Theoretically sound and data are available for many species, but stocks are generally assessed at the scale of the entire coast. |
| Population Condition | Mean length of species | 5 | 1 | 5 | Lengths measured for many species, but there may be limited data on unassessed species. |
| Population Condition | Rebuilding timeline | 3 | 7 | 5 | Available for overfished species. Most species stop declining, but some have not increased. |

POPULATION SIZE

STOCK ASSESSMENT ESTIMATES OF BIOMASS AND DEPLETION

Stock assessment trends in estimate spawning stock biomass are well established measures of the size of the many commercially important species and are subject to intense peer review. Assessments are tied directly to management efforts and provide quota levels for various fisheries. Changes in assessed populations often reflect changes in the relative abundance of individuals collected in bottom trawl surveys. When management restrictions are established, assessed populations generally stop declining. Many species begin to recover and experience population growth according to the assessments, but there are other species that appear to respond slowly to management actions (see Miller et al. 2009). Assessments provide estimates of stock status relative target and limit reference points established by the Pacific Fishery Management Council for assessed species. The target reference point is B40, 40% of the unfished spawning biomass, the level of spawning stock biomass at which stocks are considered at their optimal yield (B25% for flatfish). The limit reference point is B25, 25% of unfished spawning biomass, the level of spawning stock biomass at which stocks are overfished (5% for flatfish). However, only 30 of 90-plus species within the Pacific Coast Groundfish Fishery Management Plan (PCGFMP) have been assessed and there are generally hundreds of species of fish detected each year in the groundfish trawl surveys that have been conducted off of the U.S. west coast (e.g., Keller et al. 2008).

Stock assessments use data from multiple sources, but the primary sources of data are from two fishery independent surveys. The first survey was conducted by the Alaska Fisheries Science Center's (AFSC), the triennial bottom trawl survey, and covers every third year from 1977 to 2001. During 2004 the Northwest Fisheries Science Center (NWFSC) conducted the final year of the triennial bottom trawl survey. The NWFSC has conducted a separate fishery independent bottom trawl survey annually from 2003 to 2011. The spatial extent and timing of the AFSC triennial survey varied over time. The NWFSC annual survey has a consistent random-stratified design by depth that samples the entire U.S. West Coast from depths of 50 to 1,280 m (Figure GF1). Assessments use multiple data sources incorporating length frequencies, diet, age structure, and fecundity measures when available. Analyses used to generate time series data generally use the same stock assessment framework (Stock Synthesis, e.g., Stewart 2009). Assessments generally use multiple data sources across the range of each stock (e.g., Gertseva et al. 2009, Stewart et al. 2009); however, some species (i.e., cabezon [*Scorpaenichthys marmoratus*] and bocaccio [*Sebastes paucispinis*]) are only assessed in specific regions along the West Coast (Cope and Key 2009, Field et al. 2009).

The major findings of a stock assessment are routinely used by the public and policy makers (i.e., a population is declining, increasing, or overfished). Assessments are typically done for species that are commercially important fishery targets, for species that may be subject to bycatch in targeted fisheries, or for species for which good fishery independent survey data exists. Since assessments estimate spawning biomass, it is generally an assessment of processes that have already taken place (i.e., the impact of fishing and variable recruitment strengths), so this is generally a lagging indicator.

BOTTOM TRAWL RELATIVE SURVEY BIOMASS

The AFSC triennial trawl and the NWFSC annual trawl surveys are well established and analysis methods for these surveys have been developed with input by stock assessment scientists and through outside peer review during the PFMC stock assessment review (STAR) and science and statistical committee review (SSC) processes. The major objective of these surveys is to provide the fishery-independent data necessary to conduct formal stock assessments of fish species managed within the PFMC GFMP (Keller et al.

2008). The NWFSC annual survey collects data in trawlable habitats from the U.S.-Canada border to the U.S.-Mexico border between the months of May to October. Each trawl is 15 minutes in duration and total counts and aggregate weights by species are recorded for all species. Subsamples of targeted species (generally consisting of the 90 managed species) are randomly selected for individual measurements of length and weight, removal of age structures, and sex determination. In a typical year, approximately 600 trawls are successfully conducted; approximately 150,000 fish are individually measured for weight and length, and more than 20,000 otoliths are removed for aging (Keller et al. 2008). Some species are sampled for genetics, stomach contents, maturity level, and toxicology as special projects. These data are in a Fishery Resource Analysis and Monitoring Division (FRAM) database at NWFSC.

The trawl survey data allow for estimates of density and biomass and evaluation of the relative change in population size over time for many more species than are assessed through formal stock assessments (e.g., Levin et al. 2006). As noted, only 30 of the 90-plus managed species on the U.S. West Coast are formally assessed, while there are hundreds of species or groups of fish detected each year during the NWFSC annual trawl survey. One caveat to the bottom trawl survey is that the data are biased towards species that occupy trawlable habitats in depths from 50 to 1,280 m (there is no near shore groundfish survey) and those life history stages that are selected to the survey trawl gear. Most small individuals, either young individuals or smaller species, are not captured by the bottom trawl survey because they are in shallower water as juveniles or they escape through the net mesh. Moreover, species that move into rockier and untrawlable habitats with increasing age or size are not sampled by the bottom trawl survey. The bottom trawl survey is also not a good indicator of Pacific hake biomass, which is a more pelagic species and comprises the largest component of the groundfish population in the CCLME from a fisheries standpoint (Miller et al. 2009). This holds true for other pelagic species as well.

Estimates of relative abundance calculated from trawl surveys are easily understood by the public and have been used historically by policy makers for regulatory and legislative purposes. The estimates of relative abundance from the trawl survey are generally a lagging indicator of past stock population dynamics (i.e., what were the conditions of the ecosystem that allowed recruitment to be good or bad). Many species are captured in the survey prior to being selected by the commercial fishery, providing early information about the strength of incoming recruitments for the fishery.

Biomass. Biomass is a standard measurement of population size and is cited voluminously in the indicator literature (e.g., Fulton et al. 2005, Link 2005). Biomass is the metric estimated in formal stock assessments and the metric used for harvest rates of individual species in West Coast fisheries. However, an aggregate estimate groundfish biomass is not necessarily indicative of the state of the groundfish community, because this information will be biased towards a few large components of the community. For example, Pacific hake is the most abundant groundfish species detected in the NWFSC annual trawl survey and variation in hake abundance can swamp detectable variation in the rest of the groundfish community. Thus any groundfish community indicator will need to identify species of interest or representatives of different functional groups to monitor changes over time. Alternatively, multivariate measurements of the groundfish community will need to be developed to detect meaningful changes in the population size of groundfish.

HAKE ACOUSTIC SURVEY BIOMASS

The Pacific hake integrated acoustic and trawl survey has been conducted since 1977 to assess the size and distribution of the population in the CCLME (Helser and Martell 2007, Helser et al. 2008). The joint survey between the United States and Canada has taken place in 1977, 1980, 1983, 1986, 1989, 1992, 1995, 1998, 2001, 2003, 2005, 2007, 2009, and 2011. The survey is generally conducted between June and August

along the continental slope and shelf from Monterey, California (lat 35.7°N), to the Dixon Entrance in northern British Columbia (lat 54.8°N). However, survey methods have varied over time, particularly prior to 1995. During the survey, hydroacoustics are used to measure acoustic backscatter that is attributed to hake. The size and age structure of the acoustic backscatter is quantified by trawl tows over a subset of the areas that have been identified as hake, and then numbers (or biomass) are calculated. This survey is a single species survey that does not provide adequate information for other groundfish species. In addition, massive northward movements of Humboldt squid (*Dosidicus gigas*) can complicate the survey (e.g. the 2009 survey year). Since it is very difficult to distinguish between Pacific hake and Humboldt squid with the current acoustic survey methodologies, changes in the spatial distribution and frequency of occurrence of Humboldt squid in the survey area may pose problems in the future.

Similar to the bottom trawl surveys, the acoustic survey produces data that are commonly used by the public and policy makers, have been used historically, and are compatible with measurements used by other regions and nations.

NUMBER OF GROUPS BELOW MANAGEMENT THRESHOLDS

A simple indicator of the status of assessed groundfish species is the number of species that are currently below various management thresholds. The Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA) requires fishery conservation and management measures that prevent overfishing, while achieving optimum yield on a continuing basis (16 U.S.C. §1851a1). Overfishing occurs when the actual catch of a species exceeds the allowable catch for that species. The MSFCMA also requires that fishery management plans specify objective and measurable criteria for identifying when a fishery is overfished and contain conservation and management measures to prevent or end overfishing and rebuild the fishery (16 U.S.C. §1853a10). Under the PFMCF GFMP, a species (or stock) is considered overfished when its current spawning stock biomass is assessed to be below that limit reference point. NMFS's national standard guidelines clarify that "overfished" relates to biomass of a stock or stock complex, while "overfishing" pertains to a rate or level of removal from a stock or stock complex (50 CFR 600.310(e)(2)). Estimates of spawning stock biomass and unfished biomass are calculated during the formal stock assessment analysis.

Data to measure the overfishing threshold is available for all stocks that have an identified allowable catch. Approximately 30 of the 90-plus managed groundfish species can be evaluated for the overfished threshold. Data from the NWFSC annual trawl survey alone are not sufficient for evaluating whether or not a stock is below the overfished threshold because the survey covers a small fraction of the exploitation history for most groundfish stocks, making the determination of overfished status from survey data alone subject to the problem of shifting baselines (i.e. it is impossible to estimate the size of the stock at the beginning of its exploitation with survey data that begins at a much later date).

Policy makers have used whether a species is above or below specific management thresholds for regulatory and legislative purposes. Other nations have similar thresholds in their management frameworks (Gray et al. 2010).

POPULATION CONDITION

AGE STRUCTURE OF POPULATIONS

The longevity of many groundfish species allows them to allocate their reproductive output across many years. This strategy is particularly important when environmental conditions are unfavorable for survival of larvae or new recruits (Leaman and Beamish 1984, Berkeley 2004a). In addition, there is growing support in the literature that older fish produce more fit eggs and larvae (Hislop 1988, Berkeley 2004a, Wright and Gibb 2005, Sogard 2008), suggesting that older individuals may produce offspring that will survive and recruit to the population in higher proportions than offspring from younger individuals. Higher survival rates for offspring from older individuals could be particularly true during years when environmental conditions are less than optimal. Thus populations with a truncated age structure (fewer older individuals) may have more difficulty sustaining current population levels. For many groundfish species, the largest and oldest individuals have been historically targeted and removed by fishing practices; many groundfish species subjected to fishing pressure have a truncated size (and age) structure compared to historical, unfished size (age) structure (Jennings and Blanchard 2004, Blanchard et al. 2005). Reference points have not been established for size (age) structure indicators, but similar reference points have been suggested for the indicator mean size that would set reference points at the median size (age) of maturity.

The NWFSC annual groundfish survey collects otoliths for most managed species and age structures should be available for beginning in 2003. Data for other species varies, but are typically limited to small spatial scales and to single data collections in time. The temporal variability in age structure is a function of fishing as well as other less clearly understood factors, spatial variability is not well understood in the CCLME for most species.

The importance of age structure to the success of fish populations, older individuals are generally larger and generally produce more and stronger offspring, is recognized by policy makers. Age structure is inherently used by policy makers because stock assessments use spawning stock biomass as the fundamental metric, which is related to the age of individuals when they mature.

REBUILDING TIMELINE

For groundfish species in the PFMC GFMP, if a species population size is assessed to be below the limit reference point it is declared overfished and a rebuilding plan must be developed. A rebuilding plan establishes an allowable harvest rate that will enable the species to rebuild to its target reference point within an adequate period of time based on the minimum time of recovery, assuming no fishing (PFMC (Pacific Fishery Management Council) 2010). The rebuilding timeline varies dramatically among species. For example, under current management harvest rates, cowcod (*Sebastes levis*) are predicted to rebuild by 2071, while widow rockfish (*Sebastes entomelas*) were declared rebuilt during 2011. When management action is taken, such as reductions in harvest rate, most species stop declining, but the rate at which they rebuild varies (Miller et al. 2009). Rebuilding timelines are only developed for those species declared overfished, so there are a limited number of species with rebuilding rate calculations.

This indicator is commonly used by the public and policy makers. It is also easy to understand which species are having a difficult time rebounding from historical pressures.

SPATIAL STRUCTURE OF POPULATIONS

The spatial structure is a measure of the geographic range and distribution of a species or stock. Most groundfish species in the PFMG GFMP are managed as a single stock, but there is some evidence that the genetic composition of recruits may be quite complicated spatially (Larson and Julian 1999, Berkeley 2004b). The youngest recruits are found to have different genetic diversity and haplotypes from older year-classes or adults. This suggests that the geographic source of successful recruits may differ from year to year and that some populations may be reproductively isolated depending on oceanic conditions (Miller et al. 2005). Thus understanding how spatial structure may have changed over time may help our understanding of the connectivity of species across large spatial scales such as the CCLME. Distributional shifts are hypothesized to occur for either of two reasons climatic or exploitation, but the difference is difficult to distinguish. Perry et al. (2005) showed large latitudinal shifts correlated with changes in temperature due to climate change. Changes in latitudinal and depth distribution of groundfish assemblages can be due to ontogenetic movement, fishing pressure, and changes in climate (Fairweather et al. 2006, Coetzee et al. 2008, Dulvy et al. 2008).

As predicted, the geographic ranges of many overexploited species typically shrink, and stocks are concentrated into smaller regions following population declines (Atkinson et al. 1997, Garrison and Link 2000). Moreover, shrinking spatial distribution may limit the ability of a population to find suitable environmental conditions for offspring (Berkeley 2004b). Some changes in species spatial distributions may even result in population extinctions (Thomas et al. 2004, Drinkwater 2005). Reference points for distributional shifts are not currently defined or used and would be difficult to measure unless species were divided into and managed as distinct population segments.

The AFSC triennial survey and the NWFSC annual survey have collected data on the density and distribution of the CCLME groundfish assemblages for nearly 30 years. However, due to different survey methods these two surveys cannot be treated as a single time series. At this time, Pacific hake is the only species known to shift distribution with changes in environmental conditions, exploitation, or changes in population condition (Ressler et al. 2007).

In general, shifting or changing patterns of spatial distribution are easily understood by the public and policy makers. Spatial distribution data have been transmitted to the public in the past in the context of invasive species for terrestrial, freshwater, and marine systems. For example, the annual variability in the northern extent of the geographic range of Humboldt squid may have strong trophic impacts in the CCE. The ability to detect spatial shifts in distribution or range is likely to occur at long time scales for noninvasive species, so spatial structure should be a lagging indicator of changes in the population condition.

MEAN SIZE OF ALL SPECIES

The mean size (measured by length or weight) of all species caught in fishery-independent surveys, fishery-dependent surveys, or landings has been used to evaluate changes in an ecosystem (Link and Brodziak 2002, Link et al. 2002, Rochet and Trenkel 2003, Nicholson and Jennings 2004, Sala et al. 2004). A decrease in mean size is expected and has been observed in heavily fished systems (Haedrich and Barnes 1997, Levin et al. 2006, Methratta and Link 2006). However, the sensitivity of changes in mean size to environmental conditions is not well understood (Rochet and Trenkel 2003). One study suggests changes greater than 30% in mean length from one year to the next be set as a reference point (Link 2005), while another study suggests the reference point be set at the median length at maturity (Caddy and Mahon 1995).

In the groundfish trawl surveys, subsamples of targeted species (up to 100 per trawl) are individually measured for length and weight. In order to monitor this indicator with fishery-independent

data, all species would need to be sampled and measured in some fashion. However, this metric can be calculated using fisheries landings data (Link 2005), so historical data are available via Pacific Fisheries Information Network (PacFIN, <http://pacfin.psmfc.org/>).

This indicator is easily understood and is being used in other regional ecosystems (Link 2005). Similar to other indicators, mean size of all species is most likely to be a lagging indicator of the population condition because the size structure may be the result of environmental conditions acting on each individual since it was born.

AGE AT MATURITY

Population parameters such as age and size at maturity are adaptive traits and there is increasing support in the literature for rapid evolution of these life history characteristics (Haugen and Vollestad 2001, Stockwell et al. 2003). As with the discussion of age structure as an indicator, significant changes in a population's age at maturity can signal extreme pressures that may have significant impact on a population's ability to sustain itself and ought to be cause for concern (Olsen et al. 2004). Declines in age-at-first-maturity have been commonly associated with compensatory responses to a reduction in population size (Trippel 1995, Berkeley 2004b). There are multiple examples in which age at maturity has declined in heavily exploited groundfish populations such as Atlantic cod (*Gadus morhua*) (Beacham 1983a), haddock (*Melanogrammus aeglefinus*) (Beacham 1983b), American plaice (*Hippoglossoides platessoides*) (Trippel 1995), and community-wide measurements (Greenstreet and Rogers 2006). In most studies, age at maturity declined during periods of exploitation, as evolutionary theory would predict, but striped bass (*Morone saxatilis*) in coastal Rhode Island showed a 15% increase in age at maturity over a 46-year period (Berlinsky et al. 1995). Olsen et al. (2004) provide a framework for Atlantic cod reference points that would provide managers with early warning signals about changes in this indicator.

Estimates of age at maturity exist for most managed groundfish species, but sampling generally occurred across short temporal scales (Gunderson et al. 1980, Echeverria 1987, Love et al. 2002, Thompson and Hannah 2010). There are a few examples of multiple studies that measured age at maturity at various points in time at different locations within the CCLME, for example, canary rockfish (*Sebastes pinniger*) from California, Oregon, Washington, and British Columbia at various times between 1960 and 1982 (Phillips 1964, Westrheim 1975, Gunderson et al. 1980, Echeverria 1987). Age structures (otoliths, dorsal spines, and fin rays) are collected from targeted species during the NWFSC annual trawl survey and gonads are collected as special projects from time to time. However, most groundfish are in need of new data on maturity and fecundity relationships, because methods have been inconsistent across studies and there are few examples of estimates over time (Stewart 2008). Age at maturity is an easy indicator to understand for the public and policy makers, but this indicator has not been used because of the general lack of data over time for most species.

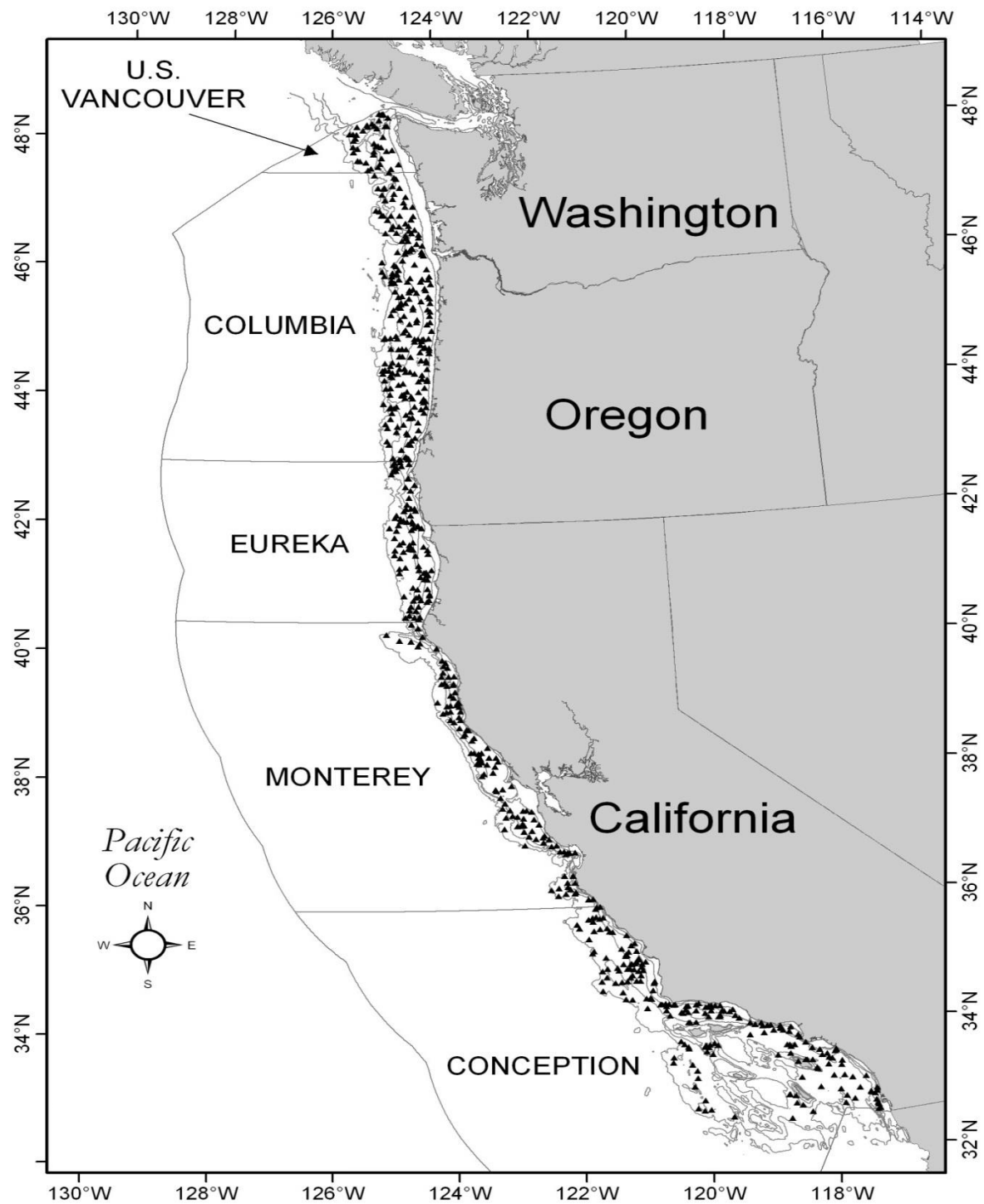


Figure GF1. Example of the number and spatial extent of locations (triangles) surveyed by the West Coast groundfish trawl survey each year during 2003–2010. (Reprinted from Keller et al. 2008.)

FINAL SUITE OF INDICATORS

The list of species for which each indicator will be calculated is located in Table GF5.

ATTRIBUTE 1 - POPULATION SIZE

From the eight indicators in the top quartile for population size, two are used as indicators for population size of groundfish in the CCLME (Table GF6):

CHANGE IN TIME SERIES OF GROUNDFISH BIOMASS

Groundfish biomass was used relative to either the estimate of the unfished biomass from a stock assessment or trends in relative abundance from the survey time series (stock depletion):

- a. Tier 1: Modeled estimates of stock depletion based on estimates of spawning biomass from assessments beginning in 2007 as earlier assessments are out of date.
- b. Tier 2: Trends in stock depletion based on relative biomass estimates from the NWFSC annual trawl survey.

NUMBER OF ASSESSED SPECIES BELOW MANAGEMENT THRESHOLDS: END YEAR POINT ESTIMATES OF STOCK DEPLETION

Two tiers are specified for biomass of groundfish in as a measure of abundance because stock assessments provide the best available estimates of spawning stock biomass and depletion, integrating all of the available data on each stock over the full exploitation history of each stock, while in the absence of a stock assessment the survey relative biomass index provides the best information available to estimate trends in the stock size, albeit over a shorter time series in comparison to the stock assessments. As stock assessments are generally updated on a 2-6 year cycle, for stock assessments that do not extend beyond 2007 the IEA is providing both the time series of spawning stock biomass from the assessment as well as the trend in biomass from the survey are presented. As hake cannot be monitored for trends via the NWFSC annual bottom trawl survey due to likely annual changes in availability to the survey gear, the hake acoustic relative survey biomass is used as an alternative. The number of species below management thresholds was chosen because it is an easy measure of species or stocks that have typically been doing poorly in the past, but we recognize that documents (Miller et al. 2009) already exist that communicate this information. Thus this indicator may not be necessary in a final status report of the CCLME.

ATTRIBUTE 2 - POPULATION CONDITION

From the five indicators in the top quartile for population condition, one is used as indicators for population condition of groundfish in the CCLME (Table GF6):

METRICS OF POPULATION AGE (OR SIZE IN THE ABSENCE OF AGE) STRUCTURE

Tier 1: Modeled estimates of age structure (or size structure in the absence of age) from assessments beginning in 2007 as earlier assessments are out of date.

Tier 2: Age structure (or size structure in the absence of age) from the NWFSC annual survey

These indicators are in the top indicators evaluated. Rebuilding timeline was not chosen as one of the final indicators because it is only available for species which have been formally considered overfished; thus it is only useful for a small number of species that are already below the target reference point. Using age structure accounts for many of the ecological processes that would affect age at maturity, so age at maturity is eliminated from the final indicator suite. Where available age structure is used as the indicator; however, size structure has been used in lieu of age structure where age data are not available. Size structure was not in the top quartile for population condition indicators, but it is the top-ranked indicator in the second quartile and missed the top quartile by 0.03 points.

POPULATION AGE OR SIZE STRUCTURE

The mean age or size of all species caught in either fishery-independent surveys, fishery-dependent surveys, or landings is thought to be a useful and simple indicator to evaluate the overall effects of fishing (e.g., changes in rates of mortality) on an ecosystem (Fulton et al. 2005, Link 2005, Coll et al. 2009). Age and size-based metrics respond to fishing impacts because age and body size determines the vulnerability of individuals, populations, and communities (Jennings and Dulvy 2005). Others contend that there are very few examples where length-based analysis leads to useful management advice, in part because of the need for age and gear selectivity information, and because size related changes in distribution will influence data (Hilborn and Walters 1992). Additionally, older individuals tend to be more fecund and some fish species produce larvae that have a higher survival rates than larvae from younger fish (Berkeley 2004b, Bobko and Berkeley 2004). Age and size based metrics are thought to better support medium-term rather than year-to-year management evaluation, because the response to management actions often cannot be quantitatively interpreted for contributing causal factors without extensive additional research (Jennings and Dulvy 2005).

Fish population age and size structure has been linked to scientifically defined reference points or progress targets. Some have based these on a decline in mean size of greater than 30% (warning or precautionary threshold) or greater than 50% (limiting reference point), the latter of which was chosen because it corresponds to an observed doubling in the time series of length after fishing has decreased (Link 2005). Others suggest that practical issues currently preclude the development and adoption of firm reference points for size-based indicators, although an appropriate target would be a reference direction that is consistent with a decline in the overall human impacts of fishing on the community, and thereby on the ecosystem (Jennings and Dulvy 2005). Similar reference points could be defined for mean population age.

The principal attraction of size-based metrics is the widespread availability of species size and abundance data collected during ongoing monitoring programs (Jennings and Dulvy 2005). Many monitoring programs collect a more limited but potentially more informative set of age data. The AFSC triennial survey and NWFSC annual survey have collected size data from a large array of species, and age data from a more limited set of species. The NWFSC annual survey collects up to 100 length measurements, sex determinations, and individual weights, and up to 25 age structures per trawl haul for key species, and more recently for all groundfish species of management concern (Keller et al. 2008). There are well recognized gear-selectivity issues associated with age and size data (Hilborn and Walters 1992) and ideally indicators should be calculated for age and size classes that are well selected by the gear. Fish population age and size structure has been used as an indicator in a variety of other ecosystems, including the Celtic Sea (Blanchard et al. 2005), northeastern U.S. continental shelf (Link and Brodziak 2002), and eastern Bering Sea (AFSC 2009).

Table GF5. List of groundfish for which the aforementioned indicators will be calculated. This list is composed of species in assemblages identified in Cope and Haltuch (2012), species with quantitative stock assessments completed from 2007-2011, and species that are well surveyed by the NWFSC annual trawl survey. Note that due to limited data availability yelloweye rockfish would be removed from this species list without the results of a current stock assessment. Pacific hake would also be removed from this list without a current stock assessment because the trawl survey data alone are subject to changes in hake availability over time. However, as hake is currently assessed every year, hake should remain on the species list.

| Species | Scientific name | Assessment Years |
|-----------------------|--------------------------------|------------------------------|
| Pacific hake | <i>Merluccius productus</i> | 2007, 2008, 2009, 2010, 2011 |
| Stripetail rockfish | <i>Sebastes saxicola</i> | |
| Sablefish | <i>Anoplopoma fimbria</i> | 2011 |
| Dover sole | <i>Microstomus pacificus</i> | 2011 |
| Redstripe rockfish | <i>Sebastes proriger</i> | |
| Splitnose rockfish | <i>Sebastes diploproa</i> | 2009 |
| Rex sole | <i>Glyptocephalus zachirus</i> | |
| Chilipepper rockfish | <i>Sebastes goodei</i> | 2007 |
| Spiny dogfish | <i>Squalus acanthias</i> | 2011 |
| Shortbelly rockfish | <i>Sebastes jordani</i> | 2007 |
| Arrowtooth flounder | <i>Atheresthes stomias</i> | 2007 |
| Darkblotched rockfish | <i>Sebastes crameri</i> | 2007, 2009, 2011 |
| Canary rockfish | <i>Sebastes pinniger</i> | 2007, 2009, 2011 |
| Lingcod | <i>Ophiodon elongatus</i> | 2009 |
| Longnose skate | <i>Raja rhina</i> | 2007 |
| Yelloweye rockfish | <i>Sebastes ruberrimus</i> | 2007, 2009, 2011 |
| Aurora Rockfish | <i>Sebastes aurora</i> | |

| | | |
|-----------------------|----------------------------------|------------------|
| Blackgill Rockfish | <i>Sebastes melanostomus</i> | 2011 |
| Bocaccio Rockfish | <i>Sebastes paucispinis</i> | 2007, 2009, 2011 |
| English Sole | <i>Parophrys vetulus</i> | 2007 |
| Flathead Sole | <i>Hippoglossoides elassodon</i> | |
| Greenstriped Rockfish | <i>Sebastes elongatus</i> | 2009 |
| Pacific Ocean Perch | <i>Sebastes alutus</i> | 2007, 2009, 2011 |
| Pacific Sanddab | <i>Citharichthys sordidus</i> | |
| Petrale Sole | <i>Eopsetta jordani</i> | 2009, 2011 |
| Spotted Ratfish | <i>Hydrolagus colliei</i> | |
| Yellowtail Rockfish | <i>Sebastes flavidus</i> | |
| Black Rockfish | <i>Sebastes melanops</i> | 2007 |
| Widow Rockfish | <i>Sebastes entomelas</i> | 2007, 2009, 2011 |
| Greenspotted Rockfish | <i>Sebastes chlorostictus</i> | 2011 |

Table GF6. Top indicators for Attributes 1 and 2.

| Attribute | Indicator | Definition and source of data | Time series | Sampling frequency |
|----------------------|--|---|-----------------------------|---|
| Population Size | Groundfish biomass | Tier 1: Modeled estimates of spawning biomass as measured by stock depletion from assessments beginning in 2007 as methods have been most stable during the 2007 – present. | Tier 1: Variable by species | Annual estimate from both Tier 1 and 2 indicators |
| | | Tier 2: Relative biomass estimates as measured by the trend in the NWFSC annual survey | Tier 2: 2003-2011 | |
| Population Size | Number of assessed species below management thresholds | Number of species below the PFMC overfished level and currently subject to rebuilding plans | N/A | Biannual rebuilding analyses |
| Population Condition | Population age (or size) structure | Tier 1: Modeled estimates of age structure (or size structure in the absence of age) from assessments beginning in 2007 as methods have been most stable during the 2007 – present. | Tier 1: Variable by species | Annual estimate from both Tier 1 and 2 indicators |
| | | Tier 2: Age structure (or size structure in the absence of age) from the NWFSC annual survey | Tier 2: 2003-2011 | |

STATUS AND TRENDS

MAJOR FINDINGS

Stock status (based on biomass trends) and population demographic condition (as measured by proportion mature and of maximum age or size) were summarized for 30 groundfishes. Most assessed groundfishes are above the biomass limit reference point, and are thus not overfished (Figure GF2). The four assessed stocks currently in an overfished state are all rockfishes. All assessed groundfishes are below their target catch, thus overfishing is not occurring in these stocks. Regarding population condition measures, age or length structure tended to show more changes, usually declines, over time than proportion mature. Non-elasmobranch groundfishes tended to see the most changes over time in both measures, with rockfishes being most sensitive to demographic changes.

INDICATOR #1: RELATIVE TRENDS IN BIOMASS TRAJECTORIES

SUMMARY

Biomass trajectories are a commonly used indicator of fisheries population dynamics and show the details of how the population biomass has changed over time. Trends in the time series of abundance smooth

out the dynamics to offer a directional summary of the changes. And while absolute biomass trends can be used, it is more common to consider the change in biomass relative to unfished condition, termed “depletion”. A stock is considered more depleted when this ratio is relatively smaller, and less depleted when it is relatively larger. This ratio has particular meaning in groundfish management, where status reference points are based on depletion. For groundfishes other than flatfishes, the target depletion is 40% of unfished levels and the limit reference point (the value under which stocks are considered overfished) is 25% unfished levels. For the flatfishes, the target and limit reference points are 25% and 12.5%, respectively. All subsequent biomass measures are the mature female biomass, also called “spawning biomass”, which is the commonly used biomass metric of age-structured stock assessments.

Ideally one would be able to census a population over a long period of time to get a direct measure of stock status for that period. Such detailed population information is not available for any Pacific coast groundfishes, so the next best source of status information is to use the population biomass estimates from age-structured stock assessments. Age-structured stock assessments combined fishery removals, abundance indices, size composition data, and life history information to reconstruct an estimation of how the population biomass changed over time. Barring the availability of stock assessment information, trends in indices of abundance as measured by a fishery-independent survey (specifically, the annual groundfish trawl survey conducted by the Northwest Fisheries Science Center since 2003) were considered. Of the 90+ groundfish species in the groundfish Fishery Management plan, 30 species contain either of these data sources, and thus were considered for status determination. The current development of data-limited methods (Cope 2012; Dick and McCall) may allow more groundfishes to be included in this summary in future iterations of the IEA.

For the analysis of groundfish status, we considered stock assessments from 2007 to 2011 to derive relative biomass trajectories. This was available for 21 of the 30 groundfishes considered. For the remaining 9 stocks, NWFSC trawl survey indices of abundance were used. Stocks with assessments only up until 2007 were also supplemented with the results of the survey abundance. Because the survey indices are limited in temporal coverage, relative trends in abundance rather than depletion are used and the change in index trend compared to the average biomass value and variance over the last 5 years are used instead of depletion reference points. Current population dynamics in the relative biomass trajectories were also evaluated for the last 5 years of the time series. Groundfish stocks were considered in 4 major groups: 1) Elasmobranchs, 2) Flatfishes, 3) Rockfishes and 4) Roundfishes. Within the first three groups, depth was used to distinguish three additional ecological categories: 1) nearshore, 2) shelf, and 3) slope. In general, there are very few nearshore representatives given the lack of assessments of nearshore species and the inadequacy of the trawl survey to sample the nearshore environment, so this status analyses is mostly limited to shelf and slope species. Full time series are provided for each series, but the last 5 years are used to determine the most recent trends.

Overall, most assessed groundfishes are above the biomass limit reference point, and are thus not overfished (Figure GF2). The only assessed stocks currently below the overfished status reference point are all rockfishes. All assessed groundfishes are below their target catch, thus overfishing is not occurring in these stocks. Many of the stocks show biomass around or above the target reference point as well as stable or increasing in the short term (Table GF7). **Elasmobranchs (Figures GF3-GF6; Table GF7):** Assessed elasmobranch stocks are all above target depletion levels, while all stocks presented show stable population dynamics over the last 5 years.

Flatfishes (Figures GF7-GF15; Table GF7): Two of the three assessed flatfishes were above the target depletion level with one between the target and limit status reference levels. All of the species showed either increasing or stable population dynamics over that past 5 years. The shelf stock represented were either

above target and/or demonstrated stable dynamics over the last five years. There is some indication that rex sole is in a slightly downward trend over the last five years, but is currently within the stable limit.

Rockfishes (Figures GF16-GF34; Table GF7): All categories of rockfishes show a similar pattern of historical declines with contemporary population increases. Black rockfish (Figure GF16) is the only representative of the nearshore rockfish complex, and it shows a recent increase with the population above the target level. Because of the diversity of life histories and fisheries interactions in the nearshore environment, black rockfish cannot be used as a proxy for the other species. The shelf species also show increasing or steady populations in recent years, though current status ranges from well above the target (greenstriped rockfish; Figure GF24) to well below the limit (cowcod; Figure GF21). Slope species, with generally higher longevities, show a variety of population responses and tend to have status below targeted levels.

Roundfishes (Figures GF35-GF38; Table GF7): The roundfishes category is an amalgam of species with very different life histories and adult habitat. The group tend to be at around the target biomass levels with increasing population trajectories, except for sablefish (Figure GF38), which is both below target and trending downward.

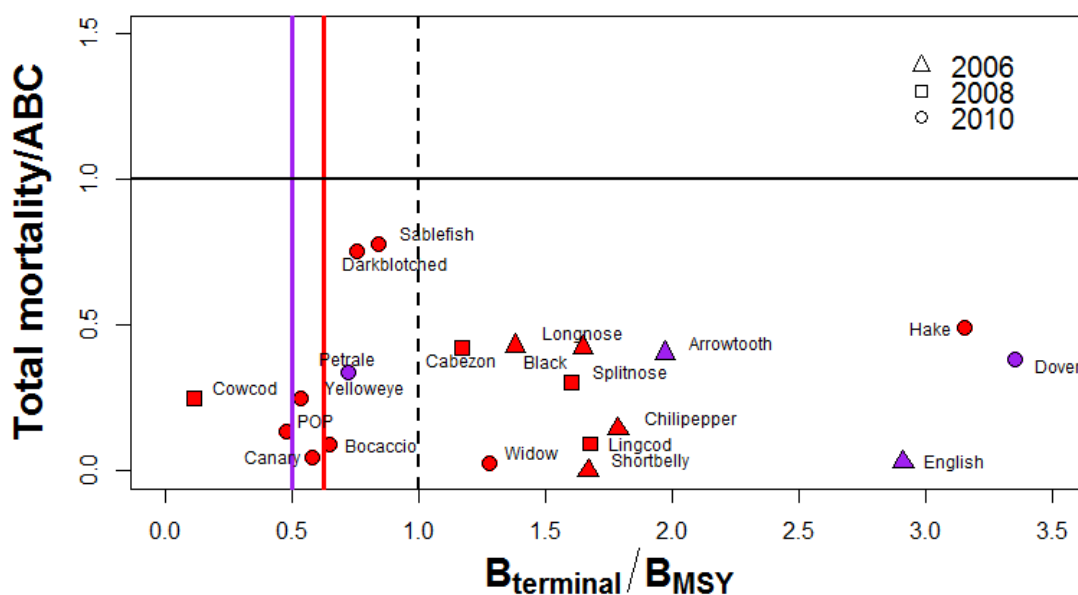


Figure GF2. Stock status plot relative to being overfished (x-axis) and overfishing (y-axis) for all species assessed since 2007. Vertical broken line indicates the target biomass reference point. Vertical solid line indicates the limit reference point indicating an overfished status (red for elasmobranchs, rockfishes, and roundfishes; purple for flatfishes). Horizontal blue line indicates overfishing wherein total mortality exceeds the allowable biological catch (ABC). Symbols indicate the terminal year of the assessment in which the reference points are determined.

Table GF7. Results for each stock evaluated for each of two status indicators: 1) Biomass and 2) Population structure. Two sources of information were used: 1) Stock assessments and 2) Northwest Fisheries Science Center (NWFSC) shelf-slope trawl survey, when assessments were not available, or older than 2007.

“Depletion” refers to the relative change in spawning biomass; “5-year trend” is the trend in the last 5 years of the time series (details found in the text). “B final year” is the biomass value in the final year compared to the 5-year average. “Prop. mature” is proportional of the population mature relative to the beginning of the time series; “95% cum.” refers to the 95% cumulative age or length of the population relative to the beginning of the time series. +: above target limit or increasing; ●: between target and limit or stable; -: below limit or decreasing. Blank spaces indicate no information reported.

| Taxa | Stock | Biomass | | | | Population structure | | | |
|--------------|---------------------|------------|------------|--------------|------------|----------------------|--------------|--------------|--------------|
| | | Assessment | | NWFSC Survey | | Assessment | | NWFSC Survey | |
| | | Depletion | 5-yr trend | B final year | 5-yr trend | Prop. mature | 95% cum. age | Prop. mature | 95% cum. lt. |
| Elasmobranch | Longnose skate | + | ● | ● | ● | ● | - | ● | ● |
| | Spiny dogfish | + | ● | | | ● | - | | |
| | Spotted ratfish | | | ● | ● | | | + | ● |
| Flatfishes | Arrowtooth flounder | + | + | ● | + | - | - | - | - |
| | English sole | + | + | ● | ● | + | - | - | ● |
| | Pacific sanddab | | | ● | + | | | ● | ● |
| | Petrale sole | ● | ● | | | - | - | | |
| | Dover sole | + | ● | | | ● | ● | | |
| | Flathead sole | | | ● | ● | | | ● | ● |
| | Rex sole | | | ● | - | | | ● | ● |
| | | | | | | | | | |
| Rockfishes | Black | + | + | | | - | - | | |
| | Bocaccio | ● | ● | | | - | - | | |
| | Canary | - | ● | | | - | - | | |
| | Chilipepper | + | ● | ● | ● | ● | - | - | + |
| | Cowcod | - | ● | | | - | - | | |
| | Darkblotched | ● | + | | | - | - | | |
| | Greenspotted | ● | + | | | - | - | | |
| | Greenstriped | + | + | | | ● | - | | |
| | Pacific Ocean perch | - | ● | | | - | - | | |
| | Redstriped | | | + | ● | | | - | ● |
| | Shortbelly | | | ● | ● | | | ● | ● |
| | Stripetail | | | ● | ● | | | ● | ● |
| | Widow | + | + | | | ● | - | | |
| | Yellowtail | | | ● | ● | | | ● | ● |
| | Aurora | | | - | - | | | - | ● |
| | Blackgill | ● | ● | | | - | - | | |
| | Splitnose | + | + | | | - | - | | |
| | Yelloweye | - | ● | | | - | - | | |
| Roundfishes | Cabazon | + | + | | | - | - | | |
| | Lingcod | + | + | | | - | - | | |
| | Pacific Hake | + | + | | | | | | |
| | Sablefish | ● | - | | | ● | ● | | |

SPECIFIC TIME SERIES

Interpreting biomass time series plots: Green area is above relative target spawning biomass, red is below the limit relative target spawning biomass, and yellow is between the target and limit values. Gray shaded area indicates the last 5 years. Significant population increases were defined as more than 1% per year, while significant decreases were less than 1% a year. No change was less than 1% either way per year. A 1% threshold was chosen arbitrarily and would lead to a minimum of a 10% increase in a decade's time. An up-arrow, down-arrow, and dot indicate increasing, decreasing, and stable population dynamics over the last 5 years, respectively, for the stock assessment derived data. For the shorter survey data time series, two different measures of relative change and trend are used. The mean (solid line) and ± 1 standard deviation (broken lines) for the full time series is calculated and shown in green. A linear trend is fit to the last five years and the change in biomass over that trend is compared to 1 standard deviation from the mean. Arrows up, down, or level indicate increasing, decreasing, or steady trends, respectively. The average biomass for the last 5 years is also calculated and compared to the full time series mean. A plus or minus indicates a change greater than 1 standard deviation from the full time series mean in either the positive or negative direction, while a dot indicates a change smaller than 1 standard deviation.

Elasmobranchs (N=3)

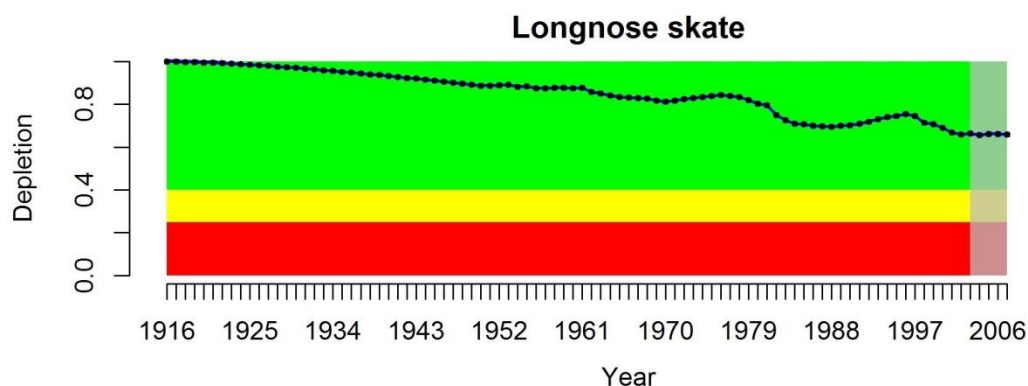


Figure GF3. Relative abundance trajectory 1916-2007 for longnose skate.

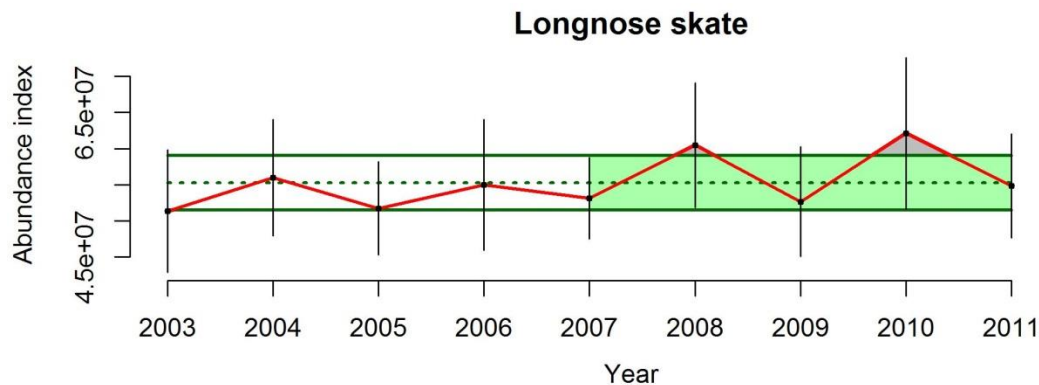


Figure GF4. Trawl survey design-based estimates of longnose skate biomass for years 2003-2011.

Summary: Longnose skate has shown a slow decline over the length of the time series, but with stable population dynamics in the most recent 5 years. Relative biomass appears to have maintained a level above the target biomass in all years.

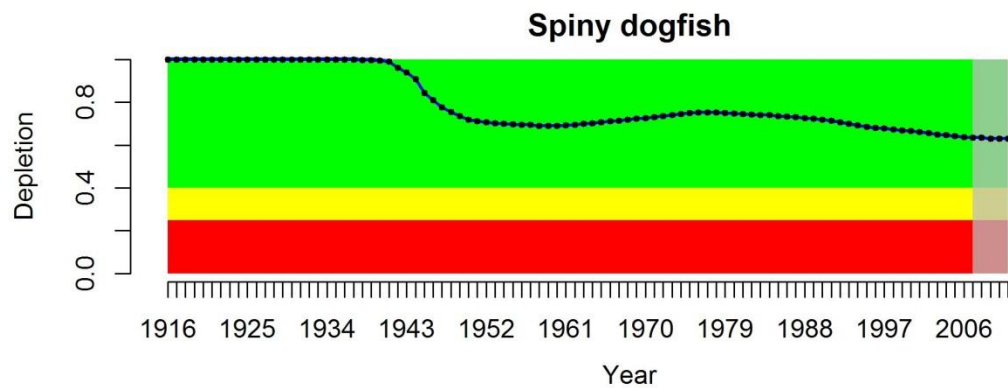


Figure GF5. Relative abundance trajectory 1916-2011 for spiny dogfish.

Summary: After an initial steep decline in the 1940s, relative spiny dogfish abundance has slowed in decline or remained stable in recent years. The population appears to have been above the target relative biomass reference point in all years.

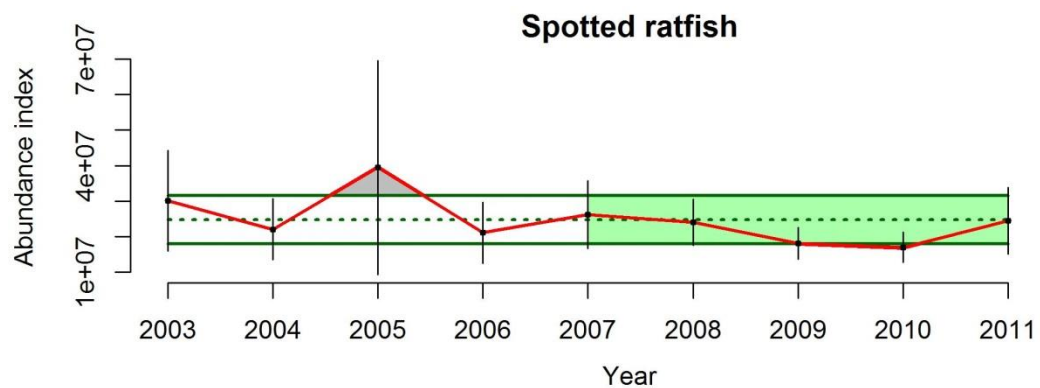


Figure GF6. Trawl survey design-based estimates of spotted ratfish biomass for years 2003-2011.

Summary: No stock assessment for spotted ratfish is available, so no baseline information can be interpreted for this stock at this time. For the most recent years, spotted ratfish appear to have a stable population abundance.

Flatfishes (N=7)

Shelf

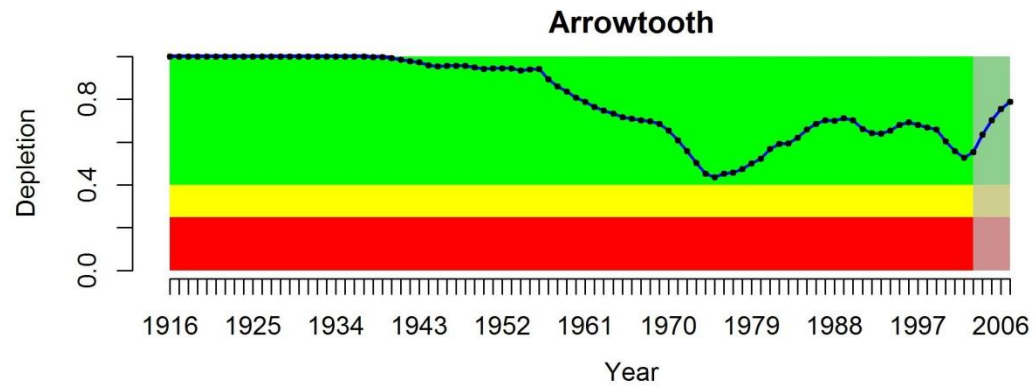


Figure GF7. Relative abundance trajectory 1916-2007 for arrowtooth flounder.

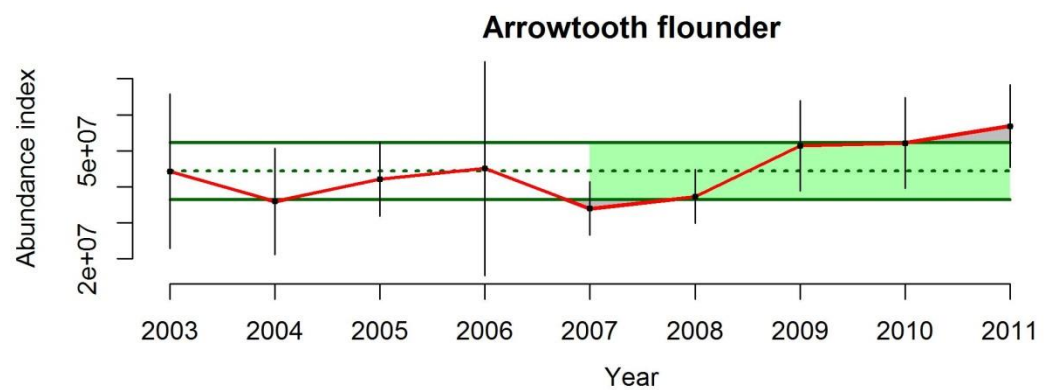


Figure GF8. Trawl survey design-based estimates of arrowtooth flounder biomass for years 2003-2011.

Summary: Arrowtooth flounder demonstrated its greatest decline from the 1950s to the 1970s. It has since increased and continues to show increase in the most recent years. At no point has it been recorded to have gone below the target relative biomass.

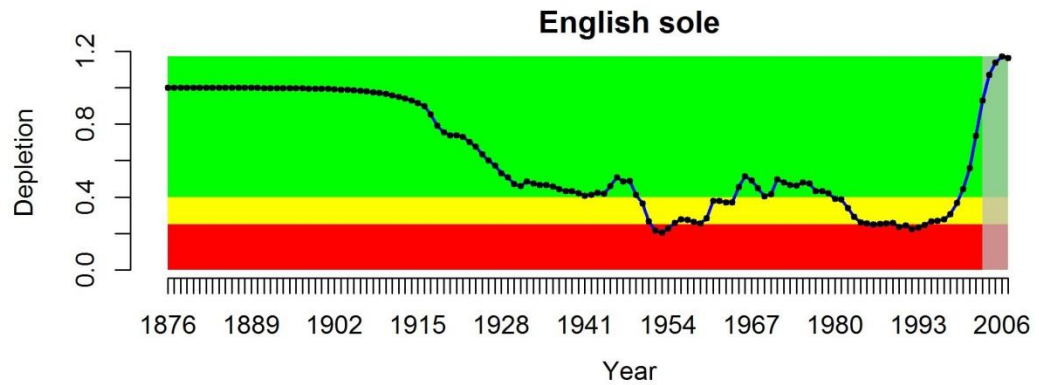


Figure GF9. Relative abundance trajectory 1876-2007 for English sole.

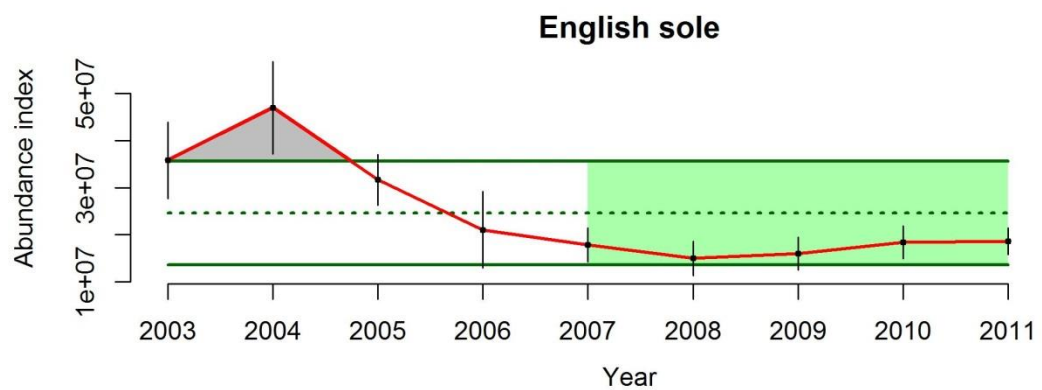


Figure GF10. Trawl survey design-based estimates of English sole biomass for years 2003-2011.

Summary: English sole demonstrated large declines in the early 20th-century, at times dropping below the target relative biomass level. Recent years indicate a large increase, with an increasing or stabilizing trend in the last 5 years.

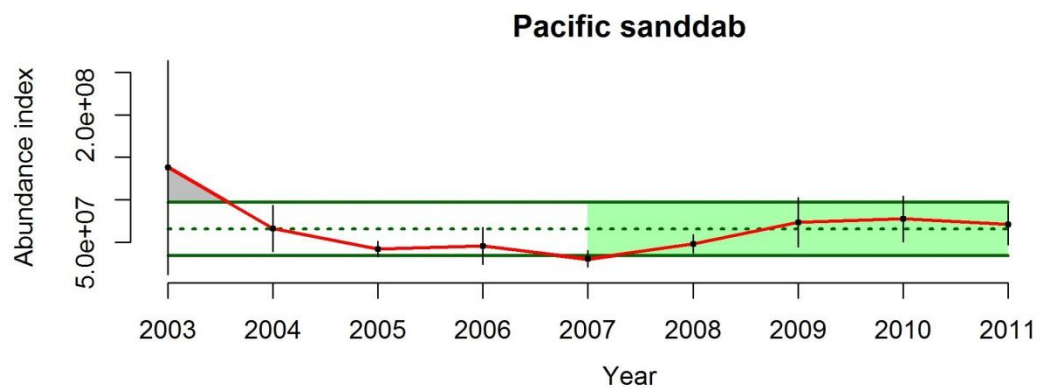


Figure GF11. Trawl survey design-based estimates of English sole biomass for years 2003-2011.

Summary: No stock assessment is available for Pacific sanddab, so no baseline information on abundance exists. Recent years indicate an increasing trend in survey abundance.

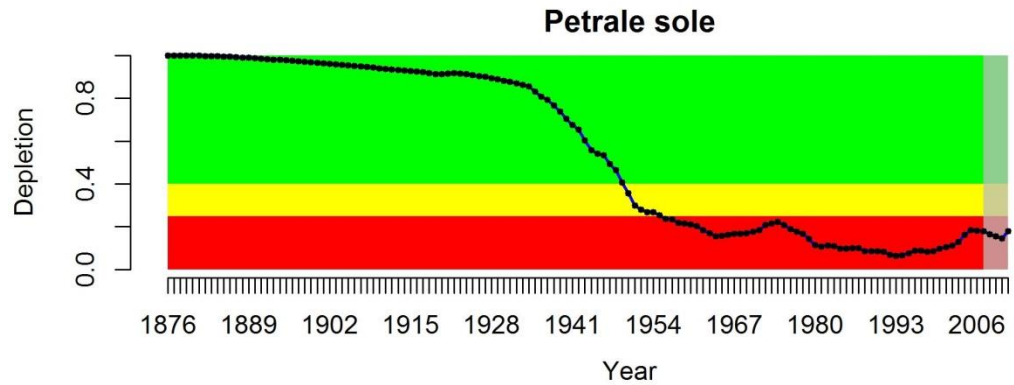


Figure GF12. Relative abundance trajectory 1876-2011 for Petrale sole.

Summary: Petrale sole abundance dropped sharply from the late 1930s to the 1950s, with a steady decline through the 1990s, bring the population below the relative biomass limit. Recent years have shown an uptick with a steady population over the last 5 years.

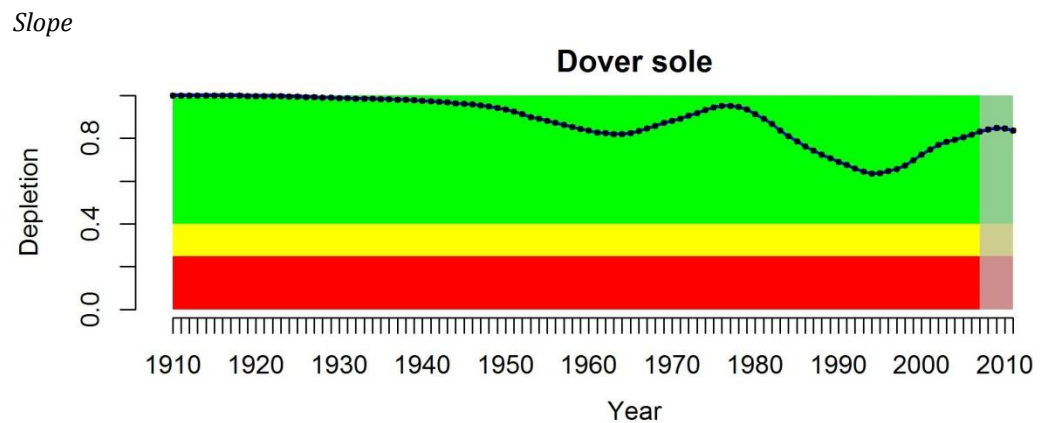


Figure GF13. Relative abundance trajectory 1910-2011 for Dover sole.

Summary: Dover sole populations have shown only slight declines over the time series. Relative biomass has stayed above target levels in all years and is steady over the last 5 years.

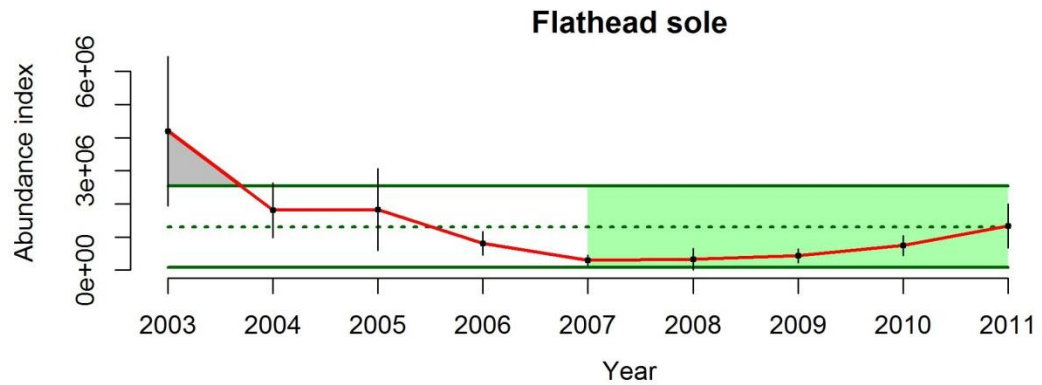


Figure GF14. Trawl survey design-based estimates of flathead sole biomass for years 2003-2011.

Summary: No flathead sole assessment is available, so no baseline information on abundance exists. Recent years indicate a steady trend in survey abundance.

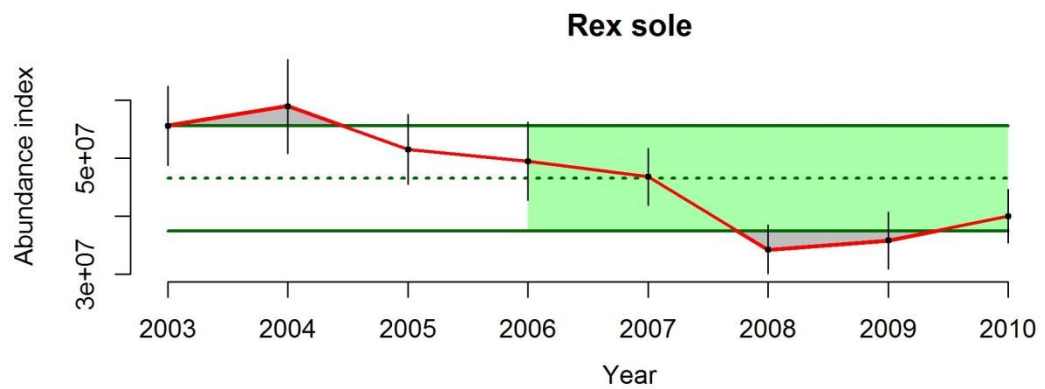


Figure GF15. Trawl survey design-based estimates of English sole biomass for years 2003-2011.

Summary: No rex sole assessment is available, so no baseline information on abundance exists. Recent years indicate a declining trend in survey abundance.

Rockfishes (N=18)

Nearshore

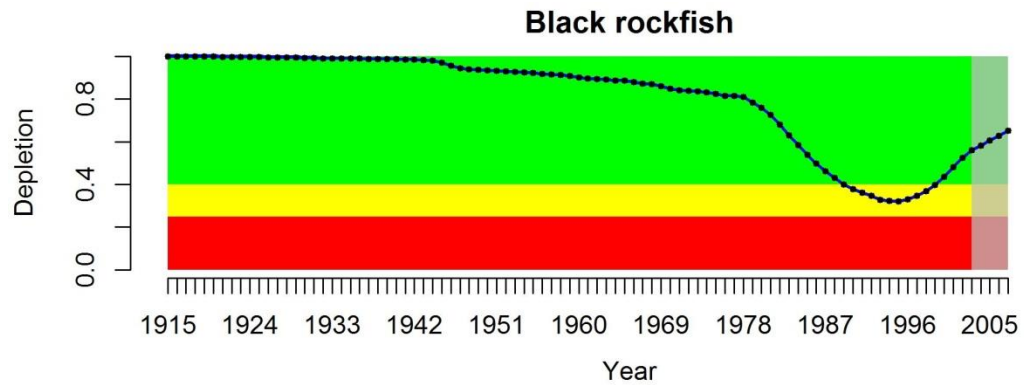


Figure GF16. Relative abundance trajectory 1916-2009 for black rockfish.

Summary: Black rockfish shows a consistent decline until the late 1990s, where in the population starts to grow. Relative biomass dropped below the target relative biomass level for most of the 1990s. Recent years show an increasing trend in population abundance.

Shelf

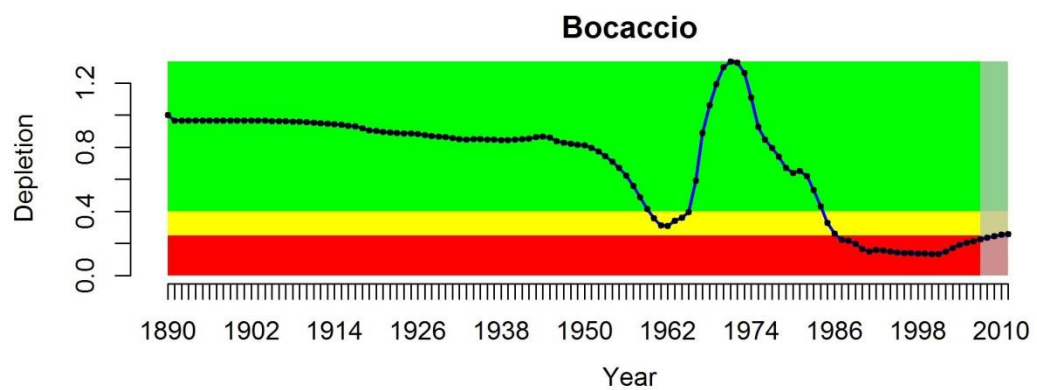


Figure GF17. Relative abundance trajectory 1890-2011 for bocaccio.

Summary: Bocaccio abundance has been highly dynamic over the time series, dropping to levels below the relative biomass limit in recent years. The population trend over that last 5 years is steady.

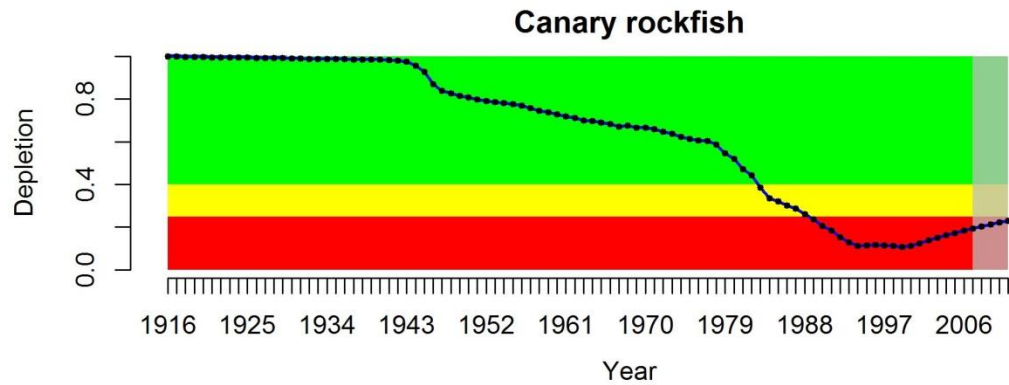


Figure GF18. Relative abundance trajectory 1916-2011 for canary rockfish.

Summary: Large declines in population abundance have been witnessed in canary rockfish, enough to drop the relative abundance below the relative biomass limit. Recent years show very slow growth and an overall stable population.

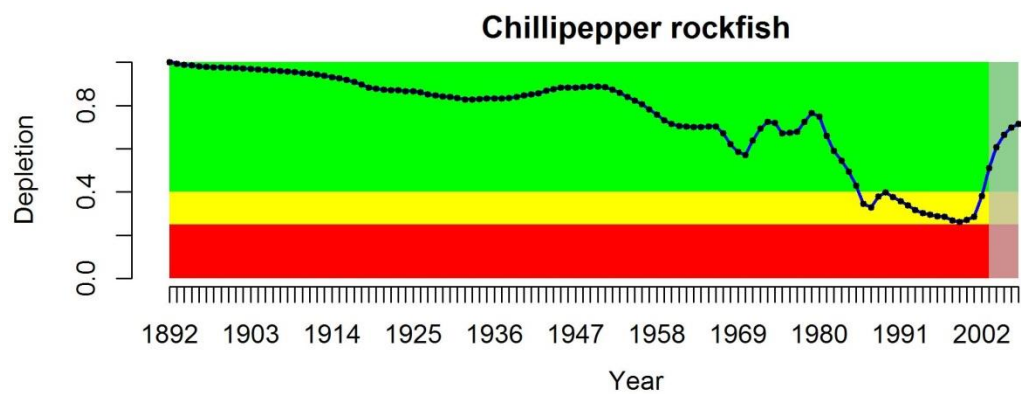


Figure GF19. Relative abundance trajectory 1892-2011 for chilipepper.

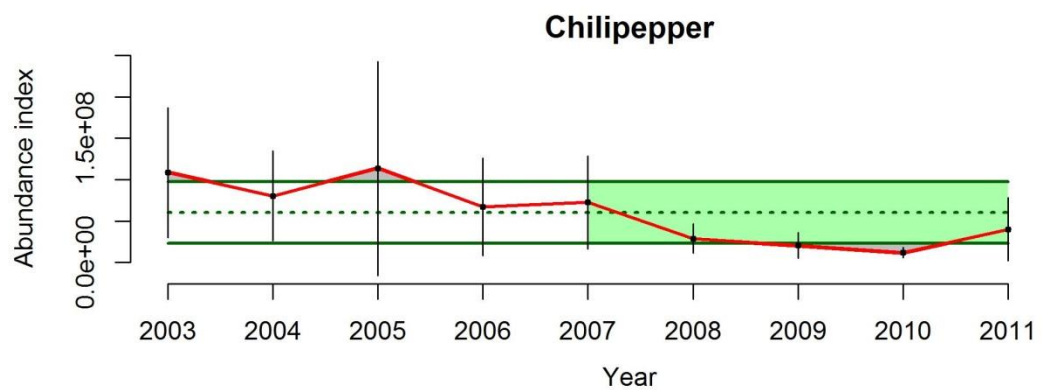


Figure GF20. Trawl survey design-based estimates of chilipepper biomass for years 2003-2011.

Summary: Chilipepper biomass declined below the relative biomass target limit after 1980, then increased substantially in the 2000s. The short-term trawl survey information indicates a stable population in recent years.

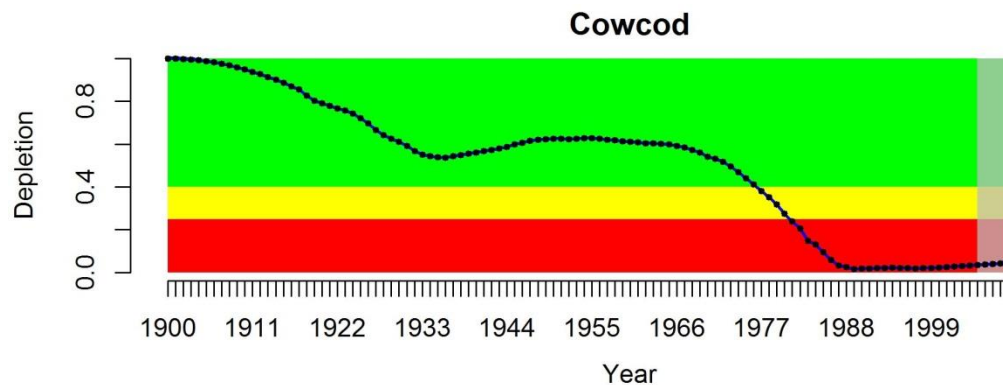


Figure GF21. Relative abundance trajectory 1900-2011 for cowcod.

Summary: Cowcod relative biomass is well below the limit reference point and has very slow growth in recent years, indicating a stable, but low population in recent years.

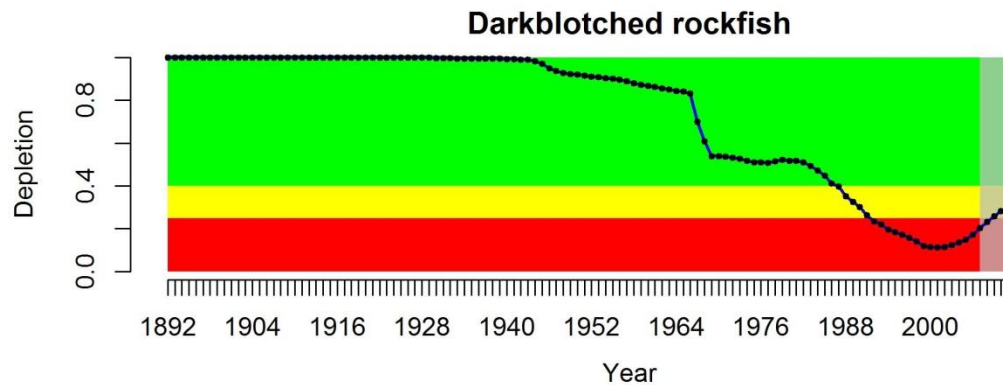


Figure GF22. Relative abundance trajectory 1910-2011 for darkblotched rockfish.

Summary: Darkblotched rockfish showed historical declines in population below relative biomass limits, but recent years show population increase above the limit.

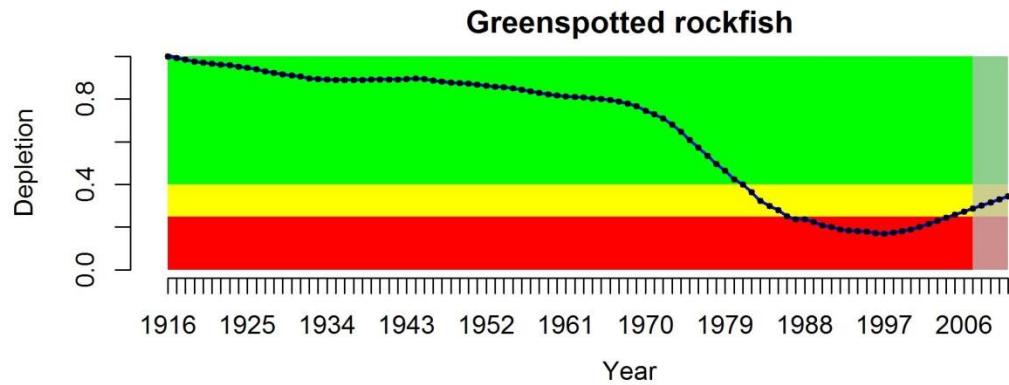


Figure GF23. Relative abundance trajectory 1916-2011 for greenspotted rockfish.

Summary: Greenspotted rockfish abundance historically dropped below the limit reference point, but is recently increasing and near the target relative biomass level.

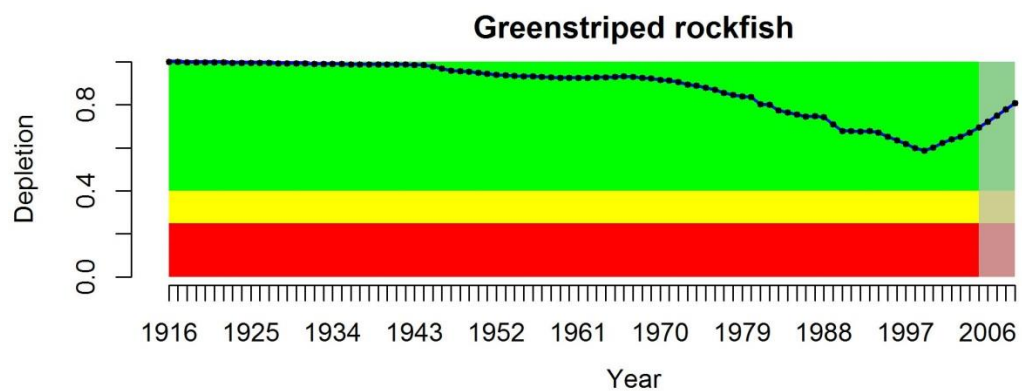


Figure GF24. Relative abundance trajectory 1910-2011 for greenstriped rockfish.

Summary: Greenstriped rockfish has stayed above the target relative biomass level with increasing biomass in the most recent years.

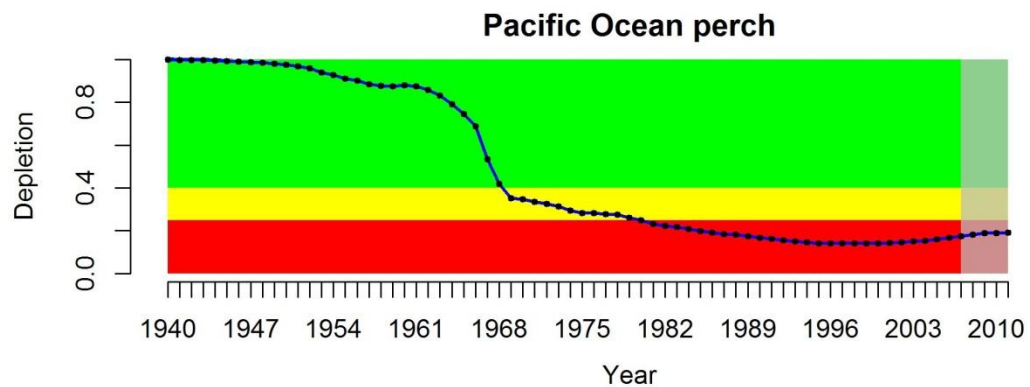


Figure GF25. Relative abundance trajectory 1940-2011 for Pacific Ocean perch.

Summary: Pacific Ocean perch biomass has shown a large historical decline and is currently below the relative biomass limit, though the population is steady in the most recent years.

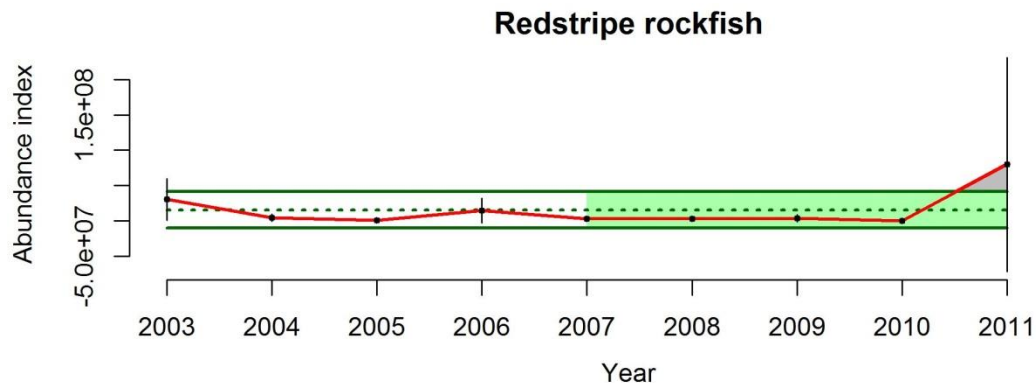


Figure GF26. Trawl survey design-based estimates of restripe rockfish biomass for years 2003-2011.

Summary: No redstripe rockfish assessment is available, so no baseline information on abundance exists. Recent years indicate a stable trend in survey abundance (the last relatively high point has large uncertainty).

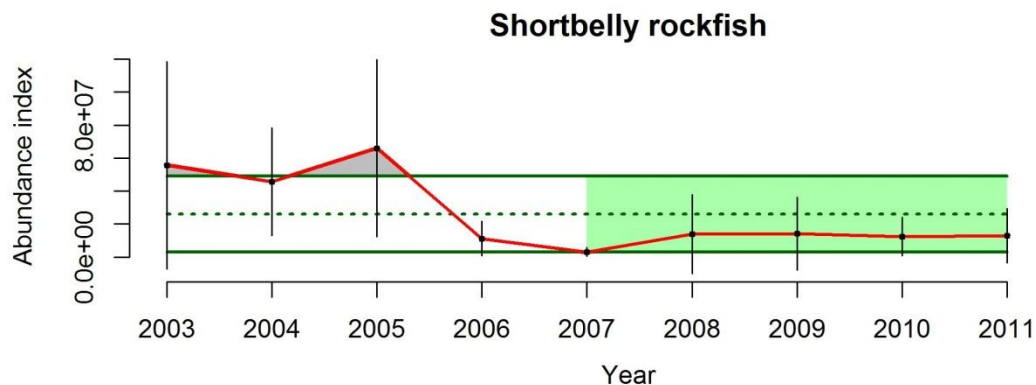


Figure GF27. Trawl survey design-based estimates of shortbelly rockfish biomass for years 2003-2011.

Summary: No shortbelly rockfish assessment is available, so no baseline information on abundance exists. Recent years indicate a stable trend in survey abundance.

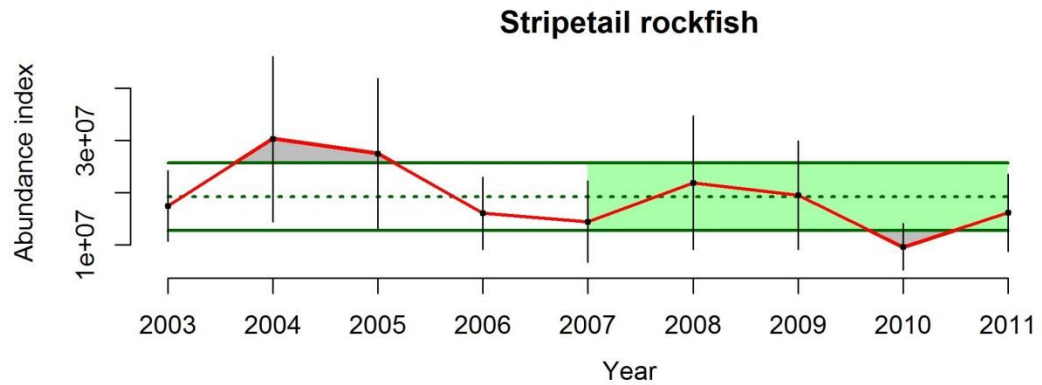


Figure GF28. Trawl survey design-based estimates of stripetail rockfish biomass for years 2003-2011.

Summary: No stripetail rockfish assessment is available, so no baseline information on abundance exists. Recent years indicate a stable trend in survey abundance.

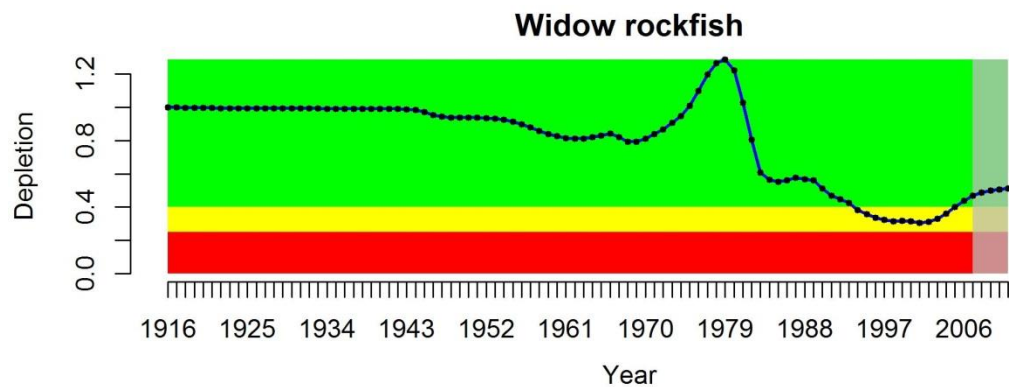


Figure GF29. Relative abundance trajectory 1916-2011 for widow rockfish.

Summary: Widow rockfish historically declined to below the target relative biomass level, but is currently increasing and is above the target.

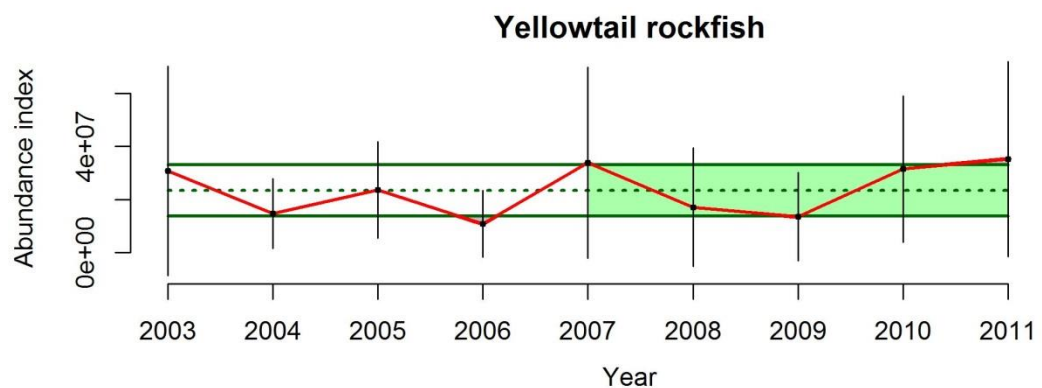


Figure GF30. Trawl survey design-based estimates of yellowtail rockfish biomass for years 2003-2011.

Summary: No yellowtail rockfish assessment is available, so no baseline information on abundance exists. Recent years indicate a stable trend in survey abundance.

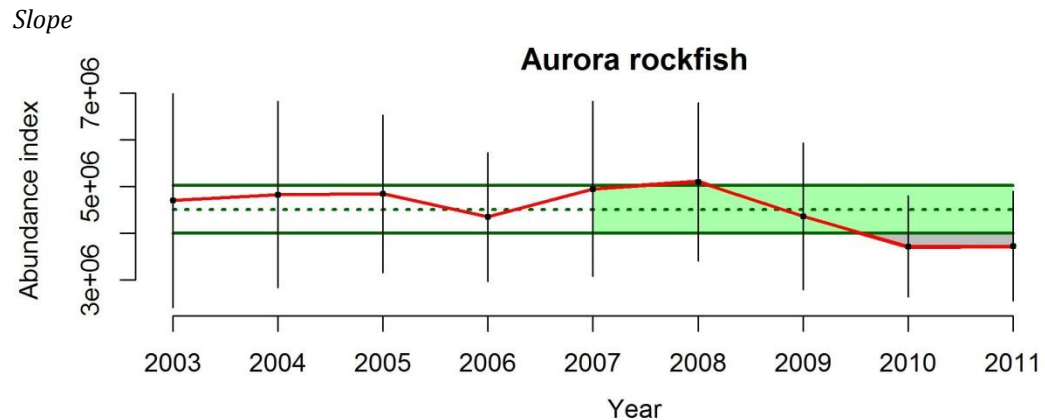


Figure GF31. Trawl survey design-based estimates of aurora rockfish biomass for years 2003-2011.

Summary: No aurora rockfish assessment is available, so no baseline information on abundance exists. Recent years indicate a declining trend in survey abundance.

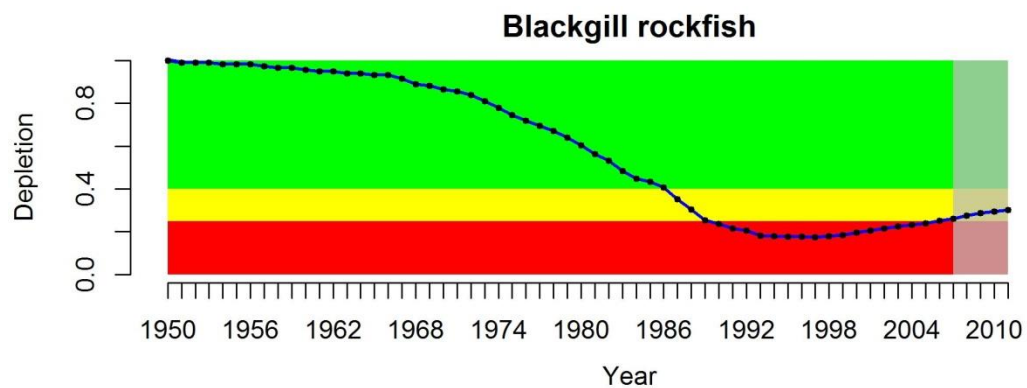


Figure GF32. Relative abundance trajectory 1950-2011 for Dover sole.

Summary: Blackgill rockfish show a historical decline below the limit relative abundance reference point with a slight increase over the last 10 years. The last 5 years show a stable population.

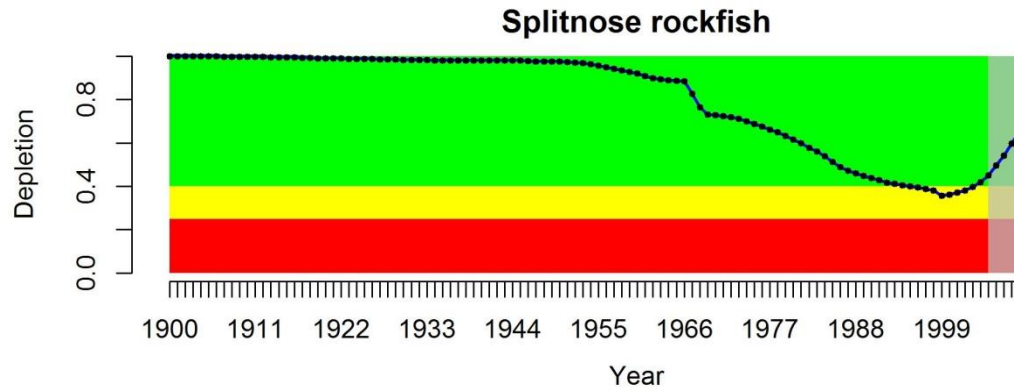


Figure GF33. Relative abundance trajectory 1910-2011 for Dover sole.

Summary: The splitnose rockfish population declined to below the target relative biomass in the late 1990s, but are currently increasing.

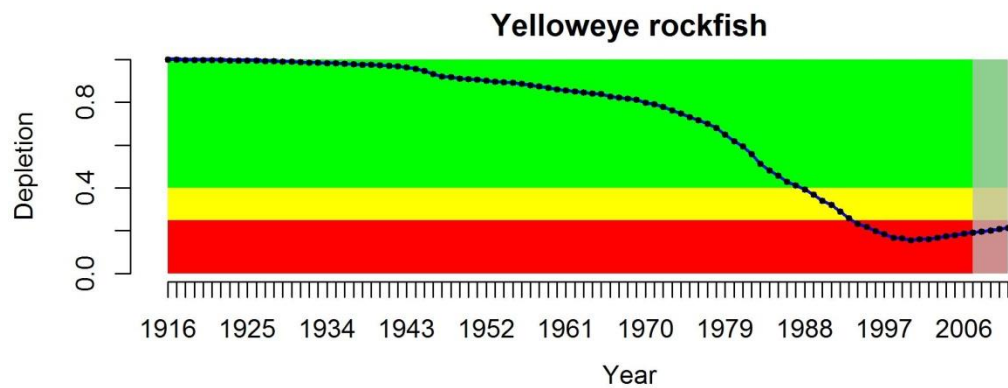


Figure GF34. Relative abundance trajectory 1916-2011 for yelloweye rockfish.

Summary: Yelloweye rockfish declined to below the limit relative biomass level and has stayed below since. Currently, the population is stable.

Roundfishes (N=4)
Nearshore

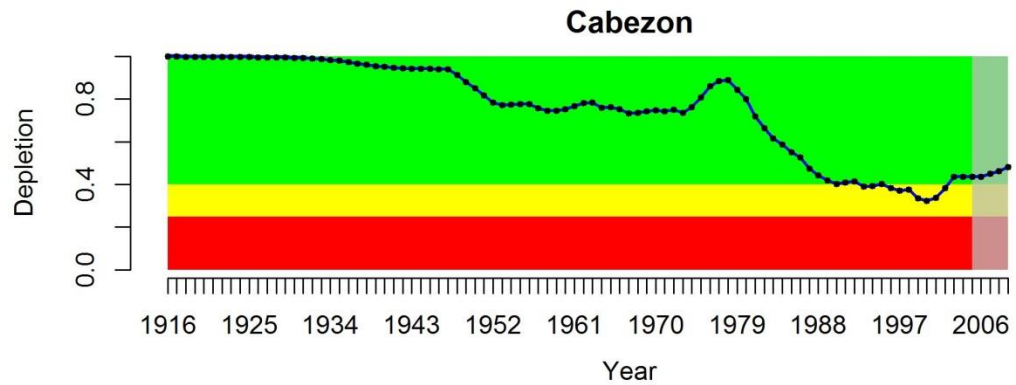


Figure GF35. Relative abundance trajectory 1910-2011 for cabezon.

Summary: Cabezon biomass had declined over the time series to below the relative biomass target level, but has since increased over the most recent years.

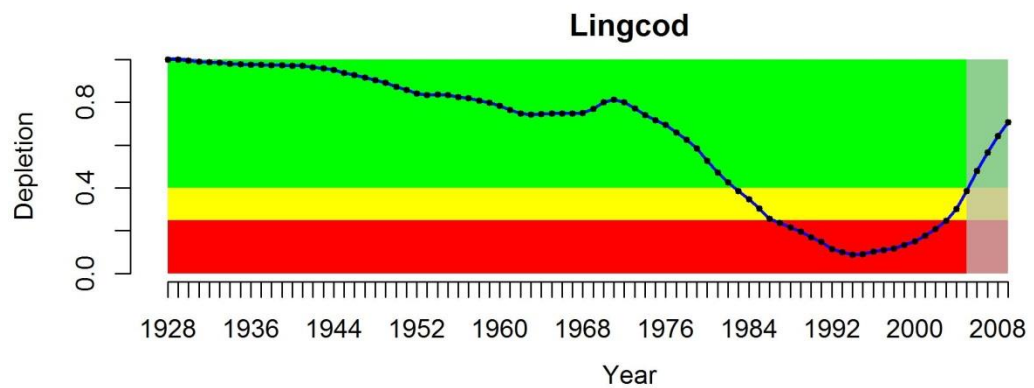


Figure GF36. Relative abundance trajectory 1910-2011 for lincod.

Summary: Lingcod biomass had declined over the time series to below the relative biomass limit reference point, but has since increased over the most recent years.

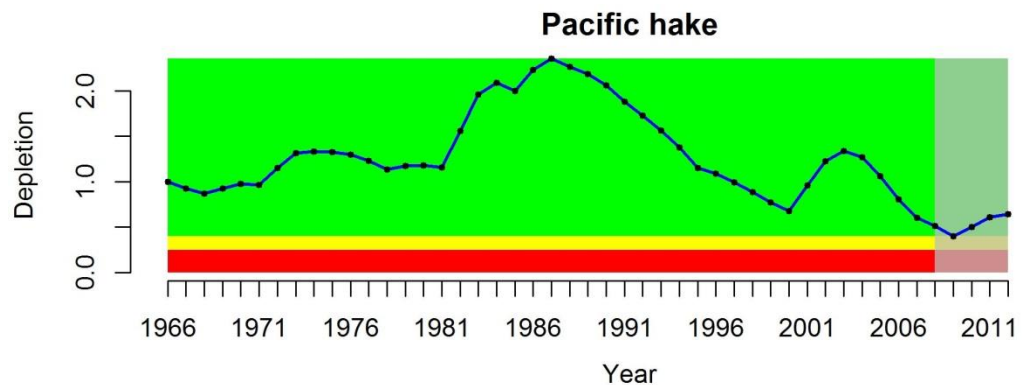


Figure GF37. Relative abundance trajectory 1910-2011 for Pacific hake.

Summary: Pacific hake biomass is very dynamic and is currently above the target relative biomass reference point with a recent increasing biomass trend.

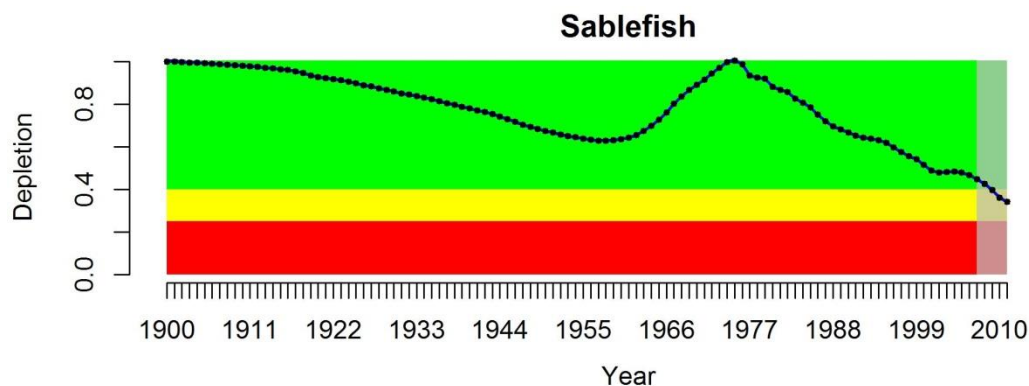


Figure GF38. Relative abundance trajectory 1910-2011 for sablefish.

Summary: Sablefish biomass is very dynamic and is currently below the target relative biomass reference point with a recent decreasing biomass trend.

INDICATOR #2: AGE AND LENGTH STRUCTURE

SUMMARY

The first indicator used female mature biomass as a status indicator, but biomass is made up of individuals with important characteristics such as age, size, and maturity status. This demographic structure of the stock is the second set of groundfish status indicators employed. Proportion maturity gives the percent of the population mature in a given year. The 95% age or length cumulative value indicates at which age or length 95% of the population is below, and thus is a measure of age/length truncation or expansion. All of the above values can be compared to the earliest value in the time series to indicate if they have changed over time. Female age and lengths are used exclusively to be comparable to the spawning biomass in the abundance trends indicator.

As with biomass, stock assessments are used as the primary source of information for maturity and age structure. If no stock assessment was available, trawl survey length compositions were used. Analyses of stocks with stock assessment from 2007 were also supplemented by the trawl length compositions. The same species grouping as used in the abundance indicators are also used to organize stock results.

Overall, age or length structure tended to show more changes over time than proportion mature (Table GF7). Long-term time series comparisons generally showed declines in these indicators, whereas short-term comparisons demonstrated more stability (Table GF7), suggested most change happened earlier in the fishery histories of these stocks. Non-elasmobranch groundfishes tended to see the most changes over time in both measures, with rockfishes being most sensitive to demographic changes (Table GF7). And though

it reasonable to expect these age/length-based indicators to be sensitive to yearly recruitment fluctuations, particularly large recruitments, changes in these indicators seemed more consistent with declines in spawning biomass, and thus deeper population structure changes, than recruitment variability.

Elasmobranchs (Figures GF39-GF42; ; Table GF7): Age or length structure showed little change in these elasmobranchs, but maturity did change in species with long time series. All measures were stable in the most recent years.

Flatfishes (Figures GF43-GF51; Table GF7): Flatfishes in the shelf showed decreases over time in both measures, while the deeper slope species showed little change over time in either measure.

Rockfishes (Figures GF52-GF70; Table GF7): Rockfishes showed a general decline in both measures through time, regardless of the adult habitat. Chilipepper was the one exception, which shows little change over the entire time series. Greenstriped (Figure GF60) and widow rockfish (Figure GF65) show contemporary measures have increased near initial conditions after historical declines. Stripetail (Figure GF64) and yellowtail rockfish (Figure GF66) show little change in the trawl survey lengths, but there is no historical baseline to interpret these values. Overall, rockfishes were the most sensitive species group to demographic changes.

Roundfishes (Figures GF71-GF73; Table GF7): Two of three roundfishes (cabezon and lingcod, both shallow egg-layers with nest-guarding males) showed declines in both measures, whereas sablefish showed little change over time. Lingcod has shown recent increases in both measures.

SPECIFIC TIME SERIES

Elasmobranchs (N=3)

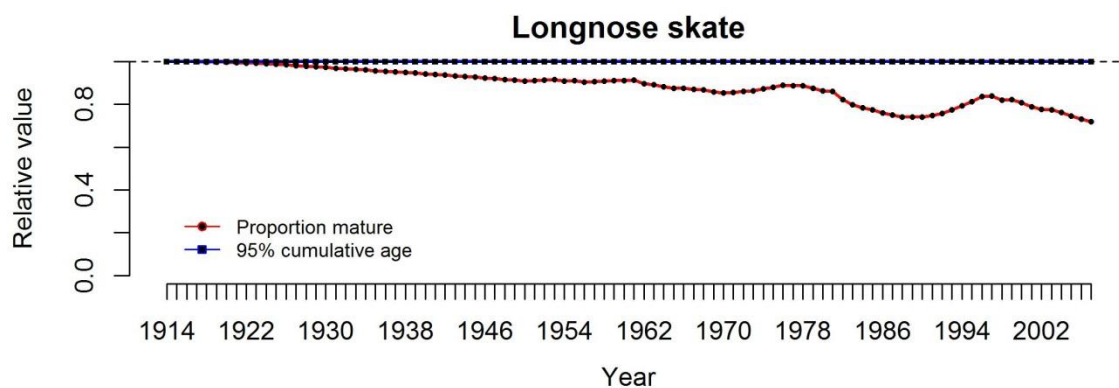


Figure GF39. Proportion of the longnose skate population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

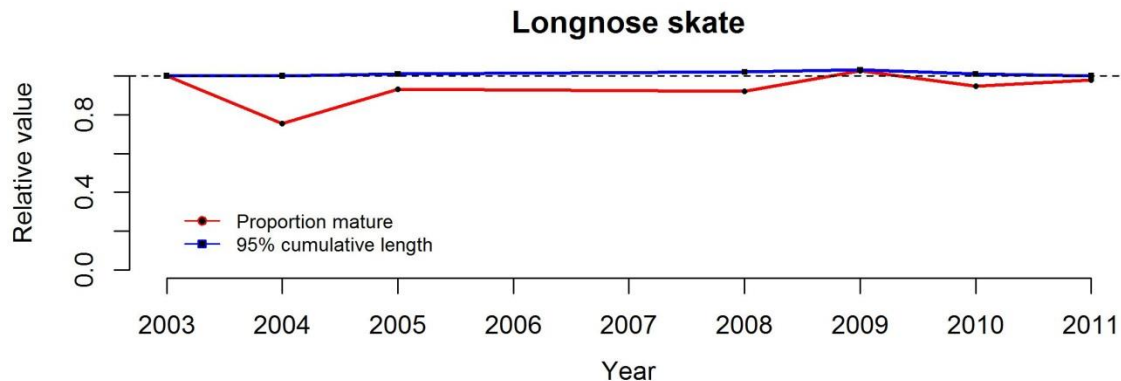


Figure GF40. Proportion of the longnose skate population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: Longnose skate have shown no decline in the proportion of the oldest ages and largest lengths, but proportion mature has declined somewhat over the length of the time series.

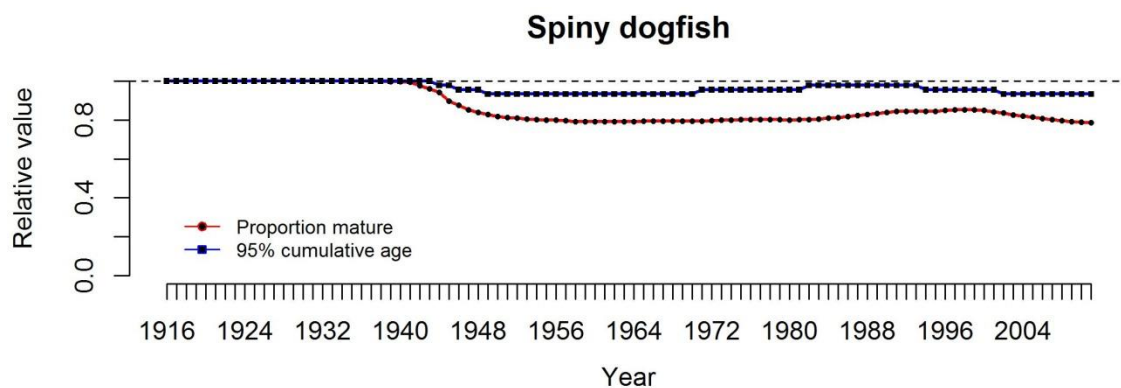


Figure GF41. Proportion of the spiny dogfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Spiny dogfish show only small declines in proportion mature and proportion of the oldest ages that have mostly stabilized since the decline in the 1940s.

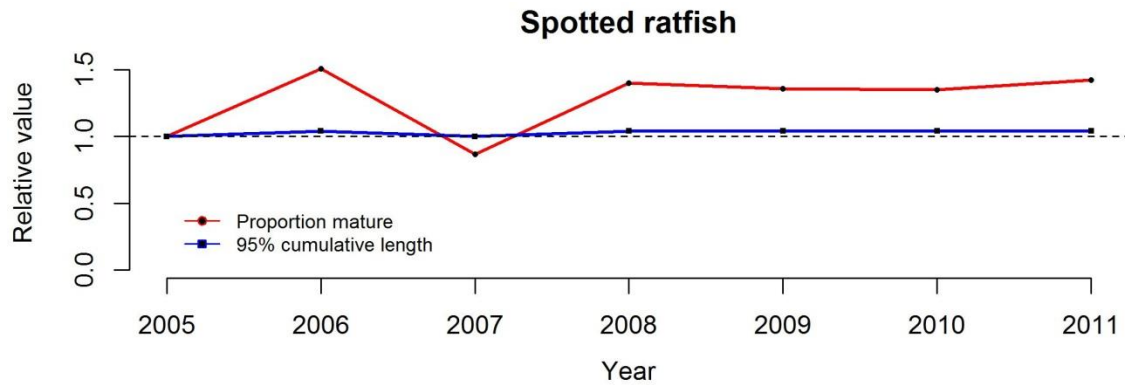


Figure GF42. Proportion of the spotted ratfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2005) of the trawl survey time series.

Summary: No stock assessment is available for spotted ratfish so no baseline information on demographic structure is available. No declines in either maturity or proportion of the largest sizes are apparent from the trawl survey data.

Flatfishes (N=7)

Shelf

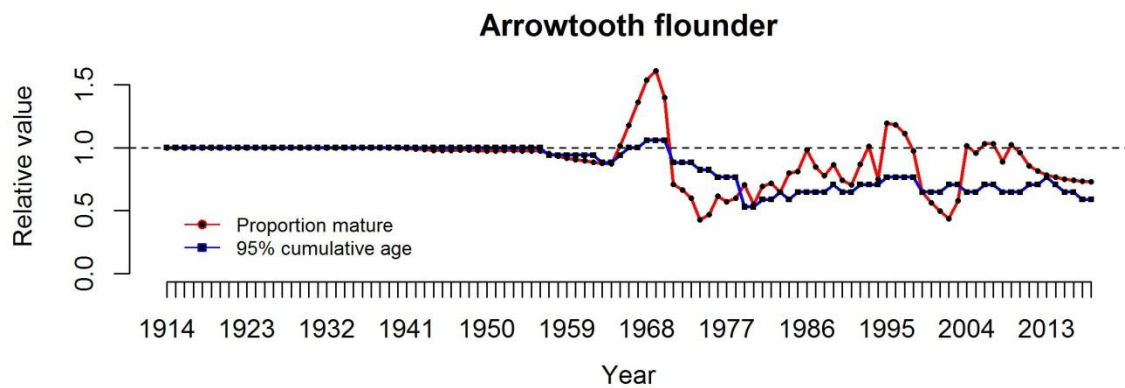


Figure GF43. Proportion of the arrowtooth flounder population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

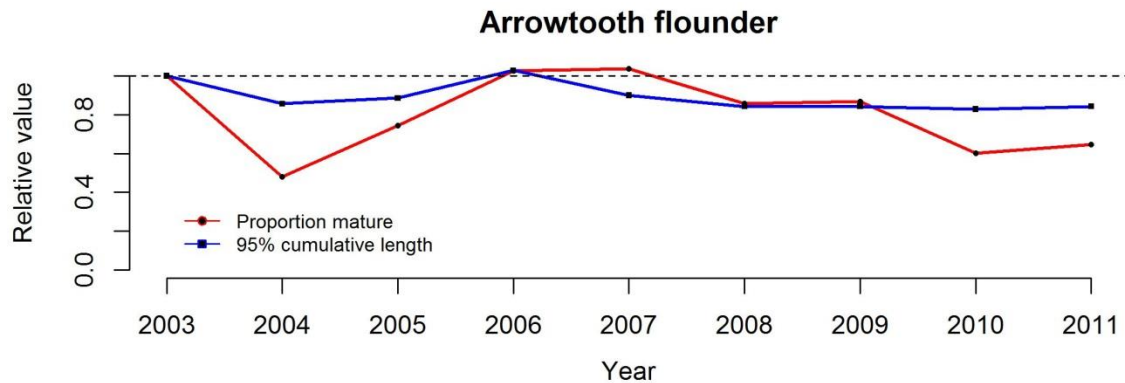


Figure GF44. Proportion of the arrowtooth flounder population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: Arrowtooth flounder show declines in proportion mature and proportion of the oldest ages and largest lengths over the length of the time series.

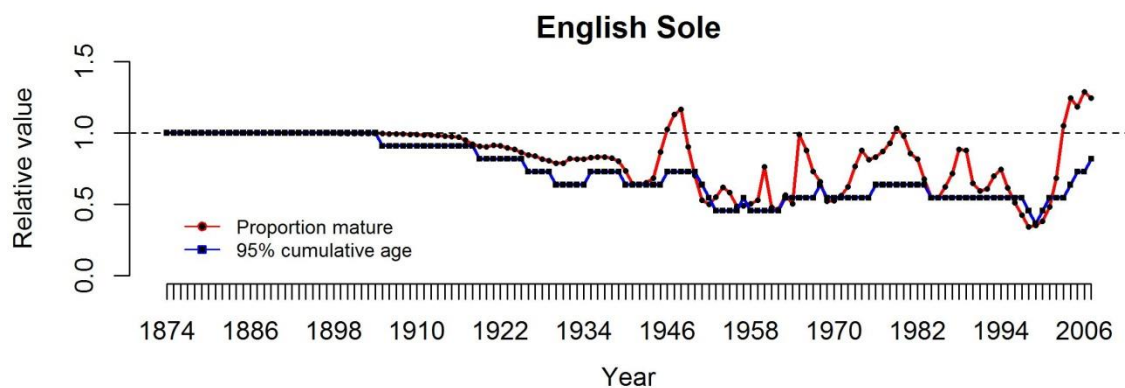


Figure GF45. Proportion of the English sole population mature (red) and at the 95% cumulative age (blue) relative to the first year (1876) of the time series.

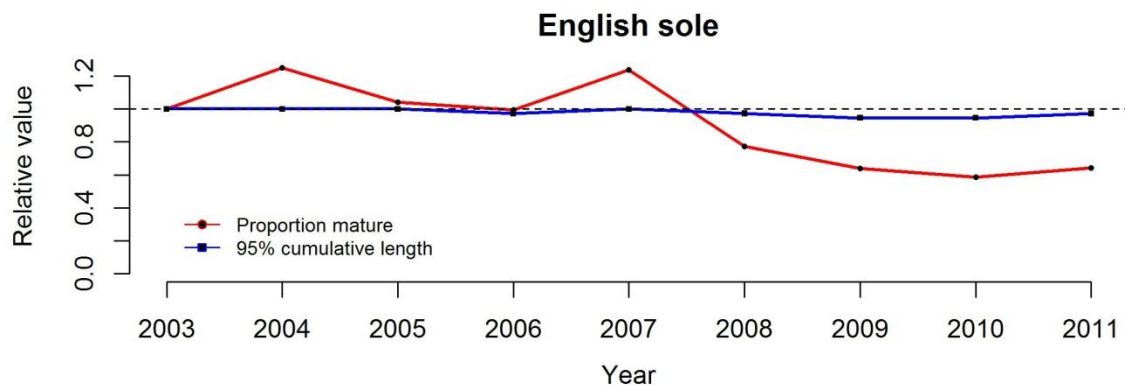


Figure GF46. Proportion of the English sole population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: English sole show a slight decline in proportion of the oldest ages, but not in proportion mature, over the length of the time series. Recent survey trends in proportion mature are downward.

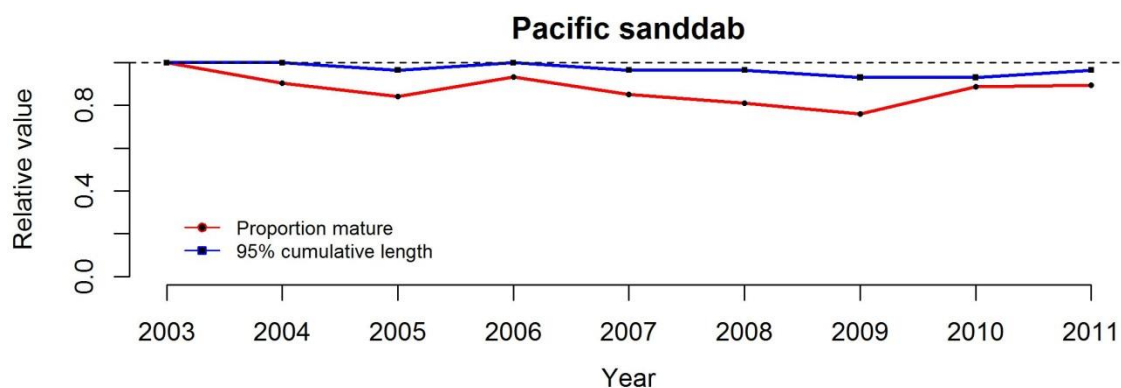


Figure GF47. Proportion of the Pacific sanddab population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: No stock assessment is available for Pacific sanddab so no baseline information on demographic structure is available. No declines in either maturity or proportion of the largest lengths are apparent from the trawl survey data.

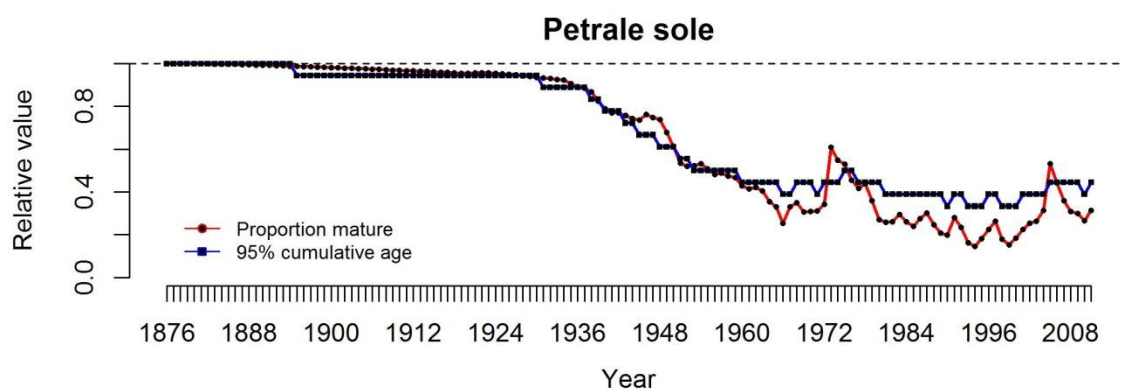


Figure GF48. Proportion of the Petrale sole population mature (red) and at the 95% cumulative age (blue) relative to the first year (1876) of the time series.

Summary: Petrale sole shows notable declines in proportion mature and proportion of the oldest ages over the length of the time series.

Slope

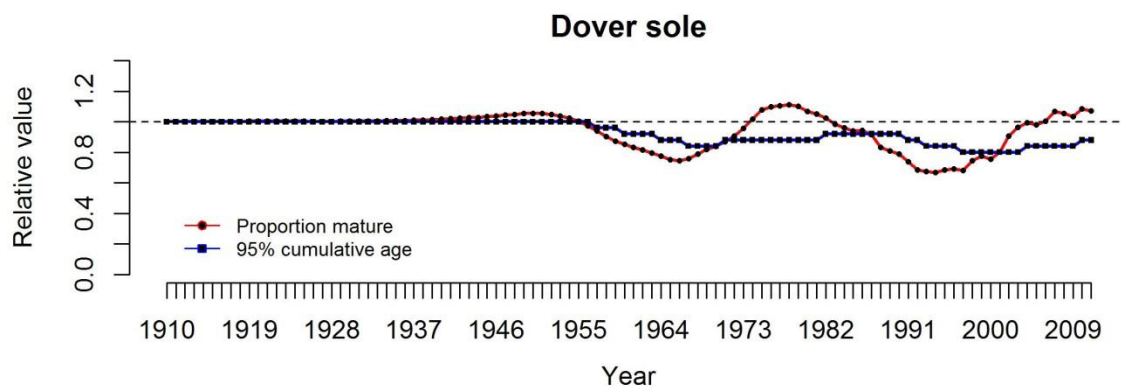


Figure GF49. Proportion of the Dover sole population mature (red) and at the 95% cumulative age (blue) relative to the first year (1910) of the time series.

Summary: Dover sole do not show any notable changes in proportion mature and proportion of the oldest ages and largest lengths over the length of the time series.

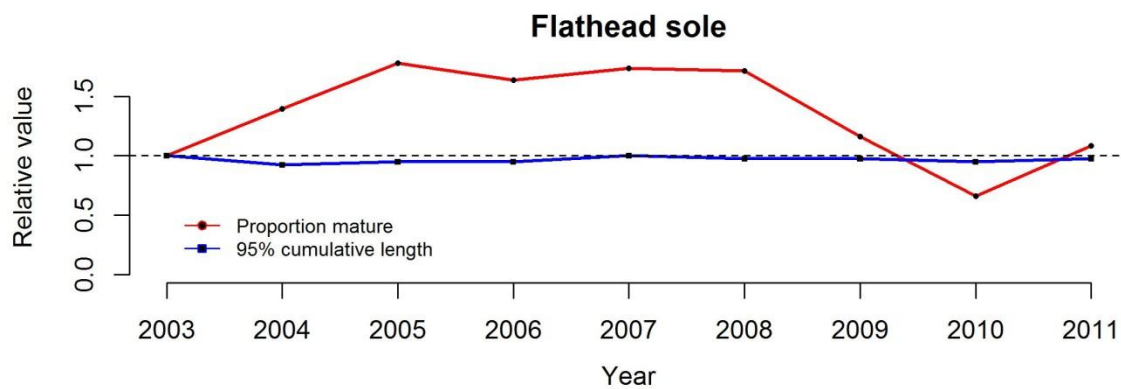


Figure GF50. Proportion of the flathead sole population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: No stock assessment is available for flathead sole so no baseline information on demographic structure is available. No declines in either maturity or proportion of the largest lengths are apparent from the trawl survey data.

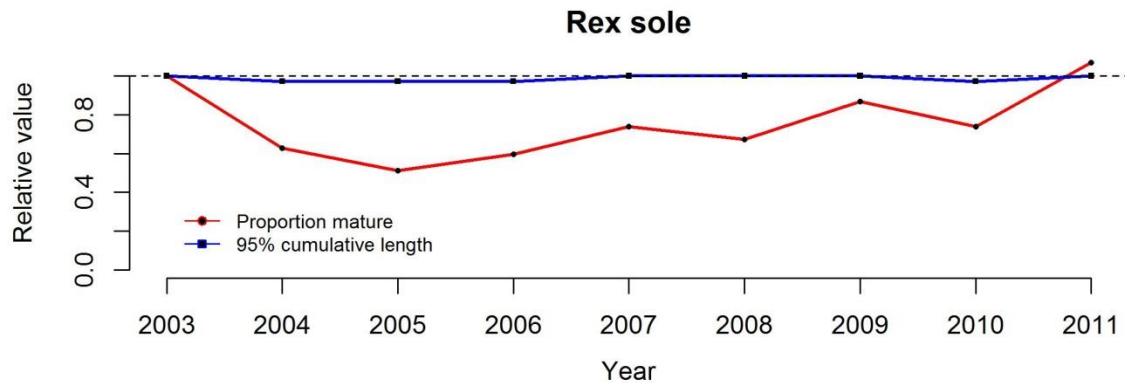


Figure GF51. Proportion of the rex sole population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: No stock assessment is available for rex sole so no baseline information on demographic structure is available. No declines in either maturity or proportion of the largest lengths are apparent from the trawl survey data.

Rockfishes (N=18)

Nearshore

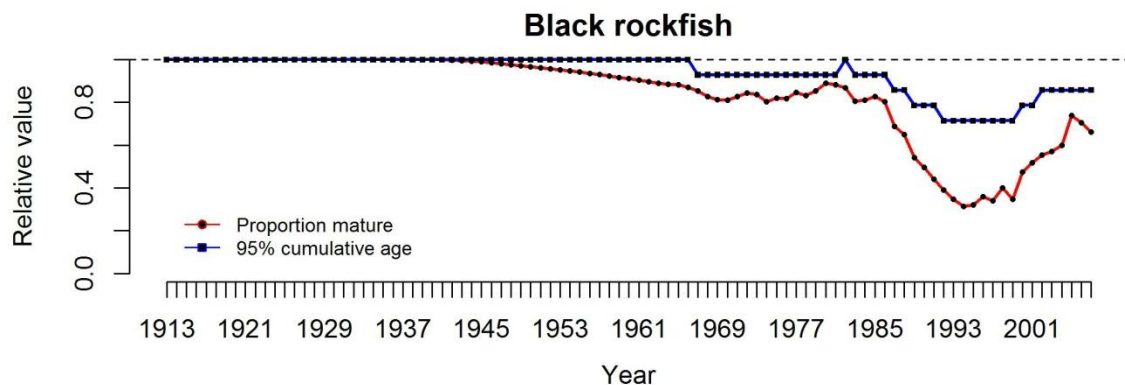


Figure GF52. Proportion of the black rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Black rockfish show notable declines in proportion mature and slight declines in proportion of the oldest ages over the length of the time series.

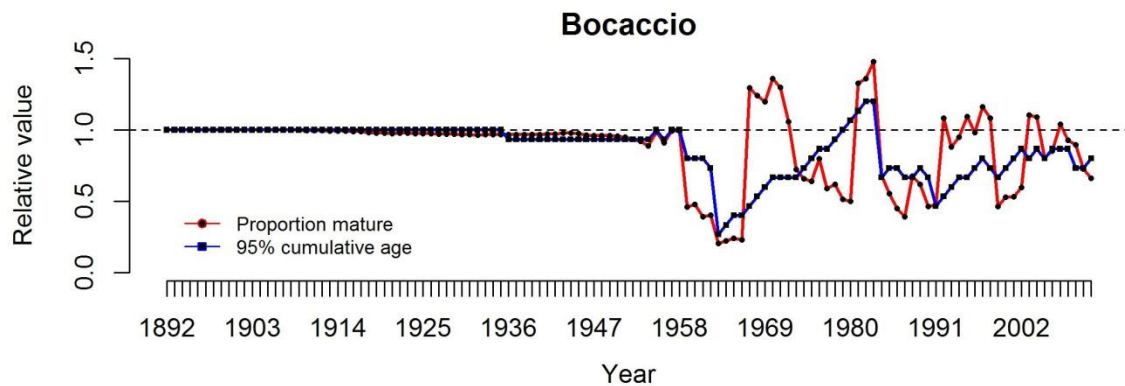


Figure GF53. Proportion of the bocaccio population mature (red) and at the 95% cumulative age (blue) relative to the first year (1895) of the time series.

Summary: Bocaccio show high variation in the proportion mature and proportion of the oldest ages over the length of the time series. The most recent measure are below historical reference levels. Fluctuations may be due to high but sporadic

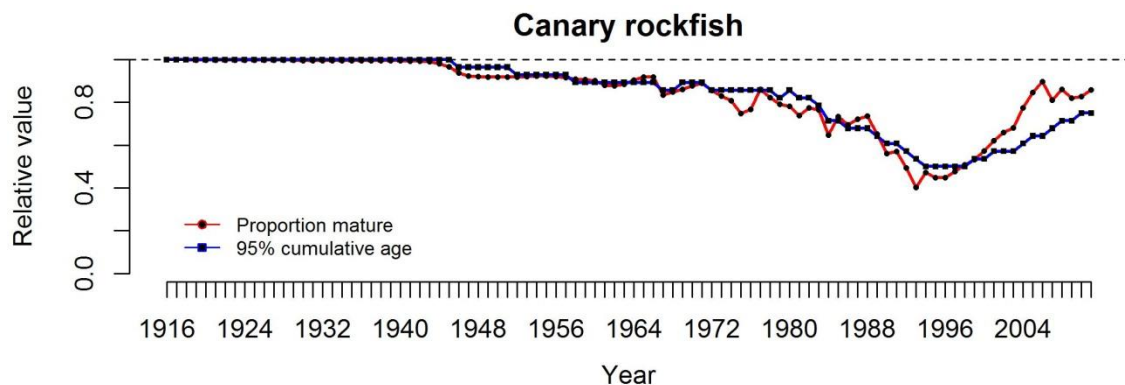


Figure GF54. Proportion of the canary rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Canary rockfish show declines in proportion mature and proportion of the oldest ages over the length of the time series, but current years demonstrate a building up of both metrics.

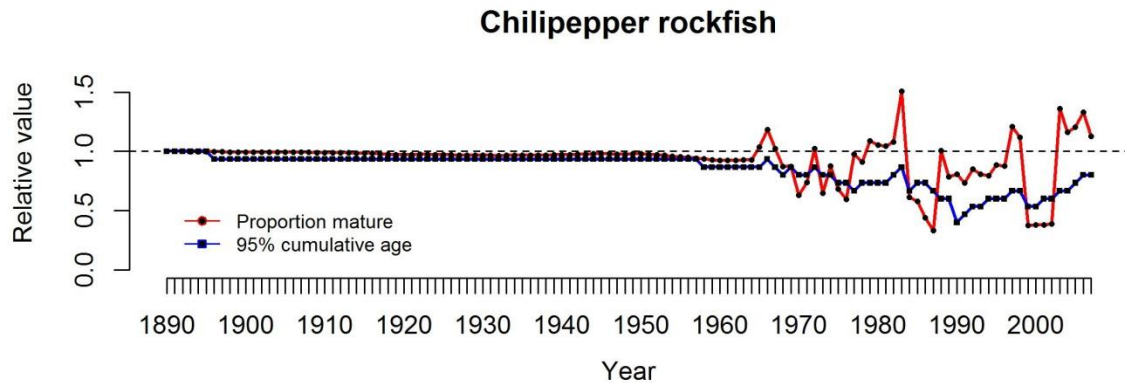


Figure GF55. Proportion of the chilipepper rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1892) of the time series (1892-2007).

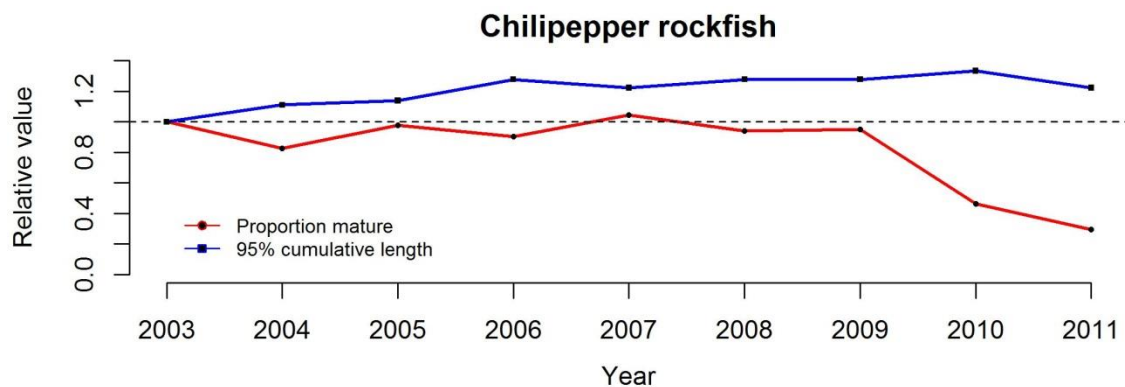


Figure GF56. Proportion of the chilipepper rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series (2003-2007).

Summary: Chilipepper rockfish show decreases in proportion mature and proportion of the oldest ages and largest lengths over the length of the time series. The short-term series shows a relative changes consistent with the long-time series when the same relative time frame is considered.

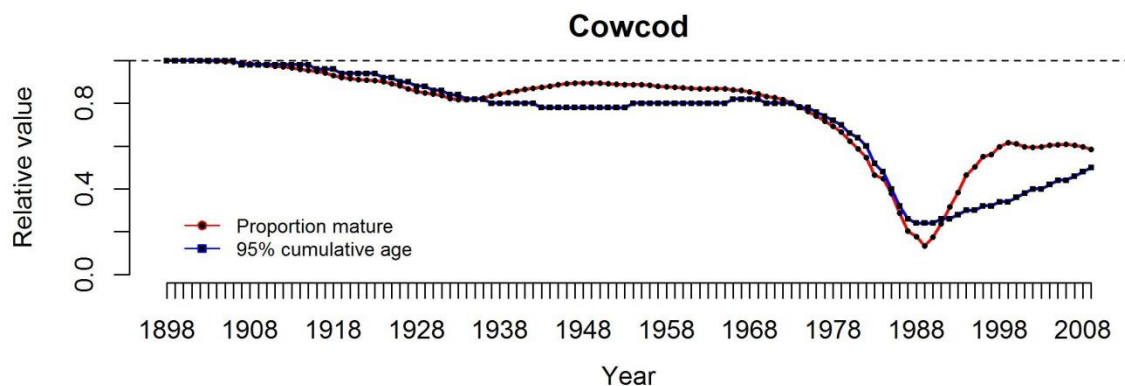


Figure GF57. Proportion of the cowcod population mature (red) and at the 95% cumulative age (blue) relative to the first year (1900) of the time series.

Summary: Cowcod show declines in proportion mature and proportion of the oldest ages over the length of the time series.

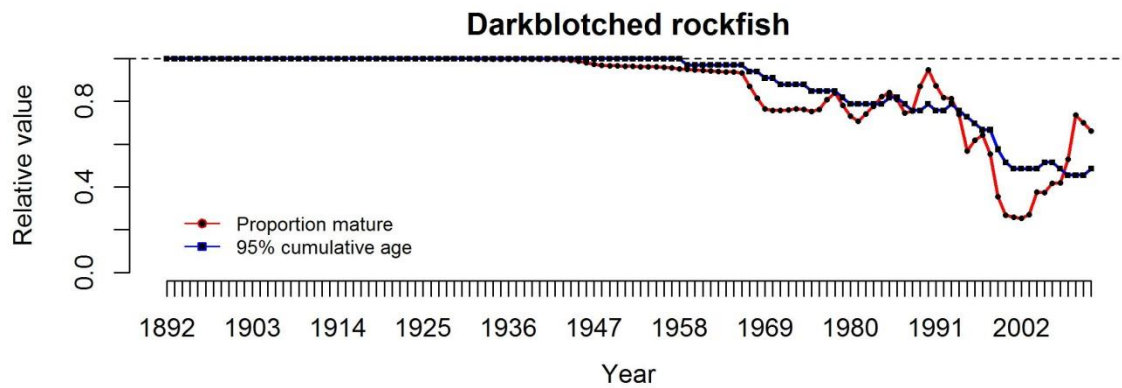


Figure GF58. Proportion of the darkblotched rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1895) of the time series.

Summary: Darkblotched rockfish show declines in proportion mature and proportion of the oldest ages over the length of the time series.

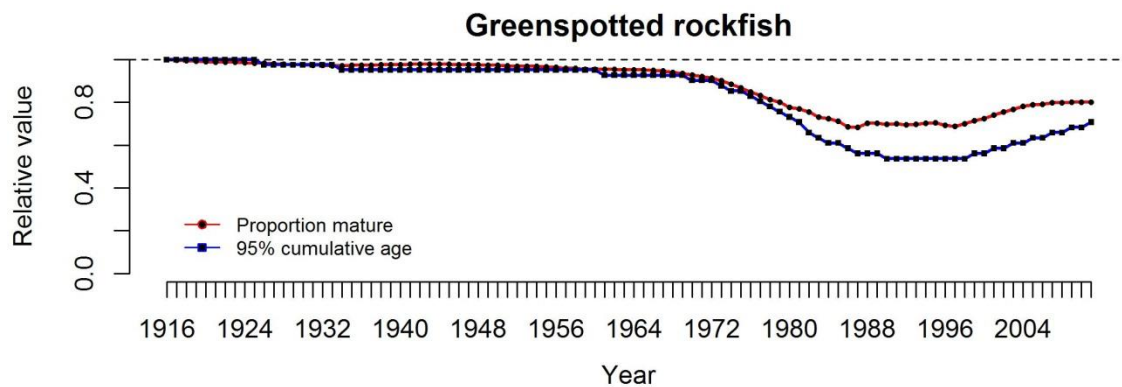


Figure GF59. Proportion of the greenspotted rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Greenspotted rockfish show declines in proportion mature and proportion of the oldest ages over the length of the time series.

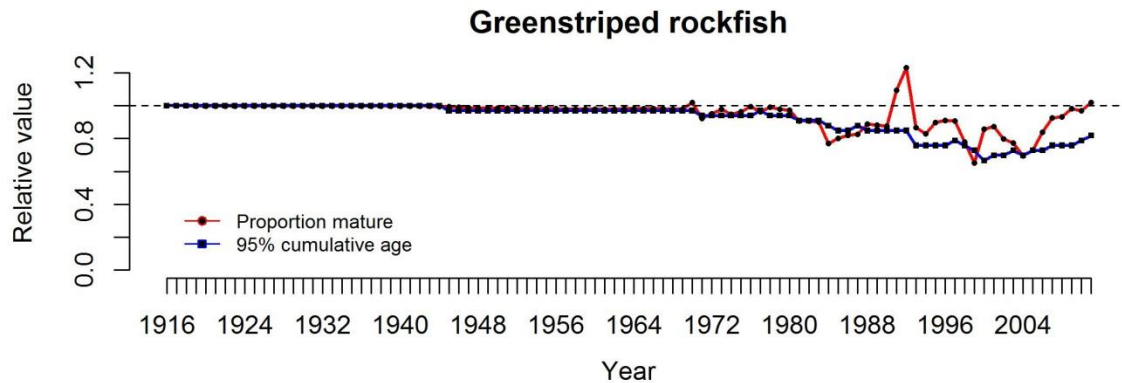


Figure GF60. Proportion of the greenstriped rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Greenstriped rockfish show little change in proportion mature and proportion of the oldest ages over the length of the time series, with only a small decrease in population structure.

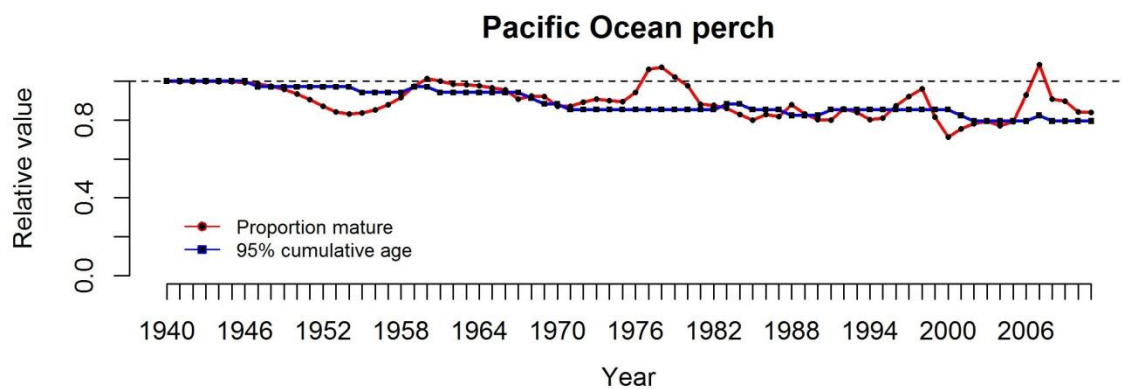


Figure GF61. Proportion of the Pacific Ocean perch population mature (red) and at the 95% cumulative age (blue) relative to the first year (1940) of the time series.

Summary: Pacific Ocean perch show low levels of decline in proportion mature and proportion of the oldest ages over the length of the time series.

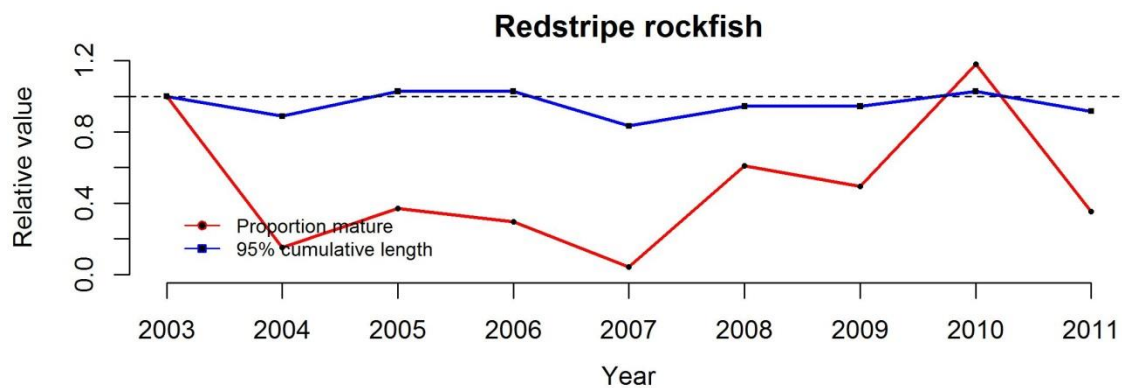


Figure GF62. Proportion of the redstripe rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: No stock assessment is available for redstripe rockfish so no baseline information on demographic structure is available. No declines in maturity are apparent from the trawl survey data, though proportion of the largest lengths is variable across years with a notable decline.

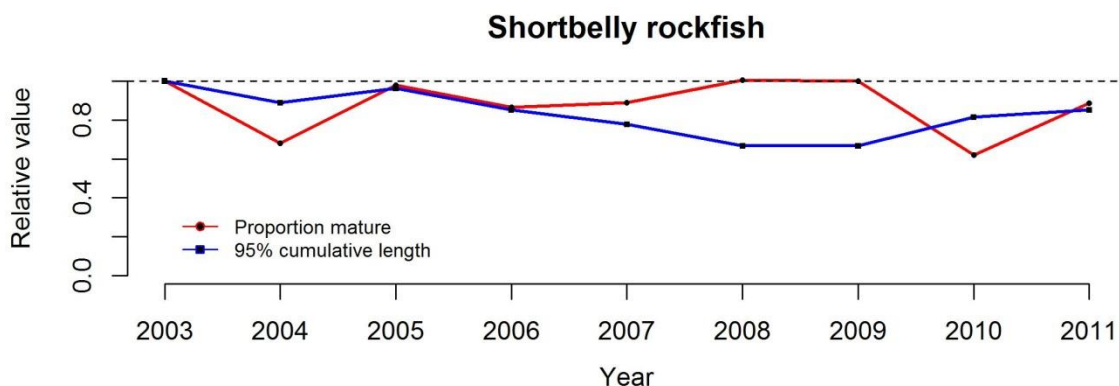


Figure GF63. Proportion of the shortbelly rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: Only modest declines in either maturity or proportion of the largest lengths are apparent from the trawl survey data for shortbelly rockfish.

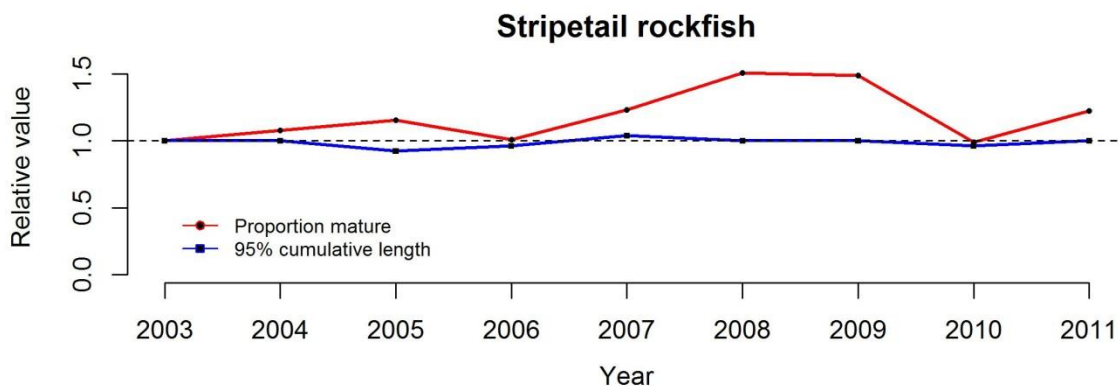


Figure GF64. Proportion of the stripetail rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: No stock assessment is available for stripetail rockfish so no baseline information on demographic structure is available. No declines in either maturity or proportion of the largest lengths are apparent from the trawl survey data.

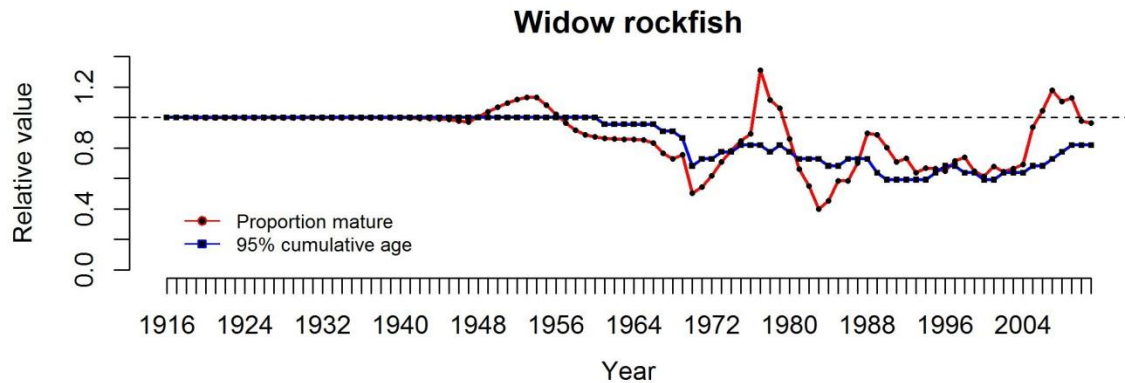


Figure GF65. Proportion of the widow rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Widow rockfish show no declines in proportion mature and population structure over the length of the time series that have returned or are building back towards historical levels.

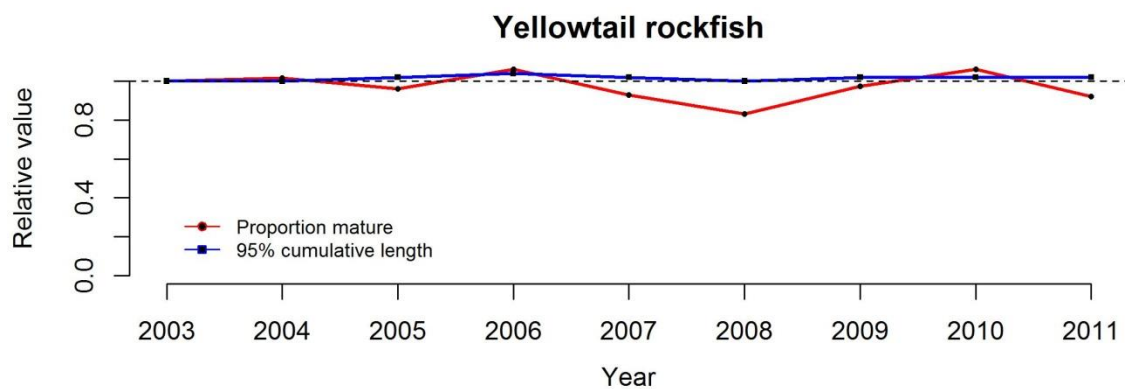


Figure GF66. Proportion of the yellowtail rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: No declines in either maturity or proportion of the largest lengths are apparent from the trawl survey data for yellowtail rockfish.

Slope

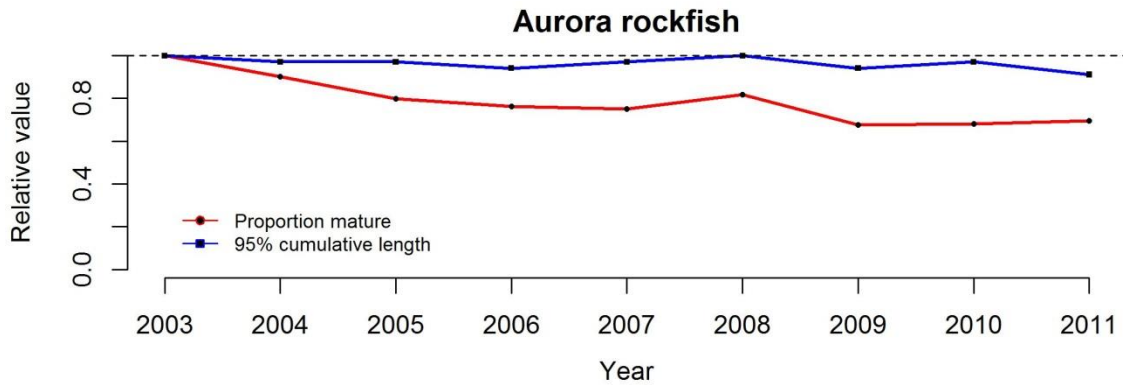


Figure GF67. Proportion of the aurora rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: No stock assessment is available for aurora rockfish so no baseline information on demographic structure is available. Modest declines proportion of the largest lengths, but stronger declines in proportion mature are apparent from the trawl survey data.

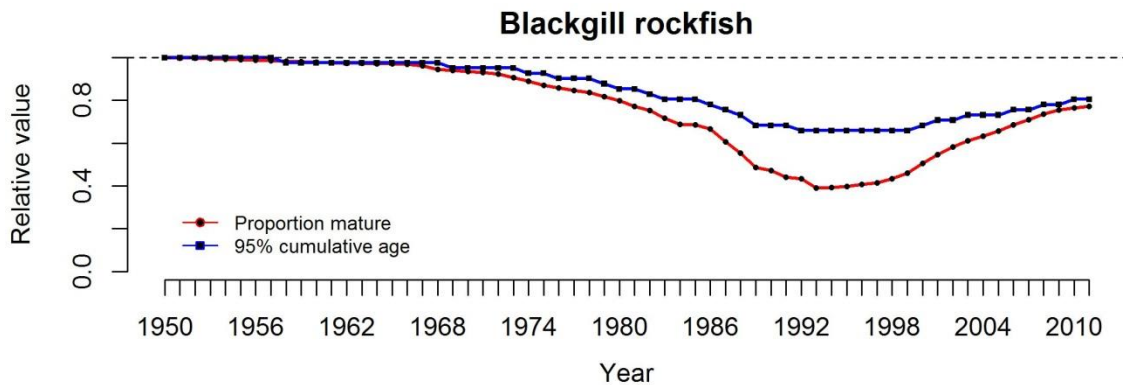


Figure GF68. Proportion of the blackgill rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1950) of the time series.

Summary: Blackgill rockfish show declines in proportion mature and proportion of the oldest ages over the length of the time series.

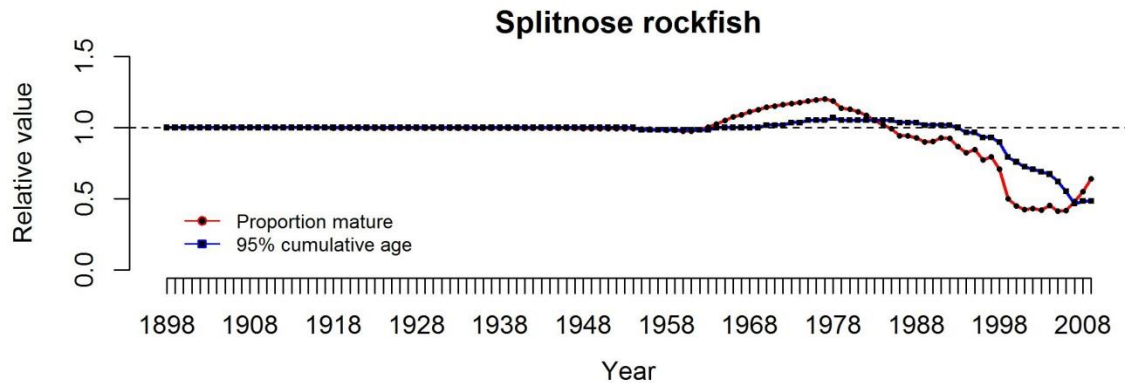


Figure GF69. Proportion of the splitnose rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1900) of the time series.

Summary: Splitnose rockfish show declines in proportion mature and proportion of the oldest ages over the length of the time series.

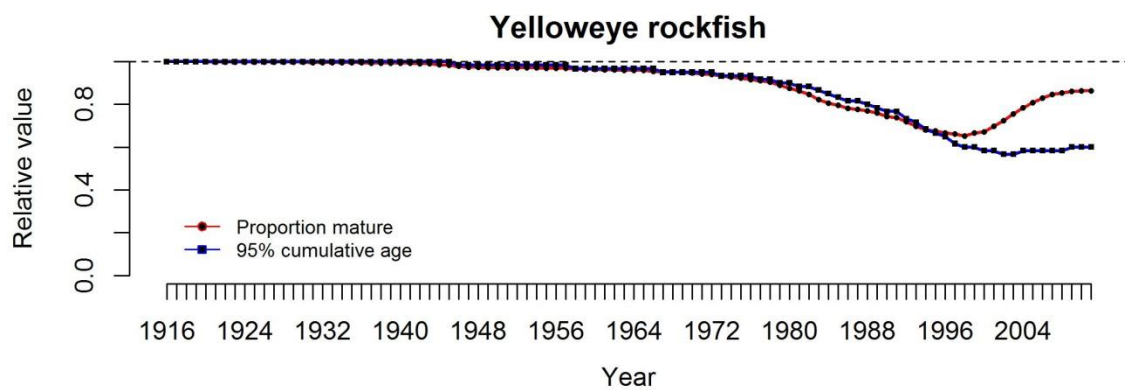


Figure GF70. Proportion of the yelloweye rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Yelloweye rockfish show have declines in proportion mature and proportion of the oldest ages over the length of the time series.

Other fishes (n=3)

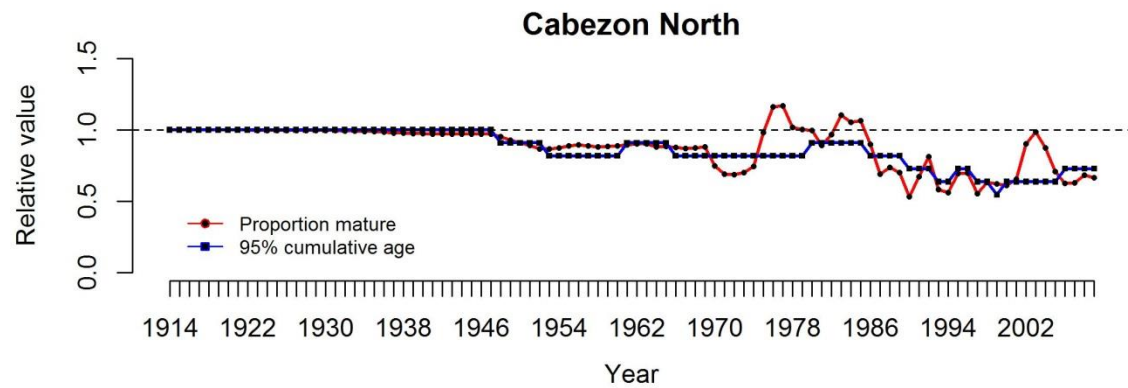


Figure GF71. Proportion of the cabezon population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Cabezon show declines in proportion mature and proportion of the oldest ages over the length of the time series.

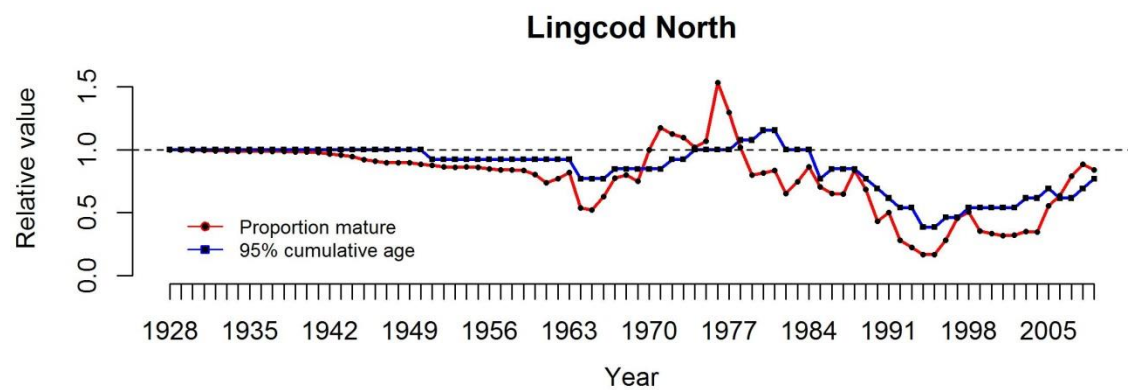


Figure GF72. Proportion of the lingcod population mature (red) and at the 95% cumulative age (blue) relative to the first year (1930) of the time series.

Summary: Lingcod show declines in proportion mature and proportion of the oldest ages that have recently shown increases towards historical levels.

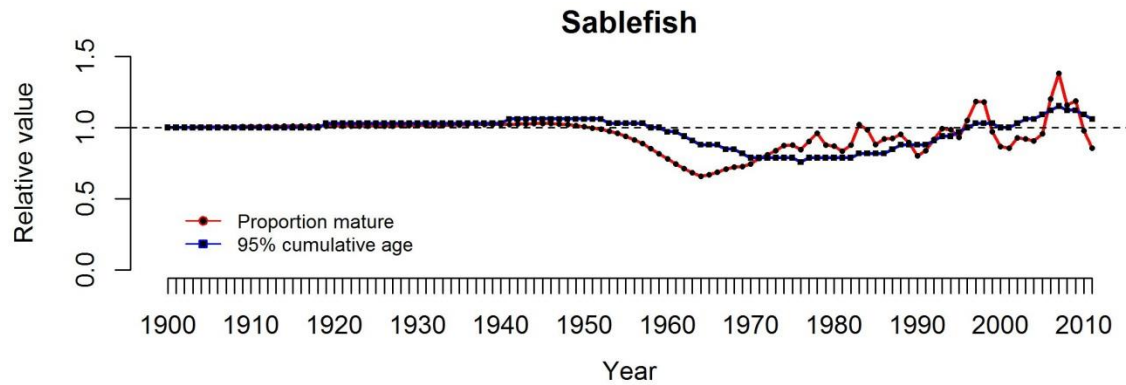


Figure GF73. Proportion of the sablefish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1900) of the time series.

Summary: Sablefish show little change in proportion mature and proportion of the oldest ages over the length of the time series.

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RESILIENT AND ECONOMICALLY VIABLE COASTAL COMMUNITIES

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TABLE OF CONTENTS (HD)

| | |
|-----------------------------------|-----|
| Executive summary..... | 476 |
| Detailed report..... | 479 |
| Indicator selection process | 481 |
| Indicator evaluation..... | 481 |
| Status and trends..... | 490 |
| Major findings..... | 490 |
| Summary and status of trends..... | 490 |
| Risk..... | 497 |
| Literature Cited | 497 |

LIST OF TABLES AND FIGURES (HD)

| | |
|---|-----|
| Figure HD. Trends in average diversification for US West Coast and Alaskan fishing vessels with over \$5k in average revenues (top left panel) and for vessels with 2010 West Coast revenue >\$5k (top right and bottom panels)..... | 478 |
| Figure HD1. Diagram of indices and factor analysis approach toward vulnerability for the human communities of the CCLME. *As determined by the U.S. Census (i.e. U.S. Census-Designated Places [CDPs] with coastal counties adjacent to the CCLME)..... | 480 |
| Table HD1: Species groups used for diversification indices..... | 489 |
| Figure HD 2. Trends in average diversification for US West Coast and Alaskan fishing vessels (left panels) and the 2010 West Coast Fleet (right panel) filtered by all vessels with over \$5k in average revenues (top panel), by average gross revenues classes (middle panel) and by vessel length classes (bottom panel)..... | 492 |
| Figure HD3. Histograms showing percentage of vessels by ranges of Herfindahl-Hirschman index scores for US West Coast and Alaskan fishing vessels (left panels) and the 2010 West Coast Fleet (right panel) filtered by all vessels with over \$5k in average revenues (top panel), by average gross revenues classes (middle panel) and by vessel length classes (bottom panel). | 494 |
| Figure HD4. Fitted relationships between the coefficient of variation (CV) of gross revenues for US West Coast and Alaskan fishing vessels (left panels) and the 2010 West Coast Fleet (right panel) filtered by all vessels with over \$5k in average revenues (top panel), by average gross revenues classes (middle panel) and by vessel length classes (bottom panel)..... | 495 |
| Table HD2. Predicted coefficient of variation (CV) of gross fishery revenue for Herfindahl-Hirschman index scores associated with alternative diversification schemes for groupings of WC and AK fishing vessels. | 496 |

OVERVIEW

The current fleet of vessels off the U.S. West Coast and Alaska is less diverse than at any point in the past 30 years.

EXECUTIVE SUMMARY

Human dimensions of the California Current Large Marine Ecosystem are included in several sections of the IEA. Human activities that potentially influence the status of natural ecosystem components are detailed in the Human Pressures section of the report. In the Management Evaluation and Scenarios section of the IEA, human dimensions are included as economic outcomes of alternative futures. Additionally, in this section, we used economic models to predict the response of fishing fleets to various management options.

In this section, we focus on the status and trends of coastal communities that are dependent on the natural resources of the California Current. This new research is ongoing and incomplete, and here we provide details of what we have completed thus far and future research directions.

In this chapter, we identify a set of proposed indices. Each index is a composite of 3 to 5 metrics and, considered together, the indices focus on the degree to which coastal communities rely on marine resources and are socioeconomically vulnerable. The integrated ecosystem assessment focuses on status and trends in focal species and focal components. In much the same way, a focus on those coastal communities most directly linked to the ecosystem via fishing provides a first step in index selection.

The California Current lies adjacent to and is a part of the socioeconomic fabric of numerous coastal communities in Washington, Oregon and California, including 123 communities previously identified as “dependent” on or “engaged” in West Coast fisheries through a fisheries Data Envelopment Analysis methodology (Sepez et al. 2007). While coastal communities are linked to the California Current ecosystem in numerous ways, in the context of the IEA we will initially focus on the communities linked to the CCLME via fishing. The Commercial Fishing Reliance Index was adapted from work on the U.S. East Coast because the index allows for the integration of readily accessible data on a multi-year basis, therefore elucidating long term trends, and also identifies communities that are expected to respond to changes in environmental conditions, potentially proving to be salient in vulnerability analyses (Clay and Olson 2008, Colburn and Jepson 2012).

Once the communities most reliant on commercial fisheries are identified, statistical analyses of subsequent indices can assess these communities in terms of their socioeconomic vulnerability. Our indices of socioeconomic vulnerability include a Population Composition Index, Poverty Index, Personal Disruptions Index and a Fishery Income Diversification Indicator.

While much of this approach has been successfully developed and implemented for coastal communities on the U.S. East Coast and Gulf Coast (Jacob et al. 2012), this method of measuring and evaluating socioeconomic resilience is still in the early stages of data collection and data organization for the communities of the coastal portions of the California Current Large Marine Ecosystem and the coastal communities of Alaska. These indices seek to account for socioeconomic vulnerability of California Current coastal communities and may therefore be linked to the CCIEA. Since data collection, organization, and analyses are ongoing, status and trends for these indices are not yet determined.

The Fishery Income Diversification Indicator presents a final single indicator, rather than an index, and is measured at the vessel level, as opposed to the community level. However this indicator provides some indication of status and trends for those individuals engaged with West Coast fisheries. Catches and prices from many fisheries exhibit high inter-annual variability leading to high variability in fishermen's income. Kasperski (AFSC) and Holland (NWFSC) recently examined more than 30,000 vessels fishing off the West Coast and Alaska over the last 30 years. This work shows that variability of annual revenue can be reduced by diversifying fishing activities across multiple fisheries or regions. Diversification can be measured with the Herfindahl-Hirschman Index (HHI), which ranges from a high 10,000 for vessel that derives all its income from a single fishery and declines toward zero as revenues are spread more evenly across more fisheries.

Levels of diversification for groupings of vessels vary greatly, and levels of diversification for these vessel groupings exhibit different trends over time. The current fleet of vessels fishing off the coasts of the U.S. West Coast and Alaska (those that fished in 2010) is less diverse than at any point in the past 30 years. The trends over time are due to both entry and exit of vessels and changes for individual vessels. Over time, less diversified vessels have been more likely to exit the fishery, which increases the average diversification level (decreases HHI). However, vessels that remain in the fishery have become less diversified, at least since the mid 1990s, and newer entrants have generally been less diversified than earlier entrants. The overall result is a moderate decline in average diversification (increase in HHI) since the mid 1990s or earlier for most vessels groupings. Notwithstanding these trends in average diversification, there are wide range of diversification levels and strategies within as well as across vessel classes and some vessels remain highly diversified.

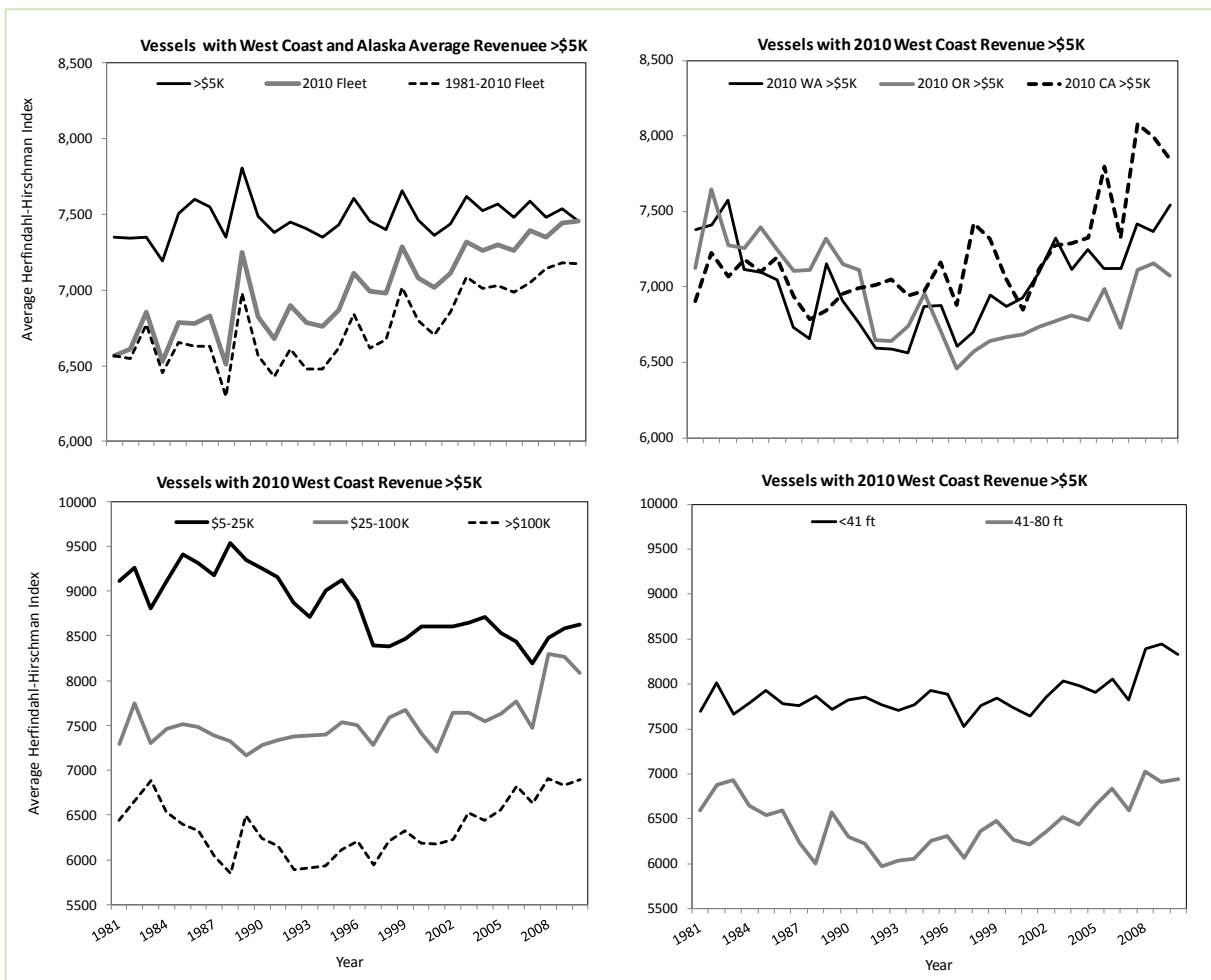


Figure HD. Trends in average diversification for US West Coast and Alaskan fishing vessels with over \$5k in average revenues (top left panel) and for vessels with 2010 West Coast revenue >\$5k (top right and bottom panels).

DETAILED REPORT

We evaluate a set of four indices and one indicator, designed to measure status and trends of fishing reliance and socioeconomic vulnerability within the coastal human communities of the CCLME. Through this effort, we identify currently available data useful in assessing vulnerability, discuss current efforts that will be useful for future analyses, and identify gaps in human dimensions research.

The indices described here have been developed for and applied to a separate vulnerability assessment process for the coastal communities of the Southeast and Northeast regions (Jepson and Colburn in press), building upon prior social indicators work in coastal and fisheries contexts (Cutter 1996, Cobb and Rixford 1998, Pollnac et al. 2006, Jepson and Jacob 2007, Cutter et al. 2008). These vulnerability indices and vulnerability analyses of coastal communities will be replicated for the human communities adjacent to and integrated with the CCLME. Similar assessments of fishing reliance and socioeconomic vulnerability are already underway in the Alaska region and, through the development of this work nation-wide, a standardized approach to coastal community vulnerability will be applied throughout the U.S. fisheries management regions of North America.

As with Colburn and Jepson's (Colburn and Jepson 2012) use of Walker et al. (Walker et al. 2004), we take the view that resilience refers to the adaptive capacity of a community faced with socioeconomic and ecological duress. The indicators here are oriented toward measures of community vulnerability, and a vulnerable status for any one community or set of communities could be offset by community-level resilience. Analytical measures of resilience typically require an examination of changes and responses over time and are often highlighted in the data by noticeable perturbations and disaster events. For example, faced with socioeconomic vulnerability evident in income and sociological health measures, communities of Columbia River gillnetters have nonetheless exhibited resilience in maintaining their livelihoods, even in the face of emergent environmental and policy challenges (Martin 2008).

Much of the socioeconomic data necessary for each of the evaluated indices is available nationally through the U.S. Census's American Community Survey (ACS). Fisheries data used in the indices are collected for fisheries management needs in each region, and regional fisheries information networks such as the Pacific Fisheries Information Network (PacFIN) and Alaska Fisheries Information Network (AKFIN) maintain similar data necessary for the fishing reliance and vulnerability indices and analyses. For this reason, the coastal community vulnerability analysis approach pioneered in the Southeast and Northeast regions of the U.S. (Colburn and Jepson 2012, Jacob et al. 2012, Jepson and Colburn in press) is appropriate in other regional contexts, including within the coastal communities of the U.S. West Coast (i.e. the coastal portion of the CCLME).

The Fishery Income Diversification Indicator is the final indicator discussed in this work and it presents something of a special case as compared to the first four indices presented. The Fishery Income Diversification Indicator is measured at the fishing vessel level, as compared to the other four indices of *community level* vulnerability. However, Fishery Income Diversification is nevertheless worthy of inclusion because this indicator provides information on the status and trends for this particular aspect of CCLME-dependent human activities, where the other included indices of vulnerability are still in the relatively early phases of data collection and analysis.

The first of the four indices discussed here, the Commercial Fishing Reliance Index, will initially be applied to the complete set of coastal communities designated by the U.S. Census as "places," and geographically located within the CCLME's coastal counties. Once the set of communities reliant on commercial fishing are determined through a confirmatory factor analysis applied to the data included with

the Commercial Fishing Reliance Index, we can apply a subsequent set of factor analyses with reference to the additional indices of socioeconomic vulnerability. In this way, we will have some measures of vulnerability status for a set of coastal communities integrated with the CCLME (Figure HD1).

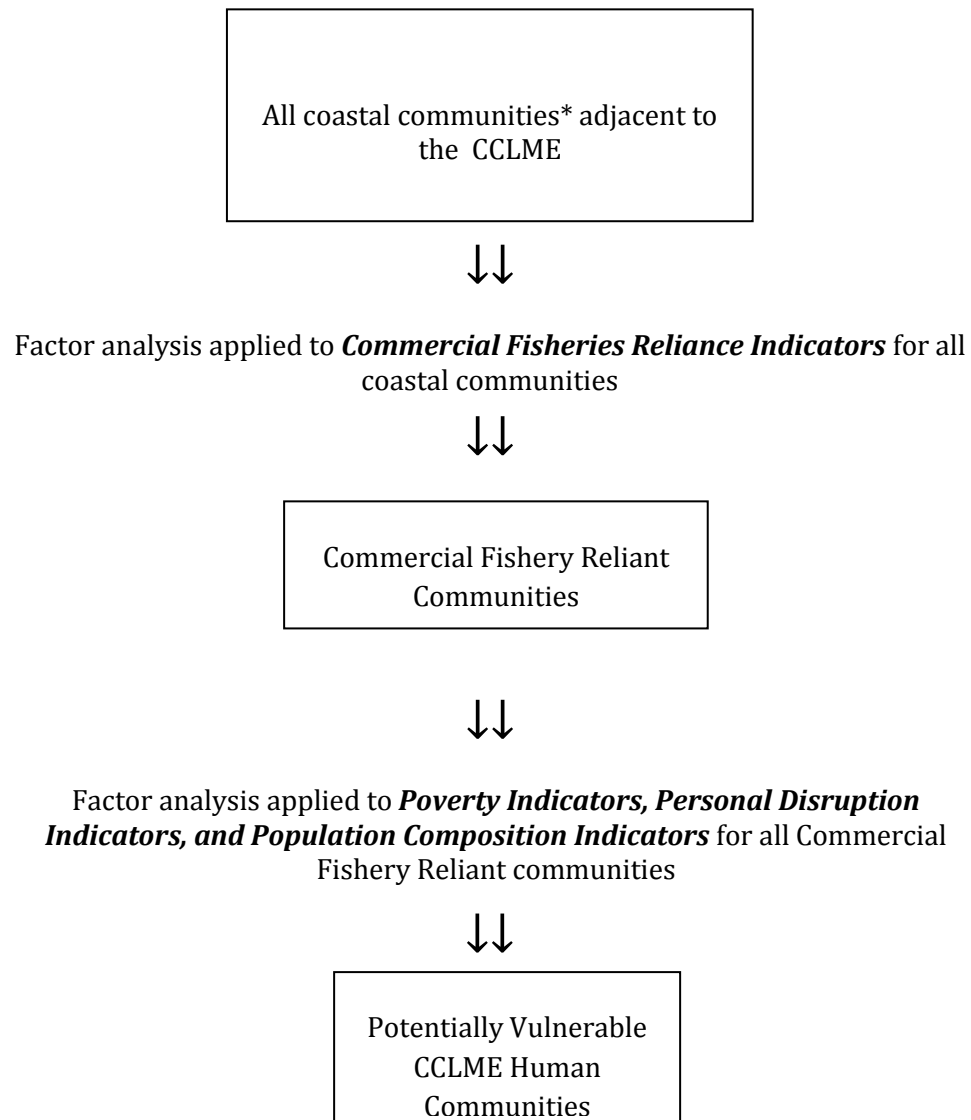


Figure HD1. Diagram of indices and factor analysis approach toward vulnerability for the human communities of the CCLME. *As determined by the U.S. Census (i.e. U.S. Census-Designated Places [CDPs] with coastal counties adjacent to the CCLME)

INDICATOR SELECTION PROCESS

INDICATOR EVALUATION

COMMERCIAL FISHING RELIANCE INDEX

The California Current Large Marine Ecosystem supports and is integral to a diverse set of human communities and human activities. There are 1912 census-designated places (CDPs) in Washington, Oregon and California, of which an as yet unidentified number may currently be defined as dependent on commercial fishing to meet their socioeconomic needs. For the purposes of the CCIEA, the most productive approach is to initially focus on those CDPs which are directly linked to the CCLME's marine species through commercial fishing, thereby identifying the communities of interest. While broader measures of ecosystem-oriented human well-being, ecosystem interaction and community resilience may eventually be developed, such measures are often difficult to identify at the community level, do not as often rest on consistently collected data and are sometimes challenged for their limited utility and applicability (Raudsepp-Hearne et al. 2010, Duraipah 2011).

The Commercial Fishing Reliance Index allows for the selection of communities most reliant on commercial fishing and therefore of particular interest to the CCIEA. The indicators included in the Commercial Fishing Reliance Index are primarily available as annually collected fisheries data maintained by the Pacific Fisheries Information Network (PacFIN), and employment data collected by the U.S. Census' American Community Survey (ACS). The indicators incorporated into the Commercial Fishing Reliance Index are the:

- Value of commercial fisheries landings per capita for each community
- Number of commercial fishing permits per capita for each community
- Processors with landings per capita for each community
- Percent employed in agriculture, fishing and forestry

We selected this index because it requires the integration of readily accessible data on a multi-year basis, and the index further points to communities that are expected to respond to changes in environmental conditions and potentially exhibit salience in concomitant vulnerability analyses. In the subsequent indices, we provide proposed indicators of condition for the selected commercial fishing-reliant communities (Kershner et al. 2011).

DEFINING 'COMMUNITY' AND COMMUNITIES OF INTEREST

Northwest Fisheries Science Center and Alaska Fisheries Science Center social scientists previously identified 123 Washington, Oregon and California coastal communities "dependent on" or substantially "engaged in" commercial fishing to meet their socioeconomic needs through a Data Envelopment Analysis (Sepez et al. 2007). The Commercial Fishing Reliance Index evaluated here was developed in part to establish standard social indicators for coastal communities nation-wide, and the application of this index to the communities of the CCLME will likely result in a set of fishing-dependent communities similar to those identified by the prior technical memorandum (Norman et al. 2007). The NWFSC technical memorandum identified 40 Washington communities, 31 Oregon communities, and 52 California communities that were

either substantially “dependent on” or “engaged in” commercial fisheries, including CCLME fisheries in particular, to meet their social and economic needs. Although the Data Envelopment Analysis (DEA) approach employed in the technical memorandum has been successfully employed in other fisheries and regions (Alsharif and Miller 2012), the advantage of the Commercial Fishing Reliance Index evaluated here is that it presents a novel organization and analysis of existing fisheries data that simplifies the DEA approach (Sepez et al. 2007) by reducing the component indicators of the community fishing dependence measure from 15 to just 4 (Jacob et al. 2010, Jepson and Colburn in press). Both the DEA approach to identifying communities of interest and the Commercial Fishing Reliance Index utilized here rest on a place-based definition of “community” that applies the U.S. Census’ Census-Designated Place (CDP) approach to community identification (Sepez et al. 2007; Colburn and Jepson 2012). Examining CCLME human communities as West Coast places designated by the U.S. Census allows for ready use of the extensive demographic and socioeconomically important information available through the American Community Survey (ACS).

The California Current is an important marine ecosystem for coastal communities engaged in commercial fisheries. In prior socioeconomic profiles of West Coast fishing communities in Washington, Oregon and California, community profile selection thresholds were based upon fishery landings, permits and landings value data inputs for seven CCLME fishery management groups of commercial interest (Norman et al. 2007), including:

- Crab
- Shrimp
- Groundfish
- Highly Migratory Species
- Coastal Pelagic Species
- Salmon
- Shellfish

The Commercial Fishing Reliance Index will similarly account for fishing indicators relative to each of the above fishery management and species group. Following the IEA indicator evaluation approach described with respect to the Puget Sound case study (Kershner et al. 2011), the selection of the Commercial Fishing Reliance Index rests on peer-reviewed literature or specific management mandates. Relevant indicator selection considerations for the Commercial Fishing Reliance Index include the following evaluation criteria:

- Theoretically-sound
 - Assessment of U.S. Gulf Coast communities (Jacob et al. 2010)
- Relevant to management concerns
 - Executive Order 12898
 - MSFCMA National Standard 8
- Concrete & Numerical
 - Factor loading on the indicators presents defensible results for other coastal communities (Jacob et al. 2010, Jepson and Colburn in press)
- Historical data or information available
 - ACS data available on 5-year, 3-year and annual cycle, depending on community size, beginning in 2000
 - Fisheries data available annually through PacFIN beginning in 1981
- Operationally simple

- (Jacob et al. 2010)
- Broad spatial coverage
 - (Jacob et al. 2010)
- Continuous time series
 - ACS data available on 5-year, 3-year and annual cycle, depending on community size
 - Fisheries data available annually at minimum through PacFIN
- Understood by the public & policymakers
 - Community vulnerability and viability has been a prominent concern for the public within fishing and coastal communities (Hall-Arber et al. 2001, Colburn and Jepson 2012)
 - Policy makers have incorporated fishing community viability into federal management statutes (MSFCMA National Standard 8)
- History of reporting
 - Fishing community dependence, reliance and engagement reported in multiple contexts (Sepez et al. 2005, Norman et al. 2007)
- Cost-effective
 - Based upon freely available ACS and PacFIN data
- Regionally/nationally/internationally compatible
 - Replicates approach for U.S. East Coast, Gulf Coast and Alaska Communities (Jacob et al. 2010, Jepson and Colburn in press)

POPULATION COMPOSITION INDEX

Once commercial fishing reliant-communities are identified by factor analyses applied to the Commercial Fishing Reliance Index, those communities which are likely to be most vulnerable to management changes, natural hazards and ecosystem pressures will be made apparent through factor analyses applied to indices of socioeconomic vulnerability. Since these indices are meant to very broadly measure community-level socioeconomic vulnerability, they then identify a set of communities vulnerable to a broad range of disruptions and pressures, both socioeconomic (e.g. fishery closures) and natural (e.g. large-scale coastal windstorms) in origin. The Population Composition Index, which quantitatively describes the social make-up of the human communities reliant on the fisheries of the CCLME, is the first such vulnerability index. The indices of socioeconomic vulnerability, including the Population Composition Index, rely on community-specific data pulled from Annual American Community Survey (ACS) datasets as maintained by the U.S. Census. The use of ACS data allows for the use of regularly updated data for each community identified by the Commercial Fishing Reliance Index. The Population Composition Index combines ACS data on race, gender and other demographics including:

- Percent of community identifying racially as “white alone”
- Percent of community with female single headed households
- Population age 0-5
- Percent that speak English less than well

For the Population Composition Index and subsequent indices developed from ACS and other sources, the U.S. Census’s ACS data provides a source of secondary data that is regularly updated, allowing for trend monitoring for each community and index. However, ACS data are collected and released based upon community population thresholds. Coastal communities with populations equal to 65,000 and above feature ACS estimates on an annual basis. Communities home to populations equal to 20,000 and above are updated every three years. Communities with fewer than 20,000 people feature data releases which are updated every

five years. In terms of consistent analyses and trend monitoring, these size-based data differences may prove problematic within the framework of the CCIEA.

For example, prior fishing community profiling work (Norman et al. 2007) identified coastal communities of varying population sizes in determining which U.S. West Coast communities were most “dependent” on, or “engaged” in commercial fisheries to meet their socioeconomic needs. Included among the set of communities that scored highly on DEA measures of fisheries dependence and engagement were large coastal communities like San Diego, California, with a population equal to 1,326,179, and Port Orford, Oregon, with a population of 1,133 (Sepez et al. 2007). Due to their large differences in population size, and the accompanying constraints connected to ACS data releases, these two probable communities of CCIEA interest could only be considered together within the socioeconomic vulnerability factor analyses every five years.

Following the indicator evaluation approach described with respect to the Puget Sound case study (Kershner et al. 2011), the selection of the Population Composition Index rests on peer-reviewed literature or specific management mandates. Relevant indicator selection considerations include the following evaluation criteria:

- Theoretically-sound
 - Assessment of U.S. East Coast communities (Jacob et al. 2012)
- Relevant to management concerns
 - Executive Order 12898
 - MSFCMA National Standard 8
- Concrete & Numerical
 - Factor loading on the indicators presents defensible results for other coastal communities (Jacob et al. 2012)
- Historical data or information available
 - ACS data available on 5-year, 3-year and annual cycle, depending on community size, beginning in 2000
- Operationally simple
 - (Jacob et al. 2012)
- Broad spatial coverage
 - (Jacob et al. 2012)
- Continuous time series
 - ACS data available on 5-year, 3-year and annual cycle, depending on community size
- Understood by the public & policymakers
 - Community vulnerability and viability has been a prominent concern for the public within fishing and coastal communities (Hall-Arber et al. 2001, Colburn and Jepson 2012)
 - Policy makers have incorporated fishing community viability into federal statutes (MSFCMA National Standard 8)
- History of reporting
 - General demographic profiles reported in fishing community reporting contexts (Sepez et al. 2005, Norman et al. 2007)
- Cost-effective
 - Based upon freely available ACS and PacFIN data
- Regionally/nationally/internationally compatible
 - Replicates approach for U.S. East Coast, Gulf Coast and Alaska Communities (Jacob et al. 2010, Jepson and Colburn in press)

POVERTY INDEX

In addition to the Population Composition Index, factor analyses on poverty indicators can offer assessments of socioeconomic vulnerability for coastal communities. A Poverty Index developed by fisheries social scientists in the Southeast and Northeast regions, following prior work on community vulnerability to natural hazards (Cutter 1996, Cutter et al. 2000, Jacob et al. 2012) provides a means of assessing relative well-being, vulnerability and resilience potential of fishing reliant communities.

The Poverty Index, employed in measuring socioeconomic vulnerability of coastal communities, includes indicators that account for the:

- Percent within the community receiving assistance
- Percent of families within the community living below the poverty level
- Percent of the community over 65 years old living in poverty
- Percent of the community under 18 years old living in poverty

For coastal communities, socioeconomic vulnerability indices were selected based upon ongoing, national research and the peer-reviewed support and management contexts for this approach is described below. Relevant indicator selection considerations include the following evaluation criteria with respect to the Poverty Index:

- Theoretically-sound
 - Assessment of U.S. East Coast communities (Jacob et al. 2012)
- Relevant to management concerns
 - Executive Order 12898
 - NOAA MSFCMA National Standard 8
- Concrete & Numerical
 - Factor loading on the indicators presents defensible results for other coastal communities (Jacob et al. 2012)
- Historical data or information available
 - ACS data available on 5-year, 3-year and annual cycle, depending on community size, beginning in 2000
- Operationally simple
 - (Jacob et al. 2012)
- Broad spatial coverage
 - (Jacob et al. 2012)
- Continuous time series
 - ACS data available on 5-year, 3-year and annual cycle, depending on community size
- Understood by the public & policymakers
 - Community vulnerability and viability has been a prominent concern for the public within fishing and coastal communities (Hall-Arber et al. 2001, Colburn and Jepson 2012)
 - Policy makers have incorporated fishing community viability into federal statutes (MSFCMA National Standard 8)
- Cost-effective
 - Based upon freely available ACS and PacFIN data
- Regionally/nationally/internationally compatible
 - Replicates approach for U.S. East Coast, Gulf Coast and Alaska Communities (Jacob et al. 2010, Jepson and Colburn in press)

PERSONAL DISRUPTIONS INDEX

As a companion to the Poverty Index, the Personal Disruptions Index developed by fisheries social scientists in the Southeast and Northeast regions, following prior work on community vulnerability (Cutter 1996, Jacob et al. 2012), provides a means of assessing relative well-being of commercial fishing reliant communities. Relatively frequent personal disruptions within the community are linked to increased overall vulnerability to natural hazards and other events and changes associated with livelihood and social impacts (Cutter et al. 2000, Jacob et al. 2012).

The Personal Disruptions Index, employed as a way of measuring socioeconomic vulnerability, includes indicators that account for:

- Percent within the community unemployed
- Uniform Crime Reporting (UCR) crime statistics index
- Percent of the community with no diploma
- Percent of the community living in poverty
- Percent of separated females in the community

For coastal communities, socioeconomic vulnerability indices were selected based upon ongoing, national research, and the peer-reviewed support and management contexts for this approach are described below. Relevant indicator selection considerations for the Personal Disruptions Index include the following evaluation criteria:

- Theoretically-sound
 - Assessment of U.S. East Coast communities (Jacob et al. 2012)
- Relevant to management concerns
 - Executive Order 12898
 - NOAA MSFCMA National Standard 8
- Concrete & Numerical
 - Factor loading on the indicators presents defensible results for other coastal communities (Jacob et al. 2012)
- Historical data or information available
 - ACS data available on 5-year, 3-year and annual cycle, depending on community size , beginning in 2000
 - UCR crime statistics index available annually beginning in 1930
- Operationally simple
 - (Jacob et al. 2012)
- Broad spatial coverage
 - (Jacob et al. 2012)
- Continuous time series
 - ACS data available on 5-year, 3-year and annual cycle, depending on community size
 - UCR crime statistics index available annually
- Understood by the public & policymakers
 - Community vulnerability and viability has been a prominent concern for the public within fishing and coastal communities (Hall-Arber et al. 2001, Colburn and Jepson 2012)
 - Policy makers have incorporated fishing community viability into federal statutes (MSFCMA National Standard 8)
- Cost-effective

- Based upon freely available ACS, PacFIN data and FBI Uniform Crime Reporting (UCR) data
- Regionally/nationally/internationally compatible
 - Replicates approach for U.S. East Coast, Gulf Coast and Alaska Communities (Jacob et al. 2010, Jepson and Colburn in press)

FISHERY INCOME DIVERSIFICATION INDICATOR

Catches and prices from many fisheries exhibit high inter-annual variability leading to variability in the income derived by fishery participants. The economic risk posed by this variability might be mitigated in some cases if individuals participate in several different fisheries; particularly if revenues from those fisheries are uncorrelated or vary asynchronously. Crop diversification is a common means of reducing risk in agriculture taking advantage of asynchronous variation in yields response and prices to minimize idiosyncratic risk (Heady 1952, Johnson 1967). Another common strategy in agriculture, particularly in semiarid regions with high fine scale variation in rainfall, is to farm a number of geographically separated plots to ensure some will be in areas with sufficient rainfall (Rosenzweig and Binswanger 1993). McCloskey argues that risk reduction was the motivation for English farmers for “scattering each man’s holdings in dozens of small strips” was, though inefficient, practiced as a risk reduction strategy (McCloskey 1976). A number of authors have argued that common property provides an important means risk reduction that may be undermined by privatization (Bromley and Chavas 1989, Thompson and Wilson 1994, Nugent and Sanchez 1998). This literature relates primarily to grazing lands held in common to protect against the potential spatial for variation in rainfall that would impact small private holdings but smooth risk for herders utilizing a much larger area held in common, but similar strategies apply to fishermen. While formal insurance programs do not exist, fishermen’s fishing strategies provide a means to reduce risk, in particular by diversifying their fishing activity across a variety of fisheries or areas (Oostenbrugge et al. 2002, Minnegal and Dwyer 2008). There is also a growing literature that suggests that fishermen should adopt portfolio approaches to their species composition to achieve the lowest variance in income for any level of expected return (Smith and McKelvey 1986, Baldursson and Magnusson 1997, Hilborn et al. 2001, Perusso et al. 2005, Sethi et al. 2012).

We measure diversification of West Coast and Alaskan fishing vessel’s gross revenues across species groups and regions each year. We utilize the Herfindahl-Hirschman Index (HHI) defined as:

$$H = \sum_{i=1}^{S_j} \sum_{j=1}^4 p_{ij}^2, \quad (1)$$

where p_{ij} represents percent (ranging from 0 to 100) of an individual’s total gross revenues derived from species group i in region j . We define p_{ij} to be the percent of a vessel’s total annual gross revenue from one of 40 different species groupings in one of four regions – the Bering Sea/Aleutian Islands, Gulf of Alaska, Alaskan in-state waters, and the WC (Table HD1). Not every species group is caught in each region, so there are a total of 84 total region-specific species groupings. HHI theoretically ranges from zero when revenues are spread amongst an infinite number of fisheries to 10,000 for a fishing operation that derives all revenue for a single fishery. Thus, the less diversified an individual’s revenue sources are, the higher the HHI. We evaluate how diversification has changed over time for various fleet groups. To explore how diversification of fishery income affects year-to-year variation and thus financial risk, we estimate the statistical relationship between HHI and the coefficient of variation (CV) of gross revenues for each vessel across years.

Socioeconomic vulnerability indicators were selected based upon ongoing, national research, and the peer-reviewed support and management contexts for this approach are described below. Relevant indicator selection considerations for the Fishery Income Diversification Indicator include:

- Theoretically-sound
 - (Perusso et al. 2005, Sethi et al. 2012)
- Relevant to management concerns
 - Relevant to fishery management plans and the groundfish trawl catch share program
- Responds predictably & is sufficiently sensitive to changes in a specific management action or pressure
 - Decreased diversification seems a likely result of recent management shifts
- Concrete & Numerical
 - Estimate based upon the statistical relationship between the Herfindahl-Hirschman Index (HHI) and the coefficient of variation (CV) of gross revenues for each vessel across years
- Historical data or information available
 - PacFIN data available beginning in 1981
- Operationally simple
- Broad spatial coverage
 - Applied on the vessel level across the CCLME
- Continuous time series
 - PacFIN data available annually at minimum
- Understood by the public & policymakers
 - Portfolio diversification typically understood as desirable in financial and agricultural contexts
- Cost-effective
 - Based upon freely available PacFIN fisheries data
- Regionally/nationally/internationally compatible
 - Similar analysis conducted relative to Alaskan fisheries, and is possible for other regions

Table HD1: Species groups used for diversification indices

| West Coast | Alaska |
|------------------------------------|--------------------------|
| Pacific Whiting | Pacific Cod |
| Dover Sole, Thornyheads, Sablefish | Flatfish |
| Rockfish and Flatfish | Rockfish |
| Skate, Dogfish, Sharks | Atka Mackerel |
| Pacific Halibut | Pollock |
| California Halibut, Croaker | Other Groundfish |
| Pink Shrimp | Sablefish |
| Other Prawns and Shrimp | Pacific Halibut |
| Crab | Herring |
| Salmon | Chinook Salmon |
| Tuna | Sockeye Salmon |
| Herring | Coho Salmon |
| Coastal Pelagics | Pink Salmon |
| Echinoderms | Chum Salmon |
| Other Shellfish | Other Salmon |
| Squid | Red King Crab |
| Other Species | Other King Crab |
| | Opilio Crab |
| | Other Snow Crab (Bairdi) |
| | Other Crab |
| | Scallops |
| | Other Shellfish |
| | Other Species |

STATUS AND TRENDS

MAJOR FINDINGS

Estimating trends in both the reliance on commercial fishing and the general vulnerability of coastal communities to ecosystem shifts and other stresses is challenging. Prior researchers have linked natural resource dependence, including fishing, to community vulnerability (Jacob et al. 2012). For this reason, we must first assess which west coast communities are most reliant on fishing, and then determine which communities among these are most vulnerable according to factor analyses of the concomitant indices of population composition, poverty and personal disruptions. Measures of fishery income diversification provide additional opportunities to assess vulnerability to risk.

However, as a single indicator, the Fishery Income Diversification indicator is somewhat problematic within the context of the CCIEA. Because fishermen can access the resources of multiple ecosystems, the Fishery Income Diversification Indicator requires the inclusion of North Pacific fisheries data in order for it to broadly account for income diversification. This capacity for multiple ecosystem access may render this indicator challenging in terms of its connection to the CCLME. The mobility of fishermen confounds at least one of the indicator evaluation criteria presented in the Puget Sound indicator evaluation approach (Kershner et al. 2011), that of predictability and sensitivity to changes in a specific ecosystem attributes.

SUMMARY AND STATUS OF TRENDS

COMMERCIAL FISHING RELIANCE INDEX, POPULATION COMPOSITION INDEX, POVERTY INDEX, PERSONAL DISRUPTIONS INDEX

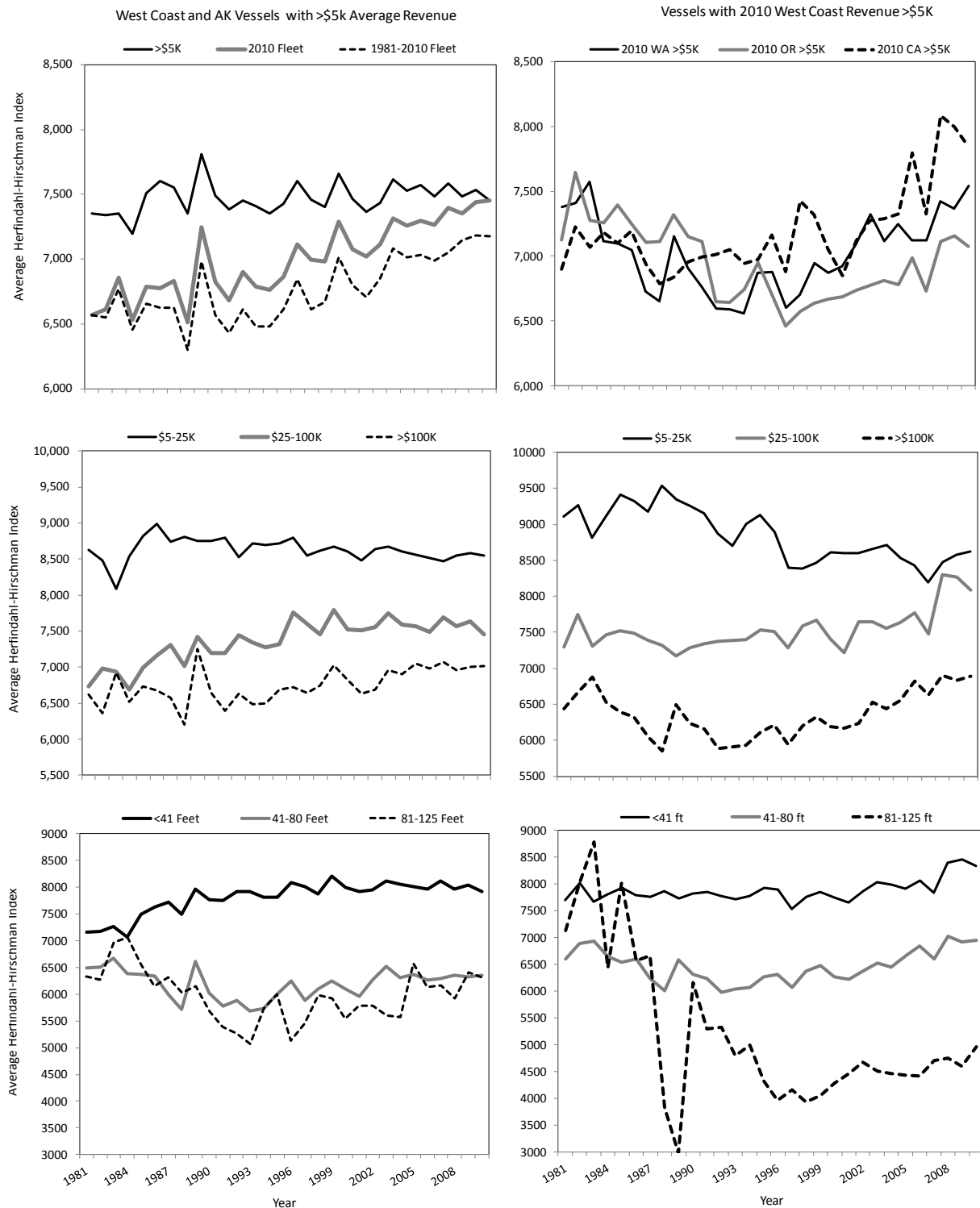
Until we can determine which communities are most reliant on commercial fisheries through a factor analysis applied to the Commercial Fishing Reliance Index, and data for each of the connected vulnerability indices is developed for those communities, complete information on coastal community status and trends will not be available for the CCLME.

FISHERY INCOME DIVERSIFICATION INDICATOR

We work with a large data set that includes annual landings and revenues between 1981 and 2010 by species, port and vessel from all commercial fisheries in the US EEZ off the West Coast and Alaska. We present analysis based on 30,757 vessels with average fishing revenues over \$5000 (adjusted to 2005 values) and at least two years of documented landings. The large data set enables us to identify trends in diversification and relationships between diversification and variation in revenues despite the relationship being very noisy. We also consider a number of subsets of the larger fleet categorized by average revenues, length and whether they had landings in West Coast states (i.e., excluding vessels with revenue only from Alaska).

Average fishery revenue diversification of West Coast and Alaskan fishing vessels is variable but shows distinct trends over time (Figure HD2). The HHI for most vessel groups, though erratic, has generally been increasing over time meaning that diversification of fishery income has been declining. The current fleet of vessels on the US West Coast and in Alaska (those that fished in 2010) is less diverse than at any point in the past 30 years. For smaller vessels diversification has generally been declining (HHI increasing) since 1981. For larger vessels, diversification increased through the early 1990s but has mostly declined since. The

causes of the decline in diversification are not completely clear and probably vary by fleet sector. One likely factor that correlates with the observed trend is the successive implementation and tightening of limited access programs and later individual quota programs. By the mid-1990s, entry into new fisheries was no longer possible for most vessels since nearly all fisheries had moratoriums on entry and many were beginning to reduce fleets through attrition, vessel buybacks or catch share programs. These programs limit fishermen's ability to move into new fisheries and often push out less active participants from a fishery. This is often necessary to limit catch and improve economic viability of the remaining participants, but it can also result in decreased diversification. Vessels that were in the fishery since 1981 have maintained a higher level of diversification than the overall fleet while vessels that entered later tend to be less diversified possible due to entry moratoriums in many fisheries. Diversification trends are somewhat more erratic for the current fleet of West Coast vessels -- i.e., vessels with at least \$5000 in revenues from landings in WA, OR or CA in 2010. For some vessels categorizations (e.g., larger vessels and each state's fleet overall) diversification tended to increase (HHI decline) through the mid-1990s and then trend upward thereafter.



While we can see some clear trends in diversification for various classes of vessels over time, there is wide variation in the degree of diversification across vessels within each class (Figure HD3). Higher earning and large vessels tend to be more diversified on average than smaller vessels and those with lower earnings. With the exception of the largest vessels, the current 2010 West Coast fleet appears to be less diversified on average than the larger fleet which includes all vessels from the West Coast and Alaska, and both current and former participants.

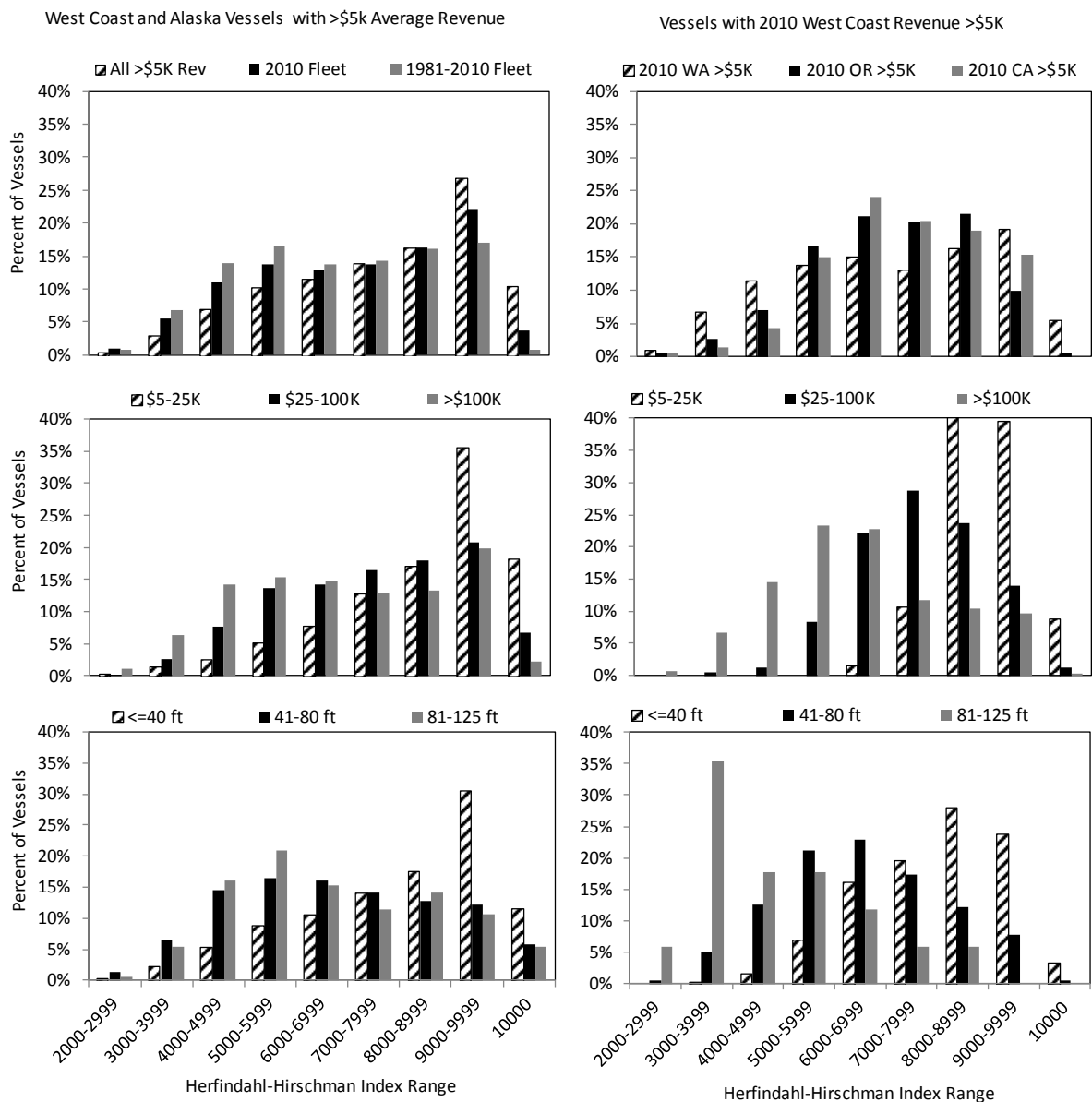


Figure HD3. Histograms showing percentage of vessels by ranges of Herfindahl-Hirschman index scores for US West Coast and Alaskan fishing vessels (left panels) and the 2010 West Coast Fleet (right panel) filtered by all vessels with over \$5k in average revenues (top panel), by average gross revenues classes (middle panel) and by vessel length classes (bottom panel).

If vessels are able to diversify into multiple fisheries whose revenues vary independently or asynchronously, they should experience a reduction in volatility of revenues and thus financial risk. This is confirmed for all of our fleet groupings by estimating quadratic regressions of the CV of gross fishery revenue as a function of HHI and HHI squared. Our analysis indicates a dome shaped relationship between variability of individuals' income and income diversification which implies that a small amount of diversification actually increases risk for some fleet categories, but moderate amounts of diversification can substantially reduce the variability of income that individuals receive from fishing. The decrease in CV with increased diversification varies substantial across vessel categories (Figure HD4 and Table HD2), but for nearly all vessel categories there is a substantial decrease in CV when moving from a low level of diversification (e.g. a 90-10 split in revenues between two fisheries) to a high level of diversification (e.g., and 50-25-25 split). Annual revenues for fishing vessels in our sample have an average coefficient of variations of 0.78. To illustrate how the decrease in CV associated with diversification affects the range of annual income a vessel might expect, we calculate the 50th percentile range of gross revenues for four hypothetical diversification schemes based on the functional relationship between HHI and CV for all vessels with mean annual revenues greater than \$5,000. The 50th percentile range of expected revenues contracts from a range of \$67,000 to \$244,000 with no diversification to a range of \$107,000 to \$207,000 with a 50-25-25 split of revenues across three fisheries.

Diversification across multiple fisheries can reduce variation and the associated financial risk. It can also increase the minimum annual revenue relative to average revenue, which should reduce the risk of a business failure. The ability of fishermen to diversify may be limited (or facilitated) by management approaches and regulatory actions. This should be a consideration when evaluating management actions, though in some cases management actions that reduce diversification are needed to remove excess capacity and promote efficiency.

There are a number of factors that may limit the feasibility or desirability of greater diversification. In many cases different fisheries require different gear that must be purchased and there are often costs of acquiring licenses and, increasingly, quota. It may also be the case that a vessel that can participate in several fisheries may be less efficient than more specialized vessels creating a trade-off between risk reduction through diversification and fishing efficiency. Exploration of this potential tradeoff would be an important extension of our research. Owners of multiple vessels can diversify by having individual vessels to specialize in different fisheries. Some fishermen may diversify their income with non-fishing sources. This seems particularly likely for vessels with low levels of revenue. We were unable to explore the degree or effects of this type of diversification due to a lack of data on non-fishing income. We hope to collect data on non-fishery income in future to explore this issue.

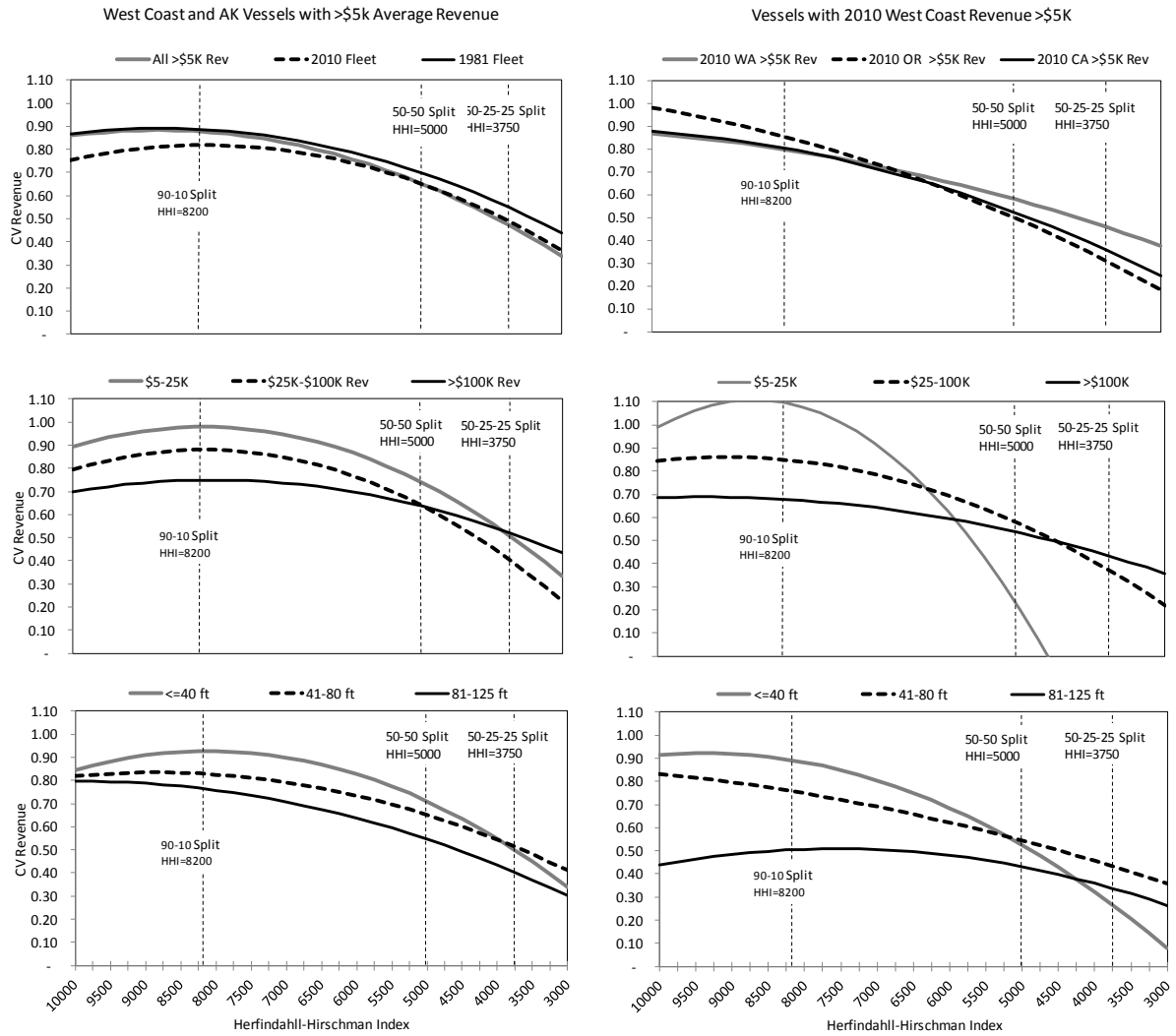


Figure HD4. Fitted relationships between the coefficient of variation (CV) of gross revenues for US West Coast and Alaskan fishing vessels (left panels) and the 2010 West Coast Fleet (right panel) filtered by all vessels with over \$5k in average revenues (top panel), by average gross revenues classes (middle panel) and by vessel length classes (bottom panel).

Table HD2. Predicted coefficient of variation (CV) of gross fishery revenue for Herfindahl-Hirschman index scores associated with alternative diversification schemes for groupings of WC and AK fishing vessels.

| Vessel Category | Predicted CV Herfindahl Index | | | | %Drop Single Fishery to 50-25-25 | Sample Size | Mean Revenue (\$1000) |
|-----------------------|-------------------------------|-------------|----------------|-------------------|--|----------------|-----------------------------|
| | Single Fishery | 90-10 Split | 50-50 Split | 50-25-25 Split | | | |
| All >\$5K Rev | 0.84 | 0.90 | 0.69 | 0.49 | 41% | 30,757 | \$ 155 |
| 2010 Fleet >\$5K | 0.75 | 0.82 | 0.65 | 0.49 | 35% | 8,288 | \$ 272 |
| 1981-2010 Fleet >\$5K | 0.87 | 0.89 | 0.70 | 0.55 | 37% | 3,880 | \$ 224 |
| \$5K-\$25K Rev | 0.89 | 0.98 | 0.74 | 0.51 | 43% | 13,088 | \$ 12 |
| \$25K-\$100K Rev | 0.78 | 0.90 | 0.68 | 0.46 | 42% | 10,081 | \$ 56 |
| >\$100K Rev | 0.68 | 0.77 | 0.65 | 0.52 | 24% | 7,588 | \$ 534 |
| <40Feet | 0.85 | 0.93 | 0.71 | 0.50 | 41% | 23,905 | \$ 49 |
| 40-80 Feet | 0.82 | 0.83 | 0.65 | 0.51 | 37% | 5,868 | \$ 201 |
| 80-125 Feet | 0.80 | 0.76 | 0.51 | 0.40 | 49% | 617 | \$ 993 |
| 2010 WA >\$5K | 0.87 | 0.80 | 0.58 | 0.46 | 47% | 404 | \$ 280 |
| 2010 OR >\$5K | 0.98 | 0.86 | 0.50 | 0.31 | 68% | 455 | \$ 194 |
| 2010 CA >\$5K | 0.88 | 0.81 | 0.52 | 0.36 | 59% | 460 | \$ 201 |
| 2010 WC \$5-25K | 0.99 | 1.09 | 0.20 | n.a. | n.a. | 162 | \$ 16 |
| 2010 WC \$25-100K | 0.85 | 0.85 | 0.57 | 0.37 | 56% | 452 | \$ 59 |
| 2010 WC >\$100K | 0.68 | 0.68 | 0.53 | 0.43 | 37% | 531 | \$ 380 |
| 2010 WC <40 Feet | 0.91 | 0.89 | 0.52 | 0.27 | 71% | 561 | \$ 90 |
| 2010 WC 41-80 Feet | 0.83 | 0.76 | 0.54 | 0.43 | 48% | 567 | \$ 283 |
| 2010 WC 81 -125 Feet | 0.44 | 0.50 | 0.43 | 0.34 | 23% | 17 | \$ 1,177 |

RISK

Until the community selections based upon the Commercial Fishing Reliance Index are complete, and the data extractions and analyses for the socioeconomic vulnerability indices are complete, we are not directly evaluating risk in this IEA. Fishery income diversification may provide some protection against risk and fisheries and income volatility, and the trends reflect decreasing diversification on the west coast, but the exposure and risks involved need further evaluation. In future, we will provide additional information on risks posed to fishermen and fishing communities according to their relative socioeconomic vulnerability measures.

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ECOLOGICAL INTEGRITY RISK ASSESSMENT

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TABLE OF CONTENTS (ENR)

| | |
|--|-----|
| Executive Summary | 505 |
| Detailed Report..... | 507 |
| General description and background..... | 507 |
| Data and methodology | 507 |
| Data-based risk assessment: activities, pressures, and habitats..... | 507 |
| Expert-based risk assessment | 520 |
| Comparison of Data- and expert-based risk assessments | 521 |
| Results..... | 521 |
| Data-based risk assessment | 521 |
| Expert-based risk assessment | 558 |
| Comparison of Data- and expert-based risk assessments | 567 |
| Conclusions – risk assessment..... | 570 |
| References cited..... | 571 |

LIST OF TABLE AND FIGURES (ENR)

| | |
|---|-----|
| Data-based assessment of risk to marine habitats in the Monterey Bay National Marine Sanctuary. BF = bottom-tended fishing, CE = coastal engineering, IP = inorganic pollution, NP = nutrient pollution, OP = organic pollution, SST = sea surface temperature, SD = sediment decreases, SI = sediment increases, SH = shipping. | 505 |
| Table EN.R.1. Activities and pressures evaluated as part of the risk assessment for ecosystem integrity in Monterey Bay National Marine Sanctuary..... | 508 |
| Table EN.R.2. The risk assessment for ecosystem integrity in Monterey Bay National Marine Sanctuary focused on the habitats listed below. Note that corals and sponges are biogenic features within hard bottom habitats, and that the nearshore/offshore designation denotes the location in which the habitat is predominantly found..... | 508 |
| Figure EN.R.1. Overview of habitats within the Monterey Bay National Marine Sanctuary on which the risk assessment focused. Data sources provided in Table EN.R.2..... | 509 |
| Figure EN.R.2. Conceptual flow for data-based habitat risk assessment. The exposure and sensitivity of each habitat to each activity or pressure was used to estimate the risk of a reduction in the quantity or quality of habitats to the point where their ecosystem functions were impaired. Figure credit: J. Samhour, G. Williams, J. Davies. | 510 |
| Figure EN.R.3. Exposure of hard and soft bottom habitats to bottom-tended fishing (trawling; BF), sediment increases (SI), and changes in sea surface temperature (SST) within the Monterey Bay National Marine Sanctuary. Categories of exposure are relative and are based on quantiles from intensity scores for MBNMS as originally presented in Halpern et al. (2009). | 512 |
| Figure EN.R.4. Exposure of coral and sponge habitats to bottom-tended fishing (trawling; BF), sediment increases (SI), and changes in sea surface temperature (SST) within the Monterey Bay National Marine Sanctuary. Categories of exposure are relative and are based on quantiles from intensity scores for MBNMS as originally presented in Halpern et al. (2009) | 513 |
| Figure EN.R.5. Exposure of beach, rocky intertidal, and kelp habitats to nutrient pollution (NP) and organic pollution (OP) within the Monterey Bay National Marine Sanctuary. Categories of exposure are relative and are based on quantiles from intensity scores for MBNMS as originally presented in Halpern et al. (2009)..... | 515 |
| Figure EN.R.6. Exposure of beach, rocky intertidal, and kelp habitats to sediment decreases (SD) and coastal engineering (CE) within the Monterey Bay National Marine Sanctuary. Categories of exposure are relative and are based on quantiles from intensity scores for MBNMS as originally presented in Halpern et al. (2009). A ccomodate | 516 |
| Table EN.R.3. Sensitivity criteria and scoring descriptions..... | 518 |
| Table EN.R.4. Data quality ratings and descriptions. | 520 |

| | |
|--|-----|
| Figure EN.R.7. Relative risk to (a) beaches, (b) corals, (c) hard bottom, (d) kelp forests, (e) offshore pelagic waters, (f) rocky intertidal, (g) seamounts, (h) soft bottom, and (i) sponges in the Monterey Bay National Marine Sanctuary due to 9 different pressures. BF = bottom-tended fishing, CE = coastal engineering, IP = inorganic pollution, NP = nutrient pollution, OP = organic pollution, SST = sea surface temperature, SD = sediment decreases, SI = sediment increases, SH = shipping | 523 |
| Figure EN.R.8. Average land- vs. sea-based risk scores for habitats in the Monterey Bay National Marine Sanctuary. Bars represent means \pm 1SE. * indicates $p < 0.05$ | 524 |
| Figure EN.R.9. Relative risk due to bottom-tended fishing in the Monterey Bay National Marine Sanctuary for the following habitats: B = beaches, C = corals, HB = hard bottom, KF = kelp forest, OP = offshore pelagic, RI = rocky intertidal, S = sponges, SB = soft bottom, SM = seamount..... | 525 |
| Figure EN.R.10. Average nearshore vs. offshore risk due to different activities and pressures in the Monterey Bay National Marine Sanctuary. Bars represent means \pm 1SE. * indicates $p \leq 0.05$ | 526 |
| Figure EN.R.11. Map highlighting locations where habitats within MBNMS experience relatively high exposure (scores of 3-4) from three activities and pressures. For beaches, kelp forests, and the rocky intertidal, this analysis focused on nutrient pollution (NP), organic pollution (OP), and sediment decreases (SD). For hard and soft bottom habitats, including locations known to have corals and sponges, this analysis focused on bottom-tended fishing (BF), sea surface temperature changes (SST), and sediment increases (SI). | 527 |
| Table EN.R.5. Relative exposure, sensitivity, and risk due to different activities and pressures for each habitat..... | 529 |
| Table EN.R.6. Relative exposure, sensitivity, and relative risk to each habitat from different activities and pressures..... | 531 |
| Table EN.R.7. Scores, rationale, and references for pressure-invariant sensitivity criteria. | 536 |
| Table EN.R.8. Scores, rationale, and references for the pressure-specific sensitivity criteria, change in area. | 539 |
| Table EN.R.9. Scores, rationale, and references for the pressure-specific sensitivity criteria, change in structure..... | 546 |
| Table EN.R.10. Scores, rationale, and references for the pressure-specific sensitivity criteria, frequency of natural disturbance..... | 552 |
| Figure EN.R.12. Expert-based assessment of the current status of habitats in the MBNMS. | 559 |
| Figure EN.R.13. Expert-based assessment of the relative intensity of different activities and pressures throughout the MBNMS..... | 560 |
| Figure EN.R.14. Expert-based assessment of risk to habitats within the MBNMS due to (a) bottom-tended fishing, and (b) coastal pollution. Data points represent average scores across respondents. B = beaches, C = corals, DS = deep sea, HB = hard bottom, KF = kelp forest, OP = offshore pelagic, RI = rocky intertidal, S = sponges, SB = soft bottom..... | 561 |

| | |
|---|-----|
| Figure EN.R.15. Expert-based assessment of exposure of habitats to bottom-tended fishing within the MBNMS, based on the spatial footprint of (top) and the temporal overlap with (bottom) bottom-tended fishing. | 562 |
| Figure EN.R.16. Expert-based assessment of sensitivity of habitats to bottom-tended fishing within the MBNMS, based on the expected degree of habitat loss (top) and the recovery rate (bottom) from bottom-tended fishing. | 563 |
| Figure EN.R.17. Expert-based assessment of exposure of habitats to coastal pollution within the MBNMS, based on the spatial footprint of (top) and the temporal overlap with (bottom) coastal pollution. | 564 |
| Figure EN.R.18. Expert-based assessment of sensitivity of habitats to coastal pollution within the MBNMS, based on the expected degree of habitat loss (top) and the recovery rate (bottom) from coastal pollution. | 565 |
| Figure EN.R.19. Expert self-assessment of uncertainty regarding responses related to risk to habitats within the MBNMS from bottom-tended fishing (top) and coastal pollution (bottom). | 566 |
| Figure EN.R.20. Discrepancies between data- and expert-based risk assessment for coastal pollution in MBNMS. (a) Exposure, (b) Sensitivity, (c) Risk. The line represents the 1:1 line, such that positive deviations indicate that expert-based assessment was greater than data-based assessment, and vice versa. | 568 |
| Figure EN.R.21. Discrepancies between data- and expert-based risk assessment for bottom-tended fishing in MBNMS. (a) Exposure, (b) Sensitivity, (c) Risk. The line represents the 1:1 line, such that positive deviations indicate that expert-based assessment was greater than data-based assessment, and vice versa. | 569 |
| Table EN.R.11. Activities and pressures posing the greatest relative risk to individual habitats within Monterey Bay National Marine Sanctuary (also see Fig. EN.R.7). Results come from the data-based assessment. | 570 |

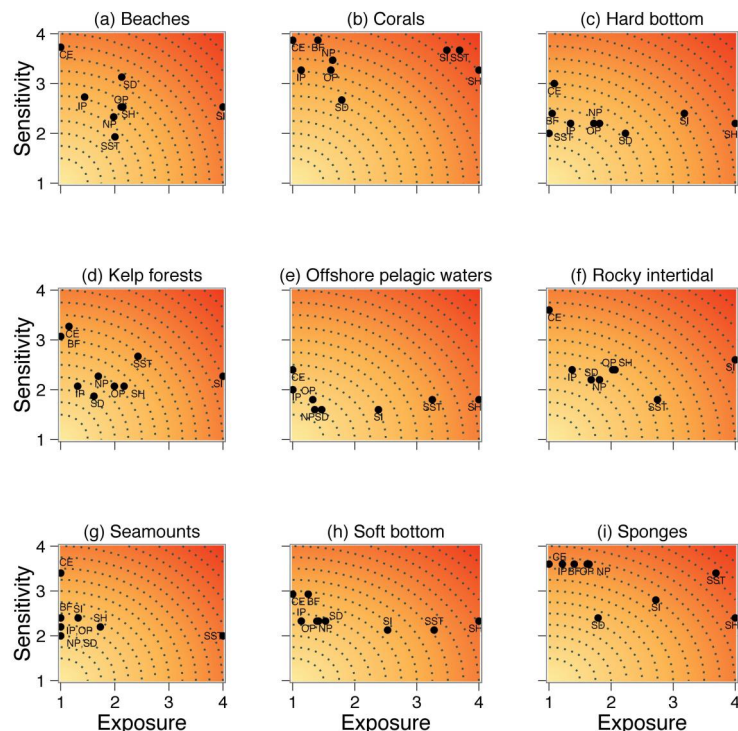
OVERVIEW

Coastal habitats-- including kelp forests, beaches, and rocky shorelines-- in the Monterey Bay National Marine Sanctuary were at highest risk due to human modifications, pollution, and climate.

EXECUTIVE SUMMARY

An ecosystem risk assessment can provide insight into the potential negative effects of drivers and pressures on ecosystem components. We assessed the environmental risks to marine habitats in a case study region, the Monterey Bay National Marine Sanctuary. A detailed look at coast-wide trends is also provided in the Human Pressures chapter of this report. This habitat risk assessment served as a proxy approach to understanding risk to ecological integrity, under the assumption that habitats act as umbrellas for communities of species and a variety of ecosystem processes. We focused the risk assessment on evaluating the potential for a reduction in the quantity or quality of habitats to the point where their ecosystem functions (e.g., water filtration, current or wave attenuation, nurseries) are impaired.

Using a data-based approach, we found that some habitats, like those containing corals and sponges, were at higher risk to many drivers and pressures, while others, like kelp forests and soft bottom habitats, experienced high risk due to a more limited subset. For each habitat, sea-based pressures, such as sea surface temperature changes and shipping, tended to exceed land-based pressures like coastal pollution. At the same time, individual drivers and pressures posed greater risk to nearshore habitats (beaches, kelp forests, rocky shores) than offshore habitats (soft bottom and offshore pelagic habitats). Comparison of these results with those from an expert-based survey showed general agreement, though there was a fair degree of uncertainty associated with survey responses. Furthermore, the expert-based risk assessment was generally less conservative than the data-based assessment in relation to pressures caused by bottom-tended fishing, but more conservative in relation to pressures resulting from coastal pollution.



Data-based assessment of risk to marine habitats in the Monterey Bay National Marine Sanctuary. BF = bottom-tended fishing, CE = coastal engineering, IP = inorganic pollution, NP = nutrient pollution, OP = organic pollution, SST = sea surface temperature, SD = sediment decreases, SI = sediment increases, SH = shipping.

We evaluated risk as a function of the exposure and sensitivity of each habitat to each activity or pressure in order to provide insight into potential mitigation measures. Habitats at high risk due to high

exposure (e.g., sediment increases in habitats containing sponges) lend themselves to management interventions focused on reducing exposure. In contrast, where habitats were at high risk due to high sensitivity (e.g., corals in habitats exposed to coastal pollution), managers might do better to focus on preventing increased exposure or preparing for habitat decline if exposure is already high. In the future, we hope to integrate our synthesis of the information available in the scientific and management literature with expert perceptions in order to generate a single, cohesive ecosystem risk assessment.

GENERAL DESCRIPTION AND BACKGROUND

Risk is defined as the likelihood that a subject will experience adverse consequences due to exposure to particular hazards (Burgman 2005). A risk assessment is an analytical approach for quantifying that likelihood and those consequences. In the context of the CCIEA, a risk assessment evaluates the degree to which pressures associated with human activities or natural processes interfere with the achievement of management objectives related to particular ecosystem components (Levin et al. 2009, Samhoury and Levin 2012). We define a pressure as a natural or human-induced element of a system that precipitates an unwanted outcome, like the decline in abundance of a population or a reduction in the quantity or quality of a habitat. Ecosystem components, defined in the Preface, are the biological, physical, or human dimension entities that policy makers, managers, or citizens are trying to manage or conserve. Unlike management scenario evaluations, risk assessment does not make projections about future states. Rather, it uses our best understanding of current linkages between pressures and states to evaluate risk to ecosystem components over a short time horizon (5-10 years). In that sense, this section represents a way of linking the chapter on Anthropogenic Drivers and Pressures to the status of the Ecological Integrity component.

The Ecological Integrity component refers to the structure and function of marine and coastal ecosystems and ecological communities. This risk assessment is thus one way of linking the chapter on Anthropogenic Drivers and Pressures to other CCIEA components. Assessing the risk of marine habitat decline is one proxy approach to understanding risk to ecosystem structure and function, because habitats serve as umbrellas for communities of species and a variety of ecosystem processes (Hayes and Landis, 2004, Tett et al. 2007, Halpern et al. 2009, Stelzenmüller et al. 2010). We focused on evaluating the potential for a reduction in the quantity or quality of habitats to the point where their ecosystem functions (e.g., water filtration, current or wave attenuation, nurseries) are impaired.

Here we demonstrate the utility of applying one specific risk assessment framework to marine habitats within the Monterey Bay National Marine Sanctuary (MBNMS). Methodologies for risk assessment are diverse and rapidly evolving. Our application provides a template for future risk assessments that would span all of the CCIEA components.

DATA AND METHODOLOGY

We conducted the risk assessment using two techniques. The first technique relied on data and literature that described associations between human activities, pressures, and habitats. The second technique was based on elicitation of expert opinion regarding the risk posed to habitats within MBNMS due to human activities and pressures. By evaluating risk using these two different approaches, we hoped to gain an understanding of how synthesis of information available in the scientific and management literature compares and complements expert perceptions.

DATA-BASED RISK ASSESSMENT: ACTIVITIES, PRESSURES, AND HABITATS

For the data-based risk assessment, we quantified the risk that three categories of pressures—modifications to the ocean bottom, pollution, and climate change—will lead to negative effects on nine habitat types within the MBNMS. This subset of pressures was selected based on an extensive dialogue with managers and scientists at the MBNMS, and represents regional concerns. Though we recognize the importance of historical pressures, our analysis focused on present-day pressures to which the habitats have been exposed within the past ten years. The specific pressures on which we focused are listed in Table EN.R.1, and the habitats are listed in Table EN.R.2 and displayed in Figure EN.R.1.

We purposefully did not assess risk to beach, rocky intertidal, and offshore pelagic habitats from bottom-tended fishing. We made this choice to avoid confusion, as modifications to the ocean bottom due to trawling are physically impossible (or nearly so) in these habitats. In contrast, other pressures have clear potential to generate risk to habitats (e.g., bottom-tended fishing in coral and sponge habitats, pollutants associated with ship traffic that may create risk for intertidal and pelagic habitats).

Table EN.R.1. Activities and pressures evaluated as part of the risk assessment for ecosystem integrity in Monterey Bay National Marine Sanctuary.

| Activity or pressure | Land- or sea-based | Data source |
|---|--------------------|--|
| <i>Modifications to the ocean bottom</i> | | |
| Bottom trawling | SB | California logbook trawl data, 2004-2009* |
| Increases and decreases in sediment loads | LB | SRTM60plus, PRISM, Syvitski et al. 2003, Halpern et al. 2009 |
| Coastal engineering | LB | NOAA ESI |
| <i>Pollution</i> | | |
| Organic pollution | LB | Halpern et al. 2008 |
| Inorganic pollution | LB | NGDC, EPA, Halpern et al. 2009 |
| Nutrient pollution | LB | USGS, NADP, Halpern et al. 2009 |
| Ship traffic | SB | CalTrans, WADOT, Halpern et al. 2009 |
| <i>Climate</i> | | |
| Sea surface temperature changes | SB | Halpern et al. 2008 |

*Includes vessels fishing for California halibut whether or not they have limited entry permits. Does not include microblocks with 1-2 vessels, or effort data from demersal seine and mid-water trawls. As presented at the PFMC meeting on 4 Nov 2010 (http://www.pcouncil.org/wp-content/uploads/HC_AGENDA_NOV2010BB.pdf). Credit: J. Mason, SWFSC

Table EN.R.2. The risk assessment for ecosystem integrity in Monterey Bay National Marine Sanctuary focused on the habitats listed below. Note that corals and sponges are biogenic features within hard bottom habitats, and that the nearshore/offshore designation denotes the location in which the habitat is predominantly found.

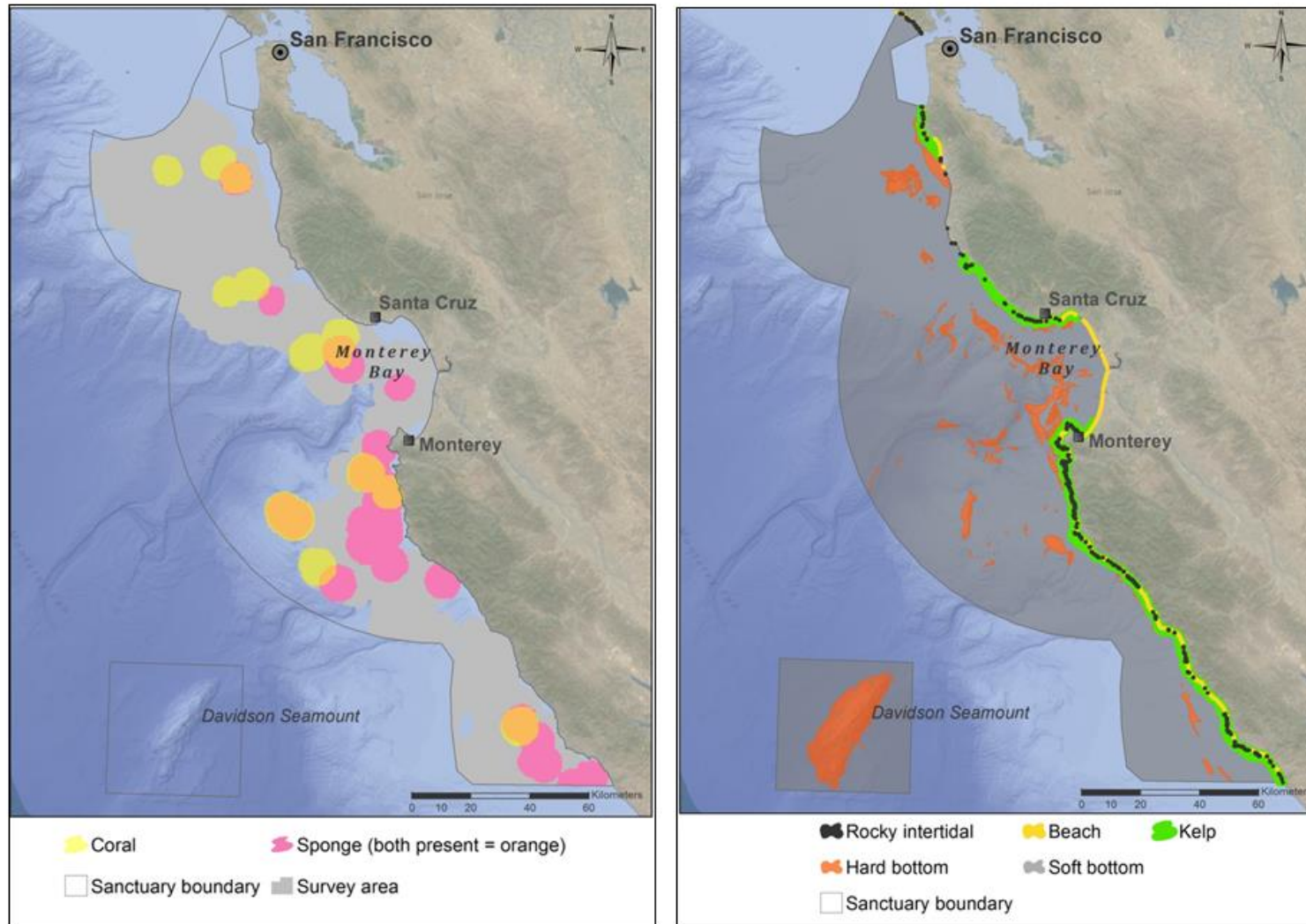
| Habitat | Nearshore or Offshore | Data source |
|---------------------------|-----------------------|---|
| Beaches | N | NOAA Environmental Sensitivity Index maps |
| Corals [§] | O | NWFSC West Coast Groundfish Bottom Trawl Survey* |
| Hard bottom | N | Moss Landing Marine Laboratory |
| Kelp forests [§] | N | California Department of Fish and Game |
| Offshore pelagic | O | all waters surrounding benthos >30 m depth [¶] |
| Rocky intertidal | N | NOAA Environmental Sensitivity Index maps |
| Seamount | O | National Centers for Coastal Ocean Science |
| Soft bottom | O | Moss Landing Marine Laboratory |
| Sponges [§] | O | NWFSC West Coast Groundfish Bottom Trawl Survey* |

*Credit: K. Whitmire

[§]Denotes living habitat.

[¶]S. DeBeukelaer, personal communication

Figure EN.R.1. Overview of habitats within the Monterey Bay National Marine Sanctuary on which the risk assessment focused. Data sources provided in Table EN.R.2.



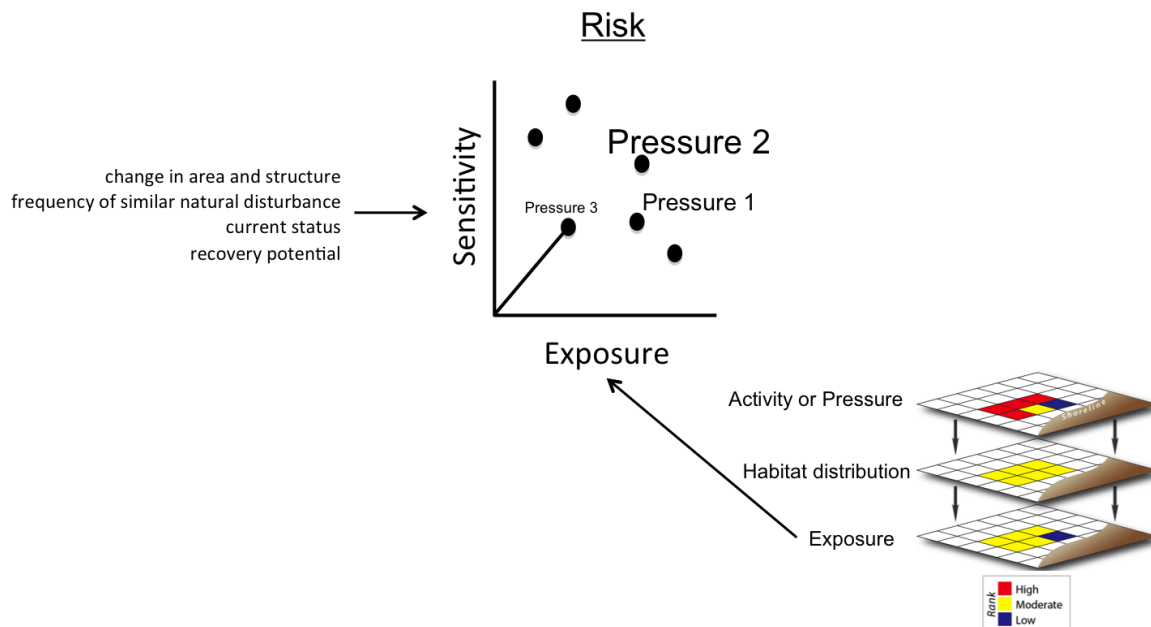
DATA-BASED RISK ASSESSMENT: ESTIMATING RISK

Our data-based risk assessment was based on the exposure E and the sensitivity S of each habitat to the activities and pressures listed in Table EN.R.1. The relative risk R_{ij} to habitat i from pressure j was calculated as:

$$R_{ij} = \sqrt{(E_{ij} - 1)^2 + (S_{ij} - 1)^2}, \quad (1)$$

implying that risk increases with Euclidean distance from the origin and each axis receives equivalent weight (see Fig. EN.R.2). We evaluated risk over the next 5 – 10 years, assuming that activities and pressures continue unchanged. Note that the assessment focused on the risk of decline of each habitat within the MBNMS, rather than the risk of decline throughout a broader geographic range. As mentioned above, we defined habitat decline as a reduction in the quantity or quality of habitats to the point where their ecosystem functions (e.g., water filtration, current or wave attenuation, nurseries) are impaired. More details about the mechanics of the framework are provided in Andrews et al. (2011) and Samhoury and Levin (2010). For a similar treatment, also see Tallis et al. (2011).

Figure EN.R.2. Conceptual flow for data-based habitat risk assessment. The exposure and sensitivity of each habitat to each activity or pressure was used to estimate the risk of a reduction in the quantity or quality of habitats to the point where their ecosystem functions were impaired. Figure credit: J. Samhoury, G. Williams, J. Davies.



EXPOSURE

We estimated exposure quantitatively and in a spatially explicit manner for all habitats. Specifically, we measured exposure as the overlap between the spatial distribution of each habitat and the intensity of each activity or pressure using GIS data. Intensity was scored as a continuous variable with values in the range 0-1; values were rescaled to the maximum on the original scale. Details about how intensity values

were generated are described more fully in Halpern et al. (2009). All activity/pressure data layers were converted from raster grid format to shape format. We used ESRI ArcGIS version 10 to obtain an exposure value by completing a union of each habitat data layer with each activity or pressure data layer. This procedure effectively weighted the activity intensity scores by the occurrence of each habitat within the MBNMS. For the final exposure score, we summed the area-weighted exposure intensity values for each habitat-activity/pressure combination. To evaluate relative risk to each habitat from the nine activities and pressures, we standardized the weighted sums across all activities and pressures within each habitat to values between 1 (minimal exposure) and 4 (maximal exposure). To characterize the habitat at greatest relative risk from each activity or pressure, we standardized the weighted sums across all habitats within each activity or pressure to values between 1 (minimal exposure) and 4 (maximal exposure). Figures EN.R.3-6 represent the unions of habitat and activity/pressure layers for several example combinations in nearshore and offshore regions of the MBNMS.

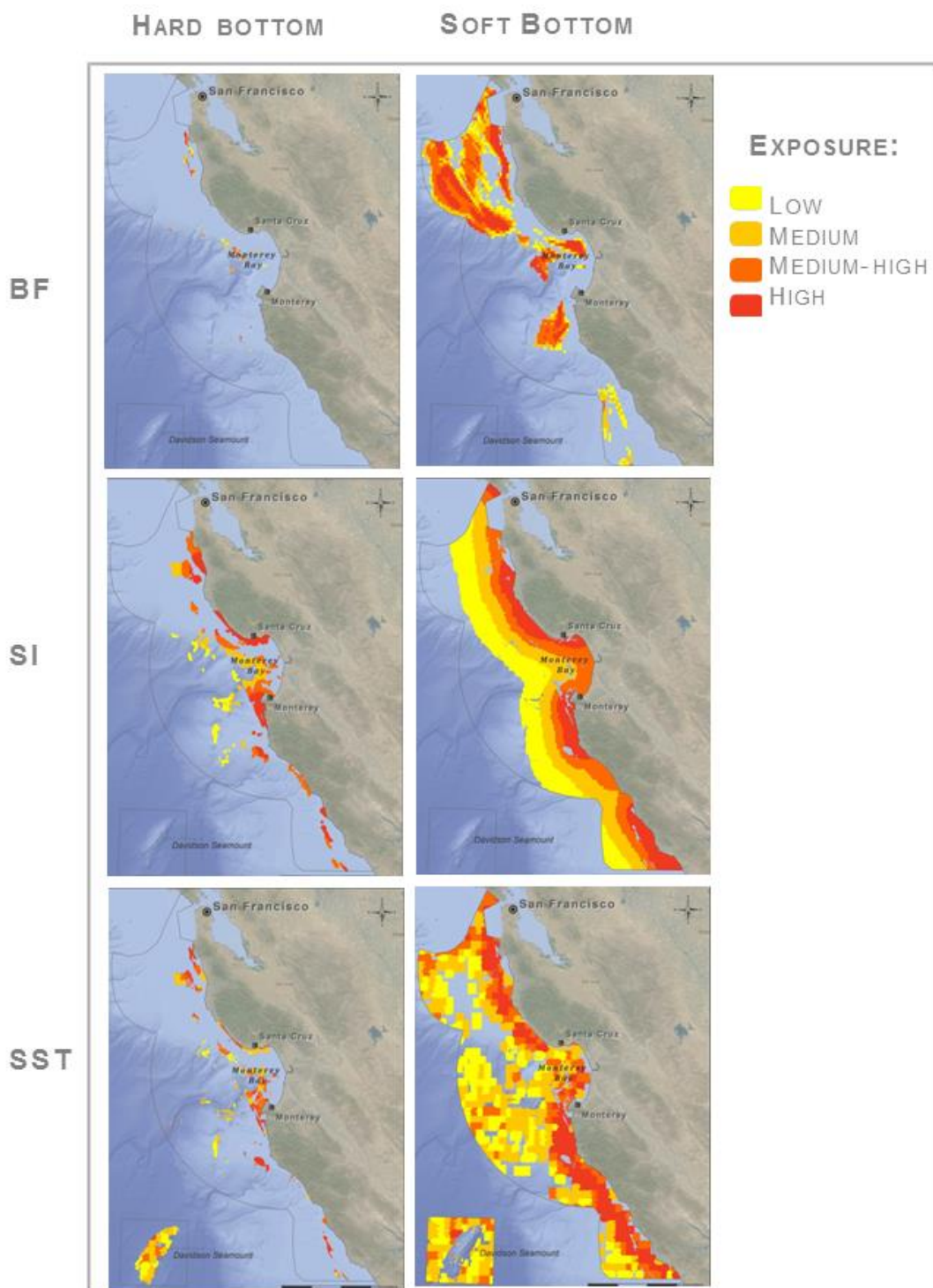


Figure EN.R.3. Exposure of hard and soft bottom habitats to bottom-tended fishing (trawling; BF), sediment increases (SI), and changes in sea surface temperature (SST) within the Monterey Bay National Marine Sanctuary. Categories of exposure are relative and are based on quantiles from intensity scores for MBNMS as originally presented in Halpern et al. (2009).

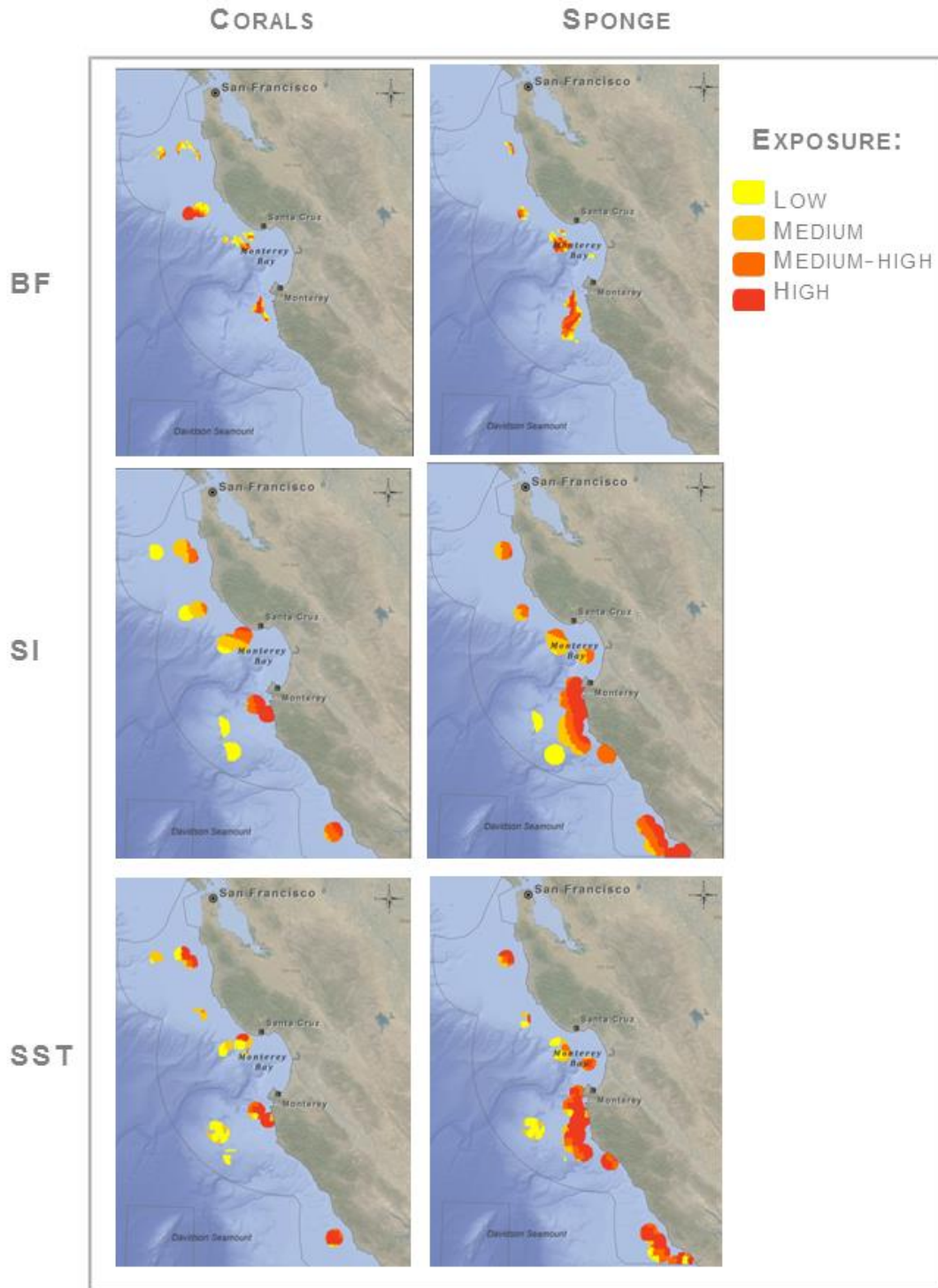


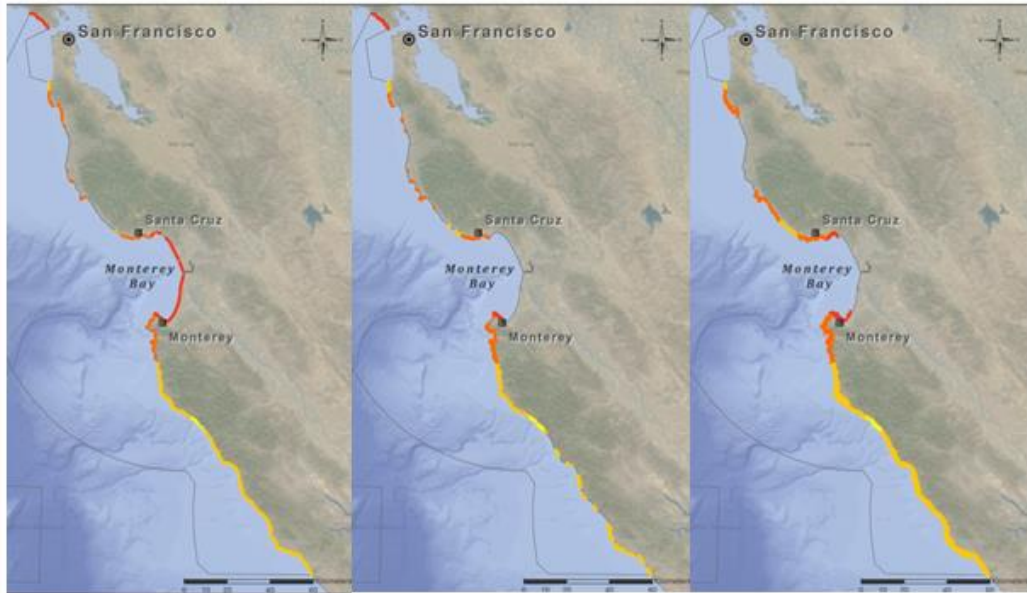
Figure EN.R.4. Exposure of coral and sponge habitats to bottom-tended fishing (trawling; BF), sediment increases (SI), and changes in sea surface temperature (SST) within the Monterey Bay National Marine Sanctuary. Categories of exposure are relative and are based on quantiles from intensity scores for MBNMS as originally presented in Halpern et al. (2009).

BEACH

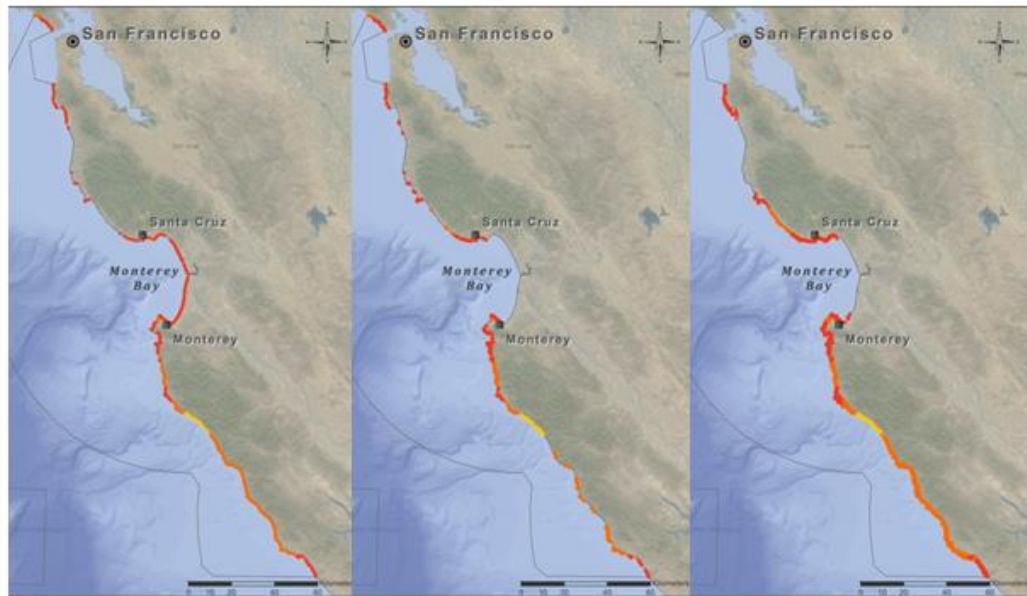
ROCKY INTERTIDAL

KELP

NP



OP



EXPOSURE:



Figure EN.R.5. Exposure of beach, rocky intertidal, and kelp habitats to nutrient pollution (NP) and organic pollution (OP) within the Monterey Bay National Marine Sanctuary. Categories of exposure are relative and are based on quantiles from intensity scores for MBNMS as originally presented in Halpern et al. (2009).

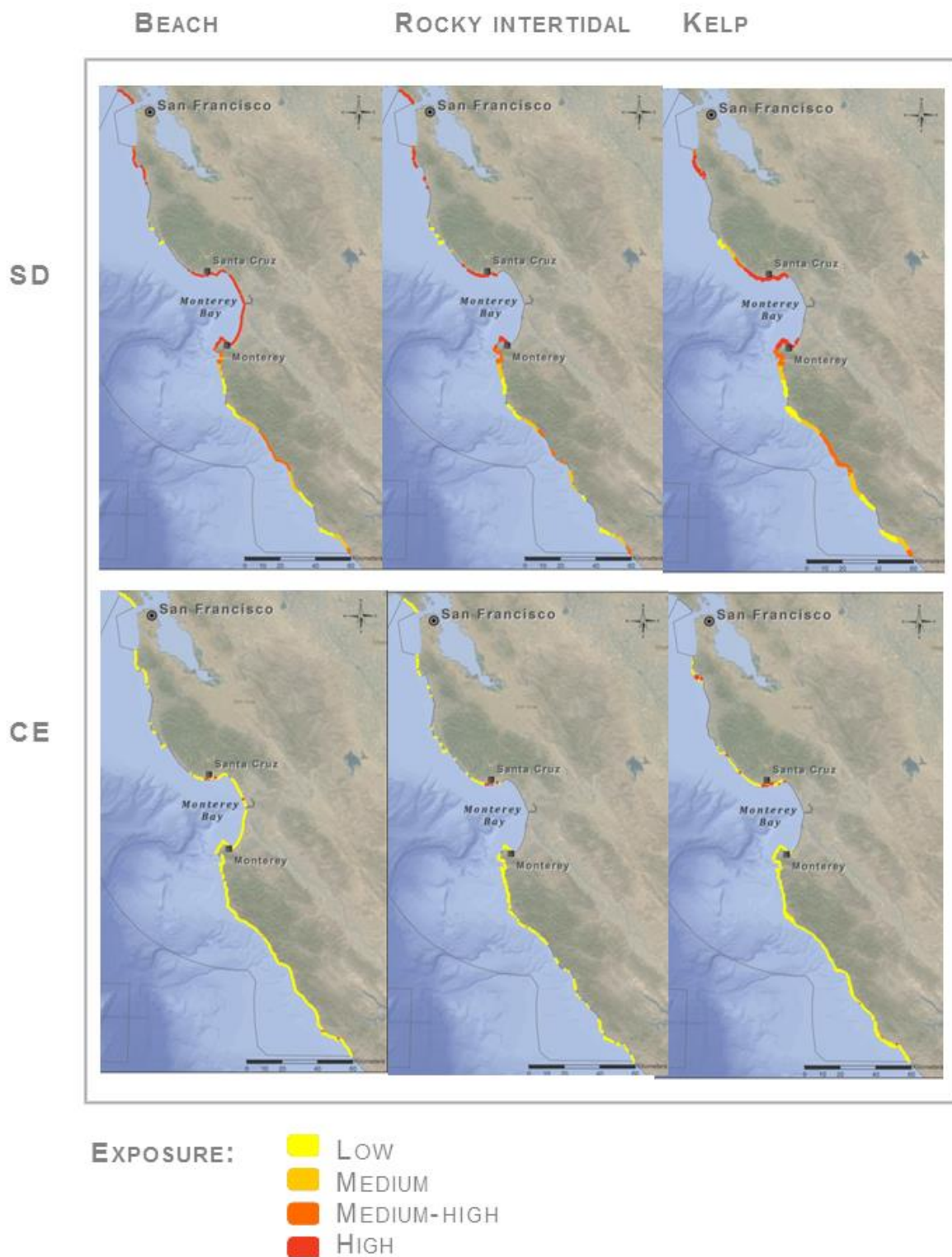


Figure EN.R.6. Exposure of beach, rocky intertidal, and kelp habitats to sediment decreases (SD) and coastal engineering (CE) within the Monterey Bay National Marine Sanctuary. Categories of exposure are relative and

are based on quantiles from intensity scores for MBNMS as originally presented in Halpern et al. (2009). A accommodation

SENSITIVITY

We estimated sensitivity qualitatively based on three activity/pressure-specific criteria, in addition to criteria that were invariant across activities and pressures (Table EN.R.3). Each habitat-activity/pressure combination was scored on a scale from 1 (low) to 4 (high) for each criterion; categories were based on Tallis et al. (2012). Scores for all criteria were assigned based on inferences from the primary and grey literatures about the expected responses of habitats if they were exposed to activities and pressures over the next 5 – 10 years. In all cases we attempted to provide ratings specific to the California Current; however, paucity of regional data did not always allow for that. We included a data quality rating (1: low, 4: high) for each criterion as a means of portraying uncertainty related to scoring assignments (Table EN.R.4). To calculate the final Sensitivity score, we first averaged the 3 sub-criteria related to the ability of a habitat to recover from perturbation (5a-c in Table EN.R.3), and then averaged this composite criterion with the other four criteria listed in Table EN.R.3 (change in area, change in structure, frequency of natural disturbance, and current status) for each habitat-activity/pressure combination.

Table EN.R.3. Sensitivity criteria and scoring descriptions.

| Sensitivity criteria | Description | 1 (low) | 2 | 3 | 4 (high) |
|--|--|---|---|---|---|
| 1. Change in area ^a | The percent change in areal extent of a habitat when exposed to a given pressure | 0 - 10% loss in area | 10 - 30% loss in area | 30 - 50% loss in area | >50% loss in area |
| 2. Change in structure ^a | For biotic habitats, the change in structure is the percentage change in structural density of the habitat when exposed to a given pressure. For abiotic habitats, the change in structure is the amount of structural damage sustained by the habitat when exposed to a given pressure. | 0 - 10% loss in structure | Low loss in structure (for biotic habitats, 10-30% loss in density, for abiotic habitats, little to no structural damage) | Moderate loss in structure (for biotic habitats, 30-50% loss in density, for abiotic habitats, partial structural damage) | High loss in structure (for biotic habitats, >50% loss in density, for abiotic habitats, total structural damage) |
| 3. Frequency of natural disturbance ^a | The frequency of natural disturbances of a similar type to the pressure; habitats subject to regular disturbance similar in kind to a pressure should be more resistant to it | Daily | Weekly to monthly | Monthly to annually | Annually or less often |
| 4. Current status | The regional status of the habitat; increasingly critical status signifies a decrease in the ability of the habitat to recover from the impacts of the pressure | No concern; negligible difference from historical | Low concern (eg, impact studies exist but do not reveal major problems); somewhat degraded compared to historical | Moderate concern (including threatened status); substantially degraded compared to historical | High concern (endangered); unrecognizable compared to historical status |
| 5a. Replenishment ^b | Includes natural recruitment rate, or the rate at which new propagules enter a population. For abiotic habitats, sensitivity is assumed to be high as replenishment only occurs on geological time scales. | Recruitment events more often than annually | Recruitment events annually | Recruitment events every 1-2 years | Recruitment events less frequently than every 2 years |

| | | | | | |
|---|---|---|--|---|--|
| 5b. Recovery time ^b | For biotic habitats, we refer to recovery time of the habitat as a whole (e.g., a mature kelp forest) rather than recovery time of individuals. For abiotic habitats, shorter recovery times for habitats such as mudflats decrease the sensitivity of exposure to human activities, whereas for habitats made of bedrock, recovery will occur on geological time scales. | Recovery time <1 year | Recovery time 1-10 years | Recovery time >10 years | Recovery time >100 years |
| 5c. Population connectivity ^b | Realized exchange with other populations based on spatial patchiness of distribution, degree of isolation, and potential dispersal capability; based on monitoring surveys, and population genetic or direct tracking estimates. For abiotic habitats, sensitivity is assumed to be high as connectivity is only relevant on geological time scales. | Regular movement/exch ange between the focal regional population and other populations; high dispersal distance (>100km) | Occasional movement/exchan ge between the focal regional population and other populations; moderate dispersal distance (10- 100km) | Low movement/exch ange between the focal regional population and other populations; low dispersal distance (1- 10km) | Lowest movement/exch ange between the focal regional population and other populations; low dispersal distance (<1km) |

^aIndicates criterion varies among activities and pressures; all other criterion are invariant across activities and pressures.

^bThese criteria were averaged to create a composite criterion representing the ability of a habitat to recover from perturbation.

Table EN.R.4. Data quality ratings and descriptions.

| Data Quality | Description | Example |
|--------------|---|--|
| 1 | Very limited data. Information based on expert opinion surveys or on general literature reviews from a wide range of habitats. | No empirical literature exists to justify scoring for a focal habitat in relation to a particular activity/pressure but reasonable inference can be made by the person conducting the risk assessment. |
| 2 | Limited data. Estimates with high variation and limited confidence, or based on studies of similar habitats or of the focal habitat in other regions. | Scoring based on a study of a similar habitat outside of the study region. |
| 3 | Adequate data. Information is based on limited spatial or temporal coverage, moderately strong or indirect statistical relationships, or for some other reason is deemed not sufficiently reliable to be designated as "best data." | Use of presence-absence data from ad hoc sampling efforts; use of relatively old information; etc. |
| 4 | Best data. Substantial information exists to support the score and is based on data collected for the habitat in the study region. | Data-rich assessment of habitat status, with reference to historical extent and current trends. |

SYNTHESIS

In addition to evaluating risk for each activity/pressure–habitat combination, we highlighted locations within the Monterey Bay National Marine Sanctuary where risk scores for particular habitats were uniformly high across multiple activities and pressures. We also tested for differences in risk due to land- versus sea-based activities and pressures across habitats and for differences in risk due to alternative activities and pressures in nearshore versus offshore habitats considered collectively.

For beaches, kelp forests, and the rocky intertidal, we mapped locations characterized by medium-high to high exposure (a score between 3-4) for each of the following: nutrient pollution, organic pollution, and sediment decreases. For hard and soft bottom habitats, including locations known to have corals and sponges, we mapped locations characterized by medium-high to high exposure for each of the following: bottom-tended fishing, sea surface temperature changes, and sediment increases.

For the land- vs. sea-based and nearshore vs. offshore risk comparisons, we conducted the analyses using generalized linear models in R, and corrected for multiple comparisons using the `glht` function in the `multcomp` package. .

EXPERT-BASED RISK ASSESSMENT

The expert-based risk assessment was conducted in collaboration with managers and scientists at MBNMS. To protect their privacy, survey respondents remain anonymous. As in the data-based risk assessment, the conceptual approach was to elicit expert perceptions of exposure and sensitivity of MBNMS habitats to a variety of activities and pressures. Exposure questions addressed the spatial footprint of activities and pressures within habitats in addition to the temporal overlap of activities and pressures with

habitats. Sensitivity questions addressed the degree of loss and rate of recovery of habitats if exposed to activities and pressures.

The survey focused predominantly on the risk posed by coastal pollution and bottom-tended fishing to habitats within the MBNMS. We did not include comprehensive questions about other activities and pressures in order to constrain the total amount of time required to complete the survey. A few questions focused on other activities and pressures including those addressed in the data-based risk assessment described above and: aquaculture, invasive species, marine debris, ocean acidification, and ocean-based pollution. The habitats included all of those listed in Table EN.R.2 except seamounts, in addition to the deep sea. Respondents were asked about their level of certainty regarding the survey questions. [The full survey can be found here](#). The survey includes the exact information respondents were given regarding definitions of habitat types, activities, and pressures.

Respondents were asked to provide categorical responses to the survey questions. In the analyses below, we have tried to represent these answers in two ways. First, we simply illustrate the number of respondents choosing each level of categorical response for each question. Second, we associated integer scores between 1 and 4 with each level of categorical response for each question, such that a score of 1 indicated least exposed or sensitive and a score of 4 indicated most exposed or sensitive. Using this second approach, overall risk was calculated according to Equation 1. We recognize that the arbitrary scaling we have chosen for these categorical responses has a direct influence on assessment of risk levels and that variation among experts in their responses can be, but has not been, incorporated directly in the estimation of risk (Kuhnert et al. 2010). These challenges will be confronted in future versions of the CCIEA.

COMPARISON OF DATA- AND EXPERT-BASED RISK ASSESSMENTS

We compared the data- and expert-based risk assessments by plotting exposure, sensitivity, and risk scores derived from each method against one another. Because experts appeared to interpret a survey question regarding the degree of habitat loss expected due to bottom-tended fishing and coastal pollution in terms of the living communities associated with each habitat (see responses in Figs. EN.R.16, 18 below), we eliminated this question from the comparison of the data- and expert-based risk assessments. All other questions in our survey were clearly focused on the physical habitats, so we have retained them in the comparison of results from the data- and expert-based assessments. Positive deviations from a 1:1 line in the figures associated with these comparisons indicated that the expert-based assessment was greater (more conservative) than the data-based assessment, and vice versa.

RESULTS

DATA-BASED RISK ASSESSMENT

Relative risk to each habitat

For each habitat within MBNMS, relative risk due to the different activities and pressures varied substantially (Fig. EN.R.7). Some habitats, like corals and sponges (Figs. EN.R.7a, i), tended to be at higher risk to multiple activities and pressures, while other habitats, like kelp forests and soft bottom habitats (Figs. EN.R.7d, h), experienced high risk due to some activities and pressures but not others. Habitats assessed with consistently high risk across activities and pressures often showed high sensitivity scores, whereas exposure scores spanned a wide range for habitats experiencing risk that varied widely in intensity across activities and pressures.

Risk due to land-based activities and pressures differed among habitats ($p = 0.04$ for glm including interaction between habitat and land/sea pressures), though there was a general tendency for risk due to sea-based activities and pressures to exceed that due to land-based activities and pressures, except in beach

and rocky intertidal habitats (Figure EN.R.8). However, the only statistically significant difference occurred in offshore pelagic habitats where sea-based risk surpassed land-based risk ($p = 0.01$ for term representing interaction between land/sea pressures and offshore pelagic habitat). Summary scores for exposure, sensitivity, and risk can be found in Table EN.R.5.

Relative risk from each activity and pressure

Relative risk due to each individual activity and pressure varied across the habitats we evaluated in the MBNMS (Fig. EN.R.9). For instance, risk scores tended to be consistently high across nearly all habitats for coastal engineering (Fig. EN.R.9b), sea surface temperature changes (Fig. EN.R.9f), and shipping (Fig. EN.R.9i), but more variable for bottom-tended fishing (Fig. EN.R.9a), organic pollution (Fig. EN.R.9e), and sediment decreases (Fig. EN.R.9g). For the higher risk activities and pressures, comparable risk scores were generated more by exposure in some cases (e.g., sea surface temperature changes; Fig EN.R.9f) and by sensitivity in others (e.g., coastal engineering; Fig EN.R.9b). Summary scores for exposure, sensitivity, and risk can be found in Table EN.R.6.

Differences in risk to nearshore and offshore habitats varied among pressures ($p = 0.01$ for glm including interaction between pressure and nearshore/offshore habitat), though there was a general tendency for risk in nearshore habitats to exceed risk in offshore habitats except in the case of bottom-tended fishing gear (Figure EN.R.10). However, risk in nearshore habitats was statistically significantly greater than in offshore habitats only for sediment increases, sediment decreases, and organic pollution ($p = 0.03$, $p=0.052$, and $p =0.053$, respectively, for interaction terms between habitat type and pressures).

Habitats were highly exposed to multiple activities and pressures in a restricted set of areas within MBNMS (Figure EN.R.11). Coastal habitats including beaches, kelp forests, and the rocky intertidal were characterized by medium-high to high exposure scores at the northern boundary of the Sanctuary and within Monterey Bay. Hard and soft bottom habitats, including those with sponges and corals, were characterized by medium-high to high exposure scores offshore from Half Moon Bay (north of Santa Cruz) and southwest of Carmel Bay (south of Monterey).

Sensitivity scores

Details about the sensitivity scores, rationale, and references can be found in Table EN.R.7 (for activity/pressure invariant criteria) and Tables EN.R.8-10 (for criteria scores are activity/pressure specific).

Figure EN.R.7. Relative risk to (a) beaches, (b) corals, (c) hard bottom, (d) kelp forests, (e) offshore pelagic waters, (f) rocky intertidal, (g) seamounts, (h) soft bottom, and (i) sponges in the Monterey Bay National Marine Sanctuary due to 9 different pressures. BF = bottom-tended fishing, CE = coastal engineering, IP = inorganic pollution, NP = nutrient pollution, OP = organic pollution, SST = sea surface temperature, SD = sediment decreases, SI = sediment increases, SH = shipping

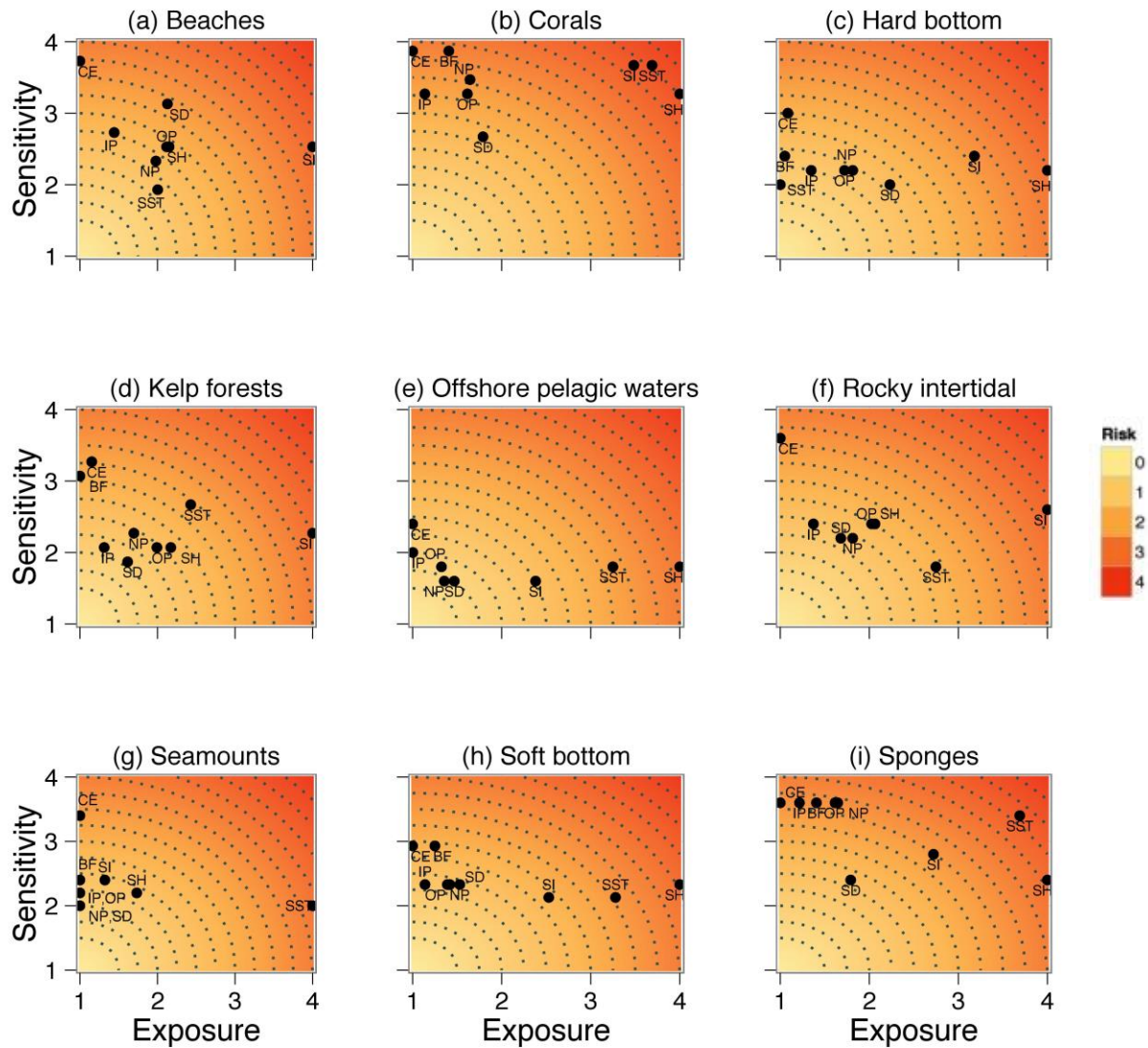


Figure EN.R.8. Average land- vs. sea-based risk scores for habitats in the Monterey Bay National Marine Sanctuary. Bars represent means \pm 1SE. * indicates $p < 0.05$.

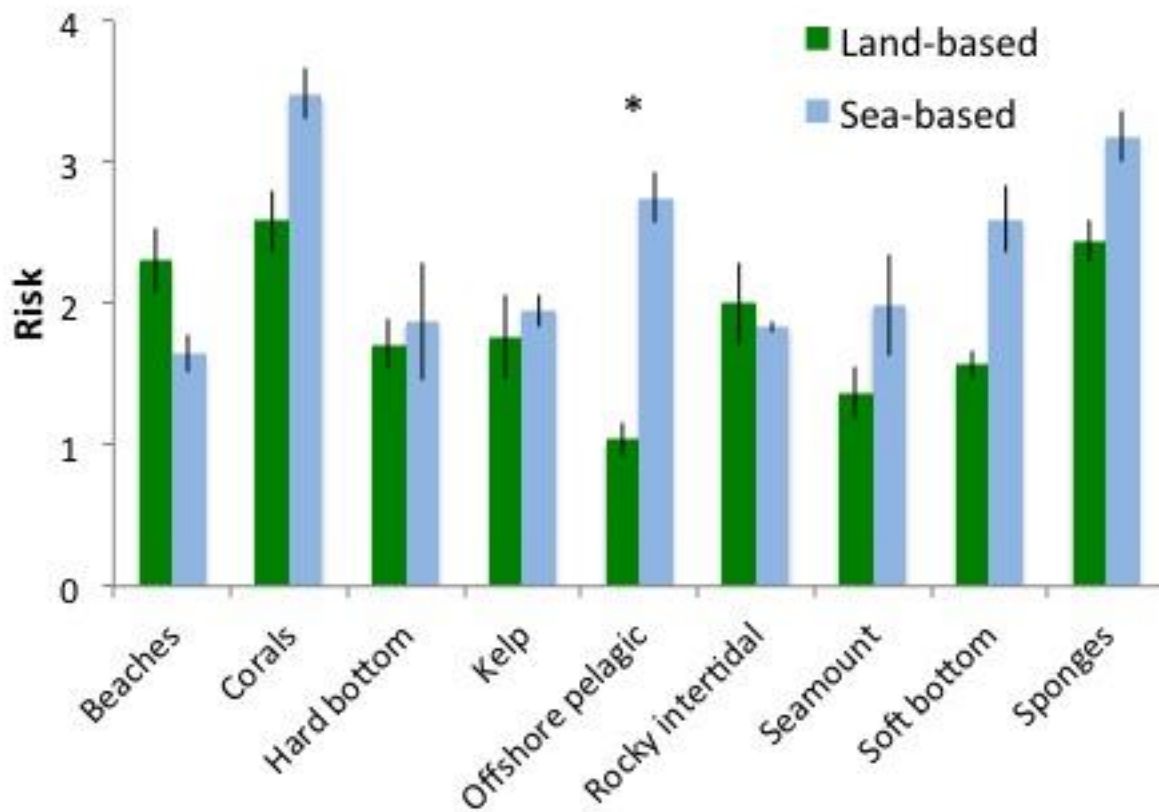


Figure EN.R.9. Relative risk due to bottom-tended fishing in the Monterey Bay National Marine Sanctuary for the following habitats: B = beaches, C = corals, HB = hard bottom, KF = kelp forest, OP = offshore pelagic, RI = rocky intertidal, S = sponges, SB = soft bottom, SM = seamount.

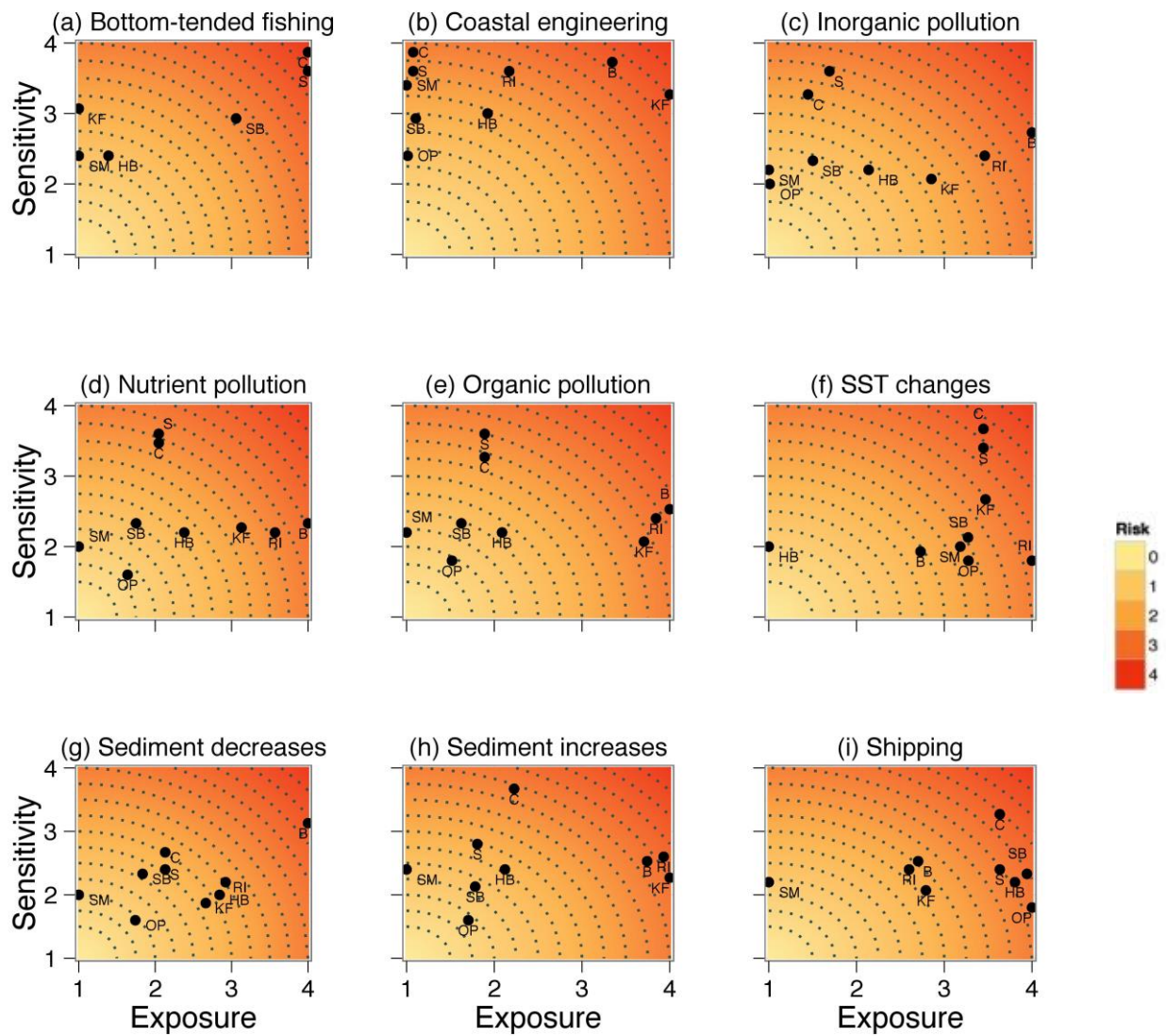
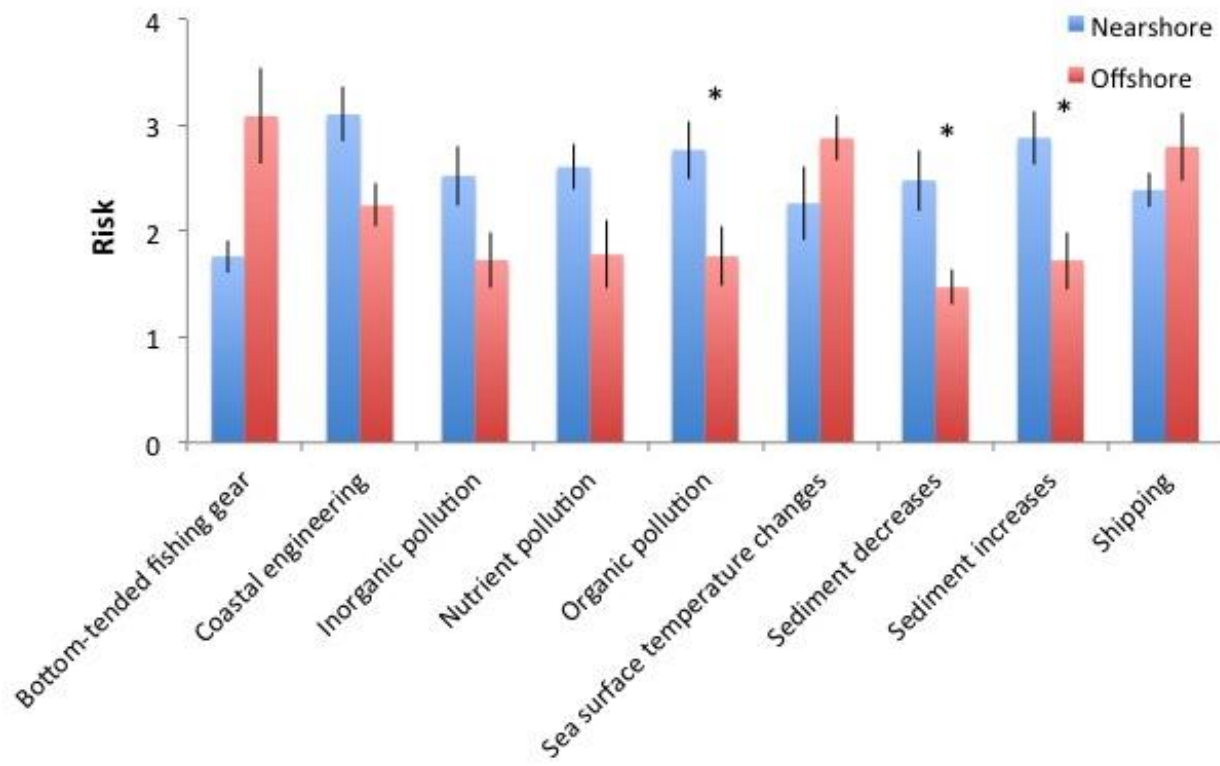


Figure EN.R.10. Average nearshore vs. offshore risk due to different activities and pressures in the Monterey Bay National Marine Sanctuary. Bars represent means \pm 1SE. * indicates $p \leq 0.05$.



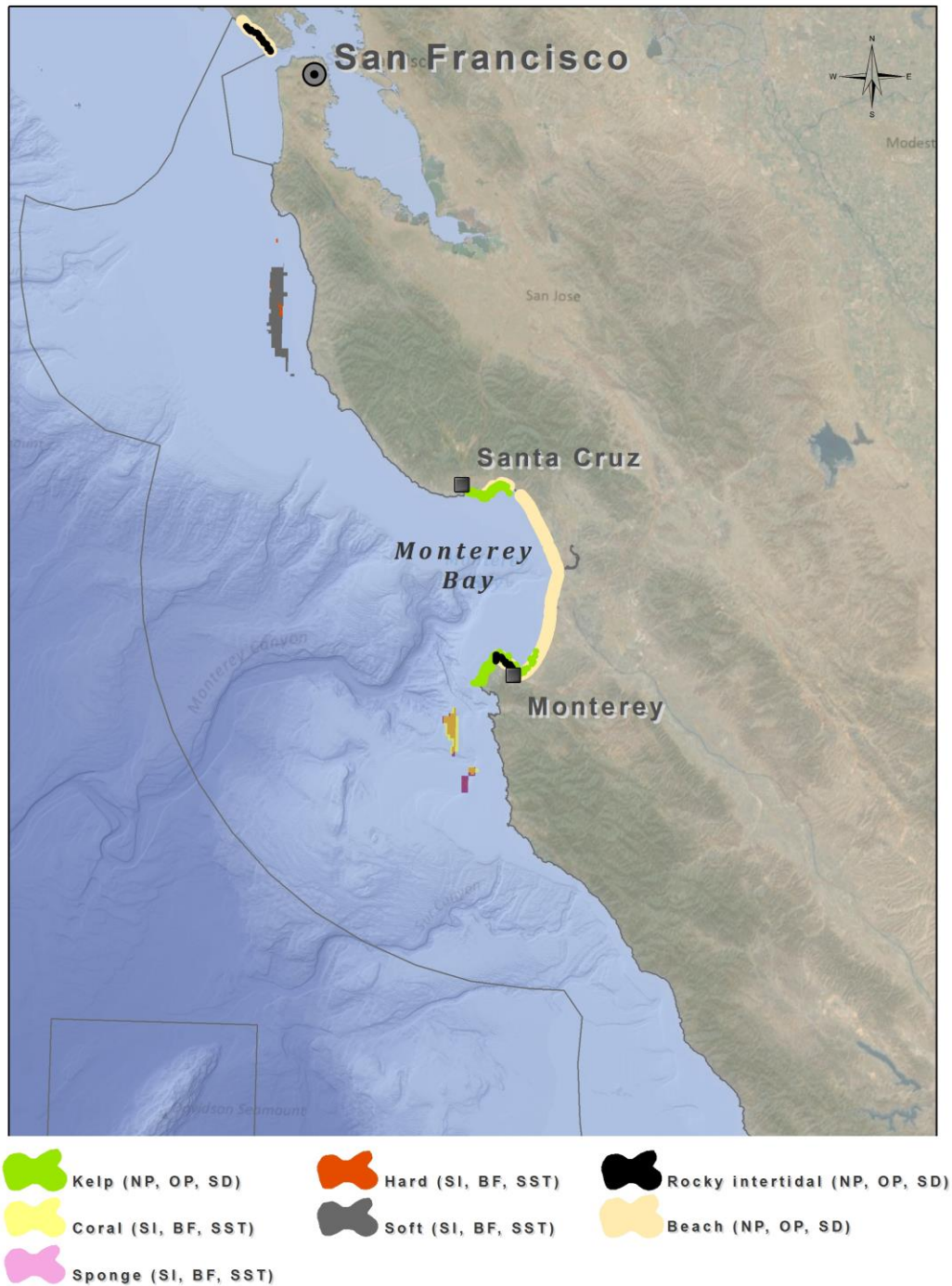


Figure EN.R.11. Map highlighting locations where habitats within MBNMS experience relatively high exposure (scores of 3-4) from three activities and pressures. For beaches, kelp forests, and the rocky intertidal, this analysis focused on nutrient pollution (NP), organic pollution (OP), and sediment decreases (SD). For hard and soft bottom habitats, including locations known to have corals and sponges, this analysis focused on bottom-tended fishing (BF), sea surface temperature changes (SST), and sediment increases (SI).

Table EN.R.5. Relative exposure, sensitivity, and risk due to different activities and pressures for each habitat.

| Habitat | Pressure | Land- or sea-based | Relative to other pressures for each habitat | | |
|------------------|---------------------------------|--------------------|--|-------------|------|
| | | | Exposure | Sensitivity | Risk |
| Beaches | Bottom-tended fishing gear | SB | n/a | n/a | n/a |
| Beaches | Coastal engineering | LB | 1.00 | 3.73 | 2.73 |
| Beaches | Inorganic pollution | LB | 1.44 | 2.73 | 1.79 |
| Beaches | Nutrient pollution | LB | 1.98 | 2.33 | 1.65 |
| Beaches | Organic pollution | LB | 2.12 | 2.53 | 1.90 |
| Beaches | Sea surface temperature changes | SB | 2.00 | 1.93 | 1.37 |
| Beaches | Sediment decreases | LB | 2.13 | 3.13 | 2.41 |
| Beaches | Sediment increases | LB | 4.00 | 2.53 | 3.37 |
| Beaches | Shipping | SB | 2.15 | 2.53 | 1.92 |
| Corals | Bottom-tended fishing gear | SB | 1.41 | 3.87 | 2.90 |
| Corals | Coastal engineering | LB | 1.00 | 3.87 | 2.87 |
| Corals | Inorganic pollution | LB | 1.14 | 3.27 | 2.27 |
| Corals | Nutrient pollution | LB | 1.64 | 3.47 | 2.55 |
| Corals | Organic pollution | LB | 1.62 | 3.27 | 2.35 |
| Corals | Sea surface temperature changes | SB | 3.69 | 3.67 | 3.79 |
| Corals | Sediment decreases | LB | 1.79 | 2.67 | 1.84 |
| Corals | Sediment increases | LB | 3.48 | 3.67 | 3.64 |
| Corals | Shipping | SB | 4.00 | 3.27 | 3.76 |
| Hard bottom | Bottom-tended fishing gear | SB | 1.05 | 2.40 | 1.40 |
| Hard bottom | Coastal engineering | LB | 1.08 | 3.00 | 2.00 |
| Hard bottom | Inorganic pollution | LB | 1.35 | 2.20 | 1.25 |
| Hard bottom | Nutrient pollution | LB | 1.81 | 2.20 | 1.45 |
| Hard bottom | Organic pollution | LB | 1.72 | 2.20 | 1.40 |
| Hard bottom | Sea surface temperature changes | SB | 1.00 | 2.00 | 1.00 |
| Hard bottom | Sediment decreases | LB | 2.23 | 2.00 | 1.59 |
| Hard bottom | Sediment increases | LB | 3.18 | 2.40 | 2.59 |
| Hard bottom | Shipping | SB | 4.00 | 2.20 | 3.23 |
| Kelp | Bottom-tended fishing gear | SB | 1.00 | 3.07 | 2.07 |
| Kelp | Coastal engineering | LB | 1.15 | 3.27 | 2.27 |
| Kelp | Inorganic pollution | LB | 1.31 | 2.07 | 1.11 |
| Kelp | Nutrient pollution | LB | 1.70 | 2.27 | 1.44 |
| Kelp | Organic pollution | LB | 1.99 | 2.07 | 1.46 |
| Kelp | Sea surface temperature changes | SB | 2.43 | 2.67 | 2.19 |
| Kelp | Sediment decreases | LB | 1.61 | 1.87 | 1.06 |
| Kelp | Sediment increases | LB | 4.00 | 2.27 | 3.26 |
| Kelp | Shipping | SB | 2.17 | 2.07 | 1.58 |
| Offshore pelagic | Bottom-tended fishing gear | SB | n/a | n/a | n/a |

| | | | | | |
|------------------|---------------------------------|----|------|------|------|
| Offshore pelagic | Coastal engineering | LB | 1.00 | 2.40 | 1.40 |
| Offshore pelagic | Inorganic pollution | LB | 1.00 | 2.00 | 1.00 |
| Offshore pelagic | Nutrient pollution | LB | 1.36 | 1.60 | 0.70 |
| Offshore pelagic | Organic pollution | LB | 1.32 | 1.80 | 0.86 |
| Offshore pelagic | Sea surface temperature changes | SB | 3.25 | 1.80 | 2.39 |
| Offshore pelagic | Sediment decreases | LB | 1.47 | 1.60 | 0.76 |
| Offshore pelagic | Sediment increases | LB | 2.38 | 1.60 | 1.51 |
| Offshore pelagic | Shipping | SB | 4.00 | 1.80 | 3.10 |
| Rocky intertidal | Bottom-tended fishing gear | SB | n/a | n/a | n/a |
| Rocky intertidal | Coastal engineering | LB | 1.00 | 3.60 | 2.60 |
| Rocky intertidal | Inorganic pollution | LB | 1.37 | 2.40 | 1.45 |
| Rocky intertidal | Nutrient pollution | LB | 1.81 | 2.20 | 1.45 |
| Rocky intertidal | Organic pollution | LB | 2.03 | 2.40 | 1.74 |
| Rocky intertidal | Sea surface temperature changes | SB | 2.75 | 1.80 | 1.92 |
| Rocky intertidal | Sediment decreases | LB | 1.68 | 2.20 | 1.38 |
| Rocky intertidal | Sediment increases | LB | 4.00 | 2.60 | 3.40 |
| Rocky intertidal | Shipping | SB | 2.06 | 2.40 | 1.76 |
| Seamount | Bottom-tended fishing gear | SB | 1.00 | 2.40 | 1.40 |
| Seamount | Coastal engineering | LB | 1.00 | 3.40 | 2.40 |
| Seamount | Inorganic pollution | LB | 1.00 | 2.20 | 1.20 |
| Seamount | Nutrient pollution | LB | 1.00 | 2.00 | 1.00 |
| Seamount | Organic pollution | LB | 1.00 | 2.20 | 1.20 |
| Seamount | Sea surface temperature changes | SB | 4.00 | 2.00 | 3.16 |
| Seamount | Sediment decreases | LB | 1.00 | 2.00 | 1.00 |
| Seamount | Sediment increases | LB | 1.32 | 2.40 | 1.44 |
| Seamount | Shipping | SB | 1.73 | 2.20 | 1.41 |
| Soft bottom | Bottom-tended fishing gear | SB | 1.25 | 2.93 | 1.95 |
| Soft bottom | Coastal engineering | LB | 1.00 | 2.93 | 1.93 |
| Soft bottom | Inorganic pollution | LB | 1.14 | 2.33 | 1.34 |
| Soft bottom | Nutrient pollution | LB | 1.42 | 2.33 | 1.40 |
| Soft bottom | Organic pollution | LB | 1.39 | 2.33 | 1.39 |
| Soft bottom | Sea surface temperature changes | SB | 3.28 | 2.13 | 2.55 |
| Soft bottom | Sediment decreases | LB | 1.53 | 2.33 | 1.43 |
| Soft bottom | Sediment increases | LB | 2.53 | 2.13 | 1.90 |
| Soft bottom | Shipping | SB | 4.00 | 2.33 | 3.28 |
| Sponges | Bottom-tended fishing gear | SB | 1.41 | 3.60 | 2.63 |
| Sponges | Coastal engineering | LB | 1.00 | 3.60 | 2.60 |
| Sponges | Inorganic pollution | LB | 1.22 | 3.60 | 2.61 |
| Sponges | Nutrient pollution | LB | 1.64 | 3.60 | 2.68 |
| Sponges | Organic pollution | LB | 1.62 | 3.60 | 2.67 |
| Sponges | Sea surface temperature changes | SB | 3.69 | 3.40 | 3.60 |
| Sponges | Sediment decreases | LB | 1.79 | 2.40 | 1.61 |
| Sponges | Sediment increases | LB | 2.72 | 2.80 | 2.49 |

| | | | | | |
|---------|----------|----|------|------|------|
| Sponges | Shipping | SB | 4.00 | 2.40 | 3.31 |
|---------|----------|----|------|------|------|

Table EN.R.6. Relative exposure, sensitivity, and relative risk to each habitat from different activities and pressures.

| | | | Relative to other habitats for each pressure | | |
|----------------------------|------------------|-----------------------|--|-------------|------|
| Pressure | Habitat | Nearshore or offshore | Exposure | Sensitivity | Risk |
| Bottom-tended fishing gear | Beaches | N | n/a | n/a | n/a |
| Bottom-tended fishing gear | Corals | O | 4.00 | 3.87 | 4.15 |
| Bottom-tended fishing gear | Hard bottom | N | 1.39 | 2.40 | 1.45 |
| Bottom-tended fishing gear | Kelp | N | 1.00 | 3.07 | 2.07 |
| Bottom-tended fishing gear | Offshore pelagic | O | n/a | n/a | n/a |
| Bottom-tended fishing gear | Rocky intertidal | N | n/a | n/a | n/a |
| Bottom-tended fishing gear | Seamount | O | 1.00 | 2.40 | 1.40 |
| Bottom-tended fishing gear | Soft bottom | O | 3.06 | 2.93 | 2.82 |
| Bottom-tended fishing gear | Sponges | O | 4.00 | 3.60 | 3.97 |
| Coastal engineering | Beaches | N | 3.35 | 3.73 | 3.60 |
| Coastal engineering | Corals | O | 1.08 | 3.87 | 2.87 |
| Coastal engineering | Hard bottom | N | 1.92 | 3.00 | 2.20 |
| Coastal engineering | Kelp | N | 4.00 | 3.27 | 3.76 |
| Coastal engineering | Offshore pelagic | O | 1.01 | 2.40 | 1.40 |
| Coastal engineering | Rocky intertidal | N | 2.17 | 3.60 | 2.85 |

| | | | | | |
|---------------------|------------------|---|------|------|------|
| Coastal engineering | Seamount | O | 1.00 | 3.40 | 2.40 |
| Coastal engineering | Soft bottom | O | 1.10 | 2.93 | 1.94 |
| Coastal engineering | Sponges | O | 1.08 | 3.60 | 2.60 |
| Inorganic pollution | Beaches | N | 4.00 | 2.73 | 3.46 |
| Inorganic pollution | Corals | O | 1.45 | 3.27 | 2.31 |
| Inorganic pollution | Hard bottom | N | 2.14 | 2.20 | 1.65 |
| Inorganic pollution | Kelp | N | 2.85 | 2.07 | 2.14 |
| Inorganic pollution | Offshore pelagic | O | 1.01 | 2.00 | 1.00 |
| Inorganic pollution | Rocky intertidal | N | 3.46 | 2.40 | 2.83 |
| Inorganic pollution | Seamount | O | 1.00 | 2.20 | 1.20 |
| Inorganic pollution | Soft bottom | O | 1.50 | 2.33 | 1.42 |
| Inorganic pollution | Sponges | O | 1.69 | 3.60 | 2.69 |
| Nutrient pollution | Beaches | N | 4.00 | 2.33 | 3.28 |
| Nutrient pollution | Corals | O | 2.05 | 3.47 | 2.68 |
| Nutrient pollution | Hard bottom | N | 2.38 | 2.20 | 1.83 |
| Nutrient pollution | Kelp | N | 3.13 | 2.27 | 2.48 |
| Nutrient pollution | Offshore pelagic | O | 1.64 | 1.60 | 0.88 |
| Nutrient pollution | Rocky intertidal | N | 3.57 | 2.20 | 2.84 |
| Nutrient pollution | Seamount | O | 1.00 | 2.00 | 1.00 |
| Nutrient pollution | Soft bottom | O | 1.75 | 2.33 | 1.53 |

| | | | | | |
|---------------------------------|------------------|---|------|------|------|
| Nutrient pollution | Sponges | O | 2.05 | 3.60 | 2.80 |
| Organic pollution | Beaches | N | 4.00 | 2.53 | 3.37 |
| Organic pollution | Corals | O | 1.89 | 3.27 | 2.44 |
| Organic pollution | Hard bottom | N | 2.09 | 2.20 | 1.62 |
| Organic pollution | Kelp | N | 3.71 | 2.07 | 2.91 |
| Organic pollution | Offshore pelagic | O | 1.52 | 1.80 | 0.95 |
| Organic pollution | Rocky intertidal | N | 3.84 | 2.40 | 3.17 |
| Organic pollution | Seamount | O | 1.00 | 2.20 | 1.20 |
| Organic pollution | Soft bottom | O | 1.62 | 2.33 | 1.47 |
| Organic pollution | Sponges | O | 1.89 | 3.60 | 2.75 |
| Sea surface temperature changes | Beaches | N | 2.73 | 1.93 | 1.97 |
| Sea surface temperature changes | Corals | O | 3.45 | 3.67 | 3.62 |
| Sea surface temperature changes | Hard bottom | N | 1.00 | 2.00 | 1.00 |
| Sea surface temperature changes | Kelp | N | 3.47 | 2.67 | 2.98 |
| Sea surface temperature changes | Offshore pelagic | O | 3.28 | 1.80 | 2.41 |
| Sea surface temperature changes | Rocky intertidal | N | 4.00 | 1.80 | 3.10 |
| Sea surface temperature changes | Seamount | O | 3.18 | 2.00 | 2.40 |
| Sea surface temperature changes | Soft bottom | O | 3.27 | 2.13 | 2.54 |
| Sea surface temperature changes | Sponges | O | 3.45 | 3.40 | 3.43 |
| Sediment decreases | Beaches | N | 4.00 | 3.13 | 3.68 |

| | | | | | |
|--------------------|------------------|---|------|------|------|
| Sediment decreases | Corals | O | 2.13 | 2.67 | 2.01 |
| Sediment decreases | Hard bottom | N | 2.84 | 2.00 | 2.10 |
| Sediment decreases | Kelp | N | 2.66 | 1.87 | 1.87 |
| Sediment decreases | Offshore pelagic | O | 1.74 | 1.60 | 0.95 |
| Sediment decreases | Rocky intertidal | N | 2.92 | 2.20 | 2.27 |
| Sediment decreases | Seamount | O | 1.00 | 2.00 | 1.00 |
| Sediment decreases | Soft bottom | O | 1.84 | 2.33 | 1.57 |
| Sediment decreases | Sponges | O | 2.13 | 2.40 | 1.80 |
| Sediment increases | Beaches | N | 3.74 | 2.53 | 3.14 |
| Sediment increases | Corals | O | 2.23 | 3.67 | 2.94 |
| Sediment increases | Hard bottom | N | 2.12 | 2.40 | 1.79 |
| Sediment increases | Kelp | N | 4.00 | 2.27 | 3.26 |
| Sediment increases | Offshore pelagic | O | 1.70 | 1.60 | 0.93 |
| Sediment increases | Rocky intertidal | N | 3.93 | 2.60 | 3.34 |
| Sediment increases | Seamount | O | 1.00 | 2.40 | 1.40 |
| Sediment increases | Soft bottom | O | 1.78 | 2.13 | 1.38 |
| Sediment increases | Sponges | O | 1.81 | 2.80 | 1.97 |
| Shipping | Beaches | N | 2.70 | 2.53 | 2.29 |
| Shipping | Corals | O | 3.63 | 3.27 | 3.47 |
| Shipping | Hard bottom | N | 3.81 | 2.20 | 3.05 |

| | | | | | |
|----------|------------------|---|------|------|------|
| Shipping | Kelp | N | 2.79 | 2.07 | 2.08 |
| Shipping | Offshore pelagic | O | 4.00 | 1.80 | 3.10 |
| Shipping | Rocky intertidal | N | 2.60 | 2.40 | 2.13 |
| Shipping | Seamount | O | 1.00 | 2.20 | 1.20 |
| Shipping | Soft bottom | O | 3.94 | 2.33 | 3.23 |
| Shipping | Sponges | O | 3.63 | 2.40 | 2.98 |

Table EN.R.7. Scores, rationale, and references for pressure-invariant sensitivity criteria.

| Habitat | Score | Data Quality | Rationale | Reference |
|------------------------------|-------|--------------|---|--|
| <u>Current status</u> | | | | |
| Beaches | 3 | 4 | Best available evidence suggests significant short term erosion. Though erosion is a natural process, sea level rise due to climate change and coastal armoring impede retreat of beach habitats landward. | Moore and Griggs 2002, Stamski 2005, Hapke et al. 2009 |
| Corals | 4 | 1 | Very little information available about status; however, many feel that coral destruction is commonplace due to human activities such as bottom trawling. In addition, changes in ocean chemistry due to climate change threaten persistence. | Guinotte et al. 2006, ONMS 2009 |
| Hard bottom | 1 | 2 | Information on status and trends is sparse; influenced by changes in sediment deposition, some burial has occurred due to landslides near Big Sur, but exhumation of nearshore hard substrate appears more common. | ONMS 2009, Storlazzi et al. 2011 |
| Kelp | 2 | 4 | Best available evidence suggests a decline over the last 40 years statewide; harvest is permitted. | Bedford 2001 |
| Offshore pelagic | 2 | 2 | Information on status and trends is sparse, however, water quality issues give reason for concern. | ONMS 2009 |
| Rocky intertidal | 2 | 2 | Modest, localized impacts due to past landslide disposal in these habitats; land and ocean based warming due to climate change suggest cause for future concern. | ONMS 2009 |
| Seamount | 1 | 1 | Considered relatively pristine and currently protected, but information on status and trends is sparse. | DeVogelaere et al. 2005 |
| Soft bottom | 2 | 3 | Clear effects of bottom trawling and other disturbances in these habitats; influenced by changes in sediment deposition. | de Marignac et al. 2008 |

| | | | | |
|----------------------------------|---|---|--|--|
| Sponges | 3 | 1 | Very little information available about status; however, many feel that sponge destruction is commonplace due to human activities such as bottom trawling and oil and gas development. | ONMS 2009 |
| <u>Replenishment rate</u> | | | | |
| Beaches | 4 | 4 | California beaches are eroding in the long-term, and in central California they are at best not changing in size. | Hapke et al. 2009 |
| Corals | 3 | 2 | Recruitment assumed to be low and episodic like tropical scleractinian corals | Consalvey et al. 2006 |
| Hard bottom | 4 | 3 | Hard bottom habitat replenishment occurs on geological time scales | Storlazzi et al. 2011 |
| Kelp | 1 | 4 | Seasonal in spring and summer | Graham et al. 1997 |
| Offshore pelagic | 1 | 4 | Water mass residence time is measured on time scales of days | Broenkow and Smethie 1978, Graham and Largier 1997 |
| Rocky intertidal | 4 | 4 | Sediment deposition and removal depends on episodic nature of storms, but in general replenishment occurs on geological time scales of at least decades. | Storlazzi and Field 2000 |
| Seamount | 4 | 3 | Seamount formation occurs on geological time scales. | Davis et al. 2002 |
| Soft bottom | 4 | 3 | Soft bottom habitat replenishment occurs on geological time scales | Greene et al. 2002 |
| Sponges | 3 | 3 | Deep-water sponge recruitment is episodic at best | Leys and Lauzon 1998 |
| <u>Recovery time</u> | | | | |
| Beaches | 3 | 4 | California beaches are eroding in the long-term, and in central California they are at best not changing in size. | Hapke et al. 2009 |
| Corals | 3 | 4 | Corals are slow-growing and long-lived, and likely to recover slowly from perturbations. | Andrews et al. 2005 |
| Hard bottom | 4 | 3 | Hard bottom habitat recovery occurs on geological time scales | Storlazzi et al. 2011 |
| Kelp | 2 | 4 | Natural, strong disturbances due to wave action and subsequent recoveries are common and well-studied, occurring on time scales of years. | Reed et al. 2011 |
| Offshore pelagic | 1 | 4 | Water masses are exchanged with those outside the MBNMS region on time scales of days | Broenkow and Smethie 1978, Graham and Largier 1997 |

| | | | | |
|----------------------------|---|---|--|--|
| Rocky intertidal | 4 | 3 | Sediment deposition and removal depends on episodic nature of storms, but in general recovery rates of rocky shores occur on geological time scales. | Storlazzi and Field 2000 |
| Seamount | 4 | 3 | Seamount recovery occurs on geological time scales, if at all. | Davis et al. 2002 |
| Soft bottom | 3 | 3 | Soft bottom habitat recovery occurs on geological time scales | Greene et al. 2002 |
| Sponges | 3 | 3 | Deep-water sponges are slow-growing and long-lived, and likely to recover slowly from perturbations. | Leys and Lauzon 1998 |
| <u>Connectivity</u> | | | | |
| Beaches | 4 | 3 | Beach habitat connectivity only relevant on geological time scales | Moore and Griggs 2002, Hapke et al. 2009 |
| Corals | 4 | 4 | Coral dispersal distances average 0.044–0.785 km | Kinlan and Gaines 2003 |
| Hard bottom | 4 | 3 | Hard bottom habitat connectivity only relevant on geological time scales | Storlazzi et al. 2011 |
| Kelp | 4 | 4 | <i>Macrocystis pyrifera</i> dispersal distances average 10–40m | Shanks et al. 2003 |
| Offshore pelagic | 1 | 4 | Water mass exchange outside of the MBNMS region is on time scales of days | Broenkow and Smethie 1978, Graham and Largier 1997 |
| Rocky intertidal | 4 | 3 | Rocky intertidal habitat connectivity only relevant on geological time scales | Storlazzi and Field 2000 |
| Seamount | 4 | 4 | Seamount habitat connectivity not relevant | Davis et al. 2002 |
| Soft bottom | 4 | 3 | Soft bottom habitat connectivity only relevant on geological time scales | Greene et al. 2002 |
| Sponges | 3 | 4 | Coral dispersal distances average 1–4 km | Kinlan and Gaines 2003 |

Table EN.R.8. Scores, rationale, and references for the pressure-specific sensitivity criteria, change in area.

| Habitat | Pressure | Score | Data Quality Score | Change in area | |
|---------|---------------------------------|-------|--------------------|--|--------------------------|
| | | | | Rationale | Reference |
| Beaches | Bottom-tended fishing gear | N/A | N/A | N/A | N/A |
| Beaches | Coastal engineering | 4 | 4 | Coastal engineering prevents landward retreat of beaches | Stamski 2005 |
| Beaches | Inorganic pollution | 1 | 1 | Pollutants would not have effects on beach area | N/A |
| Beaches | Nutrient pollution | 1 | 1 | Pollutants would not have effects on beach area | N/A |
| Beaches | Organic pollution | 1 | 1 | Pollutants would not have effects on beach area | N/A |
| Beaches | Sea surface temperature changes | 1 | 1 | No known effect of sea surface temperatures on areal extent of beaches | N/A |
| Beaches | Sediment decreases | 4 | 4 | Reduction in sediment loads to beaches would reduce areal extent | Willis and Griggs 2003 |
| Beaches | Sediment increases | 4 | 4 | Increase in sediment loads to beaches would increase areal extent | Willis and Griggs 2003 |
| Beaches | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on beach area | N/A |
| Corals | Bottom-tended fishing gear | 4 | 4 | Bottom-tended fishing gear would significantly reduce areal extent of corals | Whitmire and Clarke 2007 |
| Corals | Coastal engineering | 4 | 2 | Coastal engineering would directly reduce the areal extent of habitat-forming corals | Stamski 2005 |

| | | | | | |
|-------------|---------------------------------|---|---|--|---------------------------------|
| Corals | Inorganic pollution | 4 | 2 | Pollutants can significantly reduce area of tropical corals by increasing mortality | Fabricius 2005 |
| Corals | Nutrient pollution | 4 | 2 | Pollutants can significantly reduce area of tropical corals by increasing disease prevalence and associated mortality | Bruno et al. 2003 |
| Corals | Organic pollution | 4 | 2 | Pollutants can significantly reduce area of tropical corals by increasing mortality | Firman 1995 |
| Corals | Sea surface temperature changes | 4 | 2 | Sea surface temperature increases could influence coral calcification rates, physiology, and biochemistry, and enhance mortality | Guinotte et al. 2006 |
| Corals | Sediment decreases | 1 | 2 | Reduction in sediment loads to corals may increase areal extent | Fabricius 2005 |
| Corals | Sediment increases | 3 | 2 | Increase in sediment loads to corals may reduce areal extent | Fabricius 2005 |
| Corals | Shipping | 4 | 2 | Shipping-associated pollutants can significantly reduce area of tropical corals by increasing mortality | Fabricius 2005 |
| Hard bottom | Bottom-tended fishing gear | 1 | 2 | No known effect of bottom-tended fishing gear on areal extent of hard bottom habitats | Auster 1998, Turner et al. 1999 |
| Hard bottom | Coastal engineering | 4 | 4 | Coastal engineering would significantly modify the areal extent of hard bottom habitat | Stamski 2005 |
| Hard bottom | Inorganic pollution | 1 | 1 | Pollutants would not have effects on hard bottom habitat area | N/A |
| Hard bottom | Nutrient pollution | 1 | 1 | Pollutants would not have effects on hard bottom habitat area | N/A |
| Hard bottom | Organic pollution | 1 | 1 | Pollutants would not have effects on hard bottom habitat area | N/A |

| | | | | | |
|-------------|---------------------------------|---|---|---|---|
| Hard bottom | Sea surface temperature changes | 1 | 1 | No known effect of sea surface temperatures on areal extent of hard bottom habitats | N/A |
| Hard bottom | Sediment decreases | 1 | 4 | Reduction in sediment loads to hard bottom habitat may increase areal extent | Storlazzi and Field 2000, Storlazzi et al. 2011 |
| Hard bottom | Sediment increases | 2 | 4 | Increase in sediment loads to hard bottom habitat may reduce areal extent | Storlazzi and Field 2000, Storlazzi et al. 2011 |
| Hard bottom | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on hard bottom habitat area | N/A |
| Kelp | Bottom-tended fishing gear | 4 | 2 | Bottom-tended fishing gear would significantly reduce areal extent of kelp forests | Auster 1998, Turner et al. 1999 |
| Kelp | Coastal engineering | 4 | 2 | Coastal engineering would reduce the areal extent of kelp forests | Stamski 2005 |
| Kelp | Inorganic pollution | 1 | 1 | Pollutants would not have effects on kelp forest area | N/A |
| Kelp | Nutrient pollution | 2 | 4 | Pollutants (eutrophication) could significantly reduce kelp forest area, though seasonal upwelling and nutrient-rich waters are common in the MBNMS | Zimmerman and Kremer 1984, Dayton 1985, Cloern 2001 |
| Kelp | Organic pollution | 1 | 1 | Pollutants would not have effects on kelp forest area | N/A |
| Kelp | Sea surface temperature changes | 4 | 4 | Sea surface temperature increases could increase kelp mortality | Dayton et al. 1992, Graham et al. 2007 |
| Kelp | Sediment decreases | 1 | 3 | Reduction in sediment loads to kelp forests may increase areal extent | Reed et al. 1988 |
| Kelp | Sediment increases | 2 | 3 | Increase in sediment loads to kelp forests may reduce areal extent | Reed et al. 1988 |
| Kelp | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on kelp forest area | N/A |

| | | | | | |
|------------------|---------------------------------|-----|-----|---|--------------|
| Offshore pelagic | Bottom-tended fishing gear | N/A | N/A | N/A | N/A |
| Offshore pelagic | Coastal engineering | 1 | 1 | Coastal engineering would not reduce the areal extent of offshore pelagic habitat significantly | N/A |
| Offshore pelagic | Inorganic pollution | 1 | 1 | Pollution would not reduce the areal extent of offshore pelagic habitat significantly | N/A |
| Offshore pelagic | Nutrient pollution | 1 | 1 | Pollution would not reduce the areal extent of offshore pelagic habitat significantly | N/A |
| Offshore pelagic | Organic pollution | 1 | 1 | Pollution would not reduce the areal extent of offshore pelagic habitat significantly | N/A |
| Offshore pelagic | Sea surface temperature changes | 1 | 1 | Sea surface temperature changes would not reduce the areal extent of offshore pelagic habitat significantly | N/A |
| Offshore pelagic | Sediment decreases | 1 | 1 | Sediment decreases would not reduce the areal extent of offshore pelagic habitat significantly | N/A |
| Offshore pelagic | Sediment increases | 1 | 1 | Sediment increases would not reduce the areal extent of offshore pelagic habitat significantly | N/A |
| Offshore pelagic | Shipping | 1 | 1 | Shipping-associated pollutants would not reduce the areal extent of offshore pelagic habitat significantly | N/A |
| Rocky intertidal | Bottom-tended fishing gear | N/A | N/A | N/A | N/A |
| Rocky intertidal | Coastal engineering | 4 | 4 | Coastal engineering would directly reduce the areal extent of rocky intertidal habitat | Stamski 2005 |

| | | | | | |
|------------------|---------------------------------|---|---|--|---|
| Rocky intertidal | Inorganic pollution | 1 | 1 | Pollutants would not have effects on rocky intertidal habitat area | N/A |
| Rocky intertidal | Nutrient pollution | 1 | 1 | Pollutants would not have effects on rocky intertidal habitat area | N/A |
| Rocky intertidal | Organic pollution | 1 | 1 | Pollutants would not have effects on rocky intertidal habitat area | N/A |
| Rocky intertidal | Sea surface temperature changes | 1 | 1 | No known effect of sea surface temperatures on areal extent of rocky intertidal habitats | N/A |
| Rocky intertidal | Sediment decreases | 1 | 4 | Reduction in sediment loads to rocky intertidal habitat may increase areal extent | Storlazzi and Field 2000 |
| Rocky intertidal | Sediment increases | 2 | 4 | Increase in sediment loads to rocky intertidal habitat may reduce areal extent | Storlazzi and Field 2000 |
| Rocky intertidal | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on rocky intertidal habitat area | N/A |
| Seamount | Bottom-tended fishing gear | 1 | 1 | No known effect of bottom-tended fishing gear on areal extent of seamount habitats | Auster 1998, Turner et al. 1999, Whitmire and Clarke 2007 |
| Seamount | Coastal engineering | 4 | 2 | Coastal engineering would directly reduce the areal extent of seamount habitat | Stamski 2005 |
| Seamount | Inorganic pollution | 1 | 1 | Pollutants would not have effects on seamount habitat area | N/A |
| Seamount | Nutrient pollution | 1 | 1 | Pollutants would not have effects on seamount habitat area | N/A |
| Seamount | Organic pollution | 1 | 1 | Pollutants would not have effects on seamount habitat area | N/A |
| Seamount | Sea surface temperature changes | 1 | 1 | No known effect of sea surface temperatures on areal extent of seamount habitats | N/A |

| | | | | | |
|-------------|---------------------------------|---|---|---|---|
| Seamount | Sediment decreases | 1 | 4 | Reduction in sediment loads to seamount habitat may increase areal extent | Menard 1955 |
| Seamount | Sediment increases | 2 | 4 | Increase in sediment loads to seamount habitat may reduce areal extent | Menard 1955 |
| Seamount | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on seamount habitat area | N/A |
| Soft bottom | Bottom-tended fishing gear | 1 | 2 | No known effect of bottom-tended fishing gear on areal extent of soft bottom habitats | Auster 1998, Turner et al. 1999 |
| Soft bottom | Coastal engineering | 4 | 2 | Coastal engineering would directly reduce the areal extent of soft bottom habitat | Stamski 2005 |
| Soft bottom | Inorganic pollution | 1 | 1 | Pollutants would not have effects on soft bottom habitat area | N/A |
| Soft bottom | Nutrient pollution | 1 | 1 | Pollutants would not have effects on soft bottom habitat area | N/A |
| Soft bottom | Organic pollution | 1 | 1 | Pollutants would not have effects on soft bottom habitat area | N/A |
| Soft bottom | Sea surface temperature changes | 1 | 1 | No known effect of sea surface temperatures on areal extent of soft bottom habitats | N/A |
| Soft bottom | Sediment decreases | 2 | 4 | Reduction in sediment loads to soft bottom habitat would reduce areal extent | Menard 1955, Greene et al. 2002 |
| Soft bottom | Sediment increases | 1 | 4 | Increase in sediment loads to soft bottom habitat would increase areal extent | Menard 1955, Greene et al. 2002 |
| Soft bottom | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on soft bottom habitat area | N/A |
| Sponges | Bottom-tended fishing gear | 4 | 4 | Bottom-tended fishing gear would reduce the areal extent of habitat-forming sponges | Auster 1998, Turner et al. 1999, Whitmire and Clarke 2007 |

| | | | | | |
|---------|---------------------------------|---|---|--|-----------------------|
| Sponges | Coastal engineering | 4 | 2 | Coastal engineering would reduce the areal extent of habitat-forming sponges | Stamski 2005 |
| Sponges | Inorganic pollution | 4 | 2 | Pollutants can significantly reduce area of sponge habitat by increasing mortality via disease | Webster 2007 |
| Sponges | Nutrient pollution | 4 | 2 | Pollutants can significantly reduce area of sponge habitat by increasing mortality via disease | Webster 2007 |
| Sponges | Organic pollution | 4 | 2 | Pollutants can significantly reduce area of sponge habitat by increasing mortality via disease | Webster 2007 |
| Sponges | Sea surface temperature changes | 4 | 2 | Sea surface temperature increases could increase mortality of habitat-forming sponges directly or indirectly (via <i>Vibrio</i> virulence) | Olsvig-Whittaker 2010 |
| Sponges | Sediment decreases | 1 | 2 | Reduction in sediment loads to habitat-forming sponges may increase areal extent | Airoidi 2003 |
| Sponges | Sediment increases | 2 | 2 | Increase in sediment loads to habitat-forming sponges may reduce areal extent | Airoidi 2003 |
| Sponges | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on areal extent of habitat-forming sponges | N/A |

Table EN.R.9. Scores, rationale, and references for the pressure-specific sensitivity criteria, change in structure.

| Habitat | Pressure | Change in structure | | | |
|---------|---------------------------------|---------------------|--------------------|--|--------------------------|
| | | Score | Data Quality Score | Rationale | Reference |
| Beaches | Bottom-tended fishing gear | N/A | N/A | N/A | N/A |
| Beaches | Coastal engineering | 4 | 4 | Coastal engineering (armoring) significantly modifies beach structure | Stamski 2005 |
| Beaches | Inorganic pollution | 2 | 2 | Some pollutants (e.g., plastics) can modify beach structure | Defeo et al. 2009 |
| Beaches | Nutrient pollution | 1 | 1 | Pollutants would not have effects on beach structure | N/A |
| Beaches | Organic pollution | 1 | 1 | Pollutants would not have effects on beach structure | N/A |
| Beaches | Sea surface temperature changes | 1 | 1 | No known effect of sea surface temperatures on structure of beaches | N/A |
| Beaches | Sediment decreases | 4 | 4 | Reduction in sediment loads to beaches would modify structure | Willis and Griggs 2003 |
| Beaches | Sediment increases | 1 | 4 | Increase in sediment loads to beaches would not modify structure (rugosity) | Willis and Griggs 2003 |
| Beaches | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on structure | N/A |
| Corals | Bottom-tended fishing gear | 4 | 4 | Bottom-tended fishing gear would significantly damage coral structure | Whitmire and Clarke 2007 |
| Corals | Coastal engineering | 4 | 2 | Coastal engineering would damage the structure of habitat-forming corals significantly | Stamski 2005 |
| Corals | Inorganic pollution | 1 | 1 | Pollutants would not have effects on coral structure | N/A |
| Corals | Nutrient pollution | 2 | 2 | Pollutants (eutrophication) could cause reductions in coral structural complexity (rugosity) | Miller and Hay 1996 |
| Corals | Organic pollution | 1 | 1 | Pollutants would not have effects on coral structure | N/A |

| | | | | | |
|-------------|---------------------------------|---|---|--|---|
| Corals | Sea surface temperature changes | 4 | 2 | Sea surface temperature increases could influence coral calcification rates, physiology, and biochemistry, and enhance mortality | Guinotte et al. 2006 |
| Corals | Sediment decreases | 1 | 2 | Reduction in sediment loads to corals would not modify or perhaps enhance structural complexity (rugosity) | Roberts et al. 2006 |
| Corals | Sediment increases | 4 | 2 | Excessive increase in sediment loads to corals would reduce structural complexity (rugosity) | Roberts et al. 2006 |
| Corals | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on structure | N/A |
| Hard bottom | Bottom-tended fishing gear | 2 | 2 | Modest effects of bottom-tended fishing gear on structure of hard bottom habitats | Auster 1998, Turner et al. 1999 |
| Hard bottom | Coastal engineering | 2 | 2 | Coastal engineering could modify the structural complexity (rugosity) of hard bottom habitat, increasing it on pavement and reducing it on rocky substrate | Stamski 2005 |
| Hard bottom | Inorganic pollution | 1 | 1 | Pollutants would not have effects on hard bottom habitat structure | N/A |
| Hard bottom | Nutrient pollution | 1 | 1 | Pollutants would not have effects on hard bottom habitat structure | N/A |
| Hard bottom | Organic pollution | 1 | 1 | Pollutants would not have effects on hard bottom habitat structure | N/A |
| Hard bottom | Sea surface temperature changes | 1 | 1 | No known effect of sea surface temperatures on structure of hard bottom habitats | N/A |
| Hard bottom | Sediment decreases | 1 | 4 | Reduction in sediment loads to hard bottom habitat would not modify or perhaps enhance structural complexity (rugosity) | Airoidi 2003, Storlazzi et al. 2011 |
| Hard bottom | Sediment increases | 2 | 4 | Increase in sediment loads to hard bottom habitat would reduce structural complexity (rugosity) | Storlazzi and Field 2000, Storlazzi et al. 2011 |
| Hard bottom | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on hard bottom habitat structure | N/A |
| Kelp | Bottom-tended fishing gear | 4 | 2 | Bottom-tended fishing gear would significantly reduce structural complexity (rugosity) of kelp forests | Auster 1998, Turner et al. 1999 |

| | | | | | |
|------------------|---------------------------------|-----|-----|--|---|
| Kelp | Coastal engineering | 4 | 2 | Coastal engineering would significantly damage kelp forests | Stamski 2005 |
| Kelp | Inorganic pollution | 1 | 1 | Pollutants would not have effects on kelp forest structure | N/A |
| Kelp | Nutrient pollution | 2 | 4 | Pollutants (eutrophication) would initially enhance and eventually significantly reduce kelp structural complexity (rugosity); seasonal upwelling and nutrient-rich waters are common in the MBNMS | Zimmerman and Kremer 1984, Dayton 1985, Cloern 2001 |
| Kelp | Organic pollution | 1 | 1 | Pollutants would not have effects on kelp forest structure | N/A |
| Kelp | Sea surface temperature changes | 4 | 4 | Sea surface temperature increases could increase kelp mortality | Dayton et al. 1992, Graham et al. 2007 |
| Kelp | Sediment decreases | 1 | 3 | Reduction in sediment loads to kelp forests would not modify or perhaps enhance structural complexity (rugosity) | Reed et al. 1988 |
| Kelp | Sediment increases | 2 | 3 | Increase in sediment loads to kelp forests would reduce structural complexity (rugosity) | Reed et al. 1988 |
| Kelp | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on kelp forest habitat structure | N/A |
| Offshore pelagic | Bottom-tended fishing gear | N/A | N/A | N/A | N/A |
| Offshore pelagic | Coastal engineering | 4 | 2 | Coastal engineering would significantly modify the structure of offshore pelagic habitat | Stamski 2005 |
| Offshore pelagic | Inorganic pollution | 2 | 3 | Some pollutants (e.g., plastics) can modify offshore pelagic structure | Thompson et al. 2004 |
| Offshore pelagic | Nutrient pollution | 1 | 1 | Pollution would not reduce the structure of offshore pelagic habitat significantly | N/A |
| Offshore pelagic | Organic pollution | 1 | 1 | Pollution would not reduce the structure of offshore pelagic habitat significantly | N/A |
| Offshore pelagic | Sea surface temperature changes | 4 | 4 | Sea surface temperature changes would significantly modify the structure of water masses and physical forcing in offshore pelagic habitat | Di Lorenzo et al. 2005 |
| Offshore pelagic | Sediment decreases | 1 | 1 | Sediment decreases would not reduce the structure of offshore pelagic habitat significantly | N/A |

| | | | | | |
|------------------|---------------------------------|-----|-----|---|---|
| Offshore pelagic | Sediment increases | 1 | 1 | Sediment increases would not reduce the structure of offshore pelagic habitat significantly | N/A |
| Offshore pelagic | Shipping | 1 | 1 | Shipping-associated pollutants would not modify the structure of offshore pelagic habitat significantly | N/A |
| Rocky intertidal | Bottom-tended fishing gear | N/A | N/A | N/A | N/A |
| Rocky intertidal | Coastal engineering | 4 | 4 | Depending on the type of coastal engineering, it could enhance or reduce the structural complexity (rugosity) of rocky intertidal habitat | Stamski 2005 |
| Rocky intertidal | Inorganic pollution | 1 | 1 | Pollutants would not have effects on rocky intertidal habitat structure | N/A |
| Rocky intertidal | Nutrient pollution | 1 | 1 | Pollutants would not have effects on rocky intertidal habitat structure | N/A |
| Rocky intertidal | Organic pollution | 1 | 1 | Pollutants would not have effects on rocky intertidal habitat structure | N/A |
| Rocky intertidal | Sea surface temperature changes | 1 | 1 | No known effect of sea surface temperatures on structure of rocky intertidal habitats | N/A |
| Rocky intertidal | Sediment decreases | 1 | 4 | Reduction in sediment loads to rocky intertidal habitat would not modify or perhaps enhance structural complexity (rugosity) | Storlazzi and Field 2000 |
| Rocky intertidal | Sediment increases | 2 | 4 | Increase in sediment loads to rocky intertidal habitat would reduce structural complexity (rugosity) | Storlazzi and Field 2000 |
| Rocky intertidal | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on rocky intertidal habitat structure | N/A |
| Seamount | Bottom-tended fishing gear | 2 | 1 | Modest effects of bottom-tended fishing gear on structure of seamount habitats | Auster 1998, Turner et al. 1999, Whitmire and Clarke 2007 |
| Seamount | Coastal engineering | 4 | 2 | Coastal engineering could alter the structural complexity (rugosity) of seamount habitat | Stamski 2005 |
| Seamount | Inorganic pollution | 1 | 1 | Pollutants would not have effects on seamount habitat structure | N/A |
| Seamount | Nutrient pollution | 1 | 1 | Pollutants would not have effects on seamount habitat structure | N/A |

| | | | | | |
|-------------|---------------------------------|---|---|---|---|
| Seamount | Organic pollution | 1 | 1 | Pollutants would not have effects on seamount habitat structure | N/A |
| Seamount | Sea surface temperature changes | 1 | 1 | No known effect of sea surface temperatures on structure of seamount habitats | N/A |
| Seamount | Sediment decreases | 1 | 2 | Reduction in sediment loads to seamount habitat would not modify or perhaps enhance structural complexity (rugosity) | Tittensor et al. 2009 |
| Seamount | Sediment increases | 2 | 2 | Increase in sediment loads to seamount habitat would reduce structural complexity (rugosity) | Tittensor et al. 2009 |
| Seamount | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on seamount habitat structure | N/A |
| Soft bottom | Bottom-tended fishing gear | 4 | 4 | Bottom-tended fishing gear significantly modifies the structure of soft bottom habitats | Engel and Kvitek 1998 |
| Soft bottom | Coastal engineering | 1 | 2 | Coastal engineering would increase the structural complexity (rugosity) of soft bottom habitat | Stamski 2005 |
| Soft bottom | Inorganic pollution | 1 | 1 | Pollutants would not have effects on soft bottom habitat structure | N/A |
| Soft bottom | Nutrient pollution | 1 | 1 | Pollutants would not have effects on soft bottom habitat structure | N/A |
| Soft bottom | Organic pollution | 1 | 1 | Pollutants would not have effects on soft bottom habitat structure | N/A |
| Soft bottom | Sea surface temperature changes | 1 | 1 | No known effect of sea surface temperatures on structure of soft bottom habitats | N/A |
| Soft bottom | Sediment decreases | 1 | 4 | Reduction in sediment loads to soft bottom habitat would not modify or perhaps enhance structural complexity (rugosity) | Menard 1955, Greene et al. 2002 |
| Soft bottom | Sediment increases | 1 | 4 | Increase in sediment loads to soft bottom habitat would not modify structural complexity (rugosity) | Menard 1955, Greene et al. 2002 |
| Soft bottom | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on soft bottom habitat structure | N/A |
| Sponges | Bottom-tended fishing gear | 4 | 4 | Bottom-tended fishing gear would significantly reduce structural complexity (rugosity) of habitat-forming sponges | Auster 1998, Turner et al. 1999, Whitmire and Clarke 2007 |

| | | | | | |
|---------|---------------------------------|---|---|--|-----------------------|
| Sponges | Coastal engineering | 4 | 2 | Coastal engineering would significantly damage habitat-forming sponges | Stamski 2005 |
| Sponges | Inorganic pollution | 4 | 2 | Pollutants can significantly reduce structural complexity of sponge habitat by increasing mortality via disease | Webster 2007 |
| Sponges | Nutrient pollution | 4 | 2 | Pollutants can significantly reduce structural complexity of sponge habitat by increasing mortality via disease | Webster 2007 |
| Sponges | Organic pollution | 4 | 2 | Pollutants can significantly reduce structural complexity of sponge habitat by increasing mortality via disease | Webster 2007 |
| Sponges | Sea surface temperature changes | 4 | 2 | Sea surface temperature increases could increase mortality of habitat-forming sponges directly or indirectly (via <i>Vibrio</i> virulence) | Olsvig-Whittaker 2010 |
| Sponges | Sediment decreases | 1 | 2 | Reduction in sediment loads to habitat-forming sponges would not modify or perhaps enhance structural complexity (rugosity) | Airoidi 2003 |
| Sponges | Sediment increases | 2 | 2 | Increase in sediment loads to habitat-forming sponges would reduce structural complexity (rugosity) | Airoidi 2003 |
| Sponges | Shipping | 1 | 1 | Shipping-associated pollutants would not have effects on structure of habitat-forming sponges | N/A |

Table EN.R.10. Scores, rationale, and references for the pressure-specific sensitivity criteria, frequency of natural disturbance.

| Habitat | Pressure | Score | Frequency of natural disturbance | | |
|---------|---------------------------------|-------|----------------------------------|--|------------------------|
| | | | Data Quality Score | Rationale | Reference |
| Beaches | Bottom-tended fishing gear | N/A | N/A | N/A | N/A |
| Beaches | Coastal engineering | 4 | 4 | Coastal engineering is a completely artificial occurrence, except in cases of major seismic activity on geological time scales | Stamski 2005 |
| Beaches | Inorganic pollution | 4 | 2 | High concentrations of inorganic pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Beaches | Nutrient pollution | 3 | 4 | Seasonal coastal upwelling introduces high nutrient concentrations | Huyer 1983 |
| Beaches | Organic pollution | 4 | 2 | High concentrations of organic pollutants (pesticides) are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Beaches | Sea surface temperature changes | 1 | 2 | Periodic warming and cooling of sea surface temperatures is characteristic of the California Current ecosystem (e.g., ENSO) | Bograd and Lynn 2001 |
| Beaches | Sediment decreases | 1 | 4 | Variable sediment dynamics are a natural process | Willis and Griggs 2003 |
| Beaches | Sediment increases | 1 | 4 | Variable sediment dynamics are a natural process | Willis and Griggs 2003 |
| Beaches | Shipping | 4 | 2 | High concentrations of pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Corals | Bottom-tended fishing gear | 4 | 1 | There is no natural analog to bottom-tended fishing gear in coral habitat | N/A |
| Corals | Coastal engineering | 4 | 4 | Coastal engineering is a completely artificial occurrence, except in cases of major seismic activity on geological time scales | Stamski 2005 |
| Corals | Inorganic pollution | 4 | 2 | High concentrations of inorganic pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Corals | Nutrient pollution | 4 | 4 | Seasonal coastal upwelling introduces high nutrient concentrations in shallow waters, | Pilskaln et al. 1996 |

| | | | | | |
|-------------|---------------------------------|---|---|--|-----------------------|
| | | | | but these effects attenuate at deeper depths | |
| Corals | Organic pollution | 4 | 2 | High concentrations of organic pollutants (pesticides) are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Corals | Sea surface temperature changes | 3 | 4 | Seasonal variability in temperature is modest in sub-surface waters off of MBNMS | Lynn and Simpson 1987 |
| Corals | Sediment decreases | 4 | 2 | Sediment increases and decreases occur naturally but at much slower rates than anthropogenically forced sediment dynamics | Roberts et al. 2006 |
| Corals | Sediment increases | 4 | 2 | Sediment increases and decreases occur naturally but at much slower rates than anthropogenically forced sediment dynamics | Roberts et al. 2006 |
| Corals | Shipping | 4 | 2 | High concentrations of pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Hard bottom | Bottom-tended fishing gear | 4 | 1 | There is no natural analog to bottom-tended fishing gear in hard bottom habitat | N/A |
| Hard bottom | Coastal engineering | 4 | 4 | Coastal engineering is a completely artificial occurrence, except in cases of major seismic activity on geological time scales | Stamski 2005 |
| Hard bottom | Inorganic pollution | 4 | 2 | High concentrations of inorganic pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Hard bottom | Nutrient pollution | 4 | 4 | Seasonal coastal upwelling introduces high nutrient concentrations in shallow waters, but these effects attenuate at deeper depths | Pilskaln et al. 1996 |
| Hard bottom | Organic pollution | 4 | 2 | High concentrations of organic pollutants (pesticides) are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Hard bottom | Sea surface temperature changes | 3 | 4 | Seasonal variability in temperature is modest in sub-surface waters off of MBNMS | Lynn and Simpson 1987 |
| Hard bottom | Sediment decreases | 3 | 4 | Sediment increases and decreases occur naturally but at much slower rates than anthropogenically forced sediment dynamics | Storlazzi et al. 2011 |

| | | | | | |
|------------------|---------------------------------|-----|-----|--|--------------------------------------|
| Hard bottom | Sediment increases | 3 | 4 | Sediment increases and decreases occur naturally but at much slower rates than anthropogenically forced sediment dynamics | Storlazzi et al. 2011 |
| Hard bottom | Shipping | 4 | 2 | High concentrations of pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Kelp | Bottom-tended fishing gear | 3 | 4 | The only natural analog to effects from bottom-tended fishing is storms, but the impacts of storms on the benthos are more transient | Dayton et al. 1992, Reed et al. 2011 |
| Kelp | Coastal engineering | 4 | 4 | Coastal engineering is a completely artificial occurrence, and has no natural analog except on geological time scales | Graham et al. 2003 |
| Kelp | Inorganic pollution | 4 | 2 | High concentrations of inorganic pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Kelp | Nutrient pollution | 3 | 4 | Seasonal coastal upwelling introduces high nutrient concentrations | Huyer 1983 |
| Kelp | Organic pollution | 4 | 2 | High concentrations of organic pollutants (pesticides) are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Kelp | Sea surface temperature changes | 1 | 3 | Kelp is highly accustomed to sea surface temperature variation through space and time | Jackson 1977 |
| Kelp | Sediment decreases | 3 | 3 | Kelp forests are characterized by episodic delivery and removal of sediments via storms | Reed et al. 1988 |
| Kelp | Sediment increases | 3 | 3 | Kelp forests are characterized by episodic delivery and removal of sediments via storms | Reed et al. 1988 |
| Kelp | Shipping | 4 | 2 | High concentrations of pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Offshore pelagic | Bottom-tended fishing gear | N/A | N/A | N/A | N/A |
| Offshore pelagic | Coastal engineering | 4 | 4 | Coastal engineering is a completely artificial occurrence, and has no natural analog | Stamski 2005 |
| Offshore pelagic | Inorganic pollution | 4 | 2 | High concentrations of inorganic pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |

| | | | | | |
|------------------|---------------------------------|-----|-----|--|--------------------------|
| Offshore pelagic | Nutrient pollution | 3 | 4 | Seasonal coastal upwelling introduces high nutrient concentrations | Huyer 1983 |
| Offshore pelagic | Organic pollution | 4 | 2 | High concentrations of organic pollutants (pesticides) are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Offshore pelagic | Sea surface temperature changes | 1 | 2 | Periodic warming and cooling of sea surface temperatures is characteristic of the California Current ecosystem (e.g., ENSO) | Bograd and Lynn 2001 |
| Offshore pelagic | Sediment decreases | 3 | 4 | Sediment increases and decreases occur naturally but at much slower rates than anthropogenically forced sediment dynamics | Pilskaln et al. 1998 |
| Offshore pelagic | Sediment increases | 3 | 4 | Sediment increases and decreases occur naturally but at much slower rates than anthropogenically forced sediment dynamics | Pilskaln et al. 1998 |
| Offshore pelagic | Shipping | 4 | 2 | High concentrations of pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Rocky intertidal | Bottom-tended fishing gear | N/A | N/A | N/A | N/A |
| Rocky intertidal | Coastal engineering | 4 | 4 | Coastal engineering is a completely artificial occurrence, except in cases of major seismic activity on geological time scales | Stamski 2005 |
| Rocky intertidal | Inorganic pollution | 4 | 2 | High concentrations of inorganic pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Rocky intertidal | Nutrient pollution | 3 | 4 | Seasonal coastal upwelling introduces high nutrient concentrations | Huyer 1983 |
| Rocky intertidal | Organic pollution | 4 | 2 | High concentrations of organic pollutants (pesticides) are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Rocky intertidal | Sea surface temperature changes | 1 | 2 | Rocky intertidal habitats are characterized by exposure to high variability in sea surface temperatures | Helmuth et al. 2002 |
| Rocky intertidal | Sediment decreases | 3 | 4 | Sediment increases and decreases occur naturally but at much slower rates than anthropogenically forced sediment dynamics | Storlazzi and Field 2000 |

| | | | | | |
|------------------|---------------------------------|---|---|--|--------------------------|
| Rocky intertidal | Sediment increases | 3 | 4 | Sediment increases and decreases occur naturally but at much slower rates than anthropogenically forced sediment dynamics | Storlazzi and Field 2000 |
| Rocky intertidal | Shipping | 4 | 2 | High concentrations of pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Seamount | Bottom-tended fishing gear | 4 | 1 | There is no natural analog to bottom-tended fishing gear in seamount habitat | N/A |
| Seamount | Coastal engineering | 4 | 4 | Coastal engineering is a completely artificial occurrence, except in cases of major seismic activity on geological time scales | Stamski 2005 |
| Seamount | Inorganic pollution | 4 | 2 | High concentrations of inorganic pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Seamount | Nutrient pollution | 3 | 4 | Seasonal coastal upwelling introduces high nutrient concentrations | Huyer 1983 |
| Seamount | Organic pollution | 4 | 2 | High concentrations of organic pollutants (pesticides) are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Seamount | Sea surface temperature changes | 3 | 4 | Seasonal variability in temperature is modest in sub-surface waters off of MBNMS | Lynn and Simpson 1987 |
| Seamount | Sediment decreases | 3 | 4 | Monterey deep-sea fan drives sediment dynamics on Davidson seamount, but anthropogenic sediment dynamics are much faster | Menard 1955 |
| Seamount | Sediment increases | 3 | 4 | Monterey deep-sea fan drives sediment dynamics on Davidson seamount, but anthropogenic sediment dynamics are much faster | Menard 1955 |
| Seamount | Shipping | 4 | 2 | High concentrations of pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Soft bottom | Bottom-tended fishing gear | 4 | 1 | There is no natural analog to bottom-tended fishing gear in soft bottom habitat | N/A |
| Soft bottom | Coastal engineering | 4 | 4 | Coastal engineering is a completely artificial occurrence, except in cases of major seismic activity on geological time scales | Stamski 2005 |
| Soft bottom | Inorganic pollution | 4 | 2 | High concentrations of inorganic pollutants are almost exclusively a | Islam and Tanaka 2004 |

| | | | | | |
|-------------|---------------------------------|---|---|--|-----------------------|
| | | | | result of anthropogenic activities | |
| Soft bottom | Nutrient pollution | 4 | 4 | Seasonal coastal upwelling introduces high nutrient concentrations in shallow waters, but these effects attenuate at deeper depths | Pilskaln et al. 1996 |
| Soft bottom | Organic pollution | 4 | 2 | High concentrations of organic pollutants (pesticides) are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Soft bottom | Sea surface temperature changes | 3 | 4 | Seasonal variability in temperature is modest in sub-surface waters off of MBNMS | Lynn and Simpson 1987 |
| Soft bottom | Sediment decreases | 3 | 4 | Monterey deep-sea fan drives sediment dynamics in large tracts of soft bottom habitat, but anthropogenic sediment dynamics are much faster | Menard 1955 |
| Soft bottom | Sediment increases | 3 | 4 | Monterey deep-sea fan drives sediment dynamics in large tracts of soft bottom habitat, but anthropogenic sediment dynamics are much faster | Menard 1955 |
| Soft bottom | Shipping | 4 | 2 | High concentrations of pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Sponges | Bottom-tended fishing gear | 4 | 1 | There is no natural analog to bottom-tended fishing gear in sponge habitat | N/A |
| Sponges | Coastal engineering | 4 | 4 | Coastal engineering is a completely artificial occurrence, except in cases of major seismic activity on geological time scales | Stamski 2005 |
| Sponges | Inorganic pollution | 4 | 2 | High concentrations of inorganic pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Sponges | Nutrient pollution | 4 | 4 | Seasonal coastal upwelling introduces high nutrient concentrations in shallow waters, but these effects attenuate at deeper depths | Pilskaln et al. 1996 |
| Sponges | Organic pollution | 4 | 2 | High concentrations of organic pollutants (pesticides) are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |
| Sponges | Sea surface temperature changes | 3 | 4 | Seasonal variability in temperature is modest in sub-surface waters off | Lynn and Simpson 1987 |

| | | | | of MBNMS | |
|---------|--------------------|---|---|---|-----------------------|
| Sponges | Sediment decreases | 4 | 2 | Sediment increases and decreases occur naturally but at much slower rates than anthropogenically forced sediment dynamics | Roberts et al. 2006 |
| Sponges | Sediment increases | 4 | 2 | Sediment increases and decreases occur naturally but at much slower rates than anthropogenically forced sediment dynamics | Roberts et al. 2006 |
| Sponges | Shipping | 4 | 2 | High concentrations of pollutants are almost exclusively a result of anthropogenic activities | Islam and Tanaka 2004 |

EXPERT-BASED RISK ASSESSMENT

We distributed the risk survey to 43 people associated with the MBNMS Research Activity Panel, and of those, 28 provided comprehensive responses. At the broadest level, the survey responses suggested that the current status of habitats in the MBNMS is considered to be fair to good (Fig. EN.R.12). They also implied that coastal pollution and bottom-tended fishing ranked among the top pressures in the region (Fig. EN.R.13), out of a set of nine that were queried (coastal pollution, bottom-tended fishing, ocean warming, aquaculture, invasive species, marine debris, ocean acidification, ocean-based pollution, coastal engineering).

Closer inspection of responses to detailed questions about risk due to coastal pollution and bottom-tended fishing confirmed that risk to habitats due to bottom-tended fishing was considered comparable to risk due to coastal pollution (Figs. EN.R.14-19). This outcome is the result of experts generally perceiving both greater exposure (Figs. EN.R.15, 17) and greater sensitivity (Figs. EN.R.16, 18) of habitats to coastal pollution as compared to bottom-tended fishing. Experts tended to be somewhat more uncertain regarding their coastal pollution responses compared with their bottom-tended fishing responses (Fig. EN.R.19).

Figure EN.R.12. Expert-based assessment of the current status of habitats in the MBNMS.

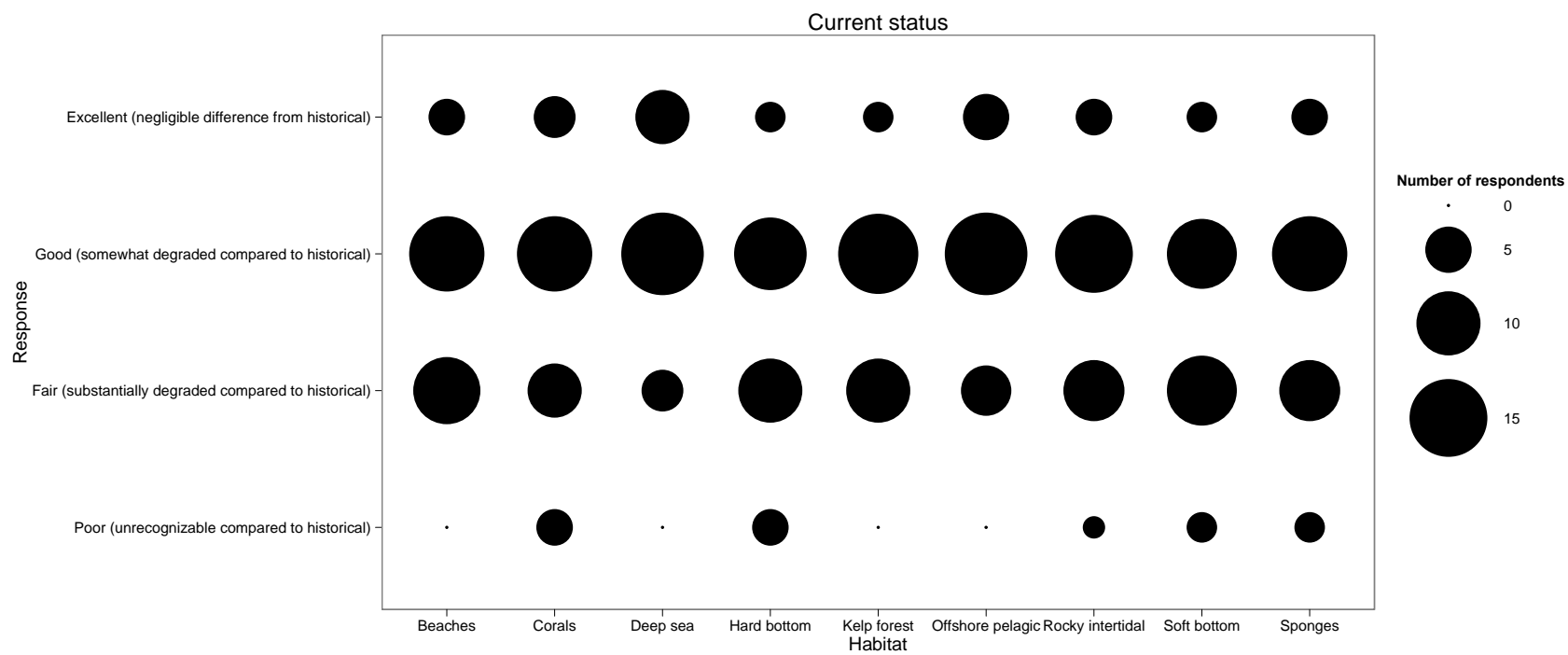


Figure EN.R.13. Expert-based assessment of the relative intensity of different activities and pressures throughout the MBNMS.

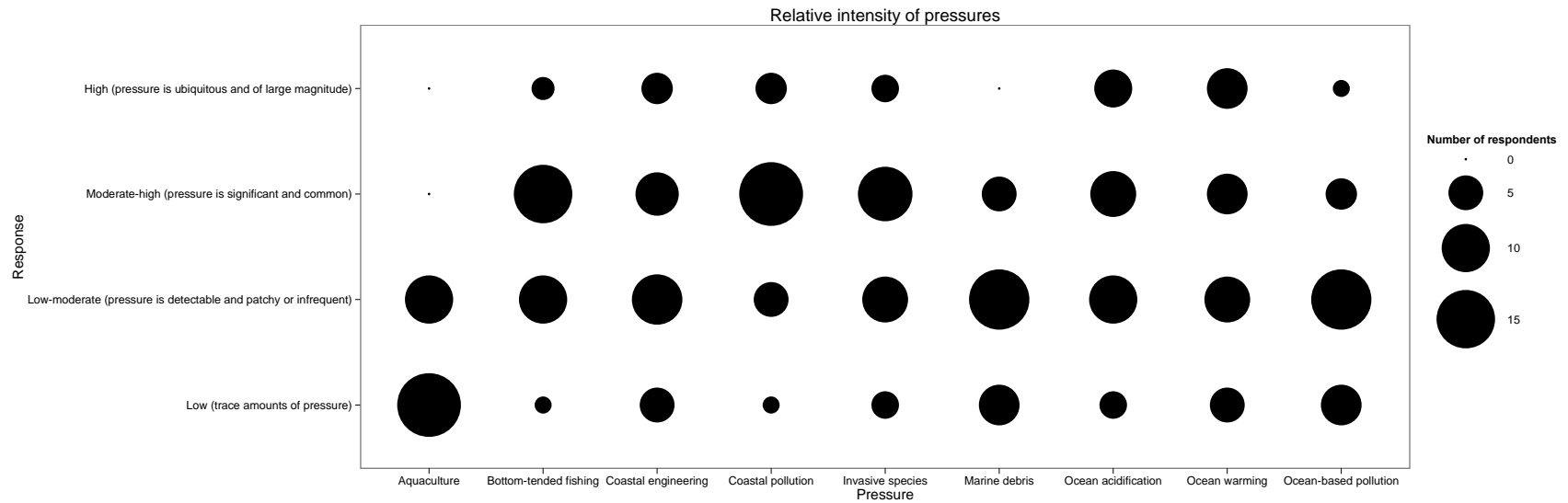


Figure EN.R.14. Expert-based assessment of risk to habitats within the MBNMS due to (a) bottom-tended fishing, and (b) coastal pollution. Data points represent average scores across respondents. B = beaches, C = corals, DS = deep sea, HB = hard bottom, KF = kelp forest, OP = offshore pelagic, RI = rocky intertidal, S = sponges, SB = soft bottom.

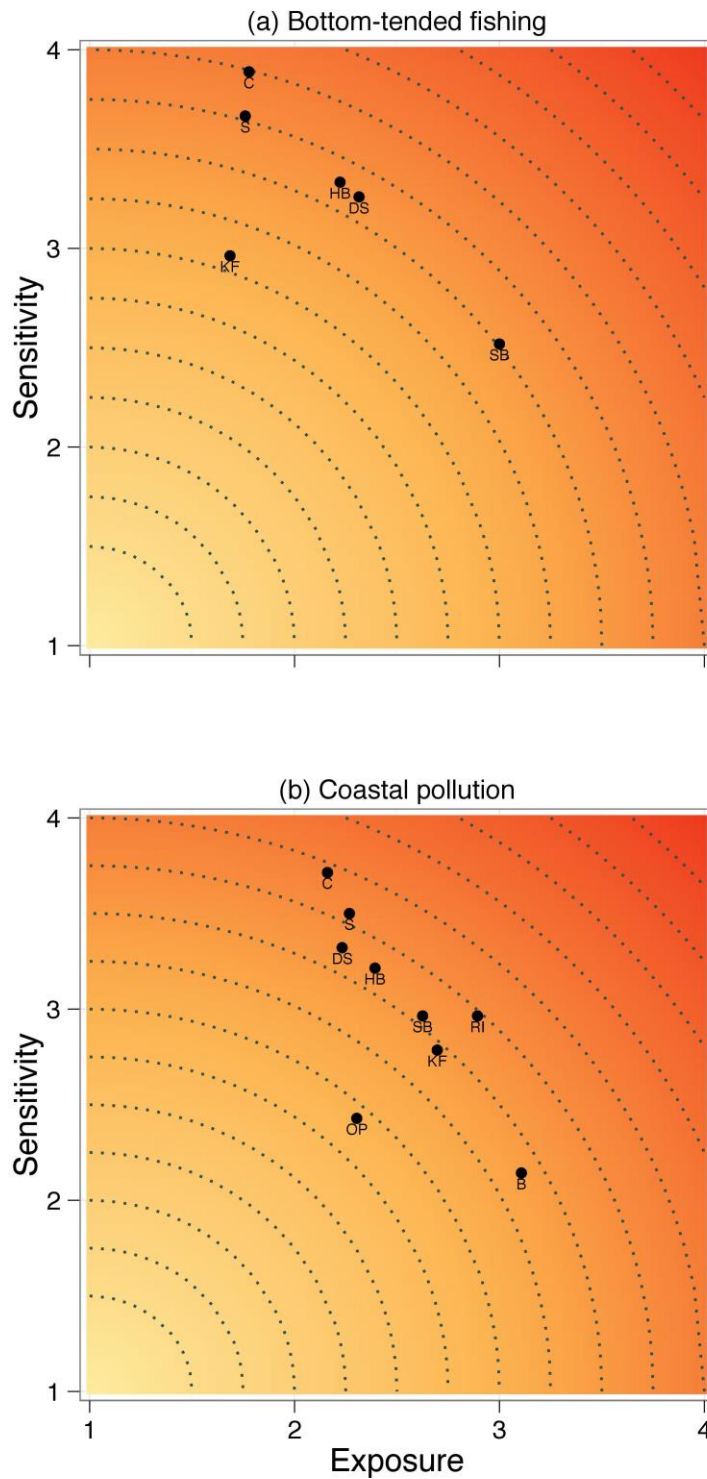


Figure EN.R.15. Expert-based assessment of exposure of habitats to bottom-tended fishing within the MBNMS, based on the spatial footprint of (top) and the temporal overlap with (bottom) bottom-tended fishing.

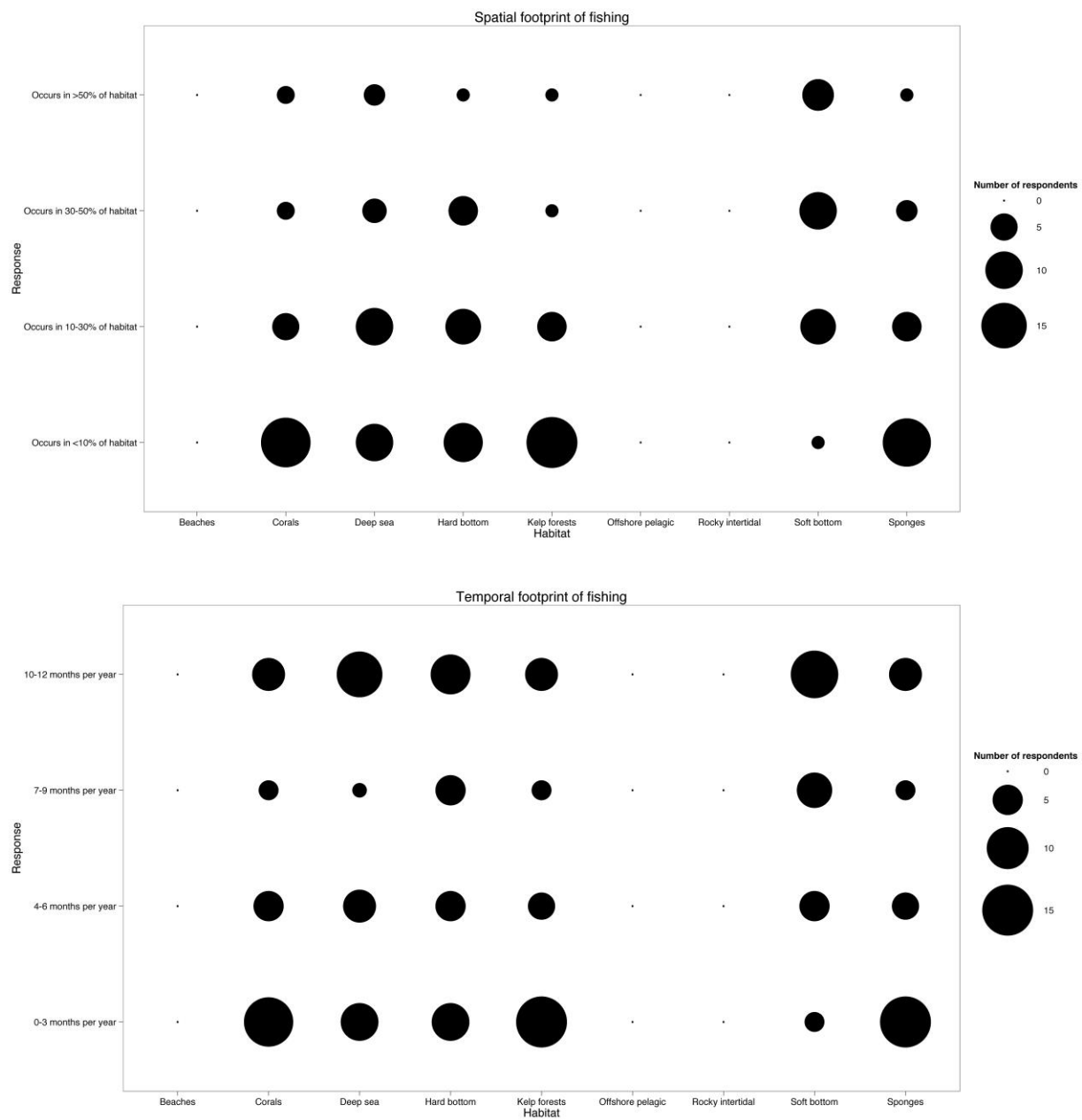


Figure EN.R.16. Expert-based assessment of sensitivity of habitats to bottom-tended fishing within the MBNMS, based on the expected degree of habitat loss (top) and the recovery rate (bottom) from bottom-tended fishing.

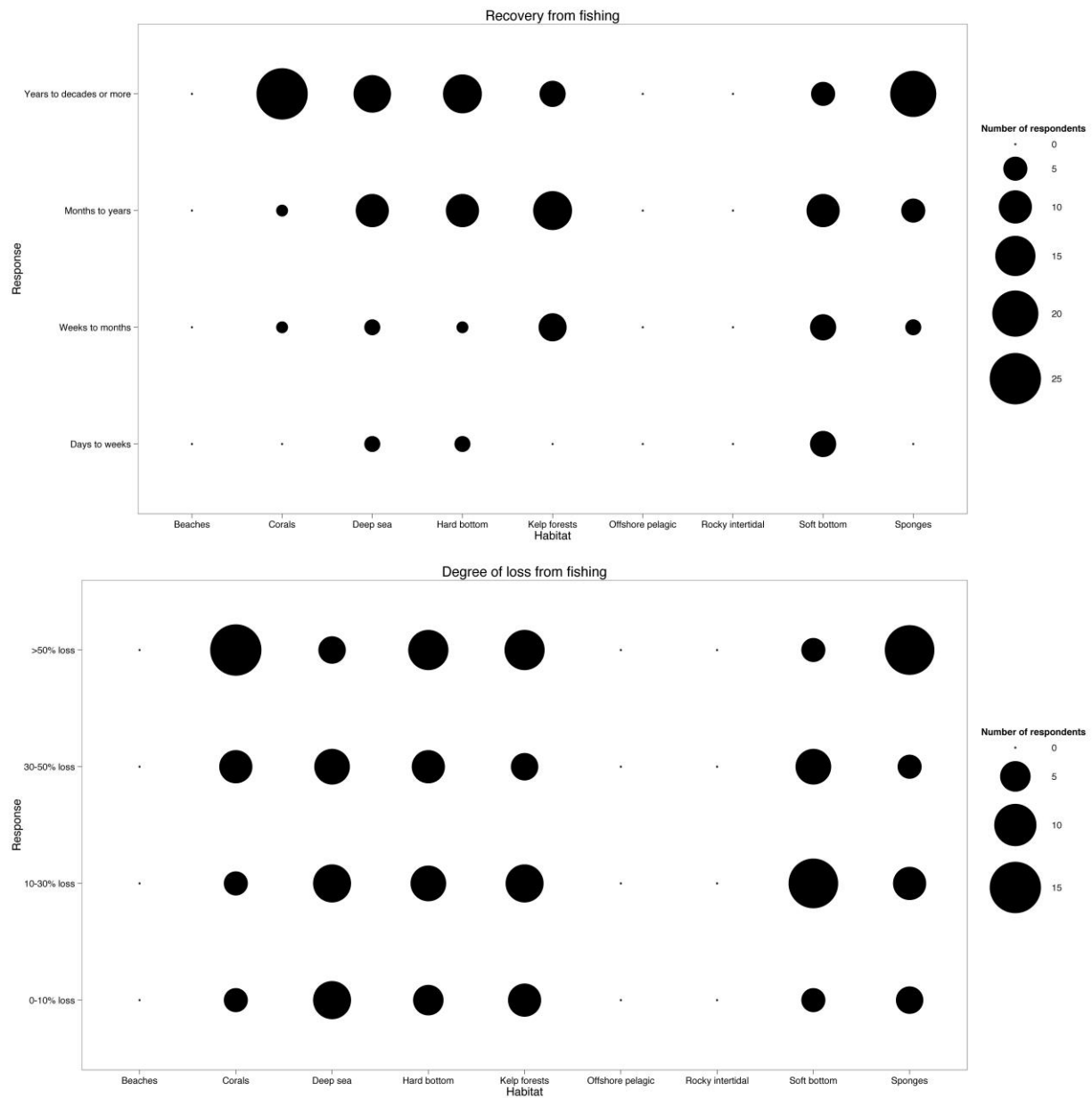


Figure EN.R.17. Expert-based assessment of exposure of habitats to coastal pollution within the MBNMS, based on the spatial footprint of (top) and the temporal overlap with (bottom) coastal pollution.

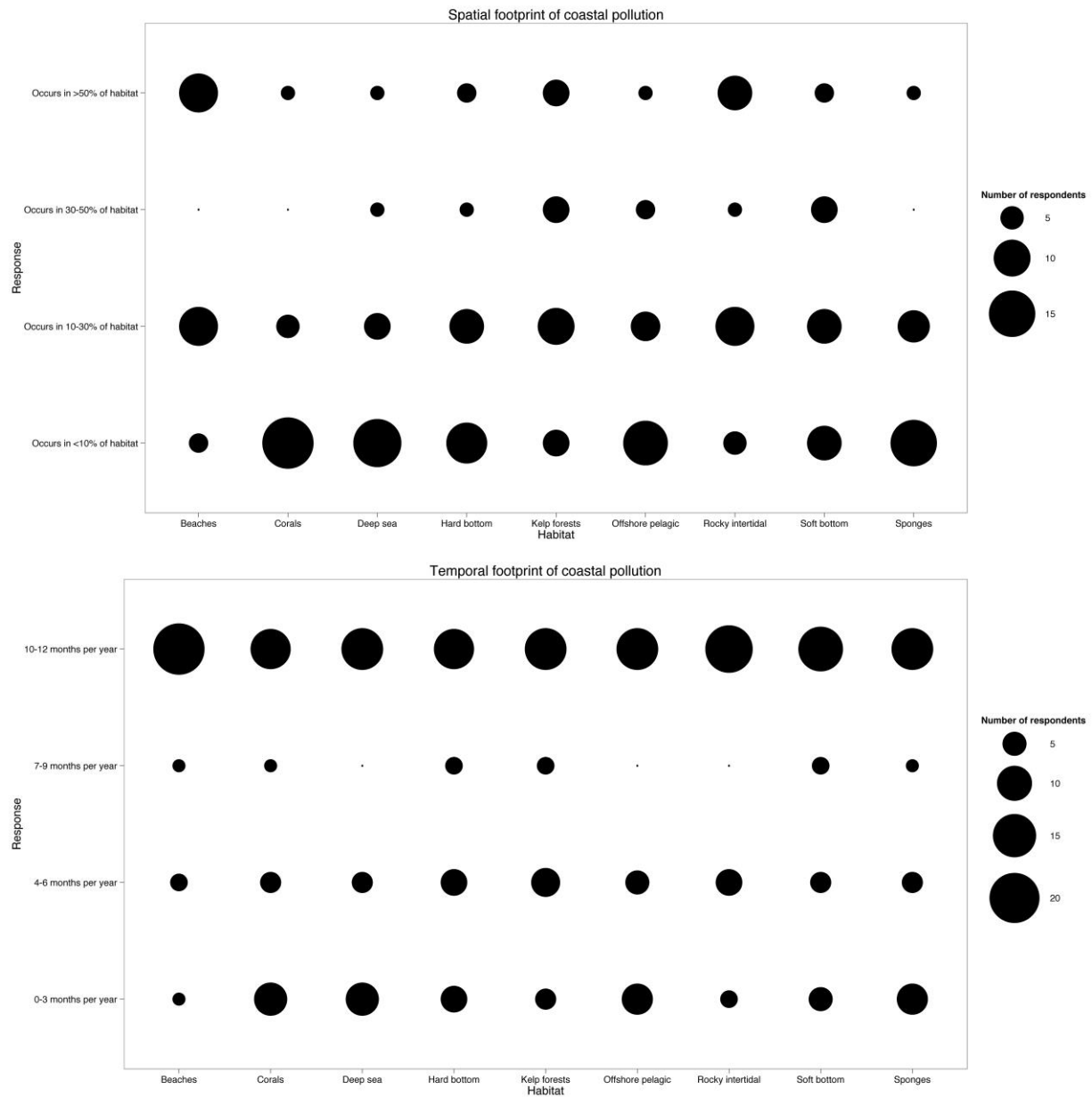


Figure EN.R.18. Expert-based assessment of sensitivity of habitats to coastal pollution within the MBNMS, based on the expected degree of habitat loss (top) and the recovery rate (bottom) from coastal pollution.

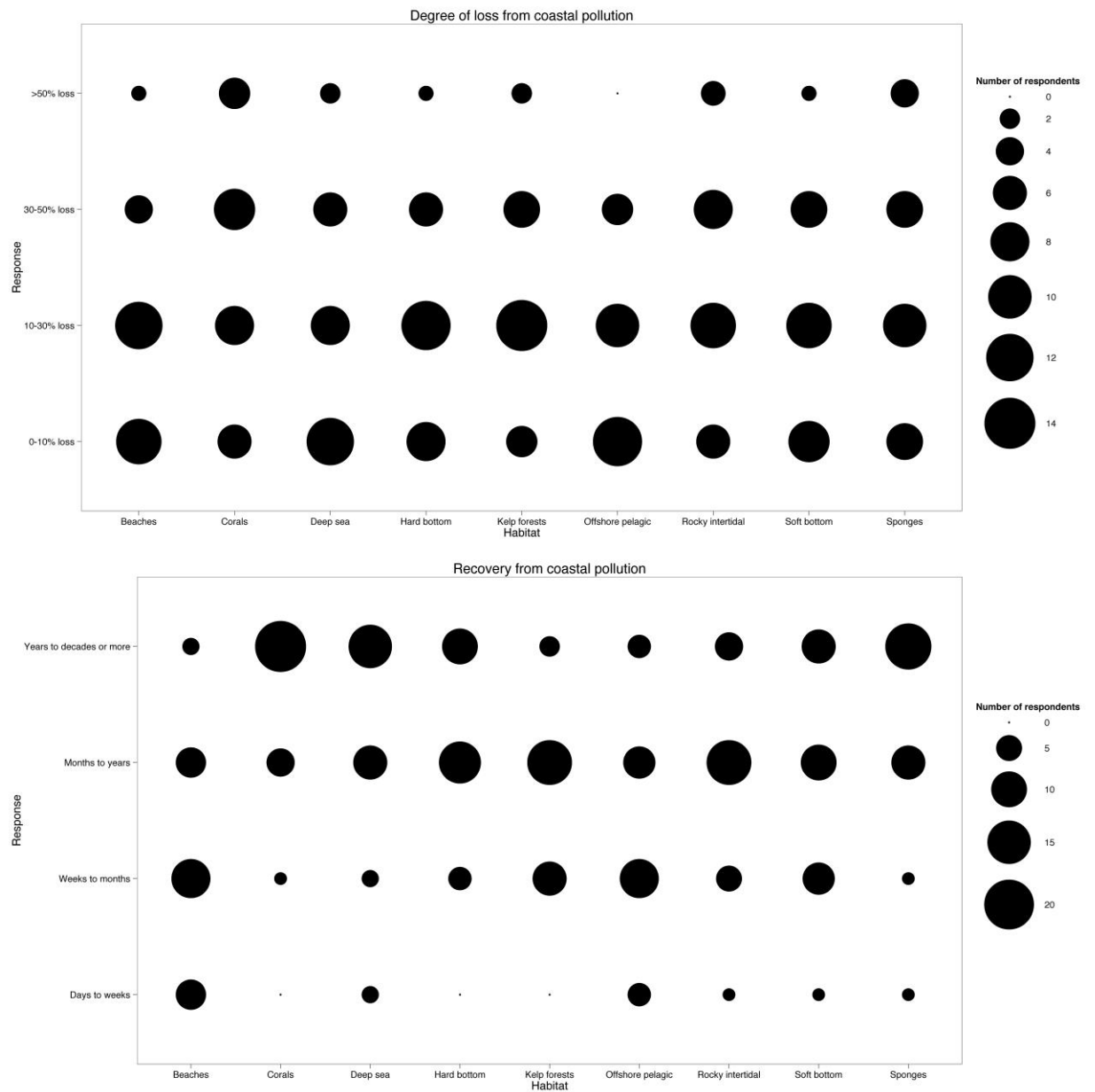
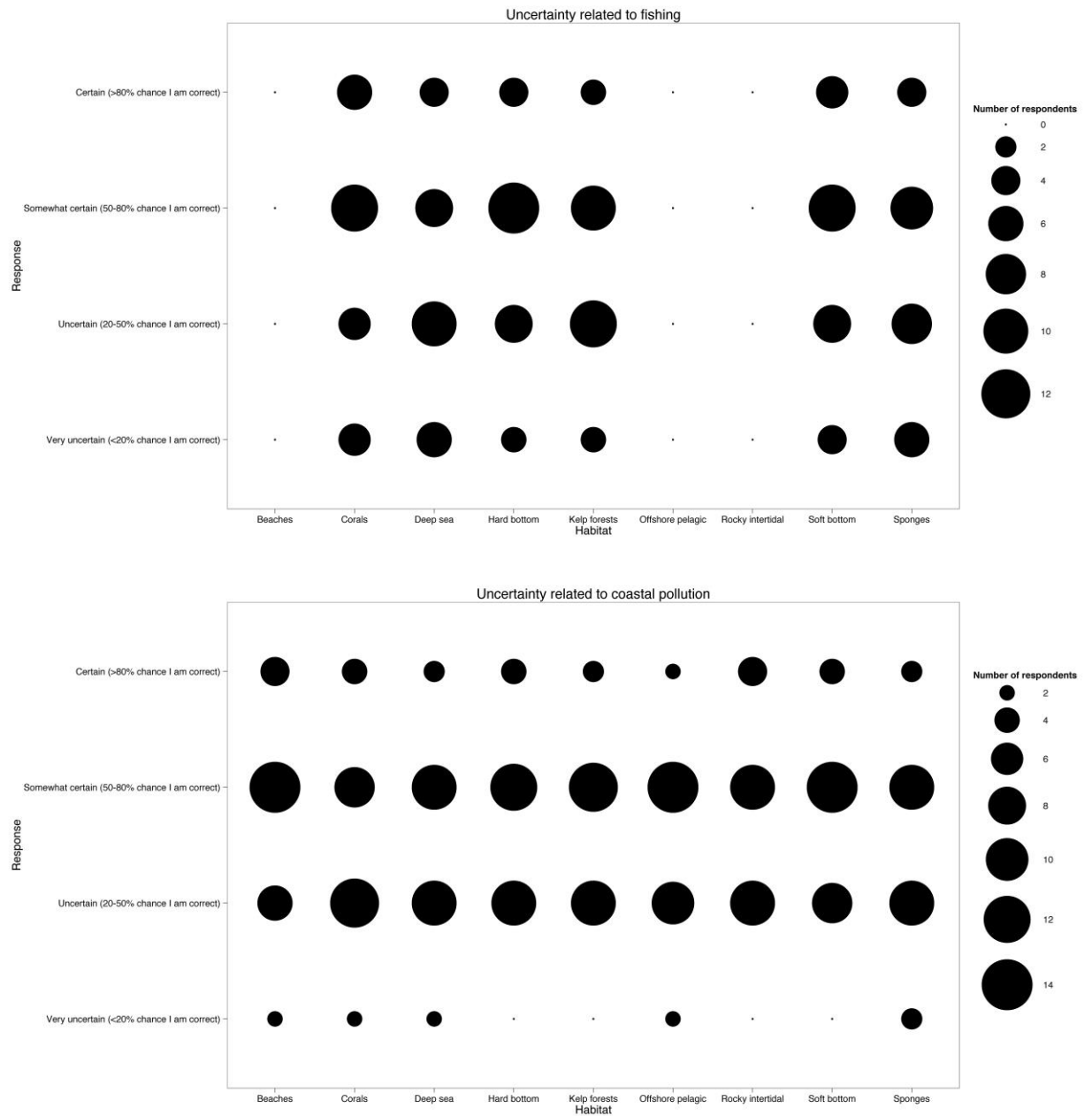


Figure EN.R.19. Expert self-assessment of uncertainty regarding responses related to risk to habitats within the MBNMS from bottom-tended fishing (top) and coastal pollution (bottom).



COMPARISON OF DATA- AND EXPERT-BASED RISK ASSESSMENTS

There was greater agreement between the data- and expert-based assessments for risk due to bottom-tended fishing, and greater disparities between the two assessments for risk due to coastal pollution (compare Figs. EN.R.20-21). Experts and data agreed remarkably well regarding risk to kelp forests from bottom-tended fishing and coastal pollution (Figs. EN.R.20-21). Lack of concordance between data- and expert-based assessments was most obvious for coral, hard bottom, and soft bottom habitats for both types of pressures, but it was not the case that risk to any of these three habitats was consistently over- or under-estimated by a specific method.

For coastal pollution (Fig. EN.R.20), experts generally perceived risk (Fig. EN.R.20c) to be higher in coral, hard bottom, offshore pelagic, and soft bottom habitats, and lower in beach habitats, than suggested by evidence in the data and literature we analyzed. In contrast, for bottom-tended fishing (Fig. EN.R.21), the data-driven assessment suggested that risk was greater for coral, soft bottom, and sponge habitats, and lower in hard bottom habitats, than suggested by the expert survey (Fig. EN.R.21c). For coastal pollution, the expert-based assessment tended to suggest greater sensitivity of habitats than the data-based assessment (Fig. EN.R.20b), but relatively lower exposure of three of the nearshore habitats (beaches, kelp forests, and rocky intertidal habitats; Fig. EN.R.20a). Exposure of corals and sponges to bottom-tended fishing was perceived to be lower by experts (Fig. EN.R.21a), and may have been overestimated in the data-driven assessment because all of the data on coral and sponge habitat locations came from trawl surveys (see Table EN.R.2). At this stage, it is not possible to say with certainty the cause of other discrepancies between these assessments, or which is closest to reality.

Figure EN.R.20. Discrepancies between data- and expert-based risk assessment for coastal pollution in MBNMS. (a) Exposure, (b) Sensitivity, (c) Risk. The line represents the 1:1 line, such that positive deviations indicate that expert-based assessment was greater than data-based assessment, and vice versa.

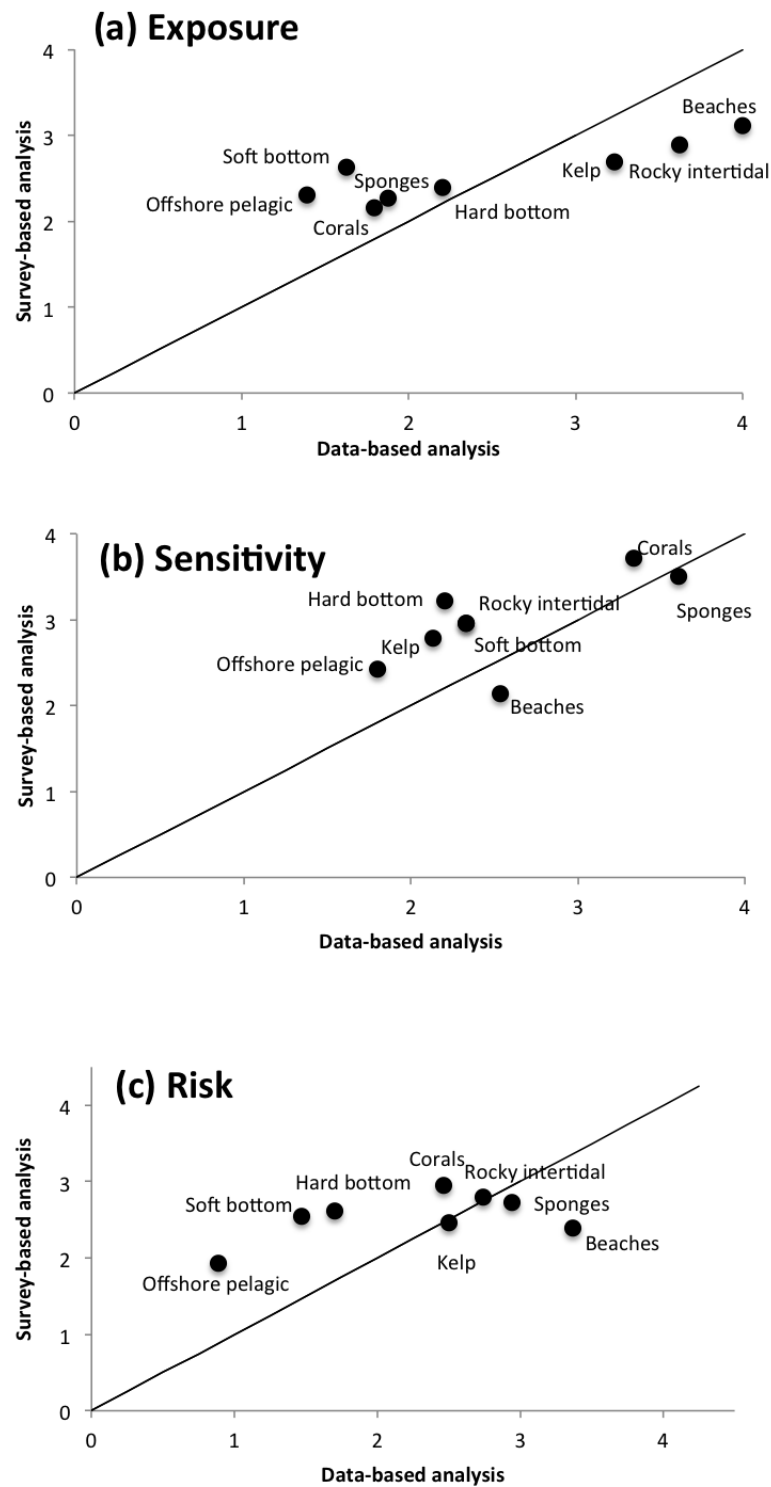
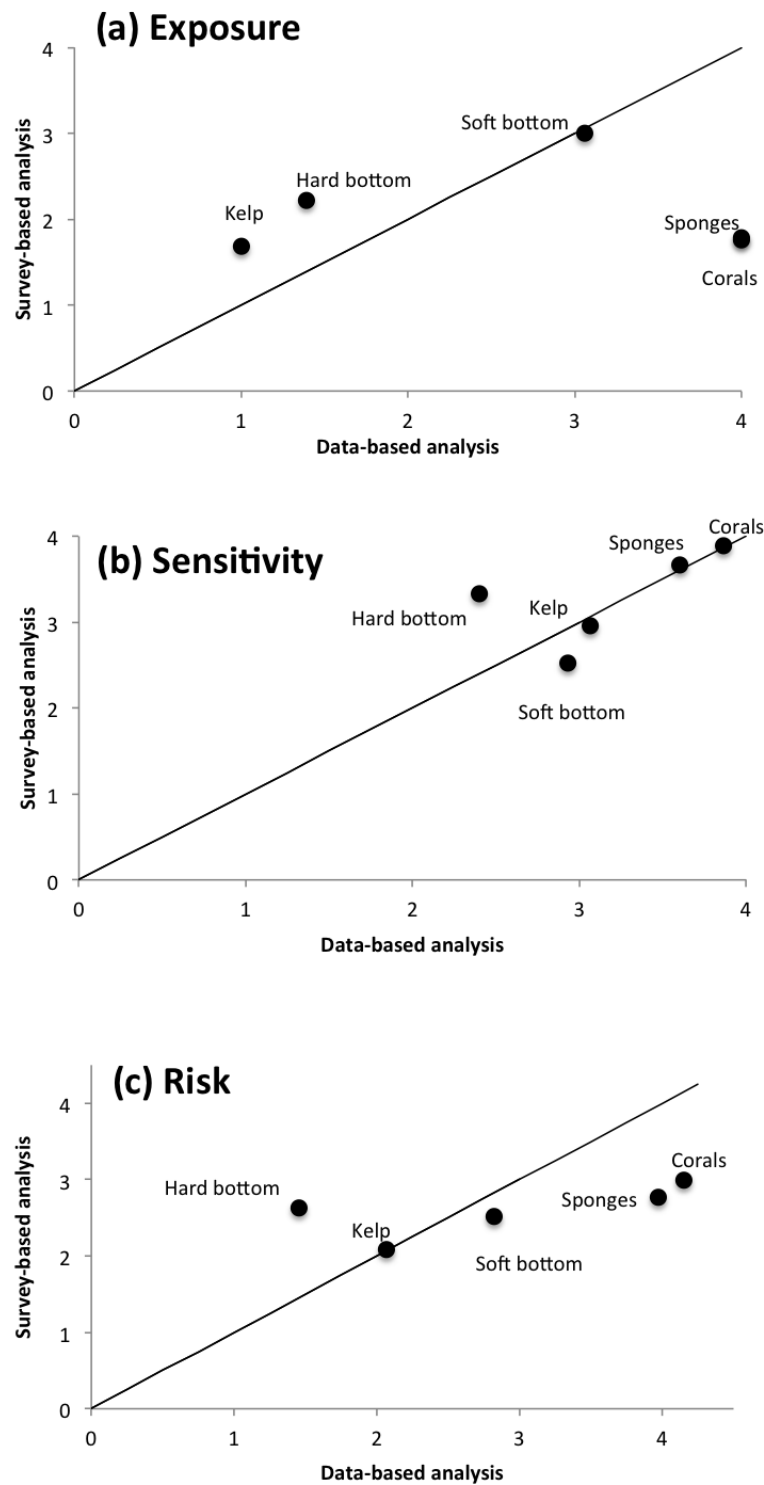


Figure EN.R.21. Discrepancies between data- and expert-based risk assessment for bottom-tended fishing in MBNMS. (a) Exposure, (b) Sensitivity, (c) Risk. The line represents the 1:1 line, such that positive deviations indicate that expert-based assessment was greater than data-based assessment, and vice versa.



CONCLUSIONS – RISK ASSESSMENT

This analysis of risk to marine habitats due to a variety of activities and pressures should provide a useful template for future iterations of the CCIEA. While it does not provide insight into the absolute risk to ecological integrity (e.g., the probability that a marine habitat will be completely destroyed or changed into an unrecognizable form), it does give a broad brush sense of which activities and pressures pose the greatest relative risk to individual habitats, and which habitats are at greatest relative risk from each activity/pressure (Table EN.R.11).

Table EN.R.11. Activities and pressures posing the greatest relative risk to individual habitats within Monterey Bay National Marine Sanctuary (also see Fig. EN.R.7). Results come from the data-based assessment.

| Habitat type | Greatest relative risk |
|-------------------------|---|
| Beaches | Sediment changes |
| Corals | Sea surface temperature changes, Sediment changes, Shipping |
| Hard bottom | Shipping, Sediment changes |
| Kelp forests | Sediment changes |
| Offshore pelagic waters | Shipping, Sea surface temperature changes |
| Rocky intertidal | Sediment changes |
| Seamounts | Sea surface temperature changes |
| Soft bottom | Shipping, Sea surface temperature changes |
| Sponges | Shipping, Sea surface temperature changes |

One apparent contradiction in the data-based risk assessment warrants discussion. Consideration of relative risk to each habitat from all pressures suggested that sea-based pressures tended to pose greater risk than land-based pressures (Fig. EN.R.8). At the same time, consideration of relative risk of different habitats to each pressure implied that nearshore habitats were at greater risk than offshore habitats (Fig. EN.R.10). However, it was not the case that risk due to sea-based pressures was greatest in nearshore habitats. Rather, sea-based pressures tended to generate greater exposure values for each habitat than did land-based pressures, so that risk from sea-based pressures was greater when each habitat was considered individually (Fig. EN.R.7). When the habitats were considered together, nearshore habitats tended to be relatively more exposed than offshore habitats across most pressures (Fig. EN.R.9). Regardless, the differences in risk due to land vs. sea-based pressures and in nearshore vs. offshore habitats were only statistically significant in a handful of cases.

Other studies, similar in kind to this one, have been conducted for the California Current. In the future, it will be useful to compare the analyses of cumulative impacts presented in Halpern et al. (2008, 2009) to this one. It will also be productive to determine the extent to which this risk assessment builds on and improves upon assessments of ecosystem condition within the MBNMS (e.g., (ONMS 2009)). In addition, it would be worth weighting risk scores by the importance of each habitat (e.g., where importance is based on habitat area, species richness, etc.) to generate ecosystem-level summary risk scores for the entire MBNMS. Finally, a variety of approaches have been established for integrating qualitative information (e.g., collected via expert elicitation) with quantitative data (Cheung et al. 2005, Teck et al. 2010, Kuhnert et al. 2010). We look forward to tackling new challenges in producing just such an integrated understanding of risk to ecological integrity in the California Current in the future.

LINKS TO DATA

[California Department of Fish and Game](#)

[Halpern et al. 2008](#)

[Halpern et al. 2009](#)

[NOAA ESI](#)

[National Centers for Coastal Ocean Science](#)

[Monterey Bay National Marine Sanctuary risk survey](#)

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MARINE MAMMALS RISK ASSESSMENT

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TABLE OF CONTENTS (MMR)

| | |
|--|-----|
| Executive Summary | 591 |
| Detailed Report..... | 594 |
| Description of ecosystem component..... | 594 |
| Data sources | 596 |
| Estimation of exposure | 600 |
| Estimation of sensitivity | 601 |
| Risk assessment results..... | 601 |
| Commercial Fishing Effort..... | 601 |
| Cetacean and Fishing Overlap Mapping | 606 |
| Blue whale..... | 606 |
| Fin whale..... | 606 |
| Baird’s beaked whale..... | 606 |
| Short-beaked common dolphin..... | 606 |
| Risso’s dolphin | 607 |
| Pacific white-sided dolphin..... | 607 |
| Northern right whale dolphin..... | 607 |
| Humpback whale..... | 607 |
| Dall’s porpoise..... | 607 |
| Sperm whale..... | 608 |

| | |
|-------------------------------|-----|
| Striped dolphin..... | 608 |
| Small beaked whales..... | 608 |
| Population Overlap Index..... | 621 |
| Cumulative Overlap Index..... | 621 |
| References cited | 626 |

LIST OF TABLES AND FIGURES (MMR)

| | |
|---|-----|
| Figure MMR-EX1. Left map: modeled humpback whale mean density expressed as the number of individuals/yr/km ² . Three narrow maps: overlap values for humpback whale with the fixed, hake and trawl fleets..... | 592 |
| Figure MMR-EX2. Modeled proportion (upper) and cumulative exposure index (lower) of each cetacean species population that overlapped with each of the three commercial fishing fleets (from 2002-2009), for each of the 12 cetacean species. B ba = Baird's beaked whale; B mu = blue whale; B ph = fin whale; D de = short-beaked common dolphin; G gr = Risso's dolphin; L bo = northern right whale dolphin; L ob = Pacific white-sided dolphin; M no = humpback whale; P da = Dall's porpoise; P ma = sperm whale; S co = striped dolphin; and, Zsm = small beaked whales..... | 593 |
| Table MMR1. Twelve species of cetaceans represented in predicted cetacean density geospatial datalayer (Barlow and Forney 2007, Barlow et al. 2009)..... | 595 |
| Table MMR2. Fixed gear fishing effort represented in West Coast Groundfish Observer Program (WCGOP) data by sector observed; including the proportion of total observed effort (cumulative hours gear was deployed) by sector from 2002-2009, the observed sector coverage rate calculated as the observed retained catch weight of target species divided by the fleet-wide landed weight of target species, and the assumed proportion of total fleet-wide effort represented in the observed data. | 597 |
| Figure MMR1. Interannual trends in fishing effort, expressed as cumulative number of hours per year (June through November months, 2002-2009) fishing gear was deployed in the water for each of the three fleet types. | 602 |
| Figure MMR2. Monthly trends in fishing effort, expressed as cumulative number of hours per month (from 2002-2009) fishing gear was deployed in the water for each of the three fleet types. Panel A = fixed; Panel B = hake; and, Panel C = trawl. | 604 |
| Figure MMR3. Patterns of fishing effort along the west coast of the United States, expressed as cumulative number of hours per gridcell (all months from 2002-2009) fishing gear was deployed in the water for each of the three fleet types. | 605 |
| Figure MMR4. Left map: modeled blue whale mean density expressed as the number of individuals/yr/km ² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for blue whale with the fixed, hake and trawl fleets..... | 609 |
| Figure MMR5. Left map: modeled fin whale mean density expressed as the number of individuals/yr/km ² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for fin whale with the fixed, hake and trawl fleets..... | 610 |
| Figure MMR6. Left map: modeled Baird's beaked whale mean density expressed as the number of individuals/yr/km ² (based on survey data collected from 1991 – 2005) within the EEZ off the west | |

coast of the United States. Three narrow maps: overlap values for Baird's beaked whale with the fixed, hake and trawl fleets. 611

Figure MMR7. Left map: modeled short-beaked common dolphin mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for short-beaked common dolphin with the fixed, hake and trawl fleets. 612

Figure MMR8. Left map: modeled Risso's dolphin mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for Risso's dolphin with the fixed, hake and trawl fleets. 613

Figure MMR9. Left map: modeled Pacific white sided dolphin mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for Pacific white sided dolphin with the fixed, hake and trawl fleets. 614

Figure MMR10. Left map: modeled Northern right whale dolphin mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for Northern right whale dolphin with the fixed, hake and trawl fleets. 615

Figure MMR11. Left map: modeled humpback whale mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for humpback whale with the fixed, hake and trawl fleets. 616

Figure MMR12. Left map: modeled Dall's porpoise mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for Dall's porpoise with the fixed, hake and trawl fleets. 617

Figure MMR13. Left map: modeled sperm whale mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for sperm whale with the fixed, hake and trawl fleets. 618

Figure MMR14. Left map: modeled striped dolphin mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for striped dolphin with the fixed, hake and trawl fleets. 619

Figure MMR15. Left map: modeled small beaked whales mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for small beaked whales with the fixed, hake and trawl fleet. 620

Figure MMR16. Modeled proportion (upper) and cumulative exposure index (lower) of each cetacean species population that overlapped with each of the three commercial fishing fleets (from

2002-2009), for each of the 12 cetacean species. B ba = Baird's beaked whale; B mu = blue whale; B ph = fin whale; D de = short-beaked common dolphin; G gr = Risso's dolphin; L bo = northern right whale dolphin; L ob = Pacific white-sided dolphin; M no = humpback whale; P da = Dall's porpoise; P ma = sperm whale; S co = striped dolphin; and, Zsm = small beaked whales623

Figure MMR17. Cumulative annual commercial fishing fleet overlap indices (from 2002-2009) for each of the 12 cetacean species. Panels A, B, and C are the fixed, hake and trawl fleets, respectively. B ba = Baird's beaked whale; B mu = blue whale; B ph = fin whale; D de = short-beaked common dolphin; G gr = Risso's dolphin; L bo = northern right whale dolphin; L ob = Pacific white-sided dolphin; M no = humpback whale; P da = Dall's porpoise; P ma = sperm whale; S co = striped dolphin; and, Zsm = small beaked whales.624

OVERVIEW

Overall, it is clear that groundfish fishing fleets overlap with various cetaceans found in the CCLME, but it is unclear whether or not this overlap presents a substantial risk to the viability of these cetacean populations.

EXECUTIVE SUMMARY

- Many cetacean populations worldwide are confronted by a multitude of anthropogenic threats. Commercial whaling, ambient ocean noise, vessel collisions, gear entanglement, resource competition, habitat disturbance and global climate change are examples of some of these threats. There is substantial evidence in the literature that cetaceans are sensitive to many of the aforementioned threats imposed by commercial fishing activities. However, few studies have addressed the potential vulnerability of a given cetacean species to an entire fishing fleet operating over a large marine ecosystem. Further, there is a paucity of data on inter-specific and -fleet overlap of commercial fishing activities on cetaceans.
- In this report, we overlaid spatially explicit predicted mean annual density of 12 cetacean species within the CCLME with observer based West Coast Groundfish Fishery (WCGF) commercial fishing effort data for fixed-gear, at-sea hake midwater trawl, and bottom trawl fleets. We quantified the vulnerability of each species to each fleet type by multiplying the predicted mean annual cetacean density by the measured fishing fleet effort (in hours) from 2002-2009 (see Figure MMR-EX1 for example map for humpback whales).
- We found that there was enormous interspecific and interfleet variability in the overlap between cetaceans and fishing fleets (Figure MMR-EX2) and this variability was not consistent over time. While many of the species had relatively low overlap rates, others had significant exposure to some of the fishing fleets. While there is not a lot of evidence of direct mortality from these fleets, our results suggest there is substantial opportunity for sublethal affects on some cetacean species.
- Our analyses are an important first step in generating formal risk assessments for quantifying the population impacts of various fishing fleets on cetaceans living in the CCLME.

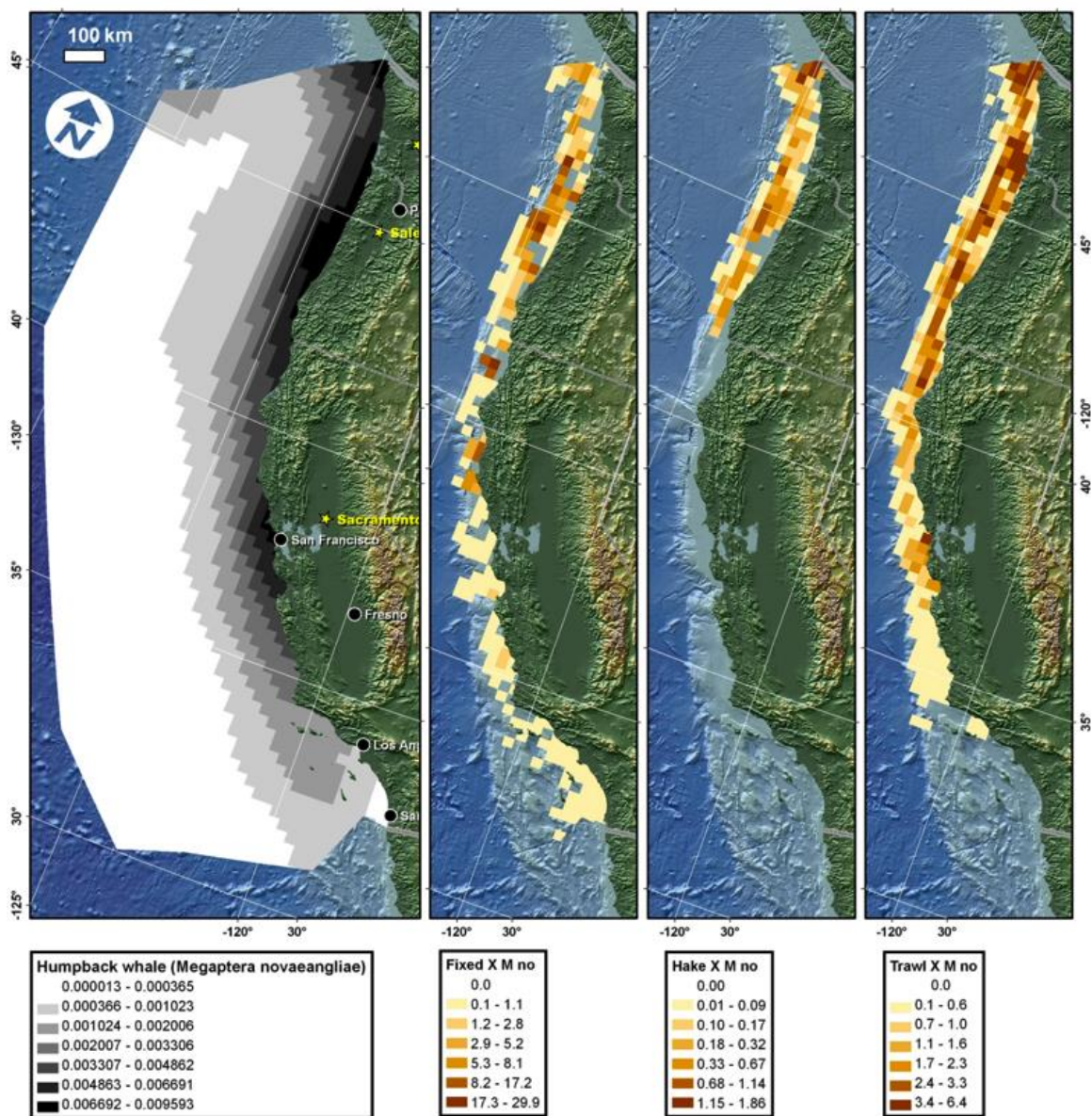


Figure MMR-EX1. Left map: modeled humpback whale mean density expressed as the number of individuals/yr/km². Three narrow maps: overlap values for humpback whale with the fixed, hake and trawl fleets

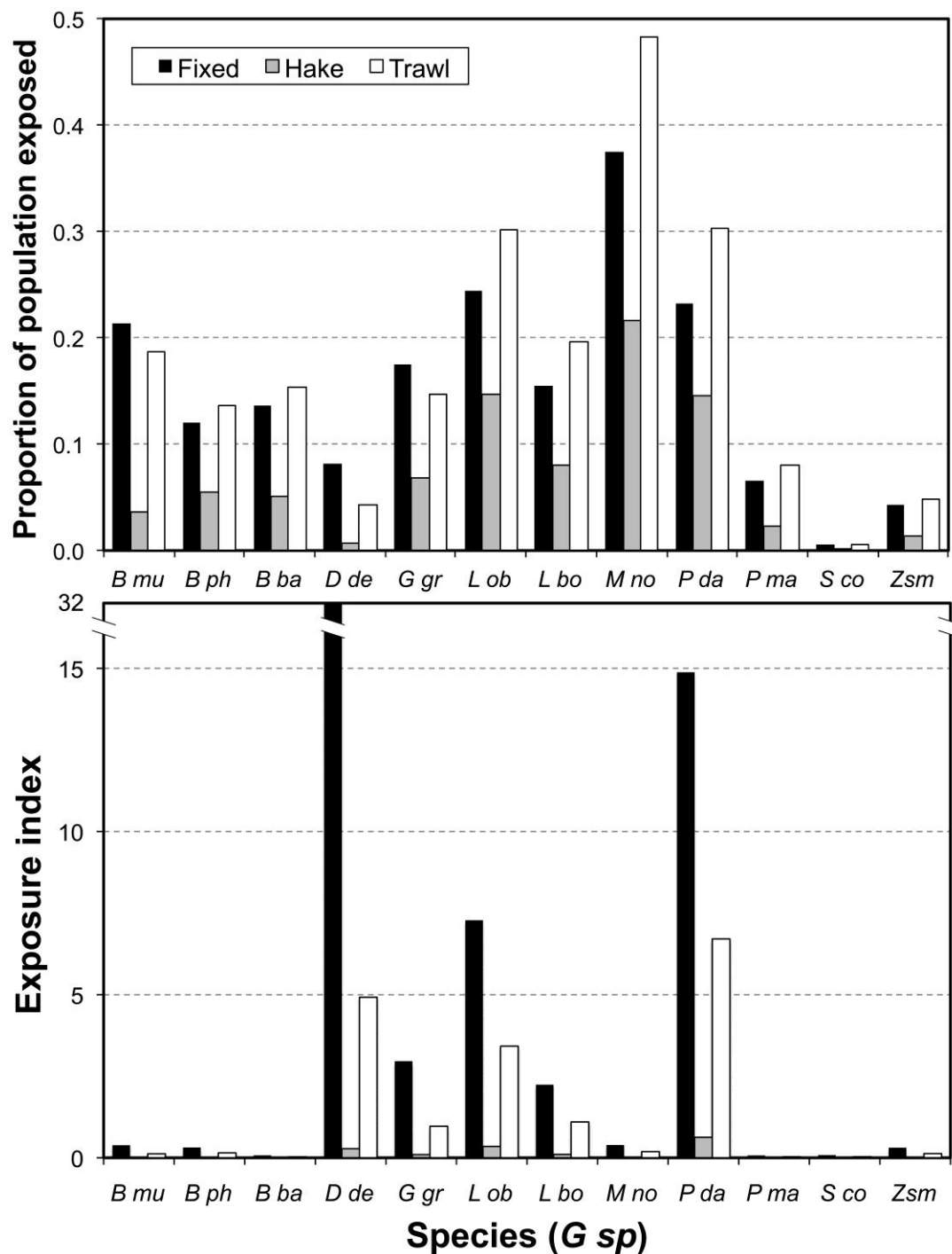


Figure MMR-EX2. Modeled proportion (upper) and cumulative exposure index (lower) of each cetacean species population that overlapped with each of the three commercial fishing fleets (from 2002-2009), for each of the 12 cetacean species. B ba = Baird's beaked whale; B mu = blue whale; B ph = fin whale; D de = short-beaked common dolphin; G gr = Risso's dolphin; L bo = northern right whale dolphin; L ob = Pacific white-sided dolphin; M no = humpback whale; P da = Dall's porpoise; P ma = sperm whale; S co = striped dolphin; and, Zsm = small beaked whales

DESCRIPTION OF ECOSYSTEM COMPONENT

This report describes a risk assessment that we ran on various cetacean species distributed in the CCLME. There are four cetacean species that are endangered and protected by the Endangered Species Act (ESA), and the other eight species are protected under the Marine Mammal Protection Act (MMPA, see Table MMR1).

The goal of this assessment was twofold. First, map and quantify the general patterns of overlap between the 12 species of cetaceans and three of the major groundfish fishing fleets operating in the CCLME. Second, map and quantify interspecific and interfleet differences in the overlap spatio-temporal patterns of overlap. From this we quantified the potential overlap (a proxy for vulnerability) for each cetacean species. The risk imposed by various groundfish fishing fleets is poorly understood in the CCLME. Given that cetaceans are a protected species and are likely key players in marine interaction webs (CIESM 2004, Paine 2006), it is important to include them in any IEA. The status of these cetacean stock ranges from unknown to endangered (Carretta et al. 2011), so it seems prudent to include these long-lived animals with low intrinsic population growth rates in this IEA.

Cetaceans around the world face a myriad stresses on their populations. Commercial whaling was once the primary threat to many cetaceans, but with the international ban on numerous whaling operations, and the Marine Mammal Protection Act (MMPA) many populations have rebounded. Nevertheless, commercial whaling activities continue in some areas and numerous lethal and sublethal anthropogenic threats to the viability of cetaceans persist. The list includes, but is not limited to, anthropogenic stress (Curry 1999, Fair and Becker 2000), vessel collisions (Panigada et al. 2006), noise (Committee on Potential Impacts of Ambient Noise in the Ocean on Marine Mammals 2003, Romano et al. 2004), exposure to toxins (hydrocarbons, exhaust, etc. (Jarman et al. 1996, Marsili et al. 2001)), entanglement with fishing gear (Read et al. 2006) and marine debris (Williams et al. 2011), resource competition and habitat disturbance from fishing (Dayton et al. 1995, DeMaster et al. 2001, Herr et al. 2009), and global climate change (MacLeod 2009).

There is substantial evidence in the literature documenting direct mortality of various cetaceans from interactions with commercial and recreational fishing gear (Read et al. 2006). For example, sperm whales (*Physeter macrocephalus*), are especially susceptible to deepwater gillnets and bottom-set longline gear (Di Natale and Notarbartolo di Sciara 1994, Haase and Felix 1994, Félix et al. 1997, Hill et al. 1999, Straley et al. 2005). They have been observed breaking through or carrying away fishing gear and may die or are seriously injured as a result. There has been considerable effort to reduce the mortality of commercial fishing activities on cetaceans (e.g., pingers on gillnets (Barlow and Cameron 2003)). However, there is plenty of opportunity for significant sublethal and injurious consequences from exposure to commercial gear of all types, and this type of interaction is poorly documented and understood.

To date, there have not been any spatial analyses run on the overlap between a multiple cetacean species (some of which are ESA/IUCN listed) and fishing fleets operating in the California Current Ecosystem. While reviews of the literature suggest cetacean mortality due to fishing gear interaction is low, there is a significant exposure rate and a better understanding of the spatio-temporal overlap dynamics (magnitude, seasonality and frequency) seems prudent. Therefore, it is useful to quantify the potential for overlap between commercial fishing activities and cetaceans. Moreover, comparing interspecific exposure rates to various fishing gear types may facilitate a better understanding of the risks imposed by commercial fishing activities on cetacean species.

Table MMR1. Twelve species of cetaceans represented in predicted cetacean density geospatial datalayer (Barlow and Forney 2007, Barlow et al. 2009).

| Cetacean | Suborder | Family | ESA Status | IUCN |
|---|----------------------|-----------------------------|-------------------|----------------|
| Baird's beaked whale (<i>Berardius bairdii</i>) | Odontoceti (toothed) | Ziphiidae (beaked) | | Data Deficient |
| Blue whale (<i>Balaenoptera musculus</i>) | Mysticeti (baleen) | Balaenopteridae | Endangered | EN |
| Fin whale (<i>B. physalus</i>) | Mysticeti (baleen) | Balaenopteridae | Endangered | EN |
| Short-beaked common dolphin (<i>Delphinus delphis</i>) | Odontoceti (toothed) | Delphinidae (dolphins) | | LC |
| Risso's dolphin (<i>Grampus griseus</i>) | Odontoceti (toothed) | Delphinidae (dolphins) | | LC |
| Northern right whale dolphin (<i>Lissodelphis borealis</i>) | Odontoceti (toothed) | Delphinidae (dolphins) | | LC |
| Pacific white-sided dolphin (<i>Lagenorhynchus obliquidens</i>) | Odontoceti (toothed) | Delphinidae (dolphins) | | LC |
| Humpback whale (<i>Megaptera novaeangliae</i>) | Mysticeti (baleen) | Balaenopteridae | Endangered | LC |
| Dall's porpoise (<i>Phocoenoides dalli</i>) | Odontoceti (toothed) | Phocoenidae (porpoises) | | LC |
| Sperm whale (<i>Physeter macrocephalus</i>) | Odontoceti (toothed) | Physeteridae (sperm whales) | Endangered | VU |
| Striped dolphin (<i>Stenella coeruleoalba</i>) | Odontoceti (toothed) | Delphinidae (dolphins) | | LC |
| Small beaked whales (<i>Ziphius</i> and <i>Mesoplodon</i>) | Odontoceti (toothed) | Ziphiidae (beaked) | | LC |

EN = endangered; LC = least concern; VU = vulnerable;

DATA SOURCES

We overlaid two different geospatial datalayer types for these analyses: modeled cetacean density and commercial fishing effort. We compared general patterns of effort by three different commercial fleets by gear type (bottom trawl, at-sea hake midwater trawl and fixed gear fleets) with general patterns of 12 cetacean species density throughout the California Current Large Marine Ecosystem (CCLME).

CETACEAN DATA

We used estimates of cetacean density based on habitat models that were generated by the National Oceanic and Atmospheric Administration (NOAA), Southwest Fisheries Science Center for an approximate 1,141,800 km² study area off the U.S. west coast (Barlow et al. 2009, Forney et al. 2012). They used data from four systematic ship-based cetacean and ecosystem assessment surveys conducted in summer and fall of 1991–2001 to build habitat-based density models for 11 species and one species guild. Models were built for striped dolphin (*Stenella coeruleoalba*), short-beaked common dolphin (*Delphinus delphis*), Risso's dolphin (*Grampus griseus*), Pacific white-sided dolphin (*Lagenorhynchus obliquidens*), northern right whale dolphin (*Lissodelphis borealis*), Dall's porpoise (*Phocoenoides dalli*), sperm whale, fin whale (*Balaenoptera physalus*), blue whale (*B. musculus*), humpback whale (*Megaptera novaeangliae*), Baird's beaked whale (*Berardius bairdii*), and a small beaked whale guild (including Cuvier's beaked whale, *Ziphius cavirostris*, and beaked whales of the genus *Mesoplodon*). Four of these species are listed as Endangered under the U.S. Endangered Species Act (Table MMR1). Generalized additive models (GAMs) were used to predict cetacean densities from habitat variables that included remotely sensed measures of sea surface temperature (SST) and the coefficient of variation of SST (to serve as a proxy for frontal regions); sea surface salinity, mixed layer depth (the depth at which temperature is 0.5°C less than surface temperature), and sea surface chlorophyll collected *in situ* during the surveys; and, water depth, bathymetric slope, and distance to the 2000 m isobaths (Barlow et al. 2009, Forney et al. 2012). Model validation was performed on a novel data set (2005), and selected models were then re-fit to the complete set of 1991–2005 data. Predicted densities for each of the five individual years (1991, 1993, 1996, 2001, and 2005) were smoothed and then averaged to produce a composite grid that represents the best estimate of average cetacean density and distribution over the past 15 years. The grids were created at a resolution of approximately 25 km and covered most of the CCLME off the coast of Washington, Oregon, and California. The predicted multi-year average densities (number of animals per km²) were used for our analyses.

COMMERCIAL FISHING EFFORT

Fishing effort was represented on either 10 km (bottom trawl fleets [herein trawl] and at-sea hake midwater trawl [herein hake] fleets) or 20 km (fixed gear fleets [herein fixed]) grids. We used data that were provided by the At-sea Hake Observer Program (A-SHOP) and the West Coast Groundfish Observer Program (WCGOP) under NOAA's Northwest Fisheries Science Center, Fishery Resource Analysis and Monitoring (FRAM) Division.

At-sea hake midwater trawl fishing effort was collected directly by the A-SHOP (National Oceanic and Atmospheric Administration 2011). The A-SHOP collects information on total catch (fish discarded and retained) from all vessels that process Pacific hake at-sea. All data were collected according to standard protocols and data quality control established by the ASHOP.

Bottom trawl fishing effort (National Oceanic and Atmospheric Administration 2010) was derived by the FRAM Division from fleet-wide logbook data submitted by state agencies to the Pacific Fisheries Information Network (PacFIN) regional database, maintained by the Pacific States Marine Fisheries Commission (PSMFC). A common-format logbook is used by Washington, Oregon, and California. Electronic logbook data is submitted by state agencies to the PacFIN regional database. Trawl logbook data is regularly used in analyses of the bottom trawl groundfish fishery observed by the WCGOP.

For both the trawl and hake survey data, a trawl towline model (line drawn from the start to end location of a trawl tow) was used to allocate data to 10 x 10 kilometer grid cells for calculation of commonly used fishing effort metrics.

Fixed gear fishing effort was collected directly by the WCGOP from the following fixed gear sectors: the limited entry sablefish primary (target – sablefish), limited entry non-sablefish endorsed (target – groundfish), open access fixed gear (target – groundfish), and Oregon and California state-permitted nearshore fixed gear (target – nearshore groundfish). The observed portion of overall fixed gear varies by coverage level in each sector (Table MMR2). Coverage rates are calculated for each sector as the observed retained catch of target species divided by the sector-wide landings of target species. Since all fishing operations are not observed, neither the maps nor the data can be used to characterize the fishery completely. Both the observed fixed gear set (start location of fishing) and haul (location of gear retrieval) were assigned to 20 x 20 kilometer grid cells for calculation. The fishing effort associated with each fixed gear fishing event was divided equally between the set and haul locations. Commonly used fishing effort metrics were then calculated for each grid cell.

Table MMR2. Fixed gear fishing effort represented in West Coast Groundfish Observer Program (WCGOP) data by sector observed; including the proportion of total observed effort (cumulative hours gear was deployed) by sector from 2002-2009, the observed sector coverage rate calculated as the observed retained catch weight of target species divided by the fleet-wide landed weight of target species, and the assumed proportion of total fleet-wide effort represented in the observed data.

| Sector (2002-2009) | % of Total Duration by Sector | Sector Coverage Rate | Proportion of Duration Represented |
|--|--|-------------------------------------|---|
| Limited Entry Sablefish Primary | 59.38% | 26.12% | 15.51% |
| Limited Entry Non-Tier-Endorsed Fixed Gear | 17.00% | 7.41% | 1.26% |
| Open Access Fixed Gear | 18.63% | 3.00% | 0.56% |

| | | | |
|---|-------|-------|-------|
| Oregon Nearshore Fixed Gear | 3.83% | 5.20% | 0.20% |
| California Nearshore Fixed Gear | 1.16% | 3.43% | 0.04% |
| Sum total percentage of duration represented = 17.57% | | | |

There are a variety of fixed gear types recorded by WCGOP, and we used the types that we deemed most likely (based on reviews of the literature) to cause harm to a cetacean, should an individual encounter that gear type. The types we used included: historic longline, vertical hook and line, other hook and line, pot, and longline (fixed hook), longline (snap gear). We decided that both poll and troll gear did not pose a significant risk to the cetaceans in this analysis, so those two gear types were excluded from the analyses.

Fishing effort was expressed as the cumulative number of hours a given fishing fleet (trawl, hake, or fixed) had gear deployed in the water. All of the fishing effort data were reported as monthly sums for each fishing gear type, so we calculated cumulative fishing effort (in hours) from June through November of each year, which corresponded to the months over which the data were collected for building the predictive cetacean model.

For the hake and trawl fleets, the data represents all (100%) of the total fishing effort. All at-sea hake vessels (catcher-processors and motherships) over 125 feet are required to carry two observers, while vessels under 125 feet carry only one. PacFIN fleet-wide logbook data is assumed to represent the entire bottom trawl fleet for our analysis. However, all fishing operations may not necessarily be recorded in logbooks and logbook submission may not be complete. For the fixed gear fleet, observers are not present on every vessel, so we calculated a correction factor (C) in order to extrapolate the effort of the entire fixed gear fleet. Catch data are reported on an annual basis, so we ran the calculation across all years (2002-2009) by multiplying the data reported for each sector by the proportion that that sector represented over the entire study area. We used the following formula to make the calculation:

$$C = \sum_{s=1}^5 \left(\frac{t_s}{T} \times \frac{W_{s(obs)}}{W_{s(land)}}$$

where s corresponded to each of the five sectors, t was the total time (in hours) a given sector was observed with gear in the water, T was the total time (in hours) all five of the sectors were observed with gear in the water, w was the total weight of fish caught on vessels with observers present (reported by sector) and W was the total weight of fish landed on all vessels (reported by sector).

The commercial fishing effort data are subject to restrictions that preserve confidentiality as required under the Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006. As such, data cannot be presented to the general public unless it represents information from three or more vessels. We ran all of the analyses in our research on the full set of fishing fleet data. However, in order to comply with confidentiality restrictions, gridcells in the final overlap maps that contained data from two or fewer boats are not displayed in this paper.

CETACEAN AND FISHERY OVERLAP

We created overlap index maps (annually and from 2002-2009) for each of the cetacean species as well as overlap index plots by year, which showed interannual variability in the overlap between the species and fleets. We also calculated the population overlap for each species with each of the three fleet types as well as a cumulative overlap index.

We used a simple formula to calculate a predicted overlap index (R , animal hours/km²):

$$R = t * \rho$$

where t is fishing effort (total time, in hours, gear was in the water), and ρ is the predicted density of cetaceans (animals/km²).

MAPS

We calculated the overlap indices for each year (2002 – 2009) and for each of the species and fleet type combinations (12 X 3 = 36) throughout the study area. Since the gridcell size of the cetacean data (~25 km) was not the same as the fishing effort data (10 or 20 km), we calculated an area weighted mean cumulative fishing effort for each year that corresponded to each respective cetacean gridcell. First, we combined the cetacean grid with the three fishing fleet grids using the INTERSECT command in ArcGIS (v. 9.3), a geographic information (GIS) software package developed by the Environmental Systems Research Institute (ESRI). Then, we used the information from this intersection to calculate an area weighted mean (AWM) fishing effort for each cetacean gridcell using the following equation:

$$t_{awm} = \left[\sum_1^n t_n(a_n) \right] / A$$

where t is the fishing effort in hours for a given portion of a given cetacean gridcell, a is the corresponding area for that effort and A is the total area of the corresponding cetacean gridcell. We repeated this procedure for each year (2002-2009) of the fishing fleet data.

Finally, we multiplied the AWM fishing effort, t , for each gridcell by the corresponding cetacean density (ρ), which yielded the final overlap index value. We used ArcGIS to join the corresponding predicted overlap index for each species and gear type combination to the original cetacean density grid in order to create 36 gridded maps, which we used to explore spatiotemporal patterns of cetacean and fishing fleet overlap.

DEFINITION OF RISK

We defined risk as the sensitivity of each of the 12 cetacean species to each of the three groundfish fishery fleets, if exposed. Our analyses focused on quantifying exposure and did not account for sensitivity. We will incorporate sensitivity in future IEA efforts.

ESTIMATION OF EXPOSURE

In order to estimate exposure, we multiplied the cumulative hours of fishing by the density of cetaceans (for each 25 x 25 km gridcell) for all cetacean species/fishing fleet combinations. We measured exposure in two different ways: a population overlap index, which represented what fraction of the population was exposed to each of the three fleets; and, a cumulative overlap index, which yielded spatially explicit exposure levels for each of the 25 x 25 km gridcells.

POPULATION OVERLAP INDEX

In order to compare inter- specific and fishery overlap relative to all of the modeled individuals in a given species, we calculated what fraction of each cetacean species' modeled population overlapped with areas where commercial fishing occurred using:

$$R_p = \sum_1^n \rho_x(a_n) / \sum_1^n P_u(a_n)$$

where ρ is the modeled cetacean density for a given gridcell that experienced commercial fishing by a given fleet, a is the area of the corresponding gridcell, and P is the modeled cetacean density for a given gridcell, regardless of whether or not that gridcell experienced commercial fishing from any of the fleets.

CUMULATIVE OVERLAP INDEX

We calculated a cumulative overlap index over the entire study area for each cetacean species/fishing fleet combination, by year and for all years from 2002-2009 using the following equation:

$$R_c = \sum_1^n R(a_n) / A$$

where R is the predicted overlap index for a given 25 km gridcell, a_n is the area of the corresponding gridcell, and A is the total area over which a given fleet operated. This allowed us to compare patterns of inter- specific, annual, and fishery overlap.

ESTIMATION OF SENSITIVITY

We did not quantify sensitivity of the cetaceans to any of the fishing fleets. However, we intend to incorporate sensitivity in future IEA products as resources and information become available. Currently, there is limited information on the sensitivity of various cetacean species to the commercial fishing fleets that we analyzed, and incorporating that information was beyond the scope of these analyses.

RISK ASSESSMENT RESULTS

COMMERCIAL FISHING EFFORT

Overall, spatial and temporal patterns of fishing effort varied widely over the study area. The cumulative level of effort during the months of June through November from 2002 – 2009 for the fixed, hake and trawl fleets was 187,015; 24,132; and, 287,886 hours, respectively.

For the fixed gear fleet, the effort captured by observers varied across sectors (Table MMR2). In general, observers captured approximately 17.57% of the total fixed gear effort (as a function of the cumulative hours gear was deployed) that occurred over the entire study area, based on the 2002-2009

proportion of effort from each observed sector and the WCGOP coverage rate of fishery landings by sector for all years combined. WCGOP coverage rates are calculated as the observed retained weight of target species divided by the fleet-wide landed weight of target species for each sector. Therefore, the overlap indices for fixed gear fishing with the various cetacean species are likely at least five times as large as the values we presented. However, this underrepresentation would not alter the proportion of each population that overlapped with the observed fixed gear fleet. We did not correct our overlap indices to account for this underrepresentation, as we do not have information about the spatial consistency of this deficiency.

INTERANNUAL PATTERNS

Cumulative annual effort varied considerably over time for each of the fleets (Figure MMR1). Observed fixed gear cumulative efforts had peaks in 2003 and 2005, with a downward trend from 2005 to 2009 (Figure MMR1). The hake fleet gradually increased in cumulative effort level until 2008 and dropped down again in 2009 (Figure MMR1). The trawl fleet had a drop in cumulative annual fishing effort in 2004, but returned to 2002 levels of effort by 2009 (Figure MMR1).

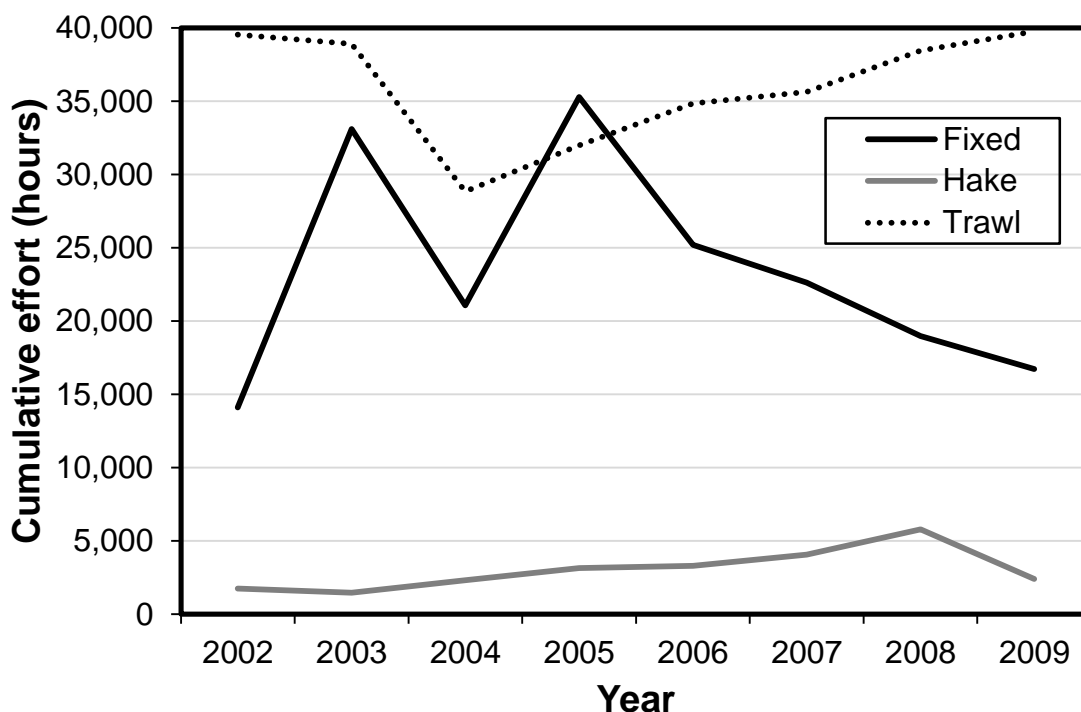


Figure MMR1. Interannual trends in fishing effort, expressed as cumulative number of hours per year (June through November months, 2002-2009) fishing gear was deployed in the water for each of the three fleet types.

MONTHLY INTER- AND INTRAANNUAL PATTERNS

There was considerable inter- and intraannual, and inter-fishery variability in the cumulative effort, based on the monthly data (Figure MMR2). The observed fixed gear fleets had the greatest interannual and intraannual variability in effort. This fleet generally had peak efforts during the summer months (Figure MMR2-A). However, there was usually a second peak of effort in the fall (Figure MMR2-A). Effort was lowest during the months of January, February, November and December (Figure MMR2-A). The hake fleet had the least interannual but the greatest intraannual variability in effort. The hake fleet does not fish from January to April each year, but they clearly have their maximum effort in May and June, with a smaller peak often occurring in the late fall (Figure MMR2-B). The trawl fleet had higher interannual but moderate intraannual variability in effort. The trawl fleet generally has considerable and consistent effort year round, but it tends to taper towards the end of the year (Figure MMR2-C). In 2002, however, there was a strong peak of effort from October through November.

SPATIAL AND TEMPORAL PATTERNS

There was considerable inter-fishery variability in the spatial extent of cumulative effort (Figure MMR3). For the period 2002-2009, various observed fixed gear efforts occurred from the US/Mexico border, north to the US/Canada border (Figure MMR3). There were concentrations of effort off the California coastal areas of Los Angeles, San Diego, Caspar, Eureka, and the northern half of the Oregon coast (Figure MMR3). The patchy distribution of the observed fixed gear fleet is assumed to be representative of overall fishing patterns, but there is a lack of logbook or other data sources to corroborate fleet-wide spatial distribution patterns. Hake fishing efforts occurred over a much smaller region, spanning Oregon and Washington (Figure MMR3). The hake fleet was not as patchy compared with the observed fixed gear fleet, but there were areas of increased effort (Figure MMR3). The trawl fleet efforts were not quite as widespread as the observed fixed gear fleet, occurring consistently from Point Conception, CA, north to the US/Canada border (Figure MMR3). Like the hake fleet, effort was more consistent along the range of activity.

Interannual spatial variability was greatest and most patchy for the observed fixed gear fleet (figures unavailable due to confidentiality restrictions). In some years (e.g., 2002), large expanses, 100s of kms or more, had no effort whatsoever. The hake fleet also became more patchy when examined on an annual basis, but there were few large areas that were unexploited in a given year (figures unavailable due to confidentiality restrictions). The trawl fleet had the most consistent efforts over space and time of the three gear types (figures unavailable due to confidentiality restrictions). However, there was still considerable interannual variability between various 10 km gridcells.

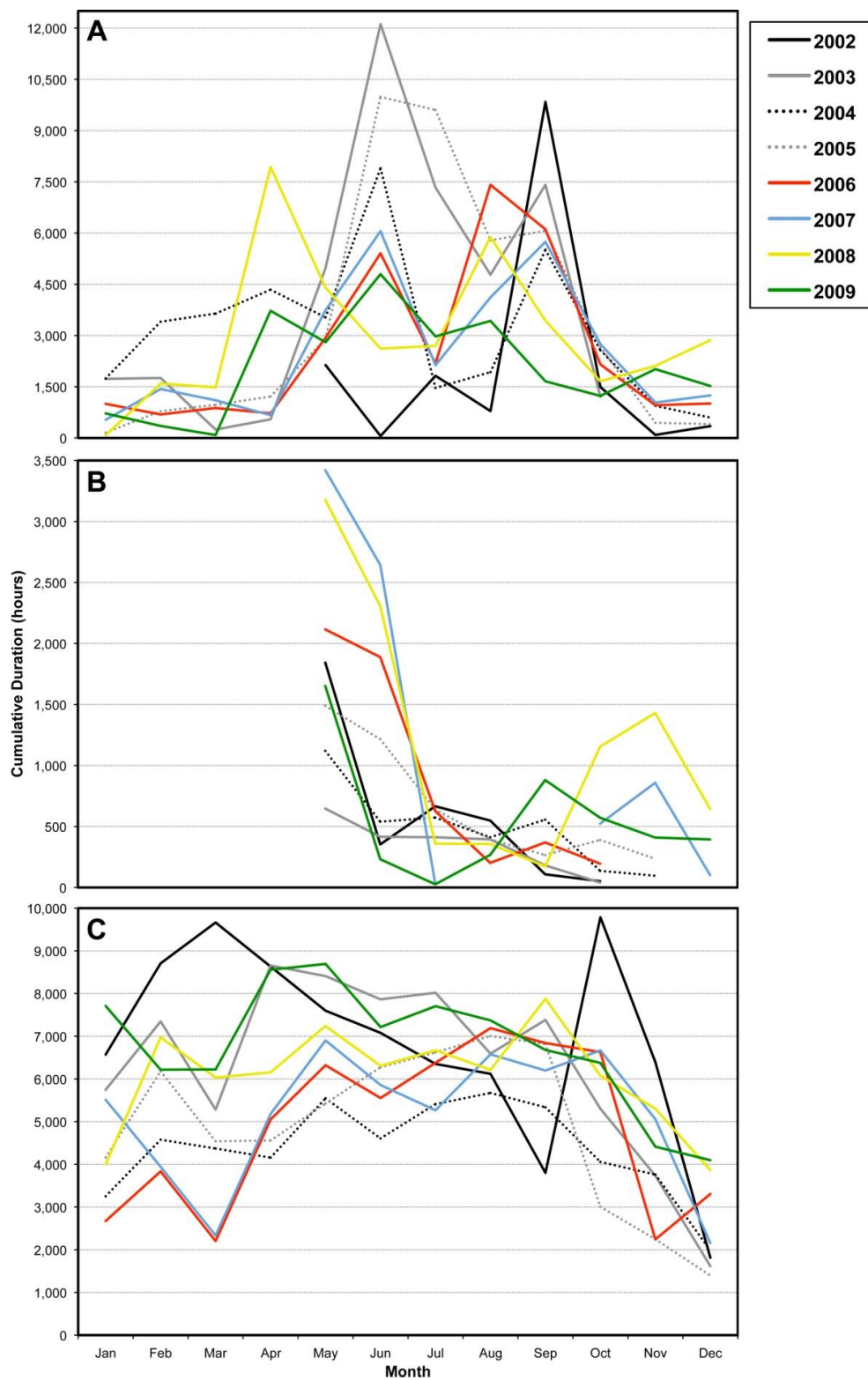


Figure MMR2. Monthly trends in fishing effort, expressed as cumulative number of hours per month (from 2002-2009) fishing gear was deployed in the water for each of the three fleet types. Panel A = fixed; Panel B = hake; and, Panel C = trawl.

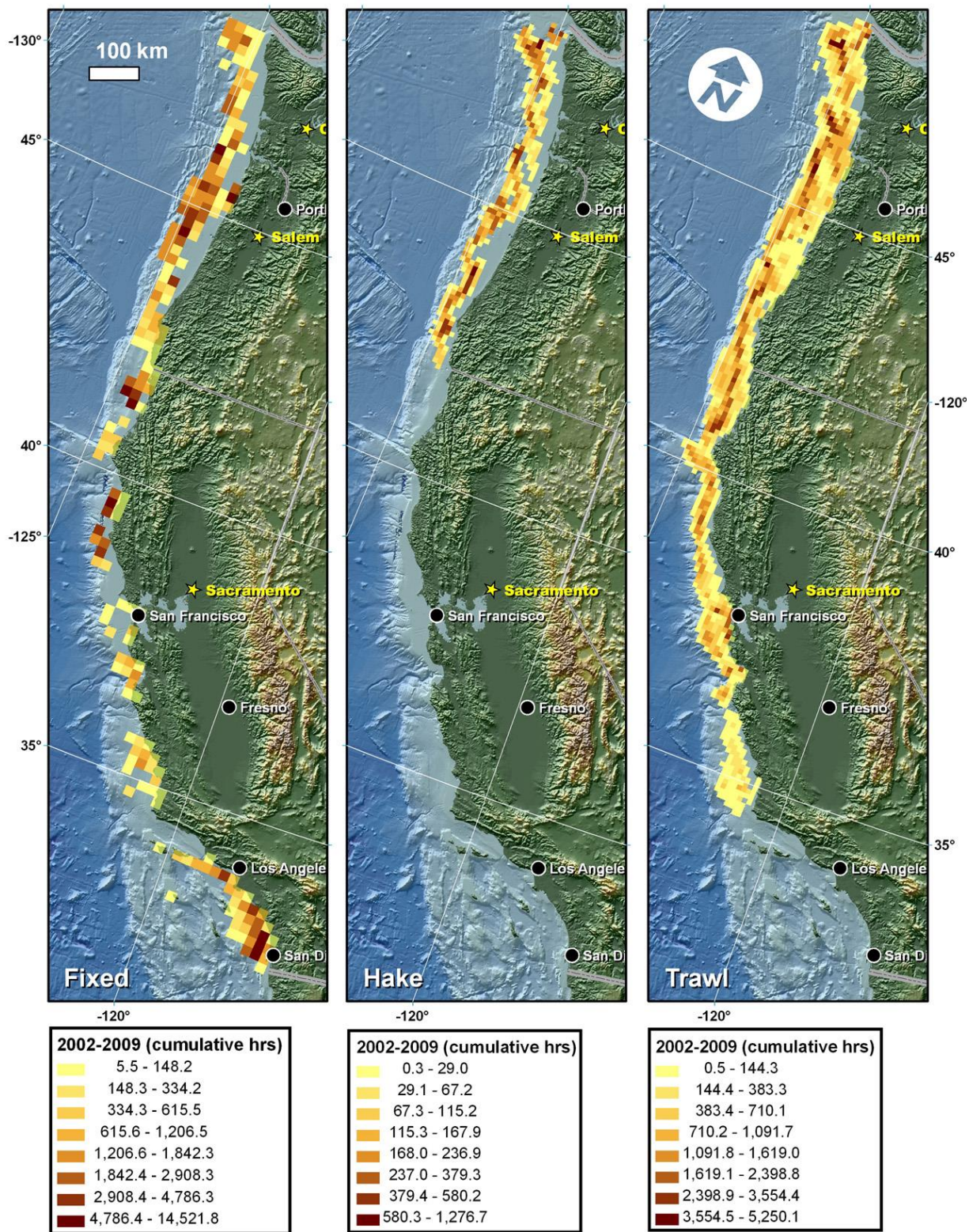


Figure MMR3. Patterns of fishing effort along the west coast of the United States, expressed as cumulative number of hours per gridcell (all months from 2002-2009) fishing gear was deployed in the water for each of the three fleet types.

CETACEAN AND FISHING OVERLAP MAPPING

Generally, there was low overlap spatially between the 12 cetacean species and the three commercial fishing fleets (Figures MMR4 to MMR15). Given that most of the fishing fleets operate within 100 km of shore, they overlap in a small portion of the modeled spatial domain of cetacean density.

Where there was overlap between the various cetacean species and the three commercial fishing fleets, there was considerable variation in the overlap index. Overall, cetacean species with higher modeled densities that coincided with longer durations of commercial fishing operations had higher overlap index scores.

BLUE WHALE

The highest degree of blue whale spatial overlap with WCGF fleets occurs with the observed fixed gear fleet, with some local overlap index values exceeding 20 animal hours/km² near San Diego, CA and just north of Cape Mendocino, CA (Figure MMR4). Overlap with the trawl fleet is much lower, with a few overlap indices exceeding ~4 animal hours/km² near Cape Mendocino, CA and off of the San Francisco Bay, CA (Figure MMR4). Overlap with the hake fleet was very limited, and was <0.5 animal hours/km² in all locations (Figure MMR4).

FIN WHALE

The highest areas of fin whale spatial overlap with the WCGF occur from the Columbia River area northward, with overlap indices for the observed fixed gear fleet of >20 animal hours/km² near the Columbia River mouth, and indices for the trawl fleet >3 animal hours/km² along the Washington coast (Figures MMR5). The highest overlap index with the hake fleet was < 2 animal hours/km², off the northern Washington coast (Figure MMR5).

BAIRD'S BEAKED WHALE

The observed fixed gear fleet overlapped the most (Figure MMR6) with Baird's beaked whale (>3.1 animal hours/km²) near the mouth of the Columbia River, the Stonewall Bank, OR, and the Trinidad Canyon, CA. Overlap with the hake fleet was considerably lower, with maxima occurring just west of Ozette Island, WA (0.239 animal hours/km², Figure MMR6). For the trawl fleet, overlap was generally higher in the northern two thirds of the fishing grounds, with maxima occurring just west of Ozette Island, WA, and north of Cape Mendocino, CA (>0.65 animal hours/km², Figure MMR6).

SHORT-BEAKED COMMON DOLPHIN

Short-beaked common dolphins overlapped the most with the fixed gear fleet from south of the Channel Islands, CA down to the US/Mexico border (>1,076 animal hours/km², Figure MMR7). Overlap with the hake fleet was greatest just west of Ozette Island, WA, near the mouth of the Columbia River and near the Astoria Sea Channel, OR (>17 animal hours/km², Figure MMR7). The trawl fleet overlapped fairly consistently along the entire fishing area, with maximum overlap occurring just west of Ozette Island, WA, just north of Cape Mendocino, CA and off the coast of San Francisco, CA (>83 animal hours/km², Figure MMR7).

RISSE'S DOLPHIN

The observed fixed gear fleet overlap with Risso's dolphin was greatest near the mouth of the Columbia River, the Stonewall Bank, OR, just north of Cape Mendocino, CA, and from the Northeast Bank, CA south to the US/Mexico border (>129 animal hours/km², Figure MMR8). Overlap with the hake fleet was greatest just west of Ozette Island, WA, and over the stretch from the mouth of the Columbia River south to the Stonewall Bank, OR, (>7 animal hours/km², Figure MMR8). Maximal overlap with the trawl fleet occurred over fairly large areas near Ozette Island, WA, and in a fairly large area at the Columbia River plume (>23 animal hours/km², Figure MMR8).

PACIFIC WHITE-SIDED DOLPHIN

Pacific white-sided dolphin overlap with the observed fixed gear fleet occurred near the mouth of the Columbia River, the Stonewall Bank, OR, and near Trinidad Canyon, CA (>289 animal hours/km², Figure MMR9). Overlap with the hake and trawl fleets was most pronounced near Neah Bay, WA (>28 and >128 animal hours/km², respectively, Figure MMR9).

NORTHERN RIGHT WHALE DOLPHIN

Maximum overlap between northern right whale dolphin and the observed fixed gear fleet occurred near the mouth of the Columbia River and Trinidad Canyon, OR (>115 animal hours/km², Figure MMR10). The hake fleet overlapped the most near Neah Bay, WA (>9 animal hours/km², Figure MMR10), and trawl fleet efforts overlapped the most near Neah Bay, WA, but had a pretty consistent overlap all the way south to Cape Mendocino and beyond (33 animal hours/km², Figure MMR10).

HUMPBACK WHALE

For the observed fixed gear fleet, peak areas of overlap with humpback whales (>17 animals hours/km²) occur north of Cape Mendocina, CA off the central Oregon coast, and off the Columbia River mouth (Figure MMR11). For the trawl fleet, the highest overlap indices occur along the northern portion of the coast from Cape Mendocina, CA to Cape Flattery, WA with areas of overlap > 3 animals hours/km² (Figure MMR11). The highest overlap indices for the hake fleet occur near Cape Flattery, WA and are < 2 animal hours/km² (Figure MMR11).

DALL'S PORPOISE

Overlap with the observed fixed gear fleet and Dall's porpoises was concentrated from the mouth of the Columbia River south to around the Stonewall Bank, OR (>630 animal hours/km², Figure MMR12). Maximum overlap with the hake fleet was near Neah Bay, WA, and in the region from the Columbia River plume south to around Heceta Valley, OR (>40 animal hours/km², Figure MMR12). The trawl fleet overlapped fairly consistently from Neah Bay, WA, all the way south to Cape Mendocino, CA (>124 animal hours/km², Figure MMR12).

SPERM WHALE

Overlap indices between the sperm whale distribution and the groundfish fisheries are generally lower compared with other whales. For the observed fixed gear fleet, the maximum values are < 6 animal hours/km², and occur in only a few places north of Cape Mendocino, CA (Figure MMR13). Overlap indices for the trawl fleet are fairly low and uniform from San Francisco, CA to Cape Flattery, WA and generally < 1 animal hours/km² (Figure MMR13). Overlap indices for the hake fleet are all < 0.3 animal hours/km² (Figure MMR13).

STRIPED DOLPHIN

Striped dolphin overlapped most with the observed fixed gear fleet near the mouth of the Columbia, Stonewall Bank, OR, Trinidad Canyon, CA, and over a fairly large area running south of Cape Mendocino down to just north of the Cordell Bank (>3 animal hours/km², Figure MMR14). In contrast, overlap with the hake fleet was concentrated over a fairly large area from the mouth of the Columbia River south to the Oregon/California border (>0.06 animal hours/km², Figure MMR14). Overlap with the trawl fleet was also fairly homogeneous, and was consistently high from 45° N latitude south to Santa Lucia Bank, CA (>0.7 animal hours/km², Figure MMR14).

SMALL BEAKED WHALES

Maximum observed fixed gear fleet overlap with small beaked whales occurred in the Columbia River plume, Stonewall Bank, OR, and the Trinidad Canyon, CA Vizcaino Knoll, and off San Diego, CA (>11 animal hours/km², Figure MMR15). Overlap coincided the most with hake fleet efforts that occurred near Neah Bay, WA, the mouth of the Columbia River and the Stonewall Bank, OR (>0.6 animal hours/km², Figure MMR15). Finally, trawl fleet operations overlapped the most near Neah Bay, WA, the Columbia River plume, Stonewall Bank, OR, Siltcoos Bank, OR, Trinidad Canyon, CA, south of Cape Mendocino, CA, and off the coast of San Francisco, CA (>2 animal hours/km², Figure MMR15).

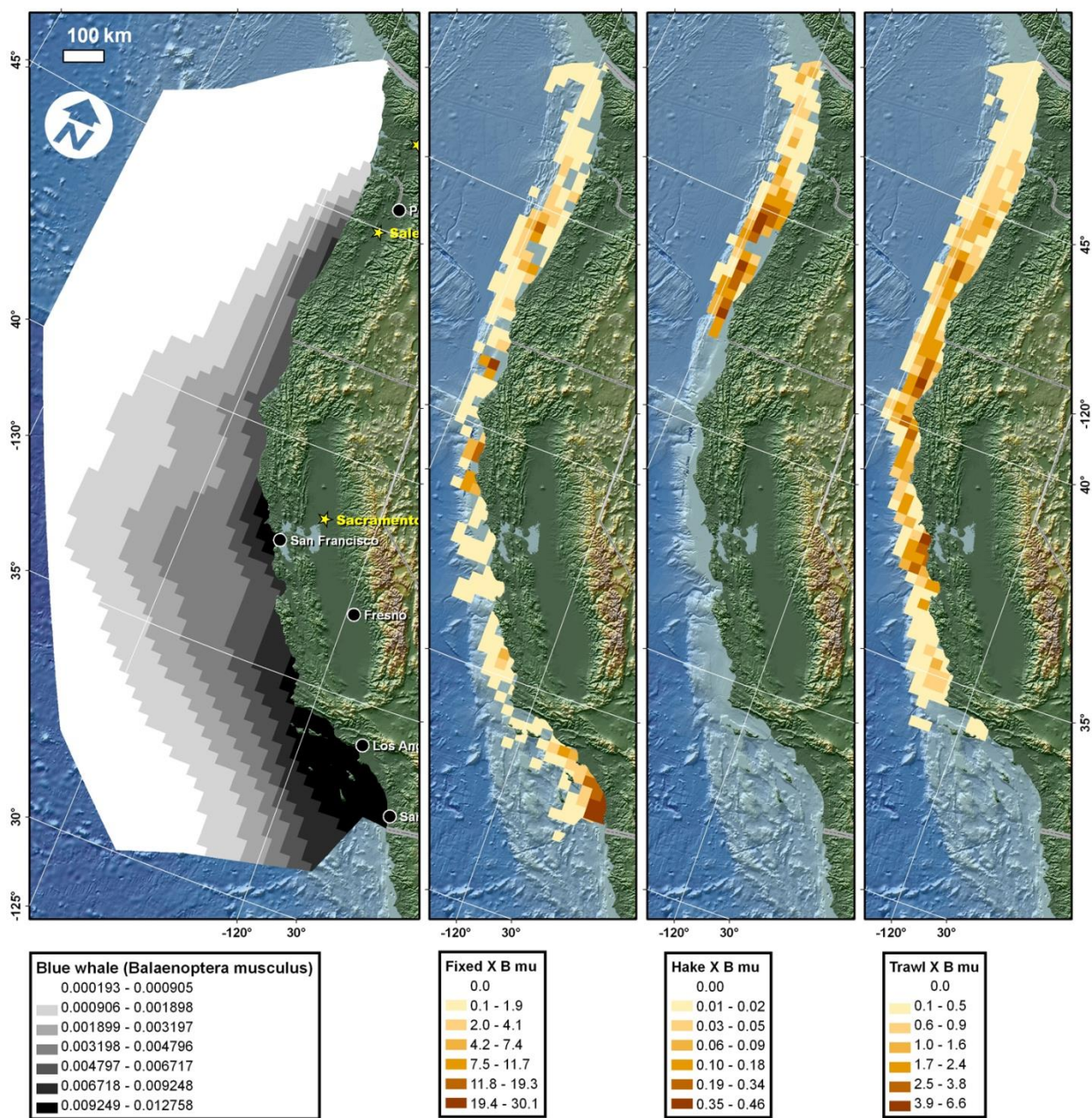


Figure MMR4. Left map: modeled blue whale mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for blue whale with the fixed, hake and trawl fleets.

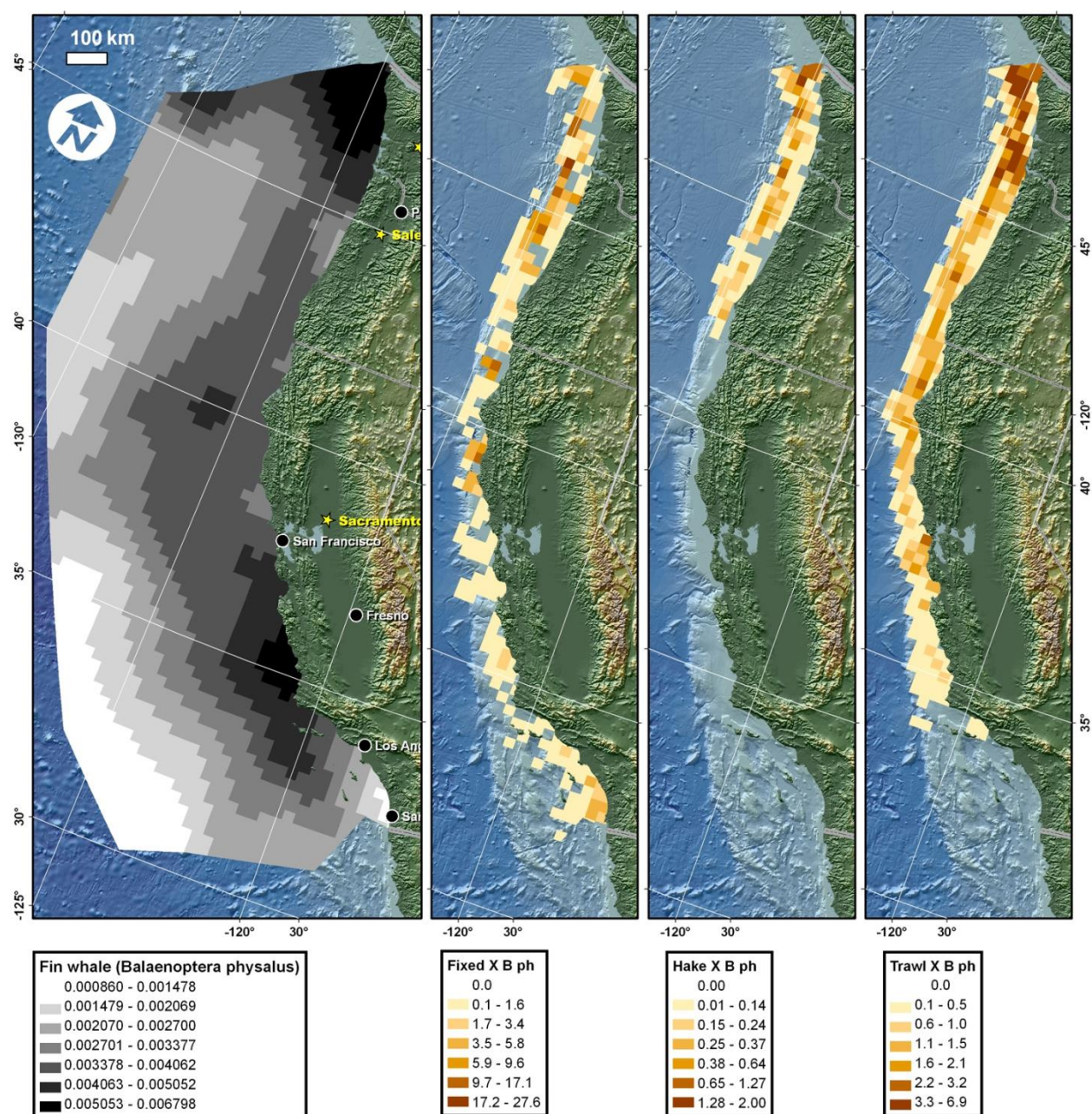


Figure MMR5. Left map: modeled fin whale mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for fin whale with the fixed, hake and trawl fleets.

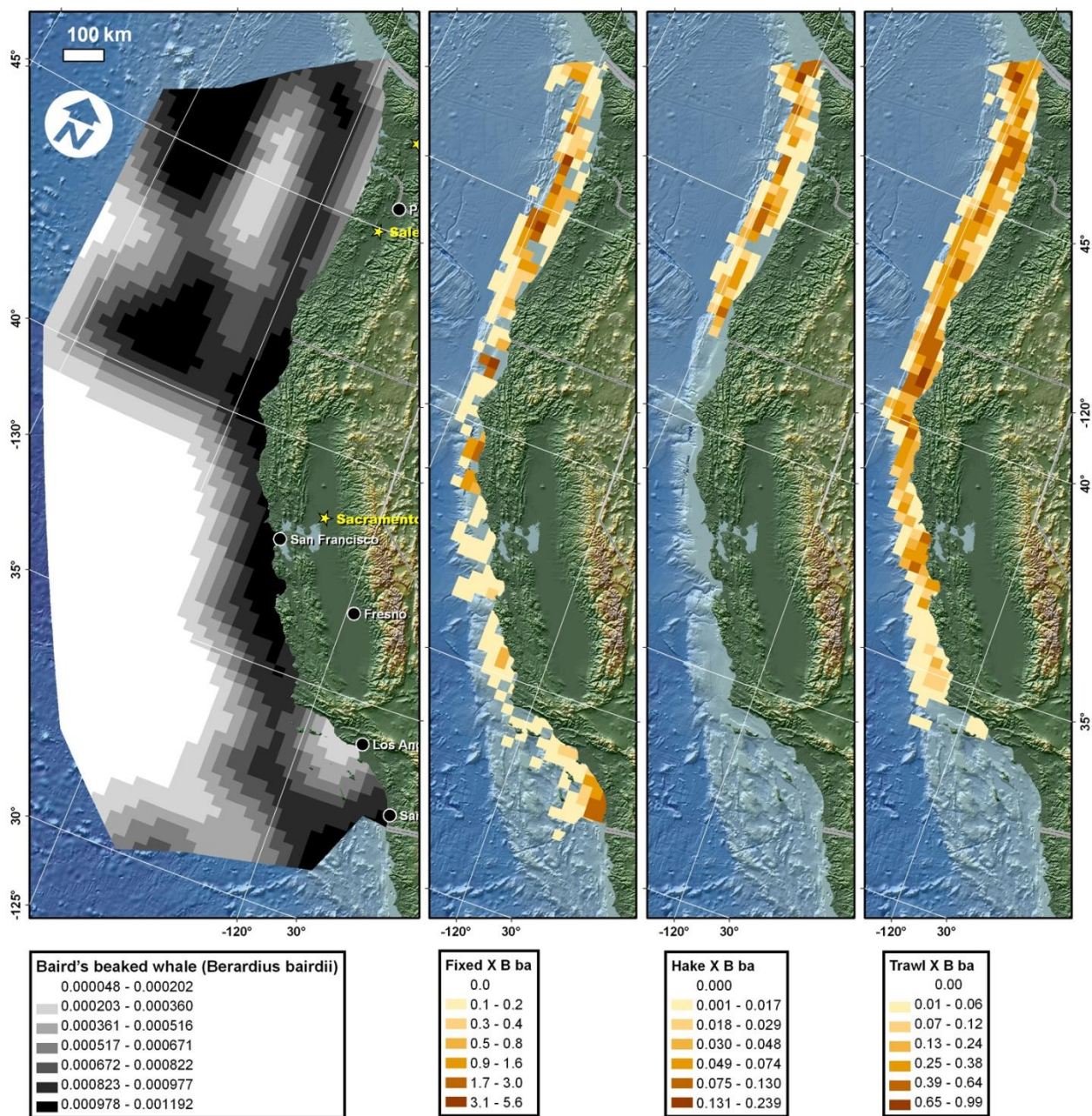


Figure MMR6. Left map: modeled Baird's beaked whale mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for Baird's beaked whale with the fixed, hake and trawl fleets.

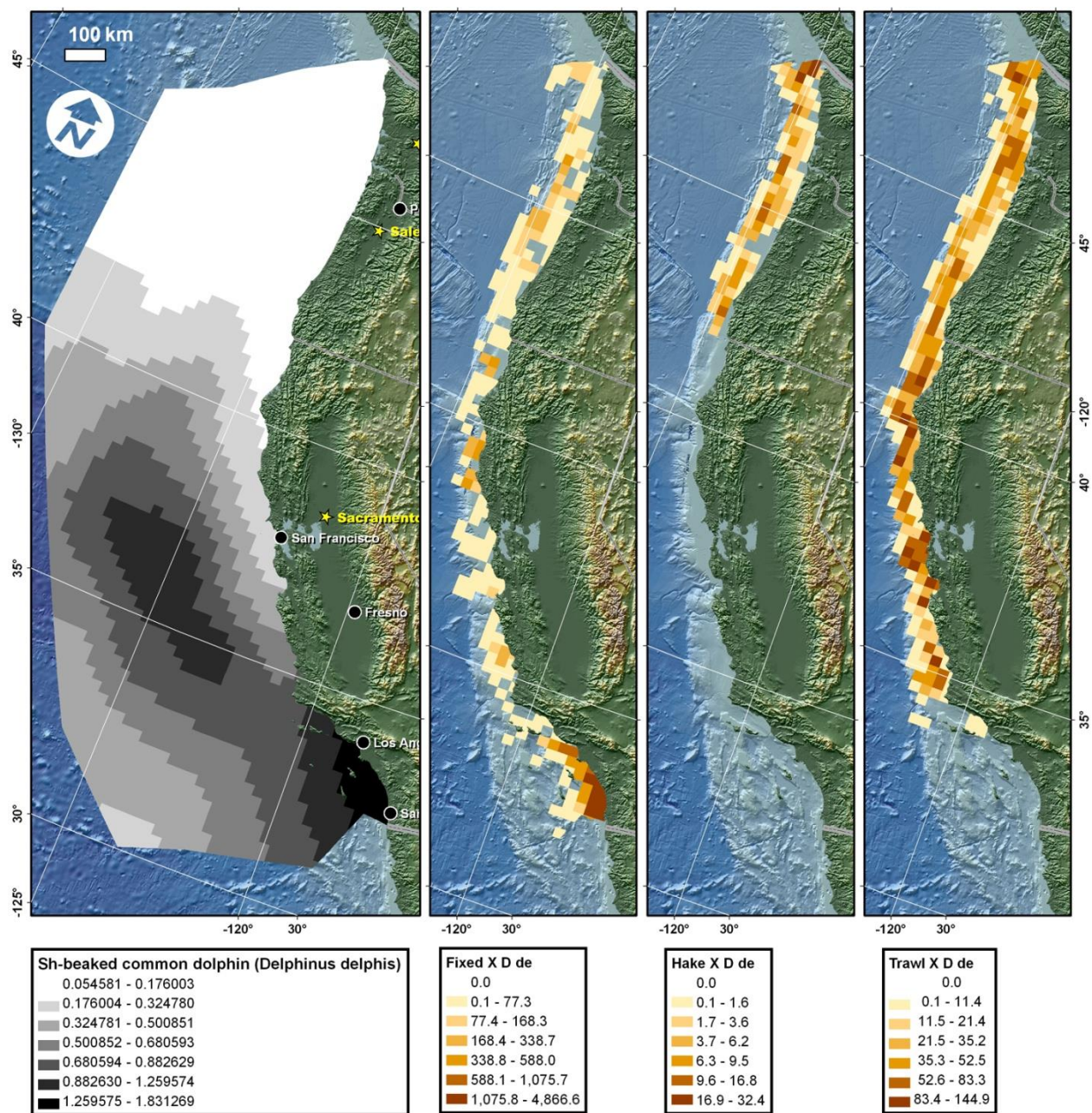


Figure MMR7. Left map: modeled short-beaked common dolphin mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for short-beaked common dolphin with the fixed, hake and trawl fleets.

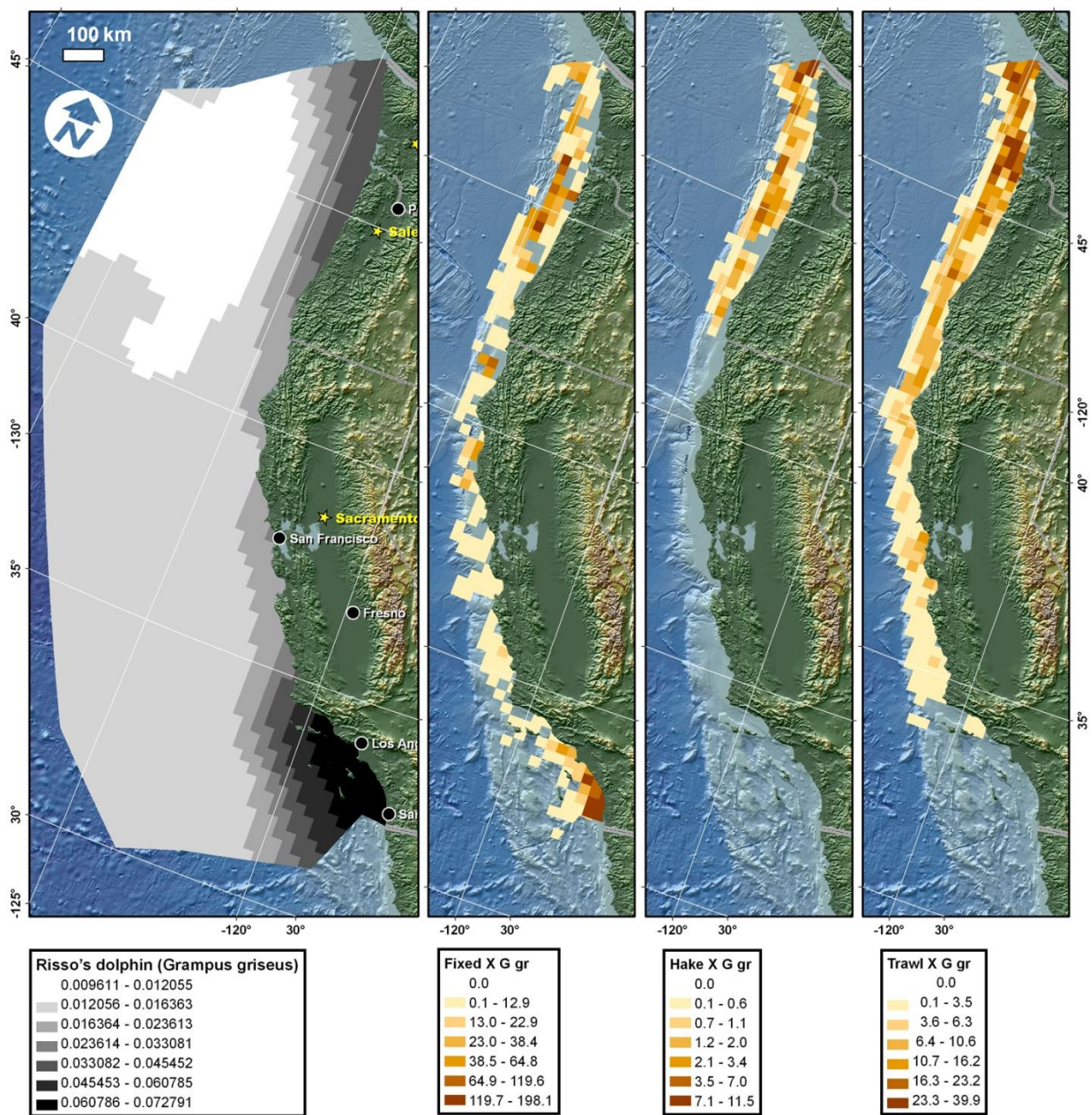


Figure MMR8. Left map: modeled Risso's dolphin mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for Risso's dolphin with the fixed, hake and trawl fleets.

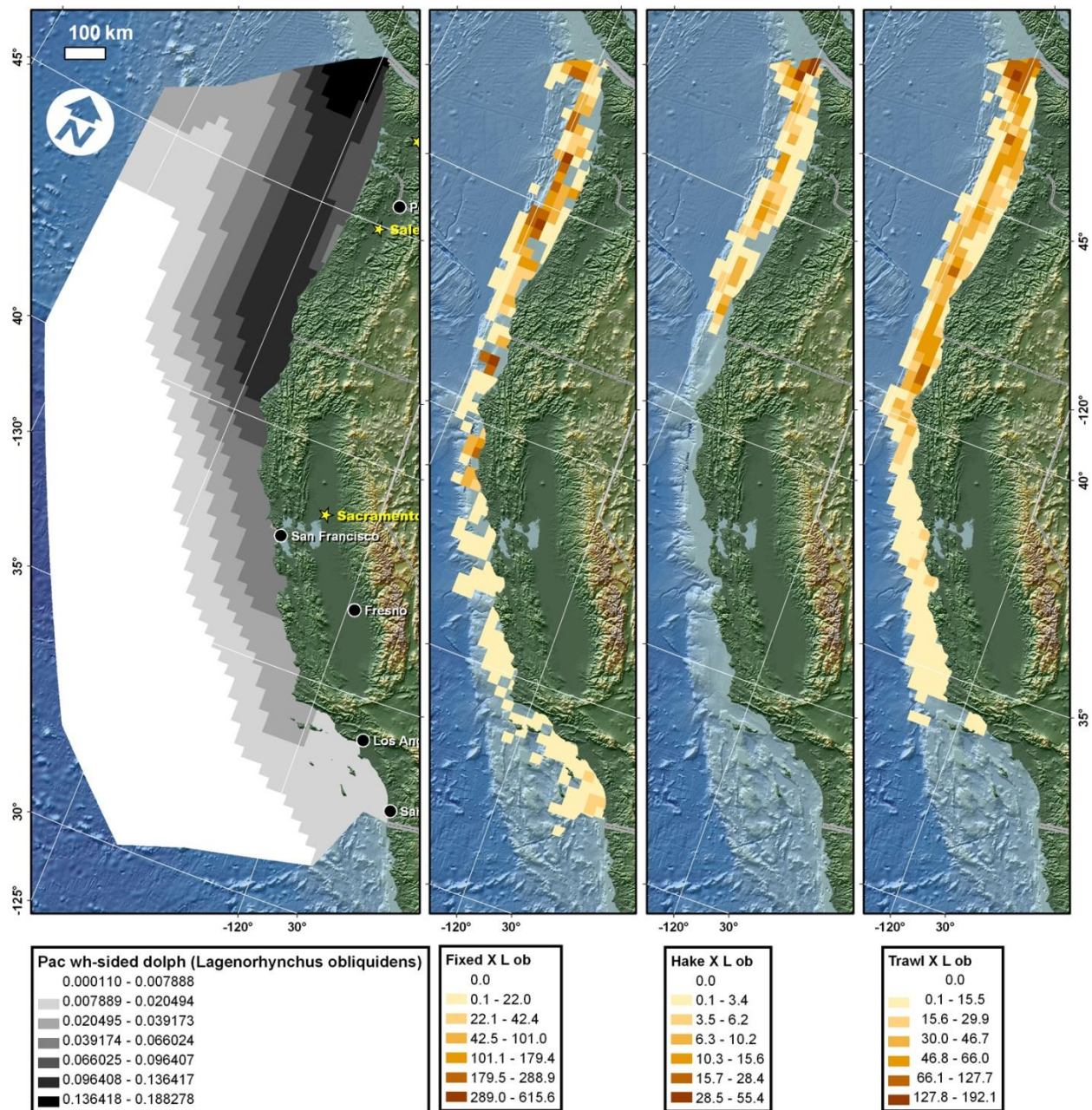


Figure MMR9. Left map: modeled Pacific white sided dolphin mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for Pacific white sided dolphin with the fixed, hake and trawl fleets.

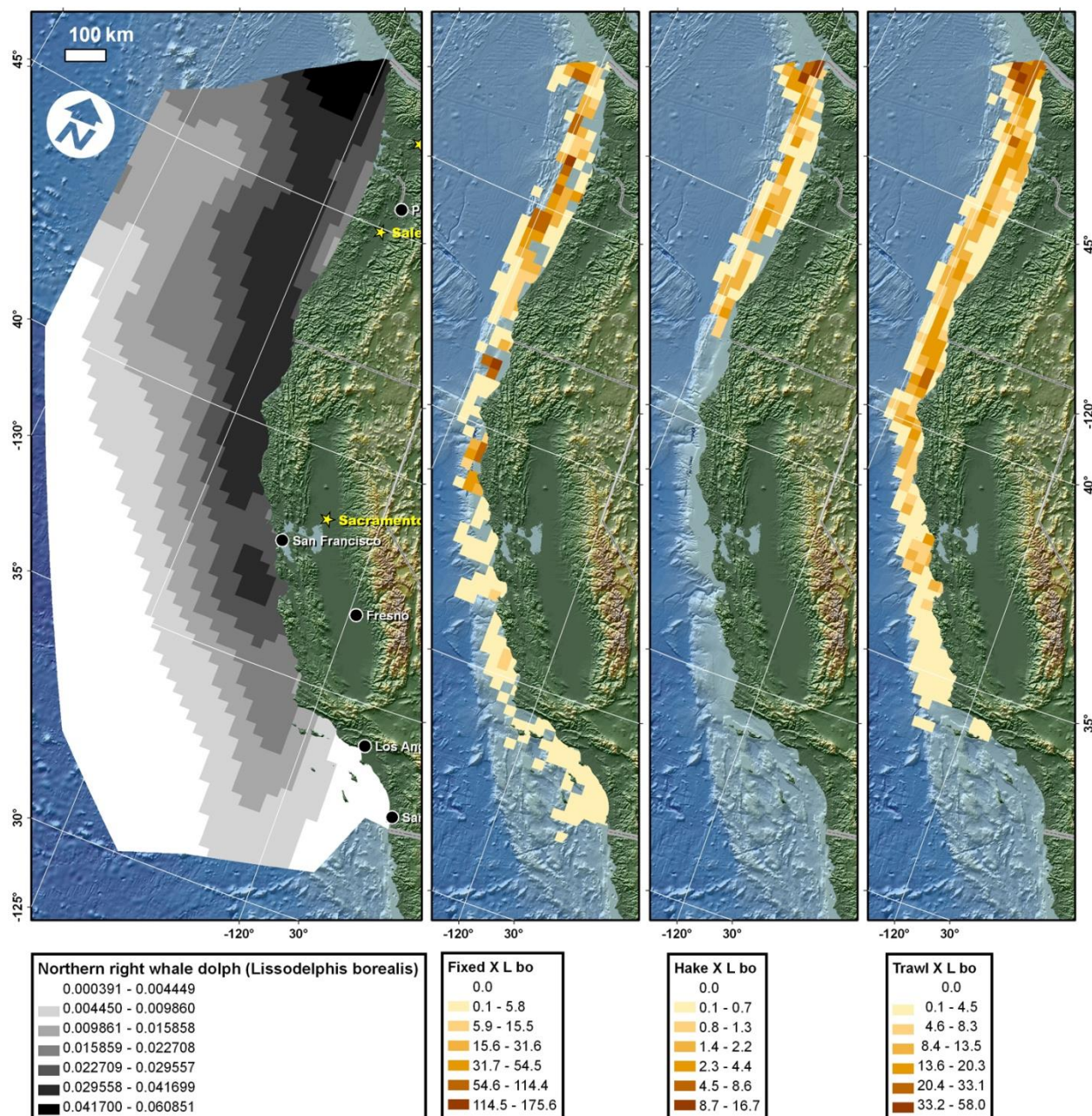


Figure MMR10. Left map: modeled Northern right whale dolphin mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for Northern right whale dolphin with the fixed, hake and trawl fleets.

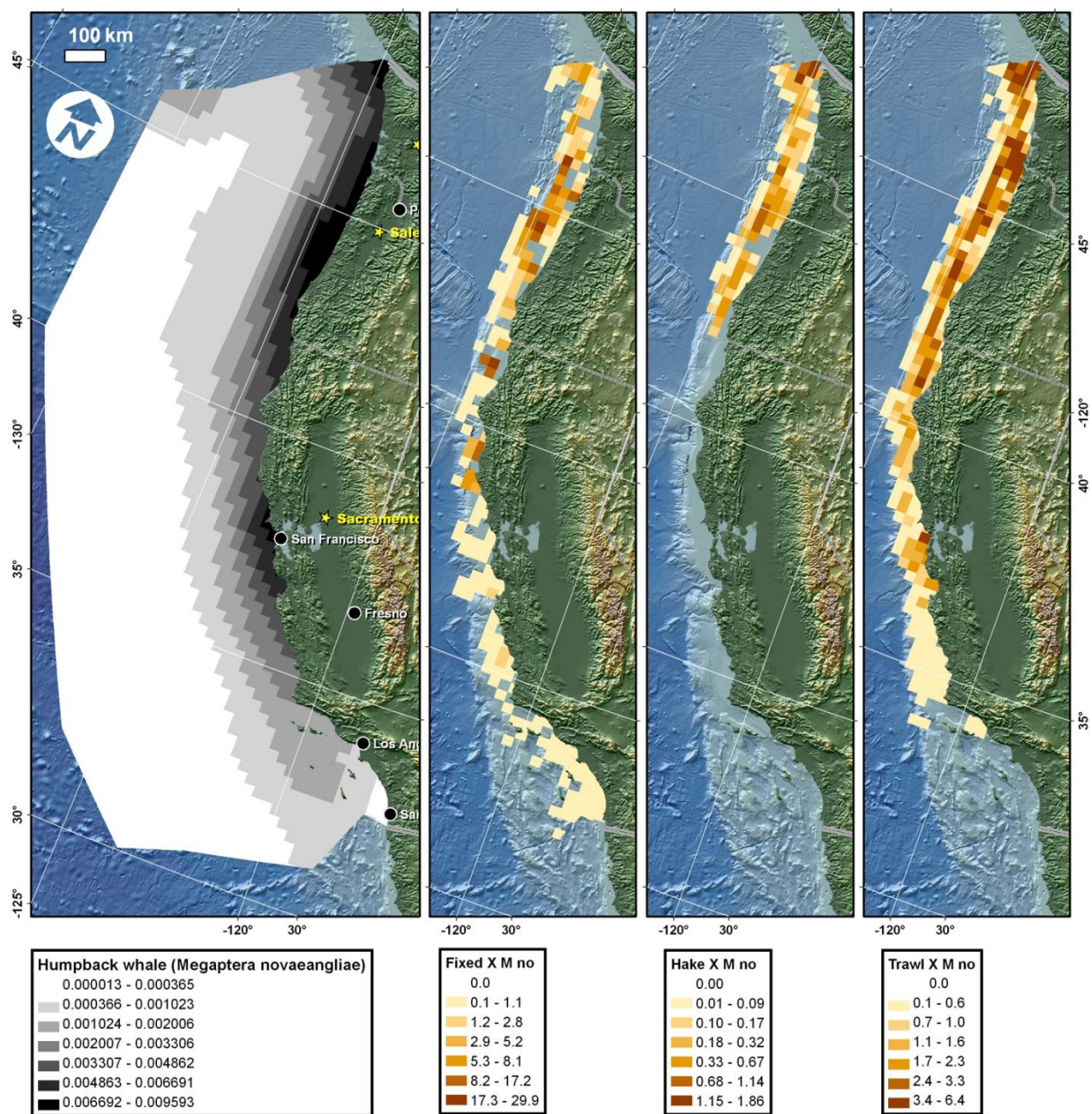


Figure MMR11. Left map: modeled humpback whale mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for humpback whale with the fixed, hake and trawl fleets.

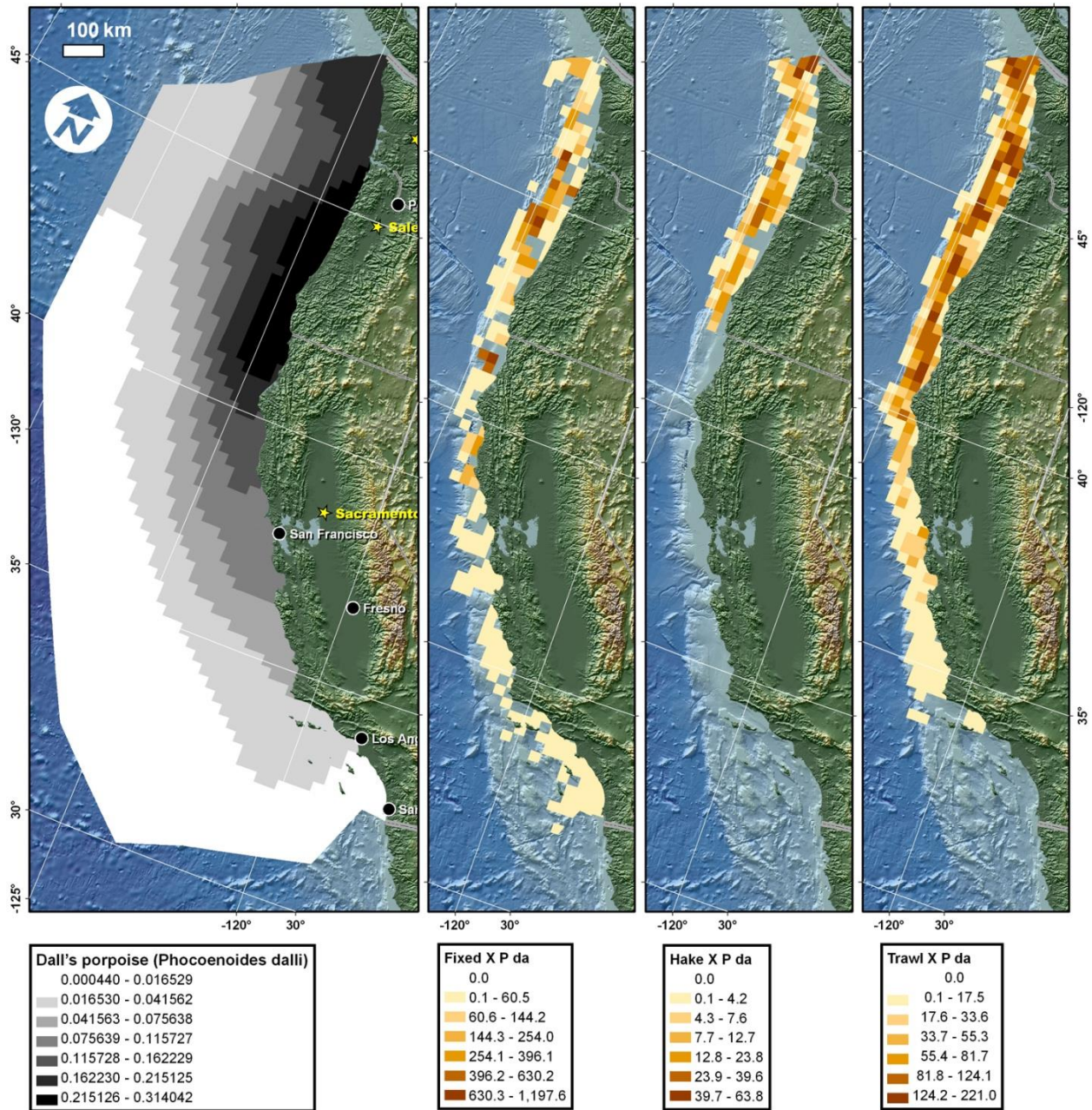


Figure MMR12. Left map: modeled Dall's porpoise mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for Dall's porpoise with the fixed, hake and trawl fleets.

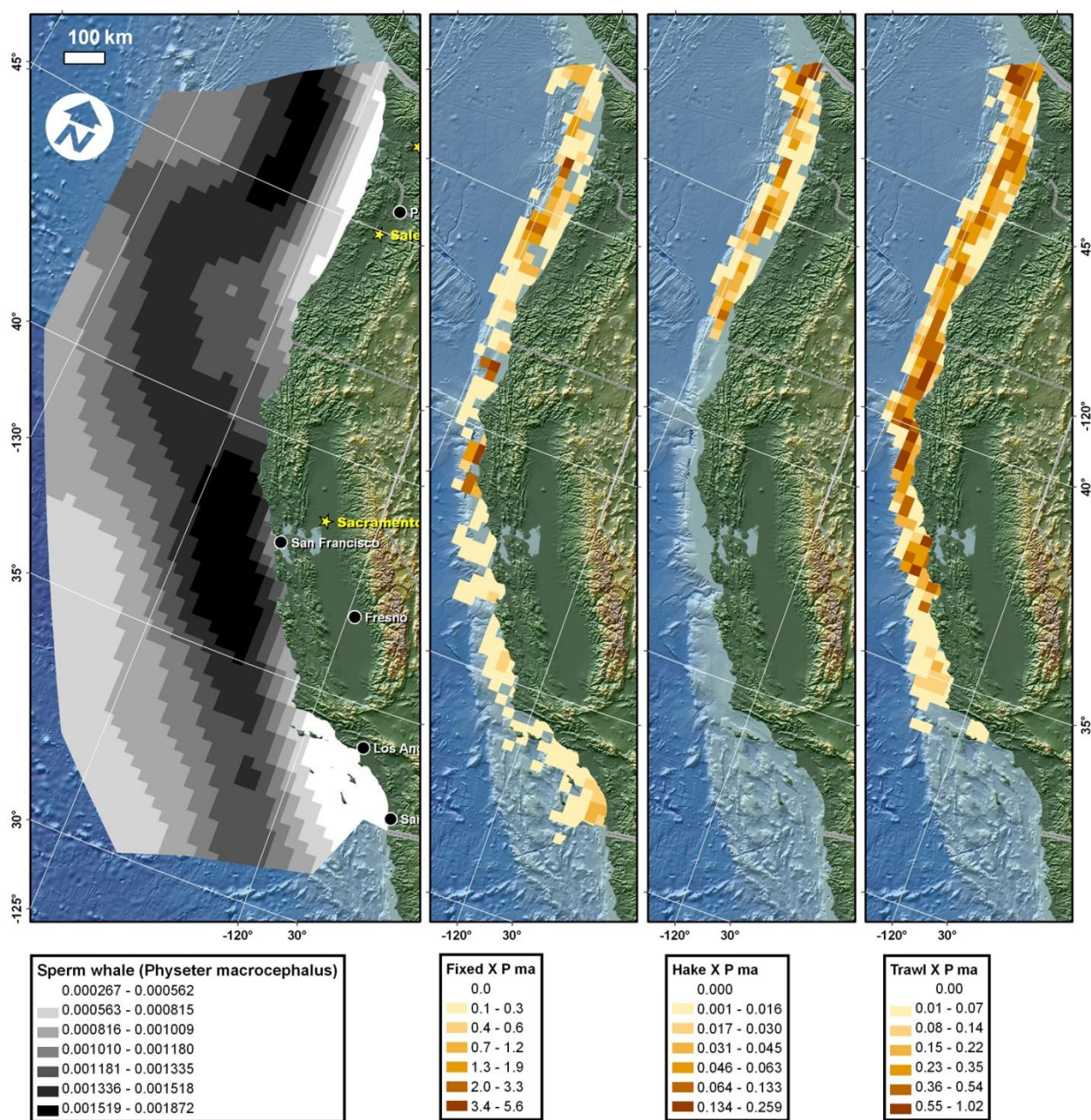


Figure MMR13. Left map: modeled sperm whale mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for sperm whale with the fixed, hake and trawl fleets.

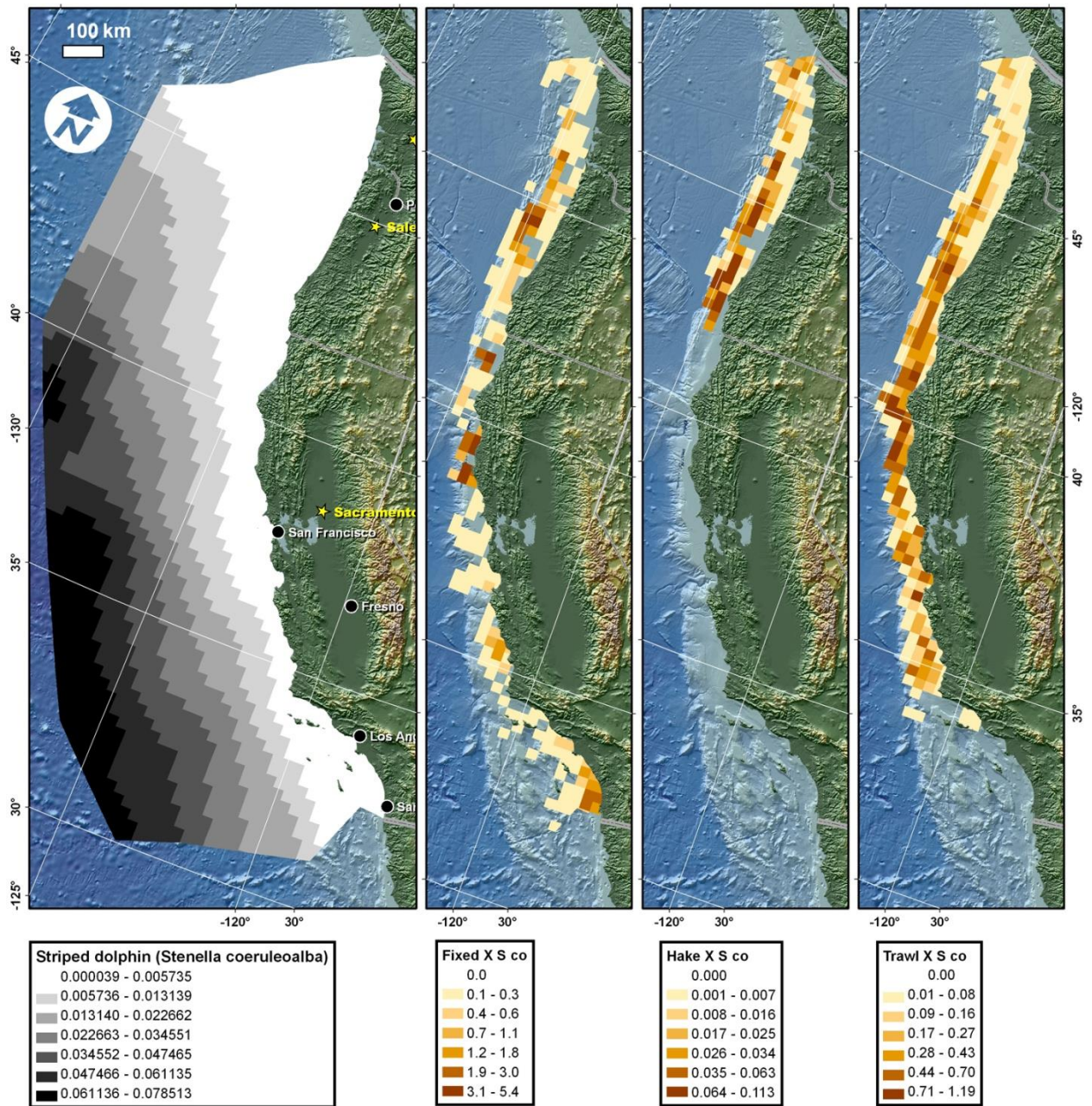


Figure MMR14. Left map: modeled striped dolphin mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for striped dolphin with the fixed, hake and trawl fleets.

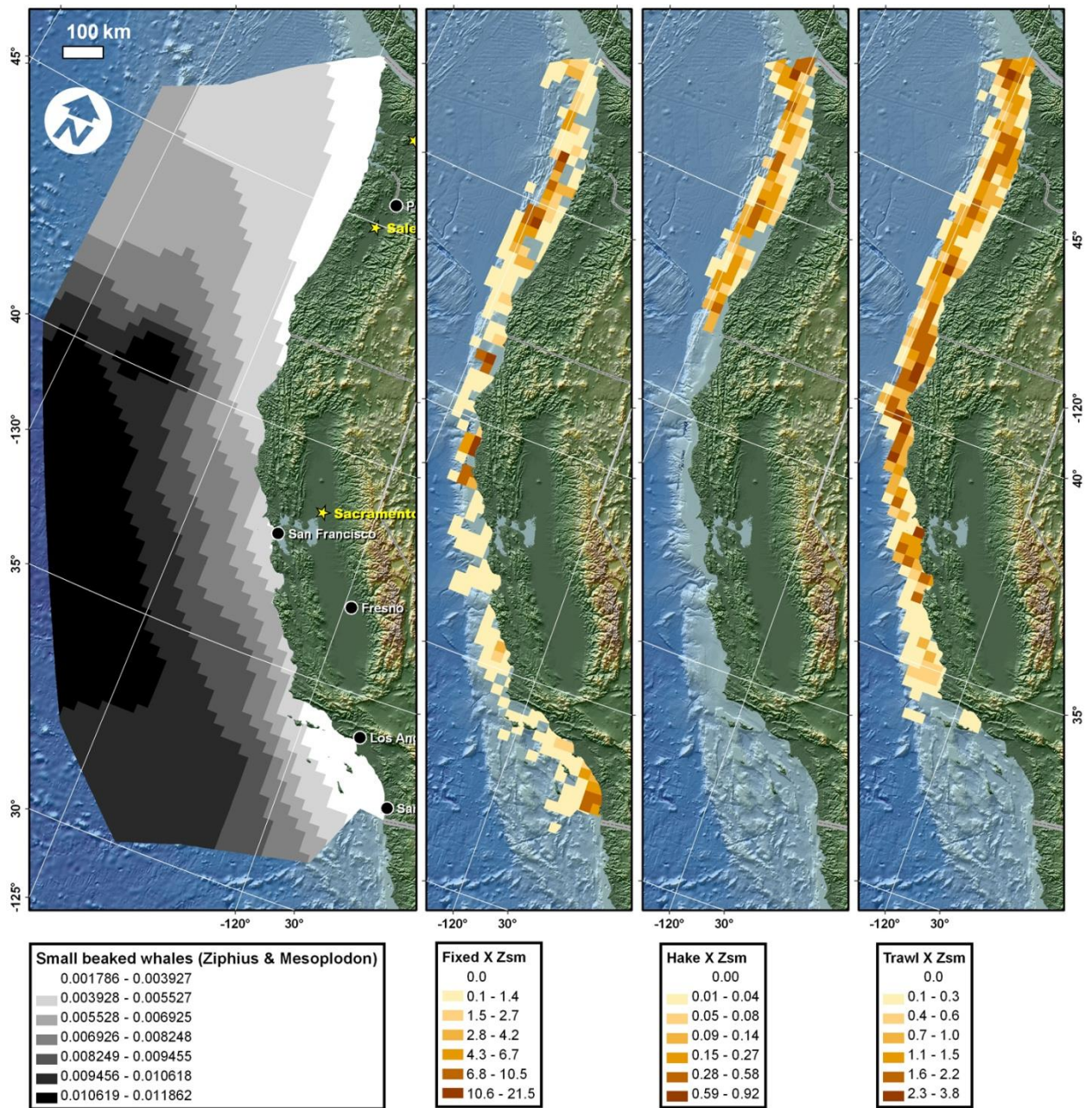


Figure MMR15. Left map: modeled small beaked whales mean density expressed as the number of individuals/yr/km² (based on survey data collected from 1991 – 2005) within the EEZ off the west coast of the United States. Three narrow maps: overlap values for small beaked whales with the fixed, hake and trawl fleet

POPULATION OVERLAP INDEX

There was considerable variability in the proportion of each modeled cetacean population that overlapped with the three fleet types for the years 2002-2009 (Figure MMR16, top panel). Overall, humpback whale, Dall's porpoise and Pacific white-sided dolphin had the greatest proportion of their populations overlapping with each of the three fleets. Population overlap was generally highest for the observed trawl fleet, but not always (i.e., short-beaked common and Risso's dolphin, Figure MMR16, top panel). It's important to note that the proportions displayed by the bars in Figure MMR16 (top panel) cannot be summed, as there was overlap between the different fleet types.

CUMULATIVE OVERLAP INDEX

OVERALL PATTERNS

Overall, there were marked differences in the overlap indices of the different cetacean species (Figure MMR16, bottom panel). The largest overlap indices occurred in the observed fixed gear fleet, which were about 40 times that of the hake fleet and 2.5 times that of the trawl fleet. Short-beaked common dolphin had the highest overlap index when combining all of fleets and Baird's beaked and sperm whales, and striped dolphin had the lowest (Figure MMR16, bottom panel). Within the three fleets, there was considerable variability in the overlap indices with dolphins and porpoises experiencing the highest overlap indices, while whales had the lowest overlap values (Figure MMR16, bottom panel).

INTERANNUAL PATTERNS

As was the case with the overall cumulative overlap indices, there was considerable interspecific variation (Figure MMR17). Overall, cumulative overlap indices (COI) were higher for the observed fixed gear fleet, compared with the hake and trawl fleets. For the observed fixed gear fleet, many cetacean species (Dall's porpoise, Pacific white-sided dolphin, northern right whale dolphin, Risso's dolphin) had marked increases in their COI in 2003 and 2005, and most species, with the exception of short-beaked common dolphin, generally had a lower COI in 2009 compared with 2002. Short-beaked common dolphin show a strong increase in the COI from 2002 to 2009, rising nearly 10 fold during this time period. Cumulative overlap indices for most species increased consistently from 2003-2008 for the hake fleet, but dropped off markedly in 2009 (Figure MMR17B). Dall's porpoise, short-beaked common dolphin and Pacific white-sided dolphin consistently had the greatest COI of all the 12 modeled cetacean species, whereas Baird's beaked whale, blue whale, fin whale, humpback whale, sperm whale, striped dolphin and small beaked whales had the lowest COI (Figure MMR17B). Finally, the trawl fleet COI were markedly different from the observed fixed gear and hake fleets. Aside from 2004, COI values were fairly consistent over time, or slightly declining (e.g., short-beaked common dolphin, Figure MMR17C). The COI for all 12 cetacean species was significantly lower in 2004, with around 20 – 30% decreases occurring in most species.

LIMITATIONS AND NEXT STEPS

There were many limitations to this risk assessment, but we focus on four broad categories that we feel were the most important and warrant the greatest attention for future improvements on this risk

assessment. Those categories were: spatial and temporal scaling, indirect groundfish fishery impacts, vulnerability vs. sensitivity, and other sources of risk.

SPATIAL AND TEMPORAL SCALING

Our approach for assessing the relative overlap between cetaceans and groundfish fleets was to compare cumulative fishing effort over all areas fished from 2002 – 2009, with best estimates of cetacean density over the past 15 years in the CCLME. Cetacean survey data are collected every 3 to 4 years, so we believe that comparing specific survey years (e.g., 2005 and 2008) with the corresponding fishing effort year might provide insight on interannual variability of overlap. Unfortunately, other scaling related problems are less easily addressed. For example, fishing effort data are available at the individual vessel level, which provides monthly, seasonal and annual patterns of effort, with high spatial and temporal precision. Acquiring comparable data for 12 species of cetaceans would require radiotagging and tracking thousands of animals, which would be logistically intractable. Therefore, generating overlap comparisons at this fine of a scale is not possible with current information, so we rely on other proxies of exposure or risk.

INDIRECT IMPACTS

Impacts from commercial fisheries on cetaceans can be direct (“operational” as described by Beverton (1985)) or indirect. Direct impacts include vessel collisions (Panigada et al. 2006), entanglement with fishing gear and “bycatch” (Lien 1994, Reeves et al. 2003, Read et al. 2006, Young and Ludicello 2007), stress (Curry 1999, Fair and Becker 2000), noise (National Research Council 2003, Romano et al. 2004, Nowacek et al. 2007), and toxins such as hydrocarbons, exhaust, etc. (Jarman et al. 1996, Marsili et al. 2001). Indirect effects of commercial fishing include exploitation competition and habitat disturbance (Dayton et al. 1995, Bearzi et al. 1999, DeMaster et al. 2001, DeMaster et al. 2006, Herr et al. 2009). While it appears as though direct impacts of the groundfish fleets on cetaceans (via bycatch and vessel collisions) are minimal at a population level (Jannot et al. 2011), indirect impacts are poorly understood. This is particularly the case with the fixed gear fleet, where observer coverage only averages about 17% across all the gear types (Table MMR2). Given the coarse nature of our analyses, it is difficult to be certain that “overlap” or “exposure”, as we have defined it, would result in harm to a given cetacean species. Rather, our analyses provide relative risks, in that some species have greater exposure to certain gear types, and certain gear types present a greater potential risk, compared with others.

Future risk analyses of the groundfish fleets should formally take in to account indirect effects from all three of the fleets considered in these analyses. However, quantifying these effects is difficult and complex, which is probably why they have not been exhaustively analyzed in the past. Further, devising better ways to estimate the potential for harm, given overlap in any given 625 km² grid cell, would greatly improve future risk assessments.

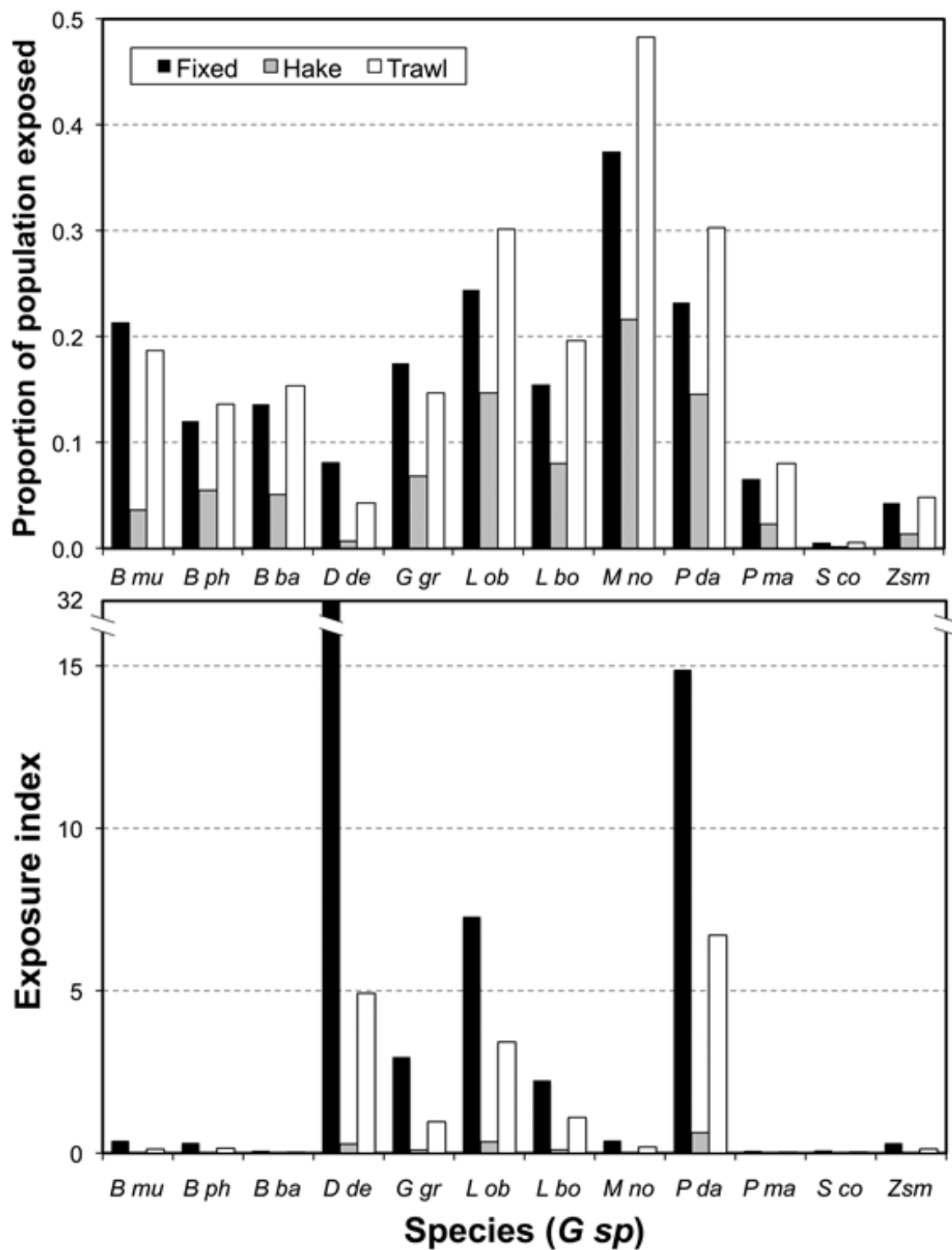


Figure MMR16. Modeled proportion (upper) and cumulative exposure index (lower) of each cetacean species population that overlapped with each of the three commercial fishing fleets (from 2002-2009), for each of the 12 cetacean species. B ba = Baird's beaked whale; B mu = blue whale; B ph = fin whale; D de = short-beaked common dolphin; G gr = Risso's dolphin; L bo = northern right whale dolphin; L ob = Pacific white-sided dolphin; M no = humpback whale; P da = Dall's porpoise; P ma = sperm whale; S co = striped dolphin; and, Zsm = small beaked whales.

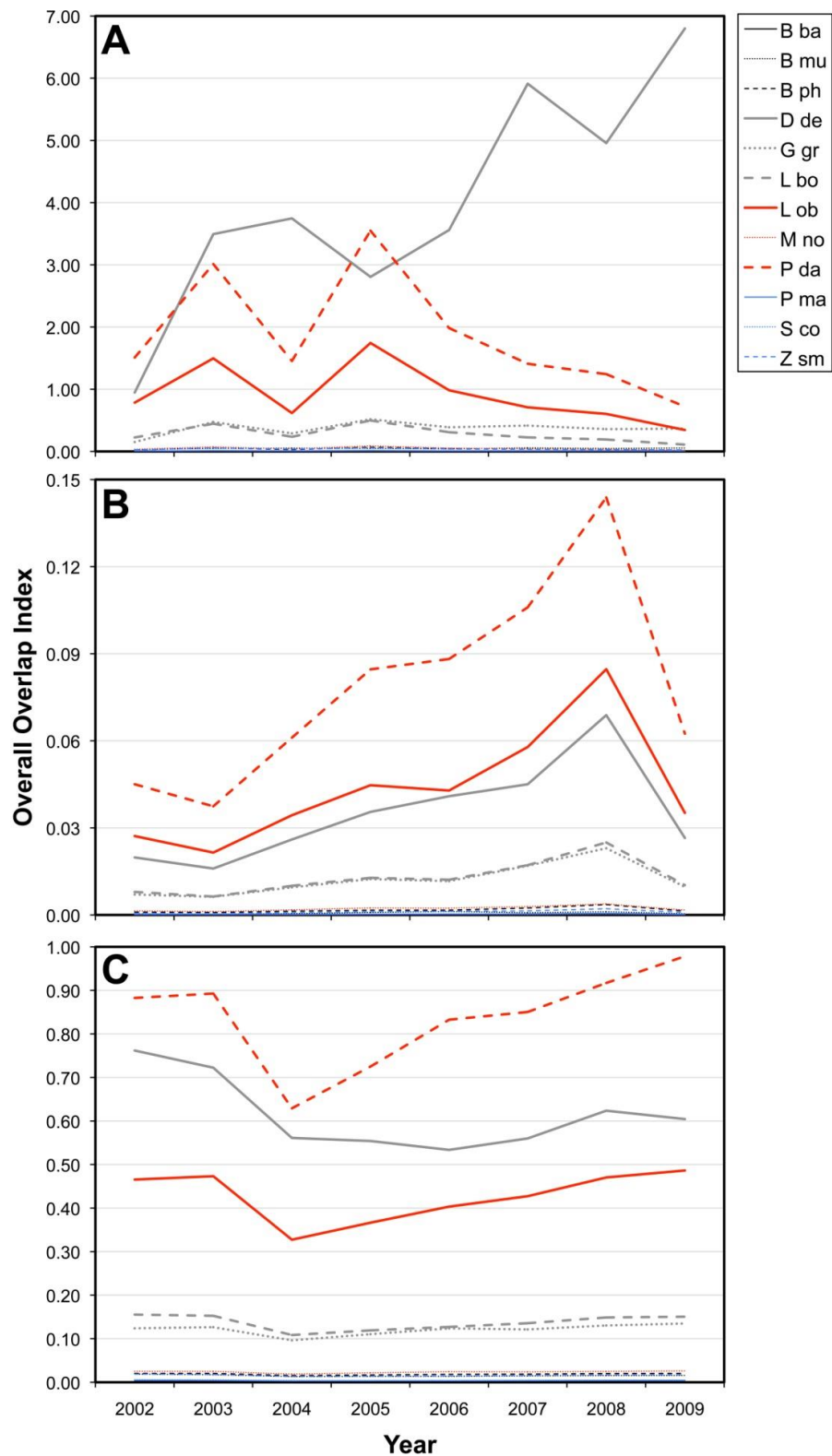


Figure MMR17. Cumulative annual commercial fishing fleet overlap indices (from 2002-2009) for each of the 12 cetacean species. Panels A, B, and C are the fixed, hake and trawl fleets, respectively. B ba = Baird's beaked whale; B mu = blue whale; B ph = fin whale; D de = short-beaked common dolphin; G gr = Risso's dolphin; L bo = northern right whale dolphin; L ob = Pacific white-sided dolphin; M no = humpback whale; P da = Dall's porpoise; P ma = sperm whale; S co = striped dolphin; and, Zsm = small beaked whales.

VULNERABILITY VS. SENSITIVITY

Rowe (1977) argued that risk is the probability that something harmful will occur, and one quantifies that probability in a risk assessment. The first step in assessing this probability is identifying vulnerability or the exposure of an organism to something that could be harmful. In this analysis, we quantified the potential risk imposed by three commercial fishing fleets on cetaceans by taking the product of cetacean density and commercial fishing effort. This was not a formal risk assessment where changes in population growth were calculated as a function of a given fishing influence. This could be viewed as a “relative” risk assessment, in that we calculated the overlap of exposure to the various fleet types. Using a common currency of fishing effort expressed as time and cetacean density expressed as the mean number of animals predicted to occupy a given area each year. We did not explicitly address the two most common aspects of a risk assessment: vulnerability and sensitivity (Zacharias and Gregr 2005). However, we argue that our analyses directly addressed vulnerability, in that a given cetacean species is vulnerable to the potential negative consequences of a given fishing fleet type when it is in fact exposed to the vessels and gear from that fleet. While further work on the sensitivity of these species to the stressors induced by commercial fishing activities is needed for a formal risk assessment, our analyses are an important first step in characterizing the spatio-temporal patterns of cetacean exposure or vulnerability to commercial fishing fleets in the California Current.

OTHER SOURCES OF RISK

There are many risks to cetacean populations occurring in the CCLME in addition to the groundfish fishery fleets. These risks include other commercial fishing fleets (e.g., drift- and gillnet fleets), anthropogenic stress (Curry 1999, Fair and Becker 2000), collisions with non-fisheries vessels (Laist et al. 2001, Jensen and Silber 2003), noise (Committee on Potential Impacts of Ambient Noise in the Ocean on Marine Mammals 2003, Romano et al. 2004), exposure to toxins (hydrocarbons, exhaust, etc. (Jarman et al. 1996, Marsili et al. 2001)), entanglement with marine debris (Williams et al. 2011), resource competition (Trites et al. 1997, Herr et al. 2009, Gomez-Campos et al. 2011) and habitat disturbance from fishing (Kaiser 1998, Watling and Norse 1998), and global climate change (MacLeod 2009). While this list is not exhaustive, it provides context for the range of risks that cetaceans are confronted with. Quantifying the vulnerability and sensitivity of cetaceans to these threats is an important next step in running a more comprehensive risk assessment.

FUTURE RISK ANALYSES

Integrating the three aforementioned broad categories would greatly improve our ability to more comprehensively run risk assessments for cetaceans occurring in the CCLME. We were unable to address these deficiencies, given limitations of resources and time, and given the lack of available data for many of the aforementioned potential risks. Given the scarcity of data available for running comprehensive risk assessments for cetaceans in the CCLME, adopting the strategies laid out by Samhoury and Levin (2012), might be a productive first step towards a more quantitative and comprehensive risk assessment. An example of an application of this methodology is described in detail in the “Risk Assessment for Habitats in the Monterey Bay National Marine Sanctuary” section of this IEA. This approach calculates “relative risk” as a function of various “drivers and pressures” or stressors, accounts for data quality, and can incorporate disparate types of quantitative data. This would be an ideal next step for cetacean risk assessments and would provide valuable insight into the deficiencies that might be preventing a more formal and comprehensive risk assessment. Finally, studies directed at quantifying the risk imposed by stressors not

considered in our assessment would be beneficial if we wish to improve our certainty regarding the risks imposed on cetaceans by various anthropogenic stressors found in the CCLME.

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GROUND FISH RISK ASSESSMENT

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TABLE OF CONTENTS (GFR)

| | |
|--|------------|
| Executive Summary | 637 |
| Detailed Report | 639 |
| Introduction | 639 |
| Methods | 639 |
| Results | 643 |
| Conclusions | 644 |
| Groundfish Appendix A | 707 |
| Groundfish Appendix B - Non-Fisheries Threats – Literature Review | 709 |
| Aquaculture..... | 709 |
| Atmospheric deposition | 709 |
| Coastal Engineering..... | 710 |
| Direct human impacts | 711 |
| Inorganic Pollution | 711 |
| Light Pollution..... | 713 |
| Nutrient input | 713 |
| Ocean-based pollution..... | 714 |
| Offshore oil activities | 715 |
| Organic Pollution | 716 |
| Coastal seawater exchange | 717 |
| Sediment decrease | 718 |
| Sediment increase | 719 |
| Shipping activity | 720 |
| Species Invasion..... | 720 |
| Coastal Trash..... | 721 |
| Climate Change threats..... | 722 |
| Ocean acidification..... | 722 |

| | |
|-------------------------------|-----|
| Sea surface temperature | 723 |
| Ultraviolet light..... | 723 |
| Other potential threats | 725 |
| References cited | 726 |

LIST OF TABLES AND FIGURES (GFR)

| | |
|--|-----|
| Figure GFRii. Exposure intensity index of ocean-based pollution for sablefish <i>Anoplopoma fimbria</i> adults. High = upper tercile, Medium = middle tercile, low = lower tercile. | 638 |
| Table GFR1. List of non-fisheries threats considered. | 645 |
| Table GFR2. Raw and final Productivity scores. A weighted average of the ten scores (Cope et al. 2011) is used, for final values ranging between 1 and 3. Boc = bocaccio <i>Sebastes paucispinis</i> ; Can = canary rockfish <i>Sebastes pinniger</i> ; Hake = Pacific hake <i>Merluccius productus</i> ; Sable = Sablefish <i>Anoplopoma fimbria</i> ; Ad = adult; Juv = juvenile. | 646 |
| Table GFR3. Definitions and scoring bins for the exposure and sensitivity criteria used in the risk assessment. Note that either 2 or alt2 and either 3 or alt3 are used. | 647 |
| Table GFR4. Summed Exposure intensities. Boc = bocaccio <i>Sebastes paucispinis</i> ; Can = canary rockfish <i>Sebastes pinniger</i> ; Hake = Pacific hake <i>Merluccius productus</i> ; Sable = Sablefish <i>Anoplopoma fimbria</i> ; Ad = adult; Juv = juvenile. | 648 |
| Table GFR5. Final Exposure scores after sums of exposure intensity values were standardized by dividing by the estimated total suitable habitat (the sum of habitat suitability probabilities (HSP)) to get a value between 0 and 1. Boc = bocaccio <i>Sebastes paucispinis</i> ; Can = canary rockfish <i>Sebastes pinniger</i> ; Hake = Pacific hake <i>Merluccius productus</i> ; Sable = Sablefish <i>Anoplopoma fimbria</i> ; Ad = adult; Juv = juvenile. | 649 |
| Table GFR6. Raw Sensitivity scores based on literature review (see Table GFR1 for definitions of factors and scoring bins; see Appendix GFR B for details and rationale for scoring). Boc = bocaccio <i>Sebastes paucispinis</i> ; Can = canary rockfish <i>Sebastes pinniger</i> ; Hake = Pacific hake <i>Merluccius productus</i> ; Sable = Sablefish <i>Anoplopoma fimbria</i> ; Ad = adult; Juv = juvenile. | 650 |
| Table GFR7. Final Sensitivity scores: The weighted average of the across the three sensitivity criteria (with mortality given twice the weight of the other two) to get a value between 1 and 3. Boc = bocaccio <i>Sebastes paucispinis</i> ; Can = canary rockfish <i>Sebastes pinniger</i> ; Hake = Pacific hake <i>Merluccius productus</i> ; Sable = Sablefish <i>Anoplopoma fimbria</i> ; Ad = adult; Juv = juvenile. | 652 |
| Table GFR8. Final Susceptibility scores: Exposure multiplied by Sensitivity to get a value between 0 and 2. Boc = bocaccio <i>Sebastes paucispinis</i> ; Can = canary rockfish <i>Sebastes pinniger</i> ; Hake = Pacific hake <i>Merluccius productus</i> ; Sable = Sablefish <i>Anoplopoma fimbria</i> ; Ad = adult; Juv = juvenile. | 653 |
| Figure GFR1. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to aquaculture as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. | 654 |
| Figure GFR2. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to atmospheric deposition as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. | |

Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 655

Figure GFR3. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to coastal engineering as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 656

Figure GFR4. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to direct human impacts as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 657

Figure GFR5. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to inorganic pollution as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 658

Figure GFR6. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to light pollution as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 659

Figure GFR7. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to nutrient input as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 660

Figure GFR8. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to ocean based pollution as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 661

Figure GFR9. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to organic pollution as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 662

Figure GFR11. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to coastal seawater exchange (including power plants) as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little

to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible..... 664

Figure GFR12. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to sediment decrease as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 665

Figure GFR13. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to sediment increase as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 666

Figure GFR14. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to shipping activity as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 667

Figure GFR17. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to ocean acidification as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 670

Figure GFR18. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to sea surface temperature as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 671

Figure GFR19. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to ultraviolet radiation as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible. 672

Figure GFR20. Habitat Suitability Probabilities for bocaccio *Sebastes paucispinis* adult. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement..... 673

Figure GFR21. Habitat Suitability Probabilities for bocaccio *Sebastes paucispinis* juvenile. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement..... 674

Figure GFR22. Habitat Suitability Probabilities for canary *Sebastes pinniger* adult. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement..... 675

Figure GFR23. Habitat Suitability Probabilities for canary *Sebastes pinniger* juvenile. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement..... 676

| | |
|---|-----|
| Figure GFR24. Habitat Suitability Probabilities for Pacific hake <i>Merluccius productus</i> adult. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement..... | 677 |
| Figure GFR25. Habitat Suitability Probabilities for Pacific hake <i>Merluccius productus</i> juvenile. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement..... | 678 |
| Figure GFR26. Habitat Suitability Probabilities for Sablefish <i>Anoplopoma fimbria</i> adult. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement..... | 679 |
| Figure GFR27. Habitat Suitability Probabilities for Sablefish <i>Anoplopoma fimbria</i> juvenile. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement..... | 680 |
| Figure GFR28. Exposure intensity index of aquaculture for Pacific hake <i>Merluccius productus</i> adult. High = upper bicile, and low = lower bicile. | 681 |
| Figure GFR29. Exposure intensity index of atmospheric deposition of pollutants for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile..... | 682 |
| Figure GFR30. Exposure intensity index of coastal engineering for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile. | 683 |
| Figure GFR31. Exposure intensity index of direct human impacts (beach trampling) for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile..... | 684 |
| Figure GFR32. Exposure intensity index of inorganic pollution for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile. | 685 |
| Figure GFR33. Exposure intensity index of light pollution for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile. | 686 |
| Figure GFR34. Exposure intensity index of nutrient runoff for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile. | 687 |
| Figure GFR35. Exposure intensity index of ocean-based pollution for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile..... | 688 |
| Figure GFR36. Exposure intensity index of offshore oil activities for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile..... | 689 |
| Figure GFR37. Exposure intensity index of organic pollution for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile. | 690 |
| Figure GFR38. Exposure intensity index of coastal seawater exchange activity for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile..... | 691 |
| Figure GFR39. Exposure intensity index of sediment runoff decrease for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile..... | 692 |
| Figure GFR40. Exposure intensity index of sediment runoff increase for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile..... | 693 |

| | |
|--|-----|
| Figure GFR41. Exposure intensity index of shipping activity for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile. | 694 |
| Figure GFR42. Exposure intensity index of species invasions for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile. | 695 |
| Figure GFR43. Exposure intensity index of coastal trash for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile. | 696 |
| Figure GFR44. Exposure intensity index of ocean acidification for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile. | 697 |
| Figure GFR45. Exposure intensity index of sea-surface temperature for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile. | 698 |
| Figure GFR46. Exposure intensity index of ultra-violet radiation for Pacific hake <i>Merluccius productus</i> adult. High = upper tercile, Medium = middle tercile, low = lower tercile. | 699 |
| Figure GFR47. Exposure intensity index of ocean-based pollution for bocaccio <i>Sebastes paucispinis</i> rockfish adults. High = upper tercile, Medium = middle tercile, low = lower tercile. | 700 |
| Figure GFR48. Exposure intensity index of ocean-based pollution for bocaccio <i>Sebastes paucispinis</i> rockfish juveniles. High = upper tercile, Medium = middle tercile, low = lower tercile. | 701 |
| Figure GFR49. Exposure intensity index of ocean-based pollution for canary <i>Sebastes pinniger</i> rockfish adults. High = upper tercile, Medium = middle tercile, low = lower tercile. | 702 |
| Figure GFR50. Exposure intensity index of ocean-based pollution for canary <i>Sebastes pinniger</i> rockfish juveniles. High = upper tercile, Medium = middle tercile, low = lower tercile. | 703 |
| Figure GFR51. Exposure intensity index of ocean-based pollution for Pacific hake <i>Merluccius productus</i> juveniles. High = upper tercile, Medium = middle tercile, low = lower tercile. | 704 |
| Figure GFR52. Exposure intensity index of ocean-based pollution for sablefish <i>Anoplopoma fimbria</i> adults. High = upper tercile, Medium = middle tercile, low = lower tercile. | 705 |
| Figure GFR53. Exposure intensity index of ocean-based pollution for sablefish <i>Anoplopoma fimbria</i> juveniles. High = upper tercile, Medium = middle tercile, low = lower tercile. | 706 |

OVERVIEW

Our initial evaluation of groundfish vulnerability to non-fisheries risks indicates that groundfish appear to be at highest risk from systemic threats such as ocean acidification and change in average sea surface temperature. This evaluation represents a first step towards evaluating the vulnerability of groundfish to such risks.

EXECUTIVE SUMMARY

A recent development in the use of risk assessment in fisheries management is the productivity-susceptibility analyses (PSA) which have been used as an evaluation of the vulnerability of fish stocks to current fisheries management practices, based upon their susceptibility to the fishery and a suite of life history traits which indicate productivity (as a main factor in the resilience of the population). We used a modified PSA approach to provide information on the relative risk imposed by the various non-fisheries threats to the four species in the California Current. Habitat Suitability Probabilities (HSPs) describe the distribution of each species/life-history stage, and the overlap of the HSPs with the spatial distribution and intensity of the threat were used to determine the exposure to each threat (e.g. Figure GFii). Exposure combined with sensitivity to each threat provides a metric of susceptibility for the PSA.

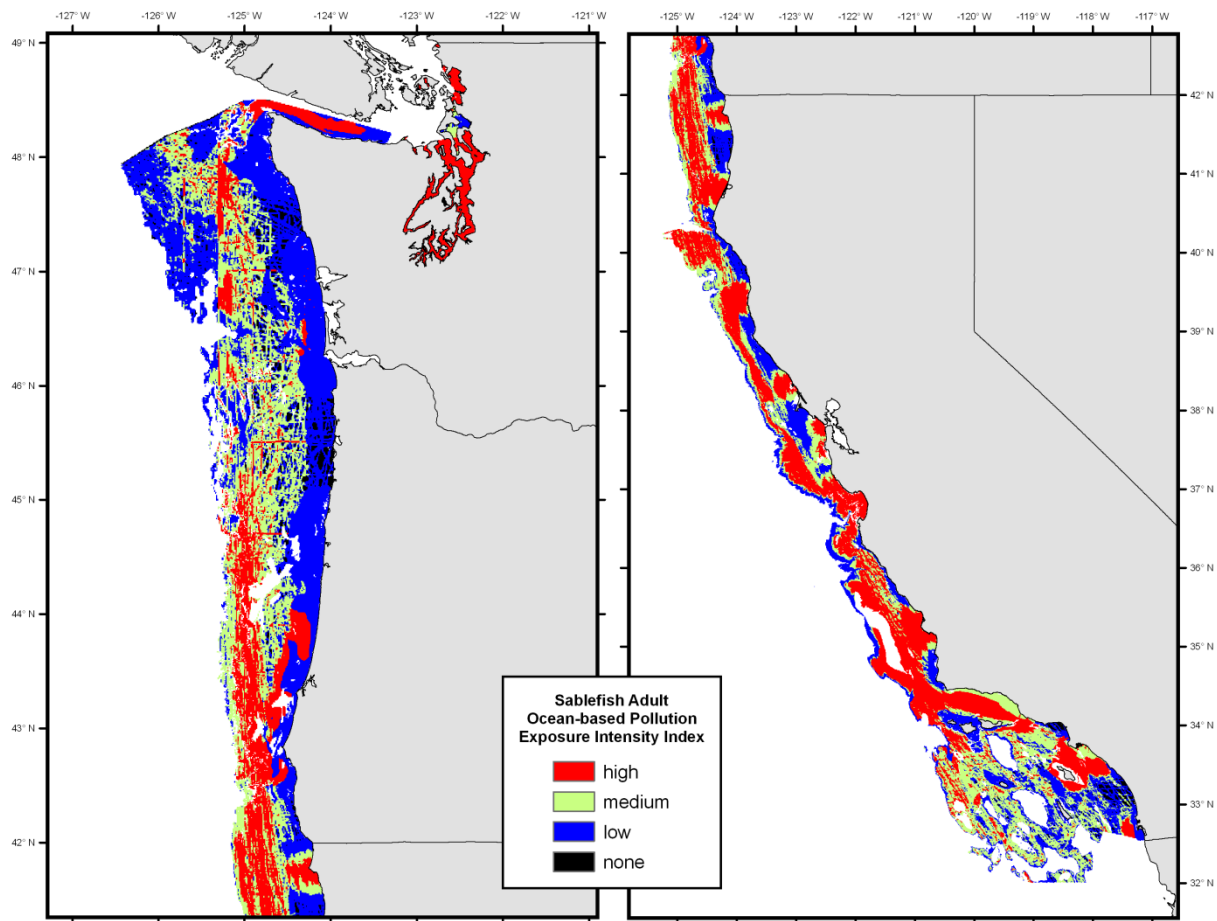


Figure GFRii. Exposure intensity index of ocean-based pollution for sablefish *Anoplopoma fimbria* adults. High = upper tercile, Medium = middle tercile, low = lower tercile.

DETAILED REPORT

INTRODUCTION

Quantitative risk assessment is a general analytical approach for describing the likelihood and magnitude of adverse consequences due to exposure to particular threats (and, if possible, cumulative impacts of multiple threats). In ecotoxicology, for example, risk is generally described using the response (or sensitivity) of a species to different levels of exposure to a threat (typically a chemical contaminant) (Suter, 2007). A recent development in the use of risk assessment in fisheries management is the productivity-susceptibility analyses (PSA) which have been used as an evaluation of the vulnerability of fish stocks to current fisheries management practices, based upon their susceptibility to the fishery and a suite of life history traits which indicate productivity (as a main factor in the resilience of the population) (Patrick et al., 2009, 2010; Hobday et al., 2011). This has been especially useful for data poor species and stock, where full assessments have not been conducted, and may not be currently feasible (Cope et al. 2011).

Both the ecotoxicological and PSA risk approaches allow an evaluation of the probability (and magnitude) of adverse effects given information about exposure to a stressor (e.g. a contaminant or a fishery) while taking into account species-specific variation in responses to the stressor (and in the case of the PSA, resilience to the impact). Information on trends is also important in evaluating whether management actions to diminish (or even stabilize) threat intensities may have been taken effectively, and this is treated elsewhere in the IEA.

In this update of the analysis on “Relative risk associated with non-fisheries threats to four focal groundfish species in the California Current” (Chapter 3 of the 2011 CCIEA), we have taken the approach of modifying the ecotoxicological/PSA approach taken last year (which was based on Samhoury and Levin, 2012) to more closely mimic the PSA approach with the goal of providing more useful and clear information on the relative risk imposed by the various non-fisheries threats to the four species in the California Current.

METHODS

FOCAL SPECIES

We re-examined the relative risk of 19 non-fisheries related threats to four groundfish species in the California current: Bocaccio (*Sebastes paucipinis*) and canary (*S. pinniger*) rockfish, Pacific hake (*Merluccius productus*), and sablefish (*Anoplopoma fimbria*). Each species is managed under the Pacific Fishery Management Council’s (PFMC) groundfish Fishery Management Plan (FMP). There are over 90 species of groundfish managed under the FMP, and the four species we examined represent species of high value (Pacific hake and sablefish) and species that are of high concern due to depleted stock levels (bocaccio and canary rockfish). These four also cover a range of productivities, variability in recruitment, migratory behavior, habitat associations, longevities, and ages at maturity, and thus are reasonably representative of the variability of life history among groundfish in the CC. For each species we examined risk to both the juvenile and adult life-stages.

Bocaccio juveniles are generally associated with inshore benthic habitats, rocks with algae, and sandy zones with eelgrass or drift algae. Juveniles gradually shift to deeper high-relief rocky habitats at depths of ~50 – 250 m; however, max depths have been reported to 478 m (Love et al. 2002).

Canary rockfish juveniles are generally associated with benthic habitats, tide pools, kelp beds, and the interface between sand and rock outcrops at depths of ~15-20 m. Juveniles shift to deeper habitat at the end of the summer and adults are commonly found near pinnacles and high-relief rocky habitats with high currents at depths of ~80 – 200m with max depths to 838 m. Canary rockfish commonly school near but not on bottom (Love et al. 2002).

Pacific hake juveniles live in shallow coastal waters, bays, and estuaries (Bailey 1981, Bailey et al. 1982, Dark 1975, Dark and Wilkins 1994, Dorn 1995, NOAA 1990, Sakuma and Ralston 1995, Smith 1995), and move to deeper water as they get older (NOAA 1990). Pacific hake school at depth during the day, then move to the surface and disperse at night for feeding (McFarlane and Beamish 1986, Sumida and Moser 1980, Tanasich et al. 1991). Adults are epi-mesopelagic (Bailey et al. 1982, NOAA 1990, Sumida and Moser 1980). Highest densities of Pacific hake are usually found between 50 and 500 m, but adults occur as deep as 920 m and as far offshore as 400 km (Bailey 1982, Bailey et al. 1982, Dark and Wilkins 1994, Dorn 1995, Hart 1973, NOAA 1990, Stauffer 1985). Spawning is greatest at depths between 130 and 500 m (Bailey et al. 1982, NOAA 1990, Smith 1995).

As juveniles, sablefish are generally found in schools near surface offshore and then migrate to inshore waters after several months (Hart 1973). As sablefish mature, they migrate offshore and live near bottom at depths to 1500 m, but are most commonly found between 366 – 915 m (Hart 1973, Schirripa 2007).

NON-FISHERIES THREATS

We continue to focus on the 19 non-fisheries related threats used in Halpern et al (2009a): aquaculture, atmospheric deposition, coastal engineering, direct human impacts, inorganic pollution, light pollution, nutrient input, ocean-based pollution, offshore oil activity, organic pollution, power plants (here referred to as “coastal seawater exchange” so as to include desalination plants, etc.), sediment runoff decrease, sediment runoff increase, shipping activity, species invasions, coastal trash, ocean acidification, sea-surface temperature anomalies, and UV radiation (see Table GFR1). These data describe the relative spatial intensity of each threat within 1-km² grid cells of the California Current. Data were downloaded from the National Center for Ecological Analysis and Synthesis website (http://www.nceas.ucsb.edu/globalmarine/ca_current_data). Each threat is described in detail in Appendix GFR B and in the supporting material of Halpern et al (2008; 2009a).

This analysis represents an attempt to synthesize and describe spatial and temporal variation in the intensity of these threats as they relate to the four groundfish species. We have highlighted particular areas (data sources, etc.) which could be improved or enhanced given sufficient time.

OVERVIEW OF RISK CALCULATION

We assess the risk that various non-fisheries threats will lead to negative effects on the adult and juvenile populations of bocaccio, canary rockfish, sablefish, and Pacific hake within the U.S. borders of the California Current Large Marine Ecosystem. As was done last year, we evaluate risk, assuming management

practices continue unchanged, based on two axes of information. However, we use different axes than were employed in that document. There, the two axes represented exposure to a threat and the sensitivity of a species/stage to that threat (from Samhouri and Levin, 2012). However, the sensitivity metric also included the intrinsic productivity of a species. The goal of risk analysis (according to NOAA Technical Memorandum NMFS-NWFSC-109, April 2011, p xvi) is “to fully explore the susceptibility of an indicator to natural or human threats, as well as the ability of the indicator to return to its previous state after being perturbed”, i.e. to assess susceptibility and resilience (or productivity). Generally these two measures have been kept separate as they represent, respectively, the effect of the threat and the intrinsic resilience of the population. Here we return to the productivity-susceptibility (PSA) approach of assessing vulnerability as put forward by Patrick et al. (2009, 2010), applying this method to non-fisheries threats.

The first axis is related to the productivity P of a species, a value based on various life history traits of the species, such as fecundity and age at maturity. The second axis is related to the susceptibility S of the population to the threats. In Patrick et al. (2009, 2010), this had to do with susceptibility to fishing, but for this risk analysis it is calculated as the product of two other values, exposure (e) and sensitivity (s) to each threat. The final value for relative risk R to each species/life history stage was then calculated as

$$R_{ij} = \sqrt{P^2 + S^2} = \sqrt{P^2 + (e * s)^2}$$

Under this framework, the risk to a species increases with Euclidean distance from the origin and productivity and susceptibility received equivalent weight in estimating risk. This is the approach developed by Patrick et al. (2009, 2010), and provides a nice visualization of the relative components of risk for each threat (e.g. Figures GFR1-19), although since the Susceptibility score is currently a relative score, the risk is not generally comparable among threats. Nor have we attempted to calculate cumulative risk in this document.

Values of P and s for each species/life history stage are averages of several sub-scores, each based on standardized set of criteria. The value for e is a product of metrics of habitat suitability and threat intensity across the area of the California Current.

PRODUCTIVITY AXIS

Productivity P for each species was taken from Cope et al. (2011), which used a weighted average of 10 criteria (The intrinsic rate of population growth, r ; maximum age; maximum size; the von Bertalanffy growth coefficient k ; natural mortality rate M ; fecundity; a metric of breeding strategy; a metric of temporal recruitment variability; age at maturity; and mean trophic level). Each criterion was designated 1, 2, or 3 (Table GFR2). Naturally, values for P varied only across species, not across life history stages within each species.

Eventually, the productivity axis could be expanded to reflect resilience to the particular threat including productivity and other factors specific to the particular threat being considered

SUSCEPTIBILITY AXIS

Susceptibility is calculated as the product of Exposure and Sensitivity. This is similar to the concept from Patrick et al. (2009; 2010) for fisheries susceptibility. In that case exposure can be thought of as the areal overlap of fishing and habitat along with the intensity of fishing, and sensitivity can be thought of as catchability and selectivity of the fisheries for that species, along with habitat impacts, etc. Here we have

instead the areal overlap of the threat and the habitat for that species/stage, along with the intensity of the threat for exposure, while the sensitivity of the species/stage to the threat represents direct and indirect impacts to that species/stage.

EXPOSURE

The value for e is a measure of overlap between each species' spatial distribution and the relative intensity of each threat across the area of the California Current. For this calculation we took advantage of two published GIS data sets. The exposure values are the same as those in the previous CCIEA, except divided by 2 to get back to the simpler scale of 0 to 1.

First, we used Habitat Suitability Probabilities to describe the distribution of each species/life-history stage (Figs. 18-25). HSP values describe the probability of occurrence of each species/life history stage within the U.S. boundaries of the California Current. Briefly, the HSP values were calculated for the National Marine Fisheries Service (NMFS) Northwest Region and the Pacific Fishery Management Council in support of an Environmental Impact Statement (EIS) to consider the designation and conservation of Essential Fish Habitat (EFH) for Pacific Coast Groundfish (<http://www.nwr.noaa.gov/Groundfish-Halibut/Groundfish-Fishery-Management/NEPA-Documents/EFH-Final-EIS.cfm>). HSP values were generated from merged habitat and bathymetry GIS data and a Bayesian Network model that incorporated information about species' habitat preferences (bottom type and depth preferences) from NMFS trawl surveys and the Habitat Use Database (see Figures GFR20-27 and Appendix GFR A for more details). We used data if HSP values were ≥ 0.01 because HSP values for habitat $< .01$ were not retained during the modeling.

Second, we used data from Halpern et al (2009a) to describe the spatial intensity of each threat throughout the distribution of each species/life history stage. These data layers provide a relative score for the intensity of each threat (log-transformed and rescaled between 0 and 1) in 1-km² grid cells across the entire California Current. The data sources and calculations for each threat are described in detail in the supporting materials of Halpern et al (2008; 2009a), and briefly outlined in Appendix GFR B.

HSP data layers for each species/life history stage and the 19 threat data layers were brought into ArcView version 9.3 for analysis. The HSP data layer was then multiplied by each threat data layer to calculate the exposure intensity (ei) for each threat across the distribution of each species/life history stage (Table GFR4). Thus, the threat j intensity scores were weighted by the probability of species/life history stage i occurring in each 1-km² cell. For each cell we then had

$$ei_{ij} = HSP_i * t_j$$

where t_j is the intensity (log-transformed and scaled 0-1) of threat j (Table GFR5).

For visual representation, we classified the distribution of ei_{ij} values into three terciles (high, medium, and low), although offshore oil activity data was divided into only high and low categories based on the median value because there were so few unique values.

For the final exposure score e , we summed all exposure intensity values for each species/life history stage i /threat j . We then scaled each sum between 0 and 1, with 1 corresponding to the sum of the HSP values for that species/life history stage (theoretically a measure of exposure if threat intensity were 1 everywhere). This is a slightly different approach than that taken in the last version of the CCIEA. There the exposure scores were rescaled between 1 and 3 (instead of 0-1) with the threat with the greatest summed exposure intensity score for each species/stage acting as the scaling factor, such that that threat would receive a 3 for that

species/stage, even if the total exposure intensity score was less than the sum of the HSP values (whereas in the current approach, a value of 1 would only be achieved if the total exposure intensity score was equal to the sum of the HSP values. i.e. if the level was the same everywhere).

SENSITIVITY

Sensitivity criteria include one criterion that describes the mortality induced by a threat and two more that describe the behavioral and physiological responses to a threat. We used the definitions in Table GFR3 to score the criteria (Table GFR6). Scoring for these criteria was based on the primary literature and is addressed in detail in Appendix GFR B. These three criteria were then averaged (with mortality given twice the weight of the other two) for each threat for each species/life history stage to arrive at the final Sensitivity scores between 1 and 3 (Table GFR7). Again, this is different than the definition of Sensitivity from the last CCIEA. In that document, the impacts of each threat on the individuals within the population and the resilience of the population (productivity) were bundled together in “Sensitivity”. However, in returning to the PSA concept, Productivity and Sensitivity scores are kept separate.

There are some rather large remaining issues with quantifying sensitivity. We do not have a direct link between the actual levels of the threat in the environment and the sensitivity of the species. Therefore, we cannot state what the true sensitivity is to the current level of threat, nor can we comfortably compare threats. The sensitivity score should be linked to either the maximum level of a threat (i.e. linked to a value of 1) or to some other consistent value across threats. We requested information on the maximum value observed for each of the threats from Halpern et al. (2009a), but they were unable to provide those values in time for this document. Future work should link the threat intensities and sensitivity as well as explore the suitability of using a $\log(x+1)$ transform for scaling the level of the threat.

RESULTS

EXPOSURE INTENSITY

The calculated exposure intensity index for each species/life-history stage/threat varied throughout the distribution of each species for most threats. As examples, Figures GFR28 – 46 show the exposure intensity for Pacific hake adults for each of the 19 threats. There are several threats that show very little overlap with hake adult habitats, e.g. aquaculture (fish farms), coastal engineering, direct human impacts (trampling), offshore oil activities, coastal seawater exchange, and coastal trash (Figures GFR28, 30, 31, 36, 38, & 43, respectively). Spatially expansive threats affect nearly the entire distribution of adult hake, e.g. atmospheric deposition, ocean-based pollution, shipping, and the three climate change threats – ocean acidification, sea surface temperature, and UV radiation (Figures GFR29, 35, 41, 44 – 46, respectively). Threats that occur as point-sources show relatively high exposure intensity in coastal areas and low or no exposure in offshore portions of their distribution, e.g. inorganic pollution, light pollution, nutrient input, organic pollution, sediment runoff decrease and increase, and species invasions (Figures GFR32 – 34, 37, 39 – 40, and 42, respectively).

Across species/life history stages, exposure intensity generally varies in relation to the offshore distribution of adult habitats and the nearshore concentration of juvenile habitats. Thus, juveniles of most species tend to be exposed to higher intensities of point-source threats because of their higher probabilities

of occurrence in nearshore habitats, while adults tend to have much broader exposure to spatially expansive threats, such as atmospheric deposition or the climate change threats. One generality among these four species may be that in the waters off Oregon and Washington, we found higher exposure intensities for juveniles as a result of their nearshore habitat, while adults experience broader, higher exposure intensities in waters off California due to broader habitat occurrence (compare Figures GFR47 & 48, 49 & 50, and 52 & 53).

RELATIVE RISK

In general, the current work indicates that the most spatially expansive threats are more likely to be of greater relative risk to each of the four species than threats related to point-sources (Figures GFR1-19). However, without a real link between the current and anticipated levels of the threat and the impacts, these results only indicate expansiveness/overlap of each threat, and not the actual potential impact.

CONCLUSIONS

Our analysis builds on the risk assessment framework of others and that of last year's CCIEA, making progress towards a method that will allow for comparison of relative risk among multiple non-fisheries threats, and potentially cumulative risk across threats. This framework will show which threats are relevant to focal species and provides a basis for prioritizing which threats are in need of management actions. Rapid assessments of other species can then be easily integrated into this framework.

Future versions of the CCIEA should further build upon this work by linking the actual current and anticipated threat exposure levels to the associated sensitivity scores, considering factors other than productivity in evaluating the resilience of a population to the effects of various threats, exploring the appropriateness of the $\log(x+1)$ transformation used before standardizing the exposure on a 0-1 scale for each threat.

Table GFR1. List of non-fisheries threats considered.

Threats

Aquaculture

Atmospheric deposition

Coastal engineering

Direct human impacts

Inorganic pollution

Light pollution

Nutrient input

Ocean-based pollution

Organic pollution

Offshore oil activities

Coastal seawater exchange

Sediment decrease

Sediment increase

Shipping activity

Species invasions

Coastal Trash

Table GFR2. Raw and final Productivity scores. A weighted average of the ten scores (Cope et al. 2011) is used, for final values ranging between 1 and 3. Boc = bocaccio *Sebastes paucispinis*; Can = canary rockfish *Sebastes pinniger*; Hake = Pacific hake *Merluccius productus*; Sable = Sablefish *Anoplopoma fimbria*; Ad = adult; Juv = juvenile.

| Factor | Weight | Boc Ad | Boc Juv | Can Ad | Can Juv | Hake Ad | Hake Juv | Sable Ad | Sable Juv |
|-------------------------|---------------|-------------------|--------------------|-------------------|--------------------|--------------------|---------------------|---------------------|----------------------|
| <i>r</i> | 2 | 1 | 1 | 1 | 1 | 1.5 | 1.5 | 1.5 | 1.5 |
| Max age | 2 | 1 | 1 | 1 | 1 | 2 | 2 | 1 | 1 |
| Max size | 1 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| <i>k</i> | 2 | 1 | 1 | 1.5 | 1.5 | 3 | 3 | 2.5 | 2.5 |
| <i>M</i> | 2 | 1 | 1 | 1 | 1 | 2 | 2 | 1 | 1 |
| Fecundity | 1 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| Breeding strategy | 2 | 1 | 1 | 1 | 1 | 3 | 3 | 3 | 3 |
| Recruitment variability | 2 | 1 | 1 | 1.5 | 1.5 | 1 | 1 | 1 | 1 |
| Age at Maturity | 2 | 2 | 2 | 1 | 1 | 2 | 2 | 1 | 1 |
| Trophic level | 2 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Weighted Average (1-3) | | 1.28 | 1.28 | 1.28 | 1.28 | 2.00 | 2.00 | 1.61 | 1.61 |

Table GFR3. Definitions and scoring bins for the exposure and sensitivity criteria used in the risk assessment. Note that either 2 or alt2 and either 3 or alt3 are used.

| Criteria | Explanation of criteria | Exposure/Sensitivity scores | | |
|--|---|---|--------------------------------------|--------------------------------|
| Exposure: Spatial intensity | The overlap between the probability of species occurrence (HSP) and the relative intensity of a threat. | Standardized distribution (scale=1-3) of the sum of species-specific exposure intensity values. | | |
| Sensitivity Factors: | | Low (1) | Moderate(2) | High(3) |
| 1. Mortality (weight = 2) | Direct effect of threat on population-wide average mortality rate of a species | Negligible | Sub-lethal | Lethal |
| 2. Behavioral response (weight =1) | Population-wide effect of threat on behavior of a species | Negligible behavioral response | Moderate behavioral response | Severe behavioral response |
| Alt 2. Effect of behavioral response (weight =1) | Population-wide change in sensitivity to threat due to behavioral response | Response reduces sensitivity | Response does not change sensitivity | Response increases sensitivity |
| 3. Physiological response (weight =1) | Population-wide effect of threat on behavior or physiology of a species | Negligible physiological response | Moderate physiological response | Severe physiological response |
| Alt 3. Effect of physiological response (weight = 1) | Population-wide change in sensitivity to threat due to physiological response | Response reduces sensitivity | Response does not change sensitivity | Response increases sensitivity |

Table GFR4. Summed Exposure intensities. Boc = bocaccio *Sebastes paucispinis*; Can = canary rockfish *Sebastes pinniger*; Hake = Pacific hake *Merluccius productus*; Sable = Sablefish *Anoplopoma fimbria*; Ad = adult; Juv = juvenile.

| Threat | Boc Ad | Boc Juv | Can Ad | Can Juv | Hake Ad | Hake Juv | Sable Ad | Sable Juv |
|---------------------------|--------|---------|--------|---------|---------|----------|----------|-----------|
| Aquaculture | 0 | 0 | 0 | 0 | 2 | 0 | 2 | 0 |
| Atmospheric deposition | 2,866 | 9,481 | 4,180 | 11,092 | 42,572 | 55 | 70,199 | 25,431 |
| Coastal engineering | 2 | 105 | 2 | 263 | 377 | 0 | 224 | 11 |
| Direct human impacts | 1 | 121 | 1 | 100 | 170 | 0 | 63 | 51 |
| Inorganic pollution | 143 | 935 | 202 | 1,505 | 1,977 | 7 | 1,142 | 421 |
| Light pollution | 173 | 913 | 189 | 1,859 | 2,657 | 7 | 2,549 | 681 |
| Nutrient input | 473 | 2,482 | 883 | 3,629 | 5,100 | 14 | 3,221 | 1,597 |
| Ocean-based pollution | 1,314 | 4,525 | 2,081 | 6,678 | 14,625 | 19 | 18,549 | 6,883 |
| Offshore oil activities | 1 | 2 | 1 | 6 | 6 | 0 | 4 | 0 |
| Organic pollution | 416 | 2,568 | 969 | 3,743 | 4,838 | 10 | 2,737 | 1,488 |
| Coastal seawater exchange | 2 | 30 | 2 | 51 | 43 | 0 | 25 | 0 |
| Sediment decrease | 689 | 3,332 | 1,282 | 5,095 | 7,562 | 18 | 5,450 | 2,427 |
| Sediment increase | 1,786 | 7,384 | 3,298 | 10,506 | 16,773 | 18 | 11,868 | 5,975 |
| Shipping activity | 6 | 254 | 8 | 397 | 2,359 | 0 | 132 | 89 |
| Species invasions | 932 | 4,231 | 1,443 | 5,359 | 10,043 | 16 | 6,715 | 3,327 |
| Coastal trash | 3 | 219 | 3 | 408 | 266 | 1 | 94 | 41 |
| Ocean Acidification | 4,579 | 12,840 | 7,778 | 20,410 | 59,300 | 65 | 104,895 | 36,161 |
| Sea Surface Temperature | 2,352 | 8,710 | 4,947 | 10,870 | 32,291 | 38 | 49,054 | 20,411 |
| UV radiation | 4,411 | 12,526 | 7,354 | 19,374 | 57,542 | 66 | 100,313 | 34,891 |

Table GFR5. Final Exposure scores after sums of exposure intensity values were standardized by dividing by the estimated total suitable habitat (the sum of habitat suitability probabilities (HSP)) to get a value between 0 and 1. Boc = bocaccio *Sebastes paucispinis*; Can = canary rockfish *Sebastes pinniger*; Hake = Pacific hake *Merluccius productus*; Sable = Sablefish *Anoplopoma fimbria*; Ad = adult; Juv = juvenile.

| Threat | Boc Ad | Boc Juv | Can Ad | Can Juv | Hake Ad | Hake Juv | Sable Ad | Sable Juv |
|---------------------------|--------|---------|--------|---------|---------|----------|----------|-----------|
| Aquaculture | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Atmospheric deposition | 0.53 | 0.62 | 0.45 | 0.45 | 0.61 | 0.72 | 0.57 | 0.60 |
| Coastal engineering | 0.00 | 0.01 | 0.00 | 0.01 | 0.01 | 0.00 | 0.00 | 0.00 |
| Direct human impacts | 0.00 | 0.01 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Inorganic pollution | 0.03 | 0.06 | 0.02 | 0.06 | 0.03 | 0.09 | 0.01 | 0.01 |
| Light pollution | 0.03 | 0.06 | 0.02 | 0.08 | 0.04 | 0.09 | 0.02 | 0.02 |
| Nutrient input | 0.09 | 0.16 | 0.09 | 0.15 | 0.07 | 0.18 | 0.03 | 0.04 |
| Ocean-based pollution | 0.24 | 0.30 | 0.22 | 0.27 | 0.21 | 0.25 | 0.15 | 0.16 |
| Offshore oil activities | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Organic pollution | 0.08 | 0.17 | 0.10 | 0.15 | 0.07 | 0.13 | 0.02 | 0.04 |
| Coastal seawater exchange | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Sediment decrease | 0.13 | 0.22 | 0.14 | 0.21 | 0.11 | 0.24 | 0.04 | 0.06 |
| Sediment increase | 0.33 | 0.48 | 0.35 | 0.43 | 0.24 | 0.24 | 0.10 | 0.14 |
| Shipping activity | 0.00 | 0.02 | 0.00 | 0.02 | 0.03 | 0.00 | 0.00 | 0.00 |
| Species invasions | 0.17 | 0.28 | 0.15 | 0.22 | 0.14 | 0.21 | 0.05 | 0.08 |
| Coastal trash | 0.00 | 0.01 | 0.00 | 0.02 | 0.00 | 0.01 | 0.00 | 0.00 |
| Ocean Acidification | 0.84 | 0.84 | 0.83 | 0.83 | 0.85 | 0.85 | 0.86 | 0.86 |
| Sea Surface Temperature | 0.43 | 0.57 | 0.53 | 0.44 | 0.46 | 0.50 | 0.40 | 0.48 |
| UV radiation | 0.81 | 0.82 | 0.79 | 0.79 | 0.82 | 0.86 | 0.82 | 0.83 |

Table GFR6. Raw Sensitivity scores based on literature review (see Table GFR1 for definitions of factors and scoring bins; see Appendix GFR B for details and rationale for scoring). Boc = bocaccio *Sebastes paucispinis*; Can = canary rockfish *Sebastes pinniger*; Hake = Pacific hake *Merluccius productus*; Sable = Sablefish *Anoplopoma fimbria*; Ad = adult; Juv = juvenile.

| Criterion | Boc Ad | Boc Juv | Can Ad | Can Juv | Hake Ad | Hake Juv | Sable Ad | Sable Juv |
|----------------------------------|--------|---------|--------|---------|---------|----------|----------|-----------|
| 1. Mortality | | | | | | | | |
| Aquaculture | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Atmospheric deposition | 2 | 3 | 2 | 3 | 2 | 3 | 2 | 3 |
| Coastal engineering | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Direct human impacts | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Inorganic pollution | 2 | 3 | 2 | 3 | 2 | 3 | 2 | 3 |
| Light pollution | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Nutrient input | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 |
| Ocean-based pollution | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Offshore oil activities | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Organic pollution | 2 | 3 | 2 | 3 | 2 | 3 | 2 | 3 |
| Coastal seawater exchange | 1 | 3 | 1 | 3 | 1 | 3 | 1 | 3 |
| Sediment decrease | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Sediment increase | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Shipping activity | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Species invasions | 2 | 3 | 2 | 3 | 2 | 3 | 2 | 3 |
| Coastal trash | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Ocean Acidification | 2 | 3 | 2 | 3 | 2 | 3 | 2 | 3 |
| Sea Surface Temperature | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| UV radiation | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 |
| 2. Behavioral response | | | | | | | | |
| Aquaculture | 3 | 3 | 3 | 3 | 2 | 2 | 2 | 2 |
| Atmospheric deposition | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Coastal engineering | 3 | 3 | 3 | 3 | 2 | 2 | 1 | 2 |
| Direct human impacts | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Inorganic pollution | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Light pollution | 2 | 2 | 2 | 2 | 2 | 2 | 1 | 1 |
| Nutrient input | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Ocean-based pollution | 3 | 3 | 3 | 3 | 2 | 2 | 2 | 2 |

| Criterion | Boc Ad | Boc Juv | Can Ad | Can Juv | Hake Ad | Hake Juv | Sable Ad | Sable Juv |
|----------------------------------|--------|---------|--------|---------|---------|----------|----------|-----------|
| Offshore oil activities | 3 | 3 | 3 | 3 | 2 | 2 | 1 | 2 |
| Organic pollution | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Coastal seawater exchange | 2 | 3 | 2 | 3 | 2 | 2 | 2 | 2 |
| Sediment decrease | 1 | 1 | 1 | 1 | 2 | 2 | 1 | 2 |
| Sediment increase | 1 | 1 | 1 | 1 | 2 | 2 | 1 | 2 |
| Shipping activity | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Species invasions | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| Coastal trash | 3 | 3 | 3 | 3 | 2 | 2 | 1 | 2 |
| Ocean Acidification | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| Sea Surface Temperature | 3 | 3 | 3 | 3 | 1 | 1 | 1 | 1 |
| UV radiation | 1 | 2 | 1 | 2 | 2 | 2 | 1 | 1 |
| 3. Physiological response | | | | | | | | |
| Aquaculture | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Atmospheric deposition | 3 | 2 | 3 | 2 | 3 | 2 | 3 | 2 |
| Coastal engineering | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Direct human impacts | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Inorganic pollution | 3 | 2 | 3 | 2 | 3 | 2 | 3 | 2 |
| Light pollution | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Nutrient input | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Ocean-based pollution | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Offshore oil activities | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Organic pollution | 3 | 2 | 3 | 2 | 3 | 2 | 3 | 2 |
| Coastal seawater exchange | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Sediment decrease | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Sediment increase | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Shipping activity | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Species invasions | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Coastal trash | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Ocean Acidification | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Sea Surface Temperature | 3 | 3 | 3 | 3 | 2 | 2 | 2 | 2 |
| UV radiation | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |

Table GFR7. Final Sensitivity scores: The weighted average of the across the three sensitivity criteria (with mortality given twice the weight of the other two) to get a value between 1 and 3. Boc = bocaccio *Sebastes paucispinis*; Can = canary rockfish *Sebastes pinniger*; Hake = Pacific hake *Merluccius productus*; Sable = Sablefish *Anoplopoma fimbria*; Ad = adult; Juv = juvenile.

| Threat | Boc Ad | Boc Juv | Can Ad | Can Juv | Hake Ad | Hake Juv | Sable Ad | Sable Juv |
|----------------------------------|--------|---------|--------|---------|---------|----------|----------|-----------|
| Aquaculture | 1.75 | 1.75 | 1.75 | 1.75 | 1.50 | 1.50 | 1.50 | 1.50 |
| Atmospheric deposition | 2.25 | 2.50 | 2.25 | 2.50 | 2.25 | 2.50 | 2.25 | 2.50 |
| Coastal engineering | 1.75 | 1.75 | 1.75 | 1.75 | 1.50 | 1.50 | 1.25 | 1.50 |
| Direct human impacts | 1.25 | 1.25 | 1.25 | 1.25 | 1.25 | 1.25 | 1.25 | 1.25 |
| Inorganic pollution | 2.25 | 2.50 | 2.25 | 2.50 | 2.25 | 2.50 | 2.25 | 2.50 |
| Light pollution | 1.50 | 1.50 | 1.50 | 1.50 | 1.50 | 1.50 | 1.25 | 1.25 |
| Nutrient input | 1.25 | 1.75 | 1.25 | 1.75 | 1.25 | 1.75 | 1.25 | 1.75 |
| Ocean-based pollution | 2.25 | 2.25 | 2.25 | 2.25 | 2.00 | 2.00 | 2.00 | 2.00 |
| Offshore oil activities | 1.75 | 1.75 | 1.75 | 1.75 | 1.50 | 1.50 | 1.25 | 1.50 |
| Organic pollution | 2.25 | 2.50 | 2.25 | 2.50 | 2.25 | 2.50 | 2.25 | 2.50 |
| Coastal seawater exchange | 1.50 | 2.75 | 1.50 | 2.75 | 1.50 | 2.50 | 1.50 | 2.50 |
| Sediment decrease | 1.25 | 1.25 | 1.25 | 1.25 | 1.50 | 1.50 | 1.25 | 1.50 |
| Sediment increase | 1.75 | 1.75 | 1.75 | 1.75 | 2.00 | 2.00 | 1.75 | 2.00 |
| Shipping activity | 1.50 | 1.50 | 1.50 | 1.50 | 1.50 | 1.50 | 1.50 | 1.50 |
| Species invasions | 2.25 | 2.75 | 2.25 | 2.75 | 2.25 | 2.75 | 2.25 | 2.75 |
| Coastal trash | 2.25 | 2.25 | 2.25 | 2.25 | 2.00 | 2.00 | 1.75 | 2.00 |
| Ocean Acidification | 2.25 | 2.75 | 2.25 | 2.75 | 2.25 | 2.75 | 2.25 | 2.75 |
| Sea Surface Temperature | 2.50 | 2.50 | 2.50 | 2.50 | 1.75 | 1.75 | 1.75 | 1.75 |
| UV radiation | 1.00 | 1.75 | 1.00 | 1.75 | 1.25 | 1.75 | 1.00 | 1.50 |

Table GFR8. Final Susceptibility scores: Exposure multiplied by Sensitivity to get a value between 0 and 2.
Boc = bocaccio *Sebastes paucispinis*; Can = canary rockfish *Sebastes pinniger*; Hake = Pacific hake *Merluccius productus*; Sable = Sablefish *Anoplopoma fimbria*; Ad = adult; Juv = juvenile.

| Threat | Boc Ad | Boc Juv | Can Ad | Can Juv | Hake Ad | Hake Juv | Sable Ad | Sable Juv |
|----------------------------------|--------|---------|--------|---------|---------|----------|----------|-----------|
| Aquaculture | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Atmospheric deposition | 0.66 | 0.93 | 0.56 | 0.68 | 0.76 | 1.08 | 0.72 | 0.90 |
| Coastal engineering | 0.00 | 0.01 | 0.00 | 0.01 | 0.00 | 0.00 | 0.00 | 0.00 |
| Direct human impacts | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Inorganic pollution | 0.03 | 0.09 | 0.03 | 0.09 | 0.04 | 0.14 | 0.01 | 0.01 |
| Light pollution | 0.02 | 0.03 | 0.01 | 0.04 | 0.02 | 0.05 | 0.01 | 0.00 |
| Nutrient input | 0.02 | 0.12 | 0.02 | 0.11 | 0.02 | 0.14 | 0.01 | 0.03 |
| Ocean-based pollution | 0.30 | 0.37 | 0.28 | 0.34 | 0.21 | 0.25 | 0.15 | 0.16 |
| Offshore oil activities | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Organic pollution | 0.10 | 0.25 | 0.13 | 0.23 | 0.09 | 0.20 | 0.03 | 0.05 |
| Coastal seawater exchange | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Sediment decrease | 0.03 | 0.05 | 0.03 | 0.05 | 0.05 | 0.12 | 0.01 | 0.03 |
| Sediment increase | 0.25 | 0.36 | 0.27 | 0.32 | 0.24 | 0.24 | 0.07 | 0.14 |
| Shipping activity | 0.00 | 0.01 | 0.00 | 0.01 | 0.02 | 0.00 | 0.00 | 0.00 |
| Species invasions | 0.21 | 0.49 | 0.19 | 0.38 | 0.18 | 0.37 | 0.07 | 0.14 |
| Coastal trash | 0.00 | 0.02 | 0.00 | 0.02 | 0.00 | 0.01 | 0.00 | 0.00 |
| Ocean Acidification | 1.05 | 1.48 | 1.04 | 1.45 | 1.06 | 1.49 | 1.07 | 1.50 |
| Sea Surface Temperature | 0.65 | 0.86 | 0.80 | 0.66 | 0.35 | 0.37 | 0.30 | 0.36 |
| UV radiation | 0.00 | 0.62 | 0.00 | 0.59 | 0.21 | 0.65 | 0.00 | 0.41 |

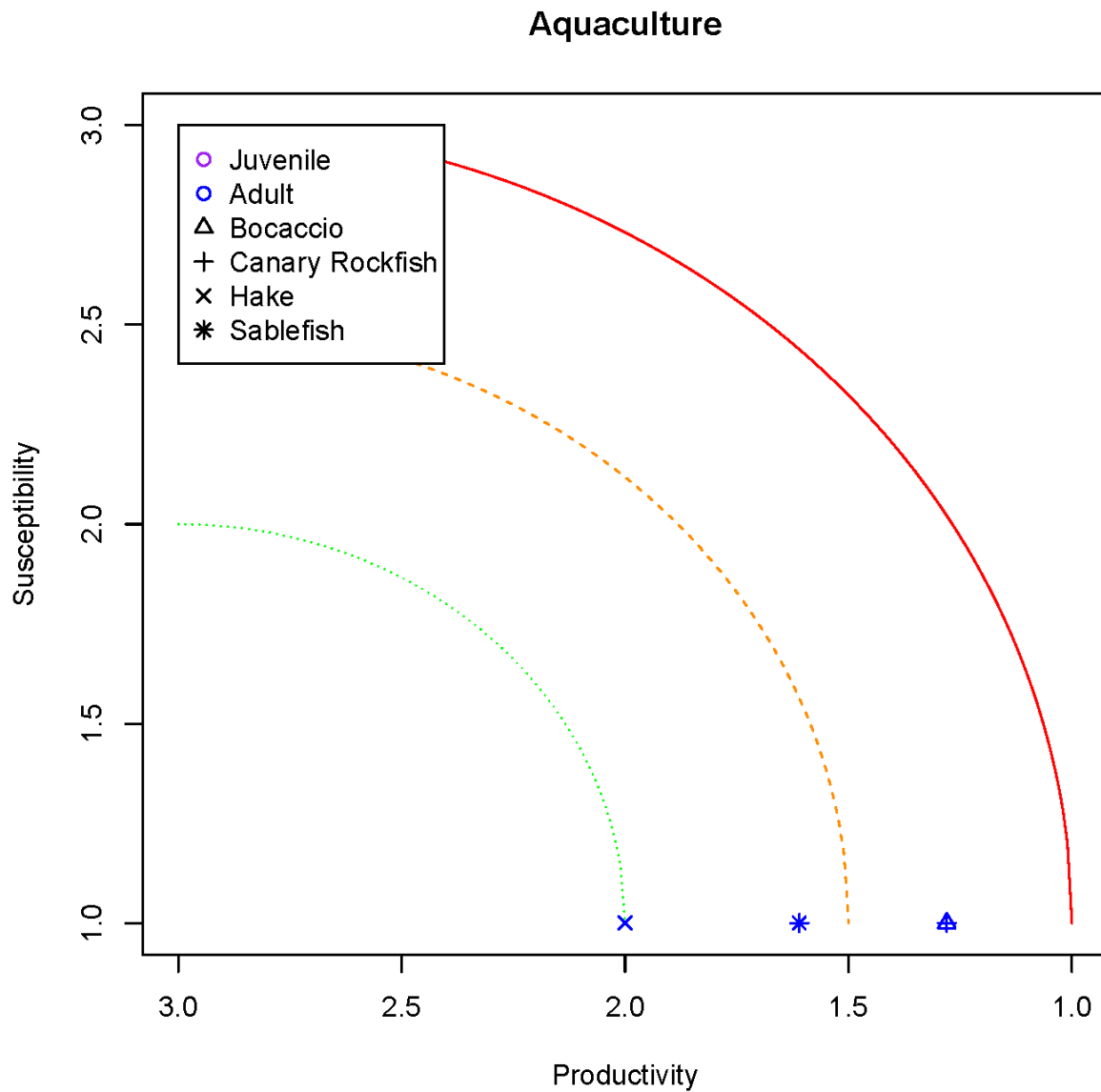


Figure GFR1. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to aquaculture as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

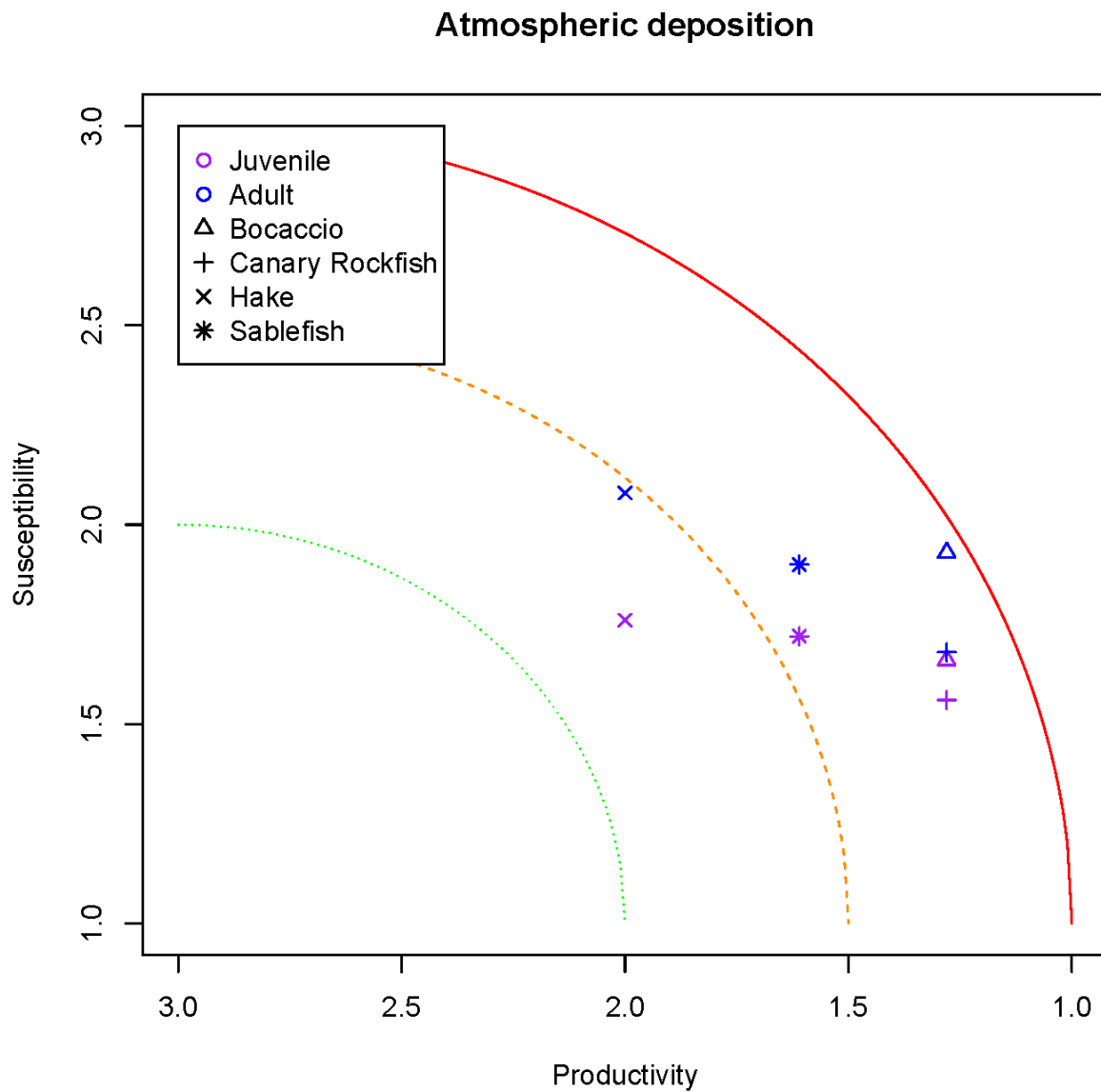


Figure GFR2. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to atmospheric deposition as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

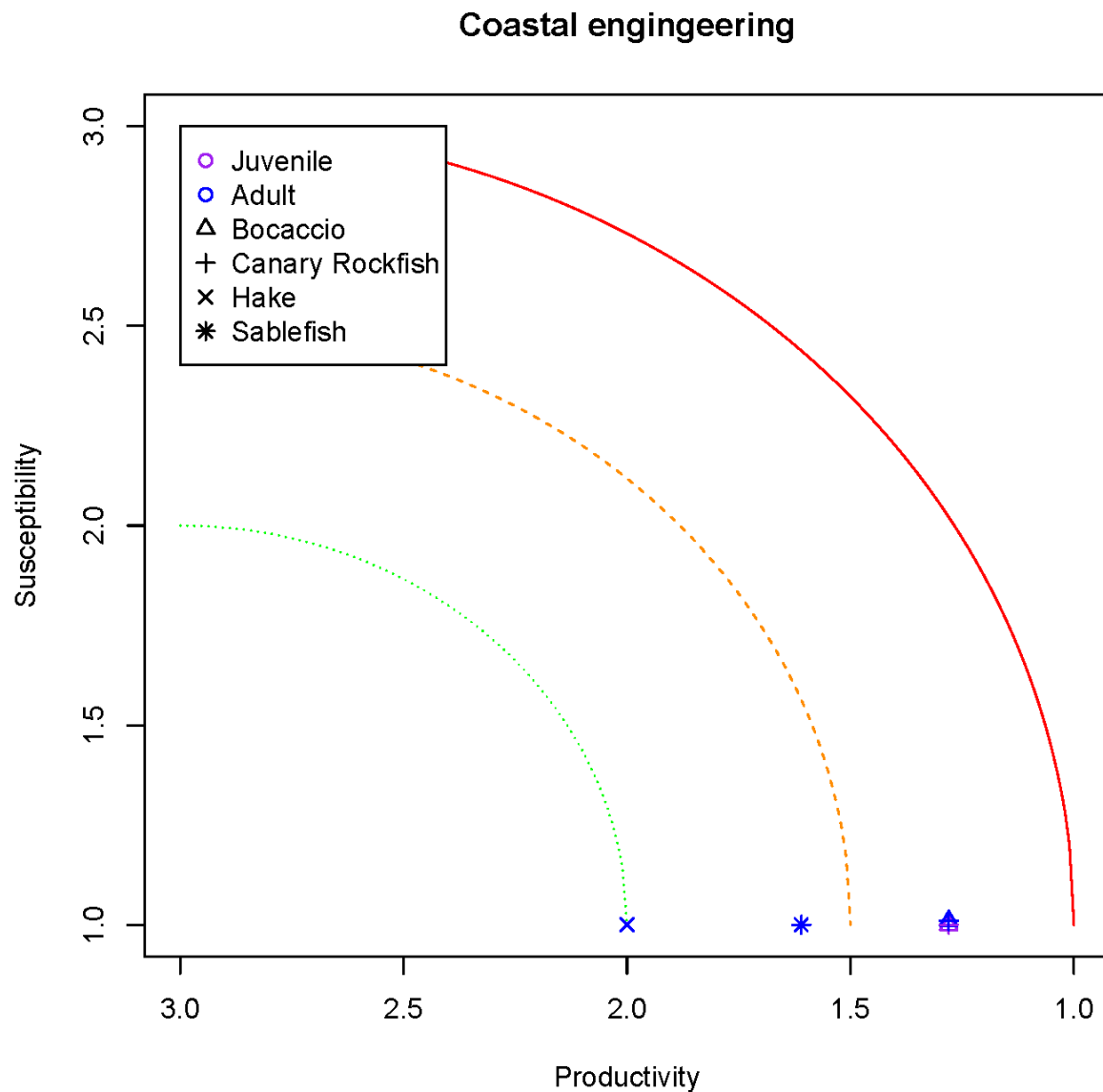


Figure GFR3. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to coastal engineering as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

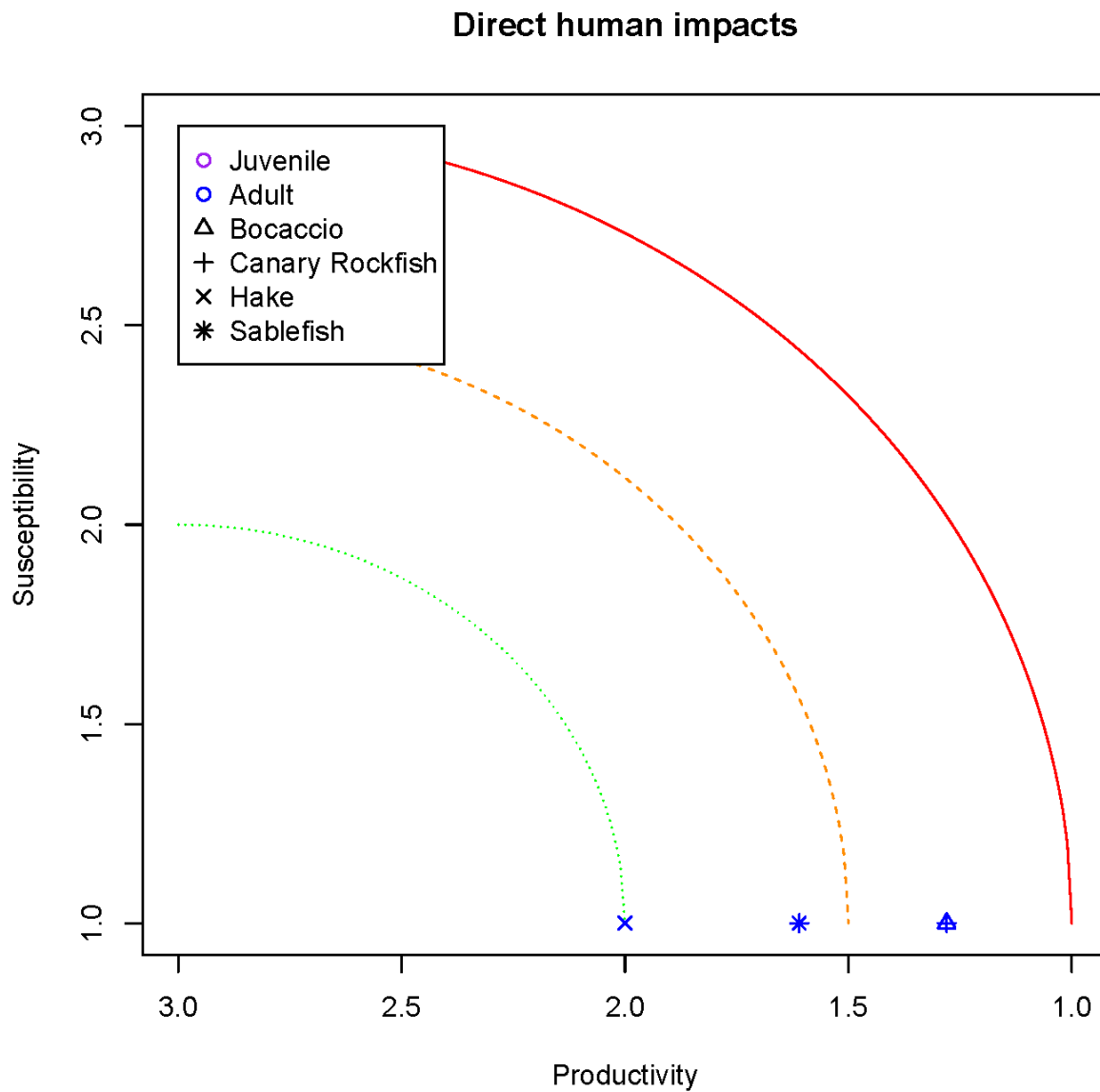


Figure GFR4. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to direct human impacts as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

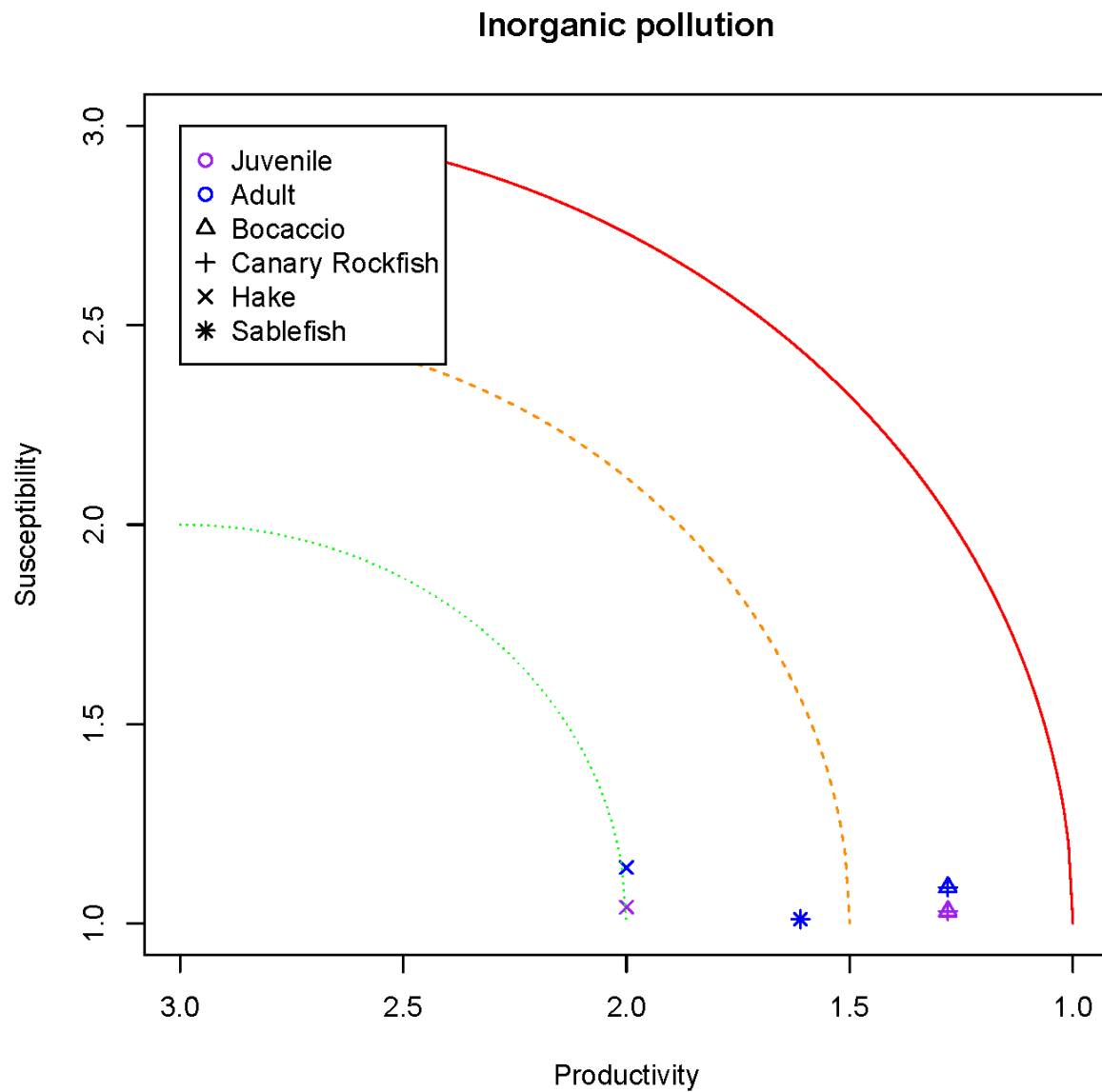


Figure GFR5. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to inorganic pollution as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

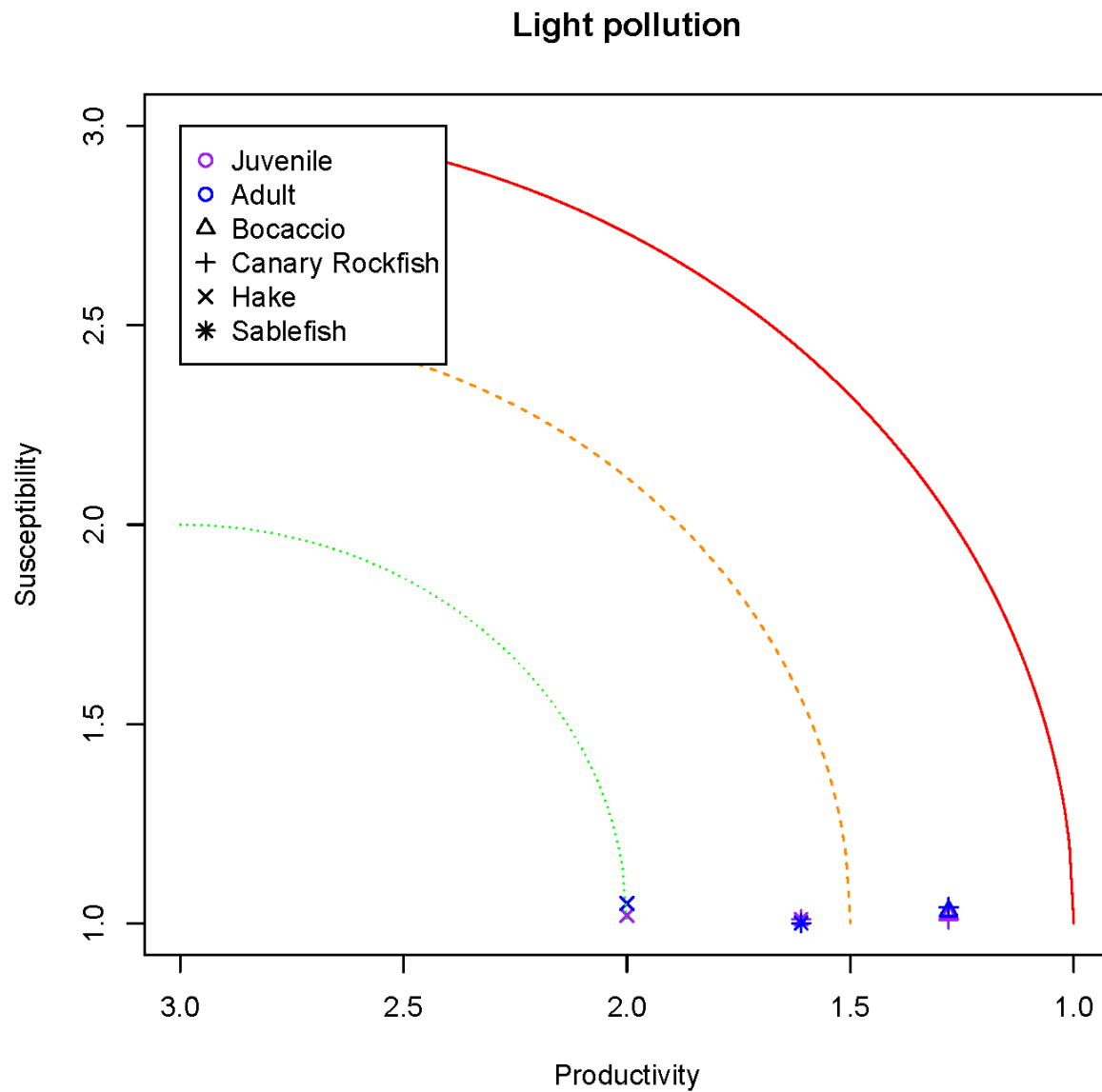


Figure GFR6. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to light pollution as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

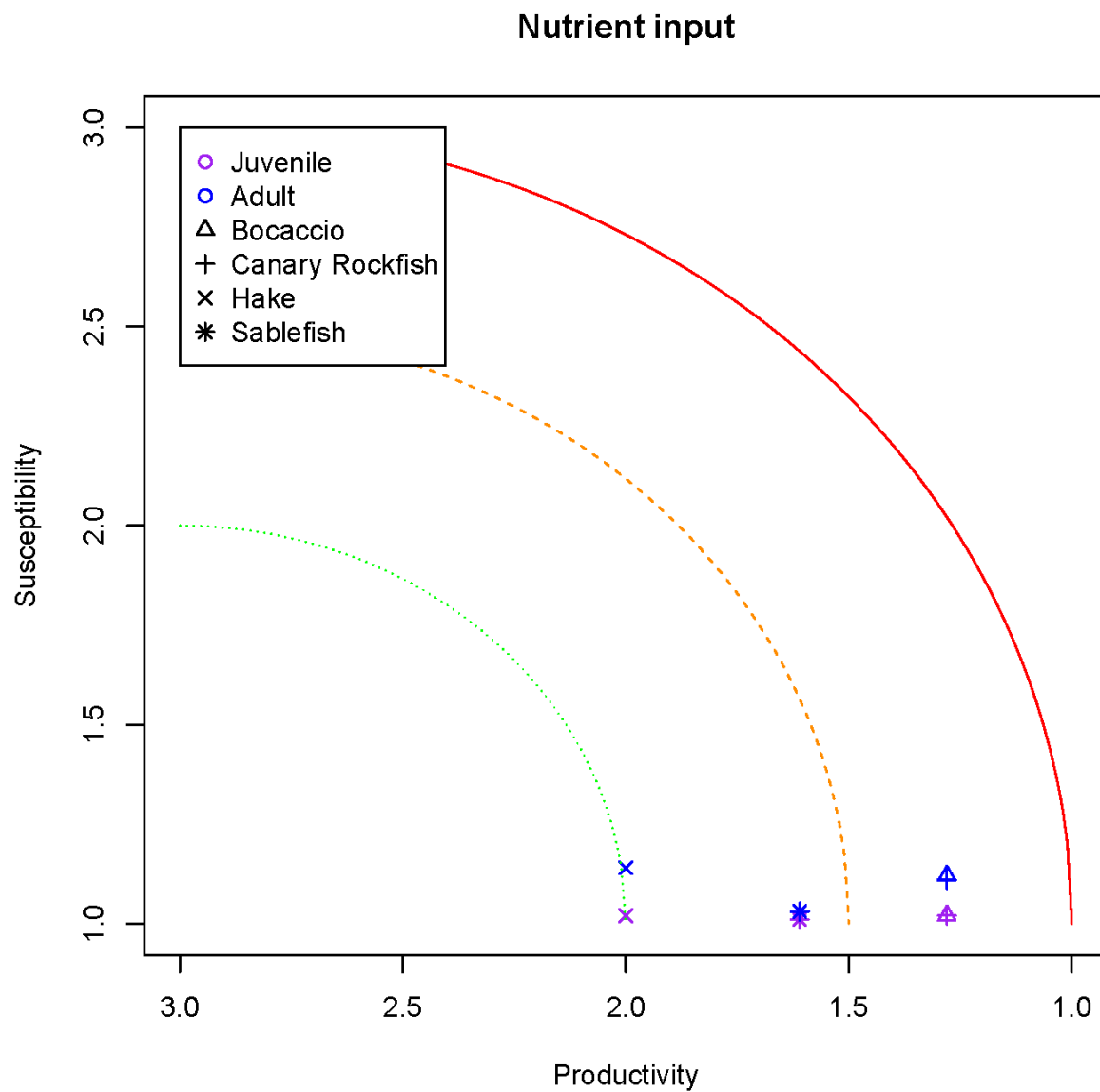


Figure GFR7. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to nutrient input as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

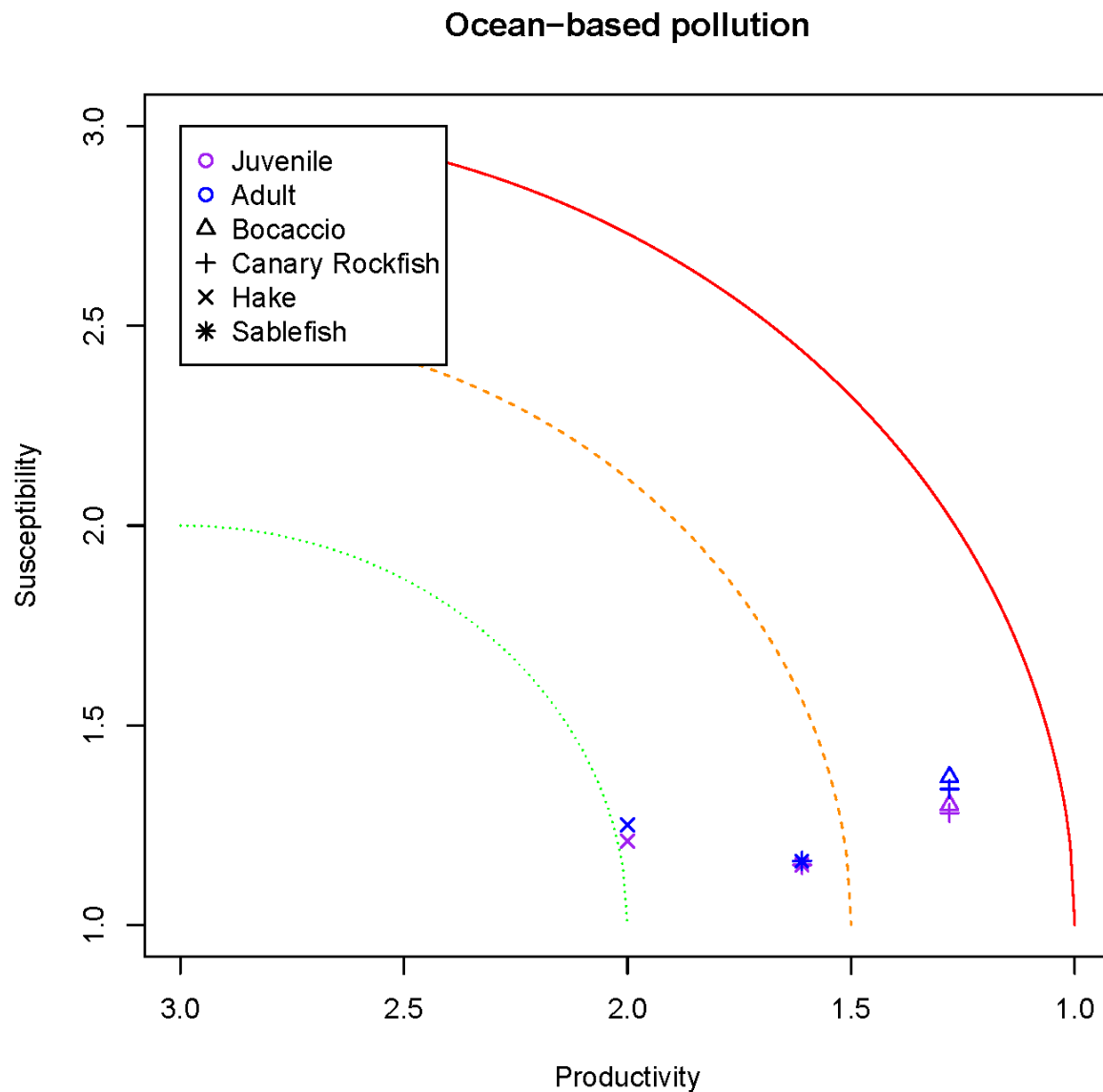


Figure GFR8. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to ocean based pollution as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

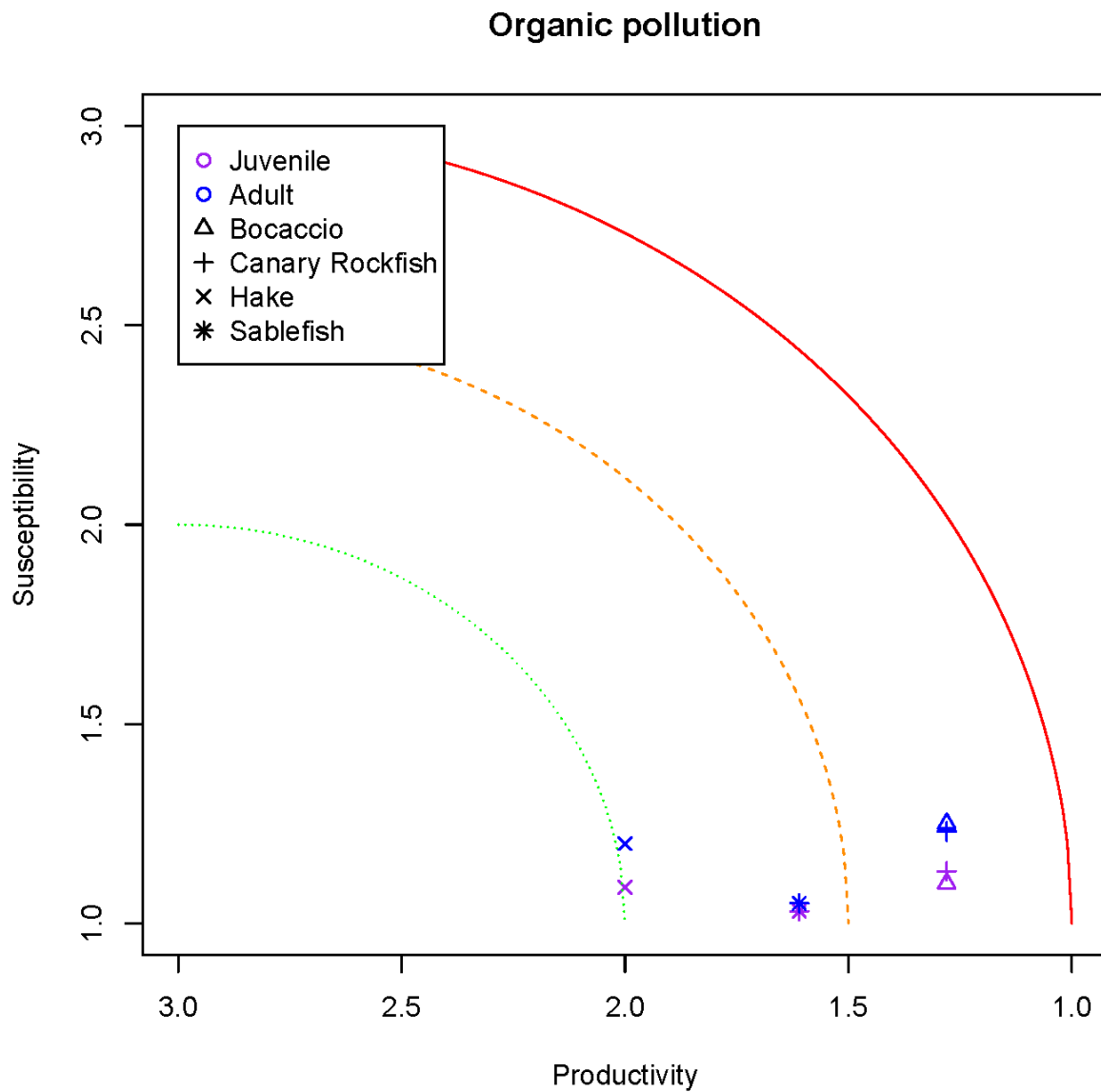


Figure GFR9. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to organic pollution as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

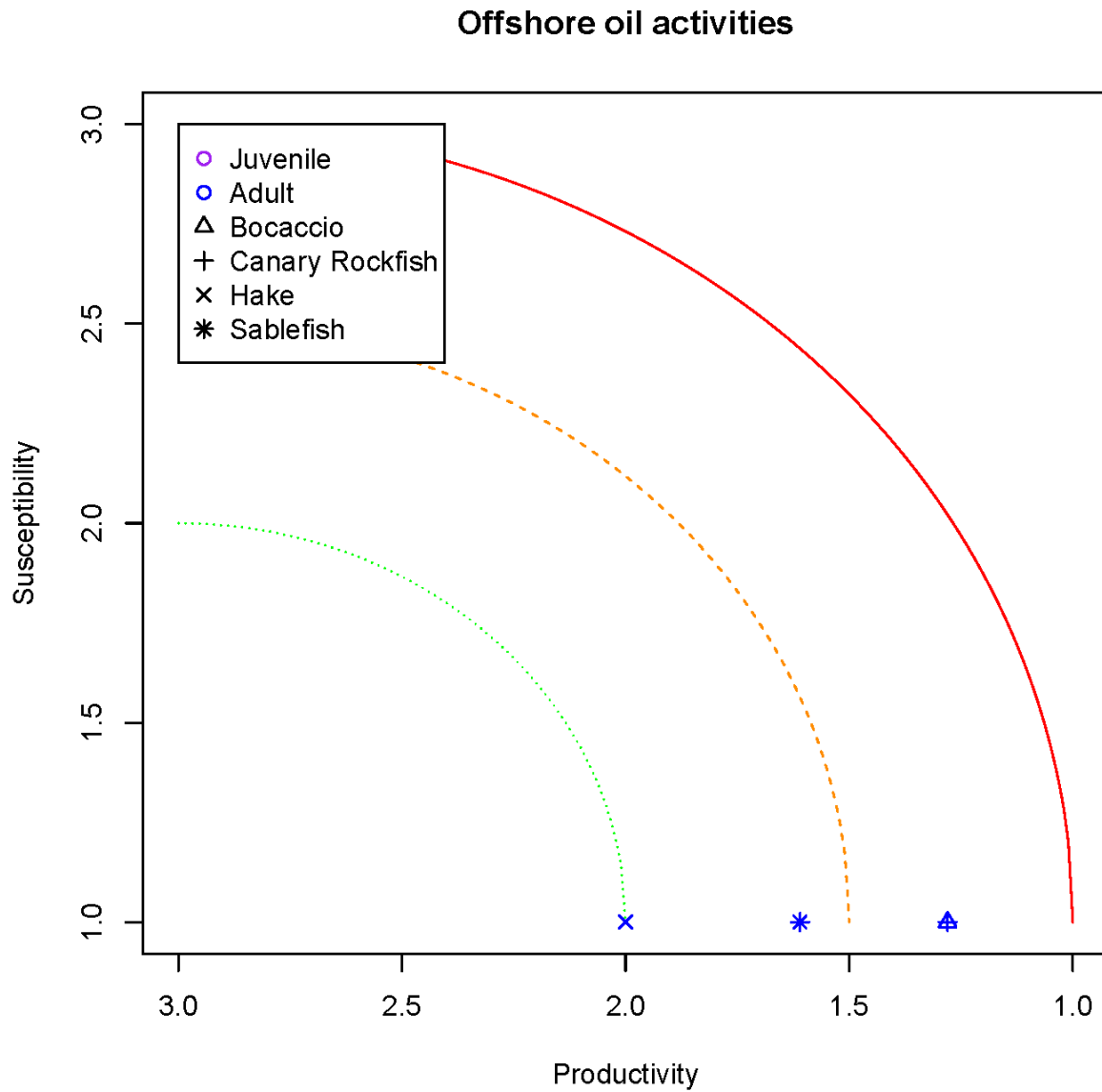


Figure GFR10. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to offshore oil activities as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

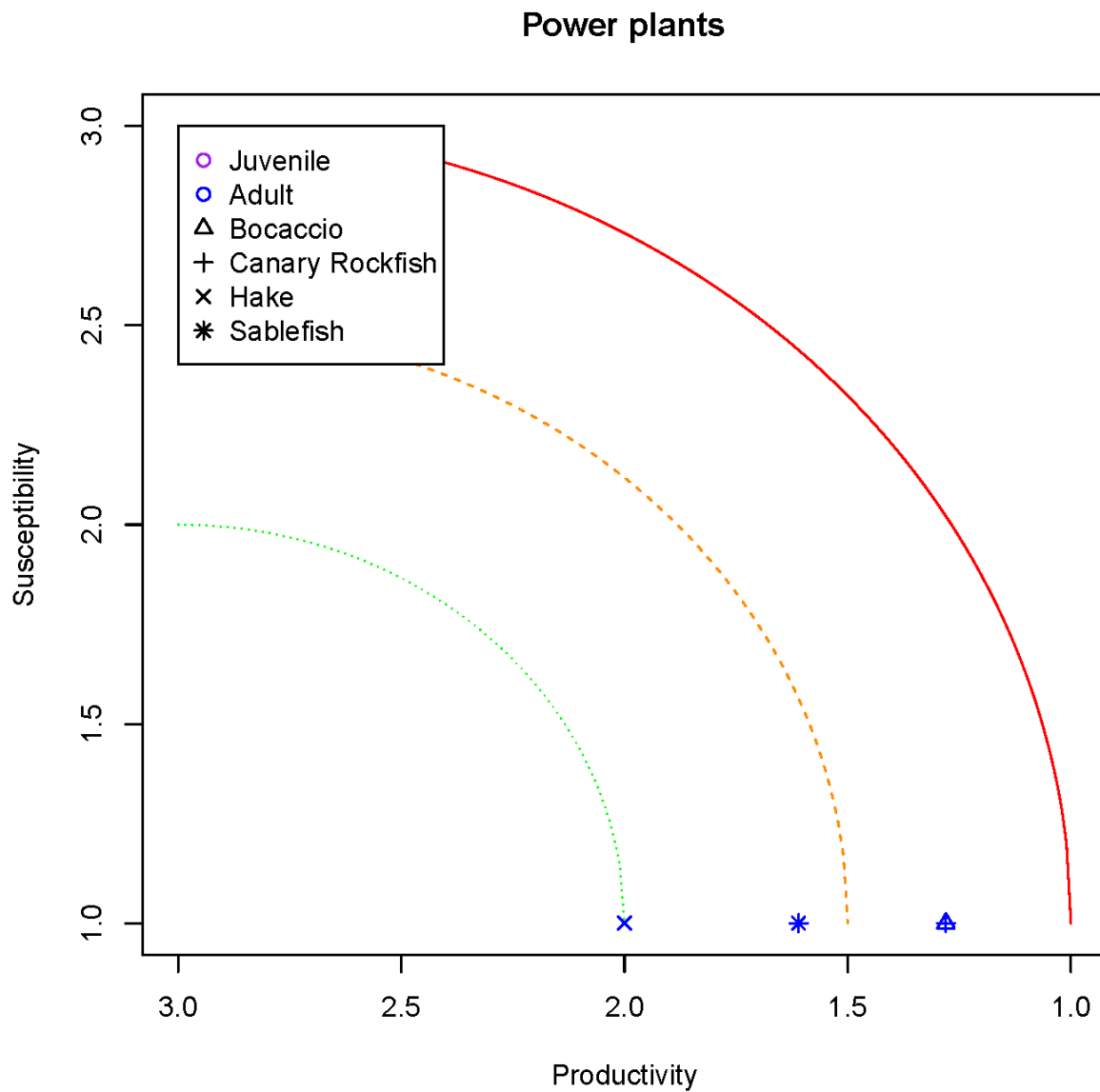


Figure GFR11. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to coastal seawater exchange (including power plants) as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

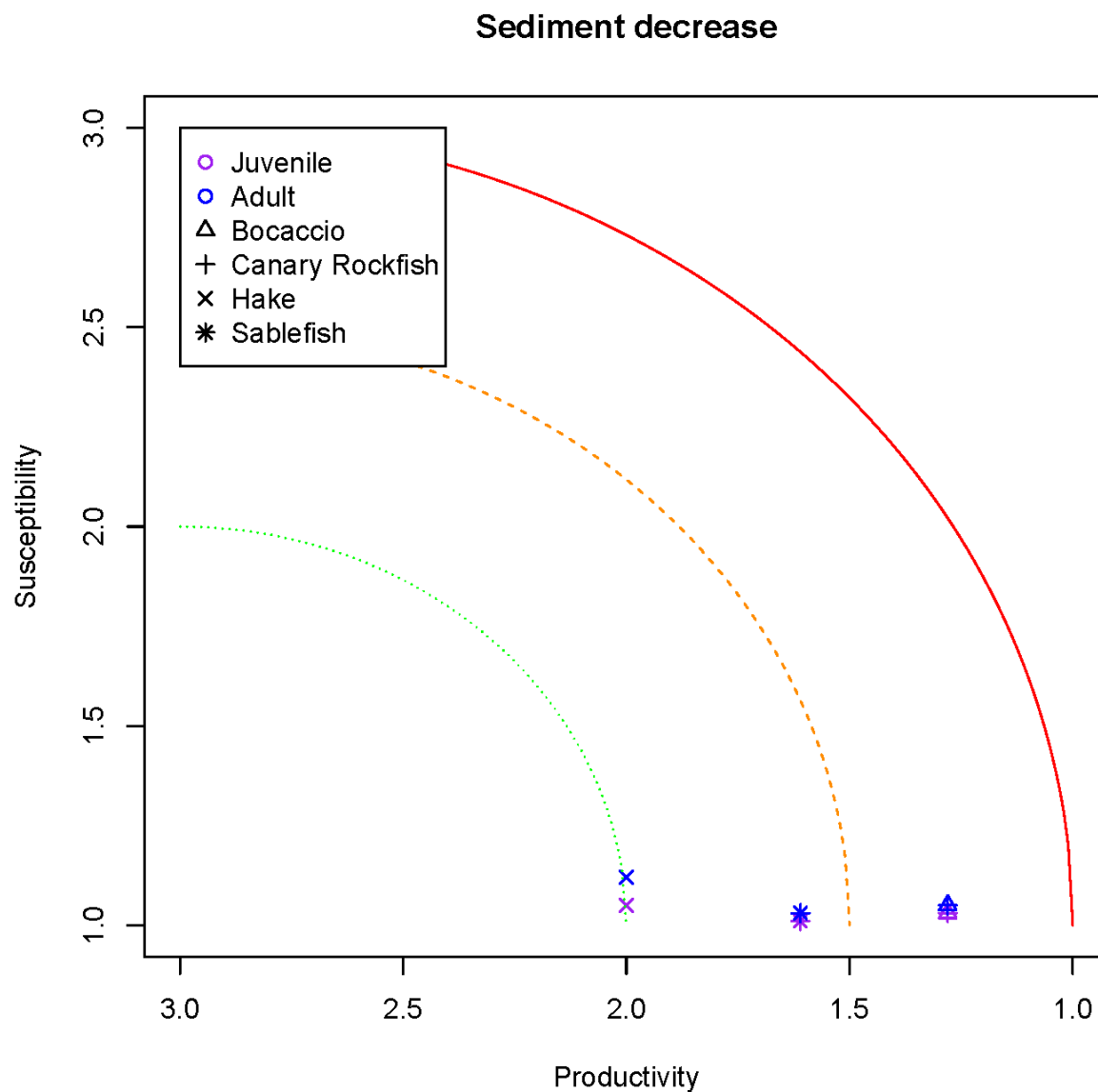


Figure GFR12. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to sediment decrease as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

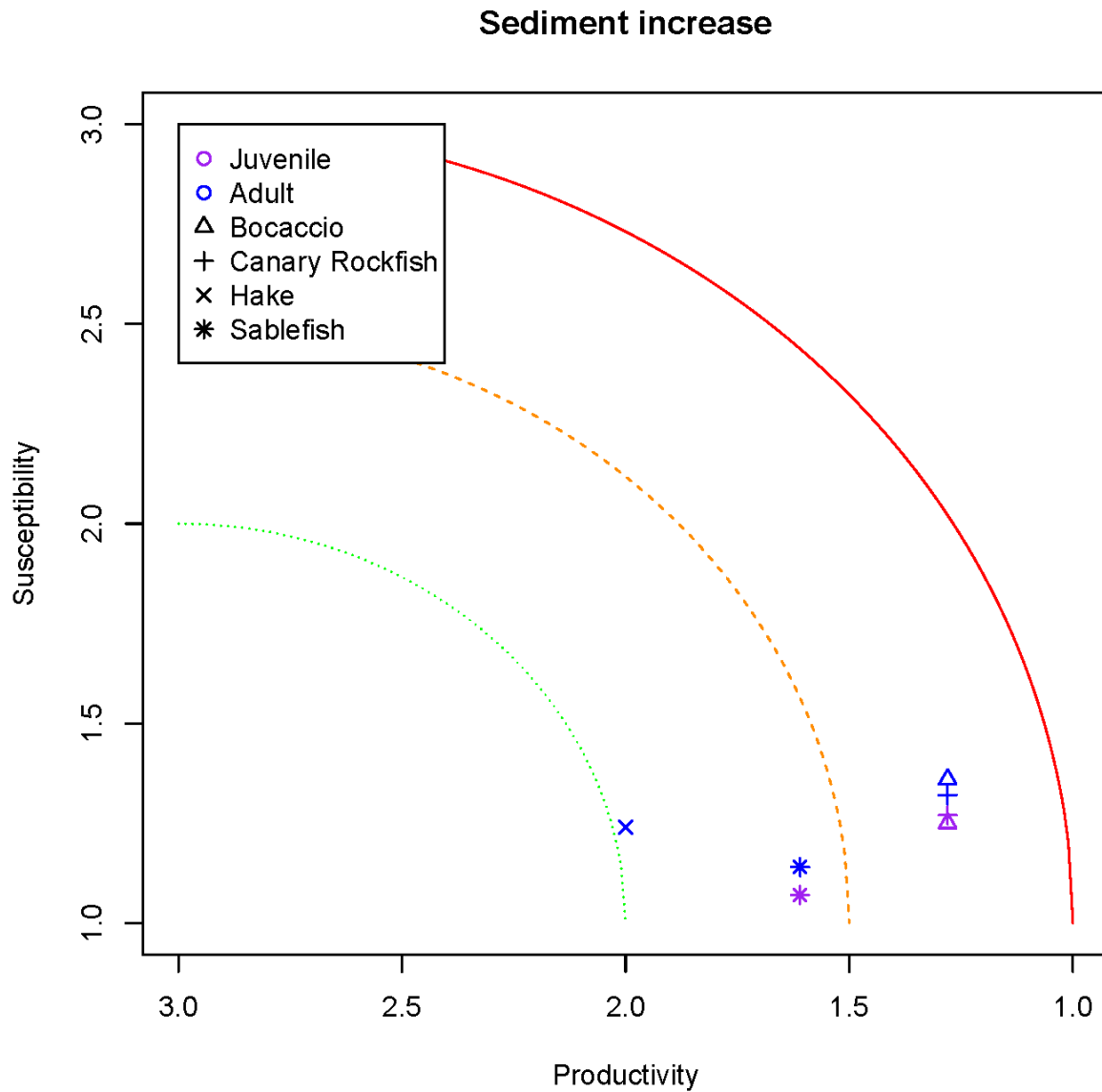


Figure GFR13. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to sediment increase as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

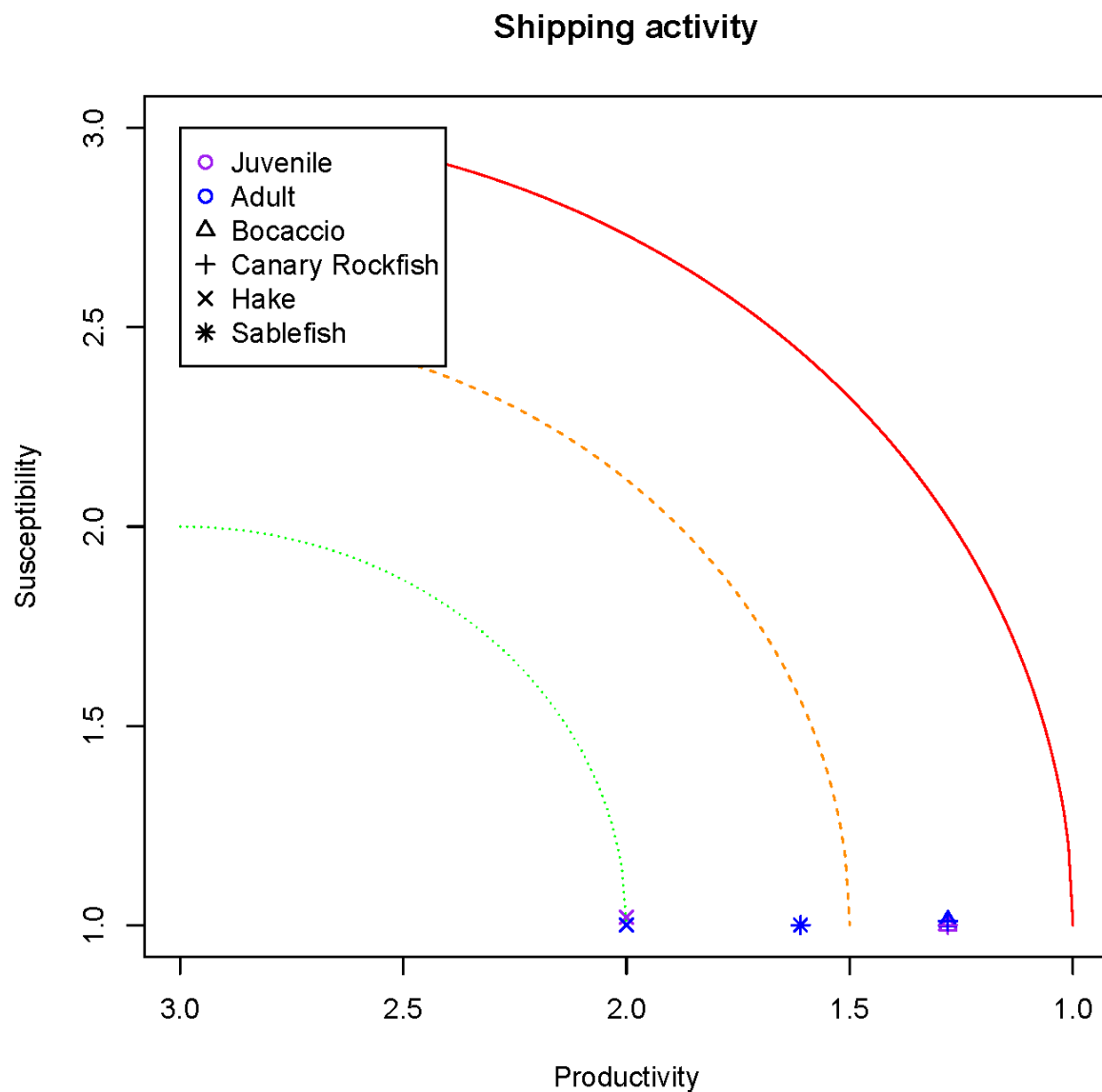


Figure GFR14. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to shipping activity as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

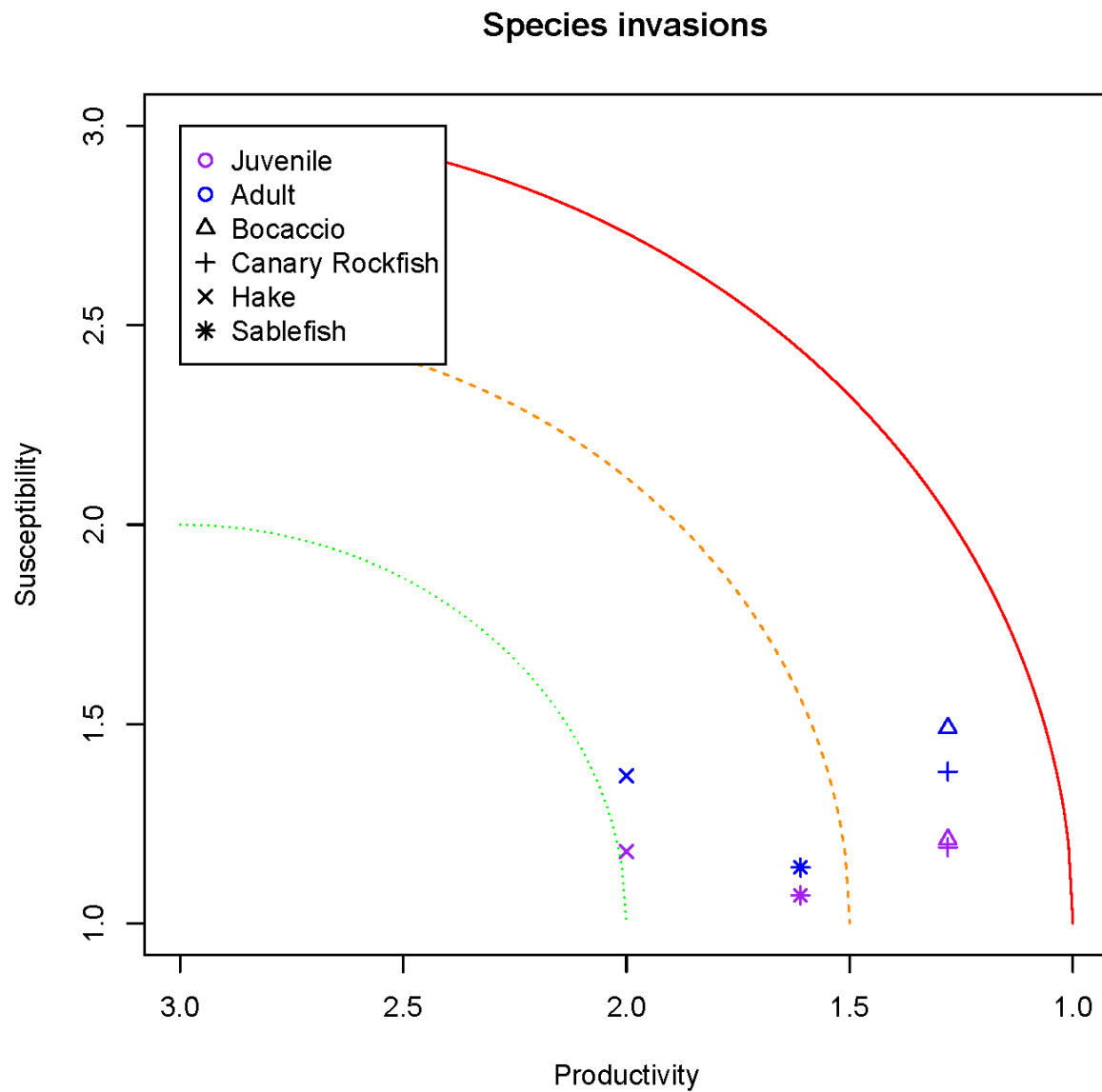


Figure GFR15. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to species invasions as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

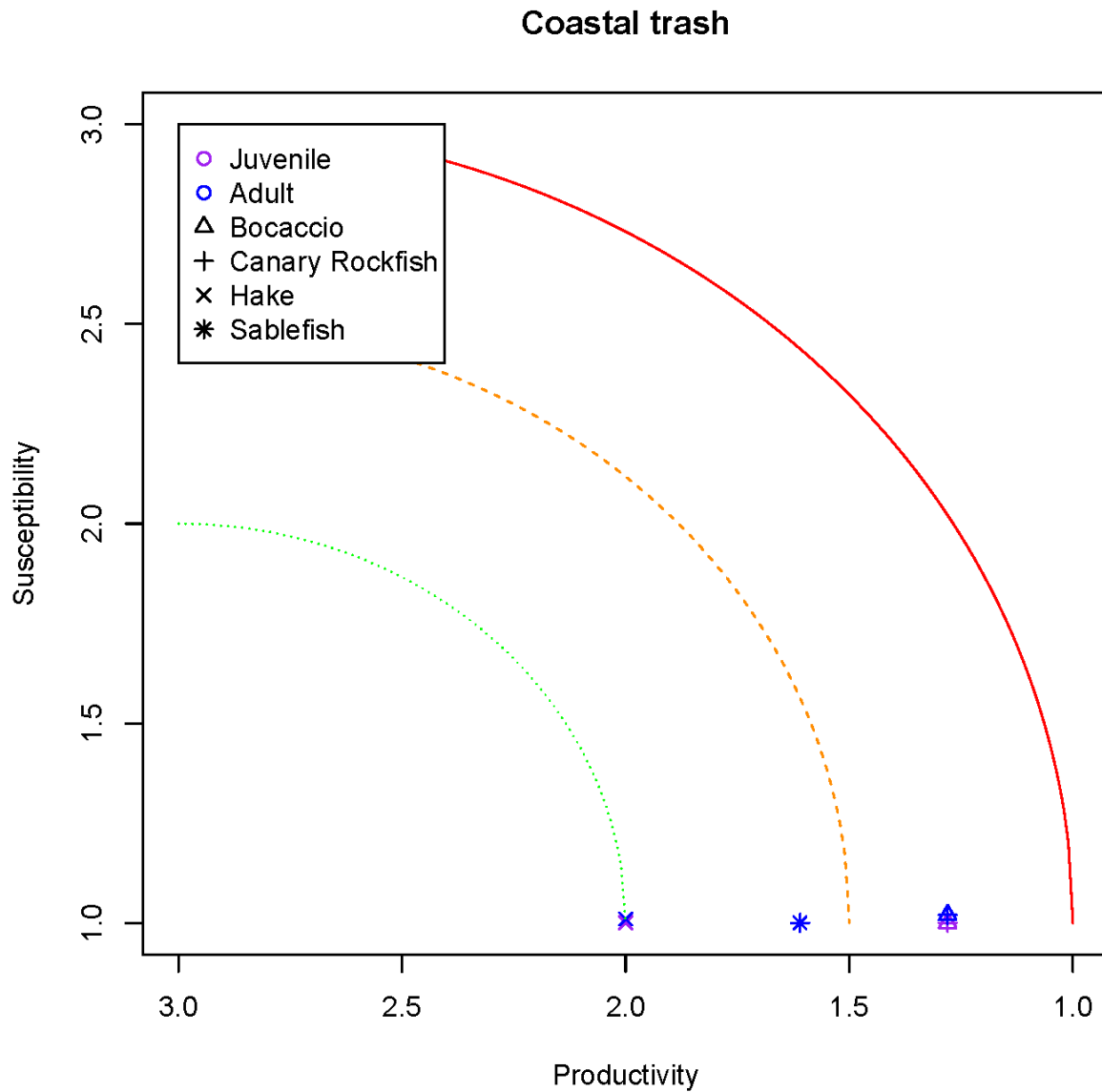


Figure GFR16. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to trash as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

Figure 1 is a graph showing the relationship between Productivity (X-axis, 1.0 to 3.0) and Susceptibility (Y-axis, 1.0 to 3.0). The graph displays three curves: a solid red curve (top), a dashed orange curve (middle), and a dotted green curve (bottom). Data points for various fish species are plotted: Juvenile (purple circle), Adult (blue circle), Bocaccio (purple triangle), Canary Rockfish (blue plus), Hake (blue cross), and Sablefish (purple asterisk). The curves represent different levels of susceptibility, with the red curve being the highest and the green curve being the lowest.

GFR - 670

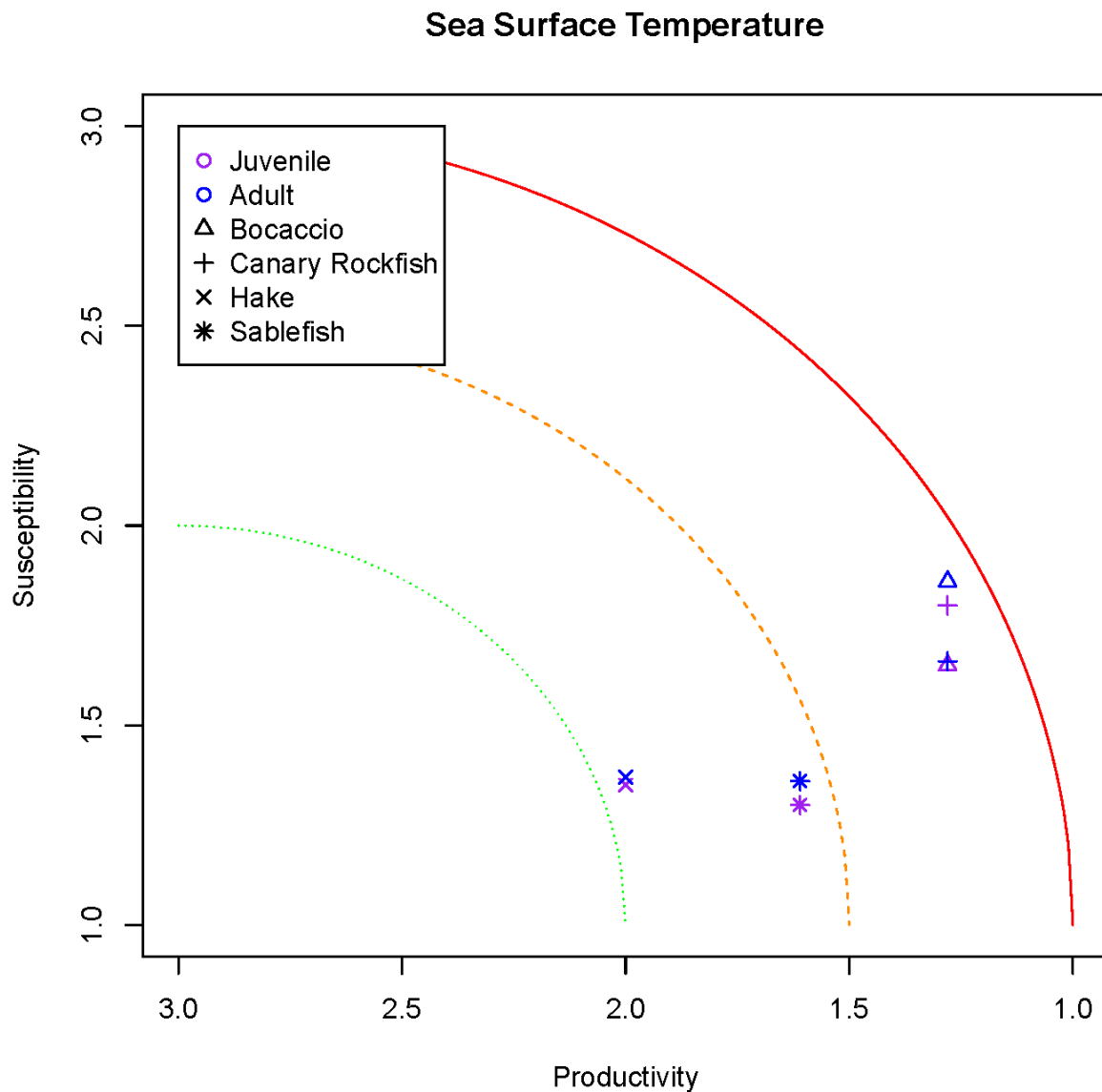


Figure GFR18. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to sea surface temperature as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

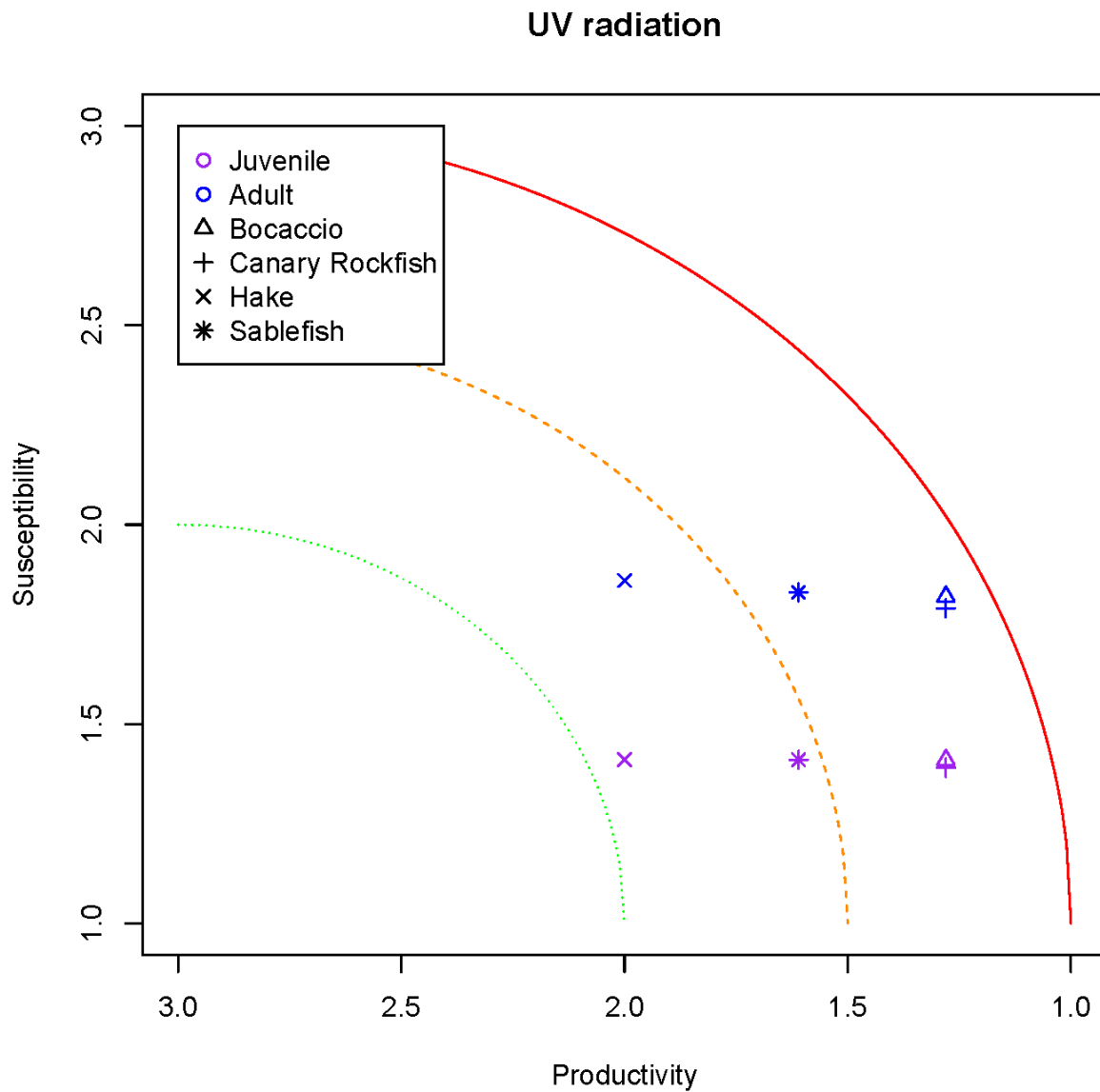


Figure GFR19. Productivity-susceptibility analysis (PSA) plot for the 8 species/stages relative to ultraviolet radiation as a threat. The susceptibility axis represents a relative score among species and stages but not among threats, though values near one indicate little to no impact in all cases. Where the adult and juvenile Susceptibility scores are identical, the symbols are on top of each other and only the adult values are visible.

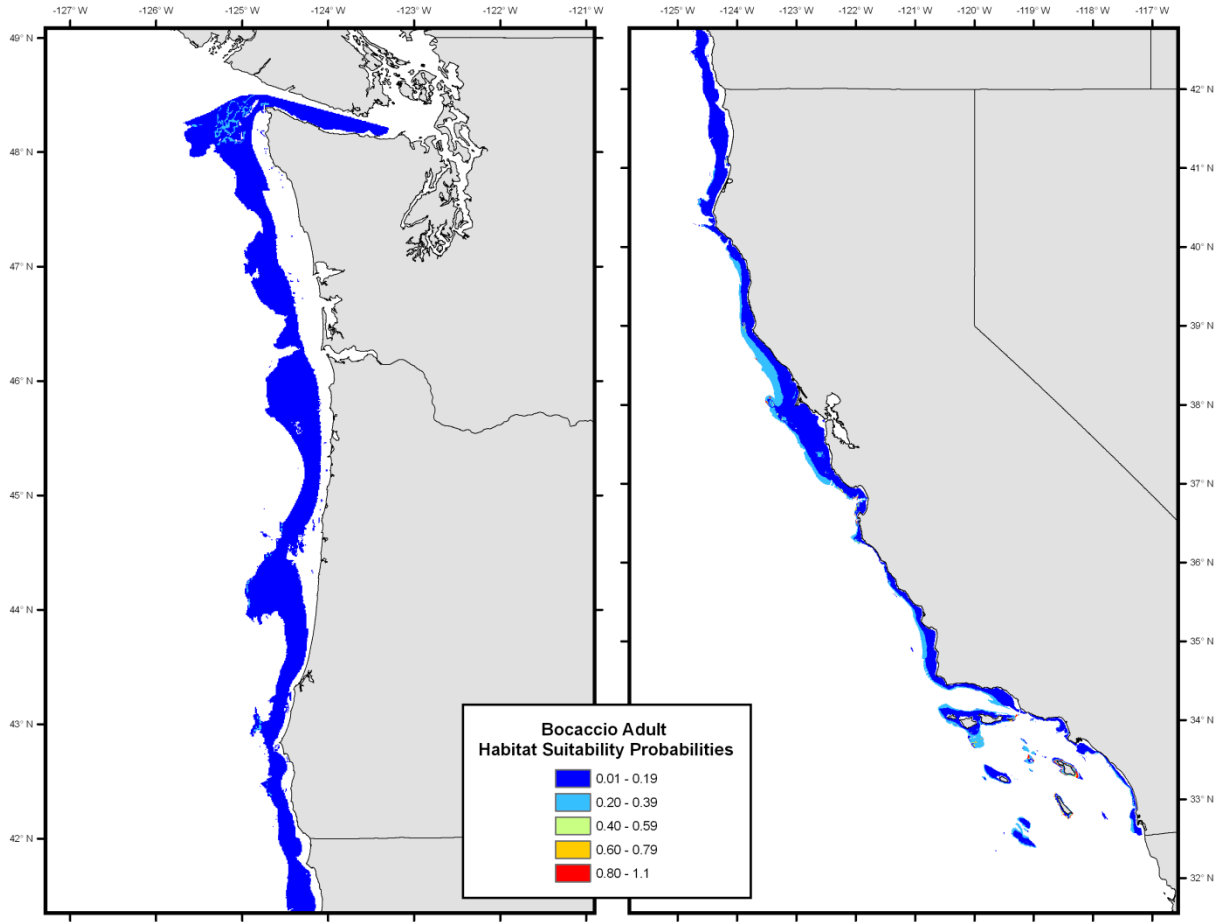


Figure GFR20. Habitat Suitability Probabilities for bocaccio *Sebastes paucispinis* adult. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement.

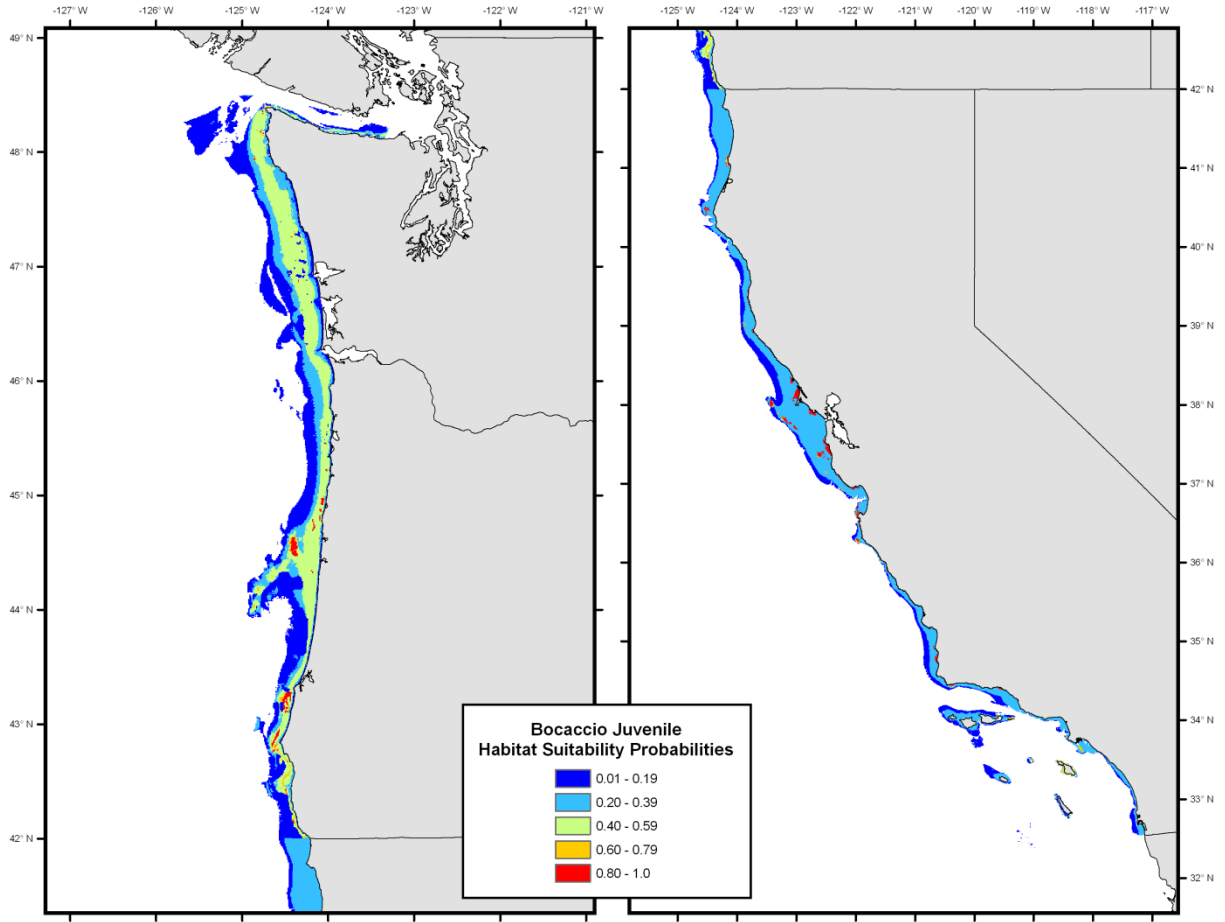


Figure GFR21. Habitat Suitability Probabilities for bocaccio *Sebastes paucispinis* juvenile. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement.

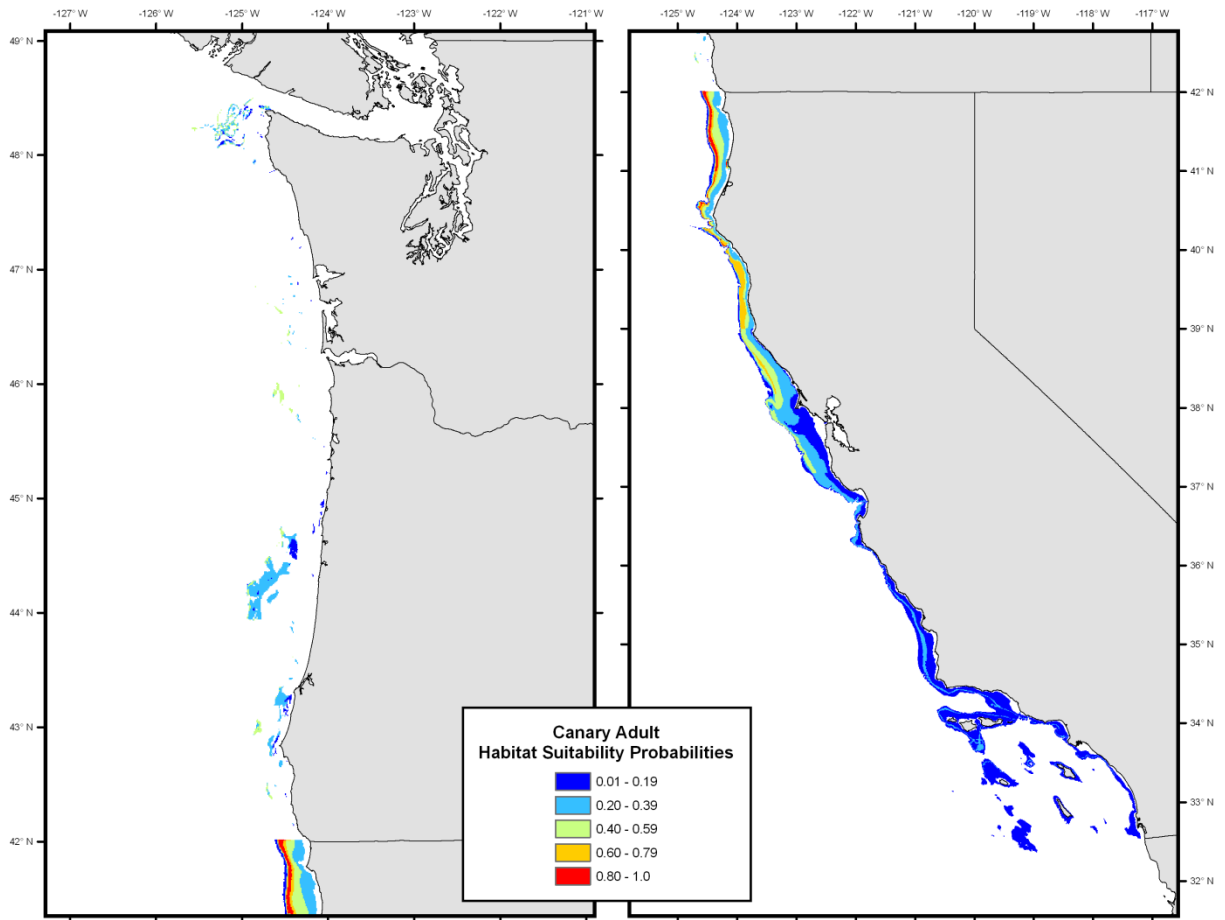


Figure GFR22. Habitat Suitability Probabilities for canary *Sebastes pinniger* adult. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement.

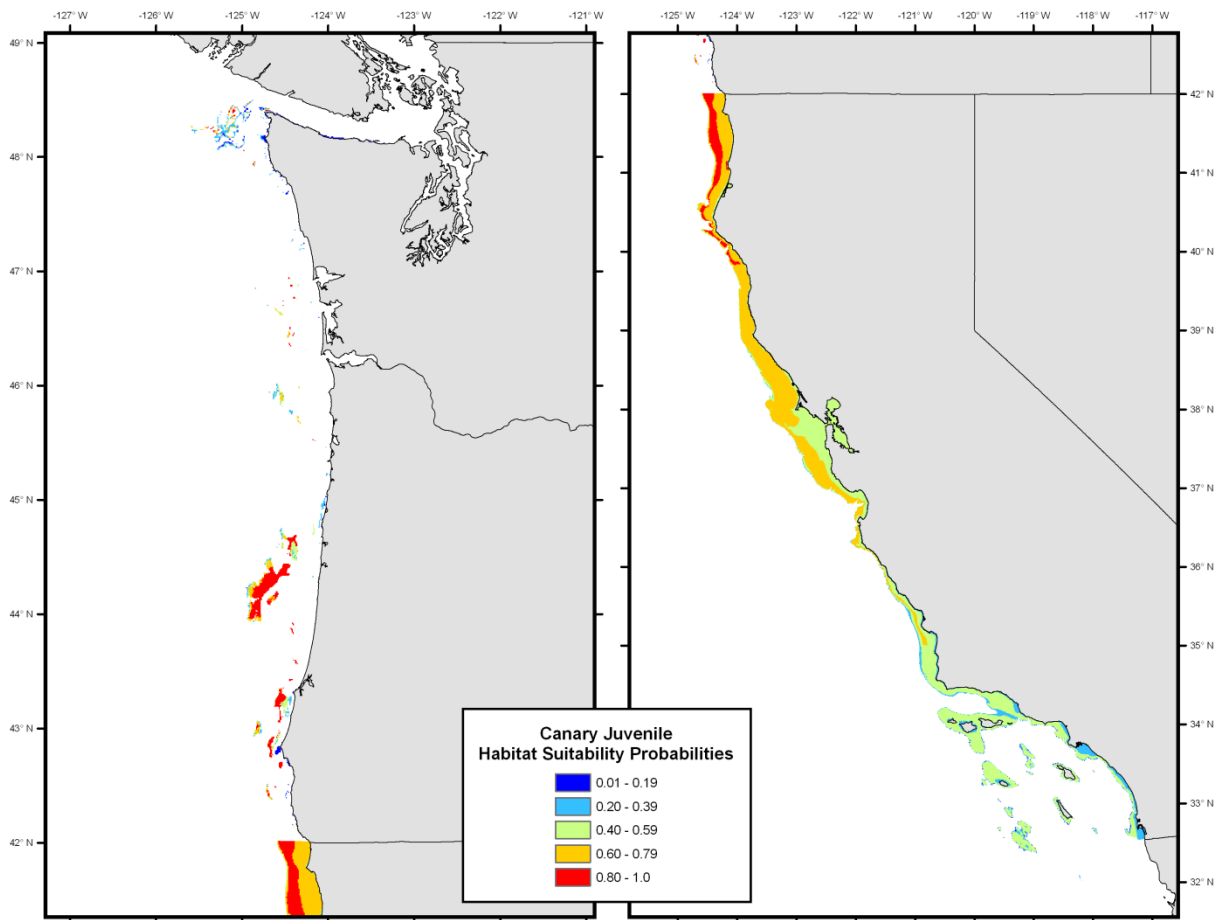


Figure GFR23. Habitat Suitability Probabilities for canary *Sebastes pinniger* juvenile. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement.

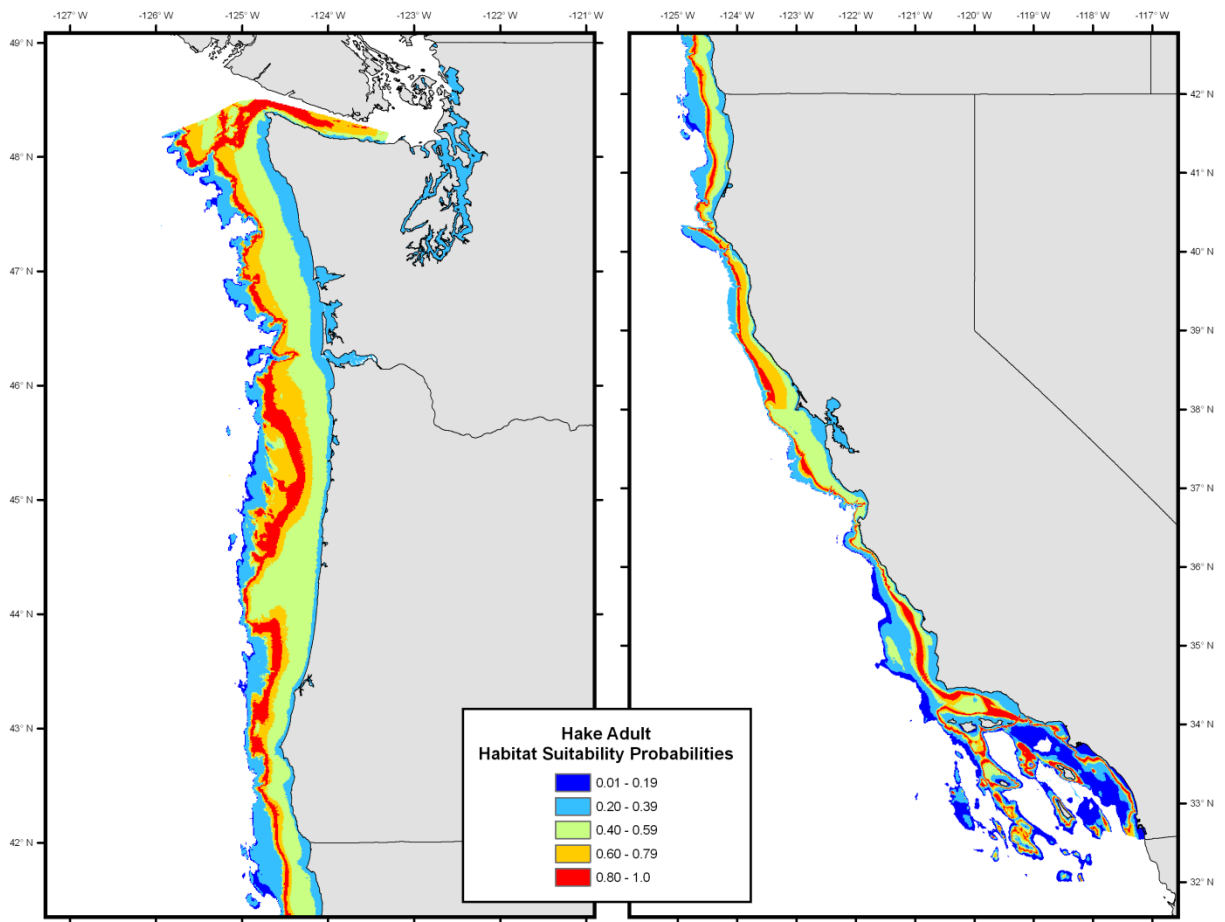


Figure GFR24. Habitat Suitability Probabilities for Pacific hake *Merluccius productus* adult. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement.

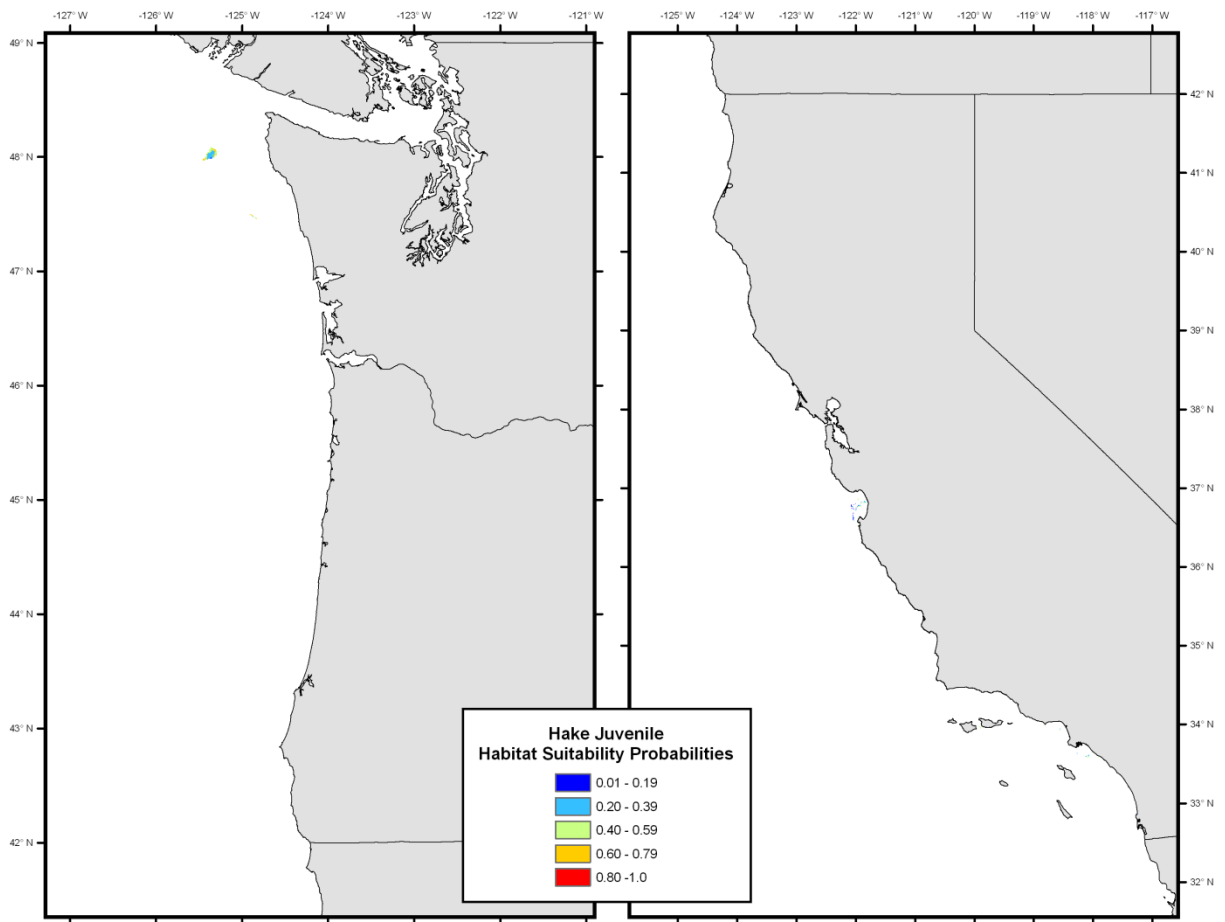


Figure GFR25. Habitat Suitability Probabilities for Pacific hake *Merluccius productus* juvenile. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement.

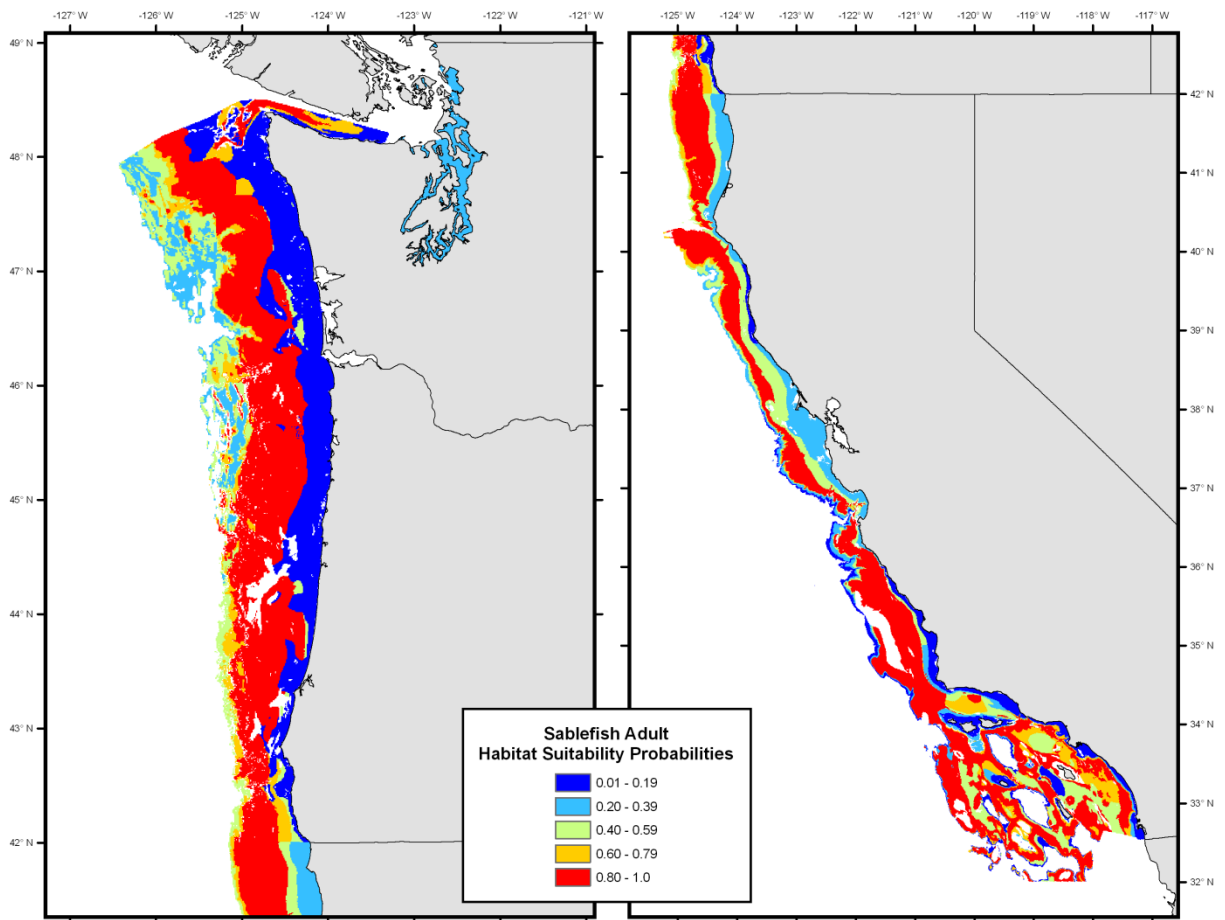


Figure GFR26. Habitat Suitability Probabilities for Sablefish *Anoplopoma fimbria* adult. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement.

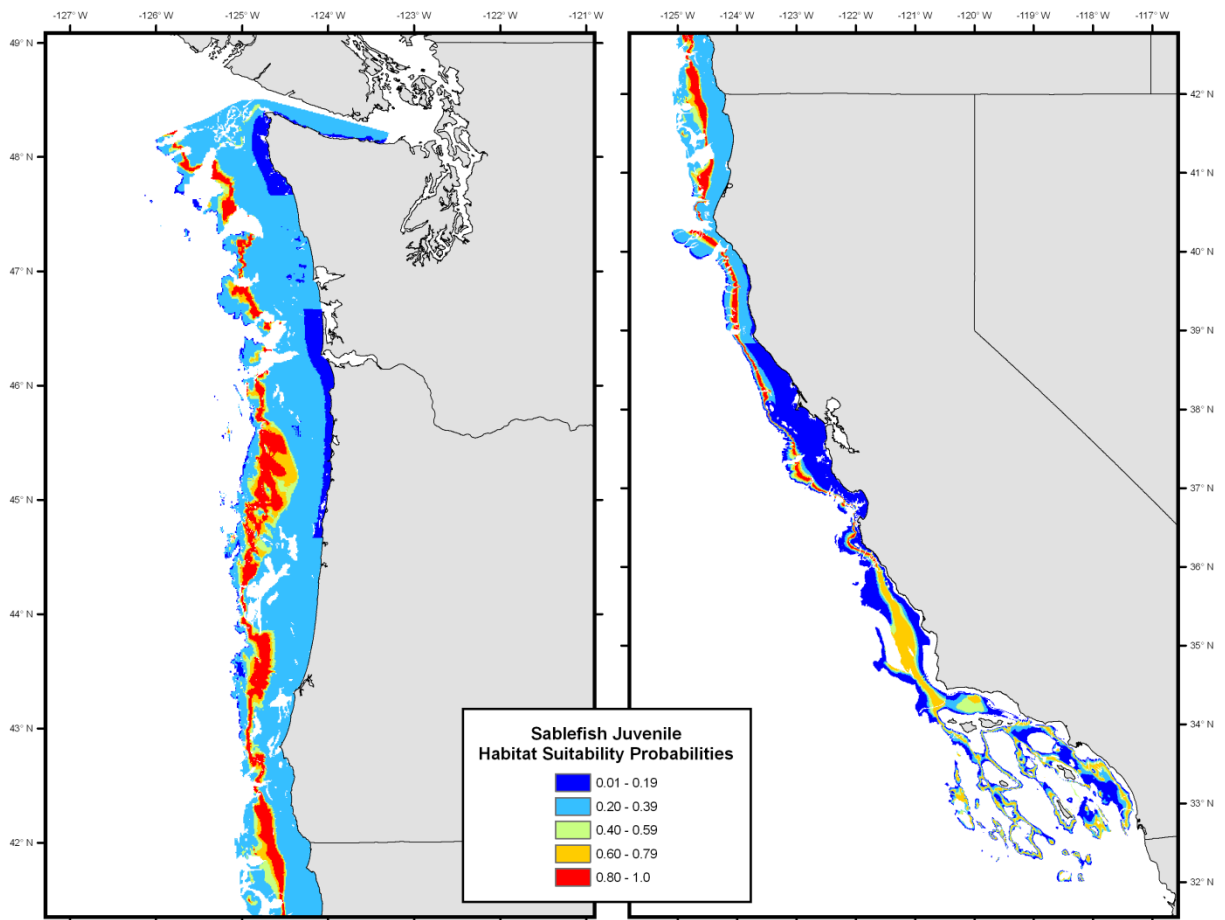


Figure GFR27. Habitat Suitability Probabilities for Sablefish *Anoplopoma fimbria* juvenile. Data from 2005 Essential Fish Habitat Final Environmental Impact Statement.

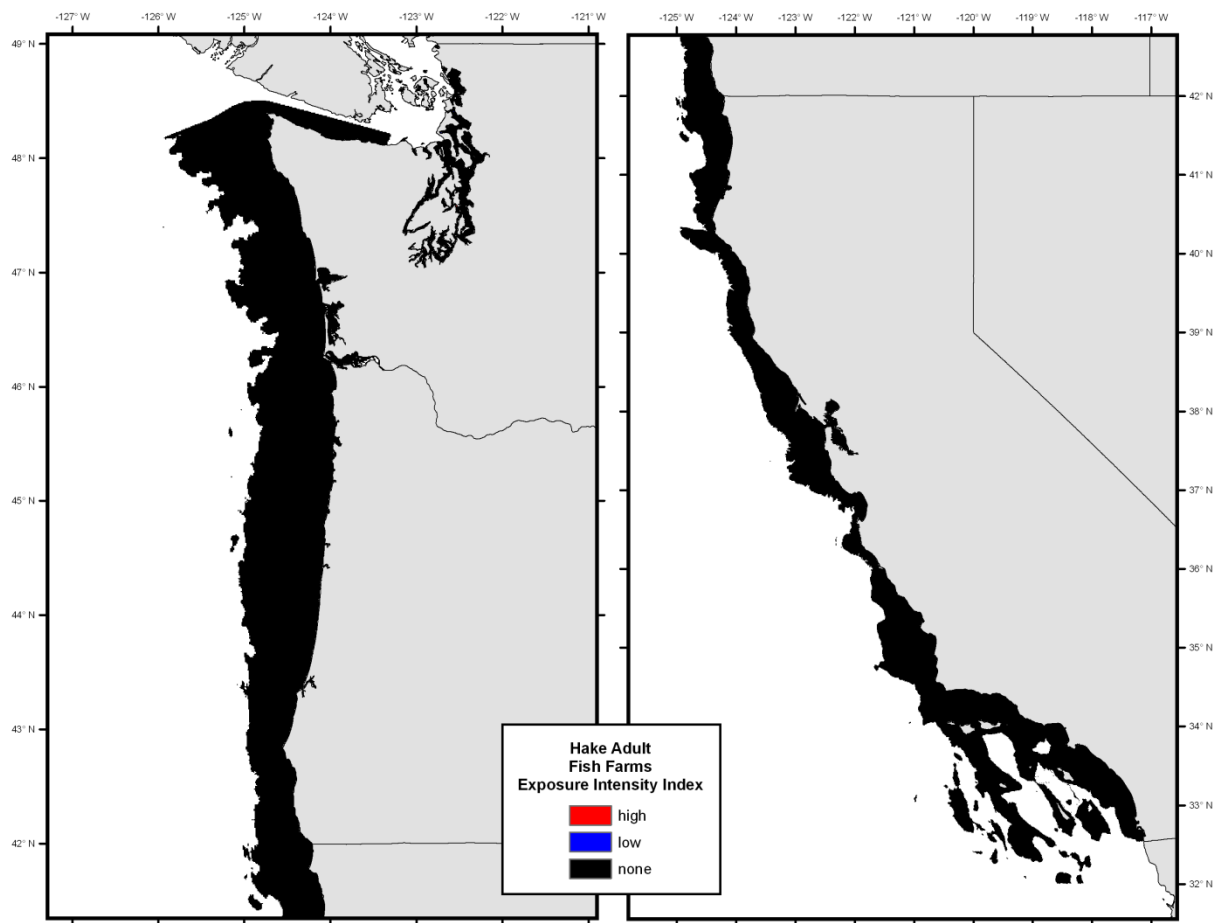


Figure GFR28. Exposure intensity index of aquaculture for Pacific hake *Merluccius productus* adult. High = upper bicile, and low = lower bicile.

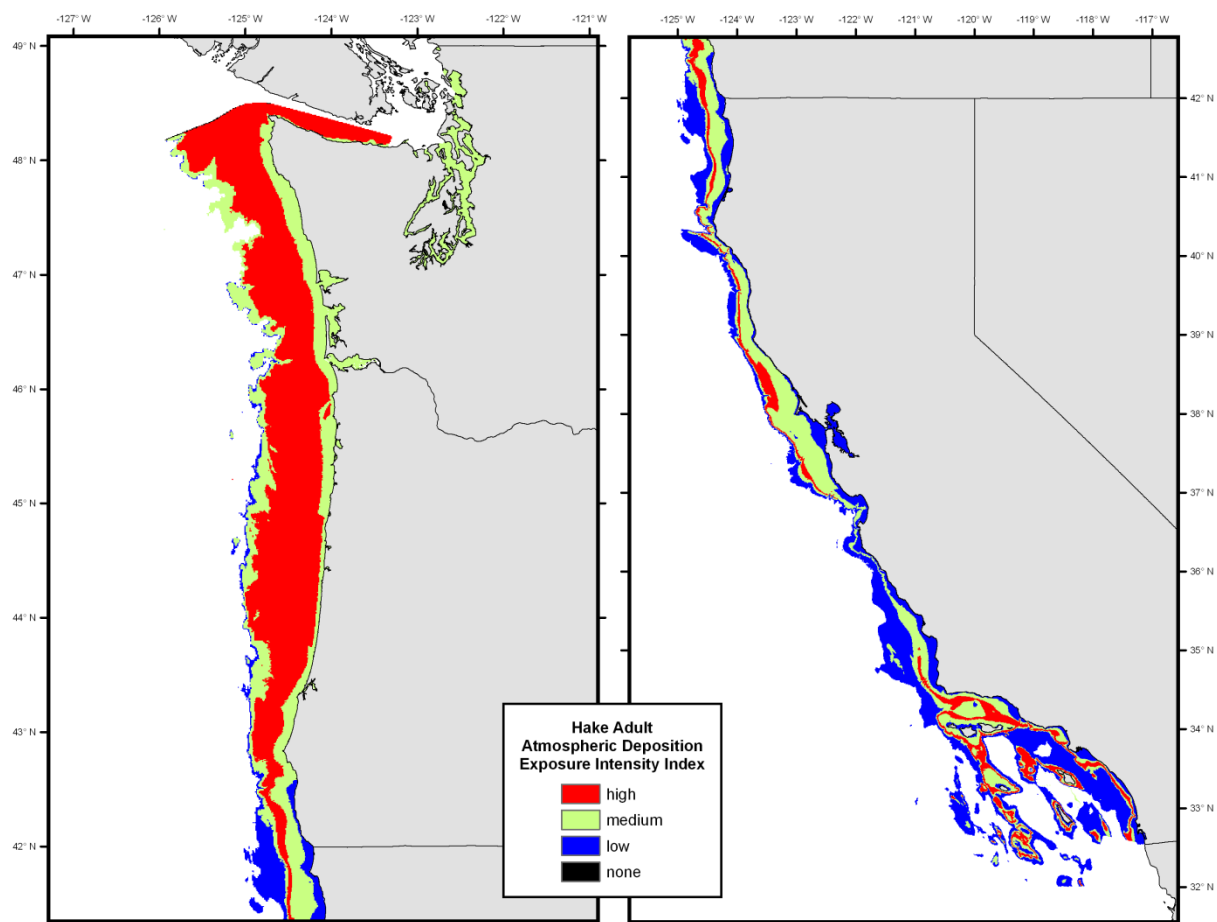


Figure GFR29. Exposure intensity index of atmospheric deposition of pollutants for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

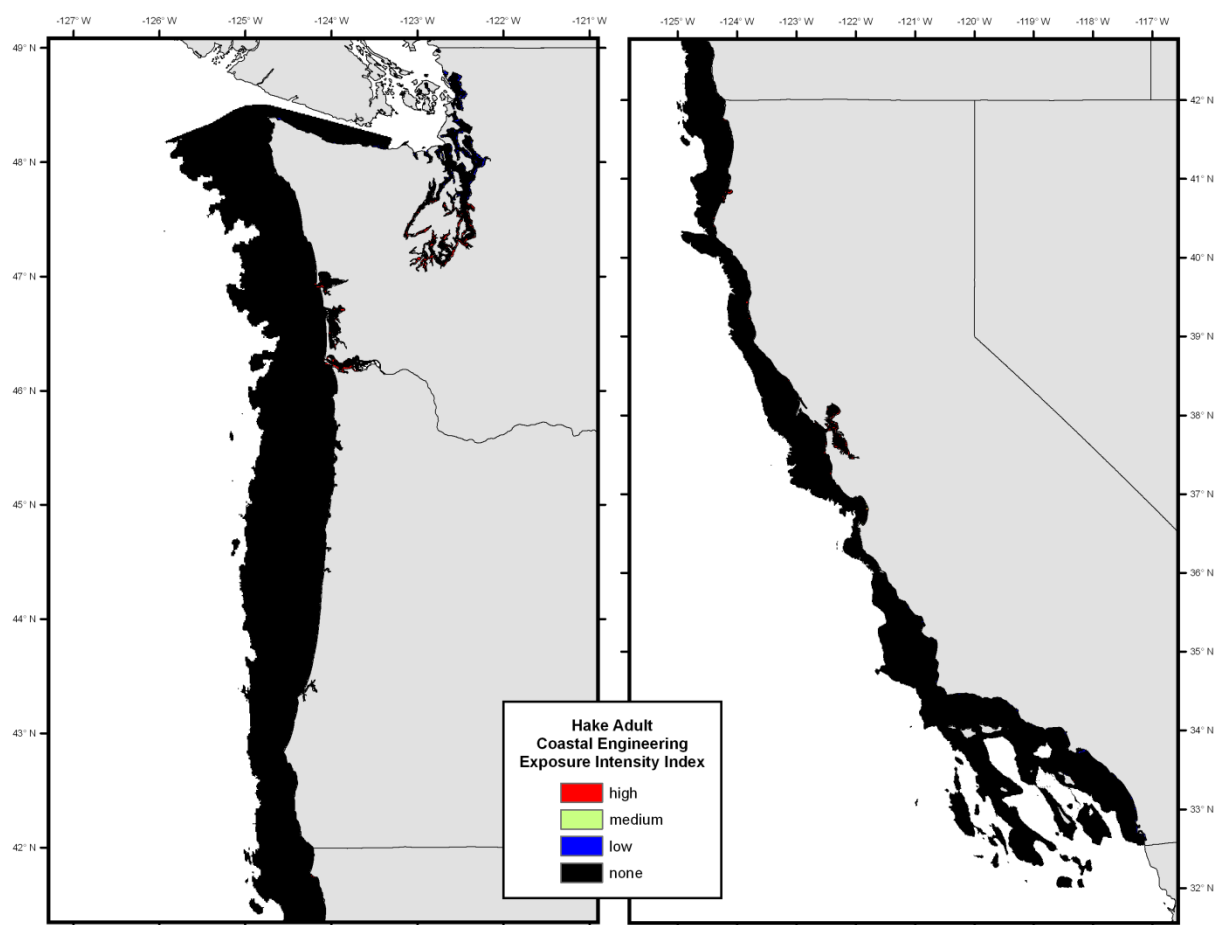


Figure GFR30. Exposure intensity index of coastal engineering for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

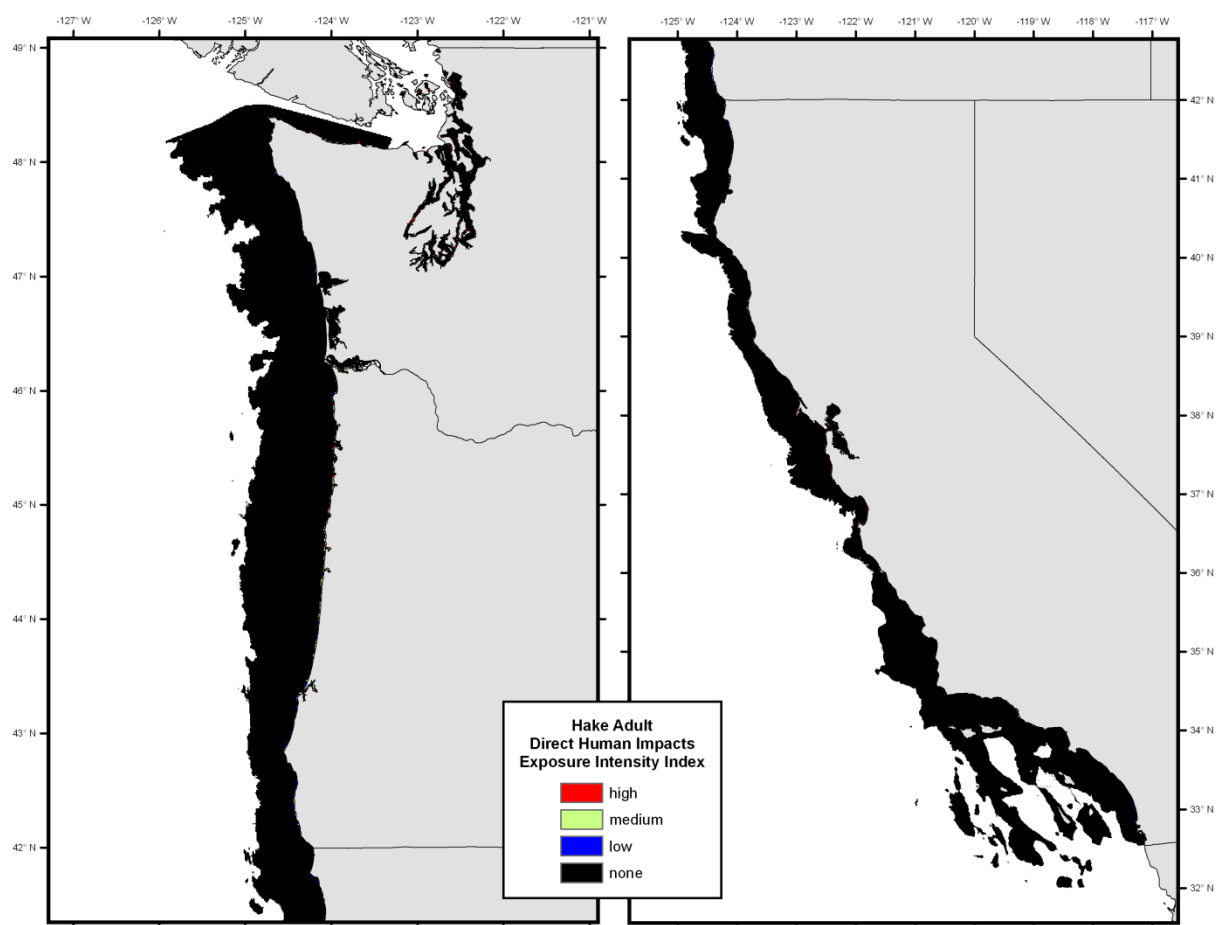


Figure GFR31. Exposure intensity index of direct human impacts (beach trampling) for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

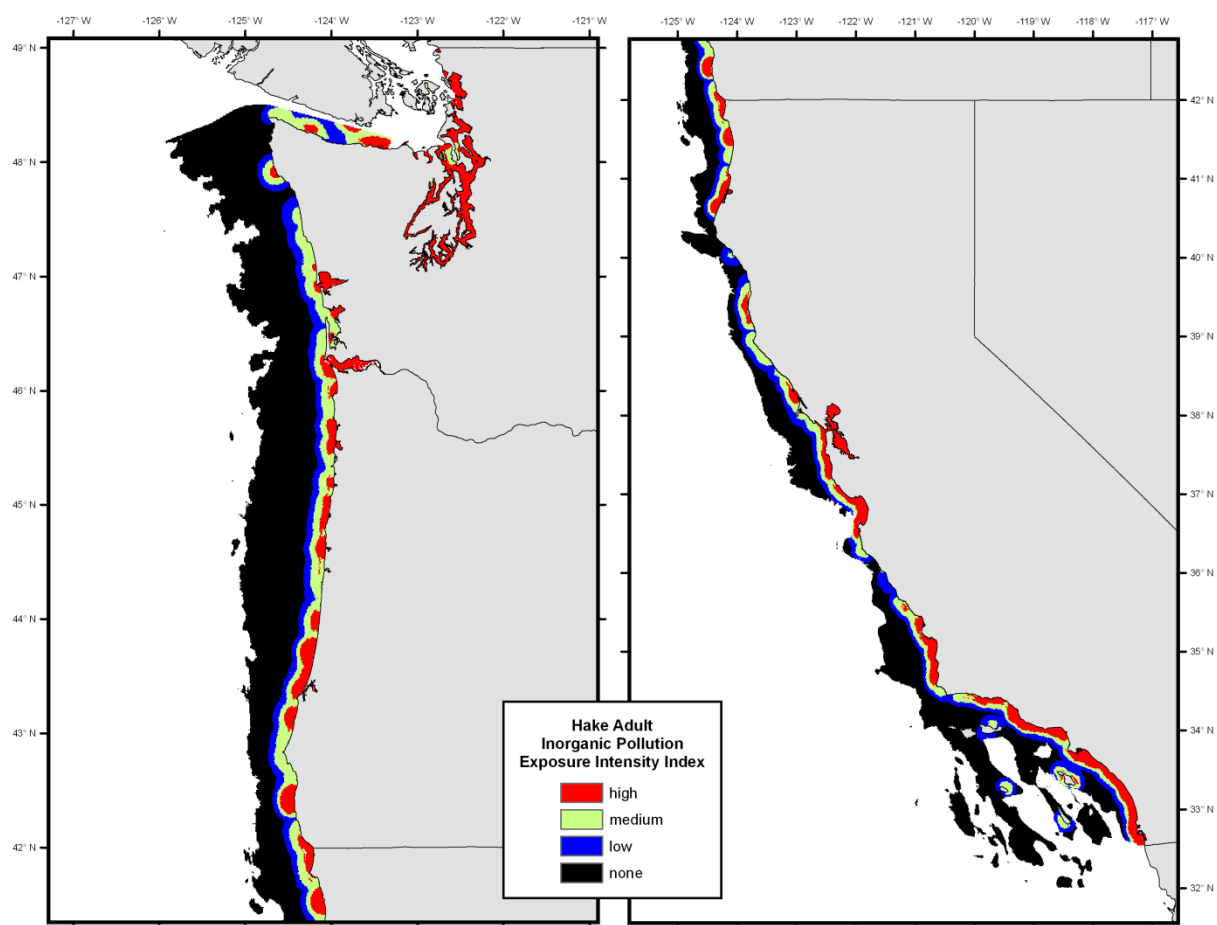


Figure GFR32. Exposure intensity index of inorganic pollution for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

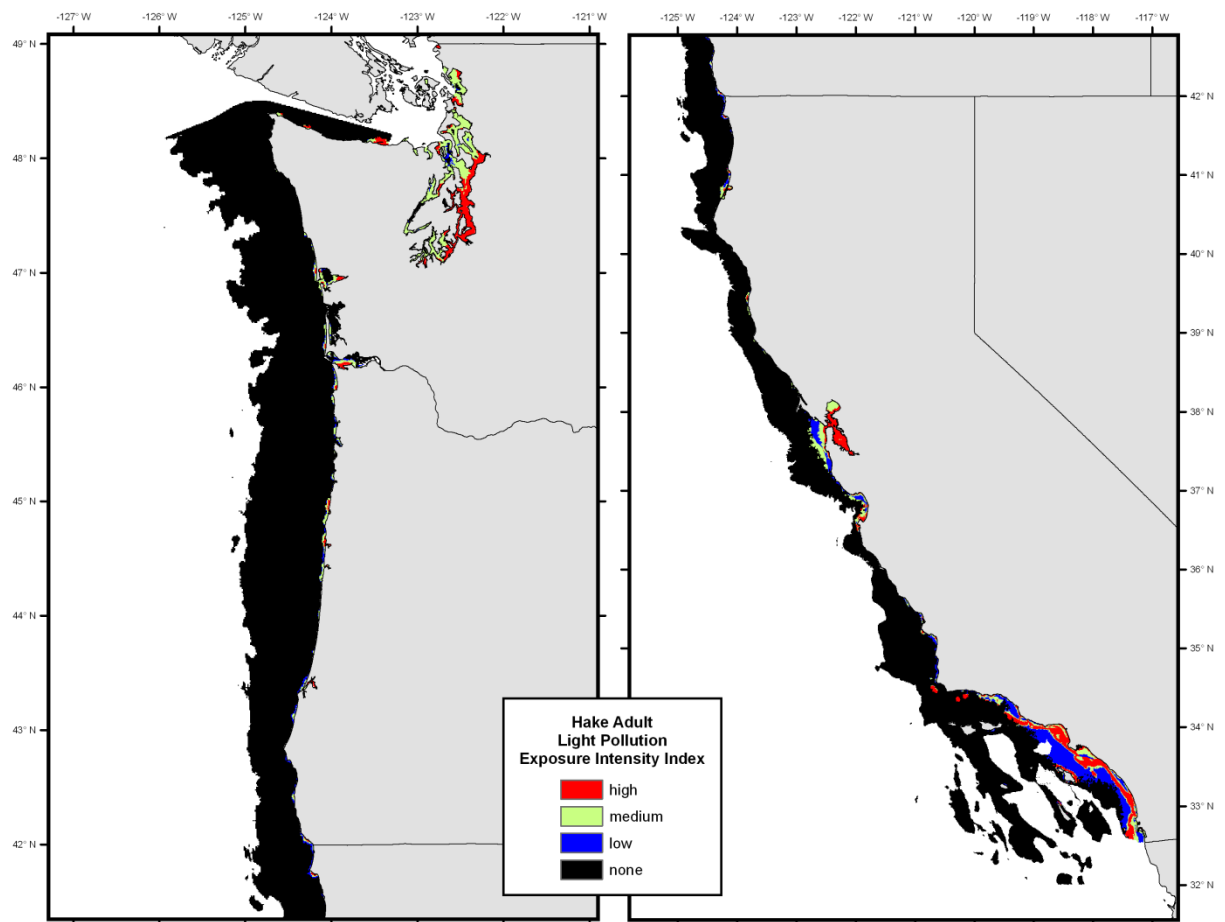


Figure GFR33. Exposure intensity index of light pollution for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

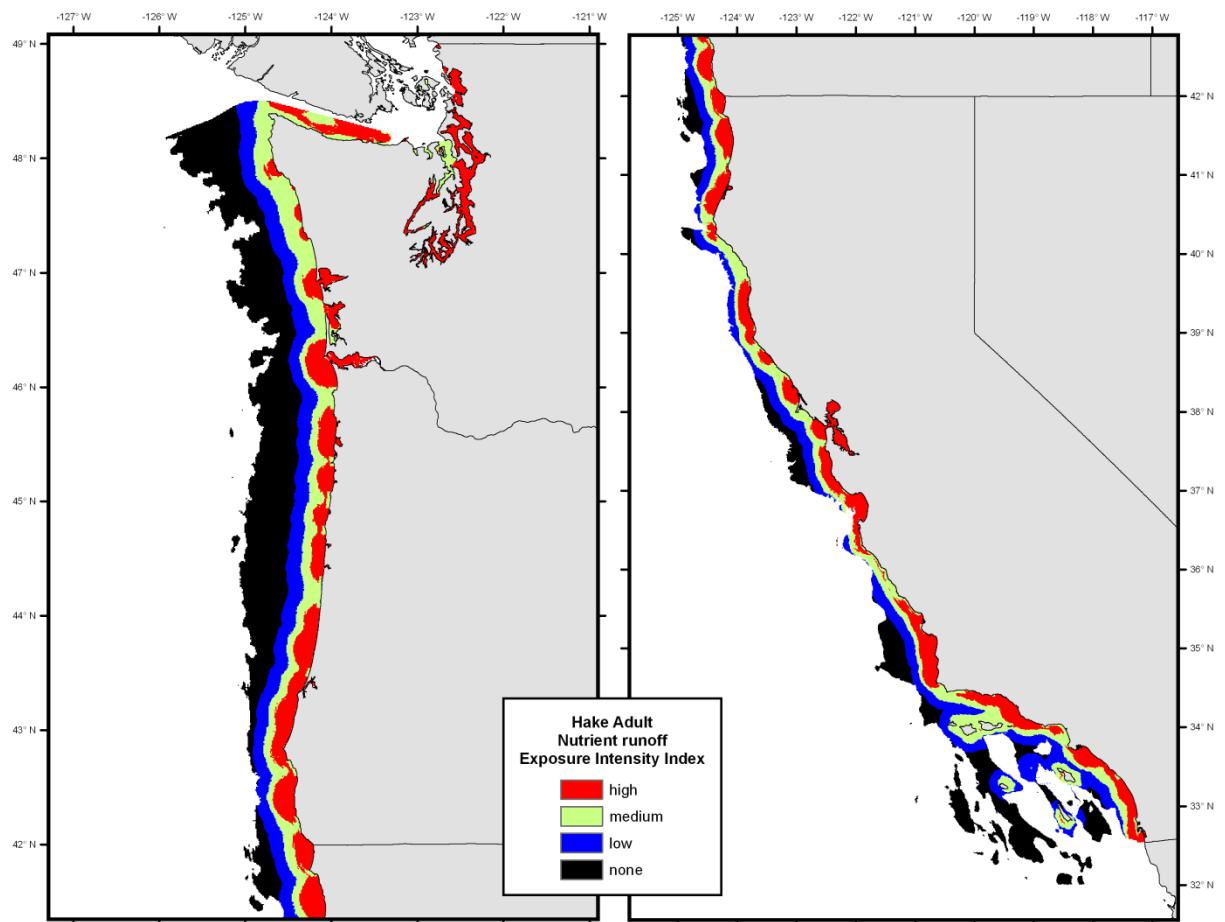


Figure GFR34. Exposure intensity index of nutrient runoff for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

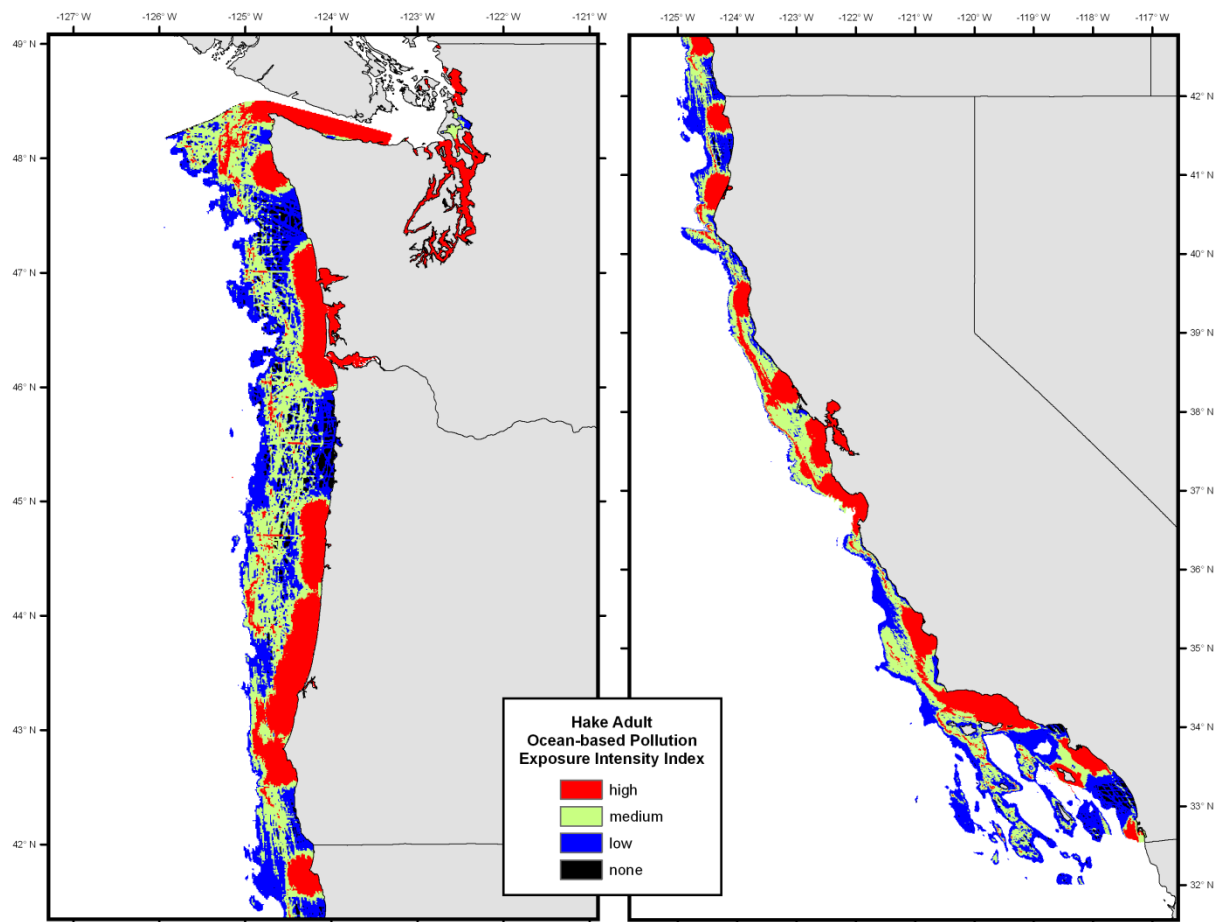


Figure GFR35. Exposure intensity index of ocean-based pollution for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

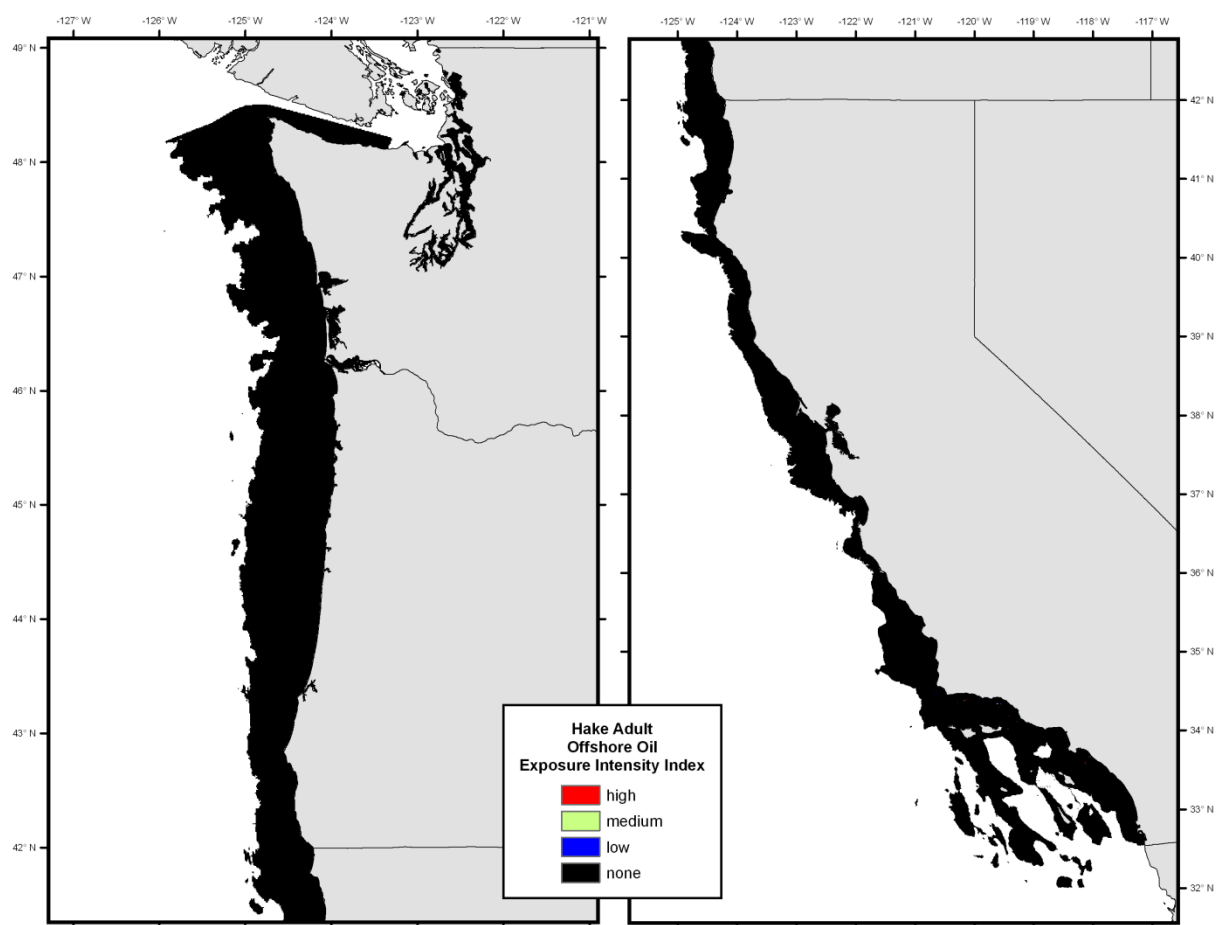


Figure GFR36. Exposure intensity index of offshore oil activities for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

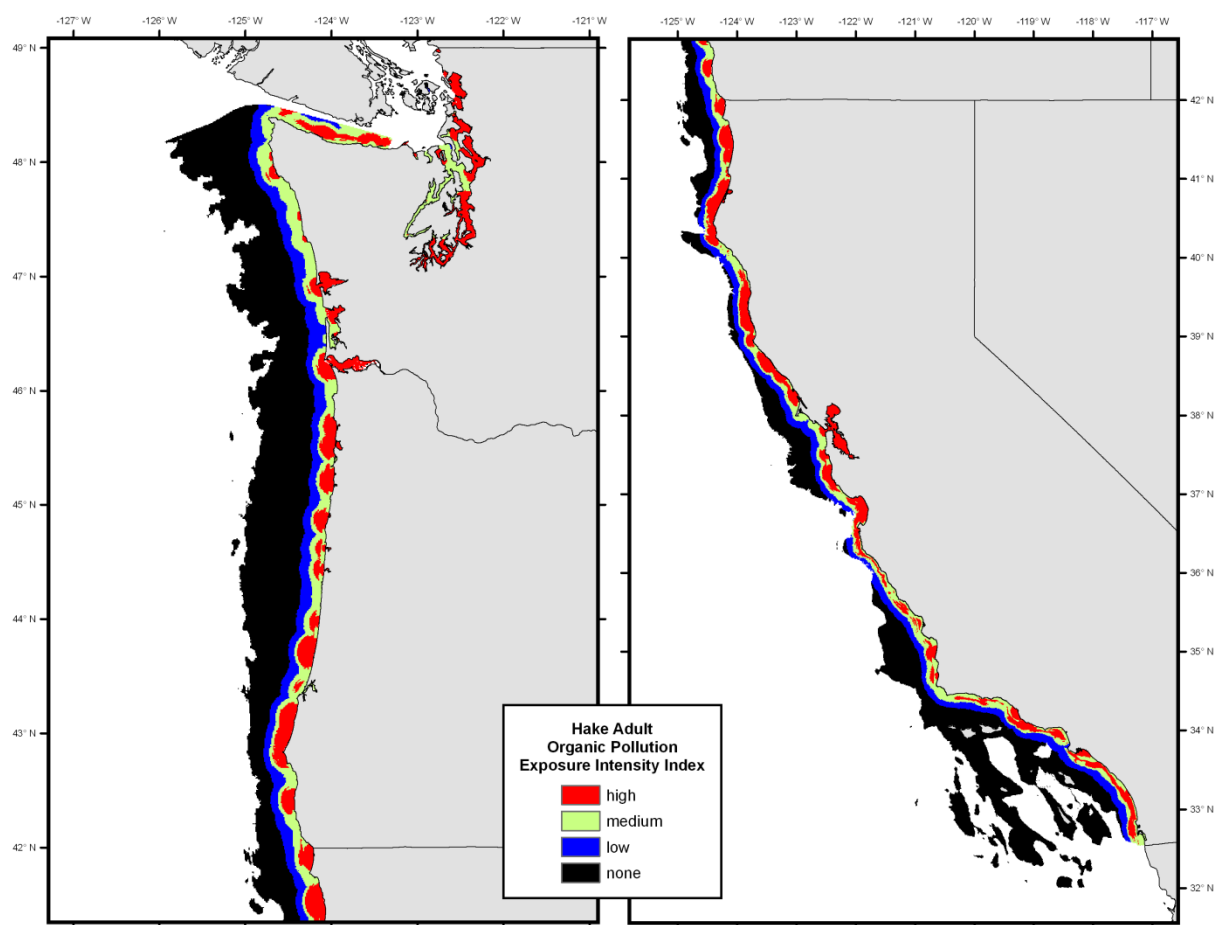


Figure GFR37. Exposure intensity index of organic pollution for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

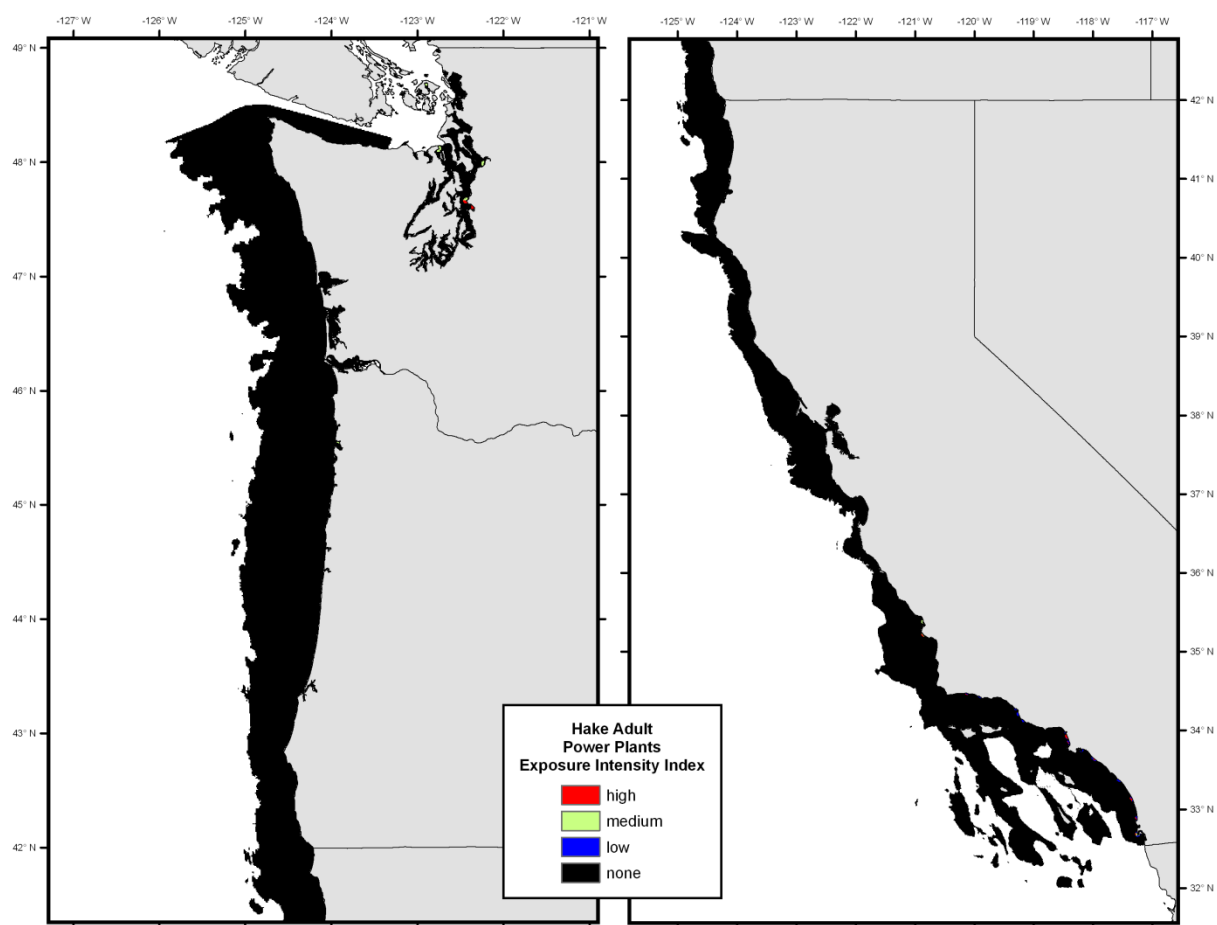


Figure GFR38. Exposure intensity index of coastal seawater exchange activity for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

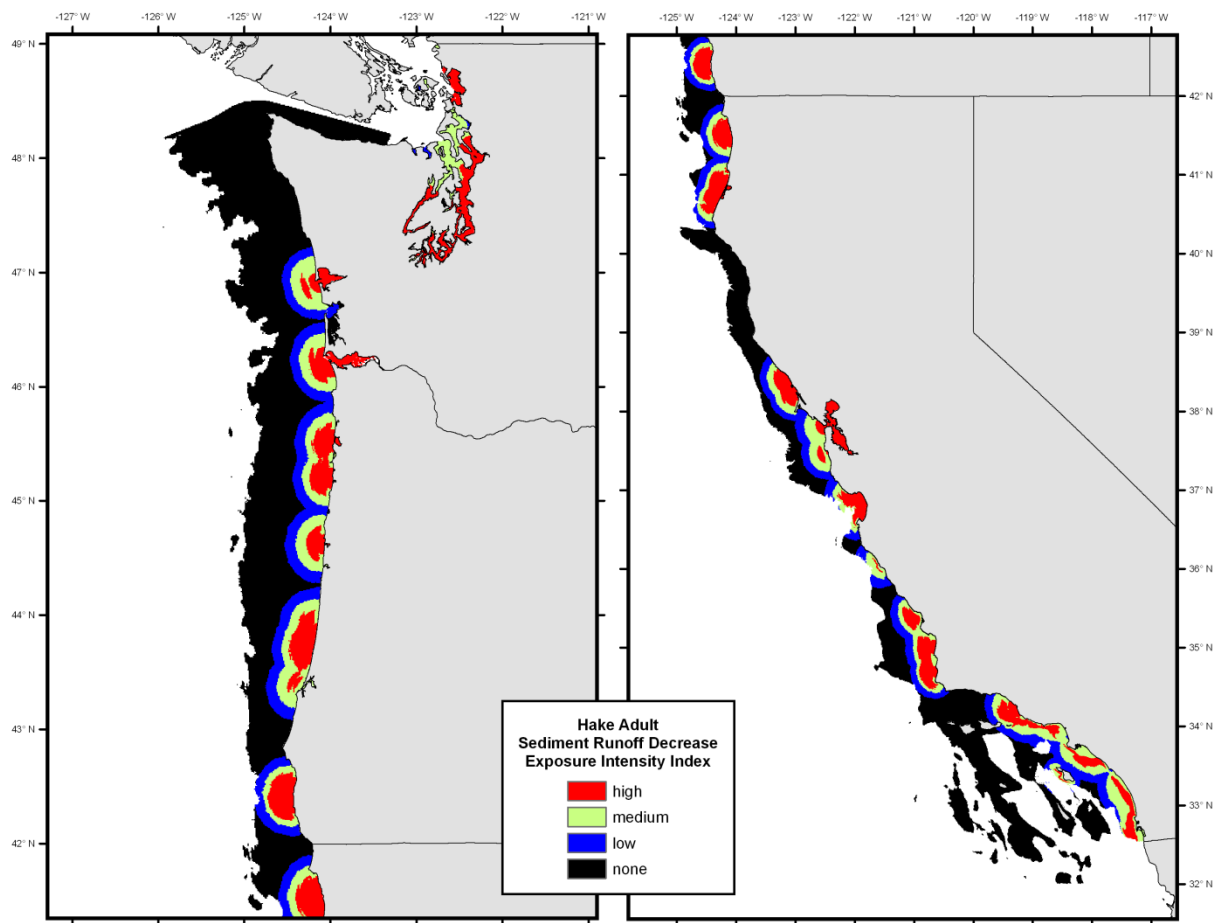


Figure GFR39. Exposure intensity index of sediment runoff decrease for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

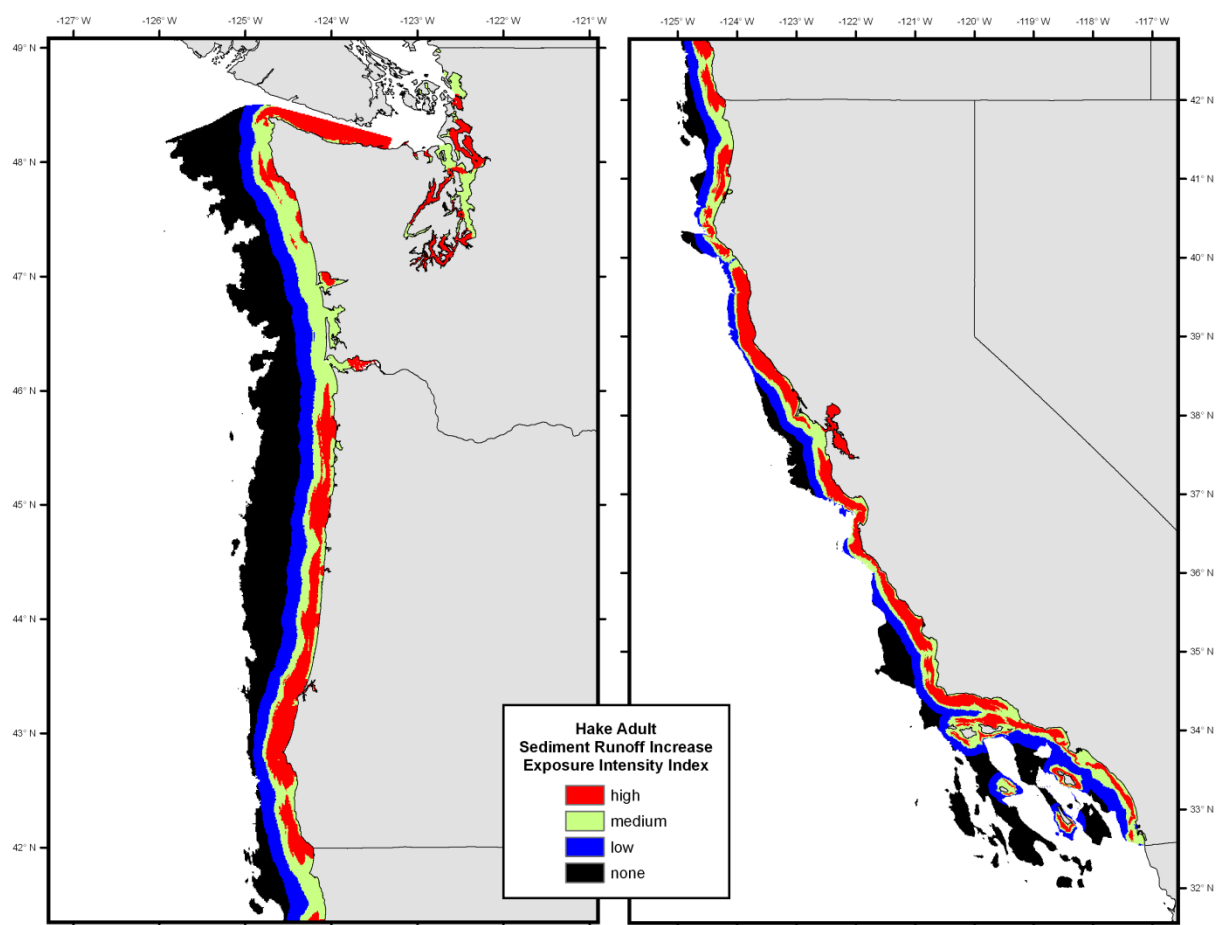


Figure GFR40. Exposure intensity index of sediment runoff increase for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

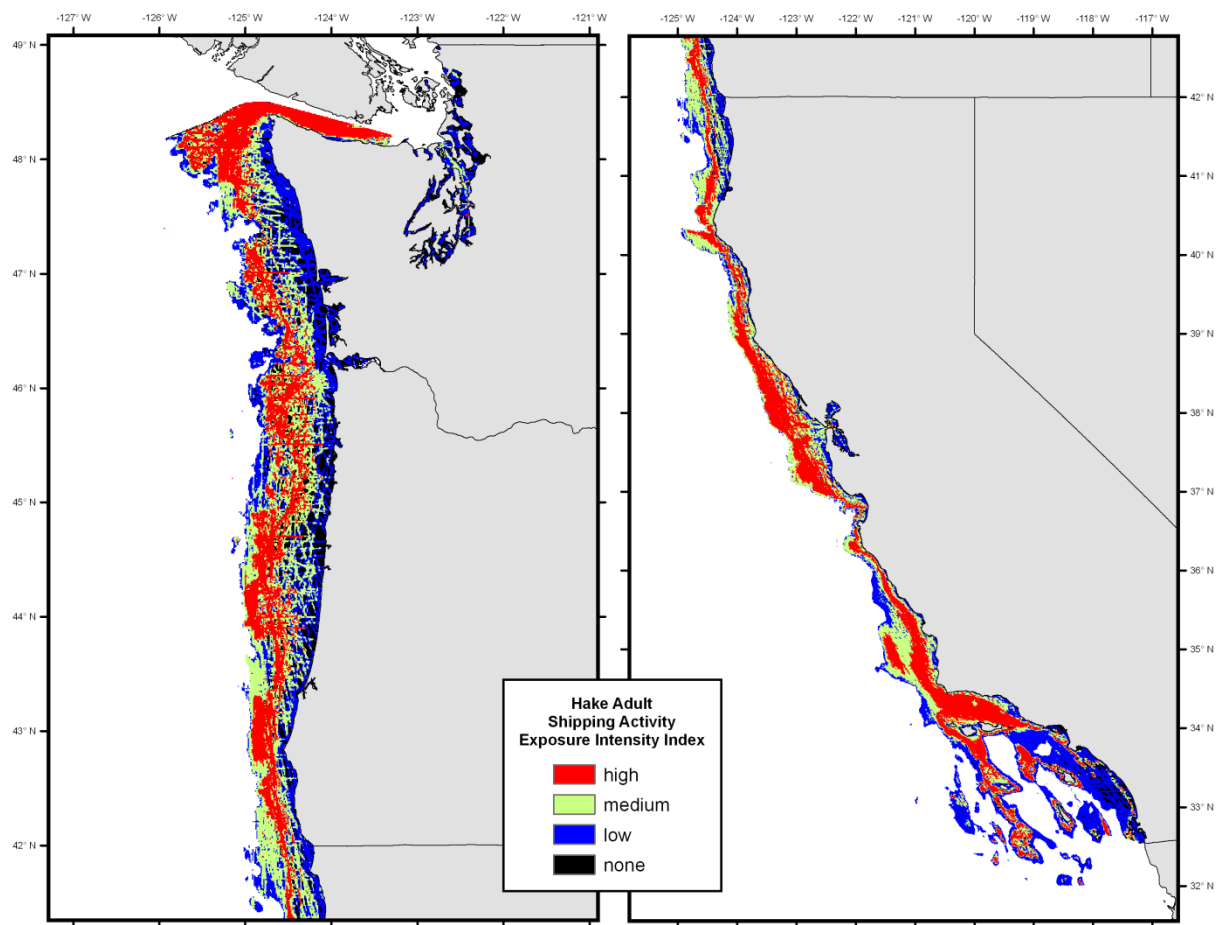


Figure GFR41. Exposure intensity index of shipping activity for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

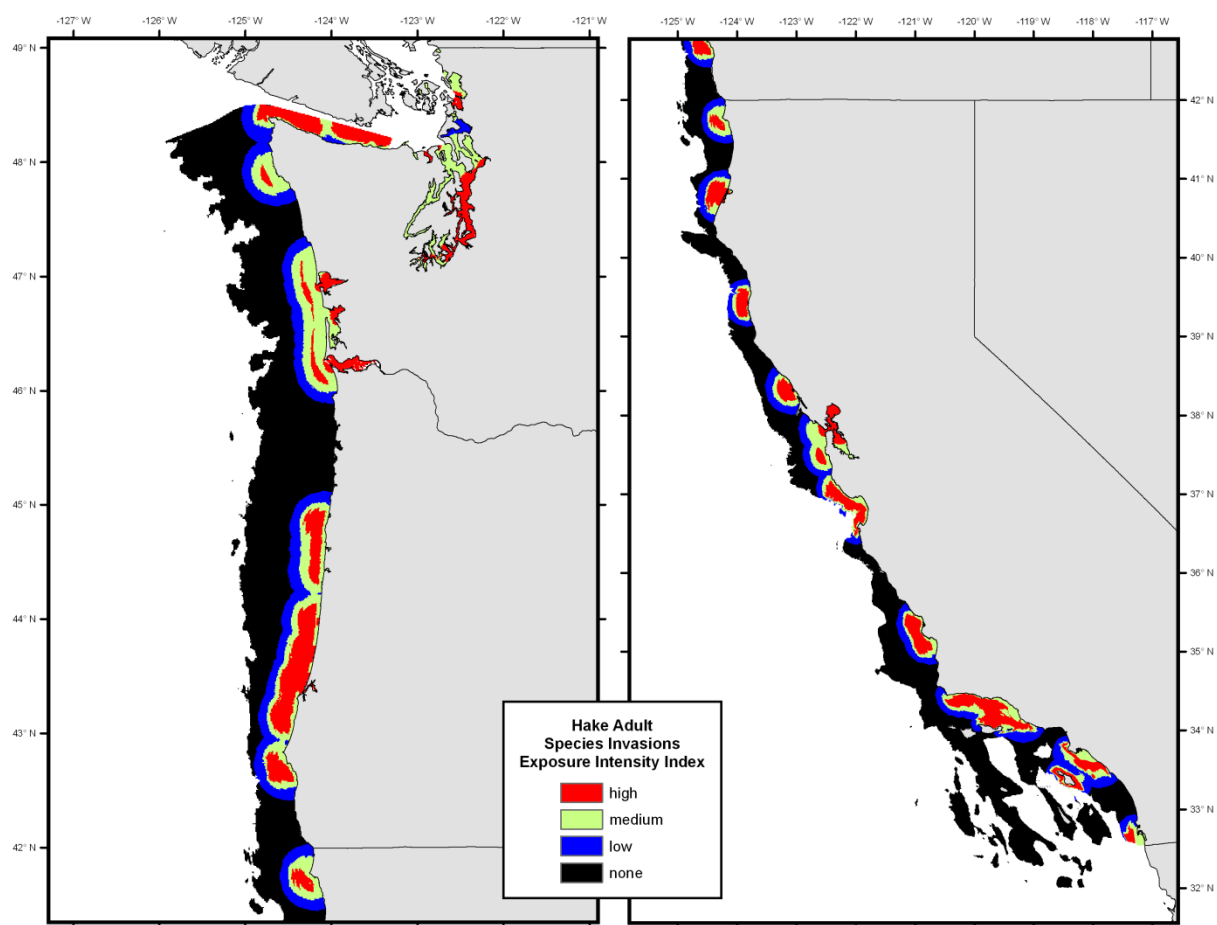


Figure GFR42. Exposure intensity index of species invasions for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

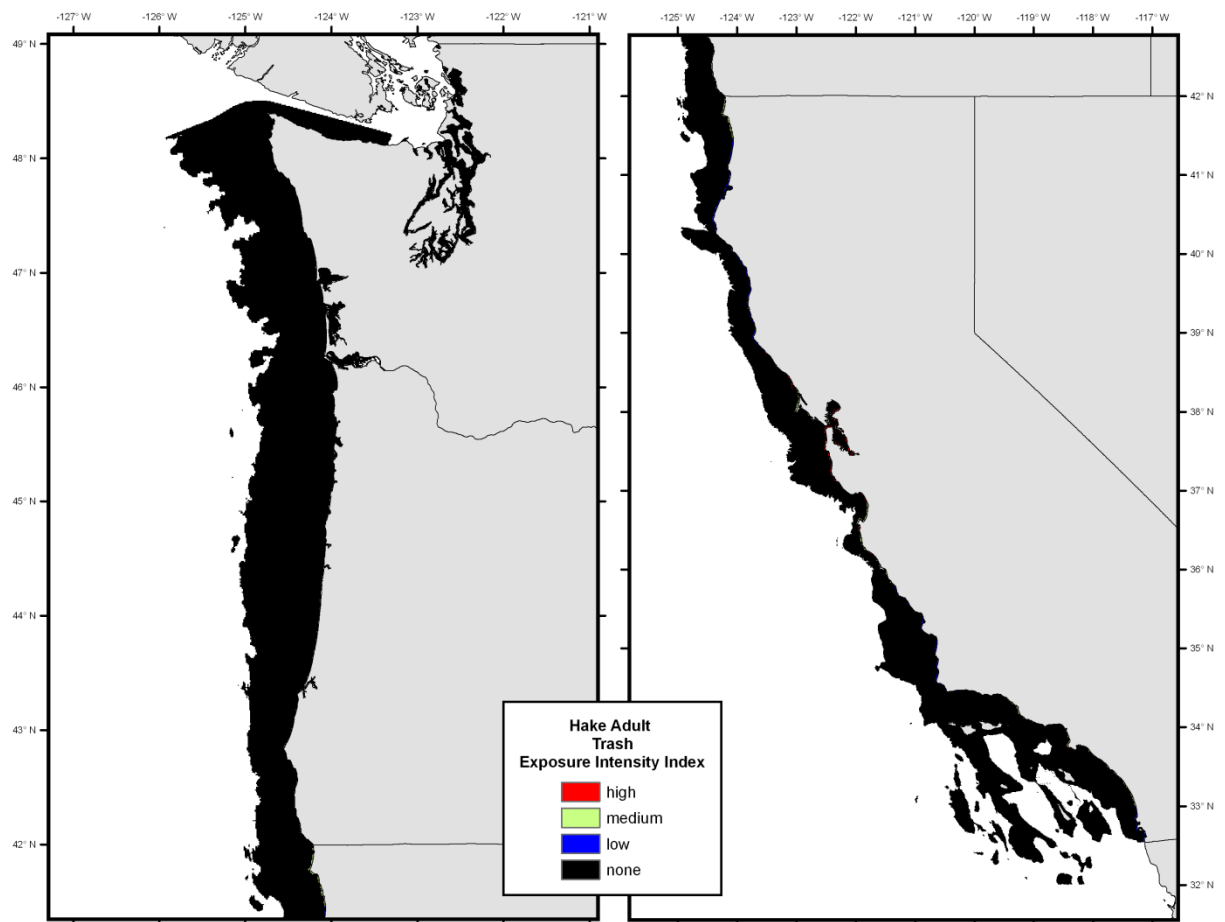


Figure GFR43. Exposure intensity index of coastal trash for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

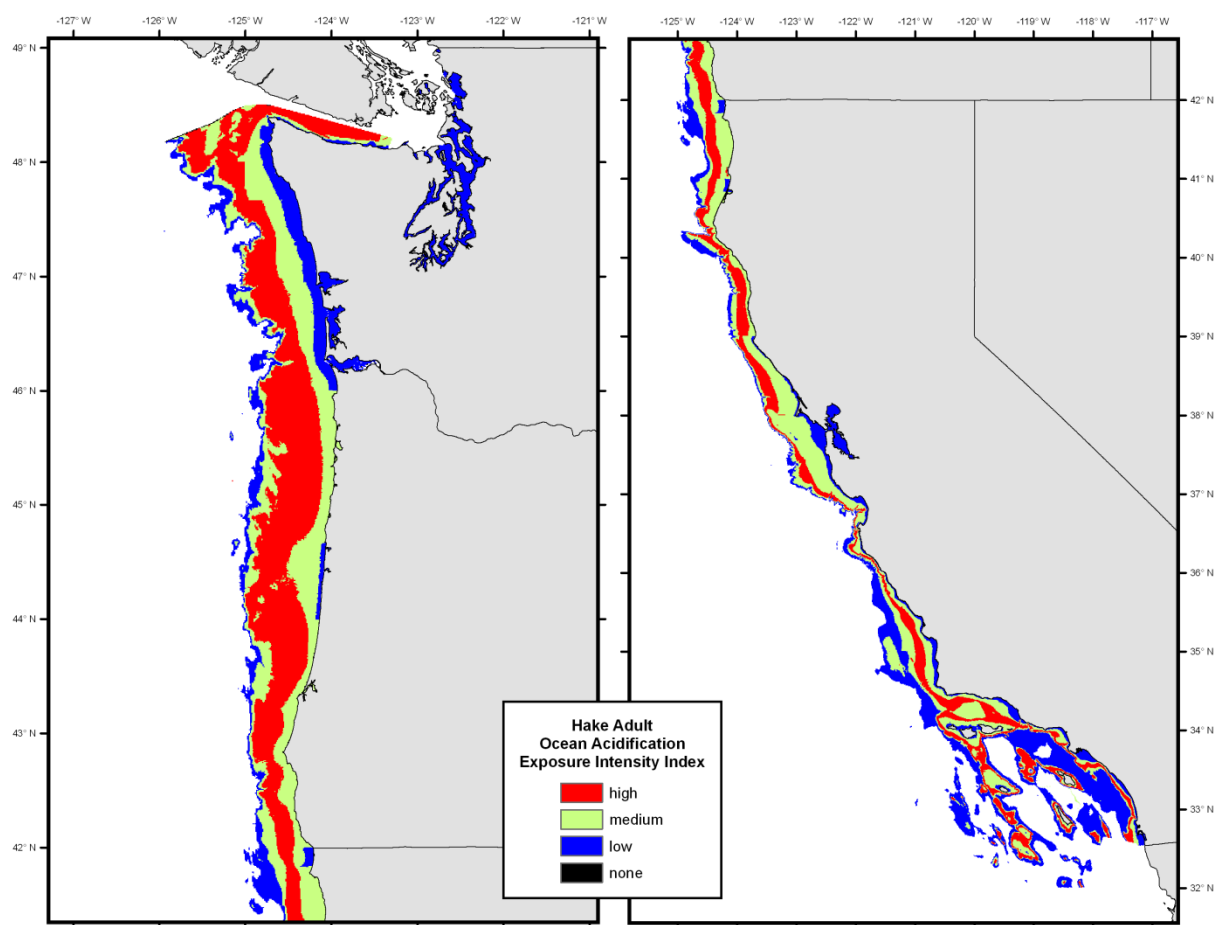


Figure GFR44. Exposure intensity index of ocean acidification for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

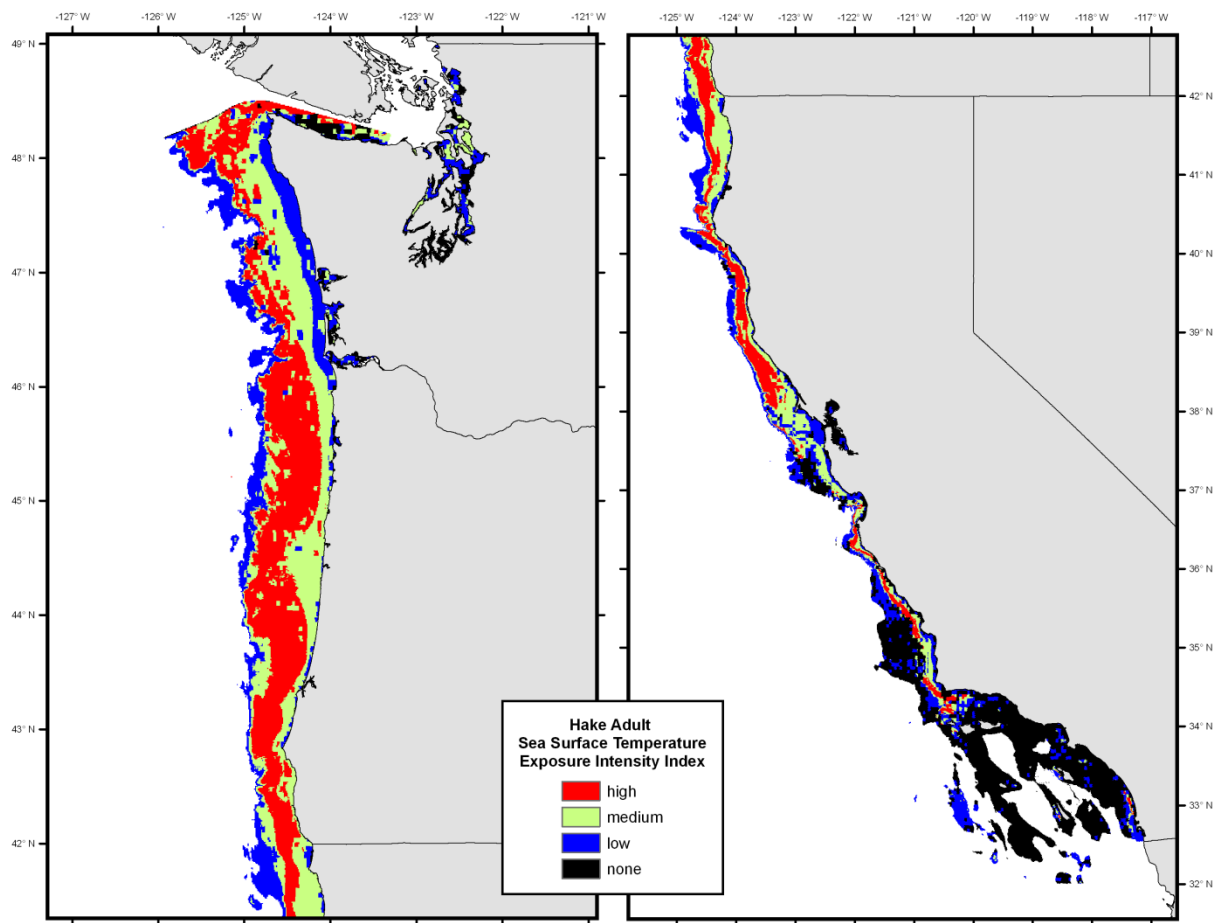


Figure GFR45. Exposure intensity index of sea-surface temperature for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

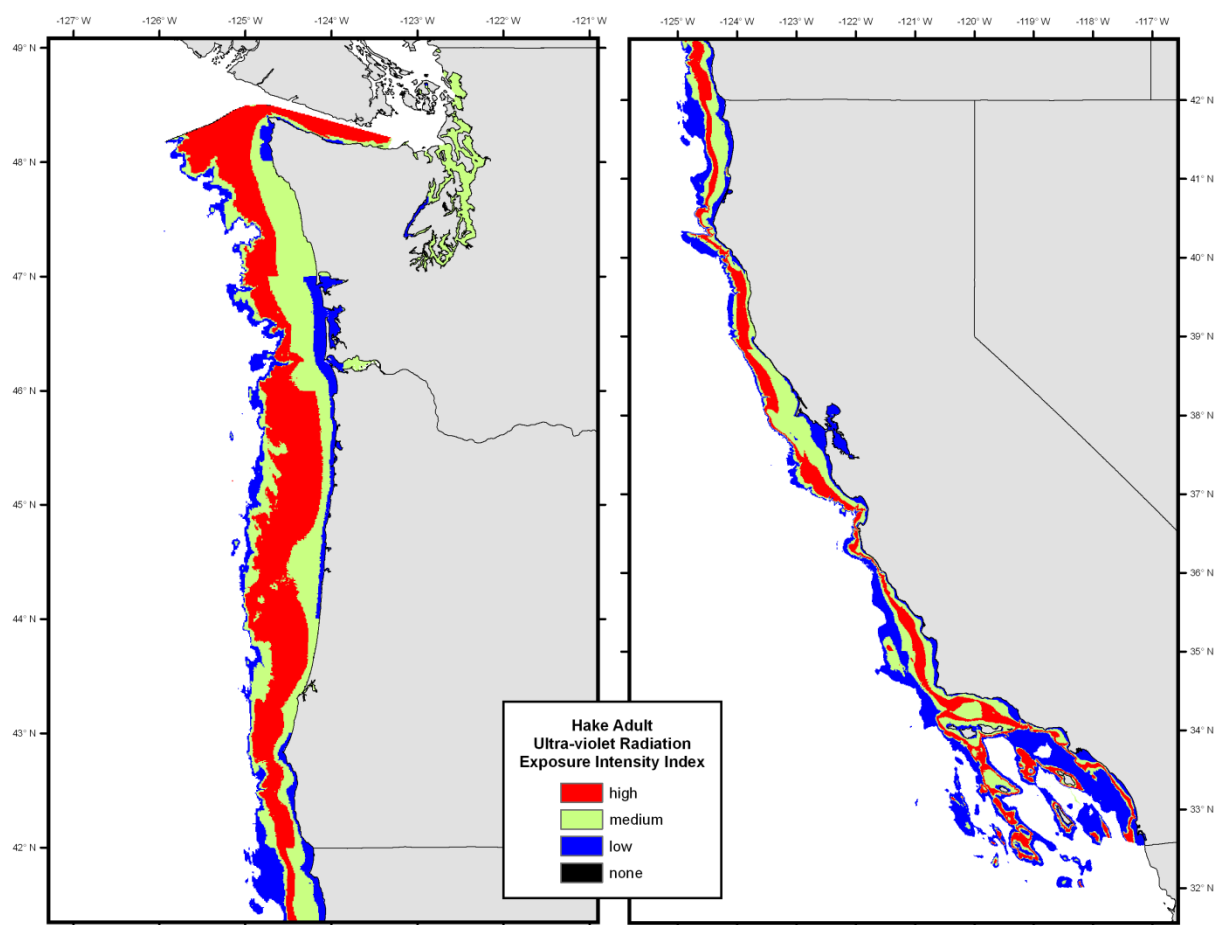


Figure GFR46. Exposure intensity index of ultra-violet radiation for Pacific hake *Merluccius productus* adult. High = upper tercile, Medium = middle tercile, low = lower tercile.

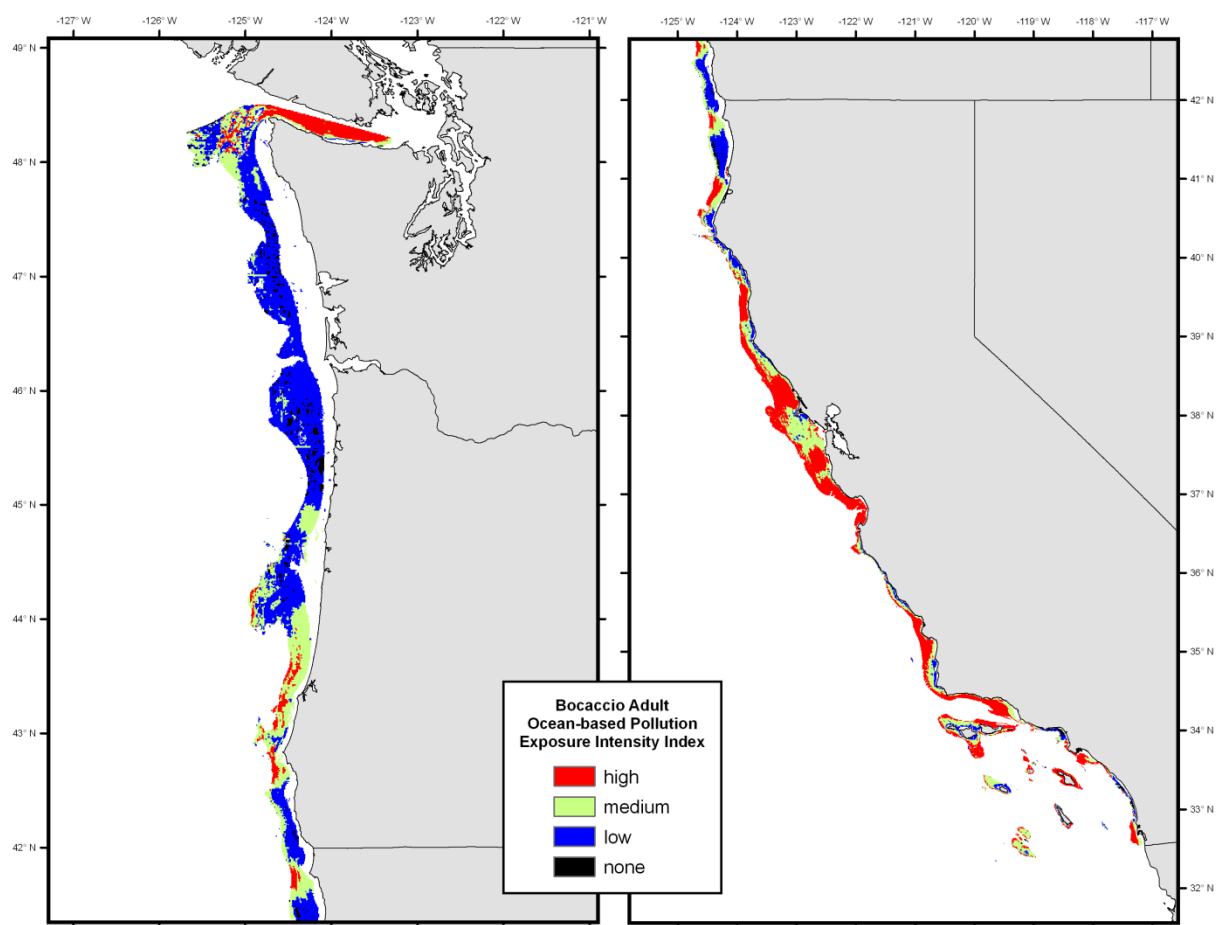


Figure GFR47. Exposure intensity index of ocean-based pollution for bocaccio *Sebastes paucispinis* rockfish adults. High = upper tercile, Medium = middle tercile, low = lower tercile.

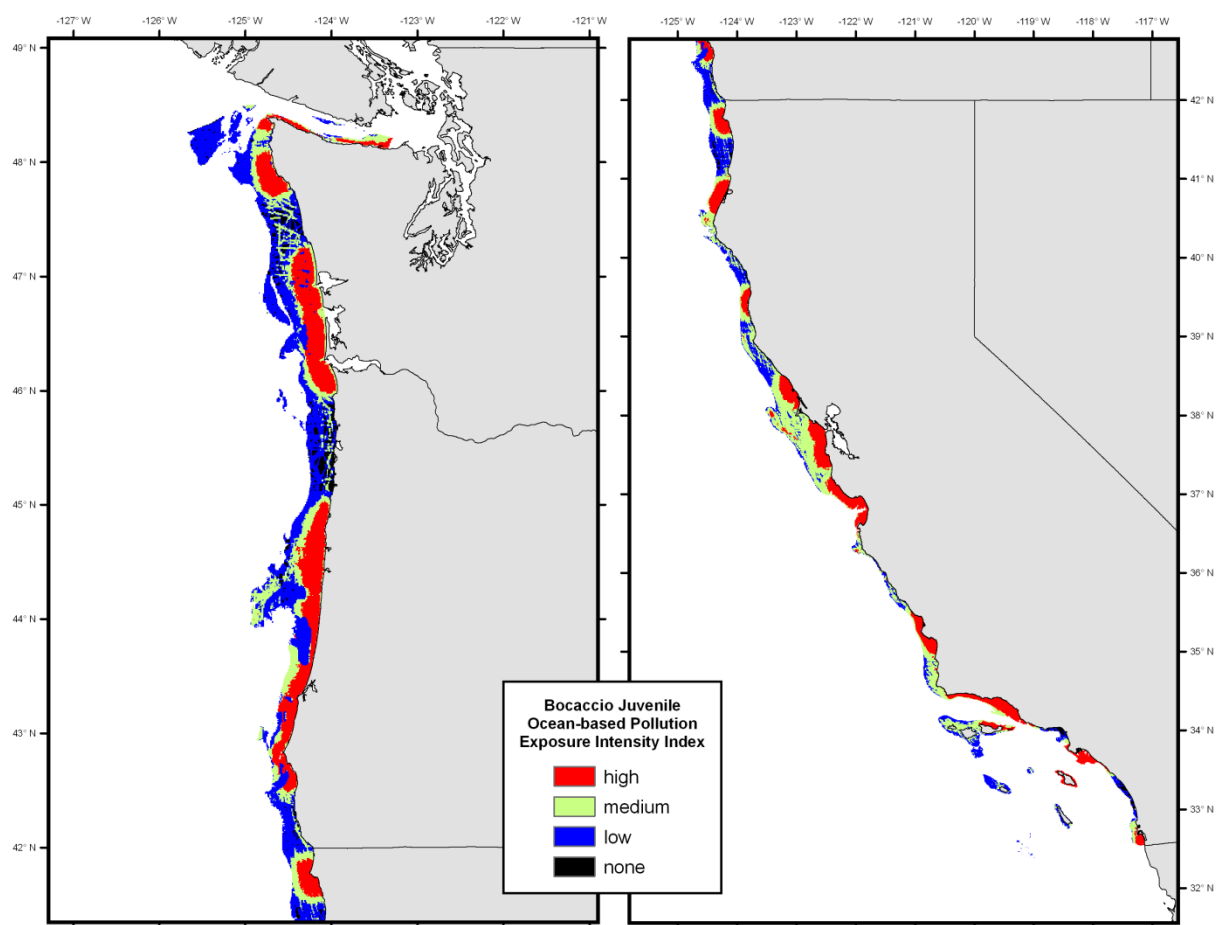


Figure GFR48. Exposure intensity index of ocean-based pollution for bocaccio *Sebastes paucispinis* rockfish juveniles. High = upper tercile, Medium = middle tercile, low = lower tercile.

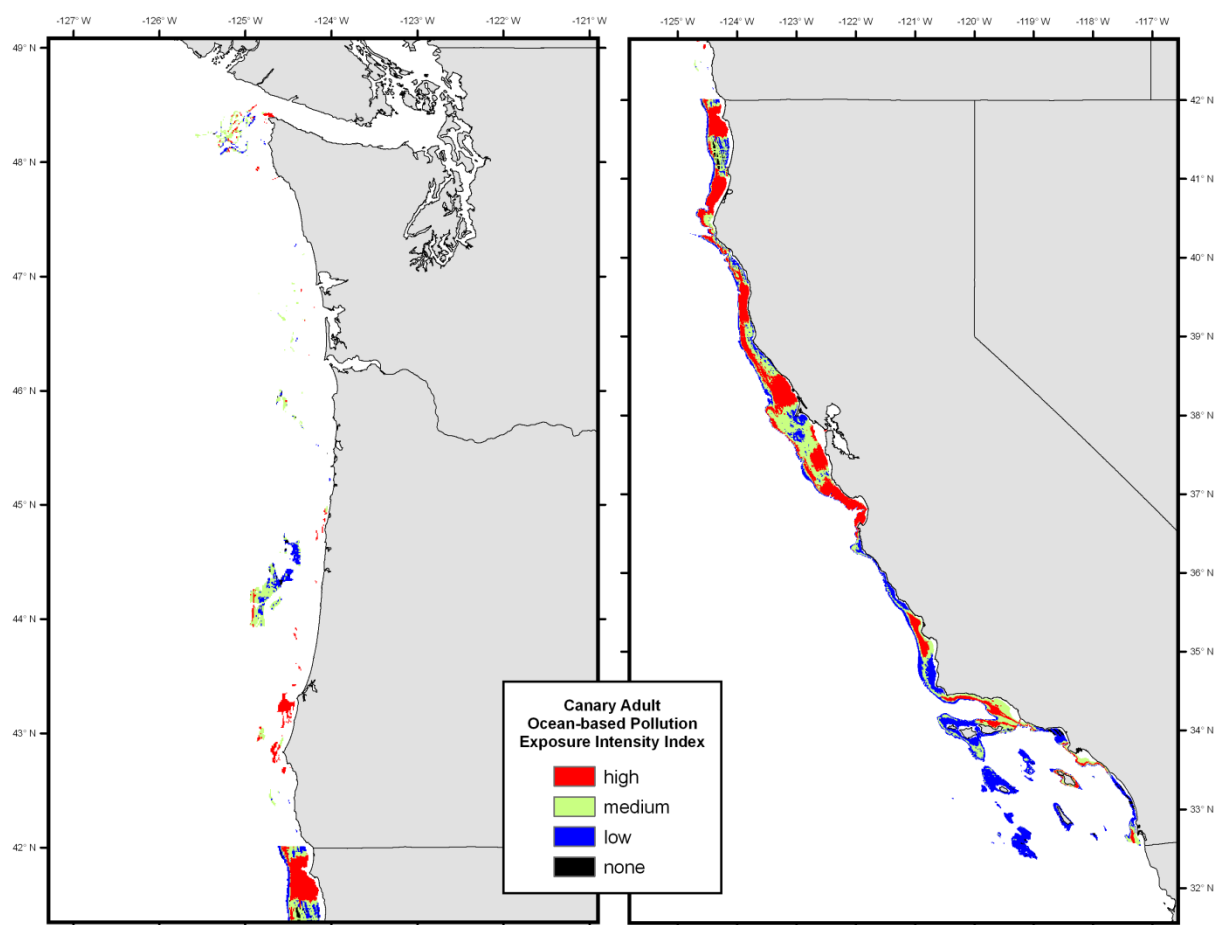


Figure GFR49. Exposure intensity index of ocean-based pollution for canary *Sebastes pinniger* rockfish adults. High = upper tercile, Medium = middle tercile, low = lower tercile.

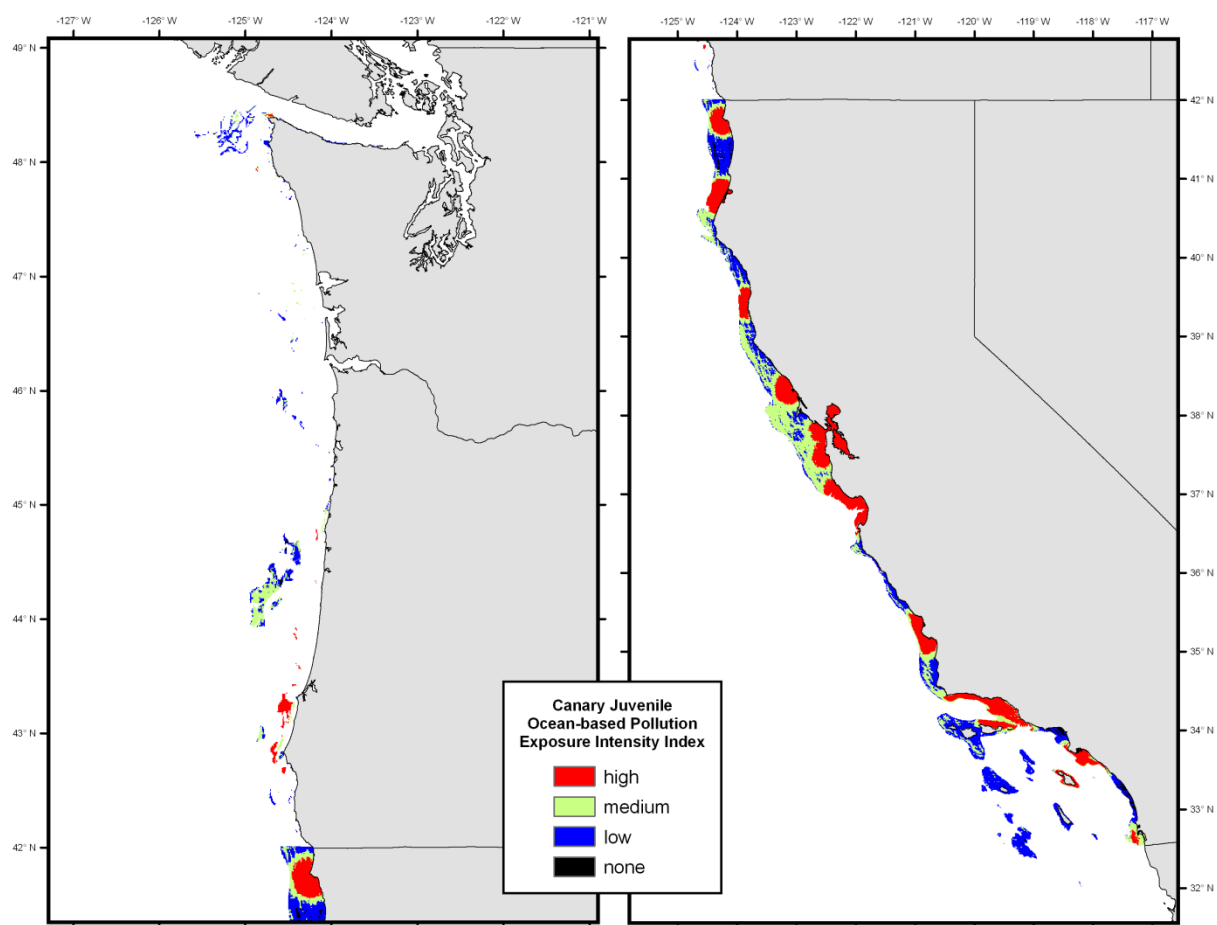


Figure GFR50. Exposure intensity index of ocean-based pollution for canary *Sebastes pinniger* rockfish juveniles. High = upper tercile, Medium = middle tercile, low = lower tercile.

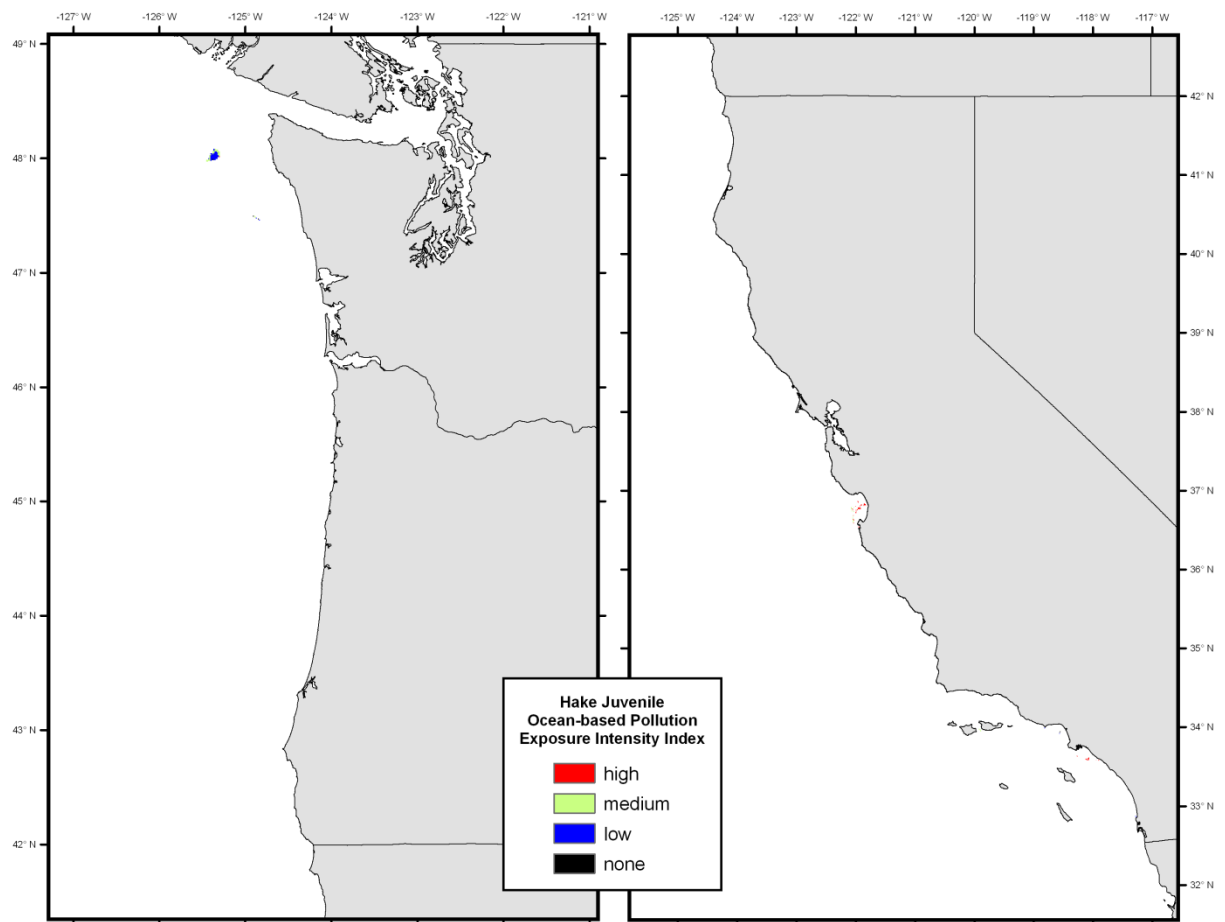


Figure GFR51. Exposure intensity index of ocean-based pollution for Pacific hake *Merluccius productus* juveniles. High = upper tercile, Medium = middle tercile, low = lower tercile.

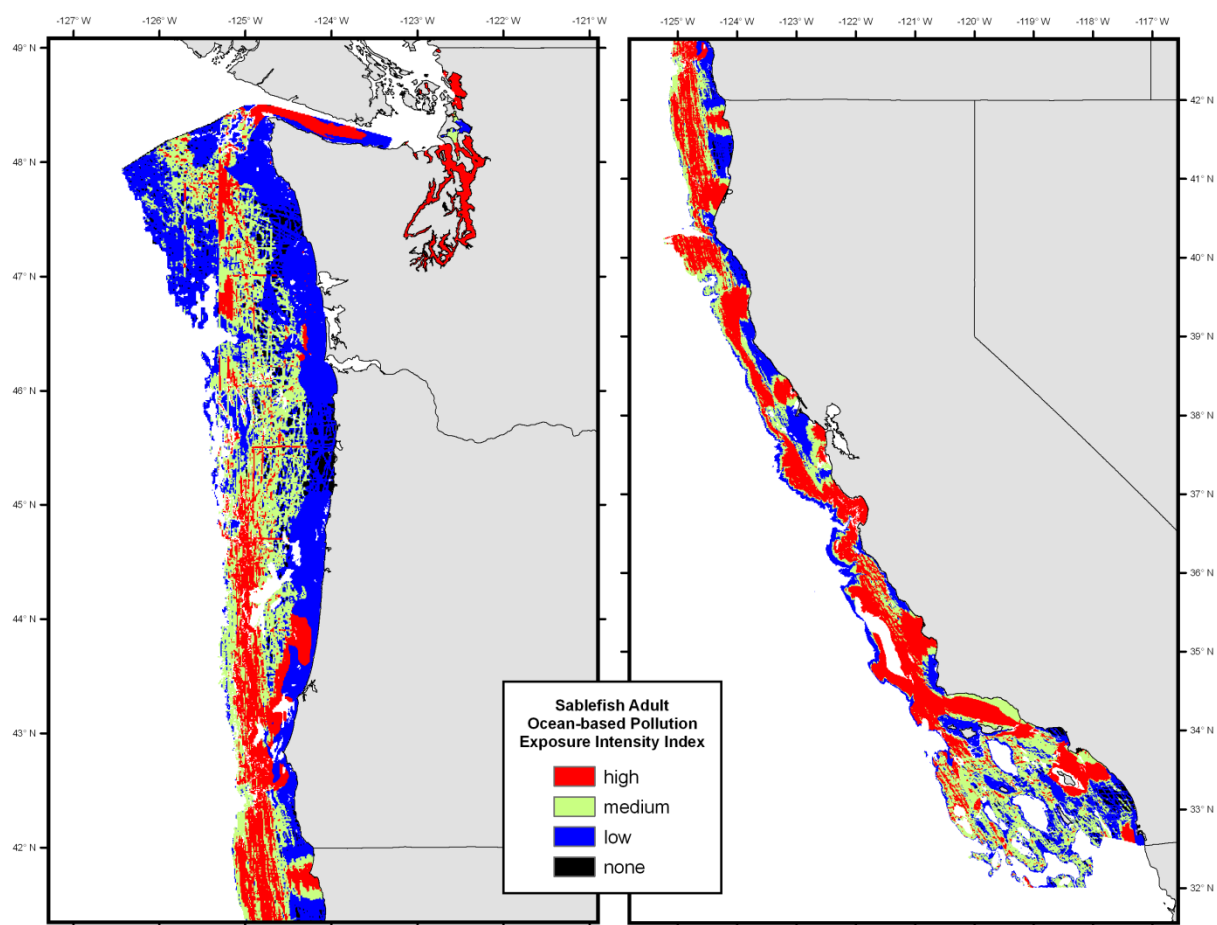


Figure GFR52. Exposure intensity index of ocean-based pollution for sablefish *Anoplopoma fimbria* adults. High = upper tercile, Medium = middle tercile, low = lower tercile.

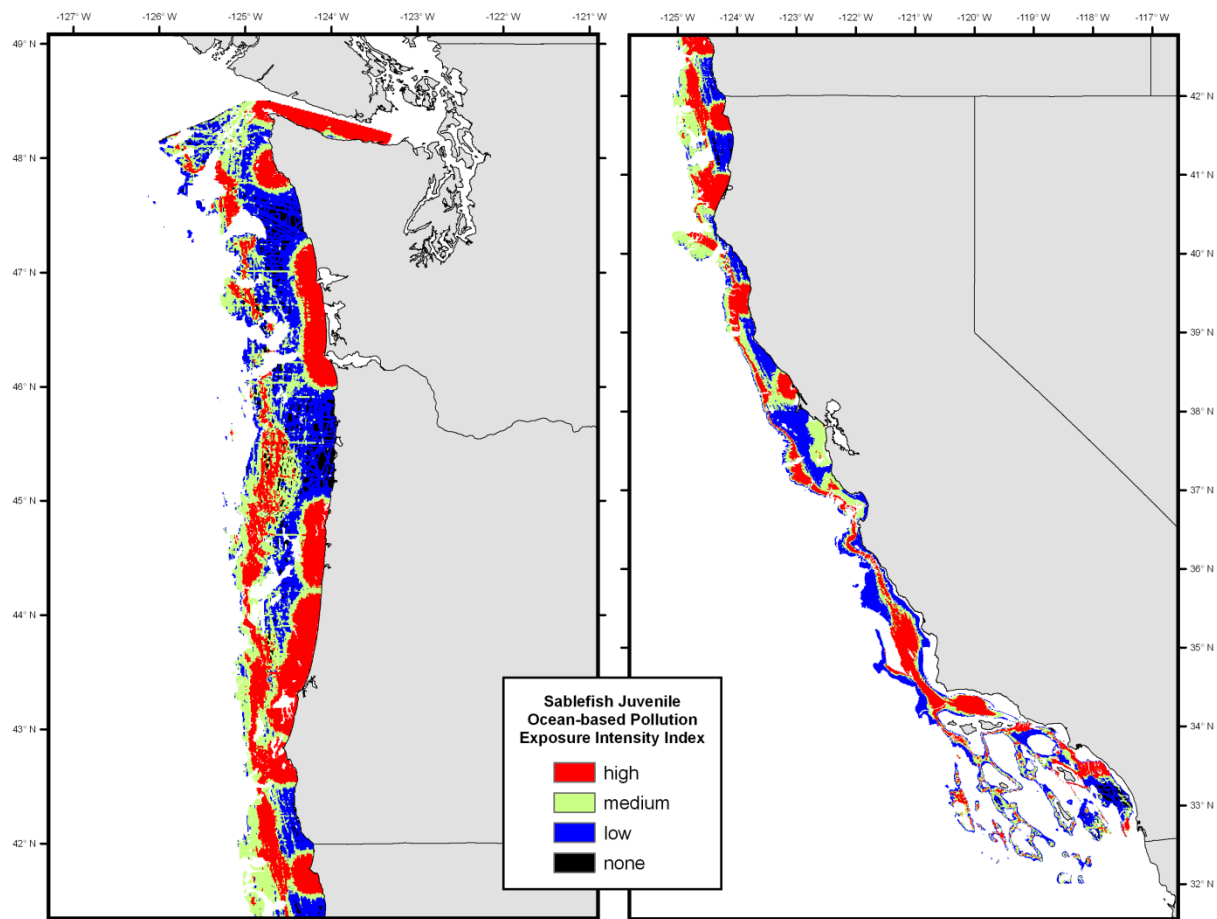


Figure GFR53. Exposure intensity index of ocean-based pollution for sablefish *Anoplopoma fimbria* juveniles. High = upper tercile, Medium = middle tercile, low = lower tercile.

GROUND FISH APPENDIX A

The HSP that we used were developed during the 2005 EFH EIS process. This work is scheduled to be updated every 5 years, so the HSP data that we used in this analysis may be updated in the near future that would improve the underlying data. Of particular interest is the HSP for juvenile Pacific hake (Fig. 6). Currently, the habitat is limited to a few locations. Depending on the definition of 'juvenile', the habitat identified for juvenile hake may be much more expansive than the current analysis.

Detailed information about the development of the data and analytical procedures used to produce the HSPs are described in the document: *Pacific States Marine Fisheries Commission. 2004. Risk Assessment for the Pacific Groundfish FMP*, which is included as Appendix A to the FEIS. Additionally, Appendix D of this document includes a *Report on Updates Made to the Production of Essential Fish Habitat Suitability Probability Maps* (<http://www.nwr.noaa.gov/Groundfish-Halibut/Groundfish-Fishery-Management/NEPA-Documents/EFH-Final-EIS.cfm>).

The shape files (GIS compatible files) for each species/life-history stage are separated into five geographic regions along the U.S. West Coast due to computer processing limitations during the analysis. We used the 'merge' command in ArcView version 9.3 to combine all regions into one combined data layer. In some of the shape files, polygons were created where HSP equaled 0. This appeared to be due to a few geographic border lines drawn that do not represent changes in HSP values. In order to keep these cells from showing up as habitat ('none' category for exposure intensity index) in further analyses, we changed all the 0 values in each HSP data layer to -9999 (represents 'no data').

NON-FISHERIES THREATS DATA

First, we downloaded the GeoTiff files projected in Arc System Zone 2 for each of the 19 non-fisheries related threats (or impacts) from the National Center for Ecological Analysis and Synthesis's website (http://www.nceas.ucsb.edu/globalmarine/ca_current_data). We created pyramids for each of the files using ArcCatalog version 9.3 and then brought each of the files into ArcView. Each file was then converted into a GRID file using the RasterToOther Conversion tool in the ArcView Toolbox.

For all threats except shipping, we assumed that the threat affected all depths of the water column. For example, if a grid cell had a value of 0.5 for organic pollution, we assumed this threat affected species inhabiting the water column at all depths including the bottom. For shipping, we made a correction to the threat value to take into account that shipping most likely affects the top 20 m of the water column, such that individuals on the bottom are not exposed to this threat. So, we limited the shipping data to depths of 20m or less for bocaccio, canary and sablefish, i.e. for grid cells that were at depths > 20 m, we multiplied the threat value by 0. For Pacific hake, we estimated a proportion of the population that migrates up into the water column at depths less than 20m based on primary literature because most surveys of hake populations do not measure the top 50 m of the water column (D. Chu, Northwest Fisheries Science Center, *pers comm.*). Juvenile hake show vertical distribution into shallow depths of the water column, particularly at night. Sakuma & Ralston (1997) present data showing that ~1/3 of juveniles collected were at 10 m, 1/3 were at 40 m, and 1/3 were found at 100 m); thus, we multiplied the threat value by 0.334 as an estimate of the proportion of juveniles that would be exposed to shipping*. For adults, some small proportion of adult hake migrate into this depth zone (0-20m) at night, typically feeding on euphausiid populations which are vertically migrating and concentrate near 20 m between 2400-0200 hrs (Alverson & Larkins 1969). Adult hake migrate on a diurnal schedule: fish are dispersed from near surface to 20- m depth at night (10 p.m. to 3 a.m.), descend quickly at dawn and form schools; and rise to the surface at night in 30-40 min (Nelson and

Larkins 1970; Ermakov 1974). These diurnal migrations have been compared to the migrations of their primary prey, euphausiids, as a causal mechanism (Alton and Nelson 1970). Because juveniles are most likely found in the upper water column at greater proportions, we used an estimate of 10% for the proportion of adult hake that migrate into the top 20m of the water column at some point*; therefore, we multiplied the shipping threat values by 0.1 in order to account for this level of exposure.

GROUND FISH APPENDIX B - NON-FISHERIES THREATS – LITERATURE REVIEW

In the sections below labeled “Threat data layer description, from Halpern et al. (2009)”, we have copied information from Halpern et al. (2009) supporting materials; thus, any use of “we” or “our” refers to analyses or work performed by the authors of the original paper.

Information on trends in the threats described below can be found in the Anthropogenic Drivers and Pressures Section of the IEA.

AQUACULTURE

Threat data layer description, from Halpern et al. (2009): Currently no data exist for the location of aquaculture facilities. Google Earth imagery was used to search the coastlines in the California Current for evidence of fish pens. This effort was focused on Puget Sound, Southern California, and Baja, Mexico where aquaculture is known to exist. Data on shellfish aquaculture facilities are not included because they do not exist at this time.

Effects: The impact of aquaculture facilities varies according to the species cultured, the type and size of the operation, and the environmental characteristics of the site (Johnson et al. 2008). Intensive cage and floating netpen systems typically have a greater impact because aquaculture effluent is released directly into the environment. The relative impact of finfish and shellfish aquaculture differs depending on the foraging behavior of the species. Finfish require the addition of a large amount of feed into the ecosystem, which can result in environmental impacts from the introduction of the feed, but also from the depletion of species harvested to provide the feed. Bivalves are filter feeders and typically do not require food additives; however, fecal deposition can result in benthic and pelagic habitat impacts, changes in trophic structure and nutrient and phytoplankton depletion. Aquaculture activities can effect fisheries at both a habitat and species-level. Typical environmental impacts resulting from aquaculture production include: (1) impacts to the water quality from the discharge of organic wastes and contaminants; (2) seafloor impacts; (3) introductions of exotic invasive species; (4) food web impacts; (5) gene pool alterations; (6) changes in species diversity; (7) sediment deposition; (8) introduction of diseases; (9) habitat replacement or exclusion; and (10) habitat conversion (Johnson et al. 2008).

SENSITIVITY SCORES

Mortality: 1 (juvenile and adult forms of all species). Mortality effects are not likely from the range of current aquaculture activities in the region.

Behavior: 3 (juvenile and adult forms of bocaccio and canary rockfish); 2 (juvenile and adult forms of hake and sablefish). Direct negative behavioral effects likely to affect species drawn to habitat structure; from the range of current aquaculture activities in the region, although indirect effects are likely via water quality, light, seafloor and related habitat impact, etc.

Physiology: 2 (juvenile and adult forms of all species). Some negative effects to physiology of all species due to exposure to parasites, nutrient input/fecal coliforms, and chemicals associated with aquaculture/net pen facilities.

ATMOSPHERIC DEPOSITION

Threat data layer description, from Halpern et al. (2009) : We used the atmospheric deposition of sulfates derived from the National Atmospheric Deposition Program (<http://nadp.sws.uiuc.edu/>), processed in the same manner as for nitrogen as described above in ‘Nutrient Input’. We used sulfate deposition as a proxy measure for the distribution and deposition of all atmospheric pollutants.

Effects: Substances such as sulfur dioxide, nitrogen oxide, carbon monoxide, lead, volatile organic compounds, particulate matter, and other pollutants are returned to the earth through either wet or dry atmospheric deposition (Johnson et al. 2008). Atmospheric pollution is a major source of many nutrient, chemical, and heavy metal pollutants whose sources can be far away from the marine ecosystems being impacted. See pollutants, above.

SENSITIVITY SCORES

Mortality: 3 (juvenile forms of all species); 2 (adult forms of all species). Scored as if inorganic/organic pollution; Sensitivity scores reflect that most fish species are particularly sensitive to contaminants/pollution during early life history.

Behavior: 2 (juvenile and adult forms of all species) There is no apparent behavioral response that would reduce or increase sensitivity to this threat.

Physiology: 3 (juvenile forms of all species); 2 (adult forms of all species). Body size, age, feeding ecology, and trophic position are some of the most important factors determining bioaccumulation in marine fishes. Most species have no apparent physiological response (i.e. they do not metabolize these pollutants to remove them) that would reduce sensitivity to this threat.

COASTAL ENGINEERING

Coastal engineering Threat data layer description, from Halpern et al. (2009) : Coastal engineering represents shore hardening of various kinds, including riprap walls, cement walls (for harbors, sediment containment, etc.), and jetties and piers. For coastlines within the United States, we extracted data from NOAA's Environmental Sensitivity Index (ESI) for California, Puget Sound and Columbia River regions (<http://response.restoration.noaa.gov>) and from The Nature Conservancy's (TNC) Pacific Northwest coast ecoregional assessment geodatabase (Ferdana et al. 2006) for Oregon and Washington. These databases classify linear segments of coast into ecosystem types and also report location of hardened shorelines. For Baja, Google Earth images were generally at high enough resolution to be able to identify human-modified shorelines, but where they were not we assumed no coastal engineering exists.

Effects: Coastal engineering structures destroy the habitat directly under them and can significantly modify surrounding ecosystems through changes in circulation patterns and sediment transport (National Research Council 2007; Halpern et al. 2009b; Shipman et al. 2010). Any structural modification of the shoreline will alter several important physical processes, and can therefore be considered an impact (Williams and Thom 2001). For the most part, impact potential can be related to the size and location of the structure and the types of physical processes it alters. Impacts may be considered direct or indirect. Direct impacts are generally associated with construction activities, including excavation, burial, and various types of pollution. Indirect impacts occur following physical disturbance, and are chronic in nature due to permanent alteration of physical processes such as sediment transport and wave energy. "Cumulative impacts" are associated with increasing number or size of indirect or direct impacts, which can have either linear or non-linear cumulative responses. Many shoreline "hardening" structures, such as seawalls and jetties, tend to reduce the complexity of habitats and the amount of intertidal habitats (Williams and Thom 2001). Differences in fish behavior and usage between modified and unmodified shorelines are caused by physical and biological effects of the modifications, such as changes in water depth, slope, substrate, and shoreline vegetation (Toft et al. 2007).

SENSITIVITY SCORES

Mortality: 1 (juvenile and adult forms of all species). We assume most of the chronic effects of coastal engineering structures on fishes will be behavioral in nature.

Behavior: 3 (juvenile and adult forms of bocaccio and canary rockfish); 2 (juvenile and adult forms of hake, juvenile form of sablefish); 1 (adult form of sablefish). Most coastal engineering impacts will affect behavior of species highly dependent on benthic habitat structure (i.e., rockfish would be attracted to structure; flatfish would avoid structure). Direct effects of construction activity (noise, disturbance) would cause avoidance behavior for all species

Physiology: 2 (juvenile and adult forms of all species). We assume most of the chronic effects of coastal engineering structures on fishes will be behavioral in nature.

DIRECT HUMAN IMPACTS

Threat data layer description, from Halpern et al. (2009) : To estimate the impact of this source of stress, we employed a 3 step process. First, we collected annual beach attendance data that are available for 98 beaches in Central and Southern California (Kildow and Colgan 2005; Dwight et al. 2007)(http://www.parks.ca.gov/?page_id=23308). Of these, only 59 have additional information on fees, facilities, and parking availability. U.S. beach access points in the California Current are reported in the MLPA database for California (<http://marinemap.org/mlpa>), the Oregon Geospatial Enterprise Office (<http://gis.oregon.gov/DAS/EISPD/GEO/alphabetical.shtml>), and Washington State Department of Ecology BEACH (Beach Environmental Assessment, Communication and Health) Program (<http://www.doh.wa.gov/ehp/TS/WaterRec/beach/default.htm>). Second, we used these actual beach attendance data to develop a predictive model of beach visitation for all access points without recorded data. Predictor variables included number of parking spaces (park), entrance fee (fee), available facilities (facils: a yes/no variable) and number of people within 50 miles of the access point (pop). Fifty miles was chosen because studies of beach attendance (in southern California) suggest most visitors are local and travel 50-80 miles from home to get to the beach (Dwight et al. 2007; Nelsen et al. 2007). Population density data come from the LandScan project (<http://www.ornl.gov/sci/landscan/index.html>) and are reported at 1km² resolution. We implemented a backwards selection procedure of a multivariate linear model on these variables, and used AIC to select the best model. The final model for predicting annual beach access (BA) was $BA = 0.1706(pop) - 16840$ ($F = 9.743$, $df = 2,94$, $p < 0.001$, adjusted $R^2 = 0.15$). We then applied this model to all beach access points without real attendance data. These annual beach access values were then used as estimates of the relative intensity of direct human impact on that pixel of coastline. Beach access point data were not available for Baja, so this impact was not estimated along the Mexican coastline.

Effects: People visiting beaches and coastal areas can impact intertidal and nearshore ecosystems through direct trampling or by disturbing or displacing species that would normally use those locations. None of these species are sessile intertidal inhabitants and therefore they would not be subject to this type of disturbance.

SENSITIVITY SCORES

Mortality: 1 (juvenile and adult forms of all species). Trampling and disturbance is not likely to affect species in water column or near bottom.

Behavior: 1 (juvenile and adult forms of all species) Trampling and disturbance is not likely to affect species in water column or near bottom.

Physiology: 2 (juvenile and adult forms of all species). The physiological response to trampling and disturbance does not change sensitivity to this threat.

INORGANIC POLLUTION

Inorganic pollution Threat data layer description, from Halpern et al. (2009): Inorganic pollution into coastal marine waters was estimated from two sources, point source pollution from factories and mines and non-point source pollution that scales with the amount of impervious (hardened) surface area. Point source

data are reported in the EPA Toxics Release Inventory (<http://www.epa.gov/tri/>). We multiplied the amount of each chemical released on-site to the ground or water (excluding aerial releases, off-site transfers, treated and recycled chemicals) by its toxicity (reported by the Indiana Clean Manufacturing Technology and Safe Materials Institute (ICMTSM) in its Indiana Relative Chemical Hazard Score (IRCHS): <https://engineering.purdue.edu/CMTI/IRCHS/>) to produce a weighted amount of inorganic pollution release from each source, and summed all values within each watershed. For those chemical compounds not listed in the IRCHS database, we applied the average score from the class of chemicals to which the missing chemical. Impervious surface area (ISA) data were processed as in the global project (Halpern et al. 2008), using the global impervious surface area data layer developed by the U.S. National Geophysical Data Center for the years 2000-2001 (http://www.ngdc.noaa.gov/dmsp/download_global_isa.html) as a proxy measure for the use and input of inorganic pollutants. The %-coverage of impervious area in each 1km² pixel was identified, and the average %-coverage for all 1km² pixels within a watershed is multiplied by the number of pixels to produce a total area (km²) of impervious surface within each watershed. Point source and ISA estimates of inorganic pollution in each watershed were then log-transformed and normalized (described below) separately, and then the two layers were summed and re-normalized to create a single inorganic pollution value for each watershed. These values were then assigned to the pour-point for each watershed.

Effects: While all pollutants can become toxic at high enough levels, there are a number of compounds that are toxic even at relatively low levels (Johnson et al. 2008). The US Environmental Protection Agency (US EPA) has identified and designated more than 126 analytes as “priority pollutants.” According to the US EPA, “priority pollutants” of particular concern for aquatic systems include: (1) dichlorodiphenyl trichloroethane (DDT) and its metabolites; (2) chlorinated pesticides other than DDT (e.g., chlordane and dieldrin); (3) polychlorinated biphenyl (PCB) congeners; (4) metals (e.g., cadmium, copper, chromium, lead, mercury); (5) polycyclic aromatic hydrocarbons (PAHs); (6) dissolved gases (e.g., chlorine and ammonium); (7) anions (e.g., cyanides, fluorides, and sulfides); and (8) acids and alkalis. While acute exposure to these substances produce adverse effects of aquatic biota and habitats, chronic exposure to low concentrations probably is a more significant issue for fish population structure and may result in multiple substances acting in “an additive, synergistic or antagonistic manner” that may render impacts relatively difficult to discern (Johnson et al. 2008).

Coastal/estuarine pollution can affect any life stage of fish, but fish can be particularly sensitive to toxic contaminants during the first year of life. Effects of pollutants on reproduction, recruitment, behavior, and survival may be particularly critical; e.g., survival may be reduced by inherited and dietary contaminants such as PCBs; reproductive rate may be a more sensitive parameter than survival.

The negative impacts of pollution on commercial fish stocks have generally not been demonstrated, largely due to the fact that only drastic changes in marine ecosystems are detectable and the difficulty in distinguishing pollution induced changes from those due to other causes (Sinderman 1994). Normally chronic and sublethal changes take place very slowly and it is impossible to separate natural fluctuations from anthropogenically caused ones. Furthermore, fish populations themselves are estimated only imprecisely, so the ability to detect and partition contaminant effects is made even more difficult.

SENSITIVITY SCORES

Mortality: 3 (juvenile forms of all species); 2 (adult forms of all species); Scoring based on assumption that most fishes are particularly sensitive to contaminants/pollution during their early life history.

Behavior: 2 (juvenile and adult forms of all species). Response behavior to inorganic pollution does not change (i.e., no avoidance) susceptibility to the toxic effects of these pollutants.

Physiology: 2 (juvenile forms of all species); 3 (adult forms of all species). Body size, age, feeding ecology, and trophic position are some of the most important factors determining bioaccumulation in marine fishes. Most species have no apparent physiological response (i.e. they do not metabolize these pollutants to remove them) that would reduce sensitivity to this threat.

LIGHT POLLUTION

Threat data layer description, from Halpern et al. (2009): Species that use coastal habitats can be impacted by noise and light pollution that emerges from coastal human populations. To estimate the distribution of this stressor, we used the stable lights at night database (http://www.ngdc.noaa.gov/dmsp/global_composites_v2.html) and isolated the light coming from coastal land area (that can be seen in ocean pixels) and offshore oil rigs (both sources of light do not move from night to night and so can be isolated, which NGDC has already processed). The files are cloud-free composites made using all the available archived DMSP-OLS smooth resolution data for 2003.

Effects: Ecological light pollution has demonstrable effects on the behavioral and population ecology of organisms in natural settings. As a whole, these effects derive from changes in orientation, disorientation, or misorientation, and attraction or repulsion from the altered light environment, which in turn may affect foraging, reproduction, migration, and communication. (Longcore and Rich 2004). Juvenile sablefish exposed to a horizontal light gradient exhibited an avoidance of bright light (Sogard and Olla 1998). While juvenile sablefish were primarily surface-oriented, they nonetheless displayed clear day/night differences in vertical distribution. Proximity to the surface and low activity at night contrasted with higher activity and the greater range of vertical movement that typified daytime behavior. Movement throughout the water column during the day and the negative phototaxis observed in a horizontal gradient suggests that juveniles in nature, at least during the day, may not be restricted to the neuston.

SENSITIVITY SCORES

Mortality: 1 (juvenile and adult forms of all species); Light pollution is generally not considered a stressor leading to the indirect/direct mortality of any of these species.

Behavior: 2 (juvenile and adult forms of bocaccio, canary rockfish, and hake); 1 (juvenile and adult forms of sablefish). Light pollution may cause some behavioral changes, such as avoidance, predator interactions, or vertical migration. Species like sablefish, which exhibit negative phototactic responses to artificial light, show a behavioral response that would reduce their sensitivity to this threat.

Physiology: 2 (juvenile and adult forms of all species). Light pollution causes minor to little physiological changes in fishes.

NUTRIENT INPUT

Nutrient Threat data layer description, from Halpern et al. (2009) : Nutrient input (considering nitrogen only here) comes primarily from three sources: farming (fertilizer application and animal farm runoff), sewage, and atmospheric deposition. Because sewage input is generally very difficult to document across larger scales, only nitrogen input from farming and atmospheric deposition was quantified. County-level fertilizer application data come from the USGS (source: "Vulnerability of Shallow Groundwater and Drinking-water Wells to Nitrate in the United States" by Bernard T. Nolan and Kerie J. Hitt) and report average annual nitrogen input from 1992-2001 in kgs/hectare. Confined manure (primarily from dairy farms) is from the same source and reported in the same units, but for the years 1992-1997. Atmospheric wet deposition of pollutants is recorded at over 100 stations within the U.S. as part of the National Atmospheric Deposition Program (<http://nadp.sws.uiuc.edu/>); data from the 19 stations along the west coast and in the Aleutian Islands was used along with spatially kriged values between the stations over the landscape and onto the waters of the California Current (including Baja), measured in kgs/yr/km².

Effects: While much of the excess nutrients within coastal waters originates from sewage treatment plants, nonpoint sources of nutrients from municipal and agricultural run-off, contaminated groundwater and sediments, septic systems, wildlife feces, and atmospheric deposition from industry and automobile emissions contribute significantly (Johnson et al. 2008). Failing septic systems contribute to non-point source pollution and are a negative consequence of urban development. The US EPA estimates that 10- 25%

of all individual septic systems are failing at any one time, introducing feces, detergents, endocrine disruptors, and chlorine into the environment. Sewage waste contains significant amounts of organic matter that cause a biochemical oxygen demand, leading to eutrophication of coastal waters.

Severely eutrophic conditions may adversely affect aquatic systems in a number of ways, including: reductions in submerged aquatic vegetation (SAV) through reduced light transmittance, epiphytic growth, and increased disease susceptibility; mass mortality of fish and invertebrates through poor water quality; and alterations in long-term natural community dynamics.

SENSITIVITY SCORES

Mortality: 2 (juvenile forms of all species); 1 (adult forms of all species). Scoring is based on assumption that fish are particularly sensitive (mortality) from eutrophic conditions / hypoxia early in their life history.

Behavior: 1 (juvenile and adult forms of all species). Avoidance response behavior to hypoxia from nutrient pollution likely increases population fitness. (Bell and Eggleston 2005)

Physiology: 2 (juvenile and adult forms of all species). Juvenile and adult fish are both sensitive to physiological effects of hypoxia that is often associated with nutrient loading in aquatic habitats. (Sinderman 1995)

OCEAN-BASED POLLUTION

Threat data layer description, from Halpern et al. (2009) : Ocean-based pollution is assumed to derive from two primary sources, commercial shipping and ports, as was done in the global project (Halpern et al. 2008). We used the shipping data described above in combination with port volume data derived largely de novo for the California Current. In all cases we used data for, or projected to, the year 2003 as this was when the largest amount of data was available. Commercial port tonnage and location data for US ports came from the US Army Corps of Engineers Navigation Data Center: <http://www.iwr.usace.army.mil/ndc/wcsc/portname03.htm>. Commercial port location data for ports in Mexico or Canada came from the Princeton University Library Digital Map and Geospatial Information Center: <http://www.princeton.edu/~geolib/gis/index.html>, with tonnage for Canadian ports from Transport Canada (http://www.tc.gc.ca/pol/en/report/anre2005/8F_e.htm) and tonnage for Mexican ports from the global project (Halpern et al. 2008). Non-commercial ports and their modeled ship traffic (measured in tonnage, but related to port facilities; see (Halpern et al. 2008)) were included from the global project. All port layers were then combined into a single layer, and this layer (log-transformed and normalized) and the shipping layer were combined and then renormalized to create a single pollution layer.

Effects: Marine trash may be ingested by some fish species, resulting in mortality, although this is most prominently reflected in the bird and sea turtle literature (Derraik 2002). The behavioral effects of marine trash or debris may be to concentrate fish both at the water's surface (FAD – floating aggregation devices) and on the bottom (artificial reefs).

Sensitivity scores

Mortality: 2 (juvenile and adult forms of all species). Scored as if solid trash/debris from commercial operations. Most likely effects of solid trash would be from ingestion or entanglement, but there are few good examples in the literature for fishes.

Behavior: 2 (juvenile and adult forms of bocaccio and canary rockfish); 2 (juvenile and adult forms of hake and sablefish). Most likely behavioral effects of solid trash would be attraction to sunken or floating debris by structure-associated species, such as rockfish, and avoidance by structure-averse species. Other species/forms would not change their behavior.

Physiology: 2 (juvenile and adult forms of all species). Scored as if solid trash/debris from commercial operations. No physiological response that would affect sensitivity to marine trash.

OFFSHORE OIL ACTIVITIES

Threat data layer description, from Halpern et al. (2009): Offshore oil rigs in the California Current are exclusively found in southern California. We obtained location information for these rigs using the same methods as described in the global project (Halpern et al. 2008), producing a total of 27 oil rigs. These locations were confirmed with the data from the California MLPA (<http://marinemap.org/mlpa>).

Effects: The environmental risks posed by offshore exploration and production are well known. They include the loss of hydrocarbons to the environment, smothering of benthos, sediment anoxia, destruction of benthic habitat, and the use of explosives (Macdonald et al. 2002). Petroleum exploration involves seismic testing, drilling sediment cores, and test wells in order to locate potential oil and gas deposits (Johnson et al. 2008). Petroleum production includes the drilling and extraction of oil and gas from known reserves. Oil and gas rigs are placed on the seabed and as oil is extracted from the reservoirs, it is transported directly into pipelines. While rare, in cases where the distance to shore is too great for transport via pipelines, oil is transferred to underwater storage tanks. From these storage tanks, oil is transported to shore via tanker. According to the MMS, there are 21,000 miles of pipeline on the United States OCS. According to the National Research Council (NRC), pipeline spills account for approximately 1,900 tonnes per year of petroleum into US OCS waters, primarily in the central and western Gulf of Mexico. Other potential negative impacts include: physical damage to existing benthic habitats within the “drop zone”, undesired changes in marine food webs, facilitation of the spread of invasive species, and release of contaminants as rigs corrode (Macreadie et al. 2011).

However, the effects of oil rigs on fish stocks is less conclusive, with these risks balanced out by the possible enhanced productivity brought about by colonization of novel habitats by structure-associated fishes and invertebrates (e.g., rockfish, encrusting organisms, etc.) (Love et al. 2006). Decommissioned rigs could enhance biological productivity, improve ecological connectivity, and facilitate conservation/restoration of deep-sea benthos (e.g. cold-water corals) by restricting access to fishing trawlers. Preliminary evidence indicates that decommissioned rigs in shallower waters can also help rebuild declining fish stocks. Petroleum extraction and transportation can lead to a conversion and loss of habitat in a number of other ways. Activities such as vessel anchoring, platform or artificial island construction, pipeline laying, dredging, and pipeline burial can alter bottom habitat by altering substrates used for feeding or shelter. Disturbances to the associated epifaunal communities, which may provide feeding or shelter habitat, can also result. The installation of pipelines associated with petroleum transportation can have direct and indirect impacts on offshore, nearshore, estuarine, wetland, beach, and rocky shore coastal zone habitats. The destruction of benthic organisms and habitat can occur through the installation of pipelines on the sea floor (Gowen 1978). Benthic organisms, especially prey species, may recolonize disturbed areas, but this may not occur if the composition of the substrate is drastically changed or if facilities are left in place after production ends. (Johnson et al. 2008).

SENSITIVITY SCORES

Mortality: 1 (juvenile and adult forms of all species). Effects of oil rigs would be primarily based on direct impacts, novel structure, noise, and addition of potential pollutants. Effects would more likely be behavioral than mortality-based effects (Macreadie et al. 2011)

Behavior: 3 (juvenile and adult forms of bocaccio and canary rockfish); 2 (juvenile and adult forms of hake, juvenile form of sablefish); 1 (adult form of sablefish). Mixed effects, depending on species and location, but more likely behavioral than mortality-based effects (Macreadie et al. 2011); more likely beneficial to rockfish, which are associated with structures.

Physiology: 2 (juvenile and adult forms of all species). Mixed effects, depending on species and location, but more likely behavioral than mortality-based effects (Macreadie et al. 2011). There is no physiological response that would enhance or reduce the sensitivity of these species to oil rigs.

ORGANIC POLLUTION

Organic pollution Threat data layer description, from Halpern et al. (2009) (Halpern et al. 2009a):

Dasymetric mapping techniques (Halpern et al. 2008) were used to estimate input rates based on national level statistics and land-use categories. Land cover data came from the U.S. Geologic Survey (<http://edcns17.cr.usgs.gov/glcc/>) for the US and Baja and from the National Atlas of Canada (<http://atlas.nrcan.gc.ca/site/english/index.html>) for those watersheds. Pesticide use statistics were reported for the US by the National Center for Food and Agricultural Policy 1997 Summary Report and by Environment Canada's Survey of Pesticide Sales and Use in British Columbia for the year 1999. These values were then distributed onto the landscape using dasymetric mapping techniques to get annual pesticide use per km². Values for Baja, Mexico were taken from the global project (Halpern et al. 2008). Data were also available at the county level within the State of California, and so we reran the dasymetric mapping for California using these county data and then compared the output to that from the national level data to test the accuracy of the broader model.

Effects: [in addition to the general pollution effects described under inorganic pollution, above. Much of the following is taken from Johnson et al. (2008)].

Pesticides - There are three basic ways that pesticides can adversely affect the health and productivity of fisheries: (1) direct toxicological impact on the health or performance of exposed fish; (2) indirect impairment of the productivity of aquatic ecosystems; and (3) loss or degradation of habitat (e.g., aquatic vegetation) that provides physical shelter for fish and invertebrates (Johnson et al. 2008). For many marine organisms, the majority of effects from pesticide exposures are sublethal, meaning that the exposure does not directly lead to the mortality of individuals. Sublethal effects can be of concern, as they impair the physiological or behavioral performance of individual animals in ways that decrease their growth or survival, alter migratory behavior, or reduce reproductive success. Early development and growth of organisms involve important physiological processes and include the endocrine, immune, nervous, and reproductive systems. Many pesticides have been shown to impair one or more of these physiological processes in fish. For example, evidence has shown that DDT and its chief metabolic by-product, dichlorodiphenyl dichloroethylene (DDE), can act as estrogenic compounds, either by mimicking estrogen or by inhibiting androgen effectiveness. DDT has been shown to cause deformities in winter flounder eggs and Atlantic cod embryos and larvae. Generally, however, the sublethal impacts of pesticides on fish health are poorly understood. The direct and indirect effects that pesticides have on fish and other aquatic organisms can be a key factor in determining the impacts on the structure and function of ecosystems. This factor includes impacts on primary producers and aquatic microorganisms, as well as macroinvertebrates that are prey species for fish. Because pesticides are specifically designed to kill insects, it is not surprising that these chemicals are relatively toxic to insects and crustaceans that inhabit river systems and estuaries.

PAH - Petroleum products, including polycyclic aromatic hydrocarbons (PAH), consist of thousands of chemical compounds which can be particularly damaging to marine biota because of their extreme toxicity, rapid uptake, and persistence in the environment (Johnson et al. 2008). PAH have been found to be significantly higher in urbanized watersheds when compared to nonurbanized watersheds. Low-level chronic exposure to petroleum components and byproducts (i.e., polycyclic aromatic hydrocarbons [PAH]) have been shown in Atlantic salmon (*Salmo salar*) to increase embryo mortality, reduce growth, and lower the return rates of adults returning to natal streams. As spilled petroleum products become weathered, the aromatic fraction of oil is dominated by PAH as the lighter aromatic components evaporate into the atmosphere or are degraded. Because of its low solubility in water, PAH concentrations probably contribute little to acute toxicity; however, lipophilic PAH (those likely to be bonded to fat compounds) may cause physiological injury if they accumulate in tissues after exposure. Even concentrations of oil that are diluted

sufficiently to not cause acute impacts in marine organisms may alter certain behavior or physiological patterns. Sublethal effects that may occur with exposure to PAH include impairment of feeding mechanisms for benthic fish and shellfish, growth and development rates, energetics, reproductive output, juvenile recruitment rates, increased susceptibility to disease and other histopathic disorders, and physical abnormalities in fish larvae. Effects of exposure to PAH in benthic species of fish include liver lesions, inhibited gonadal growth, inhibited spawning, reduced egg viability and reduced growth. Toxicity responses to winter flounder (*Pseudopleuronectes americanus*) exposed to PAH and other petroleum-derived contaminants, include: liver and spleen diseases, immunosuppression responses, tissue necrosis, altered blood chemistry, gill tissue clubbing, mucus hypersecretion, altered sex hormone levels, and altered reproductive impairments. For Atlantic cod (*Gadus morhua*) exposed to various petroleum products, responses included reduced growth rates, gill hyperplasia, increased skin pigmentation, hypertrophy of gall bladder, liver disease, delayed spermatogenesis, retarded gonadal development and other reproductive impairments, skin lesions, and higher parasitic infections. Effects from exposure of aquatic organisms to PAH include: carcinogenesis, phototoxicity, immunotoxicity, and disturbance of hormone regulation. Fuel, oil, and some hydraulic fluids contain PAH which can cause acute and chronic toxicity in marine organisms, and toxic effects of exposure to PAH have been identified in adult finfish at concentrations of 5-50 ppm and the larvae of aquatic species at concentrations of 0.1-1.0 ppm (Logan 2007). Observed effects of fish exposed to PAH include decrease in growth, cardiac disfunction, lesions and tumors of the skin and liver, cataracts, damage to immune systems, estrogenic effects, bioaccumulation, bioconcentration, trophic transfer, and biochemical changes. PAHs can be toxic to meroplankton, ichthyoplankton, and other pelagic life stages exposed to them in the water column. Short-term impacts include interference with the reproduction, development, growth, and behavior (e.g., spawning, feeding) of fishes, especially early life-history stages. Although oil is toxic to all marine organisms at high concentrations, certain species are more sensitive than others. In general, the early life stages (eggs and larvae) are most sensitive, juveniles are less sensitive, and adults least so.

There are no rockfish-specific PCB threshold data available to determine whether observed concentrations are likely to adversely affect rockfish health (West et al. 2001).

SENSITIVITY SCORES

Mortality: 3 (juvenile forms of all species); 2 (adult forms of all species); Sensitivity scores reflect that most fish species are particularly sensitive to contaminants/pollution during their early life history.

Behavior: 2 (juvenile and adult forms of all species). Response behavior to organic pollution does not change (i.e., no avoidance) sensitivity.

Physiology: 3 (adult forms of all species); 2 (juvenile forms of all species). Body size, age, feeding ecology, and trophic position are some of the most important factors determining bioaccumulation in marine fishes. Most species have no apparent physiological response (i.e. they do not metabolize these pollutants to remove them) that would reduce sensitivity to this threat.

COASTAL SEAWATER EXCHANGE

Threat data layer description, from Halpern et al. (2009) : We mapped the location of all coastal power plants that lie on the coastline from the Platts database (<http://www.platts.com/Analytic%20Solutions/Custom/gis/index.xml>), and applied a 3km buffer around these power plants as an estimate of the scale of their impact. We found 5 plants in Puget Sound, 1 in Oregon, and 17 in central and Southern California.

Effects: Coastal power plants (and deslination plants) draw in huge amounts of marine water for cooling purposes, creating an area around the intake pipes where larvae and small plants are entrained. These entrainment 'plumes' will vary in size and shape depending on ocean currents and the size of the power plant. The construction and operation of water intake and discharge facilities can have a wide range of

physical effects on the aquatic environment including changes in the substrate and sediments, water quality and quantity, habitat quality, and hydrology. Most facilities that use water depend upon freshwater or water with very low salinity for their needs (Johnson et al. 2008).

The entrainment and impingement of fish and invertebrates in power plant and other water intake structures have immediate as well as future impacts to estuarine and marine ecosystems (Johnson et al. 2008). Not only is fish and invertebrate biomass removed from the aquatic system, but the biomass that would have been produced in the future would not become available to the ecosystem. Water intake structures, such as power plants and industrial facilities, are a source of mortality for managed-fishery species and play a role as one of the factors driving changes in species abundance over time. Organisms that are too large to pass through in-plant screening devices become stuck or impinged against the screening device or remain in the forebay sections of the system until they are removed by other means.

SENSITIVITY SCORES

Mortality: 3 (juvenile forms of all species); 1 (adult forms of all species); Mortality effects would be most significant for larval or juvenile life history stages entrained in cooling system intakes.

Behavior: 3 (juvenile forms of bocaccio and canary); 2 (juvenile forms of hake and sablefish, adult forms of all species). Behavioral effects would primarily be observed in nocturnally active species that are attracted to structure or discharge plumes that increase local ocean temperatures.

Physiology: 2 (juvenile and adult forms of all species. Behavioral effects would primarily be reflected in discharge plumes that affect local ocean temperatures.

SEDIMENT DECREASE

Threat data layer description, from Halpern et al. (2009): See Sediment increase, above.

Effects: Changes in sediment regimes can affect marine ecosystems due to decreases in sediment input (largely resulting from river damming). Dams affect the physical integrity of watersheds by fragmenting the lengths of rivers, changing their hydrologic characteristics, and altering their sediment regimes by trapping most of the sediment entering the reservoirs and disrupting the sediment budget of the downstream landscape (Heinz Center 2002) (Johnson et al. 2008). Because water released from dams is relatively free of sediment, downstream reaches of rivers may be altered by increased particle size, erosion, channel shrinkage, and deactivation of floodplains (Heinz Center 2000). The consequence of reduced sediment also extends to long stretches of coastline where the erosive effect of waves is no longer sustained by sediment inputs from rivers (World Commission on Dams, 2000).

The effects to fishes of a reduced sediment regime would be indirect and primarily experienced through the long-term loss of soft-bottom habitat features and coastal landforms and/or changes to benthic habitat composition.

SENSITIVITY SCORES

Mortality: 1 (juvenile and adult forms of all species). Sediment decreases are unlikely to result in any mortality to these marine species; if there is any response, it would likely be behavioral in nature.

Behavior: 2 (juvenile and adult forms of hake, juvenile form of sablefish); 1 (adult forms of all species except hake). There are few behavioral responses that would increase sensitivity to sediment decreases, although water column species that rely on low water clarity for predation refuge may avoid these areas. This "threat" may actually open up new habitat to hard substrate site-attached species and they may move to these new habitats.

Physiology: 2 (juvenile and adult forms of all species). There is no apparent physiological response that would reduce or increase sensitivity to this threat.

SEDIMENT INCREASE

Threat data layer description, from Halpern et al. (2009) : We modeled changes in sediment regimes for all watersheds feeding in to the California Current using a 5-step process. First, we created a new, very high resolution watershed layer (see above). Second, we used the sediment release model developed by Syvitsky and colleagues (Syvitsky et al. 2003) to model natural levels of sediment runoff from these watersheds without dams in place. This model is based on 4 parameters: maximum relief, latitude, basin area, and temperature, which serves as a proxy for rainfall. Third, to calculate changes in sediment input we placed onto the landscape all moderate-sized or larger dams included in the National Inventory of Dams produced by the Army Corps of Engineers for the year 2005 (<http://www.nationalatlas.gov/index.html>). We focused on dams >50ft high and/or with a capacity >5000 acre-feet (N=809). Fourth, we reran the sediment model on the sub-watersheds to determine how much sediment reached each dam from its own sub-watershed (i.e., excluding upstream sub-watersheds), using average current temperature data from the years 1996-2006 (<http://www.prism.oregonstate.edu>) and the other parameters listed above. Finally, we applied each dam's sediment trapping efficiency rate to its sub-watershed, releasing the appropriate amount of sediment below that dam into the downstream sub-watershed, and continued this process until the sediment reached the coastal pourpoint. This analysis therefore also accounted for changes in sediment runoff from these watersheds due to changing climate (i.e. increases in precipitation correlated with rising temperature). For those watersheds without dams, this process produced a new 'natural' value of sediment input that in almost all cases was higher than the pre-industrial estimates due to climate change increasing local temperatures. Consequently, this process produced two stressor layers, increases in sediment (exclusively those watersheds without dams) and decreases in sediment (mostly watersheds with dams). Where temperature changes increased sediment but dams decreased it, the increase (always the smaller of the two) was subtracted from the decrease to produce a single value for the sub-watersheds and the final watershed pourpoint.

Effects: Changes in sediment regimes can affect marine ecosystems due to increases in sediment input (due to land use practices and climate change that can increase precipitation and runoff). Much of the available data come from bioassays that measure acute responses and required high concentrations of suspended sediments to induce the measured response, usually mortality (Wilber and Clarke 2001). Although anadromous salmonids have received much attention, little is known of behavioral responses of many estuarine fishes to suspended sediment plumes. There is a high degree of species variability in response to sedimentation; reports of "no effect" were made at concentrations as great as 14,000 mg/L for durations of 3 d and more (oyster toadfish and spot) and mortality was observed at a concentration/duration combination of 580 mg/L for 1 d (Atlantic silversides). For both salmonid and estuarine fishes, the egg and larval stages are more sensitive to suspended sediment impacts than are the older life history stages.

SENSITIVITY SCORES

Mortality: 2 (juvenile and adult forms of all species). Increases in suspended sediments could affect predator-prey interactions, whereas increased sediment loads would affect substrate composition; without accounting for loss/burial of predation refuge.

Behavior: 2 (juvenile and adult forms of hake, juvenile form of sablefish); 1 (juvenile and adult forms of bocaccio and canary rockfish; adult form of sablefish). We assume that the long-term effects of sediment increases would be to change the composition of nearshore marine habitats from coarse and rocky substrates to soft sand-mud, thereby inducing behavioral responses (attraction/avoidance) that would reduce sensitivity to this threat by marine species with specific benthic habitat preferences (i.e. rockfishes, Petrale sole, adult sablefish).

Physiology: 2 (juvenile and adult forms of all species). None of these species have apparent physiological responses that would reduce or increase sensitivity to this threat.

SHIPPING ACTIVITY

Threat data layer description, from Halpern et al. (2009) : Data was combined from the global mapping effort (Halpern et al. 2008), clipped to the California Current region, with data on ferry traffic within the region. Ferry routes were digitized, and the ferry schedule data were converted into annual ship traffic data by multiplying the number of daily ferry trips by 260 for weekdays (5 days x 52 weeks) and 104 for weekends, summed for total annual trips, and then applied to the appropriate ferry route.

Effects: Commercial shipping activity can lead to ship strikes of large animals, noise pollution, and a risk of ship groundings or sinkings. Data on effects of commercial shipping on fish suggests most responses are behavioral in nature, and mortality is not a major concern. Recent studies suggest fish are actually attracted to vessels, rather than being repelled by them; fish even appeared to be attracted to noisy commercial vessels, and recorded swimming velocities of fish schools suggest that fish do not become scared by noisy, passing ships (Rostad et al. 2006). Vessel activity in coastal waters is generally proportional to the degree of urbanization and port and harbor development within a particular area (Johnson et al. 2008). Benthic, shoreline, and pelagic habitats may be disturbed or altered by vessel use, resulting in a cascade of cumulative impacts in heavy traffic areas. The severity of boating-induced impacts on coastal habitats may depend on the geomorphology of the impacted area (e.g., water depth, width of channel or tidal creek), the current velocity, the sediment composition, the vegetation type and extent of vegetative cover, as well as the type, intensity, and timing of boat traffic. Recreational boating activity mainly occurs during the warmer months which coincide with increased biological activity in east coast estuaries. Similarly, frequently traveled routes such as those traveled by ferries and other transportation vessels can impact fish spawning, migration, and recruitment behaviors through noise and direct disturbance of the water column. Other common impacts of vessel activities include vessel wake generation, anchor chain and propeller scour, vessel groundings, the introduction of invasive or nonnative species, and the discharge of contaminants and debris.

SENSITIVITY SCORES

Mortality: 1 (juvenile and adult forms of all species). Shipping strikes, groundings, and noise pollution not likely to affect these species in the water column or near the bottom.

Behavior: 2 (juvenile and adult forms of all species). None of these species have behavioral responses that would reduce or increase sensitivity to shipping strikes, groundings, and noise pollution.

Physiology: 2 (juvenile and adult forms of all species). None of these species have apparent physiological responses that would reduce or increase sensitivity to shipping strikes, groundings, and noise pollution.

SPECIES INVASION

Threat data layer description, from Halpern et al. (2009) : The potential impact of invasive species was modeled in the same manner as in the global project (Halpern et al. 2008). Briefly, for each port, the annual tonnage of goods passed through the port (i.e., port volume) was used as a proxy measure for ship traffic and therefore probability of invasive species introduction. Past research has shown this to be a reasonable approach to estimating numbers of invasive species at a location (Carlton and Geller 1993; Drake and Lodge 2004). Port volume data were obtained from the global database (Halpern et al. 2008). These port volume values were then plumed away from each port using a diffusive model and a maximum distance of spread set at 27km for the largest port in the region, Long Beach, California.

Effects: Introductions of nonnative invasive species into marine and estuarine waters are considered a significant threat to the structure and function of natural communities and to living marine resources in the

United States (Carlton 2001; Johnson et al. 2008). The mechanisms behind biological invasions are numerous, but generally include the rapid transport of invaders across natural barriers (e.g. plankton entrained in ship ballast water, organisms contained in packing material (Japanese eelgrass *Zostera japonica*) or fouling on aquaculture shipments, aquarium trade with subsequent release to natural environments). Nonnative species can be released intentionally (i.e., fish stocking and pest control programs) or unintentionally during industrial shipping activities (e.g., ballast water releases), aquaculture operations, recreational boating, biotechnology, or from aquarium discharge.

SENSITIVITY SCORES

Mortality: 3 (juvenile forms of all species). 2 (adult forms of all species). Effects of non-native predators, competitors, prey, and/or habitat structural elements likely lethal for juveniles, sub-lethal for adults.

Behavior: 3 (juvenile and adult forms of all species). Native species are not adapted to behaviorally resist the effect of non-native predators, competitors, prey, and/or habitat-forming species. Behavioral interactions would therefore, likely increase the sensitivity of these marine species to population-wide effects.

Physiology: 2 (juvenile and adult forms of all species). None of these species have physiological responses that would reduce or increase sensitivity to invasive species.

COASTAL TRASH

Threat data layer description, from Halpern et al. (2009) : Good spatial data do not exist for marine debris at sea, but beach clean up efforts provide data for the amount of trash that ends up on (and impacts) intertidal ecosystems. The State of California collects county-level statistics on the amount of trash collected from coastal areas each year as part of the California Coastal Commission Public Education Program (<http://www.coastal.ca.gov/publiced/pendx.html>). We extracted data for the years 2003-2007 and calculated the average amount of trash collected, and then divided this county-level average by the number of coastal pixels per county to obtain the average pounds of trash collected per 1 km² of coastline. Similar data do not exist for Washington, Oregon, or Baja, but we chose to include this layer given its importance and length of the California coastline relative to the region. Intertidal ecosystems in California will have marginally higher cumulative impact scores due to this inclusion.

Effects: Marine debris causes stress to organisms that ingest it mistaking it for food, most notably sea birds, sea turtles, and some sea mammals. Ingestion by some species, resulting in mortality (Derraik 2002). Behavioral effects – may concentrate fish (FAD, Artificial reefs).

SENSITIVITY SCORES

Mortality: 2 (juvenile and adult forms of all species). Coastal trash effects were considered primarily solid trash from land-based sources. The most likely mortality effects of solid trash would be from ingestion (including minute plastic particles) or entanglement, but there are only a few good examples of this in the marine literature.

Behavior: 3 (juvenile and adult forms of bocaccio and canary rockfish); 2 (juvenile and adult forms of hake, juvenile form of sablefish); 1 (adult form of sablefish). Most likely behavioral effects of solid trash would be attraction to sunken or floating debris by structure-associated species, such as rockfish, and avoidance by structure-averse species. Other species/forms would not change their behavior.

Physiology: 2 (juvenile and adult forms of all species). There is no physiological response that would enhance or reduce these species' sensitivity to coastal trash.

CLIMATE CHANGE THREATS

We did not include time series data for these climate change threats, because they are dealt with in more precise detail elsewhere in the IEA process. They were included to provide perspective to the magnitude of other non-fisheries related threats. However, the details of the data for each threat layer are included below as well as the scoring rationale for the Sensitivity scores for each threat.

OCEAN ACIDIFICATION

Data layer description: Data for all three measures of climate change stressors (sea surface temperature anomalies, UV radiance anomalies, and ocean acidification) were taken from global data described elsewhere (Halpern et al. 2008), clipped to the California Current region. Briefly, SST anomalies measure the number of times SST was higher in the most recent five years (2000-2005) relative to the longer term (1985-2005) variance (measured as standard deviation). UV radiation anomalies were calculated in the same manner, but with a shorter range of data comparison (2000-2004 vs. the long term variance 1996-2004). Ocean acidification was modeled as the change in aragonite saturation state from pre-industrial times (1870) to modern times (2000-2009). All data layers were represented at 1km² resolution.

Effects: Increased acidity in oceans is expected to effect calcium carbonate availability in seawater, which would lower the calcification rates in marine organisms (e.g., mollusks and crustaceans, some plankton, hard corals) (IPCC 2007). Alteration of water alkalinity could have severe impacts on primary and secondary production, which have implications at the ecosystem level (Fabry et al. 2008). Increasing atmospheric carbon dioxide concentrations and altered seawater carbonate chemistry could have a range of effects, including physiological changes to marine plankton on the organismal level, changes in ecosystem structure and regulation, and large scale shifts in biogeochemical cycling (Fabry et al. 2008). For example, increased carbon dioxide concentrations are predicted to decrease the carbonate saturation state and cause a reduction in biogenic calcification of corals and some plankton, including coccolithophorids and foraminifera; however, increasing carbon dioxide concentrations could increase the rates of photosynthetic carbon fixation of some calcifying phytoplankton.

Juvenile salmon in weakly acidic freshwater streams do not respond to alarm cues (Leduc et al. 2006). The hatchling stages of some fish species appear fairly sensitive to pH decreases on the order of 0.5 or greater, but high CO₂ tolerance developed within a few days of hatching (Fabry et al. 2008).

SENSITIVITY SCORES

Mortality: 3 (juvenile forms of all species); 2 (adult forms of all species); Theoretically lethal (3) for all life history stages based on effects of ocean acidification on primary and secondary production being manifested at ecosystem level, but scored sublethal (2) for adults based on no specific literature documenting mortality in these species.

Behavior: 3 (juvenile and adult forms of all species). The current body of literature is beginning to suggest that many marine and freshwater fish species display behavioral responses (e.g., attraction to predator smells) that may increase sensitivity to ocean acidification.

Physiology: 2 (juvenile and adult forms of all species). None of these species are known to have physiological responses that would reduce or increase sensitivity to ocean acidification.

SEA SURFACE TEMPERATURE

Data layer description: Data for all three measures of climate change stressors (sea surface temperature anomalies, UV radiance anomalies, and ocean acidification) were taken from global data described elsewhere (Halpern et al. 2008), clipped to the California Current region. Briefly, SST anomalies measure the number of times SST was higher in the most recent five years (2000-2005) relative to the longer term (1985-2005) variance (measured as standard deviation). UV radiation anomalies were calculated in the same manner, but with a shorter range of data comparison (2000-2004 vs. the long term variance 1996-2004). Ocean acidification was modeled as the change in aragonite saturation state from pre-industrial times (1870) to modern times (2000-2009). All data layers were represented at 1km² resolution.

Effects: Temperature affects nearly every aspect of marine environments, from cellular processes to ecosystem function (Johnson et al. 2008). The distribution, abundance, metabolism, survival, growth, reproduction, productivity, and diversity of marine organisms will all be affected by temperature changes. Most marine organisms are able to tolerate a specific temperature range and will become physiologically stressed or die after exposure to temperatures above or below the normal range. At sublethal levels, temperature extremes can effect the growth and metabolism of organisms, as well as behavior and distribution patterns. Reproduction timing and the rates of egg and larval development are dependent upon water temperatures. The reproductive success of some cold water fish species may be reduced if water temperatures rise above the optimum for larval growth (Johnson et al. 2008). Stratification could affect primary and secondary productivity by altering the composition of phytoplankton and zooplankton, thus affecting the growth and survival of fish larvae. However, in warmer ocean areas phytoplankton became less abundant as sea surface temperatures increased further, possibly because warm water blocks nutrient-rich deep water from rising to the upper strata where phytoplankton exist; effects have been implicated as a factor in the decline in North Sea cod stocks. Impacts to the base of the food chain would not only affect fisheries but will impact entire ecosystems. Mountain (2002) predicted a northward shift in the distributional patterns of many species of fish because of increasing water temperatures in the Mid-Atlantic region as a result of climate change.

SENSITIVITY SCORES

Mortality: 2 (juvenile and adult forms of all species). Temperature is lethal (3) for all species at excessive levels and can have cascade of ecosystem effects due to changes in primary and secondary production. However, mortality risk was scored sublethal (2) based on primary responses (e.g., reduced growth, reproduction, etc.) observed in literature reviews for these species.

Behavior: 3 (juvenile and adult forms of bocaccio and canary rockfish); 1 (juvenile and adult forms of hake and sablefish). Most species display some form of behavioral thermoregulation (e.g., range shifts, vertical movement) that reduces their sensitivity to sea surface temperature change; however, some rockfish species have strong habitat preferences that may increase their sensitivity to this threat.

Physiology: 3 (juvenile and adult forms of bocaccio and canary rockfish); 2 (juvenile and adult forms of hake and sablefish). Some species, like rockfish and spiny dogfish, display physiological responses (e.g., energy budgets, growth rates) that increase their sensitivity to sea surface temperature change.

ULTRAVIOLET LIGHT

Data layer description: Data for all three measures of climate change stressors (sea surface temperature anomalies, UV radiance anomalies, and ocean acidification) were taken from global data described

elsewhere (Halpern et al. 2008), clipped to the California Current region. Briefly, SST anomalies measure the number of times SST was higher in the most recent five years (2000-2005) relative to the longer term (1985-2005) variance (measured as standard deviation). UV radiation anomalies were calculated in the same manner, but with a shorter range of data comparison (2000-2004 vs. the long term variance 1996-2004). Ocean acidification was modeled as the change in aragonite saturation state from pre-industrial times (1870) to modern times (2000-2009). All data layers were represented at 1km² resolution.

Effects: The eggs and larvae of many fish are sensitive to UV-B exposure. However, imprecisely defined habitat characteristics and the unknown effect of small increases in UV-B exposure on the naturally high mortality rates of fish larvae are major barriers to a more accurate assessment of effects of ozone depletion on marine fish populations (Hader et al. 2003). Visual predators, including most fish, are necessarily exposed to damaging levels of solar UV radiation. Skin and ocular components can be damaged by UV, but large differences are found between different species. Coral reef fishes can adapt to the UV stress by incorporating UV-absorbing substances, which they acquire through their diet, into their eyes and epidermal slime.

In addition to direct effects, including damage to biological molecules such as DNA and proteins and the generation of reactive oxygen species, photoactivation of organic pollutants and photosensitization may be detrimental (Hader et al. 2003). The damaging effects on eggs and larval stages may be enhanced by polycyclic aromatic hydrocarbons (PAHs) such as retene, which is a pollutant from pulp and paper mills. Solar UV radiation has been shown to induce DNA damage in the eggs and larvae of the Atlantic cod, where larvae were more sensitive than eggs. Artificial UV causes massive apoptosis in larval embryos of Japanese flounders. Use of video taping and measurement of oxygen consumption showed sublethal effects of UV radiation in juvenile rainbow trout. Under worst-case scenarios (60% ozone loss, sunny weather and low water turbulence), solar UV-B eliminated buoyancy and caused mortality within 1 or 2 days. Fish spawning depth strongly correlates with UV exposure. It is not known whether the fish are able to detect and avoid the high UV at shallower depths in the highUV lake or whether this spawning pattern is due simply to differential survival. A similar phenomenon has been observed in bluegill larvae (*Lepomis macrochirus*) in a UV-transparent lake where in 19% of nests the estimated UV-induced mortality of larvae exceeds 25%. Most nests are exposed to relatively low UV levels because they are either located at deeper depths or under overhanging branches (Hader et al. 2003).

SENSITIVITY SCORES

Mortality: 2 (juvenile forms of all species); 1 (adult forms of all species). Evidence of mortality in larvae and eggs, especially when exposed to PAH or other photo-activated chemicals; less obvious, sublethal effects in juveniles; negligible effect on adults.

Behavior: 2 (juvenile forms of bocaccio, canary rockfish, and hake, adult form of hake); 1 (adult forms of bocaccio, and canary rockfish, and sablefish). Deepwater, benthos-associated species/stages and species with negative phototactic response behavior would have reduced sensitivity to this threat (1); all other species/stages would show no apparent behavioral response that would reduce or increase sensitivity to this threat (2).

Physiology: 1 (juvenile and adult forms of all species). Most fish species have physiological responses (i.e., defense mechanisms that prevent or repair UVR damage) that would reduce sensitivity to UV radiation (1).

OTHER POTENTIAL THREATS (NOT USED IN THE CURRENT ANALYSIS – WAITING ON SPATIALLY-EXPLICIT DATA)*

HYPOXIA*

Data layer description: Oxygen data from 2009-2010 Pacific groundfish survey (Keller et al. in prep)

Effects: Demersal fish and benthic invertebrate communities in shallow shelf waters of the California Current were acutely affected by seasonally persistent anoxia and severe hypoxia. In August 2006, surveys along previously monitored (2000 to 2004) transect lines revealed the complete absence of all fish from rocky reefs that normally serve as habitats for diverse rockfish (*Sebastes* species) communities that are of current fishery management concern (Chan et al. 2008). Change in activity such as swimming speed and growth and avoidance of low oxygen conditions by changing the habitat have been observed in the marine environment quite frequently (Ekau et al. 2010). Sablefish, as well as a number of other fish species (e.g., Dover sole) exploit oxygen minimum zones; oxygen interfaces may be important to these species as aggregation sites or predation refugia (Levin 2003).

SENSITIVITY SCORES

Mortality: 2 (juvenile forms of all species, adult forms of all species except sablefish); 1 (adult form of sablefish). Most species, with the exception of adult sablefish, exhibit sublethal effects to hypoxia.

Behavior: 1 (juvenile forms of all species, adult forms of all species except sablefish); 1 (adult form of sablefish). Most species display some form of avoidance/movement behavior that would reduce their sensitivity to hypoxic zones (1); however, some species like rockfish may have strong habitat preference behavior that limit their mobility and may increase their sensitivity this threat (?).

Physiology: 2 (juvenile forms of all species, adult forms of all species except sablefish); 1 (adult form of sablefish). Sablefish have a physiological response that decreases their sensitivity to hypoxic zones (1); all other species do not show a response that would enhance or reduce their sensitivity (2).

HARMFUL ALGAL BLOOMS*

Data layer description: none

Effects: Mortality via direct or indirect exposure; species effect varies based on location in water column, species, mechanism, etc. (Landsberg 2002). There are few specific examples in literature that address effects on these four species, however.

SENSITIVITY SCORES

Mortality: 2 (juvenile and adult forms of all species). Theoretically lethal (3), but scored sublethal (2) based on no specific literature documenting mortality in these species.

Behavior: 2 (juvenile and adult forms of hake and sablefish); 1 (juvenile and adult forms of bocaccio and canary rockfish). Most species show no behavioral response that either reduces or increases their sensitivity to HABs. However, demersal species (e.g., rockfish, sole) have reduced sensitivity to this threat (1) due to their association with bottom habitats.

Physiology: 2 (juvenile and adult forms of all species). These species do not show a physiological response that would enhance or reduce their sensitivity to HABs (2).

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MANAGEMENT TESTING AND SCENARIOS IN THE CALIFORNIA CURRENT

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TABLE OF CONTENTS

MANAGEMENT TESTING AND SCENARIOS SECTION

| | |
|---|-----|
| CCIEA scenarios conceptual framework | 736 |
| Rationale and logic of the scenarios | 738 |
| Population Growth scenario | 739 |
| Climate change scenario | 741 |
| Conservation Demands scenario | 743 |
| Energy Crunch scenario | 745 |
| Status Quo | 747 |
| Methodology for Evaluating Scenarios | 748 |
| Scenario Assessment | 751 |
| Human population growth | 752 |
| Climate change scenario and energy crunch scenario | 756 |
| Conservation demand | 758 |
| Status Quo | 761 |
| “Natural” ecosystem components across scenarios | 763 |
| Human well-being across scenarios | 766 |
| Trade-offs among ecosystem components, including human well-being | 772 |
| Synthesis: Lessons Learned | 772 |
| References Cited | 777 |

APPENDICES (MS1 - MS7)

| | |
|--|-----|
| Appendix MS1. Assessing potential conflicts with Wave energy generation along the Oregon coast | 779 |
| Introduction..... | 779 |
| Methods and Data..... | 780 |
| Results..... | 782 |
| Discussion..... | 783 |
| References | 785 |
| Appendix A..... | 787 |
| Appendix MS2. Forecasting the response of Klamath Basin Chinook populations to dam removal and restoration of anadromy versus no action..... | 809 |
| Appendix MS3. Impacts of depleting forage species in the California Current (summary only) | 860 |
| Appendix MS4. Variable impacts of future fisheries development in the California current on ecosystem stability and spatially explicit biomass patterns..... | 861 |
| Introduction..... | 861 |
| Methods | 862 |
| Results..... | 865 |
| Discussion..... | 867 |
| References cited | 869 |
| Appendix A: Finding MSY/fishing scenarios | 883 |
| Appendix MS5. Biological and Economic Effects of Catch Changes Due to the Pacific Coast Groundfish individual quota system | 886 |
| Abstract | 886 |
| Introduction | 886 |
| Methods..... | 889 |
| Results..... | 892 |
| Discussion: A tall order, two steps at a time | 897 |
| References..... | 899 |
| Appendix MS6. Finding the accelerator and brake in an individual quota fishery: Linking ecology, economics, and fleet dynamics of US West Coast trawl fisheries (abstract only) | 901 |
| Appendix MS7. Commercial Fishing Economics Technical Report For the Secretarial Determination on Whether to Remove Four Dams on the Klamath River in California and Oregon | 902 |
| Introduction..... | 907 |
| Existing Fishery Conditions | 908 |

| | |
|---|-----|
| Biological Assumptions..... | 915 |
| Commercial Fishing Economic Value for Benefit-Cost Analysis (NED Account) | 921 |
| Commercial Fishing Expenditures for Regional Economic Impact Analysis (RED Account) | 930 |
| Summary and Conclusions | 937 |
| References | 939 |
| Appendix A. Salmon Fishery Management | 942 |
| Appendix B. Methodologies Used to Quantify Economic Effects of No Action and Alternatives | 950 |

LIST OF TABLES AND FIGURES* (MS)

| | |
|--|-----|
| Figure MS1. Schematic of Management Testing approach, where drivers are linked to pressures via narrative scenarios, and then quantitative models link pressures to responses. | 737 |
| Figure MS2. Results from preliminary engagement with managers and experts (Section 1), related to the Population Growth scenario. Blue topics are addressed with quantitative models in this IEA..... | 739 |
| Figure MS3. Results from preliminary engagement with managers and experts (Section 1), related to the Climate and Global Change scenario. Blue topics are addressed with quantitative models in this IEA..... | 741 |
| Figure MS4. Results from preliminary engagement with managers and experts (Section 1), related to the Conservation Demand scenario. Blue topics are addressed with quantitative models in this IEA..... | 743 |
| Figure MS5. Results from preliminary engagement with managers and experts (Section 1), related to the Energy Crunch scenario. Blue topics are addressed with quantitative models in this IEA. | 745 |
| Figure MS6. Results from preliminary engagement with managers and experts (Section 1), related to the Status Quo scenario. Blue topics are addressed with quantitative models in this IEA..... | 747 |
| Figure MS7. Sites for potential wave energy facilities, power grid connection points, and barge tow lanes. | 753 |
| Figure MS8. Sites for potential wave energy facilities, power grid connection points, and green sturgeon critical habitat. | 754 |
| Figure MS9. Percent of species in California Current Ecosim food web model (solid lines) and Atlantis ecosystem model (dashed lines) that exhibit changes in biomass of > 20% (either positive or negative) when forage groups are depleted below unfished levels. A value of 1.0 on the x-axis represents abundance of the forage group when it is not fished, while a value of 0.4 represents depletion of a focal forage group to 40% of unfished abundance. Focal forage groups are as follows: euphausiids -- green triangles; forage fish -- blue diamonds; mesopelagic fish -- purple crosses; mackerel -- black squares; sardines in Ecosim-- orange circles. Vertical lines of the same colors represent abundance of each forage group that leads to maximum sustainable yield in the two models (only position on the x-axis is relevant, y-position is for graphical clarity only)..... | 755 |
| Figure MS10. Number of functional groups affected by a grenadier fishery at three fishing levels (threshold of 10 percent change) by cell. Fishing scenarios represented are F75 (A, D), F40 (B, E), and F25 (C, F). Density of color indicates increasing number of functional groups affected, as indicated by legend. | 756 |
| Figure MS11. Net revenue for West Coast groundfish fleets over 30 years. Solid lines denote fuel at \$3/gallon, dashed lines at \$6/gallon. This simple metric of net revenue is gross revenue minus fixed costs (excluding capital costs) and variable costs (fuel, ice, and food, but not labor or quota costs). Details as in Appendix MS6, except that annual net revenue calculation includes adjusted variable costs to include \$6 fuel. Colors denote options for the management system: black = cumulative landings limits in place prior to 2011; grey = individual quotas with no lease price and low penalties for exceeding quota; red = individual quotas with higher lease costs and penalties. | 758 |

Table MS1. Percent increase in abundance and harvest due to performing dam removal versus no action, for two time periods: 1) prior to dam removal (2012 – 2019); and after removal of dams and cessation of active reintroduction and production of the Iron Gate Hatchery production (2030-2061). “95% CrI” is 95% credibility interval.....759

Figure MS12. Revenue in fishery sectors, and income effects in the broader West Coast economy. Year 1 predictions. Total income and revenue are represented by bars in millions of dollars (left axis). “Slightly optimistic” scenarios for individual quotas assume moderate increases in target species catch and little change in rockfish bycatch, while “Highly optimistic” scenarios for individual quotas assume large increases in target species catch with little change in rockfish bycatch.....762

Table MS2. Percent of each model polygon closed to groundfish fleets, assuming establishment of three wave energy facilities per cost scenario, with each facility closing fishing in an area 12km N-S and 2km E-W . Each model polygon spans most of the Oregon coast in the N-S direction, and is defined by depth contours indicated in the column headings.762

Table MS2: For 2006-2010, the proportion of each portgroup’s revenue derived from each species or species group. From PacFIN landings database.768

Table MS3: For 2006-2010, the proportion of revenue derived from each species or species group that is landed in each portgroup. From PacFIN landings database.769

*Does not include appendices

SUMMARY OF CONCEPTUAL FRAMEWORK

Scenarios and Management Testing aim to provide a glimpse into alternate futures for the California Current and the implications of alternate management decisions. Here we first develop narrative scenarios that consider how drivers of the system may link to pressures, for instance how human population growth increases conflicts between salmon recovery and human water needs (**Figure MS1**). We then use quantitative models to predict how changes in pressures impact attributes of interest for the IEA, such as particular protected species or human communities. The quantitative analyses are a preliminary test of the capabilities of six distinct modeling frameworks to identify and project future trends for the California Current. The scenarios and management actions that are tested in the quantitative analyses range from nearly certain to highly unlikely, given current legal frameworks and other factors. Nonetheless, the coupled scenarios and modeling analyses illustrate the impacts of both system-level drivers and potential management responses.

DESCRIPTION OF THE CONCEPTUAL FRAMEWORK

Through preliminary engagement with managers, scientists, and stakeholders we have identified potential drivers of the California Current (**Engagement section**). Other efforts within this IEA have identified patterns related to pressures, risk, status, and trends of the ecosystem (**Drivers and Pressures, Risk, and Ecosystem Components sections**). Those analyses are the motivation for Scenarios and Management Testing, which aim to provide a glimpse into alternate futures for the California Current and the implications of alternate management decisions. Scenarios and Management Testing differ from risk assessment, in that we are explicitly interested in projecting forward in time, whereas risk assessment deals with current status. Here we develop narrative scenarios that consider how drivers of the system may link to pressures, for instance how human population growth increases the demand for fresh water for urban and agricultural uses (**Figure MS1**). We then use quantitative models to predict how changes in pressures impact attributes of interest for the IEA, such as particular protected species. Timescales for the quantitative analyses are fifty years into the future or less.

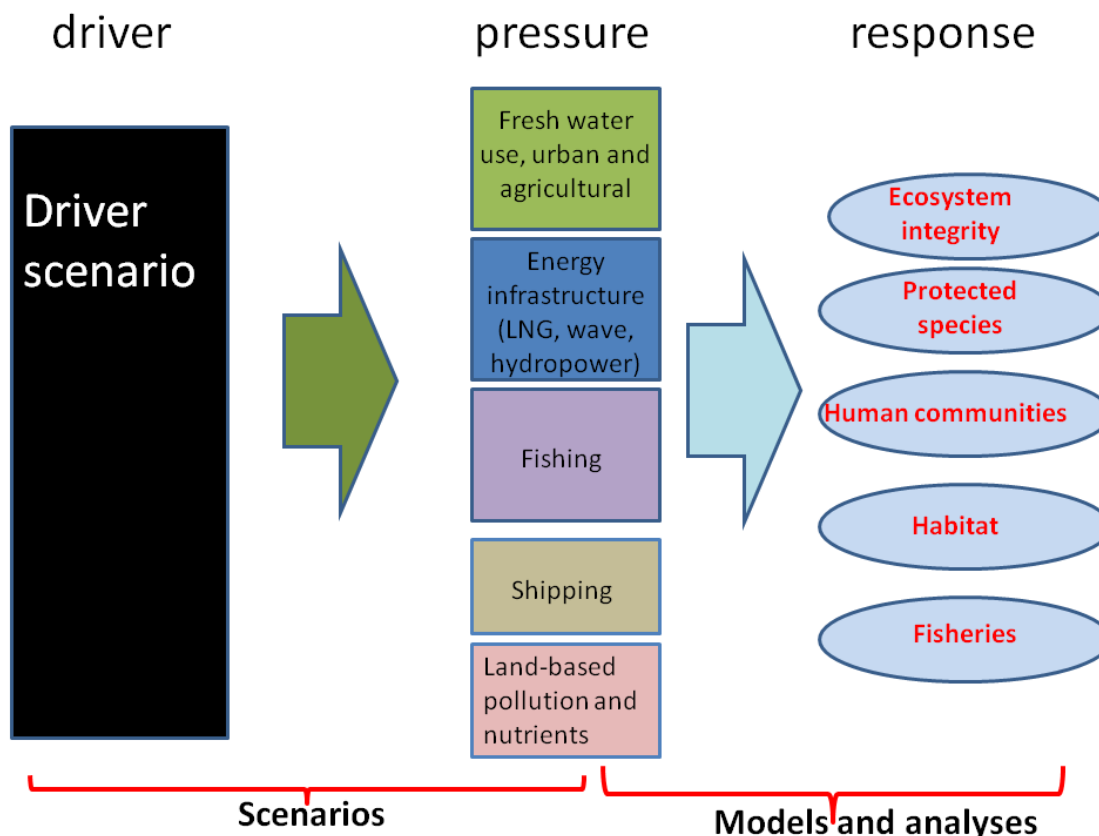


Figure MS1. Schematic of Management Testing approach, where drivers are linked to pressures via narrative scenarios, and then quantitative models link pressures to responses.

Linking from drivers to pressures (**Figure MS1**) falls outside the realm of most quantitative modeling, but can be used to inform such modeling. Scenario planning is one highly effective means of creating sensible and powerful narratives that help stakeholders envision the future, and help modelers specify meaningful measures of pressure on the ecosystem. Scenario planning has been applied to environmental issues for over 40 years (Alcamo 2008). Recently the Millenium Ecosystem Assessment (2005) successfully used scenario development to envision futures for the global environment and human populations. As described in the Millenium Ecosystem Assessment, scenarios are “plausible and often simplified descriptions of how the future may develop based on a coherent and internally consistent set of assumptions about key driving forces and relationships.” Ash et al (2010) note that “an important function of scenario analysis—particularly in the context of ecosystem assessments—is that it provides an approach to reflect on and think through the possible implications of alternative decisions in a structured manner. Simply put, a scenario exercise offers a platform that allows [decision makers] to reflect on how changes in their respective context (that is, developments not within their immediate spheres of influence) may affect their decisions.”

Scenarios are a new tool for marine resource management, but have many parallels with established approaches that are used to account for uncertainty and complex human behavior. One analogous approach from single species management is the decision table framework (Hilborn and Walters 1992) that tests performance against alternate “states of nature”, which typically bracket key uncertainties in biology, data, or fishermen’s behavior. Often these uncertainties are framed in terms of narrative “what if” scenarios posed by

expert review panels. Resource managers are also familiar with scenarios, albeit under a different terminology. For instance, given considerable uncertainty in fishermen’s behavior under a groundfish catch share program, the Pacific Fishery Management Council (2010) envisioned four sets of harvest and bycatch rates based on a blend of expert opinion and data. This approach of considering potential alternative futures is warranted when no reliable quantitative model can address a particular complex human, economic, or ecological challenge.

Though we do not have quantitative models to link all pressures to ecosystem attributes (**Figure MS1**), we can begin to apply and refine a set of relevant tools. Such quantitative tools are already in daily use by NOAA scientists and others, and include single species stock assessments (Methot 2007), GIS mapping, spatial planning tools (Tallis *et al.* 2008), food web models (Steele and Ruzicka 2011), and ecosystem models (Kaplan *et al.* 2012). Other links from pressures to impacted attributes cannot be addressed with the current generation of quantitative models.

RATIONALE AND LOGIC OF THE SCENARIOS

SUMMARY

Drawing from themes raised in our preliminary engagement with managers and other experts (**Engagement section**), we develop narrative scenarios that act as links between drivers and pressures (**Figure MS1**). These are “scenarios for drivers”, essentially “what if” stories about alternate paths that drivers and pressures may take in the future. Scenarios include drivers related to human population growth, climate change, demand for conservation, energy, and evolution of status quo management and responses to it. Scenarios detail potential effects on pressures considered in this IEA: urban and agricultural freshwater use, energy infrastructure, fishing, pollution, and shipping. The table below diagrams the major trends in pressures for each scenario, followed by a more nuanced description. Subsequent sections link selected portions of these narrative scenarios to quantitative models.

Note: The color coding below roughly indicates whether the pressure (shipping, fishing, land-based pollution, energy infrastructure, freshwater use) will **increase**, **decrease**, or **remain** at current level. For the web version of this document, [hyperlinks](#) are provided, linking to quantitative analyses (described below). Text sections lacking hyperlinks have been developed here as narratives, but lack quantitative methodologies for testing these implications of the scenarios.

| Scenario | Pressure | | | | |
|-------------------------|--|-----------------------|---------|----------------------|----------|
| | Freshwater use, urban and agricultural | Energy Infrastructure | Fishing | Land-based pollution | Shipping |
| Human Population Growth | ↑ | ↑ | ↑ | ↑ | ↑ |
| Climate Change | ↑ | ↑ | ↔ | ↔ | ↔ |
| Conservation Demands | ↓ | ↓ | ↓ | ↓ | ↓ |
| Energy Crunch | ↔ | ↑ | ↓ | ↔ | ↑ |
| Status Quo | ↔ | ↔ | ↔ | ↔ | ↔ |

FULL DESCRIPTION OF SCENARIO RATIONALE

Below, we first develop narrative scenarios that act as a link between drivers and pressures (**Figure MS1**). These are “scenarios for drivers”, essentially “what if” stories about alternate paths that drivers and pressures may take in the future. Our aim is to explore divergent paths for the California Current, not to evaluate which is most likely biologically or given legal or political constraints. We consider management actions including some that are illegal under current laws, and drivers that are possible but not necessarily likely. Importantly, not all drivers can be linked logically to each pressure, via narratives that capture our current qualitative understanding of the system. Similarly, not all pressures can be linked to impacts on each attribute, either in a logical or quantitative way. The scenarios focus on impacts related to living marine resources, with some limited consideration of other social and economic impacts. Though preliminary engagement with experts identified the drivers and pressures (**Engagement section**), the narrative scenarios are constructed by the authors.

POPULATION GROWTH SCENARIO

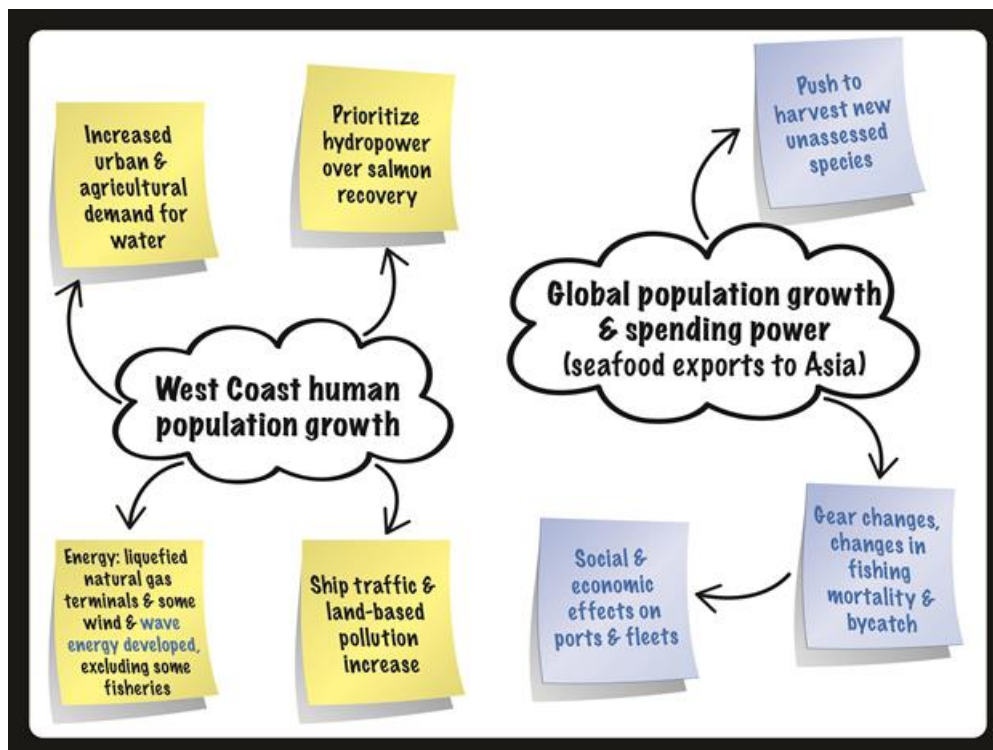


Figure MS2. Results from preliminary engagement with managers and experts (Engagement section), related to the Population Growth scenario. Blue topics are addressed with quantitative models in this IEA.

INSIGHTS FROM EXPERTS

As described in the preliminary engagement with managers and experts (**Engagement section**), human population growth on the US west coast was identified as a driver of freshwater and nearshore habitats, particularly for salmon (**Figure MS2**). Global population growth was identified as a driver of seafood demand, including demand for new species. Using themes and details from these conversations, we constructed the following narrative:

NARRATIVE FOR HUMAN POPULATION GROWTH

FRESHWATER USE, URBAN AND AGRICULTURAL: Urban demands for freshwater will increase concomitantly with the increase in human population on the West Coast. The EPA has defined baseline population growth scenarios that will increase the population of western states by 50% from 2005 to 2060 (Bierwagen 2009). This demand will compete with the needs of salmon, particularly during the summer and for “stream type” stocks (i.e. those that rear for extended periods in freshwater). Desalination plants might be built in Southern California, with local negative impacts on some plankton, fish eggs and larvae.

ENERGY INFRASTRUCTURE: The growing human population requires increased electricity production. Dam removal on major salmon rivers might be politically unviable. [Wave and wind energy installations may be built](#), but most investment focuses on LNG terminals.

FISHING: West Coast population growth does not lead to immediate increases in demand for West Coast wild seafood, primarily due to declines in US per capita seafood consumption and increased aquaculture production and imports. In a variation of this scenario, global increase in population and economic development, particularly in Asia, could drive [substantial increases in demand for West Coast seafood](#), including increased focus on species such as grenadier, crab, octopus, geoduck, and live-caught rockfish.

LAND-BASED POLLUTION: Land-based pollution, including pathogens and nitrogen inputs, is assumed to continue proportional to population growth. No major improvements in sewage or storm-water treatment are envisioned.

SHIPPING: Ship traffic is assumed to continue proportional to population growth. No major changes are envisioned related to ship speeds or shipping lanes.

See population growth graph: www.bit.ly/xZK9pW

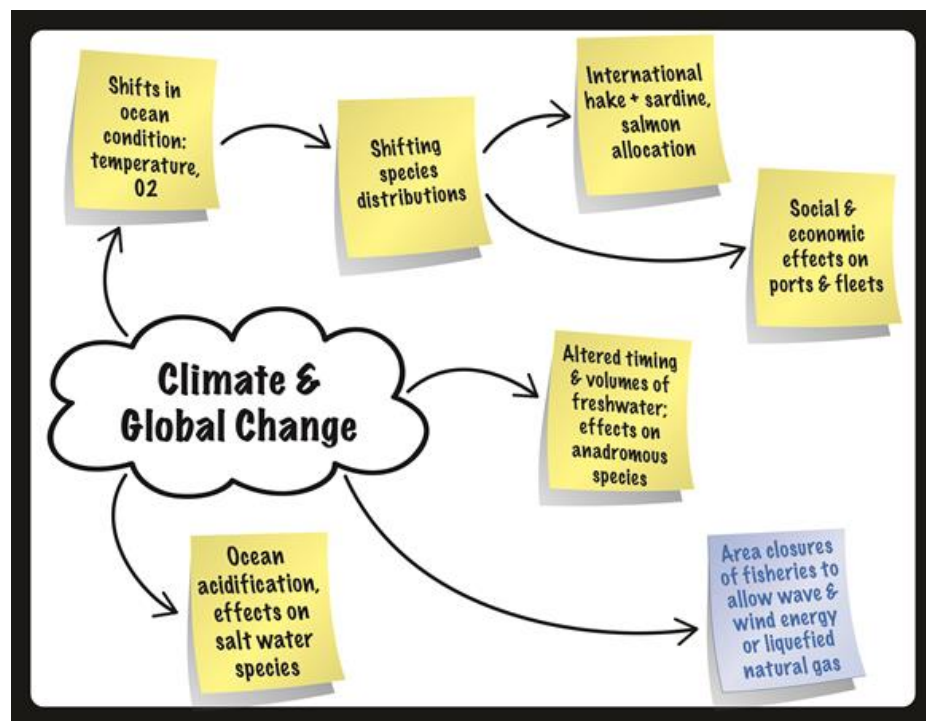


Figure MS3. Results from preliminary engagement with managers and experts (Engagement section), related to the Climate and Global Change scenario. Blue topics are addressed with quantitative models in this IEA.

INSIGHTS FROM EXPERTS

As described in the preliminary engagement with managers and other experts (**Section 1**), climate change and ocean acidification were predicted to impact salmon, sardine, anchovy, and hake (**Figure MS3**). Policy responses were limited but included altering harvest, stream restoration, and community-based management. Using themes and details from these conversations, we constructed the following narrative:

NARRATIVE FOR CLIMATE CHANGE

In the oceans, global warming may lead to a 1.8 - 4°C (3-6°F) increase in sea surface temperature this century. This may cause northward shifts in species ranges and migration patterns, changes in growth and reproductive rates, and reductions in the oxygen content of water (potentially to anoxic levels), particularly in nearshore areas <50m deep. These hypoxic or anoxic areas may lead to local die-offs of crabs or other species with limited mobility. Primary production (phytoplankton) may increase, but smaller phytoplankton may be favored, leading to less food availability for large zooplankton (e.g. krill) but more for smaller zooplankton (e.g. copepods).

Increasing fossil fuel emissions and the resulting increase in atmospheric CO₂ levels will likely lead to a decline in seawater pH of 0.3 by the year 2100. Changes to seawater pH and the saturation state of

aragonite and calcite (the minerals many organisms use to build protective structures) could lead to reduced populations of marine species including corals, crabs, shellfish, benthic invertebrates, and plankton groups such as krill. There is considerable uncertainty regarding which species will be impacted, and to what extent (National Research Council (US) 2010) .

In freshwater, global warming may reduce snowpack in mountain streams and reduced summer flows in mountain streams. Stream temperatures may be elevated in summer. These effects may lead to decreased growth and survival of juvenile salmonids, particularly Chinook salmon.

FRESHWATER USE, URBAN AND AGRICULTURAL: Reduced winter snowpack will change the timing of water demand and releases from reservoirs. Even if overall volume of water use is not changed, there could be more agricultural demand for water during the summer, in competition with some salmon stocks. “Stream type” salmon may be particularly impacted. Dams may be used to store more water during winter, rather than releasing this water for flood control purposes over the course of the winter.

ENERGY INFRASTRUCTURE: Large changes in energy infrastructure may results as a policy response to slow climate change. Low-carbon energy such as LNG, hydropower, or [wave energy may become more popular](#).

FISHING: Species distributions may shift in response to climate. Pelagic or midwater species such as hake or sardine may shift their migrations and distribution northwards. Salmonid stocks in California may decline as salmon range shifts northward. The harvest of fishing fleets (at the port level) may shift as well. [Low-carbon energy sources will exclude fishing fleets](#) from certain areas, as discussed in “Energy Crunch” scenario.

LAND-BASED POLLUTION: Changes in rainfall and river flow may alter runoff of pollutants.

SHIPPING: No direct impact expected

See related graph of yearly CO₂ emissions: www.bit.ly/zdh95M

CONSERVATION DEMANDS SCENARIO

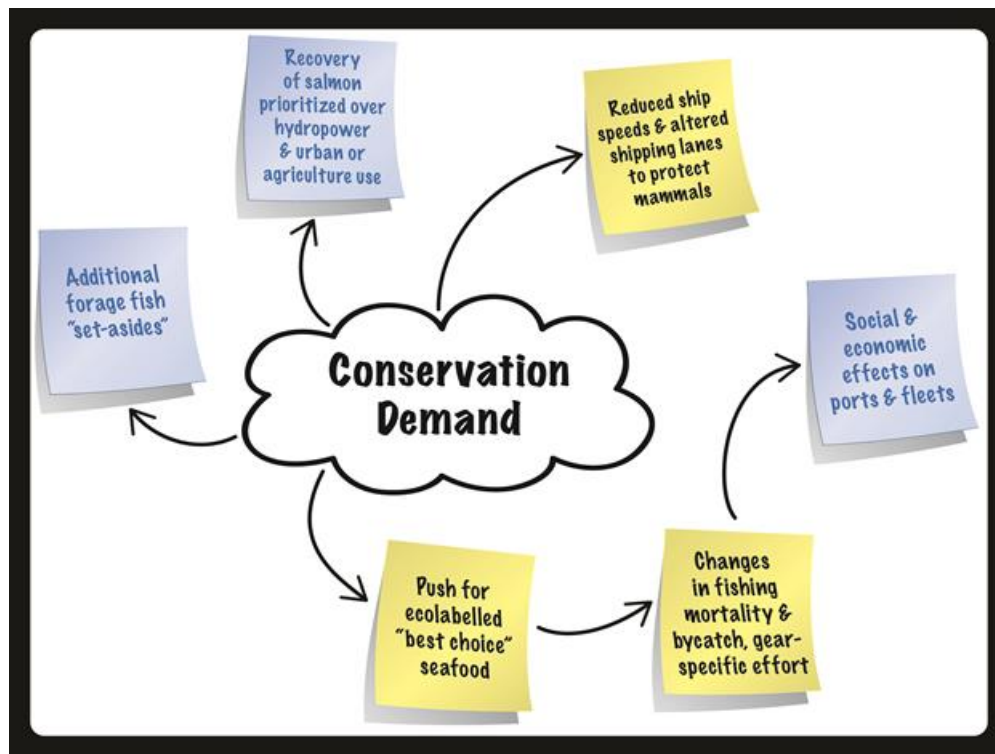


Figure MS4. Results from preliminary engagement with managers and experts (Engagement section), related to the Conservation Demand scenario. Blue topics are addressed with quantitative models in this IEA.

INSIGHTS FROM EXPERTS

As described in the preliminary engagement with managers and other experts (**Section 1**), a growing demand for conservation was envisioned to alter harvest policies, dam operation, shipping, seafood demand, and marine spatial planning (**Figure MS4**). Using themes and details from these conversations, we constructed the following narrative, which might unfold in the next 1-2 decades:

NARRATIVE SCENARIO FOR CONSERVATION DEMANDS

This scenario envisions increased demand from the public, NGOs, and stakeholders for conservation of marine resources. This may be aided by modifications to current federal, state, and tribal policies, or at the federal level by implementation of Marine Spatial Planning and National Ocean Council recommendations. At the state level and smaller scales, increased local input and cooperation between managers and stakeholders could lead to faster management responses and more local solutions and experimentation to achieve conservation goals.

FRESHWATER USE, URBAN AND AGRICULTURAL: Recovery of salmon is promoted, even above current efforts, at times limiting water available for cities and agriculture.

ENERGY INFRASTRUCTURE: Dam removal is attempted to promote recovery of certain salmon stocks. Economic and social costs of removal can be weighed against benefits to salmon stocks.

FISHING: In this scenario, harvest of forage groups (sardine, squid, mackerel) are reduced, to avoid potential negative impacts on their predators. Fishing effort shifts to only stocks that are labeled as eco-certified. A variation on this scenario keeps fishing effort on sardines (often eco-certified as a “best choice”) but avoids other forage groups. Scenario impacts may include reductions in fishing effort or fishing grounds, changes in gear that degrades bottom habitat or entangles mammals, “set-asides” of forage species for predators rather than fishermen, and possible trade-offs between stakeholders (e.g. fishermen vs. tourism) or between certain ports or regions.

SHIPPING : In this scenario, protection of marine mammals is prioritized, resulting in changes to shipping lanes and reduced ship speeds. This results in fewer ships striking mammals, and less disturbance of mammals by vessel traffic.

LAND-BASED POLLUTION: Policies reduce discharge of nitrogen and pathogens in nearshore waters, with some benefits such as reduced harmful algal blooms or reduced mortality of sea otters.

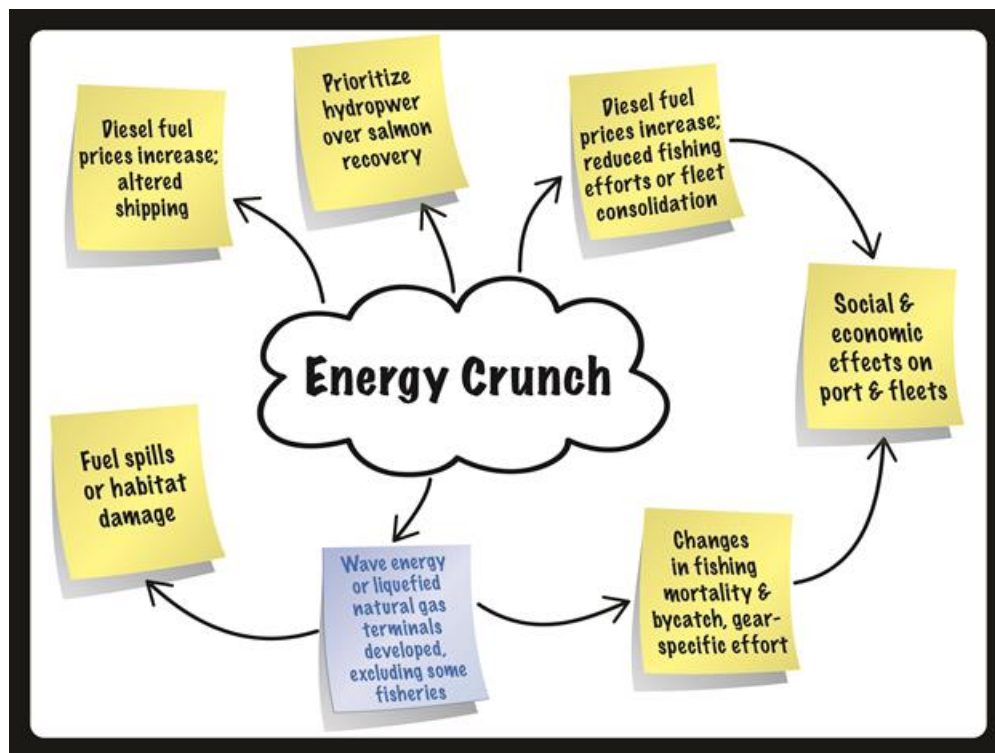


Figure MS5. Results from preliminary engagement with managers and experts (Engagement section), related to the Energy Crunch scenario. Blue topics are addressed with quantitative models in this IEA.

INSIGHTS FROM MANAGERS AND OTHER EXPERTS

As described in the preliminary engagement with managers and experts (**Section 1**), rising demand or price for energy was discussed as a driver of fishing, shipping, and the establishment of wave energy facilities. (**Figure MS5**). Using themes and details from these conversations, we constructed the following narrative, which might unfold over the next thirty years:

NARRATIVE SCENARIOS FOR ENERGY CRUNCH

"By 2015, growth in the production of easily accessible oil and gas will not match the projected rate of demand growth. ... alternative energy sources such as biofuels may become a much more significant part of the energy mix — but there is no "silver bullet" that will completely resolve supply-demand tensions."-- Shell Oil Scenarios

ENERGY INFRASTRUCTURE: The local response to rising energy demand will be to **develop wave farms**, and to exploit fuels such as liquefied natural gas (LNG). Development of LNG terminals and **wave energy installations** may lead to **exclusion of fishing gears** from portions of the coast. Increased ship activity around

these facilities could lead to fuel spills, putting vulnerable habitats or National Marine Sanctuaries at risk. The demand for hydropower will also increase, in competition with the needs of species such as salmon.

FISHING: Rising prices for diesel fuel may reduce fishing effort, cause fleet consolidation, or shift the fishing areas or methods of fleets. Fuel-intensive fleets (e.g. albacore trolling) may reduce effort substantially. This in turn could lead to social and economic impacts that vary by fleet and port. Fishery targeting may shift as profitability changes due to rising fuel costs.

SHIPPING: Shipping traffic may increase as industries push for low-cost methods (freighters, tankers) to move goods. Short-sea shipping, between existing cargo hubs and new satellite ports, may increase ship traffic in coastal areas. Increases in shipping could increase ship strikes of mammals and other vessel-related disturbance, as well as pollution discharges from ships.

LAND-BASED POLLUTION: No changes expected

FRESHWATER USE, URBAN AND AGRICULTURAL: No change expected

See graph of global energy use: www.bit.ly/S4VSfC

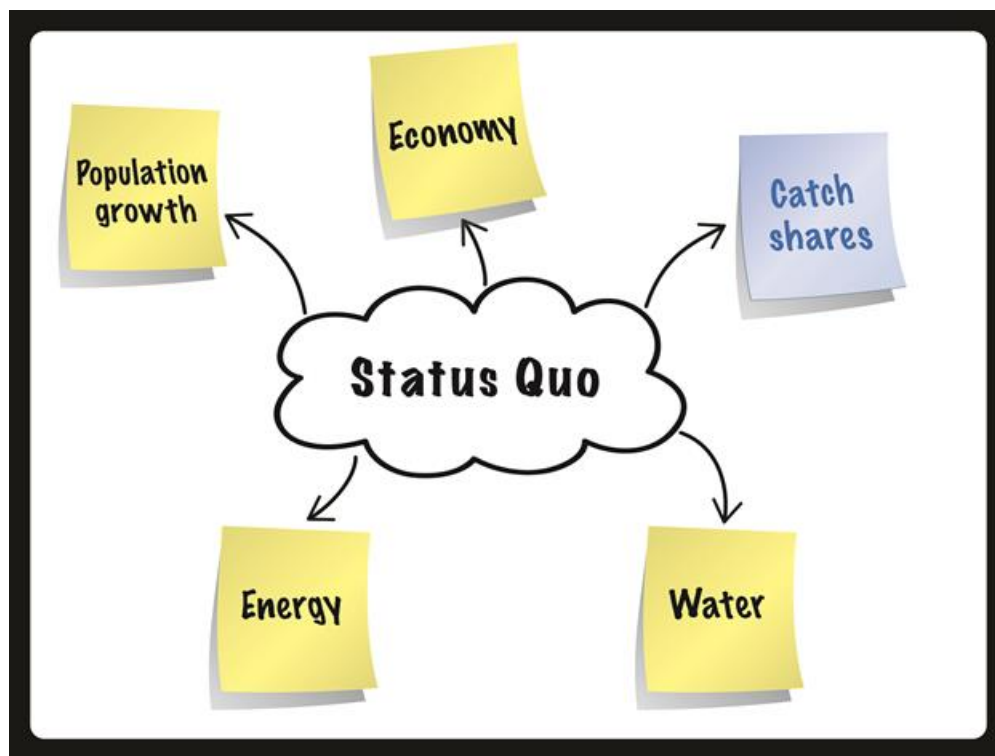


Figure MS6. Results from preliminary engagement with managers and experts (Engagement section), related to the Status Quo scenario. Blue topics are addressed with quantitative models in this IEA.

INSIGHTS FROM EXPERTS

The preliminary engagement with managers, scientists, and other experts (**Section 1**) identified key challenges with status quo fishery management, such as inflexibility, lengthy regulatory review processes, and high costs (**Figure MS6**). Additionally, the groundfish catch share program was initiated in January of 2011, and experts and managers suggested that results from the program would depend on the evolution of fishery targeting, market demand, and fleet consolidation. Using themes and details from these conversations, we constructed the following narrative:

NARRATIVE SCENARIO FOR STATUS QUO

This scenario will project current drivers and pressure on the ecosystem. Note that in some ways 10-20 year projections of this scenario are highly unrealistic if population growth continues. Nevertheless, to understand output from quantitative models, status quo can serve as a baseline that can be compared to more realistic population growth scenarios.

FRESHWATER USE, URBAN AND AGRICULTURAL: No major change in the volume or timing of demand for freshwater.

ENERGY INFRASTRUCTURE: Assume no major expansion of wave or wind energy, LNG, or changes in hydropower infrastructure or operations.

FISHING: Assume current management structure and regulations. Variants of this primarily involve different [responses of fishermen to the existing groundfish catch share](#) system, different [options to promote flexible responses](#), and how this can be altered by fuel prices and climate. This can build on an existing Environmental Impact Statement (Pacific Fishery Management Council 2010), which predicted species-level responses of several groundfish populations to different scenarios for fishermen's behavior under catch shares.

LAND-BASED POLLUTION: Left at current levels.

SHIPPING: Assume current volume of ship traffic, shipping lanes, and ship speeds

METHODOLOGY FOR EVALUATING SCENARIOS

SUMMARY

We evaluate the future system response to some of the potential pressures and management actions discussed in the scenarios. Quantitative modeling approaches include spatial analysis using GIS (geographic information systems), single species models, food web models, ecosystem models, and economic input-output analyses. This diversity of approaches is required to address specific aspects of the scenarios; there is no 'silver bullet' model that handles all pressure, drivers, and management actions.

FULL DESCRIPTION OF METHODS

Given the set of links between drivers and pressures described in the scenario narratives, we apply quantitative modeling tools to translate pressures into predicted effects on ecosystem attributes (**Figure MS1**). We tailor the predictions to species and attributes which are relevant to the IEA and for which models could be developed and applied; not all pressures can be logically or quantitatively linked to each attribute. Given the simplicity of quantitative models available for the 2012 Integrated Ecosystem Assessment, in the narratives below we treat drivers separately from one another, even though more complicated scenario planning exercises (e.g. the Millenium Ecosystem Assessment) typically create complicated scenarios that are bundles of drivers, threats, pressures, human decisions, and ecological states. Our goal is to evaluate the future system response to potential pressures and management actions, informed by consideration of drivers on the system.

Quantitative modeling approaches detailed in [Appendices MS1-MS7](#) range in complexity from spatial analysis using GIS (geographic information systems) up to very detailed modeling of species and fishing fleet dynamics. This diversity of approaches is required to address specific aspects of the scenarios; there is no 'silver bullet' model that handles all pressure, drivers, and management actions.

GIS SPATIAL MODELING

In a first step toward addressing aspects of the **Energy Crunch** scenarios and possible policy responses to **Climate Change**, we use a static, map based approach to consider spatial ramifications of wave energy (**Appendix MS1**). We apply a GIS-based decision-support tool (Marine InVEST, Tallis et al. 2011) to evaluate potential sites for wave energy conversion facilities off the coast of Oregon, and to identify spatial overlap and possible conflicts with other marine uses. Our focus on Oregon is motivated by the availability of data regarding wave energy, power infrastructure, and fishing. The wave energy model consists of three parts: 1) assessment of potential wave power based on wave conditions; 2) quantification of harvestable energy using technology specific information about a wave energy conversion device; and 3) assessment of the economic value of a wave energy conversion facility over its life span as a capital investment. We configure a wave energy facility based on previous work by the Electric Power Research Institute (Previsic, 2004b), which analyzed the system level design, performance, and cost of a commercial size offshore wave power plant installed off the coast of Oregon. Existing marine uses were fishing; transportation and utilities; and marine conservation areas. Spatial fishing effort data for 2002 – 2009 were provided by the At-sea Hake Observer Program and the West Coast Groundfish Observer Program under NOAA's Northwest Fisheries Science Center, Fishery Resource Analysis and Monitoring Division. These data produce a map of different effort levels that can be overlaid with the potential locations of wave energy facilities to reveal possible spatial conflicts. We generated additional maps of possible conflicting uses with the following data. Additional fishing effort maps were provided by Steinback et al. (2010), for several Oregon ports. For transportation, we consider general shipping lanes, and lanes established for tug and barge traffic under on ongoing agreement between tug and barge operators and crab fisherman. For utilities, submarine cable location is identified as recorded on NOAA's Electronic Navigation Charts. Finally, we consider spatial overlap between potential wave energy sites and critical habitat designated for green sturgeon (*Acipenser medirostris*) under the Endangered Species Act, and essential fish habitat conservation areas designated under the Magnuson-Stevens Fishery Conservation and Management Act. Uncertainty is considered primarily at the scenario level, by altering a key variable (cost of transmission cable) that determines the proximity of wave facilities to shore.

SINGLE SPECIES MODEL

Conservation Demand scenarios are likely to be linked to increased desire to recover individual protected species and stocks. Throughout the United States, hundreds of aging and unsafe dams have been removed, including large ones on the Sandy River in Oregon. The largest dam removal to date is in progress on the Elwha River, on the Olympic Peninsula in Washington. This dam removal is expected to increase salmon runs from current levels of several thousand to over one million. There has been considerable interest in removing four dams on the Snake River, but no progress has been made to date. Recently, work has begun to remove four dams on the Klamath River. If implemented, this would represent the largest dam removal in history. We apply a statistical single species population model to evaluate the potential impacts of the removal of the four Klamath River dams (**Appendix MS2**). The analysis evaluates the impacts of dam removal on Chinook salmon, *Oncorhynchus tshawytscha*. We forecast Chinook abundance and escapement under two alternatives (with and without dam removal) by constructing a life-cycle model composed of: 1) a stock recruitment relationship between spawners and age 3 in the ocean, which is when they are vulnerable to the fishery, and 2) a fishery model that calculates harvest, maturation, and escapement. To develop the stock recruitment relationship under assumptions of no dam removal, we estimated the historical stock recruitment relationship in the Klamath River below Iron Gate Dam in a Bayesian framework. To develop the stock recruit relationship under dam removal, we use the predictive spawner recruitment relationships in

Liermann et al. (2010) to forecast recruitment to age 3 from tributaries to Upper Klamath Lake, which is the site of active reintroduction of anadromy. We also modified the spawner recruit relationship under dam removal to include additional spawning capacity that would be added. In order to facilitate the comparison of the two alternatives, paired Monte Carlo simulations are used to forecast the levels of escapement and harvest with and without dam removal, fifty years into the future. Monte Carlo simulation was used to integrate across the uncertainty in the model parameters, and to translate these into uncertainties in model forecasts.

FOOD WEB AND ECOSYSTEM MODELS

The potential for direct and indirect effects of fishing can be identified using food web models and more detailed spatially-explicit ecosystem models. Such indirect effects of fishing are relevant to the **Human population growth scenario**, with increased demand for new species or lower trophic level species, the **Conservation Demand scenario**, which envisions changes in fishing practice to reduce negative effects on food webs, and the **Status Quo** scenario, that traces direct and indirect effects of the evolution of the groundfish individual quota (catch share) fishery. The simple food web model use here is Ecopath with Ecosim (Christensen and Walters 2004), implemented by Field et al. (2006) for the California Current. The approach begins with a simple mass-balance accounting of production and consumption of species groups (functional groups), linked by diet connections, and projects this forward in time (Ecosim) assuming predator-prey relationships. The ecosystem modeling approach we employ here is Atlantis (Fulton et al. 2011), which embeds a similar food web model in a spatial framework and links it to a physical oceanographic model. We consider two implementations of Atlantis for the California Current, one with finer scale geographic resolution in Central California (Horne et al. 2011; Kaplan et al. 2012), and another (Brand et al. 2007a; Kaplan et al. 2010) with more uniform geographic resolution that we use to dynamically model fishing fleet dynamics.

We apply Horne and colleagues' (2010) Atlantis ecosystem model and the Ecosim food web model to test the impact that depleting abundant lower trophic level forage groups has on other ecosystem components (**Appendix MS3**). We then apply a similar approach to test the implications of potential development of new fisheries, including those targeting less abundant species (**Appendix MS4**). This analysis considers area-specific responses to hypothetical fisheries that would be concentrated in particular parts of the California Current. Given a set of assumptions about future harvests by the groundfish vessels operating under an individual quota system, we then use this Atlantis model to investigate impacts on target and bycatch species biomass and harvest, as well as indirect (food web) effects (**Appendix MS5**). Finally, we apply the ecosystem model with fleet dynamics to predict the amount and location of groundfishing effort under individual quotas, and to predict the impact on target and non-target species (**Appendix MS6**). The model considers fishermen's response to quota prices for target and bycatch species, and penalties for exceeding quota. Of these four analyses involving food web and ecosystem models, the first two involve projections fifty years into the future; the other two that include more detailed modeling of fishery targeting are projected for 25 or 30 years. Uncertainty is handled primarily at the scenario level, for instance by defining alternate scenarios for future groundfish catches or for the penalties fishermen expect for exceeding quota. Effects of structural uncertainty (i.e. related to different model forms) are also considered by comparison of the joint application of Atlantis and Ecosim in Appendix MS3.

ECONOMIC INPUT/OUTPUT MODELS

All scenarios considered above will ultimately affect human communities, and here we begin to trace these effects for the portion of the **Conservation Demand scenario** related to Klamath Dam removal, and for the **Status Quo scenario** related to individual quotas (catch shares). After estimating changes in catches and

revenues associated with groundfish vessels switching to individual quotas, we apply an input-output model (Leonard and Watson 2011) to estimate how the rest of the US West Coast economy responds to these changes in fishery sector output 1, 5, 10, and 15 years in the future (Appendix MS5). These estimates include direct effects to the fishery sector, indirect effects to industries that supply the fishery sectors, and induced effects related to changes in household spending. Similarly, we apply an input-out model to estimate effects on income and employment over the course of 50 years that derive from changes in salmon harvest in response to Klamath River dam removal (Appendix MS7). Both analyses rely on IMPLAN (Impact Analysis for PLANning, <http://implan.com>), a commercially available data collection and regional modeling system commonly in use for land and resource management planning. Uncertainty is not handled explicitly in these economic analyses, but uncertainty at the scenario level (related to alternate fishery catches (Appendix MS5) or details of dam removal (Appendix MS2)) are propagated through to the economic model.

SCENARIO ASSESSMENT

SUMMARY

Quantitative analyses based on our scenarios identified the following alternate futures, vulnerabilities, and implications of alternate management decisions in the California Current.

- **The Human Population Growth scenario** can lead to potential increases in wave energy, and increased harvest of lower trophic level species and fishery targeting of new species such as grenadier and croaker. GIS mapping identified potential conflicts between wave energy and other marine uses such as tugboat lanes, sturgeon habitat, and some Oregon fishing ports. Ecosystem models suggest that large increases in harvest of lower trophic levels species (above current levels) would have substantial effects throughout the food web. However, harvest of less abundant species such as grenadier is unlikely to have large-scale effects, except at small spatial scales and for some plankton groups.
- **Climate Change and Energy Crunch scenarios may** also lead to development of wave energy and the potential conflicts listed above. Higher diesel fuel prices in the Energy Crunch scenario also affected profitability of groundfish fleets in the Status Quo scenario.
- **The Conservation Demand scenario** could involve dam removal or reductions in harvest of low-trophic level species. Dam removal on the Klamath River is likely to lead to increases in Chinook salmon abundance, and roughly a 45% increase in fishery revenue and impacts on employment, labor income, and output. Preventing increases in harvest of low-trophic level species, specifically forage fish and euphausiids, benefits their direct predators including fishery target species (in actuality, most forage species are currently unharvested or harvested at minimal rates).
- **The Status Quo scenario** investigated the new groundfish individual quota system. Results suggest that under individual quotas, the groundfish fleet could yield \$27-44 million more in revenue and \$22-36 million more in total income effects. Increased catches would primarily involve Dover sole and arrowtooth flounder, leading to moderate reductions in abundance of these stocks. Modeling of fleet dynamics under individual quotas suggests that the penalties fishermen expect for exceeding quota have the largest effect on fleet behavior, capping effort and total bycatch. Individual quota systems had high revenue per unit effort, and therefore doubling fuel costs had only moderate 10-14% impacts on net revenue. With alternative management systems (e.g. cumulative landings limits), doubled fuel costs erased all profits in some years.

Note that for these scenarios **Figures MS2-MS6** identify these quantitative analyses (blue), and other research questions for which quantitative analyses are needed (yellow). It is important to note that the scenarios and management actions that are tested in the quantitative analyses range from nearly certain to highly unlikely or illegal, given current legal frameworks and other factors.

DETAILED RESULTS

HUMAN POPULATION GROWTH

We applied quantitative models to consider three aspects of the human population growth scenario: wave energy development, increased harvest of forage fish, and increased harvest of new fishery target species.

Using a GIS-based decision support tool within the InVEST toolkit, we identified three sets of optimal locations for **wave energy facilities in Oregon (Appendix MS1)**. Development of such facilities is one avenue to address growing regional populations and power demand. We considered wave energy facilities that connect to the Tillamook, Toledo, and Tahkentic substation of the electrical power grid. Optimal locations were farther from shore in scenarios that assume lower cost of transmission lines. The average distance for the three facilities in each scenario was 16.1, 31.2, and 55.5 kms for the high, medium, and low cost scenario, respectively. There is a strong potential conflict with the tugboat and barge tow lanes for the high cost scenario (**Figure MS7**). There is also potential conflict with submarine cables connected to the Tillamook area. The locations of some wave energy facilities overlapped green sturgeon critical habitat (**Figure MS8**), particularly in the high cost scenario. For the Pacific groundfish conservation areas, there was an overlap for two of the three facilities in the low cost scenario. The medium cost scenario presented the strongest potential conflict in terms of a wave energy facility interfering with groundfish harvesting. Potential for conflict with particular ports' fishing areas is strongest for the high cost scenario, in which wave energy facilities are closest to shore. The results demonstrate how potential conflicts with existing marine uses can be identified. Simple spatial representations can present planners with a screening tool, identifying areas where a more refined investigation is worthwhile.

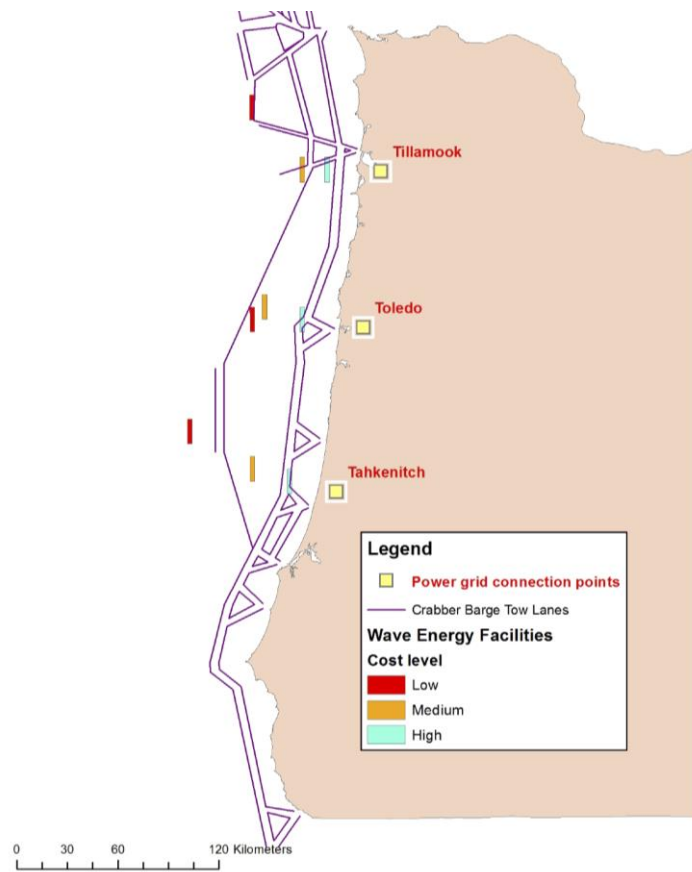


Figure MS7. Sites for potential wave energy facilities, power grid connection points, and barge tow lanes.

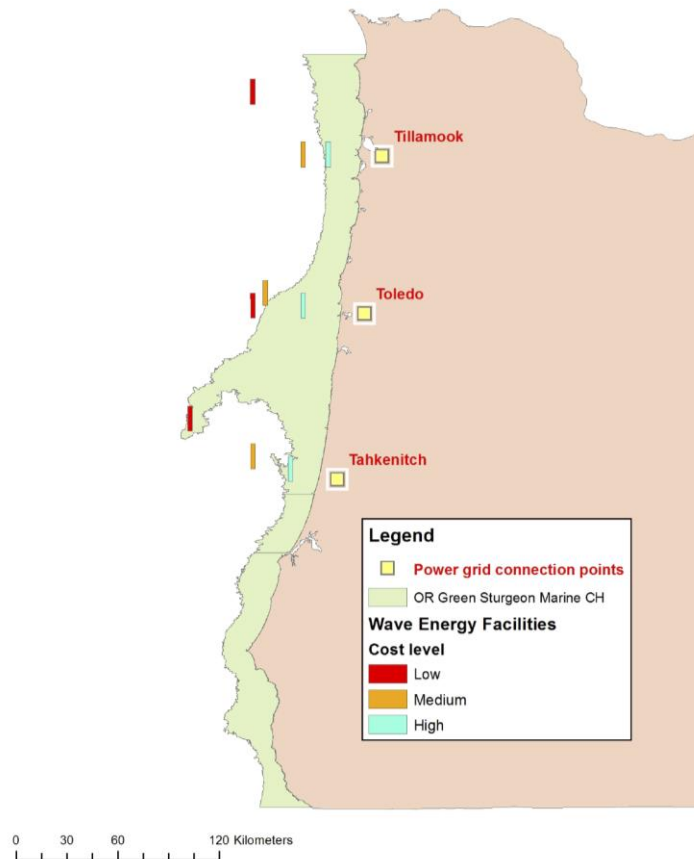


Figure MS8. Sites for potential wave energy facilities, power grid connection points, and green sturgeon critical habitat.

We applied food web and ecosystem models to identify ecosystem-level impacts due to increased demand for, and **depletion of, lower-trophic level forage species (Appendix MS3)**. Demand for harvests of forage species will increase due to global increases in population and affluence and associated demand for feed for aquaculture and livestock. Although harvest of many forage species is prohibited within the California Current, using two models we estimated the abundance that would lead to maximum sustainable yield of euphausiids, forage fish, mackerel, and mesopelagic fish (e.g. myctophids), but found that increasing harvests and depleting forage groups to these levels can have both positive and negative effects on other species in the California Current (**Figure MS9**). Though higher trophic level species such as groundfish are often managed on the basis of reference points that can reduce biomass to 40% of unfished levels, scenarios that involved depletion of forage groups to this level commonly led to impacts on predators of forage groups, some of which showed declines of >20%. Depletion of euphausiids and forage fish, which each comprise > 10% of system biomass, had the largest impact on other species. Depleting euphausiids to 40% of unfished levels altered the abundance of 13-30% of the other functional groups by >20%; while depleting forage fish to 40% altered the abundance of 20-50% of the other functional groups by >20%. The results emphasize the trade-offs between the harvest of forage groups and the ability of the California Current to sustain other trophic levels.

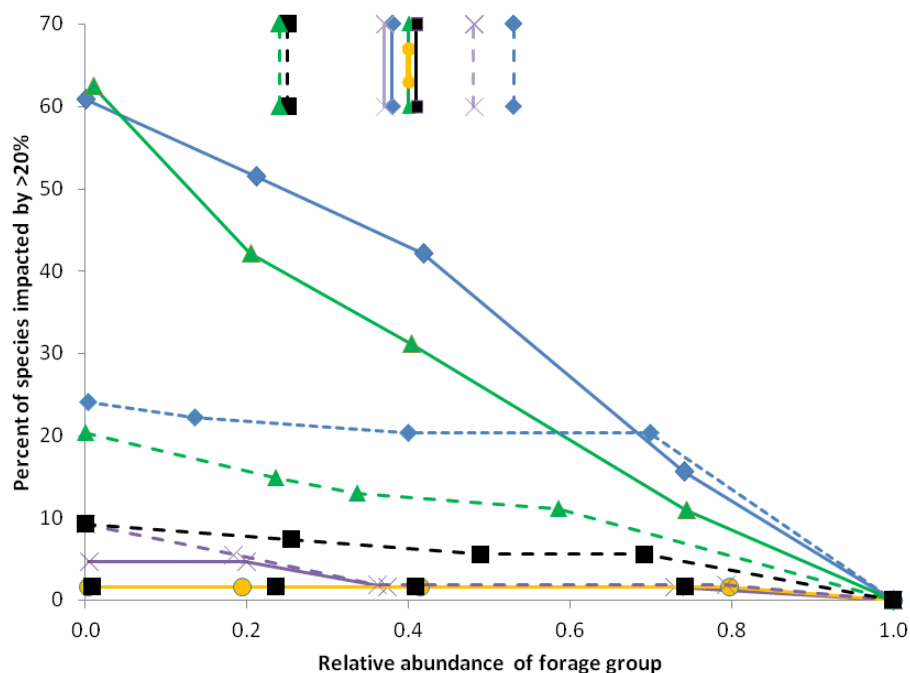


Figure MS9. Percent of species in California Current Ecosim food web model (solid lines) and Atlantis ecosystem model (dashed lines) that exhibit changes in biomass of > 20% (either positive or negative) when forage groups are depleted below unfished levels. A value of 1.0 on the x-axis represents abundance of the forage group when it is not fished, while a value of 0.4 represents depletion of a focal forage group to 40% of unfished abundance. Focal forage groups are as follows: euphausiids -- green triangles; forage fish -- blue diamonds; mesopelagic fish -- purple crosses; mackerel -- black squares; sardines in Ecosim-- orange circles. Vertical lines of the same colors represent abundance of each forage group that leads to maximum sustainable yield in the two models (only position on the x-axis is relevant, y-position is for graphical clarity only).

New fisheries could arise due to global seafood demand. Using a spatially explicit Atlantis ecosystem model, we predicted impacts of three potential fisheries targeting grenadier (*Macrouridae*), white croaker (*Genyonemus lineatus*), and shortbelly rockfish (*Sebastes jordani*) (Appendix MS4). Unlike the analysis testing effects of depleting more abundant forage species (Appendix MS3), the focus here was on low-biomass species that could arise due to niche markets and new consumer demand, rather than bulk demand for fishmeal. We explored fishing scenarios (fifty year projections) for these groups that resulted in depletion levels of 75, 40, and 25 percent. Results indicate that coast-wide the impacts of developing fisheries on these targets would be relatively small (**Figure MS10**), in terms of impacts on other species and fisheries. The spatial distribution of impacted functional groups was patchy, and concentrated in the central California region of the model. This work provides a framework for evaluating impacts of new fisheries with varying spatial distributions and suggests that regional effects should be evaluated within a larger management context.

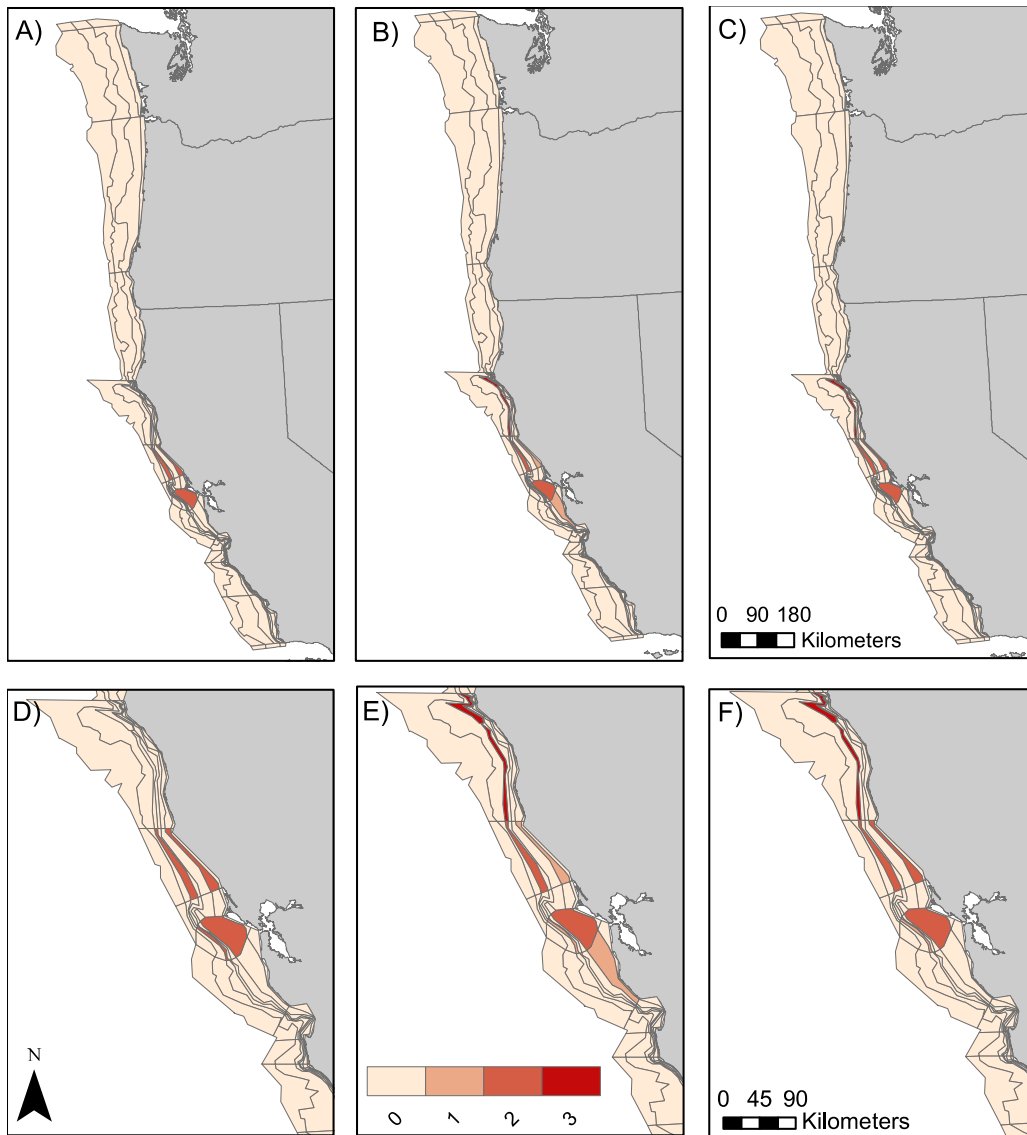


Figure MS10. Number of functional groups affected by a grenadier fishery at three fishing levels (threshold of 10 percent change) by cell. Fishing scenarios represented are F75 (A, D), F40 (B, E), and F25 (C, F). Density of color indicates increasing number of functional groups affected, as indicated by legend.

CLIMATE CHANGE SCENARIO AND ENERGY CRUNCH SCENARIO

One political and economic response to climate change may be a shift to low-carbon power, such as wave energy. Wave energy may also be a response to the energy crunch scenario, which could prompt investment in new energy sources. As noted above, we identified three sets of optimal locations for wave energy facilities in Oregon (Appendix MS1), but also identified potential conflicts with sectors such as tugboat lanes, sturgeon critical habitat, and fishing areas. The total Mwh/yr captured by all three facilities would be 3564, 3462, and 3324 Mwh/yr for the low, medium, and high cost scenarios, respectively. The average

energy captured per device also increases as lower transmission costs are assumed, which corresponds to the higher wave energy potential further offshore along the Oregon coast.

Climate change is also likely to impact small pelagic fish such as sardine and anchovy, and anadromous species such as Chinook salmon. Two avenues for research are discussed in Boxes MS1 and MS2.

Box MS1.



Analyses already exist that predict the response of particular runs of Chinook salmon to climate, and these approaches can be developed further for the IEA. Spring/summer Chinook have been shown empirically to be vulnerable to water temperature and streamflow (Crozier and Zabel 2006), and population models of Snake River and Snohomish River Chinook have been linked to downscaled global circulation models that include climate change (Battin *et al.* 2007; Crozier *et al.* 2008). Additional downscaling of climate models to predict hydrology for broad regions, and applications to

multiple salmon populations may allow an analysis of climate change at a larger scale. Climate change effects will not occur in isolation from other drivers such as population growth: streamflow will also be influenced by land use change (Battin *et al.* 2007) and human demand for water, due to predicted 50% increases in population growth over 50 years (Bierwagen 2009).

The groundfish management system is likely to influence the vulnerability of fisheries profits to energy prices (**Figure MS11**). Modeling of the groundfish fleet under the new individual quota system predicts substantial reductions in effort as compared to the previous cumulative landings limit system (Appendix MS6). Gross revenue declines only slightly under individual quotas as compared to landings limits, and net revenues (after variable costs such as fuel, and fixed costs) are typically higher under individual quotas. Our simulations assumed fuel to be \$3/gallon; diesel fuel prices for West Coast states averaged \$3.64-\$3.72 in August 2012 (<http://www.psmfc.org/efin>). Assuming \$6/gallon fuel heavily penalizes the scenario with high fishing effort (cumulative trip limits): for some years fuel costs erase all profits under cumulative landings limits. In our 30 year model projections, individual quota systems have higher revenue per unit effort and therefore fuel costs have only moderate 10-14% impacts on net revenue (profits).

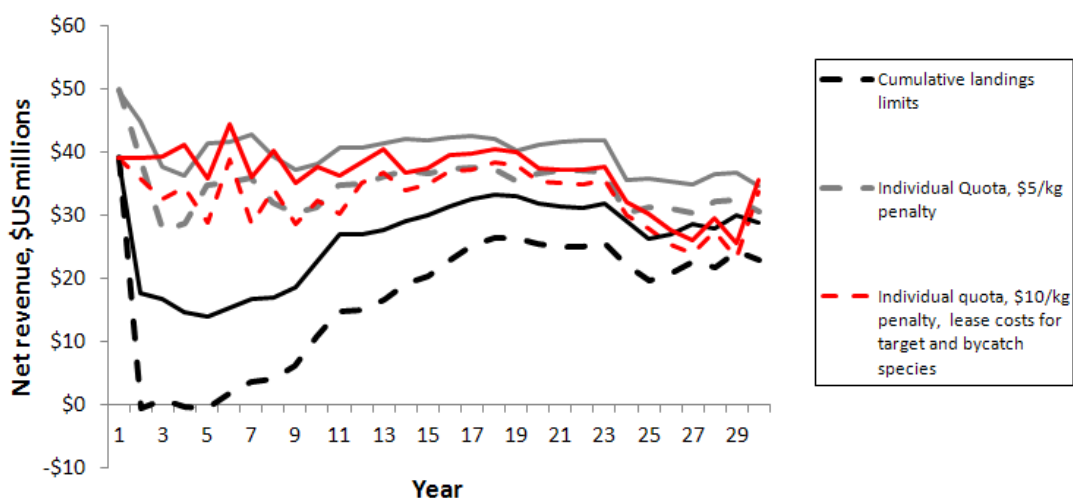


Figure MS11. Net revenue for West Coast groundfish fleets over 30 years. Solid lines denote fuel at \$3/gallon, dashed lines at \$6/gallon. This simple metric of net revenue is gross revenue minus fixed costs (excluding capital costs) and variable costs (fuel, ice, and food, but not labor or quota costs). Details as in Appendix MS6, except that annual net revenue calculation includes adjusted variable costs to include \$6 fuel. Colors denote options for the management system: black = cumulative landings limits in place prior to 2011; grey = individual quotas with no lease price and low penalties for exceeding quota; red = individual quotas with higher lease costs and penalties.

CONSERVATION DEMAND

The Conservation Demand scenario envisions increased public and political desire for species recovery and ecosystem health. Here we evaluate two facets of that: effects of dam removal, and effects of restricting harvest of forage fish.

We evaluated the impact of [Klamath River dam removal on Chinook salmon \(Appendix MS2\)](#), projecting population dynamics for the period from 2012 to 2061. Median escapements and harvest were higher under dam removal than with no action (**Table MS1**), though there was a high degree of overlap in 95% confidence intervals due to uncertainty in stock-recruitment dynamics. Still, there was a 0.75 probability of higher annual escapement and a 0.7 probability of higher annual harvest by performing dam removal relative to no action, despite uncertainty in the abundance forecasts. The median increase in escapement in the absence of fishing was 81%, the median increase in ocean harvest was 47%, and the median increase in tribal harvest was 55% under dam removal relative to no action.

Table MS1. Percent increase in abundance and harvest due to performing dam removal versus no action, for two time periods: 1) prior to dam removal (2012 – 2019); and after removal of dams and cessation of active reintroduction and production of the Iron Gate Hatchery production (2030-2061). “95% CrI” is 95% credibility interval.

| | 2012 – 2020 | | 2033-2061 | |
|--------------------------------------|-------------|-------------|-----------|-------------|
| Metric | Median | 95% CrI | Median | 95% CrI |
| Escapement in the Absence of Fishing | 11% | -80%, 493% | 81% | -60%, 881% |
| Lower Basin Escapement | 0% | -72%, 386% | 9% | -76%, 490% |
| Ocean Commercial Harvest | 9% | -87%, 836% | 47% | -69%, 1495% |
| Ocean Recreational Harvest | 9% | -87%, 836% | 47% | -69%, 1495% |
| River Harvest | 0% | -92%, 1520% | 9% | -77%, 2754% |
| Tribal Harvest | 10% | -89%, 1010% | 55% | -71%, 1841% |

Based on these projections for Chinook salmon harvest, we estimated annual changes in fishery revenue likely to derive from **Klamath dam removal, and applied an input-out model to estimate effects on income and employment (Appendix MS7)**. Higher abundance of Klamath River Chinook due to dam removal would allow more fishing on all Chinook stocks south of Cape Falcon Oregon, since harvest of all stocks in this broader region has been limited by low abundance of Klamath Chinook. We estimated \$17.1 million in annual troll fishery revenue without dam removal, and a 43% increase to \$24.4 million with dam removal. Impacts in the broader economy include an additional \$8.9 million annually in gross revenue, distributed across five management regions. For San Francisco, Fort Bragg and Central Oregon, annual impacts (depending on the area) include an additional 69 to 218 jobs, an additional \$1.05 million to \$2.56 million in labor income, and an additional \$2.41 million to \$6.6 million in output. For the Klamath Management Zones in California and Oregon, the annual impacts include an additional 11 to 19 jobs, an additional \$0.06 million to \$0.07 million in labor income, and an additional \$0.13 million to \$0.19 million in output.

Conservation demands may lead to reductions in existing harvest of forage groups. As mentioned above, we applied food web and ecosystem models to identify ecosystem-level impacts due to **a range of potential harvest rates for lower-trophic level forage species (Appendix MS3)**. Though higher trophic level species such as groundfish are often managed on the basis of reference points that can reduce biomass to 40% of unfished levels, we found that depleting forage groups to this level could have large effects on other species in the food web, with up to half of all species responding by >20%. These responses were strongest for euphausiids and forage fish, which are highly abundant and are common diet items for predators. Conservation demand scenarios to restrict harvest of these forage groups would primary benefit their direct

predators, including target fish species. Caveats include the simulation of coast-wide harvests, the aggregation of multiple species into functional groups, and the testing of a broad range of harvest rates, including rates that exceed current levels and legal limits. Other ongoing efforts (**Box MS2**) will have finer taxonomic and spatial resolution, and will also link to climate and oceanography models.

Box MS2.

An extensive collaboration between multiple researchers* has been developing a new type of model that may capture the dynamics and climate response of forage species such as California Current sardine and anchovy. For such species, managers are increasingly being asked to quantify fishing effects at the ecosystem level, present fishing impacts relative to other factors such as environmental conditions, and to project fishing effects under future, previously unobserved, conditions such as climate change. These activities require models that represent ocean circulation, lower trophic levels, a fish food web, and fishing dynamics in sufficient detail to allow for fishing to respond to changing conditions and to account for both direct and indirect effects of fishing.



Recently, advances in physics and biology have made possible end-to-end (climate-to-fish-to-fishers) ecosystem models, including fishing (humans) as a dynamical component. Our group has been developing one such end-to-end model within the widely-used ROMS (Regional Ocean Modeling System) circulation model. The concentration-based NEMURO (Nutrient-Phytoplankton-Zooplankton-type) submodel provides lower trophic level dynamics, including multiple nutrients, two phytoplankton and three zooplankton fields. A multi-species, individual-based, full life cycle submodel simulates fish population and community dynamics, including fishing fleets as one of the predator species. Our preliminary version focuses on anchovies and sardines in the California Current System. Using a 10-km resolution ROMS model, we have demonstrated proof-of-concept, how the multiple submodels can be integrated simultaneously for a multi-decadal historical simulation (1958-2006).

***Contributors**

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Pacific sardine photo courtesy of Tewey, Monterey Bay Aquarium

STATUS QUO

In our Status Quo scenario, we assume that drivers and pressures will continue at current rates or trends. However, even assuming that most other aspects of the system do not change, we expect rapid human responses to individual quotas (catch shares), the current management framework for groundfish fleets. The Pacific Fisheries Management Council implemented this individual transferrable quota (ITQ) system in 2011 for the West Coast groundfish trawl fleet. Under the ITQ system, each vessel now receives transferable annual allocations of quota for 29 groundfish species, including target and bycatch species.

Individual quotas and the new incentives they present are likely to cap most bycatch, while leading to increases in catch of target species (particularly flatfish) through changes in gear, location and timing of fishing. As part of previous work, Pacific Fishery Management Council staff developed several projections for fishery catch under varying assumptions about improvements in targeting accuracy under an **individual quota system**. In Appendix MS5, we apply these catch projections in 25 year simulations and find that target species in the California current responded directly to the imposed fishing mortality rates. Indirect (trophic) effects were minor and typically involved response of less than 10%. Relative to pre-catch share conditions, the scenarios suggest improved targeting by the groundfish fleet could yield \$27-44 million more in revenue to the fishery sectors (dockside value). At the scale of the broader West Coast economy, the IO-PAC input/output model suggests this may translate into \$22-36 million more in total income, which includes employee compensation and earnings of business owners (**Figure MS12**).

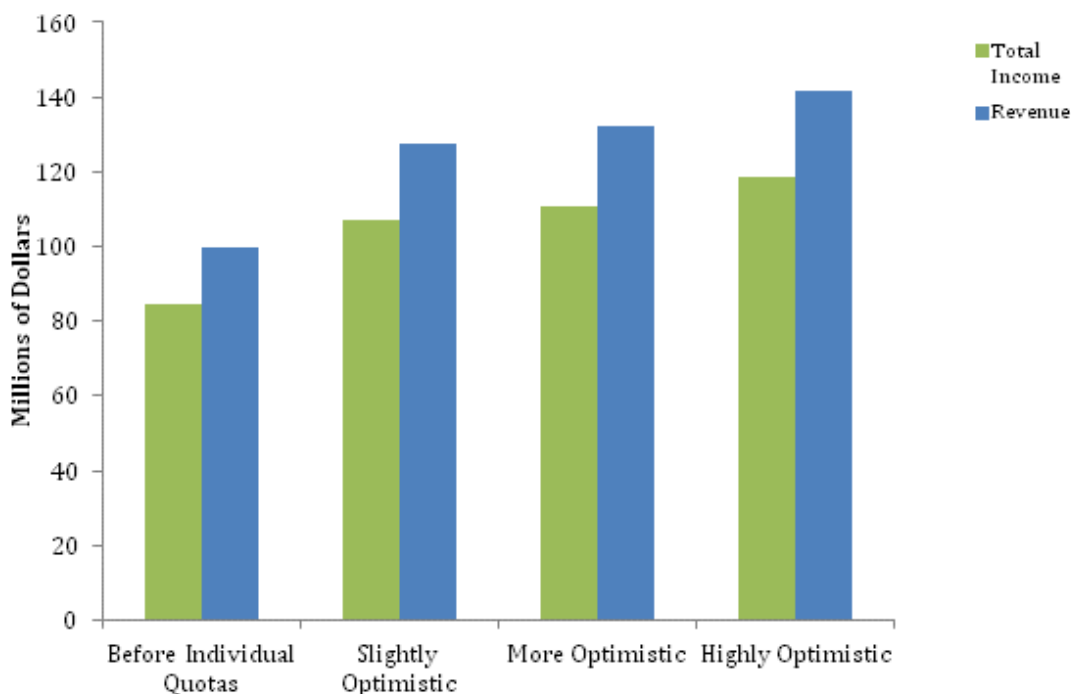


Figure MS12. Revenue in fishery sectors, and income effects in the broader West Coast economy. Year 1 predictions. Total income and revenue are represented by bars in millions of dollars (left axis). “Slightly optimistic” scenarios for individual quotas assume moderate increases in target species catch and little change in rockfish bycatch, while “Highly optimistic” scenarios for individual quotas assume large increases in target species catch with little change in rockfish bycatch.

Fishermen’s response to individual quotas is likely to evolve as a function of quota costs, enforcement, penalties for exceeding quota, initial quota allocation, and captains’ ability to target particular species. We simulated [fleet dynamics under an individual quota system \(Appendix MS6\)](#) and found that in the absence of penalties for discarding over-quota fish, removing constraints related to the previous management system (per-vessel landings limits) led to large increases in fishing effort and bycatch. The penalties fishermen expected for exceeding quota had the largest effect on fleet behavior, capping effort and total bycatch. Quota prices for target or bycatch species had lesser impacts on fishing dynamics, even up to bycatch quota prices of \$50/kg. Ports that overlapped less with bycatch species could increase effort under individual quotas, while other ports decrease effort. Relative to a prior management system, ITQs with penalties for exceeding quota led to increased target species landings and lower bycatch, but with strong variation among species. In addition to providing insights into how alternative fishery management policies affect profitability and sustainability, the model illustrates the wider ecosystem impacts of fishery management policies.

Combining some aspects of the Energy Crunch and Status Quo scenarios, we considered the potential impacts of spatial closures due to wave energy facilities in Oregon ([Appendix MS1](#)) on groundfish fleet dynamics ([Appendix MS6](#)). Resulting fleet effort and catch were predicted to vary by less than 1% due to these simulated closures. The four model regions off the Oregon coast are large relative to the size of these facilities (only 72 km² total), and closures would not exceed 2% of each region (**Table MS2**). Note that this fleet dynamics modeling is indicative of overall patterns at a fairly coarse spatial scale, and the finer scale GIS analysis ([Appendix MS1](#)) indicates potential conflicts for particular ports and gears.

Table MS2. Percent of each model polygon closed to groundfish fleets, assuming establishment of three wave energy facilities per cost scenario, with each facility closing fishing in an area 12km N-S and 2km E-W . Each model polygon spans most of the Oregon coast in the N-S direction, and is defined by depth contours indicated in the column headings.

| Oregon coast, from Columbia River to Cape Blanco Region: | | | | |
|---|---------|----------|----------|----------|
| Cost scenario | 50-100m | 100-150m | 150-200m | 200-550m |
| Low | 0.0% | 1.0% | 1.0% | 0.2% |
| Medium | 0.0% | 0.8% | 1.7% | 0.0% |
| High | 1.3% | 0.6% | 0.0% | 0.0% |

"NATURAL" ECOSYSTEM COMPONENTS ACROSS SCENARIOS

SUMMARY OF NATURAL COMPONENTS: PROTECTED SPECIES AND ECOSYSTEM INTEGRITY

The quantitative analyses do not predict how all attributes of the California Current system might respond to our scenarios, but they do make the following predictions regarding natural components:

- **Human Population Growth scenario:** Wave energy facilities built in response to increased demand for power could impact green sturgeon habitat. Increased consumer demand for trawl-caught species could lead to increased take of Steller sea lions and California sea lions. Models predict only modest indirect changes on the food web and ecosystem structure in response to three potential new fisheries. Large increases in harvest of forage species (above current levels) may restructure energy pathways related to alternate forage groups, such as copepods.
- **Climate Change and Energy Crunch scenario:** As above, wave energy facilities built to produce low-carbon power or to meet increased energy demand may impact green sturgeon habitat.
- **Conservation Demand scenario:** Dam removal on the Klamath River could increase Chinook salmon abundance. In future research, this model prediction can be compared to ongoing monitoring in the Elwha River basin, where 2 large dams have almost entirely removed. A separate food web model analysis of the California Current predicts that limiting harvest of forage species (e.g. sardine and euphausiids) to low catch levels may benefit some protected species such as seabirds and mammals; however, an ecosystem model predicts little response of protected species at the coast-wide level.
- **Status Quo:** The groundfish individual quota system includes mechanisms to reduce bycatch of rockfish and encourage their recovery; enforcement of target species quotas are the strongest such mechanism. Increased harvests of groundfish under the individual quota system could lead to increased take of Steller sea lions and California sea lions. Models predicted that at a coast-wide level, strong impacts on the food web and ecosystem typically occur at high benchmark fishing mortality rates, which exceed both current harvest rates and legal limits on catch.

PROTECTED SPECIES

In the **Human Population Growth**, **Energy Crunch**, and **Climate Change** scenarios, wave energy facilities are likely to overlap critical habitat for green sturgeon (**Appendix MS1**). The severity of the impact on sturgeon habitat is not known, but the spatial modeling suggests that if high electricity transmission costs force wave energy to be sited near shore, there is potential for overlap between sturgeon habitat and wave energy arrays.

Conservation Demand scenarios leading to dam removal on the Klamath River would increase abundance of Chinook salmon (**Appendix MS2**). Were the Klamath River dams removed, the adult salmon returned would increase by around 80% for the period 2030-2061. Lower Klamath basin escapement (returns after fishing) would be 9% higher. The analysis does not consider the effects on other anadromous species that might benefit from dam removal.

Restoring access of anadromous species such as salmon to historical spawning grounds, as discussed here for the Klamath River system, will become more common in the future. This is because many dams that

block anadromous access are aging and removing them is often a more cost effective and straightforward solution than trying to repair or refurbish them. Actual dam removal in the Klamath River system will likely require years due to such issues as funding and permitting. Thus, being able to compare model predictions of the response of anadromous species with monitoring data will require decades. However, model predictions for the Klamath can be compared to results of ongoing monitoring from the Elwha River basin, where two large dams have almost entirely been removed. Predictions of the abundance, species composition, spatial distribution, and diversity of anadromous species at various intervals following dam removal have been made and will be compared to the actual response of anadromous species, ultimately improving predictions for other rivers such as the Klamath.

The **Human Population Growth** and **Conservation Demand scenarios** considered indirect (food web) effects that would result from depleting forage groups (**Appendix MS3**). However, the impacts on protected species are equivocal, with Ecosim predicting more dynamic responses (as was typical in these model comparisons). Ecosim food web modeling predicted that depletion of forage fish would negatively impact some seabirds and marine mammals. However, the Atlantis ecosystem model did not predict strong declines in marine mammals or birds due to forage fish depletion. The Ecosim food web modeling predicted that depletion of euphausiids would lead to a shift in production towards copepods and micro-zooplankton, with subsequent increases in bird groups. The Atlantis model similarly predicted that euphausiid depletion would shift production toward copepods, but two protected groups (baleen whales and surface seabirds) that depend heavily on euphausiids had only slight declines (10% or less).

Direct impacts on protected species would also result from changes in groundfish landings. The **Status Quo scenario** included increases in landings of flatfish (**Appendix MS5**), which are likely to be associated with increased fishing effort by the groundfish trawl fleet. In the **Human Population Growth scenario**, increased harvest of grenadier (**Appendix MS4**) would also most likely involve groundfish trawl gear, with its associated bycatch of protected species. Jannot et al. (2011) estimated bycatch of marine mammals, seabirds, and sea turtles by groundfish gears for the years 2002-2009. Of all the species in these groups, California sea lions had the highest estimated bycatch, with estimated coastwide totals between 10 and 116 animals per year, with the majority of observations occurring in groundfish trawl fisheries. Steller sea lions were caught in smaller numbers, with estimated bycatch totals of 0-17 animals per year. Very few seabirds and turtles have been observed as bycatch in groundfish trawl fisheries.

Estimating the change in bycatch levels associated with increased landings depends on the spatial and temporal distribution of fishing effort and the specific fishing method. Furthermore, changes in bycatch rates that may have occurred after the implementation of the catch share system in 2011 are not reflected in the data analyzed by Jannot et al. (2011). Thus, specific estimates of increases in bycatch of sea lions or any other protected species are difficult. In the projections considered here to represent harvests under an individual quota system (**Appendix MS5**), the multipliers on fishing mortality were in the range 1-4. These values probably represent upper bounds on the increase in bycatch of protected species under these catch projections. However, the coastwide effort for many fully exploited species is not expected to increase under these scenarios, so the maximum increase in coastwide bycatch of any species is likely to be much smaller than four-fold.

ECOSYSTEM INTEGRITY

The **Human Population Growth scenario** led to investigation of the impacts of new fisheries and their potential ecosystem-level effects (**Appendix MS4**). Generally, the potential fisheries considered – grenadier, croaker, and shortbelly rockfish – would harvest low amounts of biomass, and the trophic effects of these were minimal at the coastwide scale. Food web response tended to involve plankton species such as copepods, microzooplankton, dinoflagellates, and phytoplankton, and to be concentrated in Central California.

The **Human Population Growth** and **Conservation Demand scenarios also** considered the effect on food web structure of depleting more abundant forage groups such as euphausiids (krill), mackerel, myctophids (lantern fish), and small pelagic fish (**Appendix MS3**). Two contrasting modeling approaches, Atlantis and Ecosim, both found that harvest of these forage species can have positive as well as negative effects on other species in the California Current. The most common impacts were on predators of forage groups, some of which showed declines of >20% under the scenarios that involved depletion of forage groups to typical single-species management targets. Depletion of euphausiids and forage fish, which each comprise > 10% of system biomass, had the largest impact on other species, restructuring the food web to follow energy pathways related to alternate lower-trophic level groups.

Ecosim food web modeling predicted that predators, including large piscivores (salmon, sharks, sablefish *Anoplopoma fimbria*), seabirds and marine mammals would decline in response to the depletion of forage fish. However, the model also predicted a restructuring of food web energy flow towards zooplankton: depletion of forage fish released euphausiids and copepods from predation pressure, resulting in increased abundance of those groups. This in turn provided more prey for higher trophic levels, many of which increased in abundance. The Atlantis model also predicted an increase in abundance of euphausiids in response to forage fish depletion. Unlike the Ecosim predictions, the Atlantis modeling did not predict strong declines in marine mammals or birds due to forage fish depletion.

The Ecosim food web modeling predicted that depletion of euphausiids would lead to a shift in production towards copepods and micro-zooplankton, with subsequent increases in forage fish and their predators, including several flatfish and bird groups and black rockfish (*Sebastes melanops*). The Atlantis model predicted that euphausiid depletion would cause a shift in production toward copepods, but that euphausiid removal would cause moderate declines (>20%) in many mid-trophic level groups, primarily predators on euphausiids. Euphausiid depletion also led to declines of 10% or less for two protected groups (baleen whales and surface seabirds), an overfished rockfish functional group (yelloweye and cowcod), as well as small demersal sharks and midwater rockfish.

The **Status Quo scenario** related to individual quotas for groundfish fleets caused extensive effects on the ecosystem (food web structure) only when fishing effort was allowed to rise to very high levels. In hypothetical benchmark simulations that lacked caps on effort and bycatch (**Appendix MS6**), abundance of targets species such as sablefish and large flatfish and bycatch species such as Pacific Ocean Perch and darkblotched rockfish declined. In these same benchmark simulations, over-fishing of piscivores led to a release of forage groups (small planktivores, deep vertically migrating fish, cephalopods, and nearshore fish). Thirty to sixty percent increases in these forage groups led to 10-50% increases in bird and pinniped abundance under these scenarios, since birds and mammals also consume forage species such as sardines and squid. Two highly productive invertebrate groups, shrimp and meiobenthos (flagellates, ciliates, nematodes) also responded indirectly to these benchmark ITQ cases. These benchmark high fishing

mortality rates were required for two ecosystem models (Brand *et al.* 2007b; Horne *et al.* 2010) to predict strong indirect (trophic) effects on the food web. Applying projections of catch under individual quotas, we found that functional groups that were not subject to increased fishing pressure in the catch share scenarios did not deviate more than 10% from status quo ([Appendix MS5](#)). Increases in groundfish catch caused slight increases (<6%) of three invertebrate prey groups, which ultimately led to minor increases (<10%) for some pelagic predators such as sharks and mackerel.

HUMAN WELL-BEING ACROSS SCENARIOS

SUMMARY

We have identified which ports and communities are most likely to gain or lose economic activity under these scenarios, and where possible have translated these to revenue, income, and employment both in fishery sectors and in the broader economy:

- Scenarios that involve wave energy development involve increases in non-fishery revenue near electrical substations (**e.g. Tillamook and Toledo**), but potential fishery losses for communities such as **Newport and Astoria**.
- Scenarios that vary the harvest of small pelagic fish have the strongest effects on revenue in **Central and Southern California ports**.
- Potential increase in demand for new species can lead to small but concentrated increases in fisheries revenue. For instance, increased landings of shortbelly rockfish could provide a boost (**\$~1 million in revenue**) to the relatively small fishing communities of **Central California**.
- Klamath River dam removal would cause a **42-44% increase in fishery revenue and resulting employment and income** in the broader economy. For **San Francisco, Fort Bragg and Central Oregon**, annual impacts (depending on the area) include an additional **69 to 218 jobs**, an additional **\$1.05 million to \$2.56 million in labor income**, and an additional **\$2.41 million to \$6.6 million in output**.
- The groundfish trawl fleet and associated processors and wholesalers, which are most concentrated in **Oregon and Northern California**, are projected to see long-run increases in revenues of **\$27-44 million**. At the scale of the broader West Coast economy, the economic model suggests this may translate into **\$22-36 million more in total income**.
- **Under individual quotas for groundfish, fleets that cannot stay below quotas are likely to reduce fishing effort and revenue**. In these simulations, Moss Landing, Fort Bragg, Eureka, and Coos Bay increase effort and landings, while northern fleets are more likely to cut effort. **Individual quotas have high revenue per unit effort**, and have fishery profits that are less vulnerable to increased **fuel costs**.

HUMAN WELL-BEING

Though detailed predictions related to human well-being are still in development, we can begin to identify which ports and communities are most likely to gain or lose economic activity under these scenarios. Future analyses for the IEA will build on this to predict two aspects of human well-being, resilience and vulnerability, in response to changes in port-level fishery activity and income (Jacob et al. (2012), see **Box MS3**).

Under **Human Population Growth**, **Climate Change** and **Energy Crunch** scenarios, non-fishery economic activity in Oregon is expected to increase near the Tillamook, Toledo, and Tahkentic (near Reedsport) power substations. The wave energy facility siting exercise ([Appendix MS1](#)) considered

relatively small-scale arrays, but noted that any future wave energy sites must be near these existing substations to connect to the electrical grid. Potential fishery losses might occur for the Newport fleet, based on spatial overlap with wave energy sites, and based on the large proportion of Newport revenue from groundfish fleets (**Tables MS2-MS3**). Other Oregon fleets, such as Astoria (**Tables MS2-MS3**), that harvest groundfish may also lose revenue depending on spatial overlap of fishing areas with wave energy sites.

Table MS2: For 2006-2010, the proportion of each portgroup's revenue derived from each species or species group. From PacFIN landings database.

| PORTGROUP NAME | PACIFIC WHITING | GROUND FISH TRAWL | GROUND FISH NONTRAWL | SALMON | CRAB | SHRIMP | SHELLFISH | PELAGICS | HIGHLY MIGRATORY | OTHER | PORTGROUP AVG. ANNUAL REVENUE (\$1000s) |
|---------------------------------------|-----------------|-------------------|----------------------|--------|------|--------|-----------|----------|------------------|-------|---|
| BELLINGHAM | 0% | 4% | 7% | 21% | 35% | 3% | 14% | 0% | 1% | 14% | \$ 54,977 |
| SEATTLE | 0% | 0% | 0% | 25% | 4% | 1% | 67% | 1% | 0% | 2% | \$ 33,995 |
| WESTPORT | 10% | 2% | 4% | 8% | 51% | 5% | 1% | 2% | 15% | 2% | \$ 48,185 |
| ILWACO | 3% | 0% | 7% | 14% | 32% | 2% | 0% | 1% | 37% | 2% | \$ 18,823 |
| OTHER WASHINGTON | 0% | 0% | 0% | 29% | 29% | 0% | 37% | 0% | 0% | 5% | \$ 796 |
| ASTORIA | 7% | 22% | 2% | 10% | 24% | 6% | 0% | 15% | 11% | 2% | \$ 33,901 |
| GARIBALDI | 0% | 1% | 5% | 7% | 72% | 6% | 2% | 0% | 8% | 0% | \$ 3,274 |
| NEWPORT | 10% | 12% | 8% | 2% | 44% | 9% | 0% | 0% | 13% | 2% | \$ 31,541 |
| CHARLESTON | 2% | 18% | 7% | 2% | 43% | 16% | 0% | 0% | 10% | 3% | \$ 22,907 |
| BROOKINGS | 0% | 16% | 23% | 2% | 52% | 4% | 0% | 0% | 1% | 2% | \$ 9,599 |
| CRESCENT CITY | 2% | 6% | 5% | 0% | 80% | 5% | 0% | 0% | 1% | 0% | \$ 14,542 |
| EUREKA | 2% | 26% | 5% | 1% | 58% | 1% | 0% | 0% | 3% | 3% | \$ 13,297 |
| FORT BRAGG | 0% | 30% | 17% | 12% | 17% | 0% | 0% | 0% | 1% | 22% | \$ 7,037 |
| BODEGA BAY | 0% | 2% | 3% | 18% | 73% | 0% | 0% | 0% | 0% | 3% | \$ 4,949 |
| SAN FRANCISCO | 0% | 9% | 4% | 5% | 64% | 2% | 0% | 4% | 4% | 8% | \$ 12,726 |
| MOSS LANDING | 0% | 7% | 10% | 3% | 6% | 5% | 0% | 64% | 2% | 3% | \$ 8,791 |
| AVILA | 0% | 4% | 65% | 1% | 7% | 6% | 0% | 1% | 8% | 8% | \$ 3,784 |
| SANTA BARBARA | 0% | 0% | 2% | 0% | 4% | 4% | 0% | 62% | 1% | 27% | \$ 35,356 |
| TERMINAL ISLAND | 0% | 0% | 3% | 0% | 1% | 3% | 0% | 75% | 3% | 15% | \$ 30,623 |
| OCEANSIDE | 0% | 0% | 11% | 0% | 2% | 7% | 0% | 0% | 19% | 60% | \$ 6,480 |
| OTHER CALIFORNIA | 0% | 0% | 0% | 0% | 1% | 5% | 0% | 0% | 0% | 93% | \$ 53 |
| OFFSHORE | 100% | 0% | 0% | 0% | 0% | 0% | 0% | 0% | 0% | 0% | \$ 23,046 |
| SPECIES GROUP SHARE OF ANNUAL REVENUE | 8% | 7% | 6% | 8% | 30% | 4% | 8% | 14% | 7% | 8% | \$ 418,683 |

Table MS3: For 2006-2010, the proportion of revenue derived from each species or species group that is landed in each portgroup. From PacFIN landings database.

| PORTGROUP NAME | PACIFIC WHITING | GROUND FISH TRAWL | GROUND FISH NON TRAWL | SALMON | CRAB | SHRIMP | SHELLFISH | PELAGICS | HIGHLY MIGRATORY | OTHER | PORTGROUP SHARE OF TOTAL REVENUES |
|-------------------------------------|-----------------|-------------------|-----------------------|-----------|------------|-----------|-----------|-----------|------------------|-----------|-----------------------------------|
| BELLINGHAM | 0% | 8% | 17% | 34% | 16% | 8% | 25% | 0% | 2% | 23% | 13% |
| SEATTLE | 0% | 0% | 0% | 25% | 1% | 1% | 73% | 0% | 0% | 2% | 8% |
| WESTPORT | 14% | 3% | 9% | 11% | 19% | 12% | 1% | 2% | 25% | 3% | 12% |
| ILWACO | 2% | 0% | 6% | 8% | 5% | 2% | 0% | 0% | 24% | 1% | 4% |
| OTHER WASHINGTON | 0% | 0% | 0% | 1% | 0% | 0% | 1% | 0% | 0% | 0% | 0% |
| ASTORIA | 7% | 26% | 3% | 10% | 6% | 12% | 0% | 9% | 12% | 2% | 8% |
| GARIBALDI | 0% | 0% | 1% | 1% | 2% | 1% | 0% | 0% | 1% | 0% | 1% |
| NEWPORT | 9% | 13% | 11% | 2% | 11% | 15% | 0% | 0% | 14% | 2% | 8% |
| CHARLESTON | 1% | 14% | 7% | 1% | 8% | 20% | 0% | 0% | 7% | 2% | 5% |
| BROOKINGS | 0% | 5% | 9% | 1% | 4% | 2% | 0% | 0% | 0% | 1% | 2% |
| CRESCENT CITY | 1% | 3% | 3% | 0% | 9% | 4% | 0% | 0% | 1% | 0% | 3% |
| EUREKA | 1% | 12% | 3% | 0% | 6% | 1% | 0% | 0% | 1% | 1% | 3% |
| FORT BRAGG | 0% | 7% | 5% | 2% | 1% | 0% | 0% | 0% | 0% | 5% | 2% |
| BODEGA BAY | 0% | 0% | 1% | 3% | 3% | 0% | 0% | 0% | 0% | 0% | 1% |
| SAN FRANCISCO | 0% | 4% | 2% | 2% | 7% | 1% | 0% | 1% | 2% | 3% | 3% |
| MOSS LANDING | 0% | 2% | 4% | 1% | 0% | 2% | 0% | 10% | 1% | 1% | 2% |
| AVILA | 0% | 0% | 10% | 0% | 0% | 1% | 0% | 0% | 1% | 1% | 1% |
| SANTA BARBARA | 0% | 0% | 3% | 0% | 1% | 8% | 0% | 38% | 2% | 28% | 8% |
| TERMINAL ISLAND | 0% | 0% | 4% | 0% | 0% | 5% | 0% | 40% | 4% | 14% | 7% |
| OCEANSIDE | 0% | 0% | 3% | 0% | 0% | 3% | 0% | 0% | 4% | 12% | 2% |
| OTHER CALIFORNIA | 0% | 0% | 0% | 0% | 0% | 0% | 0% | 0% | 0% | 0% | 0% |
| OFFSHORE | 65% | 0% | 0% | 0% | 0% | 0% | 0% | 0% | 0% | 0% | 6% |
| TOTAL AVG. ANNUAL REVENUE (\$1000s) | \$ 35,310 | \$ 28,577 | \$ 24,017 | \$ 34,482 | \$ 125,570 | \$ 18,685 | \$ 31,614 | \$ 57,663 | \$ 29,502 | \$ 33,262 | \$ 418,683 |

Our ability to quantify fishery economic effects on communities varies across modeling approaches due to differences in the spatial resolution of predicted landings. In some cases the quantitative analyses are at the port or local level; in other cases the analyses provide a rough idea of what gears harvest the catches but we do not attempt to explicitly model fleet dynamics and landings spatially. When we couple these catch projections with recent price data and information about the recent magnitude and distribution of revenues across species groups and port groups (**Tables MS2 and MS3**, taken from PacFIN landings database), we can, in some cases, draw at least qualitative conclusions about relative economic impacts on groups of fishing communities (grouped by port groups) along the coast.

Human Population Growth scenarios are likely to shift the regional flow of fishery revenues to particular ports. The analysis of development of new fisheries for grenadier (*Macrouridae*), white croaker (*Genyonemus lineatus*), and shortbelly rockfish (*Sebastes jordani*) (**Appendix MS4**) predicts sustainable yield coastwide yields and suggests a potential distribution of catches based on the distribution of the respective fish stocks. If catches rose to sustainable yield predictions of 2055, 2000 and 675 metric tons respectively for grenadier, white croaker and shortbelly rockfish this would translate into gross revenues of \$720 thousand, \$2.4 million and \$965 thousand respectively, based on average prices for these species between 2006 and 2010. Grenadier and white croaker are widely distributed along the coast, so we might expect landings and revenues to be spread widely as well, and the economic impacts on any specific community are unlikely to be large. Shortbelly rockfish are more concentrated in central California, and, were new landings to also concentrate there, they might provide a boost to the relatively small fishing communities there. While \$965 thousand is only a small fraction of overall fishery revenues for central California, it represents a significant increase in groundfish revenues (e.g. groundfish revenues for the Bodega Bay, San Francisco and Moss Landing port groups average less than \$6 million a year, **Tables MS2-3**). Increased revenue and catches of forage species (**Appendix MS3**) such as Pacific sardine and mackerel would be expected to accrue mainly to fleets operating out of central and southern California that dominate landings for small pelagics (**Tables MS2-MS3**).

Aspects of the **Conservation Demand scenario** identify ports and regions that could be affected by alterations to salmon harvest and purse seine fisheries. As noted above, central and southern California ports would experience changes in revenue and landings due to declines in forage fish (small pelagic species) harvest. Increased abundance of Chinook salmon associated with removal of the Klamath River dams (**Appendix MS2**) would cause a 42-44% increase in fishery revenue and resulting employment and income in the broader economy of San Francisco, Fort Bragg, Central Oregon, and the Klamath Management Zone (Humboldt and Del Norte Counties in California and Curry County Oregon, **Appendix MS7**). The additional \$8.9 million in gross revenue in these areas generates regional impacts that vary widely by area. For San Francisco, Fort Bragg and Central Oregon, annual impacts (depending on the area) include an additional 69 to 218 jobs, an additional \$1.05 million to \$2.56 million in labor income, and an additional \$2.41 million to \$6.6 million in output. For the Klamath Management Zones, the annual impacts include an additional 11 to 19 jobs, an additional \$0.06 million to \$0.07 million in labor income, and an additional \$0.13 million to \$0.19 million in output. The size of these communities and reliance on fishing might influence the effect on human wellbeing; for instance, after dam removal the largest employment effect was 218 jobs related to the San Francisco fishery, but this may have lower effect on human wellbeing than smaller employment gains in communities more reliant on fishing (e.g. 69 jobs in Fort Bragg).

Explorations of **Status Quo** management related to the evolution of fishery individual quotas point to potential benefits to groundfish fleets, but with an uneven spatial distribution. Catch projections similar to what may be expected under the new individual quota system (**Appendix MS5**) could result in up to \$44 million more in fishery sector revenue. The projections assume constant harvests and would require development of markets that can absorb higher landings, particularly of Dover sole. The projections of revenues and income from this analysis are not spatially specific. However, assuming they accrue to different port group regions in proportion to revenues from the respective gear groups (**Tables 2 and 3**), we can gain a rough idea of how impacts might be distributed. The groundfish trawl fleet, for which revenues are most concentrated in Oregon and Northern California, is projected to see long-run increases in revenues of 34-46%. The fixed gear groundfish fleets which are more broadly dispersed along the West coast see smaller gains of 6-8%. No changes are projected for the shoreside hake fleets as no direct changes in exploitation rate of hake was modeled. Changes in income effects modeled with IO-PAC are proportional to these changes in revenue.

More detailed port-level fleet dynamics under the **Status Quo** scenario's individual quotas (**Appendix MS6**) suggests that fleets (based in particular ports) that have low spatial overlap with bycatch species are most likely to increase effort and landings under an individual quota system. Other fleets that cannot avoid bycatch and cannot stay below quotas are predicted to reduce fishing effort. In these simulations, Moss Landing, Fort Bragg, Eureka, and Coos Bay increase effort and landings, while northern fleets are more likely to cut effort.

Box MS3.

Jacob and colleagues (2012) developed an approach to quantify the resilience and vulnerability of human communities in the Gulf of Mexico. Following Jacob et al. (2012), vulnerability and resilience may be related to:

- Population composition
- Poverty
- Housing characteristics
- Labor force structure
- Natural and technological disaster risk
- Labor force disruptions
- Housing disruptions
- Personal disruptions



Such an approach could be developed for the US West Coast to predict how changes in the marine and coastal economy and social conditions will influence wellbeing. Norman and colleagues' (2007) profiles of 123 fishing communities on the West Coast may be a starting point, detailing each community's demographics, history, housing, infrastructure, and involvement in fisheries.

Photo: Robert K. Brigham, NOAA Photo Library

TRADE-OFFS AMONG ECOSYSTEM COMPONENTS, INCLUDING HUMAN WELL-BEING

Here we focus on trade-offs between ecosystem components of interest for the IEA (**Figure MS1**): ecosystem integrity, protected species, human communities, habitat, and fisheries.

Our narratives related to energy illustrate potential conflicts between the need for electricity generation and other goals related to protected species, fisheries, habitat, and some metrics of human communities. Continued operation of Klamath dams (including hydropower facilities) could have negative impacts on Chinook salmon abundance and fishery economics (**Appendices MS2, MS7**), while development of wave energy sites could negatively impact sturgeon habitat, groundfish fisheries, and shipping (**Appendix MS1**). The spatial analysis illustrates areas of potential tradeoffs, but does not attempt to quantify the magnitude of these.

Most of our quantitative results do not point to stark coast-wide trade-offs between fisheries and conservation goals related to protected species and ecosystem integrity. Fishery catches similar to those currently occurring did not cause large changes in fish food webs, nor did additional harvesting of new low-biomass species (**Appendices MS4, MS5, MS6**). When these trade-offs did occur, for instance when bird and mammal abundance declined due to depletion of forage species (**Appendix MS3**), they were triggered by fishery effort much greater than current levels; such levels of depletion would be illegal under current law or harvest guidelines. Fishery and conservation goals were aligned in the case of Klamath Dam removal (**Appendices MS2, MS7**), albeit with costs incurred by other sectors. Fishery and conservation goals are also aligned in relation to groundfish catch shares, as the modeling predicts increased catches as some target stocks, with concurrent recovery of rockfish (**Appendices MS5, MS6**). Potential conflicts can arise for individual species (e.g. California and Steller sea lions), but this will be highly dependent on whether future fisheries diverge in effort, location, and gear from current practices.

Our spatial ecosystem modeling suggests that when they occur, trade-offs between fisheries and conservation goals (ecosystem integrity and protected species) are likely to be at the local scale and only in particular regions. For instance, individual quota designs that led to coast-wide increases in stocks led to local declines in fishing effort for some northern fleets (**Appendix MS6**). Similarly, harvest of new fishery targets that are sustainable when measured on a stock-wide basis can cause reconfiguration of plankton communities in Central California (**Appendix MS4**).

SYNTHESIS: LESSONS LEARNED

- The scenarios and modeling here illustrate the benefits of identifying the **“leverage points” for management actions**. This means identifying what the full response to a policy decision will be, as it plays through the human and economic portions of the system. Consideration of such leverage points is one strength of the modeling efforts here.
 - a. For instance, quantitative analyses suggest that moderate increases in one “weak stock”, Klamath River Chinook, can lead to large increases in harvest and economic benefits at the broader regional level.
 - b. On the other hand, low quotas of “weak stock” rockfish may not constrain groundfish catches. Instead, enforcement and monitoring of target species quota is more important to overall fleet behavior, revenues, bycatch, and the biological response.
- Models suggested that under most cases, harvests near current levels would not drive extreme trade-offs between fishing and conservation goals. In contrast, we illustrate **other potential trade-offs between electricity demand and shipping, fishing, and conservation of sturgeon**, based on population modeling of Chinook salmon and spatial analysis related to wave energy illustrate potential trade-offs. **Such conflicts between multiple uses in the California Current are likely to**

continue in the future, and scenario planning should therefore consider the full array of drivers and pressures.

- **A full toolbox of modeling approaches was necessary to connect drivers, pressures, and ecosystem response** in the California Current. Approaches included GIS mapping; single-species, food web, and ecosystem models; and economic input/output models. **Gaps exist in our modeling capability related to climate change, protected species, and human wellbeing.** Ongoing efforts will address some of these topics.

DETAILED ANALYSIS OF LESSONS LEARNED

Through preliminary engagement with experts and narrative scenarios we have identified drivers, pressures, and policy considerations that may shape future conditions of the California Current ecosystem. Where possible, we have applied quantitative models that evaluate management options and predict impacts of particular pressures, with the goal of demonstrating the potential to inform future management decisions. Here we present some of the key lessons learned, and surprises, regarding the following: What management actions appear to have large effects, and why? What are key trade-offs, and what modeling approaches reveal them? And what are vulnerabilities of the system that need to be considered further?

“LEVERAGE POINTS” FOR MANAGEMENT ACTIONS

Two analyses related to dam removal and groundfish individual quotas illustrate the need to identify the “leverage points” for management actions. This means identifying what the full response to a policy decision will be, as it plays through the human and economic portions of the system. With dam removal, the economic effects of moderate increases in Klamath River Chinook populations are amplified through much of Oregon and California, as Klamath Chinook are a “weak stock” and constrain fishing for other salmon runs. For groundfish fleets, our modeling argues against the *a priori* assumption that low quotas of “weak stock” rockfish would constrain catches. Instead, enforcement and monitoring of target species quota is more important to overall fleet behavior, revenues, bycatch, and the biological response. Moreover, fleets at times choose to exceed “weak stock” quotas, paying penalties or risking fines to maximize total revenue. Decision making requires understanding which management actions or policies have the largest effect on the human and economic response, and this is one strength of the modeling efforts here.

REVEALING TRADE-OFFS

Given an emphasis on models focused on fishing, we had expected to illustrate strong trade-offs between fishing and conservation goals. However, models suggested that under most cases, harvests near current levels would not drive extreme trade-offs. On the other hand, as discussed above, we illustrate other potential trade-offs between electricity demand and shipping, fishing, and conservation of sturgeon, based on population modeling of Chinook salmon and spatial analysis related to wave energy illustrate potential trade-offs. Such conflicts between multiple uses and pressures in the California Current are likely to continue in the future, and scenario planning should therefore consider the full array of drivers and pressures.

ADVANTAGES OF MODELING APPROACHES

Though scenarios exercises like those here may seem to lend themselves to complicated dynamic models, we found that simple maps were a highly effective tool for identifying trade-offs and conflicts related

to wave energy. Though these analyses do not quantify such trade-offs in detail, they are a first step toward informed decisions. The analysis identified a key axis of uncertainty, the cost of underwater transmission lines, which is likely to dictate the proximity of wave energy facilities to shore. This subsequently determines spatial overlap with gears and species, which are typically confined to certain depth zones. Additionally, the analysis points to the need for comprehensive data sets for each sector – for instance, shipping involves not just the primary shipping lanes but also specific lanes negotiated by tugs and crabbing vessels. Similar map-based analyses have had an immense impact on conservation decisions, for instance allowing tradeoffs between costs and objectives for marine reserves (Leslie et al. 2003) and terrestrial conservation (Carwardine et al. 2008).

We found that each level of model complexity was appropriate for particular questions and scenarios. We applied only one single-species model here (for Chinook salmon), in addition to comparing predictions from published stock assessments (single-species models) to ecosystem model predictions related to groundfish. Where management questions are focused on single species such as Chinook salmon, single-species models allow statistical estimation and capture the uncertainty in predictions. For higher trophic level species for which fishing causes a large portion of total mortality, our ecosystem modeling generally predicted simple, direct responses caused by harvest and bycatch, as would single-species models. The full complexity of the ecosystem and food web models was useful primarily to investigate scenarios involving lower trophic levels, spatial fishery effects, and more drastic increases in fishing rates. Additionally, spatially-explicit ecosystem modeling provided a unified view of fleet dynamics for mixed-species fleets; unlike salmon trollers groundfish fleets base their decisions on harvesting opportunities across many species, and their catches influence population dynamics of many unassessed stocks.

Predictions from the ecosystem model (Atlantis) and food web model (Ecosim) suggest distinct hypotheses regarding energy flow. Both models predict that harvest of one lower trophic level species (e.g. forage fish) will lead to increased abundance of others (e.g. euphausiids or copepods). The two models' predicted effects on predators of these species are consistent in some cases but not others; the divergent predictions are alternate hypotheses that illustrate the uncertainty in system structure and model assumptions. This paired application of modeling approaches illustrates the strength of such comparison: the ability to identify predictions that are robust to model assumptions, to highlight uncertainty in models, and to suggest alternate hypotheses that can be investigated with field data.

Overall, we found that a full toolbox of modeling approaches was necessary to begin to connect drivers, pressures, and ecosystem response in the California Current. We expect that such an approach will be necessary in the future, bringing existing tools and expertise to investigate potential scenarios.

FUTURE DIRECTIONS INDICATED BY PRELIMINARY ENGAGEMENT WITH MANAGERS, SCENARIOS, AND MODELING

The seven modeling analyses above are a first step toward linking pressures to the response of ecosystem attributes in the California Current (**Figure MS1**). However, many key species and processes were identified in the preliminary engagement with managers and other experts (**Section1**) and scenario narratives, but are not included in the quantitative analyses here. In these cases the preliminary engagement with managers and narratives are useful to at least conceptually identify potential drivers, pressures, and management options. At a minimum, this conceptual approach is informative in identifying areas of potential conflict and trade-offs and guiding future quantitative modeling. Below we discuss gaps in our existing modeling capability and avenues for future work related to climate change, protected species, and human wellbeing.

Climate change and ocean acidification were included in the conversations with experts and managers, as well as in our narrative scenarios, but were not the focus of our modeling. Wave energy development could be one response to climate change, but direct impacts might translate into shifts in river and ocean temperatures, rainfall, and freshwater volume and timing. Ocean acidification may cause declines in shelled plankton and benthic species, with indirect effects on predators. In the 2011 IEA Ainsworth and colleagues (2011) projected some aspects of climate change for marine species North Pacific, and Kaplan et al. (2010) considered effects of ocean acidification on food webs. We have not added to these capabilities here, but there are several relevant avenues of research.

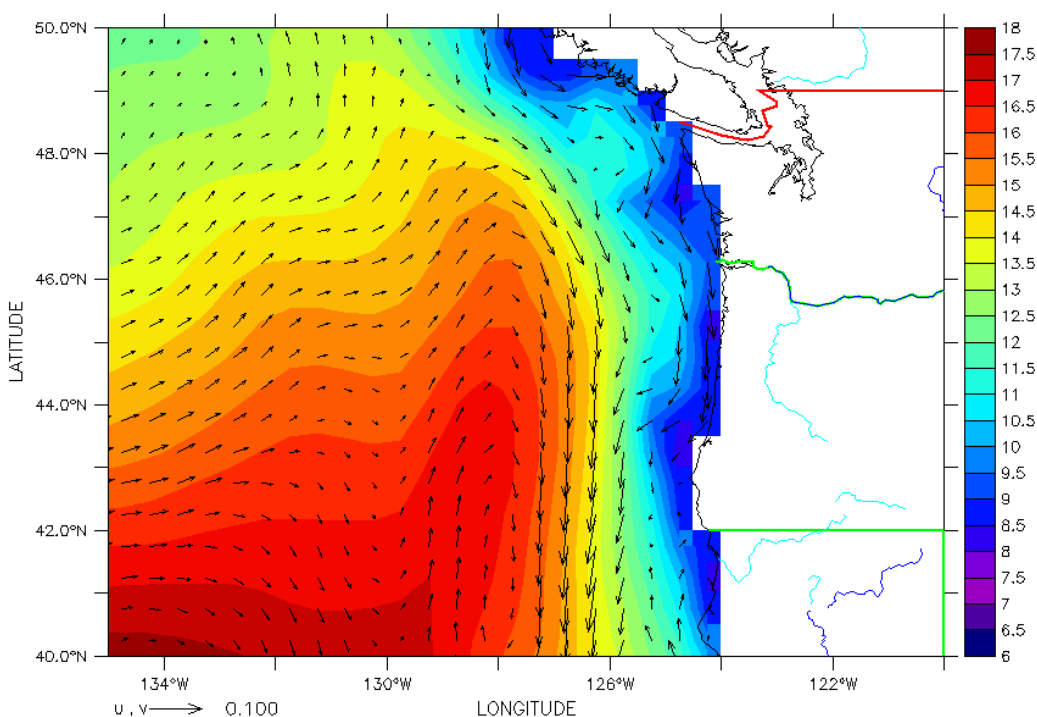
Projections of climate change can be linked to oceanographic models, and this can then be used to predict ecosystem and fishery responses. For instance, the end-to-end modeling framework being developed by Rose and colleagues (**Box MS2**) can link climate models to oceanography, plankton, small pelagic fish, and fishing fleet dynamics. Similarly, Kaplan and colleagues have begun developing the ability to link oceanographic models (Hermann *et al.* 2009) to atmospheric models forced by IPCC scenarios for carbon dioxide emissions. The oceanographic models will be linked to an Atlantis ecosystem model to yield spatial and temporal projections of the effects of global change. Such efforts may reveal local impacts of climate change, for instance at the scale of particular ports, rookeries, or National Marine Sanctuaries. In a related effort that will inform the 2013 IEA, short term climate forecasts are being used to predict metrics of ecological integrity, such as northern copepod abundance (**Ecological Integrity section**) that is positively related to salmon survival rates (Peterson and Schwing 2003) (**Box MS4**) .

Conversations with experts suggest that salmon and other anadromous species are likely to be directly influenced by climate change, due in part to shifting patterns in timing, volume, and temperature of fresh water. Preliminary engagement with experts and managers identified specific runs of salmon hypothesized to be most vulnerable to such shifts. Analyses already exist that predict the response of particular runs of Chinook salmon to climate (**Box MS1**), and these approaches can be applied to additional populations and regions.

Analysis of pressures including shipping, fishing, and energy infrastructure will necessitate additional consideration of protected species, including marine mammals and birds. The food web and ecosystem models typically require very strong, coast-wide impacts on aggregated prey groups to predict large changes in abundance of marine mammals, birds, and other protected species. We have only qualitatively identified the gears that are involved in particular scenarios and that have relatively high bycatch rates of protected species (Jannot et al. 2011). More detailed spatial consideration of hotspots of fishing and protected species (Bertrand et al. 2012) would better illustrate fishing effects on the prey base of these species. Models that predict abundance of protected species as a function of habitat (Redfern *et al.* 2006) could be used to predict current spatial distributions as well as distributions under climate change. These could be combined with dynamic projections of fishing effort to predict entanglement or take. Similarly, more refined scenarios regarding changes in shipping traffic (e.g. related to oil and gas exports or widening of the Panama Canal) could be combined with spatial abundance modeling to inform projections for ship strikes or disturbance.

Box MS4.

Work is underway to provide short term (six to nine month) forecasts of ocean conditions that are testable and relevant to annual management decisions for protected species, fisheries, and ecosystem health. The bottom-up forcing of the California Current ecosystem is predicted using the Climate Forecasting System linked to a ROMS (Regional Ocean Modeling System) with a Nutrient –Phytoplankton-Zooplankton component. The modeling predicts coastal upwelling, currents, mixed layer depths, water temperature, nitrate and oxygen concentrations, pH, and plankton distributions. A recent forecast from the CFS for the region of interest is shown below. Modeling tools and statistical relationships are available to then predict the effects of ocean condition on each of the biological components of the IEA such as protected species (salmon), fisheries (groundfish and coastal pelagic fishes), and ecosystem health.



Forecast average July 2012 temperature and velocity at 25 m

Forecast of temperature (deg C) and velocity (m/s) at 25m depth, from the Climate Forecast System. This forecast of average July 2012 conditions was produced during October 2011.

Our analyses here use modeling approaches to translate scenarios into revenue and economic impacts due to fisheries. We consider port-level or regional impacts on revenue, employment, and income. However, we do not consider the distribution of revenue and income among individuals, nor do we consider non-monetary factors related to human wellbeing. Norman and colleagues (2007) have profiled fishing communities on the west coast, detailing not only fisheries income and involvement but also each community's demographics, history, housing, and infrastructure. These data are useful for considering narrative scenarios of future change in the California Current, and could be combined with factor analysis

similar to Jacob et al. (2012) for quantitative predictions or rankings of resilience and vulnerability of human communities (**Box MS3**).

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APPENDIX MS1. ASSESSING POTENTIAL CONFLICTS WITH WAVE ENERGY GENERATION ALONG THE OREGON COAST

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INTRODUCTION

The process of coastal and marine spatial planning (CMSP) encompasses a broad array of activities that can take place in and affect large marine ecosystems such as the California Current. Assessing potential conflicts and evaluating tradeoffs among the activities is an important part of CMSP. For example, the new U.S. Ocean Policy includes a mandate for coastal and marine spatial planning (CMSP or MSP) to “reduce conflicts among uses and between using and preserving the environment to sustain critical ecological, economic, and cultural services for this and future generations” (White House Council on Environmental Quality 2010).

In this section, we focus on one activity – the generation of wave energy – and how it might conflict with other existing activities in the context of CMSP. Wave energy has the potential to generate substantial amounts of renewable electricity and provides relatively continuous and predictable power, which is advantageous for electrical grid operation. Although the technology has yet to be put into commercial production, wave energy generation costs are likely to fall over time as the underlying technologies develop and the industry expands. Although much uncertainty exists, wave energy may become economically feasible in the near future if fossil fuel energy costs continue to increase.

While waves can provide a source of clean and renewable energy, the facilities for capturing wave energy and producing electricity have a substantial footprint in the marine environment. For this reason, they can conflict with existing ocean uses or conservation strategies for protecting marine species and habitats. Wave energy facilities could hinder fishing opportunities, supplant recreational activities, diminish aesthetic views, and create navigational hazards. The existence and extent of these potential impacts are, of course, site-specific, and so analyzing the possibilities in a framework such as CMSP is desirable.

Evaluating a site’s capacity for wave energy depends on various factors, including wave power resources; the characteristics and costs of wave energy conversion devices; demand and pricing for electricity; availability of transmission networks; constraints on siting of energy conversion facilities; and compatibility with other uses or ecosystem attributes. Economic valuation of harvestable wave energy facilitates the evaluation of tradeoffs between locating a facility in a particular location for energy and the costs of installing, maintaining, and operating the facility at that location. Because technologies for wave energy production are still in the development stage, however, our focus is not on the magnitude of its economic value or even whether the value is positive or not. Instead, our intent is to find the best locations for wave energy facilities, given certain assumptions about the economic parameters that affect those

locations. These locations are then compared to the spatial distribution of existing marine uses, which enables us to (crudely) identify areas where potential conflicts exist.

We use an existing GIS-based decision-support tool to provide spatially explicit information for evaluating wave energy conversion facilities and possible conflicts with other marine uses. The tool is the Wave Energy Model (WEM) of the Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) toolkit (Tallis et al. 2011, Kim *et al.* forthcoming). The wave energy model consists of three parts: 1) assessment of potential wave power based on wave conditions; 2) quantification of harvestable energy using technology specific information about a wave energy conversion device; and 3) assessment of the economic value of a wave energy conversion facility over its life span as a capital investment. We apply this model to the siting of a potential wave energy facility along the coast of Oregon. (Our focus on Oregon is motivated by the availability of wave energy, power infrastructure, fishing, and other data specific to that state.) Below, we first discuss the application of the WEM, and then present the results of the wave energy facility analysis. Finally, we illustrate the potential for conflicts with other marine uses through a series of graphics.

METHODS AND DATA

WAVE ENERGY FACILITY LOCATIONS

Our analysis of wave energy production focuses on coastal Oregon, in an area defined by a north and south border (46° and 42°, respectively) and an east and west border defined by water depth (200m and 40m, respectively). The choice of water depths roughly bounds the range in which the wave energy device we chose (Pelamis) can operate (Pelamis Wave Power Ltd. 2010.). We configured a wave energy facility based on previous work by the Electric Power Research Institute (Previsic 2004b), which analyzed the system level design, performance, and cost of a commercial size offshore wave power plant installed off the coast of Oregon using the Pelamis device. Our configuration for an individual wave energy facility consists of four sets of 45 devices, the facilities arrayed in a north-south direction and creating a footprint 12 km long and 2 km wide. In the analysis below, we consider a set of three facilities, with each facility connected to the Bonneville Power Administration power grid at distinct locations along the Oregon coast.

As noted above, we used the InVEST WEM tool to analyze the potential electricity production and net economic value of this system of wave energy facilities. The WEM tool uses wave and water depth information to assess the potential energy that can be captured by wave energy devices. By choosing a particular device, the WEM tool can then quantify the captured wave energy and electricity production for particular locations. The economic value of energy production is estimated based on the economic costs (capital, operating, and maintenance) of the device and the transmission of the power. The location with the maximum net economic value is what we term the optimal location for the wave energy facility.

Specifically, the WEM tool uses the following input data:

- Water depth
- Wave height and power
- Performance and costs of specific wave energy conversion devices
- Electricity prices and discount rate
- Transmission line landing and power grid connection points

Table 1 lists the types and sources of data we used that are default choices for the WEM tool (version 2.2.2). For other data inputs, we chose particular values based on factors particular to Oregon or for other reasons, listed in Table 2.

Obtaining accurate input data and parameters for the economic valuation portion of the model is a significant challenge because there have been no commercial-scale wave energy facilities implemented to date. These economic parameters determine whether a wave energy facility will be economically viable – that is, whether the net present value of its construction, operation, and maintenance will be greater than zero. Of these economic parameters, however, only variation in the level of the underwater transmission line costs affects the optimal location of a wave energy facility, along with the choice of landing and power grid connection points. In our analysis, we considered three possible levels of transmission costs, which we describe as low cost, medium cost, and high cost scenarios (Table 2). Because the potential power grid connection points are largely determined by the current Bonneville Power Administration’s transmission system, we use only one set of connection points (Table 2).

EXISTING MARINE USES

We considered three sets of existing marine uses and examined how they might conflict with the optimal locations of the wave energy facilities. The existing marine uses were 1) fishing; 2) transportation and utilities; and 3) marine conservation areas (Table 3).

For fishing, we used two sources of information to locate areas along the Oregon coast where fishing effort is present and how the value of fishing varies spatially. The first source (described in Appendix A) documents fishing effort along the coast of Oregon for three different commercial fleets, distinguished by gear type (bottom trawl, at-sea hake midwater trawl and fixed gear), that could be expected to occur within each of the nine proposed wave farm sites. Fishing effort was represented on either 10 km (bottom trawl fleet [herein trawl] and at-sea Pacific hake (*Merluccius productus*) midwater trawl [herein hake] fleet) or 20 km (fixed gear fleet [herein fixed]) grids. We used data from 2002 – 2009 that were provided by the At-sea Hake Observer Program (A-SHOP) and the West Coast Groundfish Observer Program (WCGOP) under NOAA’s Northwest Fisheries Science Center, Fishery Resource Analysis and Monitoring (FRAM) Division.

Commercial fishing effort data are subject to restrictions that preserve confidentiality as required under the Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006. As such, data cannot be presented to the general public unless it represents information from three or more vessels. Therefore, we ran all of our analyses using gridcells that represented the efforts of three or more vessels, and gridcells in the overlap maps that contained data from two or fewer vessels are not displayed.

The second source of information for fishing (Steinback et al. 2010) uses the results of fisherman surveys and, in some cases, harvest data to illustrate how the use and value of fisheries vary spatially.¹ Steinback et al. (2010) collected information from commercial, charter, and recreational fisheries for several Oregon ports (Table 4). The individual sector results were normalized and then aggregated for each individual port. The results illustrate how the use and value of fishing effort in the aggregate varies spatially for a given port, but comparisons across ports are not possible.

For transportation, we considered two types of shipping corridors: 1) shipping lanes as recorded on NOAA's Electronic Navigation Charts (NOAA 2011b), and 2) lanes established for tug and barge traffic under an ongoing agreement between tug and barge operators and crab fisherman managed by the Washington Sea Grant (Washington Sea Grant 2010). For utilities, we considered submarine cables as recorded on NOAA's Electronic Navigation Charts (NOAA 2012), as these cables could conflict with the location of moorings for a wave energy facility.

Finally, we considered two types of marine conservation areas: 1) critical habitat designated under the Endangered Species Act (ESA), and 2) essential fish habitat conservation areas designated under the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA). For critical habitat, designation of an area requires federal agencies or other parties with federal permits or licenses to avoid adversely modifying that habitat. Agencies that have activities or that issue such permits or licenses are required to consult with the National Marine Fisheries Service to ensure that these actions do not have such adverse effects. Critical habitat for green sturgeon (*Acipenser medirostris*) has been designated along the Oregon coast (as well as elsewhere along the Washington and California coasts), and so we considered that designation for our analysis (NOAA 2009). Essential fish habitat conservation areas have been designated for Pacific groundfish along the Oregon coast. These areas apply to several types of fishing gear and impose various types of constraints. For our analysis, the relevant areas are ones that prohibit fishing with bottom trawl gear (NOAA 2006).

RESULTS

Based on the wave energy facility configuration and the values in Tables 1 and 2 for the WEM InVEST tool, we identified three sets of optimal locations, depending on the cost scenario (Figure 1). Across all connection points, the optimal locations for an individual wave energy facility ranges between 13.1 and 70.4 kms offshore, with facility locations farther from shore when a lower transmission cost is assumed (Table 5). The average distance for the three facilities in each scenario is 16.1, 31.2, and 55.5 kms for the high, medium, and low cost scenario, respectively. The average energy captured per device also increases as lower transmission costs are assumed, which corresponds to the higher wave energy potential further offshore

¹ Using electronic and paper nautical charts of the area, fishermen were asked to identify, by fishery, the maximum extent north, south, east, and west that they would forage or target a species. They were then asked to identify, within this maximum forage area, which areas are of critical economic importance, over their cumulative fishing experience, and to rank these using a weighted percentage—an imaginary “bag of 100 pennies” that they distribute over the fishing grounds. All maps based on Steinback et al. (2010) are considered “social” or stated importance maps, as they give equal weighting to each fishery in a sector and equal weighting to each sector when combined together. Port-level maps should not be combined with each other, and an overlap in fishing areas between maps should not be considered additive.

along the Oregon coast (Table 5). The total MWh/yr captured by all three facilities would be 3564, 3462, and 3324 MWh/yr for the low, medium, and high cost scenarios, respectively.

For fishing, the focus on particular fleets shows possible conflicts with the at-sea hake midwater trawl and bottom trawl fleets (Figures 2a and 2b). For the fixed gear groundfish fleet, the problem of missing data due to confidentiality restrictions limits any conclusions that can be drawn (Figure 2c). For the at-sea hake midwater and bottom trawl fleets combined, the medium cost scenario presents the strongest potential conflict in terms of a wave energy facility interfering with groundfish harvesting (Table 6).

Using the data on more general fishing location choices and values for specific ports, there is (unsurprisingly) a stronger possibility of conflict for ports that are close or the same as the points chosen for power grid connections (Figures 3a – 3g, esp. 3a and 3c). As has been noted, however, the methods used for constructing the underlying port-specific fishing datasets make comparisons across ports problematic. Nevertheless, in almost all cases, the potential for conflict with a particular port's fishing areas is strongest for the high cost scenario, in which wave energy facilities are closest to shore. An interesting exception is the port of Florence, where the potential for conflict is strongest for the low cost scenario due to a highly valued fishing area for that port that is relatively far from shore (Figure 3d).

For shipping and towing lanes, there is a strong potential conflict with the tugboat and barge tow lanes established off shore of all three connection points for the high cost scenario (Figure 4a), while conflicts with shipping lanes are less likely (Figure 4b). For submarine cables, there is a potential conflict with cables connected to the Tillamook area (Figure 4c). As noted above, however, the presence and extent of this conflict is speculative, as it can only be based on the mooring requirements for the wave energy device and not on the spatial location alone of the wave energy facility.

Finally, the locations of some wave energy facilities overlap green sturgeon critical habitat (Figure 5a), with each of the three facilities for the high cost scenario overlapping. This overlap could trigger requirements for federal agencies such as the Federal Energy Regulatory Commission to consult with NOAA Fisheries before licensing a wave energy facility. For the Pacific groundfish conservation areas, there is an overlap for two of the three low cost scenario facilities. Because these areas are currently managed as closures to harvest for certain groundfish fleets, the exact nature of any potential conflict is uncertain.

DISCUSSION

Using an existing GIS-based tool for evaluating potential locations of wave energy facilities, we have demonstrated how potential conflicts with existing marine uses can be identified. The variety of methods used by various data sources to measure the intensity and value of these uses makes a comparison across uses or an aggregation of the conflicts problematic. Nevertheless, a simple set of spatial representations can present planners with a screening tool, identifying areas where a more refined investigation is worthwhile.

The InVEST WEM tool has the capability of quantifying the consequences, in terms of captured wave energy and economic value, of moving the wave energy facilities to alternate locations, changing the land connection points, and so forth. Coupled with similar quantitative measures of the change in a facility's impact on existing marine uses, this capability would allow for an extended assessment of the potential tradeoffs between wave energy production and those other uses. This would provide an important analysis for CMSP.

Several deficits prevent us from exploring this issue, however. As noted above, the data sources for the existing marine uses are limited in how they spatially measure the intensity and value of those uses.

(None of the existing uses are assessed in terms of economic value.) While some conclusions can be drawn for a particular use that certain locations are likely to create “more” or “less” of a conflict, little more than that can be said. Second, for some uses, a conflict or lack of one is inferred from the presence or absence of that use in a particular location. Much more must be understood about the real nature of conflicts and the ability of various uses, including wave energy production, to coexist spatially before a viable tradeoff analysis could be conducted. And finally, many of the other uses can choose alternate locations in response to a spatial conflict. An understanding of how such choices are made and the availability and value of alternate locations would be needed, again, for a robust tradeoff analysis.

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APPENDIX A

PACIFIC GROUND FISH HARVEST EFFORT FOR THREE FISHING FLEETS

METHODS

We overlaid two different geospatial data layer types for these analyses: potential wave farm sites and cumulative observed groundfish fishery effort. We quantified the amount of fishing effort by three different commercial fleets by gear type (bottom trawl, at-sea hake midwater trawl and fixed gear) that could be expected to occur within each of the nine proposed wave farm sites.

GROUND FISH FISHERY DATA

Fishing effort was represented on either 10 km (bottom trawl fleet [herein trawl] and at-sea Pacific hake (*Merluccius productus*) midwater trawl [herein hake] fleet) or 20 km (fixed gear fleet [herein fixed]) grids. We used data from 2002 – 2009 that were provided by the At-sea Hake Observer Program (A-SHOP) and the West Coast Groundfish Observer Program (WCGOP) under NOAA's Northwest Fisheries Science Center, Fishery Resource Analysis and Monitoring (FRAM) Division.

Commercial fishing effort data are subject to restrictions that preserve confidentiality as required under the Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006. As such, data cannot be presented to the general public unless it represents information from three or more vessels. Therefore, we ran all of our analyses using gridcells that represented the efforts of three or more vessels, and gridcells in the overlap maps that contained data from two or fewer vessels are not displayed.

At-sea hake midwater trawl fishing effort was collected directly by the A-SHOP (National Oceanic and Atmospheric Administration (NOAA) 2011). The A-SHOP collects information on total catch (fish discarded and retained) from all vessels that process Pacific hake at-sea. All data were collected according to standard protocols and data quality control established by the A-SHOP (National Oceanic and Atmospheric Administration (NOAA) 2011).

Bottom trawl fishing effort (National Oceanic and Atmospheric Administration (NOAA) 2010) was derived from fleet-wide logbook data submitted by state agencies to the Pacific Fisheries Information Network (PacFIN) regional database, maintained by the Pacific States Marine Fisheries Commission (PSMFC). A common-format logbook is used by Washington, Oregon, and California. Trawl logbook data are regularly used in analyses of the bottom trawl groundfish fishery observed by the WCGOP.

For both the trawl and hake spatial data, a trawl towline model (line drawn from the start to end location of a trawl tow) was used to allocate data to the 10 x 10 km grid cells for calculation of cumulative fishing effort (hours that gear was deployed in the water).

Fixed gear fishing effort was expressed as the cumulative number of sets, as opposed to the time gear was in the water. These data were collected directly by the WCGOP from the following commercial groundfish fixed gear sectors: limited entry sablefish primary (target – sablefish), limited entry non-sablefish endorsed (target – sablefish/groundfish), open access fixed gear (target – groundfish), and Oregon and California state-permitted nearshore fixed gear (target – nearshore groundfish). Both the observed fixed gear set (start location of fishing) and haul (location of gear retrieval) were assigned to 20 x 20 km grid cells for calculation.

The fishing effort associated with each fixed gear fishing event was divided equally between the set and haul locations.

For the hake and trawl fleets, the data represents total fishing effort (100%). All at-sea hake vessels (catcher-processors and motherships) over 125 feet are required to carry two observers, while vessels under 125 feet carry one. PacFIN fleet-wide logbook data are assumed to represent the entire bottom trawl fleet for our analysis. However, all fishing operations may not necessarily be recorded in logbooks and logbook submission may not be complete. Observer data did not capture 100% of the fishing effort for the fixed gear fleet, so we calculated the proportion (C) of the fleet that was represented by the observer data:

$$C = \sum_{s=1}^5 \left(\frac{t_s}{T} \times \frac{w_{s(obs)}}{W_{s(land)}} \right)$$

s corresponded to each of the five sectors, t was the total time (in hours) a given sector was observed with gear in the water, T was the total time (in hours) all five of the sectors were observed with gear in the water, w was the total retained weight of target fish species caught on vessels with observers present (reported by sector) and W was the total landed weight of target fish species by all vessels (reported by sector).

Catch data are reported on an annual basis, so we ran the calculation across all years (2002-2009) by multiplying the data reported for each sector by the proportion that that sector represented over the entire study area. The observed portion of overall fixed gear effort varied by coverage level in each sector (Table 1). Since all fishing operations were not observed, neither the maps nor the data can be used to characterize the fishery completely.

OVERLAP WITH GROUND FISH FISHERY

We used ESRI ArcGIS (v. 9.3) to run our spatial analyses. We calculated the expected cumulative fishing effort for each of the nine proposed wave farm sites by intersecting the rectangular polygons representing each site with each of the three different commercial fishing fleet grids. Using the attribute information from the intersected polygons, we converted cumulative fishing effort (hours/10 km gridcell for the at-sea hake and bottom trawl fleets; sets/20 km gridcell for the fixed gear fleet) to cumulative effort per km². We then multiplied the effort per unit area by the total area for each proposed wave farm (30 km²), which yielded an estimate of the cumulative effort for each wave farm site.

Table 1. InVEST WEM data input default values

| Category | Item | Source |
|--------------------------------|---|----------------------------|
| Water depth | Water depth [m] | Amante and Eakins (2009) |
| Wave power | Wave height [m] | NOAA (2011c) |
| | Peak wave period [sec] | |
| Wave energy device performance | Captured wave energy for a given seastate condition defined by wave height and wave period [kW] | Previsic (2004a) |
| | Maximum capacity of device [kW] | |
| | Upper limit of wave height for device operation [m] | |
| | Upper limit of wave period for device operation [sec] | |
| Wave energy device costs | Capital cost per installed kW [\$/kW]. | Dunnett and Wallace (2009) |
| | Cost of mooring lines [\$ per m] | |
| | Cost of overland transmission line [\$ per km] | |
| | Operating & maintenance cost [\$ per kWh] | |

Table 2. InVEST WEM data input choices

| Data Input | Description |
|--|---|
| Area of Interest | North and south boundaries set at 46 degrees and 42 degrees. East and west boundaries determined by water depth, 40 and 200 meters respectively. |
| Wave energy device | Pelamis |
| Wave energy facility | 4 cluster of 45 devices |
| Cost of underwater transmission line [\$ per km] | We chose three levels of cost: Low cost scenario = \$100,000 per km Medium cost scenario = \$250,000 per km High cost scenario = \$500,000 per km Based on figures from Dunnett and Wallace (2009). |
| Landing and power grid connection | Tillamook, Toledo, and Tahkentic substation, Bonneville Power Administration, Transmission Asset Network. We chose these power grid connection points (and associated landing points) based on substation transformer capacity, not including any costs of upgrading local infrastructure to accommodate wave energy production. Source: Bonneville Power Administration (2012) |
| Price of electricity [\$ per kWh] | 5¢ / kWh. Source: U.S. Energy Information Administration (2011) |
| Discount rate | 5% |

Table 3. Existing marine uses

| Marine Use | Activity Considered | Source |
|--------------------|---|-----------------------------|
| Fishing | Fishing effort for at-sea hake midwater trawl, bottom trawl and fixed gear, fishing effort (cumulative hours fishing by 10km cell, 2002-09) | See Appendix A |
| | Fisheries Uses and Values, selected Oregon ports | Steinback et al. (2010) |
| Transportation | Shipping lanes | NOAA (2011b) |
| | Crabber-Tugboat tow lanes | Washington Sea Grant (2010) |
| Utilities | Submarine cables | NOAA (2012) |
| Conservation areas | Green sturgeon critical habitat | NOAA (2009) |
| | Pacific groundfish Essential Fish Habitat conservation areas | NOAA (2006) |

Table 4. Fisheries uses and values (Steinback et al. 2010)

| Port Group | Commercial | Charter | Recreational |
|---|--|---|---|
| Garibaldi | Dungeness Crab-Trap, Salmon-Troll, Rockfish-Fixed Gear, Shelf Bottom Trawl | N/A | Dungeness Crab, Pacific Halibut, Rockfish, Salmon |
| Depoe Bay | Dungeness Crab-Trap, Salmon-Troll, Rockfish-Fixed Gear, Urchin-Dive | Dungeness Crab, Pacific Halibut, Rockfish, Salmon | Dungeness Crab, Pacific Halibut, Rockfish, Salmon |
| Newport | Dungeness Crab-Trap, Salmon-Troll, Rockfish-Fixed Gear, Shelf Bottom Trawl | Dungeness Crab, Pacific Halibut, Rockfish, Salmon | Dungeness Crab, Flatfish, Pacific Halibut, Rockfish, Salmon |
| Florence | Dungeness Crab-Trap, Salmon-Troll | Dungeness Crab, Pacific Halibut, Rockfish, Salmon | Dungeness Crab, Pacific Halibut, Rockfish, Salmon |
| SOORC (Charleston, Coos Bay, Bandon, Winchester Bay, Reedsport) | Dungeness Crab-Trap, Salmon-Troll, Rockfish-Fixed Gear, Shelf Bottom Trawl | Dungeness Crab, Pacific Halibut, Rockfish, Salmon | Dungeness Crab, Pacific Halibut, Rockfish, Salmon |
| Port Orford | Dungeness Crab-Trap, Salmon-Troll, Rockfish-Fixed Gear, Urchin-Dive | N/A | N/A |
| Gold Beach/Brookings | Dungeness Crab-Trap, Salmon-Troll, Rockfish-Fixed Gear, Urchin-Dive | Dungeness Crab, Pacific Halibut, Rockfish, Salmon | Dungeness Crab, Pacific Halibut, Rockfish, Salmon |

Table 5. Optimal wave energy facilities

| Cost scenario | Power Grid Connection Point | Distance of facility from landing point (km) | Average energy captured per device (kWh/yr) |
|----------------------|------------------------------------|---|--|
| Low | Tillamook | 58.6 | 2,198 |
| | Toldeo | 37.4 | 2,181 |
| | Tahkenitch | 70.4 | 2,221 |
| Medium | Tillamook | 23.5 | 2,126 |
| | Toldeo | 33.6 | 2,165 |
| | Tahkenitch | 36.5 | 2,121 |
| High | Tillamook | 13.1 | 2,076 |
| | Toldeo | 16.4 | 2,061 |
| | Tahkenitch | 18.8 | 2,019 |

Table 6. Potential groundfish-wave energy conflicts

| Cost Scenario | Power Grid Connection Point | Cumulative Duration 2002-2009 (hrs/km²) | | | Cumulative Sets 2002-2009 (no./km²) |
|----------------------|------------------------------------|---|--------------------|---------------------|---|
| | | Bottom trawl | At-sea hake | Trawl + hake | Fixed gear |
| Low | Tillamook | 0.30 | conf. | 0.30 | 0.11 |
| | Toldeo | 1.65 | conf. | 1.65 | 0.18 |
| | Tahkenitch | 0.72 | 0.45 | 1.17 | 0.11 |
| Medium | Tillamook | 1.28 | 0.23 | 1.51 | conf. |
| | Toldeo | 3.77 | 0.43 | 4.20 | 0.20 |
| | Tahkenitch | 1.80 | 0.52 | 2.32 | conf. |
| High | Tillamook | 1.01 | conf. | 1.01 | 0.36 |
| | Toldeo | 1.01 | conf. | 1.01 | 0.00 |
| | Tahkenitch | 0.32 | conf. | 0.32 | conf. |

Figure 1
Optimal locations for wave energy facilities

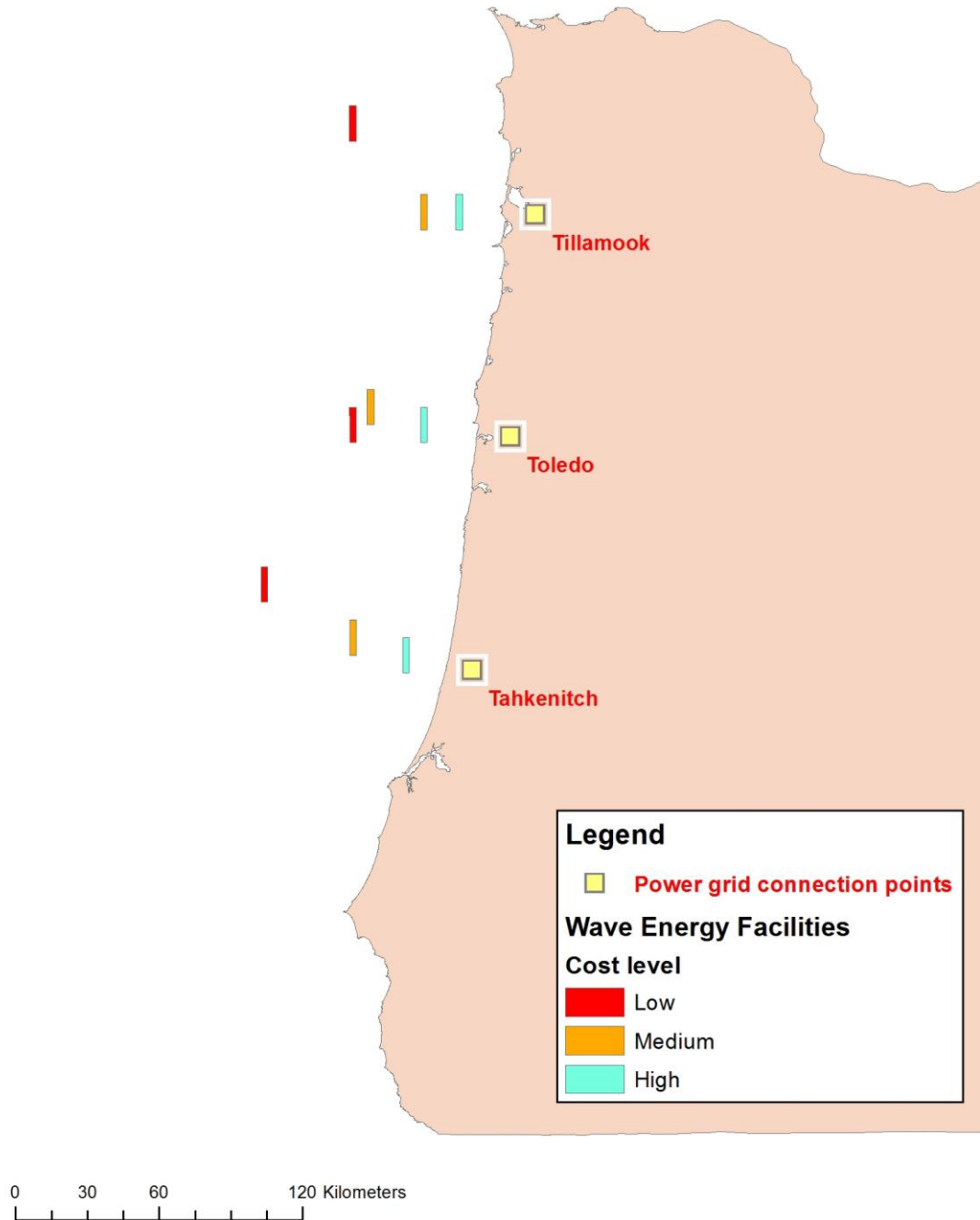
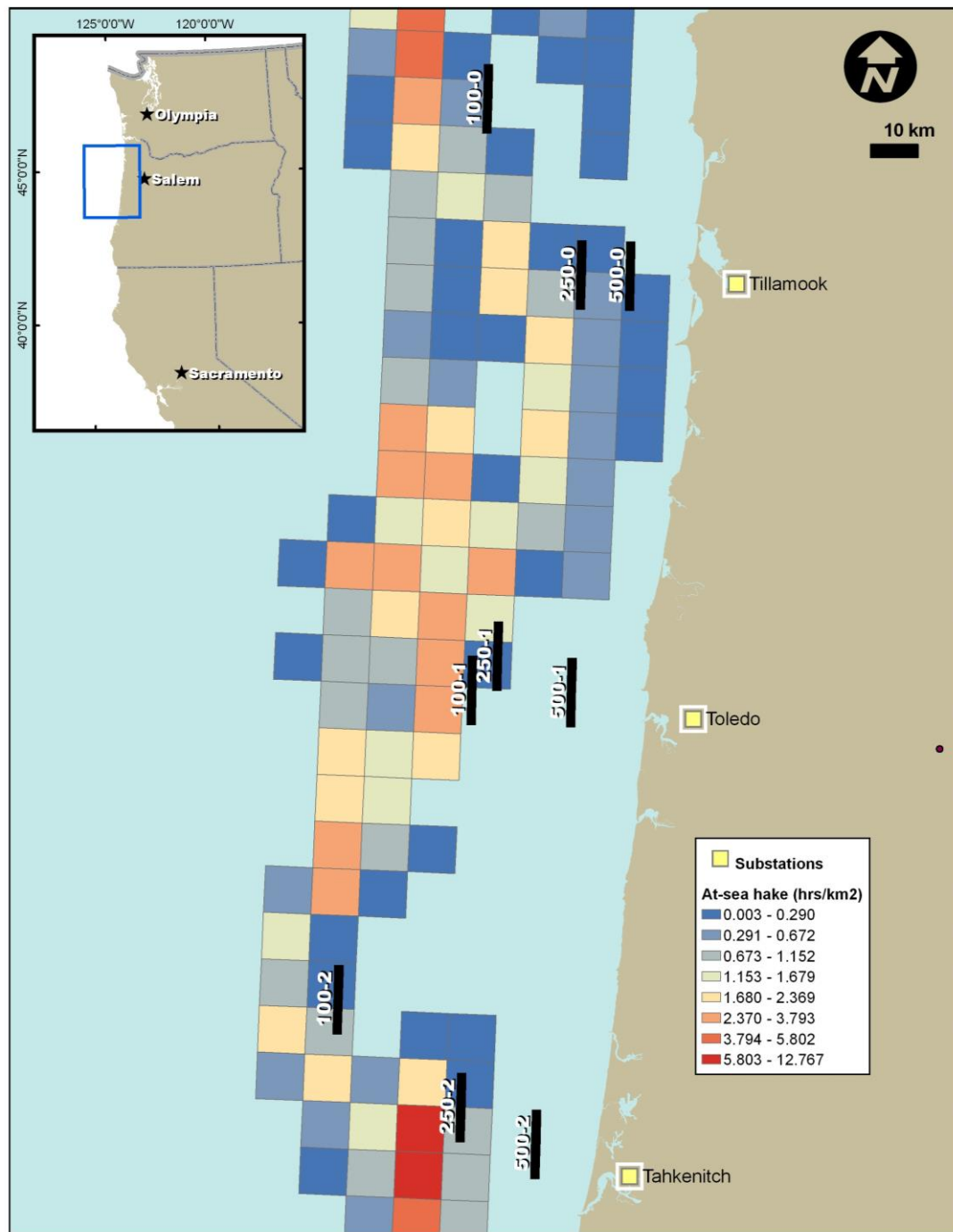


Figure 1: Using the Wave Energy Model InVEST tool, we identified three sets of optimal locations, depending on the cost scenario. The location with the maximum net economic value is what we term the optimal location for the wave energy facility.

Figure2a
Groundfish at-sea hake fishing effort



Figures 2a - 2c: We used data from 2002 - 2009 to document fishing effort along the coast of Oregon for three different commercial fleets, distinguished by gear type (bottom trawl, at-sea hake midwater trawl, and fixed gear), that could be expected to occur within each of the proposed wave farm sites. These data reveal possible conflicts with the at-sea hake midwater trawl (2a) and bottom trawl fleets (2b), while for the fixed gear groundfish fleet, the problem of missing data due to confidentiality restrictions limits any conclusions that can be drawn (2c).

Figure2b
Groundfish bottom trawl fishing effort

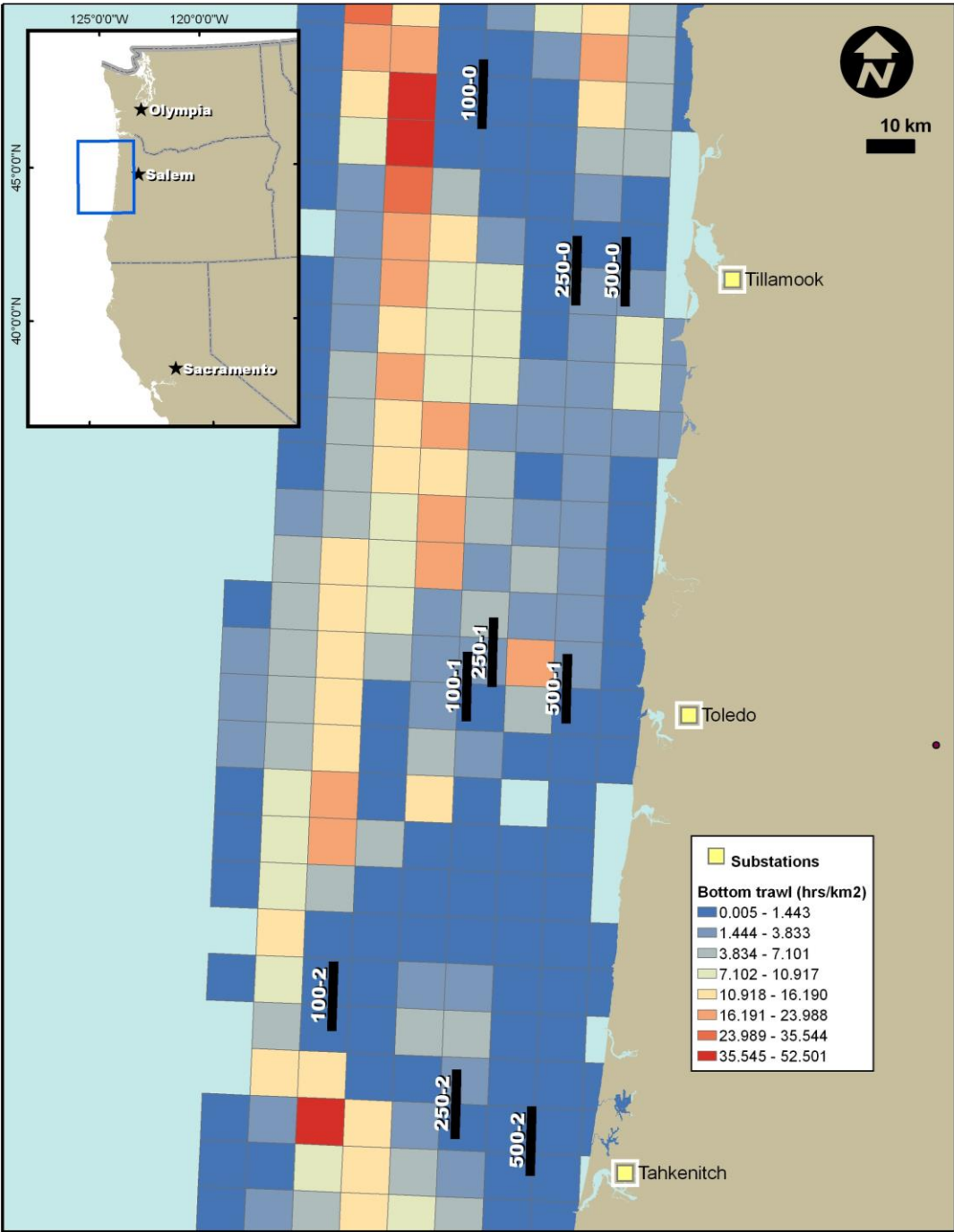


Figure2c
Groundfish fixed gear fishing effort

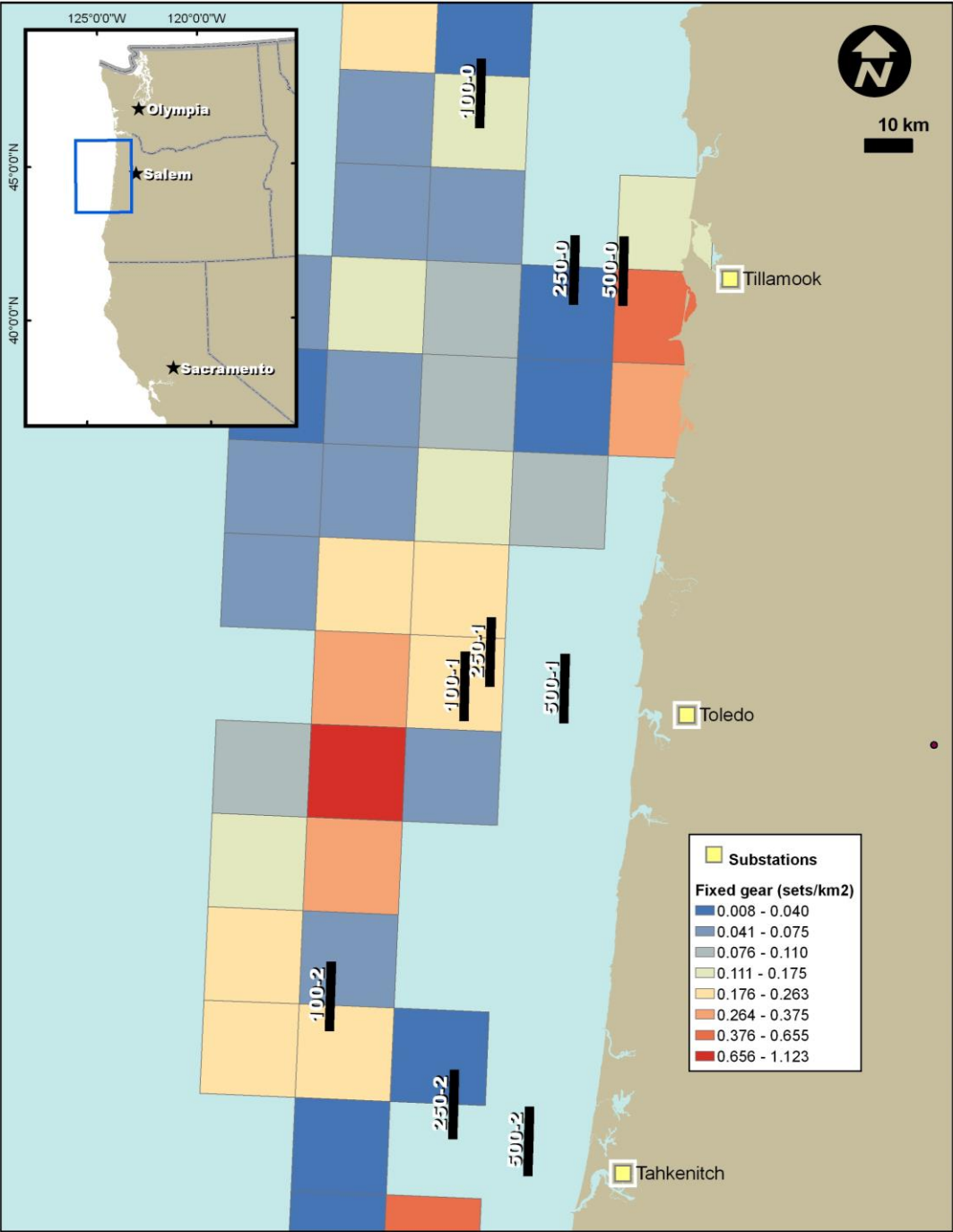


Figure3a
Garibaldi Combined Value Fishing Grounds

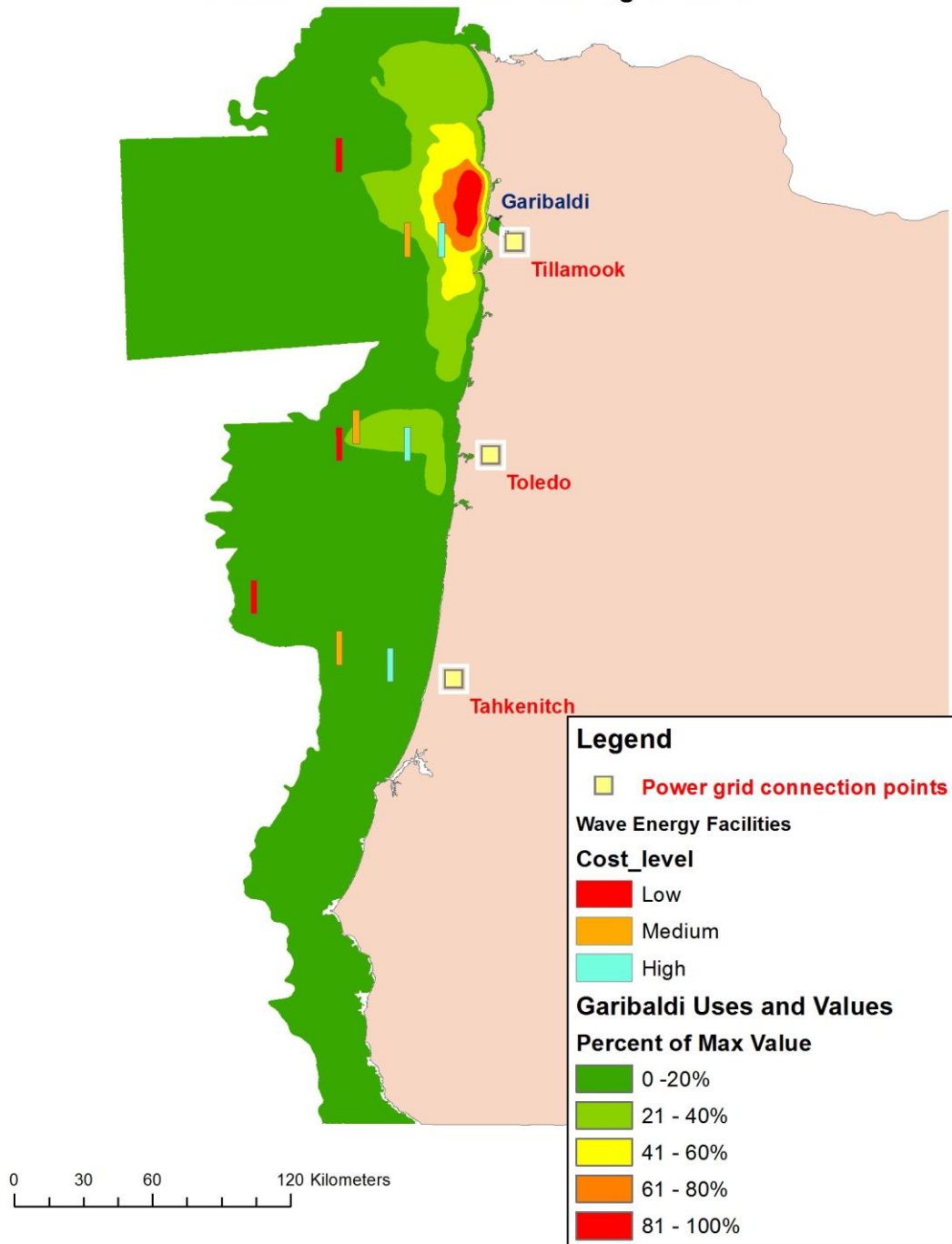


Figure 3a - 3g: Using data collected by Steinback et al. (2010) from commercial, charter, and recreational fisheries for several Oregon ports, there is a stronger possibility of conflict for ports that are close or the same as the points chosen for power grid connections (3a - 3g).

Figure3b
Depoe Bay Combined Value Fishing Grounds

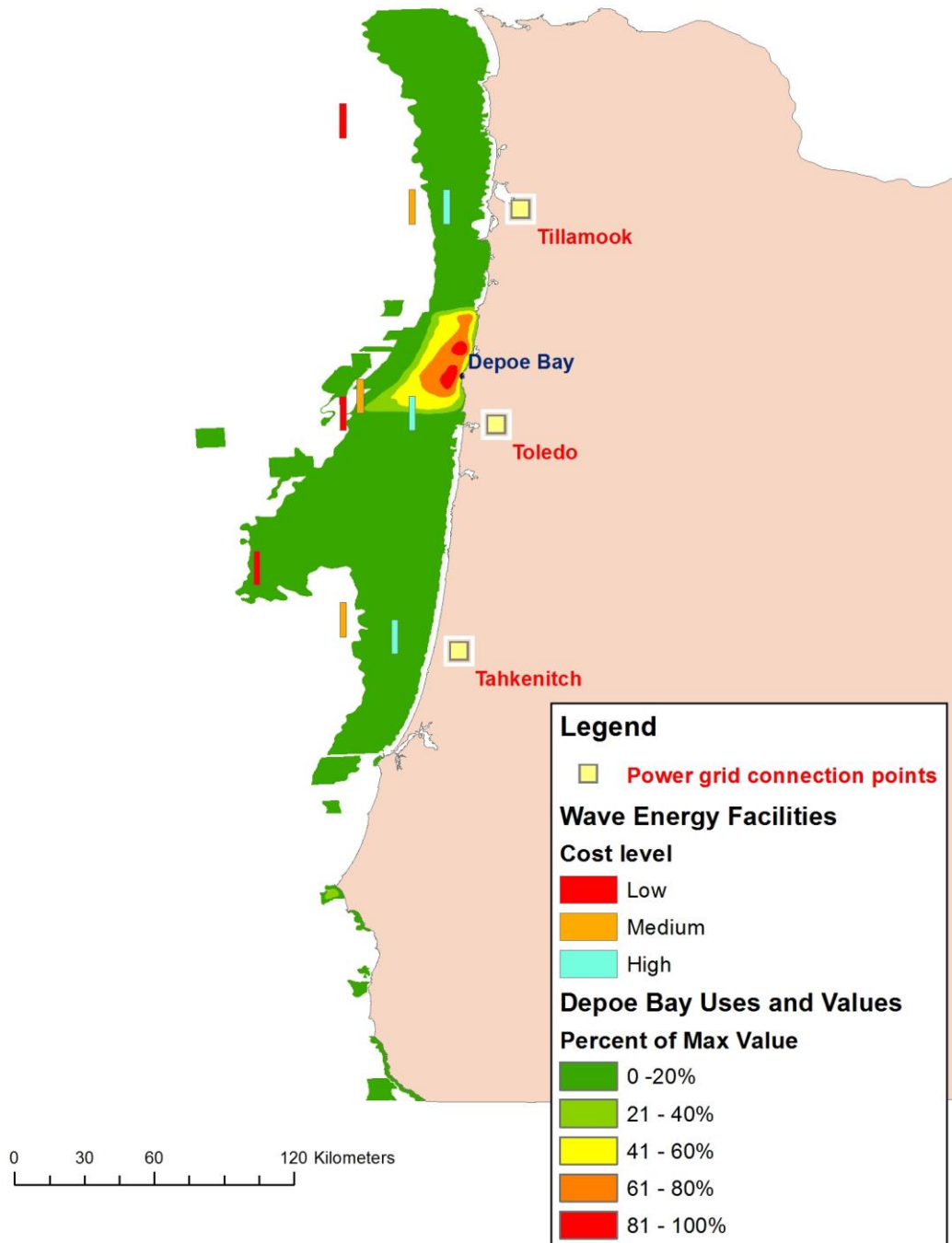


Figure3c
Newport Combined Value Fishing Grounds

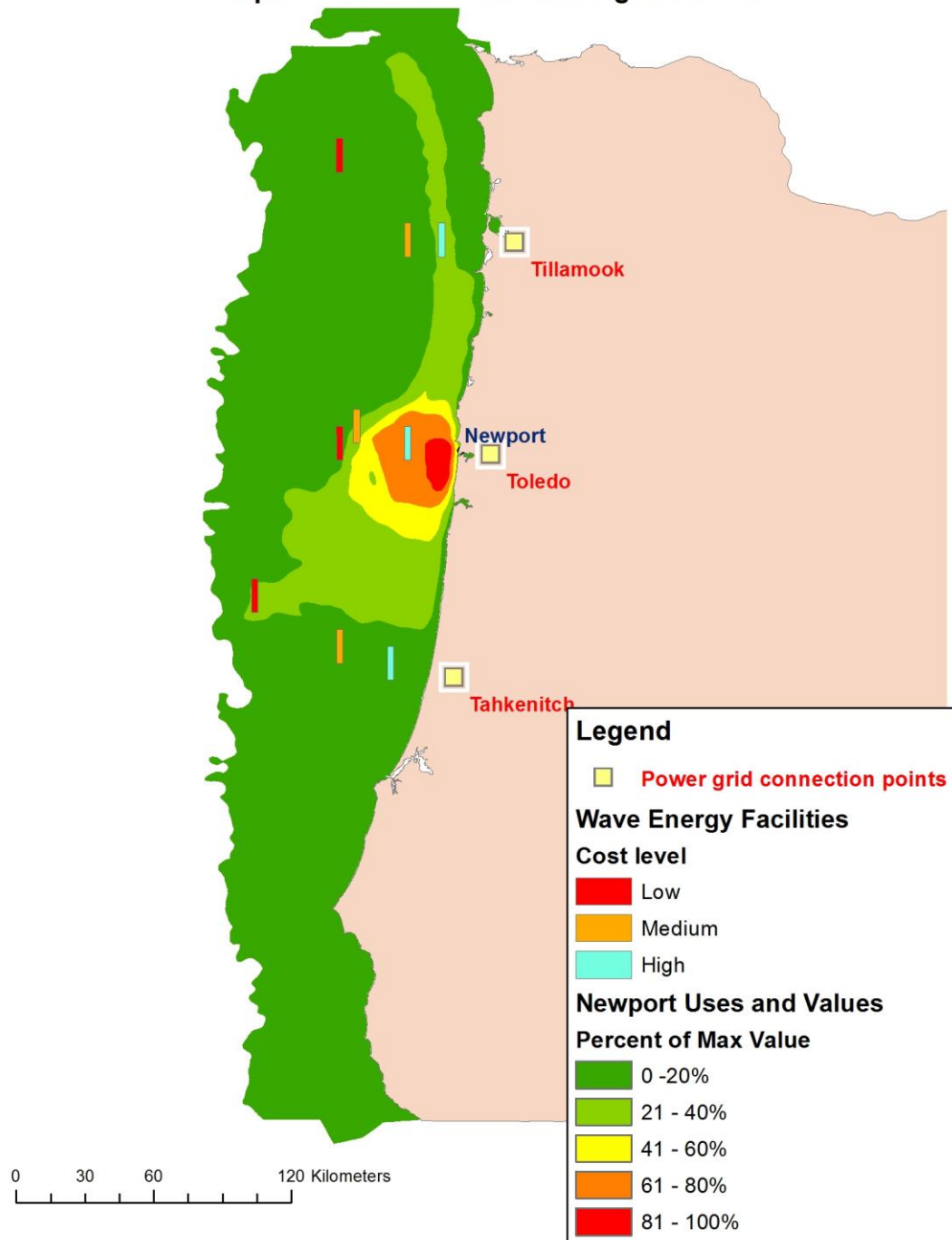


Figure3d
Florence Combined Value Fishing Grounds

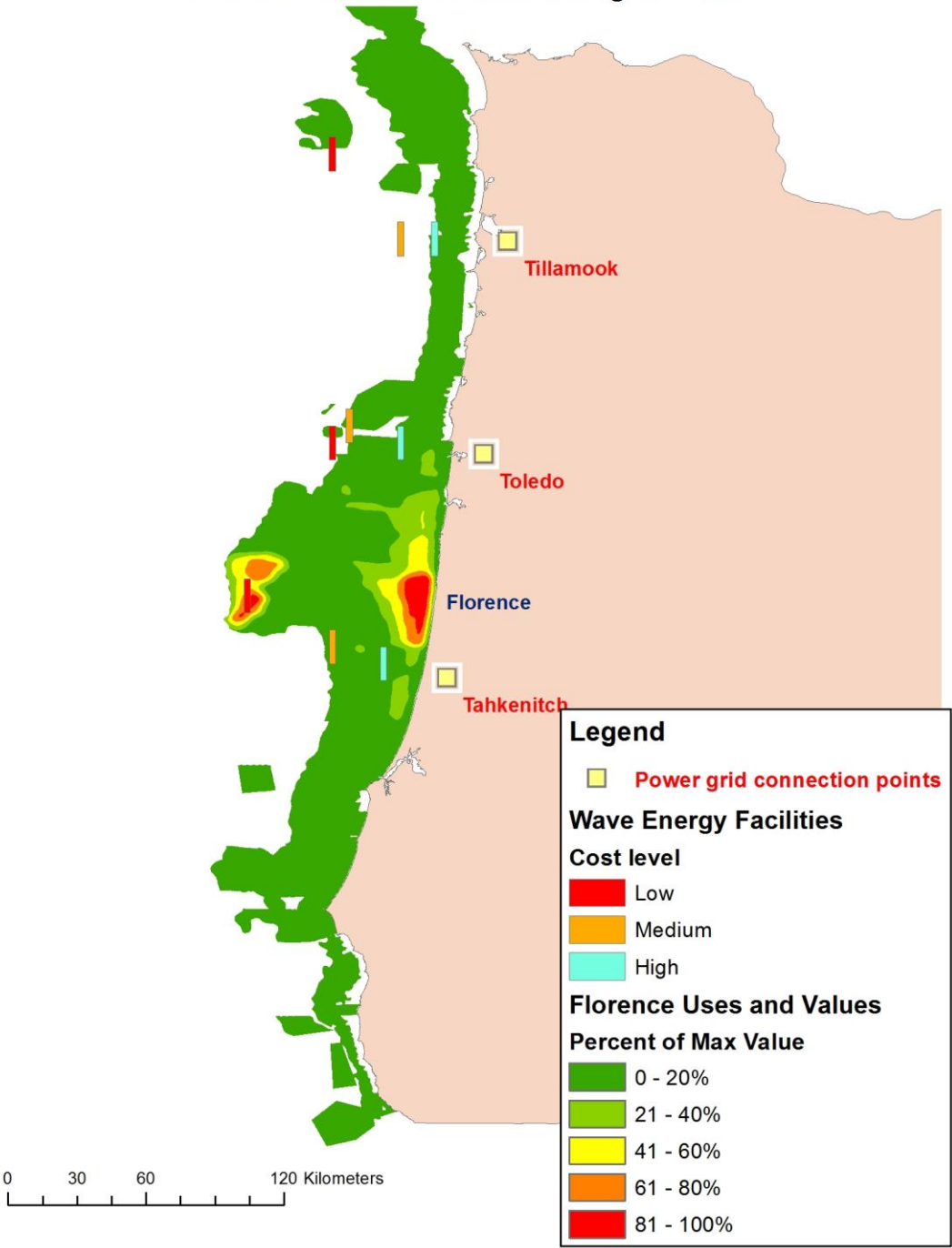


Figure3e
SOORC Combined Value Fishing Grounds

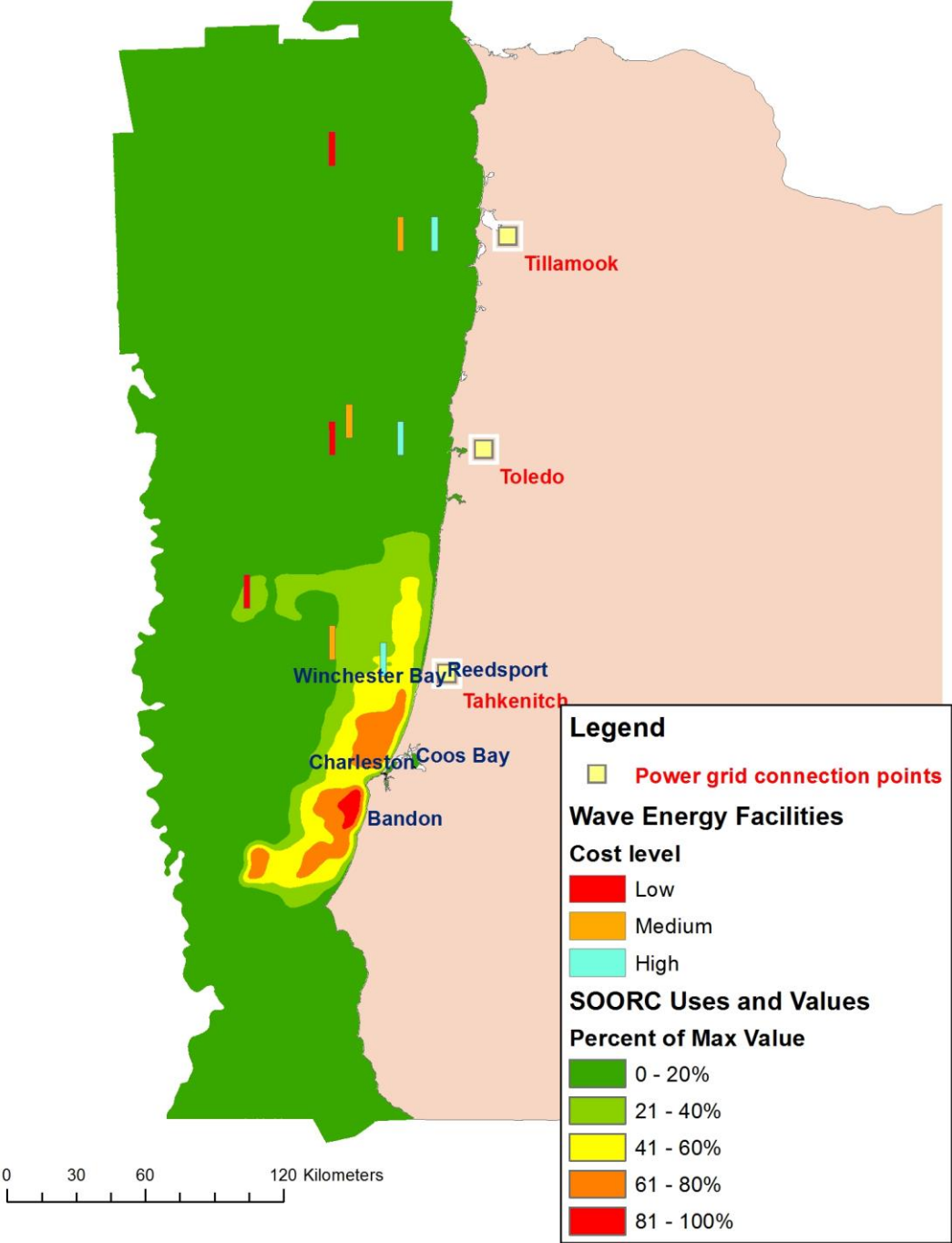


Figure3f
Port Orford Combined Value Fishing Grounds

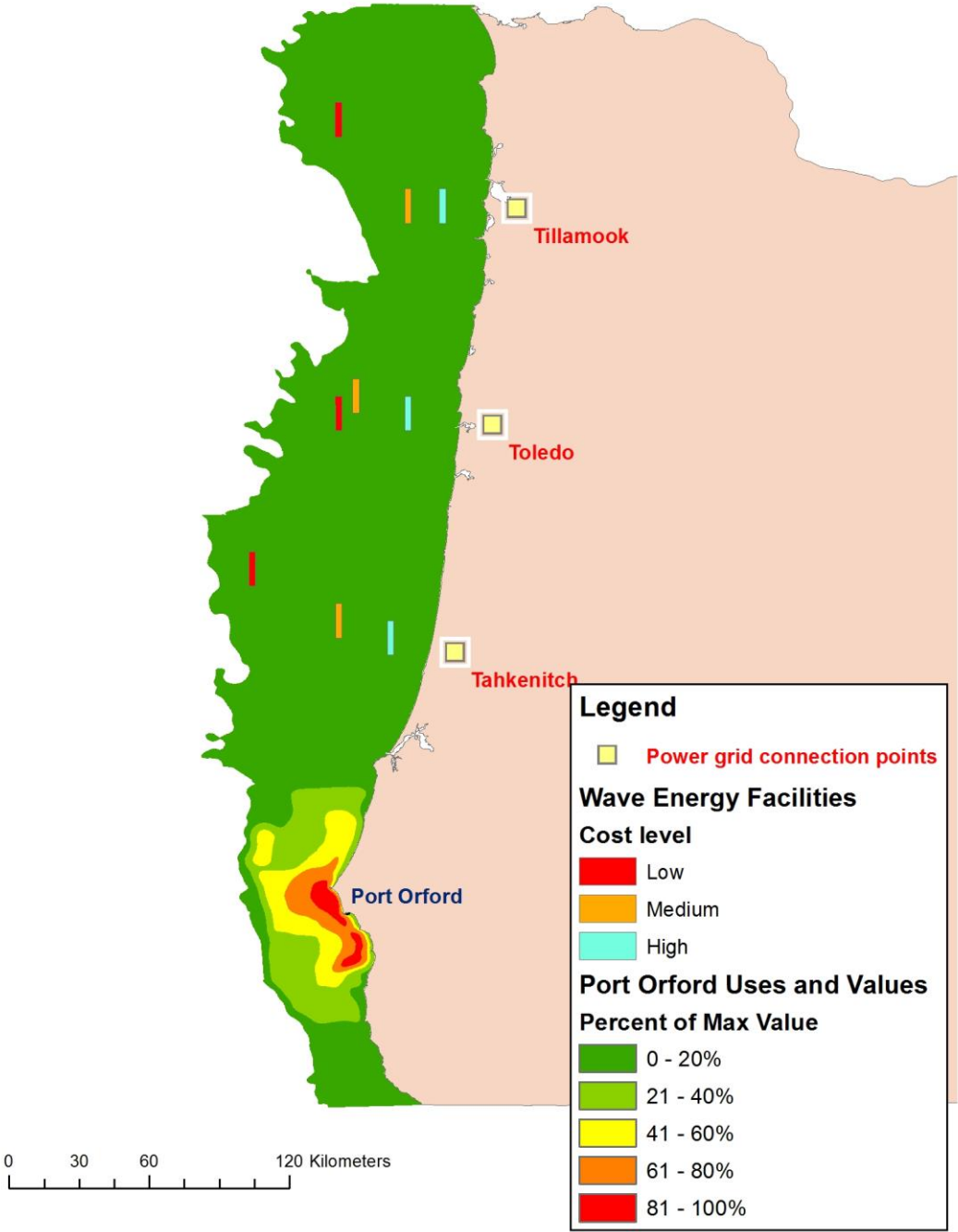
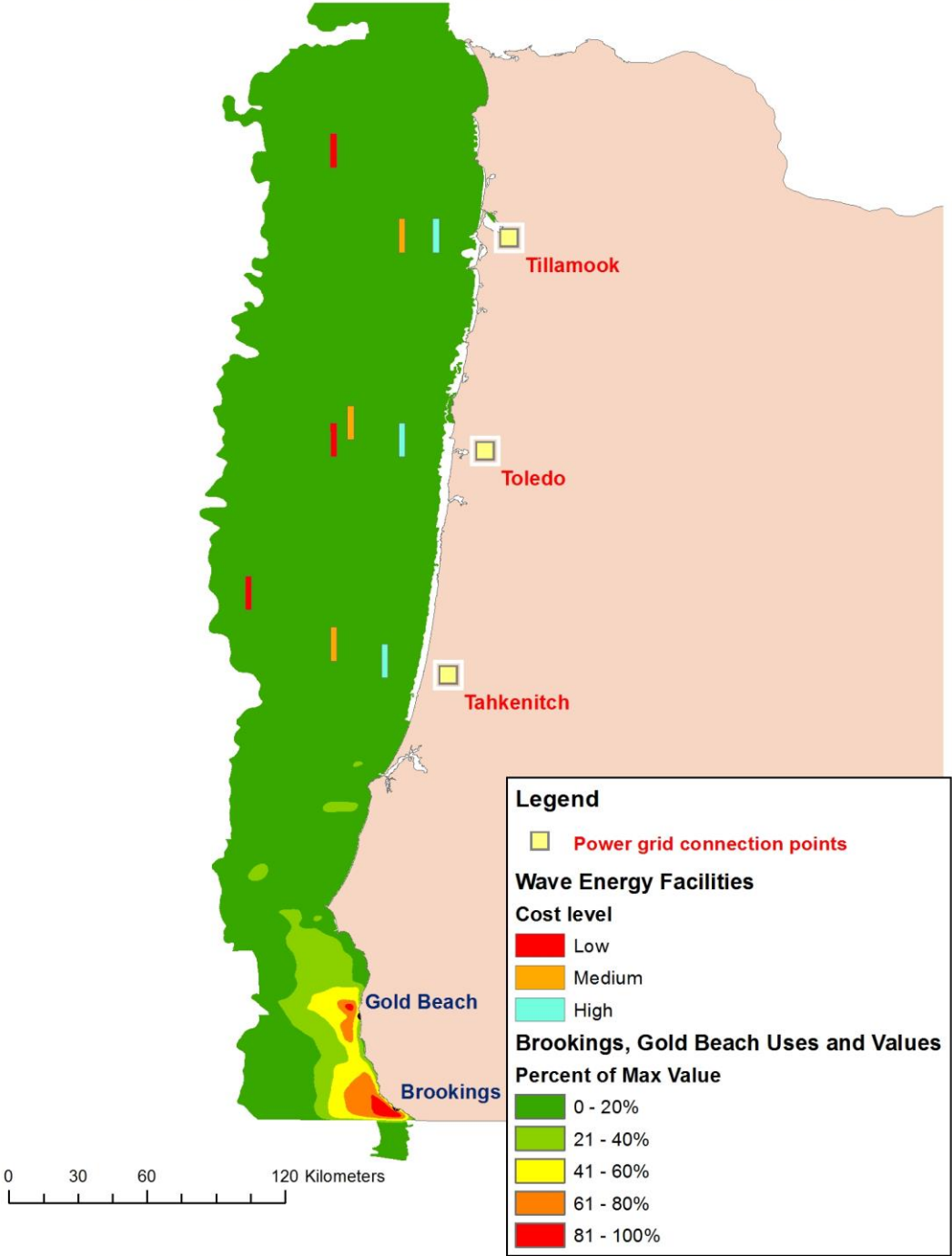


Figure3g
Brookings and Gold Beach Combined Value Fishing Grounds



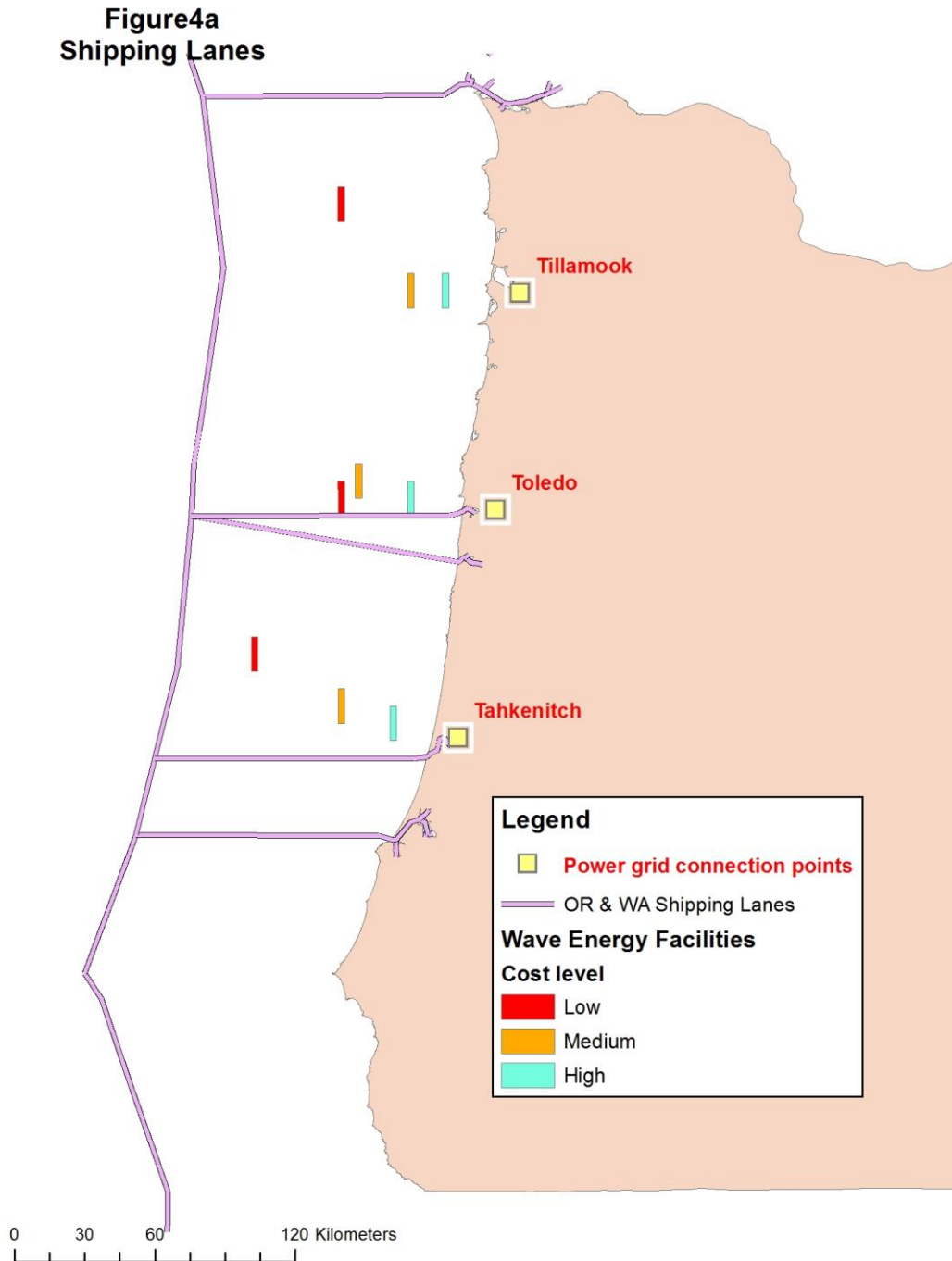


Figure 4a - 4c: For shipping lanes (4a) and towing lanes (4b), there is a strong potential conflict with the tugboat and barge tow lanes established off shore of all three connection points for the high cost scenario, while conflicts with shipping lanes are less likely. For submarine cables, there is a potential conflict with cables connected to the Tillamook area (4c).

Figure4b
Crabber-Tugboat Lanes

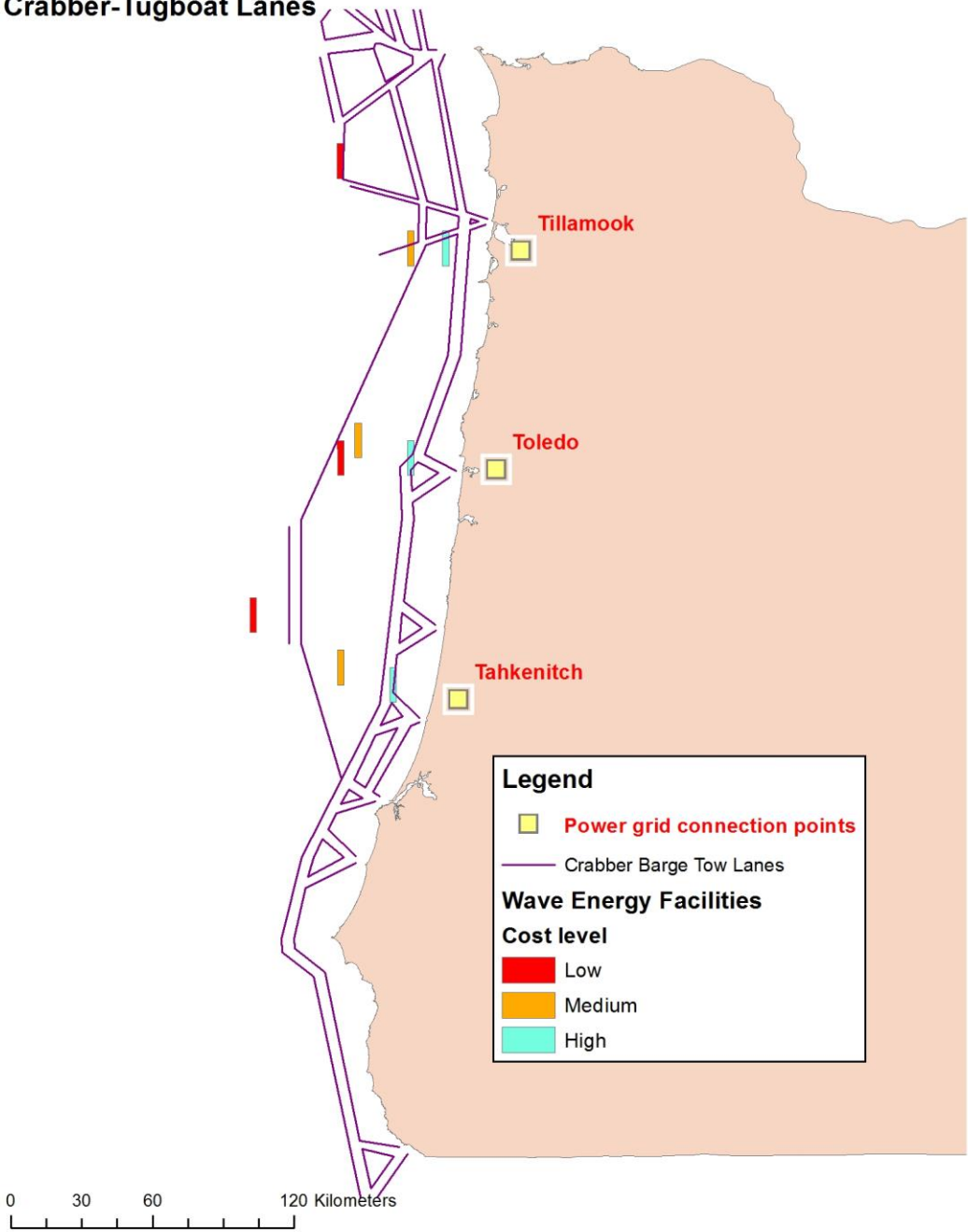


Figure4c
Underwater Utility Lines

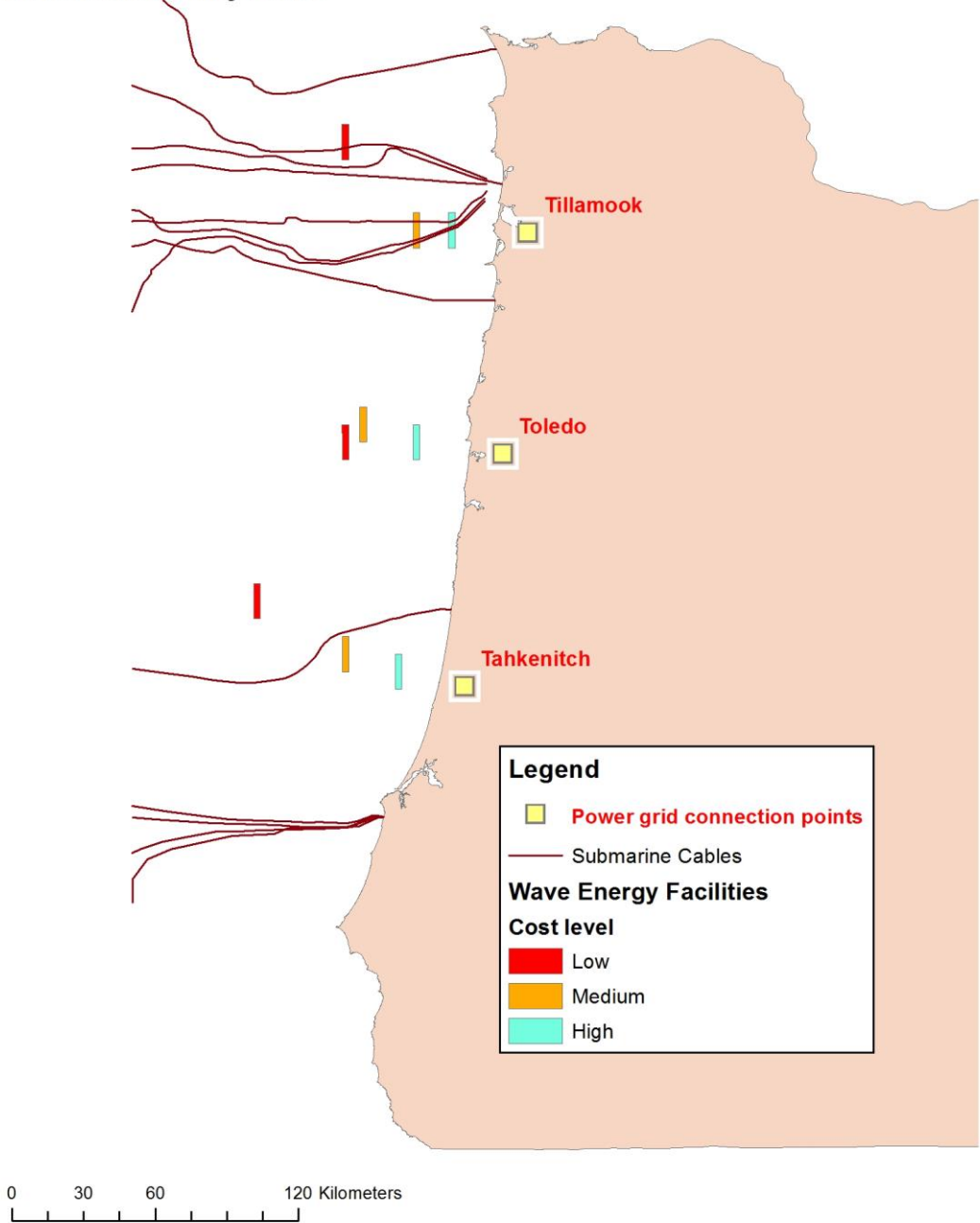


Figure5a
Green sturgeon CH

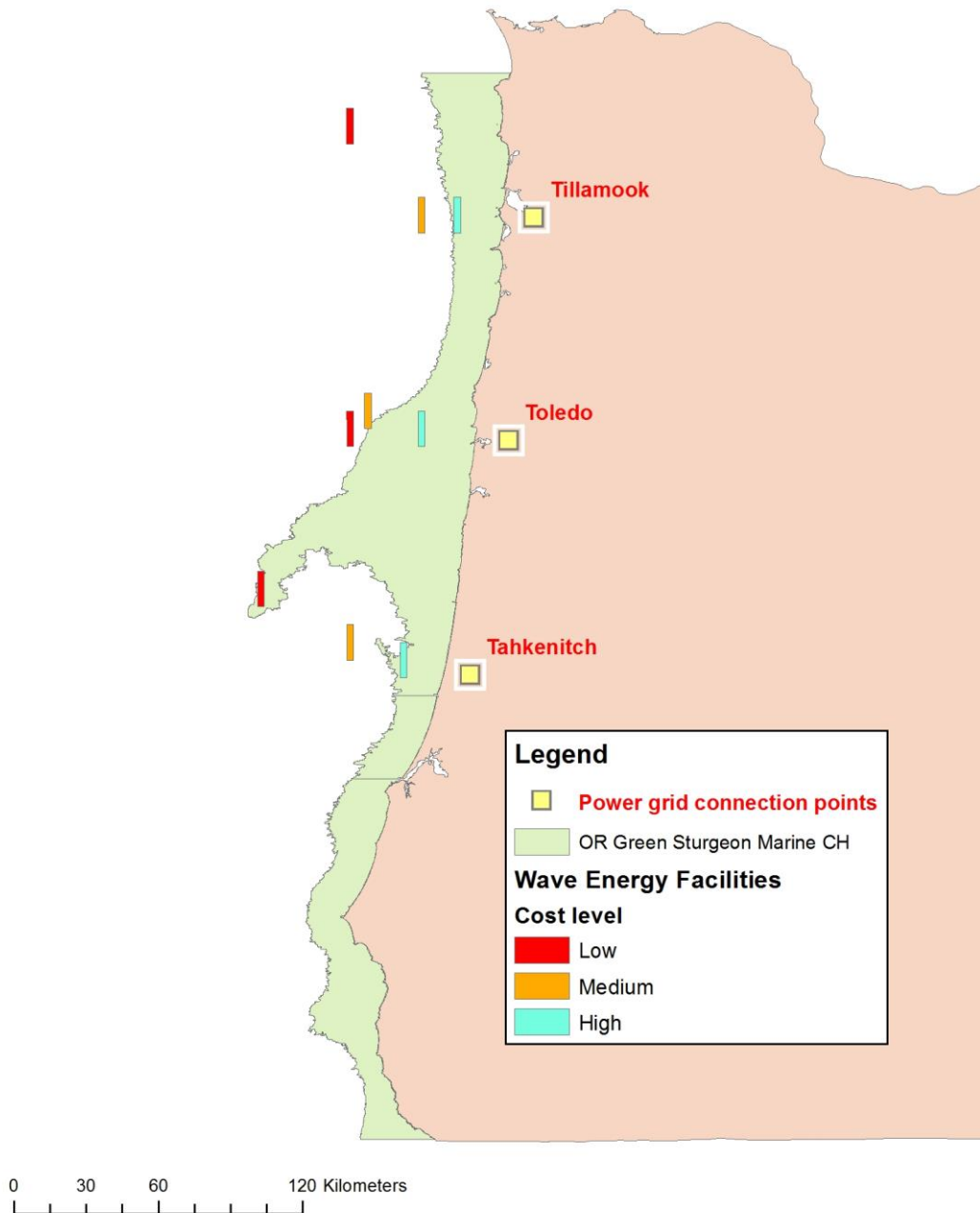


Figure 5a - 5b: The locations of some wave energy facilities overlap green sturgeon critical habitat designated under the Endangered Species Act (5a), which could trigger requirements for federal agencies such as the Federal Energy Regulatory Commission to consult with NOAA Fisheries before licensing a wave energy facility. For the Pacific groundfish conservation areas (5b), there is an overlap for two of the three low cost scenario facilities, but because these areas are currently managed as closures to harvest for certain groundfish fleets, the exact nature of any potential conflict is uncertain.

Figure5b
Pacific groundfish EFH conservation areas

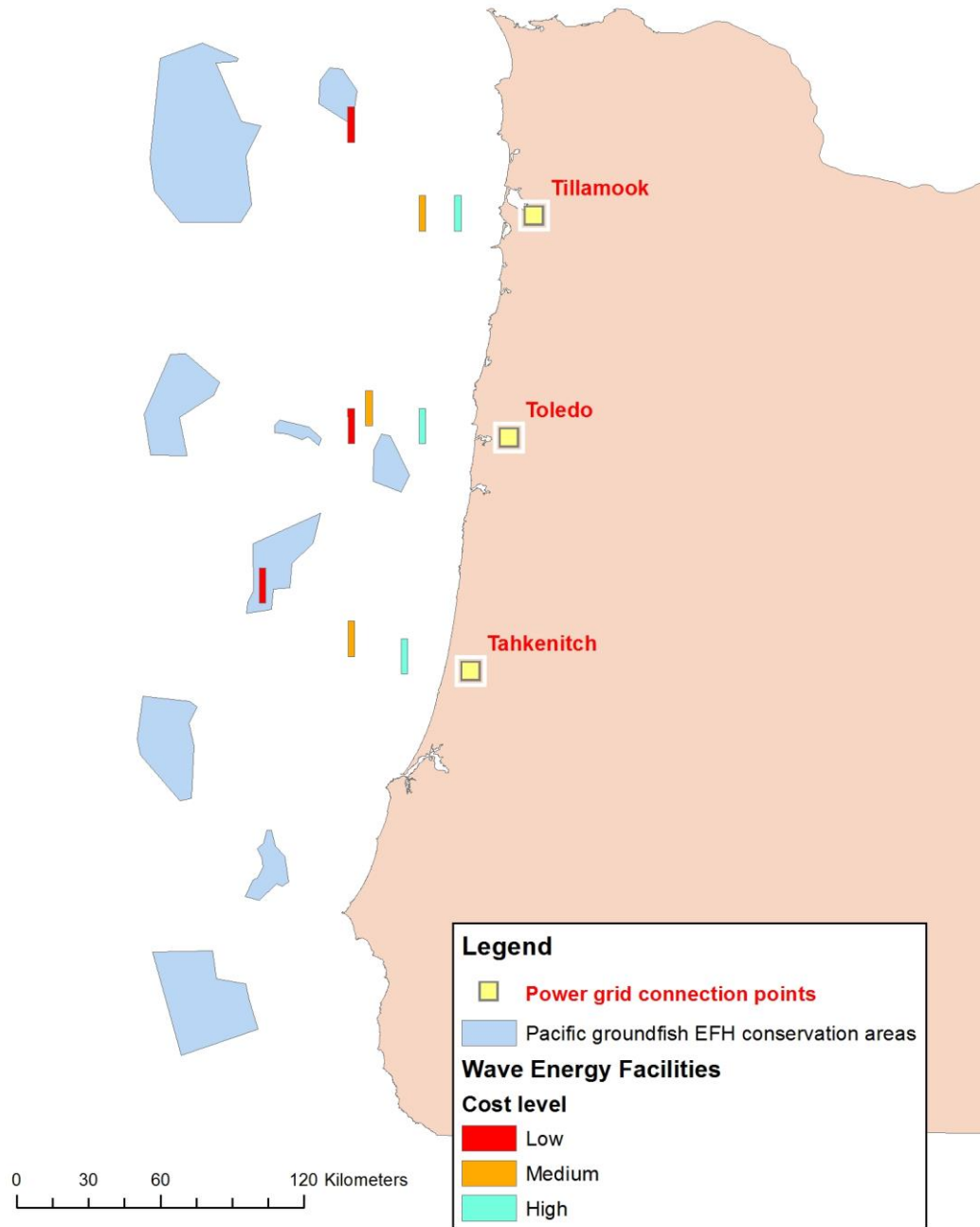


Table A1. Fixed gear fishing effort represented in West Coast Groundfish Observer Program (WCGOP) data by sector observed; including the proportion of total observed effort (cumulative hours gear was deployed) by sector from 2002-2009, the observed sector coverage rate calculated as the observed retained catch weight of target species divided by the fleet-wide landed weight of target species, and the assumed proportion of total fleet-wide effort represented in the observed data.

| Sector (2002-2009) | % of Total Duration by Sector | Sector Coverage Rate | Proportion of Duration Represented |
|---|--------------------------------------|-----------------------------|---|
| Limited Entry Sablefish Primary | 59.38% | 26.12% | 15.51% |
| Limited Entry Non-Tier-Endorsed Fixed Gear | 17.00% | 7.41% | 1.26% |
| Open Access Fixed Gear | 18.63% | 3.00% | 0.56% |
| Oregon Nearshore Fixed Gear | 3.83% | 5.20% | 0.20% |
| California Nearshore Fixed Gear | 1.16% | 3.43% | 0.04% |
| Sum total percentage of duration represented = 17.57% | | | |

APPENDIX MS2: FORECASTING THE RESPONSE OF KLAMATH BASIN CHINOOK POPULATIONS TO DAM REMOVAL AND RESTORATION OF ANADROMY VERSUS NO ACTION

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ABSTRACT

Two alternative actions are being evaluated in the Klamath Basin: 1) a No Action Alternative (NAA) and 2) removal of four mainstem dams (Iron Gate, Copco I, Copco II, and J.C. Boyle) and initiation of habitat restoration in the Klamath Basin under a Dam Removal Alternative (DRA). The decision process regarding which action to implement requires annual forecasts of abundance with uncertainty under each of the two alternatives from 2012 to 2061. I forecasted escapement for both alternatives by constructing a life-cycle model (Evaluation of Dam Removal and Restoration of Anadromy, EDRRA) composed of: 1) a stock recruitment relationship between spawners and age 3 in the ocean, which is when they are vulnerable to the fishery, and 2) a fishery model that calculates harvest, maturation, and escapement. To develop stage 1 of the model under NAA, I estimated the historical stock recruitment relationship in the Klamath River below Iron Gate Dam in a Bayesian framework. To develop stage 1 of the model under DRA, I used the predictive spawner recruitment relationships in Liermann et al. (2010) to forecast recruitment to age 3 from tributaries to Upper Klamath Lake, which is the site of active reintroduction of anadromy. I also modified the spawner recruit relationship under DRA to include additional spawning capacity between Iron Gate Dam and Keno Dam. In order to facilitate the comparison of the two alternatives, I used paired Monte Carlo simulations to forecast the levels of escapement and harvest under NAA and DRA. Median escapements and harvest were higher in DRA relative to NAA with a high degree of overlap in 95% confidence intervals due to uncertainty in stock-recruitment dynamics. Still, there was a 0.75 probability of higher annual escapement and a 0.7 probability of higher annual harvest by performing DRA relative to NAA, despite uncertainty in the abundance forecasts. The median increase in escapement in the absence of fishing was 81.4% (95% symmetric probability interval [95%CrI]: -59.9%, 881.4%), the median increase in ocean harvest was 46.5% (95%CrI: -68.7, 1495.2%), and the median increase in tribal harvest was 54.8% (95%CrI: -71.0%, 1841.0%) by performing DRA relative to NAA (estimates provided for model runs after 2033 when portion of the population in the tributaries to UKL are assumed to be established and Iron Gate Hatchery production has ceased)

1 INTRODUCTION

Evaluation of alternative actions in light of imperfect information is a dilemma commonly faced by decision makers (Berger 2006; Raifa and Schlaifer 2000). Often, there is a mismatch between the time needed to amass information through studies to provide a body of evidence for one action versus another (long time frame) and the time over which a decision is needed (short time frame). Modeling is a critical step in the decision making process and is useful for evaluating the outcome of each action, the uncertainty in the outcomes, and how those relate to the decision maker's objectives (Clemen 1996). Analyses that can improve the predictive ability of such models, such as statistical analysis, are valuable in revealing and quantifying some of the uncertainties in the decision process. Bayesian statistical analyses are particularly well suited to decision analysis given their natural approach to modeling uncertainty (Berger 2006). In the report that follows, I conducted a series of Bayesian statistical analyses and performed model forecasts in support of a decision: whether to operate the series of dams on the Klamath River consistent with recent history (the No Action Alternative) or whether to remove the four mainstem dams, restore anadromous Chinook salmon to the tributaries of Upper Klamath Lake, and initiate habitat restoration efforts in the tributaries of the Klamath Basin (the Dam Removal Alternative).

Chinook salmon (*Oncorhynchus tshawytscha*) in the Klamath River historically used the full extent of the watershed including tributaries to Upper Klamath Lake (Fortune et al. 1966; Lane and Lane Associates 1981; Moyle 2002; Hamilton et al. 2005; Butler et al. 2010). There are two distinct populations native to the Klamath Basin, namely spring and fall run. Spring run enter the river between March and July prior to maturation and hold in pools for 2 to 4 months prior to spawning, whereas fall run enter as mature adults from July through December and move directly to spawning grounds (Andersson 2003). In the tributaries of the Klamath Basin that currently have anadromy, the majority of Chinook runs are fall run (Andersson 2003), whereas spring run Chinook populations are found in the Salmon and Trinity rivers. With the potential for restoration of Chinook anadromy to the full watershed, there is interest in understanding how the levels of Chinook abundance in the Klamath Basin may change relative to the current conditions.

The objective of this effort is to develop a model that is capable of providing annual forecasts of Chinook abundance with estimates of uncertainty. The model must be able to represent the Chinook populations of the Klamath Basin using a life-cycle approach that incorporates harvest. The model must also be capable of evaluating two alternative scenarios: 1) a Dams Removal Alternative (DRA) in which the four mainstem dams (Iron Gate, Copco I, Copco II, and J.C. Boyle) are assumed to be removed in 2020, flows in the Klamath River are managed to attain hydrology as described in the Klamath Basin Restoration Agreement (KBRA), habitat improvements of spawning reaches are enacted as described in KBRA, and an active reintroduction program is implemented for the tributaries of Upper Klamath Lake (UKL); and 2) a No Action Alternative (NAA) in which the four mainstem dams remain in place and the flows in the Klamath River are managed to attain hydrology as described in the 2010 NMFS Biological Opinion (Hamilton et al. 2010). The period of record for the forecast is 2012 – 2061; thus modeling of both alternatives begins with the dams in place. The model was named EDRRA (Evaluation of Dam Removal and Restoration of Anadromy) to distinguish the work here from other models being developed in the Klamath Basin to understand the effects of dam removal, hydrology modifications, and habitat restoration.

The EDRRA model is composed of a stock production phase in which spawners generate progeny to the age 3 ocean stage. The stock production functions could potentially be derived in several ways: 1) statistical analysis of historical data, 2) literature derived values, and 3). professional judgment. Analysis of stock production relationships have been conducted periodically for Chinook of the Klamath Basin from spawner to adult recruit

(e.g., STT 2005). These data are useful for estimating a new stock production function to age 3. Further, estimation of the stock production functions in a Bayesian framework can be used to quantify the uncertainty in the stock recruitment parameters and provide predictive probability distributions for forecasting (e.g., Punt and Hilborn 1997). Where spawner and recruit data are not available, other methods must be used to make predictions of the spawner and recruitment relationship. A meta-analysis of stock-recruitment for Chinook populations throughout the western U.S. and Canada by Liermann et al. (2010) provide valuable insight into Chinook population dynamics. In particular, Liermann et al. (2010) provide posterior predictive distributions for calculating unfished equilibrium population abundance as a function of watershed size and provide posterior predictive distributions of productivity for both stream and ocean type Chinook. Such predictive distributions are valuable for making forecasts regarding the reintroduction of Chinook into tributaries to Upper Klamath Lake (UKL), where active reintroduction is planned for the Williamson, Sprague, and Wood Rivers (Hooton and Smith 2008).

To complete the life cycle, the ocean component of the life-history was needed. An “off the shelf” Klamath basin harvest model was made available by the Southwest Fisheries Science Center of NMFS (Mohr In prep). The Klamath Harvest Rate Model (KHRM), a spatially and temporally aggregated version of the Klamath Ocean Harvest Model (KOHM), calculates all sources of mortality starting at age 3. The KHRM, described in detail by Prager and Mohr (2001) and Mohr (In prep) takes as input the abundance of age 3, 4 and 5 Chinook in the ocean on September 1, and projects this population through the processes of natural mortality, ocean fishing, maturation, entry to the river, and river fisheries. Mature fish that avoid impact by river fisheries escape to spawn.

Using the EDRRA model, I compared the abundance of Chinook salmon under two alternative actions defining the future condition of the Klamath Basin. I analyzed a time series of spawner and recruitment data from 1979 to 2000 in the Lower Klamath Basin (STT 2005) in a Bayesian framework to develop a posterior predictive spawner recruitment relationship, which was used for forecasting future productivity in the lower basin. For areas of the Klamath Basin that lacked historical data, I used a spawner recruitment model that assumed capacity was related to watershed size and provided predictions of recruitment in probabilistic terms (Liermann et al. 2010). To complete the life cycle and understand the effect of the two actions on the fishery, I used the KHRM to calculate harvest and escapement. To facilitate the decision making process, I computed absolute and relative escapement and harvest metrics under NAA and DRA.

2 METHODS

2.1 RETROSPECTIVE ANALYSIS

2.1.1 STOCK RECRUITMENT DATA

Data on escapement and stock size were obtained from STT (2005). The recruitment was defined as the abundance of progeny spawned by S in calendar year BY that survive to become ocean age 3 on September 1 in calendar year $BY+3$ (STT 2005) (Table 1). The values in Table 1 were also used to compute a conversion factor (CF) from adult recruits (R) to age 3 ocean $N_{3,Sept1}$. The CF was estimated as a $N(2.03, 0.009)$ random variable, where with $N(\mu, \sigma^2)$ indicates a Normal (Gaussian) random variable with mean μ and variance σ^2 .

2.1.2 STATISTICAL MODEL

A Ricker stock-recruitment model (Quinn and Deriso 1999) was used to represent the levels of recruitment of age 3 adults in the ocean (R_t) as a function of the spawner abundance (S_t) for brood years $t = 1979, \dots, 2000$.

$$R_t = \alpha S_t e^{\{-\beta S_t + \epsilon_t\}}, \epsilon_t \sim N(0, \sigma_\epsilon^2) \quad (\text{Equation 1})$$

where ϵ_t is logNormal measurement error. The model was log transformed to obtain linearity in the relationship between log recruitment and spawning abundance given $\alpha' = \log(\alpha)$.

$$\log(R_t) = \alpha' + \log(S_t) - \beta S_t + \epsilon_t \quad (\text{Equation 2})$$

The model term $\log(S_t)$ was treated as an offset with a known coefficient value of 1 (McCullagh and Nelder 1989). Further additions to the model can be made by adding terms affecting the annual variability in the relationship between log recruitment and spawner abundance. In particular, I modeled the effect of annual variability in recruitment due to a common variability index (CVI_t) that was based on log survival rates of Iron Gate Hatchery (IGH) and Trinity River Hatchery (TRH) fingerling releases. Unlike typical covariates in a regression equation that are assumed known without error, the values of CVI_t were assumed known with error (described below). Note that the values of CVI_t were scaled to the levels of annual variability in the natural recruitment via the coefficient δ .

$$\log(R_t) = \alpha' + \log(S_t) - \beta S_t + \delta CVI_t + \epsilon_t \quad (\text{Equation 3})$$

2.1.3 COMMON VARIABILITY INDEX

The fingerling survival from IGH and TRH in the four months after release (May – Aug) for brood years 1979 to 2000 were compiled by STT (2005) to created an early-life survival index based on those data (Table 2). Instead of using the early life survival index, I used the log survival rates of fingerling Chinook released from IGH and TRH to understand the sources of annual variability in hatchery log survival rates $h_{j,t}$ for hatchery $j = \text{IGH, TRH}$, and brood year $t = 1979, \dots, 2000$.

$$h_{j,t} = \kappa_j + CVI_t + \gamma_j(Q_t) + u_{j,t}, \quad (\text{Equation 4})$$

$$CVI_t \sim N(0, \sigma_{CVI}^2)$$

$$u_{j,t} \sim N(0, \sigma_h^2)$$

where the log hatchery survival rates ($h_{j,t}$) for hatchery $j = \text{IGH, TRH}$ and brood year t were modeled as a function of a mean level of survival for each hatchery (κ_j), a random effects term representing a common source of variability to both hatchery stocks (CVI_t), a term representing the effect of summer flow in the river associated with each hatchery (γ_j) (Iron Gate Hatchery survival a function of Klamath River flow at Seiad in the first two weeks of July, USGS gage 11520500) and Trinity River survival a function of mean monthly July flow at Lewiston, USGS gage 11525500), and a residual error term $u_{j,t}$.

Coefficients in Equations 4 and 3 were estimated simultaneously in a Bayesian framework. The directed acyclic graph (DAG) for the probability model provides a mapping of the conditional relationships among the parameters (Figure 1). The values of CVI_t were not known with certainty, but rather were estimated as random effects variable in Equation 4. In equation (3), the common hatchery variability (CVI_t) is thus treated as an error in variables covariate (e.g., Congdon 2002) in the regression model for natural recruitment.

2.1.4 BAYESIAN ESTIMATION

The Bayesian paradigm estimates a probability distribution of the model parameters θ given the observed data \mathbf{R} by using Bayes' rule:

$$\pi(\boldsymbol{\theta} | \mathbf{R}) = \frac{\pi(\boldsymbol{\theta})f(\mathbf{R}|\boldsymbol{\theta})}{f(\mathbf{R})} \quad (\text{Equation 5})$$

where $\pi(\boldsymbol{\theta} | \mathbf{R})$ is the posterior probability distribution of the model parameters given the data, $\pi(\boldsymbol{\theta})$ is the prior probability distribution of the model parameters, $f(\mathbf{R}|\boldsymbol{\theta})$ is the likelihood of the data given the model parameter values, and $f(\mathbf{R})$ is the marginal probability density of the recapture data. The marginal probability density, $f(\mathbf{R})$, may also be viewed as integrating across the entire parameter space of $\boldsymbol{\theta}$; thus

$$f(\mathbf{R}) = \int \pi(\boldsymbol{\theta})f(\mathbf{R}|\boldsymbol{\theta})d\boldsymbol{\theta}.$$

Priors for the coefficients in the Bayesian estimation were non-informative (Box and Tiao 1973, Gelman et al. 2004). Priors for both the mean and the variance of the coefficients were required. Priors for the means were given normal distributions with large variances (e.g., $N(0,1000)$), whereas priors for the variance terms were given inverse gamma distributions that had approximately uniform probability density across the range of likely values (e.g., $IG(0.001, 0.001)$) (Table 3).

The posterior distributions of the model parameters $\boldsymbol{\theta}$ were estimated by drawing samples from the full conditional distributions of each parameter given values of all other parameters. This was implemented by using a Metropolis within Gibbs Markov Chain Monte Carlo (MCMC) approach (Gelman et al. 2004; Gilks et al. 1996). If the posterior distribution is a standard statistical distribution and the priors for the mean and the variance are conjugate priors, the Gibbs sampler may be used to update the samples in the Markov Chain (Roberts and Polson 1994). The non-informative priors used here were conjugate priors, thus the Gibbs sampler was used. MCMC sampling was implemented in WinBUGS 1.4.3 (Spiegelhalter et al. 2003).

Diagnostics of MCMC chains are required to ensure that the MCMC chain has converged to a stationary target distribution. Multiple chains were run using dispersed initial values for each model, and a scale reduction factor (SRF, Gelman et al. 2004), which indicates whether further sampling would improve the accuracy of draws from the target distribution, was calculated for each monitored quantity in the model. Monitored parameters in all models had SRF values that indicated samples were being drawn from the target distribution (i.e. $\text{SRF} \approx 1$) by 50 000 samples. The initial 30% of the samples were used to reach the stationary target distribution and were discarded (“burn in”) with the subsequent samples thinned to produce approximately 1,000 draws from the stationary target distributions. The 1,000 draws were used to compute the posterior mean and 95% central probability intervals or credible intervals (95% CrI). The diagnostics were implemented using the R2WinBUGS package (Sturtz et al. 2005) in R (RCDDT 2010).

I compared two models of stock recruitment; the first model was the base model (Equation 2) and a second alternative model with the common variability index (Equation 3). I used Deviance Information Criterion (DIC) to evaluate model predictive ability with a penalty for model complexity (Spiegelhalter et al. 2002).

$$DIC = \bar{D}(\mathbf{R}|\boldsymbol{\theta}) + p_D \quad (\text{Equation 6})$$

where the deviance $D(\mathbf{R}|\boldsymbol{\theta})$ is equal to $2 \times$ the negative log likelihood (e.g., $-2\log p(\mathbf{R}|\boldsymbol{\theta})$). The deviance is a measure of model fit and decreases with better fitting models. The deviance is calculated at each iteration of the MCMC chain, and the first term on the right hand side of the equation is the posterior mean of the deviance (e.g., $\bar{D} = 1/L \sum_{l=1}^L D(\mathbf{R}|\boldsymbol{\theta}^l)$). The second term on the right hand side of the equation 6 is p_D , which is the effective number of parameters. In a hierarchical model the effective number of parameters is typically less than the total number of estimated parameters, because information is being shared among random effects. The term p_D is defined as $p_D = \bar{D} - \tilde{D}$ (Spiegelhalter et al. 2002), and \tilde{D} is the deviance evaluated at the posterior mean of the model parameters (e.g., $\tilde{D} = D(\mathbf{R}|\bar{\boldsymbol{\theta}})$).

2.1.5 FISHERIES REFERENCE POINTS

Reference points of the Ricker stock recruitment relationship were calculated using the following formula (Ricker 1975):

S_{msy} is the spawner that provides maximum sustainable yield. There is no analytical solution to the equation (Quinn and Deriso 1999), thus it was solved iteratively by maximizing the yield ($R - S$), which is defined as

$$\alpha S_{msy} e^{\{-\beta S_{msy} + \delta CVI\}} - S_{msy} \quad (\text{Equation 7})$$

To calculate S_{msy} , I assumed the random effect of CVI was at its average value (i.e., $CVI = 0$)

S_{max} is the spawner abundance that provides maximum recruitment:

$$S_{max} = \frac{1}{\beta} \quad (\text{Equation 8})$$

S_{ueq} is the spawner abundance at unfished equilibrium population size, assuming recruitment is defined as adults. When the recruitment is defined as an earlier life stage, it is still useful as the spawner abundance that equals the abundance of the earlier life stage; here it is age 3 ocean fish.

$$S_{ueq} = \log(\alpha) / \beta \quad (\text{Equation 9})$$

Estimating the model parameters in a Bayesian framework facilitated the calculation of the fishery reference points as probability distributions. Distributions for fishery reference points were calculated by drawing 1000 samples from the posterior distributions of the model parameters, calculating the reference point for each of the 1000 draws and forming a probability distribution.

2.1.6 ASSUMPTIONS FOR RETROSPECTIVE ANALYSIS

The assumptions in conducting the retrospective analysis using the Ricker stock – recruitment model are the same as those enumerated in STT (2005, p. 2). In addition, I make the following assumptions in the retrospective stock recruitment analysis:

1. The flow metrics (July flow at Seiad on the Klamath River and July flow at Lewiston on the Trinity River) were representative of annual variability in flow. I evaluated multiple flow metrics in a correlation analysis to evaluate multiple flow metrics to residuals from the STT (2005) analysis (not shown). In addition, the amount of variability attributable to flow was relatively small compared to CVI; therefore, incorporation of alternative flow metrics should have a small effect on parameter estimates.
2. The Bayesian model is drawing samples from the stationary posterior distribution of model parameters (i.e., the model has converged). While there are tests for lack of convergence (i.e., SRF values) that were used here, there are no methods to guarantee convergence.

2.2 FORECASTING ABUNDANCE UNDER THE NAA AND THE DRA

Under both the NAA and the DRA, the life cycle of Chinook was completed in two stages: 1) production of natural origin age 3 ocean fish from spawners and hatchery origin age 3 ocean fish from Iron Gate and Trinity River hatcheries, and 2) calculation of harvest, maturation rates, natural mortality, and escapement by the KHRM (Mohr In prep). The production of age 3 ocean fish was implemented with Monte Carlo simulations to incorporate uncertainty in the abundance forecasts. I conducted 1000 Monte Carlo simulations to characterize the uncertainty in future productivity under each of the two alternatives. Each iteration of the Monte Carlo simulation paired the NAA and DRA forecasts; parameter draws used in the production stage under NAA and DRA (e.g., values of CVI_t) were the same under NAA and DRA for each iteration of the model. For example, the value of CVI_{2024} was the same in iteration 724 of NAA as in iteration 724 of DRA. Using the same covariate values in a given iteration allowed paired comparisons of model outputs, which were valuable for calculating the relative benefits of the two alternatives in spite of uncertainty in the absolute abundances.

I provide details on the production of age 3 ocean fish under the two alternatives below. The application of KHRM was the same between the NAA and DRA evaluations which also facilitated comparison of DRA and NAA on relative terms. In general, the default values of the KHRM were used in EDRRA. Values of the biological parameter set that were supplied for each run of KHRM were:

- 1) N_a , which was a vector of abundances consisting of: age 3 hatchery and natural origin in the ocean, age 4 hatchery and natural origin in the ocean, and age 5 hatchery and natural origin in the ocean
- 2) g_a , which was a vector of proportions of the natural origin consisting of: age 3 natural proportion, age 4 natural proportion, and age 5 natural proportion.

The KHRM operated as a deterministic harvest model with uncertainty in harvest and escapement arising only from the input of the N_a , g_a vectors only. The fishery control rule defined the harvest rates based on expected levels of escapement in the absence of harvest (Mohr In prep), and under both the NAA and DRA the fishery control rule was an updated version of the amendment 16 fishery control rule (Appendix A). The default management parameters and the fishery parameters in the KHRM were not modified; therefore, the management and fishery behavior of the KHRM model was exactly the same under both alternatives.

The role of flow in the Klamath and Trinity Rivers was expected to affect hatchery survival rates, and flow was included in the forecasted production functions to age 3. Flows for the Klamath River at Seiad were forecasted for the 50 year period (2012 to 2061) as part of flow studies on the Klamath River in support of the Secretarial Determination process (Reclamation 2010). Two flow series were used as part of the hydrological evaluation of future conditions in the Klamath Basin; these were the flows under the Biological Opinion (NMFS 2010) and the flows as recommended under KBRA. In the Ricker stock recruitment model presented here, the flow covariate was normalized to have a mean value of 0 and a standard deviation of 1. In order to use the parameter values for flow (γ_{IGH}), hydrology data for the Klamath River at Seiad was normalized using the same values as the historical data (mean of 1589.0 cfs, sd = 944.17). These normalized flows are presented in Figure 2 to provide a comparison under the two alternatives. In the Trinity River, no such flow forecasts were available; therefore, I constructed a time series of flows that were consistent with historical flows. The constructed flow series for the Trinity was used for all iterations of EDRRA under NAA and DRA.

Monte Carlo simulation was used to integrate across the uncertainty in the model parameters with the objective of translating uncertainties in model inputs into uncertainties in model outputs (Manly 1997). Monte Carlo simulation is a technique that involves using random numbers sampled from some form of a probability distribution as input to a deterministic equation or model to derive an outcome under conditions of uncertainty. As the number of outcomes in the simulations approaches infinity, the statistics (mean, standard deviation, etc.) converge to their true value (Givens and Hoeting 2005).

2.2.1 PRODUCTION TO AGE 3 IN THE OCEAN UNDER THE NO ACTION ALTERNATIVE (NAA)

Forecasted production under the NAA consisted of production of natural origin and hatchery origin age 3 ocean salmon. Forecasts of natural production were based on the results of the retrospective Ricker stock-production function described previously (Equation 3). Values of $CVI_{i,t}$ were drawn for each iteration i and year t of the model, where t is now the year when the cohort is at age 3 and S_{t-3} is the spawner abundance. The values for $CVI_{i,t}$ were drawn from a $N(0, \sigma^2_{CVI,i})$ and residual error $\epsilon_{i,t}$ from $N(0, \sigma^2_{\epsilon,i})$. The values of the parameters of the stock production function (α' , β , δ) were drawn from their Bayesian posterior distributions. In each year the hatchery was operational, the Trinity River Hatchery produced 3 million and the Iron Gate Hatchery produced 6 million fingerlings. Values of log hatchery survival were drawn from their posterior distributions (e.g., κ_j , γ_j for j = IGH, TRH) and the residual error was drawn from $N(0, \sigma^2_{h,i})$. To provide age 3 hatchery abundance, hatchery fish were assumed to have an age 2 to age 3 survival rate of 0.5 (Hankin and Logan 2010). For a more detailed description of the steps in the NAA simulation, please see Appendix B.

2.2.2 FORECASTING ABUNDANCE UNDER THE DAM REMOVAL ALTERNATIVE (DRA)

There are several substantial changes to the Klamath River system that were incorporated in the model under DRA: 1) production in the tributaries of Upper Klamath Lake (Wood, Williamson, and Sprague Rivers); 2) reintroduction of Chinook to these tributaries of UKL; 3) production in the mainstem Klamath from Iron Gate Dam to Keno Dam and tributaries (Spencer, Shovel, Jenny, and Fall creeks); 4) KBRA flows in the mainstem Klamath; and 5) KBRA habitat restoration actions in the tributaries to Upper Klamath Lake and lower basin tributaries.

2.2.2.1 Production in Tributaries to Upper Klamath Lake

I calculated the production of natural origin ocean age 3 fish from tributaries of Upper Klamath Lake (the upper basin) as described in Liermann et al. (2010). Liermann et al. (2010) used watershed size to predict the unfished equilibrium population size based on a meta-analysis of multiple stocks of Chinook salmon throughout the western United States and Canada; they also estimated the productivity for ocean-type and stream-type Chinook. I used both of these results to develop Ricker stock production functions for the upper basin.

Estimates of watershed area

The definition of usable watershed area required evaluating potential barriers to migration (Table 4). The Williamson River is the main river system in the Upper Klamath Basin that, when including the Sprague River subbasin, comprises 79 percent of the total drainage area of the Basin (Risley and Laenen 1999). The Williamson River subbasin has a drainage area of approximately 3678 km² (1,420 mi²), extending from its source on the eastern edge of the basin, and flowing through the Klamath Marsh, which covers 601 km² (232 mi²) (Risley and Laenen 1999; Conaway 2000; David Evans and Associates 2005). The area of the lower Williamson River, between the Kirk Reef and UKL, covers 311 km² (120 mi²), and is one of the major ground-water discharge areas in the upper Klamath Basin.

The Sprague River is the main tributary of the Williamson River system in the Upper Klamath Basin, comprising approximately 4,092 km² (1,580 mi²), which includes the North and South Forks, Fishhole Creek, and the Sycan River subbasins (Risley and Laenen 1999). The upper extent of the Sprague subbasin, which is upstream of Beatty Gap above the Sycan River, is approximately 1471 km² (568 mi²), and includes a portion of the Fremont-Winema National Forest. The lower extent of the Sprague subbasin below the Sycan River is approximately 1,173 km² (453 square miles in area), meandering through the lower valley for 75 miles to its confluence with the Williamson River (Conelley and Lyons 2007).

The Sycan River subbasin has a drainage area of approximately 1447 km² (559 mi²). The upper extent of the Sycan River subbasin above Sycan Marsh is approximately 103 square miles in area (Conelley and Lyons 2007).

The Sycan Marsh is predominantly a surface-water dominated wetland, measuring approximately 124.3 km² (48 mi², 30,537 acres), accepting flows not only from the Upper Sycan River, but from an additional drainage area of 456 km² (176 mi²) surrounding the marsh (USFS 2005). The lower extent of the Sycan River subbasin begins below the Sycan Marsh, and is approximately 601 km² (232 mi²) in area (Conelley and Lyons 2007).

The Wood River subbasin is located in Klamath County, Oregon approximately 40 miles north of Klamath Falls. The subbasin has a drainage area of approximately 567 km² (219 mi²) extending from the southern flanks of the Crater Lake highland within Crater Lake National Park and the Winema National Forest, and flowing southward through the Wood River Valley into Agency Lake (USBR 2005; Graham Matthews and Associates 2007).

The total estimate of watershed size for the tributaries to UKL was 4200.96 km² (Table 4). Using samples from posterior distributions provided by Martin Liermann (Martin Liermann, NWFSC NOAA, March 28, 2011 personal communication) as described in Liermann et al. (2010), a stock production function was constructed for the tributaries to UKL. Liermann et al. (2010) used a version of the Ricker stock recruitment function defined in terms of the log productivity r and the unfished equilibrium population size E (the value where recruitment abundance equals spawning abundance). Liermann et al. (2010) found that the log productivity was different for ocean type and stream type Chinook; further, they found that the relationship between watershed size and E was different for ocean and stream Chinook.

Both stream and ocean type Chinook are expected to be present in the tributaries to UKL (Dunsmoor and Huntington 2006); therefore, the production functions for the tributaries to UKL incorporated productivity (r) and unfished equilibrium population size (E) for a mixture of stream and ocean Chinook. To implement the mixture, the proportion of ocean and stream type were able to vary in each year. For each iteration i and year t of the model, a proportion of ocean Chinook $p_{i,t}$ was drawn at random from a Uniform(0,1) distribution.

The unfished equilibrium population size was calculated for stream type Chinook using Equation 8 of Liermann et al. (2010) (assuming $L = 0$ indicating stream Chinook), $E_{new\ stream}$. The unfished equilibrium population size was also calculated for ocean type Chinook using Equation 8 (assuming $L = 1$, indicating ocean Chinook), $E_{new\ ocean}$. The mixture of ocean and stream unfished equilibrium population size $E_{new, i, t}$ for iteration i and year t was calculated as follows:

$$E_{new, i, t} = p_{i, t} E_{new\ ocean, i} + (1 - p_{i, t}) E_{new\ spring, i} \quad \text{Equation (11)}$$

In a similar fashion, the values of productivity $r_{new, i, t}$ were formed as a mixture of ocean and stream type r values from Liermann et al. (2010).

$$r_{new, i, t} = p_{i, t} r_{new\ ocean, i} + (1 - p_{i, t}) r_{new\ spring, i} \quad \text{Equation (12)}$$

The values of $E_{new, i, t}$, $r_{new, i, t}$, and the spawner abundance three years previously ($S_{i, t-3}$) allowed the calculation of upper basin adult recruits in the absence of fishing via Equation 1 in Liermann et al. (2010). In addition, annual variability in recruitment was modeled with a random effect $w_{i, t}$. The random effect for annual variability in the tributaries of UKL was the same as the lower basin δ_i CVI $i, t-2$. Finally the recruitment calculated to the adult returning stage was converted from adult to 3 year ocean fish (via the $N(2.03, 0.01)$ expansion factor).

2.2.2.2 Modeling the reintroduction to tributaries of Upper Klamath Lake

The reintroduction of Chinook to the tributaries of UKL was assumed to start in 2019 with fry being planted in the tributaries to UKL prior to dam removal in 2020. The reintroduction process is expected to construct a conservation hatchery that is capable of seeding the tributaries to UKL with fry to capacity (Hooton and Smith 2008). There is no fry or other juvenile freshwater stage in the model; therefore, stocking to capacity was modeled

by assuming that the numbers of adult returns were at or above the unfished equilibrium population size $E_{new,i,t}$ from 2019 to 2029 for model iteration i and year t .

2.2.2.3 Production from Iron Gate to Keno Dam

From Iron Gate Dam to Keno Dam, the mainstem and tributaries to the mainstem (Spencer, Shovel, Jenny, and Fall creeks) watershed area was estimated at 1792.2 km² (Lindley and Davis In prep). Posterior samples from the distributions for parameters defining the relationship between watershed size and unfished equilibrium population size E were used to construct the posterior predictive distribution for E_{new} given the watershed size for Iron Gate to Keno Dam using Liermann et al. (2010) and assuming ocean type Chinook.

Further, the following steps were taken to modify the Ricker stock recruitment relationship under NAA to include the additional spawning area below Keno Dam in the DRA:

1. Calculate the distribution of unfished equilibrium population size for the Iron Gate to Keno mainstem and tributaries using Equation 8 of Liermann et al. (2010) assuming a watershed size of 1792.2 km² and ocean Chinook, $E_{Keno:IG}$
2. Multiply the unfished equilibrium population size for adult recruits by the adult recruit to age 3 ocean factor CF
3. Use the distribution of S_{ueq} calculated in Equation 9 of this document for the pre-dam removal estimate of unfished equilibrium population size (recruitment defined as age 3 ocean abundance).
4. Add the unfished equilibrium abundance for habitat from Keno to Iron Gate calculated in step 1 to the old equilibrium abundance from step 2 to calculate $S_{ueq\ new}$
5. Calculate a new distribution for the β parameter with the additional capacity by re-arranging Equation 9

$$\beta_{new} = \frac{\alpha'}{S_{ueq\ new}} \quad \text{Equation (10)}$$

Because there were 1000 posterior samples for each of these quantities ($E_{Keno:IG}$ and S_{ueq}) the above calculations were carried out 1000 times for each iteration i of the model. The 1000 samples of the distribution of β_{new} were used for the forecasting the productivity of the Klamath River below Keno Dam after 2020 (i.e., replace β in Equation 10 with β_{new}).

2.2.2.4 Modeling the effects of KBRA

Since the Fisheries Restoration Plan under KBRA has yet to be developed, specific restoration projects within each of the tributary streams currently included in the model have yet to be identified. Habitat restoration actions were specifically identified for the three major lower basin tributary streams (Scott, Shasta, and Salmon) and in the tributaries to Upper Klamath Lake. I assumed that for the purposes of this model, all of the habitat restoration actions identified will have benefits beginning in 2013 and accruing through 2061.

Stakeholders identified the likelihood that annual variability in recruitment from the tributaries to UKL could vary with Klamath River flows. The variation in production due to flow variability is not known given the lack of information on the upper basin, however. I assumed that flow variability affected outmigrating UKL fish to a similar degree as the IGH hatchery fish. Thus, the posterior distribution on γ_{IGH} was used as a posterior predictive distribution on the effect of flow on production in the tributaries to UKL. The values of the flows at Seiad used in the retrospective analysis were normalized to have a mean of 0 and a standard deviation of 1; therefore, the KBRA flows used to compute annual variability in recruitment to age 3 from tributaries to UKL under DRA were transformed using the same mean and standard deviation as the Seiad series.

Stakeholders have also identified the likelihood that KBRA actions will increase productivity between 2012 and 2061. The uncertainty in productivity was characterized by the posterior distribution of α' ; thus, the posterior distribution of α' provides a description of the range of possible productivity values in the lower basin along with the probability of observing those values (by definition of a posterior probability distribution). I implemented the improvement in productivity due to KBRA actions in EDRRA by drawing samples from a truncated distribution of productivity. By using a truncated distribution, the upper range of productivity values did not change, whereas the lower values of productivity became less likely over time. In Figure 3, the process of drawing posterior predictive samples from truncated distributions is depicted. Early in the time series, low as well as high productivity values can be drawn from the distribution; however, as the time series progresses lower values of productivity are rejected and a new draw must be made until one from the Accepted region is obtained. In practice, the draws were made from truncated Normal distributions via the package *msm* (Jackson 2011) in the statistical programming language R (RCDT 2010). The lower threshold value was set at the 0 quantile in 2012 (i.e., the full distribution was sampled) and the quantile increased linearly to 0.25 by 2061; that is, by 2061 only the upper 0.75 portion of the distribution could be sampled (lower threshold at quantile of 0.25). Draws from the truncated distribution are distinguished by an asterisk on the parameter. For example, truncated draws from the lower basin productivity α' are distinguished as α'^* .

A similar approach was implemented for the tributaries to UKL, where uncertainty was characterized through the use of posterior predictive distributions of productivity for ocean type and stream type Chinook presented in Liermann et al. (2010) (i.e., $r_{newocean}$ and $r_{newstream}$). The lower threshold for sampling in 2012 was set at the 0 quartile (the entire distribution could be sampled) and moved linearly to the 0.25 quantile by 2061 (truncated to the upper 0.75 portion of the distribution). The mixture of ocean and stream Chinook was then applied via the proportion of ocean Chinook $p_{i,t}$ after the draws from the truncated distributions of $r_{newocean}$ and $r_{newstream}$.

In a similar fashion, the values of productivity $r_{new,i,t}$ were formed as a mixture of ocean and stream type r values from Liermann et al. (2010).

$$r_{new,i,t}^* = p_{i,t} r_{newocean,i}^* + (1 - p_{i,t}) r_{newstream,i}^* \quad \text{Equation (13)}$$

Please see Appendix B for the specific steps of production of Age 3 Chinook under DRA.

2.2.3 ASSUMPTIONS TO FORECASTING UNDER DRA AND NAA

Multiple assumptions were made to forecast abundance under DRA and NAA:

1. Data used for the stock-recruit analysis and subsequent simulation modeling were based on current and past conditions and are also indicative of future conditions in the Lower Klamath Basin
2. Stock recruitment relationships developed from the retrospective analysis will be the same in the future. Any modifications to the stock recruitment relationships for the Lower Klamath Basin in the future will only occur as modeled (e.g., KBRA effects under DRA).
3. Annual variability in stock recruitment in the lower basin will be of a similar magnitude to past annual variability in stock recruitment.
4. The use of Liermann et al. (2010) work assumes that the Klamath system falls within the range of watersheds evaluated in their analysis. The Liermann et al. (2010) work was used due to its incorporation of a broad range of watersheds, inclusion of stream and ocean type Chinook, and the explicit incorporation of uncertainty in predictions for new streams. The EDRRA model assumes that production from the Klamath River at the beginning of the time series could range from the worst to the best rivers analyzed in Liermann et al (2010).
5. Conversion from adult abundance to age 3 abundance is valid based on data presented in STT (2005) (Table 1).

6. Capacity for the Iron Gate to Keno reach calculated using Liermann et al. (2010) can be added to capacity below Iron Gate estimated via the retrospective stock recruitment analysis.
7. Chinook in the Lower Basin below Keno will be predominantly ocean type.
8. Chinook in the Upper Basin above Keno will be a mixture of ocean and stream type; the relative proportion of each type will vary annually.
9. The Sycan Marsh on the Sycan River and the Klamath Marsh on the Williamson River are barriers to Chinook migration.
10. Implementation of KBRA in the EDRRA model assumes that the conditions in the Klamath River will improve over the 50 year time period of the model. This process was modeled by removing the chance for low productivity in later years of the time series. In future years, the likelihood that the Klamath would act like the worst rivers in Liermann et al. (2010) diminishes.
11. Annual variability in production of age 3 ocean recruits will be highly correlated in the upper and lower basin.
12. Flow variability in the Klamath River will affect production of Chinook in the upper basin to a similar degree as it affected survival of IGH hatchery fish. Namely, the posterior distribution on γ_{IGH} was used as a posterior predictive distribution on the effect of flow on the production in the tributaries to UKL.
13. Under the active reintroduction of the upper basin, production assumes adult abundances at or above the unfished equilibrium population size for the period 2019-2029.
14. Default values provided in the KHRM (described in Mohr et al. In prep.) for maturation rates, ocean survival rates, etc. were appropriate for future Klamath Basin Chinook stocks.
15. The fishery management is the same for DRA and NAA (please see Appendix A). Further, it is fixed for the time period of the model simulations.
16. The fishery is managed with perfect information; that is, fishery managers have perfect information of the abundances at each age and the proportion of hatchery fish in each age.
17. The fishery operates perfectly; that is, the allocated catch from the fishery managers is caught to meet the target harvest and escapement levels.

3 RESULTS

3.1 RETROSPECTIVE ANALYSIS

The Ricker stock-recruitment function with the index of common variation (CVI) provided a better explanation of the variability in the age 3 ocean recruitment ($DIC = 662.8$, $pD = 25.3$, mean deviance = 637.5) than the base model ($DIC = 683.4$, $pD = 28.5$, mean deviance = 654.9). The difference in DIC values was approximately 20 units, which is strongly supportive of the alternative model (Spiegelhalter et al. 2002). The difference in DIC values was due primarily to a decrease in mean deviance in the model, indicating an improvement in the prediction of age 3 ocean abundance by including the CVI as a covariate. Scale reduction factors indicated that samples were occurring from a stationary distribution in both models (i.e., values were near 1 for parameter estimates in both models). Observed versus predicted plots under the alternative model indicated that predicted median ocean age 3 abundances were indicative of observed abundances, but as may be expected with fitting spawner-recruit relationships (e.g., Hilborn and Walters 1992), some additional variability remained to be explained (Figure 4).

The CVI was estimated by capturing annual variability in hatchery survival common to both the IGH and TRH fingerling release groups (Figure 5). Much of the annual variability in survival of IGH and TRH releases was due to the common source of variability between the two hatcheries (Figure 6), with some remaining variability due to hatchery specific factors. Estimates of the standard deviation of the CVI provide an indicator of the magnitude of

the effect on hatchery survival. For example, TRH survival rates could vary from 4.8% to 0.38% for a 1 standard deviation increase and a 1 standard deviation decrease in the value of CVI, respectively.

Mean survival to age 2 was higher for TRH releases (1.35%) than IGH releases (0.9%) (values obtained by transforming mean values of κ in Table 5). Summer flows in the Trinity River in July at Lewiston were positively related to annual variability in survival of TRH releases; the posterior distribution of γ_{TRH} had a mean value of 0.3 (95% CrI: -0.038, 0.613, Table 5). Although the 95% CrI included zero, there was a 0.963 probability that flow was positively related to hatchery survival. Summer flows in the Klamath River (July flows at Seiad) were positively related to variability in IGH releases. The posterior distribution of γ_{IGH} was positive and the 95% CrI did not include 0 (Table 5); the probability of higher flows having a positive relationship with IGH survival in the Klamath River was ≥ 0.999 .

The common variability index (CVI) was variable among years and matched the pattern in log hatchery survival rates (Figure 6). While the pattern in the CVI may be informative, it is not known whether the magnitude of annual deviations is the same for natural recruitment to the age 3 ocean stage. A parameter was included in the model to allow the variability from the hatchery fish (CVI) to be scaled to the natural recruitment via δ . The inclusion of the δ parameter also allowed the stock recruitment function to ignore the CVI (e.g., if the δ value was 0). Median posterior estimates of δ were 0.61 (95%CrI: 0.32, 0.93) indicating that there was a positive relationship between recruitment variability and CVI, i.e., years with higher survival of TRH and IGH fingerlings were concurrent with positive deviations from the mean stock recruitment relationship.

The result of the retrospective model was a stock production function that could be used to forecast the levels of production with uncertainty for the Klamath basin below Iron Gate Dam in the No Action Alternative. The uncertainty in the stock production function is substantial, even in the absence of the CVI effect (i.e., assuming CVI = 0) (Figure 7). The fishery reference points indicate the levels of uncertainty in the stock recruitment relationships (Table 6). The spawning abundance that maximizes yield is approximately 48,000 spawners (95%CrI: 34,924, 86,141). The level of spawner abundance that maximizes recruitment has a median of 58,360 (95%CrI: 39,325, 109,167), whereas the median spawner abundance that equals the abundance of 3 years old in the ocean was estimated at 143,660 (106,407; 232,915).

The Liermann et al. (2010) model was also calculated for the lower basin assuming a watershed area of 9,653 km² (assuming a total watershed area of 12,066 km² for the Salmon, Shasta, Scott, Lower & Upper Klamath below Iron Gate and removing 20% due to watershed area draining directly into anadromous streams, D. Chow, NMFS, pers. comm.). This was completed to provide a point of comparison between the Liermann et al. (2010) approach and existing estimates of S_{msy} in the lower basin. The Liermann et al. (2010) median estimates of S_{msy} assuming a 9,653 km² watershed was 43,360 (95%CrI: 17,905, 95,500). In comparison, STT (2005) estimated S_{msy} to the adult stage as 40,700 (95% confidence interval: 32,200, 54,100). This result suggests relatively good agreement between Liermann and the STT (2005) analysis.

3.2 SPAWNER RECRUITMENT FUNCTIONS FOR DRA

3.2.1 LOWER KLAMATH BASIN

Under DRA the spawning habitat was increased by 1790 km², which equated to an adult unfished equilibrium population size of 23,613 (95% CrI: 11,063.1; 47,625.1) (Liermann et al. 2010). The adults were expanded into age 3 ocean recruits, which lead to redefining the capacity parameter in the Ricker stock recruitment relationship. The stock recruitment relationship in the lower basin shifted due to the added capacity in 2020 (Figure 8). As a result, the fishery reference points shifted to higher median levels (Table 7) with the median S_{msy} of 63,838 under DRA as compared to 48,475 under NAA and median S_{max} of 79,623 under DRA versus 58,361 under NAA.

These results were computed in the absence of KBRA to provide estimates of changes in the stock production function early in the time series.

The stock production function in the lower basin shifted over the time series due to KBRA actions affecting productivity in the lower basin tributaries (Stillwater Sciences 2010). The stock production function in 2012 was thus different than in 2061 due to the portion of the posterior distribution of α' that was sampled (Figure 9). As a result, the stock recruitment relationship shifted over the time series such that median recruitment was higher in 2055 relative to 2025, although uncertainty in recruitment remained largely unchanged (Figure 10).

3.2.2 TRIBUTARIES TO UPPER KLAMATH LAKE

The stock production function in the upper basin was derived from assuming mixed stream and ocean Chinook life history types and sampling log productivities from posterior predictive distributions provided in Liermann et al. (2010). The median log productivity from assuming the mixed life history r_{new} was 1.69 (95%CrI: 1.14; 2.24). The median estimate of unfished equilibrium population size for the tributaries to UKL using the results of Liermann et al. (2010) was 17,232 (95%CrI: 8,330; 30,439) for stream type and 53,691 (95%CrI: 23,598; 98,891) for ocean type Chinook, whereas the mixed ocean and stream type estimate was 34,350 (95%CrI: 12,964; 73,304). Restoration work in the tributaries to UKL was assumed to alter the distribution of r_{new} between 2012 and 2061 such that lower values of log productivity became less likely over this period (r_{new}^*) (Figure 11). As a result, the stock recruitment relationship (defined from spawner to age 3 in the ocean) in 2055 had higher recruitment of age 3 ocean Chinook for a given spawner abundances when compared to the stock recruitment relationship in 2025 (Figure 12). The difference between the 2025 and the 2055 stock recruitment relationships was most pronounced at spawner abundances less than approximately 33,000.

3.3 COMPARISON OF ALTERNATIVES

To support the decision process, the relative benefits of performing one action over another in the face of parametric and environmental uncertainty were calculated. Because the model iterations were paired (i.e., the same values of $CVI_{i,t}$, the same value of δ_i , the same value of $u_{i,j,t}$, etc. for hatchery j , iteration i in year t in NAA as in DRA), the probability that DRA was greater than NAA could be calculated (i.e., the number of model iterations in which DRA was greater than NAA). If there is no benefit to one action over the other, the probability will be 0.5 (i.e., 50:50 chance of higher abundance); however, if the probability is consistently greater than 0.5, then there is support for DRA despite uncertainty in the absolute abundance forecast.

I also calculated the percentage increase in abundance for each paired iteration as $(DRA - NAA)/NAA * 100\%$, which provided a quantitative estimate of the difference in abundance. There were three periods that could have different relative levels of abundance under DRA versus NAA: the period between model initiation and dam removal (2012-2020); the period after dam removal but with active reintroduction in the tributaries to UKL (2021-2032); and the final period when the population in the tributaries to UKL are assumed to be established and Iron Gate Hatchery production has ceased (2032-2061).

Escapement in the absence of fishing was calculated by the KHRM prior to determining the harvest rate, and it provided an estimate of total escapement to the Klamath Basin. The probability that forecasted escapement in the absence of fishing is higher under DRA than NAA between 2012 and 2020 is 0.54 (median of the annual probabilities from 2012-2032) (Figure 13). The probability is 0.79 from 2021-2032 and 0.78 from 2033 to 2061 that forecasted escapement under DRA was higher than NAA (Figure 13). The percentage increases in escapement of DRA relative to NAA in these three periods were 10.8% (2012-2020), 81.8% (2021-2032) and 81.4% (2033-2061) (Table 8).

Escapement to the Lower Klamath Basin was marginally higher under DRA than NAA (Figure 14). The probability that forecasted escapement to the Lower Klamath basin under DRA was greater than NAA was 0.50 between 2012 and 2020. The probability of DRA being greater than NAA was 0.54 and 0.56 for the periods 2021-2032 and 2033-2061, respectively (Figure 14). Over these three periods, the median percentage increases in escapement to the lower basin in DRA relative to NAA were approximately 7% to 9% after 2021 (Table 8).

Due to the structure of the KHRM, ocean recreational and ocean commercial harvest had the same relative response of DRA versus NAA (Figure 15 and 16). The probability of increased ocean harvest from 2012 to 2020 was 0.54. The improvement above 50% during the early period was due to KBRA restoration actions. After dam removal and during active reintroduction (2021-2032), the probability that ocean harvest was greater in DRA than NAA was 0.79. The probability of higher harvest dropped slightly to 0.72 with the cessation of active reintroduction and the loss of Iron Gate Hatchery production after 2032 (Figures 15 and 16). Median estimates of the percentage increase in ocean harvest due to DRA was approximately 9% from 2012 to 2020, rising to 63% from 2021 to 2032, and dropping to 46.5% after 2033 (Figure 15 and 16, Table 8).

Patterns in river harvest were similar to those for lower basin escapement, with relatively small increases in river harvest under DRA versus NAA (Figure 17). Prior to 2020, river harvest was roughly equivalent for NAA and DRA. The probability that DRA was greater than NAA was 0.48 prior to dam removal in 2020 (but equal to 0.5 if one includes the iterations where DRA equals NAA). After dam removal, the probability of increases in river harvest under DRA was consistent at 0.62. The pattern in river harvest was due to a 25,000 limit on capacity of recreational fishers (Mohr In prep), which minimized the amount that the DRA and NAA runs could differ. As a result, the median percentage increases in DRA relative to NAA runs were 0% during the early period (2012-2020) and increased to approximately 9% after dam removal (Table 8).

Tribal harvest was similar in pattern to ocean harvest (Figure 18), which reflected the fishery allocation rules incorporated into the KHRM. The probability of tribal harvest increasing under DRA was 0.54 prior to 2020, increasing to 0.79 during the active reintroduction period (2021-2032) and dropping down to 0.72 afterwards (Figure 18). Median estimates of the percentage increase in tribal harvest was roughly 10% before 2020, climbing to 71.5% during 2021-2032, and dropping to 54.8% thereafter.

4.0 DISCUSSION

The forecasted levels of escapement and harvest are determined by KHRM; therefore, understanding how KHRM operates provides some insight into the relative levels of escapement and harvest forecasted under NAA and DRA. The main driver of the KHRM behavior is the F- control rule, and the rule used in the forecasts under NAA and DRA is an updated amendment 16 rule (Appendix A). This rule is based on an optimal (i.e., escapement that produces maximum sustainable yield) escapement target after harvest of 40,700 (STT 2005). The updated F-control rule was developed to maximize yield under the current conditions (i.e., NAA), but it may not be optimal for DRA. The application of the updated rule to DRA affects the results here in two ways. First, given the additional recruitment to the fishery that arises from production in the Keno to Iron Gate reach and tributaries to UKL, the escapement and harvest forecasted under DRA were likely not managed optimally. Higher harvest and escapement (and potentially more consistent harvest and escapement) may be attainable by specifying an F-control rule optimized for the spawner recruitment relationships under DRA. Second, the probability of fishery closure was determined by F control rule and its escapement floor. There may be a trade-off between higher probability of closures and higher harvest rates that would need to be explored based on the spawner recruitment relationships for the lower and upper basins. Ultimately, any modification of the F-control rule would occur through a formal process under the Pacific Fishery Management Council, and modeling this process was well beyond the scope of this effort.

The KHRM was implemented in EDRRA with simplifying assumptions to highlight differences in the production under NAA and DRA. These assumptions affected the absolute estimates of harvest, and attempts to compare the harvest under NAA to historical catches may be misleading. Catch in the ocean and river fisheries between the mid 1990's through 2010 had a median value of 33,725 (PFMC 2011). Median forecasts of harvest under NAA presented here are well above the historical catches for at least two reasons. First, the ocean abundance supplied to the KHRM here is known without error; in other words, there is no error between the abundance in the preseason forecast and the postseason estimate. In reality, the level of error in preseason to postseason is not trivial, and the ratio of preseason forecast/postseason estimate of age 3 Klamath River fall Chinook has ranged from an overestimate of 2.5 to an underestimate of 0.34 in the period 1991 to 2010 (Table II-3 in PFMC 2011). As a result, the fishery management process used here was able to prescribe the exact numbers of fish to be harvested to reach the escapement objective. Second, the fishery described here operates perfectly; therefore the numbers of fish prescribed to be captured to meet the escapement objective are actually captured with perfect accuracy. The result of these two simplifying assumptions of the management and the fishery are that the escapement returning to spawn is close to 40,700 in most years (median of 42K under NAA) which means that the stock is close to S_{msy} under NAA and producing optimally.

I estimated a spawner recruitment relationship from spawners to age 3 ocean fish using historical data on the Klamath Basin that was similar in many respects to STT (2005). Although the recruitment was defined to different locations in the life history (to age 3 in the ocean here, whereas STT (2005) defined recruitment as adult escapement), the fishery reference points S_{msy} and S_{max} can be compared. The bias adjusted mean estimate of S_{msy} calculated in STT (2005) was 40,700 (95% confidence interval [CI]: 32,200; 54,100) and the bias adjusted mean estimate of S_{max} was 56,900 (95%CI: 42,400; 84,200). The reference points estimated in the Bayesian analysis here (Table 6) were higher with broader 95% *credible* intervals relative to the 95% confidence intervals in STT (2005). In particular, the median estimate of S_{msy} was 48,475 in the Bayesian analysis was higher than the bias adjusted mean estimate of 40,700 (STT 2005). If the distributions were the same, the median would be expected to be below the bias adjusted mean due to the shape of the lognormal distribution. Thus although the bias adjusted mean of S_{max} in STT (2005) and the Bayesian analysis are similar, the level of S_{max} implied by the Bayesian analysis was larger than in STT (2005). It is not surprising that the levels of S_{msy} and S_{max} differ between the two approaches. First, the estimation of the stock recruitment relationship to an earlier life stage in the Bayesian analysis (age 3 in the ocean) will affect the estimates of log productivity. Second, the annual variability in productivity was characterized differently in the Bayesian analysis than in STT (2005) which also affected log productivity estimates. Reference points that use the estimated log productivity (e.g., S_{msy}) will be affected by the difference in log productivity estimates.

Finally, one advantage of the Bayesian analysis is the incorporation of parameter uncertainty into the estimation approach as probability distributions (Gelman et al. 2004). Derived quantities of the model can then be computed as probability distributions by integrating over the uncertainty in the parameters. The full posterior distribution on the derived quantity can then be evaluated for inference (e.g., McAllister et al. 1994, Punt and Hilborn 1997, Liermann et al. 2010). Analyses of similar data sets under Bayesian and frequentist approaches may result in different results depending upon the marginal likelihood of the coefficient estimate. When the information in the data on a particular parameter value are informative, the difference between Bayesian and frequentist inference will be small; however, when the information on the parameter is limited (e.g., for parameters such as $S_{max} = \beta^1$ estimated from spawner recruitment data), the differences between the two approaches are likely to be greater. For this reason, comparison of approaches under Bayesian and frequentist approaches may provide different inference, and almost always indicate greater uncertainty in the value of the derived quantities in the Bayesian analysis (Gelman et al. 2004, Congdon 2002).

In the process of developing the tools for evaluating NAA and DRA, I computed estimates of equilibrium population sizes for the tributaries to UKL and the reach from Iron Gate Dam to Keno Dam. The median estimates

of unfished equilibrium population size using the Liermann et al. (2010) posterior distributions was approximately 23,000 ocean type Chinook in the Keno to Iron Gate reach and approximately 35,000 stream and ocean type Chinook in the tributaries to UKL. There are several other estimates of equilibrium unfished or fished population sizes for both the tributaries to UKL and the Iron Gate to Keno reach that can be used to put the estimates computed here into context. Most recently, Lindley and Davis (In prep) estimated an equilibrium *fished* population size of 720 for the Keno to Iron Gate reach and an estimate of 2372 for the tributaries of UKL (Wood, Williamson, and Sprague Rivers). Further they compare their estimates to calculations of equilibrium unfished population abundances in Liermann et al. (2010) using assumptions consistent with their model. The assumptions in Lindley and Davis (In prep) differ than those made here with respect to accessibility to portions of the watershed and the spatial structure of Chinook populations once they become established; therefore calculations using parameters in Liermann et al (2010) are not directly comparable between the two works. Finally, Dunsmoor and Huntington (2006) developed a tabular summary of aquatic habitat conditions in the Upper Klamath Basin with particular emphasis on areas above UKL. They estimated that current habitat conditions above Iron Gate Dam could support approximately 14,864 spawning fall Chinook salmon and 32,706 spawning spring Chinook salmon. Huntington (2006) developed estimates of adult Chinook to the Klamath Basin upstream of IGD using five different methods and estimated between 9,180-32,040 Chinook. These estimates are roughly comparable to the 10,000 to 50,000 levels of Chinook escapement upstream of Iron Gate Dam calculated under EDRRA.

Ultimately, the specifics of how anadromy would be restored to the Klamath Basin will require additional planning, and there are many details that were excluded from this analysis by necessity. There are several factors that have been discussed as potentially modifying the degree to which anadromy may be restored to the Upper Klamath Basin. Water quality in UKL can be problematic for salmonids with summer temperatures exceeding 25 C and dissolved oxygen levels at 4mg/L or below during the summer (Wood et al. 2006). Thus, the conditions in UKL may be a factor in determining the type of life – history strategies that are successful due to acceptable windows into and out of the tributaries to UKL. *Ceratomyxa shasta* currently affects natural origin juveniles migrating through the mainstem Klamath River. The prevalence of the disease appears to be tied to the density of the polychaete host and the flow and temperature conditions under which juveniles may be exposed to the parasite (Bartholomew and Foott 2010). The parasite *C. shasta* is also located in the Williamson River (Bartholomew and Foott 2010), although the strain there is not virulent to Chinook. It is not known whether the strain that is virulent to Chinook will become established in the tributaries to UKL and affect the production potential of those tributaries.

Still, recent studies suggest that with the provision of suitable passage facilities at downstream dams or dam removal, Chinook salmon could be re-introduced and restored to waters in the Upper Klamath Basin (Dunsmoor and Huntington 2006; Hooton and Smith 2008; Butler et al. 2010); further, substantial historical evidence shows that both Chinook salmon and steelhead trout historically used the streams of the Upper Klamath Basin for spawning and for juvenile rearing (Hamilton et al. 2005; Fortune et al. 1966). Finally, NMFS and USFWS required anadromous fish passage as a condition for issuing a Federal Energy Regulatory Commission (FERC) license to operate the dams; thus, restoration of anadromy to the upper Klamath Basin will be an important part of the FERC relicensing process.

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Table 1. The recruit and spawner data presented in Table A1 of STT (2005). BY denotes brood year; $N_{3, \text{Sept1}}$ denotes the abundance of progeny spawned by S in calendar year BY that survive to become ocean age 3 on September 1 in calendar year 3.

| BY | $N_{3, \text{Sept1}}$ | R_3 | R_4 | R_5 | R | S | R/S |
|------|-----------------------|--------|--------|-------|--------|--------|------|
| 1979 | 423701 | 42235 | 137103 | 21360 | 200698 | 30637 | 6.6 |
| 1980 | 236144 | 28082 | 56102 | 25246 | 109430 | 21484 | 5.1 |
| 1981 | 106338 | 16737 | 26354 | 7877 | 50968 | 33857 | 1.5 |
| 1982 | 277850 | 17331 | 61442 | 43414 | 122187 | 31951 | 3.8 |
| 1983 | 776743 | 73352 | 259838 | 34969 | 368159 | 30784 | 12.0 |
| 1984 | 512171 | 46576 | 181026 | 16450 | 244052 | 16064 | 15.2 |
| 1985 | 391378 | 52017 | 119909 | 16796 | 188722 | 25676 | 7.4 |
| 1986 | 256532 | 29759 | 84135 | 9353 | 123247 | 113359 | 1.1 |
| 1987 | 148910 | 20399 | 50415 | 2167 | 72981 | 101717 | 0.7 |
| 1988 | 37029 | 2871 | 13010 | 1569 | 17450 | 79385 | 0.2 |
| 1989 | 33368 | 4921 | 9962 | 1330 | 16213 | 43869 | 0.4 |
| 1990 | 85146 | 29185 | 13186 | 2539 | 44910 | 15596 | 2.9 |
| 1991 | 91590 | 29578 | 18478 | 457 | 48513 | 11649 | 4.2 |
| 1992 | 526545 | 129836 | 132474 | 7368 | 269678 | 12029 | 22.4 |
| 1993 | 177305 | 40102 | 48124 | 1984 | 90210 | 21858 | 4.1 |
| 1994 | 99535 | 24195 | 24978 | 1667 | 50840 | 32333 | 1.6 |
| 1995 | 72062 | 28271 | 10703 | 229 | 39203 | 161793 | 0.2 |
| 1996 | 74965 | 17305 | 21052 | 51 | 38408 | 81326 | 0.5 |
| 1997 | 327575 | 84784 | 76782 | 6523 | 168089 | 46144 | 3.6 |
| 1998 | 253386 | 62628 | 66021 | 1634 | 130283 | 42488 | 3.1 |
| 1999 | 406036 | 74558 | 89368 | 32271 | 196197 | 18456 | 10.6 |
| 2000 | 386121 | 60997 | 112628 | 14912 | 188537 | 82729 | 2.3 |

Table 2. Iron Gate Hatchery (IGH) and Trinity River Hatchery (TRH) fingerling early survival (May – August) after release, spawner abundance from the Klamath River (KR), Trinity River (TR), and Unknown (UN), and the weights for Klamath River (w_{KR}) and Trinity River (w_{TR}) and final survival index s' used in the STT (2005) analysis.

| BY | s'_{IGH} | s'_{TRH} | S_{KR} | S_{TR} | S_{UN} | W_{KR} | w_{TR} | s' |
|------|------------|------------|----------|----------|----------|----------|----------|--------|
| 1979 | 0.0522 | 0.0589 | 21141 | 8028 | 1468 | 0.725 | 0.275 | 0.0540 |
| 1980 | 0.0183 | 0.0071 | 12383 | 7700 | 1400 | 0.617 | 0.383 | 0.0140 |
| 1981 | 0.0329 | 0.0058 | 17517 | 15340 | 1000 | 0.533 | 0.467 | 0.0202 |
| 1982 | 0.0058 | 0.0133 | 21177 | 9274 | 1500 | 0.695 | 0.305 | 0.0081 |
| 1983 | 0.0279 | 0.0870 | 12230 | 17284 | 1270 | 0.414 | 0.586 | 0.0625 |
| 1984 | 0.0255 | 0.0656 | 9420 | 5654 | 990 | 0.625 | 0.375 | 0.0405 |
| 1985 | 0.0174 | 0.0814 | 12166 | 9217 | 4294 | 0.569 | 0.431 | 0.0450 |
| 1986 | 0.0011 | 0.0050 | 15893 | 92548 | 4919 | 0.147 | 0.853 | 0.0044 |
| 1987 | 0.0015 | 0.0047 | 26511 | 71920 | 3286 | 0.269 | 0.731 | 0.0038 |
| 1988 | 0.0010 | 0.0034 | 29783 | 44616 | 4987 | 0.400 | 0.600 | 0.0024 |
| 1989 | 0.0005 | 0.0004 | 10584 | 29445 | 3839 | 0.264 | 0.736 | 0.0004 |
| 1990 | 0.0235 | *0.0356 | 7102 | 7682 | 812 | 0.480 | 0.520 | 0.0298 |
| 1991 | 0.0045 | 0.0164 | 5905 | 4867 | 877 | 0.548 | 0.452 | 0.0099 |
| 1992 | 0.0447 | 0.0575 | 4135 | 7139 | 754 | 0.367 | 0.633 | 0.0528 |
| 1993 | 0.0018 | 0.0035 | 13385 | 5905 | 2568 | 0.694 | 0.306 | 0.0023 |
| 1994 | 0.0029 | 0.0070 | 20003 | 10906 | 1424 | 0.647 | 0.353 | 0.0043 |
| 1995 | 0.0028 | 0.0053 | 79851 | 77876 | 4067 | 0.506 | 0.494 | 0.0040 |
| 1996 | 0.0053 | 0.0106 | 31755 | 42646 | 6925 | 0.427 | 0.573 | 0.0083 |
| 1997 | 0.0668 | 0.0419 | 29015 | 11507 | 5622 | 0.716 | 0.284 | 0.0597 |
| 1998 | 0.0194 | 0.0083 | 16407 | 24460 | 1621 | 0.401 | 0.599 | 0.0128 |
| 1999 | 0.0263 | 0.0265 | 10883 | 6797 | 777 | 0.616 | 0.384 | 0.0264 |
| 2000 | 0.0123 | 0.0421 | 58388 | 24340 | 0 | 0.706 | 0.294 | 0.0211 |

* imputed value: $\hat{s}'_{TRH,1990} = \exp(0.89s'_{IGH,1990})$.

Table 3. Prior distributions for parameters in the Ricker stock recruitment function.

| Parameter | Prior |
|---------------------------------|--------------------|
| α | $N(0, 1000)$ |
| β | $N(0, 1000)$ |
| δ | $N(0, 1000)$ |
| $\kappa_i, j = \text{IGH, TRH}$ | $N(0, 1000)$ |
| $\gamma_i, j = \text{IGH, TRH}$ | $N(0, 1000)$ |
| σ_E^2 | $IG(0.001, 0.001)$ |
| σ_{CVI}^2 | $IG(0.001, 0.001)$ |
| σ_H^2 | $IG(0.001, 0.001)$ |

Table 4: Watershed area in tributaries of Upper Klamath Lake .

| Subbasin | Watershed Area in km² (mi²) |
|------------------------------------|--|
| Sycan | 1,447.2 (559) |
| Sycan downstream of the Marsh | 600.9 (232) |
| Sprague (lower, upper, and Sycan) | 4,092.2 (1,580) |
| Sprague without the Sycan | 2,644.4 (1,021) |
| Wood | 567.2 (219) |
| Williamson | 3,677.8 (1,420) |
| Williamson downstream of the Marsh | 311 (120) |

Table 5. Posterior distribution mean, median and end points for 95% credible interval (2.5% and 97.5%) for parameters in the Ricker stock recruitment function.

| Parameter | Mean | 2.5% | 50% | 97.5% |
|----------------|----------|-----------|----------|----------|
| α | 2.48 | 1.90 | 2.48 | 3.05 |
| β | 1.73e-05 | 2.54e-05 | 1.71e-05 | 9.16e-06 |
| δ | 6.12e-01 | 3.24e-01 | 6.03e-01 | 9.27e-01 |
| κ_{IGH} | -4.77 | -5.41 | -4.76 | -4.15 |
| κ_{TRH} | -4.30 | -4.89 | -4.30 | -3.74 |
| γ_{IGH} | 6.44e-01 | 3.35e-01 | 6.41e-01 | 9.41e-01 |
| γ_{TRH} | 3.06e-01 | -3.80e-02 | 3.12e-01 | 6.13e-01 |
| σ_E | 6.08e-01 | 3.65e-01 | 6.02e-01 | 8.85e-01 |
| σ_{CVI} | 1.27 | 8.78e-01 | 1.25 | 1.80 |
| σ_H | 5.07e-01 | 3.68e-01 | 4.93e-01 | 7.44e-01 |

Table 6. Probability distributions of the fishery reference points: spawner abundance that provides maximum sustainable yield (S_{msy}); spawner abundance that provides maximum recruitment (S_{max}); and the spawner abundance that is equal to recruitment at age 3 in the ocean (S_{ueq}).

| Reference Point | Median | 2.5% | 97.5% |
|------------------------|---------------|-------------|--------------|
| S_{msy} | 48,475 | 34,924.9 | 86,141.3 |
| S_{max} | 58,360.9 | 39,325.6 | 109,167.1 |
| S_{ueq} | 143,660.4 | 106,406.9 | 232,915.5 |

Table 7. Probability distributions of the fishery reference points for the Lower Klamath Basin after removing the four mainstem dams: spawner abundance that provides maximum sustainable yield (S_{msy}); spawner abundance that provides maximum recruitment (S_{max}); and the spawner abundance that is equal to recruitment at age 3 in the ocean (S_{ueq}). The stock production function used the same level of log productivity (α') as in Table 5.

| Reference Point | Median | 2.5% | 97.5% |
|-----------------|-----------|-----------|-----------|
| S_{msy} | 63,838.5 | 54,979.0 | 100,198.3 |
| S_{max} | 79,623.1 | 53,290.6 | 137,876.0 |
| S_{ueq} | 194,448.8 | 128,587.1 | 322,711.7 |

Table 8. Percent increase in abundance due to performing DRA versus performing NAA for three time periods: 1) prior to dam removal (2012 – 2019); 2) during active reintroduction in Upper Basin (2020-2029); and after active reintroduction ceases and Iron Gate Hatchery production ceases (2030-2061).

| | 2012 - 2020 | | 2021-2032 | | 2033-2061 | |
|--------------------------------------|-------------|--------------------|-----------|--------------------|-----------|--------------------|
| Metric | Median | 95%CrI | Median | 95%CrI | Median | 95%CrI |
| Escapement in the Absence of Fishing | 10.8% | -79.7%, 492.6% | 81.8% | -61.7%, 836.5% | 81.4% | -59.9%, 881.4% |
| Lower Basin Escapement | 0% | -72.2%, 385.7% | 6.7% | -77.5%, 474.8% | 9.2% | -75.8%, 489.6% |
| Ocean Commercial Harvest | 9.2% | -86.7%, 836.2% | 63.0% | -61.9%, 1618.9% | 46.5% | -68.7%, 1495.2% |
| Ocean Recreational Harvest | 9.2% | -86.7%, 836.2% | 63.0% | -61.9%, 1618.9% | 46.5% | -68.7%, 1495.2% |
| River Harvest | 0% | -92.3%, 1519.7% | 8.7% | -73.4%, 2778.1% | 9.1% | -77.4%, 2753.7% |
| Tribal Harvest | 10.3% | -88.6%, 1009.8% | 71.5% | -65.0%, 1948.2% | 54.8% | -71.0%, 1841.0% |

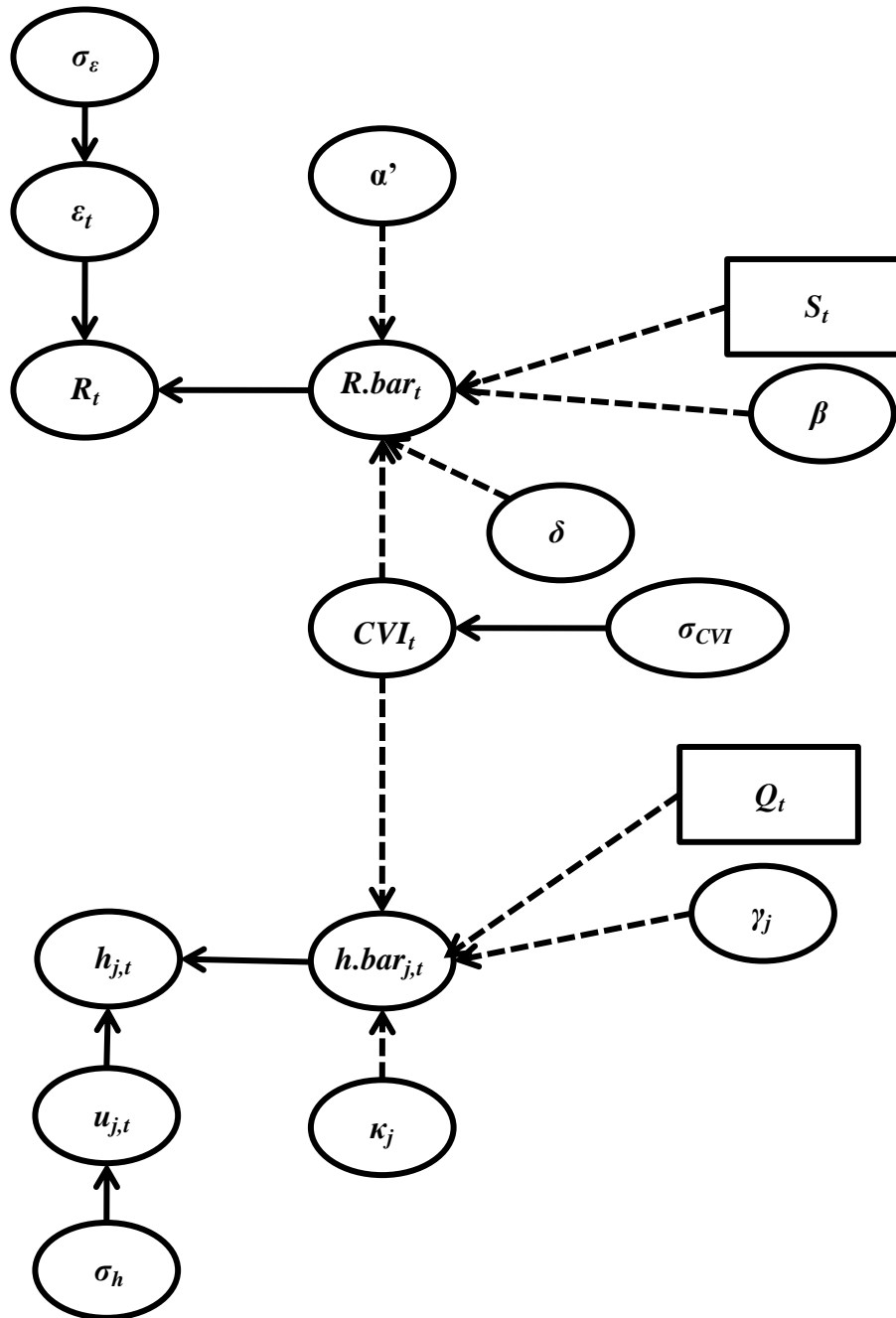


Figure 1. Directed Acyclic Graph (DAG) of the conditional relationships between coefficients in equations for estimating log hatchery survival rates (h) and natural recruitment to age 3 (R) as depicted in Equations 3 and 4. Ovals represent nodes that are calculated quantities whereas squares represent known quantities (i.e., covariates known without error). Solid lines indicate a stochastic relationship, whereas dashed lines indicate a deterministic one. All symbols the same as in Equation 3 and 4 except $h.bar_{j,t}$ which is the mean log survival rate of hatchery j in brood year t , and $R.bar_t$ which is the mean recruitment in brood year t . The figure shows the relationship of the common variability index (CVI) and its role in both the equation for Recruitment and for log hatchery survival.

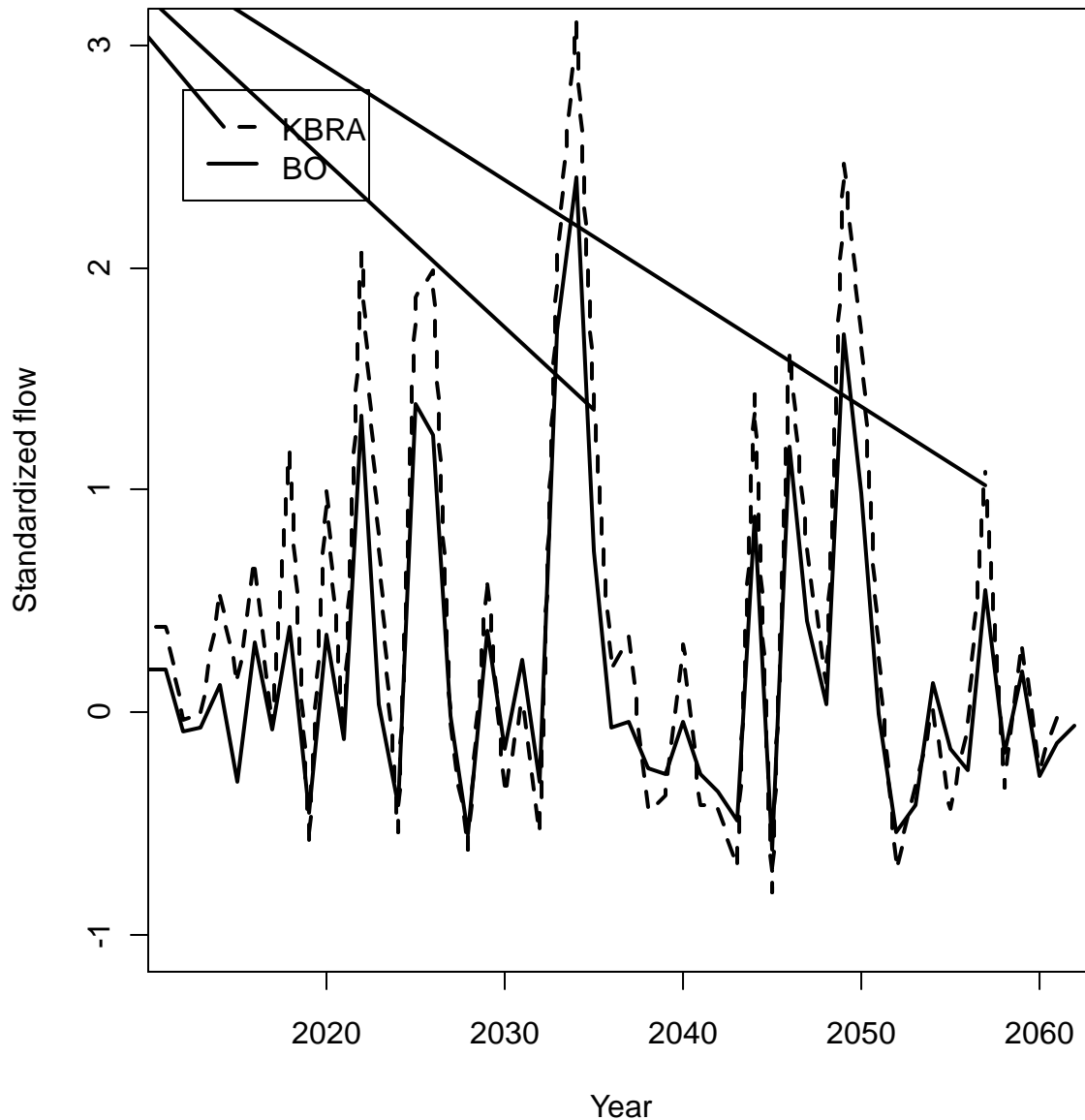


Figure 2. Flow forecasts for 2012 to 2061 under NMFS Biological Opinion (BO) and under KBRA in the Klamath River at Seiad during July. Flow values were standardized using the mean Klamath River flows in July at Seiad Valley from 1980 to 2000 (mean = 1589, sd = 944.17). The standardized flow values were incorporated in the model for forecasts of abundance and harvest under NAA and DRA.

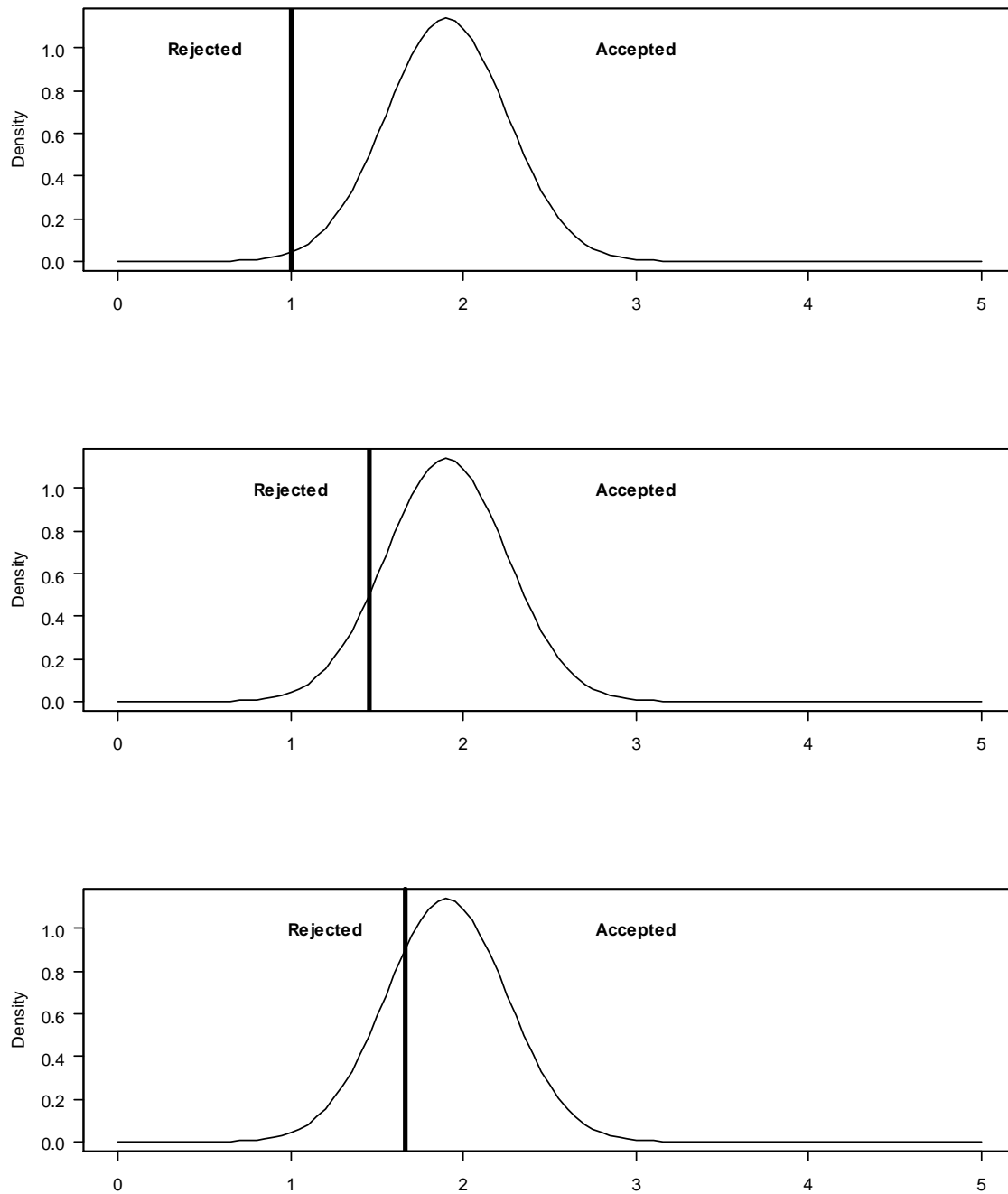


Figure 3. Depiction of sampling from higher percentiles of a hypothetical productivity distribution over time. Samples of productivity occur only from the Accepted region. Early in the time series, samples from almost the entire distribution are accepted (Top, Accepted threshold at 0.05 quantile). Later in the time series, the Accepted region is shifted to the right due to higher expected productivity (Middle, Accepted threshold at the 0.10 quantile). At the end of the time series the threshold for the Accepted region has again shifted towards higher productivity (Bottom, Accepted threshold at the 0.25 quantile).

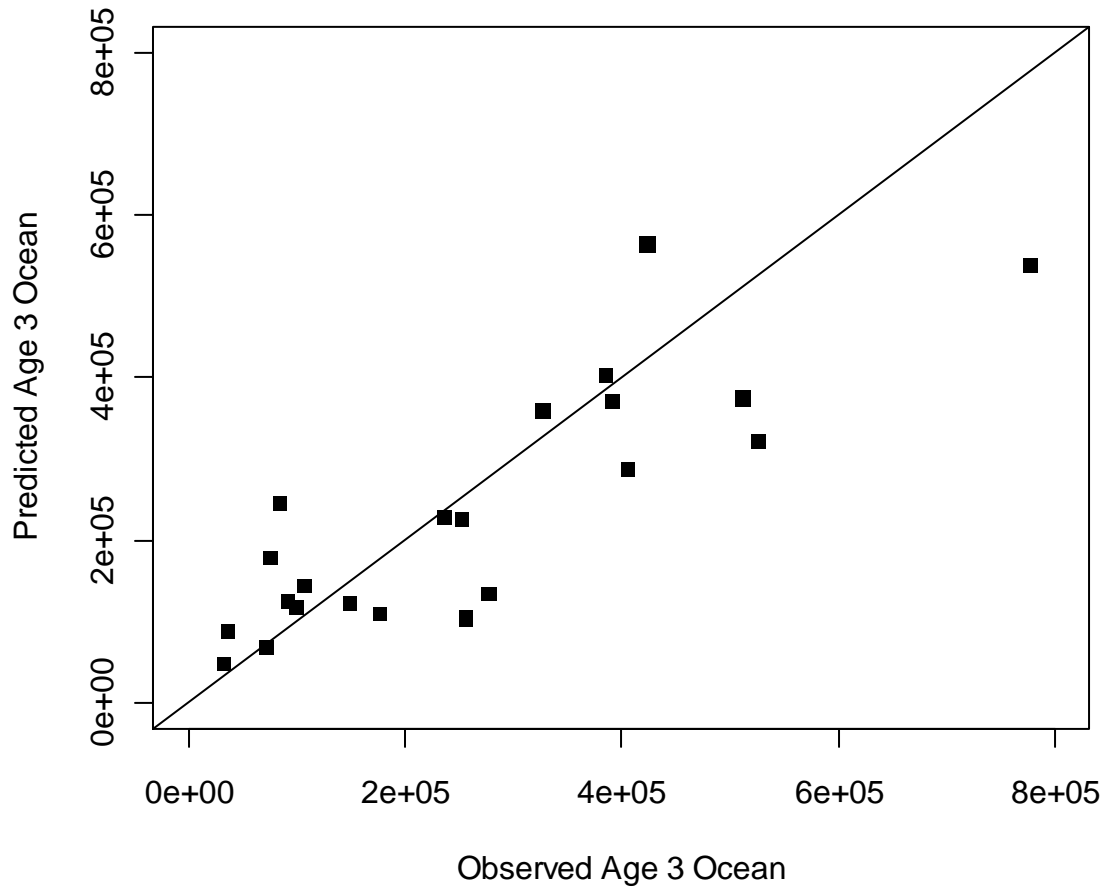


Figure 4. Median predicted ocean age 3 recruits from the Ricker stock recruitment model and observed ocean age 3.

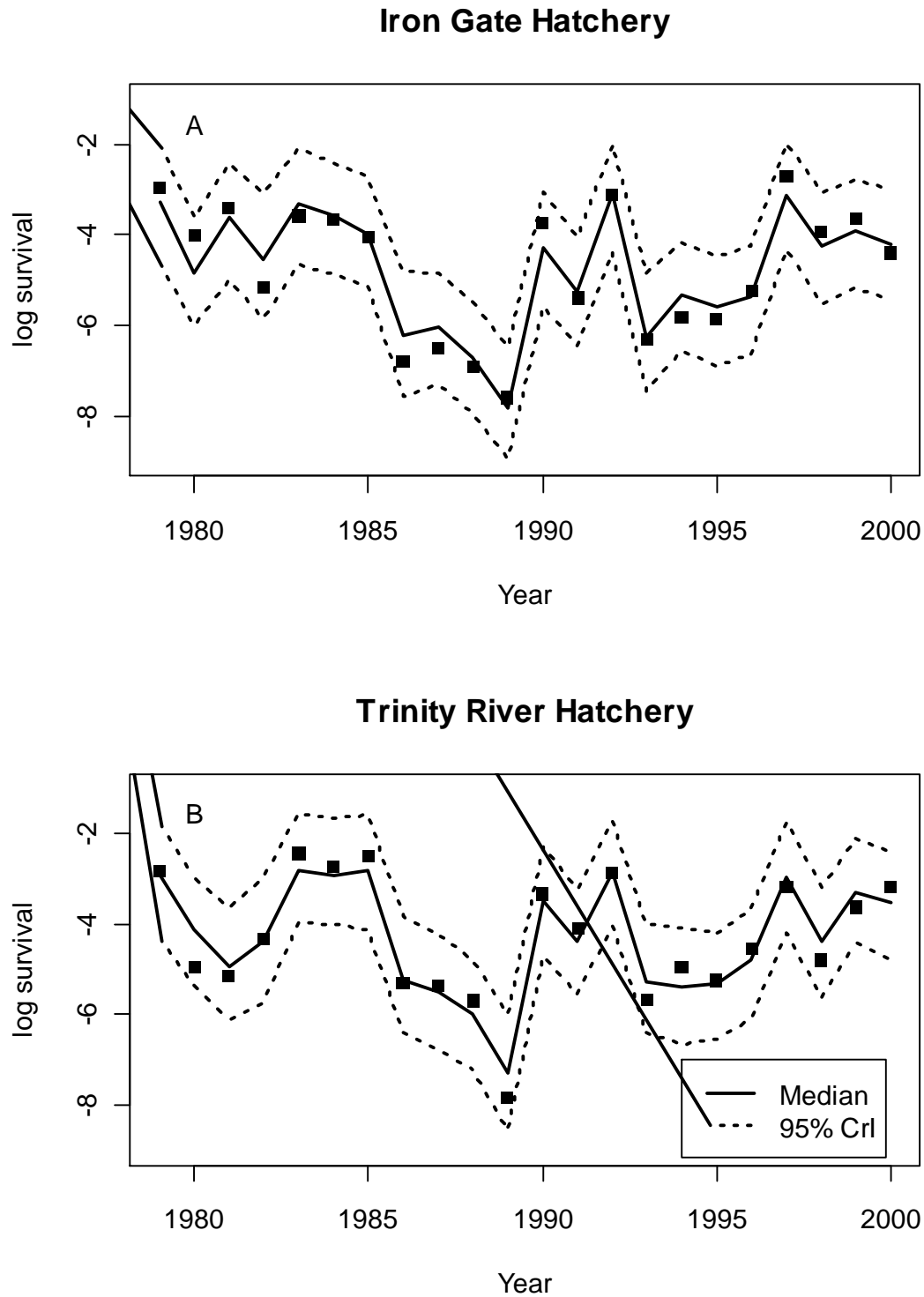


Figure 5. Observed log survival rates from Iron Gate Hatchery (A) and Trinity River Hatchery (B) with median model predictions and 95% credible intervals (95% CrI).

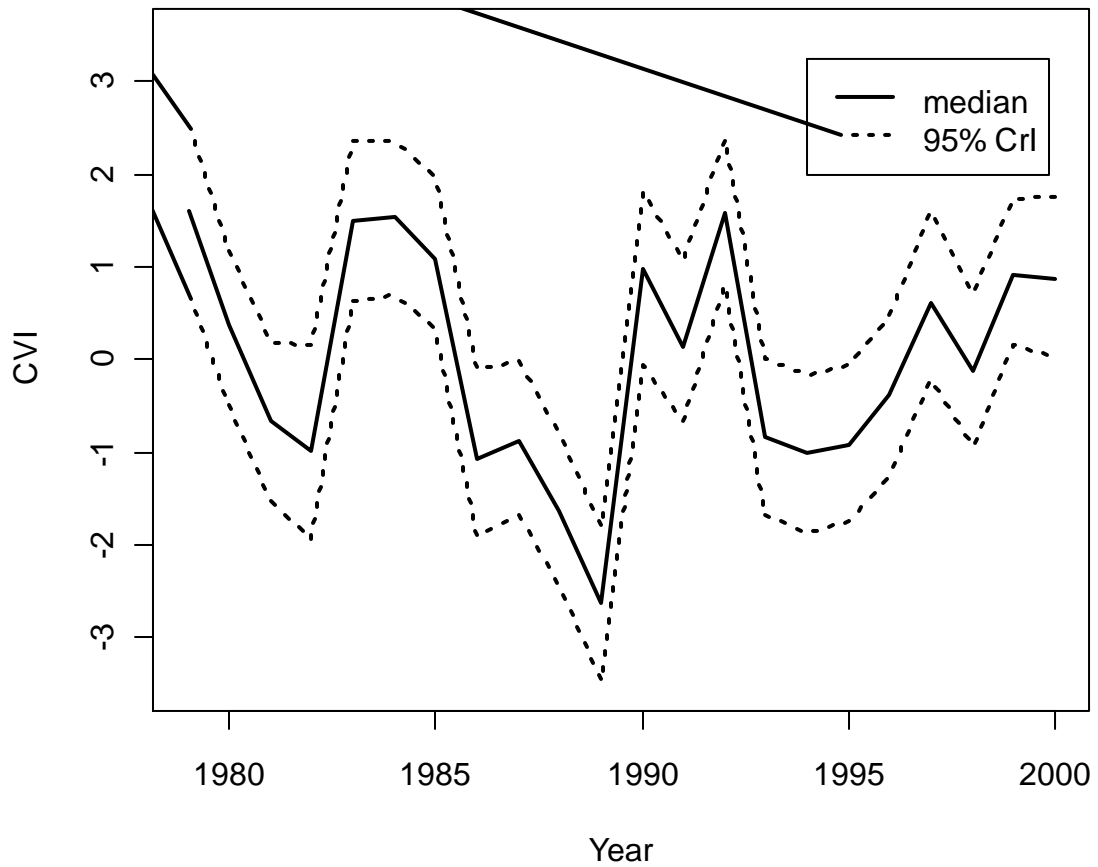


Figure 6. The common variability index (CVI) from 1979 to 2000, which is the annual variability common to both Iron Gate Hatchery and Trinity River Hatchery fingering CWT release groups estimated from log hatchery survival rates.

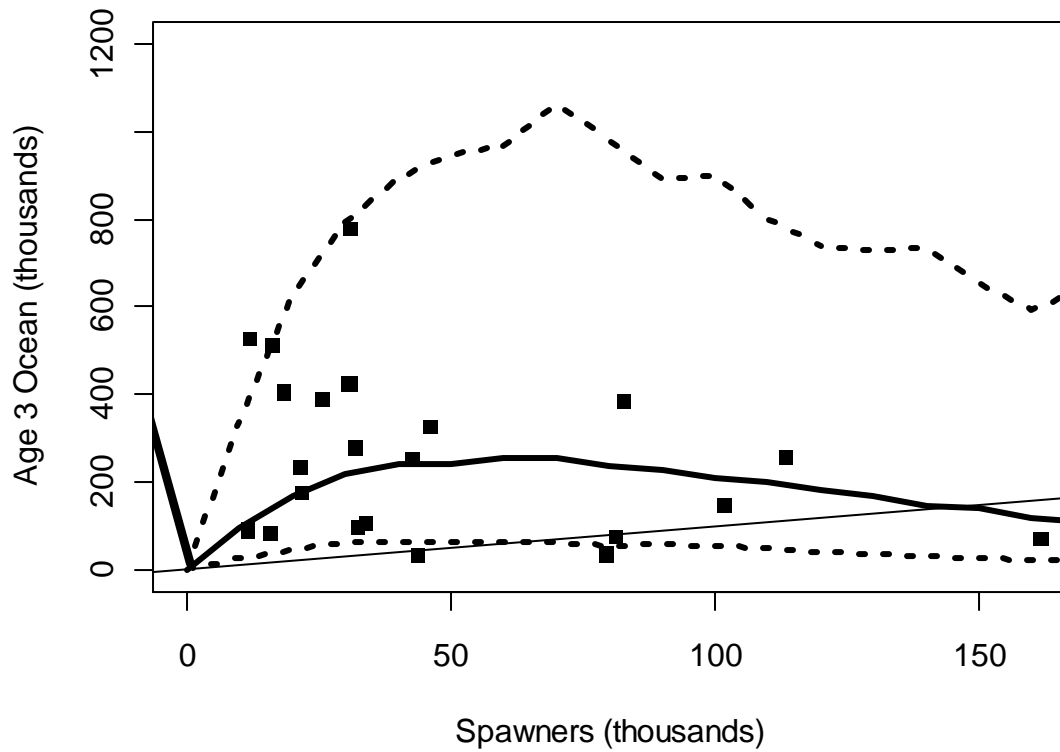


Figure 7. Estimated stock recruitment relationship between spawners and age 3 ocean abundance for brood years 1979 to 2000. Observed data (squares), median recruitment (dark solid line) and 95% credible interval (dashed lines), and the 1:1 line (thin solid line) are plotted. Model predictions assumed CVI equal 0.

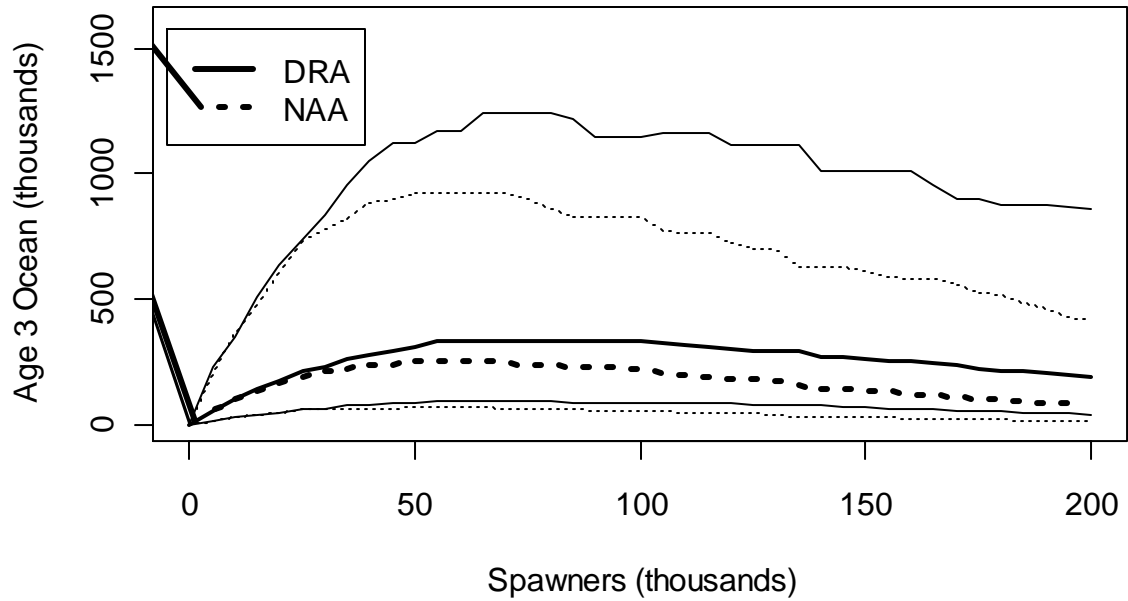


Figure 8. Lower Klamath Basin stock production relationship under the No Action Alternative (NAA) and under the Dam Removal Alternative (DRA). Median recruitment (dark lines) and 95% intervals (light lines) are plotted for production under the two alternatives. The DRA and NAA alternatives assume the same level of log productivity (α').

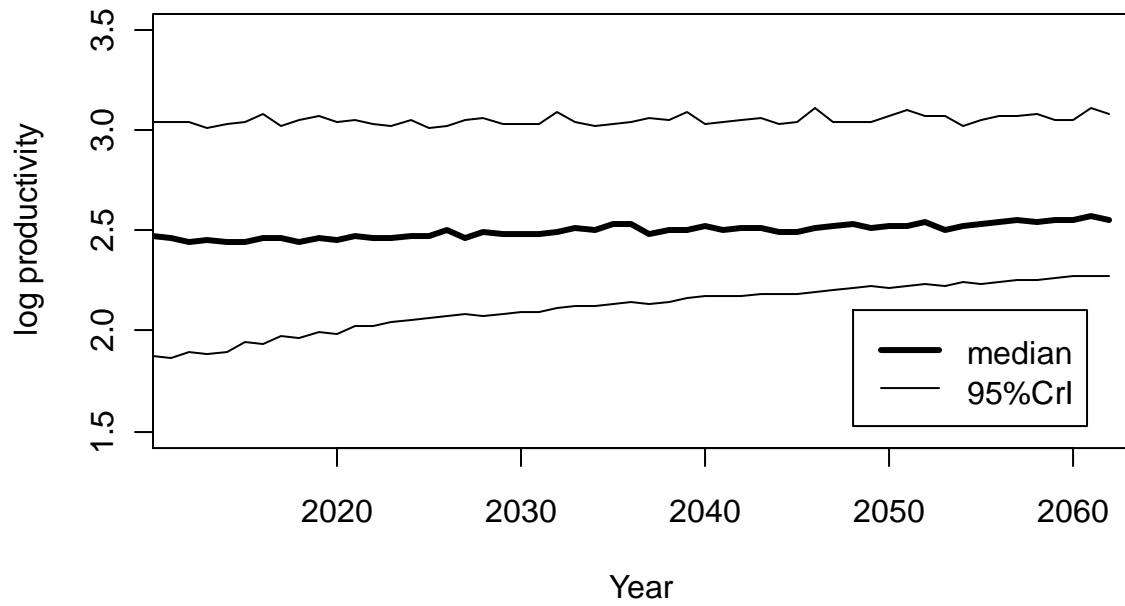


Figure 9. Distribution of .log productivity (α') in the lower Klamath Basin from 2012 to 2061 due to habitat restoration by KBRA.

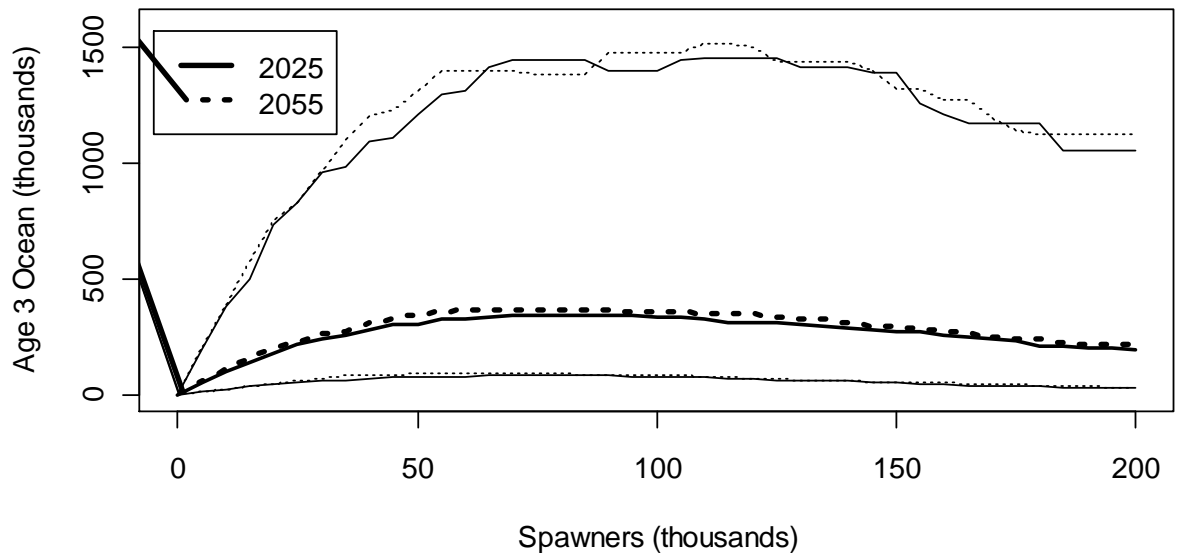


Figure 10. Stock recruitment relationship in the lower Klamath basin in 2025 and in 2055 including increase in habitat due to dam removal and KBRA actions affecting log productivity α^* . Median production (dark line) and 95%I (light line) are plotted for each of the two years.

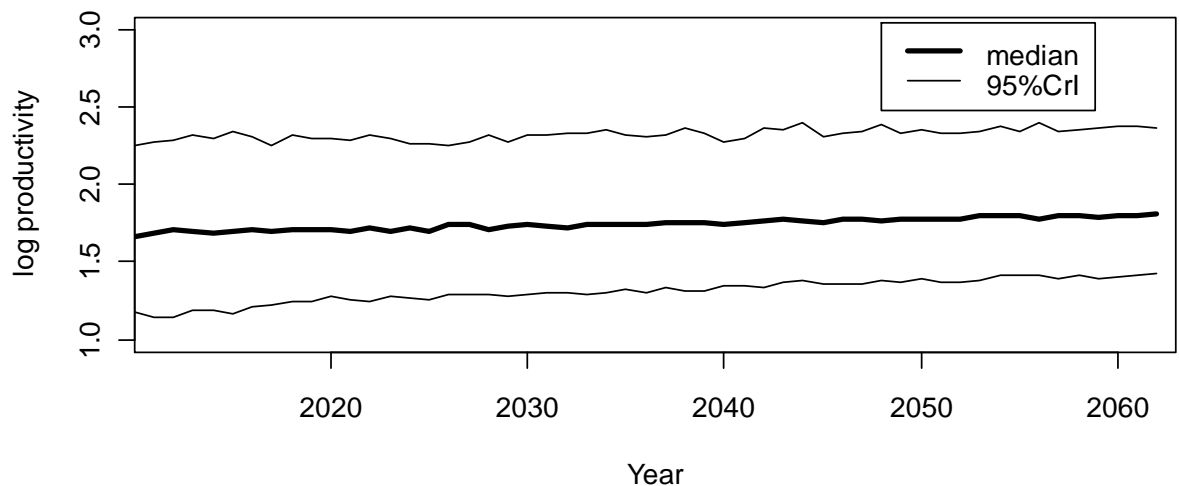


Figure 11. Distribution of log productivity (r_{new}^*) of a mixed stream and ocean type life history in tributaries to Upper Klamath Basin from 2012 to 206. Changes in log productivity over the time series are due to habitat restoration by KBRA.

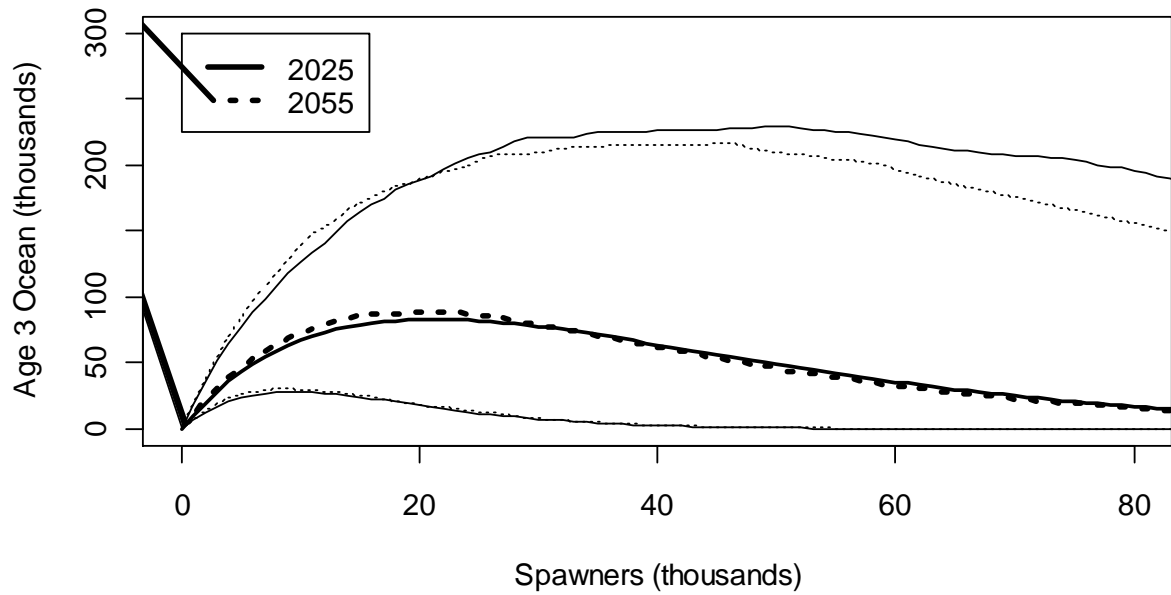


Figure 12. Stock recruitment relationship in the tributaries to Upper Klamath Lake in 2020 and in 2055 incorporating mixed stream and ocean type life history and KBRA actions affecting log productivity r_{new}^* . Median production (dark line) and 95%I (light line) are plotted for each of the two years.

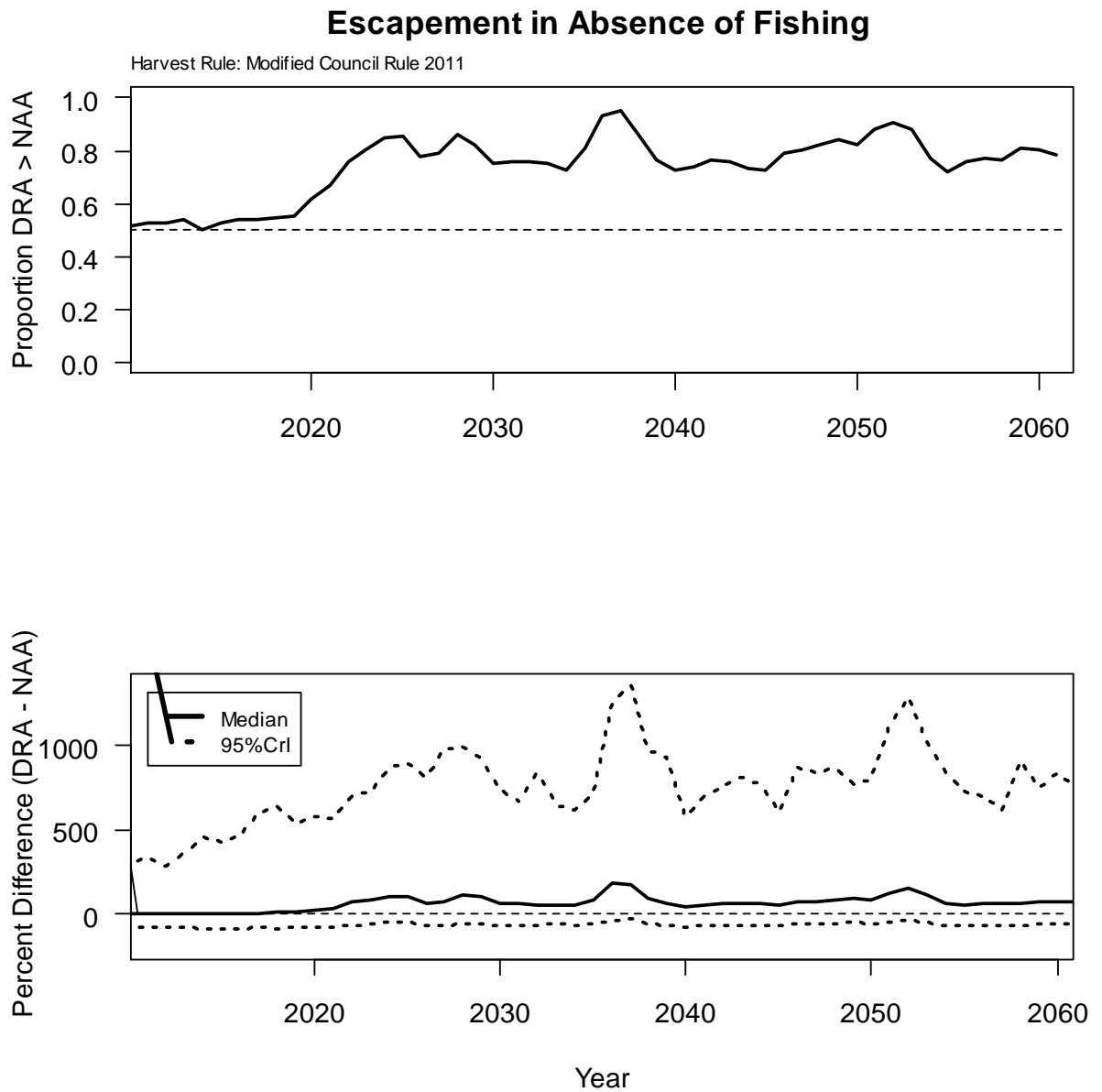


Figure 13. Probability that escapement in the absence of fishing is greater under DRA than under NAA from 2012 to 2061 (top). Dashed line represents 50/50 chance of increased abundance under DRA. Median and 95% credible intervals for the percent increase in DRA relative to NAA from 2012 to 2061 (bottom). Dashed line represents no difference between DRA and NAA.

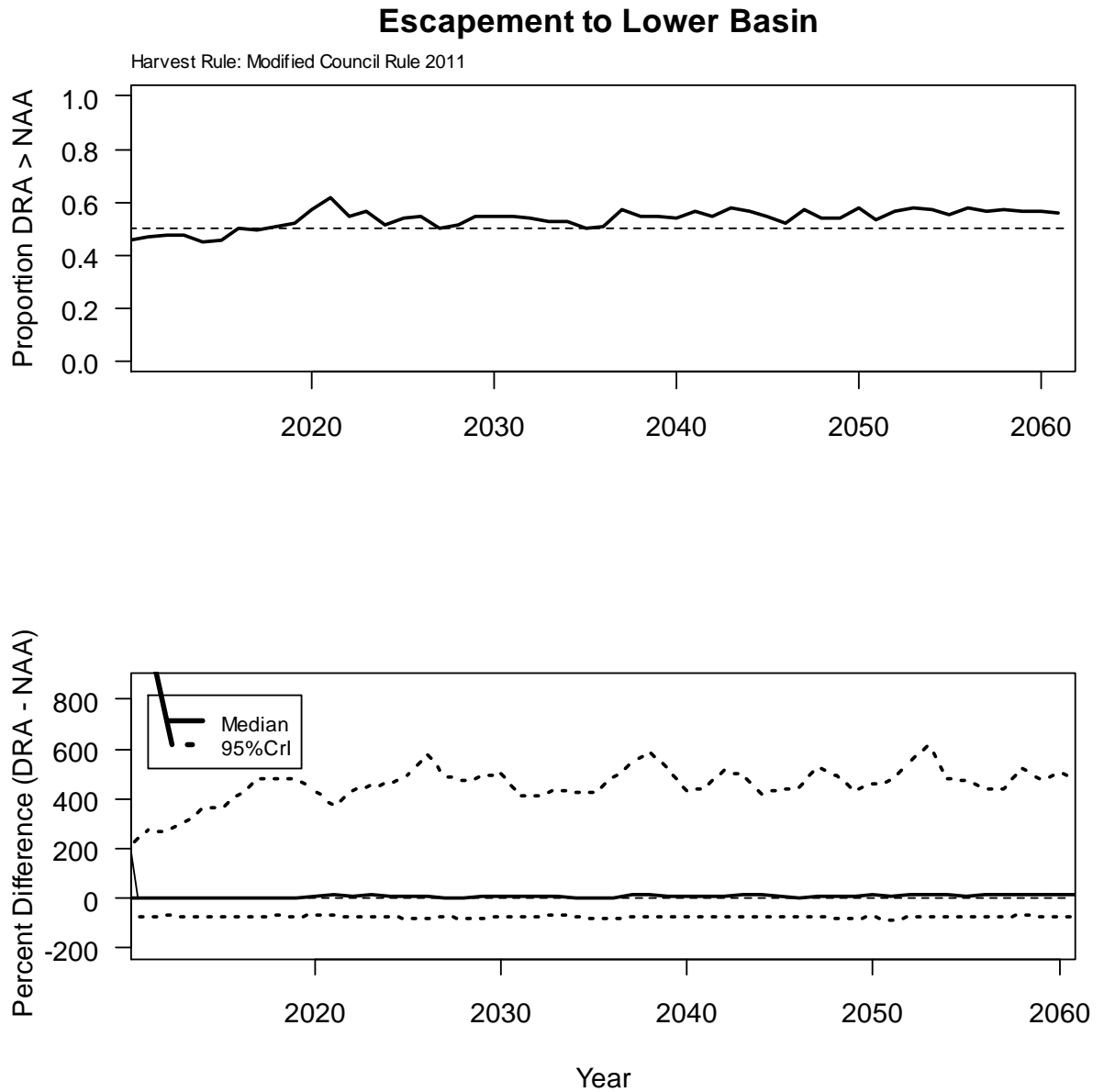


Figure 14. Probability that escapement to the Lower Klamath Basin is greater under DRA than under NAA from 2012 to 2061 (top). Dashed line represents 50/50 chance of increased abundance under DRA. Median and 95% credible intervals for the percent increase in DRA relative to NAA from 2012 to 2061 (bottom). Dashed line represents no difference between DRA and NAA.

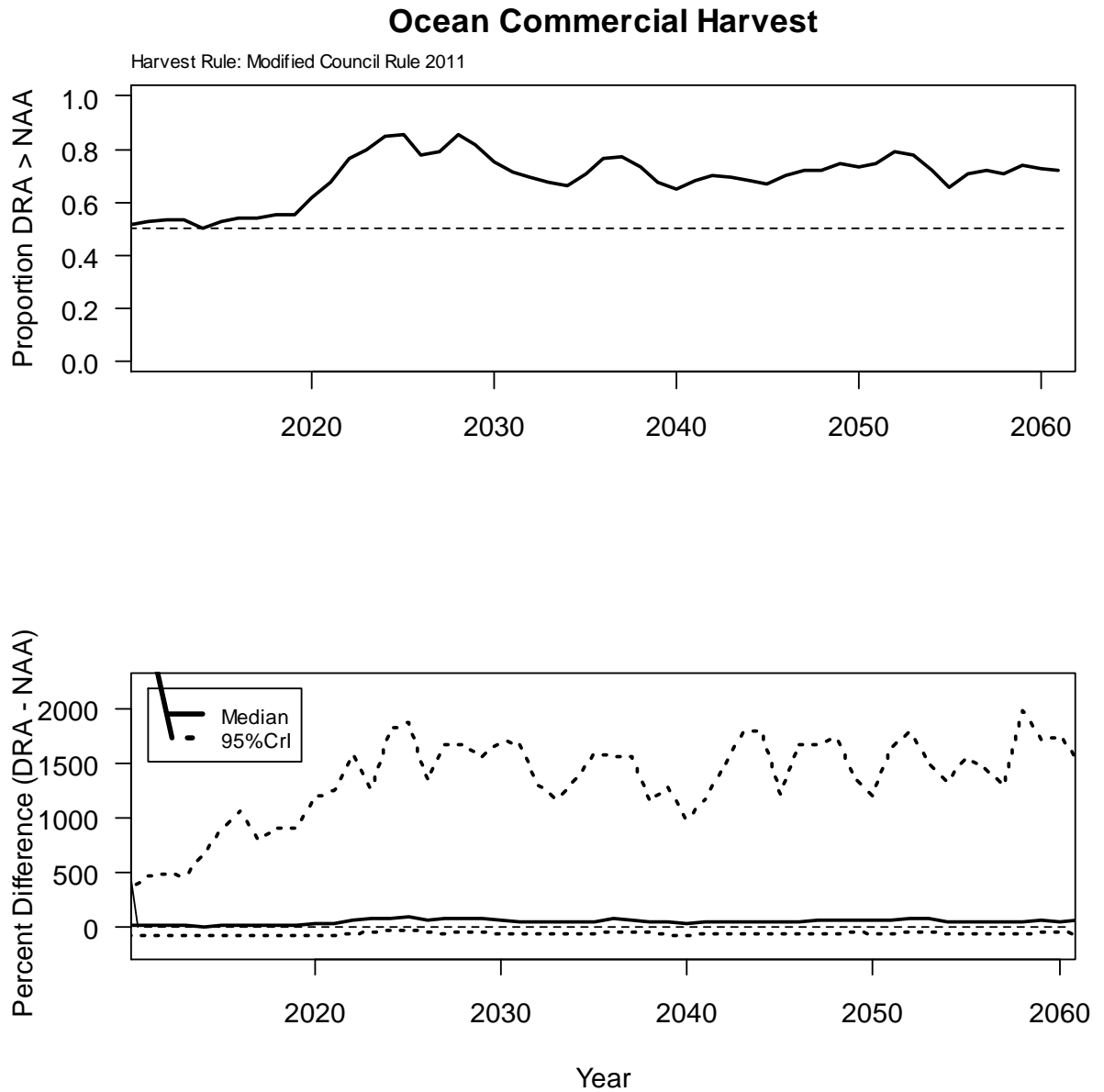


Figure 15. Probability that ocean commercial harvest is greater under DRA than under NAA from 2012 to 2061 (top). Dashed line represents 50/50 chance of increased abundance under DRA. Median and 95% credible intervals for the percent increase in DRA relative to NAA from 2012 to 2061 (bottom). Dashed line represents no difference between DRA and NAA.

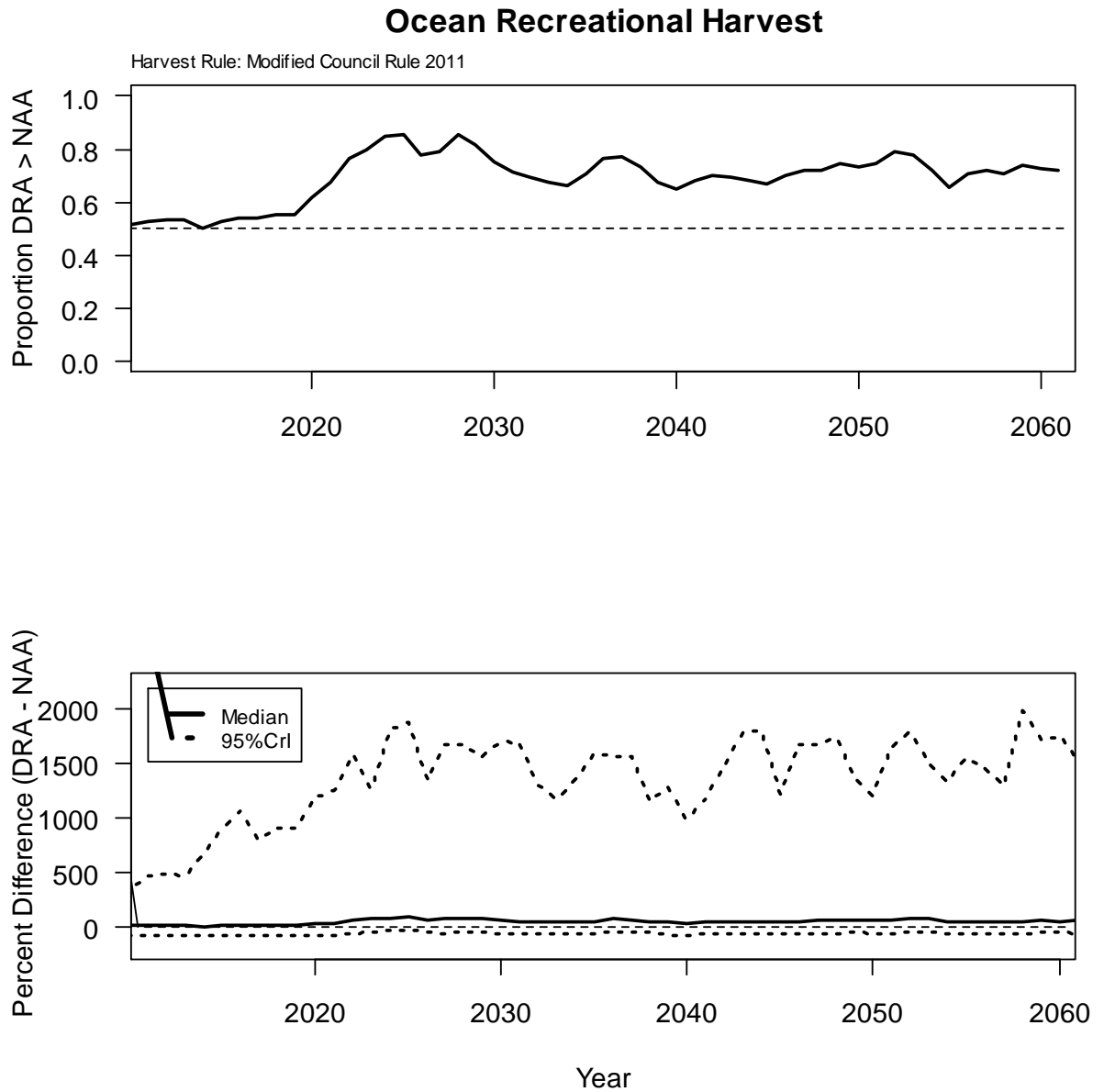


Figure 16. Probability that ocean recreational harvest is greater under DRA than under NAA from 2012 to 2061 (top). Dashed line represents 50/50 chance of increased abundance under DRA. Median and 95% credible intervals for the percent increase in DRA relative to NAA from 2012 to 2061 (bottom). Dashed line represents no difference between DRA and NAA.

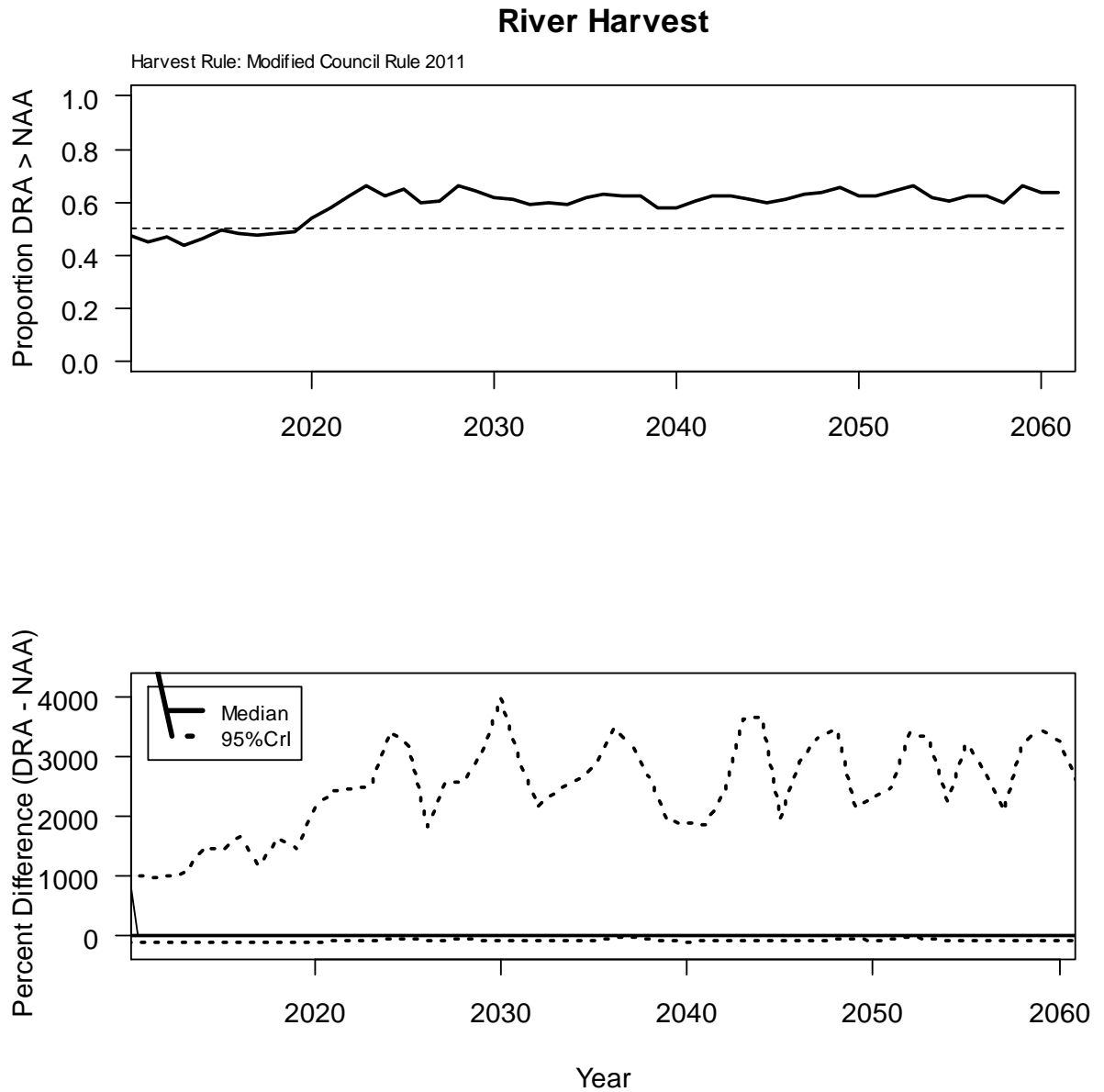


Figure 17. Probability that river harvest is greater under DRA than under NAA from 2012 to 2061 (top). Dashed line represents 50/50 chance of increased abundance under DRA. Median and 95% credible intervals for the percent increase in DRA relative to NAA from 2012 to 2061 (bottom). Dashed line represents no difference between DRA and NAA.

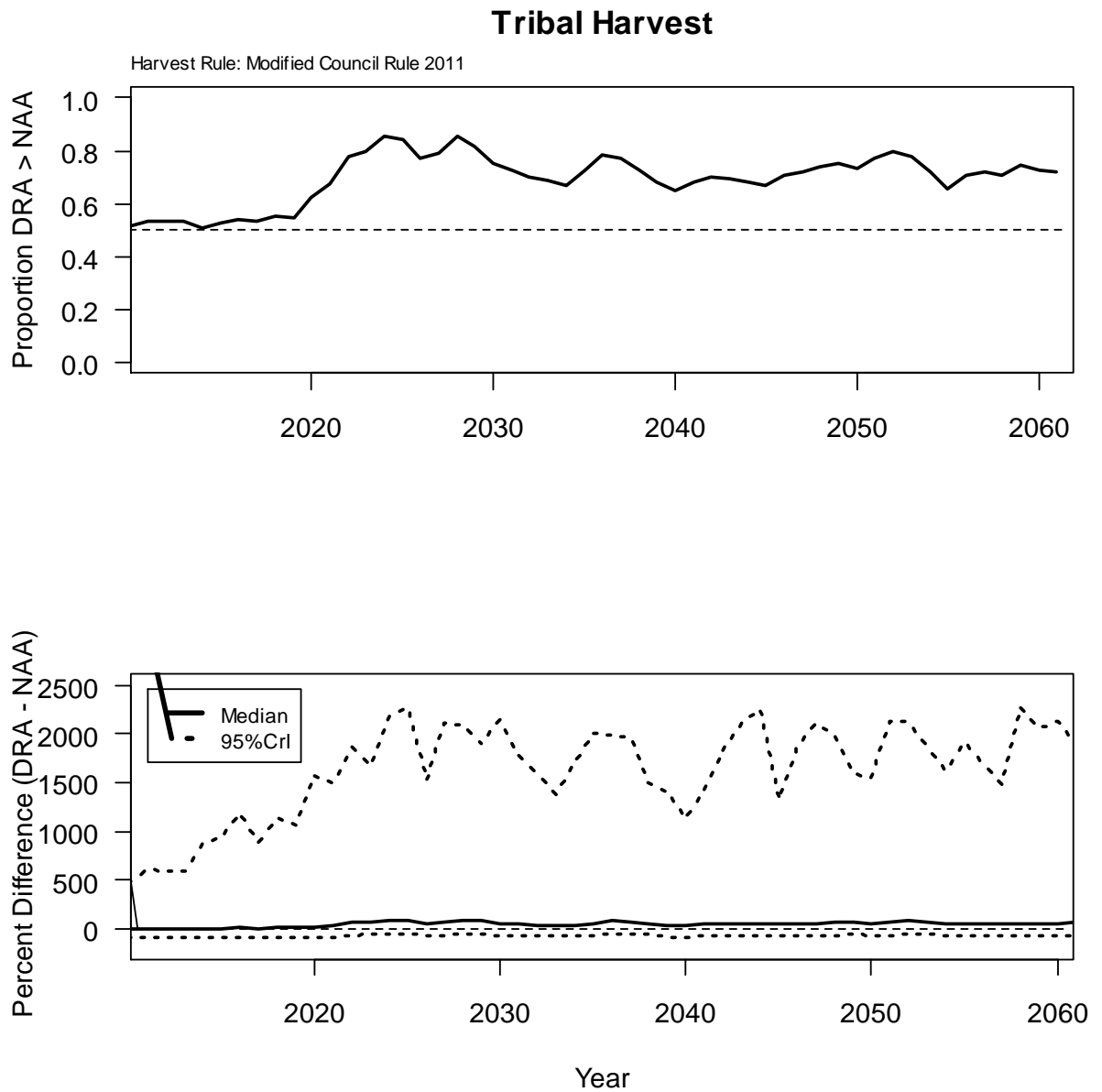


Figure 18. Probability that tribal harvest is greater under DRA than under NAA from 2012 to 2061 (top). Dashed line represents 50/50 chance of increased abundance under DRA. Median and 95% credible intervals for the percent increase in DRA relative to NAA from 2012 to 2061 (bottom). Dashed line represents no difference between DRA and NAA.

APPENDIX A. FISHERY CONTROL RULE APPLIED IN THE KLAMATH HARVEST RATE MODEL

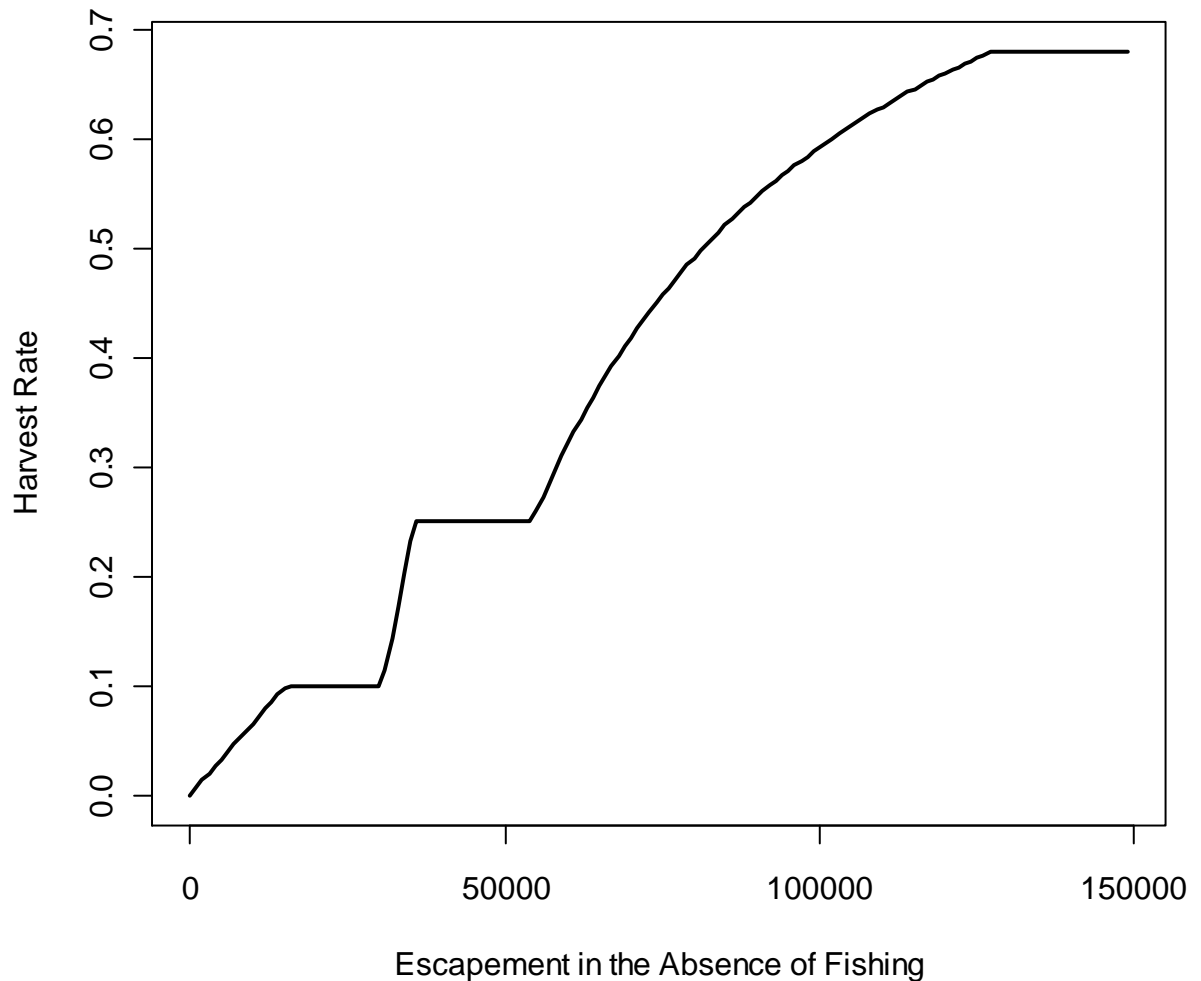


Figure A2.1. Harvest rate as a function of escapement in the absence of fishing utilized in the Klamath Harvest Rate Model (Mohr et al. in Prep).

Management of the Klamath Fishery was modeled by the Klamath Harvest Rate Model (KHRM, Mohr et al. in Prep.). Integral to the KHRM is the definition of a fishery control rule that defines the harvest rate as a function of an unfished escapement estimate (Figure 1). The fishery control rule described here provides the opportunity for a *de minimis* fishery even if the escapement in the absence of fishing is below the target stock size of 40,700 (S_{msy}).

APPENDIX B. PSEUDOCODE FOR RUNNING THE NO ACTION ALTERNATIVE AND THE DAM REMOVAL ALTERNATIVE

A. Steps to Running NAA

Set initial abundances and parameter values

The following steps were completed prior to running the annual forecasts of recruitment and harvest by drawing 1000 values from the following distributions (note that $N(\text{mean}, \text{variance})$ refers to a normal distribution with mean and variance as specified):

1. Set initial abundances from the CDFG MegaTable (CDFG 2011)
 - a. Spawning abundance in 2007 distributed as $N(61741, 25000)$
 - b. Spawning abundance in 2008 distributed as $N(48073, 25000)$
 - c. Spawning abundance in 2009 distributed as $N(52499, 25000)$
 - d. Spawning abundance in 2010 distributed as $N(49027, 25000)$
2. Set initial abundances in the ocean in 2010 (PFMC 2011 PreSeason Report)
 - a. Age 4 in the ocean in 2010 distributed as $N(66500, 25000)$
 - b. Age 5 in the ocean in 2010 distributed as $N(700, 250)$
3. Set initial proportion of natural fish in the ocean in 2010
 - a. Proportion of natural age 4 in 2010, $g_4 = 0.5$
 - b. Proportion of natural age 5 in 2010, $g_5 = 0.5$
4. Draw parameter values from samples of the posterior distribution from the Ricker stock-recruitment model for natural production (Table 4)
 - a. Productivity, α'
 - b. Ricker density dependence parameter, β
 - c. Strength of CVI on natural stocks, δ
 - d. Standard deviation of random effect CVI, σ_{CVI}^2
 - e. The values of CVI for each year of the time series, $CVI_{2007:2061} \sim N(0, \sigma_{CVI}^2)$
5. Draw parameter values from samples of the posterior distribution from the Ricker stock-recruitment model for hatchery log survival (Table 4). Hatchery production was constant over the 2007 to 2061 time series with IGH production of 6 million, and TRH production of 3 million fingerlings.
 - a. Average log hatchery survival, κ_{IGH} and κ_{TRH}
 - b. Parameter relating log survival to flow, γ_{IGH} and γ_{TRH}
 - c. Standard deviation for residual variability on log hatchery survival, σ_H^2
 - d. Unexplained variability of log hatchery survival $u_{1:2, 2010:2061}$

With the initial abundance estimates specified, and the vectors of parameter values specified, the dynamic portion of the model could be completed.

Calculate annual production and harvest

For iteration $i = 1$ to 1000 (subscript suppressed for clarity)

For years $t = 2010$ to 2061

1. Calculate natural production of the age 3 ocean fish in year t by.

$$R_t = S_{t-3} \exp\{\alpha' - \beta S_{t-3} + \delta CVI_{t-2} + \epsilon_t\}$$

Equation (A1)

2. Calculate the survival rate of IGH releases for year t , s_{IGH} , using Klamath River Biological Opinion flows .

$$s_{IGH,t} = \exp\{h_{IGH,t}\} = \exp\{\kappa_{IGH} + CVI_{t-2} + \gamma_{IGH} (Q_{KR_{BO},t-2}) + u_{IGH,t}\}$$

Equation (A2)

3. Calculate the survival rate of TRH releases for brood year t , s_{TRH} , using Trinity River flows

$$s_{TRH,t} = \exp\{h_{TRH,t}\} = \exp\{\kappa_{TRH} + CVI_{t-2} + \gamma_{TRH} (Q_{TR,t-2}) + u_{TRH,t}\}$$

Equation (A3)

4. Calculate the hatchery production to age 3 assuming age 2 survival of 0.5 (Hankin and Logan 2010) for year t

$$N_{H,t} = 0.5(s_{IGH,t} 6e06 + s_{TRH,t} 3e06)$$

Equation (A4)

5. Calculate the total abundance of year 3 ocean fish

$$N_{3,t} = R_t + N_{H,t}$$

Equation (A5)

6. Calculate the proportion of year 3 ocean fish that are natural origin

$$g_{3,t} = \frac{R_t}{R_t + N_{H,t}}$$

Equation (A6)

7. Call KHRM and pass $N_{a,t} = \{N_{3,t}, N_{4,t}, N_{5,t}\}$ and $g_{a,t} = \{g_{3,t}, g_{4,t}, g_{5,t}\}$
8. In year t KHRM returns:
 - a. Natural area escapement, E_n which is set equal to S_t
 - b. Harvest
 - i. Ocean commercial harvest, H_u
 - ii. Ocean recreational harvest, H_w
 - iii. River tribal harvest, H_t
 - iv. River recreational harvest, H_r
 - c. Ocean Abundance in year $t + 1$
 - i. 4 year old abundance in the ocean N'_4
 - ii. 5 year old abundance in the ocean, N'_5

Next year: Repeat the loop for year $t+1$ by returning to step 1 having obtained the ocean abundances for the 4 and 5 year olds returned from KHRM

Next iteration

2.2.3 Steps to Running DRA

Set initial abundances and parameter values

The following steps were completed prior to running the annual forecasts of recruitment and harvest for iterations $i = 1$ to 1000, the subscript for iteration i is suppressed for clarity.

1. Use initial abundances previously sampled for the NAA alternative
 - a. Spawning abundance in 2007
 - b. Spawning abundance in 2008

- c. Spawning abundance in 2009
 - d. Spawning abundance in 2010
2. Use initial abundances in the ocean in 2010 previously sampled for the NAA
 - a. Age 4 in the ocean in 2010
 - b. Age 5 in the ocean in 2010
3. Use initial proportion of natural fish in the ocean in 2010 from NAA
 - a. Proportion of natural age 4 in 2010, $g_4 = 0.5$
 - b. Proportion of natural age 5 in 2010, $g_5 = 0.5$
4. Lower Basin stock recruitment parameters for years 2010 to 2020
 - a. Productivity drawn from truncated α' starting in 2012 to reduce the probability of low productivity as a result of KBRA
 - b. Ricker density dependence parameter, β
 - c. Strength of CVI on natural stocks, use draws of δ from the NAA
 - d. Use the values of CVI from the NAA, $CVI_{2010:2020}$
5. Lower Basin stock recruitment parameters for 2021 to 2061
 - a. Productivity drawn from truncated α' starting in 2012 to reduce the probability of low productivity as a result of KBRA
 - b. Ricker density dependence parameter based on additional spawning habitat from Iron Gate to Keno and tributaries, β_{new}
 - c. Strength of CVI on natural stocks, use draws of δ from the NAA
 - d. Use the values of CVI from the NAA, $CVI_{2021:2061}$
6. Hatchery production from 2010 to 2028. Hatchery production was constant over the 2010 to 2020 with IGH production of 6 million, and TRH production of 3 million fingerlings.
 - a. Use draws of average log hatchery survival, κ_{IGH} and κ_{TRH} from NAA
 - b. Use draws of parameter relating log survival to flow, γ_{IGH} and γ_{TRH} from NAA
 - c. Use draws of unexplained variability of log hatchery survival $u_{1:2, 2010:2028}$ from NAA
7. Hatchery production from 2029 to 2061. Hatchery production was assumed constant at TRH with production of 3 million fingerlings, whereas production at IGH ceases after 2028.
 - a. Use draws of average log hatchery survival, κ_{TRH} from NAA
 - b. Use draws of parameter relating log survival to flow, γ_{TRH} from NAA
 - c. Use draws of unexplained variability of log hatchery survival at TRH $u_{2, 2029:2061}$ from NAA
8. Stock recruitment parameters in tributaries to UKL in years $t = 2021, \dots, 2061$
 - a. Unfished equilibrium population size, $E_{new, t}$
 - i. Draw a value of $p_{t,t}$ from a Uniform(0,1) distribution in year t
 - ii. Sample from the distribution of $E_{new\ stream}$ using the watershed size of 4200.96 km²
 - iii. Sample from the distribution of $E_{new\ ocean}$ using the watershed size of 4200.96 km²
 - iv. Calculate $E_{new, t}$ using Equation 17
 - b. Productivity, $r_{new, t}$
 - i. Sample from the truncated distribution of $r_{newocean}$ with the degree of truncation dependent upon the year
 - ii. Sample from the truncated distribution of $r_{newstream}$ with the degree of truncation dependent upon the year
 - iii. Calculate $r_{new, t}$ using Equation 18.

With the initial abundance estimates specified, and the vectors of parameter values specified, the dynamic portion of the model could be completed.

Calculate annual production and harvest

For iteration $i = 1$ to 1000 (subscript suppressed for clarity)

For years $t = 2010$ to 2020

1. Calculate natural production of the age 3 ocean fish in year t , R_t in the lower basin using Equation 10; however replace α' with the samples from the truncated α'^* (the asterisk denotes draws from a truncated distribution).
2. Calculate the survival rate of IGH releases for year t , $s_{IGH,t}$ using Equation 11 and calculate the survival rate of TRH releases for year t , $s_{TRH,t}$ using Equation 12. Note that the survival rates are the same as used in the NAA due to using the draws from the posterior distributions for parameters used in Equations 11 and 12.
3. Calculate the hatchery production to age 3 assuming age 2 survival of 0.5 (Hankin and Logan 2010) for year t using Equation 13.
4. Calculate the total abundance of year 3 ocean fish using Equation 14.
5. Calculate the proportion of year 3 ocean fish that are natural origin using Equation 15.
6. Call KHRM and pass $N_{a,t} = \{N_{3,t}, N_{4,t}, N_{5,t}\}$ and $g_{a,t} = \{g_{3,t}, g_{4,t}, g_{5,t}\}$
7. The KHRM program returns:
 - a. Natural area escapement, E_n which is set equal to S_t
 - b. Harvest
 - i. Ocean commercial harvest, H_u
 - ii. Ocean recreational harvest, H_w
 - iii. River tribal harvest, H_t
 - iv. River recreational harvest, H_r
 - c. Ocean Abundance in year $t + 1$
 - i. 4 year old abundance in the ocean N'_4
 - ii. 5 year old abundance in the ocean, N'_5

Next year: Repeat the loop for year $t+1$ by returning to step 1 having obtained the ocean abundances for the 4 and 5 year olds returned from KHRM

For years $t > 2020$

1. Calculate natural production of the age 3 ocean fish in year t , R_t in the lower basin using Equation 10; however replace α' with the samples from the truncated α'^* (the asterisk denotes draws from a truncated distribution) and the new capacity β_{new} .
2. If $t < 2032$ the reintroduction program in the tributaries to UKL provides spawners ($S_{UKL,t}$) at levels equal to or greater than capacity $S_{UKL,t} = \max(E_{new,t}, S_{UKL,t})$
3. For $t > 2022$, calculate recruitment of age 3 ocean fish from production in the tributaries to UKL ($R_{UKL,t}$) incorporating the truncated mixture of ocean and stream type Chinook, the common variability among basins (CVI), and flow related survival. Finally, recruitment to the adult stage is multiplied by an adult return to age 3 in the ocean conversion factor (CF) obtained from Table 1.

$$R_t = S_{t-3} \exp\{\alpha'^* - \beta_{new} S_{t-3} + \delta CVI_{t-2} + \epsilon_t\} \quad \text{Equation (A7)}$$

$$R_{UKL,t} = S_{UKL,t-3} \exp\left\{r_{new}^* \left(1 - \frac{S_{t-3}}{E_{new,t}}\right) + \delta CVI_{t-2} + \gamma_{IGH} Q_{KBRA,t-2}\right\} CF \quad \text{Equation (A8)}$$

4. If year $t > 2028$ IGH ceases to produce fall Chinook and hatchery production consists of TRH fish only of 3 million

$$N_{h,t} = 0.5(S_{TRH,t} 3e06) \quad \text{Equation (A9)}$$

5. Calculate total natural production of natural origin age 3 fish

$$N_{n,t} = R_t + R_{UKL,t}$$

6. Calculate the proportion of lower basin natural production relative to the total natural production for age 3 fish in year t . Note that values of $l_{4,t} = l_{3,t-1}$ and likewise $l_{5,t} = l_{4,t-1}$ so that the proportion of lower basin natural production could track the different cohorts moving through the fishery

$$l_{3,t} = \frac{R_t}{R_t + R_{UKL,t}}$$

Equation (A10)

7. Calculate total production of age 3 fish on September 1

$$N_{3,t} = N_{n,t} + N_{h,t}$$

Equation (A11)

8. Calculate the proportion of year 3 ocean fish that are natural origin

$$g_{3,t} = \frac{N_{n,t}}{N_{n,t} + N_{h,t}}$$

Equation (A12)

9. Call KHRM and pass $N_{a,t} = \{N_{3,t}, N_{4,t}, N_{5,t}\}$ and $g_{a,t} = \{g_{3,t}, g_{4,t}, g_{5,t}\}$

10. In year t KHRM returns:

- a. Age specific natural area escapement, E_{na} which is split between lower basin and UKL tributary production using the appropriate age-specific values of $l_{a,t}$. .

$$S_t = \sum_{a=3}^5 E_{n,a} l_{a,t}$$

Equation (A13)

$$S_{UKL,t} = \sum_{a=3}^5 E_{n,a} (1 - l_{a,t})$$

Equation (A14)

- b. Harvest

- i. Ocean commercial harvest, H_u
- ii. Ocean recreational harvest, H_w
- iii. River tribal harvest, H_r
- iv. River recreational harvest, H_r

- c. Ocean Abundance in year $t + 1$

- i. 4 year old abundance in the ocean N'_4
- ii. 5 year old abundance in the ocean, N'_5

Next year: Repeat the loop for year $t+1$ by returning to step 1 having obtained the ocean abundances for the 4 and 5 year olds returned from KHRM

Next iteration

APPENDIX MS3. IMPACTS OF DEPLETING FORAGE SPECIES IN THE CALIFORNIA CURRENT

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AND ANTHONY D.M. SMITH

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<http://dx.doi.org/10.1017/S0376892913000052>

The work is available through the link above, on the [California Current Integrated Ecosystem Assessment](#) website, or by email request to the first author (Isaac.Kaplan@noaa.gov).

SUMMARY

Human demands for food and fish meal are often in direct competition with forage needs of marine mammals, birds, and piscivorous harvested fish. Here we used two well-developed ecosystem models for the California Current on the U.S. West Coast to test the impacts on other parts of the ecosystem of harvesting euphausiids, forage fish, mackerel, and mesopelagic fish such as myctophids. We estimated the abundance that would lead to maximum sustainable yield for these four groups individually, but found that depleting forage groups to these levels can have both positive and negative effects on other species in the California Current. The most common impacts were on predators of forage groups, some of which showed declines of >20% under the scenarios that involved depletion of forage groups to 40% of unfished levels. Depletion of euphausiids and forage fish, which each comprise > 10% of system biomass, had the largest impact on other species. Depleting euphausiids to 40% of unfished levels altered the abundance of 13-30% of the other functional groups by >20%; while depleting forage fish to 40% altered the abundance of 20-50% of the other functional groups by >20%. Our work here emphasizes the trade-offs between the harvest of forage groups and the ability of the California Current to sustain other trophic levels. Though higher trophic level species such as groundfish are often managed on the basis of reference points that can reduce biomass to below half of unfished levels, this level of forage species removal is likely to impact the abundance of other target species, protected species, and the structure of the ecosystem.

APPENDIX MS4. VARIABLE IMPACTS OF FUTURE FISHERIES DEVELOPMENT IN THE CALIFORNIA CURRENT ON ECOSYSTEM STABILITY AND SPATIALLY EXPLICIT BIOMASS PATTERNS

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ABSTRACT

Studies have demonstrated the importance of large biomass forage groups in model food webs, but small biomass contributors are often overlooked. Here, we predict impacts of three potential fisheries targeting relatively low biomass functional groups in the California Current Atlantis Model: deep demersal fish, nearshore miscellaneous fish, and shortbelly rockfish (*Sebastes jordani*). Using a spatially explicit ecosystem model, we explored fishing scenarios for these groups that resulted in depletion levels of 75, 40, and 25 percent. We evaluated the effects of fishing on ecosystem-wide biomass and spatial distribution of biomass. We also investigated the effects of fishing on ecosystem stability using multivariate time-series methods. Results indicate that developing fisheries on the proposed targets would have low impacts on biomass of other species at the scale of the whole California Current ecosystem. Ecosystem stability declined with fishing pressure, however. The spatial distribution of impacted functional groups was patchy, and concentrated in the central California region of the model. This work provides a framework for evaluating impacts of new fisheries with varying spatial distributions and suggests that regional effects should be evaluated within a larger management context.

INTRODUCTION

Human demands on ocean production have never been higher [1]. High demands for fish and fishmeal have led to fishing activities targeting lower trophic level species than in previous decades [2-4]. Increasing demands on already taxed ecosystems can lead to difficult management decisions regarding trade-offs between consumptive and non-consumptive uses of these forage groups. Ecosystem based management is one approach that identifies trade-offs in an ecosystem context, allowing for cumulative impact assessment across sectors [5,6].

Fishery management in the U.S. has been moving towards ecosystem based management approaches for more than a decade. In 1999, a panel of experts convened by the National Marine Fisheries Service (NMFS) recommended that regional fishery management councils adopt Fishery Ecosystem Plans as a supplement to existing Fishery Management Plans [7]. The goal of a fishery ecosystem plan is to document the structure and function of the managed ecosystem, including two-way feedbacks between the ecosystem and fishing activities. Fisheries ecosystem plans have been developed for regions such as the North Sea, Aleutian Islands, Pacific Islands, and Chesapeake Bay [8-11]. The Pacific Fishery Management Council is currently developing a Fishery Ecosystem Plan [12], targeting the California Current Large Marine Ecosystem (CCLME). The plan is still in draft form, however current objectives include addressing gaps in ecosystem

knowledge with respect to effects of fishing on marine ecosystems and considering the potential of developing science and management at spatial scales relevant to stock structure [12].

The ecosystem effects of fishing high biomass, low trophic level species have been the targets of much research recently [13,14, Kaplan et al. this volume]. These groups, by definition, form the base of the pelagic food web, and are important prey species for many higher trophic level species that are of commercial importance and/or conservation concern. Fishing limits have been put into place to protect high biomass, low trophic level species such as krill, anchovy, and sardine [15,16].

Studies on the impacts of fishing forage groups often focus on species or groups that contribute large amounts of biomass to large marine ecosystem [2], while low biomass groups are more easily overlooked. Fishing on these groups may indeed have few impacts on food webs if species are functionally redundant to large biomass prey species [17]. Or, removals of low biomass groups may have disproportionate impacts, depending on their role in the ecosystem and spatial distribution and overlap of predators and prey. For example, central place foragers, like many seabirds, depend on locally abundant seasonal prey resources [14,18,19]. Fluctuations in these resources could have severe impacts on populations that rely on them, even if overall biomass is low [20].

In this study, we investigated the effects of targeted fisheries on relatively low biomass forage fish species in a large marine ecosystem. Similar to previous modeling studies we report biomass responses of species in the food web. However this work is novel in that we also describe the effects of fishing on ecosystem stability, and explore biomass impacts using a spatially explicit model to predict the regional distribution of these impacts. We explored whether fishing these species under various fishing mortality scenarios affected other species in proportion to their overall biomass in the ecosystem. We investigated target species that were broadly and narrowly distributed within the region to explore the effects of spatial variation on fishery development.

METHODS

MODEL FRAMEWORK

Atlantis is a three dimensional, spatially explicit ecosystem model, comprised of three sub-models [21]. The oceanographic sub-model simulates physical transport using output from a Regional Ocean Modeling System to track temperature, salinity, and circulation. The ecological sub-model captures nitrogen and silicon dynamics through trophic interactions among cells, representing functional groups from bacteria and plankton to fish and marine mammals. The human impacts sub-model overlays both the ecological and oceanographic sub-models, and includes fisheries, nutrient inputs, and management control rules. This framework allows for hypothesis testing of how perturbations in the food web can propagate all the way to the management arena. Fulton and colleagues [22] summarize the assumptions and options within the Atlantis code base, and detail lessons learned from 13 recent modeling efforts.

The Central California Atlantis Model (CCAM) was developed to address federal and state level management needs in the California Current Large Marine Ecosystem [23,24]. The modeled area extends from Cape Flattery in the north to Point Conception in the south and from shoreline to the 2400 m isobaths (Figure 1). There are 12 latitudinal regions, broken up longitudinally by 3 to 7 depth zones. Each of these two dimensional areas is further divided into up to seven depth bins, capturing the sediment layer to the surface layer through the water column. The central California region of the model has higher 2-d spatial resolution

in depth zones than the northern and southern regions. CCAM oceanography is based on a ROMS time-series for 1958-2004.

The ecological sub-model for CCAM compartmentalizes biomass among 62 functional groups. Species are grouped based on similar life history characteristics and diets. Details of model parameterization and calibration have been described elsewhere [23,24].

We used a “status quo” fishing scenario against which to compare all new fishing activities in the model. The status quo represented current fishing in the CCLME, and was the same as in Kaplan et al. [24]. Spatial closures represented current area-based management, and fishing mortalities were specified by targeted group and fleet and calibrated to reproduce catches from stock assessments, where applicable [23].

APPROACH

For each new target group, we created a new fleet in the model and determined the appropriate area closures based on the likely gear type. We ran fishing scenarios for each fishery addition following methods by Smith et al. [13] and Kaplan et al. (this volume). We incrementally increased the annual fishing mortality from zero until the target group was completely depleted. Each model run was 50 years, allowing functional groups to reach quasi- equilibrium; the model does not assume true equilibrium dynamics and is driven by oceanographic forcing as well as species interactions. We then used these fishing mortalities and resulting catches to determine the maximum sustained yield (MSY) for each group, as well as the fishing mortality required to obtain 3 levels of depletion relative to the status quo fishing scenario: 25 percent, 40 percent, and 75 percent of status quo biomass.

We describe the impacts of the new fisheries on equilibrium yields and biomasses of the other functional groups in the model. We averaged the last five years of each model run to represent an equilibrium catch or biomass for the majority of functional groups. However, groups with high growth rates and quick turnover tend to have flashy dynamics. For these groups (all plankton, zooplankton, and bacteria) we averaged over the last 20 years of each model run.

We used a threshold of ten percent change in catch or biomass to determine whether the new fishery impacted functional groups. The choice of threshold was somewhat arbitrary—Smith et al. used 40 percent, Kaplan et al. used 20 percent. Because our target species were lower biomass than the groups previous studies investigated, we set a lower threshold.

We also investigated the effects of fishing on ecosystem stability. Stability is a property that describes the response of an ecosystem to a perturbation [25], and may also relate to regime shifts [26]. To estimate stability, we fit multivariate auto-regressive (MAR) models to the last 10 years of Atlantis model output by cell, and estimated the ecosystem-wide community or interaction matrix for all functional groups following methods of Ives et al. [27]. We estimated three metrics of system stability derived from the community matrix (**B**)—two that relate to asymptotic stability and one that describes transient behavior after a perturbation. These well-established methods have been used to describe stability properties from time-series in both modeled and data-based food webs [28,29].

The dominant (largest) eigenvalue of the community matrix describes the rate of return of an ecosystem following a perturbation, and is the most commonly used metric of stability describing resilience (return rate). An alternative to this metric takes into account all of the eigenvalues of the community matrix: $\det(\mathbf{B})^{2/p}$ where p is the number of groups in the model [27]. We refer to this second metric as stability.

Reactivity describes transient activity immediately after a perturbation, rather than long-term patterns of return [30]. We calculated a worst-case reactivity from the community matrix: $\max(\lambda_{B'B})$ [27].

DEEP DEMERSAL FISH

The deep demersal fish group in CCAM is distributed along the continental slope (500-1200 m, Figure 1 A), and consists mostly of giant grenadier (*Albatrossia pectoralis*) and Pacific grenadier (*Coryphaenoides acrolepis*). Other species in this functional group include Pacific lamprey, eelpouts, cusk eels, and poachers [23]. The west coast groundfish fleet catches both grenadier species, and Bellman et al. [31] estimated one to two percent of the annual catch at depths greater than 250m consists of giant and Pacific grenadier, totaling 600 mt per year. These species are rarely landed because of limited market demand [32]. A pot fishery for lamprey is also included in CCAM with harvest of 1250 mt per (Table 1).

Grenadier (family Macrouridae) catches around the world have risen since the mid 1990s. Targeted fisheries currently harvest about 45000 mt of grenadier from the world's oceans each year [33]. In the North Pacific, Japanese harvested grenadier during the 1980s. They were processed into surimi, before the walleye Pollock fishery became a more marketable source [32]. Due to historical use and increasing demand for fish and fish products, we thought it would be useful to explore the potential impacts of landing this species complex on the west coast.

Natural mortality for the deep demersal fish group is low (0.1, Table 1), which suggests a priori that MSY will also be relatively low. We created a target fishery on this group that represents a fishery for grenadier using the same gear and area restrictions as the existing bottom trawl fleet [24].

NEARSHORE MISCELLANEOUS FISH

The nearshore miscellaneous fish group is a catchall group dominated by white croaker (*Genyonemus lineatus*), but also includes shallow sculpins and midshipman. This group is distributed across the nearshore model domain, with higher densities in central California than other regions (Figure 1B). Life history parameters for this group are based on white croaker [23].

We created a fishery on this group primarily to represent a fishery targeting white croaker. Croaker is a popular recreational target in California, but only small amounts are currently landed in commercial fisheries annually (3 mt in 2011) using round haul net, gill net, and hook and line gear [34]. Atlantic croaker (*Micropogonias undulatus*) is a closely related species on the east coast of the US, with similar size, life history, and habitat and food preferences [35,36]. Atlantic croaker is the target of a valuable 10000 mt fishery [33].

While the distribution of the miscellaneous nearshore fish group spans the latitudinal extent of the model domain, in reality, white croaker likely composes greater proportions of the group's biomass from San Francisco bay south to Point Conception [37]. An existing modeled recreational fishery accounts for 247 mt of biomass removed from this group each year. The natural mortality rate for the group is 0.62, suggesting it would tolerate a moderate harvest rate (Table 1). We created a target fishery for croaker using the same area closures as the existing nearshore non-fixed gear sector [24].

SHORTBELLY ROCKFISH

Shortbelly rockfish is the most abundant of the rockfish species, and in CCAM shortbelly comprise their own functional group. The most current stock assessment estimated the shortbelly stock to be 64,000 mt in 2005 [38]. Notably, modeled shortbelly biomass in our status quo scenario is roughly 25 percent of the assessed biomass (Table 1). Considerable biomass uncertainty likely results from a lack of fishery dependent data and poor catchability of shortbelly in the fisheries independent trawl survey [38]. Shortbelly rockfish density is highest in central California (Figure 1C). A few fleets unintentionally catch shortbelly, but these removals are limited to less than 1 mt per year [31]. A relatively high natural mortality rate in CCAM (0.35) suggests that this group should be able to sustain a moderate level of fishing mortality (Table 1). We modeled the shortbelly fishery as a mid-water trawl fishery, subjected to the same area closures as the existing trawl fleet [24].

Fishery interest in shortbelly rockfish has historically been quite low, at least in part because shortbelly is small-bodied (maximum size less than 30 cm) [38,39]. Lenarz [39] identified a potential pet food or surimi market for shortbelly, however he also pointed out these were not economically viable as of 1980. Currently, an annual catch target of 50 mt is in place for shortbelly. The groundfish catch regulations indicate this limit is higher than recent catches of shortbelly, but the target is set conservatively because shortbelly is an important forage species in the California Current ecosystem [40,41].

RESULTS

We found some general and some variable effects of fishing the new target groups. First, we describe overall general patterns of ecosystem response. Then, we describe specific results of fishing each target group on biomass, yields and stability.

Across all fishing scenarios and target groups, we saw limited ecosystem-wide effects of fishing on biomass or yields of other groups. The impacts we did observe were disproportionately weighted in the central California region of the model. No predators of the three target groups were affected by their removal. Nine invertebrate groups (planktonic and benthic) were affected in one or more model cells by at least one of the fishing scenarios. In some cases, affected groups were prey of target species, but in others they were more than one trophic link removed from the fished group. Likewise, not all impacted model cells contained the target species. Notably, the vast majority of impacted groups were highly productive and demonstrated oscillatory or eruptive behavior.

We attempted to explain variation in the number of groups impacted in each model cell using cell area, cell volume, total number of functional groups present, density of target group, density of prey groups (of target), and density of affected groups. However, preliminary analyses showed no relationships between any of these variables.

Fishing the deep demersal, nearshore miscellaneous, and shortbelly groups had variable effects on ecosystem stability (Table 2). The stability metric that took into account only the dominant eigenvalue of the community matrix, return rate, was least sensitive to the effects of fishing. Alternatively, the metric that weights all of the eigenvalues (what we refer to as stability) generally showed destabilizing effects of new fishing activity. Reactivity generally decreased initially with fishing effort, but increased as the target group became depleted.

DEEP DEMERSAL FISH

Simulations suggested that deep demersal fish could sustain a maximum harvest of 2055 mt per year, which required annual fishing mortality of 0.03. This level of fishing reduced the biomass of this group to 66747 mt (about 40 percent of the status quo biomass, appendix 1). The current estimate for grenadier bycatch is 600 mt [31], resulting in capacity for a fishery using the same gear as the current trawl fishery of about 2600 mt sustained yield.

Fishing deep demersals had no impact on fishery yields or abundance of any other functional group at the scale of the whole ecosystem (using a 10 percent threshold), despite the group's broad latitudinal distribution. Individual cells were affected primarily in the Central California region. There, a new fishery affected biomass of up to three invertebrate functional groups in the plankton and benthos, some of which were prey species of deep demersal fish (Figure 2). Fishing scenarios of F25, F40, and F75 varied little in their spatial impacts or number of groups affected (Figure 2). No predators of deep demersals were affected by their removal.

The qualitative effects of increasing grenadier fishing mortality varied among model cells (Figure 3). Only one of the five model cells in which two or more groups were affected had deep demersal fish present. In this cell (14), decreasing abundance of deep demersals led to increased copepod abundance, a prey species of the target group. This increase was also associated with increased microzooplankton and phytoplankton abundance (Figure 3). The direction of change for microzooplankton varied across model cells, however. Plankton groups were affected at low levels but in both directions.

Fishing the deep demersal group decreased stability and increased return rate very slightly, but only in the most severe fishing scenario (Table 2). Reactivity was lower when the target group was fished at any level, however reactivity declined to a minimum when it was fished at MSY, and increased as depletion increased. Overall, these changes in stability were quite small.

NEARSHORE MISCELLANEOUS FISH

A fishery on the nearshore demersal fish group (croaker) attained MSY of 2000 mt with an annual fishing mortality of 0.1 (appendix 1). This level of fishing reduced the biomass of the functional group to 40 percent of the status quo equilibrium biomass of 20000 mt. Fishing the target group led to increased abundance of the shrimp group, which is a prey group for nearshore demersals. The shrimp group includes all crangon, mysid, and pandalid shrimp species. This increased biomass led to higher yields of the shrimp fishery by up to 12 percent (Figure 4).

Because shrimp biomass increased with fishing the nearshore demersal group, at least one functional group was impacted in 27 model cells (the majority of the group's distribution in CCAM, Figure 5). Besides shrimp, most impacts were on invertebrate plankton groups. Benthic detritivores, benthic bacteria, and octopi were all impacted in at least one scenario. Of these, only benthic detritivores were a prey group for croaker in the model. Impacts were more concentrated in central California region, particularly in cells whose boundaries represent those of the Gulf of the Farallones and Northern Monterey Bay National Marine Sanctuaries. An intermediate fishing scenario (F40) resulted in the greatest perturbation to other functional groups (Figure 5E).

The areas of greatest perturbation occurred where densities of both shrimp and nearshore demersal fish were relatively high (Figure 6, cells 24, 39, and 46). In many cases, perturbed groups tracked the target group's productivity with greatest changes occurring when croaker were fished to B40. Overall,

microzooplankton had the largest proportional changes in biomass. These perturbations occurred in cells with very low densities of microzooplankton, however. Therefore, the change in absolute biomass was quite small.

Fishing the miscellaneous nearshore demersal fish group did not affect ecosystem return rate, but did decrease stability (increase in the second stability metric, Table 2). Reactivity declined at low fishing levels, and increased with higher fishing pressure. All levels of fishing had lower reactivity than the status quo scenario, however.

SHORTBELLY ROCKFISH

Shortbelly MSY was about 675 mt, and occurred under a fishing mortality of 0.2 per year. This coincided with a reduction in shortbelly biomass to 20 percent of the status quo. Increasing fishing mortality to 1 was required to completely deplete shortbelly (Appendix 1). A shortbelly fishery did not affect yields of any other fisheries.

Ecosystem-wide abundance of functional groups was not influenced by any shortbelly fishing scenarios. Up to four functional groups were affected in individual cells, mostly in central California where shortbelly are distributed in CCAM (Figure 1, 7). More cells and functional groups were affected as fishing mortality increased (Figure 7). As in the previous two fisheries, the greatest number of groups was affected in the Gulf of the Farallones and Northern Monterey Bay National Marine Sanctuary cells (24 and 39). Only two of the five cells in which two or more functional groups were affected overlapped with status quo shortbelly distribution in the model (Figure 8). Similar to the croaker fishery, we saw the largest proportional changes in the microzooplankton group, which was not a prey group for shortbelly. These changes occurred in cells with relatively low densities of microzooplankton, however. The greatest direct effect of removing shortbelly was increased copepod abundance. Other prey groups of shortbelly that were affected included benthic detritivores, benthic bacteria, and pelagic bacteria.

Of the three target species, fishing on shortbelly had the greatest impacts on ecosystem stability. Despite its limited distribution in the model, completely depleting shortbelly led to an increased ecosystem return rate (Table 2). Increasing fishing pressure also incrementally increased the other stability metric. Reactivity tracked with fishing pressure as in the previous two target groups. Low levels of depletion led to low reactivity, but increased fishing increased reactivity.

DISCUSSION

We explored the effects of new fishery development on three new target groups in the California Current. Overall, we found fairly low magnitude impacts on a limited number of functional groups in the model. Even the most severe fishing scenarios affected fewer than 10 percent of functional groups. We saw the most widely distributed effects on copepod abundances, across the fisheries and model domain. The effects did not propagate to higher trophic levels, however. Only one of three fisheries led to changes in fishery yields of any other functional group in the model. Despite these limited impacts, these fishing activities generally decreased ecosystem stability.

Studies focusing on large biomass low trophic level species and associated fisheries have described larger ecosystem-wide impacts of fishing on those groups [13]. Using the same model of the California Current to explore more abundant forage groups, Kaplan et al. (this volume) saw changes of greater than 20 percent in many groups, in particular predators of forage species. Our results did not show such widespread

or dramatic changes. We propose two not mutually exclusive explanations for the limited effects of fisheries for the three groups we explored here. First, and most simply, biomass for these groups is low relative to other forage groups in the model, and low relative to groups explored in previous studies. For the three cases we described, our modeling results suggest these groups may be functionally redundant with other prey species [17]. This necessarily means that fishery removals will be a smaller perturbation to ecosystem total biomass, and thus minimize impacts on other functional groups. Second, our current model's structure may be insufficient to capture local variation in space and time that could impact food web structure heterogeneously along the west coast.

The low biomass of the three functional groups we explored here identifies some constraints in the model structure that limited our ability to capture potential effects of new fisheries on these target groups. The fishery targeting nearshore demersal fish resulted in increased catches of shrimp, with no variation across individual model cells. This finding could be somewhat misleading due to Atlantis constraints on species distributions and movement. Spatial distributions of functional groups are determined seasonally in the model. These parameters allocate total biomass by functional group to individual cells proportionally. Therefore, a group could be strongly affected by fishing on the new target species within a season, but at the beginning of the next quarter, biomass is reallocated across all the cells in the model according to seasonal distribution. This limits the ability of fishing on groups with limited spatial distributions to affect densities of prey or predator species that have seasonal components under our current parameterization. These seasonal parameters apply to all vertebrate groups, euphausiids, cephalopods, and shrimp. If we could turn off seasonal movements in the model, we could test how much seasonal reallocation of biomass contributed to the changes we did (or did not) observe. Alternatively, density dependent and prey dependent movements are features of the model we have not fully explored, and these could also capture meaningful responses of locally depleted functional groups.

Similarly, seasonal constraints and the limited ability to capture spatially heterogeneous changes in functional groups may also contribute to our inability to observe changes in predator biomass of target species. Fishing shortbelly could potentially have locally negative impacts on seabirds that rely on shortbelly as a prey source during key breeding seasons, for example [14,18]. These effects could be masked in the model by re-allocation of seabird biomass across the model cells in accordance with their seasonal distribution in each quarter, or by the large size of the model cells compared to breeding grounds.

Similarly, our application of fishing mortality in this version of the model also likely constrained functional group and fishery responses. We implemented somewhat rudimentary fleet dynamics in CCAM in this study. We specified the functional groups targeted by each fleet, and area closures were implemented by fleet based on gear-type. Fishing mortality was represented by a constant (daily) rate by functional group and fleet. This resulted in a constant proportion of biomass removed across all cells that were not closed to the fishery. Therefore, catches tracked biomass linearly and proportional changes in catches had to be constant across the model domain. Small biomass groups and those with limited spatial distributions in a larger model may be particularly sensitive to these types of generalizations.

We saw disproportionately large biomass effects in central California, either in spite of or because of these model constraints. Our model predictions could have implications for the food web in this region, particularly in the Gulf of the Farallones and the northern region of the Monterey Bay National Marine Sanctuaries. These sanctuaries provide habitat for many species of conservation concern, such as seabirds and marine mammals [42,43]. However, the cause of these findings warrants further investigation before any strong conclusions can be drawn. Even if the magnitudes of the impacts of new fishing activities are underestimated or captured imperfectly by CCAM, our work identifies regions of the coast that are more

likely to be impacted. This kind of knowledge may aid regional managers in making proactive decisions, for example monitoring particular functional groups for evidence of impacts of fishing.

Impacts on invertebrate functional groups should be interpreted as qualitative expectations, rather than exact predictions, however. Nearly all functional groups that responded to reductions in target species biomass were highly productive and highly variable within a year or across years. These traits lead to quick responses to changes in the ecosystem, but also lead to dynamics that are difficult to predict, as indicated by a single functional group responding in different directions across multiple model cells.

Our work represents a first step toward understanding how fishing target species on high and low biomass groups could impact ecosystem stability and biomass distribution in a spatially explicit ecosystem model. A next step would be a comparative analysis using the findings in Kaplan et al. (this volume) to motivate a spatially explicit analysis of fishing large biomass forage groups such as sardine, myctophids, and krill. We could also compare the effects of fishing on ecosystem stability across a range of biomass removals, and explore stability in a more spatially explicit way within and across fishing scenarios.

Adopting ecosystem based management approaches and implementing fishery ecosystem plans will necessarily result in identifying trade-offs between consumptive and non-consumptive uses in large marine ecosystems. Here we demonstrated the effects of three potential fisheries that our ecosystem models suggest will have relatively low impacts on the food web at the ecosystem scale. Instead, trade-offs may occur across space, with potentially cascading effects on planktonic and benthic invertebrate groups. Our results do not provide definitive predictions on the impacts of new fisheries, but identify regions and groups that could be targeted for monitoring potential impacts if these fisheries were to develop. More importantly, this work provides a necessary framework for evaluating the effects of fishing on ecosystem stability and the distribution of biomass across a spatially heterogeneous large marine food web.

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42. Office of National Marine Sanctuaries (2010) Gulf of the Farallones National Marine Sanctuary Condition Report 2010. In: U.S. Department of Commerce NOAA, Office of National Marine Sanctuaries,, editor. Silver Spring, MD. pp. 58.
43. Office of National Marine Sanctuaries (2009) Monterey Bay National Marine Sanctuary Condition Report 2009. In: U.S. Department of Commerce NOAA, Office of National Marine Sanctuaries,, editor. Silver Spring, MD. pp. 58.

Table 1. Summary of fishing scenarios and target groups. Biomass and yield indicated are for status quo (SQ) model run. M indicates annual natural mortality rate. Maximum sustained yield (MSY) was determined using all fishing scenarios for each target group, and FMSY indicates the fishing mortality rate at MSY. Annual fishing mortality required to obtain 75, 40, 25, and 0 percent depletion is indicated by F75, F40, F25, and F0, respectively.

| Target Group | SQ Biomass (mt) | SQ Yield (mt) | M | MSY (mt) | FMSY | F75 | F40 | F25 | F0 |
|--|--------------------|------------------|------|-------------|------|------|------|------|------|
| Deep demersal fish | 183562 | 2117 | 0.1 | 2055 | 0.03 | 0.01 | 0.03 | 0.04 | 0.1 |
| Misc. nearshore fish | 20920 | 206 | 0.62 | 905 | 0.12 | 0.04 | 0.1 | 0.12 | 0.24 |
| Shortbelly rockfish (<i>Sebastes jordani</i>) | 16434 | 0.1 | 0.35 | 687 | 0.2 | 0.05 | 0.1 | 0.2 | 1 |

Table 2. Ecosystem stability metrics by target species and fishing scenario. Stability metrics were based on the community (interaction) matrix estimated from time-series models. Return rate and Stability both describe the asymptotic behavior after a perturbation and Reactivity describes short-term transient dynamics. In all cases, smaller values indicate greater stability.

| Target Species | Metric | SQ | F75 | F40 | F25 | F0 |
|----------------------|-------------|-------|-------|-------|-------|-------|
| Deep demersal fish | Return Rate | 0.754 | 0.754 | 0.754 | 0.754 | 0.755 |
| | Reactivity | 0.594 | 0.587 | 0.581 | 0.583 | 0.587 |
| | Stability | 0.045 | 0.046 | 0.046 | 0.046 | 0.047 |
| Nearshore misc. fish | Return Rate | 0.754 | 0.754 | 0.753 | 0.754 | 0.753 |
| | Reactivity | 0.594 | 0.576 | 0.581 | 0.579 | 0.586 |
| | Stability | 0.045 | 0.047 | 0.046 | 0.047 | 0.045 |
| Shortbelly rockfish | Return Rate | 0.754 | 0.753 | 0.753 | 0.754 | 0.760 |
| | Reactivity | 0.594 | 0.583 | 0.575 | 0.582 | 0.591 |
| | Stability | 0.045 | 0.045 | 0.046 | 0.046 | 0.049 |

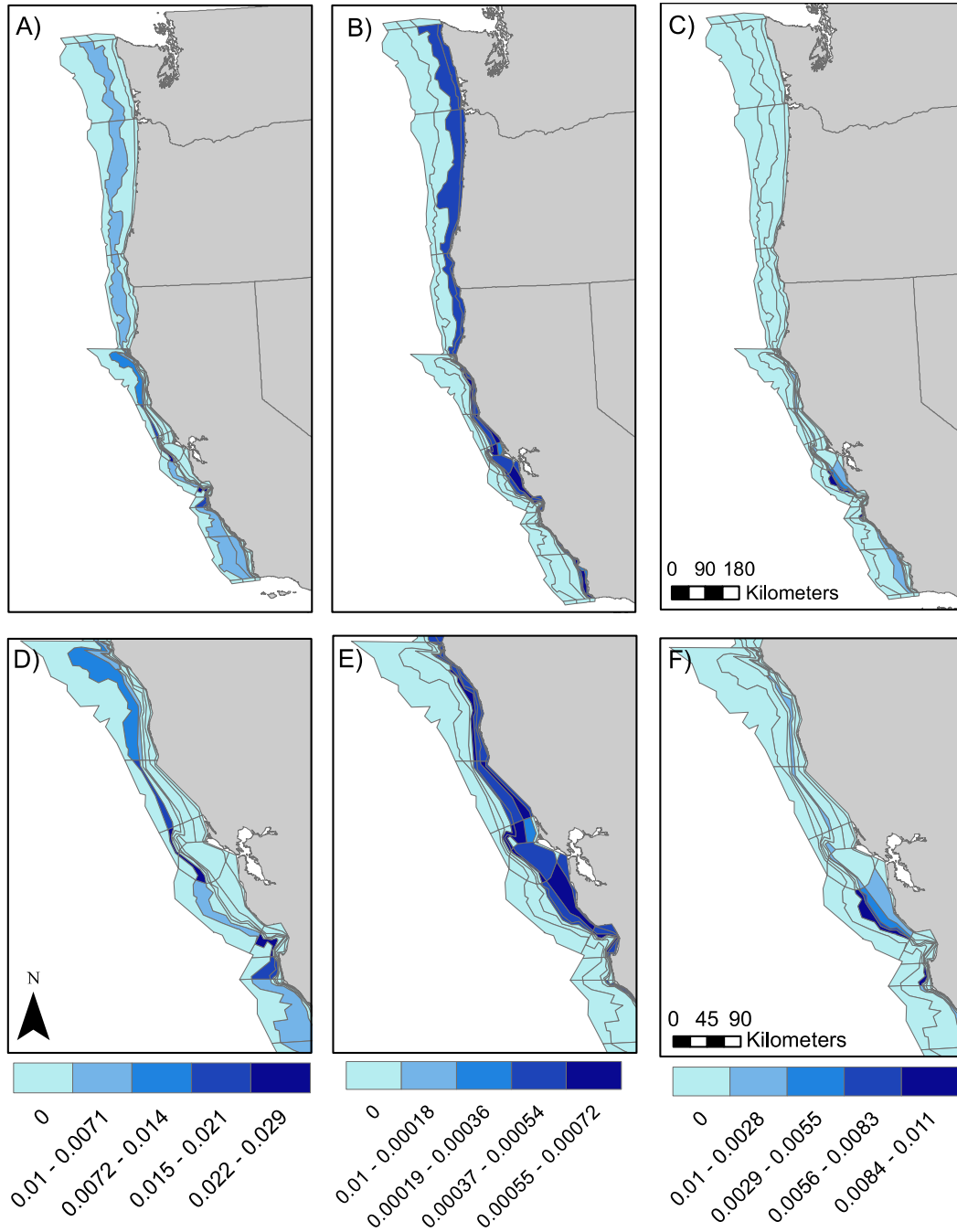


Figure 1. Status quo distribution of new potential target groups (deep demersal fish, nearshore miscellaneous fish, and shortbelly rockfish). Top panels illustrate distribution in the full model domain (A, B, C). Bottom panels show distribution within Central California region (D, E, F). Deep demersal fish densities were highest in slope cells (A, D), nearshore miscellaneous fish were limited to coastal areas (B, E), and shortbelly rockfish were concentrated in Central California (C, F). Legend below each panel indicates densities in kg/m².

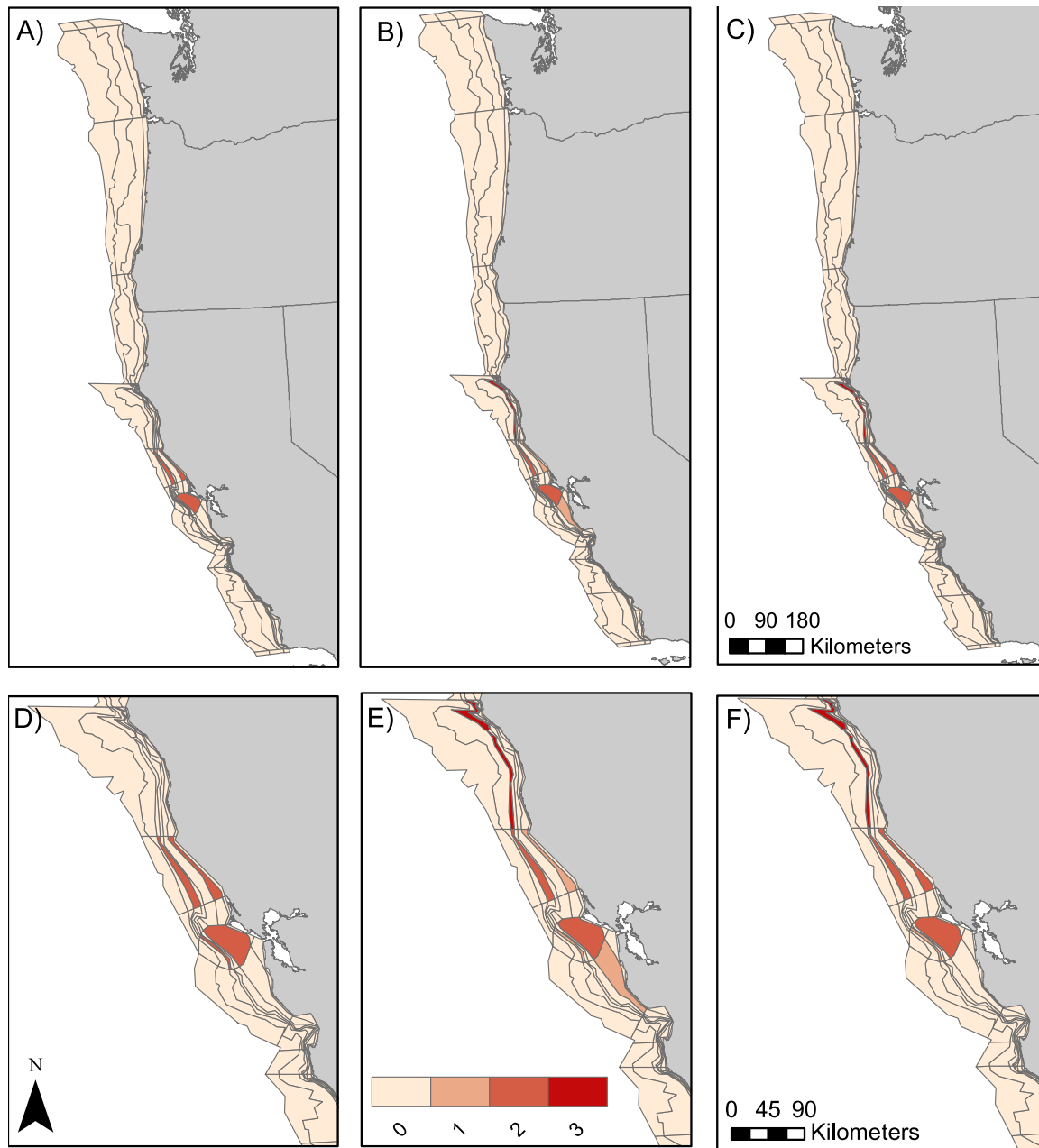


Figure 2. Number of functional groups affected by a fishing deep demersal fish at three fishing levels (threshold of 10 percent change) by cell. Fishing scenarios represented are F75 (A, D), F40 (B, E), and F25 (C, F). Density of color indicates increasing number of functional groups affected, as indicated by legend. Top and bottom panel extents as in Figure 1.

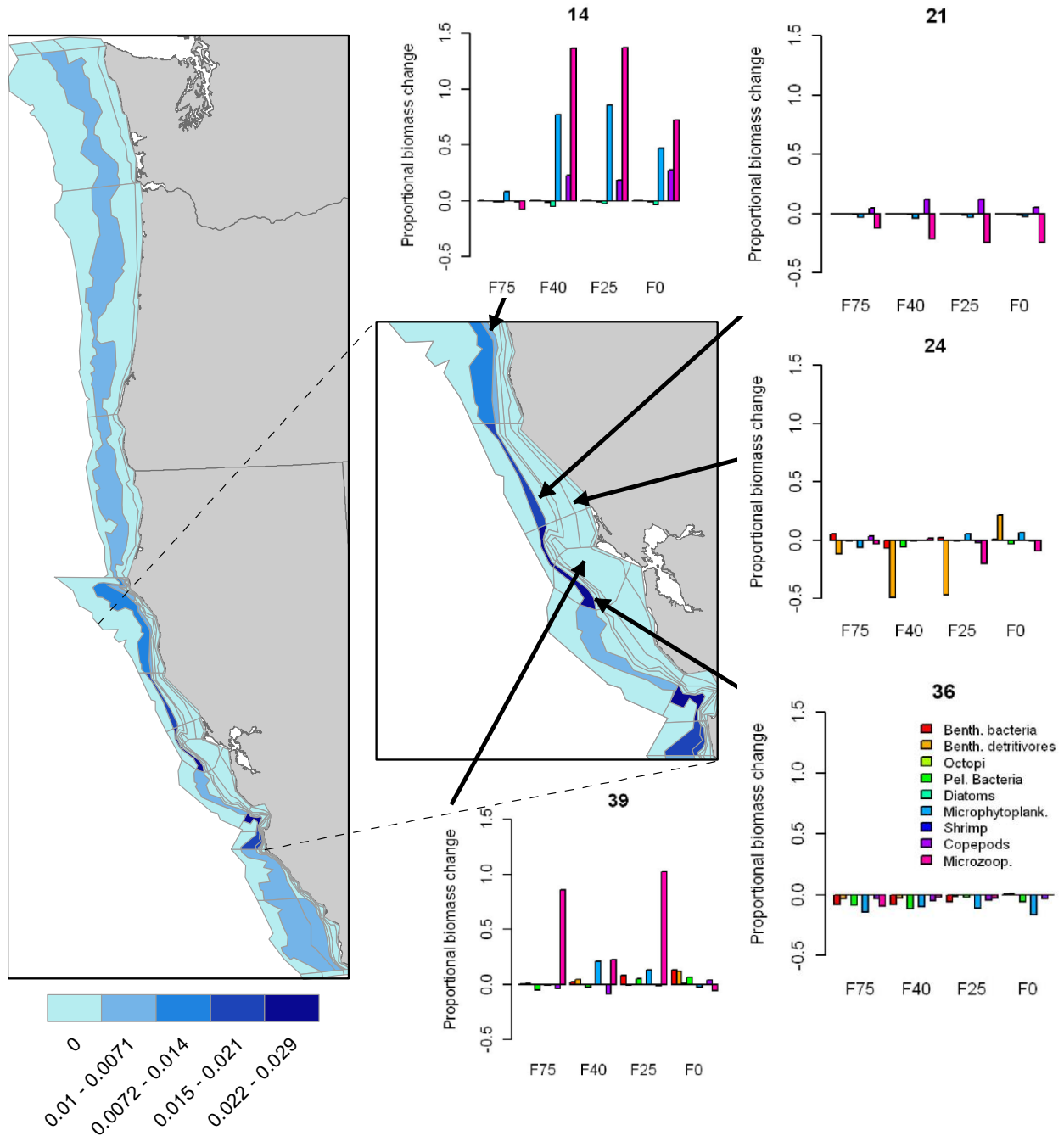


Figure 3. Effects of fishery targeting deep demersal fish on proportional biomass of other functional groups by cell. Map shading indicates deep demersal fish density, as in Figure 1. Nine functional groups were affected by at least one fishing scenario in at least one box across all three target species. For consistency, all nine groups are shown in all panels regardless of level of impact.

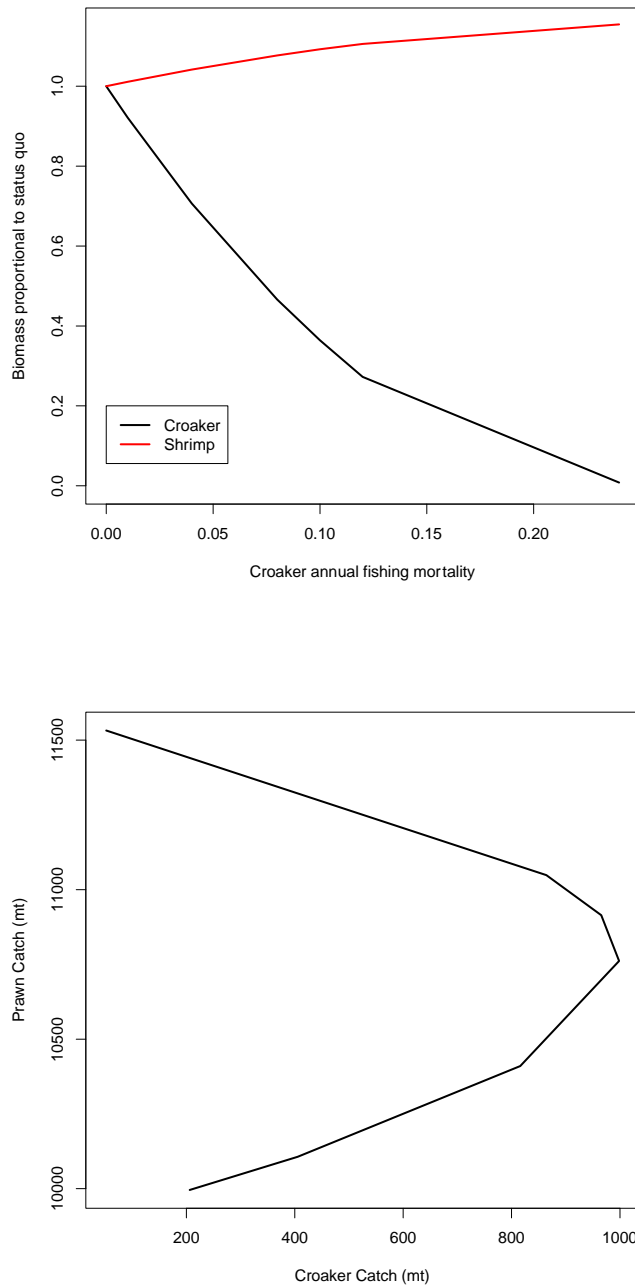


Figure 4. Changes in biomass and yield as a function of increased fishing mortality on nearshore miscellaneous fish (mostly white croaker). Shrimp biomass increased about 15 percent as white croaker biomass declined with fishing (A). Shrimp catches increased as croaker catches increased to MSY (B). Shrimp catches continued to increase as croaker catch declined and the population became depleted.

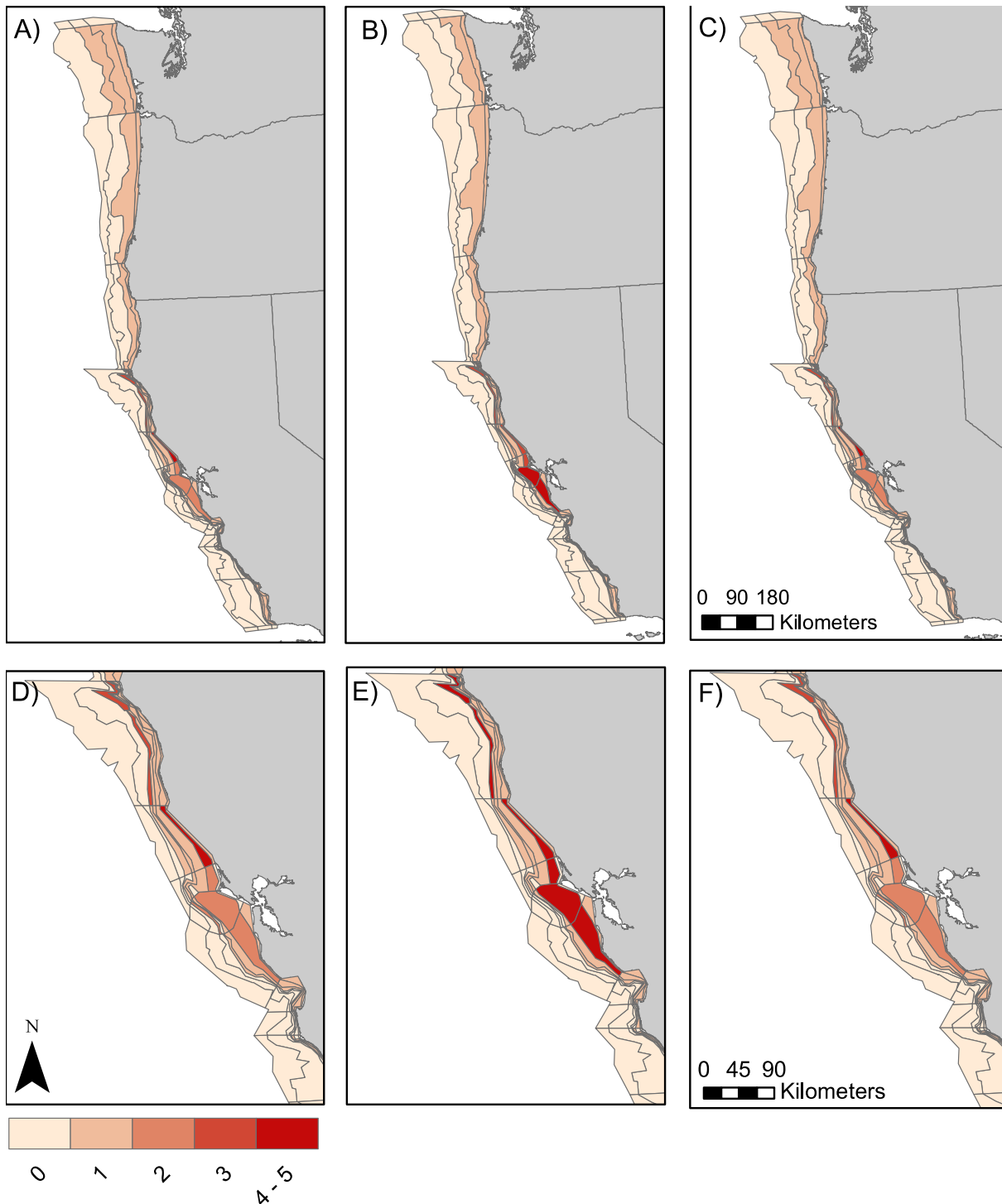


Figure 5. Number of functional groups affected (threshold ± 10 percent) by introducing a fishery targeting the nearshore miscellaneous fish group, by cell. Fishing scenarios represented are F75 (A, D), F40 (B, E), and F25 (C, F). Density of color indicates increasing number of functional groups affected, as indicated by legend. Top and bottom panel extents as in Figure 1.

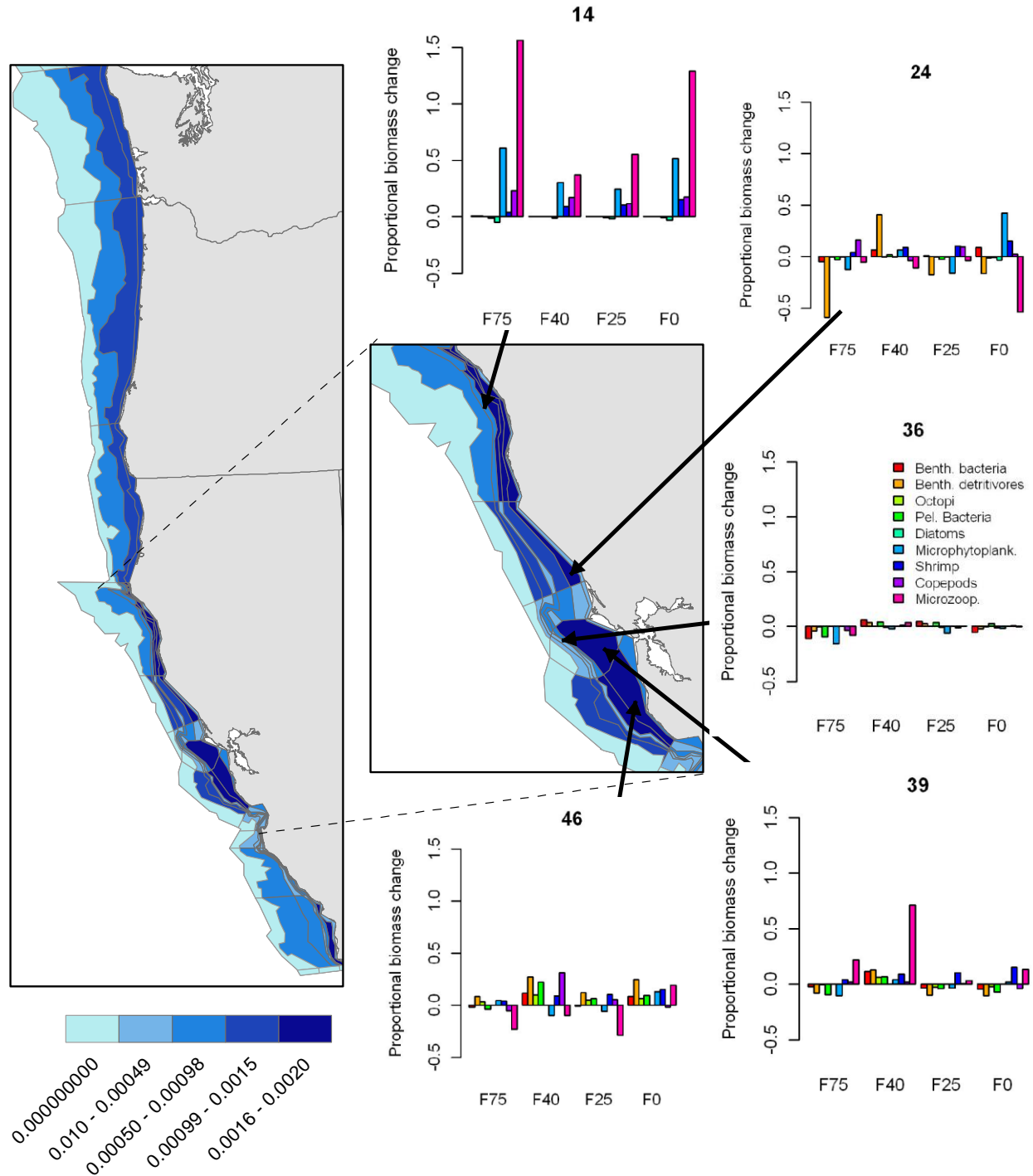


Figure 6. Effects of fishery on nearshore miscellaneous demersal fish group on functional groups by cell for cells with more than two functional groups affected. Maps show summed densities for the target fish group and prawn in status quo scenario. Surrounding plots indicate cascading effects were more common in cells with high densities of both shrimp and the targeted group. Bar coloring is consistent with Figure 4.

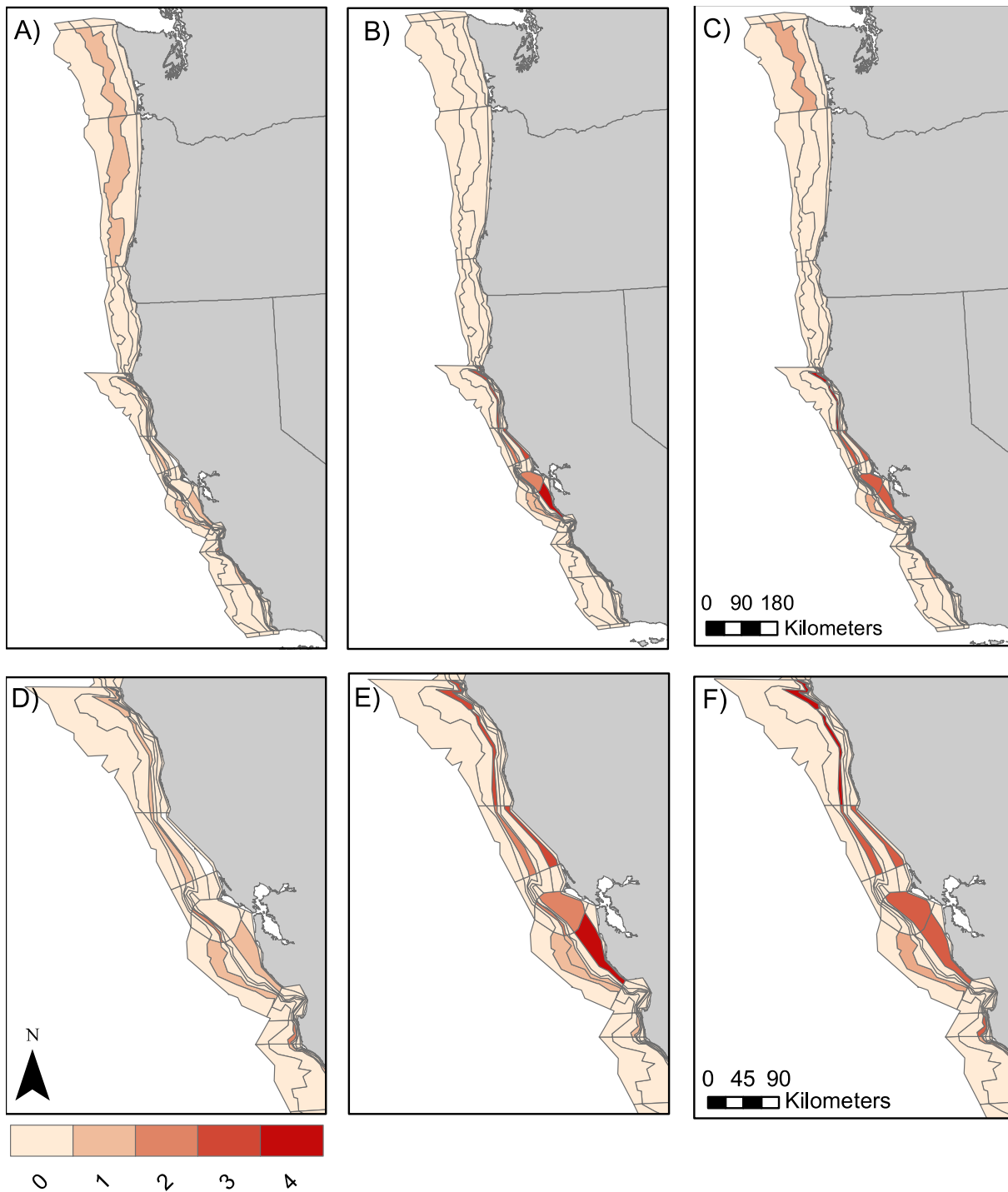


Figure 7. Number of functional groups affected (threshold ± 10 percent) by introducing a shortbelly rockfish pacific fishery, by cell. Fishing scenarios represented are F75 (A, D), F40 (B, E), and F25 (C, F). Density of color indicates increasing number of functional groups affected, as indicated by legend. Top and bottom panel extents as in Figure 1.

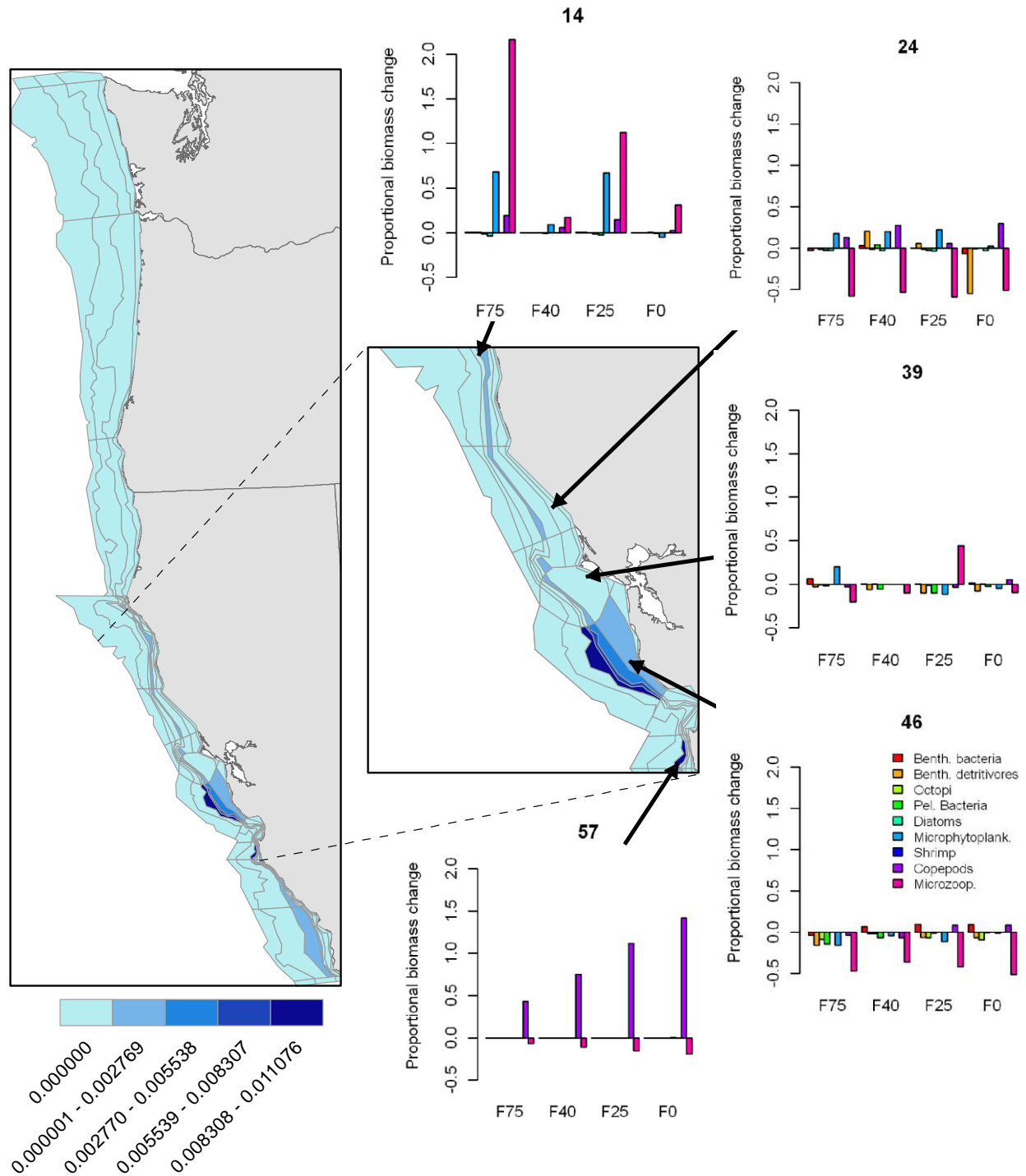


Figure 8. Effects of shortbelly rockfish fishery on proportional biomass of functional groups for cells with two or more impacted function groups. Maps indicate shortbelly density in status quo scenario as in Figure 1. Bar coloring consistent with Figure 4.

APPENDIX A: FINDING MSY/FISHING SCENARIOS

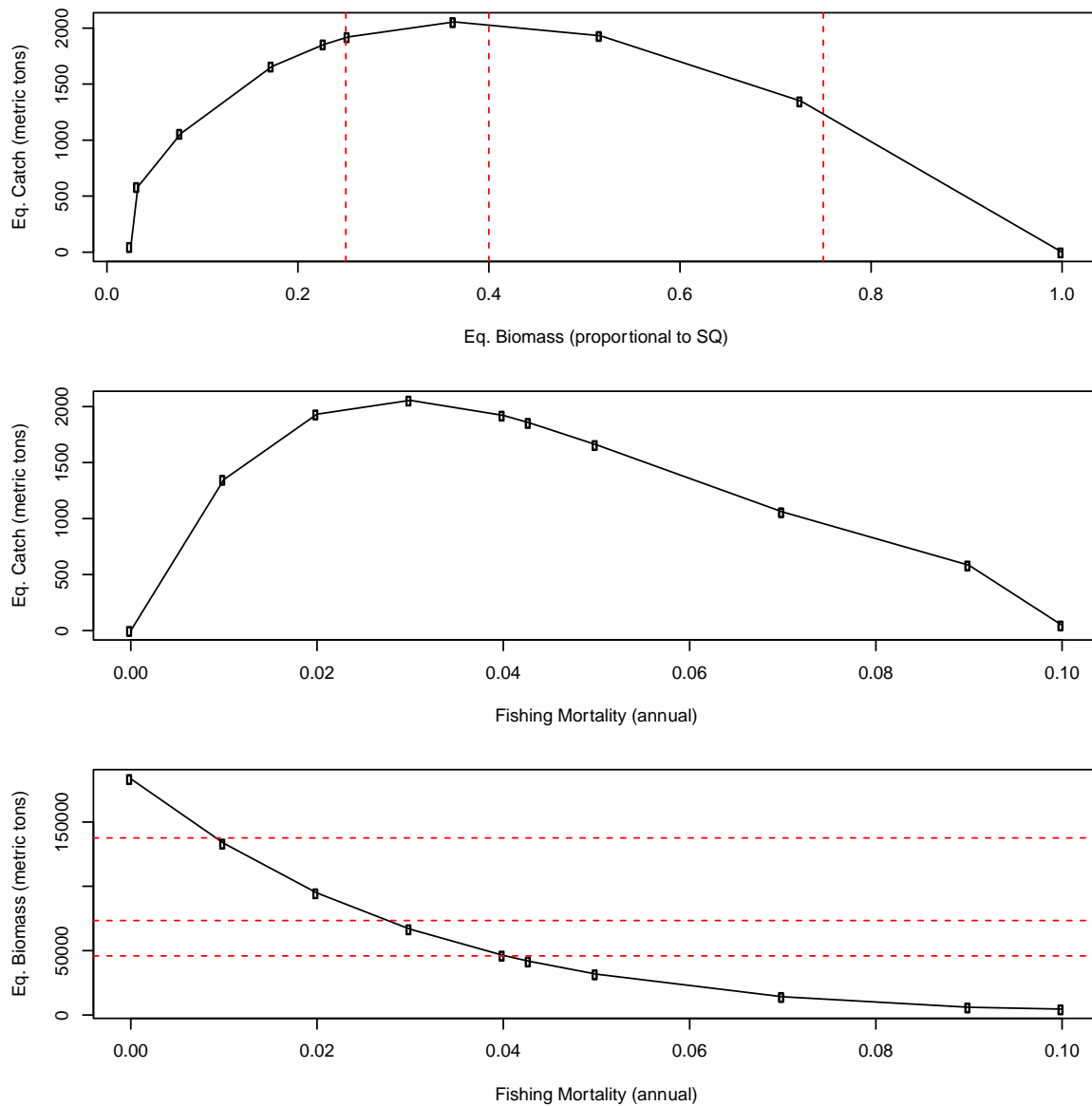


Figure A1. Fishing scenarios for deep demersal fish. Top panel shows equilibrium catch as a function of biomass proportional to the status quo scenario equilibrium. Vertical red lines indicate catch at 25, 40, and 75 percent of status quo. Middle panel shows the relationship between catch and fishing mortality. Maximum sustained yield is the peak of the curve. Bottom panel shows biomass as a function of increasing fishing mortality. Horizontal red lines indicate scenarios as in top panel, indicating fishing mortality required to meet the target biomass.

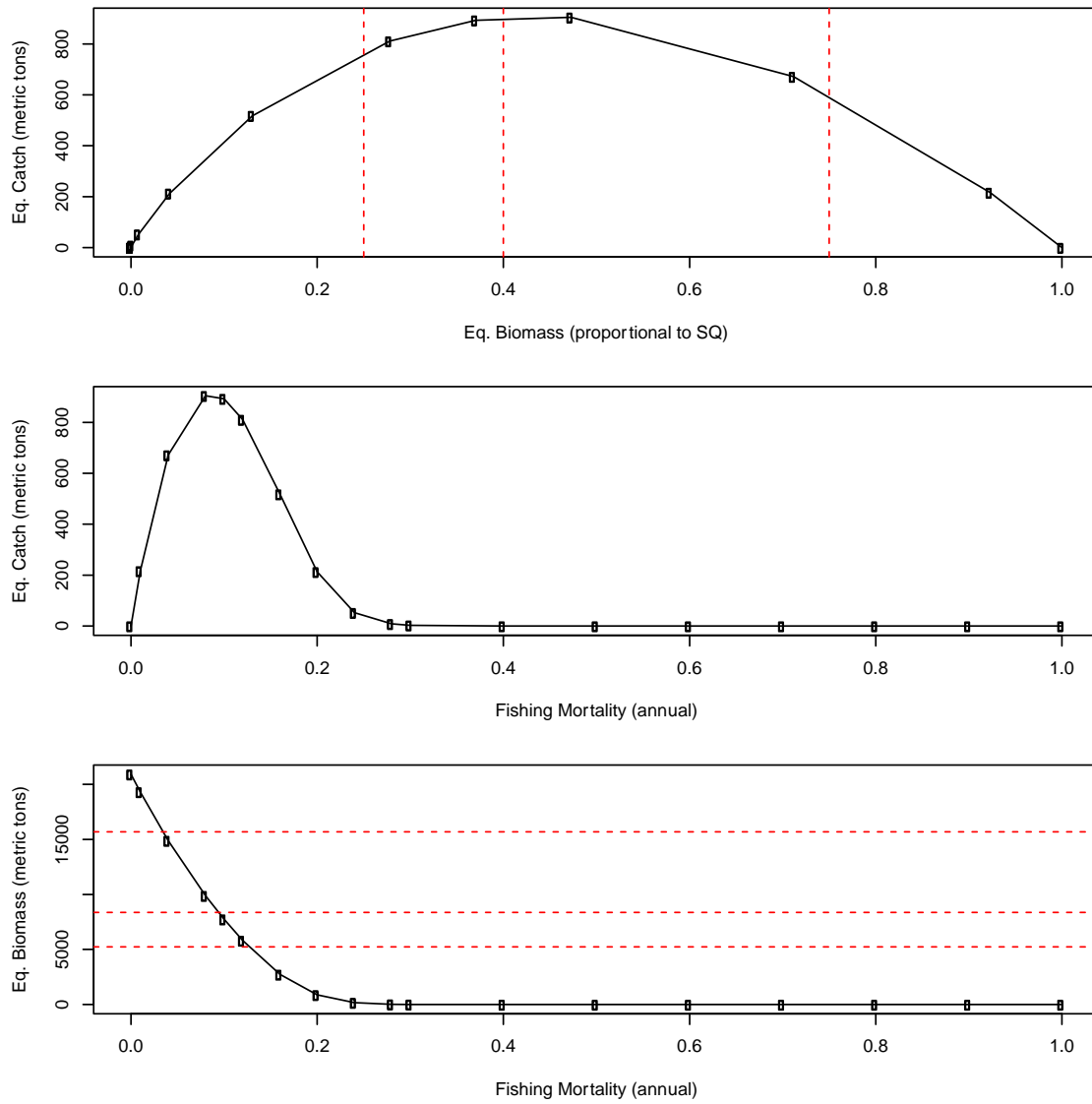


Figure A2. Fishing scenarios for nearshore miscellaneous demersal fish. Panels and axes as in Figure A1.

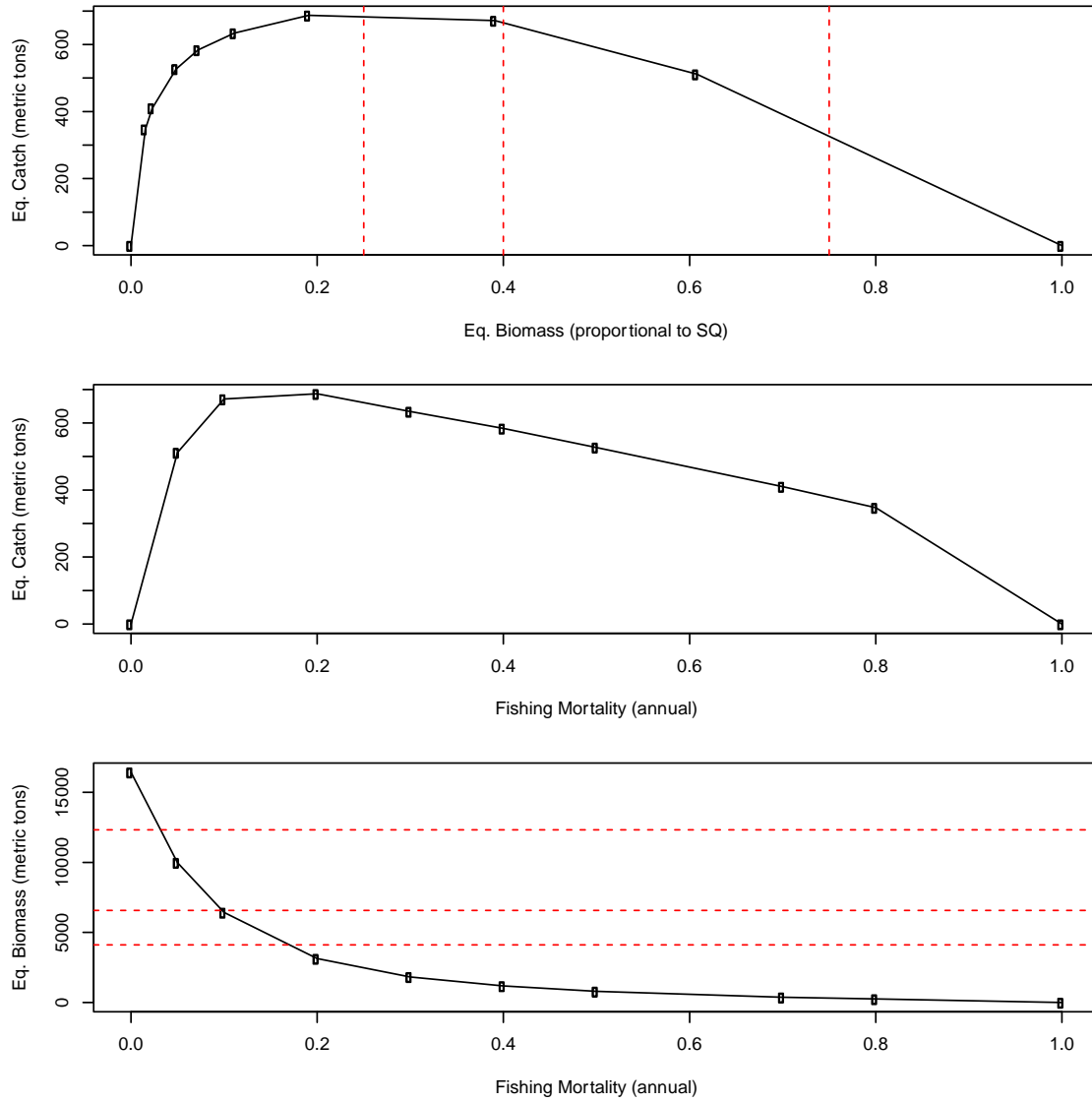


Figure A3. Fishing scenarios for shortbelly rockfish. Panels and axes as in Figure A1.

APPENDIX MS5. BIOLOGICAL AND ECONOMIC EFFECTS OF CATCH CHANGES DUE TO THE PACIFIC COAST GROUND FISH INDIVIDUAL QUOTA SYSTEM

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ABSTRACT

Instituted in 2011, the US West Coast groundfish catch shares program assigns individual groundfish vessels a portion of the quota for target and bycatch species. This new incentive is likely to cap most bycatch, while leading to increases in catch of target species (particularly flatfish) through changes in gear, location and timing of fishing. As part of previous work, Pacific Fishery Management Council staff developed several scenarios for fishery catch under varying assumptions about improvements in targeting accuracy. We investigate the effect of these suggested changes in fishery catch using an Atlantis ecosystem model and an input-output model for Pacific coast fishery economics (IO-PAC). We found that target species in the California current responded directly to the imposed fishing mortality rates. Indirect (trophic) effects were minor and typically involved response of less than 10%. Relative to pre-catch share conditions, the scenarios suggest improved targeting by the groundfish fleet could yield \$27-44 million more in revenue to the fishery sectors (dockside value). At the scale of the broader West Coast economy, the economic model suggests this may translate into \$22-36 million more in total income, which includes employee compensation and earnings of business owners.

INTRODUCTION

Catch Share Program

In 2011 the Pacific Fishery Management Council instituted a program of individual fishing quotas (catch shares) for groundfish fisheries on the US West Coast (Pacific Fishery Management Council 2010a). The individual fishing quotas allow each vessel a fixed proportion of the annual groundfish quota; full observer

coverage and accounting of bycatch is also required. This is a substantial departure from the previous system of two-month landings limits per vessel, with partial observer coverage of the fleet extrapolated to estimate bycatch and discards.

Evidence from other regions suggests that catch shares may improve management performance for target and bycatch species that fall within the individual quota program. Global meta-analyses suggest that individual fishing quotas may reduce the likelihood of fisheries collapse (Costello *et al.* 2008). Experience in British Columbia (Branch *et al.* 2006) and globally (Essington 2009), suggests that individual fishing quotas are likely to decrease discarding, particularly with full observer coverage. Essington (2009) and Melnychuk *et al.* (2011) have found that the primary effect of catch shares was to decrease variability in three metrics: landings, discard rates, or the ratio of catches to quota. There is also some evidence from US and international case studies (Branch 2009) that individual fishing quotas will promote stewardship, in terms of fishers requesting cuts to total catch. On the other hand, individual fishing quotas do not necessarily lead to improved status of non-target species (those outside the quota system) or ecosystem metrics (Gibbs 2010), and they have long been criticized for potential impacts on allocation, fleet consolidation, and economic and social equitability (McCay 1995).

Though individual fishing quotas have been in place for a full year for US West Coast groundfish fisheries, the long term consequences of this policy shift are not yet clear. This is due both to the evolution and learning that is inherent to fishing operations, and the phased implementation of catch shares. Analysis of preliminary data suggests that in 2011 fishers focused on sablefish and deeper water species, leaving a high proportion of rockfish (*Sebastes* spp.) and flatfish quotas unharvested. Depleted rockfish stocks have very low quotas, and potential for high bycatch of rockfish (particularly in shallower areas) may have constrained the ability of the fleets to fully harvest quotas of other target stocks. For example, only a small proportion of quotas of some valuable shelf species such as chilipepper rockfish and lingcod species were caught in 2011, likely due in part to individual captains' concerns about exceeding bycatch caps for several overfished rockfish species and halibut. Additionally, there is limited market demand for flatfish such as Dover sole and arrowtooth flounder, further discouraging targeting of these species. Dover sole is a potentially very large fishery, but in recent years catches have been less than half of total quotas. Total catches of potentially constraining rockfish species were only a small fraction of total quotas in 2011.

Catches of several important target species could be increased substantially depending on future demand and the ability of captains to keep rockfish catches below bycatch caps. Over time, fishermen may become less risk averse if they become more confident that they can acquire more quota to cover unexpected bycatch, and we might expect to see increases in catches of both target

and bycatch species. Conversations with experts as part of an informal scoping exercise ([Engagement section](#)) suggest that fishers are planning or undertaking experiments with gear and fishing areas, in an effort to more precisely harvest target stocks while avoiding particular rockfish species. However, failure to fully exploit quotas of many species may also be due to economic reasons – e.g., lack of demand. For these species catches may increase only if prices increase as a result of increased global demand for fish and development of new markets. Finally, phased implementation of the catch share program involves a two year moratorium on sale of quota, with leasing only during this period (Pacific Fishery Management Council 2010a); quota sales could also change the long-term incentives towards more focused targeting, specialization, and marketing efforts for stocks that were not fully harvested in 2011.

Here we investigate the potential ecological and economic effects of catch changes due to individual fishing quotas for US West Coast groundfish. By coupling an Atlantis ecosystem model (Horne *et al.* 2010; Kaplan *et al.* 2012) with an economic input/output model (Leonard and Watson 2011), we project the economic effects for 1-15 years, and the ecological effects for 1-25 years. Ecosystem dynamics are driven by four scenarios for catches (total mortality) of groundfish species, derived by the Pacific Fishery Management Council (2010b) as part of the environmental impacts statement for the individual quota system. We categorize these three scenarios as *slightly optimistic*, *more optimistic*, and *highly optimistic*, in terms of the ability of vessels to fully harvest the quota of all stocks. We also test a scenario (“*prior to catch shares*”) that represents harvests in 2007, before catch shares were implemented, and likely before any fishing activity that anticipated catch shares. The focus of the harvest increases is directed primarily at Dover sole. Other species catches projected to increase under these various levels of optimism include Arrowtooth flounder (*Atheresthes stomias*), other flatfish (mostly Rex sole, *Glyptocephalus zachirus*, and Pacific sanddab, *Citharichthys sordidus*), Shortspine thornyhead (*Sebastolobus alascanus*), Chilipepper rockfish (*Sebastes goodei*), Yellowtail rockfish (*Sebastes flavidus*), Longspine thornyhead (*Sebastolobus altivelis*), and Lingcod (*Ophiodon elongatus*). These species may experience increases in catch because they are currently harvested at levels well below the quotas; increased harvest could result from direct harvesting or incidental bycatch. These scenarios for catches (Pacific Fishery Management Council 2010b) do not specify the exact changes in fishing techniques or seafood demand that would facilitate these scenarios. Conversations with an industry representative and managers ([Engagement section](#)) suggest that they would likely involve changes in fishing practices, areas fished, or marketing opportunities for low-valued flatfish.

The ecosystem model evaluates both direct (harvest) effects and indirect (food web) effects related to these catch scenarios. We consider the impact on the full food web. Below, we compare Atlantis projections to predictions from single-species stock assessment models for a very limited set of species. The economic input-output modeling allows us to translate Atlantis output, in terms of fishery revenue, to the impact on income in the broader US West Coast economy.

METHODS

Atlantis Model

The Atlantis marine ecosystem model simulates the food web and fisheries in the California Current (Horne *et al.* 2010; Kaplan *et al.* 2012). The model is spatially explicit, and is forced by salinity, temperature, and currents driven by a Regional Ocean Modeling System (ROMS). Functional forms and data for the California Current are described in Brand *et al.* (2007), Horne *et al.* (2010), and Dufault *et al.* (2009); additional core equations are described in Fulton (2001, 2004). The Atlantis code base and recent applications have been summarized by Fulton *et al.* (2011). Additional information is available from <http://atlantis.cmar.csiro.au/>; its application by NOAA to issues in the US and Mexico is described here:

http://www.nwfsc.noaa.gov/publications/documents/atlantis_ecosystem_model.pdf. As part of the 2011 Integrated Ecosystem Assessment, this version of the model was used to screen management scenarios related to gear shifts and spatial management (Kaplan *et al.* 2011). Additionally, those management scenarios were linked to economic impacts (employment and income) by Kaplan and Leonard (Kaplan and Leonard 2012), using an approach similar to the one here.

The “*prior to catch shares*” scenario has catches of groundfish and non-groundfish fleets that match 2007 harvests, including discards where such information is available. A description of the fleets (based on gear type) and harvests under this base scenario is described elsewhere (Kaplan *et al.* 2012; Kaplan and Leonard 2012). All scenarios involved 50 year simulations of the biology, constant harvest rates (%yr⁻¹) with no additional management intervention (such as closed areas or quota reductions), and applications of the economic model to years 1- 15.

The three alternate scenarios (*slightly optimistic, more optimistic, and highly optimistic*) scale these fishing mortality rates by multipliers taken from Pacific Fishery Management Council (2010b). We calculated these multipliers as the ratio of catch per scenario divided by catch under pre-catch shares scenario. These multipliers can be found in Table 1.

| Name in Pacific Fishery Management Council (2010b) | Atlantis Functional Group | Prior to Catch Shares | Slightly Optimistic | More Optimistic | Highly Optimistic |
|--|---------------------------|-----------------------|---------------------|-----------------|-------------------|
| Chilipepper, Yellowtail | Midwater rockfish | 1.00 | 1.00 | 3.51 | 4.02 |
| Shortspine, ½ Slope rockfish | Deep large rockfish | 1.00 | 2.02 | 2.23 | 2.23 |

| | | | | | |
|-----------------------------|----------------------------|------|------|------|------|
| Longspine, ½ Slope rockfish | Deep small rockfish | 1.00 | 2.54 | 2.77 | 2.77 |
| Sablefish | Sablefish | 1.00 | 1.00 | 1.00 | 1.00 |
| Dover sole | Dover sole | 1.00 | 1.85 | 1.85 | 2.54 |
| Arrowtooth, Petrale | Large piscivorous flatfish | 1.00 | 1.38 | 1.38 | 1.38 |
| Other flatfish | Small flatfish | 1.00 | 2.03 | 3.18 | 3.18 |
| Dogfish shark | Small demersal sharks | 1.00 | 1.00 | 1.00 | 1.00 |
| Pacific hake | Pacific hake | 1.00 | 1.00 | 1.00 | 1.00 |
| Lingcod | Lingcod | 1.00 | 1.00 | 1.21 | 1.49 |

Table 1. Multipliers used to increase the fishing mortality rates for groundfish. The leftmost columns illustrate how we matched species groups reported in an environmental impact statement (Pacific Fishery Management Council 2010b) to our Atlantis model functional groups. Fifty percent of the “Slope rockfish” group from the EIS was assigned to the Atlantis deep large rockfish group, and fifty percent to the deep small rockfish group.

IO-PAC Model

We applied an input-output model for Pacific Coast Fisheries (IO-PAC, Leonard and Watson (2011)) to predict how changes in the fishery sector’s revenue would affect income at the scale of the US West Coast (Leonard and Watson, 2011). Note that revenue signifies dockside value (ex-vessel value), while income refers to employee compensation and profits to business owners. Income effects involve both direct effects (to employees and businesses in the fisheries sector), indirect effects (e.g. to shipyards or fuel suppliers), and induced effects through changes in total household spending along the US West Coast. The goal was to broaden the focus beyond the fisheries sector, to the entire West Coast economy.

The methodology follows Kaplan and Leonard (2012). We first calculate total revenue from the fisheries (large groundfish trawler, non-nearshore fixed gear, and shoreside hake midwater trawl), seafood processors, and wholesalers. We then apply IO-PAC to predict income effects 1, 5, 10, and 15 years into the future. Revenue represents all money coming into only the fishing sector (dockside or ex-vessel value of fish,

and gross receipts of seafood processors and wholesalers), while income is calculated from IO-PAC at the scale of the entire West Coast economy. Effects of any fishery sector on the west coast economy include direct effects (income by the fishery sector), indirect effects (income by supporting industries such as shipyards), and induced effects (income effects through coastwide changes in household spending). Though the biological model projects beyond 15 years, we do not apply IO-PAC beyond year 15, due to its assumptions regarding constant prices, costs, and fixed units of inputs required per unit of output. Dockside value of landed seafood is fixed at 2006 prices. We do not report employment changes due to the high uncertainty regarding fleet consolidation under catch shares (Lian *et al.* 2010) and resulting changes in employment in the fishing sectors. In reality, if consolidation occurs this may also modify costs and inputs (e.g. diesel, ice) required by seafood sectors, but for simplicity we hold these at constant values based on data collected prior to implementation of catch shares.

Revenue Comparison between Atlantis and Environmental Impact Statement (Pacific Fishery Management Council 2010b)

Comparable to our Atlantis predictions of harvests under these four scenarios, the Pacific Fishery Management Council (2010b) provides predictions of harvest per scenario. Both predictions for year 1 harvest were converted to revenue :

$$R = 2204.62 \cdot P \cdot C \cdot (1 - D)$$

Where R is revenue per species in dollars, P is the price per pound of the species (in 2006), C is the total catch in metric tons, and D is the discard ratio (Bellman 2008). The coefficient 2204.62 is the number of pounds in a metric ton. Note that since the Atlantis year 1 harvests were calibrated to match the *prior to catch shares scenario* harvests, we expect the Atlantis harvests under other scenarios to differ only slightly from PFMC 2010b harvests, due to ecological dynamics and different groupings of species (e.g. Atlantis functional groups versus PFMC 2010b aggregation at the level of species or “slope rockfish” and “shelf rockfish”).

We provide this simple comparison to illustrate that fishery sector revenue estimates are similar whether taken from the Atlantis ecosystem model or simpler predictions from the PFMC (2010b) environmental impact statement. Since IO-PAC predictions of income are simple multipliers of revenue, income is also comparable whether predicted using Atlantis or from the environmental impact statement.

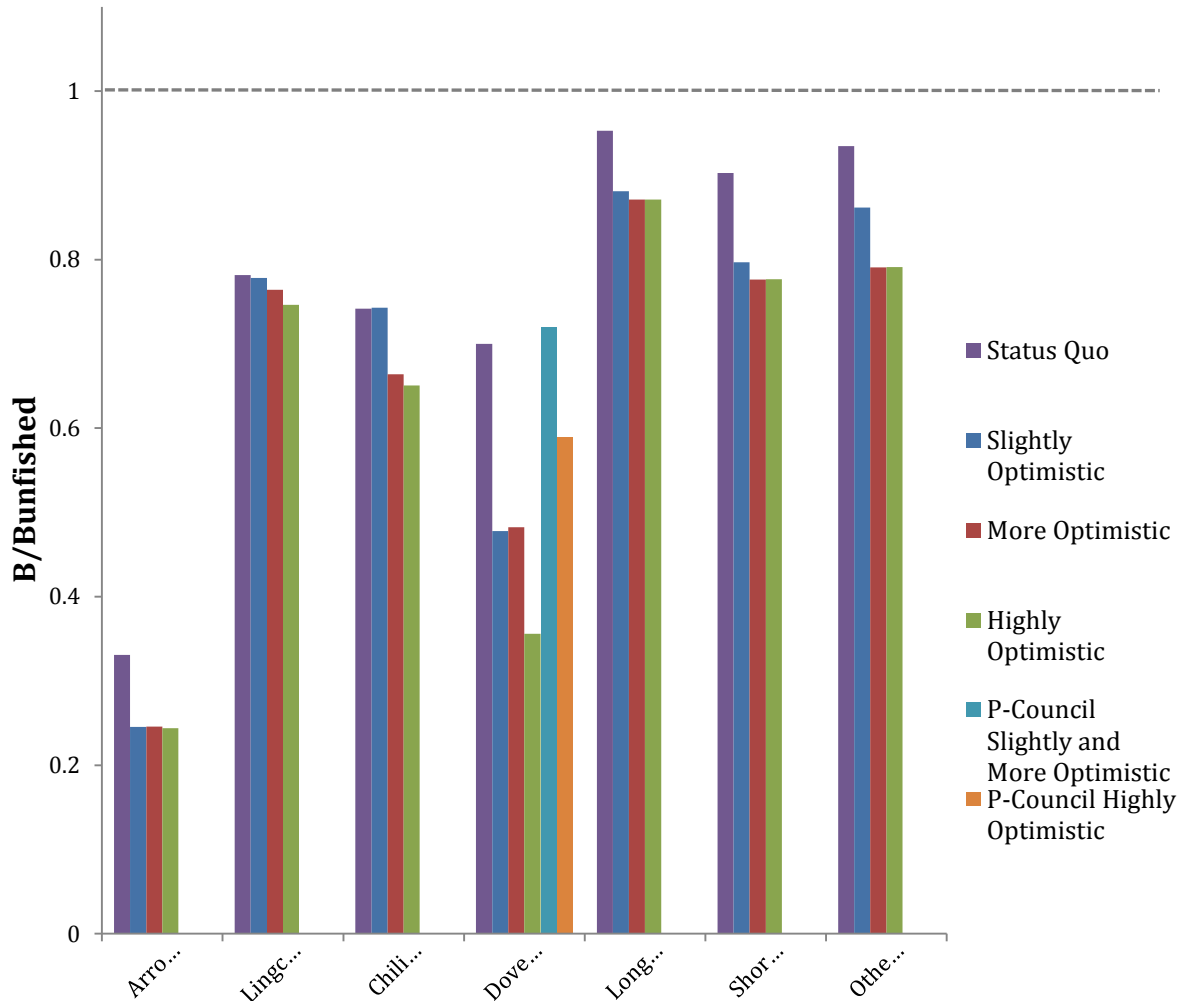


Figure 1. Relative biomass at year 25 predicted by the Atlantis ecosystem model. Also included for comparison are year 25 relative biomass values of Dover sole from a single species stock assessment (Pacific Fishery Management Council 2010b). All other functional groups varied less than 5% among scenarios.

RESULTS

Biological effects on targeted groundfish

Biomass of targeted groundfish that were the focus of the increased fishing effort decreased (Figure 1) due to direct increases in harvest rate (Figure 2). For example, harvest rate for lingcod was low (<2.5%) in the *prior to catch shares* scenario and remained low in all scenarios, which resulted in small comparative reductions in lingcod biomass over the three scenarios. By contrast, harvest rate of Dover sole increased more over the three scenarios than it did for other species, and thus Dover sole had the greatest decrease in biomass, roughly a halving of abundance at year 25. (In all scenarios Dover sole abundance remained above

the current management target, 25% of unfished spawning biomass, through year 25.) Longspine thornyhead (deep small rockfish), shortspine thornyhead (deep large rockfish), arrowtooth (large piscivorous flatfish), other flatfish, and chilipepper and yellowtail rockfish experienced lesser increases in fishing mortality, and resulting biomass reductions of 14% or less. Single-species projections from a stock assessment model also predicted that Dover sole would decline under the *highly optimistic scenario* (PFMC 2010b), but by only about 20% (Figure 2).

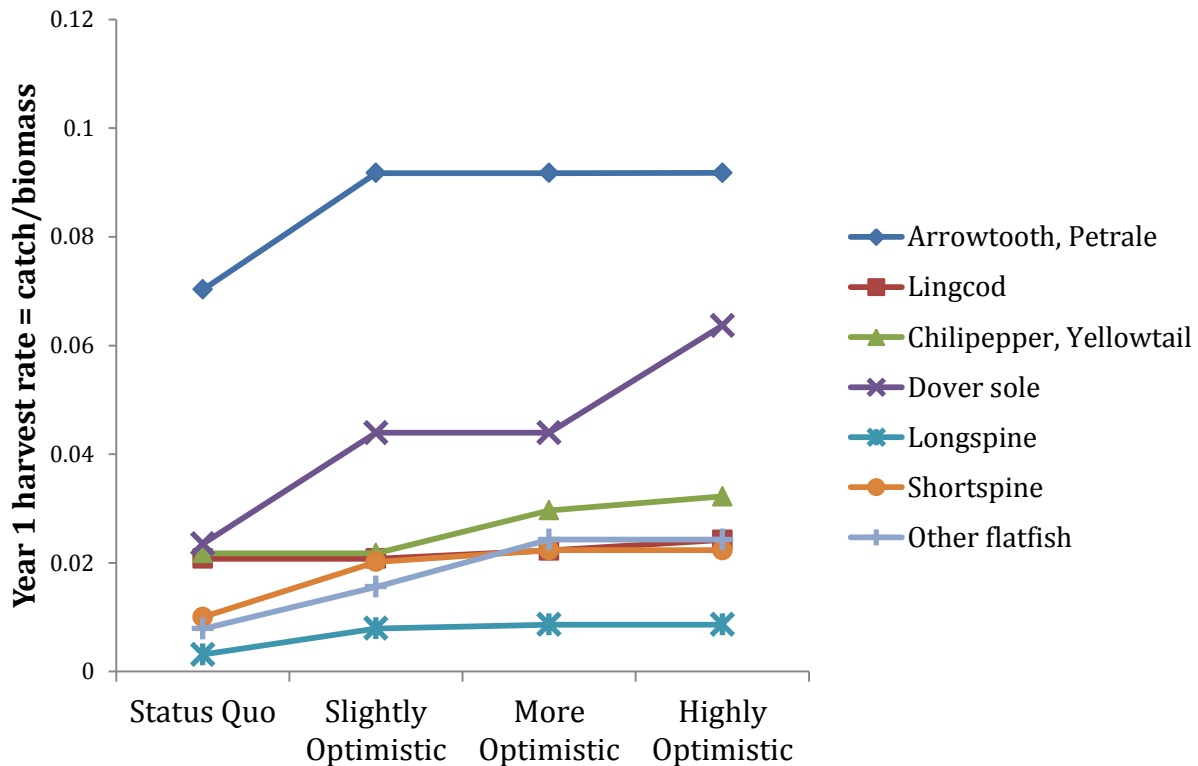


Figure 2. Harvest rate (calculated as harvest rate = catch/biomass) for each species or functional grouper, per scenario.

Trophic Effects

Indirect trophic effects of the catch share scenarios were minor. Functional groups that were not subject to increased fishing pressure in the catch share scenarios did not deviate more than 10% from status quo. The direct reduction in flatfish and some rockfish biomass led to slight reductions in predation pressure on bivalves, shrimp, and mesozooplankton. In the most extreme case (*highly optimistic scenario*, year 50) these species groups increased in biomass by 3%, 2.5%, and 6%, respectively. Predators on these invertebrates increased in abundance — mackerel by 9%, sculpin by 3%, and small shallow rockfish by 3% (a group mostly composed of stripetail and greenstriped

rockfish). Pelagic sharks are heavily dependent on mackerel as prey, and therefore exhibited a comparable increase in biomass (8%).

Economic Effects

Relative to the *prior to catch shares scenario*, all other scenarios resulted in increased revenue for fishing sectors, and related increases in total income in the broader west coast economy. However, two of the three gears exhibited little or no increase to their revenue (Table 2). The non-nearshore fixed gear fleet (longline and pot) exhibited only a 6-9% increase in revenue. This might be expected *a priori*, as this gear catches little Dover sole, and the primary target species (sablefish) for this fleet is currently harvested at close to the allowable quota. The shoreside hake fleets had no increase in revenue, since hake catches were not projected to increase (Table 1) and species other than hake that are caught by this fleet are typically discarded at sea or at the processor (V. Tuttle, NWFSC, pers. comm.). Large groundfish trawlers had markedly higher increases in revenue (34 – 72% across all scenarios and years, Table 2). This gear often targets Dover sole and other species slated for harvest increases in our scenarios.

The increase in revenue for groundfish trawlers under the catch share scenarios led to equivalent increases in terms of that fleet's contributions (direct, indirect, and induced) to coastwide total income in the first year of the most optimistic scenario (Figure 3). High fishing mortality rates (under the most optimistic scenarios) had the largest catches early in the simulations; by year 15 high fishing mortality rates caused declines in biomass, and reduced the differences between catch (or revenue) under catch shares versus the *prior to catch shares scenario* (Table 2).

| Revenue | | | | |
|---|-------------|-----------------------------------|-------------------------------|---------------------------------|
| Percent increase relative to <i>Prior to catch shares scenario</i> | | | | |
| Gear | Year | <i>Slightly optimistic</i> | <i>More optimistic</i> | <i>Highly optimistic</i> |
| Large Groundfish Trawler | 1 | 47 | 55 | 72 |
| | 5 | 42 | 51 | 64 |
| | 10 | 36 | 45 | 53 |
| | 15 | 34 | 40 | 46 |
| Non-nearshore Fixed | 1 | 7 | 9 | 9 |

| | | | | |
|----------------------------------|----|----|----|----|
| Gear | 5 | 6 | 8 | 8 |
| | 10 | 6 | 7 | 8 |
| | 15 | 6 | 8 | 8 |
| Shoreside Hake Midwater Trawl | 1 | 0 | 0 | 0 |
| | 5 | 0 | 0 | 0 |
| | 10 | 0 | 0 | 0 |
| | 15 | 0 | 0 | 0 |
| Processor | 1 | 28 | 32 | 42 |
| | 5 | 25 | 30 | 38 |
| | 10 | 22 | 28 | 33 |
| | 15 | 22 | 26 | 30 |
| Wholesaler | 1 | 28 | 32 | 42 |
| | 5 | 25 | 30 | 38 |
| | 10 | 22 | 28 | 33 |
| | 15 | 22 | 26 | 30 |
| Total | 1 | 28 | 32 | 42 |
| | 5 | 25 | 30 | 38 |
| | 10 | 22 | 28 | 33 |
| | 15 | 22 | 26 | 30 |

Table 2. Percent increase of revenue due to the effects of catch share scenarios, compared to the prior to catch shares scenario prediction for the same year. The color scheme highlights maximum (green) and minimum (red) changes. Proportional increases in income effects are identical to revenue (within 1%), since these scale linearly with revenue. We assume constant prices for seafood over the 15 years.

Overall, if fleets can increase harvests of flatfish and some rockfish to the levels suggested for the *most optimistic scenario*, fishery sector revenue will be approximately \$141.7 million, with \$118.8 million in income effects in the first year of implementation (Figure 3). This is approximately 40% above the *prior to catch share scenario* values of \$100 million in revenue and \$84 million in income effects.

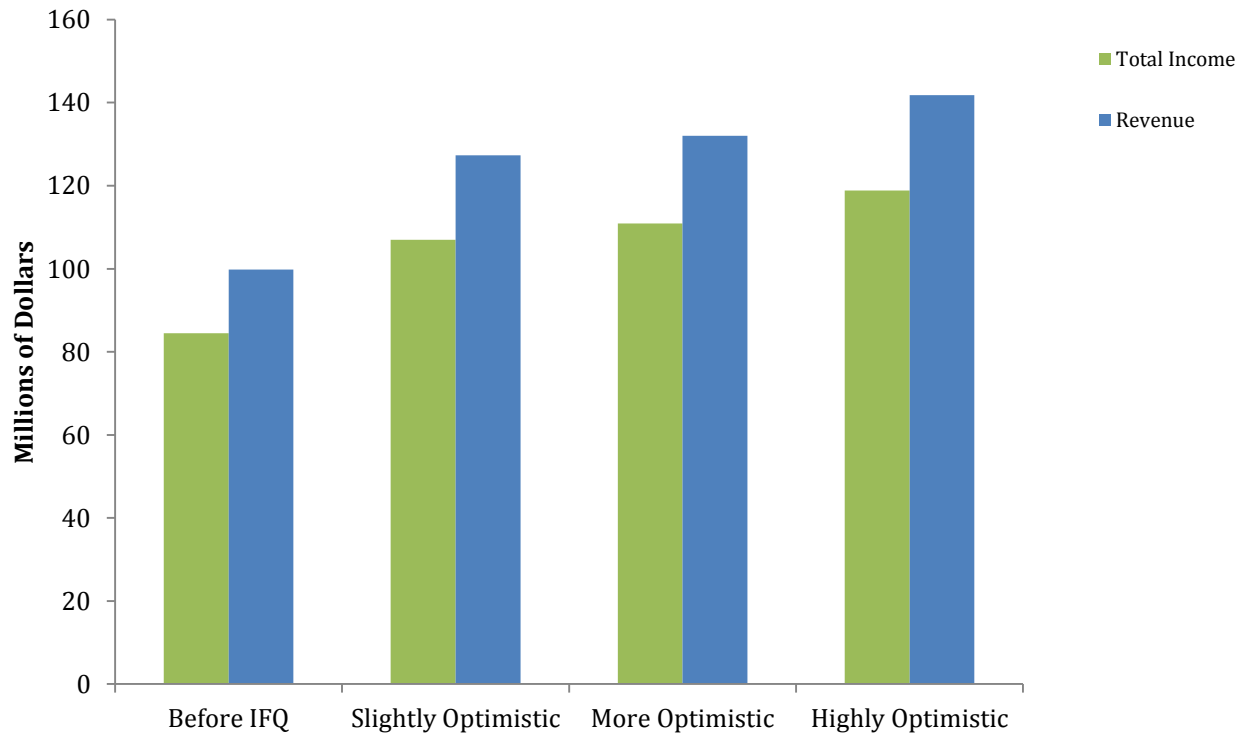


Figure 3. Revenue in fishery sectors, and income effects in the broader West Coast economy. Year 1 predictions. Total income and revenue are represented by bars in millions of dollars (left axis).

Revenue Comparison between Atlantis and Environmental Impact Statement (Pacific Fishery Management Council 2010b)

Focusing only on year 1 revenue from the three fishing fleets, catches from scenarios listed in PFMC (2010b) equate to revenue of \$77 million, \$90 million, \$95 million, and \$99 million for the four scenarios (ranging from *prior to catch shares* to *highly optimistic*). Catches from Atlantis translate into revenues of \$66 million, \$81 million, \$85 million and \$90 million, respectively. In relative terms, the year 1 PFMC (2010b) catches for the *highly optimistic scenario* have revenues 29% higher than *prior to catch shares*, while Atlantis predicts revenues 40% higher than *prior to catch shares*. The \$9-10 million difference between Atlantis and direct application of the PFMC (2010b) is due primarily to the aggregation of species into functional groups for Atlantis; each functional group must have a single (dockside) price, rather than species-level prices that

we applied to the PFMC (2010b) catches. Thus, for example, petrale sole (a valuable flatfish), is grouped with arrowtooth flounder (a low-value species with little market demand).

DISCUSSION: A TALL ORDER, TWO STEPS AT A TIME

The California Current IEA aims to evaluate the potential ecological, economic, and social impacts of management actions and future drivers such as climate change. This is a formidable task. Explicitly linking pressures (e.g. land-based pollution) to responses (e.g. status of protected species) is not always possible with the current generation of models and scientific knowledge; explicitly linking drivers (e.g. human population growth) to pressures is perhaps best handled by a challenging blend of demographic or climate forecasting and formal scenario planning exercises (e.g. Millennium Ecosystem Assessment (2005)). However, given the scope of the IEA and the drivers, pressures, and responses of interest (Figure 4), we can begin to make linkages where the scientific capacity exists. Moreover, by linking published approaches and methodologies, for particular questions we can move two steps at a time, for instance forecasting both ecological impacts and impacts on human communities.

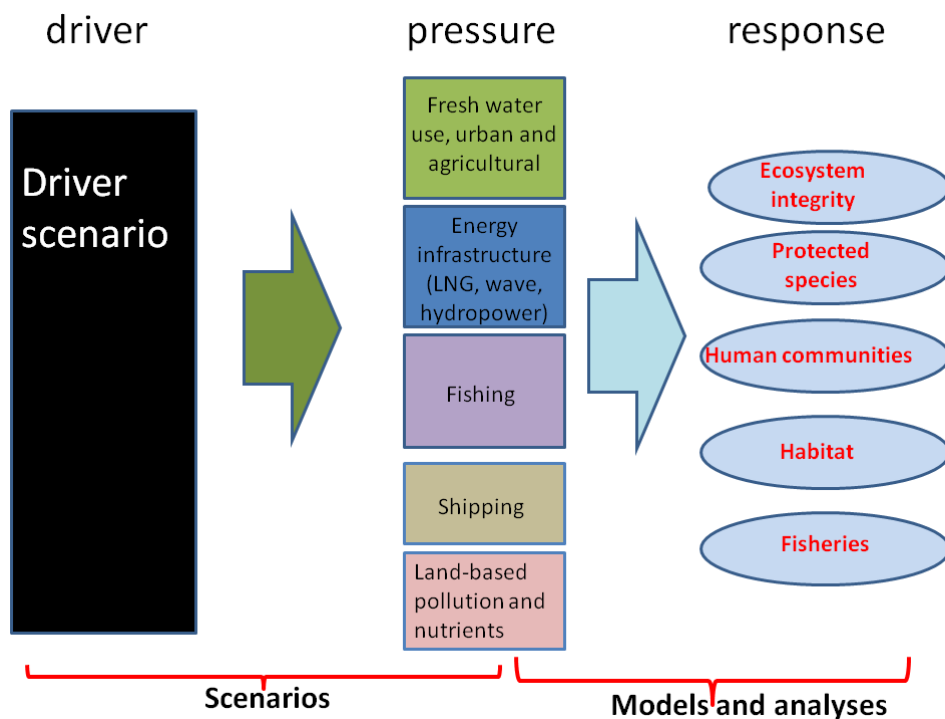


Figure 4. Schematic of Management Testing approach, where drivers are linked to pressures via narrative scenarios, and then quantitative models link pressures to responses.

Of 16 managers, stakeholders, and scientists who identified drivers and pressures relevant to the California Current, eight commented on the potential ecological and economic impacts of the new groundfish catch share program ([Engagement section](#)). Our work here addresses those questions, using two quantitative models to forecast those effects at relevant temporal scales: 1-25 years for biological variables, and 1-15 years for economic values. The Atlantis ecosystem model identifies some minor trophic effects of potential catch share scenarios, but overall suggests that major effects will only occur for fishery target species. The economic IO-PAC model predicts up to 40% increase in income effects by the seafood sectors on the broader West Coast economy, with most of this increase deriving from groundfish trawl revenue. The results can also inform future analyses related to human social wellbeing, such as those by Jacob et al (2012) that can include predictors such as fishery landings and household income.

The models here capture only some of the salient characteristics of the ecosystem, fisheries, and economy, and results should be considered strategic and comparative, rather than definitive and precise. This application of the Atlantis ecosystem model uses coarse functional groups of aggregated species, it assumes smooth recruitment relationships, and it focuses on the groundfish community rather than pelagic species. The fisheries are implemented with constant fishing harvest rates, rather than with a dynamic management response that adjusts harvest rates as biomass varies. The IO-PAC model assumes fixed costs, price, and inputs per unit of output; critically this means that all innovation and learning must be captured in the catch scenarios defined by PFMC (2010b). Other efforts are needed to capture more fine-scale fleet behavior and economic responses to catch shares ([Kaplan et al, AppendixMS6](#)), and to predict long-term economic impacts to the region (Finnoff and Tschirhart 2003). Appropriate application of such strategic models is discussed in Fulton et al. (2011), in particular for ranking management strategies and identifying the relative impacts of threats and pressures. Our results here are strengthened by a comparison to single species stock assessment for Dover sole, and simple revenue calculations that directly expand from PSMFC (2010b). This type of multi-model inference is necessary and appropriate as new models are developed that address drivers and pressures beyond simply fishing.

Though this application focused on direct fishing mortality effects for groundfish, both the Atlantis and IO-PAC frameworks are being expanded to address new drivers, pressures, and ecosystem components. This includes Atlantis forecasts related to climate change and ocean acidification, and regionalized IO-PAC applications that include fleets that harvest salmon and Dungeness crab. Both salmon and crab may be more likely than groundfish to be impacted by global change. Analyses using these tools and others can be used to screen a broad range of management scenarios and climate drivers.

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APPENDIX MS6. FINDING THE ACCELERATOR AND BRAKE IN AN INDIVIDUAL QUOTA FISHERY: LINKING ECOLOGY, ECONOMICS, AND FLEET DYNAMICS OF US WEST COAST TRAWL FISHERIES

Isaac C. Kaplan, Daniel S. Holland, and Elizabeth A. Fulton

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The work is available through <http://icesjms.oxfordjournals.org/>, or by email request to the first author (Isaac.Kaplan@noaa.gov).

ABSTRACT:

In 2011, the Pacific Fisheries Management Council implemented an individual transferrable quota (ITQ) system for the West Coast groundfish trawl fleet. Under the ITQ system, each vessel now receives transferable annual allocations of quota for 29 groundfish species, including target and bycatch species. Here we develop an ecosystem and fleet dynamics model to identify which components of an ITQ system are likely to drive responses in effort, target species catch, bycatch, and overall profitability. In the absence of penalties for discarding over-quota fish, ITQs lead to large increases in fishing effort and bycatch. The penalties fishermen expect for exceeding quota have the largest effect on fleet behavior, capping effort and total bycatch. Quota prices for target or bycatch species have lesser impacts on fishing dynamics, even up to bycatch quota prices of \$50/kg. Ports that overlap less with bycatch species can increase effort under individual quotas, while other ports decrease effort. Relative to a prior management system, ITQs with penalties for exceeding quota lead to increased target species landings and lower bycatch, but with strong variation among species. In addition to providing insights into how alternative fishery management policies affect profitability and sustainability, the model illustrates the wider ecosystem impacts of fishery management policies.

**APPENDIX MS7. COMMERCIAL FISHING ECONOMICS TECHNICAL
REPORT FOR THE SECRETARIAL DETERMINATION ON WHETHER TO
REMOVE FOUR DAMS ON THE KLAMATH RIVER IN CALIFORNIA AND
OREGON**

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ABBREVIATIONS AND ACRONYMS

| | |
|--------|--|
| DPV | Discounted Present Value |
| DRA | Dam Removal Alternative |
| EDRRA | Evaluation of Dam Removal and Restoration of Anadromy |
| EEZ | Exclusive Economic Zone |
| ESA | Endangered Species Act |
| ESU | Evolutionarily Significant Unit |
| FMP | Fishery Management Plan |
| IGD | Iron Gate Dam |
| IMPLAN | Impact Analysis for Planning |
| KBRA | Klamath Basin Restoration Agreement |
| KMZ | Klamath Management Zone |
| KMZ-CA | Klamath Management Zone – California |
| KMZ-OR | Klamath Management Zone – Oregon |
| KRFC | Klamath River Fall Chinook |
| MSFCMA | Magnuson-Stevens Fishery Conservation and Management Act |
| NAA | No Action Alternative |
| NED | National Economic Development |
| NMFS | National Marine Fisheries Service |
| PFMC | Pacific Fishery Management Council |
| RED | Regional Economic Development |
| SONCC | Southern Oregon Northern California Coast |
| SRFC | Sacramento River Fall Chinook |
| USDOI | U.S. Department of the Interior |
| USFWS | U.S. Fish and Wildlife Service |
| USWRC | U.S. Water Resources Council |

TABLE OF CONTENTS (MS7)

| | |
|---|------------|
| I. Introduction..... | 907 |
| II. Existing Fishery Conditions..... | 908 |
| III. Biological Assumptions..... | 915 |
| SONCC Coho | 915 |
| Klamath River Spring and Fall Chinook..... | 916 |
| IV. Commercial Fishing Economic Value for Benefit-Cost Analysis (NED Account)..... | 921 |
| Methodology and Assumptions..... | 921 |
| Alternative 1 – No Action | 923 |
| Alternative 2 – Full Facilities Removal of Four Dams | 925 |
| Alternative 3 – Partial Facilities Removal of Four Dams | 930 |
| V. Commercial Fishing Expenditures for Regional Economic Impact Analysis (RED Account) | 930 |
| Methodology and Assumptions..... | 930 |
| Alternative 1 – No Action | 931 |
| Alternative 2 – Full Facilities Removal of Four Dams | 933 |
| Alternative 3 – Partial Facilities Removal of Four Dams | 937 |
| VI. Summary and Conclusions | 937 |
| VII. References | 939 |
| Appendix A. Salmon Fishery Management..... | 942 |
| Appendix B. Methodologies Used to Quantify Economic Effects of No Action and Action Alternatives | 950 |

LIST OF TABLES AND FIGURES (MS7)

| | |
|--|-----|
| Figure II-1. Ocean salmon management areas south of Cape Falcon, Oregon (graphic by Holly Davis)..... | 908 |
| Figure II-2. Landings of troll-caught Chinook and coho south of Cape Falcon, Oregon (millions of fish), 1981-2010 (sources: PFMC 1990, 1991, 1998, 2009, 2010, 2011b)..... | 910 |
| Table II-1. Landings of troll-caught Chinook and coho (# fish), 1981-2010, by management area..... | 911 |
| Table II-2. Landings of troll-caught Chinook and coho (1000s of pounds dressed weight), 1981-2010, by management area | 912 |
| Table II-3. Ex-vessel value of troll-caught Chinook and coho (\$1000s, base year=2012), 1981-2010, by management area | 913 |
| Table II-4. Average annual harvest of troll-caught Chinook and coho during 1985-1989 – pounds, numbers of fish, and percent of total pounds and fish consisting of coho, by management area..... | 914 |
| Figure III-1. Harvest control rule used in the EDRRA model (En0 = annual escapement to natural areas prior to ocean or inriver harvest, F = harvest rate) (graphic by Michael Mohr, NMFS). | 917 |
| Table III-1. EDRRA model results for the troll fishery under the no action alternative (NAA) and dam removal alternative (DRA) ¹ | 918 |
| Table IV-1. Projected average annual ocean troll harvest of Klamath Chinook and total Chinook (all stocks) attributable to Klamath Chinook abundance, and associated gross and net revenues under Alternative 1 – by management area. ¹ | 923 |
| Table IV-2. Projected average annual ocean troll harvest of Klamath Chinook, total Chinook (all stocks) attributable to Klamath Chinook abundance, and gross and net revenues under Alternative 2, and change in net revenue from Alternative 1 – by management area. | 925 |
| Figure IV-1. Projected annual net revenue under Alternatives 1 and 2 during 2012-61 (calculated according to the methodology described in Appendix B-2)..... | 927 |
| Table IV-3. Discounted present value of the increase in net revenue under Alternative 2 relative to Alternative 1 (2012\$), calculated to illustrate the sensitivity of the estimates to alternative discount rates..... | 928 |
| Figure IV-2. Annual discounted values of the increase in net revenue under Alternative 2 relative to Alternative 1 (2012\$) during the projection period 2012-61, calculated on the basis of alternative discount rates of 0% (no discounting), 2%, 4.125%, 6%, and 8%..... | 929 |
| Table V-1. Average annual gross revenue under Alternative 1, by management area ¹ | 931 |
| Table V-2. Annual regional economic impacts associated with average annual gross revenue projected for Alternative 1, by management area..... | 931 |
| Table V-3. Average annual gross revenue under Alternative 2 and change from Alternative 1 – by management area..... | 933 |

| | |
|---|-----|
| Table V-4. Annual regional economic impacts associated with projected average annual increase in ex-vessel revenue under Alternative 2 relative to Alternative 1, by management area. | 934 |
| Figure A-1. Klamath River adult natural spawner escapement, 1978-2010. Dotted line represents 35,000 escapement floor in effect during 1989-2006 (source: PFMC 2011a) | 943 |
| Figure A-2. Sacramento River adult spawner escapement (natural + hatchery), 1978-2010. Dotted lines represent PFMC escapement goal of 122,000-180,000 (source: PFMC 2011a). | 944 |
| Table A-1. Years of no coho retention (NoCoho), closure of both Chinook and coho fisheries (Closure), and closure of Crescent City portion of KMZ-CA (ClosureCC) ¹ in the troll fishery south of Cape Falcon, 1990-2010, by management area. | 946 |
| Figure A-3. Klamath River fall Chinook ocean abundance index (millions of fish), 1986-2010 (source: PFMC 2011a)..... | 948 |
| Figure A-4. Sacramento River fall Chinook ocean abundance index (1000s of fish), 1983-2010 (source: PFMC 2011a)..... | 949 |
| Table B-1. Equations used to project average annual troll harvest of Klamath Chinook and total Chinook and associated gross and net revenues, by management area i and year t (2012-61), under no action alternative (NAA) and dam removal alternative (DRA). | 950 |
| Table B-2. Parameter values used to estimate Klamath Chinook and total Chinook harvest (all stocks), and gross and net revenue by management area under the no-action and action alternatives..... | 952 |
| Figure B-1. Imports of edible salmon products into the U.S., 1975-2010 (source: NOAA National Marine Fisheries Service, Office of Science and Technology, Silver Spring, MD). | 954 |
| Figure B-2. Ex-vessel prices of troll-caught Chinook in California and Oregon south of Cape Falcon during 1981-2010 and in Southeast Alaska during 1984-2009 (2012\$) (sources: PFMC 1998, 2011b; ADFG 2009). | 955 |
| Figure B-3. Annual landings (pounds dressed weight) and ex-vessel price (2012\$) of troll-caught Chinook south of Cape Falcon, Oregon, 1981-2010 (sources: PFMC 1990, 1991, 1998, 2001, 2011b). | 956 |

I. INTRODUCTION

In March 2012, the Secretary of the Interior – in consultation with the Secretary of Commerce – will make a determination regarding whether removal of four Klamath River dams (Iron Gate, Copco 1, Copco 2 and J.C. Boyle) owned by the utility company PacifiCorp advances restoration of salmonid fisheries and is in the public interest. One of the fisheries potentially affected by the Secretarial Determination is the ocean commercial salmon fishery. This report analyzes the economic effects on that fishery of three alternatives that will be considered by the Secretary:

Alternative 1 – No Action: This alternative involves continued operation of the four dams under current conditions, which include no fish passage and compliance with Biological Opinions by the U.S. Fish and Wildlife Service (USFWS) and NOAA National Marine Fisheries Service (NMFS) regarding the Bureau of Reclamation's Klamath Project Operation Plan.

Alternative 2 – Full Facilities Removal of Four Dams: This alternative involves complete removal of all features of the four dams, implementation of the Klamath Basin Restoration Agreement (KBRA 2010), and transfer of Keno Dam from PacifiCorp to the U.S. Department of the Interior (USDOl).

Alternative 3 – Partial Facilities Removal of Four Dams: This alternative involves removal of selected features of each dam to allow a free flowing river and volitional fish passage for all anadromous species. Features that remain in place (e.g., powerhouses, foundations, tunnels, pipes) would be secured and maintained in perpetuity. The KBRA and transfer of Keno Dam are also part of this alternative.

Throughout this report, Alternative 1 is referred to as the no action alternative and Alternatives 2 and 3 as the action alternatives.

Section II describes existing conditions in the ocean commercial (troll) fishery and Section III describes the biological sources of information underlying the economic analysis of fishery effects. Sections IV and V respectively analyze the alternatives in terms of two 'accounts' specified in guidelines provided by the U.S. Water Resources Council (USWRC 1983): Net Economic Development (NED) and Regional Economic Development (RED). NED pertains to analysis of economic benefits and costs from a national perspective and RED pertains to analysis of regional economic impacts in terms of jobs, income and output. Sections VI summarizes results and conclusions of the previous sections, and Section VII provides a list of references cited in the report.

II. EXISTING FISHERY CONDITIONS

The particular salmon stocks influenced by the no action and action alternatives are the two component populations of the Upper Klamath-Trinity Evolutionarily Significant Unit (ESU)² (Klamath River fall and spring Chinook) and the Southern Oregon Northern California Coast (SONCC) coho ESU. These stocks (like other salmon stocks that originate in rivers south of Cape Falcon, Oregon) generally limit their ocean migration to the area south of Cape Falcon. The area south of Falcon is divided into six fishery management areas: Monterey, San Francisco, Fort Bragg, Klamath Management Zone (KMZ), Central Oregon, and Northern Oregon. For purposes of this analysis, the KMZ (which straddles the Oregon-California border) is divided at the border into two areas: KMZ-OR and KMZ-CA (Figure II-1). To the extent possible, the effects of the alternatives are analyzed separately for each area (including KMZ-OR and KMZ-CA).



Figure II-1. Ocean salmon management areas south of Cape Falcon, Oregon (graphic by Holly Davis).

SONCC coho and Klamath Chinook co-mingle with other salmon stocks in the ocean commercial fishery. The Pacific Fishery Management Council (PFMC) manages such 'mixed stock' fisheries on the principle of 'weak stock management' whereby harvests of healthier stocks are constrained more by the need to protect weaker stocks than by their own abundance (see Appendix A for

² An Evolutionarily Significant Unit is a population or group of populations that is reproductively isolated and of substantial ecological/genetic importance to the species (Waples 1991).

detailed description of PFMC management).³ The implications of weak stock management as it relates to SONCC coho and Klamath Chinook are as follows.

PFMC-managed ocean fisheries south of Cape Falcon are subject to consultation standards for two Chinook and four coho ESUs listed under the Endangered Species Act (ESA) – including the SONCC coho ESU (listed in 1997). To meet consultation standards for the coho ESUs, the PFMC has banned coho retention in the troll fishery in KMZ-CA and KMZ-OR since 1990 and in all other management areas south of Cape Falcon since 1993 (with the exception of limited fisheries in 2007 and 2009 in Central and Northern Oregon).

The major salmon stocks targeted by ocean fisheries south of Cape Falcon are Sacramento River fall Chinook (SRFC) and Klamath River fall Chinook (KRFC). For most of the past three decades, KRFC has been more constraining on the troll fishery than SRFC. Because SRFC and KRFC intermix in the troll harvest, regulations devised to limit harvest of KRFC necessarily constrain SRFC harvest as well to levels below what would have been allowed in the absence of the KRFC constraint.

Figure II-2 describes harvest trends over the past 30 years. Troll harvests south of Cape Falcon declined markedly from the 1980s to the 1990s. A number of factors contributed to that decline – e.g., the more conservative harvest control rule for KRFC adopted in 1989, implementation of weak stock management policies in the 1990s, the spate of ESA listings that occurred during the 1990s, and the 50-50 tribal/non-tribal allocation of Klamath-Trinity River salmon implemented in 1993. These regulatory changes were compounded by drought and El Niño conditions during 1991-92 and 1997-98 that contributed to low Chinook and coho returns and prompted major fishery restrictions during the 1990s. The 1990s were followed by a period of more stable, moderate harvests during 2001-05. During 2006-10 landings fell to record low levels due to low KRFC abundance in the mid-2000s and record low SRFC abundance in the late 2000s. The lack of coho landings since 1993 is due to the non-coho retention policy adopted in that year (Appendix A).

³ See Appendix A for a description of PFMC salmon management.

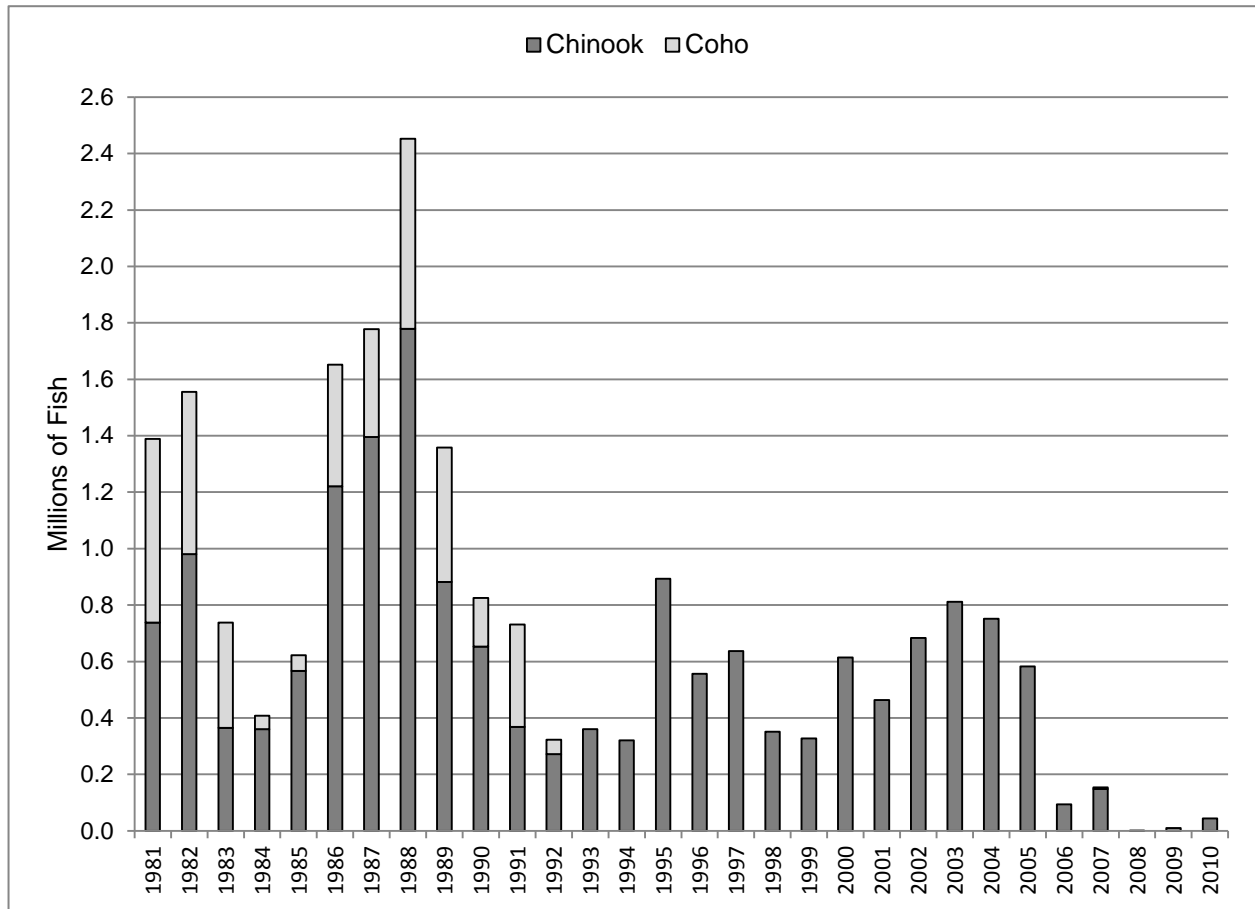


Figure II-2. Landings of troll-caught Chinook and coho south of Cape Falcon, Oregon (millions of fish), 1981-2010 (sources: PFMC 1990, 1991, 1998, 2009, 2010, 2011b).

Tables II-1 and II-2 summarize trends in troll landings (numbers and pounds of fish) by management area. Landings are generally highest in San Francisco and lowest in KMZ-CA and KMZ-OR. Landings reductions began occurring in KMZ-CA and KMZ-OR in the mid-1980s to address conservation concerns for KRFC; low landings remain a persistent features in those areas. The precipitous decline in landings after 2005 was felt in all areas.

Table II-1. Landings of troll-caught Chinook and coho (# fish), 1981-2010, by management area

| Year(s) | Management Area | | | | | | | |
|----------|-----------------|----------|----------|---------|--------|-----------|---------|-----------|
| | Monterey | San Fran | Ft Bragg | KMZ-CA | KMZ-OR | CentralOR | NorthOR | Total |
| 81-85Avg | 85,260 | 186,680 | 124,320 | 124,020 | 61,320 | 170,560 | 190,200 | 942,360 |
| 86-90Avg | 146,460 | 360,480 | 278,380 | 56,120 | 33,920 | 385,940 | 351,700 | 1,613,000 |
| 91-95Avg | 137,720 | 205,480 | 14,760 | 1,540 | 1,000 | 36,820 | 128,240 | 525,560 |
| 96-00Avg | 156,305 | 195,662 | 12,529 | 3,505 | 3,542 | 36,042 | 89,479 | 497,065 |
| 01-05Avg | 64,827 | 210,228 | 96,466 | 12,401 | 5,245 | 117,529 | 151,698 | 658,393 |
| 06-10Avg | 5,330 | 24,806 | 7,906 | 1,752 | 1,188 | 7,736 | 11,598 | 60,315 |
| 2001 | 35,940 | 136,630 | 14,993 | 5,523 | 3,599 | 72,272 | 195,001 | 463,958 |
| 2002 | 69,980 | 242,872 | 65,336 | 13,467 | 6,803 | 122,174 | 162,415 | 683,047 |
| 2003 | 36,099 | 202,876 | 248,875 | 4,044 | 5,072 | 132,156 | 182,066 | 811,188 |
| 2004 | 64,707 | 298,229 | 107,259 | 31,915 | 8,484 | 140,142 | 100,965 | 751,701 |
| 2005 | 117,408 | 170,531 | 45,869 | 7,054 | 2,266 | 120,900 | 118,044 | 582,072 |
| 2006 | 11,204 | 47,689 | 10,835 | 0 | 738 | 1,979 | 21,759 | 94,204 |
| 2007 | 14,009 | 75,254 | 16,116 | 8,762 | 4,097 | 24,096 | 11,393 | 153,727 |
| 2008 | 0 | 0 | 0 | 0 | 236 | 208 | 76 | 520 |
| 2009 | 0 | 0 | 0 | 0 | 0 | 979 | 8,738 | 9,717 |
| 2010 | 1,435 | 1,086 | 12,577 | 0 | 869 | 11,418 | 16,022 | 43,407 |

Sources: PFMC 1990, 1991, 1998, 2009, 2010, 2011b.

Table II-2. Landings of troll-caught Chinook and coho (1000s of pounds dressed weight), 1981-2010, by management area

| Year(s) | Management Area | | | | | | | |
|----------|-----------------|----------|----------|--------|--------|-----------|---------|--------|
| | Monterey | San Fran | Ft Bragg | KMZ-CA | KMZ-OR | CentralOR | NorthOR | Total |
| 81-85Avg | 748 | 1,849 | 1,218 | 967 | 495 | 1,140 | 1,080 | 7,497 |
| 86-90Avg | 1,601 | 3,700 | 2,434 | 624 | 537 | 2,765 | 2,259 | 13,921 |
| 91-95Avg | 1,350 | 1,949 | 194 | 31 | 32 | 339 | 869 | 4,764 |
| 96-00Avg | 1,699 | 2,155 | 146 | 37 | 92 | 435 | 861 | 5,425 |
| 01-05Avg | 756 | 2,704 | 1,268 | 149 | 204 | 1,124 | 1,605 | 7,809 |
| 06-10Avg | 54 | 318 | 163 | 24 | 40 | 86 | 156 | 841 |
| 2001 | 418 | 1,735 | 192 | 64 | 152 | 776 | 1,898 | 5,235 |
| 2002 | 912 | 3,060 | 872 | 162 | 218 | 1,223 | 1,722 | 8,169 |
| 2003 | 498 | 2,753 | 3,096 | 45 | 142 | 1,353 | 1,890 | 9,777 |
| 2004 | 853 | 3,712 | 1,292 | 373 | 267 | 1,214 | 1,256 | 8,967 |
| 2005 | 1,098 | 2,258 | 889 | 102 | 239 | 1,054 | 1,259 | 6,899 |
| 2006 | 87 | 684 | 273 | 0 | 45 | 56 | 290 | 1,435 |
| 2007 | 165 | 888 | 357 | 115 | 101 | 246 | 160 | 2,032 |

| | | | | | | | | |
|------|----|----|-----|---|----|-----|-----|-----|
| 2008 | 0 | 0 | 0 | 0 | 8 | 0 | 20 | 28 |
| 2009 | 0 | 0 | 0 | 0 | 5 | 5 | 82 | 92 |
| 2010 | 20 | 16 | 187 | 4 | 43 | 122 | 226 | 618 |

Sources: PFMC 1990, 1991, 1998, 2001, 2011b.

Table II-3 summarizes trends in salmon ex-vessel revenue⁴ by management area. Revenues (like landings) are generally highest in San Francisco and lowest in KMZ-CA and KMZ-OR. Revenues are influenced by ex-vessel prices² as well as landings. Price declines during 1981-2002 accentuated the landings declines that occurred during the 1980s and 1990s; price increases since 2003 have tended to offset (albeit modestly) the landings declines that occurred after 2005.

Table II-3. Ex-vessel value of troll-caught Chinook and coho (\$1000s, base year=2012), 1981-2010, by management area

| Year(s) | Management Area | | | | | | | |
|----------|-----------------|----------|----------|--------|--------|-----------|---------|--------|
| | Monterey | San Fran | Ft Bragg | KMZ-CA | KMZ-OR | CentralOR | NorthOR | Total |
| 81-85Avg | 3,671 | 9,170 | 5,881 | 4,536 | 2,426 | 4,637 | 3,965 | 34,286 |
| 86-90Avg | 7,003 | 16,751 | 10,884 | 2,736 | 2,219 | 10,983 | 8,128 | 58,703 |
| 91-95Avg | 4,095 | 6,097 | 670 | 104 | 98 | 899 | 2,349 | 14,312 |
| 96-00Avg | 3,755 | 4,912 | 340 | 81 | 217 | 1,038 | 1,950 | 12,292 |
| 01-05Avg | 2,129 | 7,422 | 3,371 | 440 | 608 | 3,206 | 4,280 | 21,456 |
| 06-10Avg | 307 | 1,797 | 925 | 134 | 243 | 500 | 834 | 4,740 |
| 2001 | 1,051 | 4,362 | 483 | 161 | 311 | 1,586 | 3,878 | 11,831 |

⁴ Ex-vessel revenue pertains to the value of fish landed dockside and ex-vessel price to the price received by fishermen for those landings.

| | | | | | | | | |
|------|-------|--------|-------|-------|-------|-------|-------|--------|
| 2002 | 1,766 | 5,927 | 1,689 | 314 | 420 | 2,354 | 3,309 | 15,778 |
| 2003 | 1,164 | 6,432 | 7,233 | 105 | 342 | 3,260 | 4,539 | 23,076 |
| 2004 | 2,912 | 12,672 | 4,411 | 1,273 | 1,096 | 4,982 | 5,096 | 32,442 |
| 2005 | 3,754 | 7,719 | 3,039 | 349 | 872 | 3,846 | 4,577 | 24,156 |
| 2006 | 497 | 3,911 | 1,561 | 0 | 275 | 342 | 1,757 | 8,344 |
| 2007 | 925 | 4,981 | 2,002 | 645 | 607 | 1,451 | 789 | 11,400 |
| 2008 | 0 | 0 | 0 | 0 | 62 | 0 | 150 | 212 |
| 2009 | 0 | 0 | 0 | 0 | 27 | 11 | 188 | 226 |
| 2010 | 114 | 91 | 1,063 | 23 | 245 | 696 | 1,286 | 3,517 |

Sources: PFMC 1990, 1991, 1998, 2001, 2011b.

The effects of the coho non-retention policy implemented in the KMZ in 1990 and in all other areas south of Cape Falcon in 1993 have been disproportionately felt in Oregon. In the five years prior to implementation of this policy (1985-89), coho dependence was most pronounced (both absolutely and as a proportion of total salmon landings) in Central and Northern Oregon. This dependence is somewhat higher when considered in terms of numbers of fish rather than pounds, as weight per fish is lower for coho than Chinook (Table II-4).

Table II-4. Average annual harvest of troll-caught Chinook and coho during 1985-1989 – pounds, numbers of fish, and percent of total pounds and fish consisting of coho, by management area.

| Management Area | 1000s of Pounds Dressed Weight | | | Number of Fish | | |
|-----------------|--------------------------------|------|------------------------|----------------|------|-------------------------|
| | Chinook | Coho | Coho as % of Total Lbs | Chinook | Coho | Coho as % of Total Fish |
| Monterey | 1,403 | 3 | 0.002 | 124,560 | 500 | 0.004 |

| | | | | | | |
|---------------|-------|-------|-------|---------|---------|-------|
| San Francisco | 3,685 | 26 | 0.007 | 345,360 | 4,120 | 0.012 |
| Fort Bragg | 2,532 | 124 | 0.051 | 266,420 | 22,440 | 0.083 |
| KMZ-CA | 537 | 63 | 0.106 | 45,740 | 9,700 | 0.179 |
| KMZ-OR | 444 | 65 | 0.110 | 29,580 | 5,140 | 0.097 |
| Central OR | 2,119 | 643 | 0.217 | 249,400 | 129,700 | 0.318 |
| Northern OR | 1,072 | 1,114 | 0.448 | 107,800 | 231,960 | 0.597 |

Sources: PFMC 1990, 1991, 1998, 2001, 2011b.

III. BIOLOGICAL ASSUMPTIONS

The economic effects of the no action and action alternatives on the troll fishery are largely driven by the effects on fish populations. This section discusses the biological effects of the alternatives on the SONCC coho ESU and Klamath River fall and spring Chinook.

SONCC COHO

The status of SONCC coho is discussed here in the context of NMFS' viability criteria and conclusions of the Biological Subgroup for the Secretarial Determination and an Expert Panel convened in December 2010 to evaluate the effects of the alternatives on steelhead and SONCC coho.

The SONCC coho ESU consists of 28 coho population units that range from the Elk and Rogue Rivers in southern Oregon to the Eel River in Northern California, and includes the coho populations in the Klamath Basin. NMFS' framework for assessing the biological viability of the SONCC coho ESU involves categorization of these component populations into seven diversity strata that reflect the environmental and genetic diversity across the ESU. Risk of extinction is evaluated on the basis of measurable criteria that reflect the biological viability of individual populations, the extent of hatchery influence, and the diversity and spatial structure of population units both within and across diversity strata (Williams *et al.* 2008).

The Klamath diversity stratum includes five population units, three of which (Upper Klamath, Shasta, Scott) are potentially affected by the action alternatives. According to the Biological Subgroup, "None of the population units of Klamath River coho salmon is considered viable at this point in time" (Biological Subgroup 2011, p 89) and "...all five of these Population Units have a high risk of extinction under current conditions" (Biological Subgroup 2011, p 90).

According to the Coho/Steelhead Expert Panel, adverse effects of dam removal on coho would likely be short-lived:

“The short-term effects of the sediment release ... will be injurious to upstream migrants of both species [coho and steelhead].... However, these high sediment concentrations are expected to occur for periods of a few months in the first two years after the beginning of reservoir lowering and sediment flushing. For a few years after that period, suspended sediment concentrations are expected to be higher than normal, especially in high flow conditions, but not injurious to fish (Dunne *et al.* 2011, pp 18-19).

The Expert Panel noted the likely continuation of poor coho conditions under the no action alternative and a modest to moderate response of coho under the action alternatives (the moderate response being contingent on successful KBRA implementation):

“Although Current Conditions will likely continue to be detrimental to coho, the difference between the Proposed Action and Current Conditions is expected to be small, especially in the short term (0-10 years after dam removal). Larger (moderate) responses are possible under the Proposed Action if the KBRA is fully and effectively implemented and mortality caused by the pathogen *C. shasta* is reduced. The more likely small response will result from modest increases in habitat area usable by coho with dam removal, small changes in conditions in the mainstem, positive but unquantified changes in tributary habitats where most coho spawn and rear, and the potential risk for disease and low ocean survival to offset gains in production in the new habitat. Very low present population levels and low demographic rates indicate that large improvements are needed to result in moderate responses. The high uncertainty in each of the many individual steps involved for improved survival of coho over their life cycle under the Proposed Action results in a low likelihood of moderate or larger responses....Nevertheless, colonization of the Project Reach between Keno and Iron Gate Dams by coho would likely lead to a small increase in abundance and spatial distribution of the ESU, which are key factors used by NMFS to assess viability of the ESU” (Dunne *et al.* 2011, p ii).

The Biological Subgroup also notes the benefits of the action alternatives on coho viability:

“Reestablishing access to historically available habitat above IGD will benefit recovery of coho salmon by providing opportunities for the local population and the ESU to meet the various measures used to assess viability (e.g., abundance, productivity, diversity, and spatial structure (Williams *et al.*, 2006). Thus there would be less risk of extinction when more habitat is available across the ESU” (Biological Subgroup 2011, p 92).

The action alternatives are expected to improve the viability of coho populations in the Klamath Basin and advance the recovery of the SONCC coho ESU. However, since the action alternatives do not include coho restoration actions outside the Klamath Basin, they alone will not bring about the conditions that would warrant de-listing of the SONCC coho ESU throughout the species range. The potential for coho harvest under the no action and action alternatives is evaluated in the context of this conclusion.

KLAMATH RIVER SPRING AND FALL CHINOOK

Biological effects of the no action and action alternatives on Klamath River Chinook are evaluated on the basis of two models – the Evaluation of Dam Removal and Restoration of Anadromy Model (Hendrix 2011) and a habitat-based model (Lindley and Davis 2011) – and conclusions of the Biological Subgroup (Hamilton *et al.* 2011) and an Expert Panel convened in January 2011 to evaluate the effects of the alternatives on Klamath River Chinook (Goodman *et al.* 2011).

EVALUATION OF DAM REMOVAL AND RESTORATION OF ANADROMY (EDRRA) MODEL

The Evaluation of Dam Removal and Restoration of Anadromy (EDRRA) model (Hendrix 2011) is a simulation model that provides 50-year projections of Klamath Chinook escapement, as well as separate

harvest projections for the ocean troll, ocean recreational, inriver recreational and tribal fisheries under the no action alternative and dam removal alternatives (denoted as NAA and DRA respectively by Hendrix). Projections from the EDRRA model begin in 2012 (the year of the Secretarial Determination) and span the period 2012-61. The harvest projections for the DRA reflect the following assumptions: (i) active introduction of Chinook fry to the Upper Basin beginning in 2011, (ii) short-term effects on Chinook of sedimentation associated with dam removal, (iii) gains in the quantity and quality of salmonid habitat associated with dam removal and KBRA, and (iv) loss of Iron Gate as a production hatchery in 2028.

The 50-year escapement and harvest projections provided by the model were each iterated 1000 times to capture the influence of uncertainties in model inputs on model outputs. The harvest projections pertain to Klamath/Trinity River Chinook and do not distinguish between spring and fall runs. Klamath/Trinity Chinook harvest (all fisheries combined) is estimated for each simulated year on the basis of the KRFC harvest control rule recommended by the PFMC to NMFS in June 2011 as part of a pending amendment to the Pacific Salmon FMP (Figure III-1). As an added constraint, the model also caps the forecast harvest rate for age-4 KRFC in the ocean fishery at 16 percent to address the consultation standard for California Coastal Chinook (listed as 'threatened' in 1999 – see Appendix A).

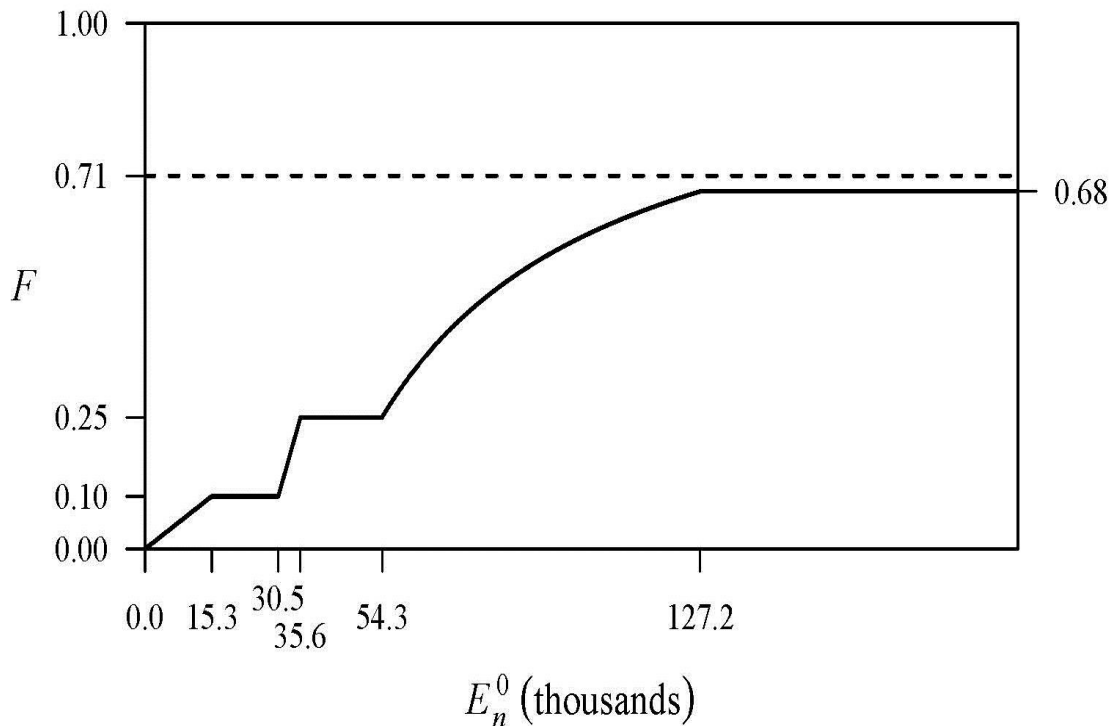


Figure III-1. Harvest control rule used in the EDRRA model (E_n^0 = annual escapement to natural areas prior to ocean or inriver harvest, F = harvest rate) (graphic by Michael Mohr, NMFS).

As reflected in Mohr (in prep) and consistent with PFMC practice, the model distributes the allowable harvest among fisheries as follows: 34.0 percent to the ocean commercial fishery, 8.5 percent to the ocean recreational fishery, 7.5 percent to the inriver recreational fishery (up to a maximum of 25,000 fish – with any surplus above 25,000 allocated to escapement), and 50.0 percent to tribal fisheries. The 50 percent tribal

share is a 'hard' allocation specified by the Department of the Interior (USDOI 1993) on behalf of the Yurok and Hoopa Valley Tribes. The distribution of the remaining 50.0 percent among the three non-tribal fisheries represents customary practice rather than mandatory conditions (Appendix A).

Table III-1 summarizes model results for the entire 50-year projection period (2012-61) and for the following subperiods: (i) 2012-20 (pre-dam removal, hatchery influence); (ii) 2021-32 (post-dam removal, continued hatchery influence), and (iii) 2033-61 (post-dam removal, no hatchery influence).⁵

Table III-1. EDRRA model results for the troll fishery under the no action alternative (NAA) and dam removal alternative (DRA)¹

| Model Results | Time Period | | | |
|--|-------------|---------|---------|---------|
| | 2012-61 | 2012-20 | 2021-32 | 2033-61 |
| 50 th percentile harvest: % diff between NAA and DRA ¹ | +43% | +7% | +60% | +47% |
| 5 th percentile harvest: % diff between NAA and DRA ¹ | -57% | -77% | -46% | -55% |
| 95 th percentile harvest: % diff between NAA and DRA ¹ | +725% | +421% | +821% | +780% |
| Average # years when DRA harvest > NAA harvest: % diff between NAA and DRA ² | 70% | 54% | 78% | 71% |
| Average # years when pre-harvest adult natural spawning escapement ≤ 30,500: % diff between NAA and DRA ³ | -66% | -4% | -79% | -80% |

¹ Source: EDRRA model outputs provided by Hendrix (2011). Derivation provided in Appendix B.1.b.

² Derivation provided in Appendix B.3.

³ Derivation provided in Appendix B.4.

2012-61: 50-year projection period

2012-20: pre-dam removal

2021-32: post-dam removal, hatchery influence

2033-61: post-dam removal, no hatchery influence

⁵ The model assumes that Iron Gate would cease to operate as a production hatchery in 2028. Hatchery influence on the fishery would continue for another 3-4 years (the length of the life cycle of the last year class released from the hatchery).

The EDRRA model assumes that ocean abundance is known without error and that the harvest control rule exactly achieves the escapement objective (Hendrix 2011). Given that the absolute harvest projections provided by the model are an idealized version of real world conditions, model results are best considered in terms of relative rather than absolute differences between alternatives. The average percent difference between EDRRA's 50th percentile harvest projections for the NAA and DRA is +43 percent for the troll fishery. The annual increase varies by subperiod, with harvest increasing by +7 percent prior to dam removal (2012-2020), peaking at +60 percent during the 12 years after dam removal when the fishery is still influenced by hatchery production (2021-32), then diminishing somewhat to +47 percent during 2033-61 after hatchery influence dissipates in 2032 (Table III-1).

EDRRA model results indicate that the 5th percentile harvest value for the DRA is 57 percent lower than the 5th percentile value for the NAA and that the 95th percentile harvest value is 725 percent higher; that is, the DRA harvest distribution is positively skewed and exhibits a high degree of overlap with the NAA harvest distribution. The EDRRA model also provides information regarding the percent of simulated years in which DRA harvest exceeds NAA harvest (50 percent indicating no difference between the two alternatives). These paired comparisons were made possible by applying the parameter draws associated with each iteration of the simulation to both the NAA and DRA. The results in Table III-1 indicate virtually no difference between the alternatives during 2012-20 (54 percent) but higher harvests under DRA in the two subsequent subperiods (2021-32 and 2033-61) in a notable majority of years (78 percent and 71 percent respectively).

The harvest control rule incorporated into the EDRRA model (Figure III-1) limits the harvest rate to 10 percent or less when pre-harvest escapements fall below 30,500 adult natural spawners. Escapements this low would likely be accompanied by major regulatory restrictions and adverse economic conditions for the fishery. Such conditions occur in 66 percent fewer years under the DRA than the NAA – with the greatest declines (-79 percent during 2021-32, -80 percent during 2033-61) occurring in the post-dam removal years (Table III-1).

BIOLOGICAL SUBGROUP

According to the Biological Subgroup, the action alternatives are expected to provide habitat favorable to spring Chinook:

“If dams were removed it is reasonable to expect reestablished spring-run Chinook salmon to synchronize their upstream migration with more natural flows and temperatures. The removal of Project reservoirs would also contribute important coldwater tributaries (e.g., Fall Creek, Shovel Creek) and springs, such as the coldwater inflow to the J.C. Boyle Bypassed Reach, to directly enter and flow unobstructed down the mainstem Klamath River, thereby providing thermal diversity in the river in the form of intermittently spaced patches of thermal refugia. These refugia would be useful to migrating adult spring-run Chinook salmon by extending opportunities to migrate later in the season. The thermal diversity would also benefit juvenile salmon” (Hamilton *et al.* 2011, p 87).

LINDLEY/DAVIS HABITAT MODEL

The Lindley/Davis habitat model focuses on potential Chinook escapement to the Upper Basin above Iron Gate Dam (IGD). The analytical approach involved compilation of escapement and watershed attribute data for 77 fall and spring Chinook populations in various watersheds in Washington, Oregon, Idaho and Northern California, and comparison of those attribute sets with the attributes of Upper Basin watersheds.

Based on their analysis, the authors concluded that Upper Basin attributes fall well within the range of spring bearing watersheds. According to Lindley and Davis:

“Our model predicts a fairly modest increase in escapement of Chinook salmon to the Klamath basin if the dams are removed. The addition of several populations of spring-run Chinook salmon with greater than 800 spawners per year to the upper Klamath would significantly benefit Klamath Chinook salmon from a conservation perspective, in addition to the fishery benefits....The last status review of the UKTR [Upper Klamath and Trinity Rivers] ESU expressed significant concern about the very poor status of the spring-run component of the ESU (Myers *et al.* 1998). Viable populations of spring-run Chinook salmon in the upper Klamath would increase the diversity and improve the spatial structure of the ESU, enhancing its viability (McElhaney *et al.*, 2000) and improving the sustainability of the ESU into the uncertain future” (Lindley and Davis 2011, p 13).

CHINOOK EXPERT PANEL

The Chinook Expert Panel concluded that “The Proposed Action offers greater potential for increased harvest and escapement of Klamath Chinook salmon than the Current Conditions” (Goodman *et al.* 2011, p 16). More specifically, the Panel noted that

“...a substantial increase⁶ in Chinook salmon is possible in the reach between Iron Gate Dam and Keno Dam. A modest or substantial increase in Chinook upstream of Keno Dam is less certain. Within the range of pertinent uncertainties, it is possible that the increase in Chinook salmon upstream of Keno Dam could be large, but the nature of the uncertainties precludes attaching a probability to the prediction by the methods and information available to the Panel. The principal uncertainties fall into four classes: the wide range of variability in salmon runs in near-pristine systems, lack of detail and specificity about KBRA, uncertainty about an institutional framework for implementing KBRA in an adaptive fashion, and outstanding ecological uncertainties in the Klamath system that appear not to have been resolved by the available studies to date” (Goodman *et al.* 2011, p 7).

With regard to spring Chinook, the Panel noted:

“The prospects for the Proposed Action to provide a substantial positive effect for spring Chinook salmon is much more remote than for fall Chinook. The present abundance of spring Chinook salmon is exceptionally low and spawning occurs in only a few tributaries in the basin. Under the Proposed Action, the low abundance and productivity (return per spawner) of spring Chinook salmon will still limit recolonization of habitats upstream of IGD. Intervention would be needed to establish populations in the new habitats, at least initially. Harvests of spring Chinook salmon could occur only if spring Chinook salmon in new and old habitats survive at higher rates than at present. Therefore, habitat quality would need to be higher than at present, and KBRA actions would need to greatly improve survival of existing populations of spring Chinook salmon. Factors specifically affecting the survival of spring Chinook salmon have not been quantified” (Goodman *et al.* 2011, p 25).

⁶ The Panel defined the term ‘substantial increase’ to mean ‘a number of fish that contributes more than a trivial amount to the population’ and cited 10 percent of the average number of natural spawners or 10,000 fish as a rough approximation to what they mean by ‘substantial’. As indicated in their report, “The Panel does not suggest that this figure is a likely increase or a minimum increase that is expected. It is only used as a benchmark for our discussions and to provide a basis for interpreting our response to the question” (Goodman *et al.* 2011, p 7, footnote 3).

IV. COMMERCIAL FISHING ECONOMIC VALUE FOR BENEFIT-COST ANALYSIS (NED ACCOUNT)

METHODOLOGY AND ASSUMPTIONS

The economic analysis provided here assumes that the troll fishery will continue to be constrained by consultation standards associated with ESA listings and that KRFC will continue to be a binding constraint in most areas south of Cape Falcon. This has been the case in most years since the PFMC initiated its weak stock management policy in the early 1990s. Notable exceptions occurred in the late 2000s, when abundance of SRFC fell to record low levels and SRFC became the binding constraint on the troll fishery in all areas south of Cape Falcon. However, as indicated in Appendix A, it is not clear whether such low SRFC abundances signal a future pattern of persistent low abundances, are part of a cyclical pattern, or are events that may recur on a rare or occasional basis.

SONCC COHO

As indicated in Section II.A, the SONCC coho ESU is listed as ‘threatened’ under the ESA. This ESU includes coho populations both inside and outside the Klamath Basin. The action alternatives are expected to increase the viability of Klamath River coho populations and advance recovery of the ESU (Hamilton *et al.* 2011, Dunne *et al.* 2011). However, since the action alternatives do not include coho restoration outside the Klamath Basin, they alone will not create conditions that would warrant de-listing of the SONCC coho ESU throughout its range. Thus, while they are expected to provide long term, positive biological effects, the action alternatives are not likely to affect the availability of coho to the troll fishery.

KLAMATH RIVER SPRING AND FALL CHINOOK

The EDRRA model (Hendrix 2011) is the basis for the quantitative projections of harvest, gross revenue and net revenue used to compare the no action and action alternatives. These variables were estimated as follows:⁷

As indicated in Section III.B.1, the absolute harvest projections provided by the EDRRA model reflect idealized rather than real world conditions. Thus model results are best considered in terms of relative rather than absolute differences between alternatives. To anchor EDRRA projections to the real world, average annual troll harvest of Klamath Chinook during 2001-05 (35,778 fish, according to PFMC 2011) was used to characterize the no action alternative. Annual harvest under the DRA (51,082 fish) was estimated by scaling average 2001-05 harvest upward, based on the difference between EDRRA’s 50th percentile harvest projections for the NAA and DRA (+43 percent, according to Table III-1). The years 2001-05 were selected as the base period for the following reasons: KRFC fell within a moderate range of abundance during those years (Figure A-3); abundance of SRFC (which is targeted along with KRFC in the troll fishery south of Cape Falcon) also fell within a moderate range (Figure A-4); and management constraints and policies that are likely to continue into the future – e.g., policies established in the 1990s to protect weaker stocks (including

⁷ See Appendix B for more details regarding the methods and assumptions underlying the harvest and revenue projections for each alternative.

ESA-listed stocks), the 50-50 tribal/non-tribal harvest allocation – were well established by that time. Record low fishery conditions experienced after 2005 made those years unsuited for base period characterization.⁸

(ii) Harvest of Klamath River Chinook varies by management area due to factors such as the biological distribution of the stock and fishery regulations. To reflect the influence of these factors, annual average Klamath Chinook harvest projected under the no action and action alternatives was distributed among management areas, based on the relative geographic distribution of KRFC harvests experienced in the troll fishery during the 2001-05 base period (data source: Michael O'Farrell, NMFS).⁹

In San Francisco, Fort Bragg, KMZ-CA, KMZ-OR and Central Oregon, KRFC is managed as a 'constraining stock'; that is, the amount of Chinook harvest (all stocks) made available to the troll fishery is contingent on the allowable harvest of KRFC. To estimate average annual Chinook harvest (all stocks) attributable to the availability of Klamath Chinook in each of these areas, average annual Klamath Chinook harvest projected for each area under the no action and action alternatives was divided by an area-specific expansion factor – calculated as the average ratio of annual Chinook harvest (all stocks) to annual Klamath Chinook harvest during 2001-05 (data source: Michael O'Farrell, NMFS). For Monterey and Northern Oregon, Klamath Chinook is not a constraining stock except in years of very low Klamath Chinook abundance. For these latter two areas, the expansion factor was set equal to 1.000 to reflect the fact that Klamath Chinook availability in these areas does not affect the troll fishery's access to other stocks; thus Klamath Chinook harvest is treated as a simple addition to total harvest under the no action and action alternatives.¹⁰

Total Chinook harvest (all stocks) in each area attributable to the availability of Klamath Chinook was converted from numbers of fish to pounds dressed weight, based on the 2001-05 mean weight of troll-caught Chinook south of Cape Falcon (11.9 pounds according to PFMC 2011b).

Total Chinook harvest (all stocks) was converted from pounds to gross revenue, based on the 2004-05 average ex-vessel price of troll Chinook landings south of Cape Falcon (\$3.59 per pound dressed weight according to PFMC 2011b, calculated in 2012 dollars). This average price was calculated based on fishery data for 2004-05 – a period when prices reflect recent consumer preferences and more normal fishery conditions than 2006-10 (Appendix B.1.c).

(vi) The economic value of the fishery was measured in terms of net revenue (gross revenue minus trip expenses). Net revenue was estimated as 81.3 percent of gross ex-vessel revenue – based on survey data indicating that salmon troll trip costs (fuel, food/crew provisions, ice, bait) comprise 18.7 percent of gross revenue (source: Jerry Leonard, NMFS).

⁸ The decades prior to the 2000s were also deemed unsuitable for characterizing the no action alternative. The 1980s pre-date current weak stock management policies. The 1990s was a period of adjustment to constraints that are expected to continue into the future (e.g., consultation standards for ESA-listed stocks, 50-50 tribal/non-tribal allocation) and also includes years of unusually low landings.

⁹ Distribution of troll harvests of KRFC during 2001-05 was as follows: Monterey 4.7 percent, San Francisco 34.4 percent, Fort Bragg 17.9 percent, KMZ-CA 4.3 percent, KMZ-OR 1.9 percent, Central Oregon 27.8 percent, Northern Oregon 9.0 percent.

¹⁰ The expansion factors used in the analysis are as follows: Monterey 1.000, San Francisco 0.058, Fort Bragg 0.065, KMZ-CA 0.199, KMZ-OR 0.107, Central Oregon 0.062, Northern Oregon 1.000.

Harvest projections provided by the EDRRA model do not differentiate between spring and fall Chinook. However, actual harvest opportunities may differ somewhat by fishery – depending on the extent to which the harvestable surplus includes spring Chinook. The Biological Subgroup indicates that the action alternatives will result in expansion and restoration of habitat beneficial to spring Chinook. The Lindley/Davis model anticipates positive conservation benefits in terms of returning spring Chinook to Upper Basin watersheds and enhancing the viability of the Klamath/Trinity Chinook ESU, as well as modest fishery benefits. The Chinook Expert Panel indicates that a ‘substantial increase’ in Chinook between IGD and Keno Dam is possible but is more cautious regarding the possibility of successful Chinook introduction above Keno Dam and benefits to spring Chinook (Section III.B). The Biological Subgroup, Lindley/Davis and Expert Panel results are used here to qualify and expand on the EDRRA results by considering what the availability of modest amounts of spring Chinook in the harvestable surplus might mean for the troll fishery.

ALTERNATIVE 1 – NO ACTION

SONCC COHO

As indicated in Section II, coho retention has been prohibited in the troll fishery south of Cape Falcon since 1993 to meet consultation standards for SONCC coho and three other coho ESUs listed under the ESA. Little improvement in the status of the SONCC coho ESU is expected under Alternative 1. Thus current fishery prohibitions on coho retention are likely to continue into the future under this alternative.

KLAMATH RIVER SPRING AND FALL CHINOOK

Under Alternative 1, annual Klamath Chinook harvest is 35,778 fish and annual Chinook harvest (all stocks) attributable to the availability of Klamath Chinook is 491,100 fish. In all areas except Monterey and Northern Oregon, total Chinook harvest (all stocks) is higher than Klamath Chinook harvest, due to the use of expansion factors to account for total harvest of all stocks associated with the availability of Klamath Chinook. In Monterey and Northern Oregon, Klamath Chinook is not a constraining stock; that is, increases in Klamath Chinook harvest represent a simple addition to total harvest and do not yield benefits in terms of increased access to other stocks.¹¹ Average annual gross and net revenue under Alternative 1 (all areas) are \$21.0 million and \$17.1 million respectively (Table IV-1).

Table IV-1. Projected average annual ocean troll harvest of Klamath Chinook and total Chinook (all stocks) attributable to Klamath Chinook abundance, and associated gross and net revenues under Alternative 1 – by management area.¹

| | # Klamath | # Chinook | Gross Revenue | Net Revenue |
|--|-----------|-----------|---------------|-------------|
|--|-----------|-----------|---------------|-------------|

¹¹ It is important to note that total Chinook harvest (all stocks) and gross revenues reported in Table IV-1 pertain only to harvest and revenues that are attributable to the availability of Klamath Chinook. Because Klamath Chinook is not normally a constraining stock (i.e., does not affect access to other stocks) in Monterey and Northern Oregon, harvest and revenues in those areas attributable to Klamath Chinook (Table IV-1) are much less than actual harvest and revenues during the 2001-05 base period (Tables II-1 and II-3).

| Management Area | Chinook | (All Stocks) | (2012\$) | (2012\$) |
|-----------------|---------|--------------|------------|------------|
| Monterey | 1,671 | 1,671 | 71,367 | 58,021 |
| San Fran | 12,312 | 213,608 | 9,125,553 | 7,419,075 |
| Fort Bragg | 6,413 | 98,382 | 4,202,992 | 3,417,033 |
| KMZ-CA | 1,530 | 7,691 | 328,574 | 267,131 |
| KMZ-OR | 667 | 6,247 | 266,894 | 216,985 |
| Central OR | 9,963 | 160,274 | 6,847,058 | 5,566,658 |
| Northern OR | 3,223 | 3,223 | 137,696 | 111,946 |
| Total | 35,778 | 491,097 | 20,980,134 | 17,056,849 |

¹ Calculations based on methodology discussed in Section IV.A.2.

It is also important to note that troll harvest of Klamath Chinook consists almost exclusively of fall run fish. This stock composition is expected to persist into the future under Alternative 1.

ALTERNATIVE 2 – FULL FACILITIES REMOVAL OF FOUR DAMS

SONCC COHO

Alternative 2 is expected to improve the viability of coho populations in the Klamath stratum of the SONCC coho ESU but is unlikely to lead to de-listing, since the ESU also includes stocks outside the Klamath Basin whose viability is not affected by this action (Section III.A). Thus Alternative 2 will yield little change in coho harvest opportunities. Coho retention will likely continue to be prohibited in the California and Oregon troll fisheries south of Cape Falcon.

KLAMATH RIVER SPRING AND FALL CHINOOK

EFFECTS ON ANNUAL HARVEST AND GROSS AND NET REVENUE

Under Alternative 2, annual average salmon harvest is projected to include 51,082 Klamath Chinook and 701,162 total Chinook (all stocks). In all areas except Monterey and Northern Oregon, total Chinook harvest (all stocks) is higher than Klamath Chinook harvest, due to the use of expansion factors to estimate total harvest of all stocks attributable to the availability of Klamath Chinook in those areas. In Monterey and Northern Oregon, increases in Klamath Chinook harvest represent a simple addition to total harvest and do not yield benefits in terms of increased access to other stocks.¹² Associated gross and net revenues (all areas) are \$30.0 million and \$24.4 million respectively. Average annual net revenue is higher under Alternative 2 (relative to Alternative 1) by \$7.3 million (Table IV-2).

Table IV-2. Projected average annual ocean troll harvest of Klamath Chinook, total Chinook (all stocks) attributable to Klamath Chinook abundance, and gross and net revenues under Alternative 2, and change in net revenue from Alternative 1 – by management area.

| Management Area | # Klamath Chinook ¹ | # Chinook (All Stocks) ¹ | Gross Revenue (2012\$) ¹ | Net Revenue (2012\$) ¹ | Change in Net Revenue ² |
|-----------------|--------------------------------|-------------------------------------|-------------------------------------|-----------------------------------|------------------------------------|
| Monterey | 2,385 | 2,385 | 101,894 | 82,840 | 24,819 |
| San Fran | 17,578 | 304,979 | 13,028,998 | 10,592,576 | 3,173,501 |

¹² It is important to note that total Chinook harvest (all stocks) and gross and net revenues reported in Table IV-2 pertain only to harvest and revenues that are attributable to the availability of Klamath Chinook. Because Klamath Chinook is not normally a constraining stock (i.e., does not affect access to other stocks) in Monterey and Northern Oregon, harvest and revenues attributable to Klamath Chinook in those areas are likely much less than actual total harvest and revenues (all stocks) that would occur under the Klamath Chinook conditions projected for Alternative 2.

| | | | | | |
|-------------|--------|---------|------------|------------|-----------|
| Fort Bragg | 9,156 | 140,465 | 6,000,817 | 4,878,665 | 1,461,632 |
| KMZ-CA | 2,184 | 10,981 | 469,121 | 381,396 | 114,265 |
| KMZ-OR | 952 | 8,920 | 381,058 | 309,800 | 92,815 |
| Central OR | 14,225 | 228,831 | 9,775,879 | 7,947,790 | 2,381,132 |
| Northern OR | 4,602 | 4,602 | 196,595 | 159,831 | 47,885 |
| Total | 51,082 | 701,162 | 29,954,363 | 24,352,897 | 7,296,049 |

¹ Calculations based on methodology described in Section IV.A.2.

² Difference in net revenue between Alternative 2 (column 5 of this table) and Alternative 1 (column 5 of Table IV-1).

To the extent that spring Chinook production increases sufficiently to provide a harvestable surplus, the EDRRA projections (which include but do not distinguish between spring and fall Chinook) may over-estimate troll harvest. The reason for this has to do with the timing of the run relative to the timing of the fishery. Specifically, the troll fishery north of Point Arena, California does not open until April 1; the troll fishery south of Point Arena (which includes the San Francisco and Monterey management areas) does not open until May 1 to meet the consultation standard for ESA-listed Sacramento River winter Chinook (PFMC 2011). Given this season structure, the harvest potential of spring Chinook may be limited for the troll fishery, as a large portion of the spring run will have returned to the river by the time the season opens.

DISCOUNTED PRESENT VALUE OF CHANGE IN NET REVENUE

Figure IV-1 depicts the annual trajectory of net revenues for Alternatives 1 and 2 during 2012-61. These annual values were derived by multiplying average annual net revenue (all areas) associated with each alternative (Tables IV-1 and IV-2 respectively) by an annual adjustment factor that reflects the variation in annual Klamath Chinook harvest relative to mean 2012-61 harvest – as projected by the EDRRA model (Appendix B.2). As indicated in Figure IV-1, the difference between the two alternatives diverges considerably after dam removal.

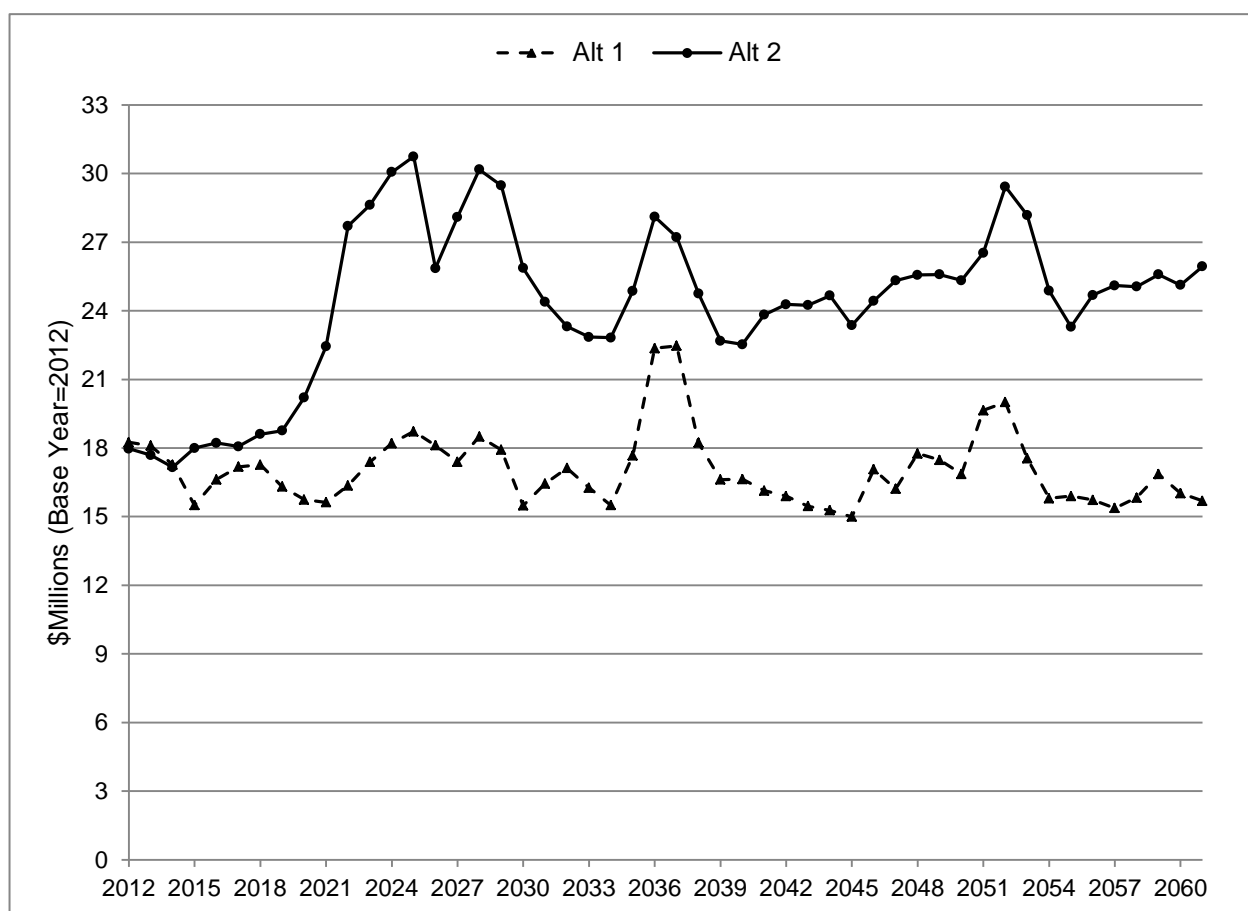


Figure IV-1. Projected annual net revenue under Alternatives 1 and 2 during 2012-61 (calculated according to the methodology described in Appendix B-2).

Results of the NED analysis provided here are also included in two summary reports (Reclamation 2011a, 2011b) that describe all quantifiable economic benefits and costs in terms of discounted present value (DPV). Discounting is based on the premise that benefits that occur more immediately are preferred to benefits that occur farther into the future. Discounting has the effect of attaching progressively smaller weights to changes in net economic value that occur later in the time series, with diminution of these weights becoming more rapid at higher discount rates. The discount rate used in the NED analysis is 4.125 percent, the rate currently prescribed for Federal water resources planning (Reclamation 2010).

DPV for the troll fishery was calculated by applying a discount factor to each of the annual net revenue estimates provided in Figure IV-1, then summing the results (Appendix B-2). Table IV-3 provides estimates of DPV associated with the prescribed 4.125 percent rate and several rates lower and higher than 4.125 percent (including 0.000 percent – no discounting). DPV associated with the 4.125 percent discount rate is \$134.5 million, which is 37 percent of the undiscounted present value (discount rate of 0.000 percent) and twice the value of DPV associated with the 8.000 percent discount rate.

Table IV-3. Discounted present value of the increase in net revenue under Alternative 2 relative to Alternative 1 (2012\$), calculated to illustrate the sensitivity of the estimates to alternative discount rates.

| Discount Rate | Discounted Present Value (2012\$) |
|---------------|-----------------------------------|
| 0.000% | 364,801,854 |
| 2.000% | 216,684,556 |
| 4.125% | 134,494,901 |
| 6.000% | 93,378,408 |
| 8.000% | 66,327,564 |

Calculations based on methodology described in Appendix B.2.

Figure IV-2 depicts the stream of the annual discounted increases in net revenue that were summed to derive the DPV estimate associated with each of the discount rates in Table IV-3. As indicated in the figure, changes in net revenue are relatively insensitive to the choice of discount rate in the first decade of the time series but can diverge rather widely in subsequent decades. The differences in the DPV estimates shown in Table IV-3 are influenced by the fact that changes in net revenue under Alternative 2 do not increase appreciably until after dam removal, which does not occur until close to the end of the first decade of the projection period 2012-61.

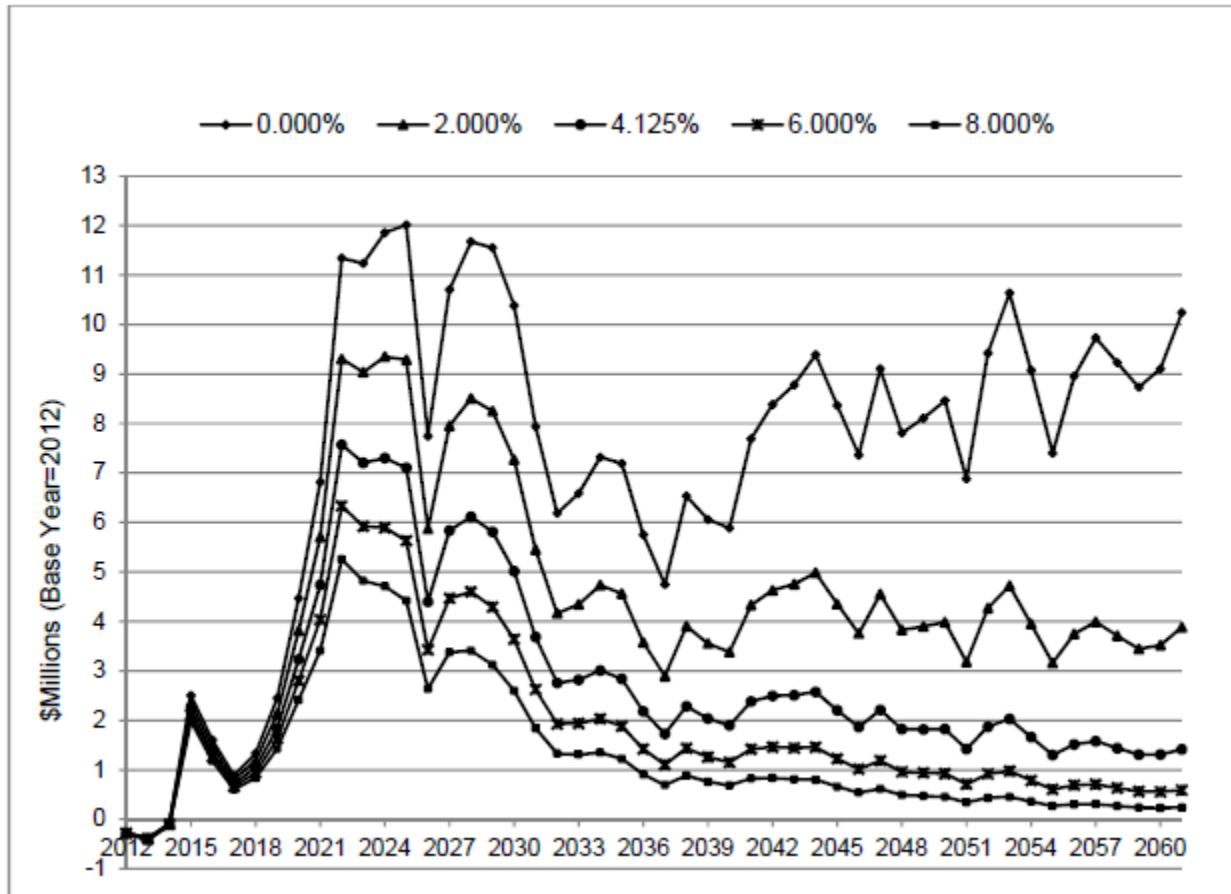


Figure IV-2. Annual discounted values of the increase in net revenue under Alternative 2 relative to Alternative 1 (2012\$) during the projection period 2012-61, calculated on the basis of alternative discount rates of 0% (no discounting), 2%, 4.125%, 6%, and 8%.

EFFECTS AT LOW LEVELS OF ABUNDANCE

Economic effects pertain not only to how harvest opportunity is affected on an average basis but also under more unusual conditions. As indicated in Figure III-1, the KRFC harvest control rule adopted by the PFMC in June 2011 limits the harvest rate to 10 percent or less when pre-harvest escapements fall below 30,500 adult natural spawners. Escapements this low would be accompanied by adverse economic conditions that are reminiscent of the situation in 2006, when actions to protect KRFC required major reductions in harvest of all salmon stocks in all areas south of Cape Falcon (including Monterey and Northern Oregon, where KRFC does not normally constrain harvest of other stocks). Salmon troll landings and revenues were 18 percent and 39 percent respectively of their 2001-05 average values (Tables II-2 and II-3), and \$60.4 million in Commercial Fishery Disaster Assistance was provided to affected businesses and communities. Results of the EDRRA model indicate that pre-harvest escapements below 30,500 would occur in 66 percent fewer years under Alternative 2 than Alternative 1, with the greatest decline (-79 percent) occurring in the post-dam removal years (Table III-1). While the quantitative economic results provided in

Sections IV.C.2.a and IV.C.2.b pertain to how the action alternatives would affect fishery conditions at moderate levels of abundance, it is important to note that Alternative 2 will also reduce the incidence of low abundances and associated adverse effects on the troll fishery.

ALTERNATIVE 3 – PARTIAL FACILITIES REMOVAL OF FOUR DAMS

Alternative 3 is intended to provide the same habitat conditions as Alternative 2 – i.e., fish passage unencumbered by dams and a free-flowing river, as well as benefits of the KBRA. Therefore the effects of this alternative on salmon populations and the salmon troll fishery are expected to be the same as Alternative 2.

V. COMMERCIAL FISHING EXPENDITURES FOR REGIONAL ECONOMIC IMPACT ANALYSIS (RED ACCOUNT)

METHODOLOGY AND ASSUMPTIONS

Regional economic impacts pertain to effects of the no action and action alternatives on employment, labor income and output in the regional economy. These impacts include: direct effects on the economy as trollers spend their revenues on labor shares and payments to support businesses that provide food/crew provisions, fuel, ice, boat maintenance/repair, moorage, and the like; indirect effects as payments by fishery support businesses to their vendors generate additional economic activity; and induced effects associated with changes in household spending by workers in all affected businesses. Estimation of this so-called multiplier effect is based on assumptions such as constant returns to scale, no input substitution, no supply constraints, and no price or wage adjustments. Thus regional impacts as estimated here are more suggestive of the economy's short-term response rather than long-term adjustment to infusions of money into the economy.

Regional impacts were estimated using Impact Analysis for Planning (IMPLAN) software and data and are based on the makeup of the economy at the time of the underlying IMPLAN data (2009). The applicability of the impacts thus estimated to any particular year of the 50 year study period is affected by the extent to which the underlying economy in that year deviates from the economy in 2009. The employment impacts include full time, part time, and temporary positions. These impacts may not be fully realized to the extent that businesses deal with changes in demand by adjusting the workload of existing employees or increasing their use of capital relative to labor rather than hiring new employees.

The regional economic analysis provided here is based on average annual gross revenues projected for the no action and action alternatives. About 99 percent of revenues from Chinook harvest (all stocks) that are attributable to the availability of Klamath Chinook is concentrated in five of the seven management areas under the no action and action alternatives (Tables IV-1 and IV-2). Thus the regional economic analysis focuses on those five areas: San Francisco (San Mateo, San Francisco, Marin and Sonoma Counties), Fort Bragg (Mendocino County), KMZ-CA (Humboldt and Del Norte Counties), KMZ-OR (Curry County), and Central Oregon (Coos, Douglas and Lane Counties). Revenues spent in the region and the multipliers used to estimate the impacts of these expenditures will vary, depending on how the affected region is defined. Thus regional impacts will differ, depending on whether impacts are (i) estimated separately for each of the five areas or (ii) estimated for a single study area defined as the aggregation of all five areas. Because the impacts provided here were estimated in the manner of (i), summing those impacts across areas will not provide an

accurate estimate of the impacts in all areas combined. More detailed documentation of the methods used to estimate regional impacts is provided in Reclamation (2011a).

ALTERNATIVE 1 – NO ACTION

Table V-1 describes average annual gross revenue in each of the five management areas covered by the regional economic analysis. These revenue estimates were used in conjunction with IMPLAN software and data to analyze the regional impacts of Alternative 1 in each area.

Table V-1. Average annual gross revenue under Alternative 1, by management area¹

| Management Area | Gross Revenue (2012\$) |
|-----------------|------------------------|
| San Francisco | 9,125,553 |
| Fort Bragg | 4,202,992 |
| KMZ-CA | 328,574 |
| KMZ-OR | 266,894 |
| Central Oregon | 6,847,058 |

¹ Extracted from Table IV-1.

The associated impacts of Alternative 1 on employment, labor income and output are shown in Table V-2 by management area. Consistent with the revenue pattern (Table V-1), impacts are highest in San Francisco and lowest in KMZ-CA and KMZ-OR.

Table V-2. Annual regional economic impacts associated with average annual gross revenue projected for Alternative 1, by management area

| San Francisco | | | |
|---------------|----------------------|------------------------------|------------------------|
| Impact Type | Employment (Jobs) | Labor Income (\$Millions) | Output (\$Millions) |
| Direct | 480.0 | 4.27 | 9.13 |

| | | | |
|-------------|----------------------|------------------------------|------------------------|
| Indirect | 8.0 | 0.56 | 2.70 |
| Induced | 22.0 | 1.27 | 3.69 |
| Total | 510.0 | 6.10 | 15.52 |
| Fort Bragg | | | |
| Impact Type | Employment (Jobs) | Labor Income (\$Millions) | Output (\$Millions) |
| Direct | 150.0 | 1.98 | 4.20 |
| Indirect | 1.4 | 0.07 | 0.18 |
| Induced | 10.6 | 0.40 | 1.24 |
| Total | 162.0 | 2.45 | 5.62 |
| KMZ-CA | | | |
| Impact Type | Employment (Jobs) | Labor Income (\$Millions) | Output (\$Millions) |
| Direct | 43.0 | 0.15 | 0.33 |
| Indirect | 0.1 | 0.01 | 0.02 |
| Induced | 0.9 | 0.03 | 0.10 |
| Total | 44.0 | 0.19 | 0.45 |
| KMZ-OR | | | |
| | Employment | Labor Income | Output |

| Impact Type | (Jobs) | (\$Millions) | (\$Millions) |
|----------------|----------------------|------------------------------|------------------------|
| Direct | 25.0 | 0.13 | 0.27 |
| Indirect | 0.1 | 0.00 | 0.01 |
| Induced | 0.5 | 0.02 | 0.05 |
| Total | 25.6 | 0.15 | 0.33 |
| Central Oregon | | | |
| Impact Type | Employment (Jobs) | Labor Income (\$Millions) | Output (\$Millions) |
| Direct | 293.0 | 3.21 | 6.85 |
| Indirect | 4.1 | 0.17 | 0.46 |
| Induced | 21.8 | 0.77 | 2.24 |
| Total | 318.9 | 4.15 | 9.55 |

Source: Reclamation 2011b, presented in 2012 dollars.

Employment measured in number of jobs. Labor income is dollar value of total payroll (including benefits) for each industry in the analysis area plus income received by self-employed individuals in the analysis area. Output represents dollar value of industry production.

ALTERNATIVE 2 – FULL FACILITIES REMOVAL OF FOUR DAMS

Table V-3 describes average annual gross revenue in each of the five management areas covered by the regional economic analysis. The changes in gross revenue from Alternative 1 to Alternative 2 was used in conjunction with IMPLAN software and data to estimate the regional impacts associated with Alternative 2.

Table V-3. Average annual gross revenue under Alternative 2 and change from Alternative 1 – by management area.

| | | |
|------------|--|--|
| Management | | |
|------------|--|--|

| Area | Gross Revenue (2012\$) ¹ | Change from Alternative 1 ² |
|----------------|-------------------------------------|--|
| San Francisco | 13,028,998 | 3,903,445 |
| Fort Bragg | 6,000,817 | 1,797,825 |
| KMZ-CA | 469,121 | 140,547 |
| KMZ-OR | 381,058 | 114,164 |
| Central Oregon | 9,775,879 | 2,928,821 |

¹ Extracted from Table IV-3.

² Difference in gross revenue between Alternative 2 (column 2 of this table) and Alternative 1 (Table V-1).

The impacts of the increase in troller revenues under Alternative 2 on employment, labor income and output are shown in Table V-4 for each management area. The increases in employment, labor income and output relative to Alternative 1 are 42 to 43 percent in each area.

Table V-4. Annual regional economic impacts associated with projected average annual increase in ex-vessel revenue under Alternative 2 relative to Alternative 1, by management area.

| San Francisco | | | | | | |
|---------------|------------|---------------------|--------------|---------------------|------------|---------------------|
| Impact Type | Employment | | Labor Income | | Output | |
| | Jobs | % change from Alt 1 | \$Millions | % change from Alt 1 | \$Millions | % change from Alt 1 |
| Direct | 205.0 | | 1.79 | | 3.90 | |
| Indirect | 3.5 | | 0.24 | | 1.15 | |
| Induced | 9.3 | | 0.53 | | 1.55 | |
| Total | 217.8 | 42.7 | 2.56 | 42.0 | 6.6 | 42.6 |

| | | | | | | |
|-------------|------------|---------------------|--------------|---------------------|------------|---------------------|
| Fort Bragg | | | | | | |
| Impact Type | Employment | | Labor Income | | Output | |
| | Jobs | % change from Alt 1 | \$Millions | % change from Alt 1 | \$Millions | % change from Alt 1 |
| Direct | 64.0 | | 0.85 | | 1.80 | |
| Indirect | 0.5 | | 0.03 | | 0.08 | |
| Induced | 4.5 | | 0.17 | | 0.53 | |
| Total | 69.0 | 42.7 | 1.05 | 42.8 | 2.41 | 42.8 |
| KMZ-CA | | | | | | |
| Impact Type | Employment | | Labor Income | | Output | |
| | Jobs | % change from Alt 1 | \$Millions | % change from Alt 1 | \$Millions | % change from Alt 1 |
| Direct | 18.0 | | 0.06 | | 0.14 | |
| Indirect | 0.1 | | 0.00 | | 0.01 | |
| Induced | 0.4 | | 0.01 | | 0.04 | |
| Total | 18.5 | 41.7 | 0.07 | 42.0 | 0.19 | 42.6 |
| KMZ-OR | | | | | | |
| Impact Type | Employment | | Labor Income | | Output | |
| | | % change from Alt 1 | | % change from Alt 1 | | % change from Alt 1 |

| | Jobs | | \$Millions | | \$Millions | |
|----------------|------------|---------------------|--------------|---------------------|------------|---------------------|
| Direct | 11.0 | | 0.05 | | 0.11 | |
| Indirect | 0.0 | | 0.00 | | 0.00 | |
| Induced | 0.2 | | 0.01 | | 0.02 | |
| Total | 11.2 | 43.8 | 0.06 | 42.8 | 0.13 | 42.8 |
| Central Oregon | | | | | | |
| Impact Type | Employment | | Labor Income | | Output | |
| | Jobs | % change from Alt 1 | \$Millions | % change from Alt 1 | \$Millions | % change from Alt 1 |
| Direct | 125.0 | | 1.35 | | 2.93 | |
| Indirect | 1.8 | | 0.07 | | 0.20 | |
| Induced | 9.1 | | 0.32 | | 0.94 | |
| Total | 135.9 | 42.6 | 1.74 | 42.0 | 4.07 | 42.6 |

Source: Reclamation 2011b, presented in 2012 dollars.

Employment measured in number of jobs. Labor income is dollar value of total payroll (including benefits) for each industry in the analysis area plus income received by self-employed individuals in the analysis area. Output represents dollar value of industry production.

ALTERNATIVE 3 – PARTIAL FACILITIES REMOVAL OF FOUR DAMS

Alternative 3 is intended to provide the same habitat conditions as Alternative 2 – i.e., fish passage unencumbered by dams and a free-flowing river, as well as benefits of the KBRA. Therefore the effects of this alternative on salmon populations and the salmon troll fishery are expected to be the same as Alternative 2.

VI. SUMMARY AND CONCLUSIONS

The particular salmon stocks influenced by the no action and action alternatives are the SONCC coho ESU (which is listed under the ESA) and Klamath River fall and spring Chinook. Economic effects of the no action and action alternatives on the troll fishery as they relate to these stocks are as follows:

SONCC COHO ESU

Coho retention has been prohibited in the troll fishery south of Cape Falcon since 1993 to meet consultation standards for SONCC coho and three other coho ESUs listed under the ESA. Little improvement in the status of the SONCC coho ESU is expected under the no action alternative. Thus current fishery prohibitions on coho retention are likely to continue into the future under this alternative. The action alternatives are expected to yield similar improvements in the viability of Klamath coho populations and advance the recovery of the SONCC coho ESU, but are unlikely to lead to de-listing since the ESU also includes stocks outside the Klamath Basin whose viability is not affected by this action. Thus coho retention will likely continue to be prohibited in the California and Oregon troll fisheries south of Cape Falcon under these alternatives.

KLAMATH RIVER CHINOOK

Economic benefits: Under the no action alternative, average annual troll harvest of Klamath Chinook is estimated to be similar to what it was during 2001-05 (35,778 fish). Reflecting the constraining influence of Klamath Chinook on the availability of Chinook (all stocks) in the San Francisco, Fort Bragg, KMZ-CA, KMZ-OR and Central Oregon management areas, Klamath Chinook harvest of 35,778 provides the opportunity for the troll fishery to harvest 491,100 Chinook (all stocks) south of Cape Falcon, Oregon. Average annual net revenue associated with such harvest is \$17.1 million.

Under the action alternatives, annual salmon troll harvest is estimated to increase by an average of 43 percent over the 2012-61 projection period. Average annual harvest under these alternatives is projected to include 51,082 Klamath Chinook and 701,162 total Chinook (all stocks), with associated net revenue of \$24.4 million. The increase in annual net revenue under the action alternatives relative to no action is \$7.3 million. The discounted present value of this increase over the 2012-61 period is \$134.5 million (based on a discount rate of 4.125 percent).

The harvest control rule underlying the Klamath Chinook harvest projections limits the harvest rate to 10 percent or less in years when pre-harvest escapements fall below 30,500 adult natural spawners. Escapements this low would likely be accompanied by major regulatory restrictions and adverse economic conditions similar to what was experienced in 2006. Such low escapements would occur in 66 percent fewer

years under the action alternatives, with the greatest decline (-79 percent) occurring in the post-dam removal years.

Economic impacts: Regional economic impacts associated with the no action and action alternatives are largely concentrated in the five management areas where Klamath Chinook is the constraining stock. Regional impacts associated with the \$20.8 million in gross revenue generated in those five areas under the no action alternative vary widely by area. For San Francisco, Fort Bragg and Central Oregon, annual impacts (depending on the area) include 162 to 510 jobs, \$2.45 million to \$6.10 million in labor income, and \$5.62 million to \$15.52 million in output. For KMZ-CA and KMZ-OR, annual impacts include 26 to 44 jobs, \$0.15 million to \$0.19 million in labor income, and \$0.33 million to \$0.45 million in output.

The additional \$8.9 million in gross revenue in the same five areas under the action alternatives generates regional impacts that vary widely by area. For San Francisco, Fort Bragg and Central Oregon, annual impacts (depending on the area) include an additional 69 to 218 jobs, an additional \$1.05 million to \$2.56 million in labor income, and an additional \$2.41 million to \$6.6 million in output. For KMZ-CA and KMZ-OR, the annual impacts include an additional 11 to 19 jobs, an additional \$0.06 million to \$0.07 million in labor income, and an additional \$0.13 million to \$0.19 million in output.

Main areas of uncertainty in this analysis include natural variability in biological and environmental parameters, uncertainty regarding future harvest management policies, and uncertain ex-vessel prices (which are affected by global supply and demand for farmed as well as wild salmon).

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APPENDIX A. SALMON FISHERY MANAGEMENT

In 1976 the U.S. Congress implemented the Magnuson Fishery Conservation and Management Act (now the Magnuson-Stevens Fishery Conservation and Management Act or MSFCMA), which established eight regional fishery management councils whose mandate was to phase out foreign fishing and manage domestic fisheries in the U.S. Exclusive Economic Zone (EEZ).¹³ The Pacific Fishery Management Council (PFMC) is the entity responsible for management of EEZ fisheries off the coasts of Washington, Oregon and California. The PFMC implemented the Pacific Coast Salmon Fishery Management Plan (FMP) in 1978. The FMP addresses management needs of multiple salmon stocks that originate in rivers along the Pacific coast. The PFMC and its member states manage the troll fishery south of Cape Falcon with regulations such as area closures, season closures, gear restrictions, minimum size limits, vessel landing limits, stock retention prohibitions, and mark-selective fishing.¹⁴

Salmon stocks that originate in rivers south of Cape Falcon, Oregon generally limit their ocean migration to the area south of Falcon. The major salmon species harvested in the south-of-Falcon fishery are Chinook (*Oncorhynchus tshawytscha*) and coho (*O. kisutch*). The area south of Falcon is divided into six management areas: Monterey, San Francisco, Fort Bragg, Klamath Management Zone (KMZ), Central Oregon, and Northern Oregon. For purposes of this analysis, the KMZ (which straddles the Oregon-California border) is divided at the border into two areas: KMZ-OR and KMZ-CA.

Management of the troll fishery is complicated by the fact that multiple salmon stocks with different conservation objectives mix in the ocean harvest. These 'mixed stock' fisheries are managed on the general principle of 'weak stock' management, whereby harvest opportunity for more abundant stocks is constrained by the need to meet conservation objectives for weaker stocks.

PFMC management reflects conservation objectives for targeted stocks, consultation standards for weak stocks, and harvest allocation requirements (PFMC 2011):

Targeted stocks: For ocean fisheries south of Cape Falcon, the major targeted stocks are Sacramento River fall Chinook (SRFC) and Klamath River fall Chinook (KRFC). Conservation objectives for these stocks¹⁵ are as follows:

¹³ The EEZ includes waters that extend 3-200 miles from the U.S. coast.

¹⁴ A mark selective fishery is a fishery in which hatchery fish are marked in a visually identifiable manner (e.g., by clipping the adipose fin), thereby allowing fishermen to selectively retain marked fish and release unmarked (wild) fish.

¹⁵ The conservation objectives for KRFC and SRFC discussed here are intended to facilitate interpretation of historical fishery trends. In June 2011 the PFMC recommended modifications to these objectives to address new requirements of the MSFCMA; these changes will likely become effective in 2012.

In 1989, following a period of sizeable KRFC harvests, low KRFC escapements and a major El Niño in 1982-83, the PFMC adopted more conservative harvest policies for KRFC, including a return of 34-35 percent of adult natural spawners and an escapement floor of 35,000 adult natural spawners (Klamath River Technical Team 1986, PFMC 1988). Figure A-1 depicts KRFC escapements during 1978-2010 relative to the escapement floor that was in effect during 1989-2006. In 2007 the floor was increased to 40,700 to help rebuild KRFC after the stock collapsed in 2006.

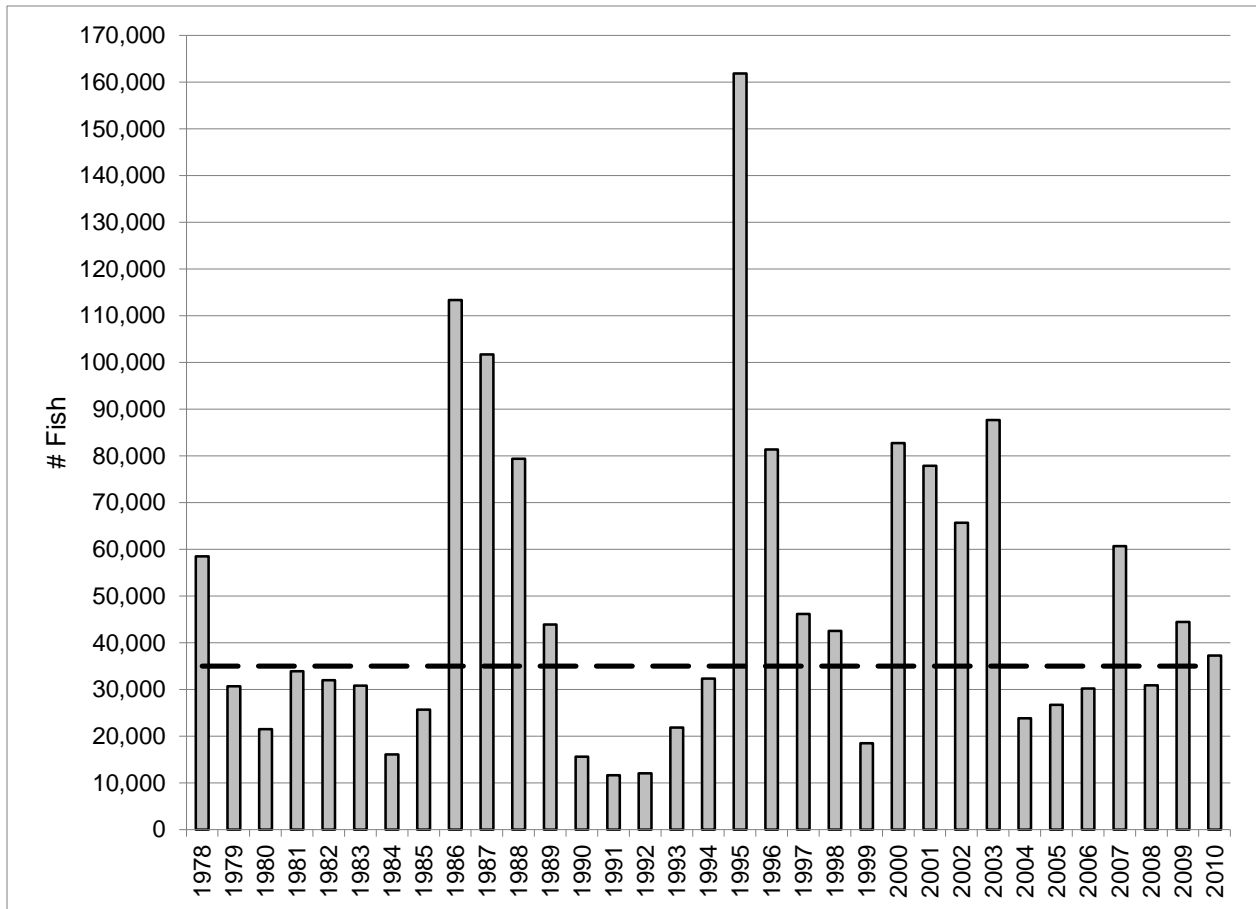


Figure A-1. Klamath River adult natural spawner escapement, 1978-2010. Dotted line represents 35,000 escapement floor in effect during 1989-2006 (source: PFMC 2011a)

The conservation objective for SRFC is a spawner escapement goal of 122,000-180,000 hatchery and natural area adults. Figure II-2 depicts SRFC escapements during 1978-2010 relative to the escapement goal, which has been in effect since 1978.

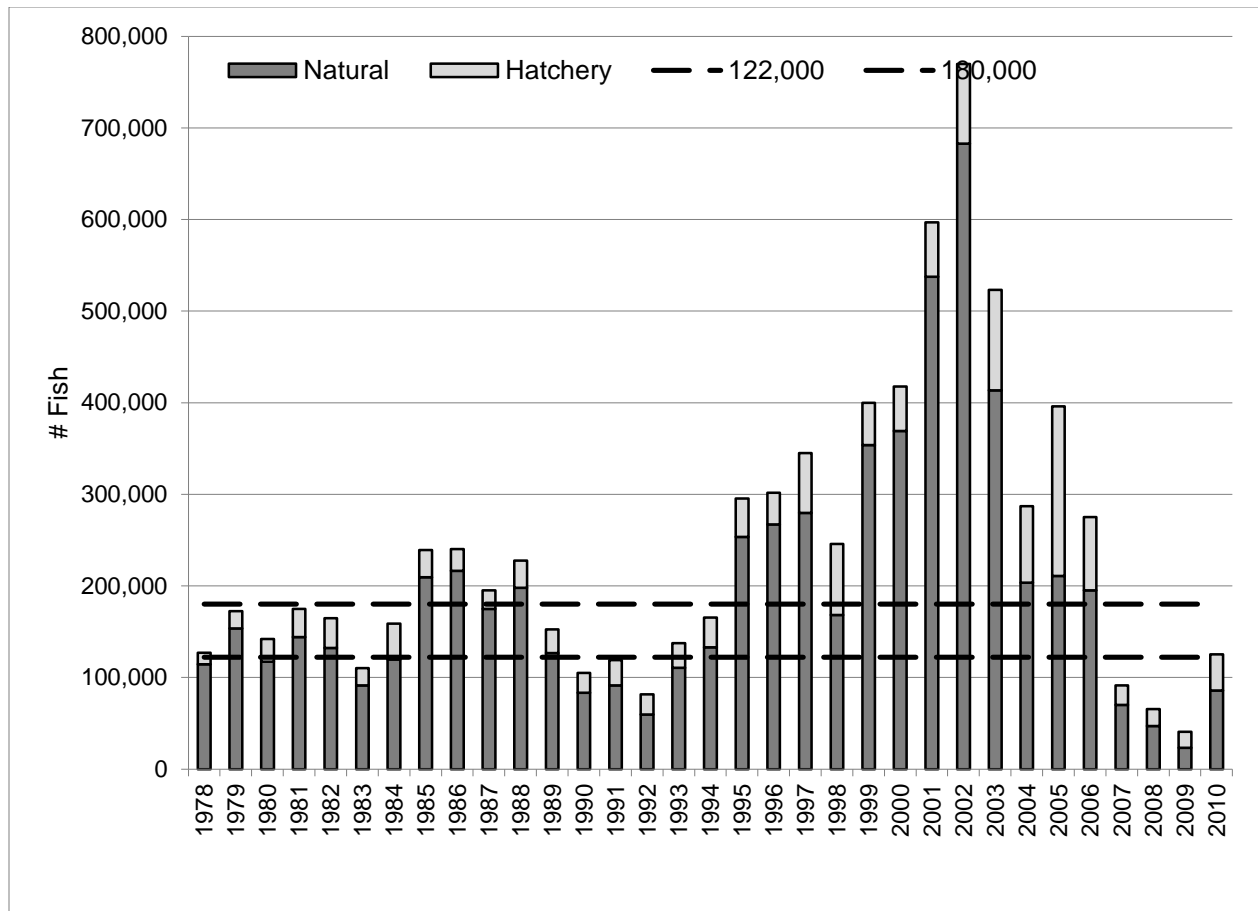


Figure A-2. Sacramento River adult spawner escapement (natural + hatchery), 1978-2010. Dotted lines represent PFMC escapement goal of 122,000-180,000 (source: PFMC 2011a).

Stocks listed under the Endangered Species Act (ESA): The PFMC is bound by consultation standards for six ESA-listed Chinook and coho stocks that occur in the ocean fishery south of Cape Falcon.¹⁶

Sacramento River winter Chinook was listed as ‘threatened’ in 1989 and reclassified as ‘endangered’ in 1994. The current consultation standard includes area, season and size limit restrictions for ocean commercial and recreational fisheries from Point Arena, California to the U.S./Mexico border.

Central California Coast coho was listed as ‘threatened’ in 1996 and reclassified as ‘endangered’ in 2005. The consultation standard is a ban on coho retention in all commercial and recreational fisheries in California.

¹⁶ A seventh stock – Central Valley spring Chinook – was listed as ‘threatened’ in 1999. NMFS determined that PFMC-managed fisheries presented ‘no jeopardy’ to this stock.

SONCC coho was listed as ‘threatened’ in 1997. The consultation standard caps the marine exploitation rate on Rogue/Klamath River hatchery coho at 13 percent.

Oregon Coastal Natural (OCN) coho was listed as ‘threatened’ in 1998, de-listed in 2006 following a NMFS update of all its listing determinations, and re-listed in 2008 after the de-listing was successfully challenged in Court. OCN coho is managed on the basis of exploitation rates that vary with habitat production potential (freshwater and marine) – measured by parent spawner status and smolt-to-adult marine survival (PFMC 1999, OCN Work Group 2000).

California Coastal Chinook was listed as ‘threatened’ in 1999. Using KRFC as an indicator stock, the consultation standard for California Coastal Chinook caps the forecast harvest rate for age-4 KRFC in the ocean fishery at 16 percent.

Lower Columbia Natural coho was listed as ‘threatened’ in 2005. The consultation standard is a maximum exploitation rate of 15 percent (marine and Columbia River combined).

Stock rebuilding: The PFMC designates a ‘conservation alert’ when a stock fails to meet its conservation objective in a single year and a ‘conservation concern’ when this happens in three consecutive years. A conservation alert may warrant precautionary management in the year of the alert, while a conservation concern (which is more indicative of a downward trend) may require a longer-term management strategy – including a stock rebuilding plan (PFMC 2003).

Allocation: In 1993, the Department of the Interior, Office of the Solicitor issued an opinion requiring that 50 percent of Klamath-Trinity River salmon be reserved for the Yurok and Hoopa Valley Tribes (USDOI 1993). This was considerably higher than the 30 percent tribal reserve that was in effect during 1987-91 (Pierce 1998) and required reduced allocations to non-tribal fisheries. The 50-50 tribal/non-tribal allocation remains in effect today.

Table A-1 identifies periods of particularly stringent troll regulations associated with low coho and/or Chinook abundances. The table illustrates the long-term nature of non-retention policies to protect coho and the frequency of fishery closures, which tend to occur when Chinook abundance is also low.

Table A-1. Years of no coho retention (NoCoho), closure of both Chinook and coho fisheries (Closure), and closure of Crescent City portion of KMZ-CA (ClosureCC)¹ in the troll fishery south of Cape Falcon, 1990-2010, by management area.

| Year | Management Area | | | | |
|---------|--------------------|----------|-------------------|---------|----------------------|
| | SanFran & Monterey | Ft Bragg | KMZ-CA | KMZ-OR | CentralOR & North OR |
| 1990 | | | NoCoho | NoCoho | |
| 1991 | | | NoCoho, ClosureCC | NoCoho | |
| 1992 | | Closure | Closure | Closure | |
| 1993 | NoCoho | NoCoho | Closure | Closure | NoCoho |
| 1994 | NoCoho | NoCoho | Closure | NoCoho | NoCoho |
| 1995 | NoCoho | NoCoho | Closure | NoCoho | NoCoho |
| 1996 | NoCoho | NoCoho | NoCoho | NoCoho | NoCoho |
| 1997-98 | NoCoho | NoCoho | NoCoho, ClosureCC | NoCoho | NoCoho |
| 1999-05 | NoCoho | NoCoho | NoCoho | NoCoho | NoCoho |
| 2006 | NoCoho | NoCoho | Closure | NoCoho | NoCoho, |
| 2007 | NoCoho | NoCoho | NoCoho | NoCoho | |
| 2008 | Closure | Closure | Closure | NoCoho | NoCoho |
| 2009 | Closure | Closure | Closure | Closure | |
| 2010 | NoCoho | NoCoho | Closure | NoCoho | NoCoho |

Sources: PFMC 1998, 2009, 2010, 2011b.

¹ KMZ-CA includes Crescent City and Eureka-area ports.

Circumstances underlying the regulatory restrictions identified in Table A-1 are as follows:

Periods of drought and El Niño conditions during 1991-92 and 1997-98 contributed to low Chinook and coho returns and prompted major fishery restrictions during the 1990s – including Commercial Fishery Disaster Assistance in 1994 (\$15.7 million), 1995 (\$13.0 million) and 1998 (\$3.5 million) (pers. comm. Stephen Freese, NMFS). Actions taken by the PFMC to deal with the persistent decline in coho stocks included a ban on coho retention in KMZ-CA and KMZ-OR since 1990 and in all other management areas south of Cape Falcon since 1993, with the exception of limited fisheries in 2007 and 2009 in Central and Northern Oregon.

Fishery closure (all stocks) generally occurs when conservation concerns for SRFC and/or KRFC occur in conjunction with the prohibition on coho retention. During 1990-92, KRFC and SRFC failed to reach their respective conservation objectives – triggering a conservation concern for both stocks (Klamath River Fall Chinook Review Team 1994, Sacramento River Fall Chinook Review Team 1994). Major fishery restrictions including closures in Fort Bragg in 1992, KMZ-CA during 1992-95, and KMZ-OR during 1992-93.

During the prolonged drought in the 2000s, KRFC failed to achieve its conservation objective for three consecutive years (2004-06). Subsequent fishery restrictions – including closure of KMZ-CA in 2006 – prompted \$60.4 million in Commercial Fishery Disaster Assistance in 2007 (Upton 2010). The PFMC also increased the adult natural spawner escapement floor from 35,000 to 40,700 as a rebuilding strategy.

Failure of SRFC to achieve its conservation objective during 2007-09 triggered a conservation concern (Lindley *et al.* 2009). Historically unprecedented restrictions were imposed on the troll fishery (including complete closure of the California fishery in 2008-09. Congress appropriated \$170 million in Commercial Fishery Disaster Assistance, of which \$117 million was disbursed in 2008 and \$53 million in 2009 (Upton 2010; pers. comm. Stephen Freese, NMFS).

It is important to note that KRFC natural spawner escapement – as depicted in Figure A-1 – does not necessarily reflect stock abundance. Ocean abundance pertains to the number of fish that migrate to the ocean and (i) are harvested in ocean or inriver fisheries, (ii) contribute to natural or hatchery escapement, (iii) remain unharvested in the ocean, or (iv) are subject to natural mortality or non-retention (hooking and dropoff) mortality.¹⁷ Figure A-3 provides an index of KRFC abundance that includes the escapement and harvest components of abundance (unharvested migrants and natural and non-retention mortality being more difficult to estimate).¹⁸ The size of the escapement and harvest components of Figure A-3 depends on

¹⁷ Natural mortality is the mortality associated with factors such as disease and non-human predation. Hooking mortality pertains to fish that die after being hooked and released. Dropoff mortality pertains to fish that die after being dropped from the fishing gear as a result of such encounters with the gear.

¹⁸ The escapements depicted in Figures A-1 and A-3 are not comparable. Figure A-1 includes natural escapement only, while Figure A-3 includes both natural and hatchery escapement.

factors such as the extent of hatchery production, how much of the ocean abundance is made available for harvest, and how the available harvest is distributed among fishery sectors (ocean and inriver).

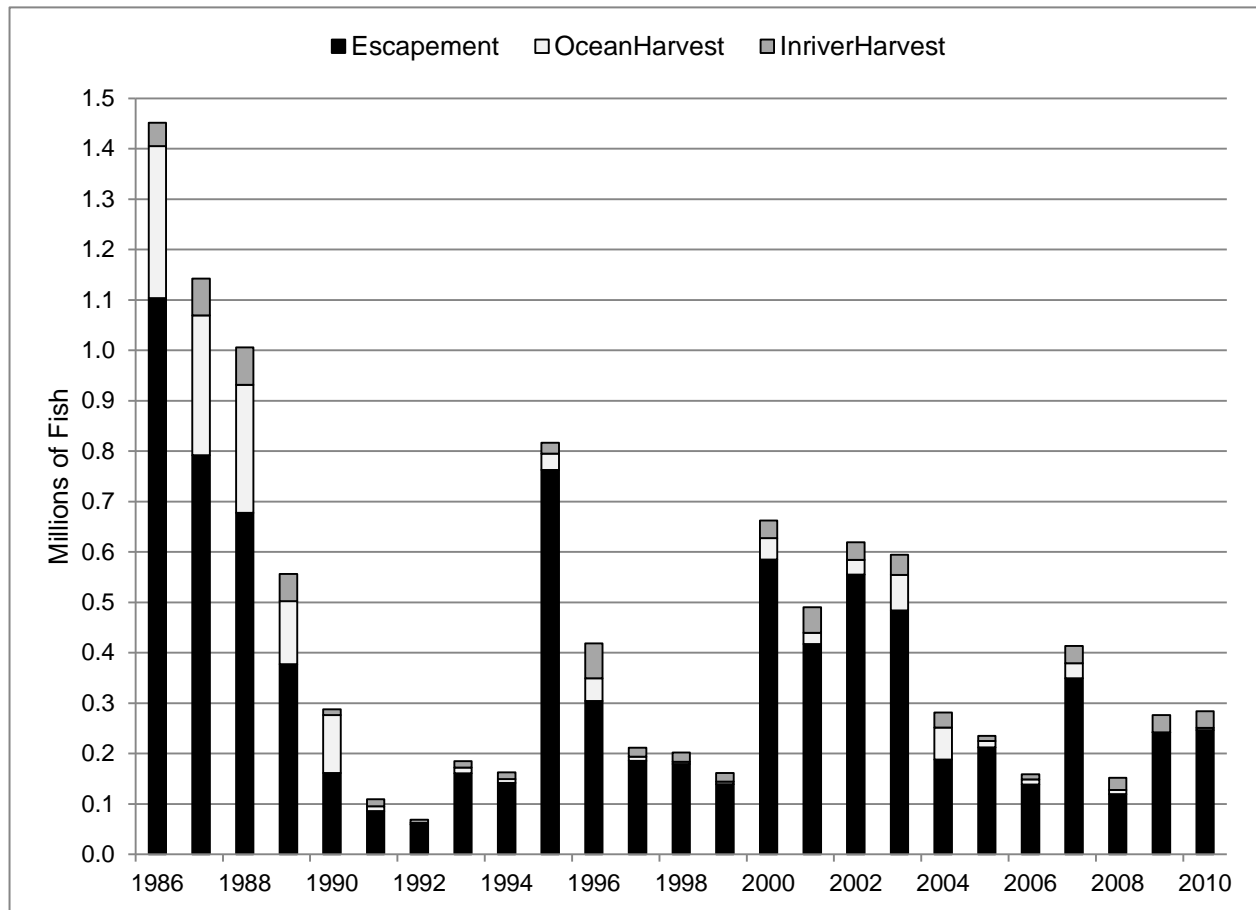


Figure A-3. Klamath River fall Chinook ocean abundance index (millions of fish), 1986-2010 (source: PFMC 2011a).

As with KRFC, SRFC adult spawner escapement – as depicted in Figure A-2 – is not necessarily indicative of stock abundance. Figure A-4 provides an index of ocean abundance for SRFC that includes the two major components of abundance (escapement and harvest).¹⁹ The pattern of abundance differs considerably from the escapement pattern.

¹⁹ The escapement portion of Figure A-4 is comparable to escapement as depicted in Figure A-2, as both figures include both natural and hatchery escapement.

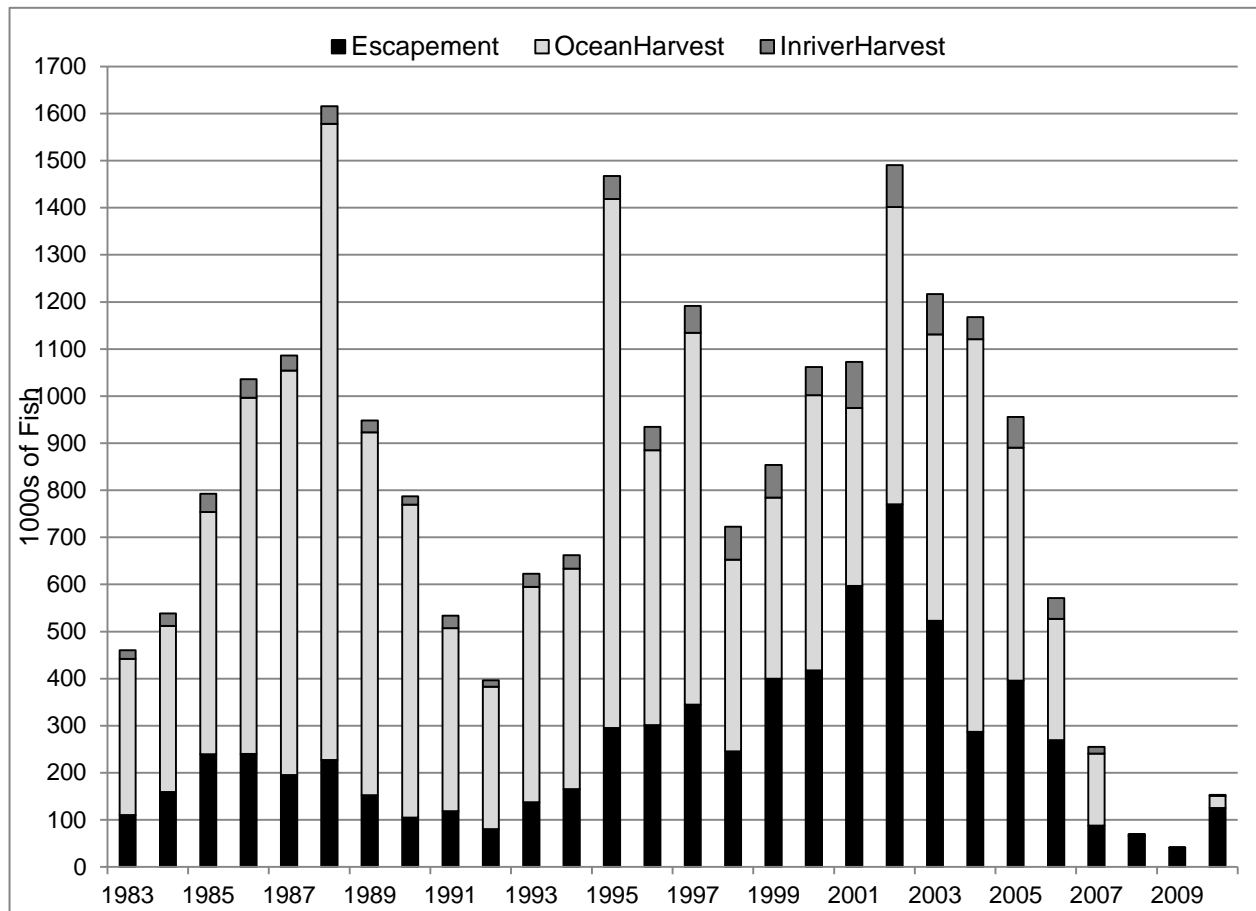


Figure A-4. Sacramento River fall Chinook ocean abundance index (1000s of fish), 1983-2010 (source: PFMC 2011a).

Escapement as a proportion of the SRFC abundance index increased from an annual average of 21 percent during 1981-95 to 40 percent during 1996-2007 to 91 percent during 2008-10 – reflecting the effect of more conservative harvest policies over time (Figure A-4). The 91 percent estimate reflects the effects of stringent fishery regulations associated with record low stock conditions during 2008-10. It is not clear whether the record low SRFC abundances experienced in recent years signal a future pattern of persistently low abundances, are part of a cyclical pattern, or are events that may recur on a rare or occasional basis.

APPENDIX B. METHODOLOGIES USED TO QUANTIFY ECONOMIC EFFECTS OF NO ACTION AND ACTION ALTERNATIVES

This appendix provides documentation of how EDRRA model projections were used in combination with fishery data to quantify the economic effects of the no action and action alternatives on the troll fishery.

ESTIMATION OF ANNUAL HARVEST AND GROSS AND NET REVENUE

Table B-1 describes the equations used to estimate Klamath Chinook harvest, total Chinook harvest (all stocks), and gross and net revenues under the no action and action alternatives. The net revenue estimates are inputs in the Net Economic Development (NED) analysis (Section IV); the gross revenues are inputs in the Regional Economic Development (RED) analysis (Section V). Numeric values of the parameters that appear in Table B-1 (α_i , EXPAND_i, LBFISH, PRICE, PCTREV) are provided in Table B-2. Derivation of the variable PCTHARV (row #1 of Table B-1) is discussed in Appendix B.1.b. Derivation of the variable PRICE (row #5 of Table B-1) is discussed in Appendix B.1.c.

EQUATIONS AND PARAMETER VALUES

Table B-1. Equations used to project average annual troll harvest of Klamath Chinook and total Chinook and associated gross and net revenues, by management area *i* and year *t* (2012-61), under no action alternative (NAA) and dam removal alternative (DRA).

| # | No-action alternative (NAA/Alternative 1) | Dam removal alternative (DRA/Alts 2 and 3) |
|--|---|---|
| 1 | $KLAMCHNK^{NAA} = KLAMCHNK_{mean(01-05)}$ | $KLAMCHNK^{DRA} = KLAMCHNK^{NAA} \times PCTHARV$ |
| 2 | $KLAMCHNK_i^{NAA} = \alpha_i \times KLAMCHNK^{NAA}$ | $KLAMCHNK_i^{DRA} = \alpha_i \times KLAMCHNK^{DRA}$ |
| 3 | $TOTCHNK_i^{NAA} = KLAMCHNK_i^{NAA} / EXPAND_i$ | $TOTCHNK_i^{DRA} = KLAMCHNK_i^{DRA} / EXPAND_i$ |
| 4 | $TOTCHNKLB_i^{NAA} = TOTCHNK_i^{NAA} \times LBFISH$ | $TOTCHNKLB_i^{DRA} = TOTCHNK_i^{DRA} \times LBFISH$ |
| 5 | $GROSSREV_i^{NAA} = TOTCHNKLB_i^{NAA} \times PRICE$ | $GROSSREV_i^{DRA} = TOTCHNKLB_i^{DRA} \times PRICE$ |
| 6 | $NETREV_i^{NAA} = GROSSREV_i^{NAA} \times PCTREV$ | $NETREV_i^{DRA} = GROSSREV_i^{DRA} \times PCTREV$ |
| Note: Variables with subscripts NAA and DRA pertain to outputs of the economic analysis. Variables with asterisked versions of these superscripts (NAA* and DRA*) pertain to outputs of the EDRRA model. | | |

$KLAMCHNK^{NAA}$ = average annual troll harvest of Klamath River Chinook under NAA (# fish, all areas).

$KLAMCHNK_{mean(01-05)}$ = average troll harvest of Klamath River Chinook during 2001-05 (# fish, all areas).

$KLAMCHNK^{DRA}$ = average annual troll harvest of Klamath River Chinook under DRA (# fish, all areas).

$PCTHARV$ = percent increase in Klamath Chinook harvest under DRA, as projected by EDRRA model (see Appendix B.1.b).

$KLAMCHNK_i^{NAA}$ = annual harvest of Klamath River Chinook (# fish) in area i under NAA.

$KLAMCHNK_i^{DRA}$ = annual harvest of Klamath River Chinook (# fish) in area i under DRA.

α_i = proportion of troll-caught Klamath River Chinook harvest occurring in area i under NAA and DRA (see Table B-2)

$TOTCHNK_i^{NAA}$ = annual Chinook harvest (# fish, all stocks) in area i under NAA

$TOTCHNK_i^{DRA}$ = annual Chinook harvest (# fish, all stocks) in area i under DRA

$EXPAND_i$ = expansion factor used to project Chinook harvest (all stocks) associated with access to Klamath Chinook in each area i under NAA AND DRA (see Table B-2)

$TOTCHNKLB_i^{NAA}$ = annual Chinook harvest (# pounds dressed weight, all stocks) in area i under NAA

$TOTCHNKLB_i^{DRA}$ = annual Chinook harvest (# pounds dressed weight, all stocks) in area i under DRA

$LBFISH$ = average pounds dressed weight per Chinook (see Table B-2)

$GROSSREV_i^{NAA}$ = annual gross ex-vessel revenue (all stocks, 2012\$) in area i under NAA

$GROSSREV_i^{DRA}$ = annual gross ex-vessel revenue (all stocks, 2012\$) in area i under DRA

$PRICE$ = ex-vessel price per pound dressed weight (2012\$) (see Table B-2)

$NETREV_i^{NAA}$ = annual net revenue (all stocks, 2012\$) in area i under NAA

$NETREV_i^{DRA}$ = annual net revenue (all stocks, 2012\$) in area i under DRA

$PCTREV$ = net revenue as percent of gross revenue (see Table B-2)

Table B-2. Parameter values used to estimate Klamath Chinook and total Chinook harvest (all stocks), and gross and net revenue by management area under the no-action and action alternatives.

| Parameter | Management Area | | | | | | |
|--|-----------------|---------|---------|--------|--------|-----------|------------|
| | Monterey | SanFran | FtBragg | KMZ-CA | KMZ-OR | CentralOR | NorthernOR |
| α_i | 0.047 | 0.344 | 0.179 | 0.043 | 0.019 | 0.278 | 0.090 |
| EXPAND _i | 1.000 | 0.058 | 0.065 | 0.199 | 0.107 | 0.062 | 1.000 |
| LBFISH | 11.9 | 11.9 | 11.9 | 11.9 | 11.9 | 11.9 | 11.9 |
| PRICE | 3.59 | 3.59 | 3.59 | 3.59 | 3.59 | 3.59 | 3.59 |
| PCTREV | 0.813 | 0.813 | 0.813 | 0.813 | 0.813 | 0.813 | 0.813 |
| <p>α_i = proportion of Klamath River Chinook harvested by troll fishery in management area I, estimated using 2001-05 fishery data (data source: Michael O'Farrell, NMFS).</p> <p>EXPAND_i = ratio of total Chinook harvest (all stocks) to Klamath Chinook harvest in management area i, estimated using 2001-05 fishery data (data source: Michael O'Farrell, NMFS).</p> <p>LBFISH = mean weight (pounds dressed weight) per troll-caught Chinook south of Cape Falcon during 2001-05 (data source: PFMC 2011b).</p> <p>PRICE = mean ex-vessel price per pound dressed weight of troll-caught Chinook south of Cape Falcon, estimated using 2004-05 fishery data (data source: PFMC 2011b).</p> <p>PCTREV = estimated percent of gross salmon troll revenue remaining after payment of trip expenses (source: Jerry Leonard, NMFS)</p> | | | | | | | |

DERIVATION OF PCTHARV

The percent increase in Klamath Chinook harvest between the NAA and DRA projected by the EDRRA model (PCTHARV) was estimated by Hendrix (2011) as follows:

$$PCTHARV = 1/T \sum_{t=1, \dots, T} \{ \text{Median}_{t,j=1, \dots, 1000} [(KLAMCHNK_{t,j}^{DRA*} - KLAMCHNK_{t,j}^{NAA*}) /$$

$$KLAMCHNK_{t,j}^{NAA*}] \}$$

[B1]

where

$KLAMCHNK_{t,j}^{NAA*}$ = troll harvest of Klamath Chinook projected for year t and iteration j under the NAA by the EDRRA model;

$KLAMCHNK_{t,j}^{DRA*}$ = troll harvest of Klamath Chinook projected for year t and iteration j under the DRA by the EDRRA model;

the term in $[\]$ is the percent difference between DRA harvest and NAA harvest projected by the EDRRA model for each iteration $j=1,...,1000$ and year $t=1,...,T$;

$Median_{t,j=1,...,1000} [\]$ is the median of the 1000 values of $[\]$ generated for year t ;

$1/T \sum_{t=1,...,T} \{Median_{t,j=1,...,1000} [\]\}$ is the mean of the median values of $[\]$, calculated over the years $t=1,...,T$.

DERIVATION OF PRICE

Over the past three decades, ex-vessel salmon prices have been heavily influenced by national and international market conditions. The relatively low prices of farmed salmon and the rapid increase in farmed salmon imports since the 1980s (Figure B-1) resulted in declining prices for both west coast and Alaska salmon (Figure B-2). The reversal of this trend, which began in 2002, is attributed to a number of factors, including increasing prices of farmed salmon compounded by growing consumer differentiation between wild and farmed salmon.

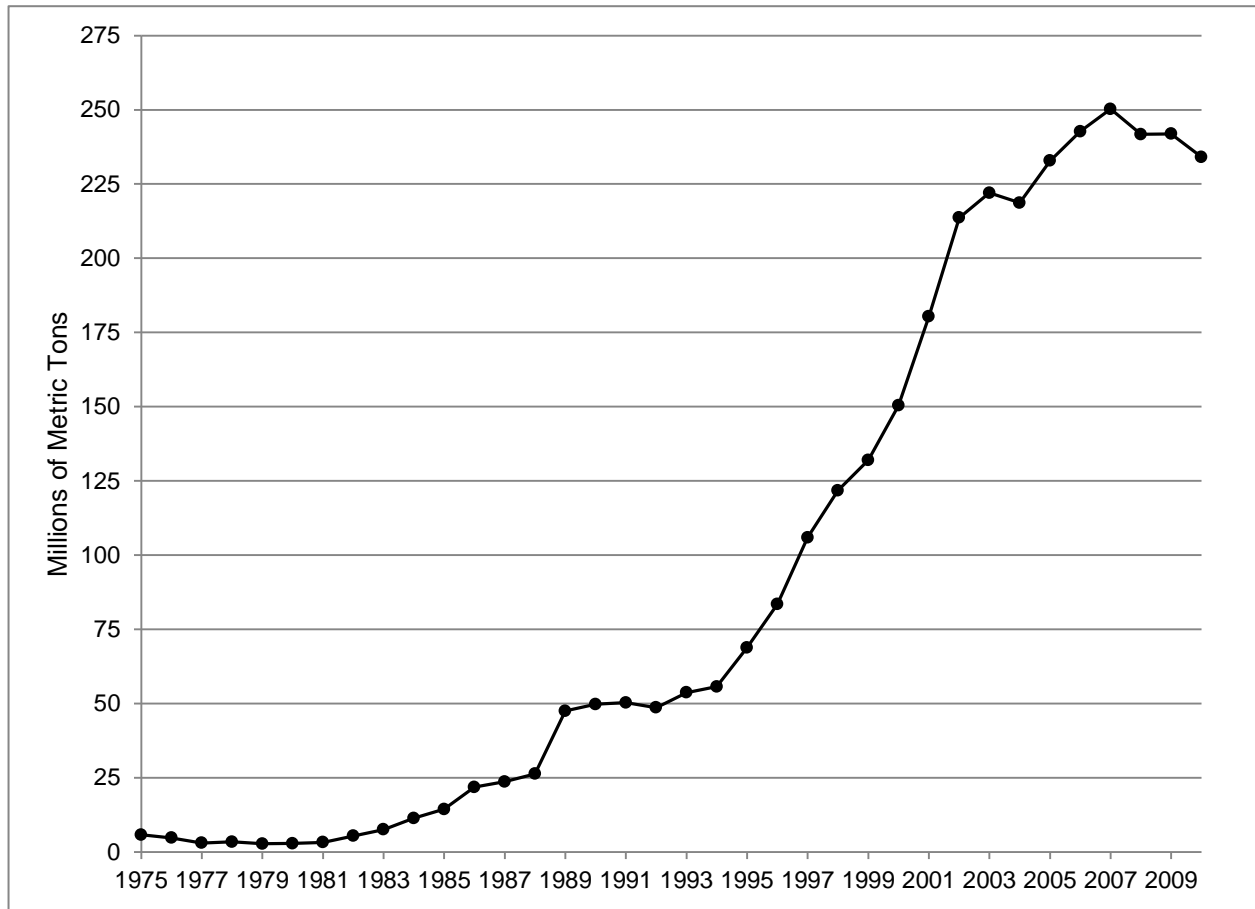


Figure B-1. Imports of edible salmon products into the U.S., 1975-2010 (source: NOAA National Marine Fisheries Service, Office of Science and Technology, Silver Spring, MD).

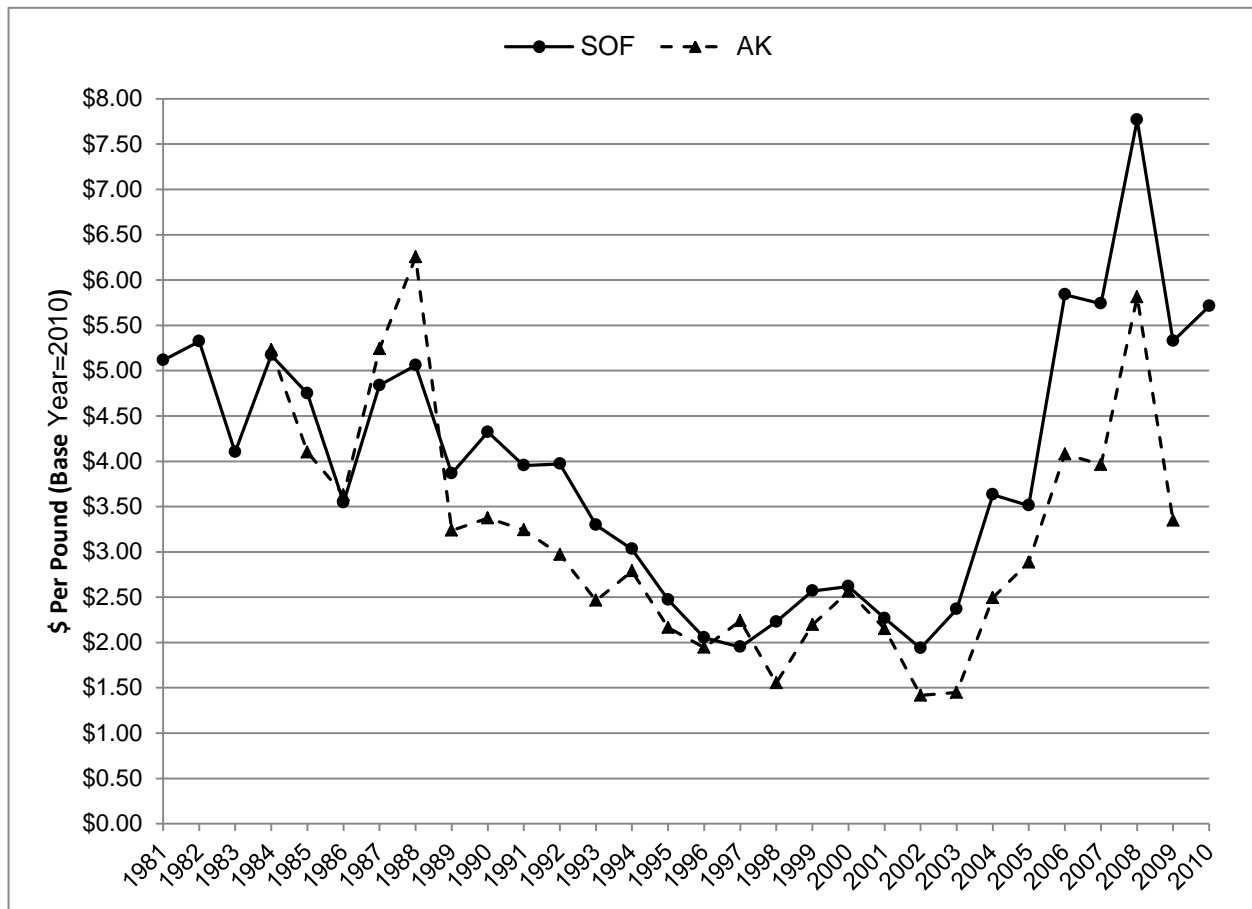


Figure B-2. Ex-vessel prices of troll-caught Chinook in California and Oregon south of Cape Falcon during 1981-2010 and in Southeast Alaska during 1984-2009 (2012\$) (sources: PFMC 1998, 2011b; ADFG 2009).²⁰

The record high prices during 2006-10 coincided with years of record low landings on the west coast (Figure B-3), suggesting that the precipitous landings decline in those years was sufficiently large to have its own influence on prices. PRICE (the ex-vessel price of troll-caught Chinook south of Cape Falcon, Oregon) was calculated based on fishery data for 2004-05 – a period where prices reflect recent consumer preferences and more moderate fishery conditions than 2006-10.

²⁰ To help ensure comparability with prices of troll-caught Chinook south of Cape Falcon, Oregon, Alaska prices pertain to Chinook harvested in Southeast Alaska, where a large majority of the commercial Chinook harvest is caught with troll gear (85 percent in 2010, according to Skannes *et al.* 2011).

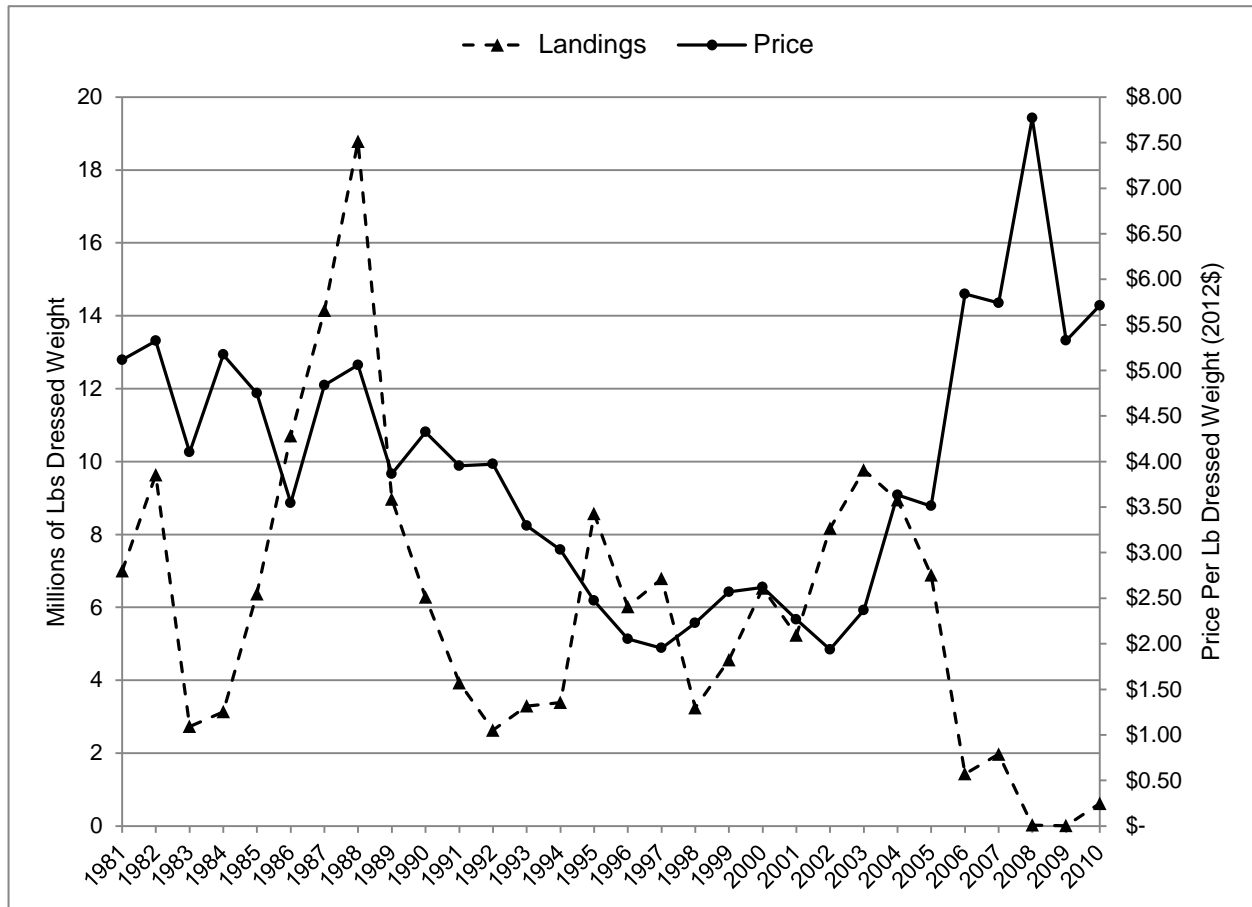


Figure B-3. Annual landings (pounds dressed weight) and ex-vessel price (2012\$) of troll-caught Chinook south of Cape Falcon, Oregon, 1981-2010 (sources: PFM 1990, 1991, 1998, 2001, 2011b).

ESTIMATION OF DISCOUNTED PRESENT VALUE OF NET REVENUE

The NED analysis (Section IV) involved estimation of the discounted present value of net revenues; this requires that a discount factor be applied to net revenue in each year of the 50-year projection period. In order to estimate net revenue for each year t , average annual net revenue (all areas) projected for Alternative 1 (Table IV-1) was multiplied by a factor that reflects the interannual variation in Klamath Chinook harvest relative to mean harvest – as projected by the EDRRA model under the NAA. This factor is applicable to net revenues as well as harvest, due to the proportional relationship between harvest and net revenues. Specifically:

$$\text{NETREV}_{t\text{Alt}1} = \text{NETREVA}_{t1} \times \text{KLAMCHNK}_{t\text{NAA}}^* / \text{KLAMCHNK}_{\text{mean}(12-61)\text{NAA}}^* \quad [\text{B2}]$$

where

NETREVA_{t1} = average annual net revenue (all areas) under Alternative 1 (\$17.1 million, according to Table IV-1), and

$KLAMCHNK_{tNAA}^* / KLAMCHNK_{mean(12-61)NAA}^*$ = the ratio of Klamath Chinook harvest in each year t to annual Klamath Chinook harvest averaged over the projection period $t=2012,...,2061$, as projected by the EDRRA model for the NAA.

Annual net revenue for each year t under Alternative 2 ($NETREV_{tAlt2}$) was similarly calculated, as follows:

$$NETREV_{tAlt2} = NETREV_{Alt2} \times KLAMCHNK_{tDRA}^* / KLAMCHNK_{mean(12-61)DRA}^* \quad [B3]$$

where

$NETREV_{Alt2}$ = average annual net revenue (all areas) under Alternative 2 (\$24.4 million, according to Table IV-2), and

$KLAMCHNK_{tDRA}^* / KLAMCHNK_{mean(12-61)DRA}^*$ = the ratio of Klamath Chinook harvest in each year t to annual Klamath Chinook harvest averaged over the projection period $t=2012,...,2061$, as projected by the EDRRA model for the DRA.

The discounted present value (DPV) of future increases in net revenue under Alternative 2 relative to Alternative 1 was estimated as follows:

$$DPV = \sum_{t=2012,...,2061} [(NETREV_{tAlt2} - NETREV_{tAlt1})] (1+r)^{-t} \quad [B4]$$

where

$NETREV_{tAlt1}$ and $NETREV_{tAlt2}$ = net revenue projection in year t for Alternatives 1 and 2 respectively, calculated on the basis of equations [B2] and [B3] above; and

r = discount rate.

ESTIMATION OF PERCENT OF YEARS WHEN DRA HARVEST > NAA HARVEST

The percent of years in which DRA harvest exceeds NAA harvest (PCTYRS) was estimated from EDRRA model outputs as follows:

$$PCTYRS = 1/T \sum_{t=1,...,T} \{ (1/1000) \text{COUNT}_{t,j=1,...,1000} [KLAMCHNK_{tj}^{DRA*} > KLAMCHNK_{tj}^{NAA*}] \} \quad [B5]$$

where

$KLAMCHNK_{tj}^{NAA*}$ = troll harvest of Klamath Chinook projected by EDRRA model for year t and iteration j under the NAA;

$KLAMCHNK_{tj}^{DRA*}$ = troll harvest of Klamath Chinook projected by EDRRA model for year t and iteration j under the DRA;

$\{ (1/1000) \text{COUNT}_{t,j=1,...,1000} [] \}$ = percent of iterations $j=1,...,1000$ when DRA harvest > NAA harvest, estimated separately for each year t . $[]$ is shorthand for what appears in brackets in equation [B5];

$1/T \sum_{t=1,...,T} \{ (1/1000) \text{COUNT}_{t,j=1,...,1000} [] \}$ = mean of $\{ (1/1000) \text{COUNT}_{t,j=1,...,1000} [] \}$ over years $t=1,...,T$.

ESTIMATION OF PERCENT DIFFERENCE IN FREQUENCY OF PRE-HARVEST ESCAPEMENT $\leq 30,500$

The percent difference between the NAA and DRA in the frequency of pre-harvest adult natural spawner escapements $\leq 30,500$ (PCTDIFF) was estimated from EDRRA model outputs as follows:

$$\begin{aligned} \text{PCTDIFF} = & 1/T \sum_{t=1, \dots, T} \{ [\text{COUNT}_{t,j=1, \dots, 1000}^{\text{DRA}^*} (\text{ESCAPE}_{t,j}^{\text{DRA}^*} \leq 30,500) \\ & - \text{COUNT}_{t,j=1, \dots, 1000}^{\text{NAA}^*} (\text{ESCAPE}_{t,j}^{\text{NAA}^*} \leq 30,500)] / \\ & \text{COUNT}_{t,j=1, \dots, 1000}^{\text{NAA}^*} (\text{ESCAPE}_{t,j}^{\text{NAA}^*} < 30,500) \} \end{aligned} \quad [\text{B6}]$$

where

$\text{ESCAPE}_{t,j}^{\text{NAA}^*}$ = pre-harvest escapement of Klamath Chinook projected by the EDRRA model for year $t=1, \dots, T$ and iteration $j=1, \dots, 1000$ under the NAA;

$\text{ESCAPE}_{t,j}^{\text{DRA}^*}$ = pre-harvest escapement of Klamath Chinook projected by the EDRRA model for year $t=1, \dots, T$ and iteration $j=1, \dots, 1000$ under the DRA;

$\text{COUNT}_{t,j=1, \dots, 1000}^{\text{NAA}^*} (\text{ESCAPE}_{t,j}^{\text{NAA}^*} \leq 30,500)$ = number of iterations j in year t when $\text{ESCAPE}_{t,j}^{\text{NAA}^*} \leq 30,500$ under the NAA;

$\text{COUNT}_{t,j=1, \dots, 1000}^{\text{DRA}^*} (\text{ESCAPE}_{t,j}^{\text{DRA}^*} \leq 30,500)$ = number of iterations j in year t when $\text{ESCAPE}_{t,j}^{\text{DRA}^*} \leq 30,500$ under the DRA;

$[\text{COUNT}_{t,j=1, \dots, 1000}^{\text{DRA}^*} () - \text{COUNT}_{t,j=1, \dots, 1000}^{\text{NAA}^*} ()] / \text{COUNT}_{t,j=1, \dots, 1000}^{\text{NAA}^*} ()$ = percent difference between DRA and NAA in number of iterations when pre-harvest adult natural spawner escapement $\leq 30,500$, estimated separately for each year t . () is shorthand for what appears in parentheses in equation [B6];

$$1/T \sum_{t=1, \dots, T} \{ [\text{COUNT}_{t,j=1, \dots, 1000}^{\text{DRA}^*} () - \text{COUNT}_{t,j=1, \dots, 1000}^{\text{NAA}^*} ()] / \text{COUNT}_{t,j=1, \dots, 1000}^{\text{NAA}^*} () \}$$

= mean of percent differences over years $t=1, \dots, T$.