MANAGEMENT TESTING AND SCENARIOS IN THE CALIFORNIA CURRENT

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*Does not include appendices

CCIEA SCENARIOS CONCEPTUAL FRAMEWORK

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.

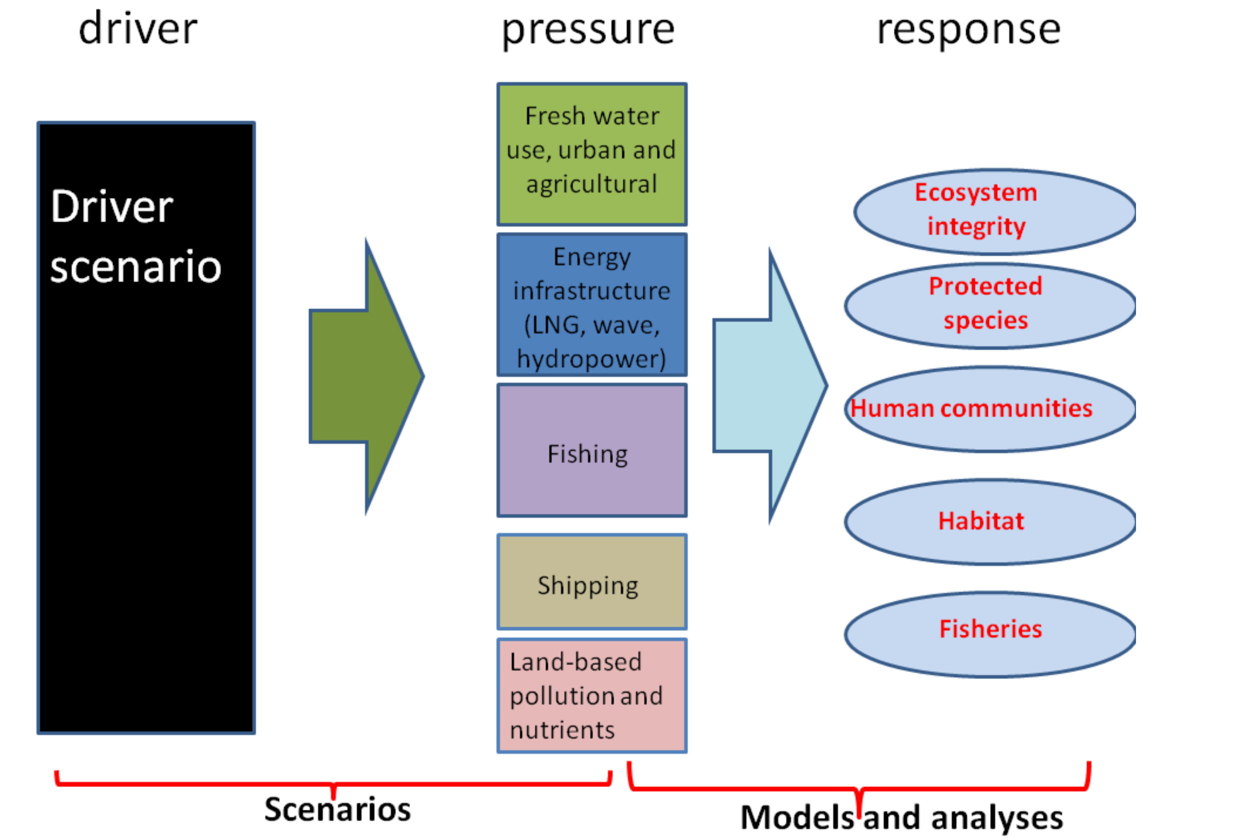


Figure MS1. Schematic of Management Testing approach, where drivers are linked to pressures via narrative scenarios, and then quantitative models link pressures to responses.

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.

	Pressure				
Scenario	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

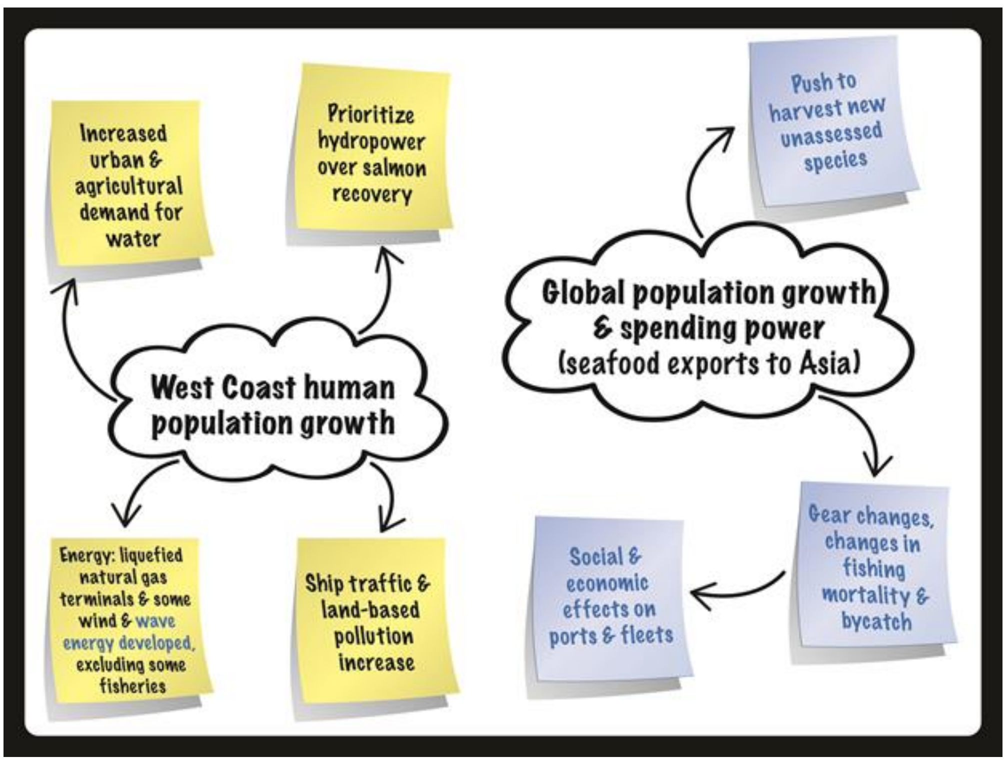


Figure MS2. Results from preliminary engagement with managers and experts (Engagement section), related to the Population Growth scenario. Blue topics are addressed with quantitative models in this IEA.

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

CLIMATE CHANGE SCENARIO

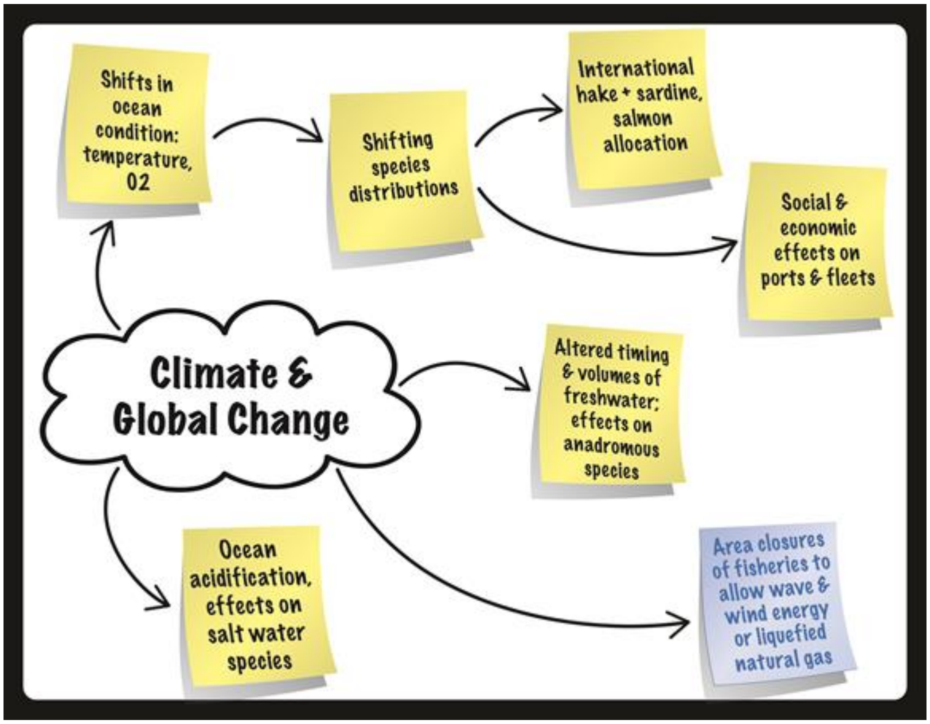


Figure MS3. Results from preliminary engagement with managers and experts (Engagement section), related to the Climate and Global Change scenario. Blue topics are addressed with quantitative models in this IEA.

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 CO_2 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 CO^2 emissions: www.bit.ly/zdh95M

CONSERVATION DEMANDS SCENARIO

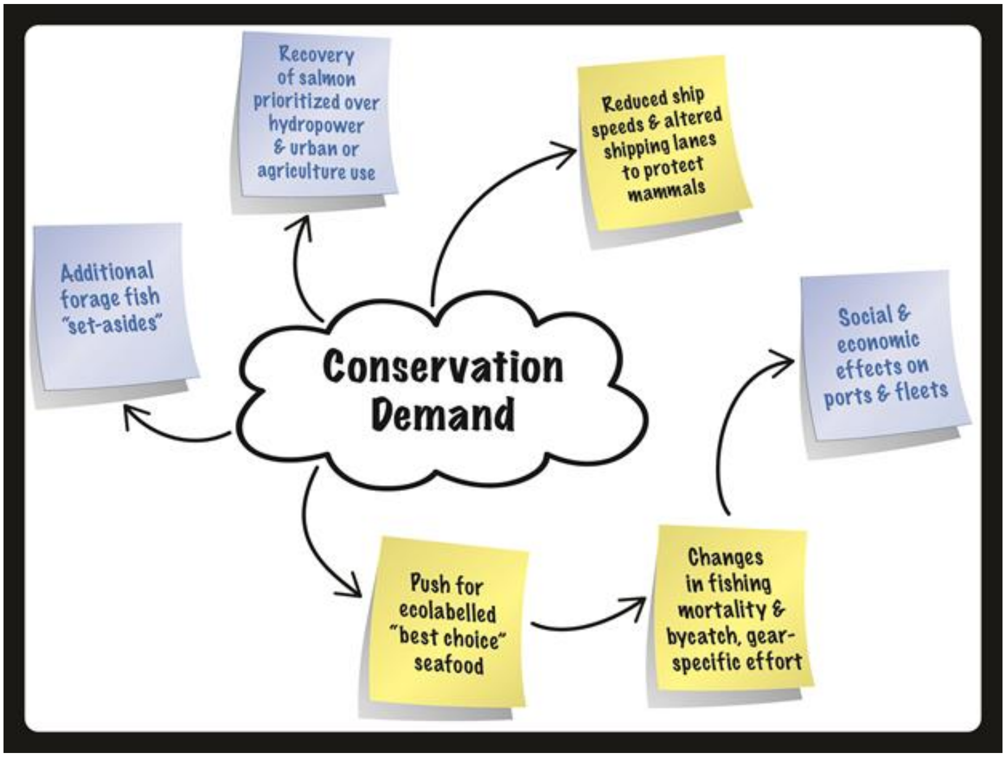


Figure MS4. Results from preliminary engagement with managers and experts (Engagement section), related to the Conservation Demand scenario. Blue topics are addressed with quantitative models in this IEA.

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.

ENERGY CRUNCH SCENARIO

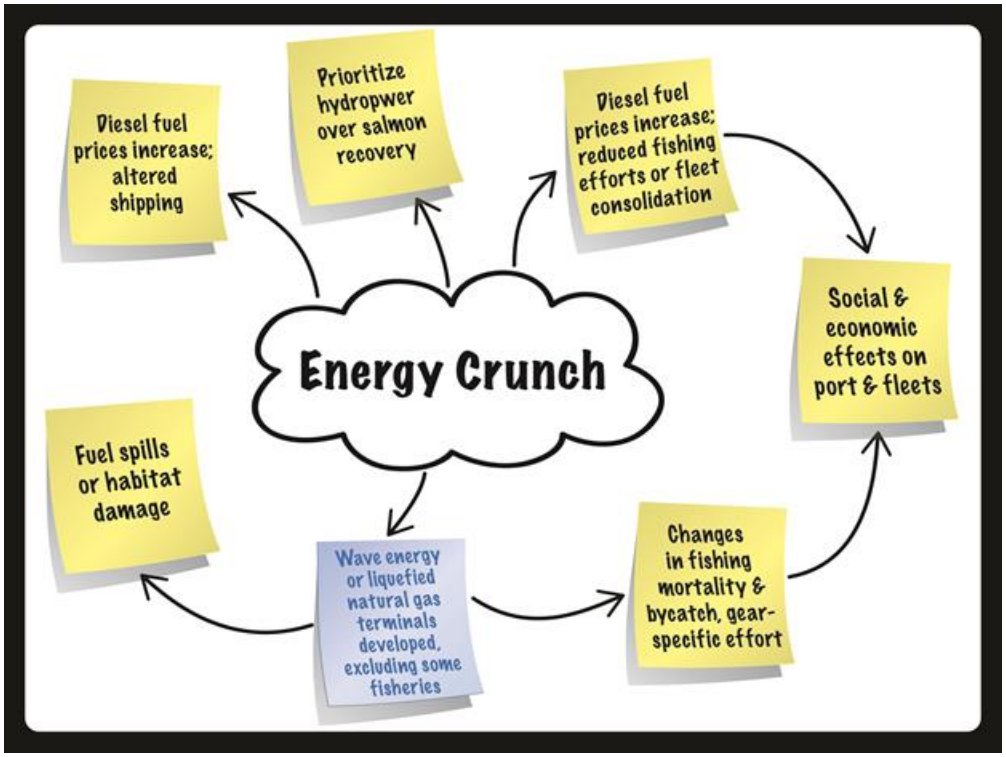


Figure MS5. Results from preliminary engagement with managers and experts (Engagement section), related to the Energy Crunch scenario. Blue topics are addressed with quantitative models in this IEA.

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

STATUS QUO

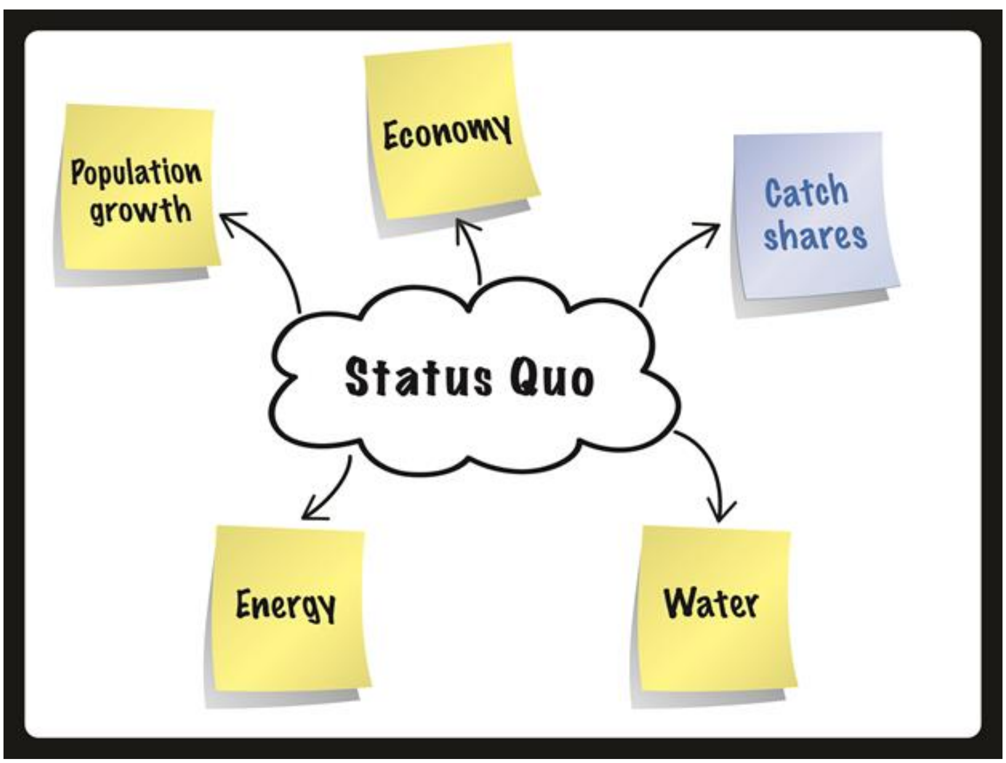


Figure MS6. Results from preliminary engagement with managers and experts (Engagement section), related to the Status Quo scenario. Blue topics are addressed with quantitative models in this IEA.

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 Tahkentich substations 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.

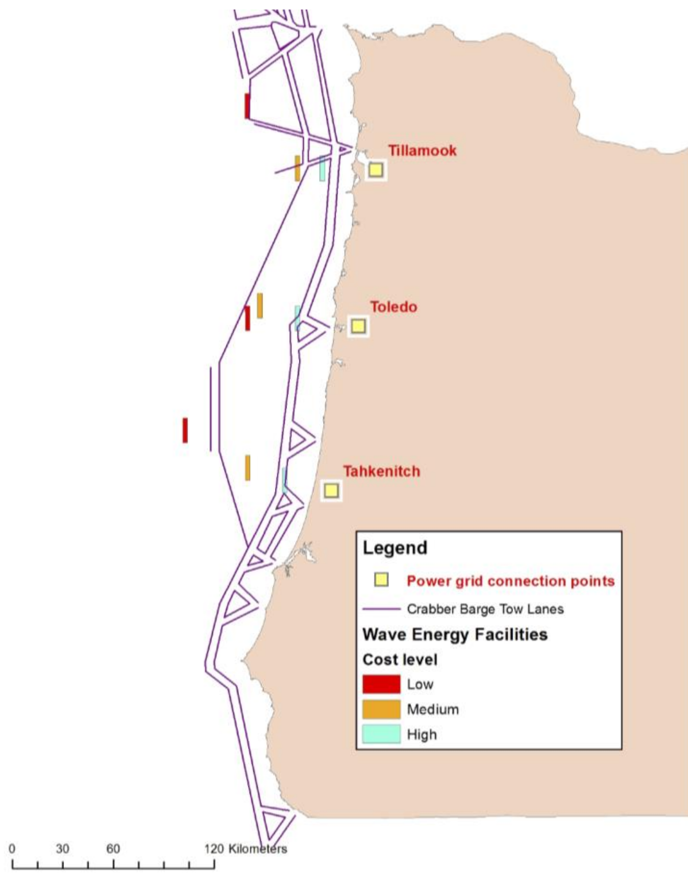


Figure MS7. Sites for potential wave energy facilities, power grid connection points, and barge tow lanes.

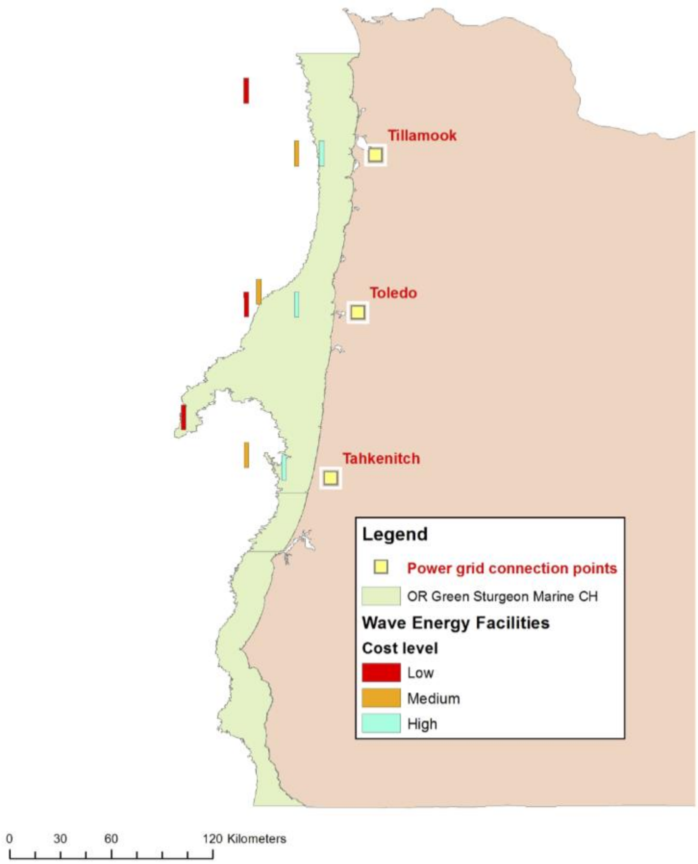


Figure MS8. Sites for potential wave energy facilities, power grid connection points, and green sturgeon critical habitat.

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.

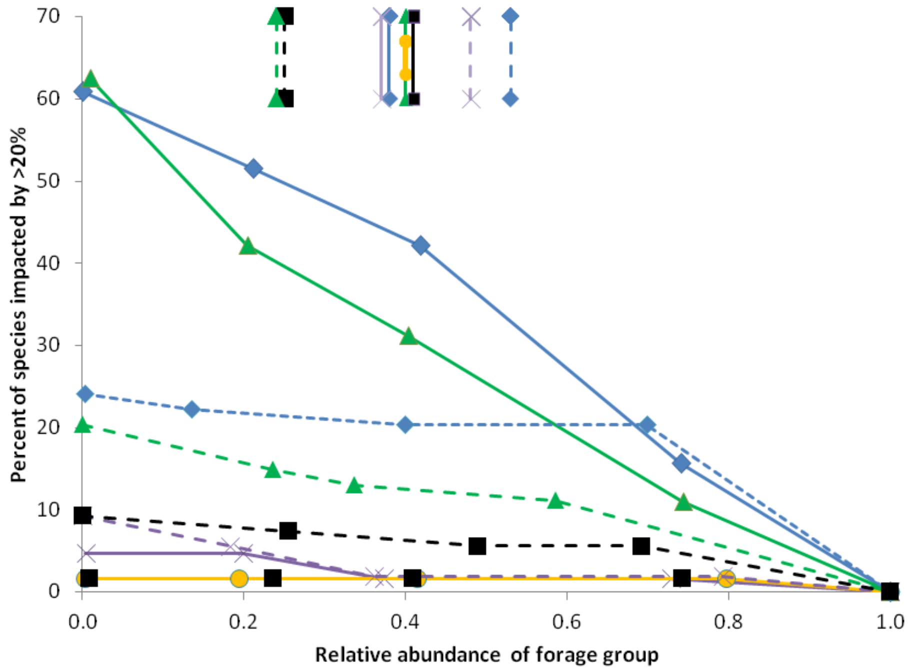


Figure MS9. Percent of species in California Current Ecosim food web model (solid lines) and Atlantis ecosystem model (dashed lines) that exhibit changes in biomass of > 20% (either positive or negative) when forage groups are depleted below unfished levels. A value of 1.0 on the x-axis represents abundance 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.

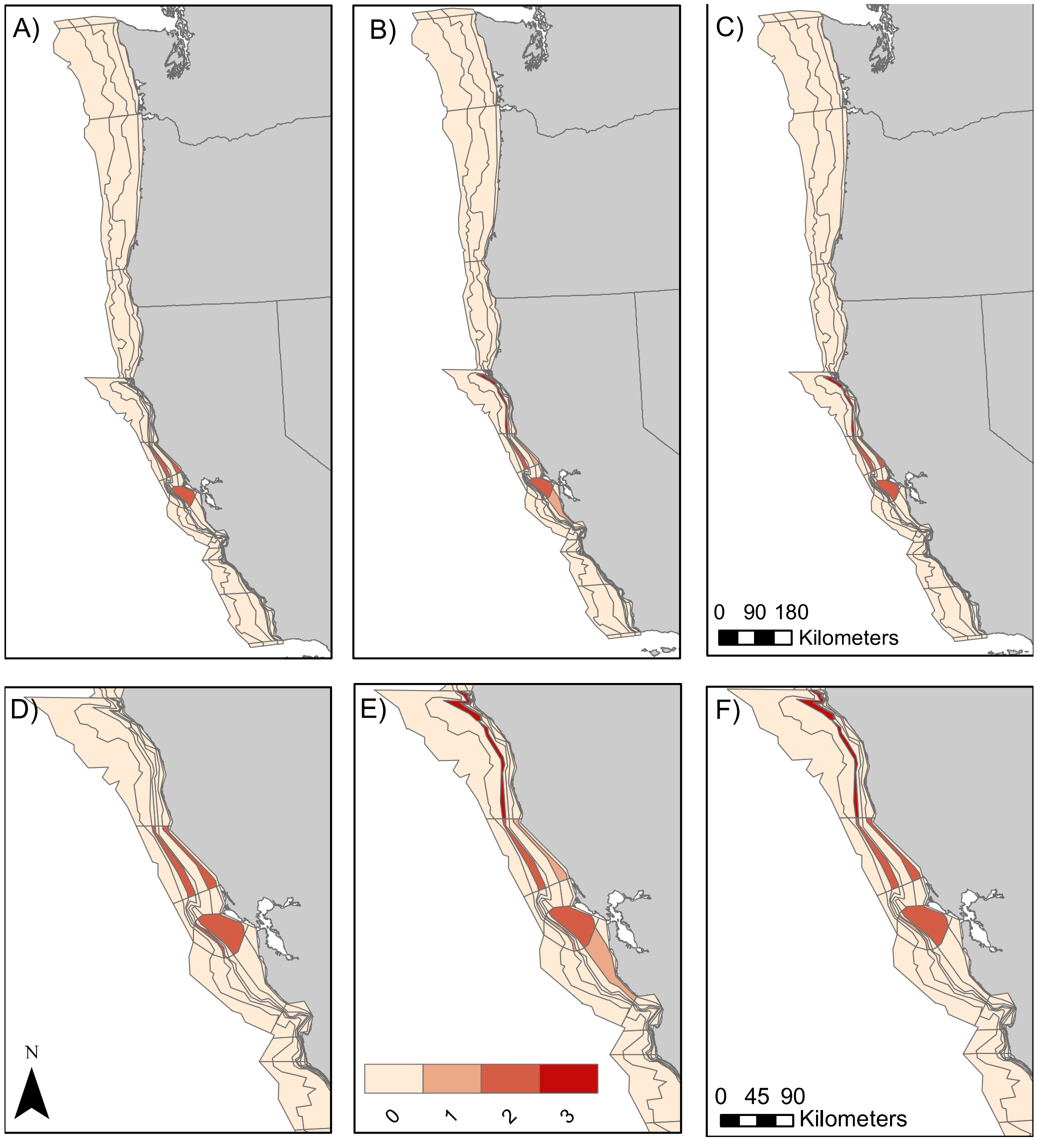


Figure MS10. Number of functional groups affected by a grenadier fishery at three fishing levels (threshold of 10 percent change) by cell. Fishing scenarios represented are F75 (A, D), F40 (B, E), and F25 (C, F). Density of color indicates increasing number of functional groups affected, as indicated 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).

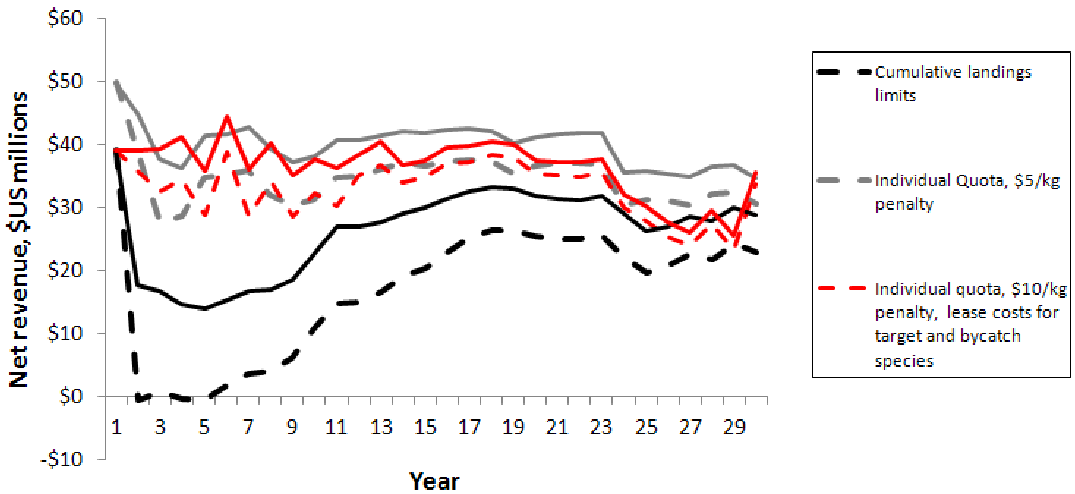


Figure MS11. Net revenue for West Coast groundfish fleets over 30 years. Solid lines denote fuel at $3/gallon, dashed lines at $6/gallon. This simple metric of net revenue is gross revenue minus fixed costs (excluding capital costs) and variable costs (fuel, ice, and food, but not labor or quota costs). 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.

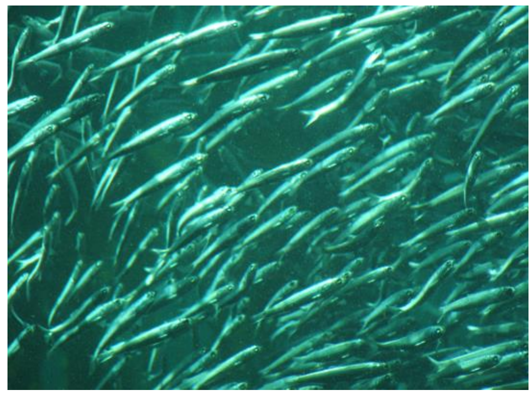
Table MS1. Percent increase in abundance and harvest due to performing dam removal versus no action, for two time periods: 1) prior to dam removal (2012 – 2019); and after removal of dams and cessation of active reintroduction and production of the Iron Gate Hatchery production (2030-2061). "95% CrI" 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 sytem. 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**).

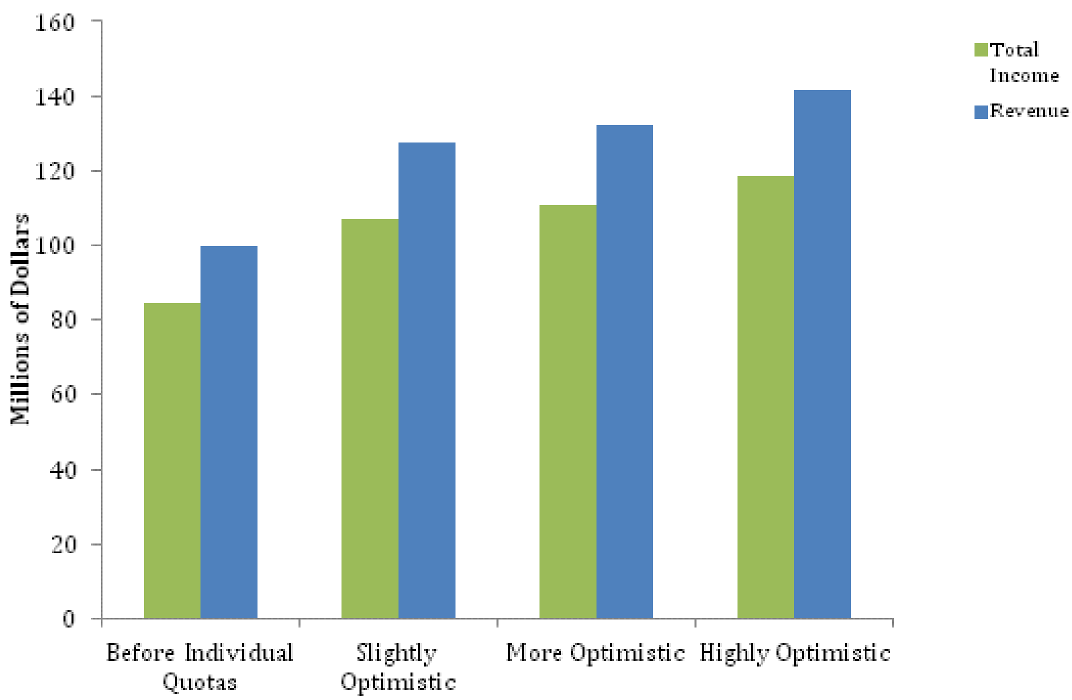


Figure MS12. Revenue in fishery sectors, and income effects in the broader West Coast economy. Year 1 predictions. Total income and revenue are represented by bars in millions of dollars (left axis). "Slightly optimistic" scenarios for individual quotas assume moderate increases in target species catch 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 km^2 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 groups (baleen whales and surface seabirds) that depend heavily on ephausiids 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.

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, microzooplantkon, 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 Tahkentich (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	GROUNDFISH TRAWL	GROUNDFISH 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	GROUNDFISH TRAWL	GROUNDFISH NONTRAWL	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.

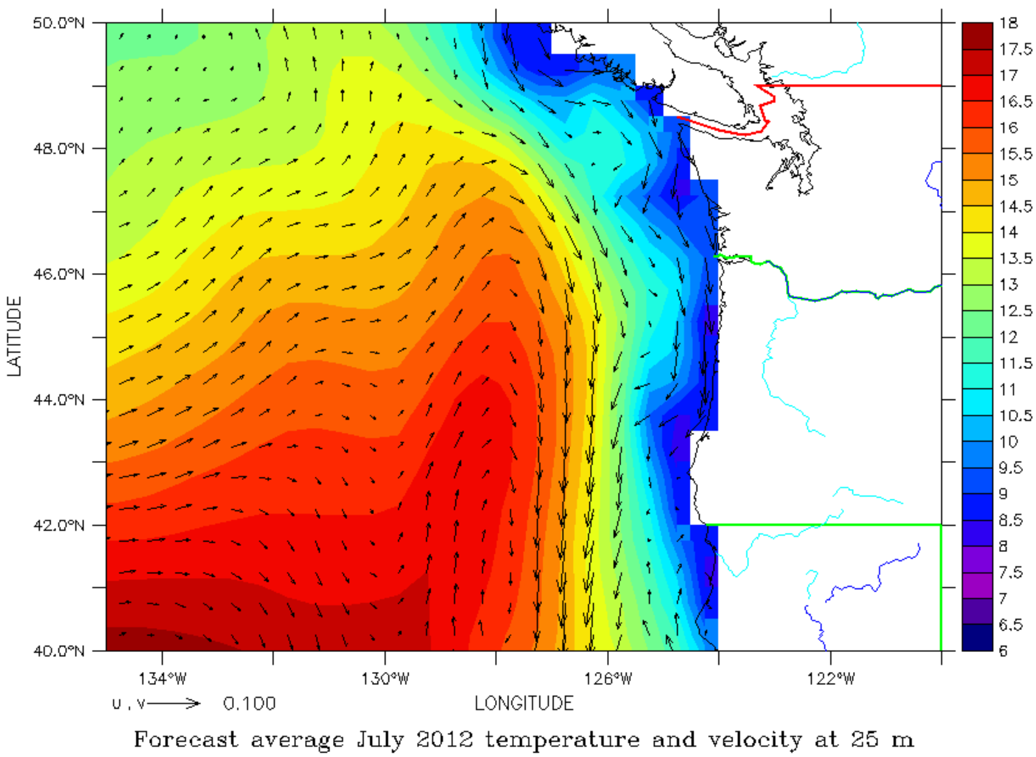
Projections of climate change can be linked to oceanographic models, and this can then be used to predict ecosystem and fishery responses. For instance, the end-to-end modeling framework being developed by Rose and colleagues **(Box MS2)** can link climate models to oceanography, plankton, small pelagic 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 heath.



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 feature 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

[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.

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 on 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 to 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 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 GROUNDFISH 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.

GROUNDFISH 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 ASHOP (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 GROUNDFISH 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 km^2. We then multiplied the effort per unit area by the total area for each proposed wave farm (30 km^2), 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 = $500,000 per km Based on figures from Dunnett and Wallace (2009).
Landing and power grid connection	Tillamook, Toledo, and Tahkentich substations, Bonneville Power Administration, Transmission Asset Network. We chose these power grid connection points (and associated landing points) based on substation transformer capacity, not including any costs of upgrading local infrastructure to accommodate wave energy production. Source: Bonneville Power Administration (2012)
Price of electricity [$ 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

		Cumulative Duration 2002-2009 (hrs/km²)			Cumulative Sets 2002-2009 (no./km²)
Cost Scenario	**Power Grid Connection Point**	**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.

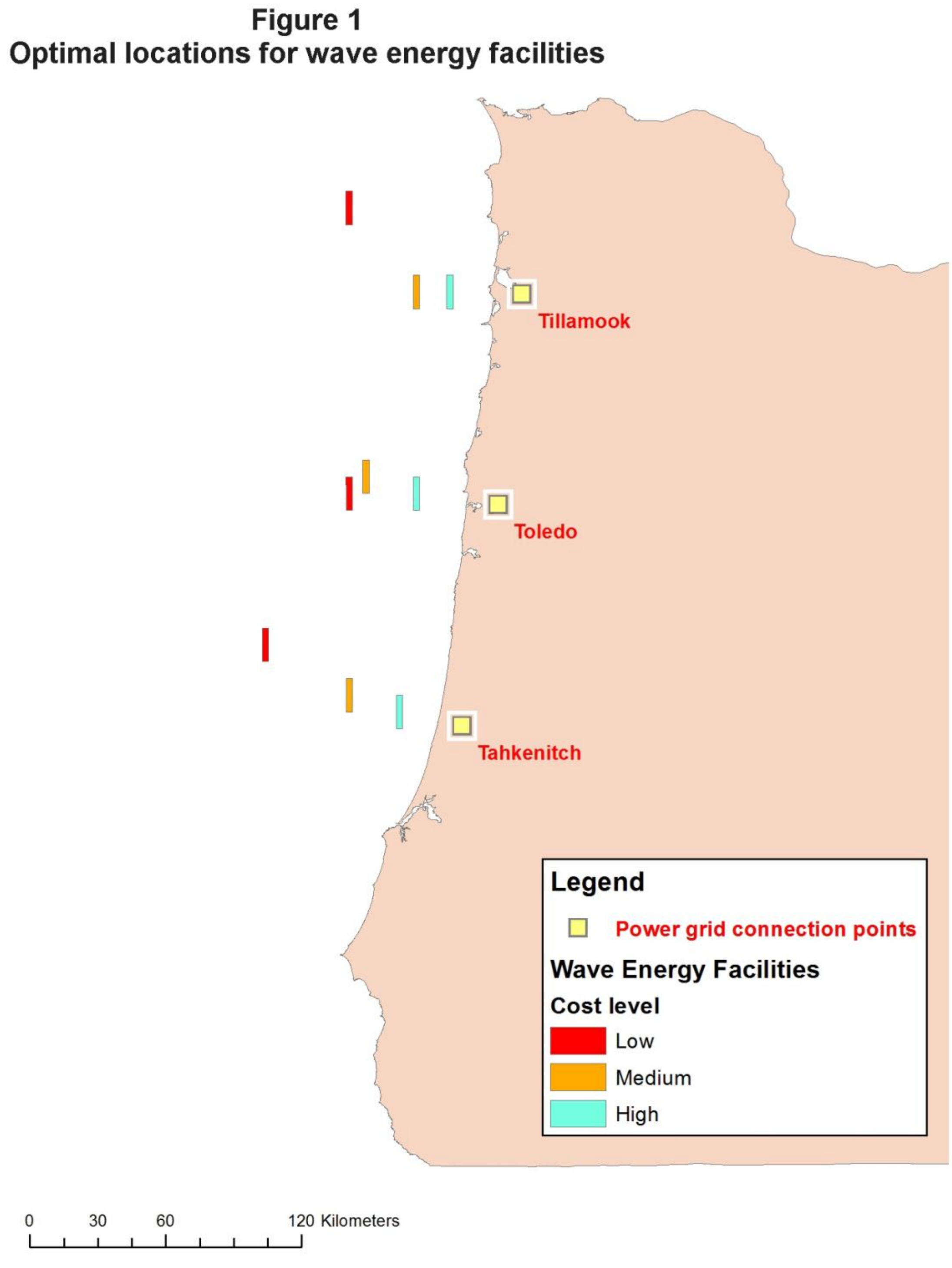
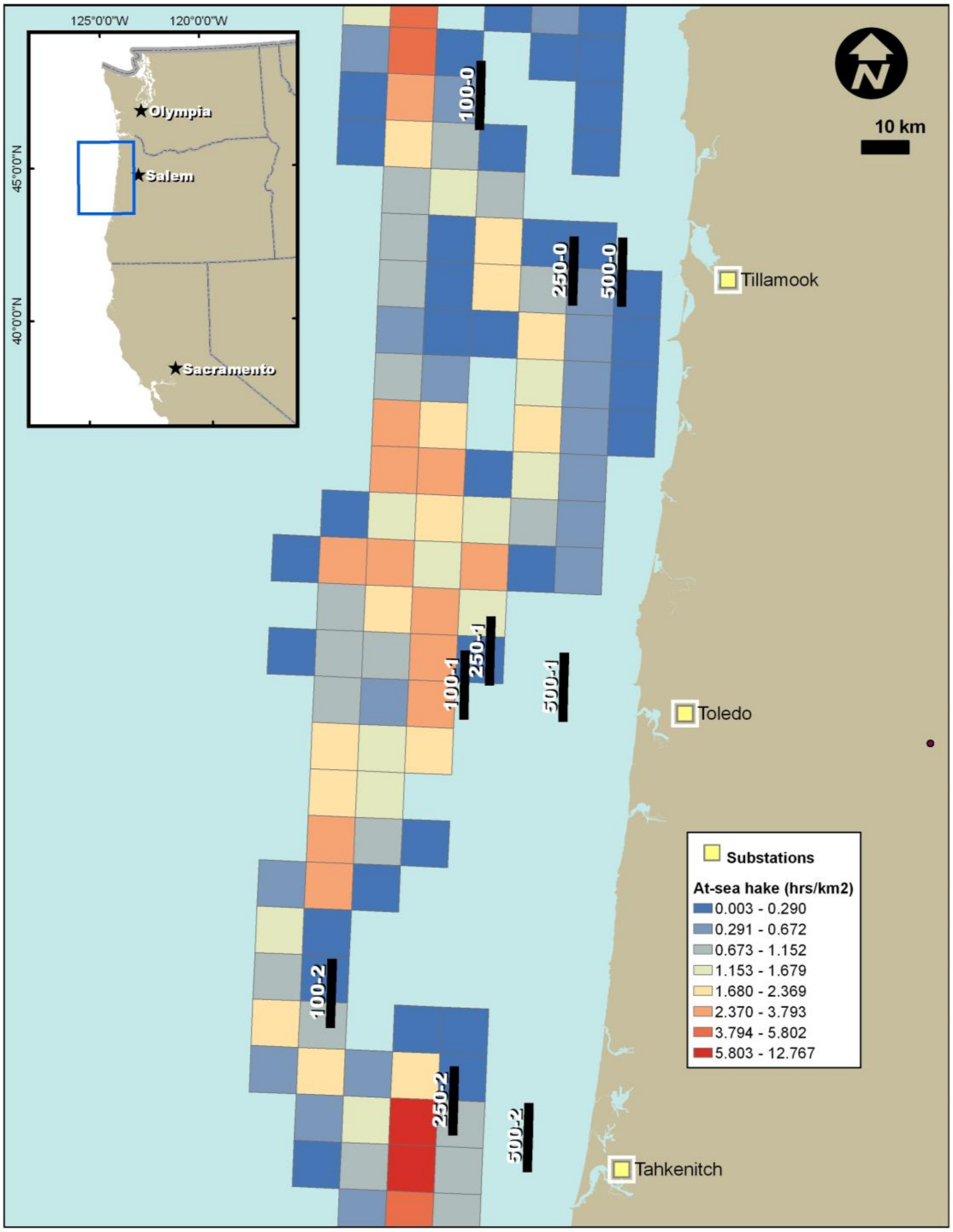


Figure 1: Using the Wave Energy Model InVEST tool, we identified three sets of optimal locations, depending on the cost scenario. The location with the maximum net economic value is what we term the optimal location for the wave energy facility.

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

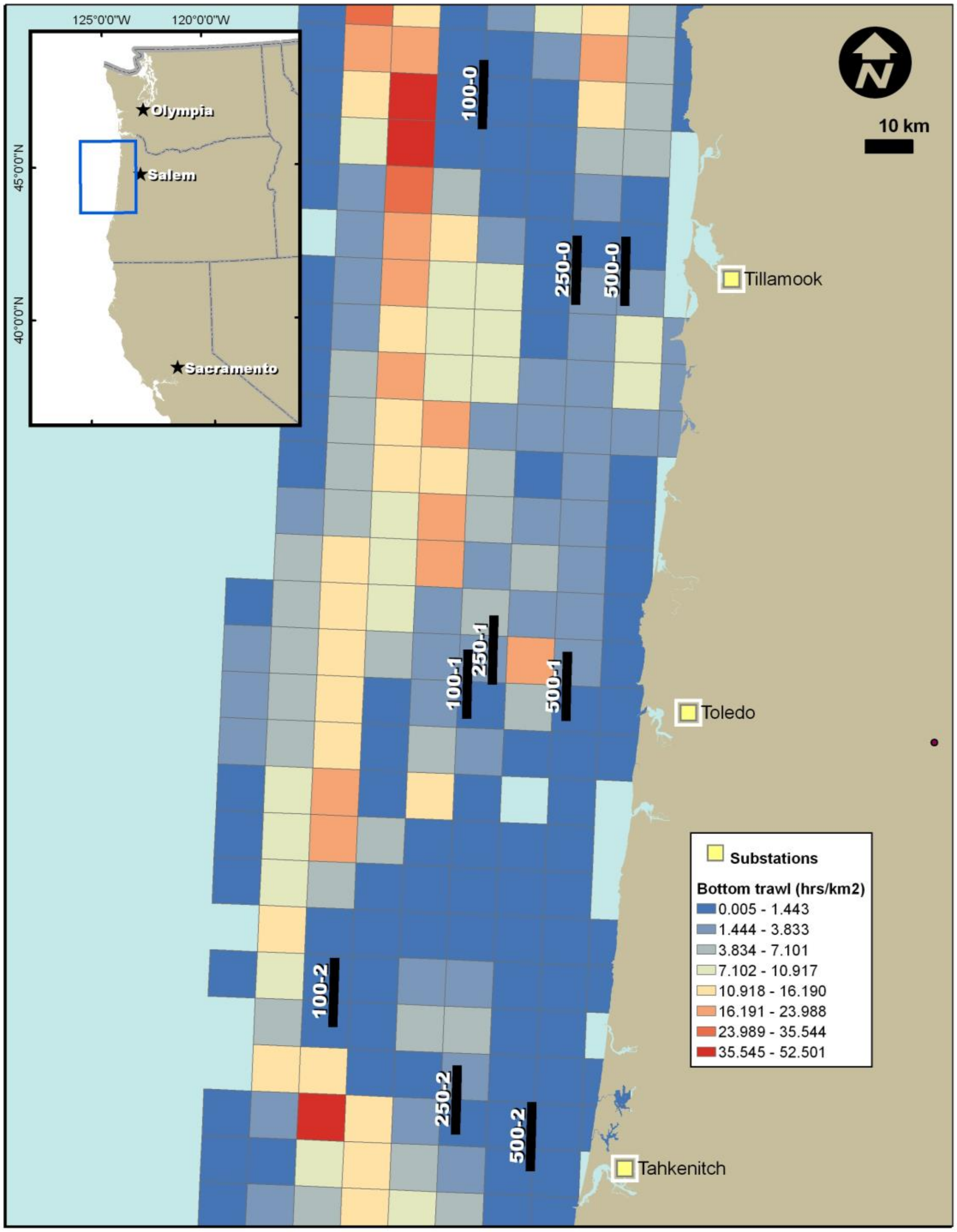
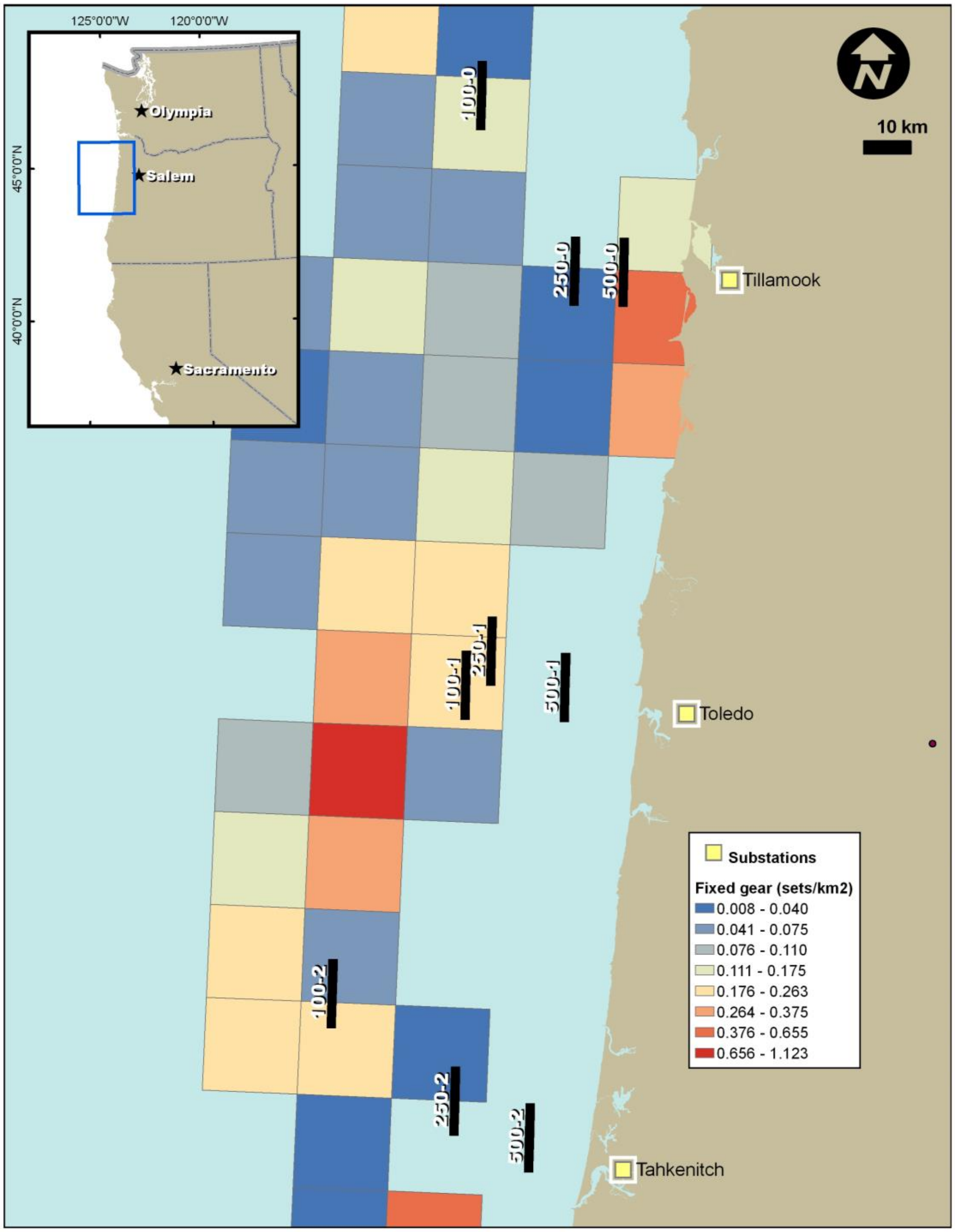


Figure2c
Groundfish fixed gear fishing effort



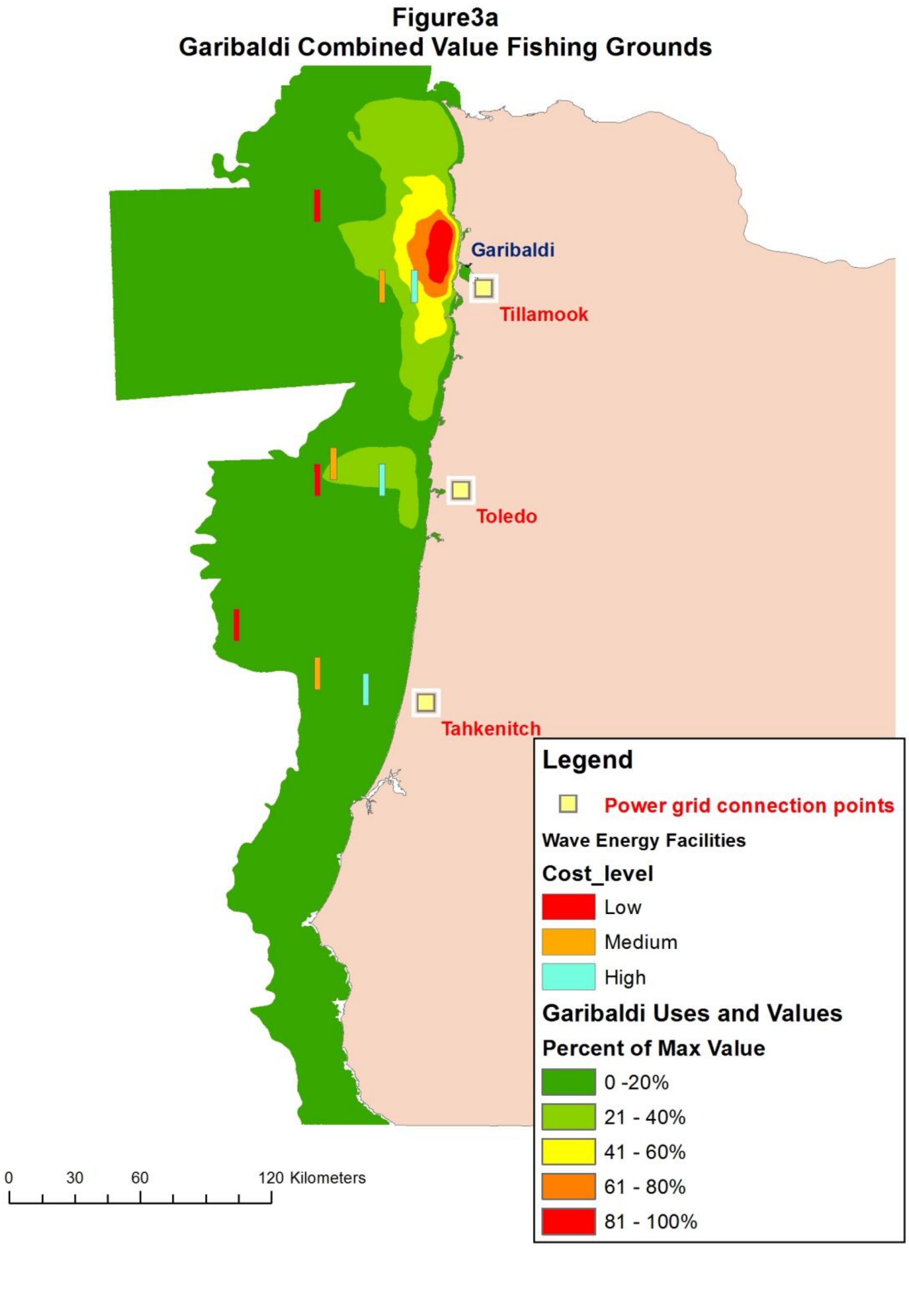


Figure 3a - 3g: Using data collected by Steinback et al. (2010) from commercial, charter, and recreational fisheries for several Oregon ports, there is a stronger possibility of conflict for ports that are close or the same as the points chosen for power grid connections (3a - 3g).

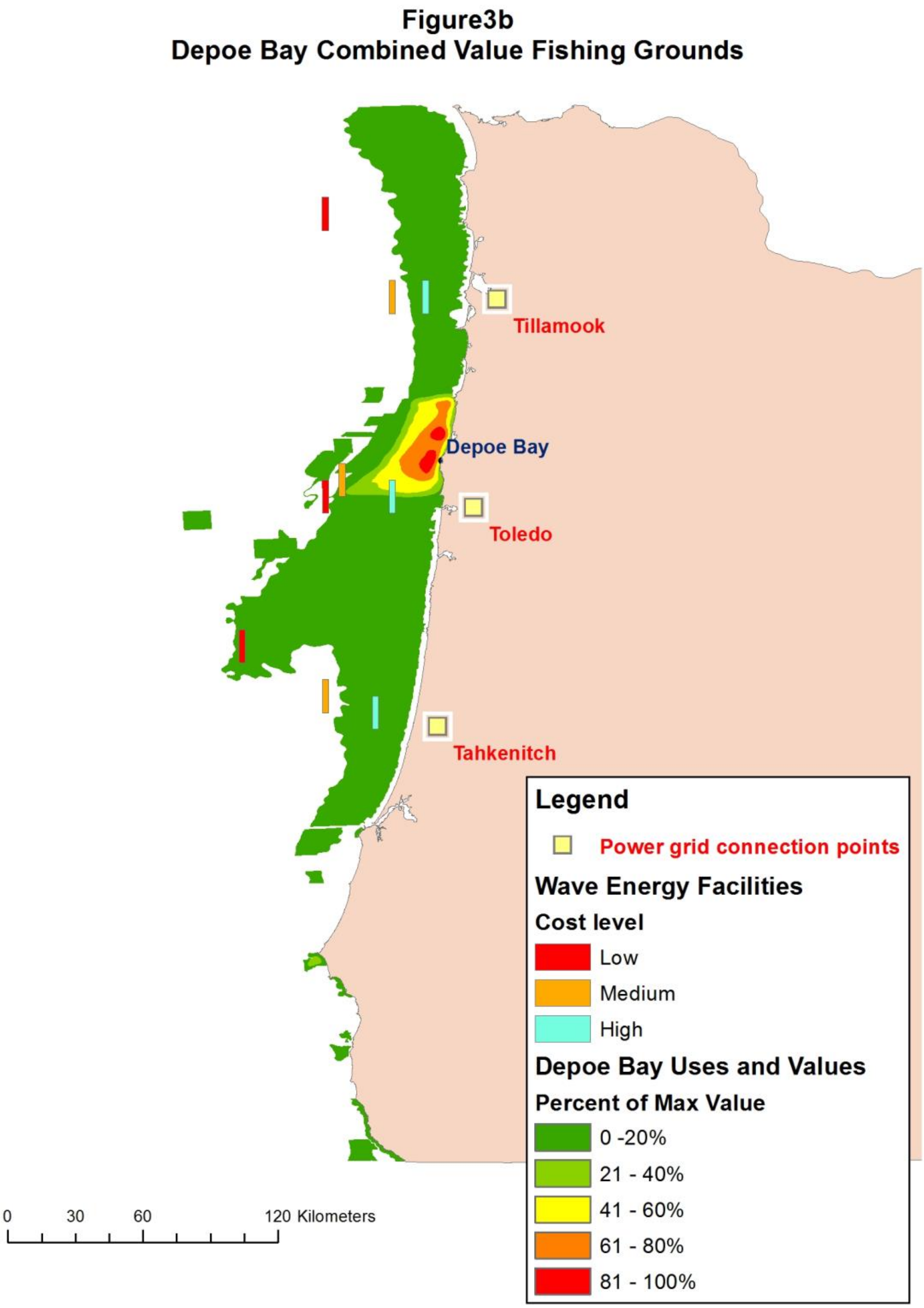
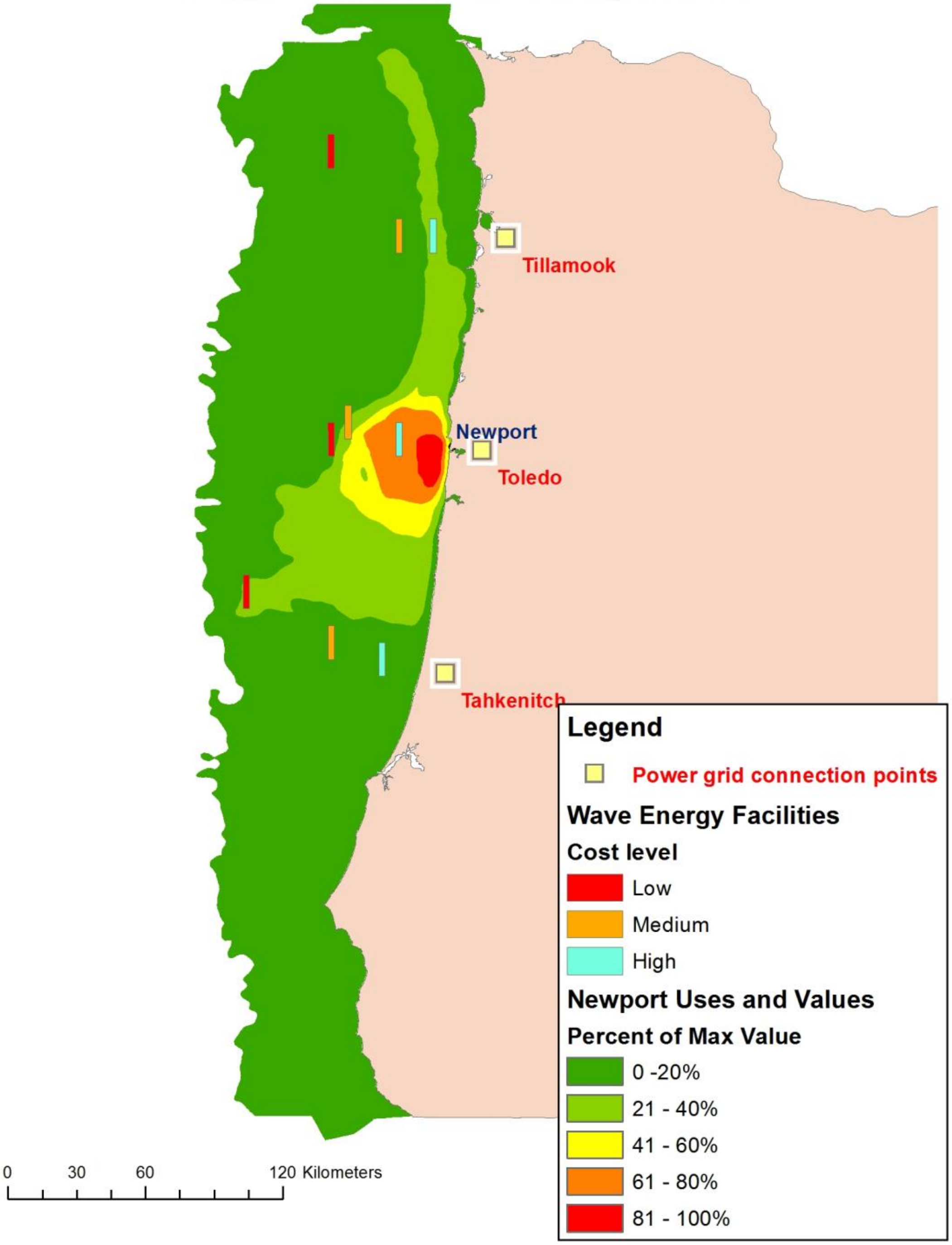


Figure3b
Depoe Bay Combined Value Fishing Grounds
Tillamook
Depoe Bay
Toledo
Tahkenitch
Legend
Power grid connection points
Wave Energy Facilities
Cost level
Low
Medium
High
Depoe Bay Uses and Values
Percent of Max Value
0 -20%
21 - 40%
41 - 60%
61 - 80%
81 - 100%
0
30
60
120 Kilometers

Figure3c
Newport Combined Value Fishing Grounds



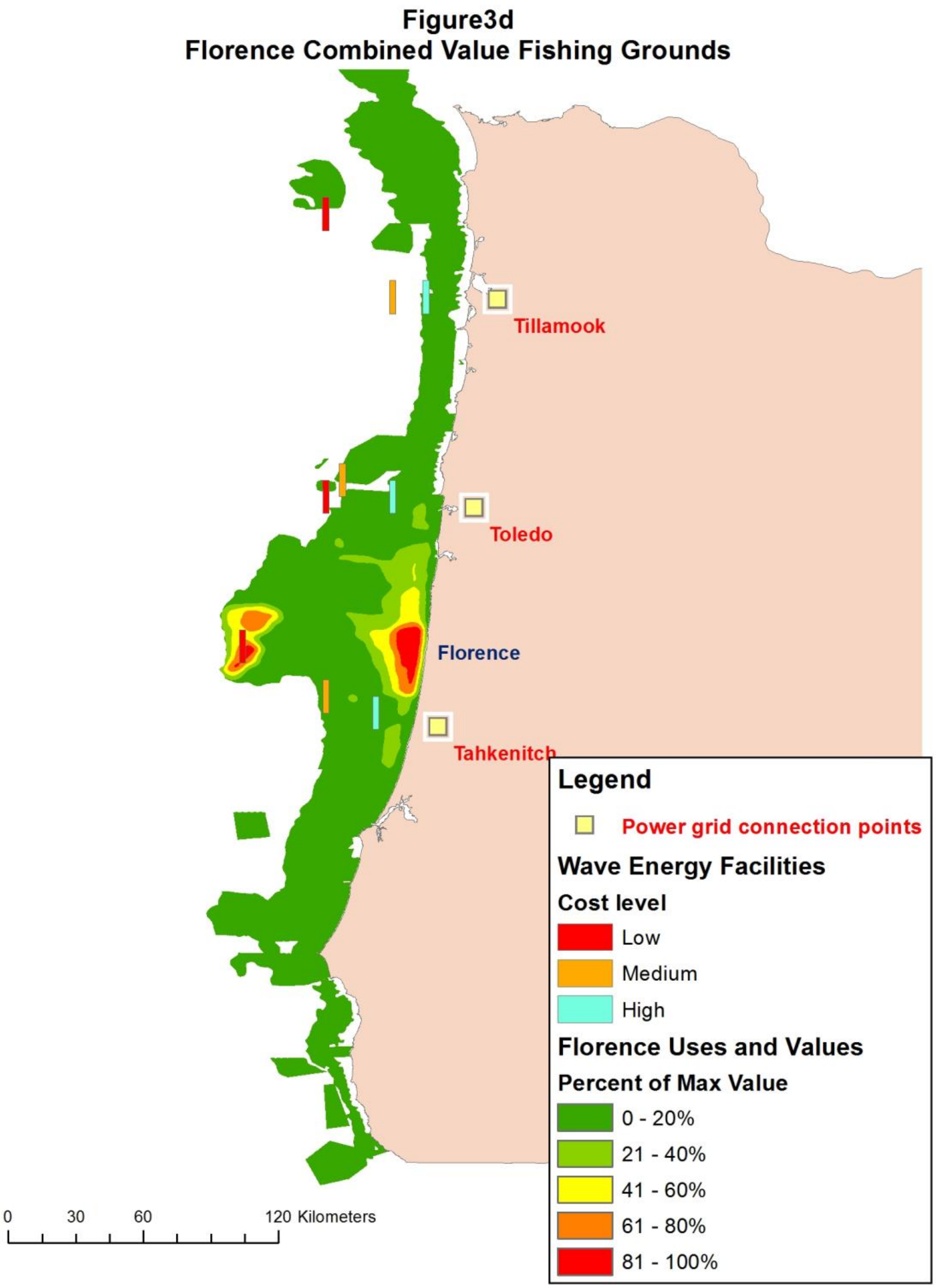


Figure3d
Florence Combined Value Fishing Grounds
Tillamook
Toledo
Florence
Tahkenitch
0 30 60 120 Kilometers
Legend
Power grid connection points
Wave Energy Facilities
Cost level
Low
Medium
High
Florence Uses and Values
Percent of Max Value
0 - 20%
21 - 40%
41 - 60%
61 - 80%
81 - 100%

Figure3e
SOORC Combined Value Fishing Grounds

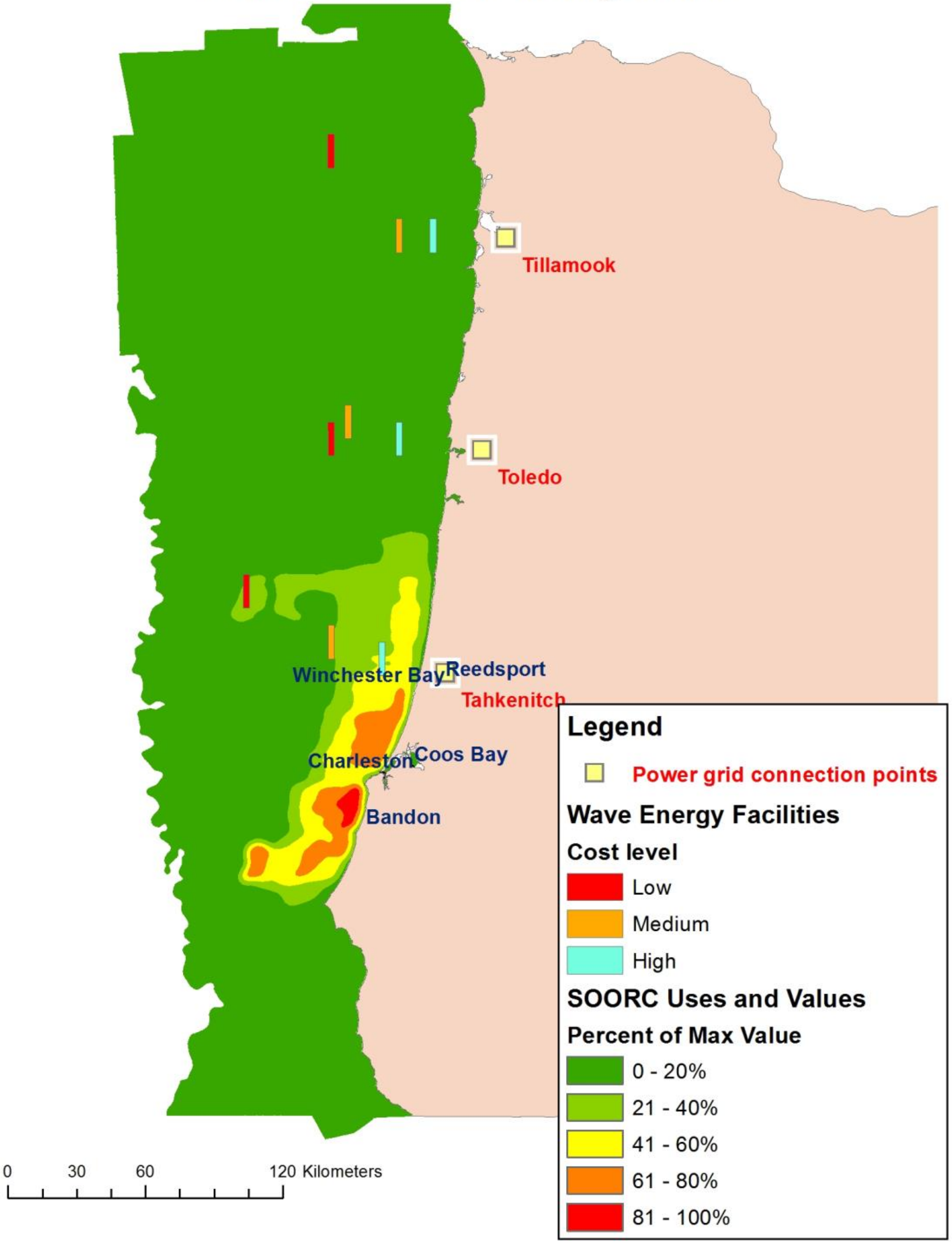


Figure3f
Port Orford Combined Value Fishing Grounds

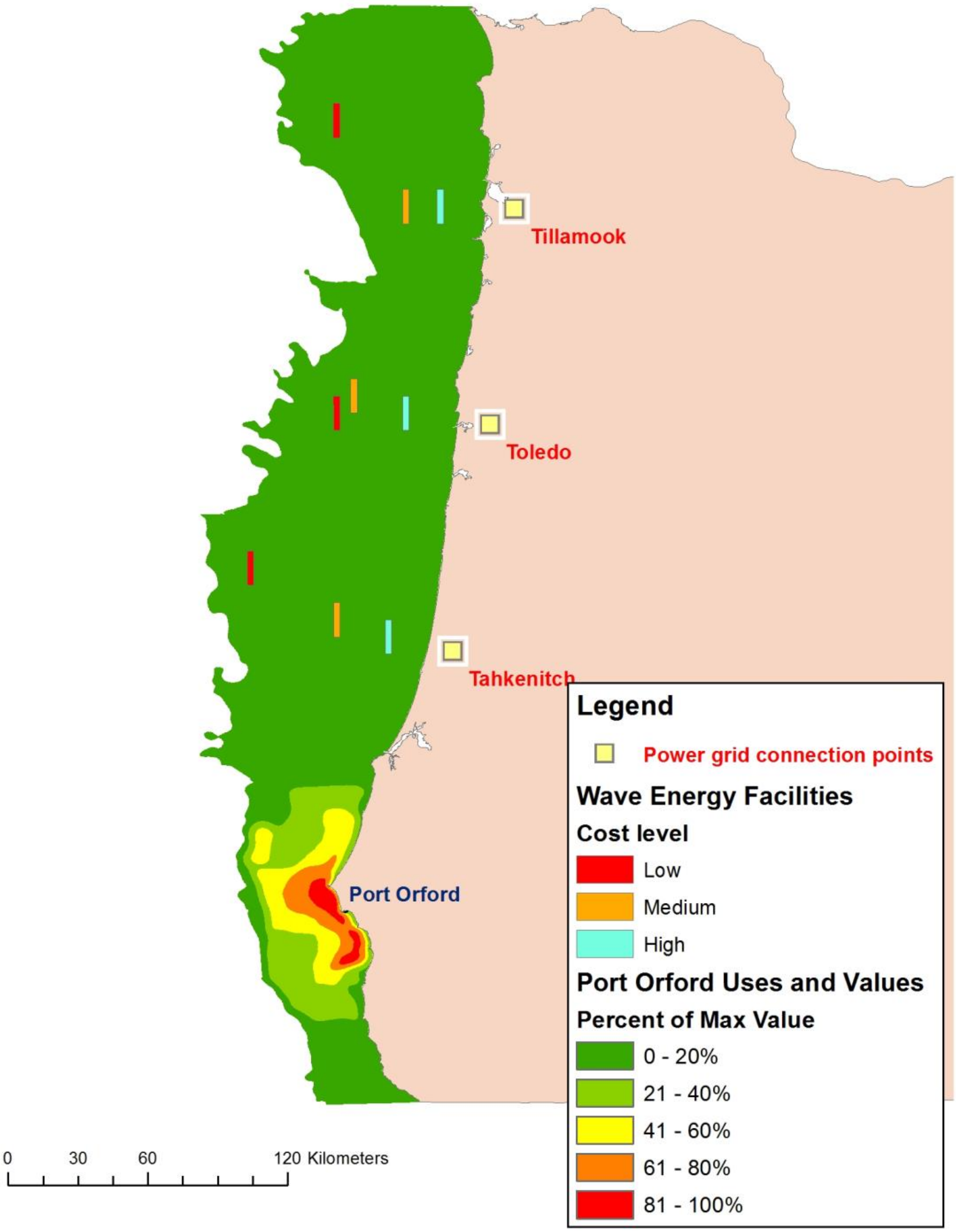
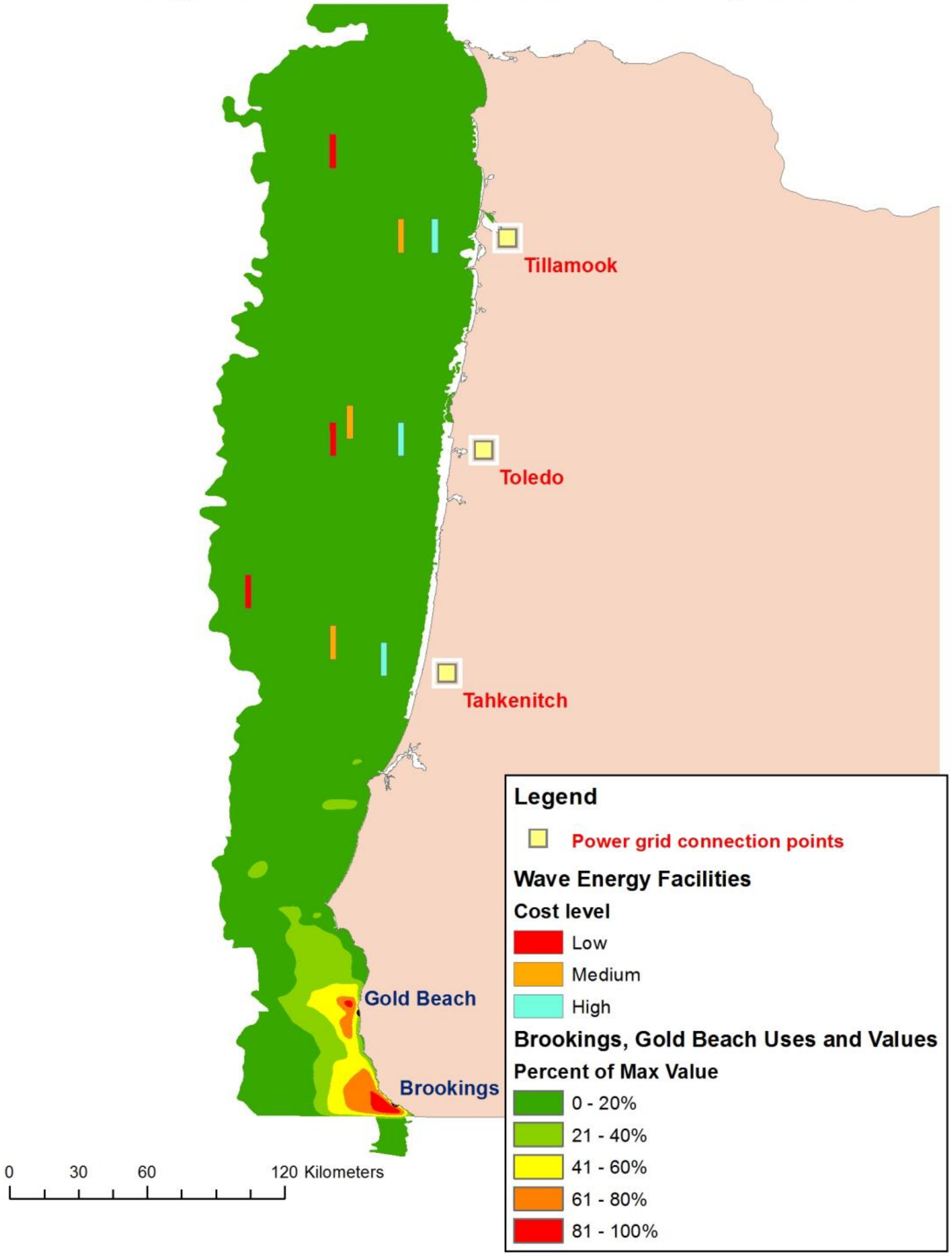


Figure3g
Brookings and Gold Beach Combined Value Fishing Grounds



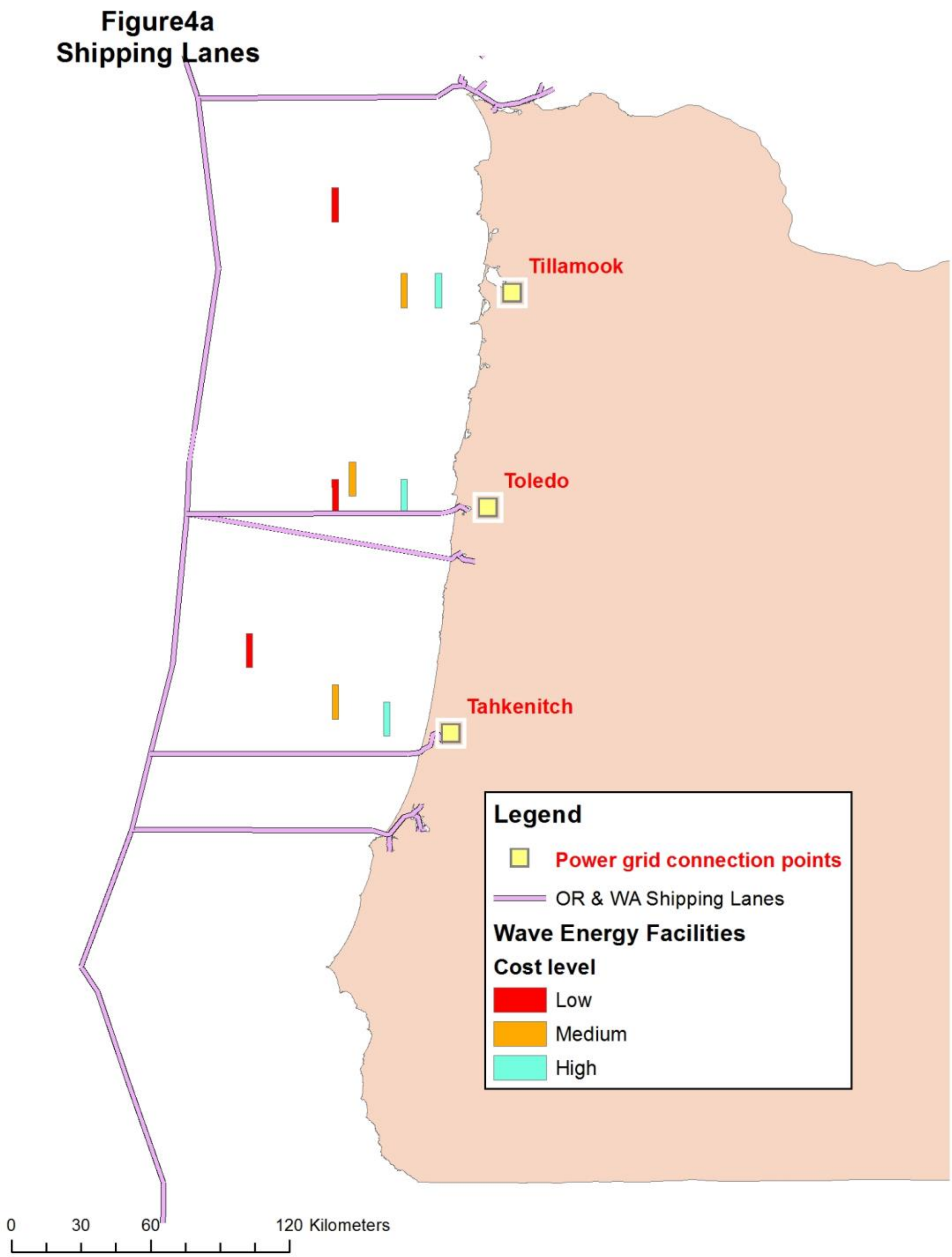


Figure 4a - 4c: For shipping lanes (4a) and towing lanes (4b), there is a strong potential conflict with the tugboat and barge tow lanes established off shore of all three connection points for the high cost scenario, while conflicts with shipping lanes are less likely. For submarine cables, there is a potential conflict with cables connected to the Tillamook area (4c).

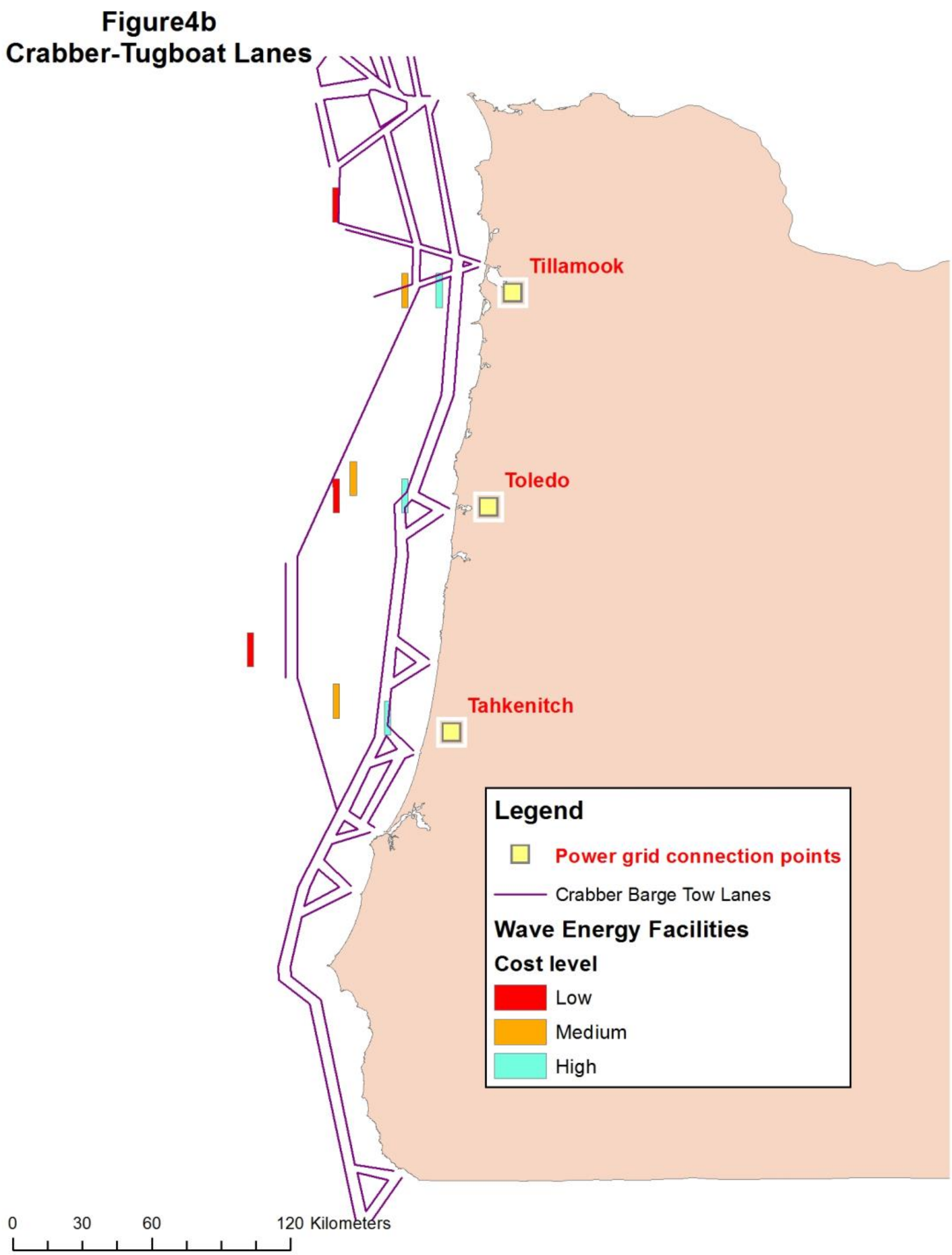


Figure4b
Crabber-Tugboat Lanes
Tillamook
Toledo
Tahkenitch
Legend
Power grid connection points
Crabber Barge Tow Lanes
Wave Energy Facilities
Cost level
Low
Medium
High
0
30
60
120 Kilometers

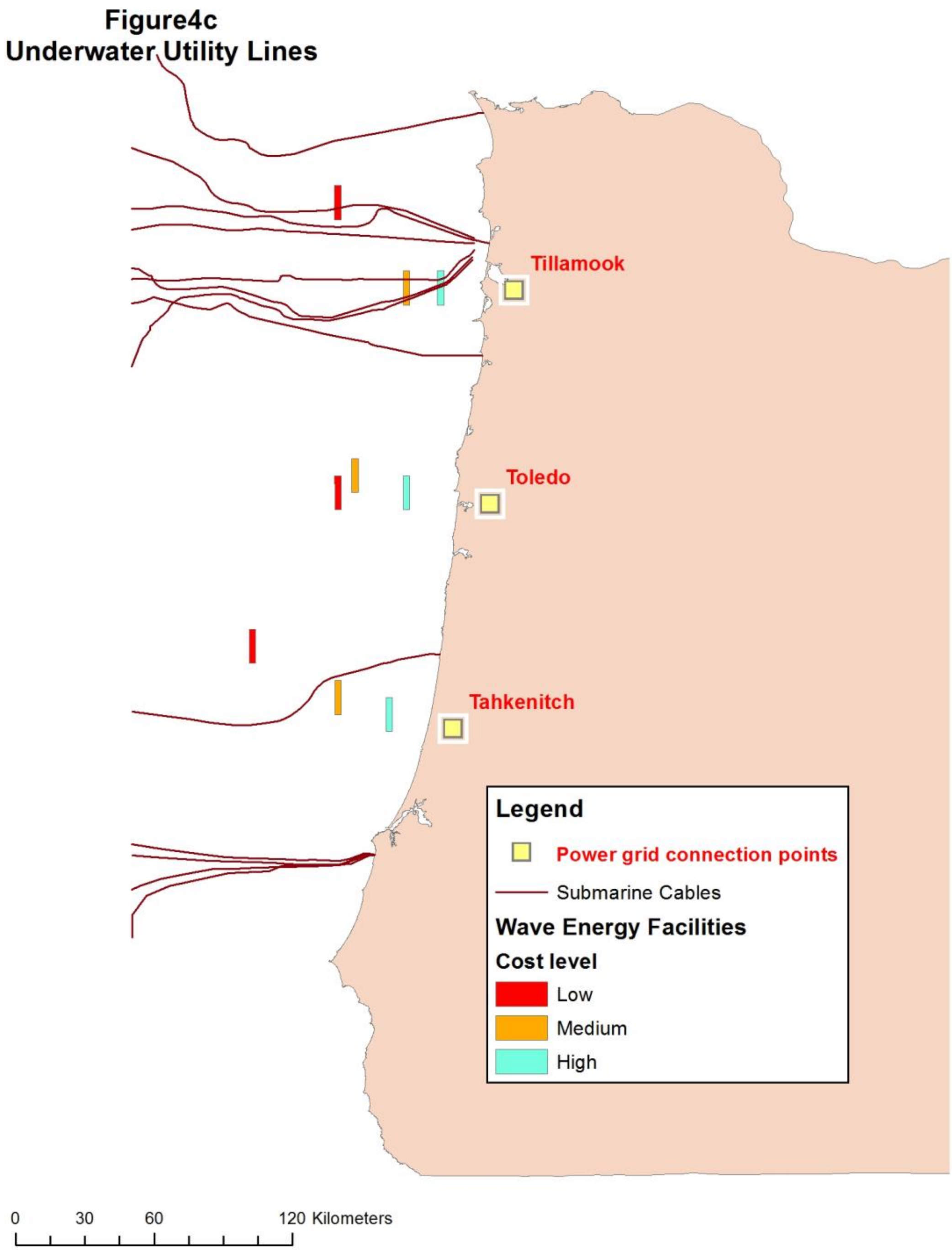


Figure4c
Underwater Utility Lines
Tillamook
Toledo
Tahkenitch
Legend
Power grid connection points
Submarine Cables
Wave Energy Facilities
Cost level
Low
Medium
High
0 30 60 120 Kilometers

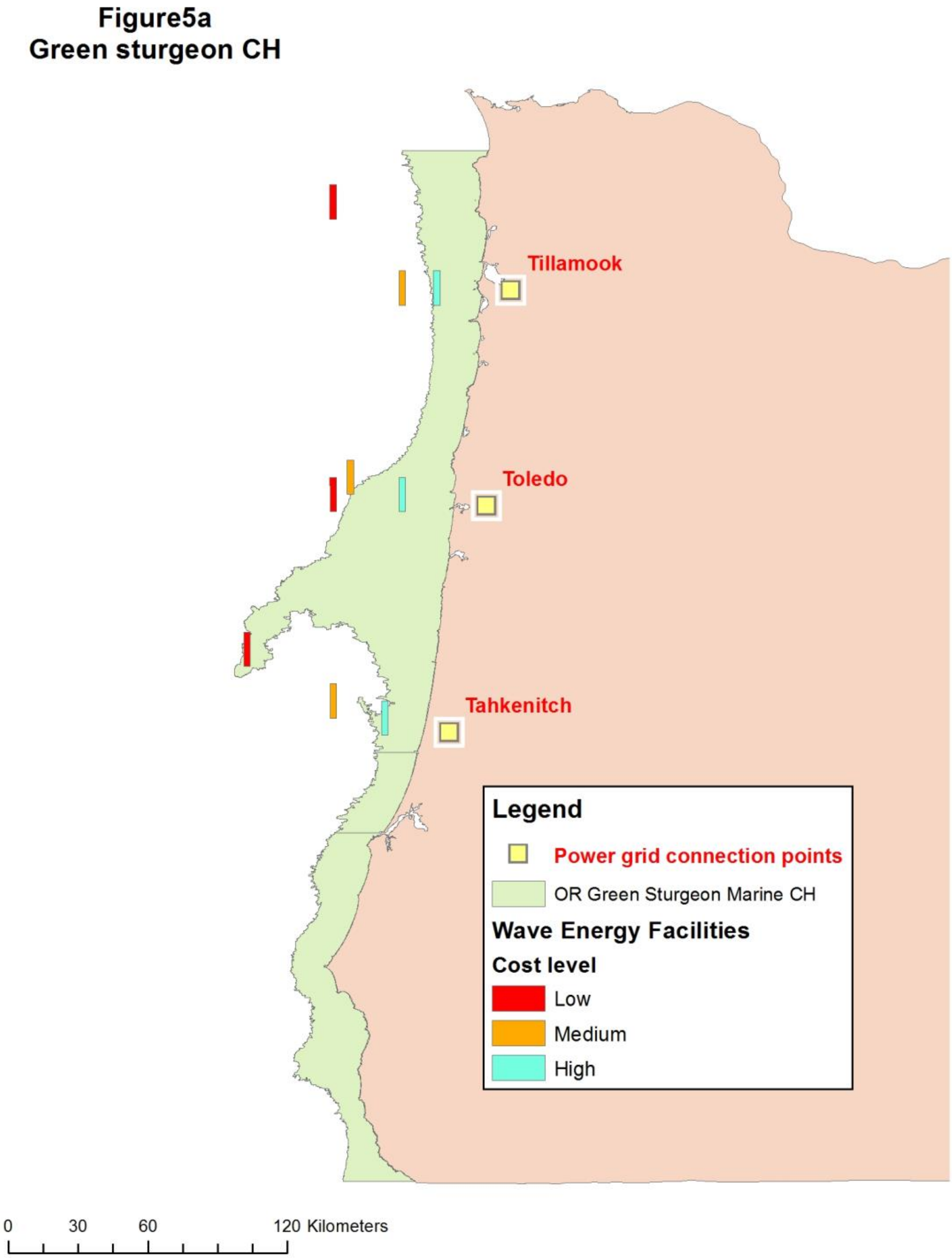


Figure 5a - 5b: The locations of some wave energy facilities overlap green sturgeon critical habitat designated under the Endangered Species Act (5a), which could trigger requirements for federal agencies such as the Federal Energy Regulatory Commission to consult with NOAA Fisheries before licensing 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

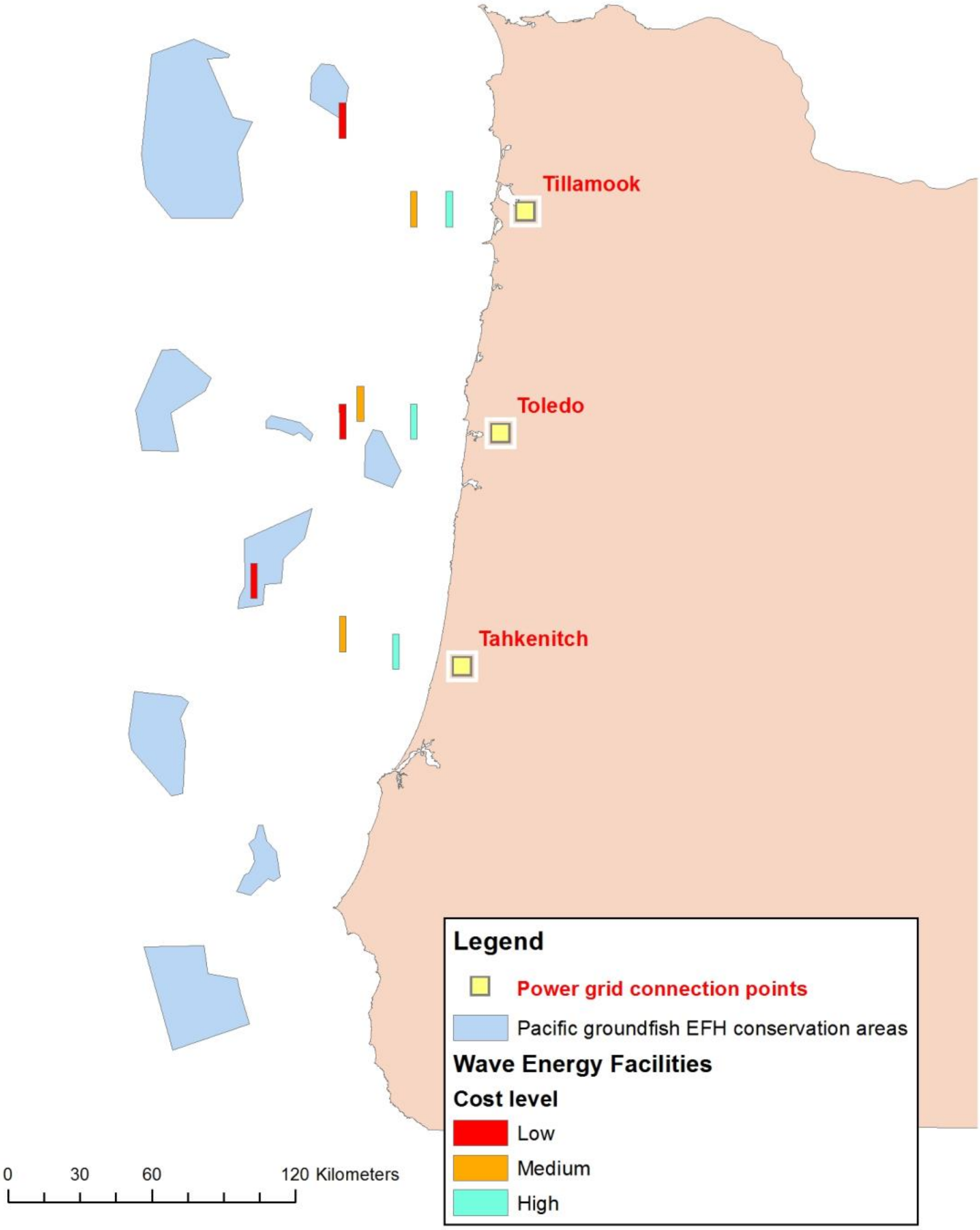


Table A1. Fixed gear fishing effort represented in West Coast Groundfish Observer Program (WCGOP) data by sector observed; including the proportion of total observed effort (cumulative hours gear was deployed) by sector from 2002-2009, the observed sector coverage rate calculated as the observed retained 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 Sclaifer 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,…,2000.

$$R_t = \alpha S_t e^{\{-\beta S_t + \epsilon_t\}}, \epsilon_t \sim N(0, \sigma_\epsilon^2) \qquad \text{(Equation 1)}$$

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 α' = log(α).

$$\log(R_t) = \alpha' + \log(S_t) - \beta S_t + \epsilon_t \qquad \text{(Equation 2)}$$

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.

$$\log(R_t) = \alpha' + \log(S_t) - \beta S_t + \delta CVI_t + \epsilon_t \qquad \text{(Equation 3)}$$

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 = IGH, TRH, and brood year t = 1979,…, 2000.

$$h_{j,t} = \kappa_j + CVI_t + \gamma_j\,(Q_t) + u_{j,t}, \qquad \text{(Equation 4)}$$

$$CVI_t \sim N(0, \sigma^2_{CVI})$$

$$u_{j,t} \sim N(0, \sigma^2_h)$$

where the log hatchery survival rates ($h_{j,t}$) for hatchery j = 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 $\boldsymbol{\theta}$ given the observed data $\mathbf{R}$ by using Bayes' rule:

$$\pi(\boldsymbol{\theta}\,|\mathbf{R}) = \frac{\pi(\boldsymbol{\theta}\,)f(\mathbf{R}|\boldsymbol{\theta}\,)}{f(\mathbf{R})} \qquad \text{(Equation 5)}$$

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. SRF ≈ 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 (RCDT 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).

$$DIC = \bar{D}(\mathbf{R}|\boldsymbol{\theta}\ ) + p_D \qquad \text{(Equation 6)}$$

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 *pD*, 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 $pD = \bar{D} - \tilde{D}$ (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

$$\alpha S_{msy} e^{\{-\beta S_{msy} + \delta \overline{CVI}\}} - S_{msy} \quad \text{(Equation 7)}$$

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 $\varepsilon_{i,t}$ from $N(0, \sigma^2_{\varepsilon,i})$. 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 km^2 (1,420 mi^2), extending from its source on the eastern edge of the basin, and flowing through the Klamath Marsh, which covers 601 km^2 (232 mi^2) (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 km^2 (120 mi^2), 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 km^2 (1,580 mi^2), 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 km^2 (568 mi^2), 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 km^2 (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 km^2 (559 mi^2). 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 km^2 (48 mi^2, 30,537 acres), accepting flows not only from the Upper Sycan River, but from an additional drainage area of 456 km^2 (176 mi^2) surrounding the marsh (USFS 2005). The lower extent of the Sycan River subbasin begins below the Sycan Marsh, and is approximately 601 km^2 (232 mi^2) 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 km^2 (219 mi^2) 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 km^2 (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:

$$E_{new,i,t} = p_{i,t}E_{new\ ocean,i} + (1 - p_{i,t})E_{new\ spring,i} \qquad \text{Equation (11)}$$

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).

$$r_{new\ ,i,t} = p_{i,t}r_{new\ ocean,i} + (1 - p_{i,t})r_{new\ spring,i} \qquad \text{Equation (12)}$$

The values of $E_{new,i,t}$, $r_{new\ ,i,t}$, and the spawner abundance three years previously ($S_{i,t\text{-}3}$) 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 basinδ_i CVI $_{i,\ t\text{-}2}$. 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 km^2 (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 km^2 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 capacityby re-arranging Equation 9

$$\beta_{new} = \frac{\alpha'}{S_{ueq\ new}} \qquad \text{Equation (10)}$$

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 form 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 $\alpha'$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).

$$r^{*}_{new\,,i,t} = p_{i,t} r^{*}_{new\ ocean,i} + (1 - p_{i,t}) r^{*}_{new\ spring,i} \qquad \text{Equation (13)}$$

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 $\geq$ 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 km^2 (assuming a total watershed area of 12,066 km^2 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 km^2 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 km^2, 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. *Ceratomixta 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 toChinook 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; $N_{3,Sept1}$ 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	$N_{3,Sept1}$	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: $\hat{s}'_{TRH,1990} = \exp(0.89 s_{IGH,1990})$.								

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)
κ_j, j = IGH, TRH	N(0,1000)
γ_j, j = 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 km^2 (mi^2)**
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%

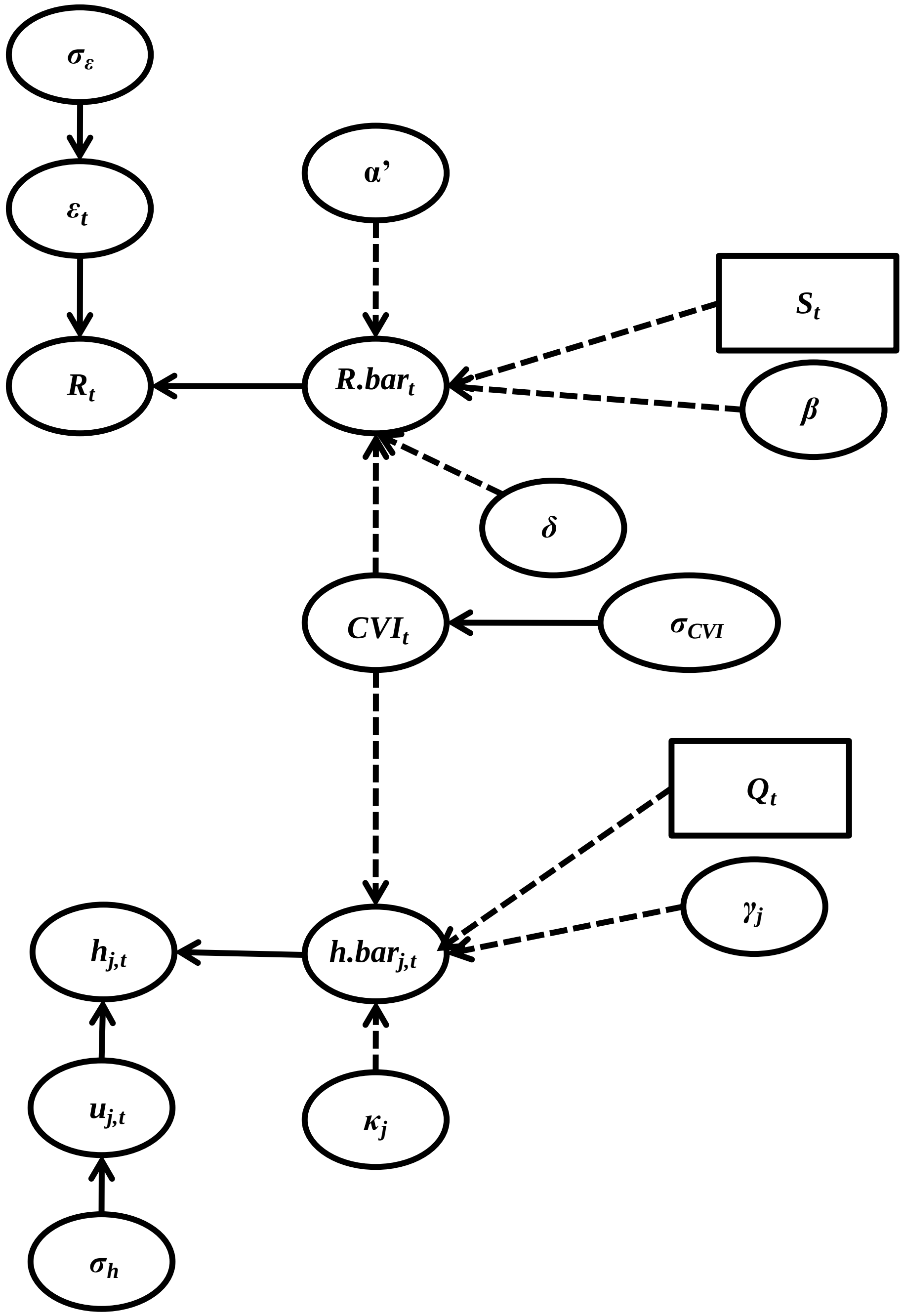


Figure 1. Directed Acyclic Graph (DAG) of the conditional relationships between coefficients in equations for estimating log hatchery survival rates (*h*) and natural recruitment to age 3 (*R*) as depicted in Equations 3 and 4. Ovals represent nodes that are calculated quantities whereas squares represent 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.

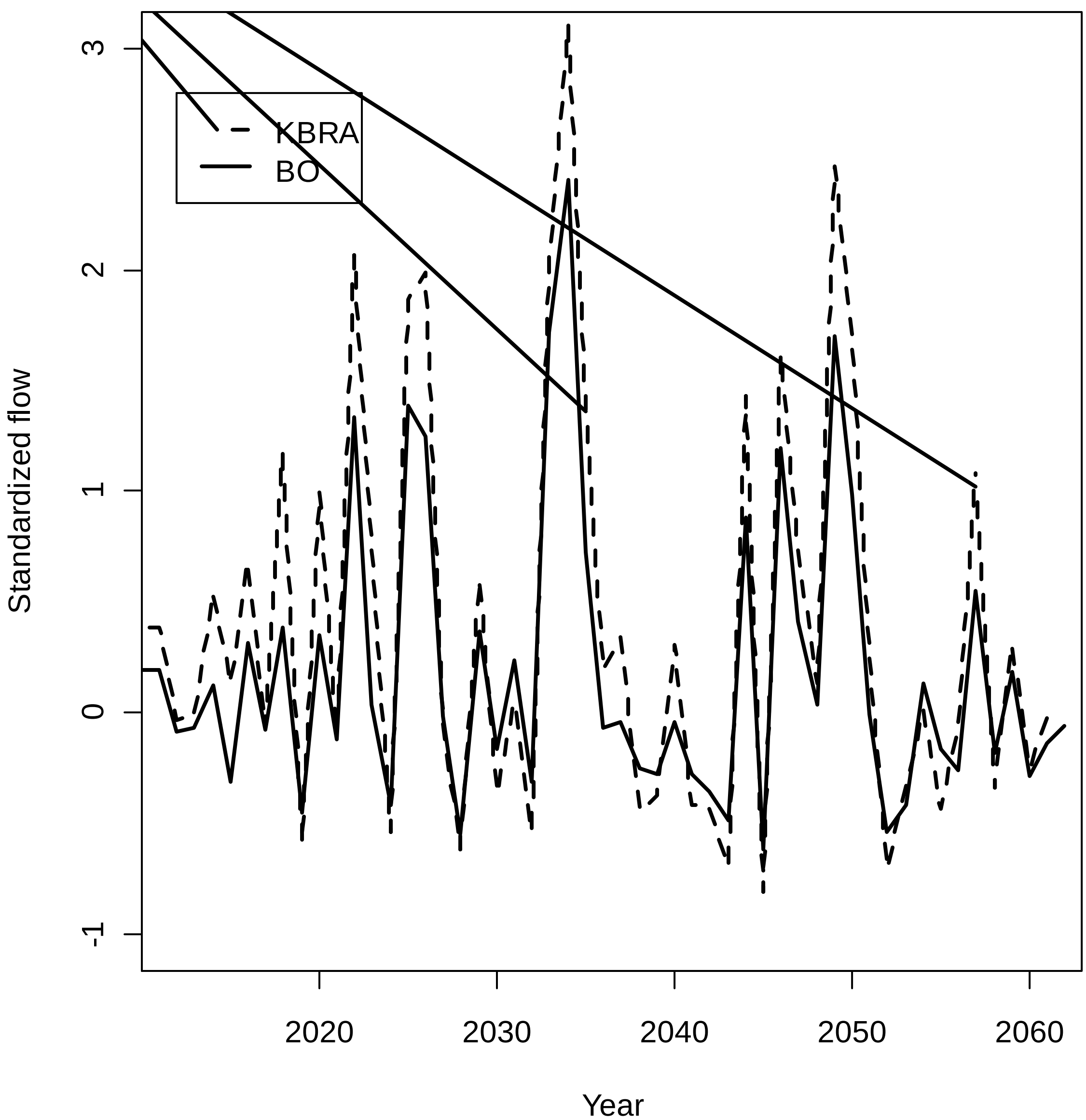


Figure 2. Flow forecasts for 2012 to 2061 under NMFS Biological Opinion (BO) and under KBRA in the Klamath River at Seiad during July. Flow values were standardized using the mean Klamath River flows in July at Seiad Valley from 1980 to 2000 (mean = 1589, sd = 944.17). The standardized flow values were incorporated in the model for forecasts of abundance and harvest under NAA and DRA.

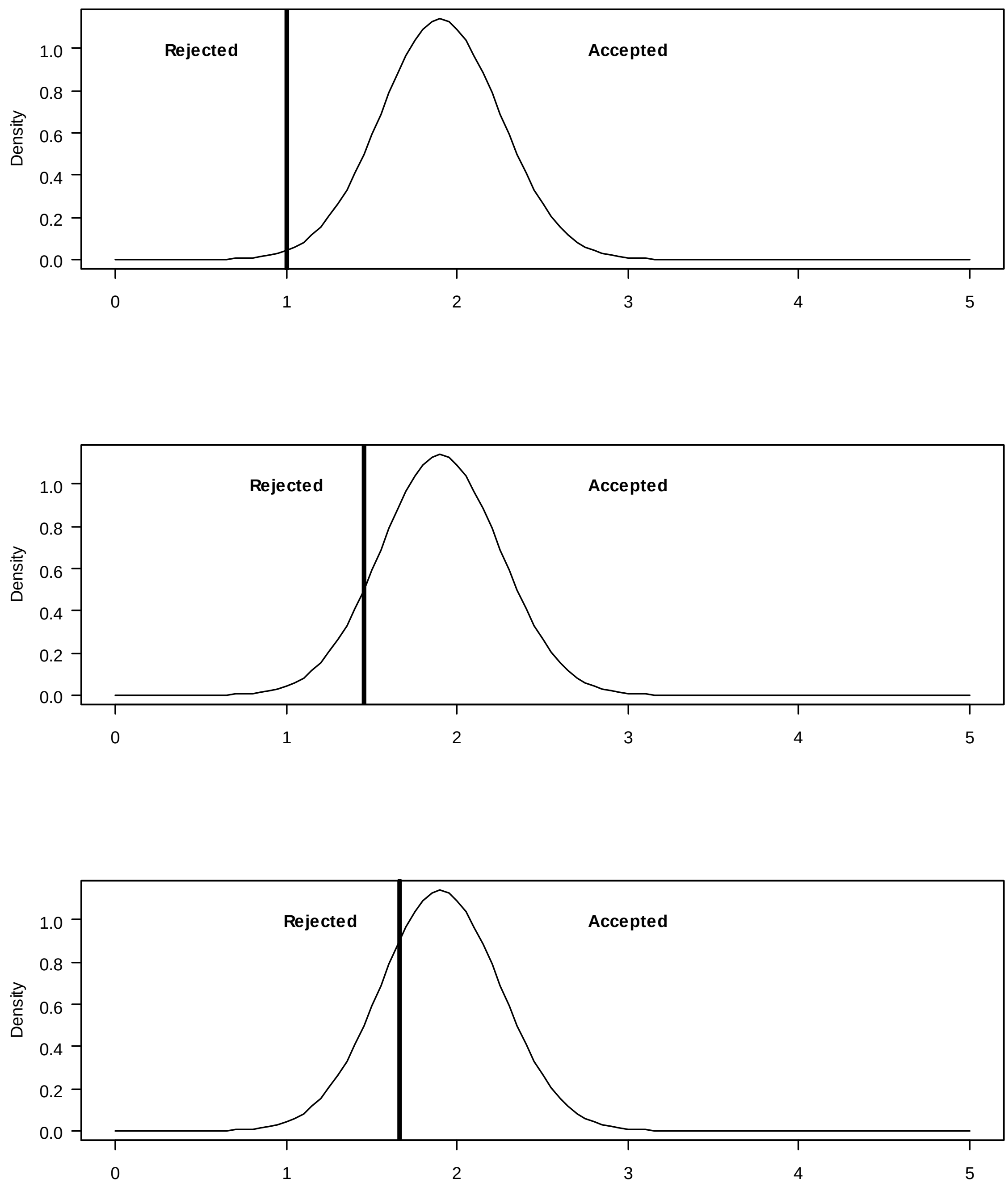


Figure 3. Depiction of sampling from higher percentiles of a hypothetical productivity distribution over time. Samples of productivity occur only from the Accepted region. Early in the time series, samples from almost the entire distribution are accepted (Top, Accepted threshold at 0.05 quantile). Later 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).

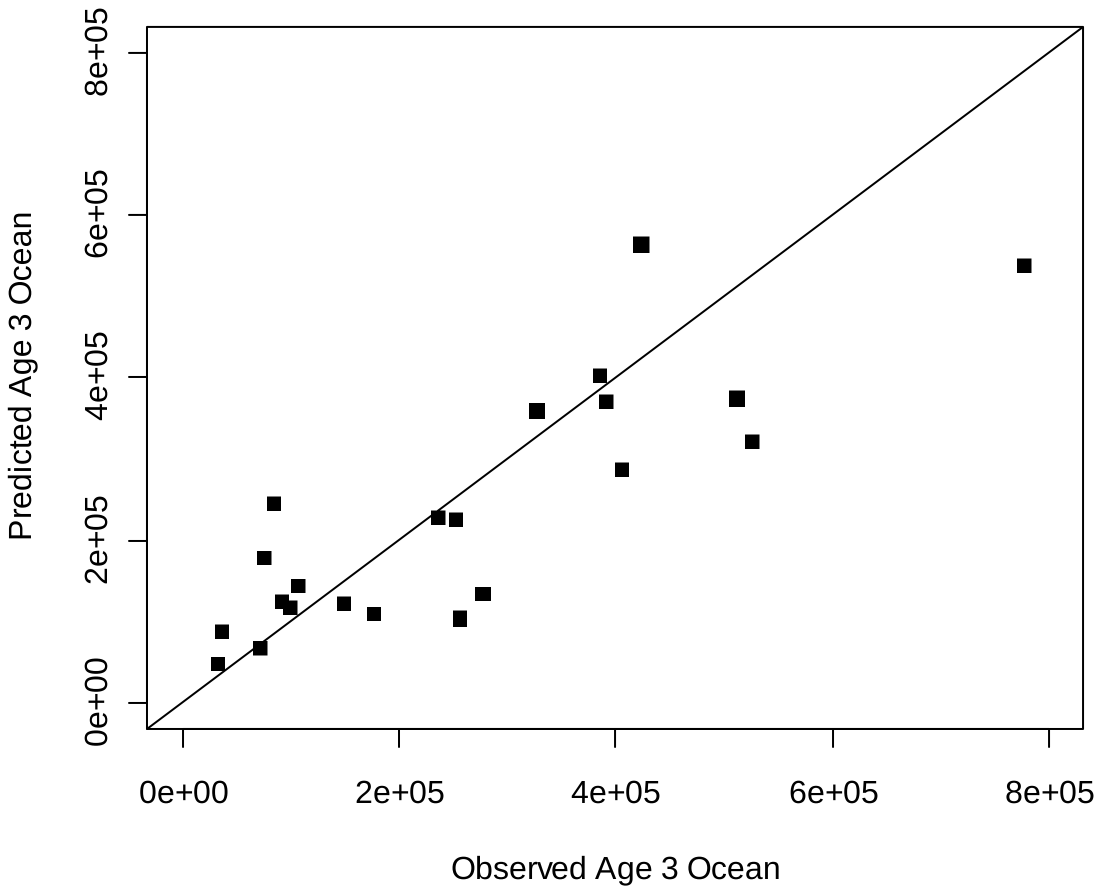
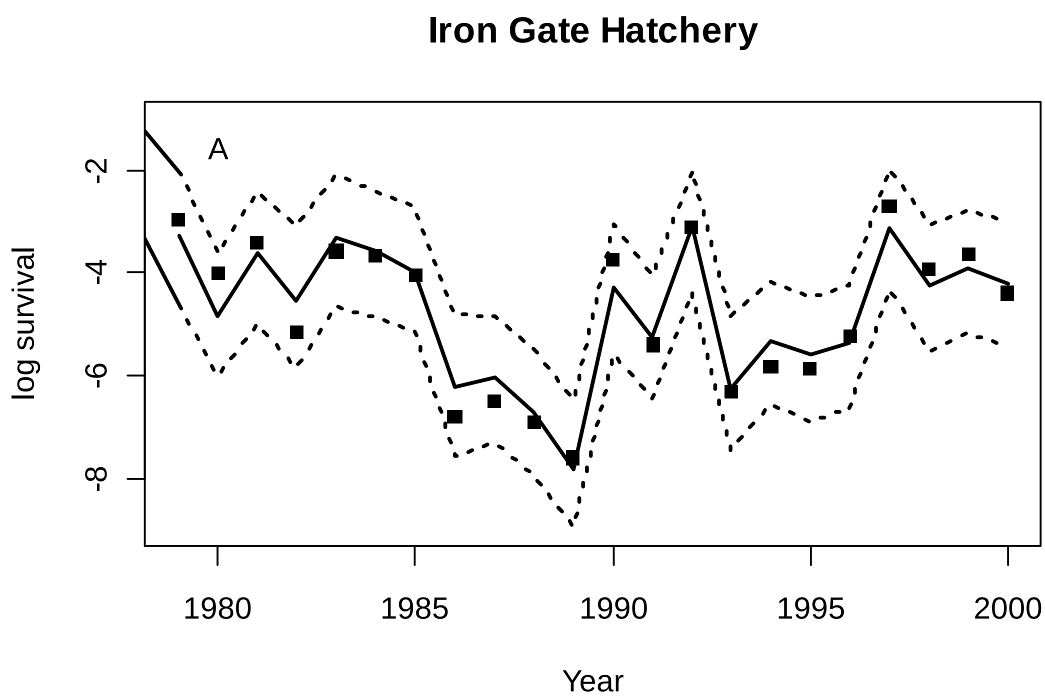


Figure 4. Median predicted ocean age 3 recruits from the Ricker stock recruitment model and observed ocean age 3.



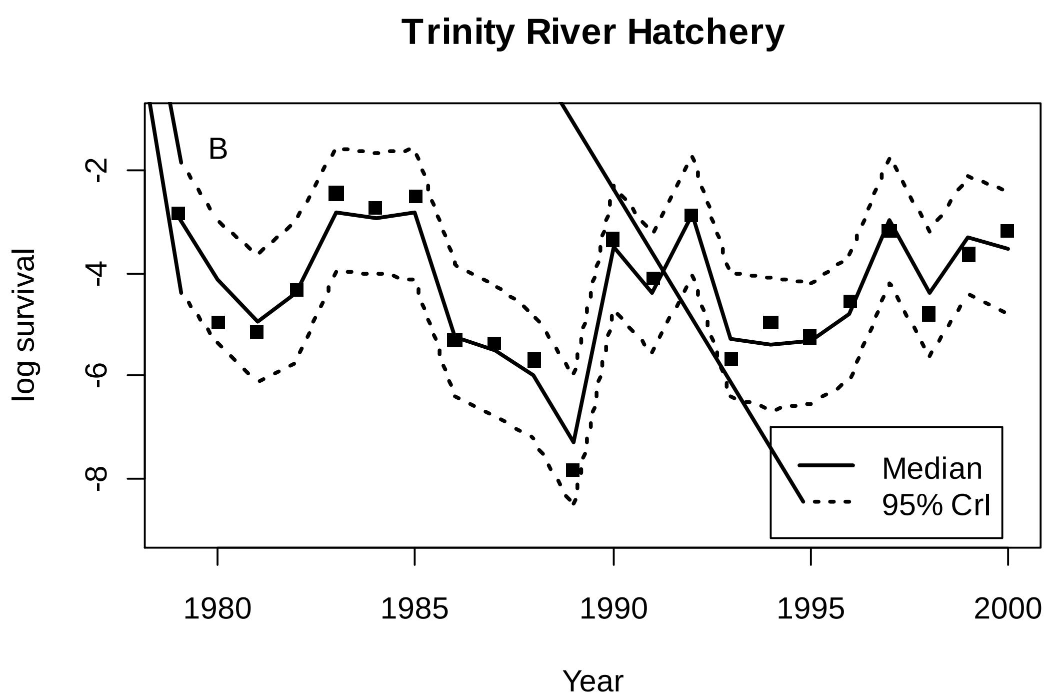


Figure 5. Observed log survival rates from Iron Gate Hatchery (A) and Trinity River Hatchery (B) with median model predictions and 95% credible intervals (95% CrI).

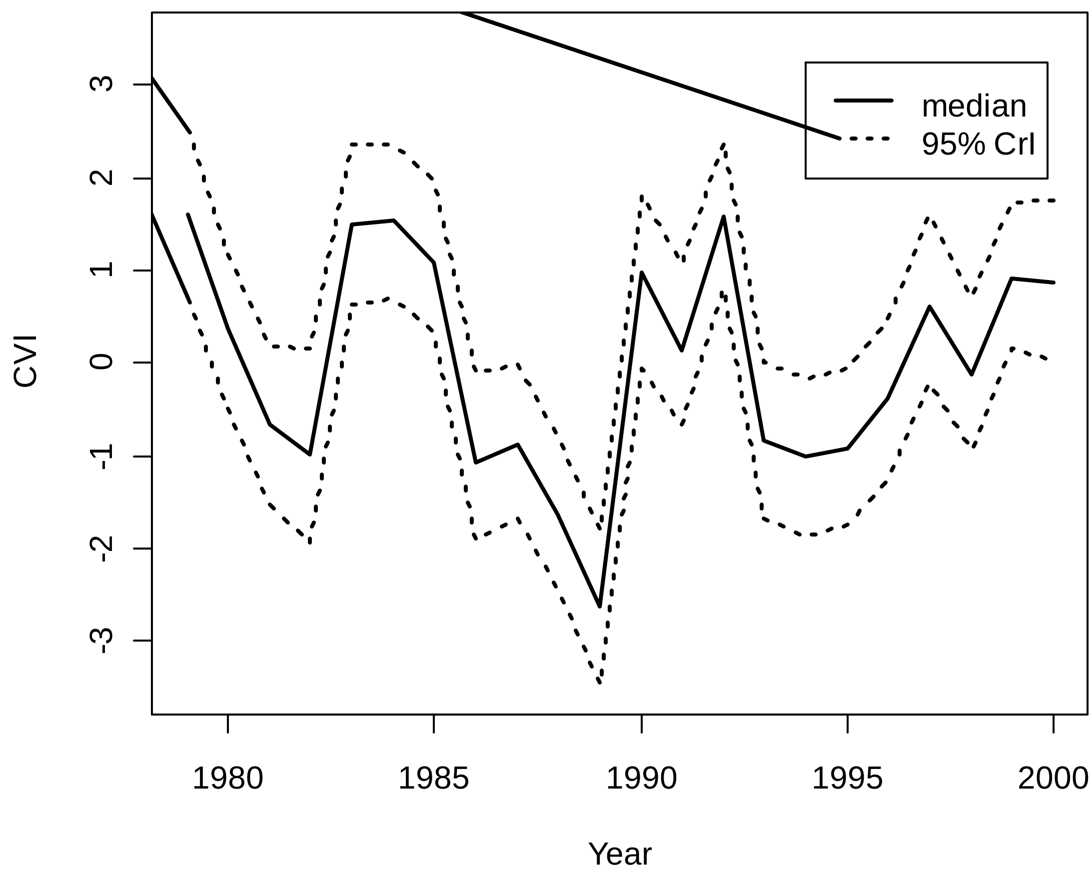


Figure 6. The common variability index (CVI) from 1979 to 2000, which is the annual variability common to both Iron Gate Hatchery and Trinity River Hatchery fingering CWT release groups estimated from log hatchery survival rates.

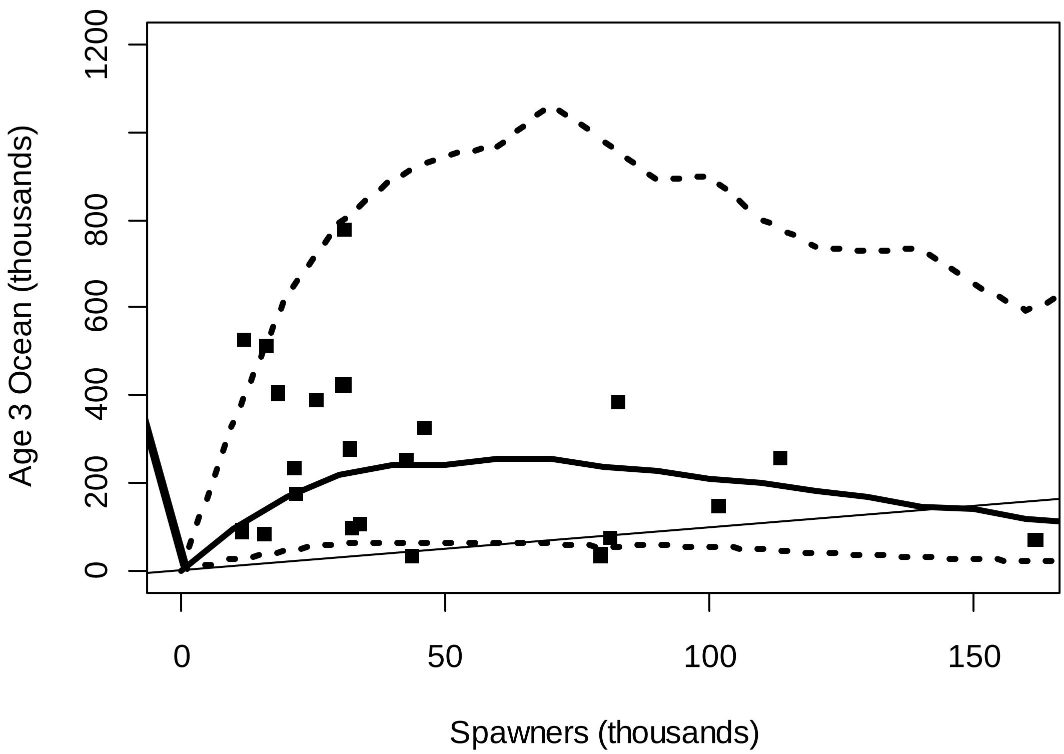


Figure 7. Estimated stock recruitment relationship between spawners and age 3 ocean abundance for brood years 1979 to 2000. Observed data (squares), median recruitment (dark solid line) and 95% credible interval (dashed lines), and the 1:1 line (thin solid line) are plotted. Model predictions assumed CVI equal 0.

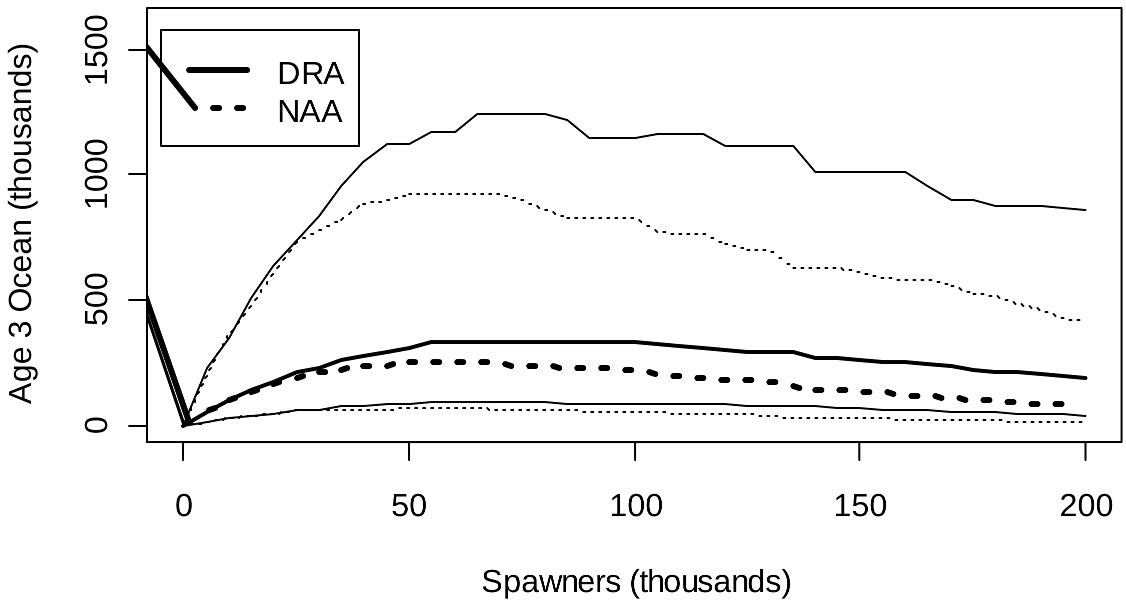


Figure 8. Lower Klamath Basin stock production relationship under the No Action Alternative (NAA) and under the Dam Removal Alternative (DRA). Median recruitment (dark lines) and 95% intervals (light lines) are plotted for production under the two alternatives. The DRA and NAA alternatives assume the same level of log productivity (α').

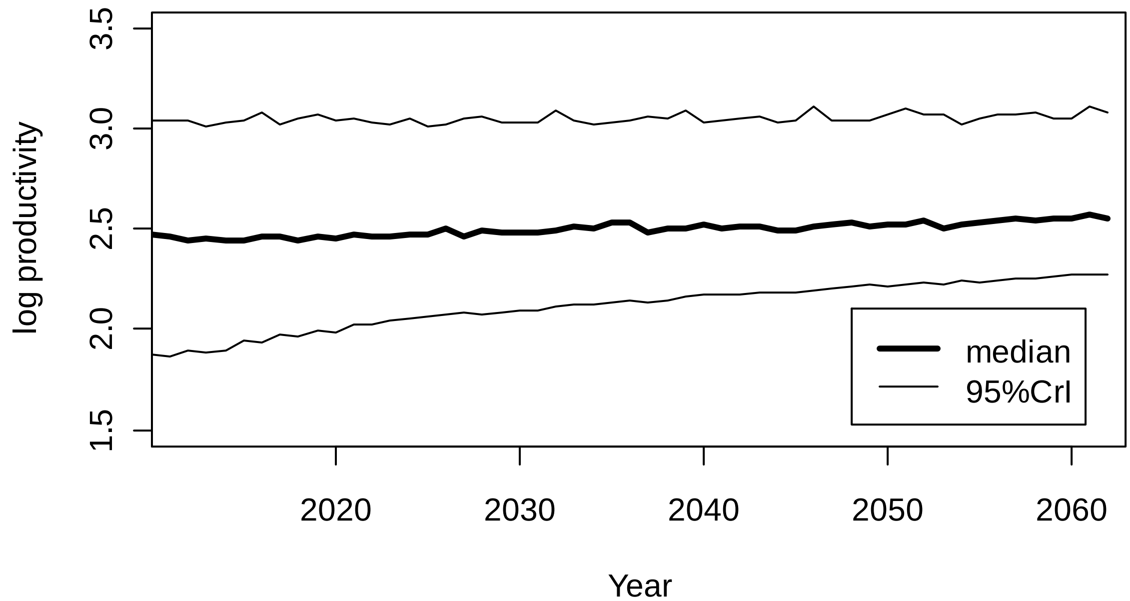


Figure 9. Distribution of .log productivity (α') in the lower Klamath Basin from 2012 to 2061 due to habitat restoration by KBRA.

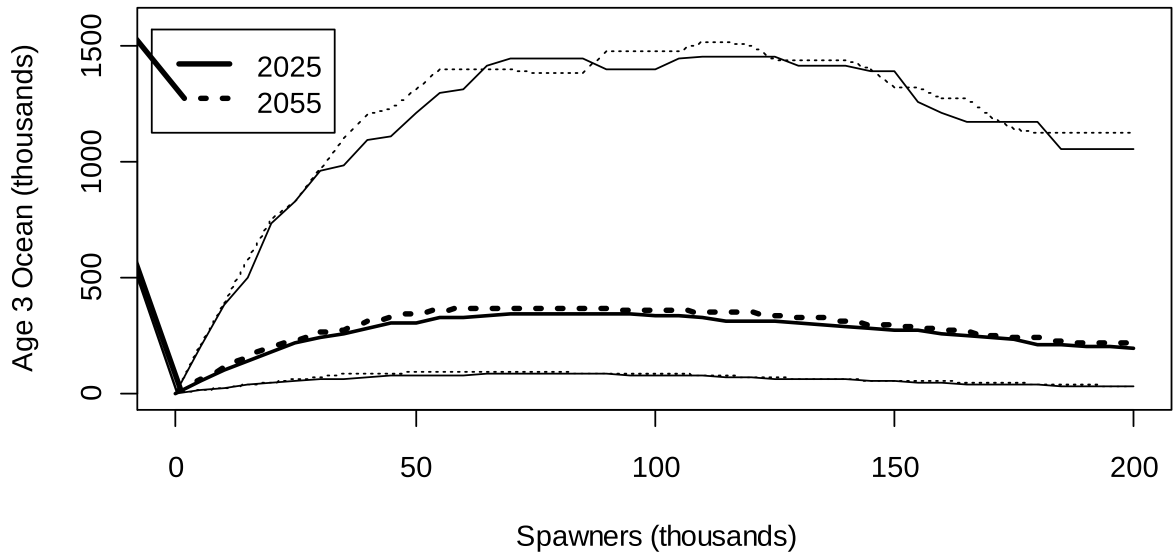


Figure 10. Stock recruitment relationship in the lower Klamath basin in 2025 and in 2055 including increase in habitat due to dam removal and KBRA actions affecting log productivity α'*. Median production (dark line) and 95%I (light line) are plotted for each of the two years.

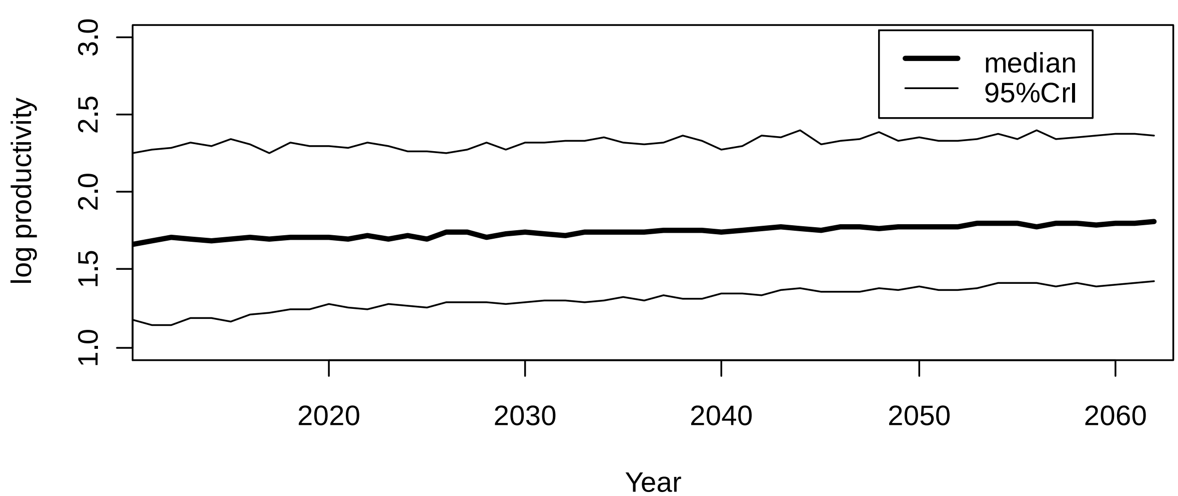


Figure 11. Distribution of log productivity (rnew*) of a mixed stream and ocean type life history in tributaries to Upper Klamath Basin from 2012 to 206. Changes in log productivity over the time series are due to habitat restoration by KBRA.

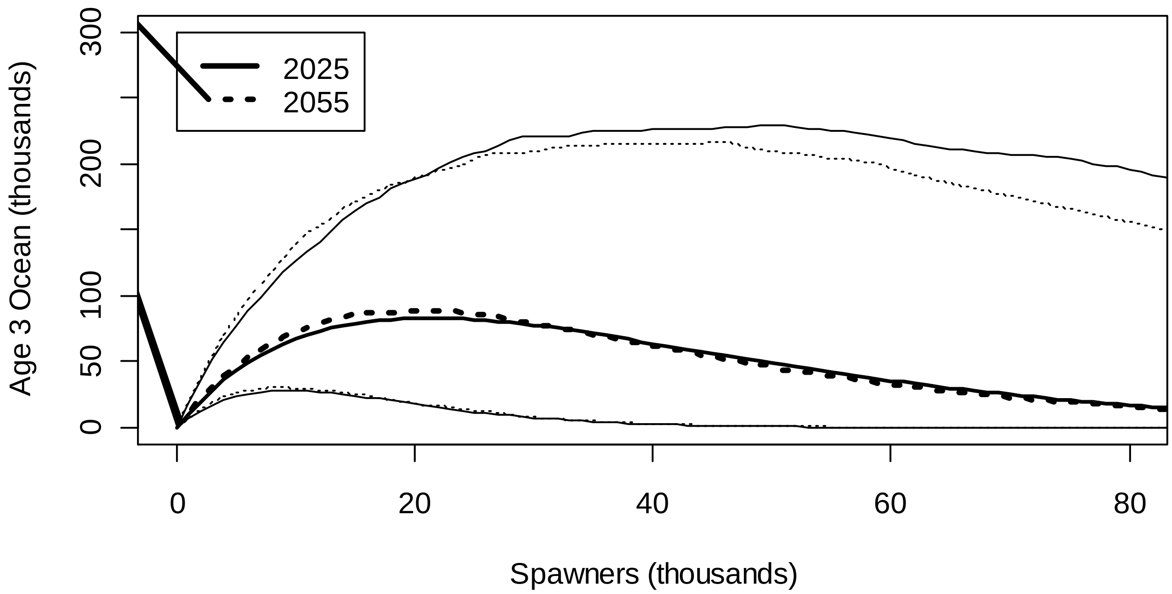


Figure 12. Stock recruitment relationship in the tributaries to Upper Klamath Lake in 2020 and in 2055 incorporating mixed stream and ocean type life history and KBRA actions affecting log productivity r_{new}^{*}. Median production (dark line) and 95%I (light line) are plotted for each of the two years.

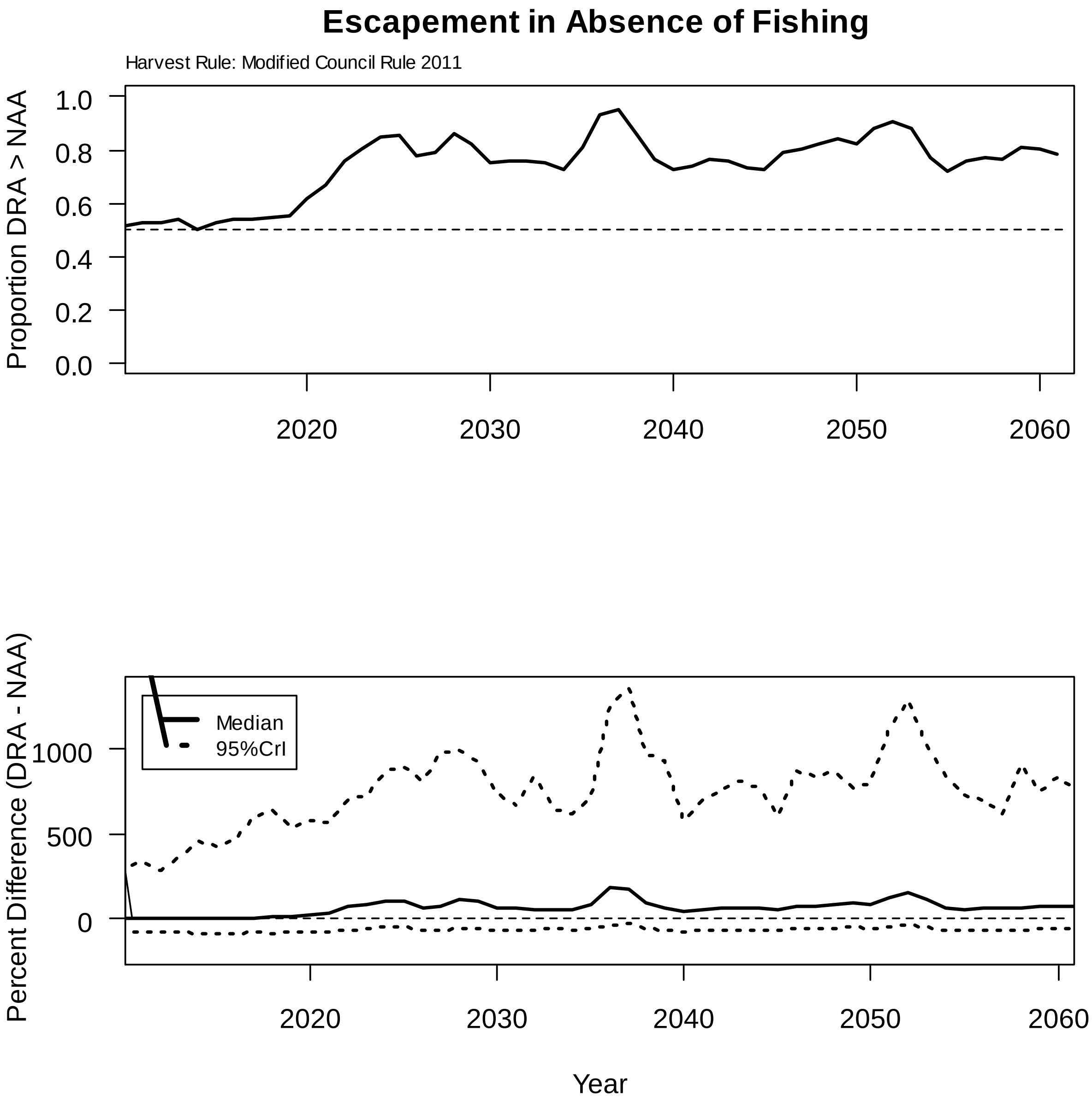


Figure 13. Probability that escapement in the absence of fishing is greater under DRA than under NAA from 2012 to 2061 (top). Dashed line represents 50/50 chance of increased abundance under DRA. Median and 95% credible intervals for the percent increase in DRA relative to NAA from 2012 to 2061 (bottom). Dashed line represents no difference between DRA and NAA.

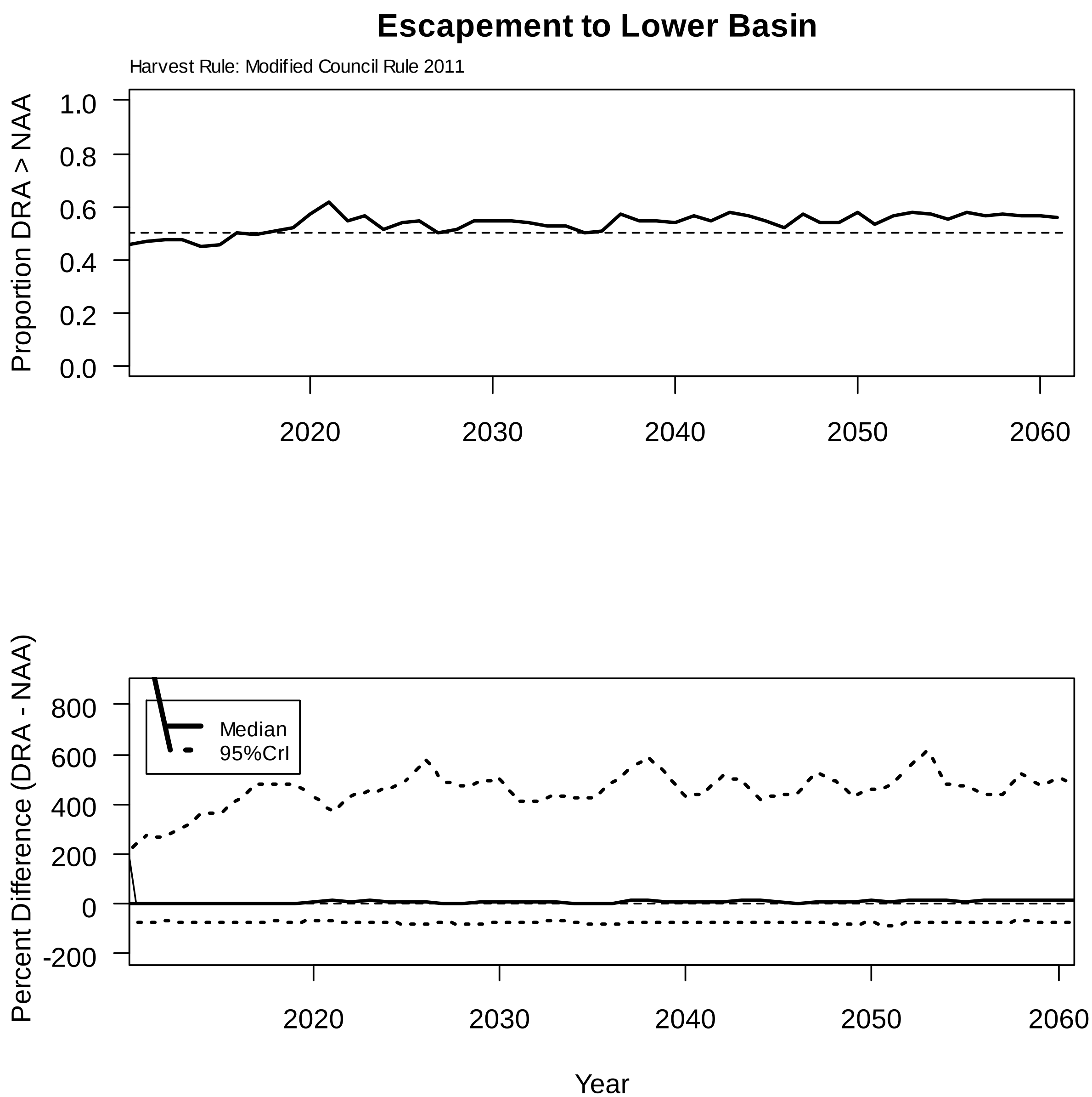


Figure 14. Probability that escapement to the Lower Klamath Basin is greater under DRA than under NAA from 2012 to 2061 (top). Dashed line represents 50/50 chance of increased abundance under DRA. Median and 95% credible intervals for the percent increase in DRA relative to NAA from 2012 to 2061 (bottom). Dashed line represents no difference between DRA and NAA.

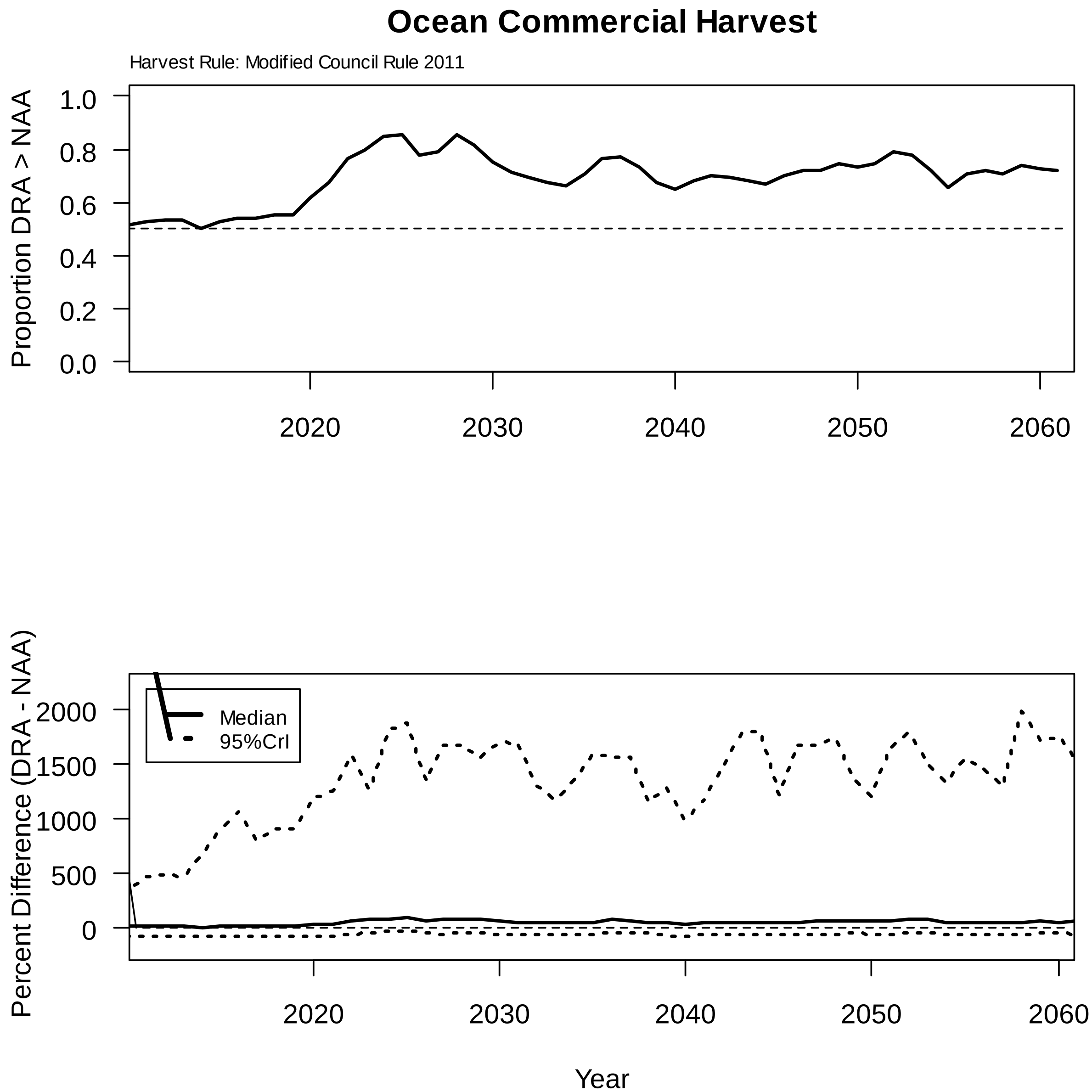


Figure 15. Probability that ocean commercial harvest is greater under DRA than under NAA from 2012 to 2061 (top). Dashed line represents 50/50 chance of increased abundance under DRA. Median and 95% credible intervals for the percent increase in DRA relative to NAA from 2012 to 2061 (bottom). Dashed line represents no difference between DRA and NAA.

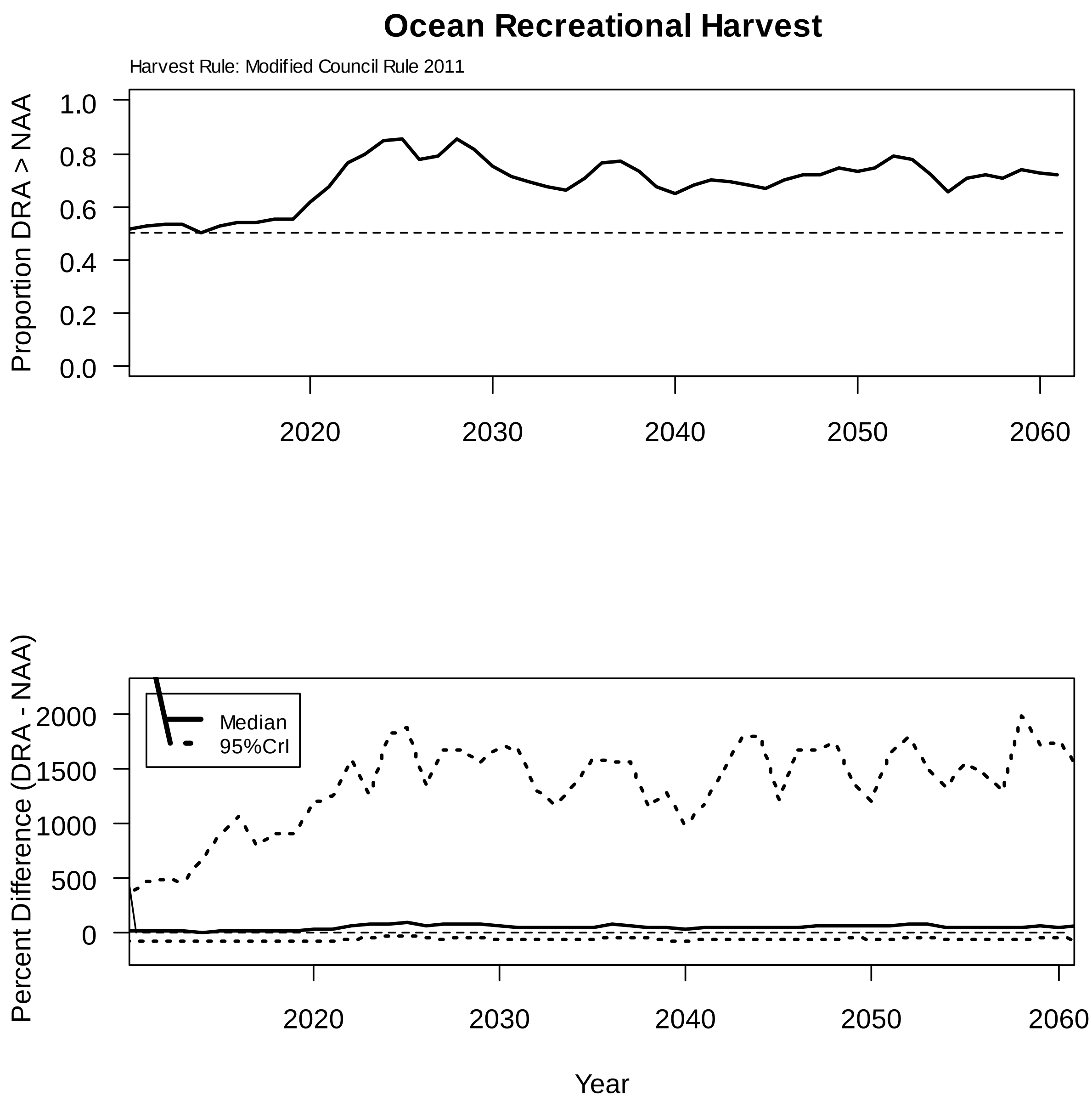


Figure 16. Probability that ocean recreational harvest is greater under DRA than under NAA from 2012 to 2061 (top). Dashed line represents 50/50 chance of increased abundance under DRA. Median and 95% credible intervals for the percent increase in DRA relative to NAA from 2012 to 2061 (bottom). Dashed line represents no difference between DRA and NAA.

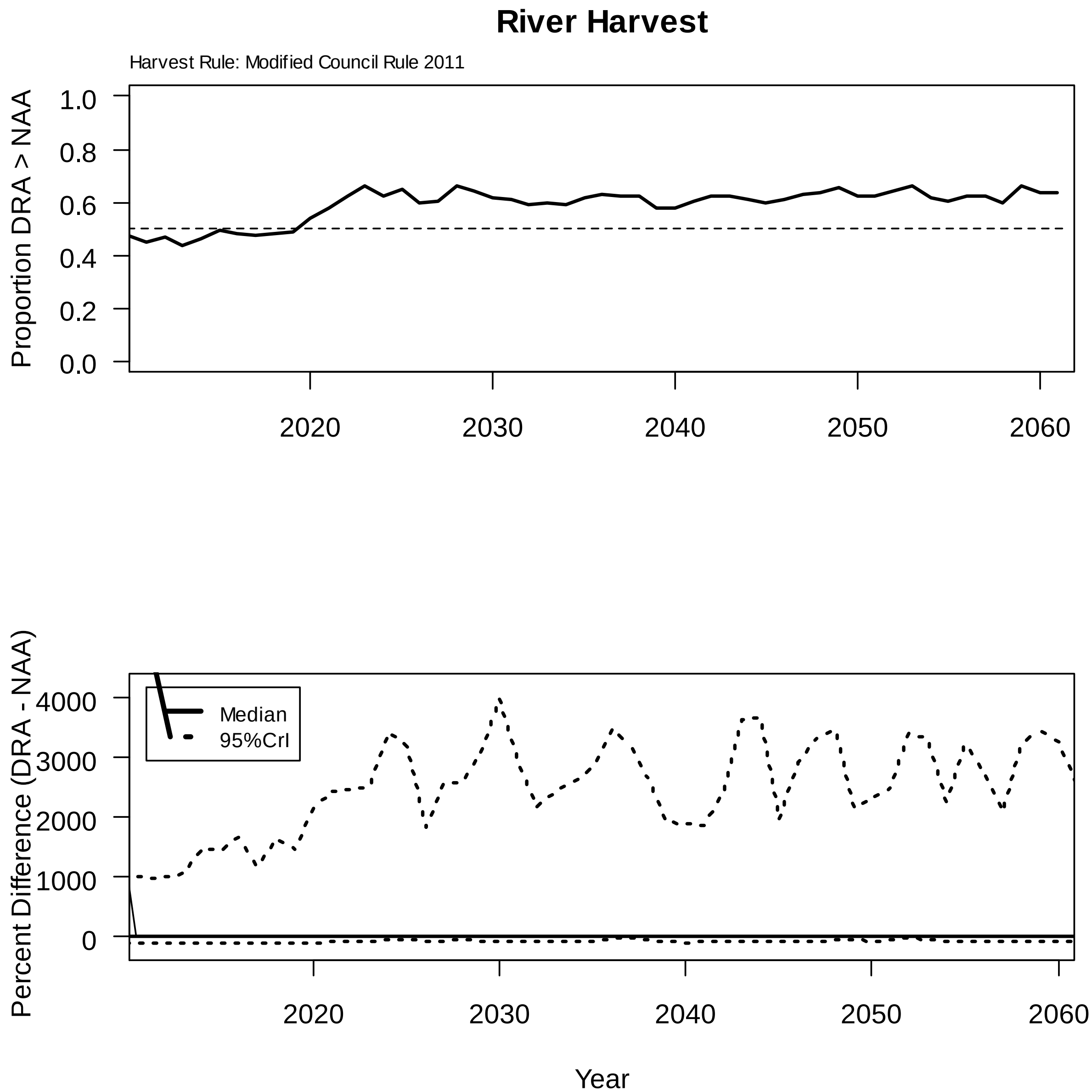


Figure 17. Probability that river harvest is greater under DRA than under NAA from 2012 to 2061 (top). Dashed line represents 50/50 chance of increased abundance under DRA. Median and 95% credible intervals for the percent increase in DRA relative to NAA from 2012 to 2061 (bottom). Dashed line represents no difference between DRA and NAA.

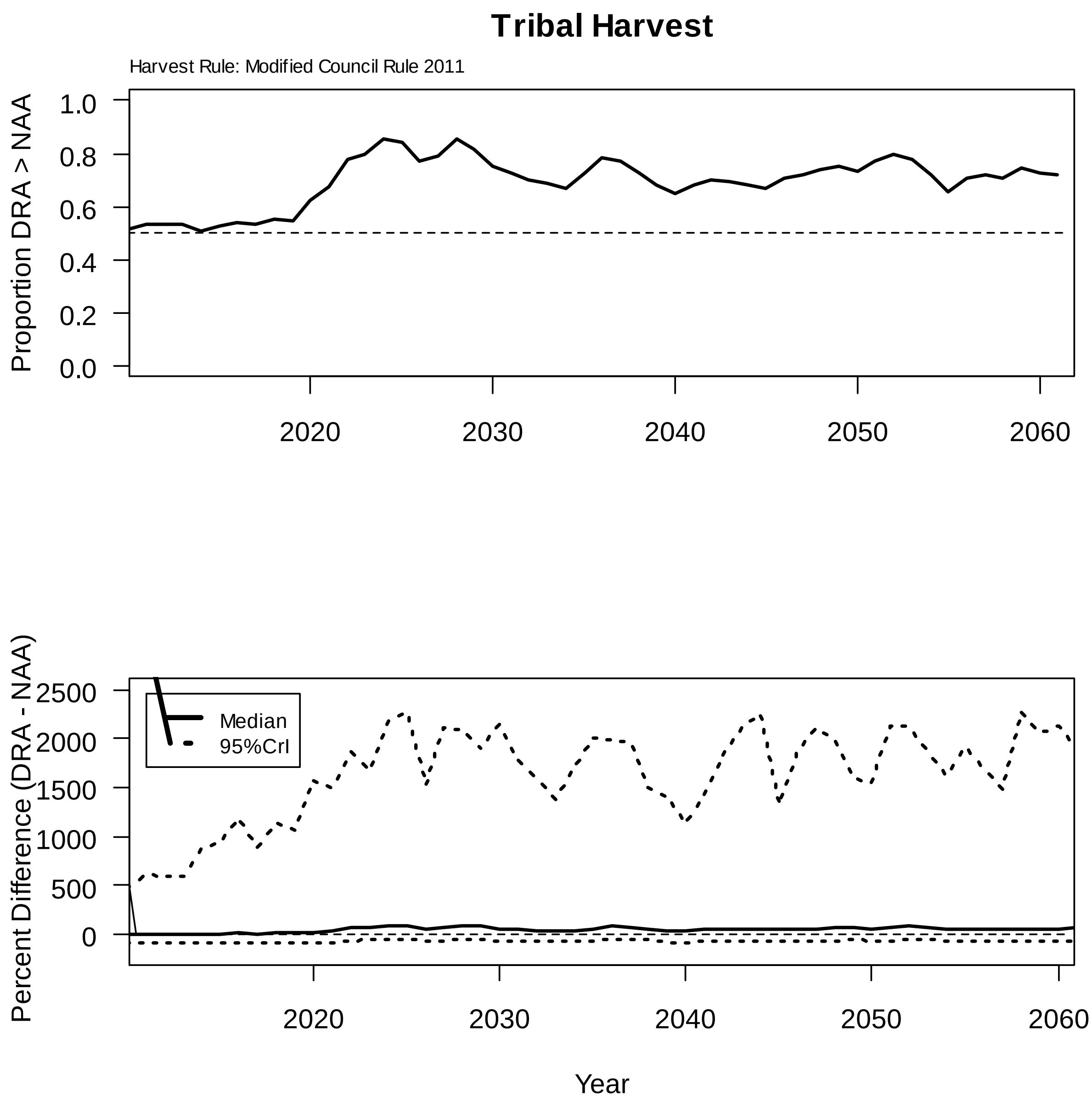


Figure 18. Probability that tribal harvest is greater under DRA than under NAA from 2012 to 2061 (top). Dashed line represents 50/50 chance of increased abundance under DRA. Median and 95% credible intervals for the percent increase in DRA relative to NAA from 2012 to 2061 (bottom). Dashed line represents no difference between DRA and NAA.

APPENDIX A. FISHERY CONTROL RULE APPLIED IN THE KLAMATH HARVEST RATE MODEL

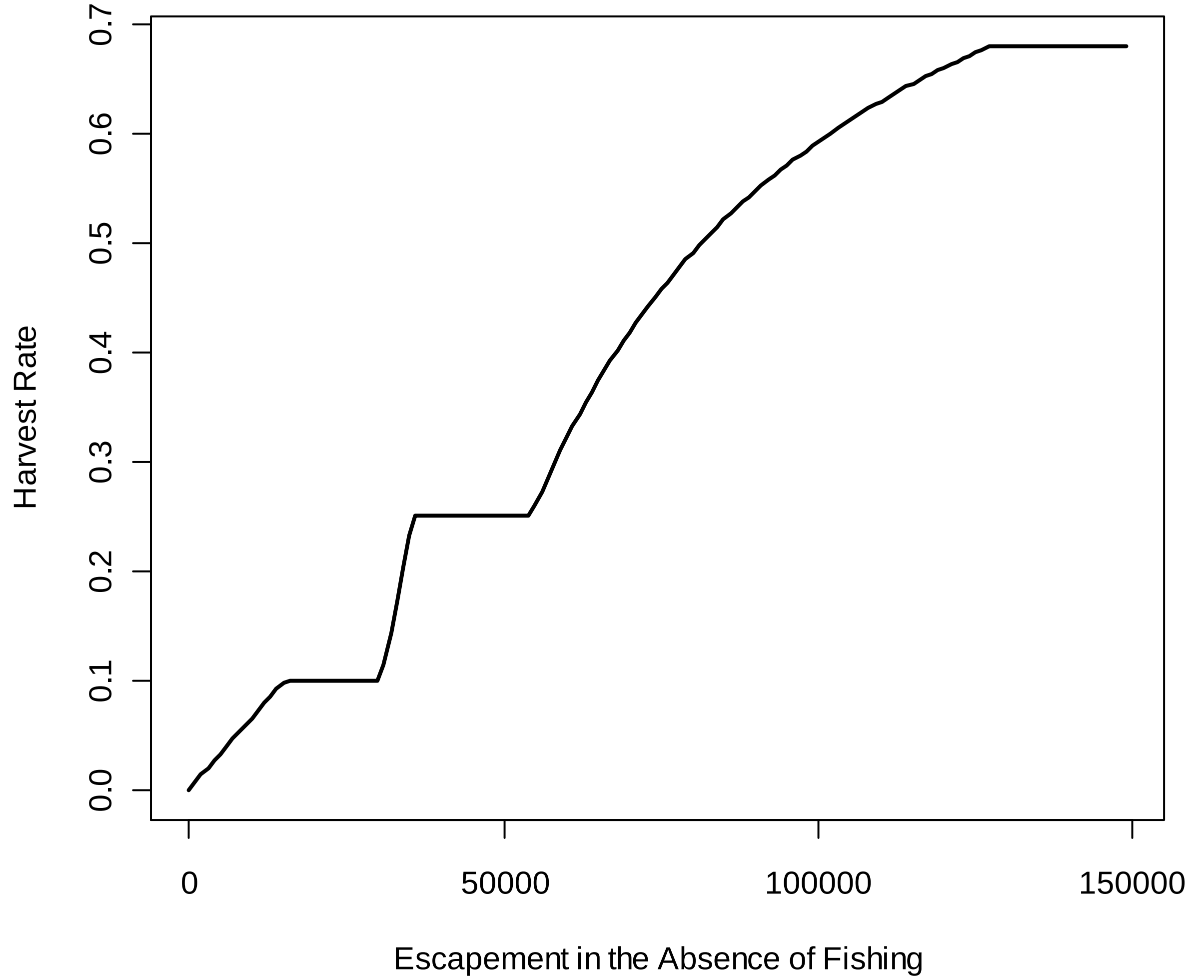


Figure A2.1. Harvest rate as a function of escapement in the absence of fishing utilized in the Klamath Harvest Rate Model (Mohr et al. in Prep).

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*(mean, 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 5in 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, $\sigma_{CVI}{}^2$
    e. The values of CVI for each year of the time series, $CVI_{2007:2061}$~$N(0, \sigma_{CVI}{}^2)$
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 clairity)

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 .

$$s_{IGH,t} = \exp\{h_{IGH,t}\} = \exp\{\kappa_{IGH} + CVI_{t-2} + \gamma_{IGH}(Q_{KR\_BO,t-2}) + u_{IGH,t}\} \quad \text{Equation (A2)}$$

3. Calculate the survival rate of TRH releases for brood year t, $s_{TRH,}$ using Trinity River flows

$$s_{TRH,t} = \exp\{h_{TRH,t}\} = \exp\{\kappa_{TRH} + CVI_{t-2} + \gamma_{TRH}(Q_{TR,t-2}) + u_{TRH,t}\} \quad \text{Equation (A3)}$$

4. Calculate the hatchery production to age 3 assuming age 2 survival of 0.5 (Hankin and Logan 2010) for year t

$$N_{H,t} = 0.5(s_{IGH,t}\, 6e06 + s_{TRH,t}\, 3e06) \quad \text{Equation (A4)}$$

5. Calculate the total abundance of year 3 ocean fish

$$N_{3,t} = R_t + N_{H,t} \quad \text{Equation (A5)}$$

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 5in 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, $\mathrm{CVI}_{2010:2020}$
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, $\mathrm{CVI}_{2021:2061}$
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, …, 2061
   a. Unfished equilibrium population size, $E_{new,\ t}$
      i. Draw a value of $p_{,t}$ from a Uniform(0,1) distribution in year t
      ii. Sample from the distribution of $\mathrm{E}_{\text{new stream}}$ using the watershed size of 4200.96 km^2
      iii. Sample from the distribution of $\mathrm{E}_{\text{new ocean}}$ using the watershed size of 4200.96 km^2
      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 clairity)

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_{TRHt} 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}.

$$R_t = S_{t-3}\exp\{\alpha'^* - \beta_{new}S_{t-3} + \delta CVI_{t-2} + \epsilon_t\} \qquad \text{Equation (A7)}$$

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.

$$R_{UKL,t} = S_{UKL,t-3}\exp\left\{r.new^*\left(1 - \frac{S_{t-3}}{E_{new,t}}\right) + \delta CVI_{t-2} + \gamma_{IGH}Q_{KBRA,t-2}\right\}CF \qquad \text{Equation (A8)}$$

4. If year t > 2028 IGH ceases to produce fall Chinook and hatchery production consists of TRH fish only of 3 million

$$N_{h,t} = 0.5(s_{TRH,t}\, 3e06) \qquad \text{Equation (A9)}$$

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_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

APPENDIX MS3. IMPACTS OF DEPLETING FORAGE SPECIES IN THE CALIFORNIA CURRENT

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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 moralities 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 ($\mathbf{B}$)—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 (*Micropogonia 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, euphausids, 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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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 (*Sebastes jordani*)	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

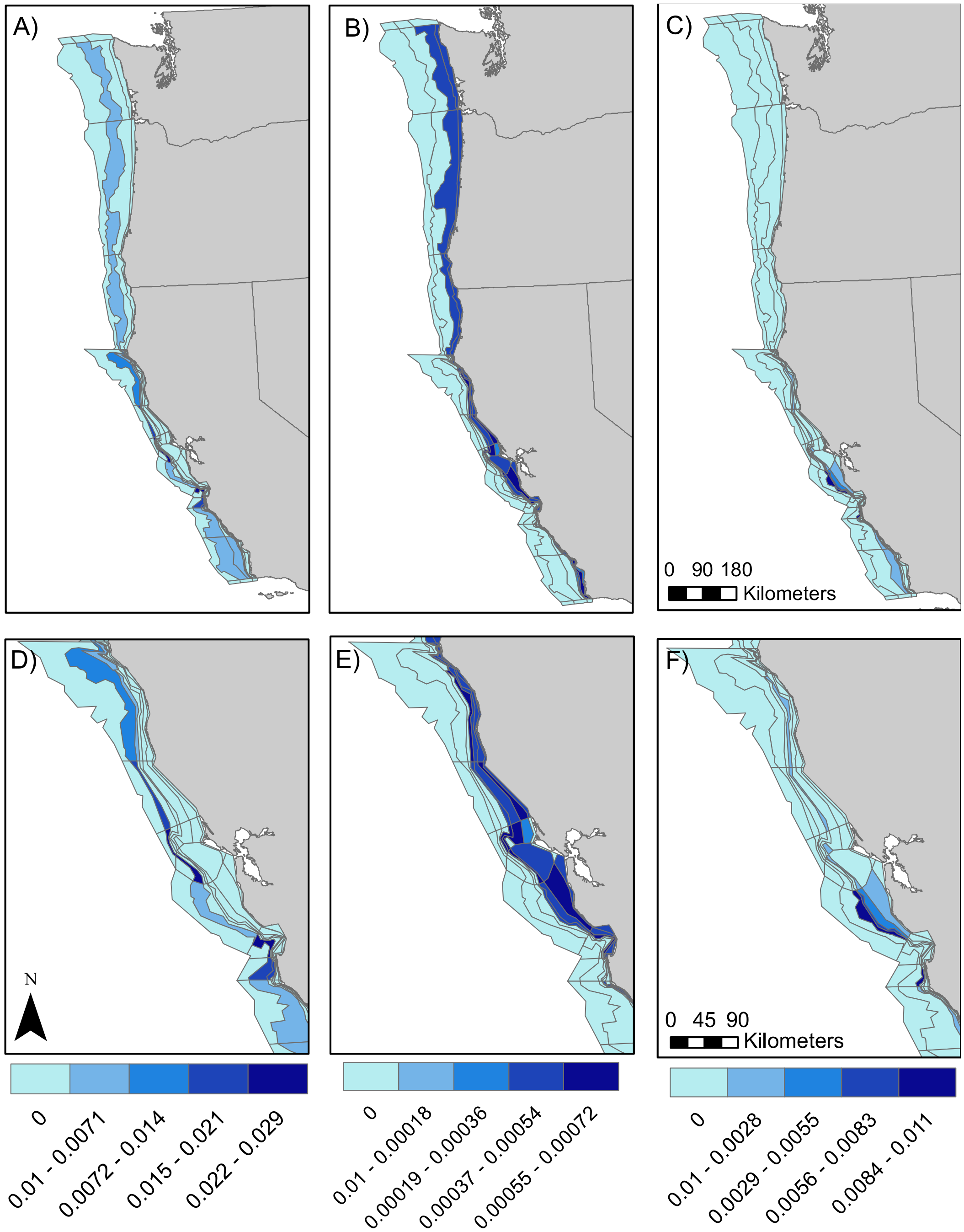

Figure 1. Status quo distribution of new potential target groups (deep demersal fish, nearshore miscellaneous fish, and shortbelly rockfish). Top panels illustrate distribution in the full model domain (A, B, C). Bottom panels show distribution within Central California region (D, E, F). Deep demersal 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/m^2.

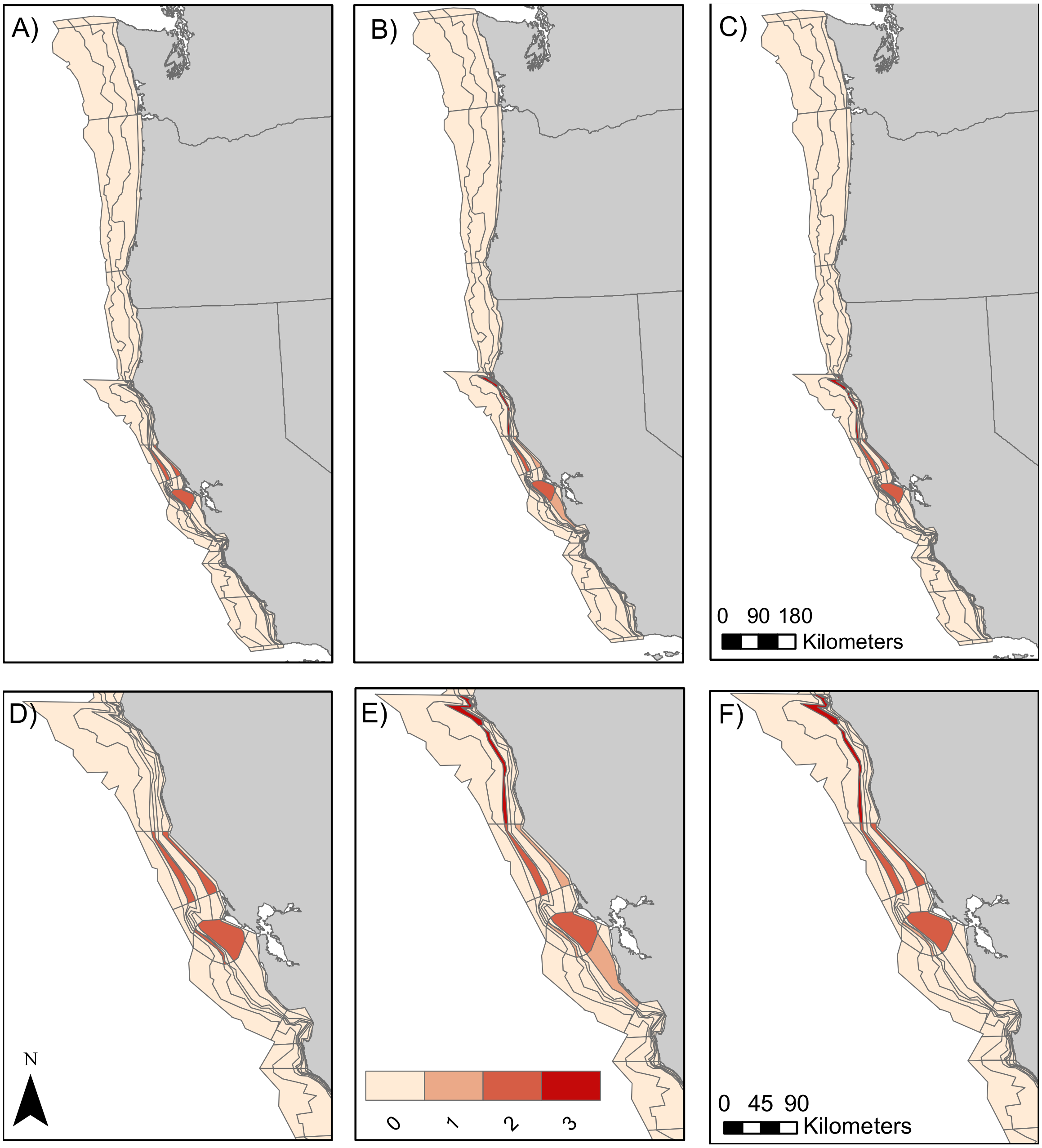

Figure 2. Number of functional groups affected by a fishing deep demersal fish at three fishing levels (threshold of 10 percent change) by cell. Fishing scenarios represented are F75 (A, D), F40 (B, E), and F25 (C, F). Density of color indicates increasing number of functional groups affected, as indicated by legend. Top and bottom panel extents as in Figure 1.

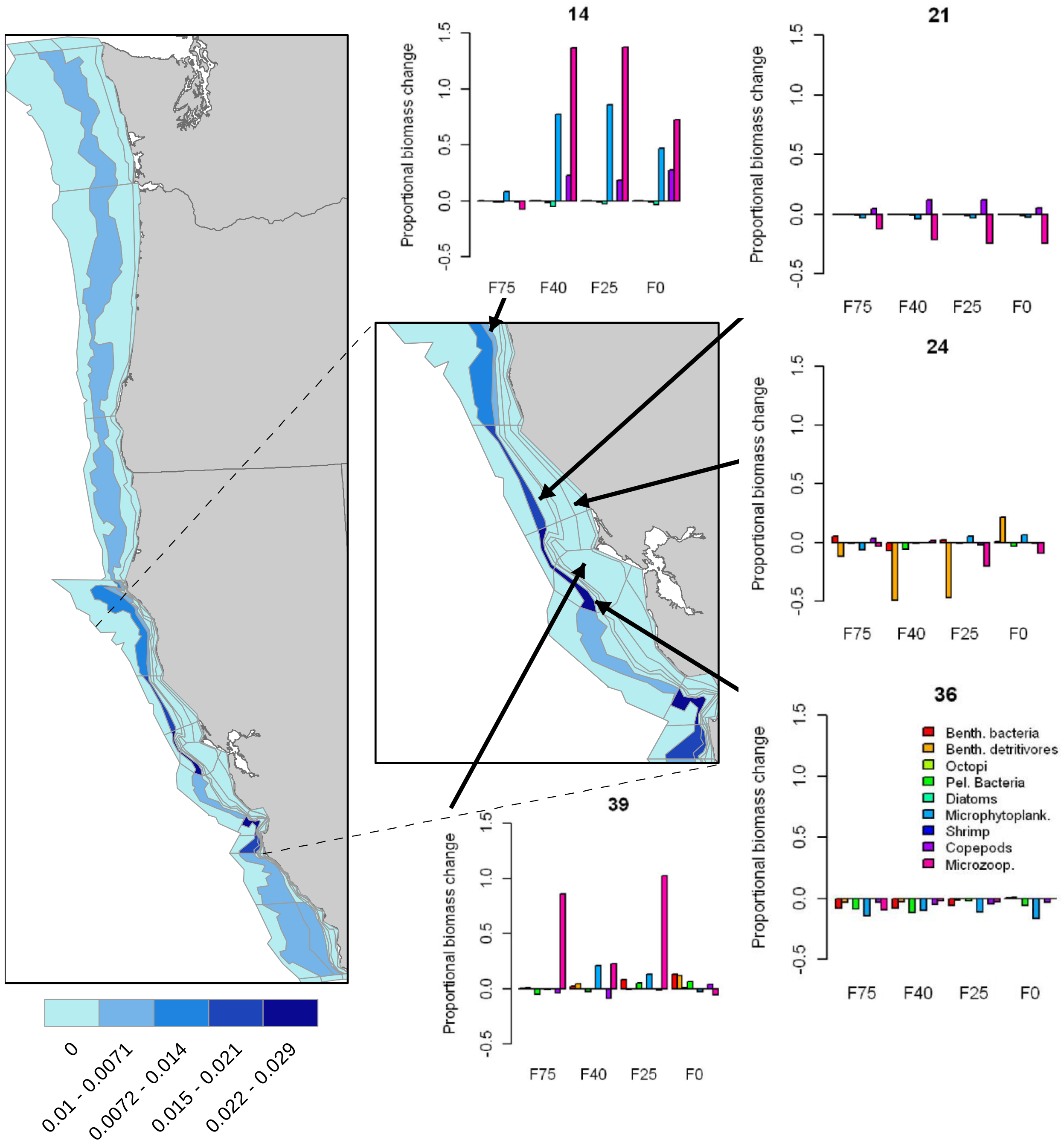

Figure 3. Effects of fishery targeting deep demersal fish on proportional biomass of other functional groups by cell. Map shading indicates deep demersal fish density, as in Figure 1. Nine functional groups were affected by at least one fishing scenario in at least one box across all three target species. For consistency, all nine groups are shown in all panels regardless of level of impact.

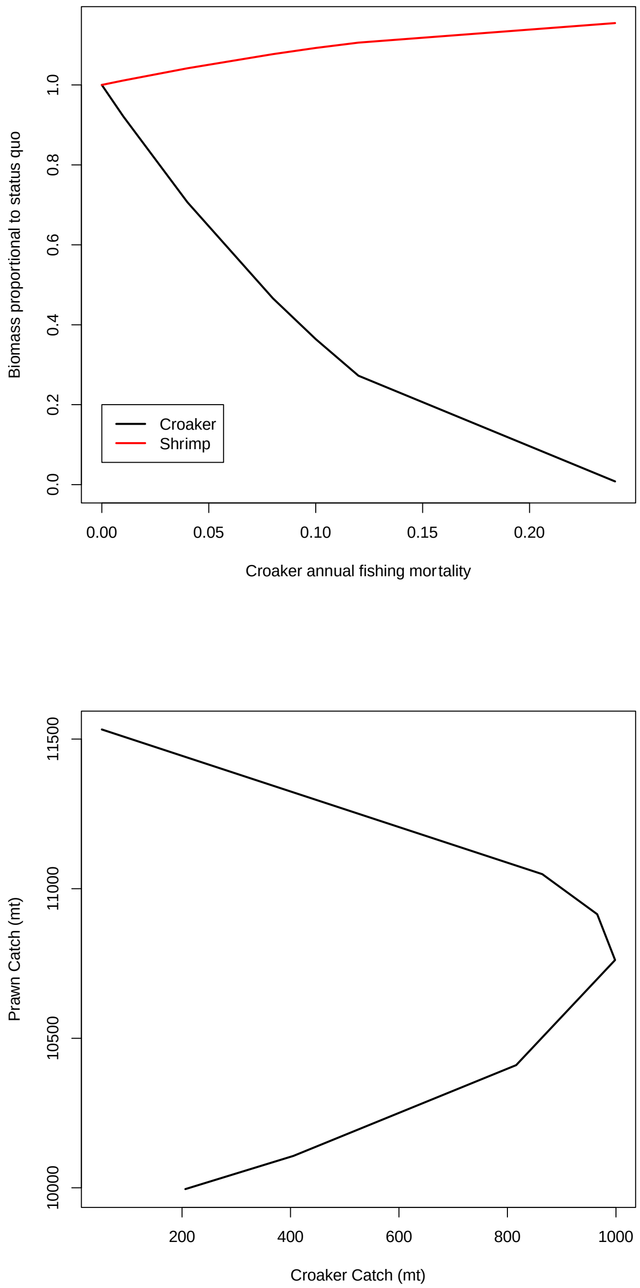

Figure 4. Changes in biomass and yield as a function of increased fishing mortality on nearshore miscellaneous fish (mostly white croaker). Shrimp biomass increased about 15 percent as white croaker biomass declined with fishing (A). Shrimp catches increased as croaker catches increased to MSY (B). Shrimp catches continued to increase as croaker catch declined and the population became depleted.

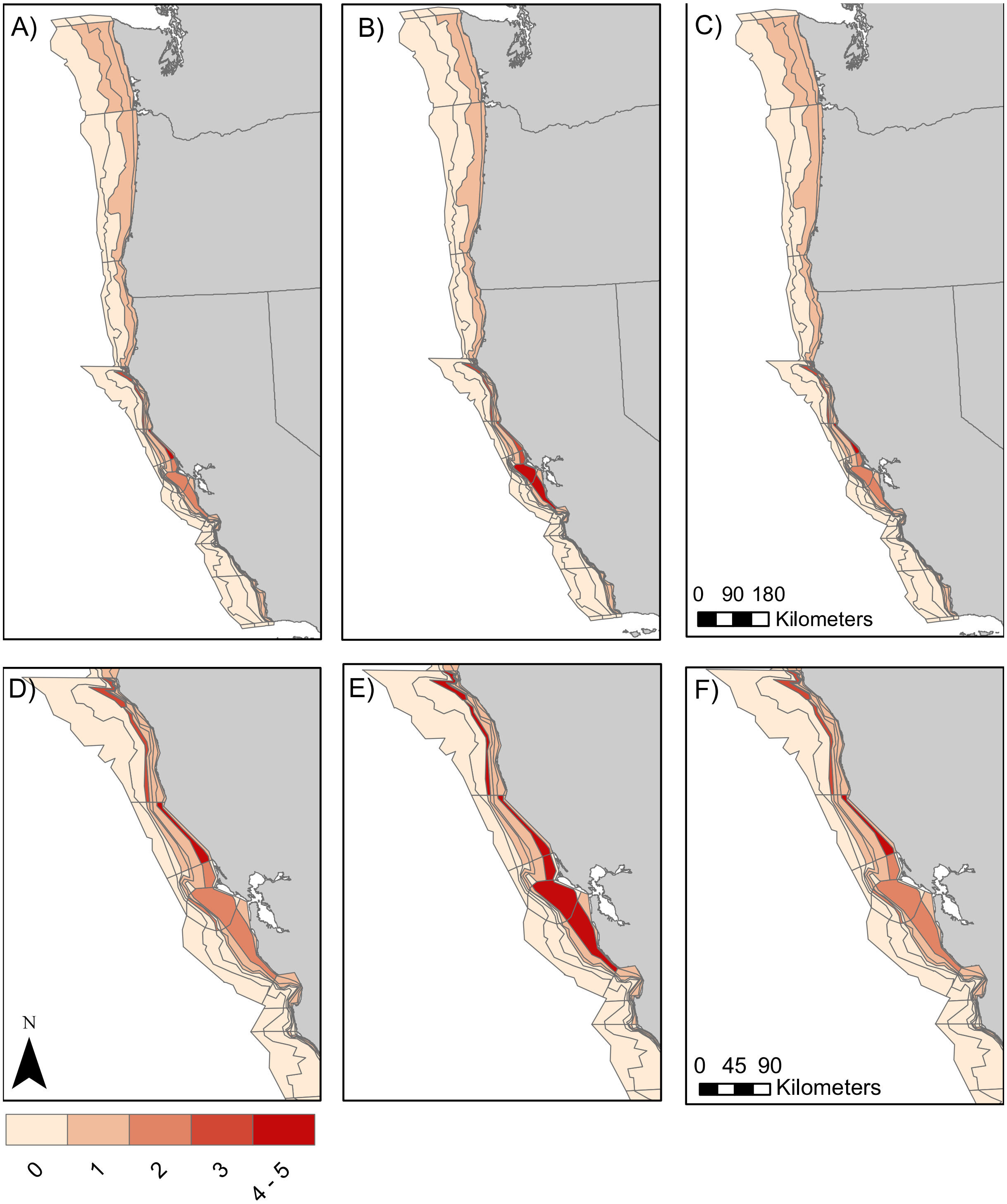

Figure 5. Number of functional groups affected (threshold +/- 10 percent) by introducing a fishery targeting the nearshore miscellaneous fish group, by cell. Fishing scenarios represented are F75 (A, D), F40 (B, E), and F25 (C, F). Density of color indicates increasing number of functional groups affected, as indicated by legend. Top and bottom panel extents as in Figure 1.

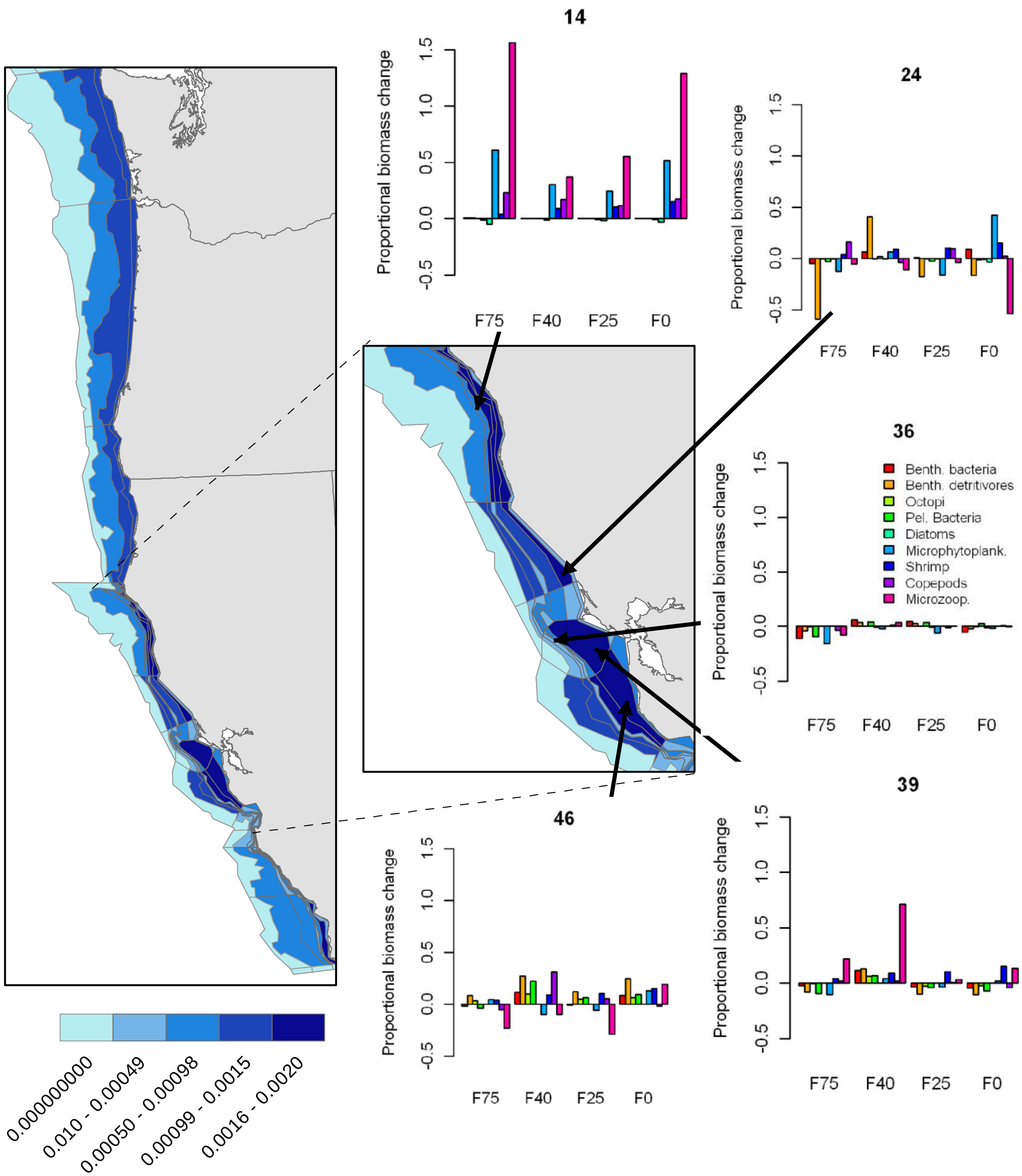

Figure 6. Effects of fishery on nearshore miscellaneous demersal fish group on functional groups by cell for cells with more than two functional groups affected. Maps show summed densities for the target fish group and prawn in status quo scenario. Surrounding plots indicate cascading effects were more common in cells with high densities of both shrimp and the targeted group. Bar coloring is consistent with Figure 4.

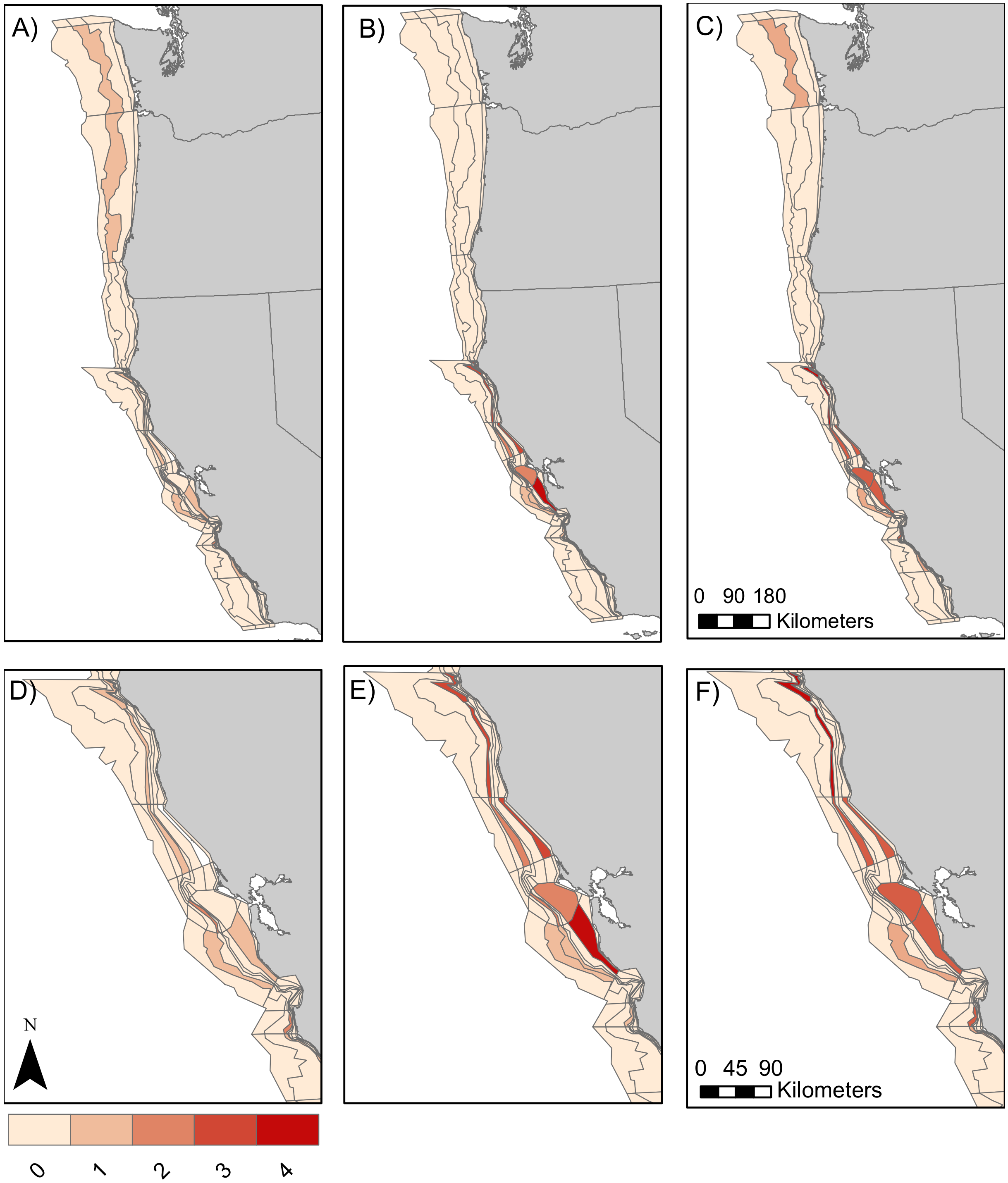

Figure 7. Number of functional groups affected (threshold +/- 10 percent) by introducing a shortbelly rockfish pacific fishery, by cell. Fishing scenarios represented are F75 (A, D), F40 (B, E), and F25 (C, F). Density of color indicates increasing number of functional groups affected, as indicated by legend. Top and bottom panel extents as in Figure 1.

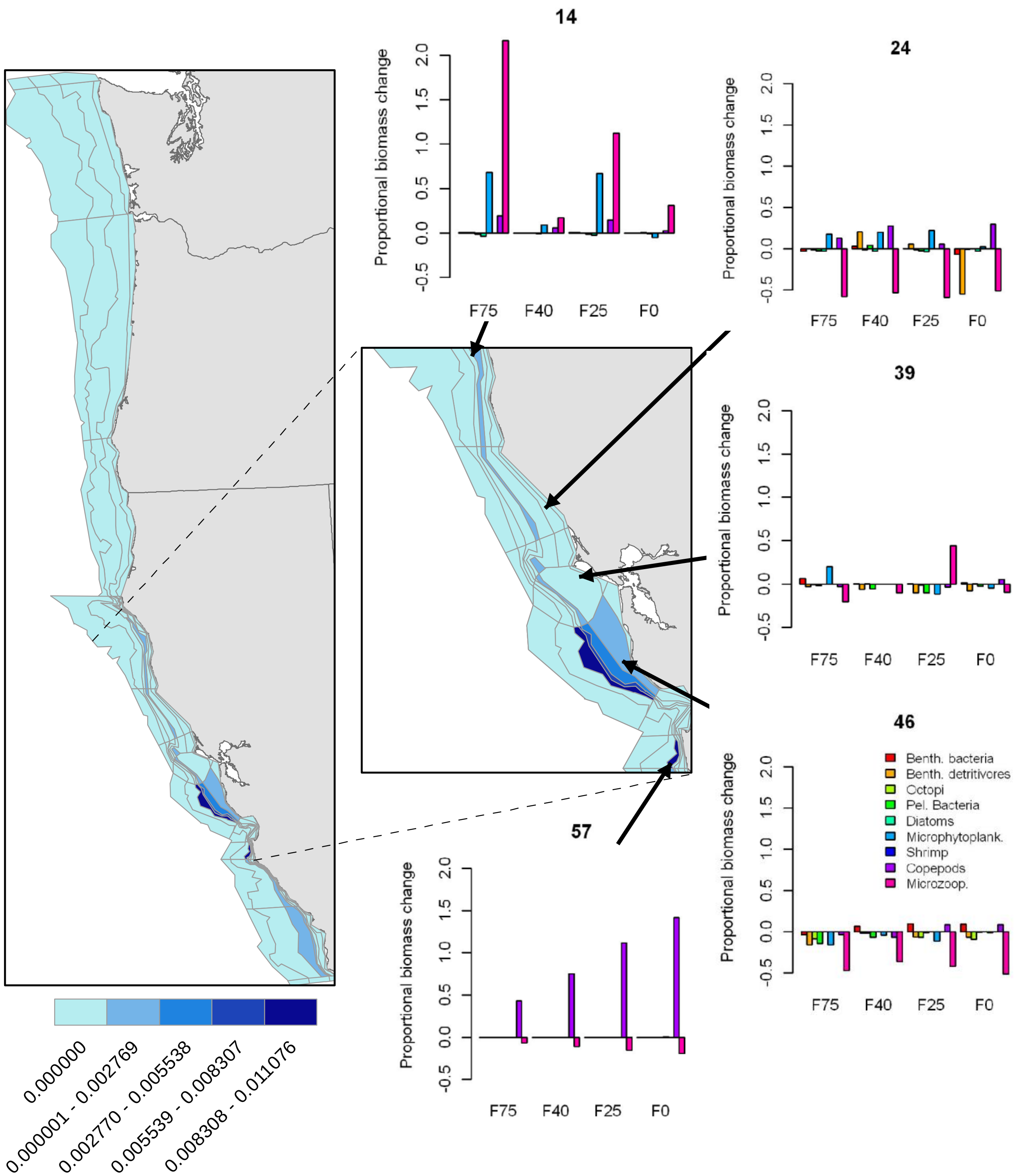

Figure 8. Effects of shortbelly rockfish fishery on proportional biomass of functional groups for cells with two or more impacted function groups. Maps indicate shortbelly density in status quo scenario as in Figure 1. Bar coloring consistent with Figure 4.

APPENDIX A: FINDING MSY/FISHING SCENARIOS

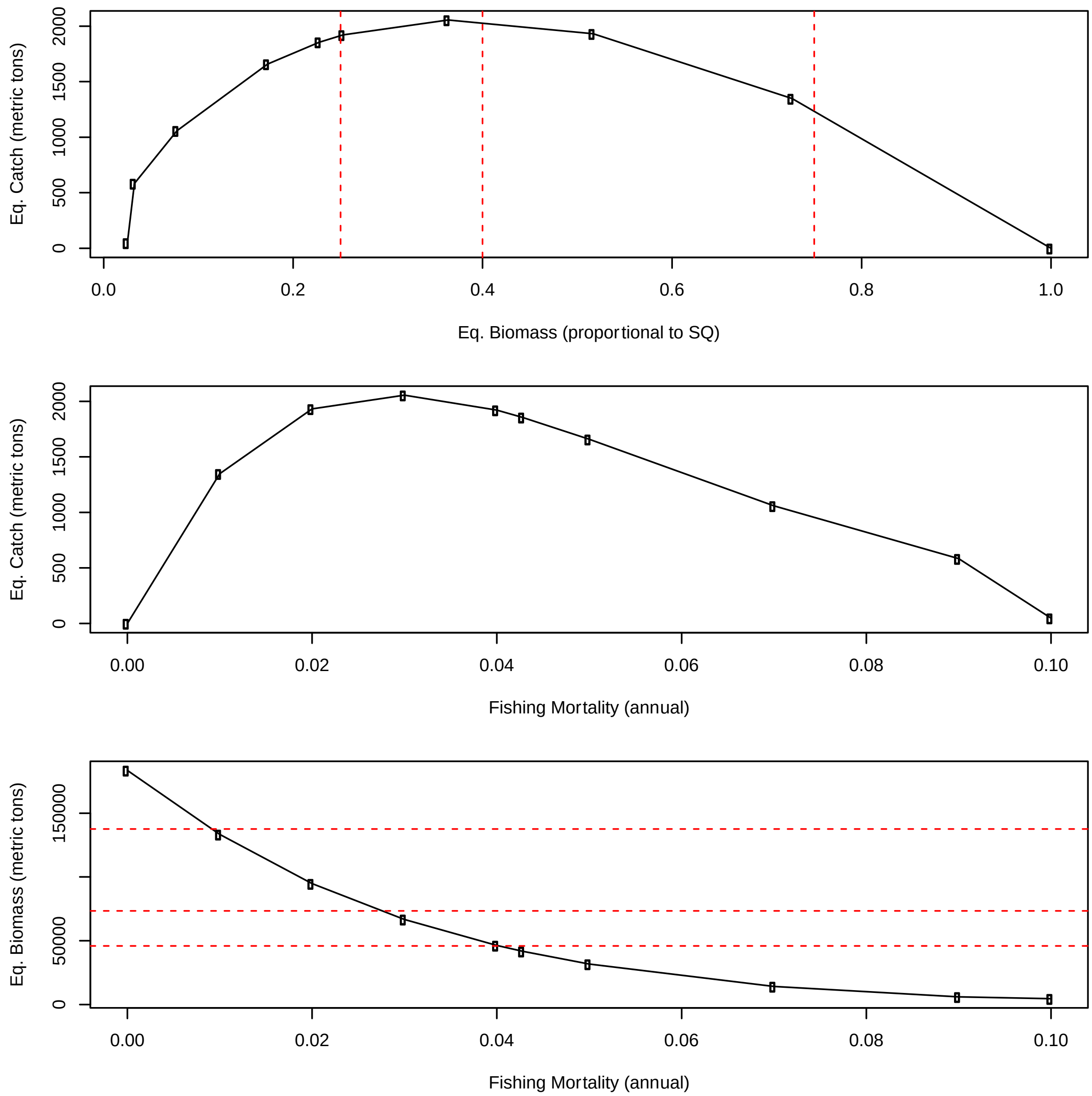

Figure A1. Fishing scenarios for deep demersal fish. Top panel shows equilibrium catch as a function of biomass proportional to the status quo scenario equilibrium. Vertical red lines indicate catch at 25, 40, and 75 percent of status quo. Middle panel shows the relationship between catch and fishing 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.

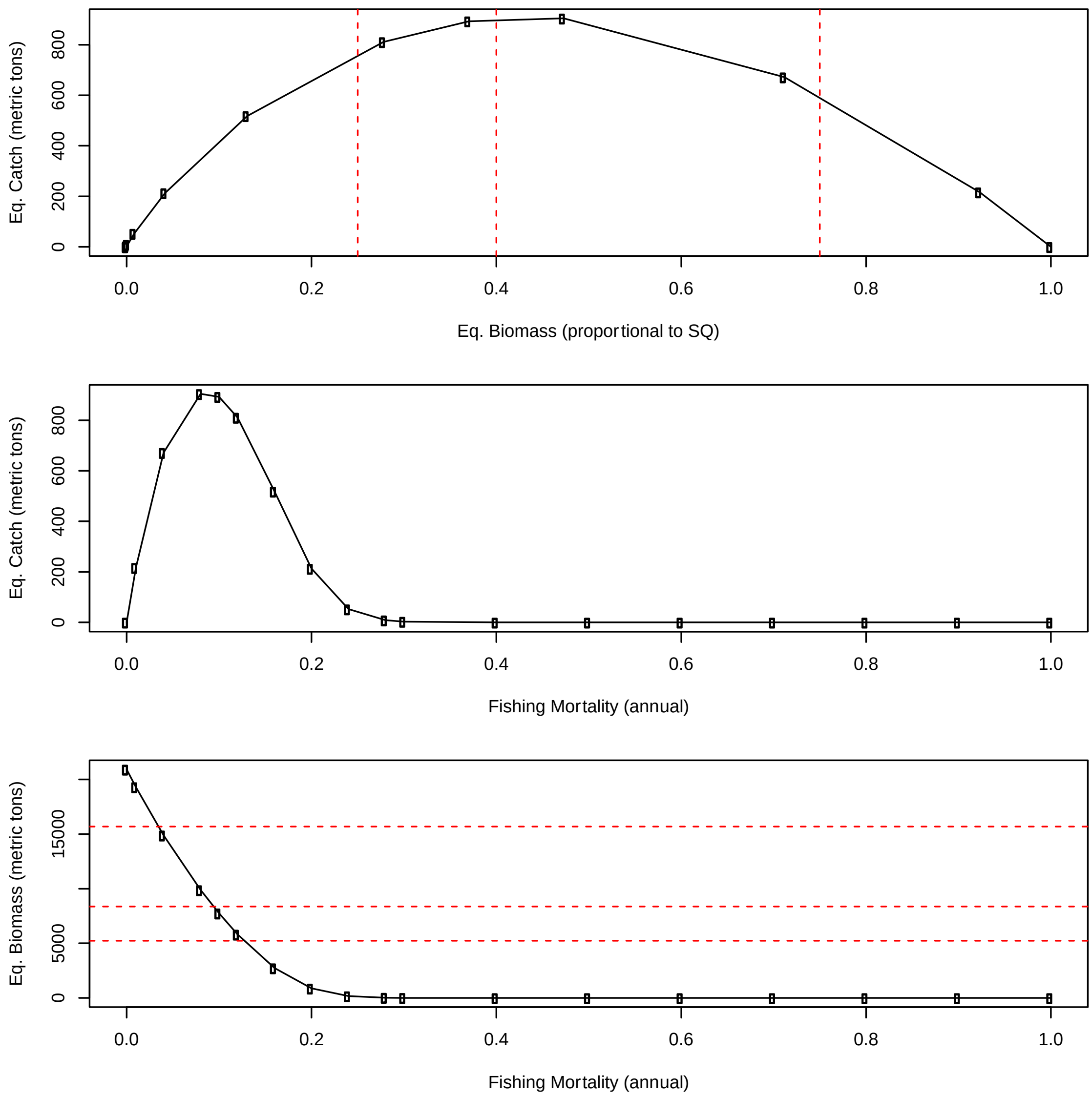

Figure A2. Fishing scenarios for nearshore miscellaneous demersal fish. Panels and axes as in Figure A1.

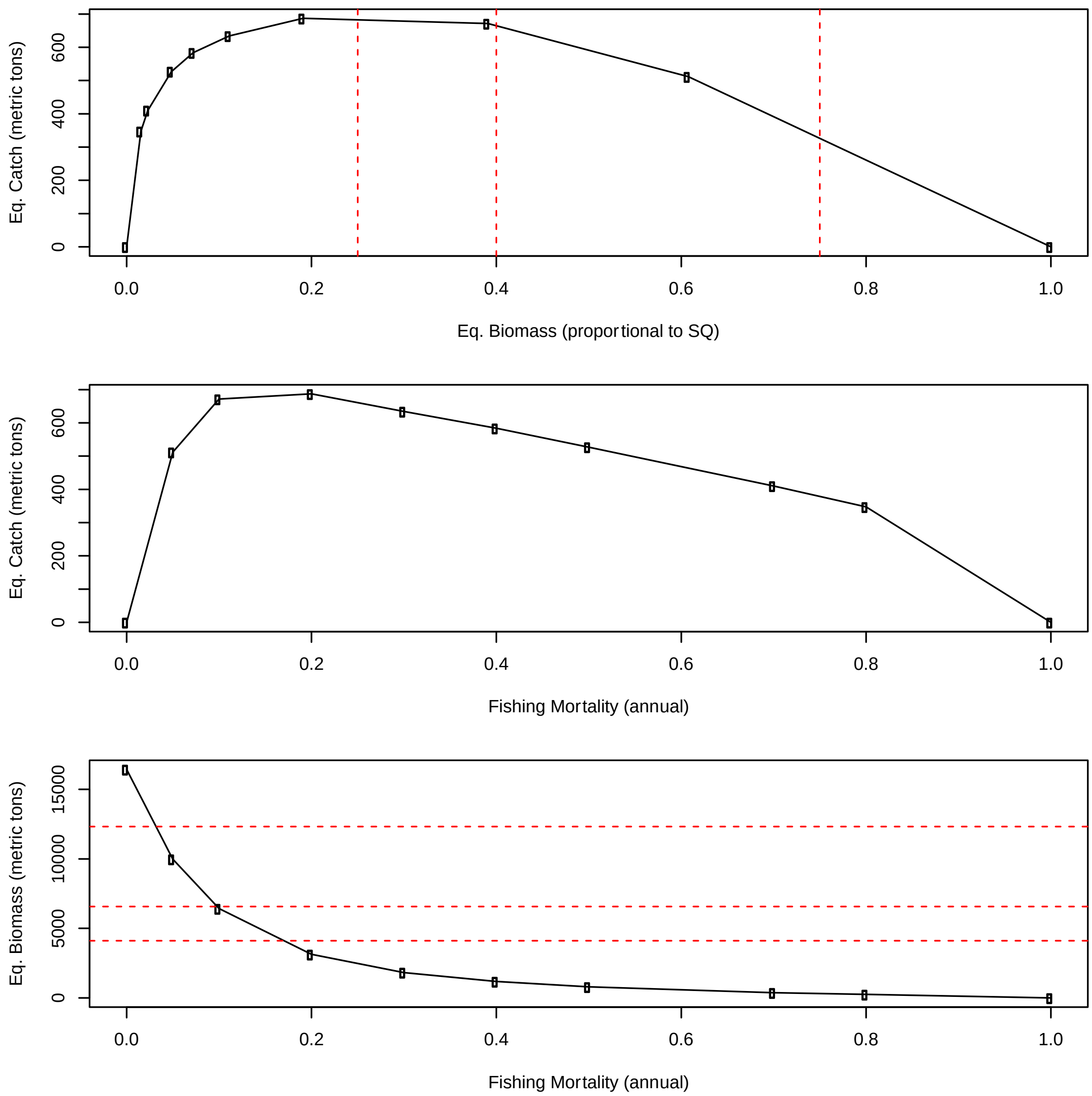

Figure A3. Fishing scenarios for shortbelly rockfish. Panels and axes as in Figure A1.

APPENDIX MS5. BIOLOGICAL AND ECONOMIC EFFECTS OF CATCH CHANGES DUE TO THE PACIFIC COAST GROUNDFISH 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.

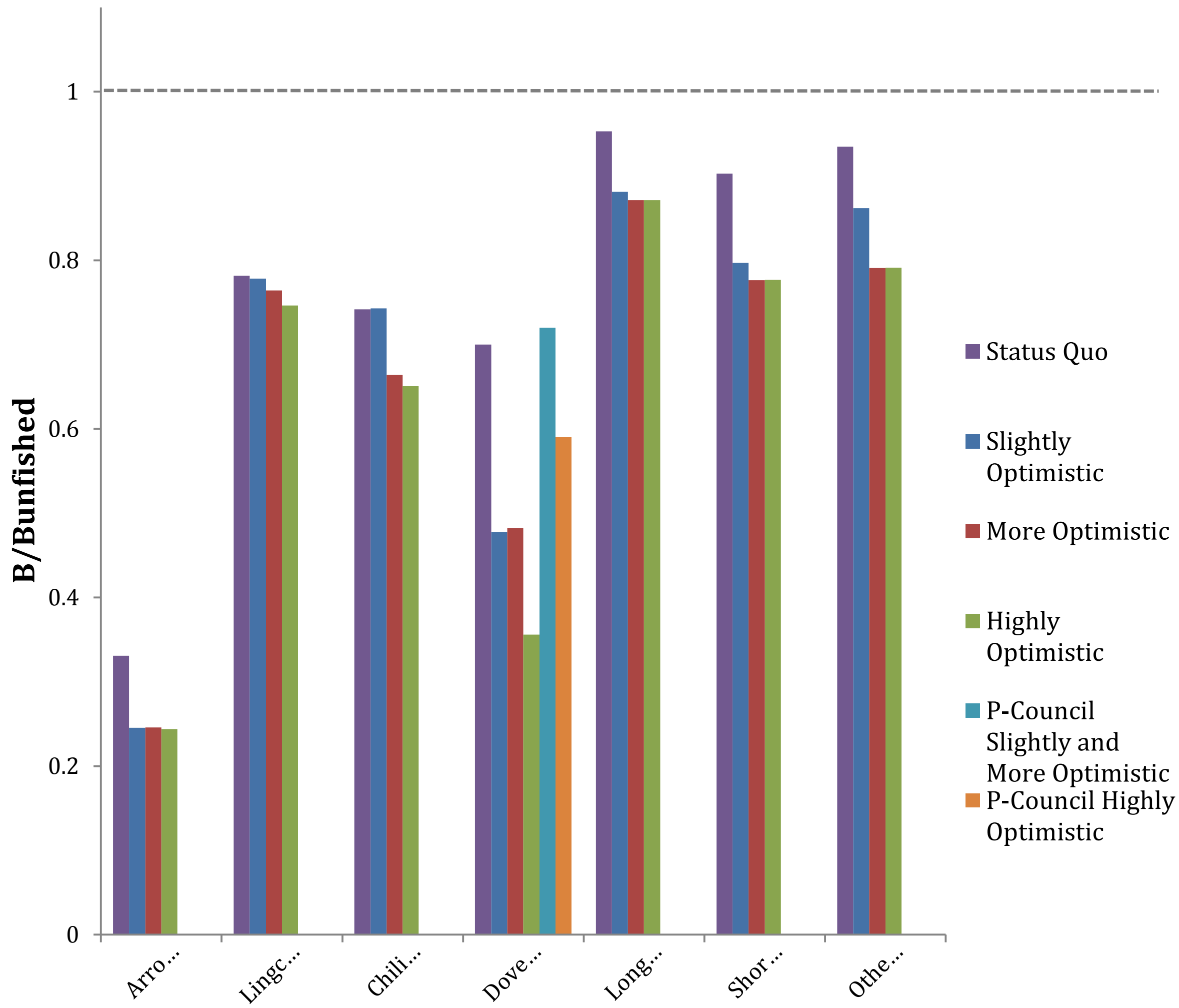

Figure 1. Relative biomass at year 25 predicted by the Atlantis ecosystem model. Also included for comparison are year 25 relative biomass values of Dover sole from a single species stock assessment (Pacific Fishery Management Council 2010b). All other functional groups varied less than 5% among scenarios.

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).

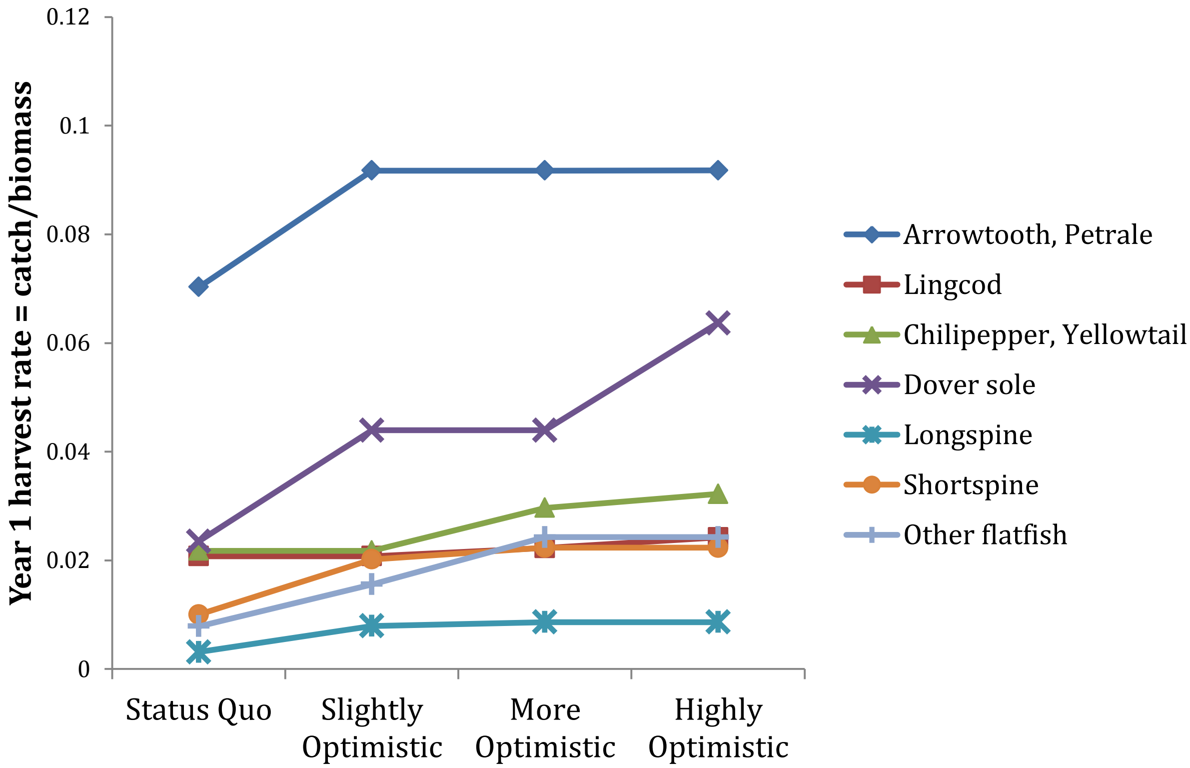

Figure 2. Harvest rate (calculated as harvest rate =catch/biomass) for each species or functional grouper, per scenario.

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 *Prior to catch shares scenario*		
Gear	**Year**	***Slightly optimistic***	***More optimistic***	***Highly optimistic***
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.

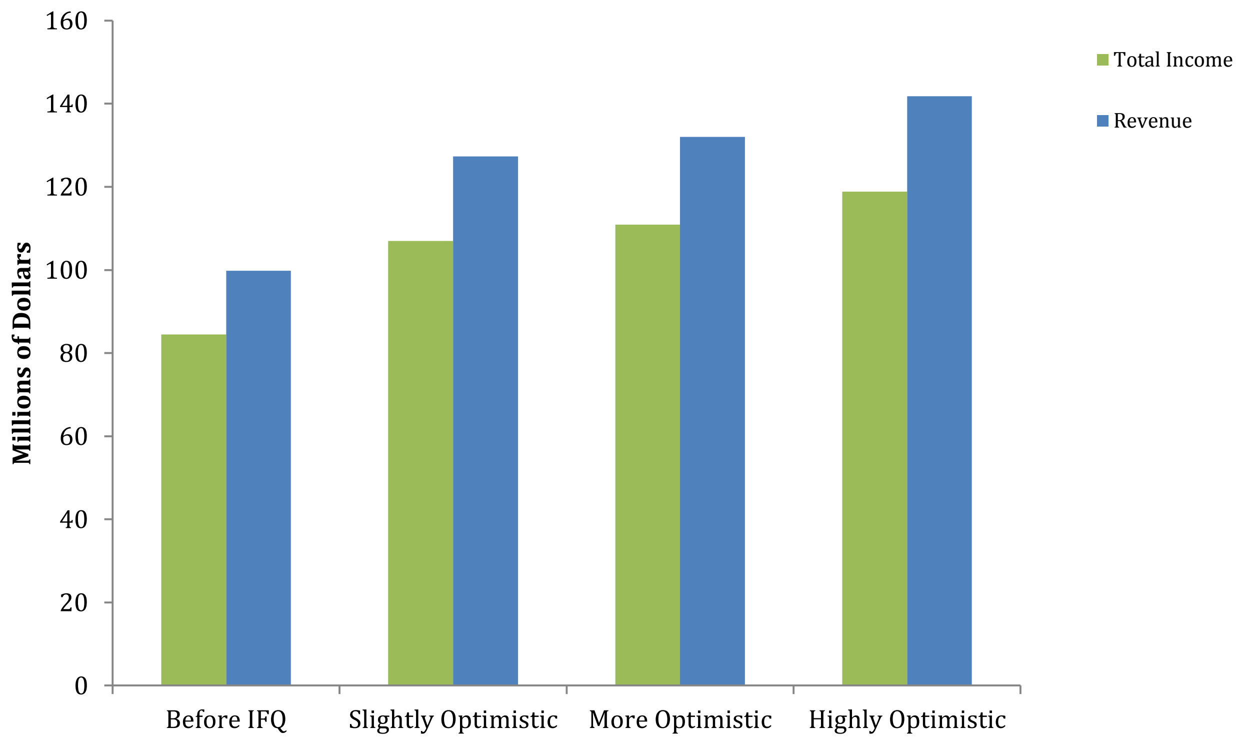

Figure 3. Revenue in fishery sectors, and income effects in the broader West Coast economy. Year 1 predictions. Total income and revenue are represented by bars in millions of dollars (left axis).

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.

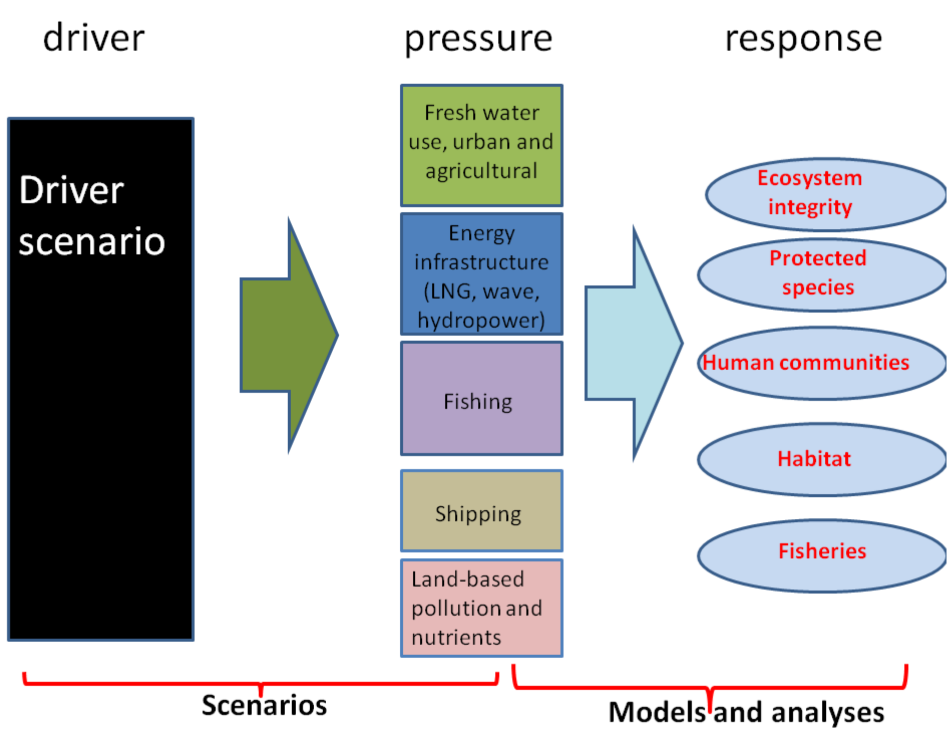

Figure 4. Schematic of Management Testing approach, where drivers are linked to pressures via narrative scenarios, and then quantitative models link pressures to responses.

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

The full version of this work is published as:

Kaplan, I.C., D. S. Holland, and E. A. Fulton. In press. Finding the accelerator and brake in an individual quota fishery: Linking ecology, economics, and fleet dynamics of US West Coast trawl fisheries. ICES Journal of Marine Science.

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

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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:

<u>Alternative 1 – No Action:</u> 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.

<u>Alternative 2 – Full Facilities Removal of Four Dams:</u> 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 (USDOI).

<u>Alternative 3 – Partial Facilities Removal of Four Dams:</u> 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).

[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).

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 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.

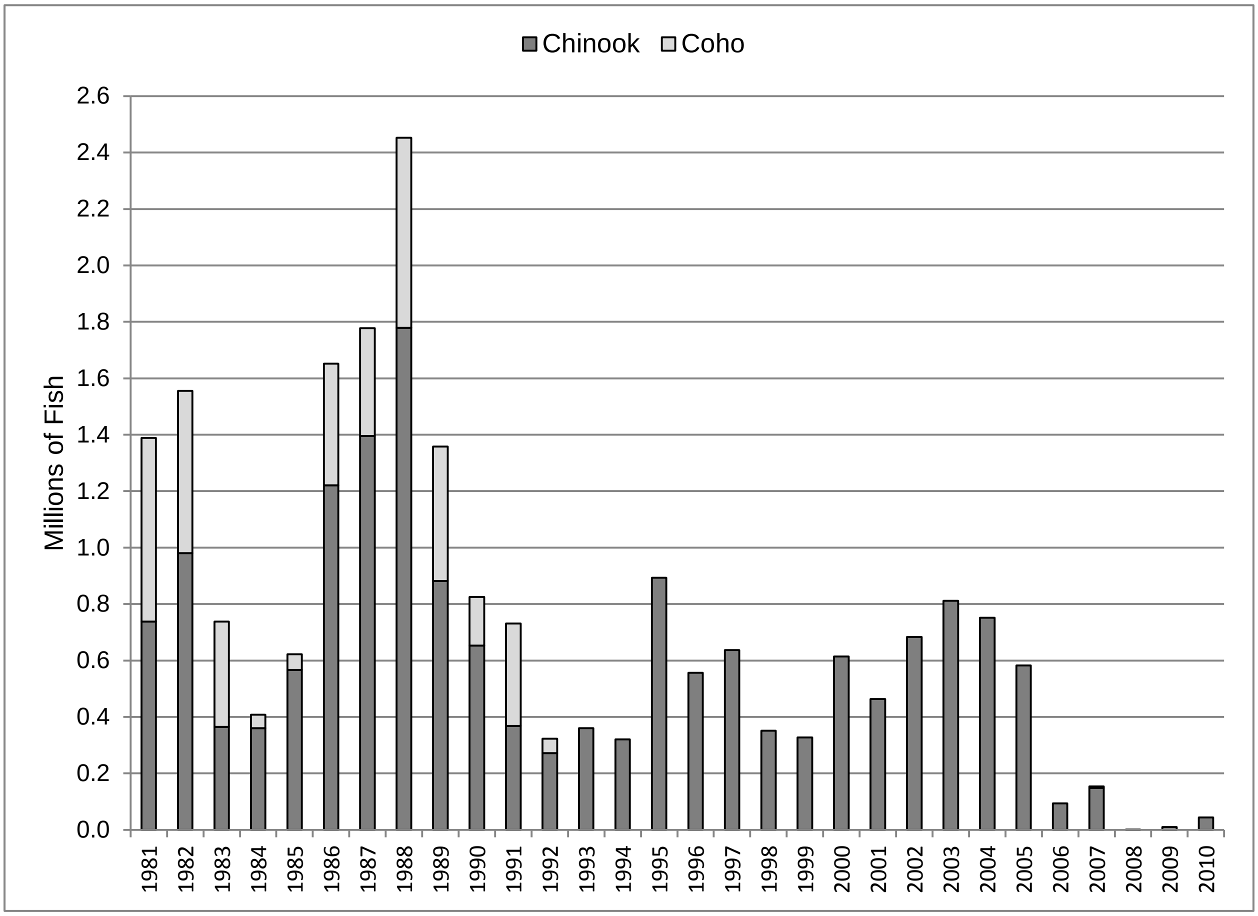

Figure II-2. Landings of troll-caught Chinook and coho south of Cape Falcon, Oregon (millions of fish), 1981-2010 (sources: PFMC 1990, 1991, 1998, 2009, 2010, 2011b).

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 revenue[4] 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 prices[2] 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).

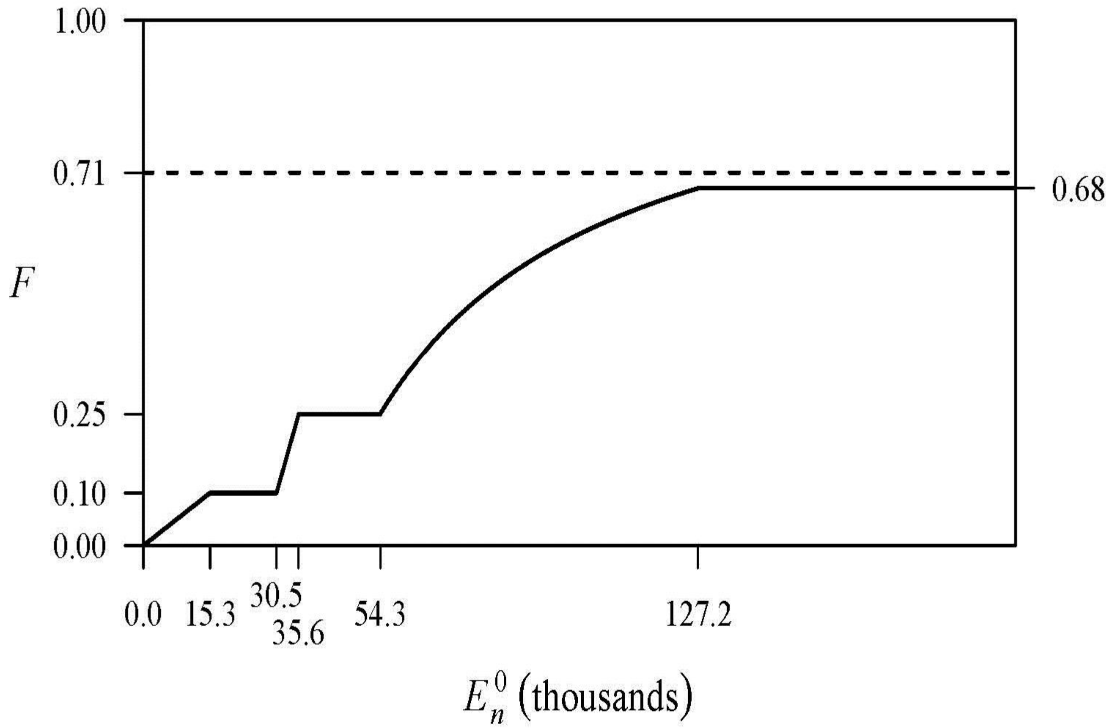

Figure III-1. Harvest control rule used in the EDRRA model (En0 = annual escapement to natural areas prior to ocean or inriver harvest, F = harvest rate) (graphic by Michael Mohr, NMFS).

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 increase[6] 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).

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[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).

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

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[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.

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]

Management Area	# Klamath Chinook	# Chinook (All Stocks)	Gross Revenue (2012$)	Net Revenue (2012$)
Monterey	1,671	1,671	71,367	58,021

[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).

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
Fort Bragg	9,156	140,465	6,000,817	4,878,665	1,461,632

[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.

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.

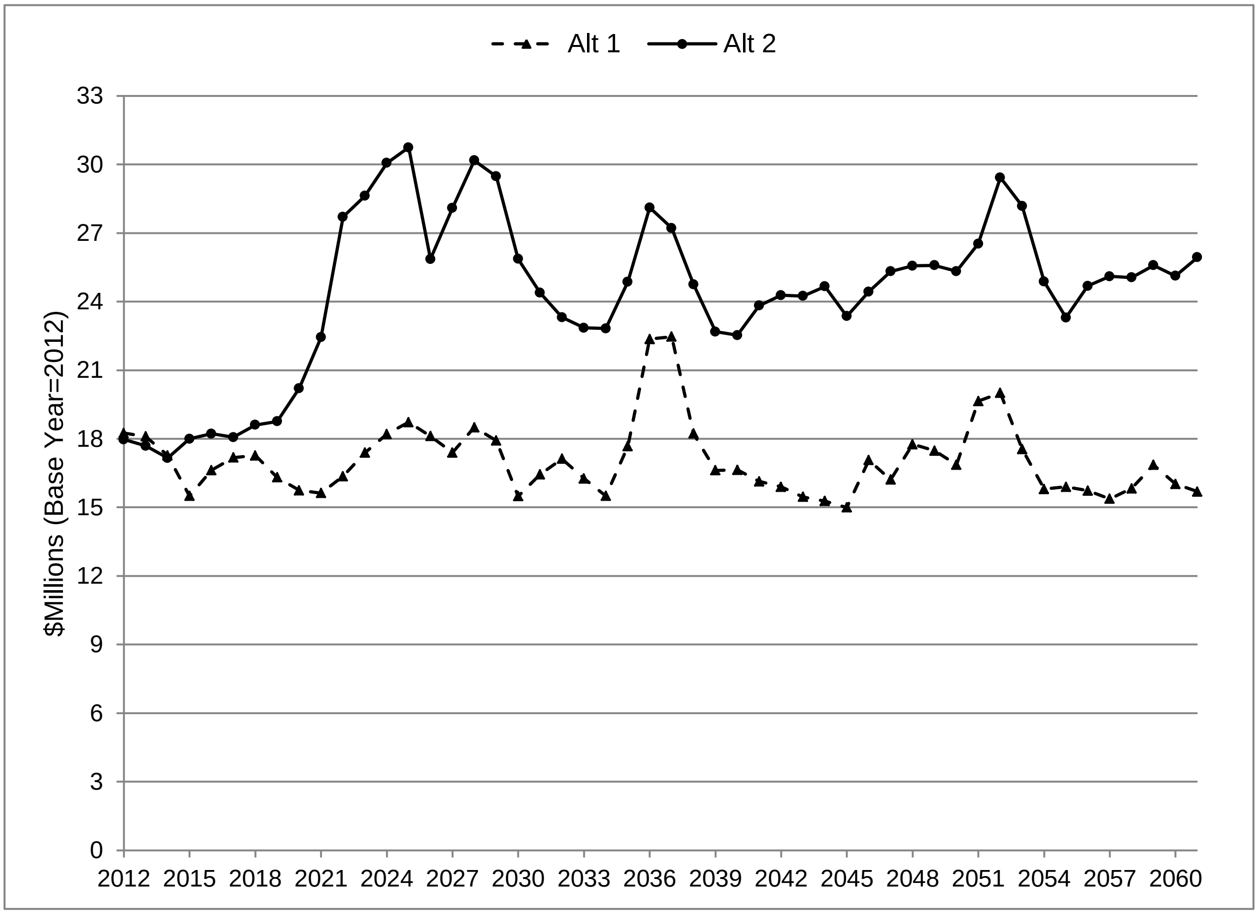

Figure IV-1. Projected annual net revenue under Alternatives 1 and 2 during 2012-61 (calculated according to the methodology described in Appendix B-2).

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.

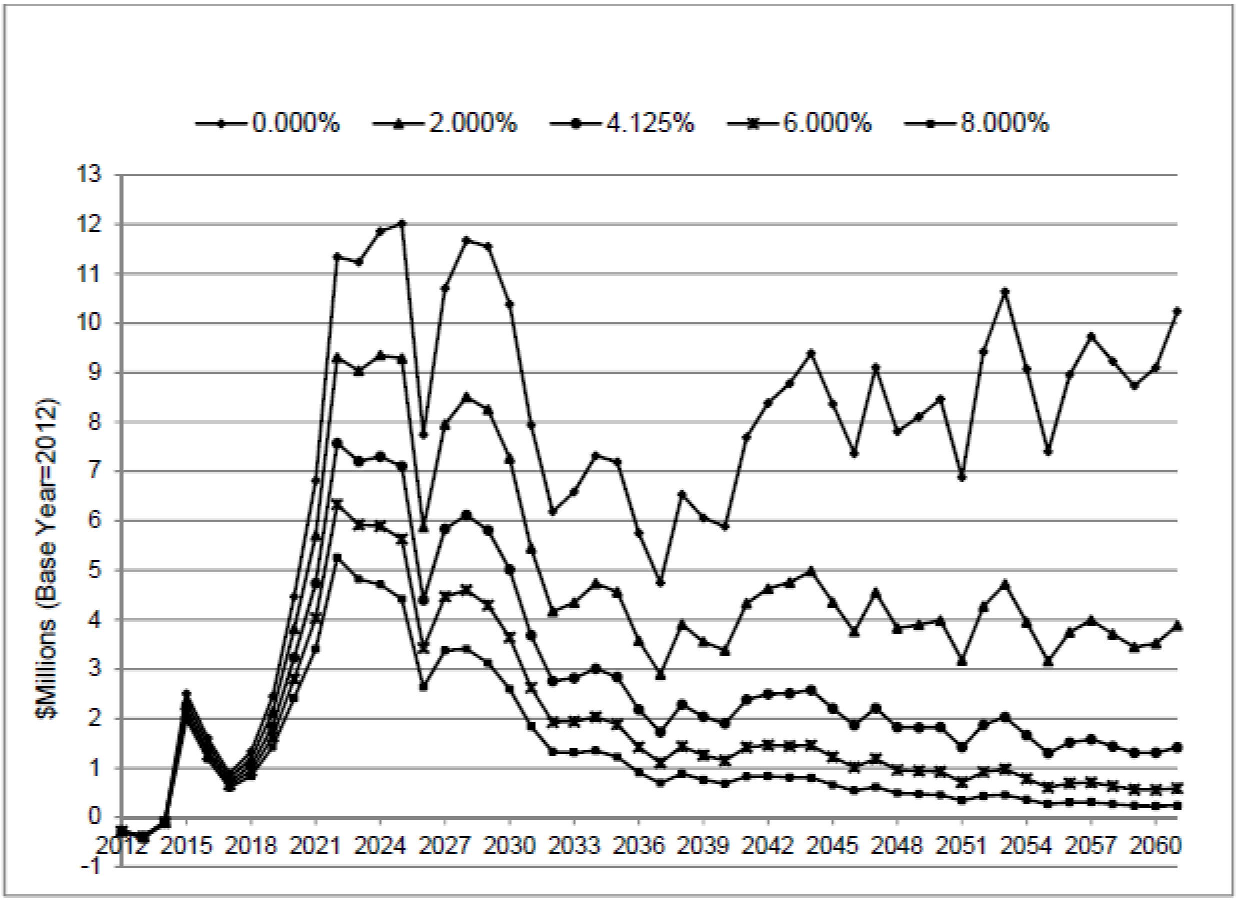

Figure IV-2. Annual discounted values of the increase in net revenue under Alternative 2 relative to Alternative 1 (2012$) during the projection period 2012-61, calculated on the basis of alternative discount rates of 0% (no discounting), 2%, 4.125%, 6%, and 8%.

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 area[1]

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			
Impact Type	Employment (Jobs)	Labor Income ($Millions)	Output ($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	
	Jobs	% change from Alt 1	$Millions	% change from Alt 1	$Millions	% change from Alt 1

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 stocks[15] are as follows:

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

[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.

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.

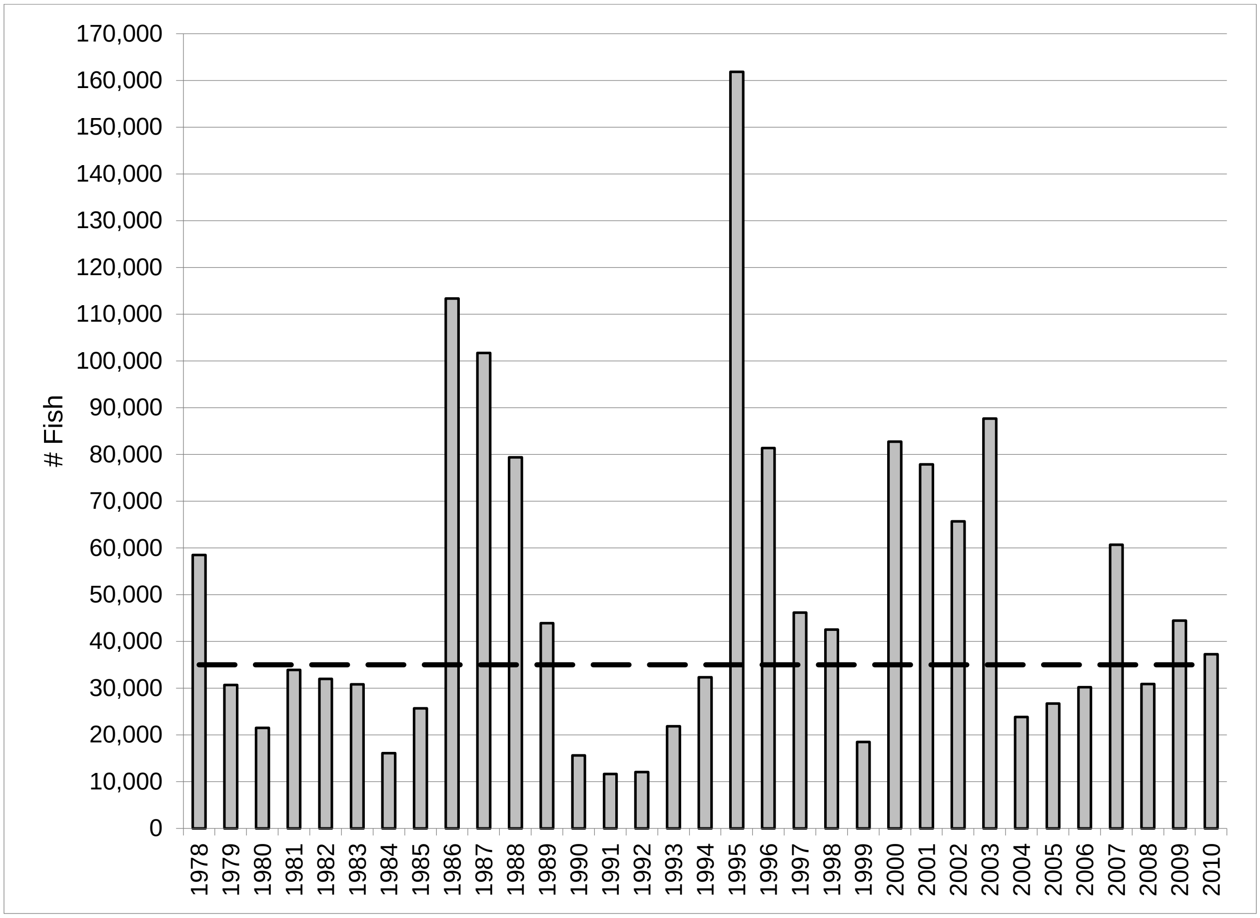

Figure A-1. Klamath River adult natural spawner escapement, 1978-2010. Dotted line represents 35,000 escapement floor in effect during 1989-2006 (source: PFMC 2011a)

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.

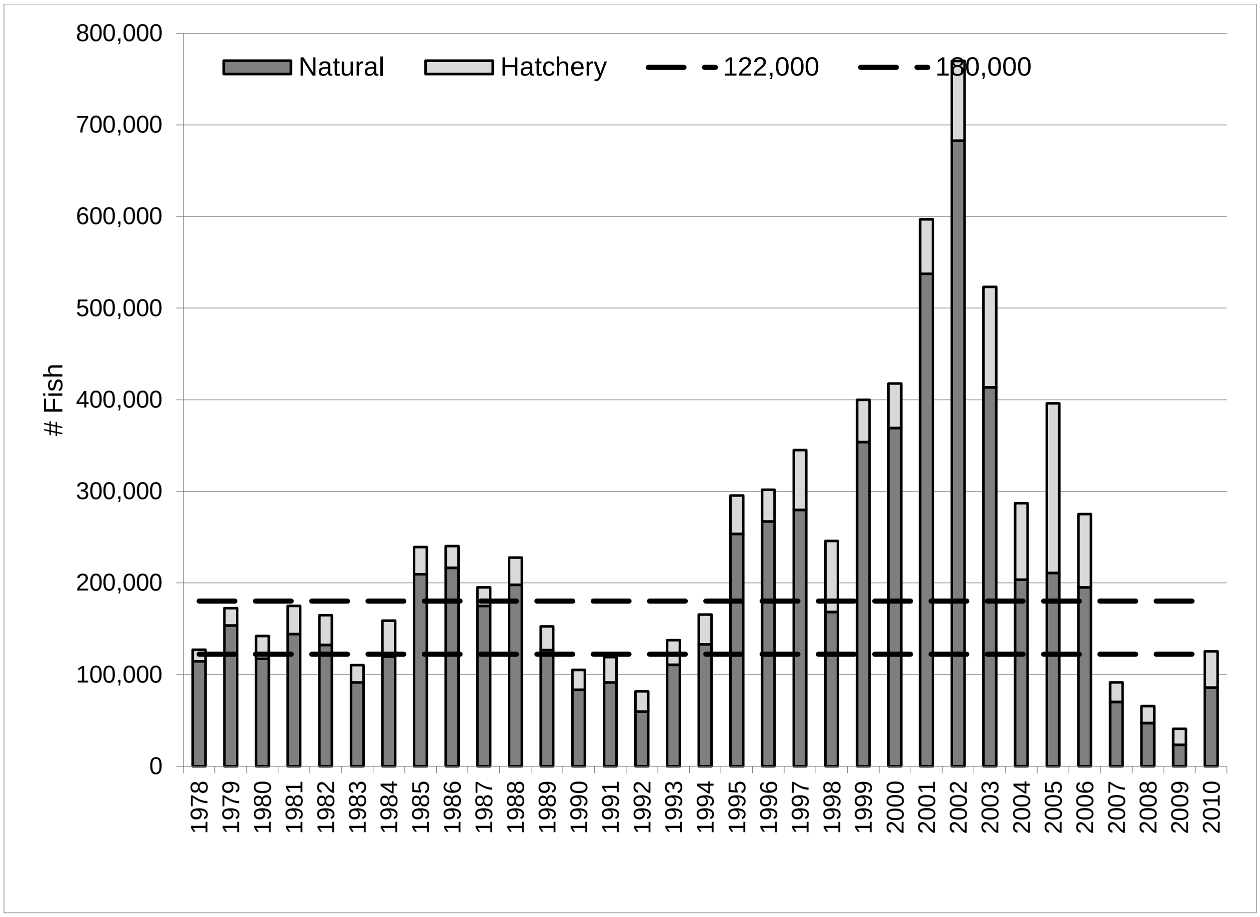

Figure A-2. Sacramento River adult spawner escapement (natural + hatchery), 1978-2010. Dotted lines represent PFMC escapement goal of 122,000-180,000 (source: PFMC 2011a).

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.

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.

[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.

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 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).

[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.

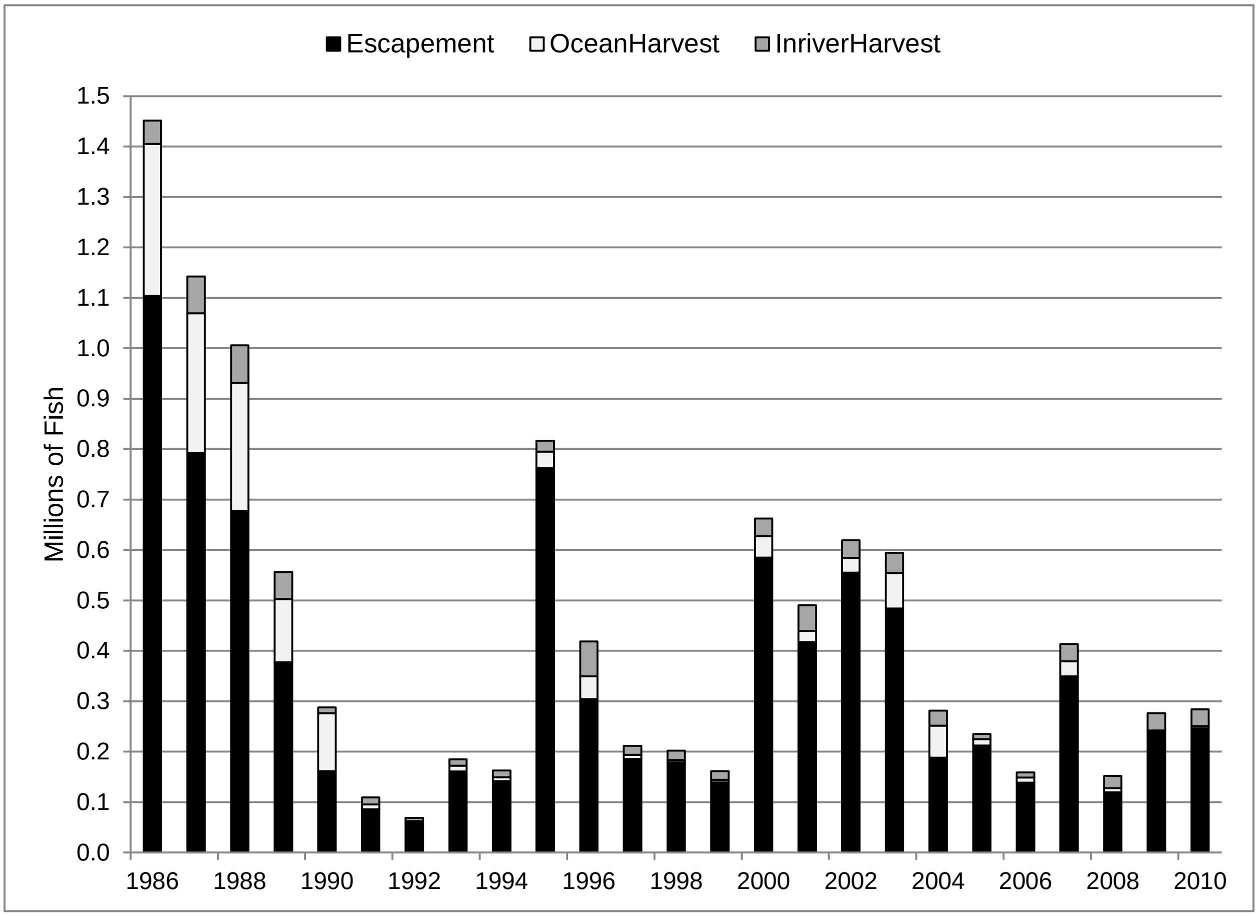

Figure A-3. Klamath River fall Chinook ocean abundance index (millions of fish), 1986-2010 (source: PFMC 2011a).

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.

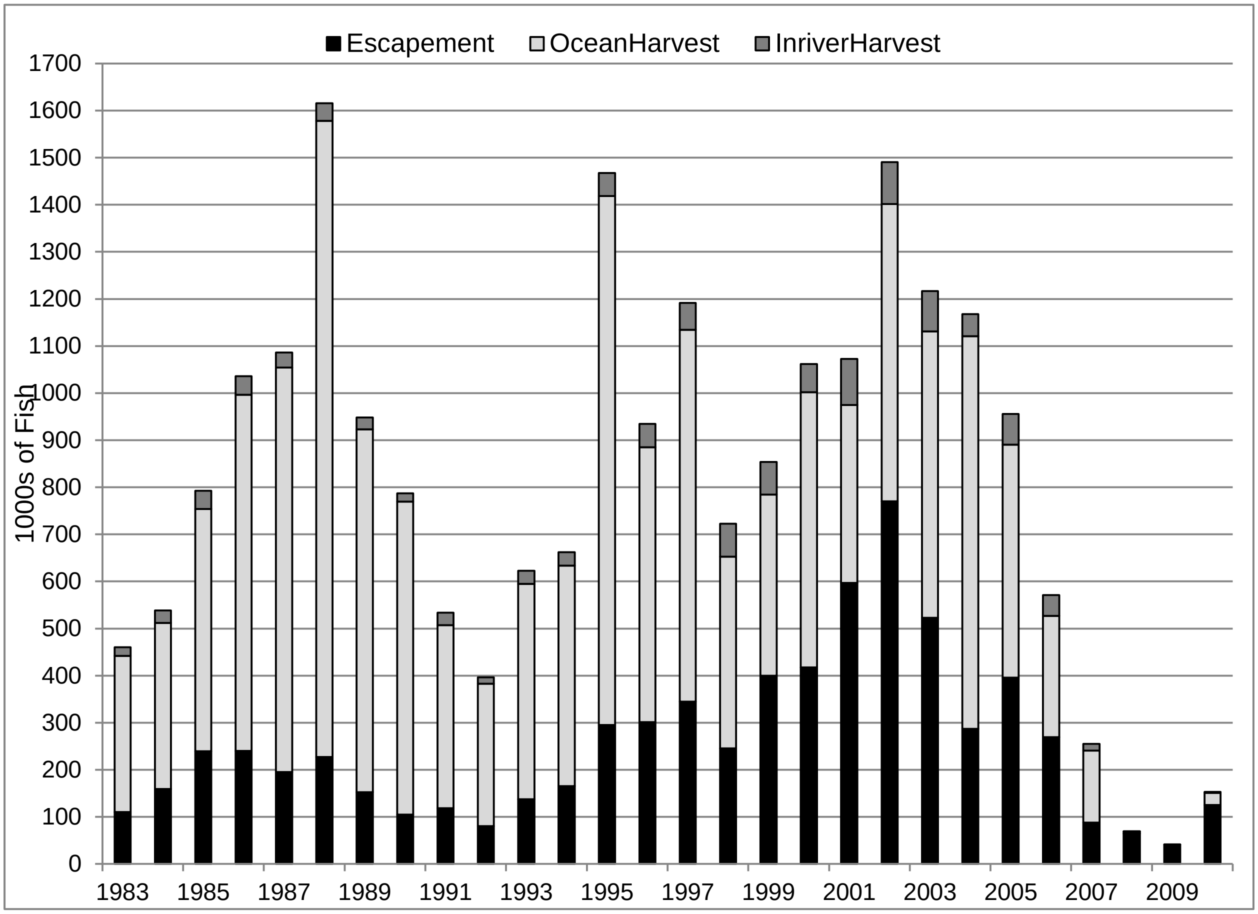

Figure A-4. Sacramento River fall Chinook ocean abundance index (1000s of fish), 1983-2010 (source: PFMC 2011a).

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	$KLAMCHNK^{NAA} = KLAMCHNK_{mean(01\text{-}05)}$	$KLAMCHNK^{DRA} = KLAMCHNK^{NAA} \times PCTHARV$
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

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).

$EXPAND_i$ = 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).

LBFISH = mean weight (pounds dressed weight) per troll-caught Chinook south of Cape Falcon during 2001-05 (data source: PFMC 2011b).

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).

PCTREV = estimated percent of gross salmon troll revenue remaining after payment of trip expenses (source: Jerry Leonard, NMFS)

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:

$$PCTHARV=1/T \sum_{t=1,\ldots,T} \{Median_{t,j=1,\ldots,1000} [(KLAMCHNK_{t,j}^{DRA*} - KLAMCHNK_{t,j}^{NAA*})/ KLAMCHNK_{t,j}^{NAA*}]\} \quad [B1]$$

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;

$Median_{t,j=1,\ldots,1000}$ [ ] is the median of the 1000 values of [ ] generated for year t;

$1/T \sum_{t=1,\ldots,T}$ $\{Median_{t,j=1,\ldots,1000}$ [ ]} 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.

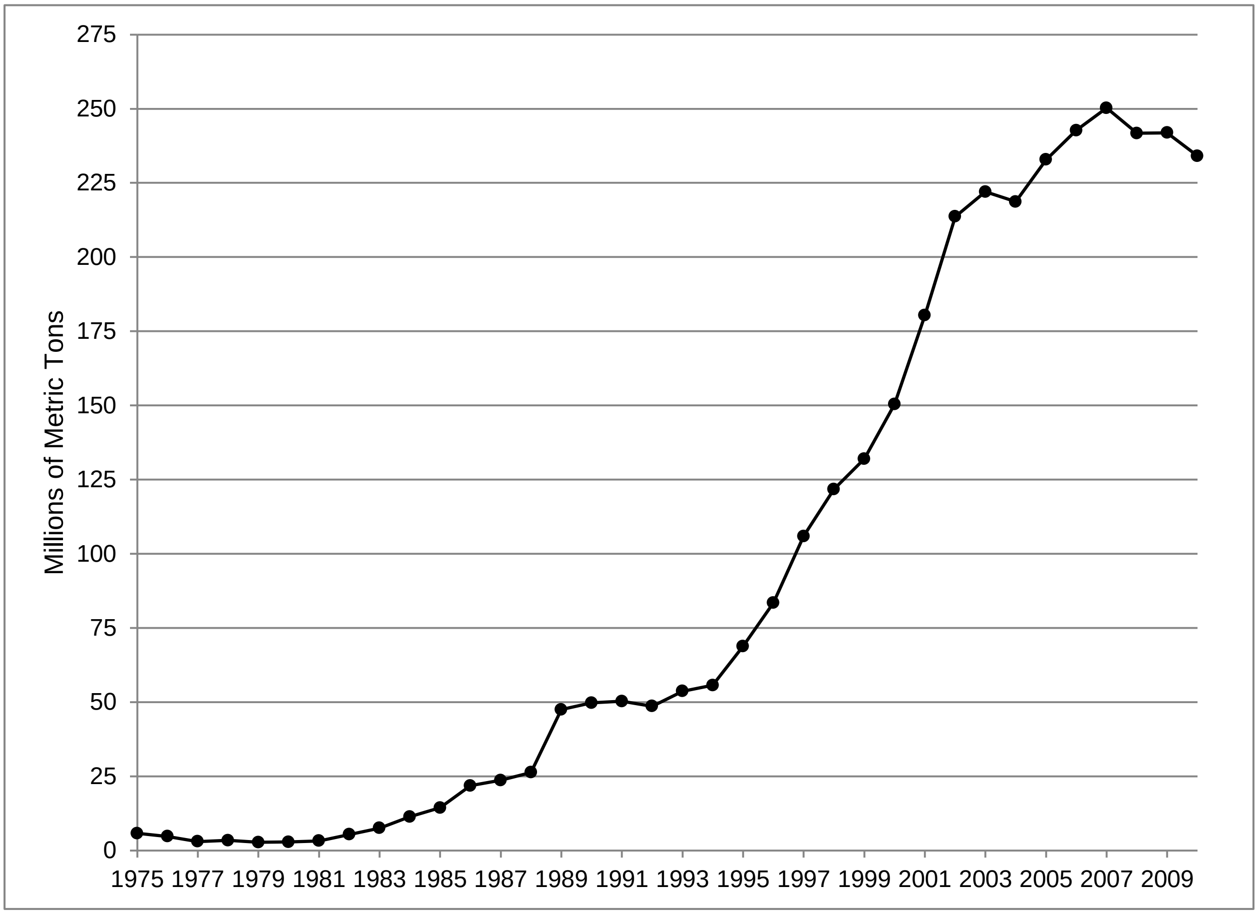

Figure B-1. Imports of edible salmon products into the U.S., 1975-2010 (source: NOAA National Marine Fisheries Service, Office of Science and Technology, Silver Spring, MD).

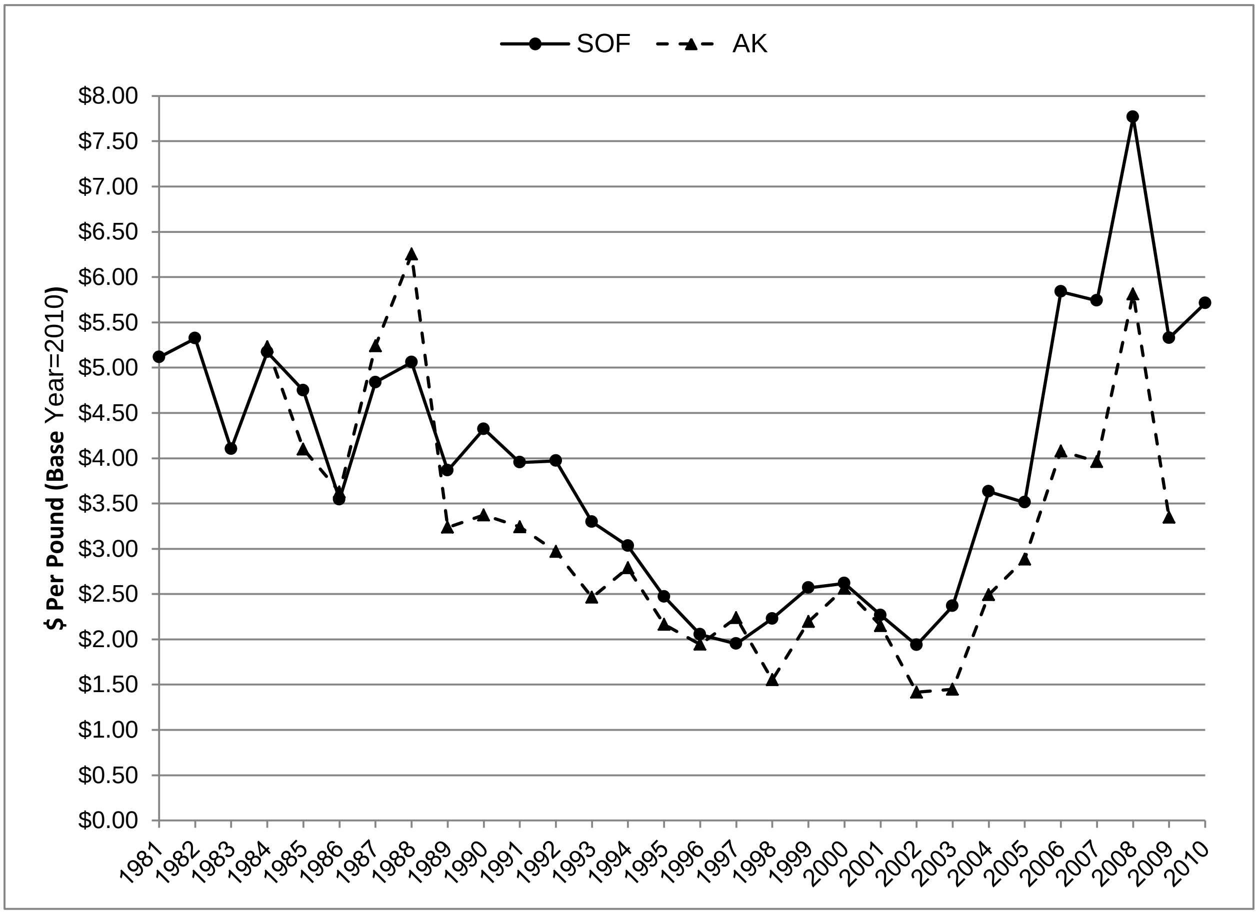

Figure B-2. Ex-vessel prices of troll-caught Chinook in California and Oregon south of Cape Falcon during 1981-2010 and in Southeast Alaska during1984-2009 (2012$) (sources: PFMC 1998, 2011b; ADFG 2009).[20]

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).

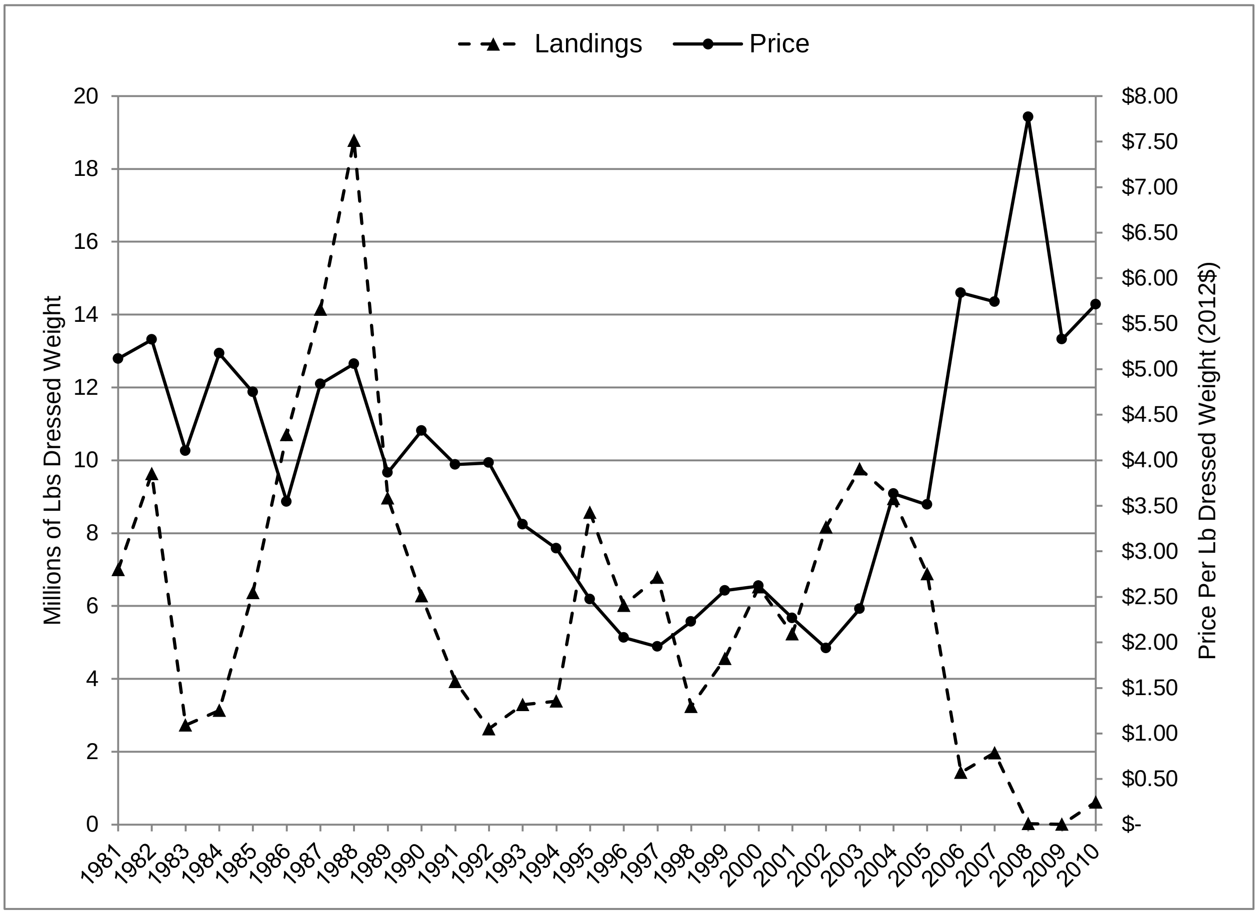

Figure B-3. Annual landings (pounds dressed weight) and ex-vessel price (2012$) of troll-caught Chinook south of Cape Falcon, Oregon, 1981-2010 (sources: PFMC 1990, 1991, 1998, 2001, 2011b).

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:

NETREVtAlt1 = NETREVAlt1 x KLAMCHNKtNAA* / KLAMCHNKmean(12-61)NAA* [B2]

where

NETREVAlt1 = average annual net revenue (all areas) under Alternative 1 ($17.1 million, according to Table IV-1), and

KLAMCHNKtNAA* / KLAMCHNKmean(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 (NETREVtAlt2) was similarly calculated, as follows:

NETREVtAlt2 = NETREVAlt2 x KLAMCHNKtDRA* / KLAMCHNKmean(12-61)DRA* [B3]

where

NETREVAlt2 = average annual net revenue (all areas) under Alternative 2 ($24.4 million, according to Table IV-2), and

KLAMCHNKtDRA* / KLAMCHNKmean(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= ∑t=2012,…,2061 [(NETREVt Alt2 - NETREVt Alt1)] (1+r)-t [B4]

where

NETREVt Alt1 and NETREVt Alt2 = 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:

$$PCTYRS=1/T \sum_{t=1,\ldots,T} \{(1/1000)\ COUNT_{t,j=1,\ldots,1000}\ [KLAMCHNK_{tj}^{DRA*}>KLAMCHNK_{t,j}^{NAA*}]\} \qquad [B5]$$

where

$KLAMCHNK_{t,j}^{NAA*}$ = troll harvest of Klamath Chinook projected by EDRRA model for year t and iteration j under the NAA;

$KLAMCHNK_{t,j}^{DRA*}$ = troll harvest of Klamath Chinook projected by EDRRA model for year t and iteration j under the DRA;

$\{(1/1000)\ COUNT_{t,,j=1,\ldots,1000}\ [\ ]\}$ = 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]);

$1/T \sum_{t=1,\ldots,T} \{(1/1000)\ COUNT_{t,,j=1,\ldots,1000}\ [\ ]\}$ = mean of $\{(1/1000)\ COUNT_{t,,j=1,\ldots,1000}\ [\ ]\}$ over years t=1,…,T.

ESTIMATION OF PERCENT DIFFERENCE IN FREQUENCY OF PRE-HARVEST ESCAPEMENT ≤ 30,500

The percent difference between the NAA and DRA in the frequency of pre-harvest adult natural spawner escapements ≤ 30,500 (PCTDIFF) was estimated from EDRRA model outputs as follows:

PCTDIFF = $1/T \sum_{t=1,\ldots,T} \{[\mathrm{COUNT}_{t,j=1,\ldots,1000}^{DRA*}(\mathrm{ESCAPE}_{tj}^{DRA*} \leq 30{,}500)$

$- \mathrm{COUNT}_{t,j=1,\ldots,1000}^{NAA*}(\mathrm{ESCAPE}_{tj}^{NAA*} \leq 30{,}500)]/$

$\mathrm{COUNT}_{t,j=1,\ldots,1000}^{NAA*}(\mathrm{ESCAPE}_{tj}^{NAA*} < 30{,}500)\}$ [B6]

where

$\mathrm{ESCAPE}_{t\,j}^{NAA*}$ = pre-h arvest escapement of Klamath Chinook projected by the EDRRA model for year t=1,…,T and iteration j=1,…,1000 under the NAA;

$\mathrm{ESCAPE}_{t\,j}^{DRA*}$ = pre-harvest escapement of Klamath Chinook projected by the EDRRA model for year t=1,…,T and iteration j=1,…,1000 under the DRA;

$\mathrm{COUNT}_{t,j=1,\ldots,1000}^{NAA*}(\mathrm{ESCAPE}_{t,j}^{NAA*} \leq 30{,}500)$ = number of iterations j in year t when $\mathrm{ESCAPE}_{t\,j}^{NAA*} \leq 30{,}500$ under the NAA;

$\mathrm{COUNT}_{t,j=1,\ldots,1000}^{DRA*}(\mathrm{ESCAPE}_{t,j}^{DRA*} \leq 30{,}500)$ = number of iterations j in year t when $\mathrm{ESCAPE}_{t\,j}^{DRA*} \leq 30{,}500$ under the DRA;

$[\mathrm{COUNT}_{t,j=1,\ldots,1000}^{DRA*}(\,) - \mathrm{COUNT}_{t,j=1,\ldots,1000}^{NAA*}(\,)]/\mathrm{COUNT}_{t,j=1,\ldots,1000}^{NAA*}(\,)$ = percent difference between DRA and NAA in number of iterations when pre-harvest adult natural spawner escapement ≤ 30,500, estimated separately for each year t. ( ) is shorthand for what appears in parentheses in equation [B6];

$1/T \sum_{t=1,\ldots,T} \{[\mathrm{COUNT}_{t,j=1,\ldots,1000}^{DRA*}(\,) - \mathrm{COUNT}_{t,j=1,\ldots,1000}^{NAA*}(\,)]/\mathrm{COUNT}_{t,j=1,\ldots,1000}^{NAA*}(\,)\}$

= mean of percent differences over years t=1.,…,T.