INTRODUCTION

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PREFACE

In 2010, scientists representing NOAA line offices along the US West Coast initiated the California Current Integrated Ecosystem Assessment (CCIEA), a structured effort to organize and analyze scientific information in the context of ecosystem-based management of the California Current Large Marine Ecosystem (CCLME). The challenging task of assembling and interpreting large volumes of data from a broad range of disciplines, locations and time frames engages over 50 scientists from NOAA's Northwest and Southwest Fisheries Science Centers, other NOAA offices and colleagues from academia and non-governmental entities. The CCIEA team has taken an iterative approach for this work, with this being the third report. The first CCIEA report described the scope and conceptual underpinnings of the CCIEA, and presented preliminary findings on status, risk and management of salmon, groundfish, green sturgeon, and overall ecosystem health as of 2010 (Levin and Schwing 2011). The second "Phase II" report extended previous findings to the year 2012; expanded the range of focal components to include coastal pelagic species, marine mammals, seabirds, and coastal communities; expanded the list of drivers and pressures; and described further risk assessments and potential management strategy alternatives in the CCLME (Levin et al. 2013).

Here, we introduce the "Phase III" report, which describes our understanding of physical, chemical, ecological, and socioeconomic conditions in the California Current through the year 2013. We also formally introduce two major new components into the CCIEA effort: Habitat, the matrix within which ecological interactions occur; and Human Dimensions, the interface between humans and the other components (living and non-living) of the CCLME. We also further advance the effort to make this work truly *integrative* across components, rather than a series of parallel condition reports on co-occurring species and processes; examples of this are perhaps most clearly seen in sections on Salmon, Risk Assessment, and Management Strategy Evaluation.

The Phase III report is presented as a series of time-stamped documents in downloadable formats with accompanying web-based materials available at the CCIEA website (http://www.noaa.gov/iea/regions/california-current-region/). As with prior CCIEA reports, all chapters and appendices in Phase III have been peer-reviewed, and we gratefully thank colleagues who provided their time and expertise in the review process.

WHAT IS AN IEA?

As in previous iterations of the CCIEA (Levin and Schwing 2011, Levin et al. 2013), we follow the NOAA definition of an IEA: a formal synthesis and quantitative analysis of information on relevant natural and socioeconomic factors in relation to specific ecosystem management goals. NOAA defines an ecosystem as a geographically specified system of environments, habitats, processes, and organisms. Importantly, "organisms" explicitly include the humans that live in or near an ecosystem and benefit from its structure and functions, and the "environments" explicitly include social conditions as well as the physical, chemical and biological conditions in which organisms dwell. Ideally, the products of an IEA provide science support for the process of ecosystem-based management (EBM) of resources and resource use.

The general IEA approach has four primary steps: (1) *scoping*, where policymakers, managers, stakeholders and researchers collaborate to identify and articulate management objectives, ecosystem boundaries, key ecosystem attributes and important stressors; (2) *indicator development*, where scientists identify and test indicator variables that are suitable proxies for ecosystem attributes and thus reflect the status of ecosystem conditions relative to management decision rules; (3) *risk analysis*, where indicators are analyzed to determine their exposure and sensitivity to natural and human stressors; and (4) *management strategy evaluation*, where potential management strategies are assessed to determine their effectiveness at meeting management objectives while also identifying potential tradeoffs across different ecosystem components. Most of the work outlined below and detailed elsewhere in the report describes efforts on steps 2-4, which reflects a conscious effort to build up our IEA "science toolkit" in advance of formal scoping; we intend to direct more effort toward scoping in coming years.

EXECUTIVE SUMMARY OF THE 2013 CALIFORNIA CURRENT IEA

OVERALL ASSESSMENT

Recent indicator values leading up to 2013 point to a relatively productive period in the CCLME. Large scale climate and ocean indices all pointed to average or above average conditions for primary production (see Tables 1 and 2 and the end of this chapter). In all regions except southern California (south of 36° N), the Bakun Upwelling Index was the highest on record. Measures of chlorophyll-*a*, determined from satellite imagery, were low in the north and above average in the middle and southern regions. One concern was that strong upwelling might quickly transport primary production offshore, and therefore lower secondary production. This did not appear to happen, as both zooplankton (particularly lipid-rich northern copepods) and coastal pelagic fish populations appeared to be relatively productive and abundant (Tables 2, 5 and 8), except possibly anchovies and sardines.

At higher trophic levels, where longer lifespans and population doubling times lead to temporally lagged responses to changes in production, indicators of population abundance and condition were more mixed. Chinook and coho salmon populations were generally within the bounds of long-term averages, though many populations showed positive or negative short-term trends (Table 6). Groundfish abundance status and trends were generally encouraging, with only a few populations (all rockfish) below the overfished threshold, although the indicators of population structure suggest considerable truncation of age structure among most taxonomic groups (Table 7). The status of birds and mammals is harder to ascertain; although the few species for which we have data appear to have stable or even increasing populations, we lack data or suitable monitoring plans for most species. Of special concern is an unusual mortality event of California sea lion pups that occurred off southern California in 2012-2013.

Total commercial fishery landings increased in recent years, primarily driven by increases in Pacific hake and shrimp, and secondarily by landings of coastal pelagic species and crabs being above the long-term averages (Table 3). However, landings in some commercial fisheries were near historic lows (salmon, groundfish), and the diversity of landings by fishing vessels and ports continued to decrease, which may reflect greater vulnerability to revenue swings. Declines in bottom trawling targeting groundfish may have lessened groundfish mortality and impacts on benthic habitats. Other anthropogenic activities in the CCLME had mixed trends (Table 4); many large-scale human activities were in decline, such as recreational beach use, shipping, and certain forms of pollution; these declines may be related to the recent downturn in US economic conditions and warrant monitoring during economic recovery.

STATUS AND TRENDS OF THE CALIFORNIA CURRENT THROUGH 2013

An IEA goal is to analyze the connectivity between indicators of different environmental drivers and trophic levels to determine how to best represent critical ecosystem status parameters. The ultimate goal is to define the indices which best help describe conditions for any specific ecosystem attribute, e.g., a species, community, habitat, fishery or element of human wellbeing.

The CCIEA has now selected 174 suitable indicators to analyze for conditions and trends. The range is from basic environmental parameters up through top predators and human dimensions. Tables 1 through 8 at the end of this chapter are presented as visual summaries of the indicators, where the arrows indicate the recent trend, the symbols (•,+,-) indicate the index value relative to the long term statistics, and the colors represent our (Harvey and Garfield) qualitative judgment on the trend or status reflecting “good,” "neutral" or "poor" conditions for overall ecosystem processes and functions. Where possible, we break these status and trends indicators out by season or one of three latitudinal ecoregions (Figure 1): the northern California Current region north of Cape Mendocino; the central California Current region between Cape Mendocino and Point Conception; and the southern California Current region, south of Point Conception. There can exist large spatial variation in physical and biological indicators among and within the subregions. Overall, most indicators are within the range of ± 1 standard deviation (s.d.) of the long-term mean, which is taken as within the normal range for the indicator. There are few significant outliers in the last five years, which suggest that conditions were fairly stable. The full temporal variation of each indicator is provided within each relevant chapter of the full Phase III report.

DRIVERS AND PRESSURES

We generally categorize indicators of drivers and pressures in two categories: (1) physical, chemical and climate drivers and pressures, i.e., forcing that is largely driven by natural processes; and (2) anthropogenic drivers and pressures, i.e., forcing that is of human origin. Both types of forces operate across a range of spatial and temporal scales.

PHYSICAL, CHEMICAL AND CLIMATE DRIVERS

The environmental indices at large (Multivariate El Niño Index (MEI), the Northern Oscillation Index (NOI), the North Pacific Gyre Oscillation (NPGO), and the Pacific Decadal Oscillation (PDO)), regional (Eddy Kinetic Energy (EKE)) and local (Upwelling index (UI), Sea Surface Temperature (SST), coastal sea surface height, meridional winds, and pycnocline depth and strength) scales generally remained within the mean range defined by the long term mean ± 1 s.d. (Table 1; see Hazen et al. 2014a). The exception is that the NPGO remained above the mean, which is indicative of stronger gyre circulation that generally favors productivity. Upwelling trends were stable or positive (Tables 1 and 2).

Chemical indices are water column nutrients, represented by nitrate plus nitrite, and dissolved oxygen, which can also serve as a proxy for ocean acidification. Similar to the environmental indicators, both the nutrient and oxygen indices are within the long term mean range with no strong trends (Table 1).

A new index, the Multivariate Ocean Climate Index (MOCI) is introduced in this Phase III report; it is composed of multiple indices and provides a broad perspective on the status of the ecosystem (Hazen et al. 2014a). Similar to the other ecosystem indices, the recent status value is within 1 s.d. of the long-term mean (Table 2).

ANTHROPOGENIC DRIVERS

The 23 anthropogenic drivers and pressures examined show considerable variation among different sectors (Andrews et al. 2014). Among fisheries, landings of demersal groundfish are historically low while those of coastal pelagic species and crabs are higher than average; landings of hake, shrimp, and all fisheries combined are increasing (Table 3). Non-fisheries activities varied widely (Table 4). Some indicators were above average but level (e.g., coastal engineering, power plant activity, sediment retention) while others were below average (offshore oil and gas activity, benthic structures). Many activities had negative trends (e.g., shipping, invasive species, beach use, several forms of pollution), which may be related to weak economic conditions. A few showed positive trends (dredging, shellfish aquaculture), and high or increasing indicators of total fishery landings, aquaculture production and seafood demand warrant continued attention. Anthropogenic activity indicators presently at declining or low levels should also be watched carefully as national and global economic trends change.

HABITAT

A formal selection of indicators of habitat quantity and quality is a new addition to this year's CCIEA. Given the important relationships that habitat types have with all drivers, species, ecological processes and human wellbeing in the CCLME, this is a significant step for the CCIEA. Using the standard CCIEA methods for indicator selection, we identified 33 high priority habitat indicators, relating to the quantity and quality of freshwater, nearshore/estuarine, pelagic, and seafloor habitats, as well as some anthropogenic pressures that are particularly focused on these habitats (Greene et al. 2014). In the coming years, these 33 indicators will be quantified and analyzed for spatial and temporal trends. They will also contribute to synthetic analyses in the IEA framework, such as spatially based risk assessments and management strategy evaluations.

KEY ECOSYSTEM COMPONENTS

ZOOPLANKTON

Zooplankton abundance and composition represent conditions near the base of the food chain, and as such can be used as one of the indicators of ecosystem health. Peterson et al. (2014) have developed a suite of zooplankton indicators based on samples collected monthly along the Newport hydrographic line. Copepod biomass and composition provide an indication of the abundance of the prey resource for higher trophic levels. Copepod composition is further separated into northern and southern copepod assemblages that indicate lipid-richness (northern > southern). All four indices remain within long term ranges and do not show trends during the last five years (Table 2; see also Hazen et al. 2014a).

COASTAL PELAGIC SPECIES

Data describing the abundance of pelagic forage species (e.g., schooling pelagic fishes and squids) are generally obtained from fishery independent surveys, and sampling methods are different in the three regions of the CCLME. Not all of the same species are sampled across regions. In general terms, in the northern and central CCLME the forage community dependent on cool productive conditions became more abundant or remained stable (Table 5; see also Wells et al. 2014a). However, sardine abundance was low throughout much of the CCLME. Anchovy in the fishery-independent sampling off central and southern California remained at a low abundance. In contrast, the biomasses of Pacific mackerel and sardines as derived from formal stock assessments are within ± 1 s.d. of the long-term mean, both slightly above the mean.

SALMON

A general statement on salmon is difficult given the diversity of riverine populations and the timing of the various runs. Here we report on 14 Chinook salmon and 4 coho salmon data-rich populations, separated into Evolutionarily Significant Units (ESUs) based on rivers and reproductive isolation (Wells et al. 2014b). Species abundance is the most common index of condition, although age diversity, percent natural population (versus hatchery) and population growth rate are indices available for some ESUs. Since salmon populations have suffered such historically significant declines, data for determining trends start with 1985; if earlier data were included, many of the current abundance indicators would be well below the long-term means (Wells et al. 2014b).

In general, California Chinook ESUs were within ± 1 s.d. of the 1985 – present data (Table 6). Central Valley winter-run Chinook salmon abundances were quite low. In the Columbia/Snake basin, Chinook salmon stocks were near the mean and showed both increasing and decreasing trends. Coho salmon stocks were also near their long term mean, again showing different trends in abundance among regions.

GROUNDFISH

Of the 90+ shelf and slope groundfish species that are managed in the CCLME, 36 had sufficient data to use as indicators of groundfish community abundance and condition (Cope and Haltuch 2014). A strong majority of these 36 species had stable or increasing population trends and spawning stock biomasses that are above target levels, and all species have fishing mortality rates that are below overfishing limits (Table 7). Biomasses of three rockfish (*Sebastes*) species were below the minimum limit reference point, indicating overfished status; in addition, several species, mostly rockfish, also have experienced long-term truncations in age distribution and declines in proportions of females that are mature (Cope and Haltuch 2014).

SEABIRDS

No status and trend updates for seabirds were conducted for the CCIEA in 2013. The most recent CCIEA review of seabirds, from last year's CCIEA Phase II report (Zamon et al. 2013), examined recent at-sea abundance trends of three indicator species (common murre *Uria aalge*, sooty shearwater *Puffinus griseus*, and Cassin's auklet *Ptychoramphus aleuticus*) in different seasons in the northern and southern regions of the CCLME. These are common birds but are a fraction of the 75+ seabird species present in the region.

Common murre abundance at sea was stable or increasing, sooty shearwater abundance was stable, and Cassin's auklets increased in the north but were stable or decreasing in the south. However, given the small number of indicators, our understanding of seabird status and trends in the CCIEA context is largely inconclusive at this time.

MARINE MAMMALS

No comprehensive marine mammal surveys were conducted in the California Current in 2013, and thus we cannot update the status and trends indicators from past CCIEA reports. The most recent CCIEA review of marine mammal population status, from last year's CCIEA Phase II report (Redfern et al. 2013), noted that coastwide survey frequencies, survey designs, and protracted marine mammal life histories preclude discernment of meaningful short-term trends. Analyses by various investigators suggest several indicator populations are increasing (e.g., humpback whales, fin whales, gray whales, California sea lions), and that apparent decreases in some populations (e.g., blue whales) likely result from distributional shifts, not from changes in abundance. However, an unusual mortality event (UME) among California sea lion pups in 2012-2013 may be evidence of episodic changes in local sea lion feeding conditions (Wells et al. 2013).

INDICATORS OF "ECOLOGICAL INTEGRITY"

CCIEA scientists evaluate many integrative indicators of "ecological integrity," by which we mean the ability of an ecosystem to support and maintain communities that are comparable to those in less-disturbed reference habitats in the same region (Parrish et al. 2003). We are following indicators of two main aspects of ecological integrity: trophic structure and biodiversity.

Indicators of trophic structure reflect average to above-average conditions for consumers in the CCLME through 2013 (Table 8; see also Williams et al. 2014). The biomass anomaly of northern copepods, which are an energy-rich food source for planktivores, was relatively high off Oregon; in contrast, biomasses of gelatinous zooplankton species, which may prey on fish larvae or compete with forage fish for prey, were generally near long-term averages and showed negative trends in some areas and seasons. The proportion of scavengers increased relative to total demersal consumer biomass, largely driven by increased biomass of crabs. Finally, the mean trophic level of groundfishes was relatively low coastwide and even declining south of Cape Mendocino, due to relatively low abundances of two predators, Pacific hake *Merluccius productus* and spiny dogfish *Squalus suckleyi*. Reduced abundances of these predators may further promote good feeding conditions for competitors such as salmon, tunas, and seabirds.

Biodiversity indices (evenness and species richness) for the groups examined were within ±1 s.d. of the long-term mean, although some groups show significant trends in recent years (Table 8). Summer copepod biodiversity had a recent declining trend, consistent with greater amounts of the relatively less-diverse northern copepods that are richer energy sources, suggesting good feeding conditions for higher trophic levels in the pelagic community. Ichthyoplankton biodiversity had a recent increasing trend at a coast-wide scale, but a declining trend in the northern sampling locations, suggesting possible differences in ichthyoplankton ecology in the northern and southern regions of the system. Groundfish diversity has declined recently at the coast-wide scale, driven most strongly by declines south of Cape Mendocino, but evenness of groundfish has increased.

HUMAN DIMENSIONS

Human dimensions include archaeological and historical heritage, contemporary demographic patterns such as population growth and migration, individual and community behaviors, cultural values and trends, social relationships and social movements, political and economic systems, institutions and governance, and perhaps most importantly in this context, the many ways that humans are connected to the environment (Breslow et al. 2014). Because of the significant role that humans play as consumers of ecosystem services and engineers of ecosystem structure, and because of legislative mandates that require consideration of societal impacts of resource management decisions, human dimensions are essential attributes to include in a true ecosystem assessment. The CCIEA has only recently begun identifying and ranking indicators of human wellbeing. Thus we cannot yet comprehensively assess the status and trends of human wellbeing in coastal communities of the CCLME, apart from what might be assumed from the indicators of anthropogenic drivers and pressures alluded to earlier. However, at least one potential indicator within the fisheries sector implies declining wellbeing for some stakeholders: an annual index of diversity of fishery revenue sources is declining across regions, vessel sizes, and vessel income levels (Holland and Kasperski 2014). Lower revenue diversity is consistent with greater variability in annual income, and thus greater financial risk.

ASSESSMENT OF RISK OF KEY ECOSYSTEM COMPONENTS

Modeled response of ten coastal pelagic species to rising sea surface temperature and accompanying variability in chlorophyll-*a* concentrations to represent conditions in 2100 were used to assess risk due to climate change (Samhouri et al. 2014). The results suggest that risk for coastal pelagic species was highest in northern, coastal areas of the California Current and lower in southerly, offshore waters. The sensitivity of individual species to those changes was an order of magnitude greater than the exposure. The findings suggest that higher resolution climate models may be necessary to better resolve the variations.

Cumulative risks of 24 anthropogenic stressors to eight top predators (marine mammals, sea turtles and seabirds) in the U.S. west coast exclusive economic zone (EEZ) were assessed (Hazen et al. 2014b). Cumulative risks were greatest in nearshore areas, particularly within National Marine Sanctuary boundaries (in part because the Sanctuaries correspond with areas frequently used by top predators) and in hotspots near Point Arena and Monterey Bay. Climate change-related stressors posed the greatest risk due to their widespread distribution. The Sanctuary program may provide a basis for increased protection of top predators from human activities.

EVALUATION OF POTENTIAL MANAGEMENT STRATEGIES

Management strategy evaluation efforts in the Phase III CCIEA are focused on narrative scenarios that explore alternative future states of climate change, ocean acidification, and shipping activities (Kaplan et al. 2014). These scenarios, developed through scoping with resource managers, were evaluated through both qualitative and quantitative analyses. The key findings of these management strategy evaluations are summarized below.

Four studies considered the potential impacts of climate change. Two studies focused on how management could mitigate climate impacts on ESA-listed Chinook salmon (*Oncorhynchus tshawytscha*) in the Columbia River system. Crozier and Zabel (2014) found that spring/summer Chinook salmon in the Snake River system face high extinction risk if poor marine conditions (positive signs of the PDO index) increase in frequency; however, that risk can be mitigated almost entirely by actions that increase survival of smolts through dams in the Snake and Columbia rivers. Jorgensen et al. (2014) found that management to improve freshwater survival of pre-smolt juveniles was the best means of mitigating climate effects on Wenatchee River spring Chinook salmon; however, while cumulative management actions could mitigate moderate climate change effects, this population appears vulnerable to severe climate change. A third study, by Ruzicka (2014), examined the effects of interannual variability in a food web model of the northern California Current. Variability was imposed on key pelagic groups that are particularly sensitive to short-term climate variation: phytoplankton, copepods, large jellies, and forage fish. Variability in phytoplankton, due to forcing such as PDO and ENSO dynamics and upwelling, was a dominant structuring force, and strong community responses were also evoked by variability in jellies and forage fish. Interannual variability also affected fisheries: high forage fish years produced higher landings for gears targeting pelagic predators, while high euphausiid years supported greater landings for gears targeting hake and sablefish. These results serve as valuable hypotheses of how local climate conditions, climate variability, and community structure affect different ecosystem properties and fishery production.

The CCLME is potentially vulnerable to the ecological effects of ocean acidification (OA), a lowering of ocean pH and carbonate saturation due to increases in anthropogenic CO_2 (Busch et al. 2014). As part of the CCIEA Phase III report, Hodgson et al. (2014) present a risk analysis for different life history stages of Dungeness crab (*Metacarcinus* [formerly *Cancer*] *magister*) and pink shrimp (*Pandalus jordani*) to the effects of OA. Larval pink shrimp and post-settled megalops of Dungeness crab were the most vulnerable stages based on future spatial projections of OA effects; furthermore, all other life history stages of both species will also be exposed to OA. The effects are predicted to be worse in areas off California than off Washington, implying that fisheries effects will be felt strongest by fleets sailing from California ports.

Management strategy evaluation related to shipping first involved informal discussions with eight experts, who provided insight on expected shipping trends over the next 5-30 years (Kaplan et al. 2014). These discussions led to five potential scenarios that warrant more formal analysis and predictive modeling regarding their effects on California Current resources and human wellbeing. The scenarios were: (1) higher fuel prices, which would sustain reduced ship speeds but would not increase intra-national shipping between US ports; (2) economies of scale, which shift shipping fleets to relatively small numbers of very large ships that concentrate in the largest ports; (3) the widened Panama Canal will shift a large portion of container traffic from the US West Coast to the US East Coast; (4) clean fuel requirements, which will alter shipping routes and reduce ship speeds; and (5) North American energy development increases energy exports from the Pacific Northwest.

A more complete shipping scenario evaluation considers the potential for ship strikes on large whales in the Southern California Bight. Ship traffic in these waters shifted due to recent regulations requiring cleaner burning fuels in coastal waters; the revised routes are closer to military ranges and may also change the risk of ship strikes to several whale species. Redfern (2014) examined ship strike risk in several alternative routes and determined that a new southerly route could lower risk to fin whales (*Balaenoptera physalus*) and humpback whales (*Megaptera novaeangliae*) and also reduce use conflict with other sectors; however, risk to blue whales (*Balaenoptera musculus*) could not be lessened, which is problematic because blue whale mortalities may exceed allowable limits.

NEXT STEPS

This report, along with the initial CCIEA report by Levin and Schwing (2011) and the Phase II report (Levin et al. 2013), have contributed to defining and establishing the basic IEA tool kit of identifying and quantifying good indicators of key ecosystem attributes, developing methods to assess the risk of ecosystem components to natural and anthropogenic stressors, and building quantitative models for evaluating effectiveness and tradeoffs in different management strategies.

Our next effort, the Phase IV report, is targeted for completion in the summer of 2016. Provided that agency resources are suitable for continued CCIEA work, we hope to achieve several major goals in Phase IV, including:

- The first set of indicator time series for habitats;
- An expanded set of time series of human dimensions indicators, including the first set of human wellbeing indicators;
- Greater emphasis on management-relevant, integrated products, including quantitative analysis of relationships between indicators, more risk analyses, and more management strategy evaluations; and
- Products serving broader constituent needs—continuing to expand beyond just fisheries-focused products to serve management needs related to other sectors (e.g., shipping, energy development, etc.), protected resources, and National Marine Sanctuaries within the CCLME.

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FIGURE AND TABLES

For reasons of formatting and readability, Figure 1 and Tables 1-8 are presented on the following pages rather than being embedded within the text of this relatively short chapter.

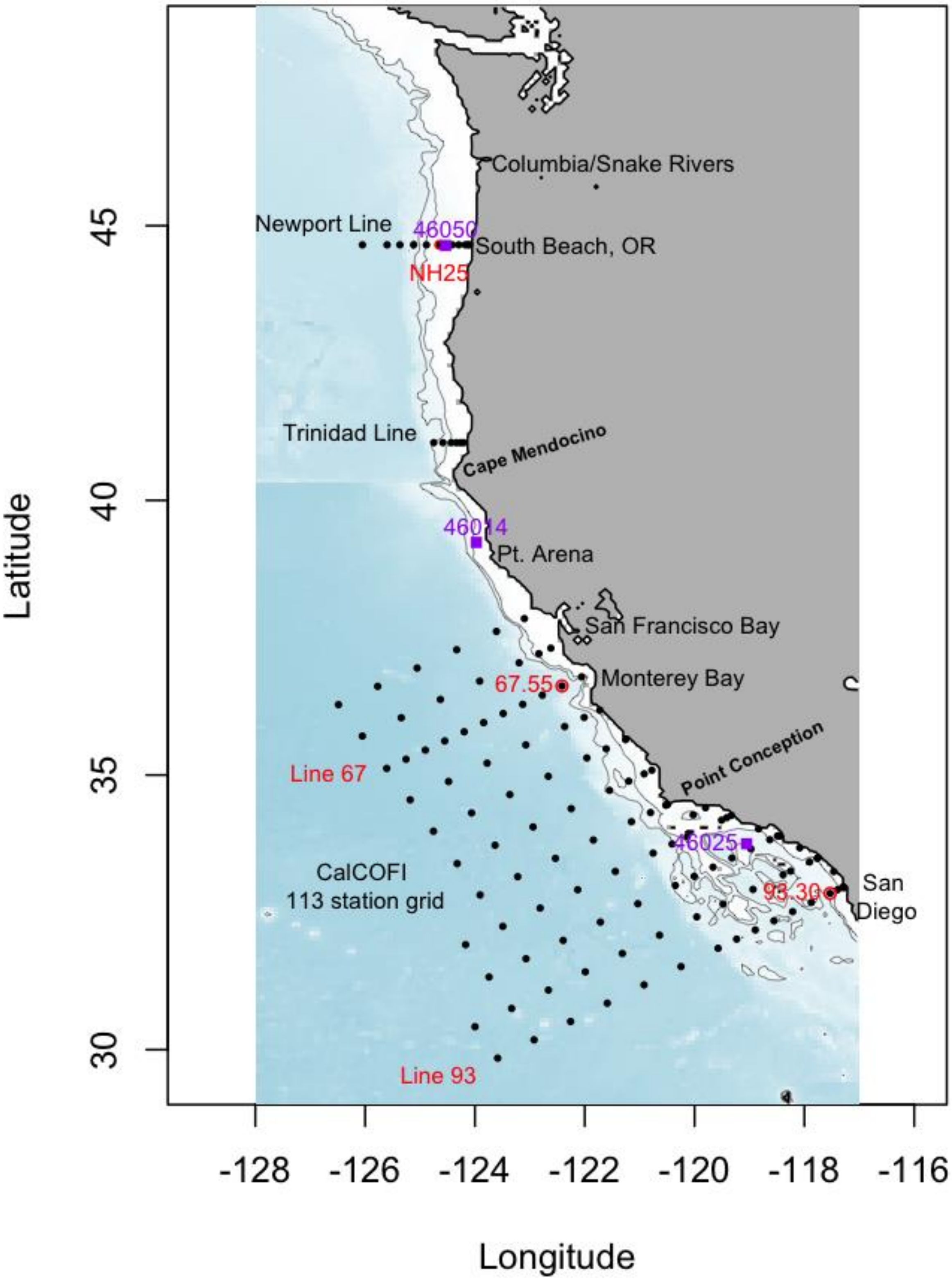

Figure 1. Map of the U.S. waters of the California Current large marine ecosystem (CCLME). Major headlands that demark ecoregional boundaries are labeled (Cape Mendocino, Point Conception), as are the locations of key sampling points that are referred to in Tables 1-8 or elsewhere in the text. Figure credit: Andrew Leising.

Table 1. Trends and status of physical, chemical and climate indicators in the CCLME. Indicators are sorted by season (columns) and location (rows, north to south except basin-scale indicators). Arrows represent the most recent 5-year trend (↗ increasing, ↔ no trend, ↘ decreasing); symbols represent status, i.e., the most recent 5-year mean relative to the long-term mean (– more than 1 s.d. below, ● within ± 1 s.d., + more than 1 s.d. above); colors indicate authors Harvey and Garfield's qualitative appraisal of trend or status as an indicator of overall ecosystem health (green: "good"; blue: "neutral"; red: "poor"; uncolored: inconclusive). Details and figures are in the chapter by Hazen et al. (2014).

		Temporal resolution						
		Monthly		Winter		Summer		Fig. in report
Indicator	Site	Trend	Status	Trend	Status	Trend	Status	
Multivariate El Niño Index	basin-scale	↔	●	↔	●	↔	●	OC27
Northern Oscillation Index	basin-scale	↔	●	↗	●	↔	●	OC8
North Pacific Gyre Oscillation	basin-scale	↔	+	↔	●	↔	●	OC28
Pacific Decadal Oscillation	basin-scale	↔	●	↔	●	↔	●	OC7
Eddy kenetic energy	45°N	↘	●	↔	●	↘	●	OC15
	39°N	↔	●	↔	●	↔	●	OC16
	33°N	↔	●	↔	●	↔	●	OC17
Upwelling Index	45°N	↔	●	↔	●	↗	●	OC18
	39°N	↔	●	↗	●	↗	●	OC19
	33°N	↔	●	↔	●	↗	●	OC20
Sea level height	So. Beach, OR	↔	●	↘	●	↔	●	OC1
	San Francisco	↔	●	↔	●	↔	+	OC2
	San Diego	↔	+	↔	+	↔	+	OC3
Sea surface temperature	NOAA Buoy 46050	↔	●	↔	●	↔	●	OC4
	NOAA Buoy 46014	↔	●	↘	●	↘	●	OC5
	NOAA Buoy 46025	↔	●	↔	●	↘	●	OC6
Meridional winds	NOAA Buoy 46050	↔	●	↔	●	↔	●	OC24
	NOAA Buoy 46014	↔	●	↘	●	↔	●	OC25
	NOAA Buoy 46025	↔	●	↔	●	↗	●	OC26
Pycnocline depth	NH25	↘	●	↔	●	↘	●	OC9
	CalCOFI 67.55	↔	●	↔	●	↘	●	OC10
	CalCOFI 93.30	↔	●	↔	●	↔	●	OC11
Pycnocline strength	NH25	↔	●	↗	●	↘	●	OC12
	CalCOFI 67.55	↔	●	↔	●	↘	●	OC13
	CalCOFI 93.30	↔	●	↔	●	↘	●	OC14
$NO_2 + NO_3$ @ 150 m	NH25	↔	●	↔	●	↔	●	OC29
	CalCOFI 67.55	↔	●	↗	●	↗	●	OC30
	CalCOFI 93.30	↔	●	↔	●	↗	●	OC31
Dissolved oxygen @ 150 m	NH25	↔	●	↔	●	↔	●	OC35
	CalCOFI 67.55	↔	●	↘	●	↘	●	OC36
	CalCOFI 93.30	↔	●	↔	●	↘	●	OC37

Table 2. Trends and status of additional physical, chemical and climate indicators in the CCLME (these indicators could not be sorted in the same seasonal and spatial manner as indicators in Table 1, and hence are presented separately). Arrows, symbols and colors are as in Table 1. Details and figures are in the chapter by Hazen et al. (2014).

Indicator	Site or season	Trend	Status	Figure in report
Spring transition Julian date	45°N	↗	●	OC21
	39°N	↘	●	OC21
	33°N	n/a		OC21
Length of upwelling season	45°N	↘	●	OC22
	39°N	↗	●	OC22
	33°N	n/a		OC22
Total upwelling magnitude	45°N	↗	●	OC23
	39°N	↗	●	OC23
	33°N	↗	●	OC23
Monthly total copepod biomass	NH Line	↔	●	OC32
Monthly copepod community composition	NH Line	↔	●	OC32
Monthly northern copepod biomass anomaly	NH Line	↔	●	OC33
Monthly southern copepod biomass anomaly	NH Line	↔	●	OC34
Multivariate Ocean Climate Index (MOCI)	Winter	↗	●	OC38
	Spring	↔	●	OC38
	Summer	↘	●	OC38
	Fall	↔	●	OC38

Table 3. Trends and status of indicators of fishery-related anthropogenic activities in the CCLME. Arrows and symbols are as in Table 1. No colors are used to qualitatively appraise these indicators' relationships to overall ecosystem health because such appraisals reflect value judgments that are societal rather than scientific in nature (for example, an increase in fishing could reflect a positive economic effect for a fleet but a negative effect on the population of the targeted fish species). Details and figures are in the chapter by Andrews et al. (2014).

Indicator	Site	Trend	Status	Figure in report
Total annual fisheries landings	Coast-wide	↗	●	AP0
Commercial fisheries				
Groundfish landings (w/o hake)	Coast-wide	↔	–	AP1
Pacific hake landings	Coast-wide	↗	●	AP2
Coastal pelagic species landings	Coast-wide	↔	+	AP3
Highly migratory species landings	Coast-wide	↔	●	AP4
Salmon landings	Coast-wide	↔	●	AP5
Crab landings	Coast-wide	↔	+	AP6
Shrimp landings	Coast-wide	↗	●	AP7
Shellfish landings	Coast-wide	↔	●	AP8
Other species landings	Coast-wide	↔	●	AP9
Total trawl landings	Coast-wide	↗	●	AP11
Shrimp trawl landings	Coast-wide	↗	●	AP12
Hook and line landings	Coast-wide	↔	–	AP13
Net gear landings	Coast-wide	↔	●	AP14
Pot and trap landings	Coast-wide	↔	+	AP15
Troll landings	Coast-wide	↔	●	AP16
Other miscellaneous gear landings	Coast-wide	↔	●	AP17
Total fishing mortality				
Groundfish (w/o hake)	Coast-wide	↘	●	AP19
Pacific hake	Coast-wide	↔	●	AP20
Fishing effects on habitat				
Total distance disturbed	Coast-wide	↘	●	AP22
Disturbance to shelf, hard substrate	Coast-wide	↔	●	AP23
Disturbance to shelf, mixed substrate	Coast-wide	↔	●	AP24
Disturbance to shelf, soft substrate	Coast-wide	↘	●	AP25
Disturbance to upper slope, hard substrate	Coast-wide	↔	●	AP26
Disturbance to upper slope, mixed substrate	Coast-wide	↘	●	AP27
Disturbance to upper slope, soft substrate	Coast-wide	↘	●	AP28
Disturbance to lower slope, hard substrate	Coast-wide	↔	●	AP29
Disturbance to lower slope, soft substrate	Coast-wide	↔	●	AP30

Table 4. Trends and status of indicators of non-fishery related anthropogenic activities in the CCLME. Arrows and symbols are as in Table 1. No colors are used to qualitatively appraise these indicators' relationships to overall ecosystem health because such appraisals reflect value judgments that are societal rather than scientific in nature (for example, an increase in an activity could reflect a positive economic effect for a sector but a negative effect on populations of some marine species). Details and figures are in the chapter by Andrews et al. (2014).

Attribute	Indicator	Site	Trend	Status	Figure in report
Aquaculture	Aquaculture production (finfish)	Coast-wide	↔	+	AP33
	Aquaculture production (shellfish)	United States	↗	+	AP34
Atmospheric pollution	Sulfate deposition	Coast-wide	↘	●	AP35
Benthic structures	# of offshore oil and gas wells	Coast-wide	↔	–	AP36
Coastal engineering	Coastal population	Coast-wide	↔	+	AP37
Commercial shipping	Vol. water disturbed in transit	Coast-wide	↘	●	AP38
Dredging	Vol. dredged sediments	Coast-wide	↗	●	AP39
Freshwater retention	Vol. freshwater stored behind dams	Coast-wide	↔	+	AP40
Inorganic pollution	Toxicity-weighted chemical releases	Coast-wide	↔	●	AP42
Invasive species	Tons of cargo moved through ports	Coast-wide	↘	●	AP43
Light pollution	Average nighttime light	Coast-wide	↔	●	AP45
Marine debris	Predicted debris counts	Northern CC	↗	●	AP46
		Southern CC	↔	●	AP46
Nutrient input	N + P fertilizer applications	Coast-wide	↘	+	AP47
Ocean-based pollution	Vol. water disturbed and cargo moved by shipping activities	Coast-wide	↘	●	AP48
Oil and gas activity	Oil and gas production	California	↔	–	AP49
Organic pollution	Toxicity-weighted pesticide conc.	Coast-wide	↘	●	AP50
Power plants	Vol. saline water withdrawals	Coast-wide	↔	+	AP51
Recreation	Beach attendance	Coast-wide	↘	●	AP52
Seafood demand	Consumption of fisheries products	United States	↔	+	AP53
Sediment retention	Vol. freshwater impoundments	Coast-wide	↔	+	AP54

Table 5. Trends and status of the abundance of pelagic forage in the CCLME, based on data from multiple monitoring programs. Results are sorted into northern, central, and southern regions of the CCLME. Arrows, symbols and colors are as in Table 1. Details and figures are in the chapter by Wells et al. (2014a). Blanks indicate insufficient data.

	Region						
	North		Central		South		
Indicator	Trend	Status	Trend	Status	Trend	Status	Fig. in report
Anchovy	↔	●	↔	●	↔	–	C2, C5, C7
Sardine	↔	●	↔	●	↔	●	C2, C5, C7
Pacific hake			↔	●	↗	–	C2, C5
Pacific sanddab larvae			↗	●	↗	●	C2, C5
Jack mackerel	↘	●			↔	●	C2, C7
Shortbelly rockfish larvae					↔	●	C2
Cool-water larvae					↗	●	C2
Warm-water larvae					↔	●	C2
Rockfish spp. larvae			↗	●			C2, C5, C7
Market squid			↗	●			C5
Krill			↔	●			C5
Pacific herring	↘	●					C7
Whitebait smelt	↗	●					C7

Table 6. Trends and status of the abundance and population condition of salmon in the CCLME. Populations are sorted from north to south. Arrows, symbols and colors are as in Table 1, except salmon status is based on the most recent 10 years of data (rather than 5 years as in Table 1). Condition indicators include the percent of spawners that are of natural origin, the population growth rate, and the diversity of age structure. Details and figures are in the chapter by Wells et al. (2014b). Blanks indicate insufficient data.

	Abundance		Condition						
			% natural		Pop. growth rate		Age diversity		
Species/population	Trend	Status	Trend	Status	Trend	Status	Trend	Status	Fig. in report
Chinook salmon									
Upper Columbia R. spring	↗	●	↔	●	↔	●	↔	●	S2, S4
Snake R. spr/sum	↗	●	↔	●	↔	●	↗	●	S2, S4
Snake R. fall	↗	●	↔	●	↔	●	↔	●	S2, S4
Willamette R. spring	↘	●	↗	●	↘	●	↗	●	S2, S4
Lower Columbia R.	↘	●	↘	–	↘	●	↘	●	S2, S4
S. OR / N. CA Coasts	↘	●	↔	●			↔	●	S2, S4
Klamath R. fall	↔	●	↗	●	↔	●	↔	●	S2, S4
California Coast	↔	●							S2
Central Valley winter	↘	●							S2
Central Valley late	↔	●							S2
Central Valley spring	↘	●							S2
Central Valley fall	↘	●	↘	●	↗	●			S2, S4
Coho salmon									
Lower Columbia R.	↗	●							S6
Oregon Coast	↔	●	↔	●	↔	●			S6, S8
S. OR / N. CA Coasts	↘	●							S6
California Coast	↘	●							S6

Table 7. Trends and status of abundance and population condition of groundfish in the CCLME. Indicators are derived from stock assessments or from trawl surveys conducted by the Northwest Fisheries Science Center (NWFSC). Indicators reflect biomass, the proportion of females that are mature, the cumulative 95% age distribution, and the cumulative 95% length distribution. Arrows, symbols and colors are as in Table 1. Details and figures are in the chapter by Cope and Haltuch (2014).

	Biomass				Population structure				
	Assessment		NWFSC survey		Assessment		NWFSC survey		Fig. in report
Species	Trend	Status	Trend	Status	p(mature)	95% age	p(mature)	95% length	
Elasmobranchs									
Longnose skate	↔	+	↔	●	↔	-	↔	↔	GF3, GF4, GF42, GF43
Spiny dogfish	↔	+			↔	-			GF5, GF44
Spotted ratfish			↔	●			+	↔	GF6, GF45
Flatfishes									
Arrowtooth fl.	+	+	+	●	-	-	-	-	GF7, GF8, GF46, GF47
Dover sole	↔	+			↔	↔			GF12, GF52
English sole	+	+			+	-	-	↔	GF9, GF48, GF49
Flathead sole			↔	●			↔	↔	GF13, GF53
Pacific sanddab			+	●			↔	↔	GF10, GF50
Petrale sole	↔	●			-	-			GF11, GF51
Rex sole	+	+					↔	↔	GF14, GF54
Rockfishes									
Aurora	↔	+			-	↔			GF31
Black	+	+			-	-			GF15
Blackgill	↔	●			-	-			GF32
Bocaccio	+	●			-	-			GF16
Canary	↔	-			-	-			GF17
Chilipepper	↔	+	↔	●	↔	-	-	+	GF18, GF19
Cowcod	↔	-			-	-			GF20
Darkblotched	+	●			-	-			GF21
Greenspotted	+	●			-	-			GF22
Greenstriped	+	+			↔	-			GF23
Pac. ocean perch	↔	-			-	-			GF24
Redstripe			↔	+			-	↔	GF25
Rougheye	+	+			↔	↔			GF33
Sharpchin	+	+							GF26
Shortbelly			↔	●			↔	↔	GF27
Splitnose	+	+			-	-			GF34
Stripetail			↔	●			↔	↔	GF28
Widow	+	+			↔	-			GF29
Yelloweye	↔	-			-	-			GF35
Yellowtail			↔	↔			↔	↔	GF30
Thornyheads									
Longspine	+	+			↔	↔			GF36
Shortspine	-	+			↔	↔			GF37
Roundfishes									
Cabezon	+	+			-	-			GF38
Lingcod	+	+			-	-			GF39
Pacific hake	+	+							GF40
Sablefish	-	●			↔	↔			GF41

Table 8. Trends and status of indicators of ecological integrity in the CCLME, arranged by community, site and/or season. Arrows, symbols and colors are as in Table 1. Details and figures are in the chapter by Williams et al. (2014).

Attribute/indicator	Site	Season	Trend	Status	Fig. in report
Trophic structure, pelagic community					
Northern copepod biomass anomaly	NH line	winter	↔	●	EI5
Northern copepod biomass anomaly	NH line	summer	↔	●	EI6
Aurelia abundance	Central CA	--	↔	●	EI7
Chrysaora abundance	Central CA	--	↔	●	EI7
Chrysaora abundance	OR/WA	June	↘	●	EI8
Chrysaora abundance	OR/WA	Sept	↘	●	EI9
Aequorea abundance	OR/WA	June	↗	●	EI8
Aequorea abundance	OR/WA	Sept	↔	●	EI9
Trophic structure, demersal community					
Groundfish mean trophic level	coast-wide	--	↔	●	EI10
Groundfish mean trophic level	N of Cape Mendocino	--	↔	●	EI11
Groundfish mean trophic level	S of Cape Mendocino	--	↘	●	EI12
Scavenger:total biomass ratio	coast-wide	--	↗	●	EI13
Scavenger:total biomass ratio	N of Cape Mendocino	--	↗	●	EI14
Scavenger:total biomass ratio	S of Cape Mendocino	--	↔	●	EI15
Crab scavengers:total biomass ratio	coast-wide	--	↗	●	EI16
Crab scavengers:total biomass ratio	N of Cape Mendocino	--	↗	●	EI16
Crab scavengers:total biomass ratio	S of Cape Mendocino	--	↔	●	EI16
Finfish scavengers:total biomass ratio	coast-wide	--	↔	●	EI17
Finfish scavengers:total biomass ratio	N of Cape Mendocino	--	↔	●	EI17
Finfish scavengers:total biomass ratio	S of Cape Mendocino	--	↔	●	EI17
Biodiversity, pelagic community					
Copepods, Simpson diversity	NH line	winter	↔	●	EI18
Copepods, Simpson diversity	NH line	summer	↘	●	EI18
Copepods, species richness	NH line	winter	↔	●	EI28
Copepods, species richness	NH line	summer	↘	●	EI29
Ichthyoplankton, Simpson diversity	Southern California	spring	↗	●	EI23
Ichthyoplankton, Simpson diversity	Southern California	summer	↗	●	EI24
Ichthyoplankton, Simpson diversity	Oregon	spring	↘	●	EI25
Ichthyoplankton, Simpson diversity	Oregon	summer	↔	●	EI26
Ichthyoplankton, species number	Southern California	spring	↗	●	EI38
Ichthyoplankton, species number	Southern California	summer	↗	●	EI39
Ichthyoplankton, species number	Oregon	spring	↔	●	EI40
Ichthyoplankton, species number	Oregon	summer	↘	●	EI41
Coastal pelagic fish, Simpson diversity	N of Cape Mendocino	June/Sept	↔	●	EI19
Coastal pelagic fish, species number	N of Cape Mendocino	June/Sept	↘	●	EI27
Biodiversity, demersal community					
Groundfish, Simpson diversity	coast-wide	--	↗	●	EI20
Groundfish, Simpson diversity	N of Cape Mendocino	--	↗	●	EI21
Groundfish, Simpson diversity	S of Cape Mendocino	--	↔	●	EI22
Groundfish, species richness	coast-wide	--	↘	●	EI30
Groundfish, species richness	N of Cape Mendocino	--	↔	●	EI31
Groundfish, species richness	S of Cape Mendocino	--	↘	●	EI32

ANTHROPOGENIC DRIVERS AND PRESSURES

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OVERVIEW

Fisheries landings of crab and shrimp have increased in recent years, while landings of salmon and groundfish remain at historically low levels. Many non-fisheries pressures (e.g., shipping activity, industrial pollution, recreational use) have decreased over the short term, possibly reflecting slowing economic conditions; however, seafood demand, dredging, and shellfish aquaculture may be increasing to historically high levels if short-term trends persist over the next few years. Methods were developed to examine these pressures as a whole in a way that could be used to investigate linkages and thresholds between multiple pressures and ecosystem components.

EXECUTIVE SUMMARY

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

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

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

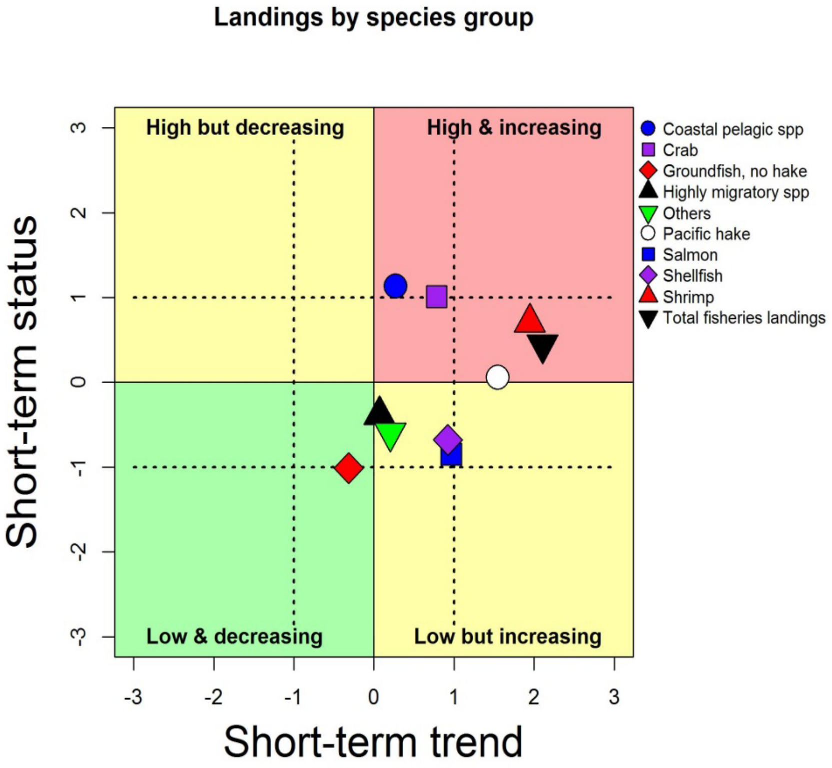


Figure AP.S.1. Short-term status and trends of annual landings (1981 – 2012) by species groups in the CCLME. Prior to plotting, time series data were normalized to place them on the same scale. The short-term trend indicates whether landings increased, decreased, or remained the same over the last five years. The short-term status represents the difference between the mean of the last five years and the mean of the full time series. Data points outside the dotted lines (1.0 standard deviation) are considered to be increasing or decreasing in the short term or show that the current status is lower or higher than the long-term mean of the time series.

Fisheries provide important services to society, including production of food, employment, livelihood, and recreation, but can also affect the ecosystem by directly removing individual fish and by disturbing habitat from the use of bottom trawls and other bottom-tended gear. Total mortality estimates are the best indicator of fisheries removals, but data are limited to very few years and are only calculated for groundfish species. Thus, we evaluated landings of catch as the best indicator of fisheries removals across the entire CCLME (Fig. AP.S.1). Landings of coastal pelagic species and crab were higher than historic levels over the last five years; Pacific hake, shrimp and total fisheries landings from commercial and recreational fishing increased over the short term; and landings of groundfish species (excluding hake) were at historically low levels for the last five years.

All other species groups were within historic landing levels. In addition, trawling effort showed a shift among habitat types, which corresponded, in part, to depth-related spatial closures implemented by the Pacific Fishery Management Council to reduce fisheries' impact on depleted species.

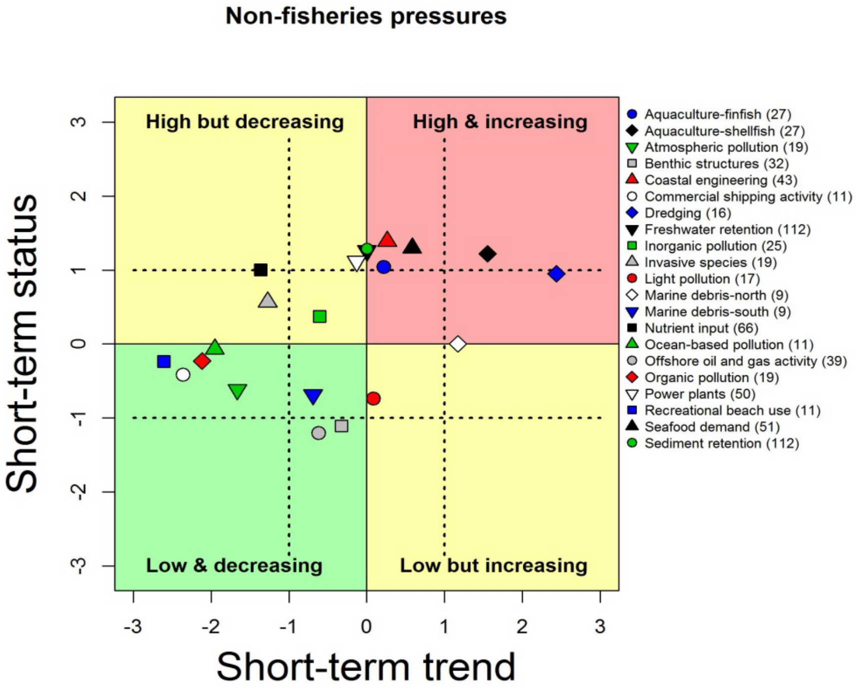


Figure AP.S.2. Short-term status and trends of non-fisheries pressures in the CCLME. See Fig. AP.S.1 for description of axes and interpretation of data points. Numbers in parentheses in the legend are the number of years in the time series for each pressure.

Most indicators of non-fisheries related pressures showed either significant short-term trends or their current status was at historically high or low levels (Fig. AP.S.2). Indicators of atmospheric, organic and ocean-based pollution, nutrient input, commercial shipping activity, recreational beach use and invasive species have all decreased over the short-term, while indicators of dredging, shellfish aquaculture, and marine debris (in the northern CCLME) increased. Indicators of seafood demand, finfish aquaculture, sediment and freshwater retention, power plant activity and coastal engineering remained relatively constant over the short-term, but were above historic levels, while indicators of offshore oil and gas activity and related benthic structures were constant over the short-term, but at historically low levels. Shellfish aquaculture is both at historically high levels and has been increasing over the last five years, whereas nutrient input is at historically high levels but has been decreasing over the last five years of the dataset.

Taken together, these results support two primary conclusions: 1) decreasing trends of several non-fisheries pressures (e.g., shipping related indicators, industrial pollution and recreational activity) potentially reflect slowing economic conditions over the last few years and 2) non-fisheries pressures at historically high levels have leveled off and are not continuing to increase, although seafood demand, shellfish aquaculture and dredging will likely be at historically high and increasing levels if current trends continue for the next couple of years (see specific time series data for each pressure in the *detailed report*).

The interpretation of the status and trends of these pressures may differ depending on the EBM component of interest. For example, a decreasing trend in fisheries removals may be "good" for rebuilding stocks of protected resources or it could be "bad" for the economies of vibrant coastal communities. In addition, none of these pressures act upon the ecosystem individually, and we have little understanding about whether the overall effects of multiple pressures will be additive, synergistic, or antagonistic on populations of interest. Nevertheless, we have developed methodology for reducing the large number of anthropogenic pressures into a smaller set of shared trends that could potentially be used to investigate linkages and thresholds between pressures and ecosystem components (see "Appendix AP1"). In addition, subsequent sections of the IEA begin to integrate the cumulative effects of multiple pressures on multiple EBM components (see "risk" sections for each EBM component and the various management strategy evaluations in the rest of the CCIEA). Moreover, these anthropogenic pressures will interact with the underlying effects of climatic and oceanographic pressures (detailed in *Oceanographic and Climatic Drivers and Pressures*). The integration of anthropogenic, oceanographic, and climatic pressures on multiple EBM components can now be modeled using various "end-to-end" ecosystem models (e.g., Atlantis; Fulton et al. 2011), but marine ecologists and fisheries scientists need to develop creative methods in the field to test the validity of these models' hypotheses and increase managers' confidence in decision making.

DETAILED REPORT

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

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

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

ANTHROPOGENIC DRIVERS AND PRESSURES

As human population size and demand for seafood increases globally and within the CCLME, numerous human activities in the ocean (e.g., fishing and shipping activity) and on land (e.g., agricultural and industrial activities) need to be recognized and incorporated into management of marine resources. However, data on the status and trends of these human-related pressures are often buried in state agency reports, described at small spatial scales and measured inconsistently among local, state and federal entities. Here, we attempted to gather and produce the best available time series data on anthropogenic pressures across the entire CCLME. These data sets are intended to quantify relative changes in anthropogenic pressures and provide the foundation for subsequent integrative analyses of cumulative effects on multiple EBM components (e.g., Appendix AP1, risk analysis and management strategy evaluations).

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

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

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

General ecosystem impact	Type	Identified pressures
Habitat loss	Smothering	Benthic structures Dredging Sediment input
	Obstruction	Benthic structures Coastal engineering Ocean mining
Habitat modification	Siltation	Freshwater retention Sediment input Dredging Coastal engineering Ocean mining
	Abrasion	Commercial shipping activity
	Conversion	Habitat destruction Dredging Aquaculture
Non-physical disturbance	Noise	Commercial shipping activity Tourism
	Visual	Recreational use Light pollution Coastal engineering Tourism
Toxic contamination	Introduction of synthetic compounds	Inorganic pollution Atmospheric pollution Marine debris Ocean-based pollution
	Introduction of non-synthetic compounds	Offshore oil and gas activity Ocean-based pollution
Non-toxic contamination	Nutrient enrichment	Nutrient input
	Organic enrichment	Organic pollution
	Changes in thermal regime	Power plants
	Changes in turbidity	Freshwater retention Power plants Sediment input Dredging
	Changes in salinity	Freshwater retention Power plants
Biological disturbance	Introduction of microbial pathogens	Aquaculture
	Introduction of non-native species	Invasive species
	Translocations or aggregation of individuals	Coastal engineering Benthic structures Offshore oil & gas activity Marine debris Ocean mining
	Extraction of species	Fisheries removals Seafood demand

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

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

DATA ANALYSIS AND PRESENTATION

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

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

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

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

Importantly, the pressures identified below do not act upon the ecosystem individually, but collectively. Pressures from terrestrial-based pollution, shipping, offshore energy development, fisheries and coastal development exert cumulative effects on the ecosystem and should be managed in a holistic way (Vinebrooke et al. 2004, Crain et al. 2008, Halpern et al. 2008, Curtin and Prellezo 2010, Stelzenmüller et al. 2010). However, quantifying the cumulative effects of these pressures is a difficult task primarily because our understanding of whether effects are additive, synergistic or antagonistic is relatively poor (Darling and Côté 2008, Hoegh-Guldberg and Bruno 2010). To conclude this section on anthropogenic pressures (see Appendix AP1), we employ three methods to summarize the temporal patterns of anthropogenic pressures as a whole in the CCLME. First, we create two cumulative pressures indices across a time period for which we have data for the greatest number of pressures. We rely on the work by Halpern et al. (2009, 2012) and Teck et al. (2010) to develop these indices. We then use two different types of dimension-reducing analyses—principal components analysis (Link et al. 2002) and dynamic factor analysis (Zuur et al. 2003a, 2003b) — to identify correlations and common trends among pressures and to reduce the number of multivariate dimensions to a smaller set that explains most of the variance across all pressures.

Two goals for future iterations of the CCIEA will be to (1) identify and evaluate the 'status' of a pressure relative to specific target levels for each indicator, and (2) identify thresholds of pressures that may identify 'tipping points' in indicators of other EBM components of the CCIEA. Establishing specific target levels of a pressure (e.g., fisheries landings quotas or concentration of nitrogen in coastal waters) is a critical step in the management and policy planning process (Samhouri et al. 2012). Placing the current status of an indicator into context with historic levels or with management goals allows managers to determine whether the current status and trend of a specific pressure is moving in the right direction or whether alternative management strategies are necessary. Target levels have been established for many of these pressures in general terms (Halpern et al. 2012), and we will refine these values specifically for the CCLME.

Thresholds represent a level of a pressure (oceanographic or anthropogenic) at which small changes produce large changes in some metric of interest. In this case, we would want to identify thresholds of anthropogenic pressures (e.g., nutrient loading) that affect specific indicators of EBM components in the CCIEA. This could be done using individual pressures or the results from our cumulative pressures indices or the results from our dimension-reducing analyses. We propose to identify nonlinearities in the relationships between indicators of EBM components and pressures (Samhouri et al. 2010).

FISHERIES PRESSURES

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

FISHERY REMOVALS

BACKGROUND

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

Fisheries are rarely selective enough to remove only the desired targets (Garcia et al. 2003), and they often take other species incidentally, along with targets. Even though incidentally taken fish (often referred to as bycatch) are routinely discarded, discard mortality can be quite high, especially for deep-water species. Therefore, fisheries can significantly reduce abundance of bycatch species associated with removals of targeted resources as well. Unintended removals can be also be facilitated by lost (or dumped) fishing gear, particularly pots, traps and gillnets, which may cause entanglement of fish, marine mammals, turtles and sea birds. The extent of such “ghost” fishing in the CCLME is unknown, but studies conducted elsewhere suggest that the impact might be non-trivial (Fowler 1987, Goni 1998, Garcia et al. 2003).

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

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

EVALUATION AND SELECTION OF INDICATORS

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

The best source for information on stock-specific fishery removals is typically stock assessments that report landings, estimate amount of discard, and evaluate discard mortality. Stock assessments also provide the longest time series of removals, commonly dating back to the beginning of exploitation. Stock assessments conducted for CCLME species are available via the Pacific Fishery Management Council website (http://www.pcouncil.org) by species and year of assessment. However, not all species from each fishery have been assessed. For non-assessed stocks, information on fishery removals can be obtained from a variety of state and federal sources. The most detailed and reliable CCLME fishery landing data are summarized in the Pacific Fisheries Information Network (PacFIN) (http://pacfin.psmfc.org), a regional fisheries database that manages fishery-dependent information in cooperation with the National Marine Fisheries Service (NMFS) and West Coast state agencies. The data in PacFIN go back to 1981. NMFS and its predecessor agencies, the U.S. Fish Commission and Bureau of Commercial Fisheries, has also been reporting fishery landing statistics collected via comprehensive surveys of all U.S. coastal states conducted since 1951. These data are available via NMFS Science and Technology website at (http://www.st.nmfs.noaa.gov/st1/commercial/index.html. Recreational catches since the late 1970’s can be found in the Recreation Fisheries Information Network (RecFIN) (http://www.recfin.org), a project of the Pacific States Marine Fisheries Commission.

There have been a few historical studies conducted to evaluate discard in CCLME fisheries (Pikitch et al. 1988, Sampson 2002), but those studies focused on specific areas and/or species groups, so that thorough analysis would be needed to extrapolate those estimates to other areas, species and years. Currently there are two observer programs operated by the NMFS NWFSC on the U.S. West Coast. These programs include the At-Sea Hake Observer Program (A-SHOP), which monitors the at-sea hake processing vessels, and the West Coast Groundfish Observer Program (WCGOP), which monitors catcher vessels that deliver their catch to a shore-based processor or a mothership. The A-SHOP dates back to the 1970s, while WCGOP was implemented in 2001. The WCGOP began with gathering data for the limited entry trawl and fixed gear fleets. Observer coverage has expanded to include the California halibut trawl fishery, the nearshore fixed gear and pink shrimp trawl fishery. Since 2011, the U.S. West Coast groundfish trawl fishery has been managed under a new groundfish catch share program. The WCGOP provides 100% at-sea observer monitoring of catch for the new, catch share based Individual Fishing Quota (IFQ) fishery, including both retained and discarded catch.

Since 2002, the WCGOP has been generating estimates of groundfish total fishing mortality from commercial, recreational and research sources including incidental catch from non-groundfish fisheries. For groundfish, WCGOP total fishing mortality estimates were selected as an indicator of fishery removal, recognizing that the data to inform this indicator are only available for the most recent years. For other species groups, the PacFIN landings were selected as the best long-term fishery removal indicator, since they represent the bulk of removals for most species and have been routinely reported. However, if available, a total mortality estimate would be the preferred indicator for all species groups, due to its higher evaluation in the “Primary considerations” criteria (Table AP2).

STATUS AND TRENDS

The status of total removals was measured using: 1) combined commercial and recreational landings of all taxa and fishing gears as reported by the Pacific Fisheries Information Network (PacFIN) at http://pacfin.psmfc.org and by the Recreational Fisheries Information Network (RecFIN) at http://www.recfin.org for Washington, Oregon, and California; 2) commercial landings, by species group (groundfish, coastal pelagic species, highly migratory species, salmon, crab, shrimp, shellfish and others) and gear (trawl, shrimp trawl, hook and line, net gear, pot and trap, troll, and other miscellaneous gear), as reported by PacFIN for Washington, Oregon and California, and 3) for groundfish, total fishing mortality estimates generated and provided by the West Coast Groundfish Observer Program (WCGOP; Table AP3).

Total fisheries landings – This indicator represents all commercial and recreational landings reported to PacFIN and RecFIN. These estimates represent the best estimate of total fisheries removals from waters off the U.S. West Coast. These data do not include estimates of bycatch that are often discarded at sea; however, comparison of the trends in commercial landings data (e.g. Figs. AP1 & AP2) and total mortality estimates (e.g. Figs. AP19 & AP20) for groundfish and Pacific hake show similar trends. This suggests that landings data are able to capture much of the annual variability in total mortality for targeted species.

Figure AP0 shows that total fisheries landings have increased over the last five years in the CCLME, and the short-term mean was within one standard deviation of the long-term mean of the entire time series. This increasing trend is likely the result of a large rebound in landings of Pacific hake *Merluccius productus* from 2009 to 2013 (see Fig. AP2). Commercial fisheries landings drive the status and trends of this indicator; thus recreational fisheries landings may warrant their own indicator in future iterations of the CCIEA.

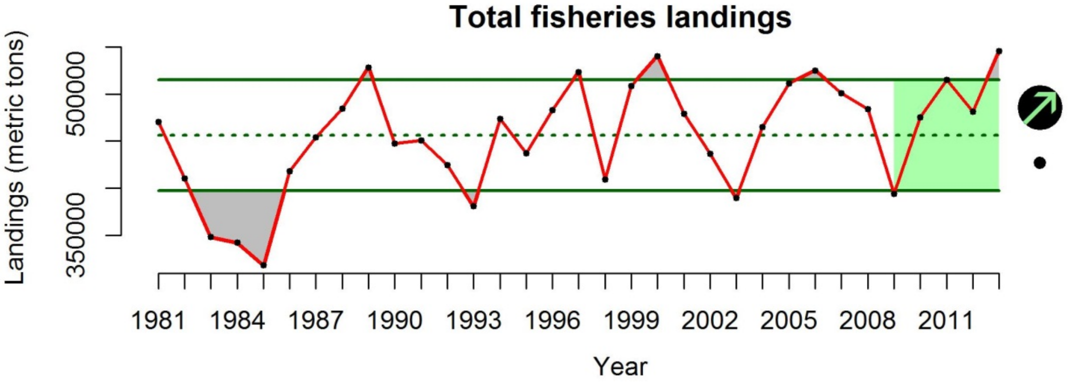


Figure AP0. Annual landings of all commercial and recreational fishing in the CCLME from 1981 – 2013.

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

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

Since 1981, commercial landings of groundfish species (other than Pacific hake), salmon and shellfish have generally decreased, in part due to management measures (Figs. AP1, AP5, AP8). Pacific hake, coastal pelagic species and crab have exhibited a positive long-term trend in landings (Figs. AP2, AP3, AP6), although over the short-term Pacific hake (Fig. AP2) and shrimp (Fig. AP7) have been increasing. Highly migratory species did not change significantly over the last 40 years, apart from the peak reported in the early 1980s (Figs. AP4). Relative to the mean of the entire time series, landings of coastal pelagic species and crab have been higher over the last five years, and landings for groundfish excluding hake have been at consistently low levels over the last five years. All other species groups have been relatively constant within historic landing levels (Fig. AP10).

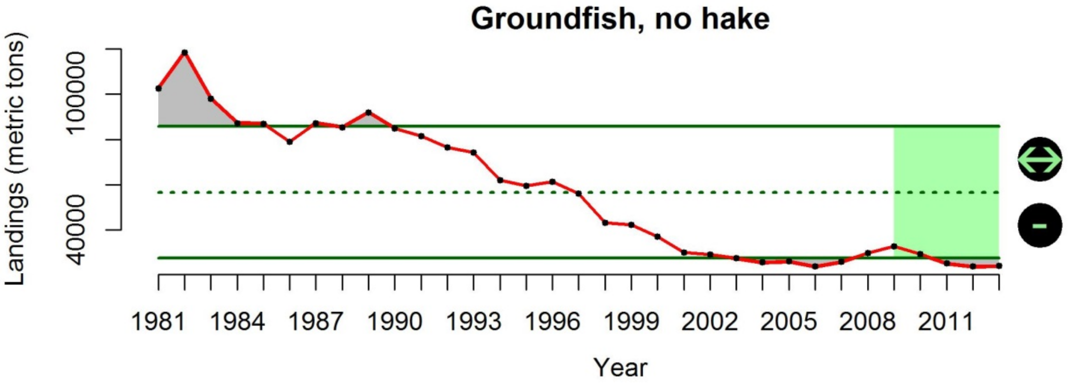


Figure AP1. Annual landings of groundfish in the CCLME from 1981 – 2013 (Pacific hake *Merluccius productus* excluded).

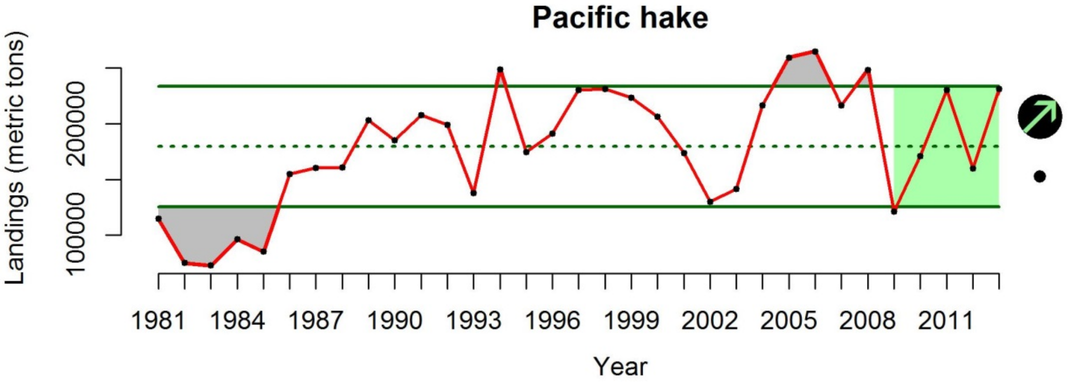


Figure AP2. Annual landings of Pacific hake *Merluccius productus* in the CCLME from 1981 – 2013.

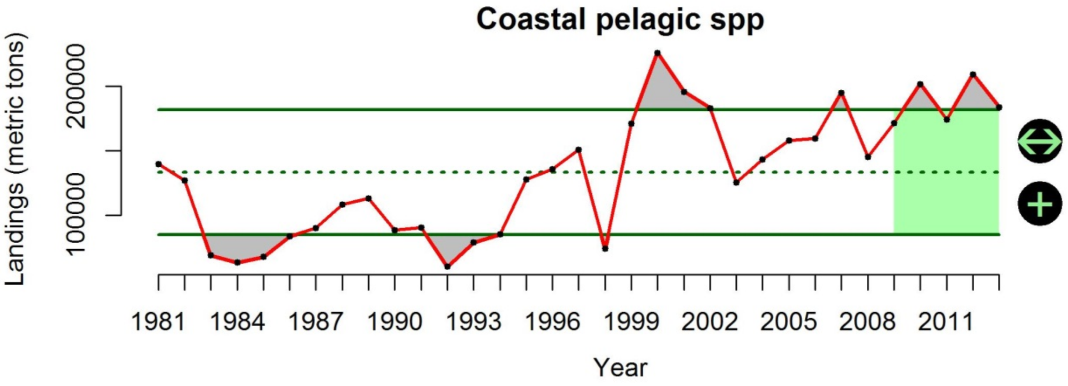


Figure AP3. Annual landings of coastal pelagic species (CPS) in the CCLME from 1981 – 2013. CPS include Pacific sardine *Sardinops sagax*, Pacific mackerel *Scomber japonicus*, northern anchovy *Engraulis mordax*, jack mackerel *Trachurus symmetricus*, and market squid *Loligo opalescens*).

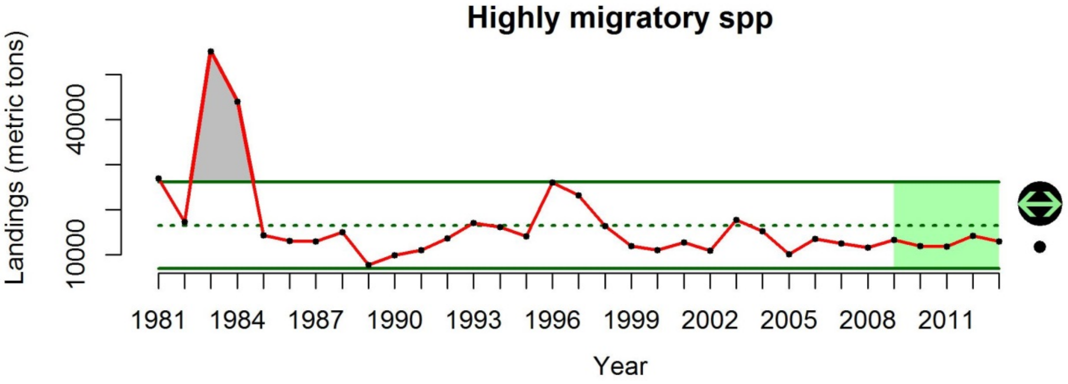


Figure AP4. Annual landings of highly migratory species (HMS) in the CCLME from 1981 – 2013. HMS include tunas, sharks, billfish/swordfish and dorado *Coryphaena hippurus*.

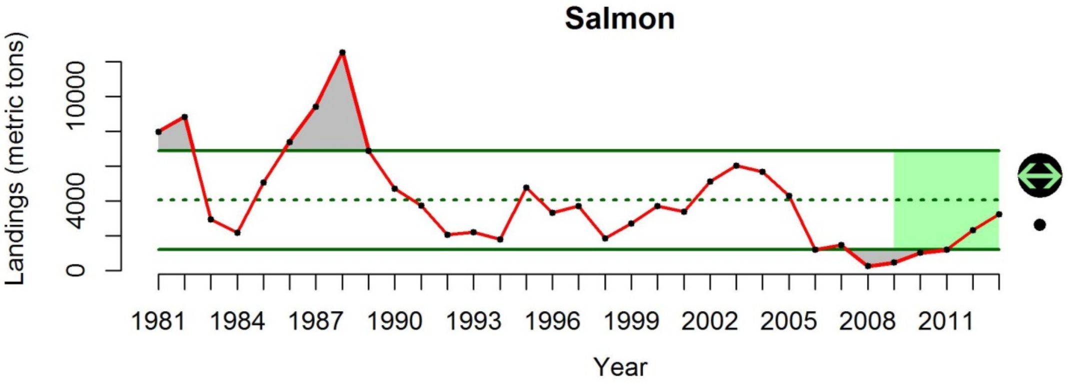


Figure AP5. Annual landings of salmon in the CCLME from 1981 – 2013.

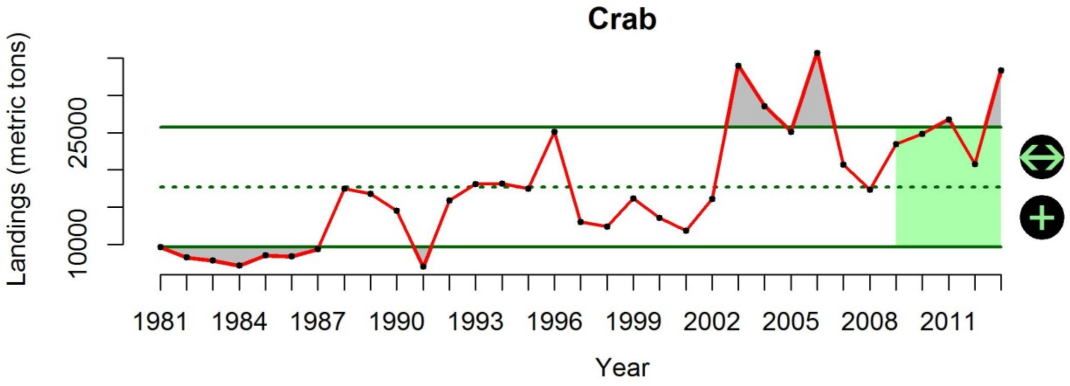


Figure AP6. Annual landings of crab in the CCLME from 1981 – 2013.

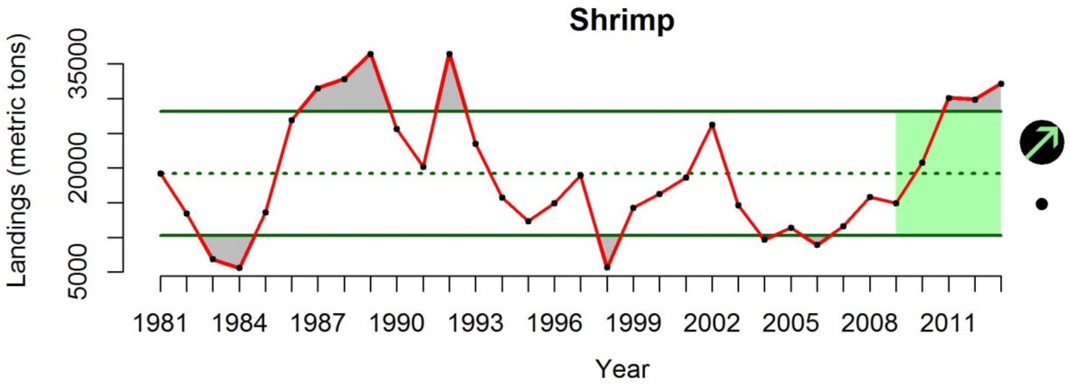


Figure AP7. Annual landings of shrimp in the CCLME from 1981 – 2013.

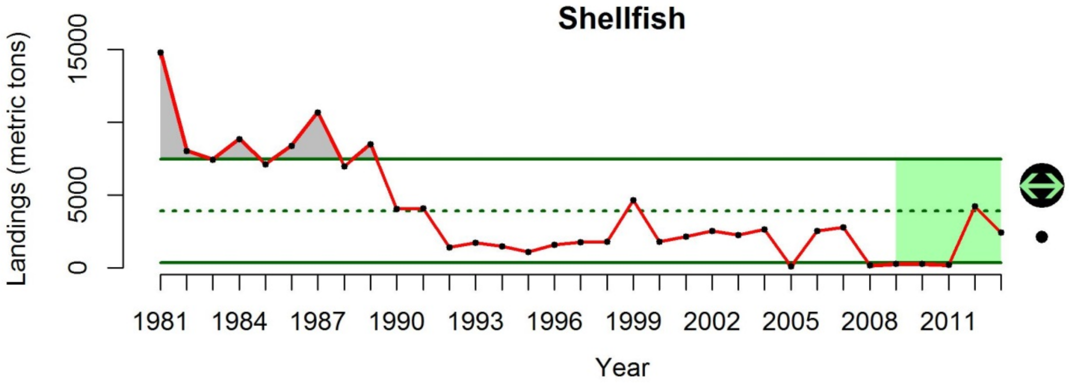


Figure AP8. Annual landings of shellfish in the CCLME from 1981 – 2013.

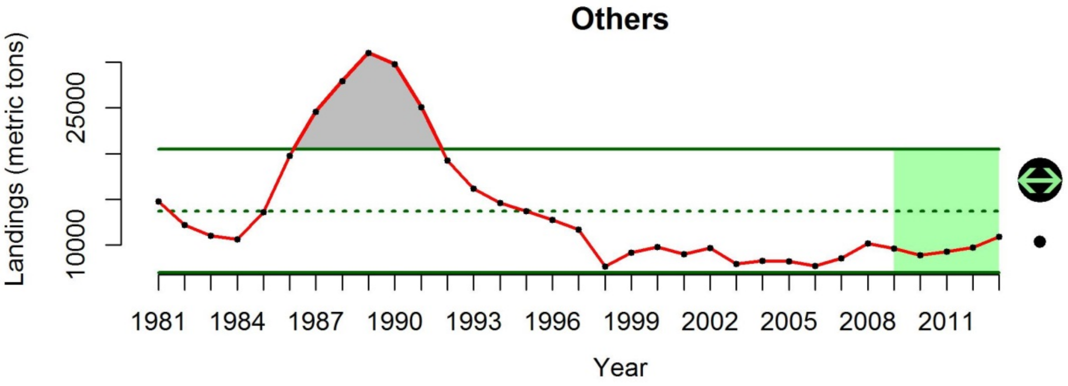


Figure AP9. Annual landings of all other species in the CCLME from 1981 – 2013.

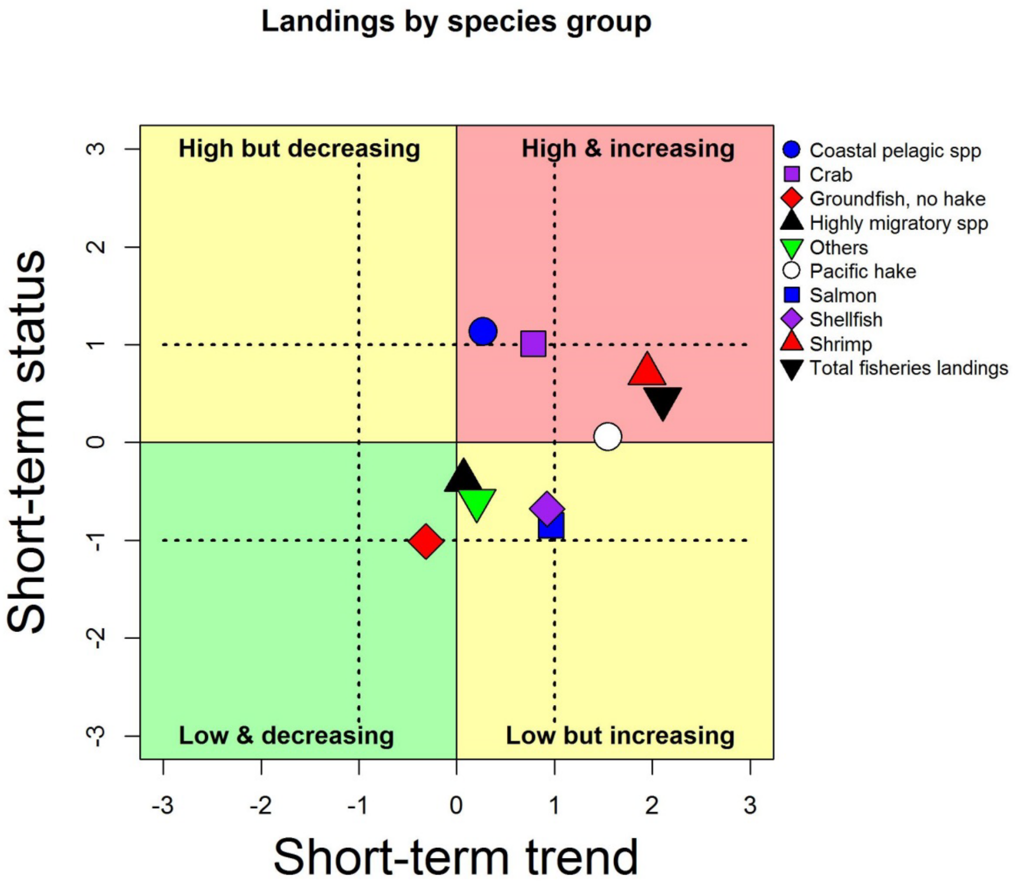


Figure AP10. Short-term status and trends of annual landings (1981 – 2013) by species groups in the CCLME. Prior to plotting, time series data for each indicator were normalized to place them on the same scale. The short-term trend indicates whether landings increased, decreased or remained the same over the last five years. The short-term status represents the difference between the mean of the last 5 years and the mean of the full time series. The dotted lines represent ± 1.0 SD; thus, data points outside the dotted lines are considered to be increasing or decreasing over the short term or the current status is lower or higher than the long-term mean of the time series.

Landings made by most gear types varied considerably over the last 40 years (Figs. AP11 – AP17), but hook-and-line landings (Fig. AP13) exhibited a decreasing trend since the late-1980's while net gear (Fig. AP14) and trolling (Fig. AP16) landings have steadily increased since the early 1990's. Over the last five years, trawl and shrimp trawl landings increased (Figs. AP11 & AP12), while landings made by other gear types did not exhibit clear trends. Hook-and-line landings were below historical landing levels (Fig. AP18) while pot and trap landings were above historical landing levels (Fig. AP15).

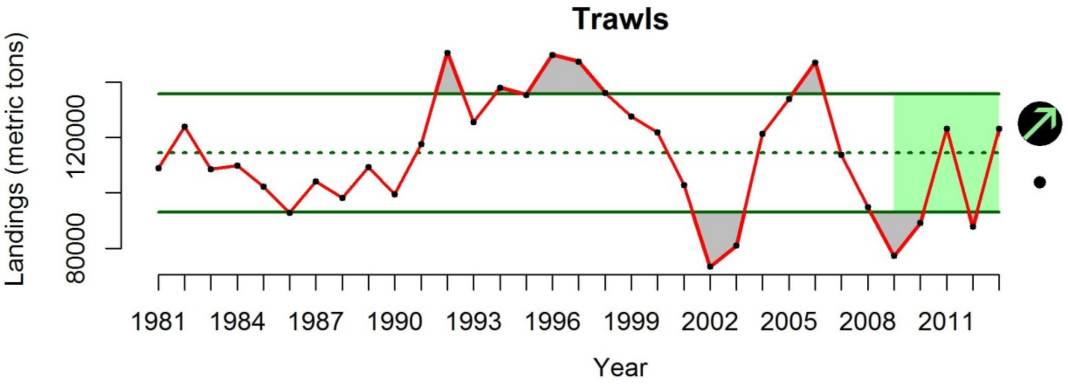


Figure AP11. Annual commercial trawl landings in the CCLME from 1981 – 2013.

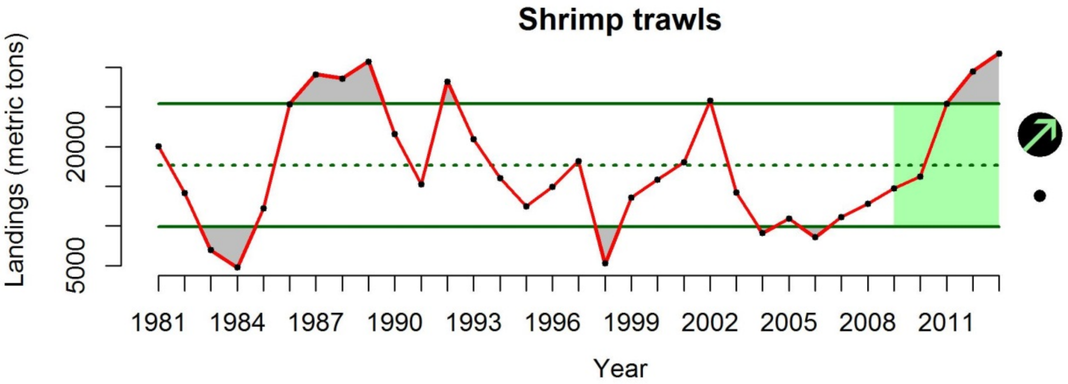


Figure AP12. Annual commercial shrimp trawl landings in the CCLME from 1981 – 2013.

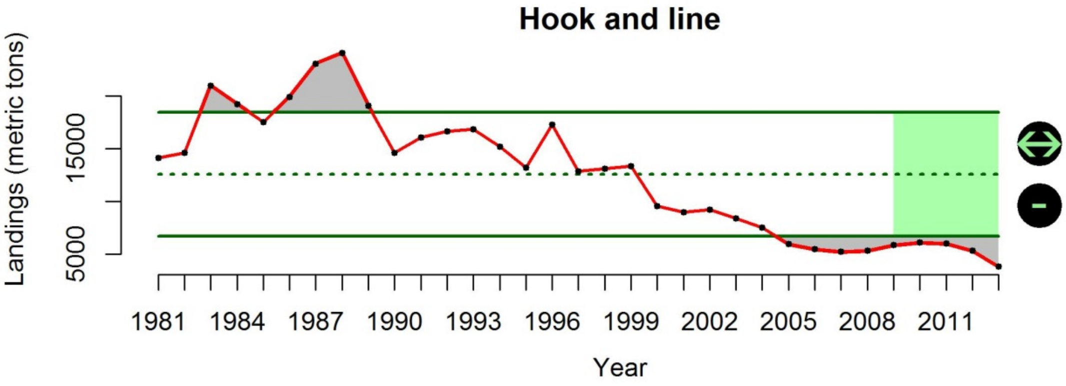


Figure AP13. Annual hook-and-line landings in the CCLME from 1981 – 2013.

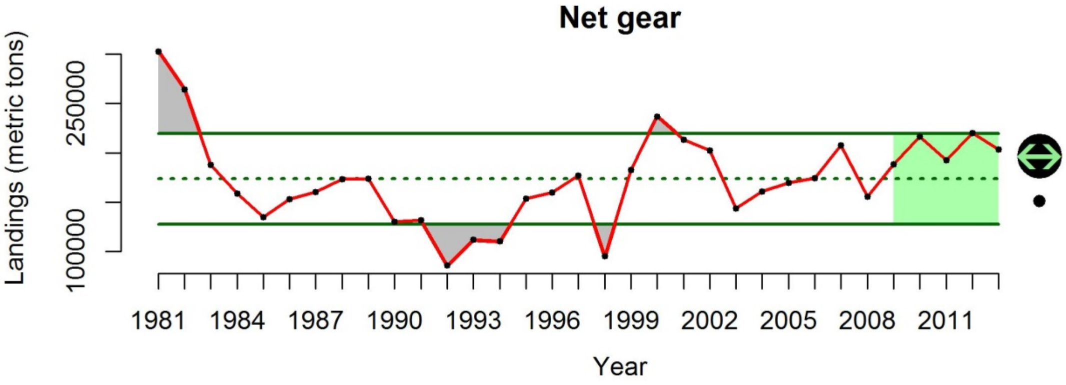


Figure AP14. Annual net-gear landings in the CCLME from 1981 – 2013.

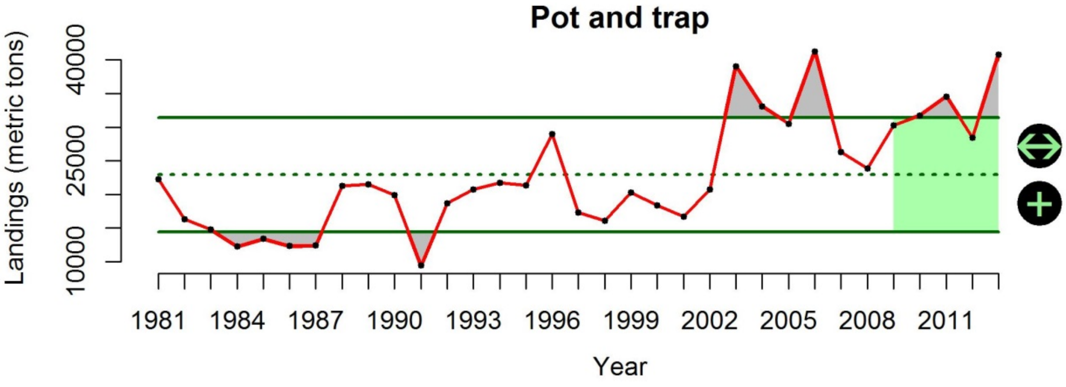


Figure AP15. Annual pot and trap landings in the CCLME from 1981 – 2013.

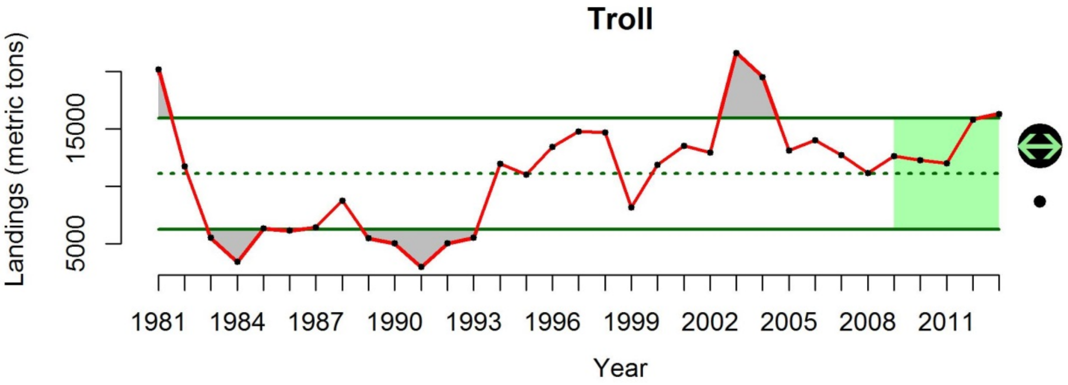


Figure AP16. Annual troll-caught landings in the CCLME from 1981 – 2013.

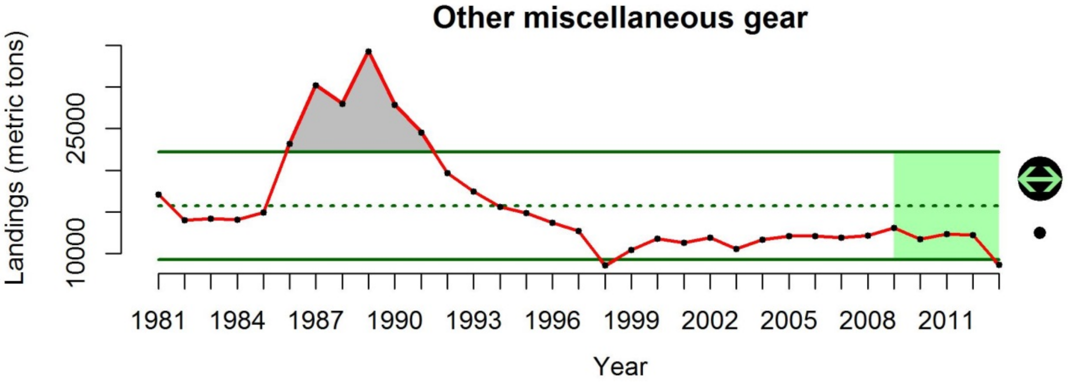
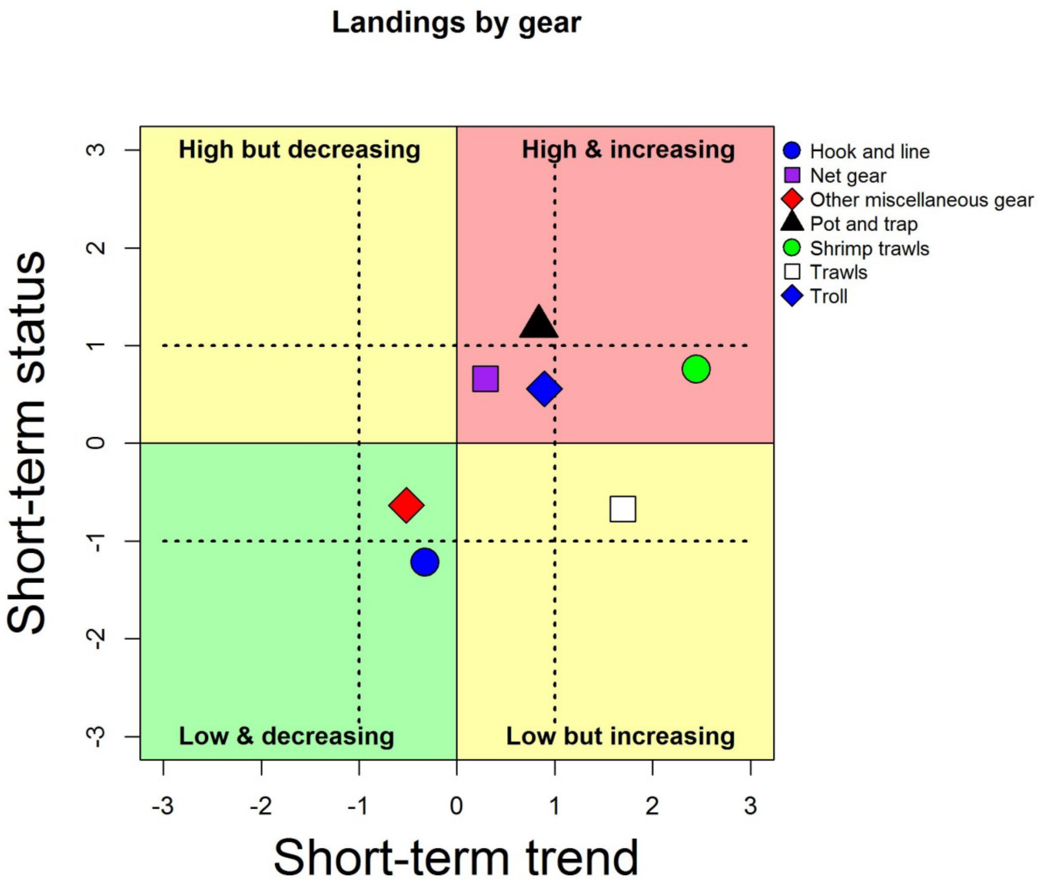


Figure AP17. Annual landings of all other miscellaneous gear in the CCLME from 1981 – 2013.



Figures AP18. Short-term status and trends of annual landings (1981 – 2013) by gear type in the CCLME. Prior to plotting, time series data for each indicator were normalized to place them on the same scale. The short-term trend indicates whether landings increased, decreased or remained the same over the last five years. The short-term status represents the difference between the mean of the last 5 years and the mean of the full time series. The dotted lines represent ± 1.0 SD; thus, data points outside the dotted lines are considered to be increasing or decreasing over the short term or the current status is lower or higher than the long-term mean of the time series.

Total fishing mortality estimates (groundfish only)– This indicator represents the total removals of groundfish species from a suite of fishery-dependent and fishery-independent sources, including shoreside commercial fisheries and at-sea hake removals, tribal and recreational catches, as well as incidental catch of groundfish in non-groundfish fisheries. It also includes removals from the research surveys conducted within the CCLME. As in the case of groundfish landings, total fishing mortality estimates of Pacific hake are reported separately. The Pacific hake fishery is the largest (in weight) on the U.S. West Coast, and, when combined with other species, total mortality of the Pacific hake overwhelms the total mortality for the entire group, and obscures changes in catch of other groundfish species. Over the last 5 years, total fishing mortality estimates for groundfish species decreased (Fig. AP19), while those of Pacific hake showed no change (Fig. AP20). The trends associated with estimates for this indicator are nearly identical to the trends found in commercial landings for these two groups across these years (see Figs. AP1 &

AP2). This is also evident in Fig. AP21, which compares short- versus long-term trends in total fishing mortality estimates for Pacific hake and other groundfish species.

Figure AP19. Total fishing mortality estimates of groundfish (Pacific hake *Merluccius productus* excluded) in the CCLME from 2002 - 2012.

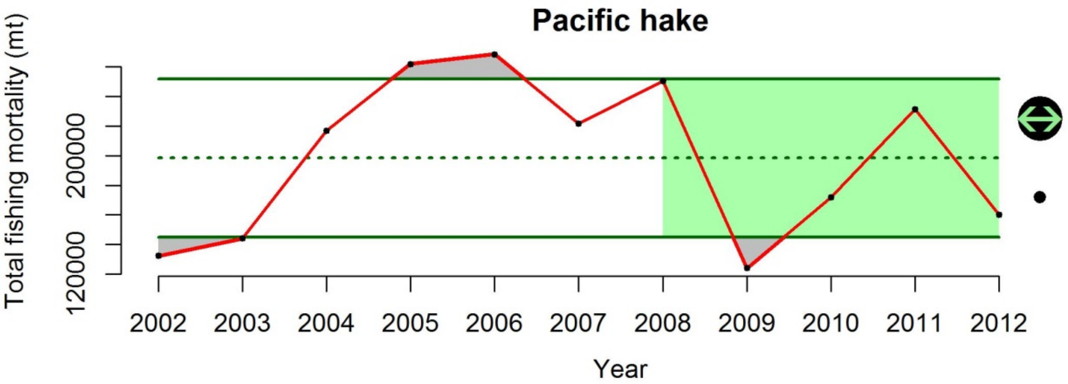


Figure AP20. Total fishing mortality estimates of Pacific hake *Merluccius productus* in the CCLME from 2002 - 2012.

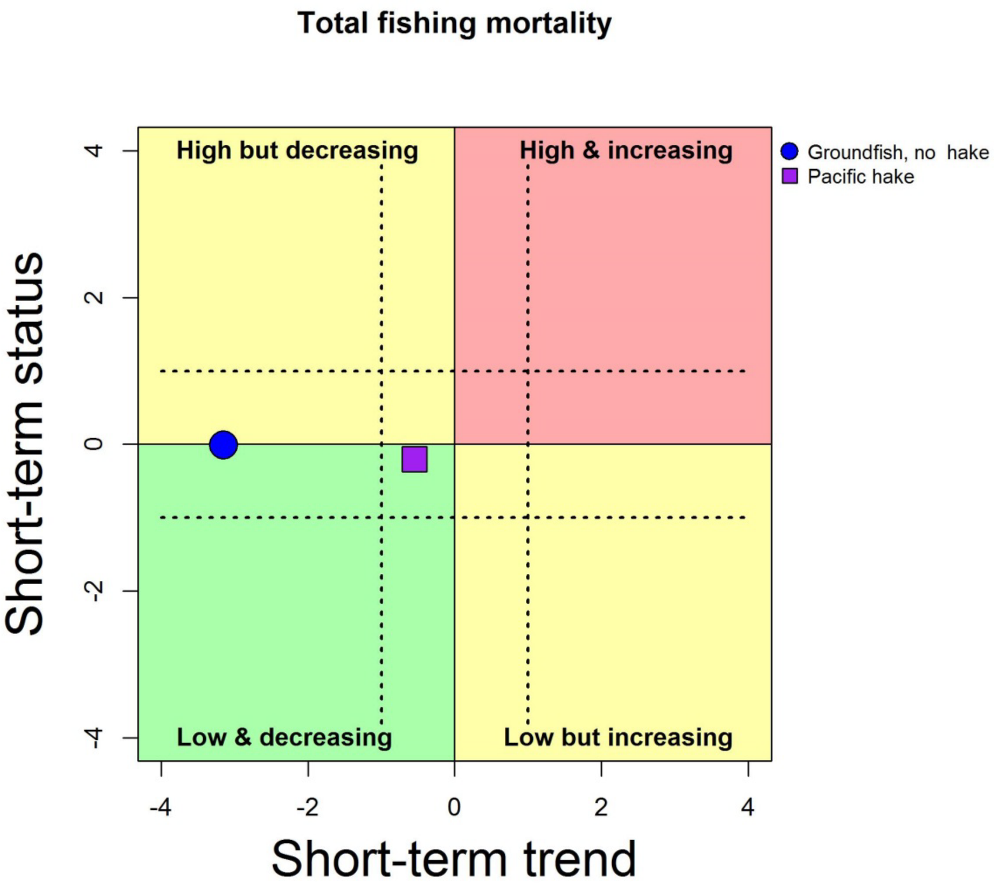


Figure AP21. Short-term status and trends of annual total fishing mortality (2002 – 2012) by species groups in the CCLME. Prior to plotting, time series data for each indicator were normalized to place them on the same scale. The short-term trend indicates whether total fishing mortality increased, decreased or remained the same over the last five years. The short-term status represents the difference between the mean of the last 5 years and the mean of the full time series. The dotted lines represent ± 1.0 SD; thus, data points outside the dotted lines are considered to be increasing or decreasing over the short term or the current status is lower or higher than the long-term mean of the time series.

HABITAT MODIFICATION

BACKGROUND

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

In the CCLME, implementation of Essential Fish Habitats (EFH), areas necessary for fish spawning, breeding, feeding, or growth to maturity, and Marine Protected Areas (MPA), in combination with gear regulation measures, have been used to reduce adverse impact of fisheries on vulnerable habitats. Also, the introduction of the Cowcod Conservation Area (CCA) and Rockfish Conservation Areas (RCAs) as management measures to prevent overfishing makes additional areas along the coast inaccessible to fishing during some or all of the year.

EVALUATION AND SELECTION OF INDICATORS

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

Bellman and Heppell (2007) estimated distance trawled within the limited entry groundfish trawl fishery in the U.S. West Coast by habitat type, defined based on type of bottom substrate. The habitat types considered were of four basic categories, including shelf, slope, basin and ridge, and two subcategories, rocky and sedimentary. Logbook data were used to obtain information on vessel, date, time and location of each individual tow as well as gear used (Bellman and Heppell 2007). These data were then overlaid with GIS seafloor habitat maps off Washington, Oregon and California compiled by Goldfinger et al. (2003), Romsos (2004) and Green & Bizzarro (2003). In addition, logbook data on trawling and fixed gear locations from 2002 – 2012 were entered into the same GIS framework (NMFS 2013).

We used estimates of coast-wide distances trawled from 1999 – 2004 (Bellman et al. 2007) and 2002 – 2012 (NMFS 2013) as an indicator for habitat modification (Table AP3). The estimates from 2002 – 2012 also include estimates of habitat modified by fixed fishing gear. Set and retrieval location of pot, trap and longline gear allowed for an estimate of the amount of bottom habitat disturbed (NMFS 2013). Distances for bottom trawling and fixed gear were summed to determine total amount of habitat modification from 2002 to 2012. Estimates from 1999 to 2004 did not include fixed gear distances estimates, but the overall distances of fixed gear are approximately 1% of the distances trawled; thus we simply incorporated the estimates for 1999 – 2001 from the previous data set into the more complete data from 2002 – 2012. Different habitat substrate types were used in the classification of the two data sets, so we limited habitat specific data to the longest data set

(NMFS 2013), while including data from both data sets in the total habitat disturbed estimate.

STATUS AND TRENDS

The status and trends of habitat modification were measured using distance trawled and distance disturbed by fixed gear by habitat type, made by the groundfish bottom-trawl fishery and the fixed-gear fishery, as estimated by Bellman and Heppell (2007) and NFMS (2013). Overall, distance trawled declined coast-wide over the last five years (Fig. AP22). During this period, the majority of habitat modification occurred in soft upper slope habitat (Fig. AP28), followed by the soft shelf habitat (Fig. AP25). A shift in trawling effort between habitat types was observed during the mid-2000's (Figs. AP23 to AP30), which in part corresponded to depth-related spatial closures implemented by the Pacific Fishery Management Council to reduce fisheries' impacts on depleted species (Bellman and Heppell 2007). If compared to the mean for the entire time series, the distance disturbed across all habitats has been within historic levels; however, if the magnitude of disturbance continues at similarly low levels as observed over the last two years (2011 – 2012), habitat modification may be at historically low levels in a few years (Fig. AP22 & AP31). Moreover, the implementation of a new management framework (i.e. Individual Transferable Quota system) in January 2011 has caused some of the fishing community to switch from trawling gear to fixed gears. Fixed gear has less impact on bottom habitats than trawling gear (NMFS 2013), further reducing the overall impact of fishing gear on habitats necessary for the diversity of seafloor communities. Reductions in the pressure may not coincide with recovery times of habitat depending on how fast recovery happens, which is likely to differ among habitat types (e.g., hard and mixed habitats will take longer to recover than soft habitat).

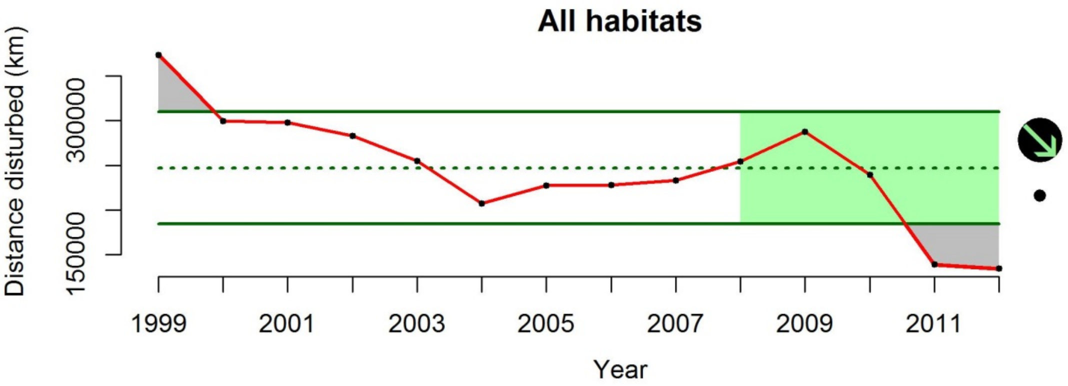


Figure AP22. Total distance disturbed (km) across all habitat types along the coast of Washington, Oregon and California by bottom-trawl and fixed-gear fisheries.

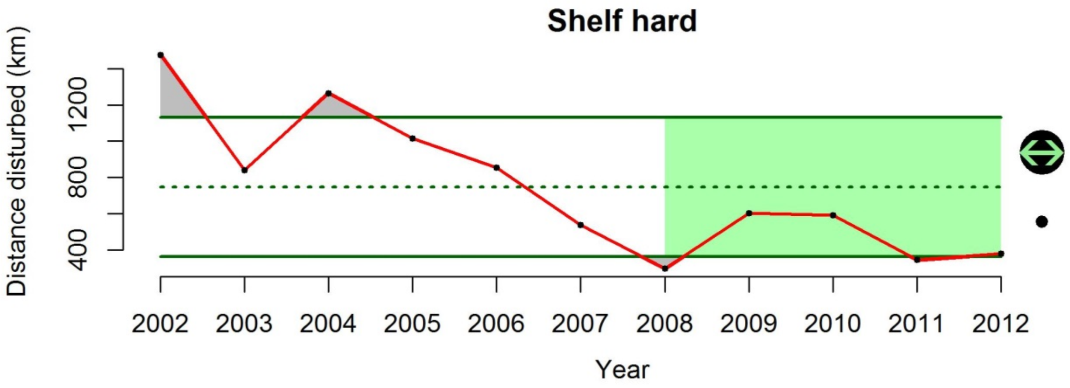


Figure AP23. Distance disturbed (km) within hard, shelf habitats along the coast of Washington, Oregon and California by bottom-trawl and fixed-gear fisheries.

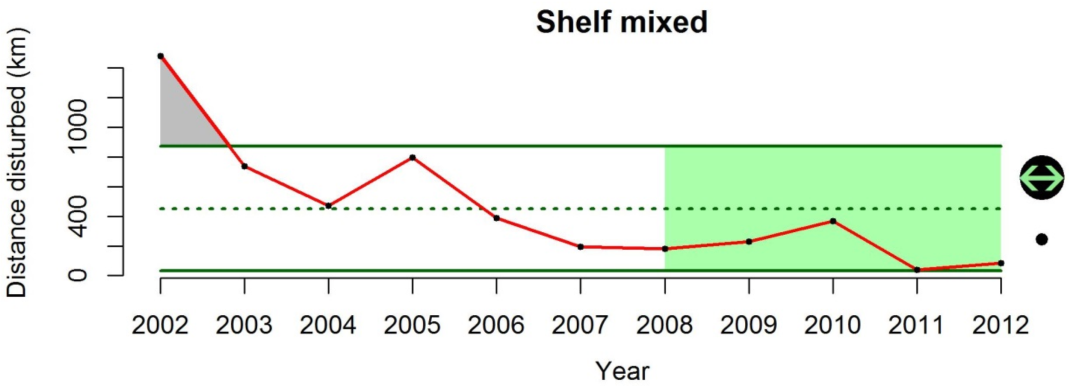


Figure AP24. Distance disturbed (km) within mixed, shelf habitats along the coast of Washington, Oregon and California by bottom-trawl and fixed-gear fisheries.

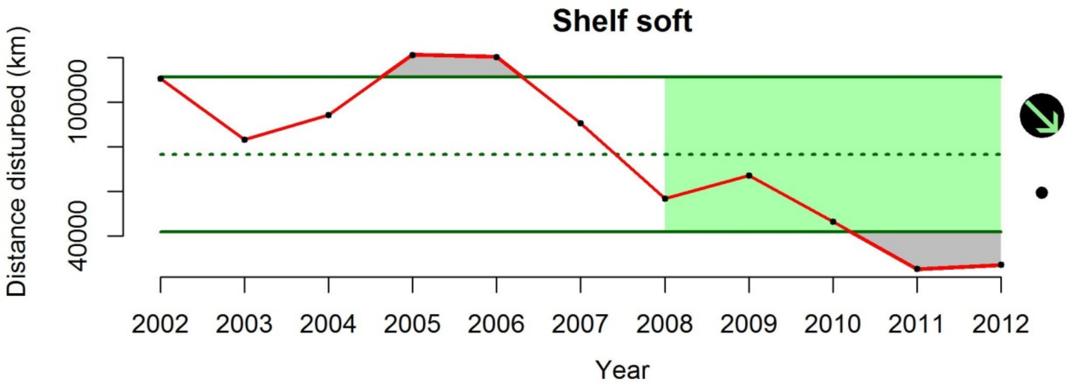


Figure AP25. Distance disturbed (km) within soft, shelf habitats along the coast of Washington, Oregon and California by bottom-trawl and fixed-gear fisheries.

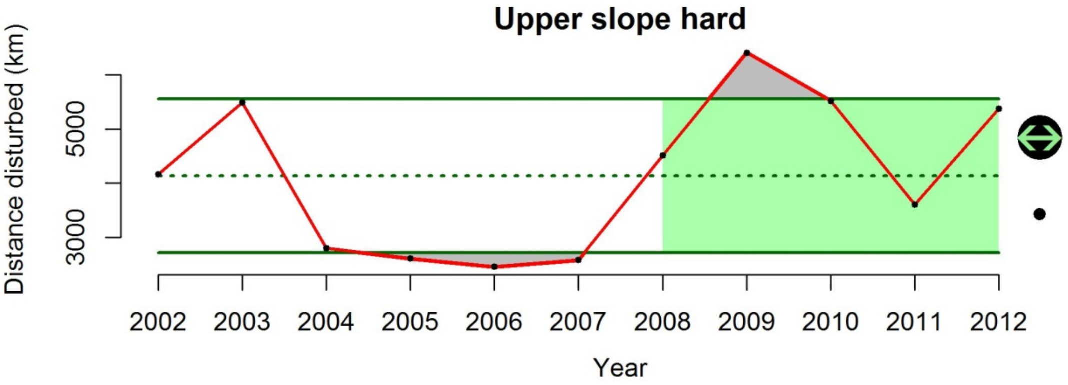


Figure AP26. Distance disturbed (km) within hard, upper slope habitats along the coast of Washington, Oregon and California by bottom-trawl and fixed-gear fisheries.

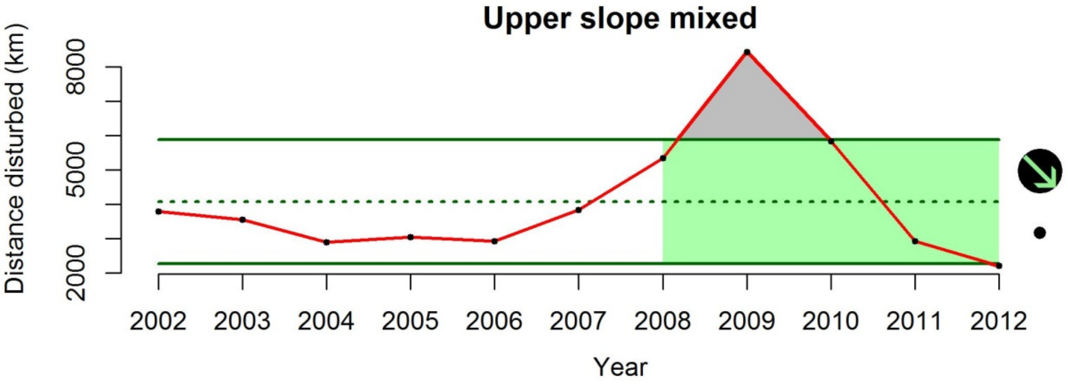


Figure AP27. Distance disturbed (km) within mixed, upper slope habitats along the coast of Washington, Oregon and California by bottom-trawl and fixed-gear fisheries.

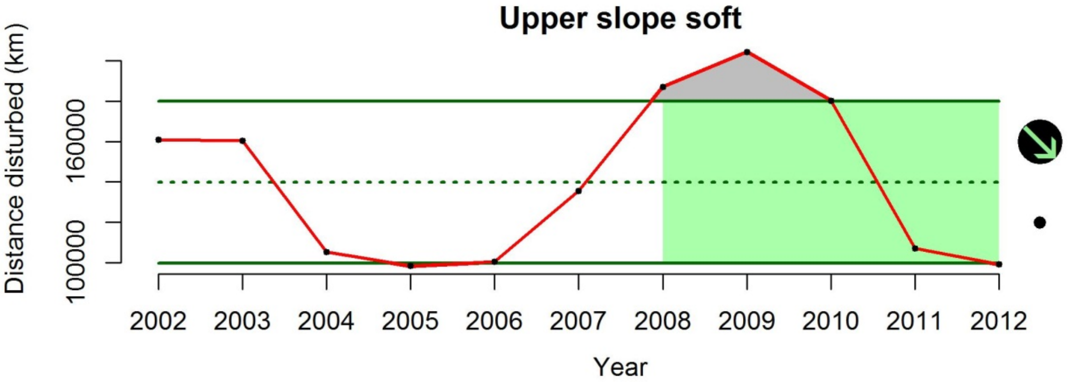


Figure AP28. Distance disturbed (km) within soft, upper slope habitats along the coast of Washington, Oregon and California by bottom-trawl and fixed-gear fisheries.

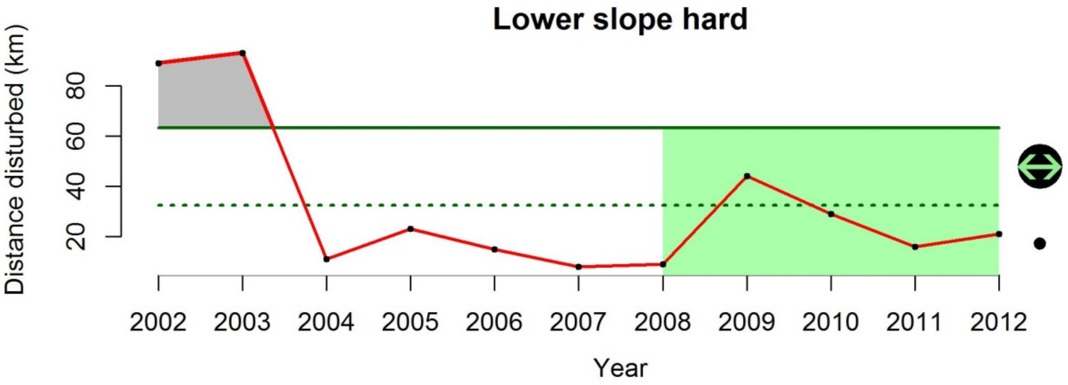


Figure AP29. Distance disturbed (km) within hard, lower slope habitats along the coast of Washington, Oregon and California by bottom-trawl and fixed-gear fisheries.

Figure AP30. Distance disturbed (km) within soft, lower slope habitats along the coast of Washington, Oregon and California by bottom-trawl and fixed-gear fisheries.

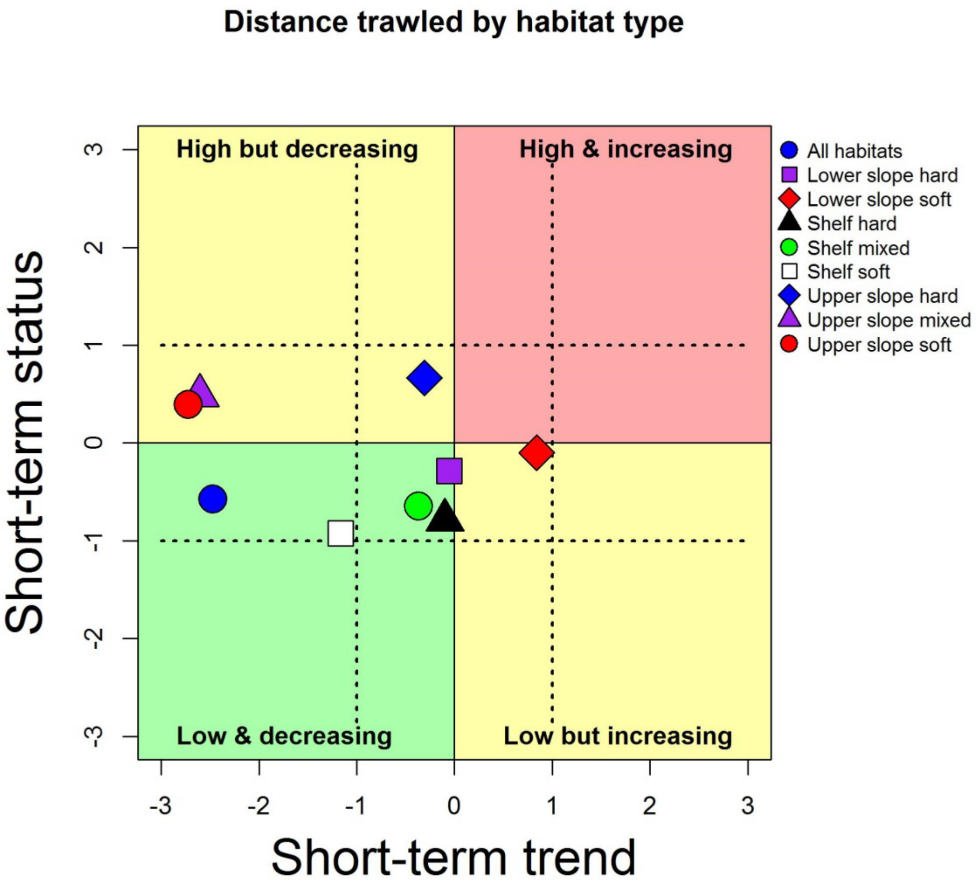


Figure AP31. Short-term status and trends of total distance disturbed across all habitats (1999 – 2012) and by habitat type (2002 – 2012) in the CCLME. Prior to plotting, time series data for each indicator were normalized to place them on the same scale. The short-term trend indicates whether distance trawled increased, decreased or remained the same over the last five years. The short-term status represents the difference between the mean of the last 5 years and the mean of the full time series. The dotted lines represent ± 1.0 SD; thus, data points outside the dotted lines are considered to be increasing or decreasing over the short term or the current status is lower or higher than the long-term mean of the time series.

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

Pressure	**Indicator**	**Primary consider-ations (5)**	**Data consider-ations (7)**	**Other consider-ations (6)**	**Summary comments**
Fisheries removals	Landings	3	7	4	Commercial landings represent the majority of removals for most species. This metric does not include discarded catch. Landings records from 1981 forward are available via http://pacfin.psmfc.org.
Fisheries removals	Groundfish total fishing mortality estimates	5	4	4	Groundfish total fishing mortality estimates are generated by the West Coast Groundfish Observer Program. These estimates are for groundfish only. The data are available from 2002 forward.
Habitat modification	Distance trawled	2	2	1	Coast-wide estimates of distance trawled by habitat type were generated by Bellman and Heppell (2007) and NMFS (2013) distance disturbed by bottom-trawl and fixed-gear fisheries based on logbook data on each individual tow (or set) and GIS seafloor habitat maps. These estimates are available from 1999 to the present.

Table AP3. Top indicators for fisheries pressures.

Pressure	Indicator	Definition and source of data	Time series	Sampling frequency
Fishery removals	Landings	Metric tons and pounds of the species landed by commercial fisheries in CA, OR and WA. Data are available from the Pacific Fisheries Information Network at http://pacfin.psmfc.org.	1981 – Present	yearly
Fishery removals	Total mortality estimates	Metric tons and pounds of the groundfish species removed by commercial, recreational and research sources as well as incidental catch from non-groundfish fisheries in CA, OR and WA. Data are available from the West Coast Groundfish Observer Program in the FRAM division of NOAA's Northwest Fisheries Science Center.	2002 – Present	yearly
Habitat modification	Distance trawled	Kilometers (km) disturbed by bottom-trawl and fixed-gear fisheries in CA, OR and WA by habitat type. Data are available from Bellman and Heppell (2007) and NMFS (2013).	1999-present	yearly

NON-FISHERIES PRESSURES

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

SUMMARY OF NON-FISHERIES PRESSURES

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

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

The interpretation of the status and trends of these pressures may differ depending on the EBM component of interest. For example, a decreasing trend in fisheries removals may be “good” for rebuilding stocks of Protected Resources or it could be “bad” for the economies of Vibrant Coastal Communities. In addition, none of these pressures act upon the ecosystem individually (i.e. many pressures are acting simultaneously on populations), and we have little understanding about whether the cumulative effects of multiple pressures will be additive, synergistic or antagonistic on populations of interest. Subsequent sections of the IEA begin to integrate the cumulative effects of multiple pressures on multiple EBM components (see “risk” sections for each EBM component and the various management strategy evaluations). Moreover, these anthropogenic pressures will interact with the underlying effects of climatic and oceanographic pressures (detailed *in Oceanographic and Climatic Drivers and Pressures*). The integration of anthropogenic, oceanographic and climatic pressures on multiple EBM components can now be modeled using various “end-to-end” ecosystem models (e.g., Atlantis; Fulton et al. 2011), but marine ecologists and fisheries scientists need to develop creative methods in the field to test the validity of these models’ hypotheses and increase managers’ confidence in decision making.

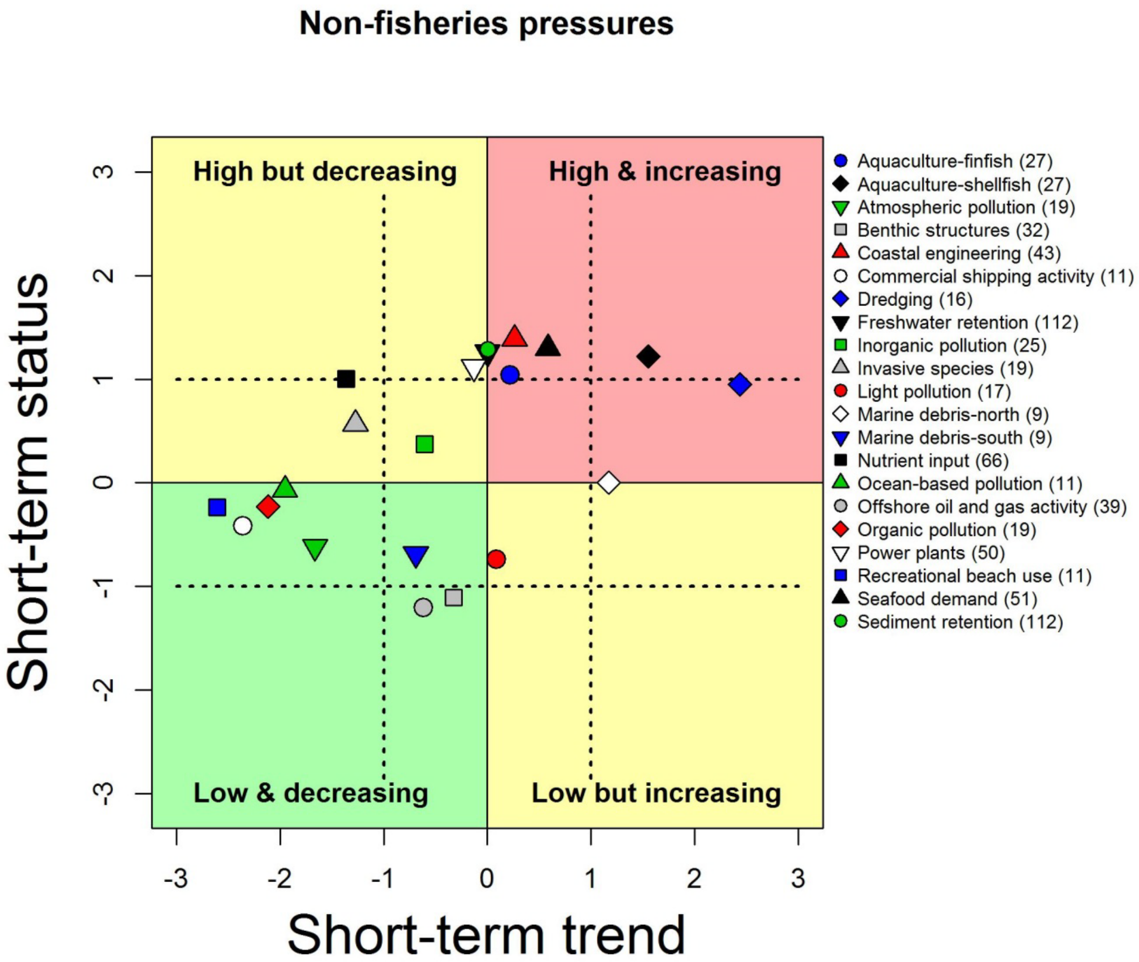


Figure AP32. Short-term status and trend of non-fisheries pressures in the CCLME. Prior to plotting, time series data for each indicator were normalized to place them on the same scale. The short-term trend indicates whether the indicator increased, decreased or remained the same over the last five years.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

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

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

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

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

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

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

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

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

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

Pressure	Indicator	Definition and source of data	Time series	Sampling frequency
Aquaculture: finfish	Finfish production	Washington state estimates (from WDFW) of Atlantic salmon aquaculture production (kg).	1986 – 2012	yearly
Aquaculture: shellfish	U.S. Shellfish production	Total U.S. shellfish production: Fisheries of the United States 2010: http://www.st.nmfs.noaa.gov/st1/publications.html. Using only “clams”, “mussels” & “oysters” estimates.	1985 – 2011	yearly
Atmospheric pollution	Atmospheric deposition of sulfate	Annual precipitation-weighted mean concentrations of sulfate measured at sites in CA, OR, and WA from the National Atmospheric Deposition Program (http://nadp.sws.uiuc.edu/data/ntndata.aspx)	1994 – 2012	yearly
Benthic structures	# offshore oil & gas wells	Total number of offshore oil and gas wells in production: Annual reports of the California State Department of Conservation’s Division of oil, gas, and geothermal resources (ftp://ftp.consrv.ca.gov/../pub/oil/annual_reports/).	1981 - 2012	yearly
Coastal engineering	Human coastal population	Population size of coastline counties in CA, OR, WA; U.S. Census Bureau (http://www.census.gov/prod/2010pubs/p25-1139/p25-1139st1.csv)	1970 – 2012	yearly
Commercial shipping activity	Volume of water disturbed	Calculated using draft, breadth and distance traveled within CCLME while in transit between shipping and receiving ports for domestic (data from USACE Navigation Data Center, New Orleans, LA) and foreign (http://www.ndc.iwr.usace.army.mil/data/dataclen.htm) vessels.	2001 – 2011	yearly
Disease/ pathogens	No appropriate indicator data available.			
Dredging	Dredge volumes	U.S. Army Corps of Engineers navigation data center dredging information system: http://www.ndc.iwr.usace.army.mil/data/datadrgsel.htm; data includes dredge volumes for individual private contracts and Corps operated dredge projects from 1997 through 2011 in WA, CA, and OR.	1997 – 2012	yearly
Freshwater retention	Impoundment volume	Total reservoir storage area in CA and Pacific Northwest water resource regions; data from state agency databases, which include information on construction date and impoundment area/volume for all dams (California: http://cdec.water.ca.gov/misc/resinfo.html, Idaho: http://www.usbr.gov/projects/FacilitiesByState.jsp?StateID=ID, Oregon: http://www.usbr.gov/projects/FacilitiesByState.jsp?StateID=OR, Washington: https://fortress.wa.gov/ecy/publications/summarypages/94016.html).	1900 – 2011	yearly

Pressure	Indicator	Definition and source of data	Time series	Sampling frequency
Inorganic pollution	ISA-toxicity-weighted chemical releases	Total pounds of inorganic pollutants disposed of or otherwise released on site to the ground or water for '1988 core chemicals'; Environmental Protection Agency, Toxics Release Inventory (http://www.epa.gov/tri/). These release values were weighted by toxicity scores (Indiana Relative Chemical Hazard Score) and impervious surface area in the drainage watersheds of the CCLME (http://www.ngdc.noaa.gov/dmsp/download_global_isa.html).	1988 – 2012	yearly
Invasive species	Tons of cargo	Total tons of cargo moved through ports in CA, OR and WA; Data from U.S. Army Corps of Engineers Navigation Data Center (http://www.ndc.iwr.usace.army.mil/data/datawcus.htm)	1993 – 2011	yearly
Light pollution	Average nighttime visible light	Data are cloud-free composites of average visible nighttime lights made using all the available archived DMSP-OLS smooth resolution data for each calendar year. Data grid cell size is 1 km^2 at the equator ; NOAA's National Geophysical Data Center's Version 4 DMSP-OLS Nighttime Lights Time Series Average Lights X Pct (http://www.ngdc.noaa.gov/dmsp/downloadV4composites.html)	1994 – 2010	yearly
Marine debris	Predicted counts of debris	The National Marine Debris Monitoring Program established standardized sampling of coastal trash along the Pacific coast. Ribic et al. (2012) modeled the predicted counts of debris in the northern and southern CCLME. This provides a standardized method that is not biased by number of volunteers or by type of debris collected.	1999 – 2007	yearly
Nutrient input	Nitrogen and phosphorus input from fertilizers	Total nitrogen and phosphorus input from fertilizer use by county has been summarized from 1987 – 2006 by the USGS (Ruddy et al. 2006, Gronberg and Spahr 2012). We use these data along with nationwide data (1945 – 2001) to develop an index for the CCLME across the longer time series. County-level data are available at: http://water.usgs.gov/lookup/getspatial?sir2012-5207_county_fertilizer. Nationwide data are from Ruddy et al. (2006)	1945 – 2010	yearly
Ocean-based pollution	Commercial shipping activity combined with tons of cargo	This indicator combines two previously used indicators. See "Commercial shipping activity" and "Invasive species" for details of data.	2001 – 2011	yearly
Offshore oil activities	Offshore oil & gas production	Number of barrels of oil/ft^3 of gas produced: Annual reports of the California State Department of Conservation's Division of oil, gas, and geothermal resources (ftp://ftp.consrv.ca.gov/../pub/oil/annual_reports/); verified by National Ocean Economics Program at the Monterey Institute of International Studies (http://www.oceaneconomics.org/Minerals/oil_gas.asp).	1974 – 2012	yearly

Pressure	**Indicator**	**Definition and source of data**	**Time series**	**Sampling frequency**
Organic pollution	Toxicity-weighted concentrations	Data are toxicity-weighted concentrations of 16 pesticides measured in water samples from stream-water sites in WA, OR and CA; U.S. Geological Survey Scientific Investigations Report 2010-5139 (http://pubs.usgs.gov/sir/2010/5139/).	1992 – 2010	yearly
Power plants	Saline water withdrawal volumes	Average daily withdrawal volumes (millions of metric tons per day) of saline water from all thermoelectric power plants on the west coast of North America (Pacific Northwest and California regions, from Kenny et al. (2009) and other previous USGS water use reports (http://water.usgs.gov/watuse/50years.html).	1955 – 2005	Every 5 years
Recreational beach use	Beach attendance	Summed beach attendance from CA, OR, and WA based on data from California State Park System Annual Statistical Reports, Oregon Parks and Recreation Dept., and Annual Attendance Reports from the Washington State Parks and Recreation Commission.	2002 – 2012	yearly
Seafood demand	Total consumption	Total consumption or utilization of edible and non-edible fisheries products as reported by annual NOAA Fisheries of the United States reports: (http://www.st.nmfs.noaa.gov/st1/publications.html)	1962 – 2012	yearly
Sediment input	Impoundment area	Same as “Freshwater input”	1900 – 2011	yearly

AQUACULTURE

BACKGROUND

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

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

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

EVALUATION AND SELECTION OF INDICATORS

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

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

STATUS AND TRENDS

The status and trends of aquaculture were divided into an indicator for finfish aquaculture and an indicator for shellfish aquaculture. The status and trends of finfish aquaculture were measured using estimates of Atlantic salmon aquaculture production in the state of Washington (Table AP5) because there are no other commercial marine netpen aquaculture operations along the U.S. West Coast. Using this dataset, finfish aquaculture over the last five years has been constant and at levels greater than the long-term average (Fig. AP33).

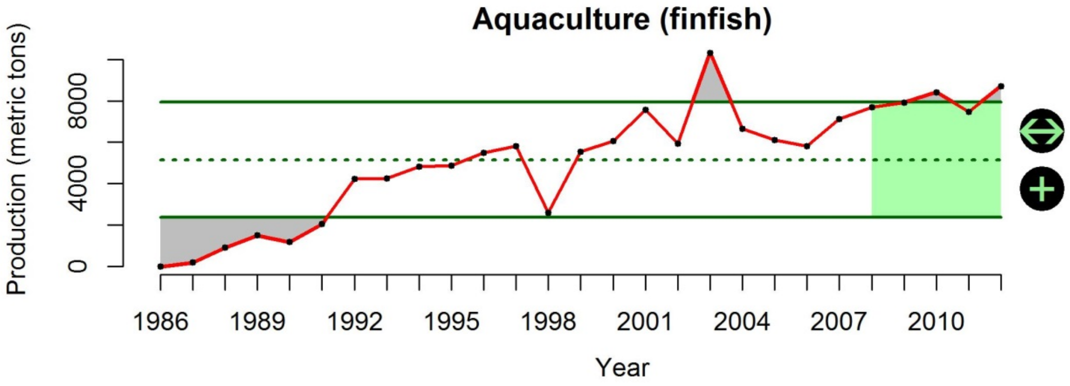


Figure AP33. Production of finfish aquaculture occurring in marine waters of the CCLME.

The status and trends of shellfish aquaculture were measured using estimates of U.S. shellfish production (Table AP5) because estimates of shellfish production in Washington State are not readily available and because Washington produces the most shellfish in the entire U.S. Using this dataset, shellfish aquaculture has increased significantly over the last five years, and the short-term average is greater than the long-term average (Fig. AP34).

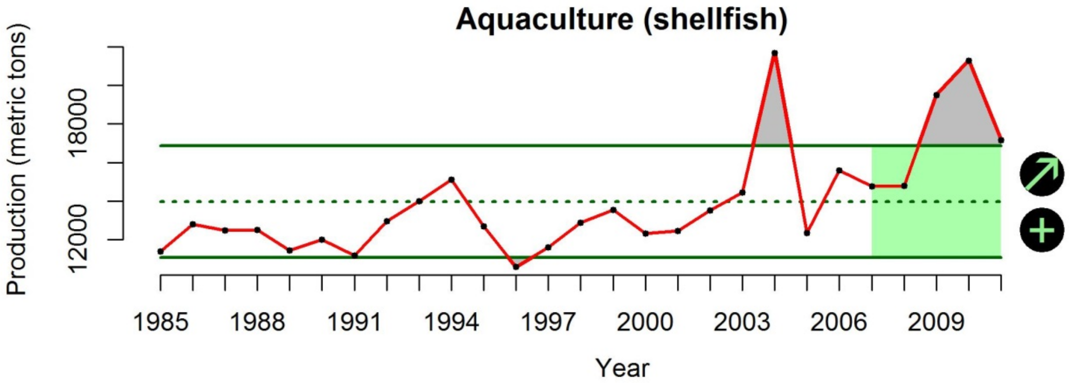


Figure AP34. U.S. production of shellfish (clams, mussels and oysters) aquaculture.

ATMOSPHERIC POLLUTION

BACKGROUND

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

EVALUATION AND SELECTION OF INDICATORS

We evaluated only one indicator for atmospheric deposition: the mean concentration of sulfates monitored by the National Trend Network (NTN) of the National Atmospheric Deposition Program (Table AP4). The NTN provides a long-term record of precipitation chemistry for sites located throughout the U.S. Data have been consistently collected weekly using the same protocols since 1994. Specific ions that are measured include calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}), potassium (K^{+}), sulfate (SO_4^{2-}), nitrate (NO^{3-}), chloride (Cl^{-}), and ammonium (NH_4^{+}) ions. These data are easily accessible via the NADP website (http://nadp.sws.uiuc.edu/data/ntndata.aspx). This indicator of atmospheric deposition rated very high under all criteria categories (Table AP4).

STATUS AND TRENDS

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

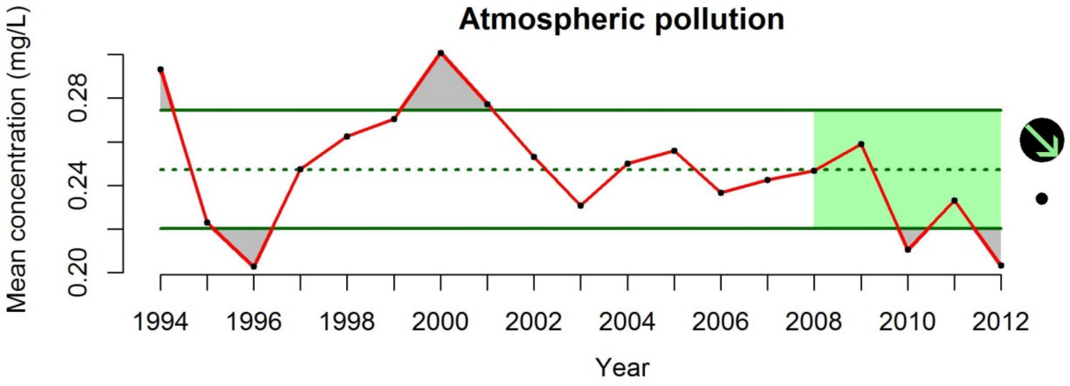


Figure AP35. Precipitation-weighted mean concentration (mg/L) of sulfates deposited out of the atmosphere at sites located within watersheds of the CCLME.

BENTHIC STRUCTURES

BACKGROUND

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

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

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

EVALUATION AND SELECTION OF INDICATORS

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

STATUS AND TRENDS

The status and trends of benthic structures were measured using the number of oil and gas wells in offshore waters of the CCLME (Table AP5). These data are available in annual reports from the California Department of Conservation's Oil, Gas and Geothermal Resources Division from 1981 – 2012 (ftp://ftp.consrv.ca.gov/pub/oil/annual_reports/). We summed the number of state and federal offshore wells "producing" and "shut-in" (i.e. temporarily sealed up). The number of benthic structures in the CCLME has been constant over the short term, but has been greater than 1SD below the long-term average of the entire time series for the last decade (Fig. AP36).

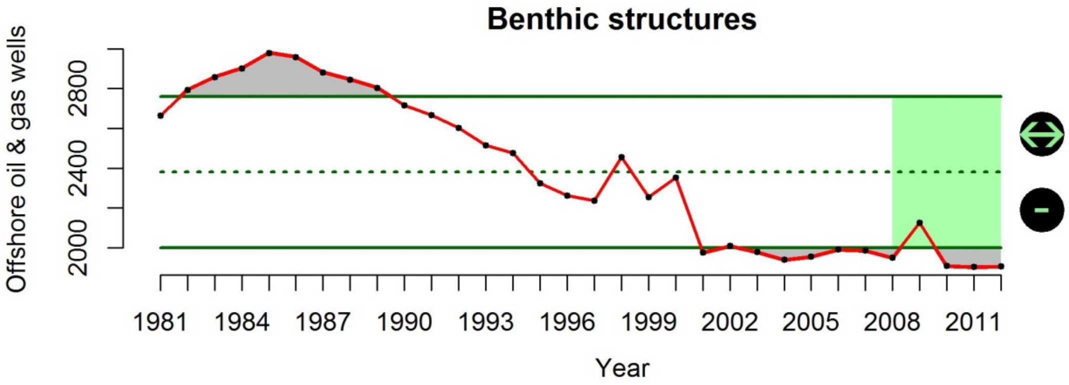


Figure AP36. The number of offshore oil and gas wells in production or shut-in in the CCLME.

COASTAL ENGINEERING

BACKGROUND

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

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

Many shoreline “hardening” structures, such as seawalls and jetties, tend to reduce the complexity of habitats and the amount of intertidal habitats (Williams and Thom 2001, Bulleri and Chapman 2010). Because shorelines are highly diverse in their geologic nature and wave climate, acceptable ranges of armoring likely differ significantly from one location to another (Shipman et al. 2010). The definition of acceptable also will vary depending on the ecosystem response variable of interest. Differences in fish behavior and usage between modified and unmodified shorelines are caused by physical and biological effects of the modifications, such as changes in water depth, slope, substrate, and shoreline vegetation (Toft et al. 2007, Morley et al. 2012). Urban infrastructure supports different epibiota and associated assemblages and does not function as a surrogate of natural rocky habitats (Bulleri and Chapman 2010). Its introduction in the intertidal zone or in nearshore waters can cause fragmentation and loss of natural habitats. Furthermore, the novel hard substrata along sedimentary shores can alter local and regional biodiversity by modifying natural patterns of dispersal of species, or by facilitating the establishment and spread of exotic species.

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

EVALUATION AND SELECTION OF INDICATORS

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

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

The rate of shoreline armoring has been shown to correspond with the rate of population growth in coastal areas (Douglass and Pickel 1999), and in the absence of good time series of geospatial data for hardened shorelines, coastal population data for the coastline counties of the West Coast of the United States provides a good proxy for this stressor. Population density has a long history of reporting and is known to affect coastal regions disproportionately (Crossett et al. 2005). Population density is becoming increasingly understood in some regions as an agent of shoreline change (e.g. Puget Sound Partnership; http://www.psp.wa.gov/vitalsigns/shoreline_armoring.php). Coastline counties of the United States, located along the country's saltwater edges, account for just 254 of the nation's 3,142 counties yet contain 29 percent of its population, 5 of its 10 most populous cities, and 7 of its 10 most populous counties (Wilson and Fischetti 2010). To qualify as coastline, a county has to be adjacent to water classified as either coastal water or territorial sea. Transformation of coastal landscapes in response to urbanization also affects the intertidal zone and nearshore estuarine and marine waters, which are also increasingly altered by the loss and fragmentation of natural habitats and by the proliferation of a variety of built structures, such as breakwaters, seawalls, jetties and pilings. Unclear however, at this time, is the explicit relationship between coastal population levels and the relative amount of shoreline affected by coastal engineering structures; this data gap is likely driven by the lack of good time series data on the latter.

STATUS AND TRENDS

The status and trends of coastal engineering were measured using estimates of human population in counties classified as "coastline" in WA, OR and CA (Table AP5). Data for coastline population estimates were retrieved from county estimates from the U.S. Census Bureau (2010 – 2012; http://www.census.gov/popest/data/datasets.html) and the National Bureau of Economic Research (1970 – 2009; http://www.nber.org/data/census-intercensal-county-population.html). Using this indicator, coastal engineering has been increasing steadily over the entire time series. Over the last five years of this dataset, however, there was no change, but the current status is >1SD above the long-term average (Fig. AP37). Populations along the coast continue to increase and the ultimate driver of many non-fisheries related pressures will continue to increase for the foreseeable future.

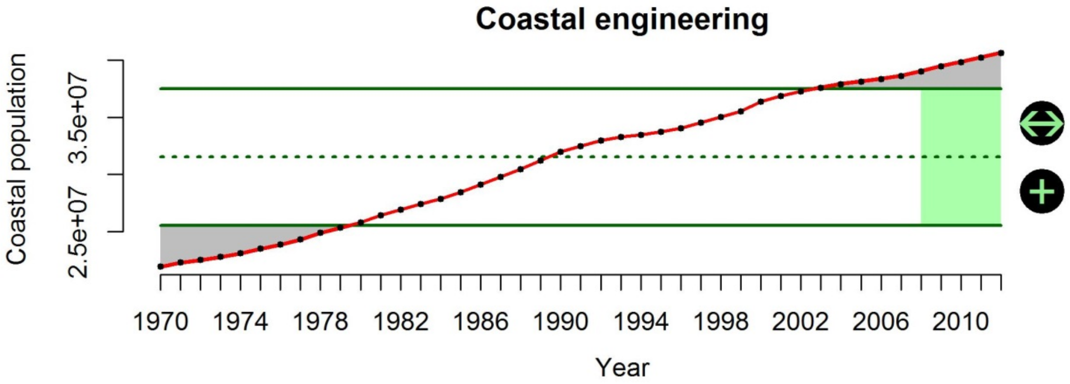


Figure AP37. U.S. population in coastline counties of WA, OR and CA.

COMMERCIAL SHIPPING ACTIVITY

BACKGROUND

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

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

The effects of commercial shipping activity on fish populations are not very well understood, but some data suggest responses will be behavioral in nature (e.g. Rostad et al. 2006) and related to loss of habitat (Uhrin and Holmquist 2003, Eriksson et al. 2004) or noise pollution (Slabbekoorn et al. 2010). Some fish species may be attracted to vessels, rather than repelled by them, and are not bothered by noisy, passing ships (Rostad et al. 2006). However, frequently traveled routes such as those traveled by ferries and other transportation vessels may impact fish spawning, migration, communicative, and recruitment behaviors through noise and direct disturbance of the water column (Barr 1993, Codarin et al. 2009).

EVALUATION AND SELECTION OF INDICATORS

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

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

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

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

We categorized trips into two categories. If the shipping and receiving port was the same (i.e., the vessel was moving from one dock to another or moving a barge within the port), this was categorized as "port" traffic, while all other trips were categorized as "coastal" traffic. For this analysis, we removed all "port" traffic because this pressure is defined as a measure of the risk of vessels striking marine mammals, causing noise pollution, and modifying coastal habitat. We include "port" traffic in the indicator for ocean-based pollution below. In order to calculate the distance traveled within the CCLME for each vessel, we used distances between ports as measured by NOAA's Office of Coast Survey and documented in USDOC (2012). For trips that traveled outside of the CCLME, we used the distance from the port within the CCLME to the boundary of the CCLME following the major shipping lane pathways. For example, if a vessel traveled from San Diego, CA to Houston, TX, we calculated the distance from San Diego to the southern boundary of the CCLME on the vessel's way toward the Panama Canal (estimated at 602 nm (1115 km)).

These distances were then multiplied by the vessel's shipping draft (m) and breadth (m) to give a volume (m^3) of water directly disturbed by the vessel during transit through the CCLME. Obviously the wake of a vessel will disturb more than our calculated volume, so this is a conservative estimate of absolute volume, but the trends over time will be relative.

STATUS AND TRENDS

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

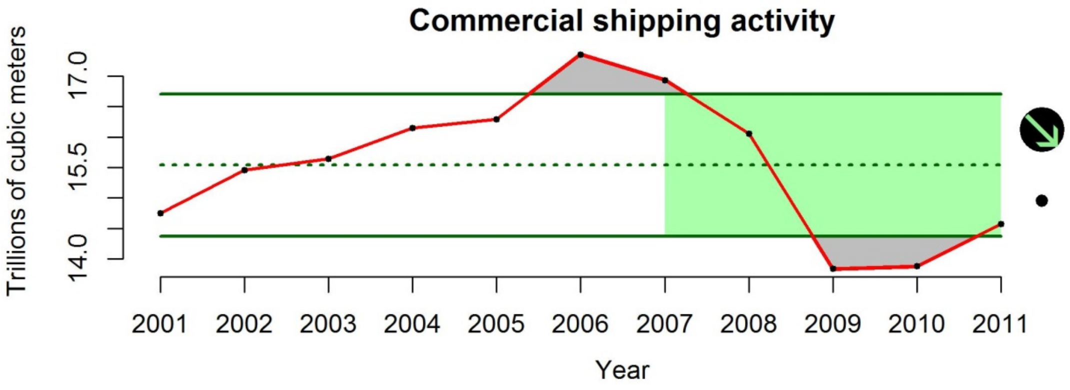


Figure AP38. Volume (trillions m^3) of water disturbed during transit of commercial shipping vessels along the coast of the CCLME.

DISEASE/PATHOGENS

BACKGROUND

The last few decades have seen a worldwide increase in the reports of disease in the marine environment (Harvell et al. 1999), though these increases appear to be taxa related (Ward and Lafferty 2004). Diseases are thought to be fostered by increases in climate variability and human activity as many outbreaks are favored by changing environmental conditions that increase pathogen transmission or undermine host resistance (Anderson 1998). Marine flora and fauna serve as hosts for numerous parasites and pathogens that may affect the host populations as well as have cascading effects throughout the ecosystem.

For example, the near elimination of seagrass (*Zostera marina*) beds from many North Atlantic U.S. coastlines in the 1930's due to wasting disease (thought be caused by a pathogenic strain of *Labyrinthula*, which has since been confirmed and identified in eelgrass beds in the 1980's on both coasts of the United States (Short et al. 1987)) was responsible for numerous alterations to coastal habitats (Rasmussen 1977) and fauna, including a reduction or loss of migratory waterfowl populations (Addy and Aylward 1944) and the loss of the scallop fishery in the mid-Atlantic coast of the U.S. (Thayer et al. 1984).

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

EVALUATION AND SELECTION OF INDICATORS

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

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

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

DREDGING

BACKGROUND

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

EVALUATION AND SELECTION OF INDICATORS

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

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

These data are accessible through the U.S. Army Corps of Engineers navigation data center dredging information system: http://www.navigationdatacenter.us/data/datadrgsel.htm. There are two sources of data: 1) Dredging contracts and 2) Corps-owned dredges. Data include dredge volumes, locations, and costs for individual private contracts and Corps operated dredge projects from 1997 through 2012 nationwide. We summarized annual dredge volumes (converted to cubic meters) for all private contracts conducted in California, Oregon, and Washington. We summarized annual dredge volumes (converted to cubic meters) for all dredging activities performed by the "Portland" Division which represents the only dredging performed by the Corps along the U.S. West Coast. Annual offshore dump volumes are not summarized and reported separately, but can be determined with some data manipulation from this database. In some locations, dredge dump volumes are also reported to give an indication of the extent of, and trends in dredging activities (e.g., Annual OSPAR Reports on the Dumping of Wastes at Sea).

STATUS AND TRENDS

The status and trends of dredging in the CCLME were measured using dredged volume (millions of m^3) of sediments from projects originating in WA, OR and CA waters (Table AP5). Using this indicator, dredging has increased over the last five years, but the short-term average is still within 1SD of the long-term average of the entire time series (Fig. AP39). If dredging activities within the CCLME remain at current levels or increase, the short-term status of this indicator will be greater than the long-term average by 2013.

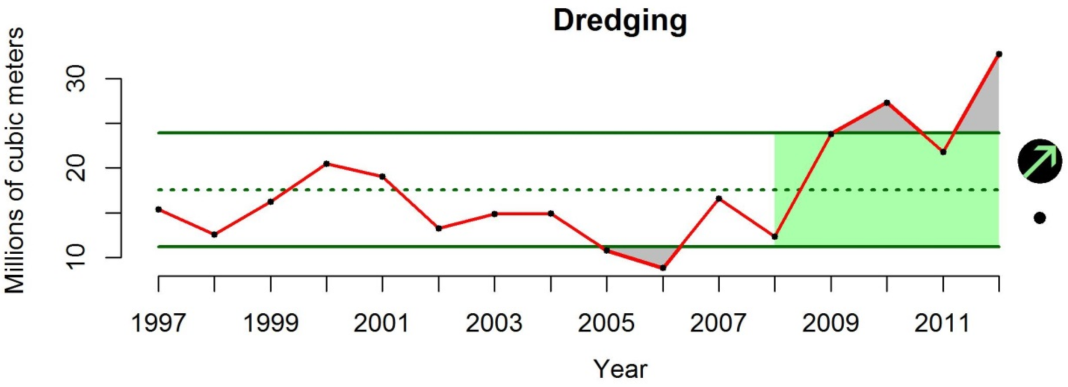


Figure AP39. Volume (millions m^3) of dredged sediments from projects originating in WA, OR and CA.

FRESHWATER RETENTION

BACKGROUND

As the world's population grows and its demands for freshwater increase, interannual variability and long-term changes in continental runoff are of great concern to water managers (Dai et al. 2009). Freshwater flow also affects fisheries and ESA-listed species. River discharge into many estuaries and coastal marine areas has been substantially altered by diversion for human use (Vorosmarty et al. 2000). Water withdrawals for public-supply and domestic uses have increased steadily since estimates began, with freshwater withdrawals of almost 1.32 billion m^3/d in 2005. Thermoelectric-power generation (see Power Plants, below) and irrigation withdrawals have generally been the two largest human use categories since these estimates were made. Hydropower is considered an "in-stream use" of freshwater, but associated dams and dam operations also alter flow patterns, volume, and depth of water within and below impoundments. Dam projects operating as "store and release" facilities drastically affect the magnitude, timing, and duration of downstream water flow and depth, resulting in dramatic deviations to natural fluctuations in habitat accessibility, acute temperature changes, and overall water quality.

Modified freshwater flow regimes change the salinity gradient and pattern in salinity variation within estuaries and coastal systems, and can induce large shifts in community composition and ecosystem function (Gillanders and Kingsford 2002). These ecosystems often respond most strongly on an interannual timescale to variability in freshwater flow. Several mechanisms for positive or negative flow effects on biological populations in estuaries have been proposed (Kimmerer 2002). Positive effects appear to operate mainly through stimulation of primary production, with effects propagating up the food web. Overall impacts on the biota are generally considered negative, however, with documented changes to migration patterns, spawning habitat, species diversity, water quality, and distribution and production of lower trophic levels (Drinkwater and Frank 1994). For freshwater systems, a framework has been developed for assessing environmental flow needs for many streams and rivers to foster implementation of environmental flow standards at the regional scale (Poff et al. 2010). Studies focused on reductions in freshwater flow have generally shown detrimental ecosystem effects and altered community composition (Gillanders and Kingsford 2002). However, freshwater subsidies to estuaries or hypersaline lagoons have also been shown to cause major shifts in vegetation, fish, and macroinvertebrate assemblages (Nordby and Zedler 1991, Strydom et al. 2002, Rutger and Wing 2006).

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

EVALUATION AND SELECTION OF INDICATORS

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

River runoff (R) can also be expressed as the difference between precipitation (P) and the sum of evapo-transpiration (ET), storage (S) (e.g., groundwater), and consumption (C) (e.g., irrigation) (Milliman et al. 2008). Therefore, data series associated with the anthropogenically-derived parameters, C and S, likely provide some of the best indicators of human impacts to freshwater input. Freshwater storage (S) data are accessible and can be obtained on an annual basis from state agency databases, which include information on construction date and impoundment area/volume for all dams (California: http://cdec.water.ca.gov/misc/resinfo.html; Idaho: http://www.usbr.gov/projects/FacilitiesByState.jsp?StateID=ID; Oregon: http://www.usbr.gov/projects/FacilitiesByState.jsp?StateID=OR; Washington: https://fortress.wa.gov/ecy/publications/summarypages/94016.html). Furthermore, large-scale hydrological alterations are known to cause a variety of downstream habitat changes, such as deterioration and loss of river deltas and ocean estuaries (Rosenberg et al. 2000).

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

STATUS AND TRENDS

The status and trends of freshwater retention in the CCLME were measured using the total impoundment volume (millions m^3) of freshwater stored behind dams in CA, OR, ID and WA (Table AP5). Using this dataset, the storage of freshwater has been relatively constant for the last 40 years, but the short-term average was greater than 1SD above the long-term average of the entire time series (Fig. AP40). This time series reflects the large increases in reservoir impoundment during the period of major dam building from the 1940's to the early 1970's with relatively little change since then. This indicator highlights the legacy of historical pressures, but the relative stability of this indicator in the short-term may not provide a useful indicator of change in freshwater retention moving forward. Further development of indicators for this pressure is likely necessary.

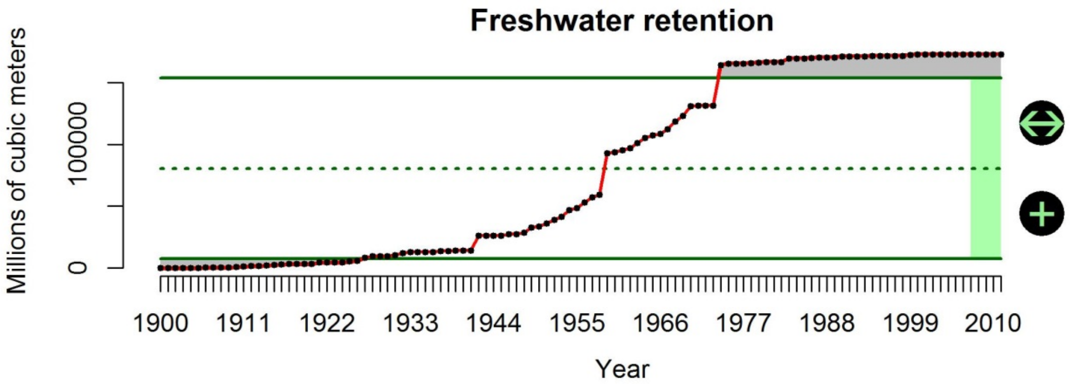


Figure AP40. Volume (millions m^3) of freshwater stored behind dams in WA, OR and CA.

INORGANIC POLLUTION

BACKGROUND

Tens of thousands of chemicals are used by industries and businesses in the United States for the production of goods on which our society depends. Many of the chemicals used in the manufacturing and production of these goods are toxic at some level to humans and other organisms and some are inevitably released into the environment. The production, use and release of various toxic chemicals have changed over time depending on economic indices, management methods (recycling and treatment of chemicals), and environmental regulations (USEPA 2010). The pathway of these chemicals to estuarine and marine environments can be direct (e.g., wastewater discharge into coastal waters or rivers) or diffuse (e.g., atmospheric deposition or urban runoff). Over the past 40 years, direct discharges have been greatly reduced; however, the input of pollutants to the marine environment from more diffuse pathways such as runoff from land-based activities is still a major concern (Boesch et al. 2001).

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

Coastal and estuarine pollution can affect all life stages of fish, but fish can be particularly sensitive to toxic contaminants during the first year of life (Rosenthal and Alderdice 1976). Over time, organisms will accumulate contaminants from water, sediments or food in their tissues, which then transfers to offspring through reproduction and throughout the food web via trophic interactions. Negative impacts of pollution on commercial fish stocks have generally not been demonstrated, largely due to the fact that only drastic changes in marine ecosystems are detectable and the difficulty in distinguishing pollution-induced changes from those due to other causes (Sindermann 1994). Normally, chronic and sublethal changes take place very slowly and it is impossible to separate natural fluctuations from anthropogenic causes. Furthermore, fish populations themselves are estimated only imprecisely, so the ability to detect and partition contaminant effects is made even more difficult. However, measurements of marine biodiversity have shown that species richness and evenness are reduced in areas of anthropogenic pollution (Johnston and Roberts 2009).

EVALUATION AND SELECTION OF INDICATORS

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

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

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

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

In order to calculate this indicator, we downloaded data from 1988 – 2012 from the TRI EZ search database (http://www.epa.gov/enviro/facts/tri/ez.html) using the “Flat (Denormalized) Form R”. We selected the following data columns for download: “TRI Facility Id”, “Reporting Year”, “Chemical Name”, “TRI Chemical Id”, “County Name”, “State Abbreviation”, “Facility Latitude”, “Facility Longitude”, “Land Total Release” and “Water Total Release” and selected for states that occur in watersheds that drain into the CCLME (Fig. AP41). Only facilities located within CCLME watersheds were used to sum all releases to land and water. Data (lbs of releases) for each chemical were converted to kg and summed across each release category. In order to weight each chemical by its relative toxicity, we multiplied the amount of releases for each chemical by its score in the Indiana Relative Chemical Hazard Ranking Score (IRCHS; http://cobweb.ecn.purdue.edu/CMTI/IRCHS/) divided by 100:

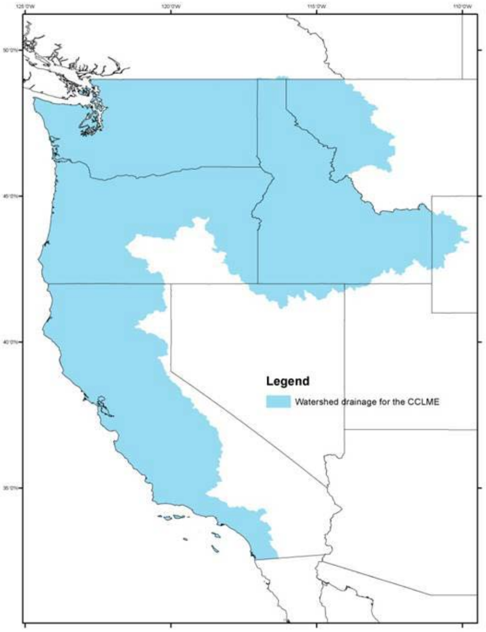


Figure AP41. Polygon of the watershed that drains into the CCLME and used to clip impervious surface area data layers (based on Halpern et al. 2009).

Toxicity-weighted releases = chemical releases * (IRCHR/100)

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

In order to provide weightings of ISA for each year, we used the ISA GIS data layers developed by the U.S. National Geophysical Data Center for the years 2000-2001 (global estimates) and January – June 2010 (estimates for the United States only). These data are available at http://ngdc.noaa.gov/eog/dmsp/download_global_isa.html. We used the watershed drainage boundary for the CCLME developed by Halpern et al. (2009) to delineate the watersheds in which ISA values would be summed across (Fig. AP41). The 2000 – 2001 and 2010 ISA data layers were clipped to the watershed boundary polygon and then ISA values were summed across all cells. Because there were only two years of ISA data, we assumed a linear relationship between 2001 and 2010 and simply extrapolated summed ISA values to the remaining years between 1988 and 2012 based on this linear assumption. Summed ISA values were then standardized as a proportion of the maximum value (i.e., summed ISA value each year/maximum summed ISA value) such that the year with the highest summed ISA value had a weighting of 1 and all others were a proportion. Toxicity-weighted releases were then multiplied by the corresponding ISA weighting for each year. Finally, the ISA-Toxicity-weighted releases were normalized.

In 1998, the EPA began collecting pollution information from the commercial hazardous waste treatment sector. Because of this change during our time series, there was a very large change in the magnitude of reported chemicals in the TRI database. To account for this magnitude shift, the ISA-Toxicity-weighted releases were normalized independently across the two time periods. Data from 1987 – 1997 were normalized and data from 1998 – 2012 were normalized and then appended to each other to create a continuous time series from 1988 – 2012. We investigated the influence of different chemicals being added to or removed from the list reported by TRI by calculating the exact same time series as described above using only chemicals from the 1988 Core Chemical list. This resulted in differences at the beginning of the time series (1988 – 1993), but had virtually no effects on the status and trends of the rest of the time series; thus, we decided to include all chemicals reported by the TRI database into the calculation of this indicator.

STATUS AND TRENDS

The status and trends of inorganic pollution in the CCLME were measured using ISA-toxicity-weighted chemical releases from data collected by the Environmental Protection Agency and reported by the Toxics Release Inventory (TRI) Program (Table AP5). This indicator incorporates the amount and toxicity of chemicals released into water and onto land by industrial facilities as well as the amount of impervious surface area in the CCLME's drainage basin. Using this indicator, inorganic pollution has not changed over the last five years, and is within 1SD of the long-term average of the entire time series (Fig. AP42).

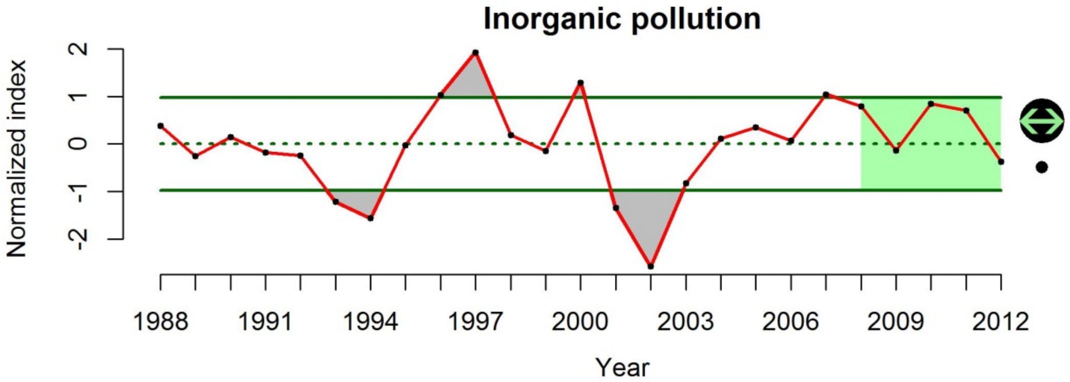


Figure AP42. Normalized index of ISA-toxicity-weighted chemical releases on land or into water by industrial facilities within watersheds of the CCLME.

INVASIVE SPECIES

BACKGROUND

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

EVALUATION AND SELECTION OF INDICATORS

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

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

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

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

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

The U.S. Department of Transportation projects that, compared to 2001, total freight moved through U.S. ports will increase by more than 50 percent by 2020 and the volume of international container traffic will more than double (American Association of Port Authorities Fact Sheet 2011: http://www.aapa-ports.org/files/PDFs/facts.pdf). In order to estimate the potential for species invasions, we used data on the total amount of shipping cargo (thousands of short tons converted to millions of metric tons) that moved through each port along the Pacific Coast of the United States. These data were available from the U.S. Army Corps of Engineers Navigation Data Center (http://www.navigationdatacenter.us/data/datawcus.htm). CSV files were available for years 1993 – 2011. Ports in the states of California, Oregon and Washington were used to calculate the sum of cargo being shipped and received in ports within the CCLME.

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

STATUS AND TRENDS

The status and trends of invasive species in the CCLME were measured using the amount of cargo moving through coastal ports of the CCLME (Table AP5). Using this indicator, the number of potentially invasive species entering ports along the CCLME has decreased over the last five years, but the short-term average is still within 1SD of the long-term average of the entire time series (Fig. AP43). This decreasing trend will quickly revert to an increasing trend if port volumes continue to increase as they have over the last two years of the dataset. In addition to using this indicator, it would be good to develop an index that combines port volume and aquaculture as a more thorough indicator of the status and trends of invasive species.

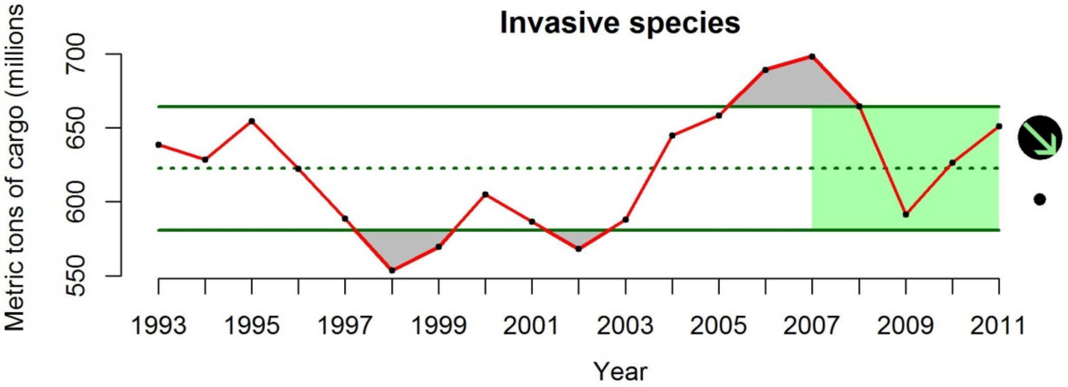


Figure AP43. Metric tons (millions) of cargo moved through ports in WA, OR and CA.

LIGHT POLLUTION

BACKGROUND

Ecological light pollution has demonstrable effects on the behavioral and population ecology of organisms in natural settings (Rich and Longcore 2006). As a whole, these effects derive from changes in orientation, disorientation, or misorientation, and attraction or repulsion from the altered light environment, which in turn may affect foraging, reproduction, migration, and communication (Longcore and Rich 2004). Many nocturnally migrating birds die or lose a large amount of their energy reserves during migration as a result of encountering artificial light sources (Poot et al. 2008). Marine zooplankton and numerous fish species are known to vertically migrate in the water column (Cushing 1951, Enright and Hamner 1967). This diel pattern of behavior allows zooplankton to avoid many visually-based predators while foraging in productive waters at night (Zaret and Suffern 1976). Diel vertical migration to avoid predation is also widespread among pelagic marine fishes (Neilson and Perry 1990, Watanabe et al. 1999). Even intertidal organisms display patterns of movement that are related to abiotic conditions, including patterns of light (Warman et al. 1993). In their early pelagic larval stages, more than 80% of fish and invertebrate species respond positively to light and migrate to the surface layers (Thorson

1964), thus changes in ambient light may have significant influence on the settlement patterns of these species.

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

EVALUATION AND SELECTION OF INDICATORS

We evaluated only one indicator of light pollution in the CCLME: a normalized index of nighttime light pixels present in waters of the CCLME (Table AP4). This indicator is based on data collected by the U.S. Air Force Weather Agency and processed by NOAA's National Geophysical Data Center (NGDC). This dataset is available from 1992 – 2010 on the NGDC's website: http://www.ngdc.noaa.gov/dmsp/downloadV4composites.html. Specifically, we used the "Average Lights x Pct" (average nighttime lights, hereafter) data layers for satellites F12-18 and years 1994 – 2010 (we deleted data from satellite F10 based on recommendations from Elvidge et al. (2009)). These data layers were derived from the average visible band digital number (DN) of cloud-free light detections multiplied by the percent frequency of light detection. The inclusion of the percent frequency of detection term normalized the resulting digital values for variations in the persistence of lighting. For instance, the value for a light only detected half the time is discounted by 50%. Note that this product contains detections from fires and a variable amount of background noise.

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

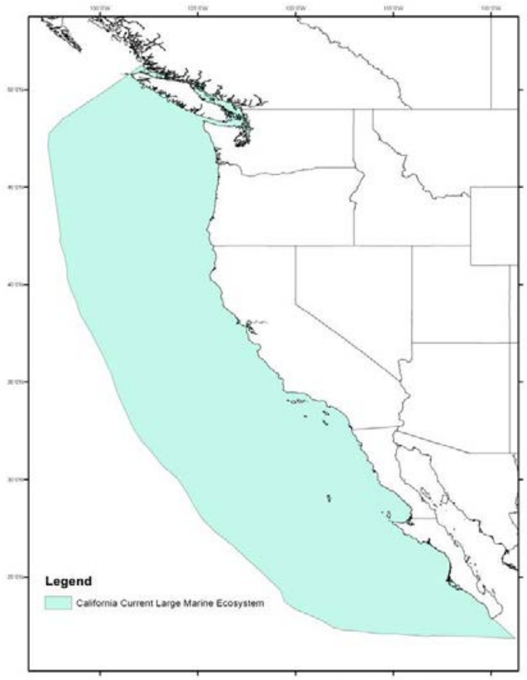


Figure AP44. Polygon of the CCLME used to clip all nighttime lights data layers

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

STATUS AND TRENDS

The status and trends of light pollution in the CCLME were measured using a normalized index of the sum of average nighttime lights (Table AP5). These data were processed and made available by the U.S. Geophysical Data Center. According to this indicator, light pollution has been constant over the last five years and is within 1SD of the long-term average of the time series (Fig. AP45). This result is a little unexpected due to the contrasting increases observed in coastline populations. The overall time series showed that light pollution steadily decreased from 1995 – 2004 within the CCLME and has been at these relatively low levels ever since.

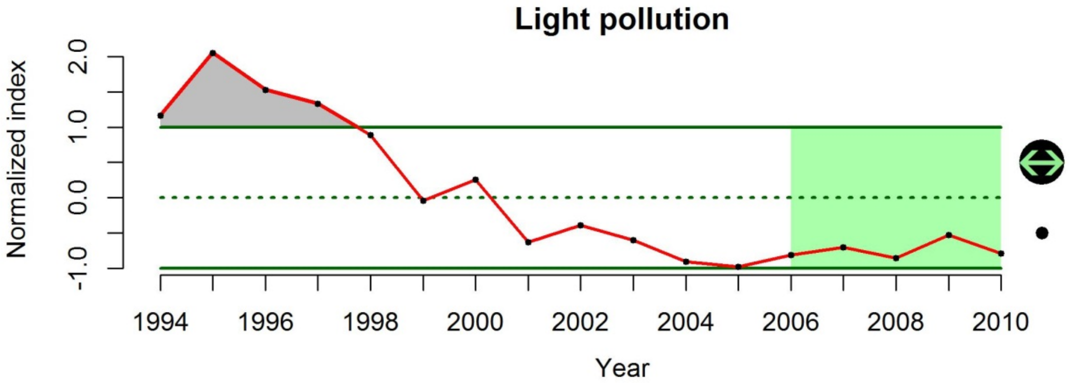


Figure AP45. Normalized index of the sum of average nighttime lights in waters of the CCLME.

MARINE DEBRIS

BACKGROUND

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

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

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

EVALUATION AND SELECTION OF INDICATORS

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

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

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

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

STATUS AND TRENDS

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

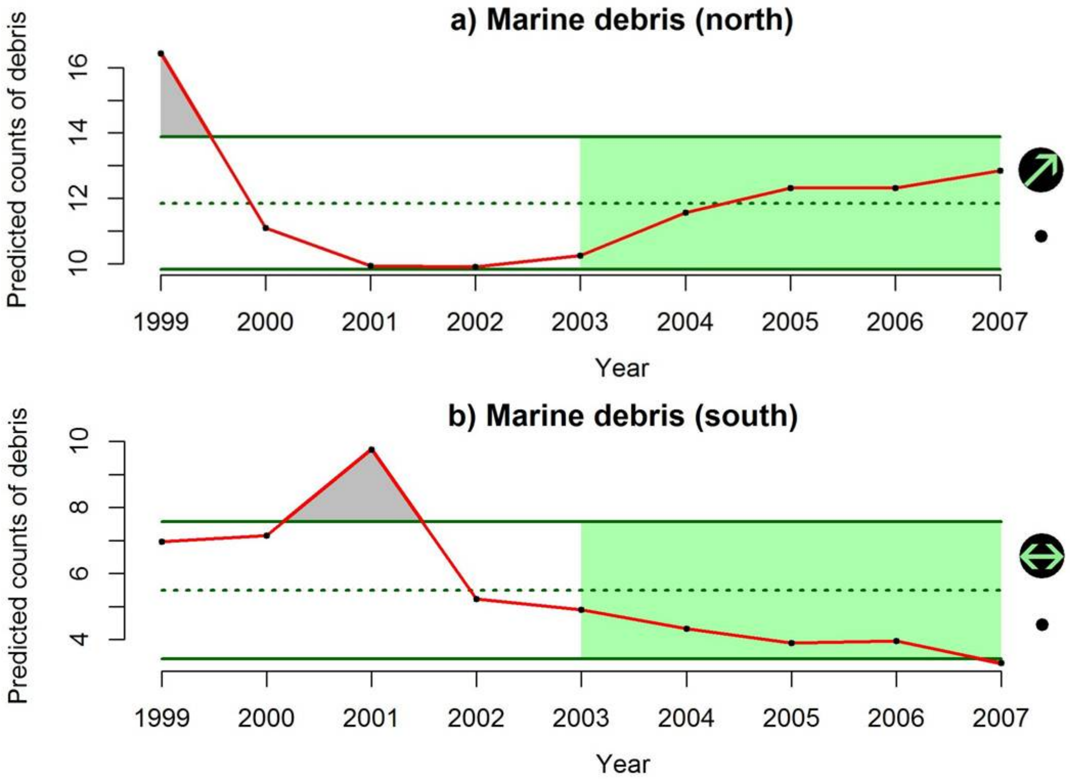


Figure AP46. Predicted counts of debris along the a) northern and b) southern coasts of the CCLME (Point Conception separates the regions). Data provided by Christine Ribic (Ribic et al. 2012).

NUTRIENT INPUT

BACKGROUND

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

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

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

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

EVALUATION AND SELECTION OF INDICATORS

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

Halpern et al. (2009) used time series data from Nolan and Hitt (2006) on county-level fertilizer application data from 1992-2001 (kgs/hectare) and confined manure (primarily from dairy farms) from 1992-1997. These files (http://water.usgs.gov/GIS/dsdl/gwava-s/index.html) (Nolan and Hitt 2006) have a relatively limited temporal range (between 1992 – 2001). A comparable alternative would be to compile county-level estimates of nutrient inputs (kg/km^2) to the land surface of the conterminous United States, presented from 1982-2006 based on fertilizer use, livestock manure, and atmospheric deposition (Ruddy et al. 2006, Gronberg and Spahr 2012)). An older time series (1945-1986) of nationwide fertilizer application data (Ruddy et al. 2006, Dubrovsky et al. 2010) could expand the time series further by assuming that watersheds bordering the Pacific Coast follow the same historic trends in fertilizer applications. More recent data (2007 – 2010) are expected in a forthcoming analysis and summary (N. Dubrovsky, USGS, *pers comm*). Models have been used to predict the probability of nitrate contamination in ground waters of the United States based on fertilizer loading and other factors (Nolan and Hitt 2006). It is unclear how this relates to coastal systems, however.

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

Nutrient trends in West Coast rivers (1993-2003) have been summarized using similar methods in a recent report by Sprague et al (2009), which showed that flow adjusted trends in total phosphorus concentrations were generally upward or non-significant at sites in the Southwestern U.S. and non-significant in the Northwestern U.S.

Trends in total nitrogen concentrations generally were downward or non-significant at sites in the Northwestern U.S., but mixed in all other regions. Regional reports include an analysis of trends (1993 – 2003) in the Columbia River and Puget Sound basins (Wise et al. 2007) and the Sacramento, San Joaquin, and Santa Ana Basins, California (Kratzer et al. 2011). In the Pacific Northwest study, point-source nutrient loads generally were a small percentage of the total catchment nutrient loads compared to nonpoint sources, with most of the monitoring sites showing decreasing trends in TN and TP, indicating that inputs from nonpoint sources of nutrients probably have decreased over time in many of the catchments (Wise et al. 2007). In the California study, most trends in flow-adjusted concentrations of nutrients in the Sacramento Basin and Santa Anna River were downward, whereas nitrogen trends in the San Joaquin Basin were upward, especially over the 1975–2004 time period (Kratzer et al. 2011). As all of these studies note, fertilizer use, livestock manure, atmospheric deposition, population growth, and source loading (e.g., wastewater treatment plants) are all known nutrient sources that can contribute to increasing nutrient stream loads. However, basin slope, basin area, mean annual precipitation, and soil type may also interact with these sources, and flow-adjusted trends in concentration can also be complex, as there often are multiple and possibly counteracting anthropogenic factors influencing nutrient source and transport in a watershed. Without detailed knowledge of all important factors in each watershed, it may be difficult to discern the specific cause(s) of a trend.

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

There were also statewide preliminary data available from the USGS (*pers comm* J. Gronberg) for 2007 – 2010. Because these data were at the state level, we calculated the proportion of statewide data that was likely contributed by counties within watersheds of the CCLME. In order to do this, we used statewide data from the 1987 – 2006 dataset for each state containing watersheds of the CCLME (CA, OR, WA, ID, MT, WY) and calculated the proportion of farm and non-farm nitrogen and phosphorus that was contributed by counties in watersheds of the CCLME for each year. We then multiplied the average of these proportions and the statewide data from 2007 – 2010 to calculate estimates of nitrogen and phosphorus in the CCLME. These data were appended to the data from 1945 – 2006 to create a full time series from 1945 – 2010. We then normalized the time series data for nitrogen and phosphorus separately, summed the normalized values for each year, and then re-normalized these sums across all years to get a single normalized index of the sum of nitrogen and phosphorus input from fertilizers across counties that drain into the California Current.

STATUS AND TRENDS

The status and trends of nutrient input into the CCLME were measured using a normalized index of the sum of nitrogen and phosphorus applied to lands as fertilizers in counties that drain into the California Current (Table AP5). Using this dataset, nutrient input has decreased over the last five years of the dataset (2006 – 2010) but the short-term average was > 1SD of the long-term average of the time series (Fig. AP47). Overall, the application of nitrogen and phosphorus increased steeply since the beginning of this time series until the early 1980's. Input of these nutrients seemed to plateau through the 1980's and 1990's before increasing again in the 2000's. The most recent decline was due to a large decrease in the amount of phosphorus from farms in California in 2009.

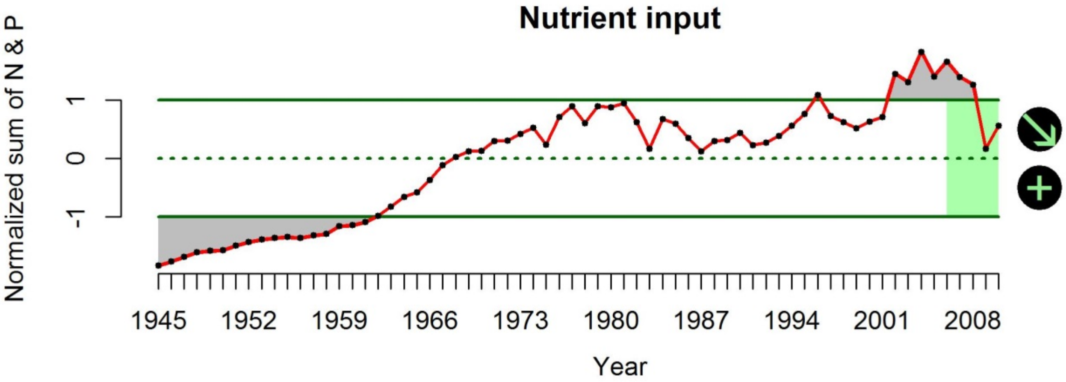


Figure AP47. Normalized index of the sum of nitrogen and phosphorus applied as fertilizers in WA, OR and CA.

OCEAN-BASED POLLUTION

BACKGROUND

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

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

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

EVALUATION AND SELECTION OF INDICATORS

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

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

STATUS AND TRENDS

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

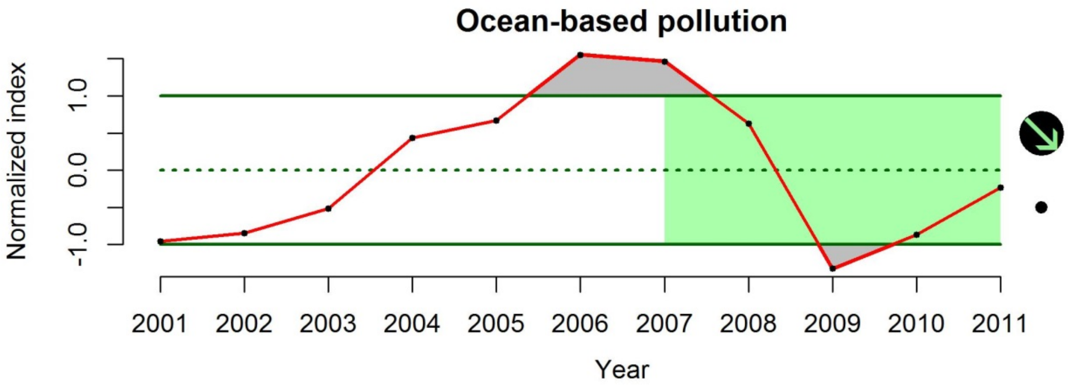


Figure AP48. Normalized index that combines the volume (millions m^3) of water disturbed by vessels during transit in port and along the coast and the volume of cargo (millions of metric tons) moving through ports in the states of CA, OR and WA.

OCEAN MINING

BACKGROUND

This pressure has not been evaluated to date.

EVALUATION AND SELECTION OF INDICATORS

This pressure has not been evaluated to date.

STATUS AND TRENDS

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

OFFSHORE OIL AND GAS ACTIVITY

BACKGROUND

The environmental risks posed by offshore exploration and production of oil and gas are well known. They include the release of hydrocarbons to the environment, smothering of benthos, sediment anoxia, destruction of benthic habitat, and the use of explosives (Macdonald et al. 2002). Petroleum exploration involves seismic testing, drilling sediment cores, and test wells in order to locate potential oil and gas deposits (Johnson et al. 2008). Petroleum production includes the drilling and extraction of oil and gas from known reserves. Oil and gas rigs are placed on the seabed and as oil is extracted from the reservoirs, it is transported directly into pipelines. While rare, in cases where the distance to shore is too great for transport via pipelines, oil is transferred to underwater storage tanks. From these storage tanks, oil is transported to shore via tanker. According to the Minerals & Management Service, there are 21,000 miles (~38,000 km) of pipeline on the United States outer coastal shelf (OCS). According to the National Research Council (NRC), pipeline spills account for approximately 1,900 tons per year of petroleum into U.S. OCS waters, primarily in the central and western Gulf of Mexico. Other potential negative impacts include physical damage to existing benthic habitats within the "drop zone", undesired changes in marine food webs, facilitation of the spread of invasive species, and release of contaminants as rigs corrode (Macreadie et al. 2011).

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

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

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

EVALUATION AND SELECTION OF INDICATORS

To estimate the temporal trend in activities related to offshore oil and gas activities off California, we evaluated two indicators: oil and gas production and the number of oil and gas wells in the CCLME (Table AP4). Both indicators have long time series of data available and are easily understood by the public and policymakers. However, the number of oil and gas wells may not reflect how much continuous activity surrounds each oil platform or well, and thus may not capture the variability associated with impact to the seafloor. Production of oil and gas from producing wells will capture the potential effects of continued activities (e.g., new anchorings, drilling, or maintenance of wells) on the seafloor. In addition, available data for production values have a broader temporal extent (1970 – 2012) than number of wells (1981 – 2012), thus production values rated higher and will be used to measure status and trends of this pressure.

We retrieved state and federal offshore oil and gas production data from reports of the California State Department of Conservation's Division of Oil, Gas, and Geothermal Resources (ftp://ftp.consrv.ca.gov/../pub/oil/annual_reports/) for the years 1981 – 2009. A second on-line data resource, the National Ocean Economics Program at the Monterey Institute of International Studies (http://www.oceaneconomics.org/Minerals/oil_gas.asp), was used to verify these numbers and expand the temporal extent of the production rate data series from 1970 to 2012. Estimates of natural gas production for state and federal offshore wells were accessible through the U.S. Energy Information Administration (http://www.eia.gov/dnav/ng/ng_prod_sum_dcu_rcatf_a.htm). Total oil production and total gas production were normalized independently, summed together and renormalized to create an index of oil and gas production in the CCLME.

STATUS AND TRENDS

The status and trends of offshore oil and gas activity in the CCLME were measured using a normalized index of oil and gas production from offshore wells in state and federal waters in California (Table AP5). Offshore oil and gas activity in the CCLME has been stable over the last five years, but the short-term mean was more than 1 SD below the long-term mean (Fig. AP49). Oil and gas production has declined steadily since the mid-1990's.

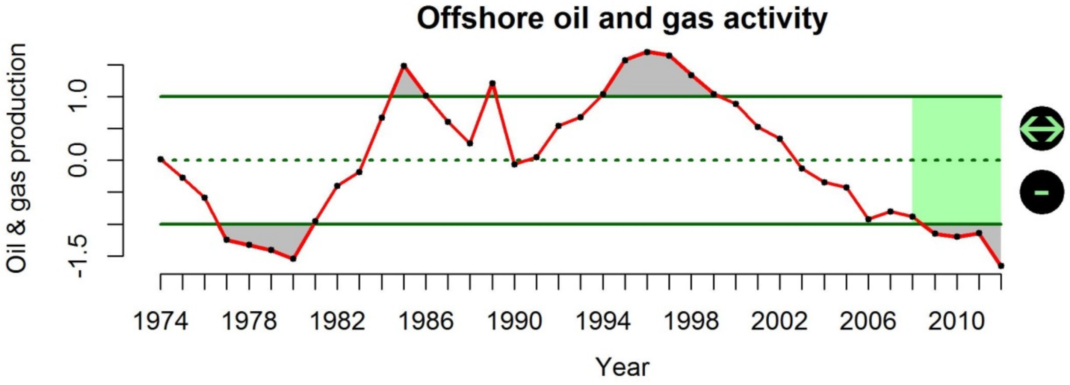


Figure AP49. Normalized index of the sum of oil and gas production from offshore wells in CA.

ORGANIC POLLUTION

BACKGROUND

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

Pesticides can affect the health and productivity of biological populations in three basic ways: (1) direct toxicological impact on the health or performance of exposed individuals; (2) indirect impairment of the productivity of the ecosystem; and (3) loss or degradation of vegetation that provides physical structure for fish and invertebrates (Hanson et al. 2003, Johnson et al. 2008). For many marine organisms, the majority of effects from pesticide exposures are sublethal, meaning that the exposure does not directly lead to the mortality of individuals. Sublethal effects can be of concern, as they impair the physiological or behavioral performance of individual animals in ways that decrease their growth or survival, alter migratory behavior, or reduce reproductive success (Hanson et al. 2003, Johnson et al. 2008), but in general the sublethal impacts of pesticides on fish health are poorly understood. Early development and growth of organisms involve important physiological processes and include the endocrine, immune, nervous, and reproductive systems. Many pesticides have been shown to impair one or more of these physiological processes in fish (Gould et al. 1994, Moore and Waring 2001). The direct and indirect effects that pesticides have on fish and other aquatic organisms can be a key factor in determining the impacts on the structure and function of ecosystems (Preston 2002). One of the most widely recognized effects of organic pollution was the decline of bald eagles and brown pelicans during the 1960's and 1970's. These birds accumulated DDT in their tissues, which changed their ability to metabolize calcium, which resulted in birds producing abnormally thin eggshells that led to reproductive failure (Hickey and Anderson 1968, Blus et al. 1971).

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

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

EVALUATION AND SELECTION OF INDICATORS

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

Recovery-adjusted concentrations (micrograms/liter) of 16 pesticides detected most frequently in urban streams were assessed by the U.S. Geological Survey using data from sites all across the United States (Ryberg et al. 2010, Martin et al. 2011). These data are easily accessible from the U.S. Geological Survey (http://pubs.usgs.gov/ds/655/). We used data identified for trend analysis (trend = "KEEP" from USGS data) and from sites located in watersheds that drain into the CCLME (states of WA, OR, ID and CA). We calculated the mean recovery-adjusted concentration across all samples within a site for each pesticide for each year (1992 – 2010). We then averaged the mean site values for each pesticide across all sites to provide a final average for each pesticide for each year. Because three of the pesticides (fipronil, desulfinylfipronil, and fipronil sulfide) did not have data prior to 2002, we eliminated them. We then multiplied the averaged concentrations by their toxicity score and summed these values across all pesticides for each year. The toxicity score was calculated by dividing the pesticide's Indiana Relative Chemical Hazard Score (https://engineering.purdue.edu/CMTI/IRCHS/) by 100 (maximum value of the scoring system). For pesticides that were not in the IRCHS list, we used the average value of the other pesticides in our dataset.

STATUS AND TRENDS

The status and trends of organic pollution in the CCLME were measured using a toxicity-weighted index of recovery-adjusted concentrations of 13 pesticides measured in streams in watersheds that drain into the CCLME (Table AP5). Using this indicator, organic pollution has decreased over the last five years of the dataset, and the short-term average is within 1SD of the long-term average of the time series (Fig. AP50). Prior to this most recent trend, organic pollution showed large increases in the mid-2000's.

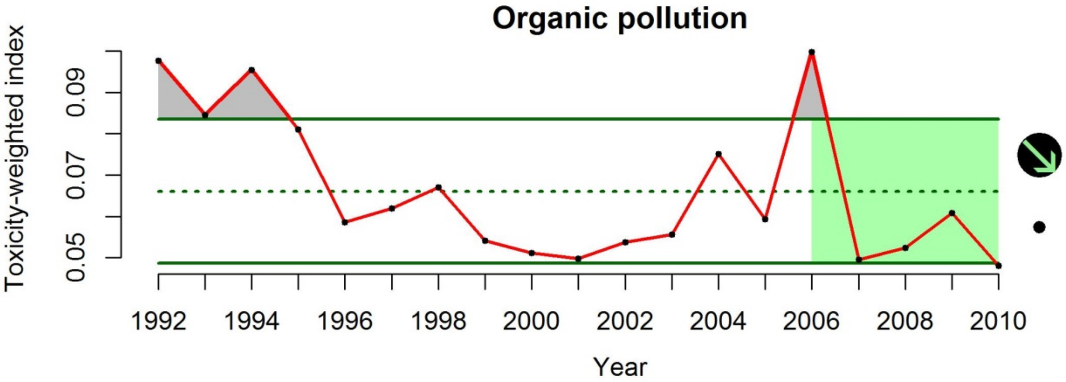


Figure AP50. Toxicity-weighted index of recovery-adjusted concentrations of 13 pesticides measured in streams in watersheds that drain into the CCLME.

POWER PLANTS

BACKGROUND

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

The entrainment and impingement of fish and invertebrates in power plant and other water intake structures have immediate as well as future impacts to estuarine and marine ecosystems (Johnson et al. 2008). Most of the immediate impact is removal of eggs, larvae and juveniles; not only is that fish and invertebrate biomass removed, but the biomass that would have been produced in the future will not become available to the ecosystem. Water intake structures, such as power plants and industrial facilities, are a source of mortality for managed-fishery species and play a role as one of the factors driving changes in species abundance over time. Organisms that are too large to pass through in-plant screening devices become stuck or impinged against the screening device or remain in the forebay sections of the system until they are removed by other means.

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

EVALUATION AND SELECTION OF INDICATORS

We evaluated two indicators of power plant activity in the CCLME: 1) average daily saline water withdrawal volumes and 2) daily entrainment mortality (Table AP4). The largest total thermoelectric withdrawals on the West Coast are in California, where nearly all of the water was withdrawn from marine surface waters for use in once-through cooling systems (Kenny et al. 2009). Washington and Oregon thermoelectric power withdrawals rely almost exclusively on fresh surface waters. In 2005, the total daily water withdrawals for thermoelectric power generation from all West Coast states combined (WA, OR, CA) equaled over 49 million m^3/d, with the vast majority (96%; 47.7 million m^3/d) attributed to CA marine surface water withdrawals. Over the course of record-keeping, marine surface water withdrawals from California have consistently represented more than 80% of West Coast thermoelectric water withdrawals.

The USGS has conducted water-use compilations in the United States every 5 years since 1950 (http://water.usgs.gov/watuse/50years.html), and thermoelectric power has represented the largest total category of water withdrawals in every compilation since 1960 (Hutson et al. 2005, Kenny et al. 2009). Withdrawals by thermoelectric-power plants across the entire U.S. have ranged from a low of 151 million m^3/d during 1950 to a high of 794 million m^3/d in 1980. In 2005, thermoelectric water withdrawals totaled 760 million m^3/d and comprised 49 percent of total water use across the entire U.S. Declines in thermoelectric-power water withdrawals from 1980 to present are primarily a result of Federal legislation requiring stricter water-quality standards for return flow and by limited water supplies in some areas of the United States. Consequently, power plants have increasingly been built with or converted to closed-loop cooling systems or air-cooled systems instead of using once-through cooling systems. By 2000, an alternative to once-through cooling was used in about 60 percent of the installed steam-generation capacity in the power plants (Hutson et al. 2005).

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

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

STATUS AND TRENDS

The status and trends of power plants in the CCLME were measured using the volume (millions of m^3) of saline water withdrawn daily by thermoelectric power plants in WA, OR and CA (Table AP5). Because these data were sampled every 5 years, we interpolated the annual value over the last five years (asterisks in Fig. AP51) assuming a linear relationship between the last two data points in order to keep the short-term status (most recent five years) consistent with the other pressure indicators. The mean and SD of the dataset were calculated using the original dataset. Power plant activity was stable over the last five years of the dataset (2000 – 2005), but the short-term average was >1SD above the long-term average (Fig. AP51). Trends of water withdrawals by thermoelectric power plants have been stable or decreasing across the U.S. since the 1980's (Kenny et al. 2009), so the CCLME may have slightly elevated its power plant activity compared to the rest of the U.S. in the early 2000's. The 2010 report on estimated use of water in the United States was delayed and data availability is not expected until late in 2014.

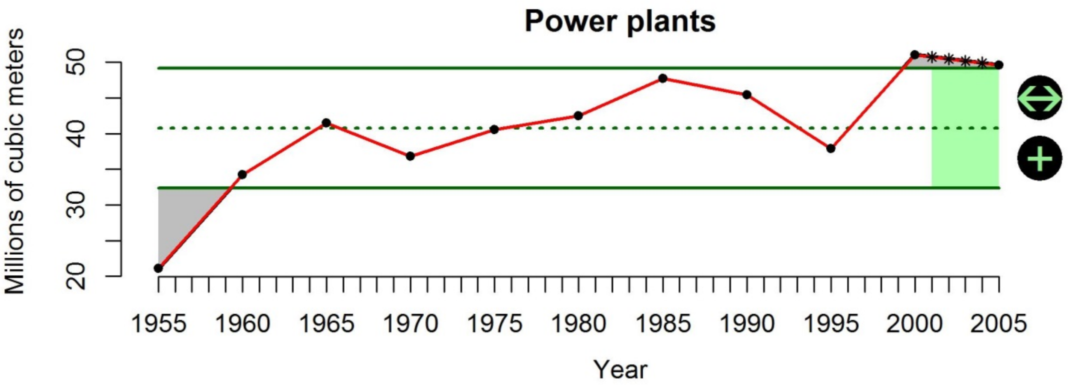


Figure AP51. Daily saline water withdrawals (millions m^3) from thermoelectric power plants in CA, OR and WA. Asterisks are interpolated values, but used to calculate short-term status and trends since this indicator is only measured every 5 years.

RECREATIONAL USE

BACKGROUND

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

EVALUATION AND SELECTION OF INDICATORS

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

For California, we extracted total visitor attendance at 48 California state parks identified as "State Beach" from the California State Park System Annual Statistical Reports: 2002 -2012 (http://www.parks.ca.gov/?page_id=23308). For Oregon, the only measure of annual beach attendance is collected by the Oregon Parks and Recreation Department's Stewardship Division for the years 2002 – 2013. This estimate is measured using automated car counters in the parking lots of coastal state parks. These estimates are based on the assumption that there are on average four occupants per vehicle (based on results of a statewide visitor survey). These measures are likely an overestimate of actual pressure on the associated beaches as some people use the parking lots and do not go to the beach. For Washington, the Washington State Parks and Recreation Commission collects attendance data at parks with ocean beach access and these data are available in annual "Attendance Reports". We limited these datasets to years in which data were available for all three states (2002 – 2012) and to parks/beaches that were open and censused in all years (i.e. if a state park was closed at some point during the time series, this park was excluded from the analysis). We normalized each state's attendance independently across the time series, summed these normalized values across all states for each year, and then renormalized these data for the final time series. Using the normalized sums of attendance (instead of only the sums of attendance) provided an estimate that weighted changes in annual attendance equally among all states. Otherwise, changes in beach attendance in California would completely drive the final time series due to the much larger magnitude of beach attendance in California.

STATUS AND TRENDS

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

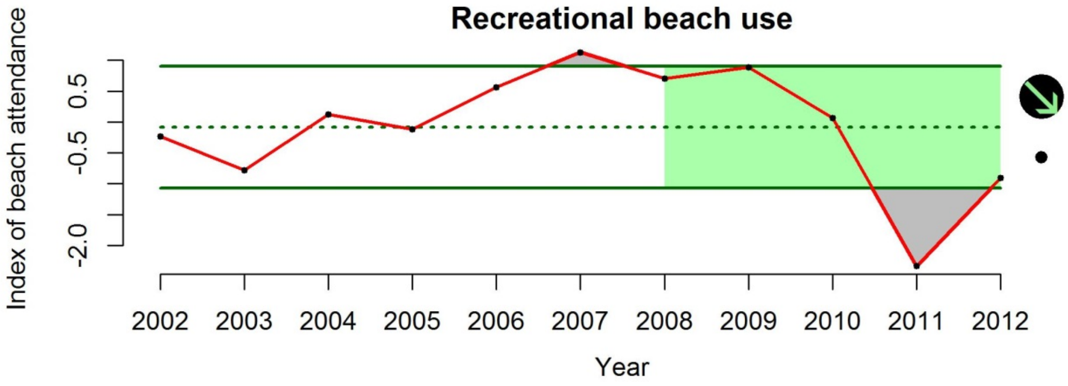


Figure AP52. Index of annual beach attendance at state parks and beaches with access points to a beach in WA, OR and CA.

SEAFOOD DEMAND

BACKGROUND

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

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

EVALUATION AND SELECTION OF INDICATORS

We identified two primary indicators of seafood demand: total consumption and per capita consumption (Table AP4). Both indicators are published in NOAA's "Fisheries of the United States" annual reports to describe the utilization of fisheries products (http://www.st.nmfs.noaa.gov/st1/publications.html). Total edible and non-edible seafood demand evaluates higher (Table AP4) because fundamentally total consumption provides a concrete estimate of what is being used, whereas per capita consumption is simply based on the total consumption estimates divided by the population of the U.S.

We retrieved total consumption estimates (billion pounds) of total (imports and commercial landings) edible and non-edible seafood from each of the Fisheries of the United States annual reports which provided data from 1962 – 2012. Data were converted to millions of metric tons.

STATUS AND TRENDS

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

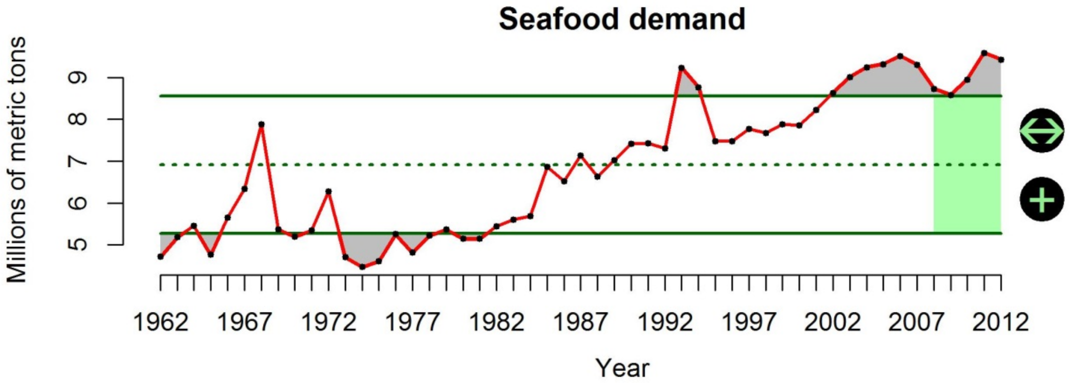


Figure AP53. Total consumption of edible and non-edible fisheries products across the United States.

SEDIMENT INPUT

BACKGROUND

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

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

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

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

EVALUATION AND SELECTION OF INDICATORS

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

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

A recent report from USGS summarizes the annual mean loads for SS in the Puget Sound and Columbia River basins using the USGS computer program Load Estimator (LOADEST), which uses a linear regression model that incorporates flow, time, and seasonal terms to estimate loads of mass over specified time periods (for this study, annual loads) (Wise et al. 2007). During water year 2000, considered an average streamflow year in the Pacific Northwest, the Columbia River discharged about 12,700 metric tons per day of SS to the Pacific Ocean. For most catchments between water years 1993-2003, the net change in non-hydrologic characteristics (land use and other human activities) was not great enough to cause any significant ($p \leq 0.05$) flow-adjusted trend in concentration (FATC) for suspended sediment (SS). Nineteen of the 48 sites available for SS trend analysis had significant FATC for SS (4 increasing, 15 decreasing), seven sites showed significant trend in load for SS (1 increasing, 6 decreasing), and more than 65 percent of the sites had decreasing (but not necessarily significant) FATC and trend in load for SS. There is currently no comparable analysis available for California basins.

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

STATUS AND TRENDS

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

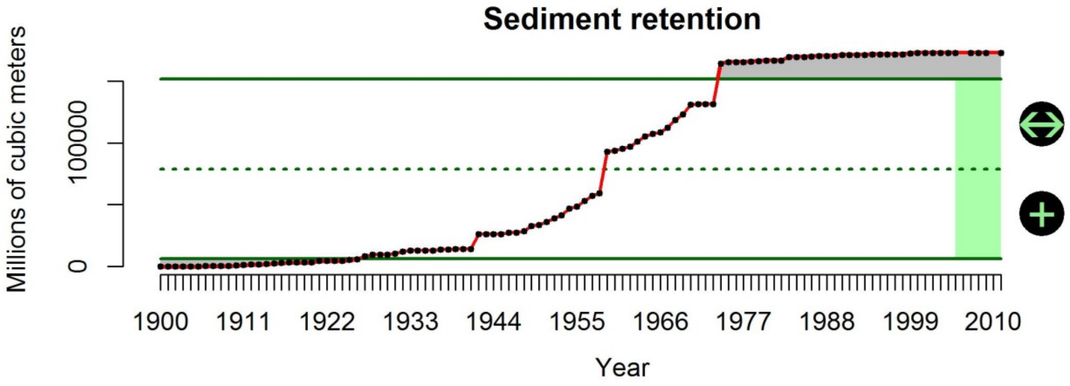


Figure AP54. Volume (millions m^3) of freshwater impoundments in WA, OR and CA (increasing freshwater storage is a proxy for decreasing sediment input).

TOURISM

BACKGROUND

This pressure has not been evaluated to date.

EVALUATION AND SELECTION OF INDICATORS

This pressure has not been evaluated to date.

STATUS AND TRENDS

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

LINKAGES BETWEEN DRIVERS AND EBM COMPONENTS

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

The linkage between fisheries and several IEA EBM components is direct: fishery removals decrease abundance of targeted fisheries as well as some protected species via directed removals and bycatch. The Pacific Fishery Management Council uses biological reference points to determine whether a stock is in an overfished state, and whether overfishing is occurring. For groundfish, for instance, the former is determined using an estimated depletion level, which is the ratio of spawning stock output (number of eggs or embryos) in the fished condition, to the spawning output in the unfished condition. The latter is determined by a fishing mortality rate (F), expressed based on spawning potential ratio (SPR). This ratio is the number of eggs produced by an average recruit over its lifetime when the stock is fished, divided by the same metric when the stock is unfished. The SPR is based on the principle that certain proportions of fish have to survive in order to spawn and replenish the stock at a sustainable level. When removals or fishing mortality exceed established reference points, management measures are implemented to correct the issue. There had been significant declines in a number of groundfish species managed by the Pacific Fishery Management Council. Since implementing the Magnuson-Stevens Act (MSA) of 1976, the Sustainable Fisheries Act (SFA) of 1996, and the reauthorization of MSA in 2006, many species have increased their abundance toward levels where they are not considered overfished, and overfishing of these species is not occurring (Miller et al. 2009). For example, lingcod, which dropped below 10% of its unfished biomass in 1986, was fully rebuilt in 2005, four years earlier than the target year established in the species rebuilding plan (Hamel et al. 2009). Based on the most recent rebuilding analyses, all groundfish species that are still considered overfished exhibit upward trends, with three species (yelloweye rockfish, bocaccio and darkblotched rockfish) being ahead of their rebuilding plan schedules (Field 2011, Stephens 2011, Taylor 2011).

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

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APPENDIX AP1. CUMULATIVE INDICES, CORRELATIONS, AND COMMON TRENDS AMONG ANTHROPOGENIC PRESSURES WITHIN THE CCLME

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*Note: This appendix provides a methodological framework for calculating cumulative indices using the time series of all indicators. It also examines methods that reduce the dimensionality of the dataset, so that multiple pressures could be incorporated into other science-based management tools. The analyses performed in this section used time series that had not been updated with 2012 data.

SUMMARY

As human population size and demand for seafood and other marine resources increase, the influence of human activities in the ocean (e.g., fishing and shipping activity) and on land (e.g., industrial and agricultural activities) is increasingly critical to the management and conservation of marine resources. In order to make management decisions related to anthropogenic pressures on marine ecosystems, we need to understand the links between pressures and ecosystem components, and we cannot draw those linkages unless we know how pressures have been changing over time. We developed indicators and time series of indicators for 22 anthropogenic pressures at the scale of the U.S. portion of the California Current ecosystem. Time series suggest that seven pressures have decreased and two have increased over the short term, while five pressures were above and two pressures were below long-term means. Cumulative indices of anthropogenic pressures suggest a slight decrease in pressures in the 2000's compared to the preceding few decades. Dynamic factor analysis revealed four common trends that sufficiently explained the temporal variation found among all anthropogenic pressures. Using this reduced set of time series will be useful when trying to determine whether links exist between individual or multiple pressures and various ecosystem components.

INTRODUCTION

Human activities in, on, and around the ocean—from shipping and fishing to urbanization, oil extraction, aquaculture, and coastal agriculture—are varied and growing. These activities generate many benefits, including production of food, employment, energy, and livelihoods (Guerry et al. 2012). However, they are also associated with anthropogenic pressures on the natural ecosystem that have a variety of consequences, such as loss or modification of habitat, extractions and introductions of species, visual and auditory disturbances, and the introduction of toxic and non-toxic contamination (Eastwood et al. 2007). Despite widespread recognition of the increasing importance of these diverse influences (Wilson et al. 2005, Halpern et al. 2007), it is rare to find a full accounting of how anthropogenic pressures in the marine environment have changed over time.

In contrast, recent spatial analyses of anthropogenic activities have revealed hotspots of individual and overlapping pressures in ecosystems across the globe (Ban and Alder 2008, Halpern et al. 2008, Halpern et al. 2009, Stelzenmüller et al. 2010, Hayes et al. 2012). These maps show patterns of spatial variation among individual and cumulative pressures that provide a framework scientists and managers can use to focus limited resources on areas of concern. They also beg the question of how anthropogenic pressures in specific locations have changed over time. Without an understanding of the legacy of anthropogenic pressures in an area, it is difficult to interpret current and potential future conditions. For instance, the ecological consequences of oil extraction in a previously untouched area like the North Slope of Alaska are likely to be very different than in a historically high-use environment such as the North Sea. Unfortunately, time series data for many human-related pressures are often buried in state and federal agency reports, described at small spatial scales, and measured inconsistently among local, state and federal entities. Thus, it would be helpful to develop a standardized set of time series that reflect the status and trends of these pressures at scales appropriate for management.

Importantly, pressures do not act upon the ecosystem independently, but rather collectively. Pressures are disparate and broadly based, ranging from terrestrial-based pollution, commercial shipping activities, and offshore energy development to fisheries and coastal development, all of which exert cumulative effects on the ecosystem and could benefit from a holistic management approach (Vinebrooke et al. 2004, Crain et al. 2008, Halpern et al. 2008, Curtin and Prellezo 2010). Quantifying the cumulative effects from multiple pressures is a challenging task, however, because we have a limited understanding of how pressures interact and whether the cumulative effects are additive, synergistic or antagonistic (Darling and Côté 2008, Hoegh-Guldberg and Bruno 2010)? Moreover, the results of these interactions may have different consequences for different taxa or ecosystem components (Crain et al. 2008). Additionally, the status and trends of many anthropogenic pressures are likely correlated with each other due to ultimate drivers such as human population growth, seafood demand or economic conditions, and so are best understood in the context of one another (e.g., Link et al. 2002).

Recent studies that aim to evaluate the effect of cumulative pressures on marine ecosystems have assumed that pressures are additive (Halpern et al. 2009, Stelzenmüller et al. 2010); however, the relative importance of each pressure on a given habitat, region, or ecosystem was incorporated into the calculation by assigning relative weightings to each individual pressure (based on expert opinions (e.g., Teck et al. 2010) or based on spatial extent of a pressure (e.g., Stelzenmüller et al. 2010)). These methodologies may also be used to calculate cumulative pressures to determine the relative status and trends over time.

Here, we developed standardized time series of indicators for 22 anthropogenic pressures acting across the entire U.S. portion of the California Current Large Marine Ecosystem (hereafter, the California Current ecosystem (CCE)). These time series were used to quantify the status and temporal trends of each pressure. We then used several approaches to describe the relative status and trends of anthropogenic pressures as a whole. First, we used simple additive models to quantify the relative status and trends of anthropogenic pressures in the California Current ecosystem. Second, we used multivariate models to determine (1) whether pressures were correlated, (2) how the composition of pressures changed over time, (3) whether there were shared trends in the time series of anthropogenic pressures, and (4) whether these trends were related to specific drivers such as coastal population abundance or economic activity. Our synthesis, and corresponding methodological approaches to quantify the status and trends of these pressures, provides a foundation for future integrative analyses on ecological components (e.g., risk analysis and management strategy evaluations) across the CCE.

METHODS

INDICATORS OF ANTHROPOGENIC PRESSURES

We developed indicators for 22 anthropogenic pressures in the California Current ecosystem (CCE). The pressures selected were derived primarily from those identified in spatial analyses by Halpern et al. (2009) and by vulnerability analyses by Teck et al. (2010); they ranged in scope from land-based pressures such as inorganic pollution and nutrient input to at-sea pressures such as commercial shipping and offshore oil and gas activities. Ultimately, we evaluated 41 different indicators and selected the best indicator to describe the status and trends of each pressure. Indicators were evaluated (see “Detailed Report” above) using the indicator selection framework developed and used by Levin et al. (2011), Kershner et al. (2011) and James et al. (2012). Briefly, we evaluated each indicator according to 18 criteria using the scientific literature to determine whether there was support for each criterion for each indicator. This resulted in a matrix of references and notes with a corresponding value of literature support (1 for "support", 0.5 for "ambiguous support", 0 for "no support"). These values of literature support were summed across criteria for each indicator and the highest scoring indicator was chosen for each pressure.

Data for all indicators were compiled from various state and federal reports and databases to create the longest possible time series for each pressure (Table AP1-1). Compatible data from the states of California, Oregon and Washington were pooled to characterize pressures at the scale of the entire CCE. In some instances (see descriptions of individual pressures in "Detailed Report" above), data from other states were included if watersheds in other states drained into the Pacific Ocean. To alleviate some of the complexities associated with different institutional data standards, governing jurisdictions, and geographic discrepancies, we limited our analysis to U.S. data and did not include data from portions of the CCE in Canada or Mexico.

The status of each indicator was evaluated against two criteria: recent short-term trend (increasing, decreasing or remaining the same over the last five years) and status relative to the mean and variance of long-term conditions (short-term status was higher than, lower than or within historic levels) (Levin and Wells 2012). An indicator's current trend was considered to have changed in the short-term if the modeled trend over the last five years of the time series showed an increase or decrease of more than 1.0 standard deviation (SD) of the mean of the entire time series. An indicator's current status was considered to be above or below historical levels if the mean of the last five years was greater than or less than 1.0 SD from the mean of the full time series, respectively. Defining the "short-term" as the last five years of the dataset is consistent with other management review processes that occur at the scale of large marine ecosystems (e.g., National Marine Fisheries Service's Essential Fish Habitat reviews (NMFS 2013) and the National Oceanic and Atmospheric Administration's Integrated Ecosystem Assessments (Levin and Schwing 2011, Levin and Wells 2012)).

Table AP1-1. Top indicators for anthropogenic pressures in the California Current ecosystem (CCE). See "Detailed Report" for "Anthropogenic Drivers and Pressures" above for evaluation and selection, source of data and calculations of indicators for each pressure.

Pressure	Indicator	Definition	Time series	Sampling frequency
***Aquaculture: finfish**	Finfish production	Estimates of Atlantic salmon production in CCE waters.	1986 – 2011	yearly
***Aquaculture: shellfish**	Shellfish production	U.S. shellfish (clams, mussels & oysters) production.	1985 – 2010	yearly
***Atmospheric pollution**	Deposition of sulfate	Annual precipitation-weighted mean concentrations of sulfate measured at sites in CA, OR, and WA.	1994 – 2010	yearly
***Benthic structures**	# offshore oil & gas wells	Total number of offshore oil and gas wells in production.	1981 - 2009	yearly

Pressure	Indicator	Definition	Time series	Sampling frequency
***Coastal engineering**	Human coastal population	Population size of coastline counties in CA, OR, WA.	1970 – 2012	yearly
Commercial shipping activity	Volume of water disturbed	Calculated using draft, breadth and distance traveled within CCE of domestic and foreign vessels.	2001 – 2010	yearly
Dredging	Dredge volumes	Dredge volumes for individual private contracts and Army Corps operated dredge projects in WA, CA, and OR.	1997 – 2011	yearly
***Fishery removals**	Total Landings	Metric tons of all species landed by commercial and recreational fisheries in CA, OR and WA.	1981 – 2011	yearly
***Freshwater retention**	Impoundment storage volume	Total reservoir storage volume in CA and Pacific Northwest water resource regions.	1900 – 2011	yearly
Habitat modification	Distance trawled	Kilometers trawled by the limited-trawl groundfish fishery in CA, OR and WA.	1999-2004	yearly
***Inorganic pollution**	ISA-toxicity-weighted chemical releases	Total pounds of inorganic pollutants disposed of or released on site to the ground or water for '1988 core chemicals' weighted by toxicity scores and impervious surface area (ISA) in the drainage watersheds of the CCE.	1988 – 2010	yearly
***Invasive species**	Tons of cargo	Tons of cargo moved through ports in CA, OR and WA.	1993 – 2010	yearly
***Light pollution**	Average nighttime visible light	Data are cloud-free composites of average visible nighttime lights made using all the available archived DMSP-OLS smooth resolution data for each calendar year.	1994 – 2010	yearly
Marine debris	Predicted counts of debris	Estimates from the National Marine Debris Monitoring Program separated into north and south CCE estimates.	1999 – 2007	yearly
***Nutrient input**	Nitrogen and phosphorus input	Total farm and non-farm nitrogen and phosphorus input from fertilizer used in counties within CCE watersheds.	1945 – 2010	yearly
Ocean-based pollution	Commercial shipping activity combined with tons of cargo	Combines "Commercial shipping activity" and "Invasive species" datasets.	2001 – 2010	yearly
***Offshore oil activities**	Offshore oil & gas production	Normalized sum of the number of barrels of oil and cubic feet of gas produced by offshore wells in CA.	1970 – 2010	yearly
***Organic pollution**	Toxicity-weighted concentrations	Toxicity-weighted concentrations of 16 pesticides measured in water samples from stream-water sites in WA, OR and CA	1993 – 2008	yearly
Power plants	Saline water withdrawal volumes	Average daily withdrawal volumes of saline water from thermoelectric power plants in the Pacific Northwest and California regions.	1955 – 2005	Every 5 years
Recreational beach use	Beach attendance	Summed beach attendance from CA, OR, and WA	2002 – 2011	yearly
***Seafood demand**	Total consumption	Total consumption of edible and non-edible fisheries products in the U.S.	1962 – 2011	yearly
***Sediment retention**	Impoundment storage volume	Same as "Freshwater retention"	1900 – 2011	yearly

*Pressures used in cumulative pressures index and principal components analysis

The historical status of each indicator should be placed in context with the amount of data available for each time series. For example, the entire time series for one indicator (habitat modification) was only six years while the time series for other indicators (e.g., freshwater and sediment retention) was > 100 years. For shorter time series, the mean of the last five years (short-term) was not likely different from the mean of the entire time series; thus, the relative status for indicators with short time series was more related to the availability of data and not historic trends. However, indicators were chosen because they were the most fundamentally-sound datasets based on 18 evaluation criteria, only 7 of which were related to data availability (see "Detailed Report" above). Moreover, most of the indicators chosen will continue to be measured, thus providing meaningful comparisons into the future.

SUMMARIZING ANTHROPOGENIC PRESSURES AS A WHOLE

We employed three different methods to examine the status and trends of pressures as a whole. First, we calculated a cumulative pressures index using a subset of pressures. Second, we used principal components analysis to examine correlations and temporal shifts among pressures. Last, we used dynamic factor analysis to determine whether the 22 pressures could be reduced to a smaller number of common trends.

CUMULATIVE PRESSURES INDEX

In order to calculate a cumulative pressures index, we determined the longest period for which there were the most pressures with continuous indicator data available. For the years 1994 – 2008, we had annual data available for 15 of the 22 pressures (Table AP1-1). Data from these 15 time series were normalized (mean = 0, standard deviation = 1) across the years 1994 – 2008 so that all pressures were on the same scale. We then used two methods to calculate a cumulative pressures index. The first method was simply an additive model in which all 15 normalized pressure values were summed for each year (an equal weighting of "1.0" for each pressure).

The second method weighted the relative importance of each pressure according to mean vulnerability scores determined by Teck et al. (2010). Briefly, we normalized mean vulnerability scores of all pressures to a scale of 0 to 1 and used the scores relevant to our 15 pressures as weightings. Mean vulnerability scores were averaged across pressure categories when more than one related to one of our 15 pressures (e.g., four nutrient input pressures were identified in Teck et al. (2010)). Finally, we multiplied each pressure value in the time series by its respective weighting value and summed across all pressures for each year.

CORRELATIONS AND TEMPORAL SHIFTS AMONG PRESSURES

We used principal components analysis (PCA; PRIMER 6.0; Clarke and Gorley 2006) to identify correlations among pressures and to reduce the number of multivariate dimensions to a smaller set that explained most of the variance of the data sets. Because PCA cannot accommodate missing values, we used the same set of 15 pressures from 1994

– 2008 that we used to calculate the cumulative pressures index to get the greatest number of pressures across the longest period of time. Loadings (correlations between the original time series and a principal component axis) greater than 0.30 were considered to have relevance for interpretation of the results (Tabachnick and Fidell 1996). We used the principal component scores across years to examine how the importance of each axis changed over time.

COMMON TRENDS AMONG PRESSURES

We used dynamic factor analysis (DFA; Zuur et al. 2003a, 2003b) to characterize underlying common trends among the time series of anthropogenic pressures. The objective of DFA is similar to PCA; to reduce the number of multivariate dimensions needed to describe patterns in data. However, DFA is based on time series models that explicitly account for temporal autocorrelation common in time series data; PCA does not. The DFA framework consists of two models: it combines (1) a random-walk model that captures the underlying shared trends among a set of time series and any covariates and (2) a model that describes how well each time series is described by each underlying trend.

In the DFA framework, a set of one or more hidden common trends (linear combinations of a set of random walks) shared by the time series data explains their temporal variations (Zuur et al. 2003a). DFA is particularly useful for our time series because it can account for missing values; thus, we can incorporate a larger number of pressures across a longer period than was possible for the calculation of the cumulative pressures index or the principal components analysis. Because DFA allows for the inclusion of covariates, we could also explore explanatory drivers of the pressures such as population size or economic growth.

Using the MARSS package in R (Holmes et al. 2012, R Development Core Team 2012), we tested models with 1 – 5 common trends and models including zero, one or two covariates (coastal human population abundance and gross domestic product of the U.S. West Coast). Preliminary analyses tested five commonly used variance-covariance matrix structures available in the MARSS package (Holmes et al. 2012) and suggested 'diagonal and equal' was the most appropriate for this data set (*see* "Supplementary Material" below). This model structure had observation variances (along the diagonal) that were equal and covariances that were equal to zero (Holmes et al. 2012).

Prior to the analysis, time series of all 22 pressures (Table AP1-1) were normalized across the period of interest (1985 – 2011). We limited the time series to this period because longer time series have proportionately greater influence than shorter time series in determining common trends and only a third of the indicators had longer time series (*see* individual pressures in "Detailed Report" above). We used Akaike's model selection criterion (AICc; Burnham and Anderson 1998) values to determine the fewest common trends and covariates required to explain the full set of time series of anthropogenic pressures in the CCE. We used an oblique rotation method (promax) to calculate factor loadings as it helped separate factor loadings among trends a little better than the default orthogonal method (varimax). DFA factor loadings > 0.2 were considered relevant for interpreting whether pressures were represented by a specific trend (Zuur et al. 2003b). Loading values represent coefficient values that when multiplied by the respective trend value and summed across all trends produce fitted values for each year for each pressure (i.e. model fits shown in Fig. AP1-6).

For the covariate ‘coastal population abundance’, we used data from the U.S. Census Bureau (2010 – 2012: http://www.census.gov/popest/data/datasets.html) and the National Bureau of Economic Research (1970 – 2009: http://www.nber.org/data/census-intercensal-county-population.html). We limited data to ‘coastal’ counties in California, Oregon and Washington as defined by the National Oceanic and Atmospheric Administration (http://www.census.gov/geo/landview/lv6help/coastal_cty.pdf). For the covariate ‘gross domestic product’, data were summed annually across the states of California, Oregon and Washington from 1963 – 2011 (Bureau of Economic Analysis; http://www.bea.gov/iTable/index_nipa.cfm) using “Regional Data” by state across all industries.

RESULTS

INDICATORS OF ANTHROPOGENIC PRESSURES

Indicators of anthropogenic pressures in the California Current ecosystem (CCE; Table AP1-1) were chosen based on rankings in the indicator evaluation matrix (see “Detailed Report” above). Descriptions, status and trends of individual indicators are described in the “Detailed Report” above, but examples of indicator time series show that the short- and long-term status and trends of anthropogenic pressures in the CCE varied widely (Fig. AP1-1). Most indicators showed either significant short-term trends or their current status was at historically high or low levels (Fig. AP1-2). Indicators of inorganic, organic and ocean-based pollution, commercial shipping activity, recreational use, invasive species and habitat modification have all decreased over the short-term, while indicators of dredging and marine debris (in the northern CCE) increased; all of these pressures, though, remained within historic levels. In contrast, indicators of seafood demand, sediment and freshwater retention, power plant activity and coastal engineering remained relatively constant over the short-term, but were above historic levels, while indicators of offshore oil and gas activity and related benthic structures were at historically low levels. Nutrient input and shellfish aquaculture were at historically high levels, but nutrient input has decreased over the last five years of its time series (Figs. AP1-1 & AP1-2), while shellfish aquaculture has continued to increase (Fig. AP1-2).

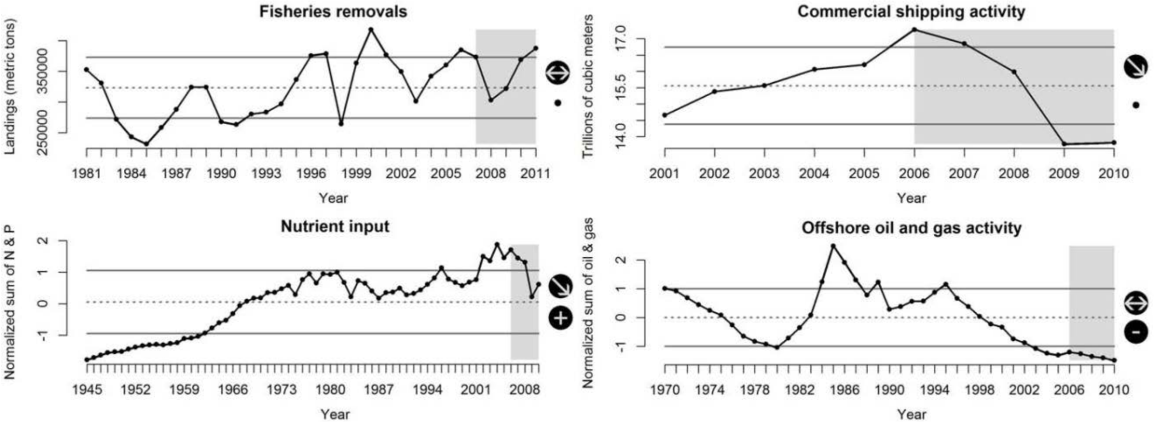


Figure AP1-1. Examples of the status and trends of anthropogenic pressures in the California Current ecosystem. Each pressure is represented by specific indicator data sets described in Table AP1-1 and the "Detailed Report". Arrows to the right of each panel represent whether the modeled trend over the last five years (shaded) increased (↗) or decreased (↘) by more than 1 SD or was within 1 SD (↔) of the long-term trend. Symbols below the arrows represent whether the mean of the last five years was greater than (+), less than (-) or within (•) 1 SD of the mean of the full time series (dotted line). Solid lines are ±1 SD of the mean of the full time series.

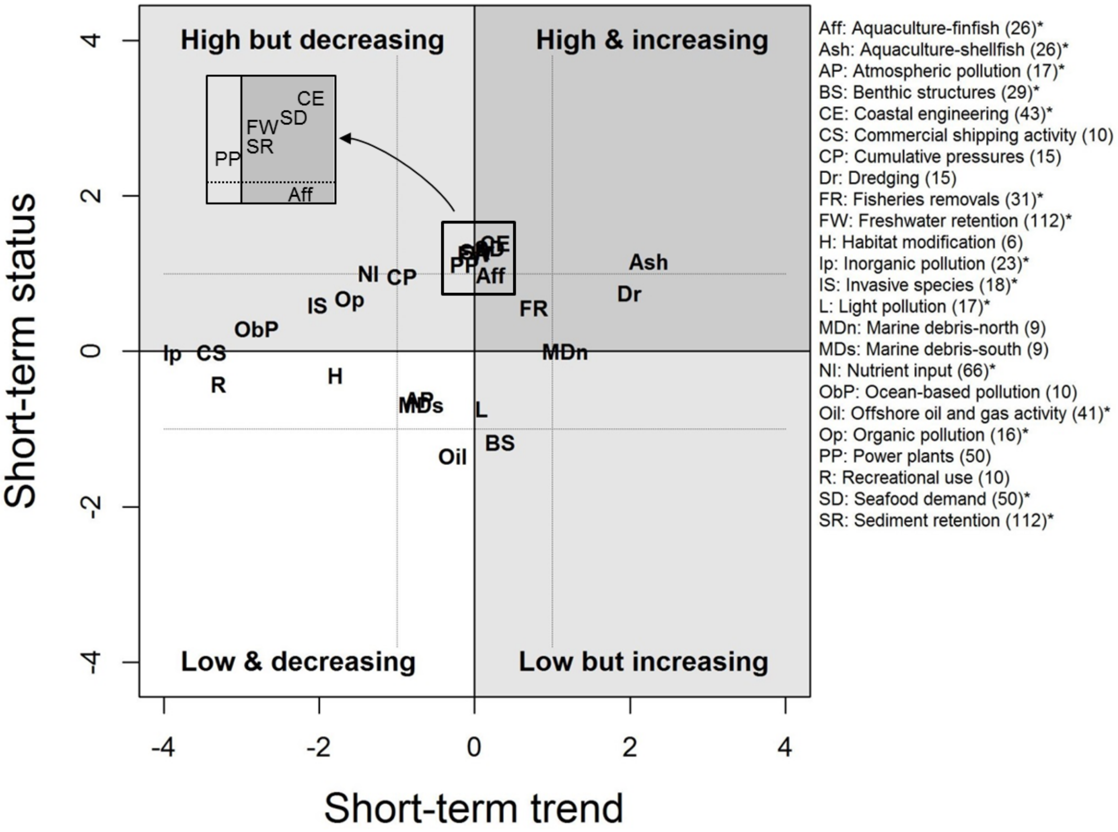


Figure AP1-2. Short-term status and trends of anthropogenic pressures in the California Current ecosystem. Prior to plotting, time series data for each indicator were normalized to place them on the same scale. The short-term trend indicates whether the indicator increased, decreased or remained the same over the last five years. The short-term status indicates whether the mean of the last 5 years was higher, lower, or within historical levels of the full time series. Data points outside the dotted lines (± 1.0 SD) are considered to be increasing or decreasing over the short term or the current status is higher or lower than the long-term mean of the time series. Numbers in parentheses in the legend are the number of years of data for each pressure. The "Cumulative pressures" indicator (see Figure AP1-3) is the additive sum of 15 of these pressures which had annual data from 1994 – 2008 (asterisks).

CUMULATIVE PRESSURES INDEX

The period of 1994 – 2008 provided the longest continuous period of data for the most indicators (15 of 22) to be included in the cumulative pressures index. The 'additive' and 'weighted' methods provided qualitatively similar estimates over this period (Fig. AP1-3). However, the additive index showed a positive trend (adjusted r^2: 0.51, $F_{1,13} = 15.7$, $p = 0.002$), whereas the weighted index showed no trend (adjusted r^2: 0.12, $F_{1,13} = 2.9$, $p = 0.110$) across the entire period. Using the same criteria to define the recent short-term status and trends of individual pressures, there was a short-term decrease in cumulative pressures using the weighted index, whereas there was no significant change in the short-term trend using the additive index (Fig. AP1-3). The short-term status for both indices was within historic levels of this time series.

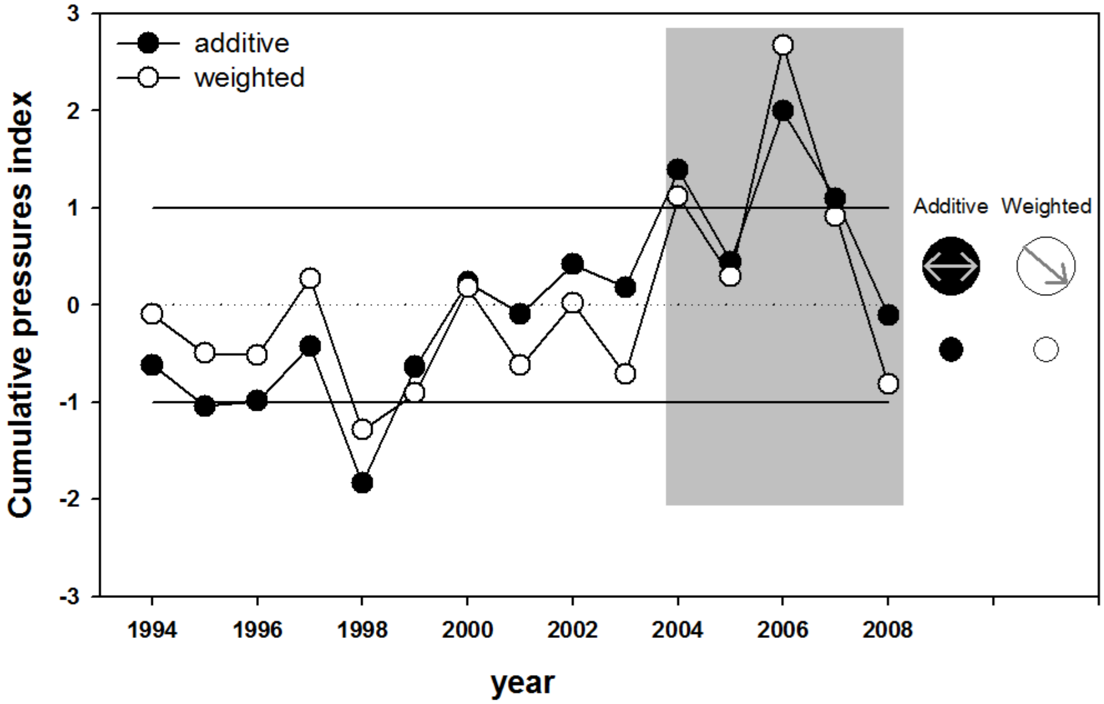


Figure AP1-3. Indices of cumulative pressures from 1994 – 2008 using 15 pressures which had data during this period: atmospheric, light, inorganic and organic pollution, nutrient input, shellfish and finfish aquaculture, invasive species, oil & gas activities, benthic structures, freshwater and sediment retention, coastal engineering, seafood demand and fisheries removals. Each index was normalized prior to plotting to place them on the same scale. 'Additive' is the sum of all pressure values each year; 'Weighted' is the sum of pressure values multiplied by their respective weighting values derived from Teck et al. (2010). See Figure AP1-1 for description of symbols, lines, and shading.

CORRELATIONS AND TEMPORAL SHIFTS AMONG PRESSURES

The first two axes of the principal components analysis explained ~68% of the total variation in the same 15 time series used to calculate the cumulative pressures index from 1994 to 2008, and the first four axes explained 86% (Fig. AP1-4). Plotting the scores of the first two principal components across time showed clear changes in the composition of pressures over this period (Fig. AP1-5). In the 1990's, there was strong influence by oil and gas activities, light pollution and benthic structures, while coastal engineering, seafood demand, nutrient input, aquaculture and organic and inorganic pollution became more important to this multivariate measurement in the 2000's. The spike observed in 2002 can be attributed to a particularly large increase in atmospheric pollution that year and the large change that occurred in 2006 was related to large increases of inorganic and organic pollution.

Sediment retention and freshwater input also loaded heavily on PC1, but in the complete time series for these pressures, they are relatively stable from 1994 to 2008 and thus would have little influence on any changes in cumulative pressure if the entire time series could have been used. Interestingly, 'fisheries removals', which was quite variable during this time period, was the only pressure that did not load significantly on PC1 or PC2, but instead loaded heavily on PC3 (Table AP1-2).

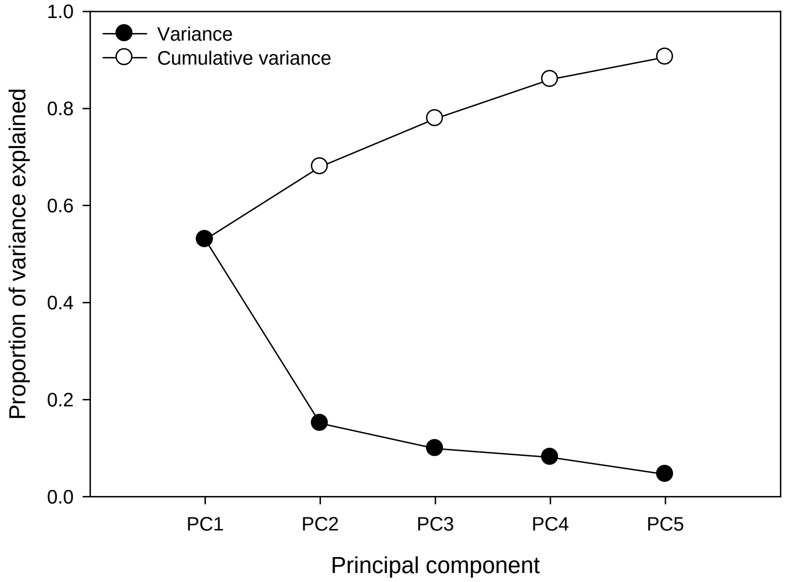


Figure AP1-4. Scree plot of principal components. PC5 had an eigenvalue < 1.0 suggesting that only PC1-4 were statistically relevant.

Table AP1-2. Principal component loadings for 15 pressures that had data from 1994 to 2008. Bold values indicate the principal component that each pressure is most closely correlated with.

Pressure	PC1	PC2	PC3	PC4
Aquaculture: finfish	**-0.64**	0.22	0.14	0.48
Aquaculture: shellfish	**-0.54**	-0.22	0.51	-0.35
Atmospheric pollution	-0.10	**0.76**	-0.22	-0.49
Benthic structures	**0.91**	-0.01	0.00	-0.13
Coastal engineering	**-0.95**	0.07	0.05	0.10
Fisheries removals	-0.21	-0.14	**-0.85**	0.29
Freshwater retention	**-0.90**	0.32	-0.10	0.16
Inorganic pollution	**-0.54**	-0.53	-0.47	-0.32
Invasive species	-0.08	**-0.80**	0.16	0.39
Light pollution	**0.95**	-0.21	-0.04	0.02
Nutrient input	**-0.81**	-0.32	0.14	-0.14
Oil & gas activities	**0.96**	-0.14	-0.04	-0.01
Organic pollution	**-0.56**	-0.48	-0.31	-0.40
Seafood demand	**-0.85**	-0.20	0.23	-0.17
Sediment retention	**-0.90**	0.32	-0.10	0.16

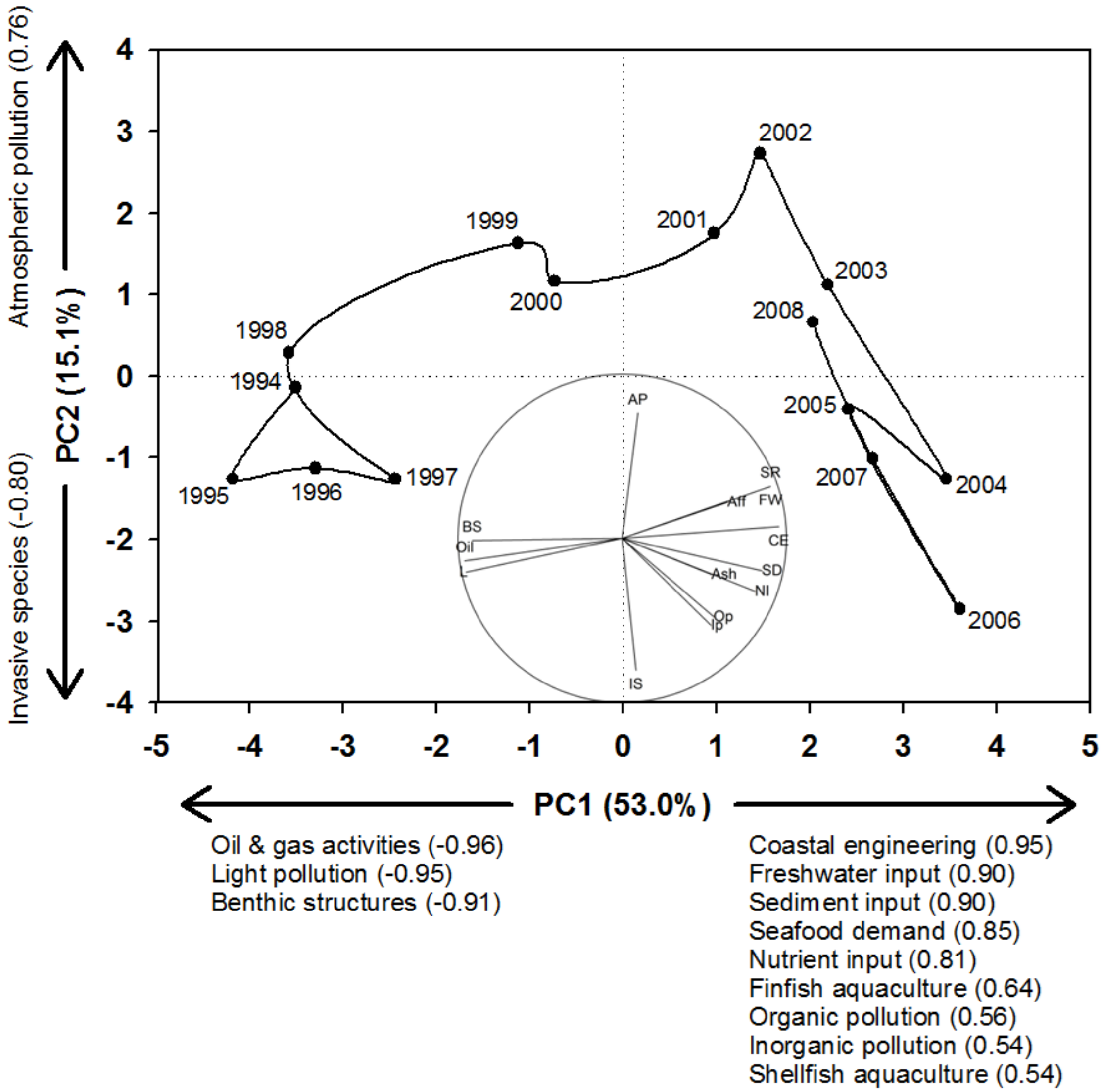


Figure AP1-5. Principal components analysis using indicators of 15 anthropogenic pressures which had data from 1994 – 2008: atmospheric, light, inorganic and organic pollution, nutrient input, shellfish and finfish aquaculture, invasive species, oil & gas activities, benthic structures, freshwater and sediment retention, coastal engineering, seafood demand and fisheries removals. Pressures identified along each axis had eigenvectors > 0.3 for one of the first two principal components, while the values in parentheses are the loading values for the predominant principal component for each pressure. See Figure AP1-2 for abbreviations.

COMMON TRENDS

Using dynamic factor analysis, we were able to include all anthropogenic pressures and data from 1985 to 2011. There were eight pressures having data prior to 1985, but including this data resulted in model convergence problems. Nonetheless, using DFA allowed us to include 7 additional pressures and 12 additional years of data compared to the cumulative pressures index or the principal components analysis. Model selection revealed a model with either 4 or 5 common trends with no covariates sufficiently explained the time series of pressure indicators (Table AP1-3). Because the model with 4 trends was more than twice as likely to be the best model as the two models with 5 trends, we used the 4-trend model to describe the common trends below. The 4-trend model had tight fits with most of the indicator time series, though a notable exception was "Fisheries removals" (Fig. AP1-6).

Table AP1-3. Model selection criteria from the top ten dynamic factor analysis models using all 23 indicator time series from 1985 to 2011 and comparing among different variance-covariance structures (R matrix), 1-5 trends and with 0-2 covariates.

R matrix	Trends	Covariate(s)	K	AICc	ΔAICc	Akaike weight	Cumulative Akaike weight
diagonal and equal	4	none	87	875.5	0.00	0.49	0.49
equal variance-covariance	5	none	107	877.2	1.68	0.21	0.70
diagonal and equal	5	none	106	877.4	1.89	0.19	0.89
diagonal and equal	3	population	90	879.6	4.12	0.06	0.95
equal variance-covariance	4	none	88	881.9	6.42	0.02	0.97
equal variance-covariance	3	population	91	882.7	7.19	0.01	0.98
diagonal and equal	2	both	92	884.5	8.97	0.01	0.99
diagonal and equal	4	population	110	885.4	9.90	0.00	0.99
diagonal and equal	3	gdp	90	885.8	10.30	0.00	1.00
equal variance-covariance	2	both	93	887.3	11.75	0.00	1.00

K = number of parameters; AICc = Akaike information criterion corrected for small sample sizes; ΔAICc = difference between each model and the lowest AICc from all possible models; population = coastal population abundance estimate; gdp = gross domestic product of U.S. West Coast states. See "Supplementary Material" below for description of each R matrix structure.

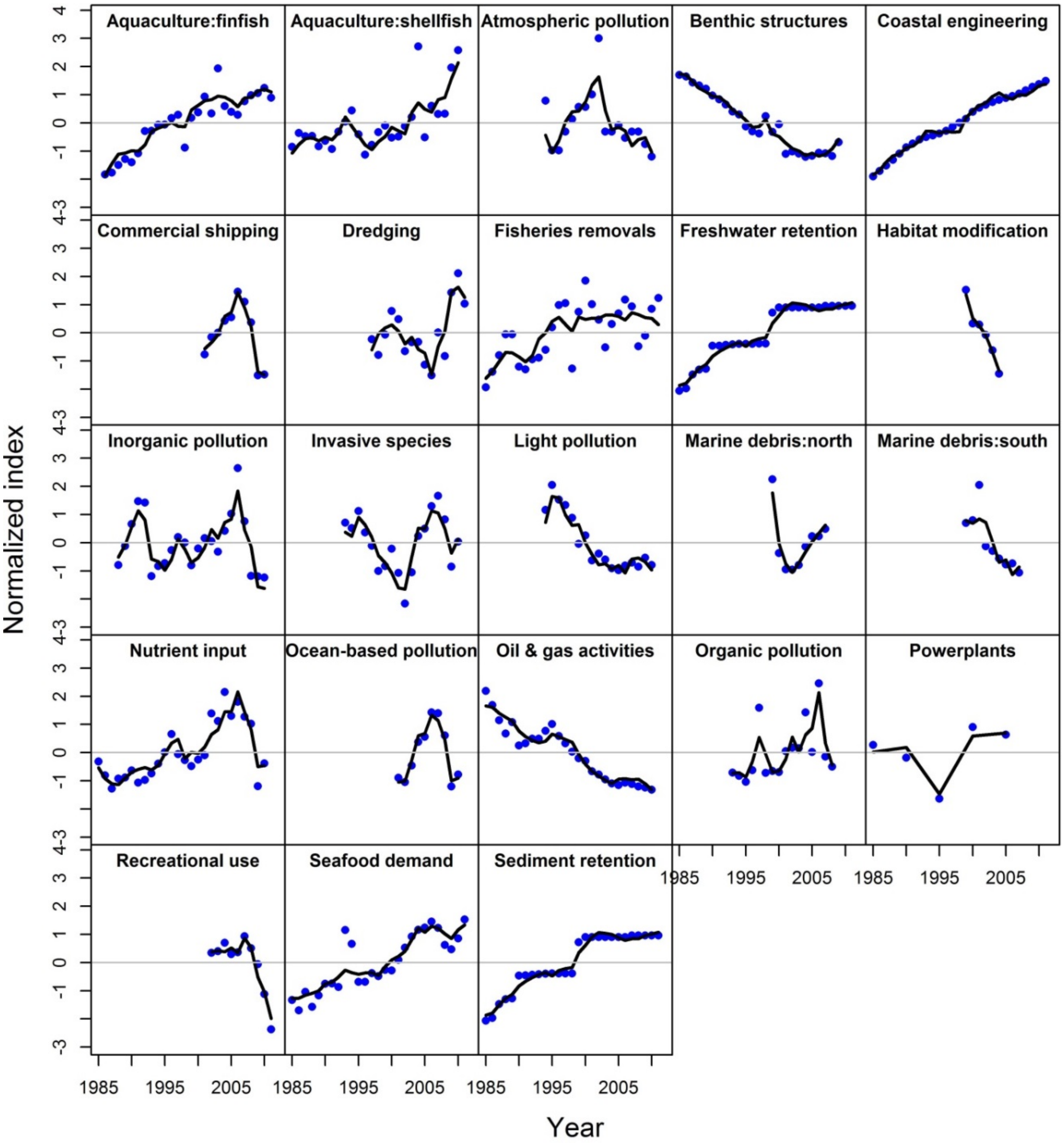
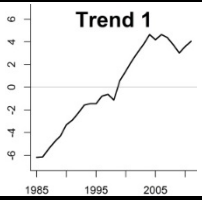
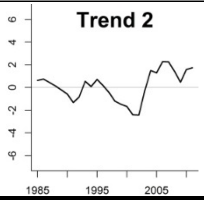
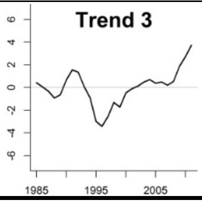
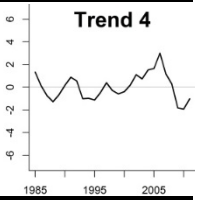


Figure AP1-6. Model fits (black lines) to each pressure time series (blue points) for the dynamic factor analysis model with four common trends, 'diagonal and equal' R matrix and no covariates. Gray line shows the zero-line.

Trend 1 showed a relatively monotonic increase from 1985 to the early 2000's followed by a more variable period during the rest of the 2000's (Table AP1-4). Eight pressures had their highest loadings on this trend and were not related to any other trend. These pressures were related to activities associated with food supply, construction and energy production. Most of these pressures were positively correlated with trend 1, but oil and gas activities and related benthic structures were negatively correlated (Table AP1-4, Fig. AP1-7). Trends 2 – 4 showed a variety of peaks and valleys at various times throughout the period. Six of eight pressures that loaded heavily on trend 2 also loaded heavily on trend 3 or 4 (Table AP1-4), suggesting a fair amount of correlation among these three trends at various time lags. Pressures associated with transportation and coastal disturbance tended to have higher loadings on trend 3, while pressures associated with the input of terrestrial pollutants into the marine environment were generally related to trend 4 (Table AP1-4).

Table AP1-4. Common trends and factor loadings identified from the 4-trend dynamic factor analysis model using 23 pressures and time series data from 1985 to 2011. Bold values indicate which pressures were related to each trend (absolute value of factor loadings >0.2). Boxes indicate which trend was most related to each pressure. Negative loadings mean that a pressure is related to the inverse of the trend shown above each column. Factor loadings are the coefficients that when multiplied by the trend value and summed across all trends produce predicted values for each pressure.

Broad category of pressures	**Pressures**				
Terrestrial pollutants	Atmospheric pollution	0.01	$\boxed{\mathbf{-0.53}}$	0.12	**0.28**
	Inorganic pollution	-0.12	0.01	0.09	$\boxed{\mathbf{0.77}}$
	Organic pollution	-0.19	-0.01	0.00	$\boxed{\mathbf{1.02}}$
	Nutrient input	0.17	0.12	-0.19	$\boxed{\mathbf{0.39}}$
Transportation	Dredging	0.05	-0.03	0.14	$\boxed{\mathbf{-0.58}}$
	Commercial shipping	-0.01	**0.27**	$\boxed{\mathbf{-0.43}}$	**0.36**
	Ocean-based pollution	-0.01	**0.47**	$\boxed{\mathbf{-0.48}}$	0.17
	Invasive species	-0.08	$\boxed{\mathbf{0.60}}$	-0.15	0.07
Coastal disturbance	Marine debris (south)	0.02	$\boxed{\mathbf{-0.34}}$	-0.11	-0.13
	Marine debris (north)	0.00	**0.38**	$\boxed{\mathbf{-1.36}}$	0.04
	Recreational use	**0.26**	0.05	$\boxed{\mathbf{-0.89}}$	-0.18
	Light pollution	-0.10	0.08	$\boxed{\mathbf{-0.41}}$	-0.20
	Habitat modification	-0.09	-0.18	$\boxed{\mathbf{-0.62}}$	-0.14
Food	Fisheries removals	$\boxed{\mathbf{0.22}}$	-0.01	-0.19	-0.14
	Shellfish aquaculture	0.15	**0.22**	**0.25**	$\boxed{\mathbf{-0.31}}$
	Finfish aquaculture	$\boxed{\mathbf{0.29}}$	-0.06	-0.05	-0.20
	Seafood demand	$\boxed{\mathbf{0.22}}$	0.11	0.06	-0.01
Construction	Coastal engineering	$\boxed{\mathbf{0.27}}$	-0.01	0.04	-0.13
	Freshwater retention	$\boxed{\mathbf{0.28}}$	-0.12	0.03	-0.08
	Sediment retention	$\boxed{\mathbf{0.28}}$	-0.12	0.03	-0.08
	Benthic structures	$\boxed{\mathbf{-0.27}}$	0.03	0.11	-0.01
Energy	Oil & gas activities	$\boxed{\mathbf{-0.26}}$	0.04	-0.12	0.07
	Power plant activity	0.08	**-0.45**	0.14	$\boxed{\mathbf{0.54}}$

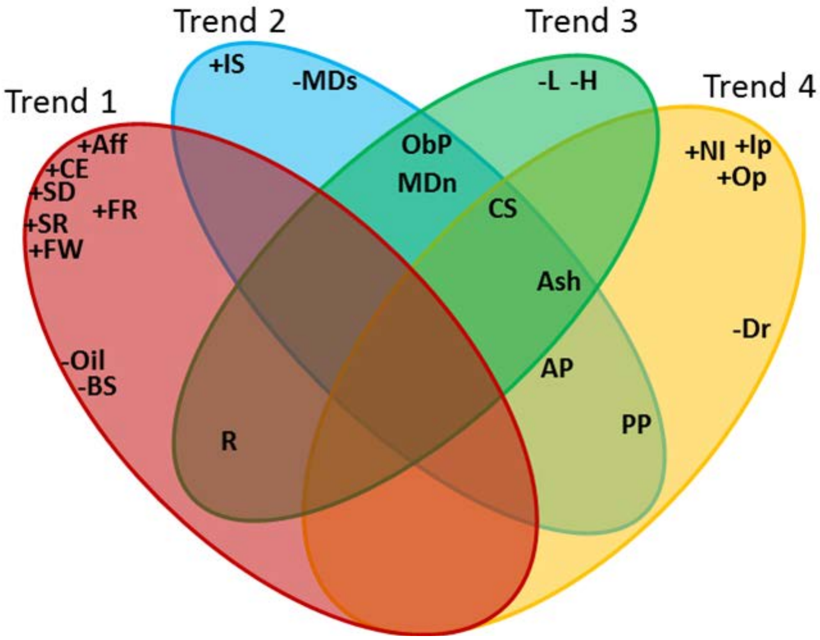


Figure AP1-7. Venn diagram showing factor loadings for each pressure relative to all four trends. Positive (+) or negative (-) loadings are distinguished for pressures that loaded significantly (>2) on only one trend. See Figure AP1-2 for abbreviations.

Because all four trends were estimated simultaneously, we cannot statistically determine which trend was most important; however, some insight can be gained by comparing the results from models with one, two and three common trend(s) with the trends found in the 4-trend model (Zuur et al. 2003a). These comparisons suggested that trend 1 was the most important as it was nearly identical to the trend found in the 1-trend model and other monotonic trends found in the 2- and 3-trend models (Fig. AP1-8).

It is important to note that the strength of the relationship between each pressure and each common trend is a function of the length of each time series. For example, the time series for marine debris in the northern CCE was strongly related to the inverse of trend 3 and less positively related to trend 2 for only a short period of that trend (data for marine debris only available from 1999 to 2007; Tables AP1-1 & AP1-4). In contrast, the time series for seafood demand (data available from 1962 to 2011; Table AP1-1) was related to trend 1 across the entire period of the trend (1985 – 2011; Table AP1-4).

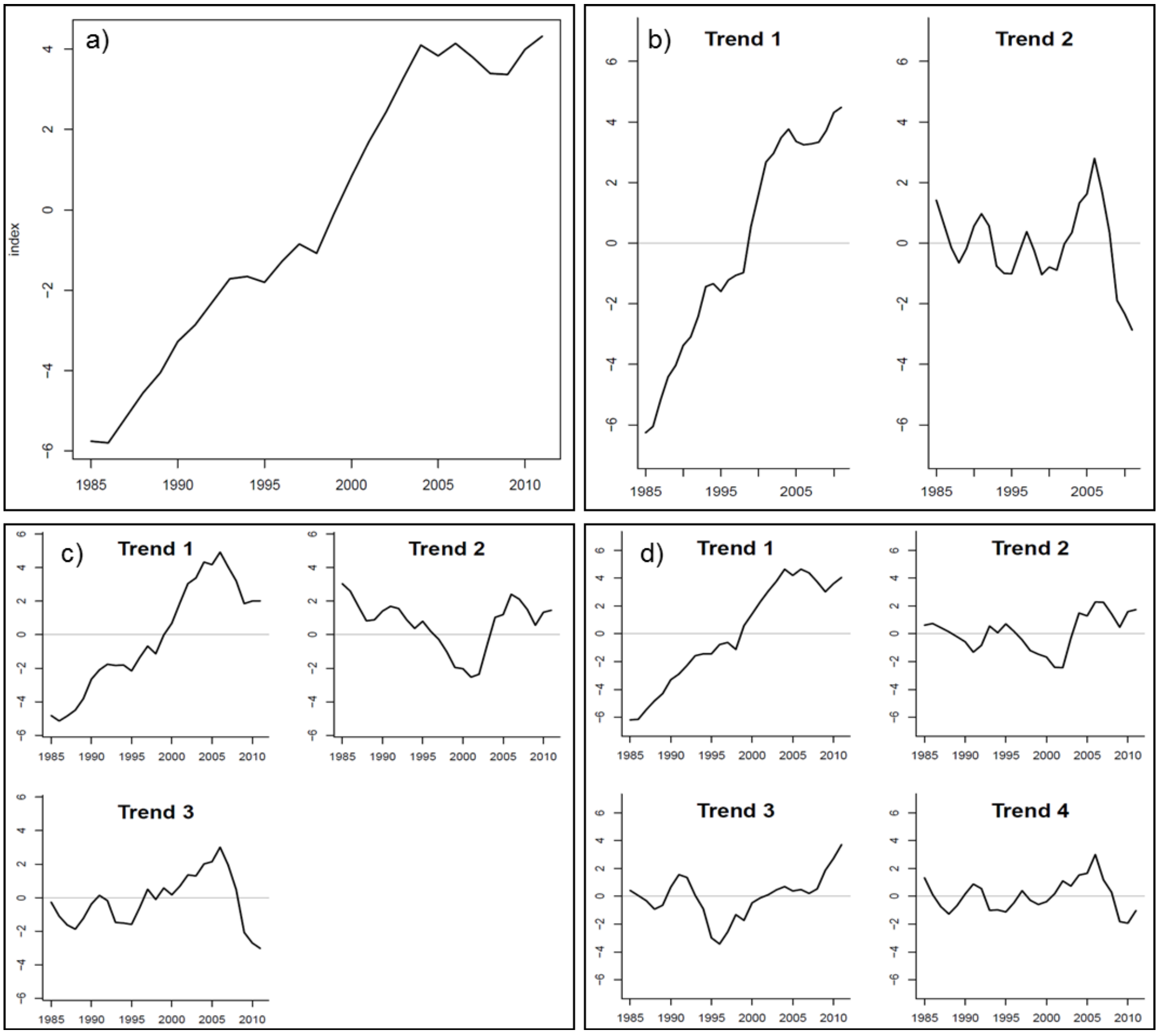


Figure AP1-8. Common trends in dynamic factor analysis models using all 23 anthropogenic pressure indicator time series, 'diagonal and equal' R matrix, no covariates, and a) one, b) two, c) three or d) four common trends. The four common trends model was the best model based on model selection criteria (AICc). Because all trends are estimated simultaneously, we cannot statistically determine which trend is most important; however, it appears that trend 1 explains the greatest amount of variation in this set of time series since it is the trend identified in the 1-trend model and remained relatively unchanged in the 2-, 3- and 4-trend models (Zuur et al. 2003a).

DISCUSSION

One of the central tenets of ecosystem-based management is to address the multiple activities, occurring both on land (e.g., agricultural and industrial practices) and in the ocean (e.g., fishing and energy exploration), that affect various components of marine ecosystems (Leslie and McLeod 2007). Spatial analyses have quantified individual and cumulative pressures across the California Current ecosystem (CCE; Halpern et al. 2009), but prior to this work we are unaware of companion analyses to determine the temporal status and trends of these anthropogenic pressures.

In this study, we evaluated 43 candidate indicators across 22 anthropogenic pressures in the CCE, and developed time series for those that ranked highest for each pressure. Most indicators showed either significant short-term trends or their current status was at historically high or low levels. Taken together, these results support two primary conclusions: 1) decreasing trends of several pressures (e.g., shipping related indicators, industrial pollution and recreational activity) potentially reflect slowing economic conditions during the 'Great Recession' that began around December 2007 (e.g., Grusky et al. 2011), and 2) most pressures at historically high levels have leveled off and are not continuing to increase. Exceptions to these general conclusions are that shellfish aquaculture continues to increase despite being at historically high levels and the time series for seafood demand and dredging suggest these pressures will be increasing at historically high levels if current trends continue over the next few years. In addition to these pressures, relatively new pressures related to wind/wave/tidal energy will need to be incorporated into this framework as activities associated with these technologies will undoubtedly increase over the next decades.

Because each of the pressures we catalogued is associated with one or more human activities, the connotation of their status and trends depends on one's perspective. For example, a decreasing trend in fisheries removals may be "good" for some conservation outcomes, while at the same time, it could be "bad" for human well-being in coastal communities (Levin et al. 2009). Understanding the trade-offs resulting from dynamic changes in these pressures for the social, economic, and biological components of the ecosystem is essential for making informed management decisions (Link 2010, Kaplan and Leonard 2012). The time series we developed here can be used to inform such decisions in the U.S. portion of the CCE, and to populate science-based decision support tools that link biological components of marine ecosystems with human communities and economies.

In addition to quantifying the status and trends of individual pressures, the ultimate goal of this work was to reduce the large number of pressures to a manageable number of trends that could subsequently be used in integrative analyses that investigate linkages between pressures and state variables across the CCE. Our first method, calculated two indices of cumulative pressures across the CCE. Although we did find statistical differences in the status and trends between the additive and weighted models, they provided qualitatively similar results. These results suggest that, at the scale of the U.S. portion of the CCE, either model could be useful for capturing the overall variation in cumulative pressures. The weighted model may be most useful when examining the relationship between cumulative pressures and specific species where the sensitivity of a species to each pressure could be used as weightings. For resource managers interested in the potential impacts of these pressures in specific habitats, habitat-specific vulnerability scores for each pressure identified by Teck et al. (2010) could be used instead of the average vulnerability score across all habitats. The habitat-specific vulnerability scores would be weighted by the proportion of area of each habitat within the region of interest in order to calculate the weighting for each pressure. However, our analysis suggests that if interactions between pressures are not assumed to be synergistic or antagonistic, the qualitative trends will not differ substantially between additive and weighted models.

A clear limitation of any analysis attempting to combine multiple pressures into a cumulative index is the lack of data on the strength and form of interactions between them. Without a clear understanding of the potential synergistic and antagonistic interactions among multiple pressures (Crain et al. 2008, Darling and Côté 2008, Brown et al. 2013), an additive index can be used to describe the cumulative effect of multiple pressures acting on the system (Halpern et al. 2009). However, there is an increasing body of work being performed to more realistically describe the effects of multiple pressures on fish populations as well as on fisheries (Kaplan et al. 2010, Ainsworth et al. 2011, Brown et al. 2013), and there has been an increasing effort to empirically evaluate the strength and direction of interactions among multiple pressures (Lefebvre et al. 2012, Lischka and Riebesell 2012, Sunda and Cai 2012). This research will help better understand cumulative effects of multiple pressures on various species, habitats and ecosystems and reduce uncertainty in quantifying these effects.

We then used two multivariate approaches to reduce the number of pressures into a manageable number of trends. Principal components (PC) analysis is a commonly employed dimension-reducing method that allowed us to reduce a set of 15 pressures down to two principal components that explained 68% of the variation. The analysis showed large changes in the composition of pressures during the period 1994 to 2008. Oil and gas activities, benthic structures and light pollution had significant influence at the beginning of this period, but pressures such as coastal engineering, seafood demand, and nutrient input were more influential in the latter part of the time series. The relative changes among pressures may reflect changes in regulatory actions, business practices, economic activity, technological advances or social norms over this period. The principal component score framework has been suggested as a way to measure the relative status of an ecosystem and to derive specific control rules, analogous to single species management (Link et al. 2002). As the PC score moves around in multidimensional space, managers could determine whether this point falls outside of acceptable conditions (Rockström et al. 2009a, Rockström et al. 2009b, Samhouri et al. 2011, Samhouri et al. 2012). Once this occurs or is approached, pressures that are correlated with the movement outside the acceptable range could be subject to regulatory actions or incentives to reduce these pressures on the marine ecosystem.

However, we caution the use of multivariate analyses as a way to reduce or combine multiple variables when those variables are time series (e.g., Link et al. 2002, Sydeman et al. 2013) for two primary reasons: (1) PC analysis assumes that each year is independent from the year before and after, thus it does not account for autocorrelation that is present in time series data, and (2) PC analysis does not allow for missing data, which can be quite common in time series data, thus reducing the set of time series that can potentially be used. In contrast, dynamic factor analysis (DFA) is an analogous dimension-reducing methodology that explicitly accounts for the nature of time series data and can explicitly account for missing data as well as incorporate the effects of explanatory variables (Zuur et al. 2003b, Holmes et al. 2012).

Using DFA, we were able to include all 23 pressure time series and increase the number of years in the analysis from 15 to 27 compared to the cumulative pressures index and the PC analysis. The DFA reduced the 23 pressure time series to four underlying common trends. Ideally, this analysis would remove the effects of assumed drivers (covariates) and then reveal correlations between each pressure and one common trend. In our analysis, the covariates did not help remove underlying variation, but only 7 of the 23 pressures were related to multiple common trends, making interpretation of the results reasonable. One of the central goals of ecosystem-based management is to identify thresholds and/or reference points of pressures that affect ecosystem state variables (Samhouri et al. 2012, Large et al. 2013). Recent studies have begun to identify thresholds for individual pressures on ecosystem components (Samhouri et al. 2010, Large et al. 2013), but there has been no attempt at identifying thresholds across multiple pressures. Reducing 23 pressure time series to 4 common trends provides a way forward to identify relationships, including thresholds, between pressures and ecosystem components.

It was surprising that coastal population abundance and economic activity did not significantly improve the fit of DFA models to the pressures. However, the trend (trend 1) that appeared to explain the greatest amount of variation across the set of pressures was highly correlated with both covariates (population abundance vs. trend 1: r = 0.98; gdp vs. trend 1: r = 0.95). This result supports the hypothesis that coastal population abundance and gross domestic product were underlying drivers of anthropogenic pressures as a whole in the CCE and that institutional controls (laws and governance), market forces, technological advances and/or cultural norms likely interacted with these drivers at various times during this period to modify the relationship between pressures and drivers. For example, implementation of the Clean Water Act over the years has provided incentives and regulations which reduced the magnitude of certain industrial pollutants (Adler et al. 1993, Houck 2002, Smail et al. 2012), even though it likely reduced profits in the short-term. Similarly, social norms have changed the way some people feel about littering our roadways and waterways (Lee and Kotler 2011, Naquin et al. 2011), thus reducing per-capita littering in some regions even though the amount of waste we produce has continued to increase over time (USEPA 2011, Brogle 2012). At some point, we expect our governing institutions or social awareness to modify the effects of pressures ultimately caused by increases in the number of humans on the planet.

CONCLUSIONS

Despite the uncertainties about the strength and direction of interactions among pressures, it is useful to understand how the magnitudes of multiple pressures are changing over time. The presence of common trends among pressures can help reduce the number of variables included in ecosystem assessments and may help identify common drivers for multiple pressures. Incorporating numerous anthropogenic pressures into the framework of ecosystem-based management is necessary to understand linkages between these pressures and various biological components, and more importantly, will allow us to identify thresholds (Samhouri et al. 2010, Large et al. 2013) and consider trade-offs among socioeconomic, cultural and biological components of the ecosystem (Rosenberg and McLeod 2005, Link 2010). Combining spatial and temporal patterns of anthropogenic pressures will provide a better understanding of how pressures are changing over time and space and allow managers to make better use of limited funding and resources. Moreover, these anthropogenic pressures interact with the underlying oceanographic conditions and climate change. Recently developed “end-to-end” ecosystem models (e.g., Atlantis; Fulton et al. 2011) and coupled ecological/economic models (Kaplan and Leonard 2012) allow examination of the effects and interactions of anthropogenic, oceanographic and climatic pressures on multiple ecological components and human communities. Now, marine ecologists, fisheries scientists, and social scientists need to develop creative methods to test the validity of these models’ results in the field in order to increase resource managers’ and stakeholders’ confidence in their use as part of the decision-making process.

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The variance-covariance matrix (R matrix) in the DFA describes the observation error structure of the set of time series. In the MARSS package (Holmes et al. 2012), there are five common R matrix structures built-in: identity, diagonal and equal, equal variance-covariance, diagonal and unequal, and unconstrained. The simplest is 'identity' which is an identity matrix in which the response variables (each time series) all have variance of 1 and are uncorrelated:

$$R = \begin{bmatrix} 1 & 0 & 0 \\ 0 & 1 & 0 \\ 0 & 0 & 1 \end{bmatrix}$$

'Diagonal and equal' is a diagonal R matrix in which the response variables all have the same variance and are uncorrelated:

$$R = \begin{bmatrix} \sigma^2 & 0 & 0 \\ 0 & \sigma^2 & 0 \\ 0 & 0 & \sigma^2 \end{bmatrix}$$

'Equal variance-covariance' is a diagonal R matrix in which the response variables all have the same variance and are correlated with the same covariance:

$$R = \begin{bmatrix} \sigma^2 & \beta & \beta \\ \beta & \sigma^2 & \beta \\ \beta & \beta & \sigma^2 \end{bmatrix}$$

'Diagonal and unequal' is a diagonal R matrix in which the response variables have unique variances and are uncorrelated:

$$R = \begin{bmatrix} \sigma_1^2 & 0 & 0 \\ 0 & \sigma_2^2 & 0 \\ 0 & 0 & \sigma_3^2 \end{bmatrix}$$

'Unconstrained' is a non-diagonal R matrix in which there are unique variance and covariance values for each response variable:

$$R = \begin{bmatrix} \sigma_1^2 & \sigma_{1,2} & \sigma_{1,3} \\ \sigma_{1,2} & \sigma_2^2 & \sigma_{1,2} \\ \sigma_{1,3} & \sigma_{2,3} & \sigma_3^2 \end{bmatrix}$$

We tested the appropriateness of each R matrix structure to determine which best explained our set of time series. The indicator time series for anthropogenic pressures consist of data measured and sampled using numerous methods across various scales of time and space. Some of these indicators take advantage of similar data sets and may be correlated. Thus, our expectation was that the 'unconstrained' R matrix would be most appropriate. However, the 'unconstrained' structure caused the solution to become unstable and parameters were not identifiable in all models. We attempted to limit the dataset by removing time series that did not resemble a random-walk (e.g., freshwater retention, coastal engineering), but even the model with no covariates and 1 trend became unstable and provided no solution. It is likely that we did not have enough data in several of the time series to estimate the large number of parameters in this type of unconstrained model. Due to these limitations, we removed 'unconstrained' from the analysis.

Models using the 'diagonal and unequal' R matrix suffered from similar issues. Models with 2 or fewer trends with and without covariates could be solved when we limited the dataset by removing time series that did not resemble a random walk, but models with > 2 trends became unstable as estimates of variance for various pressures became negative. We attempted to solve this problem by fixing the variance of pressures that went negative to very small values (0.00001), but subsequently the variance of other pressures went negative, the models became unstable and crashed. Due to these complications, we removed 'diagonal and unequal' from the analysis also.

The final set of models tested and presented in the main text of the manuscript compared the remaining three R matrix structures ('identity', 'diagonal and equal', and 'equalvarcov'). It is plausible that the more complex 'unconstrained' or 'diagonal and unequal' R matrix structures would be most appropriate for an analysis of common trends among time series that no doubt vary dramatically in observation and measurement error. However, for various reasons (perhaps lack of data to estimate the large number of parameters) these time series could not be fit to a full set of models (using 1-5 trends) using these error structures, so we used simpler error structures to determine the best model in our final results.

Of the 'diagonal and unequal' models that ran (1-2 trends) using a subset of pressures (removed freshwater and sediment retention and coastal engineering), the best model was 2 trends with population as a significant covariate. This model produced a solution with common trends (Fig. AP1-S1) that were similar to the common trends we found in the best 'diagonal and equal' model (4 trends with no covariates; Table AP1-4). Thus, we feel that limited data in some of the indicator time series may have precluded the use of the more complex R matrix structures, but it did not change the ultimate results we found using the less complex R matrix structure ('diagonal and equal').

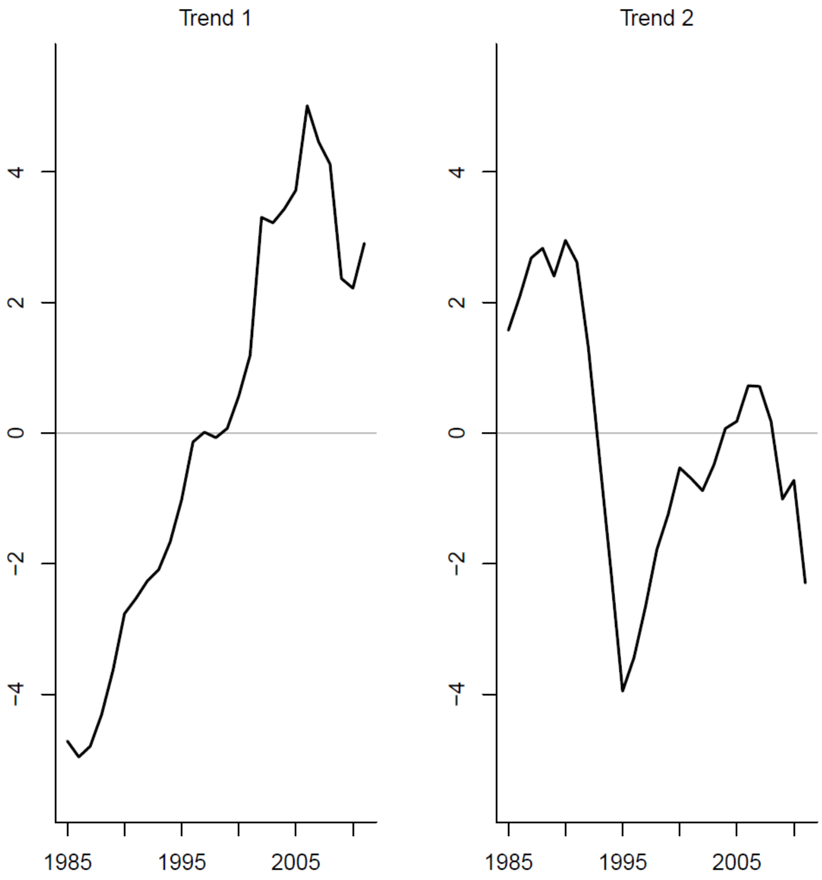

Figure AP1-S1. Common trends identified from dynamic factor analysis using 20 pressures (removed freshwater and sediment retention and coastal engineering) and time series data from 1985 to 2011.

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OCEANOGRAPHIC AND CLIMATIC DRIVERS AND PRESSURES

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SOUND BITE

The past year (2013) was characterized by record cumulative upwelling in the central CCLME leading to below average SSTs and localized regions of high Chlorophyll-a biomass, although the ecosystem implications of these anomalies remain unclear. Recent data from the equatorial Pacific suggest that an El Niño will affect the CCLME in the coming year.

EXECUTIVE SUMMARY

The California Current Large Marine Ecosystem (CCLME) is primarily driven by bottom-up physical processes; thus trends in the physical and biogeochemical state can inform the management of ecosystem services. The Pacific Decadal Oscillation (PDO) and the North Pacific Gyre Oscillation (NPGO), respectively indicators of sea surface temperature and changes in source water, continue to describe a cool phase that has largely persisted since 1999 in the CCLME (Figure E1). From late 2009 to early 2010, a short duration El Niño with stronger than average downwelling-favorable winds was observed and was quickly followed by La Niña conditions in the summer of 2010. Currently, an El Niño event is developing in the equatorial Pacific that will likely have ecosystem implications in the coming year. From 2009 to 2013, the CCLME has been characterized by periods of strong and persistent upwelling from central California to Oregon (Figure E2). In 2013, an early onset and long upwelling season led to the highest cumulative annual upwelling index (TUMI) on record, resulting in negative SST anomalies and a few localized areas of increased chlorophyll-a (chl-a), overlaid on a background of generally decreased chl-a (Figure E2). The full ecosystem implications of these upwelling anomalies are as yet unclear; although most of the physical indicators suggest that primary productivity should have been high during 2013, chl-a biomass was below average most areas. There may be an optimal window in both the timing and magnitude of upwelling that maximizes ecosystem productivity, and the extremely high upwelling during 2013 may have actually led to increased offshore transport and loss of plankton to coastal food webs. The trend of decreasing oxygen content (DO) continues to suggest increased habitat compression for pelagic species and more severe hypoxic events on the shelf that can lead to physiological stress or large scale die-offs.

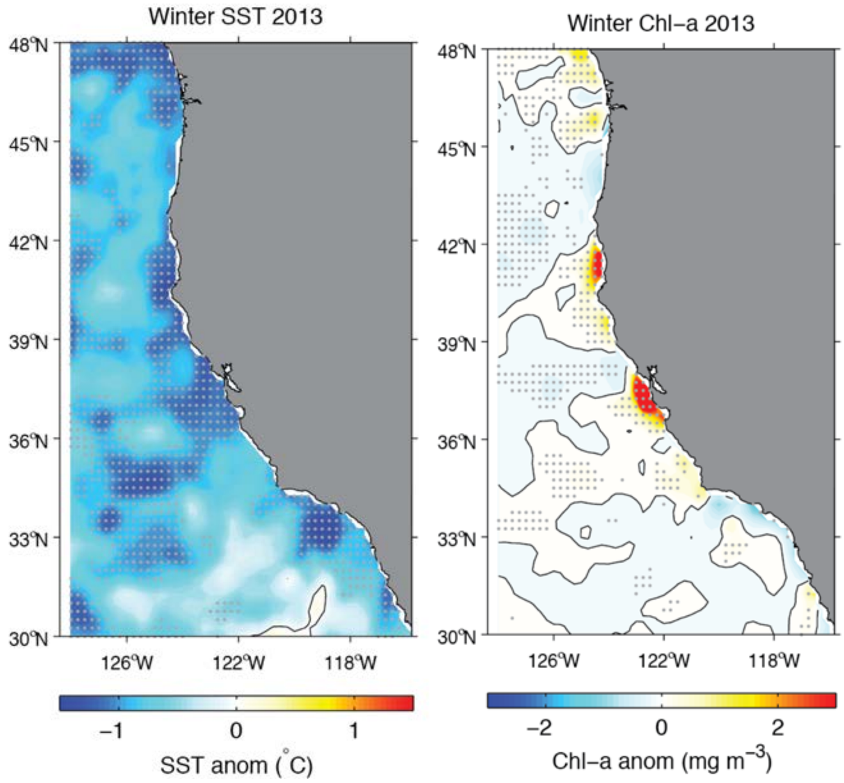


Figure E1. Winter (Jan-Mar) 2013 anomaly maps (deviation from mean over 2003-2013) of sea surface temperature (SST; left) and surface chlorophyll-a (chl-a; right). Points above or below 1 standard deviation are marked in grey.

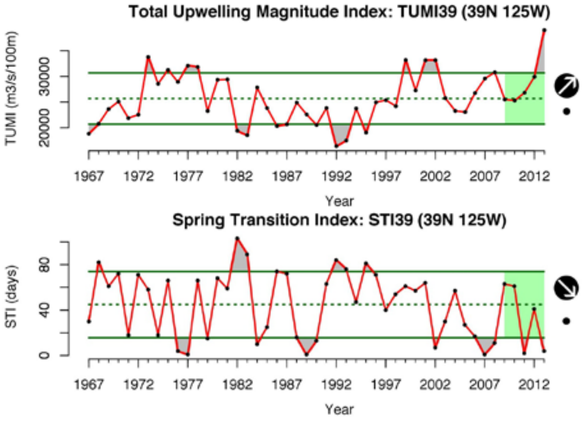


Figure E2. The Total Upwelling Magnitude Index (TUMI; top) and Spring Transition Index (STI, bottom) at 39°N.

DETAILED REPORT

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

Indicators are proxies that serve as measures of either drivers or pressures. Indicators were developed by first identifying a suite of drivers and pressures that were most closely associated with impacts and changes to the different EBM components in the California Current IEA. We used several publications (Halpern et al. 2008, Sydeman and Elliott 2008, Halpern et al. 2009, Sydeman and Thompson 2010, Teck et al. 2010, Peterson et al. 2012) to develop potential pressures on the CCLME. During reviews of the literature, we identified 32 primary groups of pressures on the CCLME, and these were categorized as “oceanographic and climatic” or “anthropogenic.” Indicators for each of these pressures were then evaluated using the indicator selection framework developed by Levin et al. (2011) and Kershner et al. (2011) and used in the previous version of NOAA’s Integrated Ecosystem Assessment for the California Current (Levin and Schwing 2011). The second step was to develop time-series for each of the top indicators for each pressure. These time-series were used to determine the current status, short-term trends, and five-year anomalies for each pressure in the CCLME. In this IEA, we use the same indicators as the 2013 CCIEA, along with a new additional multivariate ocean climate index (MOCI)(Sydeman et al. 2014), and spatially explicit satellite remote sensed sea surface temperature (SST) and chlorophyll-a (chl-a). **Changes in the current IEA versus previous reports are highlighted by BOLD**, *with new interpretations in italics*.

OCEANOGRAPHIC AND CLIMATIC DRIVERS AND PRESSURES

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

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

There is a close mechanistic link between coastal upwelling and ecosystem productivity on seasonal, annual, and interannual scales (Hickey 1979, Checkley and Barth 2009). Upwelling in the central-northern CCLME occurs in two distinct seasonal modes (winter and summer), with certain biological processes being more sensitive to one or the other (Black et al. 2011, Thompson et al. 2012). Thus in this section we present indicators when there is monthly data as winter and summer means. Summer means were calculated from June 1st - August 31st and winter means were calculated from January 1st - March 31st of the current year; thus *winter precedes summer for each index*. Indicator selection followed the IEA framework and identified datasets with the most relevance to the pressure, and had the longest and most complete time series. Indicator evaluation, data indices and sources are summarized in Table OC1.

IMPLICATIONS OF CLIMATE DRIVERS FOR COASTAL AND MARINE SPATIAL PLANNING

There are regional differences within the CCLME in climate forcing (Mendelssohn et al. 2003, García-Reyes and Largier 2012) and ecosystem response (Checkley and Barth 2009). Therefore, patterns in the southern California Current region may vary substantially from patterns in the northern California Current. When considering an overall IEA for the CCLME, it may prove most useful to separately evaluate each ecoregion/subecosystem initially. In no single region, however, are all the desired physical and biological attributes available for comprehensive analyses. Therefore, to understand ecosystem form, function, and controls, to the extent possible we must combine information between regions. We have examined three primary regions in the CCLME using cruise data such as CALCOFI (California Cooperative Oceanic and Fisheries Investigations) off southern and central California and the Newport line off Oregon. More holistic data are provided by buoy data (National Buoy Data Center), and satellite products.

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

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

CHANGE IN SEA LEVEL

BACKGROUND

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

Coastal sea level is used as a proxy for nearshore surface current strength and direction. In the winter, sea levels are high due to the poleward flowing counter current (Davidson Current). With the onset of upwelling winds in the spring, sea levels lower and the current is directed equatorward; the equatorward flow is dominant in the spring and summer (King et al. 2011).

EVALUATION AND SELECTION OF INDICATORS

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

STATUS AND TRENDS

Since 1950, there have been increasing sea level trends, particularly until 1977 with more numerous and extreme positive anomalies (Figures OC1 – OC3). Over the past five summers, the San Diego station values have been greater than one standard deviation from the mean although there were no significant short-term trends throughout the California Current. Coastal sea level trends have been somewhat muted since 1980 due to wind changes and the Pacific Decadal Oscillation (PDO) masking any upper-ocean temperature steric effect (Bromirski et al. 2012). The summer records show a long-term trend of sea level rising at about 2 mm yr^{-1}. This trend isn't evident in either the monthly or winter plots. **The greatest difference between this year's and last year's status, is that sea level in winter 2012/2013 at South Beach OR was much lower than the immediate previous years**, however, the following summertime sea level at this location was similar to previous years. *This observation is most likely due to an earlier onset of upwelling at this location*, as evidenced by an earlier STI during 2013 (Fig OC21).

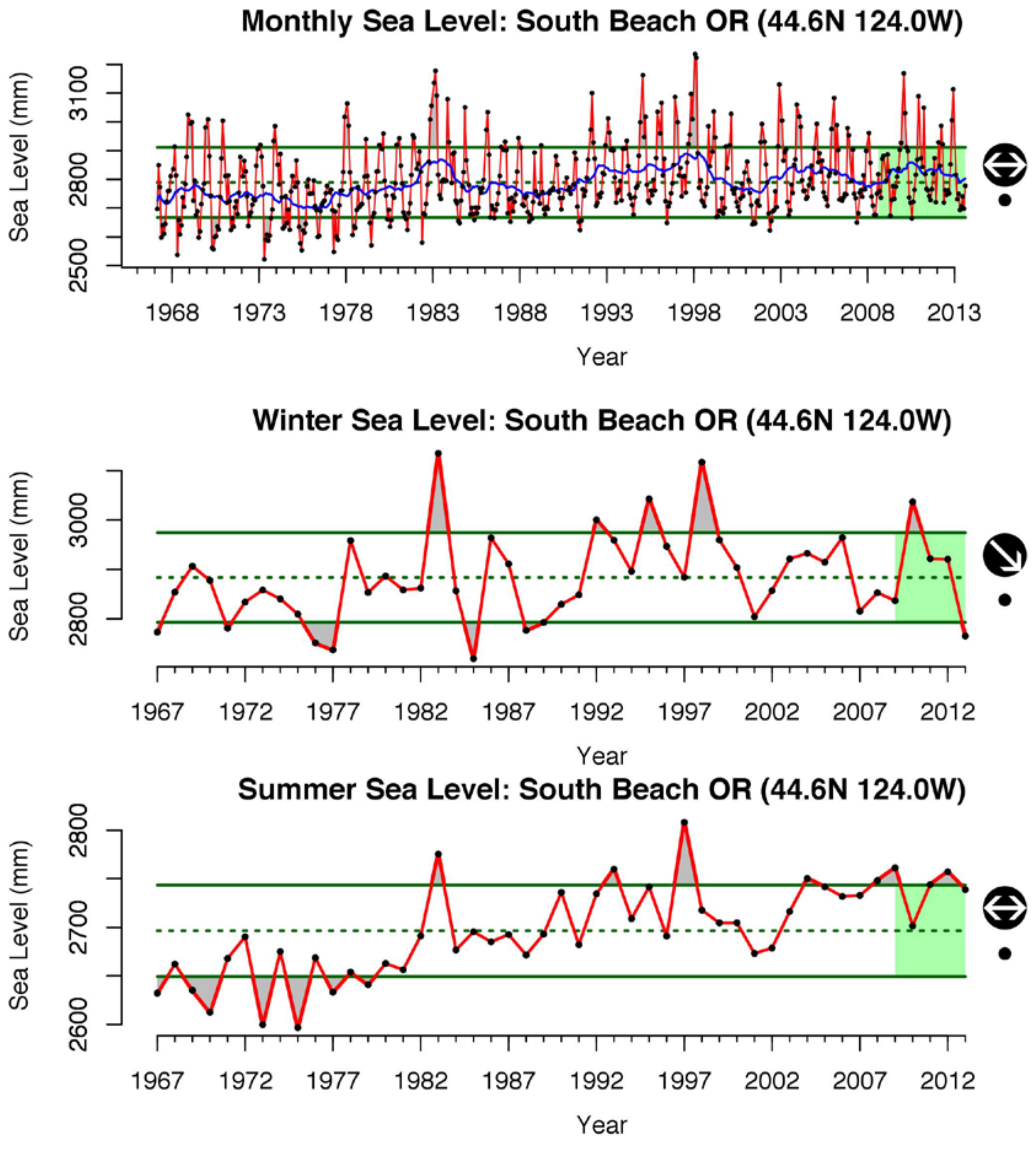


Figure OC 1. Coastal sea level heights from 1967-2013 for monthly, winter, and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. South Beach, Oregon coastal sea level illustrates patterns in the northern portion of the CCLME.

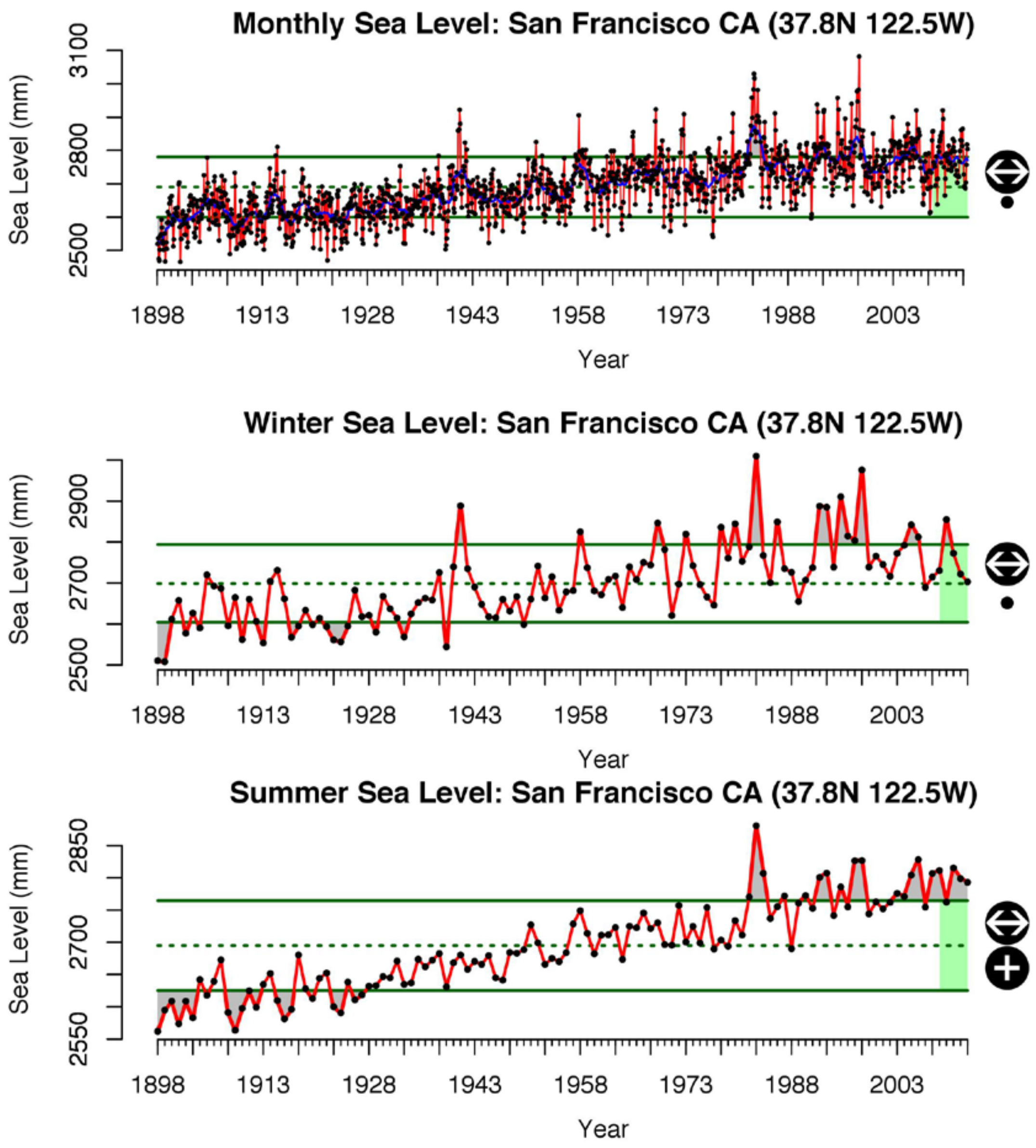


Figure OC 2. Coastal sea level heights from 1898-2013 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. San Francisco coastal sea level illustrates patterns in the central portion of the CCLME.

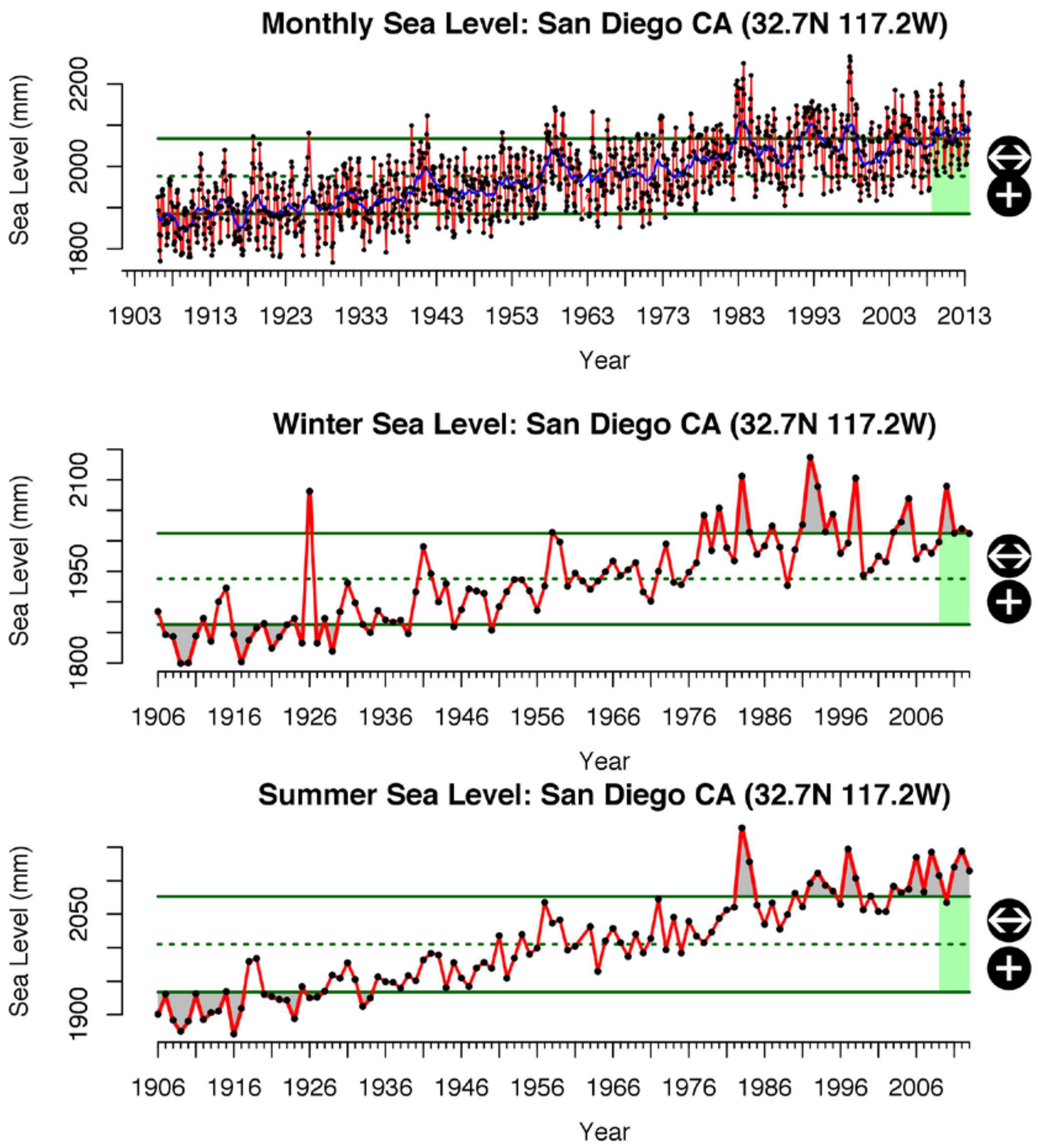


Figure OC 3. Coastal sea level heights from 1906-2013 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. San Diego coastal sea level illustrates patterns in the southern portion of the CCLME.

CHANGE IN SEA SURFACE TEMPERATURE

BACKGROUND

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

EVALUATION AND SELECTION OF INDICATORS

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

STATUS AND TRENDS

SST

Cold sea surface temperature (SST) from upwelled water often results in high productivity but nutrient content depends upon remotely forced state of the ocean, which can be indicated by large-scale climate indices (North Pacific gyre Oscillation (NPGO), PDO, MEI, and NOI). Negative NPGO, positive PDO, and positive MEI would act in concert to create an extremely warm, low-productivity regime in the CCLME. According to many long-term data sets from the open ocean, SSTs have increased by 0.5°C to 1.0°C over the past 50 years (IPCC 2007, Levitus et al. 2009). SST from three NOAA National Data Buoy Center (NDBC) buoys showed highs in 1983 and 1998 corresponding with increased MEI values (Figures OC4 – OC6). Most SST values in the past 5 years were lower than the 20-year mean average at all stations and for both seasons. The exceptions were warmer than average SSTs in winter 2010 at all three stations which had high SSTs due to the short duration El Niño, and the summer of 2011 in central California as well. **For 2013, SST values remained as cold or colder than the immediately previous years (except the winter SST for the northernmost station),** *reinforcing the evidence that 2013 had an early onset, and stronger overall upwelling than 2012*.

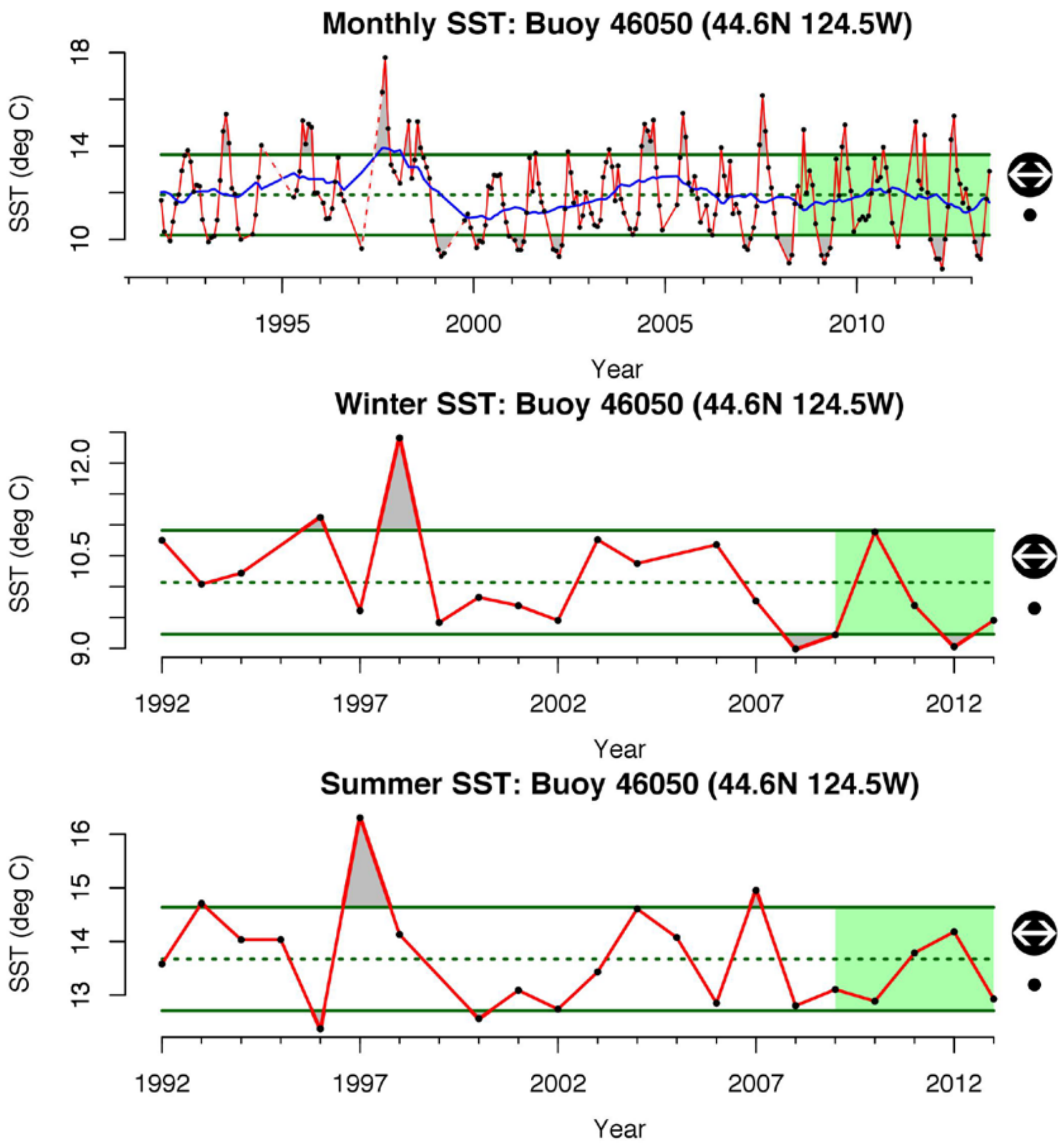


Figure OC 4. Sea surface temperature (SST) buoy data from early 1990 -2013 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Buoy 46050 illustrates patterns in the northern portion of the CCLME.

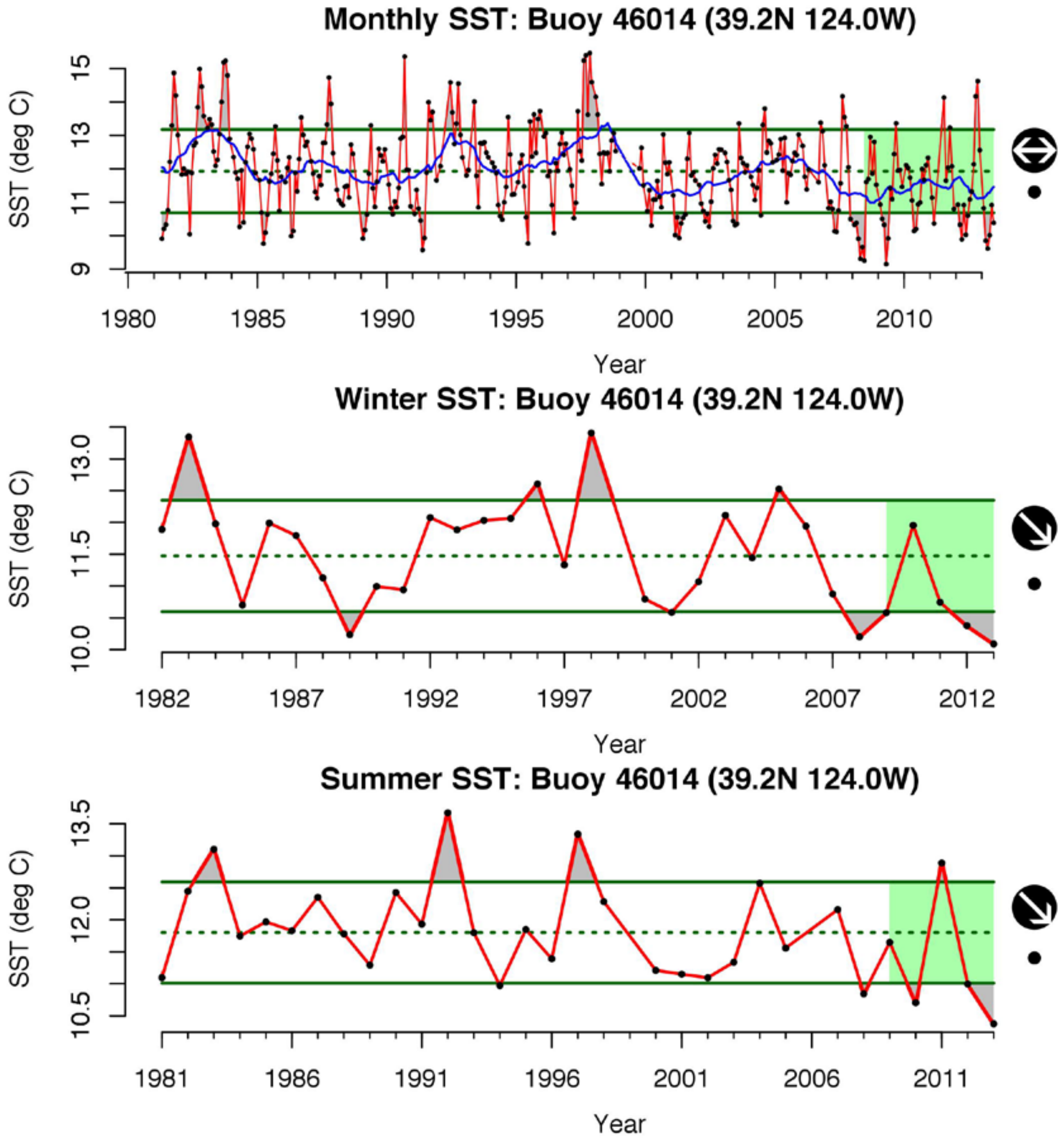


Figure OC 5. Sea surface temperature (SST) buoy data from early 1990 -2013 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Buoy 46014 illustrates patterns in the central portion of the CCLME.

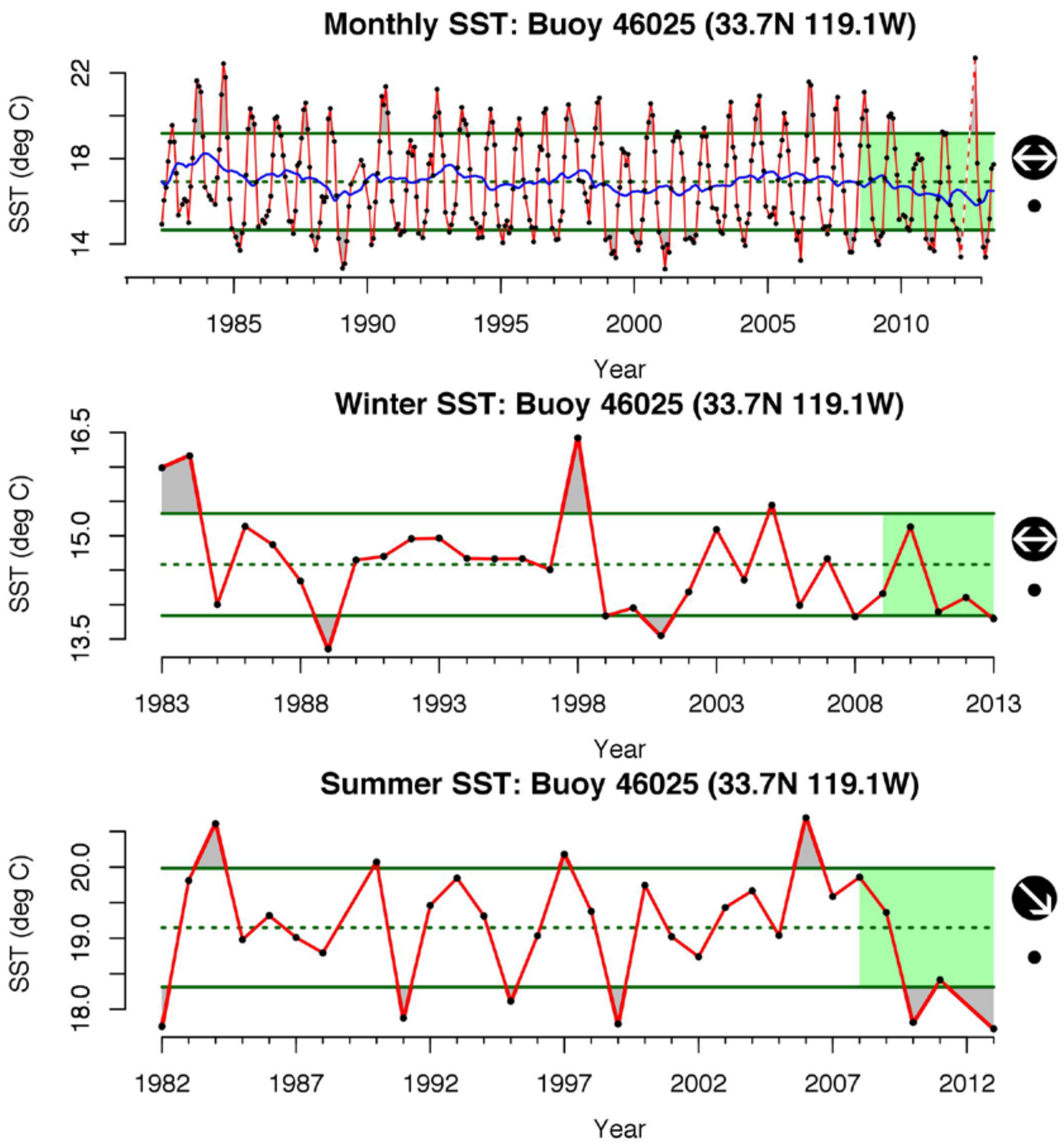


Figure OC 6. Sea surface temperature (SST) buoy data from 1990-2013 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Buoy 46025 illustrates patterns in the southern portion of the CCLME.

PDO

Pacific decadal oscillation (PDO) is a low frequency signal in North Pacific sea surface temperatures that affects biological productivity in the Northeast Pacific (Mantua et al. 1997). Cold (negative values of the PDO) periods are associated with enhanced productivity in the CCLME and vice versa (King et al. 2011). The PDO index has been largely in a positive (i.e., warm California Current and Northeast Pacific) state since late 1977, resulting in warmer waters along the coast of the CCLME with a negative phase since 1998 with occasional warm episodes from El Niños (Figure

OC7). Over the past 5 years, the winter and summer indices have remained relatively low, except a higher value in 2010 for winter. **Values for 2013 have remained low, similar to the immediately previous years,** *supporting the observations that 2013 was a strong upwelling year*.

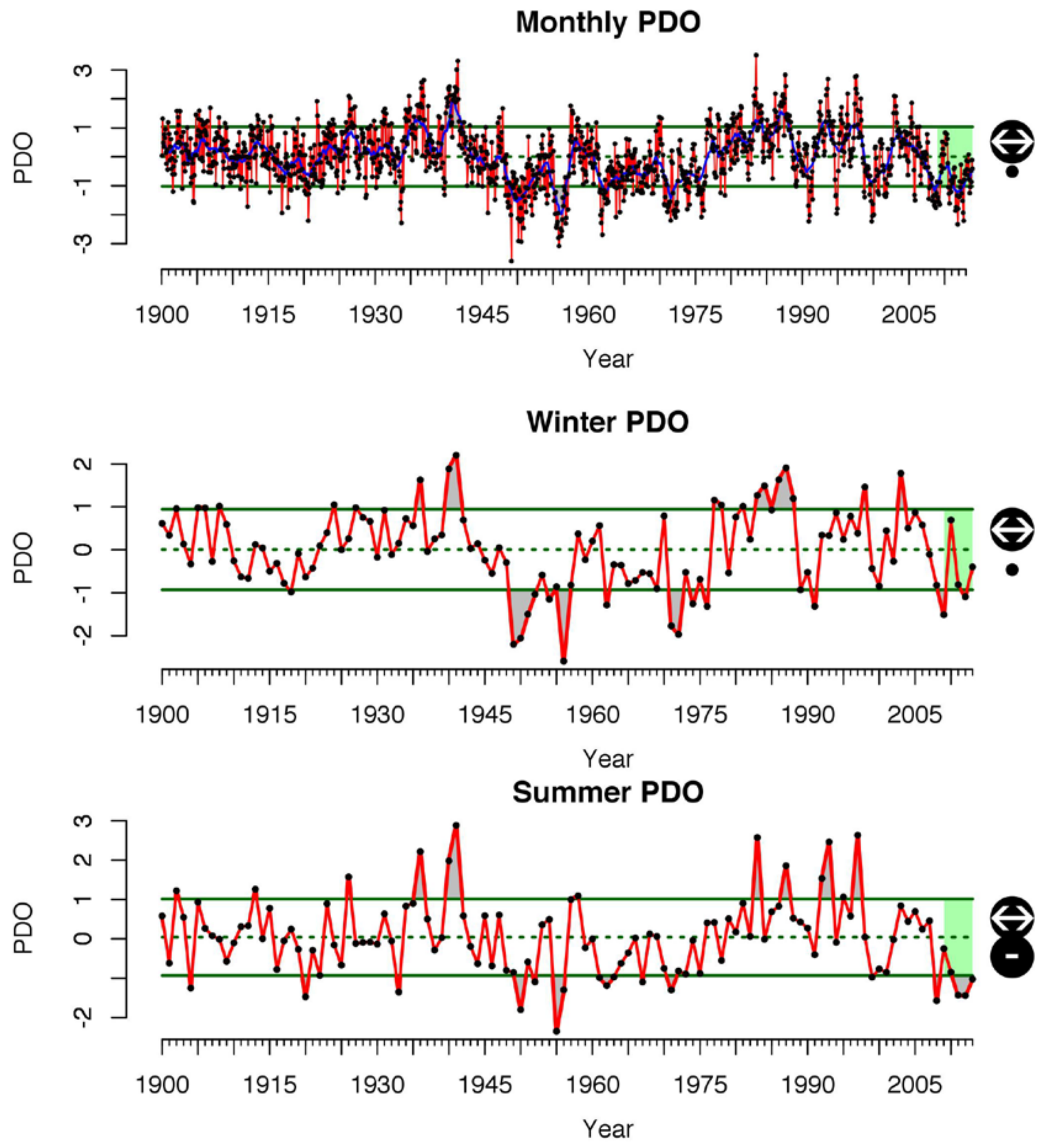


Figure OC 7. Pacific Decadal Oscillation (PDO) index values from 1900-2013 for monthly, winter and summer means. For the monthly PDO the blue line shows a running annual average.

NOI

Northern oscillation index (NOI) is the sea level pressure difference between the climatological mean position of the North Pacific High and Darwin, Australia (Schwing et al. 2002).

NOI describes the strength of atmospheric forcing between the equatorial Pacific and the North Pacific, particularly in terms with ENSO. Positive values of the NOI are related to a more intense North Pacific High and stronger north winds over the CCS, and stronger northeasterly trade winds in the subtropics resulting in cooler waters. NOI was largely positive from 1950 to 1977, and then switched to more negative values until 1998 (Figure OC8). In the winter, NOI values were positive from 2006 to 2009 with a drop and overall negative trend in 2010 representing the brief El Niño event. In the summer of 2010, NOI values became strongly positive which should result in increased coastal upwelling in the California Current, and have since returned to near neutral values. **In 2013, the NOI remained similar to the previous years, with a slight increase for the winter average;** *such atmospheric forcing is consistent with the early setup of upwelling and lower SST seen in 2013***.**

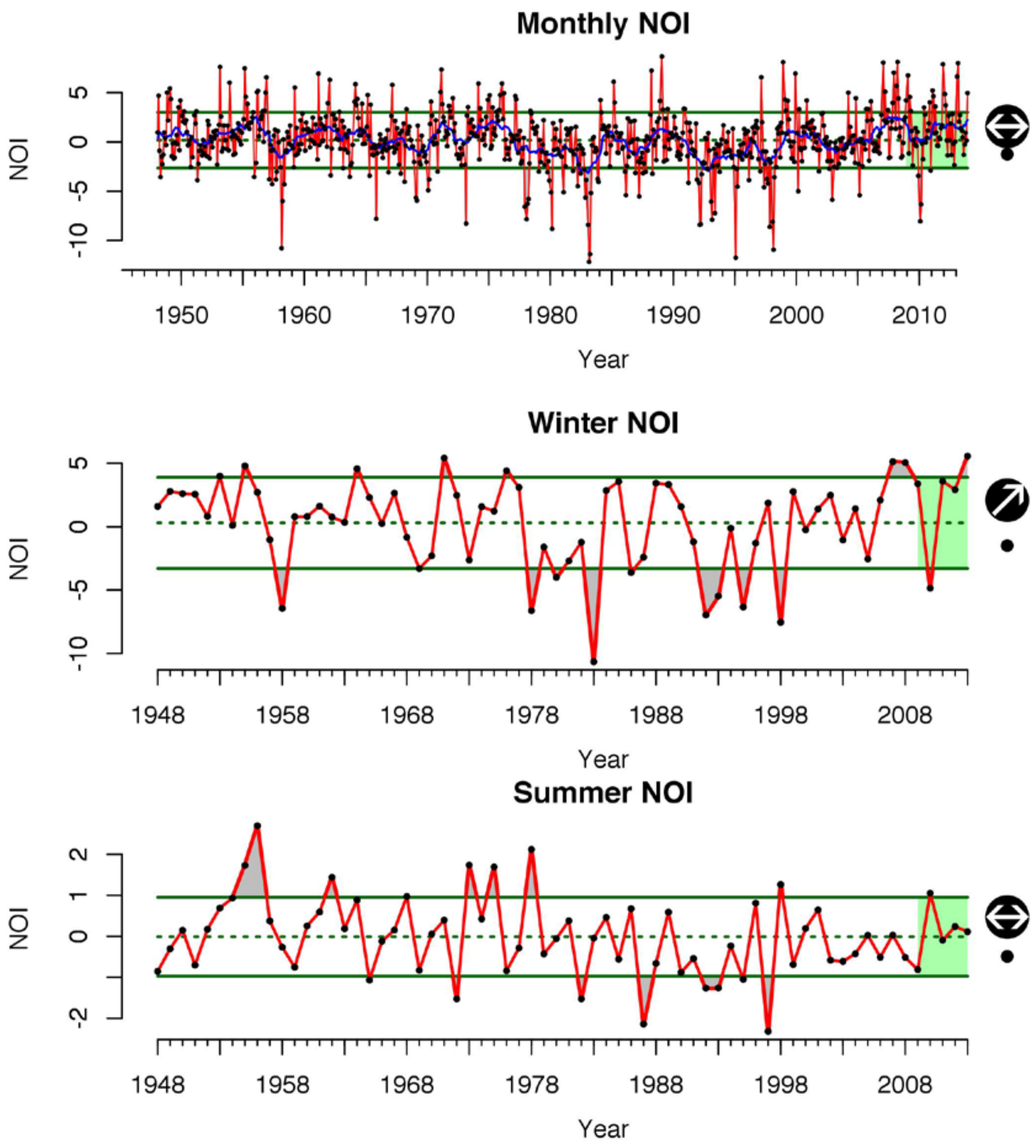


Figure OC 8. Northern Oscillation Index (NOI) values from 1948-2013 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average.

MEI

See Timing and Frequency of El Niño events.

WATER COLUMN STRUCTURE

BACKGROUND

The ocean is vertically stratified into large horizontal layers of water with different properties, such as nutrients, oxygen, temperature, salinity and density. For the water column structure attribute, we will focus on stratification due to density differences. Layers of more dense water lie below less dense water and the boundary between the layers acts as a barrier to mixing. Mixing between layers is easier when the density difference (e.g. the strength of stratification) between the layers is small. The formation of the layers is due to several different geo-physical processes, which act on different spatial and temporal timescales. For example, any physical processes that can change the water density, such as wind mixing of adjacent layers, fresh water inputs and atmospheric thermal heating/cooling, will affect water column stratification. The effectiveness of upwelling winds in the CCLME can be reduced if the water column is highly stratified thus limiting the injection of nutrients from deep water into the surface euphotic zone (Palacios et al. 2004, Behrenfeld et al. 2006). In this report we will characterize the water column structure by quantifying information of the upper surface water mass. Two variables of interest are the mixed layer depth (pycnocline depth) and the strength of the stratification (the gradient between the density of the surface layer and the adjacent lower layer). Buoyancy frequency, or Brunt–Väisälä frequency, can be used to define water column density stratification. The buoyancy frequency is proportional to vertical changes in density; the largest buoyancy frequency will mark the pycnocline (Pond and Pickard 1983). Upwelling can be constrained if the pycnocline depth is deep and the strength of stratification is strong.

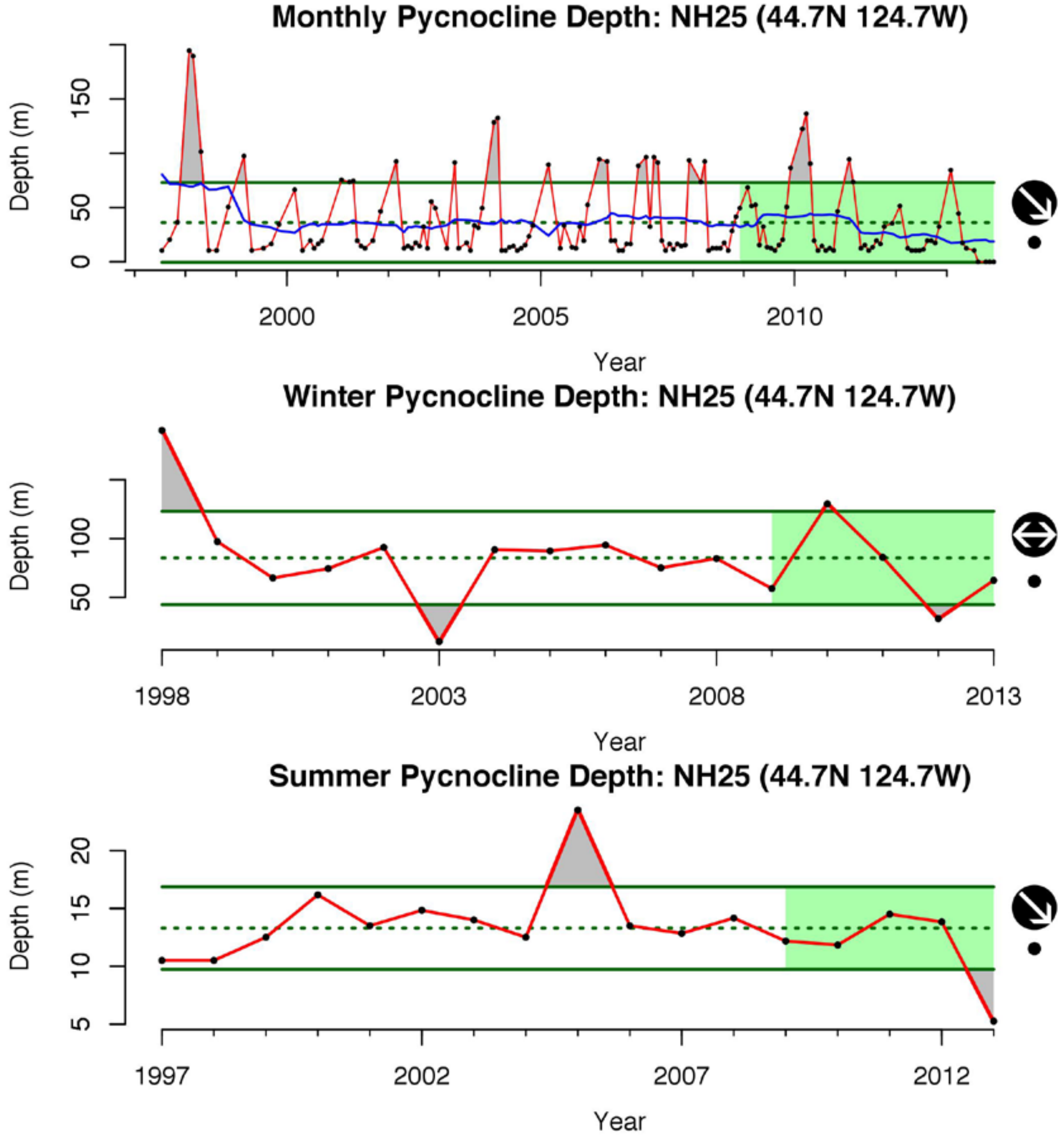
EVALUATION AND SELECTION OF INDICATORS

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

STATUS AND TRENDS

PYCNOCLINE DEPTH

Pycnocline depth, the depth at which there is the greatest rate of change in density in the vertical water column, represents the separation between warmer nutrient-poor surface waters and cooler nutrient-rich deep waters. The shallower the pycnocline, the more nutrients are available to the photic zone. From 2007-2011, pycnocline depth decreased steadily at station 67.55 in central California for both summer and winter (Figure OC10). In southern California (station 93.30), pycnocline depth is highly variable with no clear trend over this period (Figure OC11). In the northern California current (station NH25), the pycnocline has become deeper in the winter but has no clear trend in the summer (Figure OC9). Note that at particular stations, either samples were not taken or data has yet to be processed at the time of writing of this report, thus limiting our ability to comment on the most recent trends in this indicator. **For 2013, where there are data, wintertime pycnocline depth remained similar to immediately previous years (NH 25 and CalCOFI 67.55), but markedly decreased during the summer at NH 25 (the only location with summer 2013 data).** *The shallower pycnocline depth during the summer at NH 25 supports the evidence for strong upwelling and likely enhanced productivity during summer 2013.*


Monthly Pycnocline Depth: NH25 (44.7N 124.7W)

Winter Pycnocline Depth: NH25 (44.7N 124.7W)

Summer Pycnocline Depth: NH25 (44.7N 124.7W)

Figure OC 9. Pycnocline depth data from 1998-2013 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Newport line station NH25, illustrates patterns in the northern portion of the CCLME.

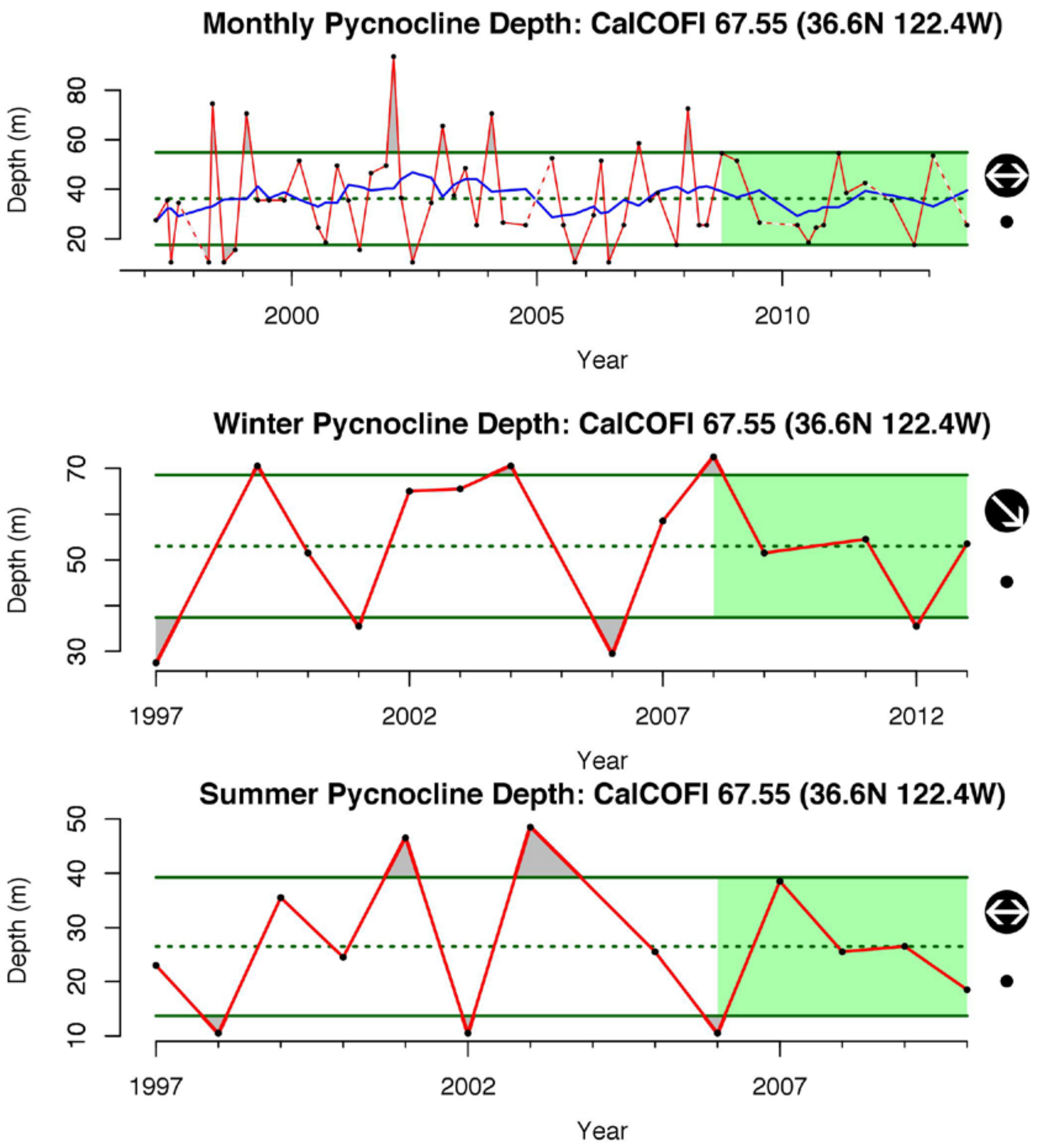


Figure OC 10. Pycnocline depth data from 1998-2013 (where available) and for monthly, winter and summer. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Station 67.55 illustrates patterns in the central portion of the CCLME.

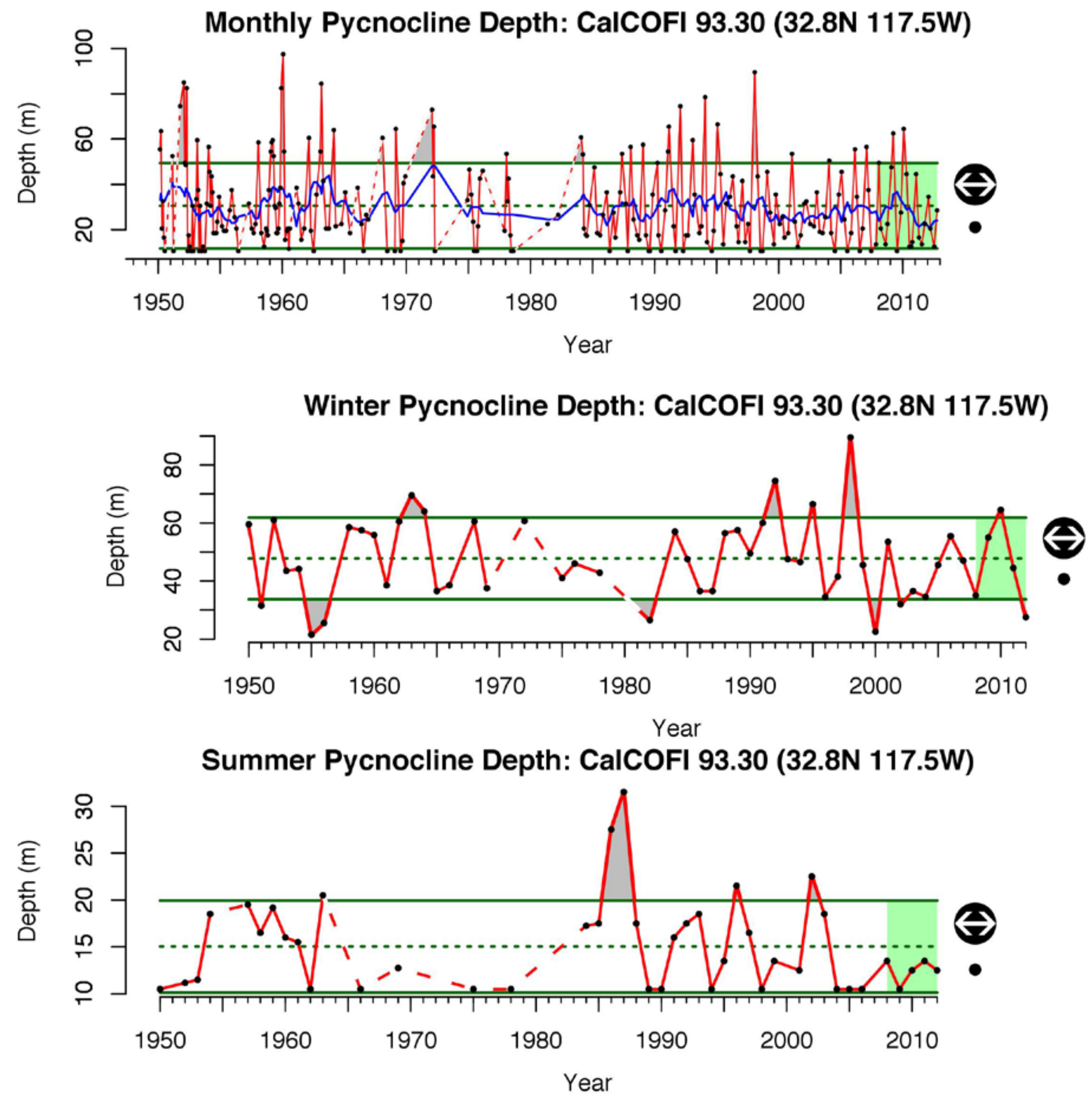


Figure OC 11. Pycnocline depth data from 1950-2012 and for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Station 93.30 illustrates patterns in the southern portion of the CCLME. Dashed lines show data gaps of greater than 2 years.

PYCNOCLINE STRENGTH

The BVF (Brunt–Väisälä frequency) value indicates the strength of the density gradient in the vertical water column. The stronger the pycnocline, the less mixing of nutrients occurs across the pycnocline. From 2007-2011, pycnocline strength has increased steadily at station 67.55 in central California for both summer and winter (Fig. OC13). In southern California (station 93.30), thermocline strength has been highly variable with no clear trend over this same period (Fig. OC14).

In the northern California current (station NH25), the pycnocline has strengthened (Fig. OC12). As noted above, due to cruise limitations, more recent samples are missing from many stations, or have yet to be processed at the time of writing this report, hence limiting our ability to update these trends. **For 2013, where there are data (NH25 and Winter, CalCOFI 67.55) pycnocline strength decreased, as compared to 2012, and was particularly weak at NH25 during the summer**. *Thus the increased upwelling during 2013 should have been very effective at carrying nutrients to the surface, and thus increasing productivity.*

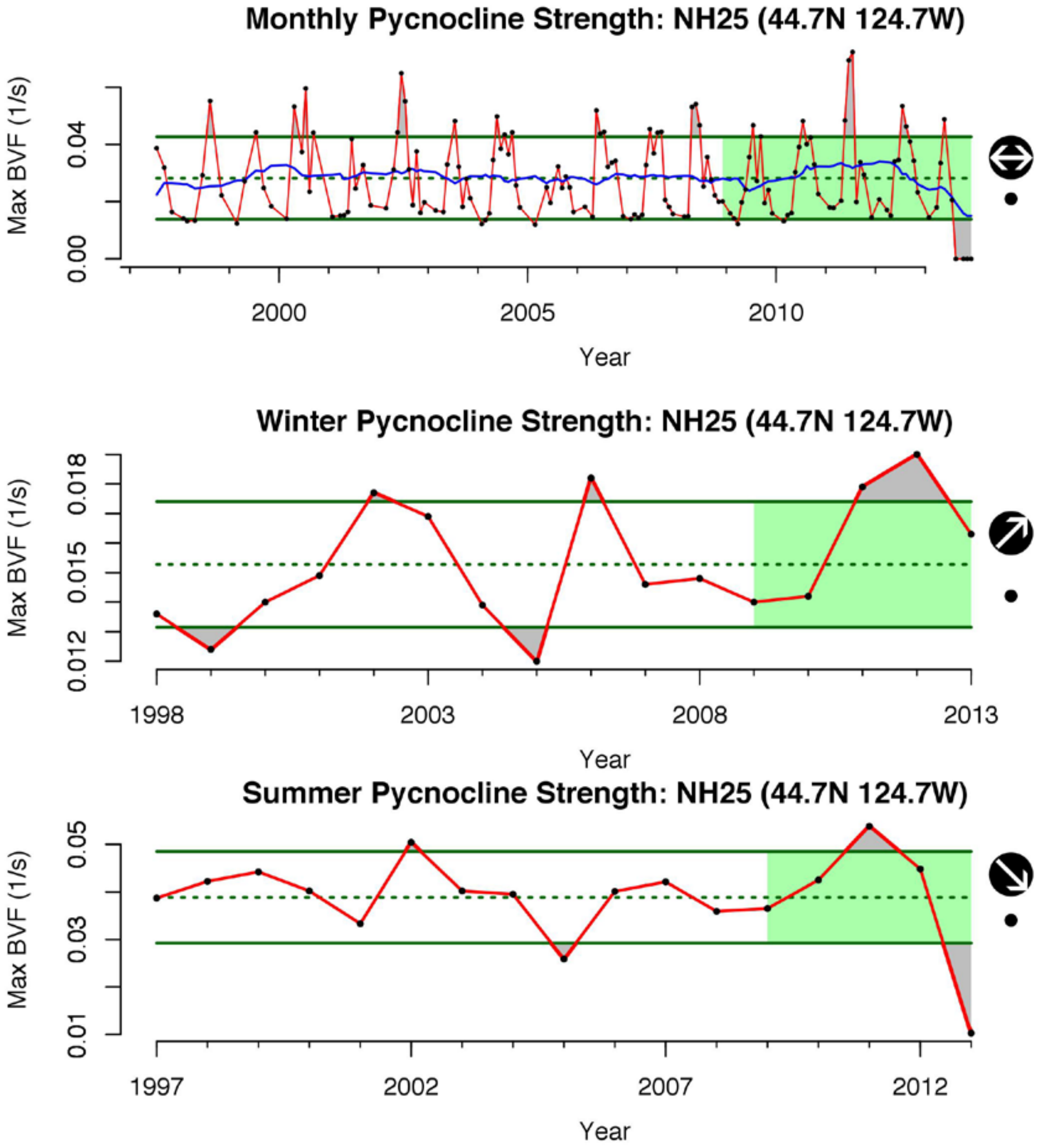


Figure OC 12. Pycnocline strength data from 1998-2013 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Newport line station NH25 illustrates patterns in the northern portion of the CCLME.

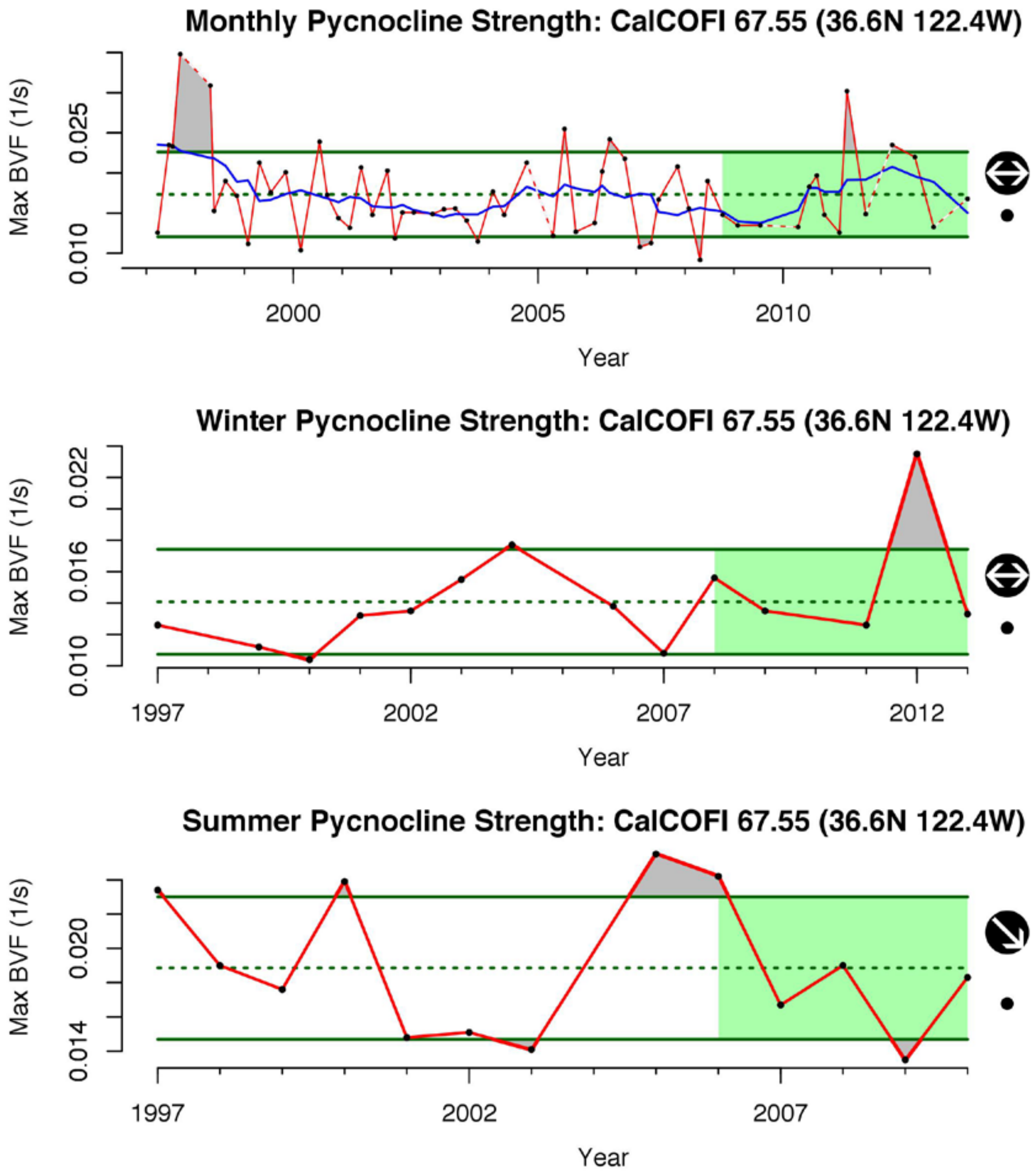


Figure OC 13. Pycnocline strength data from 1998-2013 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Station 67.55 illustrates patterns in the central portion of the CCLME.

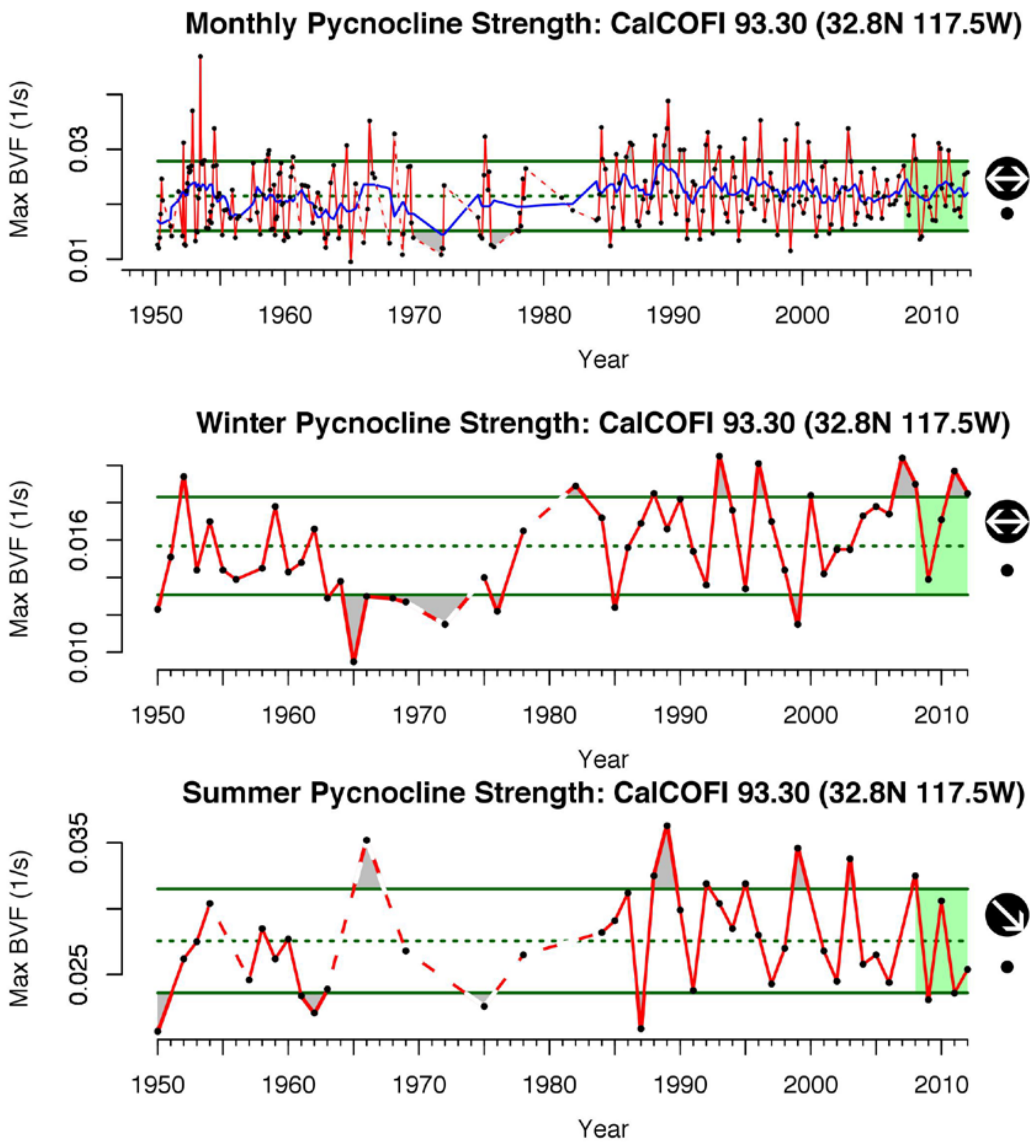


Figure OC 14. Pycnocline strength data from 1950-2012 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Station 93.30 illustrates patterns in the southern portion of the CCLME. Dashed lines identify data gaps of greater than 2 years.

MEI

See Timing and Frequency of El Niño events.

CHANGES IN CALIFORNIA CURRENT TRANSPORT AND MESOSCALE ACTIVITY

BACKGROUND

The major currents of the CCLME are the equatorward flowing California Current and coastal jet, the poleward flowing Undercurrent and Davidson Current, and the Southern California Eddy (Checkley and Barth 2009, King et al. 2011). These currents strengthen at particular times during the year due to local and remote forcing. Embedded in the slow flowing (<5 cm/s) California Current are mesoscale eddies, upwelling filaments, and jets (Checkley and Barth 2009). The geostrophically balanced California Current is present throughout the year, and is surface intensified. In winter, a broad northward flowing current, called the Davidson Current, forms when upwelling inducing winds diminish in strength. The source waters of the California Current and Undercurrent are different, with the California Current being fed by the low-salinity, high-oxygen and high-nitrate water from the North Pacific Current and the Undercurrent's source waters originating from the eastern tropical North Pacific, which are high-salinity, low-oxygen and low-nitrate. Changes in the volume transport of the California Current can result from changes to the North Pacific Current, which is affected by variations in the sea level height over the Northeast Pacific (Cummins and Freeland 2007). The North Pacific Gyre Oscillation (NPGO) index represents variations in the eastern and central regions of the North Pacific Gyre circulation. Since the NPGO is significantly correlated with nutrients and chl-a in the southern CCLME, it also provides a rough index of California Current transport (Di Lorenzo et al. 2008). Eddies and fronts provide important habitat for top predators in the California Current through prey aggregation (Wells et al. 2008, Kappes et al. 2010). We have indexed mesoscale activity using remotely-sensed measures of eddy kinetic energy (EKE) calculated from altimetry data (Strub and James 2000, Haney et al. 2001).

EVALUATION AND SELECTION OF INDICATORS

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

STATUS AND TRENDS

EKE

Eddy Kinetic Energy (EKE) is a measure of mesoscale activity calculated from the square of the zonal and meridional geostrophic flow. High EKE values indicate more mesoscale activity (front, eddies, jets) with much of the eastern Pacific having low EKE values (<300 cm^2/s^2). EKE has not shown a long-term trend at any of the three locations (33°N, 39°N, and 45°N) in winter or summer (Figures OC15 – OC17). Since 2009, EKE has been variable with no clear trend at 33°, decreased at 39°N, and decreased in the summer but not winter at 45°N. **Summer of 2013 showed generally decreased EKE at all three stations as compared to the immediately previous years.** *These observations suggest that 2013 appears to have had lower than average mesoscale activity*.

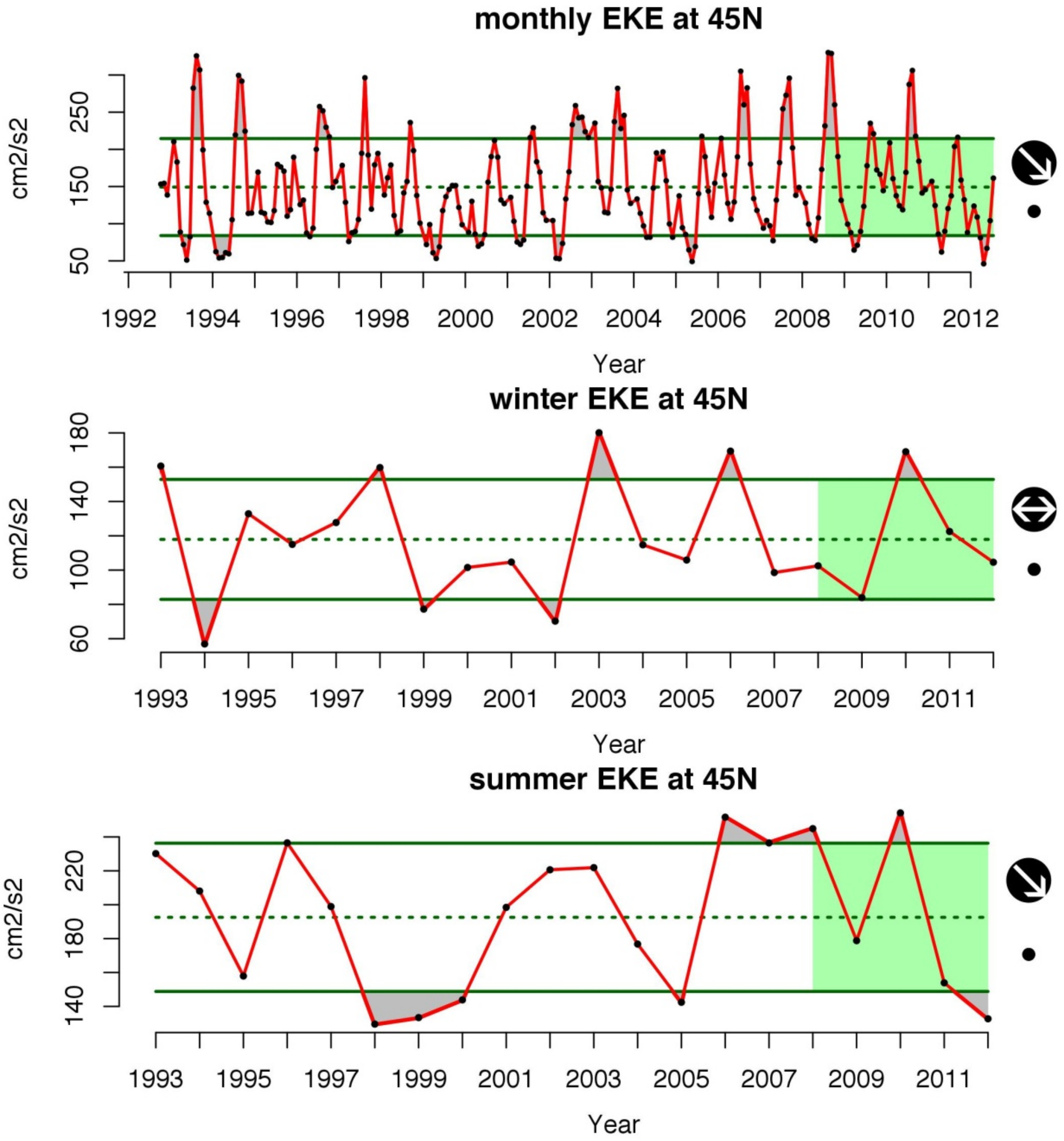


Figure OC 15. Satellite altimetry determined Eddy Kinetic Energy from 1992-2012 at 45°N for monthly, winter and summer means. Monthly values are included to show seasonal and long term variability. The region centered on 45°N illustrates patterns in the northern portion of the CCLME.

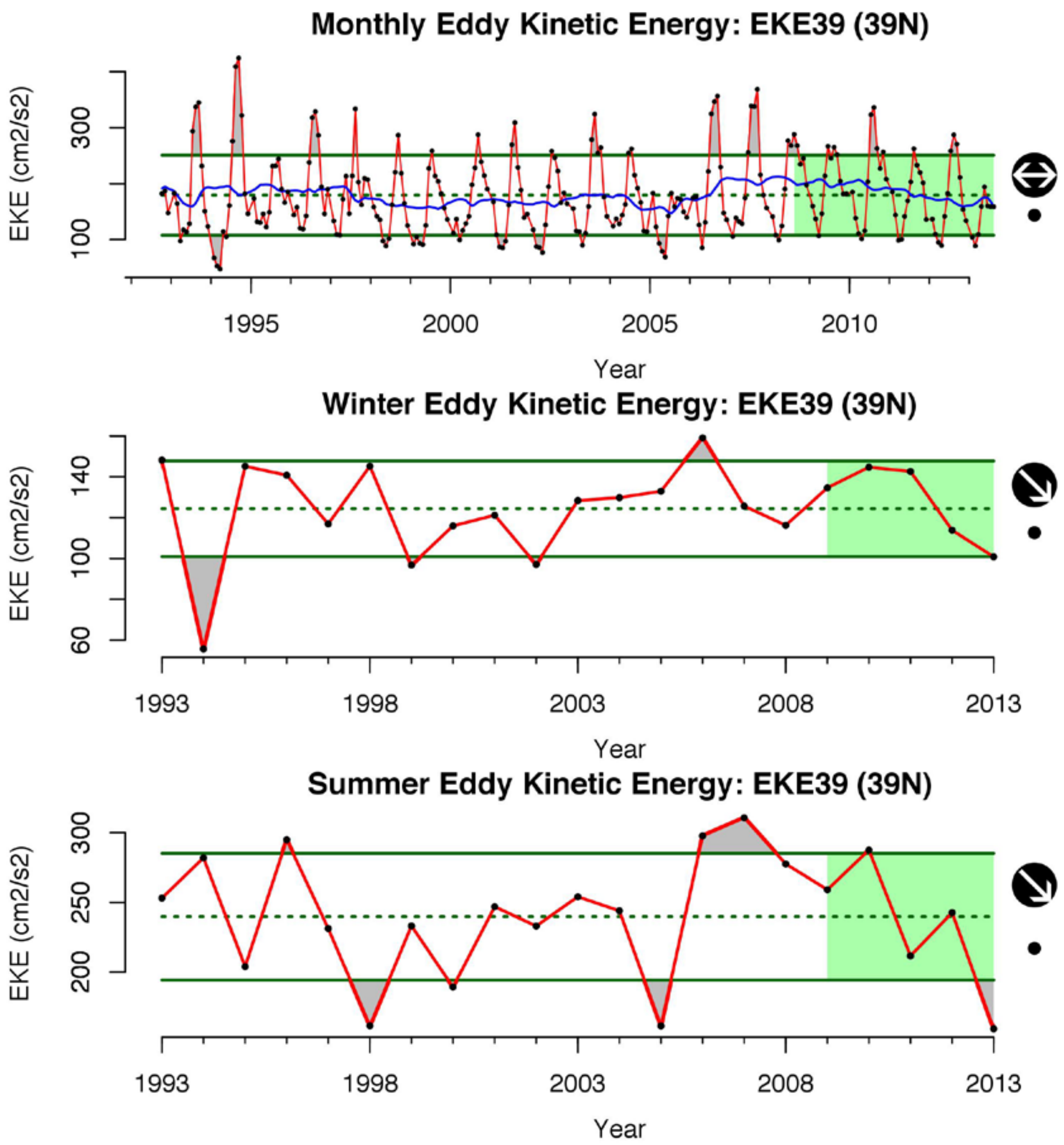


Figure OC 16. Eddy Kinetic Energy satellite data from 1992-2013 at 39°N for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. The region centered on 39°N illustrates patterns in the central portion of the CCLME.

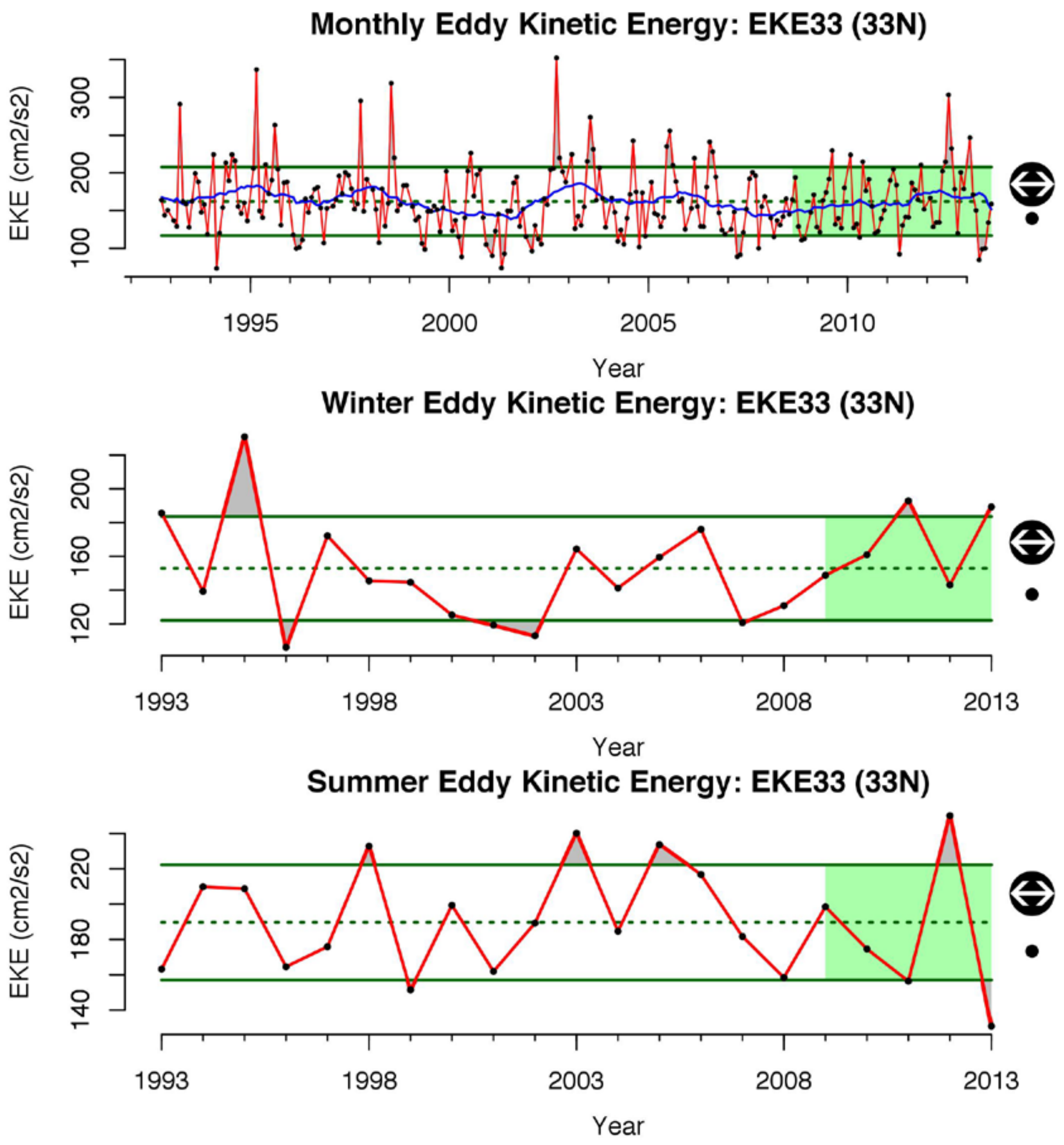


Figure OC 17. Eddy Kinetic Energy satellite data from 1992-2013 at 33°N for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. The region centered on 33°N illustrates patterns in the southern portion of the CCLME.

TIMING AND STRENGTH OF UPWELLING

BACKGROUND

Upwelling is critically important to productivity and ecosystem health in the CCLME (Huyer 1983). The strength and duration of upwelling in the CCLME is highly variable, and is forced by large-scale atmospheric pressure systems. More specifically, the pressure gradient between the oceanic North Pacific High and continental Low situated over the southwestern United States drives upwelling-favorable northerly winds. The interaction (friction and Coriolis force) of the northerly winds and the water surface moves water offshore in the surface layer, and this water is replaced by water upwelled from depths of greater than 50 - 100 m. The upwelled water is cooler, saltier and higher in nutrient concentrations than the surface water it replaces. The onset and duration of the upwelling season varies latitudinally, starting earlier and lasting longer in the southern CCLME (Bograd et al. 2009).

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

EVALUATION AND SELECTION OF INDICATORS

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

STATUS AND TRENDS

UI

Upwelling index (UI). The 2005 upwelling season was unusual in terms of its initiation, duration, and intensity. In 2005 upwelling was delayed or interrupted and SSTs were approximately 2-6°C warmer than normal (Barth et al. 2007). The situation in the southern ecoregion was different in both 2005 and 2006, as average upwelling and SST prevailed (Peterson et al. 2006). Other than a brief period of weaker than normal upwelling in the summer of 2008, west coast upwelling has been increasing since the late summer of 2006 (Figures OC18 – OC20). Wind patterns in early 2009 reflect anomalously strong high pressure over the Northeast Pacific and very high upwelling while early to mid 2010 appears to be a below average upwelling year at lat 35–45°N. **For 2013, the UI increased or stayed relatively high for all stations, as compared to the immediately previous years.** *This supports the evidence that 2013 was a year of strong upwelling.*

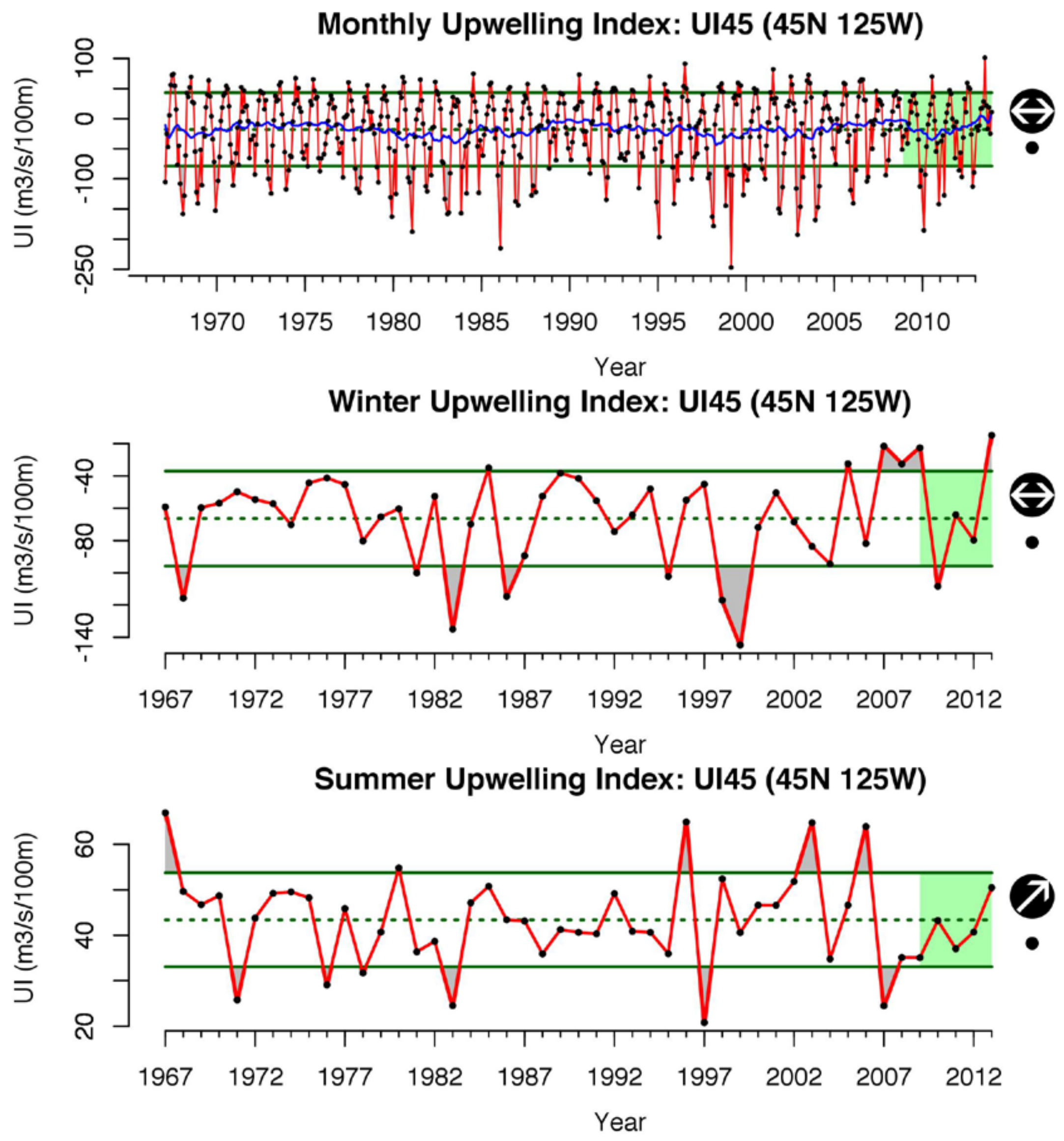


Figure OC 18. The Upwelling Index calculated from 1967-2013 at 45°N for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. The UI at 45°N was illustrates patterns in the northern portion of the CCLME.

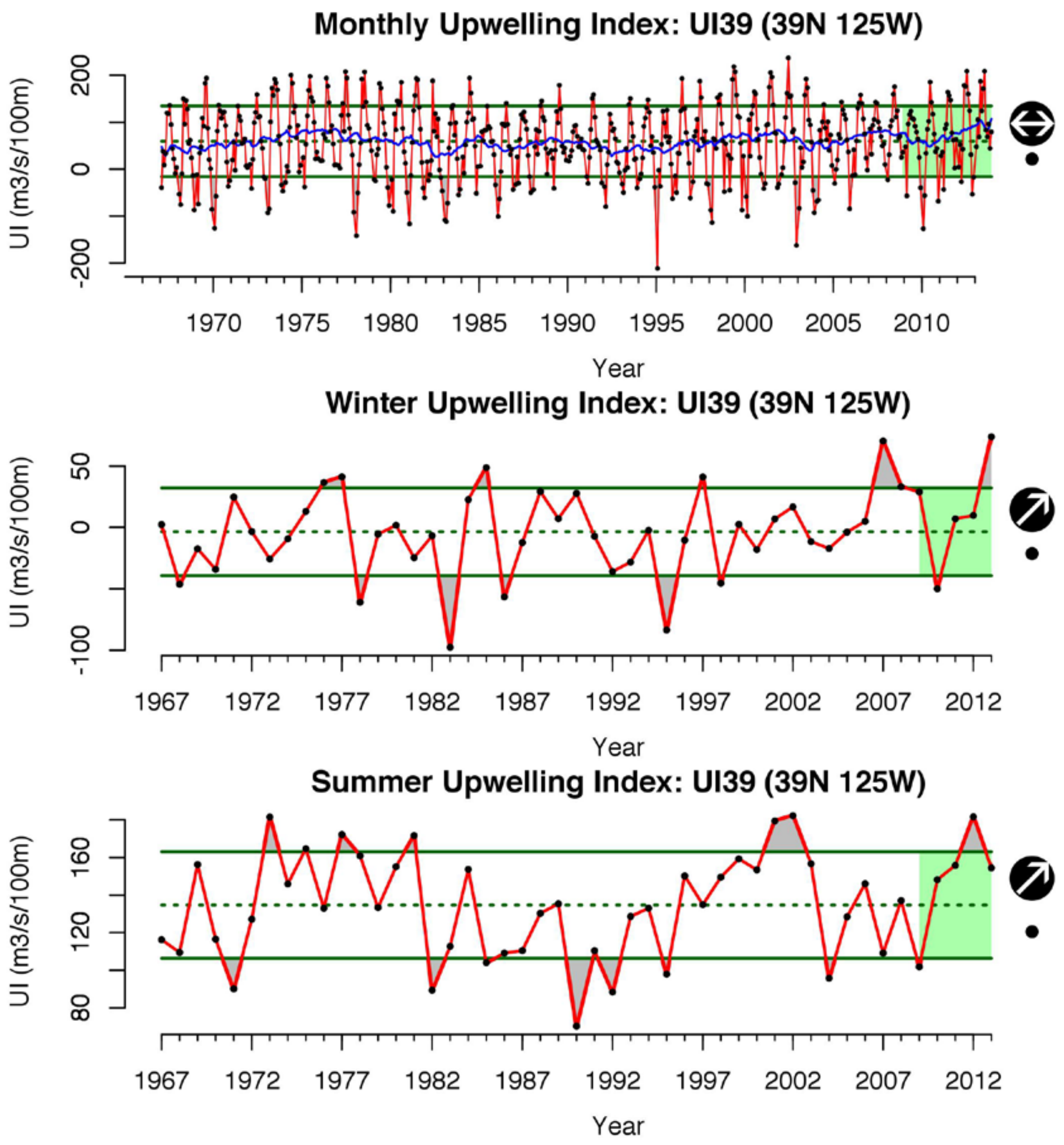


Figure OC 19. The Upwelling Index calculated from 1967-2013 at 39°N for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. The UI at 39°N illustrates patterns in the central portion of the CCLME.

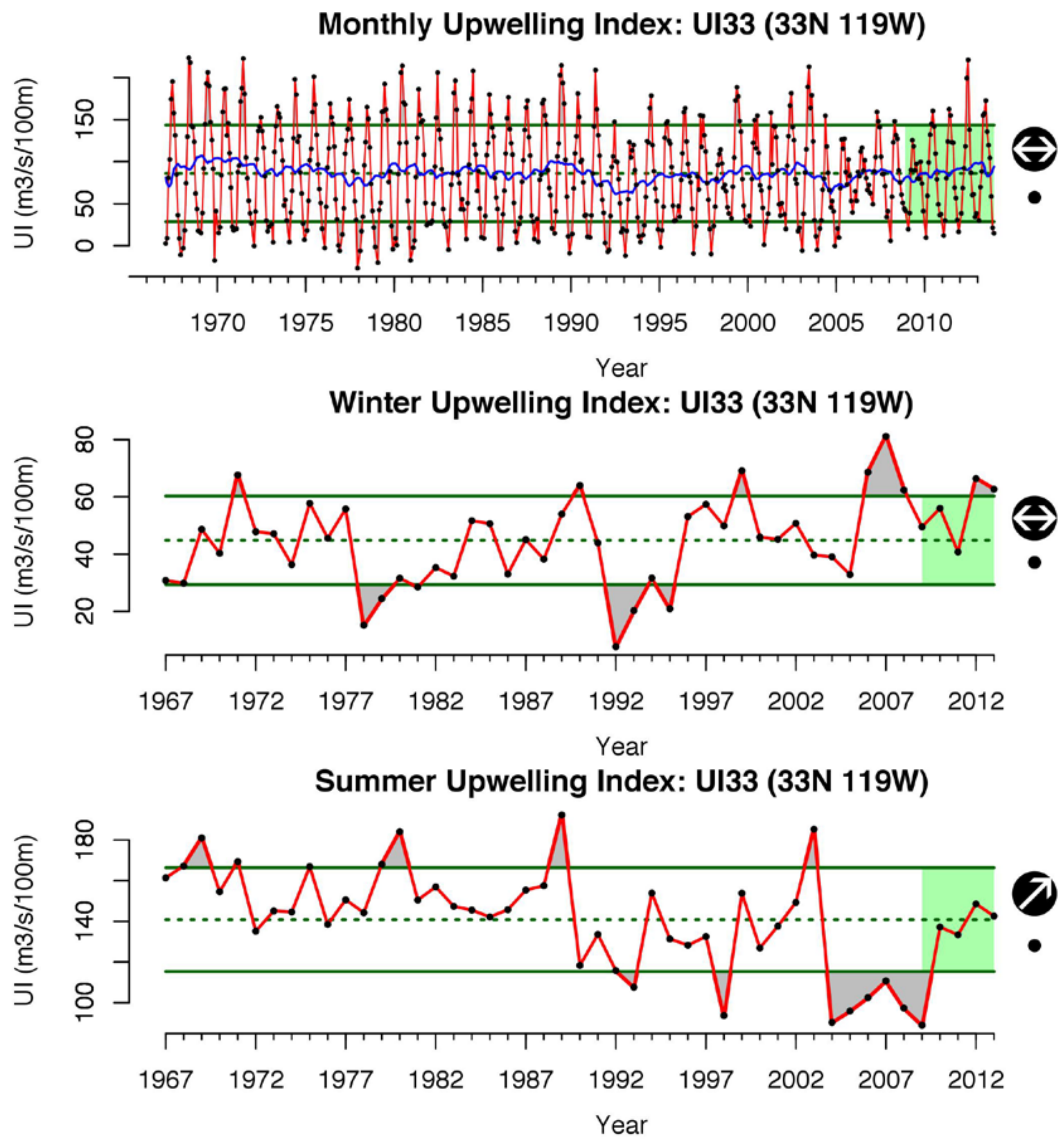


Figure OC 20. The Upwelling Index calculated from 1967-2013 at 33°N for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. The UI at 33°N illustrates patterns in the southern portion of the CCLME.

STI

The spring transition index (STI) indicates roughly the start of the upwelling season. It is defined by the date the annual cumulative upwelling index (CUI) reaches its minimum value (Bograd et al. 2009). The STI fluctuates around 10 days past March 1st with a few extremely early or late years. In the early 90's and in 2005 anomalously late upwelling occurred with a severe effect on many biological time series. The past 5 years have had variable STI values ranging from ± 1 standard deviation (Figure OC21). Given the UI often remains positive at 33°, confounding the calculation of the STI, we have excluded this time series. **For 2013, the STI decreased for both locations as compared to 2012**. *The earlier transition to upwelling in 2013 supports the evidence that total upwelling during 2013 was higher than previous years*.

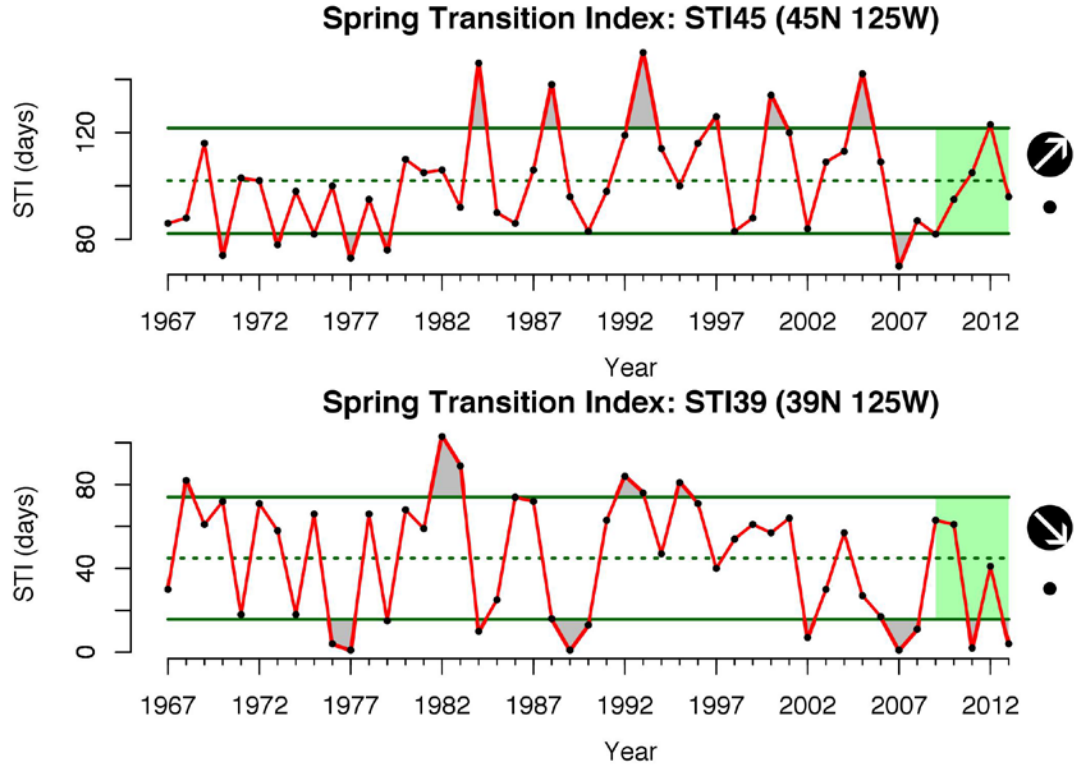


Figure OC 21. The Spring Transition Index (STI) calculated yearly from 1967-2013 at 45°N and 39°N. The STI at 33°N is not included because there is not an extended downwelling phase during a year at this latitude.

LUSI

The length of the upwelling season (LUSI) is determined by the date of the STI until the date of the CUI maximum. This length of upwelling season indicates how long the upwelling favorable conditions persisted over the year. Over the past 5 years, LUSI has been highly variable at 39° while showing a declining trend at 45°N (Figure OC22). **For 2013 vs 2012, the length of the upwelling season was longer at the southern station, and similar at the northern station.** *This supports the evidence that 2013 was in general a stronger year of upwelling, although the intensification seems to have been greater in the southern portions of the current.*

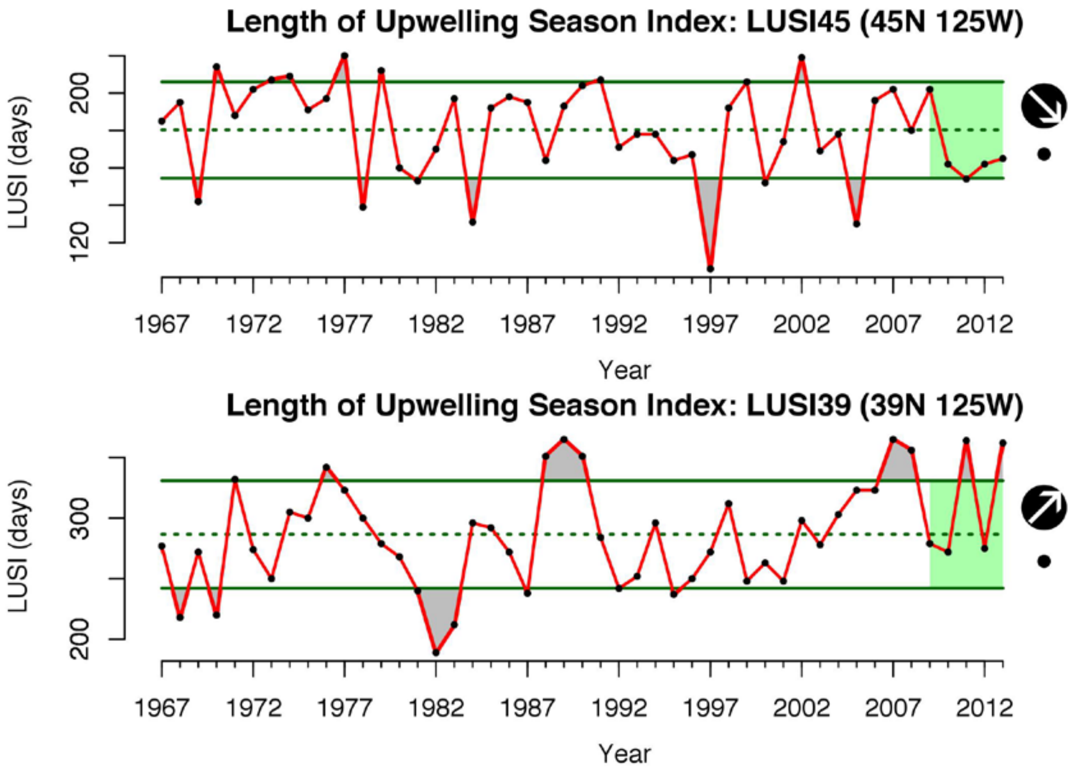


Figure OC 22. The Length of the Upwelling Season Index (LUSI) calculated yearly from 1967-2013 at 45°N and 39°N. The LUSI at 33°N is not included because there is not an extended downwelling phase during a year at this latitude.

TUMI

The total upwelled magnitude index (TUMI) is the sum of the UI over the duration of the upwelling season (e.g. LUSI). This index represents the total amount of upwelled water as an indicator of total upwelled nutrient availability to the photic zone for the year. At the southernmost station, TUMI has been variable with minima in 1992-1993 and 2004-2005 although no clear trend since 2007 (Figure OC23). TUMI at 39° N shows a decadal pattern with high values in the 1970s, low values in the 1980s-1998 and high values since 1999 with the exception of 2003-2004. At 45° N, TUMI had a minima in 1997 and a maxima in 2006. Since 2006, values have been below the mean but not extremely so. For the past 5 years, values have increased at all three locations. **For 2013, TUMI was high at all three locations, with central CA showing the highest value on record**. *This further supports the evidence that 2013 was a year of high total upwelling as compared to previous years, with the southern portion being more intensified than the north*.

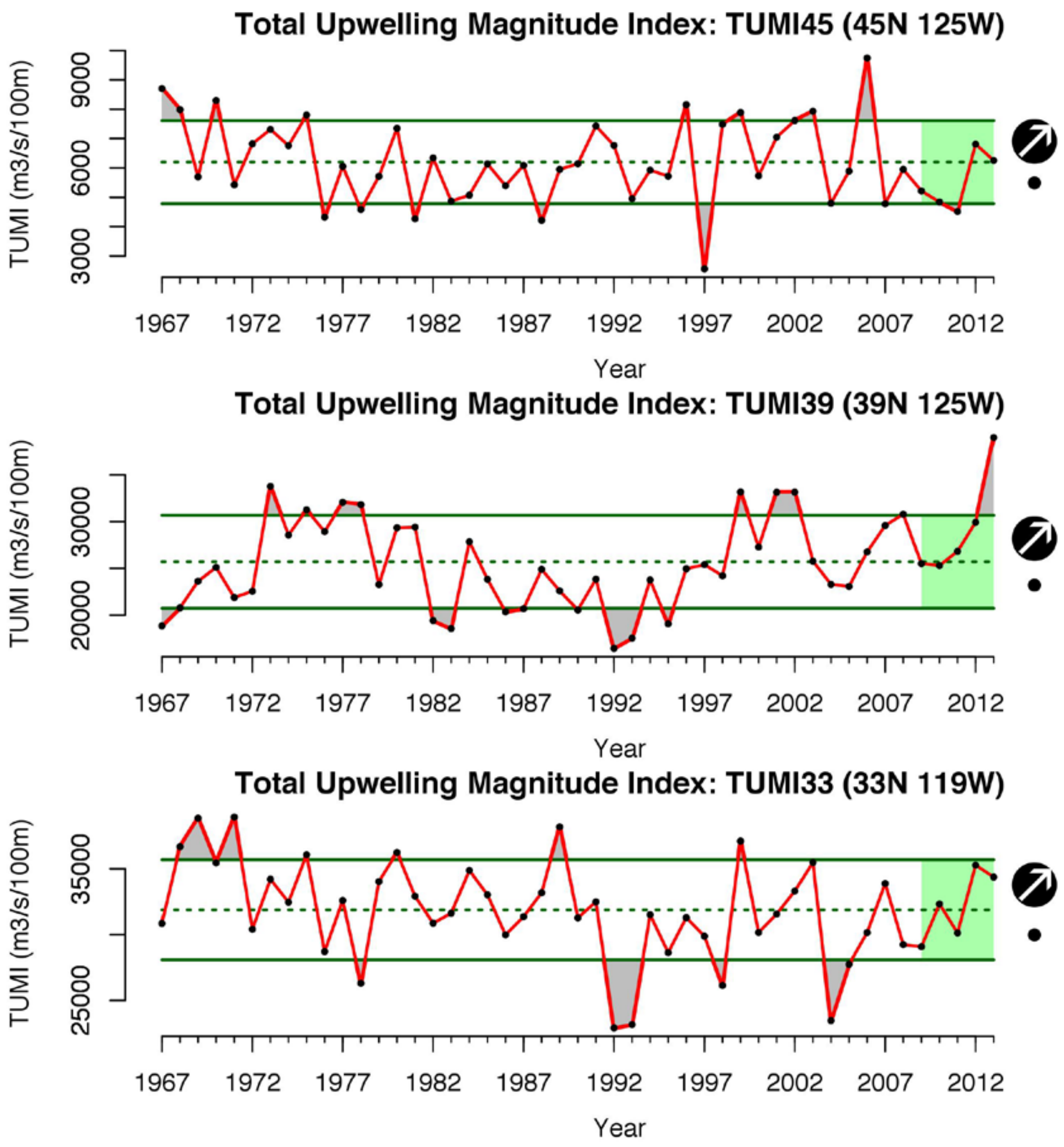
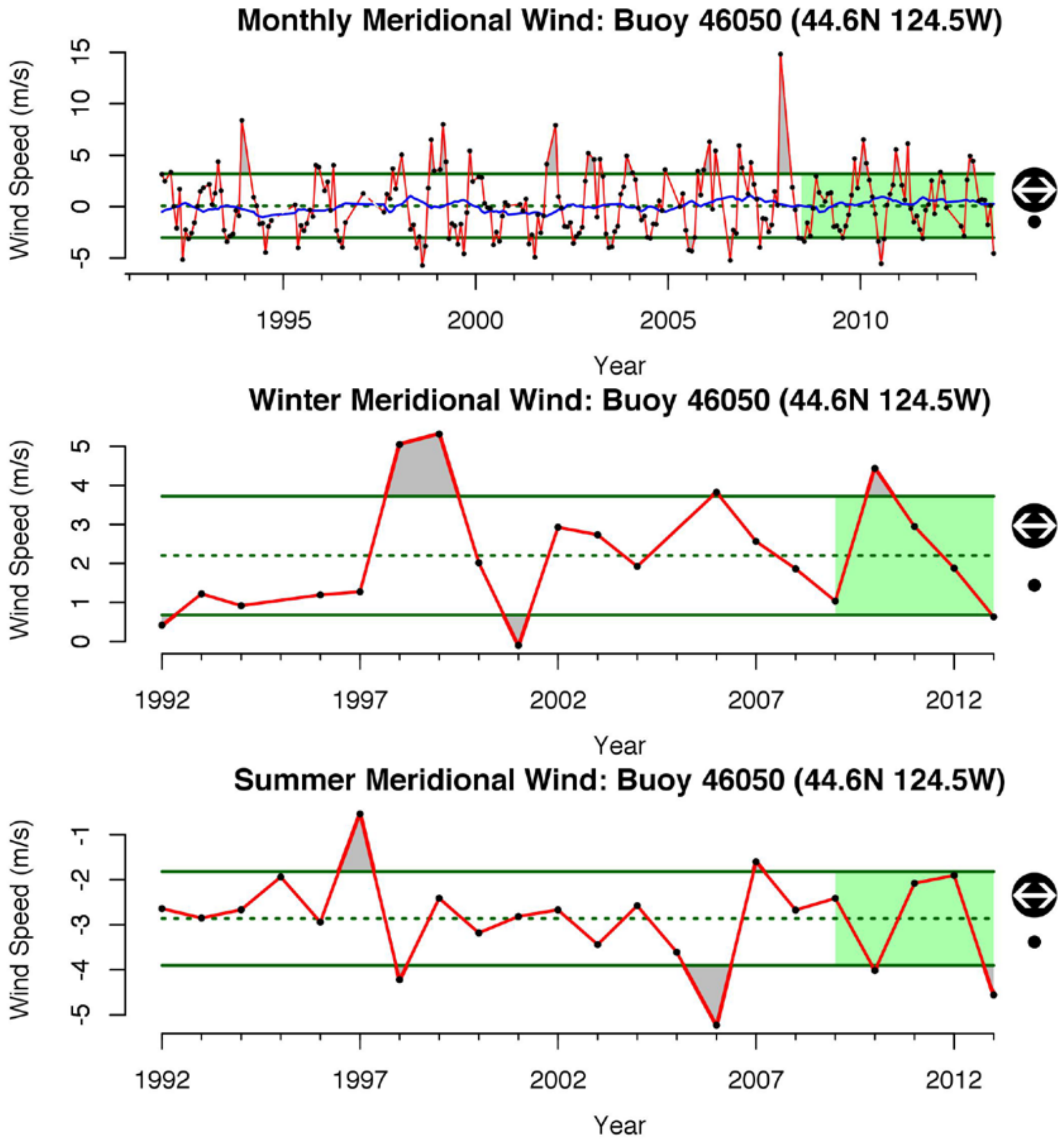


Figure OC 23. The total upwelled magnitude index (TUMI) calculated yearly from 1967-2013 at 45°N, 39°N and 33°N.

WINDS

Northerly winds in the CCLME result in offshore transport and upwelling of cold, nutrient rich water into the photic zone. In the winter, meridional (north/south) winds were consistently northward in 1998 and 2010, indicative of downwelling favorable conditions (positive MEI and NOI; Figures OC24 – OC26). In winter 2006, winds were also indicative of downwelling although less extreme than 1998 and 2010. In summer 2006 and winter 2007, there were highly favorable upwelling winds at the northern buoys (A and B). In summer 2010, upwelling favorable winds dominated all three buoys, although they declined at 39° N in 2011. **For 2013, meridional winds were more southward (e.g. more strongly towards negative values) except for the**

summertime values at the southernmost station, which were anomalously strongly northwards. *These observations support the evidence that 2013 was a year of atmospherically forced strong upwelling, which began early for most of the CCS, albeit with an altered anomalous state to the south.*


Monthly Meridional Wind: Buoy 46050 (44.6N 124.5W)

Winter Meridional Wind: Buoy 46050 (44.6N 124.5W)

Summer Meridional Wind: Buoy 46050 (44.6N 124.5W)

Figure OC 24. Alongshore, meridional winds buoy data from early 1990 -2013 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Buoy 46050 was chosen to illustrate patterns in the northern portion of the CCLME.

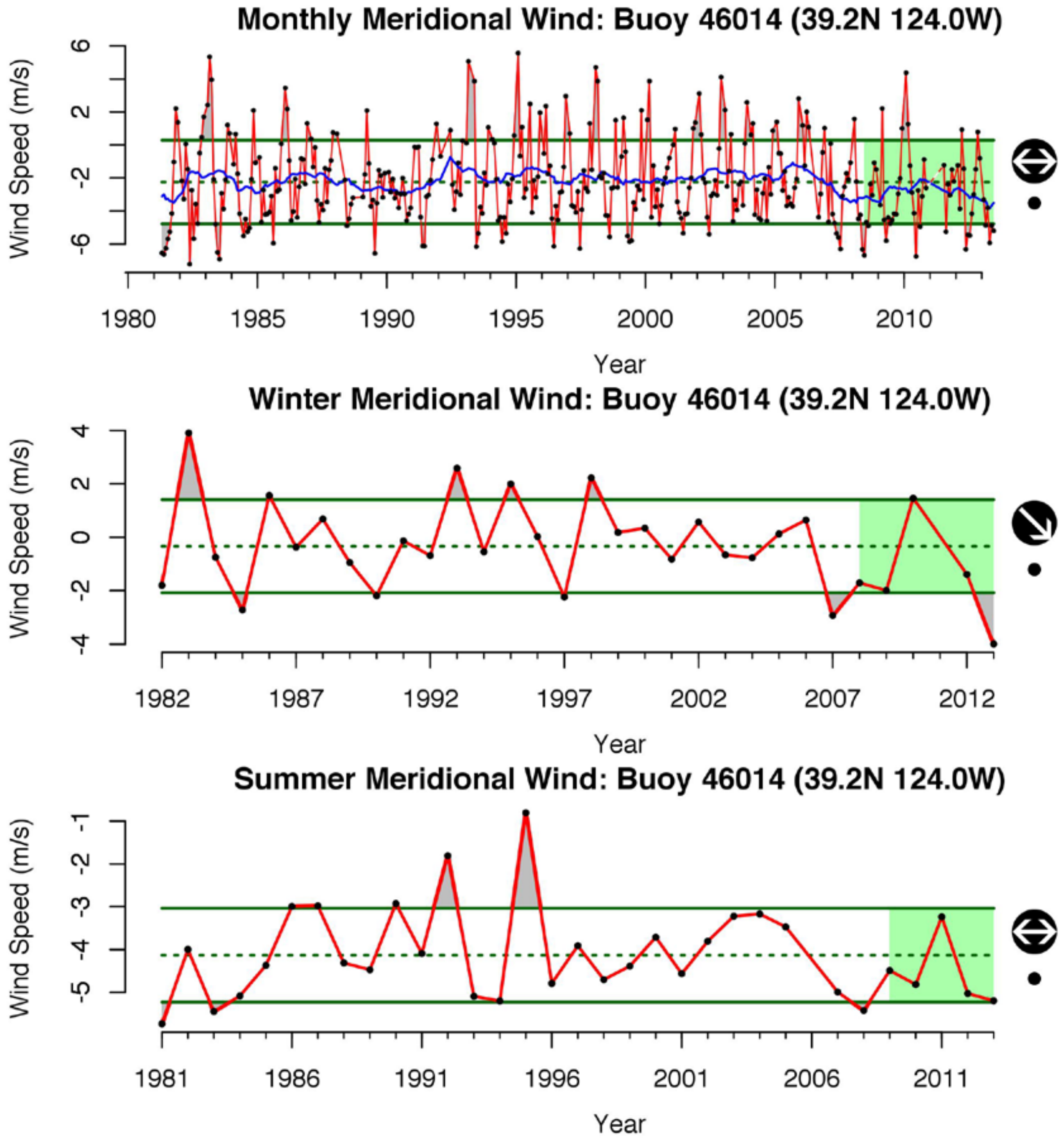


Figure OC 25. Alongshore, meridional winds buoy data from early 1990 -2013 for: monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Buoy 46014 was chosen to illustrate patterns in the central portion of the CCLME.

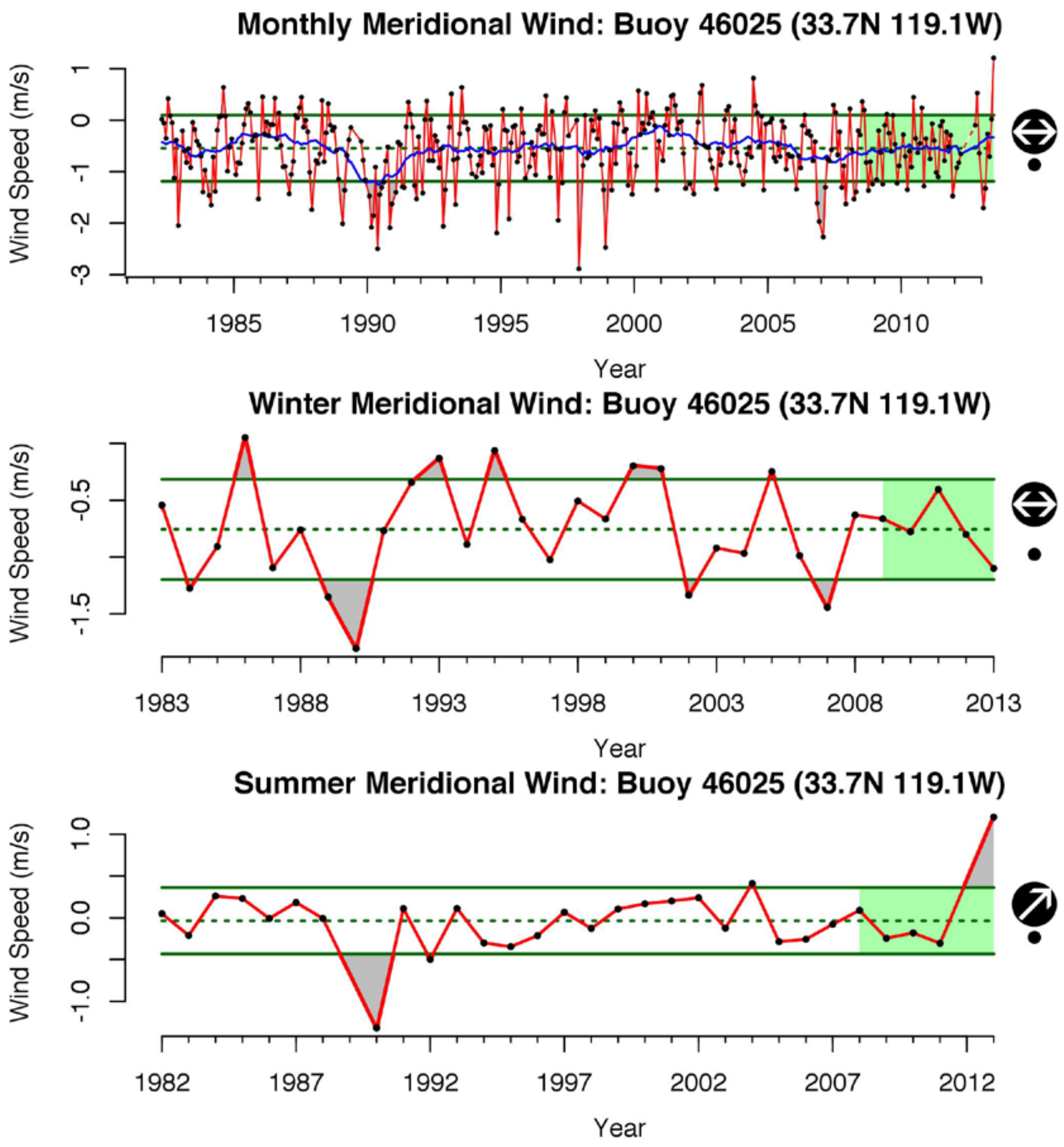


Figure OC 26. Alongshore, meridional winds buoy data from early 1990 -2013 for: monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Buoy 46025 was chosen to illustrate patterns in the southern portion of the CCLME.

TIMING AND FREQUENCY OF EL NIÑO EVENTS

BACKGROUND

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

EVALUATION AND SELECTION OF INDICATORS

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

STATUS AND TRENDS

MEI

The Multivariate ENSO Index (MEI) describes ocean-atmosphere coupling in the equatorial Pacific. Positive values of the MEI represent El Niño conditions while negative values represent La Niña conditions. El Niño conditions in the CCLME are associated with warmer surface water temperatures and weaker upwelling winds. The MEI also had an increasing trend, with more positive values since 1977 (Figure OC27). Most recently, the MEI had a relatively strong negative value in the winter of 2008 indicating La Niña conditions that typically favor ocean/atmospheric teleconnections and high productivity coupled with subarctic conditions in the CCS. The MEI switched to positive indicating El Niño conditions in the beginning of 2010, which switched to a negative value in the summer of 2010. La Niña conditions continued through mid-2011 and have begun to return to neutral in late 2011. **The summer of 2012 saw higher values of the MEI, with average to low values of the MEI in the following summer and winters of 2013**. *Based on these recent MEI values, 2013 was not an El Niño year, which is supported by the relatively lower SSTs also observed during 2013.*

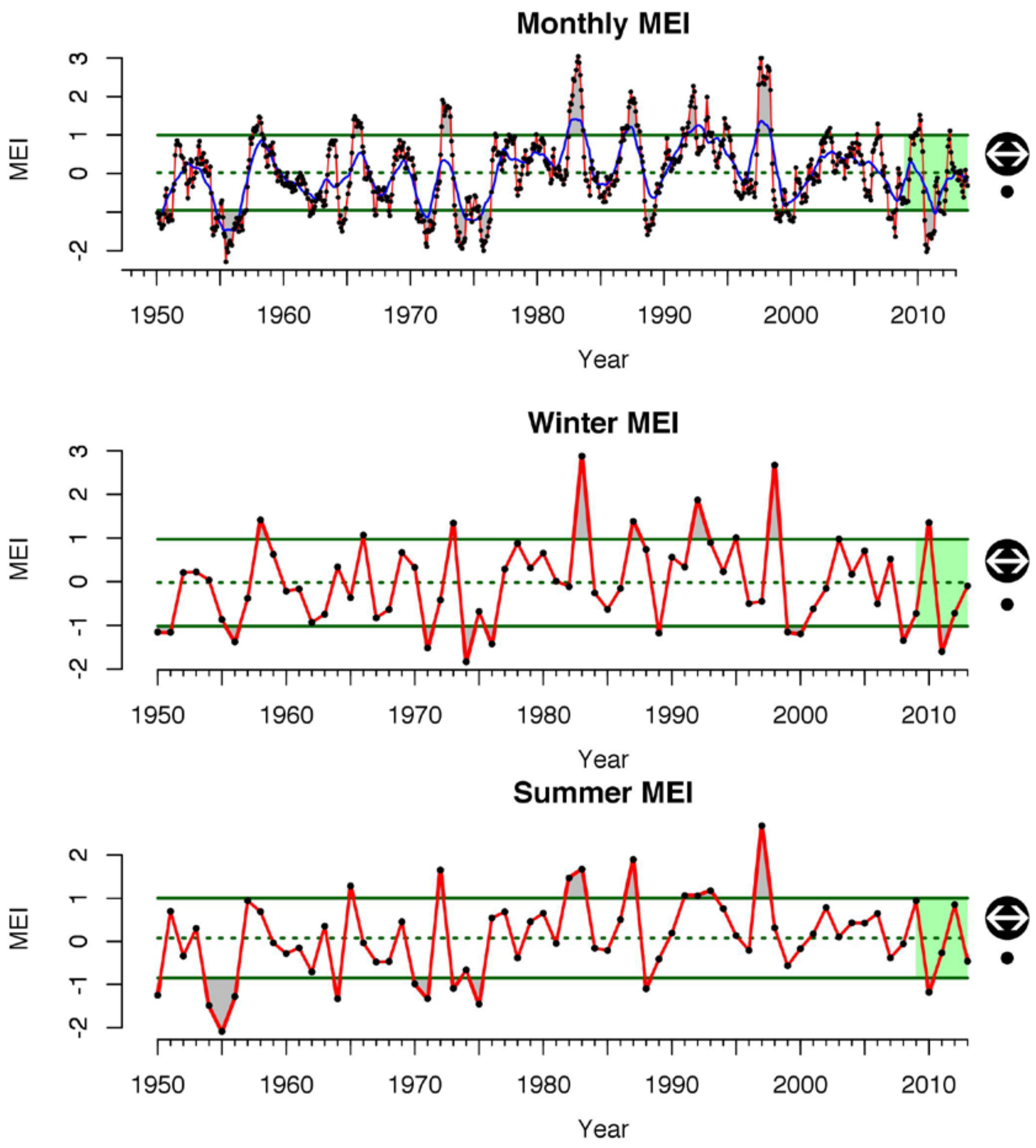


Figure OC 27. Multivariate ENSO Index values (MEI) from 1950-2013 for monthly, winter and summer means. For the monthly MEI the blue line shows a running annual average.

NOI

See sea surface temperature change above.

CHANGES IN SOURCE WATERS

BACKGROUND

Subarctic and tropical waters are important contributors of source waters to the CCLME at the upstream end and through local upwelling cells. Variations in the volume of subarctic waters occur both at the origination through ventilation (Bograd et al. 2008), transport eastward (Di Lorenzo et al. 2008) in the North Pacific Current (NPC) and as a function of where the NPC approaches the continental shelf and bifurcates into the southward-flowing California Current and the northward-flowing Alaska Current (Bi et al. 2011, Sydeman et al. 2011). Broad scale changes in nutrients and hypoxia in the California current are a function of source water changes and we have observed increased nutrients and decreased oxygen in the Southern California Bight over the past 25 years (Bograd et al. 2008). Earth system models have predicted further decreases in nutrients and oxygen over the next century (2001-2100) in part due to changes in offshore stratification and ventilation (Rykaczewski and Dunne 2011). Broad scale forcing (e.g. indexed by the Pacific Decadal Oscillation, PDO and North Pacific Gyre Oscillation, NPGO) can influence both the strength of transport and the location of bifurcation in the CCLME with downstream ecosystem consequences (King et al. 2011). Increases in subarctic source waters can result in changes in the food-web as cooler arctic waters carry larger, lipid-rich copepods and other plankton, compared to the smaller, often lipid-poor warm water copepods found offshore and to the south. Differences in copepod species composition can serve as ecological corroboration of changes in source water (Peterson and Keister 2003). The result is different trophic structure near the bifurcation (Bi et al. 2011). Dissolved Oxygen (discussed below) can also be used as an indicator of changes in source water (Bograd et al. 2008, Pierce et al. 2012)

EVALUATION AND SELECTION OF INDICATORS

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

STATUS AND TRENDS

NPGO

The North Pacific Gyre Oscillation (NPGO) is a low frequency signal in sea surface heights over the Northeast Pacific. Positive (negative) values of the NPGO are linked with increased (decreased) surface salinities, nutrients, and chl-a values in the CCLME (Di Lorenzo et al. 2008). Many NPGO events since 1975 seem to have been more extreme or had a longer duration than those earlier in the time series (Figure OC28). Winter and summer trends were very similar with a broad low from 1991 to 1997 and a peak from 1998 to 2004. Since 2006, values have been increasing with the past 5 years falling around or above 1 standard deviation from the mean. **For 2013, the NPGO remained high and roughly similar to the previous several years.** *This suggests 2013 should have* *had high surface salinity, high nutrients, and resulting high chl-a values, further supporting the trends in upwelling strength that 2013 should have been a highly productive year.*

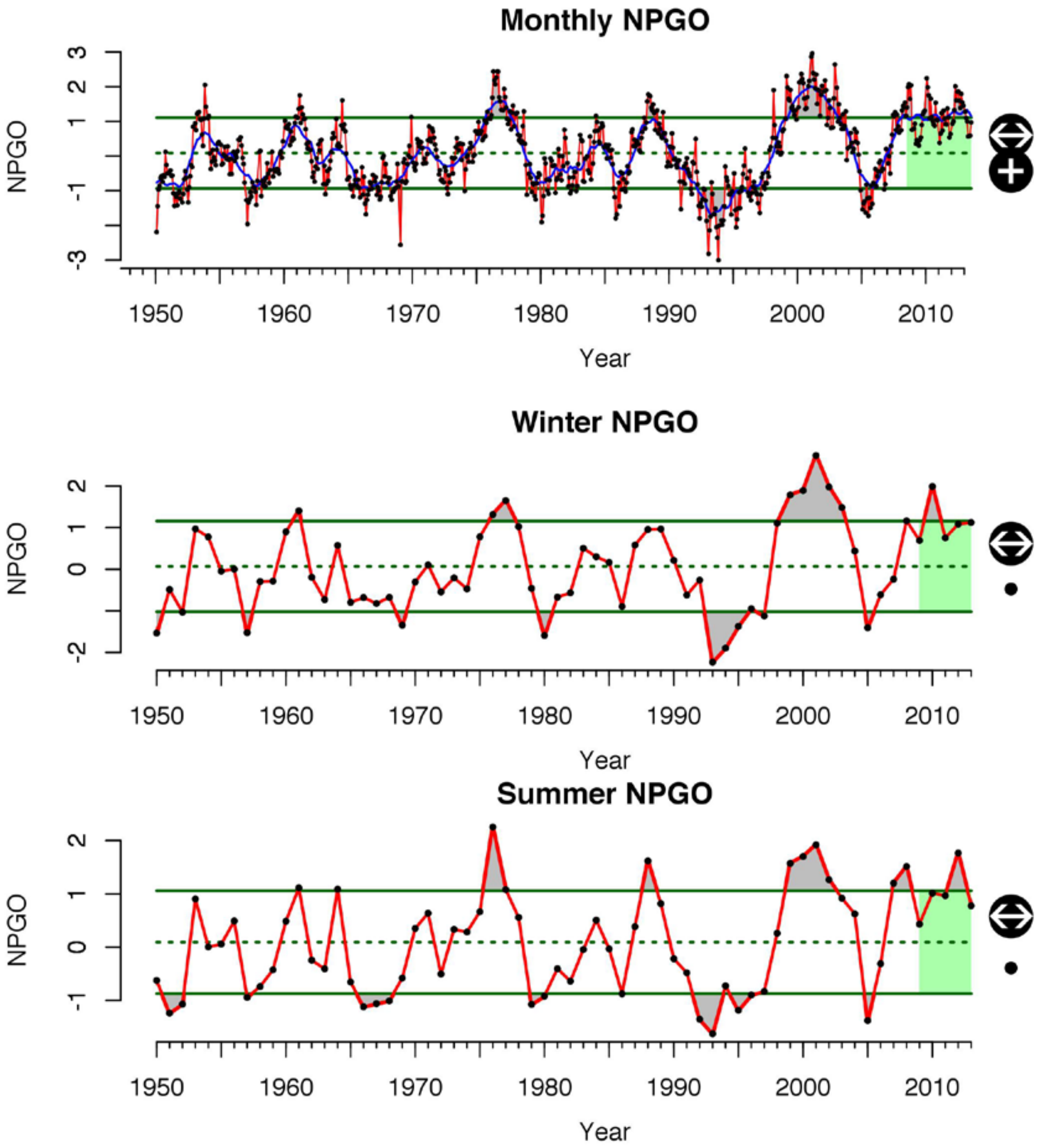


Figure OC 28. North Pacific Gyre Oscillation values (NPGO) from 1950-2013 for monthly, winter and summer means. For the monthly NPGO the blue line shows a running annual average.

NUTRIENT CONTENT

Nutrient content (represented by nitrate plus nitrite, NO_2 and NO_3) is a function of upwelling intensity and stratification, but also depends on the source waters that are upwelled. Deep casts at stations 93.30 in CALCOFI and NH25 (150 m) reflect the status of the source. CALCOFI nutrients in central (station 67.55) and southern (station 93.30) California at 150 m depth show no long-term trend from the data available., but have generally increased over the past 5 years Central California nutrients peaked in 2009 and have declined since, while southern California nitrate and nitrite values had a large drop in 2008 and have increased through 2012 (Figures OC29-OC31). **For 2013, when and where we have data, nutrients remained high.** *High nutrients during summer of 2013 support the contention that 2013 should have been a highly productive year.*

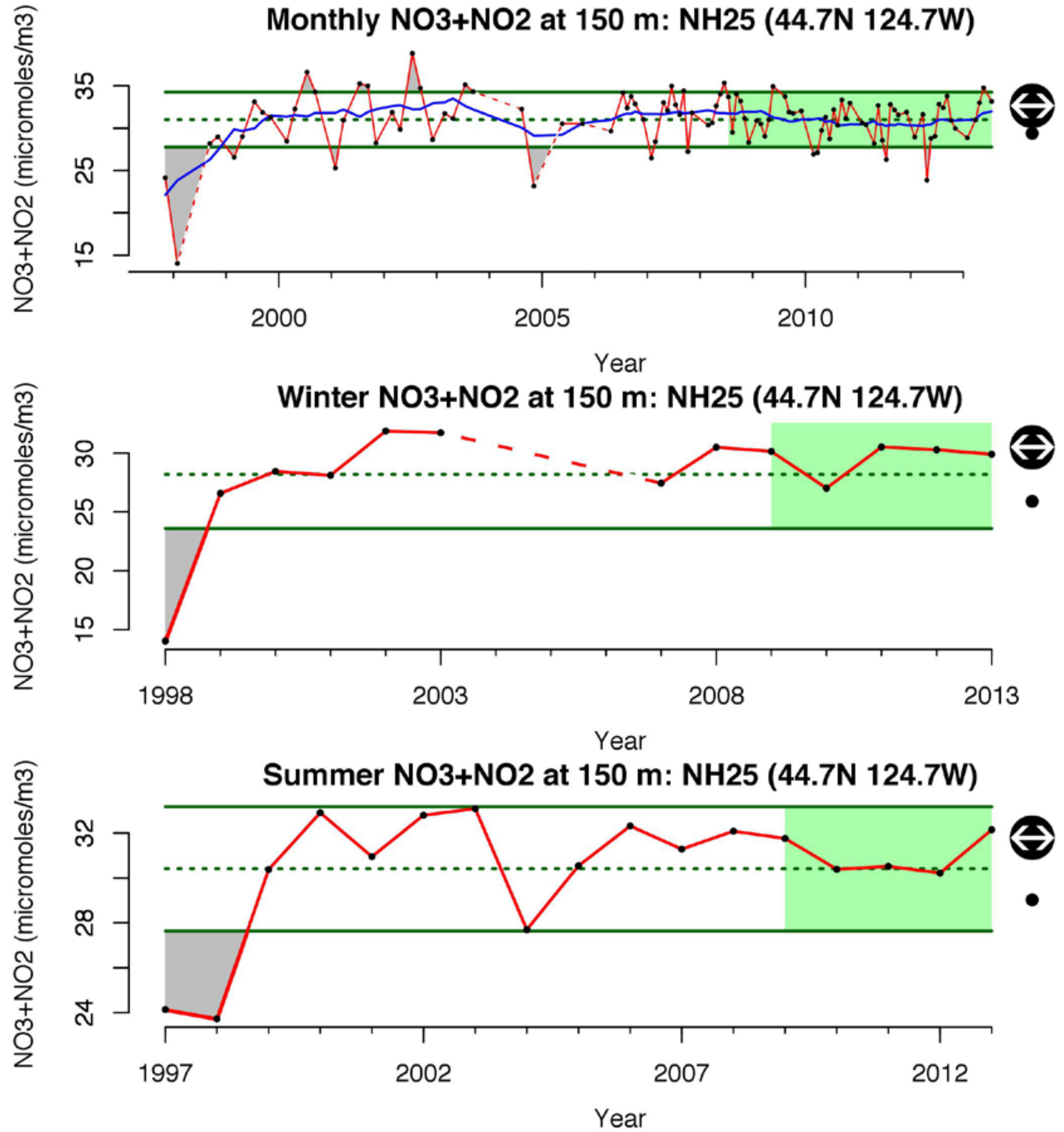


Figure OC 29. Nutrient data (nitrate + nitrite) at 150 m from 1997-2013 at station NH25 from the Newport line in the northern CCLME for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Station NH25 illustrates patterns in the northern portion of the CCLME. Dashed lines show data gaps of greater than 2 years.

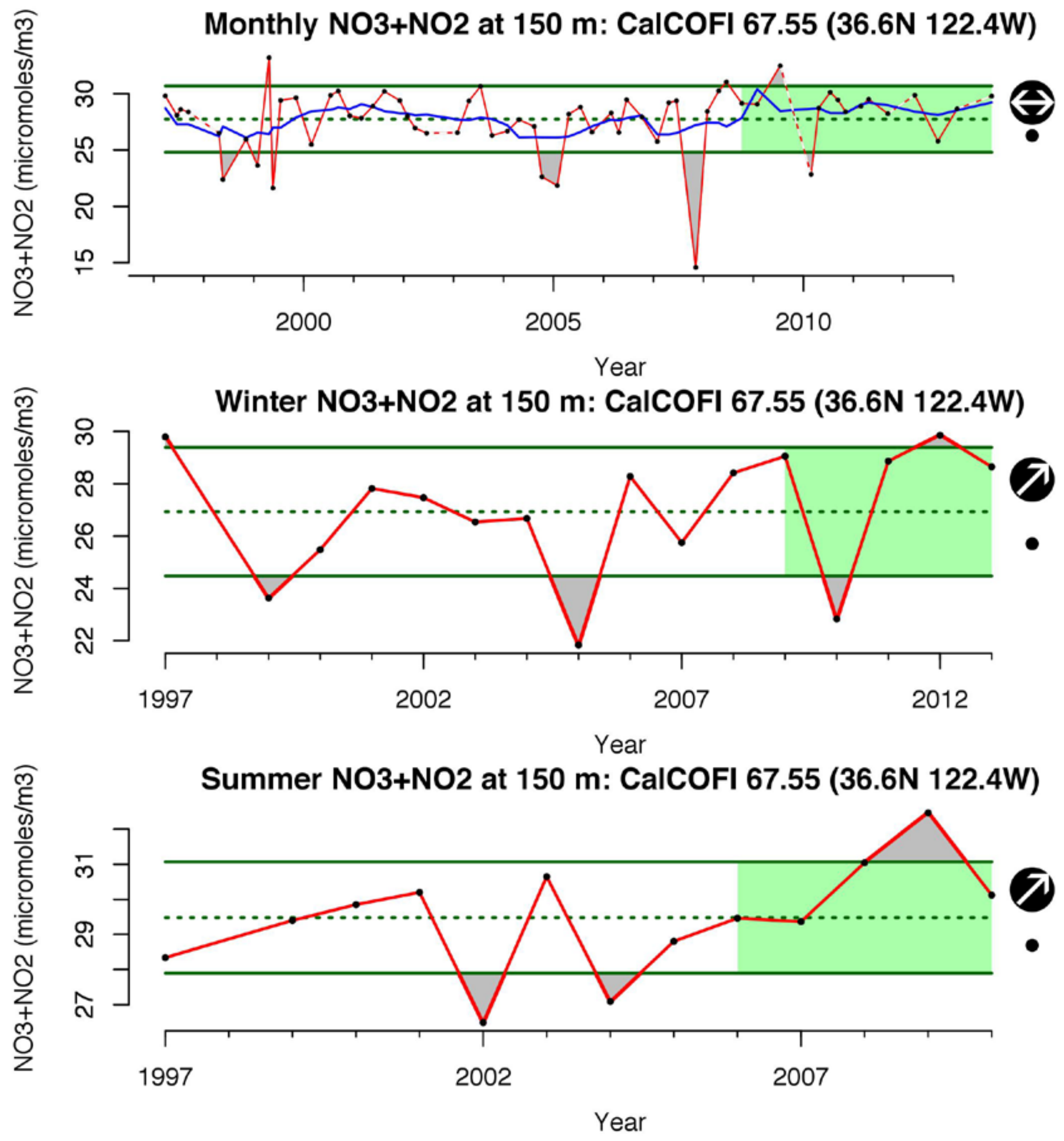


Figure OC 30. Nutrient data (nitrate + nitrite) at 150 m from 1997-2013 at CalCOFI station 67.55 representing central California for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Station 67.55 illustrates patterns in the central portion of the CCLME. For the last 3 years sampling has not done at station 67.55 in the summer (Jun-Aug).

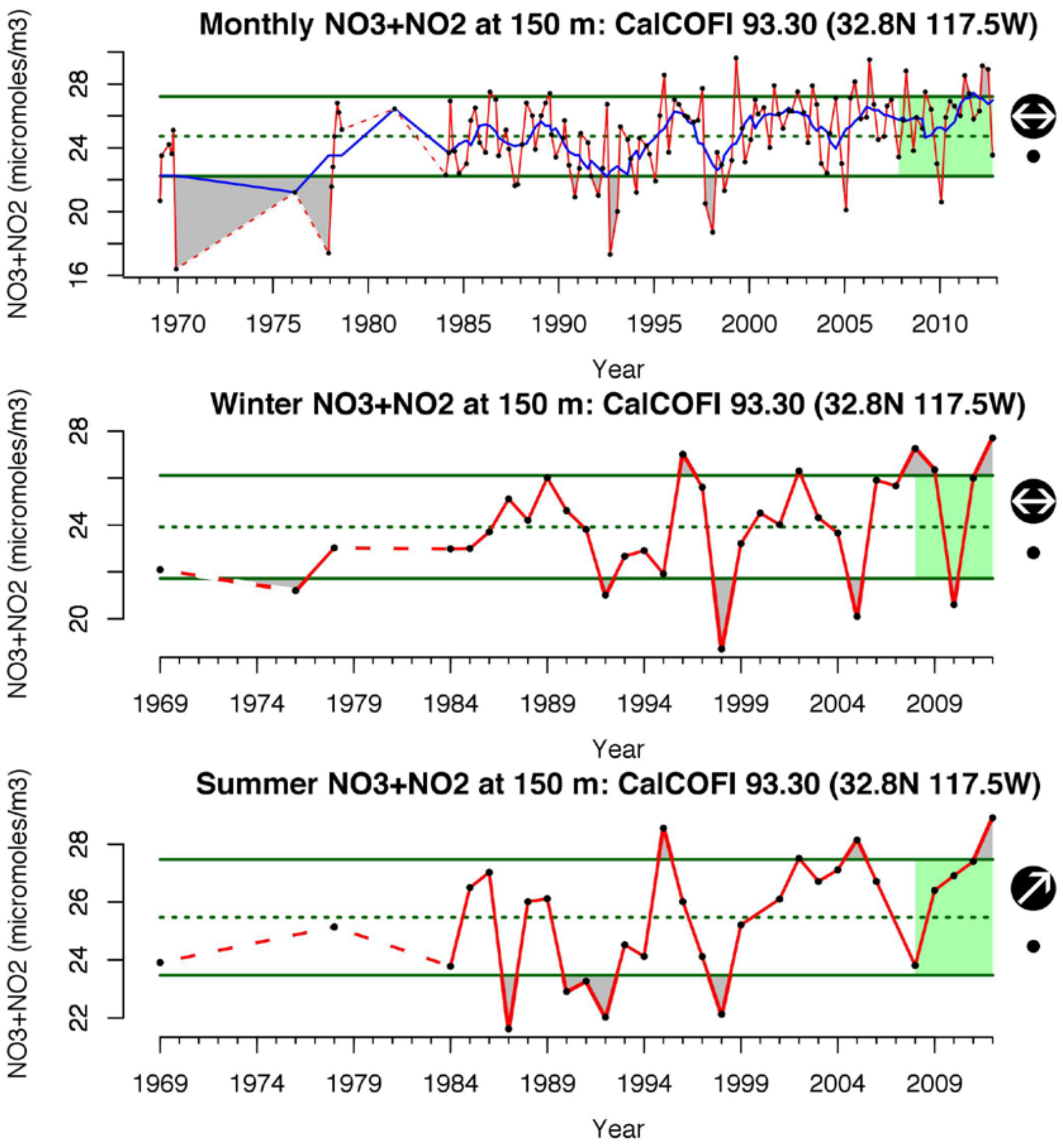


Figure OC 31. Nutrient data (nitrate + nitrite) at 150 m from 1997-2013 at CalCOFI station 93.30 representing southern California for: monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Station 93.30 illustrates patterns in the southern portion of the CCLME. Dashed lines show data gaps of greater than 2 years.

TOTAL COPEPOD BIOMASS AND SPECIES COMPOSITION

Copepod biomass and species composition vary seasonally with the highest biomass occurring in the summer months, when food is most plentiful, and the lowest biomass in the winter months (Figures OC32). Copepods are transported to the Oregon coast, either from the north/northwest or from the west/south. Copepods that arrive from the north are cold–water species with higher lipid stores and result in greater productivity of downstream predators; these are referred to as the northern copepods. Copepods that arrive from the west or south are referred to as the southern copepods and are less rich in lipids. The cold–water group, the northern copepods, usually dominates the Washington/Oregon coastal zooplankton community in summer, whereas the warm–water southern copepods group usually dominates during winter (Peterson and Miller 1977, Peterson and Keister 2003, Peterson and Schwing 2003). However, the northern and southern copepod anomalies track the PDO and MEI fairly closely, thus this seasonal pattern in species composition can be altered during El Niño events or during periods when the PDO is consistently positive or negative. The copepod community index (CCI) is the x-axis score of a 2-dimensional ordination of the copepod species abundance data. The index has a strong seasonal cycle similar to the cyclicity in the monthly total copepod biomass shown below, however when the seasonal cycle is removed (by calculating monthly anomalies) as shown below, the monthly data (red line) track the PDO (compare to Fig OC7. This means that when the PDO is in positive phase, a community dominated by more southern copepods is present (indexed by positive values of the CCI); when the PDO is in negative phase, the copepod community is dominated by more northern species (indexed by negative values of the CCI). Note that during the large El Niño event of 1997-98 and during the summer of 2005 when upwelling was delayed that copepod biomass was low and the CCI was strongly positive. In general, higher abundances of the northern copepods (negative anomalies of the CCI) are indicative of favorable conditions for many upper trophic-level species, including salmon and seabirds. **Both copepod total biomass and community index remained similar to the past several years.**

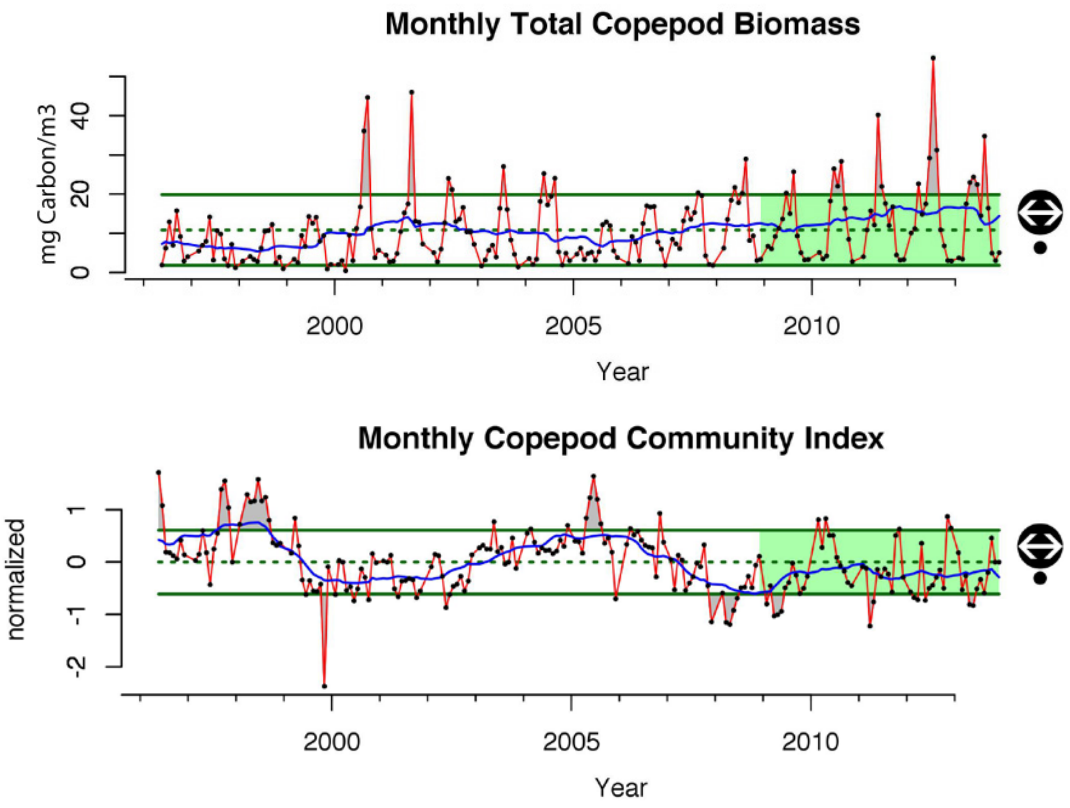


Figure OC 32. Top. total copepod biomass, and bottom. monthly anomaly of the copepod community index from 1996-2013 in the northern California current. The blue line shows a running annual average.

NORTHERN COPEPOD ANOMALY

During the 1997-98 El Niño event, the biomass anomalies of northern copepods was one order of magnitude lower than normal (Figure OC33). With the change in sign of the PDO from positive to negative in mid 1999, the northern copepods responded by showing consistently positive anomalies that prevailed through 2002 (Peterson et al. 2002, Peterson and Keister 2003). In late 2002, the PDO and MEI turned positive (indicating El Niño conditions) and the northern copepods showed negative anomalies. The anomalies were strongly negative during the summer of 2005, a summer characterized by a two-month delay to the start of upwelling (Kosro et al. 2006) and anomalous species composition among the zooplankton (Mackas et al. 2006). Over the past few years, the northern species have predominated with increases in biomass beginning in late 2006. High biomass values were observed for northern species both in 2008 and 2009 with a brief period of negative anomalies during the small El Niño from May 2009 through May 2010. The highest anomalies in the northern copepod biomass time series (since 1996) were observed in March and April 2011 and also the beginning of 2012, coincident with strongly negative PDO values. **For 2013, northern copepod biomass anomaly was again high.**

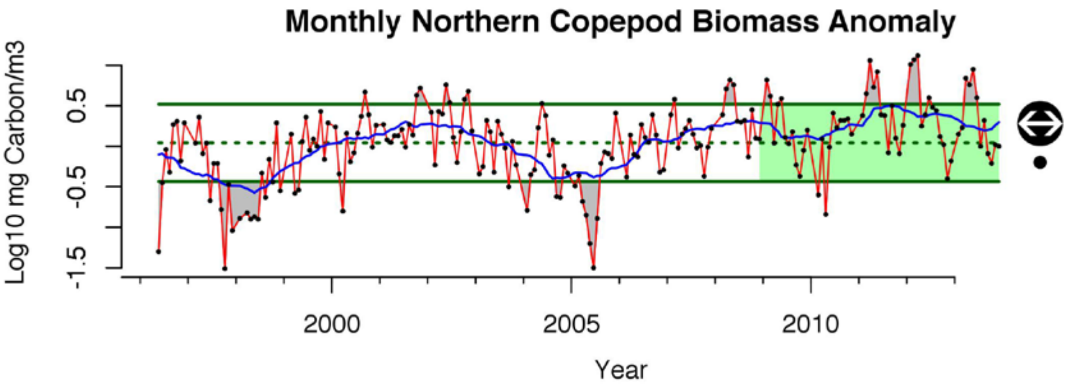


Figure OC 33. Northern Copepod Biomass Anomaly index monthly from 1996-2012 in the northern California current. The blue line shows a running annual average.

SOUTHERN COPEPOD ANOMALY

The highest positive anomalies of the southern species were observed during the 1997-1998 El Niño (Figure OC34). Consistently high positive anomalies of southern species were also observed from 2003 through 2006 coinciding with a period of positive PDO and mostly positive MEI. Over the past few years (since mid-to-late 2009) ocean conditions have been unsettled in that recently there was another small El Niño at the equator. MEI values were positive from May 2009 through May 2010 and the southern copepods responded quite strongly, having anomalies that were similar to those observed during the 1998 and extended (2003-2006) El Niño events. Both the MEI and PDO returned to negative values (signaling a cold ocean) in June 2010 and the southern copepod biomass anomaly became negative in early 2011 following the PDO and MEI sign change by about six months. **For 2013, southern copepod biomass anomaly was similarly low as compared to previous years.**

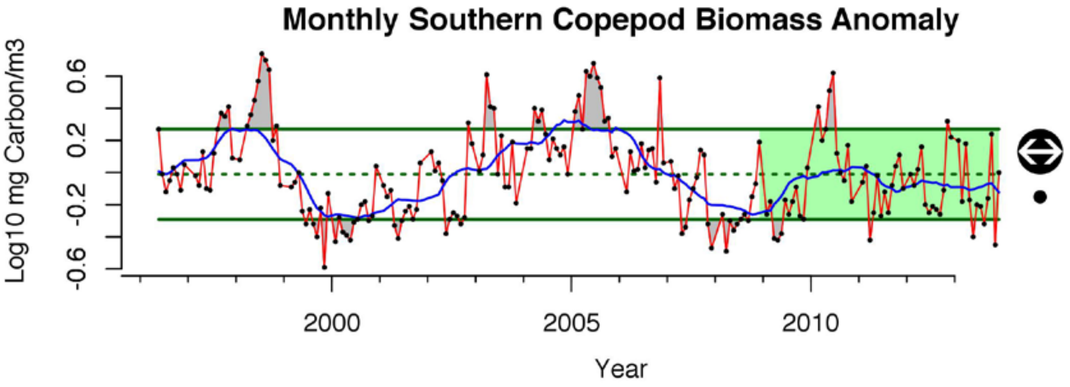


Figure OC 34. Southern Copepod Biomass Anomaly index monthly from 1996-2013 in the northern California current. The blue line shows a running annual average.

OCEAN ACIDIFICATION

BACKGROUND

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

EVALUATION AND SELECTION OF INDICATORS

The saturation state of aragonite and calcite, the pH, and the dissolved oxygen (DO) of waters in the California current all can serve as indicators of ocean acidification. It is likely that synergistic responses among these indicators will be quite difficult to isolate. Although some time series of calcium-carbonate chemistry (e.g. pCO2, pH, alkalinity) have been started, we do not have enough data yet to say anything about status and trends. Because increases in CO_2 occur along with decreases in dissolved oxygen, we can use DO as a proxy for acidification in the California current. DO serves as an indicator of multiple pressures and also has a longer time series available than the other indicators of ocean acidification. Data are obtained from Newport, Oregon's NH Line 25 nautical miles (46 km) offshore, from the central California CALCOFI station 67.55, and from the southern California Bight via CalCOFI's station 93.30 at 22 km offshore.

DISSOLVED OXYGEN AND HYPOXIC EVENTS

BACKGROUND

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

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

EVALUATION AND SELECTION OF INDICATORS

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

STATUS AND TRENDS

The northern CCLME has had increased continental shelf hypoxia and shoaling of the hypoxic boundary resulting from enhanced upwelling, primary production, and respiration over the past 15 years (Pierce et al. 2012). Severe and persistent anoxic events have had downstream effects on both demersal fish and benthic invertebrate communities off Oregon (Keller et al. 2010). For example, during a severe anoxic event in August 2006, surveys found an absence of rockfish on rocky reefs and a large mortality event of macroscopic benthic invertebrates (Chan et al. 2008). Seasonality in oxygen concentrations show oxygenated summer waters along the Newport Hydrographic Line since September 2005 (Figure OC35). In 2007, low oxygen concentrations were observed in the summer although the mean was above 1.4 ml/L. The 2011 data point had lower than average oxygen at NH25. Despite higher than average upwelling in 2008, boundary waters remained well oxygenated save two occasions.

In the southern CCLME (Bograd et al. 2008), deepening of the thermocline and decreased oxygen in deep source waters have resulted in decreased subsurface oxygen through 2007 (Figures OC35-OC37). Contrary to the past five years, large-scale wind forcing models predict hypoxia will continue to expand under Intergovernmental Panel on Climate Change warming scenarios (Rykaczewski and Checkley 2008). **In the north the 2012 and 2013 DO values were slightly below average in the winter and increased to slightly above average in the summer. While in the central and southern regions the 2012 DO values were below average – the winter and summer values in 2012 for CALCOFI station 93.30 are below the long term mean.**

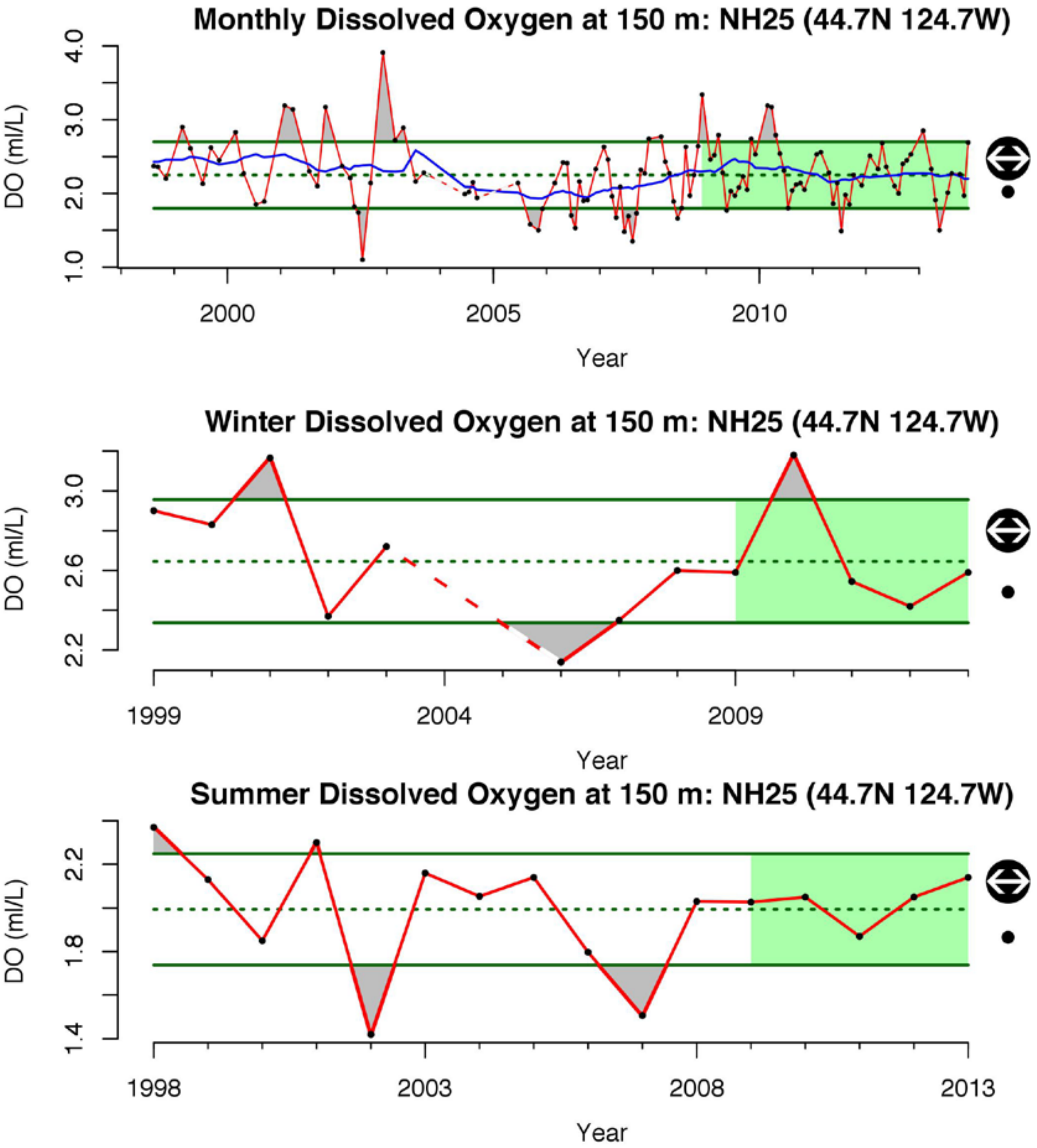


Figure OC 35. Newport line (Newport, Oregon NH25) dissolved oxygen at 150 m from 1999-2013 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Station NH25 illustrates patterns in the northern portion of the CCLME. Dashed lines show data gaps of greater than 2 years.

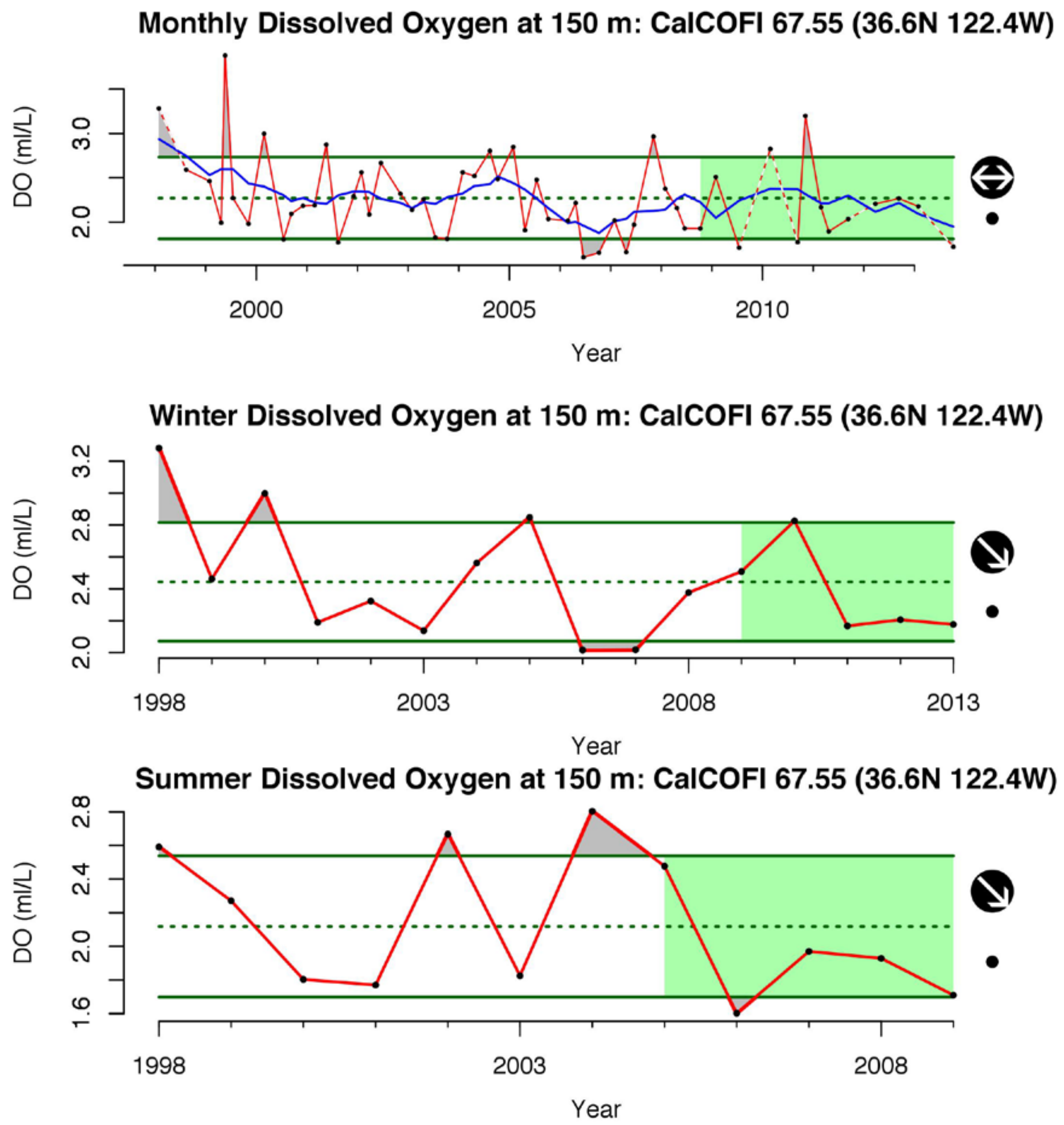


Figure OC 36. CALCOFI station 67.55 dissolved oxygen at 150 m from 1998–2012 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Station 67.55 illustrates patterns in the central portion of the CCLME.

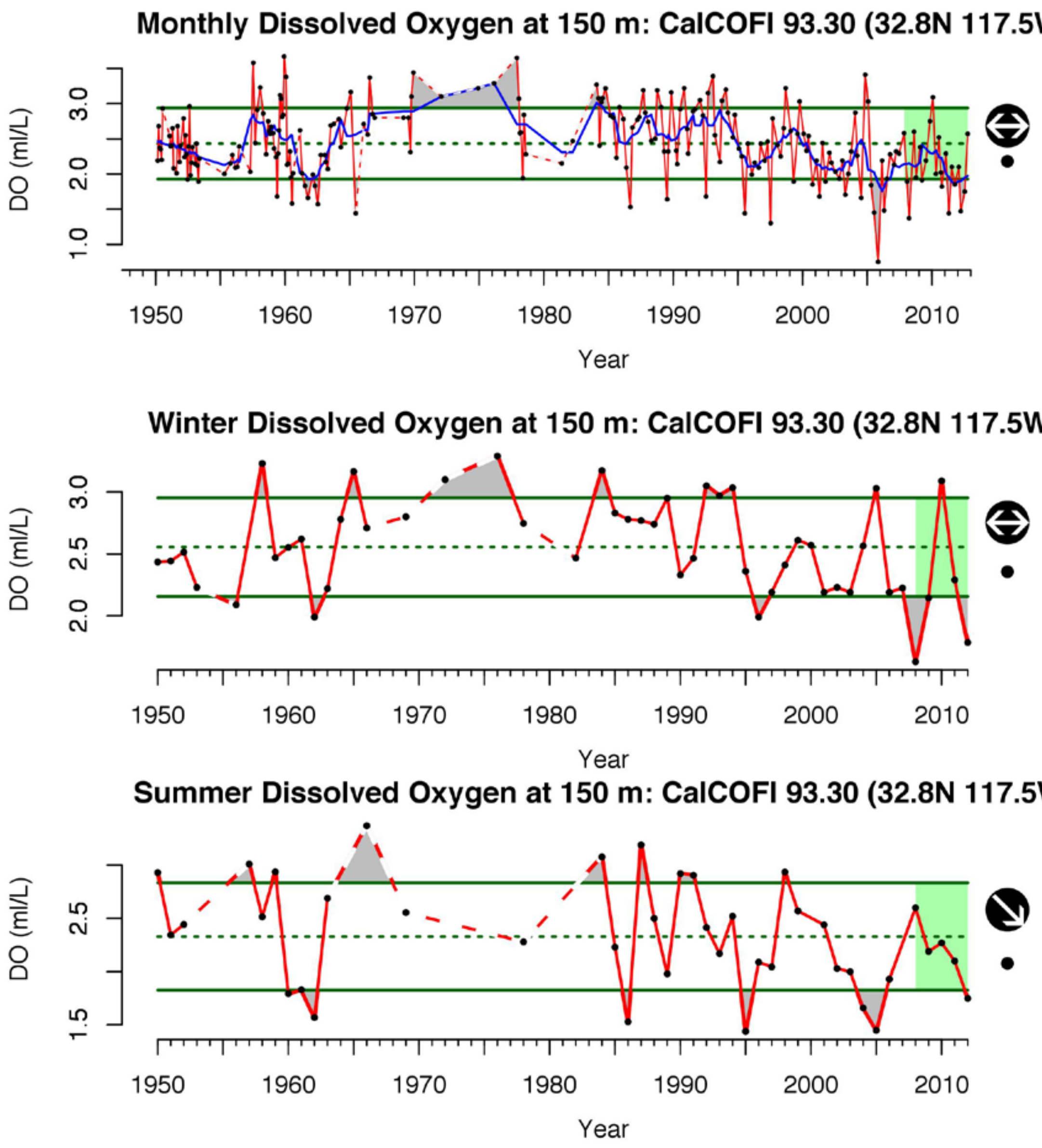


Figure OC 37. CALCOFI station 93.30 dissolved oxygen at 150 m from 1950–2012 for monthly, winter and summer means. Monthly values are included to show seasonal cycles and a continuous time series, and the blue line shows a running annual average. Station 93.30 illustrates patterns in the southern portion of the CCLME. Dashed lines show data gaps of greater than 2 years.

MULTIVARIATE OCEAN CLIMATE INDEX – MOCI

Towards furthering our goal of examining the cross-linkages between climate drivers and ecosystem response, a suite of 35 regional and local-scale anomaly indices were combined and examined using principal component analyses (PCAs) to create a **single** environmental indicator of ecosystem state in the central-northern ecoregion of the California Current (Sydeman et al. 2014). Generally, positive values of the MOCI represent a stronger subtropical influence, with warmer temperatures, higher precipitation and generally weak winds with poor upwelling, while negative values indicate subarctic influence with cooler temperatures, stronger winds, and upwelling events. While MOCI is not well correlated with some variables such as bulk chl-a from remotely sensed datasets, and krill density for shipboard surveys, it is highly correlated with important biological time series such as the northern copepod index (Fig OC38) and the reproductive success of multiple species of seabirds (Sydeman et al. 2014). The seasonal MOCI represents important components of environmental variability in the CCE and could be a useful tool in understanding or potentially forecasting physical-biological interactions in the system.

From 2006-2010, the dominant signals were the strong upwelling and cold SST winters of 2007 and 2008, as well short-duration El Niño conditions quickly followed by La Niña conditions in late 2010. This is indicated by the positive trend in MOCI winter axis 1 towards warm, weaker upwelling conditions and negative trends in MOCI summer axis 1 towards stronger upwelling conditions. The utility of MOCI depends on continued monitoring of environmental variables and re-calculation of MOCI parameters beyond 2010 for future status reports.

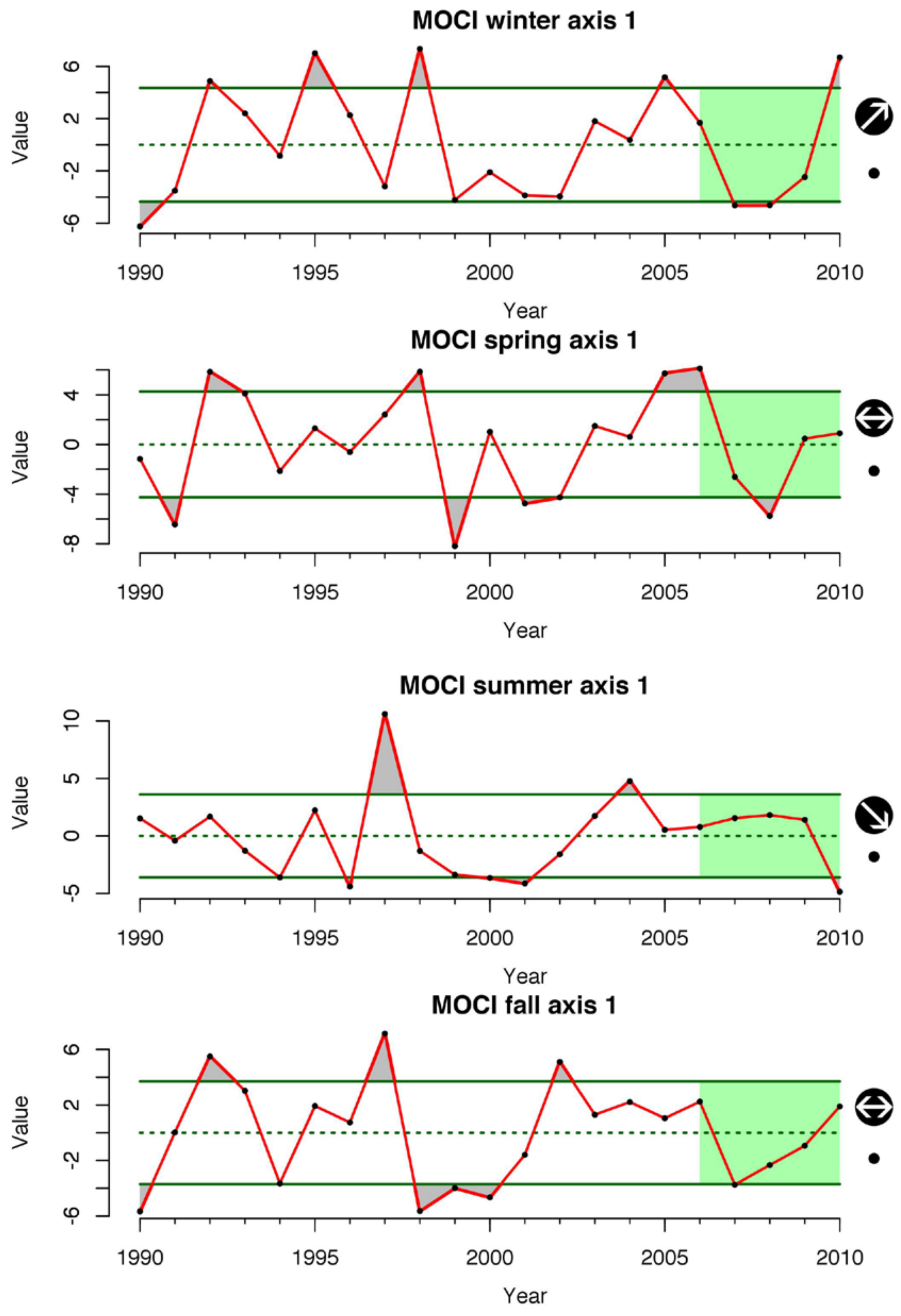


Figure OC 38. Multivariate Ocean Climate Index, axis 1, for winter, spring, summer and fall for the central-northern ecoregion of the CCLME from 1990-2010.

SPATIAL SATELLITE SEA SURFACE TEMPERATURE AND CHLOROPHYLL-A

BACKGROUND

So far in this section, time series from specific locations in the CCLME have been presented to establish the mean and trend over the last 5 years (2009-2013) for different physical processes. Satellite sampling allows for the extension of the IEA's indicator analysis to the whole CCLME. We will focus on satellite measured SST and chlorophyll-a (chl-a) sampled over 30-48°N and 130-116°W as indicators of productivity. The SST product, developed by the NOAA west coast Coastwatch node, is a blend of SST measurements from MODIS, AVHRR, and GOES satellite instruments and has 5-day means from July 2002 to the present. Chl-a data are daily data from July 2002 to the present and are measured by the Aqua MODIS satellite instrument. Monthly averages were used for both SST and chl-a to construct maps of anomalies for 2013, means over the last 5 years, and trends over the last 5 years for winter (Jan-Mar) and summer (Jun-Aug).

STATUS AND TRENDS

SST

The sea surface temperature (SST) for the winter of 2013 (Figure OC39) was cooler than the long-term mean (2002-2013), with SST anomalies of 1 °C or cooler occurring over a wide extent of coastal areas. The mean of the last 5 years (2009-2013) was slightly cooler than the long-term average but displayed no areas when the mean was below 1 standard deviation from the long-term mean. The trend over the last 5 years did display areas when the difference of the start and end dates of the trend was below 1 standard deviation. These areas occurred south of 40°N especially along the coast from 30-34°N. The cause of these negative trends is due to warm SST during the mild El Niño in the winter of 2010 being followed by exceptionally cool temperature in 2013 (see buoy SST above).

The SST anomalies for the summer of 2013 (Figure OC40) are generally warmer for the whole area except for cool areas along the coast in the north (42-45°N) and south (30-34°N). The anomalies for the satellite SST are in slight disagreement to the buoys. This is due to the fact that the summer means of the buoy data for July and August 2013 are missing and hence the buoy data missed a large increase in SST that occurred between July and August of 2013. The mean over the last 5 years was cooler for the whole region but without any grid cells below 1 standard deviation of the long-term mean. The trend over the last 5 years shows mostly positive trends in areas along the coast and areas of negative trends offshore in the south (centered around 32°N 122°W). The positive trends are due to the cool conditions experienced in the summer of 2010 followed by the warmer conditions in the summer of 2013.

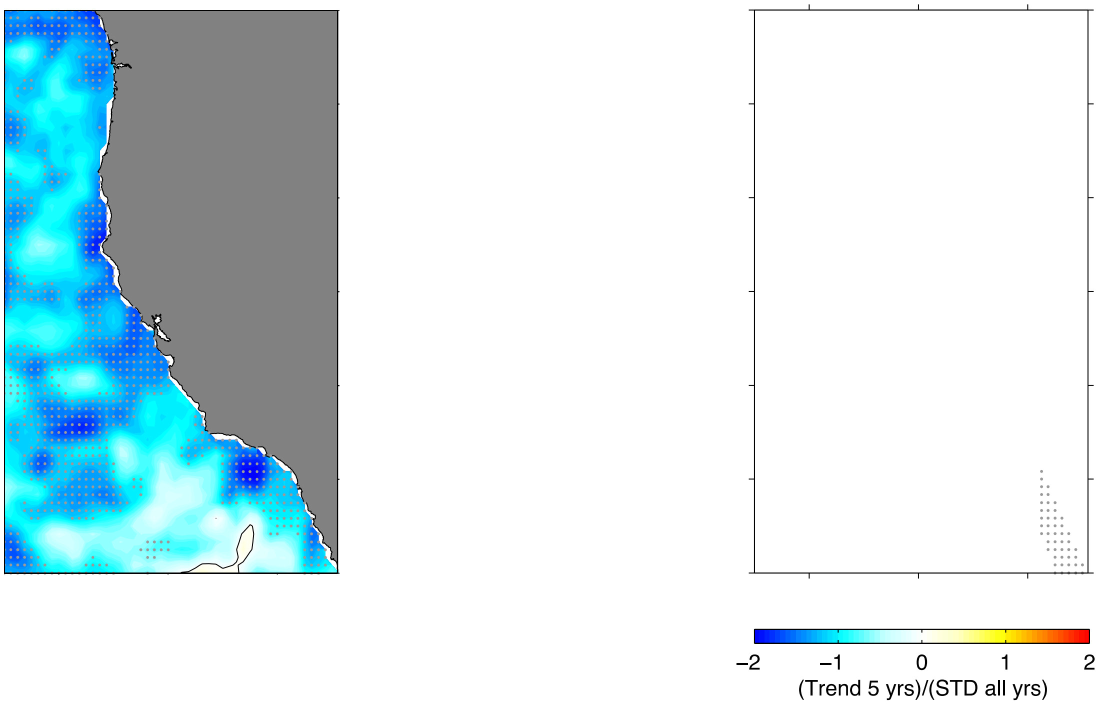


Figure OC 39. Blended satellite SST for: (left) winter 2013 SST anomalies, (middle) 5-year (2009-2013) winter SST means relative to the long-term standard deviation and (right) 5-year (2009-2013) winter SST trends relative to the long-term standard deviation. The value of each grid cell in the mean (center) and trend (right) maps has been normalized by the long-term standard deviation of the winter time series at that grid cell. In the anomaly map the zero contour is drawn in black and a gray dot marks a grid cell where the 2013 anomaly exceeds 1 standard deviation of the long-term mean. The plus/minus 1 contour is drawn in black for the trend map and a gray dot marks any grid location that has a trend exceeding 1 standard deviation from the long-term value.

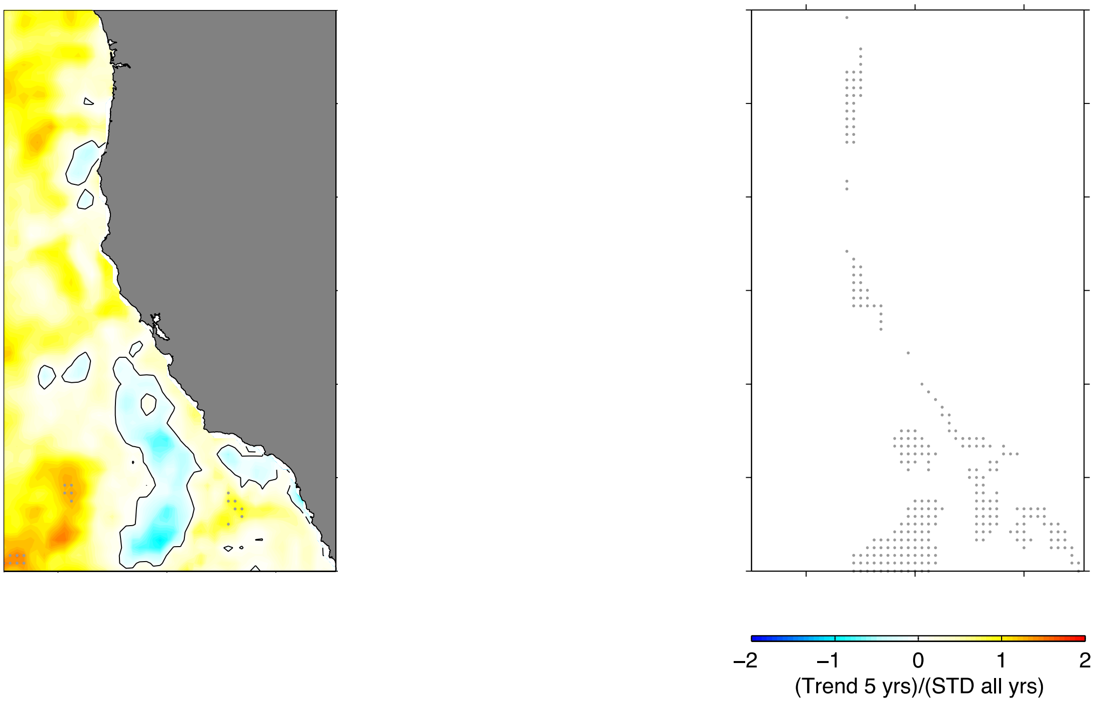


Figure OC 40. Blended satellite SST for: (left) summer 2013 SST anomalies, (middle) 5-year (2009-2013) summer SST means relative to the long-term standard deviation and (right) 5-year (2009-2013) summer SST trends relative to the long-term standard deviation. The value of each grid cell in the mean (center) and trend (right) maps has been normalized by the long-term standard deviation of the summer time series at that grid cell. In the anomaly map the zero contour is drawn in black and a gray dot marks a grid cell where the 2013 anomaly exceeds 1 standard deviation of the long-term mean. The plus/minus 1 contour is drawn in black for the trend map and a gray dot marks any grid location that has a trend exceeding 1 standard deviation from the long-term value.

CHL-A

The winter chl-a anomalies for 2013 (Figure OC41) were exceptionally high for three areas along the coast (47-48°N, 39-42°N, and 36-38°N). The winter mean for the last 5 years generally had more positive than negative values, but the mean did not exceed 1 standard deviation of the long-term mean for any grid cell. The winter trend over the last 5 years had positive trends along the coast and negative trends offshore. The coastal areas when the difference of the start and end dates of the trend was above 1 standard deviation occurred in the same three locations as the positive anomalies for 2013. These positive trends are due to the high chl-a values experienced in the winter of 2013. The offshore negative trends are due to high chl-a experienced during the winter of 2010 being followed by low chl-a values in the winters of 2012 and 2013.

The summer chl-a anomalies for 2013 (Figure OC42) have low anomalies along the coast between 40-47°N and 37-38°N. High anomalies along the coast between 47-48°N persisted in this region from the winter. The summer mean for the last 5 years was mostly below the long-term mean for areas along the coast. Though, none of these means exceeded 1 standard deviation of the long-term mean. Negative trends of the last 5 years were observed along the coast between 43-47°N and 30-33°N, while positive trends were found along the coast in areas of strong upwelling (38-39°N and 35-36°N). The areas of positive summer trends resulted due to low chl-a values in 2009 followed by the high values in 2013.

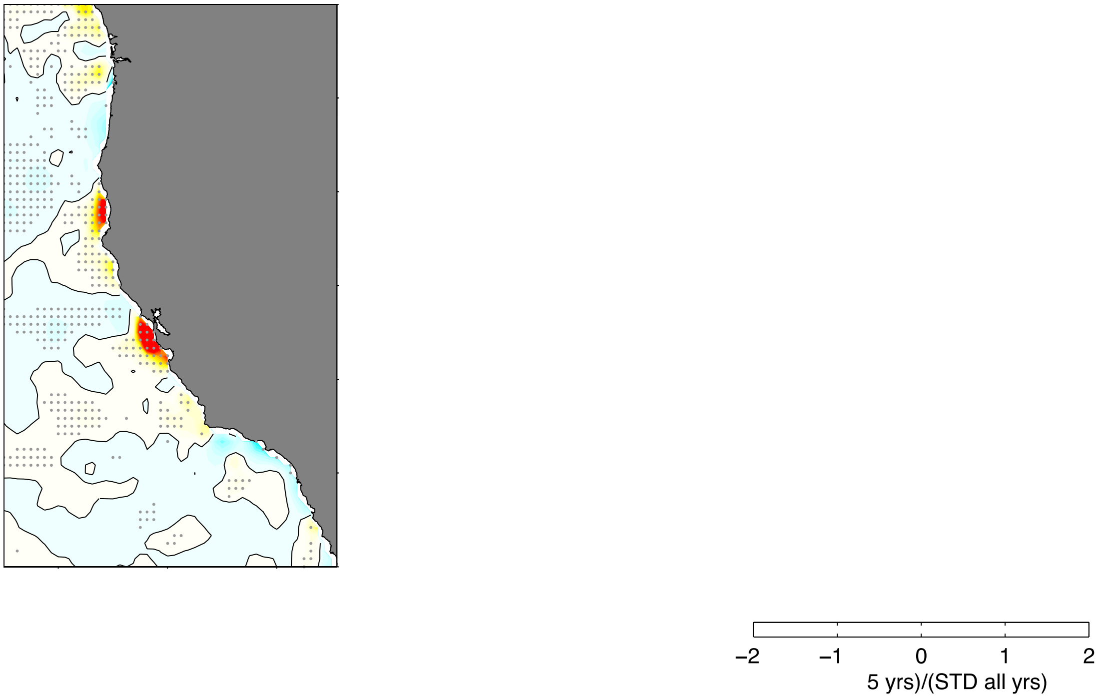


Figure OC 41. Aqua MODIS satellite chl-a for: (left) winter 2013 chl-a anomalies, (middle) 5-year (2009-2013) winter chl-a means relative to the long-term standard deviation and (right) 5-year (2009-2013) winter chl-a trends relative to the long-term standard deviation. The value of each grid cell in the mean (center) and trend (right) maps has been normalized by the long-term standard deviation of the winter time series at that grid cell. In the anomaly map the zero contour is drawn in black and a gray dot marks a grid cell where the 2013 anomaly exceeds 1 standard deviation of the long-term mean. The plus/minus 1 contour is drawn in black for the trend map and a gray dot marks any grid location that has a trend exceeding 1 standard deviation from the long-term value.

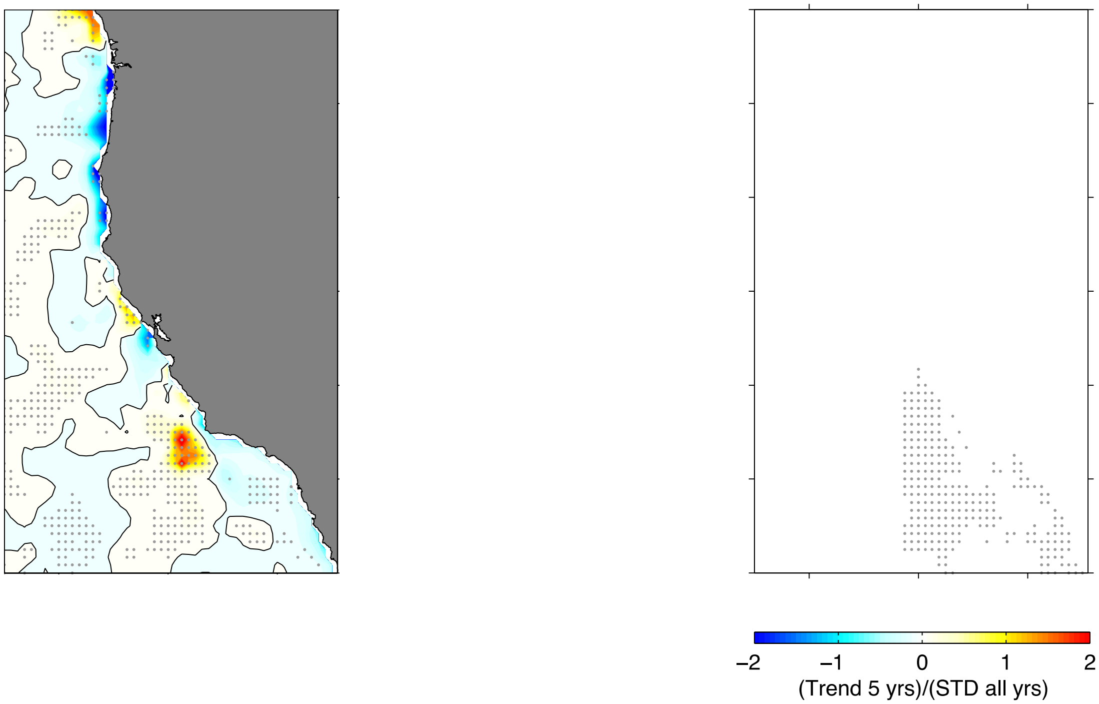


Figure OC 42. Aqua MODIS satellite chl-a for: (left) summer 2013 chl-a anomalies, (middle) 5-year (2009-2013) summer chl-a means relative to the long-term standard deviation and (right) 5-year (2009-2013) summer chl-a trends relative to the long-term standard deviation. The value of each grid cell in the mean (center) and trend (right) maps has been normalized by the long-term standard deviation of the summer time series at that grid cell. In the anomaly map the zero contour is drawn in black and a gray dot marks a grid cell where the 2013 anomaly exceeds 1 standard deviation of the long-term mean. The plus/minus 1 contour is drawn in black for the trend map and a gray dot marks any grid location that has a trend exceeding 1 standard deviation from the long-term value.

EFFECTS OF ANTHROPOGENIC CLIMATE CHANGE

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

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

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

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

LINKS TO DATA, AS APPROPRIATE

Table OC 1. Top indicators for ocean and climatic pressures. Three stations were chosen when possible for northern, central, and southern portions of the California current. Time series availability often differed across the three locations.

Pressure	Indicator	Definition and source of data	Time series	Sampling frequency
Ocean acidification	DO	Newport line station NH25 and at 150 meters deep as representative of the northern CCLME.	1998 - 2011	monthly
		CALCOFI station 93.30 at 150 meters as representative of the southern CCLME.	1984 - 2012	quarterly
		CALCOFI station 67.55 at 150 meters as representative of the central CCLME.	1998 - 2011	quarterly
Decreasing oxygen	DO	Newport line station NH25 and at 150 meters as representative of the northern CCLME.	1998 - 2013	monthly
		CALCOFI station 93.30 at 150 meters as representative of the southern CCLME.	1984 - 2012	quarterly
		CALCOFI station 67.55 at 150 meters as representative of the central CCLME.	1998 - 2013	quarterly
Sea level rise	Coastal Sea Level	Sea Level measured by tide gauges at South Beach, OR.	1967 – 2013	daily
		Sea Level measured by tide gauges at San Francisco, CA.	1897 – 2013	daily
		Sea Level measured by tide gauges at San Diego, CA. Data were obtained from Uhawaii Sea Level Center: http://uhslc.soest.hawaii.edu/home	1906 - 2013	daily
Temperature change	Sea surface buoy temperatures	Sea surface temperatures measured by NDBC buoy 46050 (44.639° N 124.534° W; 37 km from land).	1991 - 2013	hourly
		Sea surface temperatures measured by NDBC buoy 46014 (39.235° N 123.974° W; 17 km from land).	1981 - 2012	hourly
		Sea surface temperatures measured by NDBC buoy 46025 (22.749° N 119.053° W; 40 km from land). Data are available at National Data Buoy Center: http://ndbc.noaa.gov	1982 - 2013	hourly
	PDO	Pacific Decadal Oscillation (PDO) is the dominant pattern of North Pacific SST anomalies. Data are available at: http://jisao.washington.edu/pdo/	1900 - 2013	monthly
	NOI	Northern Oscillation Index (NOI) measures atmospheric teleconnections between North Pacific High and northeast Pacific. Data are available at: http://coastwatch.pfeg.noaa.gov/erddap/index.html	1967 - 2013	monthly

	MEI	Multivariate ENSO Index (MEI) reports on the status of the coupled ocean-atmophere ENSO events. Data are available at: http://www.esrl.noaa.gov/psd/enso/mei/mei.html	1950 - 2013	monthly
	Satellite sea surface temperatures	Blend of SST measurements from MODIS, AVHRR, and GOES satellite instruments. Dara are available at: http://coastwatch.pfeg.noaa.gov/erddap/griddap/erdBAssta5day.graph	2002-2013	5-day means
Water column structure	Pycnocline depth	Three stations, Newport NH25, CALCOFI 93.30, and 67.55 were used for water column structure.	1984 - 2013	quarterly
	Pycnocline strength	Three stations, Newport NH25, CALCOFI 93.30, and 67.55 were used for water column structure.	1984 - 2013	quarterly
Changes in source waters	Nutrient content (NO2+NO3)	Three stations, Newport NH25, CALCOFI 93.30, and 67.55 were used for water column structure. Nitrate+nitrate concentrations at 150 m show variations in source water.	1984 - 2013	quarterly
	Zooplankton community structure	Newport line data are compiled into four indices (total biomass, northern anomaly, southern anomaly, and copepod index).	1998 - 2013	monthly
	NPGO	North Pacific Gyre Oscillation (NPGO) explains variations in the circulation of the North Pacific Gyre. The NPGO describes nutrient concentrations in the CCS. Data are available at: http://www.o3d.org/npgo/	1950 - 2013	monthly
Changes in CC transport & mesoscale variability	EKE	Eddy Kintectic Energy (EKE) was calculated over three spatial locations (6 degree mean), at 33°, 39° and 45° N. Meridional and zonal geostrogphic velocities used in the EKE calculations are distributed by Aviso at: http://www.aviso.oceanobs.com/duacs/	1992 - 2013	daily
Timing and strength of upwelling	meridional	North winds are drive much of coastal upwelling and are measured by NDBC buoy 46050 (44.639° N 124.534° W; 37 km from land)	1991 - 2013	hourly
		North winds are measured by NDBC buoy 46014 (39.235° N 123.974° W; 17 km from land)	1981 - 2013	hourly
		North winds are measured by NDBC buoy 46025 (22.749° N 119.053° W; 40 km from land). Data are available at National Data Buoy Center: http://ndbc.noaa.gov	1982 - 2013	hourly
	UI	Upwelling Index (UI) denote the strength of coastal upwelling and downwelling; data are presented at 33°, 39° and 45° N.	1967-2013	daily
	STI	Spring Transition Index (STI) denotes the start of the upwelling season .It is derived from the daily UI and data are presented at 33°, 39° and 45° N.	1967-2013	yearly
	TUMI	Total Upwelling Magnitude Index (TUMI) is the amount of upwelling between the spring and fall transition dates. It is derived from the daily UI at 33°, 39° and 45° N.	1967-2013	yearly

	LUSI	Length of Upwelling Index (LUSI) is the number of days during the upwelling season. It is derived from the daily UI at 33°, 39° and 45° N. Data are available at: http://www.pfeg.noaa.gov/products/las.html	1967-2013	yearly
Timing and frequency of El Niño events	MEI	Multivariate ENSO Index (MEI) measures the magnitude and duration of El Niño and La Niña events. Data are available at: http://www.esrl.noaa.gov/psd/enso/mei/	1950 - 2012	monthly
	NOI	Northern Oscillation Index (NOI) measures atmospheric teleconnections between North Pacific High and northeast Pacific. Large negative values usually occur during an El Niño event. Data are available at http://coastwatch.pfeg.noaa.gov/erddap/index.html	1950 - 2012	monthly
Productivity	MOCI	Multivariate Ocean Climate Index (MOCI) is a PCA of 35 separate regional and local indicators of ecosystem status for the central-northern ecoregion of the California Current.	1990-2010	yearly
Primary Productivity	CHL-A	Chlorophyll-a measured by the Aqua MODIS satellite data available at: http://coastwatch.pfeg.noaa.gov/erddap/griddap/erdMHchla1day.graph	2002-2013	daily

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ECOLOGICAL INTEGRITY

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OVERVIEW

Indicators of Ecological Integrity suggest neutral to good feeding conditions (trophic structure) in the California Current through 2013, although low mean trophic level for groundfishes merits watching. Indicators of biodiversity were neutral or mixed in the short-term. All were within long-term norms of the respective time series..

EXECUTIVE SUMMARY

Ecological integrity is "the ability of an ecological system to support and maintain a community of organisms that has a species composition, diversity, and functional organization comparable to those of natural habitats within a region" (Parrish et al. 2003). We identified and evaluated potential indicators of ecological integrity across a variety of species and foraging guilds, using the ecological literature as a basis for their rankings. We selected the mostly highly ranked indicators to track two aspects of the California Current Large Marine Ecosystem (CCLME):

- **Trophic structure**: mean trophic level, scavenger biomass ratio, biomass of gelatinous zooplankton, and the northern copepod biomass anomaly
- **Biodiversity:** Simpson's diversity, species richness or species number for multiple taxa.

The indicators reported in this section are designed to be integrative, community-based measures that draw information from across the taxonomic spectrum. Indicators derive from monitoring time series through recent years (2011-2013, depending on the time series; see Table EI 1). Indicators specific to individual ecological components, such as coastal pelagic species, groundfishes, and protected species (marine mammals, seabirds, and Pacific salmon), also provide information that can influence ecological integrity and are covered in other sections in this report.

The spatial extent of CCLME data coverage varies among taxa. The groundfish data span the U.S. West Coast (~32 to 48 ^{o}N, ~50-1200 m depths) and conclusions related to this dataset (mean trophic level, scavenger biomass, species richness, species density, and Simpson diversity) are applicable to the full CCLME. Note, however, that the trawl survey does not adequately sample complex, rocky habitats and any conclusions are limited to trawlable areas. Data for ichthyoplankton are drawn from southern California and Oregon survey transect lines, while those for gelatinous zooplankton are taken from surveys conducted off central California and the Oregon/Washington coasts. Data for coastal pelagic fishes are also drawn from the Oregon/Washington survey, whereas the copepod data are limited to survey stations in waters off of central Oregon. Thresholds and targets are not currently set for indicators of ecological integrity, and time series are evaluated based on internal statistical properties (detailed below).

TROPHIC STRUCTURE

Indicators of **trophic structure** suggest neutral to good conditions in the CCLME in recent years (Figure EI 1). All indicators were within long-term norms (±1.0 s.d. of the long-term mean), although groundfish MTL was relatively low coastwide and showed a decrease south of Cape Mendocino.

High abundance of gelatinous zooplankton is generally considered a sign of poor conditions because they clog fishing nets, prey on fish larvae and compete with forage fishes for food. Abundances of gelatinous zooplankton (*Aequorea, Aurelia* and *Chrysaora*) were near long-term averages with *Chrysaora* decreasing in abundance in the short-term off of Oregon. However, *Aequorea* increased in June surveys off of Oregon in the short-term. September values showed no trend.

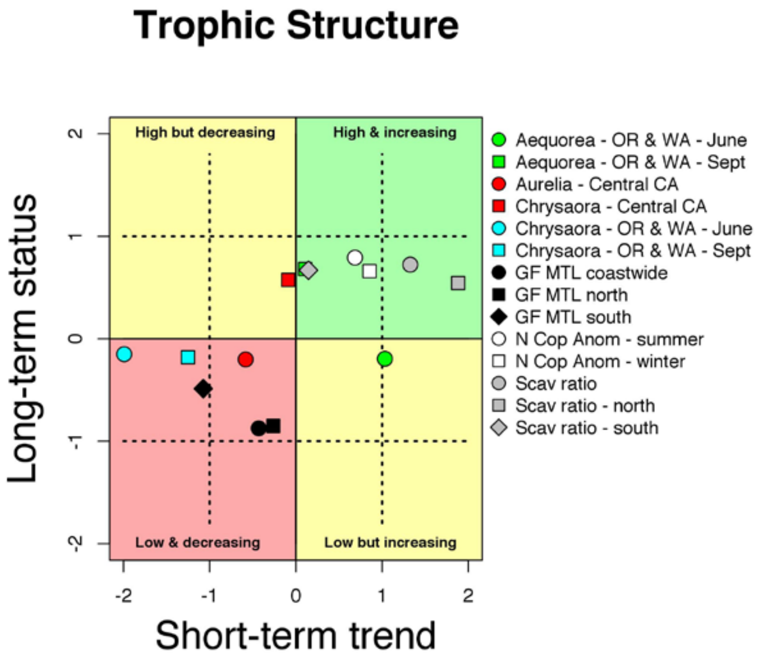

Figure EI 1. Short and long-term status of indicators of Trophic Structure for the Ecological Integrity for the California Current Large Marine Ecosystem. Prior to plotting, time series were normalized to place them on the same scale. The short-term trend indicates whether the indicator increased or decreased over the last 5 years of the time series relative to the long-term mean. The long-term axis compares the mean of the last 5 years to the mean of the full time series. Changes or differences of more than 1.0 standard deviation of the full time series (dotted lines) are considered to show an effect. GF MTL = groundfish mean trophic level, N Cop Anom = northern copeopod anomaly, Scav ratio = ratio of scavengers:total biomass, CA = California, OR = Oregon, WA = Washington. *Aequorea, Aurelia and Chrysoara* are gelatinous zooplankton (jellies). For GF MTL and Scav kg, north and south indicate north and south of Cape Mendocino.

The northern copepod anomaly was relatively high but within long-term historical norms. High abundance of energetically rich northern copepod species generally indicates good feeding conditions for many species.

The ratio of scavenger biomass to total biomass for groundfishes and crabs increased in the short-term coast-wide and in water north of Cape Mendocino. The increase appears to have been caused by an increase in crab biomass.

Groundfish mean trophic level (MTL) declined south of Cape Mendocino. While stable overall and north of Cape Mendocino, current MTL was near relatively low but still within 1.0 s.d. of the long-term mean. Low MTL is generally considered an indication of reduced abundance of top predators, and therefore, top-down forcing in the system. However, low groundfish MTL may make food resources (forage fishes and krill) available to groundfish competitors like salmon, seabirds and tuna, indicating good feeding conditions for these species. Previous work has shown that the decline in MTL was caused by a decrease in the abundance of Pacific hake and dogfish.

BIODIVERSITY

Biodiversity indicators showed mixed results (Figure EI 2). No indicators showed changes relative to their long-term trends. Six diversity measures increased in the short-term, while seven decreased. Simpson diversity (~evenness, technically equitability) increased in the short-term for groundfishes coast wide. This rise was driven by changes north of Cape Mendocino. Simpson diversity south of Mendocino did not increase. All measures of ichthyoplankton biodiversity increased in the short-term for the California Current. Conversely, ichthyoplankton spring Simpson diversity and summer species richness both declined in Oregon suggesting different trends in northern and southern regions. Groundfish species richness declined coastwide as did species richness south of Mendocino. North of Mendocino richness also declined but by less than the threshold value. In all three cases, richness was within long-term norms but above the long-term mean. Earlier declines in MTL were caused by loss of Pacific hake and spiny dogfish biomass, and the increase in Simpson diversity may be linked to these trends.

Copepod biodiversity in the summer declined as did species number for coastal pelagic fishes. Seven diversity indicators decreased in the short-term. While decreased diversity is typically considered a negative indication of ecosystem status, low copepod diversity is linked to high abundance of northern, energy-rich species and indicates overall good feeding conditions for species like forage fishes and salmon.

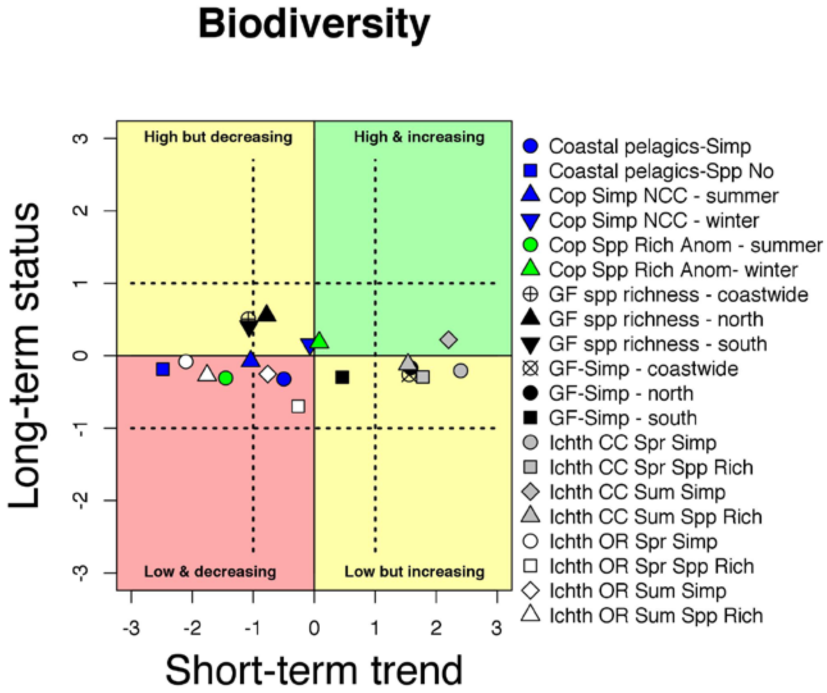

Figure EI 2. Short and long-term status of indicators of Biodiversity for the Ecological Integrity for the California Current Large Marine Ecosystem. Prior to plotting, time series were normalized to place them on the same scale. The short-term trend indicates whether the indicator increased or decreased over the last 5 years of the time series relative to the long-term mean. The long-term axis compares the mean of the last 5 years to the mean of the full time series. Changes or differences of more than 1.0 standard deviation of the full time series (dotted lines) are considered to show an effect. Cop = copepod, GF = groundfishes, Ichth = ichthyoplankton, Simp = Simpson diversity, Spp No = species number (not rarefied), Spp Rich = species richness (rarefied), Anom = anomaly, CC = CalCOFI (southern California Current), OR = Oregon, NCC = northern California Current.

DETAILED REPORT

BACKGROUND - ECOLOGICAL INTEGRITY

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

There are numerous publications that cite indicators of ecosystem health or ecological integrity in marine systems. As the basis for the initial indicator selection effort (Levin et al. 2011), we relied on several core references from the literature to develop a list of potential indicators (Jennings and Kaiser 1998, Link et al. 2002, Rochet and Trenkel 2003, Fulton et al. 2005, Jennings 2005, Jennings and Dulvy 2005, Link 2005, Shin et al. 2005, Samhouri et al. 2009, Sydeman and Thompson 2010). In many cases, authors chose indicators identified in the literature based on expert opinion or the context of the researchers' expertise. For example, many reviews of marine ecosystem indicators are put into the context of fisheries (e.g., Fulton et al. 2005, Link 2005) and ask the question: Which indicators reflect changes in the population as a result of fishing pressure?

INDICATOR SELECTION PROCESS

EVALUATION OF POTENTIAL INDICATORS

The process for selecting indicators of ecological integrity began in 2010 during the initial CCIEA and continued into 2012, using a standardized framework grounded in work developed by Kershner et al. (2011). For specific details related to the methods used in the CCIEA indicator selection process, see Levin et al. (2011) and Williams et al. (2013). In this version of the IEA, we expand the final suite of indicators by one to a total of five (5) based on recent work by Samhouri et al. (2014), who conducted a supplemental evaluation focused on those indicators derived specifically from coastal pelagic data sets. The goal of the Samhouri et al. (2014) study was to complement and balance the previous suite of indicators, which was considered to be heavily reliant on benthic-dwelling taxa of trophic level >3.0. Gelatinous zooplankton (jellyfish) biomass emerged as a promising indicator from this analysis

FINAL SUITE OF INDICATORS

Recent analyses have shown that a single indicator is not sufficient to provide a complete picture of ecosystem state (Fulton et al. 2005); conversely, too many indicators can lead to too many conflicting signals, which may lead to indecision. Therefore, we ranked the evaluation scores of all indicators for the ecological integrity goal and selected five of those ranked in the top quartile. Below, we list the five-indicator portfolio chosen to represent the ecological integrity of the California Current ecosystem during 2013:

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

What follows are brief descriptions of the five (5) top-ranked indicators composing our ecosystem integrity portfolio, generally organized under the larger ecosystem concepts of biodiversity and trophic structure.

BIODIVERSITY

Species diversity is an integrative measure that encompasses species richness - the number of species in the ecosystem, and species evenness - how individuals or biomass are distributed among species within the ecosystem (Pimm 1984). Diversity has remained a central theme in ecology and is frequently seen as an indicator of the wellbeing of ecological systems (Magurran 1988).

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

Diversity indices can be used with a variety of existing survey data, including: groundfish trawl surveys (Weinberg et al. 2002, Keller et al. 2010), pelagic or midwater trawl surveys (Brodeur et al. 2003, Sakuma et al. 2006), reef fish surveys conducted by trained divers (REEF 2008), zooplankton surveys (e.g., NWFSC Newport Line, CalCOFI), invertebrates from benthic grabs conducted by the EPA[1], and a variety of seabird and marine mammal surveys (Ainley et al. 1995, Barlow and Forney 2007, Carretta et al. 2007, McClatchie et al. 2009). Other possible data sources include intertidal invertebrate surveys from 2002 to 2010 (PISCO[2]) and datasets available at smaller spatial and temporal scales (e.g., National Park Service kelp forest monitoring program in the Channel Islands). Many of these data would need to be combined to investigate trends over time across the entire scale of the CCLME.

SIMPSON'S INDEX

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

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[1] http://www.epa.gov/emap/index.html

[2] http://www.piscoweb.org/

SPECIES RICHNESS

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

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

TROPHIC STRUCTURE

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

ZOOPLANKTON SPECIES BIOMASS ANOMALY (NORTHERN COPEPOD BIOMASS)

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

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

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

MEAN TROPHIC LEVEL

Mean trophic level (MTL) is the biomass-weighted average trophic level of all species in an ecosystem. Mean trophic level provides a synoptic view of the organization of trophic structure in marine ecosystems, and is a pervasive and heavily discussed indicator used to measure marine ecosystem status, especially in communities dominated by exploited species (Pauly and Watson 2005, Essington et al. 2006, Branch et al. 2010). Conceptually, MTL is linked to top-down control and trophic cascades; a decline in MTL represents a decrease in the ability of predators to 'control' prey populations and may have far-reaching consequences to ecological communities (Daskalov 2002, Estes et al. 2004, Pauly and Watson 2005, Baum and Worm 2009). Theoretical modeling results have been used to show that mean trophic level can be a good univariate indicator of fishing effects on an ecosystem, although it may be sensitive to data quality (e.g., landings v. survey data) (Fulton et al. 2005, Samhouri et al. 2009). Trends in 'catch' MTL, estimated from fisheries landings and other fishery-dependent data sources may not provide a good indicator of actual changes in the ecosystem. Instead, 'ecosystem' MTL, estimated from data sources like fisheries-independent surveys, is indicative of current ecosystem status especially when coupled with an exploration of the processes responsible for such patterns (Branch et al. 2010, Tolimieri et al. 2013).

A decrease in MTL is generally considered to be undesirable, as it may represent a loss of high trophic level predators, which are often the target of intensive fisheries (Pauly and Watson 2005, Essington et al. 2006, Branch et al. 2010). However, the causes and consequences of changes in MTL are complex. A decrease can be the result of a loss of top predators or an increase in the abundance of lower trophic level species. Regardless, a drop in MTL indicates a change in trophic structure and probable decrease in the strength of top-down control. While a decrease in MTL may indicate an undesirable trend for the taxa in question (e.g., groundfishes), the effect on other species can be positive if those are competitors or prey of the high TL species. For example, modeling work has shown that a decrease in groundfish MTL should correlate with positive responses in competitors like crabs, squid, salmon, tuna and seabirds (Tolimieri et al. 2013). Thus, determining the 'desired status' for MTL may include trade-offs between multiple taxonomic groups.

Mean trophic level is an operationally simple, concrete, numerical indicator, calculated each year using the simple mean of biomass-weighted trophic levels within an ecosystem. Trophic levels can be estimated for species worldwide from Fishbase[3], an online database; species biomass can be obtained from historical, annual estimates derived from standardized surveys throughout the California Current (various groundfish, zooplankton, marine mammal, and seabird surveys, etc.). These surveys are generally continuous, have broad spatial coverage, and are designed with appropriate power to have a high signal-to-noise ratio for most species. The spatiotemporal variation in these time series is becoming increasingly understood as more data are collected each year.

Although included here as a measure of trophic structure, mean trophic level is included in the list of provisional indicators for assessing progress toward the 2010 biodiversity target, proposed by the Convention of Biological Diversity[4]. As such, it is understood by the public and policymakers, considered internationally compatible, and demonstrates a relatively recent history of reporting (Pauly and Watson 2005, Stergiou and Tsikliras 2011). Mean trophic level can be estimated in a cost-effective manner using existing survey data (various groundfish Stock Assessments, REEF.org, etc.).

RATIO OF SCAVENGER BIOMASS TO TOTAL BIOMASS

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

When evaluating this indicator, we used the definition of scavenger from the Atlantis ecosystem models for the California Current (Brand et al. 2007, Horne et al. 2010). Further detail was taken from Yeh and Drazen (2011) who used baited-cameras to evaluate scavenger ecology on the California slope, and from Buckley et al. (1999) who examine food habits of several groundfishes. Detectable changes in the community composition may be a result of changes in various foraging guilds, but a change (or no change) in a single guild may not be indicative of the ecosystem as a whole. Fisheries-based reference points include B40 (target level where production is predicted to be greatest) and B25 (overfished). These single-species reference points could be adapted and used for foraging guilds such as scavengers. Alternatively, Link (2005) describes a framework of reference points that could be applied to most any indicator.

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[3] http://www.fishbase.org/search.php)

[4] http://www.cbd.int/

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

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

Using raw biomass (kg per km^{-2}) would not separate an increase in scavenger biomass from an increase in the biomass of all species. Instead we use the ratio of scavenger biomass to total biomass in the trawl survey to test for a change in the trophic structure because it indicates whether scavengers are more or less prevalent in the assemblage than in previous years.

GELATINOUS ZOOPLANKTON BIOMASS (JELLIES)

Gelatinous zooplankton (jellyfish) are a lower trophic level, high-productivity functional group with important effects on ecosystem trophic structure. High abundance of jellies may 'interfere' with the transfer of biomass, nutrients and energy from zooplankton up the food web to taxa important to human activities (fishes, squids, birds and marine mammals). In the Northern California Current (NCC), early stages of euphausiids, gelatinous taxa, and cladocerans are particularly vulnerable to predation by jellyfish (Suchman et al. 2008). Gelatinous zooplankton are increasingly thought to be a keystone group in some systems (Pauly et al. 2009). The abundance of gelatinous zooplankton has been linked to overfishing, eutrophication, habitat modification (shoreline armoring), climate change and several other human activities (Purcell et al. 2007, Pauly et al. 2009, Richardson et al. 2009, Purcell 2012).

Jellyfish populations can grow quickly in response to abundant prey, producing jellyfish "blooms." Because of fast growth rates and one-year life cycles, gelatinous zooplankton respond quickly to variability in local or regional environmental conditions, but general abundance patterns and the mechanisms responsible for those patterns have been difficult to discern (Suchman et al. 2012). The highest catches of medusae in the NCC appear correlated with cool spring–summer conditions, or negative anomalies of the Pacific Decadal Oscillation, and low winter–summer runoff from the Columbia River (Brodeur et al. 2008, Suchman et al. 2012). Recent publications suggest they have increased in abundance throughout world, and human problems with jellyfish have increased and have captured public attention (Purcell et al. 2007, Richardson et al. 2009). However, Condon (2013) suggests there is no robust evidence for a long-term, global increase in jellyfish (ostensibly due to global warming); rather, jellyfish populations undergo larger, worldwide oscillations with an approximate 20-year periodicity.

Jellyfish biomass can be a sensitive indicator of changing ecosystem status (Richardson et al. 2009). Jellyfish biomass served as the best proxy for ecosystem attributes related to community energetics using seven food web models from the North Pacific and the Baltic Sea (Samhouri et al. 2009). Increases in jellyfish are generally associated with negative impacts on ecosystem attributes (Pauly et al. 2009, Ruzicka et al. 2012). There are also numerous negative effects on humans including interference with tourism (stinging swimmers), fishing (clogging nets), aquaculture (killing fish in net-pens), and power plants (clogging cooling-water intake screens) (Purcell et al. 2007). Median biovolume of gelatinous zooplankton has been included in suites of indicators used for decision criteria (Link 2005).

INDICATOR DATA SOURCES

The data sources we propose for these indicators, including extent of time series and sampling frequency, are documented in Table EN1. The indicators we selected integrate a variety of time series from among several components of the ecosystem (i.e., pelagic and demersal communities). For the *diversity indicator*, the relative coverage of the ecosystem is obviously driven by the time series used. The *copepod biomass anomaly* indicator focuses on a single, critical component known to form the foundation of the ocean food web, linking oceanographic conditions and primary production to upper trophic levels. *Scavenger biomass* is a benthic/demersal indicator of trophic structure, which has been shown to respond to various fishing activities; it also serves to integrate data on crustacean populations, which can be responsive to top-down effects in the food web and predatory finfish populations. At this point in time, the *mean trophic level* indicator is focused on the demersal community associated with the West Coast groundfish trawl time series; future iterations will integrate other trophic levels and communities (e.g., seabirds and marine mammals). Finally, *standardized gelatinous zooplankton biomass* represents a pelagic, lower trophic level, high-productivity functional group that shows relatively strong correlations with at least half of the ecosystem attributes in a food-web modeling exercise that evaluated the performance of candidate indicators of ecosystem structure and function (Samhouri et al. 2009).

Table EI 1 Top-ranked indicators for Ecosystem Integrity and corresponding data time series.

Attribute / Guild	Indicator	Definition and source of data	Region (State)	Time series	Sampling frequency
Biodiversity	Simpson's index & Species Richness	Index of zooplankton community composition; measures dominance & number of species present in study area (Peterson et al., NOAA)	North (OR)	1996 – 2013	Biweekly
		Index of ichthyoplankton community composition using CalCOFI and BPA time series (Thompson et al.)	North & South (OR/CA)	2004-2011	Quarterly
		Index of pelagic nekton species community composition (Brodeur et al., NOAA)	North (WA/OR)	1998 – 2013	June, Sept; Annual
		Index of groundfish community composition (Keller et al. NWFSC)	Entire	2003 - 2013	Summers, Annual
		Index of seabird community composition (Zamon et al. NWFSC; Sydeman et al.) (*Not updated here. Currently being revised. See previous IEA reports for trends through 2011).*	North & South (WA/OR; CA)	2004 - 2012; 1987 - 2012	Summers, Annual
Trophic structure	Mean trophic level	Trophic structure of groundfish community (Keller et al. NWFSC)	Entire	2003 - 2013	Summers, Annual
		Trophic structure of coastal pelagic fish community (*currently in development*) (Brodeur et al., NOAA)	North	1998 – 2013	June, Sept; Annual
		Trophic structure of seabird community (*currently in development*)			
		Trophic structure of marine mammal community (*currently in development*)			
Trophic structure	Scavenger biomass	Relative biomass of scavengers, as defined by esp. Brand et al. (2007), from fishery independent surveys (Keller et al. NWFSC)	Entire	2003 - 2013	Summers, Annual
Trophic structure	Northern copepod anomaly	Monthly anomalies in the relative biomass of copepods with cold-water affinities off Newport, OR (Peterson et al., NOAA);	North (OR)	1996 – 2013	Biweekly
Trophic structure	Gelatinous zooplankton biomass	Standardized abundance or biomass of jellyfish associated with near-surface waters (Brodeur et al., NOAA; Field et al., NOAA))	North & South (OR/WA; cent. CA)	1998 – 2013; 1986-2013	June, Sept; Annual; Annual

DATA ANALYSIS AND PRESENTATION

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

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

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

Northern and Southern Trends. Some datasets have limited range and describe trends only in certain regions. Other datasets span the entire CCLME. For the latter, we present three trends: coastwide, northern and southern. Northern trends are the area north of Cape Mendocino (40.4^{o} N), an important biogeographic break point. Southern trends are for the area south Cape Mendocino. In many cases, regional trends do not match the coastwide pattern.

MAJOR FINDINGS

Indicators for **Ecological Integrity** are ecosystem and community level indices that were chosen to track two community level aspects of the CCLME: **trophic structure** (mean trophic level, scavenger biomass, gelatinous zooplankton, and the northern copepod anomaly) and **diversity** (Simpson diversity, species richness for multiple taxa). The extent to which the data for these indicators cover the California Current Large Marine Ecosystem (CCLME) varies among taxa. The groundfish data span the U.S. West Coast (~32 to 48 ^{o}N, ~50-1200 m depths). Thus, conclusions for indicators based on the groundfish dataset (MTL, scavenger biomass, species richness, species density and Simpson diversity) are applicable to the full extent of the CCLME. Data for ichthyoplankton are drawn from southern California and Oregon survey transect lines, while those for gelatinous zooplankton are taken from surveys conducted off central California and the Oregon/Washington coasts. Data for coastal pelagic fishes are also drawn from the Oregon/Washington survey, whereas the copepod data are limited to survey stations in waters off of central Oregon. See the Ecological Indicators: Data Sources and Methodology for a more complete discussion of the datasets.

TROPHIC STRUCTURE

Indicators of trophic structure suggest neutral to good conditions in the CCLME (Figure EI 3). All indicators were within long-term norms although groundfish MTL was relatively low coastwide and north of Cape Mendocino and decreased south of Mendocino.

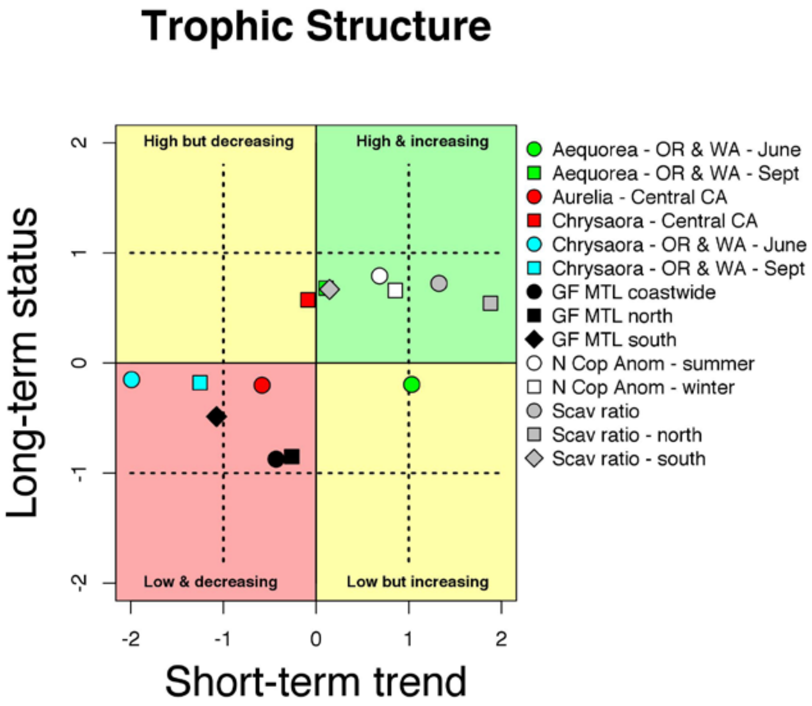

Figure EI 3. Short and long-term status of indicators of Trophic Structure for Ecological Integrity in the California Current Large Marine Ecosystem. Prior to plotting, time series were normalized to place them on the same scale. The short-term trend indicates whether the indicator increased or decreased over the last 5 years of the time series relative to the long-term mean. The long-term axis compares the mean of the last 5 years to the mean of the full time series. Changes or differences of more than 1.0 standard deviation of the full time series (dotted lines) are considered to show an effect. GF MTL = groundfish mean trophic level, N Cop Anom = northern copepod anomaly, Scav ratio = ratio of scavengers:total biomass, CA = California, OR = Oregon, WA = Washington. *Aequorea, Aurelia and Chrysoara* are gelatinous zooplankton (jellies). For GF MTL and Scav kg, north and south indicate north and south of Cape Mendocino.

Abundances of gelatinous zooplankton (*Aequorea, Aurelia* and *Chrysaora*) were near long-term average with *Chrysaora* decreasing in abundance in the short-term off of Oregon.

However, *Aequorea* increased in June surveys off of Oregon in the short-term. September values showed no trend. High abundance of gelatinous zooplankton is generally considered undesirable because they clog fishing nets, prey on fish larvae, and compete with forage fishes for food.

The northern copepod anomaly was relatively high but within long-term historical norms (within 1.0 s.d. of the long-term mean). High abundance of northern copepod species generally indicates good feeding conditions for many species.

The ratio of scavenger biomass to total biomass for groundfishes and crabs increased in the short-term coast-wide and in waters north of Cape Mendocino. The increase appears to have been driven by an increase in crab biomass.

Groundfish mean trophic level (MTL) declined south of Cape Mendocino. While stable overall and north of Cape Mendocino, current MTL was relatively low but still within 1.0 s.d. of the long-term mean. Low MTL is generally considered to be an indication of reduced abundance of top predators, and therefore, top-down forcing in the system. However, low groundfish MTL may make food resources (forage fishes and krill) available to groundfish competitors like salmon, seabirds and tuna, indicating good feeding conditions for these species.

BIODIVERSITY

Biodiversity indicators showed mixed results (Figure EI 4). No indicators showed changes relative to their long-term means. However, six diversity measures increased in the short-term, while seven decreased.

Copepod biodiversity in the summer declined as did species number for coastal pelagic fishes. Three diversity indicators related to copepods and coastal pelagic fishes decreased in the short-term. While decreased diversity is typically considered a negative indication of ecosystem status, low copepod diversity is linked to high abundance of northern, energy-rich species and indicates overall good feeding conditions for species like forage fishes and salmon.

All measures of ichthyoplankton biodiversity increased for the California Current as a whole. Conversely, ichthyoplankton spring Simpson diversity and summer species richness both declined in Oregon, suggesting different trends in northern and southern regions.

For groundfishes, coastwide Simpson diversity (~evenness, technically equitability) increased in the short-term. This rise was driven by changes north of Cape Mendocino. Simpson diversity south of Cape Mendocino did not increase. Groundfish species richness declined coastwide, as did species richness south of Cape Mendocino. North of Cape Mendocino richness also declined but by less than the threshold value. In all three cases, richness was within long-term norms but above the long-term mean. Declines in groundfish MTL noted earlier were caused by loss of Pacific hake and spiny dogfish biomass, and the increase in Simpson diversity may be linked to these trends as well. It is not clear at present what caused the decline in groundfish species richness south of Cape Mendocino, but the trend bears watching.

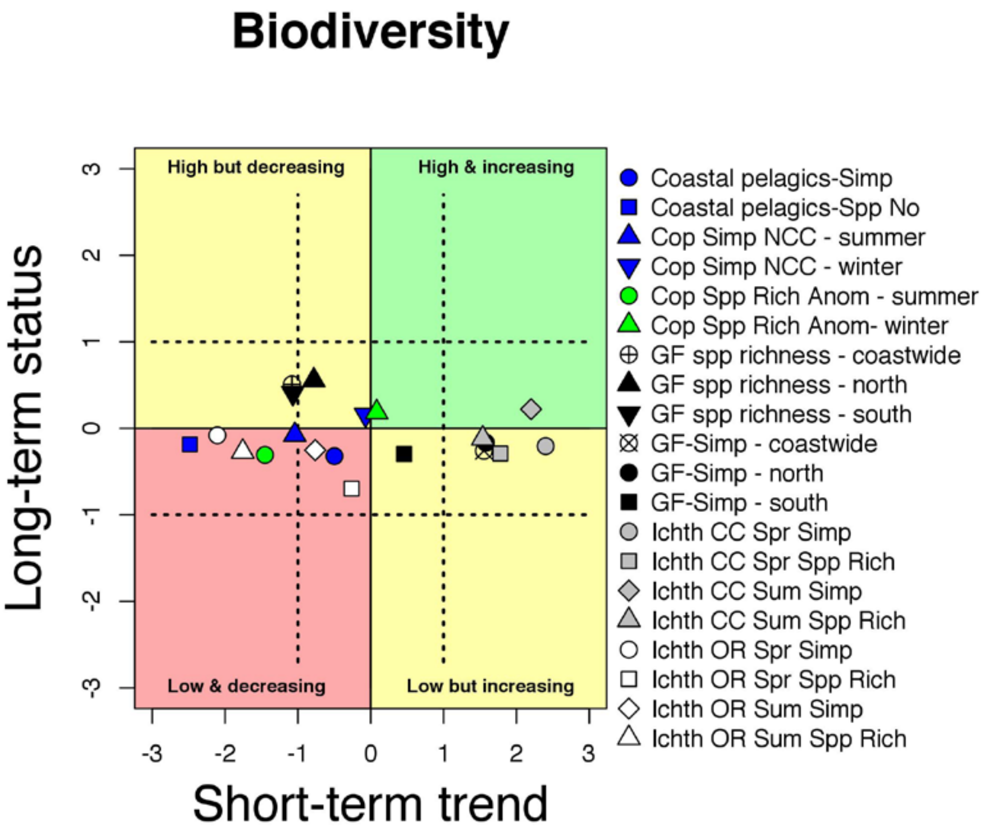

Figure EI 4. Short and long-term status of indicators of Biodiversity for Ecological Integrity in the California Current Large Marine Ecosystem. Prior to plotting, time series were normalized to place them on the same scale. The short-term trend indicates whether the indicator increased or decreased over the last 5 years of the time series relative to the long-term mean. The long-term axis compares the mean of the last 5 years to the mean of the full time series. Changes or differences of more than 1.0 standard deviation of the full time series (dotted lines) are considered to show an effect. Cop = copepod, GF = groundfishes, Ichth = ichthyoplankton, Simp = Simpson diversity, Spp No = species number (not rarefied), Spp Rich = species richness (rarefied), Anom = anomaly, CC = CalCOFI (southern California Current), OR = Oregon, NCC = northern California Current.

LOOKING TO THE FUTURE

Seabird diversity indices were not included in the present report because data were not available to update the previously reported time series. See the 2011 report for these indicators[5]. Work is ongoing to consolidate disparate datasets and produce a more succinct and cogent set of seabird indicators for future IEA reports.

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

At present, most of the Ecological Integrity indicators do not have thresholds or targets, and temporal trends are evaluated with regards to the statistical properties of the time series in question. Future work should seek to establish thresholds and targets for each indicator.

STATUS AND TRENDS: TIME SERIES DATA

In this section, we present the status and trends of each of the five ecological integrity indicators for the California Current ecosystem during 2013, as derived from time series data. Most time series are plotted in a standard format: Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last five years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the modeled trend over the last five years increased (↗), or decreased (↘) by more than 1.0 s.d., or was within 1.0 s.d. (↔) of the long-term trend. The lower symbol indicates whether the mean of the last five years was greater than (+), less than (-), or within (·) 1.0 s.d. of the long-term mean. In some cases, background analyses and interpretation of related information are also included.

NORTHERN COPEPOD BIOMASS ANOMALY

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

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[5] http://www.noaa.gov/iea/CCIEA-Report/index.html

compounds appear to be essential. Beamish and Mahnken (2001) provide an example of this for coho salmon (see http://www.nwfsc.noaa.gov/research/divisions/fed/oeip/eb-copepod-anomalies.cfm for further detail).

The northern copepod anomaly has fluctuated between 1996 – 2013. The most recent available values for both the winter (Figure EI 5) and summer (Figure EI 6) are relatively high —approximately 1.0 s.d. above the mean of the full time series—indicating generally good conditions. There were no trends in either case. Threshold values for the anomaly have not been set. However, positive values in the summer period are correlated with stronger returns of fall and spring ocean-type Chinook to Bonneville dam, and values greater than 0.2 are associated with better survival of coho salmon. Overall the high anomalies in recent years, especially for the summer data, suggest that ocean conditions are in a generally good state.

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

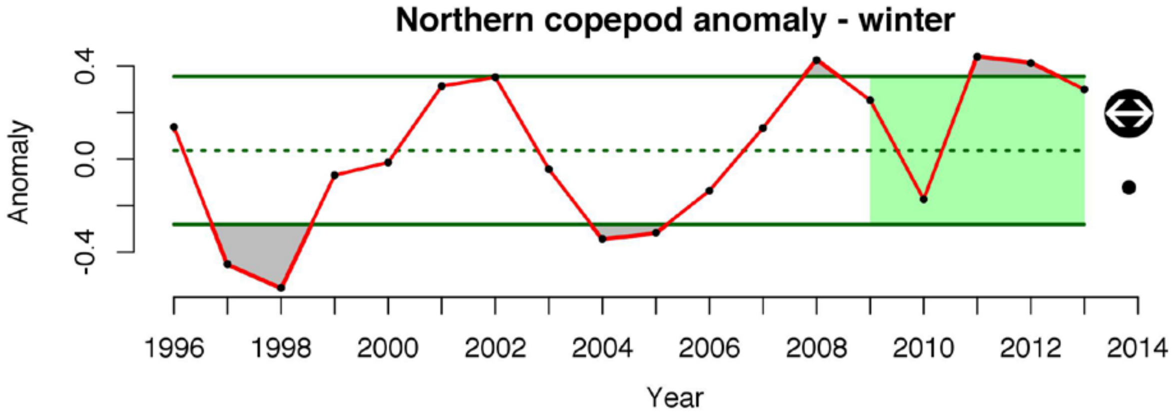

Figure EI 5. Northern copepod biomass anomaly for 1996-2013 in the waters off of Oregon during the winter (Oct-April). Data courtesy of Bill Peterson (bill.peterson@noaa.gov).

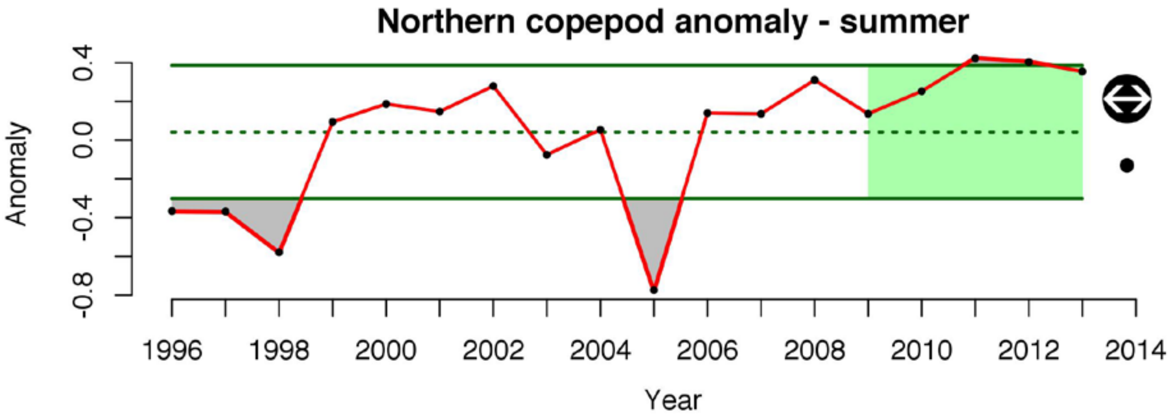

Figure EI 6. Northern copepod biomass anomaly for 1996-2013 in the waters off of Oregon during the summer (May - September). Data courtesy of Bill Peterson (bill.peterson@noaa.gov).

ABUNDANCE OF GELATINOUS ZOOPLANKTON (JELLIES)

Gelatinous zooplankton (jellyfish) are a lower trophic level, high-productivity functional group, potentially with important effects on the transfer of nutrients and energy from lower trophic levels to higher ones. High abundance of gelatinous zooplankton indicates potentially poor conditions for other taxa within the CCLME. The large medusa quantified here, *Chrysoara* sp. and *Aequorea* sp., are highly opportunistic and respond quickly to regional and local forcing factors (Suchman et al. 2012).

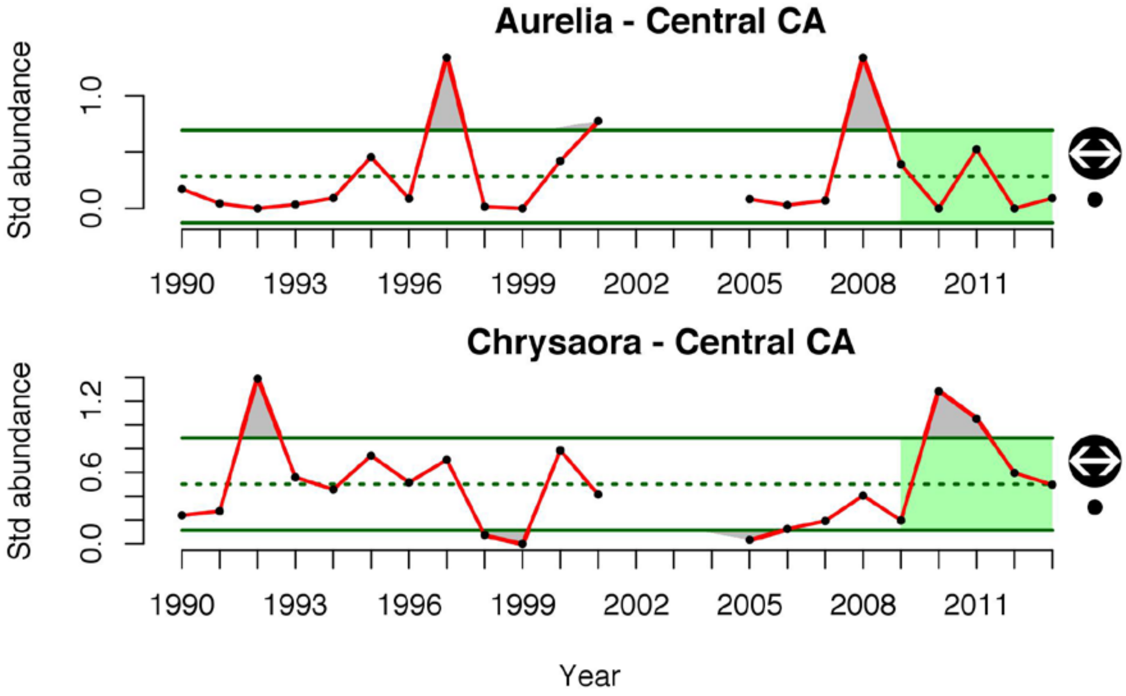

Figure EI 7 Standardized abundance of *Aurelia* and *Chrysaora* jellies in central California waters from 1990-2013. Data are courtesy of John Field (john.field@noaa.gov).

In waters off of central California, jelly abundance has fluctuated since the early 1990's (Figure EI 7). At present both *Aurelia* and *Chrysaora* are near their long-term means and showed no trends over the last five years. Both taxa have, however, decreased in abundance relative to recent peaks in 2008 and 2010 respectively. Both peaks were more than 1.0 s.d. above the long-term mean. Values for 2013 were near the long-term mean, suggesting typical conditions.

Surveys from Oregon and Washington waters showed mixed results (Figure EI 8, Figure EI 9). *Aequorea* abundance increased in the short-term in June surveys but showed no trend over the last five years in September surveys. However, *Chrysoaroa* abundance declined in the short-term in both June and September surveys.

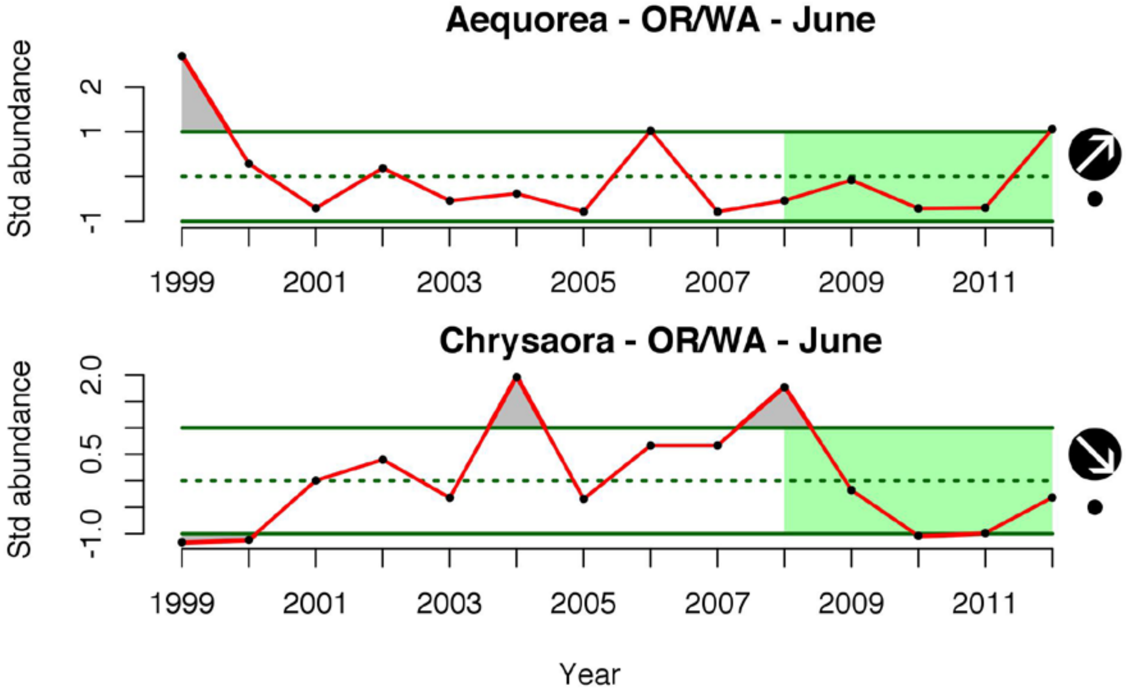

Figure EI 8 Standardized biomass of *Chrysoara* and *Aequorea* jellies in June surveys in the NCC from Newport, OR (44.6°N, 124.0° W) to Tatoosh Island, WA (48.4 N°, 124.7° W). Data are courtesy of Ric Brodeur (rick.brodeur@noaa.gov).

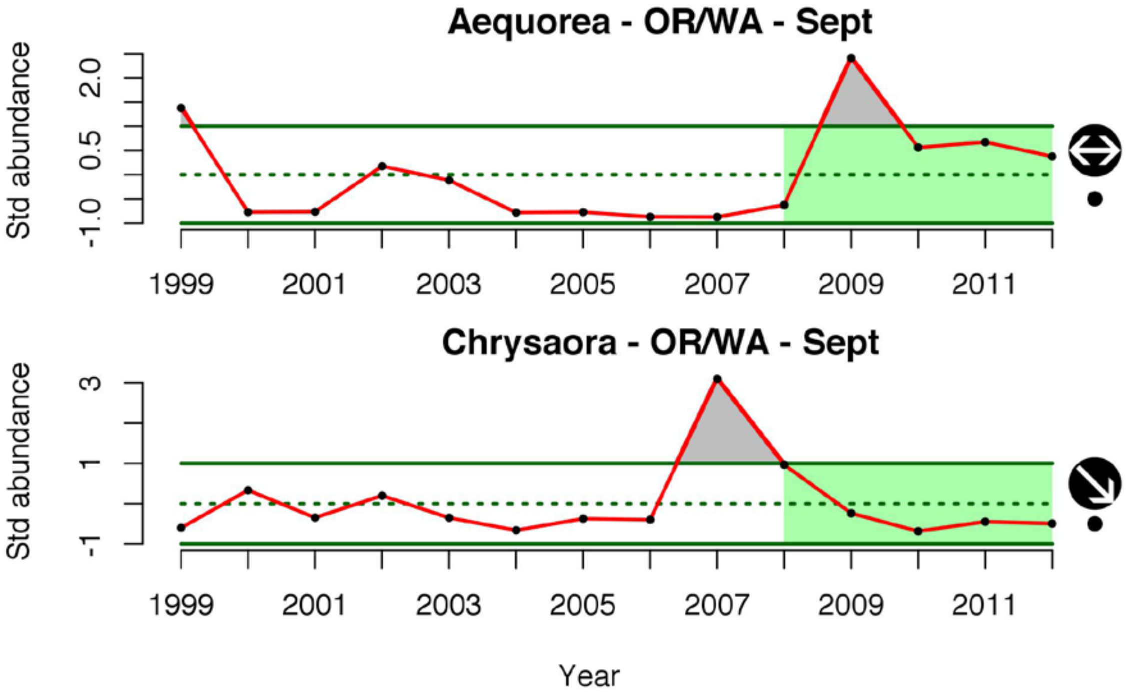

Figure EI 9 Standardized biomass of *Chrysoara* and *Aequorea* jellies in September surveys in the NCC from 1999 to 2012 off Newport, OR (44.6ºN, 124.0º W) to Tatoosh Island, WA (48.4 Nº, 124.7º W). Data are courtesy of Ric Brodeur (rick.brodeur@noaa.gov).

MEAN TROPHIC LEVEL (GROUNDFISHES)

Mean trophic level (MTL) is the biomass-weighted average of the trophic levels of the species in a sample (Pauly et al. 1998). It is widely used as an indicator of change in trophic structure (Pauly and Watson 2005). MTL is conceptually linked to trophic cascades (Estes et al. 2011). A drop in MTL is generally considered a negative indicator of ecosystem status, as it should result in a decrease in the strength of top-down forcing. However, a fall in MTL of one component (e.g., groundfishes) of the ecosystem may make prey resources available to competing taxa (e.g., salmon, seabirds and tuna), especially in wasp-waist systems where many predators rely on a small suite of prey (Tolimieri et al. 2013).

MTL comes in two forms (Branch et al. 2010). 'Catch' MTL is calculated from fisheries-dependent data and reflects changing fishing practices and availability of target species. 'Ecosystem MTL' is calculated from fisheries-independent data and represents changes in the ecosystem. Here we report 'Ecosystem' MTL for West Coast groundfishes. MTL was calculated from the West Coast Groundfish Bottom Trawl Survey. Trends are presented for the entre CCLME and for northern and southern regions, separated by Cape Mendocino (40.4ºN).

MTL for groundfishes declined from 2003 until 2010 and has remained low since (Figure EI 10). The fluctuation over the entire time series was approximately 0.077, from a high of 3.72 in 2004 to a low of approximately 3.64 in both 2010 and 2012. This decline represents a ~25% decrease in the primary productivity required to support a given amount of catch (Pauly and Christensen 1995, Essington et al. 2006). While threshold values for MTL have not been set, future work could set thresholds based on changes in necessary primary production.

Previous reports document a decline in MTL from 2003 to 2010 and 2011 largely due to a decrease in the abundance of Pacific hake, *Merluccius productus* (Keller et al. 2012, Tolimieri et al. 2013). However, over the last five years of the time series (2008-2012), groundfish MTL has been low but stable with no further decline. The mean of the last five years of the time series is within 1.0 s.d. of the full time series, but MTL for 2012 was below 1.0 s.d. of the full time series and bears watching in the future. Comparisons with other long-term datasets suggest that fluctuations in MTL are not uncommon (Branch et al. 2010).

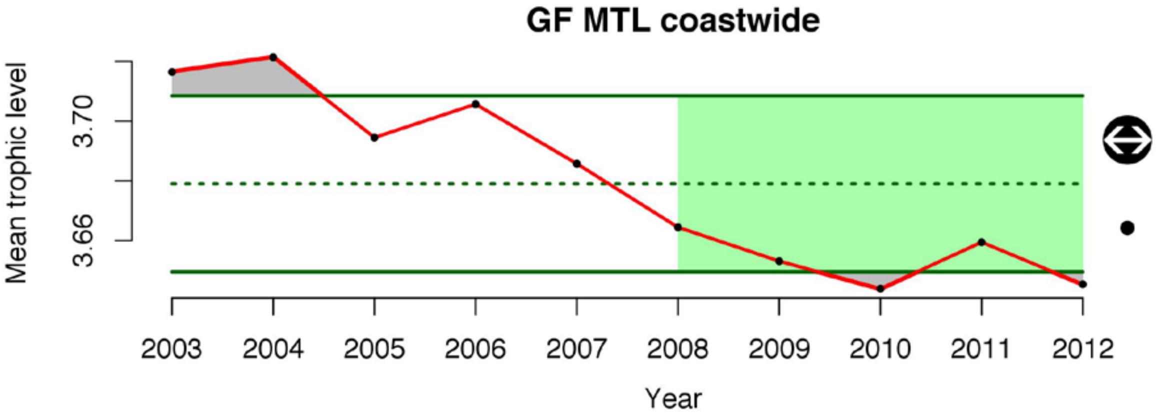

Figure EI 10. Area-weighted mean trophic level (MTL) for West Coast groundfishes from 2003 – 2012. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness.

In the region north of Cape Mendocino, MTL declined steadily from 2003 to 2010 from approximately 3.76 to 3.66 in 2010 (Figure EI 11). However over the last five years, MTL has remained low but fairly stable with no short-term trend (change over the last five years was less than 1.0 s.d. of the full time series). The mean of the last five years was also within 1.0 s.d. of the long-term mean.

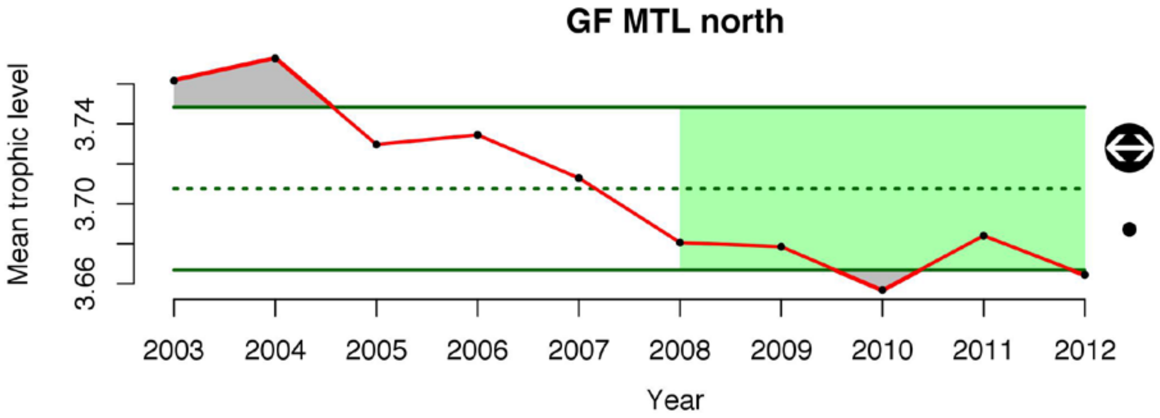

Figure EI 11. Area-weighted mean trophic level (MTL) for West Coast groundfishes north of Cape Mendocino (40.4° N) from 2003 – 2012. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

South of Cape Mendocino, MTL initially increased from 2003 to 2006 but then declined until 2012, with the last five years declining more than 1.0 s.d. of the long term mean (Figure EI 12). However, the mean of the last five years was within 1.0 s.d. of the long-term mean and the value in 2012 was similar to that in 2003. Most of the decline occurred from 2008 to 2009 and MTL has largely been low but stable since.

Low groundfish MTL may indicate good conditions for the competitors of groundfishes. Many predators in the CCLME eat krill and forage fishes. Food web modeling suggests that a drop in groundfish MTL due to a loss of higher TL species makes these prey available to other taxa such as squid, salmon, tuna and seabirds leading to positive population forcing for these taxa (Tolimieri et al. 2013). Therefore, setting targets for groundfish MTL may entail making trade-offs with these other species.

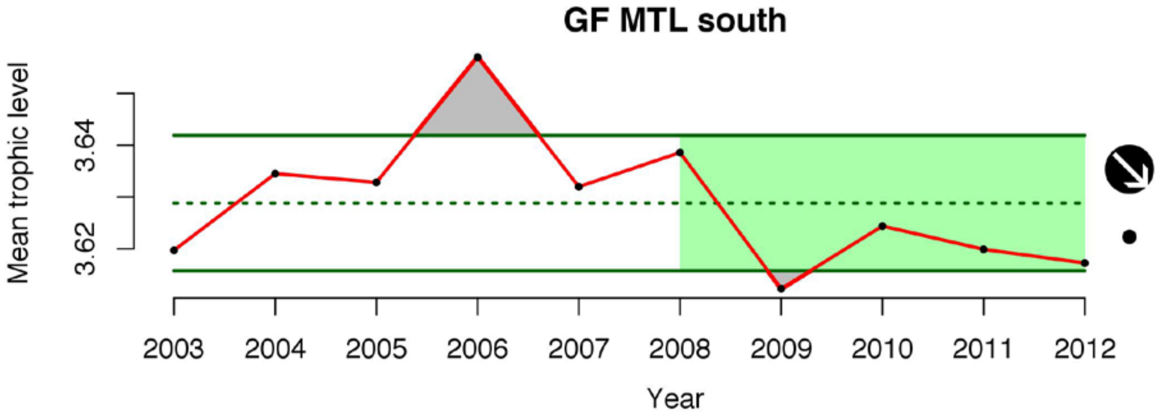

Figure EI 12. Area-weighted mean trophic level (MTL) for West Coast groundfishes south of Cape Mendocino (40.4° N) from 2003 – 2012. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

RATIO OF SCAVENGER BIOMASS: TOTAL BIOMASS (GROUNDFISHES & CRABS)

Scavengers are generally defined as active foragers of carrion (Britton and Morton 1994). Changes in the biomass of scavengers have been related to responses to fisheries discards and disturbance of bottom habitat due to trawling. The indicator presented here includes multiple groundfishes and three species of crab quantified in the West Coast Groundfish Bottom Trawl Survey (see Table EI 4, Data Sources and Methodology for a list of included taxa). Because using raw biomass (kg per km^{-2}) would not separate an increase in scavenger biomass from an increase in the biomass of all species, we use the ratio of scavenger biomass to total biomass in the trawl survey to test for a change in the trophic structure. This modification provides a more sensitive indication of whether 'scavengers' are more or less prevalent in the assemblage than in previous years.

The ratio of scavengers to total biomass increased from 2008 to 2012 (Figure EI 13) with the trend over the last five years showing an increase of more than 1.0 s.d. of the full time series. However, the ratio declined from a high of 0.27 in 2010 to just over 0.25 in 2012. The mean of the last five years was within 1.0 s.d. of the long-term mean.

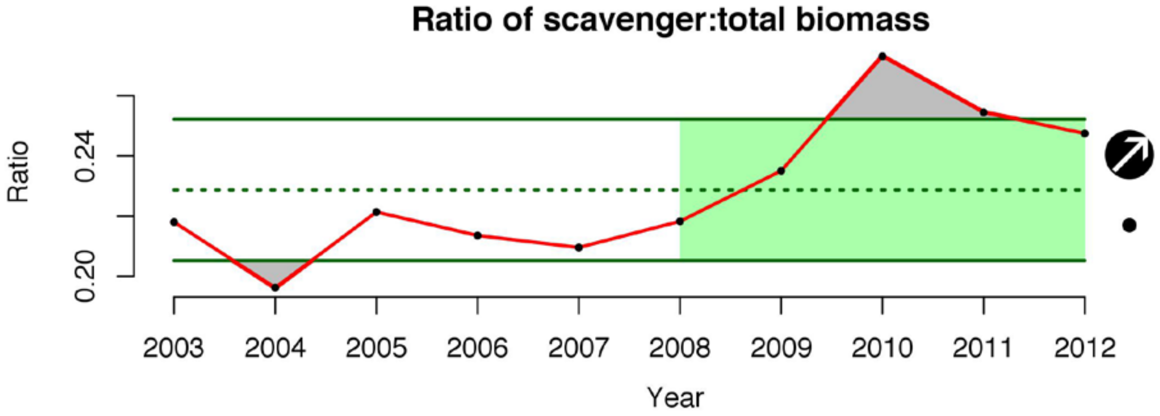

Figure EI 13. Ratio of groundfish and crab scavengers to total biomass for the West Coast shelf and slope from 2003-2012. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

Patterns north and south of Cape Mendocino (40.4^{o} N) differed. Trends north of Cape Mendocino mirrored the coastwide pattern with an increase from just over 0.16 to over 0.22. (Figure EI 14).

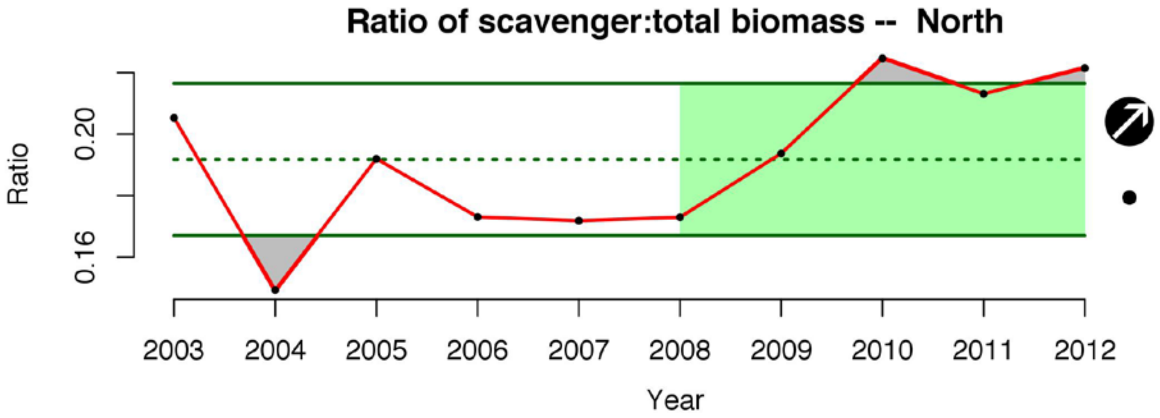

Figure EI 14. Ratio of groundfish and crab scavengers to total biomass for the West Coast shelf and slope north of Cape Mendocino (40.4º N) from 2003-2012. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

South of Cape Mendocino, the ratio of scavenger to total biomass initially increased from 2003 to a high of approximately 0.35 in 2010 (Figure EI 15). However, over the last five years of the time series, the ratio, while fluctuating, showed no trend and the 2012 value is more or less the same as in 2008 at 0.28. The mean over the last five years was within 1.0 s.d. of the long-term mean.

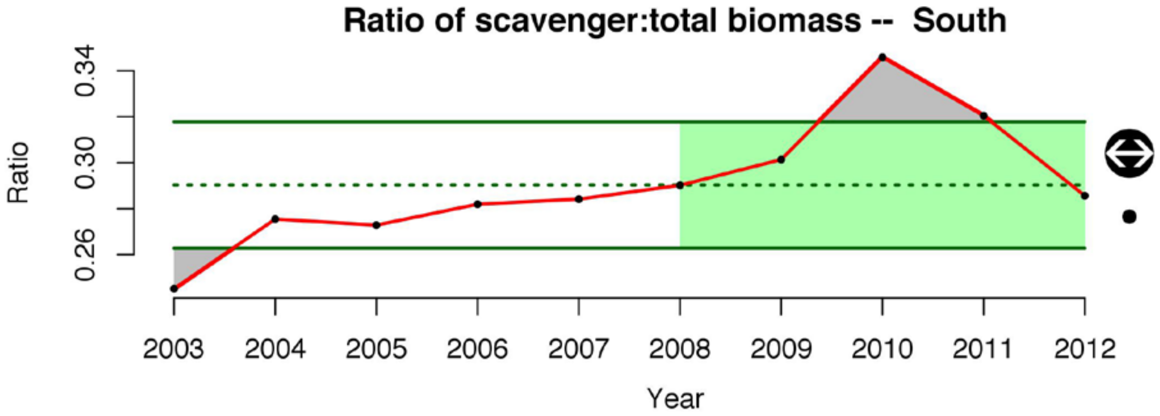

Figure EI 15. Ratio of groundfish and crab scavengers to total biomass for the West Coast shelf and slope north of Cape Mendocino (40.4º N) from 2003-2012. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov)

The increase in the ratio of scavenger to total biomass coastwide and in northern waters appears to be due to an increase in the abundance of crabs (Figure EI 16). In both time series crab to total biomass ratio increased from 2008 – 2012, peaking in 2010 for before declining somewhat. Trends in the south were similar but did not meet the threshold of a change of 1.0 s.d. or more because the southern time series dropped more from 2010 to 2012 than did the northern or coastwide one finising near the long-term mean.

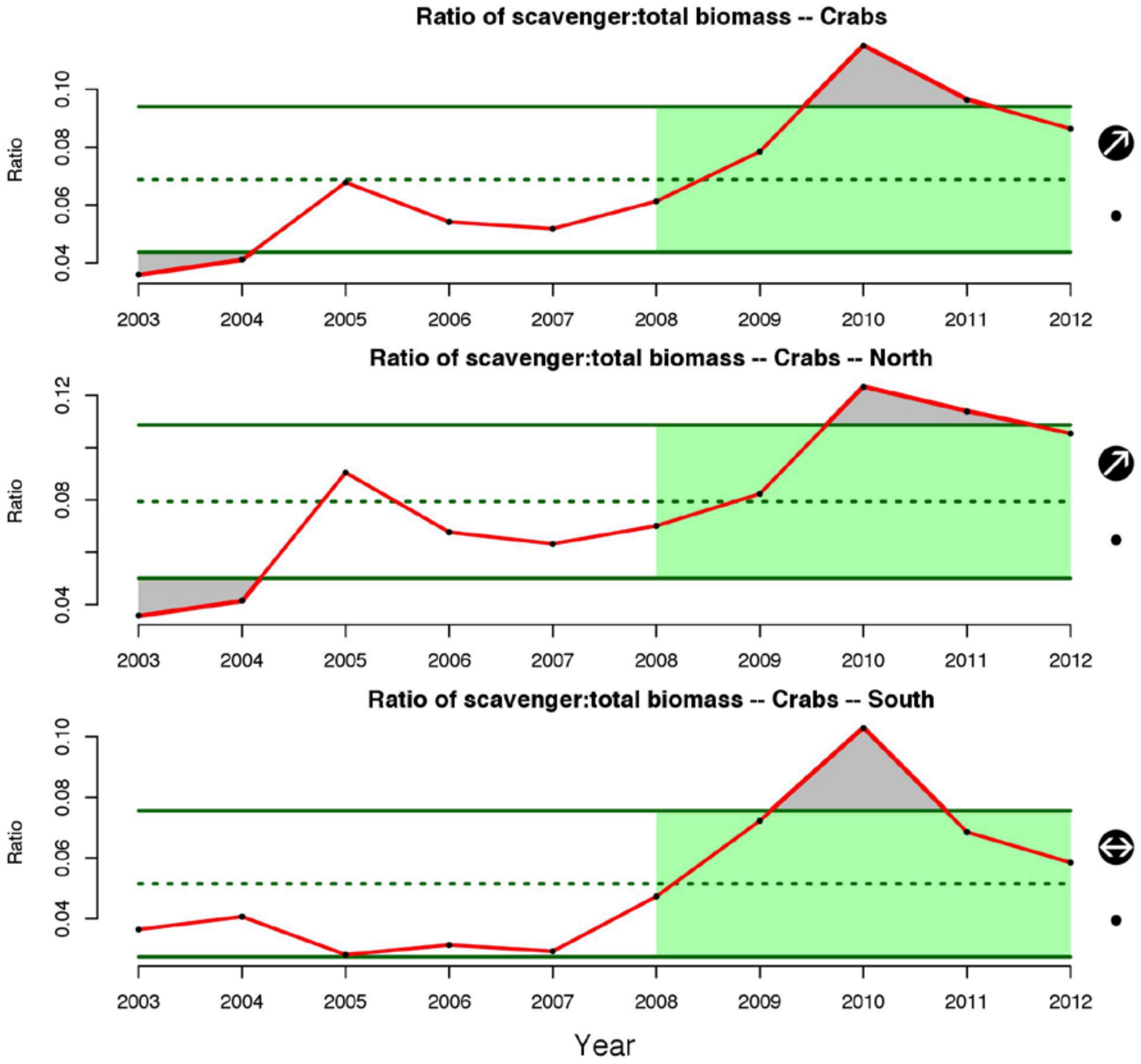

Figure EI 16 Ratio of crab scavengers to total biomass for the West Coast shelf and slope coast-wide, and north or south of Cape Mendocino (40.4° N) from 2003-2012. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov)

The ratio of fish scavengers to total biomass showed no trends over the last five years of the data (Figure EI 17). Coastwide and in northern waters the time series showed little fluctuation. However, in waters south of Cape Mendocino there was substantially more variation in the time series.

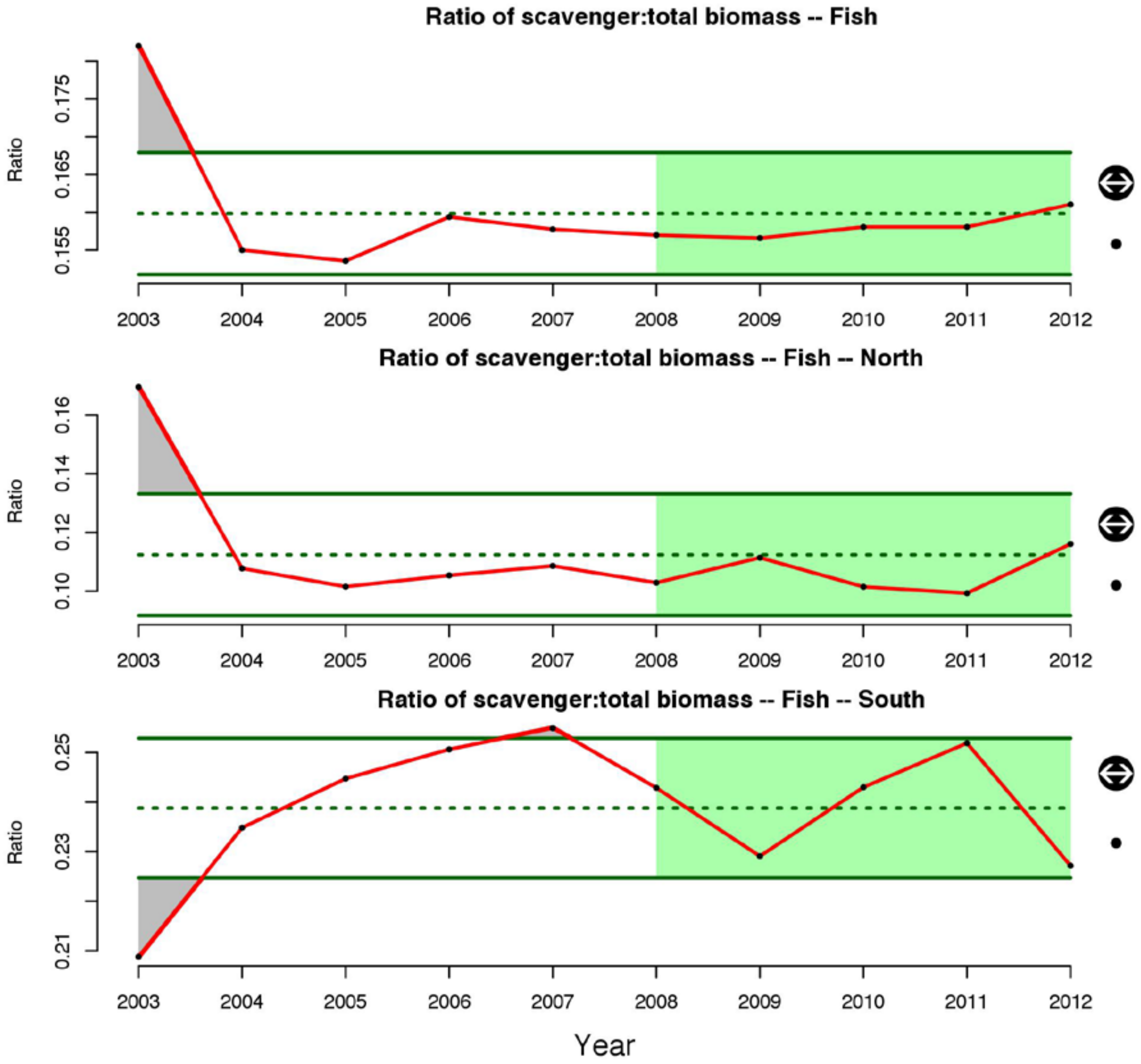

Figure EI 17 Ratio of groundfish scavengers to total biomass for the West Coast shelf and slope coast-wide, and north or south of Cape Mendocino (40.4° N) from 2003-2012. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov)

SIMPSON DIVERSITY (COPEPODS, COASTAL PELAGIC FISHES & GROUNDFISHES)

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

SIMPSON DIVERSITY - COPEPODS

Simpson diversity for copepods in the northern California Current was calculated by season using the same seasons as Peterson (2009) (Figure EI 18). For both seasons, Simpson diversity was variable through time. Simpson diversity for summer (May – Sept) assemblages decreased in the short-term (5-year trend showed a decrease of greater than 1.0 s.d. of the full time series), but the mean of the last five years was within 1.0 s.d. of the full time series. Simpson diversity for winter (Oct – April) assemblages showed no short-term trend, and the mean of the last five years was within historical norms.

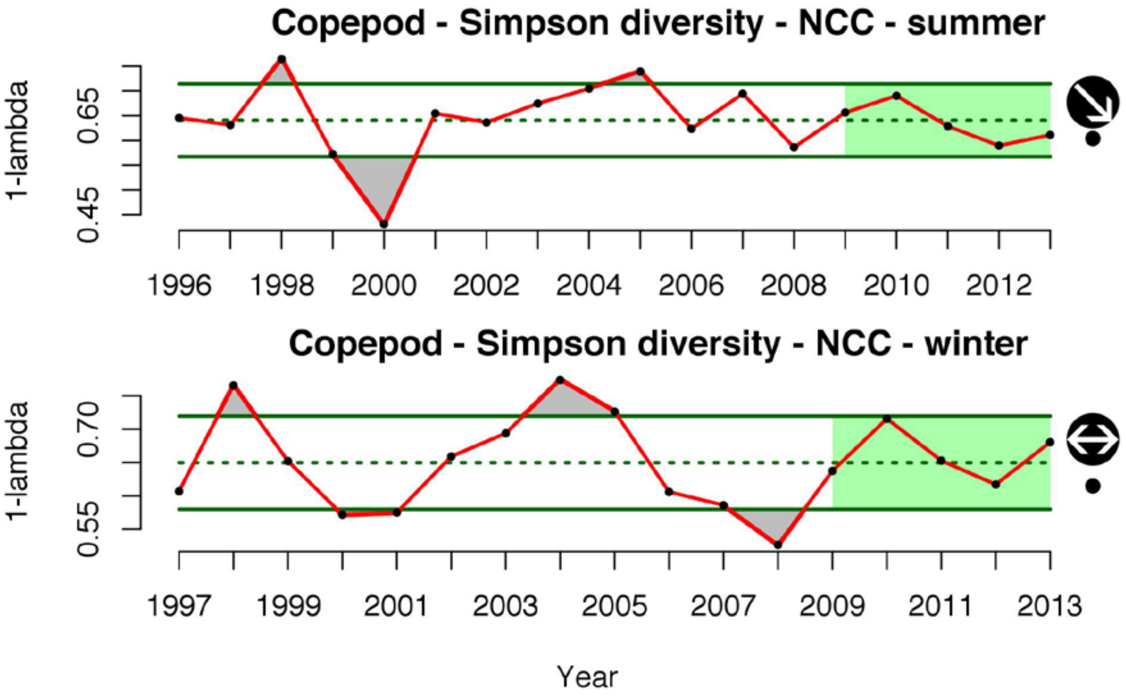

Figure EI 18. Time series of Simpson diversity (1-λ) from 1997 – 2013 for summer (May -- Sept) and winter (Oct–April) for West Coast copepods in the northern California Current (NCC). Data courtesy of Bill Peterson (bill.peterson@noaa.gov).

SIMPSON DIVERSITY - COASTAL PELAGIC FISHES

Simpson diversity for coastal pelagic species has fluctuated through time on approximately a 4-5 year cycle with highs in 2002, 2007-8 and 2011, and lows in 2000, 2005 and 2009-10 (Figure EI 19). Over the last five years, however, there has been no directional trend, and the mean of the last five years is within 1.0 s.d. of the long-term mean. However, peak values have decreased successively since 2002, being well above 1.0 s.d. of the time series in 2002, about 1.0 s.d. above the mean in 2007 and 2008 and around 0.66 s.d. above the mean in 2011.

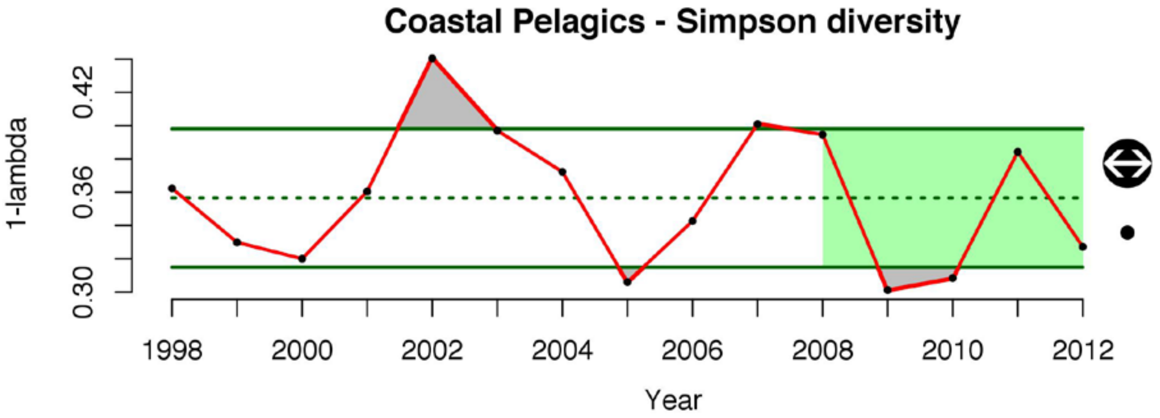

Figure EI 19. Simpson diversity (1-λ, Gini-Simpson index) for coastal pelagic fishes in the Northern California Current from 1998-2012. Data are combined June and September samples. Data courtesy of Richard Brodeur(Rick.Brodeur@noaa.gov).

SIMPSON DIVERSITY - GROUNDFISHES

Simpson's index (1-λ) for West Coast groundfishes decreased between 2003 and 2009 (Figure EI 20). Over the last five years (2008-2012), Simpson's index increased by more than one standard deviation (s.d.) of the complete time series. However, much of this evenness was lost in 2012 when Simpson diversity declined markedly. The mean of the last five years is within 1.0 s.d. of the long-term mean. North of Cape Mendocino (40.4^{o} N) the trend was more or less similar to the full, coastwide pattern (Figure EI 21). South of Cape Mendocino, the pattern differed somewhat with a peak in 2007 and lower values since (Figure EI 22). Southern Simpson diversity has remained more or less stable over the last five years.

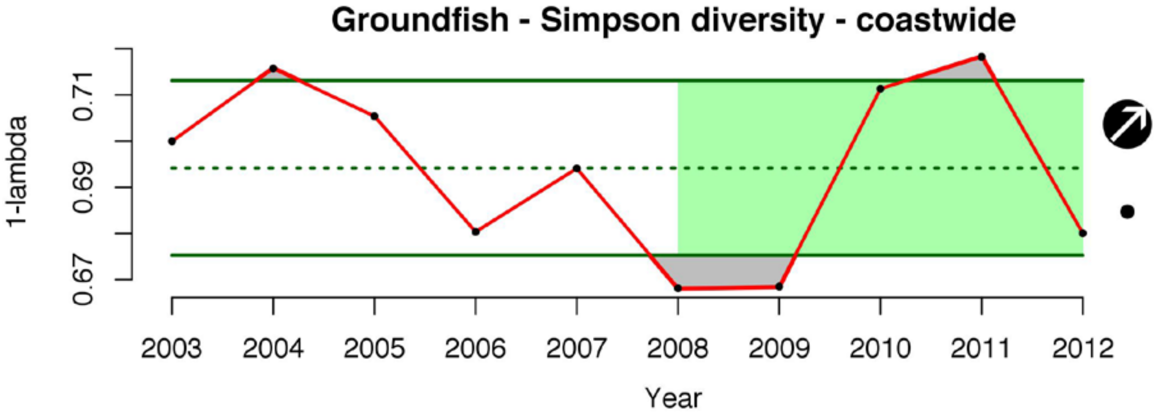

Figure EI 20. Simpson diversity (1-λ) for West Coast groundfishes from 2003 – 2012. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy Beth Horness (Beth.Horness@noaa.gov).

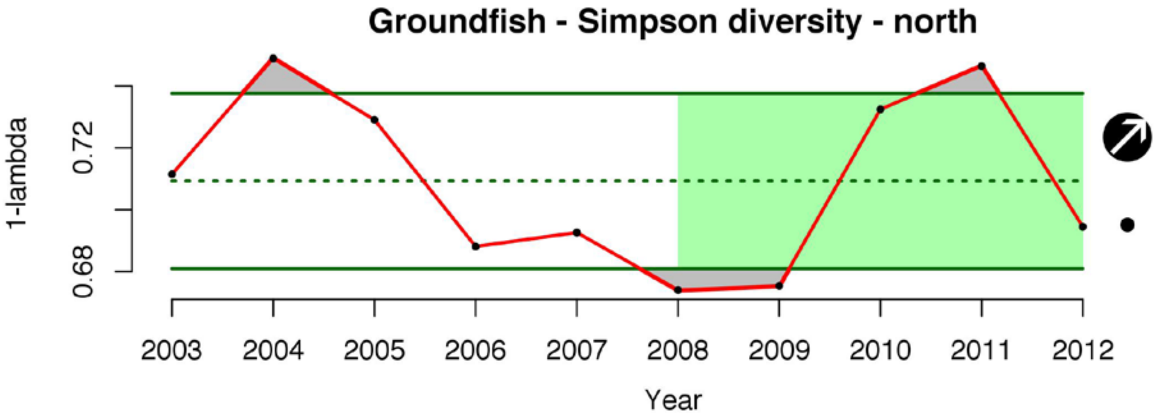

Figure EI 21. Simpson diversity (1-λ) for West Coast groundfishes from 2003 – 2012 north of Cape Mendocino (40.4° N). Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy Beth Horness (Beth.Horness@noaa.gov).

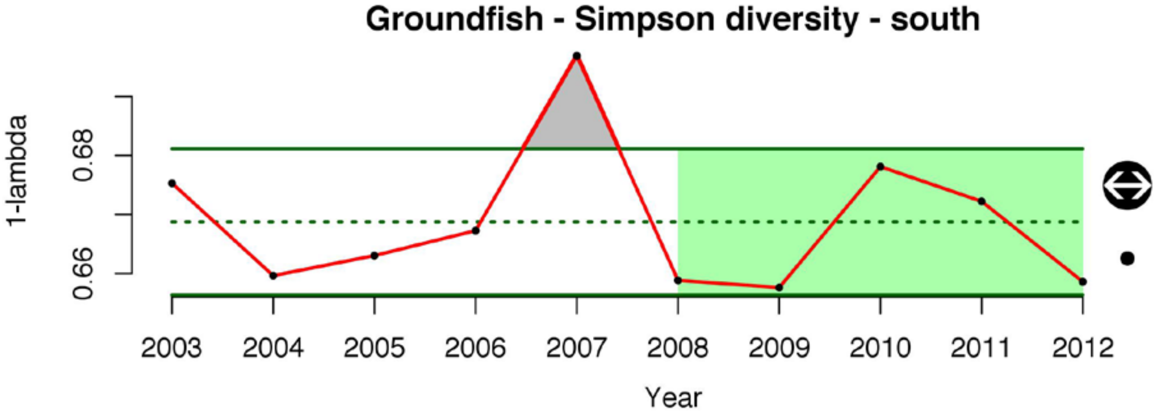

Figure EI 22. Simpson diversity (1-λ) for West Coast groundfishes from 2003 – 2012 south of Cape Mendocino (40.4° N). Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy Beth Horness (Beth.Horness@noaa.gov).

SIMPSON DIVERSITY – ICHTHYOPLANKTON

Oregon and CalCOFI data show opposite trends in diversity of ichthyoplankton, suggesting that different processes occur in the two locations. Simpson diversity (1- λ) in the spring ichthyoplankton from the CalCOFI surveys in summer California during the spring declined from over 0.7 in 2004 to a low of less than 0.5 in 2007 (Figure EI 23). It then increased over the next five years by more than 1.0 s.d. of the full time series to approximately 0.7.

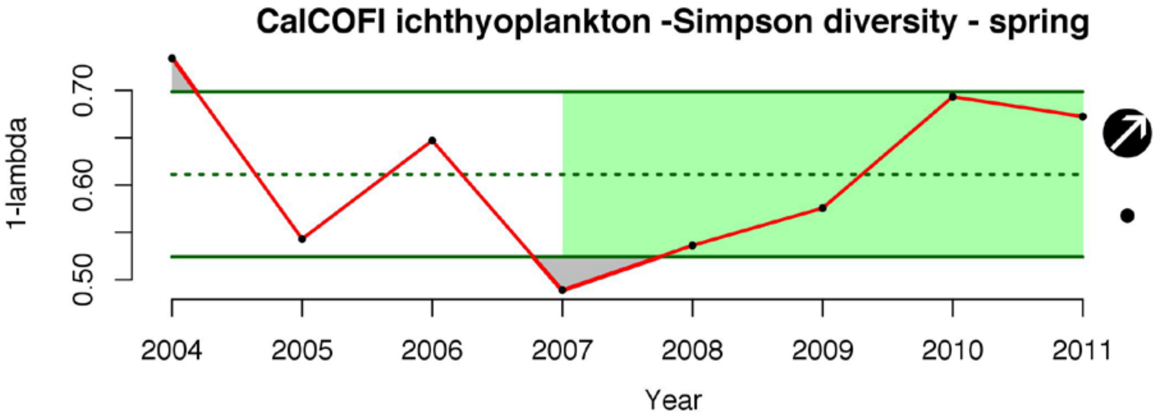

Figure EI 23. Simpson diversity (1-λ) of ichthyoplankton from the CalCOFI surveys in southern California during the spring from 2004 – 2011. Data courtesy of Andrew Thompson (andrew.thompson@noaa.gov).

Simpson diversity (1- λ) in the summer for ichthyoplankton off of southern California in the CalCOFI surveys declined from ~0.64 in 2004 to less than 0.54 in 2007 (Figure EI 24). It then increased to 2011 by more than 1.0 s.d. of the dataset before returning to values similar to 2004. It was highest in 2010 at approximately 0.66.

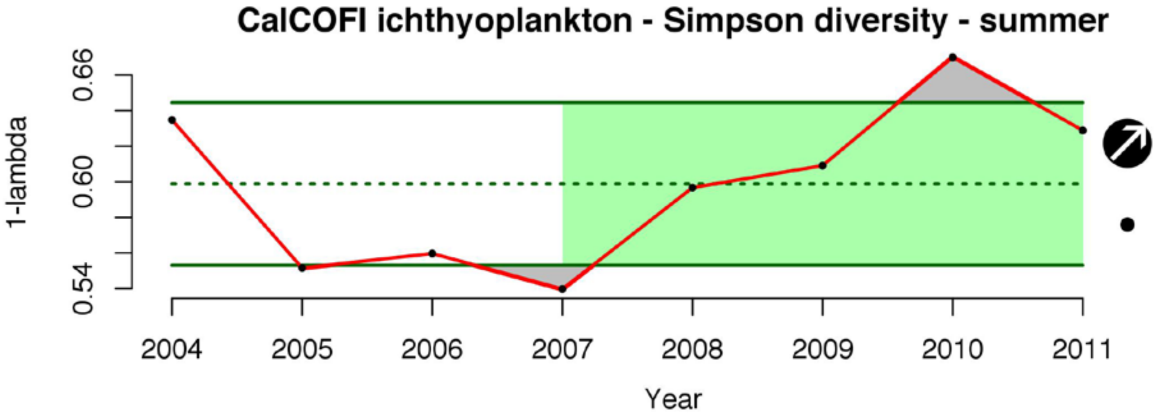

Figure EI 24. Simpson diversity of ichthyoplankton from the CalCOFI surveys in southern California during the summer from 2004 – 2011. Data courtesy of Andrew Thompson (andrew.thompson@noaa.gov).

Interestingly, Simpson diversity in the Oregon ichthyoplankton survey showed the reverse trend to that off of California. Spring Simpson diversity increased to a high of just more than 0.6 in 2007 before declining to as low as 0.4 in 2010 (Figure EI 25), resulting in a decrease over the last five years. However, the value for 2011 was approximately the same as the mean of the full time series.

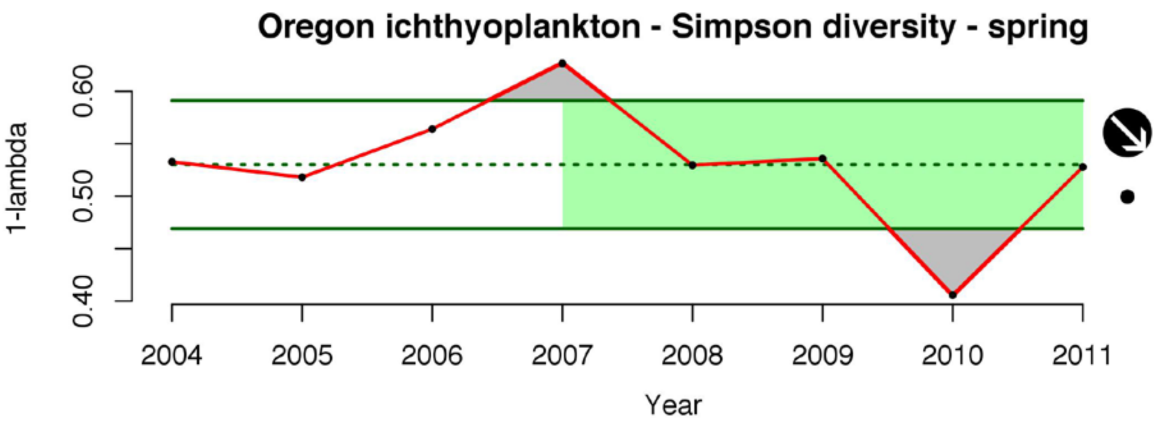

Figure EI 25. Simpson diversity of ichthyoplankton off Oregon in the spring from 2004 – 2011. Data courtesy of Andrew Thompson (andrew.thompson@noaa.gov).

Summer values off Oregon followed a similar trend with an extreme low in 2010 followed by a rebound to the vicinity of the long-term mean in 2011 (Figure EI 26). However, given the increase over the last year of the time series, there was no overall trend over the last five years and the mean was within 1.0 s.d. of the full time series.

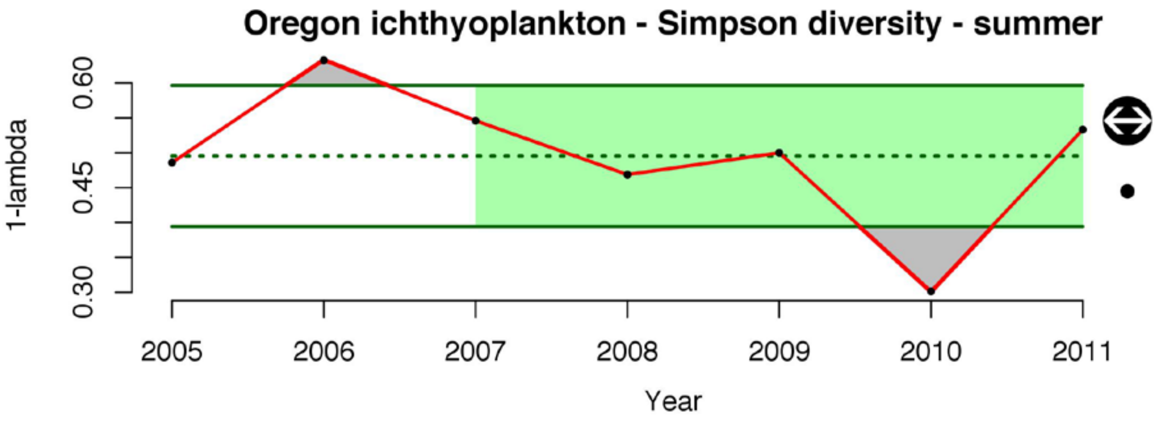

Figure EI 26. Simpson diversity of ichthyoplankton off Oregon in the summer from 2005 – 2011. Data courtesy of Andrew Thompson (andrew.thompson@noaa.gov).

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

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

Because sample effort, whether the number of individuals collected or area surveyed or both, has strong, non-linear effects on the number of species encountered, estimates of richness need to be scaled to a common effort level through rarefaction

(Gotelli and Colwell 2001). The number of species in a community can then be presented in two forms: species richness and species density. Species richness in the rarefaction sense is the number of species observed for some number of individuals collected. Species density is the number of species per some unit area. Both are relevant to different questions and purposes. Since most theoretical models in ecology are based on *per capita* interactions, species richness is relevant to these models. At the same time, species density is important to conservation and applied purposes since it measures the number of species in a given area. Here, both species richness and species density are reported where possible based on the attributes of the particular dataset. See Gotelli and Colwell (2001) for further discussion of rarefaction, species richness and species density.

SPECIES NUMBER - COASTAL PELAGIC FISHES

Data for coastal pelagic fishes were not rarefied due to the data format. Mean number of species per sample for coastal pelagic fishes was variable through time with lows in 1999, 2000, 2005 and 2012 and highs in 2003, 2004 and 2008 (Figure EI 27). While the mean of the last five years is within 1.0 s.d. of the long-term mean, species number declined rapidly from 2008 to 2012 by more than 2.0 s.d. of the full time series. Species number as of 2012 was the lowest over the 15-year time series, dropping below the previous lowest year of 1999.

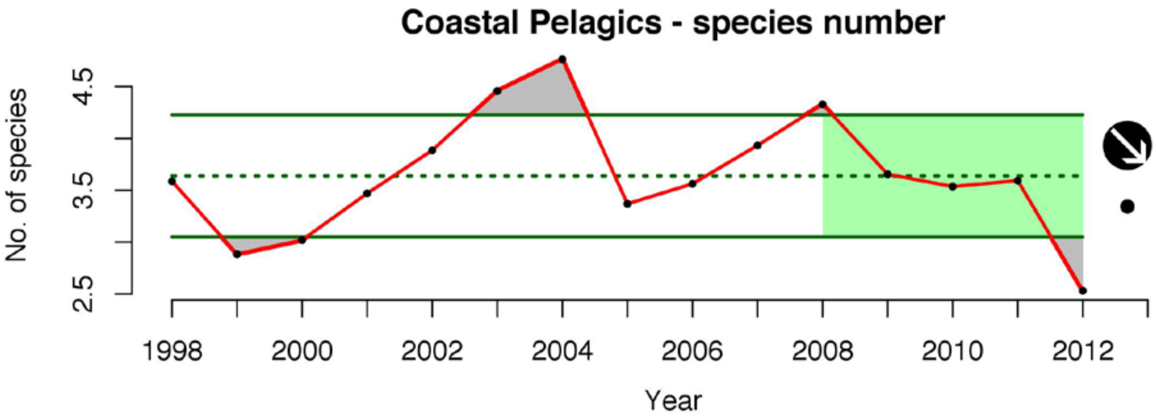

Figure EI 27. Number of species per sample for coastal pelagic fishes in the Northern California Current from 1998-2012. Data are combined June and September samples. Data courtesy of Richard Brodeur (Rick.Brodeur@noaa.gov).

SPECIES RICHNESS - COPEPODS

Copepod species richness has been tied to food chain structure and survival of coho salmon in the California Current (Peterson 2009). Low species richness is correlated with the southern transport of northern waters, high abundance of lipid-rich northern copepods and increased growth and survival of some species (Peterson 2009).

The species richness anomaly for copepods was highly variable over time. Species richness for the winter assemblage (Figure EI 28) showed no trend in the short-term, and the mean of the last five years was within 1.0 s.d. of the long-term mean. Copepod species richness in the summer declined over the last five years of the data series by more than 1.0 s.d. of the long-term mean (Figure EI 29), suggesting generally good conditions for northern copepods and their predators. However, the mean of the last five years was within 1.0 s.d. of the full time series. The value for summer 2013 was below 1.0 s.d. of the full time series.

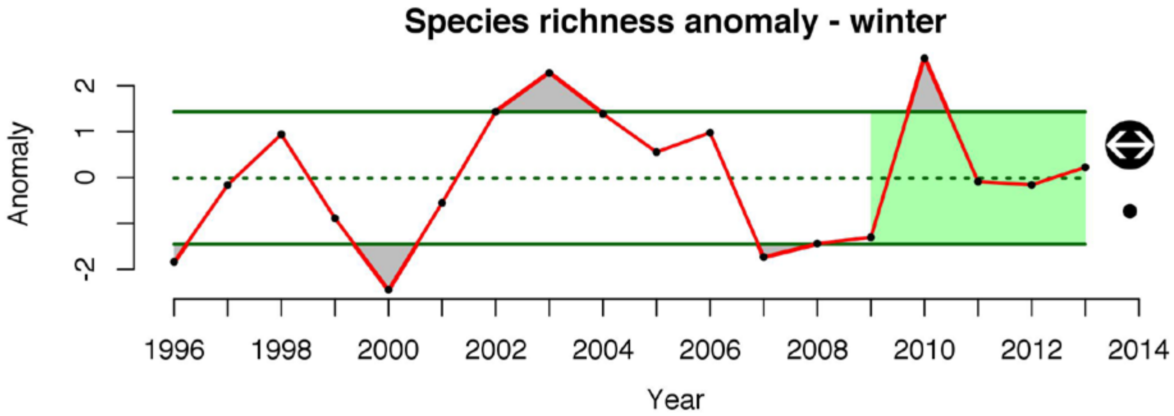

Figure EI 28. Species richness anomaly for copepods in the Northern California Current off Oregon during winter months (October – April) from 1996 to 2013. Data courtesy of Bill Peterson (bill.peterson@noaa.gov).

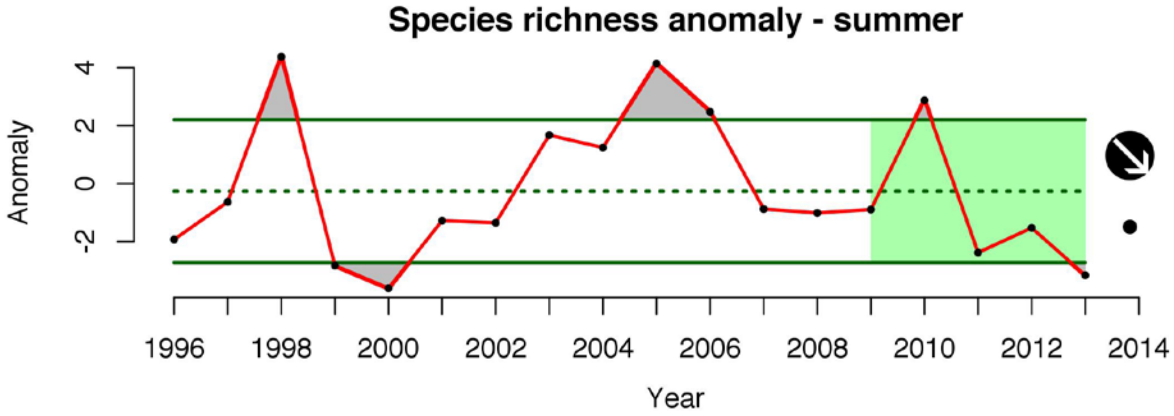

Figure EI 29. Species richness anomaly for copepods in the Northern California Current off Oregon during Summer months (May - September) from 1996 to 2013. Data courtesy of Bill Peterson (bill.peterson@noaa.gov).

SPECIES RICHNESS & DENSITY - GROUNDFISHES

Species richness for West Coast groundfishes increased steadily from just below 28 species per 3900 individuals in 2003 to over 32 species in 2009 (Figure EI 30). However, over the last five years, richness declined more than 1.0 s.d of the long-term mean to approximately 29 species per 3900 individuals. Nevertheless, the mean of the last five years was within 1.0 s.d. of the mean of the full time series. Given the fairly recent implementation of the West Coast Groundfish Bottom Trawl Survey (< 10 years), the increase in richness in the early years may be related in part to better species identification over the development of the survey. It is not clear why richness declined sharply from 2011 to 2012, and the trend should be examined in more detail.

As in other cases, trends north (Figure EI 31) and south (Figure EI 32) of Cape Mendocino differed somewhat. In both areas, richness increased initially before stabilizing. North of Cape Mendocino there was no trend over the last five years. However, south of Cape Mendocino richness declined over the last five years of the time series—markedly between 2011 and 2012.

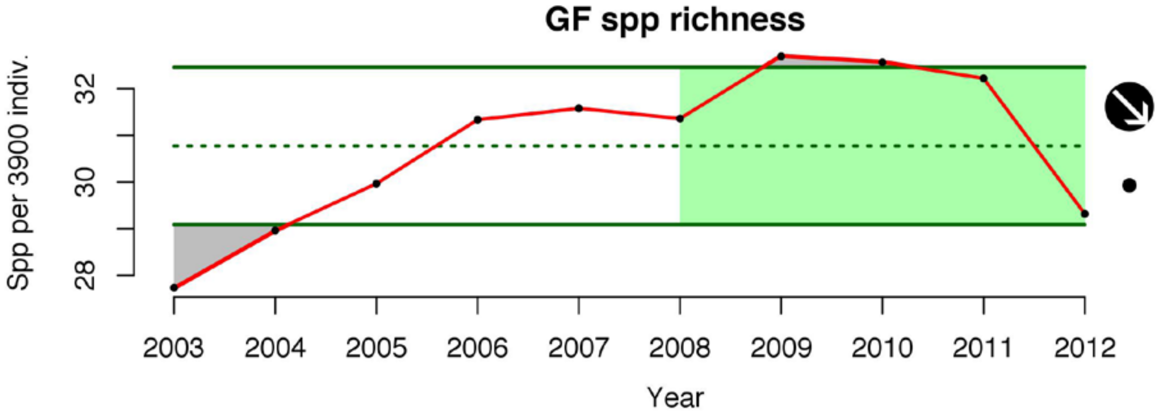

Figure EI 30. Species richness for groundfishes on the West Coast from the West Coast Groundfish Bottom Trawl Survey. Data underwent sample-based rarefaction and were then scaled to 3900 individuals to produce richeness estimates. Data courtesy of Beth Horness (Beth.Horness@noaa.gov).

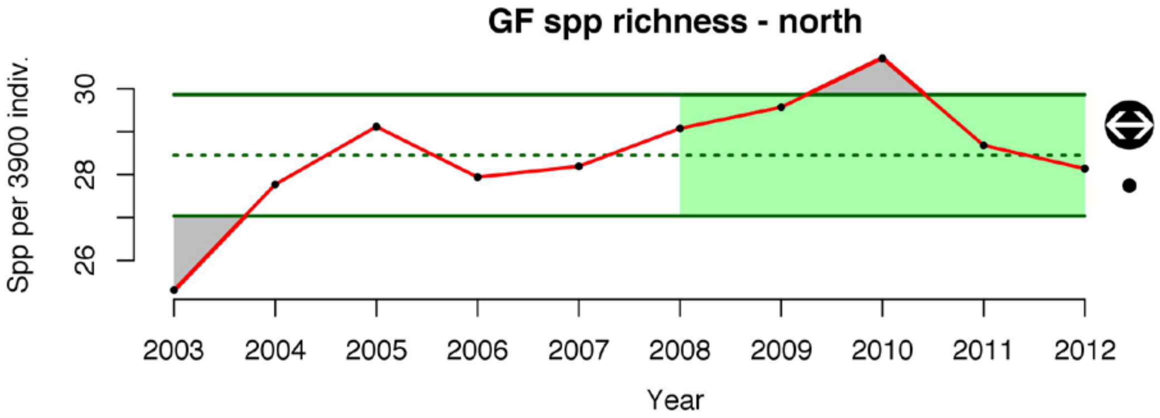

Figure EI 31. Species richness for groundfishes on the West Coast north of Cape Mendocino from the West Coast Groundfish Bottom Trawl Survey. Data underwent sample-based rarefaction and were then scaled to 3900 individuals to produce richness estimates. Data courtesy of Beth Horness (Beth.Horness@noaa.gov).

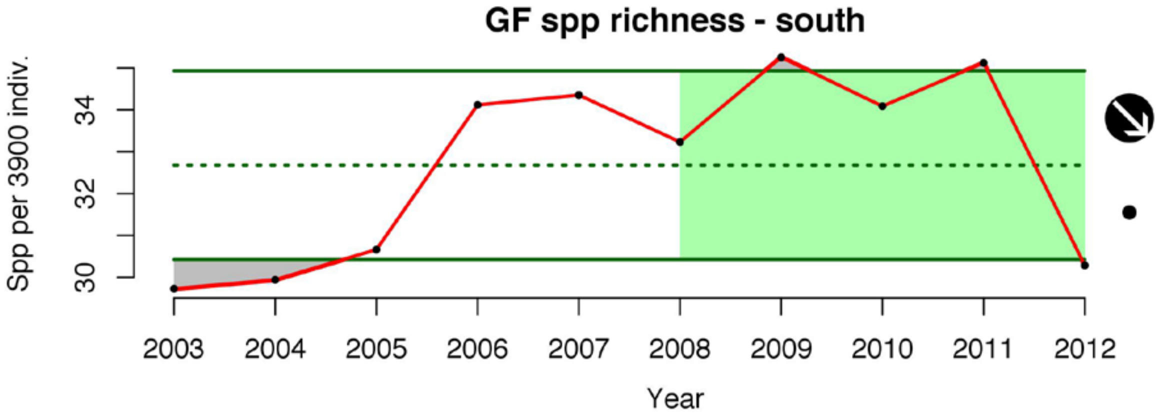

Figure EI 32. Species richness for groundfishes on the West Coast south of Cape Mendocino from the West Coast Groundfish Bottom Trawl Survey. Data underwent sample-based rarefaction and were then scaled to 3900 individuals to produce richness estimates. Data courtesy of Beth Horness (Beth.Horness@noaa.gov).

Species density initially increased from 2003 – 2006 then decreased sharply in 2007-2008 (Figure EI 33). Over the last five years, species density fluctuated but showed no overall trend, and the mean of the last five years was within 1.0 s.d. of the long-term mean. North of Cape Mendocino, there was a slight increase in species density over the last five years, but the increase was within 1.0 s.d. of the long-term mean (Figure EI 34). Trends south of Cape Mendocino were similar to the overall, coastwide tread (Figure EI 35).

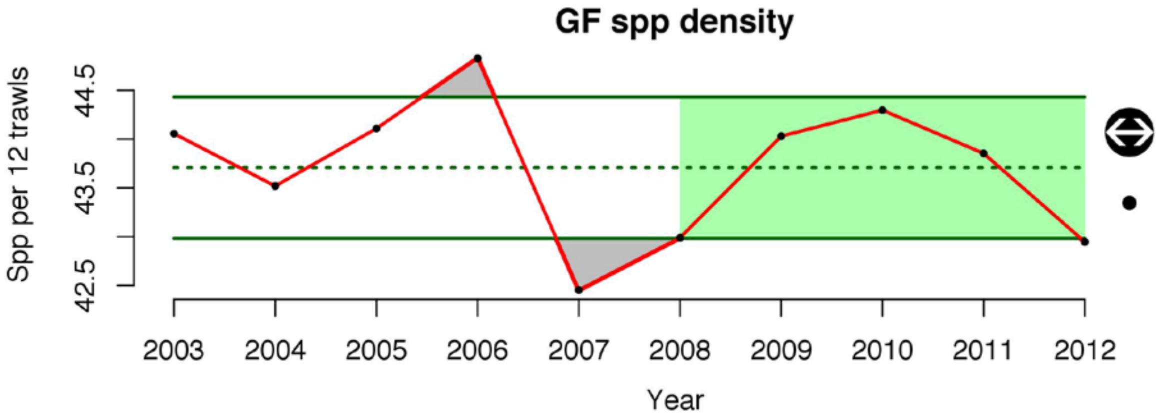

Figure EI 33. Area-weighted mean number of groundfish species per 12 trawls for 2003-2012 from the from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

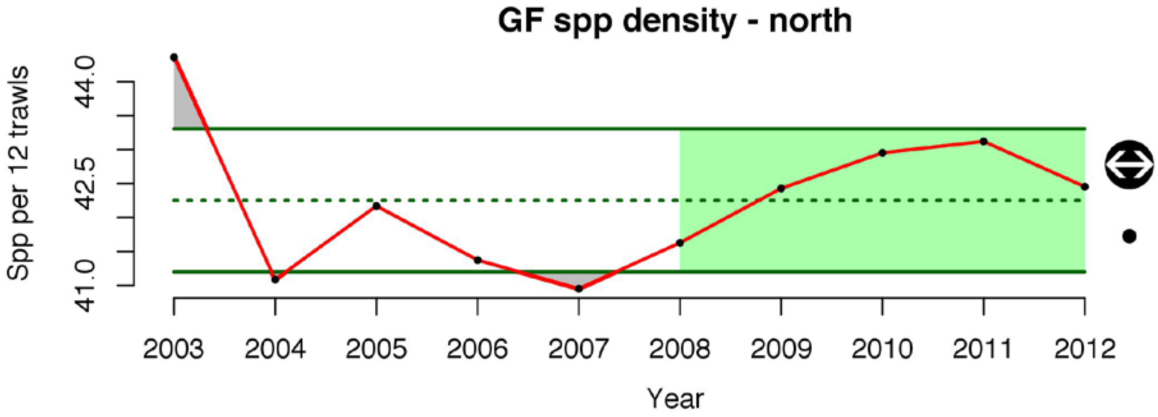

Figure EI 34. Area-weighted mean number of groundfish species per 12 trawls for 2003-2012 north of Cape Mendocino, from the West Coast Groundfish Bottom Trawl Survey; data courtesy of Beth Horness (Beth.Horness@noaa.gov).

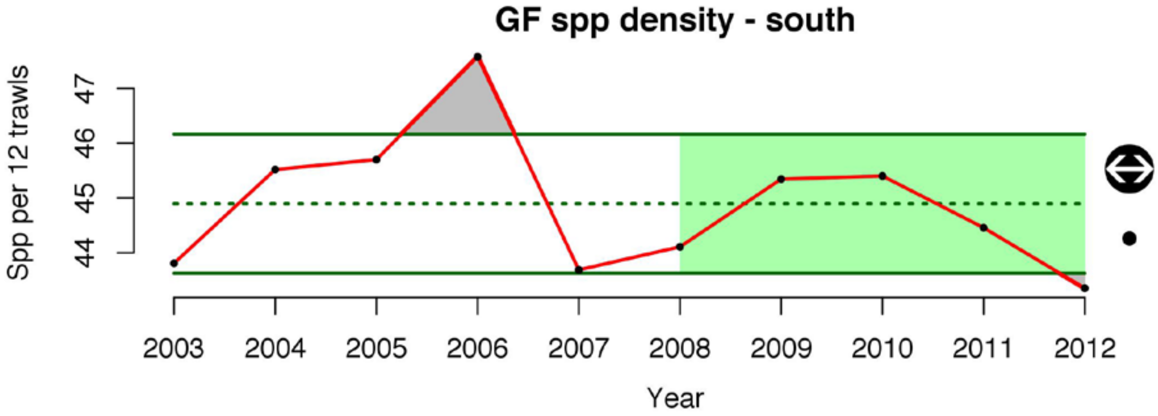

Figure EI 35. Area-weighted mean number of groundfish species per 12 trawls for 2003-2012 south of Cape Mendocino from the from the West Coast Groundfish Bottom Trawl Survey; data courtesy of Beth Horness (Beth.Horness@noaa.gov).

The differences between groundfish species richness and species density trends seen above are likely driven by the changing number of fishes captured in the trawl survey (Figure EI 36, Figure EI 37). Both the mean number fishes per trawl and the median number of fishes per trawl declined from 2003 to 2007, after which they remained stable. From 2007 – 2011 species density increased. During this period the number of individuals per haul remained stable, suggesting the increase was due to other processes.

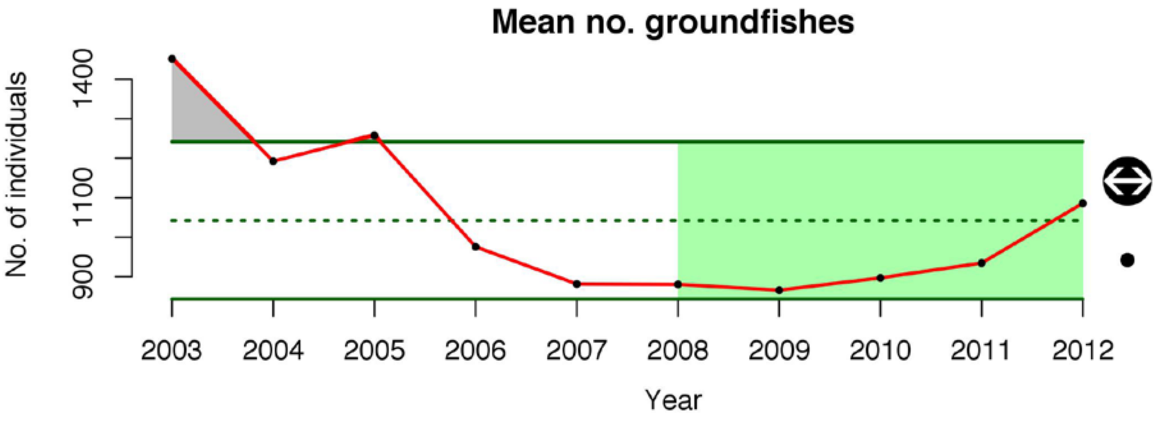

Figure EI 36. Mean number of groundfish individuals per trawl 2003-2012 from the from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

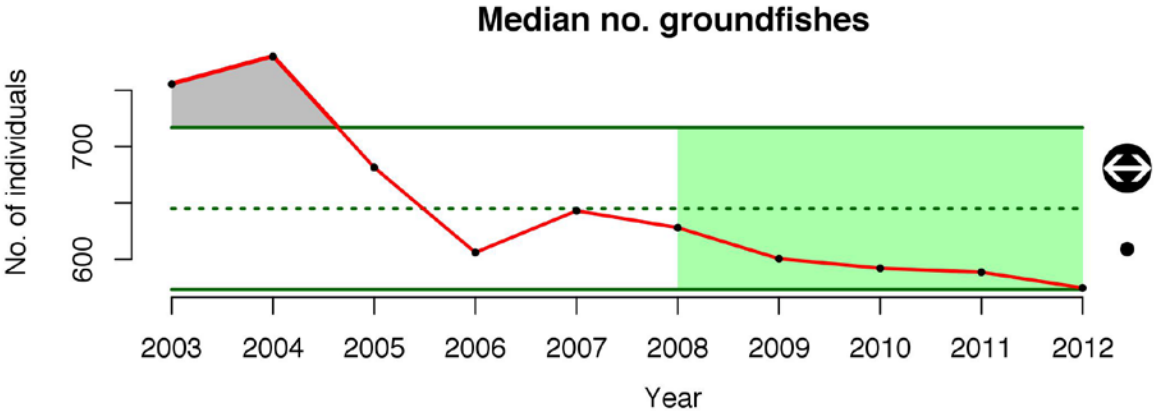

Figure EI 37. Median number of groundfish individuals per trawl 2003-2012 from the from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

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

SPECIES NUMBER—ICHTHYOPLANKTON

Species number in the CalCOFI spring ichthyoplankton surveys declined from a high in 2005 to low values between 2007 and 2010 (Figure EI 38). However, it increased rapidly in 2011 leading to an overall increase of more than 1.0 s.d. of time series. The mean of the last five years was within 1.0 s.d. of the full time series. However, given the short duration of the dataset, more emphasis should be based on the trend, not mean, over the last five years.

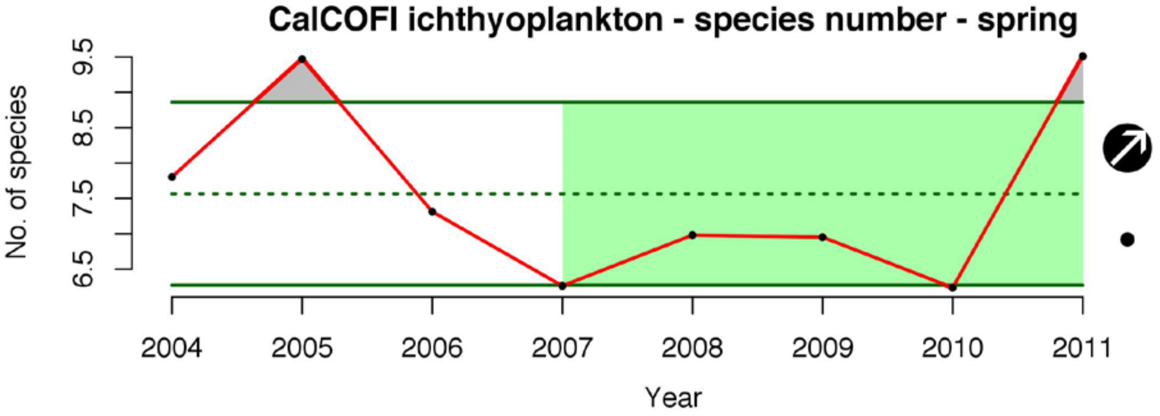

Figure EI 38. Number of species in CalCoFI ichthyoplankton surveys in the spring from 2004 – 2011. Data courtesy of Andrew Thompson (andrew.thompson@noaa.gov).

Species number in the summer CalCOFI surveys also declined to a low in 2007 prior to rebounding over the last five years (Figure EI 39). However, the final data point (2011) showed a decline to slightly lower than the long-term mean and the metric bears watching.

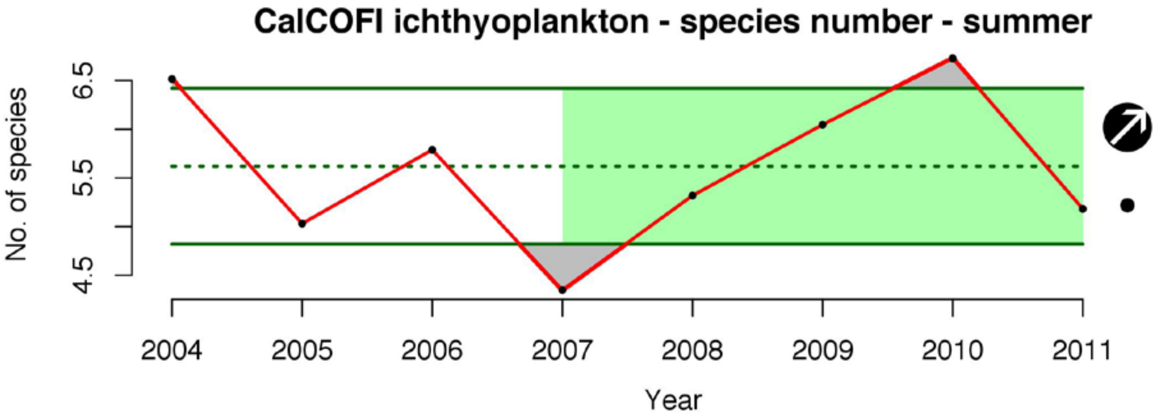

Figure EI 39. Number of species in CalCoFI ichthyoplankton surveys in the summer from 2004 – 2011. Data courtesy of Andrew Thompson (andrew.thompson@noaa.gov).

Ichthyoplankton species number off of Oregon in the spring also declined from high values in 2004 at the beginning of the time series to lows in 2009 and 2010 (Figure EI 40). While low over the last five years of the time series, species number is within 1.0 s.d. of long-term mean and showed no trend over the final five years.

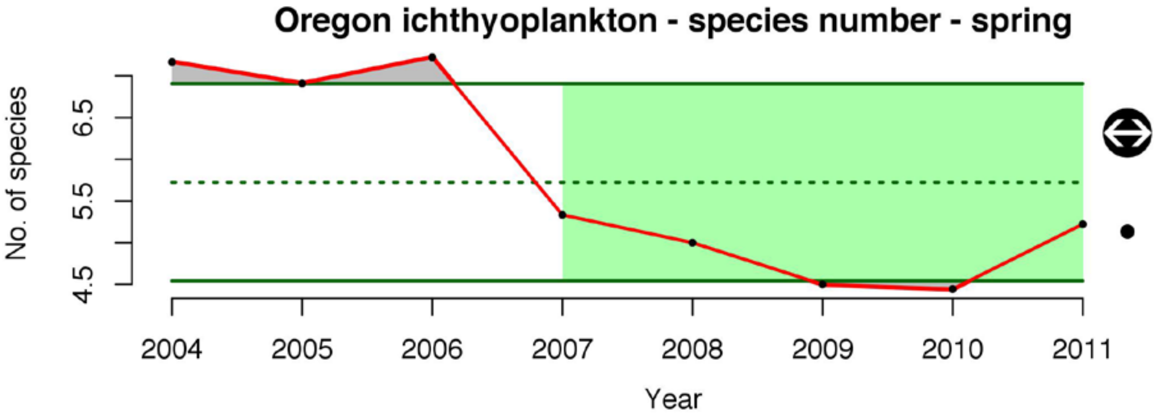

Figure EI 40. Number of species in ichthyoplankton surveys off Oregon in the spring from 2004 – 2011. Data courtesy of Andrew Thompson (andrew.thompson@noaa.gov).

Species number off Oregon in the summer declined over the last five years (2007-2011) by more than 1.0 s.d. of the full dataset (Figure EI 41). The 2011 value was well below 1.0 s.d. of the long-term mean.

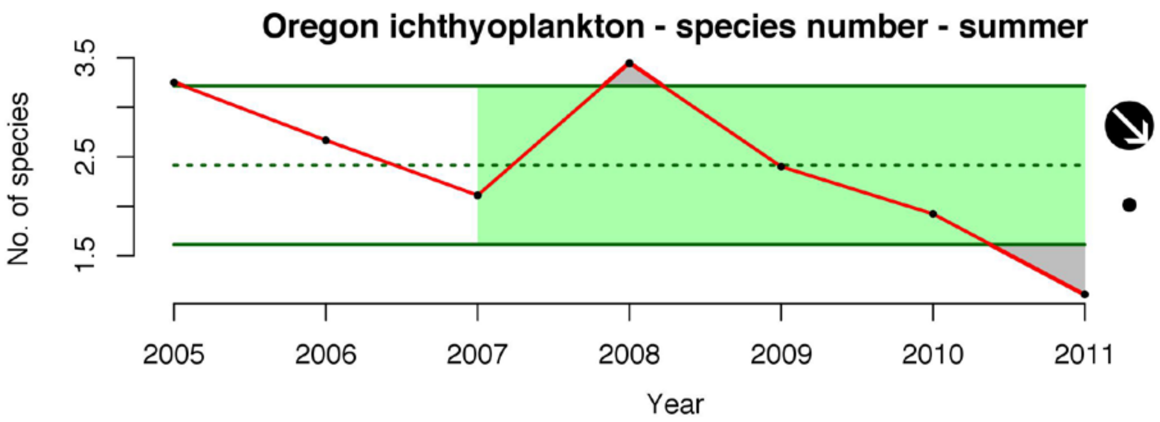

Figure EI 41. Number of species in surveys off Oregon in the spring from 2005 – 2011. Data courtesy of Andrew Thompson (andrew.thompson@noaa.gov).

DATA SOURCES AND METHODOLOGY

COASTAL PELAGIC FISHES

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

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

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

COPEPODS

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

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

Simpson Diversity (1-λ, Gini-Simpson index) was calculated for each month then averaged for each year by season: winter (Oct - April) and summer (May – Sept). Data were number of individuals by taxa per m^3. Winter means included data from the previous calendar year. That is, winter 2000 was the average of data from Oct – Dec 1999 and Jan – April 2000.

Species richness estimates of species per sample were not subject to rarefaction as in the case of groundfish. Enumeration of zooplankton data uses subsamples of a generally consistent number of individuals (200-400 individuals per sample for copepods, Peterson 2009), and therefore, does not require rarefaction to account for differences in sampling effort.

Northern copepod biomass anomaly—Data are courtesy of Bill Peterson (bill.peterson@noaa.gov). Seasonal estimates of the anomaly were calculated in winter (Oct - April) and summer (May – Sept). Winter means included data from the previous calendar year; for example, winter 2000 was the average of data from Oct – Dec 1999 and Jan – April 2000. See http://www.nwfsc.noaa.gov/research/divisions/fed/oeip/eb-copepod-anomalies.cfm for a discussion of the mechanisms behind the northern copepod biomass anomaly and for methodology in calculating the index.

GELATINOUS ZOOPLANKTON (JELLIES)

Data for gelatinous zooplankton come from two sources:

Oregon & Washington: Gelatinous zooplankton were surveyed in surface, shelf waters of the NCC from Newport, OR (44.6°N, 124.0° W) to Tatoosh Island, WA (48.4 N°, 124.7° W) along ten transect lines (Suchman et al. 2012). Five to ten stations were sampled on each transect line. At each station a Nordic 264 rope trawl (30 m wide x 19 m deep) was towed in surface waters for 30 min at 1.5-2.0 m s^{-1}. Mesh size was 162.6 cm at the throat to 9.8 cm at the cod end with a 6.1 m long, 0.87-cm mesh liner sewn into the cod end. Medusae were identified, counted and measured at sea. Total number of medusae per haul was estimated based on the total weight of species for the haul and mean weight of a subsample of at least 50 individuals. Number per haul was converted to number per km^2 based on the haul length and width of the net. The data presented here are for two dominant taxa (*Chrysaora fuscescens* and *Aequorea* sp.) for surveys conducted in June and September. Time series were standardized by subtracting the mean and dividing by the standard deviation. See Suchman et al. 2012 for further detail. Data are courtesy of Ric Brodeur (rick.brodeur@noaa.gov).

Central California: Data come from the central California rockfish recruitment survey (Wells et al. 2013). Data processing is detailed in Wells et al. (2013). In brief, time series data were standardized by subtracting the mean and dividing by the standard deviation.

Data are courtesy of John Field (john.field@noaa.gov).

GROUNDFISHES

Data for the groundfish time series come from the Northwest Fisheries Science Center's annual West Coast Bottom Trawl Survey (WCBTS, data courtesy of Beth Horness, (beth.horness@noaa.gov) (Keller et al. 2008). The survey is a depth-stratified, random sample that spans approximately 32 - 48.5° N and 55-1200 m depth for 2003-2012. Data were limited to those hauls deemed acceptable for stock assessment. Hauls from areas subsequently closed to sampling were not included in analyses. The data include 6338 trawls/hauls from 2003 - 2012 and 324 fish taxa identified to species. Of these, 3435 trawls were from north of Cape Mendocino and 2953 trawls were from south of Cape Mendocino. Two pairs of rockfishes were combined because of difficulty in discriminating between each species pair in the field: sunset rockfish *Sebastes crocotulus* and vermilion rockfish *S. miniatus* were combined into one taxon, and blackspotted rockfish S. *melanostictus* and rougheye rockfish *S. aleutianus* were combined into another taxon. Both combined taxa were included in species level analyses.

AREA-WEIGHTED MEANS (GROUNDFISHES)

Area-weighted means were calculated for mean trophic level, scavenger biomass ratio[6], Simpson diversity, and species richness because some areas of the shelf and slope are more heavily sampled than others and because the total bottom area of the shelf and slope for any given depth range varies with latitude (Table EI1).

Data (for both groundfishes and bottom area) were binned into five depth zones (<200, 201-600, 600-1200 m depths) and four latitude regions (south of Point Conception [32 - 34.5° N], Point Conception to Cape Mendocino [40.4° N], Cape Mendocino to Cape Blanco [42.5° N], and Cape Blanco to Cape Flattery [to 48.4511° N—the extent of the groundfish data]) based on previous analyses of groundfish assemblage structure (Tolimieri and Levin 2006, Tolimieri 2007). The areal extent of each depth x region bin was calculated from the U.S. Coastal Relief Model:

http://www.ngdc.noaa.gov/mgg/coastal/crm.html

The native coordinate system of these bathymetry data does not conserve area throughout the study region (e.g., a 1 X 1 degree area in the south is larger than a 1 X 1 degree area to the north). To correct this problem, we created a regular 0.1-degree grid over the study area and then re-projected this grid to a Cylindrical Equal-Area projection

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[6] Scavenger biomass ratio includes crab biomass but is included here since the data come from the groundfish trawl survey.

(units = meters, projection type = 3, longitude of the center of projection = -122 0' 0.00", latitude of the center of projection = 56 30' 0.00", Azimuth = 120.95, and Scale factor = 1). The new data layer had the correct area for each 0.1-degree grid cell. The total area of a given depth x region bin was calculated by summing the area of the relevant grid cells. Each depth x region bin was then assigned a weight equal to its proportion of the total area of all depth x region bins. These weights were then used to calculate the area-weighted mean for each groundfish-based indicator of ecological integrity.

Table EI 2. Depth and latitude bins uses in the area-weighted analyses showing total area of the bins, corresponding weights and total number of trawls from 2003 – 2012. See text for depth and latitude borders.

Depth zone	Latitude zone	Total area (km^2)	Weight	Trawls
Shelf	Flattery	36,394	0.231	1422
Shallow slope	Flattery	11,020	0.070	806
Deep slope	Flattery	10,916	0.069	496
Shelf	Blanco	5,407	0.034	264
Shallow slope	Blanco	2,182	0.014	200
Deep slope	Blanco	5,258	0.033	247
Shelf	Mendocino	16,689	0.106	763
Shallow slope	Mendocino	8,326	0.053	575
Deep slope	Mendocino	12,518	0.079	466
Shelf	Conception	10,176	0.064	364
Shallow slope	Conception	11,702	0.074	479
Deep slope	Conception	27,243	0.173	306

GROUNDFISH INDICATORS

Mean trophic level (MTL, Pauly and Watson 2005, 2010) was calculated as the biomass-weighted mean trophic level for each haul (Table EI 3), which was then used to calculate an area-weighted mean for the West Coast shelf and slope (see Area-weighted means, below). Information on trophic level was taken from Fishbase.org. Taxa included in the analyses were all fishes identified to the species level. Data were CPUE biomass (kg per km^2) by species per haul.

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

Area-adjusted MTL for groundfishes is presented in the results section above. For comparison, the raw MTL trend is shown below (Figure EI 42). While the overall trend is similar (a decline since 2003), there are important differences. Most importantly the absolute level of decline is quite different: 0.077 if adjusting for area versus 0.190 if not. A decrease in MTL of ~0.15 represents a decrease of 50% in the primary production required to support the assemblage. Therefore, failing to account for sample area overestimates the change in the trophic structure and energy requirements for the assemblage in question.

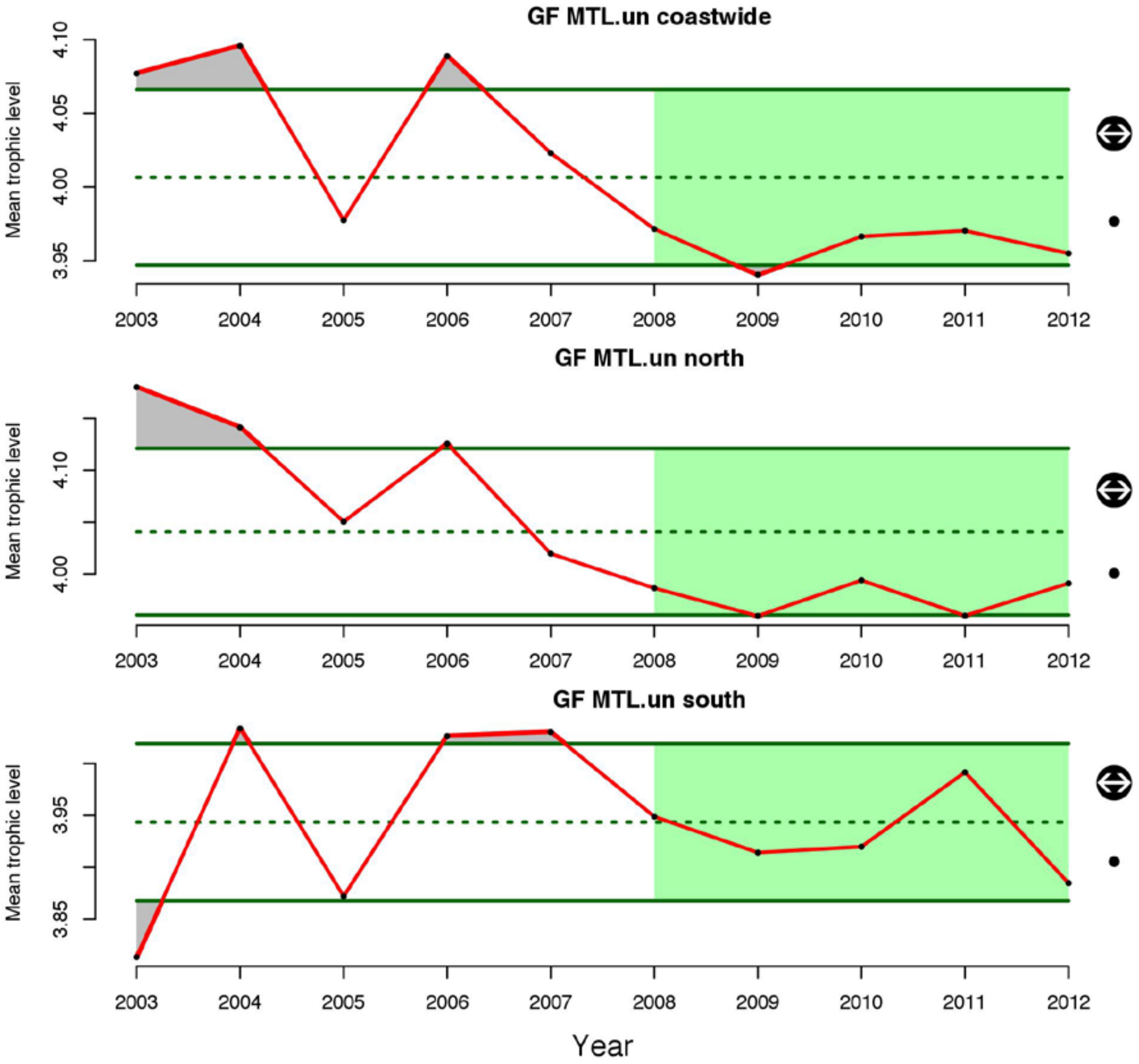

Figure EI 42a-c. Mean trophic level for West Coast groundfishes from 2003 – 2012. MTL was calculated for survey data without adjusting for sampling effort in different depth x latitude strata. The top pane shows coastwide MTL. The lower two panes show MTL for the regions north and south of Cape Mendocino (40.4° N). Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

Table EI 3. Groundfish species included in the analysis of mean trophic level. TL = Trophic level from Fishbase.org. KG = total biomass in the West Coast Groundfish Bottom Trawl Survey for 2003-2012 for depths ≤ 1200 m. Data are courtesy of Beth Horness (Beth.Horness@noaa.gov).

Order	Family	Species	Common Name	TL	KG
Pleuronectiformes	Pleuronectidae	*Microstomus pacificus*	Dover sole	3.3	285170.06
Gadiiformes	Merlucciidae	*Merluccius productus*	Pacific hake	4.3	119343.88
Scorpaeniformes	Scorpaenidae	*Sebastolobus altivelis*	longspine thornyhead	3.4	113339.45
Squaliformes	Squalidae	*Squalus acanthias*	spiny dogfish	4.3	99876.12
Scorpaeniformes	Anoplopomatidae	*Anoplopoma fimbria*	sablefish	3.8	88377.85
Pleuronectiformes	Paralichthyidae	*Citharichthys sordidus*	Pacific sanddab	3.5	72783.45

Order	Family	Species	Common Name	TL	KG
Scorpaeniformes	Scorpaenidae	*Sebastes goodei*	chilipepper	3.5	69512.06
Rajiformes	Rajidae	*Raja rhina*	longnose skate	3.8	63856.20
Pleuronectiformes	Pleuronectidae	*Atheresthes stomias*	arrowtooth flounder	4.3	61488.71
Pleuronectiformes	Pleuronectidae	*Glyptocephalus zachirus*	rex sole	3.2	59942.57
Scorpaeniformes	Scorpaenidae	*Sebastes diploproa*	splitnose rockfish	3.7	53479.68
Scorpaeniformes	Scorpaenidae	*Sebastolobus alascanus*	shortspine thornyhead	3.6	39965.62
Scorpaeniformes	Scorpaenidae	*Sebastes zacentrus*	sharpchin rockfish	3.6	34050.38
Chimaeriformes	Chiamaeridae	*Hydrolagus colliei*	spotted ratfish	3.7	32683.24
Scorpaeniformes	Hexagrammidae	*Ophiodon elongatus*	lingcod	4.3	30049.90
Gadiiformes	Macrouridae	*Coryphaenoides acrolepis*	Pacific grenadier	3.8	28005.09
Scorpaeniformes	Scorpaenidae	*Sebastes flavidus*	yellowtail rockfish	4.1	27934.79
Pleuronectiformes	Pleuronectidae	*Parophrys vetulus*	English sole	3.4	27348.26
Scorpaeniformes	Scorpaenidae	*Sebastes jordani*	shortbelly rockfish	3.2	25528.40
Scorpaeniformes	Scorpaenidae	*Sebastes saxicola*	stripetail rockfish	3.6	24686.86
Pleuronectiformes	Pleuronectidae	*Eopsetta jordani*	petrale sole	4.1	24563.38
Scorpaeniformes	Scorpaenidae	*Sebastes proriger*	redstripe rockfish	3.7	19659.21
Scorpaeniformes	Scorpaenidae	*Sebastes elongatus*	greenstriped rockfish	3.6	19280.61
Scorpaeniformes	Scorpaenidae	*Sebastes pinniger*	canary rockfish	3.8	17866.10
Gadiiformes	Macrouridae	*Albatrossia pectoralis*	giant grenadier	4.3	15871.14
Scorpaeniformes	Scorpaenidae	*Sebastes alutus*	Pacific ocean perch	3.5	15103.42
Scorpaeniformes	Scorpaenidae	*Sebastes crameri*	darkblotched rockfish	3.7	14637.72
Scorpaeniformes	Scorpaenidae	*Sebastes semicinctus*	halfbanded rockfish	3.5	12533.42
Rajiformes	Rajidae	*Raja binoculata*	big skate	3.92	11349.26
Argentiniformes	Alepocephalidae	*Alepocephalus tenebrosus*	California slickhead	3.5	10779.92
Pleuronectiformes	Pleuronectidae	*Lyopsetta exilis*	slender sole	3.4	10335.98
Pleuronectiformes	Pleuronectidae	*Hippoglossus stenolepis*	Pacific halibut	4.1	8605.01
Rajiformes	Rajidae	*Bathyraja kincaidii*	Bering skate	3.4	7826.25
Carcharhiniformes	Scyliorhinidae	*Apristurus brunneus*	brown cat shark	3.6	7661.75
Pleuronectiformes	Pleuronectidae	*Embassichthys bathybius*	deepsea sole	3.3	6613.33
Gadiiformes	Gadidae	*Gadus macrocephalus*	Pacific cod	4	6409.07
Perciformes	Zoarcidae	*Lycodes cortezianus*	bigfin eelpout	3.5	5455.26
Scorpaeniformes	Scorpaenidae	*Sebastes aurora*	aurora rockfish	3.6	4943.46
Perciformes	Sciaenidae	*Genyonemus lineatus*	white croaker	3.4	4364.22
Scorpaeniformes	Scorpaenidae	*Sebastes helvomaculatus*	rosethorn rockfish	3.6	4016.01
Scorpaeniformes	Scorpaenidae	*Sebastes paucispinis*	bocaccio	3.5	3942.88
Carcharhiniformes	Scyliorhinidae	*Parmaturus xaniurus*	filetail cat shark	3.8	3921.85
Perciformes	Zoarcidae	*Bothrocara brunneum*	twoline eelpout	3.6	3331.56
Scorpaeniformes	Scorpaenidae	*Sebastes melanostomus*	blackgill rockfish	3.7	3042.29
Scorpaeniformes	Scorpaenidae	*Sebastes entomelas*	widow rockfish	3.7	2922.55
Rajiformes	Rajidae	*Raja inornata*	California skate	3.7	2597.02
Gadiiformes	Moridae	*Antimora microlepis*	Pacific flatnose	3.5	2328.77
Scorpaeniformes	Scorpaenidae	*Sebastes chlorostictus*	greenspotted rockfish	3.7	2325.36

Order	Family	Species	Common Name	TL	KG
Scorpaeniformes	Scorpaenidae	*Sebastes caurinus*	copper rockfish	4.1	2324.90
Perciformes	Zoarcidae	*Lycodes diapterus*	black eelpout	3.4	2021.75
Torpadiniformes	Torpedinidae	*Torpedo californica*	Pacific electric ray	4.5	1776.32
Scorpaeniformes	Scorpaenidae	*Sebastes wilsoni*	pygmy rockfish	3.5	1700.66
Batrachoidiformes	Batrachoididae	*Porichthys notatus*	plainfin midshipman	4	1681.95
Scorpaeniformes	Scorpaenidae	*Sebastes melananostictus.aleutianus*	Sebastes melanostictus or Sebastes aleutianus	3.65	1670.77
Pleuronectiformes	Pleuronectidae	*Hippoglossoides elassodon*	flathead sole	3.6	1645.74
Perciformes	Embiotocidae	*Zalembius rosaceus*	pink seaperch	3.3	1618.05
Perciformes	Zoarcidae	*Lycenchelys crotalinus*	snakehead eelpout	3.5	1495.36
Perciformes	Stromateidae	*Peprilus simillimus*	Pacific pompano	4.1	1387.61
Clupeiformes	Clupeidae	*Alosa sapidissima*	American shad	3.5	1227.52
Scorpaeniformes	Scorpaenidae	*Sebastes ensifer*	swordspine rockfish	3.6	1110.36
Perciformes	Zoarcidae	*Lycodes pacificus*	blackbelly eelpout	3.3	1094.90
Scorpaeniformes	Sebastidae	*Sebastes crocotulus.miniatus*	vermilion_sunset	3.8	1088.49
Scorpaeniformes	Scorpaenidae	*Sebastes rufus*	bank rockfish	3.7	1052.72
Scorpaeniformes	Scorpaenidae	*Sebastes ruberrimus*	Yelloweye rockfish	4.4	943.38
Scorpaeniformes	Scorpaenidae	*Sebastes hopkinsi*	squarespot rockfish	3.6	907.70
Scorpaeniformes	Scorpaenidae	*Sebastes babcocki*	redbanded rockfish	3.7	877.35
Scorpaeniformes	Liparidae	*Careproctus melanurus*	blacktail snailfish	3.4	870.58
Pleuronectiformes	Pleuronectidae	*Pleuronichthys decurrens*	curlfin sole	3.8	850.75
Pleuronectiformes	Pleuronectidae	*Lepidopsetta bilineata*	southern rock sole	3.2	780.37
Perciformes	Carangidae	*Trachurus symmetricus*	jack mackerel	3.6	725.02
Clupeiformes	Engraulidae	*Engraulis mordax*	northern anchovy	3	716.64
Scorpaeniformes	Scorpaenidae	*Sebastes brevispinis*	silvergray rockfish	3.8	692.05
Scorpaeniformes	Scorpaenidae	*Scorpaena guttata*	California scorpionfish	3.8	671.78
Scorpaeniformes	Cottidae	*Icelinus filamentosus*	threadfin sculpin	3.5	620.37
Clupeiformes	Clupeidae	*Clupea pallasi*	Pacific herring	3.2	529.82
Scorpaeniformes	Scorpaenidae	*Sebastes rosenblatti*	greenblotched rockfish	3.7	394.77
Pleuronectiformes	Pleuronectidae	*Platichthys stellatus*	starry flounder	3.3	370.27
Carcharhiniformes	Scyliorhinidae	*Apristurus kampae*	longnose cat shark	3.7	326.38
Scorpaeniformes	Scorpaenidae	*Sebastes levis*	Cowcod	3.8	292.57
Scorpaeniformes	Scorpaenidae	*Sebastes reedi*	yellowmouth rockfish	3.71	292.13
Rajiformes	Rajidae	*Raja stellulata*	starry skate	3.7	281.88
Gadiiformes	Macrouridae	*Nezumia stelgidolepis*	California grenadier	4.4	276.99
Squantiformes	Squantinidae	*Squatina californica*	Pacific angel shark	4.1	269.53
Pleuronectiformes	Paralichthyidae	*Hippoglossina stomata*	bigmouth sole	3.8	260.89
Myxiniformes	Myxinidae	*Eptatretus deani*	black hagfish	3.8	235.67
Argentiniformes	Alepocephalidae	*Talismania bifurcata*	threadfin slickhead	3.3	226.57
Rajiformes	Rajidae	*Bathyraja abyssicola*	deepsea skate	3.99	212.82
Carcharhiniformes	Triakidae	*Galeorhinus galeus*	soupfin shark	4.2	207.48
Mylobatiformes	Myliobatidae	*Myliobatis californicus*	bat Ray	3.14	201.80

Order	Family	Species	Common Name	TL	KG
Gadiiformes	Gadidae	*Microgadus proximus*	Pacific tomcod	3.6	196.07
Pleuronectiformes	Pleuronectidae	*Psettichthys melanostictus*	sand sole	4.1	192.54
Pleuronectiformes	Pleuronectidae	*Pleuronichthys verticalis*	hornyhead turbot	3.1	186.89
Scorpaeniformes	Hexagrammidae	*Zaniolepis latipinnis*	longspine combfish	3.1	179.82
Pleuronectiformes	Paralichthyidae	*Paralichthys californicus*	California halibut	4.5	155.76
Scorpaeniformes	Hexagrammidae	*Hexagrammos decagrammus*	kelp greenling	3.6	154.97
Carcharhiniformes	Triakidae	*Mustelus henlei*	brown smoothhound	3.6	148.93
Gadiiformes	Macrouridae	*Nezumia liolepis*	smooth grenadier	3.3	143.45
Scorpaeniformes	Scorpaenidae	*Sebastes maliger*	quillback rockfish	3.8	137.37
Gadiiformes	Melanonidae	*Melanonus zugmayeri*	arrowtail	3.51	134.48
Rajiformes	Rajidae	*Bathyraja aleutica*	Aleutian skate	4.14	120.36
Gadiiformes	Gadidae	*Theragra chalcogramma*	walleye pollock	3.5	119.09
Aulopiformes	Synodontidae	*Synodus lucioceps*	California lizardfish	4.5	118.90
Scorpaeniformes	Scorpaenidae	*Sebastes borealis*	shortraker rockfish	3.9	116.06
Scorpaeniformes	Scorpaenidae	*Sebastes ovalis*	speckled rockfish	3.7	114.43
Perciformes	Embiotocidae	*Cymatogaster aggregata*	shiner perch	3	114.19
Myxiniformes	Myxinidae	*Eptatretus stouti*	Pacific hagfish	4.24	111.01
Scorpaeniformes	Cottidae	*Enophrys taurina*	bull sculpin	3.2	109.05
Hexanchiformes	Hexanchidae	*Hexanchus griseus*	sixgill shark	4.3	107.35
Pleuronectiformes	Paralichthyidae	*Citharichthys xanthostigma*	longfin sanddab	3.5	106.57
Osmeriformes	Osmeridae	*Thaleichthys pacificus*	eulachon	3.3	101.24
Pleuronectiformes	Pleuronectidae	*Isopsetta isolepis*	butter sole	3.6	95.79
Perciformes	Icosteidae	*Icosteus aenigmaticus*	ragfish	4.5	94.04
Scorpaeniformes	Hexagrammidae	*Zaniolepis frenata*	shortspine combfish	3.4	85.29
Scorpaeniformes	Cottidae	*Leptocottus armatus*	Pacific staghorn sculpin	3.5	75.13
Scorpaeniformes	Scorpaenidae	*Sebastes rubrivinctus*	flag rockfish	3.7	69.46
Argentiniformes	Argentinidae	*Argentina sialis*	Pacific argentine	3.1	67.33
Carcharhiniformes	Scyliorhinidae	*Cephaloscyllium ventriosum*	swell shark	3.9	66.98
Scorpaeniformes	Liparidae	*Careproctus cypselurus*	blackfin snailfish	3.32	64.93
Ophidiiformes	Ophidiidae	*Chilara taylori*	spotted cusk-eel	4.1	62.33
Scorpaeniformes	Scorpaenidae	*Sebastes auriculatus*	brown rockfish	4	61.95
Perciformes	Anarhichadidae	*Anarrhichthys ocellatus*	wolf-eel	3.5	59.10
Perciformes	Serranidae	*Paralabrax nebulifer*	barred sand bass	3.5	59.05
Osmeriformes	Osmeridae	*Allosmerus elongatus*	whitebait smelt	3.2	56.88
Clupeiformes	Clupeidae	*Sardinops sagax*	Pacific sardine	2.4	49.71
Scorpaeniformes	Scorpaenidae	*Sebastes rosaceus*	rosy rockfish	3.6	49.49
Perciformes	Cryptacanthodidae	*Cryptacanthodes giganteus*	giant wrymouth	3.27	49.30
Gadiiformes	Macrouridae	*Coryphaenoides cinereus*	popeye grenadier	3.66	41.86
Ophidiiformes	Ophidiidae	*Lamprogrammus niger*	paperbone cusk-eel	3.72	33.48
Scorpaeniformes	Scorpaenidae	*Sebastes umbrosus*	honeycomb rockfish	3.6	33.40
Scorpaeniformes	Scorpaenidae	*Sebastes simulator*	pinkrose rockfish	3.6	32.02
Gadiiformes	Moridae	*Physiculus rastrelliger*	hundred fathom codling	3.4	30.28

Order	Family	Species	Common Name	TL	KG
Perciformes	Scombridae	*Scomber japonicus*	chub mackerel	3.1	30.20
Scorpaeniformes	Psychrolutidae	*Psychrolutes phrictus*	blob sculpin	3.5	26.34
Carcharhiniformes	Triakidae	*Mustelus californicus*	gray smoothhound	3.5	26.31
Argentiniformes	Microstomatidae	*Leuroglossus stilbius*	California smoothtounge	3.26	26.28
Scorpaeniformes	Scorpaenidae	*Sebastes emphaeus*	Puget Sound rockfish	3.23	25.36
Scorpaeniformes	Scorpaenidae	*Sebastes dalli*	calico rockfish	3.53	24.01
Carcharhiniformes	Carcharhinidae	*Prionace glauca*	blue shark	4.2	22.95
Stomiiformes	Stomiidae	*Tactostoma macropus*	longfin dragonfish	4.2	22.79
Lampriformes	Trachipteridae	*Trachipterus altivelis*	king-of-the-salmon	3.9	21.29
Squaliformes	Etmopteridae	*Centroscyllium nigrum*	combtooth dogfish	3.9	21.03
Scorpaeniformes	Agonidae	*Bathyagonus nigripinnis*	blackfin poacher	3.25	20.70
Argentiniformes	Microstomatidae	*Bathylagus milleri*	robust blacksmelt	3.21	19.39
Scorpaeniformes	Scorpaenidae	*Sebastes constellatus*	starry rockfish	3.7	18.92
Stomiiformes	Stomiidae	*Stomias atriventer*	blackbelly dragonfish	4	18.48
Perciformes	Uranoscopidae	*Kathetostoma averruncus*	smooth stargazer	4.3	17.59
Argentiniformes	Microstomatidae	*Bathylagus pacificus*	Pacific blacksmelt	3.3	13.60
Osmeriformes	Osmeridae	*Spirinchus starksi*	night smelt	3.5	11.97
Scorpaeniformes	Scorpaenidae	*Sebastes nigrocinctus*	tiger rockfish	3.5	11.65
Pleuronectiformes	Pleuronectidae	*Pleuronichthys ritteri*	spotted turbot	3.2	10.50
Gadiiformes	Macrouridae	*Malacocephalus laevis*	softhead grenadier	4.2	8.76
Scorpaeniformes	Scorpaenidae	*Sebastes macdonaldi*	Mexican rockfish	3.7	8.02
Scorpaeniformes	Scorpaenidae	*Sebastes mystinus*	blue rockfish	2.8	7.85
Scorpaeniformes	Scorpaenidae	*Sebastes lentiginosus*	freckled rockfish	3.5	7.71
Pleuronectiformes	Paralichthyidae	*Xystreurys liolepis*	fantail sole	3.5	7.59
Scorpaeniformes	Triglidae	*Prionotus stephanophrys*	lumptail searobin	3.5	7.39
Scorpaeniformes	Liparidae	*Paraliparis dactylosus*	red snailfish	3.46	7.19
Perciformes	Trichiuridae	*Lepidopus xantusi*	silver scabbardfish	3.85	5.92
Pleuronectiformes	Pleuronectidae	*Lepidopsetta polyxystra*	northern rock sole	3.29	5.90
Scorpaeniformes	Agonidae	*Xeneretmus latifrons*	blacktip poacher	3.2	5.79
Scorpaeniformes	Cottidae	*Hemilepidotus spinosus*	brown Irish lord	3.5	5.70
Perciformes	Embiotocidae	*Hyperprosopon anale*	spotfin surfperch	3.3	5.62
Stomiiformes	Stomiidae	*Aristostomias scintillans*	shining loosejaw	3.5	5.58
Beryciformes	Anoplogastridae	*Anoplogaster cornuta*	fangtooth	4	5.53
Scorpaeniformes	Agonidae	*Bathyagonus pentacanthus*	bigeye poacher	3.2	5.39
Scorpaeniformes	Liparidae	*Elassodiscus caudatus*	humpback snailfish	3.31	5.05
Scorpaeniformes	Cottidae	*Scorpaenichthys marmoratus*	cabezon	3.6	4.95
Myctophiformes	Myctophidae	*Stenobrachius leucopsarus*	northern lampfish	3.2	4.70
Scorpaeniformes	Cottidae	*Bolinia euryptera*	broadfin sculpin	3.45	4.60
Scorpaeniformes	Cottidae	*Hemilepidotus hemilepidotus*	red Irish lord	3.5	4.50
Scorpaeniformes	Cottidae	*Icelinus fimbriatus*	fringed sculpin	3.7	4.47
Stephanoberyciformes	Melamphaidae	*Poromitra crassiceps*	crested bigscale	3.1	4.33
Scorpaeniformes	Agonidae	*Chesnonia verrucosa*	warty poacher	3.2	4.25

Order	Family	Species	Common Name	TL	KG
Tetradontiformes	Molidae	*Mola mola*	ocean sunfish	4	4.15
Scorpaeniformes	Scorpaenidae	*Sebastes melanops*	black rockfish	4.4	4.11
Rajiformes	Rhinobatidae	*Zapteryx exasperata*	bandedguitarfish	3.5	4.10
Osmeriformes	Osmeridae	*Hypomesus pretiosus*	surf smelt	3.4	3.90
Scorpaeniformes	Liparidae	*Careproctus gilberti*	smalldisk snailfish	3.3	3.72
Scorpaeniformes	Liparidae	*Paraliparis rosaceus*	rosy snailfish	3.7	3.56
Scorpaeniformes	Cottidae	*Chitonotus pugetensis*	roughback sculpin	3.5	3.41
Argentiniformes	Platytroctidae	*Sagamichthys abei*	shining tubeshoulder	3.1	3.35
Scorpaeniformes	Cottidae	*Enophrys bison*	buffalo sculpin	3.3	3.23
Scorpaeniformes	Scorpaenidae	*Sebastes gilli*	bronzespotted rockfish	3.8	3.20
Scorpaeniformes	Cottidae	*Radulinus asprellus*	slim sculpin	3.4	3.08
Perciformes	Trichiuridae	*Aphanopus carbo*	black scabbardfish	4.48	3.08
Scorpaeniformes	Scorpaenidae	*Sebastes carnatus*	gopher rockfish	3.6	2.97
Perciformes	Embiotocidae	*Amphistichus argenteus*	barred surfperch	3.5	2.85
Perciformes	Centrolphidae	*Icichthys lockingtoni*	medusafish	3.7	2.79
Stomiiformes	Stomiidae	*Idiacanthus antrostomus*	Pacific blackdragon	3.8	2.75
Ophidiiformes	Bythitidae	*Cataetyx rubrirostris*	rubynose brotula	3.5	2.74
Perciformes	Embiotocidae	*Phanerodon furcatus*	White Surfperch	3.4	2.67
Scorpaeniformes	Cottidae	*Gymnocanthus tricuspis*	arctic staghorn sculpin	3.46	2.35
Perciformes	Zoarcidae	*Lycodapus fierasfer*	blackmouth eelpout	3.3	2.32
Scorpaeniformes	Cottidae	*Clinocottus acuticeps*	sharpnose sculpin	3.5	2.26
Perciformes	Bathymasteridae	*Ronquilus jordani*	northern ronquil	3.1	2.26
Argentiniformes	Opisthoproctidae	*Macropinna microstoma*	barreleye	3.3	2.22
Ophidiiformes	Bythitidae	*Brosmophycis marginata*	red brotula	3.5	2.21
Perciformes	Embiotocidae	*Damalichthys vacca*	pile perch	3.68	2.12
Scorpaeniformes	Psychrolutidae	*Malacocottus kincaidi*	blackfin sculpin	3.39	1.97
Scorpaeniformes	Agonidae	*Odontopyxis trispinosa*	pygmy poacher	3.2	1.97
Anguilliformes	Serrivomeridae	*Serrivomer sector*	sawtooth eel	3.8	1.95
Argentiniformes	Microstomatidae	*Leuroglossus schmidti*	northern smoothtongue	3.12	1.91
Chimaeriformes	Rhinochimaeridae	*Harriotta raleighana*	Pacific longnose chimaera	3.55	1.80
Scorpaeniformes	Cottidae	*Icelinus burchami*	dusky sculpin	3.5	1.73
Scorpaeniformes	Cottidae	*Jordania zonope*	longfin sculpin	3.4	1.63
Perciformes	Chiasmodontidae	*Chiasmodon niger*	black swallower	4.2	1.54
Scorpaeniformes	Liparidae	*Paraliparis cephalus*	swellhead snailfish	3.38	1.50
Anguilliformes	Nemichthyidae	*Avocettina infans*	blackline snipe eel	3.5	1.40
Osmeriformes	Osmeridae	*Spirinchus thaleichthys*	longfin smelt	3.2	1.32
Aulopiformes	Paralepididae	*Magnisudis atlantica*	duckbill barracudina	4.1	1.27
Scorpaeniformes	Liparidae	*Paraliparis pectoralis*	broadfin snailfish	3.6	1.23
Petromyzontiformes	Petromyzontidae	*Lampetra tridentata*	Pacific lamprey	4.5	1.23
Perciformes	Zoarcidae	*Bothrocara molle*	soft eelpout	3.4	1.20
Stomiiformes	Stomiidae	*Borostomias panamensis*	Panama snaggletooth	3.1	1.20

Order	Family	Species	Common Name	TL	KG
Stomiiformes	Sternoptychidae	*Argyropelecus affinis*	slender hatchetfish	3.1	1.18
Anguilliformes	Nettastomatidae	*Facciolella gilbertii*	dogface witch-eel	3.4	1.16
Scorpaeniformes	Agonidae	*Agonopsis vulsa*	northern spearnose poacher	3.3	1.12
Perciformes	Zoarcidae	*Lycodapus endemoscotus*	deepwater eelpout	3.3	1.10
Squaliformes	Somniosidae	*Scymnodon squamulosus*	velvet dogfish	4	1.00
Argentiniformes	Alepocephalidae	*Bajacalifornia burragei*	sharpchin slickhead	3.3	0.99
Lophiiformes	gigantactinidae	*Gigantactis vanhoeffeni*	whipnose	4.51	0.96
Myctophiformes	Lampanyctinae	*Nannobrachium ritteri*	broadfin lampfish	4.4	0.92
Aulopiformes	Scopelarchidae	*Benthalbella dentata*	northern pearleye	4.5	0.89
Pleuronectiformes	Pleuronectidae	*Pleuronichthys coenosus*	C-O sole	3.2	0.85
Myctophiformes	Neoscopelidae	*Scopelengys tristis*	blackchin	3.1	0.78
Stephanoberyciformes	Melamphaidae	*Melamphaes lugubris*	highsnout bigscale	3.8	0.78
Gadiiformes	Moridae	*Halargyreus johnsoni*	slender codling	3.38	0.68
Anguilliformes	Nemichthyidae	*Nemichthys scolopaceus*	slender snipe eel	3.5	0.67
Scorpaeniformes	Triglidae	*Bellator xenisma*	splitnose searobin	3.4	0.62
Osmeriformes	Osmeridae	*Osmerus mordax*	rainbow smelt	3	0.61
Scorpaeniformes	Cottidae	*Icelinus tenuis*	spotfin sculpin	3.6	0.60
Perciformes	Zoarcidae	*Lycenchelys camchatica*	Kamchatka eelpout	3.3	0.57
Perciformes	Embiotocidae	*Phanerodon atripes*	sharpnose surfperch	3.4	0.56
Scorpaeniformes	Liparidae	*Rhinoliparis attenuatus*	slim snailfish	3.36	0.52
Perciformes	Trichiuridae	*Lepidopus fitchi*	scabbardfish	4.1	0.50
Perciformes	Zoarcidae	*Lycodapus mandibularis*	pallid eelpout	3.3	0.49
Myctophiformes	Myctophidae	*Tarletonbeania crenularis*	blue lanternfish	3.1	0.48
Gadiiformes	Macrouridae	*Coryphaenoides filifer*	filamented grenadier	4.5	0.47
Scorpaeniformes	Liparidae	*Rhinoliparis barbulifer*	longnose snailfish	3.31	0.46
Perciformes	Embiotocidae	*Rhacochilus toxotes*	rubberlip Surfperch	3.41	0.41
Gadiiformes	Macrouridae	*Coelorinchus scaphopsis*	shoulder spot grenadier	3.55	0.40
Perciformes	Caristiidae	*Caristius macropus*	manefish	4.2	0.40
Scorpaeniformes	Agonidae	*Xeneretmus leiops*	smootheye poacher	3.3	0.39
Argentiniformes	Microstomatidae	*Nansenia candida*	bluethroat argentine	3.3	0.38
Scorpaeniformes	Scorpaenidae	*Sebastes variegatus*	harlequin rockfish	3.6	0.38
Myctophiformes	Myctophidae	*Symbolophorus californiensis*	California lanternfish	3.1	0.37
Perciformes	Sciaenidae	*Seriphus politus*	queenfish	3.7	0.35
Anguilliformes	Nettastomatidae	*Venefica tentaculata*	Venefica tentaculata	3.48	0.34
Stomiiformes	Sternoptychidae	*Sternoptyx diaphana*	longspine hatchetfish	3.4	0.34
Anguilliformes	Nemichthyidae	*Nemichthys larseni*	pale snipe eel	3.42	0.32
Lophiiformes	Oneirodidae	*Chaenophryne draco*	smooth dreamer	3.86	0.32
Scorpaeniformes	Agonidae	*Podothecus acipenserinus*	sturgeon poacher	3.39	0.30
Scorpaeniformes	Hemitripteridae	*Nautichthys oculofasciatus*	sailfin sculpin	4.1	0.28
Lophiiformes	Oneirodidae	*Oneirodes acanthias*	spiny dreamer	3.1	0.28
Aulopiformes	Paralepididae	*Arctozenus risso*	ribbon barracudina	3.2	0.27

Order	Family	Species	Common Name	TL	KG
Aulopiformes	Notosudidae	*Scopelosaurus harryi*	scaly paperbone	3.1	0.26
Scorpaeniformes	Psychrolutidae	*Dasycottus setiger*	spinyhead sculpin	3.54	0.25
Lophiiformes	Melanocetidae	*Melanocetus johnsonii*	common blackdevil	4.1	0.24
Anguilliformes	Serrivomeridae	*Serrivomer jesperseni*	crossthroat Snipe Eel	3.69	0.24
Osmeriformes	Opostoproctidae	*Dolichopteryx longipes*	brownsnout spookfish	3	0.24
Perciformes	Zoarcidae	*Lycodema barbatum*	bearded eelpout	3.3	0.24
Argentiniformes	Platytroctidae	*Maulisia mauli*	Maulisia mauli	3.25	0.22
Perciformes	Embiotocidae	*Amphistichus rhodoterus*	redtail surfperch	3.4	0.22
Perciformes	Tetragonuridae	*Tetragonurus cuvieri*	smalleye squaretail	3.8	0.22
Perciformes	Stichaeidae	*Poroclinus rothrocki*	whitebarred prickleback	3.1	0.20
Pleuronectiformes	Cynoglossidae	*Symphurus atricauda*	California toungefish	3.39	0.20
Perciformes	Zoarcidae	*Lycodes palearis*	wattled eelpout	3.48	0.19
Ophidiiformes	Ophidiidae	*Dicrolene filamentosa*	threadfin cusk-eel	3.61	0.19
Lophiiformes	Ceratiidae	*Cryptopsaras couesii*	triplewart sea devil	4.5	0.18
Scorpaeniformes	Rhamphocottidae	*Rhamphocottus richardsoni*	grunt sculpin	3.42	0.17
Scorpaeniformes	Agonidae	*Agonopsis sterletus*	southern spearnose poacher	3.2	0.16
Ophidiiformes	Ophidiidae	*Ophidion scrippsae*	basketweave cusk-eel	3.5	0.16
Scorpaeniformes	Cottidae	*Triglops macellus*	roughspine sculpin	3.32	0.15
Aulopiformes	Paralepididae	*Lestidiops ringens*	slender barracudina	4.1	0.15
Perciformes	Chiasmodontidae	*Kali indica*	shortnose swallower	3.47	0.15
Lophiiformes	Oneirodidae	*Oneirodes thompsoni*	Oneirodes thompsoni	4.2	0.14
Stomiiformes	Stomiidae	*Bathophilus flemingi*	highfin dragonfish	3.5	0.14
Lophiiformes	Caulophrynidae	*Caulophryne jordani*	fanfin seadevil	4.02	0.12
Scorpaeniformes	Scorpaenidae	*Sebastes serriceps*	tree rockfish	3.6	0.10
Scorpaeniformes	Liparidae	*Careproctus colletti*	Alaska snailfish	3.34	0.10
Perciformes	Sciaenidae	*Cheilotrema saturnum*	black croaker	3.6	0.10
Perciformes	Embiotocidae	*Embiotoca lateralis*	striped surfperch	3.4	0.09
Perciformes	Zoarcidae	*Melanostigma pammelas*	midwater eelpout	3.1	0.09
Scorpaeniformes	Liparidae	*Elassodiscus tremebundus*	blacklip snailfish	3.57	0.09
Scorpaeniformes	Liparidae	*Liparis fucensis*	slipskin snailfish	3.5	0.08
Perciformes	Trichodontidae	*Trichodon trichodon*	Pacific sandfish	3.7	0.08
Perciformes	Ammodytidae	*Ammodytes hexapterus*	Pacific sand lance	3.1	0.07
Pleuronectiformes	Paralichthyidae	*Citharichthys stigmaeus*	speckled sanddab	3.4	0.06
Perciformes	Chiasmodontidae	*Kali normani*	needletooth swallower	3.43	0.06
Scorpaeniformes	Cottidae	*Radulinus taylori*	spinynose sculpin	3.25	0.06
Perciformes	Cryptacanthodidae	*Lyconectes aleutensis*	dwarf wrymouth	3	0.06
Scorpaeniformes	Scorpaenidae	*Sebastes serranoides*	olive rockfish	3.9	0.05
Argentiniformes	Opisthoproctidae	*Bathylychnops exilis*	javelin spookfish	4.1	0.04
Scorpaeniformes	Cottidae	*Paricelinus hopliticus*	thornback sculpin	3.4	0.04
Osmeriformes	Platytroctidae	*Platytroctes apus*	legless searsid	3.2	0.04
Perciformes	Trichiuridae	*Benthodesmus pacificus*	North-Pacific frostfish	4.3	0.04

Order	Family	Species	Common Name	TL	KG
Scorpaeniformes	Hexagrammidae	*Oxylebius pictus*	painted greenling	3.4	0.04
Perciformes	Chaenopsidae	*Neoclinus blanchardi*	sarcastic fringehead	2.2	0.04
Lampriformes	Trachipteridae	*Desmodema lorum*	whiptail ribbonfish	4.2	0.04
Scorpaeniformes	Liparidae	*Nectoliparis pelagicus*	tadpole snailfish	3.3	0.03
Argentiniformes	Alepocephalidae	*Bajacalifornia erimoensis*	Bajacalifornia erimoensis	3.35	0.02
Perciformes	Zoarcidae	*Lycodapus dermatinus*	looseskin eelpout	3.2	0.02
Stomiiformes	Sternoptychidae	*Argyropelecus lychnus*	tropical hatchetfish	3	0.02
Perciformes	Percichthyidae	*Howella sherborni*	Howella sherborni	3.1	0.02
Scorpaeniformes	Psychrolutidae	*Psychrolutes paradoxus*	tadpole sculpin	3.17	0.02
Scorpaeniformes	Cottidae	*Icelinus borealis*	northern sculpin	3.6	0.02
Perciformes	Howellidae	*Howella brodiei*	pelagic basset	3.23	0.02
Perciformes	Clinidae	*Gibbonsia metzi*	striped kelpfish	3.39	0.02
Perciformes	Stichaeidae	*Plectobranchus evides*	bluebarred prickleback	3.1	0.02
Perciformes	Zoarcidae	*Maynea californica*	persimmon eelpout	3.285	0.01
Perciformes	Zoarcidae	*Lycodes brevipes*	shortfin eelpout	4.01	0.01
Lophiiformes	Oneirodidae	*Chaenophryne longiceps*	Chaenophryne longiceps	4.1	0.01
Atheriniformes	Atherinopsidae	*Atherinops affinis*	topsmelt	2.8	0.01

Ratio of scavenger biomass to total biomass was calculated from the West Coast Bottom Trawl Survey (WCBTS), which provides quantitative biomass data for groundfishes, as well as for several crab species for 2003-2011. We followed Brand et al. (2007) and Horne et al. (2010) in defining large crabs, large demersal sharks and grenadiers as scavengers (Table EI4). Further detail was taken from Yeh and Drazen (2011) who used baited-cameras to evaluate scavenger ecology on the California slope, and from Buckley et al. (1999) who examine food habits of several groundfishes. Scavengers are generally defined as active foragers of carrion (Britton and Morton 1994). Many of the species on the list are predators that responded strongly to baited cameras (grenadiers) or had large amounts of fisheries offal in their diet (thornyheads and sablefish). While carrion may not normally make up a substantial portion of the diets of these animals in the absence of anthropogenic influences, part of the objective of monitoring scavenger biomass is to track the effects of fisheries on the ecosystem. Therefore, it is relevant to include taxa that respond strongly to these activities. Scavenger biomass ratio was calculated for each haul by dividing the sum of scavenger biomass by the sum of total biomass for each haul. This ratio was then used to calculate the area-weighted mean scavenger: total biomass as for other groundfish indicators (see above).

Table EI 4. Groundfish and decapod taxa included in the quantification of scavenger biomass 2003-2012. Data are from the West Coast Groundfish Bottom Trawl Survey, courtesy of Beth Horness (Beth.Horness@noaa.gov).

Species	Family	Order	Class	Total catch (kg)
Albatrossia pectoralis	Macrouridae	Gadiiformes	Actinopterygii	19451.3
Coelorinchus scaphopsis	Macrouridae	Gadiiformes	Actinopterygii	0.4
Coryphaenoides acrolepis	Macrouridae	Gadiiformes	Actinopterygii	33158.8
Coryphaenoides cinereus	Macrouridae	Gadiiformes	Actinopterygii	42.3
Coryphaenoides filifer	Macrouridae	Gadiiformes	Actinopterygii	0.5
Macrouridae	Macrouridae	Gadiiformes	Actinopterygii	0.0
Malacocephalus laevis	Macrouridae	Gadiiformes	Actinopterygii	8.8
Nezumia liolepis	Macrouridae	Gadiiformes	Actinopterygii	149.9
Nezumia stelgidolepis	Macrouridae	Gadiiformes	Actinopterygii	277.0
Antimora microlepis	Moridae	Gadiiformes	Actinopterygii	2951.1
Anoplopoma fimbria	Anoplopomatidae	Scorpaeniformes	Actinopterygii	88740.3
Bolinia euryptera	Cottidae	Scorpaeniformes	Actinopterygii	4.6
Chitonotus pugetensis	Cottidae	Scorpaeniformes	Actinopterygii	3.4
Clinocottus acuticeps	Cottidae	Scorpaeniformes	Actinopterygii	2.3
Enophrys bison	Cottidae	Scorpaeniformes	Actinopterygii	3.2
Enophrys taurina	Cottidae	Scorpaeniformes	Actinopterygii	109.0
Gymnocanthus tricuspis	Cottidae	Scorpaeniformes	Actinopterygii	2.4
Hemilepidotus hemilepidotus	Cottidae	Scorpaeniformes	Actinopterygii	4.5
Hemilepidotus spinosus	Cottidae	Scorpaeniformes	Actinopterygii	5.7
Icelinus borealis	Cottidae	Scorpaeniformes	Actinopterygii	0.0
Icelinus burchami	Cottidae	Scorpaeniformes	Actinopterygii	1.7
Icelinus filamentosus	Cottidae	Scorpaeniformes	Actinopterygii	620.4
Icelinus fimbriatus	Cottidae	Scorpaeniformes	Actinopterygii	4.5
Icelinus tenuis	Cottidae	Scorpaeniformes	Actinopterygii	0.6
Jordania zonope	Cottidae	Scorpaeniformes	Actinopterygii	1.6
Leptocottus armatus	Cottidae	Scorpaeniformes	Actinopterygii	75.1
Paricelinus hopliticus	Cottidae	Scorpaeniformes	Actinopterygii	0.0
Radulinus asprellus	Cottidae	Scorpaeniformes	Actinopterygii	3.1
Radulinus taylori	Cottidae	Scorpaeniformes	Actinopterygii	0.1
Scorpaenichthys marmoratus	Cottidae	Scorpaeniformes	Actinopterygii	5.0
Triglops macellus	Cottidae	Scorpaeniformes	Actinopterygii	0.2
Sebastolobus alascanus	Scorpaenidae	Scorpaeniformes	Actinopterygii	40274.5
Sebastolobus altivelis	Scorpaenidae	Scorpaeniformes	Actinopterygii	114933.6
Sebastolobus sp.	Scorpaenidae	Scorpaeniformes	Actinopterygii	0.0
Hexanchus griseus	Hexanchidae	Hexanchiformes	Chondrichthyes	107.4
Scymnodon squamulosus	Somniosidae	Squaliformes	Chondrichthyes	2
Eptatretus deani	Myxinidae	Myxiniformes	Myxini	239.2
Eptatretus sp.	Myxinidae	Myxiniformes	Myxini	2.0

Species	Family	Order	Class	Total catch (kg)
Eptatretus stouti	Myxinidae	Myxiniformes	Myxini	111.2
Cancer magister	Canceridae	Decapoda	Malacostraca	47360.4
Chionoecetes bairdi	Oregoniidae	Decapoda	Malacostraca	16.1
Chionoecetes tanneri	Oregoniidae	Decapoda	Malacostraca	30716.5

Simpson Diversity (1-λ, Gini-Simpson index) was calculated (using the vegan package in R 2.15 (R Development Core Team 2012)) for each haul and then used to calculate an area-weighted mean for the West Coast shelf and slope. Taxa included in the analyses were all fishes identified to the species level (324 species). Data were number of individuals by species per haul.

Species Richness and Species Density were calculated for each depth x latitude bin and then used to derive the area-weighted mean for the combined West Coast shelf and slope. Because the number of species per sample will increase non-linearly in relation to sampling effort (either area or number of individuals), the groundfish data required rarefaction to standardize annual estimates of species richness and species density (Gotelli and Colwell 2001). Individual based rarefaction assumes that individuals are randomly distributed in space or time (Gotelli and Colwell 2001), and is, therefore, not the best approach for fishes since many species school. Individual-based rarefaction will tend to overestimate species richness when individuals are patchily distributed (Colwell et al. 2004). Instead, sample-based rarefaction was used to calculate species density curves by pooling samples (trawls) in depth x latitude bins. Rarefaction was conducted using the 'specaccum' function in the 'vegan' package for R 2.15 (Mao Tau, Colwell et al. 2004, R Development Core Team 2012).

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

For background the raw number of species per trawl is presented here (Figure EI 43) and could be considered analogous to species density since the data are species per area (trawls of more or less consistent swept area). Raw species density declined around 2007-2008 but then subsequently increased by more than 1.0 s.d. of the long-term mean over the last five years.

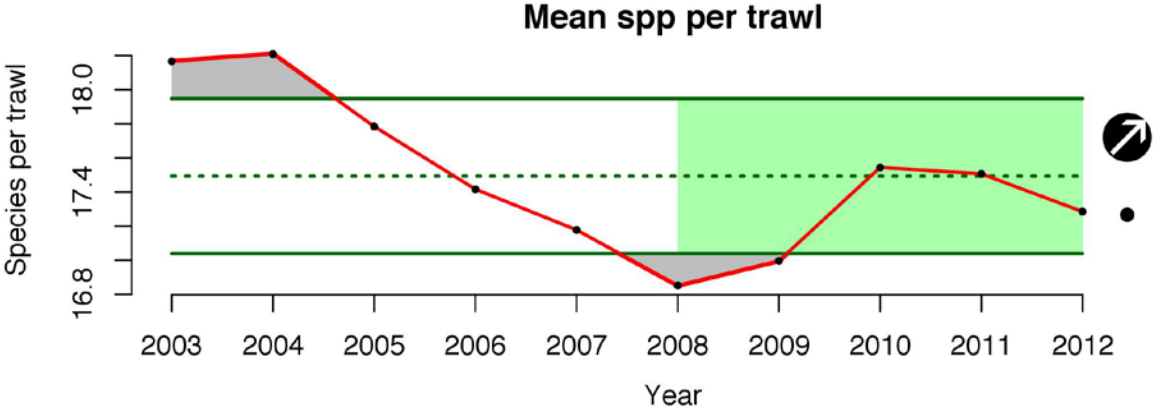

Figure EI 43. Mean number of groundfish species per trawl for 2003-2011 from the West Coast Groundfish Bottom Trawl Survey. Species number values were not subjected to rarefaction. Data courtesy of Beth Horness (Beth.Horness@noaa.gov).

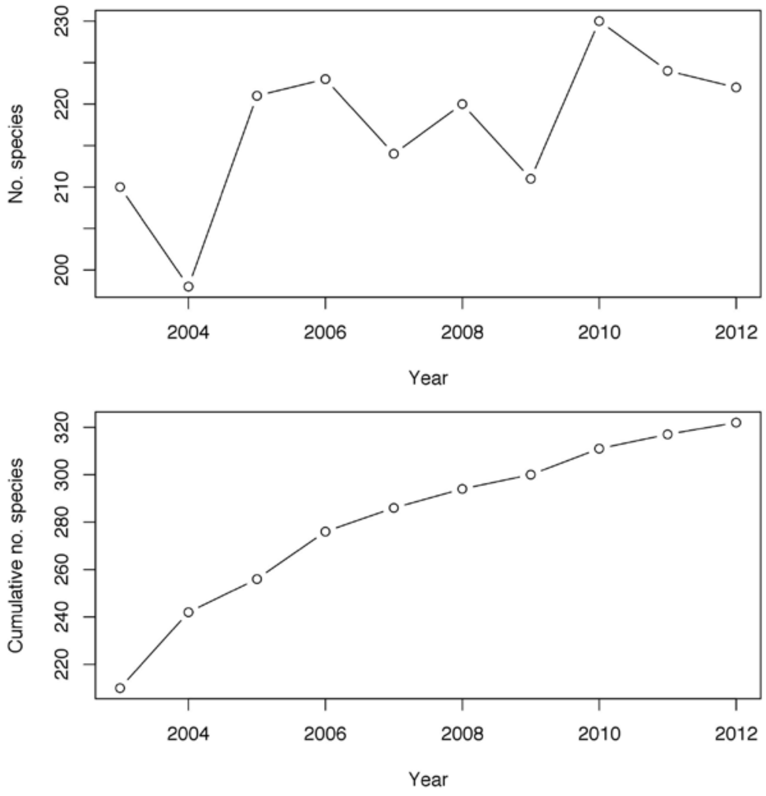

Figure EI 44. (A) Number of species identified in any given year by the trawl survey, (B) cumulative number of species identified by the trawl survey.

The initial decrease in species per trawl is probably related to a decline in the number of individuals per trawl (see Species Richness & Species Density - Groundfishes). The increase in species richness later in the time series may be attributable to better species identification. The number of species identified in any given year has increased over the time series by about 25-30 species from 2003 to 2012 (Figure EI 44a). The total number of species recorded by the trawl survey has steadily increased as expected by species-area relationships (Figure EI 44b).

ICHTHYOPLANKTON

We analyzed ichthyoplankton samples collected along two and six transects running perpendicular to shore in Oregon and California, respectively, between 2004 and 2011 in spring and summer. Stations within the Oregon lines were separated by between 8 and 20 km, while the California stations were separated by between 5 and 60 km (shoreward stations are closer together than seaward stations). The California samples comprised the suite of 66 stations sampled quarterly by the California Cooperative Oceanic Fisheries Investigations (CalCOFI) program (Hewitt 1988). Spring samples were collected in April and May in California and Oregon, respectively, and summer samples in August in both regions. Due to weather and logistic constraints, not each targeted station was sampled in each year. On average, 9 stations per season per year were sampled in Oregon and 64 stations were sampled in California.

Obliquely-towed bongo nets were used to collect ichthyoplankton following standard CalCOFI protocols (Kramer D. et al. 1972, Smith and Richardson 1977). The nets had 71-cm diameter openings in California and 60-cm diameter openings in Oregon with 0.505-mm mesh in both regions. Nets were fished to 100 m (or 5 m from the bottom at shallow stations) and 212 m (or 15 m from the bottom) in Oregon and California, respectively. Although the discrepancy in tow depth among regions potentially biases inter-region comparisons of assemblage structure, depth-stratified ichthyoplankton distributions in both Oregon and California showed that most individuals are found in the upper 100 m, thus likely minimizing impact of the different methodologies (Ahlstrom 1959, Moser and Smith 1993, Auth et al. 2007). Nets were equipped with flowmeters to determine the amount of water filtered during each tow.

Ichthyoplankton samples were preserved at sea in buffered formalin and identified to the lowest possible taxonomic level in the laboratory. Most taxa were identified to the species level but a few were only discernible to genus. For example, all rockfishes (*Sebastes* spp.) in Oregon were identified only to genus, but in California, a few were taken to species (*S. aurora*, *S. diploproa*, *S. goodei*, *S. jordani*, *S. levis*, and *S. paucispinis*), while the rest fell into the *Sebastes* spp. category. Second, although two species of sanddabs are found in both Oregon and California, they were not consistently identified to species and were thus termed *Citharichthys* spp.

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APPENDIX EI-1. EVALUATION OF INDICATORS FOR ECOLOGICAL INTEGRITY USING THE BONNEVILLE POWER ADMINISTRATION DATA SET

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OVERVIEW

As currently reported within the CCIEA, the Ecological Integrity goal relies heavily (but not exclusively) on evaluations of indicators derived from time series of benthic-dwelling taxa of trophic level ≥3. All of the current Ecological Integrity indicators have been evaluated with respect to 17 separate considerations (categorized as 'theoretical', 'data', and 'other') proposed by Kershner et al. (2011) and in previous versions of the CCIEA. However, none of these evaluations has focused on indicators derived specifically from data sets collected as part of surveys of Coastal Pelagic Species (CPS). To fill this gap, we evaluated 15 candidate indicators of Ecological Integrity derivable from a CPS data set developed with funding from the Bonneville Power Administration (BPA). In this pilot evaluation, we evaluated these indicators specifically with respect to the Ecological Integrity attribute *community composition* and data available from the BPA Plume Survey, a systematic 15-year time series of surface trawls made along the Oregon and Washington coast (see description of survey methods in Brodeur et al. 2005). In each detailed indicator summary (as well as in the Table of Contents above), the name of each indicator we evaluated is listed, followed by its score in brackets [ ].

Our approach differs from previous work in that we focused specifically on indicators populated with data for CPS. The indicators we evaluated included: forage fish biomass (in aggregate), jellyfish biomass (in aggregate), squid biomass (in aggregate), Humboldt squid, euphausiid biomass (in aggregate), species richness, Shannon diversity, (ecosystem) mean trophic level, ratio of forage fish to jellyfish, ratio of piscivorous to zooplanktivorous fish biomass, ratio of finfish to crustacean biomass, zooplanktivorous fish biomass, piscivorous fish biomass, biomass of top predators (trophic level > 4), and scavenger or detritivore biomass.

We adopted a tiered approach to the evaluation of each indicator, similar to the approach used for indicator evaluations in the Groundfish section of the CCIEA:

1. Evaluate the indicator generically, without respect to taxa or data sets, for theoretical and other considerations.
2. Modify the indicator evaluation for theoretical and other considerations based on specific information related to CPS taxa.
3. Evaluate the indicator with respect to specific CPS data sets for data considerations.

For example, for the indicator *(ecosystem) mean trophic level*, in the first tier of our evaluation, we focused on the criteria listed under theoretical considerations and other considerations, without concern for the data or taxa on which this indicator would be based. After this initial evaluation, we modified the supporting documentation, references, and scoring for this indicator so that the theoretical considerations and other considerations were evaluated for CPS taxa specifically. Finally, we evaluated the data considerations criteria with respect to a specific CPS data set (that derived from the BPA Plume survey; Brodeur et al. 2005).

The management goals outlined in the 2014 CCIEA are essentially framed by NOAA's mission to understand, conserve, and manage coastal and marine ecosystems and resources. As a result, fisheries or protected species (e.g., Groundfish, Salmon, Marine Mammals, and CPS) represent significant focal points of the assessment. One of the main roles of the Ecological Integrity goal in the CCIEA is to highlight or emphasize indicators that help us understand the web of ecosystem interactions, especially those not evaluated and reported for other CCIEA goals. While coastal pelagic species indicators, such as spatial structure, mean length (size), size structure, and age structure are potentially informative of ecosystem structure or function, we felt that they were best evaluated as candidate CPS indicators, rather than as candidate Ecological Integrity indicators. Similarly, we did not evaluate the reliability of individual coastal pelagic species as indicators of the Ecological Integrity goal, as doing so would require detailed knowledge of the relationship between each species and the Ecological Integrity attributes (ecosystem trophic structure and biodiversity), which is beyond the scope of our current expertise.

Our evaluation suggests several promising indicators of Ecological Integrity that can be derived from the BPA Plume survey data set, including aggregate biomass of forage fish, zooplanktivorous fish, and jellyfish. Of these, **jellyfish biomass is most complementary to existing Ecological Integrity indicators within the CCIEA**.

SUMMARY OF INDICATOR EVALUATIONS

Table 1. Summary of indicator evaluations for the Ecological Integrity goal of the CCIEA. The numerical value that appears under each of the considerations represents the summed scores for the criteria evaluated for each type of consideration. Criteria with full support in the peer-reviewed literature received a score of 1, those with partial support received a score of 0.5, and those with little or no support received a score of 0.

CCIEA Goal: Attribute	Indicator	Primary considerations (5)	Data considerations (7)	Other considerations (5)	Summary comments
Ecological Integrity: Community Composition	Forage fish biomass	4	5	4.5	Good indicator of changing ecosystem state and important to global fish landings and food webs. However, aggregating the biomass of all forage species may reduce its sensitivity as an indicator, and it may not respond to management actions if a threshold shift has occurred in the ecosystem. Concrete, operationally simple data, although forage fish naturally have highly variable populations (high signal:noise ratio). Changes in trends of aggregate groups will be concurrent, at best.
Ecological Integrity: Community Composition	Jellyfish biomass, status and trends	4.5	5	4	Good indicator of trophic energy transfer and pelagic community composition. Abundance can be linked to human activities, although there is contradictory guidance on reference condition or threshold. Some taxa are not sampled easily, though there is good understanding of seasonal and annual cycles for three dominant species in the CCIEA. Human problems with jellyfish have captured public attention; considered a good anticipatory indicator based on rapid growth rates and one-year life history.

CCIEA Goal: Attribute	Indicator	Primary consider-ations (5)	Data consider-ations (7)	Other consider-ations (5)	Summary comments
Ecological Integrity: Community Composition	Squid biomass	3.5	5	1	Market squid are considered part of the forage community and represent one of the most lucrative CA fisheries. Elusive as adults, squid have poor history of reporting due to sampling difficulties; this has led to a limited understanding of population dynamics, hampering assessments of stock health and related ecosystem attributes. Recent evidence suggests paralarval abundance may provide a better fishery-independent index of stock biomass.
Ecological Integrity: Community Composition	Seasonal migrants – e.g., Humboldt squid	4	0	3.5	The range expansion of seasonal migrants like Humboldt squid possibly indicates shifts in climate regimes, ocean circulation, and ecosystem-wide food webs. However, like market squid they are not well sampled in the BPA plume time-series. Increasingly appreciated as important by the public and managers, this indicator has a mixed history of reporting. Seasonal migrants are considered a good anticipatory indicator, often with short generation times and the ability to quickly respond to changes at the base of the food web.
Ecological Integrity: Community Composition	Euphausiid abundance	4.5	0	3	Euphausiids are a critical link in the food web for many commercially important taxa and considered a predictable and sensitive indicator of ocean conditions in the CA Current. They are patchily distributed and poorly sampled in the BPA plume time-series, leading to shortcomings in their reporting history and understanding by policy makers.
Ecological Integrity: Community Composition	Species richness	2.5	6	3.5	Species richness is a crucial property of ecological communities that is highly relevant to management concerns and public perception, but relationships to ecosystem function are complex. There is no support that the richness of pelagic nekton communities respond to management actions or pressures. The BPA dataset provides a useful platform for evaluating species richness of the pelagic nekton community; little evidence it could be used as an anticipatory indicator.

CCIEA Goal: Attribute	Indicator	Primary consider-ations (5)	Data consider-ations (7)	Other consider-ations (5)	Summary comments
Ecological Integrity: Community Composition	Shannon diversity	3.5	6	3	Shannon diversity estimates show habitats occupied by pelagic nekton species can expand and contract in relation to the dynamic nature of the California Current at both seasonal and interannual periodicities. However, linking Shannon diversity to targets or reference points is difficult, making this indicator difficult to interpret or contextualize. The BPA time series meets most criteria for Shannon diversity data considerations; although there is little evidence pelagic nekton Shannon diversity metrics could be used as an anticipatory indicator.
Ecological Integrity: Community Composition	Mean trophic level	3.5	4	2.5	The diet of many pelagic nekton species change on an annual basis, making trophic level assignments difficult and the data signal noisy, and often confounded by major shifts of some abundant taxa. MTL is increasingly used as an ecosystem indicator, but cannot be considered an anticipatory indicator in this context.
Ecological Integrity: Community Composition	Forage fish: jellyfish ratio	4.5	5	2	Both forage fish and jellyfish biomass score highly individually as indicators, and the data are readily obtained from the BPA data set. In addition, the conceptual underpinnings of this ratio are clear: as the biomass of forage fish increases relative to that of jellyfish, there is more production available to fisheries and to larger predatory fish, mammals, and birds. However, the statistical properties of the ratio of these 2 quantities are understudied, making it more difficult to use the ratio of forage fish to jellyfish to determine appropriate reference points, communicate to the public, and compare across ecosystems.

CCIEA Goal: Attribute	Indicator	Primary consider-ations (5)	Data consider-ations (7)	Other consider-ations (5)	Summary comments
Ecological Integrity: Community Composition	Piscivorous: Zooplanktivorous fish ratio	4	0	1	The ratio of piscivorous fish to zooplanktivorous fish biomass is in theory a sound indicator, albeit with challenging statistical properties that limit ease of communication and setting of reference points. However, there are very few piscivorous fishes represented in the CPS taxa sampled by the BPA survey. Thus, this indicator is likely to be inappropriate for use with the BPA data set and most others geared to sample CPS.
Ecological Integrity: Community Composition	Finfish: Crustacean Biomass Ratio	3	0	1	The conceptual underpinnings of this indicator are clear. In general, finfish tend to be represented on a higher mean trophic level than crustaceans. Thus if fishing down or through the food web occurs, the ratio of finfish to crustacean biomass should decline. However, this indicator is usually intended to include benthic animals as part of the crustacean biomass in the denominator, and so is not well-suited to derivation from CPS data sets like the BPA survey.
Ecological Integrity: Community Composition	Zooplanktivorous fish biomass	4.5	5	4.5	Reliable indicator of changing ecosystem state and important to global fish landings and food webs. However, aggregating the biomass of all zooplanktivoroush fish species may reduce its sensitivity as an indicator, and it may not respond to management actions if a threshold shift has occurred in the ecosystem. This indicator is concrete and simple, although zooplanktivorous fishes naturally have highly variable populations (high signal:noise ratio). Changes in trends of aggregate groups will be concurrent, at best.

CCIEA Goal: Attribute	Indicator	Primary consider-ations (5)	Data consider-ations (7)	Other consider-ations (5)	Summary comments
Ecological Integrity: Community Composition	Piscivorous fish biomass	3	0	3	In principle, piscivorous fish biomass could serve as a reliable indicator of Ecological Integrity, as it is well supported theoretically as a metric that should decline if fishing down/through the food web occurs (Pauly et al. 1998, Essington et al. 2006). In addition, this indicator is reported in many ecosystems and is relatively easy to communicate. However, the representation of piscivorous fishes in CPS data sets is minimal, reference points for this indicator have yet to be established, and as an aggregate group it is unlikely to serve as a leading indicator of ecosystem changes.
Ecological Integrity: Community Composition	Top predator biomass (trophic level > 4)	3	0	3	Like piscivorous fish biomass, in principle top predator biomass could serve as a reliable indicator of Ecological Integrity, as it is well supported theoretically as a metric that should decline if fishing down/through the food web occurs (Pauly et al. 1998, Essington et al. 2006). In addition, this indicator is reported in many ecosystems and is relatively easy to communicate. However, the representation of top predator biomass in CPS data sets is minimal, reference points for this indicator have yet to be established, and as an aggregate group it is unlikely to serve as a leading indicator of ecosystem changes.
Ecological Integrity: Community Composition	Scavenger or detritivore biomass	3	0	2	Scavenger biomass is in principle a theoretically sound indicator of ecological integrity, as it is expected to increase as fisheries discards and other human pressures accumulate. However, it is probably best represented with the inclusion of benthic detritivores and is at best partially tracked within CPS data sets like the BPA survey. Furthermore, it is an indicator that is likely to respond to ecosystem changes rather than lead them and does not have a long history of reporting in previous ecosystem assessments.

DETAILED INDICATOR EVALUATIONS FOR THE ECOLOGICAL INTEGRITY GOAL

FORAGE FISH BIOMASS, IN AGGREGATE [13.5]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 13.5

Summary: Good indicator of changing ecosystem state and important to global fish landings and food webs. However, aggregating the biomass of all forage species may reduce its sensitivity as an indicator, and it may not respond to management actions if a threshold shift has occurred in the ecosystem. Concrete, operationally simple data, although forage fish naturally have highly variable populations (high signal:noise ratio). Changes in trends of aggregate groups will be concurrent, at best.

Criterion	Score	Explanation
(a) Theoretically sound	1	• Changes in the biomass of pelagic species may indicate changes in ecosystem state as a function of fishing down the food web, predatory release of prey populations (forage fish), or insufficient forage base for top predators (Link 2005). • Zooplanktivorous fish has been described as the best indicator of total biomass and net primary production in the system using 7 food web models from the North Pacific and the Baltic Sea (Samhouri et al. 2009). • Threshold-like shifts in Baltic Sea pelagic ecosystems appear to be driven by planktivore abundance (sprat, *Sprattus sprattus*) that separates 2 ecosystem configurations in which zooplankton dynamics are driven by either hydroclimatic forces or predation pressure (Casini et al. 2009). • CPS and/or forage species: often present in high abundance, feed on plankton for a portion of their life cycle and form dense schools or aggregations (e.g., anchovy, sardine, herring, mackerel, squid and krill) (CPS Management Plan, http://www.pcouncil.org/wp-content/uploads/CPS_FMP_as_Amended_thru_A13_current.pdf).

(b) Relevant to management concerns	1	• Forage fish account for over 30% of global fish landings, and play an important role in marine food webs because they are the principal means of transferring production from plankton to larger predatory fish and to marine mammals and seabirds (Smith et al. 2011, Ruzicka et al. 2012, Sydeman et al. 2013). • Fishery Management Plans for assessed species (e.g., sardine), as well as entire CalCOFI sampling program, attest to management importance.
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	0.5	• Zooplanktivorous fish represents the best indicator of total biomass in an ecosystem based on 7 food web models from the North Pacific and the Baltic Sea (Samhouri et al. 2009). • Small forage-fish biomass in the northeast Pacific appears to increase during cold ocean conditions as a result of zooplankton assemblage composition change, though there are differences between the SCC and NCC (http://www.nwfsc.noaa.gov/research/divisions/fe/estuarine/oeip/ha-under-development.cfm). • Chavez et al. (2003) documented an Inverse relationship between sardines and anchovies, suggesting that aggregate forage fish biomass may make it insufficiently sensitive and predictable with respect to changes in community composition. • Forage fish diets are variable from year to year, reflecting changes in the abundance of prey in the ecosystem (Hill et al. submitted JMS). • Seasonal abiotic forcing predicts changes in the abundance of forage fish species (Litz et al. 2014).
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	0.5	• Management action: the pressure of fishing these species at conventional maximum sustainable yield (MSY) levels can have large impacts on other parts of the ecosystem, particularly when they constitute a high proportion of the biomass in the ecosystem or are highly connected in the food web (Smith et al. 2011; Kaplan et al. 2013; Pikitch et al. 2014). • Precautionary management will theoretically affect population, though not if threshold shift has occurred or environmental drivers have changed (Casini et al. 2009). • There are differences between the SCC and NCC in forage fish community composition and responses to fishing pressure.
(e) Linkable to scientifically-defined reference points & progress targets	1	• Link (2005) set warning and limit reference points as follows: if $B_{pelagic} > 75\% B_{total}$, or if $B_{pelagic} < 20\% B_{total}$ in any given year, then a warning threshold has been exceeded. The LRPs are set at $B_{pelagic} > 85\% B_{total}$, and at $B_{pelagic} < 10\% B_{total}$.

		• Casini et al. 2009: Shifts in Baltic Sea pelagic ecosystems appear to be driven by planktivore abundance (the clupeid, sprat) that separates 2 ecosystem configurations in which zooplankton dynamics are driven by either hydroclimatic forces or predation pressure; identified an ecological threshold, corresponding to a planktivore abundance of 17 x 10^{10} individuals. • Fishery Management Plan reference points for managed species (B_{40} and B_{25} rules) have been applied as reference targets, but simulation models by Kaplan et al. (2013) show that this level of forage species removal is likely to impact the abundance of other target species, protected species, and the structure of the ecosystem. • Cury et al. 2011 demonstrated a threshold in forage fish abundance (1/3 max historic prey biomass) below which seabirds experience reduced/more variable productivity. • Also see Lenfest report (cf. Pikitch et al. 2014).
(g) Concrete & Numerical	1	• Large trawls (336-m^2 opening) in the upper 18-20 m of the water column at every station. • Mesh sizes ranged from 162.6 cm in the throat of the trawl near the jib lines to 8.9 cm in the cod end. • To maintain catches of small fish and squids, a 6.1 m long, 0.8 cm knotless liner was sewn into the cod end. • Total counts and weights are available for each nekton taxon caught. • Length data are recorded for a subset of individuals (n=50 per station) of all species.
(h) Historical data or information available	0.5	• Since 1998; Annual systematic surveys have been conducted using consistent methodology in June and September of each year.
(i) Operationally simple	1	• Measurements of hydrography, zooplankton and pelagic fishes are made at each station. • Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.
(k) Broad spatial coverage	0.5	• Mid-Oregon to North WA coast (Newport to Cape Flattery; along nine transect lines at 50-55 stations, ranging from 45 to 48 °N).
(l) Continuous time series	1	• 15 y (1998-2012), in June and September.
(m) Spatial & temporal variation understood	0.5	• Annual and summer/spring variability well-understood for most taxa; seasonal less so. • Sampling occurs during day or crepuscular periods so diel patterns are less clear and myctophids are not well characterized.

(n) High signal-to-noise ratio	0.5	- Sampled taxa have naturally highly variable populations; for some of the best sampled taxa signal to noise is high or can be discerned, although poor for myctophids.
(o) Understood by the public & policymakers	1	- Yes; relative trends in biomass of each component of the community is easily understood and increasingly reported/embraced.
(p) History of reporting	1	- Reported in Status of California Current 2012 (CalCOFI). Link 2005; Samhouri 2009, Fulton et al. 2005.
(q) Cost-effective	1	- Assessment data already collected for many of these species; data mining is all that is needed.
(r) Anticipatory or leading indicator	0.5	- Changes in trends of aggregate groups will always be concurrent at best. - Rapid response of forage fishes and other nekton (distributional anomalies) to delayed upwelling (Brodeur et al. 2006). - forage fish generally show a between a change in ocean phase and population response (http://www.nwfsc.noaa.gov/research/divisions/fe/estuarine/oeip/ha-under-development.cfm), but this is relatively fast compared to other North Pacific fish populations (Yatsu et al. 2008) and early life history stages may respond more quickly.
(t) Regionally/nationally/internationally compatible	1	- Pikitch et al. 2014, Link 2002, Methratta & Link 2006.
TOTAL SCORE	13.5	

JELLYFISH BIOMASS, IN AGGREGATE [13.5]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 13.5

Summary: Good indicator of trophic energy transfer and pelagic community composition. Abundance can be linked to human activities, although there is contradictory guidance on reference condition or threshold. Some taxa are not sampled easily, though there is good understanding of seasonal and annual cycles for three dominant species in the CCIEA. Human problems with jellyfish have captured public attention; considered a good anticipatory indicator based on rapid growth rates and one-year life history.

Criterion	**Score**	**Explanation**
(a) Theoretically sound	1	• Gelatinous zooplankton blooms are typically associated with overfishing, climate change, or eutrophication (Arai 2001, Purcell 2005, Purcell et al. 2007, Richardson et al. 2009). • Jellyfish biomass can be one of the most sensitive indicators of changing ecosystem status (Richardson et al. 2009). • Median biovolume of gelatinous zooplankton included in list of indicators translated into decision criteria (Link et al. 2005). • Jellyfish should be reliable indicators of trophic energy transfer and pelagic community composition. For example, modeling exercises by Ruzicka et al. (2013) showed that in the Northern California Current large scyphozoan jellyfish are important consumers of plankton production, but can divert energy from the rest of the food web when abundant. • Lower trophic level, high productivity functional groups like jellyfish biomass showed relatively strong correlations with at least half of the ecosystem attributes in a food-web modeling exercise that evaluated the performance of candidate indicators of ecosystem structure and function (Samhouri et al. 2009).
(b) Relevant to management concerns	1	• Yes, jellyfish biomass and abundance is likely to be particularly relevant to understanding community composition and species interactions; often integrated into ecosystem models (Brand et al. 2007; Pauly et al. 2009, Ruzicka et al. 2012). • Numerous negative effects on human enterprise; specifically, they interfere with tourism by stinging swimmers, fishing by clogging nets, aquaculture by killing fish in net-pens and power plants by clogging cooling-water intake screens (Purcell et al. 2007). • In the NCC, early stages of euphausiids, gelatinous taxa, and cladocerans were particularly vulnerable to predation by jellyfish (Suchman et al. 2008). • Overfishing of small pelagic fishes in the northern Benguela Current may have promoted jellyfish abundance and possibly led to irreversible jellyfish dominance (Flynn et al. 2012, Roux et al. 2013).

(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	1	- Jellyfish biomass served as the best proxy for ecosystem attributes related to community energetics using 7 food web models from the North Pacific and the Baltic Sea (Samhouri et al. 2009). - Increases in jellyfish are generally associated with negative impacts on ecosystem attributes, models offer opportunity to investigate this problem (Pauly et al. 2009, Ruzicka et al. 2012). - Highest catches of medusae in the NCC correlated with cool spring–summer conditions, or negative anomalies of the Pacific Decadal Oscillation, and low winter–summer runoff from the Columbia River (Suchman et al. 2012). - Jellyfish increased steeply and then declined precipitously in the Eastern Bering Sea in the 1990s and 2000s, coinciding with wide-scale climate regime shifts (Brodeur et al. 2008c).
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	1	- Jellyfish abundance can be linked to fishing impacts, eutrophication, habitat modification (shoreline armoring), and several other human activities (Purcell et al. 2007; Richardson et al. 2009, Pauly et al. 2009, Purcell et al. 2012). - Overfishing of small pelagic fishes in the northern Benguela Current may have promoted jellyfish abundance and possibly led to irreversible jellyfish dominance (Flynn et al. 2012, Roux et al. 2013).
(e) Linkable to scientifically-defined reference points & progress targets	0.5	- Link (2005) set warning thresholds of 100% above the median biovolume for gelatinous zooplankton ($V_{jelly\text{-}med}$) and a limit reference point of 200% above $V_{jelly\text{-}med}$ - Condon et al. (2013) suggest there is no robust evidence for global increase in jellyfish; rather, jellyfish populations undergo larger, worldwide oscillations with an approximate 20-y periodicity.
(g) Concrete & Numerical	0.5	- Top three species individually; mesh size too large for smaller taxa.
(h) Historical data or information available	0.5	- Since 1998; Annual systematic surveys have been conducted using consistent methodology in June and September of each year.
(i) Operationally simple	1	- Measurements of hydrography, zooplankton and pelagic fishes are made at each station. - Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.
(k) Broad spatial coverage	0.5	- Mid-Oregon to North WA coast (Newport to Tatoosh Island). - Along nine transect lines at 50-55 stations, ranging from 45 to 48 °N).

(l) Continuous time series	1	• 15 y (1998-2012), in June and September.
(m) Spatial & temporal variation understood	0.5	• Limited to spring / summer samples. • Annual and seasonal cycles described by Suchman et al. (2012), Brodeur et al. (in review).
(n) High signal-to-noise ratio	1	• High signal to noise ratio for 3 top jellyfish species (Suchman et al. 2012).
(o) Understood by the public & policymakers	1	• Increasingly regarded as a keystone species/group in some systems (Pauly et al. 2009). • Raft of recent studies/reviews suggest they have increased in abundance throughout world. • Human problems with jellyfish have increased and have captured public attention (Purcell et al. 2007, Richardson et al. 2009).
(p) History of reporting	0.5	• The paucity of long-term data makes it difficult to draw definitive conclusions on the status of jellyfish populations (Purcell et al. 2007). • But see Brodeur et al. 2002 (Bering Sea), Pauly et al. (2009), Mackas et al. (2001), Suchman et al. (2012), and Brodeur et al. (in review MEPS) for evidence of changing trend. • Recent analyses of thirty-seven datasets between 1790 and 2011, representing 1,140 observation-years of jellyfish abundance, with a mean length of 31 y, show jellyfish populations undergo larger, worldwide oscillations with an approximate 20-y periodicity (Condon et al. 2013, JEDI database).
(q) Cost-effective	0.5	• There are a variety of survey techniques available, some of which are more costly to implement than others (evaluated by Bamstedt et al. 2006). • Several sampling challenges make quantifying populations of large medusae particularly difficult (Purcell, 2009).
(r) Anticipatory or leading indicator	1	• Jellyfish populations can grow quickly in response to abundant prey, producing jellyfish "blooms." • Because of fast growth rates and one-year life cycle, gelatinous zooplankton respond quickly to variability in local or regional environmental conditions, but general abundance patterns and the mechanisms responsible for those patterns have been difficult to discern (Suchman et al. 2012). • Autocorrelation among years in jelly abundance can be high (Richardson and Gibbons 2008).

(t) Regionally/nationally/internationally compatible	1	• Yes (Link et al. 2005; Pauly et al. 2009).
TOTAL SCORE	13.5	

SQUID BIOMASS, IN AGGREGATE [9.5]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 9.5

Summary: Market squid are considered part of the forage community and represent one of the most lucrative CA fisheries. Elusive as adults, squid have poor history of reporting due to sampling difficulties; this has led to a limited understanding of population dynamics, hampering assessments of stock health and related ecosystem attributes. Recent evidence suggests paralarval abundance may provide a better fishery-independent index of stock biomass.

Criterion	**Score**	**Explanation**
(a) Theoretically sound	1	• Can be considered part of the forage/pelagic community (see Forage Fish Biomass section, above). • Changes in the biomass of pelagic species may indicate changes in ecosystem state as a function of fishing down the food web, predatory release of prey populations (forage species), or insufficient forage base for top predators (Link 2005). • However, little is known about the present size, age structure, or status of the market squid population. • At present, no direct, statistically valid population estimates are available (PFMC 2010). • Historically, abundance/biomass estimates estimated from commercial landings, but new research shows that paralarval abundance (CalCOFI) provides a fishery-independent index

		of stock biomass (Koslow and Allen 2011). • Model simulations suggest, however, that squids are able to benefit from a general increase in fishing pressure, mainly due to predation release, and quickly respond to changes triggered by the environment. • Squids may thus be very sensitive to the effects of fishing and climate change (Coll et al. 2013).
(b) Relevant to management concerns	1	• As a forage species they play an important role in marine food webs as the principal means of transferring production from plankton to larger predatory fish and to marine mammals and seabirds (Smith et al. 2011). • CPS Management Plans, attest to management importance, however there is large variation in relative abundance of squid between NCC and SCC. • One of the largest and most lucrative California fisheries. • In the mid-1990s, market squid became the largest California fishery in terms of both landings and revenue. (Koslow and Allen 2011). • Key component of marine food webs and of increasing economic importance as evidenced by rapid rise in global landings (Hunsicker et al. 2010). • Humboldt squid *Dosidicus gigas* range expansion into CCLME coincided with declines in valuable target species such as Pacific hake *Merluccius productus* and other top predators (Zeidberg and Robison 2007).
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	0.5	• Paralarval abundance provides a fishery-independent index of stock biomass, and these statistical relationships suggest that the ENSO and PDO indices can be used for adaptive management of the market squid fishery (Koslow and Allen 2011). • However there is still limited understanding of market squid population dynamics, which would hamper assessing the status (health) of related ecosystem attributes (PFMC 2010). • New evidence suggests that the market squid center of biomass shifted northward between 1999-2012 (Barcelo et al. in prep).
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	0.5	• Unclear, except from ecological model simulations. • The role of squids in open ocean and upwelling ecosystems appear more constrained to a bottom-up impact on their predators, suggesting large removals of squids will likely have large-scale effects on marine ecosystems (Coll et al. 2013). • Simulations confirm that squids are able to benefit from a general increase in fishing pressure, mainly due to predation release, and quickly respond to changes triggered by

		the environment. Squids may thus be very sensitive to the effects of fishing and climate change. • CPUE seems to have deficiencies as an index of stock biomass because the fishery is carried out on spawning aggregations (Koslow and Allen 2011). • Owing to the shorter life cycle of cephalopods, and rapid turnover, and lower standing stocks than for longer-lived taxa; strong circumstantial evidence that fishing pressure has changed ecological conditions and cephalopod stocks have increased as predatory fish have declined (Caddy and Rodhouse 1998). • Interconnectedness of commercial cephalopods and fishes is only recently being recognized - may help promote sustainable fishing in these ecosystems under increased levels of exploitation (Hunsicker et al. 2010).
(e) Linkable to scientifically-defined reference points & progress targets	0.5	• There is a limited understanding of market squid population dynamics, which has hampered assessing the status (health) of this valuable marine resource (PFMC 2010). • The fishery is therefore "monitored" without a formal stock assessment to guide management; catch limits are based on the approximate maximum landings obtained in three seasons since 1998–2000 (107,048 mt; 118,000 st), but landings over the past decade have been mostly about half the total allowable catch. • Catch per unit effort (CPUE) has been relatively steady, but CPUE is generally unreliable as a proxy for stock biomass, particularly for a fishery with evolving gear technology and that targets spawning aggregations (Koslow and Allen 2011). • For pelagic species, Link (2005) set warning and limit reference points as follows: if $B_{pelagic} > 75\% B_{total}$, or if $B_{pelagic} < 20\% B_{total}$ in any given year, then a warning threshold has been exceeded. • The LRPs are set at $B_{pelagic} > 85\% B_{total}$, and at $B_{pelagic} < 10\% B_{total}$; Fishery Management Plan reference points for managed species (B_{40} and B_{25} rules) have been applied as reference targets, but simulation models by Kaplan et al. (2013) show that this level of forage species removal is likely to impact the abundance of other target species, protected species, and the structure of the ecosystem.
(g) Concrete & Numerical	1	• Large trawls (336 m^2 mouth opening) in the upper 18-20 m of the water column at every station. • Mesh sizes ranged from 162.6 cm in the throat of the trawl near the jib lines to 8.9 cm in the cod end. • To maintain catches of small fish and squids, a 6.1 m long, 0.8 cm knotless liner was sewn

		into the cod end. • Total counts and weights are available for each nekton taxon caught. • Length data are recorded for a subset of individuals (n=50 per station) of all species.
(h) Historical data or information available	0.5	• Since 1998; Annual systematic surveys have been conducted using consistent methodology in May, June, and September of each year.
(i) Operationally simple	1	• Measurements of hydrography, zooplankton and pelagic fishes are made at each station. • Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.
(k) Broad spatial coverage	0.5	• Mid-Oregon to North WA coast (Newport to Cape Flattery; along nine transect lines at 50-55 stations, ranging from 45 to 48 °N).
(l) Continuous time series	1	• 15 y (1998-2012), in June and September.
(m) Spatial & temporal variation understood	0.5	• Annual and summer/spring variability well-understood; seasonal (winter) less so. • Appear to be similarly abundant during day and night (C. Barcelo personal communication). • Sampled during day or crepuscular periods and not during winter. • New evidence suggests that market squid center of biomass shifted northward between 1999-2012 (Barcelo et al. in prep).
(n) High signal-to-noise ratio	0.5	• Often difficult to sample, resulting in high noise. • Patchily distributed.
(o) Understood by the public & policymakers	0.5	• Yes, but perhaps not as resonant for their ecosystem integrity value as other indicators. • Relative trends in biomass of squid are easily understood by the public and policymakers. • Abundance/biomass estimates were once determined from commercial landings, but new research shows that paralarval abundance (CalCOFI) can provide a fishery-independent index of stock biomass (Koslow and Allen 2011).
(p) History of reporting	0	• Not extensive because squid biomass estimates have been notoriously difficult to determine. • Information about the effects of fishing and the environment on squid populations is especially lacking, probably because they are difficult to sample, and they have complex life cycles and distribution patterns (Coll et al. 2013). • Historically, abundance/biomass estimates were determined from commercial landings, but new research shows that paralarval abundance (CalCOFI) provides a fishery-

		independent index of stock biomass (Koslow and Allen 2011) and acoustic methods may prove promising (Zeidberg,, personal communication).
(q) Cost-effective	0	• Stock assessments of market squid are unlikely due to sampling difficulties; more likely to rely on data already collected by CalCOFI on paralarva abundance, which has been correlated with stock biomass (Koslow and Allen 2011).
(r) Anticipatory or leading indicator	0.5	• The market squid lives only 6–9 months (Butler et al. 1999), and the population fluctuates markedly from year to year, largely in apparent response to environmental factors. • However, limited understanding of market squid population dynamics (PFMC 2010) hampers understanding of its potential role as leading indicator.
(t) Regionally/nationally/internationally compatible	0	• Few compatible indicators found in use elsewhere to reflect ecosystem integrity.
TOTAL SCORE	9.5	

SEASONAL MIGRANTS - HUMBOLDT SQUID [7.5]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 7.5

This is an example of how an individual species could be evaluated as an indicator for the Ecological Integrity goal. It is the only individual CPS we evaluated this year for the Ecological Integrity goal.

Summary:

The range expansion of seasonal migrants like Humboldt squid (*Dosidicus gigas*) possibly indicates shifts in climate regimes, ocean circulation, and ecosystem-wide food webs. However, like market squid they are not well sampled in the BPA plume time-series. Increasingly appreciated as important by the public and managers, this indicator has a mixed history of reporting. Seasonal migrants are considered a good anticipatory indicator, often with short generation times and the ability to quickly respond to changes at the base of the food web.

Criterion	**Score**	**Explanation**
(a) Theoretically sound	1	• Unusual occurrences of marine life such as Humboldt squid can be an indicator of changing ocean conditions. • Considered as upper level predator that quickly responds to changing ocean conditions with distributional shifts (Field et al. 2013). • Zeidberg & Robison 2007: Range expansion related to changes in climate and correlated with reduction in top predators; correlative information only. • Caddy & Rodhouse 1998: cephalopod landings have increased as groundfish landings decreased. • Modeling by Rosas-Luis et al. 2008 for the Gulf of California suggest the effects of Humboldt squid can be substantial.. • Tolerant of oxygen minimum zones, which appear to greatly favor *D. gigas* (Gilly et al. 2012). • The CPS management plan considers Humboldt squid as an indicator (PFMC 2010).
(b) Relevant to management concerns	1	• Yes. Zeidberg & Robison 2007. • Considered to have an important ecological role in pelagic ecosystems due to its high abundance and wide distribution (Rosas-Luis et al. 2008). • Presence indicates shifts in climate regimes, ocean circulation and potentially ecosystem wide food webs. • Cephalopod predation is an important variable affecting natural mortality and recruitment success of many fish stocks, particularly clupeids, scombrids, and gadoids in continental shelf ecosystems (Rodhouse and Nigmatullin 1996). Also see Miller et al. (2013) and Field et al. (2013) for predation data relevant to northern California Current. • Though predators of commercially important species, they are principally prey to the same in the Eastern Tropical Pacific (Field et al. 2007). • NCC invasion coincident with decline in Pacific hake - the most important commercial groundfish species off western North America (Coll et al. 2013).

(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	1	• Range expansion (distribution shifts) related to changes in climate and correlated with reduction in top predators (Zeidberg & Robison 2007, Stewart et al. 2014). • Thermal tolerance and plastic life history of *D. gigas* provide a parsimonious explanation of both the squid's historical visits to waters off central California and its range expansion (Watters et al. 2008). • Growth and physiological challenges may be related to climate change, changing CO_2 levels, and ocean acidification (Rosas-Luis et al. 2008). • Coincidence of poleward range expansions of Humboldt squid in both hemispheres suggests a physically-induced forcing mechanism (Field et al. 2007). • Tolerance of oxygen minimum zone appears to greatly favor *D. gigas*, and it is clearly an environment that this species inhabits and utilizes (Gilly et al. 2012, Stewart et al. 2014).
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	0.5	• Unclear, except from ecological model simulations. • Current variation in population appears to be physically-based (Field et al. 2007). • Results from a model for the eastern equatorial Pacific suggest that squids are controlled more by bottom-up forcing than by top-down cascades from tuna fishing (Watters et al. 2003). • The role of squids in open ocean and upwelling ecosystems appeared more constrained to a bottom-up impact on their predators, suggesting large removals of squids will likely have large-scale effects on marine ecosystems (Coll et al. 2013). • Simulations confirm that squids are able to benefit from a general increase in fishing pressure, mainly due to predation release, and quickly respond to changes triggered by the environment. • The impact of the fishery on the population is not understood, and there are concerns about overexploitation during the periodic downturns of the population (Koslow and Allen 2011).
(e) Linkable to scientifically-defined reference points & progress targets	0.5	• Generally characterized by presence/absence in time-series data from Zeidberg & Robison (2007) and Field et al. (2007). • No scientifically defined reference points or progress targets, per se.
(g) Concrete & Numerical	0	• No data available from BPA surveys.
(h) Historical data or information available	0	• No data available from BPA surveys.

(i) Operationally simple	0	• No data available from BPA surveys.
(k) Broad spatial coverage	0	• No data available from BPA surveys.
(l) Continuous time series	0	• No data available from BPA surveys.
(m) Spatial & temporal variation understood	0	• No data available from BPA surveys. • Highly mobile. • Poorly characterized with gear.
(n) High signal-to-noise ratio	0	• No data available from BPA surveys. • Notoriously difficult to sample, resulting in high noise.
(o) Understood by the public & policymakers	1	• Yes. See recent articles in press and Scientific American (2010) and videos by BBC/National Geographic; focus on low oxygen zones and competition with other commercial species (salmon, tuna).
(p) History of reporting	0.5	• Currently, lack of abundance/biomass information and incomplete knowledge of how movement and food habits may differ across seasons and between inshore and offshore waters (Field et al. 2007). • Trends in abundance estimated using landings information from commercial and recreational fisheries, resource surveys, food habits studies, discussions with fishermen (Field et al. 2007). • Unique and long time series of video observations over 16 y (Zeidberg and Robison 2007).
(q) Cost-effective	0.5	• Currently data from existing surveys and landings are probably sufficient to describe regime shift type effects (Field et al. 2007, Zeidberg & Robison 2007). • Better biomass estimates would be desired if it becomes an established indicator.
(r) Anticipatory or leading indicator	1	• Short generation times make it sensitive to change (Zeidberg & Robison 2007), but more of a concurrent indicator than an anticipatory one. • Modeling results illustrate that squids have the ability to quickly respond to changes at the base of the food web that may be triggered by environmental changes (Coll et al. 2013). • Considered as upper level predator that quickly responds to changing ocean conditions with distributional shifts.
(t) Regionally/nationally/internationally compatible	0.5	• West Coast only. • Could be considered regionally compatible as upper level predator (invasive species?) that

		quickly responds to changing ocean conditions with distributional shifts.
TOTAL SCORE	7.5	

EUPHAUSIID BIOMASS, IN AGGREGATE [7.5]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 7.5

Summary: Euphausiids are a critical link in the food web for many commercially important taxa and considered a predictable and sensitive indicator of ocean conditions in the CA Current. They are patchily distributed and poorly sampled in the BPA plume time-series, leading to shortcomings in their reporting history and understanding by policy makers

Criterion	Score	Explanation
(a) Theoretically sound	1	• Can be considered part of the forage/pelagic community (see forage section, above; Smith et al. 2012). • Species can be found well poleward of their usual range during strong El Niño events, some because they are transported by anomalously poleward geostrophic currents, and some because normally cool areas warm and become habitable (Keister et al. 2005). • Biomass anomalies for the euphausiid *Nyctiphanes simplex* in the California Current System as an indicator of plankton biomass (deYoung et al. 2008). • Changes in southern California Current are characterized by an increase in the abundance of southern and tropical euphausiid species in the southern California Bight, and a decreased abundance of a northern species (Peterson 2009). • Euphausiids also constitute an important part of the diet of cod larvae and juveniles. In particular, euphausiids have a high-energy content and constitute an important source of vitamin A for fish such as cod, which cannot synthesize this vitamin (Beaugrand et al. 2003)

		(Mackas and Beaugrand 2010). • Euphausiid abundance data from broadly based California Cooperative Oceanic Fisheries Investigation surveys in California and Baja California sectors of the California Current provided a time series distinguishing periodic, rhythmic and irregular species patterns (Brinton and Townsend 2003).
(b) Relevant to management concerns	1	• Critical link in food web. • PFMC preemptively outlawed a krill fishery because they recognized the importance of krill as forage for other fishery species (groundfish, HMS, CPS). • Many fish, mammals, and birds depend on krill directly or indirectly as a primary food resource, e.g. Pacific hake *Merluccius productus*, juvenile rockfish *Sebastes* spp., salmonids, whales, and auklets (Sydeman et al. 2011; Beaugrand et al. 2003). • Grouped in large carnivorous zooplankton group of Atlantis ecosystem model of CCS, based on densities from Newport line (Brand et al. 2007; Horne et al. 2009) or DFO surveys in Barkley Sound (Sydeman et al. 2011). • Major predators of mesozooplankton (Mackas and Beaugrand 2010). • Field and Francis (2006) reported that a large proportion of the energy flux in the CCE flows through krill, underscoring the critical role krill play in regulating ecosystem productivity (Croll 2005).
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	1	• Yes - Community composition related to climate or regime shifts (Peterson 2009; Mackas and Beaugrand 2010, Gomez-Gutierrez et al. 2005), although understanding of the spatio-temporal dynamics of krill populations remains fragmentary. • Comparisons with environmental indexes indicate significant correlations with biomass of warmwater species, most notably in coastal *Nyctiphanes simplex* (Brinton and Townsend 2003; deYoung et al. 2008). • Positive euphausiid anomalies were correlated with anomalously weak summer upwelling off of Vancouver Island (Mackas et al. 2001).
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	1	• Ecosystem models suggest impacts of harvesting krill (large zooplankton) to be medium to high among global marine systems (Smith et al. 2011; Kaplan and Leonard 2012).
(e) Linkable to scientifically-defined reference points & progress targets	0.5	• Yes. Tropical warm-species vs. cold-species relative abundance / ratio (anomaly). • Lower counts and higher patchiness of euphausiids in samples may increase variability. • Fishery Management Plan reference points for managed pelagic species (B_{40} and B_{25} rules)

		have been applied as reference targets, but simulation models by Kaplan et al. (2013) and Kaplan and Leonard (2012) show that this level of forage species removal is likely to impact the abundance of other target species, protected species, and the structure of the ecosystem and suggest an alternative.
(g) Concrete & Numerical	0	• Poorly sampled in BPA Plume survey.
(h) Historical data or information available	0	• Poorly sampled in BPA Plume survey.
(i) Operationally simple	0	• Poorly sampled in BPA Plume survey.
(k) Broad spatial coverage	0	• Poorly sampled in BPA Plume survey.
(l) Continuous time series	0	• Poorly sampled in BPA Plume survey.
(m) Spatial & temporal variation understood	0	• Mesh size too large; poorly characterized with gear.
(n) High signal-to-noise ratio	0	• Poorly sampled, resulting in high noise.
(o) Understood by the public & policymakers	0.5	• Equivocal. Relative importance of krill/euphausiids seems to be rising among scientists and policy-makers, in part due to improved acoustic sampling methods of this patchily distributed species, but public likely unaware of food-web links.
(p) History of reporting	0.5	• Reporting not as extensive as copepods; perhaps related to poor sampling methods (Hewitt and Demer 2000). • Copepods appear to be focus of most zooplankton sampling, although CalCOFI has tracked euphausiids for decades, and there is information available in the NCC (Mackas et al 2001).
(q) Cost-effective	0.5	• Patchily distributed - requiring acoustic surveys. • Data mining from relatively new large-scale hydroacoustic surveys (Santora et al. 2011) and older, station-based zooplankton net samples (e.g., CalCOFI) has been the best source of data. • Capture efficiency of euphausiids is believed to differ between day and night samples due to visual avoidance of the net (e.g., Shaw and Robinson 1998). • Existing samples from BPA trawls in northern CC use large mesh sizes that do not adequately sample euphausiids well (R. Brodeur, pers. obs.).

(r) Anticipatory or leading indicator	0.5	- Possibly, due to position in food web and rapid response to environmental / ocean conditions (e.g., Miller et al. 2010).
(t) Regionally/nationally/internationally compatible	1	- Korean Peninsula (Rebstock and Kang 2003); N. Atlantic (Beaugrand et al. 2003); Alaska (AFSC 2010); Antarctic (Reid et al. 2005); British Columbia (Mackas et al. 2001).
TOTAL SCORE	7.5	

DIVERSITY INDICES – SPECIES RICHNESS [12]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 12

Summary: Species richness is a crucial property of ecological communities that is highly relevant to management concerns and public perception, but relationships to ecosystem function are complex. There is no support that the richness of pelagic nekton communities responds to management actions or pressures. The BPA dataset provides a useful platform for evaluating species richness of the pelagic nekton community; little evidence it could be used as an anticipatory indicator.

Criterion	**Score**	**Explanation**
(a) Theoretically sound	0.5	- Substantial literature on biodiversity and ecosystem function. - Species diversity is a crucial property of ecological communities: it is the primary descriptor of community structure, and it is generally believed to be a major determinant of the functioning and the dynamics of ecological communities (Wilson 1999 and others). Therefore, diversity measurement is often a first step in characterizing an ecological community (e.g., Magurran 2004). - Species richness is the number of different species represented in a particular ecological community, landscape, or region. Often requires correction for sampling effort

		(rarefaction). • Stachowicz et al. 2007 note that relationships between richness and ecosystem function are complex, but that communities are more stable at higher richness. Populations can be more variable but community level processes are more stable. • Suggested from microbial communities that one cannot reliably estimate the absolute and relative number of species present in a community without making unsupported assumptions about species abundance distributions because sample data do not contain information about the number of rare species in the tail of abundance distributions (Haegeman et al. 2013).
(b) Relevant to management concerns	1	• Yes. Biodiversity, composed of Richness and Evenness, is often a stated goal of ecosystem management (Palumbi et al. 2009; Gislason et al. 2000; Samhouri et al. 2009) • World leaders committed to reduce biodiversity loss via the Convention on Biological Diversity (CBD) in 2002.
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	0.5	• May not be highly sensitive to change. • Species-sampling intensity relationships require rarefraction because a decrease in abundance of all spp will lead to lower richness per sample (e.g., haul). • In Gulf of Maine, changes in species diversity (both Shannon's and richness) were greatest on interannual scales, intermediate on seasonal scales, and smallest across regions, in contrast to abundance patterns, suggesting that zooplankton diversity may be a more sensitive indicator of ecosystem response to interannual climate variation than zooplankton abundance (Johnson et al. 2011). • Richness was used by Reese and Brodeur (2006) to identify areas of biological activity, with flow and circulation suggested as the primary drivers of these patterns.
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	0	• Little support for this in the literature related to CPS. • Other studies on benthic invertebrate communities do show relationship between species richness and fishing /dredging (Reiss et al 2009, Gaspar et al 2009).
(e) Linkable to scientifically-defined reference points & progress targets	0.5	• Both Link (2005) and Dulvy et al. (2006) note linking diversity indices to targets or reference points is difficult. • Link et al. 2005 give a criterion for linking Richness to Reference Points but the choice is somewhat arbitrary. • Natural/base-line levels of richness may vary so absolute values may not be comparable in terms of thresholds.

(g) Concrete & Numerical	1	- Large trawls (336 m^2 mouth opening) in the upper 18-20 m of the water column at every station. - Mesh sizes ranged from 162.6 cm in the throat of the trawl near the jib lines to 8.9 cm in the cod end. - To maintain catches of small fish and squids, a 6.1 m long, 0.8 cm knotless liner was sewn into the cod end. - Total counts and weights are available for each nekton taxon caught. - Length data are recorded for a subset of individuals (n=50 per station) of all species.
(h) Historical data or information available	0.5	- Since 1998; annual systematic surveys have been conducted using consistent methodology in May, June, and September of each year.
(i) Operationally simple	1	- Measurements of hydrography, zooplankton and pelagic fishes are made at each station. - Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.
(k) Broad spatial coverage	0.5	- Mid-Oregon to North WA coast (Newport to Tatoosh Island; along nine transect lines at 50-55 stations, ranging from 45 to 48 °N).
(l) Continuous time series	1	- 15 y (1998-2012), in June and September.
(m) Spatial & temporal variation understood	1	- Brodeur et al. 2005, Thompson et al. in review.
(n) High signal-to-noise ratio	1	- Brodeur et al. 2005, Thompson et al. in review.
(o) Understood by the public & policymakers	1	- Yes. Thompson and Starzomski 2007 provide a general review that shows a general understanding and positive impression towards 'biodiversity' among the public. - World leaders committed to reduce biodiversity loss via the Convention on Biological Diversity (CBD) in 2002.
(p) History of reporting	0.5	- Biodiversity measures have been widely used as indicators of ecosystem response/state (e.g., Chesapeake Bay index of biotic integrity, Australia's state of the environment, etc.), but CPS are generally not the focal species group of these reporting efforts.
(q) Cost-effective	1	- Can be calculated from current data sets and monitoring programs.
(r) Anticipatory or leading indicator	0.5	- Uncertain. - Could be monitored annually to detect change. - Changes in species diversity were greatest on interannual scales, intermediate on

		seasonal scales, and smallest across regions, in contrast to abundance patterns, suggesting that zooplankton diversity may be a more sensitive indicator of ecosystem response to interannual climate variation than zooplankton abundance (Johnson et al. 2011).
(t) Regionally/nationally/internationally compatible	0.5	• Yes. Species richness used worldwide as an indicator of ecosystem health, though often used in combination with other measures of diversity (e.g., evenness) (http://www.environment.gov.au/soe/2011/report/biodiversity/index.html). Few regions, however, focus on pelagic nekton (but see Johnson et al. 2011, Gulf of Maine); rather, benthic invertebrate (e.g., Chesapeake Bay; http://ian.umces.edu/ecocheck/report-cards/chesapeake-bay/2012/indicators/benthic_index/) are generally used.
TOTAL SCORE	12	

DIVERSITY INDICES – SHANNON DIVERSITY [12.5]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 12.5

Summary: Shannon diversity estimates show habitats occupied by pelagic nekton species can expand and contract in relation to the dynamic nature of the California Current at both seasonal and interannual periodicities. However, linking Shannon diversity to targets or reference points is difficult, making this indicator difficult to interpret or contextualize. The BPA time series meets most criteria for Shannon diversity data considerations; although there is little evidence pelagic nekton Shannon diversity metrics could be used as an anticipatory indicator.

Criterion	Score	Explanation

(a) Theoretically sound	0.5	• Substantial literature on biodiversity and ecosystem function • Species diversity is a crucial property of ecological communities: it is the primary descriptor of community structure, and it is generally believed to be a major determinant of the functioning and the dynamics of ecological communities (Wilson 1999 and others). Therefore, diversity measurement is often a first step in characterizing an ecological community (e.g., Magurran 2004). • Shannon Diversity is a diversity measure that incorporates both richness and evenness. • Modeling results from seven different Ecopath models in different systems showed detrivore, adult sablefish, and marine mammal biomass were positively correlated with consumption and negatively correlated with Shannon Diversity (Samhouri et al 2009).
(b) Relevant to management concerns	1	• Biodiversity, composed of Richness and Evenness, is often a stated goal of ecosystem management (Palumbi et al 2009; Gislason et al 2000; Samhouri et al. 2009). • World leaders committed to reduce biodiversity loss via the Convention on Biological Diversity (CBD) in 2002.
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	1	• Brodeur et al. (2008a) showed ichthyoplankton species composition/diversity was an indicator of warm and cold phases in northeast Pacific Ocean. • Analysis of diversity / species composition showed habitats occupied by pelagic nekton species expand and contract in relation to the dynamic nature of the California Current and are affected by changing ocean conditions at both seasonal and interannual periodicities (Brodeur et al. 2005). • In Gulf of Maine, changes in species diversity (both Shannon's and richness) were greatest on interannual scales, intermediate on seasonal scales, and smallest across regions, in contrast to abundance patterns, suggesting that zooplankton diversity may be a more sensitive indicator of ecosystem response to interannual climate variation than zooplankton abundance (Johnson et al. 2011).
(d) Responds predictably & is sufficiently sensitive to changes in a specific	0.5	• Samhouri et al. (2010) used ecological modeling to show Shannon diversity declined with increasing fishing pressur • However, Samhouri et al. note that the indicator-attribute relationship can switch

management action(s) or pressure(s)		depending upon the type of fishing pressure used in the model. This result might make the indicator-attribute relationship unpredictable in the real world.
(e) Linkable to scientifically-defined reference points & progress targets	0.5	• Samhouri et al. (2010) discuss identifying thresholds based on the relationship between the indicator or attribute and fishing pressure • Samhouri et al. (2010) did not find nonlinearity in the relationship between Shannon diversity and nearshore habitat pressure, but did find nonlinearity in the relationship between Shannon diversity and fishing. • Both Link (2005) and Dulvy et al (2006) note linking diversity indices to targets or reference points is difficult.
(g) Concrete & Numerical	1	• Large trawls (336-m^2 mouth opening) in the upper 18-20 m of the water column at every station. • Mesh sizes ranged from 162.6 cm in the throat of the trawl near the jib lines to 8.9 cm in the cod end. • To maintain catches of small fish and squids, a 6.1-m long, 0.8-cm knotless liner was sewn into the cod end. • Total counts and weights are available for each nekton taxon caught. • Length data are recorded for a subset of individuals (n=50 per station) of all species.
(h) Historical data or information available	0.5	• Since 1998; Annual systematic surveys have been conducted using consistent methodology in May, June, and September of each year
(i) Operationally simple	1	• Measurements of hydrography, zooplankton and pelagic fishes are made at each station. • Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.
(k) Broad spatial coverage	0.5	• Mid-Oregon to North WA coast (Newport to Tatoosh Island; along nine transect lines at 50-55 stations, ranging from 45 to 48° N.)
(l) Continuous time series	1	• 15 y (1998-2012), in June and September

(m) Spatial & temporal variation understood	1	• Brodeur et al. 2005, Thompson et al. in review MEPS
(n) High signal-to-noise ratio	1	• Brodeur et al. 2005, Thompson et al. in review MEPS
(o) Understood by the public & policymakers	0.5	• Somewhat understood; Biodiversity has entered the common lexicon. • While widely used by ecologists, Shannon diversity is influenced by both richness (# of spp) and evenness making the interpretation of specific changes difficult without further analyses or context. • Moreover, the numerical value is dependent upon the log-base used in calculation and other factors. The result is that the numerical value has meaning only within the context of the data set and the log-base used and is not directly interpretable to the general public. • World leaders committed to reduce biodiversity loss via the Convention on Biological Diversity (CBD) in 2002.
(p) History of reporting	0.5	• Biodiversity measures have been widely used as indicators of ecosystem response/state (e.g., Chesapeake Bay index of biotic integrity, Australia's state of the environment, etc.), but CPS are generally not the focal species group of these reporting efforts.
(q) Cost-effective	1	• Can be calculated from current data sets and monitoring programs
(r) Anticipatory or leading indicator	0.5	• Uncertain. • Could be monitored annually to detect change. • Changes in species diversity were greatest on interannual scales, intermediate on seasonal scales, and smallest across regions, in contrast to abundance patterns, suggesting that zooplankton diversity may be a more sensitive indicator of ecosystem response to interannual climate variation than zooplankton abundance (Johnson et al. 2011).
(t) Regionally/nationally/internationally compatible	0.5	• Yes. Shannon diversity is used worldwide as an indicator of ecosystem health, though often used in combination with other measures of diversity (e.g., richness) (http://www.environment.gov.au/soe/2011/report/biodiversity/index.html). Few

		regions, however, focus on pelagic nekton (but see Johnson et al., Gulf of Maine); rather, benthic invertebrates (e.g., Chesapeake Bay; http://ian.umces.edu/ecocheck/report-cards/chesapeake-bay/2012/indicators/benthic_index/) are generally used.
TOTAL SCORE	12.5	

(ECOSYSTEM) MEAN TROPHIC LEVEL [10]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 10

Summary: The diet of many pelagic nekton species change on an annual basis, making trophic level assignments difficult and the data signal noisy, and often confounded by major shifts of some abundant taxa. MTL is increasingly used as an ecosystem indicator, but cannot be considered an anticipatory indicator in this context.

Criterion	**Score**	**Explanation**
(a) Theoretically sound	0.5	• Theoretically could be done, may be especially informative when integrating pelagic estimates with other ecosystem components (e.g., demersal / benthic community). • Ecosystem MTL is calculated from Fisheries Independent data, including surveys and assessments; in a worldwide review of multiple ecosystems, survey MTL and assessment MTL were generally higher than catch MTL, reflecting the greater focus of surveys and stock assessments on bottom-dwelling high-trophic-level fish species that account for only a moderate proportion of total catch weight (Branch et al. 2010) • Average trophic level showed potential as an indicator if, and only if, good diet data

		are available; i.e., high potential but sensitive to data quality; alternatively, size-based indicators of MTL are easily collected and less error-prone (Fulton et al. 2005) • Shown to be a good indicator for demersal fish (Samhouri et al. 2009) • MTL was an ambiguous measure of invertebrate exploitation compared to indicators of mortality rate or biomass (Caddy & Garibaldi 2000) • Furthermore, diet/trophic level assignment of many forage (pelagic nekton) species change from year to year - attributed to variations in a few key prey species due to environmental variability (McFarlane and Beamish 2001, Brodeur and Pearcy 1992).
(b) Relevant to management concerns	0.5	• Ecosystem models used to examine systemic effects of fishing LTL species found variation in impact of harvest has potentially important management implications. • Generic properties that explain and predict the variation in response were primarily relative abundance of the group in the ecosystem and connectivity of the group in the food web, whereas trophic level was NOT a good predictor (Smith et al. 2011). • MTL is probably more relevant to management concerns for conservationists than for fisheries concerns.
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	1	• Diet/trophic level assignment of many forage species change from year to year - attributed to variations in a few key prey species due to environmental drivers (McFarlane and Beamish 2001, Brodeur and Pearcy 1992). • Debate continues over the best means of calculating MTL in an effort to minimize data requirements (Rochet and Trenkel 2003). • While the metric tracks changes in trophic structure it is often necessary to distinguish between changes in high and low TL taxa. For example, a decrease in MTL could be the result of a decrease in high TL fishes or a increase in the abundance of low TL fishes or both (Pauly et al. 1998, Pauly & Watson 2005, Essington 2006, Branch et al. 2010, Stergiou & Tsikliras 2011, Tolimieri et al. 2013).
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	0.5	• No consistency in comparisons between catch, survey and assessment MTL in individual ecosystems. Catch MTL is negatively correlated with survey MTL for 13 of 29 surveys, and negatively correlated with assessment MTL in 4 of 9 ecosystems (Branch et al. 2010). • While the metric tracks changes in trophic structure it is often necessary to

		distinguish between changes in high and low TL taxa. For example, a decrease in MTL could be the result of a decrease in high TL fishes or a increase in the abundance of low TL fishes or both (Pauly et al. 1998, Pauly & Watson 2005, Essington 2006, Branch et al. 2010, Stergiou & Tsikliras 2011). • Diet/trophic level assignment of many forage species change from year to year - attributed to variations in a few key prey species due to environmental variability (McFarlane and Beamish 2001, Brodeur and Pearcy 1992, Tolimieri et al. 2013).
(e) Linkable to scientifically-defined reference points & progress targets	1	• Essington et al. 2006 indicate that a 0.1 change in MTL indicates a 50% change in the primary production required to support a given level of fisheries harvest (also see Christensen and Pauly 1995, Pauly et al. 1998, Pauly & Watson 2005, Essington 2006, Branch et al. 2010, Stergiou & Tsikliras 2011).
(g) Concrete & Numerical	1	• Large trawls (336-m^2 mouth opening) in the upper 18-20 m of the water column at every station. • Mesh sizes ranged from 162.6 cm in the throat of the trawl near the jib lines to 8.9 cm in the cod end. • To maintain catches of small fish and squids, a 6.1-m long, 0.8-cm knotless liner was sewn into the cod end. • Total counts and weights are available for each nekton taxon caught. • Length data are recorded for a subset of individuals (n=50 per station) of all species.
(h) Historical data or information available	0.5	• Since 1998; Annual systematic surveys have been conducted using consistent methodology in May, June, and September of each year.
(i) Operationally simple	1	• Measurements of hydrography, zooplankton and pelagic fishes are made at each station. • Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.
(k) Broad spatial coverage	0.5	• Mid-Oregon to North WA coast (Newport to Tatoosh Island; along nine transect lines at 50-55 stations, ranging from 45 to 48° N.)

(l) Continuous time series	1	• 15 y (1998-2012), in June and September
(m) Spatial & temporal variation understood	0	• trophic levels vary inter-annually and seasonally and spatially, difficult to assign
(n) High signal-to-noise ratio	0	• difficult to distinguish signal from noise because of uncertainty in TL assignment
(o) Understood by the public & policymakers	1	• MTL included in Convention of Biological Diversity list of indicators (Pauly & Watson 2005, Stergiou & Tsikliras 2011)
(p) History of reporting	0.5	• Increasingly used as an ecosystem indicator, although catch MTL (not ecosystem MTL) was the primary marine index chosen by the Convention on Biological Diversity to measure global biodiversity, and has been applied widely to report on the state of the marine environment (Pauly & Watson 2005, Stergiou & Tsikliras 2011, Branch et al. 2010) • MTL generally based on demersal trawl surveys or assessments, not pelagic community components.
(q) Cost-effective	0.5	• Some existing pelagic surveys could be used (e.g., CalCOFI, Field et al., Brodeur et al.), though most existing calculations based on groundfish data from Stock Assessments and Keller et al. 2008 (NWFSC trawl survey, 1998-present), Weinberg et al. 2002 (AFSC triennial survey, 1977-2001); REEF.org (for data at diver depths.) • Would need to evaluate diets of species by age class, as well as relative biomass of survey.
(r) Anticipatory or leading indicator	0	• Diet/trophic level assignment of many forage (pelagic nekton) species change from year to year - attributed to variations in a few key prey species due to environmental variability (McFarlane and Beamish 2001, Brodeur and Pearcy 1992) • However, understanding the anticipatory nature of dietary changes is somewhat different than understanding the nature of MTL classification.
(t) Regionally/nationally/internationally compatible	0.5	• Catch MTL (not ecosystem MTL) included in Convention of Biological Diversity list of indicators (Pauly & Watson 2005, Stergiou & Tsikliras 2011) but exact values will depend on local food web. Global comparison of various ecosystem MTL values

		completed by Branch et al. (2010).
TOTAL SCORE	10	

FORAGE FISH / JELLYFISH RATIO [11.5]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 11.5

Summary: Both forage fish and jellyfish biomass score highly individually as indicators, and the data are readily obtained from the BPA data set. In addition, the conceptual underpinnings of this ratio are clear: as the biomass of forage fish increases relative to that of jellyfish, there is more production available to fisheries and to larger predatory fish, mammals, and birds. However, the statistical properties of the ratio of these 2 quantities are understudied, making it more difficult to use the ratio of forage fish to jellyfish to determine appropriate reference points, communicate to the public, and compare across ecosystems.

Criterion	**Score**	**Explanation**
(a) Theoretically sound	1	• As the biomass of forage fish increases relative to that of jellyfish, there is more production available to fisheries and to larger predatory fish, mammals, and birds. • Fulton et al. 2005; Samhouri et al. 2009; Kaplan et al. 2011; Shannon et al. 2009; Daskalov et al. 2007
(b) Relevant to management concerns	1	• Caddy 2000; Fulton et al. 2003; De Leiva Moreno et al 2000 • Also, forage fish account for over 30% of global fish landings, and play an important role in marine food webs because they are the principal means of transferring

		production from plankton to larger predatory fish and to marine mammals and seabirds (Smith et al. 2011) • Fishery Management Plans for assessed species (e.g., sardine), as well as entire CalCOFI sampling program, attest to management importance.
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	1	• Forage fish/jellyfish ratio was highly correlated with Simpson diversity, mean trophic level, target group biomass (Samhouri et al. 2009; Kaplan et al. 2011; Daskalov et al. 2007).
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	1	• Changes in the ratio of forage fish to jellyfish biomass across seven food web models (six in the north Pacific Ocean) were correlated with measures of diversity after perturbing the systems with various levels of fishing pressure (Samhouri et al. 2009). Also see Purcell 2012; Kaplan et al. 2011; Richardson et al. 2009.
(e) Linkable to scientifically-defined reference points & progress targets	0.5	• Changes in the ratio of these biomass groups could be used as a target, but none have been used to date
(g) Concrete & Numerical	1	• Large trawls (336-m^2 mouth opening) in the upper 18-20 m of the water column at every station. • Mesh sizes ranged from 162.6 cm in the throat of the trawl near the jib lines to 8.9 cm in the cod end. • To maintain catches of small fish and squids, a 6.1-m long, 0.8-cm knotless liner was sewn into the cod end. • Total counts and weights are available for each nekton taxon caught. • Length data are recorded for a subset of individuals (n=50 per station) of all species.
(h) Historical data or information available	0.5	• Since 1998; Annual systematic surveys have been conducted using consistent methodology in May, June, and September of each year
(i) Operationally simple	1	• Measurements of hydrography, zooplankton and pelagic fishes are made at each station. • Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.

(k) Broad spatial coverage	0.5	• Mid-Oregon to North WA coast (Newport to Tatoosh Island; along nine transect lines at 50-55 stations, ranging from 45 to 48° N.)
(l) Continuous time series	1	• 15 y (1998-2012), in June and September
(m) Spatial & temporal variation understood	0.5	• Annual and summer/spring variability well-understood for most taxa; seasonal less so. • Sampling occurs during day or crepuscular periods so diel patterns are less clear and myctophids are not well characterized.
(n) High signal-to-noise ratio	0.5	• Sampled taxa have naturally highly variable populations • For some of the best sampled taxa signal to noise is high or can be discerned. Poor for myctophids.
(o) Understood by the public & policymakers	0	• Ratios of groups are not easily understood
(p) History of reporting	0	• No
(q) Cost-effective	1	• Data mining by group is all that is needed. Forage fishes and jellyfishes often caught in surveys by identical gear (Brodeur et al. 2004, 2006).
(r) Anticipatory or leading indicator	0.5	• Evidence exists suggesting that forage fish and jellyfish individually could be leading indicators, though this ratio indicator is understudied (Anderson and Piatt 1999)
(t) Regionally/nationally/internationally compatible	0.5	• This ratio and its constituent parts are increasingly reported around the world (Pauly et al. 2009, Condon et al. 2013)
TOTAL SCORE	11.5	

PISCIVOROUS FISH / ZOOPLANKTIVOROUS FISH RATIO [5]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 5

Summary: The ratio of piscivorous fish to zooplanktivorous fish biomass is in theory a sound indicator, albeit with challenging statistical properties that limit ease of communication and setting of reference points. However, there are very few piscivorous fishes represented in the CPS taxa sampled by the BPA survey. Thus, this indicator is likely to be inappropriate for use with the BPA data set and most others geared to sample CPS.

Criterion	Score	Explanation
(a) Theoretically sound	0.5	• Fulton et al. 2005; Samhouri et al. 2009; Kaplan et al. 2011. • CPS data sets could include the following piscivorous fish species: hake, dogfish, adult salmon, jack mackerel, and sharks (Brodeur et al. 2006). • However, the representation of piscivorous fishes in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be piscivores, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to piscivory.
(b) Relevant to management concerns	1	• The zooplanktivores and piscivores included in some CPS data sets are of critical management importance, as evidenced by their inclusion in PFMC Fishery Management Plans (hake, salmon, etc.; www.pcouncil.org)
(c) Responds predictably & is sufficiently sensitive to changes in a specific	1	• Changes in the ratio of piscivorous to zooplanktivorous biomass will obviously affect community composition, but variation in overall community composition may not be detected by variation in the ratio of these two groups alone (Fulton et al. 2005;

ecosystem attribute(s)		Samhouri et al. 2009; Kaplan et al. 2011).
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	1	• Changes in the ratio of Piscivorous to Zooplanktivorous fish biomass across seven food web models (six in the north Pacific Ocean) were correlated with measures of diversity after perturbing the systems with various levels of fishing pressure (Fulton et al. 2005; Samhouri et al. 2009; Kaplan et al. 2011).
(e) Linkable to scientifically-defined reference points & progress targets	0.5	• Changes in the ratio of these biomass groups could be used as a target, but none have been used to date
(g) Concrete & Numerical	0	• The representation of piscivorous fishes in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be piscivores, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to piscivory.
(h) Historical data or information available	0	• The representation of piscivorous fishes in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be piscivores, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to piscivory.
(i) Operationally simple	0	• The representation of piscivorous fishes in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be piscivores, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to piscivory.
(k) Broad spatial coverage	0	• The representation of piscivorous fishes in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be piscivores, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to piscivory.
(l) Continuous time series	0	• The representation of piscivorous fishes in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be piscivores, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to piscivory.

(m) Spatial & temporal variation understood	0	• poorly characterized with gear • tow speeds are too slow to capture most highly mobile, top predators
(n) High signal-to-noise ratio	0	• poorly sampled, resulting in high noise;
(o) Understood by the public & policymakers	0	• Ratios of groups are not easily understood
(p) History of reporting	0	• No
(q) Cost-effective	0.5	• Data mining by group is needed. However, ideally different gear types would be used to survey pelagic piscivores and pelagic zooplanktivores, which would make comprehensive surveys expensive.
(r) Anticipatory or leading indicator	0	• Unknown
(t) Regionally/nationally/internationally compatible	0.5	• This ratio and its constituent parts are increasingly reported around the world (Caddy and Garibaldi 2000, Fulton et al. 2005, Kaplan et al. 2011)
TOTAL SCORE	5	

FINFISH / CRUSTACEAN BIOMASS RATIO [4]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 4

Summary: The conceptual underpinnings of this indicator are clear. In general, finfish tend to be represented on a higher mean trophic level than crustaceans. Thus if fishing down or through the food web occurs, the ratio of finfish to crustacean biomass should decline. However, this indicator is usually intended to include benthic animals as part of the crustacean biomass in the denominator, and so is not well-suited to derivation from CPS data sets like the BPA survey.

Criterion	**Score**	**Explanation**
(a) Theoretically sound	0	- The conceptual underpinnings of this indicator are clear. In general, finfish tend to be represented on a higher mean trophic level than crustaceans. Thus if fishing down or through the food web occurs (Pauly et al. 1998, Essington et al. 2006), the ratio of finfish to crustacean biomass should decline. - This indicator is usually intended to include benthic animals as part of the crustacean biomass in the denominator (Kaplan et al. 2011), and so is not well-suited to derivation from CPS data sets.
(b) Relevant to management concerns	0.5	- This indicator is usually intended to include benthic animals as part of the crustacean biomass in the denominator (Kaplan et al. 2011), and so derivation from CPS data sets would only be relevant if larval stages of crabs, lobsters, etc were included in the samples and known to be related to benthic population dynamics. - However, krill (and pink shrimp) are obviously important forage/pelagic crustaceans that are not mentioned here; though there is little evidence in the literature comparing the relative biomass of these groups to pelagic finfish.
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	1	- Changes in the ratio of finfish to crustacean biomass will obviously affect community composition, but variation in overall community composition may not be detected by variation in these groups alone (Kaplan et al. 2011, Worm and Myers 2003, Zhang and Chen 2007).
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	1	- The ratio of finfish to crustacean biomass can be dynamic and related to human pressures on the ecosystem, such as fishing (Kaplan et al. 2011, Worm and Myers 2003, Zhang and Chen 2007).
(e) Linkable to scientifically-defined	0.5	- Changes in the ratio of these biomass groups could be used as a target, but none have

reference points & progress targets		been used to date
(g) Concrete & Numerical	0	• This indicator is usually intended to include benthic animals as part of the crustacean biomass in the denominator (Kaplan et al. 2011), and so derivation from CPS data sets would only be relevant if larval stages of crabs, lobsters, etc were included in the samples and known to be related to benthic population dynamics.
(h) Historical data or information available	0	• This indicator is usually intended to include benthic animals as part of the crustacean biomass in the denominator (Kaplan et al. 2011), and so derivation from CPS data sets would only be relevant if larval stages of crabs, lobsters, etc were included in the samples and known to be related to benthic population dynamics.
(i) Operationally simple	0	• This indicator is usually intended to include benthic animals as part of the crustacean biomass in the denominator (Kaplan et al. 2011), and so derivation from CPS data sets would only be relevant if larval stages of crabs, lobsters, etc were included in the samples and known to be related to benthic population dynamics.
(k) Broad spatial coverage	0	• This indicator is usually intended to include benthic animals as part of the crustacean biomass in the denominator (Kaplan et al. 2011), and so derivation from CPS data sets would only be relevant if larval stages of crabs, lobsters, etc were included in the samples and known to be related to benthic population dynamics.
(l) Continuous time series	0	• This indicator is usually intended to include benthic animals as part of the crustacean biomass in the denominator (Kaplan et al. 2011), and so derivation from CPS data sets would only be relevant if larval stages of crabs, lobsters, etc were included in the samples and known to be related to benthic population dynamics.
(m) Spatial & temporal variation understood	0	• mesh size too large; poorly characterized with gear
(n) High signal-to-noise ratio	0	• poorly sampled, resulting in high noise;
(o) Understood by the public &	0.5	• Rose 2007 citing Worm & Myers 2003, but ratios of biomass are a little obscure to the general public

policymakers		
(p) History of reporting	0	- No
(q) Cost-effective	0	- Because the crustacean groups of interest are largely sampled using different gear than is used for surveying pelagic finfish, sampling would likely be expensive.
(r) Anticipatory or leading indicator	0	- Unknown
(t) Regionally/nationally/internationally compatible	0.5	- This ratio and its constituent parts are increasingly reported around the world (Myers and Worm 2003, Zhang and Chen 2007, Kaplan et al. 2011)
TOTAL SCORE	4	

ZOOPLANKTIVOROUS FISH BIOMASS [14]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 14

Summary: Reliable indicator of changing ecosystem state and important to global fish landings and food webs. However, aggregating the biomass of all zooplanktivorous fish species may reduce its sensitivity as an indicator, and it may not respond to management actions if a threshold shift has occurred in the ecosystem. This indicator is concrete and simple, although zooplanktivorous fishes naturally have highly variable populations (high signal:noise ratio). Changes in trends of aggregate groups will be concurrent, at best.

Criterion	Score	Explanation
(a) Theoretically sound	1	• Changes in the biomass of pelagic species may indicate changes in ecosystem state as a function of fishing down the food web, predatory release of prey populations (forage fish), or insufficient forage base for top predators (Link 2005). • Zooplanktivorous fish has been described as the best indicator of total biomass and net primary production in the system using 7 food web models from the North Pacific and the Baltic Sea (Samhouri et al. 2009) • Threshold-like shifts in Baltic Sea pelagic ecosystems appear to be driven by planktivore abundance (the clupeid, sprat) that separates 2 ecosystem configurations in which zooplankton dynamics are driven by either hydroclimatic forces or predation pressure (Casini et al. 2009).
(b) Relevant to management concerns	1	• Many zooplanktivorous fish are forage fish, accounting for over 30% of global fish landings, playing an important role in marine food webs because they are the principal means of transferring production from plankton to larger predatory fish and to marine mammals and seabirds (Smith et al. 2011) • Fishery Management Plans for assessed species (e.g., sardine), as well as entire CalCOFI sampling program, attest to management importance.
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	1	• Zooplanktivorous fish represents the best indicator of total ecosystem biomass based on 7 food web models from the North Pacific and Baltic Sea (Samhouri et al. 2009) • Change in zooplanktivorous fish biomass may be a result of incremental changes in all species groups, but a change (or no change) in a single species may not be indicative of the community as a whole • The biomass of small, zooplanktivorous forage fish in the northeast Pacific appears to increase during cold ocean conditions as a result of zooplankton assemblage composition change (http://www.nwfsc.noaa.gov/research/divisions/fe/estuarine/oeip/ha-under-development.cfm)
(d) Responds predictably & is sufficiently sensitive to changes in a specific	0.5	• Mgt. action: the pressure of fishing these species at conventional maximum sustainable yield (MSY) levels can have large impacts on other parts of the ecosystem, particularly when they constitute a high proportion of the biomass in the ecosystem

management action(s) or pressure(s)		or are highly connected in the food web (Smith et al. 2011) • Precautionary management will theoretically affect population, though not if threshold shift has occurred or environmental drivers have changed.
(e) Linkable to scientifically-defined reference points & progress targets	1	• Link (2005) set warning and limit reference points as follows: if $B_{pelagic}$ > 75% B_{total}, or if $B_{pelagic}$ < 20% B_{total} in any given year, then a warning threshold has been exceeded. The LRPs are set at $B_{pelagic}$ > 85% B_{total}, and at $B_{pelagic}$ < 10% B_{total}. • Casini et al. 2009: Shifts in Baltic Sea pelagic ecosystems appear to be driven by planktivore abundance (the clupeid, sprat) that separates 2 ecosystem configurations in which zooplankton dynamics are driven by either hydroclimatic forces or predation pressure; identified an ecological threshold, corresponding to a planktivore abundance of 17 x 10^{10} individuals. • Fishery Management Plan reference points for managed species (B40 and B25 rules) have been applied as reference targets, but simulation models by Kaplan et al. (2013) show that this level of forage species removal is likely to impact the abundance of other target species, protected species and the structure of the ecosystem.
(g) Concrete & Numerical	1	• Large trawls (336-m^2 mouth opening) in the upper 18-20 m of the water column at every station. • Mesh sizes ranged from 162.6 cm in the throat of the trawl near the jib lines to 8.9 cm in the cod end. • To maintain catches of small fish and squids, a 6.1-m long, 0.8-cm knotless liner was sewn into the cod end. • Total counts and weights are available for each nekton taxon caught. • Length data are recorded for a subset of individuals (n=50 per station) of all species.
(h) Historical data or information available	0.5	• Since 1998; Annual systematic surveys have been conducted using consistent methodology in May, June, and September of each year
(i) Operationally simple	1	• Measurements of hydrography, zooplankton and pelagic fishes made at each station. • Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.

(k) Broad spatial coverage	0.5	• Mid-Oregon to North WA coast (Newport to Tatoosh Island; along nine transect lines at 50-55 stations, ranging from 45 to 48° N.)
(l) Continuous time series	1	• 15 y (1998-2012), in June and September
(m) Spatial & temporal variation understood	0.5	• Annual and spring/summer variation well-understood for most taxa; seasonal less so. • Sampling occurs during day or crepuscular periods so diel patterns are less clear and myctophids are not well characterized.
(n) High signal-to-noise ratio	0.5	• Sampled taxa have naturally highly variable populations • For some of the best sampled taxa signal to noise is high or can be discerned, although poor for myctophids.
(o) Understood by the public & policymakers	1	• Yes; relative trends in biomass of each component of the community are easily understood.
(p) History of reporting	1	• Link 2005; Samhouri 2009, Fulton et al. 2005;
(q) Cost-effective	1	• Assessment data already collected for many of these species; data mining is all that is needed
(r) Anticipatory or leading indicator	0.5	• Changes in trends of aggregate groups will always be concurrent at best; • Rapid response of zooplanktivorous fishes and other nekton (distributional anomalies) to delayed upwelling (Brodeur et al. 2006); zooplanktivorous fishes generally show a 1-yr lag between a change in ocean phase and population response (http://www.nwfsc.noaa.gov/research/divisions/fe/estuarine/oeip/ha-under-development.cfm), but this is relatively fast compared to other North Pacific fish populations (Yatsu et al. 2008).
(t) Regionally/nationally/internationally compatible	1	• Link 2002, Methratta & Link 2006
TOTAL SCORE	14	

PISCIVOROUS FISH BIOMASS [6]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 6

Summary: In principle, piscivorous fish biomass could serve as a reliable indicator of Ecological Integrity, as it is well supported theoretically as a metric that should decline if fishing down/through the food web occurs (Pauly et al. 1998, Essington et al. 2006). In addition, this indicator is reported in many ecosystems and is relatively easy to communicate. However, the representation of piscivorous fishes in CPS data sets is minimal, reference points for this indicator have yet to be established, and as an aggregate group it is unlikely to serve as a leading indicator of ecosystem changes.

Criterion	**Score**	**Explanation**
(a) Theoretically sound	0.5	• Fulton et al. 2005; Samhouri et al. 2009; Kaplan et al. 2011. • CPS data sets could include the following piscivorous fish species: hake, dogfish, adult salmon, jack mackerel, and sharks (Brodeur et al. 2006, Brodeur et al. in prep). • However, the representation of piscivorous fishes in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be piscivores, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to piscivory.
(b) Relevant to management concerns	1	• The piscivores included in some CPS data sets are of critical management importance, as evidenced by their inclusion in PFMC Fishery Management Plans (hake, salmon, etc.; www.pcouncil.org)

(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	0.5	• Piscivorous fish biomass tends to be more strongly correlated with a narrower range of ecosystem attributes than other indicators that have been tested using ecosystem model simulations (Fulton et al. 2005; Samhouri et al. 2009; Kaplan et al. 2011).
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	0.5	• As the majority of piscivorous fishes are the targets of fisheries, their biomass is expected to decline as fishing pressure increases. Similarly, as higher trophic level predators, bioaccumulation of toxins can lead to population declines (Fulton et al. 2005; Samhouri et al. 2009; Kaplan et al. 2011).
(e) Linkable to scientifically-defined reference points & progress targets	0.5	• It is feasible to identify thresholds, but none have been defined to date in the California Current. • Samhouri et al. 2010 showed that the biomass of a dominant piscivore was linkable to ecosystem thresholds; • Micheli et al. 2004 suggested a threshold time to recovery for pisicvores in no-take areas
(g) Concrete & Numerical	0	• No data available in BPA data set
(h) Historical data or information available	0	• No data available in BPA data set
(i) Operationally simple	0	• No data available in BPA data set
(k) Broad spatial coverage	0	• No data available in BPA data set
(l) Continuous time series	0	• No data available in BPA data set
(m) Spatial & temporal variation understood	0	• poorly characterized with gear; • tow speeds are too slow to capture most highly mobile, top predators
(n) High signal-to-noise ratio	0	• poorly sampled, resulting in high noise;
(o) Understood by the public &	1	• Yes; relative trends in biomass of each component of the community is easily

policymakers		understood, and a focus on predators like piscivores is particularly compelling.
(p) History of reporting	0.5	- Top predator measures like piscivore biomass have been widely used as indicators of ecosystem response/state (e.g., Branch et al. 2010, Estes et al. 2011, Fulton et al. 2005, Samhouri et al. 2009, Kaplan et al. 2011), although the representation of piscivorous fishes in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be piscivores, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to piscivory.
(q) Cost-effective	0.5	- Data mining by group is needed. - However, ideally multiple gear types would be used to survey pelagic piscivores, which could make comprehensive surveys expensive.
(r) Anticipatory or leading indicator	0	- Changes in trends of aggregate groups likely to be coincident at best; piscivores tend to be larger bodied and often longer lived, slower responders.
(t) Regionally/nationally/internationally compatible	1	- Methratta & Link 2006, Fulton et al. 2005, Kaplan et al. 2011
TOTAL SCORE	6	

TOP PREDATOR BIOMASS (TROPHIC LEVEL > 4) [6]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 6

Summary: Like piscivorous fish biomass, in principle top predator biomass could serve as a reliable indicator of Ecological Integrity, as it is well supported theoretically as a metric that should decline if fishing down/through the food web occurs (Pauly et al. 1998, Essington et al. 2006). In addition, this indicator is reported in many ecosystems and is relatively easy to communicate. However, the representation of top predator biomass in CPS data sets is minimal, reference points for this indicator have yet to be established, and as an aggregate group it is unlikely to serve as a leading indicator of ecosystem changes.

Criterion	**Score**	**Explanation**
(a) Theoretically sound	0.5	• The removal of top predators from ecosystems can result in a trophic cascade in which consumers are released from pressure and decrease the abundance of heterotrophs in the system, altering community composition (Samhouri et al. 2009; Estes et al. 2011). • CPS data sets could include the following predatory species: hake, dogfish, adult salmon, jack mackerel, and sharks (Brodeur et al. 2006). • However, the representation of top predators in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be top predators, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to top predator status.
(b) Relevant to management concerns	1	• Top predators have strong effects in marine ecosystems, and tend to be the targets of fisheries, making them highly relevant to management concerns (Jennings et al. 1995, Estes et al. 2011).
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	0.5	• Top predator response times can be slow because of long lifespans (Estes et al. 2011). • However, sometimes top predators are sensitive indicators (Pauly et al. 1998; Ward & Myers 2005).
(d) Responds predictably & is sufficiently sensitive to changes in a specific	0.5	• Top predator response times can be slow because of long lifespans (Estes et al. 2011). • However, some times top predators are sensitive indicators (Pauly et al. 1998; Ward

management action(s) or pressure(s)		& Myers 2005).
(e) Linkable to scientifically-defined reference points & progress targets	0.5	- Link (2005): thresholds in top predator biomass could be used as a target, but none has been used to date in the California Current
(g) Concrete & Numerical	0	- The representation of top predators in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be top predators, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to top predator status.
(h) Historical data or information available	0	- The representation of top predators in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be top predators, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to top predator status.
(i) Operationally simple	0	- The representation of top predators in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be top predators, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to top predator status.
(k) Broad spatial coverage	0	- The representation of top predators in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be top predators, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to top predator status.
(l) Continuous time series	0	- The representation of top predators in CPS data sets is not likely to be great for 2 reasons: (i) sampling gears not typically intended to target larger bodies pelagic fishes that may be top predators, and (ii) many of the CPS sampled will be younger and smaller individuals that have not made the ontogenetic shift to top predator status.
(m) Spatial & temporal variation understood	0	- poorly characterized with gear; - tow speeds are too slow to capture most highly mobile, top predators

(n) High signal-to-noise ratio	0	- poorly sampled, resulting in high noise;
(o) Understood by the public & policymakers	1	- Yes; trends in biomass of each component of the community is easily understood, and a focus on top predators is particularly compelling (Estes et al. 2011).
(p) History of reporting	0.5	- see above; Link 2005; Pauly et al. 1998
(q) Cost-effective	0.5	- Data mining by group is all that is needed
(r) Anticipatory or leading indicator	0	- Top predator response times can be slow because of long lifespans (Estes et al. 2011). - Even when top predators are sensitive indicators (Pauly et al. 1998; Ward & Myers 2005) they are likely to be coincident, not anticipatory indicators.
(t) Regionally/nationally/internationally compatible	1	- Link 2005, Fay et al. 2013
TOTAL SCORE	6	

SCAVENGER OR DETRITIVORE BIOMASS [5]

CCIEA Goal: Ecological integrity

Attribute: Community composition

Total score: 5

Summary: Scavenger biomass is in principle a theoretically sound indicator of ecological integrity, as it is expected to increase as fisheries discards and other human pressures accumulate. However, it is probably best represented with the inclusion of benthic detritivores and is at best partially tracked within CPS data sets like the BPA survey. Furthermore, it is an indicator that is likely to respond to ecosystem changes rather than lead them and does not have a long history of reporting.

Criterion	**Score**	**Explanation**
(a) Theoretically sound	0	- This indicator probably best represented with the inclusion of benthic detritivores (Ramsay et al. 1998, Samhouri et al. 2009). - Jellyfish, sampled in some CPS data sets, eat detritus (Ruzicka et al. 2007, Pauly et al. 2009), but it is not their primary dietary item (Brodeur et al. 2008), which is also the case for other CPS. - Therefore, while scavenger biomass is in principle a theoretically sound indicator of ecological integrity, it is at best partially tracked within CPS data sets.
(b) Relevant to management concerns	1	- Scavenger biomass is expected to increase as fisheries discards and other human pressures accumulate. - reviewed by Britton and Morton 1994, and by Purcell et al. 2007, 2012 for jellyfish
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	0.5	- Changes in scavenger biomass will obviously affect community composition, but variation in overall community composition may not be detected by variation in this trophic level alone.
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	1	- Acute increases in scavenger population after fishing activities are reasonably well documented and there is also some evidence that fishing activities induce chronic increases in scavenger populations. - Ramsay et al. 1998 or Purcell 2012 for jellyfish specifically.
(e) Linkable to scientifically-defined reference points & progress targets	0.5	- It is feasible to identify thresholds, but none have been defined to date in the California Current. - Samhouri et al. (2010) showed jellyfish biomass was linkable to ecosystem thresholds using an ecosystem model for British Columbia
(g) Concrete & Numerical	0	- Scavenger biomass is at best partially tracked within CPS data sets, as these data sets do not include benthic detritivores.

(h) Historical data or information available	0	• Scavenger biomass is at best partially tracked within CPS data sets, as these data sets do not include benthic detritivores.
(i) Operationally simple	0	• Scavenger biomass is at best partially tracked within CPS data sets, as these data sets do not include benthic detritivores.
(k) Broad spatial coverage	0	• Scavenger biomass is at best partially tracked within CPS data sets, as these data sets do not include benthic detritivores.
(l) Continuous time series	0	• Scavenger biomass is at best partially tracked within CPS data sets, as these data sets do not include benthic detritivores.
(m) Spatial & temporal variation understood	0	• Scavenger biomass is at best partially tracked within CPS data sets, as these data sets do not include benthic detritivores.
(n) High signal-to-noise ratio	0	• Scavenger biomass is at best partially tracked within CPS data sets, as these data sets do not include benthic detritivores.
(o) Understood by the public & policymakers	0.5	• Relative trends in biomass of each component of the community is easily understood, though pelagic scavengers may be a more challenging concept to communicate.
(p) History of reporting	0	• No
(q) Cost-effective	1	• Data mining by group is all that is needed
(r) Anticipatory or leading indicator	0	• No, scavengers usually respond to a change in the ecosystem that has occurred (Britton and Morton 1994, Ramsay et al. 1998), rather than anticipating one.
(t) Regionally/nationally/internationally compatible	0.5	• Changes in scavenger biomass are increasingly reported around the world (eg, due to fisheries discards, ghostfishing, competitive and predatory release).
TOTAL SCORE	5	

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CCIEA PHASE III REPORT 2013: ECOSYSTEM COMPONENTS, FISHERIES – COASTAL PELAGICS AND FORAGE

COASTAL PELAGIC AND FORAGE FISHES

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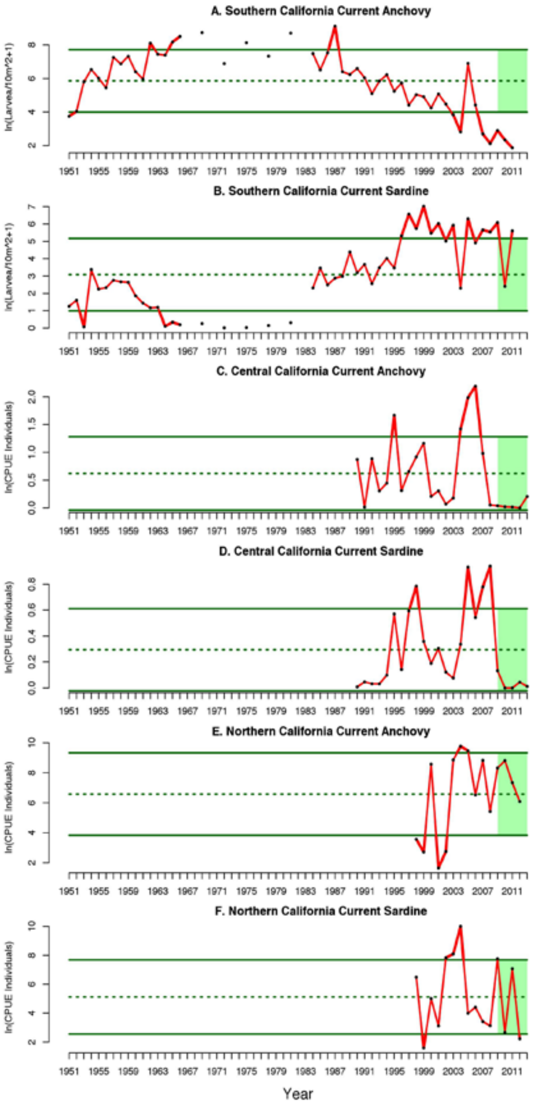
OVERVIEW

In the central and northern California Current regions, the forage community dependent on cool productive conditions became more abundant or remained stable. However, sardine abundance was low along much of the CCLME, but in the context of the longer time frame, abundance of sardines is about average. Anchovy in southern and central California remained at a low abundance. In southern California the low abundances observed follow a 30-year decline in abundance.

EXECUTIVE SUMMARY

Here, we examine trends in abundance and condition of coastal pelagic species and additional forage species collected from cruises along the California Current Large Marine Ecosystem (CCLME). Primarily we rely on the data collected from fishery independent surveys in southern California (1951-2011), central California (1990-2013), and Washington and Oregon (1998-2012). Given the differences in methods, catchability, and timing, these surveys are not directly comparable and for that reason are presented in separate figures. As well, it is important to recognize that these trends are not necessarily representative of abundance of the complete forage community. However, the trends are representative of the communities residing in the cruise areas during the timing of the cruises and a wealth of research has made the connections between the forage community dynamics in these regions and predator responses (e.g., Emmett et al. 2005, Daly et al. 2009, Phillips et al. 2009, Santora et al. 2011, Santora et al. 2012, Thompson et al. 2012b,



Abundance time series for anchovy and sardine from three regions of the California Current system. Absence of red line indicates years of no survey results , green area indicates the last five years of the data series, dashed green line indicates mean and solid green lines indicate +/- 1 s.d.

Wells et al. 2012, Koslow et al. 2013, Wells et al. 2013). We also use assessment reports of the Pacific Fisheries Management Council (1929-2013; Crone et al. 2011, Hill et al. 2013) to estimate trends in biomass and age structure of assessed coastal pelagic species.

There is substantial regional variability in the forage base dynamics in the California Current system. Generally, in the central and northern California Current regions, the forage community dependent on cool productive conditions became more abundant or remained stable. However, sardine abundance was low along much of the CCLME, but in the context of the longer time frame (multiple decades), abundance of sardines is about average. Anchovy in southern and central California remained at a low abundance. In southern California the low abundances observed follow a 30 decline in abundance (data currently available to 2011). The accompanying figure can be used to demonstrate these points.

DETAILED REPORT

The purpose of this chapter of the CCIEA is to examine trends in available indicators relevant to coastal pelagic species and additional forage fishes along the California Current and to qualitatively evaluate variability in the forage community relative to pressure. It is important to recognize that we refer to “status” here quite differently than the Pacific Fisheries Management Council (PFMC), and any difference between our status statements and those should not be considered a conflict. We are not using similar models nor benchmarks as those traditionally used. Our purpose is to set the framework for evaluating the forage community from an ecosystem perspective. This approach starts with a simple selection of indicators and evaluation of the trends. As well, we use these biological indicators in combination with indicators of environmental and anthropogenic pressures to evaluate potential risk to the forage community. Indicators for various pressures can be found in other chapters of the full CCIEA (e.g., Anthropogenic Drivers and Pressures, Oceanographic and Climatic Drivers and Pressures).

Coastal pelagic species (CPS) and forage species support important commercial fisheries as well as a number of higher trophic level species including those that are commercially exploited (e.g., rockfish, salmon) and/or legally protected (e.g., salmon, marine mammals, seabirds). In the context of this report, we consider species to be forage if they are often present in high abundance, feed on plankton for a portion of their life cycle and form dense schools or aggregations (e.g., anchovy, sardine, herring, mackerel, as well as invertebrate species such as squid and krill). Such species are often the principal means of transferring production from primary and secondary tropic levels (typically phytoplankton and zooplankton) to larger predatory fish, marine mammals and seabirds. Although the potential dynamics between the forage base and ecosystem integrity is not the primary aim of this section, we note that recent work Smith et al. (2011) and Kaplan et al. (2013) demonstrates the likely negative effects on the ecosystem caused by reductions in abundance of lower trophic level species.

Here, we define coastal pelagic species as recognized by the PFMC: northern anchovy, Pacific sardine, jack mackerel, Pacific mackerel, market squid, and krill. However, when data are available, we also include trends in other fishes that make up the forage complex including juvenile groundfish, herring, whitebait smelt, sanddabs, and selected mesopelagic assemblages. It is important to also recognize that these indices represent the temporal-spatial restrictions of the cruises. Therefore, we refrain from extending our cruise indicators to the full population dynamics, as they may not be well represented. However, there is a wealth of research connecting the data series we use here to the environmental drivers that determine the variability in the time series and resultant variability in upper predators reliant on the forage communities (e.g., Emmett et al. 2005, Daly et al. 2009, Phillips et al. 2009, Santora et al. 2011, Santora et al. 2012, Thompson et al. 2012b, Wells et al. 2012, Koslow et al. 2013, Wells et al. 2013).

INDICATOR SELECTION: SOUTHERN CALIFORNIA CURRENT, CALCOFI

INDICATOR EVALUATION: SOUTHERN CALIFORNIA CURRENT, CALCOFI

We considered a number of indicators to represent the coastal pelagic larval and forage assemblage in southern California. Our choice of indicators of trend was based on relative abundances, time series length and availability. As well, the literature indicates that unexploited oceanic assemblages are more sensitive to climatic effects than coastal and/ or exploited species (Hsieh and Ohman 2006). Data sources potentially included: 1. estimates of small pelagic fish biomass from acoustics (MacLennan and Simmonds 1992, Demer and Zwolinski 2012, Zwolinski et al. 2012), and 3. Daily Egg Production Method (DEPM) surveys for sardine (Lasker 1985, Lo et al. 1996). Although these series are valuable and both the acoustic surveys and the DEPM surveys produce biomass or spawning biomass estimates, results from these surveys are integrated in the sardine stock assessment (Crone et al 2011, Hill et al. 2011) , and we therefore do not use them individually in this report.

An additional data source, the California Cooperative Oceanic Fisheries Investigations (CalCOFI) provides the longest and most complete estimates of abundance of over 400 combined fish and cephalopod species (Table C1). Here we use CalCOFI ichthyoplankton data from 1951 to 2011 collected through oblique vertical plankton tows as described by Kramer et al (1972) and Smith and Richardson (1977).

Table C1: List of mesopelagic and coastal pelagic species from CalCOFI surveys used in this report. Subcategory lists mesopelagic species associated with warm or cool water conditions in the Southern California Bight. As well, "Trend" and "PCA" indicate species used in those analysis, respectively. All species were captured as larvae and enumerated in units of mean larvae/10m^2 captured in the CalCOFI core area within three month periods (i.e., quarters) and summed over all four quarters for a year.

Genus species	**Common name**	**Subcategory**	**Trend**	**PCA**
Bathylagus pacificus	slender blacksmelt	cool-water	X	
Bathylagus wesethi	snubnose blacksmelt	warm-water	X	X
Ceratoscopelus townsend	fangtooth lanternfish	warm-water	X	X
Citharichthys sordidus	Pacific sanddab		X	X
Citharichthys stigmaeus	speckled sanddab			X
Cyclothone signata	showy bristlemouth			X
Diaphus theta	California headlight fish			X
Diogenichthys atlanticus	longfin lanternfish	warm-water	X	X
Diogenichthys laternatus	diogenes laternfish	warm-water	X	
Engraulis mordax	northern anchovy		X	X
Idiacanthus antrostomus	Pacific black dragon			X
Leuroglossus stilbius	California	cool-	X	X

	smoothtongue	water		
Lipolagus ochotensis	eared blacksmelt	cool-water	X	X
Merluccius productus	hake		X	X
Nannobrachium ritteri	broadfin lampfish			X
Protomyctophum crockeri	California flashlightfish	cool-water	X	X
Sardinops sagax	Pacific sardine		X	X
Sebastes jordani	shortbelly rockfish		X	X
Stenobrachius leucopsarus	northern lampfish	cool-water	X	X
Symbolophorus californiensis	bigfin laternfish	warm-water	X	X
Tarletonbeania crenularis	blue laternfish	cool-water	X	X
Trachurus symmetricus	Pacific jack mackerel			X
Triphoturus mexicanus	Mexican lampfish	warm-water	X	X
Vinciguerria spp.	lightfishes	warm-water	X	X

We have restricted our analysis to the most abundant and potentially influential CPS and forage species for which we have data. To provide an integrated measure of large-scale responses to environmental variability, we aggregated the mesopelagic fishes into cool- and warm-water groups following Hsieh et al. (2005). These groups are likely to reflect general trends in the ecosystem better than time series for individual species, some of which are relatively data poor. The species and groups analyzed were Pacific sardine, northern anchovy, hake, jack mackerel, Pacific sanddab, shortbelly rockfish, cool-water mesopelagics, and warm-water mesopelagics.

To evaluate community variability we performed principal component analysis of the Hellinger-transformed ichthyoplankton assemblage (#indiv/10 m^2) collected from core

CalCOFI stations during spring and summer cruises (Thompson et al. 2012a). This analysis included all the species in Table C1. Such an indicator allows fo the interpretation of similarity and dissimilarity between years over which the survey occurred. Principle component analysis was performed using the years 1993-2011. During this time a greater number of species was enumerated. Those that significantly contribute to the PCA were included in an examination of community variability.

Summary of indicators: Southern California Current, CalCOFI

1. All data are from the core CalCOFI sampling area (lines 76.7-93.3, stations 28.0 – 120.0; Figure C1) for years when the core area was sampled during each quarter of the year. Mean larval abundances (larvae/10 m^2) were estimated for each 3.3-line by 10-station cell in the core area for each quarter, and then cells were summed over the year. Means across the entire time series were then calculated using the delta-lognormal distribution (Pennington 1983). This procedure standardized the data given unequal sampling effort during some cruises, many zero catches, and seasonal but variable patterns of spawning for the fishes analyzed.
2. Trends of individual species analyzed were Pacific sardine, northern anchovy, Pacific hake, jack mackerel, Pacific sand dab, and shortbelly rockfish.
3. The cold- and warm-water associated mesopelagic species were summed for each net tow and then analyzed as groups following the same method described above for individual species.
4. Principal component analysis of the Hellinger-transformed ichthyoplankton assemblage (#indiv/10 m^2) collected from core CalCOFI stations during spring and summer cruises.

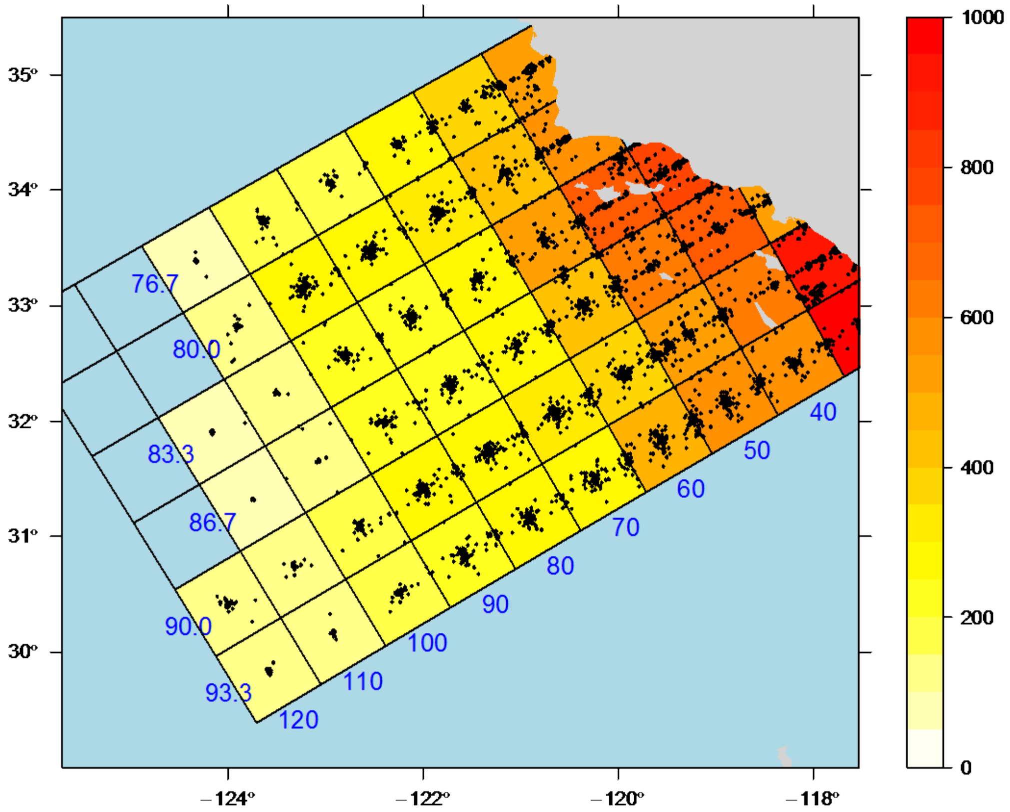
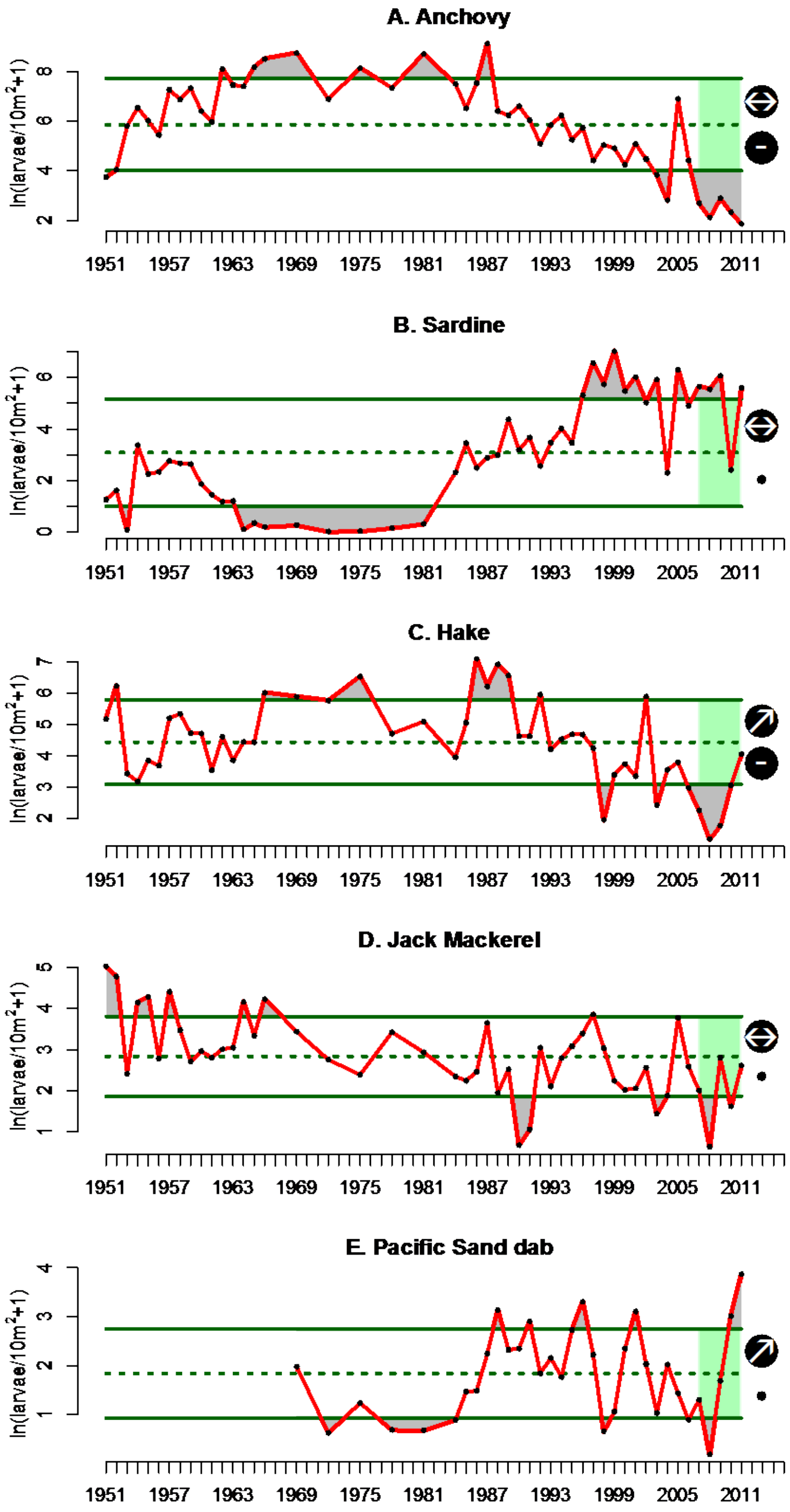


Figure C1. *CalCOFI Sampling Pattern for Oblique Net Tows.* Grid pattern of 3.3-line by 10-station cells in the core CalCOFI sampling area (lines 76.7-93.3) used for analysis of Southern California forage. Color key indicates actual number of samples collected within each cell for the period 1951-2011. Black dots indicate actual sample locations.


A. Anchovy
ln(larvae/10m²+1)
2
4
6
8
B. Sardine
ln(larvae/10m²+1)
0
2
4
6
C. Hake
ln(larvae/10m²+1)
2
3
4
5
6
7
D. Jack Mackerel
ln(larvae/10m²+1)
1
2
3
4
5
E. Pacific Sand dab
ln(larvae/10m²+1)
1
2
3
4
1951
1957
1963
1969
1975
1981
1987
1993
1999
2005
2011

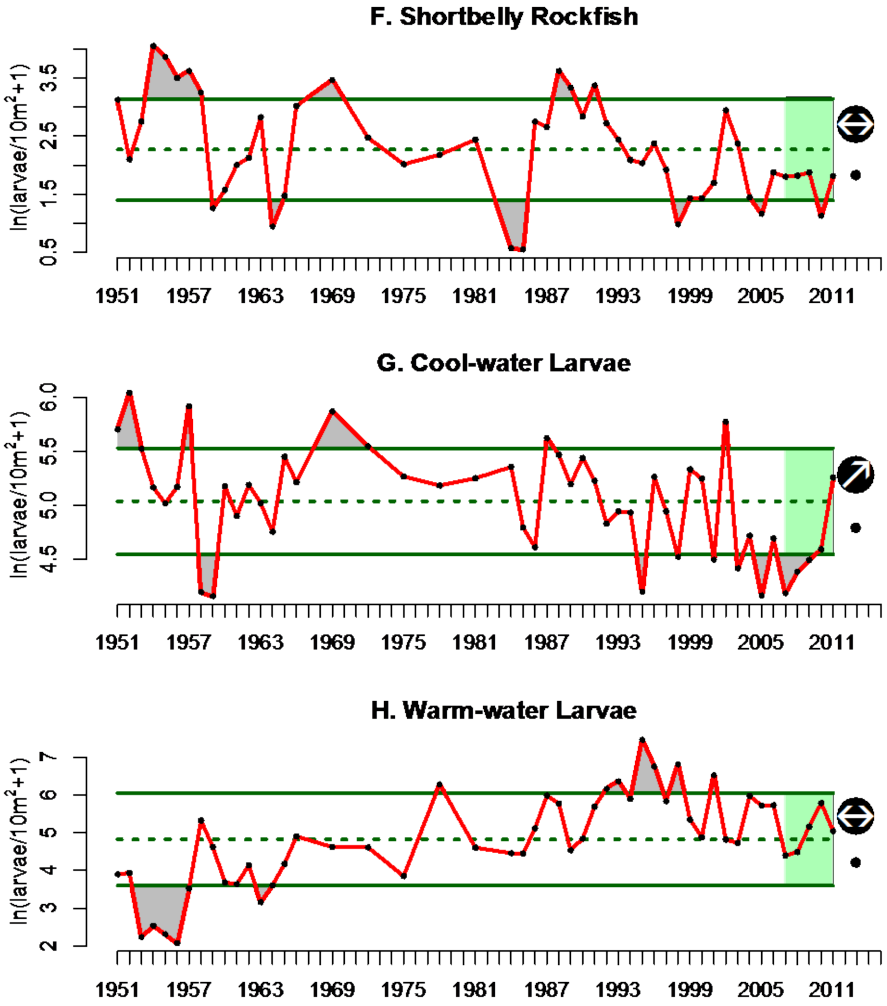


Figure C2. *Southern California Forage, CalCOFI.* Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d. , or was within one 1.0 s.d of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-)or within (·) one s.d. of the long-term mean.

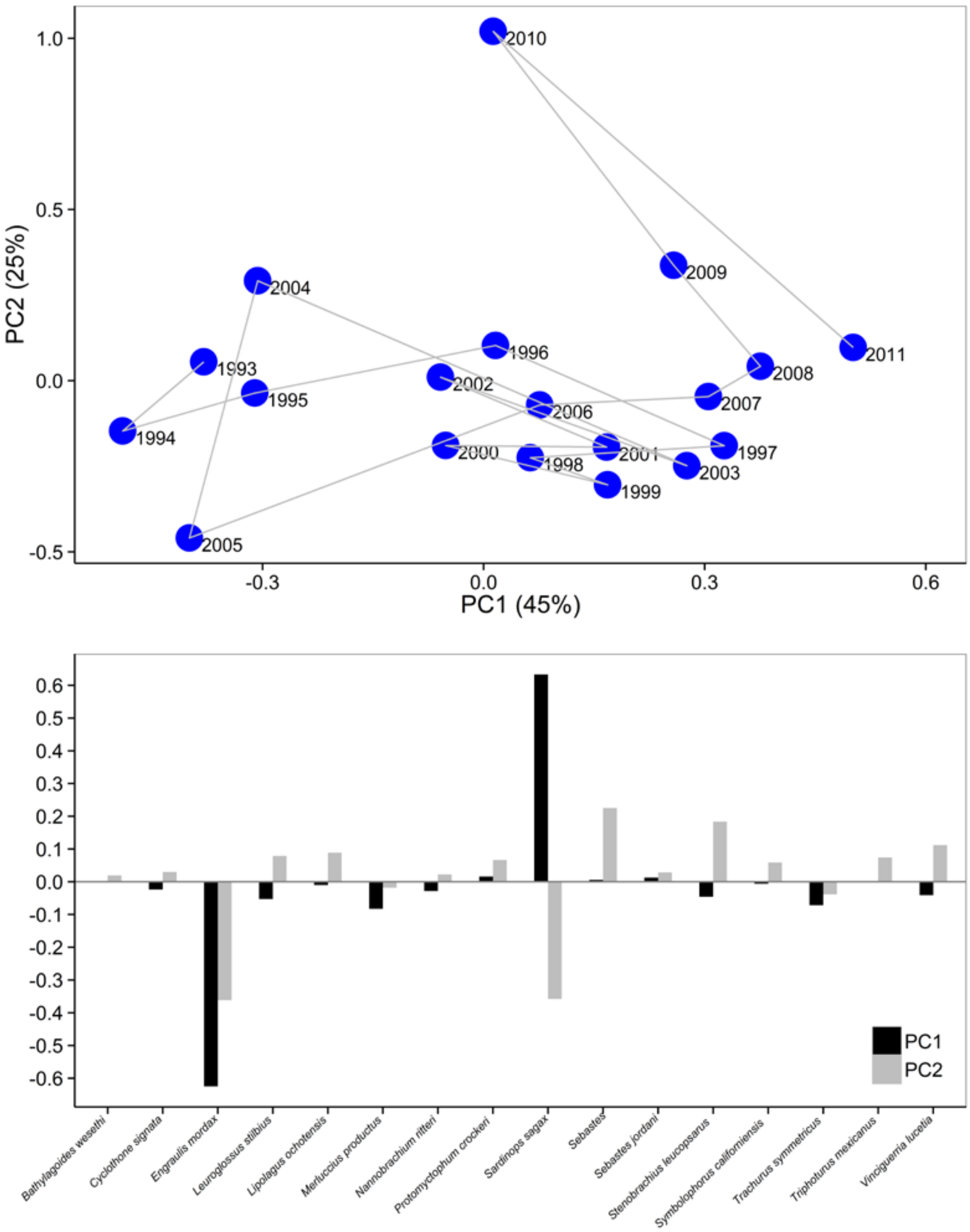


Figure C3. *Spring southern California Ichthyoplankton from oblique bongo net tows.* The top plot illustrates the loadings of years on principle components (PC) 1 and 2. The bottom plot shows the most abundant species loaded onto components 1 and 2.

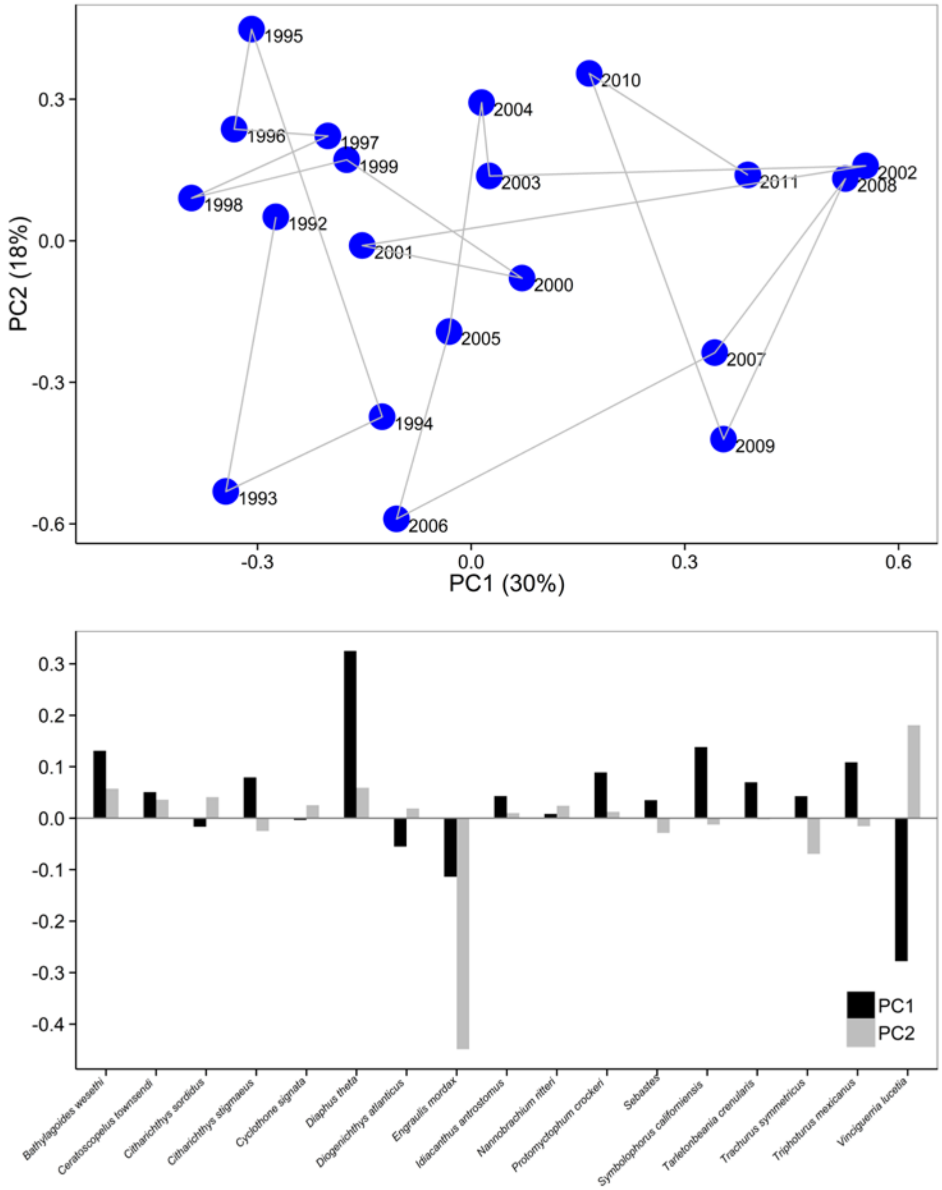


Figure C4. *Summer southern California ichthyoplankton from oblique bongo net tows.* The top plot illustrates the loadings of years on principle components (PC) 1 and 2. The bottom plot shows the most abundant species loaded onto components 1 and 2.

STATUS AND TRENDS: SOUTHERN CALIFORNIA CURRENT, CALCOFI

MAJOR FINDINGS: SOUTHERN CALIFORNIA CURRENT, CALCOFI

Since 1951 the 6 species indicators and 2 species group indicators have shown high variability and limited covariation (Figure C2). Larval anchovy abundance continued a declining trend over the last thirty years to the lowest abundance since 1951 (last data available is 2011). Sardine has been above the long-term average since 1996, minus 2004 and 2010 during which the abundance was average. Fish larvae dependent on cool productive conditions demonstrated average to above average abundance (e.g., hake, sanddabs, rockfish, and the consolidate cool-water species group).

SUMMARY AND STATUS OF TRENDS: SOUTHERN CALIFORNIA CURRENT, CALCOFI

We report both long-term means and recent trends in this status review. Under the current framework, an indicator is considered to have changed in the short-term if there are significant increasing or decreasing trends over the last five years. An indicator is considered to be above or below long-term norms if the mean of the last five years of the time series differs from the mean of the full time series by more than 1.0 standard deviation.

Anchovy, hake, and cool-water mesopelagics have generally decreased over the last 30 years (Figure C2). Anchovy abundance ending in 2011 was the lowest abundance recorded. Sardine abundance was above its long-term average. Sardine larvae show a different trend to anchovy, and it has been postulated (Chavez et al. 2003) that abundance peaks of these species alternate at decadal time scales (although the CalCOFI time series is too short to evaluate this hypothesis (McClatchie 2012). Sardine larvae in the 1980s and 90s have increased from the collapse of the stock in the 1950s. Although there has been a minor decline in sardine larval abundance since 2000, sardine larval abundance has remained above the mean of the last 60 years (Fig. C2).

Hake and cool-water mesopelagic larvae have increased substantially in the last five years and are near their long-term average values (Fig. C2). In addition to hake and cool-water mesopelagics, sanddab larvae are also increasing in the last five years. The remaining species examined (jack mackerel, rockfish, and warm-water mesopelagics) remained stable near average values (Fig. C2).

Principal component analysis of the Hellinger-transformed ichthyoplankton assemblage (#indiv/10 m^2) collected from core CalCOFI stations during spring cruises demonstrated PC1 and PC2 capture approximately 70% of the total variance (Figure C3).

PC1 mainly characterizes a gradient between assemblages dominated by anchovy (negative loadings) and sardine (positive loading). Years with high loadings on PC2 are characterized by greater influence of rockfish and northern lampfish relative to other taxa. Anchovy had the greatest influence in the years 1993-1995 and 2004-2005 while sardines were the main taxa in 1997, 2003, 2007-2009, and 2011. In 2010 both anchovy and sardine larvae were scarce and the most important taxa were rockfish and northern lampfish.

Principal component analysis of the summer CalCOFI ichthyoplankton assemblage indicated PC1 and PC2 explain approximately 48% of the total variance (Figure C4). PC1 mainly characterizes a gradient between assemblages dominated by Panama lightfish (negative loadings) and California headlightfish (positive loading). Both of these utlize mesopelagic habitats but Panama lightfish have a more southern biogeographic range than California headlight fish. Years with low loadings on PC2 are characterized by greater influence of anchovy. California headlight fish tended to be most important in the recent years of 2002, 2007,2008, 2010 and 2011 while Panama lightfish were most influential in the previous decade (1992-1999). Anchovy exhibited greater influence in the summers of 1993 and1994 as well as 2006, 2007, and 2009.

INDICATOR SELECTION: CENTRAL CALIFORNIA CURRENT, MIDWATER TRAWL SURVEY

INDICATOR EVALUATION: CENTRAL CALIFORNIA CURRENT, MIDWATER TRAWL SURVEY

General description: Central California, midwater trawl survey

We evaluated a number of indicators to represent the abundance of young-of-the-year (YOY) groundfish, coastal pelagic species and other micronekton in the coastal and offshore waters of central California. Data are based on mid-water trawl collections, as described in Ralston et al. (2013b). Available data for coastal pelagic forage species include krill (Euphausiids*)*, market squid, anchovy, and sardine. In addition, numerous other members of the forage community are available including pelagic young-of-the-year (YOY) rockfish, pelagic YOY sanddabs, YOY hake, octopus, seregestid shrimp and numerous mesopelagic species (Santora et al. 2012). Although the time series of the YOY groundfish extends back to 1983 (Ralston et al. 2013b), many of the other micronekton species have only been reliably quantified since 1990. For analysis of trends we focus here on the most abundant, continuously present, and available species: anchovy, sardine, market squid, krill, YOY rockfishes, YOY sand dabs, and YOY hake (Table C2). Four of these, anchovy, sardine, market squid and krill represent the CPS. Importantly, the abundance of anchovy and sardine from this survey in central California is not likely to correlate to overall population strengths as much as it represents variability in the distribution throughout the CCE latitudinally and longitudinally (Bjorkstedt et al. 2012, Song et al. 2012). As a consequence, unlike the overall trend in CCE, anchovy and sardine are positively correlated for the majority of the time series. We did not include juvenile salmon because the net is inefficient at collecting salmon.

For our multivariate analysis we use a broader suite of fishes including those more rarely captured but necessary for a evaluation of the community variability. The multivariate approach allows for a comparison of community structure variability and an evaluation of the similarity between like years.

Samples were collected during the May-June period, the peak of the abundance of pelagic YOY rockfish that are the focus of the survey, and are limited spatially to the region between southern Monterey Bay (approximately 36 N) and just north of Point Reyes, CA (~ 38 N). Since the early 2000s both this and a comparable survey have operated from the U.S./Mexico border to Cape Flattery, WA (see Ralston and Stewart 2013), however only rockfish data from the expanded range have been rigorously analyzed and other forage species from the broader range will be discussed in future reports. Samples are collected using a modified Cobb midwater trawl, with a head rope depth of 30 m (the average depth of the thermocline in the region) at a speed of ~2 knots for 15 minutes at depth, with the exception of stations that were too shallow (< ~60m) such as those in the Gulf of the Farallones for which the headrope depth was 10 m (Sakuma et al. 2006). In all cases, samples represent catch per standard 15 minute trawl (CPUE). The data was log-transformed data because it was log-normally distributed.

Table C2. Species collected and enumerated (geometric mean of catch per unit effort) in the mid-water trawl survey along Central California.

Genus species	**Common name**	**Stage**	**Trend**	**PCA**
Bathylagus pacificus	blacksmelt	juvenile		X
Citharichthys sordidus	Pacific sanddab	juvenile	X	X
Citharichthys stigmaeus	speckled sanddab	juvenile		X
Engraulis mordax	northern anchovy	adult	X	X
Euphausiids	krill	adult	X	X
Glyptocephalus zachirus	rex sole	juvenile		X
Leuroglossus stilbius	smoothtongue	juvenile		X
Loligo opalescens	market squid	juvenile, adult	X	X
Merluccius productus	Pacific hake	juvenile	X	X
Myctophum punctatum	blue lantern fish	juvenile		X
Octopus spp.	octopus	juvenile		X
Sardinops sagax	Pacific sardine	adult	X	X
Sebastes spp.	rockfishes	juvenile	X	X
Sergestidae	sergestids	juvenile	X	X
Stenobrachius leucopsarus	northern lampfish	juvenile		X

Appropriate indicators: Central California Current, midwater trawl survey

We examined trends in anchovy and Pacific sardine. Along the CCE northern anchovy abundance variability tends to be positively related to warmer, less productive conditions. In central California, temporal dynamics of northern anchovy abundance likely reflect a change in the distribution relative to CCE as a whole (for the May-June period in which the survey is conducted) rather than overall changes in the stock. However, while the anchovy abundance variability is poorly correlated to ocean temperatures in central California, during times of low productivity across the CCE northern anchovy make up a greater proportion of the diets of seabirds locally and, therefore, their relative abundance in the forage community can indicate overall productivity conditions. In central California, the relative abundance of Pacific sardine in late spring likely represents a change in the average distribution that relates to ocean conditions.

As well, we examined trends in a number of additional fishes that during a period of their life cycle are important contributors to the forage community, including: juvenile Pacific hake, juvenile rockfish, and juvenile Pacific sanddabs. Currently the factors that drive variability in Pacific hake abundance in this survey are not entirely clear, as hake typically spawn in southern California and northern Mexico waters, high numbers may represent a strong year class or a shift in the distribution of young-of-the-year. Juvenile and sub-adult hake are an important prey for many other higher trophic level predators. Juvenile rockfish captured in this data series represent juveniles spawned in the current winter (e.g., young-of-the-year individuals). While pelagic, they represent a critical prey resource for predators such as Common murre, rhinoceros auklets and Chinook salmon, and there is a significant relationship between juvenile rockfish abundance and breeding success of many central California seabirds (Wells et al. 2008a, Field et al. 2010). Pacific sanddabs, when juveniles, are pelagic and represent a moderately important prey resource for many seabirds and other predators in the region.

Krill is a reasonable indicator of local environmental quality. Krill abundance is known to increase during productive conditions with optimal winds (Cury and Roy 1989). Central California represents a region with several well known krill hot spots (Santora et al. 2011) where seabirds, mammals, salmon (adult and juveniles), rockfishes and a number of other species feed on krill. Wells et al. (2008b) and Wells et al (2012) demonstrate the critical role of krill on seabirds, rockfish and salmon. Here, we do not separate the two dominant species of krill in central California (*Euphausia pacifica* and *Thysanoessa spinifera*) because they were not identified to the species level until 2002. However, the two species generally occupy different habitats (inner-shelf vs outer-shelf, (Santora et al. 2012))and have different life-histories.

Monterey Bay is a spawning ground for market squid, and this species forms one of the largest and most lucrative California fisheries (although the greatest landings are typically in southern California). Both juvenile and adult squid make up a significant proportion of the diets of many predators. High market squid abundance is generally positively associated with cool, productive conditions. Data series are log-normally distributed so in these analyses we log-transformed the data.

A multivariate indicator can be used to indicate the overall forage assemblage characteristics and similarity to previous years. Beyond the seven species examined specifically, we included a suite of species commonly collected including rockfish, speckled sanddab, Pacific sanddab, market squid, Pacific hake, octopus, krill, rex sole, Seregestids, smooth tongue, blacksmelt, blue lantern, anchovy, and sardine. These species were included in a principle component analysis and the results demonstrate patterns in forage complex over the years.

STATUS AND TRENDS: CENTRAL CALIFORNIA CURRENT, MIDWATER TRAWL SURVEY

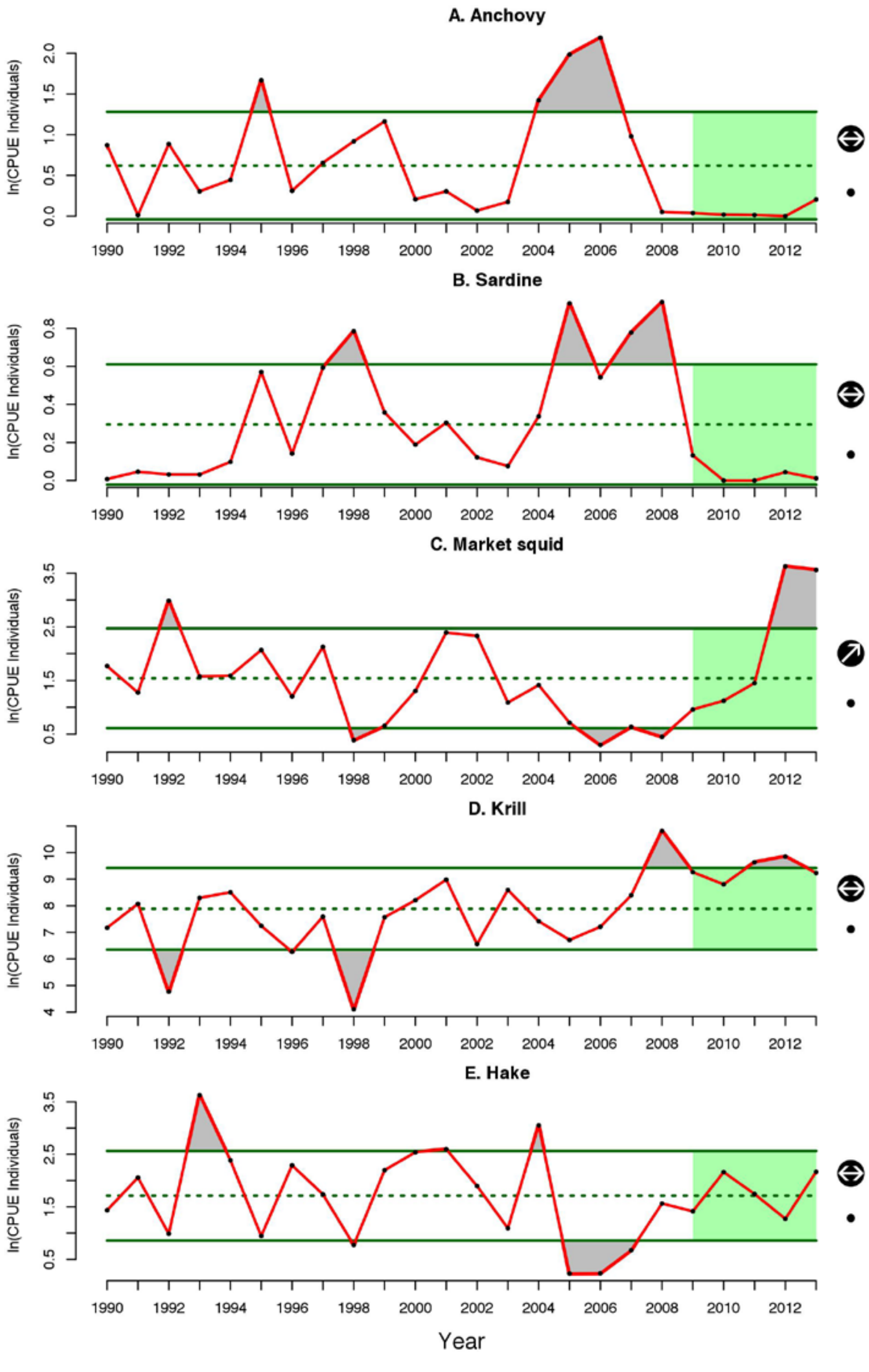
MAJOR FINDINGS: CENTRAL CALIFORNIA CURRENT, MIDWATER TRAWL SURVEY

Overall, these data series suggest that recent years have been conducive to more production and improved forage abundance for onshelf species, in agreement with Bjorkstedt et al. (2012). However, while fish reliant on productive onshelf conditions experienced increased production, those further offshore were at record low values (Wells et al. 2013).

SUMMARY AND STATUS OF TRENDS: CENTRAL CALIFORNIA CURRENT, MID-WATER TRAWL SURVEY

Anchovy and sardine continued a period of low abundance with no indication of a declining nor increasing trend over the last 5 years (Figure C5). However, rockfish, sanddabs, and market squid have recovered from record low levels observed in 2005 and 2006, and were all at record high levels leading to a positive trend in the last five years. These 3 groups are favored under high transport (advection) conditions that are associated with cool and productive conditions on the shelf (Wells et al. 2013). Likewise, krill and hake abundances have trended upward since 2005. In fact, krill achieved record levels in 2008 and has maintained relatively high abundance since (Figure C5). Finally, market squid is presently experiencing the greatest of the three boom periods of the last 20 years leading to a significantly positive five-year tend (Figure C5). This trend is consistent with consistently high landings of market squid throughout California waters in recent years.

The trends observed in the 7 indicators shown in Figure C3 are consistent with trends across a broader suite of taxa within this region, with the first and second components (of a principle components analysis of 15 of the dominant taxon) explaining approximately 36% and 16% of the variance in the data respectively. Loadings of these groups indicate strong covariance among young-of-the-year groundfish (rockfish, sanddabs and Pacific hake), cephalopods and euphausiids, which in turn tend to be negatively correlated over time with coastal pelagic and mesopelagic species. Specifically, in the central California region, those species loading positive high on the first component represent productive onshelf conditions. Principle component 2 is less obvious in its interpretation. 2012 and 2013 continued to indicate a pelagic micronekton community structure conditions similar to those seen in the early 1990s and early 2000s (Figure C6).


A. Anchovy
B. Sardine
C. Market squid
D. Krill
E. Hake
ln(CPUE Individuals)
0.0
0.5
1.0
1.5
2.0
0.2
0.4
0.6
0.8
2.5
3.5
4
5
6
7
8
9
10
1990
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2006
2008
2010
2012
Year

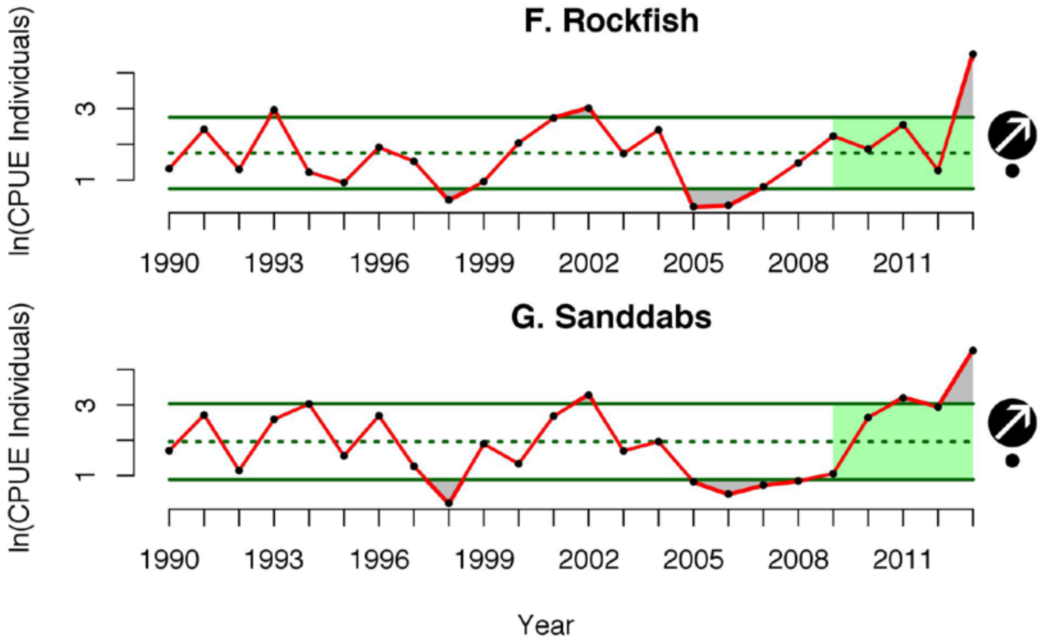

Figure C5. *Central California Forage, mid-water trawl.* Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d., or was within one 1.0 s.d of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-)or within (·) one s.d. of the long-term mean.

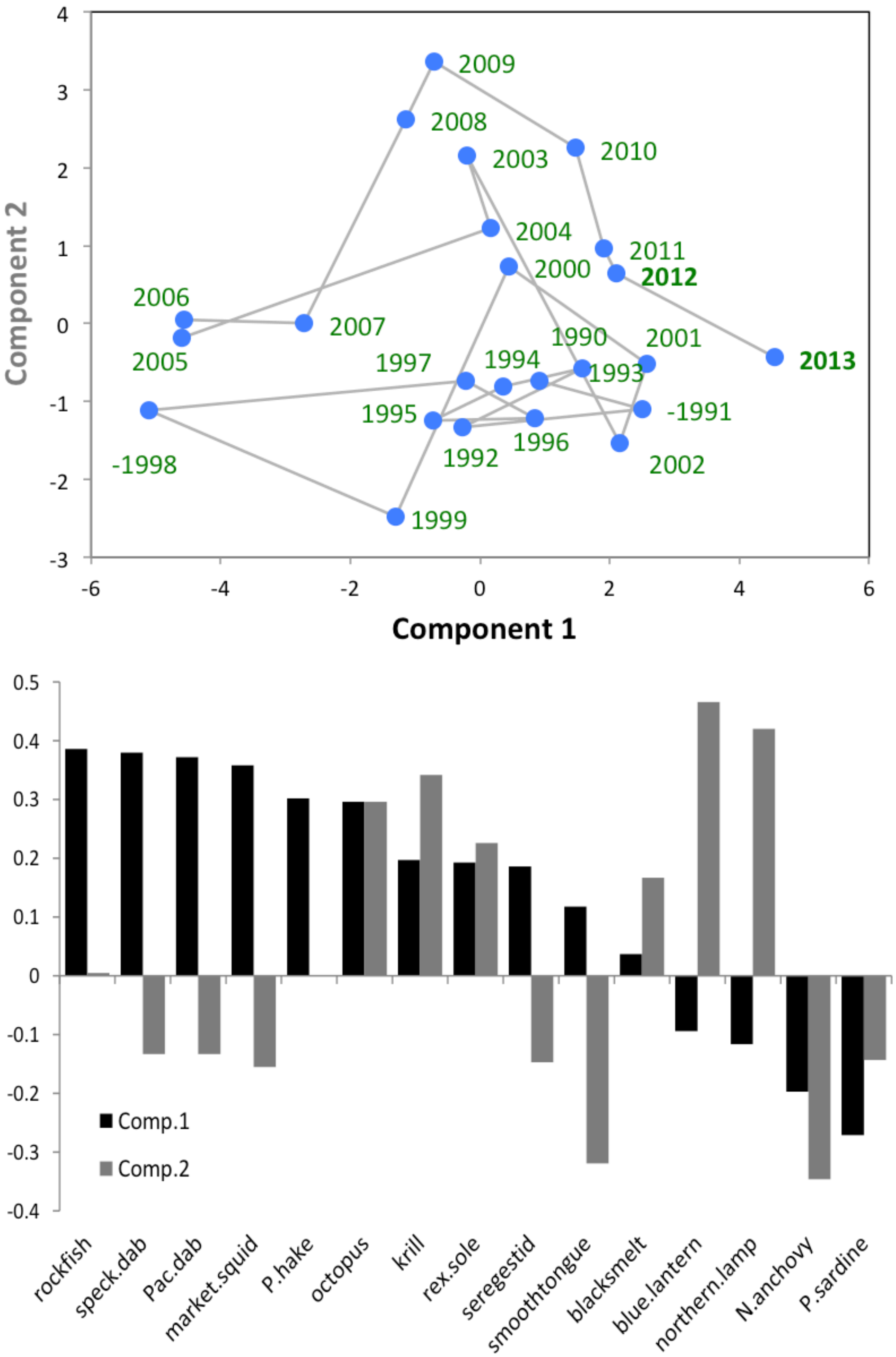

Figure C6. *Central California Forage, mid-water trawl.* The top plot demonstrates the similarity between forage communities between years. The bottom plot shows how the individual species loaded onto these components. Note, 2013 is characterized by very good production of onshelf forage fishes (as indicated by positive loadings on component 1).

INDICATOR EVALUATION: NORTHERN CALIFORNIA CURRENT

General description: Northern California Current

Pelagic nekton catch data were collected by the NWFSC-NOAA Bonneville Power Administration survey surface trawls on standard transects and stations between Tatoosh Island, WA and Cape Perpetua, OR in June from 1998 to 2012. All tows were made during the day at predetermined locations along transects extending off the coast to the shelf break (Brodeur et al. 2005). Numbers of individuals were recorded for each species caught in each haul and were standardized by the horizontal distance sampled by the towed net as CPUE (#/km^2 towed). Yearly abundance data were obtained by combining (summing) the standardized count data of each species captured during June for each year.

Table C3. Species collected in the surface trawl of the Northern California Current survey (#indiv/km2) (using a $\log_{10}(x+1)$ transformation).

Genus species	Common name	Stage	Trend	PCA
Allosmerus elongatus	whitebait smelt	juvenile, adult	X	X
Anarrhichthys ocellatus	wolf eel			X
Anoplopoma fimbria	sablefish	juvenile		X
Clupea pallasii	Pacific herring	juvenile, adult	X	X
Cololabis saira	Pacific saury	juvenile, adult		X
Engraulis mordax	northern anchovy	juvenile, adult	X	X
Galeorhinus galeus	soupfin shark			X
Oncorhyncus keta	chum salmon	juvenile		X
Oncorhyncus kisutch	coho salmon	juvenile		X
Oncorhynchus nerka	sockeye salmon	juvenile		X

Oncorhyncus tshawytscha	Chinook salmon	juvenile		X
Osmeridae	smelts	juvenile, adult		X
Sardinops sagax	Pacific sardine	juvenile, adult	X	X
Scomber japonicus	Pacific mackerel	juvenile, adult		X
Trachurus symmetricus	Jack mackerel	juvenile, adult	X	X

Appropriate indicators: Northern California Current

Time series plots of standardized yearly abundance data are presented for each of the five most dominant and consistently collected forage species measured (jack mackerel, Pacific sardine, northern anchovy, Pacific herring and whitebait smelt; Table C4). Although other forage species are caught in these surveys (see multivariate analysis of community), these five species represent the bulk of the forage fish catch in surface waters during the day. They include migratory species (sardines and some anchovies) that may spawn off the Pacific Northwest or migrate from California (Emmett et al. 2005, Litz et al. 2008). Jack mackerel can be a forage fish at younger ages but off Oregon and Washington are too large to be fed upon by a number of predators such as seabirds or adult rockfishes. They spawn off southern California and arrive during summer to feed off Oregon and Washington. Herring and whitebait smelt are likely spawned locally. A number of these species may have seasonal trends in abundance (Emmett et al. 2005) so may have different trends than taken twice a year but over a broader geographical area. Because the data are log-normally distributed they were log-transformed for this analysis.

We also characterized the variability in the community makeup. A PCA allows for a comparison between years. We examined the 15 most dominant taxa sampled in the plume survey (log10(x+1) transformation).

MAJOR FINDINGS: NORTHERN CALIFORNIA CURRENT

The environment has fluctuated during the period since 1998 between relatively cool years (2008, 2011, 2012) to warm years (2010) (Bjorkstedt et al. 2012), likely leading to great variability in jack mackerel, Pacific herring, and sardine. Notably herring and jack mackerel catch per unit effort in June were exceptionally low in 2012.

SUMMARY AND STATUS OF TRENDS: NORTHERN CALIFORNIA CURRENT

Jack mackerel has generally decreased over the last 5 years and is currently at its record low CPUE (Figure C7). Herring shows a declining trend over the last six years and experienced a decline to low abundance in 2012 (Figure C7). The whitebait smelt population appears to be generally stable through time with an increase in recent years following some years (1999, 2000, 2002, 2006, 2008) of below average values (Figure C7). Anchovy has been relatively stable since its highest abundance in 2004, and recent values remain greater than the low values demonstrated in the late 1990s and early 2000's (Figure C7). Sardine have had fluctuating abundances in recent years after the high abundances observed in 2003 and 2004. In the last 5 years sardine abundance has been highly variable with two of the three lowest years in the full series occurring in 2010, and 2012, only slightly higher than the lowest level in 1999.

Principal component analysis of 15 of the dominant taxa abundances (#indiv/km2) (using a $\log_{10}(x+1)$ transformation) quantitatively sampled in the BPA plume survey between 1998 and 2012 explained a total of 44.8% of the variability (PC 1 and 2 explained 28% and 16.8%, respectively, Figure C8). Years with similar community structures include 2001, 2011, and 2012, whereas years 1998, 2005 and 2008 have very distinct communities from other years. Salmonid species show strong loadings on PC 1 and 2 and smelts (Osmeridae spp., and whitebait smelt) are negatively loaded onto PC 1. Forage species (sardine, anchovy, saury and sablefish) are positively loaded onto PC1 and 2. In general, smelts are negatively correlated with pelagic forage species over time on both components.

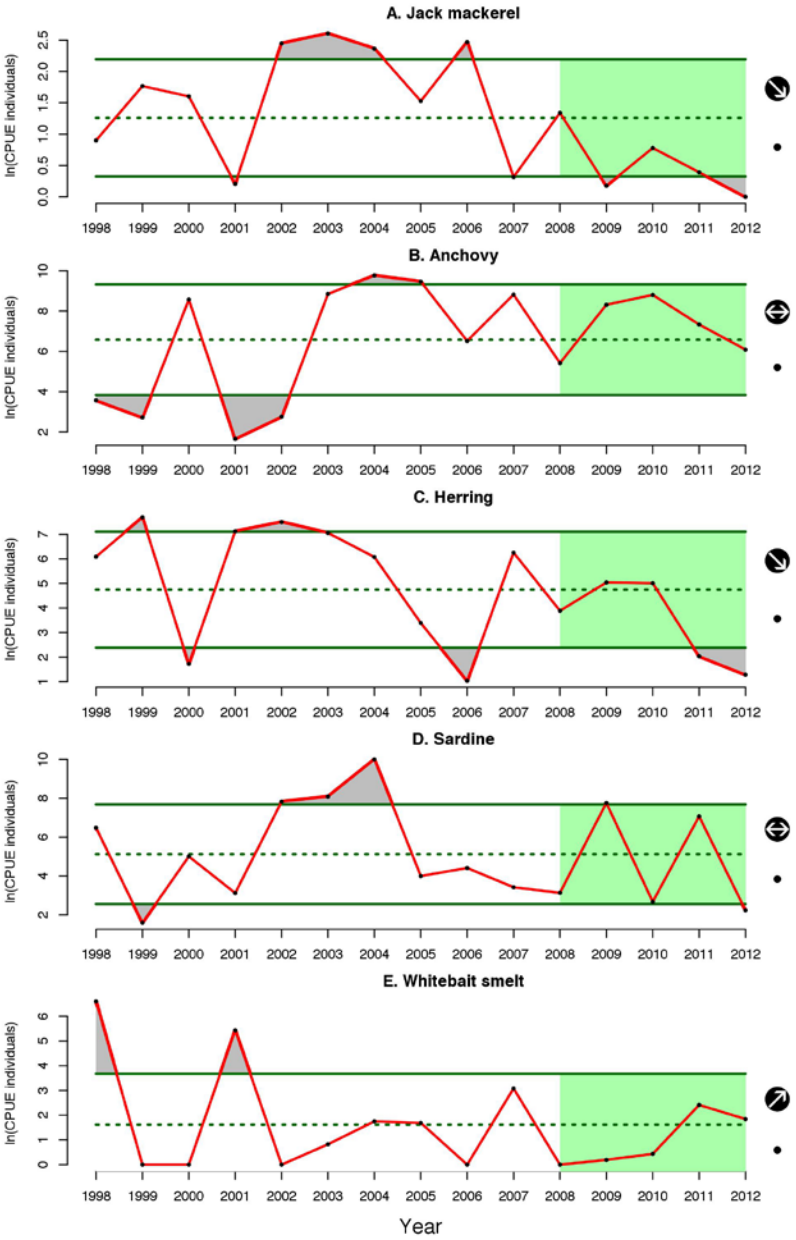

Figure C7. *Northern California Forage (NWFSC/BPA)*. Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d., or was within one 1.0 s.d of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-) or within (.) one s.d. of the long-term mean

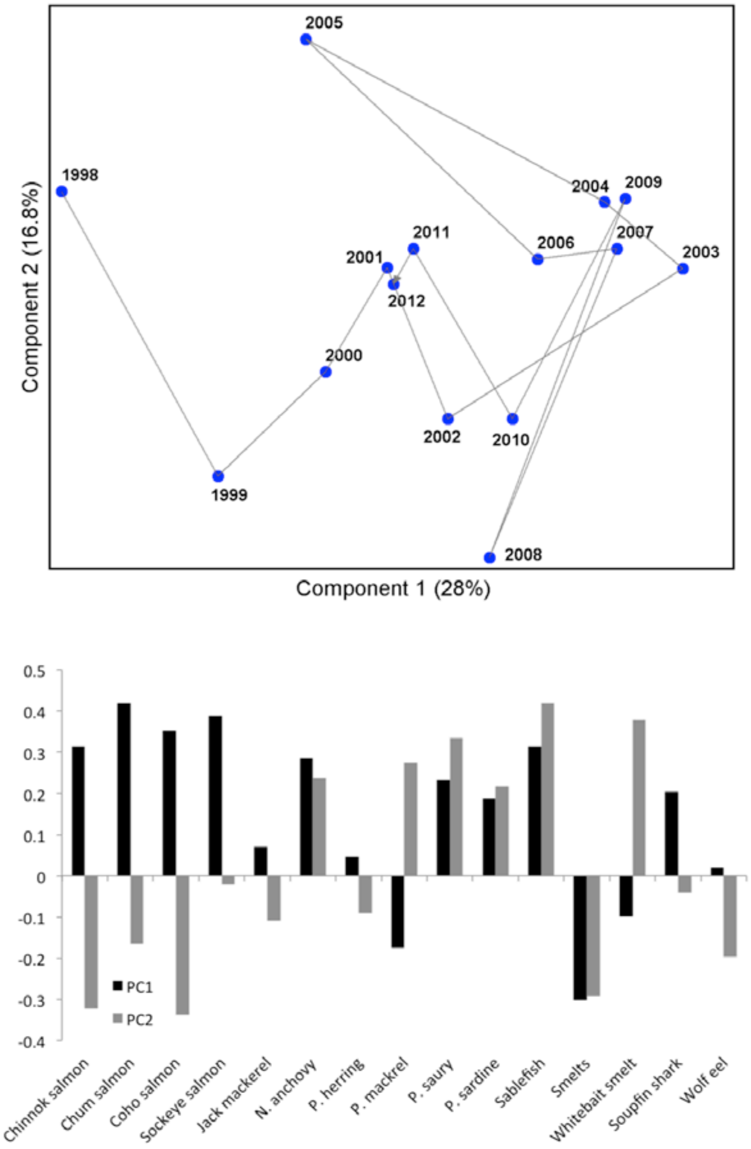

Figure C8. *Northern California Pelagic Community, surface trawl.* The top plot illustrates the similarity between years in terms of their pelagic communities. The bottom plot shows the most abundant species in June loaded onto components 1 and 2. Notably, 2001, 2011 and 2012 have very similar communities, and years 1998, 2005 and 2008 are distinct in their community composition from other years of the survey

INDICATOR SELECTION: ASSESSMENTS

INDICATOR EVALUATION: ASSESSMENTS

General description: Assessments

Pacific mackerel and sardine assessments are prepared for the PFMC to be used for developing harvest specifications. The assessments incorporate data from a number of sources and determine the biomass and age distribution of the populations along the coast. They represent the most complete analysis of recent abundance trends across the CCE. Therefore, we use recent assessments (Crone 2013, Hill et al. 2014), along with previously reviewed assessments (Murphy 1966, MacCall 1979, Dorval et al. 2008, Crone et al. 2009, Hill et al. 2010, Crone et al. 2011), to guide our estimation of long-term population trends of abundance and current stock status for these two species.

The Pacific mackerel assessment is an age-structured model incorporating information on catch, length and age distributions, and recreational fishery surveys (Crone et al. 2011). Full model details, problems and uncertainties are disclosed at http://www.pcouncil.org/coastal-pelagic-species/stock-assessment-and-fishery-evaluation-safe-documents/ and in Crone et al. (2011).

The sardine assessment includes fishery and survey data (egg production, and acoustic estimates of biomass). Full model details, problems, uncertainties are disclosed at http://www.pcouncil.org/coastal-pelagic-species/stock-assessment-and-fishery-evaluation-safe-documents/ and in Hill et al. (2014).

Appropriate indicators: Assessments

We focus on three indicators representing abundance and condition of Pacific mackerel and sardine. To estimate abundance trends we evaluate the biomasses of the two species.

1. The biomass time series for Pacific sardine is compiled from the most recent stock assessment (Hill et al. 2014) appended with previous assessments covering earlier periods of time (Murphy 1966, MacCall 1979, Hill et al. 2010). Biomass units are for sardine ages 2 and older, log-transformed metric tons.

2. Biomass time series for Pacific mackerel is the most recent stock assessment (Crone 2013), appended with previously reviewed assessments covering the historic period (Dorval et al. 2008, Crone et al. 2009). Biomass units are for mackerel ages 1 and older, log-transformed metric tons.

MAJOR FINDINGS BASED ON INDICATORS

In recent years the biomasses of Pacific mackerel and sardine have been average relative to the long-term mean yet, for sardine, the recent values are greater than the period following the population crash between 1950 and the early 1990s.

SUMMARY AND STATUS OF TRENDS BASED ON INDICATORS

In the first half of the 20th century both Pacific mackerel and sardine were relatively abundant. In the late 1970s and 1980s Pacific mackerel demonstrated above average production but production has declined in the past two decades. In the last five years population estimates of biomass are within 1 s.d. of the long-term mean and there is no apparent trend (Figure C9). Similarly, sardine experienced near-above average production in the past 10-20 years yet the estimates of biomass are with 1 s.d. of the long-term mean suggesting that, while the abundance in greater in recent years, it is still only a portion of that observed in the earlier part of the 20th century (Figure C9).

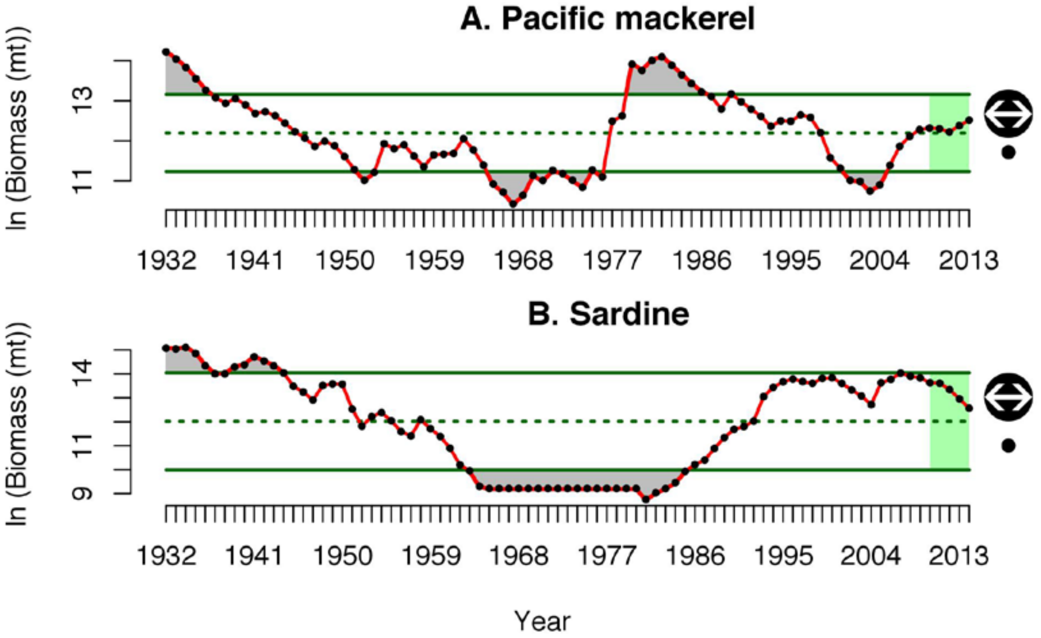

Figure C9*. Assessment biomass.* Most time series are plotted in a standard format. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 5-years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend over the last 5-years increased, or decreased by more than 1.0 s.d., or was within one 1.0 s.d of the long-term trend. The low symbol indicates whether the mean of the last 5 years was greater than (+), less than (-)or within (·) one s.d. of the long-term mean.

POSSIBLE FORAGE AND PREDATOR RESPONSES TO RECENT OCEAN CONDITIONS

Comparison of the abundances of various species across the three parts of the CCE indicates that their dynamics are not necessarily in sync across regions. Similarly, a recent analysis of ichthyoplankton assemblage structure between 2004 and 2011 in the northern and southern CCE showed that fish larvae respond to environmental variation very differently in California and Oregon, and that there was no correlation in the abundance of species found in both areas (e.g., northern anchovy) through time (Thompson et al. 2014) This emphasizes the need to sample widely throughout the system to understand how forage species in the broad ecosystem respond to environmental variability induced by El Niños.

In 2012 and 2013 the basin-scale indices and conditions from regional surveys indicated that oceanographic characteristics of the CCE were similar to recent cool years., Hereafter, we reference figures from the 'Oceanographic and Climatic Drivers and Pressures' (Hazen et al. 2014) chapter of this web report , indicated by the prefix 'OC'. The Pacific Decadal Oscillation (PDO) and Northern Oscillation Index (NOI) signaled a continued pattern of increased production and cool ocean waters and the NPGO was consistent with strong southward transport (see Figures OC7, OC8, and OC28 in Hazen et al. 2014, Ocean and Climate Drivers section of this report). Consistent with these large-scale signals, the CCE was cooler than typical 2009-2013, but an increasing trend is noted for the summer period (Fig.s OC39, OC40) (Hazen et al. 2014).

The timing of the spring transition is important to the development of forage community; early stronger winds are typically associated with improved production of the coastal forage community. At 45°N the spring transition occurred later 2008-2012 but was average in 2013 (Fig. OC21) (Hazen et al. 2014). By contrast, at 39°N the spring transition has been trending toward occurring earlier (Fig. OC21) (Hazen et al. 2014). Upwelling in the northern CCE was substantial, especially north of 39°N and the trend has been positive over the last five years (Fig.s OC18, OC19) South of 39°N upwelling winds have been at or slightly below typical (Fig. OC20) (Hazen et al. 2014). These regional and basin-scale conditions are conducive to an increased and more diverse coastal forage community (Brodeur and Pearcy 1992, Wells et al. 2008a, Santora et al. 2009, Sydeman et al. 2009, Santora et al. 2011, Santora et al. 2012, Thompson et al. 2012b, Schroeder et al. 2013, Wells et al. 2013).

There are limitations and differences between data series represented here, but in 2012 from our available observations, a CCE-wide pattern emerged with reduction of northern anchovy larvae (Figures C2, C5, and C7) and, to some degree, Pacific sardine larvae (Figures C5, C7, and C9). Catches of larval anchovy in the southern CCE waters have declined over the last three decades with the lowest densities recorded in the recent five years ending in 2011 (the last year of available data). The three possible causes of these trends are a reduction in spawning stock biomass, early survival, or increased advection from the region (Bakun and Parrish 1982).

Generally, in central and northern CCE those fishes whose abundance is reliant more on local (typically onshelf) conditions of production (Emmett et al. 2006, Santora et al. 2012) exhibited improved production/abundance in 2012 (Figures C6 and C8). For instance, in central CCE, a micronekton assemblage of rockfish, market squid, euphausiids, and flatfishes had improved production since 2005, consistent with increased local upwelling and productive shelf conditions. Similarly, whitebait smelt abundance (Emmett et al. 2006) was at average levels in the north in contrast to the declining abundances of northern anchovy and clupeids.

The reductions of Pacific sardine and northern anchovy larvae and the improved production of the forage reliant on shelf productivity may point to variability in the quality of the shelf and offshelf habitats. Namely, over much of the range of northern anchovy, the fish feed, and may even spawn, at and beyond the shelf break (Kramer and Ahlstrom 1968, Smith 1972). In part, the northern anchovy may be held offshore by advection (Bakun and Parrish 1982), however the scale of upwelling is only of the order 50 km. This is clear in the central CCE region where, even during the cool, productive conditions that benefit northern anchovy production (Lindegren et al. 2013), the northern anchovy are not overwhelmingly abundant in the survey region. It is only when upwelling subsides, or during relatively unproductive years associated with reduced winds (e.g., 2005 and 2006, Figure C5) (Peterson et al. 2006, Schwing et al. 2006) that northern anchovy become increasingly available to the trawls and the inshore environment. Pacific sardine, as well, reside more offshore at or beyond the shelf break (Kramer 1970). By contrast, the fishes reliant on productive, cool waters inshore have had improved production recently. These fishes, such as rockfish, market squid, flatfishes and others, reside largely in the productive cool nearshore waters during upwelling periods (Figure C5).

With 2013 came an exceptionally strong winter and spring upwelling period that acted predictably on the regional hydrography; salinities were greater and surface temperatures lower (Wells et al. 2013). Biological data, for the most part, has yet to be processed, therefore, the biological signal will be discussed in greater detail in the next report. However, the May-June juvenile rockfish survey did report record numbers of young-of-the-year pelagic rockfish, and high abundances of many other micronekton forage species as well (other juvenile groundfish, krill, and market squid).

EL NIÑO

The Climate Prediction Center (CPC) of the National Weather Service is forecasting a reasonable likelihood of an El Niño event in the near future. The CPC posted on the El Niño /Southern Oscillation (ENSO) Diagnostics Discussion board (http://www.cpc.ncep.noaa.gov/products/analysis_monitoring/enso_advisory/ensodisc.html) 10 April 2014 "While ENSO-neutral is favored for Northern Hemisphere spring, the chances of El Niño increase during the remainder of the year, exceeding 50% by summer." Importantly, El Niño periods have the potential to affect the forage community along the CCE and the predators that rely on them.

In southern CCE, El Niño events (e.g., 1983, 1992, 1998, and 2004) can potentially lead to variability in the productivity and distribution of the forage community. Sardine spawning habitat is expected to increase (Song et al. 2012). During the 1998 El Niño event, an increase in sardine catches was, indeed, noted (Figure C2). As well, there were decreases in fishes associated with cooler, typically more coastal waters such as hake, sanddab, rockfishes, and cool-water mesopelagics (Figure C2). In general, however, across the CalCOFI sampling region, variability in forage assemblages due to typical El Niño events may be modest but with an observed increase in oceanic contributors as warmer, offshore oceanic waters encompass a larger portion of the sampling area (Thompson et al. 2012a). An examination of Figures C3 and C4 demonstrate that there has not been a dramatic signal in larval fish assemblage structure representing El Niño years; although summer of 1998 had the lowest PC1 score of the years examined (largely reflecting extremely high abundances of Panama lightfish) it is not greatly unlike other more average years nor similar to 2003 and 2004 El Niño years. Further, spring of 1998 did not stand out from the other years. At a smaller scale, however, variations in assemblage makeup and distribution can be dramatic. Specifically, along more inshore regions of southern CCE, coastally-dependent fishes (e.g., northern anchovy, smoothtongue, and hake) can experience reduced production while oceanic-dependent fishes (e.g., broadfin lampfish, California flashlightfish, and bigfin laternfish, and longfin laternfish) increase in relative abundance as they become impinged closer to the coast (Thompson et al. 2012a). Such variability can impact foragers and lead to dependence on inferior prey. For instance, California sea lions, while dependent on sardine, also rely on coastal forage such as squid which is drastically reduced in the diets during El Niño events (Lowry and Carretta 1999). As demonstrated by the 1998 El Niño event, if prey availability is reduced to lactating females, their dependent pups may be in poor condition (Melin et al. 2010, Melin et al. 2012)

In central California, during strong El Niño events there have been reduced catches of pelagic juvenile rockfish and other juvenile groundfishes (Figure C5)(Ralston et al. 2013a, Ralston and Stewart 2013). However, in explaining the common trend in juvenile rockfish abundance there has been stronger correlation with relative sea level (as an indicator of transport, Fig. OC1) (Hazen et al. 2014) rather than either the MEI or other El Niño indices (Ralston et al. 2013a). This is largely due to the relatively poor correlation to years of high abundance, although some of the years of the highest abundance (e.g., 1999, Figure C5) follow strong El Niño events (and/or anomalously high northward transport in winter).

During El Niño events there has been a reduction in other coastally-dependent forage species (e.g., krill, particularly *Thysaonessa spinifera,* sanddabs, and market squid, Figure C5) and an associated increased abundance of sardine and mesopelagic fishes in the core survey area. Variability of sardine and mesopelagic fish abundances largely reflects changes in distribution and timing of movement patterns (so relates to local availability rather than coast-wide abundance). Such variability in forage dynamics can translate into availability of forage for predators such as seabirds and salmon, that subsequently rely on a different suite of prey species during El Niño years, with resultant real impacts to their productivity (Ainley et al. 1995, Sydeman et al. 1997, Sydeman et al. 2001, Sydeman et al. 2006, Wells et al. 2008a, Sydeman and Bograd 2009, Sydeman et al. 2009, Cury et al. 2011, Thompson et al. 2012b, Wells et al. 2012, Wells et al. 2013, Thayer et al. 2014). The multivariate methodology shown in Figure C6 demonstrates the dichotomy between the coastally-dependent forge species (e.g., rockfish) and those originating from more oceanic waters (e.g., sardines). It is clear from this analysis that the 1998 El Niño resulted in a dramatically different forage community from the average years. It was similar to only two other years (2005 and 2006) that, while not El Niño years, did demonstrate a similar ecosystem condition (Peterson et al. 2006, Schwing et al. 2006).

Along the northern CCE, during low upwelling, like 2005 and typical of El Niño years (Peterson et al. 2006, Schwing et al. 2006), there has been relatively low nutrients on the shelf leading to a dinoflagellate dominated phytoplankton community rather than a diatom phytoplankton community. This is likely to lead to longer food webs (more intermediate trophic levels) resulting in a less productive system (Brodeur and Pearcy 1992). As well, a relatively large euphausiid, *Thysaonessa spinifera,* is replaced by the southern smaller *Nyctiphanes simplex* and the large northern copepods are replaced with small southern ones. The impact of these changes is realized in the lipid and fatty acid composition of forage fish (Litz et al. 2010) that makes them less desirable as prey for higher trophic levels. The forage species' growth is also reduced leading to smaller size at age (Takahashi et al. 2012). These overall forage conditions lead to poor recruitment and condition of top predators such as salmon (Peterson and Keister 2003).

Consider, however, in the northern CCE, spawning of species like anchovy may actually be earlier and over a broader area in the warm El Niño years, leading to a potential good year class if conditions improve by late summer (Takahashi et al. 2012). In general, total larvae are more abundant during El Niño and/or low upwelling years and more larval and juvenile rockfish are found over the shelf rather than offshore (Auth 2008, 2011, Thompson et al. 2014) likely as a result of onshore impingement and increased ocean temperatures, which may expand spawning habitat and lead to better survival of those the impinged fishes. For the species we examined here, anchovy abundance was relatively low but sardine and whitebait smelt abundances increased during the 1998 El Niño (Figure C7). This was a particularly dramatic El Niño and, as indicated by multivariate analysis, 1998 was significantly separated from the other years on record as a result, in part, by a reduction in salmon and increase in the availability of more offshore-dependent species (e.g., sardine) (Figure C8).

CLIMATE CHANGE

Climate change has the potential to impact the forage community in a variety of ways. Some examples of these impacts are explored within the Management Strategy Evaluation (MSE) chapters of this report (e.g., Ruzicka 2014).

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APPENDIX C-1. EVALUATION OF INDICATORS FOR COASTAL PELAGIC AND FORAGE SPECIES USING THE BONNEVILLE POWER ADMINISTRATION DATA SET

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OVERVIEW

As currently reported within the CCIEA, the Coastal Pelagic and forage species (CPS) goal of the CCIEA reports on several indicators, but none have been evaluated systematically according to the same considerations suggested by Kershner et al. (2011) and in previous versions of the CCIEA (e.g., Groundfish section), or hierarchically with respect to particular attributes of the CPS goal. To fill these gaps, we evaluated 5 candidate indicators of Coastal Pelagic Species derivable from a data set developed with funding from the Bonneville Power Administration (BPA). The BPA Plume Survey has generated a systematic 15-year time series of surface trawls made along the Oregon and Washington coast (see description of survey methods in Brodeur et al. 2005).

To initiate an indicator evaluation and selection framework that parallels the Groundfish section (and others), we evaluated 5 candidate indicators with respect to 2 CPS attributes: population size and population condition. We defined these attributes as in other CCIEA sections (e.g., see the Groundfish goal). One candidate indicator, forage fish biomass (in aggregate), was evaluated with respect to the CPS attribute of population size. The remaining four candidate indicators were evaluated with respect to the population condition attribute for CPS, and included: spatial structure, mean length (size), size structure, and age structure. In the future, it would be worth evaluating additional CPS indicators of population size, such as survey-specific biomass or abundance, and population condition, like age and size at maturity. In the Table of Contents below, the name of each indicator we evaluated is listed, followed by its score in brackets [].

We adopted a tiered approach to the evaluation of each indicator, similar to the approach used for indicator evaluations in the Groundfish section of the CCIEA:

1. Evaluate the indicator generically, without respect to taxa or data sets, for theoretical and other considerations.
2. Modify the indicator evaluation for theoretical and other considerations based on specific information related to CPS taxa.
3. Evaluate the indicator with respect to specific CPS data sets for data considerations.

For example, for the indicator *age structure*, in the first tier of our evaluation, we focused on the criteria listed under theoretical considerations and other considerations, without concern for the data or taxa on which this indicator would be based. After this initial evaluation, we modified the supporting documentation, references, and scoring for this indicator so that the theoretical considerations and other considerations were evaluated for CPS taxa specifically. Finally, we evaluated the data considerations criteria with respect to a specific CPS data set (that derived from the BPA Plume survey; Brodeur et al. 2005).

While coastal pelagic species indicators, such as spatial structure, mean length (size), size structure, and age structure are potentially informative of ecosystem structure or function, we felt that they were best evaluated as candidate CPS indicators, rather than as candidate Ecological Integrity indicators. Similarly, we did not evaluate the reliability of individual coastal pelagic species as indicators of the Ecological Integrity goal, as doing so would require detailed knowledge of the relationship between each species and the Ecological Integrity attributes (ecosystem trophic structure and biodiversity), which is beyond the scope of our current expertise.

All 5 of the candidate CPS goal indicators received scores ≥12, suggesting that they would all serve reasonably well. However, we only evaluated one candidate indicator of the CPS population size attribute, and suspect that estimates of the biomass of individual species would perform better than forage fish biomass in aggregate (see Groundfish goal for parallel evaluation). **The highest scoring indicator of CPS population condition was age structure.**

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SUMMARY OF INDICATOR EVALUATIONS

Table 1. Summary of indicator evaluations for the Coastal Pelagic Species goal of the CCIEA. The numerical value that appears under each of the considerations represents the summed scores for the criteria evaluated for each type of consideration. Criteria with full support in the peer-reviewed literature received a score of 1, those with partial support received a score of 0.5, and those with little or no support received a score of 0.

CCIEA Goal: Attribute	Indicator	Primary considerations (5)	Data considerations (7)	Other considerations (5)	Summary comments
Coastal Pelagic Species: Population Size	Forage fish abundance; in aggregate	4	5	3	Highly important to global fish landings and food webs. Stock assessments are conducted on most important taxa to guide harvest. Aggregating the abundance of all forage species may reduce its sensitivity as an indicator, and population size may not respond to management actions if a threshold shift has occurred in the ecosystem. Changes in trends of aggregate groups will be concurrent, at best. Most commonly reported indicators involving forage species are based on biomass.
Coastal Pelagic Species: Population Condition	Spatial structure of population	3.5	5	4.5	Changing spatial distributions of pelagic nekton have been predicted with some climate change scenarios, showing corresponding impacts to associated commercial fisheries. It is difficult to specifically attribute these changes to particular causes, however. Though not necessarily anticipatory, this indicator is well understood by policymakers and compatible with a number of other ecosystem assessments.

CCIEA Goal: Attribute	Indicator	Primary consider-ations (5)	Data consider-ations (7)	Other consider-ations (5)	Summary comments
Coastal Pelagic Species: Population Condition	Mean length (size)	3	5	4	Length-based indicators receive contradictory theoretical support from the literature about population condition, unless they are tied closely to age information. The BPA survey does have size information for a 15-y time series of data in the region. Though used to assess year class strength, mean length is better for measuring medium term trends.
Coastal Pelagic Species: Population Condition	Size structure	3	5	4	Size- based indicators receive contradictory theoretical support from the literature about population condition, unless they are tied closely to age information. The BPA survey does have size information for a 15-y time series of data in the region. Though used to assess year class strength, size structure is better for measuring medium term trends.
Coastal Pelagic Species: Population Condition	Age structure	4.5	5	4	Age structure is the basis for most ecological models and stock assessments, and generally has well-defined progress targets. Age truncation is known to have severe consequences for fish population sustainability. This indicator is well understood and highly compatible, with a variety of other assessments.

DETAILED INDICATOR EVALUATIONS FOR THE ECOLOGICAL INTEGRITY GOAL

FORAGE FISH ABUNDANCE; IN AGGREGATE [12]

CCIEA Goal: Coastal Pelagic Species

Attribute: Population Size

Total score: 12

Summary: Highly important to global fish landings and food webs. Stock assessments are conducted on most important taxa to guide harvest. Aggregating the abundance of all forage species may reduce its sensitivity as an indicator, and population size may not respond to management actions if a threshold shift has occurred in the ecosystem. Changes in trends of aggregate groups will be concurrent, at best. Most commonly reported indicators involving forage species are based on biomass.

Criterion	**Score**	**Explanation**
(a) Theoretically-sound	1	Most indicators of the pelagic fish community are based on biomass, not numbers, although stock assessments of Pacific sardine and mackerel generally estimate both biomass and age-based population size (Crone et al. 2011, Hill et al. 2011) Survey abundance may be a better indicator of relative spatial distribution than population strength (Bjorkstedt et al. 2012, Song et al. 2012) Abundance surveys show distribution shifts of sardine into NW waters in the mid-1990's (Emmett et al. 2005) and Baltic Sea pelagic ecosystems appear to be driven by planktivore abundance (Casini et al. 2009).
(b) Relevant to management concerns	1	Forage fish account for over 30% of global fish landings, and play an important role in marine food webs because they are the principal means of transferring production from plankton to larger predatory fish and to marine mammals and seabirds (Smith et al. 2011) Fishery Management Plans for assessed species (e.g., sardine, CPS Fishery Management Plan; State plans for Herring assessment and management), as well as entire CalCOFI sampling program, attest to management importance.
(c) Responds predictably & is	1	Climate or regime shifts; Survey abundance of anchovy and sardine in California current represent variability in the distribution throughout the

sufficiently sensitive to changes in a specific ecosystem attribute(s)		CCE (Bjorkstedt et al. 2012, Song et al. 2012) Pelagic nekton responded to anomalous environmental conditions in CCE in 2005, including widespread onshore and poleward displacement of taxa to new geographic areas, population changes within the normal range, and reduced productivity of early life stages based on larval and juvenile surveys (Brodeur et al. 2006) Abundance surveys show distribution shifts of sardine into NW waters in the mid-1990's, likely in response to regional warming trends (Emmett et al. 2005) Between 1998 and 2002, species composition shifted from a community dominated by southern species (mackerels and hake) to one dominated by northern species (squid, smelts, and salmon) (Brodeur et al. 2005).
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	0.5	Mgt. action: the pressure of fishing these species at conventional maximum sustainable yield (MSY) levels can have large impacts on other parts of the ecosystem, particularly when they constitute a high proportion of the biomass in the ecosystem or are highly connected in the food web (Smith et al. 2011; Kaplan et al. 2013) Precautionary management will theoretically affect population, though not if threshold shift has occurred or environmental drivers have changed (Casini et al. 2009) Survey abundance of anchovy and sardine in California current is not likely to represent overall population strength as much as variability in the distribution throughout the CCE (Bjorkstedt et al. 2012, Song et al. 2012).
(e) Linkable to scientifically-defined reference points & progress targets	0.5	Most reference points are based on population biomass (Link 2005). Fishery Management Plan reference points for managed species have been applied as reference targets for sardines (Hill et al. 2011), but simulation models by Kaplan et al. (2013) show that groundfish (B25, B40) levels of forage species removal is likely to impact the abundance of other target species, protected species, and the structure of the ecosystem.
(g) Concrete & Numerical	1	Large trawls (336 m2 mouth opening) in the upper 18-20 m of the water column at every station. Mesh sizes ranged from 162.6 cm in the throat of the trawl near the jib lines to 8.9 cm in the cod end. To maintain catches of small fish and squids, a 6.1 m long, 0.8 cm knotless liner was sewn into the cod end. Total counts and weights are available for each nekton taxon caught.

		Length data are recorded for a subset of individuals (n=50 per station) of all species.
(h) Historical data or information available	0.5	Since 1998; Annual systematic surveys have been conducted using consistent methodology in May, June, and September of each year
(i) Operationally simple	1	Measurements of hydrography, zooplankton and pelagic fishes are made at each station. Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.
(k) Broad spatial coverage	0.5	Mid-Oregon to North WA coast (Newport to Tatoosh Island; along nine transect lines at 50-55 stations, ranging from 45 to 48 deg N.)
(l) Continuous time series	1	15 y (1998-2012), in June and September
(m) Spatial & temporal variation understood	0.5	Annual and summer/spring variability well-understood for most taxa; seasonal less so. Sampling occurs during day or crepuscular periods so diel patterns are less clear and myctophids are not well characterized.
(n) High signal-to-noise ratio	0.5	Sampled taxa have naturally highly variable populations; for some of the best sampled taxa signal to noise is high or can be discerned, although poor for myctophids.
(o) Understood by the public & policymakers	1	Yes; relative trends in abundance of each component of the community is easily understood.
(p) History of reporting	0.5	Biomass is usually the most commonly reported indicator involving pelagic / zooplanktivorous fishes (Link 2005, Samhouri et al. 2009, Fulton et al. 2005) Historic time series provide relative CPUE estimates for regions of the CCE (e.g., Brodeur et al. 2005, Emmett et al. 2005).
(q) Cost-effective	1	Assessment data or other pelagic surveys are already conducted for many of these species; data mining is all that is needed.

(r) Anticipatory or leading indicator	0.5	Changes in trends of aggregate groups will always be concurrent at best Rapid response of forage fishes and other nekton (distributional anomalies) to delayed upwelling (Brodeur et al. 2006) Forage fish generally show a short lag between a change in ocean phase and population response (http://www.nwfsc.noaa.gov/research/divisions/fe/estuarine/oeip/ha-under-development.cfm), but this is relatively fast compared to other North Pacific fish populations (Yatsu et al. 2008).
(t) Regionally/ nationally/ internationally compatible	0	No. Most commonly reported indicators involving pelagic / zooplanktivorous fishes are based on biomass (Link 2005, Samhouri et al. 2009, Fulton et al. 2005)
TOTAL SCORE	12	

SPATIAL STRUCTURE OF POPULATION (CENTER OF DISTRIBUTION [LATITUDE OR DEPTH]) [13]

CCIEA Goal: Coastal Pelagic Species

Attribute: Population condition

Total score: 13

Summary: Changing spatial distributions of pelagic nekton have been predicted with some climate change scenarios, showing corresponding impacts to associated commercial fisheries. It is difficult to specifically attribute these changes to particular causes, however. Though not necessarily anticipatory, this indicator is well understood by policymakers and compatible with a number of other ecosystem assessments.

Criterion	**Score**	**Explanation**
(a) Theoretically-sound	1	Spatial structure of biomass considered as attribute by Fulton et al. (2005); Reese and Brodeur (2006) used species concentration/composition to identify areas of biological activity, with flow and circulation suggested as the primary drivers of these patterns. Climate change may lead to large-scale redistribution of global catch potential, with an average of 30–70% increase in high-latitude regions and a drop of up to 40% in the tropics (Cheung et al. 2010). As an example, Barcelo et al. (in prep) show a northward shift in the center of distribution of market squid. Northward shifts in species distributions were matched by corresponding northward shifts in fisheries (Pinsky and Fogarty 2012).
(b) Relevant to management concerns	1	Fishery Management Plans of assessed CPS species like Pacific sardine and mackerel (Hill et al. 2011; Crone et al. 2011). Spatial changes may have impacts on the nature and value of these commercial fisheries (Perry et al. 2005). Changing species distributions are predicted to result in

		species extinctions on land and population extinctions in the sea (Thomas et al. 2004; Drinkwater 2005); Climate change may lead to large-scale redistribution of global catch potential, with an average of 30–70% increase in high-latitude regions and a drop of up to 40% in the tropics (Cheung et al. 2010). Northward shifts in species distributions were matched by corresponding northward shifts in fisheries, with fisheries lagging by 10-30%; lags may lead to overfishing and population declines (Pinsky and Fogarty 2012).
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	0.5	Distributional shifts are hypothesized to occur for either two reasons - climatic or exploitation - but the difference is difficult to distinguish. Dulvy et al. (2008) suggests changes in depth distribution of an assemblage of North Sea groundfish is due to climate while, latitudinal shifts may be caused by either - (similar to Coetzee 2008 & Fairweather et al. 2006). Distributions of North Sea fishes responded markedly to increases in sea temperature, with nearly two-thirds of species shifting in mean latitude or depth or both over 25 years; all but one shifted northward. Shifting species were those with faster life cycles (Perry et al. 2005). In the northern California current, models have identified significant relationships between sardine eggs and sea surface temperature, chlorophyll a concentration, and the gradient of sea surface altitude. The model accurately predicts the habitat and seasonal migration pattern of sardine, irrespective of spawning condition (Zwolinski et al. 2011). Reese and Brodeur (2006) used species composition and concentration to identified areas of biological activity, with flow and circulation suggested as the primary drivers of these patterns.
(d) Responds predictably & is sufficiently sensitive to changes in a specific management action(s) or pressure(s)	0.5	Distributional shifts are hypothesized to occur for either two reasons - climatic or exploitation - but the difference is difficult to distinguish. Perry et al. 2005 showed large latitudinal shifts correlated with changes in temperature. Dulvy et al. (2008) suggests changes in depth distribution of an assemblage of North Sea groundfish is due to climate while, latitudinal shifts may be caused by either - similar to

		Coetzee 2008 & Fairweather et al. 2006. The geographic ranges of overexploited species typically decline, and stocks are concentrated into smaller regions following population declines (Atkinson et al., 1997; Garrison & Link 2000).
(e) Linkable to scientifically-defined reference points & progress targets	0.5	Reference points for distributional shifts are not used and would be difficult to measure unless species were divided into Distinct Population Segments and shifts away from one segment triggered management actions Management should strive to preserve a minimal spawning biomass throughout the geographic range of the stock (Berkeley et al. 2004). Spatial management tools need to formulate more specific targets (Backcock et al. 2005). However, historical spatial center could serve as a straightforward reference point (Cheung et al. 2010).
(g) Concrete & Numerical	1	Large trawls (336 m2 mouth opening) in the upper 18-20 m of the water column at every station. Mesh sizes ranged from 162.6 cm in the throat of the trawl near the jib lines to 8.9 cm in the cod end. To maintain catches of small fish and squids, a 6.1 m long, 0.8 cm knotless liner was sewn into the cod end. Total counts and weights are available for each nekton taxon caught. Length data are recorded for a subset of individuals (n=50 per station) of all species.
(h) Historical data or information available	0.5	Since 1998; Annual systematic surveys have been conducted using consistent methodology in May, June, and September of each year
(i) Operationally simple	1	Measurements of hydrography, zooplankton and pelagic fishes are made at each station. Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.
(k) Broad spatial coverage	0.5	Mid-Oregon to North WA coast (Newport to Tatoosh Island; along nine transect lines at 50-55 stations, ranging from 45 to 48 deg N.)

(l) Continuous time series	1	15 y (1998-2012), in June and September
(m) Spatial & temporal variation understood	0.5	Annual and summer/spring variability well-understood for most taxa; seasonal less so. Sampling occurs during day or crepuscular periods so diel patterns are less clear and myctophids are not well characterized.
(n) High signal-to-noise ratio	0.5	Sampled taxa have naturally highly variable populations; for some of the best sampled taxa signal to noise is high or can be discerned, although poor for myctophids.
(o) Understood by the public & policymakers	1	Yes, Dulvy et al. 2008
(p) History of reporting	1	Yes, stock assessments, bottom trawl survey, and landings by area or by location from trawl logbooks; CPS analyses generally divided by region.
(q) Cost-effective	1	Yes, stock assessments, bottom trawl survey, and landings by area or by location from trawl logbooks
(r) Anticipatory or leading indicator	0.5	Not necessarily, although some estimates of changing species distributions could be anticipatory.
(t) Regionally/ nationally/ internationally compatible	1	Yes; Perry et al. 2005, Coetzee 2008, Dulvy et al. 2008, Garrison & Link 2000.
TOTAL SCORE	13	

MEAN LENGTH (SIZE) [12]

CCIEA Goal: Coastal Pelagic Species

Attribute: Population condition

Total score: 12

Summary: Length-based indicators receive contradictory theoretical support from the literature about population condition, unless they are tied closely to age information. The BPA survey does have size information for a 15-y time series of data in the region. Though used to assess year class strength, mean length is better for measuring medium term trends.

Criterion	**Score**	**Explanation**
(a) Theoretically-sound	0.5	Mean size of all species caught in either fishery-independent surveys, fishery-dependent surveys, and/or landings is a useful and simple indicator to evaluate the overall effects of fishing on an ecosystem (Link 2005; Jennings & Kaiser 1998; Rochet & Trenkel 2003); Average size or distribution of sizes in catch is often taken as an indicator of population status...[but, there are] very few examples where length-based analysis leads to useful management advice (Hilborn and Walters 1992)
(b) Relevant to management concerns	0.5	Only if size representative of age. See references and notes for (a).
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	0.5	Attempts to use length-based analysis to formulate management advice for [some] species... is misguided and fundamentally hopeless (you need age information, gear selectivity and size related changes in distribution will influence data) (Hilborn and Walters 1992); Mean length (size) metrics... (Link & Brodziak 2002; Link et al. 2002; Rochet & Trenkel 2003; Nicholson & Jennings 2004). Fish size responds predictably to dietary ration and temperature.
(d) Responds predictably & is sufficiently sensitive to	0.5	Mean size (l, as length) of all species caught in either fishery-independent surveys, fishery-dependent surveys, and/or

changes in a specific management action(s) or pressure(s)		landings is a useful and simple indicator to evaluate the overall effects of fishing on an ecosystem... even if the change cannot be directly attributed to fishing, the indicator should still be monitored more closely, with initial steps taken to mitigate the change (Link & Brodziak 2002; Link et al. 2002; Rochet & Trenkel 2003; Nicholson & Jennings 2004). Attempts to use length-based analysis to formulate management advice for [some] species... is misguided and fundamentally hopeless (you need age information, gear selectivity and size related changes in distribution will influence data) - (Hilborn and Walters 1992); Length-based indicators weighted by biomass, rather than abundance, were more sensitive and specific to fishing pressure (Houle et al. 2012). Exploitation influences community structure directly through preferential removal of larger-bodied fishes and indirectly because larger-bodied fishes may exert top-down control upon other community members (Dulvy et al. 2004); A decrease of mean length in the population is expected and has been observed under the effects of fishing (Haedrich and Barnes 1997; Babcock et al. 1999). "...because of methodological difficulties and despite accumulated experience, it is currently not possible to state precisely how fishing affects size spectra" (Rochet and Trenkel 2003.)
(e) Linkable to scientifically-defined reference points & progress targets	1	If there is a decline in mean size of more than 30% between years, then a warning or precautionary threshold (30%) has been exceeded; the indicator should still be monitored more closely, with initial steps taken to mitigate the change. The limit reference point (LRP) has been set at a value of 50% decline, and the control rule would be to enlarge mesh size or to take similar action to alter fishing gear; 50% was chosen because it corresponds to an observed doubling in the time-series of l after fishing decreased (Link 2005 and references therein); Attempts to use length-based analysis to formulate management advice for [some] species... is misguided and fundamentally hopeless (you need age information, gear selectivity and size related changes in distribution will influence data) (Hilborn and Walters 1992); reference directions provide alternative medium-term

		management targets (Jennings and Dulvy 2005)
(g) Concrete & Numerical	1	Large trawls (336 m^2 mouth opening) in the upper 18-20 m of the water column at every station. Mesh sizes ranged from 162.6 cm in the throat of the trawl near the jib lines to 8.9 cm in the cod end. To maintain catches of small fish and squids, a 6.1 m long, 0.8 cm knotless liner was sewn into the cod end. Total counts and weights are available for each nekton taxon caught. Length data are recorded for a subset of individuals (n=50 per station) of all species.
(h) Historical data or information available	0.5	Since 1998; Annual systematic surveys have been conducted using consistent methodology in May, June, and September of each year
(i) Operationally simple	1	Measurements of hydrography, zooplankton and pelagic fishes are made at each station. Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.
(k) Broad spatial coverage	0.5	Mid-Oregon to North WA coast (Newport to Tatoosh Island; along nine transect lines at 50-55 stations, ranging from 45 to 48 deg N.)
(l) Continuous time series	1	15 y (1998-2012), in June and September
(m) Spatial & temporal variation understood	0.5	Annual and summer/spring variability well-understood for most taxa; seasonal less so. Sampling occurs during day or crepuscular periods so diel patterns are less clear and myctophids are not well characterized.
(n) High signal-to-noise ratio	0.5	Sampled taxa have naturally highly variable populations; for some of the best sampled taxa signal to noise is high or can be discerned, although poor for myctophids.
(o) Understood by the public & policymakers	1	Yes. Link 2005; Fulton et al. 2005

(p) History of reporting	1	Yes. Link & Brodziak 2002
(q) Cost-effective	1	Yes, data collected, used to assess year class strength, included in stock assessments and FMPs for assessed groundfish species; http://www.pcouncil.org/wp-content/uploads/fmpthru19.pdf
(r) Anticipatory or leading indicator	0	No; better for measuring medium term (5-10 y) trends (Jennings and Dulvy 2005)
(t) Regionally/ nationally/ internationally compatible	1	Methratta & Link 2006; Fulton et al. 2005
TOTAL SCORE	12	

SIZE STRUCTURE [12]

CCIEA Goal: Coastal Pelagic Species

Attribute: Population condition

Total score: 12

Summary: Size- based indicators receive contradictory theoretical support from the literature about population condition, unless they are tied closely to age information. The BPA survey does have size information for a 15-y time series of data in the region. Though used to assess year class strength, size structure is better for measuring medium term trends.

Criterion	**Score**	**Explanation**
(a) Theoretically-sound	0.5	Population size structure has been evaluated as a good indicator of fishing by Fulton et al. (2005; L_{max}) and Coll et al. (2012, mean fish size); At large scales(ecosystems), size-based indicators show reliable responses to changes in rates of fishing mortality (Jennings and Dulvy 2005 and refs therein). Average size or distribution of sizes in catch is often taken as an indicator of population status...[but, there are] very few examples where length-based analysis leads to useful management advice (Hilborn and Walters 1992)
(b) Relevant to management concerns	0.5	Only if size representative of age. See references and notes for (a).
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	0.5	Attempts to use length-based analysis to formulate management advice for [some] species... is misguided and fundamentally hopeless (you need age information, gear selectivity and size related changes in distribution will influence data) (Hilborn and Walters 1992); (Link & Brodziak 2002; Link et al. 2002; Rochet & Trenkel 2003; Nicholson & Jennings 2004). Fish size responds predictably to dietary ration and temperature.
(d) Responds predictably & is sufficiently sensitive to	0.5	Attempts to use length-based analysis to formulate management advice for [some] species... is misguided and

changes in a specific management action(s) or pressure(s)		fundamentally hopeless (you need age information, gear selectivity and size related changes in distribution will influence data) - (Hilborn and Walters 1992); Size-based metrics would better support medium-term rather than year-on-year management evaluation, because (1.) they are unlikely to be appropriate for detecting responses to management action on time scales <5 years, and (2.) the response to management action cannot be quantitatively decomposed in the contributing causal factors without extensive additional research (Jennings and Dulvy 2005). Population size structure has been evaluated as a good indicator of fishing by Fulton et al. (2005; Lmax) and Coll et al. (2012, mean fish size); at large scales (ecosystems), size-based indicators show reliable responses to changes in rates of fishing mortality (Jennings 2005 and refs therein). Length-based indicators weighted by biomass, rather than abundance, were more sensitive and specific to fishing pressure (Houle et al. 2012). Exploitation influences community structure directly through preferential removal of larger-bodied fishes and indirectly because larger-bodied fishes may exert top-down control upon other community members (Dulvy et al. 2004)
(e) Linkable to scientifically-defined reference points & progress targets	1	If there is a decline in mean size of more than 30% between years, then a warning or precautionary threshold (30%) has been exceeded; the indicator should still be monitored more closely, with initial steps taken to mitigate the change. The limit reference point (LRP) has been set at a value of 50% decline, and the control rule would be to enlarge mesh size or to take similar action to alter fishing gear; 50% was chosen because it corresponds to an observed doubling in the time-series of l after fishing decreased (Link 2005 and references therein); Attempts to use length-based analysis to formulate management advice for [some] species... is misguided and fundamentally hopeless (you need age information, gear selectivity and size related changes in distribution will influence data) (Hilborn and Walters 1992); Reference directions provide alternative medium-term management targets (Jennings and Dulvy 2005)

(g) Concrete & Numerical	1	Large trawls (336 m^2 mouth opening) in the upper 18-20 m of the water column at every station. Mesh sizes ranged from 162.6 cm in the throat of the trawl near the jib lines to 8.9 cm in the cod end. To maintain catches of small fish and squids, a 6.1 m long, 0.8 cm knotless liner was sewn into the cod end. Total counts and weights are available for each nekton taxon caught. Length data are recorded for a subset of individuals (n=50 per station) of all species.
(h) Historical data or information available	0.5	Since 1998; Annual systematic surveys have been conducted using consistent methodology in May, June, and September of each year
(i) Operationally simple	1	Measurements of hydrography, zooplankton and pelagic fishes are made at each station. Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.
(k) Broad spatial coverage	0.5	Mid-Oregon to North WA coast (Newport to Tatoosh Island; along nine transect lines at 50-55 stations, ranging from 45 to 48 deg N.)
(l) Continuous time series	1	15 y (1998-2012), in June and September
(m) Spatial & temporal variation understood	0.5	Annual and summer/spring variability well-understood for most taxa; seasonal less so. Sampling occurs during day or crepuscular periods so diel patterns are less clear and myctophids are not well characterized.
(n) High signal-to-noise ratio	0.5	Sampled taxa have naturally highly variable populations; for some of the best sampled taxa signal to noise is high or can be discerned, although poor for myctophids.
(o) Understood by the public & policymakers	1	Yes, used to assess year class strength, included in stock assessments and FMPs for assessed groundfish species; http://www.pcouncil.org/wp-content/uploads/fmpthru19.pdf

(p) History of reporting	1	Yes, used to assess year class strength, included in stock assessments and FMPs for assessed groundfish species; http://www.pcouncil.org/wp-content/uploads/fmpthru19.pdf
(q) Cost-effective	1	Yes, data collected, used to assess year class strength, included in stock assessments and FMPs for assessed groundfish species; http://www.pcouncil.org/wp-content/uploads/fmpthru19.pdf
(r) Anticipatory or leading indicator	0	No; better for measuring medium term (5-10 y) trends (Jennings and Dulvy 2005)
(t) Regionally/ nationally/ internationally compatible	1	Celtic Sea (Blanchard et al. 2005); Northeastern groundfish stocks (Link 2005), Eastern Bering Sea community size structure (AFSC 2009)
TOTAL SCORE	12	

AGE STRUCTURE [13.5]

CCIEA Goal: Coastal Pelagic Species

Attribute: Population condition

Total score: 12.5

Summary: Age structure is the basis for most ecological models and stock assessments, and generally has well-defined progress targets. Age truncation is known to have severe consequences for fish population sustainability. This indicator is well understood and highly compatible, with a variety of other assessments.

Criterion	**Score**	**Explanation**
(a) Theoretically-sound	1	Sardine (Hill et al. 2011, McClatchie et al. in prep) and Pacific mackerel (Crone et al. 2011) (and theoretically, all other) stock assessments; Atlantis is based on age-structured models - Fulton et al. 2005; Age truncation commonly induced by fisheries may have severe consequences for long-term sustainability of fish populations (Berkeley et al. 2004). Clupeids (e.g. Pacific sardine, herring), smelts [e.g. eulachons, and other forage fishes such as northern lampfish and Pacific sandlance, are short-lived with a small body size and size at maturation, low fecundity, high growth rates and small eggs. They are also surface and mid-water pelagic species that exhibit little if any parental investment and are planktivores or lower-order carnivores - considered opportunistic strategists; population responses tend to be large in amplitude and species grouped according to this life history strategy have been classified as having either cyclical, irregular, or spasmodic population patterns (King and MacFarlane 2003);
(b) Relevant to management concerns	1	Fishery Management Plans for sardine - Stock assessments have been performed for sardine since 1982 by using an age-structured population model incorporating both fishery-dependent and fishery-independent data (U.S. Department of Commerce, Technical Memorandum NOAA-

		TM-NMFS-SWFSC-396) (Hill et al. 2011); "A full and stable age distribution indicates that a population has had recruitment successfully for a number of years, older large fish are present in the population, and the fishery may be less prone to collapse. However following a few years of reduced recruitment (and typically lower biomass) the age structures can be weighted more toward older fish. Likewise, when there is a boom year the vast majority of the biomass will shift to younger fish that could destabilize the age structure for a number of years following (McClatchie et al. *in prep*); Age truncation commonly induced by fisheries may have severe consequences for long-term sustainability of fish populations (Berkeley et al. 2004). Opportunistic strategists (like most pelagic nekton) should be managed to maintain a critical minimum spawning biomass; Spawning stock biomass is the basis of management targets and "spawning-stock" is based on age of fish (King and MacFarlane 2003)
(c) Responds predictably & is sufficiently sensitive to changes in a specific ecosystem attribute(s)	1	Asymmetrical age structure may also be a sign of extended period of poor recruitment. Annual recruitment is estimated using a cohort analysis of aged fish; full and stable age distribution indicates that a population has had recruitment successfully for a number of years, older large fish are present in the population, and the fishery may be less prone to collapse ; broad age distribution can also reduce recruitment variability (All stock assessments, Berkeley et al. 2004); Abundance and distribution of opportunistic strategists (e.g. Pacific sardine and Pacific herring) are known to fluctuate concurrently with climate–ocean regimes (McFarlane & Beamish 2001). Population responses tend to be large in amplitude and species grouped according to this life history strategy have been classified as having either cyclical, irregular or spasmodic population patterns (King and MacFarlane 2003).
(d) Responds predictably & is sufficiently sensitive to changes in a specific	0.5	One of the more predictable effects of fishing is the reduction or removal of the older age classes, i.e., age truncation. Age truncation commonly induced by fisheries

management action(s) or pressure(s)		may have severe consequences for long-term sustainability of fish populations (Berkeley et al. 2004). However, this may be more relevant to species with different life history strategy (e.g., long-lived groundfish). In CCE, Pacific herring represent one of the only forage species for which fishing is considered to be contributing to truncated age structure (Pikitch et al. 2014)
(e) Linkable to scientifically-defined reference points & progress targets	1	All stock assessments, Spawning stock biomass is the basis of management targets and "spawning-stock" is based on age of fish.
(g) Concrete & Numerical	1	Large trawls (336 m^2 mouth opening) in the upper 18-20 m of the water column at every station. Mesh sizes ranged from 162.6 cm in the throat of the trawl near the jib lines to 8.9 cm in the cod end. To maintain catches of small fish and squids, a 6.1 m long, 0.8 cm knotless liner was sewn into the cod end. Total counts and weights are available for each nekton taxon caught. Length data are recorded for a subset of individuals (n=50 per station) of all species.
(h) Historical data or information available	0.5	Since 1998; Annual systematic surveys have been conducted using consistent methodology in May, June, and September of each year
(i) Operationally simple	1	Measurements of hydrography, zooplankton and pelagic fishes are made at each station. Vertical profiles of temperature, salinity, fluorescence, oxygen and turbidity are taken at each station.
(k) Broad spatial coverage	0.5	Mid-Oregon to North WA coast (Newport to Tatoosh Island; along nine transect lines at 50-55 stations, ranging from 45 to 48 deg N.)
(l) Continuous time series	1	15 y (1998-2012), in June and September
(m) Spatial & temporal variation understood	0.5	Annual and summer/spring variability well-understood for most taxa; seasonal less so. Sampling occurs during day or crepuscular periods so diel patterns are less clear and myctophids are not well

		characterized.
(n) High signal-to-noise ratio	0.5	Sampled taxa have naturally highly variable populations; for some of the best sampled taxa signal to noise is high or can be discerned, although poor for myctophids.
(o) Understood by the public & policymakers	1	Sardine stock assessment and FMP
(p) History of reporting	1	Sardine stock assessment and FMP; Even if not all CPS have age structure reported historically, the concept is highly familiar to the PFMC
(q) Cost-effective	0.5	Sardine stock assessment and FMP
(r) Anticipatory or leading indicator	0.5	Concurrent; (Jennings and Dulvy 2005)
(t) Regionally/ nationally/ internationally compatible	1	Celtic Sea (Blanchard et al. 2005); Northeastern groundfish stocks (Link 2005), Eastern Bering Sea community size structure (AFSC 2009)
TOTAL SCORE	13.5	

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GROUNDFISH

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OVERVIEW

Groundfish abundance trends are generally upward or stable, though the underlying population structure, which builds more slowly than overall spawning biomass, shows most stocks are below indicator target levels.

EXECUTIVE SUMMARY

Groundfish are an important component of the California Current. Over 46 indicators of groundfish population size and population condition were evaluated for use on the 90+ groundfish stocks of the California Current during 2012 (Cope et al. 2013). We used two indicators of groundfish population size: 1) biomass of groundfish relative to either the estimate of unfished biomass (when a stock assessment is available) or trends in the survey time series, and 2) the number of assessed species below management thresholds. Additionally, two indicators of groundfish population condition were selected: 1) the proportion of the population mature (using ages or size in the absence of ages), and 2) the 95% cumulative age or length of the population. We summarized the status of stocks (based on biomass trends) and population demographic condition (as measured by the percentage of mature individuals and of maximum age or size) for 36 groundfishes; the remaining species did not have sufficient data to determine their status at this time. We found that most assessed groundfishes are above the biomass limit reference point, and thus are not overfished (Figure GF 1; Table GF 1). The three assessed stocks currently in an overfished state are all rockfishes. All assessed groundfishes are below their target catch, thus overfishing is not occurring in these stocks. With respect to population condition measures, we discovered that age or length structure tended to show more changes (usually declines) over time than the proportion that is sexually mature. We also found that non-elasmobranch groundfishes tended to exhibit the most changes over time in both measures, with rockfishes being most sensitive to demographic changes. The development of additional data-limited methods may allow more species to be included in future iterations of the IEA.

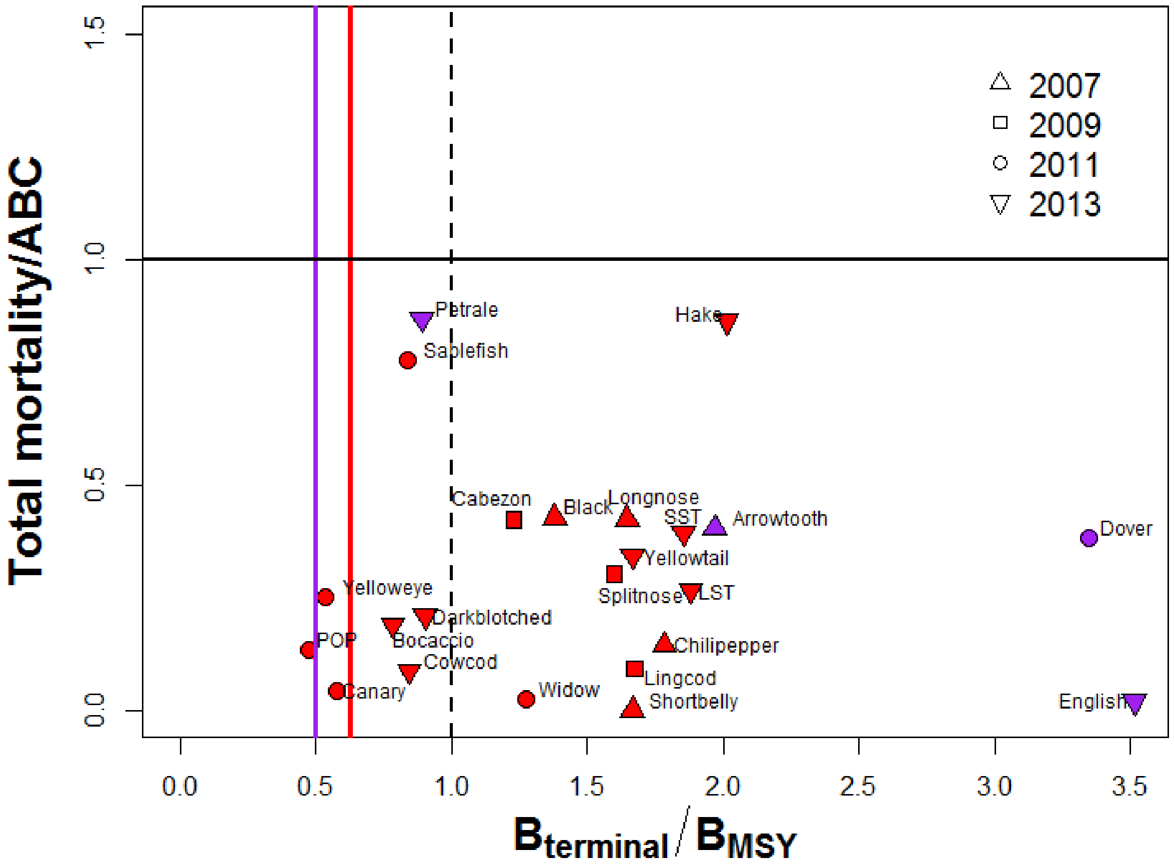

Figure GF1. Stock status plot relative to being overfished (x-axis) and overfishing (y-axis) for all species assessed since 2007 that were not managed in a stock complex at the time of the assessment. Vertical broken line indicates the target biomass reference point. Vertical solid line indicates the limit reference point indicating an overfished status (red for elasmobranchs, rockfishes, and roundfishes; purple for flatfishes). Horizontal line indicates overfishing wherein total mortality exceeds the allowable biological catch (ABC). For example, sablefish is below the target (black vertical broken line), but above the limit (red vertical solid line) biomass target, and below the overfishing limit (horizontal solid line). Symbols indicate the terminal year of the assessment in which the reference points are determined.

Table GF1. Results for each stock evaluated for each of two status indicators: 1) Biomass and 2) Population structure. Two sources of information were used: 1) Stock assessments and 2) Northwest Fisheries Science Center (NWFSC) shelf-slope trawl survey, when assessments were not available, or older than 2007. "Depletion" refers to the relative change in spawning biomass; "5-year trend" is the trend in the last 5 years of the time series (details found in the text). "B final year" is the biomass value in the final year compared to the 5-year average. "Prop. mature" is proportion of the population mature relative to the beginning of the time series; "95% cum." refers to the 95% cumulative age or length of the population relative to the beginning of the time series. +: above target limit or increasing over last 5 years; ●: between target and limit or stable; -: below limit or decreasing. Blank spaces indicate no information reported.

		Biomass				Population structure			
		Assessment		NWFSC Survey		Assessment		NWFSC Survey	
Taxa	Stock	Depletion	5-yr trend	B final year	5-yr trend	Prop. mature	95% cum. age	Prop. mature	95% cum. lt.
Elasmobranch	Longnose skate	+	●	●	●	●	-	●	●
	Spiny dogfish	+	●			●	-		
	Spotted ratfish			●	●			+	●
Flatfishes	Arrowtooth flounder	+	+	●	+	-	-	-	-
	Dover sole	+	●			●	●		
	English sole	+	+			+	-	-	●
	Flathead sole			●	●			●	●
	Pacific sanddab			●	+			●	●
	Petrale sole	●	●			-	-		
	Rex sole	+	+					●	●
Rockfishes	Aurora	+	●			-	●		
	Black	+	+			-	-		
	Blackgill	●	●			-	-		
	Bocaccio	●	+			-	-		
	Canary	-	●			-	-		
	Chilipepper	+	●	●	●	●	-	-	+
	Cowcod	-	●			-	-		
	Darkblotched	●	+			-	-		
	Greenspotted	●	+			-	-		
	Greenstriped	+	+			●	-		
	Pacific Ocean perch	-	●			-	-		
	Redstripe			+	●			-	●
	Rougheye	+	+			●	●		
	Shaprchin	+	+						
	Shortbelly			●	●			●	●
	Splitnose	+	+			-	-		
	Stripetail			●	●			●	●
	Widow	+	+			●	-		
	Yelloweye	-	●			-	-		
	Yellowtail			●	●			●	●
Thornyheads	Longspine	+	+			●	●		
	Shortspine	+	-			●	●		
Roundfishes	Cabezon	+	+			-	-		
	Lingcod	+	+			-	-		
	Pacific Hake	+	+						
	Sablefish	●	-			●	●		

DETAILED REPORT

BACKGROUND - GROUNDFISH

Groundfish are generally defined as a community of fishes that are closely associated with the ocean bottom. In the CCLME, some of the better known species include the rockfishes (Scorpaenidae), flatfishes (Pleuronectidae and Bothidae), sculpins (Cottidae), Pacific hake (*Merluccius productus*), sablefish (*Anoplopoma fimbria*), greenlings and lingcod (Hexagrammidae), skates (Rajidae), and benthic sharks (PFMC 2008). Similar to most fishes, many groundfish species have a planktonic larval and young-of-year life history stage in which young fish inhabit surface waters and feed on a diet of zooplankton. After a few months in the plankton, most species settle to the bottom, generally moving to deeper waters and they age/grow. Groundfish vary across a wide range of trophic levels and inhabit all types of habitats (e.g., rocky, sandy, muddy, kelp) from the intertidal zone to the abyss and have generally variable recruitment, often mature late, and are long lived.

This community of fishes constitutes a large biomass in the CCLME and provides the economic engine for coastal communities in Washington, Oregon, and California. The Pacific Fishery Management Council (PFMC) manages a subset of groundfish species that are typically captured during fishing operations along the U.S. West Coast. Those species caught in the Pacific groundfish trawl fishery were worth approximately $40 million in 2009 (NOAA Press Release 2010). Thus, understanding how groundfish populations fare over time is of great interest to ecosystem managers and the coastal communities that derive much of their wealth from this assemblage of fishes.

INDICATOR SELECTION

Forty six potential indicators of groundfish population size and condition were evaluated using the ecological literature as a basis for their rankings (for detailed methods, see Levin and Schwing 2011, Cope et al. 2013). For population size, the top ranked indicators included: 1) biomass of groundfish relative to either the estimate of unfished biomass (when a stock assessment is available) or trends in the NWFSC annual groundfish trawl survey time series, and 2) the number of assessed species below management thresholds (Table GF 2). For groundfish population condition, the selected indicators were: 1) the proportion of the population mature (using ages or size in the absence of ages) and 2) the 95% cumulative age or length of the population (Table GF 2). These indicators are described briefly below.

Table GF2. Top indicators for groundfish population size (attribute 1) and population condition (attribute 2).

Attribute	Indicator	Definition and source of data	Time series	Sampling frequency
Population Size	Groundfish biomass	Tier 1: Modeled estimates of spawning biomass as measured by stock depletion from assessments beginning in 2007 as methods have been most stable during the 2007 – present. Tier 2: Relative biomass estimates as measured by the trend in the NWFSC annual survey	Tier 1: Variable by species Tier 2: 2003-2011	Annual estimate from both Tier 1 and 2 indicators
Population Size	Number of assessed species below management thresholds	Number of species below the PFMC overfished level and currently subject to rebuilding plans	N/A	Biannual rebuilding analyses
Population Condition	Population age (or size) structure	Tier 1: Modeled estimates of age structure (or size structure in the absence of age) from assessments beginning in 2007 as methods have been most stable during the 2007 – present. Tier 2: Age structure (or size structure in the absence of age) from the NWFSC annual survey	Tier 1: Variable by species Tier 2: 2003-2011	Annual estimate from both Tier 1 and 2 indicators

ATTRIBUTE 1 -POPULATION SIZE

CHANGE IN GROUNDFISH BIOMASS

Groundfish biomass was used relative to either the estimate of the unfished biomass from a stock assessment or trends in relative abundance from the survey time series (stock depletion):

a. Tier 1: Modeled estimates of stock depletion based on estimates of spawning biomass from assessments beginning in 2007 as earlier assessments are out of date.

b. Tier 2: Trends in stock depletion based on relative biomass estimates from the NWFSC annual trawl survey.

NUMBER OF ASSESSED SPECIES BELOW MANAGEMENT THRESHOLDS

Two tiers are specified for biomass of groundfish as a measure of abundance. Stock assessments provide the best available estimates of spawning stock biomass and depletion, because they integrate all of the available data on each stock over the full exploitation history of each stock. In the absence of a stock assessment, the NWFSC annual survey relative biomass index provides the best information available to estimate trends in the stock size, albeit over a shorter time series in comparison to the stock assessments. As stock assessments are generally updated on a 2-6 year cycle, for stock assessments that do not extend beyond 2007 the IEA is providing both the time series of spawning stock biomass from the assessment as well as the trend in biomass from the survey. As hake cannot be monitored for trends via the NWFSC annual bottom trawl survey due to likely annual changes in availability to the survey gear, the hake acoustic relative survey biomass is used as an alternative. The number of species below management thresholds was chosen because it is an easy measure of species or stocks that have typically been doing poorly in the past, but we recognize that documents (Miller et al. 2009) already exist that communicate this information. Thus this indicator may not be necessary in a final status report of the CCLME.

ATTRIBUTE 2 - POPULATION CONDITION

METRICS OF POPULATION AGE (OR SIZE IN THE ABSENCE OF AGE) STRUCTURE

a. Tier 1: Modeled estimates of age structure (or size structure in the absence of age) from assessments beginning in 2007 as earlier assessments are out of date.

b. Tier 2: Age structure (or size structure in the absence of age) from the NWFSC annual survey

These indicators are among the top indicators evaluated. Rebuilding timeline was not chosen as one of the final indicators because it is only available for species which have been formally considered overfished; thus it is only useful for a small number of species that are already below the target reference point. Using age structure accounts for many of the ecological processes that would affect age at maturity, so age at maturity is eliminated from the final indicator suite. Where available age structure is used as the indicator; however, size structure has been used in lieu of age structure where age data are not available. Size structure was not in the top quartile for population condition indicators, but it is the top-ranked indicator in the second quartile.

POPULATION AGE OR SIZE STRUCTURE

The mean age or size of all species caught in either fishery-independent surveys, fishery-dependent surveys, or landings is thought to be a useful and simple indicator to evaluate the overall effects of fishing (e.g., changes in rates of mortality) on an ecosystem (Fulton et al. 2005, Link 2005, Coll et al. 2009). Age and size-based metrics respond to fishing impacts because age and body size determines the vulnerability of individuals, populations, and communities (Jennings and Dulvy 2005). Others contend that there are very few examples where length-based analysis leads to useful management advice, in part because of the need for age and gear selectivity information, and because size related changes in distribution will influence data (Hilborn and Walters 1992). Additionally, older individuals tend to be more fecund and some fish species produce larvae that have a higher survival rates than larvae from younger fish (Berkeley 2004, Bobko and Berkeley 2004). Age and size based metrics are thought to better support medium-term rather than year-to-year management evaluation, because the response to management actions often cannot be quantitatively interpreted for contributing causal factors without extensive additional research (Jennings and Dulvy 2005).

Fish population age and size structure has been linked to scientifically defined reference points or progress targets. Some have based these on a decline in mean size of greater than 30% (warning or precautionary threshold) or greater than 50% (limiting reference point), the latter of which was chosen because it corresponds to an observed doubling in the time series of length after fishing has decreased (Link 2005). Others suggest that practical issues currently preclude the development and adoption of firm reference points for size-based indicators, although an appropriate target would be a reference direction that is consistent with a decline in the overall human impacts of fishing on the community, and thereby on the ecosystem (Jennings and Dulvy 2005). Similar reference points could be defined for mean population age.

The principal attraction of size-based metrics is the widespread availability of species size and abundance data collected during ongoing monitoring programs (Jennings and Dulvy 2005). Many monitoring programs collect a more limited but potentially more informative set of age data. The AFSC triennial survey and NWFSC annual survey have collected size data from a large array of species, and age data from a more limited set of species. The NWFSC annual survey collects up to 100 length measurements, sex determinations, and individual weights, and up to 25 age structures per trawl haul for key species, and more recently for all groundfish species of management concern (Keller et al. 2008). There are well recognized gear-selectivity issues associated with age and size data (Hilborn and Walters 1992) and ideally indicators should be calculated for age and size classes that are well selected by the gear. Fish population age and size structure has been used as an indicator in a variety of other ecosystems, including the Celtic Sea (Blanchard et al. 2005), northeastern U.S. continental shelf (Link and Brodziak 2002), and eastern Bering Sea (AFSC 2009).

STATUS AND TRENDS: GROUNDFISH

MAJOR FINDINGS

Stock status (based on biomass trends) and population demographic condition (as measured by proportion mature and of maximum age or size) were summarized for 36 groundfish species (Table GF3). Most assessed groundfishes were above the biomass limit reference point, and are thus not overfished (Figure GF2). The three assessed stocks currently in an overfished state are all rockfishes. All assessed groundfishes are below their target catch, thus overfishing is not occurring in these stocks. Regarding population condition measures, age or length structure tended to show more changes, usually declines, over time than proportion mature. Non-elasmobranch groundfishes tended to see the most changes over time in both measures, with rockfishes being most sensitive to demographic changes.

Table GF3. List of groundfish for which indicators were calculated. This list is composed of species in assemblages identified in Cope and Haltuch (2012), species with quantitative stock assessments completed from 2007-2013, and species that are well surveyed by the NWFSC annual trawl survey. Note that due to limited data availability, yelloweye rockfish would be removed from this species list without the results of a current stock assessment. Pacific hake would also be removed from this list without a current stock assessment because the trawl survey data alone are subject to changes in hake availability over time. However, as hake is currently assessed every year, hake should remain on the species list.

Species	Scientific name	Assessment Years
Longnose skate	*Raja rhina*	2007
Spiny dogfish	*Squalus acanthias*	2011
Spotted Ratfish	*Hydrolagus colliei*	
Arrowtooth flounder	*Atheresthes stomias*	2007
Dover sole	*Microstomus pacificus*	2011
English sole	*Parophrys vetulus*	2013
Flathead sole	*Hippoglossoides elassodon*	
Pacific Sanddab	*Citharichthys sordidus*	2013
Petrale sole	*Eopsetta jordani*	2009, 2011, 2013
Rex sole	*Glyptocephalus zachirus*	
Aurora rockfish	*Sebastes aurora*	2013
Black rockfish	*Sebastes melanops*	2007
Blackgill rockfish	*Sebastes melanostomus*	2011
Bocaccio rockfish	*Sebastes paucispinis*	2007, 2009, 2011, 2013
Canary rockfish	*Sebastes pinniger*	2007, 2009, 2011
Chilipepper rockfish	*Sebastes goodei*	2007
Cowcod	*Sebastes levis*	2013
Darkblotched rockfish	*Sebastes crameri*	2007, 2009, 2011, 2013
Greenspotted rockfish	*Sebastes chlorostictus*	2011
Greenstriped rockfish	*Sebastes elongatus*	2009
Pacific Ocean Perch	*Sebastes alutus*	2007, 2009, 2011
Redstripe rockfish	*Sebastes proriger*	
Rougheye rockfish	*Sebastes aleutianus*	2013
Sharpchin rockfish	*Sebastes zacentrus*	
Shortbelly rockfish	*Sebastes jordani*	2007
Splitnose rockfish	*Sebastes diploproa*	2009
Stripetail rockfish	*Sebastes saxicola*	
Widow rockfish	*Sebastes entomelas*	2007, 2009, 2011
Yelloweye rockfish	*Sebastes ruberrimus*	2007, 2009, 2011

Yellowtail rockfish	*Sebastes flavidus*	
Longspine Thornyhead	*Sebastolobus altivelis*	2013
Shortspine Thornyhead	*Sebastolobus alascanus*	2013
Cabezon	*Scorpaenichthys marmoratus*	
Lingcod	*Ophiodon elongatus*	2009
Pacific hake	*Merluccius productus*	2007-2013
Sablefish	*Anoplopoma fimbria*	2011

ATTRIBUTE 1 -POPULATION SIZE

SUMMARY

Biomass trajectories are a commonly used indicator of fisheries population dynamics and show the details of how population biomass has changed over time. Trends in the time series of abundance smooth out the dynamics to offer a directional summary of the changes. And while absolute biomass trends can be used, it is more common to consider the change in biomass relative to unfished condition, termed "depletion". A stock in considered more depleted when this ratio is relatively smaller, and less depleted when it is relatively larger. This ratio has particular meaning in groundfish management, where status reference points are based on depletion. For groundfishes other than flatfishes, the target depletion is 40% of unfished levels and the limit reference point (the value under which stocks are considered overfished) is 25% unfished levels. For the flatfishes, the target and limit reference points are 25% and 12.5%, respectively. All subsequent biomass measures are the mature female biomass, also called "spawning biomass", which is the commonly used biomass metric of age-structured stock assessments.

Ideally one would be able to census a population over a long period of time to get a direct measure of stock status for that period. Such detailed population information is not available for any Pacific coast groundfishes, so the next best source of status information is to use the population biomass estimates from age-structured stock assessments. Age-structured stock assessments combined fishery removals, abundance indices, size composition data, and life history information to reconstruct an estimation of how the population biomass changed over time. Barring the availability of stock assessment information, trends in indices of abundance as measured by a fishery-independent survey (specifically, the annual groundfish trawl survey conducted by the Northwest Fisheries Science Center since 2003) were considered. Of the 90+ groundfish species in the groundfish Fishery Management plan, 36 species had either of these data sources available, and thus were considered for status determination. The current development of data-limited methods (e.g., Cope 2013) may allow more groundfishes to be included in this summary for future iterations of the IEA.

For the analysis of groundfish status, we considered stock assessments from 2007 to 2013 to derive relative biomass trajectories. This was available for 28 of the 36 groundfishes considered. For the remaining 8 stocks, NWFSC trawl survey indices of abundance were used. Stocks with assessments only up until 2007 were also supplemented with the results of the survey abundance. Because the survey indices are limited in temporal coverage, relative trends in abundance rather than depletion are used and the change in index trend compared to the average biomass value and variance over the last 5 years are used instead of depletion reference points. Current population dynamics in the relative biomass trajectories were also evaluated for the last 5 years of the time series. Groundfish stocks were considered in 5 major groups: 1) Elasmobranchs, 2) Flatfishes, 3) Rockfishes, 4) Thornyheads, and 5) Other or Roundfishes. Within the first three groups, depth was used to distinguish three additional ecological categories: 1) nearshore, 2) shelf, and 3) slope. In general, there are very few nearshore representatives given the lack of assessments of nearshore species and the inadequacy of the trawl survey to sample the nearshore environment, so this status analysis is mostly limited to shelf and slope species. Full time series are provided for each series, but the last 5 years are used to determine the most recent trends.

Overall, most assessed groundfishes are above the biomass limit reference point, and are thus not overfished (Figure GF2). The only assessed stocks currently below the overfished status reference point are all rockfishes. All assessed groundfishes are below their target catch, thus overfishing is not occurring in these stocks. Many of the stocks show biomass around or above the target reference point as well as stable or increasing in the short term (Table GF4).

ELASMOBRANCHS (FIGURES GF3-GF6; TABLE GF4)

Assessed elasmobranch stocks are all above target depletion levels, while all stocks presented show stable population dynamics over the last 5 years.

FLATFISHES (FIGURES GF7-GF14; TABLE GF4)

Two of the three assessed flatfishes were above the target depletion level with one between the target and limit status reference levels. All of the species showed either increasing or stable population dynamics over that past 5 years. The shelf stock represented were either above target and/or demonstrated stable dynamics over the last five years. There is some indication that rex sole is in a slightly downward trend over the last five years, but is currently within the stable limit.

ROCKFISHES (FIGURES GF15-GF35; TABLE GF4)

All categories of rockfishes show a similar pattern of historical declines with contemporary population increases. Black rockfish (Figure GF15) is the only representative of the nearshore rockfish complex, and it shows a recent increase with the population above the target level. Because of the diversity of life histories and fisheries interactions in the nearshore environment, black rockfish cannot be used as a proxy for the other species. The shelf species also show increasing or steady populations in recent years. A recent assessment of Cowcod; (Figure GF20) suggests that what was once thought to be a drastically reduced population was only moderately reduced, and is now increasing. Slope species, with generally higher longevities, show a variety of population responses and tend to have below targeted level status.

THORNYHEADS (FIGURES GF36-GF37; TABLE GF4)

Thornyheads, while a target of recent live-fish enterprises, have not demonstrated major declines from pre-exploitation levels, both species indicate relative biomass is well above target reference points.

ROUNDFISHES (FIGURES GF38-GF41; TABLE GF4)

The roundfishes category is an amalgam of species with very different life histories and adult habitat. The group tends to be at around the target biomass levels with increasing population trajectories, except for sablefish (Figure GF41), which is both below target and trending downward.

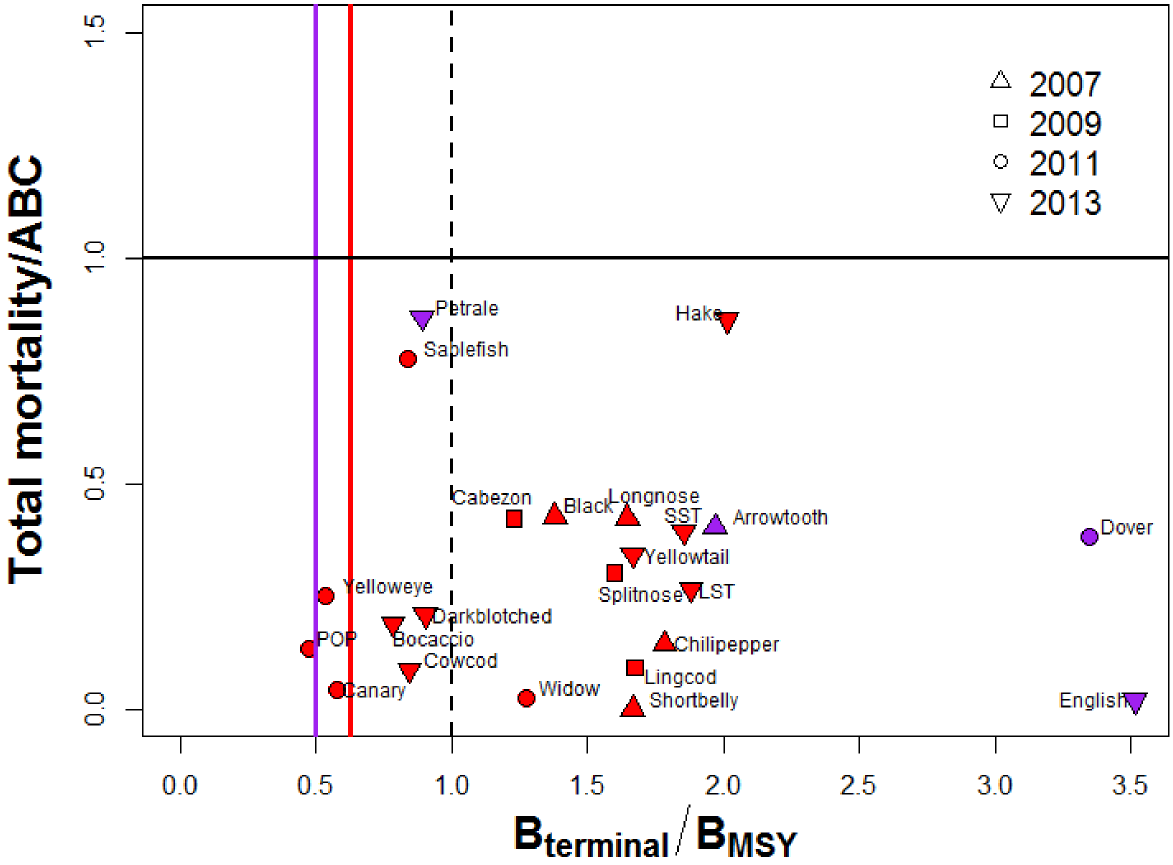

Figure GF2. Stock status plot relative to being overfished (x-axis) and overfishing (y-axis) for all species assessed since 2007. Vertical broken line indicates the target biomass reference point. Vertical solid line indicates the limit reference point indicating an overfished status (red for elasmobranchs, rockfishes, and roundfishes; purple for flatfishes). Horizontal blue line indicates overfishing wherein total mortality exceeds the allowable biological catch (ABC). Symbols indicate the terminal year of the assessment in which the reference points are determined.

Table GF4. Results for each stock evaluated for each of two status indicators: 1) Biomass and 2) Population structure. Two sources of information were used: 1) Stock assessments and 2) Northwest Fisheries Science Center (NWFSC) shelf-slope trawl survey, when assessments were not available, or older than 2007. “Depletion” refers to the ratio of spawning biomass in the most current year relative to initial population spawning biomass ; “5-year trend” is the trend in the last 5 years of the time series (details found in the text). “B final year” is the biomass value in the final year compared to the 5-year average. “Prop. mature” is proportional of the population mature relative to the beginning of the time series; “95% cum.” refers to the 95% cumulative age or length of the population relative to the beginning of the time series. +: above target limit or increasing over last five years; ●: between target and limit or stable; -: below limit or decreasing. Blank spaces indicate no information reported.

		Biomass				Population structure			
		Assessment		NWFSC Survey		Assessment		NWFSC Survey	
Taxa	Stock	Depletion	5-yr trend	B final year	5-yr trend	Prop. mature	95% cum. age	Prop. mature	95% cum. lt.
Elasmobranch	Longnose skate	+	●	●	●	●	-	●	●
	Spiny dogfish	+	●			●	-		
	Spotted ratfish			●	●			+	●
Flatfishes	Arrowtooth flounder	+	+	●	+	-	-	-	-
	Dover sole	+	●			●	●		
	English sole	+	+			+	-	-	●
	Flathead sole			●	●			●	●
	Pacific sanddab			●	+			●	●
	Petrale sole	●	●			-	-		
	Rex sole	+	+					●	●
Rockfishes	Aurora	+	●			-	●		
	Black	+	+			-	-		
	Blackgill	●	●			-	-		
	Bocaccio	●	+			-	-		
	Canary	-	●			-	-		
	Chilipepper	+	●	●	●	●	-	-	+
	Cowcod	-	●			-	-		
	Darkblotched	●	+			-	-		
	Greenspotted	●	+			-	-		
	Greenstriped	+	+			●	-		
	Pacific Ocean perch	-	●			-	-		
	Redstripe			+	●			-	●
	Rougheye	+	+			●	●		
	Shaprchin	+	+						
	Shortbelly			●	●			●	●
	Splitnose	+	+			-	-		
	Stripetail			●	●			●	●
	Widow	+	+			●	-		
	Yelloweye	-	●			-	-		
	Yellowtail			●	●			●	●
Thornyheads	Longspine	+	+			●	●		
	Shortspine	+	-			●	●		
Roundfishes	Cabezon	+	+			-	-		
	Lingcod	+	+			-	-		
	Pacific Hake	+	+						
	Sablefish	●	-			●	●		

SPECIFIC TIME SERIES

INTERPRETING BIOMASS TIME SERIES PLOTS

Green area is above the relative target spawning biomass, red is below the limit relative target spawning biomass, and yellow is between the target and limit values. Gray shaded area indicates the last 5 years. Significant population increases were defined as more than 1% per year, while significant decreases were more than -1% a year. No change was less than 1% either way per year. A 1% threshold was chosen arbitrarily and would lead to a minimum of a 10% increase in a decade's time. If an assessment was done in 2007 or not available, current survey trends were provided when available. Because the survey data time series is significantly shorter than the stock status time series, two different measures of relative change and trend are used. The mean (solid line) and +/- 1 standard deviation (broken lines) for the full trawl survey time series is calculated and shown in green. A linear trend is fit to the last five years and the change in biomass over that trend is compared to 1 standard deviation from the mean. The average biomass for the last 5 years is also calculated and compared to the full time series mean.

ELASMOBRANCHS (N=3)

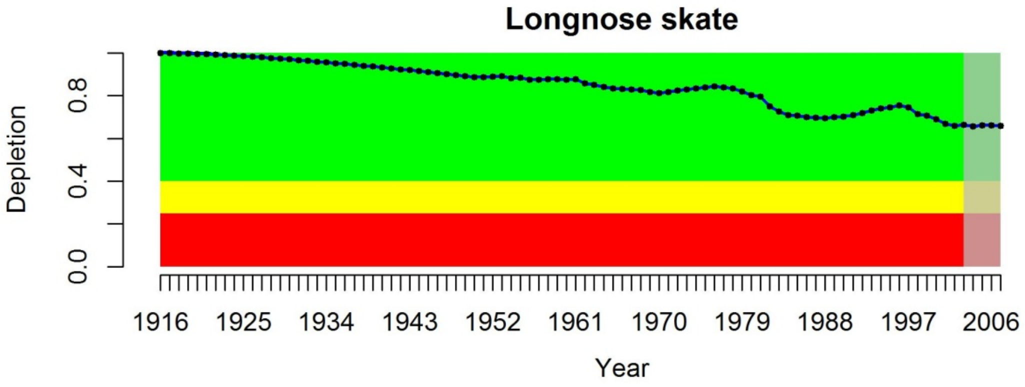

Figure GF3. Relative abundance trajectory 1916-2007 for longnose skate.

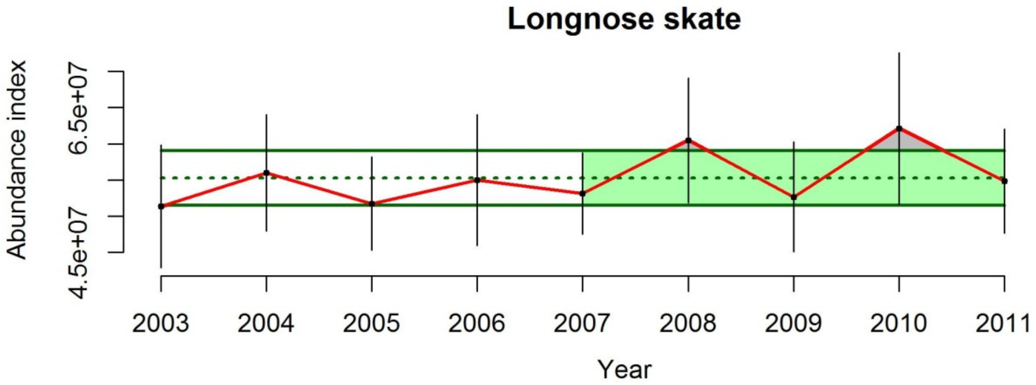

Figure GF4. Trawl survey design-based estimates of longnose skate biomass for years 2003-2011.

Summary: Longnose skate has shown a slow decline over the length of the time series, but with stable population dynamics in the most recent 5 years. Relative biomass appears to have maintained a level above the target biomass in all years.

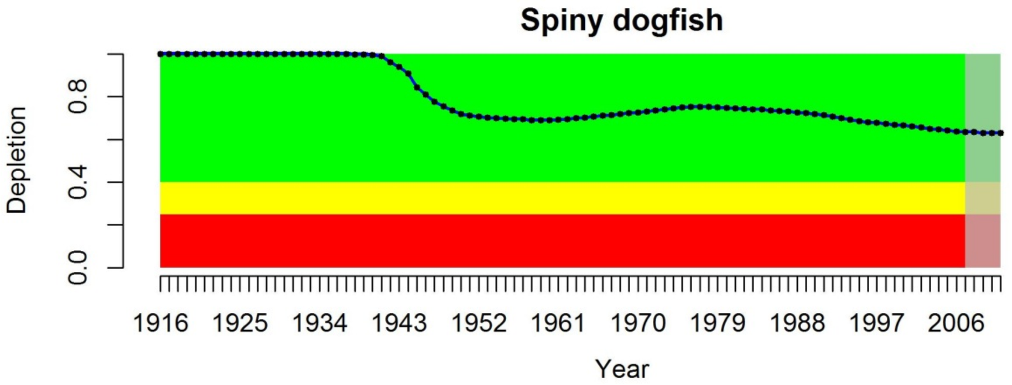

Figure GF5. Relative abundance trajectory 1916-2011 for spiny dogfish.

Summary: After an initial steep decline in the 1940s, relative spiny dogfish abundance has slowed in decline or remained stable in recent years. The population appears to have been above the target relative biomass reference point in all years.

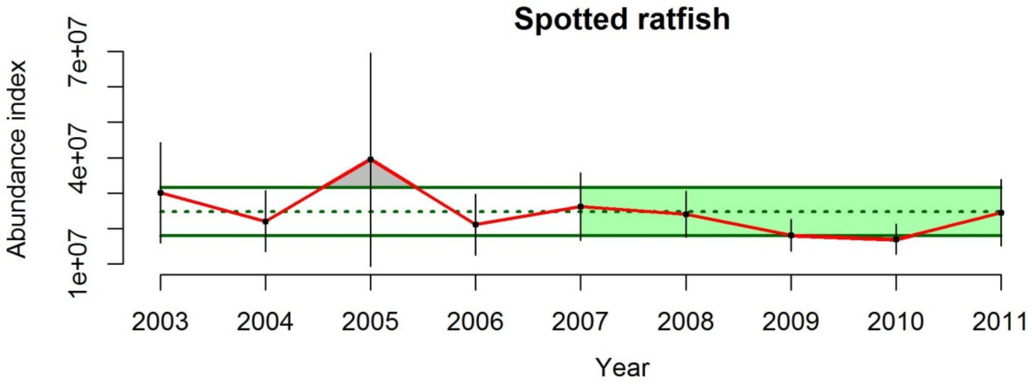

Figure GF6. Trawl survey design-based estimates of spotted ratfish biomass for years 2003-2011.

Summary: No stock assessment for spotted ratfish is available, so no baseline information can be interpreted for this stock at this time. For the most recent years, spotted ratfish appear to have a stable population abundance.

FLATFISHES (N=7)

Shelf

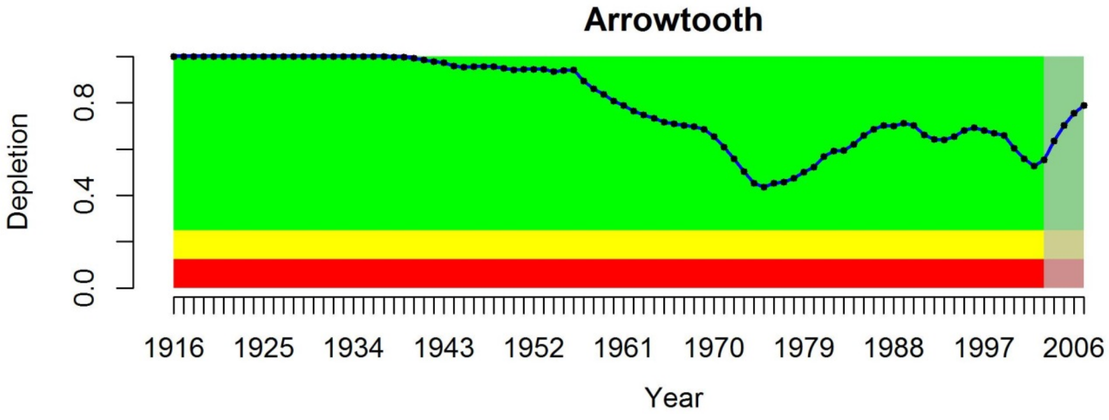

Figure GF7. Relative abundance trajectory 1916-2007 for arrowtooth flounder.

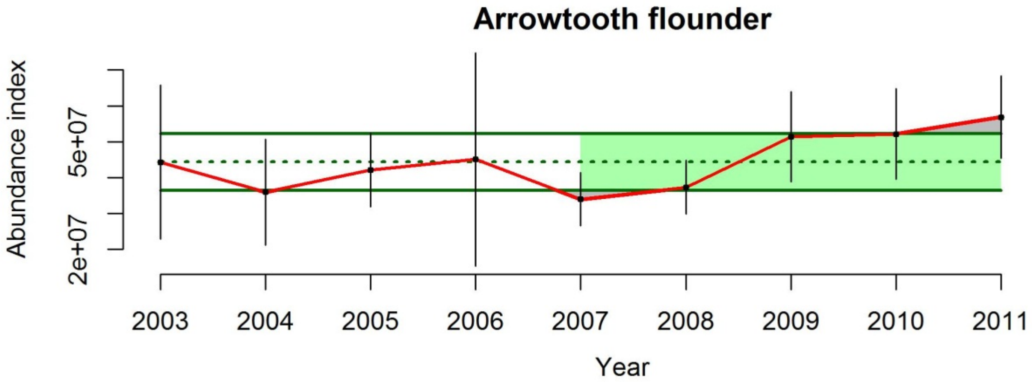

Figure GF8. Trawl survey design-based estimates of arrowtooth flounder biomass for years 2003-2011.

Summary: Arrowtooth flounder demonstrated its greatest decline from the 1950s to the 1970s. It has since increased and continues to show increase in the most recent years. At no point has it been recorded to have gone below the target relative biomass.

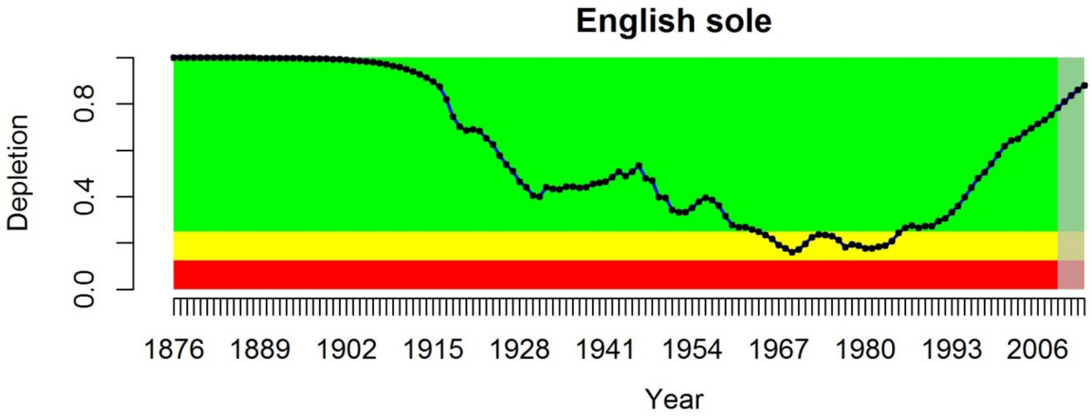

Figure GF9. Relative abundance trajectory 1876-2013 for English sole.

Summary: English sole demonstrated large declines in the early 20th-century, at times dropping below the target relative biomass level. Recent years indicate a large increase, with an increasing trend in the last 5 years.

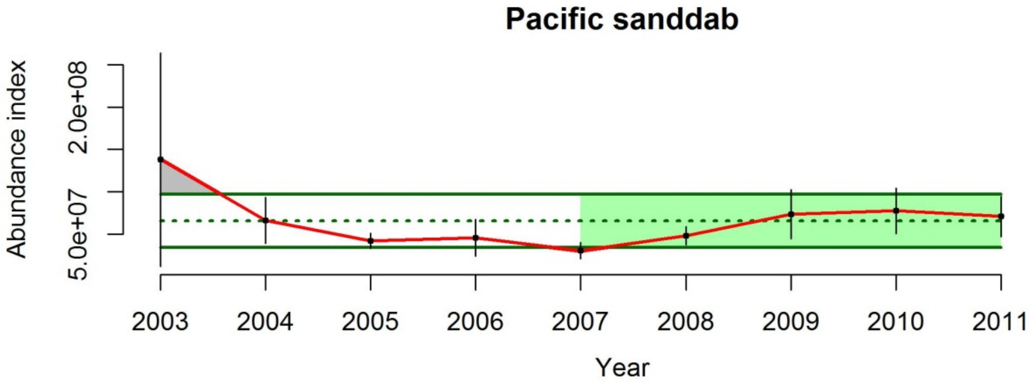

Figure GF10. Trawl survey design-based estimates of English sole biomass for years 2003-2011.

Summary: No stock assessment is available for Pacific sanddab, so no baseline information on abundance exists. Recent years indicate an increasing trend in survey abundance.

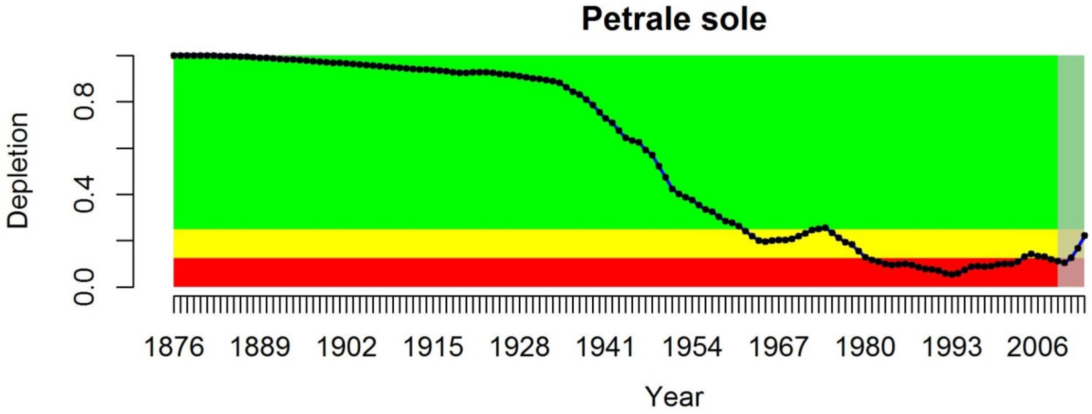

Figure GF11. Relative abundance trajectory 1876-2013 for Petrale sole.

Summary: Petrale sole abundance dropped sharply from the late 1930s to the 1950s, with a steady decline through the 1990s, bringing the population below the relative biomass limit. Recent years have shown an uptick with a steady population over the last 5 years.

Slope

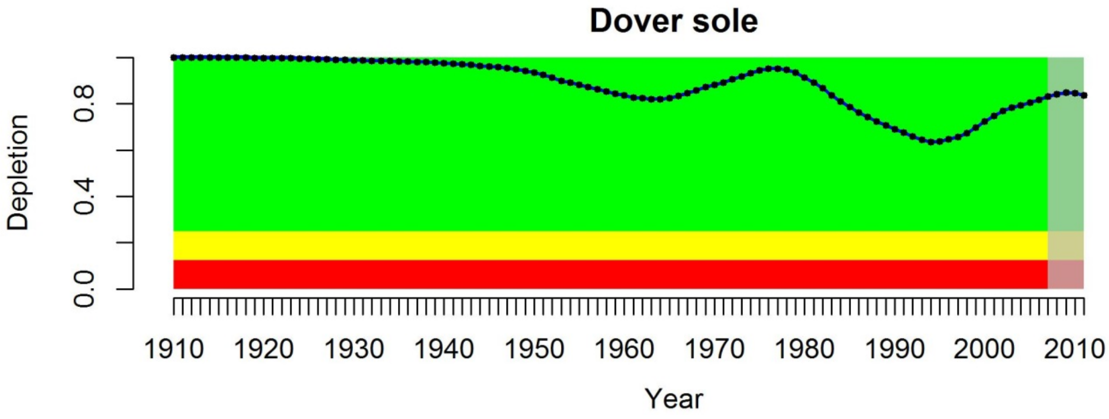

Figure GF12. Relative abundance trajectory 1910-2011 for Dover sole.

Summary: Dover sole populations have shown only slight declines over the time series. Relative biomass has stayed above target levels in all years and is steady over the last 5 years.

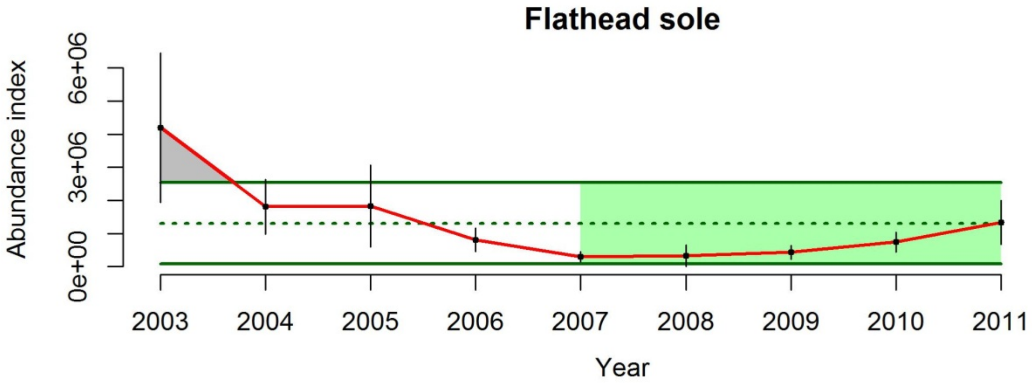

Figure GF13. Trawl survey design-based estimates of flathead sole biomass for years 2003-2011.

Summary: No flathead sole assessment is available, so no baseline information on abundance exists. Recent years indicate a steady trend in survey abundance.

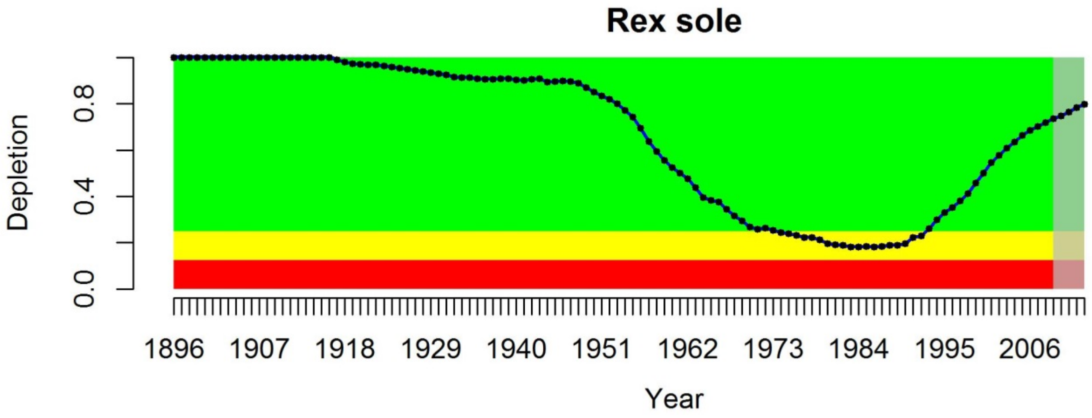

Figure GF14. Relative abundance trajectory 1896-2013 for rex sole.

Summary: Rex sole demonstrated large declines in the 1970s, even dropping below the limit reference point for relative biomass level. Recent years indicate a large increase, with an increasing trend in the last 5 years.

ROCKFISHES (N=21)

Nearshore

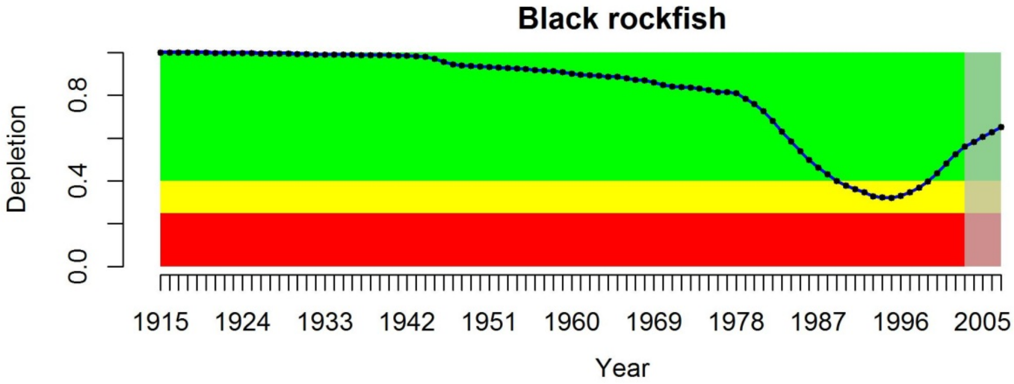

Figure GF15. Relative abundance trajectory 1916-2009 for black rockfish.

Summary: Black rockfish shows a consistent decline until the late 1990s, where in the population starts to grow. Relative biomass dropped below the target relative biomass level for most of the 1990s. Recent years show an increasing trend in population abundance.

Shelf

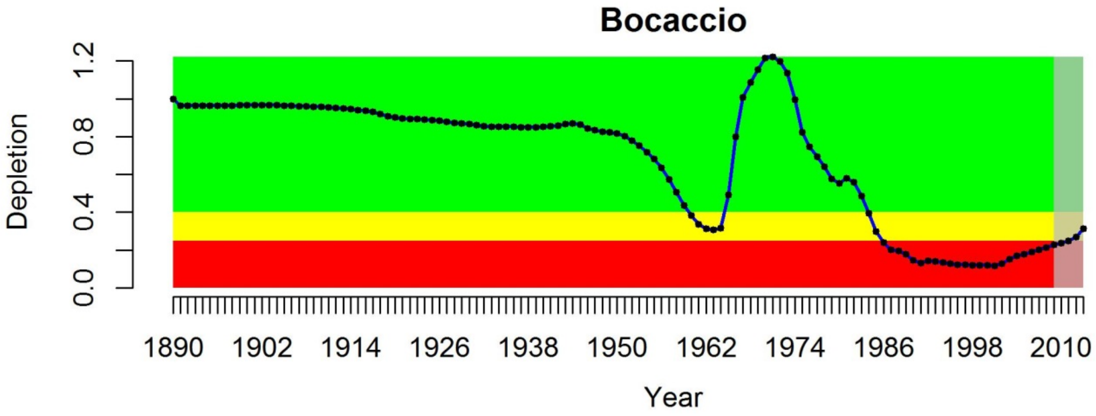

Figure GF16. Relative abundance trajectory 1890-2011 for bocaccio.

Summary: Bocaccio abundance has been highly dynamic over the time series, dropping to levels below the relative biomass limit in recent years. The population trend over that last 5 years is increasing.

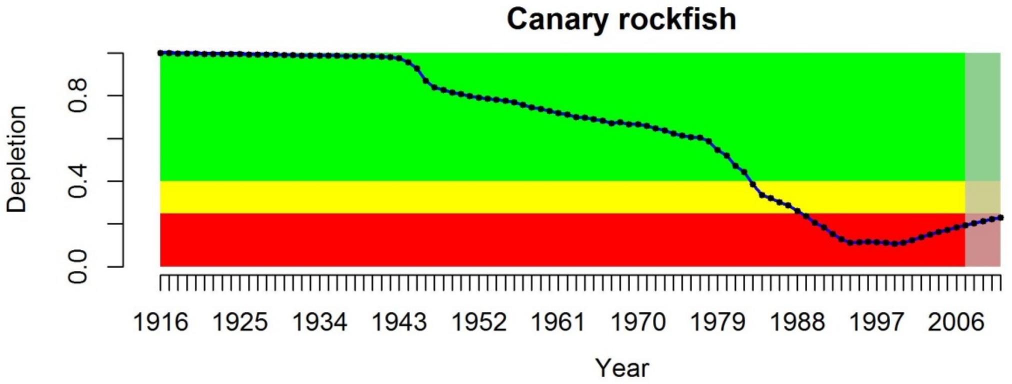

Figure GF17. Relative abundance trajectory 1916-2011 for canary rockfish.

Summary: Large declines in population abundance have been witnessed in canary rockfish, enough to drop the relative abundance below the relative biomass limit. Recent years show very slow growth and an overall stable population.

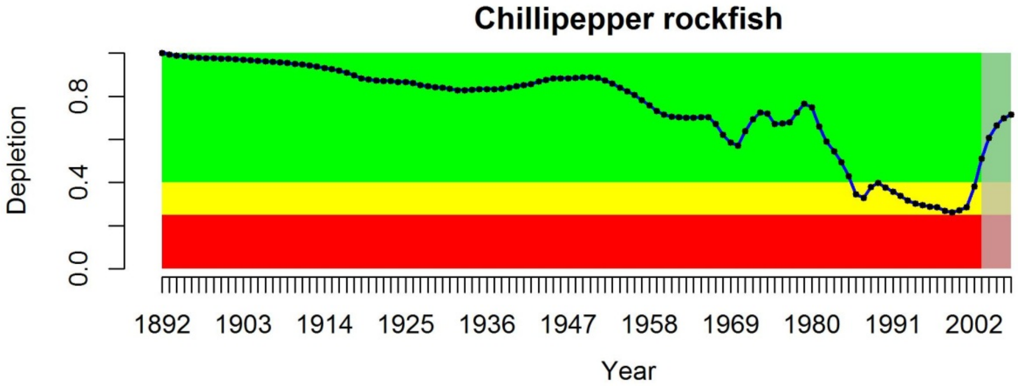

Figure GF18. Relative abundance trajectory 1892-2011 for chilipepper.

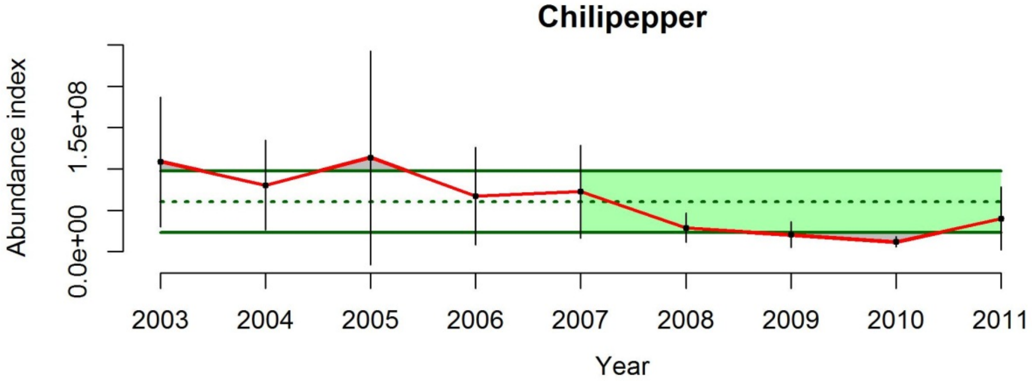

Figure GF19. Trawl survey design-based estimates of chilipepper biomass for years 2003-2011.

Summary: Chilipepper biomass declined below the relative biomass target limit after 1980, then increased substantially in the 2000s. The short-term trawl survey information indicates a stable population in recent years.

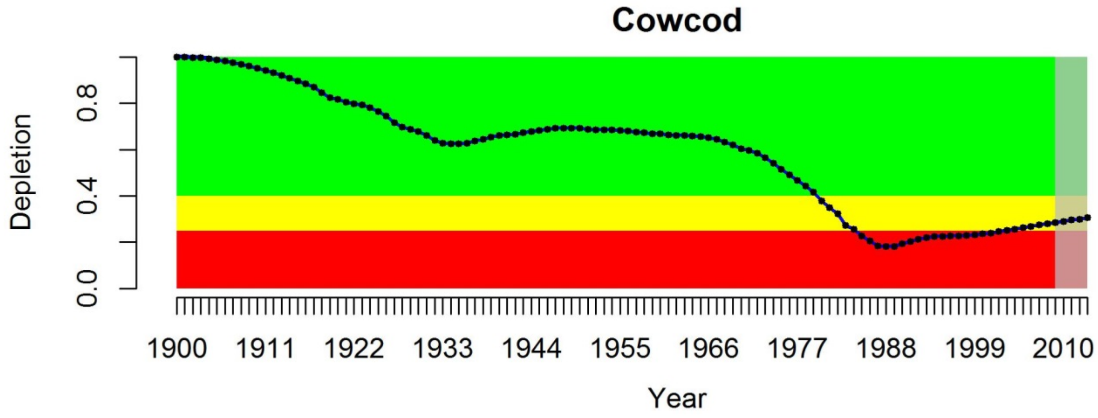

Figure GF20. Relative abundance trajectory 1900-2013 for cowcod.

Summary: The view of cowcod relative biomass has changed greatly since the past assessment in 2011. Once thought to be well below the limit reference point with very slow growth, the new status trends show a stock in the precautionary zone of relative biomass, with an increasing trend over the last five years, and only a small historical period of being below the relative biomass limit reference point.

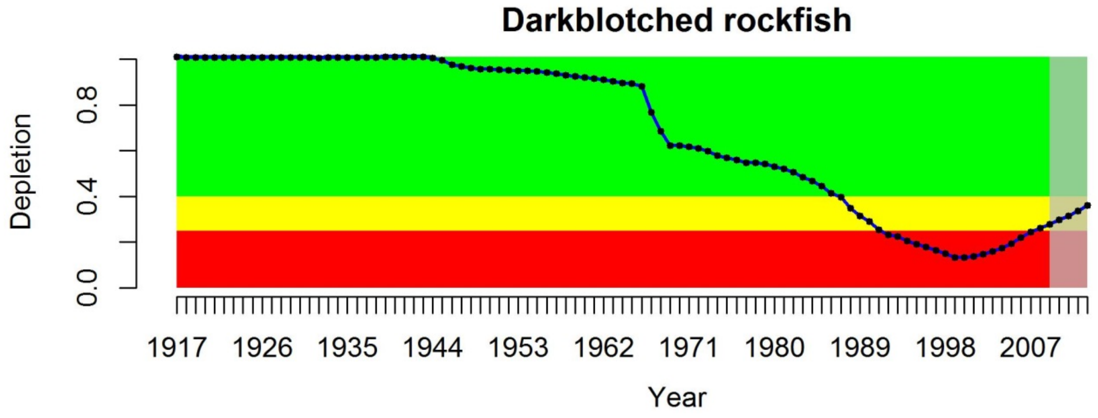

Figure GF21. Relative abundance trajectory 1910-2013 for darkblotched rockfish.

Summary: Darkblotched rockfish showed historical declines in population below relative biomass limits, but recent years show population increase above the limit, nearing the rebuilding target relative biomass.

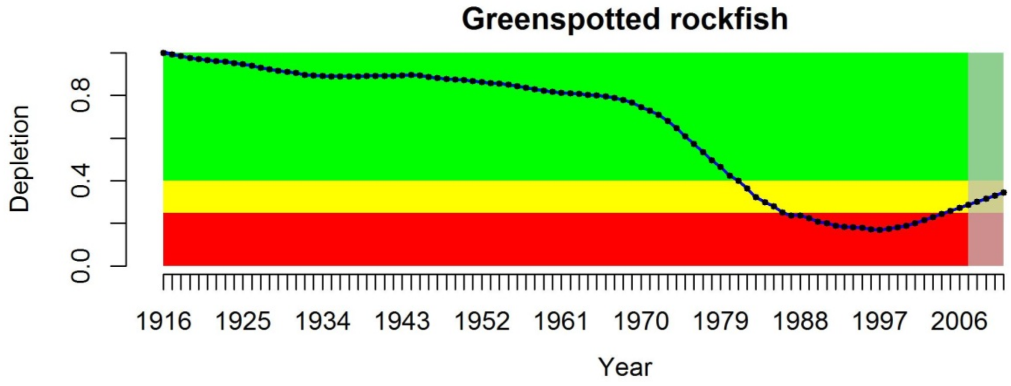

Figure GF22. Relative abundance trajectory 1916-2011 for greenspotted rockfish.

Summary: Greenspotted rockfish abundance historically dropped below the limit reference point, but is recently increasing and near the target relative biomass level.

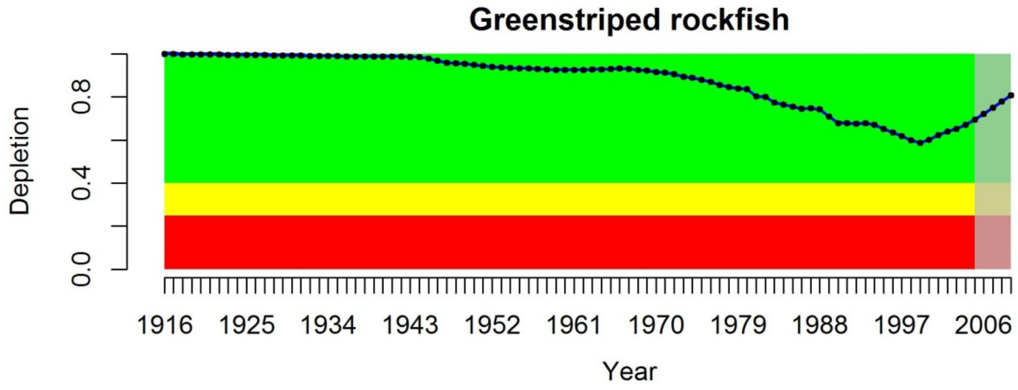

Figure GF23. Relative abundance trajectory 1910-2011 for greenstriped rockfish.

Summary: Greenstriped rockfish has stayed above the target relative biomass level with increasing biomass in the most recent years.

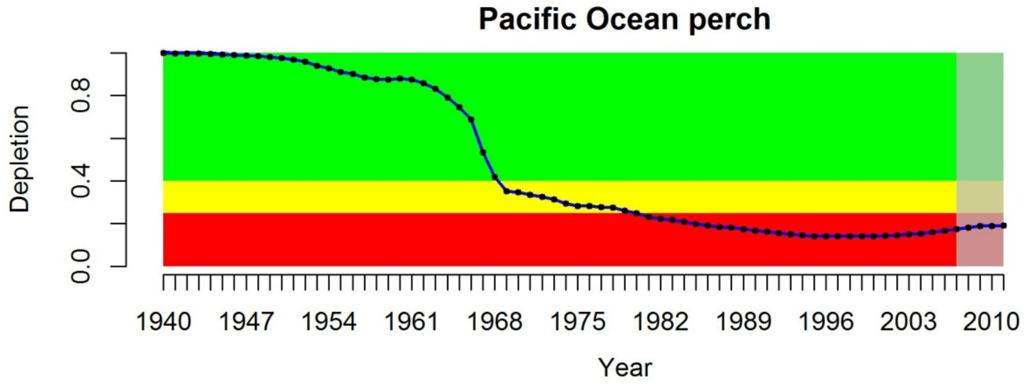

Figure GF24. Relative abundance trajectory 1940-2011 for Pacific Ocean perch.

Summary: Pacific Ocean perch biomass has shown a large historical decline and is currently below the relative biomass limit, though the population is steady in the most recent years.

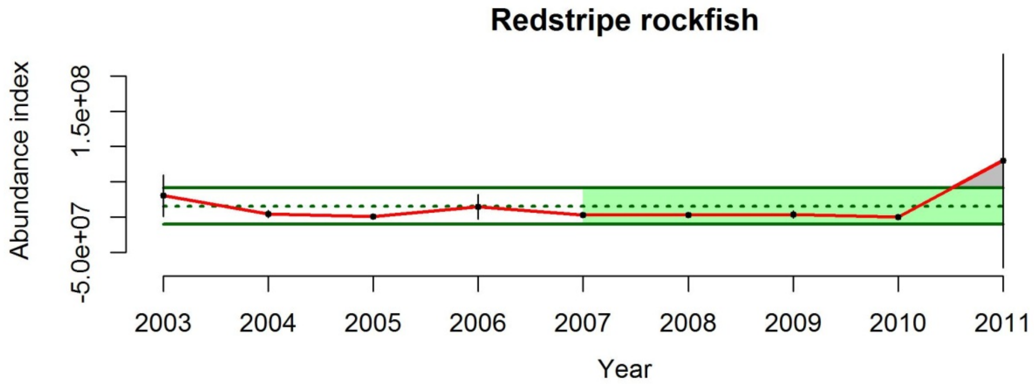

Figure GF25. Trawl survey design-based estimates of restripe rockfish biomass for years 2003-2011.

Summary: No redstripe rockfish assessment is available, so no baseline information on abundance exists. Recent years indicate a stable trend in survey abundance (the last relatively high point has large uncertainty).

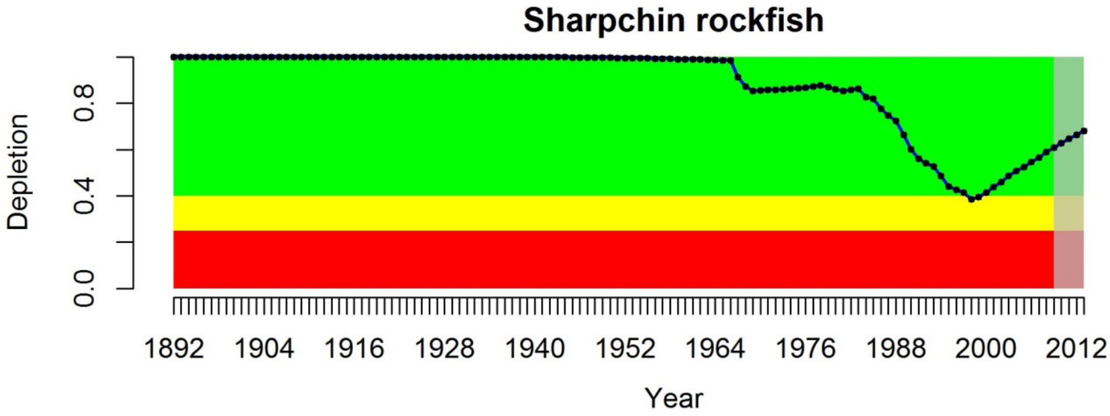

Figure GF26. Relative abundance trajectory 1892-2013 for sharpchin rockfish.

Summary: Sharpchin rockfish has not had a targeted fishery, but demonstrated historical declines in the 1980s. The population has subsequently increased and is well above the target relative biomass.

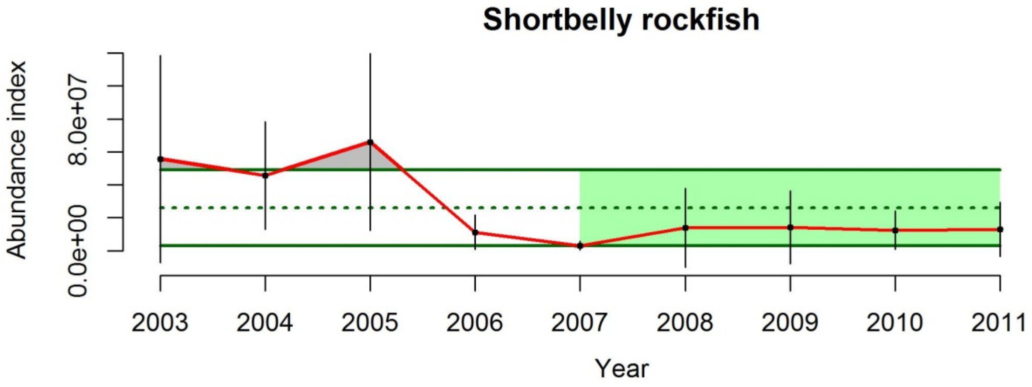

Figure GF27. Trawl survey design-based estimates of shortbelly rockfish biomass for years 2003-2011.

Summary: No shortbelly rockfish assessment is available, so no baseline information on abundance exists. Recent years indicate a stable trend in survey abundance.

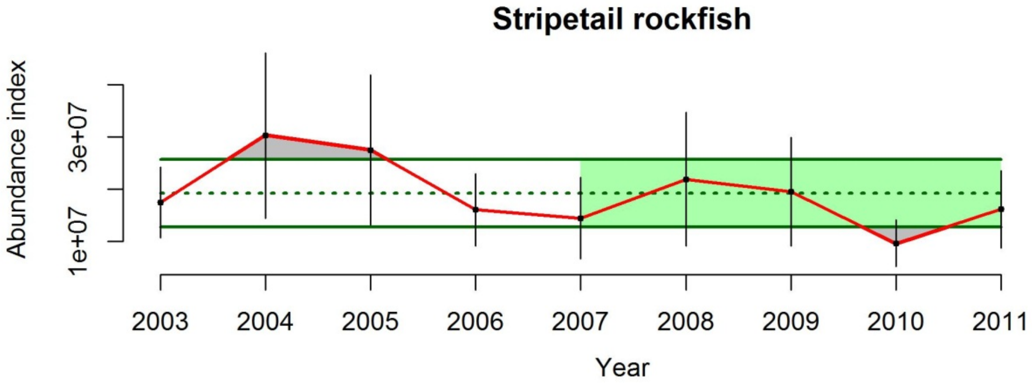

Figure GF28. Trawl survey design-based estimates of stripetail rockfish biomass for years 2003-2011.

Summary: No stripetail rockfish assessment is available, so no baseline information on abundance exists. Recent years indicate a stable trend in survey abundance.

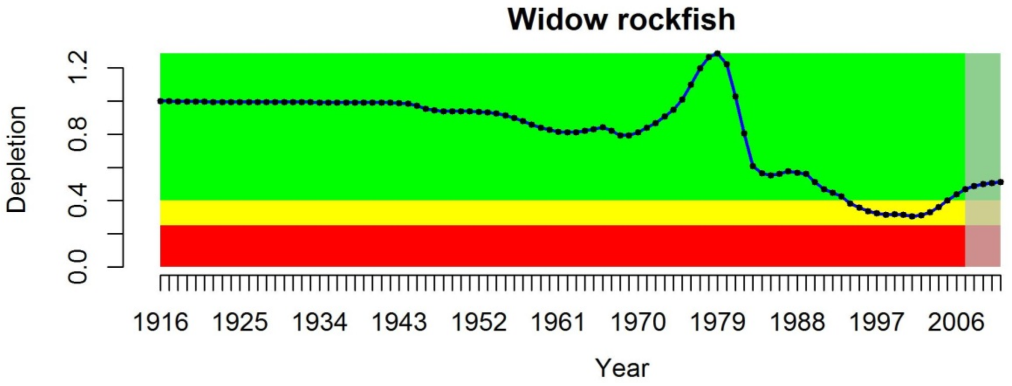

Figure GF29. Relative abundance trajectory 1916-2011 for widow rockfish.

Summary: Widow rockfish historically declined to below the target relative biomass level, but is currently increasing and is above the target.

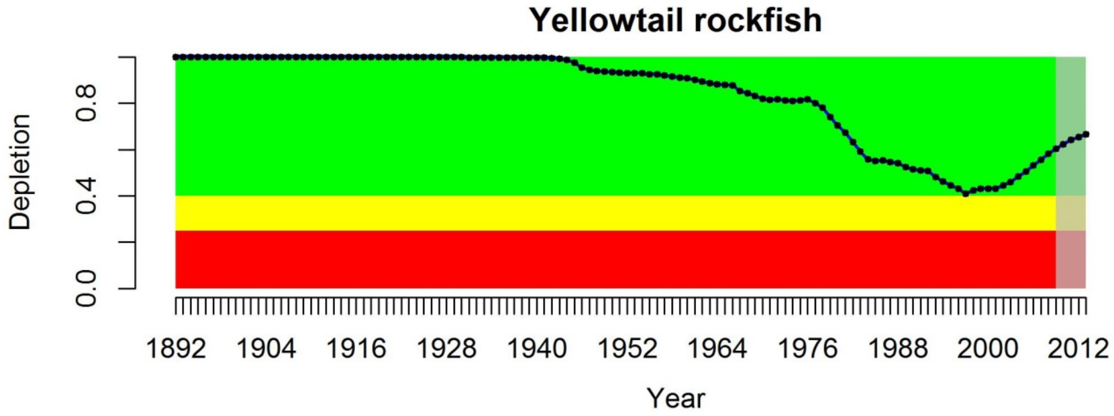

Figure GF30. Relative abundance trajectory 1891-2013 for yellowtail rockfish.

Summary: Yellowtail rockfish show similar temporal declines as other shelf rockfishes, but not the same extent. Yellowtail rockfish remain well above the relative target biomass levels and are increasing.

Slope

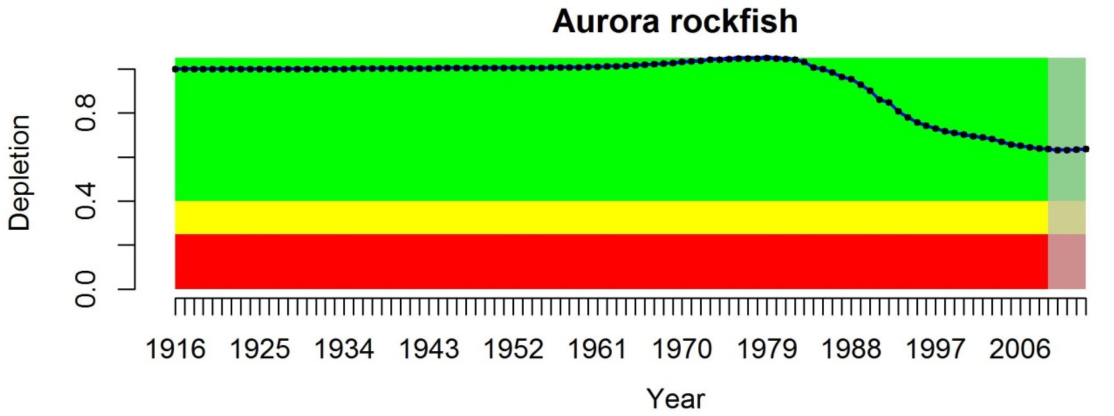

Figure GF31. Relative abundance trajectory 1916-2013 for aurora rockfish.

Summary: Aurora rockfish has not been a consistent or widespread fishery target and thus has shown little decline over the last century or removals. The biomass is steady and above the target relative biomass.

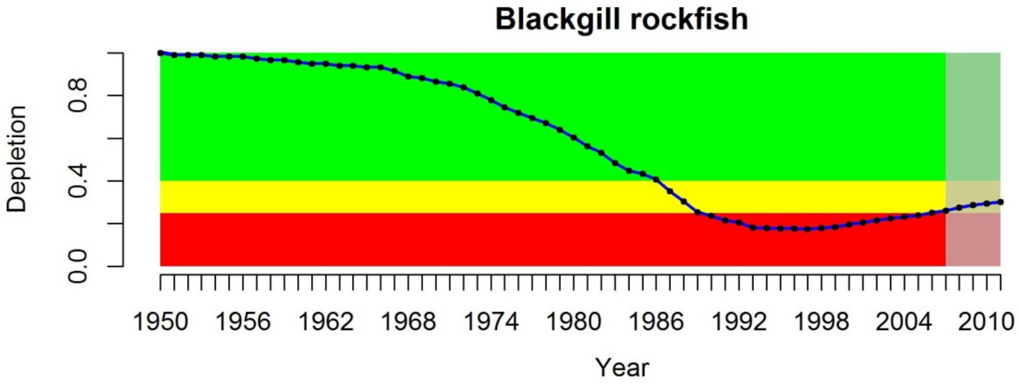

Figure GF32. Relative abundance trajectory 1950-2011 for blackgill rockfish.

Summary: Blackgill rockfish show a historical decline below the limit relative abundance reference point with a slight increase over the last 10 years. The last 5 years show a stable population.

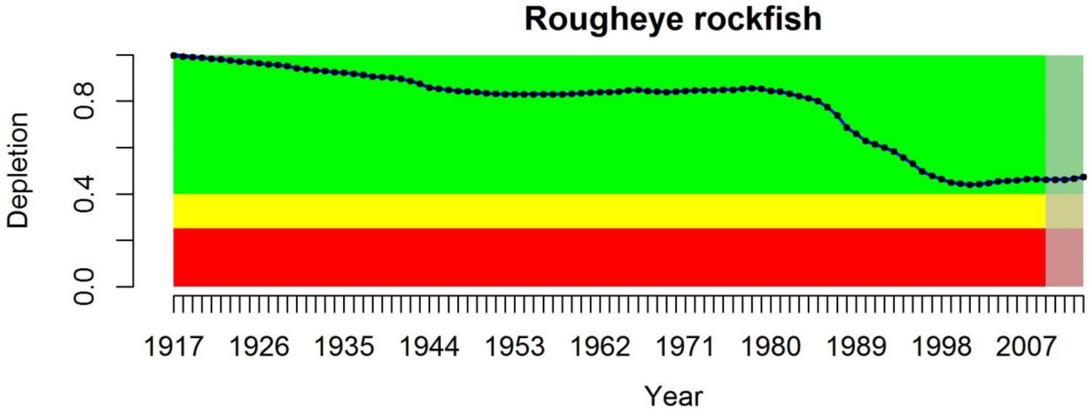

Figure GF33. Relative abundance trajectory 1917-2013 for rougheye rockfish.

Summary: Rougheye rockfish demonstrates the typical rockfish population trajectory, with declines in the 1980s and stabilizing to increasing trends in the most recent years. Assessment results indicate that the rougheye rockfish stock is near, but above the relative biomass target level.

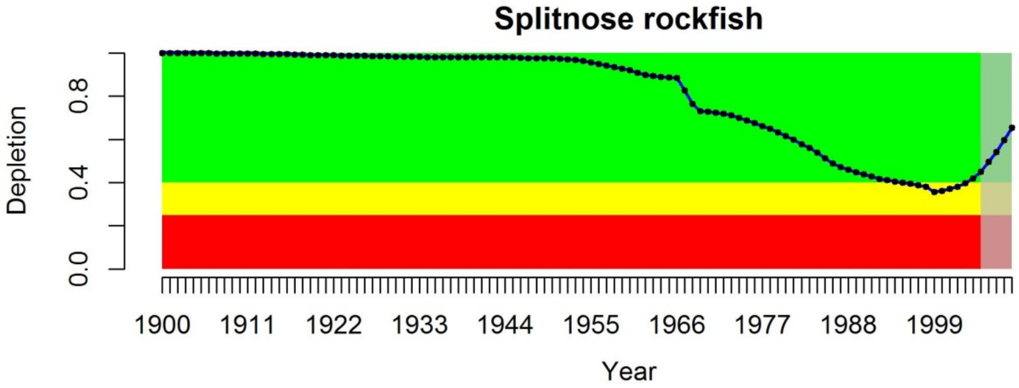

Figure GF34. Relative abundance trajectory 1910-2011 for splitnose rockfish.

Summary: The splitnose rockfish population declined to below the target relative biomass in the late 1990s, but is currently increasing.

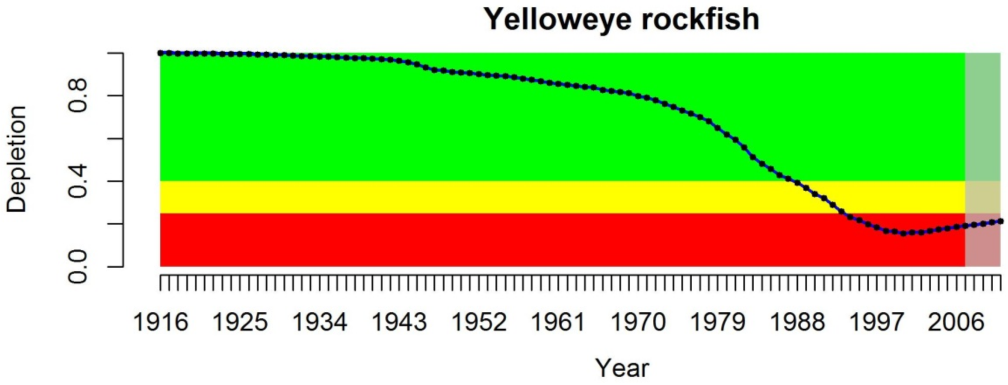

Figure GF35. Relative abundance trajectory 1916-2011 for yelloweye rockfish.

Summary: Yelloweye rockfish declined to below the limit relative biomass level and has stayed below since. Currently, the population is stable.

THORNYHEADS (N=2)

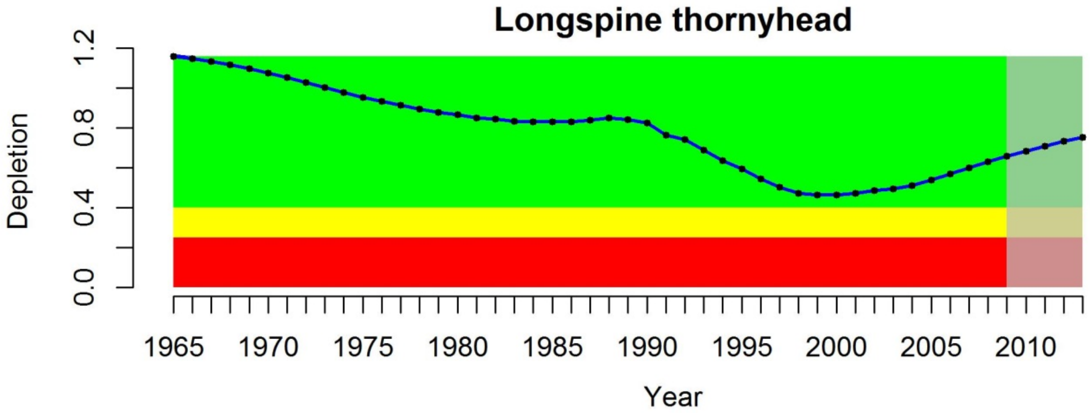

Figure GF36. Relative abundance trajectory 1965-2013 for longspine thornyhead.

Summary: Longspine thornyhead show initial declines, followed by several recent years of increasing relative population size. The population seems to have never dropped below the relative biomass target.

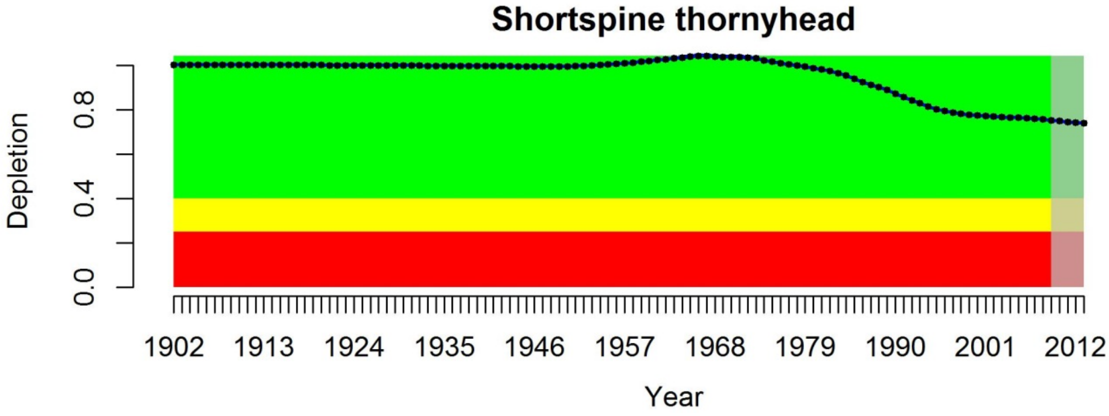

Figure GF37. Relative abundance trajectory 1902-2013 for shortspine thornyhead.

Summary: Longspine thornyhead show recent, but slow, population decline. The population seems to have never dropped below the relative biomass target.

ROUNDFISHES (N=4)

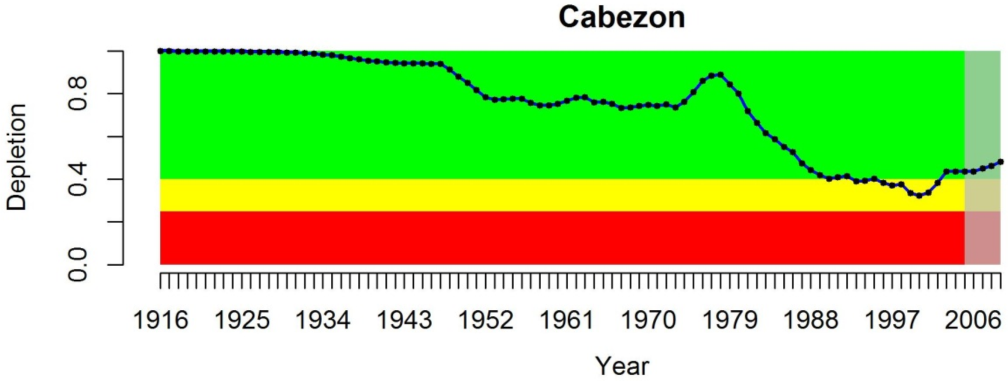

Figure GF38. Relative abundance trajectory 1910-2011 for cabezon.

Summary: Cabezon biomass had declined over the time series to below the relative biomass target level, but has since increased over the most recent years.

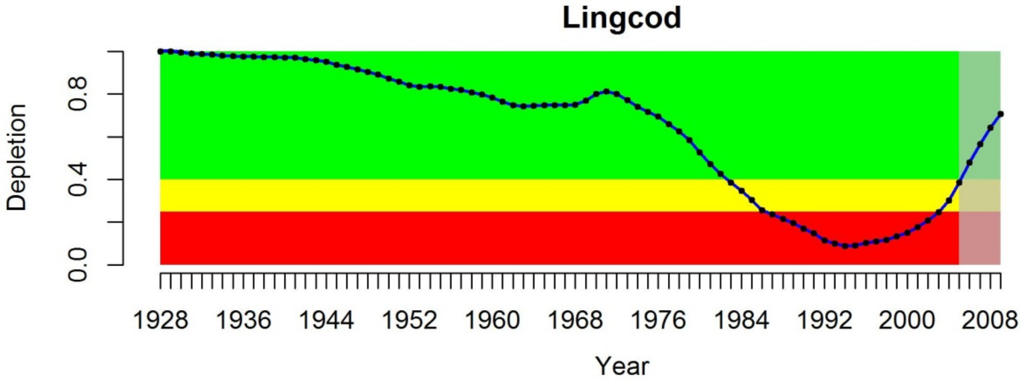

Figure GF39. Relative abundance trajectory 1910-2011 for lingcod.

Summary: Lingcod biomass had declined over the time series to below the relative biomass limit reference point, but has since increased over the most recent years.

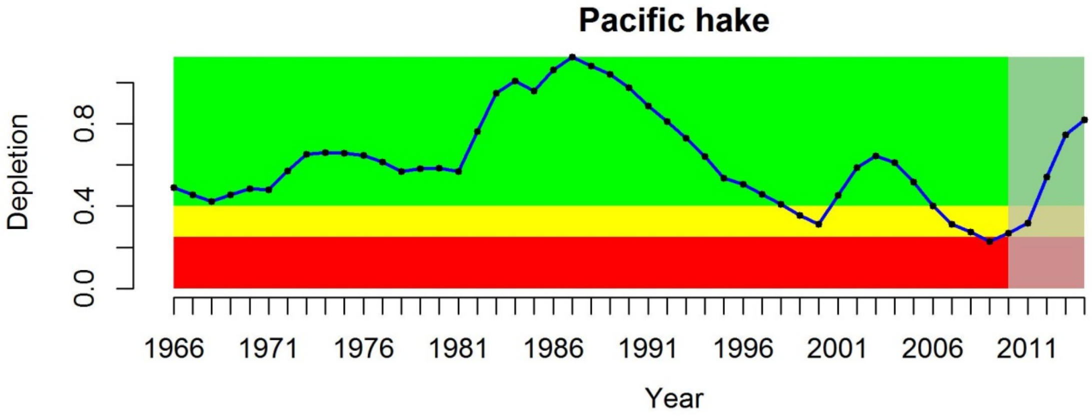

Figure GF40. Relative abundance trajectory 1910-2011 for Pacific hake.

Summary: Pacific hake biomass is very dynamic and is currently above the target relative biomass reference point with a recent increasing biomass trend.

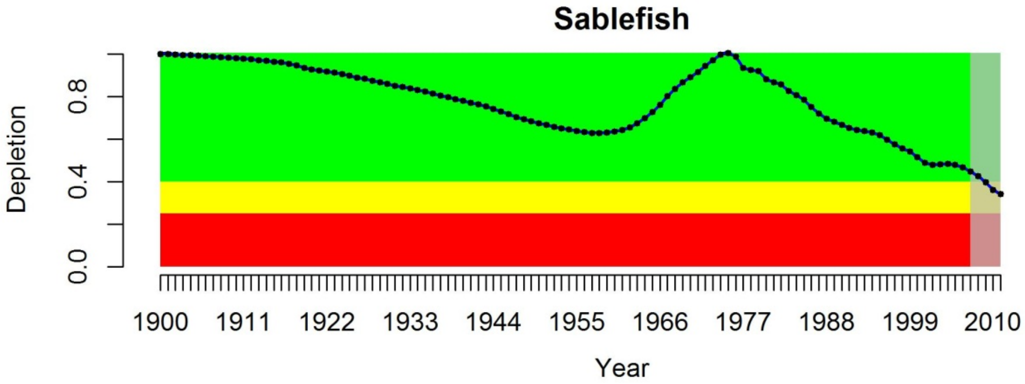

Figure GF41. Relative abundance trajectory 1910-2011 for sablefish.

Summary: Sablefish biomass is very dynamic and is currently below the target relative biomass reference point with a recent decreasing biomass trend.

ATTRIBUTE 2 - POPULATION CONDITION

SUMMARY

The first groundfish population condition indicator uses female mature biomass as a status indicator, but biomass is a broad term that obscures other important information (e.g. age and size compositions). In order to capture this additional population dimension, demographic structure of each stock is considered as another status indicator. Proportion maturity gives the percent of the population mature in a given year. The 95% age or length cumulative value indicates at which age or length 95% of the population is below, and thus is a measure of age/length truncation or expansion. All of the above values can be compared to the earliest value in the time series to indicate if they have changed over time. Female age and lengths are used exclusively to be comparable to the spawning biomass in the abundance trends indicator.

As with biomass, stock assessments are used as the primary source of information for maturity and age structure. If no stock assessment was available, trawl survey length compositions were used. Analyses of stocks with their most recent stock assessment in 2007 were also supplemented by the trawl length compositions. The same species grouping used in the abundance indicators are also used to organize stock results. Note that indicators of population age or size structure were not estimated for three species for the following reasons: Pacific hake length data were not deemed reliable, length data were not used in the sharpchin rockfish assessment, and no age/length data were available for shortbelly rockfish.

Overall, age or length structure tended to show more changes over time than proportion mature (Table GF4). Long-term time series comparisons generally showed declines in these indicators, whereas short-term comparisons demonstrated more stability (Table GF4), suggesting most change happened earlier in the fishery histories of these stocks. Non-elasmobranch groundfishes tended to show the most changes over time in both measures, with rockfishes being most sensitive to demographic changes (Table GF4). Though it is reasonable to expect these age/length-based indicators to be sensitive to yearly recruitment fluctuations, particularly large recruitments, changes in these indicators seemed more consistent with declines in spawning biomass, and thus deeper population structure changes, than recruitment variability.

ELASMOBRANCHS (FIGURES GF42-GF45; TABLE GF4)

Age or length structure showed little change in these elasmobranchs, but maturity did change in species with long time series. All measures were stable in the most recent years.

FLATFISHES (FIGURES GF46-GF54; TABLE GF4)

Flatfishes on the shelf showed decreases over time in both measures, while the deeper slope species showed little change over time in either measure.

ROCKFISHES (FIGURES GF55-GF74; TABLE GF4)

Rockfishes showed a general decline in both measures through time, regardless of the adult habitat. Chilipepper (Figure GF59) and rougheye (Figure GF72) are exceptions which show little change over the entire time series. Greenstriped (Figure GF63) and widow rockfish (Figure GF68) show contemporary measures have increased near initial conditions after historical declines. Stripetail (Figure GF67) and yellowtail rockfish (Figure GF68) show little change in the trawl survey lengths, but there is no historical baseline to interpret these values. Overall, rockfishes were the most sensitive species group to demographic changes.

THORNYHEADS (FIGURES GF75-GF76; TABLE GF4)

Thornyheads show little change in either population measure over the period of each stock assessment.

ROUNDFISHES (FIGURES GF77-GF79; TABLE GF4)

Two of three roundfishes (cabezon and lingcod, both shallow egg-layers with nest-guarding males) showed declines in both measures, whereas sablefish showed little change over time. Lingcod has shown recent increases in both measures.

SPECIFIC TIME SERIES

ELASMOBRANCHS (N=3)

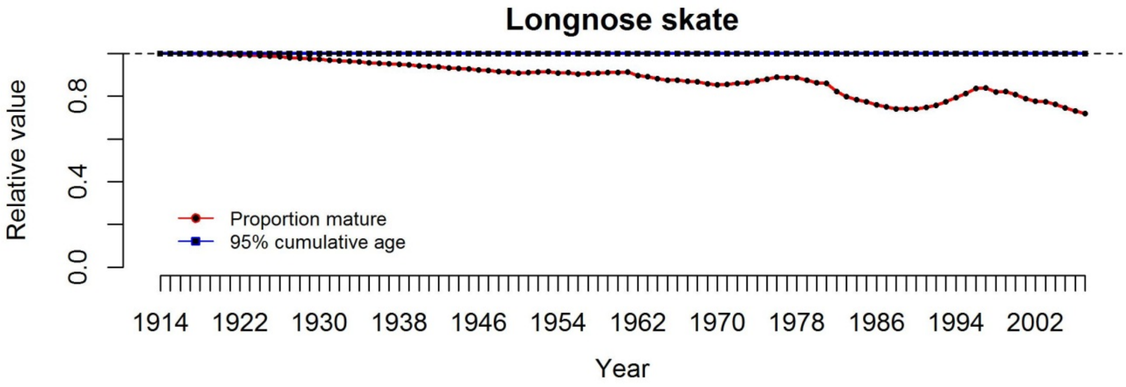

Figure GF42. Proportion of the longnose skate population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

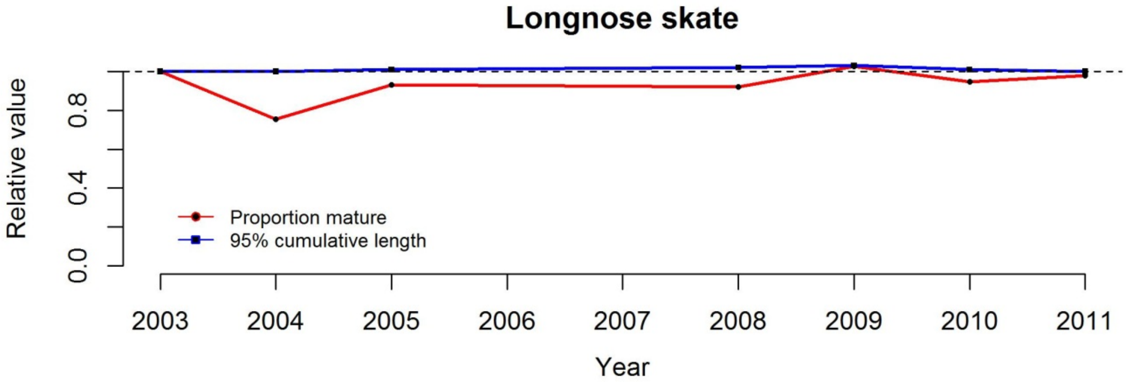

Figure GF43. Proportion of the longnose skate population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: Longnose skate have shown no decline in the proportion of the oldest ages and largest lengths, but proportion mature has declined somewhat over the length of the time series.

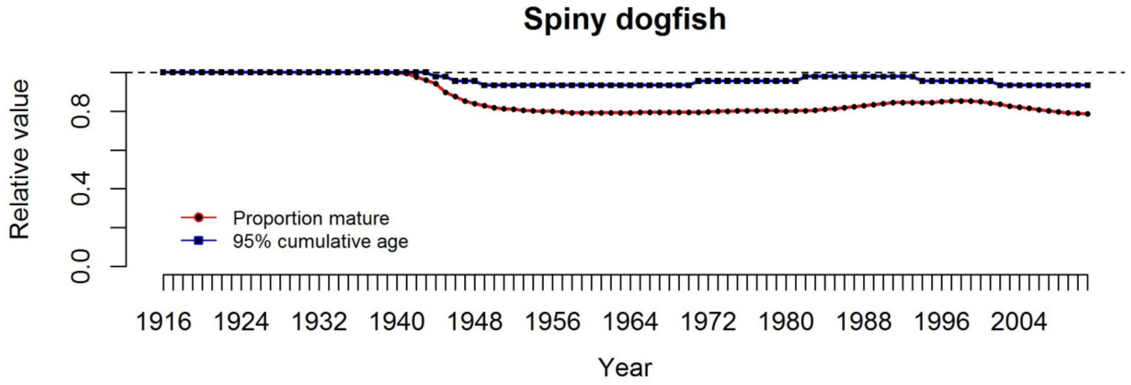

Figure GF44. Proportion of the spiny dogfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Spiny dogfish show only small declines in proportion mature and proportion of the oldest ages that have mostly stabilized since the decline in the 1940s.

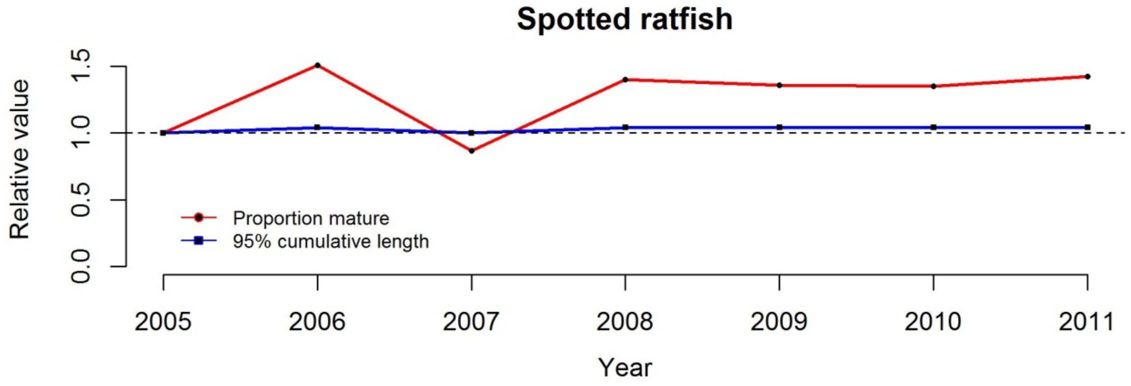

Figure GF45. Proportion of the spotted ratfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2005) of the trawl survey time series.

Summary: No stock assessment is available for spotted ratfish so no baseline information on demographic structure is available. No declines in either maturity or proportion of the largest sizes are apparent from the trawl survey data.

FLATFISHES (N=7)

Shelf

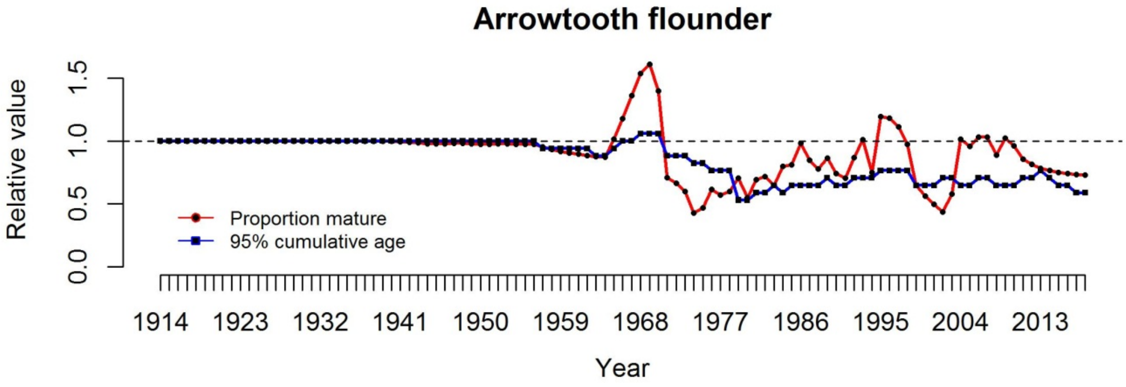

Figure GF46. Proportion of the arrowtooth flounder population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

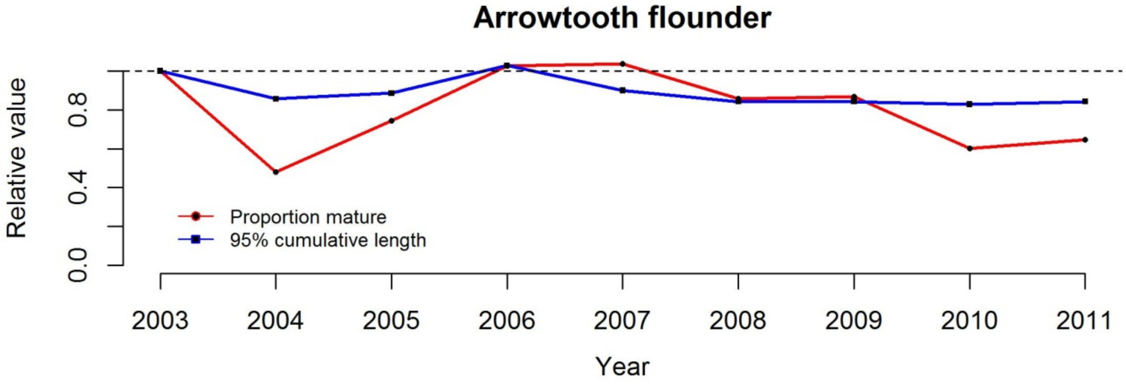

Figure GF47. Proportion of the arrowtooth flounder population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: Arrowtooth flounder show declines in proportion mature and proportion of the oldest ages and largest lengths over the length of the time series.

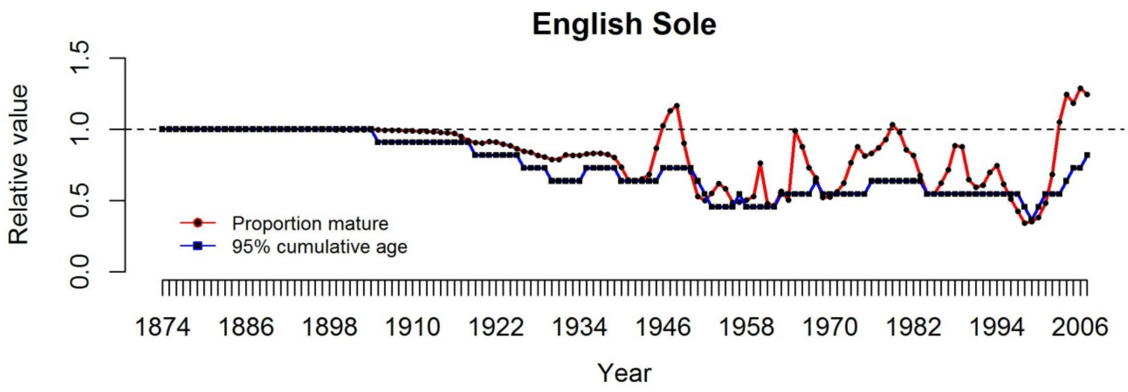

Figure GF48. Proportion of the English sole population mature (red) and at the 95% cumulative age (blue) relative to the first year (1876) of the time series.

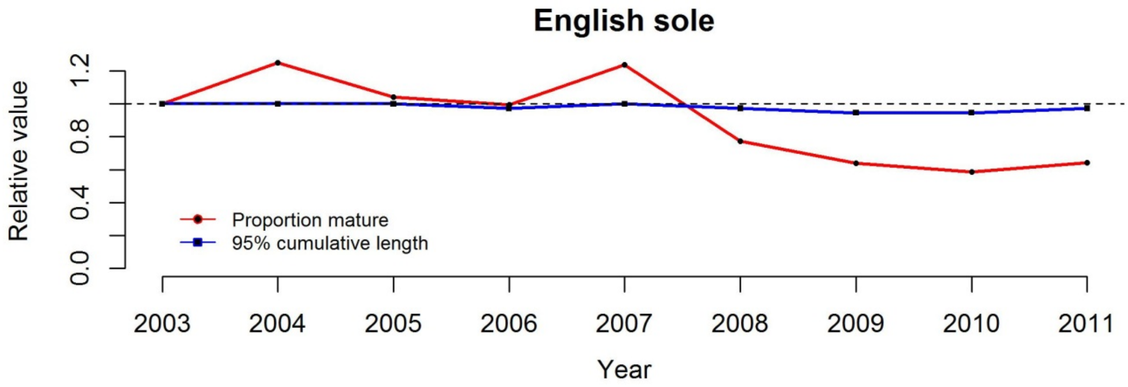

Figure GF49. Proportion of the English sole population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: English sole show a slight decline in proportion of the oldest ages, but not in proportion mature, over the length of the time series. Recent survey trends in proportion mature are downward.

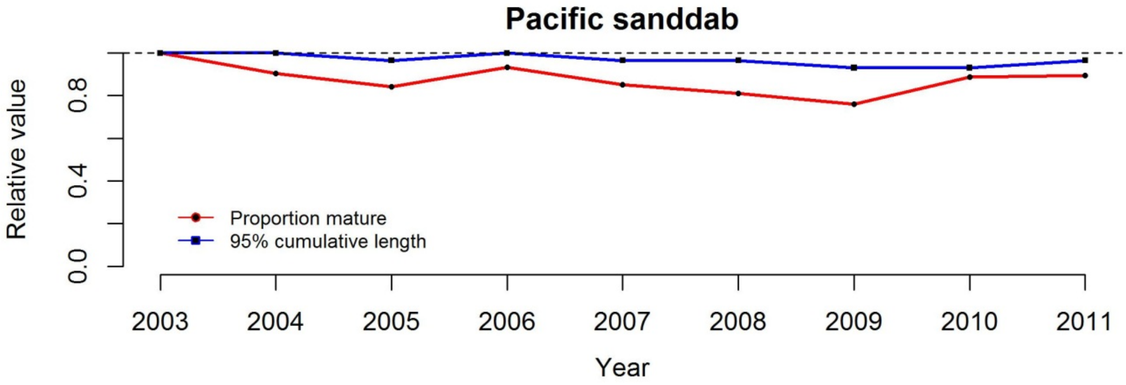

Figure GF50. Proportion of the Pacific sanddab population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: No stock assessment is available for Pacific sanddab so no baseline information on demographic structure is available. No declines in either maturity or proportion of the largest lengths are apparent from the trawl survey data.

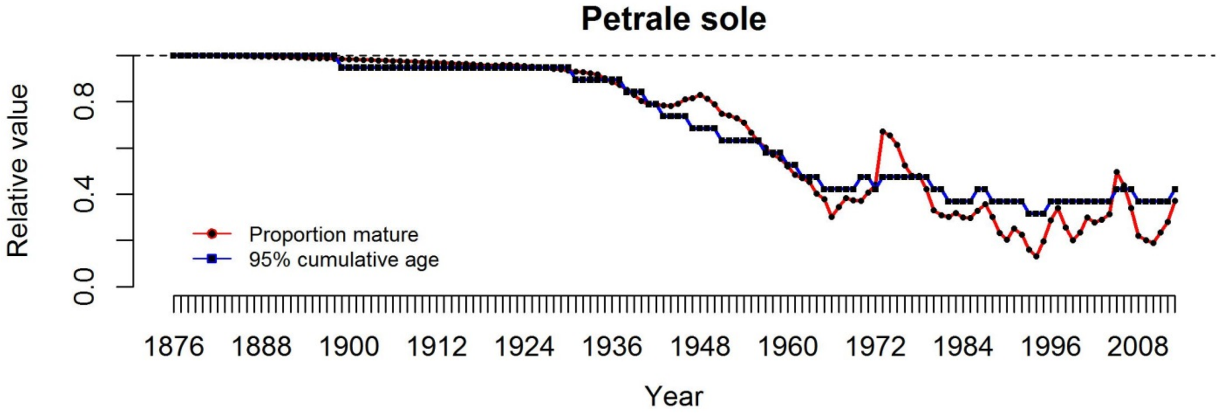

Figure GF51. Proportion of the Petrale sole population mature (red) and at the 95% cumulative age (blue) relative to the first year (1876) of the time series.

Summary: Petrale sole shows notable declines in proportion mature and proportion of the oldest ages over the length of the time series.

Slope

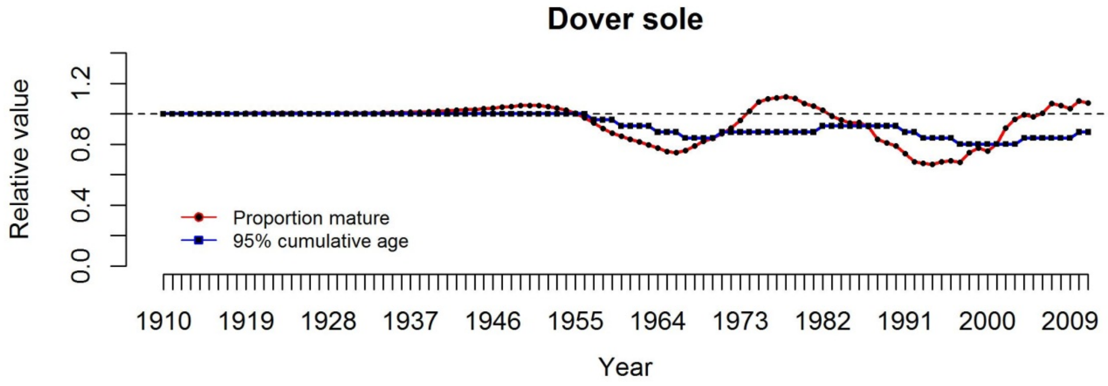

Figure GF52. Proportion of the Dover sole population mature (red) and at the 95% cumulative age (blue) relative to the first year (1910) of the time series.

Summary: Dover sole do not show any notable changes in proportion mature and proportion of the oldest ages and largest lengths over the length of the time series.

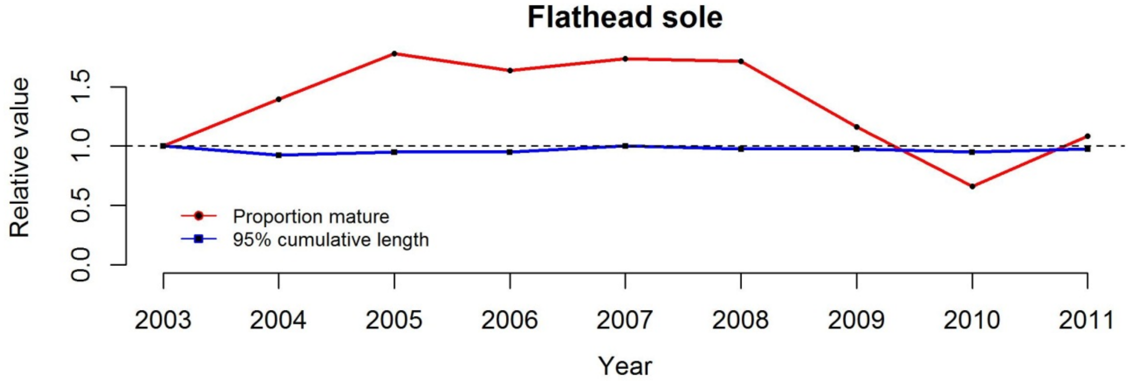

Figure GF53. Proportion of the flathead sole population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: No stock assessment is available for flathead sole so no baseline information on demographic structure is available. No declines in either maturity or proportion of the largest lengths are apparent from the trawl survey data.

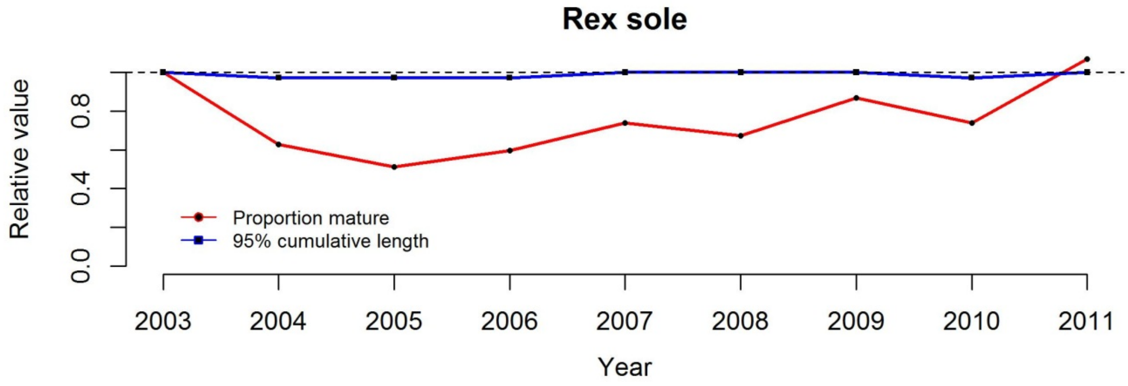

Figure GF54. Proportion of the rex sole population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: No stock assessment is available for rex sole so no baseline information on demographic structure is available. No declines in either maturity or proportion of the largest lengths are apparent from the trawl survey data.

ROCKFISHES (N=18)

Nearshore

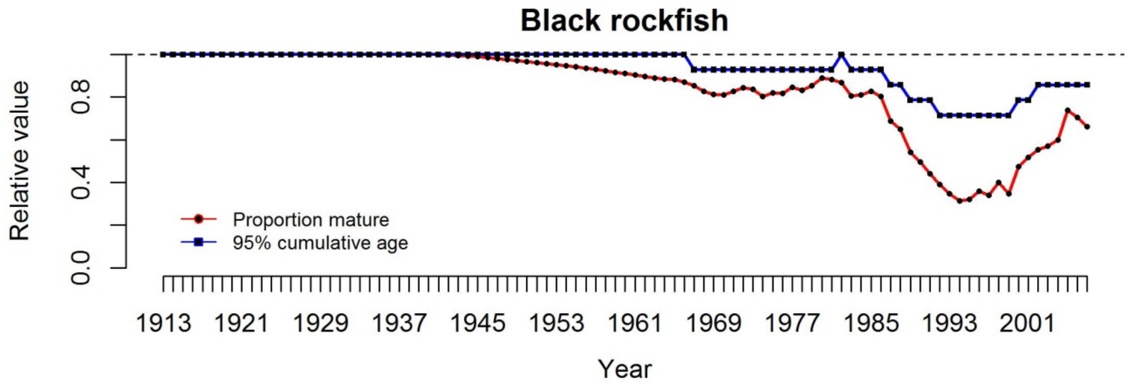

Figure GF55. Proportion of the black rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Black rockfish show notable declines in proportion mature and slight declines in proportion of the oldest ages over the length of the time series.

Shelf

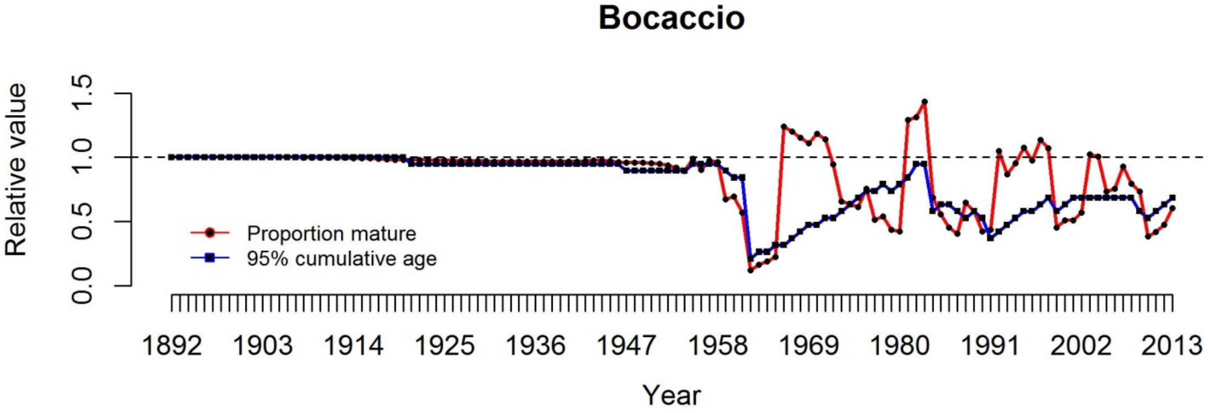

Figure GF56. Proportion of the bocaccio population mature (red) and at the 95% cumulative age (blue) relative to the first year (1895) of the time series.

Summary: Bocaccio show high variation in the proportion mature and proportion of the oldest ages over the length of the time series. The most recent measure are below historical reference levels. Fluctuations may be due to high but sporadic recruitment.

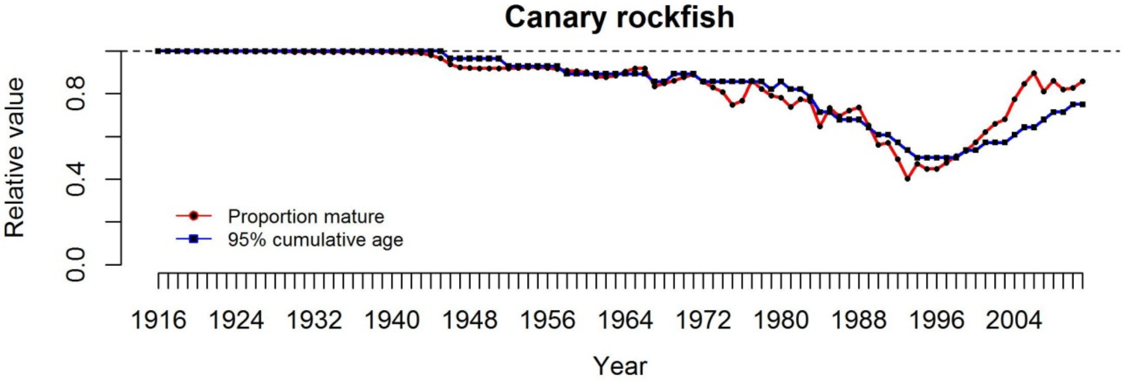

Figure GF57. Proportion of the canary rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Canary rockfish show declines in proportion mature and proportion of the oldest ages over the length of the time series, but current years demonstrate a building up of both metrics.

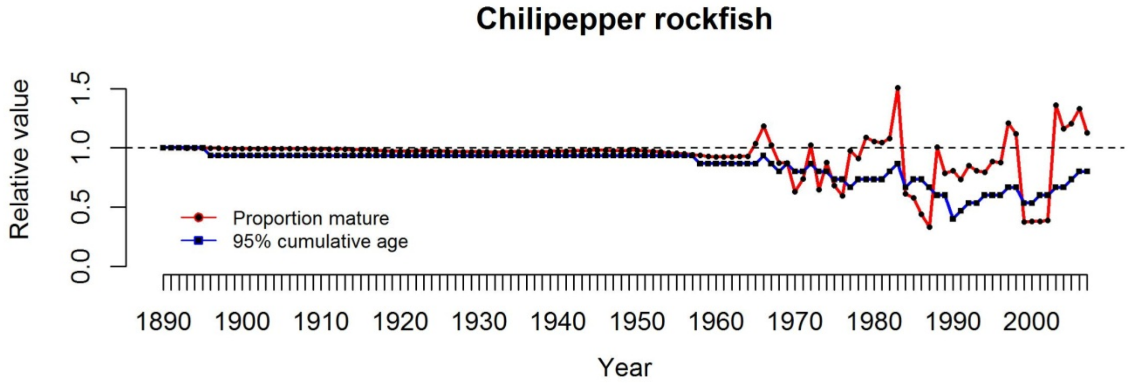

Figure GF58. Proportion of the chilipepper rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1892) of the time series (1892-2007).

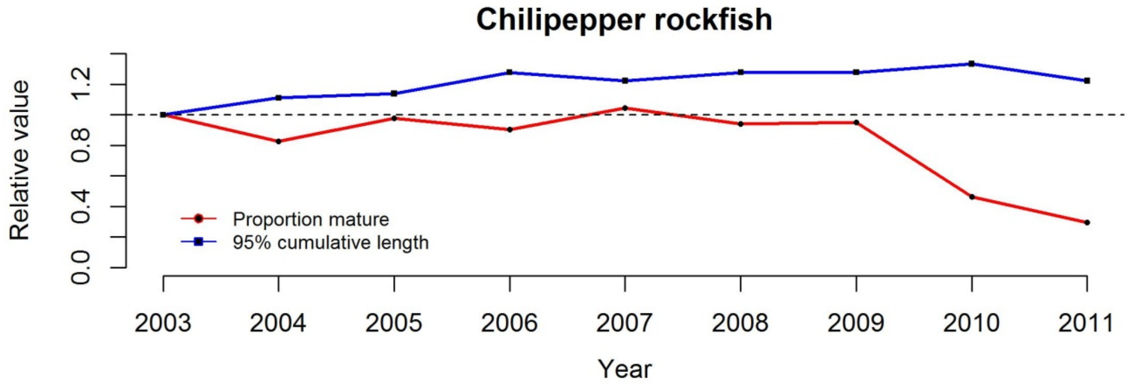

Figure GF59. Proportion of the chilipepper rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series (2003-2007).

Summary: Chilipepper rockfish show decreases in proportion mature and proportion of the oldest ages and largest lengths over the length of the time series. The short-term series shows a relative changes consistent with the long-time series when the same relative time frame in considered.

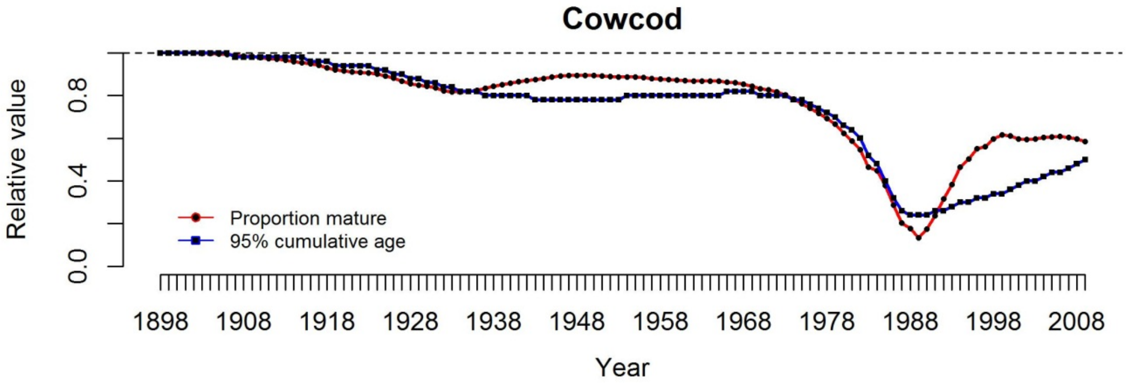

Figure GF60. Proportion of the cowcod population mature (red) and at the 95% cumulative age (blue) relative to the first year (1900) of the time series.

Summary: Cowcod show declines in proportion mature and proportion of the oldest ages over the length of the time series.

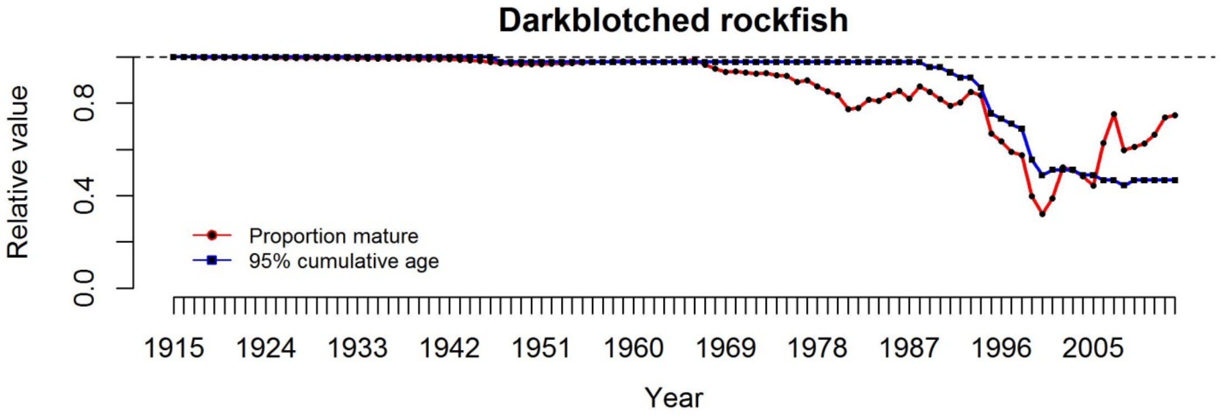

Figure GF61. Proportion of the darkblotched rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1895) of the time series.

Summary: Darkblotched rockfish show declines in proportion mature and proportion of the oldest ages over the length of the time series.

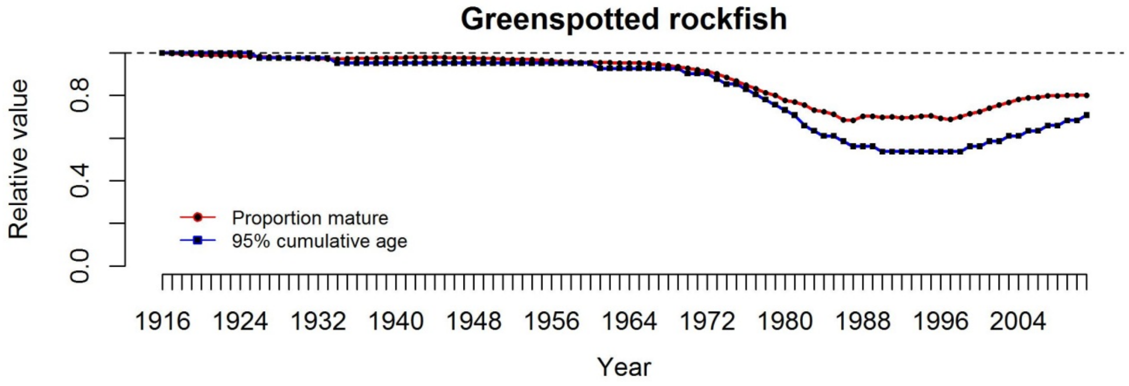

Figure GF62. Proportion of the greenspotted rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Greenspotted rockfish show declines in proportion mature and proportion of the oldest ages over the length of the time series.

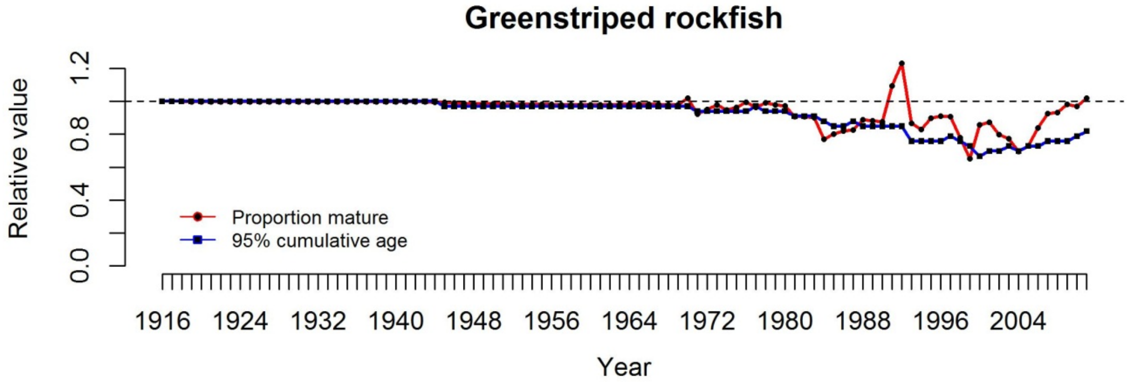

Figure GF63. Proportion of the greenstriped rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Greenstriped rockfish show little change in proportion mature and proportion of the oldest ages over the length of the time series, with only a small decrease in population structure.

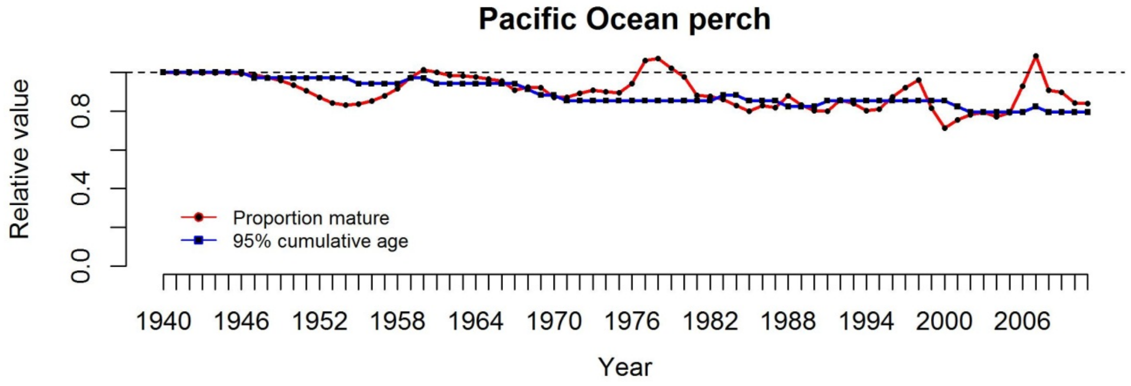

Figure GF64. Proportion of the Pacific Ocean perch population mature (red) and at the 95% cumulative age (blue) relative to the first year (1940) of the time series.

Summary: Pacific Ocean perch show low levels of decline in proportion mature and proportion of the oldest ages over the length of the time series.

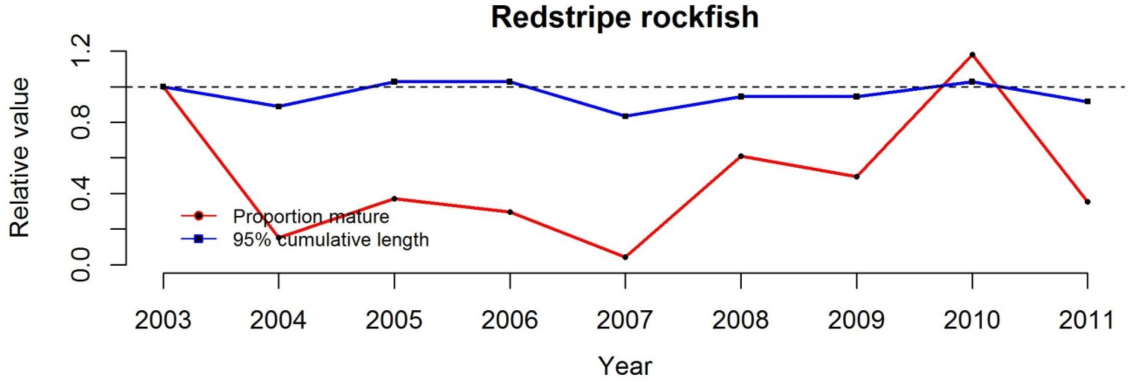

Figure GF65. Proportion of the redstripe rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary:

No stock assessment is available for redstripe rockfish so no baseline information on demographic structure is available. No declines in proportion of the largest lengths is apparent from the trawl survey data, though proportion of mature individuals is variable across years with a notable decline.

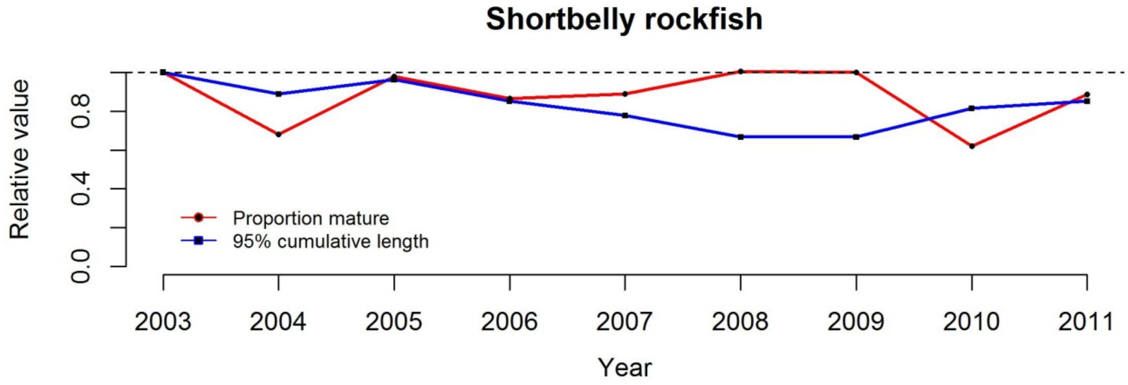

Figure GF66. Proportion of the shortbelly rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: Only modest declines in either maturity or proportion of the largest lengths are apparent from the trawl survey data for shortbelly rockfish.

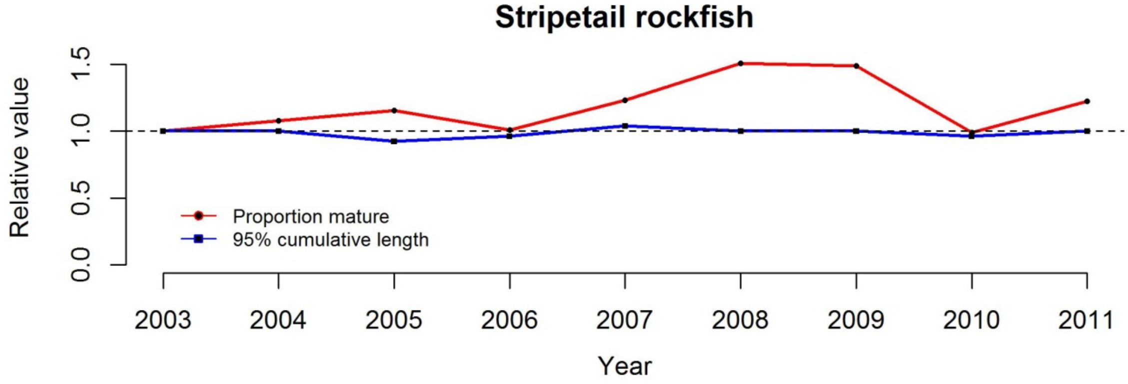

Figure GF67. Proportion of the stripetail rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: No stock assessment is available for stripetail rockfish so no baseline information on demographic structure is available. No declines in either maturity or proportion of the largest lengths are apparent from the trawl survey data.

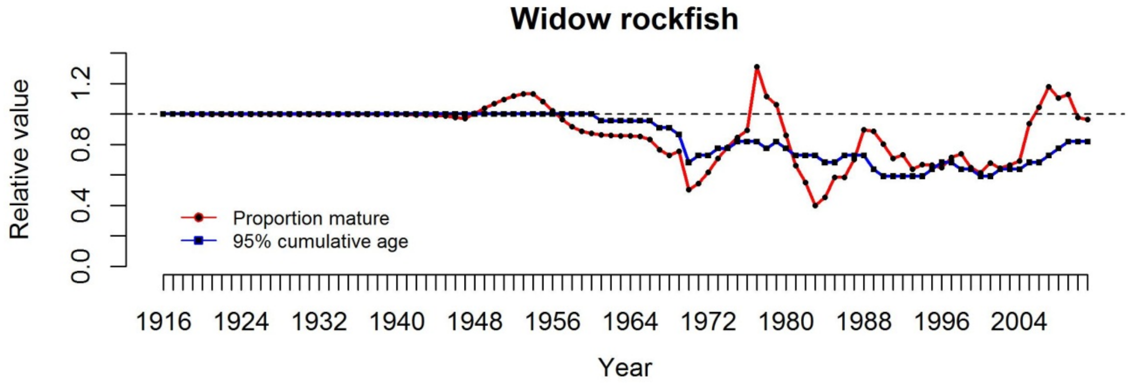

Figure GF68. Proportion of the widow rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Widow rockfish show no declines in proportion mature and population structure over the length of the time series that has returned or is building back towards historical levels.

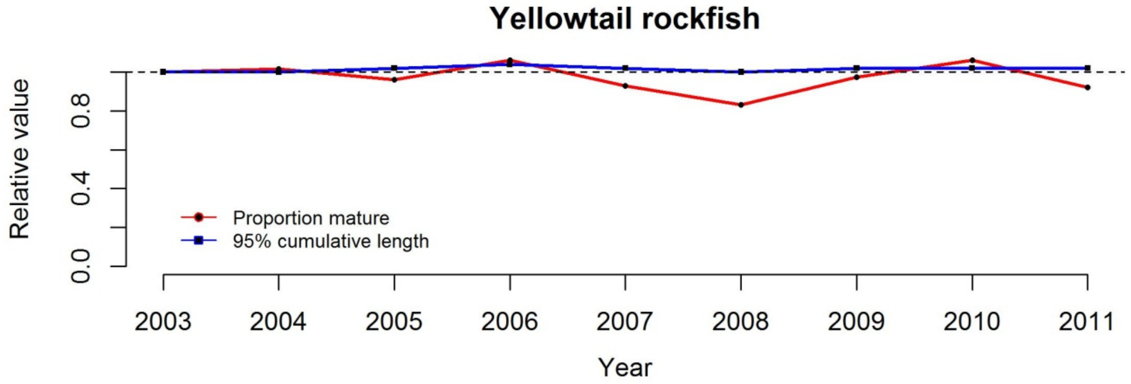

Figure GF69. Proportion of the yellowtail rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: No declines in either maturity or proportion of the largest lengths are apparent from the trawl survey data for yellowtail rockfish.

Slope

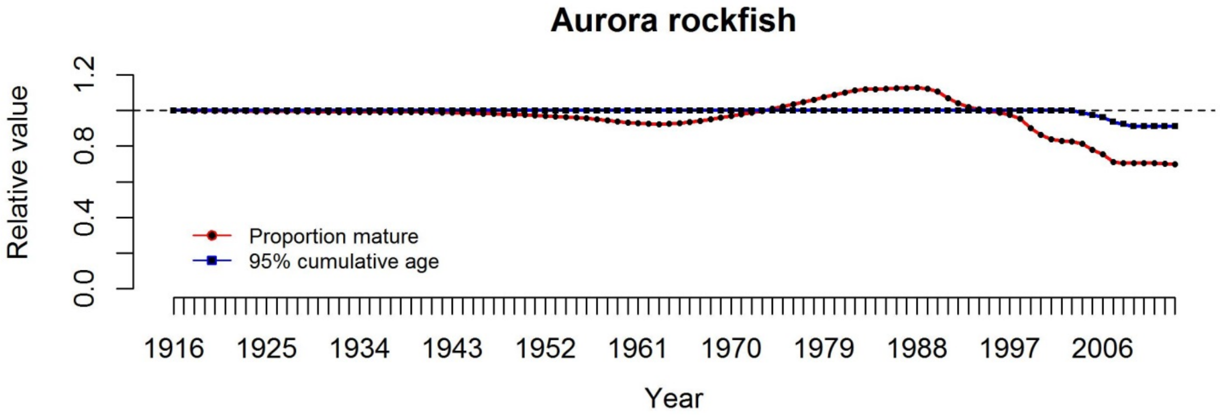

Figure GF70. Proportion of the aurora rockfish population mature (red) and at the 95% cumulative length (blue) relative to the first year (2003) of the trawl survey time series.

Summary: Aurora rockfish shows very little change in cumulative age, with greater decline in the proportion of mature females.

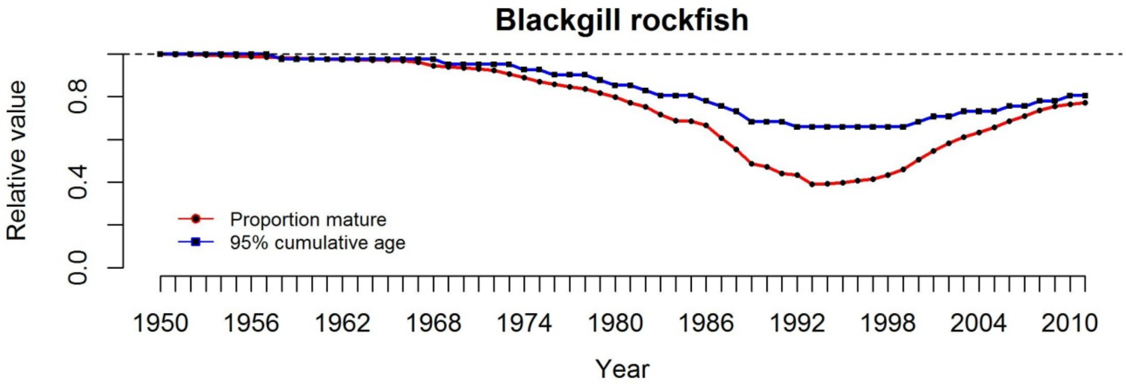

Figure GF71. Proportion of the blackgill rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1950) of the time series.

Summary: Blackgill rockfish show declines in proportion mature and proportion of the oldest ages over the length of the time series.

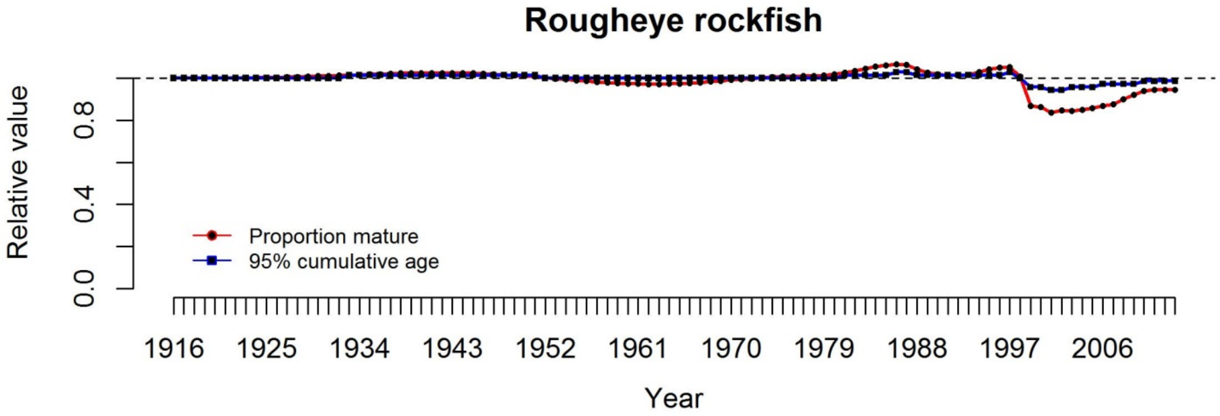

Figure GF72. Proportion of the rougheyerockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1950) of the time series.

Summary: Rougheye rockfish show little change in proportion mature and proportion of the oldest ages over the length of the time series.

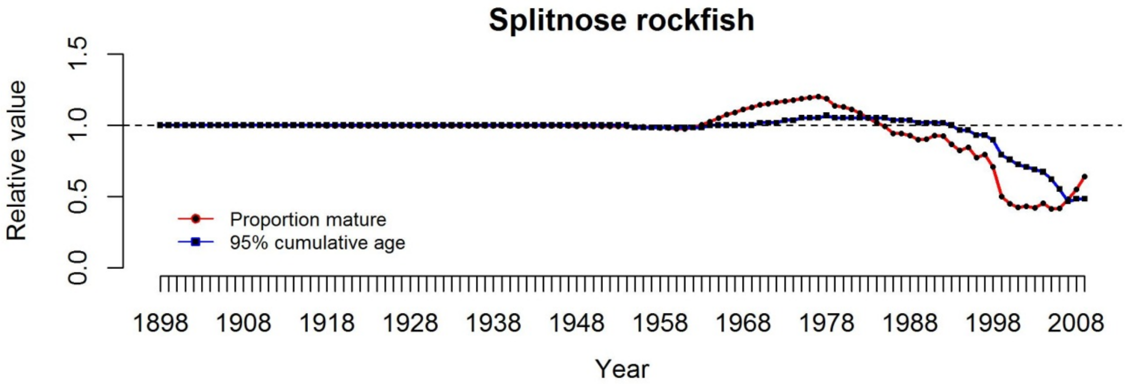

Figure GF73. Proportion of the splitnose rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1900) of the time series.

Summary: Splitnose rockfish show declines in proportion mature and proportion of the oldest ages over the length of the time series.

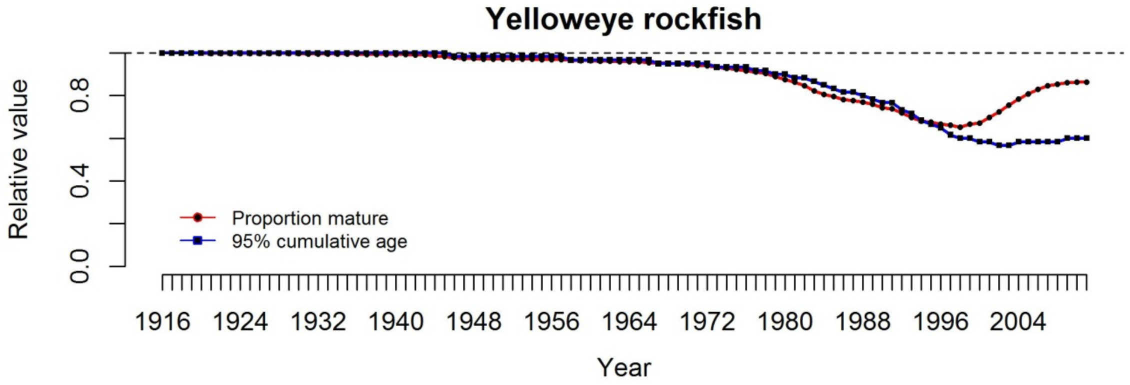

Figure GF74. Proportion of the yelloweye rockfish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Yelloweye rockfish show declines in proportion mature and proportion of the oldest ages over the length of the time series.

THORNYHEADS (N=2)

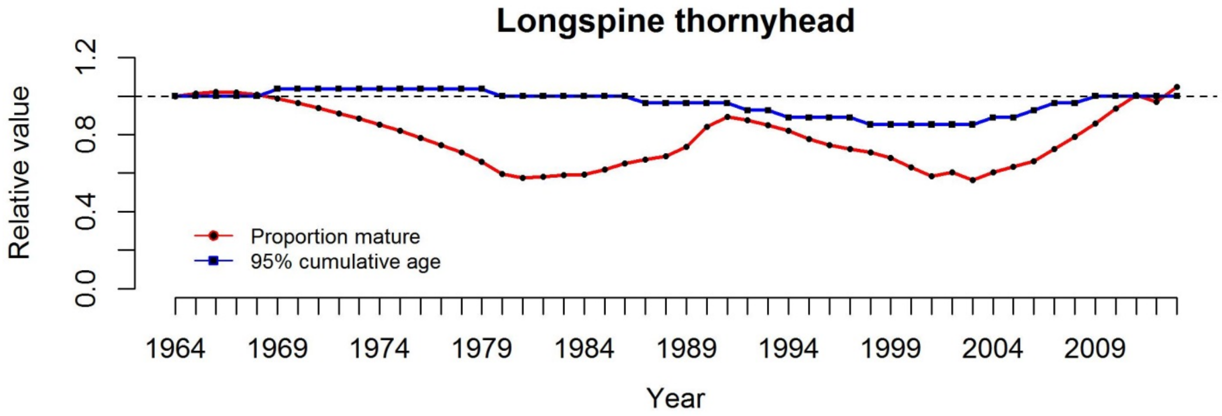

Figure GF75. Proportion of the longspine thornyhead population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Longspine thornyheads have demonstrated .some historical decline in age structure and maturity, but current populations are similar to earlier period population structure.

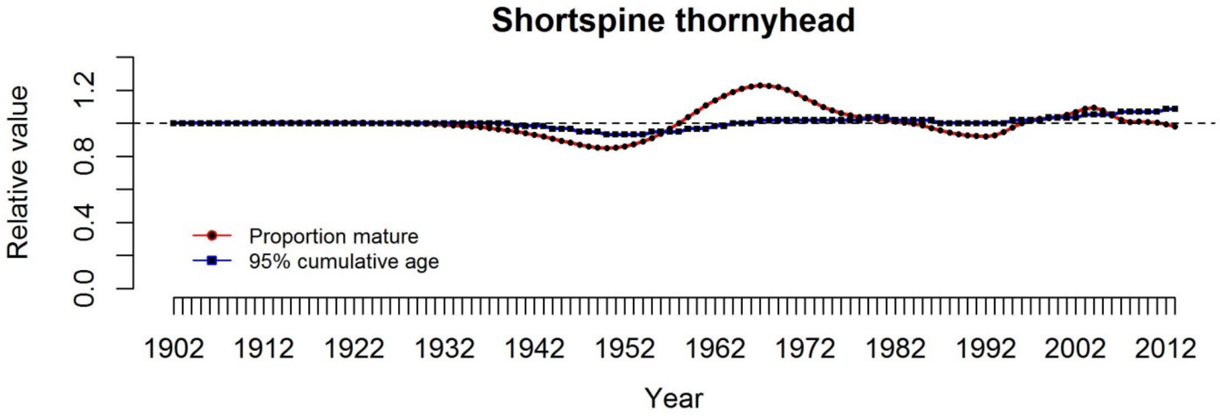

Figure GF76. Proportion of the shortspine thornyhead population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Shortspine thornyheads have demonstrated very little change in age structure and maturity over the measured time period.

ROUNDFISHES (N=3)

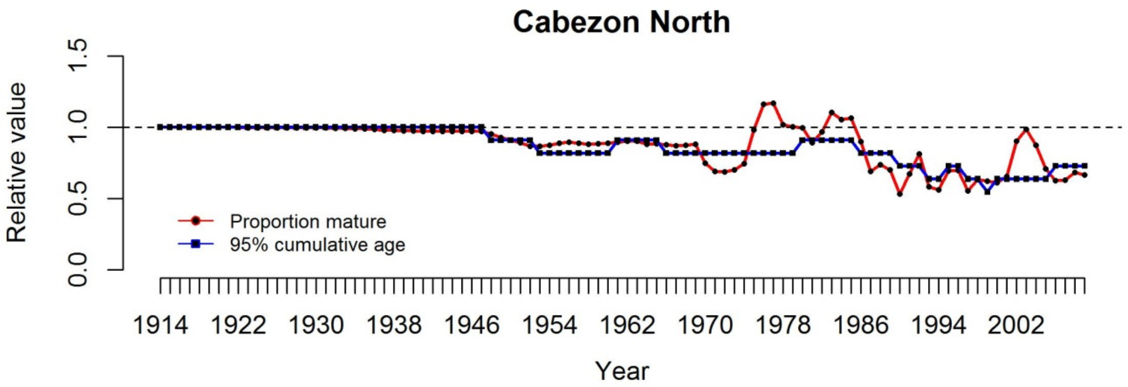

Figure GF77. Proportion of the cabezon population mature (red) and at the 95% cumulative age (blue) relative to the first year (1916) of the time series.

Summary: Cabezon show declines in proportion mature and proportion of the oldest ages over the length of the time series.

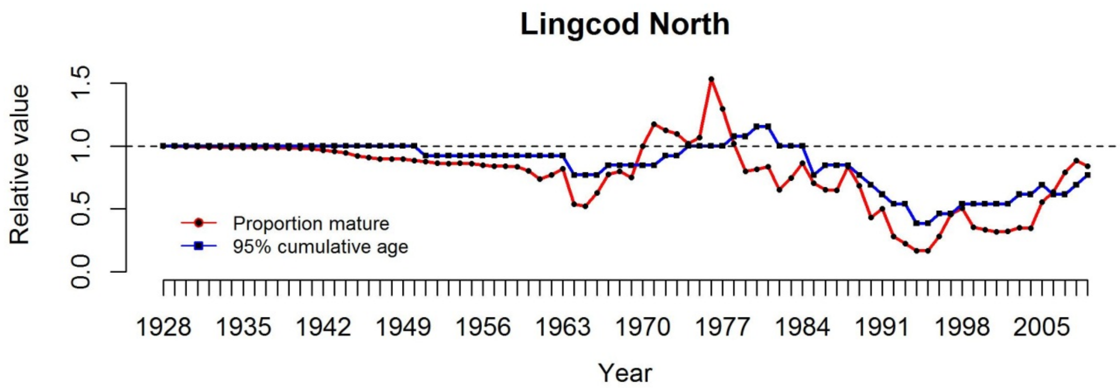

Figure GF78. Proportion of the lingcod population mature (red) and at the 95% cumulative age (blue) relative to the first year (1930) of the time series.

Summary: Lingcod show declines in proportion mature and proportion of the oldest ages that have recently shown increases towards historical levels.

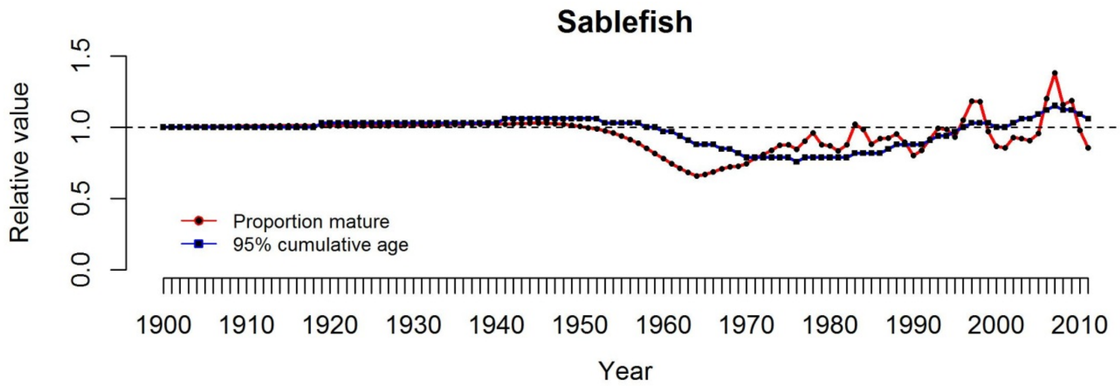

Figure GF79. Proportion of the sablefish population mature (red) and at the 95% cumulative age (blue) relative to the first year (1900) of the time series.

Summary: Sablefish show little change in proportion mature and proportion of the oldest ages over the length of the time series.

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PACIFIC SALMON

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OVERVIEW

Abundances for a number of West Coast salmon population groups declined over the last ten years. For Chinook salmon, the Lower Columbia and Willamette Spring-run data series exhibited the steepest declines, while the Central Valley Fall-run, Spring-run, and Winter-run Chinook salmon as well as the Southern Oregon/Northern California, California Coast, and Klamath fall-run Chinook salmon stocks exhibited more moderate declines. On the positive side, Snake River and Upper Columbia River Chinook salmon abundance increased. All Chinook salmon Evolutionarily Significant Units (ESUs) were near their longer-term (25-30 year) average abundance. For coho salmon, all recent abundance series were near their 25-30 year averages. The California Coast and Southern Oregon/Northern California Coasts trends in abundance declined steeply, while the trends abundance of Lower Columbia River coho salmon increased. Oregon Coast coho salmon demonstrated no significant recent trend over 2003 to 2012.

Recent ocean conditions and the forage complex indicate a likelihood of improved early marine survival of Chinook salmon and coho salmon in 2012 and 2013, suggesting improved adult returns in the next few years. In contrast, freshwater flows and temperatures suggest reduced smolt production in the near future across California. Anthropogenic climate change trends are likely to increase risks facing West Coast salmonid stocks in future decades of the 21st century.

Salmon and steelhead populations and habitat have been influenced by dynamic interactions between natural landscape features (e.g., resource abundance, climate, topography) and human activities such as fur trading, mining, logging, agriculture, dams, hatcheries, and fisheries. Historical development of these activities was largely driven by economic interests and encouraged by robust market demand and prices, improvements in extractive and processing technologies and transportation, and expansionist government policies. Most of these activities (other than fur trading) continue to the present day in some form. Public policies have changed over time, from an ethos of laissez faire resource extraction to one that also considers effects of extraction on wild salmon populations and the habitat and ecological processes that affect salmon. Such policy shifts reflect recognition that salmon and salmon habitat are components of human values and well-being.

EXECUTIVE SUMMARY

Both short- and long-term trends for salmon indicators of West Coast salmon abundance and aspects of their ecosystems are reported in this summary. An indicator is considered to have changed over the short-term if the trend over the last 10 years (2003-2012) of the series showed a significant increasing or decreasing slope. An indicator is considered to be above or below long-term norms if the mean of the last 10 years of the time series differs from the mean of the full time series by more than 1.0 s.d. of the full time series. A major motivation for presenting long- and short-term trends is to distinguish stocks/populations that were once very large and suffered historical declines as much as 100 years ago but have stabilized at lower abundances from populations with ongoing declines; we do not address issues of historical declines prior to the mid-1980s. This issue most affected populations with very long time series of abundance (e.g., certain Columbia River Chinook salmon populations). Such long time series are not available for most Californian populations. We avoided reliance on data prior to 1985 because of concerns over data quality. Therefore, references to "long-term" abundance, condition, etc. refer to periods of record from 1985. It should be noted that many if not most of these populations are now at levels far below historical values – so caution should be used when interpreting the "long-term" status in this report.

Generally, all California Chinook salmon stocks from 2004-2013 were within 1 s.d. of their longer-term average. However, during the last ten years there was a significant decline in abundance of most California populations examined, with Central Valley Winter-run Chinook salmon at extremely low abundances from 2007-2011. This relates to a reduction from series highs during the early 2000s and a return to the very low values typical of the 1980s and 1990s. For the Columbia/Snake Basin Chinook salmon stocks, recent abundances were also close to average, except for a positive deviation for the Snake River Fall-run. There is a notable contrast in recent trends between steep declines in the lower Columbia River and Willamette River stocks and increases in the upper Columbia and Snake River stocks. As for the California stocks, the observed steep declines follow peak abundances in the early 2000s.

With a few noteworthy exceptions, both the recent trends and recent average levels of condition indices for Chinook salmon have been near long-term average values. In general, there are significant downward trends in condition for Lower Columbia River and Willamette River series, with the exception of improving trends for Willamette River percent natural spawners and age diversity. Klamath River Fall-run Chinook salmon also exhibited an upward trend for percent natural spawners. Notably, Lower Columbia River percent natural reflects a long-term decline in this indicator (Figure S4). Similarly, the Central Valley Fall-run Chinook salmon falls on the border of the "low and decreasing" quadrant for percent natural spawners.

While recent abundance of all coho salmon stocks are near their long-term average, there is a sharp contrast in recent trends. The Central California Coast and Southern Oregon/Northern California Coast stocks both had steep declines following strong peaks in 2004, while the Lower Columbia River stock had a fluctuating increase in recent years. The two northern stocks (Oregon and Lower Columbia) are both well above their historic low abundances in the 1990s.

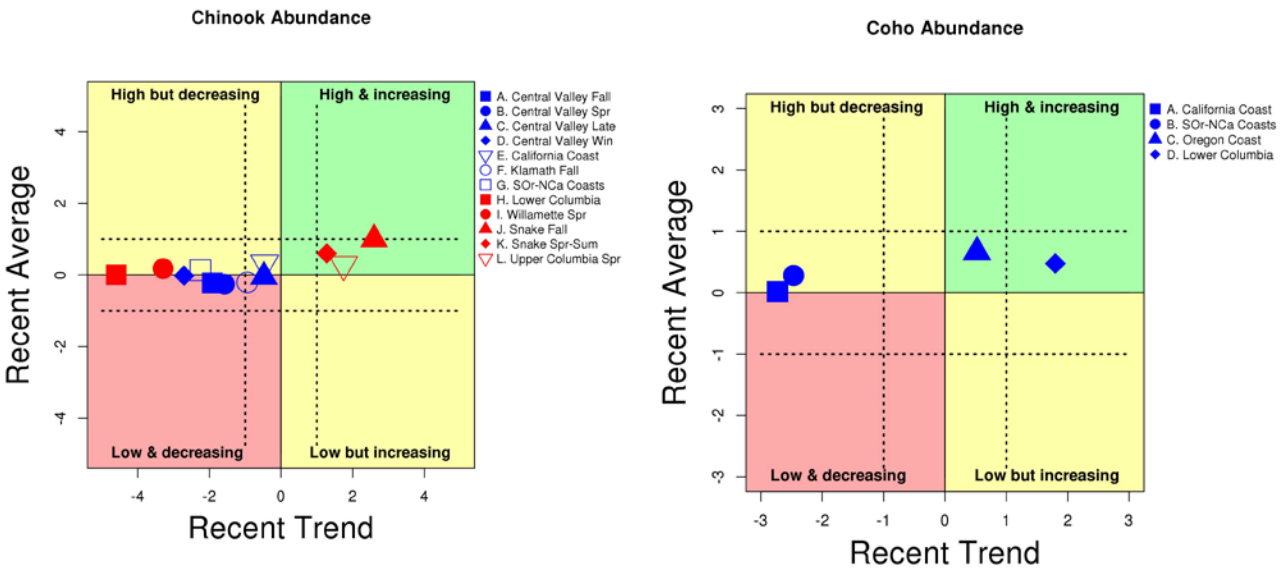
There is no condition data available for the two southernmost coho salmon ESUs, and data for the two northern ESUs are limited to percent natural spawners (both stocks) and population growth rate (for the Oregon Coast stock). None of the data series exhibit significant recent trends, and both series for the Oregon Coast stock are near 25-30 year averages. Recent percent natural spawners for the Lower Columbia River stock is higher than the longer-term average. The Oregon Coast stock exhibits an encouraging long-term upward trend in percent natural spawners.

In this report we consider those environmental factors demonstrated to affect salmon abundance and condition. We evaluate the state of the environment, its potential influence on salmon abundance and condition, and the potential for effects from future climate change. Recent ocean conditions and the forage complex indicate a likelihood of improved early marine survival of Chinook and coho salmon in 2012 and 2013, suggesting improved adult returns in the next few years. In contrast, freshwater flows and temperatures suggest reduced smolts per spanwer in the near future for the

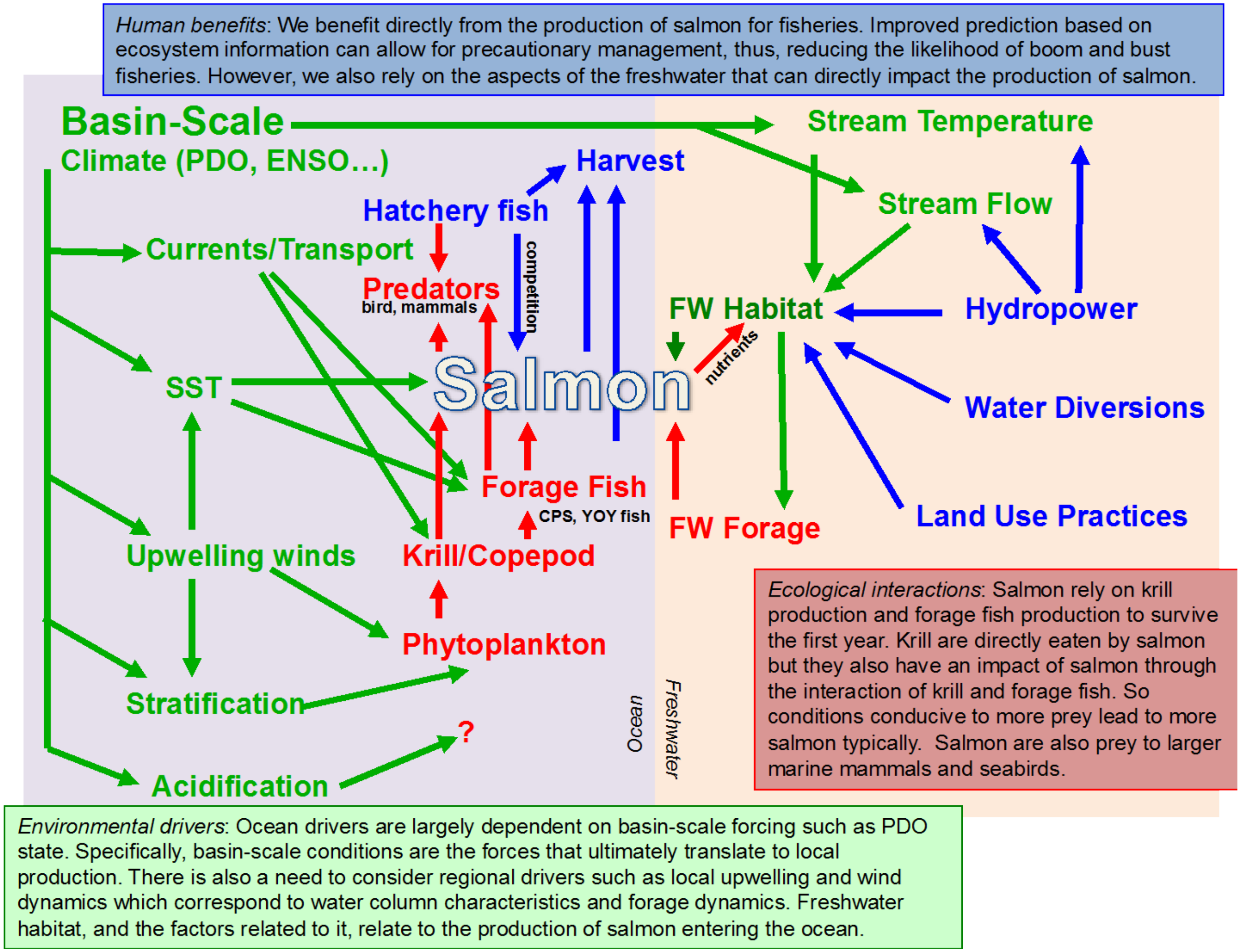
Snake River Basin and across California. Anthropogenic climate change trends are likely to increase risks facing West Coast salmon stocks over the future decades of the 21st century and beyond.

Salmonid populations and habitat have been influenced by dynamic interactions between natural landscape features (e.g., resource abundance, climate, topography) and human activities such as fur trading, mining, logging, agriculture, dams, hatcheries and fisheries. Historical development of these activities was largely driven by economic interests and encouraged by robust market demand and prices, improvements in extractive and processing technologies and transportation, and expansionist government policies. Most of these activities (other than fur trading) continue to the present day in some form. Public policies have changed over time, from an ethos of *laissez faire* resource extraction to one that also considers effects of extraction on wild salmon populations and the habitat and ecological processes that affect salmon. Such policy shifts reflect recognition that salmon and salmon habitats are components of human values and wellbeing.

Most of the quantitative information regarding anthropogenic influences on salmon pertained to outputs from commercial activities (e.g., timber production, agricultural values, salmon harvest). Additional work is needed to consider other indicators that are inclusive of other aspects of human wellbeing. An important next step toward operationalizing the CCIEA is to identify goal(s) that managers wish to achieve by considering salmon in a California Current integrated Ecosystem Assessment (CCIEA) framework, as those goals will affect model specification and the types of indicators appropriate for inclusion in the model.



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



Human benefits: We benefit directly from the production of salmon for fisheries. Improved prediction based on ecosystem information can allow for precautionary management, thus, reducing the likelihood of boom and bust fisheries. However, we also rely on the aspects of the freshwater that can directly impact the production of salmon.

Ecological interactions: Salmon rely on krill production and forage fish production to survive the first year. Krill are directly eaten by salmon but they also have an impact of salmon through the interaction of krill and forage fish. So conditions conducive to more prey lead to more salmon typically. Salmon are also prey to larger marine mammals and seabirds.

Environmental drivers: Ocean drivers are largely dependent on basin-scale forcing such as PDO state. Specifically, basin-scale conditions are the forces that ultimately translate to local production. There is also a need to consider regional drivers such as local upwelling and wind dynamics which correspond to water column characteristics and forage dynamics. Freshwater habitat, and the factors related to it, relate to the production of salmon entering the ocean.

The conceptual diagram demonstrates the various environmental and anthropogenic influences that interact to affect salmon through their life cycle. We have included information in this report on each factor when available. We discuss its history, status, and/or trend in the context of salmon and management of the ecosystem. This model should aide in the understanding of the complex web that must be considered when managing the trade-offs associated with human wellbeing and salmon viability

DETAILED REPORT

Pacific salmon (*Oncorhynchus* spp.) are iconic members of North Pacific rim ecosystems, historically ranging from Baja California to Korea (Groot and Margolis 1991). Historically, salmon supported extensive native estuarine and freshwater fisheries along the U.S. West Coast, followed more recently by large commercial marine and recreational marine and freshwater harvest. Salmon and steelhead connect marine and freshwater ecosystems through extensive migrations up to 1500 km.

The purpose of this chapter of the CCIEA is to examine trends in available indicators relevant to salmon along the California Current. It is important to recognize that we refer to population “status” quite differently than that reported by Pacific Fisheries Management Council (PFMC) and in current Endangered Species Act status reports, therefore, any difference between our status statements and those should not be considered a conflict. We use different models and benchmarks than those traditionally used by fishery managers. Our purpose is to set the framework for evaluating the salmon community from an ecosystem perspective. This approach starts with a simple selection of indicators and evaluation of the trends. Here, to a limited degree, we use these biological indicators in combination with indicators of environmental and anthropogenic pressures to evaluate potential risk to the salmon community. Indicators for various pressures can be found in other chapters of the full CCIEA (e.g., Anthropogenic Drivers and Pressures, Oceanographic and Climatic Drivers and Pressures).

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

We focus on the two most abundant salmon species in the CCLME Chinook salmon (*O. tshawytscha*) and coho salmon (*O. kisutch*), which have historically supported large fisheries and continue to support economically and culturally important fisheries when and where they remain open (Pacific Fisheries Management Council 2012). Within these species, we selected stocks that span a range of geographic and life-history variation characteristic of the broader community. Pacific salmon species have complex population structures, leading to a variety of ways of defining 'stock' (e.g., Cushing 1981, Dizon et al. 1992). We have chosen to use the Evolutionarily Significant Unit (ESU) defined by NOAA for use in Pacific salmon conservation management (Waples 1991). ESUs are defined on the basis of reproductive isolation and their contribution to the evolutionary legacy of the species as a whole, and are often composed of a number of geographically contiguous populations. They do not correspond exactly to the stock delineations that are used for harvest management; in most cases several stocks/populations make up an ESU. It is worth noting, future Phases of the CCIEA will also include more representation of steelhead(*O. mykiss*).

Both short- and long-term trends are reported in this summary. An indicator is considered to have changed over the short-term if the trend over the last 10 years (2003-2012) of the series showed a significant increasing or decreasing slope. An indicator is considered to be above or below long-term norms if the mean of the last 10 years of the time series differs from the mean of the full time series by more than 1.0 s.d. of the full time series. A major motivation for presenting long- and short-term trends is to distinguish stocks/populations that were once very large and suffered historical declines as much as 100 + years ago but have stabilized at lower abundances from populations with ongoing declines. This issue most affected populations with very long time series of abundance (e.g., certain Columbia River Chinook salmon populations). Such long time series are not available for most Californian populations. We avoided reliance on data prior to 1985 because of concerns over data quality. Therefore, references to "long-term" abundance, condition, etc. refer to periods of record from 1985. It should be noted that many if not most of these populations are now at levels far below historical values – so caution should be used when interpreting the "long-term" status in this report.

We expanded this report from CCIEA Phase II to include environmental pressures and anthropogenic activities that affect salmon abundance and condition either directly or indirectly. The state of current and potential future environments are discussed in the context of the salmon and salmon habitat. We also discuss the patterns and trends of resource-related activities occurring post-1848, including fur trading, mining, logging, farming, dams, hatcheries and fishing that provides a context for the historical declines over the past 100-150 years to most if not all salmonid populations throughout the California current. We relate these activities to major demographic, economic, social, technological and policy changes that occurred with Euro-American settlement in the region. This historical perspective considers legacy as well as ongoing effects of those activities on wild salmon and salmon habitat, and shows how more recent environmental protections have moderated the single-minded resource exploitation characteristic of earlier decades. It also provides (with the benefit of hindsight) the opportunity to illustrate, with concrete examples, the dynamic relationship between human dimensions and salmon, and may suggest ways in which such relationships can be modeled in an ecosystem context. The quantitative indicators provided here regarding trends in human activities are a first step in that direction, though much more work needs to be done.

SALMON ABUNDANCE AND CONDITION INDICATOR SELECTION PROCESS

SUMMARY OF INDICATORS

An extensive search and evaluation of indicators of salmon abundance and condition was conducted. The specifics of that search are outlined in greater detail below. In summary, Table S1 shows the relevant ESUs for Chinook salmon and coho salmon and indicators types (abundance or condition) used in this report.

Table S1. Salmon ESUs/stocks and available data. 'X' indicates that a data series is available, a blank indicates insufficient data are available**.**

		Condition Index		
Stock/ESU Name	**Abundance**	**Age Diversity**	**Percent Natural**	**Population Growth Rate**
Chinook Salmon				
A. Central Valley Fall-run	X		X	X
B. Central Valley Spring-run	X			
C. Central Valley Late Fall-run	X			
D. Central Valley Winter Run	X			
E. California Coast	X			
F. Klamath River Fall-run	X	X	X	X
G. Southern Oregon/ Northern California Coasts	X	X	X	
H. Lower Columbia River	X	X	X	X
I. Willamette River Spring-run	X	X	X	X
J. Snake River Fall-run	X	X	X	X
K. Snake River Spring-Summer-run	X	X	X	X
L. Upper Columbia Spring-run	X	X	X	X
Coho Salmon				
A. Central California Coast	X			
B. Southern Oregon/ Northern California Coasts	X			
C. Oregon Coast	X		X	X
D. Lower Columbia River	X			

INDICATOR EVALUATION

Rather than develop an unique suite of indicators for this report, we have relied on the extensive previous work in evaluating the status of salmon populations and ESUs on the Pacific coast (Allendorf et al. 1997, Wainwright and Kope 1999, McElhany et al. 2000, Lindley et al. 2007). In particular, we selected indicators that were not inconsistent with these previous efforts and also the Viable Salmon Population (VSP) characteristics (McElhany et al. 2000) that are the foundation of current conservation and recovery planning efforts for Pacific salmonids; in addition, they are the bases for on-going evaluation of status updates of Pacific salmonid populations. McElhany et al. (2000) described four characteristics of populations that should be considered when assessing viability: abundance, productivity, diversity, and spatial structure. Since a high priority of the IEA effort it to develop frameworks that can expand to include new data and address multiple issues (e.g., protected species, fisheries, and ecosystem health), we felt it most appropriate to use indicators that are used in status reviews and ESA recovery planning documents (Table S1, S2). From this list of potential indicators, we selected those with the most widespread data availability (to allow for comparisons across species and regions) and with most relevance to the state of the marine ecosystem. The following sections describe the indicators we considered as measures of stock abundance and condition.

Table S2. Key indicators for salmon, identified during the ESA listing and recovery planning processes. Indicators categories chosen for this analysis are in *bold italic* font.

Indicator	Selection/Deselection Reasoning
Abundance	
Spawning escapement	Widely measured; key measure of reproductive population
Ocean abundance (recruitment)	Requires stock-specific harvest rate estimates; not widely available
Juvenile abundance	Not widely available, but key indicator of reproduction for some ESUs
Population Condition	
Population growth rate (lambda)	Widely available, standard measure of population trend
Natural return ratio (NRR)	A measure of sustainability of the natural component of mixed hatchery-natural stocks; requires both age-structure and natural proportion data, and knowledge of the relative fitness of hatchery fish.
Intrinsic rate of increase	Widely available, but depends on a specific formulation of density dependence.
Proportion of natural spawners	Widely available
Genetic diversity	Indicator of stock genetic integrity and effectiveness of natural production
Age structure diversity	Available for most Chinook salmon stocks; a quantifiable measure of phenotypic diversity; indicator of harvest-related risk
Population spatial structure	Available for few stocks.

POTENTIAL INDICATORS FOR ASSESSING ABUNDANCE (POPULATION SIZE)

Monitoring population size provides information of use both for protected species conservation and for harvest management. We considered three primary indicators of abundance, and chose to focus on one (spawning escapement) as the most widely available and relevant (Table S2).

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

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

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

POTENTIAL INDICATORS FOR ASSESSING POPULATION CONDITION

There are a number of potential metrics for assessing the condition of a managed salmon population (Table S2). These fall into the broad categories of population growth/productivity, diversity, and spatial structure (McElhany et al. 2000). We considered the seven commonly used metrics, and based on data availability and relevance, chose three of those metrics (population growth rate, hatchery contribution, and age-structure diversity) to reflect a range of assumptions about the effects of various stressors on the populations.

1. Population growth rate–Calculated as the proportional change in abundance between successive generations, population growth rate is an indication of the population’s resilience. In addition, growth rate can act as a warning of

critical abundance trends that can be used for determining future directions in management. Also, the viability of a population is dependent in part on maintaining life-history diversity in the population. Because of limited information on hatchery fish and natural return ratio (see below) this value includes hatchery-origin fish.

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

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

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

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

6. Age structure diversity–A diverse age structure is important to improve population resilience. Larger, older Chinook salmon produce more and larger eggs (Healey and Heard 1984). Therefore, they produce a brood that may contribute proportionally more to the later spawning population than broods from younger, smaller fish. However, the diversity of ages including younger fish is important to accommodate variability in the environment. If mortality on any given cohort is great, there is benefit to having younger spawners. An individual that produces offspring that return at different adult ages (i.e., overlapping generations) may increase the likelihood of contributing to future generations when environmental conditions are less than favorable one year to the next. This bet hedging is a critical aspect of Chinook salmon that allow it to naturally mitigate year-to-year environmental variability (Heath et al. 1999). Adult age structure is not an issue for coho salmon, which in our region spawn predominantly at age three (with the exception of a small

proportion of younger male 'jacks'). While coho salmon in our region spawn predominantly at a single age, Chinook salmon typically spawn over an age range of 3 or 4 years, and exhibit differences in spawning age both among years and among populations. Data are available for most Chinook salmon populations of commercial importance or of ESA interest ESUs (e.g., Sacramento River Winter-run), although data are typically stock/population specific and might not be representative of an ESU.

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

SELECTING APPROPRIATE STOCKS/POPULATIONS FOR EVALUATION OF ABUNDANCE AND CONDITION

Stock selection was based on economic and ecosystem importance, geographic and life-history diversity, and data availability. This resulted in selections consistent with current ESU delineations. Because of regional differences in the availability of data, we considered stocks and data series separately within two regions: California (including southern Oregon south of Cape Blanco) and Oregon-Washington coasts (Cape Blanco to the mouth of the Strait of Juan de Fuca). For each ESU, a variety of data series are available; each series has been used in management documents, status reports, and/or the scientific literature. Any data series that was less than 15 years long was removed; within each ESU, all data series were truncated to match the shortest series. Available data series meeting these criteria for given ESUs are listed in Tables S3-S6. It should be noted that in many cases we used data that were not used for recent ESA status updates. Many of the available time series are at the stock or population scale and may not be representative of the whole ESU (the listing unit for ESA efforts) and therefore may not be appropriate for evaluating the status of an ESU. For our purposes we determined that development of the indicators and ecosystem models using stock/population scale measures was appropriate at this initial stage of development of IEA and we should be able to accommodate ESU representative data as rigorous monitoring programs are established.

For California ESUs (Tables S3 & S4), the data series were compiled from a variety of sources and are presented in Williams et al. (2011), PFMC (2012), and Spence and Williams (2011). Because of the diversity of data types available, indicators for each stock were selected based on their availability, time series lengths, and scientific support. Data series that were used are highlighted in the tables.

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

When data were only available for a portion of an ESU (e.g., single stream or tributary, but not necessarily representative of the whole ESU) and no ESU-wide estimates were available, we used these data as a proxy for the ESU unless it was not recent enough or was incomplete (Table S3). If data restrictions or reporting required multiple series be used for a given indicator within a single ESU, we computed an ESU-wide average (e.g., Table S3, Central Valley Spring-run). To do this, series were standardized and then averaged across populations within ESUs. These standard scores represent the index for abundance or conditions for that ESU. Data series that represented similar values (e.g., escapements) were weighted by absolute spawning abundance.

APPROPRIATE INDICATORS

We evaluated abundance using the metric of escapement of natural-origin spawners. Selection rationale for assessing only escapement and no other abundance metrics is listed in Table S2. The populations/ESUs that had sufficiently met the criteria for inclusion in the analyses are listed in Tables S3 and S5. When ESU-wide estimates were available and sufficient they were used. If data were only available at the sub-ESU level, escapement values from the component subpopulations were used. As well, we only used data beginning in 1985 so that, when possible, the longer time series could be compared equivalently among populations. Data series for multiple subpopulations were standardized by subtracting the series mean and dividing by the series standard deviation. If a consolidated index for the stock was needed we computed an annual weighted average of the standardized series, with weights proportional to the average abundance for each subpopulation.

To evaluate condition we restricted our analyses to examination of population growth rate, proportion of natural-origin spawners, and age-structure diversity. Selection rationale for assessing only these metrics of condition and no other condition metrics is listed in Table S2. The populations/ESUs that had sufficiently met the criteria for estimation of condition are listed in Tables S4 and S6.

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

Table S3. California ESUs/Stocks and data available for abundance estimates. Each of these series met the criteria for inclusion in the analyses and was used.

Population	Data Available: Escapement	Period
Chinook Salmon		
Central Valley Fall Run	Escapement to system	1983-2012
Central Valley Late Fall Run	Escapement to system	1971-2011
Central Valley Winter Run	Escapement to system, carcass survey	2001-2011
Central Valley Spring Run	Escapement Antelope Creek	~1982-2012
	Escapement Battle Creek	1989-2012
	Escapement Big Chico Creek	1970-2012
	Escapement Butte Creek	1970-2012
	Escapement Clear Creek	1992-2012
	Escapement Cottonwood Creek	1973-2012
	Escapement Deer Creek	1970-2012
	Escapement Feather River Hatcher	1970-2012
	Escapement Mill Creek	1970-2012
Klamath R. Fall Run	Escapement to system (Klamath+Trinity)	1978-Present

Population	Data Available: Escapement	Period
SOr-NCa Chinook Fall	Huntley Park (Rogue River)	1973-2013
Cal Coastal Chinook	Tomki Creek (Live/Dead Counts)	1979-Present
	Cannon Creek (live/Dead Counts)	1981-Present
	Sprowl Creek (Live/Dead Counts)	1974-Present
Coho salmon		
SOr-NCa Coho	Huntley Park (Rogue River)	1973-2013
California Coastal Coho	Lagunitas Creek coho salmon reddcounts	1995-2012

Table S4. Data series that met the criteria for inclusion in the condition analyses of California ESUs. Each of these series met the criteria for inclusion in the analyses and was used.

Population	Data Series on Condition	Period
Chinook Salmon		
CV Fall Sacramento R. Fall Run	Hatchery contribution	1983 -2012
	Population Growth Rate	1983-2012
Klamath R. Fall Run	Klam Age diversity (S-W)	1981-2012
	Hatchery contribution	1978 -2011
	Population Growth Rate	1981-2013
SOr-NCa Chinook Fall	Rogue Age Diversity	1980-2013
	Hatchery Contribution	1972-2011

Table S5. Oregon-Washington ESUs/stocks and data available for abundance estimates. Each of these series met the criteria for inclusion in the analyses and was used.

Stock/ESU	Data Available: Escapement	Period
Chinook Salmon		
Lower Columbia River ESU	Clatskanie River Fall	1974-2006
	Coweeman River Fall	1977-2010
	Elochoman River Fall	1975-2010
	Grays River Fall	1964-2010
	Kalama River Fall	1964-2010
	Kalama River Spring	1980-2008
	Lewis River	1964-2010
	Lewis River Fall	1973-2009
	Lower Cowlitz River Fall	1977-2010
	Mill Creek Fall	1980-2010
	North Fork Lewis River Spring	1980-2008
	Sandy River Fall (Bright)	1981-2009
	Sandy River Spring	1981-2008
	Toutle River Fall	1964-2009
	Upper Cowlitz River Spring	1980-2009

Stock/ESU	Data Available: Escapement	Period
	Upper Gorge Tributaries Fall	1964-2008
	Washougal River Fall	1977-2010
	White Salmon River Fall	1976-2009
Snake River Fall-run ESU	Snake River Lower Mainstem Fall	1975-2012
Snake River Spring/Summer-run ESU	Bear Valley Creek	1960-2012
	Big Creek	1957-2012
	Camas Creek	1963-2012
	Catherine Creek Spring	1955-2011
	Chamberlain Creek	1985-2012
	East Fork Salmon River	1960-2012
	East Fork South Fork Salmon River	1958-2012
	Grande Ronde River Upper Mainstem	1955-2011
	Imnaha River Mainstem	1949-2011
	Lemhi River	1957-2012
	Loon Creek	1957-2012
	Lostine River Spring	1959-2011
	Marsh Creek	1957-2012

Stock/ESU	Data Available: Escapement	Period
	Minam River	1954-2012
	Pahsimeroi River	1986-2012
	Salmon River Lower Mainstem	1957-2012
	Salmon River Upper Mainstem	1962-2012
	Secesh River	1957-2011
	South Fork Salmon River Mainstem	1958-2012
	Sulphur Creek	1957-2012
	Tucannon River	1979-2011
	Valley Creek	1957-2012
	Wenaha River	1964-2012
	Yankee Fork	1961-2011
Upper Columbia River Spring-run ESU	Entiat River	1960-2011
	Methow River	1960-2011
	Wenatchee River	1960-2011
Willamette River ESU	Clackamas River Spring	1974-2011
	McKenzie River Spring	1970-2012

Stock/ESU	Data Available: Escapement	Period
Coho Salmon		
Lower Columbia River ESU	Clackamas River	1970-2010
	Sandy River	1970-2010
Oregon Coast ESU	Alsea River	1990-2012
	Beaver Creek	1990-2012
	Coos River	1990-2012
	Coquille River	1990-2012
	Floras/New River	1990-2012
	Lower Umpqua River	1990-2012
	Middle Umpqua River	1990-2012
	Necanicum River	1990-2012
	Nehalem River	1990-2012
	Nestucca River	1990-2012
	North Umpqua River	1990-2012
	Salmon River	1990-2012
	Siletz River	1990-2012
	Siltcoos Lake	1990-2012

Stock/ESU	Data Available: Escapement	Period
	Siuslaw River	1990-2012
	Sixes River	1990-2012
	South Umpqua River	1990-2012
	Tahkenitch Lake	1990-2012
	Tenmile Lake	1990-2012
	Tillamook Bay	1990-2012
	Yaquina River	1990-2012

Table S6. Oregon-Washington ESUs/stocks and data available for condition estimates. These data series met the criteria for inclusion in the condition analyses Data types available are: HC – hatchery contribution to natural spawning; PGR – population growth rate; Age – spawning age structure. Period is the period of availability for the longest series for that population.

Stock/ESU	Population	Data Types	Period
Chinook Salmon			
Lower Columbia River ESU	Clatskanie River Fall	HC, PGR, Age	1974-2006
	Coweeman River Fall	HC, PGR	1980-2010
	Elochoman River Fall	HC, PGR	1975-2010
	Grays River Fall	HC, PGR	1964-2010
	Kalama River Fall	HC, PGR	1964-2010
	Kalama River Spring	PGR	1980-2008
	Lewis River	HC, PGR	1978-2010
	Lewis River Fall	PGR	1977-2009
	Lower Cowlitz River Fall	HC, PGR	1973-2009
	Mill Creek Fall	HC, PGR	1980-2010
	North Fork Lewis River Spring	PGR	1980-2008
	Sandy River Fall (Bright)	HC, PGR, Age	1981-2006
	Sandy River Spring	HC, PGR, Age	1981-2008
	Toutle River Fall	PGR	1964-2009

Stock/ESU	Population	Data Types	Period
	Upper Cowlitz River Spring	PGR	1980-2009
	Upper Gorge Tributaries Fall	HC, PGR	1964-2008
	Washougal River Fall	HC, PGR	1977-2010
	White Salmon River Fall	HC, PGR, Age	1976-2009
Snake River Fall-run ESU	Snake River Lower Main. Fall	HC, PGR, Age	1975-2012
Snake River Spring/Summer-run ESU	Bear Valley Creek	HC, PGR, Age	1960-2012
	Big Creek	HC, PGR, Age	1957-2012
	Camas Creek	HC, PGR, Age	1963-2012
	Catherine Creek Spring	HC, PGR, Age	1955-2011
	Chamberlain Creek	HC, PGR, Age	1985-2012
	East Fork Salmon River	HC, PGR, Age	1960-2012
	E. Fork S. Fork Salmon River	HC, PGR, Age	1958-2012
	Grande Ronde River - Upper Main	HC, PGR, Age	1955-2011
	Imnaha River Mainstem	HC, PGR, Age	1949-2011
	Lemhi River	HC, PGR, Age	1957-2012
	Loon Creek	HC, PGR, Age	1957-2012
	Lostine River Spring	HC, PGR, Age	1959-2011

Stock/ESU	Population	Data Types	Period
	Marsh Creek	HC, PGR, Age	1957-2012
	Minam River	HC, PGR, Age	1954-2012
	Pahsimeroi River	HC, PGR, Age	1986-2012
	Salmon River Lower Mainstem	HC, PGR, Age	1957-2012
	Salmon River Upper Mainstem	HC, PGR, Age	1962-2012
	Secesh River	HC, PGR, Age	1957-2011
	South Fork Salmon River Mainstem	HC, PGR, Age	1958-2012
	Sulphur Creek`	HC, PGR, Age	1957-2012
	Tucannon River	HC, PGR, Age	1979-2011
	Valley Creek	HC, PGR, Age	1957-2012
	Wenaha River	HC, PGR, Age	1964-2012
	Yankee Fork	HC, PGR, Age	1961-2011
Upper Columbia River Spring-run ESU	Entiat River	HC, PGR, Age	1960-2011
	Methow River	HC, PGR, Age	1960-2011
	Wenatchee River	HC, PGR, Age	1960-2011
Willamette River ESU	Clackamas River Spring	HC, PGR, Age	1974-2011

Stock/ESU	Population	Data Types	Period
	McKenzie River Spring	HC, PGR, Age	1970-2012
Coho Salmon			
Oregon Coast ESU	Alsea River	HC, PGR	1990-2012
	Beaver Creek	HC, PGR	1990-2012
	Coos River	HC, PGR	1990-2012
	Coquille River	HC, PGR	1990-2012
	Floras/New River	HC, PGR	1990-2012
	Lower Umpqua River	HC, PGR	1990-2012
	Middle Umpqua River	HC, PGR	1990-2012
	Necanicum River	HC, PGR	1990-2012
	Nehalem River	HC, PGR	1990-2012
	Nestucca River`	HC, PGR	1990-2012
	North Umpqua River	HC, PGR	1990-2012
	Salmon River	HC, PGR	1990-2012
	Siletz River	HC, PGR	1990-2012

Stock/ESU	Population	Data Types	Period
	Siltcoos Lake	HC, PGR	1990-2012
	Siuslaw River	HC, PGR	1990-2012
	Sixes River	HC, PGR	1990-2012
	South Umpqua River	HC, PGR	1990-2012
	Tahkenitch Lake	HC, PGR	1990-2012
	Tenmile Lake	HC, PGR	1990-2012
	Tillamook Bay	HC, PGR	1990-2012
	Yaquina River	HC, PGR	1990-2012

STATUS AND TRENDS OF SALMON ABUNDANCE AND CONDITION

MAJOR FINDINGS OF SALMON ABUNDANCE AND CONDITION

A number of salmon population groups [ESUs] have demonstrated declines over the last ten years. For Chinook salmon, the Lower Columbia and Willamette Spring-run data series exhibited the steepest declines, while the Central Valley Fall-run, Spring-run, and Sacramento River winter-run and the Southern Oregon/Northern California series exhibited more moderate declines. On the positive side, Snake River fall-run, Snake River spring/summer-run, and Upper Columbia River spring-run Chinook demonstrated increases. All Chinook salmon ESUs were near their longer-term (25-30 year) average abundance.

For coho salmon, all recent abundance averages were near their longer-term averages, but the California Coast and Southern Oregon/Northern California Coasts series demonstrated recent steep declines, while the Lower Columbia River showed an increase. Oregon Coast coho salmon demonstrated no significant recent trend.

SUMMARY AND STATUS OF TRENDS OF SALMON ABUNDANCE AND CONDITION

Both short- and long-term trends are reported in this summary. An indicator is considered to have changed over the short-term if the trend over the last 10 years (2003-2012) the series showed a significant increasing or decreasing slope. An indicator is considered to be above or below long-term norms if the mean of the last 10 years of the time series differs from the mean of the full time series by more than 1.0 s.d. of the full time series. “Long-term” trends reflect data since 1985, not historical abundance. A major motivation of presenting long- and short-term trends is to distinguish between stocks/populations that were once very large and suffered historical declines but have stabilized at lower abundances from populations with ongoing declines. This was a particular issue for populations with very long time series of abundance (e.g., certain Columbia River Chinook salmon populations). Information on historical abundances indicate that for many if not most of these populations current values are now far below historical values – so caution should be used when interpreting “long-term,” and not associate it with historically robust populations. We did not include data prior to 1985 in our analysis because data quality and consistency is much lower in the early years, and such long time series are not available for most California populations.

CHINOOK SALMON: ABUNDANCE

Generally all California Chinook salmon stocks were within 1 s.d. of their long term average (since 1985). However, during the last ten years there has been a significant decline in abundance of most California populations examined, with Central Valley Winter Run Chinook salmon at extremely low abundances from 2007-2011 (Figure S1). Largely, though, this relates to a reduction from series highs during 2000s and a return to previous values (Figure S2). For the northern Chinook salmon stocks, recent abundances were also close to average, except for a positive deviation for the Snake River Fall-run (Figure S1). There is a notable contrast in recent trends between steep declines in the lower Columbia River and Willamette River stocks and increases in the upper Columbia and Snake River stocks (Figure S1). As for the California stocks, the observed steep declines follow higher abundances in the early 2000s. This suggests that 10 years may be too short a time frame for evaluating status of these stocks.

CHINOOK SALMON: CONDITION

With a few noteworthy exceptions, both the recent trends and recent average levels of condition indices for Chinook salmon have been near long-term average values (Figure S3). In general, there are significant downward trends in condition for Lower Columbia River and Willamette River series, with the exception of improving trends for Willamette River percent natural spawners and age diversity. Klamath River Fall-run also exhibits an upward trend for percent natural spawners. Notably, Lower Columbia River percent natural spawners falls into the “low and decreasing” quadrant, reflecting a long-term decline in this indicator (Figure S4). Similarly, the Central Valley Fall-run falls on the border of the “low and decreasing” quadrant for percent natural spawners.

COHO SALMON: ABUNDANCE

While recent abundance of all coho salmon stocks are near their long-term average (since 1985), there is a sharp contrast in recent trends (Figure S5). The Central California Coast and Southern Oregon/Northern California Coast stocks both exhibit steep declines following increased abundance in 2004, while the Lower Columbia River stock exhibited a fluctuating increase in recent years (Figure S6). The two northern stocks are both well above their historical low abundances in the 1990s.

COHO SALMON: CONDITION

There is no condition data available for the two southernmost coho salmon ESUs, and data for the two northern ESUs are limited to percent natural spawners (both stocks) and population growth rate (for the Oregon Coast stock). None of the data series exhibit significant recent trends, and both series for the Oregon Coast stock are near long-term 25-30 averages (Figure S7). Recent percent natural spawners for the Lower Columbia River stock is higher than the 25-30 year average. The Oregon Coast stock exhibits an encouraging long-term upward trend in percent natural spawners (Figure S8).

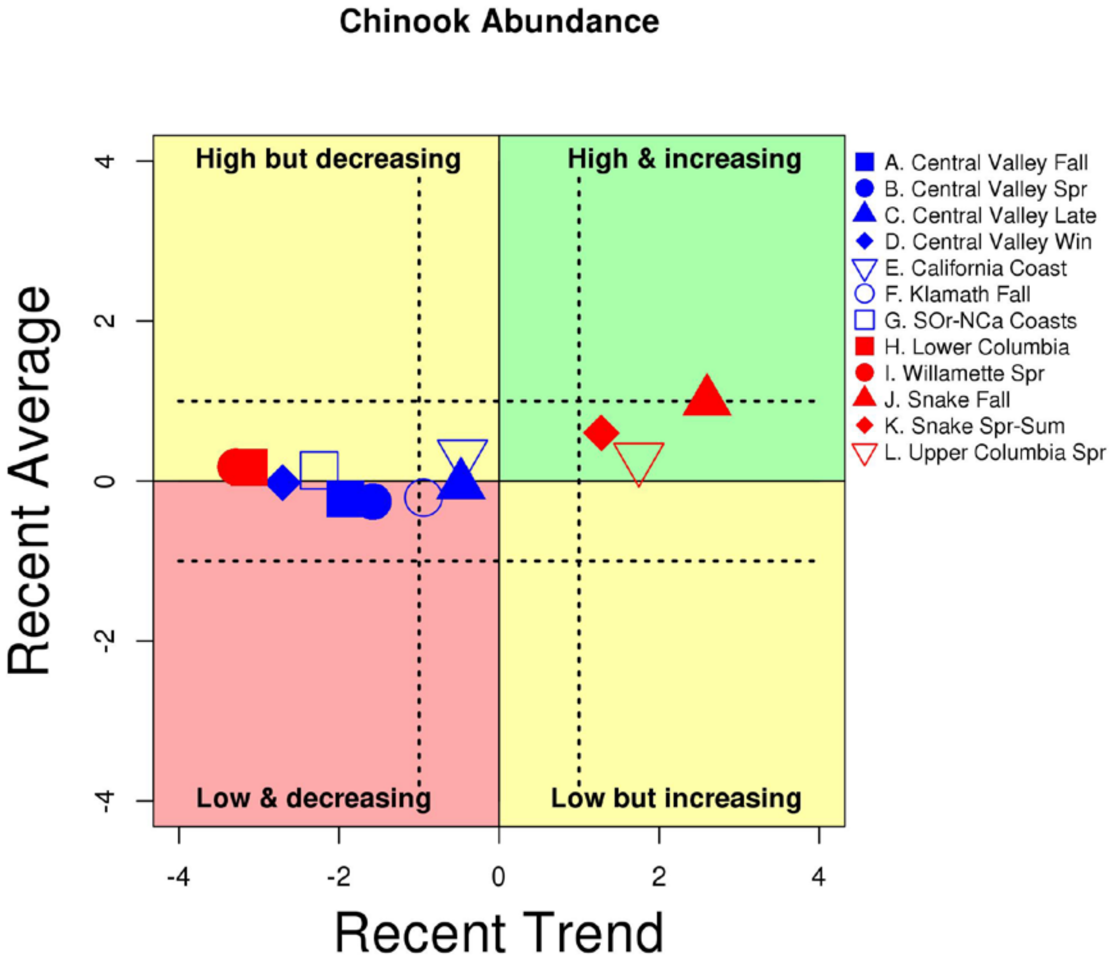
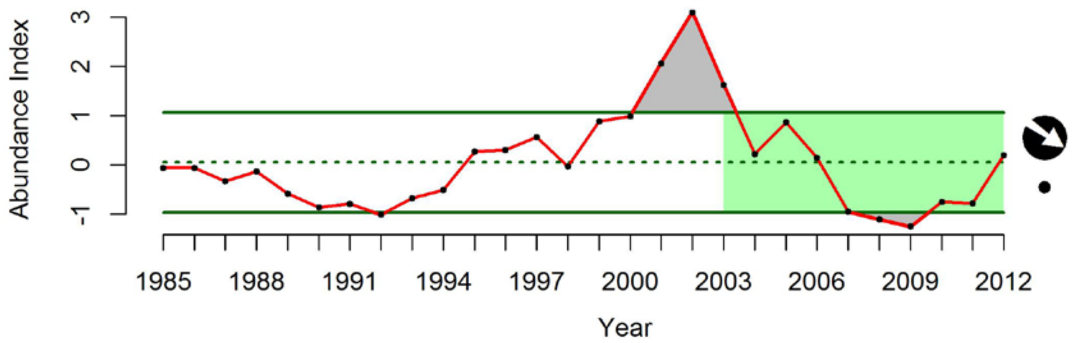
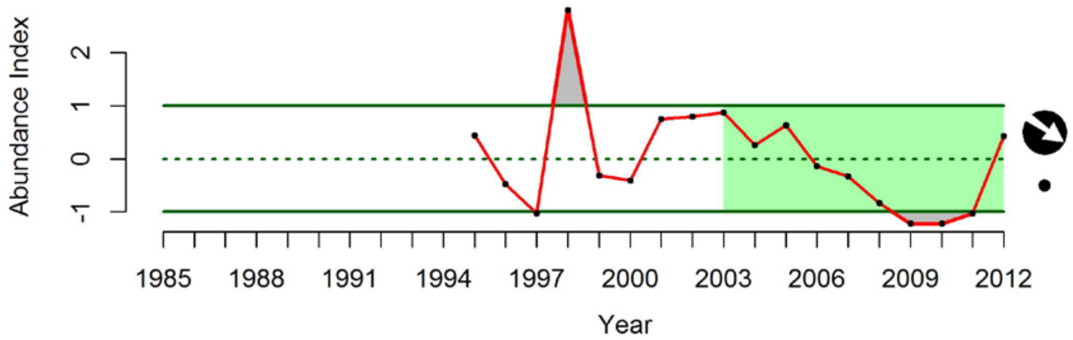


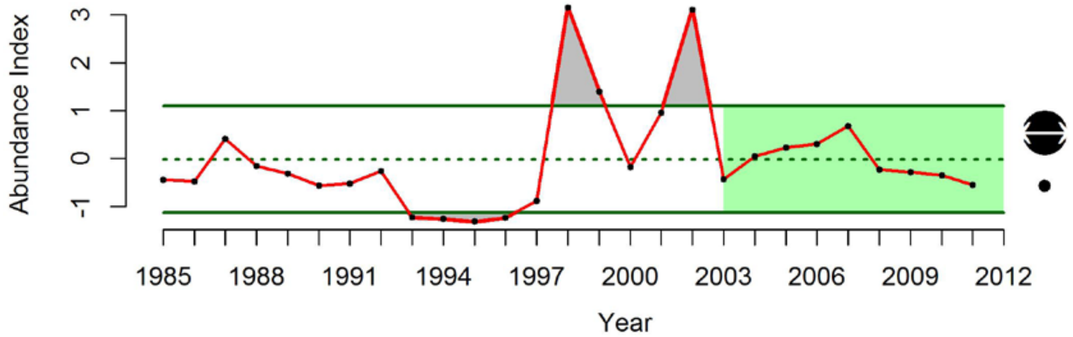
Figure S1. *Chinook salmon abundance.* Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. Dotted lines show ± 1.0 s.d. Populations listed correspond to data series in Tables S3 & S4.



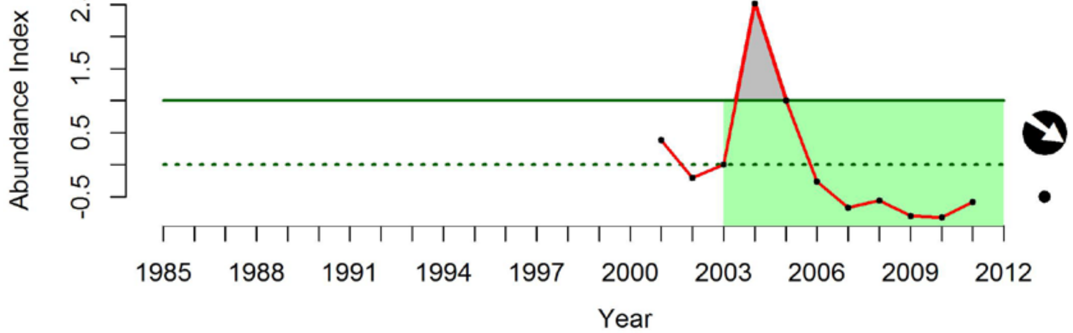
A. Central Valley Fall



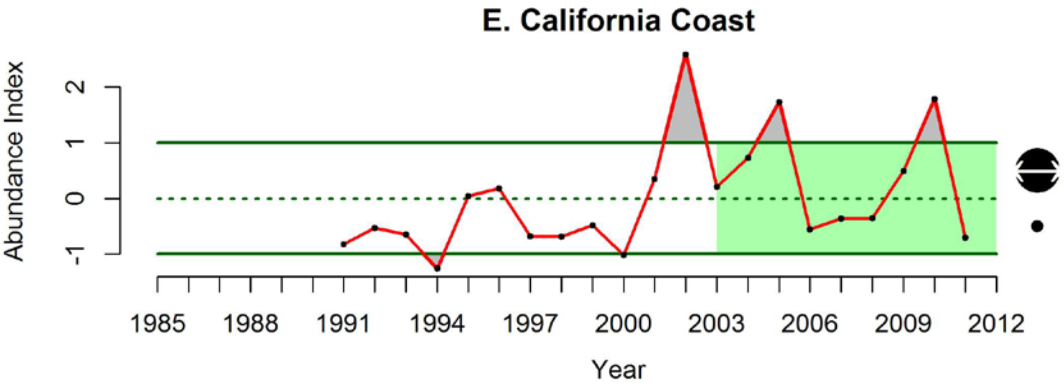
B. Central Valley Spr



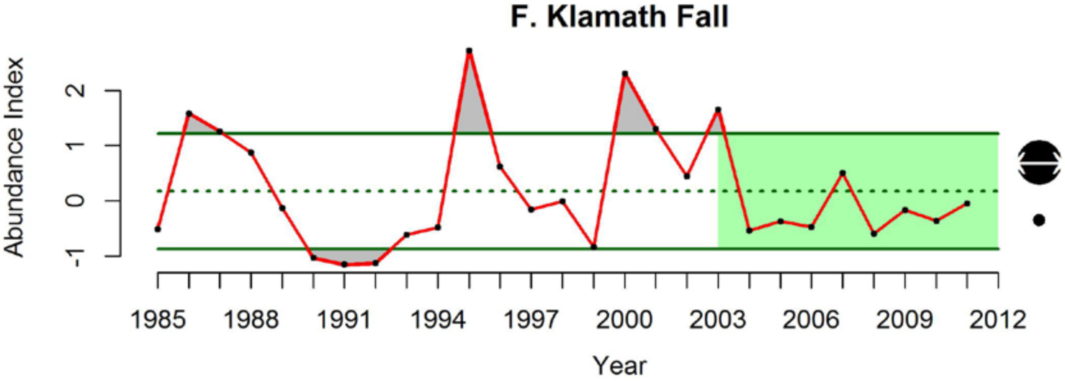
C. Central Valley Late



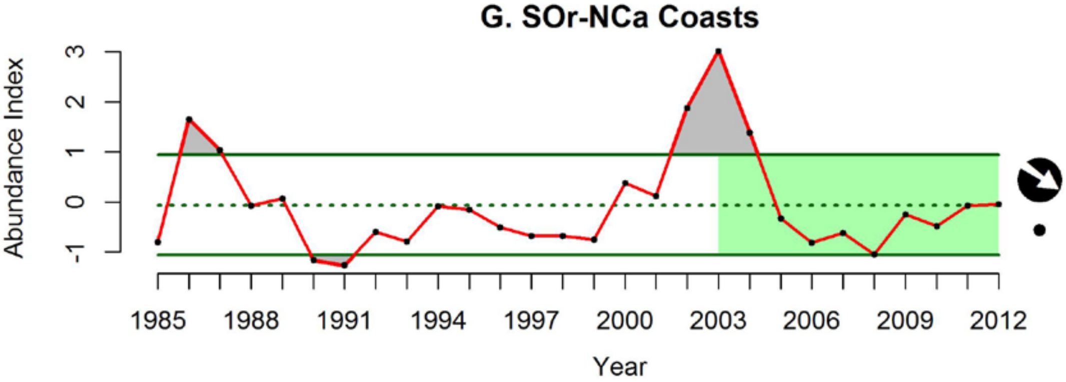
D. Central Valley Win



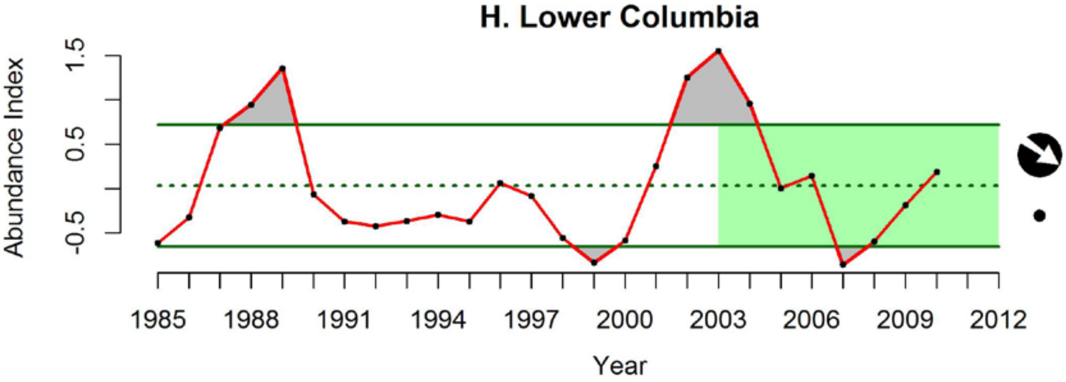
E. California Coast



F. Klamath Fall



G. SOr-NCa Coasts



H. Lower Columbia

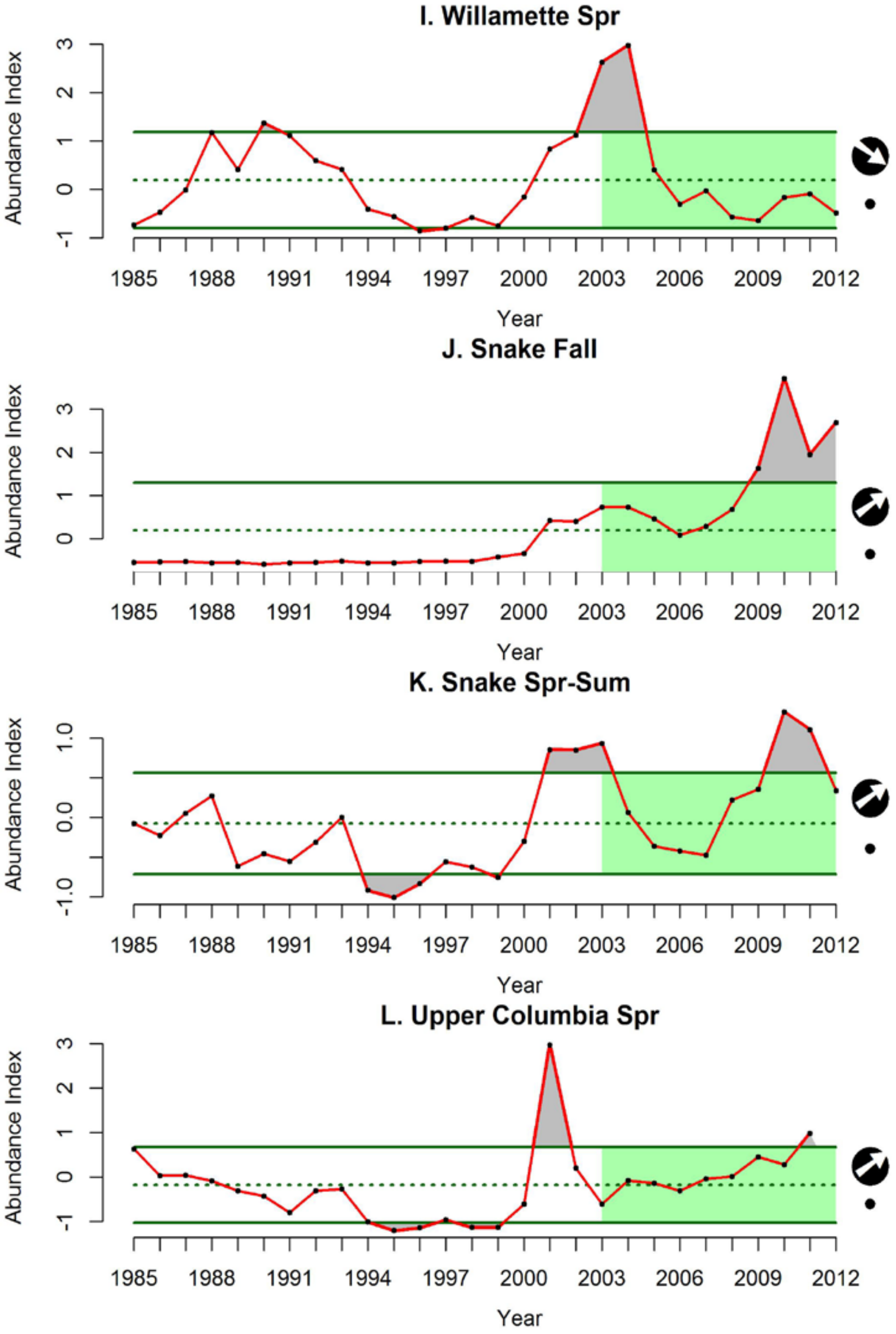


Figure S2. *Chinook salmon abundance*. The abundance index is calculated as anomalies (observed-mean/standard deviation). Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend was significant over the last 10-years. The lower symbol indicates whether the mean during the last 10 years was greater or less than or within one s.d. of the long-term mean. Population abbreviations correspond to populations listed in Tables S3 & S4. Abundances are shown as anomalies.

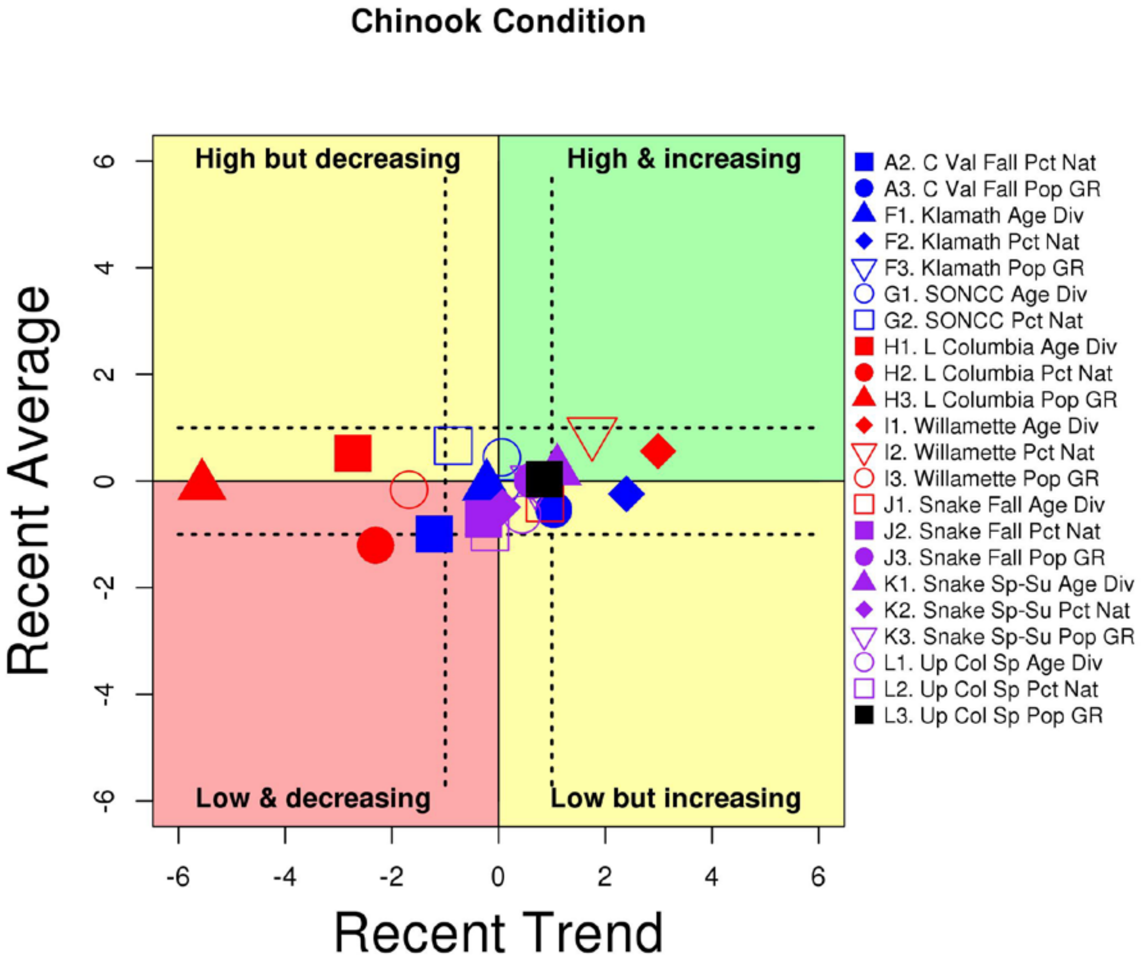
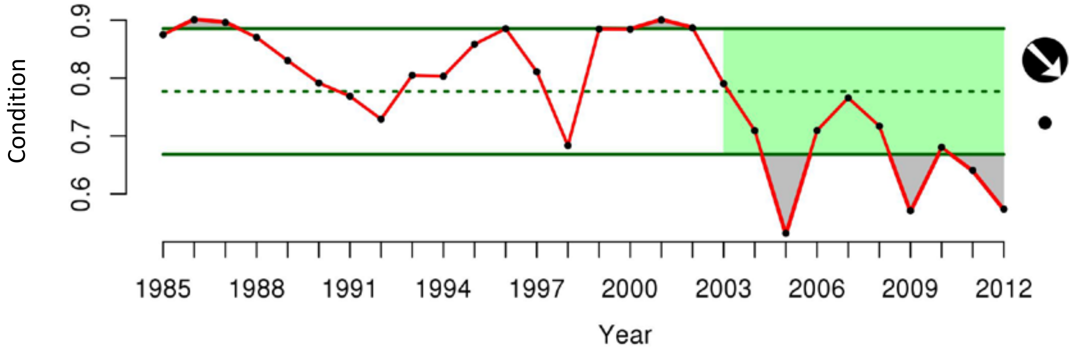
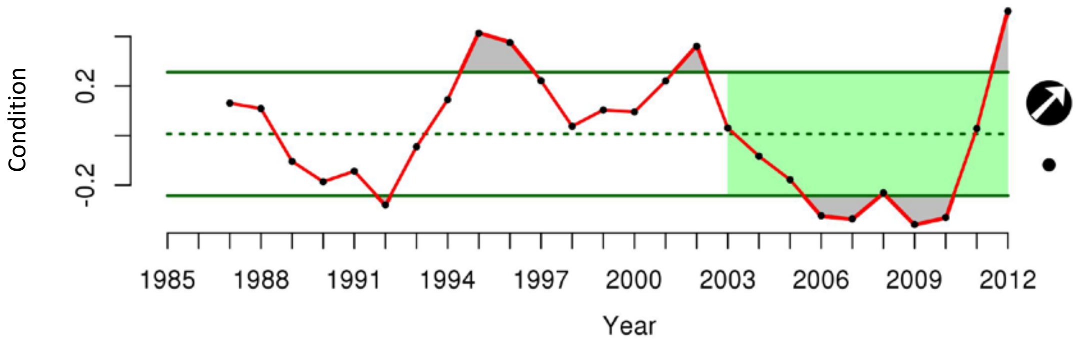


Figure S3. *Chinook salmon condition.* Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. Dotted lines show ± 1.0 s.d. When possible we evaluated percent natural spawners (Pct Nat), age-structure diversity (Age Div), and population growth rate (Pop GR).

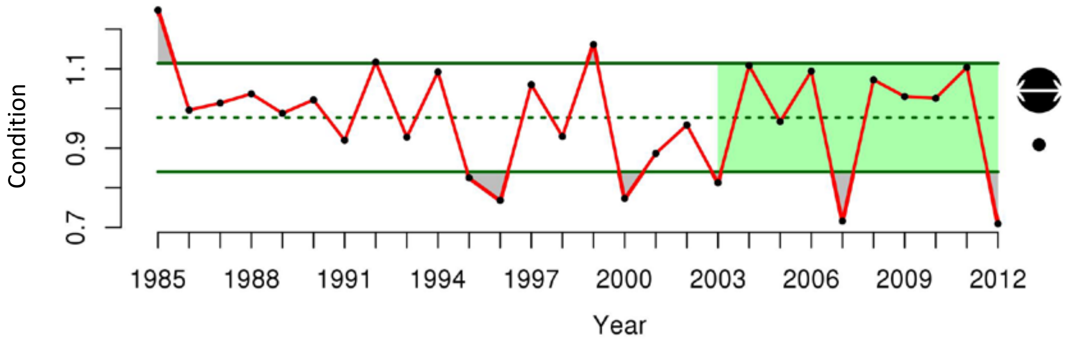
A2. C Val Fall Pct Nat



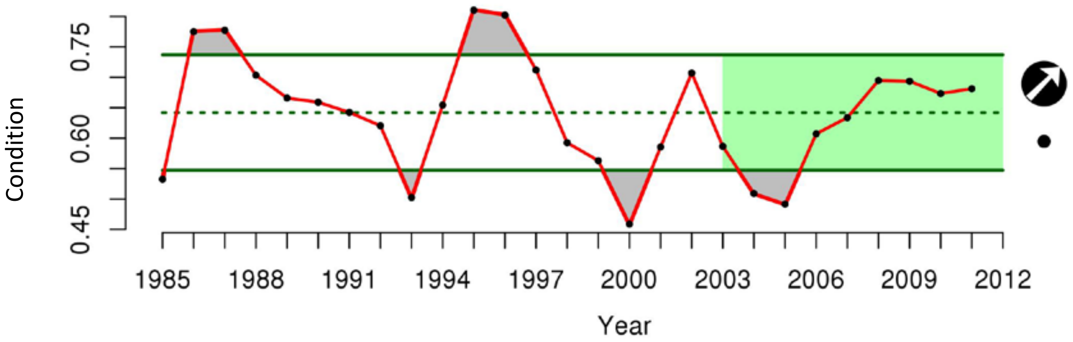
A3. C Val Fall Pop GR



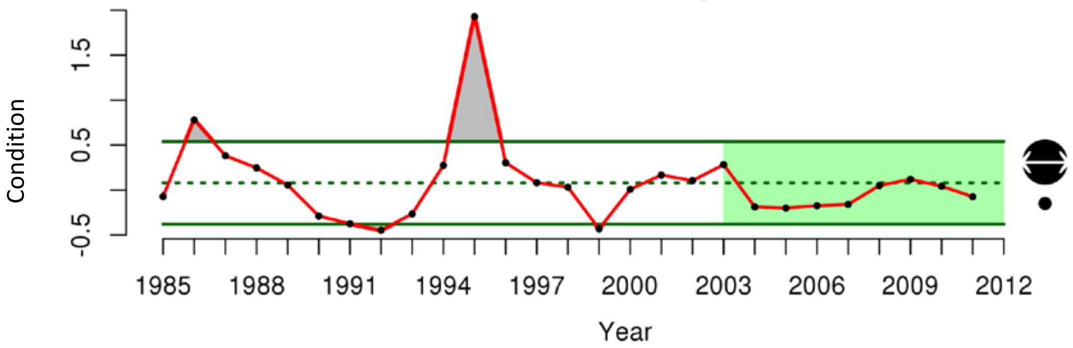
F1. Klamath Age Div



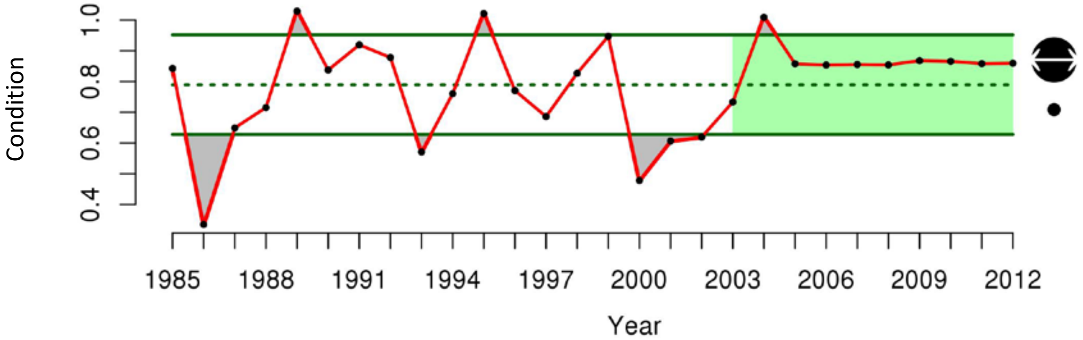
F2. Klamath Pct Nat



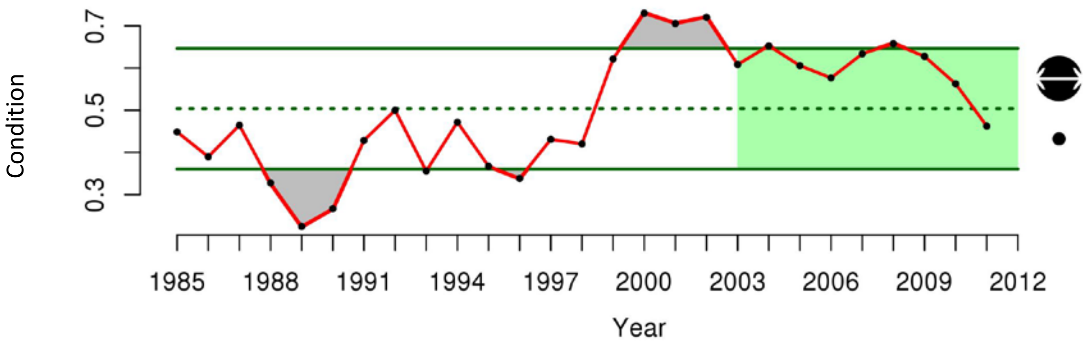
F3. Klamath Pop GR



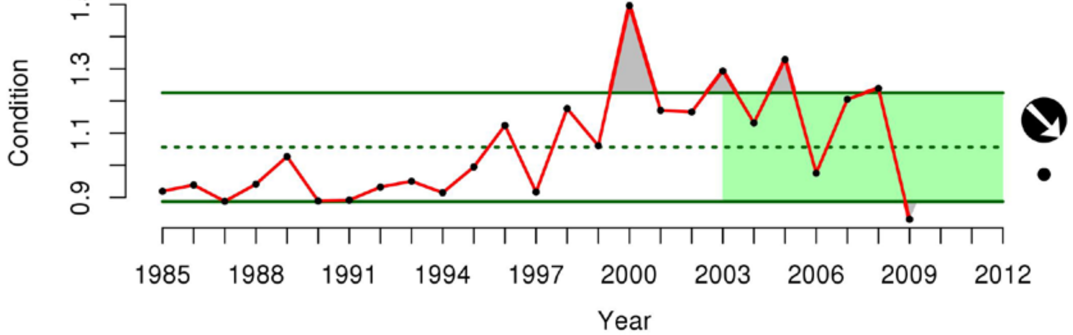
G1. SONCC Age Div

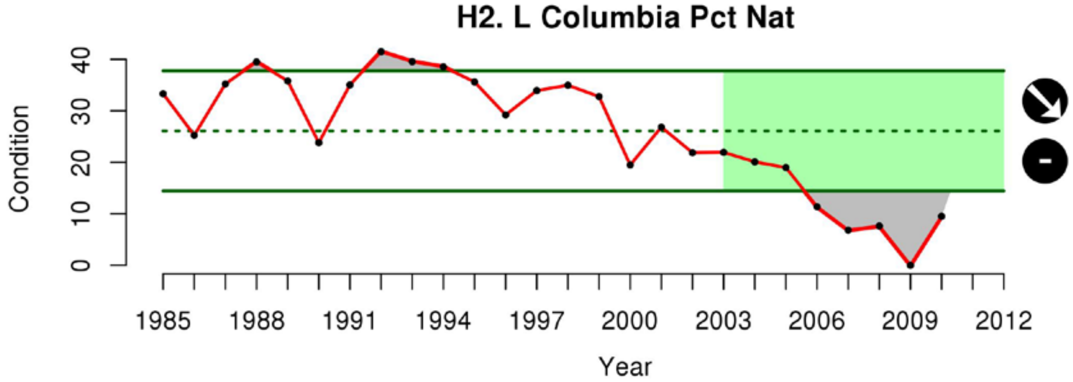


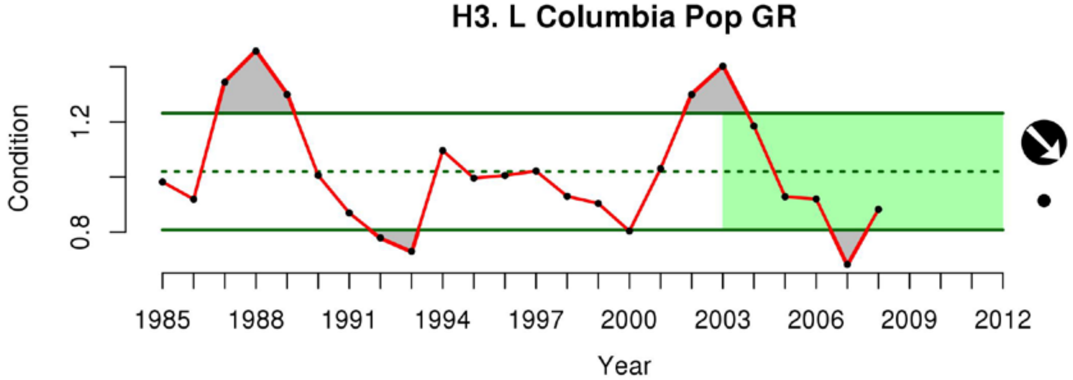
G2. SONCC Pct Nat

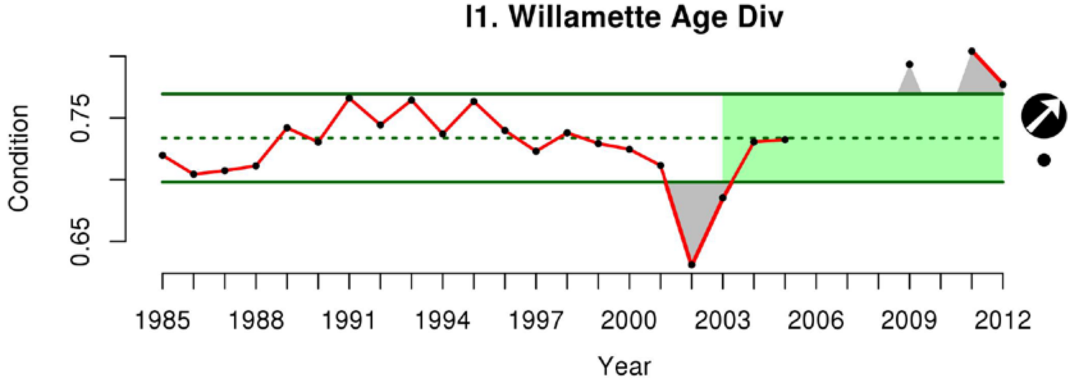


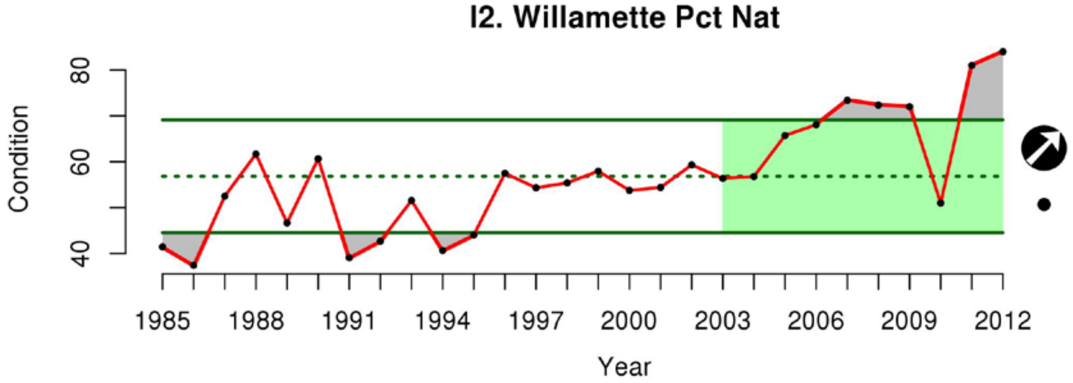
H1. L Columbia Age Div




H2. L Columbia Pct Nat


H3. L Columbia Pop GR


I1. Willamette Age Div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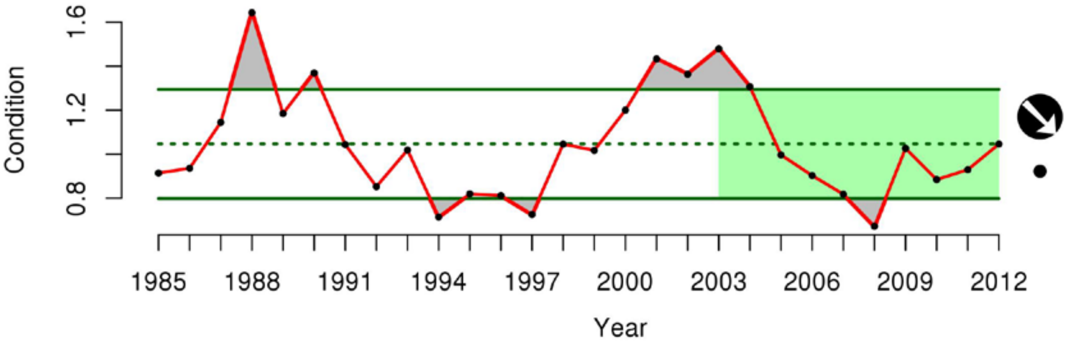
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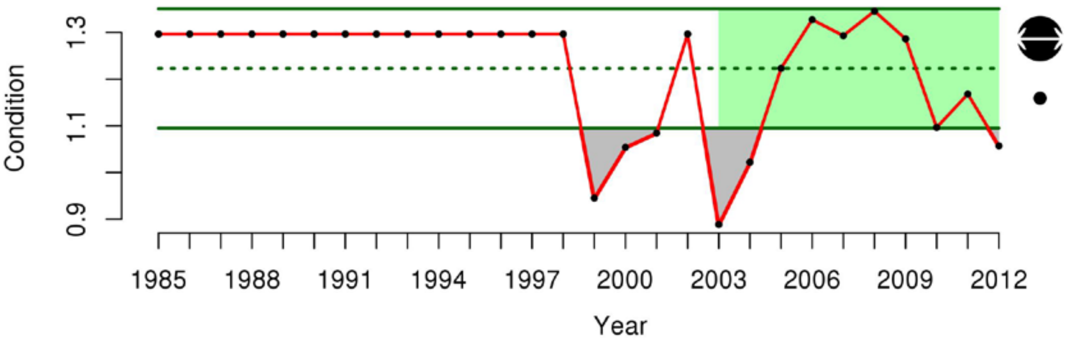
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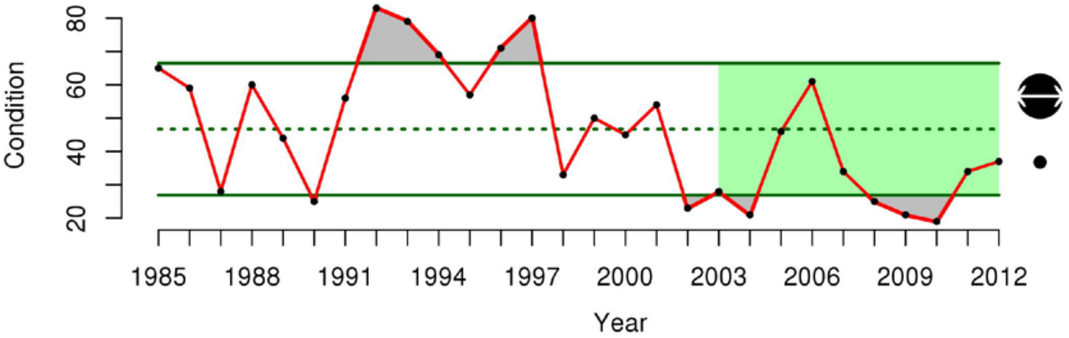
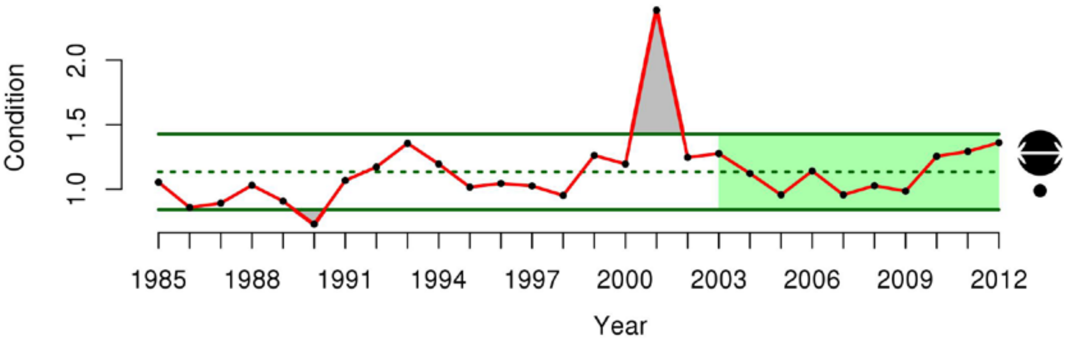
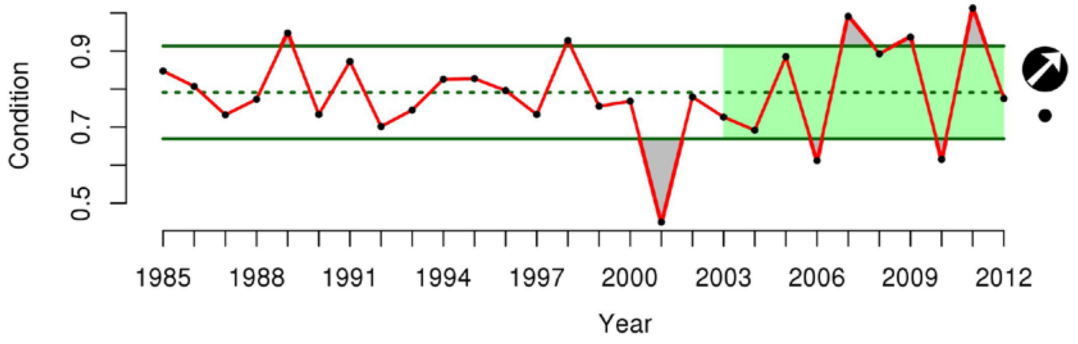
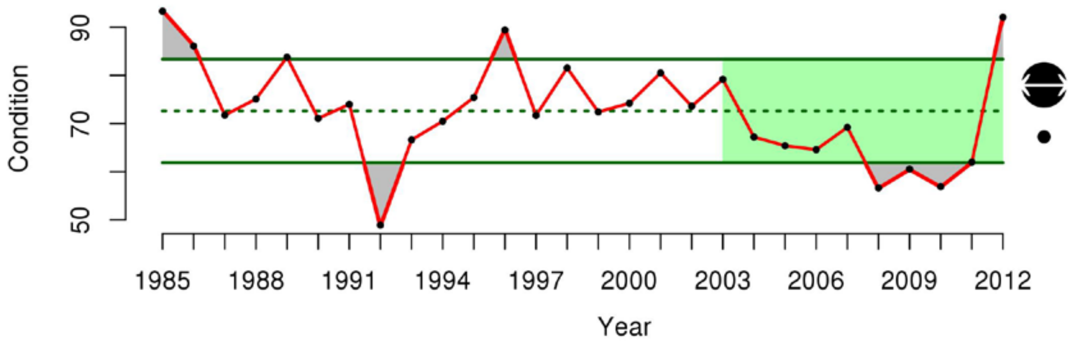

J2. Snake Fall Pct Nat


J3. Snake Fall Pop GR



K1. Snake Sp-Su Age Div



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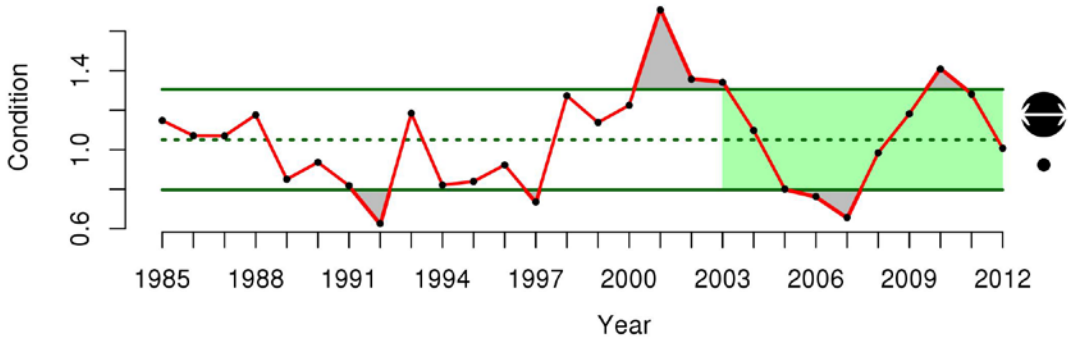
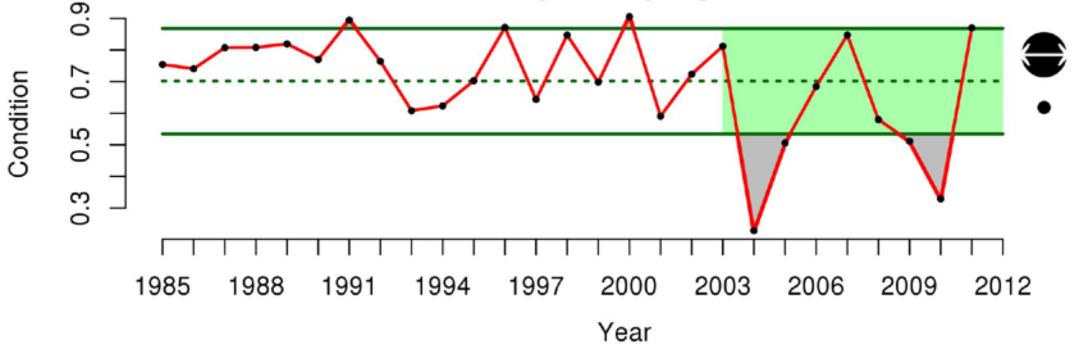


K3. Snake Sp-Su Pop GR



L1. Up Col Sp Age Div

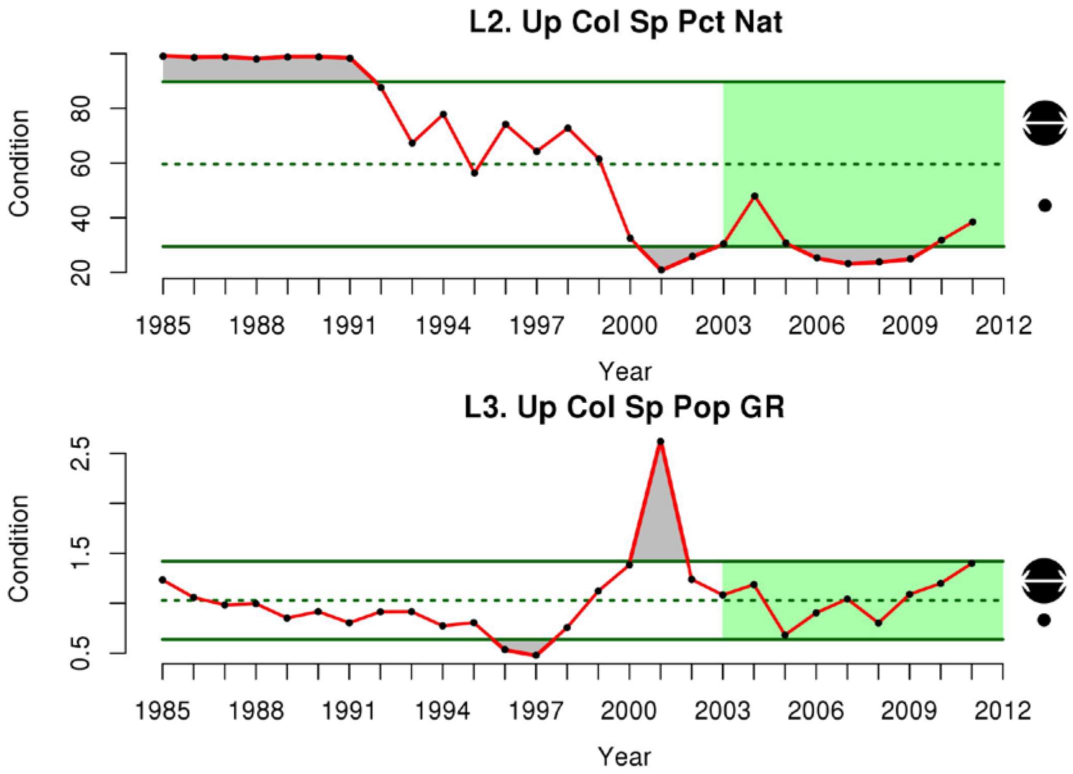


Figure S4. *Chinook salmon condition*. The series titles are titled by different populations (letters) and data series type (numbers). Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend was significant over the last 10-years . The lower symbol indicates whether the mean during the last 10 years was greater or less than or within one s.d. of the long-term mean. When possible we evaluated age-structure diversity (Age Div, 1), percent natural spawners (Pct Nat, 2), and population growth rate (Pop GR, 3).

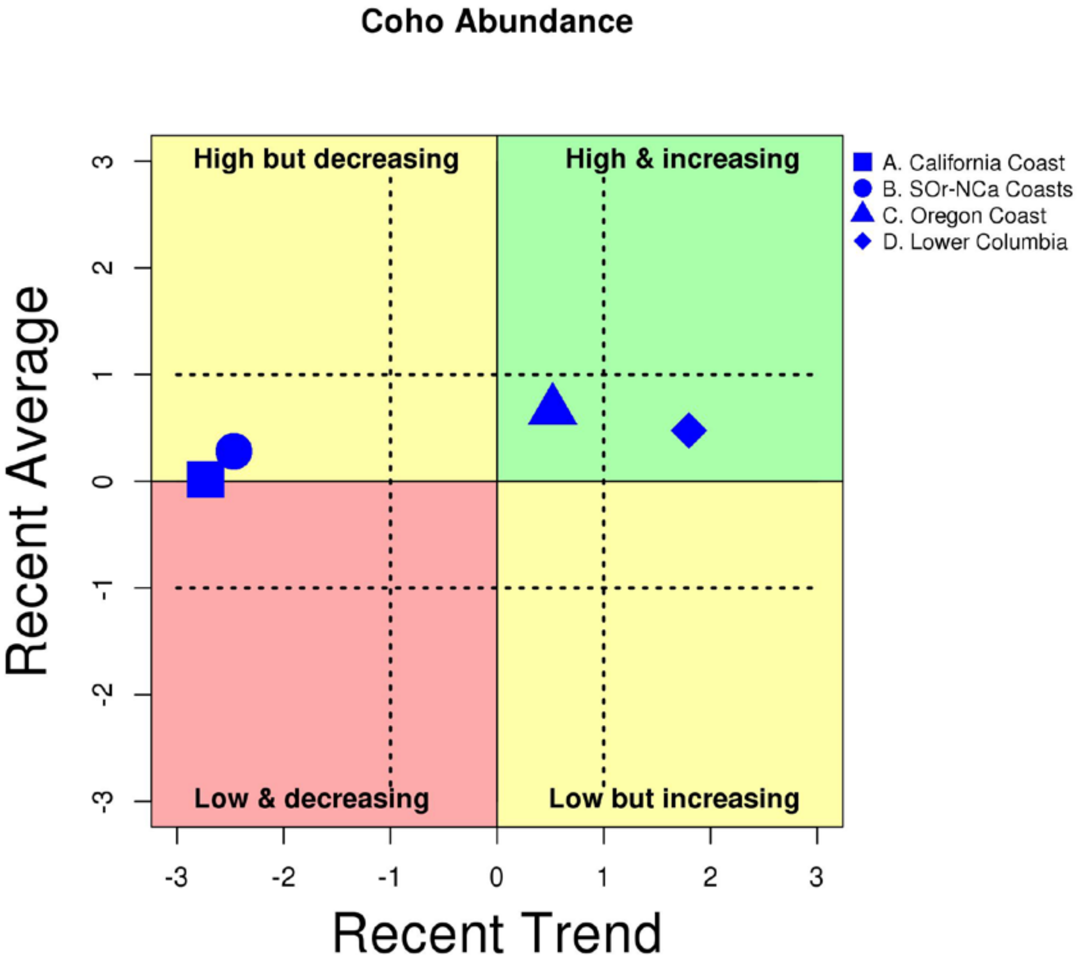


Figure S5. *Coho salmon abundance.* Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis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

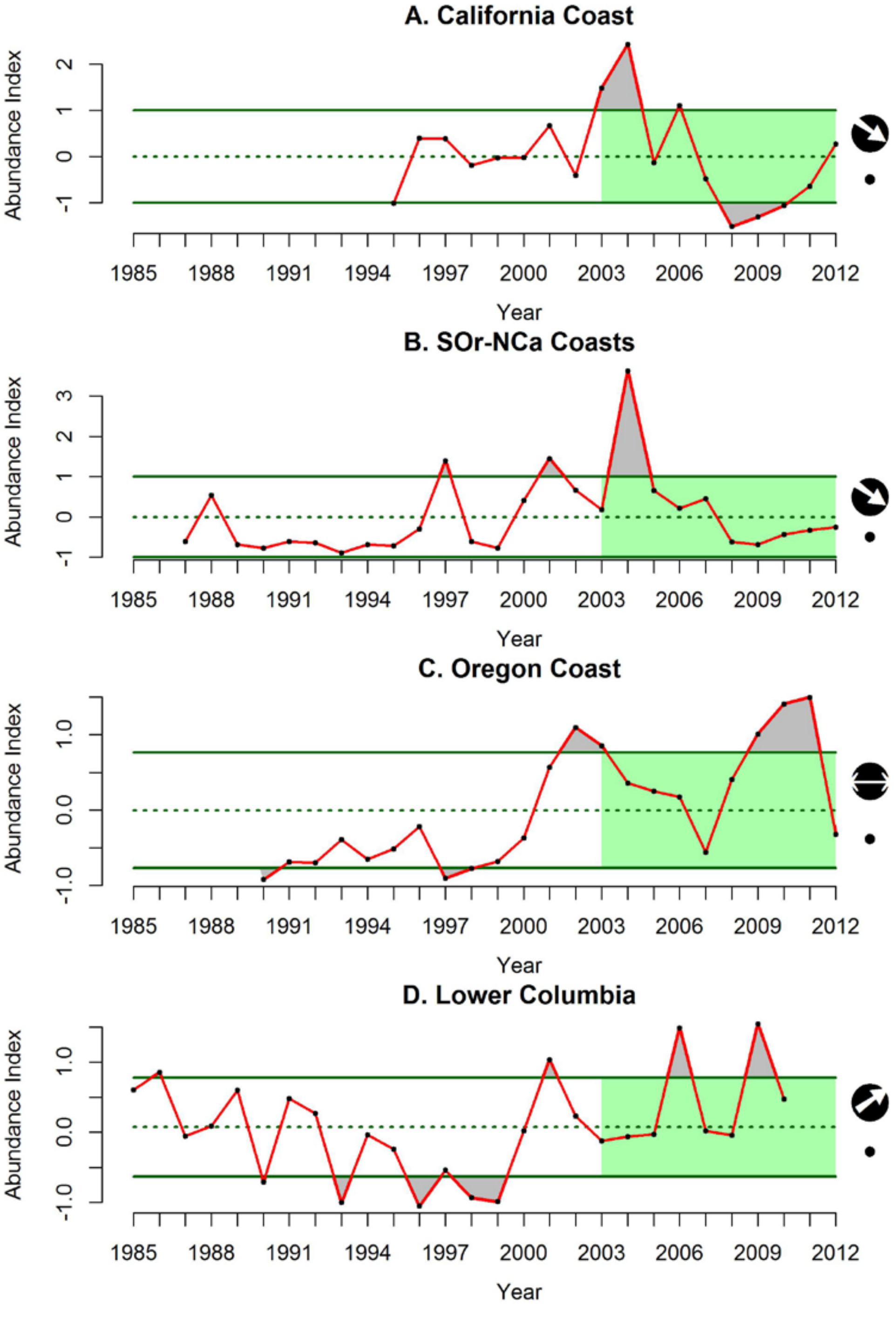


Figure S6. *Coho salmon abundance*. The abundance index is calculated as anomalies (observed-mean/standard deviation). Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend was significant over the last 10-years . The lower symbol indicates whether the mean during the last 10 years was greater or less than or within one s.d. of the long-term mean. Abundances are shown as anomalies.

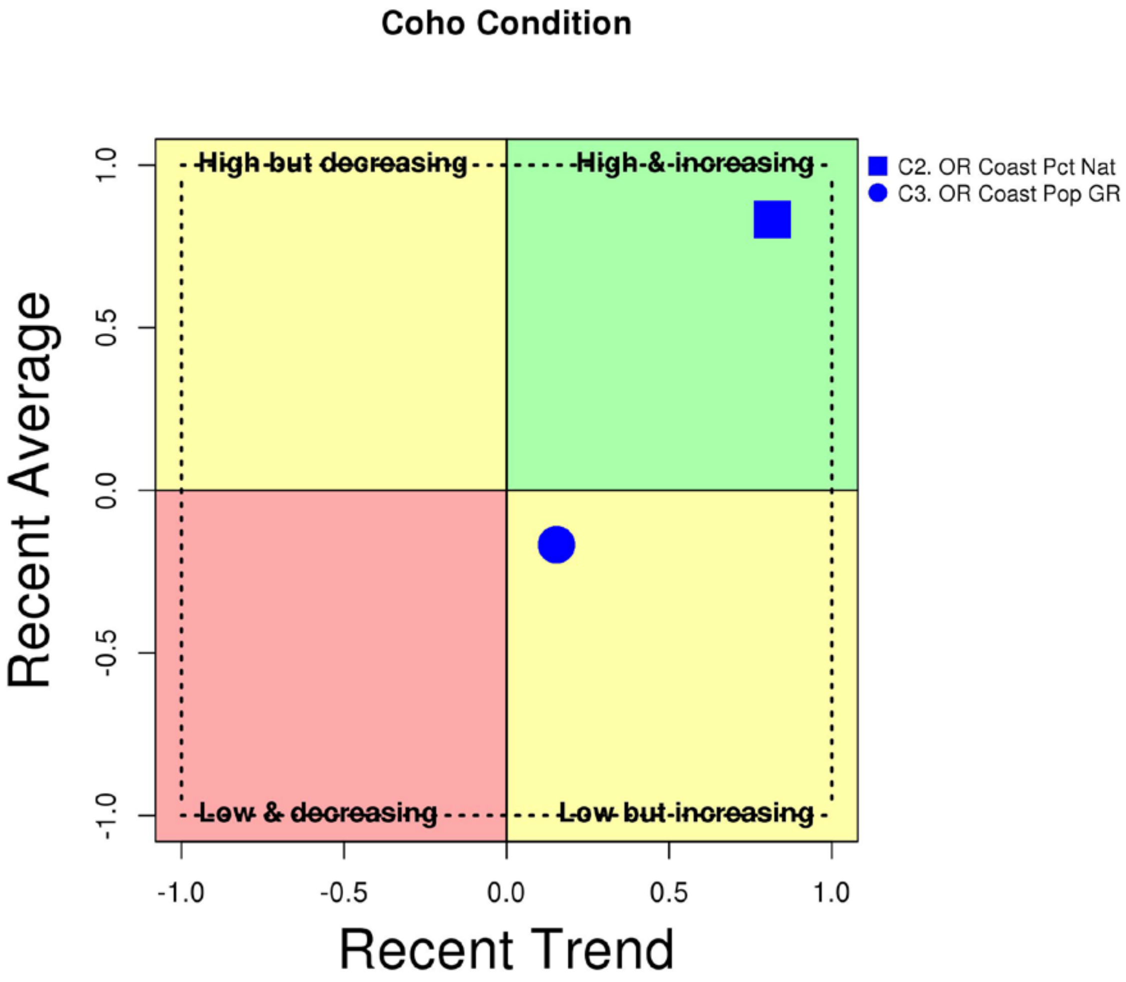


Figure S7. *Coho salmon condition.* Quadplot summarizes information from multiple time series figures. Prior to plotting time series were normalized to place them on the same scale. The short-term trend (x-axis) indicates whether the indicator increased or decreased over the last 10-years. The y-axis indicates whether the mean of the last 10 years is greater or less than the mean of the full time series. Dotted lines show ± 1.0 s.d. When possible we evaluated percent natural spawners (Pct Nat), age-structure diversity (Age Div), and population growth rate (Pop GR).

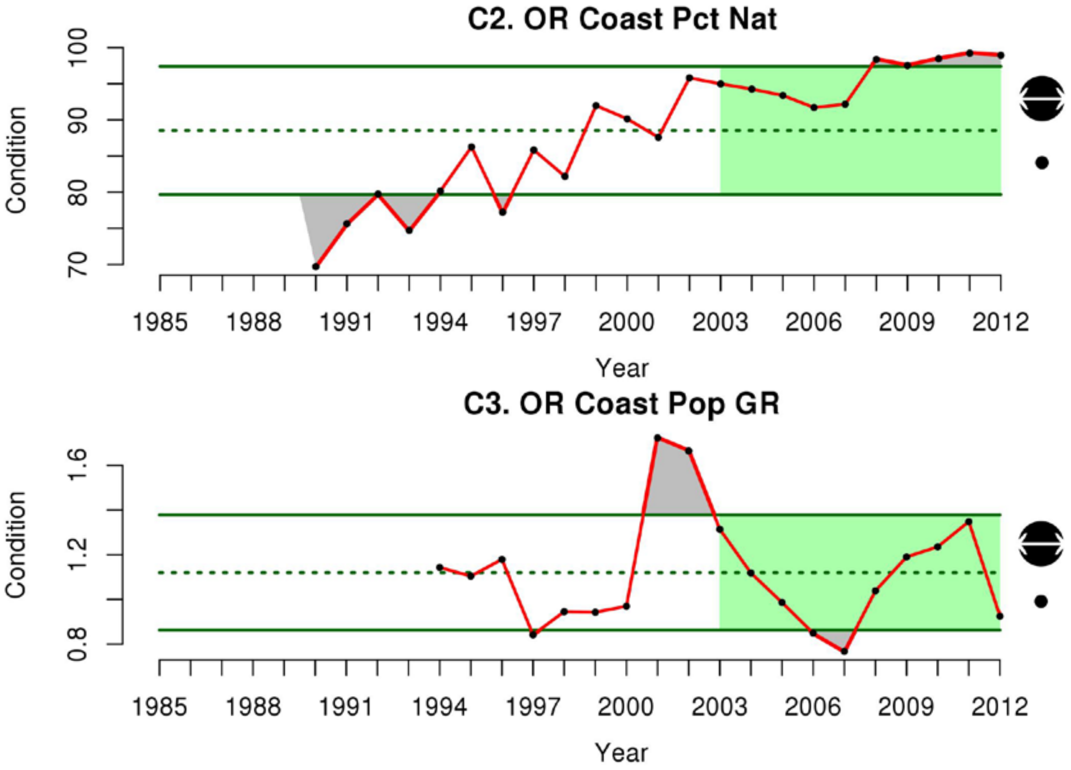


Figure S8. *Coho salmon condition*. Dark green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series. The shaded green area is the last 10-years, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the trend was significant over the last 10-years . The lower symbol indicates whether the mean during the last 10 years was greater or less than or within one s.d. of the long-term mean. When possible we evaluated percent natural spawners (Pct Nat) age-structure diversity (Age Div), and population growth rate (Pop GR).

EVNIRONMENTAL PRESSURES RELEVANT TO SALMON

Here, we briefly review recent ocean and freshwater conditions, and longer-term risks related to global climate change. Where appropriate, we reference figures from the 'Oceanographic and Climatic Drivers and Pressures' (Hazen et al. 2014) and 'Coastal Pelagic and Forage Fishes' (Wells et al. 2014) chapters of this web report , indicated by the prefixes 'OC' or 'C', respectively. In summary, recent ocean conditions indicate a likelihood of improved early marine survival of Chinook salmon and coho salmon in 2012 and 2013, suggesting improved adult returns in the next two years. Freshwater flows and temperatures in the Pacific Northwest suggest improved smolts per spawner from 2008 to 2012, but poor conditions in 2013. However, conditions have been poor from southern Oregon through California for much of the last decade. Longer-term climate change trends are likely to increase risks for most West Coast salmon stocks.

RECENT OCEAN CONDITIONS

Based on historical relationships between ocean conditions and observed Chinook salmon and coho salmon survival rates, basin-scale, regional, and local seascapes were likely conducive to improved early survival of Chinook salmon and coho salmon in 2012 and 2013.

BASIN-SCALE PROCESSES

The basin-scale forcing acting on salmon while in marine waters, in part, determines the later adult abundance (Mantua and Hare 2002, Wells et al. 2006, Wells et al. 2007, Wells et al. 2008, Black et al. 2011, Schroeder et al. 2013). In 2012 and spring 2013 the basin conditions were likely conducive to improved early salmon survival and growth. The multivariate El Niño Southern Oscillation (ENSO) index (MEI) (Wolter and Timlin 1998) transitioned from El Niño to La Niña conditions in summer of 2010 through January 2012 (see Figure OC27 in Hazen et al. 2014, Ocean and Climate Drivers section in this report). In the summer of 2012, the MEI increased but the values were too low and short-lived to be classified as an El Niño event; the values returned to neutral conditions in the spring of 2013. The Pacific Decadal Oscillation index (PDO) (Mantua and Hare 2002) became negative (cool in the CCS) coinciding with the start of the La Niña in the summer of 2010 (Figure OC7). The PDO continued in a negative phase through the summer of 2012, with a minimum in August. After October 2012, the PDO increased to slightly negative values in the winter and spring of 2013. The North Pacific Gyre Oscillation index (NPGO) (Di Lorenzo et al. 2008) was positive from the summer of 2007 to the spring 2013 with a peak value in July 2012 (Figure OC28).

LOCAL AND REGIONAL PROCESSES

Local and regional-scale coastal processes (including coastal winds, upwelling, and temperature) are the proximate influences on salmon food webs (including ecosystem structure) in the ocean (Wells et al. 2007, Black et al. 2011, Wells et al. 2012).

Spring and summer coastal upwelling drives the seasonal supply of nutrients to the CCE, and thus is an important influence on food supply for juvenile salmon. Coastal upwelling conditions were also conducive to improved salmon production in 2012 and 2013. In March 2012 upwelling winds north of 39°N were anomalously low while winds south of 39°N remained near the climatological mean. Upwelling north of 39°N did not resume again until May and for summer and fall remained at close to climatological values. In contrast, south of 39°N average upwelling prevailed from winter 2011 to April 2012, after which it intensified. Strong upwelling continued off central California until fall 2012. North of 36°N, high upwelling persisted through winter 2012 and into January-February 2013 (Figure OC19).

Phenology (seasonal timing) of winds and upwelling, particularly the timing of the spring transition, is also important in determining the productivity of the CCE (Chavez and Messie 2009, Checkley and Barth 2009) and salmon survival (Koslow et al. 2002, Logerwell et al. 2003). The cumulative upwelling index (CUI) gives an indication of how upwelling influences ecosystem structure and productivity over the course of the year (Bograd et al. 2009). At 45°N, the upwelling season began later than average from 2007 to 2012, with 2012 being the latest spring transition since 2007 (Figure OC21). The upwelling season began early in southern and central California (33°-39°N) during 2012 (Figure OC21). Strong upwelling continued into the summer off southern California (33°N) with CUI estimates at the end of July being the highest since 1999. At 36°N, the 2012 CUI values at the end of the year were the second highest on record, falling just below the high in 1999. Through mid-2013, CUI values are greater than previously observed records throughout the CCS. While there were significant regional differences in upwelling in 2012, strong upwelling occurred more widely in the CCS in winter and spring of 2013.

SALMON FORAGE IN THE OCEAN

An examination of zooplankton communities in the northern region of the CCIEA shows that secondary production was conducive to improved salmon production from 2010-2012. Examination of the copepod community can help to determine source waters and provide insights into the productivity of the system (Peterson and Keister 2003). Copepods that arrive from the north are cold–water species that originate from the coastal Gulf of Alaska and include three cold–water species: *Calanus marshallae*, *Pseudocalanus mimus*, and *Acartia longiremis*. Copepods that reside in offshore and southern waters (warm-water species) include *Paracalanus parvus*, *Ctenocalanus vanus*, *Calanus pacificus*, and *Clausocalanus* spp. among others. Copepods are transported to the Oregon coast, either from the north/northwest (northern species) or from the west/south (southern species). The Northern Copepod Index (Peterson and Keister 2003) was positive from autumn 2010 through summer 2012 (see data at http://www.nwfsc.noaa.gov/research/divisions/fe/estuarine/oeip), indicating an abundance of boreal zooplankton species. In central California, springtime krill (*Euphausia pacifica* and *Thysanoessa spinifera*) abundance was increased during 2008-2013 compared to 1990-2007, indicating good prey for salmon as they initiate their marine migration. At the time of emigration to sea has been identified as a critical period for determining later adult abundance (Wells et al. 2012, Woodson et al. 2013).

The ichthyoplankton and juvenile fish communities along the Newport Hydrographic Line off the coast of Oregon in May 2012 were similar to the average assemblages found in the same area and month during the previous five years both in terms of mean concentrations and relative concentrations of the dominant taxa (Wells et al. 2013), indicating that forage conditions were not poor. However, larval myctophids were found in the highest concentration in July 2012 of the five-year time series, while larval northern anchovy were found in higher concentrations (>3x) in July 2012 than in the same month in 2007-2010. In addition, concentrations of the dominant taxa of juvenile fish were higher in July 2012 than in the same month in the previous five years, largely due to the abnormally high concentration of juvenile rockfish found in July 2012 (>10x that of any other year in 2007-2011). No juvenile age-0 Pacific hake or northern anchovy were collected from the midwater trawl samples in May or July 2012, although age- 1 and adult specimens of both species were found. Similarly, the biomass of ichthyoplankton in 2013 from winter collections along the Newport Hydrographic Line were above average (1998-2013), which should have favored average-to-good feeding conditions for juvenile salmon during the 2013 out migration. Consistent with these results, in the region between Tatoosh Island, WA and Cape Perpetua, OR the forage community was typical and not indicative of a poor forage environment for salmon (Figure C7).

In central California, the forage assemblage in both 2012 and 2013 showed higher productivity for the species and assemblages that tend to do better with regionally cool, high southward transport conditions, including juvenile rockfish, market squid and krill (predominantly *Euphausia pacifica* and *Thysanoessa spinifera*) (Figure C5). On the shelf these species provide the bulk of prey resources to salmon (Wells et al. 2012, Thayer et al. 2014). In 2012, juvenile rockfish catches were above average, as they have been in most years since 2008, and in 2013 the highest catches of juvenile rockfish in the time series of the survey were recorded, with huge numbers of juvenile rockfish of all species (as well as young-of-year groundfish of other species, such as Pacific hake, flatfishes and lingcod, *Ophiodon elongates*) encountered throughout both the core and expanded survey areas. Market squid and krill were at very high levels in 2012 and 2013 as well. Although more northern anchovy were encountered in 2013 than in the previous five years, catches of both that species and of Pacific sardine remained well below long-term averages. As with the 2012 results, 2013 continued to indicate a pelagic micronekton community structure dominated by cool-water, high transport, high productivity forage species (like juvenile groundfish), krill and market squid (see Ralston et al. 2013).

RECENT FRESHWATER CONDITIONS

Although this IEA is focused on the marine environment, salmon forge a strong connection between marine and freshwater ecosystems, and both marine and freshwater phases of the life history are important determinants of population status and trends (Bradford 1995); for this reason, we include a review of recent freshwater conditions. The key factors discussed here are river flows and water temperatures; we leave discussions of habitat structure (e.g, woody debris, pools, gravel, side channels) to later reports. In the Pacific Northwest, indices related to freshwater conditions have been similar or slightly more favorable for many salmon stocks in the past 10 years compared with the average since 1976 and especially frorm 2009 to 2012. In California, however, recent freshwater conditions have been below average due to, among other factors, drought and resource conflicts. A number of the human activities that relate directly or indirectly to flow and temperature are discussed in the section “Human dimensions relevant to salmon abundance and condition.” In the framework of CCIEA, these human activites should be considered in the context of the environmental variability and considered in any management scenario evaluation.

Interior Columbia basin salmon generally migrate upstream as adults from spring to fall, depending on the population. Those that migrate in the summer or early fall (summer and fall Chinook salmon and sockeye salmon) can confront stressful temperatures particularly in the mainstem Columbia and Snake rivers (Crozier et al. 2008a, Crozier et al. 2011). High temperatures expose the fish to direct thermal stress, but also increase morbidity and mortality from some diseases and raise energetic costs due to the exponential rise in metabolic rate with warmer temperature (McCullough 1999). Spring Chinook salmon migrate earlier in the year and thus rarely experience high temperatures during the migration, but like summer Chinook salmon they risk prespawn mortality due to stressfully warm temperatures in the tributaries while they wait to spawn in late summer or fall. Thus high summer temperatures can increase adult mortality in these populations. Juvenile growth and survival can also be affected by summer temperature, but negative effects of warmer years are seen most in the warmer streams of spring/summer Chinook populations (Crozier and Zabel 2006, Crozier et al. 2008b). Thus summer temperatures affect all the life stages that inhabit freshwater in this season, and although most populations remain below stressful temperatures in most years, unusually high temperatures are generally negative for these cold-water fish.

Stream flow also affects multiple life stages in a complex manner. The level of flow can affect available habitat area, the distribution and availability of prey, refuges from predators, water temperature, and other factors (e.g., Arthaud et al. 2010, Poff and Zimmerman 2010). Accordingly, Snake River spring/summer Chinook salmon populations are vulnerable to low flows in the fall (Crozier and Zabel 2006, Crozier et al. 2008b). In lower elevation spawning areas (e.g., fall Chinook salmon and coho salmon), winter precipitation generally falls as rain. Heavy rains can scour nests, or bury them in sediment, thus reducing egg viability, so high flows present risks over winter. In spring, high flows are generally favorable for migrating smolts because they facilitate transit to the ocean. Increased flow increases migration speed, which decreases exposure to factors such as predation and temperature stress in reservoirs (e.g., Ferguson et al. 1998), and it affects ocean entry timing and early ocean survival (Scheuerell et al. 2009).

To describe recent freshwater conditions affecting these life stages, we summarize trends first in a composite index, the Pacific Northwest Index, then at selected locations in the Columbia River Basin where temperature and flow have been measured over multiple decades. The Pacific Northwest Index is based on 1) air temperature at Olga in the San Juan Islands, averaged annually from daily data; 2) total precipitation at Cedar Lake in the Cascade Mountains; and 3) snowpack depth at Paradise on Mount Rainier on March 15 of each year (Ebbesmeyer and Strickland 1995). Lower values of this index correspond to cooler and wetter conditions in freshwater, especially west of the Cascade Mountains. Since 1976, the five-year running mean of this index has been positive since the 1976 regime shift, and like the PDO, shifted negative in recent years (since 2006), indicating better freshwater conditions for Pacific Northwest salmon in recent years.

Long-term water temperature records are relatively scarce, and are most accessible from the mainstem dams in the Columbia and Snake Rivers. Mean August temperature has been similar or slightly cooler in the last 10 years compared with the long-term mean (since 1976) at both Bonneville Dam on the Columbia River and Ice Harbor Dam on the Snake River, but by much less than one standard deviation (0.18˚C cooler, much less than 0.5-0.7˚C s.d. in the long-term time series). However, recent summer air temperatures have been increasing, so tributary temperatures might be warmer than reflected in the mainstem.

Fall flows (average of September and October) in the Salmon River, the largest free flowing tributary to the Snake River, have averaged lower in the last 10 years than since 1976 (1976-2012: 1129 cubic feet per second (cfs) fall spring, 2003-2012: 1090 cfs), but have been climbing since 2001. Similarly, spring freshet flows have been rising recently (since 2001), but unlike fall flows, spring flows are slightly higher than their long term mean (1976-2012: 5335 cfs spring, 2003-2012: 5415 cfs spring). The good smolt migration conditions (relatively higher flows and lower temperatures) have likely contributed to improved ocean survival from 2009-2012.

However, July 2013 witnessed high mortality of adult Columbia Basin salmon on their spawning migration, attributed to an early rise in temperature (Crozier et al. 2014). Adult Snake River sockeye salmon died at high rates throughout the spawning migration compared with recent years (only 13% reached the spawning grounds in the Upper Salmon River Basin from Bonneville Dam).

Much of California experienced drier than average conditions in 2008-2010, and 2012- April 2014, and most of the state is currently under extreme or exceptional drought conditions (in May 2014; see http://droughtmonitor.unl.edu). In summer 2013, there were multiple conflicts over in-stream and out-of-stream flows that impacted California Chinook salmon stocks. For example, water temperature standards in June 2013 were relaxed for Sacramento River salmon by moving the boundary for water temperature targets upstream because of the limited supply of cold water in reservoir storage. There was conflict between the Westlands water district and fish advocates in the Klamath basin over diverting stored water in Trinity reservoir into the Trinity River and Klamath River to reduce the risk of a significant fish kill, thereby reducing exports in the Central Valley Project. As well, there was adjudication of water rights by the state of Oregon that awarded the Klamath Tribes senior water rights in the upper Klamath Basin, and the tribe's subsequent decision to exercise those rights to keep water in the river at the expense of junior rights holding irrigators. Generally speaking, the drought of 2013-2014 will likely have widespread negative impacts on the spawning and juvenile freshwater rearing success for California's natural (and possibly hatchery) spawning salmonids.

CLIMATE CHANGE

A number of studies have examined the potential effects of climate change on Pacific salmon populations (see reviews Battin et al. 2007, Independent Scientific Advisory Board 2007, Crozier et al. 2008b, Schindler et al. 2008), and concern for these effects has led to the inclusion of climate change as a risk factor in recent Endangered Species Act status reviews for salmon (e.g., Ford 2011). The overall effect of climate change on any anadromous stock must consider all habitats and life stages simultaneously and cumulatively (Crozier et al. 2008b, Wainwright and Weitkamp 2013). For example, because Pacific salmon are cold water fishes, increases in temperature—whether in freshwater, estuarine, or marine environments—are likely to be detrimental near the warmest edges of their range.

In a recent review of climate effects across the life cycle of Oregon Coast coho salmon, Wainwright and Weitkamp (2013) found that, despite substantial uncertainties in future climate scenarios and biological response, the preponderance of negative effects across the life cycle indicates a significant risk to long-term sustainability of those populations; while the details would differ by region and species, we expect that these conclusions would apply to most West Coast salmonid populations. A warming climate points to changes in freshwater, estuarine, and marine habitats that are likely to put salmon populations at greater risk. In freshwater, significant reductions in cold-water flows in summer may affect juvenile and adult migration, spawning, egg incubation, and rearing. In estuaries, rising sea levels will lead to inundation of low-lying lands and increases in salinity that will cause substantial transformations in estuary habitats. In the ocean, rising water temperatures, acidification, and changes in coastal water circulation will have both direct and indirect (via foodweb processes) effects on salmon. In all of these habitats, some effects (such as the physiological response to temperature) are predictable while others (such as interspecies competitions and changes in community structure) are uncertain both in the direction (positive or negative) and magnitude of the effect on salmon. Despite these uncertainties, the overall effect of climate change will very likely be to increase risks facing salmon populations, particularly those near the southern/warmer limits of their species' range.

Freshwater

Future climate change scenarios mostly point to degraded freshwater habitat for West Coast salmonids. These scenarios include a general pattern of increased winter precipitation in the wettest locations (northern California to BC) and reduced winter precipitation in the driest locations (southern California) (see Dalton et al. 2013, Garfin et al. 2013). However, all regions are expected to warm substantially (2 to 6 °C by 2100), and the atmospheric warming will lead directly to warmer stream temperatures (Arismendi et al. 2012, Isaak et al. 2012) and will support trends to a higher fraction of precipitation falling as rain rather than snow (Elsner et al. 2010, Mantua et al. 2010, Cloern et al. 2011). For affected watersheds, this combination causes reduced springtime snowpack and reduced snow-fed stream flow in late spring and early summer, and increased rain-fed runoff and stream flow in winter. The combination of reduced spring freshet and higher river temperatures will likely reduce smolt survival for life history types that include freshwater rearing in summer. Adult migrants might benefit energetically from weaker flows, but thermal costs to migration and longer pre-spawn periods will likely outweigh this benefit. Peak flows in winter are likely to increase, as a larger fraction of affected basins will generate runoff than in a colder climate. Some adult coho populations use fall rains to trigger their spawning migration. The impact on spawning migration timing is still uncertain. Increasing fall flows, predicted for some locations under some scenarios, might benefit juveniles (see analysis in Crozier and Zabel 2014 chapter of this web report). In summer and fall, base flows are expected to decline in many watersheds as a consequence of increased water deficits driven by warming temperatures that increase the atmosphere's demand for water. The shift to a more amplified hydrograph, both seasonally and episodically, will likely increase redd-scour in fall and winter, and reduce freshwater rearing habitat in summer and fall, which combined would reduce spawner to smolt productivity rates.

Stream temperatures are expected to warm for most locations year round, although the local sensitivity to surface warming will vary widely depending on specific watershed characteristics (deep and shallow groundwater interactions with surface flows, proximity to the coast and frequency of summer fog, channel characteristics like depth and width, vegetative cover, and water infrastructure and management). Growth rates in the coldest streams and coldest seasons will likely improve (Beer and Anderson 2013, Crozier and Zabel 2014), but growth rates, thermal stress (Crozier and Zabel 2014), and temperature and flow mediated fish kills and migration barriers in already warm streams will likely increase in frequency and distribution (McCullough 1999, Mantua et al. 2010, Cloern et al. 2011). Anoxic conditions may result from strong deep groundwater upwelling combined with weak river flows (Roegner et al. 2011). These conditions threaten juvenile outmigrants.

Estuaries

Within West Coast estuaries, changes in temperature, flow, and sea level are the primary physical factors responsive to a changing climate. Global sea level is projected to rise by 0.3 to 1.2 m by 2100, relative to 1986-2005 (Intergovernmental Panel on Climate Change 2013). Rising sea levels will favor increased seawater intrusion into estuaries, increased inundation of intertidal habitats and low-lying areas, and transformations in habitat types along the interfaces between terrestrial, freshwater and marine habitats. For key estuaries like California's Bay-Delta region and the Columbia River estuary, substantial changes in temperature, salinity, and water levels are expected by 2100 (Cloern et al. 2011).

Temperature changes can potentially affect all salmon life stages that inhabit or migrate through estuaries. First, shallow water habitats (such as tide flats and marshes) will likely warm more than other areas, which may result in seasonal shifts in habitat use, and potential reductions in total habitat available to salmon. Juveniles will leave habitats when they get too hot, potentially exposing themselves to higher predation rates in deeper water. Second, before temperatures exceed thermal limits, we can expect other thermal responses to occur. In particular, all species and life stages will respond bioenergetically. For example, yearling size fish will encounter elevated temperatures during their migration through the estuary that can affect their metabolic rates while sub yearling Chinook salmon can encounter elevated but not stressful temperatures in wetlands. At more modest temperature increases, growth rates of salmon may actually increase, assuming food resources do not diminish. It is unclear how migration and rearing timing will adapt to changes in the estuarine temperature regime. Finally, temperature-mediated shifts in species distributions will also lead to changes in community composition (Roessig et al. 2004), with unpredictable effects on salmon.

The second major type of physical change that will impact the Columbia River estuary is sea level rise. As the level of the sea rises, a number of changes can be expected. Tidal wetlands may become submerged or have longer periods of inundation than they do currently, and nearby terrestrial habitats will be flooded (Kirwan et al. 2010). In other cases, shallow water habitats such as wetlands may erode as sea levels rise. Diking and other barriers may prevent wetlands from expanding and keeping up with erosion impacts. The net effect of these processes on estuarine habitats depends on the rate of sea level rise, the rate of vegetation growth and sedimentation, and the land contours in and adjacent to the estuary (Roessig et al. 2004, Kirwan et al. 2010), but in general the global rate of sea level rise is currently faster than the colonization rate for new wetlands (Roessig et al. 2004). Also, rising sea levels will push the saline portion of the estuary upstream into freshwater areas and change the location of freshwater-saltwater ecotones (Flitcroft et al. 2013). Such changes will affect how these estuarine habitats function for salmon. For example, the head of tide in tributaries will move upstream with accompanying changes in physical structure of the estuary (due to changes in the tidal prism) and biological characteristics of these river mouth systems.

A third type of change that can affect salmon in the estuary (and the coastal plume) is flow-related changes as a result of changes in precipitation patterns, snow melt, and water management practices. As previously discussed, climate scenarios for the Pacific coast suggest that there will be a reduction in precipitation that occurs as snowfall and an increase in rainfall, which would increase winter flow levels and diminish summer flows. Coupled with increased temperatures, such a scenario could critically limit salmon migration periods. In the estuary, there is a relationship between flow, tides, and salinity at any point. Changes in salinity (e.g., either the upstream extent of measurable salinity or the regime at any particular place) will depend on freshwater flow, tides, and basic sea level rise (polar and glacial melting). Changes in salinity or water levels in the estuary will alter the biological community structure as well as accessibility of these locations to salmon. Flow changes are also important in terms of downstream fish migration rates through the estuary. Reduced flows during the time salmon are outmigating, for example, could slow the downstream migration of salmon and affect their timing of ocean entry. Changes in flow (e.g., resulting from changes in precipitation patterns) could also affect the nearshore ocean via changes in river plume characteristics (Burla et al. 2010).

The Ocean

Salmon will be affected by climate-driven changes in the ocean's physical, chemical, and biological components and processes (Doney et al. 2012, Howard et al. 2013). The major physical changes in the ocean, especially the California Current System, that are of concern for West Coast salmonids are higher ocean temperatures, intensified or weakened upwelling, delayed spring transition, intensified stratification, and increased ocean acidity (Crozier et al. 2008; Wainwright and Weitkamp 2013).

The direct and indirect effects of ocean temperature changes on salmonids have received the most attention. Water temperature has a strong effect on fish physiology, development, distribution, and behavior (e.g., Marine and Cech 2004, Richter and Kolmes 2005). As ocean temperatures warm a number of things will occur that will ultimately reduce survival rates of some anadromous populations. Effects of ocean warming are suggested for a number of different salmon and steelhead stocks including Snake River steelhead and Chinook salmon (Petrosky and Schaller 2010), Fraser River sockeye salmon (Hinch et al. 1995), Central Valley Chinook salmon (MacFarlane 2010), and steelhead in general (Welch et al. 2000, Atcheson et al. 2012) In particular, metabolic rates will likely increase such that growth will be impacted (Hinch et al. 1995, Atcheson et al. 2012). This will also affect adult size at age and age at maturity, which in turn may have consequences for fecundity, migration ability, and ability to dig spawning redds.

One of the mechanisms by which warming water temperatures will affect salmon and steelhead is by changing bioenergetics of the fish. Diet information on yearling Chinook salmon from 19 years of ocean research indicates that during warmer ocean years, juvenile salmon consume 20-29% more food than in the colder ocean years and are in significantly lower body condition (E. Daly personal communication). Research also suggests that the biomass of fish prey is reduced during warmer ocean conditions at the same time that the salmon are consuming more food (Daly et al. 2013). This food stress, along with direct effects of temperature on physiology, can shift competitive responses and increase predation risk for salmonids (Reeves et al. 1987, Marine and Cech 2004). Increasing ocean temperatures will also change food web relationships involving salmon, especially as ranges shift for predators, competitors, and prey (e.g., Murawski 1993, Hays et al. 2005, Cheung et al. 2009).

Overall, the combination of these temperature effects will result in changes in the range of the species. Temperature is a main factor determining the northern and southern limits of fishes in the California Current (Horn and Allen 1978). Climate-driven range shifts in marine fishes have been observed (Hsieh et al. 2009) and predicted for the future (Cheung et al. 2009). Warming of the upper ocean in the CCS will likely lead to poleward and shoreward shifts in the distribution of sub-tropical species (e.g., Hazen et al. 2013), which could increase competition with and predation on maturing salmon. Range shifts for Pacific salmon have been observed in past periods of climate change (Ishida et al. 2009) and are occurring now (Irvine and Fukuwaka 2011). For the future, Abdul-Aziz et al. (2011) illustrate this point for Pacific

Northwest salmon by showing how climate scenarios can result in a dramatic contraction (30-50% by the 2080s) of the summer thermal range suitable for chum salmon, pink salmon, coho salmon, sockeye salmon, and steelhead in the marine environment. They predict an especially large contraction (86-88%) of Chinook salmon summer range under two commonly-used IPCC (2007) greenhouse gas scenarios. Previous analyses focusing on sockeye salmon (Welch et al. 1998) came to similar conclusions. A consequence of northward shifts of suitable salmon marine habitats is that populations near the southern limit of their species' range will be more susceptible to climate change than those near the species' center of distribution. For example, populations of sockeye salmon in the Columbia River Basin and coho salmon and Chinook salmon in central California may be more vulnerable than populations further north. Maintaining these salmon populations under future climate conditions may require greater improvements in freshwater habitat and the river migration corridor than will be needed for more northerly populations.

Beyond water temperature, other climatological changes in the California Current are also likely to affect West Coast salmonids. The timing and intensity of upwelling has an important but complex relationship to salmon production. Changes in the intensity of upwelling winds could have profound impacts on upper ocean properties (temperature, salinity, nutrients, and primary productivity) in the CCS (Checkley and Barth 2009). There have been observed increases in upwelling intensity (Bakun 1990) and shifts in timing of spring transition and the total length of the upwelling season (Bograd et al. 2009), but analyses using climate models find little agreement on future changes in upwelling, largely because current models do not have sufficiently fine scale to resolve coastal wind and circulation processes (Diffenbaugh 2005). Upwelling of nutrient-rich water is also limited by the degree of water-column thermal stratification (Kosro et al. 2006), which is expected to increase as surface waters warm (Di Lorenzo et al. 2005). Upper ocean warming, absent substantial changes in upper ocean salinity, would increase stratification in ways that tend to reduce the upwelling of cold, nutrient rich water from greater depth, which absent other changes would lead to reduce concentrations of macronutrients and primary productivity, and a shift in the phytoplankton community structure away from large diatoms to smaller phytoplankton species (e.g., dinoflagellates). However, increased stratification in the open waters of the North Pacific might also reduce ventilation rates in upwelling source waters in ways that substantially increase nutrient concentrations of upwelled waters (Rykaczewski and Dunne 2010).

A final major issue for coastal waters is acidification as a consequence of increasing atmospheric CO_2; increasing acidity is already being observed in the California Current System (Hauri et al. 2009). Ocean acidification in the CCS will be affected by changes in global ocean acidity as well as processes that include coastal upwelling and changes in upwelling source-water chemistry (Rykaczewski and Dunne 2010). Acidification will likely have little direct effect on salmon and steelehad, with the exception of some possible biochemical stress (Fabry et al. 2008). However, it may have a dramatic impact on invertebrates that are important in salmon food webs (Fabry et al. 2008); the consequences for salmonids depend on potentially complex shifts in prey availability and abilities of salmon to shift diets.

Note, that biological effects of climate change, whether in freshwater, estuarine or ocean environments, are extremely difficult to predict. The rapid expansion of Humboldt squid—a voracious predator-- along the West Coast of North America in recent years and their population explosion in 2009 (Field et al. 2013), remind us that although physical processes are more straightforward to predict, the response of biological systems to physical changes are much more difficult to predict.

INTEGRATED ENVIRONMENTAL EFFECTS

So far, we have discussed a number of individual climate factors that affect salmon and steelhead in certain habitats or specific parts of their life cycle. In order to fully assess the consequences of climate change, we need to consider the interactions of all the individual effects as they multiply across life stages within generations and across generations within populations (Wainwright and Weitkamp 2013). While many of the effects described above are difficult to project with much certainty, most are more likely than not to have negative effects on salmonid growth and survival. Thus, the overall consequences of climate change for West Coast salmonids are likely to be negative, and will require management strategies that increase resiliency of these ESUs over the foreseeable future.

HUMAN DIMENSIONS RELEVANT TO SALMON ABUNDANCE AND CONDITION

Native Americans have lived in the Pacific region for thousands of years, and salmon have played a central role in their diet, culture and economy. By approximately three thousand years ago, Native Americans of the Northwest coast began to organize their lives around the fluctuating, seasonal runs of salmon, a primary source of protein (White 1980, Schalk 1986). Salmon runs helped determine the location of villages, as well as the timing of visits with relatives living in other watersheds (Suttles 1987). Early Northwesterners could access, enhance, and exploit such a diversity and concentration of resources – including forest game and berries, abundant salmon runs and many other species of fish, shellfish and marine mammals – that they were among the only people in the world who developed relatively settled, dense, stratified, and wealthy societies without relying on intensive agriculture (White 1980, Lichatowich 1999). Instead, they developed sophisticated salmon harvesting, drying, and storage technologies (Stewart 1977), and they enhanced the region's natural resources by actively managing the land with periodic burns that encouraged berries, bracken, and grazers, and by planting and tending camas meadows, estuarine gardens, and shellfish beds (White 1980, Boyd 1999). Most notably, Native Americans developed complex and cooperative resource ownership and access systems that enabled them to sustain and cope with the dynamism of their local resources (Suttles 1987, Singleton 1998). For example, the problem of how to allocate salmon was addressed through territorial fishing grounds, potlatches, and intermarriage among families in different river basins, which helped ensure access to such a variable and unpredictable resource (Suttles and Sturtevant 1990).

Since Euro-American settlement, Native Americans suffered enormous losses of land, people, culture, language, and spirit through disease, colonization, forced assimilation, criminalization, and discrimination (Breslow 2011). In particular, and as detailed below, industrial resource exploitation and non-Native settlement decimated the salmon runs that have been keystones in the social integrity of coastal tribes. Nevertheless, contemporary Native Americans in the

Northwest region continue to rely directly and extensively on their local environment for sources of food, ceremonial materials, spiritual power, and cultural identity (Onat and Hollenbeck 1981, Sepez 2001, Donatuto 2008). Salmon and salmon fishing remain central to their way of life and essential for overcoming historical trauma, and revitalizing cultural traditions, including indigenous food systems and resource management practices (Swinomish Tribal Mental Health Project 1991, Donatuto 2008, Brave Heart et al. 2011, Northwest Indian College 2012). The role of salmon in food practices and as iconic species and markers of regional identity is also experienced by other sociocultural groups (in addition to Native Americans) who live in the Northwest (Nabhan et al. 2010).

Sporadic encounters with foreign explorers, missionaries and entrepreneurs became more common by the 1770s (National Research Council 1995). The 1848 California Gold Rush was the start of large-scale Euro-American settlement in the Pacific region that continued at a rapid pace even after the peak of the mining boom. Factors that encouraged such exploitation included the abundance of natural resources, development of steamship and railroad lines (which facilitated westward migration and eastward market expansion), advances in extractive and processing technologies, and government policies encouraged rapid development (e.g., land grants to railroads).

The 1848 Gold Rush occurred at a time when Mexico was about to surrender California to the U.S. and the only form of U.S. authority was federal troopers. Western settlers devised their own ways of staking claims to resources. Water was an essential resource – a critical input into activities such as mining, logging, farming, hydropower production, and fishing. When California became a state in 1850, the Legislature adopted the common-law riparian doctrine already in use in the eastern states. This doctrine granted riparian landowners the right to use water flowing through their land, so long as they did not impair the rights of other riparians and regardless of whether they actually used the water. Beginning with the Gold Rush, prospectors and non-riparian farmers began claiming water by simply appropriating it (i.e., by building diversions). By the end of the nineteenth century, prior appropriation became the basis of water rights throughout the western states: claims to water must be based on a reasonable and beneficial use, rights can apply to riparian or non-riparian lands, priority of rights is determined on the basis of “first in time, first in right,” and water rights can be lost due to non-use (Gillilan and Brown 1997, Hundley 2001). As noted by Gillilan and Brown (1997), “The allocation of water to those who took it first provided incentives for settlers to take and put to use all the water that they could possibly use as quickly as possible, rather than leaving it for instream uses or for potential out-of-stream uses by future settlers.”

By 1900, practically the only rivers in the West that were undeveloped were those that were too remote or too large to be developed without public investment (Gillilan and Brown 1997). Such investment was initiated in the 1930s, with the construction of large federal dams serving multiple purposes (irrigation, flood control, navigation, municipal/industrial water supply, recreation). From the Gold Rush to the 1960s, little effort was made to mitigate effects of resource extraction on wild salmon populations. By the 1970s, growing environmental awareness led to a spate of federal legislation – e.g., 1970 National Environmental Policy Act, 1972 Clean Water Act, 1973 Endangered Species Act, 1980 Comprehensive Environmental Response, Compensation and Liability Act (CERCLA or Superfund Act) – that gave legal standing to the needs of fish and wildlife, including salmon. Prior-appropriation water rights have been limited by the

Courts (through mechanisms such as public trust doctrine, an expansive interpretation of navigable waters in the commerce clause of the U.S. Constitution, and federal reserved rights doctrine) and by legislation (e.g., appropriations of instream flow for environmental use, minimum flow standards) (Butler 1990). According to Hundley (2001), "Reserved rights are often among the most senior in a watershed, because many federal land reservations were made quite early, in the late 1800s or early 1900s. Federal reserved water rights are based on the purposes for which they were reserved rather than actual use and cannot be lost through nonuse. And, perhaps most important from the perspective of federal land management agencies, reserved rights are based on federal, rather than state, law, and are presumably not subject to diminishment by the states." A notable example of the exercise of federal reserved water rights occurred in 2013, when Oregon recognized a U.S. claim to surface water in the Klamath River basin on behalf the Klamath Tribes as the most senior water right in the Basin (Oregon Water Resources Department 2013).

This section describes post-1848 resource-related activities – fur trading, mining, logging, farming, dams, hatcheries, and fishing – and relates these activities to major demographic, economic, social, technological, and policy changes that occurred with Euro-American settlement in the region. This historical perspective considers legacy as well as ongoing effects of those activities on wild salmon and salmon habitat, and shows how more recent environmental protections have moderated the single-minded resource exploitation characteristic of earlier decades. It also provides (with the benefit of hindsight) the opportunity to illustrate, with concrete examples, the dynamic relationship between human dimensions and salmon, and may suggest ways in which such relationships can be modeled in an ecosystem context. The quantitative indicators provided here regarding trends in human activities are a first step in that direction; though much more work needs to be done. Importantly, river flows and water temperatures that are discussed in "Recent freshwater conditions" and escapement, as discussed in "Summary and status of trends of salmon abundance and condition," are reliant on many of the anthropogenic activities that we present. *Therefore, in the context of the ecosystem-based management, these human activities should be considered as part of any management scenario and the current and future environmental condition should be used to help determine appropriate trade-offs and choices.*

Figure S9 depicts current land use patterns in the Pacific region, including forests, farmland and population centers. These patterns reflect the legacy effects of early settlement and development, as well as changes in resource abundance, economic conditions, technology, and law and public policy that have occurred since the Gold Rush.

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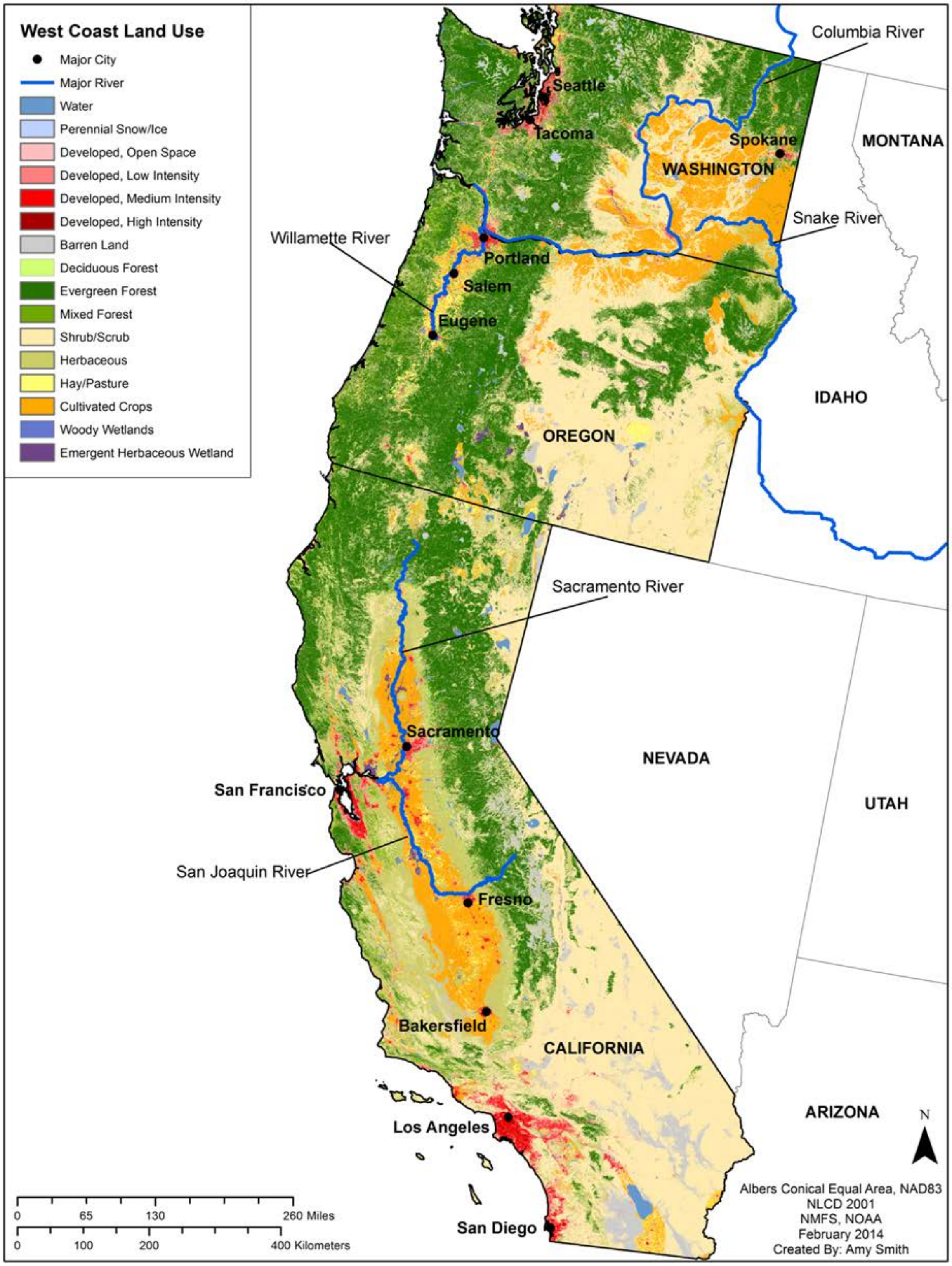

Figure S9. Land use in California, Oregon and Washington

STATUS, PATTERNS, AND TRENDS OF ACTIVITIES RELEVANT TO SALMON

FUR TRAPPING

Fur trapping and trading were lucrative enterprises from the 1780s to the 1830s, but subsequently dwindled due to declines in beaver populations and the market for pelts. The decimation of beaver populations had adverse effects on salmon, as beaver ponds provided nutrients and important rearing habitat for salmon and also stabilized habitat by dampening the effects of currents and flows (Lichatowich 1999, Taylor 1999).

MINING

In 1848, gold was discovered on the American River in California. The mining boom – which also included other metals (e.g., silver), minerals (e.g., sand, gravel) and coal – spread, albeit on a lesser scale, to other parts of California as well as Oregon and Washington Territories (Wissmar et al. 1994, Schwantes 1996). Methods of placer mining depended on water availability and included ground sluicing, dredging, diversion of streams to diggings that lacked water, and hydraulic mining that washed entire hillsides into streams (Pomeroy 1965). By 1870 almost 7000 miles of ditches had been constructed in California to move water from rivers to mines (Gillilan and Brown 1997). In 1884 the Ninth Circuit Court in San Francisco shut down the hydraulic mining industry in California, citing its destructive effects on property and the navigability of the Sacramento and Feather Rivers (Hundley 2001). Mining impeded salmon spawning, migration and survival by changing the course of rivers, stream flows, temperatures and suspended sediment, altering bottom substrate composition, damaging riparian habitats, and creating high sediment loads that clogged salmon gills and smothered salmon eggs and aquatic insects (Nelson et al. 1991, Wissmar et al. 1994, Lichatowich 1999).

The extralegal property rights system established by the miners, allowing free and open access to minerals on public lands, was codified in the 1872 Mining Act. The 1920 Mineral Leasing Act (administered by the Bureau of Land Management) provided a system for private companies to lease and develop mining interests on federal lands. Today, BLM shares authority for minerals management with four other Department of the Interior (DOI) agencies: Minerals Management Service, Office of Surface Mining and Reclamation, Bureau of Mines and U.S. Geological Survey. Major environmental laws governing mining include the Clean Water Act, the Superfund Act, and the Endangered Species Act (Klyza 1997). Acid pollution from some abandoned mines persists to the current time. For example, Iron Mountain Mine in the Klamath Mountains of northern California was designated a Superfund site in 1983. At that time, acid drainage from the mine included a ton of copper and zinc each day – about one-quarter of the total discharge of copper and zinc into surface waters by all municipal and industrial point sources in the U.S. Since 1963, spills from Spring Creek Reservoir (where the acid drainage is contained) during large storms have caused at least 20 major fish kills on the Sacramento River (U.S. Department of Justice 2000, U.S. Environmental Protection Agency 2013).

LOGGING

The Hudson Bay Company established the Pacific region's first sawmill at Fort Vancouver, Washington in 1827 (Schwantes 1996). Beginning with the 1848 Gold Rush, timber production increased rapidly to meet the demand for mining infrastructure, housing and fuel for the rapidly growing population (McKelvey and Johnston 1992). Numerous small sawmills were built in California's Sierra Nevadas to serve nearby mining towns. In 1848, there were about 22 sawmills in western Oregon and Washington. In Oregon alone, the number of mills increased from 100 in 1851 (Lichatowich 1999) to 173 in 1870 to nearly 500 by 1900 (Taylor 1999). Mills dumped tons of sawdust into streams and bays – consuming oxygen and smothering plants as well as salmon eggs and alevin (Taylor 1999).

Mills were often located near water to facilitate the transport of logs. Logs were floated down mountains in water-filled flumes or "skidded" by oxen to a river transport site, then washed downriver when flows were sufficiently high. To overcome the problem of seasonal flows, logs and water were held in splash dams and periodically released into the river for the trip downstream, with dynamite used to break up subsequent logjams (Taylor 1999, Lichatowich 1999). These practices resulted in rapid changes in flow, destruction of stream bottoms and banks, loss of spawning gravel and refugia, and direct destruction of salmon (particularly Chinook, which spawn and rear in the main stem where log drives occurred). The loss of old-growth riparian forests elevated stream temperatures, increased stream bank erosion, and reduced the supply of nutrients needed for biological productivity (Gregory and Bisson 1997, Lichatowich 1999, Taylor 1999).

The invention of steam donkeys (steam-powered engines used to yard logs) and narrow-gauge logging railroads in the 1880s allowed logging to expand into previously inaccessible mountain areas once riparian forests were cut (Lichatowich 1999). High-lead logging, developed in 1905-10 and used to elevate logs above ground-level obstructions, facilitated the cutting of forests for the next forty years. By 1909 the so-called "Kraft process" was being used to convert wood pulp into paper, and the market for plywood developed in the 1920s (Schwantes 1996). Washington was the largest lumber producer in the U.S. during 1905-1938, except for one year when it was surpassed by Oregon (Schwantes 1996). After World War II, logging railroads were replaced by logging roads traveled by heavy-duty equipment such as trucks and bulldozers. Railroad and road construction contributed to landslides that blocked streams and caused siltation that smothered salmon eggs. Old skid trails and abandoned and poorly maintained logging roads continue to create landslides and sediment problems, particularly in areas of steep terrain, heavy rainfall and sedimentary soils. Habitat problems associated with logging should be considered in the context in which they occur (National Research Council 1995).

Today, the vast majority of public forest lands are owned by the federal government and managed by the U.S. Department of Agriculture (USDA) Forest Service (with the remainder under DOI's Bureau of Land Management and Park

Service). Passage of the 1976 National Forest Management Act and the 1976 Federal Land Policy and Management Act changed the federal government's historical pro-timber orientation by requiring the Forest Service and the BLM to consider wildlife protection as part of forest management, implement forest practice regulations, and develop a new planning process that included public participation as required by the 1970 National Environmental Policy Act (Gregory 1997, Hoberg 1997). In 1990 the Forest Service developed a conservation strategy for the Northern Spotted Owl, which was listed as "threatened" under the ESA that same year. In 1994 the agency implemented its Northwest Forest Plan, which expanded the scope of its owl strategy to include "ecologically significant late-successional ecosystems, species, and processes, including but not confined to Northern Spotted Owls" (Thomas et al. 2006). Protection of old-growth forests necessitated large reductions in Northwest timber harvests. Timber harvest reductions also occurred in the Pacific Southwest Region to protect the California spotted owl. Since the early twentieth century, the U.S. Congress has allocated 25% of federal timber sale revenues to counties to compensate them for the loss of tax base associated with federal land ownership. Thus reductions in federal timber harvest have direct economic effects not only on the logging industry but on counties with acreage in federal forests (Hoberg 1997). Since the early 1970s, non-federal (state and private) forest lands have been managed on the basis of forest practices rules developed and enforced by the states. A number of state and federal programs provide technical and financial assistance for voluntary efforts by private forest landowners, conservation districts and non-profit watershed groups to improve forest management. According to Gregory and Bisson (1997), "In general, society has called for high standards of environmental protection on public and private forest lands, but management activities in public forests are restricted to a greater extent than in private forests."

Timber production is considerably higher in Oregon and Washington than California. The early 1990s (when protection of old growth forests became an important concern) marked the beginning of a decline in timber harvest in all three states (Figure S10). The percentage of timber harvest derived from public lands fell from an average of 41% in 1978-1989 to 11% in 1995-2013 in California, from 55% in 1962-1989 to 20% in 1995-2010 in Oregon, and from 33% in 1965-1989 to 21% in 1995-2012 in Washington. Timber production continues to be concentrated in the densely forested areas of northern California and coastal Oregon and Washington (Figure S11).

Interestingly, it was the ESA listing of the Spotted Owl, rather than the listing of salmonid populations themselves, that has so far enabled the most reliable and widespread protection of salmon habitat in the Pacific Northwest (Lombard 2006). The ESA listing of the spotted owl eventually led to the federal Northwest Forest Plan, state Habitat Conservation Plans, and, in Washington State, the multi-stakeholder Timber, Fish and Wildlife Agreement and Forest and Fish Law. These provisions require loggers to leave 100-300 foot-wide buffers of stream side forest in order to protect salmon and other wildlife habitat. As Lombard notes, "No representatives of any other major land use in the state [of Washington] have accepted anything remotely similar" (Lombard 2006). Furthermore, Lombard argues, the spotted owl listing led to state forestry regulations that are "arguably the strictest in the country" and that so far it is the owl, not the salmon, that has had "the greatest effect of any listed species on ecosystem protection in the Pacific Northwest," including the protection of salmon habitat (Lombard 2006).

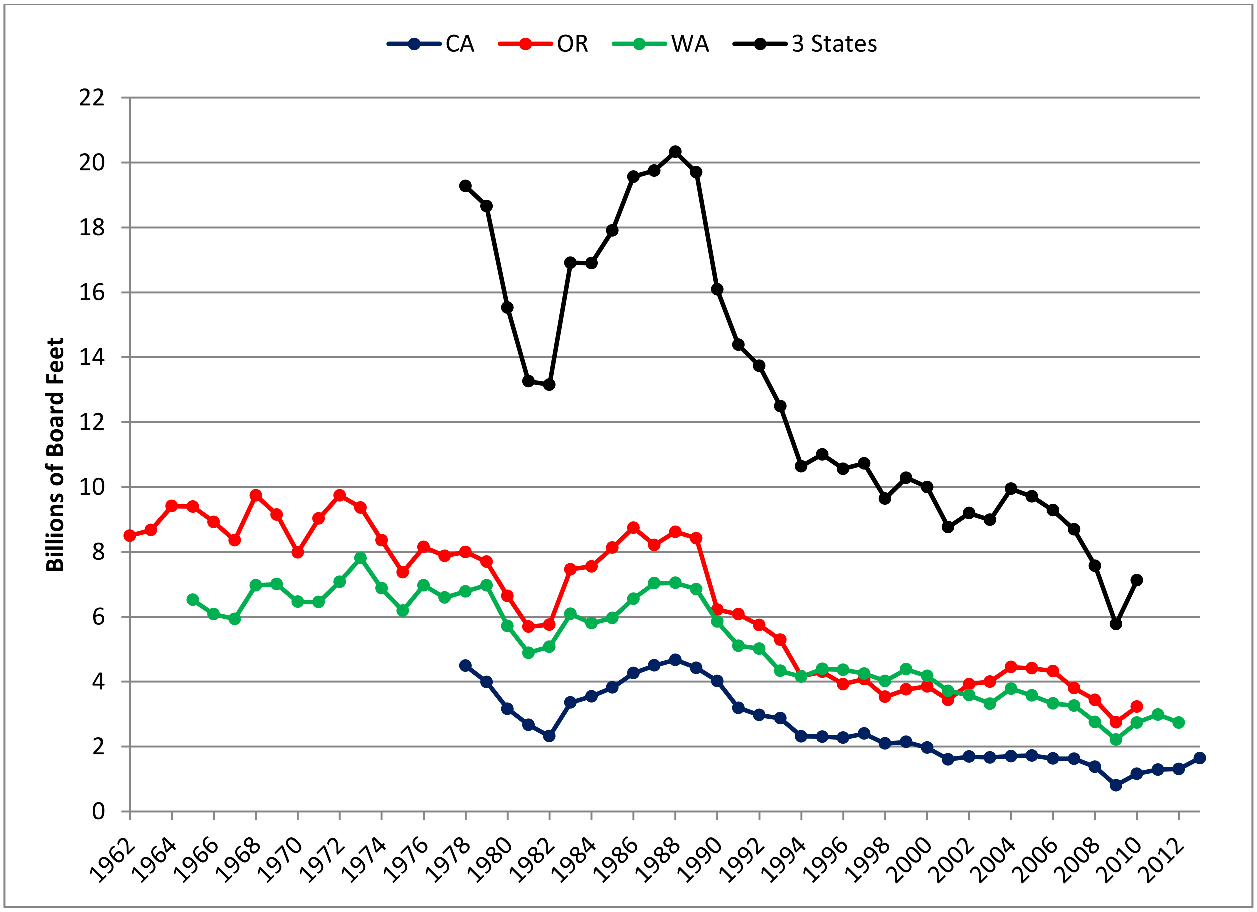

Figure S10. Timber production (billions of board feet) by state and three-state total, 1962-2013 as available (sources: California Board of Equalization, Oregon Department of Forestry, Washington State Department of Natural Resources).

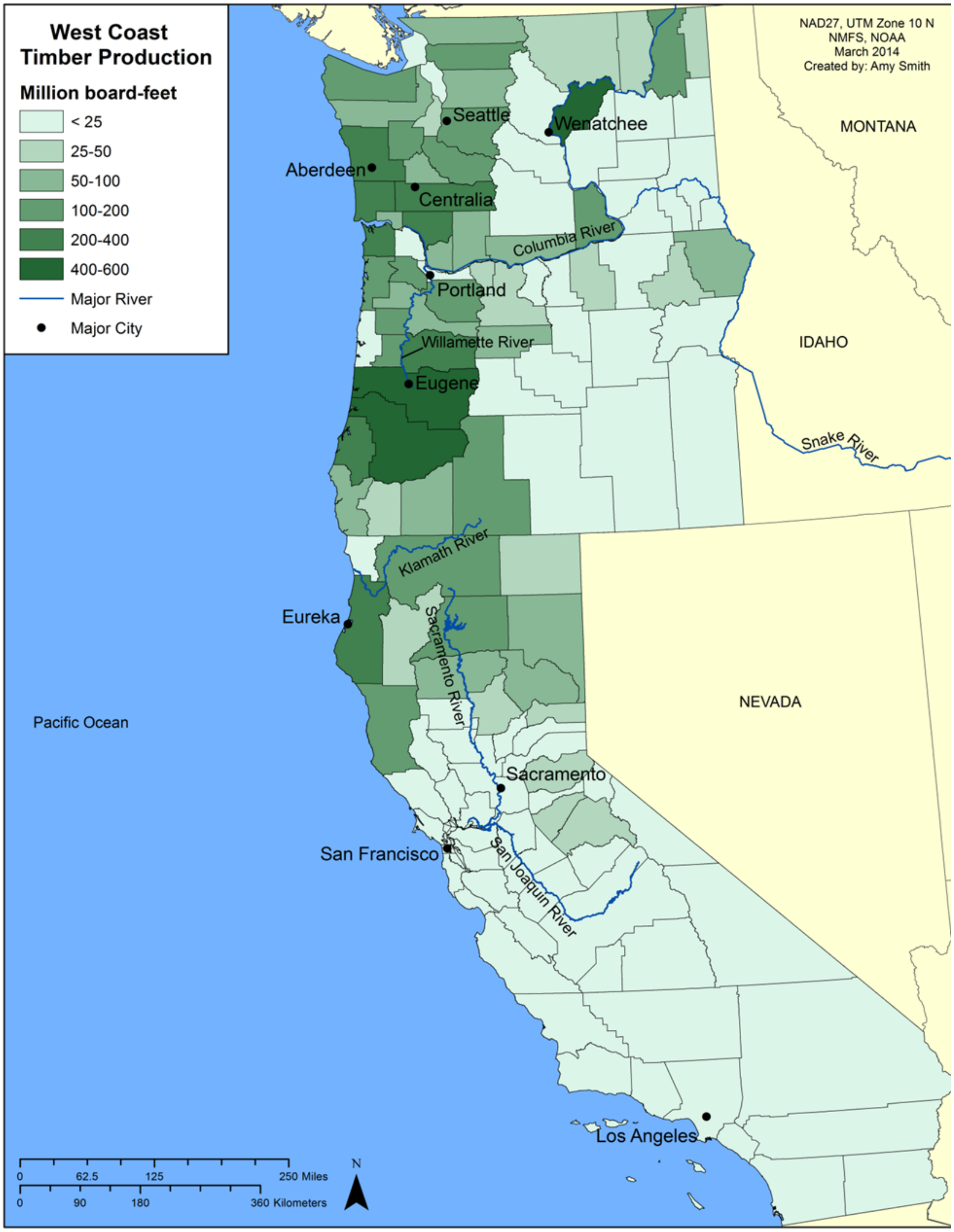

Figure S11. Counties categorized by volume of timber production

AGRICULTURE

Livestock grazing began in California in 1769 when Franciscan missionaries brought cattle and horses from Mexico to their first mission at San Diego. After winning independence from Spain in 1821, Mexico began secularizing the mission landholdings and, by the mid-1830s, had issued land grants for over 400 ranchos, largely for cattle raising (Burcham 1956). In the Pacific Northwest, Native American tribes such as the Nez Percés began grazing horses in the 1700s (Pomeroy 1965, Galbraith and Anderson 1991). White settlers began large-scale cattle raising in the 1860s from stock raised at Hudson Bay Company posts and mission stations, or from animals driven over the Oregon Trail or from California and Texas (Schwantes 1996). While grazing was well-established before the 1848 Gold Rush, demand for meat in mining towns greatly expanded the scale of livestock production. Even after the mining boom, demand for meat remained strong as railroads and the introduction of stock cars and refrigerated cars in the late 1800s opened up new markets as far away as Chicago. By the 1880s, deterioration of the range and mass starvation of cattle during severe California droughts and harsh Northwest winters led cattlemen to abandon open-range grazing in favor of summer grazing on public and private lands and containment and hay provision in the winter. As cattle herds were reduced to numbers that could be contained during the winter, less-costly sheep herds increased and by 1900 exceeded cattle in most western states. The range of cattle and sheep shrank as grazing areas were converted to cropland, settlers fenced off land and streams, and railroads received large land grants (Galbraith and Anderson 1991, Schwantes 1996). Nevertheless, livestock and livestock products remain important components of today's farm economy. Cattle are attracted to riparian areas, where they trample and eat streamside vegetation, erode and destabilize stream banks, reduce water retention by compacting the soil, increase siltation, elevate water temperatures, disturb salmon nesting areas, and deposit waste into streams (Platts 1991, Gillilan and Brown 1997).

Agriculture expanded to include not only livestock and livestock products but also grains and other commodities. The first wheat and apple harvests in the Northwest occurred in the 1820s at Fort Vancouver, Washington – the first permanent white settlement in the region (Schwantes 1996). In the 1830s emigrants from the Midwest began traveling the Oregon Trail to farm in the fertile Willamette Valley. In the mid-1800s, farming expanded east of the Cascades to the warmer, dryer Columbia Plateau; by the 1880s, wheat fields extended throughout eastern Washington and Oregon. Completion of the Northern Pacific Railway in 1883 resulted in another population boom, and refrigerator cars expanded markets for agricultural products. The first irrigation project began in 1859 in the Walla Walla River Valley, followed by similar projects in Oregon. Irrigation expanded cultivation of apples and other commercial crops, and large-scale commercial orchards became common in the valleys of Oregon and Washington by 1905-1915 (Schwantes 1996, Pomeroy 1965).

In California, large expanses of flat fertile land, combined with rainy winters and hot dry summers, were ideally suited to wheat. The wheat boom peaked in the 1880s and 1890s, facilitated by innovations such as the combined grain harvester. During 1890-1914 the farm economy – aided by irrigation – rapidly shifted from large-scale livestock and grain operations to smaller-scale, intensive crops (cotton, fruits, nuts, vegetables). Previously marginal or under-utilized lands were brought into cultivation. Dairy and poultry operations expanded rapidly, and after 1940, non-dairy livestock resurged in the form of large-scale, commercial feed-lot operations. Labor shortages, the large-scale nature of farm operations, high yields, flat landscape, and the absence of rain during harvest season were conducive to highly mechanized farms – e.g., steam and later gasoline tractors, giant combines, high capacity seeders, mechanical harvesters (cotton, sugar beets, tomatoes). Production was also enhanced by a sizeable agricultural research establishment and innovations that reduced perishability and enhanced the quality of fruits and vegetables (Olmstead and Rhode 2003).

The 1935 Rural Electrification Act provided federal loans for installation of electrical distribution systems in rural areas. Rural electrification proceeded rapidly in the Pacific region, spurred by to the large amounts of power needed to pump water for irrigation and for activities such as dairy and poultry farming. By 1939, 75% of farms in California, 50% in Oregon and 57% in Washington were electrified, compared to 22% in the U.S. as a whole (Beall 1940). Irrigation made possible the conversion of vast areas into farmland, but also had adverse effects on salmon habitat. Farmers diverted water from streams using gravity systems with head gates or low dams that often spanned entire streams. Many small dams remain undocumented and unscreened. Unscreened diversions impede salmon passage, reduce river flows (particularly during summer months), and cause high mortality of juvenile and adult fish. Return flows elevate stream temperatures and also transport silt, nutrients, and chemicals that are adverse to salmon (Taylor 1999).

Pumps were used to access groundwater as well as divert surface water. Invention of the deep-well turbine pump in 1930 allowed water to be pumped at greater depths (Faunt et al. 2009a). During 1901-1950, California accounted for about 70% of the nation’s agricultural pumps; acreage irrigated by groundwater increased more than thirty-fold, while acreage irrigated by surface water only tripled (Olmstead and Rhode 2003). Today, groundwater accounts on average for more than one-third of the water used by California’s cities and farms; much more is pumped in dry years (California Natural Resources Agency et al. 2014). The current drought in California (which began in 2012) is the most recent of many such multi-year droughts since 1900: 1918-1920, 1923-1926, 1928-1935, 1947-1950, 1959-1962, 1976-1977, 1987-1992, 2000-2002, and 2007-2009 (California Department of Water Resources 2012). Oregon and Washington manage their groundwater, while California authorizes local agencies to do so. Many groundwater basins are not managed sustainably, although the current drought is highlighting the need for a stronger state role in groundwater management (Faunt et al. 2009a, California Natural Resources Agency et al. 2014). Nearly all surface water bodies interact with groundwater. Poor groundwater management has resulted in reductions in stream flow, shrinkage of riparian areas, land subsidence, adverse effects on water quality, and more costly and energy intensive pumping due to lowering of groundwater levels (Gillilan and Brown 1997, California Natural Resources Agency et al. 2014).

The 1934 Taylor Grazing Act established the Grazing Service (later BLM) to regulate grazing on public lands (National Research Council 1995). State and local agencies are responsible for permitting, inspection and enforcement of environmental regulations on private farms and ranches; these regulations are generally based on regulatory guidance provided by the U.S. Environmental Protection Agency (EPA). The USDA Natural Resources Conservation Service provides educational outreach and technical and financial assistance to encourage farmers to adopt environmentally sustainable practices (Stubbs 2011). As indicated by Gregory and Bisson (1997), "Land use regulations for agricultural and range lands tend to be less protective of streams than forest policies."

California first led the U.S. in cash farm receipts in 1929 and has maintained that lead since 1949 (Hundley 2001). Today California produces over 350 agricultural commodities, including over one-third of the nation's vegetables and nearly two-thirds of the nation's fruits and nuts (California Department of Food and Agriculture 2013). Cash farm receipts in California have increased significantly since 1980. Washington values have increased modestly since 1970, and Oregon values (as available) are about 50% of Washington's (Figure S12). Based on 2012 data, the highest valued commodities in California are dairy, grapes, almonds, greenhouse/nursery, and cattle/calves. The top counties in terms of value are Fresno, Tulare, Kern, and Merced (San Joaquin Valley) and Monterey (central coast). Oregon's highest valued commodities are greenhouse/nursery, cattle/calves, dairy, wheat, and hay. Top counties are Marion, Washington and Clackamas (northwest Oregon) and Morrow and Umatilla (northeast Oregon). Washington's highest valued commodities are apples, dairy, wheat, cattle/calves, and potatoes. Top counties are Yakima, Benton, Grant, Franklin and Walla Walla (all south central Washington) (U.S. Department of Agriculture 2014).

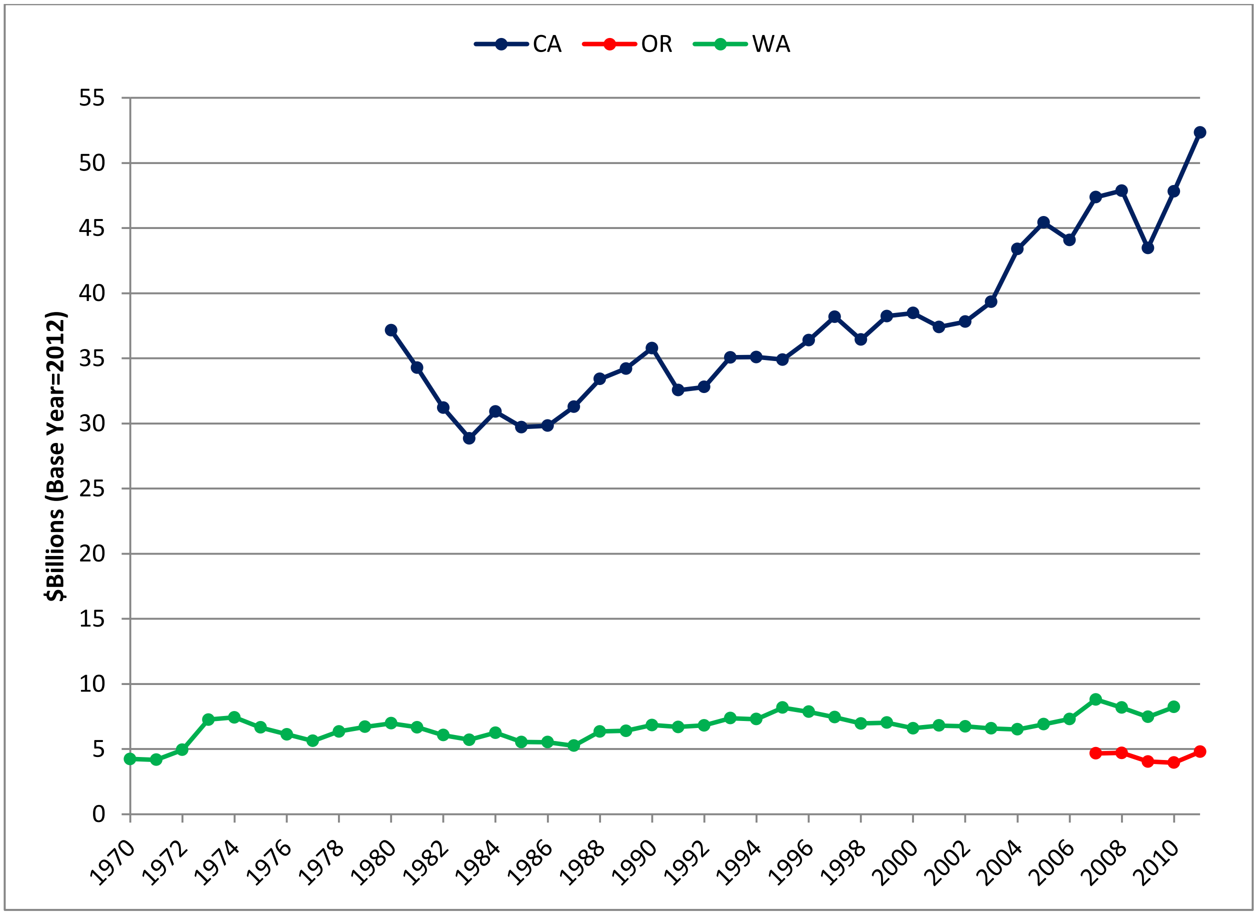

Figure S12. Cash farm receipts in California, Oregon and Washington, 1970-2011 as available (base year=2012).

DAMS

The 1902 Reclamation Act authorized construction of large federal irrigation projects in the arid western states, with construction costs to be recovered without interest from irrigators who benefit from the project. The 1939 Reclamation Project Act transformed the program by authorizing construction of water projects for multiple uses, with costs to be shared among all such uses. These Acts – as well as other laws and budget appropriations – led to construction of some of the largest water storage, withdrawal, conveyance and diversion systems in the world. These water projects stimulated population growth and greatly expanded economic opportunities in the Pacific region, but also fundamentally and adversely affected salmon populations and their habitat. According to the National Research Council (1995), "Of the various human-caused changes in the region, particularly the Columbia River basin, perhaps none has had greater impact than dams."

Dams block or impede access of salmon to historical upstream habitat. Passage facilities (when available) provide some access but also delay upstream migration, increase prespawning mortality and reduce spawning success.

Downstream migrants must pass through reservoirs, survive dams, spillways, bypass facilities and turbines, and overcome hazards such as increased predation. Disruption of the natural flow regime (in terms of timing as well as volume of flows) elevates water temperatures, affects the flow of nutrients and energy, and alters habitat of vegetation, fish and other biota. Flow changes also affect downstream channel morphology and hinder flushing of sediments, recruitment of sediments and spawning gravel, and transport of large woody debris. Dams and associated structures are highly disruptive to a river's ecological processes (Ligon et al. 1995, National Research Council 1995).

Beginning in the 1930s, a number of large federal dams were constructed in the Columbia River Basin. These included Bonneville (1937), Grand Coulee (1942), McNary (1954), Chief Joseph (1955), The Dalles (1960), and John Day (1971) on the Columbia River, and Ice Harbor (1962), Lower Monumental (1969), Little Goose (1970) and Lower Granite (1972) on the Snake River. These dams are operated by the U.S. Army Corps of Engineers except for Grand Coulee, which is operated by the U.S. Bureau of Reclamation. Construction of Bonneville and Grand Coulee helped lift the Northwest economy out of the Depression and brought electricity to rural areas (Lichatowich 1999). By the late 1970s, 14 mainstem Columbia River dams and 13 main stem Snake River dams were operating in the Columbia Basin. Today the Northwest depends on hydropower for 80% of its electricity (Foundation for Water and Energy Education 2014). Salmon passage can be particularly challenging at high dams. Upstream migration can be facilitated by trap-and-haul operations or fishways, but the sheer size of the reservoirs can hinder the ability of outmigrating smolts to find their way to sea (National Research Council 1995).

Grand Coulee (the largest dam by mass in the U.S.) is part of the Columbia Basin Project (CBP) in central Washington – the largest reclamation project in the U.S. The Project includes over 300 miles of main canals, 2000 miles of lateral irrigation canals, and 3500 miles of drains and wasteways. CBP purposes include flood control, navigation, irrigation, and municipal and industrial uses. Fish and wildlife purposes were recognized in 1980, based on a Court interpretation of the 1958 Fish and Wildlife Coordination Act (U.S. Bureau of Reclamation 2013b). Even before 1930, hundreds of smaller dams were built in the Pacific region, including 32 major dams on Columbia River tributaries (Lichatowich 1999). By 1975, about 48% of the spawning and rearing habitat accessible to Chinook in the Columbia River Basin was lost to dam construction. Habitat loss has been greater for spring and summer Chinook than for fall Chinook (Mundy 1997).

About one-sixth of the irrigated land in the U.S. is in California's Central Valley and about one-fifth of the nation's groundwater demand comes from Central Valley aquifers (Faunt et al. 2009b). High water demand and frequent droughts make "water wars" a fact of life in California (Speir et al. In review, Speir and Stradley In review). The major water projects in California are the Central Valley Project and the State Water Project. The nexus of these two projects is the Sacramento/San Joaquin Delta. The Delta is also a central corridor on the migration route of Central Valley salmonids.

Central Valley Project: The federal Central Valley Project (CVP) was authorized in 1935 to increase economic opportunities in the fertile lands of the flood-prone Sacramento Valley and the arid San Joaquin Valley by regulating flows and more equally distributing water between the two valleys (Hundley 2001). The CVP, operated by the U.S. Bureau of Reclamation, extends 400 miles from the Cascade Mountains near Redding to the Tehachapi Mountains near Bakersfield. The CVP manages about nine million acre feet (11.1 billion cubic meters) of water and includes 20 dams and reservoirs, 11 power plants, and 500 miles of major canals that manage water for irrigation, municipal and industrial use, flood control, hydropower and recreation to Central Valley communities. Today about 78% of CVP water goes to irrigation, 9% to municipal and industrial use, and 13% for release into rivers for the benefit of fish and wildlife (U.S. Bureau of Reclamation 2013a).

The CVP includes water impounded on the Sacramento, Trinity, American, Stanislaus, and San Joaquin rivers. Sacramento River water is impounded at Shasta Dam (the second largest U.S. dam in mass, after Grand Coulee) and Keswick Dam. Trinity River water is impounded at Trinity Dam, released downstream to Lewiston Dam, and then diverted through the Trinity Mountains via the Clear Creek Tunnel to the Sacramento River. Water in Sacramento River reservoirs is released (1) to the Tehama-Colusa and Corning canals to serve irrigators on the west side of the Sacramento Valley, and (2) down the Sacramento River to the Sacramento/San Joaquin Delta. Water from the central Delta is transported via the Contra Costa Canal to Contra Costa County, while water from the south Delta is intercepted at the Delta Cross Channel and pumped through the Delta Mendota Canal. The Delta Mendota Canal transports water south to the Mendota Pool near the town of Mendota on the San Joaquin River, with some water diverted to the San Luis Reservoir and other CVP reservoirs along the way; this water is allocated to San Joaquin Valley irrigators. Water from the American River (a Sacramento River tributary) is impounded at Folsom Dam for use by local communities and to augment water supplies in the rest of the CVP. San Joaquin River water is impounded at Friant Dam and diverted to irrigators in the south San Joaquin Valley via the Madera and Friant-Kern Canals. Water from the Stanislaus River (a tributary of the San Joaquin River) is impounded at New Melones Dam for eventual release to the San Joaquin (Hundley 2001, U.S. Bureau of Reclamation 2013a).

State Water Project: The State Water Project (SWP) – authorized by the 1960 California Water Resources Development Bond Act – is operated by the California Department of Water Resources. Today, the SWP includes 33 storage facilities, 21 lakes and reservoirs, and 700 miles of canals, pipelines and tunnels. The SWP stores and re-regulates about 5.8M acre-feet (7.2 billion cubic meters) of water originating from the Feather River (a tributary of the Sacramento River). This water is impounded in large reservoirs, then released into the Sacramento River and transported downstream to the Sacramento/San Joaquin Delta. From the Delta, the water is pumped to (1) the North Bay Aqueduct for delivery to Napa and Solano counties, (2) the South Bay Aqueduct for delivery to Alameda and Santa Clara counties, and (3) the 444-mile California Aqueduct for off stream storage at the San Luis Reservoir. South of the San Luis Reservoir, the Coastal Branch of the California Aqueduct diverts water to San Luis Obispo and Santa Barbara counties. The California Aqueduct then carries water 2000 feet over the Tehachapi Mountains, where water is supplied by the West Branch to Ventura and Los Angeles counties, by the East Branch to Los Angeles and areas south, and by the East Branch Extension to Riverside and San Bernardino counties. Due to its prodigious pumping requirements, the SWP is the largest single user of power in the state. About 70% of SWP water is used for municipal and industrial use in Southern California and the San Francisco Bay area, and the other 30% for irrigation (California Department of Water Resources 2008a, 2010).

Sacramento/San Joaquin Delta: The Sacramento/San Joaquin Delta is where fresh water from the Sacramento and San Joaquin rivers mingles with saltwater from the Pacific Ocean, creating the West Coast's largest estuary. It is the hub of California's water system – the Central Valley Project and the State Water Project. Historical development of the Delta was spurred by the 1850 Swamp Land Act, which provided California and 12 other wetland-abundant states with grants to reclaim wetlands through construction of levees and drains. In California, about two million acres of wetlands were reclaimed – including 500,000 acres of tidal marsh in the flood-prone Delta. Due to widespread reclamation, California has experienced the largest percentage loss of wetlands of any state (91%) (Dahl 1990). In 1861, California enacted the Reclamation and Swampland Act authorizing the creation of swampland districts (later reclamation districts) responsible for financing and providing flood control within their boundaries (Hundley 2001). Today flood control is the shared responsibility of California's Central Valley Flood Protection Board, the California Department of Water Resources, local levee maintaining agencies, and the Army Corps of Engineers (ACOE). The ACOE sets standards for levee safety, and both the Corps and the State provide rehabilitation assistance to qualifying local agencies. Comprehensive flood management is needed to address problems associated with aging levee infrastructure, susceptibility of levees to earthquakes and floods, increasing costs of levee maintenance, and growing population in flood-prone areas (California Central Valley Flood Protection Board 2013).

Today 55 islands and tracts in the Delta are protected by 1100 miles of levees. The levees protect not only farms but also hundreds of thousands of people who live and work in the Delta and $47 billion in infrastructure (e.g., highways, railroads, energy transmission lines, water and petroleum pipelines) serving the San Francisco Bay area. The levees were constructed from fertile peat soils native to the area. Land subsidence due to peat oxidation has gradually lowered the elevations of Delta islands – in some cases more than 20 feet below sea level – and increase the risk of levee failure (California Department of Water Resources 2008b). The natural intrusion of brackish water into the Delta is exacerbated by levee subsidence and reductions in freshwater inflow due to water exports. Some of this intrusion has been offset by modifications to CVP and SWP dam operations that increase freshwater inflow in dry summer months for the benefit of migrating salmon and resident Delta smelt. The powerful south Delta pumps that supply water for the CVP and SWP cause Delta water to flow from north to south instead of east to west, disrupting fish migration and causing salinity buildup in the east Delta, where salts can no longer be flushed to the ocean by natural river flows. The ecology of the Sacramento/San Joaquin Delta has been profoundly affected by the CVP and SWP and the human activities that depend on these Projects (Ingebritsen and Ikehara 1999).

Figure S13 depicts the location of CVP and SWP irrigation districts and populous cities in southern California, San Francisco Bay and the Central Valley. The map reflects the importance of the CVP and SWP to irrigated agriculture and municipal water users throughout the state.

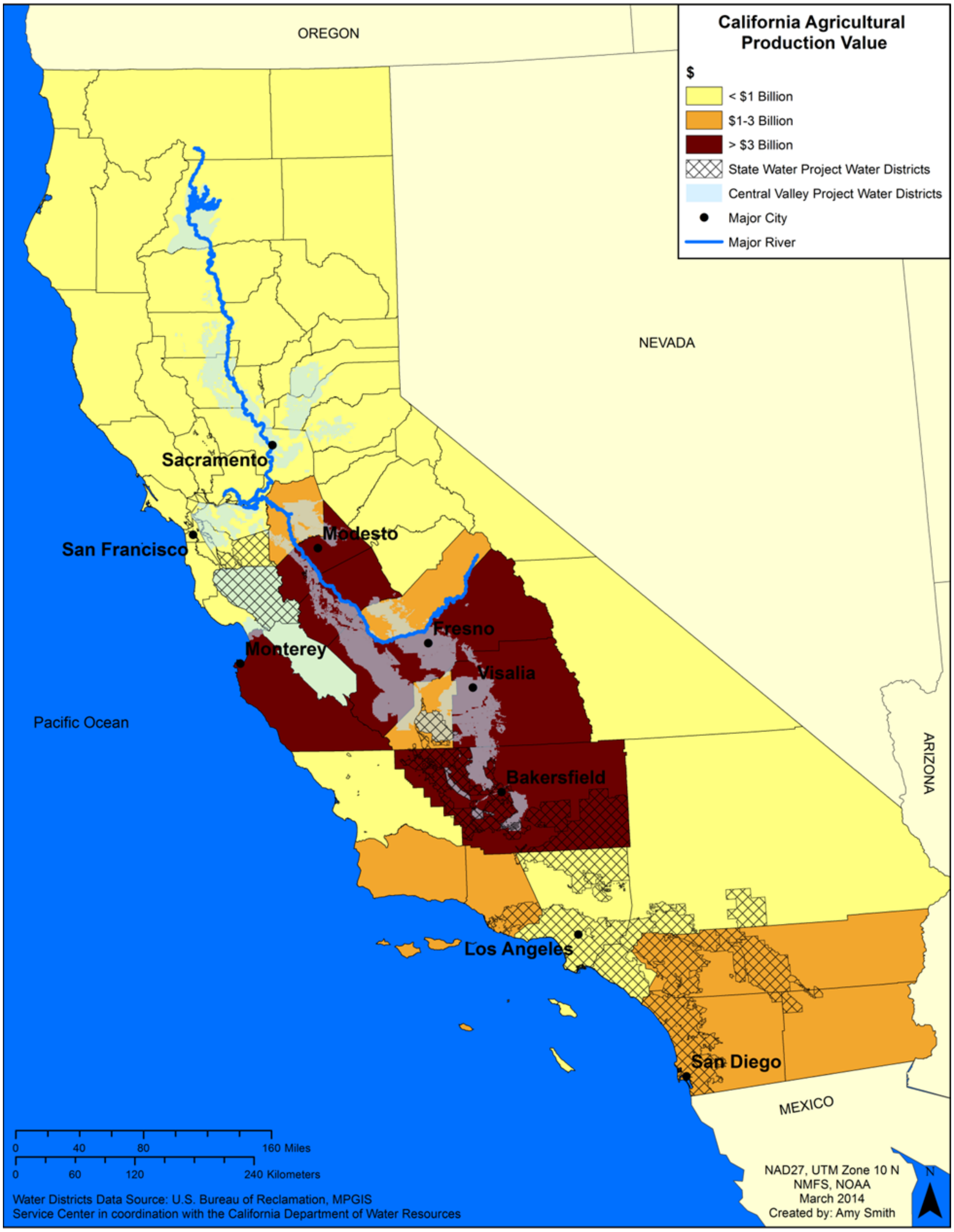

Figure S13. California counties categorized by the value of agricultural production, overlaid with location of Central Valley Project and State Water Project water districts.

In addition to government projects such as the CBP, CVP and SVP, the 1906 General Dam Act authorized the construction of hydropower facilities by private utility companies, subject to Congressional approval (Gillilan and Brown 1997). In addition, thousands of smaller diversions throughout the Pacific region provide water for irrigation, livestock and other uses. These latter dams have not been fully inventoried and the extent to which they impede passage and impair spawning and rearing habitat is not well documented (National Research Council 1995).

Effects of federal dams on salmonids are addressed via ESA Section 7 consultations. Non-federal hydropower dams are regulated by the Federal Energy Regulatory Commission (FERC) as authorized by the 1935 Federal Power Act (FPA). The FPA has received renewed interest since the 1990s, when many of the FERC licenses originally issued 40-50 years ago are getting ready to expire – providing an opportunity for NOAA to prescribe salmonid passage as a condition for relicensing (Gillilan and Brown 1997). Federal legislation intended to address more regional salmon needs include the 1980 Pacific Northwest Power Planning and Conservation Act, the 1984 Trinity River Basin Fish and Wildlife Management Act, the 1992 Central Valley Project Improvement Act, and the 2009 San Joaquin River Restoration Act (Gillilan and Brown 1997). Regional programs that include salmonid recovery as a goal have also been established, including the Puget Sound Shared Strategy, the California Bay-Delta Program (later superseded by the Delta Stewardship Council), and the Trinity River Restoration Program.

Structural and operational methods of mitigating effects of dams on anadromous salmonids include (for example) fish ladders for upstream migration, trucking and barging of downstream migrants, screens and spillways to reduce turbine mortality, reservoir drawdown to reduce travel time through reservoirs, control of predators (e.g., pikeminnow), and flow augmentation to facilitate migration and reduce water temperatures. Some of these measures (e.g., trucking and barging) have detractors as well as supporters (National Research Council 1995). Another mitigation method is dam removal. Examples of recent dam removals are Condit Dam on the White Salmon River in Washington, Elwha and Glines Canyon dams on the Elwha River in Washington, Savage Rapids Dam on the Rogue River in Oregon, and San Clemente Dam on the Carmel River in California. Consideration is being given to removal of four private hydroelectric dams on the Klamath River (USDOI undated). Hatcheries have also been used as mitigation for dams.

HATCHERIES

The first salmon hatchery in the Pacific region was established in 1872 on the McCloud River (a Sacramento River tributary) by the U.S. Fish Commission to incubate eggs to supplement declining Atlantic salmon stocks. In 1877, the Commission established a second hatchery on the Clackamas River near Portland, with partial funding from Columbia River cannery operators who were concerned about the decline of prized spring-run Chinook. Later that year, when a flood wiped out the eggs at the Clackamas hatchery, eggs were transferred from the McCloud hatchery to Clackamas (Gregory and Bisson 1997). Although the canners were critical of transplanting practices and the fish culturists' unwillingness to distinguish salmon by runs, they had no scientific basis for their objections. Federal and state hatcheries proliferated during the late nineteenth century. Hatchery releases became a means of increasing salmon abundance, and hatchery monitoring was too sporadic to demonstrate otherwise. Pacific salmon eggs were transferred to distant hatcheries in Australia, New Zealand, South America, Europe, Iowa and the Great Salt Lake (Taylor 1999). Fish culturists of the time did not understand that salmon had natal streams. The role of genetics in distinguishing races and runs and the importance of reproductive isolation for salmon survival was not known until the 1930s (Lichatowich 1999,Taylor 1999).

The 1938 Mitchell Act provided funding (derived from user fees collected from commercial fishermen) to recover salmon runs affected by water diversions, dams, pollution and logging on the Columbia River (Taylor 1999). From the 1930s through the early 1950s, support for hatcheries declined due to poor returns and disease problems (National Research Council 1995). However, a 1946 amendment to the Mitchell Act (originally intended to rebuild salmon runs upstream of Bonneville Dam) stimulated a resurgence of hatcheries in the Columbia Basin. The newly amended Act allowed hatcheries to be constructed downstream of Bonneville to mitigate for effects on salmonids upstream – including extirpation of runs above Grand Coulee Dam, which had no fish passage. These downstream hatcheries encouraged dam construction by providing a means of mitigating their effects. In the 1960s, the introduction of pasteurized and formulated feeds reduced the incidence of disease and further raised expectations that hatcheries provide would effective mitigation for dams. More than 80 hatcheries have been built in the Columbia Basin – including 39 Mitchell Act hatcheries (National Research Council 1995). Use of hatcheries, fishways, screens and spillways as dam mitigation measures has been common practice in California as well as the Northwest. Figure S14 depicts major hatcheries and dams in the Pacific region.

The location of hatcheries below dams has dramatically shifted the distribution of salmon production from upper to lower river areas. The selective preference of fishery managers for certain species (primarily coho salmon and fall-run Chinook salmon) altered relative species abundance. Between 1850 and 1977-1981, coho doubled as a proportion of total abundance while sockeye salmon and chum salmon virtually disappeared in the Columbia Basin (National Research Council 1995). Collection, mating, rearing and release practices may cause hatchery fish to experience loss of genetic variation, inbreeding depression, and poor adaptation to wild conditions. Hatchery releases can affect the effective population size of wild fish and lead to outbreeding depression through hybridization with less fit hatchery fish (Naish et al. 2008). Hatcheries can also induce genetic changes in wild populations even with no interbreeding of hatchery and wild fish, due to selective fisheries that target hatchery fish (Reisenbichler 1997). Efforts such as the Pacific Northwest Hatchery Reform Project, established by Congress in 2000, are underway to review and reform hatchery practices (Hatchery Scientific Review Group 2009).

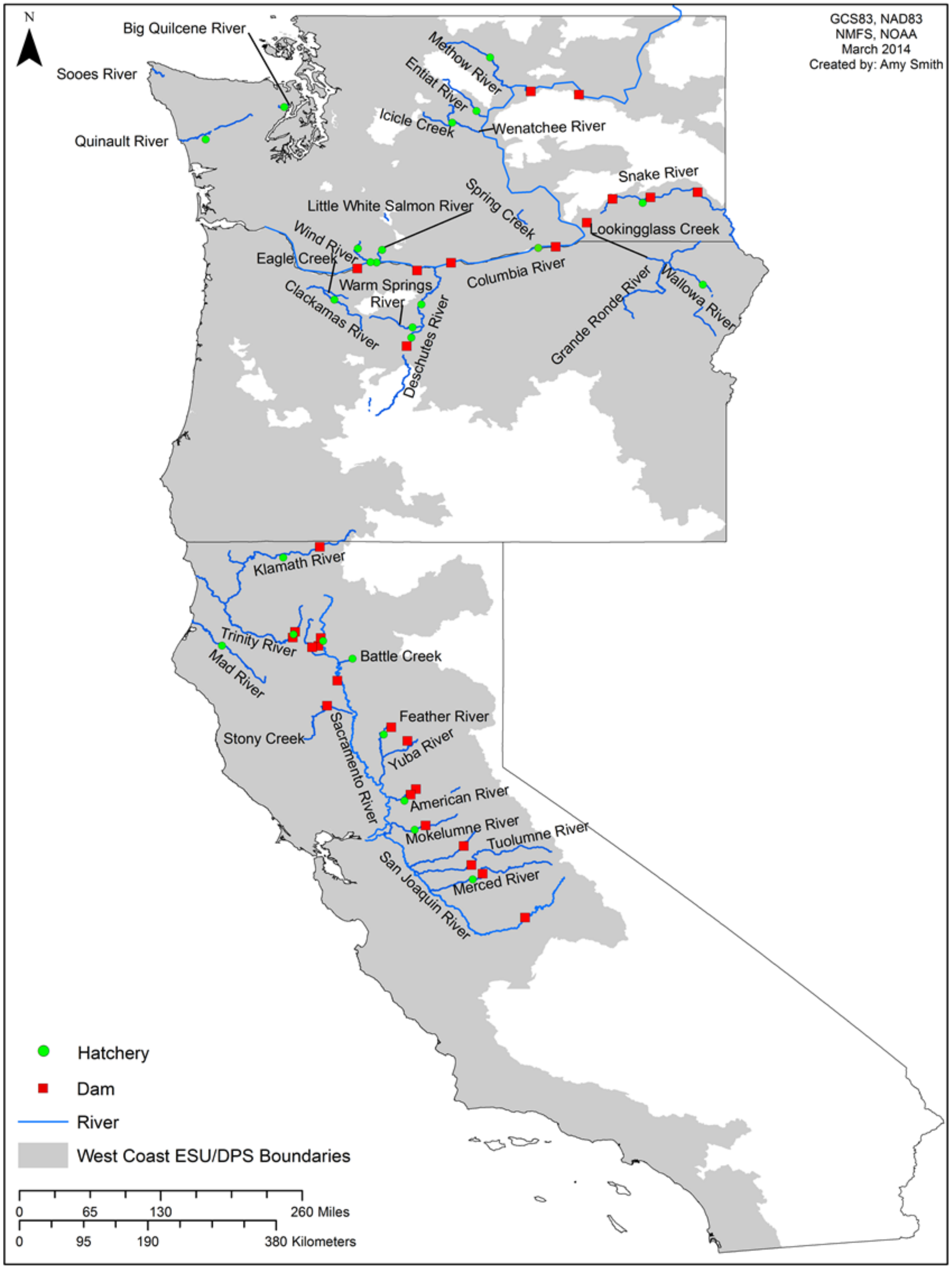

Figure S14. Distribution of major hatcheries and dams

FISHERIES

For thousands of years, salmon has been an integral part of the cultural, subsistence and ceremonial life of Native American tribes who inhabited coastal and riparian areas of the Pacific region (Taylor 1999). Traditional fishing methods included gillnets, dip nets, traps, fishing spears, and communal fishing dams (weirs); preservation methods included drying and smoking (Taylor 1999, Biedenweg et al. 2014). The first return of salmon to the river was celebrated by a First Salmon Ceremony (Lichatowich 1999). Such rituals served to reflect, reinforce, and transmit to younger generations a world view of salmon as integral to and inseparable from the river ecosystem. Historical salmon harvests by tribes in California's Central Valley are thought to have exceeded 8.5 million pounds annually. Tribes on California's north coast annually consumed up to 2000 pounds of salmon per family (Boydstun et al. 2001). Among the Northwest coastal tribes, annual per-capita consumption was more than 365 pounds (Taylor 1999). Salmon was also valued as an item of trade with non-coastal tribes. During the 1700s, it is estimated that aboriginal harvests of salmon were at least as large as contemporary peak tribal harvests (Breslow 2011). Mining, logging and agriculture interfered with the ability of Native Americans to engage in traditional subsistence and cultural practices. Native Americans found themselves at great disadvantage in the competition with Euro-Americans for resources, including salmon (McEvoy 1986), and were driven off their lands to reservations (Taylor 1999). For Euro-Americans, salmon was a commodity to be harvested rapidly and exhaustively (not moderated by long-term communal interests), processed using high-volume preservation methods (canning), and sent via high-volume transport (railroads and steamships) to distant markets (Taylor 1999).

The commercial salmon fishery developed in California several years after the 1848 Gold Rush. The fishery was initially constrained by the volatile labor market (with fishers distracted by the lure of gold) and a shortage of salt for curing. In 1864, William Hume established the first salmon cannery in the Pacific region on the Sacramento River (National Research Council 1995). Completion of the First Transcontinental Railroad in 1869 provided access to markets for cheap, nutritious canned salmon on the East coast, Britain and elsewhere. Although Hume moved his cannery to the Columbia River in 1866 to take advantage of the larger runs, 20 other canneries were established on the Sacramento/San Joaquin River system by 1880. Production peaked at 200,000 cases (from a harvest of 12 million pounds) in 1882, then declined as intensive fishing and habitat destruction from mining, logging and agriculture took their toll on the runs. Declines of spring Chinook meant greater reliance on other Chinook salmon runs and coho salmon. The last cannery closed in 1919, and California proceeded to close its river fisheries – the Mad River in 1919, the Eel River in 1922, the Smith and Klamath/Trinity rivers (including tribal fisheries) in 1933, and the Sacramento and San Joaquin rivers in 1957 (Boydstun et al. 2001).

The cannery relocated by Hume from California to the Columbia River in 1866 was the first in the Northwest. By 1883, more than 50 canneries were located on the Columbia River and its tributaries, and many others on the coastal rivers of Oregon and Puget Sound. Astoria, where fattened salmon enter the Columbia River, became a center of canning operations. Harvest of spring-run and summer-run Chinook salmon peaked in 1883. When spring-run Chinook salmon became scarcer, canneries began processing what they considered to be inferior fall-run Chinook salmon, then turned to sockeye salmon and steelhead and later coho salmon and chum salmon (Lichatowich 1999). Total cannery pack of all species on the Columbia River peaked in 1895 at 635,000 cases, then declined as the runs were diminished by habitat deterioration and overfishing (Lichatowich 1999); the last Columbia River cannery closed in 1975 (National Research Council 1995). Cannery operations on Puget Sound dated from 1877 and peaked in 1913 at 2.5 million cases (Schwantes 1996). Production in Alaska and British Columbia eventually eclipsed that of Oregon and Washington.

Advances in processing technology – e.g., mechanized fish gutting and cutting, the sanitary can (which replaced glass jars), and the double seamer (which eliminated hand soldering) – were adopted shortly after 1900 and stimulated the demand for fish to feed the cannery lines (Lichatowich 1999). The number of gillnetters on the Sacramento River increased from 60 boats in the 1850s to 1500 boats in 1884 (McEvoy 1996). The fleet expanded its range to the lower Sacramento/San Joaquin River, San Francisco Bay, and rivers north of San Francisco (Boydstun et al. 2001). In the Northwest, gillnetters attached additional webbing and weights to their nets to better entangle salmon at more river depths. In the late 1870s, poundnets and fishwheels – stationary structures that blocked river passage in areas where the current ran too slow or too fast for gillnets – were introduced into the fishery. In the 1890s, teams of horses were used along tidewater to haul beach seines packed with salmon. On the Columbia River, the number of gillnets increased from 900 in 1880 to 2,200 in 1894, traps increased from 20 in 1881 to 378 in 1895, and fishwheels increased from one in 1882 to 57 in 1895 (Taylor 1999). At times, huge piles of salmon were left to rot when harvests from large runs exceeded processing capacity (Taylor 1999).

The ocean commercial troll fishery began in Monterey Bay in the 1880s with boats powered by sails. In about 1908, several powered gillnetters from the Sacramento River entered the Monterey troll fishery. As gasoline engines replaced sails, the fleet grew to 200 boats and expanded northward to Fort Bragg, Eureka and Crescent City by 1916. During the 1920s and 1930s, the arduous work of retrieving lines, weights and fish was done by hand. Power gurdies, invented in the 1940s, were quick adopted by professional trollers. The troll fleet grew from about 570 boats in 1935 to 1,100 by 1947. By the 1970s, the fleet comprised almost 5,000 vessels, many of them part-time summer fishers who had other jobs during the year (Boydstun et al. 2001). In the Northwest, gasoline engines began replacing sails on Columbia River gillnet boats by the late 1890s and also gave rise to a mobile troll fleet at the river mouth. The Columbia River troll fleet grew from a handful of boats in 1900 to 500 boats in 1915 to 1500 boats by 1919 (Lichatowich 1999).

For many years tribal fishing rights throughout the Pacific region had been eroded by Euro-American fishing interests, backed by the power of the state. By the late 1950s, Native Americans in the Northwest were taking less than 5% of the harvest but continued to be resented and blamed for the declining runs. In 1969 Judge Robert Belloni affirmed the right of treaty Native Americans in Oregon to “a fair and equitable share of all fish.” In 1974, Judge George Boldt ruled that treaty Native American tribes in Washington were entitled to 50% of the available harvest in their "usual and accustomed areas" (Taylor 1999). In 1993, the DOI Office of the Solicitor issued an opinion requiring that 50% of the allowable harvest of Klamath-Trinity salmon be reserved for the Yurok and Hoopa Valley tribes (U.S. Department of the Interior 1993). In 1977, four Columbia River tribes with treaty fishing rights formed the Columbia River Inter-Tribal Fish

Commission to coordinate management. Today, Columbia River fisheries are managed by Oregon, and Washington state agencies and Columbia River treaty tribes under the Columbia River Compact. Puget Sound fisheries are jointly managed by Western Washington treaty tribes and the state of Washington. The Yurok Tribe and Hoopa Valley Tribe in northern California manage their own fisheries. Native American tribes are actively represented on the Pacific Fishery Management Council and Council advisory committees.

Northwest treaty tribes continue to actively pursue their right to harvest fish. In Washington State, the current major legal debate centers on whether treaty tribes have the right to the existence of salmon at all, and therefore the right to maintain healthy salmon habitat – which stretches from open ocean to mountain streams. Court rulings are so far in the tribes' favor. Note that with the potential to require landscape-wide habitat protection across all jurisdictions, the treaty fishing right is possibly the most powerful legal tool available to protect salmon habitat in this region—even more powerful than the Endangered Species Act, which is effectively limited to federal jurisdictions (Lombard 2006, Blumm and Steadman 2009). Note also that with the Boldt and Belloni decisions, the treaty right to take fish would protect the non-tribal half of salmon harvest from habitat destruction, as well (Breslow 2014b).

Commercial salmon fishing is now conducted in the ocean by trollers equipped with fish-finding sonar, radio communications and global positioning systems, and Native American gillnetters in Puget Sound and the Klamath/Trinity River. Combined tribal and non-tribal commercial salmon landings have declined since the mid-1980s, although ex-vessel value has generally increased since 2000 (Figure S15). The divergence between the landing and revenue trends is largely due to price increases over the past decade.

Since 1981, landings have generally been highest in the Native American sector of the salmon fishery and lowest in the commercial troll sector (Figure S16). Revenues have followed a similar pattern since the mid-2000s but show a less consistent pattern in previous years (Figure S17).

Since 1981, salmon landings have been consistently and considerably higher in Puget Sound than in other regions (Figure S18). Revenues show a similar but less marked pattern (Figure 11), as the species composition of Puget Sound harvest tends toward lower-priced species.

Figure S20 shows a general decline in ex-vessel salmon prices from 1981 to the early 2000s, followed by a general increase. The increase is particularly marked for troll-caught Chinook. The "missing" prices for pink salmon reflect the fact that the pink salmon fishery is open every other year.

Figure S21 depicts major salmon ports, selected on the basis of average 1981-2012 ex-vessel salmon revenue (base year=2012). The fishery extends from central California to Washington, with troll gear predominating in coastal ports in California and Oregon, net gear predominating in Columbia River and Puget Sound ports, and both gears used in coastal Washington ports. The mix of troll and net gears in Ilwaco and Neah Bay reflects the fact that both non-tribal commercial trollers and tribal gillnetters land in these ports. The importance of salmon relative to total fishing activity varies by port (Norman et al. 2007, Speir et al. 2014).

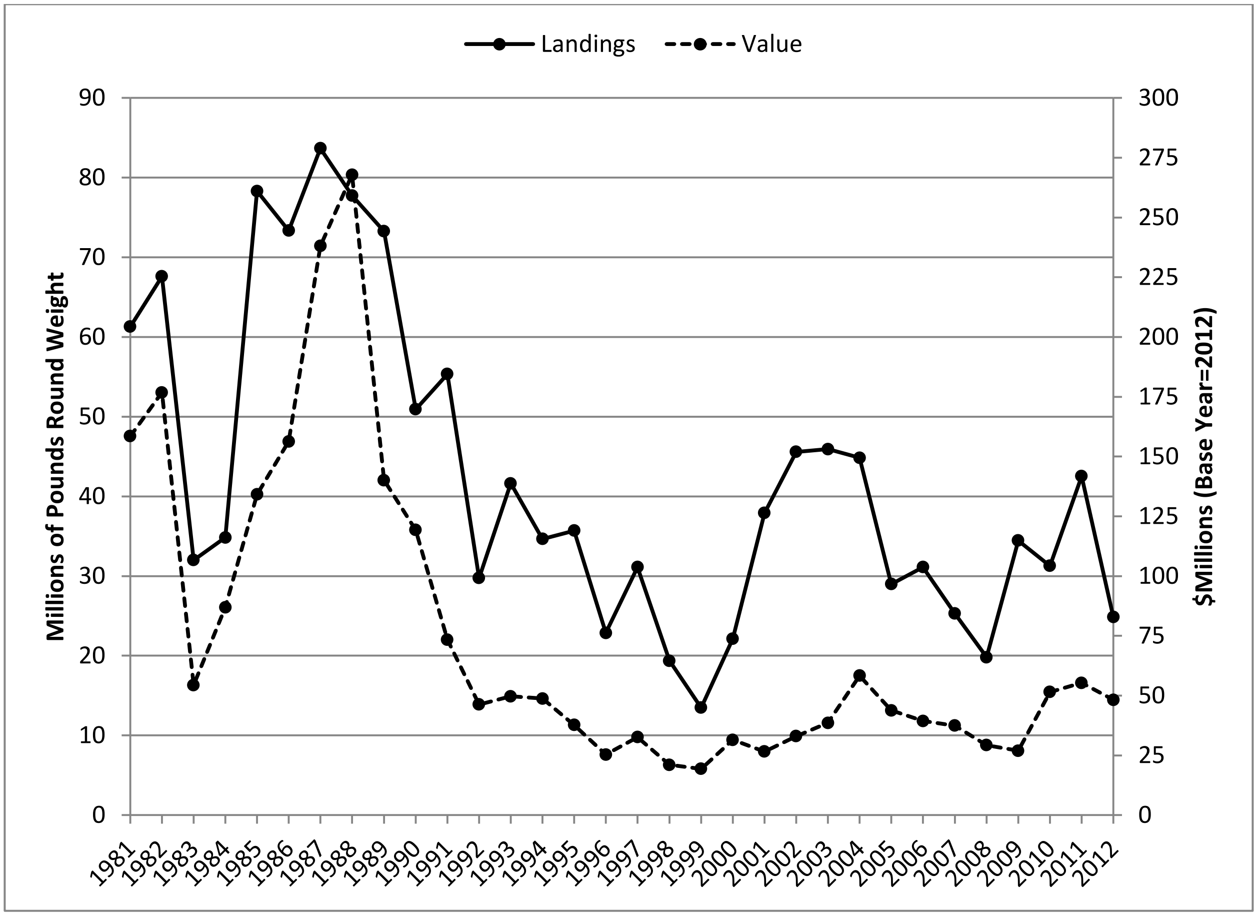

Figure S15. Landings and ex-vessel value (base year=2012) of tribal and non-tribal commercial salmon fisheries in California, Oregon and Washington, 1981-2012 (source: PacFIN).

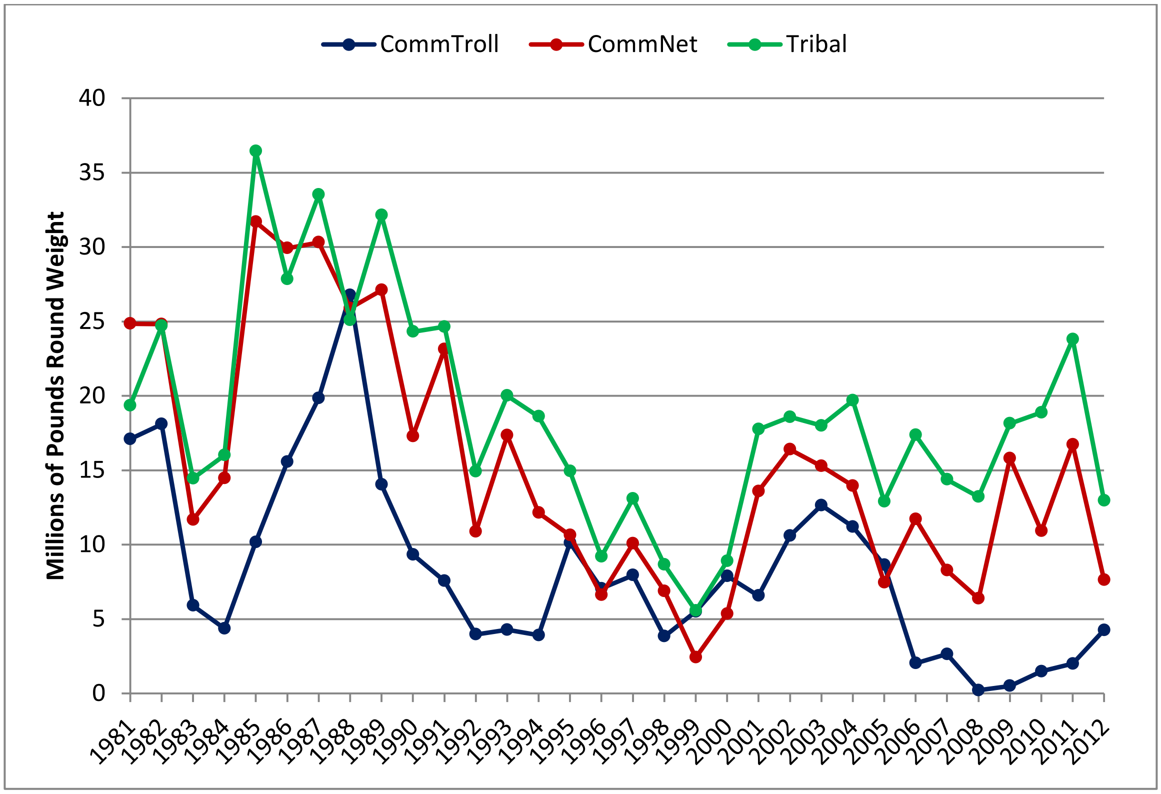

Figure S16. Coastwide tribal and non-tribal commercial salmon landings, by fishery sector, 1981-2012, combined for the three states (source: PacFIN).

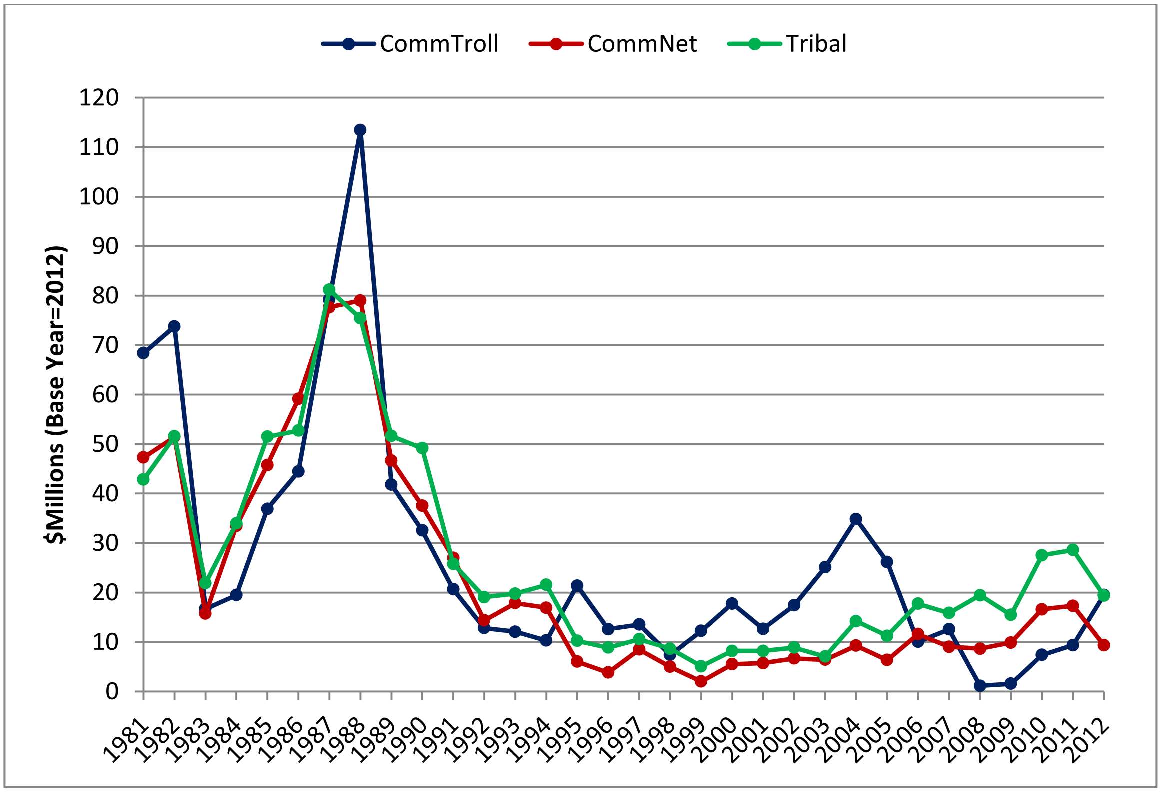

Figure S17. Ex-vessel value of coastwide tribal and non-tribal commercial salmon landings (base year=2012), by fishery sector, 1981-2012, combined for the three states (source: PacFIN).

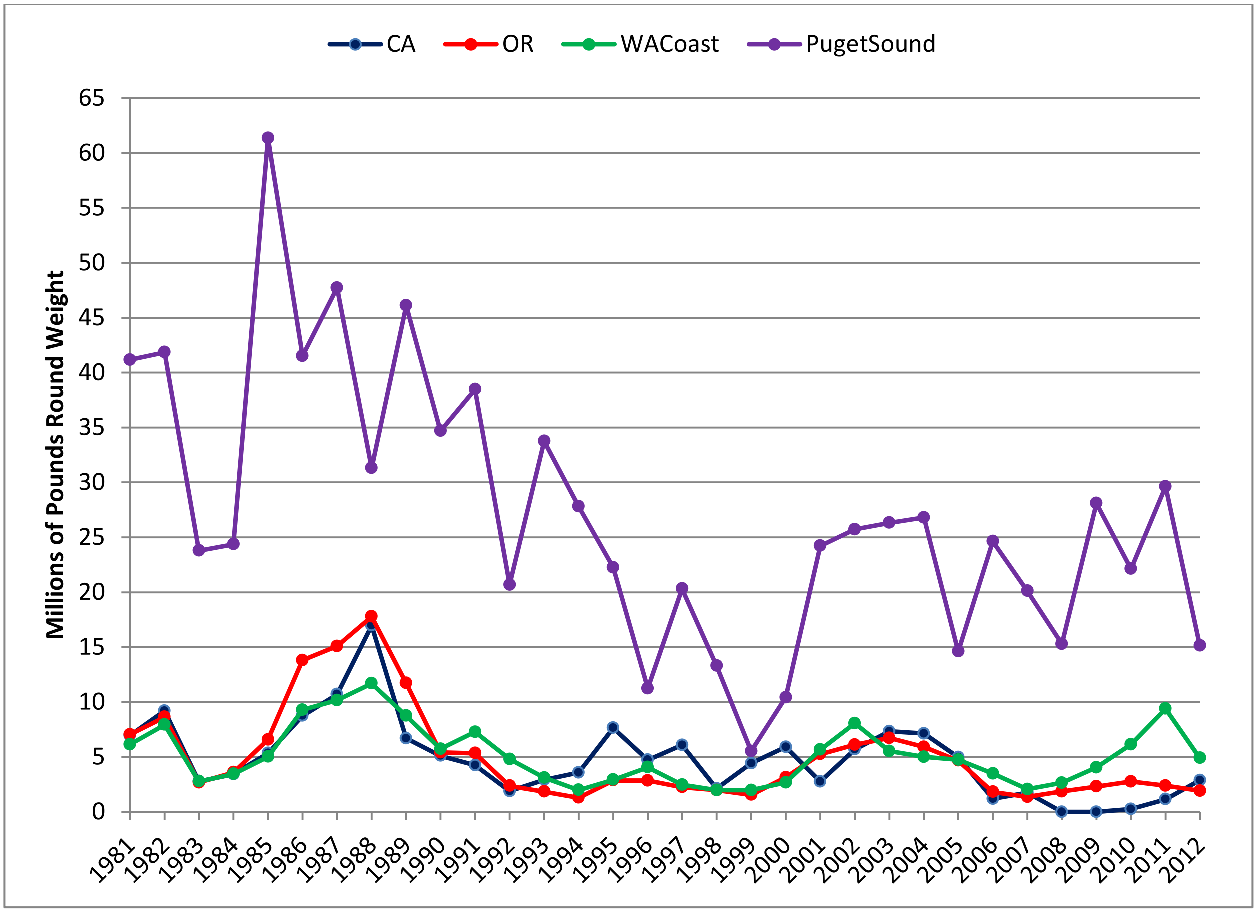

Figure S18. Tribal and non-tribal commercial salmon landings, by region, 1981-2012 (source: PacFIN).

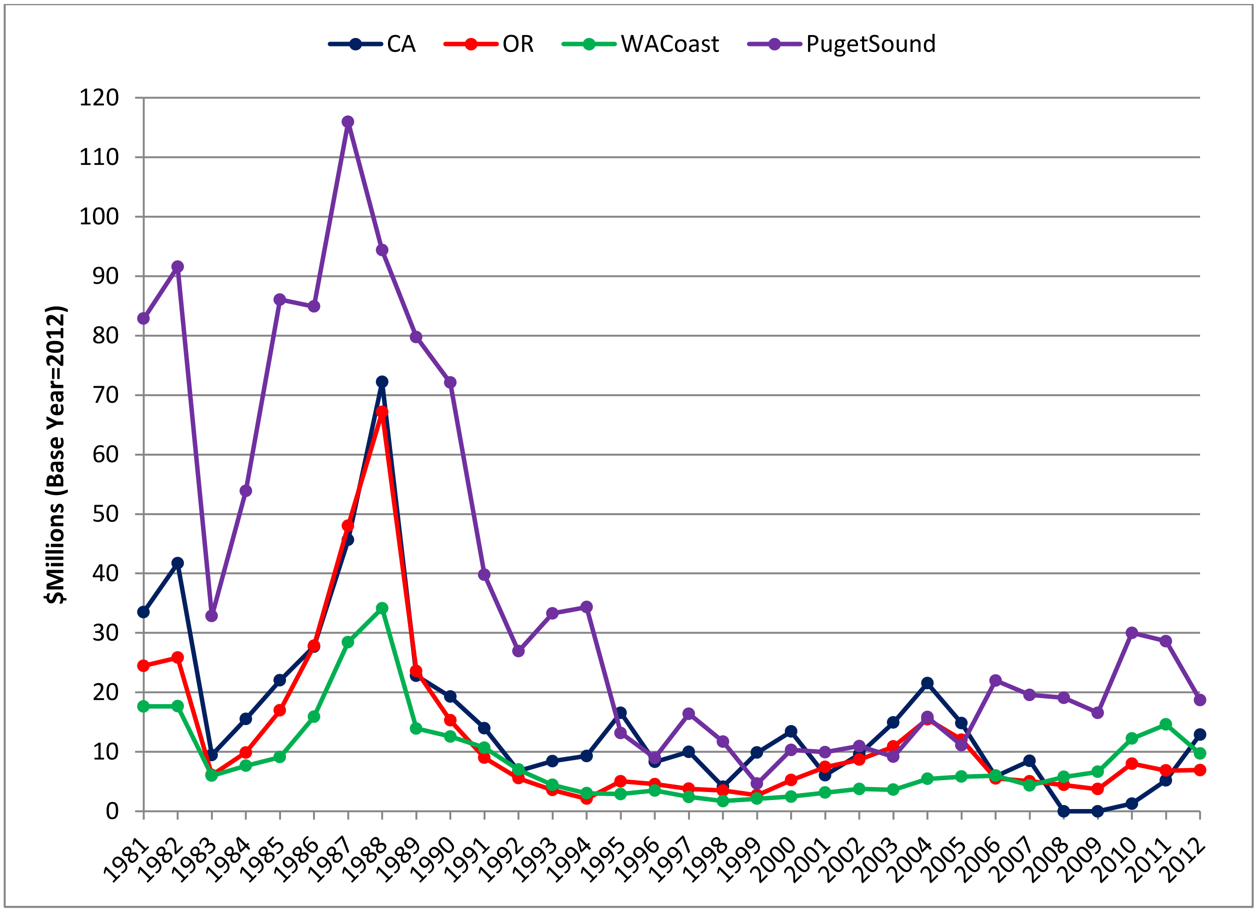

Figure S19. Ex-vessel value of tribal and non-tribal commercial salmon landings (base year=2012), by region, 1981-2012 (source: PacFIN).

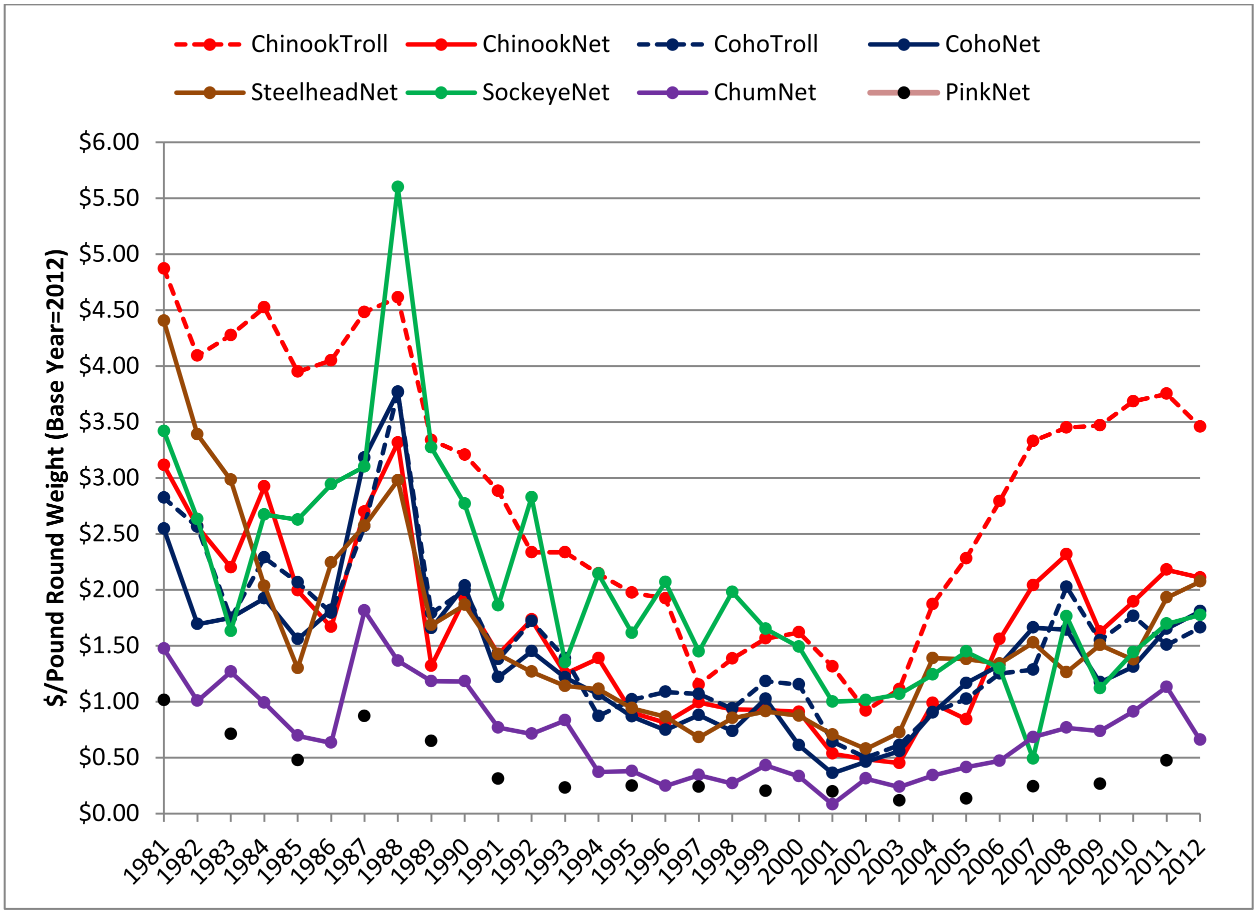

Figure S20. Ex-vessel prices of major commercial salmon stocks (base year=2012), 1981-2012, combined for the three states (source: PacFIN).

Figure S21. Major commercial salmon ports, based on average 1981-2012 ex-vessel revenue (base year=2012) (source: PacFIN).

RECREATION, TOURISM, EDUCATION, VOLUNTEERISM, MANAGEMENT, RESEARCH

Recreation, tourism, education, volunteerism, management and research are rich areas of wellbeing related to salmon and will be discussed in more detail in future iterations of the CCIEA. Salmon are an important part of the cultural heritage of the Northwest (Lichatowich 1999). Fishing, agriculture, hatcheries, and dams provide recreational as well as commercial opportunities. Recreational salmon fisheries occur both in river and in the ocean. Agricultural and fishing communities and organizations sponsor events such as salmon, garlic, artichoke and apple festivals. Farmers markets and wine tasting are venues for recreation and tourism as well as consumer purchases. Hatcheries provide tours and exhibits that educate the public regarding the salmon life cycle. People visit rivers as well as hatcheries to watch salmon spawn. Dam operators provide reservoir recreation and dam tours; dam operations also indirectly affect the timing and location of river recreation due to effects on the river's flow regime. People join organizations such as fishing clubs and 4-H, and volunteer for river cleanups and other habitat restoration activities. A large research establishment – including federal and state agencies, educational institutions (including land grant and sea grant universities), and private entities – conducts research relevant to the understanding, management and improvement of agriculture, fisheries and forestry. The complex challenge of salmon recovery has required new forms of social organization and cooperation, and has also engendered passionate debates among diverse communities in the region who are grappling with how to ensure that salmon, fishing, and other resource-based livelihoods can survive in an increasingly globalized economy and urbanizing landscape (Breslow 2014a).

HUMAN POPULATION TRENDS

Figure S22 describes population growth in the Pacific states during 1850-2010. Estimates for the earlier decades only partially reflect the Native American population, as Native Americans were not fully enumerated in the U.S. Census until 1890. Diseases brought by Euro-Americans had already caused an 80-90% decrease in the Native American population by the late 1850s (National Research Council 1995). Additional Native American mortality resulted from conflicts with the U.S. Army and white settlers, and loss of traditional modes of subsistence (Harden 1996).

Population trends (reflecting births, deaths and net migration) since 1850 represent the cumulative effect of many factors – e.g., climate and weather, natural resource abundance, economic opportunities, amenities and disamenities of various types (e.g., schools, traffic), aesthetic qualities – and are suggestive of the pressures exerted on a state's water resources. Many major metropolitan areas in the Pacific region (e.g., Willamette Valley, Puget Sound, Columbia Plateau, San Francisco Bay, Central Valley - Figure S24) are located on historical wetlands and/or near major rivers, tributary junctions and estuaries. According to Gregory and Bisson (1997), "Though forest practices and, to a much lesser degree, agricultural practices have drawn intense scrutiny resulting in more protective land-use regulations, urbanization and industrial development tend to cause the most extensive alternation of aquatic ecosystems." Population growth has been disproportionately high in California, as indicated by the growth and density trends in Figures S22 and S23. In drought-prone California, population growth exacerbates longstanding and heated conflicts over water. Due to the large water storage and conveyance systems that link the entire state, salmon is affected by population pressures even in areas well outside the range of its habitat.

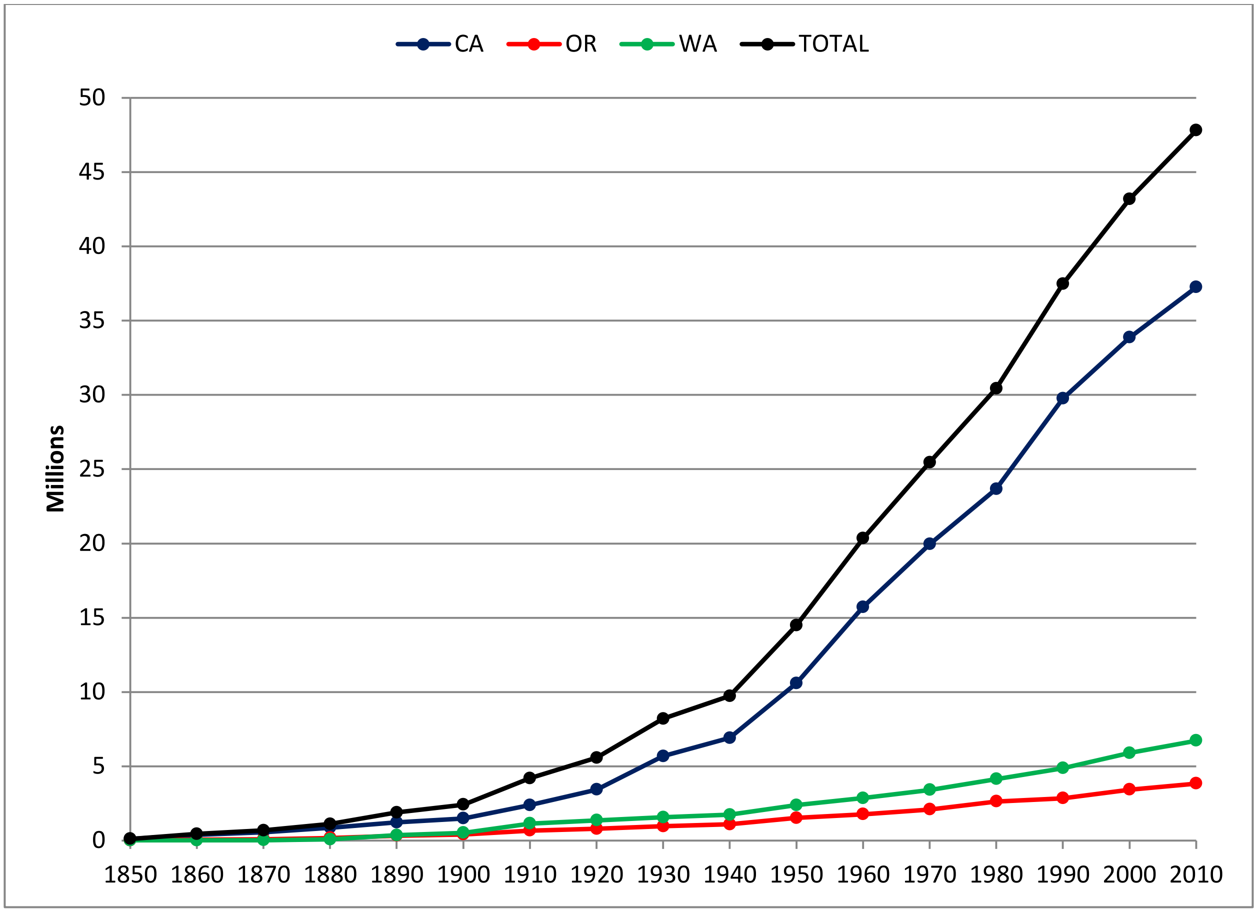

Figure S22. U.S. population by state and total all states, 1850-2010 (source: U.S. Census).

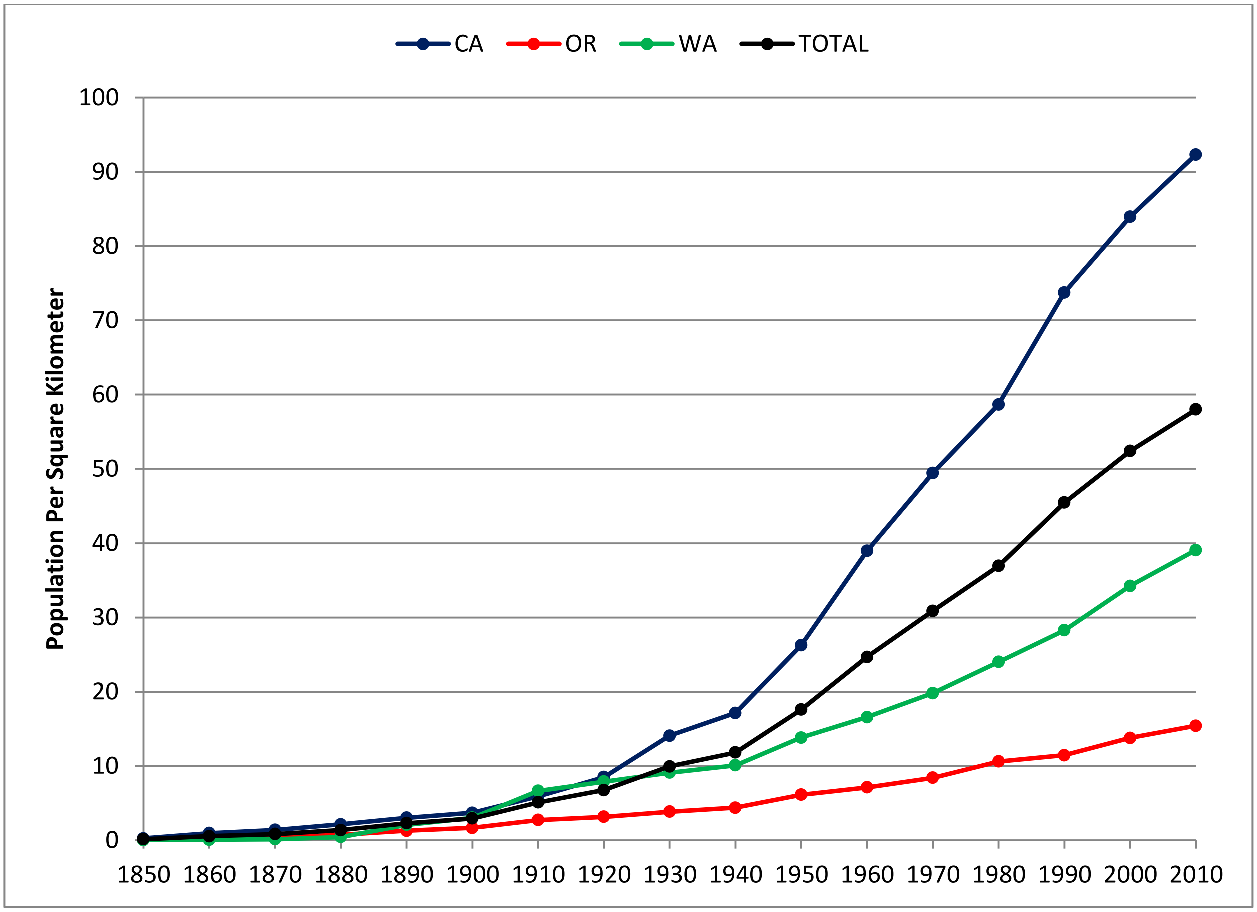

Figure S23. Population density (population per square kilometer) by state and total all states, 1850-2010 (source: U.S. Census).

Figure S24. Counties categorized by Rural-Urban Continuum Code (1= county in metro areas of 1 million population or more, 2= county in metro areas of 250,000 to 1 million population, 3= counties in metro areas of fewer than 250,000 population, >3=county in non-metro areas) (source: USDA Economic Research Service).

SUMMARY AND NEXT STEPS FOR INTEGRATING THE HUMAN DIMENSION

Salmonid populations and habitat have been influenced by dynamic interactions between natural landscape features (e.g., resource abundance, climate, topography) and human activities such as fur trading, mining, logging, agriculture, dams, hatcheries and fisheries. Historical development of these activities was largely driven by economic interests and encouraged by robust market demand and prices, improvements in extractive and processing technologies and transportation, and expansionist government policies. Most of these activities (other than fur trading) continue to the present day in some form. Public policies have changed over time, from an ethos of laissez faire resource extraction to one that also considers effects of extraction on wild salmon populations and the habitat and ecological processes that affect salmon and steelhead. Such policy shifts reflect a recognition of salmon and salmon habitat as components of human values and well-being.

Most of the quantitative information provided in this section pertained to outputs from commercial activities (e.g., timber production, agricultural values, salmon harvest). Additional work is needed to consider other indicators that are inclusive of other aspects of human well-being. An important next step toward operationalizing the CCIEA is to identify goal(s) that managers wish to achieve by considering salmon in a CCIEA framework, as those goals will affect model specification and the types of indicators appropriate for inclusion in the model.

CCIEA PHASE IV: NEXT STEPS

At this point of the CCIEA, salmon indicators have been developed for quantifying the status of a number of Chinook salmon and coho salmon populations across the California Current. We have, in this current iteration of the CCIEA, explored the potential effects of current and future environment on Chinook salmon and coho salmon populations. We also examined the temporal, spatial and demographic aspects of resource uses that interact with salmon populations. In the next iteration of the CCIEA, Phase IV, we will examine how various management strategies across the environmental and resource needs landscape may potentially affect salmon populations. Specifically, we are developing salmon life-cycle models across the California Current that can be used to simulate the potential responses by the salmon populations and to human well-being under a suite of potential management scenarios including but not limited to hatchery practices, river flow management, and fishery regulations. As well, in future reports we will include status and trends of steelhead salmon.

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CCIEA PHASE III REPORT 2013: HABITAT

SELECTING AND EVALUATING INDICATORS FOR HABITATS WITHIN THE CALIFORNIA CURRENT LARGE MARINE ECOSYSTEM

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EXECUTIVE SUMMARY

Aquatic habitats are at the heart of science and management mandates for the National Marine Fisheries Service, the National Ocean Service, and other state and federal agencies charged with natural resource management. However, our lack of understanding of the condition of aquatic habitats and their importance for living resources often hinders decisions regarding effective habitat conservation. In the face of needs for better habitat monitoring, a critical starting point is addressing what indicators of habitat should be tracked to determine ecosystem health. This is the initial stage of Integrated Ecosystem Assessments (IEAs). In addition to indicator selection, IEAs examine status of trends of indicators, analysis of risk to ecosystem components, and management strategy evaluation of potential actions society can take to facilitate a sustainable ecosystem.

This document summarizes indicator selection for habitats in the California Current large marine ecosystem (CCLME). Indicator selection followed from a conceptual model that identifies four major habitat types (freshwater, estuary and nearshore, pelagic, and seafloor environments) in the CCLME and their links to other IEA components including environmental drivers, anthropogenic pressures, species-specific ecosystem components, and human wellbeing. Given contrasting habitat needs supporting the great diversity of species inhabiting the CCLME, we subdivided habitat indicators into these four habitat types, and for each type we identified indicators of habitat quantity, habitat quality, and the main anthropogenic pressures impacting them. We rated each indicator using 18 criteria encompassing 1) primary or scientific support considerations, 2) data limitations, and 3) other considerations related to the application of indicators by society (Levin & Schwing 2011). During the evaluation, we recognized that habitat data are often limited in time or across space, so we strove to identify indicators useful for either mapping or trend analysis for quantity or quality of each habitat type.

We identified 33 priority indicators for freshwater, estuary/nearshore, pelagic, and seafloor habitats, which are listed in Table H5 on page 66. In general, metrics related to estimating areal extents of substrate or biogenic habitat were identified as priority indicators of habitat quantity, while metrics of habitat quality often focused on well-measured attributes such as temperature, dissolved oxygen, and nutrients. Common pressures were urban and agricultural land cover and effects of fishing.

The suite of priority habitat indicators addressed many of the linkages we postulated in our conceptual model, although more attention to cross-habitat linkages and connections to human wellbeing is warranted. In addition, indicator selection also provided insight into monitoring gaps for potential indicators with strong scientific support, and

points to the need for better monitoring programs, data collection, and synthesis of these sources of habitat information. Our next steps will be to summarize data for priority indicators and thereby examine status (maps and summaries of current condition) and region-specific trends over time.

DETAILED REPORT

INTRODUCTION

Habitats are the interface through which climate drivers and human activities influence biota and the matrix through which ecosystem interactions occur. Aquatic habitats are therefore at the heart of science and management mandates (e.g., Essential Fish Habitat (EFH), Critical Habitat under the Endangered Species Act) for the National Marine Fisheries Service (NMFS). Unfortunately, we are still in the foundational stages of identifying important habitats for fish and other living resources at all life stages, the extent that people have affected these habitats and the benefits they receive from them, how natural resource managers can apply habitat information, and how habitat restoration and protection stand to improve the status of commercial fisheries and other trust resources.

The importance of improving habitat information for management has been laid out in the Habitat Assessment Improvement Plan (HAIP; NMFS 2010) and NOAA's Habitat Blueprint (NMFS 2012). The HAIP's primary goals are two-fold: 1) to improve habitat assessments so that EFH can progress from presence/absence to higher information levels, and 2) to integrate habitat information into stock assessments to better assist in stock management. As noted in the HAIP and NMFS' Our Living Oceans Habitat (NMFS 2014), the numerous unanswered scientific questions concerning species-habitat interactions and habitat status hamper our ability to effectively implement actions to benefit the living marine resources managed by NOAA. Following from these directives, building a comprehensive Habitat Component into Integrated Ecosystem Assessments (IEAs) should improve their applicability to NMFS's management and by extension their utility as tools for ecosystem management (Levin et al. 2009). This document lays the groundwork for integrating a habitat component into the California Current Integrated Ecosystem Assessment (CCIEA), specifically describing those indicators of habitat that should be characterized and monitored.

As described by Levin et al. (2009), IEAs synthesize information on ecosystem attributes and associated human dimensions in order to inform ecosystem management objectives. The framework of IEAs includes scoping, indicator selection, analysis of status and trends of indicators, risk analysis, management strategy evaluation, and feedbacks for adaptive management. The CCIEA is accomplishing these steps through integration of several socio-ecological components. Components include natural drivers and human activities influencing the ecosystem, as well as benefits people derive from the CCLME. Other components address the major groups of NOAA trust resources (e.g., salmon, groundfish, marine mammals) (see 2012 web report at http://www.noaa.gov/iea/CCIEA-

Report/index.html). Because habitats physically connect trust resources with most climate drivers and human activities, and because many of NMFS management actions concern habitat conservation measures, habitat constitutes an important additional component with unique indicators, risks, and management scenarios.

Habitats for NOAA trust resources on the Pacific Coast extend from the mountains (for Pacific salmon) to ocean depths greater than 1,000 m at the seaward edge of the Exclusive Economic Zone (EEZ). A broad set of indicators will therefore be needed to adequately characterize the habitats that support the rich diversity of aquatic life on the Pacific Coast. To that end, we have developed the Habitat Component of the CCIEA in four general groupings relevant to NMFS management – freshwater, estuarine/nearshore, pelagic, and seafloor habitats.

In this report, we provide a rationale for determining indicators of habitat quantity and quality and anthropogenic activities that can impact these four habitat types. We start with a conceptual model that frames the Habitat Component in the context of other components being examined in the CCIEA. Next we describe the process by which four teams of scientists selected indicators. Finally, we describe the priority indicators of habitat quantity and quality selected for future status and trends analysis. In the parlance of ecosystem management (Levin et al. 2009, Levin & Schwing 2011, Halpern et al. 2012), these indicator datasets can be used to determine the health of habitats in the California Current Ecosystem.

CONCEPTUAL MODEL OF HABITAT FOR THE CCIEA

The aquatic habitats of the California Current Ecosystem span the Pacific Coast from Northern Washington to Southern California. Spaulding et al. (2007) and others (Parrish et al. 1981, Allen et al. 2006) divide this region into two large marine ecosystem provinces, with the boundary at Point Conception. These have subsequently been divided into four ecoregions: the Salish Sea (Puget Sound and the Straits of Georgia and Juan de Fuca); the Washington, Oregon, and Northern California Coast; Central California, and the Southern California Bight (Sullivan-Sealey and Bustamante 1999, Spaulding et al. 2007, Parrish et al. 1981, Allen et al. 2006, Longhurst 1998, Pelc et al. 2009). The boundaries of these ecoregions are based on geomorphic, hydrodynamic, and biogeographic breaks at Cape Flattery, Cape Mendocino, and Point Conception, as well as on the political borders of the United States (i.e., those with Canada and Mexico). Freshwater systems entering the two provinces have been divided into six ecoregions based on the biogeography of associated fish (Abell et al. 2008). Nevertheless, these classification systems largely are consistent in encompassing habitats of the important freshwater and marine species in the CCIEA.

Aquatic habitats constitute essential links in the broader socio-ecological conceptual framework of the CCIEA. In this framework, habitat is the matrix for interactions of physical and anthropogenic activities with living marine resources, or in IEA terms, the ‘components’ of ecological integrity. Habitat is paralleled in this larger schema with social systems and governance as the matrix for interactions by broad social and economic forces with components of human wellbeing. Our conceptual model of the Habitat Component incorporates multiple drivers, interconnections among habitat types, the living marine resources using habitats, and the benefits of these habitats to people (Fig. H1).

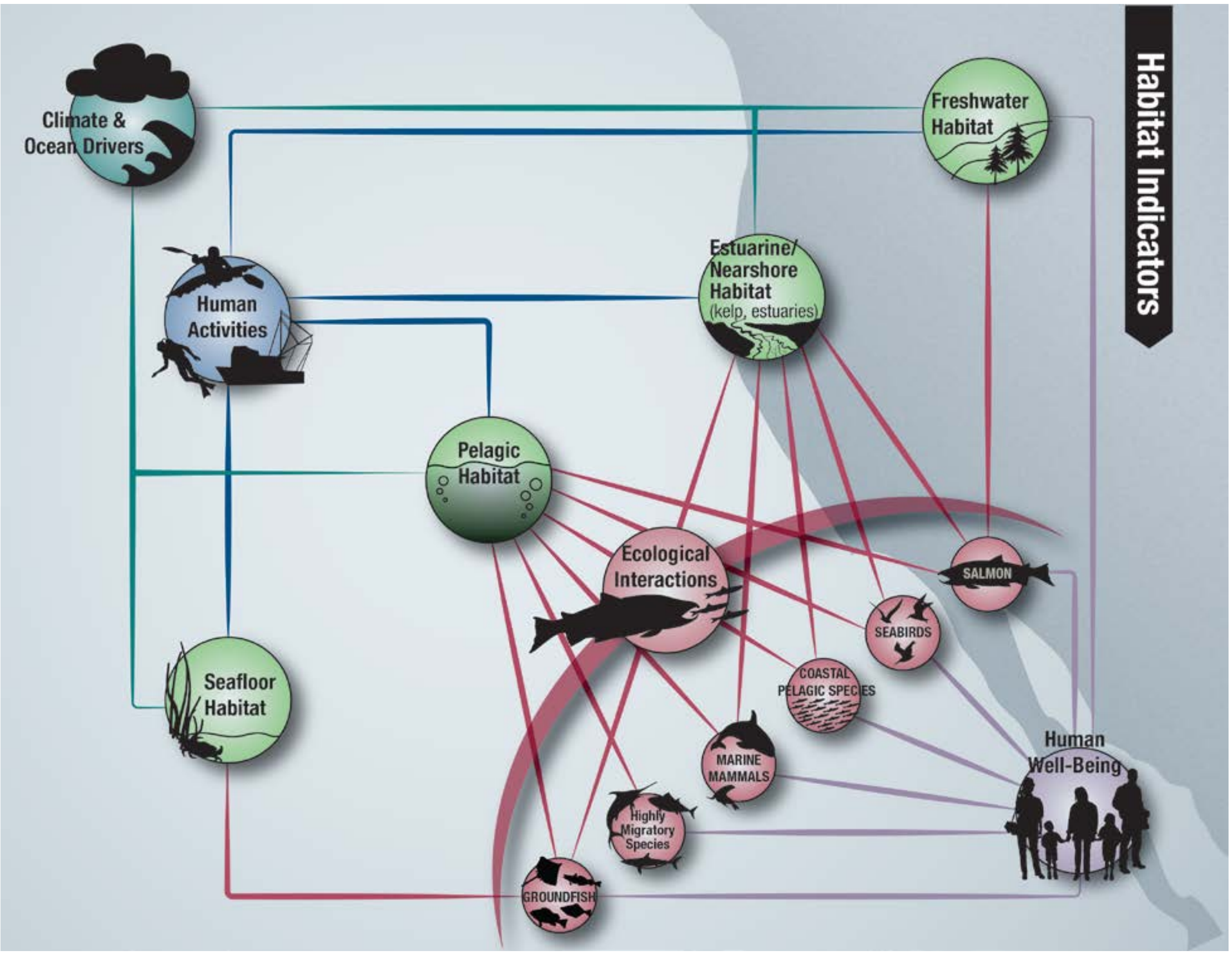

Figure H1. Conceptual model of the Habitat Component of the CCIEA. Freshwater, estuary/nearshore, pelagic, and seafloor habitats influence each other and provide the interface that affects associated organisms. Climate and ocean drivers directly affect habitats and associated organisms via the habitat interface. Human activities affect all habitats and, in turn, human wellbeing is influenced both by the habitats and the organisms (HMS = highly migratory species, CPS = coastal pelagic species) they influence. One main effect not illustrated is the direct effects of human activities on organisms via fishing.

Both freshwater and marine aquatic habitats in the California Current are the products of dynamic geologic, geomorphic, and climate processes with various time scales, many of which occur more slowly than annual rates of change. Habitat-forming processes influence the distribution of aquatic habitats, and processes interact in a manner largely following topography and bathymetry: freshwater systems influence estuary and nearshore environments, which in turn interact with pelagic environments, which subsequently influence seafloor habitats. Regional climate drivers shape temperature, precipitation, coastal storms, wind patterns, currents, and upwelling, and these natural drivers are now shifting as a result of anthropogenic climate change. In addition, people affect the quantity and quality of habitats through a number of activities occurring at more local levels. Different habitats support different complexes of species, which use multiple habitats during their life history. These species can be influenced directly by people through fisheries and indirectly by anthropogenic activities affecting habitats. People subsequently benefit from habitats directly, and indirectly from the fisheries they support. Hence, anthropogenic activities and species responses are the outcome of ecological interactions that occur in the context of habitat. This habitat context can modulate predator-prey interactions and interspecific competition, and influence the intensity of fisheries and other human activities upon NOAA trust resources.

Following from this conceptual model, fish and other species experience climate and most human activities (fisheries being a partial exception) through their interaction with multiple habitats. Consequently, efforts to rebuild imperiled stocks need to carefully consider the quantity and quality of habitats. To address the question “What is the state of the California Current Ecosystem?,” we need indicators of habitat quality and quantity and to define the anthropogenic and climate pressures directly affecting them. Hence, we use a more detailed tier of conceptual models to describe the specific climate and ocean drivers and anthropogenic activities affecting specific habitat types, the consequential effects on different fisheries, and the benefits people gain from these habitats (Fig. H2).

FRESHWATER HABITAT

Freshwater habitats linked to the CCLME include river and lake systems connecting to the Pacific Ocean, spanning the West Coast of North America from the Fraser River in the north to the Tijuana River in the south. These habitats are intimately connected to their watersheds, and habitat conditions within rivers and lakes are strongly influenced by the landscapes that surround them (Fausch et al. 2002). Broadly speaking, freshwater habitat types include streams, rivers, floodplain channels, ponds, and lakes. Headwater streams are small and generally much steeper than rivers in the lower basins, and diversity of habitats generally increases in the downstream direction because the array of lentic and lotic habitat types grows as rivers and floodplains widen. However, this general trend is often interrupted by geologic controls, tributary junctions, and glacial features, which can create local variation in habitat types and diversity (e.g., Benda et al. 2004). This diverse array of habitats supports a large number of anadromous species, including salmon, sturgeon, lamprey and others. Rivers and their floodplains also support a wide range of ecosystem services to people, including water supply, land for agriculture or development, transportation, recreation, energy generation, cultural resources, and commercial, sport, and subsistence fisheries (Zedler & Kercher 2005, Nelson et al. 2009).

Freshwater habitat conditions are controlled by a hierarchical suite of climatic, geomorphic, and biological processes (e.g., Beechie et al. 2013). The spatial structure of the river network and locations of canyons, floodplains, and tributary junctions are controlled by geology and topography (which we refer to as landscape template). This template is relatively immutable over common management timeframes, meaning that land and water uses generally do not alter the structure of the drainage network, locations of canyon and valley reaches, or the slopes of valleys. The landscape template controls the range of potential habitat conditions that can be expressed within any particular reach. Conditions that are expressed at any point in time are then controlled by watershed-scale and reach-scale processes. The key watershed-scale processes are the runoff and erosion processes that produce stream flow and sediment supply to rivers. Hydrologic processes control the flow regime and sizes of streams and rivers, whereas sediment supply exerts strong controls on channel form and dynamics. Hence, these processes control basic channel patterns in the river network, including cascade, step-pool, plane-bed, and pool-riffle channels in small streams, and straight, meandering, island-braided, and braided channels in large rivers. Smaller scale habitat features such as pools and riffles, or habitat quality attributes such as food web structure and temperature regimes, are controlled by reach-scale riparian processes including root reinforcement of stream banks, supply of wood to channels, stream shading and nutrient supply.

Anthropogenic activities include direct modification of river channels and their floodplains, alteration of stream flow and erosion regimes, removal or altering riparian forests, and addition of pollutants or pesticides. Some of the earliest alterations to freshwater habitats in the California Current were channelization of rivers and draining water from floodplains for agriculture (e.g., Beechie et al. 1994). At the same time, most of the wood in rivers was removed to facilitate navigation (e.g., Collins et al. 2002). These early modifications to river floodplains dramatically reduced freshwater habitat availability and diversity throughout the region. Sediment supply from mountain regions was increased in some areas by logging practices or hydraulic mining, and sediment supply is locally decreased downstream of numerous dams in the region. In some rivers, flows are dramatically reduced by water extraction for irrigation or municipal water supplies. Riparian alteration is ubiquitous in the region.

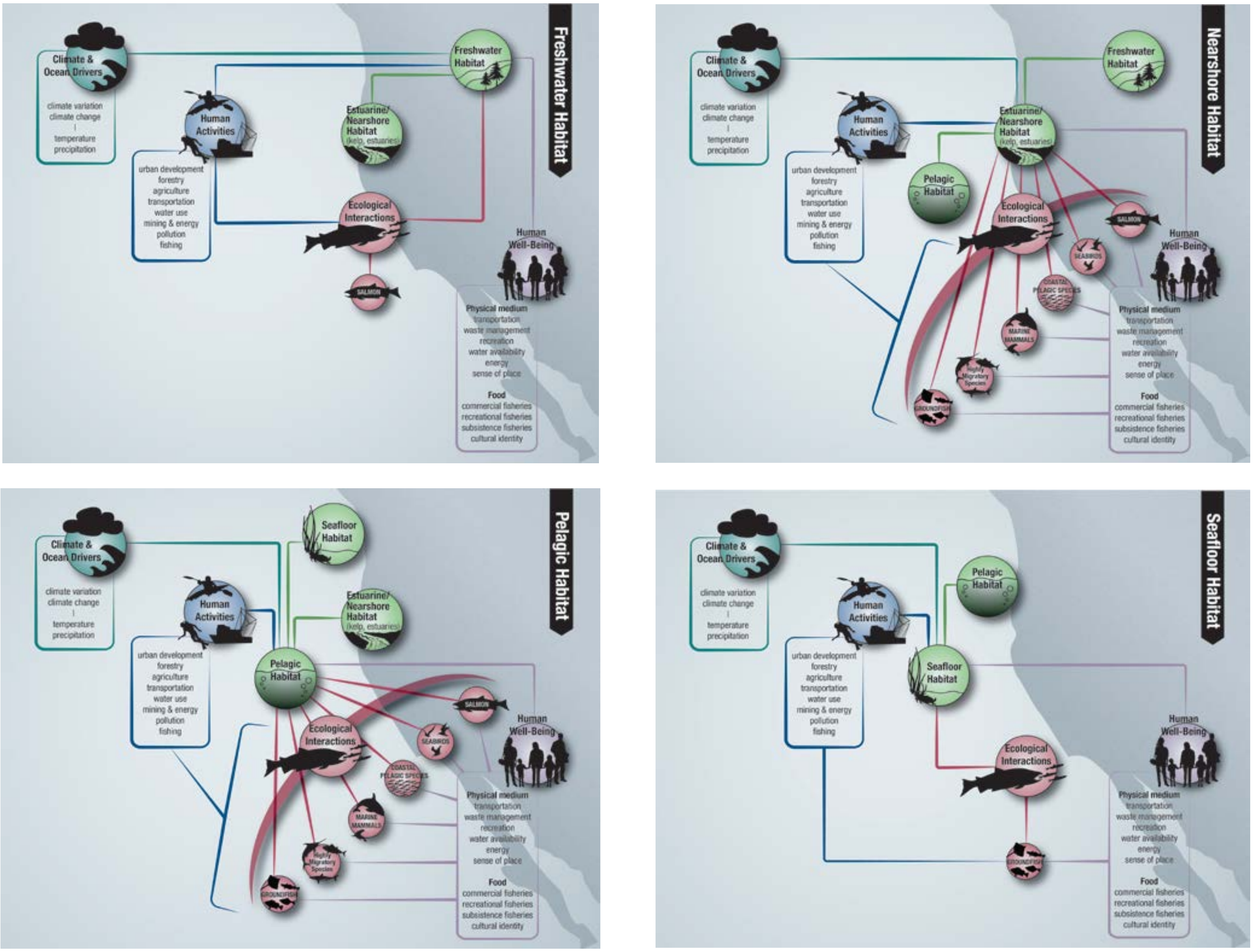

Figure H2. Application of the general conceptual model to each habitat type. The major differences among models are the between-habitat linkages, and the specific climate drivers, human activities, aspects of human wellbeing, and other ecosystem components affected.

ESTUARY AND NEARSHORE HABITAT

We define estuary and nearshore habitats as those systems that are strongly influenced by both marine and freshwater or terrestrial processes. On the Pacific Coast, the extent and variety of estuary and nearshore environments is limited by the steep topography of land and continental shelf. Estuaries include enclosed bodies of water – drowned river mouths, embayments, lagoons, and fjords – characterized by tidal influence such as water level fluctuations and daily to seasonal variation in salinity (Potter et al. 2010). Nearshore environments include rocky shores, beaches, and headlands directly adjacent to marine waters (Inman & Nordstrom 1971). The spatial extent of estuaries includes their floodplain from head of tide (the maximum upstream extent of tidal influence) to the marine shoreline, which has often been defined as the mean lower low water line (e.g., Wessel and Smith 1996) but can extend subtidally through distributary channel networks, deltaic formations, and hydrodynamic processes. Nearshore environments are intertidal and subtidal water column and benthic habitats as deep as 40 m (NMFS 2014), which define the bathymetric limit of kelp that play a key role in nearshore systems. Nearshore systems are defined laterally by littoral drift cells, which are discrete zones created by topography and longshore currents that define sediment sources (e.g., rivers, bluffs), transport, and deposition (e.g., beaches) (Inman & Nordstrom 1971). While estuary and nearshore systems on the Pacific Coast are strongly influenced by marine processes, they are differentiated from pelagic and seafloor environments by 1) the influence of terragenic geomorphic processes creating shallow and sheltered habitats, and 2) the presence of sunlight throughout the water column, creating opportunities for submerged aquatic vegetation such as eelgrass and kelp.

Estuary and nearshore habitat quality and quantity are shaped by large scale geomorphic and climate drivers as well as human activities at local spatial extents. Geomorphic processes such as river flow, tidal action, fetch, and currents (Uncles 2002) make estuary and nearshore systems highly dynamic and subject to a wide variety of climate forcings. Consequently, estuary and nearshore environments might be expected to be influenced by the gamut of climate processes affecting Pacific Coast systems, from changes in precipitation, river flow and water temperature, to variation in sea level and storm surges. In addition, estuary and nearshore environments are foci for human activities and therefore are at risk from a broad array of anthropogenic activities, including habitat loss, hardening of wetland and shoreline habitats, and water quality impairments from pollution and nutrient inputs. In addition to local drivers, estuary and nearshore habitats are expected to be influenced by freshwater processes, and to link with pelagic habitat processes.

The species benefiting from estuary and nearshore habitats include salmon, groundfish, coastal pelagics, seabirds, and marine mammals (NMFS 2014), as well as numerous other fish and invertebrates. Salmon and some groundfish and coastal pelagic stocks use estuaries and nearshore environments as rearing areas during juvenile life stages, and these consequently are hotspots for feeding by seabirds and marine mammals. All species are influenced indirectly by human activities that affect these habitats, but also directly via commercial and recreational fishing. Habitats can conceivably play a mediating role in the extent to which people can affect stocks by fishing. For example, higher habitat complexity or the remoteness of habitat areas might reduce fishing pressure. In addition to supporting fisheries and aquaculture, estuary and nearshore habitats provide a number of benefits to people as sites for transportation, alternative energy infrastructure, waste disposal and water diversions, and recreation. Additional benefits to human wellbeing include sense of place, local ecological knowledge, cultural heritage, and quality of life.

PELAGIC HABITAT

The pelagic habitat for the CCLME extends from the west coast of Vancouver Island south to the subtropical waters off Baja California, Mexico (20-25°N), offshore to the EEZ, and vertically in the water column where the bottom is deeper than 40 m. While the four ecoregions in the CCLME are based on relatively static boundaries, the resultant oceanography and pelagic habitat is a highly dynamic product of oceanic processes (e.g., frontal structure, thermocline depth). Vertically, pelagic habitat is defined as below the surface and above the bottom, but more specifically as the Epipelagic (0-200 m, euphotic), Mesopelagic (200-1000 m), and Bathypelagic (>1000 m bottom depth). The pelagic habitat is characterized by strong physical forcing at a suite of space and time scales, beginning with wind-driven upwelling, nutrient delivery to the photic zone, phytoplankton blooms and the commencement of the pelagic food-web. Bathymetric and topographical features such as capes, islands, rocky banks, and canyons and oceanographic features including eddies and fronts affect the quality of pelagic habitat and their resultant food webs. For two reviews of pelagic ecosystems, see Checkley and Barth (2009) and Bograd et al. (in press).

The base of the pelagic food web is the phytoplankton, which bloom seasonally as nutrients are upwelled into the photic zone (Kudela et al. 2008). The predominant phytoplankton groups within the California Current include diatoms, dinoflagellates (which commonly form harmful algal blooms (HABs)), and cyanobacteria. Secondary producers include microzooplankton, crustacean zooplankton, gelatinous zooplankton, euphausiids, ichthyoplankton, and small pelagic fish. Copepods serve as critical prey resources for a suite of predators. Gelatinous zooplankton have boom and bust cycles where they can serve as an important predators of zooplankton and ichthyoplankton, although the forcing of these blooms is not well understood. Euphausiids, primarily the species *Euphausia pacifica* and *Thysanoessa spinifera*, are another critical link in the food-web of the CCLME (Brinton & Townsend 2003). These species primarily eat diatoms and small zooplankton, and in turn are the food for many species of fish, birds, and marine mammals. Euphausiids often form large conspicuous schools and swarms that attract larger predators, including baleen whales (Croll et al. 2005). Due to their quick feeding rates, high growth rates, and role as a key prey resource for many species, euphausiids are a major node of energy flow in the CCLME (Field et al. 2006).

Forage fish are both iconic components of the CCLME as targets of historic fisheries, and important components of the CCLME pelagic food-web. Dominant species include northern anchovy (*Engraulis mordax*), Pacific sardine (*Sardinops sagax*), Pacific mackerel (*Scomber japonicus*), and jack mackerel (*Trachurus symmetricus*) and they feed almost exclusively on phytoplankton, zooplankton, and ichthyoplankton. The forage fish complex is prey for predatory fishes, seabirds, and marine mammals. Further offshore, particularly outside of the reach of the more productive upwelled waters, mesopelagic fish and invertebrates that vertically migrate daily (e.g. myctophids, penaeid shrimp, squids) serve as a key prey resource in the oligotrophic pelagic habitat (Brodeur & Yamamura 2005).

Many mobile species migrate seasonally throughout the CCLME (Horne and Smith, 1997; Agostini et al. 2008; Checkley & Barth, 2009; Block et al. 2011), while other species come from across the Pacific. The California Current is a hotspot for a high diversity and abundance of top predators as a result of the seasonal upwelling and nutrient-rich waters that result in an abundance of prey (Block et al. 2011). Large pelagic migratory fishes are abundant and support a number of fisheries, including hake, salmon, rockfishes, billfishes, sharks, and a few species of tuna (Field et al. 2010; Block et al. 2011; Glaser 2011; Preti et al. 2012; Wells et al. 2012). Seabird species include local breeders and oceanic migrants, both of which rely on the CCLME as their foraging grounds (Shaffer et al. 2006; Yen et al. 2006; Mills et al. 2007; Kappes et al. 2010). Six pinniped species breed on the coast of California, with many of these animals foraging in offshore waters (Antonelis & Fiscus 1980) alongside a high diversity of cetacean species (Barlow & Forney 2007). These predators have all evolved strategies to benefit from the seasonal productivity of the CCLME while minimizing interspecific competition.

Pelagic habitat is identified by predictable and persistent areas of productivity or aggregation of lower organisms at multiple trophic levels (Sydeman et al. 2006, Hazen et al. 2013). These persistent features, often called marine hotspots, are characterized by increased trophic exchange and often are of high ecosystem importance (Sydeman et al. 2006, Hazen et al. 2013). Bathymetric features such as seamounts, shelf breaks, or islands create hotspots by increasing upwelling or creating eddies (Reese & Brodeur 2006). Mesoscale features such as large eddies and fronts can entrain productivity or prey species that in turn result in increased productivity and aggregation (Logerwell and Smith 2001; Palacios et al. 2006; Yen et al. 2006). Persistent upwelling locations can result in greater productivity than surrounding areas, attracting forage species and top predators (Palacios et al. 2006 and references within). In the CCLME, three coastal hotspots of primary production are apparent via remotely sensed imagery: Cape Mendocino to Point Arena, Bodega Head to Point Sur, and Cape San Martin to Point Arguello (Palacios et al. 2006). Beyond the magnitude of chlorophyll *a* blooms, persistence indices identified similar hotspots in time and space that are productive regions for a large portion of the year, and provide reliable prey resources for seabirds in the area (Suryan et al. 2012). Much less is known about the vertical components of marine hotspots compared to the horizontal, but temperature ranges, light penetration, nutria-clines, and dissolved oxygen can all serve to define pelagic habitat. Shoaling oxyclines can lead to vertical displacement of organisms, mismatches in predator and prey based on oxygen tolerances of prey and predator (Chan et al. 2008, Stramma et al. 2011), and also create pathways for invasion by species such as for Humboldt squid, *Dosidicus gigas* (Stewart et al. 2012).

Marine hotspots are important economically, as aggregations of forage fishes and predatory fishes create reliable fishing spots. Some of the most valuable fisheries in the CCLME include forage fish (e.g. anchovies and sardines), salmon species, highly migratory fishes such as tunas and swordfish, and squids. These species are not only economically important, but they also support an associated suite of human wellbeing benefits, such as fishing heritage, sense of place and social networks within fishing communities. In addition, shipping vessels travel across the pelagic realm transporting goods to the western Pacific and beyond. In the pelagic realm, fisheries, shipping, and use by culturally important species are the primary ecosystem services. Anthropogenic pressures such as climate change, ocean acidification, ship strikes, pollution, and oil spills can affect living resources in the pelagic realm. For example, earth system models of climate change project widespread shifts in fish distribution and abundance through habitat change in the Pacific (Polovina et al. 2011, Hazen et al. 2013). There have been proposals and limited implementation of alternative energy sources including wind and tidal energy where installation could negatively impact habitat in the pelagic realm.

SEAFLOOR HABITAT

Seafloor habitats in the CCLME extend from the neritic zone (ca. 40 m water depth) to the abyssal plain (>3,000 m) at the seaward boundary of the U.S. EEZ. The geospatial framework for seafloor habitats follows on recent analyses by NMFS (2013) identifying four ecoregions and three depth zones. The three depth zones are the continental shelf, upper and lower slopes. The water depth of the continental shelf break in the region varies slightly, but is generally described to be ca. 200 m water depth. The boundary between the upper and lower continental slope was placed at 700 fathoms (1280 m) water depth, corresponding in general to the deepest extent of the groundfish fishery.

Seafloor habitats in the California Current are shaped by a diverse array of physical processes. The CCLME is located in an active tectonic region where the Pacific plate is subducting under the North American plate. Tectonic and seismic activity has transformed the continental margin in several ways over the past several millennia. The continental margin of the northern and central ecoregions is characterized by a relatively narrow (8-40 km) continental shelf and steep slope. Although the shelf is dotted with occasional rocky banks (e.g., Heceta, Cordell), it comprises mostly sandy and muddy sediments. Several large submarine canyons cut across the shelf, often ending in a sedimentary fan at the base of the slope. Other unique seafloor habitats in the northern and central ecoregions include slumps, landslides, and cold methane seeps. In the southern ecoregion, submerged islands and banks interspersed with deep basins characterize what is known as the southern California borderlands. These rocky banks support some of the most diverse assemblages of fishes and macroinvertebrates in the CCLME.

Seafloor habitats provide critical ecological services. Most importantly, the physical structure of seafloor habitat is necessary for sessile invertebrates to attach, and for sedentary invertebrates and fishes to forage and seek refuge. Habitat studies over the past few decades have greatly contributed to our knowledge of how demersal fishes and macroinvertebrates use seafloor habitats. One general conclusion from these analyses is that abundance and diversity of these organisms are influenced by primarily by physical attributes, and associations with biogenic habitat are much weaker or statistically undetectable (Tissot et al. 2006).

At a larger scale, climate and associated changes in seawater temperature and chemistry influence distribution of fishes and invertebrates. For example, Pacific hake, the most migratory species of groundfish in the CCLME, make large seasonal migrations between winter spawning and summer feeding grounds. The northern extent of these migrations is greatest during El Niño years. Other fishes, such as rockfishes and many marine macroinvertebrates are more closely associated with seafloor habitats and have much smaller home ranges. Nevertheless, these species may be affected by large-scale hypoxic events in the northern part of the CCLME.

Seafloor habitats in the CCLME support valuable fisheries, providing food, income and recreation to coastal economies. The Pacific Coast Groundfish fishery management plan of the Pacific Fishery Management Council includes 91 species, including rockfishes, flatfishes, sablefish, lingcod, hake and sharks. Most of these fishes are targeted either commercially or recreationally using bottom and pelagic trawls, traps, bottom-set longlines, and other hook-and-line gears.

Seafloor habitats in the CCLME are subject to several direct and indirect anthropogenic activities. Fishing and pollution are the most widespread stressors in the region, while dredge material disposal, undersea cable laying, mining, and offshore energy development and production have localized impacts. Fishing pressures due to bottom trawling are higher in the northern part of the CCLME. Offshore oil and gas production only occurs in the southern ecoregion, where 26 drilling platforms provide complex artificial habitats for a diverse assemblage of fishes and macroinvertebrates. Off Oregon, several sites are being proposed for offshore wave and wind energy facilities. Finally, in all parts of the CCLME, the impacts of human activities like pollution, ship traffic, and disposal of dredge material diminish with distance from shore or point source.

SELECTING INDICATORS FOR HABITAT

WHAT IS AN INDICATOR?

Indicators are quantitative biological, chemical, physical, social, or economic measurements that serve as proxies of the conditions of attributes of natural and socioeconomic systems (e.g., Landres et al. 1988, Kurtz et al. 2001, EPA 2008a, Fleishman & Murphy 2009). Ecosystem attributes are characteristics that define the structure, composition, and function of the ecosystem. These attributes are typically of scientific or management importance but insufficiently specific or logistically challenging to measure directly (e.g., Landres et al. 1988, Kurtz et al. 2001, EPA 2008, Fleishman & Murphy 2009). Thus, indicators provide a practical means to judge changes in ecosystem attributes related to the achievement of management objectives. They can also be used for predicting ecosystem change and assessing risk.

Indicators are often cast in the Driver-Pressure-State-Impact-Response (DPSIR) framework—an approach that has been broadly applied in environmental assessments of both terrestrial and aquatic ecosystems, including NOAA’s IEA (Levin et al. 2009). Drivers are factors that result in pressures that cause changes in the system. Both natural and anthropogenic forcing factors are considered; an example of the former is climate conditions while the latter include human population size in the coastal zone and associated coastal development, the desire for recreational opportunities, etc. In principle, human driving forces can be assessed and managed. Natural environmental changes cannot be controlled but must be accounted for in management.

Pressures are factors that cause changes in state or condition. They can be mapped to specific drivers. Examples include coastal pollution, habitat loss and degradation, and fishing. Coastal development results in increased coastal armoring and the degradation of associated nearshore habitat. State variables describe the condition of the ecosystem (including physical, chemical, and biotic factors). Impacts comprise measures of the effect of change in these state variables such as loss of biodiversity, declines in productivity and yield, etc. Impacts are measured with respect to management objectives and the risks associated with exceeding or returning to below these targets and limits.

Responses are the actions (regulatory and otherwise) taken in response to predicted impacts. Forcing factors under human control trigger management responses when target values are not met as indicated by risk assessments. Natural drivers may require adaptive responses to minimize risk. For example, changes in climate conditions that in turn affect the basic productivity characteristics of a system may require modifications to ecosystem reference points that reflect the shifting environmental states.

Ideally, indicators should be identified for each step of the DPSIR framework such that the full portfolio of indicators can be used to assess ecosystem condition as well as the processes and mechanisms that drive ecosystem health. State and impact indicators are preferable for identifying the seriousness of an environmental problem, but pressure and response indicators are needed to know how best to control the problem (Niemeijer & de Groot 2008). Indicators can be used as measurement endpoints for examining alternative management scenarios in ecosystem models or in emerging analyses to predict or anticipate regime shifts.

CONCEPTUAL FRAMEWORK FOR INDICATOR SELECTION

Habitat is often the focus of management efforts because natural resources or ecosystem services are generally associated with specific types of habitat (e.g., designations of essential fish habitat or critical habitat). Conservation or restoration efforts for many species are often directed toward habitats needed to support specific life-history stages, making habitat a critical component of ecosystem assessments. At the scale of the California Current, it is a significant challenge to select a suite of indicators that accurately characterize important patterns and processes among the various habitat types while also being relevant to policy concerns. A straightforward approach to overcoming this challenge is to employ a framework that explicitly links indicators to policy goals (Harwell et al. 1999, EPA 2002). This type of framework organizes indicators in logical and meaningful ways in order to assess progress towards policy goals. We use the framework within the rest of the California Current IEA as guidance. Our framework begins with the conceptual models presented above using the set of four major habitat types: freshwater, estuarine/nearshore, pelagic, and seafloor. The key attributes of these habitats are characteristics that specifically describe management-relevant aspects of the habitat. They are characteristic of the health and functioning of each habitat, and they provide a clear and direct link with the indicators. For each habitat type, we identified the same key attributes: habitat quantity, habitat quality, and anthropogenic pressures on quantity and quality.

Habitat quantity. Understanding the distribution and/or abundance of specific types of physical or biogenic habitat is important for management actions. Habitat characteristics are often used to delineate spatial management boundaries that regulate specific activities. For example, rockfish conservation areas (RCAs) designate areas that prohibit bottom trawl fishing, primarily in areas along the continental shelf break that are the main habitat for several overfished rockfish species. Habitat quantity can also be used to describe the upper limits (carrying capacity) of population size or biomass that a system can support. While this idea has been applied to a great extent in freshwater (Reeves et al. 1989) and estuary systems (Beamer et al. submitted) and underlies much of the logic for habitat restoration, it has received less attention in pelagic and seafloor environments even though the concept of carrying capacity has long been accepted in stock assessment and modeling for diverse stocks (Ricker 1954, Beverton & Holt 1957).

Habitat quality. The quality of habitat available has been shown to influence physiology, growth, and behavior of individuals, and these translate into variation in demographic rates of many aquatic organisms. Indicators related to these processes are often important for identifying mechanisms responsible for changes in population size and condition of species-of-interest or changes in ecosystem health.

Anthropogenic pressures. The CCIEA previously developed indicators for a host of anthropogenic pressures, but at the time these did not necessarily include terrestrially based pressures (Andrews et al. 2013). We have updated the indicators of anthropogenic pressures upon habitats in the CCLME to include a wider range of such pressures.

Our goal was to summarize indicators into those supporting two sets of products: spatial analyses and temporal trends. Habitat indicators are often spatially rich but lack long time series, due to the slow pace of change or poor historical monitoring. For these, maps are very important tools even if they are often static and rarely updated. Some habitat indicators are temporally dynamic and amenable to analysis of temporal trends. Examination of trends should be done in the context of the spatial framework such that heterogeneity of habitat state across the California Current Ecosystem is quantified.

EVALUATING POTENTIAL INDICATORS FOR THE CALIFORNIA CURRENT

INITIAL SELECTION OF INDICATORS

The quantity and quality of habitat have been measured in numerous ways throughout the scientific literature. During reviews of the literature, we identified 131 potential indicators of quantity and quality across all four habitat types. Indicators of habitat quantity include the measurement and spatial mapping of various physical and biogenic habitats or population size of algae, corals, sponges and other biogenic habitats. Habitat quality indicators vary widely with measurements of water quality, structural complexity, and food availability.

EVALUATION FRAMEWORK

We follow the evaluation framework established by Kershner et al. (2011) and Levin & Schwing (2011). We divide indicator criteria into three categories: primary considerations, data considerations, and other considerations. Indicators should do more than simply document the decline or recovery of the habitat; they must also provide information that is meaningful to resource managers and policy makers (Orians & Policansky 2009). Because indicators serve as the primary vehicle for communicating habitat status to stakeholders, resource managers, and policy makers, they may be critical to the policy success of EBM efforts, where policy success can be measured by the relevance of laws, regulations, and governance institutions to ecosystem goals (Olsen 2003). Advances in public policy and improvements in management outcomes are most likely if indicators carry significant ecological information and resonate with the public (Levin et al. 2010).

PRIMARY CONSIDERATIONS

Primary considerations are essential criteria that should be fulfilled by an indicator in order for it to provide scientifically useful information about the status of the ecosystem in relation to key attributes of the defined goals. They are:

1. Theoretically sound: Scientific, peer-reviewed findings should demonstrate that indicators can act as reliable surrogates for ecosystem attributes.
2. Relevant to management concerns: Indicators should provide information related to specific management goals and strategies.
3. Predictably responsive and sufficiently sensitive to changes in specific ecosystem attributes: Indicators should respond unambiguously to variation in the ecosystem

attribute(s) they are intended to measure, in a theoretically expected or empirically expected direction.

4. Predictably responsive and sufficiently sensitive to changes in specific management actions or pressures: Management actions or other human-induced pressures should cause detectable changes in the indicators, in a theoretically expected or empirically expected direction, and it should be possible to distinguish the effects of other factors on the response.
5. Linkable to scientifically defined reference points and progress targets: It should be possible to link indicator values to quantitative or qualitative reference points and target reference points, which imply positive progress toward ecosystem goals.

DATA CONSIDERATIONS

Data considerations criteria relate to the actual measurement of the indicator. These criteria are listed separately to highlight ecosystem indicators that meet all or most of the primary considerations, but for which data are currently unavailable. They are:

1. Concrete and numerical: Indicators should be directly measureable. Quantitative measurements are preferred over qualitative, categorical measurements, which in turn are preferred over expert opinions and professional judgments.
2. Historical data or information available: Indicators should be supported by existing data to facilitate current status evaluation (relative to historic levels) and interpretation of future trends.
3. Operationally simple: The methods for sampling, measuring, processing, and analyzing the indicator data should be technically feasible.
4. Broad spatial coverage: Ideally, data for each indicator should be available across a broad range of the California Current.
5. Continuous time series: Indicators should have been sampled on multiple occasions, preferably without substantial time gaps between sampling.
6. Spatial and temporal variation understood: Diel, seasonal, annual, and decadal variability in the indicators should ideally be understood, as should spatial heterogeneity and patchiness in indicator values.
7. High signal-to-noise ratio: It should be possible to estimate measurement and process uncertainty associated with each indicator, and to ensure that variability in indicator values does not prevent detection of significant changes.

OTHER CONSIDERATIONS

Other considerations criteria may be important but not essential for indicator performance. Other considerations are meant to incorporate nonscientific information into the indicator evaluation process. They are:

1. Understood by the public and policy makers: Indicators should be simple to interpret, easy to communicate, and public understanding should be consistent with technical definitions.
2. Historically reported: Indicators already perceived by the public and policy makers as reliable and meaningful should be preferred over novel indicators.
3. Cost-effective: Sampling, measuring, processing, and analyzing the indicator data should make effective use of limited financial resources.
4. Anticipatory or leading indicator: A subset of indicators should signal changes in ecosystem attributes before they occur, and ideally with sufficient lead-time to allow for a management response.
5. Lagging indicator: Reveals evidence of a failure in or to the attribute.
6. Regionally, nationally, and internationally compatible: Indicators should be comparable to those used in other geographic locations, in order to contextualize ecosystem status and changes in status.

SCORING INDICATORS

Each indicator was evaluated independently according to these 18 evaluation criteria by reviewing peer-reviewed publications and reports. The result was a matrix of indicators and criteria that contained specific references and notes in each cell, which summarized the literature support for each indicator against the criteria. This matrix can be easily reevaluated and updated as new information becomes available. The matrix of habitat indicators and indicator evaluation criteria provided the basis for scoring the relative support in the literature for each indicator (Kershner et al. 2011, Levin & Schwing 2011). For each cell in the evaluation matrix, we assigned a literature-support value of 1.0, 0.5, or 0.0 depending on whether there was support in the literature for the indicator, whether the literature was ambiguous, or whether there was no support in the literature for the indicator, respectively.

However, scoring indicators also requires careful consideration of the relative importance of evaluation criteria. The importance of the criteria will certainly vary depending on the context within which the indicators are used and the people using them.

Thus, scoring requires that managers and scientists work together to weight criteria. Failure to weight criteria is, of course, a decision to weight all criteria equally.

To determine the weightings for each of the evaluation criteria, we used weightings calculated in Levin & Schwing (2011). Briefly, the weightings were calculated by asking 15 regional resource managers, policy analysts, and scientists to rate how important each of the evaluation criteria was to them on a scale of 0 to 1. This provided an average weighting for each criterion. Each criterion was then assigned to the quartile into which its average weighting fell (1st = 0.25, 2nd = 0.50, 3rd = 0.75 or 4th = 1.0) and this was used as the weighting for each criterion.

For each cell, the literature-support value was multiplied by the weighting for the respective criterion and then summed across each indicator to yield the final score for each indicator. For each key attribute of each EBM component, we then calculated the quartiles for the distribution of scores for each indicator. Indicators that scored in the top quartile (top 25%) for each attribute of each habitat type were considered to have good support in the literature as an indicator of the attribute they were evaluated against. We describe below the results of the evaluation for each indicator that scored in the top quartile.

RESULTS OF INDICATOR EVALUATIONS

The results of our evaluation of each indicator are summarized in the tables in this section. Following the framework outlined above, we organized the results of the evaluation by habitat type. The sum-of-scores within each criteria grouping (e.g., Primary considerations) in the tables are provided along with a brief summary of why the indicator is important and how it was evaluated. Indicators that ranked in the top quartile for each key attribute of each habitat type are described in more detail in the following sections. Potential indicators that scored poorly and are unlikely to be used in the IEA are listed in the tables but not discussed further in the text.

EVALUATION OF FRESHWATER HABITAT INDICATORS

HABITAT QUANTITY

We identified nine indicators of freshwater habitat quantity (Table H1). Given the long history of studies of freshwater systems and salmonid habitats, most potential metrics had a good scientific basis. However, many potential indicators suffered data limitations due to poor sampling over time and across systems, and due to lack of historical reference points. We highlight three indicators for which these spatial or temporal challenges can largely be overcome.

Discharge: spatial and temporal patterns. Discharge (or streamflow) is considered a "master variable" in riverine ecosystems, meaning that it strongly influences many habitat attributes of rivers (e.g., temperature, channel morphology, habitat diversity) and ultimately limits the distribution and abundance of riverine species (Poff et al. 2007). There are five key discharge attributes that comprise the streamflow regime: magnitude, frequency, duration, timing, and rate of change (Poff et al. 2007). Each of these attributes can be quantified from long-term discharge records, with a range of individual metrics that can be calculated for each attribute. One such set of metrics is the Indicators of Hydrologic Alteration (IHA; Richter et al. 1996). These metrics were selected for characterizing the influence of dams on the flow regime, and for identifying how those flow changes influence physical and biological processes and the health of biota downstream of dams (Poff et al. 2007, Poff et al. 2010). The IHA includes 33 individual flow metrics, 24 of which are measures of flow magnitude for high and low flows of various durations (e.g., 1-day average, 7-day average, monthly, annual), and 2-3 metrics each for flow timing, frequency, duration, and rate of change Richter et al. 1996). We note that at least 15 years of data are required for reasonable characterization of the flow regime; describing changes in the flow regime due to a dam therefore requires 15 years of data both before and after dam construction. In this report we do not select specific indicators, as the relevant indicators depend upon the species and pressures evaluated.

The primary data source for these discharge records is the set of long-term USGS gage records. Daily discharge data for all 26,000 active and inactive gages in the US are available at http://waterdata.usgs.gov/nwis/sw.

Available habitat length (% of historical stream habitat that is currently accessible). Available habitat length or area is one of the most important habitat variables controlling population sizes of anadromous fishes (e.g., Reeves et al. 1989, Beechie et al. 1994, Sharma & Hilborn 2001). Several studies indicate that more habitat generally produces more fish, and this general relationship holds across species and spatial extents (e.g., Beechie et al. 1994, Kim and LaPointe 2010). However, the type and arrangement of habitats also influences population size (Sharma & Hilborn 2001, Kim & LaPointe 2010). The simplest indicator of habitat quantity is the length of stream or river accessible to anadromous fish relative to the amount that was historically available. This indicates whether the overall habitat capacity is significantly reduced from its natural potential. The most difficult component of this metric to measure is the historical habitat length that has been blocked by dams, although there are methods available to do so. Where dams are large and large areas of habitat are blocked, existing maps can provide a reasonable measure of blocked habitat. However, smaller barriers are often not mapped, and the cumulative length of blocked habitat on small streams and rivers is generally not available.

Hence, this metric may currently be most useful in large river basins such as the Columbia River basin or the Central Valley in California, where the majority of blocked habitat is in relatively large rivers above major dams. In smaller basins with only small dams or with habitat blocked mainly by other structures such as culverts, this metric is less likely to be useful except where full barrier inventories have been completed.

Fish distribution data downstream of major barriers can be found on StreamNet (for Washington, Oregon, Idaho; https://www.streamnet.org/mapping_apps.cfm). Data on barriers that have been removed or modified to allow passage since the inception of the Pacific Coastal Salmon Recovery Fund can be found in the PCSRF restoration project database (https://www.webapps.nwfsc.noaa.gov/apex/f?p=309:13:). State salmon passage barrier databases are:

Washington: http://geography.wa.gov/GeospatialPortal/dataDownload.shtml

Oregon: https://nrimp.dfw.state.or.us/Oregonplan/default.aspx?p=134&XMLname=44.xml

California:
http://www.calfish.org/Programs/ProgramIndex/CaliforniaFishPassageAssessmentDatabase/tabid/189/Default.aspx

Area of disconnected floodplain (% of floodplain habitat accessible vs historical habitat). Just as stream length is an important control on population size of anadromous fishes, the availability of floodplain habitats strongly influences habitat capacity and population size (e.g., Beechie et al. 1994, Burnett et al. 2007). Floodplains are dynamic environments that produce abundant and diverse habitats for salmon and other aquatic organisms (Ward et al. 2002, Beechie et al. 2001, Beechie & Imaki 2014), as well as for riparian species diversity (Beechie et al. 2006, Naiman et al. 2010). Not surprisingly, human occupation of floodplains was one of the earliest impacts on salmon habitat (Beechie et al. 1994, Beechie et al. 2001), and those impacts are wide-spread in the region (Fullerton et al 2006, Hall et al. 2007). The main effect of human occupation was the separation of rivers from their floodplains by levees, and the elimination of floodplain channels and ponds that functioned as salmon spawning and rearing habitats (Beechie et al. 1994, Beechie et al. 2001). In much of the CCLME drainage area, floodplain habitats have been virtually eliminated, and loss of floodplain habitats is by far the largest habitat influence on salmon population declines. Hence, restoration of those habitats is a critical conservation need for successful recovery of important species (Hall et al. 2007, Beechie & Imaki 2014)

There are no readily available geospatial datasets that quantify this metric, although the necessary datasets and techniques exist. Floodplains can be mapped from existing topographic datasets (including readily available 10-m DEMs and locally available LiDAR) (Hall et al. 2007, Beechie & Imaki 2014, Nagel et al 2014). The 10-m data are likely not of high enough resolution to provide accurate data to quantify the proportion of floodplains disconnected from rivers region-wide, but are sufficient for identifying extent of historical floodplains (Beechie & Imaki 2014). The human impact on connectivity will require LIDAR data, which are increasingly available and can be used to identify patches of floodplain that are disconnected by roads and levees (e.g., Konrad unpublished data). Other satellite-based methods of identifying wetted area of the landscape may eventually be useful, though they currently appear to be incapable of measuring connectivity directly (e.g., Watts et al. 2012).

HABITAT QUALITY

We identified nine indicators of freshwater habitat quality (Table H1). Like freshwater quantity, some of these indicators have benefited from a long history of freshwater studies. However, many indicators lacked ability to directly link the metric to habitat condition, or sufficiently understand variation. Habitat quality metrics also suffered from large spatiotemporal gaps and public support for monitoring. We identified one indicator that could overcome these challenges:

Water temperature: temporal and spatial patterns. Temperature controls the rates of many biological processes, and is a key driver of ecological processes controlling population and community structure in aquatic ecosystems (Allan and Castillo 2007; Webb et al. 2008). For ectotherms, water temperature regulates rates of physiological, neurological, embryological and behavioral development (Brett 1971; Ficke et al. 2007). Water temperature can alter behavior (e.g., rates of movement), and through its influence on metabolic efficiency and growth, water temperature drives the timing of ontogenetic transitions from one life stage to the next (Ward & Stanford 1979; Beacham & Murray 1990). Pacific salmon are likely influenced by both spatial and temporal patterns in altered thermal regimes (McCullough et al. 2009). At broad spatial scales, stream temperature may define species distributions. At finer scales, stream temperature can define connectivity among habitats used during different life stages (e.g., foraging, breeding) (Schlosser 1995; Armstrong et al. 2013). Extreme temperatures may create barriers to movement but spatial heterogeneity in water temperature provides pockets of refuge from unfavorable temperatures (Poole & Berman 2001; Torgersen et al. 1999). Salmon may also be influenced by temporal variability in water temperature. Angilletta et al. (2008) and Crozier et al. (2008) suggested that alterations in the magnitude and timing of stream thermal regimes could induce mismatches between evolved life-histories and current environmental conditions that may reduce survival and fitness. Emergence by Chinook salmon alevins in a laboratory was delayed when fish experienced thermal regimes with extreme daily or seasonal variation (Steel et al. 2012).

Human activities such as operation of hydropower dams, development of land adjacent to streams, and water withdrawal for irrigation alter stream thermal regimes. For many regulated rivers, water below large dams is warmer in winter and cooler in summer than in unregulated rivers, and the amplitude of variation in water temperature at finer resolutions (hourly, daily, weekly) is dampened (Steel & Lange 2007; Olden & Naiman 2010). Land conversion and water withdrawal alter stream thermal regimes by increasing the amount of surface area exposed to solar radiation (e.g., from reduced forested buffers or stream widening) or by altering the hydrologic cycle, and therefore the rate of water exchange, both overland and via groundwater recharge (Bisson et al. 2009). Each of these impacts can intensify temporal trends in water temperature causing streams to be warmer during hot periods and cooler during cool periods. Anthropogenic climate change is predicted to alter thermal regimes in streams by increasing water temperatures and decreasing summer flows (Mote et al. 2003; IPCC 2007). Coupled with natural variability and uncertainty in projections, stream temperatures may increasingly stress stream organisms (Ficke et al. 2007). In the Pacific Northwest, summer stream temperatures are expected to reach or exceed thermal tolerances for salmonids by as early as the 2020s (Mantua et al. 2010; Isaak et al. 2013). Specific metrics for temperature are not identified here, but all rely on availability of continuously recorded data. Examples of specific metrics that may be useful in include number of days or weeks above some specified threshold (Mantua et al. 2010), or maximum weekly maximum temperature (Isaak et al. 2010).

Data are increasingly available to characterize both temporal and spatial patterns in water temperature at a variety of scales. Isaak et al. (2011) are compiling temporal data collected at many point locations across the region by various entities. They have developed a geostatistical model that uses these data to make spatially continuous predictions of water temperature for both current and future scenarios (Peterson et al. 2013). Predictions can be made for different time periods (e.g., seasons), but data collected during summers are most abundant. Spatially continuous data for maximum summer stream temperature are also available for hundreds of large rivers throughout the Pacific Northwest collected using remotely sensed airborne thermal infrared (TIR) sensing (R. Faux, Watershed Sciences Inc., pers. comm.; Handcock et al. 2012).

PRESSURES

We identified 14 indicators of anthropogenic pressures (Table H1). Many of these were summarized in the anthropogenic pressures report for the CCIEA (Andrews et al.

2013). Most pressure indicators suffer from insufficient reporting and spatial and temporal gaps, which was one reason Andrews et al. (2013) used population density as the main indicator for tracking anthropogenic pressures. We identified three indicators that could be synthesized either for mapping or for tracking trends:

Number of dams. Dams can block access of fish to upstream habitats or alter flow, sediment, and temperature regimes downstream (e.g., Sheer & Steel 2006, Poff et al. 2007). Therefore, this pressure is related to the quantity indicators of discharge and percent of historical habitat length accessible to anadromous fish, as well as the quality indicator of temperature. The primary dataset available is the National Inventory of Dams maintained by the US Army Corps of Engineers (http://geo.usace.army.mil/pgis/f?p=397:1:0). One problem with this database is that it does not consistently identify whether the dam is a passage barrier, and for which species (e.g., some dams are passable to salmon via fish ladders, but are still barriers to lamprey migration). Nonetheless, even in the near term the number of dams is a coarse indicator of human pressures on riverine habitats. It may also be possible to identify whether each dam in the database is passable or not, thereby restricting the dataset to more accurately reflect dams that affect upstream passage (for the 'length of habitat' indicator). The full dataset is more likely to be appropriate as an indicator of pressure on discharge and temperature regimes downstream of dams.

Riparian vegetation. Riparian (streamside) vegetation provides many benefits to freshwater biota, both directly and indirectly. Key functions include root reinforcement of banks, stream shading, sediment retention, nutrient supply, and large wood supply to stream, lake, and wetland ecosystems (Beechie et al. 2013). Nutrients and terrestrial invertebrates can enhance aquatic food webs, and large wood provides cover and helps to form habitat units such as pools and backwater areas. Functioning riparian areas also allow interactions between a water body and its floodplain, and access by fish to important off-channel rearing habitats (Ward et al. 2002, Naiman et al. 2010). Indirect benefits include contributions to water quality and maintenance of hydrologic processes (Beechie et al. 2013). Riparian vegetation can influence water temperature via shading (reduced solar input blocked by leaves) (Moore et al. 2005). Vegetation can filter excess sediment, nutrients, and pollutants and provide stable banks (via networks of roots) that prevent large inputs of sediment. Intact riparian areas also help to control the timing and amount of runoff from precipitation events and the maintenance of adequate water table heights.

Types of native vegetation differ among regions; areas with wetter climates tend to have dense forests often with thick understory, and arid regions tend to have sparser tree cover with mostly brush and grasses underneath. The types of vegetation present (both native and nonindigenous) and alterations to riparian characteristics can decrease riparian function. Removal or reduction in coverage of riparian vegetation can lead to stream widening, which may decrease shading and increase water temperature. Wood recruitment is also reduced, which may alter stream morphology (i.e., simplifying channel characteristics and reducing habitat diversity). Bank hardening associated with developed areas (rip rap, levees) and water withdrawal or diversion may lead to stream widening or incision which reduces access by fish to off-channel habitats. Primary mechanisms of altered riparian characteristics (reduction in vegetation cover or increased representation of nonindigenous species) are anthropogenic development (e.g., development, agriculture, and road building), forestry (timber harvest, road building), grazing, and mining. Climate change may alter the type of vegetation capable of growing in a region. Beavers are also agents of riparian change, and can be used to restore riparian processes along with active riparian planting or other restoration techniques.

Quantification of changes in vegetation can be accomplished by comparing land use and land cover from satellite and aerial imagery at different spatial resolutions (Fullerton et al. 2006). Data are not summarized very frequently but there are applications available for tracking change over time, such as LandTrendr. In certain locations (e.g., west of the Cascades), vegetation has been summarized at a finer resolution and includes variables such as tree size, percent cover, and type (conifer vs. deciduous) that may be more relevant for monitoring riparian areas (i.e., the Interagency Vegetation Mapping Project).

% Urban land cover. Urban land cover has been associated with degraded habitat quantity and quality for salmon and other aquatic organisms (Booth & Jackson 1997, Booth et al. 2002, Morley & Karr 2002, Pess et al. 2002). Urbanization results in increased impervious surface area, causing increased runoff and peak flows, runoff of metals and pesticides from roads and landscaping, reduced riparian functions via vegetation removal, and in some cases channel modifications that simplify or eliminate habitat (Beechie et al. 2013). Increased runoff and flood magnitudes due to impervious surfaces (mainly pavement and rooftops) in some areas cause channel incision (Booth & Jackson 1997, Booth et al. 2002). Runoff of pesticides and metals degrades ecosystem health and can also cause pre-spawning mortality of some salmon species (e.g., Morley & Karr 2002, Scholz et al. 2011). Percent developed land cover has also been correlated directly with coho salmon population sizes (Pess et al. 2002). Therefore, trends in developed land cover are potentially a useful indicator of large-scale trends in habitat quality.

A variety of land cover classifications, land cover, and land use data are available for the USA, and can be used to assess the proportion of land in urban areas (e.g., the Human Footprint, and NOAA's C-CAP land cover dataset). NOAA's C-CAP data is available for Pacific Coast, and spans 1985-2011 (https://www.csc.noaa.gov/digitalcoast/data/ccapregional).

Table H 1 Summary of evaluations of potential freshwater indicator across five primary considerations, seven data considerations, and six other criteria. Each criterion was scored 0, 0.5, or 1 depending on the level of literature support for that criterion. The numerical value that appears under each of the criteria groupings represents the sum of those values. For example, river discharge has peer-reviewed literature strongly supporting five out of five primary considerations criteria. *Indicators in the top quartile; **Promising indicators with gaps; unmarked indicators scored poorly and will not be considered further.

Indicator	Primary (5)	Data (7)	Other (6)	Summary comments
Quantity				
River discharge (e.g., 1-day average peak flow, 7-day average low flow)*	5	6	3	Good indicator of change in ecologically important flows (e.g., low flows, flood flows). Spatial and temporal patterns of river discharge well recorded with long-term USGS gaging stations on large rivers. Less consistent coverage of smaller streams.
% of network accessible vs historical*	5	6.5	5.5	Good indicator of habitat availability at large scales (e.g., amount of habitat blocked by dams). Time series can be reconstructed but will require some additional effort. Less useful for migration barriers on small streams because field inventories are rarely complete or consistent across states or watersheds.
% of floodplain vs historical**	5	4	4.5	Good indicator of habitat availability at reach scales (e.g., amount of habitat removed by levees and floodplain modification). Time series can be reconstructed but will require some additional effort. Where LIDAR is available this can be modeled in GIS, but accuracy needs to be evaluated.
Wood counts at index sites	5	3.5	3	Wood counts are inconsistently recorded both spatially and temporally. Historical records are rare and there are few monitoring programs in place to inventory wood in streams.
Spawning gravel availability	2.5	2.5	1.5	Spawning gravel is rarely measured, and inconsistently recorded both spatially and temporally. Criteria vary by species.
Intrinsic potential	3.5	3	2	Indirect indicator and not sensitive to land use or other impacts to habitat.
Node density (Whited et al.)	5	4	5	Indicator that can be linked to habitat availability but would need additional work to be a good indicator of change over time. Remote sensing methods are available to measure this on very large rivers (e.g., Yukon), but it has been well tested in smaller rivers such as those in the CC.
% watershed restored	2.5	4	2	Very little data available, and unlikely that this could be made into a useful metric even with considerable effort and expense.
Wetland area	4	5	4.5	Some data available but accuracy is low and time series not available.

Quality				
Riparian or floodplain condition*	5	4	4	Good indicator of habitat quality at reach scales, but time series must be reconstructed and will require some additional effort. There are no automated remote sensing methods developed yet; Land cover data from LandSAT are available but coarse resolution (30-30m cells). Finer resolution data not yet able to characterize riparian types.
Temperature **	4.5	5	4	Poorer time series data, but a good indicator where available. Time series available at selected sites, but not widespread.
Upland condition	2	3	2.5	Difficult to link directly to habitat condition in many cases. The indirect is indirect and mechanistic links to habitat vary widely; the meaning of the indicator is not clear except that more human influence generally means lower quality habitat.
IBI/BIBI scores	5	3	2.5	Good indicator of ecosystem health but poor spatial and temporal coverage.
303d lists	3	2	4	These are opportunistic designations of poor water quality under the Clean Water Act, but reaches are inconsistently identified where specific perceived problems are documented. Not a systematic dataset, so not a good indicator of water quality either temporally or spatially.
Grain size/fine sediment	4.5	2	3.5	In some cases this may be a good indicator of ecosystem health (not always); these are field measurements that have poor spatial and temporal coverage and methods vary among locations and studies.
Salmon production per km	2.5	2	1.5	This may be a good indicator of ecosystem health where hatchery and harvest influences are low, but is influenced by many factors besides habitat.
# listed species	1.5	3.5	2.5	Potentially a good indicator of ecosystem health, but may be influenced by factors other than habitat.
Predators (e.g., birds, fish)	4	0.5	3	Inconsistent data coverage; more predators not necessarily negative in natural settings.
Pressures				
% developed/impervious*	4	6	4	Good indicator of pressure influencing hydrologic effects on small streams in urban areas. Also related to poor water quality as measured by multi-metric biological indicators. Generally not a wide-spread habitat problem, but locally important.
% agriculture*	3.5	6	4	Good indicator of pressure influencing sediment effects on habitat may also be related to water quality and habitat quantity changes, but specific mechanisms not well documented.

# of dams (length of habitat blocked)*	5	6.5	5.5	Good indicator of pressure that drives length of habitat accessible. Databases exist for medium to large dams.
Riparian veg condition**	5	4	4	Good indicator of pressure on stream habitat condition and stream temperature. There are no automated remote sensing methods developed yet; Land cover data from LandSAT are available but coarse resolution (30-30m cells). Finer resolution data not yet able to characterize riparian types.
Erosion rate (function of land use)	2.5	6	3	See land use pressures below (%ag land cover and forest road density). Mechanistic linkages are known but datasets are sometimes inaccurately represented in existing data (e.g., forest road data are of poor quality).
# of flow diversions	3.5	2	1.5	Indicator of pressure on stream flow, but poor spatial and temporal data availability.
% high Intrinsic potential modified by land use	4	3.5	2	Indirect indicator of pressure on habitat condition; indicator will change with land cover change.
Levees/dikes (% of bank)	4.5	3.5	3.5	Good indicator of pressure on stream habitat condition, but poor temporal and spatial coverage.
Shoreline armoring (% of bank)	4.5	3.5	3.5	Good indicator of pressure on stream habitat condition, but poor temporal and spatial coverage.
Human foot print metrics	4	5	4	Indirect indicators of pressure on habitat condition, but poor temporal coverage.
Forest road density	5	5	4.5	Data quality is inconsistent spatially and temporally, but important indicator of erosion pressure.
Point sources of pollution	3.5	4	3	Good indicator of pressure on stream habitat condition, but poor temporal and spatial coverage.
Nonpoint sources of pollution	3.5	3.5	2.5	Good indicator of pressure on stream habitat condition, but poor temporal and spatial coverage.
Non-indigenous species	4.5	0.5	2	Indirect indicator of pressure on habitat condition, but poor temporal coverage.

EVALUATION OF ESTUARINE/NEARSHORE HABITAT INDICATORS

HABITAT QUANTITY

We evaluated 16 indicators of estuary and nearshore habitat quantity (Table H2). These indicators were related to the quantification of physical and biogenic habitat types. They include indices of the areal extent of selected habitat types (e.g., areal inundated wetland coverage); indicators of energy flow and transport processes (e.g., sediment deposition); and landscape-scale metrics of drainage-basin and habitat characteristics (e.g., estuary surface area: drainage area). Among the highest ranked indicators were habitat metrics for which satellite data are available to map annual changes in extent and distribution over large areas (e.g., areal extent of macrophytes). Salinity and other physical measurements (e.g., isohaline position) also hold some promise because such metrics are routinely monitored in many systems. Landscape-scale metrics depicting the relative size or complexity of river basins, estuaries, and nearshore environments were deemed relatively insensitive to interannual variations and ranked relatively low in our assessment. We identified eight promising indicators of estuarine and nearshore habitat quantity.

River Flow. Alteration of hydrology in estuaries resulting from changes to freshwater flow can have substantial effects on tidal inundation, material delivery, degree of mixing between salt and freshwater, water residence time, and temperature. Such changes can lead to impacts to water quality, reduced connectivity, and marsh subsidence (MBNEP 2002a,b). Rivers supply an estuary with freshwater, sediment, and other materials, all of which are important for the continued functioning of estuarine processes. Like freshwater habitats, discharge (or streamflow) therefore strongly influences many habitat attributes of rivers (e.g., temperature, channel morphology, habitat diversity) and limits the distribution and abundance of estuarine species. As noted in the Freshwater indicators, the magnitude, frequency, duration, timing, and rate of change (Poff et al. 2007) of flow metrics can be quantified from long-term discharge records, and one such set of metrics that have been commonly used is the Indicators of Hydrologic Alteration (IHA; Richter et al. 1996). These metrics were developed for characterizing the influence of dams on the flow regime, and for identifying how those flow changes influence physical and biological processes and the health of biota downstream of dams (Poff et al. 2007, Poff et al. 2010). Recently, IHA metrics were used to examine impacts to estuaries on all coasts of the contiguous United States (Greene et al. in press). This analysis utilized long time series of flow records at USGS gages (http://waterdata.usgs.gov/nwis/sw) upstream of estuaries. Numerous gaps were observed across the US, although the Pacific Coast had relatively fewer gaps than other coasts.

Areal inundated wetland coverage. Tidal wetlands in estuaries provide a variety of ecosystem services, including flood and erosion control, water purification, energy production and nutrient cycling, and cover and structure for a diversity of species (Barbier et al. 2011; Zedler and Kercher 2005; Visintainer et al. 2006). Tidal wetlands produce large quantities of organic matter and prey that can be exported far from local production sites to the larger ecosystem (Ramirez 2008; Eaton 2010). For example, wetland vascular plants are a primary source supporting the estuarine food webs of juvenile salmon, conveyed through production of insect and other prey taxa (Gray et al. 2002; Maier & Simenstad 2009). Various salmon-performance metrics have been linked directly to the extent of available estuarine wetlands, including salmon rearing capacity (Greene and Beamer 2012; Beamer et al. 2013), survival (Magnusson & Hilborn 2003), life-history diversity (Bottom et al. 2005; Jones et al. 2014), and adult returns (Jones et al. 2014).

Coarse scale mapping of land cover and wetland classes is available across the United States from the National Wetland Inventory (http://www.fws.gov/wetlands/Data/) and the national land cover database (http://www.mrlc.gov/). Satellite imagery provides data to quantify annual changes in herbaceous and woody wetland cover classes (http://www.mrlc.gov/; http://www.csc.noaa.gov/digitalcoast/data/ccapregional). Historical land and hydrological survey data also have been analyzed for selected Pacific Coast estuaries, establishing a baseline for long-term changes in the composition and distribution of wetland habitat types (e.g., Collins & Sheik 2005; Marcoe & Pilson 2013).

Historical wetland losses have been substantial in most Pacific Coast estuaries (Good 2000; Collins & Sheik 2005; Cereghino et al. 2012), and wetland restoration is now recognized as a priority of various ecosystem and salmon recovery strategies (e.g., Cereghino et al. 2012; Thom et al. 2013). Thus, the areal extent of wetland habitat is not only a useful indicator of habitat quality or the potential nursery function of estuaries. It is also a useful benchmark for measuring the progress of ecosystem restoration and salmon recovery efforts.

Area of salinity zones. The composition and distribution of fish, invertebrate, and plant assemblages in estuaries have been linked to variations in salinity distribution as determined by interactions between tides and river flow in each basin (Allen 1982; Bottom & Jones 1990; Emmett et al. 1991). Salinity tolerances vary among species, and the estuary distributions of sessile plants and invertebrates may be constrained by salinity (e.g., Kentula & DeWitt 2003). In contrast, nektonic species can adjust to the physical environment, and the horizontal distribution and composition of estuarine fish assemblages has been linked to seasonal fluctuations in the salinity gradient (Allen 1982; Bottom et al. 1984; Bottom & Jones 1990). The mean areal extent of salinity zones within an estuary (e.g., oligohaline, mesohaline, euryhaline) could be a useful indicator of habitat quantity based on the tolerances or preferences of individual species and assemblages.

Salinity is recorded in many estuaries, although existing monitoring programs do not routinely report the average areal extent of particular estuary salinity zones. Several studies have reported fish assemblage or species distributions relative to broad salinity ranges (Bottom et al. 1984, Bottom & Jones 1990; Beamer et al. 2007) but different salinity classes have been chosen for these comparisons. NOAA uses a digital geographic information system (GIS) to report average annual salinity of US estuaries for three broad salinity classes: Tidal Fresh (0 - 0.5 parts per thousand), Mixing Zone (0.5 - 25 parts per thousand), and Seawater Zone (25 parts per thousand or greater; see http://catalog.data.gov/harvest/object/8ff3b448-7128-4414-a5d4-7268df7ba140/html). These same zones were used by Monaco et al. (1990) and Emmett et al. (1991) to organize general species distribution data for each estuary in the NEI Data Atlas (NOAA 1985).

Salinity is a common parameter of many ongoing estuary monitoring programs and is readily understood by the public. Yet in some estuaries short-term (i.e., tidal) salinity variations may equal or exceed seasonal fluctuations, complicating efforts to distinguish anthropogenic effects from natural variations. Salinity could be a useful indicator of estuary response to future climate changes that will likely alter hydrology, increase sea level, and thereby modify estuary circulation and salinity patterns. However, such changes also could shift fundamental relationships of species to salinity indicators (Cloern & Jassby 2012). While salinity zones may be a promising indicator, additional analysis may be needed to define the appropriate zones, estimate their areas in Pacific Coast estuaries, and evaluate their sensitivity as an indicator of ecosystem change.

Isohaline position. Some studies have used the near-bottom isohaline position as an indicator of potential physical and biotic responses to changes in salinity intrusion. In San Francisco Bay the 2 parts per thousand salinity isohaline has been related to annual measures for a variety of variables including phytoplankton supply, benthic macroinvertebrates, mysids and shrimp, larval fishes, and fish abundance (Jassby et al. 1995; Dege & Brown 2004). The position of the low-salinity isohaline has been widely used in San Francisco Bay as an indicator of the effects of flow variation and water withdrawals. Linkages between water diversion and native fish mortality, including imperiled species such as longfin and delta smelt, are now recognized in management policies for the San Francisco Bay-Delta (Cloern & Jassby 2012).

Jassby et al (1995) note that “the 2‰ value may not have special ecological significance for other estuaries…but the concept of using near-bottom isohaline position as a habitat indicator should be widely applicable.” For example, salinity intrusion length similarly has been used as an indicator of effects of flow regulation in the Columbia River estuary (Jay & Naik 2011).

The data needed to depict distribution of a particular salinity isohaline are available from a variety of monitoring programs but isohaline position is not now routinely reported for many Pacific Coast estuaries. Salinity monitoring data at multiple scales are available for Puget Sound (Moore et al. 2008a,b) and the Columbia River estuary (http://www.stccmop.org/datamart/virtualcolumbiariver). The data are less consistent for small coastal estuaries, although some long-term salinity records are available from the Oregon Department of Environmental Quality (http://deq12.deq.state.or.us/lasar2/ ; see Lee and Brown 2009).

As noted for salinity-zone metrics, isohaline position is a promising indicator of potential hydrological and biotic responses to climate or other changes. The indicator is straightforward and readily understood, and data needed to define isohaline position are available for many areas. However, further analyses may be needed to determine the isohaline value(s) that are most biologically relevant and to estimate isohaline position in each estuary.

Areal extent of physical habitat. Area of physical habitat did not evaluate in the top quartile primarily due to limitations in the amount of historical data available but did evaluate highly in four out of five primary considerations criteria. Moreover, area of physical habitat (e.g., rocky intertidal, sandy beaches) is an obvious indicator to evaluate the status and trends of quantity of nearshore habitat. Physical habitat is relevant to management concerns as boundaries of various habitat types have often been used to delineate management actions such as spatial closures or regulations associated with shoreline modification. Physical habitat on land can be quantified using remote sensing, although this will require lots of processing time to calculate across the CCLME. Subtidal physical habitat can be measured with multi-beam sonar surveys, but typically these surveys occur in offshore habitats where large ships can operate. Small-boat based surveys could map nearshore habitats, but funding to map these areas along the Pacific Coast has been an obstacle. The amount of physical habitat is easily understood by the public and often used by policymakers.

Areal extent of macrophytes. In the California Current ecosystem, the two most important submerged macrophytes are eelgrass and kelp. Eelgrass is an important structural component of subtidal and intertidal communities in shallow coastal bays, estuaries, and semi-protected soft-bottom areas of the open coast (Bernstein et al. 2011). Native eelgrass provides habitat for young-of-the-year Dungeness crab (McMillan et al. 1995), produces epibenthic prey species favored by juvenile chum salmon (Fresh 2006), and serves as spawning substrate for Pacific herring (Plummer et al. 2012). Eelgrass beds also can provide key rearing habitats for coastal cutthroat trout (Krentz 2007) and juvenile coho and Chinook salmon (Bottom et al. 2005, Jones et al 2014). Subtidal eelgrass beds adjacent to intertidal flats offer complex low-tide refugia that may support higher fish densities than other non-vegetated channels (Bottom et al. 1988).

Areal extent of eelgrass is commonly monitored in many estuaries and coastal bays as an indicator of ecosystem condition and change. Underwater surveys have been useful in deeper habitats that are not well represented in photo imagery. For example, underwater videography has been used in Puget Sound to estimate changes in areal cover of eelgrass beds at site and regional scales (Norris et al. 1997; Gaeckle et al. 2008). Aerial photography and digital mapping (GIS) have been used successfully to quantify coarse-scale changes in eelgrass coverage (Short and Burdick 1996; Robbins 1997). A combination of side-scan sonar and aerial imagery is now widely used for system-wide surveys conducted in southern California (Morro Bay in the north to Tijuana Estuary) (Bernstein et al. 2011). Use of satellite imagery should reduce future field sampling of eelgrass extent and allow for regional and national comparisons.

Historical eelgrass data are available for selected regions, including southern California Bays (Bernstein et al. 2011). Coarse resolution habitat maps produced for coastal planners in the late 1970s may provide a satisfactory baseline for monitoring changes in eelgrass extent among 16 Oregon estuaries (e.g., Bottom et al. 1979; maps available online: http://www.coastalatlas.net/index.php/tools/planners/63-estuary-data-viewer). Eelgrass monitoring data for Puget Sound is also available since 2000 (Gaeckle et al. 2009). Eelgrass extent has wide application to estuary management programs including its designation as Essential Fish Habitat (Sustainable Fisheries Act) and a Habitat Area of Particular Concern (HAPC); as an ecosystem indicator for measuring the progress of Puget Sound restoration (Puget Sound Vital Signs, http://www.psp.wa.gov/vitalsigns/); and in mitigation policies enacted in California (Southern California Eelgrass Mitigation Plan) and Oregon (Oregon Administrative Rules governing removal-fill authorizations, e.g., http://arcweb.sos.state.or.us/pages/rules/oars_100/oar_141/141_085.html).

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

1999) ecosystems. Most kelp forests exist in waters less than 60 m deep, but because of its importance as essential fish habitat for many species of concern, including young-of-year (Carr 1991), understanding the temporal variation and spatial heterogeneity (Jones 1992, Bustamante & Branch 1996) of kelp forest coverage in the CCLME should be a useful indicator of the quantity of important nearshore habitat. Following the framework of Link (2005), reference points related to percent change in areal coverage of canopy-forming kelp could be established.

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

Extent of kelp coverage along the coastline is easily understood by the public and has been used by policy makers to develop guidelines related to provisions of the Magnuson Stevens Act to identify essential fish habitat (16 U. S. C. §1855b). Changes in the extent of kelp cover affects recruitment of invertebrates and other species (e.g., Carr 1991), such that kelp coverage could anticipate recruitment of older life stages into offshore populations and into various fisheries; thus kelp coverage may not only be a good indicator for the quantity of nearshore habitat, but could also be a leading indicator for community-level attributes of the CCLME.

Macrophyte density. Whereas areal extent of macrophytes measures their exterior boundary across a large area, eelgrass density provides an index of the relative condition of eelgrass or kelp within a bed. Two types of condition indicators often have been used. Percent eelgrass coverage estimates the proportion of eelgrass patches that compose the area of a bed (e.g., 0-25%, 26 to 50%, etc.) (Bernstein et al. 2011). Eelgrass coverage at multiple scales has been estimated based on diver surveys, underwater videography, and side-scan sonar (Norris et al. 1997; Bernstein et al. 2011). Permanent plots have been established in some areas to assess rates of expansion and mortality of patches within an eelgrass meadow (Oleson and Sand-Jensen 1994).

Eelgrass condition within a defined patch is often indicated by the mean density of leaf shoots m^{-2}. Shoot density has proven a useful indicator of productivity response to environmental change and is sensitive to a wide variety of anthropogenic disturbances, including effects of commercial mussel harvest (Neckles et al. 2005), boat docks and other light-limiting obstructions (Burdick and Short 1999), eutrophication and associated macroalgal cover (Hauxwell et al. 2001; Hessing-Lewis et al. 2011), and climate change (Short & Neckles 1999). In situ measurements at representative reference and disturbed sites have been used to compare eelgrass shoot density and to quantify the extent and intensity of disturbance over larger areas (Neckles et al. 2005).

On the Pacific Coast, eelgrass coverage and density indicators have been used primarily in southern California bays (Bernstein et al. 2011) and in National Estuarine Research Reserves (e.g., Rumrill & Sowers 2008). Unlike estimates of eelgrass extent, which rely on indirect methods (i.e., imagery) to map areal distribution over large regions, monitoring protocols for eelgrass density typically involve surveying permanent plots within a bed to quantify short-term changes representative of a larger area. SeagrassNet has established standard monitoring protocols for vegetative parameters and environmental variables that allow regional and world-wide comparisons of seagrass changes through time (Short et al. 2006).

Areal coverage of biogenic species. Biogenic species other than macrophytes, such as structure-forming invertebrates, provide habitat for diverse subtidal communities (Dayton 1985, Syms & Jones 2000, Tissot et al. 2006). These communities often consist of biologically and commercially-important species of algae, invertebrates, fish, and marine mammals (Foster & Schiel 1985, Steneck et al. 2002, Tissot et al. 2006). Thus, understanding the spatial and temporal variation in the quantity of this habitat will be a useful measure of the quantity of nearshore habitat. Following the framework of Link (2005), reference points related to percent change in areal coverage of biogenic species could be established.

The distribution of biogenic species has been measured historically in numerous ways. Many historical datasets include scuba diving surveys (e.g., Partnership for Interdisciplinary Studies of Coastal Oceans [PISCO] at http://www.piscoweb.org/, U.S. National Park Service at http://www.nps.gov/chis/contacts.htm), but these are generally over small spatial and short temporal scales. Recent advances in satellite and infrared photography should allow researchers to measure areal canopy cover and biomass of kelps along much of the U.S. Pacific Coast (Deysher 1993, Cavanaugh et al. 2010), but measuring the coverage of structure-forming invertebrates will only be possible in specific areas such as oyster flats, which can be surveyed when they are exposed, or areas where long-term monitoring occurs using scuba surveys or hydroacoustic sonar methods (e.g., multi-beam, side-scan).

The areal coverage of biogenic species is easily understood by the public and has been used by policymakers to delineate essential fish habitat (e.g., Habitat Areas of Particular Concern). Changes in the coverage of biogenic species can affect recruitment of invertebrates and other species (Zimmerman et al. 1989, Carr 1991, Lenihan et al. 2001, Peterson et al. 2003), such that areal coverage of biogenic species could anticipate recruitment of older life stages into offshore populations and into various fisheries; thus areal coverage of biogenic species may not only be a good indicator for the quantity of nearshore habitat, but could also be a leading indicator for community-level attributes of the CCLME.

HABITAT QUALITY

We evaluated 14 indicators for estuarine and nearshore habitat quality (Table H2). These indicators were related to the quantification of factors affecting system productivity (e.g., dissolved oxygen and temperature) and growth of organisms inhabiting estuaries. Growth indicators had limitations with respect to primary criteria, and in terms of spatial and temporal data limitations. We identified six promising indicators of estuarine habitat quality with high spatial or temporal resolution.

Water temperature. Water temperature is an important habitat quality metric because most aquatic species exhibit temperature-dependent growth windows (e.g., Buckley et al. 2004, Hinke et al. 2005). At low temperatures metabolism is slowed, resulting in low growth rates. At higher temperatures, ectothermic aquatic organisms have a higher metabolism, and so must consume more food (Portner 2002). At physiologically stressful temperatures, organisms are unable to keep up with metabolic demands. In addition, dissolved oxygen concentrations decline at high temperatures following Boyle's Law, and organisms can expire from heart failure due to lack of aerobic scope (Farrell et al. 2008).

Water temperature has a long record of measurement across the Pacific Coast and is one of the most commonly measured water quality variables. Data varies in terms of spatial and bathymetric coverage, frequency and methods employed. Methods vary from spot surface or benthic measurements during other sampling events, monthly or other consistent periodic measurements across the water column, continuous measurements at particular depths using automated loggers, and nearly continuous water column sampling at automated buoys. Satellite datasets in the infrared spectrum also can be used to interpret surface temperature in coastal environments (Thomas et al. 2002, Franz et al. 2006, Thomas and Weatherbee 2006).

Dissolved oxygen. Dissolved oxygen in estuarine and nearshore areas has been widely acknowledged as an important indicator of habitat quality for fish. Dissolved oxygen is required for aerobic respiration, so all fish and shellfish species are sensitive to low dissolved oxygen, although some species are more sensitive to declines than others. Standards for hypoxic (< 2 mg/l) and stressful conditions (< 5 mg/l) have been long established, based on laboratory studies and documented fish kills in the field. In addition the seasonal conditions associated with low dissolved oxygen are now well understood – in the California Current, low dissolved oxygen is associated with upwelling events in the spring. However, hypoxia in some nearshore environments and deep estuary systems like Puget Sound is often most acute in the late summer and early fall, when near-bottom hypoxic water created as a consequence of microbial respiration in stratified waters undergoes mixing and affects a larger portion of the water column. In shallower systems, hypoxia can occur as a result of eutrophication and subsequent bacterial activity. Hypoxia has been linked with low pH and high carbonic acid levels; hence where these other metrics are unavailable, low dissolved oxygen has been used as an indicator of ocean acidification. As a consequence of all these factors, dissolved oxygen has been routinely measured in water quality surveys within estuaries and nearshore areas by state, federal, and other groups. In some cases, these datasets are readily accessible, but even these have key spatial and temporal gaps.

Turbidity. Turbidity is a consequence of suspended solids in the water column and is an important indicator of habitat quality for a number of species. Turbidity influences light diffusion and attenuation and hence the ability of phytoplankton and macrophytes to perform photosynthesis. Moderate levels of turbidity may reduce predation risk of planktivorous fish without impacting their ability to feed, while high levels of turbidity can clog gills. Extremely high turbidity levels can abrade tissues like eyes and gills, although these events are rare and occur primarily in freshwater under high run-off conditions. Turbidity is associated with riverine inputs, particularly during run-off events. For example, the Columbia River is well known for the relatively high turbidity levels in its plume, and fish utilization of the plume is associated with turbidity level and spatial variation. In addition to riverine inputs, high primary productivity by phytoplankton can elevate turbidity, resulting in negative feedback on primary producers such phytoplankton and submerged aquatic vegetation.

Turbidity has had a long history of being measured in estuary and nearshore environments with a Secchi disk. Increasingly turbidity is measured with optical sensors that calculate light scattering properties based on nephelometric turbidity units (NTUs). Like many other metrics measured in estuary and nearshore environments, turbidity measurements have many spatial and temporal gaps. Over the last 16 years, turbidity has been measured using NASA's SEAWIFS remote sensing data. These measurements, based on surface optical properties of turbid waters, are sensitive to reflectance and other noise created by coastal activities, and the spatial resolution is relatively coarse for estuary systems. Hence, remotely sensed turbidity measurements must be considered carefully in the context of estuary and nearshore systems.

Chlorophyll *a*. The concentration of chlorophyll *a* is a direct measure of primary production by phytoplankton and therefore a useful indicator for basal elements of food availability in aquatic environments including estuary and nearshore environments. As such, chlorophyll *a* is a leading indicator of ecosystem function, and is sensitive to anthropogenic alterations in coastal waters such as nutrient additions. However, a number of different microbes including diatoms and dinoflagellates produce chlorophyll *a*. Hence, overall concentrations of chlorophyll may not be informative for groups of species that consume specific microbes or are dependent on these consumers.

Chlorophyll production has been measured in several ways, including lab assays of concentration in water samples, fluorometric readings in automated water column profilers, and satellite-based measurements. Lab assays have the highest precision but data collection is often temporally or spatially patchy. In contrast, satellite methods have broad spatial and temporal coverage over the last 16 years, but the precision of measurements can be reduced for estuary and nearshore datasets due to reflectance and other issues. Calibrating satellite-based measurements along the coast with lab assays is currently an active area of research.

Nitrogen: Phosphorus ratio. The N:P ratio describes the ratio of two important nutrients in aquatic systems – total inorganic nitrogen (ammonium, nitrates, and nitrites), and phosphate ions (PO_4). Theoretical and experimental work has examined departures of this ratio from the ratio that primary producers uptake these nutrients (Redfield et al. 1963), the effects of anthropogenic nutrients upon this ratio (Cloern 2001), and the relationship of these nutrients with eutrophication. These nutrients are routinely measured in estuary environments, and a number of studies have documented trends in N:P in particular estuaries. They are also a component of the National Eutrophication Assessment's suite of indicators (Bricker et al. 2007) and the EPA's National Coastal Condition index. However, systematic spatiotemporally extensive measurements are much spottier (Greene et al. in press), so a fair amount of data synthesis may be required for systems not covered by previous national and state-wide assessments.

Silicate: Nitrogen ratio. The Si:N ratio describes the ratio of two important inorganic nutrients in aquatic systems – silicilic acid ($Si0_4$ ions) and total inorganic nitrogen (ammonium, nitrates, and nitrites). Like the N:P ratio, the benchmark for the Si:N ratio is the rate at which phytoplankton requiring Si (diatoms, most notably) optimally consume these nutrients (Redfield et al. 1963, Cloern 2001). Departures from this ratio indicate whether Si or N is limiting in a particular environment (Cloern 2001). Consequently this metric is sensitive to anthropogenic changes such as nutrient additions, water storage, and run-off. Unlike chlorophyll *a*, Si:N is particularly reflective of potential primary production by diatoms and is therefore a good potential leading indicator of primary productivity in estuaries and nearshore systems. Si and total N are very commonly measured inorganic nutrients. However, sampling programs vary temporally and spatially; Si:N measurements therefore suffer from spatiotemporal gaps.

PRESSURES

We evaluated 17 potential indicators of anthropogenic pressures in estuarine and nearshore environments (Table H2). In the California Current, estuaries tend to be subject to greater pressures than nearshore environments, and include threats that were outlined in the Freshwater Habitat section (upland environments). Hence, the indicators of pressures we outline below are in addition to those outlined earlier, and most focus on indicators measured within estuary and nearshore environments. The best indicators as noted below outperformed others due to extensive previous research on primary, the spatial and temporal breadth of sampling, and emerging importance. Conversely, we identified a number of potential indicators of pressures that lacked good scientific backing or lacked spatiotemporally extensive data. We identified eight promising indicators of pressures on estuarine and nearshore habitat.

Eustatic sea level rise. Sea level rise from climate change is expected to accelerate in the next century. The International Panel on Climate Change (IPCC) estimates that the global average sea level will rise further between 0.6 and 2 feet (0.18 to 0.59 meters) in the next century (IPCC 2007) as a result of natural processes and anthropogenic global warming. Across the Pacific Coast, the ranges of estimated sea level rise are between 10 and 167 cm by 2100, with strong latitudinal clines (NRC 2012). At its simplest, sea level rise is due to the thermal expansion of seawater (Domingues et al. 2008) and increased freshwater inputs from melting polar and glacier ice from the continents (Radić & Hock 2011). To best estimate the rate of sea level rise, vertical movements of the land such as post-glacial rebound need to be considered to get an adequate rate (Douglas 1991). Multiple time scales are associated with sea level rise. On multidecadal timescales, steric changes in the density field are often attributed to climate variability, while seasonal to interannual time scales variations are due to atmospheric and oceanic effects that can result in geostrophic readjustments.

Records of sea level rise must be multiple decades in length to distinguish changes over naturally occurring low-frequency signals that derive from atmospheric and oceanic forcing (Parker 1991). Three tidal gauge locations within the California Current ecosystem achieve the criteria of being exceptionally long in length. They are: San Diego, CA (1906-present), San Francisco, CA (1897-present), and South Beach, OR (1967-present).

Combining coastal tide gauges with satellite altimetry (Saraceno et al. 2008) can provide a direct measure of sea level rise, although time series are limited by satellite altimetry availability.

Organic pollutants in fish and shellfish. Organic pollutants measured in fish and shellfish tissue include industrial pollutants such as polychlorinated biphenyls (PCBs), organochlorine pesticides such as DDTs, chlordane, and dieldrin, and more recently, the flame retardants, polybrominated diphenyl ethers (PBDEs). Polycyclic aromatic hydrocarbons are also organic pollutants of concern, which bioaccumulate in shellfish, but to a lesser extent in fish (Varanasi et al. 1989). Exposure to these compounds can be monitored by measuring their metabolites in fish bile (Beyer et al. 2010). Most organic pollutants are not extensively metabolized by fish and shellfish, and generally there are good correlations between levels of organic pollutants in sediments and other environmental media and concentrations in fish and shellfish from the corresponding areas. This may, however, be influenced by how resident the target fish species is at the site of collection, as well as the lipid content of the target fish species. Fish with higher lipid content generally accumulate higher concentrations of organic contaminants.

Concentrations of organic pollutants are typically measured by gas chromatography and mass spectroscopy (GC/MS) using standard protocols common to all laboratories, with some minor modifications (e.g., Sloan et al. 2004; EPA 2007a, 2007b, 2008). For classes of compounds that include multiple congeners or isomers (e.g. DDTs, PCBs, PBDEs), there may be some variability in the specific chemical congeners or isomers measured, with larger number of compounds generally being measured in more recent analyses. There may also be variation in detection limits, with higher detection limits in older data. However, total concentrations of these chemicals are often comparable, as the most commonly occurring and abundant congeners and isomers are consistently measured.

Data on concentrations of organic contaminants in fish and shellfish from Pacific Coast estuarine and nearshore environments are available from a variety of sources, including the EPA's Coastal Condition and EMAP programs (EPA 2005; Hayslip et al. 2006, 2007; EPA 2012); NOAA's Mussel Watch program (Kimbrough et al. 2008); NOAA's National Benthic Surveillance program (Brown et al, 1998; McCain et al. 2000); the California Water Resources Control Board California Surface Water Ambient Monitoring Program, SWAMP (Davis et al. 2007, 2012) and the Puget Sound Ecosystem Monitoring Program, PSEMP (West et al. 2001; PSAT 2007; West et al. 2011). Monitoring has also been conducted in the Lower Columbia River and Estuary (e.g., LCREP 2007; Nilsen et al. 2014). While the most extensive datasets are available for major urban estuaries such as Puget Sound and San Francisco Bay, there is broad coverage, with the EMAP program, for example, providing data on 410 estuaries and bays and 3,940 square miles of coastal area (EPA 2012). Some datasets include information collected as long ago as the 1970s (Davis et al. 2007).

Concentrations of contaminants in fish and shellfish are easily understood by the public and have been used by policymakers to develop fish consumption advisories, to identify impaired water bodies, and for resource damage assessment and remediation at contaminated sites. Elevated concentrations of organic contaminants in fish and shellfish pose a threat not only to the affected fish themselves but to the wildlife and humans that consume them. Moreover, the effects of contaminants on the health and productivity of estuarine species may affect fish recruitment and populations of fisheries.

Sediment quality index. Various types of sediment quality indices are widely used in estuarine and nearshore environments along the Pacific Coast. Most of these indices include three components: concentrations of chemical contaminants in sediments, sediment toxicity to benthic organisms in bioassays, and benthic community condition, evaluated by metrics such as invertebrate species diversity, or proportions of sensitive and tolerant species (Borja & Dauer 2008; Long et al. 2006; Chapman et al. 2013).

Chemical concentrations in samples are generally compared with sediment guidelines associated with the likelihood of toxicity or injury to benthic organisms. Various guidelines are used, such as the effects range low (ERL) and effects range moderate (ERM) of Long and colleagues (1995; 2006) which is used in the EPA's Coastal Condition Assessment (EPA 2012) and related assessments performed as part of the Pacific Coast EMAP program (Hayslip et al. 2006, 2007), as well as some assessments performed by the State of Washington's Department of Ecology (Dutch et al. 2009). The State of California uses two sets of guidelines (Bay and Weisberg 2012): the California Logistic Regression Model (CA LRM), a logistic regression modeling approach that estimates the probability of acute toxicity in sediments based on the chemical concentration; and the Chemical Score Indicator (CSI), which is based on the association of chemical concentration with benthic community disturbance. Based on comparison with these guidelines, areas are classified into categories such as minimally exposure, low exposure, moderate, exposure, or high exposure.

Sediment toxicity is evaluated with invertebrate bioassays. The State of California, the EPA Coastal Condition Assessment Program, and the State of Washington all use a marine amphipod survival bioassay (EPA 2012; Bay et al. 2007). Responses are assigned to categories of non-toxic, low toxicity, moderate toxicity and high toxicity, depending on how they compare with responses on uncontaminated control sediments.

Benthic community measures are also included in most sediment quality indices. The EPA Coastal Condition Assessment, for example, uses a benthic index that compares invertebrate species diversity at each site to the expected diversity for the specific salinity representative of the site (EPA 2012). In the State of California, up to four benthic community condition indices are used to determine the magnitude of disturbance to the benthos at each site (Bay & Weisberg 2012). These include the Benthic Response Index (BRI) based on the pollution tolerance of the organisms present; the Index of Benthic Biotic Integrity (IBI), which identifies community measures that have values outside a reference range of estuaries; the Relative Benthic Index (RBI), which incorporates several community metrics as well as presence or absence of both positive and negative indicators species; and the River Invertebrate Prediction and Classification System (RIVPACS), which calculates the number of reference taxa present in the test sample and compares it to the number expected to be present in a reference sample from the same habitat. The results are combined to provide an overall benthos level of effect category, with four levels ranging from reference to high disturbance. In Washington State, a benthic community condition is also assessed from a suite of indices, including total abundance, major taxa abundances, taxa richness, evenness, species dominance, and abundance of stress-sensitive and -tolerant species. These indices are compared to median values for all of Puget Sound to determine whether the invertebrate assemblages appeared to be adversely affected or unaffected by natural and/or human-caused stressors (Dutch et al 2012).

Finally, the sediment chemistry, toxicity, and benthos data are typically integrated into an overall assessment of site condition. Both the State of California and the State of Washington classify sediment quality into six categories of impact ranging from unimpacted to clearly impacted, plus an inconclusive category for cases in which the three lines of evidence conflict (Bay & Weisberg, 2012; Dutch et al., 2012). The EPA Coastal condition assessment uses good, fair and poor ratings (EPA 2012).

As the discussion above indicates, sediment quality index data are available from nearshore and estuarine sites all along the Pacific Coast. Time series data are also available for some sites and estuaries. In Puget Sound, for example, sediment quality index data are available from 1997 to the present (Dutch et al. 2012). Sediment quality indices present some challenges as indicators because the exact components included in them and their methods of calculation vary from program to program and state to state. Also, indicator reporting is often limited to proportions of samples classified as unimpacted or in good condition, possibly impacted or in fair condition, and clearly impacted or in poor condition. However, similar data are collected for all the indices, and underlying data are usually available, so a consistent methodology could be applied to generate a uniform index or classification scheme for all nearshore and estuarine sites. Indeed, the EPAs Coastal

Condition assessment has applied their index to sites from Washington, Oregon, and California.

While sediment quality indices can appear complex, their basic components of sediment contaminant concentrations, toxicity to benthic organisms, and changes to benthic communities are easily understood by the public. Sediment quality indices are used by policymakers to evaluate dredged material, to identify impaired water bodies, and for resource damage assessment and remediation at contaminated sites. Elevated concentrations of contaminants in sediments and injury to benthic communities are in themselves a concern, but have wider implications for incorporation of contaminants into estuarine and nearshore food webs, as well as potential indirect effects on fish and other aquatic organisms that use benthic invertebrates as a food source through reductions in prey quality and availability.

Eutrophic state. Eutrophication is defined as "the enrichment of water by nutrients causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned, and therefor refers to the undesirable effects resulting from anthropogenic enrichment by nutrients (OSPAR 1998). Eutrophication can lead to increases in hypoxia, fish kills, and the occurrence of harmful algae (e.g. Boesch, 2002). Various indicators of eutrophic state have been developed in Europe and the United States but common components include chlorophyll *a* as a measure of phytoplankton biomass; and several physiochemical indicators including total phosphorus (TP), total nitrogen (TN), dissolved inorganic nitrogen (DIN), and dissolved inorganic phosphorus (DIP) as indicators of nutrient levels, as well as dissolved oxygen (DO) as an indicator of potential hypoxia and water quality degradation (Ferreira et al. 2011).

Many of these parameters are routinely measured in Pacific Coast estuarine and nearshore environments as part of water quality assessments required by the EPA. However, some indices have also been developed and applied that deal specifically with eutrophication. For example, the EPA's Coastal Condition Assessment Water Quality Index includes all of these indicators and uses them to assess the extent of eutrophication in coastal estuaries (EPA 2012). Some states have components of their water quality assessment program that deal specifically with eutrophication. For example, as part of the Oregon Water Quality Index, a eutrophication sub-index is calculated based on ammonia-nitrate nitrogen and phosphorus (Cude 2001). The Puget Sound Marine Water Condition Index (Krembs 2012) also includes a Eutrophication Index which incorporates ambient changes in levels of nutrients (concentrations of nitrate, nitrate:DIN, and phosphate); nutrient enrichment (changes in ammonium, phosphate, and nitrate concentrations in estuarine or nearshore waters compared to ocean conditions); and the impact (changes in the balance of nutrients and algal biomass, as indicated by DIN:phosphate, silicate:DIN, and chlorophyll *a*).

Some coast-wide assessments of eutrophic state are available for Pacific Coast estuarine and nearshore sites. As mentioned above, eutrophication is assessed as part of EPA's National Coastal Condition Assessment (EPA 2012). Additionally, NOAA conducted a nationwide assessment of eutrophication in coastal water, including 29 estuarine and nearshore sites on the Pacific Coast in 1999, and updated in 2007 (Bricker et al. 2007). This assessment provides a rating of eutrophic condition based on common symptoms of eutrophication, including increased chlorophyll *a*, epiphytes, and macrophytes, low dissolved oxygen, loss of submerged aquatic vegetation, and increased frequency of nuisance and/or toxic algal blooms. Embayments were ranked high, moderate and low for eutrophic condition. At the state level, information on components of the Oregon Water Quality Index related to eutrophication are available in Oregon water quality index annual reports (e.g. Merrick & Hubler 2013), which date from 2001. The Puget Sound Marine Water Eutrophic Index was adopted only in 2012, but has been evaluated as far back as 1999 from previously collected data (Krembs 2012). Additionally, information on variables related to eutrophic state is widely available as part of state, federal, and local water quality assessment programs but these data are not integrated into an index or comparable comprehensive evaluation of eutrophic status.

The general concept of eutrophic state and the overall findings of evaluations using eutrophication indices are easily understood by the public and policy makers (i.e., the eutrophic condition or index score of a particular area is good or poor, high or low) but the details of index calculation and differences among indices with different components may be less clear. Since these indices measure current water quality conditions, they generally provide an assessment of changes that have already taken place (i.e., nutrient enrichment, increased algal growth, hypoxia) so this is generally a lagging indicator. Trends in some parameters, such as nutrient levels, however, may be indications of developing problems even if currently measured values would not be indicative of impaired waters.

Beach closures. Beach closures are a simple indicator relating to fecal coliform or other bacterial outbreaks or at estuarine and nearshore sites. Human activities including sewer treatment plants, failing septic systems, improper handling of boat waste, combined sewer outfalls, agricultural activities, and animal waste are major sources of bacterial contamination of aquatic environments. These microbial contaminants may include disease-carrying organisms that pose a risk to public health. For example, use of swimming beaches that do not meet water quality standards for bacterial contamination can result in gastrointestinal illnesses, respiratory illnesses, and skin infections. This is problematic not only for infected individuals, but for the economies of coastal towns that are dependent on income from tourism at coastal beaches.

Beach closures, as well an indicators related to closure, such as levels of microbial contaminants, are tracked and used as a habitat quality indicator in estuaries throughout the Pacific Coast. Much of this data is generated by states and counties in conjunction with EPA's Beaches Environmental Assessment and Coastal Health (BEACH) Program, initiated in 1999 to reduce the risk of disease to users of marine recreational beaches. The EPA provides national guidance on beach monitoring as part of its Environmental Monitoring for Public Access and Community Tracking (EMPACT) Program (EPA 2003), which is incorporated into programs administered by the states. The California beach program is the most extensive in the nation, annually sampling 656 monitoring stations at 291 beaches. The Oregon Department of Environmental Quality and Oregon Health Authority also have a program that monitors recreational water quality at ocean beaches (ODEQ 2006). The State of Washington BEACH Program (WDOE 2002; Schneider 2004) led jointly by the Washington State Departments of Ecology and Health, is comparable. In Washington, the condition of swimming beaches is a Vital Signs indicator in the Healthy Human Population component of the Puget Sound Ecosystem Monitoring Program, in which the number of beaches not meeting the EPA water quality standards for the fecal bacteria enterococcus is tracked.

Data on beach closures are available from a number of sources. The EPA releases an annual report on beach closures by state (e.g. EPA 2013b; see http://water.epa.gov/type/oceb/beaches/summarylist.cfm). Data are available as far back as 1999, though information by state may not be available prior to 2006. Beach closures are also used as an indicator in EPA's Coastal Condition Assessment reports (EPA 2012). In California, beach closure reports are issued by the counties and by the Southern California Water Resources Control Board (e.g., SCWRB 2002). California is also developing a statewide California Beachwatch database to collect all state beach water quality information. Data on beach closures are also available through the Oregon and Washington BEACH programs. In Washington, the condition of swimming beaches is an indicator in the Healthy Human Population component of the Puget Sound Ecosystem Monitoring Program. This indicator tracks the number of beaches not meeting the EPA water quality standards for the fecal bacteria enterococcus. Heal the Bay (www.healthybay.org), a non-profit organization based in Southern California, also compiles data on beach closures and other measures of beach quality, and issues a yearly beach report card. Information has been compiled for California beaches since the 1990s

(Heal the Bay 2000), and the more recent reports include Oregon and Washington (Heal the Bay 2011).

Recreational water quality standards associated with beach closures may vary somewhat from state to state. Most are based on the EPA guideline of less than 104 enterococcus bacteria per 100 ml saltwater, but some states like California have additional sampling criteria, so conditions for closures may be more stringent.

This indicator is easily understood by public and policy makers, and generally could be expected to respond in a predictable way to management actions directed toward reducing bacterial contamination. Beach closures and proportions of beaches meeting water quality standards for fecal coliform are used by policy makers and managers to regulate water quality for the protection of human health. As discussed above, these measures are also used as habitat quality indicators nationally and in state and local programs in specific estuaries, including Puget Sound.

Fish disease. Fish disease has been used as an indicator of environmental quality in a number of studies worldwide (Au et al. 2004). In some cases, as in EPA's EMAP program, assessments are made by collecting gross pathology data on parasites, visible tumors in liver, fin erosion, abrasions, and other lesions (EPA 2001). In other cases, fish tissues are examined microscopically to diagnose disease conditions based on histopathology (e.g., Murchelano 1990; Myers et al. 1998, Schwaiger et al. 2003; PSAMP 2007; Stentiford et al. 2009). The latter studies document a range of lesions in fish liver tissue, including neoplasms and pre-neoplasms that are highly correlated with exposure to carcinogenic polycyclic aromatic hydrocarbons in field studies, and that have also been induced with controlled exposure to similar chemicals in laboratory settings (Myers et al. 2003).

Fish disease monitoring with gross pathology can be somewhat problematic as an indicator of estuarine and nearshore habitat quality, as its relationship with chemical contamination and habitat degradation can be inconclusive. However, it has been used successfully in some East and Gulf Coast estuaries sampled in EPA's EMAP program (Fournie et al. 1996; Landsberg et al. 1998). Fish liver disease, on the other hand, shows a much more consistent relationship with chemical contamination, especially with exposure to PAHs (Myers et al. 2003; Logan 2007), and has been used successfully to monitor improvements in fish health and ecological condition of PAH-contaminated sites following cleanup and remediation (PSAMP 2007; Myers et al. 2008). However, this indicator also has some limitations, as there is variation in the susceptibility of different fish species to liver disease, due to differences in diet and migratory patterns that affect exposure, as well as to differences in metabolism and detoxification of PAHs (Logan 2007). The risk of liver disease also increases with age (Stentiford et al. 2010), so this factor must be taken into account in comparing lesions prevalences.

Data on gross pathology in fish have been collected along the Pacific Coast as part of EPA's EMAP program (Hayslip et al. 2006, 2007) though its relationship with other measures of environmental degradation has not been comprehensively analyzed. Histopathological data on lesions in benthic fish were collected as part of NOAA's NBSP (Myers et al. 1998). Extensive data are also available for Puget Sound, collected as part of the PSEMP program, in which liver lesions in English sole *Parophrys vetulus* are a key indicator for PAH exposure and injury (PSAT 2007). Several studies have also been conducted in specific embayments in California (e.g., Basmadiian et al. 2008).

Fish disease, including fish cancer, is easily understood by the public and has been used by policymakers to identify impaired water bodies. Fish liver disease, on the other hand, shows a much more consistent relationship with chemical contamination, especially with exposure to PAHs (Myers et al. 2003; Logan 2007), and has been used successfully to monitor improvements in fish health.

Fish vitellogenin (VTG) induction. Since the 1990s, there have been many reports of releases of synthetic and natural estrogens into river systems and marine waters (Ramirez et al. 2009), including into nearshore and estuarine sites on the Pacific Coast (Alvarez et al. 2014; Sengupta et al. 2014). Exposure to these chemicals has been associated with a number of health effects on aquatic organisms, including altered reproductive development and behavior, reduced fertility, intersex, and feminization of males (Kime 1996; Goksyr 2006).

Among the actions of estrogens in fish is the induction of the yolk protein, vitellogenin, which is incorporated into the developing egg (Tyler et al. 1990). In female fish this is a natural occurrence induced by increased levels of endogenous estrogens during the reproductive cycle. However, abnormal induction of vitellogenin may also occur in male and juvenile fish when they are exposed to estrogens or estrogen-like compounds from an exogenous source. Accordingly, the induction of vitellogenin in male or juvenile fish has become a useful environmental indicator for the presence of and exposure to environmental estrogens in aquatic life (Sumpter & Joblins 1995; Kime et al. 1999).

Vitellogenin can be measured in fish through a variety of methods (Sumpter & Jobling 1995; Jones et al. 2000). One of the most widely used in the enzyme-linked immunoassay (ELISA). Alternatively, exposure to environmental estrogens has been detected by monitoring increased expression of estrogen responsive genes, including those associated with the production of vitellogenin and zona pellucida (egg shell) proteins

(Arukwe & Goksyr 2003; Filby et al. 2007; Baker et al. 2013). One drawback of the indicator is that assays must often be developed for target species of concern (Sumpter and Jobling 1995; Tyler et al. 1996), although assay kits are increasingly available for a range of fish species, and some universal assays can be applied across species (Heppell et al. 1995; Van Veld et al. 2005). However, studies suggest that while relative levels and trends are generally consistent, there may be substantial interlaboratory variability in VTG concentrations measured by ELISA (Batelle 2003). Finally, as this indicator has been applied only relatively recently to environmental monitoring programs, long-term trends data are generally lacking.

Vitellogenin induction has been used as an indicator of xenoestrogen exposure in several Pacific Coast estuarine and nearshore sites, including Puget Sound (Johnson et al. 2008; Peck et al. 2011), San Diego, Orange County and Los Angeles (Rempel et al. 2006; Deng et al. 2007; Baker et al. 2013), and the Lower Columbia River and Estuary (Hinck et al. 2006; LCREP 2007; Jenkins et al. 2014). Results indicate widespread exposure of fish to environmental estrogens, with especially high proportions of fish affected in areas near industrial and municipal outfalls. The chemicals responsible have not always been identified, although in Puget Sound, analyses of fish bile suggest important sources of estrogen activity may be the plasticizer bisphenol A, and natural and synthetic estrogens (17-beta estradiol, estrone) often present in sewage (da Silva et al. 2013).

Fish vitellogenin induction is readily understood by the general public when explained as abnormal production of egg yolk proteins in male or juvenile fish. It is included as one of the recommended assays is EPAs endocrine disruptor screening program (EPA 2009) and is being used as an indicator in environmental monitoring programs in Puget Sound and southern California. It can also be a useful indicator to evaluate the effectiveness of toxics reduction activities, including changes in sewage treatment to reduce estrogenic compounds (Vidal-Dorsch et al. 2014).

Table H 2. Summary of estuary/nearshore indicator evaluations across five primary considerations, seven data considerations, and six other criteria. Each criterion was scored 0, 0.5, or 1 depending on the level of literature support for that criterion. The numerical value that appears under each of the criteria groupings represents the number of evaluation criteria supported by peer-reviewed literature. For example, areal extent of salinity zones has peer-reviewed literature supporting five out of five primary considerations criteria. *Indicators in the top quartile; ** Promising indicators with gaps; unmarked indicators scored poorly and will not be considered further.

Indicator	**Primary (5)**	**Data (7)**	**Other (6)**	**Summary Comments**
Quantity				
River flow *	5	6.5	6	River flow is an important component of water quantity in estuaries and influences dynamics in estuaries and nearshore areas. Many USGS gages facilitate measurement of river flow, but coverage is spotty and time series are often not extensive.
Areal inundated wetland coverage*	4.5	4	5	Areal wetland coverage is an important measure of habitat quantity for all species that are resident in estuaries. Extent can be measured using remote sensing, although the extent of freshwater tidal zones requires additional analysis and groundtruthing.
Area of salinity zones**	5	3.5	3	Salinity zones are important transitions for a number of species and drive what marsh vegetation will grow. Salinity zones are temporally dynamic, fluctuating daily and seasonally. Measures of the average extent are possible but analysis requires extensive groundtruthing over time.
Isohaline position**	5	4	4	This metric may be useful for large estuaries influenced by water diversions or storage. It is currently quantified only for the Sacramento-San Joaquin Delta, but is useful for understanding habitat available for ESA listed species there.
Area of physical habitat	4.5	4	3.5	Management protects physical habitat that may otherwise be modified or disturbed by fishing or other industrial activities; however, increases in physical habitat, such as rock are not likely possible unless sediments are scoured away, but further loss of habitat may be possible. Nearshore, subtidal estimates can be difficult as multi-beam sonar surveys are less prevalent than in offshore habitats.
Areal macrophyte extent*	5	5.5	4	Macrophytes (e.g., eelgrass and kelp) provide habitat to diverse marine communities. Extent and coverage could anticipate recruitment of fish. Recent advances in satellite imagery and algorithms can help quantify extent and biomass efficiently.
Macrophyte density*	5	5	4	Macrophytes provide habitat to diverse marine communities. Density estimates are difficult to get from satellite imagery, so diver surveys along the coast are required.
Floodplain area: drainage area	2.5	4.5	2	Facilitates comparison of floodplain area among river systems. However, it is not expected to greatly change inter- or inter-annually.
Network complexity (number of nodes)	4	5	3	Network complexity provides insight into the existing estuary distributary network. However, it is not expected to greatly change inter- or inter-annually.
Estuary surface area:drainage area	3	4.5	2.5	This metric facilitates comparison of estuarine area among river systems. However, it is not expected to greatly change inter- or inter-annually.
Detritral production	2.5	4	3	Detrital production is one variable influencing accretion in estuaries and some nearshore environments. However, it is sporadically measured with many temporal gaps, and poorly

				understood as an indicator by the public.
Sediment deposition (mm)	3.5	4.5	3	Sediment deposition is one variable influencing accretion in estuaries and nearshore environments. However, it is sporadically measured with many temporal gaps.
Structure forming invertebrate extent	4.5	4.5	4	Benthic communities are highly diverse in habitats created by sessile invertebrates. Surveys of subtidal communities exist, but at small spatial scales.
Areal coverage of biogenic species*	5	6.5	5	Biogenic species provide habitat to highly diverse communities. Mapping of kelps, seagrasses, and sessile invertebrates can be combined to develop broad calculations of habitat quantity across nearshore habitat in the CCLME.
Density of biogenic species	4.5	6	5	Biogenic species provide habitat to highly diverse communities. Similar to macrophytes, density of biogenic species will be more difficult to quantify across the entire CCLME as most estimates are at small spatial scales and the difficulty in using satellite imagery.
Un-impounded shoreline extent	4.5	4.5	2.5	Shoreline modification alters nearshore currents and coastal sediment delivery processes which can interfere with the recruitment and survival of biogenic habitat. Satellite imagery could potentially measure changes in impoundment of shoreline.
Quality				
Temperature*	5	7	5	Important indicator of growth potential in estuary and nearshore environments and of impacts of global warming in these habitats. Data collection efforts are by many different agencies.
Dissolved O2*	5	7	5	Important indicator of growth potential in estuary and nearshore environments and of hypoxic conditions. In some places, data may be limited in time or space. Some historical conditions are known through sediment cores.
Turbidity*	5	7	5	Turbidity is important in estuary and nearshore environments as an indicator of phytoplankton production, and sediment delivery. This metric is spatially and temporally patchy, although satellite data exists that may be useful in estuary and nearshore habitat if well-calibrated to field conditions.
Chl a*	4.5	6	5	Good indicator of phytoplankton biomass and amount of energy fueling the ecosystem, satellite remotely sensed chlorophyll concentration data available system wide. However, satellite data are biased for nearshore areas and ground-based methods are therefore more accurate.
N:P*	5	7	5	Important indicator of nutrients for phytoplankton production, nutrient inputs by people, and eutrophication. Data is spatially and temporally patchy.
Si:N*	5	4	3.5	Important indicator of nutrients for production by diatoms, nutrient inputs by people, and eutrophication. Data is spatially and temporally patchy.
Water quality index	5	5	4.5	This type of metric has been used to summarize multiple physical water properties. The time series is just over a decade and currently limited to Puget Sound.
fish size and growth	3.5	4	2.5	These metrics have been used to infer growth benefits to key fisheries species. However, different species have different growth controls, and measurements have many spatial and temporal gaps.
Diversity of sediment grain size	2.5	2.5	4	Variation in sediment grain size in estuaries and nearshore environments provides one metric of habitat complexity. However, it is unclear how this metric informs habitat science, particularly when variation in this metric is not well understood and poor records existing across multiple

				systems and years. Annual variation in this metric is expected to be low.
Invertebrate density (benthic core, insect fallout, bongo net)	4	6.5	4	These metrics summarize food available for fisheries at early life stages. Very few systems use multiple sampling techniques even though all sample types are relevant, and the cost of taxonomic identification is high. Detailed time series are lacking for most systems.
Rugosity of substrate	4.5	5	4	Sampling of rugosity by multi-beam sonar can be useful in nearshore systems to examine structural complexity. However, post-processing of data can be expensive, and many spatial gaps exist, most without repeated measurements over time.
Habitat connectivity/ fragmentation	3.5	4.5	3.5	Habitat fragmentation and connectivity has been widely used in terrestrial contexts but much less so in aquatic areas. Measurements have many spatial and temporal gaps and poorly estimated historic condition.
Growth of biogenic habitat (kelps, sponges, corals, oysters)	3.5	4.5	2	Growth estimates from biogenic habitat provide one possible way to infer productivity during historical periods lacking direct monitoring. They also provide estimates of recovery rate of perturbation However, this metric needs additional calibration and data collection efforts to make it an effective metric across the California Current.
Pressures				
Eustatic sea level rise*	4	6	6	Sea level rise (SLR) is an important threat to estuary and nearshore systems. Several measurements are required to estimate SLR, so many systems lack adequate data to estimate affects and to monitor continuously.
Ocean acidification (pCO2, TCO2, alkalinity, calcite & aragonite saturation state)**	4	3.5	2.5	The frequency of corrosive waters has been increasing in the Pacific Northwest, and have directly impacted aquaculture facilities. Many data gaps exist across the coast due to the challenges of measuring carbonate chemistry, although national efforts may soon improve the technology and opportunity for long-term measurements.
Impoundment releases/hydrograph changes*	4	6	5.5	Changes in patterns of flow due to water storage and releases can be used to infer impacts to estuary habitats. Records have spatial and temporal gaps and often historical reference points do not exist.
Dam/Reservoir storage volume (acre-ft)*	4.5	6.5	3.5	Data series associated with water use and storage provide some of the best indicators of human impacts to freshwater input into estuaries. Freshwater storage data are available from state agency databases, which include information on construction date and impoundment area/volume for all dams.
Organic pollutants in shellfish & fish*	5	6	4.5	Data on concentrations of organic contaminants in fish and shellfish from Pacific Coast estuarine and nearshore environments are available from a variety of state and federal sources going back as far as the 1970s. However many spatial gaps exist.
Sediment quality index (pollutants, inverts)*	5	6	4.5	Provides information on sediment toxicity and invertebrate diversity and measured by state and federal agencies. Data are available in some systems from 1997 to the present, but time series may be limited for many systems. Sediment quality indices may be qualitatively estimated.
Eutrophic state **	5	3	4.5	This multi-metric index summarizes risk of an estuary to eutrophication. While this is a useful

				metric, many systems are not included, and updating has occurred every five years and may be discontinued.
Beach closures*	5	7	5	Beach closures provide a measure of the impacts of sewage, harmful algal blooms, and other impacts to recreational beach use. This measure can be tracked over time through state alerts, at relatively local levels.
Fish disease*	5	6	5	Fish diseases are easily understood by the public and policymakers, and has been used to assess effectiveness of toxics reduction and cleanup activities. Spatial and temporal data gaps exist.
Fish VTG induction*	5	5	5	Vitellogenin induction is an indicator of xenoestrogen exposure. It has been measured in several Pacific Coast estuarine and nearshore sites, including Puget Sound, Los Angeles, and the Columbia River, but large spatial and temporal gaps exist.
Shoreline armoring (dikes, hardening)	3	4	2	Shoreline armoring datasets have been completed for the Pacific Coast of North America by a variety of federal, state, and local agencies. Most, however, provide a baseline indication of current or recent conditions and are generally unavailable coastwide or over time.
Dredging	3	7	4	The amount of material (in cubic yards - CY) dredged from all waterways off the US Pacific Coast is a concrete, spatially explicit indicator that concisely tracks the magnitude of this human activity throughout the California Current region.
Aquaculture facilities (pounds produced)	3	6	4	Production is limited to the state of WA. Production will correlate with certain aspects of the pressures (e.g., escapement, disease, nutrient input, waste, fishmeal) on the ecosystem, but specific impacts may not increase/decrease with production as new technology is used to mitigate impacts on water quality or interactions with wild stocks.
Aquaculture facilities (acreage, number)	2.5	5	3	The amount of habitat used is relevant to determine impacts on the ecosystem. However, this metric may not account for advances in technology or growing capabilities. Data are limited to net-pen dimensions of the current year's permit, so there is little temporal data.
Nonnative macrophytes and invertebrates	5	3	5	A global assessment scored and ranked invasive species impacts (http://conserveonline.org/workspaces/global.invasive.assessment/). This database serves as a baseline for invasion, is spatially coarse, and has not been updated since its creation.
Inorganic pollutants in shellfish & fish	3	5.5	4.5	Measuring concentrations of inorganic pollutants in organisms assesses the severity and potential impacts of pollutants released. However, variation in other variables will still limit the correlation between these land-based pollutants and observations in the CCLME.
Organic pollutants (point and nonpoint sources)	5	6	5	Data are collected as part of various federal monitoring programs, so data will continue to be collected using standardized methods that will be useful for temporal and spatial analyses in the future.
Dissolved organic carbon, Particulate organic matter	4	4	3.5	Poorly characterized in CCLME; however, high POM usually linked to hypoxia and dead zones.

EVALUATION OF PELAGIC HABITAT INDICATORS

HABITAT QUANTITY

We identified four indicators of pelagic habitat quantity (Table H3). Of these, two indicators – euphotic depth and thermocline depth – were selected as high priority indicators. The other two indicators are plume and eddy size. Plumes from large rivers (most notably the Columbia River and the Fraser River in the Salish Sea) create areas of lower salinity and elevated turbidity. Eddies created by currents interacting with local topography create areas of longer water residence time. The direct impacts of these eddies on ocean life are poorly known but may represent unique habitat for some marine animals (Loggerwell and Smith 2001,Trainer et al. 2002, Burger 2003, Yen et al. 2006, Pool et al. 2008). Both plume environments and eddies are spatially restricted and are not extensively characterized for the entire CCLME (especially for smaller systems), but may be important elements of pelagic habitat quantity as additional data becomes available.

Euphotic depth and **Thermocline depth.** The euphotic or epipelagic zone is defined as the uppermost layer of the pelagic zone, where solar radiation can penetrate and therefore drive primary production. The lower boundary of this zone occurs around 200 m depth, where light radiation levels reach 1% of surface radiation (Checkley & Barth 2009). The thermocline is defined as the depth of maximum change in temperature and defines the bottom depth of the mesopelagic zone, below which water ceases to be mixed regularly (Checkley & Barth 2009). The depth of both the photic zone and the mixed water layer, and its temperature and solar irradiation play a key role on the productivity of pelagic ecosystems. In the California Current, the above attributes are subject to seasonal and interannual variability. Seasonal physical forcing is determinant to replenish nutrients to the euphotic zone, which in turn dictate the condition for primary production in the following spring (Mantyla et al., 2008, Ianson & Allen, 2002) and consequently the recruitment success of many fish species. The upwelling communities appear thus to be affected by the timing and intensity of both upwelling and downwelling, several times in advanced of the spawning and recruitment seasons. Epipelagic species, in particular those with planktonic early life stages seem to be extremely dependent on the conditions of the upper mixed layer (Lasker, 1978, Parrish et al., 1981). For example temperature is known to dictate the rate of development of eggs (Zwiefel & Lasker, 1976) and hence the duration of exposure to predators. Also, turbulence can modulate the feeding ability of larvae (Lasker, 1981), and upwelling generating-winds are known to disperse and transport the eggs and larvae, onto or beyond their suitable habitat (Bakun & Parrish, 1982). Therefore monitoring of the upper water column characteristics is essential for understanding trends in recruitment and planning sustainable exploitation plans for many commercial and ecologically important species in the California Current (McClatchie, 2014).

Euphotic depth and thermocline depth are routinely measured via water column measurements of photosynthetically active radiation and temperature in the CCLME. The longest time series of water column measurements encompassing physical and biological parameters is found on the California Cooperative Fisheries Investigations (CalCOFI) surveys (McClatchie, 2014). Although the surveys originally spanned the entire California Current (Hewitt, 1988), the current survey design encompasses four surveys per year focusing on Southern California waters, from the coastline to more than 200 miles offshore. Waters off Central California to the north are surveyed on a semi-periodical basis during fisheries-oriented surveys, for example the combined Hake/Sardine survey (http://www.nmfs.noaa.gov/stories/2012/11/11_26_12sake_survey.html). Partial sampling of the California Current is performed during many other surveys, for example the Annual midwater trawl survey for juvenile rockfish (Baltz et al., 2006), acoustic trawl method surveys for coastal pelagic species (Zwolinski et al., 2012) or meso-scale midwater multi-species trawls surveys (Suchman et al., 2012) Although the combination of the above and other fisheries surveys collectively survey the physical and biological characteristics of the upper mixed layer of a large proportion of the California Current, there is not an ongoing comprehensive and synoptic survey.

HABITAT QUALITY

We evaluated 12 indicators of pelagic habitat quality (Table H3). Most indicators have been previously examined in previous indicator assessments for the IEA (Levin & Schwing 2011, Hazen et al. 2013 Williams et al. 2012), and summaries of some of these are repeated below or reframed in a habitat context. Many were theoretically sound, relevant to management, and predictably responsive tended to meet many of our data criteria (e.g., chlorophyll *a*). Those potential indicators that did not score highly either did not meet primary criteria or were not well characterized in space or time. For example, salinity is well measured and may be an important indicator for river plume environments; however, other environmental variables (oxygen, temperature) have greater direct effects on organisms in the majority of the pelagic realm. Topographic upwelling is an emerging metric of importance (Genin 2004, Santora et al. 2011), that may create biological hotspots, but the extent and dynamics of these water mass boundaries is still poorly understood.

Temperature. Water temperature is a key driver of the rates for metabolism for both primary producers and ectothermic heterotrophs, including most fish. Not surprisingly, water temperature has a long record of measurement across pelagic areas of the Pacific coast and is one of the most commonly physically measured variables. Data varies in terms of spatial and bathymetric coverage, frequency and methods employed but most pelagic measurements come from water column measurements during periodic surveys, from fixed buoys, or from satellite-based measurements. Due to latitudinal differences, weather, currents, upwelling, and mixing, temperature can exhibit strong dynamic variation across the CCLME, so not all temperature variation is readily interpretable.

Turbidity. Turbidity is generally related to riverine or estuarine outflow (see plume size and volume) and is highest in the ocean immediately offshore of river mouths. However, even episodic storm events can create turbid plumes in such generally clear coastal areas such as in the Southern California Bight (Lahet & Stramski 2010). Terragenic sediments are likely to be the major contributor to the suspended material but during major phytoplankton blooms, biogenic particles (phytoplankton and zooplankton) are also likely to increase turbidity. The Columbia River Plume transports a great amount of suspended material directly offshore and in summer, along the coast of Oregon (Banas et al. 2009). The plume has well defined lateral boundaries separating turbid plume water from clearer coastal or oceanic water and may be an important localized high abundance area for plankton and fish (De Robertis et al. 2005, Morgan et al. 2005). It has been hypothesized that the Columbia River plume may serve as a refuge from predation for juvenile salmon and small forage fishes (Emmett et al. 2005), since experimental studies have shown that planktivores are still able to feed under relatively high turbidity levels whereas piscivores are generally prevented from feeding there due to the poor visibility (De Robertis et al. 2003).

Dissolved Oxygen. Low dissolved oxygen concentrations in coastal and shelf waters of the California Current ecosystem is a relatively recent issue (Grantham et al. 2004; Bograd et al. 2008). When dissolved oxygen concentrations fall below 1.4 ml L^{-1} (=2 mg L^{-1}= 64 μM), the waters are considered to be 'hypoxic'. The drawdown of oxygen primarily occurs in bottom waters, which are isolated from atmospheric influences and where a build-up of sinking organic matter fuels microbial degradation and respiration that consumes oxygen. Within the California Current, the primary source of nutrients to the system is from deep waters that are upwelled onto the shelf. There is evidence that the frequency, duration and spatial coverage of hypoxic events has been increasing over the last 20 years (Diaz and Rosenberg, 2008), potentially due to increased stratification (reduced vertical mixing) and a decrease in the oxygen concentration of upwelled waters. In the southern portions of the California Current, the shoaling of the permanent Oxygen Minimum Zone is a contributing factor (Helly & Levin, 2004; Bograd et al. 2008). The impact of hypoxia on organisms in the California Current is poorly understood.

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

Chlorophyll *a* data from from GLOBEC sampling cruises between 1997 and 2004 and CalCOFI cruises from 2000 to 2004 have been used CCLME ecosystem model building and calibration (Brand et al. 2007). Remotely sensed chlorophyll *a* concentration (mg m^{-3}) data can be obtained at minimal cost from the Sea-viewing Wide Field-of-View Sensor (SeaWiFS at http://oceancolor.gsfc.nasa.gov/SeaWiFS/) to derive broad-scale coverage of values over the CCLME (Polovina & Howell 2005) or at smaller regional scales (Sydeman & Thompson 2010). Phytoplankton color, derived from continuous plankton recorder surveys (http://www.sahfos.ac.uk/about-us/cpr-survey/the-cpr-survey.aspx), can also be used to show intensity and seasonal extent of chlorophyll *a* (Edwards & Richardson 2004). Species or subsets of species of phytoplankton that affect chlorophyll *a* concentration can serve as an indicator of change in phytoplankton biomass, but physical measurements of upwelling intensity may provide a better leading indicator.

Coho salmon smolt-to-adult survival rate. The salmon smolt-to-adult survival rate is considered a good indicator of the state of the CCLME because salmon populations are highly influenced by ocean conditions, and coho salmon marine survival in particular is significantly and independently related to the dominant modes acting over the coastal region in the periods when the coho first enter the ocean (Koslow et al. 2002, Logerwell et al. 2003, Scheuerell & Williams 2005, Peterson et al. unpubl. manuscr.). Furthermore, salmon are of high commercial, recreational, and cultural importance along much of the Pacific coast, and therefore have high relevance in the delivery of ocean ecosystem services to the region (NRC 1996). Strong coupling has been demonstrated between smolt-to-adult survival and ocean upwelling in the spring and fall, suggesting management policies directed at conserving salmon need to explicitly address the important role of the ocean in driving future salmon survival (Scheuerell & Williams 2005). Furthermore, the salmon smolt-to-adult survival rate may affect management as it relates to using ocean conditions to determine best release date of hatchery fish.

The Oregon Production Index (OPI), defined as the smolt-to-adult return rate for coho salmon in Oregon, is currently one of several time series considered useful ecosystem indicators within the California Current region (Peterson et al. unpubl. manuscr., Sydeman and Thompson 2010). This dataset is temporally extensive and comprehensive for the central CCLME (PFMC 2010). However, it is considered a lagging or retrospective indicator of ocean conditions due to the protracted life cycle of salmon (Scheuerell & Williams 2005, Peterson et al. unpubl. manuscr.).

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

PRESSURES

We evaluated four potential indicators of pressures on pelagic habitats (Table H3). These were previously examined by Andrews et al. (2013) as potential indicators of anthropogenic pressures. Of the four examined, we recommend three metrics – commercial landings, atmospheric pollution, and vessel traffic – as the primary and measurable pressures to pelagic habitats in the CCLME.

Commercial landings. This indicator represents commercial landings of coastal pelagic species from shoreside commercial fisheries. It also includes tribal removals and catches from exempted fishing permit studies. Commercial landings represent the bulk of fishery removals for highly priced, high retention rate species, but not for bycatch species that are often discarded when caught. Status and trends of this indicator, therefore, may not thoroughly represent changes in fishery removals, and will also reflect changes in markets or/and management. Data are summarized by the Pacific Fisheries Information Network (PacFIN) at http://pacfin.psmfc.org for Washington, Oregon, and California.

Atmospheric pollution. The impact of pollutants deposited from the atmosphere on marine populations is largely unstudied; however, many nutrient, chemical and heavy-metal pollutants are introduced to marine ecosystems from sources that are geographically far away via this process (Ramanathan & Feng 2009). Substances such as sulfur dioxide, nitrogen oxide, carbon monoxide, lead, volatile organic compounds, particulate matter, and other pollutants are returned to the earth through either wet or dry atmospheric deposition (Johnson et al. 2008). Atmospheric nitrogen input is rapidly approaching global oceanic estimates for N2 fixation and is predicted to increase further due to emissions from combustion of fossil fuels and production and use of fertilizers (Paerl et al. 2002, Duce et al. 2008). Atmospheric deposition is one of the most rapidly increasing means of nutrient loading to freshwater systems and the coastal zone, as well as one of the most important anthropogenic sources of mercury pollution in aquatic systems (Johnson et al. 2008). Industrial activities have increased atmospheric mercury levels, with modern deposition flux estimated to be 3-24 times higher than preindustrial flux (Swain et al. 1992, Hermanson 1998, Bindler 2003). In the southwestern U.S., atmospheric deposition rates have been calculated at the upper end of this range, 24 times higher than pre-industrial deposition rates (Heyvaert et al. 2000). We assume these pollutants represent similar pressures on marine populations as pollutants introduced through other mechanisms (e.g., urban runoff and dumping).

We evaluated only one indicator for atmospheric deposition: the mean concentration of sulfates monitored by the National Trend Network (NTN) of the National Atmospheric Deposition Program (Table H3). The NTN provides a long-term record of precipitation chemistry for sites located throughout the U.S. Data have been consistently collected weekly using the same protocols since 1994. Specific ions that are measured include calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}), potassium (K^{+}), sulfate (SO_4^{2-}), nitrate (NO^{3-}), chloride (Cl^{-}), and ammonium (NH^{4+})ions. These data are easily accessible via the NADP website: http://nadp.isws.illinois.edu/ntn/. This indicator of atmospheric deposition evaluated very high under all criteria categories (Table H3).

Volume of water displaced by vessel traffic. Andrews et al. (2013) evaluated three indicators of commercial shipping activity in the CCLME: port volume of cargo, number of vessel trips, and the volume of disturbed water during transit. Each of these indicators is certainly correlated with some aspect of commercial shipping activity. The port volume of cargo moved through ports along the Pacific Coast of the U.S. describes the total volume moving between ports, but this value does not give us any indication of how far shipping vessels are transporting these goods throughout the CCLME. This indicator is also probably not a relevant measure that management could use to “turn the dial” up or down. Increases or decreases to port volume may not have anything to do with the risk associated with ships striking marine mammals or increases to noise pollution off the coast.

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

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

Table H3. Summary of pelagic indicator evaluations across five primary considerations, seven data considerations, and six other criteria. Each criterion was scored 0, 0.5, or 1 depending on the level of literature support for that criterion. The numerical value that appears under each of the criteria groupings represents the number of evaluation criteria supported by peer-reviewed literature. For example, plume size has peer-reviewed literature supporting five out of five primary considerations criteria. . *Indicators in the top quartile; ** Promising indicators with gaps; unmarked indicators scored poorly and will not be considered further.

Indicator	Primary (5)	Data (7)	Other (6)	Summary Comments
Quantity				
Euphotic depth*	3.5	5	5.5	Euphotic depth is measured using light sensors. These are broadly recorded on multiple surveys across the CCLME and provide a good metric of depth and by extension the volume of water where primary production can occur.
Thermocline depth*	3.5	4	4	Thermocline is routinely derived from temperature measurements by CTD casts on numerous cruises. Thermocline provides the depth and by extension the volume of water defining favorable growth conditions for primary consumers.
Plume size (surface area)	5	5	4	Large river systems in the CCLME can produce plumes of water with lower salinity and higher turbidity, which favor certain fish species. These are largely confined to the Columbia River and Strait of Juan de Fuca and their roles for smaller systems outside this region is not well understood.
Current eddy size (surface area)	4.5	5	3.5	Several large eddies exist in the CCLME. However, the size, structure, and function of these systems as habitat is not fully understood or well-monitored.
Quality				
Dissolved O_2 (mg/l)*	4	6.5	5.5	Important indicator of growth potential in pelagic environments and of hypoxic conditions. Data are commonly measured during surveys across the CCLME. In some places, data may be limited in time or space.
Temperature (deg C)*	4.5	4.5	4	Temperature is an important variable predicting production and species distributions, and is widely measured on surveys and by satellite.
Turbidity*	3.5	6	4.5	Turbidity is strongly related to coastal sediment inputs and high local productivity, and can provide a predator refuge to small pelagic fish. Satellite measurements provide good spatial and temporal coverage across the CCLME.
Chlorophyll (mg/l)*	5	6	4	Good indicator of phytoplankton biomass and amount of energy fueling the ecosystem, satellite remotely sensed chlorophyll concentration data available system wide.

Forage fish biomass (aggregate)*	4.5	5.5	5.5	Changes in a single group may or may not be indicative of entire community. Most forage fish data are fishery dependent but new surveys are coming on-line.
Salmon smolt to adult survival rate*	4	6	5.5	Related to dominant modes acting over the coastal region, extensive historical records, perhaps best as a retrospective (lagging) indicator of historic ocean conditions.
Zooplankton biomass	3	7	4	Base of food web, fundamental component of CCLME, correlated with regime shift and climate change, can be used to estimate thresholds, several ongoing long-term datasets.
Euphausid biomass	3	6	4	Indicator of plankton biomass changes, critical link in marine food web, low counts and high patchiness in samples may increase variability, data availability as above.
Sardine & anchovy biomass	2.5	5	3	These two species are often the most abundant fish in pelagic waters and therefore are important indicators of the system's productivity. However, biomass can depend on factors other than productivity, and time series across the coast are limited.
Cetacean species status	3	6	3	Theoretically sound sentinel species, but high variability in data; low sample size and numerous coverage gaps; slow population response rate.
Salinity (ppt)	4	4	5	Extensive measurements of salinity have been made during cruises, but salinity is not a major source of variation in pelagic habitat characteristics. The exception occurs at large river plumes, where salinity variation can be important for some fish species.
Topographic upwelling (alongshore distance)	2.5	3	3.5	In several coastal areas, shelf and slope topography can facilitate upwelling, creating nutrient hotspots. Several of these sites have been identified, but the extent of these locations across the CCLME is not well documented and the time course of topographic upwelling events is therefore not broadly characterized.
Pressures				
Commercial landings of coastal pelagic fisheries*	4	7	4	Commercial landings represent the majority of removals for most species. This metric does not include discarded catch. Landings records from 1981 forward are available via http://pacfin.psmfc.org.
Atmospheric pollution*	5	7	5	The concentration of sulfate deposition measured by the National Atmospheric Deposition Program is a proxy for all chemicals deposited across the landscape. This dataset has been used in multiple publications as an indicator for atmospheric pollution.

Volume of water displaced by vessel traffic*	4	6.5	5	Similar to indicators that measure habitat modification caused by bottom-trawl fishing gear. Using the actual draft and breadth of each vessel times the distance travelled each trip provides a better estimate of the risk associated with the movement of shipping vessels through the CCLME.
Marine debris	3.5	4.5	4.5	Standardized sampling programs of measuring marine debris will be better than community groups, but it is unknown whether coastal measurements correlate with ocean measurements.

EVALUATION OF SEAFLOOR HABITAT INDICATORS

HABITAT QUANTITY

We evaluated three indicators used to measure the quantity of seafloor habitat (Table H4). These indicators include the areal extent (and distribution and abundance) of seafloor substrate substrata (e.g. rock, sand, mud, gravel), spatial patterns in substratum types, and metrics quantifying coverage of live corals and sponges. Areal extent of various substratum types ranked in the top quartile of our evaluation and is discussed here as the primary indicator of change in the quantity of seafloor habitat. In general, indicator data were collected in targeted high-priority areas (e.g. Sanctuaries, state waters) and were collected once per area. Consequently, data are unevenly distributed across the shelf and slope, and are challenging to use in time series analysis.

One seafloor habitat indicator ranked in the top quartile of our evaluation:

Extent of substratum type. The extent of seafloor substrate influences the distribution and abundance of demersal fishes (Love et al. 2002; Yoklavich et al 2000; Yoklavich et al. 2002; Anderson and Yoklavich 2007; Love et al. 2009; Pearcy et al. 1989; Stein et al. 1992). Consequently, substrate data are commonly used to infer fish distributions, and to regulate and monitor ocean uses (e.g. Rockfish Conservation Areas, Essential fish Habitats and Habitat Areas of Particular Concern, Marine Life Protection Act Marine Protected Areas).

There are few areas where analysis of change in substrate types over time would be meaningful at the scale of the California Current. Historic data exist at relatively low resolution (e.g., nautical charts etc.) for most of the CCLME, and more recent mapping surveys provide new substrate data in some areas.

The need to measure the extent of substrate types and the connection between substrate and fishes is easily understood by the public and managers. For instance, most people understand that the probability of catching certain species of fish changes in relation to bottom type. Managers can influence substrate through management of anthropogenic disturbances such as benthic trawling, construction and sediment deposition, and can use qualitative reference points inferred from the relative degree of association between substrate types and demersal fish species (see Love et al. 2002 and Allen et al. 2006 for reviews).

Areal extent of various substratum types is the primary indicator of change in the quantity of seafloor habitat (Table H4). The extent of substratum types influences the distribution and abundance of demersal marine fish and invertebrate species in the CCLME are significantly influenced by extent of substratum types (Love et al. 2002; Yoklavich et al 2000; Yoklavich et al. 2002; Anderson and Yoklavich 2007; Love et al. 2009; Laidig et al.; Pearcy et al. 1989; Stein et al. 1992). The relative degree of association of substratum type and demersal fish species is known (see Love et al. 2002 and Allen et al. 2006). Accurate information on the extent of substratum types (e.g., rock outcrops, boulder fields, mud and sand) can greatly improve predictive models of abundance/biomass of these organisms. The distribution and amount of substratum types are critical components in effectively regulating and monitoring ocean use off the U.S. west coast (e.g., EFH closures; California Marine Life Protection Act Marine Protected Areas), of which one intended result is to protect and improve seafloor habitats.

The extent of substratum types can be directly measured and the metrics are generally compatible throughout the CCLME. The accuracy of the metrics depends on the resolution of the data. Substratum types are interpreted from bathymetric and backscatter acoustic data, other geologic data, and ground-truthing from visual surveys using submersibles and remotely operated vehicles and from sediment grabs. Various derived indices are used to quantify substratum types. Resolution of these types of data varies regionally. In general, the spatial coverage and resolution of substrata data is greater within state waters compared to deeper, offshore areas. For example, the seafloor has been completely mapped with high-resolution multibeam sonar inside California's 3-mile jurisdiction. (i.e., high resolution data available in California state waters; much of federal waters has low resolution of interpreted substratum types; NMFS 2013). Historic data on extent of substratum types exist at relatively low resolution (e.g., nautical charts) for most of the CCLME; recent mapping surveys provide higher resolution data on the extent of substratum types in limited areas. As survey tools and technologies to map the seafloor advance, the resolution of the extent of substratum types improves.

An assessment of change in the extent of the substratum types would be meaningful only in a few relatively small areas, and would be difficult to evaluate on the scale of the CA Current. In addition, alterations in the sensitivity of survey technologies (e.g., improved sensors and geographic positioning) and in survey methods and interpretation of substratum types present challenges in discerning real change in the extent of seafloor substratum types. That said, change in the extent of substratum types could be a lagging indicator of impacts from sedimentation, scour, ocean engineering, and fishing. Change in the extent of substratum types could be a leading indicator of change in distribution and abundance of some species. An assessment of change in the extent of the substratum types would be meaningful only in a few relatively small areas, and would be difficult to evaluate on the scale of the CA Current.

The distribution and amount of substratum types, and their importance to communities on the seafloor, are easily understood by the public and often used by resource managers. For instance, most people understand that the probability of catching certain species of fishes changes in relation to seafloor substratum type. Managers can influence substrate impacts to seafloor substratum types through management of anthropogenic disturbances such as benthic trawling, construction, and sediment deposition, and can use qualitative reference points inferred from the relative degree of association between substrate substratum types and demersal fish species (see Love et al. 2002 and Allen et al. 2006 for reviews).

HABITAT QUALITY

We evaluated six indicators to measure the quality of seafloor habitats: dissolved oxygen, seafloor temperature, ocean acidification, terrain complexity, density of prey, and sediment accumulation (Table H4). Seafloor temperature, dissolved oxygen, and terrain complexity were judged to be the three primary indicators of change in quality of seafloor habitat, and are discussed in detail below.

Seafloor temperature. Temperature is a fundamental parameter monitored in oceanography, and the physiological response of demersal marine organisms to temperature is well studied. Change in temperature of seafloor habitats can reflect atmospheric-ocean processes such as upwelling on regional and local spatial scales and on seasonal, interannual, and decadal temporal scales (with potential for longer-term trends related to climate change). Changes in ocean temperature have been linked to shifts in population abundance and community structure of many demersal organisms. Regional reference points and time series of temperature are found in oceanographic databases for specific regions of the CCLME (e.g., CalCOFI; archives of various oceanographic institutions), and have been predicted at depth from oceanographic models (such as the regional oceanographic modeling system, ROMS).

Temperature can be directly and precisely quantified using well-established methods and standards set by the oceanographic community. Historically, ocean temperature was measured using bottle casts with reversing thermometers at fixed water depths, and is now measured continuously with widely available sensors on CTD (conductivity, temperature, depth) rosettes, moorings, and autonomous vehicles. There are ocean temperature data from the early 1950s, with spatial and temporal limitations. Our current understanding of CCLME oceanography can explain diel and seasonal variability in temperature, while variability on annual, decadal, and longer temporal scales is an active area of research. Change in temperature in seafloor habitats could be a leading indicator of latitudinal and depth-related shifts in distribution and abundance of demersal species (Perry et al. 2005, Dulvy et al. 2008).

Collecting data on ocean temperature is relatively cost-effective. Temperature and other key environmental parameters currently are measured during oceanographic cruises. Temperature sensors increasingly are being integrated into autonomous gliders and mooring systems, resulting in much broader collections of temperature data throughout the CCLME. The public can easily understand the impacts of changes in ocean temperature. Explanation of decadal-scale change in temperature patterns in the CCLME, the connection between regional and global patterns, and potential impacts from global warming are areas of active research.

Dissolved Oxygen. Fishes require DO for metabolic processes, and the physiology and biochemistry of respiration in fishes is well studied. The physical chemistry of dissolved oxygen in marine systems also is well studied, and oxygen concentration varies on a seasonal, annual, and decadal time scale. There is a growing literature on the response of marine fishes and invertebrates to varying degrees of hypoxia (oxygen deficiency) in the CCLME (Grantham et al. 2004, Chan et al. 2008, Keller et al. 2010). The onset of hypoxia on the continental shelf can reflect basin-scale fluctuations in atmosphere-ocean processes that alter oxygen content of upwelled water, the intensity of upwelling wind stress, and productivity-driven increases in coastal respiration (Chan et al. 2008). Regional reference points and time series of DO are found in extensive oceanographic databases for specific regions of the CCLME (e.g., CalCOFI; NODC World Ocean Database, archives of various oceanographic institutions), and hypoxia thresholds have been reported by Chan et al. (2008) and PISCO (Partnership for Interdisciplinary Studies of Coastal Oceans).

Dissolved oxygen can be directly and precisely quantified using well established chemical methods and international standards set by the oceanographic community. Dissolved oxygen in the ocean has always been measured broadly during research cruises, first from bottle casts at fixed water depths and now continuously with widely available sensors on CTD rosettes, moorings, and autonomous vehicles. There are data on DO in seawater from the early 1950s, with spatial and temporal limitations. Decreased DO (hypoxia) can be a leading indicator of stress and mortality of seafloor organisms. The public easily understands the need for oxygen by marine organisms. Explanation of recent decadal-scale change in the distribution of DO in the CCLME, its relationship to global patterns in DO, and potential impacts from global warming are not well understood by scientists or the public.

Terrain complexity or rugosity. Rugosity and other topographic metrics such as change in slope and bathymetric position index is an index of terrain complexity. Distribution and abundance of demersal fish species are influenced by the amount and level of terrain complexity, which can indicate size and extent of available shelter (O'Connell and Carlile 1993). Accurate measures of rugosity can improve predictive models of abundance/biomass of demersal organisms, and can be a critical component in effectively regulating and monitoring ocean use in the CCLME (e.g., EFH closures; California Marine Life Protection Act Marine Protected Areas). Change in rugosity could be a lagging indicator of impacts from sedimentation, scour, ocean engineering, and fishing, and could be a leading indicator of change in distribution and abundance of some species. An assessment of change in rugosity would be meaningful on a relatively small spatial scale, and would be difficult to evaluate on the scale of the CA Current.

Rugosity can be derived from bathymetry (continuous measures of depth) or interpreted from visual observations or side scan sonar data. Accuracy is dependent on the level of resolution of the underlying data. Historic data from which to derive rugosity are available at relatively low resolution. There is comprehensive coverage of multibeam bathymetry on the shelf and upper slope within state waters in California. Much less information is available in deeper offshore areas of the CCLME. There are few (if any) relatively small areas in which rugosity can be derived from data collected over time (and none that would be meaningful on the scale of the CCLME). Rugosity and the relationship between level of complexity and distribution of demersal organisms are easily understood by the public and often used by resource managers.

PRESSURES

The first CCIEA report (Andrews et al. 2013) examined a number of anthropogenic threats. The areal extent of bottom contact fishing gear was the only indicator appropriate for spatial and temporal analyses of pressures to seafloor habitat. Other potential metrics, such as artificial structures, were regarded as poor indicators of impacts to habitat because of their small footprint. In addition, some species appear to be attracted to artificial structures (Love et al. 2005). Hence, it remains unclear whether such structures should be viewed as true pressures or as habitat improvements.

The extent of bottom contact gear. Areal extent of bottom trawl fishing is the priority indicator of anthropogenic pressure on seafloor habitats in the CCLME. Due to the size and mass of this gear, and because several parts of the gear are in direct contact with the ocean floor, bottom trawls can physically remove, disturb, or harm rocky outcrops, corals, sponges, eelgrass beds, and other components of seafloor habitats. This type of fishing gear can significantly alter the extent and function of physical and biogenic substratum types by reducing terrain complexity and structure (NRC 2002).

Bottom trawling activity in the CCLME is conducted primarily by the Pacific Coast groundfish fishery, which harvests over 90 species, and by smaller state-managed fisheries targeting shrimp, prawns, and California halibut. Mainly due to restrictions on gear configurations and size, most bottom trawling activities currently occur on soft, unconsolidated sand and mud and adjacent to hard bedrock outcrops on the continental shelf and upper slope. In consultation with treaty tribes, management of the bottom trawl fishery is executed by NMFS, the three west coast states, and the PFMC, and comprises a complicated matrix of stakeholders, seasons, and spatial limitations. The effects of trawling vary by substratum type, and the Pacific Coast Groundfish Fishery Management Plan (PFMC 2011) includes a risk assessment of bottom trawling (and other gears), and a sensitivity index and recovery rates for a variety of components of seafloor habitat. Although bottom trawling occurs throughout the region out to about 1,300 m water depth, many areas within the CCLME have been closed to bottom trawling in order to protect seafloor habitat as well as to recover overfished species. In addition, bottom trawling is prohibited entirely in state waters (to 3 nmi) off Washington and is severely restricted off California.

Change in the areal extent of bottom trawl fishing is variable in space and time, and has been evaluated as part of the 5-year review of Pacific coast groundfish essential fish habitat (NMFS 2014). Change in the areal extent of bottom trawl fishing could be a lagging indicator of management strategies, declining fish stocks, redistribution of fish stocks due to ocean conditions, or economic dynamics. Change in the extent of bottom trawl fishing, particularly due to spatial fishing closures, could be a leading indicator of change in 1) distribution and abundance of species that are targeted or removed as bycatch in the trawls, 2) condition of seafloor habitat components, and 3) changes in biodiversity, productivity, and fish yield of the area. In general, the public understands the extent of bottom trawling and potential resultant impacts to seafloor habitats.

Table H4. Summary of seafloor indicator evaluations across five primary considerations, seven data considerations, and six other criteria. Each criterion was scored 0, 0.5, or 1 depending on the level of literature support for that criterion. The numerical value that appears under each of the criteria groupings represents the number of evaluation criteria supported by peer-reviewed literature. For example, areal extent of substrate habitat type has peer-reviewed literature supporting four and a half out of five primary considerations criteria. *Indicators in the top quartile; ** Promising indicators with gaps; unmarked indicators scored poorly and will not be considered further.

Indicator	Primary (5)	Data (7)	Other (6)	Summary Comments
Quantity				
Extent of substratum type (km2)*	4.5	5	4.5	Maps of substratum type exist coast-wide for the CCLME; resolution for substratum data varies regionally; data for state waters is mostly high-resolution, while data for most federal waters is low resolution (NMFS 2013)
Live Coral/Sponge (metrics: density, % cover, diversity)	3.5	3	4.5	The occurrence of live coral/sponge is recorded from bycatch from regional bottom trawl surveys of "trawlable" habitats or during direct visual surveys of habitats "suitable" to corals and sponges (those requiring/preferring high relief, hard substrate). Direct count visual surveys occur throughout the CCLME for a variety of purposes. Metrics of relative abundance are often quantitative, but some records compiled for the region are presence only.
Spatial pattern of substratum types (e.g., number of patches)	1.5	1	1.5	Spatial pattern of substratum type are rarely quantified and reported. Reference points have not been established, and depend on high-resolution multibeam data in order to derive meaningful metrics as a habitat indicator. Multibeam data are localized except for broad coverage in CA and OR territorial seas
Quality				
Dissolved O2 (ml/l, mg/l, µmol/l, µmol/kg, % saturation)*	4.5	5.5	5.5	Regional reference points and time series of dissolved oxygen (DO) are found in extensive oceanographic databases for specific regions of the CCLME. DO measured in seawater from discrete samples, but increasingly measured continuously with sensors on CTD rosettes, moorings, midwater and bottom trawls and underwater vehicles.
Bottom temperature (deg C)*	4.5	5	4	One of the most commonly measured environmental parameters. Regional reference points and time series of temperature are found in extensive oceanographic databases for specific regions of the CCLME. Historically measured from bottle casts with reversing thermometers at fixed water depths, but now uniformly measured continuously with sensors on CTD rosettes, moorings,

				midwater and bottom trawls and underwater vehicles.
Ocean Acidification (pCO2, TCO2, alkalinity, calcite & aragonite saturation state)**	4	3.5	2.5	Regional reference points exist for pH, pCO2, and aragonite saturation state as well as other OA-relevant parameters. There are historical databases for the CCLME and various overlapping and unique academic and institutional archives; NOAA PMEL is one regional and global repository. Note – high-precision pH measurements in deep water can be difficult to achieve due to pressure changes.
Terrain complexity (e.g., rugosity)**	3.5	4.5	1.5	Rugosity can be derived from or interpreted from bathymetry, side scan sonar or visual surveys; Historic data for deriving rugosity are available at relatively low resolution, and currently the necessary comprehensive coverage of multibeam bathymetry exists only on the shelf and upper slope within state waters in California and a portion of OR
Density of Prey spp (# or biomass/km2)	3	4	5.5	Densities for megafaunal species (fishes and invertebrates) are measured during coast-wide bottom trawl surveys or local direct count visual surveys. Regional trawl surveys and direct count visual surveys occur throughout the CCLME for a variety of purposes and pelagic prey are sampled locally via several surveys.
Sediment accumulation rates (g cm-2/yr-1 or mm/yr)	3	4.5	2.5	Historical data exists, but has spatial and temporal limitations. There are reference sites along the US Pacific Coast that have been sampled repeatedly over decades.
Pressures				
Extent of bottom trawling (km2)*	4	6.5	4	Coast-wide estimates of distance trawled by habitat type were generated by Bellman and Heppell (2007) based on logbook data on each individual tow and GIS seafloor habitat maps. These estimates are available between 1999 and 2004 and have been updated through 2010 as part of the NMFS Groundfish synthesis (NMFS 2013).

SUITE OF INDICATORS FOR HABITAT IN THE CALIFORNIA CURRENT

The goal of this report was to determine a suite of indicators sufficient for monitoring habitat conditions across the California Current. We identified 33 high priority indicators to evaluate habitat status and trends across freshwater, estuary and nearshore, pelagic, and seafloor habitats (Table H5). This suite is a balancing act between the need for a relatively small indicator set (Levin et al. 2009, Levin & Schwing 2011), and the importance of adequately representing the complexity of habitat conditions that support

Table H5. Priority indicators of freshwater, estuarine/nearshore, pelagic, and seafloor habitats.

Habitat	Attribute	Spatial analysis	Trend analysis
Freshwater	Quantity	River discharge	River discharge
		% of network accessible	
	Quality	Temperature	Temperature
		Riparian condition	
	Pressures	% agriculture	% agriculture
		% developed/impervious	% developed/impervious
			Number of dams
Estuary/nearshore	Quantity	SAV extent	SAV extent
		Estuary wetland area	River discharge
		Benthic substrate extent	Sea level rise
	Quality	Temperature	Temperature
		Dissolved O_2	Dissolved O_2
		Nitrogen: Phosphorus	Nitrogen: Phosphorus
			Turbidity
			Chlorophyll a
	Pressures	% agriculture	% agriculture
		% developed/impervious	% developed/impervious
			Beach closures
Pelagic	Quantity	Euphotic depth	Euphotic depth
		Thermocline depth	Thermocline depth
	Quality	Surface temperature	Surface temperature
		Turbidity	Turbidity
		Chlorophyll a	Chlorophyll a
			Dissolved O_2
			Total forage fish biomass
			Marine survival of salmon
	Pressures	Atmospheric pollution	Atmospheric pollution
		Ship displacement volume	Ship displacement volume
			Commercial fishery landings
Seafloor	Quantity	Substratum types	--
	Quality	Temperature	Temperature
		Dissolved O_2	Dissolved O_2
		Rugosity	
	Pressures	Areal extent of bottom trawling	Areal extent of bottom trawling

the huge diversity of aquatic life on the Pacific Coast. When examined for particular species or particular habitat features at smaller spatial scales, indicators not represented on this list may be of greater relevance. Nevertheless, following the goal of representing the state of habitat for the entire California Current ecosystem, this suite represents the most appropriate, scientifically based, and well-monitored set of habitat indicators.

These indicators also relate to key linkages identified in our conceptual model (Fig. H2). In addition to the status of habitats within each habitat type, the conceptual model points to several important linkages worth tracking: ocean drivers, anthropogenic pressures, cross-habitat linkages, species responses, and human wellbeing. When categorized by these relationships, the list of priority indicators does a relatively good job in linking with other ecosystem components (Table H6). All indicators were specifically designed to capture habitat status, and the metrics listed are key examples of habitat elements of key importance within habitat types. Cross-habitat connections are most relevant for estuary/nearshore and pelagic habitats, and our list of priority indicators provides several good examples for cross-habitat linkages for estuary systems. However, indicators describing other habitat linkages were not as highly prioritized.

Table H6. How priority indicators track linkages to other elements in the conceptual model for Habitat (Fig. H2). Italicized gray terms indicate potential indicators not in the priority list.

	Freshwater	**Estuary/nearshore**	**Pelagic**	**Seafloor**
Habitat status	River discharge Riparian condition	SAV extent Benthic habitat extent Wetland area	Euphotic depth Thermocline depth Temperature Chlorophyll a	Substratum types Rugosity
Habitat linkages	*Marine-derived nutrients*	River discharge Water storage Sea level rise	Turbidity	*Sedimentation rate*
Climate and ocean drivers	River discharge Temperature	River discharge Temperature Dissolved O_2 Sea level rise	Thermocline depth Temperature Dissolved O_2	Temperature Dissolved O_2
Anthropogenic pressures	% agriculture % developed Number of dams	% agriculture % developed Nitrogen:phosphorus Water storage	Atmospheric pollution Ship displacement volume	Areal extent of bottom trawling
Species responses	% of network accessible	Wetland area SAV extent	Forage fish biomass Salmon marine survival	*Biogenic habitat*
Human wellbeing	River discharge	Beach closures	Commercial landings	*Groundfish landings*

Indicators sensitive to key climate and ocean drivers received high priority for all habitat types, particularly because these metrics generally represent the best time series. Likewise, we specifically developed indicators to capture anthropogenic pressures on habitat. We also chose indicators to be relevant habitat quantity and quality metrics for living marine resources. However, some indicators are more biologically relevant than others because they specifically examined biogenic components (e.g., submerged aquatic vegetation, salmon marine survival) or targeted a species response (e.g., % of watershed accessible for salmon migrations). We also chose several habitat metrics that people directly benefit from: i.e., water supply, beach use, and the commercial harvest.

Aquatic habitats are of course defined in part by the species that use them. Individual species have particular preferences that would be represented as ranges in habitat indicators; likewise, particular species would be expected to have variable responses to anthropogenic pressures. Hence, habitat indicators need to be tailored to species or suites of species with similar habitat preferences. Pressure indicators will likewise need to be examined in light of how pressures affect habitat for these species. Nevertheless, there may exist thresholds beyond which variation in habitat quantity or quality and concomitant anthropogenic pressures on habitat affect a broad suite of species. For example, hypoxic conditions (<5 mg/l) appears to have negative effects on a broad range of demersal fish species (Keller et al. 2010). Hypoxia represents an extremely well-studied example, and thresholds such as these are difficult to assign for many habitat metrics. Therefore, we expect additional efforts required to improve linkages between habitat indicators and species or suites of species. Explicit links between habitat and the living marine resources that depend upon habitat should improve the ability of assessments to inform habitat conservation actions as one major set of management strategies.

INDICATORS AND ADAPTIVE MANAGEMENT

Like all adaptive management programs, we recognize that as additional knowledge, know-how, and management questions arise, some indicators may change in priority for the CCIEA. For example, numerous seafloor researchers have been interested in the habitat roles of biogenic habitat (e.g., sponges, deep-sea corals, and sea pens) but data is currently insufficient for mapping or tracking availability of these habitats (NMFS 2013), and work is just developing for determining their importance for demersal stocks (Tissot et al. 2006). As findings accumulate and better sampling methods and data become available, and if questions concerning the impacts of ocean acidification on biogenic habitat were to rise in importance, we can reevaluate the indicators with the new information which may result in a higher priority for monitoring seafloor biogenic habitats.

In this respect, our indicator selection process can help shed light on priority information and data gaps. Priority indicators are those which have very good scientific support as represented by primary considerations, as well as good data quantity and quality as represented by data considerations. Lower priority indicators exist because of both poorer primary and data considerations (Fig. H3). However, those indicators with high primary considerations but low data considerations could be considered good indicators with poor data, and therefore targets for improvements in monitoring. Examples of these types of indicators were:

- Freshwater: amount of large woody debris, index of biotic integrity scores
- estuary/nearshore: areal extent of salinity zones in estuaries, nonnative plants and animals
- Pelagic habitats: plume size, eddy size
- Seafloor: Areal coverage of biogenic habitat, carbonate chemistry

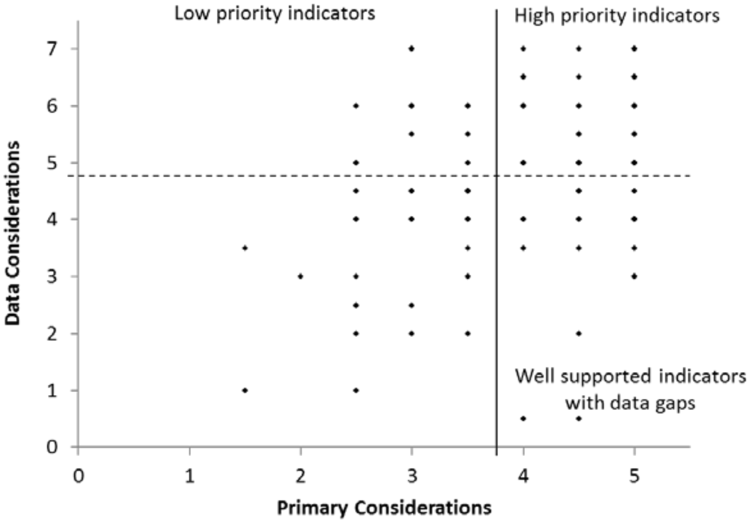

Figure H3. Primary and data considerations of habitat indicators, with lines indicating upper 25th percentile cutoffs. High priority indicators are in the upper right quadrant, while indicators with strong scientific support but lower data considerations are in the lower right hand corner.

Adaptive management requires decision points for re-evaluation of science and management programs over time. As noted in Levin et al. (2009), IEAs include multiple opportunities for adaptive management. This report constitutes an initial screening of potential habitat indicators, and recognizes that some indicators may deserve more attention due to data limitations. Better technology, additional research, and expanded monitoring should help make these better indicators. Where possible, new habitat assessment efforts should incorporate measurements for these promising additional indicators. The best opportunities for re-evaluation of indicators should occur during status and trends updates, when determinations over improvements to data considerations can be made following the same methodology used in this report.

NEXT STEPS

The main purpose of using ecosystem indicators is to evaluate ecosystem health or function (Levin & Schwing 2011). At the scale of the California Current, this question can have both spatial and temporal relevance. We anticipate that dividing status and trend analysis into mapping products and trend analyses will facilitate improvements for our understanding of habitat conditions for the CCLME's living marine resources. This will improve our ability to address where habitat is in good condition or impaired, as well as track how habitat elements are changing over time. Habitat data are well known for their information gaps, and the indicators we have selected are no exception. In this respect, selection of these indicators and tracking their status and trends can also shed light upon where additional information needs to be collected. As we synthesized indicators, we noted particular metrics for which we anticipate additional development time for analysis or data synthesis.

In addition to the indicator development outlined in this report, IEAs examine status and trends of indicators, risk analysis, and management strategy evaluation. Our assessment of indicators for the Habitat Component has highlighted important aspects that make analysis of status and trends different from other Ecosystem Components: habitat is by nature a spatially variable feature, habitats are interconnected, and changes in habitat can occur at very different time scales than living marine resources. These principles will need to be accounted for in future IEA products. Tasks for addressing these issues as part of future research include:

1) Developing a spatial framework connecting habitat types in order to facilitate spatially explicit evaluation of status and trends. Previous status and trends efforts for the CCIEA lack spatial variation, which are important to address region-specific priorities for ecosystem management.
2) Using this framework to build risk analyses relevant at multiple spatial scales (e.g., watersheds, estuaries, ecoregions), which track both local anthropogenic habitat modifications and impacts from climate change.

3) Using spatially referenced habitat indicator sets to update management strategy evaluations. Thus far, much of the ecosystem modeling has incorporated habitat changes in qualitative ways, and these models will likely be improved by quantitative measures of habitat and their effects on species groups and their interactions.
4) Improving linkages between habitat and human wellbeing beyond indicator analyses. Following from research in other marine systems (Kittinger et al. 2012), explicitly incorporating the benefits of habitat conservation to socio-economic systems will improve our ability to determine the relevance of habitat to people in the context of the CCLME. These linkages are best incorporated into management strategy evaluations of habitat conservation.

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HUMAN DIMENSIONS OF THE CCIEA

A SUMMARY OF CONCEPTS, METHODS, INDICATORS, AND ASSESSMENTS

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OVERVIEW

A conceptual model of the California Current Large Marine Ecosystem (CCLME) socio-ecological system highlights the "social" within the socio-ecological system and demonstrates that any particular management strategy can affect human wellbeing through at least two major pathways: through alterations in environmental conditions, which in turn affect human wellbeing, and through direct effects on human wellbeing. In addition to broad conceptualizations of the coast-wide system in both natural and social terms, and discussions of relevant social science approaches and frameworks, we include 5 major indicator efforts within the CCLME. These indicators cover levels of human coastal community vulnerability, vessel- and port-level fisheries diversification trends and effects, "personal use" of fisheries as a preliminary proxy for possible subsistence practices among commercial operators, the relationship between water supply and agricultural production in Central California, and a survey of marine-oriented recreational expenditures.

EXECUTIVE SUMMARY

In this chapter, we focus on the "human dimensions" of the California Current's coupled socio-ecological systems. Human dimensions include archaeological and historic heritage, contemporary demographic patterns such as population growth and migration, individual and community behaviors, cultural values and cultural trends, social relationships and social movements, political and economic systems, institutions and governance, and perhaps most importantly in this context, the many ways that humans are connected to the environment. This chapter also serves to introduce research relative to human wellbeing and, accordingly, the "social" in the socio-ecological system of the California Current.

Human wellbeing is linked to the California Current, as a large marine ecosystem, in a variety of ways. We provide brief synopses of the multiple and diverse human connections of several focal ecological components to human wellbeing published in social science literatures. Focal components included here are: groundfish, marine mammals, seabirds, forage fish, salmon, and habitat.

Prior to describing our relevant indicators, we include discussions of social science approaches to some of the human dimensions of the California Current Large Marine Ecosystem (CCLME). These approaches include frameworks aimed at capturing some the cultural connections to the CCLME, economic frameworks, social indicators and human well-being frameworks, and political ecology as a holistic approach to human-environment interactions.

In terms of CCLME human dimensions indicators, community vulnerability indices highlight both sociodemographic vulnerability and marine and fisheries-specific vulnerabilities at the community-level. Economic data at the vessel and port level provide an indicator of economic diversification, which in turn demonstrates that fisheries income variability is reduced on average if individuals diversify their income by participating in several different fisheries, though diversification in general is in decline. A personal use indicator provides information on port location and species of interest for subsistence and non-commercial harvests among commercial operators. Notably, an inland CCLME-relevant human dimensions indicator for central California is provided by research on inland agricultural activity and water use. This research indicates that reduced irrigation water supply reduces the demand for farm labor and the production of some crops over the course of a 31-year study period, and that labor demand and crop output may have become more sensitive to changes in water supply. Lastly, data from a recently completed survey will be used in estimates of West Coast consumptive and non-consumptive ocean recreational activities.

The indicators described toward the close of this section are based on available data collected and organized by the Northwest and Southwest Fisheries Science Centers, along with their research partners, and reflect the analytical work currently underway within the human dimensions and economics programs at these science centers. Many of the datasets used in developing these indicators are updated annually and therefore offer time series analysis possibilities within future iterations of the IEA. Finally, the described work of the Social Wellbeing Indicators for Marine Management (SWIMM) project is organized around developing a more refined definition of "human wellbeing" and, accordingly, improving upon and re-evaluating relevant social indicators.

DETAILED REPORT

INTRODUCTION: INTEGRATING HUMAN DIMENSIONS INTO THE CCIEA

WHAT ARE “HUMAN DIMENSIONS” AND “HUMAN WELLBEING”?

“Human dimensions” refer to all aspects of human life across time and space, including demography, behavior, cultural values, social relationships, political and economic systems, institutions and governance. In this chapter, we focus on the “human dimensions” of the California Current’s coupled socio-ecological systems. A variety of social science disciplines are used to study these different aspects of the human condition, such as anthropology, economics, sociology, political science, psychology, and geography. The contributing authors offer only a subset of social science perspectives on the human dimensions of the CCIEA.

In this chapter we discuss the concept of “human wellbeing.” Human wellbeing gained prominence as an area of interest in environmental science, policy, and management via the 2005 Millenium Ecosystem Assessment (MA) and the ecosystem services frameworks. Here, we generally use “human wellbeing” to mean happiness, health, and quality of life, both for individuals and communities. A working group of social scientists advising the Social Wellbeing Indicators for Marine Management (SWIMM) project (described in a section below) developed a more refined definition of “human wellbeing” that draws from multiple literatures and is intended to clarify its meaning in the context of ecosystem-based management:

> *Human wellbeing is a state of being with others and the environment, which arises where human needs are met, where individuals and communities can act meaningfully to pursue their goals, and where individuals and communities can enjoy a satisfactory quality of life.*

WHAT IS “SOCIAL” IN THE SOCIO-ECOLOGICAL SYSTEM?

A socio-ecological systems approach is a holistic view of interacting ecological and social phenomena in their environments, which create important functional connections across spatial and temporal scales (Berkes 2011). The diagram of the CCIEA socio-ecological system illustrates how human wellbeing is related to the marine, coastal, and associated upland environments (Fig. HD1). These relations are dependent on qualities of both the biophysical environment and the human social system. Like the natural environment, human society comprises multiple interrelated components and forces. Human wellbeing in general – including even those aspects related to environmental conditions – is always mediated by broad social forces, local social systems, and human activities.

Broad social forces – such as population growth and settlement patterns, national and global economic and political systems, historical legacies, dominant cultural values, and class systems – constrain or enable local social systems and human activities in ways that directly or indirectly affect human wellbeing. Likewise, local social systems that vary geographically and across different social groups – such as state and local laws and policies, regional economies, local institutions and social networks, local social hierarchies, diverse cultural values and norms, the built environment, and other particularities – affect human wellbeing directly or indirectly, and constrain or enable human activities related to the natural environment. Such human activities might include, for example, fishing, farming, mining, recreation, environmental research, education, activism, restoration, and resource management. Such activities generate benefits for humans, and they are also how humans affect the natural environment, in this sense often called pressures. However, the ways in which these activities, benefits, and pressures directly or indirectly affect human wellbeing and its myriad dimensions (Fig. HD1) depend on the social attributes and contexts of the humans in question – i.e. the broad social forces and local social systems in which they are embedded.

For example, in order to enjoy the nutritional and cultural wellbeing that comes with harvesting and eating Dungeness crab, a Washington State resident must be able to access the crab fishery, know how to harvest and cook crab, and possess positive cultural values toward harvesting and eating crab, among other qualities. These requirements are mediated through particular governmental, economic, social, and cultural conditions such as state fishing regulations, the affordability of fishing, an accessible launch site, and community-based cultural practices, as well as through environmental conditions such as the quality and availability of crabs themselves. Similarly, human wellbeing derived from working as crew on a trawler, watching seabirds, kayaking, conducting oceanographic research, or any other environment-related activity will be mediated by a complex matrix of social conditions, connections and capabilities.

The Integrated Ecosystem Assessment (IEA) is a tool to track the condition of the ecosystem (including people) under changing environmental conditions and management strategies. There are multiple and interrelated social and natural factors that can affect human wellbeing. Note that with respect to environmental policy and management specifically, any particular strategy can affect human wellbeing through at least two major pathways: 1) policy and management can affect environmental conditions, which in turn affect human wellbeing; and 2) policy and management can directly affect human wellbeing, such as through the nature of the political process, and how management actions affect people's access to resources. The environmental social sciences devote considerable attention to the latter pathway, i.e. how conservation and resource management directly affect people, because this can significantly affect major areas of wellbeing such as sense of control and certainty, social relationships, livelihoods, and equity. It is important to attend to both of these and other pathways to wellbeing – and not only to the connection between the natural environment and wellbeing – in order to understand the social dynamics and consequences of environmental policy and management.

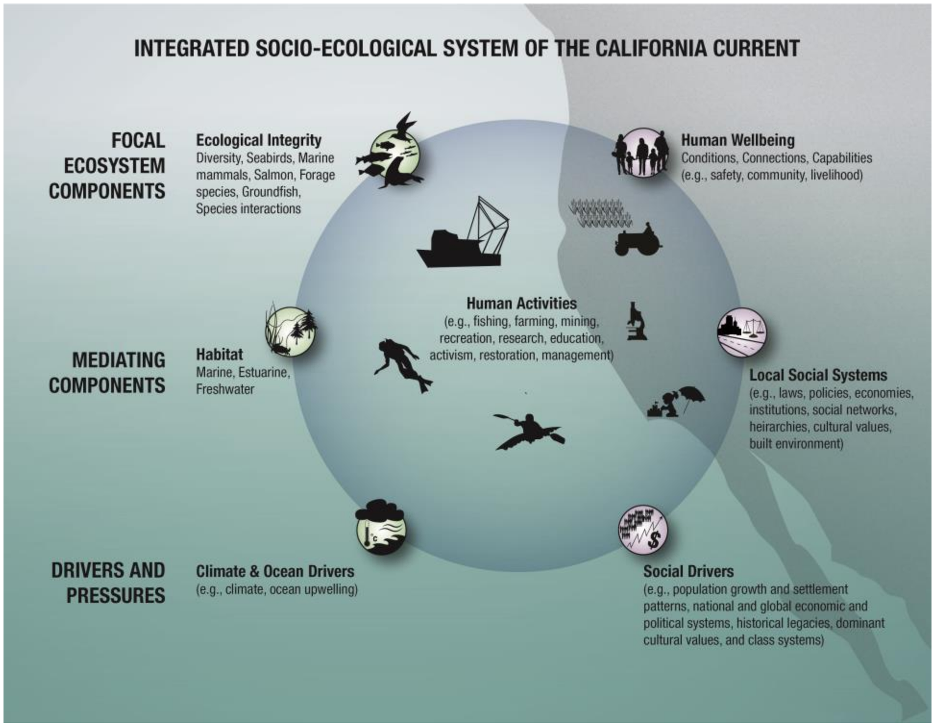


Figure HD1. Conceptual model of the California Current socio-ecological system.

HUMAN WELLBEING CONNECTIONS TO FOCAL ECOLOGICAL COMPONENTS

Each ecological component of the socio-ecological system contributes to human wellbeing in multiple ways. Previous phases of the California Current IEA captured many of the commercial benefits of some focal ecological components (e.g. salmon and groundfish). In this section, we provide brief synopses of the multiple and diverse human connections of each focal ecological component to human wellbeing published in social science literatures. Focal components included here are: groundfish, marine mammals, seabirds, forage fish, salmon, and habitat.

GROUNDFISH

Groundfish are linked with human wellbeing in a number of ways. They provide food for domestic consumption and export, and support a diverse commercial fishery that encompasses the length of the Pacific Coast and involves many species and gear types. Groundfish are also important species for recreational and subsistence fishing. Groundfish activities contribute to job satisfaction, “quality of life”, local ecological knowledge, and also play a role in building community capacity, for example through fishing cooperatives, risk pools, gear innovation, education, and training. Groundfish regulations have affected the spatial distribution as well as volume of groundfish activity, with associated effects on human wellbeing. To rebuild overfished rockfish stocks, the Total Allowable Catch (TAC) in groundfish fisheries has been reduced for both rebuilding and targeted stocks. Other management changes to groundfish fisheries include area closures and gear restrictions to reduce bycatch, heightened monitoring (observer programs, electronic vessel monitoring systems) and – for the trawl sector – an industry-funded buyback, prohibition of bottom trawling in Essential Fish Habitat, and a catch share program that enhances individual accountability for reducing bycatch and individual flexibility to harvest target species. Groundfish fisheries also face issues in common with other fisheries (e.g., graying of the fleet, aging port infrastructure). Groundfish species also indirectly affect human wellbeing through ecological interactions (e.g. as predators, competitors or prey) with culturally and economically important marine species (e.g. forage fish, salmon, seabirds, and marine mammals). Community involvement in restoration (e.g. derelict fishing gear clean-up) and conservation (marine protected areas, or MPAs) also contributes to emerging social networks, and increase engagement with decision-making.

MARINE MAMMALS

Marine mammals have social, cultural, economic, and value to humans. Some marine mammals contribute to sense of place and serve as place-based icons in coastal areas. Marine mammals such as sea otters, pinnipeds and whales are culturally important to many coastal communities’ way of life, including as subsistence resources for indigenous communities. Interactions with marine mammals occur at aquaria, zoos, and at sea where marine mammals can be experienced in their natural environment. These activities contribute to employment and income in coastal economies, as well support opportunities for marine science education. Marine mammal education and conservation activities can function to increase public knowledge and build communities with shared values. Several studies document willingness to pay (WTP) for marine mammal viewing and existence. In some cases, marine mammals have led to decreases in human wellbeing through competition and trophic interactions with fisheries, gear and property damage, and loss of catch in commercial and recreational fisheries, and predation on species of concern (e.g. listed salmonids). A wide range of human activities (e.g., fisheries, tourism, shipping, military sonar, seismic surveys associated with offshore oil and gas exploration) is regulated to reduce injury or mortality to marine mammals.

SEABIRDS

Seabirds have social, cultural, and economic value to humans. Some seabirds contribute to sense of place and serve as place-based icons in coastal areas. Interactions with seabirds occur at aquaria, zoos, and in their natural environment along coastal areas or at sea. These activities often contribute to employment and income in coastal economies, as well support opportunities for marine science education. Seabird education and conservation activities can function to increase public knowledge and build communities with shared values. To the north of the California Current, in Canada and Alaska, seabird eggs are harvested for subsistence by some indigenous communities, a practice tied to traditional ecological knowledge. Some migratory birds found seasonally in the California Current are harvested elsewhere on their migratory circuit (e.g. shearwaters by Maori communities). Seabirds can be effective indicators of the condition and health of marine systems, pollution levels, fish stock health and management, contaminants, and climate variability. In some cases, seabirds can influence human wellbeing negatively through competition for resources and trophic interactions with fisheries (in particular, predation on Pacific salmon in the Columbia River basin). Some human activities (e.g., fisheries, tourism) are regulated to reduce injury or mortality to seabirds.

FORAGE FISH

Northern anchovy, Pacific sardine, Pacific herring, and other forage fish have social, cultural, and economic value to humans. Northern anchovy was the second fish species to come under management in the United States. Northern anchovy and Pacific sardine support recreational, subsistence, and live-bait fisheries, and are especially important in the Southern areas of the California Current. Forage fish commercial activities spur many off-the-dock socioeconomic benefits (e.g., supporting local processing, transport and storage industries; creating jobs; and shaping how families structure their time throughout the year). Additionally, the skills, job satisfaction and professional identity of forage fisheries contribute to human wellbeing. Forage fish also contribute to sense of place in some coastal areas where these species have played important roles in shaping community economies and heritage (e.g. “Cannery Row” named for the sardine canning factories in Monterey, California). Forage fish, so-called for their importance as lower trophic level food to higher trophic level species, also have indirect social values owing to their role as food for iconic species such as salmon, seabirds, and marine mammals, which people also value. In addition to food for human consumption, anchovy and sardine are also processed as feed for commercial aquaculture and livestock.

Less commercially important species of forage fish contribute to wellbeing through their role as subsistence food for diverse communities along the Pacific Coast, and as traditional and ceremonial foods for indigenous communities. For example, Pacific herring is a culturally important forage fish for Northern Pacific Coast indigenous communities, particularly in the Northern Salish Sea and Vancouver Island areas. The whole fish and its eggs (e.g. roe on kelp) are used. Herring figures prominently in the origin stories and oral histories of Northern coastal cultural groups. Knowledge about harvesting techniques, locations and processing comprises part of the cultural legacy of these forage fishes' importance to coastal communities. Systems of rights, ownership, and harvesting patterns have been in place to maintain sustainable traditional harvests. Similarly, eulachon –also referred to as "ooligan"– has historically been used by many Pacific Northwest coastal indigenous communities as food, medicine, material, and trade. The nutritional content of eulachon is high in vitamins A, E, K and fatty acids, as well as calcium, iron and zinc. Eulachon has declined dramatically in the Pacific; the Southern population is listed as a threatened species under the ESA.

SALMON

Salmon play a central role in the social organization, diet, culture, ceremonial and spiritual practice, cultural identity, and economy of coastal indigenous communities of the California Current. Salmon are also important to non-indigenous residents of the larger region as food, regional identity, and an important economic resource. Historically, fluctuating seasonal runs of salmon helped determine the location of Native American villages, where sophisticated salmon harvesting, drying, and storage technologies developed, coupled with complex and cooperative resource ownership and access systems. Contemporarily, wild and hatchery salmon remain an integral part of the fishing economy, and are used for commercial and recreational fisheries and subsistence food throughout the California Current. Marine mammals that people value also prey on salmon; for example, Chinook salmon is a primary prey species for Southern resident killer whales, a culturally iconic marine mammal. Public awareness and concern for salmon protection and recovery (largely owing to reduction in salmon populations resulting from hydropower production, farming, ranching, fishing, logging, and municipal and industrial water use and supporting infrastructure (e.g., dams, water storage and transport systems, hatcheries, among other activities and pressures)) has grown. The growth in concern and awareness is reflected in participation in river restoration, educational programs, and stewardship organizations. Conflicts among competing uses are exacerbated when habitat conditions are particularly limiting (e.g., the current California drought). A large research establishment conducts research relevant to the understanding, management and improvement of salmon fisheries and other natural resources that are socio-ecologically linked to salmon and their habitat, such as agriculture and forestry. The complex challenge of salmon recovery has required new forms of social organization and cooperation, and has also engendered passionate debates among diverse communities in the region who are grappling with how to ensure that salmon, fishing, and other resource-based livelihoods can survive in an increasingly globalized economy and urbanizing landscape.

HABITAT

Habitats provide the matrix through which ecosystem interactions occur. Human wellbeing is therefore influenced directly and indirectly both by the habitats and the organisms they influence, as well as by their general characteristics that contribute to senses of place (rocky shorelines, intertidal biodiversity, sandy beaches, tide flats, the open ocean, etc.). People benefit from habitat directly and indirectly from the fisheries they support, as well as aesthetic, recreational, cultural, spiritual, and scientific reasons. The CCIEA focuses on four major habitat types: freshwater, estuarine/nearshore, pelagic, and seafloor. Freshwater habitats are crucial not only for their role in provisioning a diversity of species important to human wellbeing (e.g. fish, marine mammals, seabirds), but also for supporting a wide range of benefits to people, including water supply, land for agriculture or development, transportation, recreation, energy generation, cultural resources, and commercial and sport fisheries. Estuary and nearshore habitat directly and indirectly support fisheries and aquaculture, and they also provide a number of other benefits to people as sites for transportation, alternative energy infrastructure, waste disposal and water diversions, and recreation. In the pelagic realm, fisheries and transport are the primary human benefits. Seafloor habitats support important fisheries, providing food, income and recreation for numerous individuals and coastal economies. As well, seafloor habitats are sites for important human activities—undersea cables, oil and gas exploration and infrastructure– to name a few. Habitat is often the focus of management efforts because natural resources are generally associated with specific types of habitat (e.g., designations of essential fish habitat or critical habitat). Conservation or restoration efforts for many species is often directed to necessary habitats needed to support specific life-history stages and is thus a critical component of ecosystem assessments.

CONCEPTUAL APPROACHES

A primary challenge in accounting for human dimensions in the CCIEA is that we often lack conceptual and methodological precedents for integrating the social sciences into environmental science frameworks such as the IEA approach. To meet this challenge, we have worked with our natural science colleagues to redraw the CCIEA's overall socio-ecological system conceptual model in order to better account for the complexity of human dimensions. Some aspects of human dimensions are more suited to quantitative approaches than others, and thus, we suggest making a place in IEAs for qualitative approaches that may be most effective at shedding light on historical, cultural, and political contexts underlying peoples' experiences and values of ecological systems.

A second major challenge to integrating human dimensions into the IEA is that social data are not necessarily already available on the topics or at the resolution necessary to answer questions about the social effects of marine conditions and management strategies. There is a critical need for additional research to produce new, diverse kinds of social science information to inform ecosystem-based management.

In the following section we illustrate a number of diverse potential conceptual approaches for integrating human dimensions into the CCIEA. This is followed by a section summarizing CCIEA-specific social indicators and other types of assessments that have been produced through a number of these approaches. Together these results provide a multifaceted, though still admittedly incomplete, picture of the human dimensions of the California Current.

CULTURAL DIMENSIONS OF SOCIO-ECOLOGICAL SYSTEMS

(author: Melissa Poe, NWFSC)

Environments are complex socio-ecological systems demanding interdisciplinary research and conservation. Despite significant progress in characterizing socio-ecological complexity, cultural values and their importance to conservation remain poorly understood and inadequately accounted for in ecosystem-based management (EBM). In a recent review, Poe et al. (2014) synthesized existing social sciences to build an approach for better integrating cultural dimensions into coastal conservation. They used a focus on cultural dimensions to help identify important interactions between coastal resources and social groups, and as a means to improve socio-ecological analyses and management. Using examples from coastal ecosystems in North America, Poe et al. (2014) described cultural dimensions of a socio-ecological systems model to illustrate five key interrelated cultural aspects: (1) meanings, values, and identities; (2) knowledge and practice; (3) governance and access; (4) livelihoods; and (5) cultural interactions with biophysical environments (see Figure HD2).

It is important to consider cultural dimensions in conservation because implementation of integrated conservation programs without consideration of sociocultural dimensions provides only part of the ecosystem picture (Poe et al. 2014). Coastal environments are fundamental to the sociocultural wellbeing of people and contribute to people's sense of place, wellbeing, relationships, and community resilience. Thus, failure to consider cultural dimensions risks creating or reproducing social inequalities, diminishing community resilience, and stripping away mitigating processes (e.g., customary tenure, social norms, and knowledge systems). Moreover, omitting important cultural dimensions may create conflict, reduce trust, and hinder collaborative management. Conversely, including sociocultural dimensions in conservation may increase buy-in, reduce conflict and costs associated with negotiation, and yield better alternatives that address concerns of those most affected by environmental and institutional changes. Including meaningful sociocultural components in conservation also fulfills a number of government directives to which natural resource agencies are bound.

Meanings, Values, and Identities

- Define a person or community and constitute a 'way of life'
- Attributed to objects, places, relationships, practices, and processes
- Enlivened through language, relationships, and practices
- Develop through ecosystem interactions
- Form and informed by 'cultural models'
- Dynamic, heterogenous, changing over time and space

Local Ecological Knowledge and Practice

- Cumulative knowledge of the environment and its social and spatial conditions
- Embedded within sociocultural processes
- Continually regenerated through practical engagements with ecosystems

Livelihood Dynamics

- Formal and informal economic activities
- Noncommercial harvests for household use or exchange
- Linked to culture, knowledge, social relations, and traditions
- Job satisfaction, quality of life, and occupational and place identities

Governance and Access

- Mechanisms of control, rules of access, decision-making processes
- Tied to philosophies, norms, relationships, and knowledge systems
- Varied dynamics across spatial and organizational scales
- Entangled with political issues of power and inequalities

Bio-cultural Interactions

- Varied food web effects on sociocultural phenomena
- Cultural keystones species play fundamental roles in social systems and cultural identity
- Culturally-based restoration and management creates 'bio-cultural landscapes'
- Changing enviroments impact cultural connections to ecosystems and cultural wellbeing

Figure HD2. Cultural dimensions of socioecological systems model: key aspects and attributes

Poe et al. (2014) conclude their review by suggesting a set of guiding principles for conservation scientists and practitioners working across socio-ecological systems. These principles are: (1) Recognize the diverse cultural meanings and values embedded in human-environment interactions; (2) Protect access to resources, spaces, and processes upon which cultural wellbeing depends; (3) Involve communities who have cultural connections to ecosystems in science and management at all stages (from problem framing to assessment, to identifying and implementing solutions, to monitoring); (4) Allow for cross-scale and nested linkages when assessing and managing cultural dimensions of ecosystems; and (5) Recognize the integrated and coupled nature of sociocultural wellbeing and ecosystem health, and design conservation approaches appropriately. Joining sociocultural with ecological and economic considerations of complex socio-ecological systems can be challenging, but is necessary to manage and protect environments for human wellbeing, ecosystem integrity, and viable economies.

ECONOMIC FRAMEWORKS

(author: Dan Holland, NWFSC)

As noted by Lipton et al. (1995, p. 10), the "fundamental distinction between the way economics and other disciplines such as ecology use the term 'value' is the economic emphasis on human preferences." Economics as a discipline is anthropocentric, focusing on human behavior and wellbeing. As such, economic assessments provide a natural complement to ecological perspectives on ecosystem health and function that emphasizes functioning of natural systems and how they are impacted by humans (Holland et al. 2009). Economic analyses can assess tradeoffs between ecosystem protection and associated changes on one or more human activities—in terms of the overall impact on long-run social wellbeing.

Benefits derived from ecosystem services can be direct (e.g., beach use, commercial fish catch), or indirect (e.g., the contribution of submerged aquatic vegetation to the production of fish harvested elsewhere). Services may be traded in traditional markets with observable market prices and values (e.g., commercial fish harvest, electricity from offshore wind turbines), or may be available outside of traditional markets (e.g., recreational fishing, bird watching, coastal viewsheds). People also value things they do not use (non-use values) and may never see – e.g., the continued existence of deep water corals or an endangered seabird. Although economics is often accused of overemphasizing market activities and associated benefits, appropriate economic analysis should provide equal consideration to all short- and long-term sources of human benefit, regardless of their relationships to organized markets. There are a variety of methods that use observations of peoples' activities and choices (revealed preference methods) or surveys (stated preference methods) to determine the value people derive from ecosystem services that are not bought and sold in organized markets (including non-use values).

There are a variety of analytical frameworks used to integrate economic insight into management considerations. One common means of providing economic insight, denoted cost benefit analysis (CBA), involves either comprehensive or partial assessments of the long-term economic benefits and costs of projects or policies. Multi-attribute utility theory, or MAUT, is a cousin of CBA, in that it is designed to allow assessment policies such as EBM in which multiple attributes are affected. Like CBA, MAUT attempts to estimate a single cardinal "value" whereby policies may be ranked. However, unlike CBA, the "weights" or relative importance given to each policy attribute are not determined by economic value or willingness to pay (WTP) of affected households or individuals but are defined by decision makers, policy experts, or analysts.

In some cases the information necessary to determine the benefits of alternative actions or policies is unavailable but there is still a need to achieve a specified outcome efficiently. Cost-effectiveness analysis (CEA) can help determine the most efficient means of achieving specified management goals in cases where these goals are predetermined by legislation, prior consensus, or other means. CEA can also provide insight on the costs of obtaining various management outcomes.

Yet another economic approach sometimes used to inform management is regional economic modeling, or economic impact analysis (EIA). Unlike CBA or CEA, economic impact analysis measures changes in economic activity or indicators (e.g., regional income, gross value of landings, workers employed, gross expenditures, multipliers) related to monetary flows between economic sectors. In simple terms, EIA tracks monetary payments as they move through a regional economy — measuring the transfer of money from one sector to another. These flows provide insight into the raw quantity of economic activity within a given region and are often of interest to policy makers, but they do not measure changes in economic benefits or costs. A classic illustration of this would be measuring the economic impact of an oil spill with an EIA. Economic activity associated with clean-up activities could easily exceed the economic activity impeded by the oil spill in the short-run but the long-run costs of the oil spill in terms of loss of ecosystem services could be substantial. Of course we would never consider deliberately causing an oil spill to create jobs and income, but this example illustrates that an EIA might suggest that the spill would have positive economic impacts when a CBA would clearly show that human welfare was diminished.

At this time, the Human Dimensions chapter of the IEA and associated analyses do not undertake a comprehensive economic analysis of the net benefits humans derive from the California Current ecosystem or the impacts of human activities and policies on those benefits. Economics is arguably less useful for determining the overall benefits associated with an ecosystem than it is in evaluating how specific types of benefits change over time, or might change as a result of a policy or management action or some external driver such as climate change or an economic shock. At present we provide only a few indicators of economic benefits, such as time series of fishery revenues by community or fishery and metrics such as the fishery income diversification index, which is an indicator of financial risk for the fishing industry (see below, "Fishing Diversification," and Appendix for details). In the future, additional analyses may be added to quantify and track various types of benefits, but this will likely remain a limited set of analyses targeting specific ecosystem services and economic indicators rather than a comprehensive assessment of benefits derived from the California Current ecosystem.

SOCIAL INDICATORS AND HUMAN WELLBEING: CONCEPTS AND METHODS

(authors: Sara Breslow, Melissa Poe, Karma Norman, Phil Levin, NWFSC; Nives Dolsak, Brit Sojka, Raz Barnea, University of Washington; Penny Dalton, Washington Sea Grant)

The Social Wellbeing Indicators for Marine Management (SWIMM) project is a two-year effort supported by the NWFSC, Washington Sea Grant, and the University of Washington to improve understanding of the human dimensions of ecosystem-based management (EBM). The primary objective is to develop a suite of indicators of human wellbeing for use in NOAA's Integrated Ecosystem Assessment (IEA) of the California Current. The broader objective is to develop a generalizable social science protocol for assessing human wellbeing that can be used in other socio-ecological assessments, such as marine spatial planning and social impact assessment, in other regions of the US, and beyond.

With these multiple expectations, SWIMM aims to develop indicators of human wellbeing that: (1) integrate with the biophysical indicators that have already been developed for the CCIEA; (2) serve the needs of federal marine managers and other environmental decision-makers; (3) resonate with a broad diversity of people on the US West Coast; and (4) can be modified for other contexts. Given its scope, SWIMM is informed by local to international sources.

The overall SWIMM approach is modeled after the first two steps – scoping and indicator selection – of the process developed for other IEA indicators (Levin et al. 2009), with modifications based on insights from the social sciences and local stakeholders (Fig. HD3). Theoretical and methodological guidance is provided by an 18-member working group of interdisciplinary and international environmental social scientists who represent a broad range of applied expertise in environmental governance, human wellbeing, social impact assessment, indicator development, ecosystem services valuation, and related fields (

Table HD1). We have developed a conceptual model of human wellbeing (Fig. HD4) for the purposes of ecosystem-based management (EBM) by comparing and compiling priorities for wellbeing found in US Federal environmental policy and legislation, to serve managers' direct needs (Table HD2), and those found in existing socio-ecological indicator projects around the world, to ensure a well-rounded and generalizable definition of wellbeing (Table HD3). Finally, as a pilot study, we are seeking guidance on local issues, concerns, and definitions of wellbeing, specifically with respect to marine conditions and management, from conversations with stakeholders on the outer coast of Washington State (scoped for August 2014).

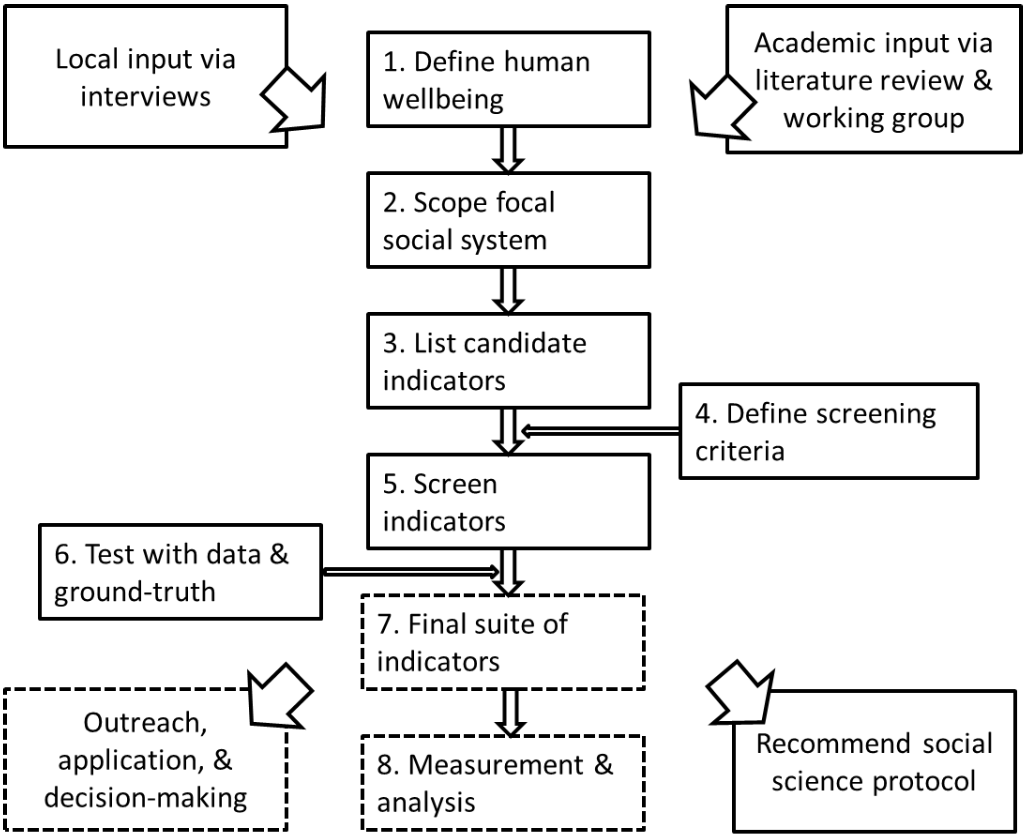


Figure HD3. Proposed approach to identifying indicators of human wellbeing for EBM. Dotted lines represent steps outside the scope of SWIMM.

Table HD1. SWIMM working group members.

Arun Agrawal, University of Michigan
Xavier Basurto, Duke University
Sara Breslow, NRC/NOAA
Courtney Carothers, University of Alaska
Susan Charnley, USFS, Portland
Sarah Coulthard, Northumbria University
Nives Dolsak, University of Washington
Jamie Donatuto, Swinomish Tribe
Carlos Garcia-Quijano, University of Rhode Island
Christina Hicks, Center for Ocean Solutions
Phil Levin, NOAA
Arielle Levine, San Diego State University
Michael Mascia, Conservation International
Karma Norman, NOAA
Melissa Poe, NOAA/Washington Sea Grant
Terre Satterfield, University of British Columbia
Kevin St. Martin, Rutgers University

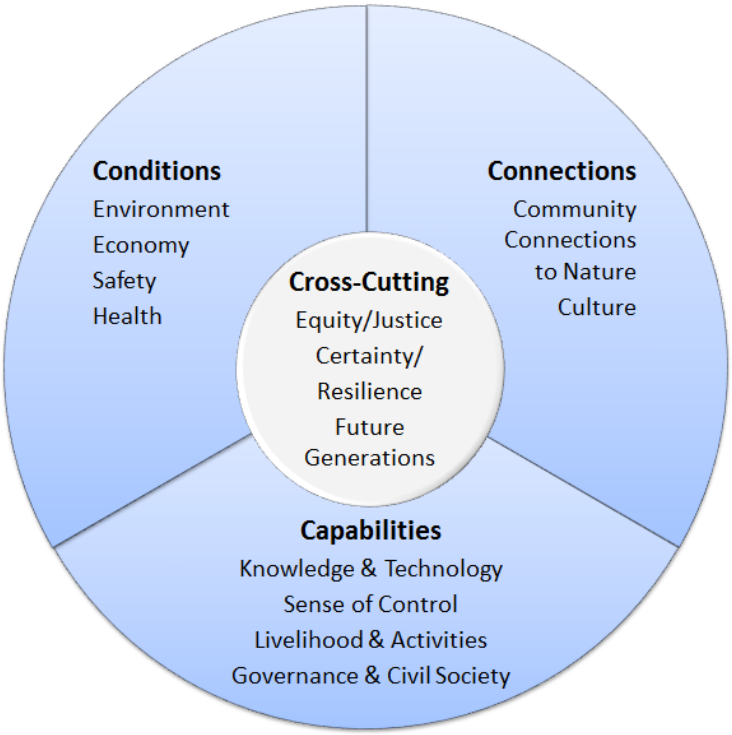


Figure HD4. The Wheel of Wellbeing: SWIMM's conceptual model of human wellbeing (in progress). The wheel and spokes suggest domains of wellbeing that are conceptually distinguishable, but in reality interdependent and dynamic. The central hub indicates domains of wellbeing that are generated by all others and which may be assessed through a cross-cutting analysis. This is a preliminary conceptual model, to be modified as research progresses.

Table HD2. Governmental legislation and policy reviewed for attributes of human wellbeing (n=21). These statutes were selected for their relevance and importance to ocean and coastal management in the United States and Canada. Attributes of wellbeing and supporting language were identified for each.

US Federal Legislation (n=7)

1. Magnuson Stevens Act 2007 - Amended
2. Clean Air Act
3. Federal Water Pollution Control Act (Clean Water Act)
4. National Environmental Policy Act (NEPA)
5. Marine Mammal Protection Act
6. Endangered Species Act
7. Coastal Zone Management Act

US Federal Policy (n=4)

8. National Ocean Policy 2013
9. Ocean Policy Task Force Final Recommendations 2010
10. Executive Order on Government to Government Relations
11. Obama 2013 Ocean Research Priorities Plan Update
12. Executive Order on Environmental Justice

US West Coast Management (n=5)

13. CCIEA Report Summary 2012
14. CCIEA Scenarios 2012
15. CCIEA 2012 Engagement Chapter
16. PFMC 2013 - Pacific Coast Ecosystem Fishery Plan
17. PFMC 2013 - Ecosystem Initiatives Appendix

US State Leg/Policy: WA, OR, CA (n=4)

18. California Ocean Protection Act
19. California Coastal Act
20. Washington Shoreline Management Act
21. Oregon Coastal Management Program

Table HD3. Applied socio-ecological projects reviewed for attributes of human wellbeing, candidate indicators, and best practices (n=52). From a list of 175 candidate projects collected through a literature review and expert consultation, 52 projects were selected for review based on 4 major criteria: 1) inclusion of social and ecological indicators, 2) real-world application, 3) thorough documentation and evaluation, and 4) influential status due to funding level, geographic scope, or presence in the media or literature.

Environmental Management Projects (n = 12)

1. Transboundary Waters Assessment Programme
2. Integrating Watershed & Coastal Areas Management in Caribbean Small Island Developing States
3. Nature Conservation and Human Well-Being in Bhutan
4. Wellbeing in Developing Countries (WeD)/Wellfish
5. Ocean Health Index
6. Millenium Ecosystem Assessment
7. Gulf Ecology Human Wellbeing Index
8. Developing Human Wellbeing Indicators for the Hood Canal Watershed
9. Vital Signs (African Monitoring System)
10. Evaluating Social and Ecological Vulnerability of Coral Reef Fisheries to Climate Change
11. Selecting Indicators to Protect and Sustain Experiences in the Eastern Arctic of Nunavut
12. Socio-economic drivers and indicators for artisan coastal fisheries in Pacific Island Countries & Territories

National Indicator Projects (n = 10)

13. Measures of Australian Progress (MAP)
14. Canadian Index of Wellbeing
15. UK Measuring National Well-being Programme
16. The State of the USA
17. European Social Survey Round 3 Wellbeing Module
18. Commission on the Measurement of Economic Performance and Social Progress
19. Bhutan's Gross National Happiness Project
20. Gallup Healthways Well-Being Index
21. Hong Kong Quality of Life Index
22. Thailand Green & Happiness Index

U.S. Federal Resource Mgmt Projects (n = 10)

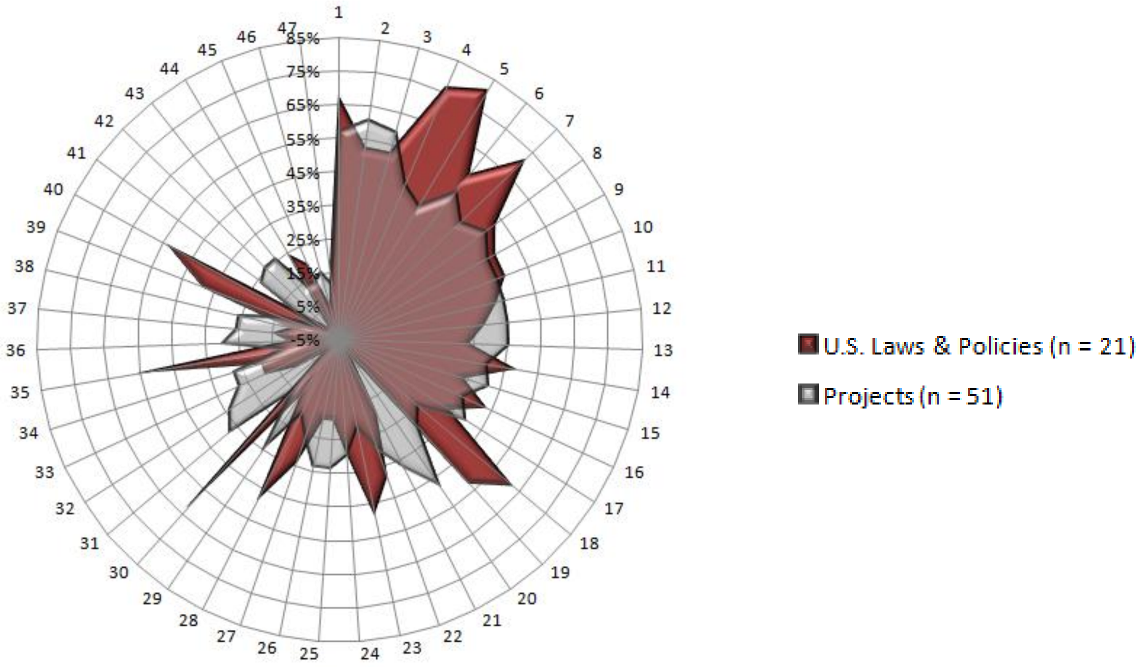
23. Large Marine Ecosystems (U.S. Federal Resource Management; UNEP/RS; GEF)
24. Evaluating Changes in Health and Well-being in Communities Affected by the Deepwater Horizon Disaster
25. Development of Social Indicators of Fishing Community Vulnerability and Resilience in the U.S. Southeast and Northeast Regions
26. Fisheries Social Impact Assessment (Pollnac et al.)
27. Measuring the social and economic performance of catch share programs: definition of metrics and application to the U.S. Northeast Region groundfish fishery
28. Marine and Estuarine Goal Setting for South Florida (MARES) - Noneconomic Indicators
29. Puget Sound Partnership
30. Socioeconomic Profiles of Fishers, their Communities and their Responses to Marine Protective Measures in Puerto Rico
31. Community Profiles for West Coast Fishing Community
32. Improving Community Profiles for the North Pacific Fisheries

Indigenous Projects (n = 10)

33. Voices From The Bay: Traditional Ecological Knowledge of Inuit and Cree in the Hudson Bay Bioregion
34. Social Indicators Study of Alaskan Coastal Villages
35. West Coast Vancouver Island Coastal Strategy & Integrated Ocean Management Plan
36. Arctic Social Indicators Project
37. Swinomish Indigenous Health Indicators
38. Te Kupenga Maori Wellbeing Survey
39. Indigenous Relational Wellbeing Index
40. First Nations Health Indicators Toolkit
41. SARD Cultural Indicators of Indigenous Peoples' food and agro-ecological systems
42. UN Permanent Forum on Indigenous Issues

Sustainability Projects (n = 10)

43. Toronto Vital Signs
44. Sustainable Consumption & Production Indicators for Developing Countries
45. SUSTAIN Partnership
46. Sustainable Neighborhoods for Happiness
47. Sustainability Monitor of the Netherlands
48. UNDESA Indicators of Sustainable Development
49. FAO Intl Guidelines on Securing Small-Scale Fisheries
50. Genuine Progress Index (GPI) Atlantic
51. Sustainable Bergslagen Cultural Indicators
52. Measuring Wellbeing: Blythe Valley Case Study



1. Infrastructure/Built Environment/Ports/Housing/Transit
2. Education/Outreach/Building Awareness/Access to Info
3. Material Wellbeing/Wealth/Prosperity/Material Security
4. Public/Political Participation
5. Environmental Quality/Habitat Health
6. Physical Health/Mortality
7. Governance/Management/Public Services
8. Pollution/Waste
9. Resource Availability & Ecosystem Distributions
10. Resource Access & Utility
11. Cultural Values/Traditions/Valued Practices
12. Food/Nutrition/Food Security
13. Civil Society
14. Future Generations' Wellbeing/Sustainability
15. Commerce/Industry/Trade/Revenue
16. Recreation and Tourism
17. Social Justice/Equity
18. Conservation/Stewardship/Environmentalism
19. Transparency in Government
20. Emotion/Attitude/Mental Health
21. Jobs/Employment
22. Access to Nature
23. Archaeological/Historic Heritage
24. Agency/Self-Governance/Sovereignty
25. Subsistence
26. Security/Peace
27. Hazards Preparedness
28. Safety
29. Demographics
30. Diversity/Multiple Users
31. Social Relationships
32. Personal Activities/Time Allocation
33. Non-Consumptive Uses
34. Place Attachment/Sense of Place/Place-Based
35. Science/Research/Production of Knowledge/Technology
36. Livelihoods
37. Local Economies/Corporate Consolidation/Economic Freedom
38. Social Capital
39. Conflict Reduction/Resolution
40. Beauty/Aesthetics
41. Wonder/Spirituality
42. Job Quality
43. Energy Production & Consumption Patterns
44. Community Vibrancy/Integrity/Stability/Adaptability
45. Resilience
46. Identity
47. Certainty/Predictability/Ability to Plan Future

Figure HD5. Percentage of reviewed US governmental documents and socio-ecological indicator projects that mention each wellbeing attribute (presence/absence).

According to lessons learned from more than a century of social indicators use and application, the most effective indicator sets do not attempt to measure all aspects of wellbeing; rather, indicators should be few in number but high in theoretical, applied, and symbolic significance (Cobb and Rixford 1998). Thus, while we have developed a robust model of wellbeing that aims to provide context and raise awareness of the multiple, interrelated dimensions of wellbeing, we are developing indicators for only a subset of its domains. The Working Group identified six priority domains that were (1) foundational to other areas of wellbeing in an EBM context, and (2) most sensitive to EBM decisions. These domains may be related to one or more attributes. While subject to change, the domains we are first focusing on are:

1. Resource access (resource access and utility, resource availability, environmental quality, etc.)
2. Self-determination (sense of control: agency, self-governance, sovereignty, political participation, government transparency, etc.)
3. Social integrity (social relationships, social capital, community integrity, etc.)
4. Job quality (jobs/employment, demographics, livelihoods, personal activities, time allocation, etc.)
5. Food systems (food resources, nutrition, food security, etc.)
6. Intangible connections to nature (sense of place, wonder and spirituality, recreation and tourism, cultural values, knowledge, etc.)

Following the IEA method (Levin et al. 2009), we have begun screening indicators of wellbeing for these domains according to predefined criteria, such as theoretical validity, geographic relevance, management relevance, local significance, and data availability. Candidate indicators are being compiled from 52 existing socio-ecological indicator projects, a literature review and local input. A next step, beyond the scope of SWIMM, will be to test the screened indicators with actual data and ground-truthing. Final indicator sets can then be selected and tailored for specific intended uses and audiences.

POLITICAL ECOLOGY: A HOLISTIC APPROACH TO SOCIO-ECOLOGICAL ANALYSIS

(author: Sara Breslow, NWFSC)

Political ecology is a well-developed field in the environmental social sciences that takes a holistic approach to analyzing the social causes and consequences of environmental problems. Primarily through case studies, political ecology explores the causal linkages among the various components of the socio-ecological system, with a focus on how local socio-ecological dynamics interact with broader political and economic forces. Collectively, these studies reveal regional to global patterns in the human dimensions of ecosystems and natural resource management. Indicators can inform or complement a political ecology analysis.

A case study of social conflict surrounding salmon habitat restoration and farmland preservation in the Puget Sound basin suggests how political ecology can inform and guide resource management. This study analyzes how "social hierarchies and mistrusts, conflicting senses of place, prevailing cultural narratives, and legal and institutional constraints contribute to the local dispute over habitat restoration." It argues that, "Closer attention to sociocultural factors such as these may help managers identify and implement locally supported recovery opportunities, facilitate cooperation among stakeholders, improve agency approaches, and reframe management agendas to better address collective needs." (Breslow 2014)

SOCIAL INDICATORS AND ASSESSMENTS

COMMUNITY VULNERABILITY ASSESSMENTS

(authors: Karma Norman, Stacey Miller, NWFSC; Stephen Kasperski, AFSC; Kristin Hoelting, Colorado State University)

This section presents a method for using secondary data to assess community-level vulnerability to ecosystem changes, as well as management, policy and other shifts. The method relies primarily on sociodemographic data derived from the U.S. Census alongside commercial fisheries data, but also includes and analyzes data from other available and relevant secondary data sources. The indices which incorporate these data have been developed for and applied to a separate vulnerability assessment process for the coastal communities of the U.S. Southeast and Northeast regions (Jepson and Colburn 2012), building upon prior social indicators work in coastal and fisheries contexts (Cutter 1996, Cobb and Rixford 1998, Pollnac et al. 2006, Jepson and Jacob 2007, Cutter et al. 2008).

The community vulnerability assessment approach is also supported by earlier efforts within fisheries social science, and within the National Marine Fisheries Service (NMFS) in particular, to define and characterize fishing communities both quantitatively and qualitatively (Acheson 1980; McCay and Cieri 2000; Gilden 1999; Norman et al. 2007; Sepez, et al. 2006; Sepez, et al. 2007). Vulnerability indices and vulnerability analyses employed for the coastal communities of the U.S. East Coast have been replicated for the human communities adjacent to and integrated with the CCLME. Similar assessments of fishing reliance and socioeconomic vulnerability are already underway in the Alaska region and, through the development of this work nation-wide, a relatively uniform approach to coastal community vulnerability will be applied throughout U.S. fisheries management regions and in multiple IEA contexts.

In order to assess and track coastal community vulnerability for the inhabited shoreline areas adjacent to the California Current Large Marine Ecosystem (CCLME), we identified a set of indices that were drawn from extant community-level data and subjected to factor analyses. This process determined which communities are potentially most reliant on fisheries and marine ecosystems, and which among these are the most socioeconomically vulnerable. While this approach has been successfully developed and implemented for coastal communities on the U.S. East Coast (Jacob et al. 2012; Jacob et al. 2010; Colburn and Jepson 2012), the method of measuring and evaluating socioeconomic resilience is still in the early stages of data collection, organization and analysis for the communities of the U.S. West Coast (i.e. the coastal portion of the California Current Large Marine Ecosystem) and Alaska. Several of these indices are developed to account for socioeconomic vulnerability of California Current coastal communities. The socioeconomic vulnerability indices provided below include a *personal disruption index*, a *population composition index* and an index of community *poverty*.

For all three of these aforementioned indices, data are provided by the U.S. Census's American Community Survey (ACS), and were organized for all census-designated place (CDP) level communities in all coastal counties in Washington, Oregon and California. In this way, this vulnerability indicator approach sought to cover the geographic breadth required of the CCLME. Relevant indicator selection considerations for the personal disruptions index were based upon an ongoing national approach along with modified indicator selection criteria described for the natural science components of the IEA (Kershner et al. 2011).

The *personal disruptions index* developed by fisheries social scientists in the Southeast and Northeast regions, following prior work on community vulnerability (Cutter 1996, Jacob et al. 2012), provides a means of assessing commercial fishing reliant communities according to one aspect of their relative socioeconomic vulnerability. Relatively frequent personal disruptions within the community are linked to increased overall vulnerability to natural hazards and other events associated with livelihood and social impacts (Cutter et al. 2000, Jacob et al. 2012). The personal disruptions index, employed as a way of measuring socioeconomic vulnerability, includes indicators that account for:

- Percent within the community unemployed
- Percent of the community with no diploma
- Percent of the community living in poverty
- Percent of separated females in the community

As a companion to the personal disruptions index, the *population composition index* quantitatively describes the social make-up of the human communities reliant on the fisheries of the CCMLE. The indices of socioeconomic vulnerability, including the population composition index, rely on community-specific data pulled from annual ACS datasets as maintained by the U.S. Census. American Community Survey data allows for the use of regularly updated data for each of the 2,529 communities within the coastal counties of interest for the CCLME. The population composition index combines ACS data on race, gender and other demographics including:

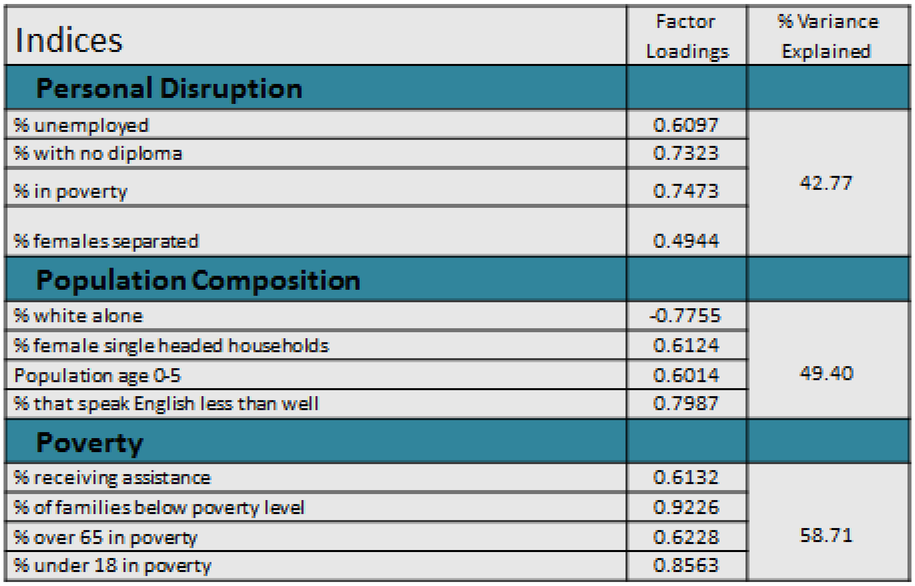
- Percent of community identifying racially as "white alone"
- Percent of community with female single headed households
- Population age 0-5
- Percent that speak English less than well

In addition to the personal disruptions index and the population composition index, factor analyses on poverty indicators can offer assessments of socioeconomic vulnerability for coastal communities. A *poverty index* developed by fisheries social scientists in the Southeast and Northeast regions, following prior work on community vulnerability to natural hazards (Cutter 1996, Cutter et al. 2000, Jacob et al. 2012), provides a means of assessing relative well-being, vulnerability and resilience potential of fishing reliant communities. The poverty index, employed in measuring socioeconomic vulnerability of coastal communities, includes indicators that account for the:

- Percent within the community receiving assistance
- Percent of families within the community living below the poverty level
- Percent of the community over 65 years old living in poverty
- Percent of the community under 18 years old living in poverty

Data for each socioeconomic vulnerability indicator, based upon the most recent U.S. Census survey of 2010, were subjected to factor analyses in order to provide single factor solutions for each index of socioeconomic vulnerability (Table HD4). Considered together, these indices provide a means of comparing socioeconomic vulnerabilities across the coastal communities of the California Current Large Marine Ecosystem (Figure HD6).

Table HD4. Factor loading results for each of the sociodemographic vulnerability indices. These were factor analyses applied to 2,529 communities in coastal counties in Washington, Oregon and California, including 1,099 for which data indicate commercial and/or recreational fishing activity.



Indices	Factor Loadings	% Variance Explained
Personal Disruption		
% unemployed	0.6097	42.77
% with no diploma	0.7323	
% in poverty	0.7473	
% females separated	0.4944	
Population Composition		
% white alone	-0.7755	49.40
% female single headed households	0.6124	
Population age 0-5	0.6014	
% that speak English less than well	0.7987	
Poverty		
% receiving assistance	0.6132	58.71
% of families below poverty level	0.9226	
% over 65 in poverty	0.6228	
% under 18 in poverty	0.8563	

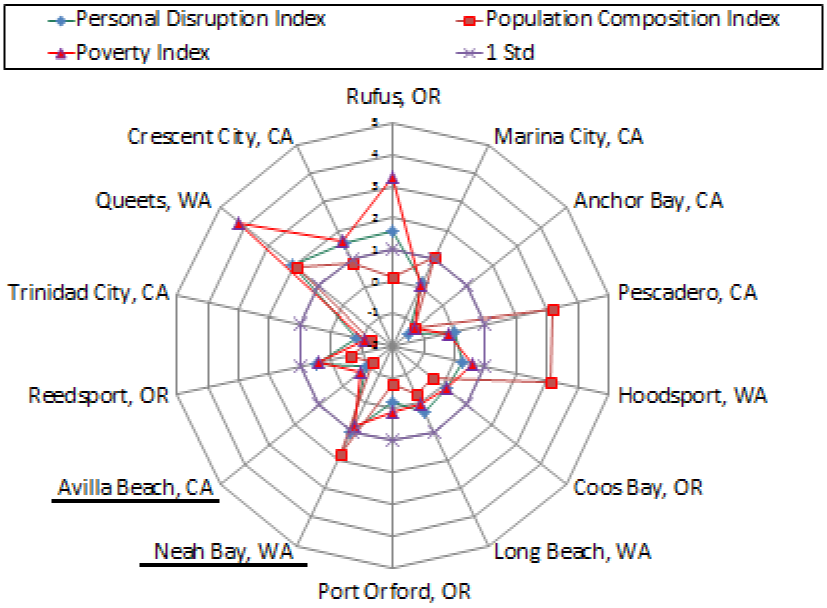


Figure HD6. Selected California Current coastal communities compared relative to one another sociodemographically. The underlined communities of Neah Bay, Washington, and Avilla Beach, California exemplify the kind of contrast that this approach helps to illuminate in the context of the IEA: Neah Bay is at least one standard deviation above the mean for all three indices of socioeconomic vulnerability, whereas Avilla Beach lies below the standard deviation for all three indices.

Similarly, additional indices are used to examine coastal communities with respect to their *reliance* on, and *engagement* with commercial fishing. The commercial fishing reliance index allows for the selection of communities most reliant on commercial fishing and therefore of particular interest to the CCIEA. The indicators included in the commercial *commercial fishing reliance index* are primarily available as annually collected fisheries data maintained by the Pacific Fisheries Information Network (PacFIN), and employment data collected by the U.S. Census' American Community Survey (ACS). The indicators incorporated into the commercial fishing reliance index are the:

- Value of commercial fisheries landings per capita for each community
- Processors with landings per capita for each community
- Percent employed in agriculture, fishing and forestry

The indicators which are included in the *commercial fishing engagement index* are:

- Value of commercial fisheries landings
- Total landings for each community
- Processors with landings

Considered in conjunction with the previously described socioeconomic vulnerability indices, commercial fishing indices allow for selection among those communities that are clearly linked to the CCLME, through data that captures commercial fishing activity, and are also potentially most socioeconomically vulnerable to exogenous shifts and events (Figure HD7).

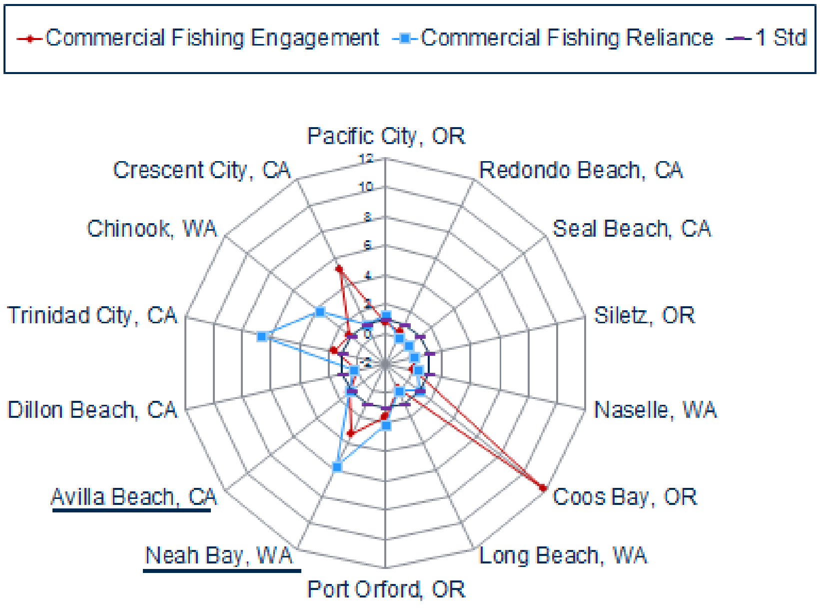


Figure HD7. Selected California Current coastal communities compared relative to one another on fisheries indices. The underlined communities of Neah Bay, Washington, and Avilla Beach, California again exemplify the kind of contrast that this approach helps to illuminate in the context of the IEA: Neah Bay is at least one standard deviation above the mean for both indices capturing commercial fishing activity, whereas Avilla Beach lies below the standard deviation for both indices.

FISHING DIVERSIFICATION

(authors: Dan Holland, NWFSC; Stephen Kasperski, AFSC)

Catches and prices from many fisheries exhibit high inter-annual variability leading to variability in the income derived by fishery participants. Our analysis indicates that income variability is reduced on average if individuals diversify their income by participating in several different fisheries. The annual variability of aggregate revenues for ports is also reduced by diversification. We utilize the Herfindahl-Hirschman Index (HHI) to measure diversification of West Coast and Alaskan entity's gross revenues across species groups and regions. HHI theoretically ranges from zero when revenues are spread amongst an infinite number of fisheries to 10,000 for an entity that derives all revenue for a single fishery. Thus, the less diversified an entity's revenue sources are, the higher the HHI. We evaluate how diversification measured at the vessel level has changed over time for various fleet groups. We also track diversification of aggregate revenues for various port groups over time. A summary of key results is provided below. A description of the methodology and more detailed reports are provided in Appendix HD-1.

Average fishery revenue diversification of West Coast and Alaskan fishing vessels is variable but shows distinct trends over time (Fig. HD8). The HHI, though erratic, has generally been increasing over time meaning that diversification of fishery income has been declining. The current fleet of vessels on the US West Coast and in Alaska (those that fished in 2012) was the least diverse at any point in the past 30 years in 2011,, but diversification increased slightly in 2012.

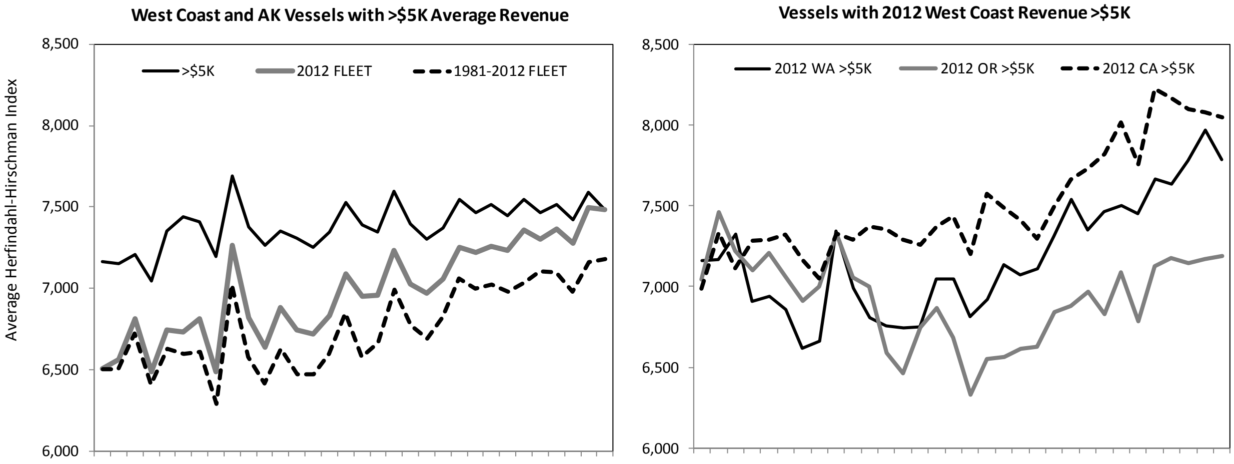


Figure HD8. Trends in average diversification for US West Coast and Alaskan fishing vessels (left panel) and the 2012 West Coast fleets by state (right panel)

Diversification across multiple fisheries can reduce variation in annual revenues and the associated financial risk. It can also increase the minimum annual revenue relative to average revenue, which should reduce the risk of a business failure (Kasperski and Holland, 2013). The ability of fishermen to diversify may be limited (or facilitated) by management approaches and regulatory actions that make it harder (easier) for fishermen to participate in multiple fisheries. There are a number of factors that may limit the feasibility or desirability of greater diversification for individual fishermen. In many cases different fisheries require different gear that must be purchased and there are often costs of acquiring licenses and, increasingly, quota. It may also be the case that a vessel that can participate in several fisheries may be less efficient than more specialized vessels creating a trade-off between risk reduction through diversification and fishing efficiency. The decrease in average diversification is due at least in part to regulations deliberately designed to reduce participation in oversubscribed and often overcapitalized fisheries. Thus, while our results suggest that the observed decrease in diversification of fishing vessels may have increased income variation and financial risk, this does not suggest a decrease in overall economic efficiency.

As is true with individual vessels, the variability of landed value at the port level is reduced with greater diversification of landings. Diversification of landed revenue for some ports has clearly declined (Fig. HD9). Examples include Seattle and most, though not all, of the ports in Southern Oregon and California. A few ports have become more diversified including Bellingham Bay in Washington and Westport, Washington which became less diversified through the mid 1990s but has since reversed that trend. Diversification scores are highly variable year-to-year for some ports, particularly those in Southern Oregon and Northern California that depend heavily on the Dungeness crab fishery which has highly variable landings.

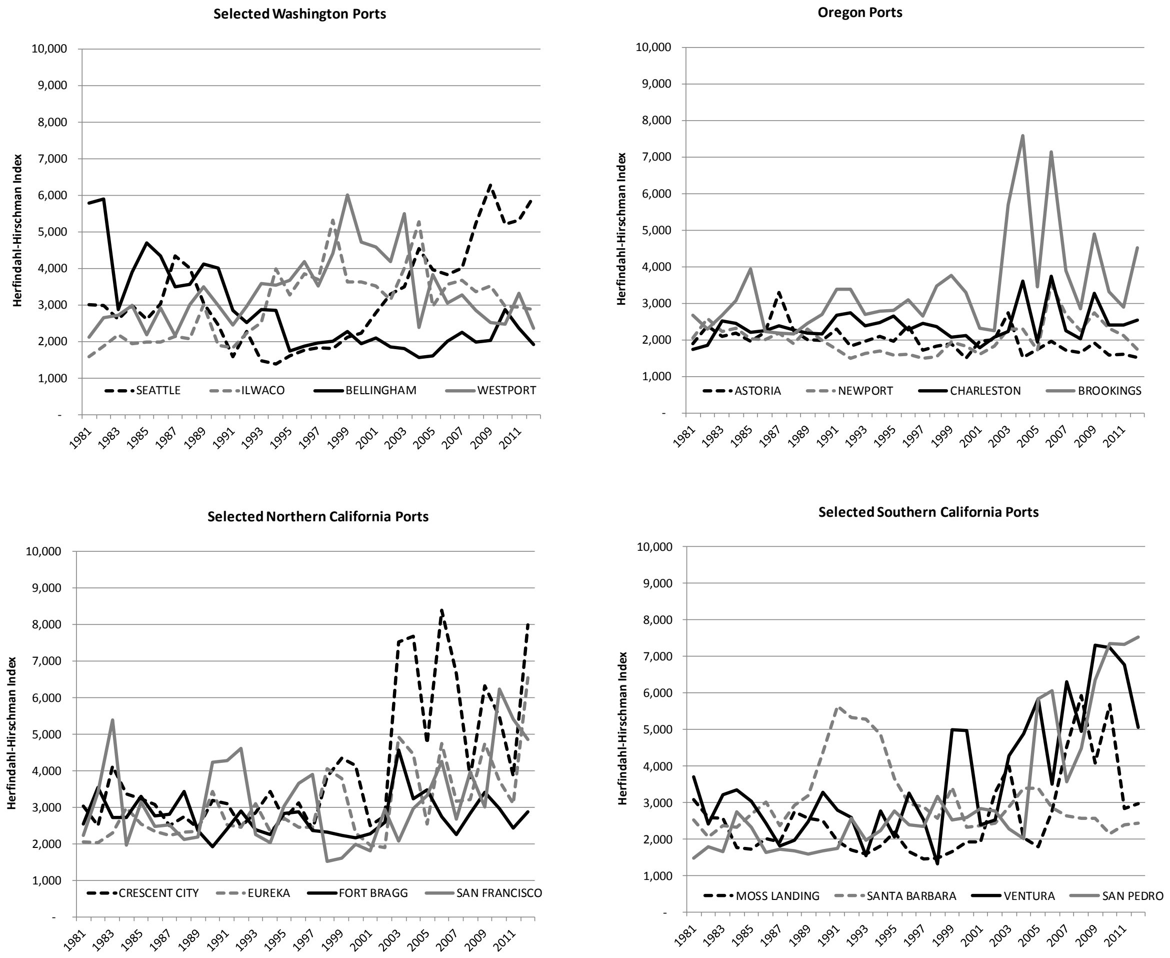


Figure HD9: Trends in diversification for selected primary West Coast ports in Washington, Oregon, and California.

It is not clear that ports could or should increase diversification to reduce variation in landed value, but it does appear that higher levels of diversification can reduce variation in landed value. High variation in overall landed value for several ports is associated with dependence on fisheries like Dungeness crab that have high variation in revenues. This variation could be socially disruptive, but this may be somewhat unavoidable if those ports want to continue to attract the landings from valuable fisheries that have highly volatile annual landings. It should also be noted that the variation in landed value at ports is not necessarily closely correlated with variation in fishing income of fishermen living in those communities since those fishermen may be landing catch in other ports. The link between diversification of individual fishermen and ports and socio-economic wellbeing of communities is one that deserves further research.

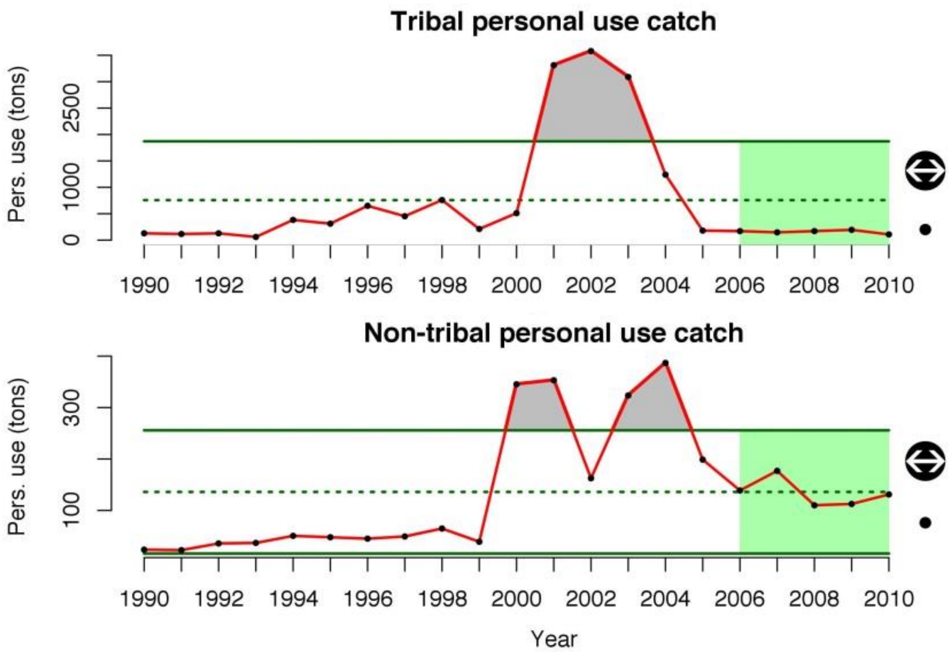
PERSONAL USE: SUBSISTENCE AND INFORMAL ECONOMIC PRACTICES AMONG COMMERCIAL FISHERIES

(authors: Melissa Poe, Nick Tolimieri, Phil Levin, Karma Norman, NWFSC)

Between 1990 and 2010, over 17 million kg of fish and shellfish (worth $116.5 million in fishing revenue) were kept by commercial fishing vessels in Washington and California USA for 'personal use', a category used as a proxy for subsistence food use (Pacific Fisheries Information Network, PacFIN). These 17 million kg of personal use constitute a fraction (0.2%) of the total catch (7.4 billion kg) landed during that same period. Although a nominal figure in the overall seafood catch, subsistence practices function to improve human wellbeing and strengthen community resilience by increasing food security. They may also be significant in the everyday lives of fishing communities for their role supporting social networks through seafood gifts and maintenance of food knowledge systems, ceremonial use, and alternatives to crew compensation. Importantly, the presence of subsistence practices among market-based commercial fishing operators reveals a more diverse array of economic systems than previously imagined.

Personal use is a category of fish biomass landed in ports by commercial vessels, which is not used for commercial or research purposes. Rather, personal use applies to the removal of wild ocean seafood species such as salmon, albacore, squid, crab, and more than a hundred other species that are kept for personal subsistence, sharing within communities, and other noncommercial purposes. In effect, personal use is a functional category identifying subsistence harvesting by commercial operators. While the actual volume of subsistence and noncommercial use is likely much larger than reported, the PacFIN personal use category is one of the few databases through which any subsistence and noncommercial fishing practices on the West Coast can be tracked systematically. The only other noncommercial harvest tracked in the rest of Western US is limited to "recreational" fishing (see RecFIN, http://www.recfin.org/). Thus, while these PacFIN data can tell us a limited amount of information about subsistence among commercial operators, they are not a substitute for a potentially much wider and more diverse set of subsistence practices for food security and cultural food systems in the US.

During the study period, rates of subsistence harvest varied across ports in Washington and California, ranging from zero personal use landings in many ports to over 10% of the relative total catch attributed to personal use in other ports, and as much as 33% in one Puget Sound, WA port. Nearly 85% (14.4 million kg) of the personal use removals is from tribal participants in WA (Fig. HD10). Slightly more than 15% of the personal use removals is from nontribal participants from both WA and CA. The majority of personal use, (over 13.8 million kg or 81.3%) was landed in Puget Sound.


Tribal personal use catch

Non-tribal personal use catch

Figure HD10. Catch retained for personal use from 1990-2010 in tons (= 2000 lbs or 907.2 kg). Green horizontal lines show the mean (dotted) and ± 1.0 s.d. (solid line) of the full time series**.** The shaded green area is the last 5 years of the time series, which is analyzed to produce the symbols to the right of the plot. The upper symbol indicates whether the modeled trend over the last 5-years increased (↗), or decreased (↘) by more than 1.0 s.d., or was within one 1.0 s.d. (↔) of the long-term trend. The lower symbol indicates whether the mean of the last 5 years was greater than (+), less than (-), or within (.) one s.d. of the long-term mean. Data courtesy of PacFIN (pacfin.psmfs.org); data not reported from OR.

Ninety-six percent of the retained catch of tribal participants is comprised of salmonids, the other top species retained by tribes for personal use include: geoduck, Dungeness crab, and Pacific halibut (see Fig. HD11). Nontribal participants retain a wider diversity (breadth) of species than their tribal counterparts; top species include: market squid, albacore, Pacific sardine, Dungeness crab, Pacific halibut, bait shrimp, and salmonids. California ports record less personal use overall than Washington ports, but the species breadth in CA is greater (e.g. in CA, 229 species were kept for personal use and in WA, 93 species were kept).

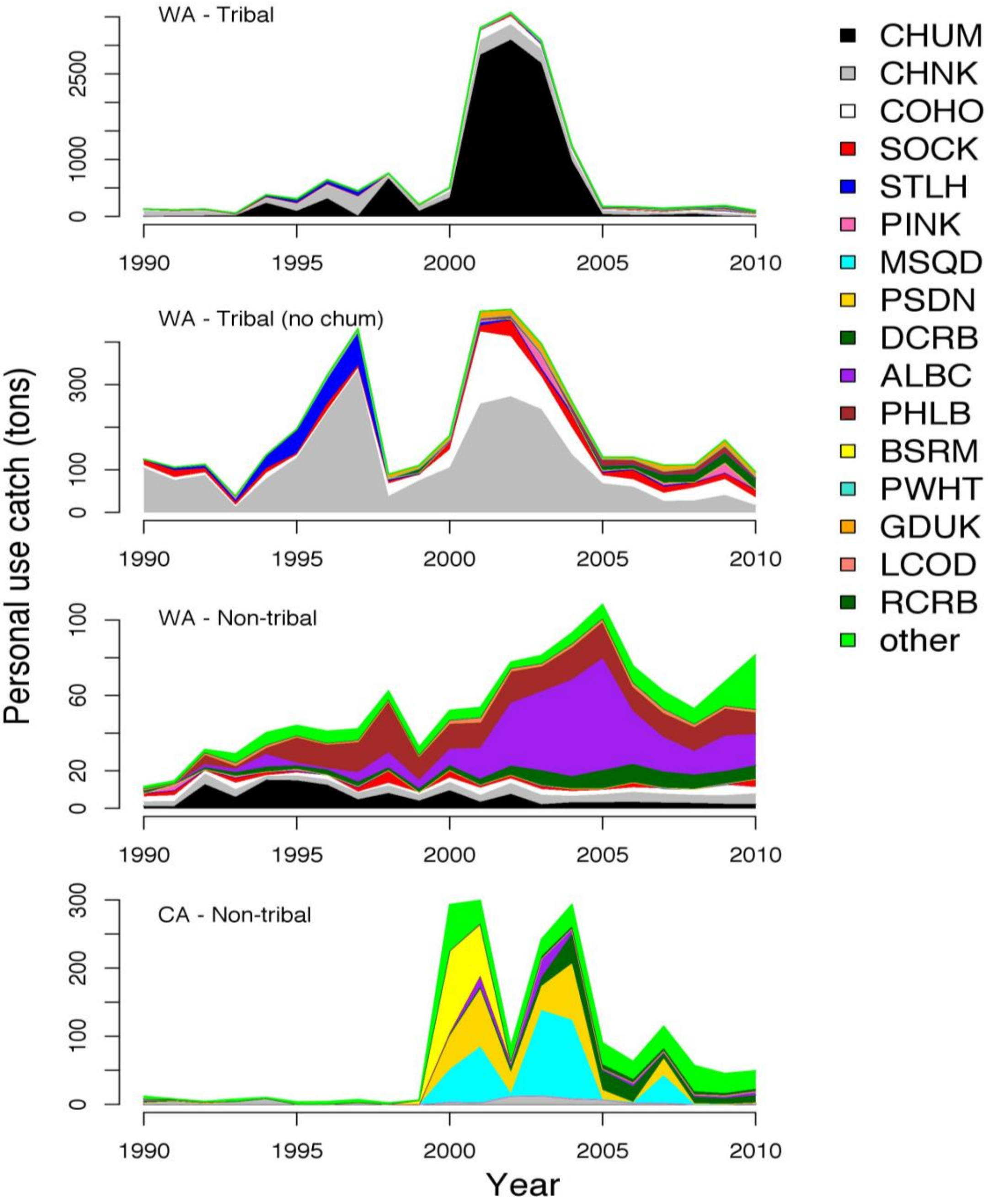


Figure HD11. Annual personal catch by species in tons (= 2000 lbs or 907.2 kg) for WA tribal fishers, WA non-tribal fishers and CA non-tribal fishers from 1990-2010. CHUM = chum salmon, CHNK = Chinook salmon, COHO = coho salmon, SOCK = sockeye salmon, STLH = steelhead, PINK = pink salmon, MSQD = market squid, PSDN = pacific sardine, DCRB = Dungeness crab, ALBC = albacore, PHLB = Pacific halibut, BSRM = unidentified bait shrimp, PWHT = Pacific whiting (hake), GDUK = geoduck, LCOD = lingcod, RCRB = rock crab. Data courtesy of PacFIN (pacfin.psmfs.org).

EFFECTS OF WATER SUPPLY ON LABOR DEMAND AND AGRICULTURAL PRODUCTION IN CALIFORNIA'S SAN JOAQUIN VALLEY

(authors: Cameron Spier, Aaron Mamula, SWFSC; Daniel Ladd, University of California-Santa Cruz)

The San Francisco Bay Delta is the central feature of California's water supply system and is the source of irrigation for about 3.75 million acres of highly productive farmland. The Delta also provides critical habitat for salmonids like Chinook salmon (*Oncorhynchus tshawytscha*) and Steelhead trout (*O. Mykiss*), listed under state and federal Endangered Species Acts. Management of water exports from the Delta is a key issue facing ecosystem restoration efforts. Increased emphasis on instream flow and episodes of drought mean that irrigation water deliveries may be periodically reduced in the future. In this study, we estimate the effects of annual changes in the quantity of water delivered to farms in the San Joaquin Valley on agricultural labor and crop production. Two water projects export water from the Delta to farms in the San Joaquin Valley: the State Water Project (SWP) and the U.S. Bureau of Reclamation's Central Valley Project (CVP).

We construct a statistical model of agricultural production in the San Joaquin Valley of California. The model uses data from 1981 through 2011 to determine how water deliveries from the CVP and SWP to farmers in the San Joaquin Valley are correlated with farm employment and production of certain crops. Our study area consists of six counties in the southern San Joaquin Valley: Stanislaus, Merced, Fresno, Kings, Tulare, and Kern. This region represents some of the most productive farmland in the United States, with all six counties ranking among the top nine in terms of market value of agricultural products sold.

The model consists of 8 equations – an agricultural labor demand equation and supply equations for 7 crop groups (Field Crops, Cotton, Tree Fruits, Grapes and Berries, Nut Orchard Crops, Vegetables, and Processing Tomatoes). To measure agricultural labor in each of the six counties, we use data on farm employment from the U.S. Bureau of Economic Analysis. To measure agricultural production and crop prices, we use data from California County Agricultural Commissioner's Reports. Data on water deliveries are from the U.S. Bureau of Reclamation and California Department of Water Resources.

Preliminary results indicate that farm employment is affected by annual water supply. These effects are relatively small but statistically significant and imply that a 10 percent change in water deliveries results in a less than 2 percent change in employment. Lower water deliveries are also associated with lower production of cotton, field crops, processing tomatoes, and vegetables. Our results also indicate that, over the 31 years of the data, labor demand and crop output may have become more sensitive to changes in the supply of water from the CVP and SWP.

2012 NATIONAL OCEAN RECREATION EXPENDITURE SURVEY

(authors: Rosemary Kosaka, Cindy Thomson, SWFSC; Scott Steinbeck, NEFSC)

In 2012, the National Ocean Recreation Survey was implemented to increase our understanding of national and regional participation in ocean recreation activities. The survey collected participation and expenditure information associated with recreational activities that occur at, in, or in view of oceans, bays, estuaries, coastal wetlands, saltwater bayous, and other seawater areas. These include:

- Recreational finfishing
- Recreational shellfishing
- Hunting waterfowl or other animals
- Viewing or photographing ocean features (e.g., waves) or wildlife (e.g., whales)
- Beachcombing, tidepooling, or collecting items
- Water contact sports such as swimming, surfing, and diving
- Boating and associated activities such as cruises, kayaking, and water skiing
- Outdoor activities not involving water contact such as walking and horseback riding

The survey period was one year, divided into six two-month waves to capture the seasonal variability in recreational activities. On the West Coast, randomly selected households in California, Oregon, and Washington participated in at least one of the six survey waves, with respondents in each wave asked questions about their activities in the previous two months.

Additionally, information was collected regarding how hypothetical changes in air temperature might influence respondents' recreational choices. Using the temperature estimate provided by each survey participant for the day(s) of their most recent ocean activity, they were asked whether they would participate in that same activity, switch to a different ocean activity, or switch to a non-ocean activity if the temperature was 5, 10, or 15°F higher or lower than what they actually experienced. The responses to these questions may contribute to our understanding of how temperature changes may influence the choice between different ocean activities (for example, from boating to swimming if temperatures were to increase) or non-ocean activities (for example, from ocean swimming to pool swimming if temperatures were to decrease).

This data collection was a cross-regional effort between the Office of Science & Technology, Southwest Fisheries Science Center, and the Northeast Fisheries Science Center. Additional partners included the Gulf States Marine Fisheries Commission, ECS (formerly OAK Management), and GfK (formerly Knowledge Networks). Data analysis is underway.

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APPENDIX HD-1. FISHERY INCOME DIVERSIFICATION AND RISK FOR FISHERMEN AND FISHING COMMUNITIES OF THE US WEST COAST AND ALASKA – UPDATED TO 2012

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INTRODUCTION AND METHODOLOGY

Catches and prices from many fisheries exhibit high interannual variability, leading to variability in the income derived by fishery participants. The economic risk posed by this variability might be mitigated in some cases if individuals participate in several different fisheries, particularly if revenues from those fisheries are uncorrelated or vary asynchronously. High annual variation in income is a common problem among natural resource-dependent individuals and communities, and there has been extensive study of risk-coping mechanisms for farmers (Alderman and Paxson, 1992; Paxson, 1992; Townsend, 1994). Crop diversification is a common means of reducing risk in agriculture, taking advantage of asynchronous variation in yields and prices to minimize idiosyncratic risk (Heady, 1952; Johnson, 1967). Another common strategy in agriculture, particularly in semi-arid regions with high fine-scale variation in rainfall, is to farm a number of geographically separated plots to ensure some will be in areas with sufficient rainfall (Rosenzweig and Binswanger, 1993). A number of authors have argued that common property provides an important means risk reduction that may be undermined by privatization (Bromley and Chavas, 1989; Nugent and Sanchez, 1998; Thompson and Wilson, 1994). This literature relates primarily to grazing lands held in common to protect against the potential spatial variation in rainfall that would impact small private holdings but smooth risk for herders utilizing a much larger area held in common. However, similar strategies and principles from this literature apply to fishermen. While formal fishing insurance programs do not exist, fishermen's fishing strategies provide a means to reduce risk, in particular by diversifying their fishing activity across a variety of fisheries or areas (Minnegal and Dwyer 2008; van Oostenbrugge et al. 2002). There is also a growing literature suggesting that fishermen should adopt portfolio approaches to their species composition to achieve the lowest variance in income for any level of expected return (Baldursson and Magnusson, 1997, Hilborn et al. 2001, Kasperski and Holland 2013, Perusso et al. 2005, Sethi 2010, Sethi et al. 2012, Smith and McKelvey 1986).

Following Kasperski and Holland (2013), we measure diversification of West Coast and Alaskan entities' gross revenues across species groups and regions each year. We consider two types of entities for this analysis: individual fishing vessels and individual fishing ports. For both types of entity, we utilize the Herfindahl-Hirschman Index (HHI), defined as:

$$H = \sum_{i=1}^{S_j} \sum_{j=1}^{4} p_{ij}^2, \qquad (1)$$

where p_{ij} represents percent (ranging from 0 to 100) of an entity's total gross revenues derived from species group i in region j. We define p_{ij} to be the percent of an entity's total annual gross revenue from one of 40 different species groupings in one of four regions – the Bering Sea/Aleutian Islands, Gulf of Alaska, Alaskan in-state waters, and the WC (Table HD1-1). Not every species group is caught in each region, so there are a total of 84 region-specific species groupings. HHI theoretically ranges from zero, when revenues are spread amongst an infinite number of fisheries, to 10,000 for an entity that derives all revenue from a single fishery. Thus, the less-diversified an entity's revenue sources are, the higher the HHI. We evaluate how diversification has changed over time for various fleet groups and ports. To explore how diversification of fishery income affects year-to-year variation and thus financial risk, we estimate the statistical relationship between HHI and the coefficient of variation (CV) of gross revenues for each entity across years.

Table HD1-1: Species groups used for diversification indices.

West Coast	Alaska
Pacific Whiting	Pacific Cod
Dover Sole, Thornyheads, Sablefish	Flatfish
Rockfish and Flatfish	Rockfish
Skate, Dogfish, Sharks	Atka Mackerel
Pacific Halibut	Pollock
California Halibut, Croaker	Other Groundfish
Pink Shrimp	Sablefish
Other Prawns and Shrimp	Pacific Halibut
Crab	Herring
Salmon	Chinook Salmon
Tuna	Sockeye Salmon
Herring	Coho Salmon
Coastal Pelagics	Pink Salmon
Echinoderms	Chum Salmon
Other Shellfish	Other Salmon
Squid	Red King Crab
Other Species	Other King Crab
	Opilio Crab
	Other Snow Crab (Bairdi)
	Other Crab
	Scallops
	Other Shellfish
	Other Species

RESULTS

We work with a large dataset that includes annual landings and revenues between 1981 and 2012 by species, port and vessel from all commercial fisheries in the US EEZ off the West Coast and Alaska. We present analysis based on 28,151 vessels with average fishing revenues over $5000 (adjusted to 2005 values) and at least two years of documented landings. The port level analysis includes 166 ports with average fishing revenues over $100,000 (adjusted to 2005 values) and includes 79 ports along the West Coast and 87 ports in Alaska. The large dataset enables us to identify trends in diversification and relationships between diversification and variation in revenues, despite the relationship being very noisy. We also consider a number of subsets of the larger fleet categorized by average revenues, length and whether they had landings in West Coast states (i.e., excluding vessels with revenue only from Alaska).

Average fishery revenue diversification of West Coast and Alaskan fishing vessels is variable but shows distinct trends over time (Figure HD1-1). The HHI for most vessel groups, though erratic, has generally been increasing over time meaning that diversification of fishery income has been declining. The current fleet of vessels on the US West Coast and in Alaska (those that fished in 2012) is less diverse than at nearly any point in the past 30 years, except that they are slightly more diverse than they were in 2011. For smaller vessels diversification has generally been declining (i.e., HHI has been increasing) since 1981. For larger vessels, diversification increased through the early 1990s but has mostly declined since. The causes of the decline in diversification are not completely clear and probably vary by fleet sector. One likely factor that correlates with the observed trend is the successive implementation and tightening of limited access programs and, later, individual quota programs. By the mid-1990s, entry into new fisheries was no longer possible for most vessels since nearly all fisheries had moratoriums on entry, and many were beginning to reduce fleets through attrition, vessel buybacks or catch share programs. These programs limit fishermen’s ability to move into new fisheries and often push out less-active participants from a fishery. This is often necessary to limit catch and improve economic viability of the remaining participants, but it can also result in decreased diversification. Vessels that were in the fishery since 1981 have maintained a higher level of diversification than the overall fleet, while vessels that entered later tend to be less diversified, possibly due to limited access programs in many fisheries. We also looked specifically at diversification trends for vessels with at least $5000 in revenues from landings in WA, OR or CA in 2012. Overall, trends for vessels fishing the West Coast are similar to those for the larger fleet of vessels fishing the West Coast and/or Alaska.

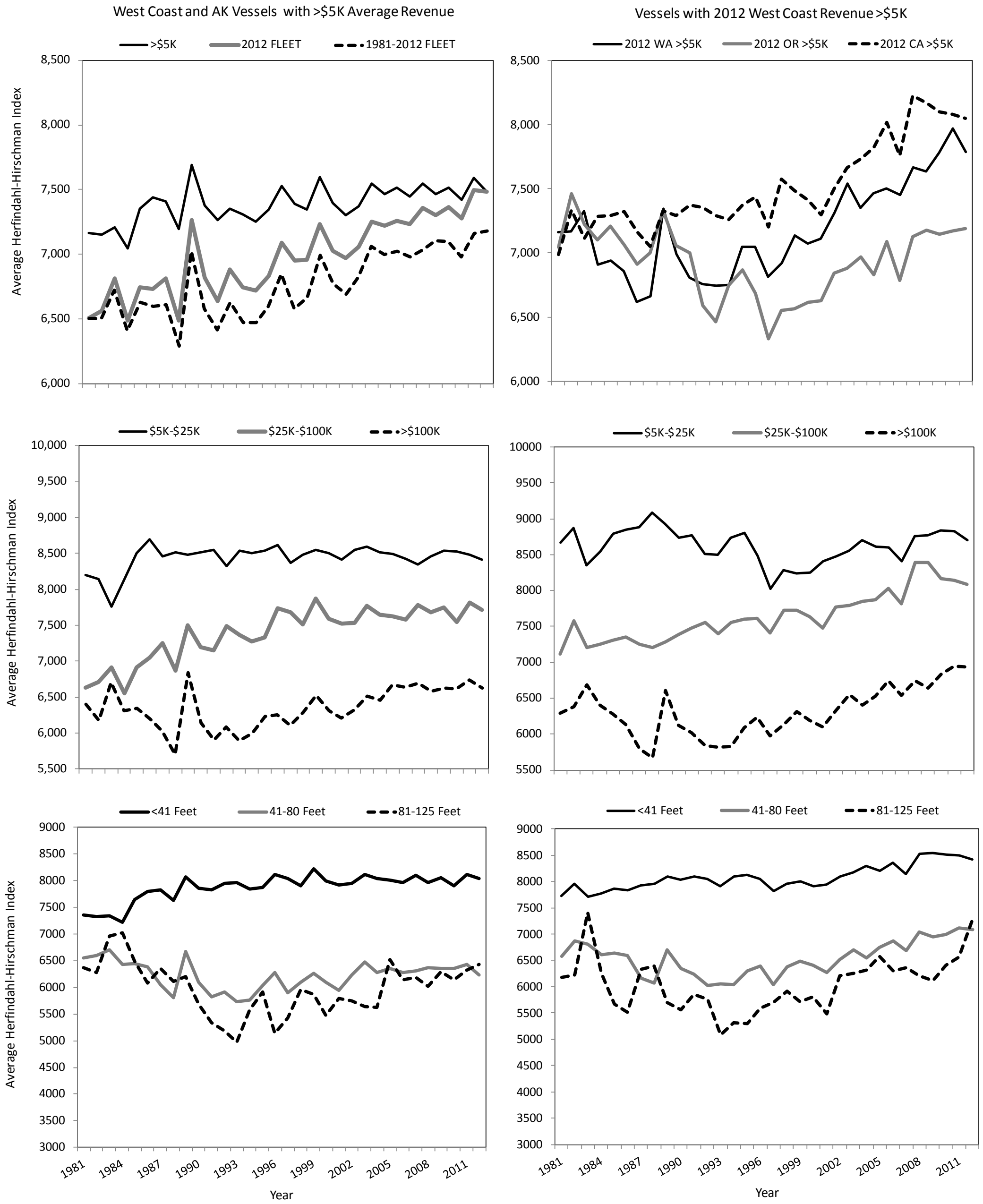

Figure HD1-1: Trends in average diversification for US West Coast and Alaskan fishing vessels (left panels) and the 2012 West Coast Fleet (right panel) filtered by all vessels with over $5,000 in average revenues (top panel), by average gross revenues classes (middle panel) and by vessel length classes (bottom panel).

While we can see some clear trends in diversification for various classes of vessels over time, there is wide variation in the degree of diversification across vessels within each class (Figure HD1-2). Higher-earning and large vessels tended to be more diversified on average than smaller vessels and those with lower earnings. The current (2012) West Coast fleet appears to be slightly less diversified on average than the larger fleet, which includes all vessels from the West Coast and Alaska, and both current and former participants.

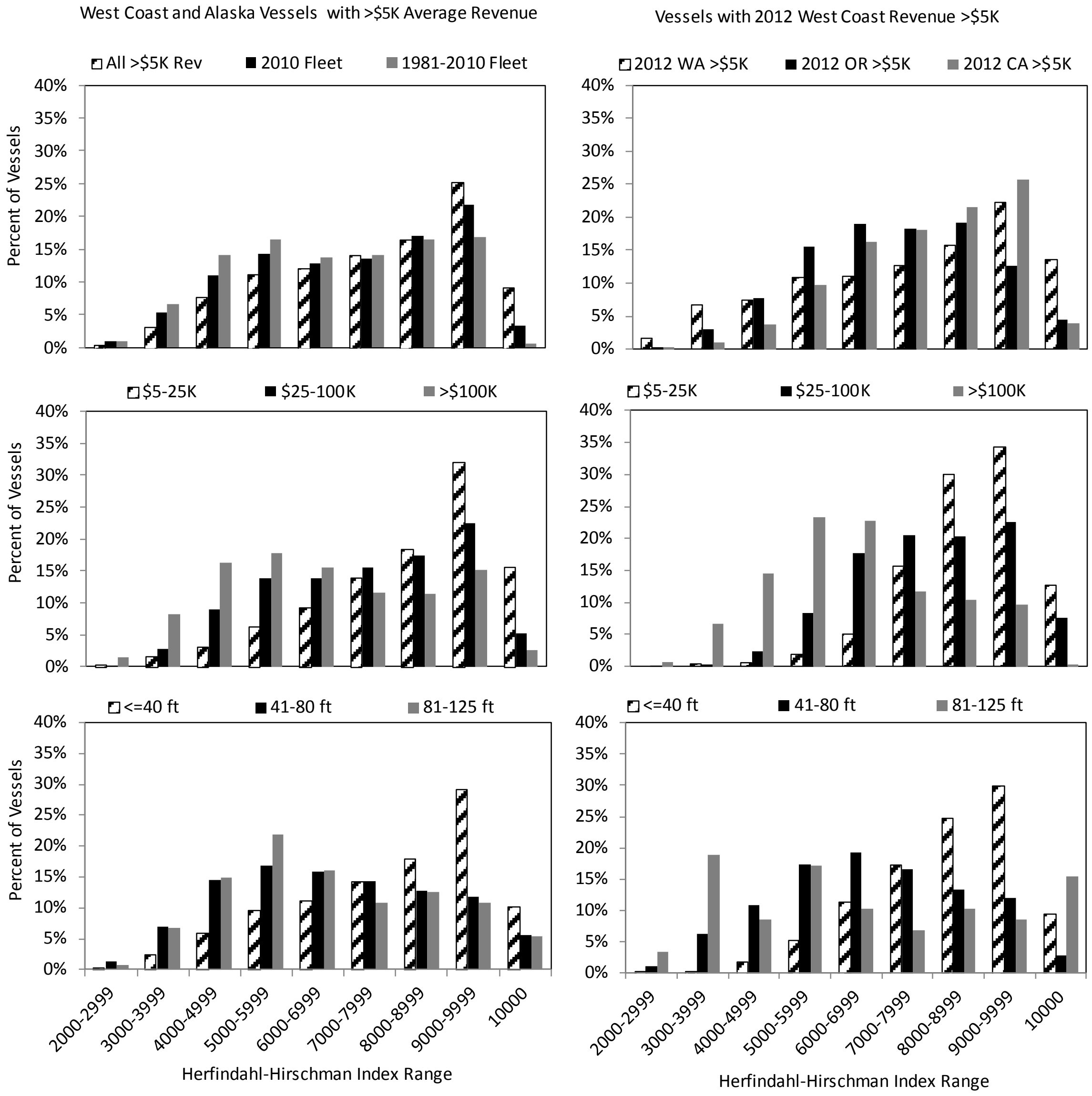

Figure HD1-2: Histograms showing percentage of vessels by ranges of Herfindahl-Hirschman index scores for US West Coast and Alaskan fishing vessels (left panels) and the 2012 West Coast Fleet (right panel) filtered by all vessels with over $5,000 in average revenues (top panel), by average gross revenues classes (middle panel) and by vessel length classes (bottom panel).

If vessels are able to diversify into multiple fisheries whose revenues vary independently or asynchronously, they should experience a reduction in volatility of revenues and thus financial risk. This is confirmed for all of our fleet groupings by estimating quadratic regressions of the CV of gross fishery revenue as a function of HHI and HHI2. Our analysis indicates a dome-shaped relationship between variability of individuals' incomes and income diversification, which implies that a small amount of diversification actually increases risk for some fleet categories, but moderate amounts of diversification can substantially reduce the variability of income that individuals receive from fishing. The decrease in CV with increased diversification varies substantially across vessel categories (Table HD1-2, Figure HD1-3), but for nearly all vessel categories there is a substantial decrease in CV when moving from a low level of diversification (e.g., a 90-10 split in revenues between two fisheries) to a high level of diversification (e.g., a 50-25-25 split between three fisheries). Annual revenues for fishing vessels in our sample have an average CV of 0.78. To illustrate how the decrease in CV associated with diversification affects the range of annual income a vessel might expect, we calculated the 50th percentile range of gross revenues for four hypothetical diversification schemes based on the functional relationship between HHI and CV for all vessels with mean annual revenues greater than $5,000. The 50th percentile range of expected revenues contracts from a range of $72,000 to $239,000, when all revenue comes from one fishery, to a range of $105,000 to $206,000 with a 50-25-25 split of revenues across three fisheries.

Table HD1-2: Predicted coefficient of variation (CV) of gross fishery revenue for Herfindahl-Hirschman index scores associated with alternative diversification schemes for groupings of WC and AK fishing vessels

	Predicted CV Herfindahl Index						
Vessel Category	Single Fishery	90-10 Split	50-50 Split	50-25-25 Split	%Drop Single Fishery to 50-25-25	Sample Size	Mean Revenue ($1000)
All >$5K Rev	0.80	0.85	0.66	0.48	23%	28,151	$ 155
2012 Fleet >$5K	0.68	0.75	0.60	0.45	33%	8,522	$ 272
1981-2012 Fleet >$5K	0.67	0.72	0.60	0.49	27%	2,577	$ 224
$5K-$25K Rev	0.86	0.94	0.75	0.55	36%	12,431	$ 12
$25K-$100K Rev	0.69	0.81	0.64	0.44	37%	10,329	$ 56
>$100K Rev	0.59	0.68	0.60	0.49	17%	5,391	$ 534
<40Feet	0.80	0.87	0.68	0.49	38%	21,848	$ 49
40-80 Feet	0.78	0.78	0.61	0.48	38%	5,269	$ 201
80-125 Feet	0.79	0.77	0.48	0.44	45%	612	$ 993
2012 WA >$5K	0.68	0.72	0.58	0.44	35%	917	$ 280
2012 OR >$5K	0.72	0.76	0.52	0.31	57%	808	$ 194
2012 CA >$5K	0.74	0.76	0.53	0.34	54%	1,359	$ 201
2012 WC $5-25K	0.79	0.90	0.50	0.14	82%	798	$ 16
2012 WC $25-100K	0.63	0.77	0.51	0.23	63%	1,048	$ 59
2012 WC >$100K	0.55	0.61	0.53	0.44	19%	898	$ 380
2012 WC <40 Feet	0.69	0.80	0.49	0.19	72%	1,618	$ 90
2012 WC 41-80 Feet	0.77	0.72	0.54	0.43	44%	1,065	$ 283
2012 WC 81 -125 Feet	0.64	0.66	0.52	0.39	38%	58	$ 1,177

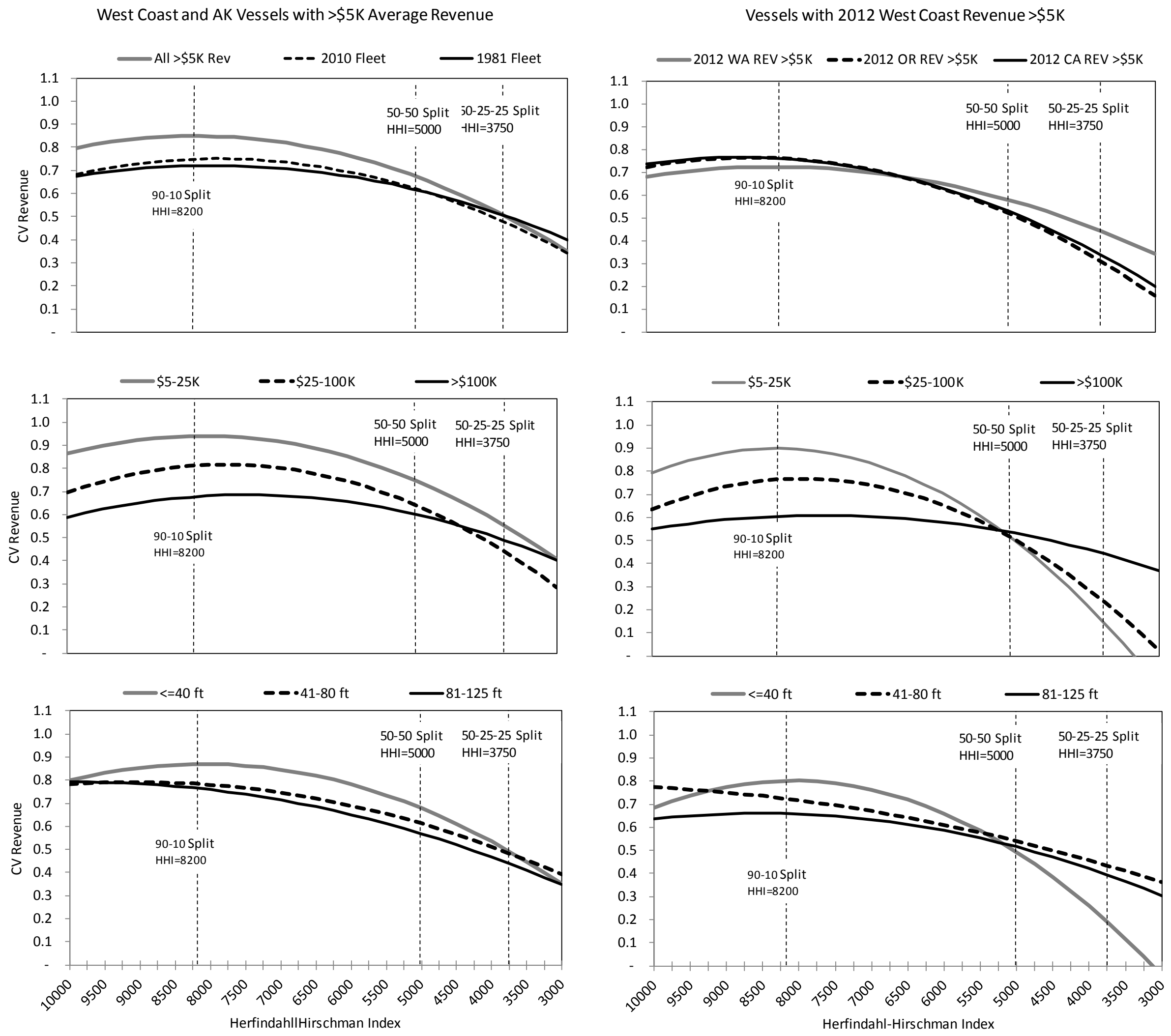

Figure HD1-3: Fitted relationships between the coefficient of variation (CV) of gross revenues for US West Coast and Alaskan fishing vessels (left panels) and the 2012 West Coast Fleet (right panel) filtered by all vessels with over $5,000 in average revenues (top panel), by average gross revenues classes (middle panel) and by vessel length classes (bottom panel).

Individual fishing ports experience a high degree of variation in diversification as well as landed revenue (Figures HD1-4 and HD1-5). Diversification of landed revenue for some ports has clearly decreased as evidenced by an increasing HHI. Examples include Seattle and most, though not all, of the ports in Southern Oregon and California. A few ports have become more diversified, including Bellingham Bay in Washington and Westport, Washington, which became less diversified through the mid-1990s but has since reversed that trend. Diversification scores at the port level are generally much lower than for individuals because port-level scores reflect landings of many different fishermen who individually may be less diversified but in aggregate land a variety of species. Diversification scores are highly variable for some ports, particularly those in Southern Oregon and Northern California that depend heavily on the Dungeness crab fishery. Crab revenue, and consequently overall landed value, in those ports over the last decade has varied dramatically year to year, which in turn drives variability in diversification (Figure HD1-5). When crab revenues are very high they dominate landed value for the port and drive up the HHI (i.e. lower diversification). HHI for Southern California ports has increased substantially in recent years as landed value from these ports has become increasingly dominated by squid.

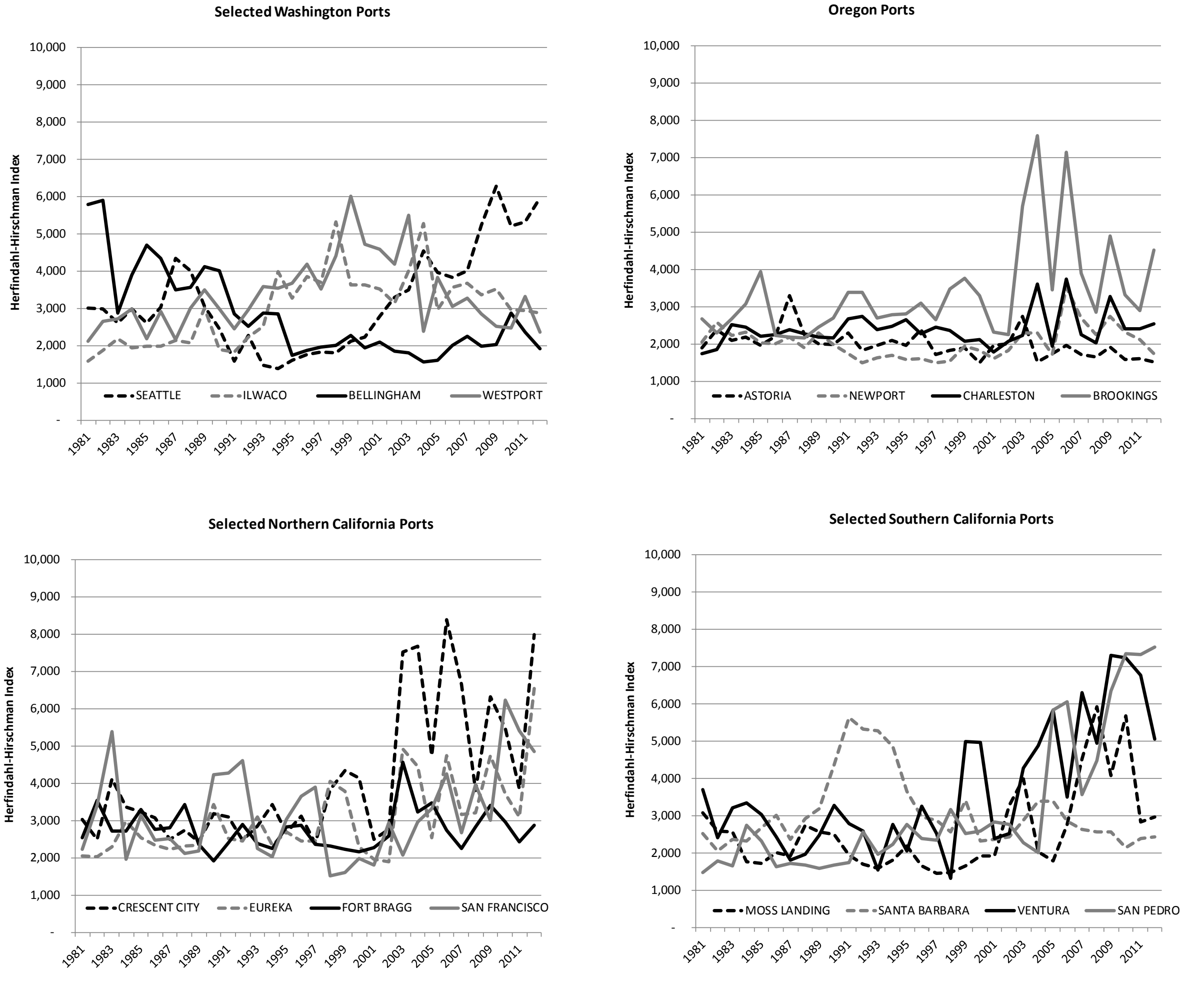

Figure HD1-4: Trends in diversification for selected primary West Coast ports in Washington, Oregon, and California.

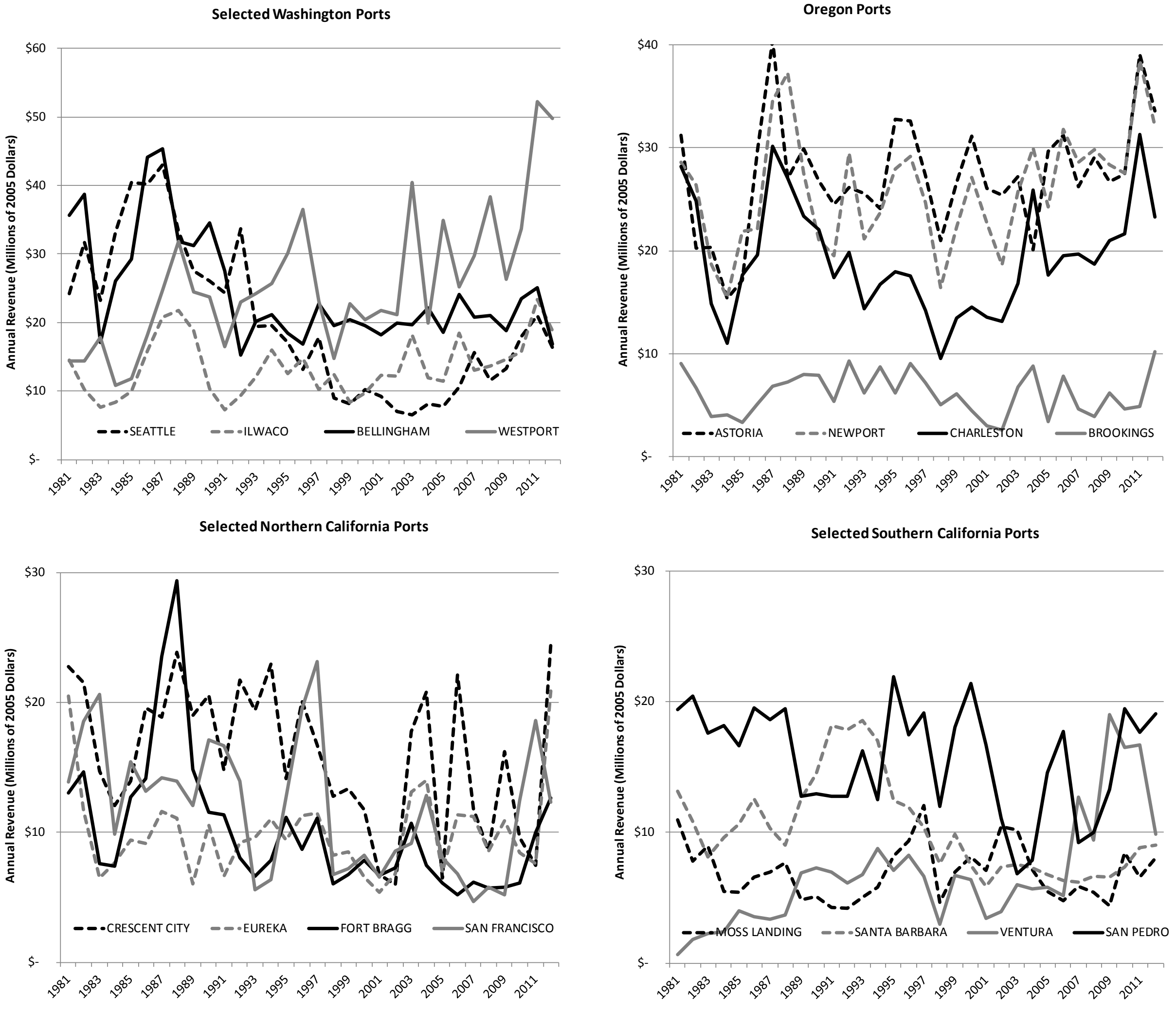

Figure HD1-5: Total landed value in 2005 dollars for selected primary West Coast ports in Washington, Oregon, and California.

As is true with individual vessels, the variability of landed value at the port level is correlated with HHI. The fitted relationship between the CV of annual landed value and HHI is domed-shape as it is for individual vessels, thus the predicted CV declines at an increasing rate as the diversification of the port increases (HHI declines) (Figure HD1-6). However, relative to the to the fitted relationship for vessels, the relationship between CV of annual landed revenues and HHI for ports has substantially more curvature and requires a much higher level of diversification to begin experiencing a decline in the CV of annual landed revenues (e.g., an HHI of 3,750, as with a 50-25-25 split).

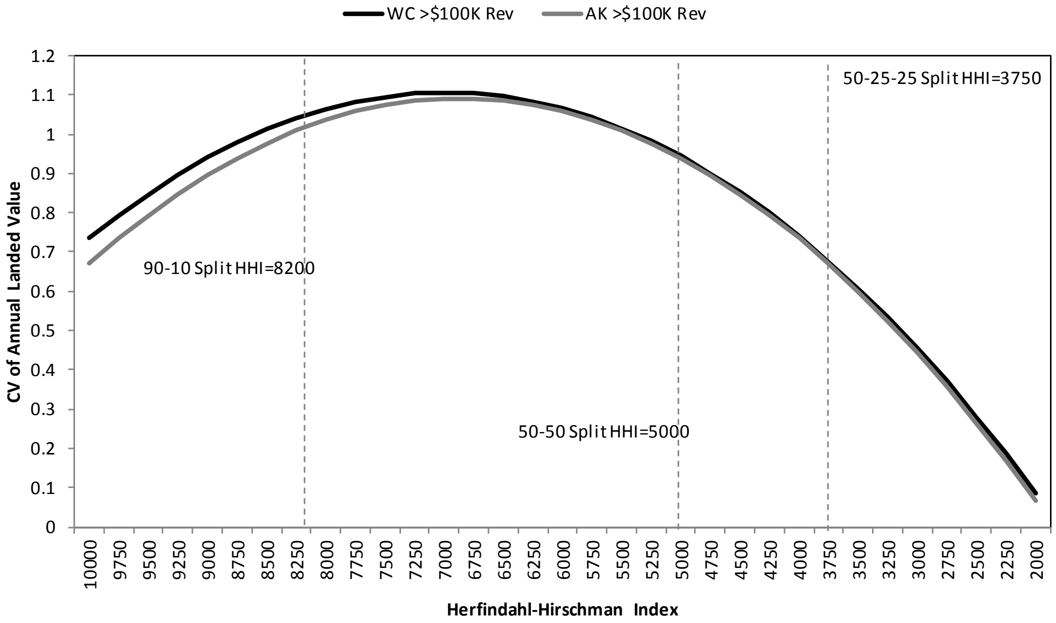

Figure HD1-6: Fitted relationships between the coefficient of variation (CV) of gross revenues for US West Coast and Alaskan fishing ports.

DISCUSSION

Diversification across multiple fisheries can reduce variation in catches and the associated financial risk. It can also increase the minimum annual revenue relative to average revenue, which should reduce the risk of a business failure (Kasperski and Holland, 2013). The ability of fishermen to diversify may be limited (or facilitated) by management approaches and regulatory actions. This should be a consideration when evaluating management actions, though in some cases management actions that reduce diversification are needed to remove excess capacity and promote efficiency.

There are a number of factors that may limit the feasibility or desirability of greater diversification. In many cases different fisheries require different gear that must be purchased and there are often costs of acquiring licenses and, increasingly, quota. It may also be the case that a vessel that can participate in several fisheries may be less efficient than more specialized vessels creating a trade-off between risk reduction through diversification and fishing efficiency. Exploration of this potential tradeoff would be an important extension of our research. Owners of multiple vessels can diversify by having individual vessels to specialize in different fisheries. Some fishermen may diversify their income with non-fishing sources. This seems particularly likely for vessels with low levels of revenue. We were unable to explore the degree or effects of this type of diversification due to a lack of data on non-fishing income. We hope to collect data on non-fishery income in future to explore this issue.

It is not clear that ports could or should increase diversification to reduce variation in landed value, but it does appear that high levels of diversification can reduce variation in landed value. High variation in overall landed value for several ports is associated with dependence on fisheries that have high variation in revenues. This variation could be socially disruptive, but this may be somewhat unavoidable if those ports want to continue to attract the landings from valuable fisheries like crab that have highly volatile annual landings. It should also be noted that the variation in landed value at ports is not necessarily closely correlated with variation in fishing income of fishermen living in those communities since those fishermen may be landing catch in other ports. The link between diversification of individual fishermen and ports and socio-economic wellbeing of communities is one that deserves further research.

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ASSESSMENT OF RISK DUE TO CLIMATE CHANGE FOR COASTAL PELAGIC SPECIES IN THE CALIFORNIA CURRENT MARINE ECOSYSTEM

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SOUND BITE

Forage fishes in northern, nearshore areas of the California Current marine ecosystem are most exposed and most sensitive to changes in sea surface temperature and chlorophyll projected to occur by 2100.

EXECUTIVE SUMMARY

In the ocean, forage species such as squid, anchovies, and sardines play a crucial role, serving as dominant lower trophic-level consumers, targets of some of the largest fisheries in the world, and essential food for higher trophic level species like marine mammals, seabirds, and larger fishes. Contemporary climate change has already changed the distribution and abundance of some of these species, while also modifying the timing and synchrony of important ecological and life history events. It has been challenging, however, to predict *a priori* which species, in which places and under which conditions, are most likely to be affected. Using projections of changes in oceanographic climate, we assessed risk to marine forage species in the California Current to the year 2100. On the basis of expected changes in the mean and variability of sea surface temperature and chlorophyll concentrations, and species-specific sensitivity to these changes, we ranked the relative risk to 15 species, many of which are or were valuable fisheries targets. We found that exposure to changes in oceanographic climate varied much less across the California Current than the sensitivity of individual species to those changes. By separating exposure and sensitivity components of risk to marine forage species, we provide insights into how to proactively develop climate change adaptation and mitigation strategies.

DETAILED REPORT

INTRODUCTION

Anthropogenic climate change is a major driver of ecological dynamics in terrestrial, freshwater, and marine ecosystems (Parmesan and Yohe 2003). Already, species have responded to changing climatic conditions in a variety of ways, including via altered phenology, distributional shifts, changes in synchrony with food and habitat resources, and in some cases extirpations and extinctions (Parmesan 2006, Doney et al. 2011). It has been challenging, however, to predict *a priori* which species, in which places and under which conditions, are most likely to be affected. Generating such expectations is a crucial first step to proactively developing climate change adaptation and mitigation strategies (Dawson et al. 2011).

Perhaps the most common method for assessing climate change impacts relies on species-distribution (or climate-envelope) models (Guisan and Thuiller 2005, Cheung et al. 2009). While one of the only tools for assessing the impacts of forecasted climate change on a wide range of species, alone such niche-based models may be too simplistic to resolve discrepancies between predicted and realized climate impacts (Dawson et al. 2011, Brander et al. 2013). More hopefully, recent evidence suggests that fine-scale climate data can help to improve on the predictions of species-distribution models by more accurately capturing location-specific changes in climate (Pinsky et al. 2013). However, these models are mostly projecting plastic responses of species to change and fail to consider species micro-evolutionary responses and adaptations.

For every rule related to expected climate change responses (e.g., expected poleward shifts and earlier spring blooms), there appear to be numerous exceptions. The same species can respond differently throughout its distribution, different species can respond differently in the same location, and responses to climate change clearly vary among communities and across ecosystems (Parmesan 2006, Burrows et al. 2011, Pinsky et al. 2013). While species-distribution models tend to account explicitly for the exposure of a species to changing climatic conditions, they generally make implicit assumptions about other aspects of a species' vulnerability, such as sensitivity and adaptive capacity (Turner et al. 2003, Williams et al. 2008).

Traditionally, ecological risk assessment models have considered a species' sensitivity equivalent to its response to a stressor visualized along a dose-response curve (Turner et al. 2003). This concept captures the idea that the identical stressor—whether it is nutrient limitation or human exploitation—can have dramatically different effects on different species, depending on their physiologies, prior conditioning, life histories, and behaviors (e.g., Vaquer-Sunyer and Duarte 2008; it is related in many ways to the evolutionary concept of phenotypic plasticity, Nussey et al. 2007). Similarly, a species' sensitivity to climate change is a function of many factors, including physiological tolerance limits, ecological traits (e.g., behavior, Kearney et al. 2009), and genetic diversity (Chevin et al. 2010, Phillimore et al. 2010). Whereas niche-based models project current and past understanding of a species-environment relationship (the plastic response), they fail to consider evolutionary constraints to adaptability. Under the assumption that adaptability is proportional to genetic diversity (A = S^2 * H * delta, where S is genetic diversity, H = heritability and delta is the selection gradient, akin to exposure), the variety of niches that a species occupies throughout its range is an expression of its genetic diversity, and therefore proportional to its adaptive capacity (Arnold 1992, Futuyma 2010, Hutchings 2011).

If species today are adapted to current environmental conditions, a reasonable first approximation is that species occurring in climatically consistent environments (through space or time) will be more sensitive (and less able to adapt) to climate change than species living in climatically variable environments (cf. Rapaport 1982, Gaston 2003, Gaston and Fuller 2009, Dickinson et al. unpublished). Thus, the climatic breadth of a species today may be inversely related to its sensitivity to, and adaptive capacity for, climate change in the future. Remotely-sensed data and global climate models, when coupled to information on contemporary species distributions, offer an unprecedented opportunity to estimate the sensitivity of many species to climate change based on their climatic breadths, using a rapid, uniform method.

In this paper, we bring together measures of both exposure and sensitivity to estimate risk of population change for marine forage species due to changes in oceanographic climate. By doing so we integrate our best understanding of potential evolutionary and ecological responses of forage species to climate change. Forage species play a crucial role in marine ecosystems, serving as dominant consumers, targets of some of the largest fisheries in the world (e.g., Peruvian anchoveta), and food for higher trophic level species like marine mammals, seabirds, and larger fishes (Pikitch et al. 2014). Populations of forage species are notoriously variable in size (Pinsky et al. 2011), and in many cases this variability is thought to be linked to changes in biophysical factors in the ocean such as sea surface temperature and primary productivity (e.g., Cushing 1988, Chavez et al. 2003, Baumgartner et al. 1992, Sydeman et al. 2013). Despite the widely acknowledged connections between climate and the dynamics of forage species, we are not aware of any other efforts to bring together climate forecasts, along with exposure and sensitivity estimates, for multiple forage species in any particular geographic domain. Our analysis thus provides insights into population- and community-level risk to marine forage species, with implications for fisheries, food webs, and the utility of alternative management responses in the California Current, a highly productive yet spatially and temporally variable upwelling ecosystem (Checkley and Barth 2009).

METHODS

We assessed risk due to climate change for marine forage species within the California Current ecosystem. Specifically, we focused on 10 coastal pelagic fishes and invertebrates (Table 1) that are well-sampled in long-term monitoring studies of pelagic nekton, during life stages in which they are considered forage species (Brodeur et al. 2003).

Risk was defined as each species' relative exposure and sensitivity to changes in oceanographic climate expected by the year 2100. Following Dawson et al. (2011), Samhouri and Levin (2012), Foden et al. (2013), and others, those species that were highly exposed and highly sensitive were considered most at risk due to climate change. In order to quantify exposure and sensitivity, we first defined each species' current distribution and the oceanographic climate within that distribution for the recent past (1976-2005). We also defined the oceanographic climate within the historic distribution at the end of this century (to 2100).

We developed an extent of occurrence (EOO; Gaston and Fuller 2008) layer to define the current distribution of each species. This layer was based on data from IUCN (www.iucnredlist.org) and AquaMaps (www.aquamaps.org), such that all 50 km x 50 km cells with a probability of occurrence ≥0.4 were scored as present for each species. In the future, we hope to develop species distribution maps based on annual and seasonal average abundance estimates from fisheries-independent surveys.

Oceanographic climate was defined for the entire planet in each 50 km x 50 km grid cell based on four variables correlated with the abundance of marine forage species within the California Current. These variables included: mean sea surface temperature μ_{SST}, mean chlorophyll *a* μ_{CHL}, standard deviation in sea surface temperature σ_{SST}, and standard deviation in chlorophyll *a* σ_{CHL} (Brodeur et al. 2005, Emmett et al. 2005, Litz et al. 2008, Kaltenberg et al. 2010, Zwolinski et al. 2011). The historic oceanographic climate was defined based on monthly averages of the Geophysical Fluid Dynamics Laboratory's (GFDL) Earth System Model 2G (ESM2G) (Dunne et al. 2012) forced with historical estimates of atmospheric composition. Here, we selected a 30-year period from 1976-2005 as representative of the current climate. The future oceanographic climate was defined based on ESM2G monthly projections following IPCC scenario rcp 8.5 for the 30-year period 2071-2100.

The exposure of each species was estimated based on the magnitude of expected change in a multivariate climatic index, given the current observed distribution of each species within the California Current ecosystem. This approach provides a species-specific estimate of the expected "climate velocity" (Burrows et al. 2011, Pinsky et al. 2013) over this century. To derive the multivariate climatic index, we conducted a principal components analysis (PCA) on global ocean values for μ_{SST}, μ_{CHL}, σ_{SST}, and σ_{CHL} using a correlation matrix. We used outputs from a PCA, rather than estimates of change in the original climate variables, in order to account for covariance between the variables. Prior to performing the PCA, we examined the pairwise relationships among oceanographic climate variables in the historic and future periods, and determined that they were qualitatively similar. Therefore, we pooled oceanographic climate data from both time periods for the PCA. Each principal component was treated as an axis of a 4-dimensional global climate space. Principal component scores were calculated from rescaled climate variables (centered on zero and scaled to standard normal). See the Appendix for additional details about the PCA.

We estimated exposure of the *i*th species E_i as the Euclidean distance between historic *h* and future *f* principal component scores in each of the four *j* dimensions of the global climate space within each grid cell of each species range within the California Current ecosystem (equation 1),

$$E_i = \sqrt{\sum_{j=1}^{4}(f_j - h_j)^2}\,. \tag{1}$$

Note that this approach is agnostic as to whether increases or decreases in principal component scores are thought to have a positive or negative influence on the probability of a species' persistence. It simply assumes that larger differences between the future and historic periods (greater climate velocities) signify greater exposure to climatic change.

The sensitivity of each species to climate change was defined as the inverse of its historic climatic breadth. Following Dickinson et al. (unpublished ms), we quantified climatic breadth as the range of values of the multivariate climatic index experienced by each species during the historic period. Species with greater climatic breadth were assumed to be less sensitive to future climatic change. This approach assumes that (i) species are adapted to current climate, and (ii) species currently exposed to a broad range of climate variation will, all other things being equal, be able to withstand a greater degree of climate change than species currently exposed to a narrow range of climate.

For each of the *i* species we estimated sensitivity S_i as the inverse of the average distance between historic PC scores (in each of the four *j* dimensions of the global climate space) for the 97.5% quantile $h_{j,upper}$ and 2.5% quantile $h_{j,lower}$ of grid cells included within each species global geographic distribution (equation 2),

$$S_i = \frac{1}{\sum_{j=1}^{4}(h_{j,upper} - h_{j,lower}) / 4}\,. \tag{2}$$

The denominator in equation 2 thus represents our estimate of climatic breadth. In the Appendix, we show that an alternative estimate of sensitivity, based simply on the geographic area of each species range, was highly correlated with climatic breadth.

The relative risk R_i to each of the *i* species was estimated as

$$R_i = \sqrt{E_i^2 + S_i^2}\,, \tag{3}$$

implying that risk was expected to increase with Euclidean distance from the origin in the exposure-sensitivity space and that each axis received equivalent weight. Note that to ensure independence between these 2 components of risk, we first tested the rank correlation between them. Below, we report the average values of E_i, S_i, and R_i across each species' distribution.

In addition to summarizing R_i values for each species individually, we also examined geographic variation in the exposure, sensitivity, and risk due to climate change for the marine forage species community. Specifically, we mapped the exposure $E_{community}$ and sensitivity $S_{community}$ of the marine forage species community in each grid cell k as the sum of exposure and sensitivity values for each of the i species, or

$$E_{community,k} = \sum_{i=1}^{N} E_i \qquad (4)$$

and

$$S_{community,k} = \sum_{i=1}^{N} S_i \,. \qquad (5)$$

We estimated risk to the marine forage species community as the Euclidean distance from the origin in the summed exposure and sensitivity space,

$$R_{community,k} = \sqrt{{E_{community,k}}^2 + {S_{community,k}}^2}\,. \qquad (6)$$

These community estimates allowed us to identify geographic concentrations of low and high exposure, sensitivity, and risk within the California Current ecosystem.

RESULTS

Oceanographic climate projections for the historic (Figs. 1a, c, e, g) and future (Figs. 1b, d, f, h) time periods showed substantial latitudinal clines for average and standard deviation values in sea surface temperature and chlorophyll a concentrations (Dunne et al. 2013, and references therein). Principal components analysis of pooled data from the historic and future periods distilled this multivariate variation into 4 new orthogonal axes. The first 2 principal components explained >80% of the variation (Fig. 2; Table 1), such that PC1 was positively correlated to warmer average sea surface temperatures and lower average chlorophyll a concentrations and PC2 was positively correlated to reduced variability in sea surface temperatures. The resulting global maps of historic and future oceanographic climate, plotted in principal component space (Figs. 1i-p), allowed us to derive a single exposure score (following equation 1) for each grid cell within the California Current (Fig. 3), and for each marine forage species within that geographic area (see Figs. 4a-b for an example with northern anchovy). We also estimated a globally-based sensitivity score for each species using these maps (following equation 2; Figs. 4a, c).

Estimates of exposure varied by ~20% among species, whereas sensitivity estimates spanned an order of magnitude more variation (~200%; Fig. 5). Exposure and sensitivity scores were not significantly correlated across species (Spearman rank correlation = -0.125, p = 0.66). Overall, risk due to climate change for each of the 10 marine forage species varied widely, with *Scomber japonicus* exhibiting the lowest risk and *Allosmerus elongatus* the highest risk (Fig. 5).

Community exposure, sensitivity, and risk showed similar geographic gradients (Fig. 6). In general, forage species were more exposed, sensitive, and at risk in northern, coastal areas of the California Current than they were in southern, offshore areas. This cline was somewhat stronger for sensitivity estimates than it was for exposure and risk, as highly exposed and at risk communities extended further offshore than did highly sensitive communities.

DISCUSSION

Climate change is expected to have major effects on the viability of populations on land and in the sea. Understanding their risk of decline due to climatic change is key to predicting the consequences for biodiversity, and for people that rely on some of these species for food directly and indirectly. Indeed, this risk analysis can be viewed both as an assessment of risk to the forage species themselves, and to the fisheries and predators that rely on them.

Risk has been assessed in multiple ways, including via climate envelope models and through vulnerability frameworks that separate exposure from sensitivity. There are pros and cons of assessing risk each way. Ensembles of approaches are best, and there is a real need to balance the vast climate envelope modeling literature with other approaches. A vulnerability framework (*sensu* Turner et al. 2003), like the one presented here, provides a convenient method for predicting climate change impacts by integrating exposure and sensitivity estimates. A species that is highly exposed but not very sensitive may have the same predicted overall vulnerability to climate change as a species that is highly sensitive but not very exposed, yet very different management actions may be warranted for these alternative scenarios (Dawson et al. 2011).

For highly sensitive species, in order to reduce the probability of a population decline, one appropriate management response may be to reduce the magnitude of non-climate stressors (eg harvest rates) on them. Doing so may mitigate negative impacts of changes in climate. Similarly, for highly exposed species, it may make sense to redistribute non-climate stressors in areas of lower exposure, where possible. Furthermore, it may make sense to displace fishing effort away from marine forage species that are highly exposed or highly sensitive to climate change, by encouraging alternative, less exposed and less sensitive target species.

A major limitation of this risk assessment, as applied here, relates to the spatial resolution of the global climate model. The model has a spatial resolution that precludes great representation of the coastal upwelling domain. It has upwelling, but not to the right magnitude; kind of smearing it out over the first grid points (hundred of km) rather than right along the coast. So it is most appropriate for asking broad-scale questions, but not necessarily for resolving different types of habitat within the upwelling system. For spatial domains like the California Current Large Marine Ecosystem, it may mean that the model cannot project the right spatial gradient in the magnitude of change in the mean and variability of SST and chlorophyll, even qualitatively. However, scientists at NOAA GFDL are in the process of developing a downscaled model for the California Current that will better capture nearshore oceanographic features. The approach presented here will be best applied when that model is ready.

Among the many caveats here is the assumption that species ranges do not change nor expand in the future. This is a very coarse assumption for mobile, pelagic species such as sardine and mackerel. For instance, King et al. (2011) considered conceptual pathways linking climate change to Pacific sardine, and suggested that in the future we may expect higher abundances of sardine off Vancouver Island, and potentially even resident populations. Such details of life history and ocean condition are not captured by the risk assessment methods here.

In a thoughtful and comprehensive chapter, Freon et al. (2009) consider potential effects of climate change on small pelagic fishes in the California Current and other systems. These authors discuss key physical processes (beyond simple warming) such as changes in stratification, upwelling, and intensity of the California Current. These processes in turn may lead to changes in temperature and primary production, but also shifts in the zooplankton food web, altered advection of fish larvae, and changes to phenology essential for successful recruitment. However, beyond recognizing the role of these processes, precise predictions of these complex interactions are not possible, given the resolution of the current generation of models such as ESM2G (50km grid scale) and the limits of present ecological knowledge. Thus we offer the present framework as a first step toward prioritizing species and regions that are most at risk, based on expected deviations of future conditions from the present.

In future risk assessments related to expected changes in climate within the California Current, it will be interesting to:

- Pair the analyses presented here with information on non-climate stressors, e.g. levels of harvesting of individual species.
- Examine changes in estimates of risk based on alternative species distribution data, beyond AquaMaps.
- Relax the assumption, implicit in our analysis, that species that occupy the same grid cell have the same climate sensitivity, regardless of differences in life history or other characteristics.
- Investigate alternative measures of community risk, including one that weights risk according to the abundance or biomass of each species.
- Determine whether greater community risk in more northern, nearshore areas is simply a consequence of more species co-occurring in that region than in southern and/or offshore areas.
- Limit or expand the analysis to the spawning period of the life-cycle for each species, in order to avoid underestimating risk for highly migratory species (e.g., sardine and hake).

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TABLES

Table 1. Marine forage species considered in the risk assessment. These species are commonly sampled in the BPA survey (Brodeur et al. 2003).

Scientific name	**Common name**
Allosmerus elongatus	Whitebait smelt
Clupea pallasii	Pacific herring
Cololabis saira	Pacific saury
Engraulis mordax	Northern anchovy
Hypomesus pretiosus	Surf smelt
Loligo. opalescens	Market squid
Merluccius productus	Pacific hake
Scomber japonicas	Pacific mackerel
Sardinops sagax	Pacific sardine
Trachurus symmetricus	Jack mackerel

Table 2. Results of principal components analysis on pooled output from historic and future projections of the GFDL model.

CORRELATION MATRIX FOR POOLED HISTORIC AND FUTURE DATA				
	sst_mean	sst_sd	chl_mean	chl_sd
sst_mean	1	-0.0773	-0.5156	-0.6810
sst_sd	-0.0773	1	0.1724	-0.1315
chl_mean	-0.5156	0.1724	1	0.6314
chl_sd	-0.6810	-0.1315	0.6314	1
EIGENVECTOR LOADINGS				
	PC1	PC2	PC3	PC4
sst_mean	0.5717	-0.0014	0.6910	0.4423
sst_sd	-0.0501	-0.9544	-0.1295	0.2641
chl_mean	-0.5573	-0.1755	0.7111	-0.3912
chl_sd	-0.6001	0.2414	0.0088	0.7626
PERCENT OF VARIATION EXPLAINED BY EACH PC				
	PC1	PC2	PC3	PC4
	55.6046	26.6206	11.8835	5.8914

FIGURES

Figure 1. (a-h) Maps of GFDL Earth System Model 2G global hindcasts (historic: 1976-2005) and forecasts (following IPCC scenario rcp 8.5, future: 2071-2100) of mean and standard deviation in sea surface temperature (SST) and chlorophyll a (CHL). (i-p) Maps of principal component scores based on principal component analysis of GFDL model output (see Methods for details).

Figure 2. Results of principal component analysis of GFDL Earth System Model 2G global hindcasts (historic: 1976-2005; black symbols) and forecasts (following IPCC scenario rcp 8.5, future: 2071-2100; blue symbols) of mean and standard deviation in sea surface temperature and chlorophyll a.

Figure 3. Exposure, the magnitude of expected change in a multivariate climatic index within the US California Current ecosystem.

Figure 4. Example (a) species distribution, and calculation of (b) exposure and (c) sensitivity for *Engraulis mordax*. We estimated exposure as the Euclidean distance between historic and future principal component scores in each of the four dimensions of the global climate space within each grid cell of each species range within the US California Current ecosystem (see equation 1). We estimated sensitivity as the inverse of the average distance between historic PC scores (in each of the four dimensions of the global climate space) for the upper 97.5% and lower 2.5 of grid cells included within each species global geographic distribution (see equation 2).

Figure 5. Preliminary risk scores.

Figure 6. Preliminary community exposure, sensitivity, and risk scores.

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Figures

Figure 1. (a-h) Maps of GFDL Earth System Model 2G global hindcasts (historic: 1976-2005) and forecasts (following IPCC scenario rcp 8.5, future: 2071-2100) of mean and standard deviation in sea surface temperature (SST) and chlorophyll a (CHL). (i-p) Maps of principal component scores based on principal component analysis of GFDL model output (see Methods for details).

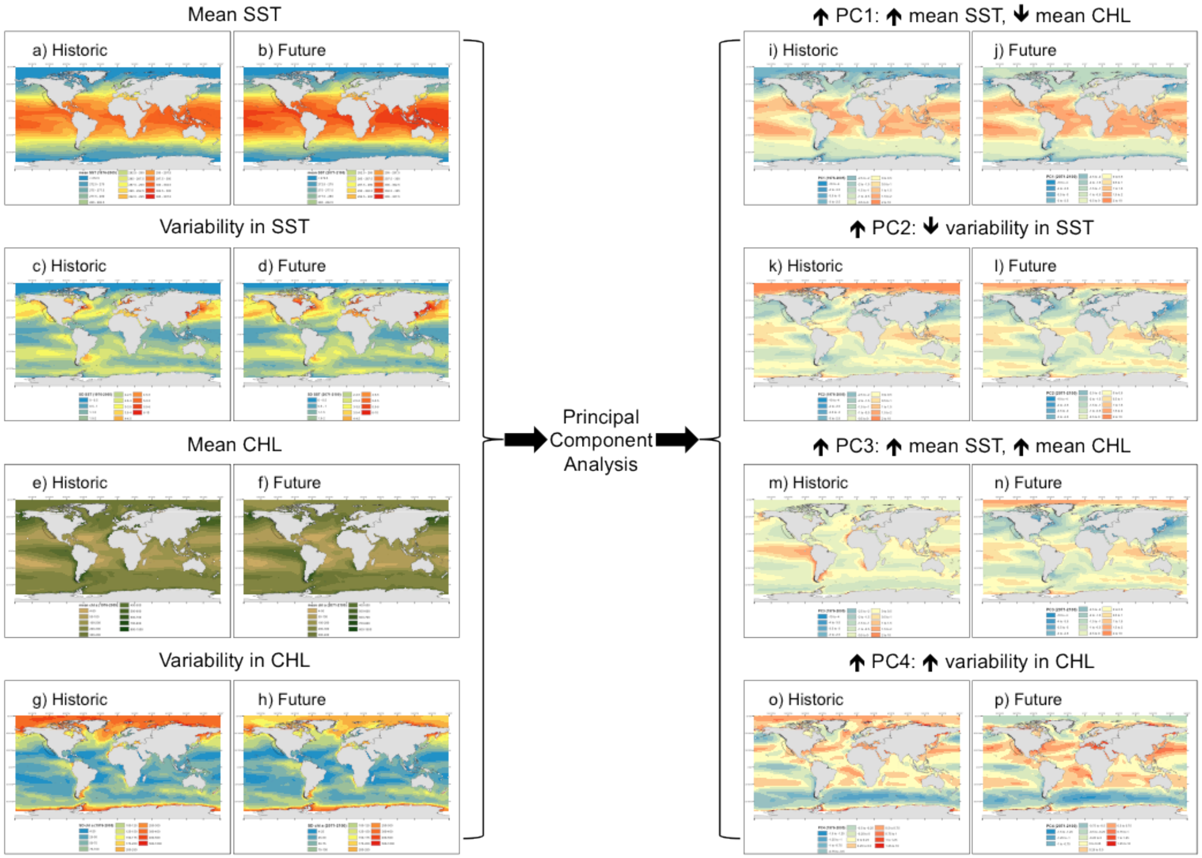
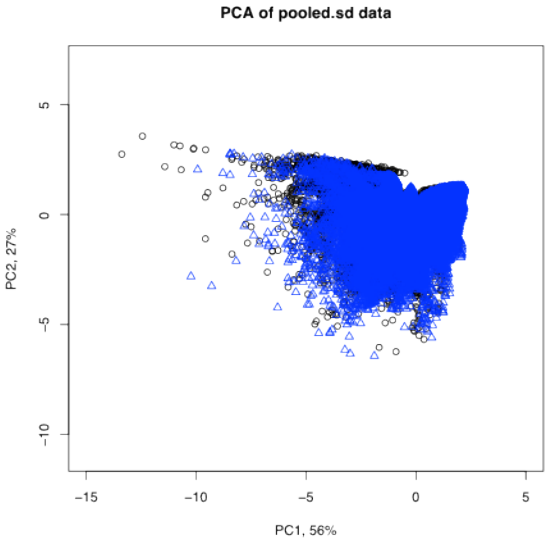
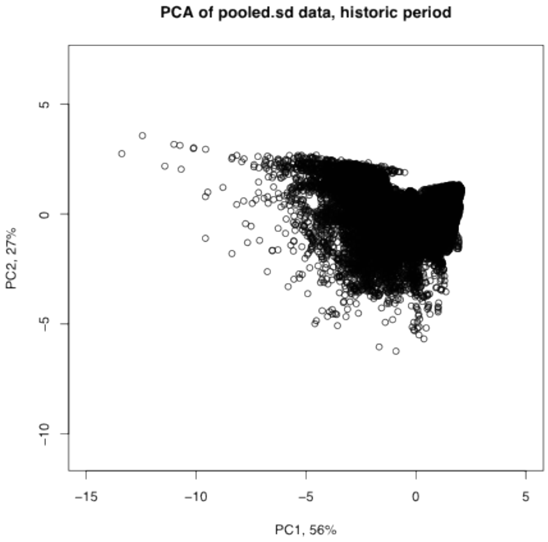


Figure 2. Results of principal component analysis of GFDL Earth System Model 2G global hindcasts (historic: 1976-2005) and forecasts (following IPCC scenario rcp 8.5, future: 2071-2100) of mean and standard deviation in sea surface temperature and chlorophyll a.

a) PCA showing historic and future periods together



b) PCA showing historic period



c) PCA showing future period

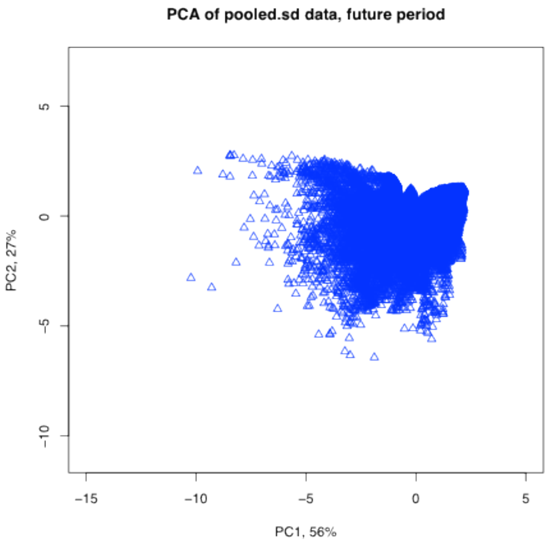

PCA of pooled.sd data, future period
PC2, 27%
PC1, 56%
5
0
−5
−10
−15
−10
−5
0
5

Figure 3. Exposure or climate velocity, the magnitude of expected change in a multivariate climatic index within the California Current ecosystem.

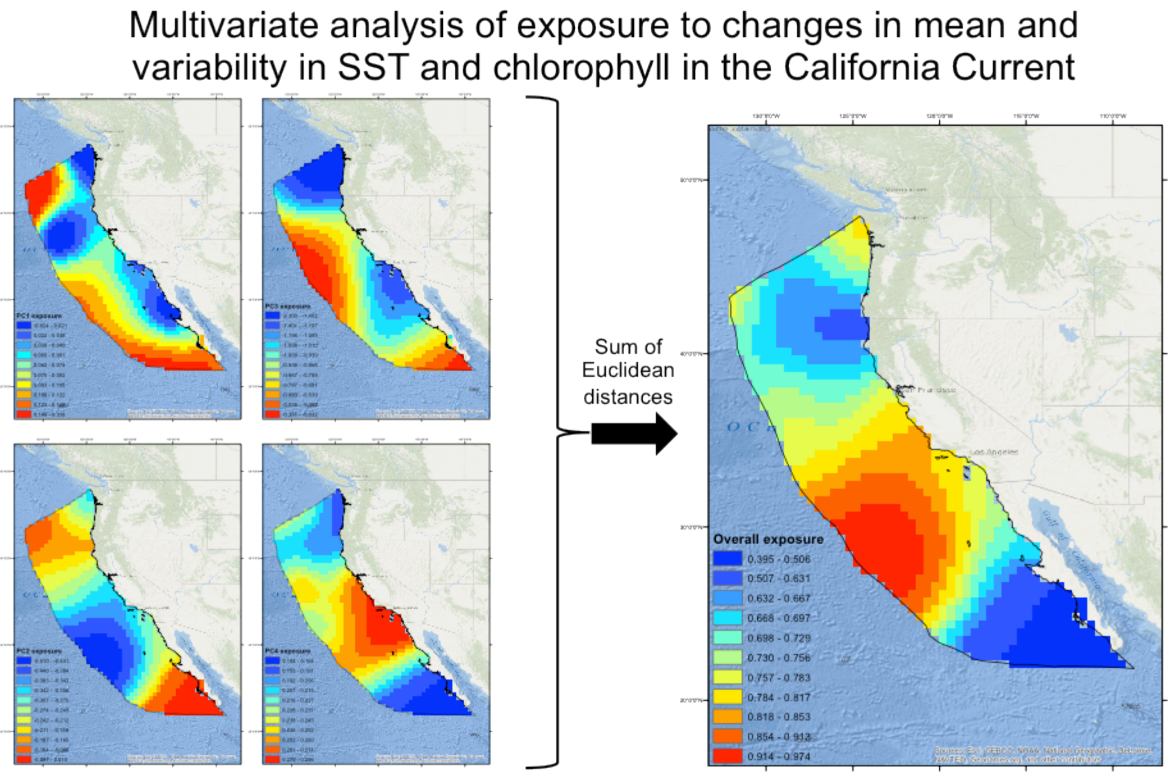
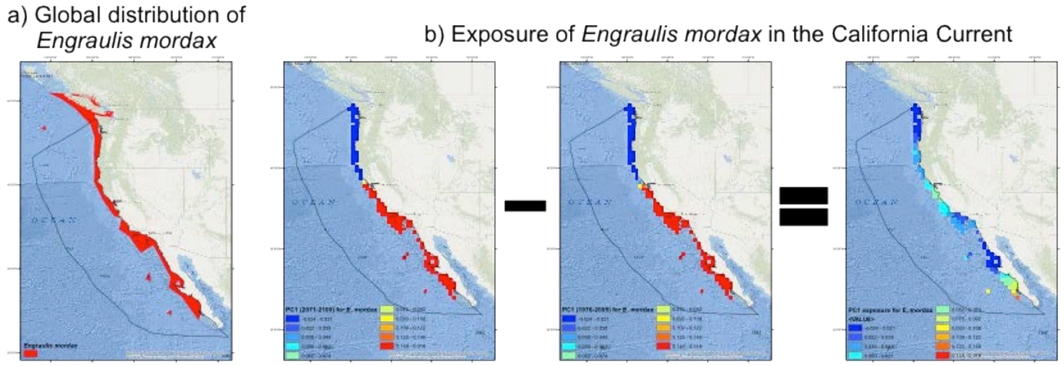
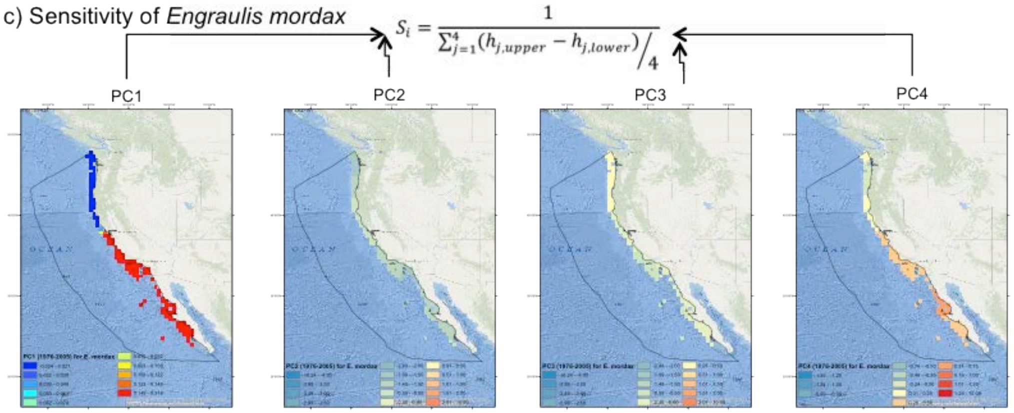


Figure 4. Example (a) species distribution, and calculation of (b) exposure and (c) sensitivity for *Engraulis mordax*. We estimated exposure as the Euclidean distance between historic and future principal component scores in each of the four dimensions of the global climate space within each grid cell of each species range within the US California Current ecosystem (see equation 1). We estimated sensitivity as the inverse of the average distance between historic PC scores (in each of the four dimensions of the global climate space) for the upper 97.5% and lower 2.5 of grid cells included within each species global geographic distribution (see equation 2).


a) Global distribution of *Engraulis mordax*

b) Exposure of *Engraulis mordax* in the California Current


c) Sensitivity of *Engraulis mordax*

$$S_i = \frac{1}{\sum_{j=1}^{4}(h_{j,upper} - h_{j,lower})/4}$$

Figure 5. Preliminary risk scores.

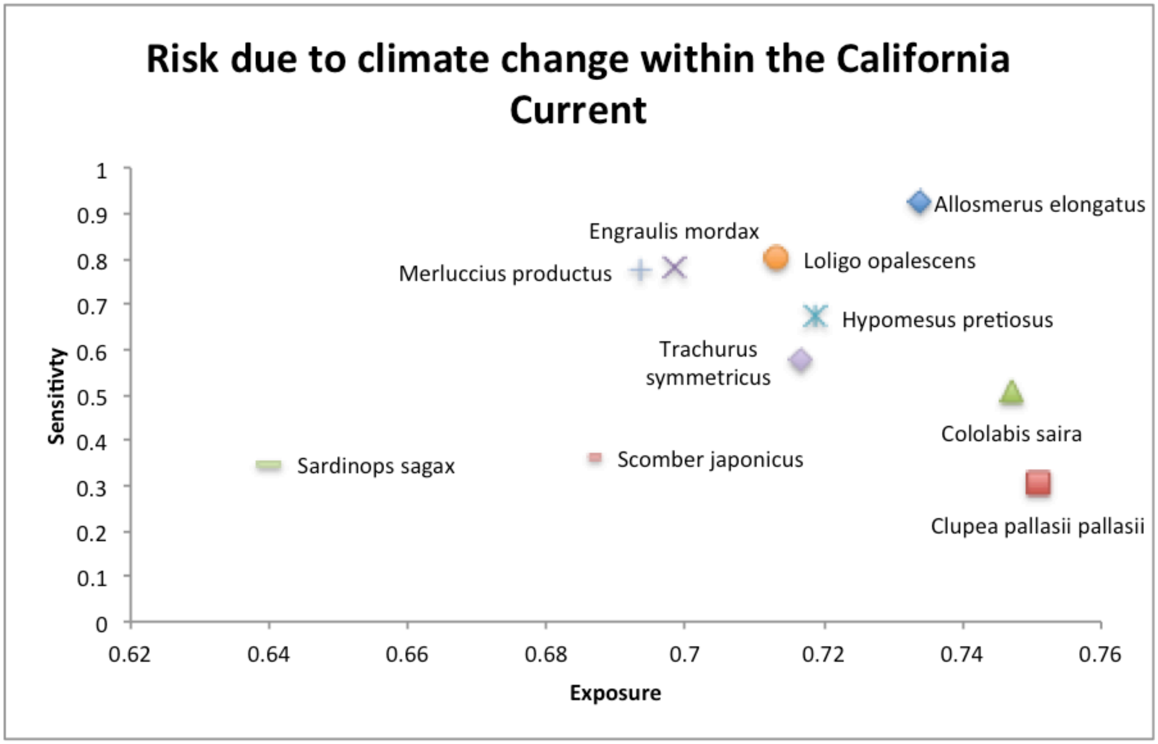
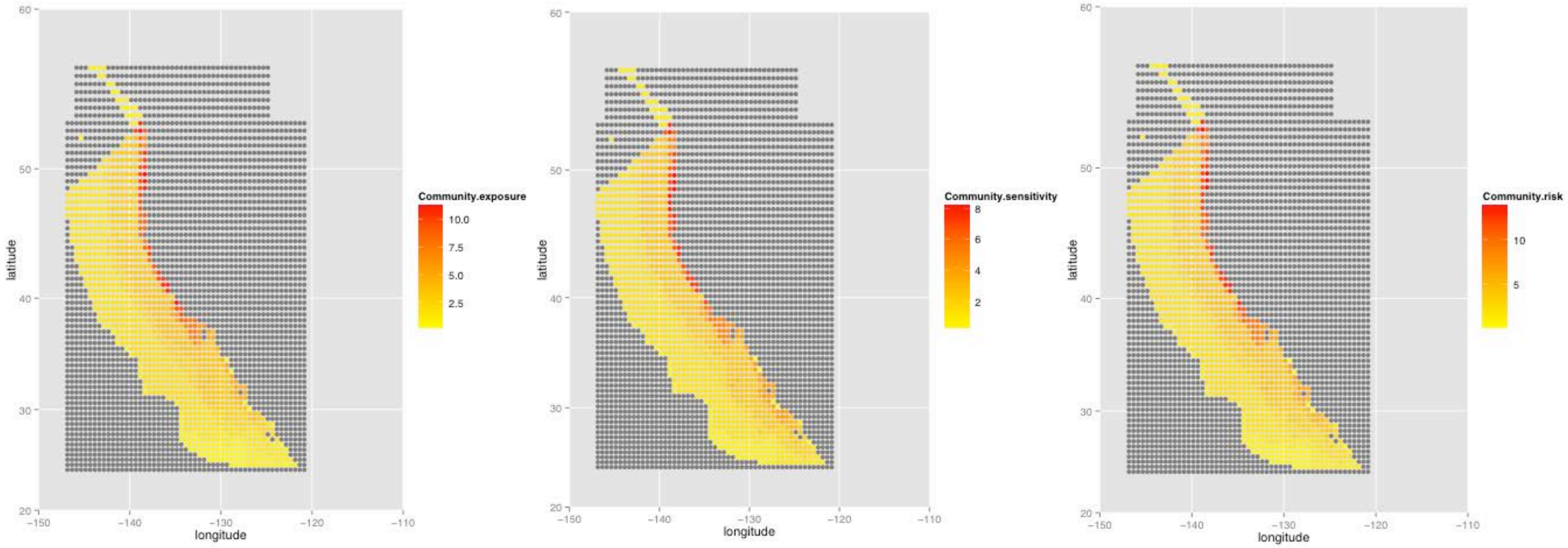


Figure 6. Preliminary community exposure, sensitivity, and risk scores.



Appendix

PCA details

Correlation matrix for mean and SD historic data:

	sst_mean	sst_sd	chl_mean	chl_sd
sst_mean	1	0.005853084	-0.554346	-0.701783
sst_sd	0.005853084	1	0.0899618	-0.292243
chl_mean	-0.55434549	0.089961798	1	0.6478051
chl_sd	-0.70178284	-0.29224314	0.6478051	1

Correlation matrix for mean and SD future data:

	sst_mean	sst_sd	chl_mean	chl_sd
sst_mean	1	-0.1703871	-0.474776	-0.6747135
sst_sd	-0.170387	1	0.2705106	0.09804175
chl_mean	-0.474776	0.27051064	1	0.62167702
chl_sd	-0.674714	0.09804175	0.621677	1

Figure A1. Draftsman plot for mean and SD historic data

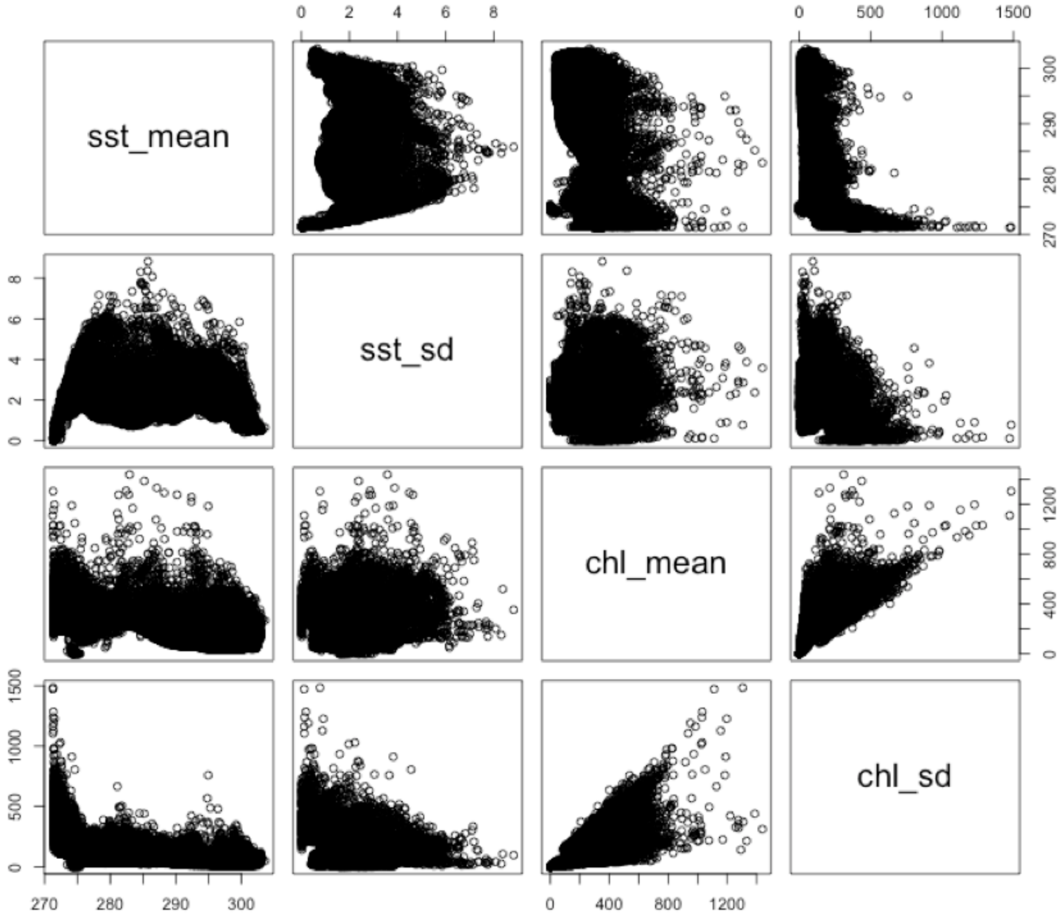
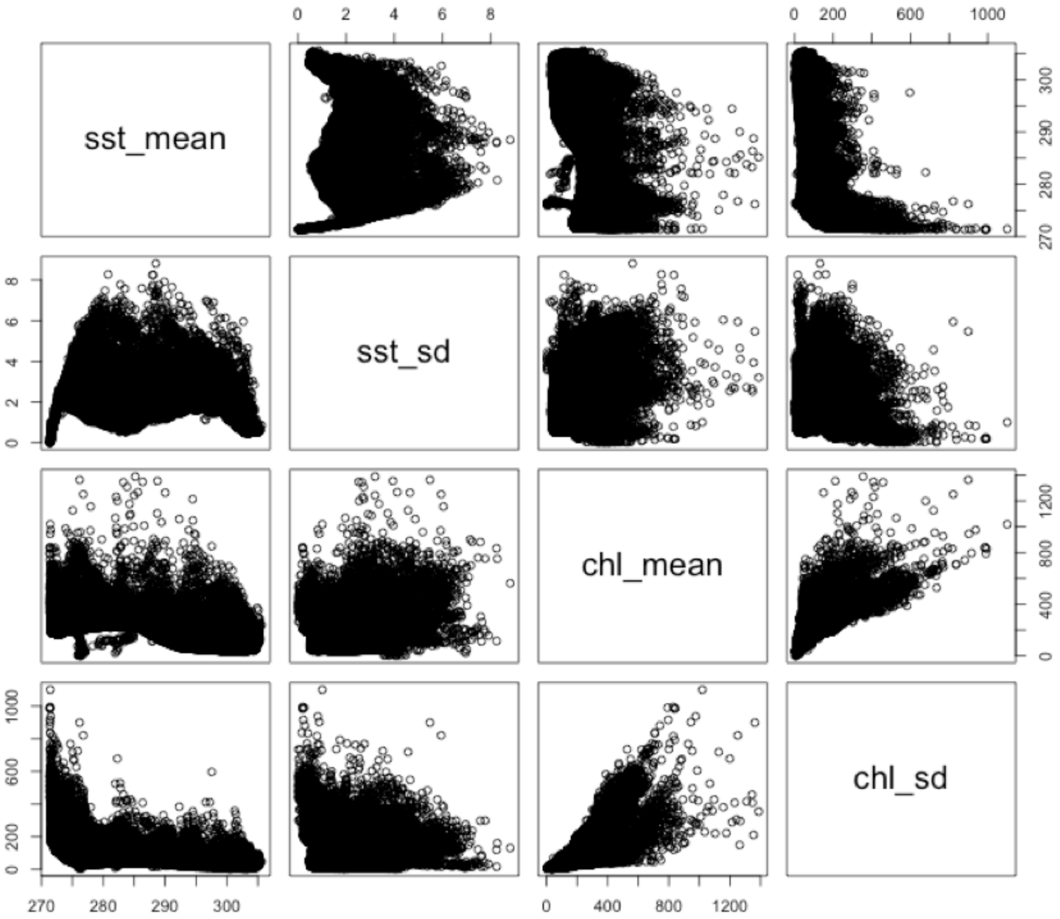


Figure A2. Draftsman plot for mean and SD future data



Sensitivity estimates

The inverse of the geographic area of a species range provides a simple, alternative estimate of sensitivity to climate change (Gaston 2003, Gaston and Fuller 2009). It is also independent of the climate forecast data we used to estimate exposure. We estimated the geographic area of each species range based on the maximum convex polygon characterizing the outermost points of its extent of occurrence layer (see *Methods*). The inverse of this estimate of range size was highly correlated with the sensitivity measure described in Equation 2 (Spearmans rank correlation = 0.71, p = 0.004; Fig. A3). Indeed, risk scores for the California Current marine forage species were qualitatively similar whether we used the inverse of climatic breadth or range size as our estimate of sensitivity (Fig. A4).

Figure A3. Relationship between 2 alternative sensitivity measures, one based on the inverse of a species range size and the other based on the inverse of its climatic breadth (see Equation 2, main text).

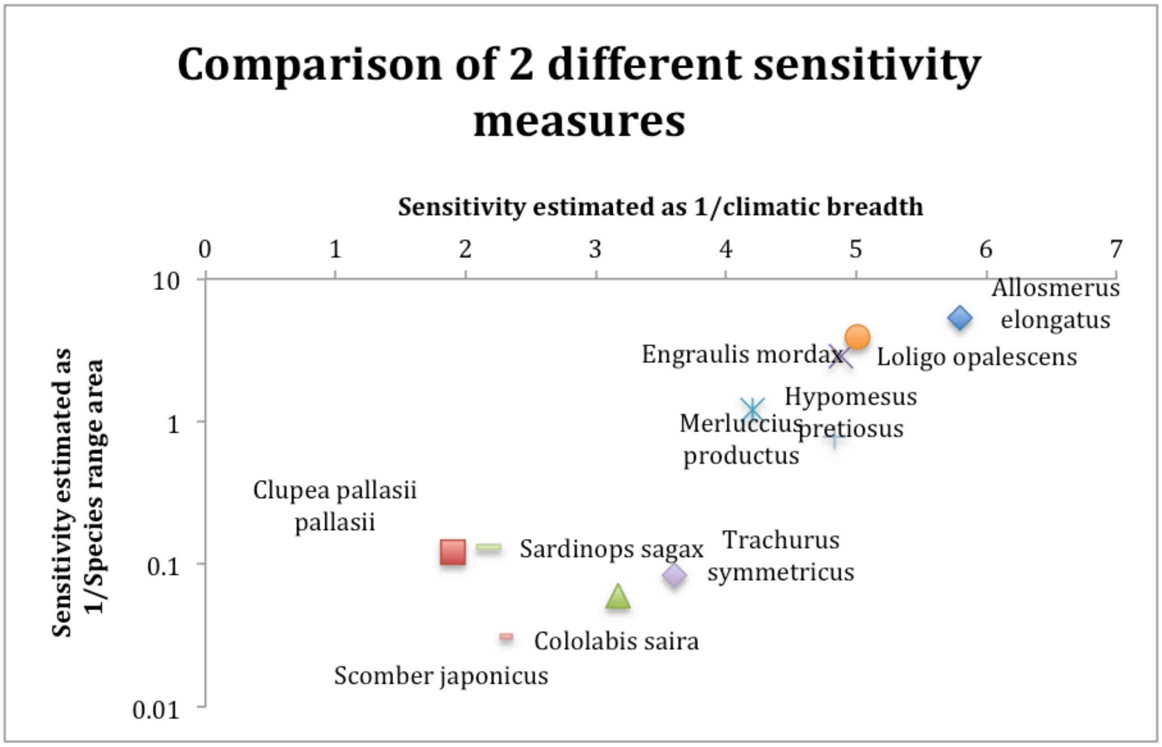
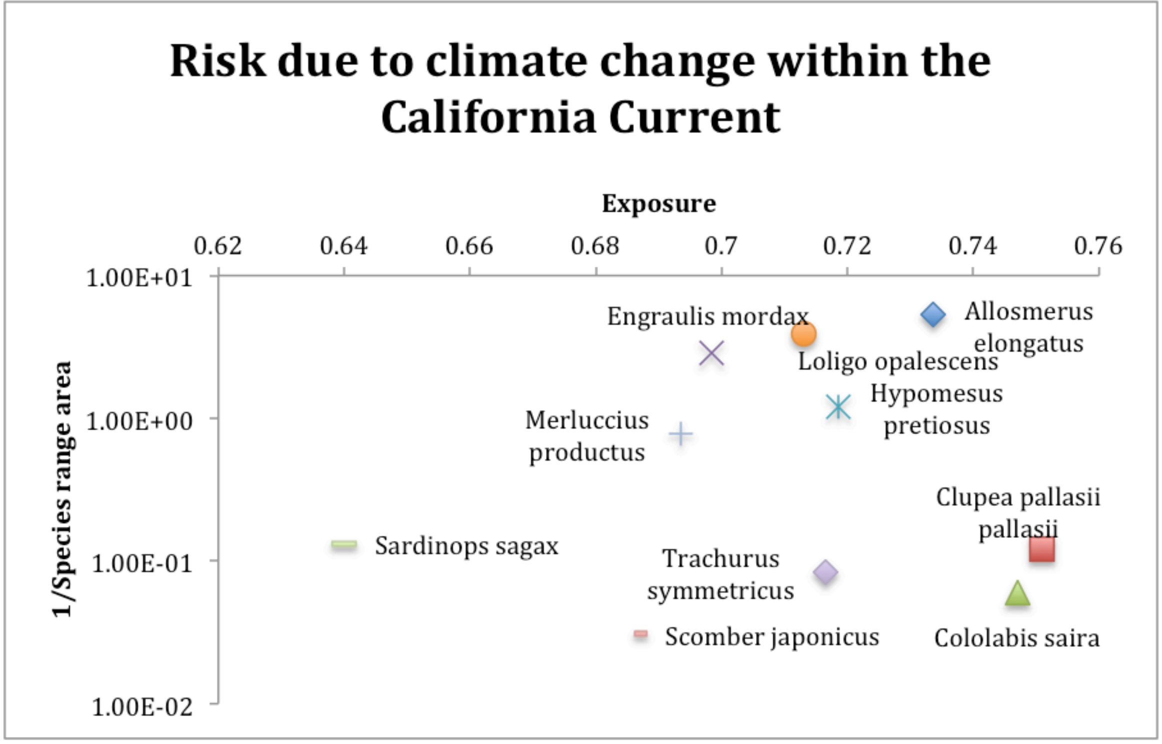


Figure A4. Preliminary risk scores based on exposure (as defined in Equation 1) and sensitivity, estimated as the inverse of a species range size.



CUMULATIVE RISKS TO MARINE MAMMALS, TURTLES, AND SEABIRDS IN THE CALIFORNIA CURRENT

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SUMMARY

In order to better understand species' use of the oceans and maximize effectiveness of marine spatial planning, we need quantitative assessments of animal distributions, human uses, and human threats. The synthesis from the Tagging of Pacific Predators (TOPP) program provided a large biologging dataset of 23 species and over 4,300 tags in the Pacific Ocean, with many species spending significant time in the California Current (Block et al. 2011). In addition, a focused study on cumulative human impacts in the California Current provided spatial layers of threat within the EEZ (Halpern et al. 2009). A recent manuscript, Maxwell et al. (2013), provides a synthetic analysis of marine mammal, turtle, and bird movement data from TOPP (8 species, 685 individuals, Maxwell et al. 2013 Table S3) relative to the 24 anthropogenic stressors previously compiled for the California Current (Figure 1, Halpern et al. 2009, Maxwell et al. 2013 Table S2). In this manuscript, the authors ranked each of the threats by species for 1) frequency of exposure, 2) directness of impact, 3) resistance to the impact, and 4) recovery time of individual, 5) reproductive impacts and 6) population level effects (Maxwell et al. 2013 Table S4). These rankings were then summed for each threat, and the intensity of the threat in each grid cell was multiplied by the ranking to come up with a spatial threat value (see diagram and equation in Figure 1). The species threats were then normalized to 1 and combined to come up with a multi-species threat value (Figure

2a). These cells are the ones with the greatest risk to the suite of examined species. Each species' utilization distribution calculated from a gridded utilization distribution method was summed by grid cell and then normalized to 1 to create a utilization map for each species. These were then summed to come up with a total multi-species utilization distribution (Figure 2b). Finally, these two surfaces (multi-species risk and use) were multiplied together to come up with a cumulative impact score to identify areas that are high use and high risk (Figure 2c).

All three guilds had high use of the U.S. West Coast EEZ, particularly on the shelf (Figure 1a; Maxwell et al. 2013 Figure 1). Seabirds had higher use offshore compared to the marine mammal and turtle guilds and all three guilds had high use in National Marine Sanctuaries (NMS) save the Olympic Coast NMS. This is not to suggest that this NMS has low densities of top predators but instead that most species and populations tagged were centered in central or southern California. Some risks were more ubiquitous throughout the EEZ such as climate change, others were more tied to the coast and river mouths such as pollution, while others were focused in hotspots such as shipping and fishing (Halpern et al. 2009 Figure 1). As part of the ranking of threats to each species, there were those that were similarly ranked across the study species (e.g. ocean acidification, pollution, shipping and invasive species), while others had particular targeted risks such as demersal fishing, coastal engineering, and beach access, which were greater only for pinnipeds that use coastal habitat regularly (Maxwell et al. 2013 Figure S2). The combined threat layers and utilization distributions show areas of overlap between high/low use and high/low risk (Figure 1c). Cumulative risk and use distributions were highest nearshore, within sanctuary boundaries, and particularly along the coast near Point Arena and Monterey Bay. Cumulative impacts and use values for seabirds were highest further north, off the coast of Oregon primarily. Climate stressors had the greatest influence across top predators, potentially due to their widespread and uniform distribution in the EEZ. The NMS were also hotspots of cumulative impact, with particularly high-risk values for climate, coastal pollution, and fishing, and high use from multiple top predator species.

Spatial differences between species use and risk intensity provides opportunities to target management action for the greatest gain. The widespread stressors, such as climate and pollution, will require a long-term effort and international cooperation, increasing the need for minimizing other threats. Mitigation may be the only solution at the local scale for the broad-scale stressors, but increased protection & risk-reduction of critical habitats (spawning habitat, migratory corridors) may be the most successful approach. Given that many of these threats may affect multiple levels of the food web, the cumulative impacts may compound across trophic levels. At the same time, mitigation for the more widespread stressors will likely have positive impacts across multiple components of the ecosystem.

One of the highest top predator use areas and highest risk areas were the suite of National Marine Sanctuaries along the coast. This pattern held true even when indirect and widespread cumulative impact layers were excluded from the analysis (Maxwell et al. 2013 Table 1 & Supp Fig. S5). Because most of the sanctuaries were key habitat for marine predators, the Sanctuary framework could offer a useful tool for providing greater protection within their boundaries. This could be a traditional closure of key threats year round, or a more targeted approach of reducing threats when top predator numbers are greatest. This dynamic management approach could use top predator data to examine spatial and temporal patterns of use relative to risk layers that may be seasonal or year-round and could restrict potentially harmful activities when critical habitat is present (Hobday et al in press – Law review). Good examples include bycatch of bluefin tuna in the yellowfin fishery in eastern Australia, where fishers without bycatch quota are excluded from high-risk areas based on seasonal and oceanographic changes (Hobday et al. 2009).

This case study compared predator tag data to a re-ranked database of threats, but many other datasets could be used to refine and expand this analysis. Tag data provides a behavioral context to top predator movement, but shipboard sightings or fisheries catch data can provide more holistic transects of biodiversity patterns. In addition, other risk datasets can offer greater temporal and spatial resolution, such as AIS data from shipping vessels or logbook data from fishers, improving the chances of finding grid cells and periods of maximum benefit. This framework of risk / use / cumulative stress can and should be applied to additional components of the ecosystem to minimize overlooked species or ecosystem components from these analyses. The case study provided here along with Maxwell et al. (2013) and Halpern et al. (2009) offers a road map towards combining use and risk calibrated by impact and intensity of perceived threat.

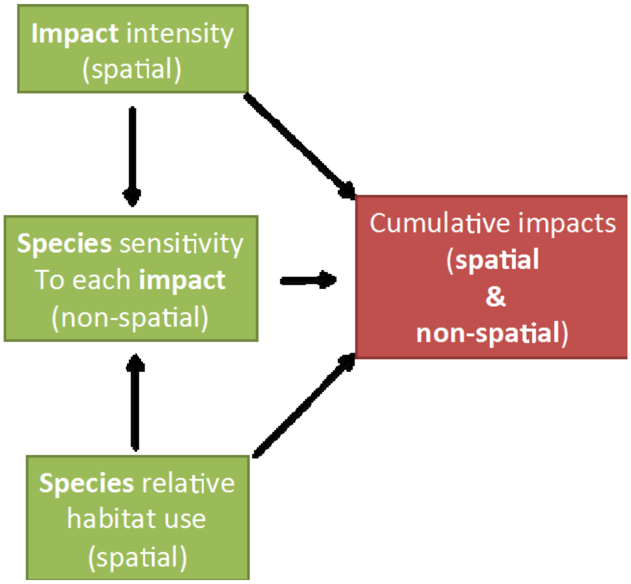

Figure 1. Diagram of cumulative impact calculation, formula shown here $CI = \sum_{i=1}^{n}\sum_{j=1}^{m} D_i + S_j + u_{i,j}$ where D_i = impact intensity, S_j = species sensitivity, and $u_{i,j}$ = relative habitat use. Impact intensity is derived from Halpern et al. 2009, species sensitivity from rankings as part of Maxwell et. al 2013, and species relative use from the TOPP project, Block et al. 2011. The weighted average of the three provide the metric of cumulative impact.

Figure 2. Spatial representations across all species for a) species relative use, b) ranked relative risk, and c) combined cumulative impact products. Each plot is demarcated by color relative to a) use, b) risk, and c) impact (white – low; grey – medium; black - high). The areas of the west coast National Marine Sanctuaries are colored blue.

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MANAGEMENT TESTING AND SCENARIOS FOR THE CALIFORNIA CURRENT

Authors (Analyses and summary): Isaac Kaplan[1], Lisa Crozier[1], Elliott Hazen[2], Dan Holland[1], Jeff Jorgensen[1], Karma Norman[1]

Authors (Analyses): Shallin Busch[1], Lisa Crozier[1], Tim Essington[3], Elliott Hazen[2], Emma Hodgson[3], Jeff Jorgensen[1], Isaac Kaplan[1], Elizabeth Petras[4], Jessica Redfern[2], James Ruzicka[5], Rich Zabel[1]

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Appendix MS2013-01. Sensitivity of the California Current ecosystem to climate change and ocean acidification. Isaac Kaplan, Shallin Busch

Appendix MS2013-02. Population responses of Snake River spring/summer Chinook salmon to freshwater and marine climate changes. Lisa Crozier and Rich Zabel.

Appendix MS2013-03. Ocean conditions and selected management options on the population dynamics of Wenatchee River spring Chinook salmon. Jeff Jorgensen

Appendix MS2013-04. Application of the Northern California Current ECOTRAN model to pelagic ecosystem scenarios for the 2013 California Current Integrated Ecosystem Assessment. James Ruzicka

Appendix MS2013-05. Assessing the risk of ocean acidification in the California Current to two key fishery species, Dungeness crab (*Cancer magister)* and pink shrimp (*Pandalus jordani).* Emma Hodgson, Tim Essington, Isaac Kaplan

Appendix MS2013-06. Scenarios for shipping on the US West Coast. Isaac Kaplan, Jessica Redfern, Elizabeth Petras

Appendix MS2013-07. Assessing the Risk of Ships Striking Large Whales in Marine Spatial Planning**.** Redfern et al (2013)

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INTRODUCTION

In preliminary engagement with managers and other experts as part of the 2012 IEA (Levin, Wells, et al. 2013), we developed narrative scenarios that acted as links between drivers and pressures on the California Current Ecosystem, for instance between increased global seafood demand and local fishing pressure. These were “scenarios for drivers”, essentially “what if” stories about alternate paths that drivers and pressures may take in the future. Scenarios included drivers related to human population growth, climate change, demand for conservation, energy, and evolution of status quo management and responses to it (Table 1). Narrative scenarios detailed potential effects on pressures considered in this IEA: urban and agricultural freshwater use, energy infrastructure, fishing, pollution, and shipping. For the 2012 IEA, we accompanied these scenarios with seven quantitative models that forecast impacts of some of these pressures on the ecosystem. The focus of our quantitative modeling in 2012 was primarily on fisheries, with one exception that focused on renewable ocean energy development.

Table MS1. *Schematic of narrative scenarios and potential impacts on five types of pressures on the California Current ecosystem (Levin and Wells 2013).*

	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	↔	↔	↔	↔	↔

For the 2013 IEA we develop modeling to consider the implications of climate change and ocean acidification, and begin considering trends and tradeoffs associated with shipping. We first focus on climate change, and its effects on the ecosystem and fisheries. In the context of salmon management, we evaluate the utility of strategies that have the potential to offset some of the effects of climate change. We consider the relative vulnerability of life stages of two major fishery species, Dungeness crab and pink shrimp, to ocean acidification. We also track how recent climate variation has altered productivity of key fisheries such as Pacific hake.

We broaden the scope of the IEA to include shipping by developing a series of narratives for shipping, based on conversations with individuals with expertise in the transportation sector. We also apply models that consider the spatial overlap of ships and whales, to identify tradeoffs and unintended consequences of clean air regulations and new shipping routes.

CLIMATE CHANGE AND OCEAN ACIDIFICATION

In the 2012 California Current IEA, preliminary engagement with managers identified climate change and ocean acidification (or more broadly, global change) as potential major drivers of the marine ecosystem. These conversations and narratives included qualitative predictions regarding impacts on salmonid survival and distribution, shifts in migrations and distribution of pelagic or midwater species such as hake or sardine, and increased mortality of shelled, calcifying organisms susceptible to acidified water (Figure **MS1**). Policy responses discussed were limited but included altering harvest, stream restoration, and community-based management. For the most part, quantitative modeling of climate change and ocean acidification was absent from the 2012 IEA, though in the 2011 IEA (Levin & Schwing 2011), Ainsworth et al. and Kaplan et al. presented quantitative ecosystem models simulating effects of global change.

For the 2013 California Current IEA, we have developed three quantitative modeling analyses that address impacts of climate and climate change on salmon and the continental shelf food web. We also present a risk assessment for two calcifying species, Dungeness crab (*Cancer magister*) and pink shrimp (*Pandalus jordani*), which may be particularly vulnerable to ocean acidification. Below, we summarize results from the four analyses, and present lessons learned from this effort. Appendices include an overview of the recent literature regarding climate change and ocean acidification (**Appendix MS2013-01**) in the California Current. Full articles or reports for the modeling analyses are found in the appendices.

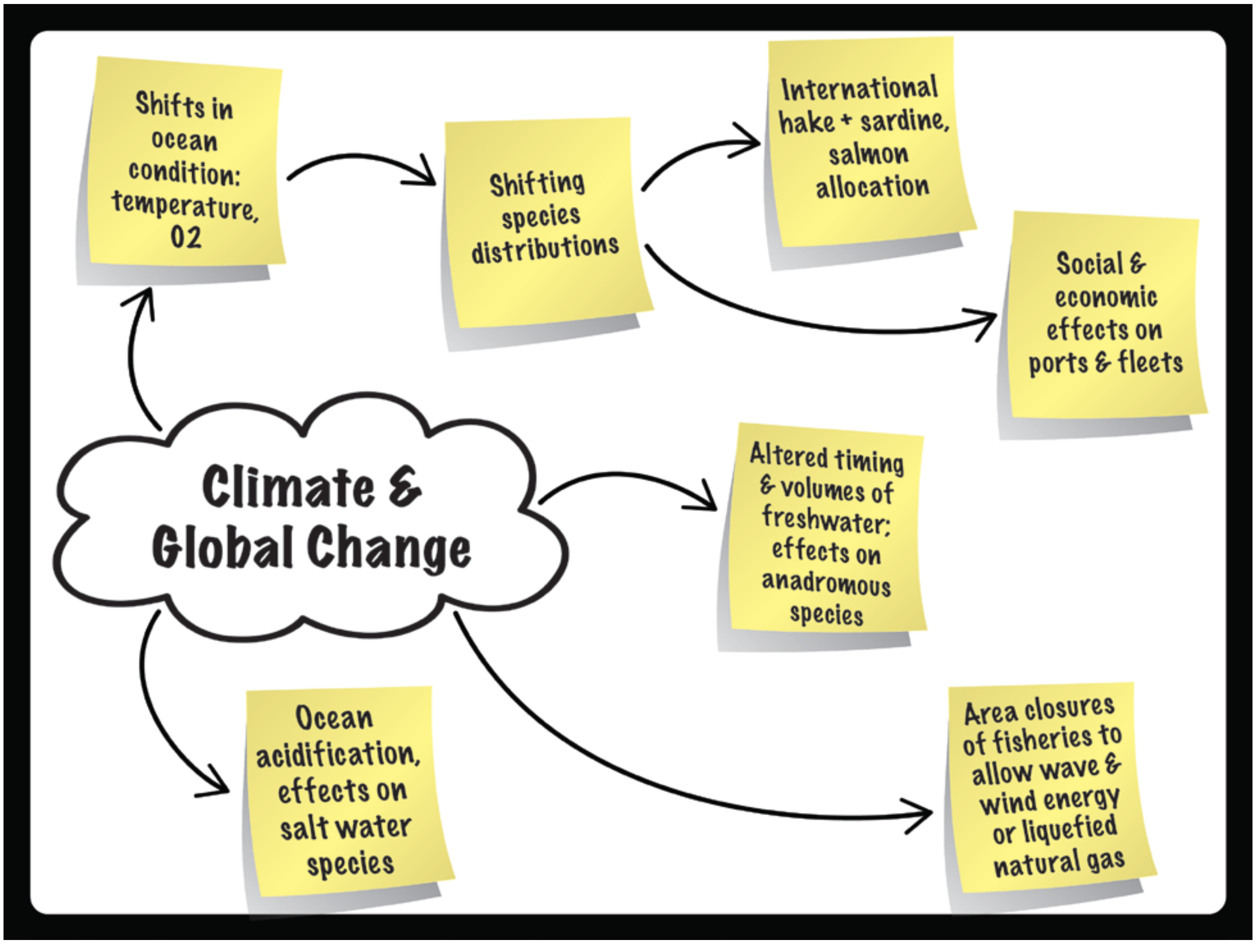

Figure MS1. Climate and Global Change scenario, with pressures (yellow notes) identified in narratives and by experts for the 2012 California Current IEA.

GENERAL APPROACH FOR CLIMATE CHANGE ANALYSES

King et al. (2011), Stock et al. (2011), and Hollowed et al. (2013) note the difficulty in inferring local patterns from coarse scale global circulation models, and they point to the need for downscaled, finer resolution oceanographic modeling to predict climate change impacts. Downscaled oceanographic models of the California Current, forced by coarser scale global models and scenarios for CO2 emissions, are in progress but not yet available to predict a full suite of ocean conditions such as temperature, upwelling, nutrients, and pH. King et al. (2011) provide conceptual diagrams illustrating the potential for climate change to lead to warmer surface waters; increased upwelling-favorable winds; a deepening thermocline; and increased coastal stratification. These authors presented logical consequences of climate change for different fish species, but were not able to make quantitative predictions (see summary, **Appendix MS2013-01**).

Punt and colleagues (2013) acknowledge this lack of quantitative predictions of climate change impacts, in the context of simulation testing fishery management strategies. Given the uncertainties related to precisely forecasting species responses to climate, these authors argue for a more general consideration of how the ecological system may change in the future, and whether management strategies are robust to this change. The three analyses below regarding climate change (**Appendices MS2013 02-04**) illustrate recent inter-annual and inter-decadal shifts in the food web and ocean conditions, and can inform how climate-driven shifts in productivity may alter fisheries and the ecosystem. In particular, a critical question for decision makers is whether potential management actions can buffer or offset changes in productivity or species survival that may stem from climate change.

Two of our analyses on climate change (**Appendices MS2013 02-03**) focus on ocean conditions for salmon. Ocean conditions have a large influence on salmon population dynamics (e.g., Koslow et al. 2002; Scheuerell and Williams 2005; Wells et al. 2008; Burke et al. 2013), and predicting future impacts of climate change on salmon populations requires forecasting ocean conditions and consideration of the implications for abundance and persistence of populations. These ocean conditions are a function of both regional and basin-scale processes (e.g., Mantua et al. 1997; Peterson et al. 2012). For instance, Jorgensen et al. (2013) and Crozier et al. (2013), respectively, have identified coastal upwelling and the Pacific Decadal Oscillation (PDO) as important determinants of ocean survival for Chinook salmon (*Oncorhynchus tshawytscha*) populations in the US Pacific Northwest (**Figure MS2**). In particular, the PDO generally indicated a period of cool, productive conditions for salmon from 1961-1976, and unfavorable warmer years from 1977-1997. Wells et al. (2008) found that Chinook salmon in the Smith River, California benefited from cool ocean temperatures and strong upwelling, wind stress, and a strong California Current.

In lieu of downscaled climate-ocean models for salmon, Crozier and Zabel (**Appendix MS 2**) and Jorgensen (**Appendix MS2013-03**) consider a range of ocean condition scenarios, and evaluate to what extent potential management options can compensate for poor ocean conditions for Chinook salmon. Ocean conditions are based on Monte Carlo resampling of years from the cool, productive phase of the PDO (1961-1976) and the phase with poorer conditions for most salmon stocks (1977-1997). Both models use a similar stochastic, age-structured salmon life cycle modeling framework developed originally by Zabel et al. (2006). Crozier and Zabel (**Appendix MS2013-02**) combine this scenario-based approach for the ocean with downscaled global circulation models applied to the fresh water, similar to other modeling efforts in rivers and streams (Battin et al. 2007; Crozier et al. 2008; Beechie et al. 2012).

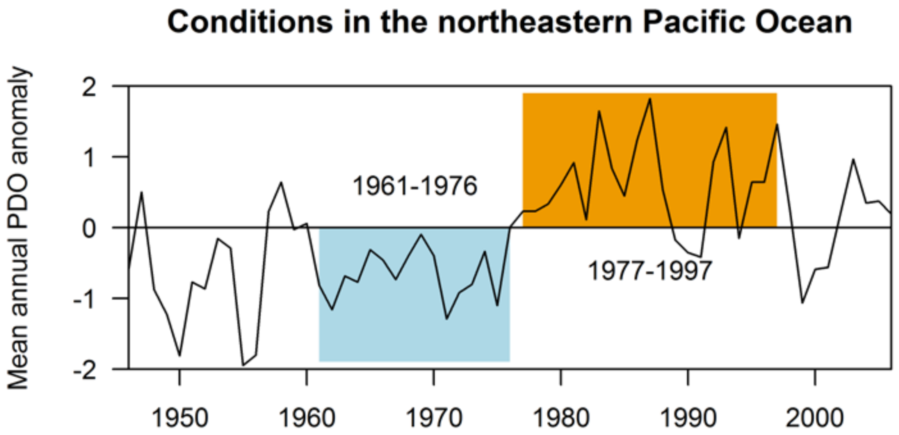

Figure MS2: Ocean conditions as measured by Pacific Decadal Oscillation anomalies in recent years, with relative periods of favorable (cooler ocean surface waters, 1961-1976; blue) and unfavorable (warmer ocean surface waters, 1977-1997; orange) conditions for Pacific salmon survival in the ocean used to develop scenarios of future ocean conditions.

SUMMARY OF INDIVIDUAL CLIMATE CHANGE ANALYSES

Crozier and Zabel (Appendix MS2013-02) employed a life cycle model to evaluate the impact of climate change on three populations of Snake River spring/summer Chinook salmon (*Oncorhynchus tshawytscha*). These three populations spawn and rear in tributaries of the Salmon River, and are listed as threatened under the Endangered Species Act. The authors used downscaled temperature and stream flow projections for the 2040s from 10 global circulation models (GCMs) and 2 emissions scenarios to characterize freshwater climate changes. They conducted a sensitivity analysis of ocean conditions by systematically varying periods of relatively favorable and unfavorable climate regimes from the historical record. Scenarios for ocean conditions consisted of alternative percentages of years when ocean conditions during early ocean entry by salmon were considered favorable (negative mean annual Pacific Decadal Oscillation [PDO] values) and unfavorable (positive PDO values) for survival, as discussed above and illustrated in Figure **MS2**.

Crozier and Zabel found that Chinook salmon populations differed in their sensitivity to freshwater change, with responses ranging from neutral to negative. In all three populations, spawner abundance declined in a relatively linear manner as the percentage of unfavorable ocean regimes increased (**Figure MS3**). However, there was a dramatic increase in extinction risk if ocean regimes shifted from 60% to 80% unfavorable. Because the 60% scenario produced very similar levels of risk and abundance as our historical scenario, this suggests these populations are already near a tipping point. Any decline in ocean conditions thus poses a very serious risk. However, the management scenarios considered (based on recent improved survival through the Columbia and Snake

Rivers), increased median population abundance 1.6-2.2 times across all climate scenarios and all populations. The maximum extinction risk dropped from 62% to 19%. Most importantly, management actions leading to higher survival through the hydrosystem (dams) successfully mitigated for the increased extinction risk due to climate conditions in all three populations. Abundance still declined from baseline under the worst ocean scenarios in two populations. Whether this recent improved survival can be sustained is not clear. But these results suggest a significant opportunity for recovery in these threatened populations.

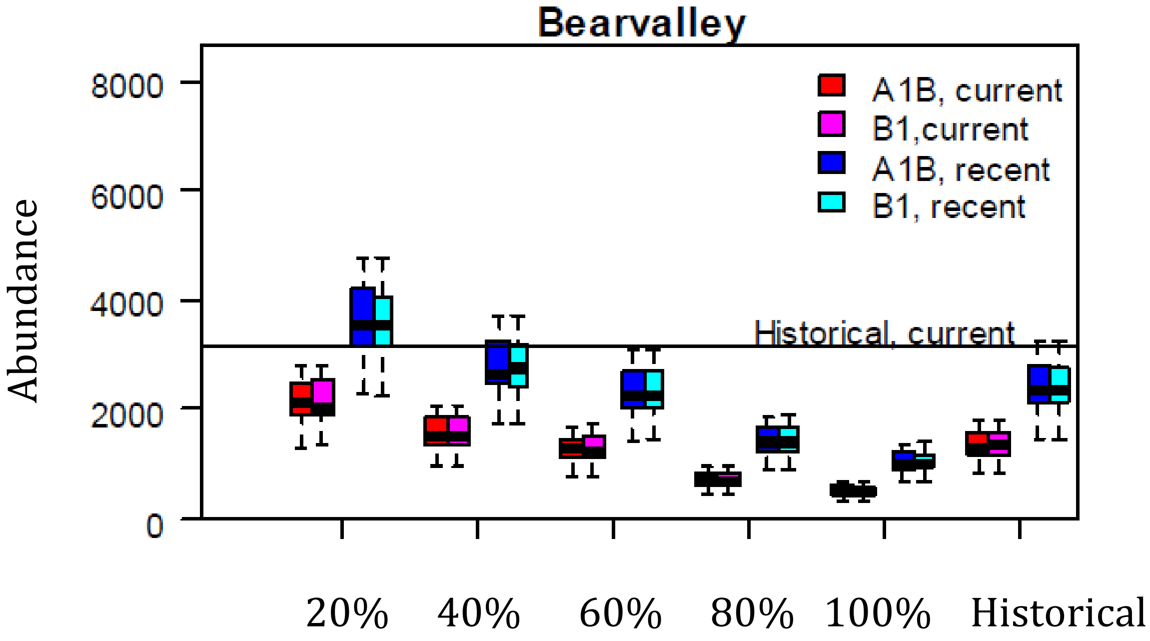

Figure MS3. *Median spawner abundance of Bear Valley Creek (Salmon River) Chinook salmon, as a function of freshwater climate scenarios (A1B or B1), hydrosystem survival ("Current", or improved survival rates labelled "recent"), and ocean conditions. Ocean conditions are characterized in terms of the percent of years with consistently positive PDO, and are compared with the actual historical time series ("Historic"). The baseline scenario used the historical freshwater and ocean conditions and the "current" hydrosystem management, and is shown by the horizontal line. The boxes show the range across all global climate models (GCMs) for a given scenario (line shows the median GCM, the boxes show the interquartile range, and the whiskers show the full range of all GCMs).*

Jorgensen (Appendix MS2013-03) applied scenarios for climate and management actions, focusing on responses of Wenatchee River spring Chinook salmon, a population listed as endangered under the Endangered Species Act. Predictions of population responses are available from a stochastic salmon life cycle model, similar to that used by Crozier and Zabel (**Appendix MS2**). Jorgensen (Appendix MS2013-03) combined scenarios of simulated future ocean conditions with estimated effects of management actions that affected the freshwater (prespawning adults, and rearing juvenile fish), mainstem (smolt migration through the Federal hydropower system), and estuary (avian predation). Similar to **Appendix MS2013-02**, scenarios for ocean conditions consisted of alternative percentages of years when ocean conditions were generally favorable for West Coast salmon (negative mean annual Pacific Decadal Oscillation [PDO] values) and unfavorable (positive PDO values) (**Figure MS2**). Compared to a benchmark scenario, in the Wenatchee River median spawners and carrying capacity declined with worsened ocean conditions. When management actions were applied individually, freshwater survival increases had the best ability to mitigate for poor ocean conditions, while mainstem hydropower dam and estuary survival improvements had a more moderate ability to mitigate for poor ocean conditions (**TableMS2**). Collectively, freshwater, mainstem, and estuary management actions offset the effects of some moderate declines in ocean condition, but not the poorest ocean conditions considered in these scenarios.

Table MS2: *Estimated impacts of management actions on the number of Wenatchee River basin wild spring Chinook salmon spawners using a life cycle model that incorporated scenarios of simulated future ocean conditions. FCRPS survival is downstream smolt survival through the dams.* $N_{100,\,50\%}$ *is 50th percentile of spawner abundance at time t = 100 years, taken across runs.* $Pr(QE)_{100}$ *is probability of quasi-extinction for simulations that ran t = 100 years. The geometric mean of the number of wild spawners for the five year period 2005-2009 was 576 spawners.*

Ocean conditions	Avian predation	FCRPS survival	Freshwater survival	$N_{100,\,50\%}$	$Pr(QE)_{100}$
Historical	Current	Current	Current	860	0.001
20% bad	Current	Current	Current	822	0.002
40% bad	Current	Current	Current	737	0.005
60% bad	Current	Current	Current	632	0.001
80% bad	Current	Current	Current	549	0.009
100% bad	Current	Current	Current	493	0.008
Historical	Current	Current	+10%	1111	0
20% bad	Current	Current	+10%	1049	0
40% bad	Current	Current	+10%	901	0
60% bad	Current	Current	+10%	859	0
80% bad	Current	Current	+10%	668	0.001
100% bad	Current	Current	+10%	606	0.001
Historical	-50% reduced	+10%	Current	1004	0
20% bad	-50% reduced	+10%	Current	976	0
40% bad	-50% reduced	+10%	Current	826	0.001
60% bad	-50% reduced	+10%	Current	734	0
80% bad	-50% reduced	+10%	Current	642	0.003
100% bad	-50% reduced	+10%	Current	541	0.004
Historical	-50% reduced	+10%	+10%	1226	0
20% bad	-50% reduced	+10%	+10%	1254	0
40% bad	-50% reduced	+10%	+10%	1055	0
60% bad	-50% reduced	+10%	+10%	970	0
80% bad	-50% reduced	+10%	+10%	811	0
100% bad	-50% reduced	+10%	+10%	700	0.001

Ruzicka (**Appendix MS2013-04**) developed an end-to-end model (Steele & Ruzicka 2011; Ruzicka et al. 2012) to estimate the ecosystem-level and functional group responses to inter-annual variability in food web structure. The NCC ECOTRAN model maps the flow of production through the food web from lower trophic-level producers to upper trophic-level consumers and fisheries. The model domain covers the Oregon and Washington continental shelf ecosystem during the summer. NCC ECOTRAN was driven by inter-annual changes over the past decade in phytoplankton production and biomass, copepod community composition and biomass, the biomass of large jellyfishes, and changes in the forage fish community. Ten parameterizations of the model, one per year for 1998 through 2007, were developed in **Appendix MS2013-04**. For this region, the inter-annual variability in the abundance of these species was likely driven by basin-scale patterns such as the PDO and El Niño, but also by local patterns involving upwelling timing and influx of cold, fresh water from the north (Venrick et al. 2003; Peterson et al. 2006).

Generally, there was correspondence between years of high phytoplankton biomass and production rates up the food web. This was largely driven by the extreme years of the time-series: the low production El Niño year of 1998 and the high phytoplankton production years of 2002, 2006, and 2007 (**Figure MS4**). Aside from these extreme years, the response of the trophic groups and fisheries depended not on mean abundance of the groups manipulated in the scenarios but on abundance of particular lower trophic level groups, and trophic interactions.

This simple scenario modeling exercise demonstrated the short-term effects of observed community changes within the plankton and forage fish community upon higher trophic levels and upon production of fished species. Primary production and food web structural variability over the past decade suggest that pelagic fishery production, a measure of energy flow to the target species, generally varied 50% - 200% about the decadal mean. Variability was higher among fisheries that target forage species. Energy flow to Pacific hake, a major fishery target species, has varied from 40% below to 50% above the decadal mean (**Figure MS4**). Energy flow to gear types that targeted hake and sablefish performed best during years of higher euphausiid production (2006-2007), with roughly 30-50% increases during these years. Though we cannot at present predict what future levels of productivity will be under climate change, this period from 1998-2008 provides a range of annual production rates that could be used in the future to bracket what may occur under climate change.

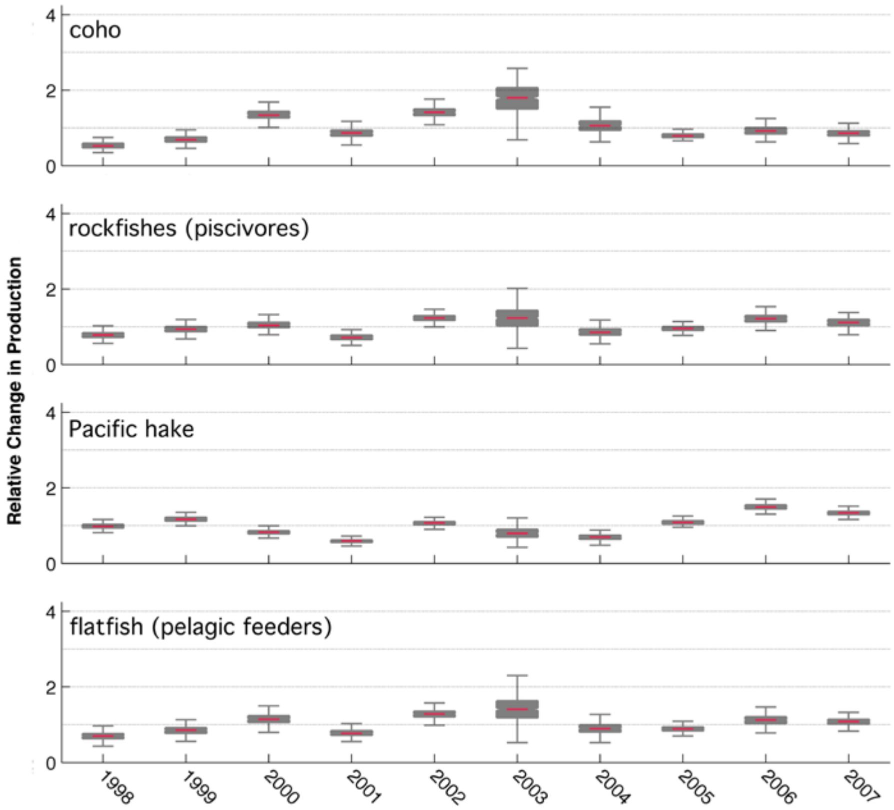

Figure MS4. *Scenarios showing effects of interannual variability among bottom- and mid-trophic level groups (phytoplankton, copepods, jellyfish, and forage fish) upon the production rates of select fish groups. Boxplots show distributions of changes in production rates relative to the inter-annual mean (ratio of scenario production rate to inter-annual mean, or 'base' model production rate). Boxplots show distributions of scenarios applied to 445 random, thermodynamically balanced model parameter configurations. A value of 1 on the y-axis represents no change from the inter-annual mean.*

SUMMARY OF OCEAN ACIDIFICATION STUDY

Hodgson et al. (**Appendix MS2013-05**) developed an ecological risk analysis of ocean acidification impacts on two species in the California Current: Dungeness crab, *Cancer magister*, and pink shrimp, *Pandalus jordani*. These species support US West Coast fisheries that were worth $174 million and $32 million in 2012, respectively. The California Current is particularly susceptible to ocean acidification, as low levels of carbonate saturation already exist within the near-shore environment (**see Appendix MS2013-01**). For each life stage of these species, Hodgson and colleagues define two components of risk: sensitivity and exposure. Sensitivity was determined from a literature review that examined the response of Dungeness crab and Pink shrimp, or related species, to acidification, typically measured in experimental conditions. Exposure is the overlap of species' distributions with pH predicted for the year 2050 (Gruber et al. 2012). The methods build on ecological risk analyses in the 2012 IEA (Levin, Wells, et al. 2013), but with a focus on these two shelled species, and these more precise predictions of future pH .

Though ocean climate change analyses such as that by Crozier and Zabel (**Appendix MS2013-02**) and Jorgensen (**Appendix MS2013-03**) necessarily relied on a scenarios approach to crudely bracket future ocean conditions, the analysis by Gruber and colleagues offers a downscaled prediction for coastal ocean acidity (but not climate shifts related to temperature or other factors), and can be applied in the risk analysis of ocean acidification (**Appendix MS2013-05**).

Hodgson et al. (**Appendix MS2013-05**) found that juvenile stages of these two species are most at risk, specifically larvae for pink shrimp and settled megalops for Dungeness crab (**Figure MS5**). Shrimp larvae are the most at risk because they are both highly sensitive and experience high levels of exposure. Within their distribution, 81.3% of their habitat at 100 m depth is predicted to be exposed to water more acidic than pH 7.7 by year 2050. (Laboratory and field studies suggest impacts on some marine species at values below, or more acidic than, pH 7.7). Hodgson and colleagues' sensitivity metric is derived from experiments on a related species, *Pandalus borealis*, that indicate impacts on development but not survival (Bechmann et al. 2011, Arnberg et al. 2012). Experimental results suggest that Dungeness crab settled megalops are only moderately sensitive to low pH, but they also have a high exposure, with 59% of waters they inhabit predicted to be more acidic than pH 7.7 by year 2050. Combining high exposure and moderate sensitivity suggests a relatively high final risk score.

All life history stages of both species are likely to experience a high degree of exposure to acidic waters (more acidic than pH 7.7 in year 2050) in >10% of their distributions. Of the eight life history stages of two species examined, six are predicted to be exposed to water more acidic than pH 7.7 in 59-89% of their distribution. This is largely due to the temporal and spatial distributions of adults and eggs of species, which are found along the bottom where pH is the lowest.

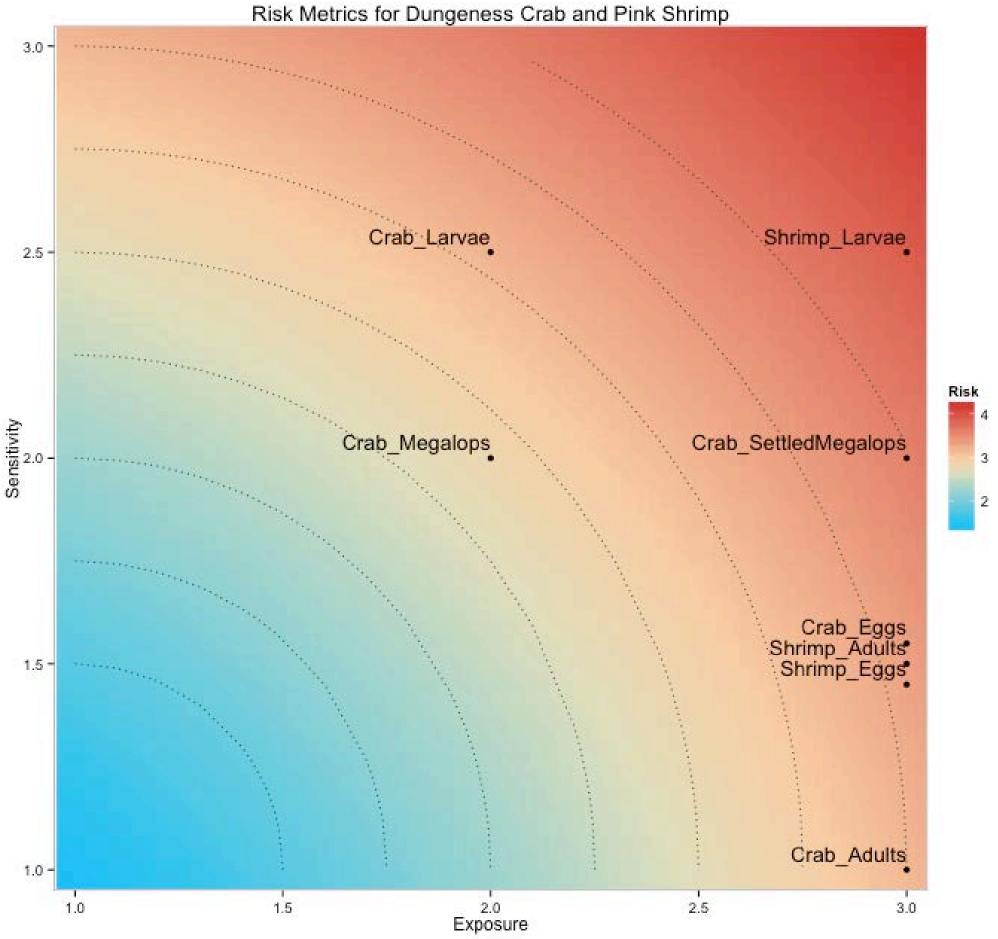

Figure MS5*. Risk plot demonstrating risk scores for each species and life history stage. Sensitivity values come from the literature and exposure values are specifically related to exposure to pH below 7.7.*

SUMMARY OF SHIPPING STUDIES

To lay the groundwork for future quantitative models that may include shipping, we conducted a series of conversations with eight individuals familiar with the shipping industry (**Appendix MS2013-06**). The goal was to understand recent and potential future trends in US West Coast shipping sectors over the next 5-30 years. These conversations outlined five trends, which are simple scenarios that are relevant to understanding or predicting shipping routes, speeds, or volumes, and may be relevant in predicting effects of shipping on various components of the ecosystem.

The first trend involved reduced ship speeds (super slow steaming). Container ships, bulk freighters, and tankers were all reported to have adopted this practice in recent years, for instance reducing ship speeds from maximum (e.g. 25 knots) to most efficient (e.g. 17 knots) speeds, with the exact speed varying by ship and engine type. Scenario 1 envisioned continuation of super slow steaming into the future. Potentially these lower speeds would reduce the probability of lethal shipstrikes on marine mammals.

The second trend, in container and tanker ships, was a large increase in ship size over the last decade, in an effort by shipping firms to maximize economies of scale. Scenario 2 envisioned a continuation of this trend toward fewer, larger vessels that would likely favor the use of the largest ports, such as LA/Long Beach. Shipping impacts on the ecosystem would likely be concentrated on these ports.

The Panama Canal is being expanded, with a new set of locks and capacity for larger ships slated for 2015. Goods manufactured in Asia and transported in containers could bypass the US West Coast and instead travel via the canal directly to markets on the East and Gulf Coasts. The most extreme outcome would be a scenario with a 50% decline in container ship traffic to West Coast ports. Impacts from shipping on the marine environment, or conflicts with other marine sectors, would likely decline near major container ship ports such as LA/Long Beach, San Francisco/Oakland, Tacoma, and Seattle.

The fourth trend involved altered spatial patterns of shipping due to new clean fuel requirements. In 2008, California began requiring the use of low sulfur fuel (clean fuel) in large vessels traveling within 24 nautical miles of the coast. The initial rules resulted in shifts in ship travel patterns: many ships moved farther offshore, in order to avoid the cleaner fuel requirement. However, by 2015, the International Maritime Organization will require clean fuel use out to 200 nautical miles, which could eliminate the advantage of these routes that were slightly offshore of 24 nautical miles. In such a scenario, a change in shipping routes would lead to changes in the overlap with habitat use by particular whale species. For instance, Redfern et al. (2013) found that humpback whales in Southern California occur in nearshore areas, while fin whales occur farther offshore.

The fifth trend involved continued development of new sources of oil, natural gas, and coal throughout the US and Canada, accompanied by increases in tanker and bulk freight cargos from ports in Oregon, Washington, and the Vancouver Canada area. This scenario envisioned continued increases in tanker and bulk freight shipping from Pacific Northwest ports, with increased potential for impacts concentrated in this region.

Future quantitative modeling of shipping, such as potential extensions of the work of Redfern et al. **(Appendix MS2013-07)**, may use these scenarios to consider how global forces translate into impacts on the local ecosystem.

Figure MS6. *Container ships. Photo: NOAA*

In **Appendix MS2013-07** and Redfern et al. (2013), the authors focus on spatial overlap between whales and shipping, and the potential for ships striking whales. As discussed in the narrative scenarios (**Appendix MS2013-06**), the California Air Resources Board recently implemented the Ocean-Going Vessel Fuel Rule. The fuel rule required large, commercial ships to use cleaner-burning fuels when traveling close to the mainland coast. Before implementation of the rule, a majority of ships traveled through the traffic separation scheme adopted by the International Maritime Organization in the Santa Barbara Channel. Following implementation, a higher proportion of ships began traveling south of the northern Channel Islands. The authors assessed the risk of ships striking humpback (*Megaptera novaeangliae*), blue (*Balaenoptera musculus*), and fin (*B. physalus*) whales in alternative shipping routes derived from patterns of shipping traffic observed before and after implementation of the fuel rule.

Redfern and colleagues (**Appendix MS2013-07** and Redfern et al. (2013)) developed models predicting habitat use by whales, and assumed ship-strike risk for the alternative shipping routes was proportional to the number of whales predicted by the models to occur within each route. The route with the lowest risk for humpback whales had the highest risk for fin whales and vice versa. Risk to both species may be ameliorated by creating a new route south of the northern Channel Islands and spreading traffic between this new route and the existing route in the Santa Barbara Channel. Creating a longer route may reduce the overlap between shipping and other uses by concentrating shipping traffic. Blue whales are distributed more evenly across the study area than humpback and fin whales; thus, risk to blue whales could not be ameliorated by concentrating shipping traffic in any of the routes we considered. Reducing ship-strike risk for blue whales may be necessary because the assessment of the potential number of strikes suggests that they are likely to exceed allowable levels of anthropogenic impacts established under U.S. laws.

SYNTHESIS: LESSONS LEARNED

IDENTIFY SCOPE FOR MANAGEMENT ADAPTATION TO CLIMATE CHANGE

Analyses related to climate change (**Appendices MS2013 02-04**) illustrate the potential for decision makers to mitigate the impacts of declining ocean conditions. Impacts of climate on salmon stocks could be mitigated by selected management practices focused on freshwater tributary, mainstem hydropower, and estuary survival (**Appendices MS2013 02-03)**. Chinook salmon life cycle modeling for the Wenatchee and Snake Rivers illustrated that improvements to fish survival rates in the rivers and estuary could compensate for moderate declines in ocean productivity. Offsetting poor ocean conditions would involve dam operations, policies related to barge transport of salmon, habitat restoration in spawning and rearing reaches, and reduced avian predation. For the Wenatchee and Salmon River populations, life cycle models suggested that stock status could be maintained or improved, despite scenarios for generally poor ocean climate. In addition to reducing extinction rates, these management policies prevented declines in abundance for some stocks when faced with a slightly higher frequency of poor ocean conditions (<60-80%) than in recent decades. Prior to these analyses, we did not anticipate that small improvements in freshwater survival could substantially buffer against moderate declines in ocean condition and survival. However, the analyses also show that some salmon populations remain at risk even with the freshwater management interventions.

Food web modeling of the Northern California Current food web (**Appendix MS2013-04**) did not explicitly test new management actions, but instead estimated four-fold interannual variability in energy flow to key fisheries species. If climate change alters these energy flows, for instance increasing years when less production is routed to species such as Pacific hake, fishery managers may need to respond by adjusting both harvests of forage fish and high trophic level species. Future work should explore the types of management actions that may be required.

RECENT DECADES ILLUSTRATE IMPACTS OF CLIMATE

Modeling of Salmon River and Wenatchee River Chinook salmon and of the Northern California Current food web (**Appendices MS2013 02-04**) illustrate the highly variable nature of local oceanography and productivity. This was driven both by large scale climate patterns (PDO, ENSO), but also by local patterns such as upwelling timing in the Northern California Current and influx of cold, fresh water from the north (**Appendix MS2013-04)**. As suggested by Punt et al. (2013), in lieu of forecasts of ocean conditions, we can consider how to devise management that is robust to recent extremes in low productivity, or extended periods of poor ocean conditions. We can use observations of outcomes during extreme conditions to understand how the ecosystem and various ecosystem components are impacted and respond to these conditions. Using models, including single-species and ecosystem models, parameterized and fitted to available data for periods with more extreme ocean conditions, we can project outcomes that may occur if these conditions persist for longer periods than have occurred to date.

CONSIDER GLOBAL DRIVERS, REGULATIONS, AND ECONOMIC TRENDS

Our narratives and analysis of shipping (**Appendices MS2013 06-07**) illustrate the extent to which the local ecosystem is influenced by global economic trends and international agreements. In particular, requirements for clean fuel and energy efficiency were expected to influence shipping pressure in California Current; clean fuel requirements at the state level have already altered ship-strike risk for marine mammals (**Appendix MS2013-07**). Global trends in fuel prices and container ship sizes and routes, for example, are also likely to alter risk of impacts such as fuel spills, ship-strikes, and ballast-water invasions. In the 2012 IEA (Levin, Wells, et al. 2013) we considered scenarios related to global population growth, seafood demand, and energy needs, focusing primarily on impacts on fisheries. However, such global trends are also relevant to protected species and to sectors beyond just fisheries. Predicting these trends is outside the scope of the IEA, but as with impacts of climate change on the ecosystem, it is useful to at least explore the impacts of particular scenarios to understand how economic changes can ripple through the ecosystem.

SCENARIOS: BOTH A STEPPING STONE, AND A LONG TERM TOOL

A useful approach to advance the ability to model impacts of climate change is the use of scenarios to bracket potential climate conditions; for example, scenarios that are based on extremes of recent ocean productivity. This illustrates trophic effects that could ripple through the food web (**Appendix MS2013-04**) and the extent to which decision makers can adapt to potential shifts (**Appendix MS2013 02-03**). We hope that the scenario approach can give way to coupling ecosystem models to downscaled atmosphere-ocean models.

The model by Gruber et al. (2012) projects coastal ocean acidification, and Hodgson et al. (**Appendix MS2013-05**) illustrate how this can be used to infer risk for species of interest. This fine-scale model differs from other global models operating on geometries as coarse as 1° latitude x 1° longitude (Dunne et al. 2012, 2013). However, the downscaled model of Gruber and colleagues is forced with a constant climate, i.e., there is no trend in atmospheric forcing except for CO2. One advantage of this model's relatively fine spatial resolution is that it can capture currents and upwelling within our region. This is particularly useful for understanding the spatial overlap of a threat (acidification) with nearshore fishery species, in this case state-managed Dungeness crab and pink shrimp fisheries that together are valued at more than $200 million annually.

In contrast to our climate change modeling, for which scenarios may be a stopgap approach, consideration of trends and future impacts of shipping may involve the scenario approach as a permanent, long term tool. Individuals with expertise in shipping (**Appendix MS2013-06**) emphasized that it is very difficult to make 5+ year forecasts of rapidly changing business practices and economic conditions for the transportation sector. We expect that future efforts to forecast complex human responses to economics will necessitate forward projections based on hypothetical scenarios, with retrospective analyses of recent data to identify trends (e.g. time series of pressures such as shipping volume reported in the IEA).

MANAGEMENT SCENARIOS AND TESTING CAN INFORM INDICATOR SELECTION AND RISK ASSESSMENT

The analyses here suggest that this IEA should include ongoing time series of certain key indicator species or biological groups. Ruzicka's (**Appendix MS2014-04**) analysis suggests that major changes in the Northern California continental shelf food web stem from altered abundances of forage fish and jellyfish. Analyses in the 2012 IEA (Levin, Wells, et al. 2013) also indicated that forage fish abundance strongly affects other components of the food web (Kaplan et al. 2013). Forage fish are sampled annually by several research groups, and time series are now included in the IEA. Jellyfish are more challenging to sample, and are not included in the 2012 IEA (Levin, Wells, et al. 2013), though the authors noted that "other indicators warrant more examination in the future, including the biomasses of jellyfish."

Our narratives and conversations suggest that the IEA should continue to include two distinct types of metrics related to shipping: both the number of ships, and the volume or amount of cargo. Respondents familiar with the container ship and tanker sectors noted large increases in ship size over the last decade, in an effort to maximize economies of scale and reduce cost per unit of cargo. Thus, we may expect diverging trends between indicators of vessel counts and cargo volume. Predicting impacts of shipping on different marine resources might require tracking different indicators of shipping activity. For instance, risk of mammal ship-strikes or likelihood of oil spills may depend on ship transits, while the potential scale of oil spills may depend on liquid cargo and fuel volumes.

Schematic diagrams portraying the IEA process (Levin et al. 2009) separate Risk Assessment from Management Testing and Scenarios, perhaps artificially. In previous California Current IEAs, Management Testing and Scenarios dealt primarily with forward projections or forecasts, while Risk Assessment focused more on spatial overlap between existing threats and particular habitats or species. Here we have begun to blend these two efforts. The risk of ocean acidification to Dungeness crab and pink shrimp (**Appendix MS2013-05**) and the risk of ship-strikes of marine mammals (**Appendix MS2013-07**) illustrate the value of combining ecological risk assessment (Hobday et al. 2011; Samhouri & Levin 2012) with scenario-based projections of climate, acidification, or shipping. A key contribution of Hodgson et al. (**Appendix MS2013-05**) is to demonstrate that impacts of ocean acidification needs to be considered in a spatial risk framework, based on maps of projected pH and species' habitat usage, and not simply from laboratory studies or meta-analyses.

EXPAND BEYOND FISHERIES TO MULTI-SECTOR MANAGEMENT AND CONSERVATION

While the focus of modeling analyses in 2012 was mainly on fishery management and impacts of fisheries, for the 2013 IEA we present analyses relevant to non-fishing drivers such as climate change and acidification. Even the single-species salmon models (**Appendices MS2013 02-03**) include detailed consideration and statistical relationships between climate or ocean conditions and ecological responses. We provide an introduction to issues related to shipping, a key non-fishing sector that may have a variety of impacts on, and risks to, the California Current. Fisheries landings and revenue are crucial metrics for the California Current, and we expect fisheries management actions to be included in Management Testing and Scenarios for future IEAs, but, we anticipate a broader, more comprehensive, approach going forward.

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APPENDIX MS2013-01. SENSITIVITY OF THE CALIFORNIA CURRENT ECOSYSTEM TO CLIMATE CHANGE AND OCEAN ACIDIFICATION

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[King et al. (2011) article summarized with permission from Jacquelynne King, Department of Fisheries and Oceans, Canada]

CLIMATE CHANGE AND THE CALIFORNIA CURRENT ECOSYSTEM

King et al. (2011) review potential implications of climate change on the California Current. Their synthesis, augmented with several other recent studies, provides a backdrop for the quantitative modeling we present here. Overall, King et al. identify trends in recent decades that may be associated with climate change; they point to specific pathways by which changes in physics and biogeochemistry may impact particular species; and they identify predictions from ensembles of global circulation models that forecast potential biophysical changes. One example of such a biophysical change impacting upper trophic levels (Woodworth-Jefcoats et al. 2013) is that large fish abundance is projected to decrease in the broader north Pacific, while in the California Current large fish are predicted to increase.

King et al. summarize empirical trends from the last 50-60 years that indicate warming of surface waters, an increase in upwelling-favorable winds particularly in the north, but increased stratification, particularly in the south (Bakun 1990, Mendelssohn et al. 2003, Palacios 2004). Increased upwelling may therefore have not led to increased primary production (McGowan et al. 2003, Palacios 2004). The biophysical system responds strongly to El Niño events, which have been increasing in intensity and frequency (An and Wang 2000). El Niño events bring warm water to the coast, and generally reduce the productivity of many coastal stocks (Checkley and Barth 2009), though migratory species such as sardine and hake may increase in abundance in the northern extent of the California Current.

King et al provide conceptual diagrams illustrating the potential for climate change to lead to: warmer surface waters; increased upwelling-favorable winds; a deepening thermocline; and increased coastal stratification that may lessen the beneficial effects of upwelling. Increased upwelling often includes increased acidification and lower dissolved oxygen, which can result in further habitat compression for hypoxia sensitive species (Stramma et al. 2011) but also reduced maximum body sizes for fish species in the CCS (Cheung et al. 2012). King et al. predicted that southern species of copepods would move north, replacing boreal copepods that provide higher energy content to predators and have been linked to high salmon production. Migratory fish such as Pacific hake and sardine could expand northward and increase the extent of annual northerly migrations. Pelagic species such as albacore tuna and blue shark could increase movements to nearshore areas. Chinook salmon freshwater habitat may have low water flow during salmon spawning and rearing phases, decreasing spawner success and juvenile survival. On the other hand, King and colleagues suggest that some long-lived groundfish may be able to withstand prolonged periods of poor recruitment, and could increase in the northern extent of the California Current. However, seabirds often have poor hatchling survival and fledging success during warm El Niño events, and this may be persistent if warmer conditions continue in the future. As illustrated by the expansion of jumbo squid in the California Current and recruitment failures of Cassin's auklet and Chinook salmon in recent years (Sydeman et al. 2006, Lindley et al. 2009, Stewart et al. 2012), changing environmental conditions will most certainly result in some populations expanding in size and habitat while others decrease.

Overland and Wang (2007) summarize climate change predictions from an ensemble of 10 atmosphere-ocean global circulation models that best fit 20th century historical data of sea surface temperature and the Pacific Decadal Oscillation. This ensemble approach leads to estimates of slight warming of approximately 1.2-1.8°C by year 2050, across the North Pacific. King et al. (2011) use a similar model ensemble, and also report some minor increases in upwelling intensity, particularly in the northern California Current. Natural variability overshadowed climate signals for many important metrics, particularly through 2040, with more substantial deviation from long term means beginning in 2040-2050. The northeast Pacific was predicted to have increases of 1.2°C for 2040-2049, relative to 1980-1999, based on ten IPCC models. In addition, global climate models have predicted a pole-ward expansion of the less productive subtropical gyre in the future (Polovina et al. 2008, 2011). The more northward position of the boundary between the subtropical and subarctic gyres, known as the transition zone, will change important north Pacific migration corridors to and from the California Current (Polovina et al. 2011, Hazen et al. 2012). These warming trends and the poleward expansion of subtropical water in recent decades are consistent with King and colleagues' conceptual pathways from climate to plankton, fish, and birds.

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SENSITIVITY OF THE CALIFORNIA CURRENT ECOSYSTEM TO OCEAN ACIDIFICATION

CURRENT AND PROJECTED FUTURE OCEAN CARBON CHEMISTRY

Worldwide, ocean chemistry is changing due to increasing atmospheric carbon dioxide concentrations (Caldeira and Wickett 2003, Feely et al. 2004, Orr et al. 2005, Doney et al. 2009). About a third of all anthropogenically-released carbon dioxide has been absorbed by the oceans since the Industrial Revolution, though ocean carbon dioxide absorption has slowed over time and oceans currently absorb only about a quarter of annual carbon dioxide emissions (Sabine et al. 2004, Canadell et al. 2007, Le Quéré et al. 2010). When carbon dioxide dissolves in seawater, it forms carbonic acid, which lowers seawater pH. Due to the accumulation of carbon dioxide in marine waters over the past ~250 years, the concentration of H^+ has increased ~30% and the average pH of global oceans has dropped from ~8.2 to ~8.1, a phenomenon known as ocean acidification (Caldeira and Wickett 2003). The accumulation of carbon dioxide in seawater also decreases the concentration of carbonate ions, which affects how readily calcium carbonate structures accrete or dissolve. Increases in carbon dioxide can reduce the saturation state for calcium carbonate structures to the point at which dissolution is chemically favored (Feely et al. 2004), which has implications for the large number of marine species that form calcium carbonate shells, tests, and skeletons (Kroeker et al. 2010). Aragonite is a form of calcium carbonate used by many marine organisms that is relatively sensitive to changes in carbon chemistry conditions.

North Pacific waters, which include the California Current Ecosystem, have relatively low seawater pH values and shallow aragonite saturation horizons due to a variety of natural oceanographic processes (Feely et al. 2004, Jepson and Jacob 2007, Feely et al. 2008, Feely et al. 2009, Hauri et al. 2009). North Pacific waters are at the end of the ocean's global conveyer belt, meaning that the waters are "old" and have an accumulation of carbon dioxide from respiration processes. Sub-surface waters (150-300 m deep) which are naturally high in carbon dioxide and nutrients and also carry anthropogenic carbon dioxide absorbed from prior contact with the atmosphere commonly upwell along the US West Coast in the summer months (Feely et al. 2008). Upwelling events, while ephemeral, amplify the acidification experienced in this region. Near coast and estuarine waters in the California Current Ecosystem also experience acidification events induced by biological processes: high nutrient loads from rivers and run-off from the land can cause phytoplankton blooms that then die and are decomposed by respiring bacteria (but see

Borges and Gypens 2010, Cai et al. 2011, Sunda and Cai 2012). Respiration of organic carbon is estimated to drive over half of the acidification that occurs in the deep waters of Puget Sound's Hood Canal during summer months (Feely et al. 2010). This and a variety of other physical and biological drivers (*e.g.*, day-night cycle of photosynthesis and respiration, tidal cycle, freshwater contributions, pollution) contribute to the wide variation in carbon chemistry conditions observed in nearshore waters (Doney et al. 2007, Hofmann et al. 2011, Barton et al. 2012).

Similar to global estimates, ocean acidification has decreased pH in the California Current Ecosystem by ~ 0.1 unit (to ~8.04) and aragonite saturation state by about 0.4 (to ~2.3) (Hauri et al. 2009, Gruber et al. 2012). This change is ten times faster than any change in ocean carbon chemistry over the past 50 million years (Pelejero et al. 2010). Ocean carbon chemistry in the region is also influenced by changes in ocean circulation due to climate change, such as those induced by the increase in upwelling favorable winds (Bakun 1990, Feely et al. 2012). Over recent decades, offshore upwelling in the southern California Current Ecosystem has intensified (Rykaczewski and Checkley 2008). Water upwelled to the surface in some parts of the California Current Ecosystem is now undersaturated with respect to aragonite due to ocean acidification (Feely et al. 2008). Without ocean acidification, undersaturated waters would be 50 m deeper than they are today (Feely et al. 2008). If carbon dioxide emissions continue as expected, globally, average surface ocean pH will decrease by ~0.3-0.4, to its lowest value in over 40 million years, and carbonate ion concentration will decrease by about 50% (Caldeira and Wickett 2003, Orr et al. 2005, Solomon et al. 2007, Pelejero et al. 2010). This change would occur ~100 times faster than the changes in ocean pH during Earth's recent glacial-interglacial transitions (Pelejero et al. 2010). By 2050, models project that over half of the nearshore water mass in the central part of the California Current Ecosystem will be undersaturated with respect to aragonite (Gruber et al. 2012). The California Current Ecosystem is one of Earth's three hot spots for the progression of ocean acidification (Gruber et al. 2012).

SPECIES RESPONSE TO OCEAN ACIDIFICATION

Laboratory and field research has found that many organisms, especially calcifiers, respond negatively to ocean acidification (Hall-Spencer et al. 2008, Kroeker et al. 2010). These changes include decreased growth and survival and altered gene and protein expression and physiology, including acid-base balance and energy metabolism (Kroeker et al. 2010, Parker et al. 2013). However, there is strong variation in response to acidification between species and even within some species (Kroeker et al. 2010, Parker et al. 2011, Kelly et al. 2013). Some primary producers (*e.g.*, seagrasses, macroalgae, and phytoplankton with low-efficiency CO_2 concentrating mechanisms) will likely benefit from ocean acidification (Palacios and Zimmerman 2007, Swanson and Fox 2007, Reinfelder 2011). The fast rate of change in ocean carbon chemistry raises the potential that some marine species harmed by ocean acidification may not be able to adapt, evolve, or adjust quickly enough to persist. Geologically induced ocean acidification events in Earth's history are contemporaneous with extinction events in some taxa, suggesting that ocean acidification may overwhelm evolutionary processes and reorganize ecosystems (Hautmann et al. 2008, Kump et al. 2009, Pelejero et al. 2010). Marine communities near natural CO_2 vents are significantly different than neighboring communities that are not exposed to elevated CO_2 levels (Hall-Spencer et al. 2008, Fabricius et al. 2011, Kroeker et al. 2011). Furthermore, ecosystem modeling suggests that trophic interactions can cause the direct impacts of ocean acidification on sensitive species to ripple through food webs, positively or negatively affecting species to which they are tropically linked (Busch et al. 2013). While the literature on the biological and ecological impacts of ocean acidification is growing rapidly, how the vast majority of economically and ecologically important species in the California Current Ecosystem will respond to ocean acidification and how acidification will affect species interactions is largely unknown. However, we do know that production of Pacific oyster (*Crassostrea gigas*) larvae in Pacific Northwest shellfish hatcheries has already been negatively affected by changes in ocean carbon chemistry (Barton et al. 2012).

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APPENDIX MS2013-02. MITIGATION FOR POTENTIAL DECLINES IN SNAKE RIVER SPRING/SUMMER CHINOOK SALMON DUE TO FRESHWATER AND MARINE CLIMATE CHANGES

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SUMMARY

We employed an established life cycle model to evaluate the impact of climate change on three populations of Snake River spring/summer Chinook salmon (*Oncorhynchus tshawytscha*), listed as threatened under the Endangered Species Act. We used downscaled temperature and stream flow projections for the 2040s from 10 global circulation models (GCMs) and 2 emissions scenarios to characterize freshwater climate changes. We conducted a sensitivity analysis of ocean conditions by systematically varying periods of relatively favorable and unfavorable climate regimes from the historical record. We found that populations differed in their sensitivity to freshwater changes from neutral to negative responses. In all populations, spawner abundance declined in an approximately linear manner as the percentage of unfavorable ocean regimes increased. However, there was a dramatic increase in extinction risk if ocean regimes shifted from 60% to 80% unfavorable. Because the 60% scenario produced very similar levels of risk and abundance as our historical scenario, these populations might already be near a tipping point. Any decline in ocean conditions thus poses a very serious risk. However, the management scenario we considered (based on recent improved survival of juveniles through the Columbia and Snake River hydrosystem), increased median population abundance 1.6-2.2 times across all climate scenarios and all populations. The maximum extinction risk, assuming increased juvenile survival, dropped from 62% to 19%. Most importantly, higher survival through the hydrosystem successfully mitigated for the increased extinction risk due to climate warming in all three populations. Abundance still declined from baseline under the worst ocean scenarios in two populations. Whether this recent high survival can be sustained is not clear, but these results suggest a significant opportunity for recovery in these threatened populations.

INTRODUCTION

For anadromous salmon and trout, limiting environmental factors vary as fish move through different habitats over their life cycle (Crozier et al. 2008a, McDaniels et al. 2010). In freshwater, stream temperature and flow govern many aspects of juvenile growth and survival. Low temperature can limit growth, especially in high mountain streams. But high temperature can lower survival and reproductive success (McCullough 1999, Martins et al. 2012). High streamflow can benefit juvenile migrants (Williams et al. 2005) but also scour nests (Goode et al. 2013) and raise energetic costs of maintaining position or swimming upstream during the adult migration (Rand et al. 2006). Environmental conditions also shape habitat structure and the quantity and quality of prey (Power et al. 2008), as well as pressure from competitors and predators (Wenger et al. 2011, Kuehne et al. 2012, Lawrence et al. 2012). Some of these pressures vary with fish density. For example, there might be a limited number of refugia from high temperatures or low flows, causing the environmental drivers to determine the carrying capacity of a given habitat.

In this report, we examine environmental drivers in freshwater and ocean stages independently. Regional climate drivers affect freshwater habitat productivity in population-specific ways, influencing relative population vulnerability to climate change (Crozier et al. 2008b). Previous analyses identified fall flow and summer temperature as strong correlates of juvenile survival in Snake River spring/summer Chinook salmon (Crozier and Zabel 2006, Crozier and Zabel 2013), which are listed as threatened under the Endangered Species Act. Climate regimes, defined largely by the Pacific Decadal Oscillation (PDO), shape salmon population dynamics at the decadal scale (Mantua et al. 1997). Cooler ocean conditions, associated with negative phases of the PDO, bring nutrient-rich prey and reduced predator populations, improving survival for Columbia River salmon. Warmer ocean conditions, on the other hand, such as the 1977-1997 positive phase of the PDO, result in lower salmon survival (Zabel et al. 2006). We explored the consequences of more or less frequent stretches of less-favorable climate regimes for population viability. Because we currently lack specific projections of the indices needed to model salmon survival from GCM and oceanographic models, we conducted a sensitivity analysis by increasing the frequency of poor-ocean periods from 20% to 80% of our time series. Models with appropriate resolution (Gruber 2011) are being developed, however, and will be incorporated into the next IEA report.

An increasingly pressing concern for the recovery of at-risk populations is whether mitigating for climate-related losses is possible. If climate change reduces population growth rates, what management actions could reduce this effect (Beechie et al. 2012)? The primary management lever under consideration for Columbia River salmon involves operation of the major Federal hydroelectric dams on the Columbia and Snake Rivers (NOAA Fisheries 2010). Here we consider a scenario of improved juvenile survival (Zabel et al. 2013b) based on recent observations (2005-2009 outmigration years). This is the first population viability analysis using this scenario. We present preliminary results to demonstrate the approach and propose further integration of the effects of changes in mainstem Columbia River conditions and additional scenarios of ocean conditions.

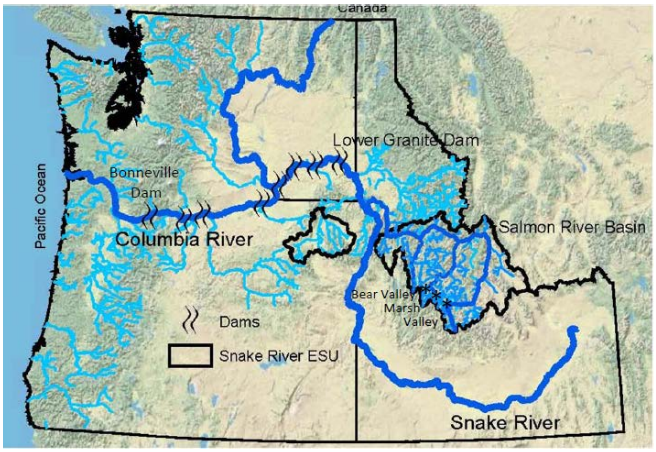

Figure 1. Map of the Columbia and Snake Rivers, with the modelled populations labelled. The eight major hydrosystem dams affecting these populations are shown.

BACKGROUND ON SNAKE RIVER SPRING/SUMMER CHINOOK SALMON

Snake River spring/summer Chinook salmon spawn and rear in tributaries of the Salmon River in central Idaho, the Grande Ronde and Imnaha rivers in northeastern Oregon, and the Tucannon River in southeast Washington (Fig. 1). In response to severe population declines throughout the second half of the 20th century, they were listed as threatened under the Endangered Species Act in 1990 (NMFS 1992). The Salmon River Basin populations considered here mostly spawn and rear at high elevation (1946-2010 meters above sea level), within legally protected wilderness areas, although agriculture and grazing affect some areas and historically mining and logging occurred in the basin. The hydrological regime is snow-dominated, with large flows in spring and low flows in late summer. Juveniles spend a full year in tributaries before migrating 1100-1500 km through the lower Snake and Columbia rivers to the Pacific Ocean. They return to natal habitats to spawn only once after 1-3 years in the ocean.

Here we focus on three populations in the Salmon River Basin with differing sensitivities to climate change, as identified by Crozier and Zabel (2013). Populations from Bear Valley and Valley creeks both have negative responses to temperature, whereas the Marsh Creek population has a neutral response (Fig. 2). Flow has a generally negative impact on the Bear Valley Creek population, but a positive impact on both Marsh and Valley Creek populations (Crozier and Zabel (2013).

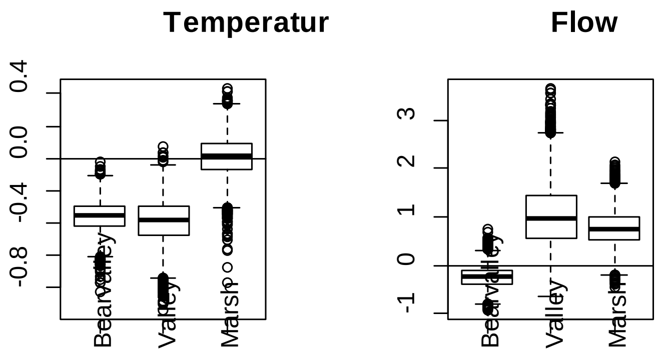

Figure 2. Box and whisker plots depicting the posterior distribution of the A. temperature and B. flow standardized regression coefficients for parr-to-smolt survival (see Crozier and Zabel 2013) based on Bayesian model analysis. The zero line is shown. Boxes show the 25th-75th quantile range; the whiskers show 1.5 times that range. The horizontal lines within each box show the median.

METHODS

We applied the life cycle modeling framework originally developed by Zabel et al (2006). The model is explained in detail elsewhere (ICTRT and Zabel 2007, Jorgensen, this report, Zabel et al. 2013a). This stochastic, age-structured matrix model includes multiple life stages in some annual steps. The spawner-to-year 1, and year 1-to-year 2 transitions are modeled as population-specific, density-dependent functions. Freshwater environmental drivers are incorporated into the asymptotic recruitment of year 1-to-year 2. Survival through the hydrosystem affects survival in year 2, and mainstem Columbia River and ocean conditions (indexed by the April upwelling and May PDO) affect survival in year 3. Fish return to spawn over years 3-5, with constant proportions in each age group.

FRESHWATER CLIMATE SCENARIOS

Freshwater climate drivers are 1) summer average air temperature across the Salmon River Basin from May through August, which largely determines growth opportunity in these mountain streams, and 2) mean flow in September and October, measured on the Salmon River at Salmon, Idaho (USGS 2005, Station ID 13302500). We established our baseline, “historical” conditions using the monthly mean temperature records from 1962 to 2005 at seven weather stations distributed around the Salmon River Basin (see Crozier et al. 2008b for station IDs and methods). The Climate Impacts Group modeled historical flows (http://www.hydro.washington.edu/2860/). We used modeled historical flows rather than observed flows for a fair comparison with future scenarios that rely on hydrological modeling. We used the climate projections for A1B and B1 emissions scenarios from 10 Global Circulation Models (GCMs) that the Climate Impacts Group downscaled using the “Hybrid Delta” method followed by the Variable Infiltration Capacity hydrological model (Fig. 3, Hamlet et al. 2013). We added the differential in air temperature projected by each GCM for the Middle Fork Lodge near Yellow Pine (SALYP 2031) to our historical time series (the “Delta” method). We analyzed projections for the “2040s”, which average over GCM output from 2029 to2059.

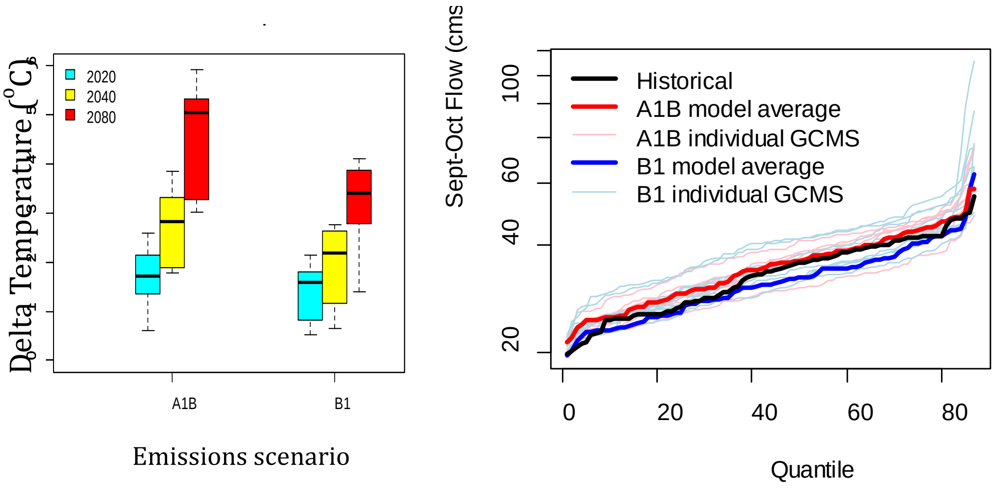

Figure 3. Freshwater climate perturbations. A. Temperature boxplots depict the offsets from all GCMs under a given emissions scenario and time frame. The median is depicted by the horizontal line, the interquartile range lies within the boxes and the whiskers show the most extreme GCMs. Only the 2040s are analyzed In this report, but we show the earlier and later periods for reference. B. A quantile plot of modeled streamflows (averaged over September and October). Note the log scale on the y axis.

“HYDRO” SCENARIOS

As juveniles migrate downstream, a variable proportion of them are transported in barges through the hydropower system and released below Bonneville Dam. Transported fish have different rates of adult return than fish that migrate in-river, based on PIT-tagged fish from 1994-2001 (Berggren et al. 2006). Within a scenario, we held the proportion transported and survival rates constant. Previous analyses have manipulated these rates under different management scenarios of the Columbia River Hydrosystem. We explored two scenarios. The first was used by ICTRT and Zabel (2007), which they called “current,” and we called the “1990s” scenario. The second was derived in Zabel (2013b) based on more recent returns. We called this alternate scenario the “2000s” because it describes survival from outmigration years 2005 to 2009. This scenario involves higher in-river survival and a much lower proportion of fish transported.

OCEAN SCENARIOS

Some of the most profound impacts of environmental conditions on salmon occur during their first year in the ocean. ICTRT and Zabel (2007, pp 23-26) identified three factors for Snake River spring/summer Chinook salmon that characterize variation in this term: mainstem Columbia River water travel time (inversely related to flow), April upwelling off the mouth of the Columbia, and the PDO index for May. Higher spring flows and high levels of spill reduce water travel time. The PDO and upwelling indices describe ocean conditions affecting juvenile salmon: cooler, high-nutrient water corresponds to a negative PDO and strong upwelling. This term varies stochastically, but we used historical conditions from 1946 to 2001 as the baseline for our simulations. Our alternate scenarios vary the proportions of favorable and unfavorable regimes within the historical record. The 1977-1997 period characterized "bad" conditions, and the 1961-1976 period characterized "good" conditions. The scenarios contained from one to four blocks of bad conditions (20%, 40%, 60% or 80%) in 20 year units (repeated as necessary) [Fig. 4]. The remainder of the 100-year time series consisted of good blocks. We alternated bad and good conditions in the intermediate scenarios, but the extremes contained one 20-year stretch of one type of conditions followed by 80 years of the other condition. We randomized the start year of the environmental time series to avoid trends due to the ordering of good and bad regimes.

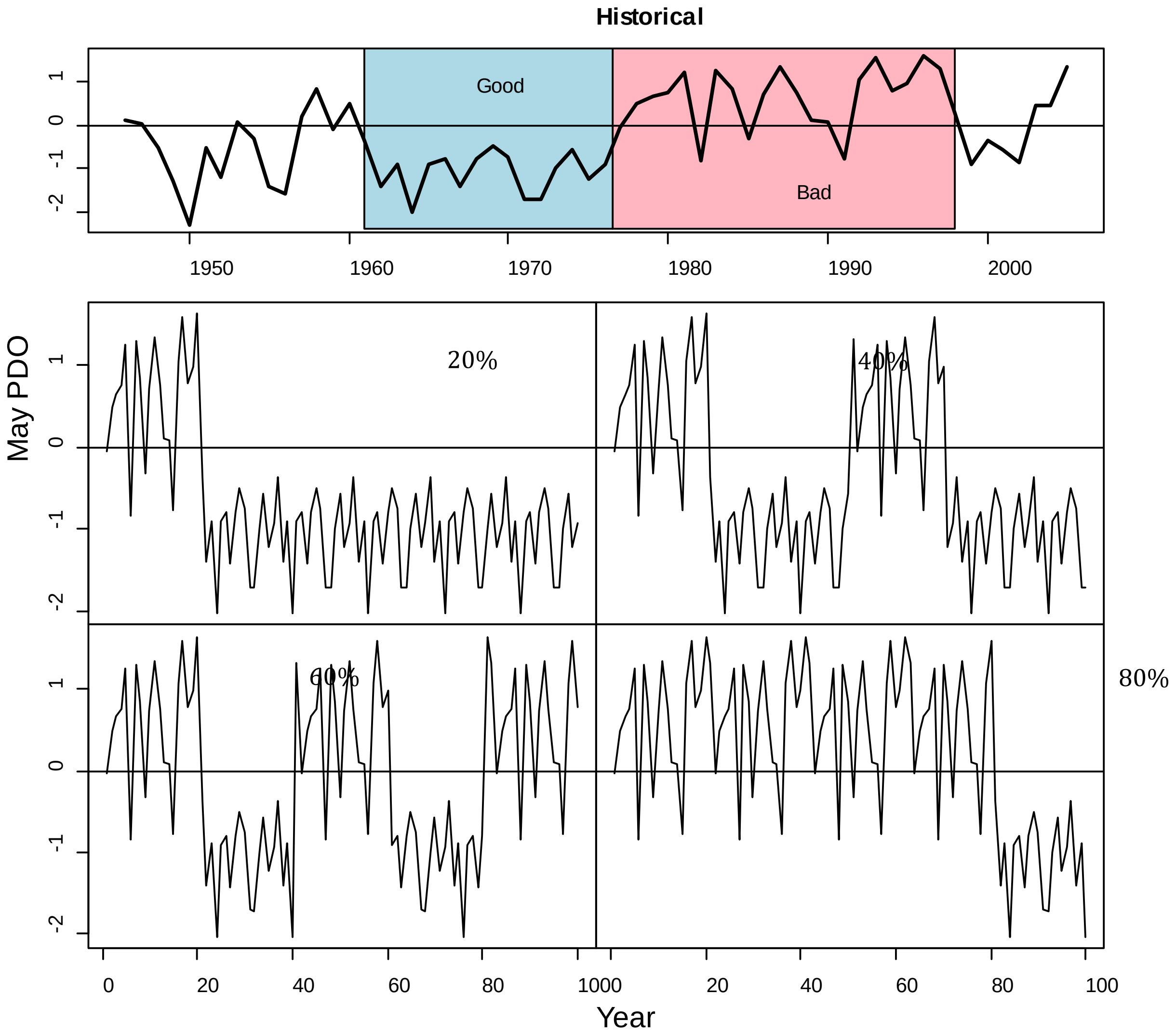

Figure 4. Ocean scenarios. Historical time-series of May PDO, with shaded boxed indicating the "Good" and "Bad" periods used in the scenarios, followed by 20%-80% "bad" regimes.

SIMULATIONS AND RESPONSE METRICS

We initiated our simulations with the mean spawner abundance observed over the last five years for each population (“N_0” in Appendix A). We ran the model for 100 years for each iteration, and ran 1000 iterations for each scenario. Our response metrics captured the core viability criteria of extinction risk and population abundance. We characterized extinction risk as the frequency across iterations within a scenario of population projections falling below a quasi-extinction threshold of 50 spawners in a 4-year running-mean. Our primary measure of abundance is the mean across iterations of the median spawner abundance within each iteration. Additional metrics are reported in Appendix A (for a full explanation of metrics, see Zabel et al. 2013a).

RESULTS AND CONCLUSION

The risk of extinction in our scenarios depended mostly on ocean conditions, but the 2000s hydrosystem management scenario had a very large mitigating effect. Our “baseline” conditions entail the historical freshwater and ocean conditions and the 1990s hydrosystem management scenario (horizontal lines in Fig. 5). Maintaining that hydrosystem management scenario, all populations showed dramatic increases in extinction risk when 80% of the years exhibited positive PDO conditions (red and pink boxes, Fig. 5A). Under the new management scenario, however, extinction risks dropped back to or below baseline conditions for all three populations, in even the worst ocean conditions (blue boxes). If ocean conditions were similar to the historical or improved (≤60% bad), the new management scenario reduced extinction risks to negligible levels. Under historical ocean conditions, one population reduced extinction risk even under the 1990s hydrosystem management scenario. Marsh Creek suffered no ill effects from rising temperature, and benefitted from the higher precipitation/high flow scenarios. Rising temperature lowered juvenile survival in the other two populations, and only excellent ocean conditions compensated for declines in freshwater. The two emissions scenarios showed very similar patterns (red vs. pink, and light vs. dark blue) because this time frame (the 2040s) precedes significant divergence in greenhouse gas concentrations.

Trends in population abundance ultimately drove changes in extinction risk, but displayed more subtle responses to the scenarios (Fig. 5B). Relative to the historical baseline, Bear Valley declined in almost all of the scenarios. The only exception was superb ocean conditions combined with high mainstem survival. The strong response to freshwater climate scenarios reflected the negative effect of rising temperature without ameliorating benefits from the high precipitation / high flow scenarios (Fig. 2). Valley Creek and Marsh Creek benefitted more from higher flows. In combination with high mainstem survival, these two populations grew as long as ocean conditions stayed at the historical quality or improved. Under the worst ocean conditions, abundance of all populations declined severely.

The three Snake River salmon populations modeled here showed high sensitivity to climate-driven changes in freshwater and ocean conditions, which could be largely mitigated by maintaining mainstem survival at recent levels. Across the wide range of ocean conditions we explored, population mean abundance declined in a relatively linear manner as ocean conditions deteriorated. However, extinction risk increased dramatically in all populations when unfavorable ocean conditions increased from 60% and 80%. We suspect that these populations might already be near this tipping point because the 60% scenario produced similar levels of risk and abundance as our historical scenario. Although changes in upwelling and the PDO are currently not well modeled by GCMs, rising sea surface temperature is very certain and a recent analysis (Burke et al. 2013) found that sea surface temperature has an even stronger effect on ocean survival than the PDO or upwelling. Our analysis indicated that any decline in ocean conditions poses a very serious risk.

Recent increases in survival through the Snake and Columbia rivers prevented climate-driven increases in extinction risk, and buffered against some declines in spawner abundance. This scenario successfully mitigated for the increased extinction risk due to climate conditions in all three populations. Abundance still declined under the worst climate scenarios in Valley Creek, and under 4 of the 6 ocean scenarios in Bear Valley Creek. Nonetheless, across all climate scenarios and all populations, median population abundance was 1.6-2.2 times higher under the new hydrosystem survival rates compared with the 1990s paradigm. The maximum extinction risk dropped from 62% to 19%. The extent to which this survival rate depends on climate conditions, however, is not entirely clear. Over the past 5 years, far fewer fish were transported and spill was high. However, in-river survival depends on fish travel time, which in turn depends on meteorological conditions (especially snowpack and spring melting) in addition to spill, so it is not entirely under the control of dam operators.

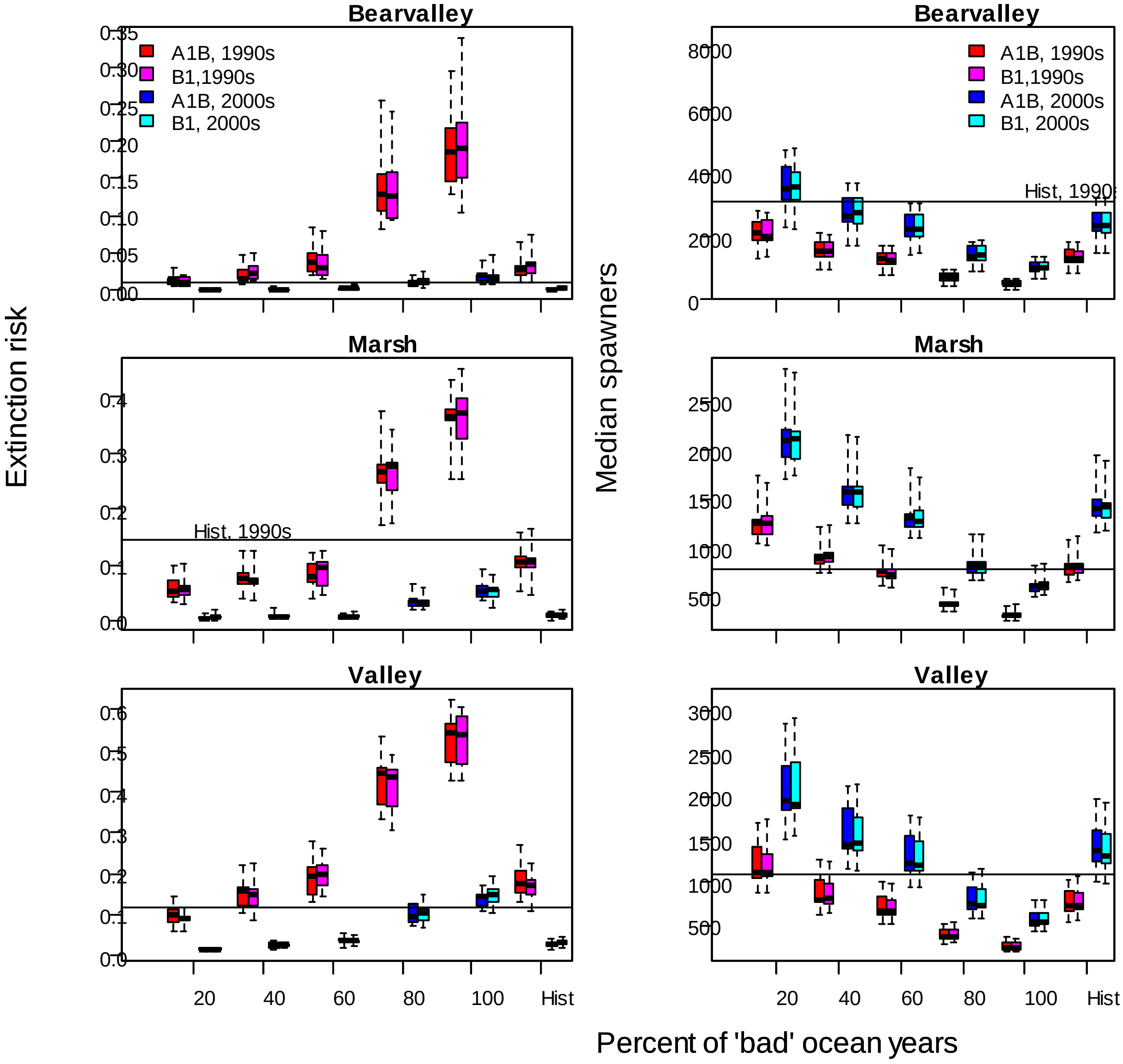

Figure 5. Probability of quasi-extinction (left) and median spawner abundance (right) for three populations as a function of freshwater climate scenarios (A1B or B1), hydrosystem survival ("Current" or "recent"), and ocean conditions. Ocean conditions are characterized in terms of the percent of years with consistently positive PDO, and are compared with the actual historical time series ("Historic"). The baseline scenario used the historical freshwater and ocean conditions and the "current" hydrosystem management, and is shown by the horizontal line. The boxes show the range across all GCMS for a given scenario (line shows the median GCM, the boxes show the interquartile range, and the whiskers show the full range of all GCMs).

Changes in freshwater conditions will likely lower population abundance and raise extinction risks for the most temperature-sensitive populations. However, because most of the habitat within the Salmon River Basin is over 1900-m elevation, populations in the coldest streams will likely benefit from warming, at least temporarily (Crozier and Zabel 2013), and stream temperatures are unlikely to exceed lethal limits in the foreseeable future. Rather, low flows pose an alternative limiting factor for many populations, so declining precipitation could hinder recovery of all of these populations (Crozier et al. 2008b). Because the climate scenarios considered here ranged from only moderate decreases to substantial increases in flow, freshwater conditions for some populations improved under the climate scenarios. Uncertainty stemming from the emissions scenario (A1B vs B1) was negligible because atmospheric CO_2 levels do not diverge between scenarios much by the 2040s. Uncertainty stemming from different GCMs generally did not affect the direction of population response, but only its magnitude.

Our conclusions include some very important caveats. First, we do not fully understand the impacts of temperature and flow from a mechanistic perspective, so our analysis is based on observed statistical relationships; these correlations might change in future decades. Second, climate change might impact other life stages, such as juvenile and adult migrants, and change population viability rates substantially. Finally, important competitors or predators in these streams are already responding to changing conditions, as will prey resources. Invading smallmouth bass and other exotic species already occupy many streams in the Columbia River Basin (Rahel and Olden 2008, Sanderson et al. 2009, Lawrence et al. 2012), with negative effects on native prey (Hughes and Herlihy 2012). Additional management “levers” to control invasive species might well be necessary. Thus our results should not be used as predictions for final decision making. Nonetheless, they do capture a mosaic of responses and point to some potential benefits of possible increases in fall precipitation, risks from warming in this region, and implications of shifting ocean conditions.

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Appendix 1. Bear Valley Creek response metrics[1]

FWclimate	CO2	Ocean	Hydro	GCM	N0	P5.25	P50.25	P95.25	P5.100	P50.100	P95.100	NL	NM	NH	PrQET25	PrQET100	PrQETNmin	MeanSurv	MeanSp
hist	hist	bad.20	current	hist	560	557.9	4599.4	22340	490.9	4484.9	22074.1	790	4848	19853	0.001	0.002	0.01	0.109	4379.9
2040	A1B	bad.20	current	modavg	560	241.8	2038.4	9220	250	2050.9	8768.4	398	2191	8133	0.005	0.012	0.053	0.0974	1997.9
2040	B1	bad.20	current	modavg	560	217.5	2083.4	8922.2	281	2009.4	9115.1	405	2200	8205	0.003	0.006	0.04	0.0971	2021
hist	hist	bad.20	20052009	hist	560	1166.5	7606.1	35357.5	1281.6	7664.9	40529.2	1597	8530	32289	0	0	0	0.0957	7866.5
2040	A1B	bad.20	20052009	modavg	560	502.9	3432.5	13138.3	627.2	3582.8	13511.8	788	3719	12881	0	0	0.007	0.0823	3458.7
2040	B1	bad.20	20052009	modavg	560	601.6	3627	15146	688.6	3570.2	14235.3	776	3732	13072	0	0	0.002	0.0826	3471.3
hist	hist	bad.40	current	hist	560	346.9	3000.6	18916.8	341	3298.2	22559	517	3495	17206	0.004	0.009	0.022	0.1186	3216.7
2040	A1B	bad.40	current	modavg	560	193.9	1539.2	7604.6	198.1	1508.2	7388.9	269	1637	6614	0.003	0.016	0.073	0.107	1486.5
2040	B1	bad.40	current	modavg	560	182	1475.1	6780.4	210.4	1452.9	7220.3	265	1592	6442	0.009	0.016	0.063	0.1066	1450.2
hist	hist	bad.40	20052009	hist	560	715	5417.8	29494.4	887.1	5714	29932.8	1035	6209	26313	0	0	0.001	0.1035	5679.9
2040	A1B	bad.40	20052009	modavg	560	356.8	2653.6	12214.3	419.5	2596.1	11486.3	527	2838	10552	0	0.001	0.004	0.0901	2604.3
2040	B1	bad.40	20052009	modavg	560	377	2556.9	11046.2	433.7	2506.7	12370.7	524	2732	10087	0	0	0.002	0.0899	2522
hist	hist	bad.60	current	hist	560	267.6	2905.6	18585.5	368.7	2965.5	18502.9	437	3007	15102	0.004	0.007	0.037	0.1224	2774.2
2040	A1B	bad.60	current	modavg	560	149.9	1406.4	6705.6	155.7	1291.1	8006.9	196	1332	6532	0.01	0.032	0.123	0.1123	1228.3
2040	B1	bad.60	current	modavg	560	146.7	1238.4	7523.6	162.2	1362	8017.2	195	1298	6460	0.017	0.029	0.095	0.1127	1206.5
hist	hist	bad.60	20052009	hist	560	627.5	5679.5	32814.1	705.6	5519.6	26887.1	906	5467	24323	0	0	0.002	0.1059	5089.3
2040	A1B	bad.60	20052009	modavg	560	319.3	2368.8	12164.3	351	2443	12404.5	390	2330	10355	0	0.001	0.01	0.0955	2177.8
2040	B1	bad.60	20052009	modavg	560	320.2	2398.2	11917.8	318.3	2426.1	11840.2	398	2374	10838	0.001	0.003	0.011	0.0952	2226.6
hist	hist	bad.80	current	hist	560	92.8	1326.3	12350.8	137.6	1636.8	12377	244	1718	10784	0.023	0.048	0.114	0.1375	1623.7
2040	A1B	bad.80	current	modavg	560	57.9	709.7	4083.1	55.9	686.4	4812	108	739	4162	0.029	0.139	0.32	0.1281	694.8
2040	B1	bad.80	current	modavg	560	63.6	683.2	4404.2	64.7	712	5282.5	112	736	4187	0.026	0.116	0.289	0.1276	702.2
hist	hist	bad.80	20052009	hist	560	364.5	2857.2	18502.6	388.6	3016.4	19882.8	543	3221	17205	0	0.001	0.009	0.1205	3099.9
2040	A1B	bad.80	20052009	modavg	560	204	1405.4	7334.4	191.8	1260.9	7452.8	265	1469	7027	0.004	0.012	0.062	0.108	1409.1
2040	B1	bad.80	20052009	modavg	560	184.9	1323.3	7512.1	182.8	1303.5	7729.1	273	1467	6996	0	0.003	0.037	0.1076	1414.8
hist	hist	BAD.iea	current	hist	560	68.9	954.5	6360.1	102.8	938.6	6425.7	165	1117	5371	0.018	0.066	0.168	0.1471	1020.5
2040	A1B	BAD.iea	current	modavg	560	41.7	434.5	2421	40.7	451	2266.2	85	514	2254	0.052	0.173	0.391	0.1399	473
2040	B1	BAD.iea	current	modavg	560	37.4	496.8	2550.9	35.4	415	2468.9	83	520	2286	0.044	0.188	0.396	0.1398	476.5
hist	hist	BAD.iea	20052009	hist	560	281.6	1987.9	9312	274.5	2155.4	11043.4	442	2388	10083	0.003	0.004	0.021	0.1277	2233.2
2040	A1B	BAD.iea	20052009	modavg	560	127.1	1006.5	4776.8	157.9	931.4	4489.6	209	1081	4056	0.004	0.016	0.071	0.1187	1002.1
2040	B1	BAD.iea	20052009	modavg	560	153.3	979.7	4294.5	132.2	1041.2	4620.1	212	1078	4000	0.005	0.015	0.053	0.1175	1001.8
hist	hist	historic.iea	current	hist	560	214	2397.2	17292.1	307.7	2579.9	16635.5	463	3139	15904	0.005	0.01	0.041	0.1208	2905.4
2040	A1B	historic.iea	current	modavg	560	129.9	1185.1	6511.1	147.6	1142	5839	228	1403	6416	0.007	0.025	0.091	0.1105	1301.8
2040	B1	historic.iea	current	modavg	560	117.2	1247	6220.1	132.8	1176.4	6284.4	234	1403	6390	0.009	0.023	0.085	0.1097	1305.6
hist	hist	historic.iea	20052009	hist	560	519	4460.7	24732.4	657.3	4999.9	26318.5	937	5475	25717	0	0.001	0.002	0.1056	5165
2040	A1B	historic.iea	20052009	modavg	560	300.6	1976.1	10883.7	307	2259.7	10481.5	468	2436	10074	0	0	0.008	0.0937	2304.7
2040	B1	historic.iea	20052009	modavg	560	299.3	1992.5	9652.4	287.8	2002	10926.1	470	2450	10199	0.002	0.002	0.011	0.0932	2312.7

The “Fwclimate” column indicates the historic freshwater conditions (“hist”), or conditions characteristic of the 2040s. “CO2” column indicates historic atmospheric conditions or the A1B or B1 emissions scenario. “Ocean” indicates the ocean scenario (see text). “Hydro” is the mainstem survival management lever. N0 is the initial number of spawners at the beginning of each simulation. The next six columns show the 5th, 50th, and 95th quantile spawner abundance across simulations at a given time point, either in simulation year 25 or year 100. NL, NM and NH are the geometric means across simulations of the 5th, 50th, and 95th quantile spawner abundance, respectively, within each simulation across the full 100 years. PrQET indicates the probability of passing the quasi-extinction threshold of 50 spawners within the first 25 years (PrQET25) or at any time within the full 100 years of simulation (PrQET100), or of passing a population-specific threshold (historical minimum) in 100 years (PrQETNmin). MeanSurv shows the average parr-to-smolt survival rate for each scenario. See Zabel et al (2013) for more explanation of these metrics.

APPENDIX MS2013-03. OCEAN CONDITIONS AND SELECTED MANAGEMENT OPTIONS ON THE POPULATION DYNAMICS OF WENATCHEE RIVER SPRING CHINOOK SALMON

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EXECUTIVE SUMMARY

Ocean conditions have a large influence on salmon population dynamics, and predicting future impacts of climate change on salmon populations requires forecasting ocean conditions and consideration of the implications for abundance and persistence of populations. However, there is much uncertainty regarding the implications of climate change on local and basin-scale oceanography. Lacking downscaled climate-ocean models relevant to salmon, we can consider a range of ocean condition scenarios, and evaluate to what extent potential management options can compensate for poor ocean conditions. Here I apply scenarios for climate and management actions, focusing on responses of Wenatchee River spring Chinook salmon, a population listed as endangered under the Endangered Species Act. Predictions of population responses are available from a stochastic Leslie matrix-type salmon life cycle model that combined scenarios of simulated future ocean conditions with estimated effects of management actions that affected freshwater (prespawning adults, and rearing juvenile fish), mainstem (smolt migration through the Federal hydropower system) and estuary (avian predation). Scenarios for ocean conditions consisted of alternative percentages of years when ocean conditions during early ocean entry by salmon were favorable (negative mean annual Pacific Decadal Oscillation [PDO] values) and unfavorable (positive PDO values) for survival. Compared to a benchmark scenario, median spawners and carrying capacity declined with worsened ocean conditions. When we applied management actions individually, freshwater survival increases had the strongest effect on mitigating for poor ocean conditions compared to the mainstem hydropower dam and estuary survival improvements. Taken together, both freshwater, mainstem, and estuary management actions offset the effects of some moderate declines in ocean condition, but not the poorest ocean conditions considered in these scenarios. Future salmon life cycle modeling should consider other aspects of potential future ocean conditions, such as the frequency and magnitude of bad and good PDO periods, upwelling, and ocean variability.

BACKGROUND/INTRODUCTION

Ocean conditions have a large influence on salmon population dynamics (e.g., Koslow et al. 2002; Scheuerell and Williams 2005; Wells et al. 2008; Burke et al. 2013), and predicting future impacts of climate change on salmon populations requires forecasting ocean conditions and consideration of the implications for abundance and persistence of populations. These ocean conditions are a function of both regional and basin-scale processes (e.g., Mantua et al. 1997; Peterson et al. 2012). For instance, Jorgensen et al. (2013) and Crozier et al. (2013), respectively, have identified coastal upwelling and the Pacific Decadal Oscillation (PDO) as correlates of ocean survival for Chinook salmon (*Oncorhynchus tshawytscha*) populations in the US Pacific Northwest. Wells et al. (2008) found that Chinook salmon in the Smith River, California benefited from cool ocean temperatures and strong upwelling, wind stress, and a strong California Current. A critical question for decision makers is whether potential management actions can buffer or offset changes to these regional and basin-scale drivers that may stem from climate change.

There is much uncertainty regarding the implications of climate change on local and basin-scale oceanography. At the scale of the entire subarctic North Pacific Ocean, Schindler and colleagues (2008) note that increases in salmon production over the last several decades are linked to cool, productive ocean conditions, which may not persist under warming trends due to climate change. King et al. (2011) summarize potential implications of climate change in the California Current, and conceptual linkages between climate change and the potential response of Chinook salmon. King and colleagues summarize ensemble results from global circulation models, which suggest the potential for slight warming by year 2050, and some minor increases in upwelling intensity, particularly in the northern California Current. They identify risks of climate change to Chinook salmon, specifically the potential for a weakened California Current that could depress fecundity and increase mortality, and for ocean warming to favor northern populations over southern populations. These authors and Hollowed et al. (2013) note the difficulty in inferring local patterns from coarse scale global circulation models, and they point to the need for downscaled, finer resolution oceanographic modeling. Downscaled oceanographic models, forced by coarser scale global circulation models under IPCC CO_2 emissions scenarios, are in progress but not yet available for the marine environment. This contrasts with more extensive downscaling of global circulation models that has been applied to freshwater portions of Chinook salmon habitat (Battin et al. 2007; Crozier et al. 2008; Beechie et al. 2012).

In lieu of downscaled climate-ocean models for salmon, we can consider a range of ocean condition scenarios, and can evaluate to what extent potential management options can compensate for poor ocean conditions for salmon. This approach is consistent with the use of scenarios for ecological assessment and planning (Millenium Ecosystem Assessment 2005; Alcamo 2008) and with the need to identify management options that are robust to a range of uncertain future ocean conditions. Management options might include freshwater habitat restoration, which could improve survival of adult spawning fish and the juvenile freshwater rearing stage, modifications to hydropower operations, and a reduction in avian predation of juvenile outmigrants in the Columbia River estuary.

Here I apply these scenarios for climate and management actions, focusing on responses of Wenatchee River spring Chinook salmon, a population listed as endangered under the Endangered Species Act. Predictions are available from a life cycle model being developed by a team of researchers as a part of the Adaptive Management Implementation Plan (AMIP) of the 2008 Federal Columbia River Power System Biological Opinion (FCRPS

Biop) (Jorgensen et al. 2013). Life cycle models are useful tools to explore environmental change and resource management options on species population dynamics. The 2008 FCRPS Biop used life cycle models of Pacific salmonid populations to examine the effects of hydropower system dam operations on population viability under a range of future climate and hydropower system operations scenarios. The AMIP, an addition to the 2008 FCRPS Biop, called for an extension of these models to include more populations, and to include several types of effects, including habitat mitigation actions, and climate (Zabel et al. 2013). The work below considers habitat actions and climate, as well as management actions related to hydropower and avian predation.

STOCHASTIC LIFE CYCLE MODEL AND SCENARIOS

In this section we briefly describe the model, outline a few model scenarios, and provide and discuss some preliminary results.

THE MODEL

The Wenatchee spring Chinook life cycle model framework is built from a Leslie matrix age-structured population model for stream-type spring Chinook salmon (Zabel et al. 2006; ICTRT and Zabel 2007; Jorgensen et al. 2013). It tracks population numbers across five life stage classes through time: parr, smolts, ocean residence (from one to three years), and tributary spawners (four and five year old fish that spent two and three years, respectively, in the ocean). The following is a brief description of the model, but see ICTRT and Zabel (2007) and Jorgensen et al. (2013) for more details. The model is coded and runs in the R statistical and programming environment (R Development Core Team 2013).

The number of individuals at time $t+1$ is represented by $\mathbf{n}$, which is a 5 x 1 vector of the number of individuals at each of five life stages, and is a product of a 5 x 5 transition matrix, $\mathbf{A}(t)$, the dimensions of which reflect the five life stages incorporated into the model and the entries of which change with t, and the number of individuals in each of the life stages, $\mathbf{n}$, at time t:

$$\mathbf{n}(t+1) = \mathbf{A}(t) \cdot \mathbf{n}(t).$$

The elements in each row of $\mathbf{A}(t)$ determine the transition of individuals at one life stage progressing through to the next life stage, from one row in the $\mathbf{n}(t+1)$ matrix down to the next:

$$\mathbf{A}(t) = \begin{bmatrix} 0 & 0 & 0 & b_4 \cdot s_A \cdot F_4(t) & s_A \cdot F_5(t) \\ s_2 & 0 & 0 & 0 & 0 \\ 0 & s_3(t) & 0 & 0 & 0 \\ 0 & 0 & (1 - b_3) \cdot s_o & 0 & 0 \\ 0 & 0 & 0 & (1 - b_4) \cdot s_o & 0 \end{bmatrix}.$$

The s_is are the survival probabilities of moving from one life stage to the next. s_2 is the survival probability of parr to the smolt stage (moving from one-year-old fish to two years old). $s_3(t)$ is the survival probability of the transition of fish from two to three years old, the period in which fish leave freshwater and enter the estuary and ocean, corresponding to their first year of ocean residency. The s_3 term accommodates stochasticity and varies in time and according to scenarios of climatic and ocean conditions. s_o represents the subsequent annual probability of ocean survival, which was fixed at 0.80 (TRT and Zabel 2007). The proportion of three and four year olds leaving the ocean and returning to spawn (their breeding propensities) are noted by b_3 and b_4, thus, the proportion of three and four year old fish remaining in the ocean is given by $(1 - b_3)$ and $(1 - b_4)$. s_A is the survival of adults from Bonneville dam to the spawning grounds, and is a product of upstream survival through the Columbia River mainstem dam system, s_u, survival after in-river harvest, $(1 - h_r)$, and survival from the upper-most dam to the Wenatchee basin, s_{sb}. Fecundity in some cases for some fish species may be different for spawning fish of different ages, and the model can accommodate this differential with a fecundity multiplier, the F_i terms.

Table 1: Parameter inputs for the Wenatchee River spring Chinook salmon matrix-type model for three major production areas: Chiwawa River, Nason Creek, and the White River.

Parameter	**Chiwawa River**	**Nason Creek**	**White River**
Spawner(t) -to- parr$(t+1)$ Beverton-Holt "a"	353.437	328.490	154.318
Spawner(t) -to- parr$(t+1)$ Beverton-Holt "b"	0.000298	0.005	0.005
$\sigma^2{}_1$	0.412	0.600	1.04
ϕ_1 (variance term)	0.1	---	---
Parr-smolt survival[1]	0.6	0.6	0.6
Hydrosystem survival	0.525	0.525	0.525
s_3 (first ocean year)	Stochastic variable, dependent on relationship to ocean conditions	Stochastic variable, dependent on relationship to ocean conditions	Stochastic variable, dependent on relationship to ocean conditions
s_o (ocean survival for years after s_3)	0.8	0.8	0.8
b_3 (propensity of 3 year olds to breed)	0.046	0.046	0.046
b_4 (propensity of 4 year olds to breed)	0.514	0.514	0.514
h_r (in-river harvest rate)	0.09	0.09	0.09
s_u (Bonneville-to-basin survival rate)	0.794	0.794	0.794
s_{sb} (pre-spawning survival rate)	0.9	0.9	0.9
Initial abundance of 4 and 5 year old tributary spawners (geometric mean of 2008-2012 counts)	406	148	38

[1] Parr-smolt survival measures survival from exiting the tributaries until reaching the mainstem Columbia, derived in original matrix model.

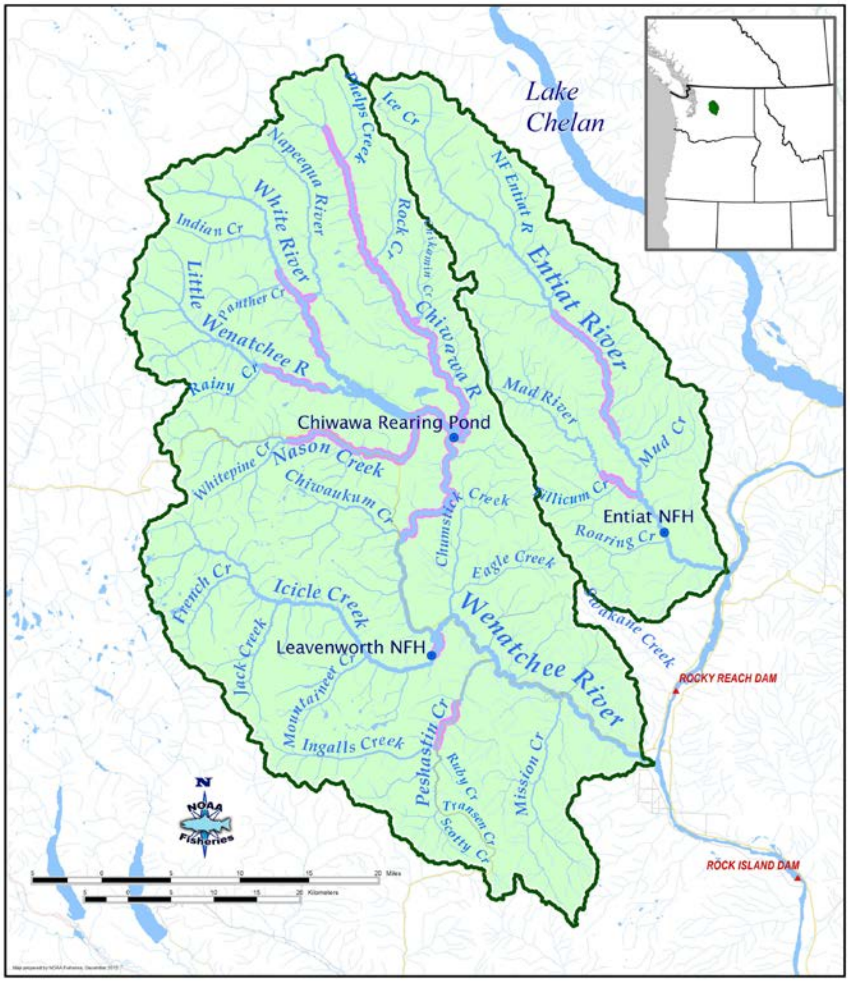


Figure 1: Wenatchee and Entiat River basins, with areas of recent spring Chinook salmon spawning and rearing indicated with highlights (in pink).

MULTIPLE PRODUCTION AREAS

We developed this life cycle model to incorporate three of the major fish production areas in the Wenatchee River basin representing the vast majority of fish production: Chiwawa River, Nason Creek, and the White River (Fig. 1). At the present time, we include three of these in the model: Chiwawa River (parameters as in the 2007 report, and shown in Table 1), Nason Creek, and the White River (parameters for both are reported in Table 1). The model essentially functions as though there are alternative transition matrices, $\mathbf{A}(t,j)$, and population vectors, $\mathbf{n}(t,j)$, for each production area, j, with production-area-specific parameters where appropriate or where data were available to estimate them (Table 1). As the model moves through time, each of the production areas' life stage transition survival calculations are handled separately, and the numbers of fish within the age classes in each of the $\mathbf{n}(t,j)$ vectors were summed to create one $\mathbf{n}(t)$ vector representing the entire Wenatchee population, which was used for calculations of overall population metrics such as the geometric mean of spawners, mean recruits per spawner, and for calculations of extinction probabilities (see explanations of these in "Model Output Response Measures" section below). As a consequence of the spatial coverage of the model's structure, we have implicitly begun to encompass two of several juvenile life history strategies. For example, juveniles from Nason Creek typically migrate to and rear in the mainstem Wenatchee River, rather than remain in their tributary to rear until they begin to migrate to the ocean. This alternative life history strategy can often lead to differential survival as they progress through subsequent freshwater and ocean life stages. As we develop the model further we will more explicitly incorporate juvenile life history variation.

SPAWNER-TO-PARR TRANSITION AND SURVIVAL

There is density dependence built into the spawner to subsequent parr transition, which was estimated for three of the five production areas by fitting a density-dependent Beverton-Holt (B-H) relationship to spawners and subsequent parr, s_1; $\text{parr}(t+1) = \left(a \cdot S(t)\right)/\left(1 + b \cdot S(t)\right)$. B-H estimates of "$a$" and "$b$" parameters for the Chiwawa River fish were from the ICTRT and Zabel (2007) Wenatchee matrix-type model, and derived by dividing recruits by the product of prespawning survival, smolt-to-adult return rates, and parr-to-smolt survival; $\text{parr}(t+1) = R_t/\left(s_{sb} \cdot \text{SAR}_{t+2} \cdot s_{p-s}\right)$. Chiwawa River estimates included a Box-Cox transformation as a way to deal with the heteroscedasticity in the data (the σ_1^2 and $\phi_1$parameters; see Zabel et al. 2006 and ICTRT and Zabel 2007 for details). Nason Creek and White River B-H models were fitted to spawner and parr estimates from those subbasins (Washington Department of Fish and Wildlife, unpublished data). The short spawner and parr time series for Nason Creek and the White River didn't allow the Box-Cox transformation and estimation of these parameters, thus the s_1 function for these production areas was of the simpler form without the Box-Cox transformation. In place of those in Table 1 for Nason Creek and White River we report the variance, σ^2, of the B-H fits.

MODEL OUTPUT RESPONSE MEASURES

The following model output summary metrics are reported:

- *X*th percentile of spawner abundance at time t = 100 years, taken across runs. The percentiles are X = 5%, 50% (median), and 95% ($N_{t,5\%}$, $N_{t,50\%}$, $N_{t,95\%}$).
- Geometric mean (taken across runs) of low (L), medium (M), and high (H) (calculated across years within a run) of spawner abundance, where $\bar{N}_{\text{L}}$ = the 5th percentile, $\bar{N}_{\text{M}}$ = the 50th percentile, and $\bar{N}_{\text{H}}$ = the 95th percentile within a run.
- Probability of quasi-extinction for simulations that ran t = 100 years (pr(QE)_t). We calculated the probability that the population would fall below 50 spawners in a moving average of four years.

We calculated two additional population dynamics metrics in response to these scenarios for purposes of comparison with other reports of this model's output on different sets of scenarios. They included productivity at low spawner abundance and carrying capacity. To calculate these metrics we first produced spawner and recruit data from model simulations for a given scenario. We ran several iterations (n = 10), and then combined all these data together. We then fit the following Beverton-Holt relationship to these data:

$$R_t = \frac{a \cdot S_t}{1 + b \cdot S_t} \cdot \exp(\varepsilon_t), \; \varepsilon \sim N(0, \sigma^2),$$

where R_t and S_t are recruits and spawners, respectively, in brood year t, a and b are model parameters, and with a multiplicative lognormal error term, ε_t. The parameter a represents maximum productivity (recruits per spawner) at low abundance, and a/b represents the maximum asymptotic recruitment. From these fits, we reported carrying capacity, K, at equilibrium population abundance where $R = S$, which is equal to $(a - 1)/b$. In some cases K was negative which indicated unfavorable conditions, as presented in the model by detrimental combinations of scenarios and parameter values. Negative K values were reported as “NA.”

Taken together, these output metrics give a snapshot of the health of a population in response to a given set of environmental and management actions.

We ran simulations for t = 100 years and for each scenario we repeated model runs for n = 1,000 times to obtain a robust estimate of quasi-extinction probabilities.

SCENARIOS: FUTURE OCEAN CONDITIONS

Ocean conditions enter the life cycle model in survival during the third year of life, $s3$, when fish migrate out of their natal tributary basin and enter the estuary and ocean and begin their ocean residency period. We do not have direct measurements of $s3$, however we can estimate it from annual measurements of smolt-to-adult survival, SAR. We estimated $s3$ from SAR data and, treating $s3$ as a response variable, we found relationships between $s3$ and ocean indices (ICTRT and Zabel 2007; Kendall et al. 2013). For Wenatchee spring Chinook salmon, spring coastal upwelling (April and May; Pacific coastal upwelling index at 45°N 125°W) and river transit time to reach the estuary were important drivers of $s3$ (ICTRT and Zabel 2007; Jorgensen et al. 2013; Kendall et al. 2013).

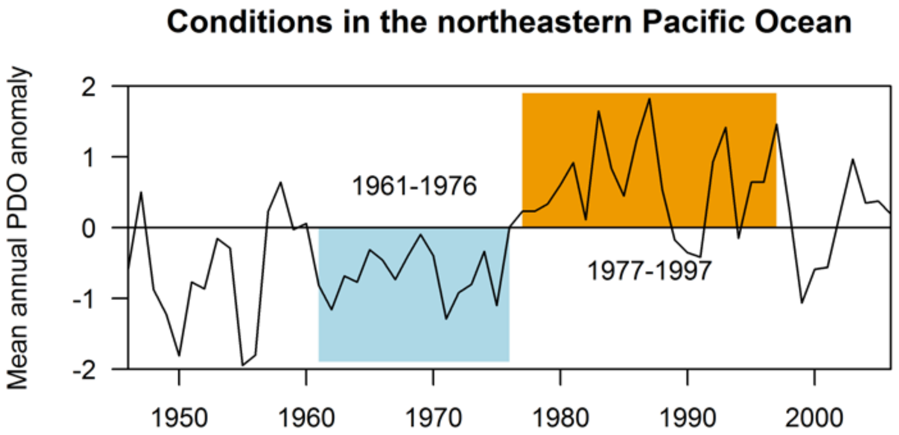


Figure 2: Ocean conditions as measured by Pacific Decadal Oscillation anomalies in recent years, with relative periods of favorable (cooler ocean surface waters, 1961-1976; blue) and unfavorable (warmer ocean surface waters, 1977-1997; orange) conditions for Pacific salmon survival in the ocean used to develop scenarios of future ocean conditions.

For the future ocean conditions scenarios, we developed time series of ocean indices for periods composed of differing amounts of "good" and "bad" periods as measured by the Pacific Decadal Oscillation anomaly index (PDO, Fig. 2; Mantua et al. 1997; http://jisao.washington.edu/pdo/PDO.latest). These time series determined third year survival, *s*3. We focused on PDO as a measure of ocean conditions to remain consistent across the ocean conditions scenarios for Chinook salmon life cycle modeling included in this report (see Crozier and Zabel, this report). Negative PDO values indicate cooler northeastern Pacific Ocean surface waters and promote conditions favorable to salmon ocean survival, whereas positive PDO values indicate warmer northeastern Pacific Ocean surface waters which are associated with conditions generally unfavorable to salmon survival in the ocean (e.g., Mantua et al. 1997; Peterson et al. 2012; Fig. 2). Our time series consisted of different percentages of good and bad years taken from these time periods (Fig. 2): 20% bad; 40% bad; 60% bad; 80% bad; 100% bad. We interleaved the good and bad year blocks (approximate 15 and 20 yr blocks, respectively; Fig. 2) to achieve the desired scenario compositions for time series of 100 years in length; thus, except for the 20% and 100% bad scenarios, blocks of bad and good years alternated through the series. In addition to these scenarios, we also ran a scenario encompassing good and bad periods and the intervening years (1946-2006; Jorgensen et al. 2013). During each run of 100 years in the model, the model randomly chose a starting point in the ocean time series as one part of the model's procedure to introduce stochasticity into third year survival (ICTRT and Zabel 2007). Due to this random starting process the time series were duplicated and stacked as necessary to allow the model to complete each 100 yr run.

The model's procedures for estimating third year survival from ocean conditions as well as the scenarios of future ocean conditions we describe above were the same as those of Crozier and Zabel (this report).

SCENARIOS: MANAGEMENT LEVERS

We combined future ocean conditions scenarios with several resource management scenarios (Table 2). These included:

Freshwater habitat improvements (s_{sb} and s_1 survival)

These parameter perturbations were used as a proxy for habitat improvements through freshwater restoration actions in adult spawning and juvenile rearing reaches, which could impact returning adult fish prior to spawning (prespawning mortality, s_{sb}) and the spawner-to-parr stage (s_1). Presently, we are developing relationships between freshwater habitat actions and fish survival. Therefore, habitat improvements were simulated by increasing survival at these life stages, and as Wenatchee model development continues, these perturbations will be replaced with relationships between freshwater habitat characteristics and in-basin survival estimates.

Improved survival in mainstem and estuary

We explored population dynamics in response to improvements in downstream smolt survival through the FCRPS dams (applied to s_2). In the same scenario, we also applied a multiplier on survival in the estuary (applied to s_3) that estimated a reduction in avian predation on smolt in the estuary (Paulsen and Zabel 2013).

Table 2: Resource management scenarios included in this study.

Survival stage	Change
Prespawning and spawner-parr (s_{sb} and s_1) survival[1,2]	+10%
FCRPS survival (s_2) and avian predation[2]	+10% FCRPS, and -50% reduction in avian predation

[1] Survival changes were applied to survival of spawners from the last dam to their tributaries (s_{sb}) and to the number of parr in $t+1$ produced by spawners in t (s_1) simultaneously.

[2] No decrement was applied to these survivals.

COMBINED EFFECTS OF FUTURE OCEAN CONDITIONS AND MANAGEMENT ACTIVITIES

The benchmark scenario consisted of typical regime shifts in ocean conditions observed over the last ~60 years, with approximately 46% unfavorable and 54% favorable years of ocean conditions for salmon from 1946-2006 as measured by the number of positive and negative PDO anomalies (Fig. 2). Median spawner abundance over the 100 year simulations was 860 individuals. The probability of quasi-extinction was small, only

0.001. However, interpretation of extinction probability as presented here must be done with care. Extinction probability in this analysis was defined solely on the frequency of falling below a low abundance threshold level (below 50 spawners in a four-year moving average). Abundance is one of several metrics used to determine species viability and population persistence (McElhany et al. 2000; ICTRT 2007). Taking into account additional measures of population persistence (i.e., abundance, productivity, spatial structure, and diversity), a recent assessment rated this population to have a high risk of extinction (ICTRT 2007; UCSRB 2007). Therefore below I focus more on metrics of abundance than of extinction.

In the absence of new management actions, ocean conditions alone drove median spawner abundance ($N_{100,\,50\%,}$) down by over 40% in the worst-case ocean scenario (Table 3). Across the range of ocean conditions tested, poorer ocean conditions led to reduced numbers of median spawners and a slightly increased probability of extinction. Carrying capacity also declined with worsening ocean conditions. Although we observed declining spawner abundance with increasingly unfavorable ocean conditions, there were little to no increase in extinction probabilities. Certainly, the combined effects of poor ocean conditions with detrimental impacts from freshwater residency, mainstem and estuary occupancy periods could have severe impacts on this population.

As management actions were applied, median spawners generally increased relative to scenarios where there were no management actions (Table 3). Response metrics generally mirror patterns in median spawner abundance, so I focus on median spawner abundance below. In the absence of changes in ocean condition, freshwater habitat actions improved median spawner abundance by approximately 30%, while improved survival during dam passage and in the estuary improved these population metrics by approximately 15%. Combining both types of actions led to nearly additive (~45%) improvements in these metrics.

Neither management actions improving freshwater habitat, nor actions to improve survival in the mainstem and the estuary, could completely reverse the impacts of the most extreme declines in ocean condition. Although freshwater habitat actions could improve median spawner abundance, habitat actions could not reverse the decline in median spawner abundance (relative to benchmark scenario) caused by the two worst ocean conditions scenarios (80-100% bad years; Table 3). Similarly, although mainstem FCRPS improvements combined with reduced avian predation in the estuary led to increased median spawner abundance, these actions could not compensate for the three worst ocean condition scenarios (60%-100% bad years).

Combining both types of management actions, under the historical ocean conditions, led median spawner abundance to increase by more than 40% (Table 3). Under the worst ocean conditions, the combination of freshwater with mainstem and estuary survival improvements had the least reduction in median spawners (~-20%) of any of the other scenario. When ocean conditions were poor at most 60% of the time, these combined management actions allowed median spawner abundance to remain above the benchmark level of median spawner abundance. Under the worst ocean conditions (80-100% poor conditions), these combined management actions led to declines in median spawner abundance and other population metrics, relative to the benchmark scenario.

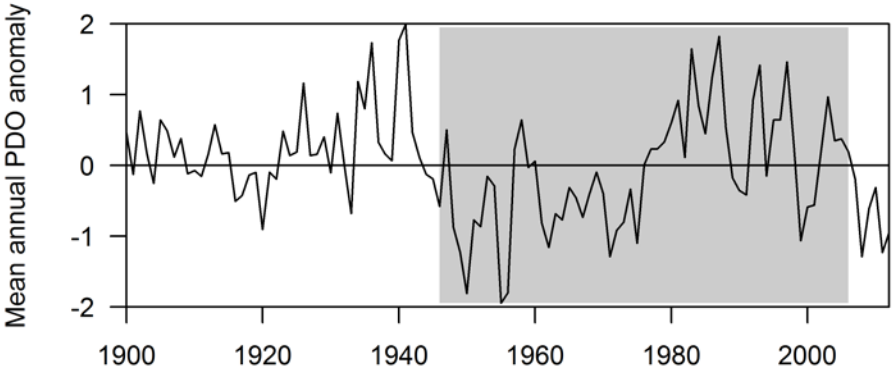
DISCUSSION

The results above suggest that if climate change increases the frequency of years with positive PDO, which are generally unfavorable to salmon survival in the ocean, the management actions considered may be able to buffer the Wenatchee River spring Chinook population to a limited extent. Drastic increases in the frequency of poor ocean conditions, i.e. positive PDO for >60% of years, could not be countered by the management actions tested – thus a key question is to what extent such poor ocean conditions are expected to occur. Improved downscaled climate-ocean modeling is needed to better forecast likely future patterns in basin-scale metrics such as PDO, as well as measures of local productivity.

The management actions tested here led to substantial increases in abundance of this Chinook population. Encouragingly, these actions were modest, involving 10% improvements in survival rates and 50% reductions in avian predation. Thus there appears to be some scope for management of Wenatchee River Chinook salmon to adapt to declining ocean conditions, though we do not evaluate the costs, tradeoffs, or other additional mitigating actions.

In the work above, the positive phase of the PDO is used as a proxy to identify years with poor ocean conditions for salmon in general. Consistent with this, Crozier et al (2013) found that PDO was a significant predictor of survival during ocean residency of some Chinook salmon stocks. However, for Wenatchee spring Chinook salmon, survival was better predicted by upwelling intensity (Jorgensen et al. 2013). Therefore, for the Wenatchee population, in addition to several other Columbia River salmon and anadromous trout populations (Kendall et al. 2013), future downscaled predictions of upwelling are particularly critical; simulations that increase frequency of years with poor upwelling may depress the population more than the simulations here that increased frequency of years with positive PDO.

Our modeled ocean conditions were developed from a recent period of observations which included cold and warm periods. However, it is difficult to predict future ocean conditions, and it is uncertain whether variability in ocean conditions will be analogous to or different from the period we used to develop our scenarios. Recent conditions in the further recent past (to 1900) had relatively more years of transition between warm and cold conditions (Fig. 3). A more variable ocean combined with more frequent and persistent warm periods could increase extinction probability. We did not explore changes in ocean variability, but this is an important aspect that should be addressed in assessing the effects of future ocean conditions on salmon with life cycle models.



356 **Figure 3:** PDO anomaly from 1900-2012, with the time period used in this study to construct scenarios of future ocean conditions boxed in gray.

Table 3: Estimated impacts of management actions on the number of Wenatchee River basin wild spring Chinook salmon spawners using a life cycle model that incorporated scenarios of simulated future ocean conditions. The geometric mean of the number of wild spawners for the five year period 2005-2009 (from the Salmon Population Summary database https://www.webapps.nwfsc.noaa.gov/apex/f?p=261:home:0) was 576 spawners.

Ocean conditions	**Avian predation**	**FCRPS survival**	**Freshwater survival**	$N_{100,\,5\%}$	$N_{100,\,50\%}$	$N_{100,\,95\%}$	$\bar{N}_{100,\,5\%}$	$\bar{N}_{100,\,50\%}$	$\bar{N}_{100,\,95\%}$	$\Pr(QE)_{100}$	K	R/S_{low}
Benchmark												
Historical	current	current	current	210	860	3979	323	843	2790	0.001	754	1.58
Altered ocean condition												
20% bad	current	current	current	220	822	3074	317	817	2177	0.002	879	1.58
40% bad	current	current	current	188	737	2907	276	735	1978	0.005	704	1.84
60% bad	current	current	current	168	632	2662	241	642	1804	0.001	654	1.97
80% bad	current	current	current	136	549	2257	211	542	1512	0.009	633	1.50
100% bad	current	current	current	123	493	1872	196	486	1214	0.008	518	1.45
Freshwater habitat actions												
Historical	current	current	+10%	280	1111	4996	423	1098	3652	0	857	1.57
20% bad	current	current	+10%	291	1049	3920	399	1052	2838	0	947	1.52
40% bad	current	current	+10%	238	901	3455	337	908	2470	0	943	1.92
60% bad	current	current	+10%	221	859	3465	317	860	2411	0	759	2.09
80% bad	current	current	+10%	172	668	2768	252	666	1894	0.001	583	1.92
100% bad	current	current	+10%	158	606	2380	246	607	1504	0.001	574	1.43
Mainstem hydrosystem and estuary actions												
Historical	-50% reduced	+10%	current	251	1004	4633	377	989	3395	0	1110	2.46
20% bad	-50% reduced	+10%	current	267	976	3756	375	985	2627	0	NA	0.92
40% bad	-50% reduced	+10%	current	213	826	3278	312	832	2252	0.001	622	1.88
60% bad	-50% reduced	+10%	current	195	734	2993	276	736	2086	0	638	1.56
80% bad	-50% reduced	+10%	current	163	642	2631	241	639	1835	0.003	646	1.87
100% bad	-50% reduced	+10%	current	139	541	2214	222	547	1355	0.004	415	1.40
All management actions combined												
Historical	-50% reduced	+10%	+10%	303	1226	5477	458	1202	4041	0	1264	1.76
20% bad	-50% reduced	+10%	+10%	351	1254	4668	481	1261	3381	0	980	1.62
40% bad	-50% reduced	+10%	+10%	280	1055	4055	393	1056	2873	0	973	2.92

Ocean conditions	**Avian predation**	**FCRPS survival**	**Freshwater survival**	$N_{100,\,5\%}$	$N_{100,\,50\%}$	$N_{100,\,95\%}$	$\bar{N}_{100,\,5\%}$	$\bar{N}_{100,\,50\%}$	$\bar{N}_{100,\,95\%}$	$\Pr(\mathrm{QE})_{100}$	K	R/S_{low}
60% bad	-50% reduced	+10%	+10%	257	970	3916	357	979	2759	0	896	2.07
80% bad	-50% reduced	+10%	+10%	206	811	3338	307	806	2287	0	748	2.05
100% bad	-50% reduced	+10%	+10%	183	700	2690	282	698	1738	0.001	443	1.56

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APPENDIX MS2013-04. APPLICATION OF THE NORTHERN CALIFORNIA CURRENT ECOTRAN MODEL TO PELAGIC ECOSYSTEM SCENARIOS FOR THE 2013 CALIFORNIA CURRENT INTEGRATED ECOSYSTEM ASSESSMENT

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SUMMARY

An end-to-end ecosystem model of the Northern California Current (NCC ECOTRAN) is used to investigate the scale of effects that the observed variability within the plankton, jellyfish, and forage fish community have upon higher trophic levels and upon fishery production. Ocean survey observations of the plankton and pelagic fish community made over the 1998 - 2007 period allow for basic inferences of inter-annual differences in food web structure. Study of the wide range of ocean and biological conditions over this decade-long period can improve our model extrapolations and strengthen our confidence in predicted ecosystem responses to long-term climate change. The simple scenarios developed here alter only a few key lower- and mid-trophic level groups, but they demonstrate that changes in community composition and trophic structure can have effects upon higher trophic levels and fisheries as important as variability in primary production levels. Thus, consideration of changes in trophic relationships, that can result both from long-term changes in local climate and inter-regional changes in migration patterns population distributions, are as necessary as the study of the impact of climate change upon individual species.

INTRODUCTION AND GOALS

Understanding the effects of climate variability and climate change is a key challenge for marine resource managers on the US West Coast and for the Integrated Ecosystem Assessment. Hollowed *et al.* (2013) identify eight foci needed to improve the projections of climate impacts on fish, fisheries, and fishery-dependent communities. Among these is improved understanding of the mechanisms underlying fish and shellfish responses to environmental drivers. These authors argue that observations and understanding of the present-day responses of prey groups, such as zooplankton and forage fish, to changes in ocean condition are needed to predict future responses to climate change.

For the Oregon and Washington continental shelf, a relatively rich data set of plankton, forage fish, and primary production is available in summer months (e.g., Brodeur *et al*., 2005; Emmett *et al*., 2006; and sources cited below). Below, I combined these data sets within the framework of an end-to-end trophic network model, the NCC ECOTRAN ecosystem model, to ask: How does inter-annual variability in food web structure affect specific groups and fisheries production? The NCC ECOTRAN model is used to estimate ecosystem-level and functional group responses to observed inter-annual changes over the past decade in phytoplankton production and biomass, copepod community composition and biomass, the biomass of large jellyfishes, and changes in the forage fish community. These scenarios provide a baseline measure of current inter-annual variability and point to considerations necessary to design scenarios predicting responses to future climate change.

Punt and colleagues (2013) discuss the simulation testing of fishery management strategies for climate change. Given the uncertainties related to precisely forecasting species responses to climate, these authors argue for a more general consideration of how the ecological system may change in the future, and whether management strategies are robust to this change. The work below illustrates recent inter-annual shifts in the food web, and can inform how climate-driven shifts in productivity may alter fisheries in the future.

MODEL STRUCTURE AND SCENARIO METHODOLOGY

The “NCC ECOTRAN” end-to-end ecosystem model describes the trophic interactions between 83 functional groups and 17 separate fisheries and gear types in the benthic and pelagic environments of the Northern California Current upwelling system (Steele & Ruzicka, 2011; Ruzicka *et al.*, 2012). The model domain covers the Oregon and Washington continental shelf ecosystem during the summer seasons (June - September).Here, ten alternate configurations of the model are developed, one representative of the food web structure for each year 1998 through 2007.

At its heart, the NCC ECOTRAN model is a map of the flow of energy through the entire food web from lower trophic-level producers to upper trophic-level consumers and fisheries. As an “end-to-end” model, it accounts for nutrient input via upwelling, includes nutrient recycling via bacterial metabolism of detritus, and can account for advective losses of plankton production. The production of each functional group within the trophic network may be driven by nutrient or plankton production input at the base of the food web — as from a plankton production model or an upwelling index time-series. The distribution of all energy consumed by each group to metabolism, to production, to each predator and fishery group, and to detritus as feces or unconsumed production is taken into account. The response of upper trophic levels to changes to plankton dynamics, community variability or other energy flow rearrangements, or to physiological changes can readily be estimated. Further, in this configuration the propagation of parameter uncertainty (including diet uncertainty) through the model is accounted for (Ruzicka *et al.*, 2011; Fig. 1).

Production of biological groups for a particular year, as estimated by NCC ECOTRAN, is the creation of new biomass – the sum of recruitment of juveniles and growth in body size during that year. In the long term, changes in production, after accounting for mortality, will lead to concomitant changes in stock size and biomass. However, on an annual scale, changes in production may not lead to substantial, immediate changes in stock size or biomass, particularly for long-lived species. For instance, a baleen whale population with a biomass production rate of only 4% yr^{-1} will not crash if that production rate is halved for a single year. However, the focus on production in NCC ECOTRAN is particularly relevant for considering climate effects, which we can envision as long term extensions of production rates that will ultimately impact stock size and biomass.

Similarly, NCC ECOTRAN focuses on annual fishery production, which is the product of biomass production of targeted species and a fishing mortality rate. It is akin to removing a fixed fraction of ‘surplus production’; it differs from removing fixed fractions of (standing stock) biomass. Over decadal scales associated with climate change, however, harvests of fixed fractions of biomass will lead to catches that track long term changes in biomass production rates.

The NCC ECOTRAN model complements other California Current ecosystem models in two areas: 1) Assessing the role of small pelagic fishes, including sardine, anchovy, herring, and juvenile salmon, and 2) providing Monte Carlo simulations to address observational uncertainties and natural variability in scenario simulations. This model has proven useful for examining scenarios of alternative (forage fish, krill, jellyfish) food web pathways (Ruzicka *et al.*, 2012).

DATA SOURCES

The NCC ECOTRAN model incorporates benthic and pelagic survey data to infer the network of trophic interactions during the productive upwelling season. Survey data include: 1) Bonneville Power Administration-sponsored pelagic fish and zooplankton surveys of the Oregon and Washington shelf (Brodeur *et al.*, 2005; Morgan *et al.*, 2005; Emmett *et al.*, 2006), and 2) the NWFSC and AFSC west coast groundfish surveys (e.g., Keller *et al.*, 2008). Phytoplankton biomass and production rates (1998 - 2007) were estimated from SeaWIFS satellite Chl a data (A. Thomas, U. Maine) and the Eppley version of the Vertically Generalized Production Model (M. Behrenfeld, Oregon State University; www.science.oregonstate.edu/ocean.productivity/standard.product.php). Fishery data were obtained from the Pacific States Marine Fisheries Commission data servers: PacFIN (pacfin.psmfc.org) and RecFIN (recfin.org). Diet information was obtained from the literature of local NCC studies. Physiological rate definitions were obtained from the global literature.

SCENARIO DESIGN

A series of model scenarios, representing individual years from 1998-2007,was produced to explore inter-annual differences in food web configuration and the short-term impact upon higher trophic levels. This set of years includes both unproductive (e.g. El Niño) years and years in which primary producers, zooplankton, and forage fish were in high abundance. Each scenario estimates changes to energy flow throughout the food web necessary to accommodate, and as consequence of, observed changes among nine manipulated lower- and mid-trophic level groups: phytoplankton, copepods, large carnivorous jellyfishes (*Chrysaora fuscescens*), and the major forage fish groups (sardine, anchovy, herring, and smelt). In each scenario, the total grazing or predation pressure on a given producer group was not changed, rather a scenario was created by changing the biomass and consumption of each manipulated group at the direct expense of any and all competitor groups. Similarly, transfer efficiencies were held constant, implying no change to physiology (assimilation efficiencies, growth efficiencies, and weight-specific production rates) nor to predation vulnerabilities.

Individual scenarios were constructed as deviations from the baseline model representing the mean Northern California Current food web structure inferred from ocean observations made across the 1998 - 2007 upwelling seasons (April - September). Inter-annual biomass anomalies are shown in Table 1. For each scenario year, these factors were applied to the baseline biomasses of nine manipulated groups (Table 1, top). Each scenario also altered the phytoplankton primary production rate, thus scenario results represent both structural changes to the trophic network and changes to the overall productivity of the ecosystem as a whole. Consequences of each scenario are expressed as changes in the production rate (t km^{-2} y^{-1}) of each functional group relative to the inter-annual mean, or ‘base’, model.

Table 1 illustrates the range of biological and oceanographic conditions experienced on the Northern California Current continental shelf during 1998-2007, which result from both local and basin-scale processes. Years in which biomass was low among the manipulated groups include both an El Niño year, 1998, and a year of delayed seasonal transition to upwelling conditions, 2005. The delayed spring-transition to local upwelling conditions had negative impacts on many species (Peterson *et al.*, 2006). In other years, such as 2003, biomasses of some mid-trophic level groups (e.g., forage fish) was high, while other lower trophic-level groups were slightly less abundant than the decadal mean. In 2002, the northern region of the California Current ecosystem supported higher-than-average biomasses of most of the manipulated groups, despite being in a slight positive (unproductive) phase of the basin-scale PDO. Local Northern California Current conditions in 2002 may have been influenced primarily by an influx of cold, nutrient-rich fresh water from the north, with the result of higher-than-average phytoplankton production and biomass (Venrick *et al.*, 2003). Applying anomalies from the decadal mean biomasses of low- and mid- trophic level groups as scenario scaling factors simulates the effects of inter-annual variability in the pelagic community structure on the upper food web and fisheries.

To account for uncertainty in scenarios, I have adapted the principles of the "ECOSENSE" simplified Bayesian Synthesis methodology developed by Aydin *et al*. (2007). The uncertainties associated with each group's biomass, diet, and physiology were defined *a priori* from observation or from a pre-established parameter "pedigree" of poorly known parameters. A series of parameter sets were randomly drawn via Monte Carlo sampling from each parameter's distribution. From among many thousands of potential models, only parameter sets that produced systems within thermodynamic balance were retained. Scenario manipulations applied consistently across this set of potential 'base' models express the range of potential system responses within the limits of the defined parameter uncertainties and parameter-set retention criteria (Ruzicka *et al*., 2013).

RESULTS

For each scenario year, I show the relative change in production of several important pelagic groups (Table 2, Fig. 2) and fisheries (Table 3, Fig. 3) relative to the decadal mean baseline model. The years 1998 (an El Niño year) and 2001 were generally bad years across multiple groups and trophic levels (Table 2). Years 2002, 2003, and 2006 were generally good years for most groups. At the base of the food web, variation in total phytoplankton production and the biomass of the small phytoplankton was comparatively small while variability among jellyfish and forage fishes was large (Table 1).

Patterns of interannual variability: Generally, there is correspondence between years of high phytoplankton biomass and production rates up the food web. This is largely driven by the extreme years of the time-series: the low production El Niño year of 1998 and the high phytoplankton production years of 2002, 2006, and 2007. For instance, in

1998 many fish groups' production declined by 10-50%, and seabird production declined by >50%. In 2002 and 2006, many fish groups increased in production by 20-50%.

Aside from these extreme years, the response of the trophic groups and fisheries depended not on mean abundance of the groups manipulated in the scenarios but on abundance of particular lower trophic level groups, and trophic interactions. For instance, 1999 had below-average abundances for every manipulated group, but a mix of above- and below-average production for other groups. The complexity of observed patterns across years show that competitive trophic interactions, direct and indirect, can have as great an impact on the production of higher trophic level groups as variability in production at the base of the food web. In particular, forced imposition of jellyfish and forage fish variability within the scenarios drives strong responses within the network of trophic relationships. First, forage fishes are both prey to higher trophic levels and competitors for lower trophic level production. When forage fish biomass is high (2000-2003) or low (1998-1999, 2006), salmon, seabirds, and marine mammal predators respond similarly. Groups that compete with forage fishes (squid, planktivorous rockfishes) respond in the opposite manner. Second, jellyfishes, particularly sea nettles (*Chrysaora fuscescens*), can become a major consumer of plankton production off the Oregon and Washington coasts in some years (Ruzicka *et al*., 2007). Model scenarios show juvenile fishes and young salmon are less productive during years of especially high jellyfish abundance (2001, 2007). Fisheries, however, appeared insensitive to jellyfish variability - yet these scenarios do not consider the effect of jellyfish on recruitment of juvenile fishes to the fisheries in subsequent years.

As noted above, the two "across-the-board" poorest years were 1998 and 2001. In 1998, scenario drivers were nearly all in alignment for forcing poor production throughout the whole food web. Both plankton production and forage fish biomasses were anomalously low, thus production through the system was low. In 2001, however, phytoplankton production was low while copepod, jellyfish, and forage fish abundances were high. The example of 2001 shows how variability of mid-trophic level energy pathways have large affects upon the rest of the food web. In these scenarios, abundant forage fish and jellyfish use a higher proportion of plankton production at the direct expense of other planktivores (e.g. production of euphausiids, juvenile fishes, small squid decline by 25-60%). Groups such as rockfishes, flatfishes, hake, and Pacific mackerel that rely more heavily upon these "alternate" planktivores than upon the forage fishes decline by 20-60%. For groups that prey directly upon forage fishes (salmon, seabirds, marine mammals), the 2001 scenario was near base model conditions.

Fishery/gear types responded in the same manner as their target groups. The behavior of the different fishery/gear types to food web variability (i.e., the nine manipulated groups) can be divided into two main types. Pelagic fishery/gear types that target forage fishes (seine, gill nets, non-trawl pelagic net gear) or that target salmon, Pacific mackerel, jack mackerel, and tuna (recreational fishers, seine, gill nets, troll gear, offshore hook & line gear) performed best during the forced high forage fish years (2000-2003), and production increased by as much as four-fold during these years. Gear types that targeted hake and sablefish (mid-water trawls, non-shrimp bottom trawls, fish pots) performed best during years of higher euphausiid production (2006-2007), with roughly 30-50% increases in production during these years. Note that the reported increases in production do not imply an equivalent increase in harvest for that year, but would lead to large increases in biomass and harvest over longer time periods.

DISCUSSION

Predicting sensitivity to future ocean variability: This simple scenario modeling exercise shows the short-term effects of observed community changes within the plankton and forage fish community upon higher trophic levels and upon production of fished species. Primary production and food web structural variability over the past decade suggests that pelagic fishery production generally varied within 50% - 200% about the decadal mean. Variability was higher among fisheries that target forage species. Energy flow (fish production) to the major fishery species within the Northern California Current ecosystem, Pacific hake (largely harvested using mid-water trawl gear), has varied from 40% below to 50% above the decadal mean (Table 3). Hake were not a forced group in these scenarios; these simulations represent the net effect of observed variability in primary production and trophic network structure.

A single year of low biomass production will not lead to immediate sharp declines in fisheries catch, but these declines will occur if such unproductive conditions are persistent under future climate scenarios. We cannot at present predict what future levels of productivity will be under climate change. However, the period from 1998-2007 provides a range of annual production rates that could be used in the future to bracket what may occur under climate change.

In the context of climate change impacts on the California Current, the results here illustrate the impacts of local conditions and the importance of understanding the trophic structure and linkages within the food web. Though considerable effort has gone into modeling climate at the scale of the North Pacific (Overland & Wang 2007), and basin-scale patterns such as the PDO are known to influence productivity in the California Current (Checkley & Barth 2009), the modeling effort here suggests that local patterns can also have profound effects on biomass and fishery productivity. For instance, the timing of upwelling off Oregon and Washington in 2005 and intrusion of cold, northerly water in 2002 may have set the stage for the observed changes in lower trophic levels and the modeled responses of the food web. Downscaled oceanographic models that can include climate change and capture these local processes will be needed to improve climate change forecasts for marine populations (Hollowed *et al*., 2013).

Except during years of extremes in lower-trophic level productivity, an understanding of the variability in the network of trophic connections is essential to predicting responses of higher trophic levels. There may not be a close correlation between plankton production and the production of particular higher trophic level species (at least on an inter-annual scale). Instead, energy flow through the food web is modified by the composition of the mid-trophic level community, leading to varied responses of higher trophic level groups. Additionally, within the Northern California Current, mid-trophic level composition is very much modified by migration of the more abundant species (e.g., hake and sardine) and the very poorly understood factors that control production of jellyfish populations. As climate change is expected to cause both changes to local plankton production, i.e., to local upwelling (Bakun, 1990) and to region-scale migration patterns and population distributions (Ainsworth *et al*., 2011; Pinsky & Fogarty, 2012), we can logically expect even larger variability in fishery production in the future. Model scenarios can be crafted to show fishery sensitivities to defined future changes in local production and shifts in population distributions.

Euphausiids: a fundamental limitation in our understanding of the Northern California Current ecosystem: Euphausiids are dominant prey species for many of the most abundant fishes (Miller *et al*., 2010) and are an influential energy transfer node in the Northern California Current ecosystem. In these scenarios, squid, rockfishes, Pacific mackerel, and Pacific hake covary with euphausiid abundance; 2006 and 2007 being years of especially high abundance, 2000, 2001, and 2004 being years of low abundance. Forage fishes and euphausiids are assumed to be competitors for plankton production and covary inversely. Thus, squid, rockfishes, Pacific mackerel, and Pacific hake also covary inversely with forage fish. Because synoptic time-series observations of euphausiid abundance along the Oregon and Washington coasts are not available, I could not force scenarios with observed euphausiid biomasses. Patterns of euphausiid variability are, and must remain, a construct of the model based on our assumptions of competition for plankton production and euphausiid responses to large-scale oceanographic processes.

From a single-species perspective, several authors have investigated optimal fishery management strategies for coping with variable climate (MacCall, 2002; King & Mcfarlane, 2006; A'mar *et al*., 2009; Haltuch *et al*., 2011). As shown by the NCC ECOTRAN work presented here, climate and interannual variability in productivity drive strong responses throughout the food web and fisheries. If climate change leads to altered trends or variability in both productivity and community composition in the California Current, we are challenged to move beyond single-species approaches to develop management strategies that are robust at the level of the whole food web.

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Table 1. Base model conditions (top row) and individual, annual anomalies used as scaling factors in scenario analyses (individual year / base). Base model conditions are the mean plankton and forage fish biomasses (t km^{-2}) observed over the 1998 - 2010 period (April - September) and the mean phytoplankton production rate (t km^{-2} y^{-1}) over the 1998 - 2007 period (April - September) as estimated from satellite data. Red shading indicates a large (≥20%) reduction relative to base model conditions, green shading indicates a large (≥20%) increase over base model conditions.

	phytoplankton			**copepods**[2]		**jellyfish**	**forage fishes**			
	production rate	**small**[1]	**large**	**small**[3]	**Large**	**sea nettle**	**sardine**	**anchovy**	**herring**	**smelt**
base biomass (t km^{-2})	7853.1 ± 948.7	4.7 ± 0.2	33.6 ± 8.9	8.3 ± 1.4	2.8 ± 1.5	7.9 ± 12.8	15.2 ± 18.0	4.3 ± 6.5	2.6 ± 3.4	1.8 ± 2.6
1998	0.87	0.95	0.75	0.27	2.25	--	0.21	0.01	0.59	0.02
1999	0.86	0.98	0.86	0.74	0.47	0.04	0.34	0.004	0.15	0.002
2000	0.89	0.98	0.89	1.60	1.43	0.01	1.63	0.27	3.93	5.20
2001	0.91	0.98	0.84	1.35	0.42	3.99	2.17	0.58	3.28	3.93
2002	1.16	1.07	1.31	1.44	0.99	1.21	0.85	3.26	1.20	0.37
2003	1.07	0.99	0.93	0.95	0.71	0.30	4.17	0.93	1.69	0.46
2004	0.98	0.96	0.77	1.47	1.20	0.21	0.37	4.85	0.31	0.77
2005	0.99	0.96	0.87	0.33	0.54	0.77	0.54	1.05	0.15	0.15
2006	1.08	1.05	1.25	1.08	0.72	0.29	0.31	0.13	0.20	0.10
2007	1.19	1.08	1.52	1.05	0.67	5.12	0.89	0.13	0.27	0.17

[1] Small phytoplankton are < 10µm

[2] The base model was built on zooplankton data from BPA cruise vertical net data while copepod scenarios were scaled based on time-series observations off the central Oregon coast (NH-Line station NH05, 9km from coast, data from W. Peterson NWFSC).

[3] Small copepods are < 0.025 mg C

Table 2: Interannual scenarios: effect of variability among bottom- and mid-trophic level groups (phytoplankton, copepods, jellyfish, forage fish) on the production of a few select groups. Values are ratios of scenario to base model production rates. Scenarios created by scaling base model biomasses of selected groups by observed biomass anomalies (Table 1). Red shading indicates ≥20% reduction and green shading indicates ≥20% increase over base model. Error terms represent 1 standard deviation of scenarios applied to 445 random, balanced model configurations. (see Fig. 2)

	1998	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**
Planktivores										
E. pacifica	0.88 ± 0.06	0.96 ± 0.03	0.78 ± 0.06	0.76 ± 0.04	1.14 ± 0.06	0.95 ± 0.01	0.71 ± 0.04	1.05 ± 0.06	1.24 ± 0.02	1.41 ± 0.06
juvenile fish	0.78 ± 0.11	1.09 ± 0.08	0.87 ± 0.10	0.42 ± 0.11	1.24 ± 0.08	0.91 ± 0.06	0.92 ± 0.10	0.84 ± 0.10	1.57 ± 0.09	0.74 ± 0.20
small squid	0.98 ± 0.07	1.29 ± 0.10	0.80 ± 0.08	0.41 ± 0.08	1.09 ± 0.07	0.78 ± 0.13	0.81 ± 0.10	1.11 ± 0.07	1.65 ± 0.10	0.95 ± 0.21
juvenile salmon										
coho yearling	0.79 ± 0.06	1.17 ± 0.09	1.03 ± 0.09	0.36 ± 0.08	1.32 ± 0.05	0.86 ± 0.08	1.09 ± 0.07	0.91 ± 0.06	1.62 ± 0.09	0.61 ± 0.14
Chinook yearling	0.81 ± 0.06	1.17 ± 0.08	1.01 ± 0.09	0.38 ± 0.08	1.29 ± 0.05	0.86 ± 0.07	1.05 ± 0.07	0.92 ± 0.06	1.61 ± 0.08	0.66 ± 0.13
Chin. subyearling	0.82 ± 0.05	1.18 ± 0.08	0.99 ± 0.08	0.39 ± 0.07	1.29 ± 0.05	0.84 ± 0.07	1.05 ± 0.06	0.95 ± 0.06	1.60 ± 0.08	0.68 ± 0.12
piscivorous fishes										
coho salmon	0.54 ± 0.08	0.70 ± 0.10	1.35 ± 0.13	0.86 ± 0.12	1.42 ± 0.14	1.76 ± 0.40	1.07 ± 0.20	0.80 ± 0.07	0.93 ± 0.12	0.86 ± 0.10
Chinook salmon	0.52 ± 0.09	0.62 ± 0.10	1.37 ± 0.15	0.91 ± 0.11	1.44 ± 0.16	1.76 ± 0.40	1.06 ± 0.22	0.77 ± 0.07	0.84 ± 0.12	0.92 ± 0.11
Pacific mackerel	0.97 ± 0.08	1.16 ± 0.06	0.83 ± 0.08	0.54 ± 0.05	1.07 ± 0.07	0.76 ± 0.15	0.70 ± 0.09	1.06 ± 0.07	1.52 ± 0.07	1.26 ± 0.11
Pacific hake	0.99 ± 0.08	1.17 ± 0.08	0.83 ± 0.07	0.59 ± 0.05	1.06 ± 0.07	0.79 ± 0.15	0.69 ± 0.08	1.09 ± 0.07	1.50 ± 0.08	1.34 ± 0.07
Rockfishes										
planktivores[4]	1.07 ± 0.07	1.37 ± 0.09	0.69 ± 0.07	0.40 ± 0.05	1.03 ± 0.07	0.63 ± 0.11	0.70 ± 0.09	1.13 ± 0.06	1.77 ± 0.10	1.17 ± 0.13
piscivores[5]	0.78 ± 0.10	0.93 ± 0.10	1.05 ± 0.11	0.72 ± 0.09	1.23 ± 0.10	1.23 ± 0.30	0.86 ± 0.13	0.96 ± 0.07	1.21 ± 0.12	1.10 ± 0.12
benthivores[6]	0.95 ± 0.06	1.24 ± 0.08	0.82 ± 0.08	0.48 ± 0.06	1.13 ± 0.06	0.77 ± 0.09	0.77 ± 0.07	1.03 ± 0.06	1.65 ± 0.10	1.12 ± 0.11
flatfishes										
pelagic feeders[7]	0.69 ± 0.10	0.84 ± 0.11	1.15 ± 0.13	0.78 ± 0.10	1.28 ± 0.12	1.41 ± 0.35	0.90 ± 0.17	0.89 ± 0.08	1.12 ± 0.13	1.08 ± 0.09
benthic feeders[8]	0.80 ± 0.05	0.99 ± 0.06	0.99 ± 0.07	0.64 ± 0.09	1.25 ± 0.04	0.88 ± 0.04	0.83 ± 0.05	0.88 ± 0.04	1.41 ± 0.08	1.17 ± 0.14
seabirds and mammals										
sooty shearwaters	0.40 ± 0.09	0.53 ± 0.13	1.51 ± 0.21	0.96 ± 0.15	1.54 ± 0.24	2.04 ± 0.55	1.20 ± 0.35	0.72 ± 0.08	0.72 ± 0.15	0.73 ± 0.15
common murre	0.45 ± 0.10	0.59 ± 0.13	1.44 ± 0.18	0.93 ± 0.15	1.47 ± 0.22	1.98 ± 0.51	1.11 ± 0.31	0.74 ± 0.08	0.79 ± 0.15	0.79 ± 0.14
small odontocetes	0.60 ± 0.11	0.69 ± 0.14	1.19 ± 0.16	0.98 ± 0.16	1.38 ± 0.15	1.55 ± 0.41	0.93 ± 0.20	0.81 ± 0.08	0.94 ± 0.16	1.22 ± 0.23
large pinnipeds	0.73 ± 0.07	0.88 ± 0.08	1.12 ± 0.10	0.76 ± 0.08	1.27 ± 0.10	1.32 ± 0.24	0.90 ± 0.12	0.91 ± 0.06	1.16 ± 0.09	1.09 ± 0.08

[4] planktivorous rockfishes include: bank, blue, darkblotched, greenstriped, redstripe, rosy, sharpchin, splitnose, shortbelly, widow, Pacific ocean perch

[5] piscivorous rockfishes include: black, bocaccio, canary, chillipepper, yelloweye, yellowtail

[6] benthivorous rockfishes include: cabezon, China, quillback, rosethorn, rougheye, shortraker, shortspine and longspine thornyhead

[7] pelagic-feeding flatfishes: Pacific halibut, arrowtooth flounder, petrale sole

[8] benthic feeding flatfishes: English sole, Dover sole, rex sole

Table 3: Results of inter-annual scenarios: estimated effects on pelagic fishery and major bottom-fish fishery production rates. Values are the ratio of the annual scenario to base model production rates. Scenarios were created by scaling base model biomasses of select bottom- and mid-trophic level groups (phytoplankton, copepods, jellyfish, and forage fish) by the observed biomass anomalies of each year (Table 1). Scenarios assume constant effort across years. Red shading indicates ≥20% reduction and green shading indicates ≥20% increase over base model. Error terms represent 1 standard deviation of scenarios applied to 445 random, thermodynamically balanced model configurations. (see Fig. 3).

	1998	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**
longline	0.93 ± 0.07	1.12 ± 0.07	0.89 ± 0.07	0.61 ± 0.05	1.09 ± 0.07	0.90 ± 0.14	0.72 ± 0.08	1.05 ± 0.06	1.44 ± 0.08	1.28 ± 0.07
troll gear	0.45 ± 0.06	0.56 ± 0.07	1.47 ± 0.17	0.94 ± 0.12	1.52 ± 0.22	1.89 ± 0.45	1.17 ± 0.32	0.74 ± 0.06	0.77 ± 0.08	0.81 ± 0.12
hook-line inshore	0.92 ± 0.07	1.11 ± 0.07	0.88 ± 0.07	0.63 ± 0.05	1.08 ± 0.07	0.99 ± 0.14	0.69 ± 0.07	1.05 ± 0.06	1.43 ± 0.07	1.29± 0.07
hook-line offshore	0.72 ± 0.06	0.88 ± 0.06	1.16 ± 0.10	0.75 ± 0.07	1.26 ± 0.11	1.32 ± 0.23	0.90 ± 0.14	0.91 ± 0.06	1.15 ± 0.07	1.08 ± 0.08
pelagic net gear	0.27 ± 0.04	0.43 ± 0.03	1.39 ± 0.08	1.28 ± 0.10	1.04 ± 0.05	3.56 ± 0.21	0.39 ± 0.06	0.58 ± 0.03	0.55 ± 0.05	1.10 ± 0.07
gill nets	0.56 ± 0.04	0.58 ± 0.06	1.93 ± 0.16	1.14 0.12	1.36 ± 0.07	1.56 ± 0.17	0.80 ± 0.10	0.62 ± 0.05	0.79 ± 0.07	0.72 ± 0.06
seine	0.21± 0.01	0.38 ± 0.02	1.37 ± 0.09	1.34 0.10	1.00 ± 0.04	3.89 ± 0.08	0.34 ± 0.03	0.55 ± 0.02	0.47 ± 0.02	1.12 ± 0.07
recreational[9]	0.61± 0.05	0.75± 0.06	1.26 ± 0.10	0.83 0.08	1.37 ± 0.09	1.58 ± 0.20	1.01 ± 0.13	0.84 ± 0.05	1.00± 0.07	0.95 ± 0.07
mid-water trawls	0.98 ± 0.08	1.17 ± 0.08	0.83 ± 0.07	0.59 ± 0.05	1.06 ± 0.07	0.80 ± 0.15	0.69 ± 0.08	1.09± 0.07	1.49 ± 0.08	1.33 ± 0.07
bottom trawls[10]	0.91 ± 0.07	1.09 ± 0.07	0.91 ± 0.07	0.63 ± 0.05	1.12 ± 0.07	0.94 ± 0.15	0.75 ± 0.08	1.04 ± 0.06	1.41 ± 0.08	1.26 ± 0.07
fish pots	0.93 ± 0.07	1.12 ± 0.07	0.89 ± 0.07	0.61 ± 0.05	1.09 ± 0.07	0.90 ± 0.14	0.72 ± 0.08	1.05 ± 0.06	1.44 ± 0.07	1.28 ± 0.07

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[9] all gear types and platform (boat vs. shore-based) combined

[10] excluding shrimp trawls

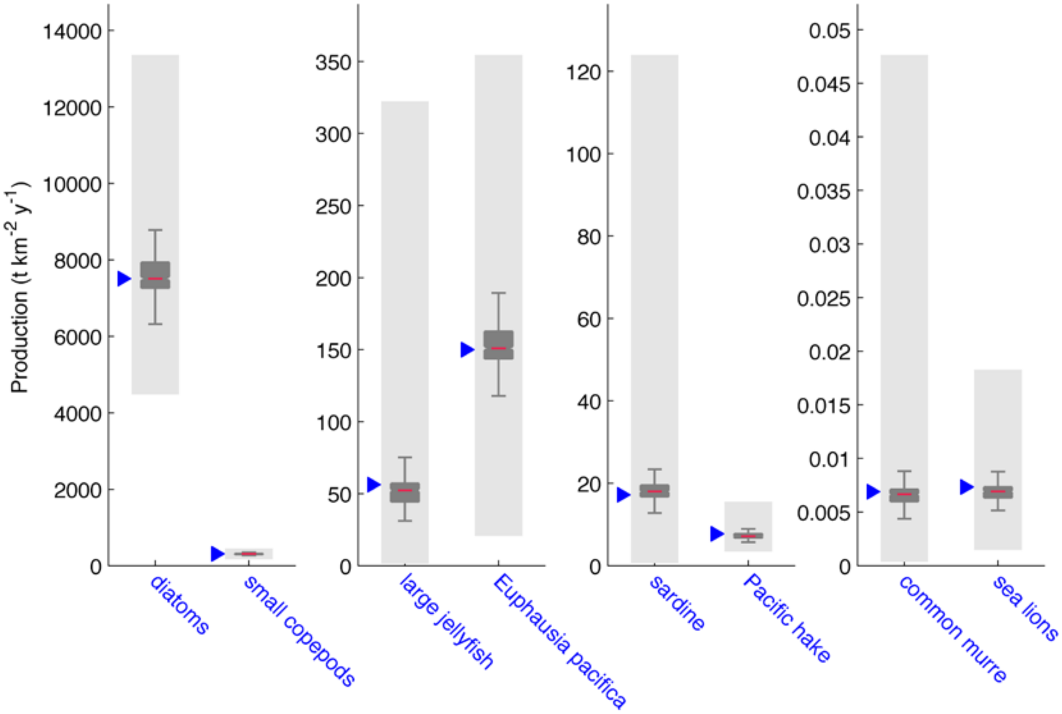

Figure 1. Production rate distributions of Monte Carlo iterations about the decadal mean, baseline model, showing a few example functional groups . Parameter sets were randomly generated by sampling within the 1 CV range of mean parameter values (biomass, production/biomass ratio, growth efficiency, assimilation efficiency, and diet preference). Gray shading shows sampling space of production rates (product of biomass and production/biomass ratio) as defined by observed inter-annual variability (or as defined by assumed parameter 'pedigrees' for poorly observed groups such as the euphausiids). Box plots show distribution of 445 thermodynamically balanced models. Blue arrows indicate the value of the defining 'type' base model.

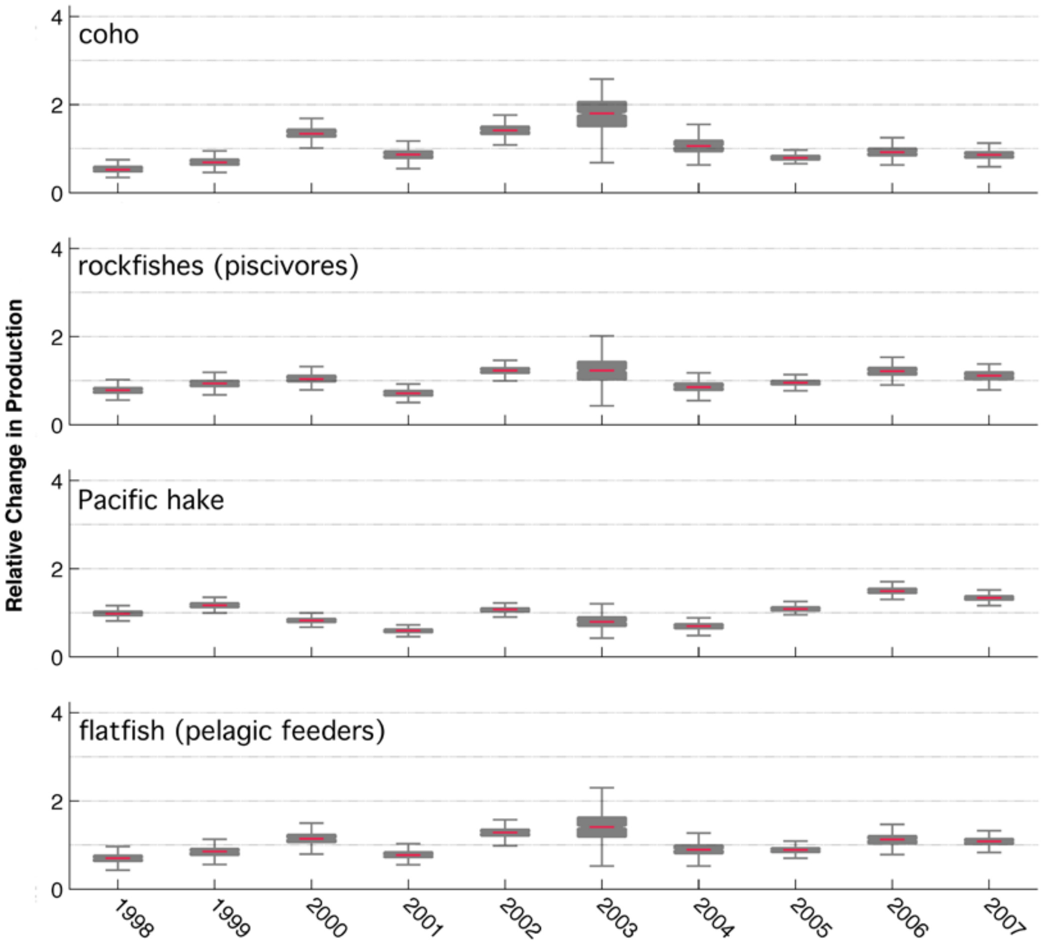

Figure 2. Scenarios showing effects of interannual variability among bottom- and mid-trophic level groups (phytoplankton, copepods, jellyfish, and forage fish) upon the production rates of select fish groups. Boxplots show distributions of changes in production rates relative to the inter-annual mean (ratio of scenario production rate to inter-annual mean, or 'base' model production rate). Boxplots show distributions of scenarios applied to 445 random, thermodynamically balanced model parameter configurations. A value of 1 on the y-axis represents no change from the inter-annual mean. (see Table 2)

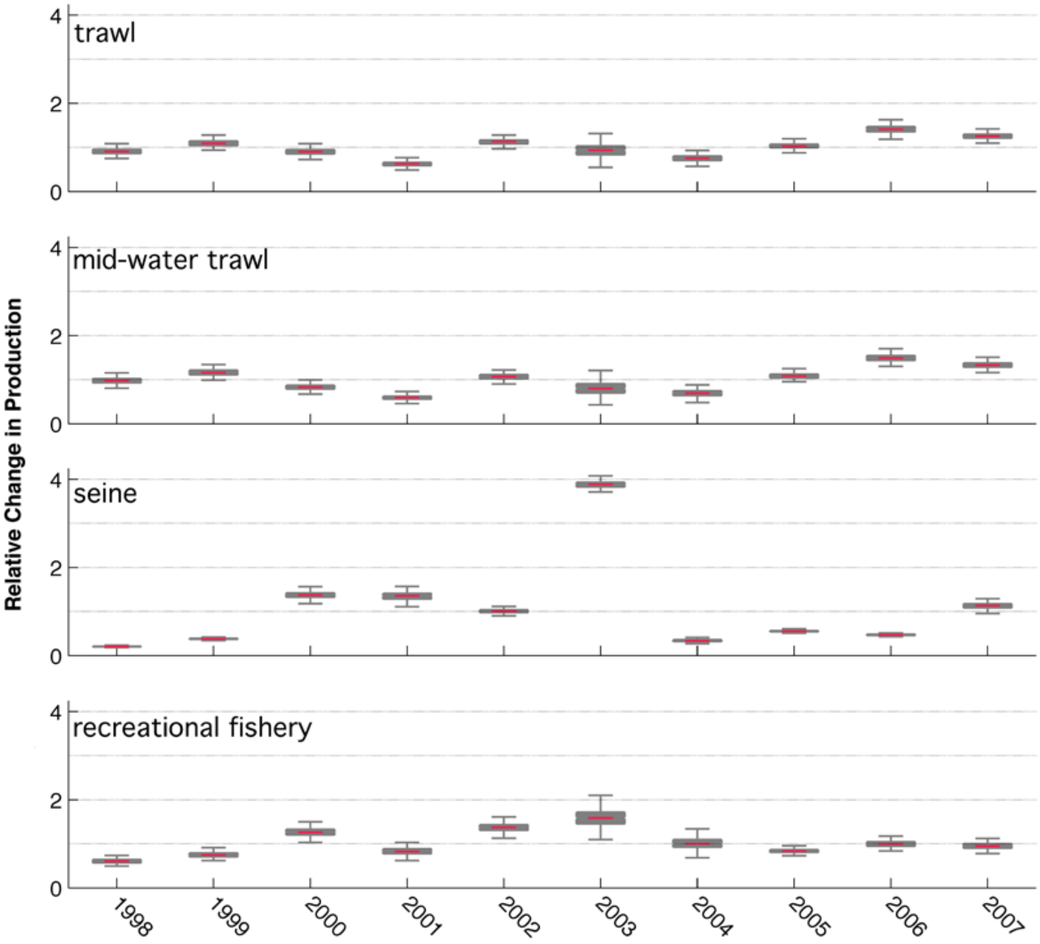

Figure 3. Scenarios showing effects of interannual variability among bottom- and mid-trophic level groups (phytoplankton, copepods, jellyfish, and forage fish) upon the production rates of select pelagic fishery groups. Boxplots show distributions of changes in production rates relative to the inter-annual mean (ratio of scenario production rate to inter-annual mean, or 'base' model production rate). Boxplots show distributions of scenarios applied to 445 random, thermodynamically balanced model parameter configurations. A value of 1 on the y-axis represents no change from the inter-annual mean. (see Table 3)

APPENDIX MS2013-05. ASSESSING THE RISK OF OCEAN ACIDIFICATION IN THE CALIFORNIA CURRENT TO TWO KEY FISHERY SPECIES, DUNGENESS CRAB (*CANCER MAGISTER)* AND PINK SHRIMP (*PANDALUS JORDANI)*

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INTRODUCTION

Marine ecosystems face stressors from multiple anthropogenic sources (Halpern et al. 2008, Halpern et al. 2009) resulting in unknown consequences on species and their interactions (Hughes et al. 2003, Turley and Gattuso 2012). While effects of anthropogenic CO_2 emissions on climate and ocean pH levels are widely acknowledged, predicting which species are most susceptible to these effects remains a challenge. Ocean acidification in particular has been demonstrated to negatively impact a number of marine organisms (Kroeker et al. 2013) but responses to date have been highly variable (Branch et al. 2012, Greene et al. 2012). Given the uncertainty, it is important to gain a better understanding of the risk faced by marine species, especially those that play an important ecological and/or economic role.

Ecological risk assessment is defined as the *assessment of environmental effects of certain stressors and their immediate and long-term damage or harm to an ecosystem* (Chen et al. 2013). Risk assessment is aimed at better identifying which species might be most adversely affected by a stressor by assessing the probability, or risk, of effects (Burgman 1993, Harwood 2000). Within the context of marine systems, risk assessment has been applied to compare the importance of individual stressors and to identify which species face the greatest threat from individual or multiple stressors (Hobday et al. 2011, Samhouri and Levin 2012). Identifying ecological risks is a crucial first step in determining where to focus future research, when considering monitoring programs and when deciding whether to implement precautionary or responsive management policies (Astles et al. 2009).

For the California Current Integrated Ecosystem Assessment (IEA; Levin et al. 2009), risk assessments are a key step to explore the susceptibility of ecosystem attributes to natural and human pressures. This work is aimed at addressing risk of future ocean acidification impacts faced by two key fishery species in the California Current: Dungeness crab, *Cancer magister*, and pink shrimp, *Pandalus jordani*. Dungeness crab and pink shrimp fisheries provided a cumulative $206.2 million USD in 2012, 32.5% of West Coast commercial fishery revenues for the year (PacFIN 2012). The California Current System is highly impacted by multiple anthropogenic stressors (Halpern et al. 2009) and is particularly susceptible to ocean acidification as low levels of carbonate saturation already exist within the near-shore environment (Gruber et al. 2012).

Here we apply an ecological risk analysis to gauge the risk of ocean acidification for each life history stage of these two species. A life-stage specific approach is important, because the sensitivity, exposure, and consequence of ocean acidification likely varies among life history stages (Kurihara 2008). Adult forms are what we harvest for most species, however the earlier life stages, such as eggs and larvae, may be more susceptible and thus a limiting factor when it comes to surviving ocean acidification (Baumann et al. 2011, Frommel et al. 2011, Hurst et al. 2013). It is important to understand the risk faced by early life stages to better understand the potential impacts on fisheries depending on these marine resources.

METHODS

The risk metric used in this analysis is similar to that used by Samhouri and Levin (2012) and for the California Current IEA (Levin & Wells 2013) including components of exposure and sensitivity, each rated on a scale of 1 (low exposure or sensitivity) to 3 (high exposure or sensitivity). The risk metric for each species, R_i, is dependent on exposure, E, and sensitivity, S such that:

$$R_i = \sqrt{(E-1)^2 + (S-1)^2}$$

Exposure to ocean acidification was determined from the overlap of species' distributions with pH predicted for the year 2050 (see methods and references below). Sensitivity was determined from a literature review of papers that have examined the responses of Dungeness crab and pink shrimp, or related species, to acidification.

DATA SOURCES

PH PREDICTIONS

Predicted pH levels for the California Current were obtained from modeling by Gruber et al. (2012) for the year 2050. Gruber et al. (2012) implemented a Regional Ocean Modeling System (Schepetkin and McWilliams 2005) for the California Current, including an ecosystem model that tracks nitrogen and marine inorganic carbon. The model is forced by the A2 high-CO_2 emissions scenario from the IPCC (Nakicenovic et al. 2000). The model assumes present day boundary conditions for all variables except CO_2 and dissolved inorganic carbon; thus it simulates ocean acidification but not potential changes in temperature, currents, or other aspects of climate change.

The Gruber et al. (2012) model predicted anomalies of high pH values off the coast of Washington, where we would expect low-pH water due to upwelling. Consequently, the present analysis may underestimate the extent of exposure to low-pH environments, particularly in waters off Washington.

SPECIES DISTRIBUTIONS

Distributions of the multiple life history stages of both Dungeness crab and pink shrimp were determined from a literature review and discussions with experts on each species. Life history stages mapped for Dungeness crab include: egg, zoeal larvae, megalopal larvae, and adults. For pink shrimp the stages were: egg, larvae, and adults. Maps were produced in ArcGIS (ESRI 2013) of the distribution of each life history stage during every month of the year. Details for species distributions came from Reilly (1983), Pauley et al. (1986), Hobbs and Botsford (1992), Hobbs et al. (1992 and Rasmuson et al. (2013) for Dungeness crab, and Dahlstrom (1970), Dahlstrom (1973), Rothlisberg and Pearcy (1976), Rothlisberg and Miller (1983) and Hannah (2011) for pink shrimp.

Details on the distributions of Dungeness crab life history stages are outlined below. Dungeness crab adults are predominantly found between 30-90 m depth; they are only occasionally found in the surfzone (Rasmuson 2013). Dungeness crabs are found from Alaska to Santa Barbara, California and for this work were mapped from the outer coast of Washington to Santa Barbara California. The eggs of Dungeness crab are retained on female crabs and are found off the coasts of Washington and Oregon between October-March and off California between September-February (Reilly 1983, Pauley et al. 1986, Rasmuson 2013). Dungeness crab larvae have exhibited movement far offshore, starting as zoeae over the continental shelf and progressively moving out past the continental shelf before metamorphosing into megalopae (Reilly 1983, Hobbs and Botsford 1992, Hobbs et al. 1992, Rasmuson 2013). Pelagic crab larvae are found down to 70 m depth. Megalopae move into the nearshore environment to settle (Reilly 1983, Rasmuson 2013). Megalopae were broken into two groups, 'megalops' and 'settled megalops' to account for their presence both in the open ocean and in benthic environments where adult crabs are found.

Pink shrimp adults are similarly found within a limited depth range, with the highest concentrations between 80-230 m depth (Hannah 2011). Like Dungeness crabs, female pink shrimp retain their eggs, and ovigerous females are found off Oregon and Washington in October-March, off Northern California in October-April and Southern California in November-June (Dahlstrom 1970, 1973). The distribution of pink shrimp larvae has been minimally investigated, with most of the research conducted in the 1970s (Rothlisberg and Pearcy 1976, Rothlisberg and Miller 1983). The estimated distribution of pink shrimp larvae therefore depends on a number of assumptions. Larvae are found within 55 km of shore for the first month present in the water column, and then out to 110 km as they disperse via advection and diffusion. Therefore larval shrimp distribution covers a large area from 2 km offshore out to 110 km from shore during most months of the year when they are present. Pink shrimp larvae have been found between the neuston and 150 m depth, with the majority above 100 m, thus the depth of 100 m was used in this analysis.

SENSITIVITY DATA

A literature review was conducted for both species to find papers examining their sensitivity to low pH waters. We also identified relevant studies of species in the same genus. Using Web of Knowledge and Scopus, the following search criteria were used for Dungeness crab: 1. "*Cancer magister*" "ocean acidification", 2. "*Cancer*" "ocean acidification" 3. "Crab" "ocean acidification". For shrimp it was similar: 1. "*Pandalus jordani*" "ocean acidification", 2. "*Pandalus*" "ocean acidification", 3. "Shrimp" "ocean acidification". Papers testing the impacts of pH were retained.

EXPOSURE METRIC

The life history stage of each species was assigned an exposure value from 1-3 based on the fraction of its distribution that will be exposed to 'low-pH' waters in 2050. The following criteria were used: 1 = 0-10% of their distribution will be exposed to low-pH waters, 2 = 10-50% will be exposed to low-pH waters and 3 = 50-100% will be exposed to low-pH waters. The 'low pH' value used was a pH of 7.7, which is at the lower end of what species experience currently (using model-predicted pH for 2013 and comparing it to species distributions). pH is a continuous metric and thus using a single value as a cutoff creates a coarse measure of exposure dependent on binary information. This cutoff value was used to provide a means to determine what is and is not low pH, however future applications of this work will aim to include a value that is not binary.

ArcGIS (ESRI 2013) was used to determine pH exposure for each species and life history stage during every month of the year. Therefore the exposure maps address both spatial and temporal exposure to low-pH waters. The pH maps obtained from Gruber et al. (2012) were in a 5 x 5 km grid, and using GIS the grid was clipped based on the area where the species' life history stage is found (e.g. Figure MS5-1). Assuming the low-pH value of 7.7, we determined the fraction of the distribution that experiences low pH, using R Statistical Software (R Development Core Team 2005).

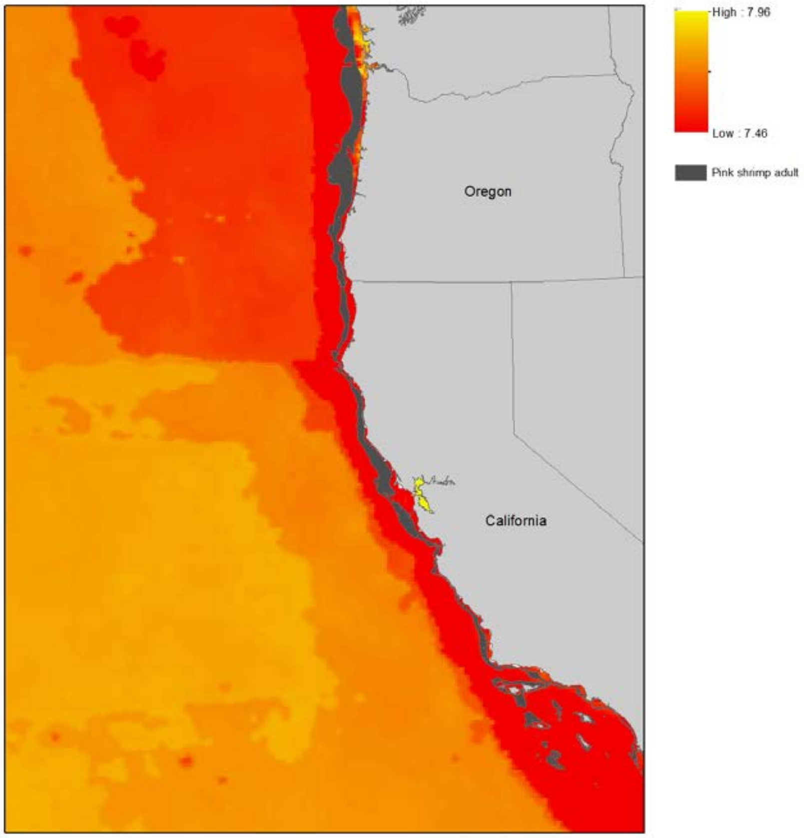

Figure MS5-1. Example exposure map for pink shrimp adults, found between 80-230 m depth (gray shaded areas), during the month of January. pH predictions from Gruber et al. (2012) for the year 2050 along the sea floor.

SENSITIVITY METRIC

Sensitivity values for each species' life history stages were assigned based on relevant literature and pre-existing knowledge about species distributions and tolerance levels. Table MS5-1 provides written categorizations. Using literature published on both species, other species in the same genus or other shrimp or crab species for general information, sensitivity metrics were determined. For those species where the only publications were on a related species, the sensitivity metric was assigned an additional 0.5 value to account for uncertainty.

Table MS5-1. Methods for categorizing species sensitivity levels.

Sensitivity	Category Description
1	High confidence in capacity of life stage to tolerate exposure. Confidence is based on direct experimentation on this or very closely related species, or based on known exposure patterns in sustained populations.
2	Some confidence in capacity of life stage to tolerate exposure, based on evidence suggesting limited, but not full, tolerance to exposure. Empirical evidence shows some effect, but effect size is moderate. Evidence may come from this or from a related species or similar life stage.
3	Little confidence in capacity of life stage to tolerate exposure. This may be based on direct demonstration of demographic effects on this or closely related species. Demographic effects are deemed to be probable based on physiology response or other lines of evidence.

PORT EXPOSURE

In addition to determining the exposure and sensitivity of the different life stages of Dungeness crab and pink shrimp to low pH waters, we also conducted a preliminary exposure assessment for four key fishery port groups on the Pacific Coast. According to PacFIN (2013) four of the port groups landing over $10 million in Dungeness crab in 2012 were: Washington coastal area ports (including Westport Washington), Crescent City area ports, Eureka area ports and San Francisco area ports. Using ArcGIS (ESRI 2013), exposure to low pH waters along the sea floor was determined for the 100-km region around the major city for each port group, for every month of the year.

RESULTS AND DISCUSSION

The life history stage most at risk for the two species examined is larvae for pink shrimp and settled megalops for Dungeness crab (Figure MS5-2, Table MS5-2). Shrimp larvae have the highest risk because they are both highly sensitive and experience high levels of exposure. Within their distribution, 81.3% of the waters they inhabit at 100 m depth are predicted to be at a pH of <7.7 by year 2050. Their sensitivity estimate is derived from experiments on a related species, *Pandalus borealis*, for which it has been found that there is no impact on survival but that development is affected (Bechmann et al. 2011, Arnberg et al. 2012) and therefore this closely related species exhibits some tolerance but not complete tolerance (sensitivity categorization based on Table MS5-1). Although Dungeness crab settled megalops are rated as having some level of tolerance to low pH waters, their categorization of 3 for exposure results in a relatively high final risk score. Settled megalops are found in low-pH waters along the bottom and thus are highly exposed: by year 2050, 59% of waters they inhabit are predicted to be more acidic than a pH of 7.7.

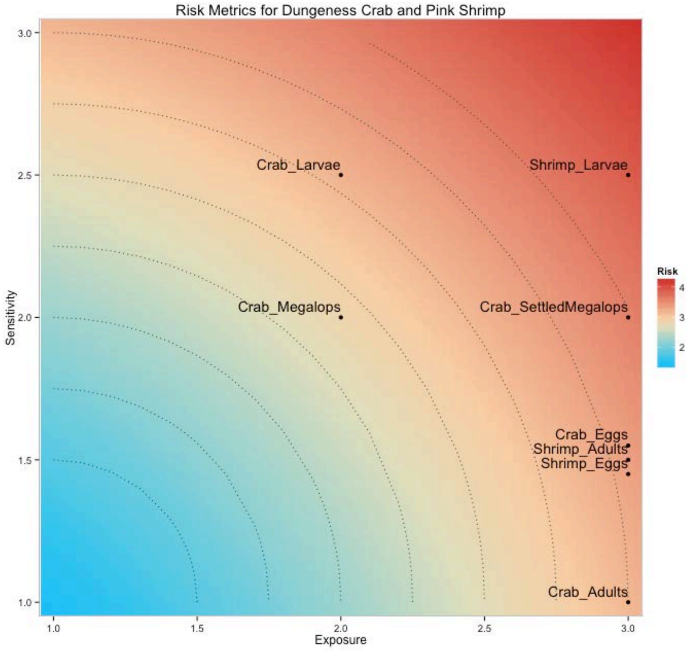

Figure MS5-2. Risk plot demonstrating risk scores for each species and life history stage. Sensitivity values come from the literature and exposure values are specifically related to exposure to pH < 7.7.

Table MS5-2. Summary of Sensitivity and Exposure ratings for each species and life history stage combination, and their final Risk Score. Sensitivity values come from the literature and exposure values are specifically related to exposure to below 7.7 pH.

Species by Life History Stage	**Sensitivity Value**	**Percent Exposure**	**Exposure Value**	**Risk Score**
Crab Eggs	1.5	87.6	3	3.4
Crab Larvae	2[1]	35.5	2	2.8
Crab Megalops	2	29.1	2	2.8
Crab Settled Megalops	2	58.6	3	3.6
Crab Adults	1[2,3]	76.2	3	3.2
Shrimp Eggs	1.5[4]	95.9	3	3.4
Shrimp Larvae	2.5[4,5]	81.3	3	3.9
Shrimp Adults	1.5[6]	89.1	3	3.4

[1] (Jason Miller, unpublished results) [2] (Pane and Barry 2007) [3](Metzger et al. 2007) [4] (Arnberg et al. 2012) [5] (Bechmann et al. 2011) [6] (Hammer 2012). Conclusions reach for sensitivities of life history stages for which there was no publication relied on information known about similar life history stages or known information about the pH in the environment where the species is currently found.

Although pink shrimp larvae and settled Dungeness crab megalops are the most at risk, all life history stages of both species are likely to experience a high degree of exposure to acidic waters (pH <7.7 in year 2050) in >10% of their distributions, falling into exposure levels 2 and 3 (Figure MS5-2, Table MS5-2). Of the eight life history stages of the two species examined, six are predicted to be exposed to pH less than 7.7 in 59-89% of their distribution (exposure level 3). This is largely due to the temporal and spatial distributions of these species. Both Dungeness crab and pink shrimp adults are found along the bottom where pH is the lowest. Since their eggs are attached to the adults, eggs also experience these low-pH conditions. Thus, although some of these life history stages are not categorized as highly sensitive, due to their exposure their final risk metric falls close to 2.

The species and life history stage combinations experiencing the lowest risk are Dungeness crab larvae and pre-settlement megalops. Their low risk score is due to a lower exposure level. Both of these life history stages are found at shallow depths, away from the low-pH waters at depth. They are also found offshore for a large proportion time, allowing them to escape nearshore, low-pH upwelling waters.

From this work we have hypothesized the most at-risk life history stage for each species. Two questions remain. One, which life history stage is the most important for the long term sustainability of each species? Two, is that life history stage able to avoid risk based on its spatial and temporal distribution? Next steps will aim to address these questions using stage-structured population models. Once models have been developed for each species, sensitivity analysis of the populations to their vital rates (e.g. survival, reproduction) will be performed to determine the relative importance of each vital rate for species survival and the relative importance of each life history stage (Morris and Doak 2002).

From the port level comparisons, the ROMS model predicts high exposure to low-pH waters for the three ports in California (Table MS5-3). Although it appears that the

Washington coastal area port has lower risk of exposure, this is likely a result of the high-pH anomalies that were observed in the Gruber et al. (2012) model but are not supported by field observations. Since exposures between 92-100% are observed for California, this analysis demonstrates that low-pH waters may be found nearby at least some major ports with high Dungeness crab landings and revenue.

Table MS5-3. Summary of exposure results for port level comparisons, values represent percent exposure to waters with pH below 7.7 for the 100-km radius around the major city in the port group.

Port Group	**Percent Exposure**
Washington area ports, WA	65.7%
Crescent City area ports , CA	100%
Eureka area ports, CA	100%
San Francisco area ports, CA	92.6%

Results from this analysis are subject to assumptions regarding the oceanographic model used and thus are a preliminary draft that should be interpreted with caution. There are a number of components that need to be refined, including further investigation of the anomalously high pH predicted by Gruber et al. (2012) near the Washington coast, and possible comparison of conclusions when using an alternative oceanographic model. Additionally, the exposure value is dependent on which pH is chosen as the 'low pH' value and thus will vary depending on the value used. Finally, data used to determine shrimp and crab distributions varied for each life history stage included. For both species, adult distributions are known because they are of fishery value and eggs are attached to adults. However, for both species data on larval distributions are limited and therefore assumptions had to be made in order to make definitive maps.

We aim to expand this work to additional species that are likely to be impacted by ocean acidification (Branch et al. 2013, Kroeker et al. 203), to allow better understanding of the relative risk for particular species and fisheries. Direct impacts of low pH can be assessed using the risk score methodology above, as well as the proposed stage-structured population models. Indirect effects propagated through the food web, for instance to predators that rely on shelled organisms, can be forecast by ecosystem models that include trophic relationships and geography. Such models (e.g. (Fulton et al. 2011)) can be used to evaluate impacts on fishery harvests and management (Kaplan et al. 2010), and potential synergisms between ocean acidification and the changing climate (Griffith et al. 2011).

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APPENDIX MS2013-06. SCENARIOS FOR SHIPPING ON THE US WEST COAST

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INTRODUCTION

As an integrated synthesis of the marine ecosystem in the California Current, the IEA focuses on five components: habitat, wild fisheries, ecosystem integrity, vibrant coastal communities, and protected resources (Levin and Schwing 2011, Levin et al. 2013). Drivers and pressures on these components that are within the scope of the IEA include:

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

Notably, shipping is potentially linked to other pressures such as coastal zone development (via dredging and construction of terminals), invasive species (via ballast water), energy development (for instance to transport oil or coal), naval exercises (that alter shipping routes), and oil spills. Protected species such as whales and turtles may be struck by ships, and both the overlap of species' habitat use with shipping routes (Redfern et al. 2013), and shipping speeds (Vanderlaan and Taggart 2007) may determine the frequency of ship strikes. Concern regarding risk of oil spills to sensitive marine species and ecosystems led NOAA, working with the U.S. Coast Guard, to request that the International Maritime Organization (IMO) designate 'an Area to be Avoided' by certain classes of ships off the Olympic Coast National Marine Sanctuary [1]. NOAA also worked with the U.S. Coast Guard and the IMO to create recommended lanes for ships carrying hazardous materials, which increased the distance that ships are to stay offshore when within Monterey Bay National Marine Sanctuary[2]. To reduce the risk of ships striking whales, NOAA provided the U.S. Coast Guard with information which they considered in their proposals to the IMO to modify the traffic separation schemes at the approaches to the Ports of Los Angeles/Long Beach and San Francisco. The traffic lane changes included measures to reduce the risk of ship strikes, such as moving vessel lanes away from known locations of whale aggregations[3]; these changes were adopted by the IMO in 2012 and went into effect June 1, 2013.

Figure 1. Container ships. Photo: NOAA

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[1] http://olympiccoast.noaa.gov/protect/incidentresponse/atba.html)

[2] http://montereybay.noaa.gov/vt/vtexec.html

[3] http://sanctuaries.noaa.gov/protect/shipstrike/policy.html

Arguably, the 2011-2012 IEAs focused more on fishing than shipping, though these ecosystem assessments do include time series of shipping volumes and tonnage beginning in 2003, and dredge volumes from 1997. One clear indication of the intersection between shipping, protected species, fisheries, and energy development was the work of Plummer and Feist (*in* Levin and Wells (2013)), which illustrated potential spatial conflicts between sites for wave energy and other uses, such as tug-and-barge lanes, critical habitat for green sturgeon, and fishing areas for Oregon fleets.

One of the goals of the IEA is to evaluate the performance of management strategies, in terms of outcomes for ecosystem components (e.g. a protected species or fish) that are influenced by the set of drivers and pressures above. To lay the groundwork for future quantitative models that may include shipping, we conducted a series of conversations with eight individuals familiar with the shipping industry. The goal was to understand recent and potential future trends in US West Coast shipping sectors over the next 5-30 years. Below, we outline five of these trends, which are simple scenarios that are relevant to understanding or predicting shipping routes, speeds, or volumes, and may be relevant in predicting effects of shipping on various components of the ecosystem.

INTENDED AUDIENCE

This summary is intended for IEA researchers who wish to consider how their research, and components of the California Current Ecosystem, fit into the context of West Coast shipping. It is targeted for fishery biologists, social scientists, and others who want a simple entrée into some of the issues driving trends in shipping. It aims to illustrate the issues and potential future trends.

METHODS

The methodology was a series of informal conversations via telephone. These conversations do not constitute formal interviews or scoping. Respondents were free to focus on shipping sectors with which they were most familiar. Initial contacts were recommended by staff from the Office of National Marine Sanctuaries (ONMS) familiar with San Francisco ship traffic; most respondents recommended additional contacts or documents. In a few cases we have supplemented responses with citations from published literature.

We present the results of the conversations as very simple scenarios, similar to those developed for other marine sectors in the 2012 California Current IEA (see Levin and

Wells (2013)). 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 plausible future states 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." The preliminary scenarios below aim to provide that context for shipping and components of the California Current ecosystem.

The eight respondents offered insights from a:

- Global shipping company
- Non-profit organization on clean transportation
- US West Coast shipping trade association
- National shipping trade association
- US West Coast tug-and-barge trade association
- Port operating association
- NOAA National Marine Fisheries Service
- NOAA National Ocean Service

A grasp of the different shipping sectors that exist on the US West Coast is useful in understanding the responses. Container ships (**Figure 1**) transport dry freight in modular shipping containers. Common routes are from China or Southeast Asia to major US ports such as LA/Long Beach, San Francisco/Oakland, Seattle/Tacoma, Vancouver and Prince Rupert Canada, and Ensenada Mexico. Tankers and barges carry liquid cargos including crude oil and refined petroleum products, for instance from Alaska to refineries in Puget Sound or San Francisco Bay. Bulk freight is dry freight that is not carried in containers, such as grain, coal or wood products, which are often exports from North America. Tug-and-barge traffic involves transport of a variety of goods both on the open ocean, in Puget Sound and San Francisco Bay, and in the Inside Passage to Alaska. Tugs also are essential for assisting larger vessels in port.

RESULTS

SCENARIO 1: HIGH FUEL PRICES SUSTAIN SUPER SLOW STEAMING, BUT DO NOT SPARK SHORT-SEA SHIPPING

Increases in fuel prices were noted to have already changed shipping patterns over the last 10 years. Super slow steaming, meaning reducing ship speeds from maximum (e.g. 25 knots) to most efficient (e.g. 17 knots) speeds was discussed as common practice, with the exact speed varying by ship and engine type. Container ships, bulk freighters, and tankers were all reported to have adopted this practice. Slower steaming may necessitate using more ships in rotation, but reduces fuel consumption that accounts for 60% of variable costs. Slower speeds were also reported as one way to improve compliance with emissions regulations. Slower ship speeds were mentioned as one response to the economic downturn of 2007-2008, low consumer demand for goods during that period, and excess capacity in the shipping fleet. Though there is some potential that ship speeds might increase with an increase in consumer demand, the consensus was that slower, more efficient speeds will likely remain common practice for the long term for container and liquid cargo vessels. One respondent noted that tugs and-barges already have low speeds (7-10 knots) and high fuel efficiency per ton of cargo. The continuation of slower speeds for large ships could be relevant to predicting risk and severity of mammal-ship strikes (Vanderlaan and Taggart 2007).

Short-sea shipping, which is the transport of goods via ship between US ports, has been suggested as one response to higher fuel costs for land transportation (Perakis and Denisis 2008). However, respondents noted that success with this was limited on the West Coast. One respondent noted that there was limited capacity for US shipyards to provide appropriate vessels, and that short-sea shipping ran counter to economies of scale (see below); another respondent noted that heavy subsidies were required to initiate short-sea shipping efforts in California. Increased handling cost (unloading each container twice, once from a trans-Pacific ship and once from a coastal ship) was cited as a major economic obstacle.

SCENARIO 2: ECONOMIES OF SCALE PUSH FOR FEWER, LARGER SHIPS CONCENTRATING IN THE LARGEST PORTS

Respondents familiar with the container ship and tanker sectors noted large increases in ship size over the last decade, in an effort to maximize economies of scale and reduce cost per unit of cargo. Thus, we may expect diverging trends between indicators of vessel counts and cargo volume, both of which have been reported in the IEA. A respondent noted that one exception is for tankers in Puget Sound, where tanker size is capped at 125,000 tons, necessitating more transits of smaller tankers. Relatively weak economic demand was cited as one reason why mid-sized vessels are being scrapped or sold earlier than is typical, to be replaced by larger ships. Widening of the Panama canal (see below) also allows larger ships in general, but the global trend toward larger vessel size has been ongoing for thirty years (Cullinane & Khanna 2000). Continuing this trend toward fewer, larger vessels would likely favor the use of the largest ports, such as LA/Long Beach (**Figure 2**) (see also Redfern et al 2013). Shipping impacts on the ecosystem would likely be concentrated on these ports. Limitations on federal infrastructure investment, compared to European ports, was mentioned as one limit to handling increased numbers of the largest vessels, and to further expansion of the largest ports at the expense of the smallest.

Predicting impacts of shipping on marine resources might require tracking different indicators of shipping activity. For instance, risk of marine mammal ship-strikes or likelihood of oil spills may depend on ship transits, while the potential scale of oil spills may depend on liquid cargo and fuel volumes.

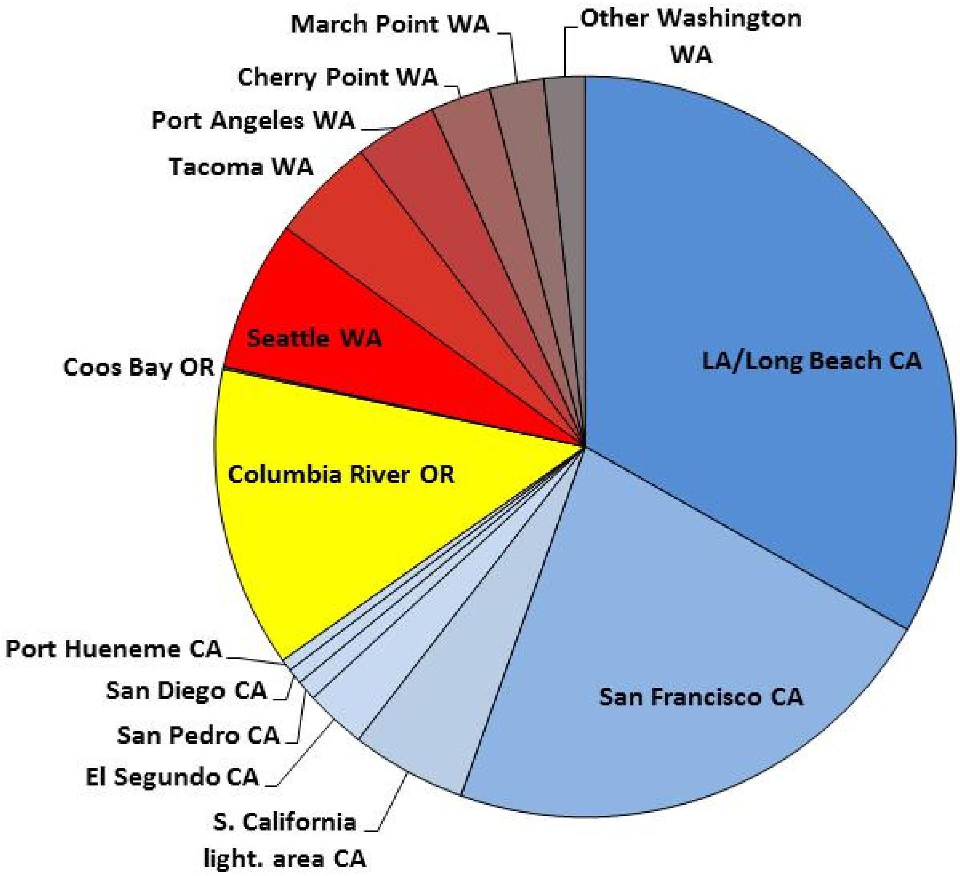

Figure 2. 2011 US West Coast Vessel traffic, proportion of deadweight tonnage x vessel calls, provided by the US Department of Transportation. Includes vessels over 10,000 deadweight tons.

SCENARIO 3: WIDENING OF PANAMA CANAL SHIFTS CONTAINER TRAFFIC FROM WEST COAST TO EAST.

The Panama Canal is being expanded, with a new set of locks and capacity for larger ships slated for 2015. Goods manufactured in Asia, transported in containers, and destined for the US East Coast or Midwest will be able to travel via the canal directly to the East and Gulf Coasts on the largest ships. This may make the canal route more cost efficient than for instance unloading in LA or Seattle, and sending goods east by rail or truck. Respondents mentioned that roughly 70% of freight into Seattle, and 50% into LA/Long Beach, is headed to consumers to the east. The most extreme outcome would be a scenario with a 50% decline in container ship traffic to West Coast ports. Impacts from shipping on the marine environment, or conflicts with other marine sectors, would likely decline near major container ship ports such as LA/Long Beach, San Francisco/Oakland, Tacoma, and Seattle.

However, respondents noted multiple reasons to doubt that the widened canal would cause a broad decline in West Coast shipping. Firstly, they noted that a substantial portion (perhaps half) of goods unloaded on the West Coast are non-discretionary, meaning that they are consumed on the West Coast. Secondly, recent major port infrastructure improvements linking ships to rail and truck service were said to make West Coast ports more cost-competitive. In contrast, East Coast ports must be deepened to allow access to the largest ships when they are fully loaded, and ports such as Jacksonville, Charleston, and Savannah were said to be competing for limited federal funds for dredging. Finally, a shift in manufacturing from China to Southeast Asia and India already favors shipping goods west, through the Suez Canal, and so further declines in east-bound traffic may be unlikely with the 2015 opening of the wider canal.

SCENARIO 4: CLEAN FUEL REQUIREMENTS CONTINUE TO ALTER SHIP ROUTES, ENCOURAGE SLOW-STEAMING

In June 2008, the State of California Air Resources Board (CARB) adopted a regulation that required the use of low sulfur fuel (clean fuel) in large vessels when traveling within 24 nautical miles of the coast. The regulation was designed to reduce particulate matter, oxides of nitrogen and sulfur oxide emissions. The initial CARB rules resulted in shifts in ship travel patterns as many ships moved farther offshore, rather than travel in established traffic separation zones (e.g., Santa Barbara Channel (McKenna et al. 2012, Redfern et al. 2013)), in order to avoid the cleaner fuel requirement within 24 nautical miles. The IMO sets emissions standards for ships internationally through the International Convention on the Prevention of Pollution from Ships, also known as MARPOL. Under MARPOL the IMO established emissions control areas (ECAs) where stringent limits on emissions are imposed and low-sulfur fuel requirements are being phased in over the next several years. The North American ECA includes most of the U.S. and Canadian Exclusive Economic Zones (i.e. within 200 nautical miles of the coast). By 2015, ships operating in the North American ECA must use fuel with sulfur content of 0.1%. The requirements for clean fuel use out to 200 nautical miles could eliminate the advantage of these routes that were slightly offshore of 24 nautical miles. In such a scenario, a change in shipping routes would lead to changes in the overlap with habitat use by particular whale species. For instance, Redfern et al. (2013) found that humpback whales in Southern California occur in nearshore areas, while fin whales occur farther offshore.

Increased fuel efficiency is mandated for all new ships by the IMO under the Energy Efficiency Design Index (EEDI) requirements, and two respondents indicated that this could also encourage the continuation of slower, more efficient steaming speeds.

To the extent that air quality regulations are already in place, there may be limited further impacts on ship routes, but additional Tier III requirements under the IMO could require further reductions in emissions, or installation of potentially costly Selective Catalytic Reduction Systems. Vessels that spend the majority of time within 200 nm, such as cruise lines or coastal freighters serving Alaska, may switch to LNG fuel to meet emissions requirements, while larger ships may alter routes.

SCENARIO 5: NORTH AMERICAN ENERGY DEVELOPMENT INCREASES TANKER AND BULK FREIGHT EXPORTS FROM THE PACIFIC NORTHWEST

The ongoing development of new sources of oil, natural gas, and coal throughout the US and Canada may lead to large increases in tanker and bulk freight cargos from ports in Oregon, Washington, and the Vancouver Canada area. For instance, one respondent familiar with the region pointed out that coal terminals are proposed or under review in Puget Sound, the Vancouver BC area, and on the Columbia River. The proposed Gateway Pacific coal terminal would result in approximately 487 freighters in Puget Sound annually (~15% increase in vessel traffic), while the Kinder-Morgan Pipeline from Alberta could result in an additional 358 tankers through the Strait of Juan de Fuca[4]. Respondents noted potential for export of US shale oil from the Dakotas via rail, to Pacific Northwest ports. They also noted that this export of energy was occurring at a time when US consumer demand for refined gasoline was falling, and production of crude oil on the North Slope of Alaska and tanker traffic between Alaska and the West Coast states, were declining.

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[4] www.seas.gwu.edu/~dorpjr/VTRA/PSP/June13/1%20-%20VTRA%202010%20%20-%20PGHSC%20JUNE-%20WHAT%20IF%20SCENARIO%20SIMULATION.pdf

OTHER FACTORS

Arctic ice melting allows increased ship traffic to Alaska:

Smith and Stephenson (2013) and others note that global climate models suggest that melting ice could allow increased shipping activity in the Arctic, with regular use of the Northwest Passage by mid-century. Most respondents were skeptical of this, or considered the timeframe to be too great for speculation. Moreover, seasonal unpredictability of trans-Arctic routes could make any commercial transit unreliable for modern shipping networks. One respondent identified a major exception to this: destination trips, for instance trips by small fuel tankers or tugs to service oil installations in Alaska. Such vessels may be based in Washington and this may lead to modest increases in West Coast traffic.

Competition between US and Canadian and Mexican ports

Competition between US ports and Canadian and Mexican ports was mentioned by port and shipping company representatives. In particular, competition for importing discretionary goods (those not consumed locally) by Vancouver and Prince Rupert Canada was cited as a major concern for Pacific Northwest ports. As discussed above, export of new energy sources (coal, oil, LNG) could occur from a variety of ports in Washington and British Columbia. Development of freight capacity in Ensenada, Mexico could shift traffic away from LA/Long Beach. Lower port fees and infrastructure were the major advantages for Mexican and Canadian ports. Given the proximity of these Canadian and Mexican ports to US waters, a shift in traffic to these ports might still involve risks from vessels transiting or operating near the US portion of the California Current.

CONCLUSION

The scenarios above may serve as catalysts for future quantitative analysis and predictive modeling through the IEA. Though the five scenarios are preliminary and intentionally broad, they could be refined into quantitative predictions relevant to particular management questions for protected resources, fisheries, habitat, ecosystem integrity, and coastal communities. The details of refined scenarios would depend upon the geographic, temporal, ecological, and social scope of models available for such management questions. For instance, concerns about risk to herring spawning habitat in Puget Sound would require quite a different scenario specification than consideration of marine mammals and shipping lanes in Southern California.

The ideas presented here are not novel, and experts have presented detailed scenarios and quantitative predictions of shipping patterns on the global scale. For instance, Tavasszy et al. (2011) considered global and European perspectives on scenarios involving super slow steaming, increased costs of land transportation, and increased use of Arctic routes. Tavasszy and colleagues predicted that super slow steaming would favor larger container ships and ports that were equipped to handle transhipments (i.e. hubs in distribution networks). Doubling land transportation costs was predicted to shift 8% more traffic to ports and shipping, including short sea shipping. Polar routes were predicted to absorb as much as 1.5% of container ship traffic, but would be particularly relevant to northern ports. A separate global scenario planning exercise for shipping was undertaken by Wartsila, the Finnish manufacturer of large marine engines and other equipment[5]. The authors of that exercise envisioned three complex scenarios: Rough Seas, Yellow River, and Open Oceans. These involve broad narratives about the location of key industries, economic development, required shipping routes and demand by ship type, and national and international responses to challenges such as climate change and piracy. For the California Current IEA, the aim is not to duplicate these efforts, but to understand how these global forces translate into impacts on local ecosystem components and ecosystem-based decision making.

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[5] http://www.shippingscenarios.wartsila.com/

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APPENDIX MS2013-07: ASSESSING THE RISK OF SHIPS STRIKING LARGE WHALES IN MARINE SPATIAL PLANNING

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SUMMARY

Marine spatial planning provides a comprehensive framework for managing multiple uses of the marine environment and has the potential to minimize environmental impacts and reduce conflicts among users. An example of the connections between users of the marine environment and the possibility for conflict recently occurred in Southern California when the California Air Resources Board implemented the ‘Ocean-Going Vessel Fuel Rule’. The fuel rule required large, commercial ships to use cleaner-burning fuels when traveling close to the mainland coast. Before implementation of the rule, a majority of ships traveled through the traffic separation scheme adopted by the International Maritime Organization in the Santa Barbara Channel. Following implementation, a higher proportion of ships began traveling south of the northern Channel Islands to reduce the time spent using more expensive fuels. This shift resulted in increased shipping traffic in military ranges and raised concerns for maritime safety; it also raised concerns about the risk of ships striking large whales.

Spatially explicit assessments of the risks to key marine species from human activities are a requirement of marine spatial planning. We assessed the risk of ships striking humpback (*Megaptera novaeangliae*), blue (*Balaenoptera musculus*), and fin (*B. physalus*) whales in alternative shipping routes derived from patterns of shipping traffic observed before and after implementation of the fuel rule. Specifically, we developed whale-habitat models and assumed ship-strike risk for the alternative shipping routes was proportional to the number of whales predicted by the models to occur within each route. This definition of risk assumes all ships travel within a single route. We also calculated risk assuming ships travel via multiple routes. We estimated the potential for conflict between shipping and other uses (military training and fishing) due to overlap with the routes. We also estimated the overlap between shipping routes and protected areas.

The route with the lowest risk for humpback whales had the highest risk for fin whales, and vice versa. Risk to both species may be ameliorated by creating a new route south of the northern Channel Islands and spreading traffic between this new route and the existing route in the Santa Barbara Channel. Creating a longer route may reduce the overlap between shipping and other uses by concentrating shipping traffic. Blue whales are distributed more evenly across our study area than humpback and fin whales; thus, risk could not be ameliorated by concentrating shipping traffic in any of the routes we considered. Reducing ship-strike risk for blue whales may be necessary because our assessment of the potential number of strikes suggests that they are likely to exceed allowable levels of anthropogenic impacts established under U.S. laws.