# Consequences of spatially variable ocean acidification in the California Current: lower pH drives strongest declines in benthic species in southern regions while greatest economic impacts occur in northern regions

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**Abstract**

Marine ecosystems are experiencing rapid changes driven by anthropogenic stressors which, in turn, are affecting human communities. One such stressor is ocean acidification, a result of increasing carbon emissions. Most research on biological impacts of ocean acidification has focused on the responses of an individual species or life stage. Yet, understanding how changes scale from species to ecosystems, and the services they provide, is critical to managing fisheries and setting research priorities. Here we use an ecosystem model, which is forced by oceanographic projections and also coupled to an economic input-output model, to quantify biological responses to ocean acidification in six coastal regions from Vancouver Island, Canada to Baja California, Mexico and economic responses at 17 ports on the US west coast. This model is intended to explore one possible future of how ocean acidification may influence this coastline. Outputs show that declines in species biomass tend to be larger in the southern region of the model, but the largest economic impacts on revenue, income and employment occur from northern California to northern Washington State. The economic consequences are primarily driven by declines in Dungeness crab from loss of prey. Given the substantive revenue generated by the fishing industry on the west coast, the model suggests that long-term planning for communities, researchers and managers in the northern region of the California Current would benefit from tracking Dungeness crab productivity and potential declines related to pH.

Key words: Atlantis; California Current; Dungeness crab; food web modeling; input-output modeling; ocean acidification.

## 1. Introduction

The oceans are experiencing warming, acidification, eutrophication, and other changes that are modifying marine ecosystems (Halpern *et al.* 2008; Ekstrom *et al.* 2015); these modifications have consequences for human communities that rely on living marine resources. Much of the research focused on the impacts of novel stressors has investigated the responses of individual species (e.g., Cooley *et al.* 2015; Dueri *et al.* 2016; Vanderplanke *et al*. 2015). However, understanding how these changes scale up to impact ecosystems and ecosystem services, such as fisheries catch and revenue, is critical to setting research priorities and making strategic marine resource management decisions. Ecosystem scale research has begun to identify broad geographic regions most at risk from modifications of the environment (Kaplan *et al.* 2010; Ainsworth *et al.* 2011; Cheung *et al.* 2011; Barange *et al.* 2014). However, impacts on human communities often depend on localized ecological change, making it critical to understand changes at a finer geographic scale (Ekstrom *et al.* 2015).

Human communities are place-based and benefit from distinct sets of species. Thus, spatially heterogeneous variation in climate change impacts on those species can have socio-economic consequences that vary across communities. Methods to understand the consequences of marine ecosystem change require an interdisciplinary approach (Bai *et al.* 2016) that includes oceanographic conditions (such as temperature and/or pH) at high resolution, spatially explicit responses of the ecosystem to those conditions and the dependencies of local human communities on marine resources (Allison *et al.* 2009).

Ocean acidification (OA) caused by increasing global carbon emissions is a stressor with both spatial and temporal variability that has the potential to restructure marine systems (Griffith *et al.* 2011; Branch *et al.* 2012; Le Quesne and Pinnegar 2012; Marshall *et al.* 2017). Both calcifying and non-calcifying species have been shown to respond directly to changing pH (Branch *et al.* 2012; Kroeker *et al.* 2013; Busch and McElhany 2016). Ocean carbonate chemistry varies globally (Orr *et al.* 2005; Friedrich *et al.* 2012) and some regions are more at risk from OA than others because of a natural occurrence of low pH water from upwelling that is expected to be exacerbated by the effects of OA (Feely *et al.* 2016). Such regions include eastern boundary currents like the California Current large marine ecosystem (Feely *et al.* 2008; Gruber *et al.* 2012).

The California Current is an upwelling system with high spatial variability in oceanographic conditions (King *et al.* 2011) and seasonally low pH in nearshore environments (Feely *et al.* 2008; Gruber *et al.* 2012). Upwelling occurs in spring and summer, bringing up low pH waters that create a temporal window of exposure to low pH that is expected to worsen with OA (Hauri *et al.* 2013). Latitudinally, the California Current can be divided into three regions (King *et al.* 2011). The northern region, defined by the area north of Cape Mendocino (or Cape Blanco) is defined by strong winter storms and substantial freshwater inputs. The regions, south to Point Conception, California, and then beyond Point Conception, experience increasing levels of upwelling, as indicated by the Bakun Cumulative Upwelling Index. Feely *et al.* (2016) report the lowest surface pH in the central region and a model by Hauri *et al.* (2013) projected this region to have the most variable pH levels (range = 7.85 to 8.15; $\sigma_{pH}$=0.1) and the lowest surface pH most months of the year. Though species in this system may have adapted to these heterogeneous conditions (Pespeni *et al.* 2013), the natural occurrence of low pH waters can be problematic because future declines may push pH beyond species' physiological tolerance thresholds (Fabry *et al.* 2008; Feely *et al.* 2008; Gruber *et al.* 2012). The oceanographic heterogeneity in the California Current will lead to heterogeneous exposure to low pH waters, and thus spatial heterogeneity in ecosystem changes.

The California Current supports a diverse food web and a multi-million-dollar fishing industry. The direct ex-vessel revenue from fisheries in the US portion of the ecosystem was worth over $450 million in 2012 (PacFIN 2013). Fisheries catches and species composition are localized in time and space (Kaplan *et al.* 2013b). For instance, fisheries in Washington State are dominated by Dungeness crab (*Metacarcinus magister,* Cancridae), sardine (*Sardinops sagax*, Clupeidae), Pacific hake (*Merluccius productus*, Merlucciidae), and shrimp (family Pandalidae), while ports off southern California catch market squid (*Doryteuthis opalescens*, Loliginidae), mackerel (family Scombridae), sardine, anchovy (*Engraulis mordax*, Engraulidae*)* and nearshore urchins (*Mesocentrotus* spp., Strongylocentrotidae) (PacFIN 2013). Given the variability of target species and total landings between ports in the California Current (Kaplan *et al.* 2013b) and heterogeneity of oceanographic conditions (King *et al.* 2011; Hauri *et al.* 2013; Feely *et al.* 2016), changes in catch from ocean acidification are likely to be regionally different, based on localized changes in organism responses. Basin-wide impacts have been projected in the California Current (Kaplan *et al.* 2010; Marshall *et al.* 2017), the southeastern Australia marine ecosystem (Griffith *et al.* 2011) and for the UK (Fernandes *et al.* 2016). To our knowledge, no work has estimated how OA may impact fisheries at finer spatial scales, such as the port-level.

Here we explore spatially heterogeneous impacts of ocean acidification on species and fisheries in the California Current. We employ oceanographic predictions from a Regional Oceanographic Modeling System (ROMS) model, forced by an earth system model (GFDL-ESM2M) under climate scenario RCP8.5; an Atlantis end-to-end marine ecosystem model (Fulton *et al.* 2011); and an economic input-output model (Leonard and Watson 2011). We projected the state of the ecosystem in the 2060s and described spatial variation in the marine species and human communities at risk. The value of ecosystem models such as this is in long term strategic management and planning (where path dependency means that actions now may influence the utility of measures into the future) as well as research prioritization; thus our work identifies where future research, monitoring and management focus is needed by exploring the potential consequences of pH changes and vulnerabilities for port revenue, income and employment in US west coast communities.

## 2. Methods

### *2.1 Overview*

We used an ecosystem model to investigate the impacts of ocean acidification in the California Current and consequences for fishing communities along the US west coast. Outputs generated by a fine-scale ocean model provided physical forcing for the Atlantis ecosystem model for the present and 50 years in the future. Outputs from Atlantis were then passed to an economic input-output model developed for the west coast of the US (IO-PAC; Leonard and Watson 2011). A schematic diagram representing the modelling approach can be found in Appendix A (Figure A1). Because the IO-PAC model and high-resolution revenue data were only available for the US portion of the model, we focused our economic analysis on the US west coast (i.e., excluding Canada and Mexico). We focused on the 17 US port groups on the outer coast (excluding Puget Sound) used by the Pacific Fisheries Information Network (PacFIN) to aggregate data and avoid issues of disclosure of confidential information (Table 8-1 of Appendix A in PFMC 2004). For all 17 port groups we simulated changes in catch, revenue, income and employment.

*2.2 Summary of Atlantis model*

We used the spatially explicit end-to-end ecosystem model Atlantis (Fulton *et al.* 2011) to simulate food web dynamics and fisheries. Atlantis represents ecosystem dynamics with three components: (i) an oceanographic sub-model (ii) an ecological sub-model and (iii) a human dynamics sub-model (Fulton *et al.* 2004a; Fulton *et al.* 2004b). Extensive documentation of the Atlantis modeling framework, for both the California Current and other ecosystems, can be found in previous publications (Fulton *et al.* 2004a; Fulton *et al.* 2004b; Kaplan *et al.* 2010; Kaplan *et al.* 2013a). Therefore, we briefly summarize the most recent version of the Atlantis model developed for the California Current (Marshall *et al.* 2017), including all three sub-models, highlighting changes we made explicitly to investigate spatial impacts of OA.

The physical domain of the Atlantis model is represented by polygons, which are defined by depth ranges and by longitudinal and latitudinal breaks. Our model includes 88 spatial polygons that span the entire domain of the California Current, from the northern extent of Vancouver Island, Canada and south to Punta Eugenia, Baja California, Mexico. Longitudinal polygon boundaries were based on bathymetry with breaks at 50m, 100m, 200m, 550m and 1200m and then finally at the 200 nautical mile Exclusive Economic Zone (EEZ; Figure 1). These breaks were chosen based on a mixture of bathymetric and biological information (representing: nearshore shelf, deeper shelf, shelf/slope break; see Appendix S1 in Marshall *et al.* 2017). Depth ranges of boxes within the model match the depths used to define bathymetric breaks.

*2.3 Oceanographic model*

The Atlantis model was forced with oceanographic outputs from an implementation of ROMS version 3.7 (Haidvogel *et al.* 2008; Moore *et al.* 2011) with biogeochemistry (Fennel *et al.*, 2006, 2008), configured to predict oceanographic conditions (currents, pH, oxygen, temperature, salinity and nutrients) for the present decade (2011-2020) and for a decade 50 years in the future (2061-2070). From these two runs, we used a single year to represent current conditions (2013) and a single year to represent future conditions (2063), see further explanation below. The two decade-long runs were used as opposed to creating a full time series of downscaled oceanographic conditions between 2011-2070 because creating a 60 year time series of ROMS was too computationally intensive.

Our implementation of ROMS has 10x10 km horizontal grid resolution, includes tidal dynamics, and uses global model outputs from the NOAA Geophysical Fluid Dynamics Laboratory (GFDL) ESM2M model with biogeochemistry (TOPAZ) (Dunne *et al.* 2012; Dunne *et al.* 2013) to derive surface forcing, initial conditions, and boundary conditions for the two simulated time periods. The climate scenario used was RCP8.5 (Moss et al. 2010; van Vuuren et al. 2011), which assumes continuation of current emissions trajectories, in our case through the 2060s. We force Atlantis with this RCP 8.5 scenario for two reasons. First, in comparison with our 2013 baseline scenario, this bounds the potential changes we expect on the biogeochemistry of the California Current by the 2060s. Second, global $CO_2$ emissions continue to rise (with no clear evidence of a rapid reversal being likely) locking in future change over the next few decades and approximating RCP 8.5 more closely than more moderate scenarios (USGCRP 2017). The two time periods were run independently in ROMS. In each case the first year (January 1$^{st}$ 2011 and 2061, respectively) was initialized through interpolation of the global ocean simulation results (the ESM2M/TOPAZ simulation, scenario RCP8.5). This higher resolution ROMS model of oceanography for the California Current was developed because upwelling in this region is not well resolved with global scale models (Orr *et al.* 2005) and upwelling plays an important role in the development and spatial extent of OA. This ROMS model does not represent nearshore environments (inshore of 50 meters) with high resolution, and therefore does not represent land-based run-off as a contributor to OA. Previous models for the southern California Current have found expected pH levels of around 7.8 in nearshore surface waters by 2050, with heterogeneity in space (by latitude and depth) and seasonally (Gruber *et al.* 2012). Thus, pH levels in this model are expected to be in a similar range. Further details on the ROMS implementation can be found in Marshall *et al.* (2017) and the associated appendices.

Oceanographic conditions were extracted from the decade-long ROMS runs and used to force Atlantis now and in the future. In each case a single year was taken from each ROMS run (2013 in the first instance and 2063 in the second because they were representative of the time period) and then looped 100 times to achieve quasi-stable forcing for the ecological model so as to achieve quasi-equilibrium ecological conditions in Atlantis. This looping of a single year, as opposed to implementing the full 10-year set of oceanographic conditions, helped control for inter-annual variability in ocean conditions.

*2.4 Ecological model*

The Atlantis modeling framework is a C++ code base that simulates multi-species ecosystem dynamics in a spatial framework, using a simple forward-difference integration scheme typically on 12-hour time-steps (Fulton *et al.* 2004a; Fulton *et al.* 2004c; Horne *et al.* 2010). Atlantis is 3-dimensional and spatially explicit, with vertebrate numbers-at-age and weights-at-age tracked per polygon and invertebrates modeled as biomass pools per polygon. The model simulates processes including primary production, growth and reproduction, trophic dynamics, movement/migration, and habitat interactions within each polygon and depth layer. These processes can be directly impacted by environmental conditions such as pH, temperature, oxygen and water flux from ROMS or other oceanographic forcing. Predator-prey dynamics allow diet switching, predator starvation, and declines in predator weight-at-age if prey decline in abundance.

The most recent California Current Atlantis model includes 82 functional groups representing detritus (2 groups), primary producers (6 groups), benthic and pelagic invertebrates (25 groups), fishes (36 groups), marine mammals (10 groups) and seabirds (3 groups). This model is an update of the Marshall *et al.* (2017) model, which was developed to better represent groups that are known to respond to low pH, including: pteropods, Dungeness crab, coral, coccolithophores and market squid. All biological parameters used in the model reported in this manuscript, such as recruitment, mortality, diets and distributions, were the same as those used in Marshall *et al.* (2017), save for a few changes discussed below.

Model calibration for the Marshall *et al.* (2017) version of the model was achieved in two phases: initial simulations with no fishing, and then secondary calibration with constant fishing pressure. We followed best practices for calibration of this type of end-to-end model, as described previously (Kaplan & Marshall 2016). We evaluated Atlantis projections of seasonal and inter-annual biomass trends, spatial distribution, vertebrate numbers-at-age and weights-at-age, in a Pattern Oriented Modeling approach (Grimm et al. 2005). Iterative, manual calibration is the best possible current approach for these complex models, for which simulations require hours to days of computing time. Tests of fishing pressure involved comparing species productivity, as measured by fishing rates sustainable in the model, to expected 'rules of thumb' based on natural mortality rates (Kenneth et al. 2001; Walters & Martell 2002).

In the course of calibration, we made slight modifications to the model parameterization used by Marshall *et al.* (2017). With our focus on the spatial effects of OA on fisheries catch, we modified parameter values for five of the 11 groups that declined to low biomass levels in the previous model under a baseline simulation: (i) we made slight modifications to feeding efficiency parameters for Dungeness crab, nearshore sea urchins and an epibenthic predatory invertebrate group comprised of sea stars and large predatory snails; and (ii) both feeding efficiencies and quadratic mortality rates were modified for market squid and pandalid shrimp. Dungeness crab, pandalid shrimp and market squid are key invertebrate fishery species. Market squid in particular is a highly important fishery species in Southern California (making up over 50% of the biomass of catch for 5 ports in 2013).

With these modifications we were able to ensure persistence of both market squid and pandalid shrimp and obtain more realistic biomass levels of Dungeness crab along the entire coast, however thirteen groups still declined to low biomass levels (<2% starting biomass). Groups declining which also declined in Marshall *et al.* (2017) included: deep demersal fish, Pacific ocean perch (*Sebastes alutus*, Sebastidae), Arrowtooth flounder (*Atheresthes stomias*, Pleuronectinae), large demersal predators, large pelagic predators, Chinook salmon (*Oncorhynchus tshawytscha*, Salmonidae), pelagic sharks, epibenthic predatory invertebrates, and large phytoplankton. Groups which declined in our model and not in Marshall *et al.* (2017) included: deep small rockfish, small demersal sharks, nearshore sea urchins, and coccolithophores. Though the groups declining made up 17% of the functional groups in the model, they represented only 2% of vertebrate biomass and <1% of consumer biomass, and accounted for 7.5% of 2013 US west coast catch. Additionally, crangon shrimp were found to have unrealistic increases in biomass and were removed from economic analyses (they accounted for 0.1% of 2013 catch). Similar challenges with persistence in complex ecosystem models have been reported by others (Gaichas *et al.* 2012; Thorpe *et al.* 2015) and remain a substantial challenge in large multi-species models. With our primary focus on effects on fisheries catch, the declining groups were not expected to greatly impact outputs, though future modeling efforts can work to increase persistence across groups.

*2.5 Fisheries parameterization*

Fishing mortality for each functional group was set at a constant rate (e.g., 0.05 $yr^{-1}$) such that the catch (metric tonnes) in the first year of the model would match the 2013 catch data from PacFIN. Our US catches were allocated spatially to 17 port groups (hereafter called 'ports') along the coast, with one additional fishery each for Canada and Mexico (Figure 1). Spatially explicit fishing mortality rates per port (or country) and functional group were then applied as constant fishing rates (see Appendix A). This assumes no management responses or shifts in regimes and/or fishery behavior in the future; while this is unrealistic, it allows for consideration of OA effects without the confounding socioeconomic dynamics which would make attribution incredibly difficult. US vessels were assumed to have access to coastal (nearshore) areas within 200km of the homeport (the main city in the port), and equal access to areas offshore to 1200m depth. Canadian and Mexican fishing mortality rates were applied equally to all Canadian and Mexican areas in the model domain, respectively.

Though our Atlantis model projects port-level catches, each of these ports has catch from a mix of vessel types (Table 1). The categories used to classify vessels were originally defined in

Leonard and Watson (2011). The vessel categories have distinct target species, economic characteristics, and subsequent vulnerabilities to ocean acidification. The port-level catches were allocated to the vessel types based on the proportion of catches per vessel type reported in PacFIN in 2013. We report results for 10 vessel types that primarily harvest groups well represented by Atlantis biomass dynamics; nine other vessels types depend substantially on species that are poorly represented in the base Atlantis simulations (i.e., functional groups that do not persist) and these were aggregated into a 'generic' coast-wide fishery.

The catch (metric tonnes) landed at each port was converted to total revenue, using 2013 price per pound data (PacFIN 2013). Prices were found for each functional group at the resolution of vessel type and port. These prices were multiplied by biomass to find total revenue by: (1) vessel type at each port and (2) functional group at each port.

*2.6 Input–Output models, IMPLAN, and IO-PAC*

We used an Input-Output (IO) model (Leontief 1951) to investigate changes in income and employment resulting from changes in ex-vessel fisheries revenue. Here, and in previous work (Kaplan and Leonard 2012), we use IO models to estimate direct, indirect, and induced effects. Direct effect refers to the change in production, such as impacts at the level of the vessel. Indirect effect refers to secondary activity caused by changing input needs of directly affected industries (e.g., changes in fleet revenue may cause a decline in output from shipyards that service those vessels). Induced effects are caused by changes in household spending from additional income generated by direct and indirect effects.

The Northwest Fisheries Science Center's input-output model for Pacific Coast fisheries (IO-PAC) was designed to estimate the gross changes in economic contributions and economic impacts resulting from policy, environmental, or other changes that affect fishery harvest (Leonard and Watson 2011). The IO-PAC was constructed by customizing Impact Analysis for Planning (IMPLAN) regional input-output (IO) software (IMPLAN Pro, 2012, MIG Inc. Hudson Wisconsin).

Development of IO-PAC included customizing IMPLAN with an addition of 19 commercial fishing vessel types. The present application used a version of IO-PAC that was developed to include the 17 ports in the Atlantis model and the subset (10) of the 19 fishing vessel types for which spatial catches were tracked (Table 1). Economic impact estimates in IO-PAC include the effects of changes in fish harvest on income and employment by harvesting vessels and processors, at the scale of port and vessel type.

There are three major assumptions of IO-PAC: (1) Supply of outputs is not constraining. An increase in demand, such as demand by the fishing sectors for engine maintenance, is always met by an increase in supply for the commodity or service demanded. (2) Prices of commodities, such as processed fish, and factors of production, such as diesel fuel, are fixed, and here are denominated in 2013 dollars. (3) There is no substitution in either production or consumption, which means that a fishery sector will always require the same set of inputs (diesel, ice, etc.) to land a dollar's worth of fish. Similarly, households always purchase the same set of commodities in the same proportions.

*2.7 Functional group responses to pH*

For this analysis, 10 functional groups were assumed to respond directly to pH. Groups were chosen based on a review of 393 papers on sensitivity of species in temperate oceans to carbon chemistry and a tailoring of species sensitivity estimates to the California Current (Busch and McElhany 2016) (Table 2). These were the same 10 groups affected by pH in the *Cumulative* scenario in Marshall *et al.* (2017) and had the strongest negative responses to ocean acidification according to the review by Busch and McElhany (2016). Functional group sensitivity was translated into species response through parameterizing pH-induced mortality (additional to any linear or quadratic mortality). All response curves were parameterized such that there was no pH-induced mortality above pH 8.0, and mortality linearly increased with declines in pH. Declines in pH would be felt by a species based on their spatial distribution (polygons in which they inhabit) and the pH in that region. The threshold of 8.0 was chosen because average pH in the region in present day conditions start just under 8.0 (at 7.95). The rate of decline (slope of the curve) was set such that the most sensitive group experienced an additional 10% total annual mortality rate for one unit change in pH (from 8.0 to 7.0; as depicted in Marshall *et al.* (2017) Figure S3). All other functional groups' pH response slopes were scaled against that group, using scaling parameters provided by Busch and McElhany (2016). Marshall *et al.* (2017) tested Atlantis model output sensitivity to mortality rates up to 10 times higher and found the directions of impact (either in the positive or negative direction) of species responses were largely insensitive to higher mortality rates.

Two sets of oceanographic outputs described above were used to run three 100-year Atlantis scenarios, isolating the effects of pH and removing the impacts that may have been caused by changes in temperature. The three Atlantis scenarios included: two baselines using 2013 and 2063 oceanography (referred to as *2013Baseline* and *2063Baseline*), where no species responded to pH, and a third run with 2063 oceanography where species responded to pH (referred to as *2063pHmortality*). We calculated the effect size for each functional group as the difference in biomass for the scenario with 2063 conditions with and without pH mortality turned on (*2063pHmortality* and *2063Baseline*, respectively), standardized by the biomass predicted in the 2013 runs (*2013Baseline*); using the following metric:

$$E_i = \frac{B_{2063pHmortality,i} - B_{2063Baseline,i}}{B_{2013Baseline,i}} \qquad \text{[eq.1]}$$

Biomass values used were an average over the final 10 years of the 100-year model run, representing quasi-equilibrium values under the oceanographic conditions used. This approach was used to remove the influence of other oceanographic changes (such as temperature) over the 50 years, so that results are an indication of the sole influence of pH. Although biological responses to changes in temperature and oxygen levels are expected to be substantial (Shaffer *et al.* 2009; Barange *et al.* 2014; Fernandes *et al.* 2016), they were not the focus of our analysis.

Revenue changes were calculated at the finest resolution of functional group by vessel type and port, multiplying biomass by price:

$$R_{i,j,k,r} = P_{i,j,k} B_{i,j,k,r} \qquad \text{[eq.2]}$$

where $R_{i,j,k,r}$ is functional group revenue calculated directly as the total biomass landed at the port, $B_{i,j,k,r}$, multiplied by the price-per-unit biomass, $P_{i,j,k}$ , for group *i,* at port *j* for vessel type *k* and for model run *r* (where the run can be *2013Baseline, 2063Baseline* or *2063pHmortality*). We investigated revenue outputs at three scales, 1) total port revenue (*RP*), 2) revenue by port and vessel type (*RF*), and 3) revenue by port and functional group (*RS*). The effect size on revenue was calculated in the same way as for biomass, the difference between revenue under 2063 conditions with and without pH mortality, standardized by 2013 revenue.

We report changes in biomass and revenue only for effect sizes greater than $|0.2|$. This is the convention with large ecosystem models, which are best used to reveal large vulnerabilities rather than fine-grained projections (Fulton *et al.* 2011; Fulton *et al.* 2014; Collie *et al.* 2016) and because a difference of less than 20% would be lost in the observation noise of most ecosystem processes.

Further economic analysis was performed using multipliers from the IO-PAC model, to consider direct, indirect and induced effects on income and employment. Multipliers for fisheries revenue and processor effect at each port were used. For example, income effects ($I$) for a single port, $j$, were the product of revenue outputs by vessel type, $k$, and port, and the two multipliers:

$$I_{j,r} = \sum_{j=1}^{j=17} RF_{j,k,r} \ (M_{I,D,j} + M_{I,P,j}) \qquad \text{[eq.3]}$$

where $M_{I,D,j}$ is the economic multiplier for revenue for income at port $j$ and $M_{I,P,j}$ is the multiplier for the processor effect on income at port $j$. The employment produced, $E_{j,r}$, was calculated the same way, using $M_{E,D,j}$ and $M_{E,P,j}$. The effect size on income and employment was calculated as the difference between scenarios in 2063 and standardizing by *2013Baseline*. Three ports do not have processors (Brookings, Crescent City and Bodega Bay), thus those ports only had $M_{I,D,j}$ and $M_{E,D,j}$ .

## 3. Results

### *3. 1 Changes in Biomass*

Demersal functional groups were most affected by changes in pH. Sixteen functional groups experienced biomass effect sizes of |0.20| or greater. Affected groups included invertebrates and demersal fishes (Figure 2). We categorize functional groups according to three classes of response: functional groups with substantial changes (magnitude > 0.2) that varied regionally, groups with substantial but spatially consistent changes, and groups with no substantial change (< |0.2|). We focus on functional groups with regional variability, and then briefly touch on the last two classes.

Most invertebrate responses varied by region, but the patterns of responses were unique to each functional group (excluding pandalid shrimp; Figure 2). For species directly responding to pH, the variation was largely driven by local pH values. On average, pH declined the most between Northern California and Vancouver Island, although species exposure to pH was dependent on the pH projected in the ROMS model within the species' latitudinal range and depth distribution. While Dungeness crab experienced the lowest pH in Canadian waters, other species (e.g., benthic herbivorous grazers and deposit feeders) experienced the lowest pH in southern California (Figure 2). As a result, species responses to lower pH values do not consistently match what might be expected from observations (Feely *et al.* 2016) that the lowest pH and most severe effects would occur between northern California and Oregon. Rather, proportional effects on some functional groups were more substantial in the southern portion of the model, with the size of response scaling with the slope of their pH mortality curve (Table 2). Notably, the crab group (excluding Dungeness) had heterogeneous responses across space, likely due to low biomass (near 7% of starting biomass) creating higher sensitivity to small differences between 2013 and 2063. Declines in benthic herbivorous grazers, bivalves, deposit feeders and benthic carnivores were not only from direct effects of changing pH, but also from indirect food web effects via changes in predator-prey dynamics (Appendix B, Table B1).

Other invertebrates indirectly responded to declines in pH in heterogeneous patterns, resulting primarily from changes in predator and prey biomass levels. Groups with indirect declines — meiobenthos, gelatinous zooplankton, microphytobenthos and black corals — all responded in limited geographical regions (southern portion of the model), with meiobenthos and gelatinous zooplankton experiencing changes in predation. Biomass increases for pandalid shrimp and microzooplankton, neither of which responded directly to pH, resulted from a release from predation.

Dungeness crab were a unique case. They were parameterized to respond directly to declines in pH, but the realized impact resulted almost entirely from indirect effects of changes in prey availability (Appendix B, Table B1). This impact was most substantial off of Oregon and northern California (Figure 2), but also led to declines in Dungeness crab biomass off Canada and Washington State. Ninety percent of the Dungeness crab biomass in the model is in these northern regions, and we did not see the same substantial declines of Dungeness crab in southern regions.

Five functional groups exhibited consistent changes across the model domain. There were four declining fish groups: Petrale sole (*Eopsetta jordani*, Pleuronectinae), Dover sole (*Microstomus pacificus*, Pleuronectinae), cowcod (*Sebastes levis*, Sebastidae), and deep large rockfish, whereas pandalid shrimps increased (Figure 2). The four fish groups declined due to decreases in prey abundance that were substantial enough to create a consistent effect across all six regions. All four fish groups consumed benthic invertebrates that declined: benthic herbivorous grazers, bivalves and crangon shrimp (not shown in Figure 2, but had 19% decline). Pandalid shrimp are preyed upon by both Dover sole and deep large rockfish in the model such that biomass declines in these two fish groups may have caused release from predation on pandalid shrimp, making increases constant across space.

Groups that did not respond > $|0.2|$ included four of the functional groups parameterized to respond directly to declines in pH (mesozooplankton, pteropods, shallow benthic filter feeders and crangon shrimp), as well as pelagic fishes, mammals, sharks and seabirds. It is likely that for groups parameterized to respond to pH that did not change substantially, a combination of high productivity and release from predation were able to counteract direct effects from induced mortality.

*3.2 Spatial Economic Impacts*

Simulated revenue from catch in our base scenario (*2013Baseline*) mostly matched recent revenue composition for 2013 from the PacFIN records (15 of 17 ports) with exceptions driven by biomass trends in the simulations (Appendix C, Table C1). Port revenue from sablefish (*Anoplopoma fimbria*, Anoplopomitidae) and Pacific hake was consistently lower in our Atlantis simulations than in 2013 landings data. Atlantis projections for North Coast, WA, Fort Bragg, CA and San Diego, CA did not accurately represent 2013 landings, as all three ports have substantial landings of species that declined in abundance in baseline Atlantis simulations. Simulated revenue per port (ratio of simulated revenue to revenue from PacFIN) ranged from 0.4 – 1.48 (mean 1.07, SD 0.39, Appendix C, Table C2). Simulated revenue was lower than baseline records at ports between Crescent City and San Francisco, CA due to the model projecting lower Dungeness crab abundance in those regions. In comparison, simulated revenue was higher than 2013 records in the southern California ports highly dependent on market squid, which were projected to increase in biomass over the course of the present day simulation.

Of the 16 major fishery target groups (i.e. groups which comprise 90% of the model port revenue), one quarter had a substantial change in revenue ($RS > |0.2|$) in response to projected pH (Figure 3). Indirect food web effects drove these changes. Declines in revenue were dominated by declines in Dungeness crab, with effect sizes between -0.42 and -1.0+ at the nine U.S. ports north of Fort Bragg, CA (see Figure 2 caption for explanation of changes greater than 1.0+). A number of ports experienced an increase in revenue from pandalid shrimp (consistently a 0.3

effect size). Seven ports experienced an effect size of -0.27 from declines in Petrale sole, some in the southern region (Figure 3). Three ports in Oregon and Northern California experienced declines in revenue from Dover sole with an effect size of -0.39.

Substantial port-level revenue changes (*RP* > |0.2|) occurred at ports north of Fort Bragg, CA that had a large proportional reliance on Dungeness crab and flatfish (Figures 3 and 4). The most substantial declines occurred where there was both large reliance on Dungeness crab (> 50% of revenue), and where crab biomass declined most severely (Oregon and Northern California). Tillamook experienced the largest decline in revenue as a result of declines in pH due to Dungeness crab, which made up 97.3% of revenue in *2013Baseline* (see impacts by functional group in Figure 3). In PacFIN Dungeness crab made up 86.3% of revenue for Tillamook in 2013 (Appendix C, Table C1). Some ports experienced relief from Dungeness crab declines with increases in pandalid shrimp revenue (most notably for Coos Bay, Oregon with equivalent reliance on Dungeness crab and pandalid shrimp, Appendix C, Table C1). Though Dover and Petrale sole both declined, neither had a strong influence on port-level revenue changes, as they made up a smaller proportion of port revenue (5.0-8.5% and 4.0-19.8% reliance on Dover and Petrale sole, respectively). The eight ports that did not experience declines greater than |0.2| nonetheless experienced modest declines in catch and revenue (Appendix D, Table D1).

Income and employment impacts of OA reflected revenue impacts at the port-level, with the effect on employment at times more exaggerated than the effect on revenue and income (Figure 4). This exaggeration is a result of differences in the income versus employment multipliers by vessel type. For example, Crescent City has catch from four vessel types (Figure 5). The income multipliers for these four vessel types are similar (Appendix D), whereas the multipliers for employment are higher for Crabbers and Other groundfish fixed gear than for Large groundfish trawlers and Shrimpers. Therefore employment declined more strongly (0.71) than revenue and income (0.58 and 0.59; Appendix D) when revenue for Crabbers was reduced. Eureka, in contrast, had similar impacts across revenue (-0.37) income (-0.36) and employment (-0.37) because multipliers were similar across vessel types landing catch at this port.

Parsing port-level revenue to vessel type suggests that nine of the ten vessel types will experience a decline > |0.2| in at least one port (Figure 5). The widespread impacts across vessel types reflect the fact that each vessel type fishes for many species throughout the year, thereby enhancing the likelihood that the catch composition of any single vessel type will include one or more species that is adversely affected by OA. For instance, hake trawlers, sablefish fixed gear vessels and crabber vessels all catch a mixture of species, with Dungeness crab making up a substantial proportion of revenue in our model. With Pacific hake and sablefish both declining over the 100-year spinup of 2063 conditions, hake trawlers and sablefish fixed gear vessels relied more strongly on Dungeness crab than might be expected. Nonetheless, in Oregon and Washington (from Brookings north), these three vessel types and large trawlers show the most detrimental effects from ocean acidification, despite crabber vessels experiencing some relief from increases in pandalid shrimp. In the south, Diver vessels experienced declines in a variety of targeted species (Appendix C, Table C3), ); however, this outcome may be confounded by the difficulty we had in maintaining initial densities of nearshore urchins during baseline model calibration (see Methods, section 2.4).

## 4. Discussion

Though increasing focus has been placed on understanding the economic consequences of climate change (Cooley and Doney 2009; Barange *et al.* 2014; Punt *et al.* 2014; Falkenberg and Tubb 2017), or the resulting vulnerability of species and human communities (Ekstrom *et al.* 2015; Hare *et al.* 2016), such work is frequently at the ecosystem scale, which is a coarser spatial scale than is relevant for many stakeholders and managers. Here, our modeling framework used a meaningful spatial resolution to project regionally varying biological and economic impacts of ocean acidification within the California Current. The model projects that the largest economic consequences may occur for central and northern west coast ports that rely on demersal species for a large portion of fisheries catch, in particular Dungeness crab. Short of immediate global actions to dramatically reduce $CO_2$ emissions, highly vulnerable ports and fisheries will need to consider resilient responses such as establishment of marine reserves, reductions in nutrient loading (Washington State Blue Ribbon Panel on Ocean Acidification 2012), or an emphasis on robust management, flexible and diversified fisheries, and increased monitoring (Pinsky and Mantua 2014; Schindler and Hilborn 2015).

The Atlantis model is best suited to questions of strategic management and exploration of ecosystem dynamics (Fulton *et al.* 2011). We use the model in the spirit of scenario analysis, a common approach used by businesses and organizations to explore possible future pathways with a recognition that there are large uncertainties when projecting forward in time (Postma and Liebl 2005; Amer *et al.* 2013) – the value of taking this approach is discussed further below. The geographic variability of impacts found here resulted from the high resolution in the three components of our modeling framework: the ROMS model, the spatially explicit ecosystem model, and the spatially resolved fishing revenues used in an IO model. Additional scenarios, either through different models, different configuration of this same model, or other types of scenario assessment will more fully scope vulnerable species and communities (Peterson *et al.* 2003).

Indirect food web effects that varied with space reflect the complex predator-prey interactions and detailed species distributions that the Atlantis model can represent. Large-scale analyses of the effects of climate change frequently include either food web dynamics (Ainsworth *et al.* 2011; Barange *et al.* 2014), or high spatial resolution (Cheung *et al.* 2010; Cheung *et al.* 2011; Fernandes *et al.* 2016), but not both. Atlantis provides the ability to consider spatially variable indirect food web effects from changing oceanographic conditions. This high spatial resolution has often not been fully utilized in previously published Atlantis analyses that have summarized findings at a coarser resolution (Kaplan *et al.* 2010; Fulton 2011; Griffith *et al.* 2011; Marshall *et al.* 2017). While some users within management agencies may directly utilize some of the finer scale information, existing publications tend to remain at the broad spatial scale.

Model outputs presented here support results from Griffith *et al.* (2011) suggesting that demersal species are most at risk from OA. Griffith *et al.* (2011) using an Atlantis modeling framework found substantial indirect impacts of OA on demersal fishes, including demersal sharks, shallow macrozoobenthos, cephalopods and benthic filter feeders. In the California Current, Kaplan *et al.* (2010) found very strong indirect effects on flatfish, namely English sole (*Parophrys vetulus*), yellowtail rockfish (*Sebastes flavidus*) and arrowtooth flounder. Unlike Kaplan *et al.* (2010), where most of the indirect effects appeared to be on demersal fishes, Marshall *et al.* (2017) and this paper found strong indirect effects on Dungeness crabs. These differences in model outcomes are not unexpected, given that the Kaplan *et al.* (2010) and the present model had different functional group structures. In contrast to the demersal community, consumers in the pelagic community (pelagic fishes, marine mammals and seabirds) experienced only small indirect effects in our model, consistent with previous results.

Our work and that of Marshall *et al.* (2017) highlight the importance of integrating vulnerability information for an individual species from a variety of methodologies to understand potential impacts across ecological scales, from physiological to population responses. For instance, previous work on Dungeness crab has provided mixed results regarding OA. Lab studies have shown that lowered pH reduces larval survival and slows development rates (Miller *et al.* 2016). In contrast, a population vulnerability assessment integrating across life stages showed that the overall population may have low vulnerability, even with high larval vulnerability (Hodgson *et al.* 2016). Here, we see that although direct effects from pH-induced mortality are insubstantial in our model, indirect food web effects (loss of prey) made Dungeness crab populations quite vulnerable. That declines in fishery revenue are so dependent on this species warrants future attention on details of Dungeness crab vulnerability and vulnerability of their primary prey resources.

Finally, the variability in pH conditions and resulting ecological impacts was driven in part by the oceanographic model (ROMS). There are a number of methods simpler than computationally intensive dynamical downscaling through ROMS, however, each alternative method has limitations. For instance, initial insights into the ecosystem impacts from OA using Atlantis employed a uniform mortality rate for groups assumed to be impacted by declines in pH (Kaplan *et al.* 2010; Griffith *et al.* 2011). Alternatively, direct use of global circulation models (GCMs) is possible but has limited utility in the California Current because they currently do not resolve upwelling regions from eastern boundary currents (Gruber *et al.* 2012) nor tidal forces on the continental shelf (Ådlandsvik and Bentsen 2007). Thus, using these models directly for projections of OA in the California Current is not yet ideal. Statistical downscaling is an alternative option, but is limited in the number of oceanographic variables produced and assumes that relationships between current climate conditions (correlations between oceanographic variables) hold under future climate conditions, which may or may not occur (Ekström *et al.* 2015). Given these varying limitations, ROMS was used here as it provides high-resolution output that resolves upwelling and includes numerous oceanographic parameters. The drawback to this approach was that it necessitated considering potential end point outcomes rather transitory effects.

### *4.1 Management Applications*

While this initial analysis was constrained by available inputs to considering a single scenario, it does form a solid basis for identifying vulnerable species and ports that warrant further scientific and management focus. As evidenced by some of the counter intuitive results and the high heterogeneity found here, simply relying on "common sense" mental extrapolation is not appropriate in path dependent instances (such as natural resource management and planning). Consequently, even uncertain results can provide useful insights and starting points for discussions, planning and strategic research prioritization.

Output from other Atlantis ecosystem models has been used to inform strategic management questions, and outputs from the present work have the potential to fit into existing management efforts within the U.S. For example, in Australia Atlantis has been used as part of management strategy evaluation exercises to compare different management levers in the process of restructuring south-eastern Australian federal fisheries (Fulton *et al.* 2011). It has also been used (alone or in combination with other models) to inform the Australian government of ecosystem responses to climate change (Fulton and Gorton 2014; Fulton et al 2018).

Within the U.S. there has been an increasing focus on both developing climate change strategies and understanding risk to different regions of the coastline to improve resilience and reduce the impact of forthcoming change. One of the seven objectives in the NOAA/National Marine Fisheries Service (NMFS) Climate Science Strategy is to develop future projections, specifically at regional scales that are relevant to management (Busch et al. 2016). Similarly, The West Coast Ocean Acidification and Hypoxia Science Panel note the importance of both higher spatial resolution in model outputs, and the need for coupling biophysical and ecosystem models to improve our understanding of potential outcomes (Chan et al. 2016). Thus, outputs from this work can feed into these management priority areas and can be used in conjunction with alternative methodologies, such as the NMFS Climate Vulnerability Assessments (NOAA 2016).

Within this advisory context, it is important to recognize that the results presented here represent a single instance of how ocean acidification may impact the biology and economics of the coastline bordering the California Current. As with any highly parameterized and complex ecosystem model, it is important to consider the results in conjunction with an ensemble of other models that use different assumptions and modeling frameworks. We are hopeful that in future ensemble projections will build on this current work, and provide alternative scenarios and projections so that we can gain a fuller picture of possible future vulnerabilities within the system. However, the reality that more models are needed does not diminish the utility of this approach, because in taking steps towards understanding vulnerability within the California Current to ocean acidification there is a need to go beyond mental extrapolation and tangibly build on past information.

*4.2 Uncertainties and Limitations*

Given the constrained nature of the work presented here it is important to be explicit about the associated uncertainties and limitations. While model complexity is useful in providing higher resolution outputs and more explicitly dealing with a broader range of specific processes, it also introduces uncertainty (Cheung *et al.* 2016). A number of elements regarding model uncertainty as defined by Hawkins and Sutton (2009) and Cheung *et al.* (2016) were addressed in Marshall *et al.* (2017), thus we focus on uncertainties as they pertain to influencing spatial impacts of OA.

The ROMS model used for present day and future oceanographic predictions was forced by a single realization of a global earth system model (GFDL-ESM2M) run under a single emissions scenario (RCP 8.5). Both of these simplifications should be addressed in future efforts. Future work would benefit from downscaling of multiple realizations and multiple global earth system models to quantify both intrinsic (natural) and model (structural) uncertainty related to physics and biogeochemistry. This could be substantially addressed by adding two or three different downscaling studies and does not necessarily require dozens of computationally intensive model iterations. Additionally, alternative emissions scenarios should be considered (Hawkins and Sutton 2009; Hollowed *et al.* 2009).

In our Atlantis ecosystem model, a substantial structural uncertainty was the representation of vertebrate spatial movements. Vertebrate biomass was distributed using prescribed seasonal shifts, with the daily distribution interpolated linearly from one season to the next. More complex movement patterns have been applied in some other Atlantis models (Fulton *et al.* 2014), including effects such as thermal tolerances, but these add substantially to model calibration. Consequently, we used prescribed seasonal movement, which effectively handles large model domains under historic conditions, but limits the evolution of spatial variability in future projections. There are two potential consequences of our approach. If a vertebrate functional group only experiences a decline in one region (e.g., -0.6 off Washington state), that effect will be smoothed out across all regions reducing the extent to which we will observe heterogeneous impacts on vertebrate species. In addition, it masks the potential for strong spatial rearrangement of species distributions in response to environmental conditions. Such redistribution might mean coastwide biomass shifts different to those seen here (e.g. a group may not have an impact exceeding our reporting threshold effect size of > |0.2| as it has moved to more hospitable areas). The economic effects of a shift may also differ to what is seen here, particularly if a fleet components is unable to match the distributional shift (regardless of the realized biomass levels).

Predator diet compositions are notoriously challenging to estimate (Baker *et al.* 2014), and in Atlantis diets evolve dynamically based on shifts in prey abundance and spatial overlap between predators and prey. For example, starting conditions allowed Dungeness crab to prey upon many functional groups, but realized Dungeness crab diets were relatively simple and consisted of mostly bivalves, benthic herbivorous grazers and Dungeness crab (which are known to cannibalize (Fernández 1999)). With decreases in two of their main prey resources, cannibalism increased, likely driving declines in biomass (Figure 2). However, it is possible that this was an oversimplification of true crab diets, making them more sensitive to changes in the model than they would be to changes in the natural environment.

In the economic analysis we assumed fixed prices from present day through to the 2060s and did not include substitution between fisheries inputs (fisheries always use the same amount of diesel, food, etc. per dollar of revenue). These are both characteristics of IO models (Seung and Waters 2006) which have been used previously for moderate future forecasts (15 years; Kaplan and Leonard 2012), and long-term forecasts over 50 years (Fernandes *et al.* 2016). There is presently no alternative for analyzing port-level outputs for the US west coast. In the future, long term projections of economic responses may be better forecasted with techniques such as a Computable General Equilibrium (CGE) model (e.g., (Finnoff and Tschirhart 2003) which factors in changing prices and input substitutions. We note that IO model estimates of impacts on employment and income did not substantially differ from changes in revenue (Figure 5); our economic outputs are most useful for identifying ports with higher relative vulnerability.

In addition to these primary limitations are a few key biological assumptions that could affect our results. We looked at ocean acidification as a single stressor, when it is well established that the California Current experiences a variety of anthropogenic impacts (Halpern *et al.* 2008). Thus future analyses would benefit from investigating cumulative effects, particularly regarding temperature which is highly correlated to observed pH (Reum *et al.* 2014). Our work assumes pH only affects mortality, though additional physiological processes have been included in other recent modeling efforts (Fernandes *et al.* 2016). Finally, we do not include the evolution of species responses or changes in species distributions given future climate conditions, and both of these are possible, if not likely (Cheung *et al.* 2010; Sunday *et al.* 2011; Lohbeck 2012; Pinsky *et al.* 2013).

### *4.3 Conclusions*

Ocean acidification has the potential to restructure marine ecosystems and strongly impact human communities dependent on marine resources (Le Quesne and Pinnegar 2012;

Ekstrom *et al.* 2015). It is critical to assess risks to prioritize research and make strategic management decisions. The California Current is already experiencing the consequences of regional variations in pH, and here we identify functional groups that are most likely to change in response to declines in pH within the region, and the ports that may experience the largest economic impacts from these changing conditions. Substantial biological impacts were projected to occur for benthic invertebrate and fish species. Economic consequences were most severe where declines in Dungeness crab biomass resulted in lost fisheries catch. Dungeness crab is a major fishery resource on the US West Coast, making up between 40-89% of revenue for 8 of the 17 ports (43% on average across all ports; Appendix C). Our results highlight the value of methods that integrate physical, ecological and economic consequences of climate change at a spatial resolution compatible with human communities and management decisions. The outcomes can prioritize further research and adaptive management approaches from state and tribal agencies in the face of these climatic changes.

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## Tables

Table 1. US west coast vessel types, as categorized by IO-PAC, with the designation use in this model: “Spatial” if the catch from the fleet was tracked by each port and “Generic” if the fleet’s catch was lumped into the generic coastwide fishery.

| IO-PAC Fleet | Atlantis Designation |
|---|---|
| Pacific whiting trawler | Spatial |
| Large groundfish trawler | Spatial |
| Small goundfish trawler | Spatial |
| Sablefish fixed gear | Spatial |
| Other groundfish fixed gear | Spatial |
| Pelagic netter | Generic |
| Migratory netter | Generic |
| Migratory liner | Generic |
| Shrimper | Spatial |
| Crabber | Spatial |
| Salmon troller | Generic |
| Salmon netter | Generic |
| Other netter | Generic |
| Lobster vessel | Generic |
| Diver vessel | Spatial |
| Other, more than 15K | Generic |
| Other, less than 15K | Generic |

Table 2. Ten functional groups directly responding to pH in our *2063pHmortality* scenario, the species making up each group, their relative survival scalar (proportional mortality for one unit decline in pH), realized pH exposure in the *2063pHmortality* scenario. Effect size for each species ($E_i$) was calculated for the whole model domain weighted by biomass within each polygon, and the min and max effect across regions. Effect sizes stronger than |0.2| are bold.

| **Functional Group Name** | **Example species** | **Relative survival scalar** | **Realized Exposure** | **OA effect size: mean (min, max)** |
|---|---|---|---|---|
| Benthic herbivorous grazers | sea urchins (*Allocentrotus fragilis,* Strongylocentrotidae), snails | 0.1 | 7.670 | **-0.74 (-0.80, -0.55)** |
| Mesozooplankton | copepods | 0.099 | 7.726 | -0.07 (-0.16, -0.04) |
| Bivalves | bivalves | 0.089 | 7.768 | **-0.29 (-0.42, -0.24)** |
| Pteropods | thecosome pteropods | 0.081 | 7.714 | -0.09 (-0.16, -0.05) |
| Crabs | crabs (excluding Dungeness crab) | 0.070 | 7.669 | **-0.09 (-0.82, 0.99)** |
| Shallow benthic filter feeders | tunicates, sponges | 0.055 | 7.770 | 0.11 (0.07, 0.13) |
| Crangon shrimp | shrimps (excluding pandalids) | 0.045 | 7.745 | -0.19 (-0.19, -0.19) |
| Dungeness crab | Dungeness crab | 0.041 | 7.770 | **-0.47 (-1+, -0.03)** |

| | | | | |
|---|---|---|---|---|
| Benthic carnivores | polychaetes, nematodes | 0.039 | 7.805 | **-0.23 (-0.24, -0.16)** |
| Deposit feeders | amphipods, isopods | 0.037 | 7.658 | **-0.27 (-0.35, -0.17)** |

## Figures

Figure 1. Atlantis polygons showing entire model domain. Ports are identified in the inset for US West Coast ports only.

Figure 2. Change in biomass for species that were found to have >|0.20| change in biomass in any of the six regions, where changes are effect sizes and are therefore unit-less. Changes <|0.20| are grey, and changes >|0.20| (either positive or negative) are colored according to the color bar. White indicates that there is no biomass for the specified functional group in the region (e.g., there are no Dungeness crab and pandalid shrimp off of Mexico). Numbers along the top of the plot indicate the change in pH within the region between 2013 and 2063 (averaged across all depths and months). Numbers in boxes are the realized exposure to pH, provided only for groups responding directly to pH. Realized exposure is the weighted mean pH species experienced based on their realized distributions in the model, weights were the relative biomass experiencing each pH. Note that the changes greater than 1.0 are possible due to the response metric which standardizes the decline in biomass (numerator) by *2013Baseline* (denominator), and in some cases the 2013 baseline had lower biomass than the *2063Baseline*.

Figure 3. Functional groups making up (in sum) a minimum of 90% of port-level fishery revenue for the base 2013 run and their change in revenue under future ocean acidification levels. Any species on the x-axis which was part of the groups making up 90% of catch for each port has a square, where the number in the square signifies the proportion of revenue from that functional group in the model. The squares are then colored according to change in biomass from our response metric, legend provided.

Figure 4. Change in revenue (A), income (B) and employment (C) by port. Changes with an absolute value less than 0.20 are grey, and changes greater than 0.20 (either positive or negative) are colored according to the color bar binned by 0.10. Port groups are labeled in Figure 1.

Figure 5. Change in fleet-specific revenue driving port-level revenue changes. Changes with an absolute value less than 0.20 are grey, and changes greater than 0.20 (either positive or negative) are colored according to the color bar binned by 0.10. White indicates that the fleet did not land any species at the identified port in 2013.

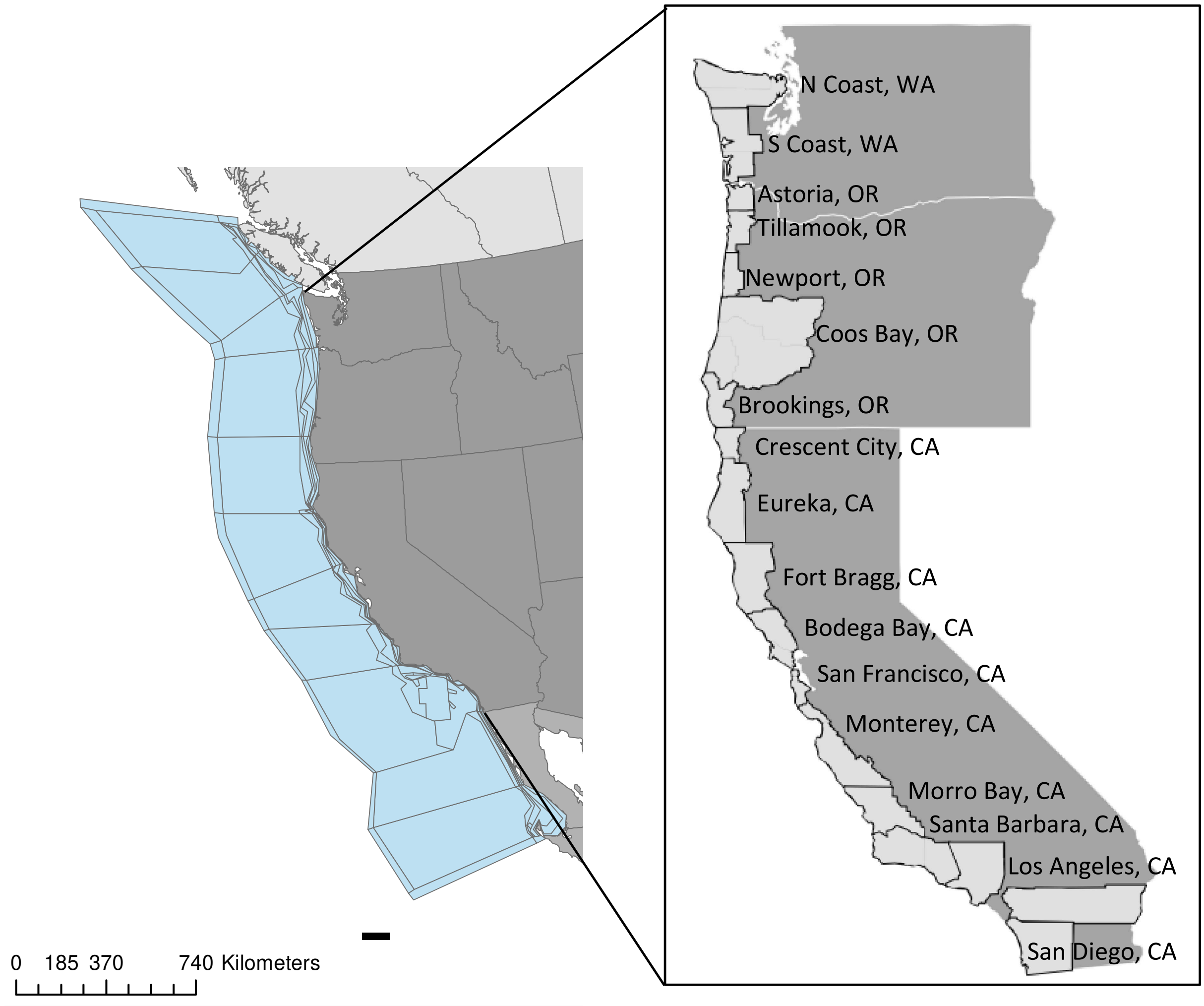
0 185 370 740 Kilometers
N Coast, WA
S Coast, WA
Astoria, OR
Tillamook, OR
Newport, OR
Coos Bay, OR
Brookings, OR
Crescent City, CA
Eureka, CA
Fort Bragg, CA
Bodega Bay, CA
San Francisco, CA
Monterey, CA
Morro Bay, CA
Santa Barbara, CA
Los Angeles, CA
San Diego, CA

Figure 1.

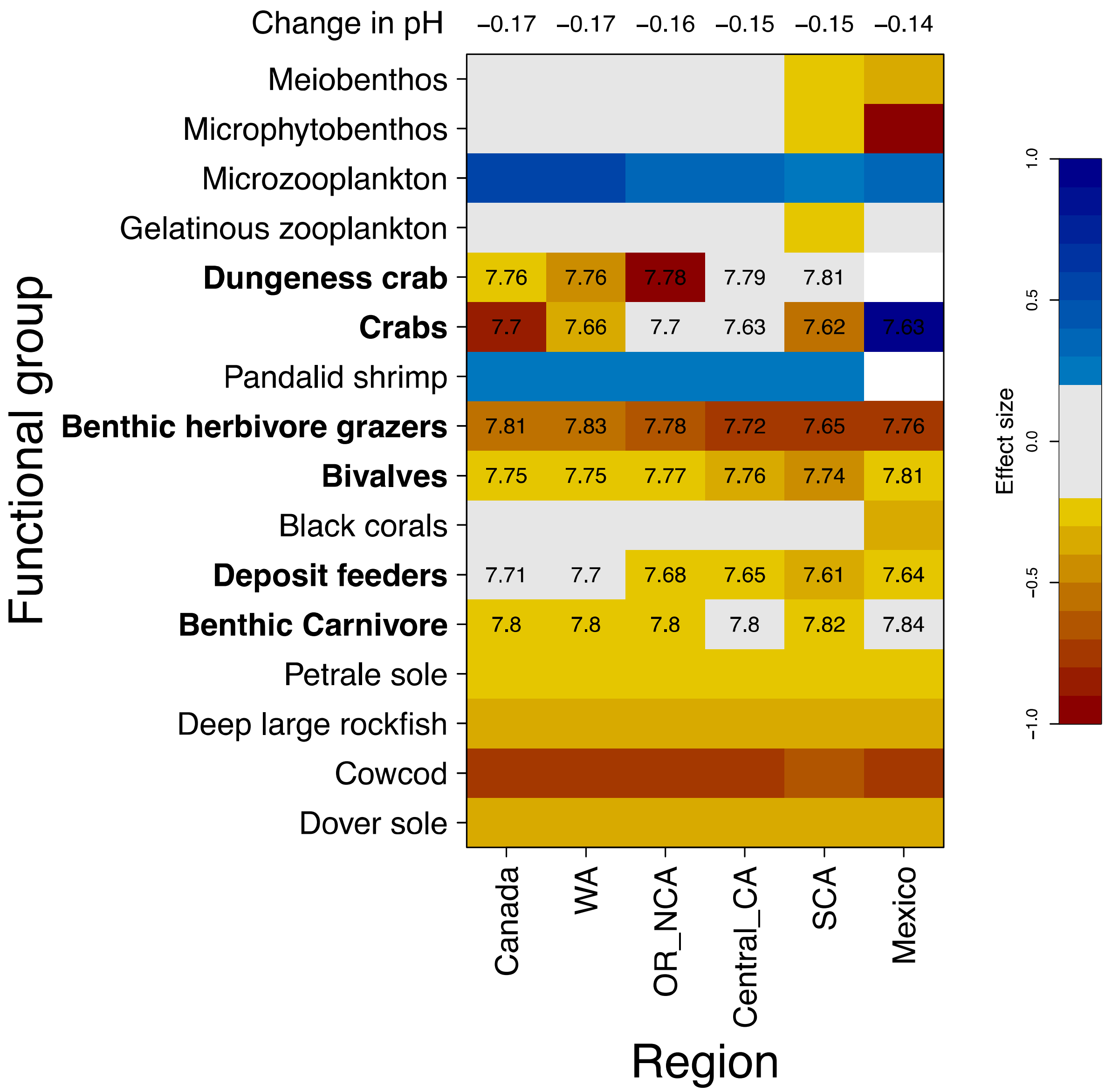
Change in pH
−0.17
−0.17
−0.16
−0.15
−0.15
−0.14
Functional group
Meiobenthos
Microphytobenthos
Microzooplankton
Gelatinous zooplankton
Dungeness crab
Crabs
Pandalid shrimp
Benthic herbivore grazers
Bivalves
Black corals
Deposit feeders
Benthic Carnivore
Petrale sole
Deep large rockfish
Cowcod
Dover sole
7.76
7.76
7.78
7.79
7.81
7.7
7.66
7.7
7.63
7.62
7.63
7.81
7.83
7.78
7.72
7.65
7.76
7.75
7.75
7.77
7.76
7.74
7.81
7.71
7.7
7.68
7.65
7.61
7.64
7.8
7.8
7.8
7.8
7.82
7.84
Canada
WA
OR_NCA
Central_CA
SCA
Mexico
Region
Effect size
1.0
0.5
0.0
−0.5
−1.0

Figure 2.

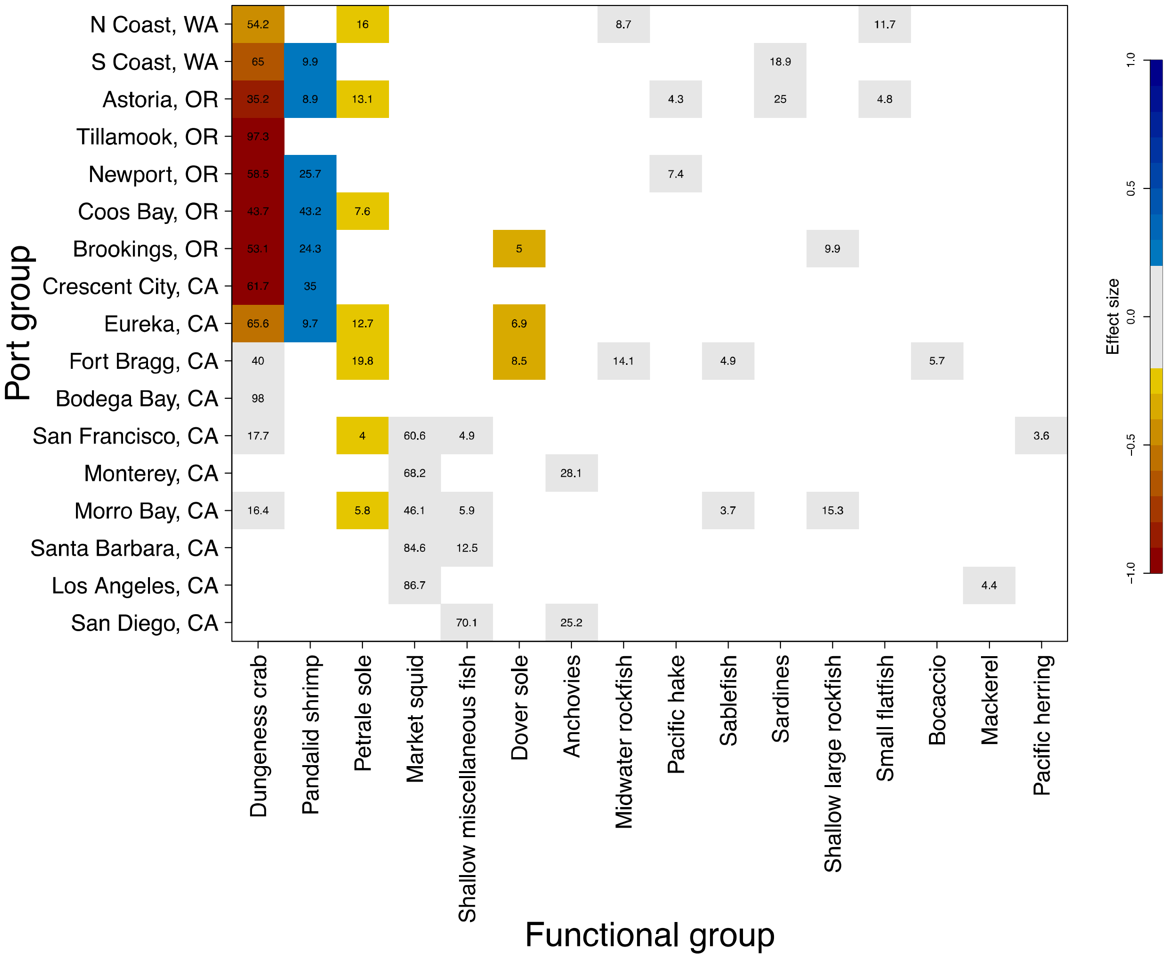
Port group
N Coast, WA
S Coast, WA
Astoria, OR
Tillamook, OR
Newport, OR
Coos Bay, OR
Brookings, OR
Crescent City, CA
Eureka, CA
Fort Bragg, CA
Bodega Bay, CA
San Francisco, CA
Monterey, CA
Morro Bay, CA
Santa Barbara, CA
Los Angeles, CA
San Diego, CA
54.2 16 8.7 11.7
65 9.9 18.9
35.2 8.9 13.1 4.3 25 4.8
97.3
58.5 25.7 7.4
43.7 43.2 7.6
53.1 24.3 5 9.9
61.7 35
65.6 9.7 12.7 6.9
40 19.8 8.5 14.1 4.9 5.7
98
17.7 4 60.6 4.9 3.6
68.2 28.1
16.4 5.8 46.1 5.9 3.7 15.3
84.6 12.5
86.7 4.4
70.1 25.2
Dungeness crab
Pandalid shrimp
Petrale sole
Market squid
Shallow miscellaneous fish
Dover sole
Anchovies
Midwater rockfish
Pacific hake
Sablefish
Sardines
Shallow large rockfish
Small flatfish
Bocaccio
Mackerel
Pacific herring
Functional group
Effect size
1.0
0.5
0.0
−0.5
−1.0

Figure 3.

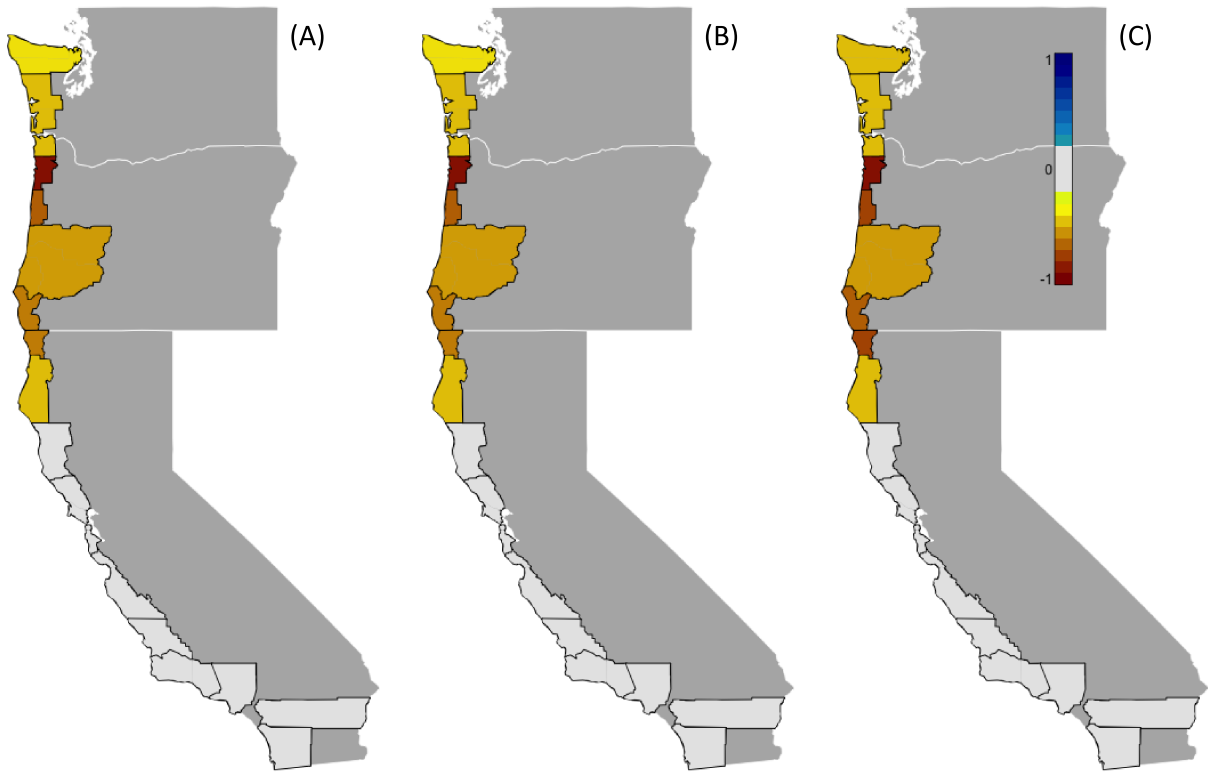


Figure 4.

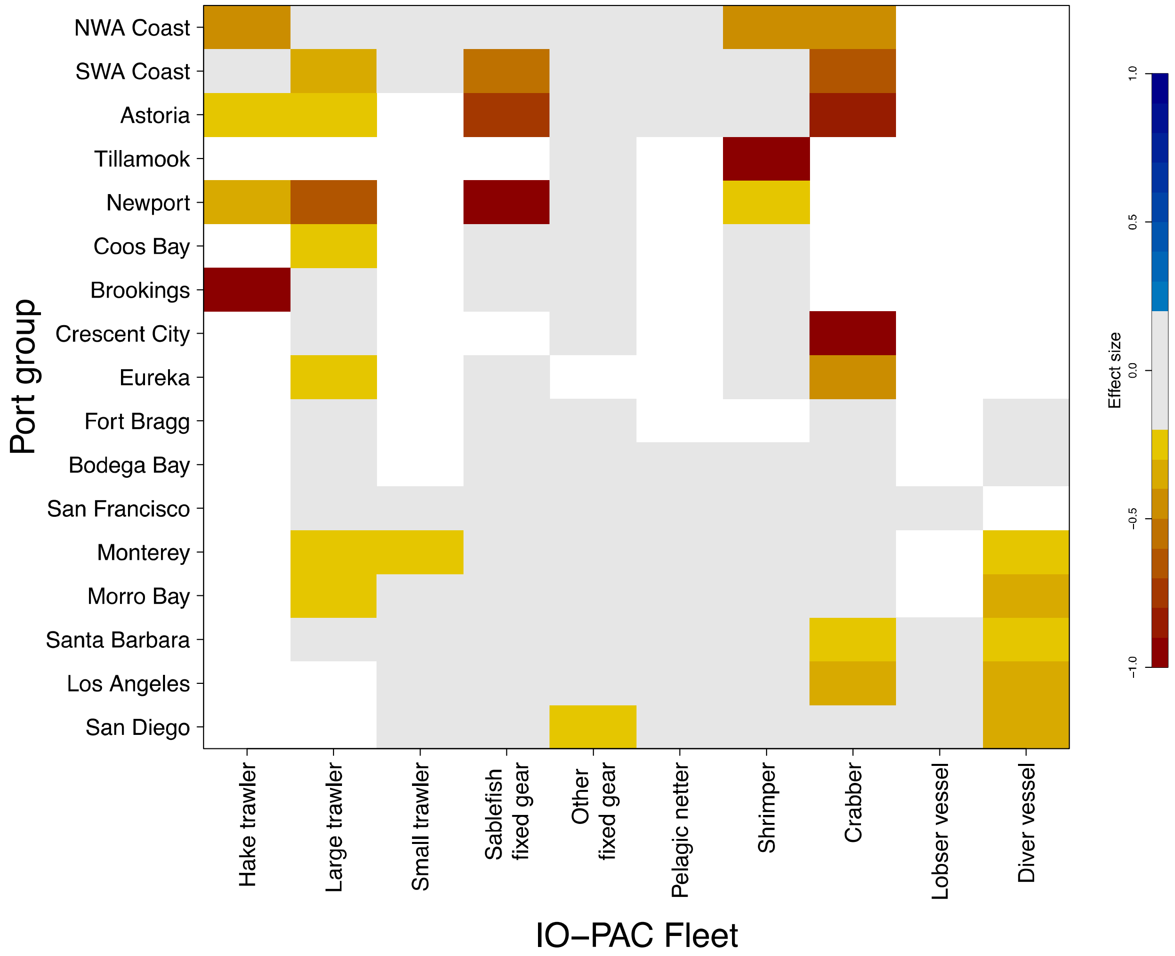
NWA Coast
SWA Coast
Astoria
Tillamook
Newport
Coos Bay
Brookings
Crescent City
Eureka
Fort Bragg
Bodega Bay
San Francisco
Monterey
Morro Bay
Santa Barbara
Los Angeles
San Diego
Port group
Hake trawler
Large trawler
Small trawler
Sablefish fixed gear
Other fixed gear
Pelagic netter
Shrimper
Crabber
Lobser vessel
Diver vessel
IO−PAC Fleet
Effect size
1.0
0.5
0.0
−0.5
−1.0

Figure 5.