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Impact of multiple stressors on juvenile fish in estuaries of the northeast Pacific

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Abstract

A key step in identifying global change impacts on species and ecosystems is to quantify effects of multiple stressors. To date, the science of global change has been dominated by regional field studies, experimental manipulation, meta-analyses, conceptual models, reviews, and studies focusing on a single stressor or species over broad spatial and temporal scales. Here, we provide one of the first studies for coastal systems examining multiple stressor effects across broad scales, focused on the nursery function of 20 estuaries spanning 1,600 km of coastline, 25 years of monitoring, and seven fish and invertebrate species along the northeast Pacific coast. We hypothesized those species most estuarine dependent and negatively impacted by human activities would have lower presence and abundances in estuaries with greater anthropogenic land cover, pollution, and water flow stress. We found significant negative relationships between juveniles of two of seven species (Chinook salmon and English sole) and estuarine stressors. Chinook salmon were less likely to occur and were less abundant in estuaries with greater pollution stress. They were also less abundant in estuaries with greater flow stress, although this relationship was marginally insignificant. English sole were less abundant in estuaries with greater land cover stress. Together, we provide new empirical evidence that effects of stressors on two fish species culminate in detectable trends along the northeast Pacific coast, elevating the need for protection from pollution, land cover, and flow stressors to their habitats. Lack of response among the other five species could be related to differing resistance to specific stressors, type and precision of the stressor metrics, and limitations in catch data across estuaries and habitats. Acquiring improved measurements of impacts to species will guide future management actions, and help predict how estuarine nursery functions can be optimized given anthropogenic stressors and climate change scenarios.

KEYWORDS
anthropogenic stress, Chinook salmon, English sole, estuary, human impacts, juvenile life stage, pollution

1 | INTRODUCTION

An important challenge in measuring impacts of global change is to determine how multiple stressors cause changes to species and ecosystem function. Estuaries and coasts are among the ecosystems that are immensely threatened yet valuable to people (Costanza et al., 1997). In these ecosystems, people and their interaction with the environment are aggregated, as are many ecosystem services such as fish production (Beck et al., 2001). Our understanding of global change in estuaries and coasts is derived from regional field studies (e.g., hypoxia effects on flatfish in Elkhorn Slough, CA, USA; Hughes et al., 2015), small-scale experiments (e.g., additive, antagonistic, or synergistic stressor effects on macroalgae; Vye, Emmerson, Arenas, Dick, & O'Connor, 2015), meta-analyses (e.g., synthesizing interactive and cumulative effects of stressors; Crain, Kroeker, & Halpern, 2008), conceptual and theoretical models (e.g., Vinebrooke et al., 2004), and reviews (e.g., syntheses of multiple stressors on coral reefs; Harborne, Rogers, Bozec, & Mumby, 2017). Studies that examine broad temporal and spatial scales tend to be conducted in freshwater or terrestrial ecosystems (e.g., Ceballos, Ehrlich, & Dirzo, 2017; Esselman et al., 2011), emphasize single species across larger geographic gradients (e.g., Cheng et al., 2015) or a single stressor across longer temporal gradients (e.g., Barceló, Ciannelli, Olsen, Johannessen, & Knutsen, 2016). Thus, we have an incomplete understanding of how threats from coastal development impact fish on large scales, and how these scalar issues relate to regional management priorities. Here, we address this knowledge gap by analyzing the effects of multiple stressors on the presence and abundance of seven fish species that rely on nursery functions of northeast Pacific coast estuaries.

We define a stressor as anthropogenic changes to environmental drivers that affect estuarine habitat quality and the species that occupy those habitats. Stressors to coasts and estuaries can take many forms, including shoreline urbanization, pollution, reduced water flows, and eutrophication (Airoldi & Beck, 2007; Greene, Blackhart, Nohner, Candelmo, & Nelson, 2015; Kennish, 2002; USEPA, 2012). Although stressors can be documented directly, significant effort is required to quantitatively compare them to the condition of species, especially given interactions among multiple stressors (Kroeker et al., 2016; Lefcheck, Wilcox, Murphy, Marion, & Orth, 2017; Vasconcelos et al., 2007). The management implications of this are real, both in developing methods to control escalating stressors, and in determining the ongoing consequences of these stressors on protected fisheries species (Kappel, 2005). Ecologists have recognized that juveniles aggregate in protective and productive shallow habitat mosaics of coastal systems, and hypothesized that shallow habitats contribute disproportionally to adult populations (Beck et al., 2001; Dahlgren, Kellison, & Adams, 2006; Nagelkerken, Sheaves, Baker, & Connolly, 2015). Thus, it is advantageous to investigate effects of stressors on fish in coastal systems because (i) juveniles rely on these systems to provide critical nursery habitats, and (ii) these systems are focal areas for conservation and management (Beck et al., 2001).

To examine the role of multiple stressors across northeast Pacific coast estuaries (U.S. states of California, Oregon, and Washington), we focused on seven fish and invertebrate species, a subset with ample available catch data of 15 focal species previously identified by Hughes, Levey, and Brown (2014). These species represent major guilds, are of commercial, recreational, and cultural importance, and have life histories that encompass large portions of northeast Pacific estuaries, spanning more than 1,600 km of coastline. Five of these have management targets through their listing under the U.S. Endangered Species Act and/or because they are important fisheries species—Chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*Oncorhynchus kisutch*), Dungeness crab (*Cancer magister*), English sole (*Parophrys vetulus*), and Pacific herring (*Clupea pallasii*). Two of the species, although not tied to such targets—shiner perch (*Cymatogaster aggregata*) and Pacific staghorn sculpin (*Leptocottus armatus*)—can play important ecological roles in estuarine food webs as secondary consumers (Hughes et al., 2014).

Our assessment builds on previous syntheses of Pacific coast estuaries (Emmett, Stone, Hinton, & Monaco, 1991; Gleason, Newkirk, & Merrield, 2011; Heady et al., 2014; Hughes et al., 2014; Monaco, Emmett, Nelson, & Hinton, 1990; Monaco, Lowry, & Emmett, 1992) by combining location-specific fish sampling with published values of stressor scores indicating levels of human impact (below, from Greene, Blackhart et al., 2015). We focused on three different classes of stressors, representing (i) land cover/land use (e.g., amount developed, change from natural estuarine classes), (ii) alteration of fluvial processes (e.g., changes in patterns of flow in rivers entering estuaries), and (iii) sources of pollution (e.g., toxic releases and pollution discharges). The large-scale stressors that we examined include habitat alterations that impact fish populations worldwide (Able, Grothues, & Kemp, 2013; Bilkovic & Roggero, 2008; Valesini, Teedley, Clarke, & Potter, 2014) due to growing human populations dependent on coastal ecosystem services (Kennish, 2002; UNEP, 2006).

Our approach was to compile raw data from beach seine collections in shallow waters to examine effects of stressors on presence and abundance of the seven focal species. Our hypothesis was that there would be lower presence and abundance of fish in estuaries with higher stressor scores for species that were most estuarine dependent and/or negatively affected by human impacts.

2 | MATERIALS AND METHODS

2.1 | Data collection

We used the network of coastal scientists and managers associated with the Pacific Marine and Estuarine Fish Habitat Partnership to generate a pool of potential data holders of nearshore and estuarine fish. From this pool, 120 managers and researchers responded to an online survey, providing descriptions of the scope and availability of their data. This contact list expanded with additional professional referrals and estuary-specific queries of federal and state data portals to fill geographic gaps. Ultimately, more than 200 individuals from 73 different agencies were contacted regarding the data request. Beach seine data, encompassing the years 1990 to 2014, were combined from 22 sampling programs representing 20 estuaries (Figure 1; Table S1). We prioritized datasets that sampled multiple species and specified life stage or length parameters. Submitted datasets were compiled and uploaded to a Microsoft Access database, based on an observations data model that provided a consistent format for the storage and retrieval of point observations in a relational database. This approach is designed to facilitate an integrated analysis of large datasets collected by multiple investigators (Horsburgh, Tarboton, Maidment, & Zazlavsky, 2008). When geographic coordinates were unavailable, fish observations were resolved to an estuary polygon, if metadata allowed. Prior to analysis, data were queried by species, life stage (when available), and location to include only samples collected within U.S. Pacific estuary boundaries using current National Oceanic and Atmospheric Administration (NOAA) designations.

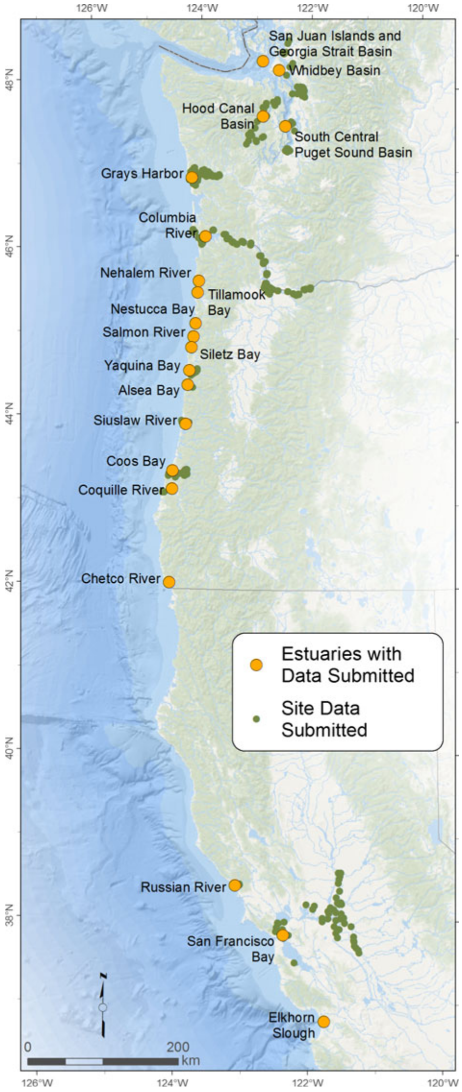

FIGURE 1 Location of the 20 estuaries analyzed, and specific sampling locations [Colour figure can be viewed at wileyonlinelibrary.com]

Throughout our data collection and analysis, we observed sources of data variability and constraints that limited the extent to which we could apply our analyses. Some estuaries were less sampled than others, and not all available datasets were submitted. Therefore, lack of data from one estuary did not necessarily indicate that data do not exist; it could also mean that the data were simply not available. Data coverage within each estuary varied, as did the precision of specific measurements of latitude and longitude where sampling occurred. Also, nontarget species from a study focusing on one group (e.g., salmonids) were often binned to general levels (e.g., flatfish, cancer crabs), and in those cases, could not be incorporated into our species-level assessment. Recent advances in quantitative ecology allowed us to integrate, reduce, and analyze large-scale datasets from varying sources, recognizing sources of variability such as differences in experimental sampling, model parameters, and functional differences in underlying biological processes (Thorson et al., 2013). Our analysis path was chosen to best account for the variability and constraints that are expected when combining data from different research groups.

2.2 | Data analysis

We compared fish abundances with anthropogenic impacts on estuaries, using the stressor scores calculated for each estuary by the 2010 National Assessment (Greene, Blackhart et al., 2015; Table 1) which included habitat alterations expected to impact fish populations. Stressor scores in Greene, Blackhart et al. (2015) combined 43 indicator datasets into four categories: (i) land cover/land use, (ii) alteration of river flows, (iii) pollution sources, and (iv) eutrophication (Table S2). Of these, the first three categories had suitable coverage for the 20 estuaries in our study and were used in subsequent analyses; eutrophication was not included because it had the greatest number of data gaps and data were not available for four of the estuaries in this analysis (Greene, Blackhart et al., 2015). Pollutants, river flow, and land use all showed correlations with eutrophication as summarized in Greene, Blackhart et al. (2015), and this can be used to infer links to eutrophication as a stressor.

The remaining three metrics were themselves indices of multiple measures (Table S2). Land use/land cover datasets evaluated recent areal land cover of agriculture, development, and estuary habitat as well as changes in these classes during the last 20 years. River flow datasets integrated numerous flow metrics, including the number of dams per watershed area as well as indicators of hydrologic alteration (Richter, Baumgartner, Powell, & Baum, 1996) from United Stations Geological Survey (USGS) river flow gages closest to head of tide, using 15-year averages of mean discharge, maximum and minimum discharge, high and low pulse duration, as well as the trend in these values across the entire time series. Pollution datasets included the number of mines, toxic release sites, pollution discharge sites, and hazardous waste sites per unit watershed area. Values of each stressor were scaled 0 to 1 based on the ranks of individual datasets

TABLE 1 Stressor scores used in analyses

Estuary	Flow	Land cover	Pollution
San Juan Islands/Georgia Strait Basin, WA	0.503	0.781	0.5
Whidbey Basin, WA	0.172	0.448	0.261
Hood Canal Basin, WA	0.483	0.019	0.275
South Central Puget Sound Basin, WA	0.39	0.516	0.903
Grays Harbor, WA	0.158	0.101	0.435
Columbia River, WA and OR	0	0.37	0.307
Nehalem River, OR	0.721	0.151	0
Tillamook Bay, OR	0.649	0.425	0.366
Nestucca Bay, OR	0.423	0.201	0
Salmon River, OR	0.367	0.274	0
Siletz Bay, OR	0.311	0.306	0.284
Yaquina Bay, OR	0.964	0.083	0.389
Alsea Bay, OR	0.039	0.119	0.247
Siuslaw River, OR	0.377	0.179	0.183
Coos Bay, OR	0.925	0.261	0.449
Coquille River, OR	0.655	0.772	0.27
Chetco River, OR	0.682	0.393	0
Russian River, CA	0.397	0.329	0.596
San Francisco Bay, CA	0.463	0.800	0.704
Elkhorn Slough, CA	0.324	0.859	0.802

The 20 Estuaries are listed from north to south, with stressors of flow, land cover, and pollution from Greene, Blackhart et al. (2015).

across estuary systems in the National Assessment, with higher values indicating greater human impacts (Greene, Blackhart et al., 2015). The land cover stressor index included amount of land developed and change from natural estuarine classes, thereby acknowledging alterations from pristine states unique to each estuary. These stressors are linked to fish habitat quality and quantity. For example, development described by stress to land cover reduces the amount of estuarine nursery habitat available to juvenile fish, pollution degrades water quality, and alteration of river flows changes downstream hydrology of estuaries. Our analysis was constructed to evaluate these associations across systems and taxa.

Several estuaries with fish data required interpretation of stressor scores from nearby sites or from newly acquired data. Monterey Bay values for flow were used in Elkhorn Slough, South and Central Bay values were averaged for San Francisco Bay, Drayton Bay values were used for San Juan Islands and Georgia Strait Basin, and Puget Sound values were used for South Central Puget Sound Basin. We estimated one value describing flow stress in the Salmon River, Oregon, by averaging the neighboring Nestucca and Siletz Rivers, which were similar in geographic location and flow stress values (≈13 km to the north and south, respectively, values of 0.423 and 0.311, average value of 0.367). We calculated flow stressor values for Coos Bay, Oregon and Yaquina Bay, Oregon using data made available after the analysis of Greene, Blackhart et al. (2015).

We quantified the relationship between species abundance and estuarine stressors using generalized additive mixed models (GAMMs; Zuur, Ieno, Walker, Saveliev, & Smith, 2009; Wood, 2011). This modeling approach was appropriate because fish catches were non-normally distributed, there were non-linear annual trends in species abundance, and species were repeatedly sampled from the same estuaries and years. As is typical in fisheries investigations, the data were zero-inflated and we therefore fit two models for each species: one describing presence or absence of a species (hereafter: presence/absence model) and the other describing catch per unit effort (CPUE) when a species was present (hereafter: CPUE model). Presence/absence models were fit using a binomial distribution and a logit-link function. CPUE models were fit using a negative binomial distribution and log-link function. We used a negative binomial distribution rather than a Poisson distribution because, as often occurs in ecological data, the variance of our count data exceeded its mean (Zuur et al., 2009).

The fixed effects considered for the models were flow, land cover, and pollution stressors, and salinity zone. We used salinity zones of freshwater tidal (<0.5 ppt), mixing (0.5–25 ppt), and seawater (>25 ppt). Salinity zones were assigned based on averaged field measurements from a location. If salinity data were unavailable, we used the NOAA 3-Zone Average Annual Salinity Digital Geography layer (spatial join w/50 m buffer). If no field data or NOAA digital data existed, the location was assigned the salinity class of the nearest classified point (nearest neighbor ID) and reviewed by a regional expert for accuracy. Each estuary and combination of estuary and calendar year was treated as a random intercept. This accounted for nonindependence of samples taken repeatedly from the same system and during the same years, due to unmeasured conditions specific to estuaries or years that were outside the scope of our study. The log-transformed length of the net was treated as an offset to account for differences in sampling intensity due to variable net sizes.

We used a smoother to account for annual trends of species abundance using the day of year as the explanatory variable. Smoothers were constrained to connect day 365 to day 1 and limited to four knots to avoid overfitting the model. The annual timing of Chinook and coho salmon differed among salinity zones, which was consistent with their anadromous life histories. Models describing these species were therefore fit with a unique smoother describing annual trends in abundance for each salinity zone. Chinook salmon and herring captured in San Francisco Bay showed unique trends in annual abundances, and sampling dates were adjusted so that peak abundances were centered to other estuaries. For Chinook salmon, this adjustment was done separately for each salinity zone to be consistent with the parameters in the model. The number of days by which to offset dates was determined by visualizing annual time series of the data via local regression and comparing annual peaks in San Francisco Bay to those for all estuaries combined.

We used multimodel inference to select and estimate the values of parameters that influenced fish abundances. First, we fitted a group of candidate models that included all combinations of model parameters describing estuarine stressors. All candidate models included smoothed day of year and salinity zone parameters because abundances of these fish were well known to vary among seasons and salinities. If fitting a candidate model produced convergence warnings, we first eliminated the random effect of year from the global model, and if warnings persisted we eliminated from consideration the candidate model. From these candidate models, we retained those within 7 AIC of the model with the lowest AIC because this range is likely to include models ranging in support from “substantial” to “considerably less,” but not “essentially none” (Burnham & Anderson, 2002). We then reported estimates of parameters calculated by averaging, when present, their values among the retained candidate models.

If parameter estimates suggested a significant ($p < .05$) effect of an estuarine stressor on fish, we performed a sensitivity analysis and cross-validated our models to examine how well they fit and predicted our data. Given large variance in sampling intensity among estuaries, we tested how sensitive our models were to the exclusion of data from each estuary. We expected parameter estimates of a robust model to change minimally depending on the presence of data from any one estuary. Next, we estimated how well our models fit and predicted the data by fitting models using 80% of the data and examining how well they predicted observations in the remaining 20% (Albouy-Boyer et al., 2016). We randomly selected portions of the data for these training and testing models, and repeated training and testing 10 times for each model to quantify how our estimates of model fit varied due to the random selection procedure. For CPUE models, we compared predicted and observed data using Spearman’s rank correlation coefficient, which ranges from −1 to 1, indicating perfectly negative or positive relationships, respectively, between observed and predicted values. For presence/absence models, we compared predicted and observed data using the true skill statistic (Allouche, Tsoar, & Kadmon, 2006), which ranges in value from −1 to 1, with values less than zero indicating no predictive ability and 1 indicating perfect predictive ability.

When models indicated a significant relationship between stressors and fish, we used their outputs to estimate predicted fish catches among estuaries. These metrics were calculated by making predictions from all candidate models within 7 AIC of the model with the lowest AIC and weighted averaging their results according to their AICs (Bartoń, 2016). These predictions were based on sampling in the mixing zone and on the median day of year sampled for each species. Values were classified using Jenks natural breaks in ArcMap 10.4.1. Modeling analysis was conducted in R version 3.2.2 (R Core Team, 2015) using the mgcv (Wood, 2015) and MuMIn (Bartoń, 2016) packages.

3 | RESULTS

3.1 | Overall data coverage

There were 205,452 individual records of the seven focal species of juvenile fish and invertebrates representing sampling events spanning the years 1990 to 2014 (Table S3). These species occurred broadly across the 20 estuaries (Figure 2), but not every combination of estuary and species was represented. There was a range of four to seven species in each estuary (mean 5.5), and a range of 13 to 19 estuaries with coverage for each species (mean 16). If a species was observed in a given salinity zone in fewer than three estuaries, these observations and thus the parameter estimate for that salinity zone were excluded from analysis.

3.2 | Data analysis

We fitted presence/absence and CPUE models for all species (Figure 3), detecting negative effects of estuarine stressors on Chinook salmon and English sole. The presence and abundance of all species were greatest in the spring and summer (e.g., Figure S1), and the presence and abundance of many species varied among salinity zones (Figure 3). Visualizations of the raw data and information describing candidate models used to select and estimate model parameters are provided in Appendix S1.

The presence of Chinook salmon was significantly lower in estuaries that were more polluted ($p = .011$; Figure 3, Appendix S1). Pollution was present in both candidate models used to estimate parameters, suggesting that pollution was a major, negative influence on the presence of Chinook salmon. No single estuary seemed to drive the negative relationship between pollution and Chinook salmon presence, but this relationship was not statistically significant when 3 of 18 estuaries were individually excluded (those with the most southern extent). Models describing the presence of Chinook salmon from part of our data were able to moderately predict observations of the remaining data (true skill statistic mean ± *SE*: 0.31 ± 0.0028). The presence of most species varied by salinity zones. For example, both anadromous salmon species were less likely to be present in the seawater zone and more likely to be in the tidal fresh zone, whereas English sole, herring, shiner perch, and staghorn sculpin were more likely to be present in the seawater zone.

When fish were present, there were significantly fewer Chinook salmon in estuaries that were more polluted ($p = .040$; Figure 3, Appendix S1) and significantly fewer English sole in estuaries with greater land cover stress ($p = .008$; Figure 3, Appendix S1). There was also a pattern of fewer Chinook salmon in estuaries with greater flow stress, but this trend was marginally insignificant ($p = .057$). Only flow and pollution were present in candidate models used to estimate parameters describing the abundance of Chinook salmon, suggesting that they had major, negative influences on the abundance of this species. No single estuary seemed to drive the negative relationship between pollution and Chinook salmon abundance, but this relationship was not statistically significant when 6 of 18 estuaries were individually excluded (including the same three as for presence/absence models). In addition, our models detected significantly fewer Chinook salmon in estuaries with greater flow stress when data were excluded from six estuaries. Of the five candidate models used to estimate parameters describing abundances of

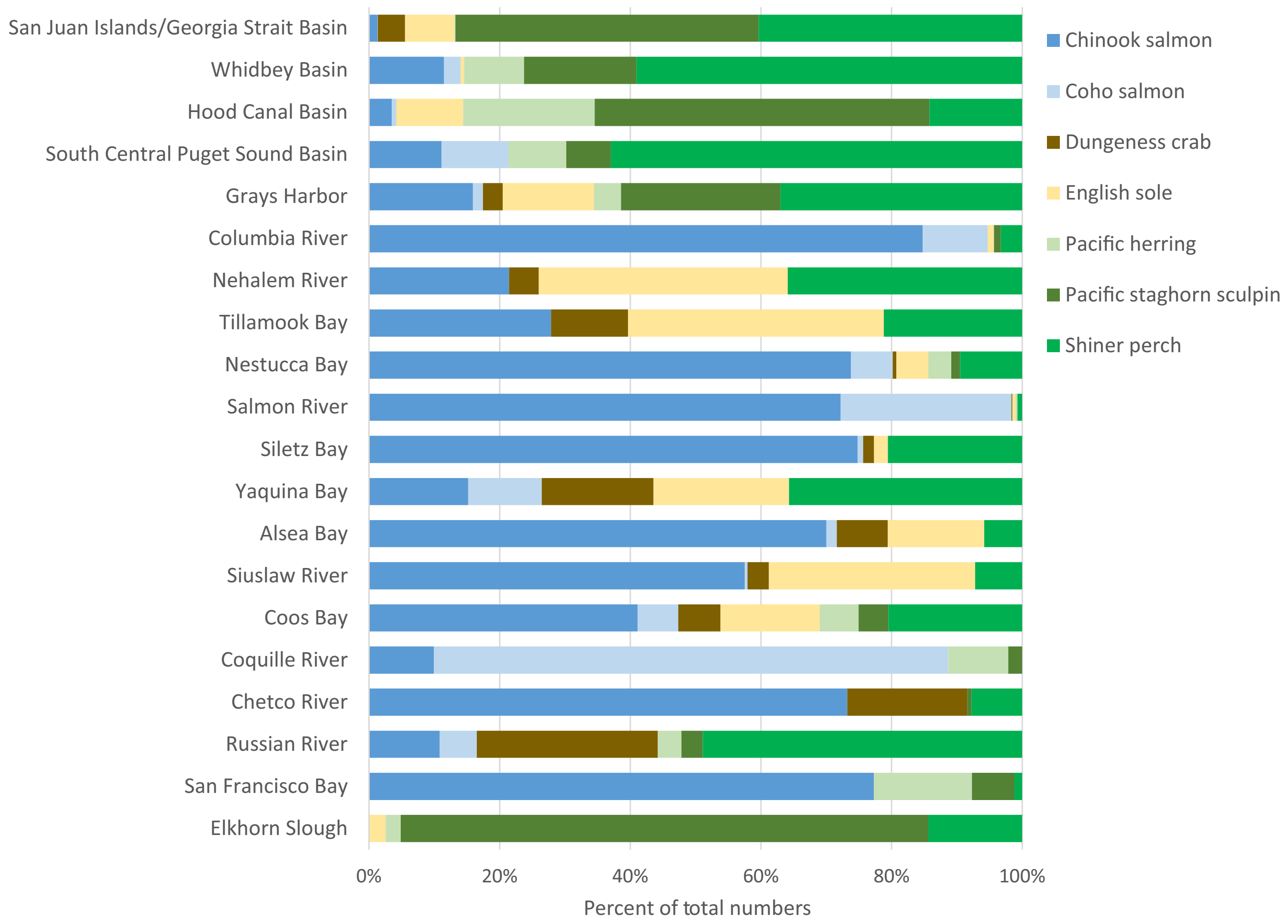

FIGURE 2 Percent composition by abundance of data on focal species in each estuary. Estuaries are sorted descending by latitude from top to bottom [Colour figure can be viewed at wileyonlinelibrary.com]

English sole, land cover was only present in one, indicating significant but limited support for the inclusion of this parameter in the final model. The negative relationship between land cover stress and the abundance of English sole was statistically significant regardless of whether data were excluded from any one estuary. Notably, our models detected significantly more English sole in estuaries with greater flow stress when data were excluded from one estuary. Models from part of our data were moderately able to predict observations of the remaining data describing abundances of Chinook salmon and, to a greater degree, English sole (Spearman's rank correlation coefficient mean ± *SE*: 0.27 ± 0.0053 and 0.46 ± 0.011, respectively). CPUE for four of the species were influenced by salinity, with similar patterns to those for species presence/absence.

We used GAMM parameter estimates and estuary-specific stressor scores to predict catches of the two fishes significantly affected by stressors (Figures 4 and 5; Table S4). The presence and CPUE of Chinook salmon were predicted to be lower among estuaries with greater stress values of pollution. Thus, GAMM outputs demonstrated that estuaries substantially affected by these stressors, such as San Francisco Bay, the South Central basin of Puget Sound, and Coos and Yaquina Bays, would have lower presence and CPUE of Chinook salmon relative to less stressed estuaries, such as Nehalem and Chetco Rivers for presence, and Alsea Bay and the Columbia River for CPUE. Predicted abundances of English sole were less intuitive because, although we detected a significantly negative relationship between their abundances and land cover stress, this relationship had limited support in candidate models. Thus, predictions of English sole CPUE reflected the conditions of these estuaries to all three stressors. Predicted abundances were highest in estuaries with great flow stress and low-to-moderate land cover stress, such as Yaquina and Coos Bays.

4 | DISCUSSION

Here we have provided new, empirical evidence that the effects of stressors from human land-use/cover, pollution, and altered river flow culminate in detectable depressions of some fish species on a coastal scale. Specifically, Chinook salmon were less likely to occur and were less abundant in more polluted estuaries. When present, Chinook salmon also tended to be less abundant in estuaries with greater flow stress, although this trend was marginally insignificant.

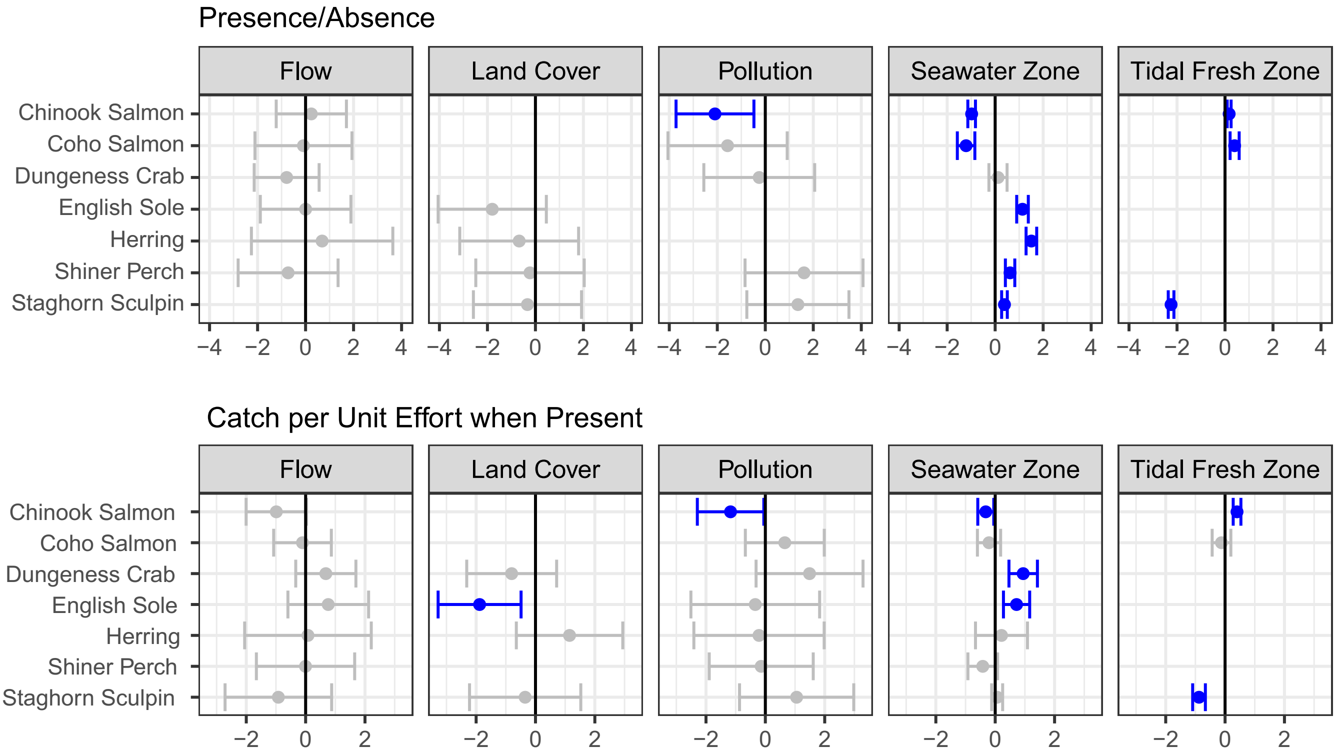

FIGURE 3 Output of GAMMs describing the stressor effect to presence/absence and catch per unit effort of seven focal species. Parameter estimates are indicated by circles and their 95% confidence intervals are indicated by bars. Statistically significant ($p < .05$) terms are highlighted. Parameter estimates of the seawater and tidal fresh zones are reported relative to estimates of the mixing zone (i.e., the mixing zone is the baseline for salinity zone parameters) [Colour figure can be viewed at wileyonlinelibrary.com]

In addition, English sole were less abundant in estuaries with greater land cover stress. Our novel findings build upon regional field studies, experimental manipulation, meta-analyses, models, reviews, and studies focusing on a single stressor or species over broad spatial and temporal components that have shown stressors can threaten fishes (Barceló et al., 2016; Cheng et al., 2015; Crain et al., 2008; Harborne et al., 2017; Hughes et al., 2015; Minello, Able, Weinstein, & Hays, 2003; Munsch, Cordell, & Toft, 2017; Vinebrooke et al., 2004; Vye et al., 2015). Estuaries are increasingly drawing conservation attention because of recognition that they serve as critical habitats for fish (Beck et al., 2001), and we demonstrate on a coast-wide basis that such efforts are for good reason.

The two species that experienced negative stressor effects are known to rely on estuarine habitats and respond on finer scales to habitat degradation. Estuaries are important nursery habitats for outmigrating juvenile Chinook salmon where they forage, acclimatize to marine environments, and avoid predators (Hughes et al., 2014; Munsch, Cordell, & Toft, 2016; Simenstad, Fresh, & Salo, 1982; Weitkamp, Goulette, Hawkes, O'Malley, & Lipsky, 2014). Wild juvenile Chinook salmon use estuarine environments more extensively than hatchery Chinook salmon (Rice et al., 2011; Roegner, McNatt, Teel, & Bottom, 2012), emphasizing the importance of these systems for naturally produced fish. Contaminants can accumulate in tissues of juvenile Chinook salmon (Meador, Yeh, Young, & Gallagher, 2016), and impediments to water flow (e.g., dams) can alter the morphology of nearshore environments and restrict access to critical upriver habitats (O'Connor, Duda, & Grant, 2015). There are also linkages between nursery habitats and stressors to land cover for English sole. For example, juvenile English sole from estuaries contribute more to adult populations than do those from coastal waters (Brown, 2006), and are unable to hide from predators by burrowing where shoreline armoring has displaced soft sediment (Munsch, Cordell, & Toft, 2015; Toft, Cordell, Simenstad, & Stamatiou, 2007). Thus, there are causal explanations that support our observed patterns between stressors and habitat use for Chinook salmon and English sole. Accordingly, managers should consider that flow, land cover, and pollution stressors are limiting estuarine habitat use and, potentially, production of these species.

Perhaps what is most surprising is that we did not detect coast-wide negative effects of stressors on five of the species. One explanation is that some of these species may depend less on estuarine habitats, or that they can use a wider range of habitats, including novel environments created by people (Hobbs et al., 2014). For example, finer-scale effects of land cover (Magnusson & Hilborn, 2003) and pollution (Johnson et al., 2007; Meador et al., 2016) are more detectable in Chinook than co-occurring coho salmon or staghorn sculpin. Juvenile coho salmon typically rear mainly in natal streams and migrate to the ocean as yearlings, and may react more to alterations in freshwater rearing areas (but see Jones, Cornwell, Bottom, Campbell, & Stein, 2014, for estuary-resident life histories).

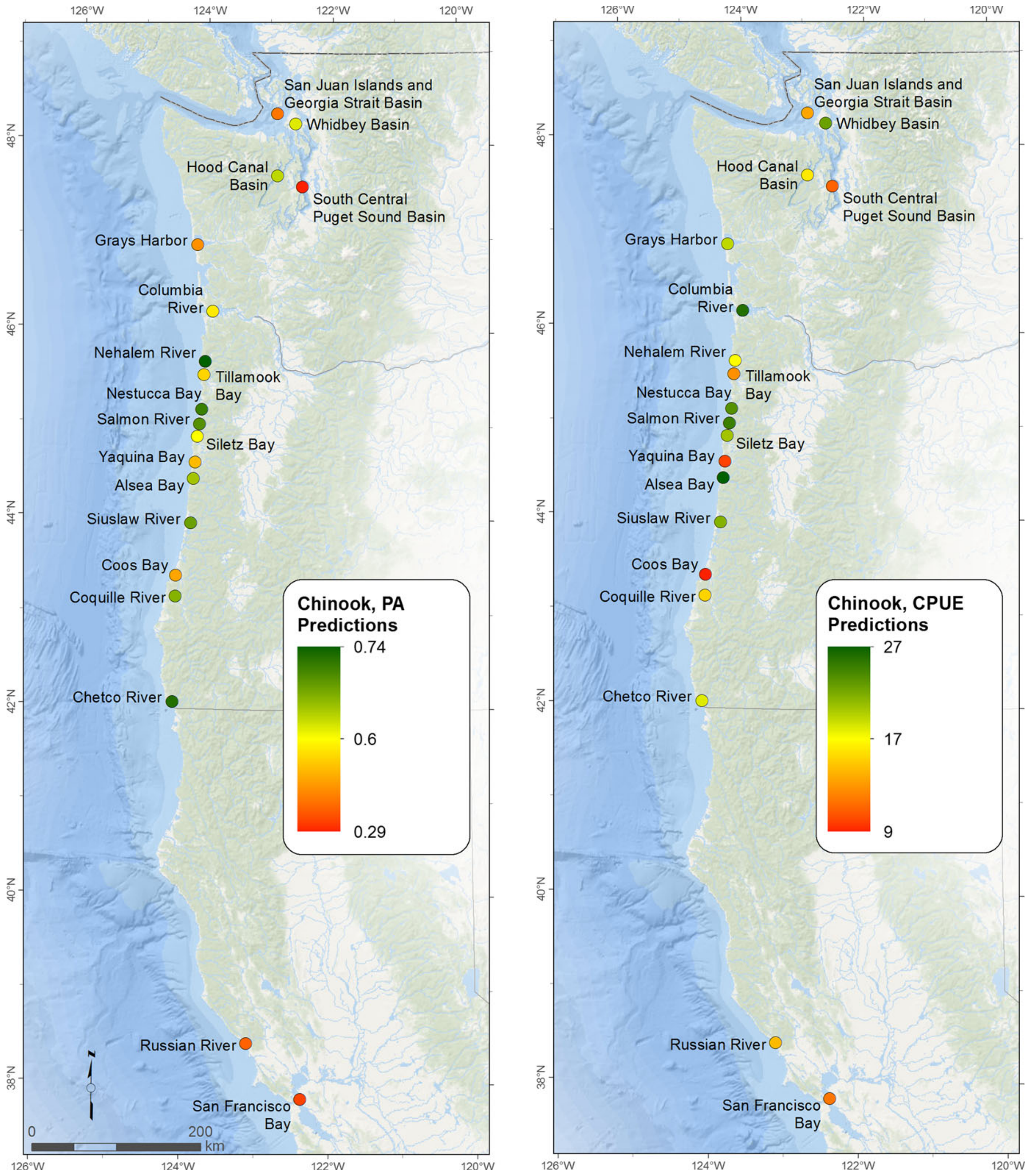

FIGURE 4 Predicted presence and CPUE, when present, of Chinook salmon among estuaries. Values were calculated using GAMM parameter estimates and estuary-specific stressor scores [Colour figure can be viewed at wileyonlinelibrary.com]

Also, documented threats to staghorn sculpin and shiner perch in estuaries are minor (Hughes et al., 2014). Another explanation for lack of trend detection in some species is limitations in our data. Fisheries data, especially when acquired opportunistically, are characterized by low ratios of signal to noise, and imbalanced representation of species, places, and times. We suspect that it is no coincidence that the clearest trends were detected in Chinook salmon, a well-studied species protected under the Endangered Species Act. A

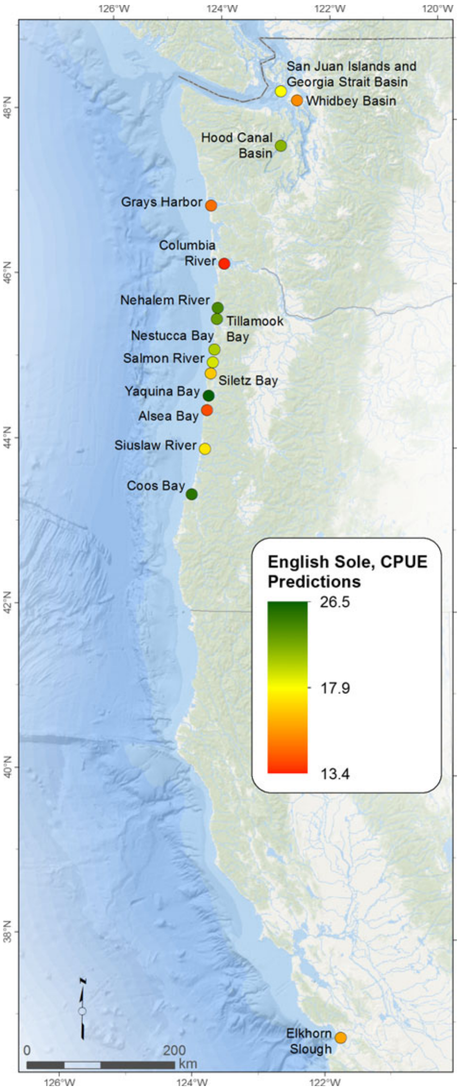

FIGURE 5 Predicted CPUE, when present, of English sole among estuaries. Values were calculated using GAMM parameter estimates and estuary-specific stressor scores [Colour figure can be viewed at wileyonlinelibrary.com]

parsimonious interpretation of our results is that (i) anthropogenic stressors on estuaries probably degrade the habitats of many species, and (ii) we detected trends between specific stressors and species that relied most on habitats altered by stressors and were sampled well enough to produce detectable trends.

Factors outside the scope of our study should also be considered in the interpretation of our results. Although we focused on juvenile stages of fish, adult fish are subject to impacts in coastal zones (Archambault, Rivot, Savina, & Le Pape, in press; Lester et al., 2010), as are eggs and larval development by loss of spawning habitat (Siple & Francis, 2016). Analysis of historic change is another approach, and has shown negative anthropogenic effects on Pacific herring (Greene, Kuehne, Rice, Fresh, & Penttila, 2015). Other environmental factors contribute, such as Dungeness crab fluctuating due to hypoxia caused by anthropogenic nutrient loading in estuaries, and upwelling conditions in open-coast systems (Grantham et al., 2004). We note that the stressors we analyzed correlate with eutrophication (Greene, Blackhart et al., 2015), and have been used in other studies as a proxy for eutrophication (e.g., land cover in Honig, Mahoney, Glanz, & Hughes, 2017), suggesting that stressors such as eutrophication are also contributing factors. Beach seines sample shallow waters, and thus implications are centered on riverine estuaries mainly in Washington and Oregon, in part because species such as Dungeness crab, English sole, and Pacific herring are more apt to occupy colder deeper waters in the southern part of their range in California. Our focus on presence and abundance of seven species also precluded an assemblage, density, or species richness analysis (Courrat et al., 2009; Harley, Myers, & Dunn, 2001; Monaco et al., 1992; Nicolas et al., 2010), which would provide additional information such as how climate-driven processes affect fish community structure (Barceló et al., 2016; Feyrer et al., 2015). All of these examples illustrate that some stressors have more identifiable impacts than others, depending on a myriad of factors that can be estuary, habitat, or species and life-stage specific, and varying on spatial biogeographic and temporal scales.

Independent of stressors, we also demonstrated associations of fish species with salinity zones, which can inform species-specific responses to climate impacts, such as changes in river flow and sea-level rise. These associations are well appreciated among estuarine biologists, but broad landscape patterns have often been based more on expert knowledge (Monaco et al., 1990) than on quantitative information across systems. Our analysis revealed strong associations consistent with contrasting life histories of our focal species: juvenile salmon enter estuaries after migrating downstream from riverine spawning grounds, and so show negative associations with the higher salinity zone as they move offshore away from shallow waters, amid other potential reasons such as mortality as they outmigrate. In contrast, marine fish, such as English sole, spawn in coastal areas, thus associations favor higher salinities. Systematic data on salinity variation would likely highlight even stronger associations, as salinity zones can change dramatically within and between years in response to changes in river hydrographs (Yang & Khangaonkar, 2008) and seasonal closures of estuaries by sand spits (Behrens, Bombardelli, Largier, & Twohy, 2013). Nevertheless, these patterns suggest that long-term impacts of climate change, such as changes in riverine hydrographs and sea-level rise, are an essential consideration for estuarine-dependent species and may alter habitat and species distributions.

Considering our findings, what can we do to promote the sustainability of fish that depend on estuaries during critical life stages? First, we can minimize or continue to ban contaminants, such as polychlorinated biphenyls (PCBs), that persist in riverine and estuarine food webs years after chemical use is discontinued (e.g., West, O'Neill, & Ylitalo, 2017). In particular, Chinook salmon near wastewater treatment plants accumulate many contaminants of emerging concern (Meador et al., 2016), including some that impair liver mitochondrial function (Yeh, Marcinek, Meador, & Gallagher, 2017). Next, we can conserve or restore biogenic fish habitats (e.g., seagrass meadows, oyster reefs) that maintain water quality for coastal systems, but have been degraded due to human activities (Lamb et al., 2017; Lotze et al., 2006). Managing water flow is an ongoing challenge in coastal systems worldwide, and maintaining a balance between human needs (e.g., water consumption and diversion; Cloern & Jassby, 2012), and fish health will be key to sustaining coastal ecosystem services. We can also reduce land cover stress by minimizing impervious surfaces, particularly shoreline armoring that degrades coastal fish habitats (Munsch et al., 2017). Finally, we can conserve or repair lost connectivity across the estuarine landscape. The size of available nursery area can lead to higher estuary production (Rooper, Gunderson, & Armstrong, 2004), and juvenile English sole may rely on different parts of the estuary at different times (Chittaro, Finley, & Levin, 2009), suggesting that they benefit from habitat mosaics that are not fragmented by threats to land cover. Overall, there are many ways that we can improve the quality of estuarine habitats, and our study suggests that certain species, particularly during life stages associated with shallow waters, may respond to conservation efforts that target specific stressors.

Targeting the stressors in our analysis as a management goal will be challenging, especially with continued coastal development in the ecotone between land and water (Halpern et al., 2009). The estuaries predicted to have lower abundance of Chinook salmon have substantial human impacts with numerous drivers of change and corresponding fish declines (e.g., San Francisco Bay and Puget Sound; Emmett, Llanso, Newton, & Thom, 2000; Brown & Moyle, 2005). Furthermore, estuaries with low stressor values and high predicted fish numbers (many on the Oregon coast) do not imply that there are no impacts. For example, flow stress in the Columbia River is represented as low (Greene, Blackhart et al., 2015) because it carries large volumes of water despite documented human impacts and lower flows than historic levels (Bottom et al., 2005). Few estuaries are unaffected by anthropogenic stress, and developing a regional network with management guidelines will be key to coordinating restoration efforts (Merrifield, Hines, Liu, & Beck, 2011).

Examining fish responses to environmental variables is a continually developing field, with many opportunities for refinement (Vasconcelos, Le Pape, Costa, & Cabral, 2013). It can be difficult to isolate effects of individual stressors, which can have interactive and cumulative effects (Crain et al., 2008; Halpern et al., 2009; Kroeker et al., 2016), and may be more easily isolated in purpose-designed (i.e., not opportunistic studies that repurpose data), finer-scale studies. For future studies, conducting fish sampling contemporaneously with updating of stressor measurements would allow for analysis of dynamic responses. Although laborious, such sampling would build upon studies in which robustness is affected by the quality of stressor scores (Greene, Blackhart et al., 2015), submitted fish capture datasets, and the trade-off between data precision and the requisite scaling necessary for large-scale studies (Halpern et al., 2009; Jordan, Smith, & Nestlerode, 2008). At a global level, patterns of biodiversity can be hierarchical from global to local parameters (Pasquaud et al., 2015; Vasconcelos et al., 2015), lending credence to continued examination of scale-dependent responses in stressors and their impacts.

In conclusion, our findings can guide management responses and monitoring programs aimed at reducing human impacts in coastal areas, specifically the effects of land cover, pollution, and flow stressors. Impacts of stressors and restoration actions are two areas of study deserving more attention as to their specific contributions to nursery functions (Munsch et al., 2017). Are there species or life-history types that have adapted to human-induced changes to shoreline and shallow water conditions of estuarine nursery habitats better than others? What anthropogenic modifications have caused the most harm to which species, and do these affect seasonal patterns, for example, causing fish to prematurely disperse from highly impacted systems? How can targeted studies address associated management concerns? Given increasing coastal urban growth and projected sea-level rise, there is great potential for restoration actions that not only enhance shoreline health but also better protect coastal communities using more natural approaches (Allan et al., 2013; Arkema et al., 2013; Shepard, Crain, & Beck, 2011; Toft, Bilkovic, Mitchell, & La Peyre, 2017). Applying these perspectives into management scenarios will be key to maintaining and enhancing sustainable coastlines for fish, as well as the growing human population that relies on these healthy systems for quality of life.

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SUPPORTING INFORMATION

Additional Supporting Information may be found online in the supporting information tab for this article.

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