1

2    MR. JASON D TOFT (Orcid ID : 0000-0002-9879-9126)

3    DR. BRENT B HUGHES (Orcid ID : 0000-0002-2263-1554)

4

5

7

8

9    **Impact of Multiple Stressors on Juvenile Fish in Estuaries of the Northeast Pacific**

10

11    **Running Head:** Multiple stressors impact estuarine fishes

12

13    Jason D. Toft[1,*], Stuart H. Munsch[1,2], Jeffery R. Cordell[1], Kiira Siitari[3], Van C. Hare[3], Brett

14    Holycross[3], Lisa A. DeBruyckere[4], Correigh M. Greene[2], and Brent B. Hughes[5,6]

15

16    [1] School of Aquatic and Fishery Sciences, University of Washington, Seattle, WA, USA

17    [2] Northwest Fisheries Science Center, National Marine Fisheries Service, Seattle, WA, USA

18    [3] Pacific States Marine Fisheries Commission, Portland, OR, USA

19    [4] Pacific Marine and Estuarine Fish Habitat Partnership, Salem, OR, USA

20    [5] Institute of Marine Sciences, University of California at Santa Cruz, CA, USA

21    [6] Nicholas School of the Environment, Duke Marine Lab, Duke University, Beaufort, NC, USA

22    *Corresponding author*

23      Phone: 1 206 221 5460

24      Fax: 1 206 685-7471

25      E-mail: tofty@u.washington.edu

26

29

**Paper Type:** Primary Research Article

# 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 1600 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.

## 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 *et al.*, 2015), meta-analyses (e.g., synthesizing interactive and cumulative effects of stressors; Crain *et al.,* 2008), conceptual and theoretical models (e.g., Vinebrooke *et al.,* 2004), and reviews (e.g., syntheses of multiple stressors on coral reefs; Harborne *et al.*, 2017). Studies that examine broad temporal and spatial scales tend to be conducted in freshwater or terrestrial ecosystems (e.g., Esselman *et al.*, 2011; Ceballos *et al.*, 2017), emphasize single species across larger geographic gradients (e.g., Cheng et al., 2015) or a single stressor across longer temporal gradients (e.g., Barceló *et al.,* 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 (Kennish, 2002; Airoldi & Beck, 2007; USEPA, 2012; Greene *et al.,* 2015b). 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 (Vasconcelos *et al.,* 2007; Kroeker *et al.,* 2016; Lefcheck *et al.*, 2017). 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

84  habitat mosaics of coastal systems, and hypothesized that shallow habitats contribute

85  disproportionally to adult populations (Beck *et al.,* 2001; Dahlgren *et al.,* 2006; Nagelkerken *et*

86  *al.,* 2015). Thus, it is advantageous to investigate effects of stressors on fish in coastal systems

87  because (1) juveniles rely on these systems to provide critical nursery habitats, and (2) these

88  systems are focal areas for conservation and management (Beck *et al.,* 2001).

89  To examine the role of multiple stressors across northeast Pacific coast estuaries (U.S.

90  states of California, Oregon, and Washington), we focused on seven fish and invertebrate

91  species, a subset with ample available catch data of 15 focal species previously identified by

92  Hughes *et al.* (2014). These species represent major guilds, are of commercial, recreational, and

93  cultural importance, and have life histories that encompass large portions of northeast Pacific

94  estuaries, spanning more than 1600 km of coastline. Five of these have management targets

95  through their listing under the U.S. Endangered Species Act and/or because they are important

96  fisheries species – Chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*Oncorhynchus*

97  *kisutch*), Dungeness crab (*Cancer magister*), English sole (*Parophrys vetulus*), and Pacific

98  herring (*Clupea pallasii*). Two of the species, though not tied to such targets – shiner perch

99  (*Cymatogaster aggregata*) and Pacific staghorn sculpin (*Leptocottus armatus*) – can play

100 important ecological roles in estuarine food-webs as secondary consumers (Hughes *et al.,* 2014).

101 Our assessment builds on previous syntheses of Pacific coast estuaries (Monaco *et al.,*

102 1990; Emmett *et al.,* 1991; Monaco *et al.,* 1992; Gleason *et al.,* 2011; Hughes *et al.,* 2014;

103 Heady *et al.,* 2014) by combining location-specific fish sampling with published values of

104 stressor scores indicating levels of human impact (below, from Greene *et al.,* 2015b). We

105 focused on three different classes of stressors, representing (1) land cover/land use (e.g., amount

106 developed, change from natural estuarine classes), (2) alteration of fluvial processes (e.g.,

107 changes in patterns of flow in rivers entering estuaries), and (3) sources of pollution (e.g., toxic

108 releases and pollution discharges). The large-scale stressors that we examined include habitat

109 alterations that impact fish populations worldwide (Bilkovic & Roggero, 2008; Able *et al.,* 2013;

110 Valesini *et al.,* 2014) due to growing human populations dependent on coastal ecosystem

111 services (Kennish, 2002; UNEP, 2006).

112    Our approach was to compile raw data from beach seine collections in shallow waters to

113  examine effects of stressors on presence and abundance of the seven focal species. Our

114  hypothesis was that there would be lower presence and abundance of fish in estuaries with higher

115  stressor scores for species that were most estuarine-dependent and/or negatively affected by

116  human impacts.


117  **Materials and Methods**


118  *Data Collection*

119  We used the network of coastal scientists and managers associated with the Pacific Marine and

120  Estuarine Fish Habitat Partnership to generate a pool of potential data holders of nearshore and

121  estuarine fish. From this pool, 120 managers and researchers responded to an online survey,

122  providing descriptions of the scope and availability of their data. This contact list expanded with

123  additional professional referrals and estuary-specific queries of federal and state data portals to

124  fill geographic gaps. Ultimately, more than 200 individuals from 73 different agencies were

125  contacted regarding the data request. Beach seine data, encompassing the years 1990 to 2014,

126  were combined from 22 sampling programs representing 20 estuaries (Fig. 1; Table S1). We

127  prioritized datasets that sampled multiple species and specified life-stage or length parameters.

128  Submitted datasets were compiled and uploaded to a Microsoft Access database, based on an

129  observations data model that provided a consistent format for the storage and retrieval of point

130  observations in a relational database. This approach is designed to facilitate an integrated

131  analysis of large datasets collected by multiple investigators (Horsburgh *et al*., 2008). When

132  geographic coordinates were unavailable, fish observations were resolved to an estuary polygon,

133  if metadata allowed. Prior to analysis, data were queried by species, life-stage (when available),

134  and location to include only samples collected within U.S. Pacific estuary boundaries using

135  current National Oceanic and Atmospheric Administration (NOAA) designations.


136    Throughout our data collection and analysis, we observed sources of data variability and

137  constraints that limited the extent to which we could apply our analyses. Some estuaries were

138  less sampled than others, and not all available datasets were submitted. Therefore, lack of data

139  from one estuary did not necessarily indicate that data does not exist; it could also mean that the

140    data were simply not available. Data coverage within each estuary varied, as did the precision of

141    specific measurements of latitude and longitude where sampling occurred. Also, non-target

142    species from a study focusing on one group (e.g., salmonids) were often binned to general levels

143    (e.g., flatfish, cancer crabs), and in those cases, could not be incorporated into our species-level

144    assessment. Recent advances in quantitative ecology allowed us to integrate, reduce, and analyze

145    large scale datasets from varying sources, recognizing sources of variability such as differences

146    in experimental sampling, model parameters, and functional differences in underlying biological

147    processes (Thorson *et al.,* 2013). Our analysis path was chosen to best account for the variability

148    and constraints that are expected when combining data from different research groups.


149    *Data Analysis*

150    We compared fish abundances with anthropogenic impacts on estuaries, using the stressor scores

151    calculated for each estuary by the 2010 National Assessment (Greene *et al.,* 2015b) (Table 1)

152    which included habitat alterations expected to impact fish populations. Stressor scores in Greene

153    *et al.* (2015b) combined 43 indicator datasets into four categories: (1) land cover/land use, (2)

154    alteration of river flows, (3) pollution sources, and (4) eutrophication (Table S2). Of these, the

155    first three categories had suitable coverage for the 20 estuaries in our study and were used in

156    subsequent analyses; eutrophication was not included because it had the greatest number of data

157    gaps and data were not available for four of the estuaries in this analysis (Greene *et al.,* 2015b).

158    Pollutants, river flow, and land use all showed correlations with eutrophication as summarized in

159    Greene *et al.,* (2015b), and this can be used to infer links to eutrophication as a stressor.


160        The remaining three metrics were themselves indices of multiple measures (Table S2).

161    Land use/land cover datasets evaluated recent areal land cover of agriculture, development, and

162    estuary habitat as well as changes in these classes during the last 20 years. River flow datasets

163    integrated numerous flow metrics, including the number of dams per watershed area as well as

164    indicators of hydrologic alteration (Richter *et al.,* 1996) from United Stations Geological Survey

165    (USGS) river flow gages closest to head of tide, using 15-year averages of mean discharge,

166    maximum and minimum discharge, high and low pulse duration, as well as the trend in these

167    values across the entire time series. Pollution datasets included the number of mines, toxic

168    release sites, pollution discharge sites, and hazardous waste sites per unit watershed area. Values

169 of each stressor were scaled 0 to 1 based on the ranks of individual datasets across estuary

170 systems in the National Assessment, with higher values indicating greater human impacts

171 (Greene *et al.,* 2015b). The land cover stressor index included amount of land developed and

172 change from natural estuarine classes, thereby acknowledging alterations from pristine states

173 unique to each estuary. These stressors are linked to fish habitat quality and quantity. For

174 example, development described by stress to land cover reduces the amount of estuarine nursery

175 habitat available to juvenile fish, pollution degrades water quality, and alteration of river flows

176 changes downstream hydrology of estuaries. Our analysis was constructed to evaluate these

177 associations across systems and taxa.

178 Several estuaries with fish data required interpretation of stressor scores from nearby sites

179 or from newly acquired data. Monterey Bay values for flow were used in Elkhorn Slough, South

180 and Central Bay values were averaged for San Francisco Bay, Drayton Bay values were used for

181 San Juan Islands and Georgia Strait Basin, and Puget Sound values were used for South Central

182 Puget Sound Basin. We estimated one value describing flow stress in the Salmon River, Oregon,

183 by averaging the neighboring Nestucca and Siletz Rivers, which were similar in geographic

184 location and flow stress values ($\approx$13km to the north and south, respectively, values of 0.423 and

185 0.311, average value of 0.367). We calculated flow stressor values for Coos Bay, Oregon and

186 Yaquina Bay, Oregon using data made available after the analysis of Greene *et al.* (2015b).

187 We quantified the relationship between species abundance and estuarine stressors using

188 generalized additive mixed models (GAMMs; Zuur *et al.,* 2009; Wood, 2011). This modeling

189 approach was appropriate because fish catches were non-normally distributed, there were non-

190 linear annual trends in species abundance, and species were repeatedly sampled from the same

191 estuaries and years. As is typical in fisheries investigations, the data were zero-inflated and we

192 therefore fit two models for each species: one describing presence or absence of a species

193 (hereafter: presence/absence model) and the other describing catch per unit effort (CPUE) when

194 a species was present (hereafter: CPUE model). Presence/absence models were fit using a

195 binomial distribution and a logit-link function. CPUE models were fit using a negative binomial

196 distribution and log-link function. We used a negative binomial distribution rather than a Poisson

197 distribution because, as often occurs in ecological data, the variance of our count data exceeded

198 its mean (Zuur *et al*., 2009).

199    The fixed effects considered for the models were flow, land cover, and pollution

200    stressors, and salinity zone. We used salinity zones of freshwater tidal (<0.5 ppt), mixing (0.5–25

201    ppt), and seawater (>25 ppt). Salinity zones were assigned based on averaged field

202    measurements from a location. If salinity data were unavailable, we used the NOAA 3-Zone

203    Average Annual Salinity Digital Geography layer (spatial join w/ 50m buffer). If no field data or

204    NOAA digital data existed, the location was assigned the salinity class of the nearest classified

205    point (nearest neighbor ID) and reviewed by a regional expert for accuracy. Each estuary and

206    combination of estuary and calendar year was treated as a random intercept. This accounted for

207    non-independence of samples taken repeatedly from the same system and during the same years,

208    due to unmeasured conditions specific to estuaries or years that were outside the scope of our

209    study. The log-transformed length of the net was treated as an offset to account for differences in

210    sampling intensity due to variable net sizes.

211    We used a smoother to account for annual trends of species abundance using the day of

212    year as the explanatory variable. Smoothers were constrained to connect day 365 to day 1 and

213    limited to four knots to avoid overfitting the model. The annual timing of Chinook and coho

214    salmon differed among salinity zones, which was consistent with their anadromous life histories.

215    Models describing these species were therefore fit with a unique smoother describing annual

216    trends in abundance for each salinity zone. Chinook salmon and herring captured in San

217    Francisco Bay showed unique trends in annual abundances, and sampling dates were adjusted so

218    that peak abundances were centered to other estuaries. For Chinook salmon, this adjustment was

219    done separately for each salinity zone to be consistent with the parameters in the model. The

220    number of days by which to offset dates was determined by visualizing annual time series of the

221    data via local regression and comparing annual peaks in San Francisco Bay to those for all

222    estuaries combined.

223    We used multi-model inference to select and estimate the values of parameters that

224    influenced fish abundances. First, we fitted a group of candidate models that included all

225    combinations of model parameters describing estuarine stressors. All candidate models included

226    smoothed day of year and salinity zone parameters because abundances of these fish were well

227    known to vary among seasons and salinities. If fitting a candidate model produced convergence

228    warnings, we first eliminated the random effect of year from the global model, and if warnings

229  persisted we eliminated from consideration the candidate model. From these candidate models,

230  we retained those within 7 AIC of the model with the lowest AIC because this range is likely to

231  include models ranging in support from "substantial" to "considerably less," but not "essentially

232  none" (Burnham & Anderson, 2002). We then reported estimates of parameters calculated by

233  averaging, when present, their values among the retained candidate models.

234      If parameter estimates suggested a significant ($P < 0.05$) effect of an estuarine stressor on

235  fish, we performed a sensitivity analysis and cross-validated our models to examine how well

236  they fit and predicted our data. Given large variance in sampling intensity among estuaries, we

237  tested how sensitive our models were to the exclusion of data from each estuary. We expected

238  parameter estimates of a robust model to change minimally depending on the presence of data

239  from any one estuary. Next, we estimated how well our models fit and predicted the data by

240  fitting models using 80% of the data and examining how well they predicted observations in the

241  remaining 20% (Albouy-Boyer *et al.,* 2016). We randomly selected portions of the data for these

242  training and testing models, and repeated training and testing 10 times for each model to quantify

243  how our estimates of model fit varied due to the random selection procedure. For CPUE models,

244  we compared predicted and observed data using Spearman's rank correlation coefficient, which

245  ranges from -1 to 1, indicating perfectly negative or positive relationships, respectively, between

246  observed and predicted values. For presence/absence models, we compared predicted and

247  observed data using the true skill statistic (Allouche *et al*., 2006), which ranges in value from -1

248  to 1, with values less than zero indicating no predictive ability and 1 indicating perfect predictive

249  ability.

250      When models indicated a significant relationship between stressors and fish, we used

251  their outputs to estimate predicted fish catches among estuaries. These metrics were calculated

252  by making predictions from all candidate models within 7 AIC of the model with the lowest AIC

253  and weighted averaging their results according to their AICs (Bartoń, 2016). These predictions

254  were based on sampling in the mixing zone and on the median day of year sampled for each

255  species. Values were classified using Jenks natural breaks in ArcMap 10.4.1. Modeling analysis

256  was conducted in R version 3.2.2 (R Core Team, 2015) using the mgcv (Wood, 2015) and

257  MuMIn (Bartoń, 2016) packages.

258 **Results**

259 *Overall data coverage*

260 There were 205,452 individual records of the seven focal species of juvenile fish and
261 invertebrates representing sampling events spanning the years 1990 to 2014 (Table S3). These
262 species occurred broadly across the 20 estuaries (Fig. 2), but not every combination of estuary
263 and species was represented. There was a range of four to seven species in each estuary (mean
264 5.5), and a range of 13 to 19 estuaries with coverage for each species (mean 16). If a species was
265 observed in a given salinity zone in fewer than three estuaries, these observations and thus the
266 parameter estimate for that salinity zone were excluded from analysis.

267 *Data Analysis*

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

274 The presence of Chinook salmon was significantly lower in estuaries that were more
275 polluted ($P = 0.011$; Fig. 3, Appendix S1). Pollution was present in both candidate models used
276 to estimate parameters, suggesting that pollution was a major, negative influence on the presence
277 of Chinook salmon. No single estuary seemed to drive the negative relationship between
278 pollution and Chinook salmon presence, but this relationship was not statistically significant
279 when three of 18 estuaries were individually excluded (those with the most southern extent).
280 Models describing the presence of Chinook salmon from part of our data were able to
281 moderately predict observations of the remaining data (true skill statistic mean $\pm$ *SE*: 0.31 $\pm$
282 0.0028). The presence of most species varied by salinity zones. For example, both anadromous
283 salmon species were less likely to be present in the seawater zone and more likely to be in the
284 tidal fresh zone, whereas English sole, herring, shiner perch, and staghorn sculpin were more
285 likely to be present in the seawater zone.

286    When fish were present, there were significantly fewer Chinook salmon in estuaries that

287    were more polluted ($P = 0.040$; Fig. 3, Appendix S1) and significantly fewer English sole in

288    estuaries with greater land cover stress ($P = 0.008$; Fig. 3, Appendix S1). There was also a

289    pattern of fewer Chinook salmon in estuaries with greater flow stress, but this trend was

290    marginally insignificant ($P = 0.057$). Only flow and pollution were present in candidate models

291    used to estimate parameters describing the abundance of Chinook salmon, suggesting that they

292    had major, negative influences on the abundance of this species. No single estuary seemed to

293    drive the negative relationship between pollution and Chinook salmon abundance, but this

294    relationship was not statistically significant when six of 18 estuaries were individually excluded

295    (including the same three as for presence/absence models). In addition, our models detected

296    significantly fewer Chinook salmon in estuaries with greater flow stress when data were

297    excluded from six estuaries. Of the five candidate models used to estimate parameters describing

298    abundances of English sole, land cover was only present in one, indicating significant but limited

299    support for the inclusion of this parameter in the final model. The negative relationship between

300    land cover stress and the abundance of English sole was statistically significant regardless of

301    whether data were excluded from any one estuary. Notably, our models detected significantly

302    more English sole in estuaries with greater flow stress when data were excluded from one

303    estuary. Models from part of our data were moderately able to predict observations of the

304    remaining data describing abundances of Chinook salmon and, to a greater degree, English sole

305    (Spearman's rank correlation coefficient mean ± $SE$: $0.27 \pm 0.0053$ and $0.46 \pm 0.011$,

306    respectively). CPUE for four of the species were influenced by salinity, with similar patterns to

307    those for species presence/absence.

308    We used GAMM parameter estimates and estuary-specific stressor scores to predict

309    catches of the two fishes significantly affected by stressors (Figs. 4 and 5; Table S4). The

310    presence and CPUE of Chinook salmon were predicted to be lower among estuaries with greater

311    stress values of pollution. Thus, GAMM outputs demonstrated that estuaries substantially

312    affected by these stressors, such as San Francisco Bay, the South Central basin of Puget Sound,

313    and Coos and Yaquina Bays, would have lower presence and CPUE of Chinook salmon relative

314    to less stressed estuaries, such as Nehalem and Chetco Rivers for presence, and Alsea Bay and

315    the Columbia River for CPUE. Predicted abundances of English sole were less intuitive because,

316    although we detected a significantly negative relationship between their abundances and land

317 cover stress, this relationship had limited support in candidate models. Thus, predictions of

318 English sole CPUE reflected the conditions of these estuaries to all three stressors. Predicted

319 abundances were highest in estuaries with great flow stress and low to moderate land cover

320 stress, such as Yaquina and Coos Bays.

321 **Discussion**

322 Here we have provided new, empirical evidence that the effects of stressors from human land-

323 use/cover, pollution, and altered river flow culminate in detectable depressions of some fish

324 species on a coastal scale. Specifically, Chinook salmon were less likely to occur and were less

325 abundant in more polluted estuaries. When present, Chinook salmon also tended to be less

326 abundant in estuaries with greater flow stress, although this trend was marginally insignificant.

327 In addition, English sole were less abundant in estuaries with greater land cover stress. Our novel

328 findings build upon regional field studies, experimental manipulation, meta-analyses, models,

329 reviews, and studies focusing on a single stressor or species over broad spatial and temporal

330 components that have shown stressors can threaten fishes (Minello *et al.,* 2003; Vinebrooke *et*

331 *al.,* 2004; Crain *et al.,* 2008; Cheng *et al*., 2015; Hughes *et al.,* 2015; Vye *et al.*, 2015; Barceló *et*

332 *al.,* 2016; Harborne *et al*., 2017; Munsch *et al*., 2017). Estuaries are increasingly drawing

333 conservation attention because of recognition that they serve as critical habitats for fish (Beck *et*

334 *al.,* 2001), and we demonstrate on a coast-wide basis that such efforts are for good reason.

335 The two species that experienced negative stressor effects are known to rely on estuarine

336 habitats and respond on finer scales to habitat degradation. Estuaries are important nursery

337 habitats for outmigrating juvenile Chinook salmon where they forage, acclimatize to marine

338 environments, and avoid predators (Simenstad *et al.,* 1982; Weitkamp *et al.,* 2014; Hughes *et al.*,

339 2014; Munsch *et al.*, 2016). Wild juvenile Chinook salmon use estuarine environments more

340 extensively than hatchery Chinook salmon (Rice *et al.,* 2011; Roegner *et al.,* 2012), emphasizing

341 the importance of these systems for naturally-produced fish. Contaminants can accumulate in

342 tissues of juvenile Chinook salmon (Meador *et al.,* 2016), and impediments to water flow (e.g.,

343 dams) can alter the morphology of nearshore environments and restrict access to critical upriver

344 habitats (O'Connor *et al.,* 2015). There are also linkages between nursery habitats and stressors

345 to land cover for English sole. For example, juvenile English sole from estuaries contribute more

346    to adult populations than do those from coastal waters (Brown, 2006), and are unable to hide

347    from predators by burrowing where shoreline armoring has displaced soft sediment (Toft *et al.,*

348    2007; Munsch *et al.,* 2015). Thus, there are causal explanations that support our observed

349    patterns between stressors and habitat use for Chinook salmon and English sole. Accordingly,

350    managers should consider that flow, land cover, and pollution stressors are limiting estuarine

351    habitat use and, potentially, production of these species.

352          Perhaps what is most surprising is that we did not detect coast-wide negative effects of

353    stressors on five of the species. One explanation is that some of these species may depend less on

354    estuarine habitats, or that they can use a wider range of habitats, including novel environments

355    created by people (Hobbs *et al.,* 2014). For example, finer-scale effects of land cover

356    (Magnusson and Hilborn, 2003) and pollution (Johnson *et al.,* 2007; Meador *et al.,* 2016) are

357    more detectable in Chinook than co-occurring coho salmon or staghorn sculpin. Juvenile coho

358    salmon typically rear mainly in natal streams and migrate to the ocean as yearlings, and may

359    react more to alterations in freshwater rearing areas (but see Jones *et al.,* 2014 for estuary-

360    resident life histories). Also, documented threats to staghorn sculpin and shiner perch in estuaries

361    are minor (Hughes *et al.,* 2014). Another explanation for lack of trend detection in some species

362    is limitations in our data. Fisheries data, especially when acquired opportunistically, are

363    characterized by low ratios of signal to noise, and imbalanced representation of species, places,

364    and times. We suspect that it is no coincidence that the clearest trends were detected in Chinook

365    salmon, a well-studied species protected under the Endangered Species Act. A parsimonious

366    interpretation of our results is that (1) anthropogenic stressors on estuaries probably degrade the

367    habitats of many species, and (2) we detected trends between specific stressors and species that

368    relied most on habitats altered by stressors *and* were sampled well enough to produce detectable

369    trends.

370          Factors outside the scope of our study should also be considered in the interpretation of

371    our results. Although we focused on juvenile stages of fish, adult fish are subject to impacts in

372    coastal zones (Lester *et al.,* 2010; Archambault *et al.,* in press), as are eggs and larval

373    development by loss of spawning habitat (Siple & Francis, 2016). Analysis of historic change is

374    another approach, and has shown negative anthropogenic effects on Pacific herring (Greene *et*

375    *al.,* 2015a). Other environmental factors contribute, such as Dungeness crab fluctuating due to

376   hypoxia caused by anthropogenic nutrient loading in estuaries, and upwelling conditions in open-

377   coast systems (Grantham *et al.,* 2004). We note that the stressors we analyzed correlate with

378   eutrophication (Greene *et al.,* 2015b), and have been used in other studies as a proxy for

379   eutrophication (e.g., land-cover in Honig *et al.,* 2017), suggesting that stressors such as

380   eutrophication are also contributing factors. Beach seines sample shallow waters, and thus

381   implications are centered on riverine estuaries mainly in Washington and Oregon, in part because

382   species such as Dungeness crab, English sole, and Pacific herring are more apt to occupy colder

383   deeper waters in the southern part of their range in California. Our focus on presence and

384   abundance of seven species also precluded an assemblage, density, or species richness analysis

385   (Monaco *et al.,* 1992; Harley *et al.,* 2001; Courrat *et al.,* 2009; Nicolas *et al.,* 2010), which

386   would provide additional information such as how climate-driven processes affect fish

387   community structure (Feyrer *et al.,* 2015; Barceló *et al.,* 2016). All of these examples illustrate

388   that some stressors have more identifiable impacts than others, depending on a myriad of factors

389   that can be estuary, habitat, or species and life-stage specific, and varying on spatial

390   biogeographic and temporal scales.

391        Independent of stressors, we also demonstrated associations of fish species with salinity

392   zones, which can inform species-specific responses to climate impacts, such as changes in river

393   flow and sea level rise. These associations are well appreciated among estuarine biologists, but

394   broad landscape patterns have often been based more on expert knowledge (Monaco *et al.*, 1990)

395   than on quantitative information across systems. Our analysis revealed strong associations

396   consistent with contrasting life histories of our focal species: juvenile salmon enter estuaries after

397   migrating downstream from riverine spawning grounds, and so show negative associations with

398   the higher salinity zone as they move offshore away from shallow waters, amid other potential

399   reasons such as mortality as they out-migrate. In contrast, marine fish, such as English sole,

400   spawn in coastal areas, thus associations favor higher salinities. Systematic data on salinity

401   variation would likely highlight even stronger associations, as salinity zones can change

402   dramatically within and between years in response to changes in river hydrographs (Yang &

403   Khangaonkar, 2008) and seasonal closures of estuaries by sand spits (Behrens *et al.*, 2013).

404   Nevertheless, these patterns suggest that long-term impacts of climate change, such as changes in

405   riverine hydrographs and sea level rise, are an essential consideration for estuarine-dependent

406   species and may alter habitat and species distributions.

407        Considering our findings, what can we do to promote the sustainability of fish that

408    depend on estuaries during critical life-stages? First, we can minimize or continue to ban

409    contaminants, such as polychlorinated biphenyls (PCBs) that persist in riverine and estuarine

410    food webs years after chemical use is discontinued (e.g., West *et al.,* 2017). In particular,

411    Chinook salmon near wastewater treatment plants accumulate many contaminants of emerging

412    concern (Meador *et al.,* 2016), including some that impair liver mitochondrial function (Yeh *et*

413    *al.,* 2017). Next, we can conserve or restore biogenic fish habitats (e.g., seagrass meadows,

414    oyster reefs) that maintain water quality for coastal systems, but have been degraded due to

415    human activities (Lotze *et al.,* 2006; Lamb *et al.,* 2017). Managing water flow is an ongoing

416    challenge in coastal systems worldwide, and maintaining a balance between human needs (e.g.,

417    water consumption and diversion; Cloern & Jassby, 2012), and fish health will be key to

418    sustaining coastal ecosystem services. We can also reduce land cover stress by minimizing

419    impervious surfaces, particularly shoreline armoring that degrades coastal fish habitats (Munsch

420    *et al.*, 2017). Finally, we can conserve or repair lost connectivity across the estuarine landscape.

421    The size of available nursery area can lead to higher estuary production (Rooper *et al.,* 2004),

422    and juvenile English sole may rely on different parts of the estuary at different times (Chittaro *et*

423    *al.,* 2009), suggesting that they benefit from habitat mosaics that are not fragmented by threats to

424    land cover. Overall, there are many ways that we can improve the quality of estuarine habitats,

425    and our study suggests that certain species, particularly during life-stages associated with

426    shallow waters, may respond to conservation efforts that target specific stressors.

427        Targeting the stressors in our analysis as a management goal will be challenging,

428    especially with continued coastal development in the ecotone between land and water (Halpern

429    *et al.,* 2009). The estuaries predicted to have lower abundance of Chinook salmon have

430    substantial human impacts with numerous drivers of change and corresponding fish declines

431    (e.g., San Francisco Bay and Puget Sound; Emmett *et al.,* 2000; Brown & Moyle, 2005).

432    Furthermore, estuaries with low stressor values and high predicted fish numbers (many on the

433    Oregon coast) does not imply that there are no impacts. For example, flow stress in the Columbia

434    River is represented as low (Greene *et al.,* 2015b) because it carries large volumes of water

435    despite documented human impacts and lower flows than historic levels (Bottom *et al.,* 2005).

436    Few estuaries are unaffected by anthropogenic stress, and developing a regional network with

437    management guidelines will be key to coordinating restoration efforts (Merrifield *et al.,* 2011).

438    Examining fish responses to environmental variables is a continually developing field,

439    with many opportunities for refinement (Vasconcelos *et al.,* 2013). It can be difficult to isolate

440    effects of individual stressors, which can have interactive and cumulative effects (Crain *et al.,*

441    2008; Halpern *et al.,* 2009; Kroeker *et al.,* 2016), and may be more easily isolated in purpose-

442    designed (i.e., not opportunistic studies that repurpose data), finer-scale studies. For future

443    studies, conducting fish sampling contemporaneously with updating of stressor measurements

444    would allow for analysis of dynamic responses. Although laborious, such sampling would build

445    upon studies in which robustness is affected by the quality of stressor scores (Greene *et al.,*

446    2015b), submitted fish capture datasets, and the trade-off between data precision and the

447    requisite scaling necessary for large-scale studies (Jordan *et al.,* 2008; Halpern *et al.,* 2009). At a

448    global level, patterns of biodiversity can be hierarchical from global to local parameters

449    (Vasconcelos *et al.,* 2015; Pasquaud *et al.,* 2015), lending credence to continued examination of

450    scale-dependent responses in stressors and their impacts.

451    In conclusion, our findings can guide management responses and monitoring programs

452    aimed at reducing human impacts in coastal areas, specifically the effects of land cover,

453    pollution, and flow stressors. Impacts of stressors and restoration actions are two areas of study

454    deserving more attention as to their specific contributions to nursery functions (Munsch *et al.*,

455    2017). Are there species or life history types that have adapted to human-induced changes to

456    shoreline and shallow water conditions of estuarine nursery habitats better than others? What

457    anthropogenic modifications have caused the most harm to which species, and do these affect

458    seasonal patterns, for example causing fish to prematurely disperse from highly impacted

459    systems? How can targeted studies address associated management concerns? Given increasing

460    coastal urban growth and projected sea level rise, there is great potential for restoration actions

461    that not only enhance shoreline health, but also better protect coastal communities using more

462    natural approaches (Shepard *et al.,* 2011; Allan *et al.,* 2013; Arkema *et al.,* 2013; Toft *et al.*,

463    2017). Applying these perspectives into management scenarios will be key to maintaining and

464    enhancing sustainable coastlines for fish, as well as the growing human population that relies on

465    these healthy systems for quality of life.

## Acknowledgements

## References

Able KW Grothues TM Kemp IM (2013) Fine-scale distribution of pelagic fishes relative to a large urban pier. *Marine Ecology Progress Series*, **476**, 186–198.

Airoldi L Beck MW (2007) Loss, status and trends for coastal marine habitats of Europe. *Oceanography and Marine Biology: An Annual Review*, **45**, 345–405.

Albouy-Boyer S Plourde S Pepin P *et al*. (2016) Habitat modelling of key copepod species in the Northwest Atlantic Ocean based on the Atlantic Zone Monitoring Program. *Journal of Plankton Research*, **38**, 589–603.

Allan JD McIntyre PB Smith SDP *et al*. (2013) Joint analysis of stressors and ecosystem services to enhance restoration effectiveness. *Proceedings of the National Academy of Sciences*, **110**, 372–377.

Allouche O Tsoar A Kadmon R (2006) Assessing the accuracy of species distribution models: prevalence, kappa and the true skill statistic (TSS). *Journal of Applied Ecology*, **43**, 1223–1232.

Archambault B Rivot E Savina M Le Pape O (in press) Using a spatially structured life cycle model to assess the influence of multiple stressors on an exploited coastal-nursery-dependent population. *Estuarine, Coastal and Shelf Science*.

495  Arkema KK Guannel G Verutes G *et al*. (2013) Coastal habitats shield people and property from

496    sea-level rise and storms. *Nature Climate Change*, **3**, 913–918.

497  Barceló C Ciannelli L Olsen EM Johannessen T Knutsen H (2016) Eight decades of sampling

498    reveal a contemporary novel fish assemblage in coastal nursery habitats. *Global Change*

499    *Biology*, **22**, 1155–1167.

500  Bartoń K (2016) MuMIn: Multi-Model Inference. R package version 1.15.6. https://CRAN.R-

501    project.org/package=MuMIn

502  Beck MW Heck Jr KL Able KW *et al*. (2001) The identification, conservation, and management

503    of estuarine and marine nurseries for fish and invertebrates. *BioScience*, **51**, 633–641.

504  Behrens DK Bombardelli FA Largier JL Twohy E (2013) Episodic closure of the tidal inlet at the

505    mouth of the Russian River–A small bar-built estuary in California. *Geomorphology*, **189**, 66–

506    80.

507  Bilkovic DM Roggero MM (2008) Effects of coastal development on nearshore estuarine nekton

508    communities. *Marine Ecology Progress Series*, **358**, 27–39.

509  Bottom DL Jones KK Burke J *et al*. (2005) Salmon at river's end: the role of the estuary in the

510    decline and recovery of Columbia River salmon. NOAA Technical Memorandum NMFS-

511    NWFSC-68, U.S. Department of Commerce, Seattle, WA.

512  Brown JA (2006) Using the chemical composition of otoliths to evaluate the nursery role of

513    estuaries for English sole *Pleuronectes vetulus* populations. *Marine Ecology Progress Series*,

514    **306**, 269–281.

515  Brown LR Moyle PB (2005) Native fishes of the Sacramento–San Joaquin drainage, California:

516    A history of decline. *American Fisheries Society Symposium*, **45**, 75–98.

517  Burnham KP Anderson DR (2002) Model Selection and Multimodel Inference: A Practical

518    Information-Theoretic Approach. Second Edition. Springer-Verlag New York.

519  Ceballos G Ehrlich PR Dirzo R (2017) Biological annihilation via the ongoing sixth mass

520    extinction signaled by vertebrate population losses and declines. *Proceedings of the National*

521    *Academy of Sciences*, **114**, E6089–E6096.

522  Cheng BS Bible JM Change AL *et al.* (2015) Testing local and global stressor impacts on a

523    coastal foundation species using an ecologically realistic framework. *Global Change Biology*,

524    **21**, 2488–2499.

Chittaro PM Finley RJ Levin PS *et al*. (2009) Spatial and temporal patterns in the contribution of fish from their nursery habitats. *Oecologia*, **160**, 49–61.

Cloern JE Jassby AD (2012) Drivers of change in estuarine-coastal ecosystems; discoveries from four decades of study in San Francisco Bay. *Reviews of Geophysics*, **50**, RG4001.

Costanza R d'Arge R, de Groot R *et al*. (1997) The value of the world's ecosystem services and natural capital. *Nature*, **387**, 253–260.

Courrat A Lobry J Nicolas D *et al*. (2009) Anthropogenic disturbance on nursery function of estuarine areas for marine species. *Estuarine, Coastal and Shelf Science*, **81**, 179–190.

Crain CM Kroeker K Halpern BS (2008) Interactive and cumulative effects of multiple human stressors in marine systems. *Ecology Letters*, **11**, 1304–1315.

Dahlgren CP Kellison GT Adams AJ (2006) Marine nurseries and effective juvenile habitats: Concepts and applications. *Marine Ecology Progress Series*, **312**, 291–295.

Emmett RL Stone SL Hinton SA Monaco ME (1991) Distribution and abundance of fishes and invertebrates in West Coast estuaries, Volume II: species life history summaries. ELMR Rep. No. 8. NOAA/NOS Strategic Environmental Assessments Division, Rockville, MD, 329 pp.

Emmett R Llanso R Newton J Thom R (2000) Geographic signatures of North American West Coast estuaries. *Estuaries*, **23**, 765–792.

Esselman PC Infante DM Wang L Wu D Cooper AR Taylor WW (2011) An index of cumulative disturbance to river fish habitats of the conterminous United States from landscape anthropogenic activities. *Ecological Restoration*, **29**, 133–151.

Feyrer F Cloern JE Brown LR Fish MA Hieb KA Baxter RD (2015) Estuarine fish communities respond to climate variability over both river and ocean basins. *Global Change Biology*, **21**, 3608–3619.

Gleason MG Newkirk S Merrield MS (2011) A conservation assessment of West Coast (USA) estuaries. The Nature Conservancy, Arlington VA. 65pp.

Grantham BA Chan F Nielsen KJ *et al*. (2004) Upwelling-driven nearshore hypoxia signals ecosystem and oceanographic changes in the northeast Pacific. *Nature*, **429**, 749–754.

Greene CM Kuehne L Rice C Fresh K Penttila D (2015a) Forty years of change in forage fish and jellyfish abundance across great Puget Sound, Washington (USA): Anthropogenic and climate associations. *Marine Ecology Progress Series*, **525**, 153–170.

555 Greene CM Blackhart K Nohner J Candelmo A Nelson DM (2015b) A national assessment of

556 stressors to estuarine fish habitats in the contiguous USA. *Estuaries and Coasts*, **38**, 782–799.

557 Halpern BS Kappel CV Selkoe KA *et al.* (2009) Mapping cumulative human impacts to

558 California current marine ecosystems. *Conservation Letters*, **2009**, 138–148.

559 Harborne AR Rogers A Bozec YM Mumby PJ (2017) Multiple stressors and the functioning of

560 coral reefs. *Annual Review of Marine Science*, **9**, 445–468.

561 Harley SJ Myers RA Dunn A (2001) Is catch-per-unit-effort proportional to abundance?

562 *Canadian Journal of Fisheries and Aquatic Sciences*, **58**, 1760–1772.

563 Heady WN O'Connor K Kassakian J *et al*. (2014) An Inventory and Classification of U.S. West

564 Coast Estuaries. The Nature Conservancy, Arlington, VA. 81pp.

565 Hobbs RJ Higgs E Hall CM *et al.* (2014) Managing the whole landscape: Historical, hybrid and

566 novel ecosystems. *Frontiers in Ecology and the Environment*, **12**, 557–564.

567 Honig SE Mahoney B Glanz JS Hughes BB (2017) Are seagrass beds indicators of

568 anthropogenic nutrient stress in the rocky intertidal? *Marine Pollution Bulletin*, **114**, 539–546.

569 Horsburgh JS Tarboton DG Maidment DR Zazlavsky I (2008) A relational model for

570 environmental and water resources data. *Water Resources Research*, **44**, W05406.

571 Hughes BB Levey MD Brown JA (2014) Nursery Functions of U.S. West Coast Estuaries: The

572 State of Knowledge for Juveniles of Focal Invertebrate and Fish Species. The Nature

573 Conservancy, Arlington, VA. 168pp.

574 Hughes BB Levey MD Fountain MC Carlisle AB Chavez FP Gleason MG (2015) Climate

575 mediates hypoxic stress on fish diversity and nursery function at the land-sea interface.

576 *Proceedings of the National Academy of Sciences*, **112**, 8025–8030.

577 Johnson LL Ylitalo GM Arkoosh MR *et al*. (2007) Contaminant exposure in outmigrant juvenile

578 salmon from Pacific Northwest estuaries of the United States. *Environmental Monitoring and*

579 *Assessment*, **124**, 167–194.

580 Jones KK Cornwell TJ Bottom DL Campbell LA Stein S (2014) The contribution of estuary-

581 resident life histories to the return of adult *Oncorhynchus kisutch*. *Journal of Fish Biology*, **85**,

582 52–80.

583 Jordan SJ Smith LM Nestlerode JA (2008) Cumulative effects of coastal habitat alterations on

584 fishery resources: toward prediction at regional scales. *Ecology and Society*, **14**, 16.

585     Kappel CV (2005) Losing pieces of the puzzle: threats to marine, estuarine, and diadromous

586        species. *Frontiers in Ecology and the Environment*, **5**, 275–282.

587     Kennish MJ (2002) Environmental threats and environmental future of estuaries. *Environmental*

588        *Conservation*, **29**, 78–107.

589     Kroeker KJ Sanford E Rose JM *et al.* (2016) Interacting environmental mosaics drive geographic

590        variation in mussel performance and predation vulnerability. *Ecology Letters*, **19**, 771–779.

591     Lamb JB van de Water JA Bourne DG *et al.* (2017) Seagrass ecosystems reduce exposure to

592        bacterial pathogens of humans, fishes, and invertebrates. *Science*, **355**, 731–733.

593     Lefcheck JS, Wilcox DJ, Murphy RR, Marion SR, Orth RJ (2017) Multiple stressors threaten the

594        imperiled coastal foundation species eelgrass (*Zostera marina*) in Chesapeake Bay, USA.

595        *Global Change Biology*, 1–10. doi:10.1111/gcb.13623

596     Lester SE McLeod KL Tallis H *et al.* (2010) Science in support of ecosystem-based management

597        for the US West Coast and beyond. *Biological Conservation*, **143**, 576–587.

598     Lotze HK Lenihan HS Bourque *et al.* (2006) Depletion, degradation, and recovery potential of

599        estuaries and coastal seas. *Science*, **312**, 1806–1809.

600     Magnusson A Hilborn R (2003) Estuarine influence on survival rates of Coho (*Oncorhynchus*

601        *kisutch*) and Chinook salmon (*Oncorhynchus tshawytscha*) released from hatcheries on the

602        U.S. Pacific coast. *Estuaries and Coasts*, **26**, 1094–1103.

603     Meador JP Yeh A Young G Gallagher EP (2016) Contaminants of emerging concern in a large

604        temperate estuary. *Environmental Pollution*, **213**, 254–267.

605     Merrifield MS Hines E Liu X Beck MW (2011) Building regional threat-based networks for

606        estuaries in the western United States. *PLoS One*, 6:e17407.

607     Minello TJ Able KW Weinstein MP Hays CG (2003) Salt marshes as nurseries for nekton:

608        testing hypotheses on density, growth, and survival through meta-analysis. *Marine Ecology*

609        *Progress Series*, **236**, 39–59.

610     Monaco ME Emmett RL Nelson DM Hinton SA (1990) Distribution and abundance of fishes

611        and invertebrates in West Coast estuaries, Volume I: Data summaries. ELMR Rep. No. 4.

612        NOAA/NOS Strategic Environmental Assessments Division, Silver Spring, MD, 232 pp.

613     Monaco ME Lowry TA Emmett RL (1992) Assemblages of U. S. West Coast estuaries based on

614        the distribution of fishes. *Journal of Biogeography*, **19**, 251–267.

615  Munsch SH Cordell JR Toft JD (2015) Effects of shoreline engineering on shallow subtidal fish

616      and crab communities in an urban estuary: A comparison of armored shorelines and nourished

617      beaches. *Ecological Engineering*, **81**, 312–320.

618  Munsch SH Cordell JR Toft JD (2016) Fine-scale habitat use and behavior of a nearshore fish

619      community: nursery functions, predation avoidance, and spatiotemporal habitat partitioning.

620      *Marine Ecology Progress Series*, **557**, 1–15.

621  Munsch SH Cordell JR Toft JD (2017) Effects of shoreline armoring and overwater structures on

622      coastal and estuarine fish: opportunities for habitat improvement. *Journal of Applied Ecology*.

623      doi: 10.1111/1365-2664.12906

624  Nagelkerken I Sheaves M Baker R Connolly RM (2015) The seascape nursery: a novel spatial

625      approach to identify and manage nurseries for coastal marine fauna. *Fish and Fisheries*, **16**,

626      362–371.

627  Nicolas D Lobry J Lepage M *et al*. (2010) Fish under influence: A macroecological analysis of

628      relations between fish species richness and environmental gradients among European tidal

629      estuaries. *Estuarine, Coastal and Shelf Science*, **86**, 137–147.

630  O'Connor JE Duda JJ Grant GE (2015) 1000 dams down and counting. *Science*, **348**, 496–497.

631  Pasquaud S Vasconcelos RP Franca S Henriques S Costa MJ Cabral HN (2015) Worldwide

632      patterns of fish biodiversity in estuaries: Effect of global vs. local factors. *Estuarine, Coastal*

633      *and Shelf Science*, **154**, 122–128.

634  R Core Team (2015) R: A language and environment for statistical computing. R Foundation for

635      Statistical Computing, Vienna, Austria. http://www.R-project.org/.

636  Rice CA Greene CM Moran P *et al*. (2011) Abundance, stock origin, and length of marked and

637      unmarked juvenile Chinook salmon in the surface waters of greater Puget Sound. *Transactions*

638      *of the American Fisheries Society*, **140**, 170–189.

639  Richter BD Baumgartner JV Powell J Baum DP (1996) A method for assessing hydrologic

640      alteration within ecosystems. *Conservation Biology,* **10**, 1163–1174.

641  Roegner GC McNatt R Teel DJ Bottom DL (2012) Distribution, size, and origin of juvenile

642      Chinook salmon in shallow-water habitats of the lower Columbia River and estuary, 2002–

643      2007. *Marine and Coastal Fisheries*, **4**, 450–472.

644 Rooper CN Gunderson DR Armstrong DA (2004) Application of the concentration hypothesis to

645     English sole in nursery estuaries and potential contribution to coastal fisheries. *Estuaries*, **27**,

646     102–111.

647 Shepard CC Crain CM Beck MW (2011) The protective role of coastal marshes: a systematic

648     review and meta-analysis. *PLoS ONE*, **6**, e27374.

649 Simenstad CA Fresh KL Salo EO (1982) The role of Puget Sound and Washington coastal

650     estuaries in the life history of Pacific salmon: an unappreciated function. In: *Estuarine*

651     *Comparisons* (ed Kennedy VS), pp. 343–364. Academic Press, New York, NY.

652 Siple MC Francis TB (2016) Population diversity in Pacific herring of the Puget Sound, USA.

653     *Oecologia*, **180**, 111–125.

654 Thorson JT Cope J Kleisner K Shelton A Samhouri J Ward E (2013) Giants' shoulders 15 years

655     later: Lessons, challenges, and guidelines in fisheries meta-analysis. *Fish and Fisheries*, **16**,

656     342–361.

657 Toft JD Cordell JR Simenstad CA Stamatiou LA (2007) Fish distribution, abundance, and

658     behavior along city shoreline types in Puget Sound. *North American Journal of Fisheries*

659     *Management*, **27**, 465–480.

660 Toft JD Bilkovic DM Mitchell MM La Peyre MK (2017) A synthesis of living shoreline

661     perspectives. In: *Living Shorelines: The Science and Management of Nature-Based Coastal*

662     *Protection* (eds Bilkovic DM Mitchell MM La Peyre MK Toft JD), pp. 481–486. CRC Press,

663     Boca Raton, FL.

664 UNEP (2006) Marine and coastal ecosystems and human wellbeing: A synthesis report based on

665     the findings of the Millennium Ecosystem Assessment. UNEP. 76 p.

666 USEPA (2012) National Coastal Condition Report IV. U.S. Environmental Protection Agency,

667     Office of Research and Development and Office of Water. EPA/842-R-10-003. 298 p.

668 Valesini FJ Teedley JR Clarke KR Potter IC (2014) The importance of regional, system-wide

669     and local spatial scales in structuring temperate estuarine fish communities. *Estuaries and*

670     *Coasts*, **37**, 525–547.

671 Vasconcelos RP Reis-Santos P Fonseca V *et al*. (2007) Assessing anthropogenic pressures on

672     estuarine fish nurseries along the Portuguese coast: A multi-metric index and conceptual

673     approach. *Science of the Total Environment*, **374**, 199–215.

674    Vasconcelos RP Le Pape O Costa MJ Cabral HN (2013) Predicting estuarine use patterns of

675      juvenile fish with generalized linear models. *Estuarine, Coastal and Shelf Science*, **120**, 64–74.

676    Vasconcelos RP Henriques S Franca S Pasquaud S Cardoso I Laborde M Cabral HN (2015)

677      Global patterns and predictors of fish species richness in estuaries. *Journal of Animal Ecology*,

678      **84**, 1331–1341.

679    Vinebrooke RD Cottingham KL Norberg J Stanley MS Dodson I Maberly SC Sommer U (2004)

680      Impacts of multiple stressors on biodiversity and ecosystem functioning: the role of species co-

681      tolerance. *Oikos*, **104**, 451–457.

682    Vye SR Emmerson MC Arenas F Dick JTA O'Connor NE (2015) Stressor intensity determines

683      antagonistic interactions between species invasion and multiple stressor effects on ecosystem

684      functioning. *Oikos*, **124**, 1005–1012.

685    West JE O'Neill SM Ylitalo GM (2017) Time trends of persistent organic pollutants in benthic

686      and pelagic indicator fishes from Puget Sound, Washington, USA. *Archives of Environmental*

687      *Contamination and Toxicology,* **73**, 207–229.

688    Weitkamp LA Goulette G Hawkes J O'Malley M Lipsky C (2014) Juvenile salmon in estuaries:

689      comparisons between North American Atlantic and Pacific salmon populations. *Reviews in*

690      *Fish Biology and Fisheries,* **24**, 713–736.

691    Wood SN (2011) Fast stable restricted maximum likelihood and marginal likelihood estimation

692      of semiparametric generalized linear models. *Journal of the Royal Statistical Society*, **73**, 3–36.

693    Wood SN (2015) Mixed GAM computation vehicle with GCV/AIC/REML smoothness

694      estimation. R package version 1.8–7. Available at: https://cran.r-

695      project.org/web/packages/mgcv/index.html.

696    Yang Z Khangaonkar T (2008) Modeling of salt intrusion, intertidal mixing, and circulation in a

697      braided estuary. *Journal of Coastal Research*, 171–180.

698    Yeh A Marcinek DJ Meador JP Gallagher EP (2017) Effect of contaminants of emerging

699      concern on liver mitochondrial function in Chinook salmon. *Aquatic Toxicology*. DOI:

700      10.1016/j.aquatox.2017.06.011

701    Zuur AF Ieno EN Walker N Saveliev AA Smith GM (2009) Mixed effects models and

702      extensions in ecology with R. Springer-Verlag, Berlin.

703

704 Table 1. Stressor scores used in analyses. The 20 Estuaries are listed from north to south, with

705 stressors of flow, land cover, and pollution from Greene *et al.* (2015b).

| 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 |

706

707

708 **Figure Captions**

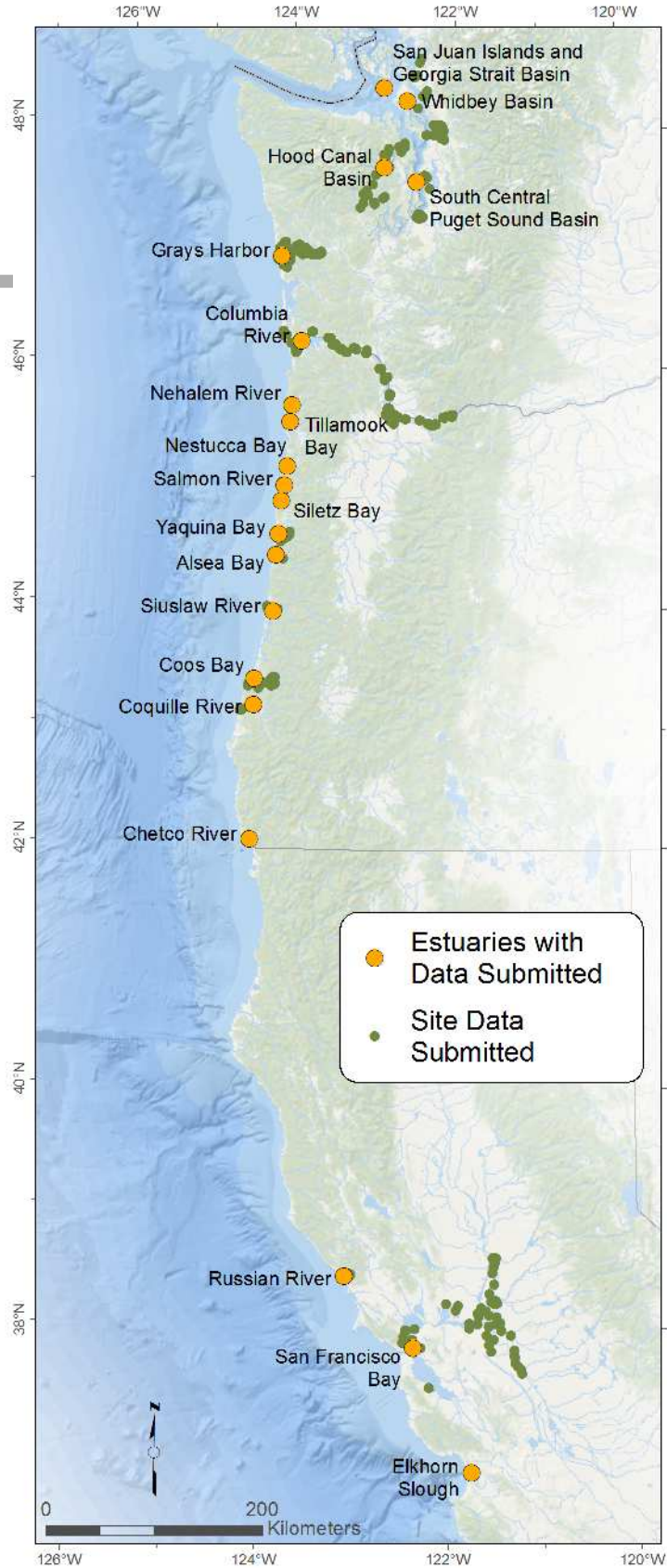709 Figure 1. Location of the 20 estuaries analyzed, and specific sampling locations.

710 Figure 2. Percent composition by abundance of data on focal species in each estuary. Estuaries

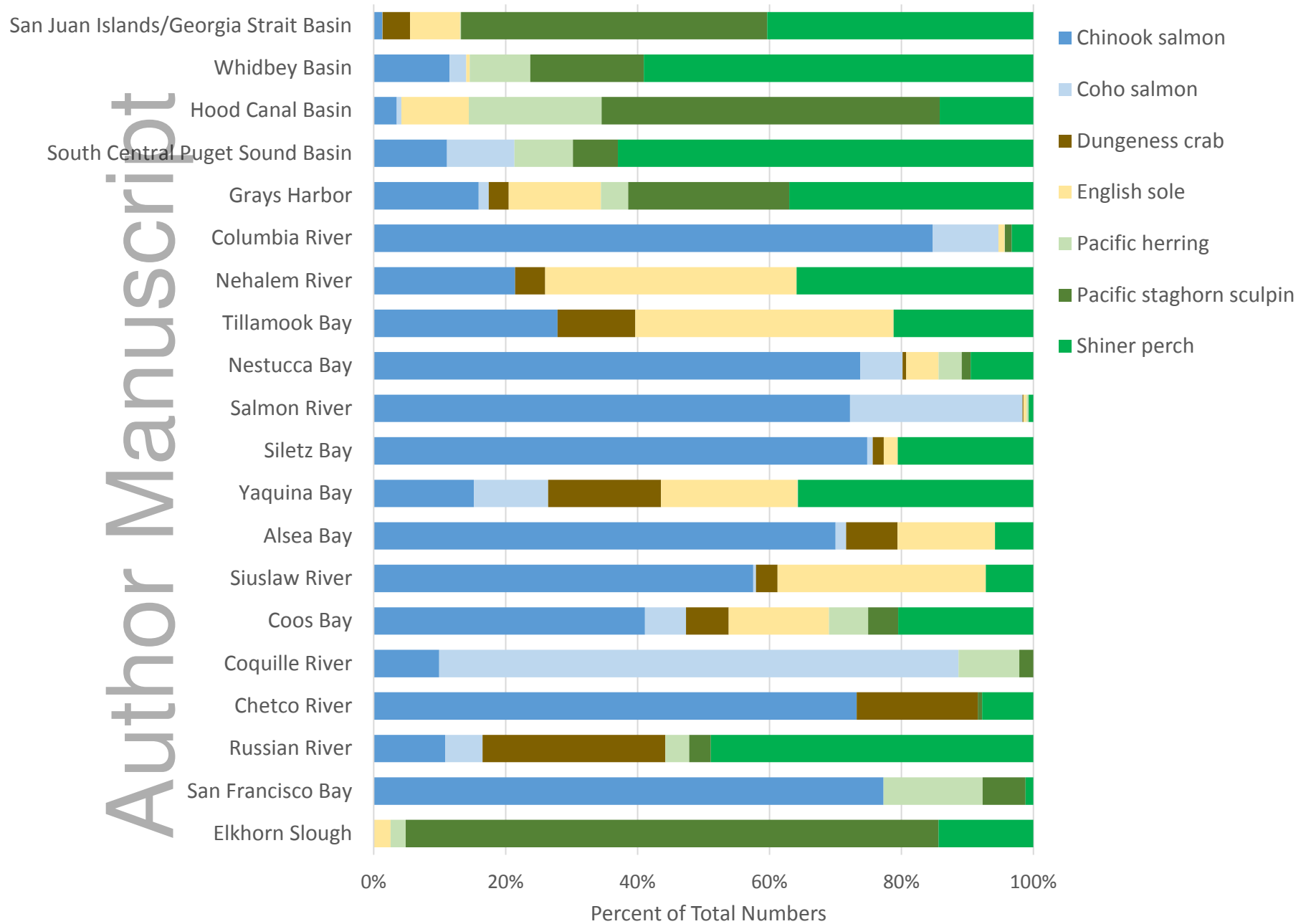711 are sorted descending by latitude from top to bottom.

712    Figure 3. Output of GAMMs describing the stressor effect to presence/absence and catch per unit

713    effort of seven focal species. Parameter estimates are indicated by circles and their 95%

714    confidence intervals are indicated by bars. Statistically significant ($P < 0.05$) terms are

715    highlighted. Parameter estimates of the seawater and tidal fresh zones are reported relative to

716    estimates of the mixing zone (i.e., the mixing zone is the baseline for salinity zone parameters).

717    Figure 4. Predicted presence and CPUE, when present, of Chinook salmon among estuaries.

718    Values were calculated using GAMM parameter estimates and estuary-specific stressor scores.
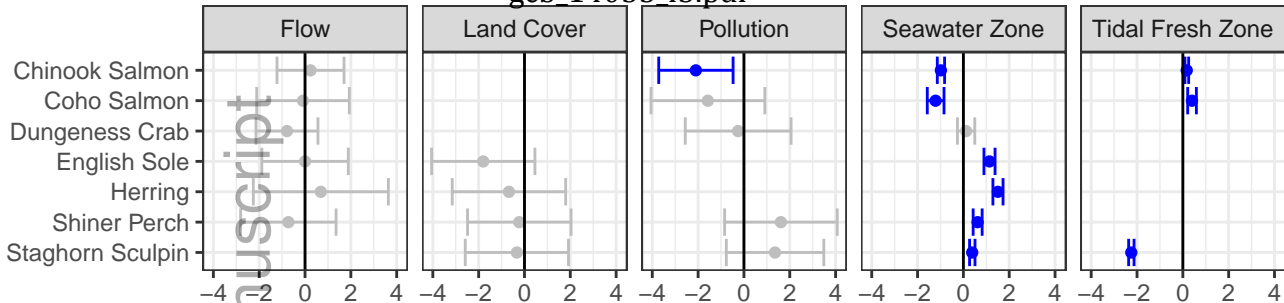
719    Figure 5. Predicted CPUE, when present, of English sole among estuaries. Values were

720    calculated using GAMM parameter estimates and estuary-specific stressor scores.

gcb_14055_s1.png
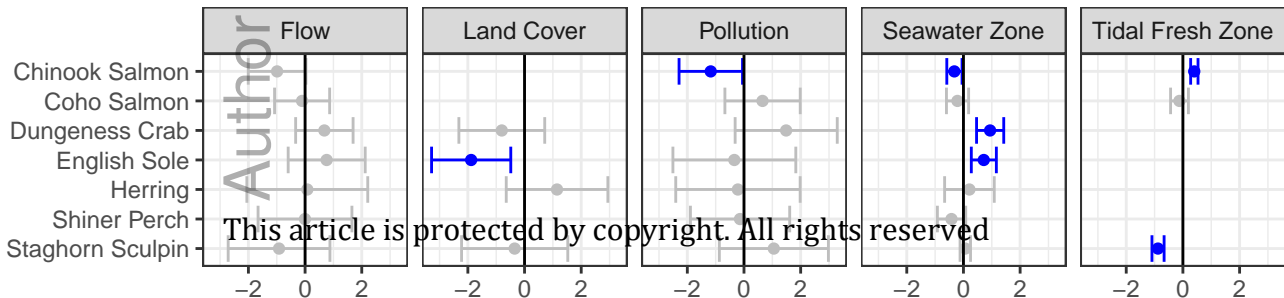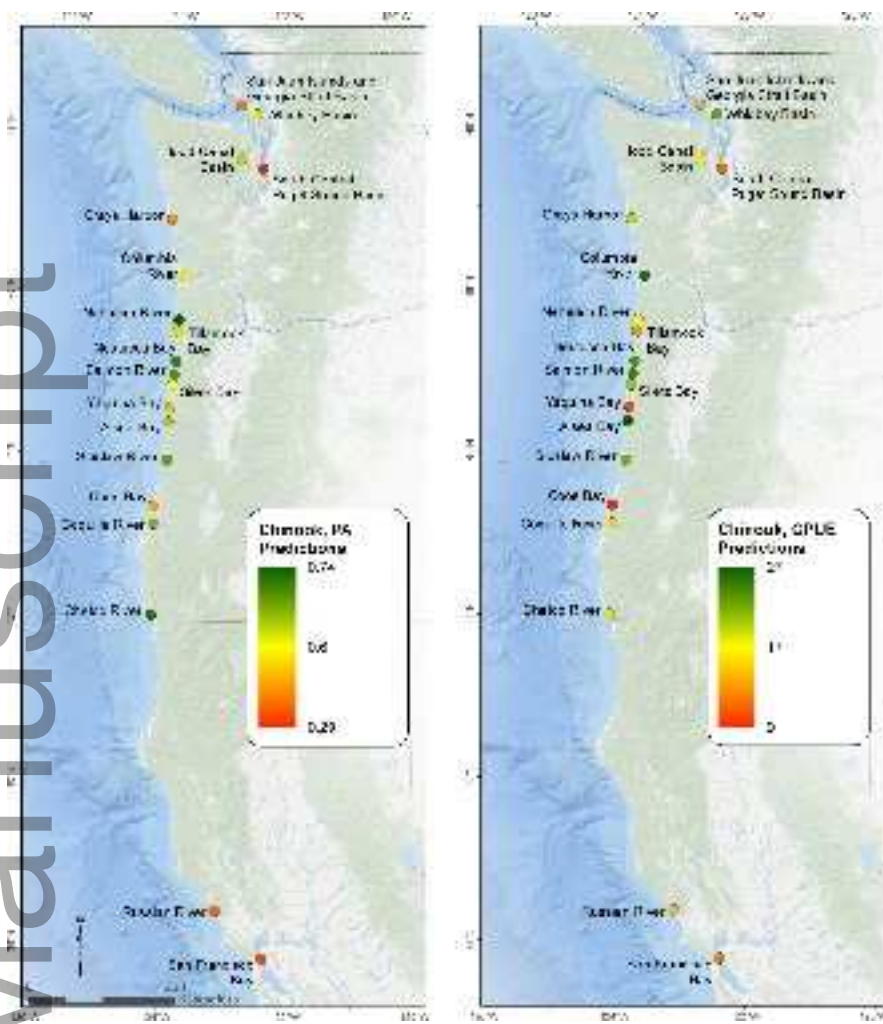
Legend:
- Chinook salmon
- Coho salmon
- Dungeness crab
- English sole
- Pacific herring
- Pacific staghorn sculpin
- Shiner perch

X-axis: Percent of Total Numbers (0% to 100%)

Categories (top to bottom):
- San Juan Islands/Georgia Strait Basin
- Whidbey Basin
- Hood Canal Basin
- South Central Puget Sound Basin
- Grays Harbor
- Columbia River
- Nehalem River
- Tillamook Bay
- Nestucca Bay
- Salmon River
- Siletz Bay
- Yaquina Bay
- Alsea Bay
- Siuslaw River
- Coos Bay
- Coquille River
- Chetco River
- Russian River
- San Francisco Bay
- Elkhorn Slough

gcb_14055_f4.png

English Sole, CPUE Predictions

26.5

17.9

13.4

gcb_14055_f5.png