Table 1. Additional information for each estuary is provided in the Supplementary data[1].

Total square metres for each estuarine exposure area was determined without considering differences in habitat type or quality. These values (converted to km^2) represent only the intertidal and subtidal areas of the local estuary plus the immediate nearshore habitat, which in some cases resulted in different values from those of Bortleson et al. (1980), who calculated total surface area of the estuary. Depths were limited to 10 m as shown on National Oceanic and Atmospheric Administration charts, the range that juvenile Chinook are known to utilize (Carter et al. 2009). The area of subtidal habitat, which can support prey for juvenile salmon, was not included in the Bortleson et al. (1980) analyses. Prey species are more abundant in some types of habitats; however, it was assumed that invertebrates from the benthos and associated water column would be available for consumption by juvenile Chinook. In many systems juvenile salmon feed on the benthos (Higgs et al. 1995; Cordell et al. 2001*a*; Fresh et al. 2006), which can contain very high densities of common prey species, even in those systems that are considered contaminated and lacking prime habitat (e.g., the Duwamish).

To determine the number of fish per square metre in local estuaries, the number of ocean-type Chinook and coho released into these systems was estimated and divided by the calculated area (Table 1). Data for the number of fish released was obtained from the Hatchery Scientific Review Group (2002), Washington Department of Fish and Wildlife (2000–2005), and RMIS database. Only ocean-type Chinook and coho were included in this analysis because they would likely compete for resources in the estuary and represent the majority of juvenile salmon in these systems. Mean values were used for fish release estimates taken from the RMIS database, which were often variable over years.

Many of the estimated density values for outmigrating fish are likely overestimates, because some hatcheries have not operated for all years of this study and many of the release estimates are current levels and were lower in previous years. The Green River – Duwamish estimate is the only one that includes both hatchery and natural production for both species. Hatchery fish likely predominate in most systems, as indicated by Rice et al. (2011) for Chinook in many of these local estuaries; however, the actual contribution of wild fish to most systems is unknown. It should be noted that these density values include outmigration for fish over several months (April–July) and are system-wide means that do not account for location-specific abundance, which can be higher (Cordell et al. 2011).

Analyses

In this analysis, hatchery was treated as a replicate (Ryding and Skalski 1999). One-way analysis of variance (ANOVA) and analysis of covariance (ANCOVA) were run on various combinations of factors. The SAR data were not normally distributed, which violates the assumption of normality for ANOVA. The Chinook data were best fit to a log-normal distribution, and the coho SAR values were arcsin square-root transformed to achieve a normal distribution. The nonparametric Wilcoxon signed-rank test (Sokal and Rohlf 1995) was also performed, which allowed year-wise comparisons and a reduction in the large variability for survival that occurs from year to year. The Wilcoxon test uses the sign and magnitude of the differences to determine trends. Also, a Mann–Kendall test was performed on SAR values over years to test for a temporal trend. Regressions were performed to test for relationships between survival, release mass, and release day of year (DoY). These were performed for combinations of hatchery–year data and for all tag code groups over different time periods. One period (release years 1985–2008) was selected because of the general decline in SAR after 1985 and to capture the most recent data. Another period selected was for the release years 1997–2008, to more closely match the analysis from Duffy and Beauchamp (2011). Standard deviations (SD) show the range in data for a parameter, and the standard error (SE) was reported when comparison of means was intended.

Results

The mean (SD) SAR, release mass, number of fish released (total and with CWT), and years of data for each hatchery are shown in Tables 3 and 4. This dataset comprises 2.3×10^8 total Chinook (21% with CWT) and 1.1×10^8 coho (30% with CWT) released over 37 years. A few hatcheries were represented most years of the analysis; however, many yielded data for a subset of this time period.

For 5 of the release years (1974, 1977–1978, and 1983–1984), Chinook SAR values were available for only one of the categories (contaminated or uncontaminated estuary) and were therefore not used in the analysis. For 9 of the years, data were available for only one hatchery (no replicates) in one of the categories. Coho SARs were represented in all release years (1973–2008) for both groups.

Estuary contamination

A comparison of concentrations in whole-body, liver, and stomach contents for fish from several hatcheries indicated that juvenile Chinook were exposed to high levels of contaminants in some estuaries (Table 2). Mean (SD) concentrations of PCBs in stomach contents were relatively high for hatchery fish (60 (39) $ng \cdot g^{-1}$ wet mass) because of a few high values in the 1980s. Over the past 20 years, hatchery feed concentrations have declined substantially to very low levels, which may be due to changes in the oil added to fish pellets (Maule et al. 2007). When available, contaminant concentrations in salmon and other species, in conjunction with sediment toxicity bioassays, sediment criteria, and the number of listed sites, all agreed (except for one minor case), supporting the designation of contaminated or uncontaminated for each estuary. For contaminated estuaries, these different lines of evidence all support the expectation of adverse effects for outmigrating salmon.

Few data exist on contaminant concentrations in juvenile coho. One study examined whole-body and stomach content concentrations in juvenile Chinook and coho from five Oregon estuaries over several years. Estuary-matched mean concentrations for whole-body total PCBs and DDTs were higher in juvenile Chinook (~2.5-fold for PCBs and ~3.2-fold for DDTs) (Johnson et al. 2007). Differences this large, and greater, for these two species were also seen for fluorescent aromatic compounds in bile (phenanthrene wavelengths) as a result of PAH exposure and for total DDTs and PCBs in stomach contents for those sites exhibiting elevated levels. The largest differences were observed for total PAHs in stomach contents, which ranged from ~10 to 200 times higher in Chinook over coho. These results are supported by another study showing that the concentrations of whole-body total PCBs in juvenile Chinook were 6.6 times higher compared with values for juvenile coho collected in Commencement Bay near the Puyallup River estuary (Olson et al. 2008). These data support the hypothesis of higher levels of toxicant exposure for Chinook compared with coho outmigrating through contaminated estuaries.

Smolt-to-adult survival

When all data were considered (all hatchery–year combinations; $n = 244$), the mean survival for juvenile Chinook released from hatcheries into contaminated estuaries was 45% lower than for fish outmigrating through uncontaminated estuaries (SAR values 0.48% versus 0.87%; $p < 0.0001$) (Table 5). The ANOVA for Chinook release masses was not significant ($p = 0.28$), indicating no difference among hatcheries and years. The ANCOVA also determined that there was no interaction between estuary contamination status and fish mass at release ($p = 0.27$). The more appropriate analysis by the nonparametric Wilcoxon signed-rank test