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ScienceDirect
Marine Pollution Bulletin 52 (2006) 892–899
MARINE POLLUTION BULLETIN
www.elsevier.com/locate/marpolbul

Results from a sixteen year study on the effects of oiling from the *Exxon Valdez* on adult pink salmon returns

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Abstract

For sixteen years following the 1989 *Exxon Valdez* oil spill adult returns of pink salmon in Prince William Sound, Alaska were monitored to assess spill effects on survival. No evidence of spill effects was detected for either intertidal or whole-stream spawning fish. From 1989 through 2004 mean densities for oiled and reference streams tracked each other, illustrating similar responses of oiled and reference stream adult populations to naturally changing oceanographic and climactic conditions. Hatchery fish strayed into the study streams, but similar incursions occurred in oiled and reference streams, and their presence was compensated for to eliminate their influence on determining the success of the returning natural populations. These results, showing no detectable effects of oiling on pink salmon spawning populations, are supported by published field studies on pink salmon incubation success in oiled streams.
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Keywords: Pink salmon; Oil effects

1. Introduction

When the *Exxon Valdez* oil spill occurred in Prince William Sound (PWS), March of 1989, there were expectations that salmon and the commercial fisheries they supported would be seriously affected. Pink salmon were considered the most vulnerable salmon species because they spawned extensively in the intertidal areas of PWS streams (Noerenberg, 1963; Thorsteinson et al., 1971), and thus adults could have potentially been exposed to oil concentrations during the entire fall of 1989, and developing embryos could have been exposed during the fall and winter incubation period in those years immediately following the spill. Contrary to those expectations, estimates by Alaska Department of Fish and Game (ADFG) of the overall impact of the spill on pink salmon were relatively small (Geiger et al., 1996), and nine of the ten largest run sizes ever recorded in PWS have occurred since the spill (Fig. 1). Oil washed ashore in just the southwestern region of the Sound and affected only 14% of the 212 identified salmon streams in the three districts of Eshamy, Southwest, and Montague (Neff and Stubblefield, 1995; Maki et al., 1995) (Fig. 2), which accounted for approximately 20% of the annual wild escapement (Donaldson et al., 1995). ("Escapement" is the number of spawning adults that escape the fishery and return to spawn in their natal streams.) Consequently, it is feasible that negative effects in those streams could have been obscured by the overall increased abundance of pink salmon throughout the Sound.

Effects of the spill on pink salmon would have originated with the embryos exposed to toxic polycyclic aromatic hydrocarbon (PAH) levels during their incubation phase, because PAH concentrations in the marine waters were too dilute to be a problem even in the first year after the spill (Neff and Stubblefield, 1995), and oil was virtually

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doi:10.1016/j.marpolbul.2005.08.028

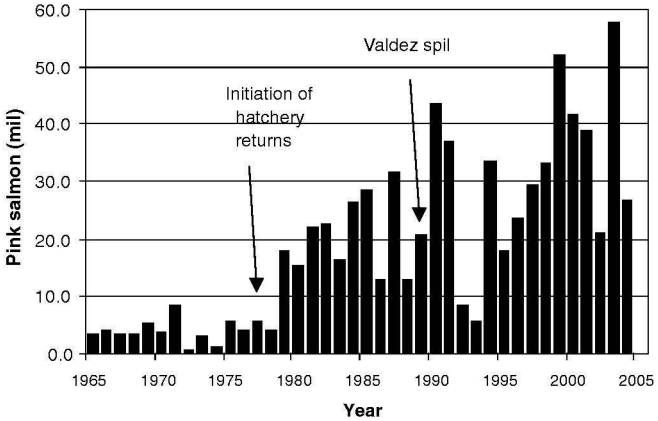

Fig. 1. Historical run sizes of pink salmon in Prince William Sound, Alaska (ADFG Commercial Fisheries). The *Exxon Valdez* oil spill occurred in 1989.

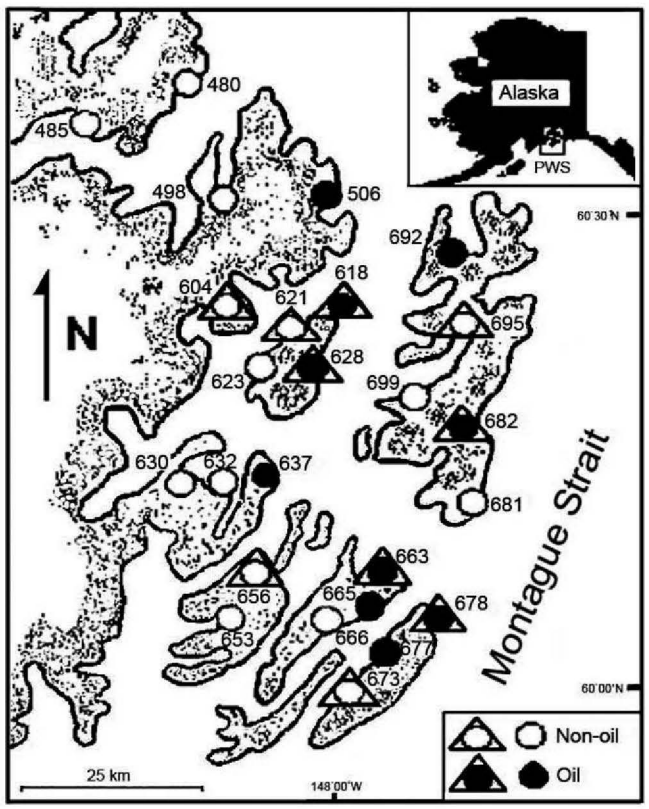

Fig. 2. Study streams in Prince William Sound and the AFK hatchery. Those with an open triangle are used by the authors and those with circles are used in studies by ADFG and NMFS, NOAA.

gone by the second year. However, oil in the streams was also at low concentrations immediately after the spill (Brannon et al., 1995), which presents a problem in attempting to reconcile allegations of damage to pink salmon. Mean TPAH concentrations in stream sediments did not exceed 267 ppb in 1989, and, with the exception of a few streams, concentrations dropped to near background levels by 1991.

Therefore, oil in stream gravel was never high enough to be toxic to incubating pink salmon. In response to those findings, it has been suggested that the risk to the embryos was actually from the highly toxic leachate originating from buried oil deposits on the adjacent beaches (Murphy et al., 1999). Based on laboratory results, very low concentrations of high molecular weight PAH compounds have been reported lethal to pink salmon embryos (Heintz et al., 1999), and these are the hydrocarbons alleged to have infiltrated the incubation streams and caused mortality of incubating eggs, translating into a high adult loss in the succeeding years (Peterson et al., 2003). If there were any long-term measurable effects of the spill on pink salmon, it would best be demonstrated in adult abundance patterns in the spill-affected stream populations.

The present study analyzed enumeration data in spill affected (oiled) and reference (non-oiled) streams to ascertain if there were any effects on returning adults in 1989 and on early life stages in years following the spill that would translate into reduced adult pink salmon productivity. Except for 1993, pink salmon spawning populations have been enumerated by the authors in the southwestern part of the Sound from 1989 to 2004, for five of the most heavily oiled streams, and, in most years, five of the unaffected reference streams in the same region.

Pink salmon have a two year lifecycle with odd and even year populations, and thus never genetically cross. The majority of spawning occurs in intertidal reaches of streams (Helle, 1970; Thorsteinson et al., 1971). Embryos and larvae develop throughout the fall and winter within stream-bed gravel. Juveniles emerge from gravel in April-June and move into estuarine habitats. They feed in nearshore areas for a few weeks, and then move into the Gulf of Alaska where they begin a 12-month counterclockwise migratory circuit around the north Pacific, before returning to their natal streams the following summer.

Helle et al. (1964) noted that earlier returning pinks tend to spawn above tidal influence, while the later spawners use the intertidal areas, although temporal segregation may not be apparent in some brood years (Helle, 1970). The later fish are also more confined to intertidal areas because of lower flows late in the season. The slightly warmer temperatures from mixing with marine waters compensate for the later spawning. Gharrett et al. (2001) also noted that pink salmon tend to return to specific reaches within their natal streams. Consequently, sub-stocks are inclined to form within stream populations, segregated by tidal elevation, with only the intertidal fish at any risk.

The presence of hatchery strays in PWS streams was also a factor in assessing return success of the natural population. The abundance of stray hatchery fish is highest in those streams closest to the hatcheries (Sharp et al., 1993; Joyce and Evans, 1999a,b), and since hatchery fish were not exposed to in-stream oiling as eggs and alevins, their relative abundance as adults would not have been affected by oiling. Consequently, it was necessary to account for hatchery fish in the analyses and interpretation of

the effects of oiling on the return success of the natural population.

2. Methods

2.1. Field sampling

Forty streams were selected for the study in 1989, based on location, potential for oil exposure, and the likelihood that pink salmon used the intertidal reaches for spawning. The selected streams were in the southwestern part of PWS, and each stream was classified as either "oiled" or "reference" on the basis of levels of oiling recorded by shoreline survey teams during the summer of 1989 (Neff and Stubblefield, 1995). Subsets of the original 40 streams were selected for continued study in succeeding years (Fig. 2), and included 5 oiled and 5 reference streams, which were also a subset of those surveyed by the *Exxon Valdez* Oil Spill Trustee Council for the egg mortality studies (Sharr et al., 1994a,b). The 5 oiled streams provided a substantial and representative sample (19%) of the oiled streams in the Southwest district. Each stream was divided into an (1) intertidal reach, from 1.2 m to 3.7 m above mean lower low water (MLLW), and (2) an upstream reach, from above +3.7 m to the farthest upstream location where pink salmon were observed. Standard visual survey counts of live pink salmon adults were made by two man crews at least weekly (Maki et al., 1995) from the first part of August to the 15th of September each year, providing at least four to ten stream visits per season.

2.2. Estimating escapement

Estimating a spawning population, i.e., total escapement, from stream counts is complicated by three issues which result from counting procedures and spawning behavior of pink salmon. First, when sampling multiple streams, it is not feasible to count each stream daily due to travel time between streams and the need to sample during low tide windows in daylight hours.

Consequently, we counted each stream four to ten times during a spawning season. Second, multiple counts of fish occur when the length of time between surveys is less than streamlife. Streamlife is defined as the number of days a fish lives in a stream (Bocking et al., 1998; Quinn and Deriso, 1999). Hence, some fish survive to be recounted on subsequent surveys, resulting in multiple counts of the same fish. Third, peak spawning, length of spawning season, and streamlife differ among streams and affect estimates of escapement (Pirtle, 1977; Sharp et al., 1993). These three issues are addressed by the following procedures.

Streamlife: We estimated streamlife by fitting the Weibull distribution (Lawless, 1982) to observed streamlife estimated from four study streams in 1989 (Maki et al., 1995). Because streamlife decreases through the spawning season (Helle et al., 1964; Neilson and Green, 1981; McCurdy, 1984) we estimated streamlife separately for six consecutive weeks. Pink salmon were captured weekly at stream mouths, tagged and released. We surveyed streams daily for live and dead tagged salmon. The number of days between tagging and recovery of dead salmon provided an estimate of observed streamlife l for each tagged fish,

$$\begin{aligned} l = {} & (\text{Julian date of tag recovery} \\ & - \text{Julian data of tag release}) \\ & - \text{estimated milling time.} \end{aligned} \tag{1}$$

Milling time was the intervals in days between release and entry of the first tagged fish of a weekly tag group. For each weekly tag group, the observed streamlife l was fit with the Weibull survival function to estimate daily survival rates. See Fig. 3a for Weibull estimates of the daily fraction of surviving fish (based on Maki et al., 1995, Fig. 4a).

Discounting: To estimate escapement, the fish counted on any the 2nd and higher surveys were discounted for the fish surviving from previous surveys. The discounted number of fish d on sampling date $i + a$ from counts c made on a previous sampling date i is

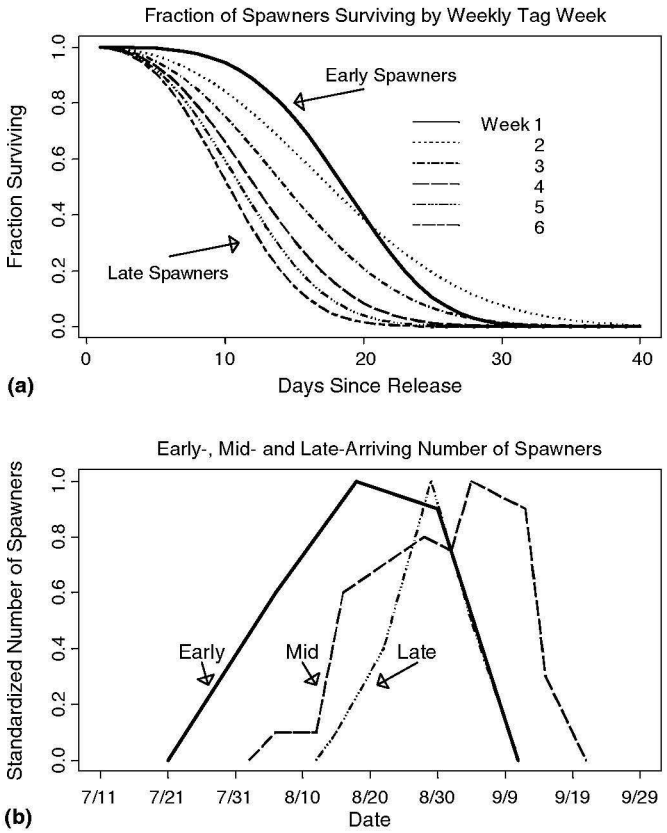

Fig. 3. (a) Weibull estimates of fraction surviving by weekly tag group. Survival rate generally decreased from early- to late-arriving spawners. (b) Count profiles for early-, mid- and late-arriving salmon in 1989 for example streams. Profiles have been standardized to peak count.

$$d_{i+a} = \left(\prod_{i}^{a} S_i\right) c. \quad (2)$$

S_i is the daily survival rate (previous paragraph). We assigned daily survival rates S_i to non-study streams (and to all streams in all years after 1989) on arrival time and duration of the spawning season for that particular stream. Since the length of the escapement time period decreased over the season, later-arriving salmon had less time to spawn, hence shorter streamlife, than earlier-arriving salmon. Longer survival was assigned to early arriving adults. See Fig. 3b for examples of streams with early-, mid-, and late arriving salmon in 1989 (based on Maki et al., 1995, Fig. 4b).

Escapement: The modified area-under the curve method was used to interpolate across non-sample days (Maki et al., 1995) and to estimate escapement. Surveys were discounted for fish counted previously and plotted across time. Escapement was estimated by summing polygons formed between adjacent survey dates (Perrin and Irvine, 1980).

2.3. Impact on productivity

The central issue for assessing effects of oiling on adult returns lies in reducing confounding effects of spatial and annual variation on the oiling signal. Spatial effects include stream size, flow volume, substrate quality, and location of the upstream barrier. Since each stream is discrete, productivity varies among streams. Superimposed on spatial variation are annual variations. Annual affects include temporally varying climactic and oceanographic conditions, as well as harvest impacts. Because annual affects are regional, influencing all streams in southwestern PWS similarly, we expect similar biennial changes in spawning density and spawner density feedback, i.e., spawner-recruitment, for oiled and reference stream categories.

We base our statistical assessment on the exponential growth model (Quinn and Deriso, 1999).

Let N_{hij} = escapement in the ith stream in the jth year for treatment h (oil, reference). Stream population trends are represented by

$$N_{hij} = N_{hi0} \exp(r_h t_j), \quad (3)$$

where N_{hi0} is the initial escapement at each stream i within each treatment h; r_h is the population growth rate for the hth treatment; t_j is the elapsed time in years j (= 1,2).

The exponential function is fundamental to most population growth models used in fisheries (e.g., the logistic is a dampened exponential, Quinn and Deriso, 1999). Our purpose here is to develop an additive statistical model which explicitly incorporates covariate and treatment factors that would affect growth rate r, rather than modeling population dynamics. We can reparametize time t and rate r such that $r_h t_j$ is expressed in terms of environmental factors which affect population growth (covariates, Cov) and *treatment* effects (oiled and reference)

$$\beta_1 t_j + \beta_2(t_j \times \text{Cov}) + \beta_3(t_j \times \text{treatment}), \quad (4)$$

where covariates Cov could be any or all of the environmental factors affecting growth rate. Eq. (3) becomes

$$N_{hij} = N_{hi0} \exp(\beta_1 t_j + \beta_2(t_j \times \text{Cov}) + \beta_3(t_j \times \text{treatment})). \quad (5)$$

Occurring in the exponential term, Cov and treatment, effects are multiplicative and represent environmental conditions which are likely to produce similar fractional changes in abundance across streams, e.g., reduction in escapement would be proportionate to loss of habitat due to oiling. Taking logarithms and substituting β_0 for $\ln N_{hi0}$

$$\ln N_{ij} = \beta_0 + \beta_1 t_j + \beta_2(t_j \times \text{Cov}) + \beta_3(t_j \times \text{treatment}), \quad (6)$$

becomes additive in coefficients for time t, Cov and treatment (oil, ref). Given additive effects, we expect the coefficient for the interaction of time and treatment $t_j \times$ treatment to be zero. That is, under the null hypothesis of no oiling effect, profiles of means at oiled and reference treatments parallel each other over years t, given *treatment*s are in dynamic equilibrium (Skalski et al., 2001; Wiens and Parker, 1995; Parker and Wiens, 2005). As argued in the first paragraph of this section, the assumption of oiled and reference streams being similarly affected by temporally varying factors (e.g., climate) is supported by the observation of similar time profiles among reference streams.

As with oiling, strays from hatcheries would also affect stream counts. Joyce and Evans (1999a,b) showed that the proportion of strays from hatcheries was negatively correlated with distance from hatcheries. Cronin and Maki (2004) used the measured proportion of hatchery fish and distance from the Armin F. Koernig (AFK) hatchery for the streams studied by Joyce and Evans (1999a,b) to characterize the relationship in proportion of hatchery fish with distance from hatchery. Strays have both spatial and temporal components. Using distance from hatcheries dist as a covariate results in the following model where den_{ij} is the density for stream i and treatment j, dist is the distance to AFK and *year* is t,

$$\begin{aligned}\text{den}_{ij} = {} & \beta_0 + \beta_1(\text{year}) + \beta_2(\text{dist}) + \beta_3(\text{year} \times \text{dist}) \\ & + \beta_4(\text{treatment}) + \beta_5(\text{year} \times \text{treatment}) + \text{error}.\end{aligned} \quad (7)$$

Main effects for interactions have been added and coefficients were renumbered from Eq. (6). Since streams differ in spawning area, density (den_{ij}) standardizes stream counts (N_{ij}) across streams and treatments. Covariate effects of time *year* and distance dist to AFK are removed prior to testing for *treatment* and the *year-by-treatment* interaction. β_1 and β_2 test for the *year* effect and the stray effect as measured by distance (dist) from AFK, and β_3 their interaction. β_4 tests for a *treatment* effect; differences in spawning density between oiled and reference treatments could be due to natural spatial variation and consequently, a significant coefficient β_4 does not in itself signal an oiling effect. β_5 tests for the *year-by-treatment* interaction: under

the null hypothesis of no oiling effect on escapement density, profiles in mean density for oiled and reference treatments will be parallel. Non-parallel profiles infer an initial oiling impact and subsequent recovery (Skalski et al., 2001; Wiens and Parker, 1995; Parker and Wiens, 2005). Data for all years as well as separately for odd and even year populations were tested.

3. Results

3.1. Productivity

Excluding 1993 when streams were not surveyed, two of the five reference streams were surveyed in all years, whereas all oiled streams were surveyed each year. On average, reference and oiled categories of streams were nearly equal in mean annual whole-stream escapements, each at about 10 K fish. Reference streams were more similar in mean escapement than were oiled steams, where means for reference streams ranged from 9.0 to 14.0 K and for oiled streams from 2.6 to 22.6 K fish. The lowest estimate of whole-stream escapement was 86 fish (1992 at stream 16180) and the highest was 39 K (1998 at stream 16820). Generally, about half the spawners were found in the intertidal areas.

Table 1 shows results of tests on effects of time, distance from AFK, treatment (oil, reference) and oiling on escapement density. There was a significantly ($\alpha = 0.05$) positive relationship between time and escapement density. The effect of strays on escapement was significant for the intertidal density, but not whole-stream density, although the p-value was relatively low for whole-stream for all years (p-value $= 0.062$). These analyses do not show evidence for a trend in the effect of strays on escapement density over years, no year × dist interaction. Treatment (oiled or reference) was not significant, perhaps due to large-scale inter-annual variability (Fig. 4). Temporal variation was large, with mean whole-stream escapement density ranging from approximately 0.2–3.2 fish/m^2. The null hypothesis for equal profiles for oiled and reference treatments was not rejected, i.e., no year × treatment interaction. Fig. 4 supports failure to reject this hypothesis of equal profiles, where oiled and reference means track temporal variation in escapement. To supplement these analyses we used a multivariate repeated measures analysis after filling in missing data with the Expectation-Maximization algorithm (Dempster et al., 1977): profiles in treatment means were also not significant for the repeated measures model, supporting above results for no effect of oiling on escapement density.

Table 1
p-Values for hypothesis on time, distance from hatchery and effects of oiling on spawning density

Coefficient[a]	Effect	p-Values for populations		
		All years	Even years	Odd years
Intertidal				
β_1	Time	0.001	0.029	0.002
β_2	Strays	0.003	0.012	0.008
β_3	Trend in strays	0.993	0.993	0.789
β_4	Treatment	0.463	0.463	0.250
β_5	Oiling[b]	0.987	0.987	0.822
Whole-stream				
β_1	Time	0.001	0.018	0.004
β_2	Strays	0.062	0.238	0.114
β_3	Trend in strays	0.994	0.991	0.689
β_4	Treatment	0.922	0.507	0.404
β_5	Oiling[b]	0.966	0.988	0.955

[a] Coefficients from Eq. (7). Null hypothesis is for zero coefficients, i.e., no effect.
[b] Test for parallel profiles in treatment (oil, reference) means.

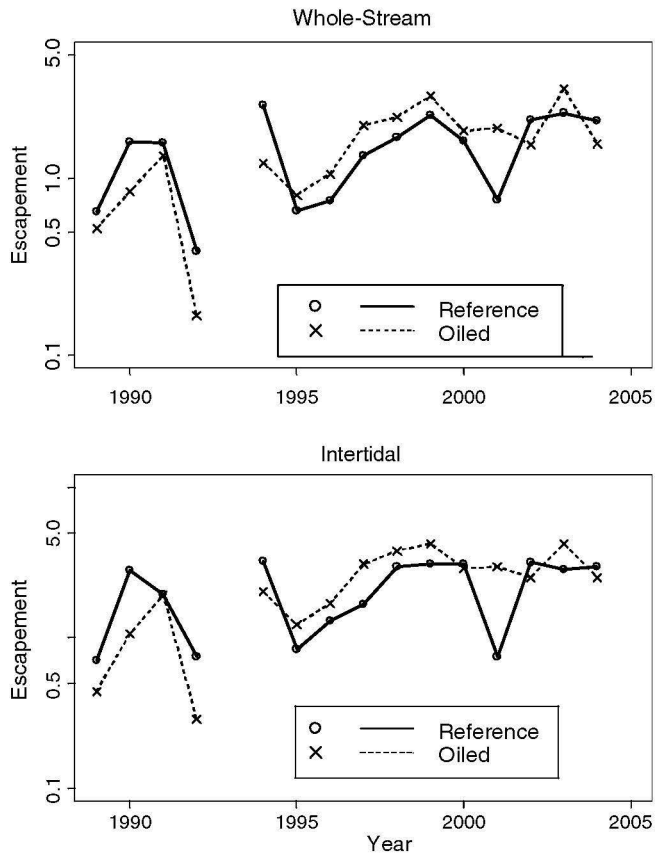

Fig. 4. Profiles of geometric mean escapements densities (numbers/m^2) for whole-stream (top) and intertidal (bottom) on log-scale. Time trajectories in escapement densities did not significantly differ for oiled and reference categories of streams at $\alpha = 0.05$. Stream surveys were not conducted in 1993.

4. Discussion

The analysis of PWS pink salmon adult return performance is conducted under the premise that annual environmental influences on populations are common across the ten survey streams of the study, even while inter-annual differences among streams can be quite large. The variability that occurs among brood year returns in PWS, and the dominant influence that marine factors have on pink

salmon abundance as reported by Cooney and Willette (1997) and Willette (1992), therefore, are accommodated by comparing profiles (annual changes) in mean adult abundance for oiled and non-oiled streams. In the absence of an oiling effect, profiles will parallel.

Commonality of stream populations within years is also evident by (1) the mixture of populations in migratory pathways before segregating to individual streams, (2) the proximity of streams in the study area, (3) the common weather patterns and similar temperature regimes of the streams, (4) the fact that major portions of the spawning areas are intertidal, (5) the segregated distribution of early fish in cooler upper stream reaches and later fish in the lower warmer stream areas, and (6) the common environment of the basin receiving emergent fry. Parallel profiles among reference streams supports these arguments. The environmental and proximate commonality of stream populations in PWS suggests that major perturbances due to oiling in the selected streams should be detectable in adult return performance.

4.1. Performance of populations in oiled and reference streams

We used three lines of evidence to assess adult returns to oiled and non-oiled streams. First, the null hypothesis for parallel profiles in mean spawner densities for oiled and reference categories of streams was not rejected for either whole-stream or intertidal areas. We conclude that parallel profiles demonstrate that escapements to oiled and non-oiled streams were similar in the years following the spill.

Second, similar profiles for intertidal and whole-stream areas provide evidence that oiled and non-oiled areas did not show differences in adult returns. This is noteworthy because upstream areas provided a non-oiled control within the same stream that experienced oil in the intertidal zone.

Third, we failed to reject the null hypothesis for parallel profiles of odd and even year populations. Assessment of odd and even-year data amounts to a long-term assessment of spawner-recruit relationships for odd and even year populations. If the 1989 spill was an influence on the stream populations, it would have affected the odd and even year populations differentially. Oiling in 1989 would have affected the 1988 brood progeny incubating in the streams at the time of the spill: effects on odd year progeny would have been on eggs incubating in highly weathered oil. Adults were not exposed to oil.

From these three lines of analysis we conclude there is no evidence that the oil spill had a detectable affect on PWS pink salmon runs. This conclusion is supported by Templin et al. (1996) who found no reduction in recruits per spawner in the Southwest district in 1990 and 1991. Templin et al. also concluded that the 1991 recruitment (from the 1989 brood year) in the southwest district was higher than the median sound-wide value, the opposite of what would have been expected if there was an impact of oil on eggs and alevins in 1989, the year of greatest oil contamination.

Further, our results are consistent with the ADFG model assessment (Geiger et al., 1996), and earlier assessments of Moulton (1996, 1998), Maki et al. (1995), and Brannon and Maki (1996). While the relatively small mortality estimated by Geiger was 2.3 million pink salmon out of a 1990–1994 multiyear population of 144 million returning adults, the mortality would have been confined only to the 30 streams in the southwest region that were oiled, and that would have represented a significant impact on those stream populations. However, our comparisons of population data in oiled and non-oiled streams, and in upstream vs. whole-stream have indicated densities and productivity to be indistinguishable between oiled and reference areas, and thus consistent with those studies that showed no oil-related incubation mortality occurred in southwestern PWS streams (Brannon et al., 1995, 2001).

These results are also consistent with Maki and Parker (1996) that showed spawner return density was unrelated to streambed sediment PAH concentrations in the spawning seasons immediately following the spill. Mean PAH levels measured in the sediments at the time of spawning (0.5–267 ppb) and in embryo tissue prior to emergence (<95) were very low (Brannon et al., 1995), and thus support the conclusion that in oiled streams PAH concentrations were not high enough to diminish adult return success.

4.2. Influence of strays on conclusions from PWS studies

Hatchery strays influenced the enumeration surveys of brood year spawners in some streams. Hatchery embryos were otolith marked, and Joyce and Evans (1999a,b) reported that hatchery strays contributed to spawning adults in PWS streams. Most of the hatchery strays are timed similar to or a little later than the native spawners. Consistent with Joyce and Evans, Collins (2002) has shown by sampling for otolith marks that hatchery fish make up the major component of some stream populations, and in some cases represent over 80% of the samples taken from the late segment of the runs.

Results were significant on the effect of strays on spawning density for only the intertidal area. Although enumeration observations have shown that the later returning hatchery fish (confirmed by otolith marks), are not confined to just the lower reaches of the streams, and thus add to the whole-stream totals, the higher proportion of strays in the intertidal zone may explain the lower p-values from our tests for intertidal over those for whole-stream. Based on significant results for the effect of strays in the intertidal area, the statistical model in Eq. (7) was rerun to test dist. × treatment in order to assess for potential differences in the effect strays had on oiled and reference streams; p-values were >0.30. The test for dist. × treatment was not run for wholestream since the effects of strays were not significant for the whole-stream.

We suggest that the demonstrated long-term parallel performance in adult returns in the oiled and reference study streams has also provided data relevant to the concerns about genetic damage or physiological injury in gametes of returning adults. The genetic damage contemplated by Bue et al. (1998) and physiological impairment suggested by Heintz et al. (1999) would have manifest themselves in adult return differences between oiled and reference stream reaches, and that wasn't apparent in the return data. Parallel return success for whole-stream and intertidal areas is consistent with the suspicion that the reported increased egg mortality, which was the primary source of mortality in the Geiger et al. (1996) model forecasting reduced adult return abundance, was caused by mechanical sampling shock of the eggs rather than oil effects (Collins et al., 2000; Brannon et al., 2001). This also conforms with the most recent analysis of ADFG, (Craig et al., 2002), where they concluded that other factors may have confounded their ability to determine the effect of oil on incubating pink salmon.

Our results also nullify some of the speculation about the spill. For example, using data from laboratory studies Peterson et al. (2003) concluded that sublethal effects from oiling on incubating eggs and on fry growth would cascade to a 50% reduction in adult returns. Our results over the sixteen year period of actual field studies and the earlier field results of Templin et al. (1996) demonstrate the high risk involved when attempting to apply laboratory results to ascertain effects of an environmental impact in the absence of field data.

5. Conclusions

The present paper is one of a series of studies that have been aimed at ascertaining the effects of the *Exxon Valdez* oil spill on the biological resources in PWS. The sixteen years of adult pink salmon survey data analyzed in this paper show that no effects of oiling were detected in the analysis of spawner density or recruits per spawner in spill-affected streams. These results are consistent with earlier work that showed the oil that reached southwestern PWS was weathered and relatively low in toxicity. These results are also consistent with studies that showed only low concentrations of toxic compounds were in the gravel due to freshwater flushing across the intertidal spawning reaches of spill affected streams. Parallel adult performance in oiled and reference streams supports the conclusion that the alleged oil related egg mortality, the ostensible factor affecting adult return success in the Geiger et al. (1996) model, was most likely caused by a problem in sampling error, rather than from oil, and thus there was no anomalous incubation mortality. We conclude from the present study that the timing and concentrations of *Exxon Valdez* oil that reached the incubation streams in PWS did not create detectable negative impacts on the incubating populations that would have translated into lower adult return performance of PWS pink salmon.

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