Vol. 248: 245–255, 2003 | MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser | Published February 20

Mechanism for transport of oil-contaminated groundwater into pink salmon redds

Mark G. Carls[1,*], Robert E. Thomas[2], Michael R. Lilly[3], Stanley D. Rice[1]

[1]US National Marine Fisheries Service, Auke Bay Laboratory, 11305 Glacier Hwy, Juneau, Alaska 99801, USA
[2]Department of Biological Sciences, California State University, Chico, California 95929, USA
[3]GW Scientific, PO Box 81538, Fairbanks, Alaska 99708, USA

ABSTRACT: Groundwater movement from oil-contaminated intertidal beaches to surface and subsurface water of salmon streams in Prince William Sound, Alaska, was studied to determine if transport of dissolved petroleum hydrocarbons to incubating pink salmon eggs (*Oncorhynchus gorbuscha*) was plausible. Beaches surrounding 31% of the streams in the Sound were extensively oiled in 1989; salmon egg mortality was elevated even though little oil was observed in stream gravel. In 2000, fluorescent tracer dyes injected into 2 of these beaches during ebb tides were subsequently observed throughout most of the intertidal portion of each watershed, including surface and subsurface (hyporheic) stream water. Mean horizontal groundwater flow was rapid through the porous gravel (4 to 7 m h^{-1}) and was driven by hydraulic gradients within beach groundwater. When different dyes were simultaneously released at ebb tide on opposite sides of a stream, each dye was detected in the beach opposite release within the first tidal ebb. Dye was moved vertically upward at least 0.5 m by subsequent incoming tides. Thus, tidal cycles and resultant hydraulic gradients provide a mechanism for groundwater transport of soluble and slightly soluble contaminants (such as oil) from beaches surrounding streams into the hyporheic zone where pink salmon eggs incubate.

KEY WORDS: Intertidal groundwater · Hydraulic gradient · Contaminant transport · Habitat damage · Pink salmon · Egg contamination · PAH

INTRODUCTION

The potential vulnerability of pink salmon *Oncorhynchus gorbuscha* habitat to pollution was evident after the 1989 'Exxon Valdez' oil spill in Prince William Sound (PWS), Alaska. Extensive areas of shoreline were coated with oil, including beaches surrounding the intertidal reaches of streams used by these fish (Brannon & Maki 1996, Murphy et al. 1999). These intertidal reaches are a critical spawning habitat for pink salmon; up to 75% of spawned eggs are deposited here (Helle et al. 1964, Heard 1991). Furthermore, approximately 31% of salmon streams in PWS were oiled (Geiger et al. 1996). Oil often penetrated deep into the gravel beaches these streams typically cross, creating the potential for years of chronic contamination (Michel & Hayes 1993). Flowing freshwater prevented oil deposition directly in stream channels. Despite this apparent protection of spawning habitat, cytochrome P4501A enzyme activity was elevated in alevins from streams in oiled areas but not in those from reference streams, suggesting that pre-emergent fish were exposed to oil (Weidmer et al. 1996).

The vulnerability of pink salmon stream habitat is controversial. Because flowing freshwater diverted oil from streams, Brannon et al. (1995) and Brannon & Maki (1996) argued that the oil concentrations in stream sediment were too low to damage pink salmon. These researchers reported that mean total polycyclic aromatic hydrocarbon (PAH) concentrations in the sediment of oiled streams were low, 0.5 to 2818 ng g^{-1} (but greater than in non-oiled reference streams, 0.2 to 64 ng g^{-1}). (Total PAH is defined as the sum of concentrations of 39 compounds ranging from naphthalene through benzo(ghi)perylene; e.g. Short et al. 1996). Except in the highest tide zone studied (3.7 m), Bran-

*Email: mark.carls@noaa.gov

non et al. (1995) failed to detect oil-related embryo mortality, but data were limited to a single year and included fewer eggs and less stream area than observed by Bue et al. (1996, 1998) and Craig et al. (1999). Studies by the latter authors had much greater statistical power (Rice et al. 2001) and reached opposite conclusions; that embryo mortality in these streams was elevated through 1993 and again in 1997 and that the key difference was oil. However, for oil to be causative, transport of hydrocarbons from heavily oiled sediment surrounding streams into pink salmon redds is necessary.

Flow of groundwater from surrounding beaches into stream beds (the hyporheic zone, where pink salmon eggs incubate), provides a plausible mechanism for the observed contamination (Heintz et al. 1999). 'Exxon Valdez' oil penetrated into sediment throughout much of the mid- to upper intertidal area surrounding oiled streams (ADFG 1989). These beaches are periodically covered with water due to tidal cycling, bringing water into contact with oiled substrate. PAH from oil-coated rock in contact with water dissolves into water (e.g. Marty et al. 1997, Short & Heintz 1997, Heintz et al. 1999). Because intertidal stream channels are eroded below beach surfaces, gravity-driven groundwater gradients lead to transport of potential contaminants from surrounding beaches into stream channels. Periodic pulses of oil-contaminated water could potentially contaminate the lipophylic salmon eggs buried in stream gravel while the gravel (which is not lipophylic) remains relatively uncontaminated. Similar groundwater flow has been demonstrated in creeks in tidal marshes and sandy beaches, with slower, less extensive flow in the less permeable marsh sediment (e.g. Agosta 1985, Harvey et al. 1987, Li et al. 1999). In marshes and sandy beaches, the hydraulic groundwater head corresponds to beach elevation on ebb tides, causing horizontal subsurface flow either seawards or towards stream channels (e.g. Lanyon et al. 1982, Harvey et al. 1987, Nuttle & Hemond 1988). Tidal movement in areas adjacent to rivers not only causes cross-shore fluctuations in groundwater, but also oscillating flows along the shore which can result in movement of contaminants from groundwater to river water (Li et al. 1999).

The objective of this study was to determine if groundwater from intertidal beaches flows into streambed gravel where pink salmon eggs incubate and the mechanism for this movement should it occur. Groundwater movement was determined by release of fluorescent tracer dyes, measurement of groundwater elevation as tides ebbed, and by relating changes in groundwater salinity and temperature to tidal cycles. Sediment texture was characterized in order to relate porosity and permeability to hydraulic conductivity, thus providing a framework for interpretation of groundwater movement and a way to compare study results with those in other environments, such as saltwater marshes.

Study area

The study was completed in June 2000 at 2 pink salmon streams in western PWS, Sleepy Creek on

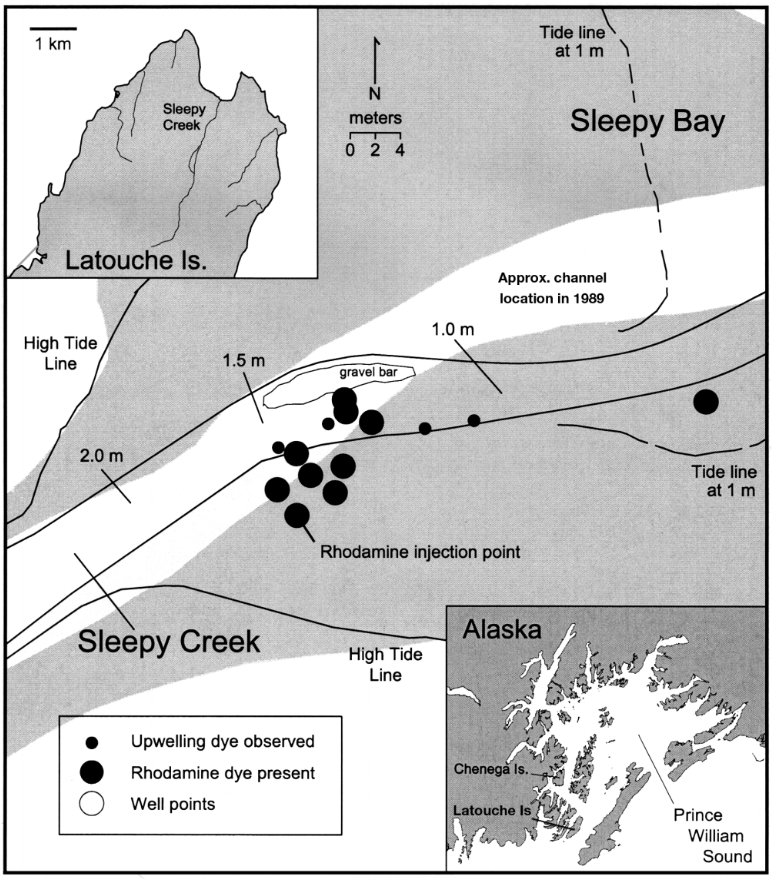

Fig. 1. Sleepy Creek, Latouche Island, Prince William Sound, Alaska: tracer dye detection area, wellpoint installations, and dye injection point. Large solid circles summarize dye detection across all observation times. Small dots indicate locations where dye upwelled from sediment into stream water (not all such observations were mapped). Vertical stream bed elevations are indicated (meters above mean lower low water). Shaded areas approximate oil distribution in 1989 (ADFG 1989)

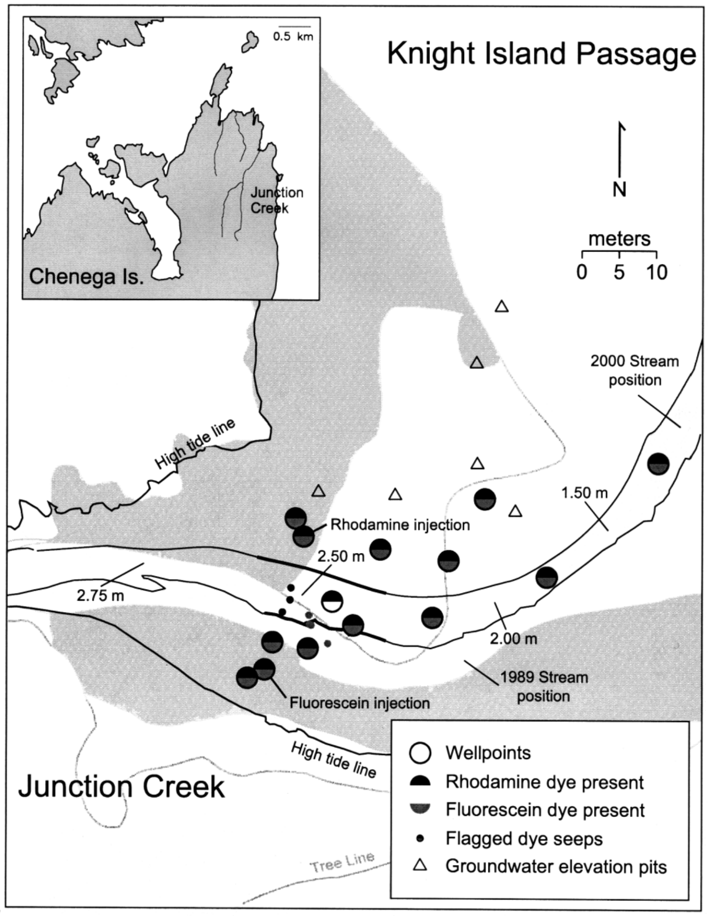

Fig. 2. Junction Creek, Chenega Island, Prince William Sound, Alaska: tracer dye detection area, wellpoint installations, dye injection points, and groundwater elevation pits. Rhodamine and fluorescein presence is summarized across all observation times. Small dots indicate locations where dye was observed upwelling from sediment into stream water (a minority of such observations were mapped). Vertical stream bed elevations are indicated (meters above mean lower low water). Shaded areas approximate oil distribution in 1989 (ADFG 1989)

Latouche Island and Junction Creek on Chenega Island (Figs. 1 & 2). Areas surrounding both streams were heavily oiled in 1989 (Gundlach et al. 1990) and studied by Brannon et al. (1995), Bue et al. (1996, 1998), Craig et al. (1999), and Murphy et al. (1999). Evidence of oil persisted in beaches around both streams through at least 1995 (Murphy et al. 1999), and pink salmon egg survival was monitored through 1997 (Bue et al. 1998, Craig et al. 1999). Dye injection wells were intentionally located in previously oiled beach areas and adjacent to reaches of stream previously monitored for egg survival. Recent maps were compared with preceding maps to describe temporal change. Beach sediment was well aerated at both sites; anoxia was never encountered in numerous pits dug to depths of 1 m.

Sleepy Creek

Sleepy Creek extends roughly 3.9 km south to north on the north end of Latouche Island (Fig. 1 inset). The stream leaves a bedrock channel just above the high tide line and angles northeast across a gravel beach at the south end of Sleepy Bay where large amounts of 'Exxon Valdez' oil penetrated sediment to depths of 50 to 125 cm (Hayes & Michel 1998). Asphalt-covered mousse was observed in intertidal sediment within 52 m of the stream at the time of our study (June 2000). Roughly two thirds of the sediment was greywacke and one third was shale. Water flow and volume were 1.0 ± 0.1 m s^{-1} and 1.25 m^3 s^{-1} where the stream entered the intertidal area. The mean stream gradient through the intertidal study area at Sleepy Creek was -2.5 ± 0.6 cm m^{-1} (range -4.1 to -0.9 cm m^{-1}). The stream channel was eroded 1.1 to 2.6 m below the surrounding beach; elevation differences were least in the lower intertidal area.

Junction Creek

Junction Creek extends primarily north for approximately 2 km to exit on the northeastern side of Chenega Island into Knight Island passage (Fig. 2 inset). The stream valley is fairly broad and Junction Creek passes through a marshy area (roughly 5000 m^2) as it enters a gravel beach. Oiling immediately surrounding Junction Creek ranged from light to heavy in 1989 and was mapped by the Alaska Department of Fish and Game (ADFG), but there was no visible evidence of this oil at the time of our study. Roughly two thirds of the sediment was shale and one third was greywacke. Stream flow and volume were 0.5 ± 0.1 m s^{-1} and 0.26 m^3 s^{-1} near the high tide line. The mean stream gradient through the intertidal study area at Junction Creek was -2.1 ± 0.4 cm m^{-1} (range -3.4 to -0.7 cm m^{-1}). The stream channel was eroded 0.9 to 2.1 m below the surrounding beach; elevation differences were least in the lower intertidal area.

MATERIALS AND METHODS

Wellpoints were installed in beach sediment and stream channels at Sleepy Creek and Junction Creek to trace groundwater movement (Fig. 3). Wellpoints, which had a hardened steel tip followed by a 60 cm

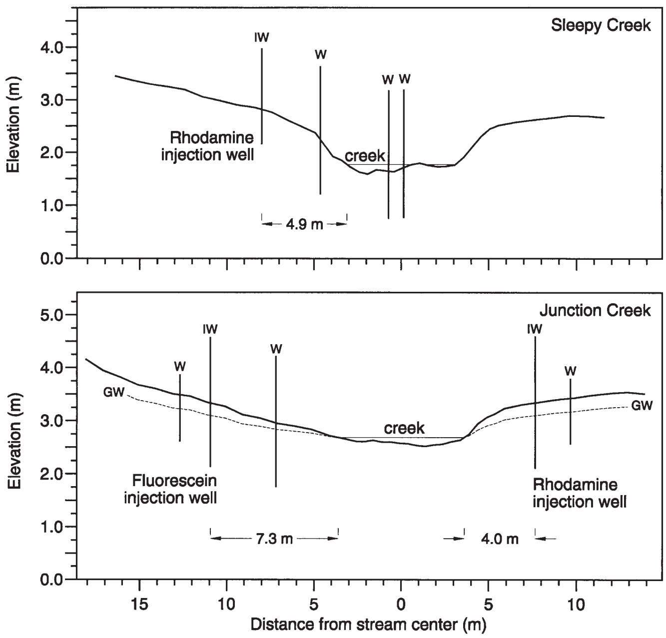

Fig. 3. Example cross-section profiles at Sleepy Creek (on Latouche Island) and Junction Creek (on Chenega Island), including injection wells (IW) and sample wells (W) (facing upstream). Distances from injection wells to stream margins are noted. An estimation of groundwater elevation (GW) is included for Junction Creek, calculated using equations relating beach surface and groundwater elevations (see text). Depicted stream surface elevations are at ebb tide

screened section, were threaded onto variable lengths of 3.2 cm diameter steel pipe. Polyethylene tubing, 1.3 cm in diameter, was threaded through the top to within approximately 8 cm of the bottom of the screened area. Wellpoint water samples were withdrawn from this tube. A second polyethylene tube to sample intermediate depths was extended along the outside of the pipe and ended 0.3 m below the sediment surface (for beach installations only). The end of this tube was packed in gravel to prevent plugging. Before rising tides flooded the wells, the central tubing was removed and the pipes were capped to prevent water entry from the upper end. Tubing was marked to ensure reinstallation in the original pipes at the proper depth.

Wellpoint installation at Sleepy Creek resulted in considerable local sediment disturbance because boulders were frequently encountered, but sediment disturbance was minimized at Junction Creek. Sleepy Creek pits were dug by shovel as deep as practical and wellpoints were hammered in as far as possible beyond the pit bottom. Pits were partially backfilled with original sediment, enough to cover the intermediate depth tube. Digging at Junction Creek was limited to 0.3 m, enough to place the intermediate collection tubes, and wellpoints were hammered to a constant depth of 1.2 m in beaches or 0.9 m in the stream channel. Pits were backfilled with original sediment. Wells uphill from each injection point did not have wellpoints and were installed by digging to a depth of 0.9 m. Rock was packed around the end of these 2 pipes to prevent plugging and the holes were refilled with original substrate.

Sleepy Creek was the first stream sampled, and the experience gained at this site helped us perform a more comprehensive survey at Junction Creek. Most of the wellpoints at Sleepy Creek were clustered within a 5 m radius, except one pipe placed 33.7 m downstream of the dye injection point (Fig. 1). The total area monitored was 130 m^2, including 29 m of stream channel. There were 4 wellpoints in the stream channel, 1 on the stream margin, and 5 in the beach on one side of the stream (including the injection well). Liquid rhodamine WT dye, 473 ml, specific gravity 1.03, was poured into the injection well on a falling tide at 14:10 h on June 14, 2000 (Figs. 1 & 3). Predicted high tide was 2.7 m at 13:12 h for Latouche Island. The injection well was located 7.9 m from the high tide line and 4.9 m from the stream. Fluorescence, temperature, and salinity sampling began before injection (at 12:37 h on June 14), and continued periodically through 17:00 h on June 15. Wellpoint and intermediate depth samples were collected by peristaltic pump; surface samples were dipped from water adjacent to pipes (when present). Fluorescence samples were put in 120 ml glass bottles with Teflon lids, immediately placed in the dark, kept cool, and periodically transferred to a support vessel for analysis. Temperature ($\pm$0.1°C) and salinity ($\pm$0.5%) were measured with a handheld meter. Sampling cycled through all available wells; the rate of sampling was controlled by the rate at which samples could be collected (about 3 min per sample $\times$ 3 samples per pipe) for 3 to 7 h periods (Fig. 4).

Both sides of the stream were equipped with wellpoints at Junction Creek and an area of 577 m^2 was monitored (Fig. 2). Four wells were placed on the south beach (including a fluorescein injection well), plus 1 on the stream margin. Five additional wells were installed on the north beach (including a rhodamine injection well). Four wells were placed in the stream channel, spanning a distance of 51 m. Both injection wells were installed at the same elevation, 3.3 m above mean lower low water (MLLW) (Figs. 2 & 3). Liquid rhodamine WT and fluorescein dyes, 473 ml each, specific gravity 1.01 to 1.03, were simultaneously injected on a falling tide at 03:48 h on June 18, 2000, just after the water level fell below these pipes. Predicted high tide was 3.6 m at 02:28 h for Chenega Island. The rhodamine injection well was about 14 m from the high tide line, 4.0 m from the stream, and 2.2 m inside the area identified by ADFG as heavily oiled in 1989 (Fig. 2). The fluorescein injection well was about 6 m from the high tide line, 7.3 m from the stream, and 0.4 m inside the area heavily oiled in 1989. Wells were periodically sampled as described for Sleepy Creek, except sample times were constrained to 2 falling and 1 rising tide cycle, and no surface samples were collected (Fig. 4). Analysis of fluorescence samples from Junction Creek was delayed until return to the laboratory, but they were stored in the dark and kept cool to ensure that the dye did not degrade before analysis.

Groundwater elevations during 2 receding tides were determined in 6 open pits in the north beach of Junction Creek (Fig. 2). Measurements were completed to the nearest 3 mm from reference marks of known elevation on pipes placed in each pit. Observations were cycled among accessible pits over a 2.3 to 2.7 h period starting when the tide began to ebb. Elevations at common times were determined by best-fit linear regression modeling; r^2 ranged from 0.988 to >0.999 for these estimates, except $r^2 = 0.968$ in 1 pit on June 18. The first measurement in this pit was apparently in error and was excluded from final analysis. Groundwater elevations at common times (for overlapping observations only) were regressed against corresponding beach elevations; best fit models were accepted. (Regression models included x-ladder of powers, y-ladder of powers, exponential and power).

Sediment grain size was characterized along transects placed near the wellpoints. A 36 m transect at Sleepy Creek was extended parallel to the stream and sediment was collected every 3 m from approximately 0.25 × 0.25 m × 0.25 m holes (n = 12). Each location was photographed prior to disturbance; dimensions of the largest rocks were determined from these photos. Masses of boulders too large for the balance to weigh were estimated by water displacement and a field-determined density of 2.78 g cm^{-3}. Sediment was collected every 5 m along two 25 m transects at Junction Creek, located on opposite sides ($n_{total} = 12$). Wet sediment was sieved through 16, 32, 63, and 100 mm mesh, weighed to the nearest 5 g, and discarded. The smallest fraction (<16 mm) was weighed (wet), collected, dried, passed through 0.125, 0.25, 0.5, 1, 2, 4, and 8 mm mesh, and also weighed to the nearest 5 g. Dry- and wet-weight ratios were determined for wet-sieved data and applied to allow combination of the 2 data sets. Assuming a potential maximum mesh size of 500 mm (based on rock size measurements), mid-grain size was estimated as described by Buchanan (1984); geometric means were the retransformed means of logarithmically transformed sizes. Porosity was estimated by placing 600 ml unconsolidated sediment subsamples (<16 mm fraction) in a 1 l graduated cylinder and measuring the volume of water required to fill the interstitial space. Porosity measures were repeated with consolidated sediment; ~100 ml aliquots of sediment were sequentially tamped into the cylinder until 600 ml were present; roughly 25 to 33 % more sediment was present than in unconsolidated tests. *In situ* sediment packing was not measured. Organic content was determined in the <16 mm fraction by drying samples at 77°C for several days, then combusting at 500°C for 5 h.

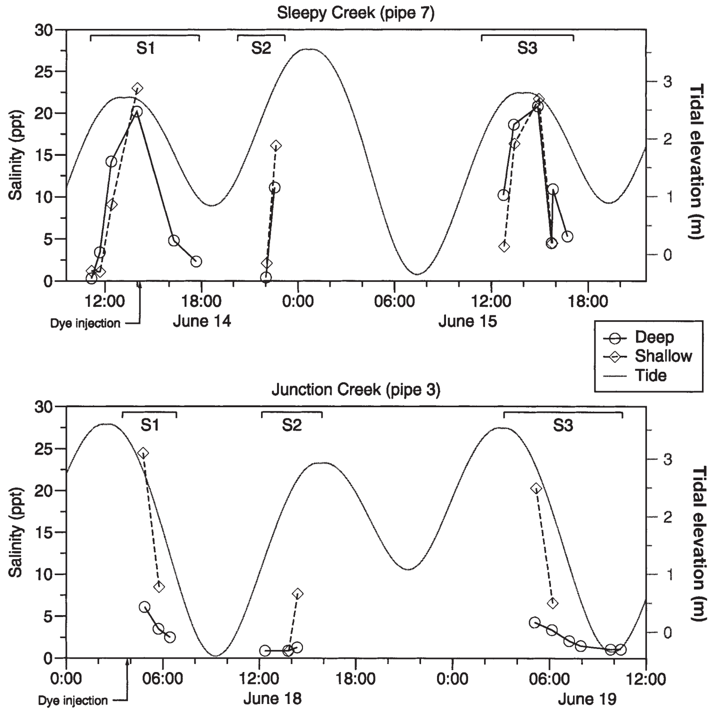

Fig. 4. Salinity in deep and shallow groundwater at Sleepy Creek and Junction Creek compared to the tidal cycle. Data were collected within 3 discrete time intervals (S1 to S3) at each site

Fluorescence analysis. A filter fluorometer (Model 10AU, Turner Designs) was used to determine the presence of rhodamine WT (Fluorescent FWT, Red) and fluorescein (Fluorescent FLT, Yellow/Green) in water samples. Excitation and emission wavelengths were 550 ± 10 and >570 nm for rhodamine, and 486 ± 10 and 510–700 nm for fluorescein. Samples were centrifuged at 600 × *g* for 3 min to remove suspended material, decanted into 25 × 150 mm test tubes, and equilibrated to 21°C in a water bath before measurement. pH, determined for each sample at the time fluorescence was measured, averaged 7.8 at Sleepy Creek (range 7.1 to 8.4), and 7.6 (range 6.8 to 8.6) at Junction Creek. Rhodamine WT, 5% by weight, and fluorescein, 7.5% by weight, dyes were obtained in liquid form from Kingscote Chemicals. Standard solutions, prepared by serial dilution of stock solutions with distilled water and also equilibrated to 21°C, were used for calibration and to determine when quenching could be expected. Background fluorescence was subtracted from sample fluorescence using blank solutions collected intermittently in the field. Water samples were collected in duplicate at each sampling; paired fluorescence values were averaged before further data analysis. Mean coefficient of variation in paired samples with detectable fluorescence was 25%: concentrations >0.005 ppb were considered detectable. Only a few intensely colored injection well samples required dilution to avoid quenching. Analysis of samples where both dye colors were present did not result in false detections.

Mapping. Streams were mapped to describe the sites and for comparison with post-'Exxon Valdez' spill conditions (ADFG 1989). Primary mapping was by triangulation from fixed points; mapped points were subsequently verified with a global positioning system including ground-based differential correction (20 cm accuracy; July 2000). Additional upstream detail and high tide positions were digitized from an aerial photograph of Junction Creek (May 2000); scaling was determined by overlaying the map on the photograph. A 1989 Junction Creek map (ADFG 1989) was corrected for an apparent 10% north-south optical distortion (also determined by map overlay) and combined with our map (Fig. 2). Our Sleepy Creek field map was similarly overlaid on a 1989 map (ADFG 1989), but correspondence between current and past maps is likely less accurate because not all fixed details could be matched exactly.

Vertical elevations of well points, stream stations, and stream cross-sections were determined with a surveyor's level. The elevation of a fixed reference point was determined with respect to known low tide levels, and periodically rechecked to ensure that the level remained calibrated. Five cross-sections were completed at Sleepy Creek, and 7 at Junction Creek. The elevation of the lowest points in the stream channel were estimated from cross-section profiles.

Reported distances and areas were determined from final map positions. Estimated areas are based on projection to a flat plane and do not account for changes in surface topography. Stream gradients (change in elevation/change in distance) were estimated by determining the distance between lowest channel points along cross-sections.

Groundwater flow rates. First arrival times of tracer dyes at known distances allowed estimation of groundwater flow rates. First arrival was recorded exactly for a subset of dye plumes emanating from stream gravel at measured distances from appropriate dye injection points. The time of first dye arrival in fixed well points was also determined, but these rate estimates are conservative because first detection of the dye was dependent on sampling frequency and exact arrival times were unknown. Data were first analyzed by 3-way ANOVA (dye color, stream, and vertical placement). Classifications were simplified for final analysis where initial statistical results were clearly not significant ($p > 0.5$; e.g. vertical placement was dropped at Junction Creek).

Hydraulic conductivity (K) was estimated in 2 ways, from grain size analysis and from tracer dye movement (Krumbein & Monk 1942, Millham & Howes 1995, Baird & Horn 1996). Grain size data were used to compute intrinsic permeability, from which K was estimated: $K = 760\ (GM_d)^2\ e^{-1.31\sigma}\ \rho_w\ g\upsilon^{-1}$, where GM_d = the geometric mean grain diameter (mm), σ = the standard deviation in phi units [$-\log_2$(diameter in mm)], ρ_w = density of water (0.999849 to 0.999947 g cm^{-3}), g = acceleration due to gravity (980.6 cm s^{-2}), and υ = viscosity of water (0.01386 to 0.01481 g s^{-1} cm^{-1}). Observation of dye movement allowed *in situ* calculations: $K = vn_c\ (dh/dl)^{-1}$, where v = dye velocity (m h^{-1}), n_c = consolidated porosity (0.32 to 0.35, dimensionless), and dh/dl = hydraulic gradient (0.03 to 0.27, dimensionless). Estimates of dh/dl were based on an observed correlation between beach elevation and groundwater elevation.

RESULTS

Drainage area affected by tracer dyes

Fluorescent tracer dye was detected in groundwater within a single tidal cycle throughout most of the area monitored at both streams, including groundwater in banks opposite of dye release (Figs. 1 & 2). Dye upwelled from stream beds into stream water within 0.5 h and colored plumes washed downward into the bays. At Sleepy Creek, rhodamine was detected in all wellpoints in place during the first ebb tide ($n = 8$). At Junction Creek, rhodamine was detected in 8 of 10 wellpoints during the first tidal outflow but fluorescein was detected in just 3 wellpoints. During the first tidal outflow, fluorescein was also detected in 3 of 10 shallow pits on the bank opposite release and 8 of 12 pits on the same side: rhodamine was also detected in both banks at this time (Fig. 5). By the time the first tide cycle (ebb plus flood) was complete at Junction Creek, rhodamine had been detected in 9 of 11 wellpoints, and fluorescein in 10 of 11 wellpoints.

Ultimately, single-point dye releases were detected in groundwater throughout most of the area surveyed for dye presence including groundwater in beaches and groundwater below both stream channels (Figs. 1 & 2). In the deeper Junction Creek samples, rhodamine was detected over the entire area monitored,

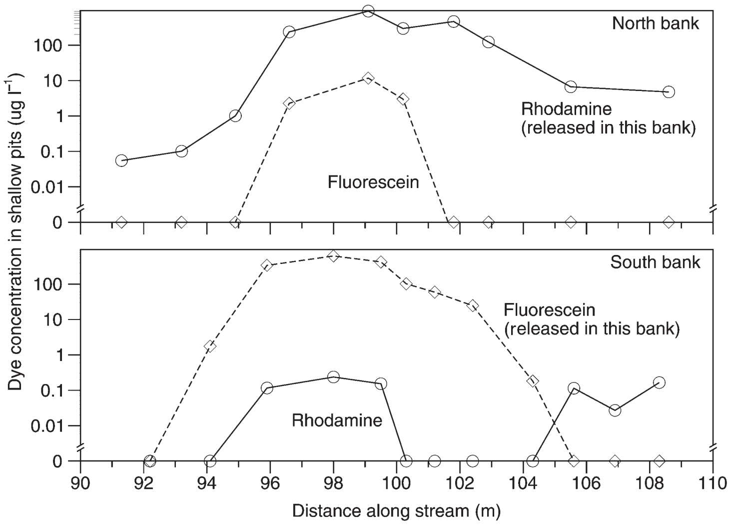

Fig. 5. Tracer dye concentrations in shallow pits located on north and south stream banks immediately adjacent to stream water at Junction Creek. Rhodamine dye was released in the north bank, fluorescein in the south; pit samples were collected 3 h later. Pits were positioned linearly in the vicinity of 2.50 m MLLW (along thickened stream margin illustrated in Fig. 2)

577 m^2, including all 61 m of monitored stream channel. Similarly, fluorescein was detected over a 438 m^2 area, including 57 m of stream channel. Dyes were frequently detected in groundwater on the side opposite of release at Junction Creek, and generally co-occurred below stream channels. At Sleepy Creek, where the area monitored was smaller and constrained to a single bank plus stream bed, dye was detected in groundwater throughout the entire area monitored, 130 m^2, and extended at least halfway under the stream channel (the furthest extent observed). Dye was also detected at all monitored intermediate and surface depths at this site, indicating vertical water movement. At Junction Creek, dye was usually, but not always, found at intermediate depths where that particular dye was detected in deeper sediment. Dye was transported vertically at least 0.5 m uphill by incoming tides; uphill concentrations were moderately high (9 ppb rhodamine, 22 ppb fluorescein).

Sediment characteristics

Beach sediment surrounding each stream was coarse (Table 1). At Sleepy Creek, cobbles and boulders comprised 46 ± 5% of the mass (n = 12). Estimated mid-grain size ranged from 39 to 375 mm; little sediment was ≤0.25 mm (0.3 ± 0.1%). Mean porosity estimates ranged from 32 to 40% (±3%); permeability was $1.6 \times 10^{-5} \pm 5.4 \times 10^{-6}$ cm^2. At Junction Creek, the frequency of cobbles and boulders on the south beach was 34 ± 4% (n = 6), but fewer were present on the north beach (9 ± 2%, n = 6). Estimated mid-grain size ranged from 18 to 108 mm; little sediment was ≤0.25 mm (0.6 ± 0.1%). Mean porosity estimates ranged from 35 to 46% (±0.7%); permeability was $2.4 \times 10^{-6} \pm 3.5 \times 10^{-7}$ cm^2.

Groundwater movement was rapid

Horizontal groundwater flow, estimated by timing tracer dye movement, was rapid (1.3 to 12.4 m h^{-1}) through the porous beach substrate (Table 1). Dye began upwelling from stream beds into stream water 23 to 34 min after subsurface injection into beaches at Junction Creek, and 26 min after injection at Sleepy Creek. The mean flow rate at Sleepy Creek, 4.5 ± 1.2 m h^{-1} (n = 7), was less than at Junction Creek, 7.0 ± 0.7 m h^{-1} (n = 11, p = 0.054). The type of dye did not influence rate estimates (p = 0.940). Flow rate estimates from wellpoints and shallow groundwater samples at Junction Creek were clearly similar (p = 0.505) and were combined in this analysis. Flow rate estimates based on seepage into the stream channel at Junction Creek were higher, 9.5 ± 0.8 m h^{-1} (n = 7, p = 0.059).

Estimates of hydraulic conductivity (a measure of the ability of sediment to transmit water) ranged from 0.05 to 4.5 cm s^{-1}. Estimates based on permeability (0.05 to 4.15 cm s^{-1}) were very similar to estimates based on dye movement and consolidated sediment porosity (0.06 to 4.5 cm s^{-1}). These estimates were within the range predicted by Freeze & Cherry (1979) for clean sand and gravel and indicate great capacity for fluid transmission.

Table 1. Summary of sediment characteristics

	Sleepy Creek	Junction Creek
Organic content (%)	1.5 ± 0.1	0.7 ± 0.1
Porosity		
Unconsolidated (%)	40 ± 0.3	46 ± 0.7
Consolidated (%)	32 ± 0.3	35 ± 0.3
Permeability (cm^2)	1.6×10^{-5}	2.4×10^{-6}
Geometric mean grain size (mm)	57	25
Cobbles and boulders (≤63 mm) (%)	46 ± 5	22 ± 4
Fine sediment (≤0.25 mm) (%)	0.3 ± 0.1	0.6 ± 0.1
Hydraulic conductivity (cm s^{-1})		
In situ estimate	0.4 ± 0.3	1.6 ± 0.4
Estimated from permeability	1.1 ± 0.4	0.2 ± 0.02

Mechanisms

The primary mechanism of dye dispersal was groundwater movement, evident because dye concentrations were consistently greater in groundwater than in paired water samples collected above the sediment. These paired observations were limited to Sleepy Creek, where dye concentrations in surface water averaged 7 ± 6% of concentrations in corresponding wellpoints (range 0 to 91%, n = 16). Colored surface water flushed into bays where it was diluted and there was no evidence that dye in surface water recontaminated study areas during subsequent flood tides.

Tidal cycles, the ability of sediment to transmit water, and gravity resulted in hydraulic gradients within these intertidal beaches, which in turn was responsible for lateral water movement. Groundwater elevations declined as tides receded, but were correlated with beach elevations ($0.89 \leq r^2 \leq 0.999$), thus creating a hydraulic gradient (Figs. 3 & 6). The downslope direction of these hydraulic gradients caused groundwater to flow toward stream channels, as previously illustrated by dye movement. Although not quantified, hydraulic gradients were clearly reversed during flood tides, as demonstrated by the upslope transport of fluorescent dye.

Salinity and temperature

Salinity and temperature in groundwater increased rapidly with incoming tides and declined with outgoing tides, further evidence of groundwater movement and demonstrating the dynamic between salt and freshwater sources (e.g. Fig. 4). Wellpoint salinity ranged from 0.1 to 27.6 at Sleepy Creek and 0.4 to 24.9 at Junction Creek, but was sufficiently site-specific that individual patterns were obscured when all data were combined (for each creek), even when they were grouped by closely adjacent wells. Overall salinity ranges in stream water (0.0 to 27.6 at Sleepy Creek and 0.0 to 32.6 at Junction Creek) were slightly larger than in wellpoints, but closely overlapped them. Salinity at high tide at Junction Creek was generally lower 1.2 m below the surface (mean 11, range 4 to 22) than at 0.3 m (mean 22, range 19 to 26) ($p < 0.001$), but at Sleepy Creek salinity tended to be higher in the deepest groundwater than in shallow groundwater or surface water.

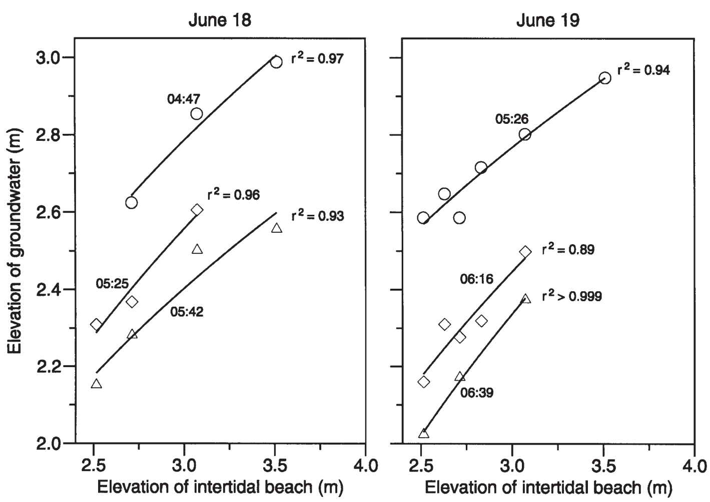

Fig. 6. Relationship between beach elevation and groundwater elevation during ebb tides at Junction Creek. Time of observation is printed by each regression; independent data (beach elevation) were log-transformed. (○) First daily observation; (◇) second; (△) third. High tide was at 02:28 h on June 18 and 03:03 h on June 19, 2000

DISCUSSION

Tidal motion and hydraulic gradients provide a mechanism for groundwater transport of soluble and slightly soluble contaminants (such as oil) in beaches surrounding streams into streambed water and gravel where pink salmon eggs incubate. Fluorimeteric tracer dyes injected into gravel beaches surrounding streams during ebb tides were rapidly observed throughout most of the intertidal portion of each watershed, including water below stream channels. Within half an hour, dyes were observed upwelling from streambed gravel into stream water. Mean horizontal groundwater flow rates were rapid (4 to 7 m h^{-1}), as expected for sand and gravel beaches with high hydraulic conductivity, and were driven by hydraulic gradients within beach groundwater. Dyes were consistently detected in wellpoints located in the hyporheic zone and in surrounding beaches. At Junction Creek, where different dyes were simultaneously released on opposite sides, each dye was detected in the beach opposite release within the first tidal ebb (3 h). Dyes remained detectable in subsequent tidal cycles (observations were discontinued after 2 or 3 cycles) and dye was forced vertically uphill by tidal action (0.5 m elevation increase, the upper extent of observations).

Groundwater elevations, measured only at Junction Creek, increased as sediment elevation increased, providing evidence of the hydraulic gradient responsible for lateral subsurface water movement. This observation is consistent with water table observations in intertidal salt marshes (e.g. Agosta 1985, Nuttle & Harvey 1988, Howes & Goehringer 1994), in sandy beaches (e.g. Ericksen

1970, Lanyon et al. 1982), and unconfined coastal aquifers (e.g. Todd 1964). When a beach surface is flooded, infiltration from an essentially unlimited source of water fills the sediment to capacity and the depth-averaged head becomes equal to the elevation of the sediment surface (Nuttle & Hemond 1988). When the surface is exposed, the hydraulic head corresponds to sediment elevation, thus the greatest horizontal water movement occurs where bank slopes are steepest (Harvey et al. 1987, Nuttle & Hemond 1988). Harvey et al. (1987) found that subsurface flow was primarily horizontal, occurred when marsh surfaces were not flooded, and was directed toward creek banks. Our results, both visual and fluorimetric, clearly indicate lateral and downslope movement, but also demonstrate that groundwater can be transported uphill by incoming tides when the direction of the hydraulic gradient is temporarily reversed.

The rapid horizontal water flow at our study sites (4 to 7 m h^{-1}) indicates that potentially all groundwater in each intertidal drainage basin might enter the stream during typical tidal cycles. At Sleepy Bay, estimated drainage distance during a typical tide cycle is 48 m at +3 m MLLW (10.8 h mean emergence time × 4.5 m h^{-1}). Potential drainage distance at low elevations are smaller, e.g. 22 m at +1.6 m MLLW. This suggests that groundwater throughout the entire intertidal Sleepy Creek drainage basin may potentially flow into the stream as tides ebb, an area >1300 m^2. Similarly, groundwater can potentially drain from the entire intertidal portion of the drainage basin at Junction Creek into the stream channel during a typical tide cycle. The capacity for groundwater drainage in these gravel beaches ($5 \times 10^{-2} \le K \le 4 \times 10^{0}$ cm s^{-1}) is greater than in saltwater marshes where sediment is fine grained and hydraulic conductivity is correspondingly lower ($2 \times 10^{-6} \le K \le 7 \times 10^{-3}$ cm s^{-1}) and drainage may be limited to within 4 to 15 m of creek banks (Agosta 1985, Nuttle & Harvey 1988, Nuttle & Hemond 1988). The estimated hydraulic conductivity range for sandy shorelines, 2×10^{-2} to 9×10^{-2} cm s^{-1} (Lanyon et al. 1982, Millham & Howes 1995), overlaps the lower portion of our observed range.

Implications for eggs of anadromous salmon

Habitat characteristics selected by pink salmon for spawning are the same characteristics that facilitate exposure of eggs to contaminants in surrounding beaches. The porous gravel beach sediment surrounding the 2 anadromous salmon streams observed in this study typifies the substrate surrounding anadromous salmon streams in Prince William Sound and is the habitat selected by pink salmon for spawning. For successful incubation, pink salmon eggs require good intragravel flow of oxygenated water; thus coarse gravel with little silt in high-flow areas are utilized (Heard 1991). Pink salmon spawning habitat is remarkably similar throughout western PWS and hydraulic conductivity is undoubtedly similarly high throughout this habitat. The coarse nature of the spawning substrate, placement of eggs in gravel below stream channels (20 to 25 cm depths are typical, Helle et al. 1964, Heard 1991), and the high frequency of intertidal spawning place these eggs at risk of exposure to any contaminants that might be contained in surrounding beaches.

The flow of groundwater containing PAH dissolved from oiled intertidal beaches surrounding anadromous streams into the hyporheic zone explains how pink salmon eggs became contaminated after the 'Exxon Valdez' oil spill. Although flowing freshwater did not allow oil to directly contaminate stream channels, shorelines surrounding both study streams and many others in western PWS were coated with oil (Brannon & Maki 1996, Bue et al. 1996, 1998, Geiger et al. 1996, Craig et al. 1999, Murphy et al. 1999). Aliphatic and aromatic hydrocarbons dissolve into water surrounding oil-coated rock at rates related to the energy required for escape (Short & Heintz 1997), and the resultant contamination is sufficient to cause toxicity (Marty et al. 1997, Heintz et al. 1999, 2000). Dissolved PAH are most likely responsible for this toxicity, which increases with molecular weight (Rice et al. 1977, Neff 1979, Black et al. 1983, Heintz et al. 1999, 2000). Murphy et al. (1999) detected high molecular weight PAH in sediment surrounding streams in PWS and concluded that concentrations at some streams were likely toxic through 1993. For example, mean total PAH concentrations in intertidal sediments adjacent to 7 oiled spawning streams in 1989 ranged from 13 to 823 µg g^{-1} (Murphy et al. 1999), greater than concentrations known to damage pink salmon embryos (3.8 µg g^{-1}, Heintz et al. 1999). Of primary toxic concern are naphthalenes through chrysenes, which dissolve into water in contact with oiled sediment. Larger molecular weight PAH are rare or absent in Alaska North Slope crude oil (e.g., Short et al. 1996) and monoaromatics generally evaporated before shoreline oiling (Wolfe et al. 1994). The presence of PAH in the hyporheic zone allows pink salmon eggs to accumulate PAH, which explains why Weidmer et al. (1996) observed elevated cytochrome P4501A levels in eggs from oiled streams. Because these eggs are lipophilic, they accumulate much higher hydrocarbon concentrations than are present in the surrounding water (e.g. Heintz et al. 2000). In contrast, stream gravel, which is not lipophilic, is not likely to accumulate dissolved hydrocarbons and this explains why Brannon & Maki (1996) found relatively few contaminant hydrocarbons in gravel from these streams. Reduced embryo survival resulted from exposure through 1993 (Bue et al. 1996, 1998). Mortality between oiled and reference streams did not differ significantly from 1994 to 1996, but survival was lower again in 1997 (Craig et al. 1999). Shifts in stream courses may have increased the flow of dissolved hydrocarbons into stream water; studies in 2000 and 2001 continued to find persistent oil at many sites within PWS (Rice et al. 2001) and 'Exxon Valdez' oil was detected in the intertidal hyporheic zone at Sleepy Creek in 1999 (M. Carls unpubl. data).

Shifts in intertidal stream channels are not unusual and increase the potential for transport of contaminants into stream water and stream beds. Michel & Hayes (1993) report that Sleepy Creek constantly shifts position: estimates from their maps suggest 1 to 11 m annual movement is possible in the vicinity of our study (Michel & Hayes 1994). An anchor installed at +2 m MLLW in Sleepy Creek in the fall of 1999 could not be located in June 2000, apparently because the stream had shifted approximately 4 m and deposited a thick layer of gravel over it. Changes at Junction Creek offered more definitive evidence of short-term channel shifts; anchors placed at +2 m MLLW in the stream center in August 1999 were on the bank in June 2000, indicating that the stream channel had moved approximately 3 to 4 m. A longer-term change in the Junction Creek channel position is evident by comparing the 1989 position mapped by ADFG with the 2000 position (Fig. 2). Large-scale stream shifts may increase the potential for contaminant transport by functionally increasing the area of the drainage basin. At smaller scales, reworking of contaminated sediment may bring such material in direct contact with the stream channel and mechanical movement may expose fresh surfaces on oil-coated material, accelerating the weathering process and increasing downstream and hyporheic contaminant concentrations.

Conclusions

Groundwater in intertidal gravel beaches surrounding streams drains into stream water and the hyporheic zone below streams during ebb tides. Direct evidence that groundwater from intertidal beaches enters stream water and the hyporheic zone was provided by tracer dyes. Secondary evidence of groundwater water movement included (1) lack of anoxia in beach sediment, (2) the cyclic presence and absence of subsurface water, and (3) cyclic changes in temperature and salinity of groundwater corresponding to tidal cycles. Lateral groundwater flow, driven by hydraulic gradients related to watershed topography and tidal cycles, may extend throughout the entire intertidal portion of drainage basins during each ebb tide because hydraulic conductivity is high in gravel beaches, hence flow rates are rapid. Drainage of groundwater from beaches into surface and subsurface stream water provides a mechanism for contaminant exchange, and together with results from previous publications that demonstrate water flowing through oiled gravel accumulates toxic hydrocarbons, explains how pink salmon eggs in Prince William Sound became contaminated by 'Exxon Valdez' oil. The stream substrate characteristics selected by spawning adult pink salmon and required for successful incubation of eggs (including high porosity and water exchange) are the same characteristics that permitted contaminated water ready access to these eggs.

Acknowledgements. We thank J. & K. Mendez for field assistance, D. & A. Janka for vessel support, and Dr. Bruce Richmond (USGS) for manuscript review. We thank the 'Exxon Valdez' Oil Spill Trustee Council for support of this research. However, the findings and conclusions presented by the authors are their own, and do not necessarily reflect the view or position of the Trustee Council.

LITERATURE CITED

ADFG (1989) Alaska Department of Fish and Game stream survey maps. Cordova, AK

Agosta K (1985) The effect of tidally induced changes in the creekbank water table on porewater chemistry. Estuar Coast Shelf Sci 221:389–400

Baird AJ, Horn DP (1996) Monitoring and modelling groundwater behaviour in sandy beaches. J Coast Res 12: 630–640

Black JA, Birge WJ, Westerman AG, Francis PC (1983) Comparative aquatic toxicology of aromatic hydrocarbons. Fund Appl Toxicol 3:353–358

Brannon EL, Maki AW (1996) The 'Exxon Valdez' oil spill: Analysis of impacts on the Prince William Sound pink salmon. Rev Fish Sci 4:289–337

Brannon EL, Moulton LL, Gilbertson LG, Maki AW, Skalski JR (1995) An assessment of oil-spill effects on pink salmon populations following the 'Exxon Valdez' oil spill—Part I: Early life history. In: Wells PG, Butler JN, Hughes JS (eds) 'Exxon Valdez' oil spill: fate and effects in Alaskan waters. ASTM STP 1219, American Society for Testing and Materials, Philadelphia, p 548–584

Buchanan JB (1984) Sediment analysis. In: Holme NA, McIntyre AD (eds) Methods for the study of marine benthos. Blackwell Scientific Publications, Boston, p 41–65

Bue BG, Sharr S, Moffitt SD, Craig A (1996) Effects of the 'Exxon Valdez' oil spill on pink salmon embryos and preemergent fry. Am Fish Soc Symp 18:619–627

Bue BG, Sharr S, Seeb JE (1998) Evidence of damage to pink salmon populations inhabiting Prince William Sound, Alaska, two generations after the 'Exxon Valdez' oil spill. Trans Am Fish Soc 127:35–43

Carls MG, Rice SD, Hose JE (1999) Sensitivity of fish embryos to weathered crude oil: Part 1. Low level exposure during incubation causes malformations and genetic damage

in larval Pacific herring (*Clupea pallasi*). Environ Toxicol Chem 18:481–493
Craig AK, Willette M, Evans DG, Bue BG (1999) Injury to pink salmon embryos in Prince William Sound—Field monitoring. Restoration Report 98191A-1 'Exxon Valdez' Oil Spill Final Report. Alaska Department of Fish and Game, Anchorage
Erikson NJ (1970) Measurement of tide induced changes to water table profiles in coarse and find sand beaches along Pegasus Bay, Canterbury. Earth Sci J 4:24–31
Freeze RA, Cherry JA (1979) Groundwater. Prentice-Hall, Englewood Cliffs, NJ
Geiger HJ, Bue BG, Sharr S, Wertheimer AC, Willette TM (1996) A life history approach to estimating damage to Prince William Sound pink salmon caused by the 'Exxon Valdez' oil spill. Am Fish Soc Symp 18:487–498
Gundlach ER, Bauer J, Bayliss R, Provant S, Kendziorek M (1990) Response to the 'Exxon Valdez' oil spill by the Alaska Department of Environmental Conservation. In: Spaulding ML, Reed M (eds) Oil spills management and legislative implications: Proceedings of the conference, May 15–18, 1990, Newport, Rhode Island, USA. American Society of Civil Engineers, New York, p 471–478
Harvey JW, Germann PF, Odum WE (1987) Geomorphological control of subsurface hydrology in the creekbank zone of tidal marshes. Estuar Coast Shelf Sci 25:677–691
Hayes MO, Michel J (1998) Evaluation of the condition of Prince William Sound shorelines following the 'Exxon Valdez' oil spill and subsequent shoreline treatment: 1997 Geomorphological monitoring survey. US Dept Commer, NOAA Tech Memo NOS ORCA 126
Heard WR (1991) Life history of pink salmon (*Oncorhynchus gorbuscha*). In: Groot C, Margolis L (eds) Pacific salmon life histories. Department of Fisheries and Oceans, Biological Sciences Branch, Pacific Biological Station, Nanaimo, British Columbia, Canada. University of British Columbia Press, Vancouver, p 121–230
Heintz R, Short JW, Rice SD (1999) Sensitivity of fish embryos to weathered crude oil: Part II. Increased mortality of pink salmon (*Oncorhynchus gorbuscha*) embryos incubating downstream from weathered 'Exxon Valdez' crude oil. Environ Toxicol Chem 18:494–503
Heintz RA, Rice SD, Wertheimer AC, Bradshaw RF, Thrower FP, Joyce JE, Short JW (2000) Delayed effects on growth and marine survival of pink salmon *Oncorhynchus gorbuscha* after exposure to crude oil during embryonic development. Mar Ecol Prog Ser 208:205–216
Helle JH, Williamson RS, Bailey JE (1964) Intertidal ecology and life history of pink salmon at Olsen Creek, Prince William Sound, Alaska. US Fish Wild Serv Spec Sci Rep Fish 483
Howes BL, Goehringer DD (1994) Porewater drainage and dissolved organic carbon and nutrient losses through the intertidal creekbanks of a New England salt marsh. Mar Ecol Prog Ser 114:289–301
Krumbein WC, Monk GD (1942) Permeability as a function of the size parameters of unconsolidated sand. Trans Am Inst Mining Metallurgical Engineers 151:153–163
Lanyon JA, Eliot IG, Clarke DJ (1982) Groundwater-level variation during semidiurnal spring tidal cycles on a sandy beach. Aust J Mar Freshw Res 33:377–400
Li L, Barry DA, Stagnitti F, Parlange JY (1999) Tidal alongshore groundwater flow in a coastal aquifer. Environ Modeling Assess 4:179–188
Marty GD, Short JW, Dambach DM, Willits NH, Heintz RA, Rice SD, Stegeman JJ, Hinton DE (1997) Ascites, premature emergence, increased gonadal cell apoptosis, and cytochrome P4501A induction in pink salmon larvae continuously exposed to oil-contaminated gravel during development. Can J Zool 75:989–1007
Michel J, Hayes MO (1993) Evaluation of the condition of Prince William Sound shorelines following the 'Exxon Valdez' oil spill and subsequent shoreline treatment. Vol. 1: Summary of results—geomorphological shoreline monitoring survey of the 'Exxon Valdez' spill site, Prince William Sound, Alaska September 1989–August 1992. US Dept Commer, NOAA Tech Memo NOS ORCA 73
Michel J, Hayes MO (1994) Evaluation of the condition of Prince William Sound shorelines following the 'Exxon Valdez' oil spill and subsequent shoreline treatment. Vol. II: 1994 Geomorphological monitoring survey. NOAA Tech Memo NOS ORCA 91
Millham NP, Howes BL (1995) A comparison of methods to determine K in a shallow coastal aquifer. Ground Water 33:40–57
Murphy ML, Heintz RA, Short JW, Larsen ML, Rice SD (1999) Recovery of pink salmon spawning areas after the 'Exxon Valdez' oil spill. Trans Am Fish Soc 128:909–918
Neff JM (1979) Polycyclic aromatic hydrocarbons in the aquatic environment. Sources, fates, and biological effects. Applied Science Publishers, London
Nuttle WK, Harvey JW (1988) Geomorphological controls on subsurface transport in two salt marshes. In: Kusler JA, Brooks G (eds) Proceedings of the National Wetland Symposium: Wetland hydrology, September 16–18, 1987. Association of State Wetland Managers, New York, p 253–261
Nuttle WK, Hemond HF (1988) Salt marsh hydrology: implications for biogeochemical fluxes to the atmosphere and estuaries. Global Biogeochem Cycles 2:91–114
Rice SD, Short JW, Karinen JF (1977) A review of comparative oil toxicity and comparative animal sensitivity. In: Wolfe DA (ed) Fate and effects of petroleum Hydrocarbons in marine organisms and ecosystems. Pergamon Press, New York, p 78–94
Rice SD, Thomas RE, Carls MG, Heintz RA, Wertheimer AC, Murphy ML, Short JW, Moles A (2001) Impacts to pink salmon following the 'Exxon Valdez' oil spill: persistence, toxicity, sensitivity, and controversy. Rev Fish Sci 9: 165–211
Short J, Heintz R (1997) Identification of 'Exxon Valdez' oil in sediments and tissues of PWS. Environ Sci Technol 31: 2375–2384
Short JW, Jackson TJ, Larsen ML, Wade TL (1996) Analytical methods used for the analysis of hydrocarbons in crude oil, tissues, sediments, and seawater collected for the natural resources damage assessment of the 'Exxon Valdez' oil spill. Am Fish Soc Symp 18:140–148
Smart PL, Laidlaw IMS (1977) An evaluation of some fluorescent dyes for water tracing. Water Resour Res 13:15–33
Todd DK (1964) Groundwater. In: Chow VT (ed) Handbook of applied hydrology. McGraw-Hill Book Company, New York, p 13-1–13-55
Wiedmer M, Fink MJ, Stegeman JJ, Smolowitz R, Marty GD, Hinton DE (1996) Cytochrome P-450 induction and histopathology in preemergent pink salmon from oiled spawning sites in Prince William Sound. Am Fish Soc Symp 18: 509–517
Wolfe DA, Hameedi MJ, Galt JA, Watabayashi G and 8 others (1994) The fate of the oil spilled from the 'Exxon Valdez'. Environ Sci Technol 28:561A–8A

Editorial responsibility: Otto Kinne (Editor), Oldendorf/Luhe, Germany

Submitted: April 25, 2002; Accepted: September 5, 2002
Proofs received from author(s): January 23, 2003