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Simulations of the Effects of Potential Oil Spill Scenarios on Juvenile and Adult Sockeye Salmon *(Oncorhynchus nerka)* Migrating through Bristol Bay, Alaska

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SIMULATIONS OF THE EFFECTS OF POTENTIAL OIL SPILL SCENARIOS ON JUVENILE AND ADULT SOCKEYE (*ONCORHYNCHUS NERKA*) MIGRATING THROUGH BRISTOL BAY, ALASKA

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ABSTRACT

Simulation models were designed to represent the movements of juvenile and adult sockeye salmon (Oncorhynchus nerka) through Bristol Bay. The first model describes the timing and distribution of the fish on a per river basis; the second the local movements of the salmon as they come into contact with an oil spill. Parameters in the models were chosen to maximize the potential effects of the described spills and the spills were assumed to occur at the peak migration times of the juveniles and of the adults. Avoidance of the spill commensurate with available theoretical studies was allowed the salmon, and simulations were also run with no avoidance to provide upper and lower predicted bounds of contamination and mortality.

A tanker accident releasing 240,000 bbl of No. 2 diesel fuel off of either Port Moller or Port Heiden was predicted to cause mortalities of 7 to 35% of the juveniles migrating through the area at the time of the spill, which would represent 3 to 13% of the total migrating juvenile population. At the same time 3 to 18% of the surviving juveniles in the spill area or 1 to 6% of the total population were predicted to be tainted at above 0.6 ppm. From 2 to 18% of adults passing through the tanker spill area were predicted to die, which represented 1 to 5% of the whole population. Mortalities could be twice as high as these mean values for adults returning to the Ugashik River as these would be closer to shore in the area of the spill.

Tainting levels could reach 3 to 7% of the population migrating through the spill area or 1 to 2% of the whole population. The percents tainted were again specific to the river of return and the degree of tainting on the fishing grounds off the various rivers is strongly dependent on the travel time for the adults

between the spill area and the fishing grounds. Thus under the assumption of no change in chemical composition of the oil over the duration of the spill, up to 30% of the adults returning to the Ugashik River fishing grounds on any one day could be tainted at above 0.6 ppm after passing through a spill off of Port Heiden, and up to 9% of those returning following a spill off of Port Moller could be tainted.

Simulated effects of a blowout of crude oil of 20,000 bbl/day were much lower with expected mortalities reaching a maximum of 1% for juveniles in the area of the Port Moller blowout and tainting less than 1% in all instances.

Results from these simulations are highly dependent on the physical location of the oil spill and its projected movements. Effects on adults are greatly increased for spills located further into Bristol Bay, and effects on juveniles are greatly increased for spills occurring closer to shore.

ACKNOWLEDGEMENTS

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INTRODUCTION

The most valuable concentrations of salmon (Oncorhynchus spp.) in Alaska pass through Bristol Bay on their way to and from the Bering Sea and Pacific Ocean. Sockeye salmon (O. nerka) are the most abundant of the Bristol Bay salmon, comprising 89% of the adult returns since 1951 (Rogers 1977). They are also the species for which most Bristol Bay data are available. The life histories of Bristol Bay sockeye salmon are variable, but can be generalized as follows from Royce et al. (1968)[1/].

In August and September the adults spawn in the extensive Bristol Bay river systems. The resulting fry emerge mainly in the following June and move to a lake where they spend 1 or 2 winters. They migrate out of the lakes following the breakup of lake ice, mainly in June, and into Bristol Bay. Their length at this point ranges on average from 7 to 12 cm, depending on age and river system. Most of the sockeye salmon spend 2 or 3 years (60-80% spend 2 years) at sea so that at maturity they are usually 4, 5, or 6 years old. In the first few months as they migrate through Bristol Bay they feed mainly on larval fish and euphausiids and remain within 54 km of the southern shore (Straty 1974). It is postulated that they remain in the Bering Sea until at least autumn, probably reaching the western Bering Sea before proceeding southwards into the North Pacific Ocean and the Subarctic current. By the next spring and summer (the beginning of their second ocean year) they are in their characteristic summer migration with the Alaska stream along the south side of the Aleutian Islands, their distribution extending up to 185 km offshore, but with the major abundances

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1/ Original references from Royce et al. will not be given here.

within 56 km. At this point the fish have grown to an average 35 cm, and are feeding actively on various crustaceans (primarily euphausiids) and larval fish. In the offshore regions, squid form a major part of their diet in addition to the larval fish and crustaceans (LeBrasseur 1966). The salmon repeat this elongated east-west course, which extends from about long. 165°E to 140°W, once or twice before returning to the estuaries of their natal rivers in Bristol Bay. The returns extend from about June 20 through July 25, with the peak in returns varying from July 2 to July 9 in different years. On return, the fish which have spent 2 years at sea average 51 to 53 cm, and are thoroughly mixed with those which have spent 3 years at sea and average 57 to 59 cm. During these migrations, their rate of migration has increased from an estimated 18 km/d in the first year to 31 km/d in the second year and again to 46 to 56 km/d in the final 30 to 60 days at sea.

The above generalized life history description conceals the variability within the Bristol Bay salmon; thus as late as May 1 in their final year they have been identified (by subsequent recaptures of marked fish) from 2200 km west of Bristol Bay (off the Kamchatka Peninsula), to 2200 km to the east (in the central Gulf of Alaska). Despite this widespread distribution the return timing is finely delimited.

Annual run sizes ranged from 2.4 to 53.1 million between 1951 and 1974, much of this variability being caused by the cyclical variability in returns to the Iliamna Lake in the Naknek/Kvichak district (Rogers 1977). The same author estimates the average number of sockeye smolts leaving the lake systems of the four major rivers from 1950 to 1974 at 209 million. The coefficient of variation of the number of smolts emigrating (1.29) was greater than that for number of returning adults (0.75).

In six of the years 1950-1974 the combined estimate of numbers of pink (O. gorbuscha) and chum (O. keta) salmon juveniles (average numbers 57 and 37 million, respectively) was greater than that estimated for the sockeye salmon smolts. Unfortunately, the analysis of the effects of the hypothetical oil spill on these species is precluded by the lack of consideration of the distribution of oil in nearshore waters.

CHARACTERISTICS OF STOCKS

Adult Migrations Through Bristol Bay

The timing and numbers of adult salmon returning to Bristol Bay have been summarized by Rogers (1977, 1984); he emphasizes that the catch statistics used to index abundance may not accurately measure total abundance because fishing effort has been variable. Catches until 1951 were comprised of 95% sockeye salmon. Since 1951 this percentage has decreased to about 89%. Their.estimated abundance has ranged from 2 to over 60 million fish since 1951, the high annual variability being partly a consequence of cyclic variability in runs to Iliamna Lake in the Naknek-Kvichak district. From 1956 to 1977 average abundance of the main age groups was 15.4 million fish. Since 1977 the average abundance has increased to 36.6 million fish. On average, 54% of the adult sockeye salmon returning to Bristol Bay return to the Naknek-Kvichak district although this district, on average, produces only 36% of the outmigrating sockeye salmon. The majority of the returning adults have spent 1 or 2 years in freshwater, and 2 or 3 years at sea. Since 1952, on average, 50% spent 1 year in freshwater (range 19 to 89%) and, on average, 57% spent 2 years at sea (range 28 to 77%). Mean lengths on return are dependent on the age of the fish; annual means have ranged from 48.5 - 52.5 cm for adults of age 1.2 to 56.5 - 59.5 for age 2.3. The average mean lengths tend to show a negative correlation with run size and have been at the higher end of the quoted range since 1977 (Rogers 1984). Rogers suggests that the increase in abundance and the higher average lengths in recent years are associated with higher winter temperatures while the fish were at sea.

Return times of adult salmon to Bristol Bay follow a very consistent pattern with 80% of the run passing the fishery over a 12.9 day period (SD 1.58 days) from 1956 to 1976 (Burgner 1980, Fig. 4). The mean time of return over the same period was July 4 (extremes June 28 - July 10; SD 2.92 days); fifty percent of this annual variability in timing can be explained by the correlation with Adak-Cold Bay air temperatures (Burgner 1980). Adults start arriving at outer Bristol Bay (Port Moller) around June 15 in an average year and it takes them about 6 days to migrate the 300 km to the inner fishing districts. Tagging of returning adults has indicated a sequential correspondence between the times of tagging in the Aleutian Islands area and the times of recovery in Bristol Bay, with a tendency for the later migrants to move at a more rapid rate (Hartt 1966, cited by Royce 1968).

Straty (1975) collated the available information on the distribution and movements of adult sockeye salmon from experimental fishing and from mark and recapture data. Adult sockeye were not caught inshore with exploratory fishing, at least until east of Cape Seniavin. These data corroborate conclusions of Hartt (1966, cited by Straty 1975) that adult sockeye bound for Bristol Bay from the Pacific Ocean traveled north into the Bering Sea before turning east into Bristol Bay. Exploratory fishing further suggested that adult sockeye cross the S.E. Bering Sea shelf in two bands, one north and one south of the Pribilof Islands. The Bristol Bay stocks, with the possible exception of returns to the Wood River System, were expected to be more abundant in the southern band.

Offshore distribution of adult salmon within the Port Moller/Port Heiden area has been studied using exploratory gillnet fishing (Straty 1975). I have reanalyzed the data taken on a transect from Port Moller to Cape Newenham (161°30'W) in 1965 and 1966 (12 sets), and on a transect from Cape Seniavin to Cape Newenham in 1939 (16 sets) (Table 1). The midpoint of the offshore distribution (largest catch) is marked and the proportion of the total catch in each series caught at each location computed (data from the 6/27 - 7/7 series in 1939 were omitted because of their apparent bimodality). The data set was divided into the northern and southern halves of the distribution and the proportions caught plotted against distance from shore (Fig. 1). Regressions for the two sets intersected at 63 km offshore, indicating the average location for the center of the distribution. Similarly the x intercepts of the two regressions at 13 and 178 km indicate the inshore and offshore limits of the distribution, respectively.

Adult sockeye salmon were tagged from 1957 to 1965 by U.S. and Japanese researchers in one area from the Pribilof Islands south to the Aleutian Islands and in another area in outer Bristol Bay between Cape Seniavin and Port Heiden. The rivers to which the tagged fish returned were tabulated and the results analyzed by Straty (1975) to determine if the different stocks migrated at distinct distances from the southern shore. To summarize his results I have subdivided the two areas into north (offshore) and south (inshore): 0 to 175 km and 225 to 350 km from shore for the area on the outer shelf, and 0 to 30 km and 75 to 125 km from shore for the area in outer Bristol Bay.

Table 1. Catches of adult sockeye salmon from exploratory gill net fishing as a function of distance from shore. Data as summarized by Straty (1975).

Location	Date	Distance from shore (km)	Number	Midpoint of distribution	Proportion from midpoint to one end South	North
161°30'W Port Moller to Cape Newenham	6/24-7/6, 1965	108	5			0.02
		83	46			0.18
		70	43			0.17
		65	82	*	0.32	0.32
		45	56		0.22	
		35	27		0.10	
	6/24-7/6, 1966	122	6			0.05
		97	11			0.09
		88	14			0.12
		75	52	*	0.43	0.43
		60	29		0.24	
		30	9		0.07	
Cape Seniavin to Cape Newenham	6/27-7/7, 1939	217	5			0.03
		162	2			0.01
		119	55			0.28
		80	20	?	0.10	0.10
		49	100		0.51	
		17	16		0.08	
	7/10-7/14, 1939	194	11			0.04
		163	12			0.04
		129	65			0.22
		99	122	*	0.41	0.41
		66	85		0.29	
	7/17-7/23, 1939	147	10			0.04
		115	17			0.07
		82	145	*	0.58	0.58
		49	80		0.32	
		15	0		0.0	

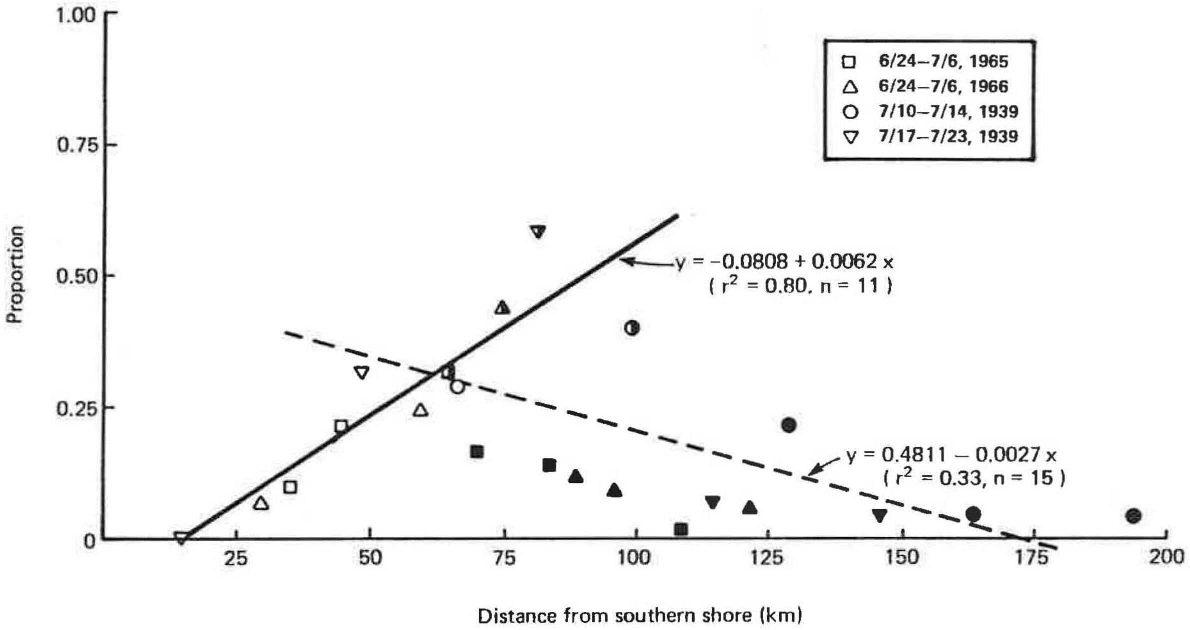

Figure 1.--Proportions of total sockeye salmon caught in exploratory gill net surveys in Bristol Bay, that were caught at varying distances from shore. Data from Straty (1975).

RIVER OF RETURN

	Nushagak	Nak-Kvi	Egegik	Ugashik
Area 1 - Pribilofs south to Aleutians				
South	1	55	22	7
North	3	23	17	7

Chi-square = 6.1055; p>0.10

	Nushagak	Nak-Kvi	Egegik	Ugashik
Area 2 - Cape Seniavin to Port Heiden				
South	4	3	5	5
North	4	5	1	0

Chi-square = 6.8096; 0.10>p>0.05

Neither data set showed significant difference in the contribution of the stocks from the different rivers with changing distance from shore, suggesting that the adults are mixed at these two tagging locations. The numbers of returns are small, however, and fish tagged in the northern area of outer Bristol Bay had very few returns to the Egegik and Ugashik rivers, suggesting that adults returning to these rivers may already have started to separate out from the remaining adults, and to move in a more southerly direction towards their natal streams.

Straty (1975) also reports the recapture of fish tagged within and adjacent to the four fishing areas in inner Bristol Bay from 1955-1957 and 1959. Adult sockeye were fairly well segregated at the fishing areas, the predominant tag returns coming from the nearest rivers. There was, however, a spreading out of the distribution around a river; thus returns to the Ugashik River came from fish tagged further out in the bay (as far as the Cinder River) and from the inner bay (to midway between the Egegik and Naknek rivers). Figure 22 from Straty (1975) is reproduced here (Fig. 2) and represents an overall summary of the hypothesized distributions of the adult sockeye salmon in Bristol Bay

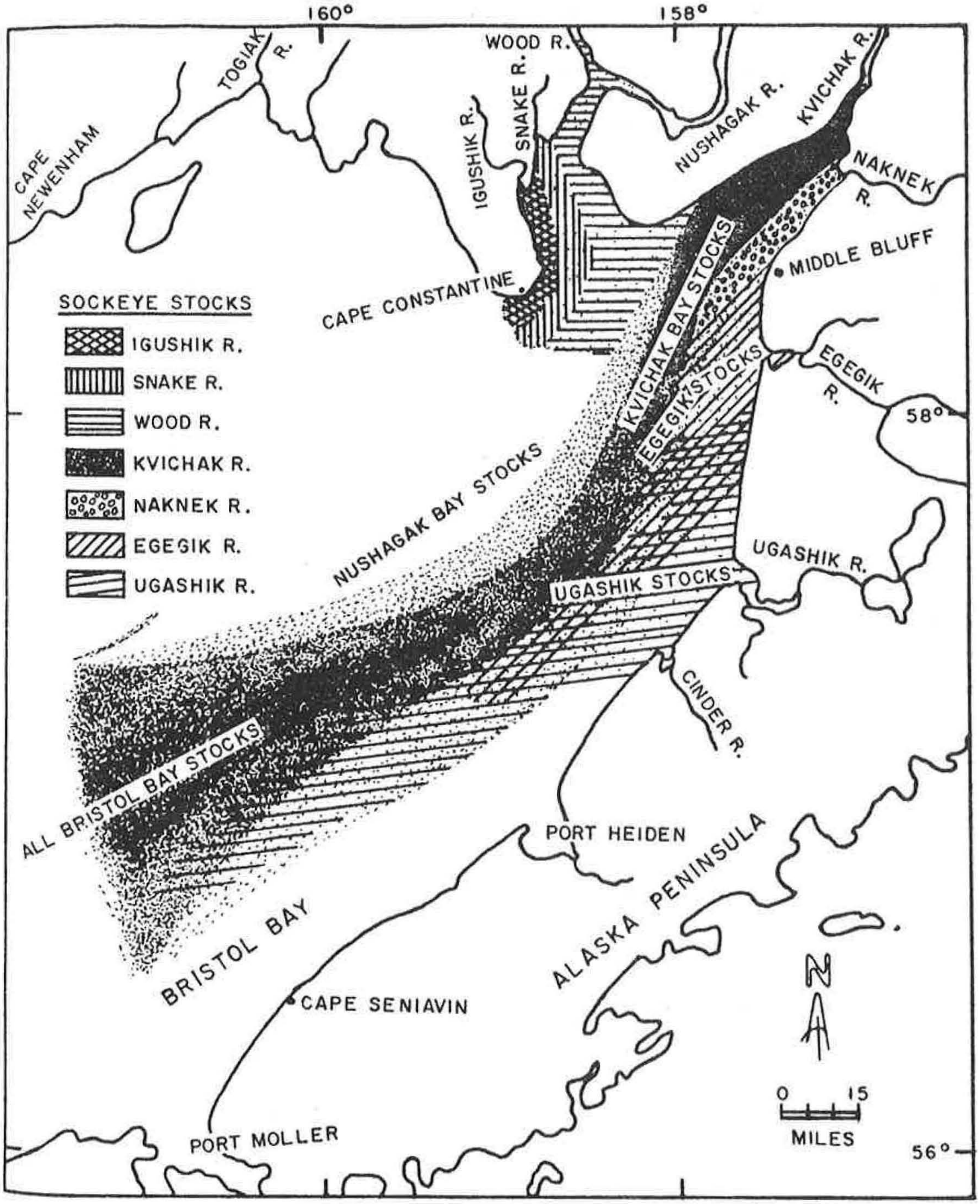

Figure 2.--Migration pathways of adult sockeye salmon returning to Bristol Bay. (Figure 22 from Straty (1975), reproduced with permission of author.)

as determined from tagged fish. Some interpolations were necessary in the middle of the inner Bay where no tags had been released. The lack of returns to the Kvichak and Naknek rivers from fish tagged close to shore near the Ugashik and Egegik rivers, suggests that the Kvichak and Naknek fish remained in the middle of inner Bristol Bay until close to their home rivers. This corroborates the earlier data for the Ugashik River which indicated that the adults did not move directly toward the shore until they were in the proximity of their natal river.

Direct interpretation of the results from adult tagging implies a migration rate of about 50 km/d through Bristol Bay. However, several studies indicate that migration is not in a straight line during this period. Adult chum and Atlantic salmon (Salmo salar) returning to their spawning streams have been observed (using ultrasonic telemetry) to undergo vertical migrations, the frequency of which increases as they close on the coast (Ichihara and Nakamura 1980, Westerberg 1983). The latter author suggests that these vertical migrations are used to traverse the fine structure of the coastal environment and identify the natal river water in the vertical stratification using their highly developed olfactory sense. The direction of movement of their natal river water relative to adjacent layers would provide information on the necessary direction of movement. Similar vertical migrations occur in the open ocean, and so other factors may also be involved.

Lateral variability in migration direction of Atlantic salmon was observed by Westerberg (1983) and Brawn (1982), the latter finding migration suspended in the estuary for up to 3 1/2 months. It is possible that these observations on returning adults captured in fresh or brackish water, and released further out in the estuary may have been experimental artifacts; however, Groot et al. (1975) who observed possible estuarine movement with the tides for sockeye salmon returning to the Fraser River, suggested this was not the case and cites similar experiments by Madison et al. (1972), where sockeye salmon released 93 km from the river mouth migrated actively towards the river. Smith et al. (1981) observed Atlantic salmon adults and grilse to maintain a relatively constant heading independent of both speed and direction of tidal flow. This migration superimposed on the changing tidal flow led to an overall elipsoid movement towards the coast.

These studies suggest that although migration of adult sockeye salmon through Bristol Bay may be fairly direct (at least in the horizontal plane), on arrival near their natal river estuary migration could become passive as the adults delay upstream migration. In this situation the adults would be particularly susceptible to contamination by oil in the area.

Timing of Smolt Outmigrations

The timing of outmigration of smolts from the rivers determines the time at which they enter Bristol Bay, and assuming a constant relationship between migration rate and fish size, the size of the fish at outmigration determines their initial migration rate. Together with the spread of the run over the time, these factors determine the proportion of an outmigration that will be exposed to a specified oil spill. Eggers and Rogers (1978; their Appendix 1) describe the collection of the smolt data and its condensation to a useful format for simulation for four rivers entering Bristol Bay:

"Data were condensed from hourly counts to an evening index count, a daily index count, and a random site count where available. The index count was generated by placing an "index net" (typically a fyke net with a cod end) in the region of the stream bed where the majority of smolts were believed to pass, and sampling over a 2-6 hour period of peak daily migration. The daily or 24-hour count was performed by counting migrating smolts passing through the index net over a 24-hour period. Random site counts required a set of five or six fyke nets placed uniformly across the stream bed. At the beginning of the period of peak migration, sometime between 2000 and 0200, a net was selected at random and smolts passing through were counted for an hour; then the process was repeated until the end of the migration period.

Also recorded were the average snout to fork tail length of the smolts in millimeters, the number of fish sampled for the length calibrations, and the percentage of fish in a second sample which had spent one year in the lake and river system as well as the sample size for age calibrations.

The Ugashik samples [total counts] were taken at a point 150 yeards below the outlet of the lower Ugashik Lake. The Naknek sampling [total counts] was performed 8-3/4 miles below the outlet of Nakek Lake. The Kvichak samples [index counts] were obtained 4 miles downstream from the Iliamne Lake outlet. The Wood River sampling [index counts] was done at Mosquito Point.

The data were obtained from four major sources: Alaska Department of Fish and Game leaflets; Alaska Department of Fish and Game technical reports; Fishery Bulletins; and records compiled by Dr. Donald E. Rogers, Fisheries Research Institute, University of Washington."

Continued sampling since the conclusion of Eggers and Rogers (1978) report has included sonar counts of absolute numbers on some rivers. An index of abundance was estimated for the Kvichak River from 1971 to 1976 with a fyke net at the same time that sonar counts were made; there was little correlation between the two measures ($r=0.39$; $p>0.20$, $n=6$). The sonar counts were added to the data base of Eggers and Rogers (1978) which was updated to include the more recently available data. Data were provided by Dr. Rogers from the same sources as detailed previously.

The timing of outmigration of the smolts varies between rivers, between age classes, and between years. Egegik and Ugashik smolts enter Bristol Bay first, followed by those from the Naknek-Kvichak, and then those from the Wood (Nushagak) River. Smolts migrate over a shorter time period from the Ugashik or Kvichak rivers than from the Naknek or Wood river systems because there are fewer sockeye rearing lakes in the former systems. Annual timing of smolt migrations is strongly influenced by spring weather conditions, e.g., time of ice breakup in the lakes and rivers. Summary statistics for the four major rivers are given in Table 2 .

Larger smolts are likely to migrate out of a lake earlier in the outmigration period, thus the smolts with 2 years of freshwater life are likely to migrate out earlier than those smolts with only one year. Thus there is a tendency for the size of the smolts to decrease over the period of outmigration. This decrease can be offset, however, by the extra growth in freshwater during this delay of migration. The following unweighted regressions illustrate the change in mean lengths of the outmigrants for the four rivers under consideration in 1982.

Table 2.--Summary of outmigrations of sockeye salmon smolts from four rivers entering Bristol Bay.

a) Abundance, Age and Size

River	Kvichak	Naknek	Ugashik	Wood
Gear	Sonar	Fyke Net	Fyke Net	Sonar
Years (n)	1971-1983 (13)	1957-1976 (20)	1958-1975 (15)	1975-1982 (8)
Total smolts x 10^6 (range)[1/]	130 (16-269)	9 (<1-25)	12 (<1-70)	65 (34-106)
Percentage I (S.D. annual means)	48 (38)	45 (21)	56 (28)	86 (10)
Length I (S.D. annual means)	88 (5)	101 (6)	91 (4)	82 (6)
Length II (S.D. annual means)	108 (8)	113 (4)	114 (5)	96 (3)

[1/] range given because numbers not normally distributed.

b) Timing

River	Kvichak	Naknek	Ugashik	Wood
Years (n)	1955-1976	1958-1975	1958-1974	1951-1966 1975-1976
10% migration (range)	5/30 (5/21-6/10)	6/4 (5/28-6/9)	5/24 (5/16-6/2)	6/12 (6/2-6/26)
50% migration (range)	6/2 (5/23-6/11)	6/12 (6/3-6/21)	5/31 (5/27-6/12)	6/24 (6/10-7/14)
90% migration (range)	6/9 (5/27-6/24)	6/29 (6/13-7/10)	6/10 (6/4-6/20)	7/7 (6/15-7/29)
Duration of middle 80%	10	22	17	24

River	Age	Number	Intercept A	Slope B	R
Kvichak	I	22	86.40	-0.11	-0.40
	II	24	108.36	-0.42	-0.64***
Ugashik	I	3	90.74	-1.13	-0.68
	II	3	119.91	-4.43	-0.77
Wood	I	12	75.58	0.14	0.62*
	II	12	91.10	0.17	0.49
Naknek	I	47	93.73	0.02	0.09
	II	38	112.95	-0.12	-0.24

* - $p<0.05$

*** - $p<0.001$

Juvenile Sockeye Salmon Outmigrations Through Bristol Bay

Sockeye salmon juveniles migrating through Bristol Bay come principally from the Kvichak, Naknek, Wood, Egegik, and Ugashik river systems. Early reports suggested that their distribution in Bristol Bay was discontinuous with all the smolts passing Cape Seniavin between 31 and 56 km from shore, and most 35 to 38 km from shore (Aspinwall and Tetsell 1966; cited by Straty 1974). Hartt (1980) concluded from his analysis of 188 purse seine sets made in Bristol Bay from 1956 to 1970 that substantial numbers of juvenile sockeye salmon were present throughout the summer (July to September) between 160°W and 164°W with the distribution extending up to about 78 km from the southern shore. The most extensive work on the juvenile sockeye migrations in this area was conducted by Straty and Jaenicke from 1966 to 1971. This work has been reported several times; the following information is taken from Straty (1974).

Smolts reach Bristol Bay several days after leaving the lake outlets, and are present across the width of the inner Bay[2/] in the early and late summer. They are most abundant on the southeast side of the inner and outer Bay with abundance declining with distance from this shore; no juveniles were found further than 56 km offshore in the outer Bay and most were within 40 km. Juveniles from the Wood River in the northern inner Bay move southwards across the inner Bay and then continue their migration along the southeastern shore with the smolts from the other rivers, remaining within 56 km of shore perhaps as far as Unimak. Specific data on the distribution of other north shore stocks are not available; however Straty (1974) concluded that their migration patterns would likely be similar to those originating from the Wood River.

In southeastern Alaska the band of juvenile sockeye, pink, and chum salmon along the coastal belt was very narrow (Hartt 1980); the mean catches were 434 within 7 to 17 km of shore, 267 between 20 and 30 km, and 1.3 between 42 and 50 km. Hartt speculated that this close coastal association was caused by the continental shelf which narrows to less than 37 km in the sampling area; the width of the band of juveniles in the northern area of the Gulf was wider, commensurate with the wide continental shelf (Royce et al., 1968). By comparison, the data for Bristol Bay as reported by Straty do not indicate a similar decline in abundance with distance from shore within the 40 km where they were most abundant.[3/] Neither Hartt nor Straty reports on the distribution of juveniles close inshore, although Straty did catch juvenile sockeye salmon within Port

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[2/] Straty (1974) defines inner Bristol Bay as that portion of the Bay east of a line drawn from Port Heiden to Hagemeister Island.

[3/] Preliminary data from 1984 surveys suggest that abundances of sockeye salmon within the first 40 km may decline with increasing distance from shore in Bristol Bay (Steve Parker, FRI, Univ. Washington, Seattle, WA 98195: pers. comm.)

Moller bay. Manzer (1956) sampled the movements of juvenile Pacific salmon along Chatham Sound and the approaches to the Queen Charlotte Straits, and concluded that the juvenile sockeye salmon were present "on the beaches" (i.e., caught with the beach seine) in June and July, but found that by mid-August to early September the only recaptures were taken over deep water, implying that offshore movement had occurred (but only into the area where Hartt and Straty conducted all their sampling). Straty suggests that offshore movement begins in Bristol Bay once the juveniles reach Port Moller, although he does not present data to substantiate this. It is certain that at some time in the autumn or winter the juvenile sockeye must leave their coastal areas since age 0.1 sockeye (presumably of Bristol Bay origin) have been captured in the central Bering Sea in February and March (Bakkala 1969, as cited by Straty 1974).

Juvenile sockeye salmon were schooled as they passed through Bristol Bay, with perhaps a more contagious distribution in the inner Bay. They were found most abundant in the top 3 m of the water column at daytime with perhaps a few as deep as 6 m. This agrees with the vertical distributions of juvenile salmon, including sockeye salmon, in the southern Straits of Georgia (Barraclough and Phillips 1978). Birtwell and Harpo (1980) indicate that juvenile chum salmon were surface orientated during their study and some of the juveniles would enter surface waters polluted with pulp mill effluent, even when comparatively cleaner deeper waters were available. These results suggest that surface orientation of the juvenile smolt may be obligatory at this stage of their life history and it is not at all certain that they would attempt to avoid an oil spill through vertical migration.

Growth of the juvenile sockeye salmon in Bristol Bay is not well defined and comparable data are not available for other systems because of the juveniles rapid seaward migration (e.g. Healey 1978). The juveniles average 35 cm at the beginning of their second summer (Royce et. al. 1968), implying a growth of 25 cm (0.7 mmd^{-1}) in the year since their entry into Bristol Bay at about 10 cm; however, seasonal fluctuations in growth rates (c.f. annual growth rings on salmon scales) suggest rates higher than the mean in the first summer and correspondingly lower rates in the following winter. Straty (1974) found the typical widely-spaced marine growth rings on juveniles to be absent until they entered outer Bristol Bay, implying little or no growth in the inner Bay. With the assumption of no growth in the inner Bay he concluded that juveniles from the Wood River may have increased in length by 50% over 4 weeks in the outer Bay, and those from the Kvichak River by 100% over 8 weeks. Assuming size at outmigration to be about 10 cm, this implies a daily growth rate of 1.8 mmd^{-1}. Given the seasonal variations in growth included in the mean annual growth rate of 0.7 mmd^{-1}, this estimate from Straty's (1974) paper does not appear unreasonable.

There is a separation between the time of entry of the major stocks entering Bristol Bay, with 50% of the smolts on average passing the lake outlets of the Ugashik, Kvichak, Naknek, and Wood rivers by May 31, June 2, June 12, and June 24, respectively (Rogers 1977). There is annual variability around these average timings. This order would be maintained throughout the juveniles passage through Bristol Bay if they migrated at similar speeds. Some information on the rate of migration for the different stocks can be obtained from comparing the time of release of marked fish with their time of recapture. Straty (1974, Table 14.5) provides data on the time and place of release and recapture for 26 juvenile sockeye salmon marked with fluorescent pigment, although time of release is not exact as it covered an extended period. These data are tabulated in Table 3 , and the migration rate of each fish estimated. The absolute migration rate in centimeters per sec increased with increasing size of the fish (Fig. 3) and the average relative migration rate in body lengths per second was 0.9 (S.E. 0.4). Relative migration rates within a stock ranged over a wider interval (e.g., Ugashik 1970: 0.2 to 1.6) than did the means between stock and year (0.7 to 1.4), thus there was no indication of interstock differences in relative migration rate. The relationship between the absolute migration rate and body length indicates that the rivers producing larger smolts would have juveniles with an overall higher absolute migration rate. Similarly, the larger age 2 smolts would migrate faster than the smaller age 1 smolts. These size differences will cause either intermingling or separation of the stocks migrating through Bristol Bay, depending on the actual stocks and age classes concerned.

A second factor affecting stock intermingling would be a change in spread of the stocks as they moved along. The age 2 smolts are larger and in general migrate out sooner than the age 1 smolts so an increase in spread of the migrants from each river is to be expected. A comparison of the timespan over which the smolts were marked with the timespan over which they were recaptured will detect any changes in distribution, although this will be an underestimate as it is unlikely that sampling would cover the entire distribution of the juveniles. The standard deviation of the distribution of marking dates was computed for each river and year for which there were recaptures, under the assumption that the reported spread in marking encompassed 95% of the juveniles (4SD). Sample standard deviations of the distribution of recaptures were computed and compared to those at marking (Table 4). On average, the standard deviation in the samples was increased by a factor of 4.8 over that at release. There was

Table 3 . Times and places of release and recapture for 26 juvenile sockeye salmon marked with fluorescent pigment, and estimates of their straight-line rate of migration (data from Straty 1974).

Year	Point of release	Point of[1] recapture	Distance travelled (km)	Date of release	Date of recapture	Elapsed[2] time (d)	Absolute migration rate (km/d)	Absolute migration rate (cm/sec)	Age of fish (yr)	Estimated fish size (cm)	Relative migrations rate (1/sec)	Mean migration rates	
												Stocks	Recapture locations
1967	Kvichak	P. Moller	440	5/26-6/11	7/4	31	14.2	16.4	2	11.8	1.4	–	
	Ugashik	P. Moller	290	5/27-30	7/9	42	6.9	8.0	1	9.2	0.9	0.7	
		P. Moller	290	5/27-30	7/19	52	5.6	6.5	2	11.5	0.6		
		Cold Bay	415	5/27-30	9/7	103	4.0	4.7	1	9.2	0.5		
1969	Naknek	P. Heiden	225	5/30-6/14	6/27	21	10.7	12.4	1	9.6	1.3		
		Egegik	95	5/30-6/14	7/6	30	3.2	3.7	1	9.6	0.4		0.8
		P. Heiden	225	5/30-6/14	7/13	37	6.1	7.0	1	9.6	0.7		
		P. Moller	360	5/30-6/14	7/17	41	8.8	10.2	1	9.6	1.1	0.8	
		P. Moller	360	5/30-6/14	7/27	51	7.1	8.2	1	9.6	0.9		0.8
		P. Moller	360	5/30-6/14	8/2	53	6.8	7.9	2	11.2	0.7		
		P. Moller	360	5/30-6/14	8/2	53	6.8	7.9	1	9.6	0.8		
		Cold Bay	400	5/30-6/14	8/26	77	5.2	6.0	1	9.6	0.6		
	Wood	Egegik	120	6/14-7/1	6/30	8	15.0	17.4	1	8.3	2.1		
		Egegik	120	6/14-7/1	7/6	14	8.6	9.9	1	8.3	1.2		1.4
		Egegik	120	6/14-7/1	7/9	17	7.1	8.2	1	8.3	1.0	1.2	
		P. Heiden	225	6/14-7/1	7/13	21	10.7	12.4	1	8.3	1.5		0.9
		P. Heiden	225	6/14-7/1	8/11	50	4.5	5.2	1	8.3	0.6		
		P. Heiden	225	6/14-7/1	8/15	54	4.2	4.8	1	8.3	0.6		
1970	Ugashik	P. Moller	290	5/17-6/5	6/13	17	17.0	19.7	2	12.5	1.6		
		P. Moller	290	5/17-6/5	6/15	19	15.3	17.7	2	12.5	1.4	0.9	
		P. Moller	290	5/17-6/5	6/23	27	10.7	12.4	2	12.5	1.0		
		P. Moller	290	5/17-6/5	9/13	109	2.7	3.1	2	12.5	0.2		
		P. Moller	290	5/17-6/5	9/13	109	2.7	3.1	1	9.7	0.3	–	
	Wood	P. Moller	360	6/6-6/22	8/11	58	6.2	7.2	1	8.3	0.9	0.8	
		P. Moller	360	6/6-6/22	8/16	63	5.7	6.6	1	8.3	0.8		
		P. Moller	360	6/6-6/22	8/31	78	4.6	5.3	1	8.3	0.6		

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[1] Approximate locations. See Straty (1974), Table 14.5 for more detail.
[2] From midpoint of marking period.

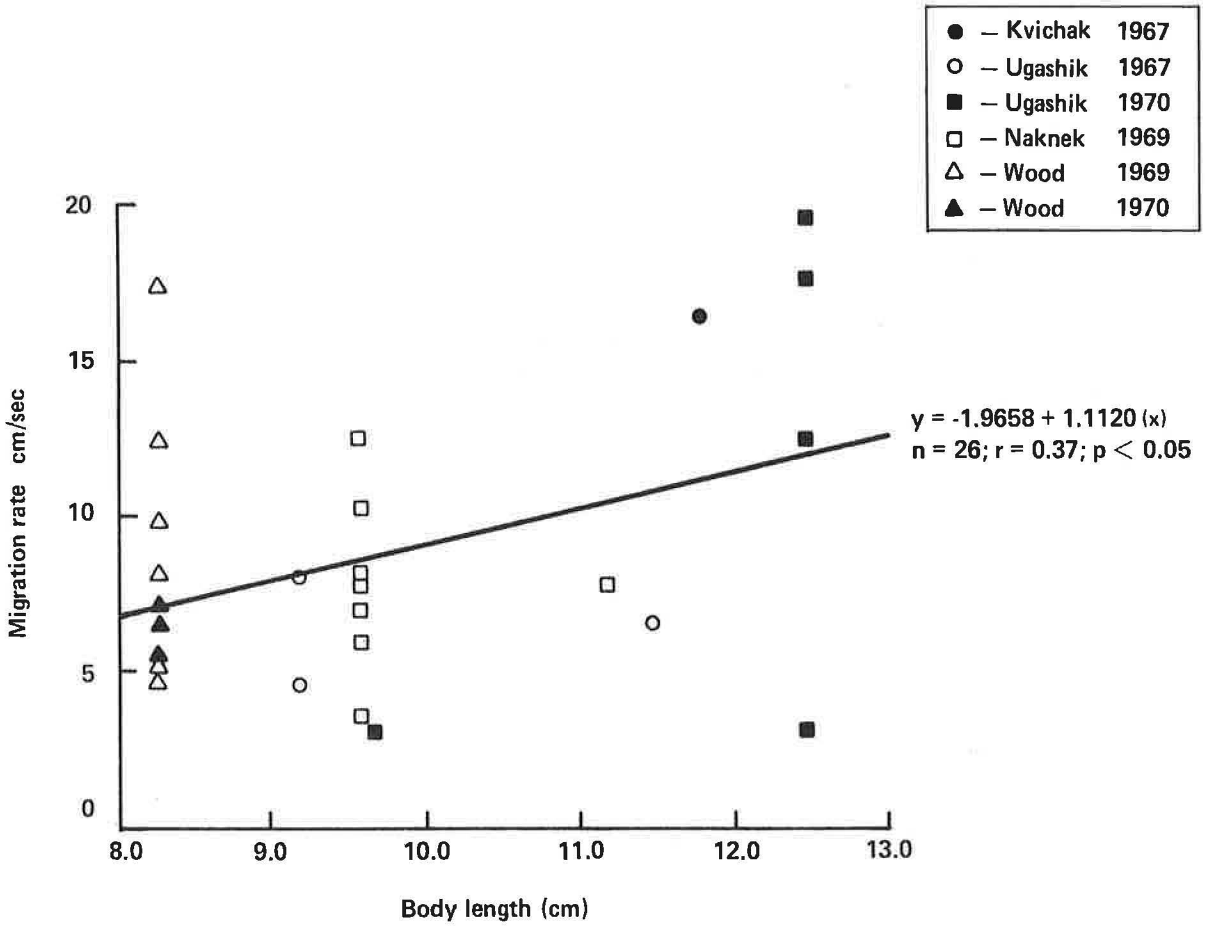

Figure 3.--Average migration rates of marked juvenile sockeye salmon from river to recapture in Bristol Bay. (Data from Straty 1974.)

Table 4. Spread in timing of groups of juvenile sockeye salmon at release and recapture in Bristol Bay (data from Straty 1974).

Year	River of release	Recapture location	Number of juveniles	Timespan of marking (d)	S_m[1]	Elapsed times to recaptured (d)	Mean elapsed time (d)	S_R	S_R/S_m
1967	Ugashik	P. Moller	2	3	0.75	42, 52	47	7.1	9.5
1969	Naknek	P. Heiden	2	16	4.0	21, 37	29	11.3	2.8
		P. Moller	4	16	4.0	41, 51, 53, 53	49	5.7	1.4
	Wood	Egegik	3	17	4.25	8, 14, 17	13	4.6	1.1
		P. Heiden	3	17	4.25	21, 50, 54	42	18.0	4.2
1970	Ugashik	P. Moller	5	19	4.75	17, 19, 27, 109, 109	56	48.3	10.2
	Wood	P. Moller	3	16	4.0	58, 63, 78	66	10.4	2.6

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[1] Timespan of subsequently recaptured fish assured to cover 95% (4.S.D.) of release distribution.

considerable variability in this increase factor, the maximum two values resulting from Ugashik River fish recaptured at Port Moller in 1967 and 1970 (9.5 and 10.2, respectively). Increase factors for the Wood River and Naknek stocks ranged from 1.1 to 4.2 and did not appear to increase with the distance traveled (Fig. 4), suggesting that the spreading of their distributions occurred soon after marking. The implied higher spreading of the Ugashik River stocks could indicate stock-specific differences in migration patterns; however, effects of the sampling design cannot be eliminated.

Sockeye juveniles were found by purse seining to be most abundant in inner Bristol Bay from late May through early August (2 - 2 1/2 mos); after early August the major stocks will be further seaward (Straty 1974). These conclusions agree with the estimates of distribution derived from marked fish. Thus, applying a migration rate of 0.9 L/sec to the fish emigrating from the lake outlets (data from Rogers 1977), suggests that 50% of the smolts from the Ugashik, Naknek, Kvichak, and Wood rivers would be past Port Heiden (i.e., out of the inner Bay) by July 1, 19, 27, and August 13, respectively. Straty further suggests that the migration rate is faster through the inner Bay where the stomach fullness of the captured smolts is less, and where no marine growth was detected. However, there is no correlation between the rate of migration and the distance traveled by the recaptured juveniles (r=0.06, n=26, p>0.50). The purse seine catches in 1969 (Straty 1974, Table 14.2) appeared higher at Port Moller in both June-July and August, but no sample statistics are given.

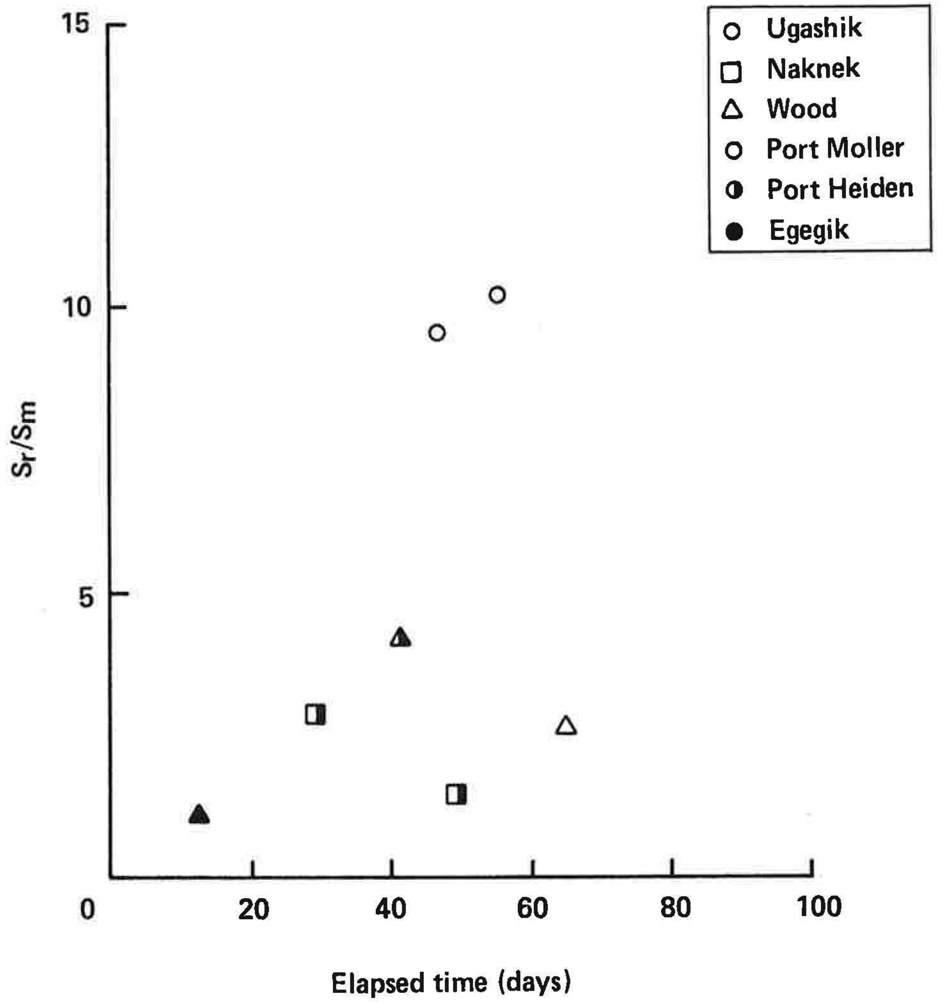


Figure 4.--Ratio of standard deviation of recaptures to standard deviation of releases for marked juvenile sockeye salmon released into rivers leading to Bristol Bay. (Data from Straty 1974. Symbols denote river of release; shading point of recapture.)

The implied higher catches could indicate a longer residence around Port Moller causing the increased availability of the juveniles to the sampling gear, but the data are not different from those expected if the juveniles migrated continuously at 0.9 L/sec. Given the available data, there is insufficient evidence to conclude that the mean migration rate is slower for the juvenile sockeye salmon in the inner Bay. It seems likely, however, that the spreading out of the fish between marking in the rivers and recapture in the Bay occurs soon after saltwater entry, implying that some of the juveniles may reside temporarily near the river estuary while others migrate away directly. Similar variability in migration patterns upon saltwater entry have been observed for juvenile chum salmon (Bax 1983, 1984).

On top of these seasonal variations in abundance of smolts in Bristol Bay there may also be annual variations. Straty suggests that in a cold year migration rate could be decreased (c.f. Mohr 1969, Brett and Glass 1973). Thus, in 1971 when the long term surface temperature anomaly in the southeastern Bering Sea was -2.1°C (compared to 0.8 in 1969, Ingraham, NWAFC, personal communication), Straty reports the absence of juvenile sockeye salmon off Port Moller in late June and July, the area where in 1969 the juveniles were indicated to be most abundant at this time. This delay would have been at least partly caused by the delayed outmigration from the lakes (the times of 50% outmigration were 9 and 4 days later for the Kvichak and Naknek rivers in 1971 than in 1969; Rogers 1977), although it seems unlikely that these delays could account for the virtual absence of the juveniles off Port Moller in early July. There is a significant positive relationship between the 60 min sustained swimming speed (V) of sockeye salmon smolts and the ambient temperature (T): $V = 3.05 + 0.17T$; $n = 23$, $r = 0.70$, $p < 0.001$ (data from Brett and Glass 1973 for smolts between 1 and 55 grams). Long term mean monthly surface temperatures over the Bering Sea shelf north of the Aleutians range from approximately 5.5 in June to 10.0 in August with a mean for the four months of approximately 8.5 (Ingraham, personal communication). Thus, if mean temperatures during the migration were 9.3 in 1969 and 6.4 in 1971, the maximum 60 min sustained swimming speeds would have been 4.6 and 4.1, or a difference of approximately 10%. It is possible then that the migration rates of the same size fish could have been 10% slower in 1971 than 1969 due to the colder water temperatures in Bristol Bay.

On top of these two factors we need to consider the effects of decreased smolt size at outmigration in cold years: the smolts outmigrating from the Kvichak were, on average, 10.1 cm in 1969, but only 9.1 cm in 1971, a decrease of 10%. Together with the temperature decrease in relative swimming speed this could lead to an absolute decrease in swimming speed of about 20%, at least, until the size at outmigration is no longer the principal determinant of the juveniles length. This reduced migration speed would apply to juveniles already delayed by up to at least 9 days in migration. Together these three factors could have produced the delays suggested in 1971. The longer residence in Bristol Bay in cold years would increase the changes of exposure to an oil spill, and this would occur at the same time that the juveniles' ability to cope with the effects. Elimination of oil would also be reduced due to the lower temperatures (Korn et al. 1979, Moles et al, 1979, Thomas and Rice 1979).

Relationship Between Smolt and Adult Abundances

Of crucial concern in the estimation of juvenile sockeye salmon oil-induced mortalities is to what extent these mortalities will be evidenced in the resultant adult population. If, for example, the abundance of the adult population is determined at some later time (perhaps only a fixed number can survive the first winter at sea), then oil-induced mortalities might be considered to have little effect on subsequent adult returns unless the number of smolt was decreased below some lower limit. An alternative assumption would be one of no density dependent mortality: that is, for every x% decrease in the smolt population there would be a corresponding x% decrease in the adult population. Rogers (1980) concluded that density dependent factors were important for Bristol Bay sockeye salmon during their last summer at sea, and reported a negative correlation between the abundance at this time and the weight of the returning adults. He postulated that this density-dependent growth occurred when the maturing sockeye are concentrated north of the Aleutian Islands on their migration back to Bristol Bay. Rogers (1984) also reports a correlation between the abundance of Bristol Bay adult return and the temperature while the fish were at sea, especially the winter months. He postulates that this results from the adults distribution in the Gulf of Alaska compressing in cold years, making them more vulnerable to predation by marine mammals.

Rogers (1977, and unpublished data) and technical data reports from the Alaska Department of Fish and Game provide data on estimated smolt numbers leaving four principal rivers in Bristol Bay - Kvichak, Naknek, Wood, and Ugashik - and the resulting run sizes. Until the 1970's the outmigrants were sampled with fyke nets although not all rivers were sampled in all years. Beginning in the 1970's the censusing changed over to sonar counts which were considered to be more accurate. Sonar counts began in 1971 for the Kvichak, 1973 for the Wood, and 1982 for the Egegik and Naknek. The number of smolts migrating down the Kvichak were estimated with both fyke net and sonar from 1971 to 1976; there was no significant correlation between the estimates from the two methods (r=0.39, p>0.20, n=6); however, sample size was small. Obviously the lack of any correlation between smolt abundance and subsequent adult returns may be as much a function of sampling variability as the lack of a real correlation.

With the above proviso I have regressed the estimated number of adults resulting from a year's outmigration on the estimated number of smolts outmigrating. The regressions were performed on each river for smolts which had spent one year in freshwater, smolts which had spent two years in freshwater, and the combined estimates. The data are presented for fyke net counts and for sonar counts in Table 5 .

In all instances where there was a significant relationsip between smolts and adults the relationship was stronger for the smolts which had spent 2 years in freshwater, and the indicated slope appeared higher, suggesting that 2-yr-old smolts are more likely to return as adults than 1-yr-old smolts, and that variability in survival is likely to be less for the 2-yr-old smolts. The two data sets where total smolt estimates were significantly correlated with adult returns were the Kivichak River sonar counts (1971-1981) and the Ugashik River fyke net counts (1958-1974). Estimated slopes ranged from 0.06 to 0.4, indicating that, on average, between 9 and 17% of the smolts survived to return as adults.

This analysis assumed a linear relation between smolts and adults (no assumption was made that the regression line would pass through the origin, however), and potentially other forms of curve may be more appropriate. The

Table 5. Relation between abundance (or index of abundance) of outmigrating smolts and resulting abundance of returning adults (data multiplied by 10^{-4} before analyses).

River	Years	Gear used to sample smolts	Freshwater age	Sample size	a	b	r[1]
Kvichak	1955-1976	Fyke net[2]	1	22	-60.1	0.05	0.814***
			2	22	-46.2	0.07	0.849***
			All	22	-126.9	0.06	0.805***
	1971-1981	Sonar	1	11	73.2	0.06	0.662*
			2	11	-28.6	0.11	0.962***
			All	11	-306.0	0.11	0.844**
Wood	1955-1970	Fyke net[3]	1	15	110.6	<0.01	0.141
			2	15	10.6	0.02	0.509*
			All	15	132.1	<0.01	0.105
	1975-1981	Sonar	1	7	112.5	0.04	0.718
			2	7	29.3	0.02	0.452
			All	7	124.2	0.04	0.618
Naknek	1957-1977	Fyke net	1	21	78.9	0.02	0.105
			2	21	120.0	<0.01	0.00
			All	21	235.4	-0.03	0.173
Ugashik	1958-1974	Fyke net	1	15	11.9	0.06	0.590*
			2	15	3.5	0.09	0.941***
			All	15	24.6	0.06	0.666**

[1] * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$

[2] 24 hr index count.

[3] 2 hr index count in 100's.

data are plotted in Fig. 5. There is not sufficient evidence to justify fitting a more sophisticated model, perhaps with the exception of the Wood River stock where percentage returns were indicated to have decreased with increasing smolt numbers.

Data for the Naknek River indicated little relationship between smolt numbers and adult numbers. If the Naknek River smolt-adult survival rates were swamped by the effects of the Kvichak River sockeye, then a multiple regression analysis would yield an improved model. Neither the inclusion of Kvichak smolt numbers, nor Kvichak adult numbers as independent variables resulted in a significant model (Table 6). No significant improvement in fit was noted for the Wood River stocks using a similar model.

The clear correlation between smolts and returning adults from the Kvichak and Ugashik rivers indicate that if an oil spill effected mortality on the smolts soon after censusing we could expect a mean loss of 1 returning adult for every 6 smolts lost from the Ugashik (95% C.I. 2-10), and a mean loss of 1 returning adult for every 11 smolts lost from the Kvichak (95% C.I, 6-16). These ratios would decrease throughout the life span of the sockeye salmon as the probability of surviving to return as an adult increased. The form of this change in ratio is significant - if it is linear then the result of oil-induced mortality on juveniles in Bristol Bay would be similar to mortalities on the smolts. If, on the other hand, the form of mortality declines exponentially as suggested by the high coastal mortalities of pink salmon (Parker 1968) and the high estuarine mortalities of chum salmon (Bax 1983), then the loss of a specified number of juveniles in Bristol Bay could have a considerably greater impact than the loss of the same number of smolts. There is reason to believe that early marine mortalities are high for Bristol Bay sockeye salmon; predation by Belukka whales in Kvichak Bay was estimated at 6 million smolts or 5% of the total run. The indications are that any oil-induced mortalities on the juveniles would operate on the survivors of an already thinned population.

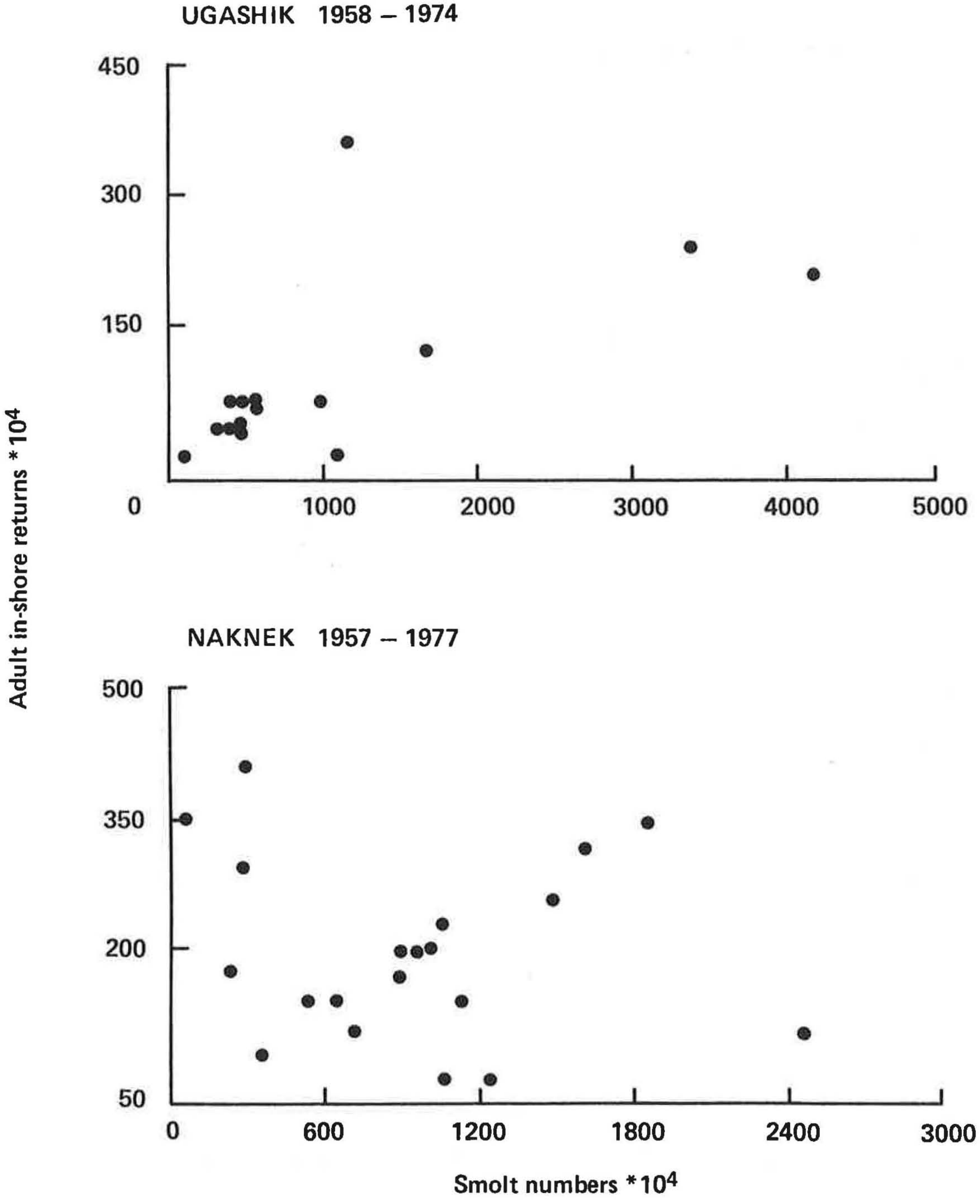
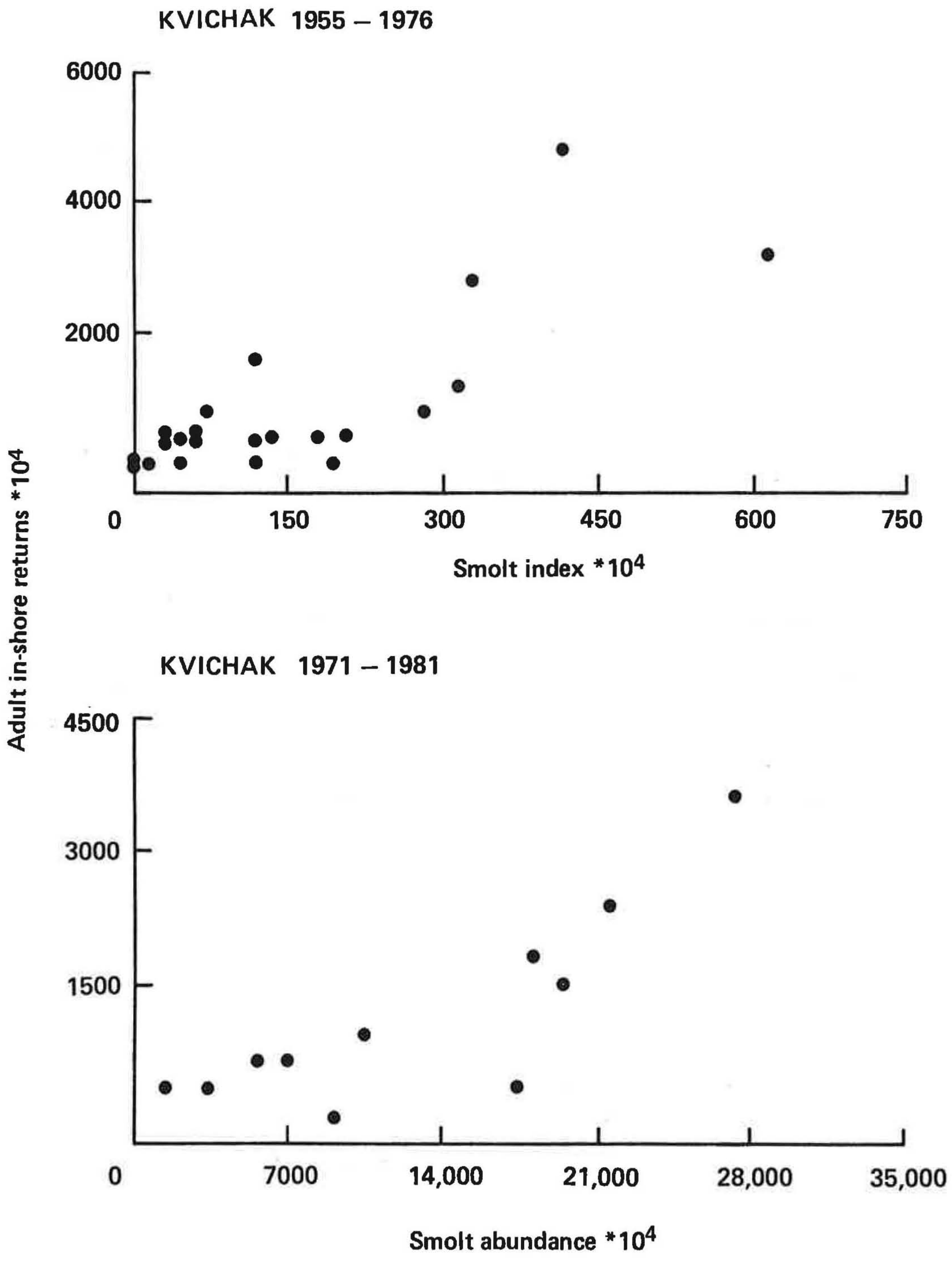


Figure 5.--Returns of adult Bristol Bay sockeye salmon and their estimated numbers (or index) as smolts outmigrating from the rivers. (Data sources detailed in text.)

Figure 5 continued



KVICHAK 1955 – 1976

KVICHAK 1971 – 1981

Figure 5 continued

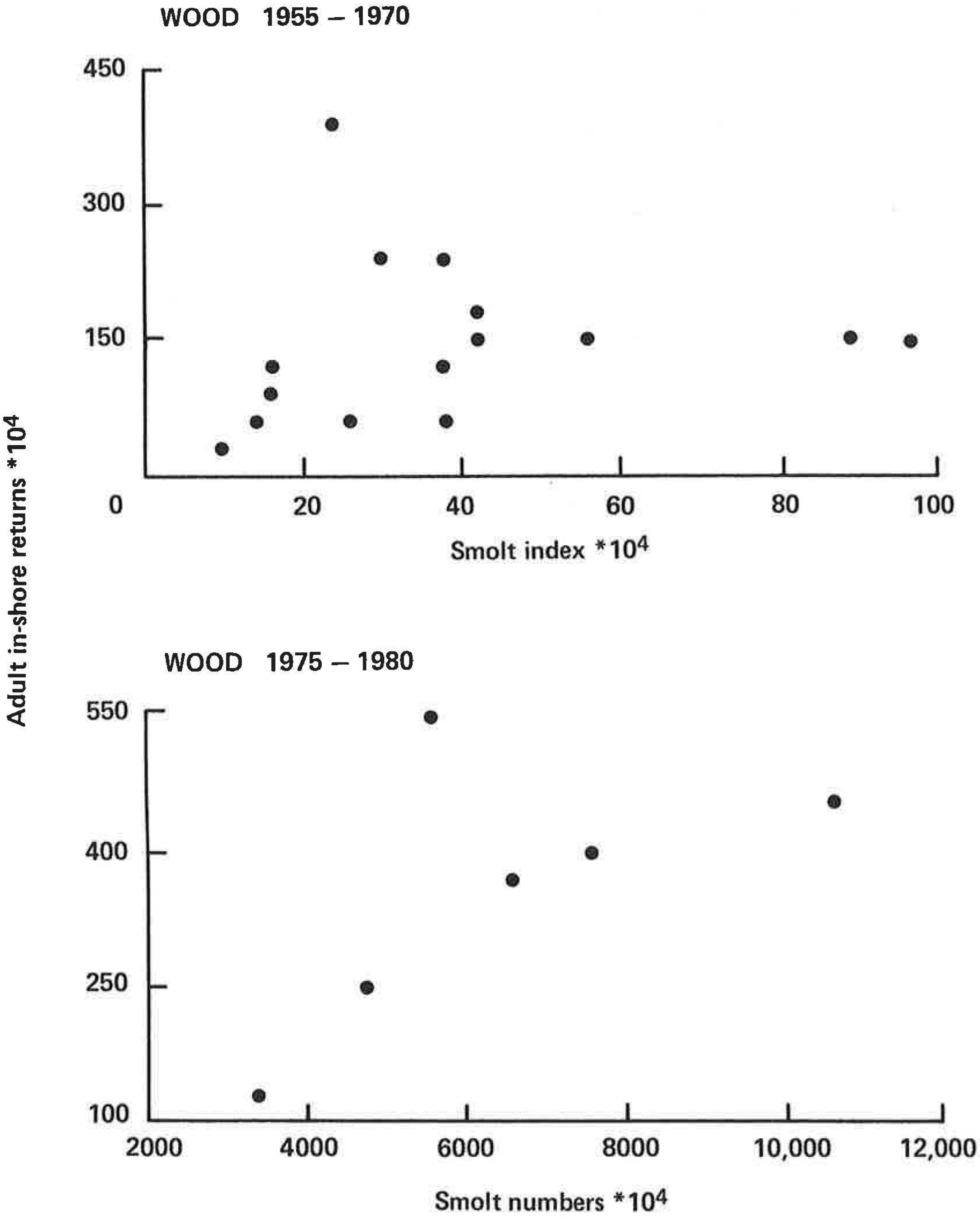


Table 6 . Relation between total abundance (or index of abundance) of smolts migrating for the Wood and Naknek river systems and the resulting total abundance of adults, taking the abundance of Kvichak River smolts or adults into consideration.

River	Years	Sample size	Independent variable	a	b	r
Naknek	1957-1976	20	Kvichak smolts	211.0	0.201	.276
			Naknek smolts		-0.049	.276
		20	Kvichak adults	217.0	<-0.01	
			Naknek smolts		-0.02	.114
Wood	1955-1970	15	Kvichak smolts	13.9	-0.09	
			Wood smolts		<0.01	.176
		15	Kvichak adults	129.1	0.01	
			Wood smolts		<0.01	.141

REACTIONS TO, AND EFFECTS OF, OIL

Avoidance

Sublethal effects in general, and avoidance in particular, are difficult factors to measure satisfactorily. In all except isolated field circumstances where fish have chosen between a polluted and a nonpolluted area and are subsequently accurately censused, e.g., adult salmon returning to their natal rivers, it is not practical to attempt to measure avoidance in the field. The alternative method, laboratory measures of avoidance, suffers from an unknown, but probable, lack of applicability to field situations.

It is fortunate that the salmon, with their well-defined life history and migratory patterns are more amenable to measurements of avoidance than other fish species, and both laboratory and field studies of avoidance have been conducted. The long history of raising juvenile salmon in hatcheries makes them a logical choice for laboratory studies, while the precise homing of the adult salmon to restricted areas which can be routinely censused, i.e., their natal streams and rivers, makes them a logical choice for field studies.

The 50% avoidance level of smolt and presmolt coho salmon in the laboratory to an approximation of the water soluble fraction of Prudhoe Bay crude was reported as 1.9 to 3.7 mg/l, respectively, in freshwater by Maynard and Weber (1981). Avoidance to the mixture was at a lower concentration than avoidance to any one of the individual components, suggesting an interaction between the different components in the WSF. Rice (1973) has shown the dependence of the avoidance response of pink salmon fry on time of year, and/or salinity, and/or temperature (the experimental design did not distinguish between these possible effects). Thus, hatchery-reared pink salmon fry exposed to Prudhoe Bay crude showed avoidance at 497 ppm in freshwater at 5°C on the 5th of June (no avoidance at 96 hr TLM), avoidance at 16.0 ppm in seawater at 7.5°C on the same date (1/13 of 96 hr TLM), and avoidance at 1.6 ppm in seawater at 11.5°C on August 27th (1/69 of 96 hr TLM). These oil concentrations reported by Rice are the total concentration of oil added to water, which was then shaken and the dissolved fraction removed and added to the test tank so the concentrations to which the fry were exposed were considerably less than reported. Maynard and Weber (1981) ruled out temperature as the influencing factor, and concluded that smolting coho salmon were twice as sensitive as presmolts. Sprague and Drury (1969) reported that the avoidance of rainbow trout and Atlantic salmon to pollutants (detergent, chlorine, and phenol) was specific to the pollutant; thus phenol was not avoided at sublethal levels, and the fish were actually attracted to the 4d sublethal level of chlorine. Similarly, Westlake et al., (1983) found no significant preference or avoidance to treated refinery effluent present at concentrations which would have led to sublethal physiological effects. There were indications of reduced activity at the higher concentrations of effluent. Morrow (1974) also reported attraction of sockeye and coho salmon fry to the oil inlet during mortality studies at 4.8 and 12.1 ppm of Prudhoe Bay crude (same reporting of concentrations as used by Rice, above), but this may have been an artifact of the tank design. Similar attraction was reported for chinook salmon by Bean et al. (1974; cited by Maynard and Weber 1981). It is possible that changes in acitvity level, mediated through the effects of hydrocarbons on the central nervous system could lead to indirect avoidance. Plaice larvae exposed to horizontal gradients of oil dispersants in a fluvarium, tended to the clean water channel at all times, but as a result of an orthokinetic response (increased swimming activity) not chemotaxis (Wilson 1974). Herring larvae in the same study showed no avoidance of vertical gradients but remained in the dispersant layer until narcotized, then sank into clear water where they recovered before swimming up again into the dispersant layer.

There are no studies of the avoidance of juvenile salmon to petroleum products during active migration; however, there have been several studies on the avoidance of other pollutants. The high variability associated with sampling juvenile salmon, engendered by their highly clumped distribution, can frequently result in no rejection of the null hypothesis of no avoidance of an area; thus apparent lack of avoidance to pollutants in the field is often more a reflection of the difficulties of sampling than a real distributional event. Poulin and Oguss (1981) concluded that there was lateral avoidance of pulp mill effluent in Neroutsus Inlet, British Columbia, but Birtwell and Harbo (1980) who studied the effects of pulp mill effluent on the migration of juvenile Pacific salmon at Port Alberni and Port Mellon in British Columbia, found no evidence of continuous avoidance of the surface waters in the proximity of the effluent outflow, even though it was at acutely toxic levels. Using live boxes the latter authors studied the volitional vertical distribution of juvenile chum salmon in polluted areas, where fish staying above 3 m would die and those staying below 3 m would survive. They found that all fish were observed below 3 m, but mortality was 25%, indicating that some of the juveniles had at some stage entered the upper, toxic 3 m. Birtwell and Harbo concluded that "...perhaps the inherent requirements of the juvenile salmon to enter surface waters overcame the avoidance response to acutely toxic effluent concentrations, resulting in mortality." Salo et al., (1979) recorded avoidance of juvenile chum salmon to glacial tills from dredging at 35 to 190 mg/l whereas the 96 hr LC50 ranged from 15.8 to 32.9 mg/l. The same authors suggested, although the data are not conclusive, that avoidance of the same materials in the field occurred at 2.0 to 9.8 mg/l. In a later report (Salo et al., 1980) avoidance of nearshore habitat disturbed by dredging or offshore pile driving by migrating juvenile chum salmon was observed. In both cases the measured response was not an all or nothing response, rather it was dependent on the individual fish and the habitat in which they were studied. The importance of individual variation in avoidance response to pollutants was also remarked on by Bohle (1982), who observed juvenile cod (Gadus morhua) in a trilobed aquarium to avoid petroleum hydrocarbons at 50 to 100 mg/ml, but with some exceptions.

In comparison to the studies on juveniles which include controlled laboratory experiments, the few studies of avoidance of pollutants by adult salmon have been conducted in the field. Westerberg (1983) observed retrograde movement by ultrasonically tagged adult Atlantic salmon released in a polluted branch of the Lule estuary (dredging and effluents from a steelworks and a coke-oven plant), whereas tagged adults released in an unpolluted branch of the same estuary showed a slow, but upstream, progress. Similarly, Elson et al., (1972) suggested avoidance by Atlantic salmon of a branch of the upper Miramichi estuary polluted primarily with pulp mill effluent, although the data were not definitive. Perhaps the best study to date on the effects of pollutants on avoidance of salmonids is that of Weber et al., (1981), who measured the avoidance of returning adult Pacific salmon (99% coho salmon) to parallel fish ladders on their natal stream. Avoidance to the ladder polluted with a close approximation to the WSF of Prudhoe Bay crude reached 50% when the concentration reached 3.2 mg/l at the top and middle of the ladder. The regression of percent of adults ascending the polluted ladder on the concentration of hydrocarbons was $Y=50.11-8.04X$ (correlation coefficient = 0.92).

Avoidance of pollutants by adult salmon is obviously a real and measurable quantity. In the above studies, the avoidance has been measured in restricted waterways compared to the coastal or open oceans and it could be argued that the avoidance is peculiar to these environments; however, it needs be remembered that this avoidance was measured for fish which were at the end of a long migration and would be expected to have the maximum migratory momentum. Taking both of these factors into account, it would appear reasonable to conclude that avoidance in the coastal waters would occur at or before the concentrations recorded by Weber et al., (1981). This contrasts with the National Academy of Science's (1983) report which concluded that there was little evidence to support the claim that adult salmon can avoid spills; however, they were unable to consider the more recent work. This conclusion modifies that of Thorsteinson and Thorsteinson (1982), studying a hypothetical spill, who concluded that "...adult salmon...migrate through surface layers but would avoid lethal concentrations or move rapidly through sublethal concentrations...", in that some avoidance is anticipated to occur at sublethal concentrations of oil.

The avoidance of pollutants by juvenile salmon in coastal waters is less easy to conjecture. Avoidance in the vertical direction was well demonstrated by Birtwell and Harbo (1980), avoidance in the horizontal direction was suggested by the studies of Salo et al., (1979, 1980), although avoidance was not complete in either instance; the lack of complete vertical avoidance of toxic pulp mill effluent led to mortalities of 25% of the test fish. Maynard and Weber (1981) found 50% avoidance of smolt and presmolt coho salmon in the laboratory at concentrations of dissolved hydrocarbons within the range where 50% avoidance was found for the adult coho. In both cases, however, it is not clear that there is a strictly linear relationship between avoidance and hydrocarbon concentration. No measurements of avoidance of 75% or greater were made for the adult coho and it is not certain that the linear relationship proposed by Weber et al., would apply at the higher levels of pollutants required to produce these levels of avoidance. For both the juveniles and adults it will be assumed that maximum avoidance is 90%, i.e., 10% of the population will not attempt to avoid polluted waters.

Rates of Uptake and Depuration

The concentration and subsequent elimination, or depuration, of hydrocarbons from the salmon is of interest for two reasons: tainting from hydrocarbons within the muscle, and lethal or non-lethal stress resulting from uptake into other organs, for example, the liver and gall bladder. Accumulation rate is not the same for the muscles and organs; maximum levels of napthalenes were found in the liver of adult Atlantic salmon exposed to approximately 40-50 ppb crude oil in a flow through vessel after 7 hr (at 42 ppm or 1000 times external concentration), while the maximum levels in the muscle were after 6-8 d (0.5 ppm or 10 times external concentration; Brandl et al., 1976). Organoleptic analyses also indicated that maximum muscle tainting occurred at 6-8 d; no mortalities occurred. Depuration began at an earlier stage in the liver, reducing napthalene levels to 2-3 ppm after 6-8 d, the time of maximum levels in the muscle, which in turn depurated to 0.02 ppm after 39 d. I emphasize that during depuration the salmon were still exposed to the flow-through water polluted at 40-50 ppb, although the concentrations of the more volatile hydrocarbons had decreased. Roubal et al. (1978) exposed juvenile coho salmon to 0.9-1.0 ppm of the WSF of Prudhoe Bay crude in flow-through containers over 6 weeks. Testing the fish after 2, 3, 5, and 6 weeks indicated maximum levels in the muscle at 5 weeks (10.35 +/- 1.83 ppm, overall bioconcentration 10; bioconcentration of napthalenes 117). No hydrocarbons were detected in the muscle after 1 week depuration in clean water. The difference in times of maximum bioconcentration between the two studies may be due to the loss of the more volatile aromatics in Brandl et al., (1976) experiment; however, it is not clear that similar, though smaller, losses did not occur in Roubal et al., (1978) study. An extreme example of this effect of experimental design is provided by Rice et al., (1977), who concluded the maximum bioconcentration in the gills, gut, and muscle of pink salmon fry exposed to No. 2 fuel oil to occur after 10 hr, decreasing by 24 hr. However, after 96 hr the concentration of oil in the exposure water had decreased to only 20% of its original level, negating these results. Experimental determination of depuration rates is further complicated by compound specific variability in depuration; some napthalenes and higher molecular weight aromatics can remain after other compounds have declined to undetectable levels (Lee 1977, Connell and Miller 1981).

None of the above studies addressed the uptake pathways for the oil into the flesh; however, NAS (1975) concluded that for animals with respiratory surfaces in contact with seawater, the partitioning between animal and water may be the most important avenue for uptake and loss of hydrocarbons; this avenue supercedes any food web magnification. Morrow (1974) found no significant mortalities of young coho and sockeye salmon force fed with 1 g oil per 100 g body weight per day. The relative lack of importance of feeding to oil uptake may be especially applicable to juvenile salmon; coho smolts were reported to stop feeding at 320 ppb of Cook Inlet WSF (Thorsteinson and Thorsteinson 1984).

The crude oil used in the experiments of Brandl et al., (1976), was originally left open for 3 d to simulate the changes in an offshore oil spill while approaching the coast, and subsequently constantly recirculated; thus the change in the aromatic content of the oil over the course of the experiment may have approximated that occurring during the weeks following a single-source oil spill. Bioconcentration of napthalenes in this study was 10 after one week, which does not seem unreasonable compared to Roubal et al., (1978) values for bioconcentration of napthalenes of 34, 49, and 117 after 2, 3, and 5 weeks, respectively.

Studies on juvenile Pacific salmon indicate that most deaths that occur during 96 hr bioassays, occur during the first 12 or 24 hrs (Morrow 1974, Rice et al., 1975, Moles et al., 1979). Thus, for pink salmon fry exposed to No. 2 fuel oil, the 24 hr TLm, at 0.89 ppm, was not dissimilar from the 96 hr TLm, at 0.8 ppm (Rice et al., 1977). Similarly, sublethal responses, including coughing, and increased opercular rates occur within the first 12 hr of exposure and subsequently decline (Thomas and Rice 1975, cited by Patten 1977; Rice et al., 1977). Brocksen and Bailey (1973) measured the increase in respiration of juvenile chinook salmon exposed to 5 ppm benzene in a continuous flow respirometer, and found the maximum at 48 hr; after 10 d in clear water the rate had returned to initial level.

There appear to be two phases of oil uptake in the juvenile salmon, with the sensitivity of the fish decreasing during exposure as the detoxification or excretion rate increases with the increasing activity of the enzymes in the liver that metabolize aromatic hydrocarbons (Moles et al., 1981). Other workers cited by these authors have shown that there is an increase in aryl hydrocarbon hydroxylase in coho salmon after their exposure to crude oil or aromatic hydrocarbons. Rice et al. (1977) speculate that there is a large energy requirement on initial exposure to oil, in order to synthesize the large quantities of enzymes needed to metabolize hydrocarbons into excretable forms, and if the fish are not overwhelmed by this initial exposure they can rid themselves of the toxic compounds. There may still be sublethal effects after this initial exposure due to the continued elevated metabolism, requiring "increased food intake which puts the fish at a disadvantage in the struggle for survival" (loc. cit.).

Oil Induced Mortalities

Delineation of the percent mortalities caused by different concentrations of oil is complicated by varying experimental designs and the different measurement techniques used to estimate the oil concentrations (IR, UV, or GC). Further complications arise when considering fish at different life history stages, and/or at different temperatures.

Sensitivity of juvenile chum, pink, and sockeye salmon to oil was increased in a more saline environment (Cardwell 1973, Rice et al., 1975, Moles et al., 1979); however, this increase in sensitivity might be negated after more than several weeks acclimation in seawater (Moles et al., 1979), or even reversed (Rice 1973). This increased sensitivity might be a function of increased physiological stress during saltwater adaptation (Moles et al., 1979), or by increased gulping of water in response to the hyperosmotic environment (Cardwell 1973). Decreases in ambient temperature also increased the sensitivity of pink salmon fry exposed to toluene in saltwater (Korn et al., 1979, cited by Moles et al., 1979). Interspecific differences in sensitivity of Pacific salmon to the water soluble fraction of diesel oil were noted by Cardwell (1973). Oil composition affects mortalities; no significant mortalities occurred for young coho and sockeye salmon exposed to crude oil weathered for 30 d, whereas significant mortalities occurred following exposure to the fresh oil (Morrow 1974).

Time spans over which oil-induced mortalities operate are more rapid than the time spans for tainting. Thus, although experimental results are often recorded as the concentration necessary to produce 50% mortalities after 96 hr (96 hr Tlm or LC 50), the majority of mortalities occur within the first 12 or 24 hrs (Cardwell 1973, Moles et al., 1979, Moles et al., 1981, Rice et al., 1977). Less than one percent of the mortalities of coho fry exposed to various concentrations of toluenes and napthalenes occurred after 96 hr (Moles et al., 1981), and Davis (1976) reported no differences between mortalities of Pacific salmon fry exposed to kraft pulp mill effluent for 200 d, and mortalities for those exposed for only 96 hr. Woodward et al. (1981), however, found the survival of cutthroat trout exposed to 520 ppb Wyoming crude oil for 90 d to be only 52%, whereas the 96 hr Tlm was 2.4 ppm; less than 2% mortalities were reported for 90 d exposures at 100-450 ppb.

The above studies have concentrated on the direct uptake of oil from the water, and do not report any effects of feeding on oiled organisms. It is possible that the gulping of hyperosmotic water observed by Cardwell (1973) would supercede any feeding effects; Morrow (1974) found no significant mortalities of young coho and sockeye salmon force fed with 1 g oil per 100 g body weight per day. This lack of documentation of the effects of feeding may prove to be inconsequential if, as Thorsteinson and Thorsteinson (1982) indicate, juvenile salmon (coho) stop feeding at 320 ppb of the WSF of Cook Inlet crude.

Several estimates of the 96 hr Tlm of juvenile salmon exposed to oil are available in the literature, with variability between the studies at least partly accounted for by the different analytical methods. I am not aware of any studies on oil-induced mortalities for adult salmon. Since most mortalities occur during the first 24 hr of exposure, a distinction between flow-through and static experiments does not seem necessary. Results from the published studies on Pacific salmon are reproduced in Table 7. Estimates of the 96 hr Tlm for sockeye salmon smolts exposed to crude oil vary from 1.0 to 4.0 ppm, the lowest value being for smolts in seawater after a 6 d acclimation period designed to simulate the sequential increase in salinities experienced by naturally migrating smolts. The 96 hr Tlm in fuel oil No. 2 was slightly lower at 0.8 ppm. With the exception of the data from Rice et al., (1975) results

Table 7. Summary of concentrations of oil in water reported to cause 50% mortalities of juvenile salmon.

Species of salmon	Oil Type	Salinity (fresh or sea)	Measurement of oil	Concentrations for 50% Mortalities (PPM)				Temperature (°C)	Notes
				24 hr	48 hr	72 hr	96 hr		
Pink fry	Prudhoe Bay crude	FW	IR	41	16	13	10	4	Rice et al. 1975. ppm extrapolated from Table 1 - mechanical mixing
		SW	IR	13	13	11	11	5	
Pink fry	Cook Inlet crude	SW	IR	4.1 (3.5-4.8)			2.9 (2.6-3.2)	10-12	Rice et al. 1977 95% confidence limits in parenthesis
	Prudhoe Bay crude	SW	IR	1.6 (1.4-1.7)			1.6 (1.4-1.7)	10-12	
	Fuel Oil No. 2	SW	IR	0.9 (0.8-1.0)			0.8 (0.7-0.9)	10-12	
Sockeye fry	Prudhoe Bay crude	FW	GC				4.0 (3.5-4.6)	7	Moles et al. 1979 95% fiducial limits in parenthesis
Sockeye smolts	Prudhoe Bay crude	FW	GC				2.7 (2.4-3.0)	8	
		SW	GC				1.0 (0.8-1.4)	8	Value for sockeye smolts in SW, estimated from relationship of total aromatics (GS) to paraffins (IR) for other bioassays: GC=1.72 + 1.30 IR
Chinook fry	Prudhoe Bay crude	FW	GC				3.6 (3.1-4.1)	6	
Coho fry	Prudhoe Bay Crude	FW	GC				3.7 (3.3-4.1)	8	
Coho fry	Toluene	FW	UV				9.8		Moles 1980
	Napthalene	FW	UV				8.8		
	Prudhoe Bay crude	FW	UV				3.0		
Coho fry	Toluene	FW	UV				6.3	7.6-10.4	Moles et al. 1981
	Napthalene	FW	UV				2.1	7.6-10.4	
Pink fry	Napthalene	SW	UV				1.2	8	Moles et al. 1983 40d LC_{50} also 1.2 for both.
	CookInlet crude	SW	GC				1.2	8	
Cutthroat trout-fry	Wyoming crude	FW	GC				2.4		Woodward et al. 1981 90d 52% survival at .52 ppm.

from all studies are within an order of magnitude (Rice et al., report in their conclusions that the 96 hr Tlm was 6 ppm, but this does not follow from the data they present earlier in their paper). Dr. Adam Moles of the Auke Bay Laboratory, NOAA, provided me with data on mortalities of outmigrant sockeye salmon exposed to varying levels of Prudhoe Bay crude oil and Fuel Oil No. 2 in seawater. These data are reproduced in Table 8 and Figure 6, where a regression has been fitted. Mortalities were approximately an order of magnitude higher for salmon exposed to Fuel Oil No. 2, compared with fish exposed to the same concentration of Prudhoe Bay crude.

Table 8. Mortalities of outmigrant sockeye salmon exposed to varying levels of Prudhoe Bay crude and fuel oil in seawater. Data provided by Adam Moles (NOAA) from static tests. Concentrations measured with GC.

Initial concentration (ppm)	N	Number of Mortalities 24 hr	48 hr	72 hr	96 hr
		Prudhoe Bay Crude in Seawater			
5.05	10	10	10	10	10
3.57	10	10	10	10	10
2.50	10	10	10	10	10
1.81	10	6	7	7	7
1.40	10	6	6	7	7
0.70	10	2	2	2	2
0	10	0	0	0	0
		Fuel Oil in Seawater			
0.600	10	10	10	10	10
0.450	10	10	10	10	10
0.339	10	6	8	10	10
0.257	10	4	7	8	8
0.161	10	2	3	3	5
0.080	10	0	0	0	0
0	10	0	0	0	0

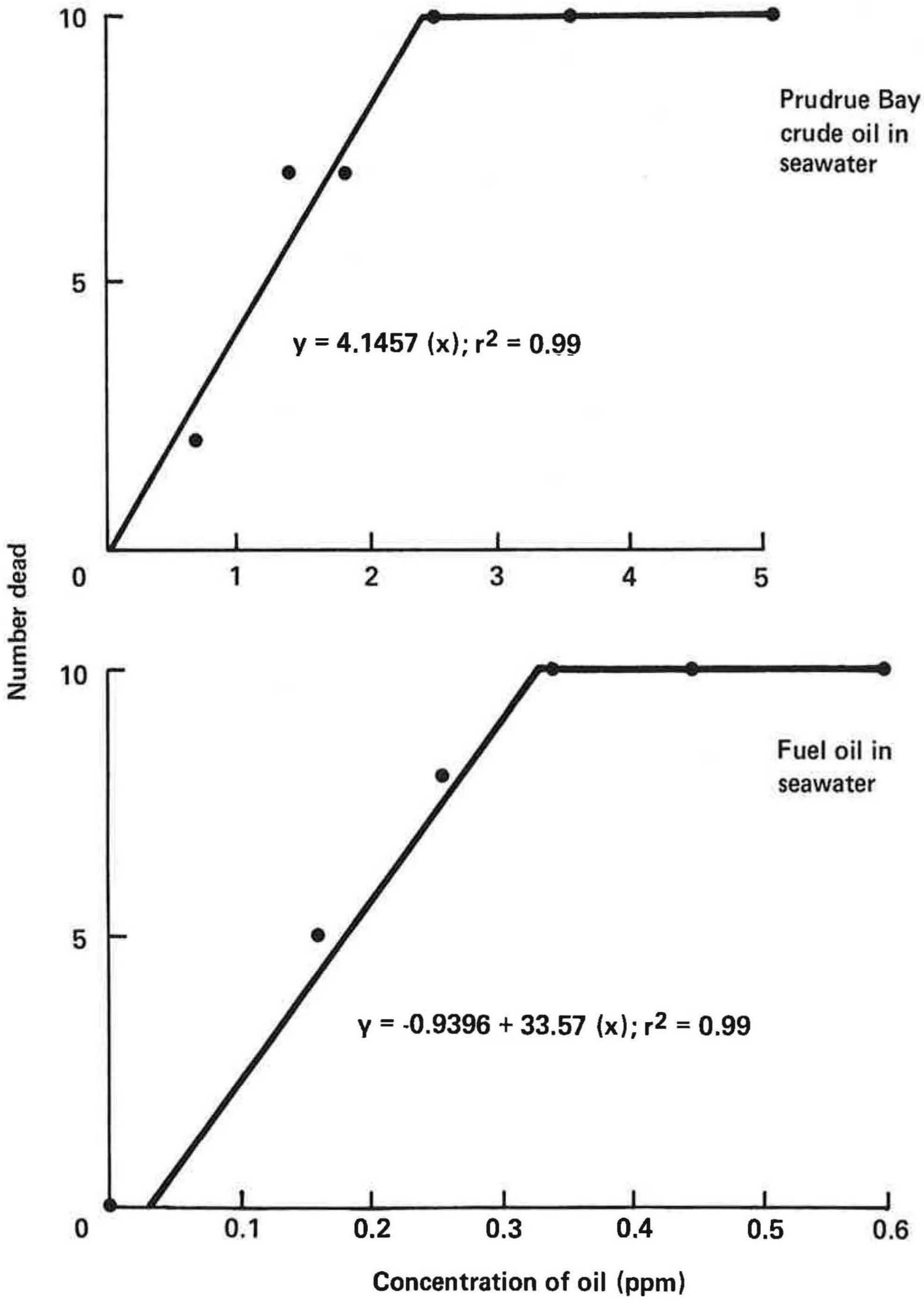


Figure 6.--Number of sockeye salmon smolts dying (out of 10) following exposure to oil in static seawater tests for 96 hr. Data from Adam Moles (NOAA).

Tainting by Petroleum Products

There have been many reports of tainting in fish flesh following their exposure to petroleum products, although the precise relationship between the implicated petroleum product and the subsequent tainting has rarely been defined. In fact, there are several instances of "oily taints" being falsely attributed to petroleum when the source was in fact dimethyl sulphide (DMS), resulting from the thermal decomposition of dimethyl-propiothetin (DMP), which occurs naturally in phytoplankton. DMS is the source of the well-known "petroleum odour" of canned salmon (Motohiro 1962, cited by Howgate et al., 1977), the "blackberry" condition of Labrador coastal cod, and the "gunpowder" condition of fish in certain Norwegian and Greenland waters (Ackman et al., 1967, cited by Howgate et al., 1967).

The degree of tainting following exposure to petroleum products may depend on the type of petroleum involved. Thus, Kerkhoff (1974, cited by IMCO 1977) reports that the middle distillate fraction of crude oil, e.g., diesel fuel, contains many of the odorous components present in the crude, and whilst diesel oil in water can be detected nasally at 0.5 ppb, fuel and crude oils can only be detected at 0.1 to 0.5 ppm (Martin 1970, cited by IMCO 1977). Reports of tainting have followed spills of diesel fuel. A spill of 2,200 tons of diesel oil near Finnsnes, northern Norway, resulting from the grounding of a tanker in 1973, was spread by wind and currents in the Gisund, a narrow strait. In the ensuing two months fishermen reported the smell and taste of oil in the fish - cod, saithe, haddock, herring, flounder, sea trout, and salmon. Two months after the spill sampled fish were tasted by a trained taste panel and the flesh tested with gas chromatography and mass spectrometry. The results were in agreement with the reported tainting; levels of hydrocarbons were still in the order of 150-200 ppb in the liver of the cod even though the components of diesel oil were no longer present in the samples of water and sediments taken at the site (Palmork and Wilhelmsen 1974, cited by IMCO 1977). Eleven days after a spillage of diesel oil into a river containing brown trout "both the fish after cooking and the hydrocarbon fraction isolated from the uncooked fish were found to smell and taste in a manner reminiscent of the fuel oil" (Mackie et al., 1972). A "very crude" estimate of the contamination in the trout flesh was 300 ppm. Following the Torrey Canyon oil spill, tainting of mackerel and sea trout was reported; however, there is some question as to whether the tainting resulted from the original spill or from the petroleum dispersants used to clean up the spill (Simpson 1968, cited by IMCO 1977).

Several laboratory studies further delineate the hydrocarbons responsible for tainting as the napthalenes and perhaps the benzenes. Anderson and Neff (1974, cited by IMCO 1977) exposed the killifish, Fundulus simulus, to the water soluble fraction of No. 2 fuel oil in synthetic sea water for 2 to 24 hr, and then transferred them to oil-free seawater recirculated through activated charcoal. Total napthalenes were accumulated very rapidly with the maximum tissue concentration reaching 23 times the exposure concentration (1.9 ppm) after 2 hours. Depuration of the total napthalenes resulted in undetectable levels after 14 d and complete depuration within 3 to 4 weeks. An extensive study by Shumway and Palensky (1973), in which rainbow trout were exposed to individual organics in a flow-through chamber, indicated no further significant increases in the mean off-flavor index after 1.7 hrs. Significant losses in flavor impairment occurred after 6.5 hrs in fresh water, with no detectable impairment of flavor remaining after 33.5 hrs. Howgate et al., (1977) reported an oily taint in plaice kept for 1 to 2 days on oiled sediment. The equivalent oil concentration in their flesh at this time was approximately 160 ppm which did not change over the next 8 days the fish were retained on the oiled sediment.

A well executed study designed to simulate the effects of an oil spill weathered for several days on entrained fish was conducted by Brandl et al., (1976). One hundred saithe (3-400 g), and 50 salmon (100 g) were kept in a flow through vessel (turnover rate 30 min) for 68 d, during which time they were not fed. The water, oil, and the liver and muscle of the fish were analyzed using gas chromatography coupled with mass spectrometry. Organoleptic analysis of individual fish wrapped in aluminum foil and steam boiled in separate pans was conducted by a taste panel of trained personnel. Tainting in the salmon was first observed after 4 d exposure to the water contaminated with an average 40 to 50 ppb hydrocarbon. The tainting became obvious after 6 d, had returned to the 4 d level after 13 d and after 30 d was identical with the controls. The saithe on the other hand had a slight indication of a taint after 22 d, but no trace of a taint after 28 d. The only components of significant concentration found in the fish were napthalenes and benzene compounds, the latter being too volatile for quantitative determination; the authors suggest that the benzenes and benzene compounds were present at levels roughly equivalent to those of the napthalenes. The concentration of napthalenes reached a maximum of 0.5 ppm after 6 to 8 d for the salmon muscle (bioconcentration of 10), and depurated to 0.03 ppm after 39 d. The liver of the salmon showed higher concentrations of napthalenes, reaching a maximum of 42 ppm (bioconcentration of 1000) after 7 hr, decreasing to 2 to 3 ppm after 6-8 d. The saithe showed a similar trend over time for concentrations of napthalenes in the muscle and the liver, with maximum concentrations of 0.09 ppm and 10 ppm, respectively. The authors conclude that, therefore, a concentration of 0.3 ppm napthalenes is necessary for tainting to be observed.

The 0.3 ppm tainting threshold level suggested by Brandl et al., 1976 is an order of magnitude lower than the other available estimates as reviewed by IMCO 1977 which range from 5 ppm of gas oil in spiked tissue from muscles (Kerkhoff 1974), to 4 to 12 ppm of diesel oil components in lobster (Paradis and Ackman 1975), to 10 to 30 ppm of crude oil in spiked tissue (Whittle and Mackie 1974). However, Shumway and Palensky (1973) found that the threshold tainting levels for 34 organics in rainbow trout ranged over 5 orders of magnitude, from 0.4 ppb for 2, 4 - Dichlorophenol to 95 ppm for formaldehyde. Some compounds, e.g., acetone, did not even impair flavor at levels up to 1000 ppm. The same authors concluded that the flavor impact of organic compounds are not additive, although some interactions may occur. IMCO 1977 also cites a personal communication that a trained taste panel could detect no further increases in tainting above 200 to 300 ppm of crude oil in spiked tissue.

The importance of tainting to commercial fisheries following exposure of the resource to hydrocarbons will depend not only on the severity of the contamination itself, but also on the rate of uptake and the rate of depuration. These factors have been considered in an earlier section. Another component of tainting is diffusion of tainted fish through a larger population. Thus a kerosene-like taint in the sea mullet (Mugil cephalus) thought to be due to their exposure to refinery effluents, resulted in the condemnation of 78 short tons near Brisbane (Grant 1969, cited by IMCO 1977); the tainted fish were spread over 100 miles of coast mingling with untainted fish, resulting in the condemnation of entire catches (Connell 1974, cited by IMCO 1977). For this reason the average tainting in a population, especially a highly migratory population like the Pacific salmon, is an insufficient measure of tainting; proportions of the population that would be tainted and the degree of tainting of these different proportions will be required for effective decision making.

Additional Sublethal Effects

Many sublethal effects of oil (and other pollutants) on salmonids have been suggested, but it is often unclear to what extent these effects would result in a decrease in the overall fitness of the population. Reduced growth rates of juvenile coho salmon in the laboratory followed their exposure to 32% of their LC_{50} of toluene and napthalene (Moles et al., 1981), and reduced growth rates are expected for outmigrant sockeye salmon exposed to oil concentrations as low as 100 ppb (Moles, personal communication). Growth rates of flatfish in the area at the time of the Amoco Cadiz spill were reduced (Gundlach et al., 1983), thus this effect is not restricted to laboratory situations. The effects of a reduced growth rate are not clear since growth compensation may occur at a later time (Mount 1968, cited by Sprague 1971); however, reduced growth rates initially would cause the juveniles to be susceptible to size selective predators over an extended period of time. Additional sublethal effects such as a decrease in responsiveness to fright stimuli (Cardwell 1973), or the development of eye lesions, to which salmonid eyes are particularly susceptible (Hawkes 1977, Woodward et al., 1981), would also increase susceptibility to predators; Thorsteinson and Thorsteinson (1982) report that increased predation of coho salmon on juvenile chum salmon occurred during their first three days of exposure to Cook Inlet crude oil, but the data source is not provided. Development of eye lesions could also lead to behavioral changes leading in turn to increased predation; thus, Mace (1983) found that in a year when poor nutrition led to visual deficiencies in juvenile chinook salmon the rate of bird predation on these fish over a 2 km riverine outmigration increased from 10-12% to 21-32%.

If chemoreception is reduced in salmonids following their exposure to oil, as found for two estuarine fish (Solangi 1981), then together with a reduced visual acuity this could change schooling behavior (thought to be an important anti-predator mechanism) and other behavioral mechanisms necessary to the highly evolved migratory and reproductive behavior in salmonids. Moore and Dwyer (1974) consider that feeding, reproduction, and social behaviors in fish can be disrupted by "soluble aromatic derivative" concentrations as low as 10-100 ppb, however, their data sources are not clear. Disrupted schooling behavior, including swimming to the surface, and/or the source of the oil in bioassays have been reported by several researchers. Migration patterns could be disrupted, with the possibility of an increased spatial overlap with predators, if temperature preferences were affected by exposure to oil as noted by Oligivie and Anderson (1965, cited by Sprague 1971) for Atlantic salmon exposed to DDT. However, Salo et al. (1983) found that the brief exposure of returning adult chinook salmon to crude oil at concentrations representative of oceanic spills had no detectable effect on their homing abilities over the final 7 km of freshwater migration.

Exposure of the adults to oil might affect the viability of subsequent offspring; thus, Gundlach et al. (1983) report that following the Amoco Cadiz spill flatfish showed an increase in reproductive pathologies in addition to reduced growth, and that there was an almost complete absence of juveniles in the year following the spill. Burdick et al. (1964) found that trout fry from parents exposed to DDT died at a certain stage in their development due to DDT received from their parents in the yolk. Contrasting with these studies is that of Hodgkins et al. (1977) who found that trout fed 1 g of Prudhoe Bay crude oil per kg food for 6-7 months before maturity showed no changes in reproductive timing, fertility, fry survival, or in the gross morphology of the fry.

Exposure of salmonids to oil in the water can result in an increased metabolism (i.e., increased respiration - Brocksen and Bailey 1977, Rice et al., 1977) and a decrease in the glycogen reserves in the liver (Hawkes 1977). This could be especially significant for returning adult salmon which are on a fixed energy budget (no further food intake) in the final stages of their migration and can be expending energy at a rate equivalent to 80% of that at the maximum swimming speed (Brett 1965). Sockeye salmon are recorded as using 78-96% of their protein and 31-61% of their fat reserves by completion of spawning (Idler and Clemens 1959). Their stamina, necessary in the final migration might be reduced; Adams (1975) found the stamina of brook trout (Salvelinus fontinalis) to be significantly less than controls following an exposure of 2 and 3 weeks to a 14.3 ppb lead, and 0.5-5 ppm hydrocarbons. Reduced stamina and an increased metabolism might be significant for the juveniles which stop feeding at low concentrations of oil (320 ppb for coho salmon - Thorsteinson and Thorsteinson 1982) and indeed feeding might be the most sensitive indicator of the sublethal effects of pollution in salmon (Bull and McInerney 1974).

Reductions in the ability to withstand further stress following the exposure of estuarine fish to oil was noted by Solangi (1980), and an epidemic of lethal ulceration of Atlantic salmon, probably triggered by a surge of Cu - Zn pollution and high water temperatures noted by Pippy and Hare (1969, cited by Sprague 1971). This reduced resistance to infection could also result from an altered microflora on the epidermis as noted for striped mullet exposed to crude oil (Giles et al., 1978, cited by Woodward et al., 1981).

It is difficult to evaluate the importance of sublethal effects of oil on salmon. In years or areas where the fish are exposed to few concurrent, or subsequent, stresses their importance might be reduced, although permanent histological changes such as eye lesions would be expected to be continuously detrimental. Maurin (1981) considered the medium and long term damage of the Amoco Cadiz spill to be more important than the immediate effects, and indeed for the returning salmon a cessation of harvesting following reports of tainted fish can be hypothesized as an important medium and long term effect of an oil spill.

OVERVIEW OF SIMULATIONS

INTRODUCTION

Juvenile and adult sockeye salmon are present at specific areas in Bristol Bay for only a short time each year, and consequently it would be inappropriate to arbitrarily select a single time of the year in which to simulate the effects of an oil spill. Instead two models were used to simulate the maximum effects of a specific oil spill scenario on the migrating salmon. The first simulation estimates the maximum proportion of the Bristol Bay sockeye salmon that could be at the longitude of the spill area over a 10 d period, and the second simulation estimates the proportion of the population passing through the spill area in 10 d that would be affected by that spill. Together with data on the inshore/offshore distribution of the salmon, these two simulations provide an estimate of the maximum impact of a specific oil spill scenario. Parameterization of the models tends towards maximizing potential effects of the oil spill when data are ambiguous. Both simulations are run separately for the adults and juveniles.

PRESENCE IN SPILL AREA

Juveniles

Available data from the sampling of juvenile sockeye salmon in Bristol Bay (Straty 1974, Hartt 1980) indicate substantial numbers of juveniles in the Inner Bay from late May to early August, and in the Outer Bay from as early as June to at least September (no data are available for distributions after September). Annual variations in the timing occur within the above ranges. While these data indicate the period of the year during which an oil accident could be expected to exert a direct effect on the juveniles migrating through Bristol Bay, they are insufficient to estimate the proportion of the juveniles potentially affected.

Data on the timing and numbers (or relative numbers) of sockeye salmon smolts leaving the Kvichak, Naknek, Ugashik, and Wood rivers are available for most years from 1955 to present (Rogers 1977). These data together with the migration rate of the juveniles through Bristol Bay derived from Straty's (1974) data on marked fish, can be used to simulate the time that specific proportions of juvenile sockeye salmon from each river pass designated areas on the southeastern shore of Bristol Bay in any of the years for which data are available.

The above approach was used by Eggers and Rogers (1978) and I have modified and reparameterized their original simulation in this study. In this simulation the proportions of the total smolt outmigration passing the counting station in any 5 d period are read in, together with an estimate (or index) of the total numbers, their mean size and age composition. Additionally, the estimated adult returns to the Bristol Bay inshore fishery from these outmigrants are input at this time. Computations in the simulation start at May 15, with the smolts (if any) outmigrating from each river within the 10 d period to May 24 entering Bristol Bay at the estuary of their natal stream. Migration through the Bay is simulated as a straight line migration (no dimensions of offshore or vertical distribution are included) along the southeastern shore. Simulated migrations of the Wood River smolts include the initial southerly migration to reach the southeastern shore described by Straty (1974).

For the simulated time periods beginning May 25 and subsequently, smolts from each river enter Bristol Bay at the estuary of their natal river when the simulation time corresponds to their passage past the counting station. Any smolts which entered the simulation in previous time steps are moved along the straight line migration route at 0.9 $BLsec^{-1}$ and their new location recorded. For convenience Naknek estuary is defined as 0 km; locations further into the Bay (including the Wood River) are negative and those further out of the Bay are positive. The distance moved in any time step will depend on the size of the juveniles. Data input to the simulation include the mean size of the smolt outmigrants from each river and year, and since no consistent trend in size over the duration of the outmigration was found in earlier analyses, it will be assumed that this mean size is equally applicable to all portions of the outmigration. Straty (1974) concluded that no (or very limited) growth occurred in inner Bristol Bay, but that in the outer Bay growth could result in a doubling of fish size in 8 weeks. For a 100 mm fish this suggests a growth rate of 1.8 mmd^{-1}. In this simulation once the migration of a group of juveniles has taken them past Port Heiden and into the outer Bay their size is considered to increase by 18 mm for each 10 d period.

At any timestep in the simulation, and for any year's outmigration data, the distribution of the individual proportions of the outmigration by river and age class are available as a function of their distance along the migration route through Bristol Bay (Fig. 7). From these outputs it is a simple matter to compute the time and magnitude of the maximum proportion that would be at a designated area over a 10 d period in any year, either by river and age class, or as the combined maximum proportion for all rivers, where the proportions from individual rivers are weighted by the numbers of the resulting adult returns.

Adults

The timing of adult sockeye salmon returning to Bristol Bay is both well defined and limited to a short time period. Eighty percent of run passed the fishery over a 12.9 d period (S.D., 1.58 d) from 1955 to 1976 (Burgner 1980); mean time of return over the same period was July 4 (extremes June 28-July 10; S.D., 2.92 d).

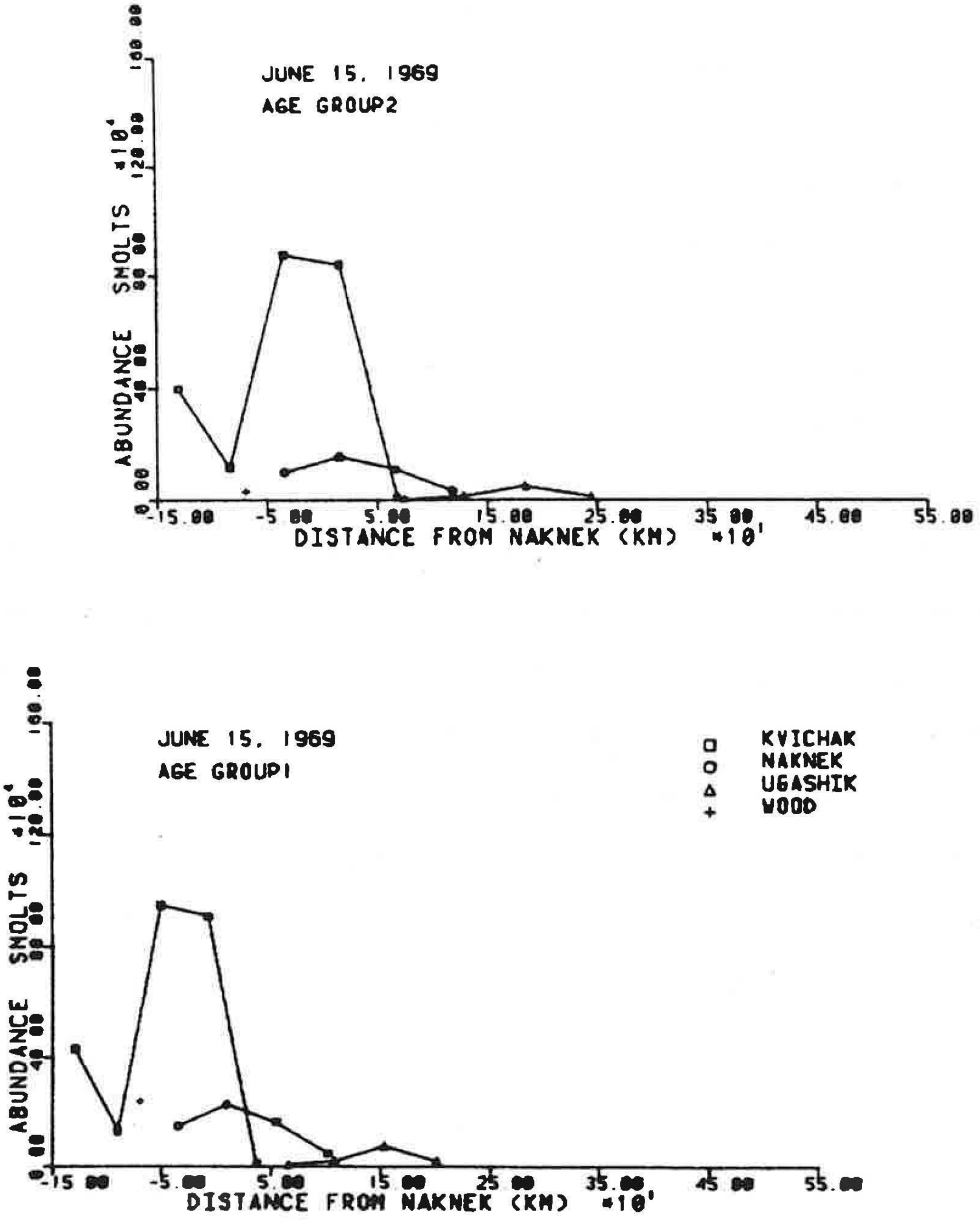


Figure 7.--Simulated abundances of juvenile sockeye salmon migrating through Bristol Bay. (No allowance made for mortality.)

In a year when the return timing was highly aggregated (lower 95% C. L. for duration of returns), 80% of the run would pass the fishery over a 9.8 d, or approximately over a 10 d period. Migration of the adult salmon from the time they enter outer Bristol Bay until they reach the fishing districts in inner Bristol Bay appears reasonably direct (300 km in 6 d, with the sequence of arrivals at Port Moller corresponding to that at the fishing districts), so these data on return timings are applicable to the oil spill scenario areas. This district run timing obviates the need for a simulation to estimate the presence of adult sockeye salmon in the spill areas.

EFFECTS OF OIL SPILL ON MIGRATING SALMON

Effects of the oil spill scenarios developed by SAI on adult and juvenile sockeye salmon are simulated using one model with only minor reparameterization for the two life history stages (e.g., migration direction, fish size, inshore distribution). A general description of the simulation follows and is illustrated in Fig. 8. Fish, or schools of fish[1/], are assigned to the squares in the grid in proportion to their historic probability of occurrence, with the maximum probability receiving 5 fish for each square. Each fish is recorded individually and will be treated individually throughout the simulation. Final output is as the statistical means and accompanying variances of these fish.

For each fish in each timestep a working grid is set up composed of the eight adjacent squares (Figs. 8 and 9). Values for probabilities of occurrence and oil concentrations are transferred from the main grid to those working grids at each timestep. The probability of migration in the direction of any of the

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[1/] The individual units of fish biomass move independently of one another and can be considered as individuals or highly cohesive schools.

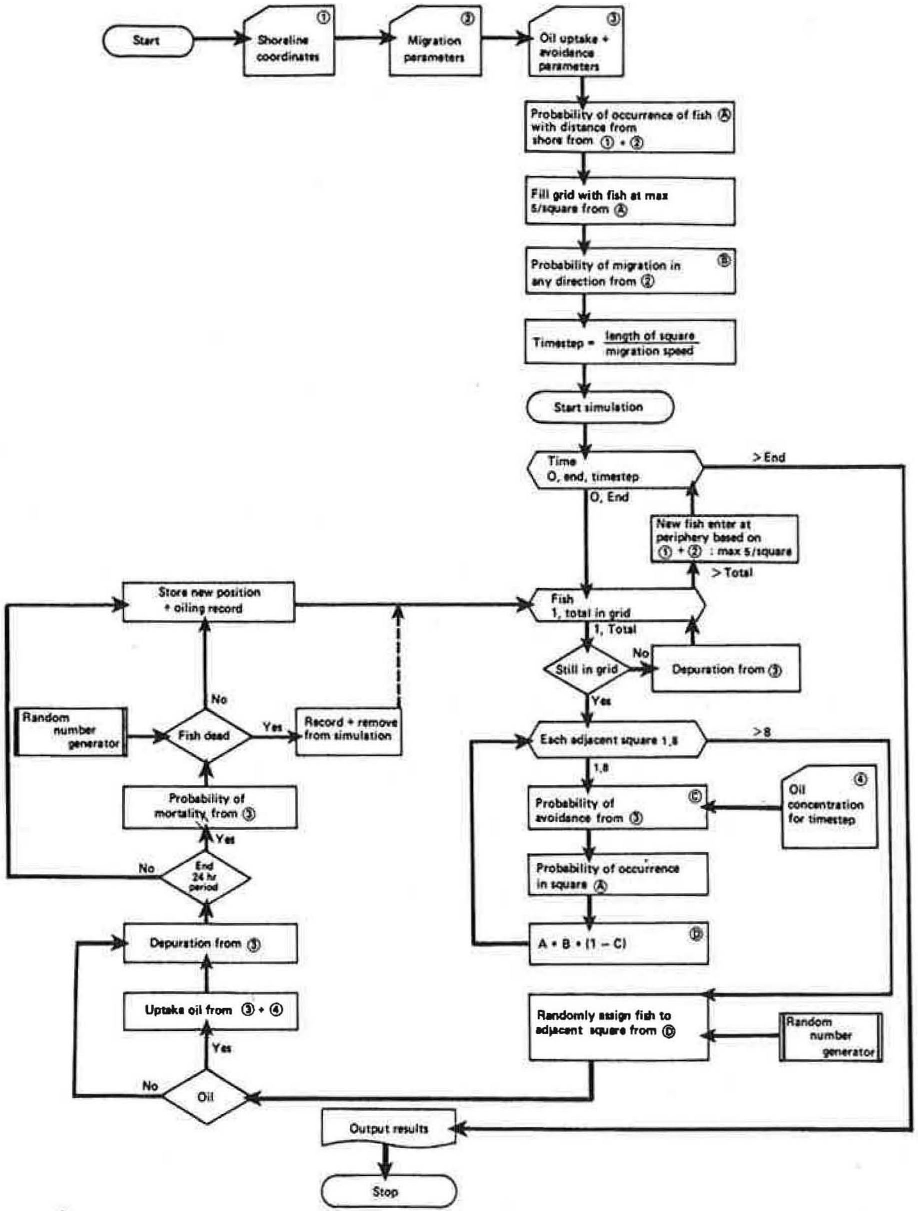


Figure 8.--Flow chart of processes involved in simulating the movement of sockeye salmon through an oil spill in Bristol Bay.

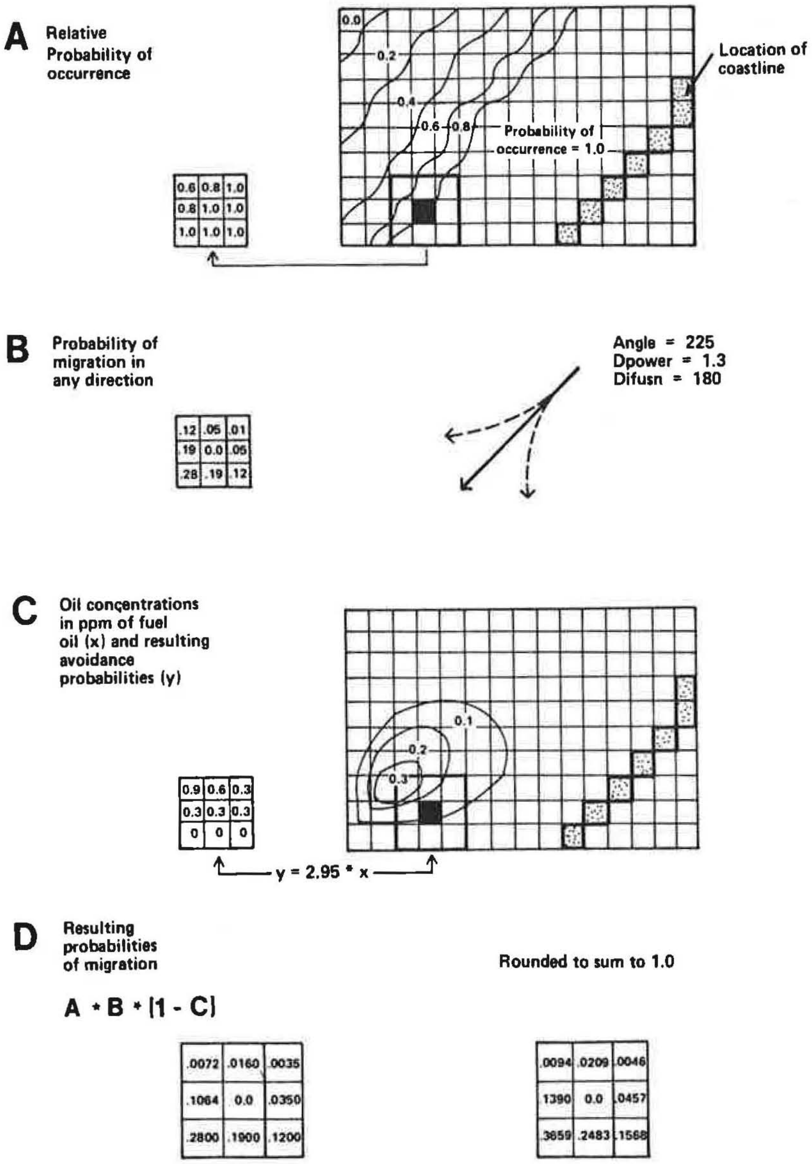


Figure 9.--Schematic of the simulation of movement of one juvenile sockeye salmon over one timestep in the presence of oil in Bristol Bay.

adjacent squares in the working grid is computed at this time. The timestep in the simulation is set to equal the time taken for a fish to move to an adjacent square (i.e., the length of an individual square (2 km), divided by the migration speed in km/hr). At each timestep the fish must move to an adjacent square, the selection of the square being dependent on the product of the historical probabilities of occurrence, the probabilities of migration in the direction of each square, and the probabilities of avoidance of the ambient oil concentration (if any) in each square.

Two parameters are required in this simulation to compute the probability of migration in any direction - the preferred migration direction (ANGLE) and the spread around this preferred direction. In the simplest instance the preferred migration direction is set constant all over the grid; parallel to the shore and out of Bristol Bay for the juveniles (225°), parallel to the shore and into Bristol Bay for the adults (45°). For a more involved simulation area, e.g., migration into and out of Ugashik Bay, a grid containing the preferred migration direction from individual squares is read in. Two options are available to simulate the spread around the preferred migration direct. The first option (direct migration) permits no spread around the preferred direction, and consequently no avoidance of an oil spill. The second option assigns probabilities of migration (COMPROB) in any direction of the compass (COMPAS) from the following equation:

$$\mathrm{COMPROB} = ((\mathrm{DIFUSN} - |\mathrm{COMPAS{-}ANGLE}|)/\mathrm{DIFUSN})^{\mathrm{DPOWER}};\ \mathrm{min}\ 0.01$$

where both the total angle of diffusion (DIFUSN) and the gradient of probabilities (DPOWER) can be adjusted. Examples of this equation are:

(i) DIFUSN = 180.
DPOWER = 1.3
ANGLE= 225.

0.12	0.05	<0.01
0.19		0.05
0.28	0.19	0.12

(ii) DIFUSN = 180.
DPOWER = 3.0
ANGLE = 45.

0.06	0.20	0.47
0.01		0.20
<0.01	0.01	0.06

where the probabilities in each instance have been rounded down to sum to 1.0.

There are insufficient data available to describe the small-scale coastal migration patterns of Pacific salmon. It is a common observation that the juveniles are to be found within a short distance of the shoreline, but movement of individual fish or fish schools within this area is not detailed. Vertical migration patterns of returning adult chum and Atlantic salmon have been described by Ichihara and Nakamura (1980) and Westerberg (1983), and indicate that the migration route of returning adult salmon would not be adequately described by a straight line. For neither the juvenile nor the adult sockeye salmon are empirical data sufficient to estimate the degree of diffusion around the preferred (or mean) migration direction. Available data indicate that both adult and juvenile sockeye salmon migrate approximately parallel to the southeastern Bristol Bay shoreline and that their mean migration rate into and out of the Bay, respectively, is 0.9 BL/sec. Many combinations of swimming speed and diffusion could simulate these migration characteristics; in this simulation I have defined only the upper and lower limits of swimming speed.

The lower limit of swimming speed would be 0.9 BL/sec under direct migration when the swimming speed would equal the mean migration rate. Under this condition no avoidance of the spill would be possible. An upper limit to swimming speed was defined from the theoretical studies of Weihs (1975), Trump and Leggett (1980), and Wakeman and Wohlschlag (1981) which suggest a maximum likely swimming speed of 2 BL/sec. In combination with a total angle of diffusion (DIFUSN) of 180° in either direction and a gradient of probabilities (DPOWER) set at 1.3 (see (i) above) this swimming speed of 2 BL/sec will produce the desired mean migration rate of 0.9 BL/sec. Under the above assumptions this combination will result in the maximum possible avoidance of the spill in this simulation.

In the simplest situation where there was no oil in the adjacent square and the historical probability of occurrence was equal for all squares, the above formulations would determine the probability of movement in any direction. More involved situations arise when the historical probability of occurrence is not equal in all directions. These historical probabilities are read into the working grid from the master grid. Their derivation in the master grid is based on the distance of each square from the shoreline, since historical catch data suggest that the abundance of juveniles is constant from the coastline to 40 km offshore, thereafter dropping to zero at 54 km offshore (Straty 1974), and that the abundance of adults increases from zero at 13 km to a maximum at 64 km offshore, thereafter dropping to zero at 178 km (Straty 1975). These functions are illustrated in Fig 10.

The last factor that would affect the migration direction is the presence of oil in any of the adjacent squares. Oil concentrations from the master grid for the appropriate timestep are transferred to the working grid. Avoidance (AVOID) of that level of crude (OIL) is determined from regression functions fitted to the data of Weber et al., (1981) for the adults:

$$\text{AVOID} = 0.16 * \text{OIL} \qquad ; \text{max } 0.90$$

and to the data of Maynard et al., (1981) for the juveniles:

$$\text{AVOID} = 0.295 * \text{OIL} \qquad ; \text{max } 0.90$$

Maximum avoidance is limited to 0.90 in both instances, because there are indications of variability in the avoidance response of individual fish, and the two above sources provide no data on potential avoidance above 0.75.

No comparable data are available for the potential avoidance of concentrations of fuel oil. I assume that the sevenfold greater sensitivity to fuel oil compared

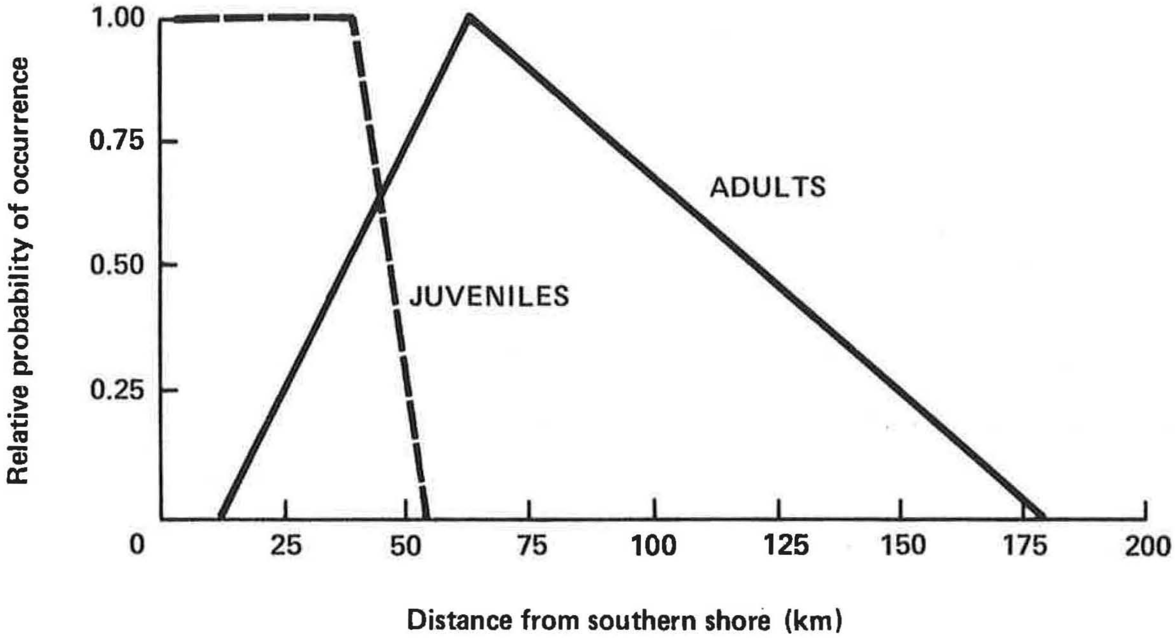


Figure 10.--Functions describing the relative possibilities of occurrence of sockeye salmon with distance from shore between Port Heiden and Port Moller in Bristol Bay.

to crude oil as measured by the probabilities of mortality (described later) can be transferred directly to the probabilities of avoidance. Thus in the simulation fuel oil will be avoided with a sevenfold greater probability than crude oil at the same concentration.

The three probability distributions described above - migration direction, historical probability of accurrence, and avoidance of oil - are multiplied together and the product rounded to equal 1.0 (Figs. 8 and 9). Based on this final probability distribution the fish is randomly assigned to one of the adjacent squares. If the square to which the fish has been moved contains oil then this oil is taken up into the flesh in a prescribed manner and in either case any existing oil in the flesh is depurated. Uptake of oil and its depuration are based on the experiments of Brandl et al. (1976) which measured uptake to, and depuration from, the flesh of Atlantic salmon contained in a flow-through chamber designed to replicate the oil concentrations following an actual oil spill. In the absence of any evidence to the contrary, these rates of uptake and depuration are considered identical for the juveniles and adults. Uptake of oil to the flesh over the course of a single time step is equal to the product of the external concentration and the bioconcentration rate per timestep. A bioconcentration rate of 3.0 is suitable to simulate the results of Brandl et al. (1976). This bioconcentration rate is increased to 21.0 for fuel oil, the justification for this adjustment having been presented previously.

Depuration rates as approximated from Brandl et al. (1976) results will include compensation for the loss of the more volatile components from the dissolved and emulsified oil which is not allowed for in the oil spill scenarios provided. The major facet of depuration which is set to increase by 3% per day from zero on the day of first contact with oil to a maximum of 90% loss of oil per day after 30 days, simulates the increase in appropriate enzyme activity by the affected fish. Together the expressions for uptake and depuration produce maximum concentrations of oil in the flesh after 7 days, with maximum bioconcentrations of 11 and 77 for crude and fuel oil, respectively (Fig. 11). Internal concentrations of oil in the flesh decrease to 50% of the original ambient levels after approximately 30 days. These two expressions determine the amount of tainting predicted by the simulation, where any fish with an internal flesh concentration greater than, or equal to, 0.6 ppm is considered tainted.

Direct mortality from exposure to oil is handled separately from the uptake and depuration of oil to the flesh. While several researchers have described the mortality of juvenile salmon at various oil concentrations, no published studies are available relating these mortalities to concentration levels in the various organs. Accordingly in this simulation I represent oil-induced mortalities as a direct function of the exposure history of the fish rather than attempting to simulate the uptake of oil to sensitive organs and the internal lethal levels. A frequent manner of reporting oil-induced mortalities in experimental situations is as the external concentration that will cause 50% mortalities after 96 hr, this despite the fact that most mortalities that are going to occur in a 96 hr exposure will have occurred in the first 24 hours. In this simulation the mean exposure concentration over the previous 24 hr period is computed and the probability of mortality at this mean exposure level computed according to functions derived from Moles (unpublished data).

(i) FUEL OIL $P(\text{MORTALITY}) = 0.09396 + 3.357(\text{XCON})$

(ii) CRUDE OIL $P(\text{MORTALITY}) = 0.41457\ (\text{XCON})$

where XCON is the mean external concentration of oil in ppm experienced by the fish over the previous 24 hrs (Fig. 12). Two constraints are placed on these

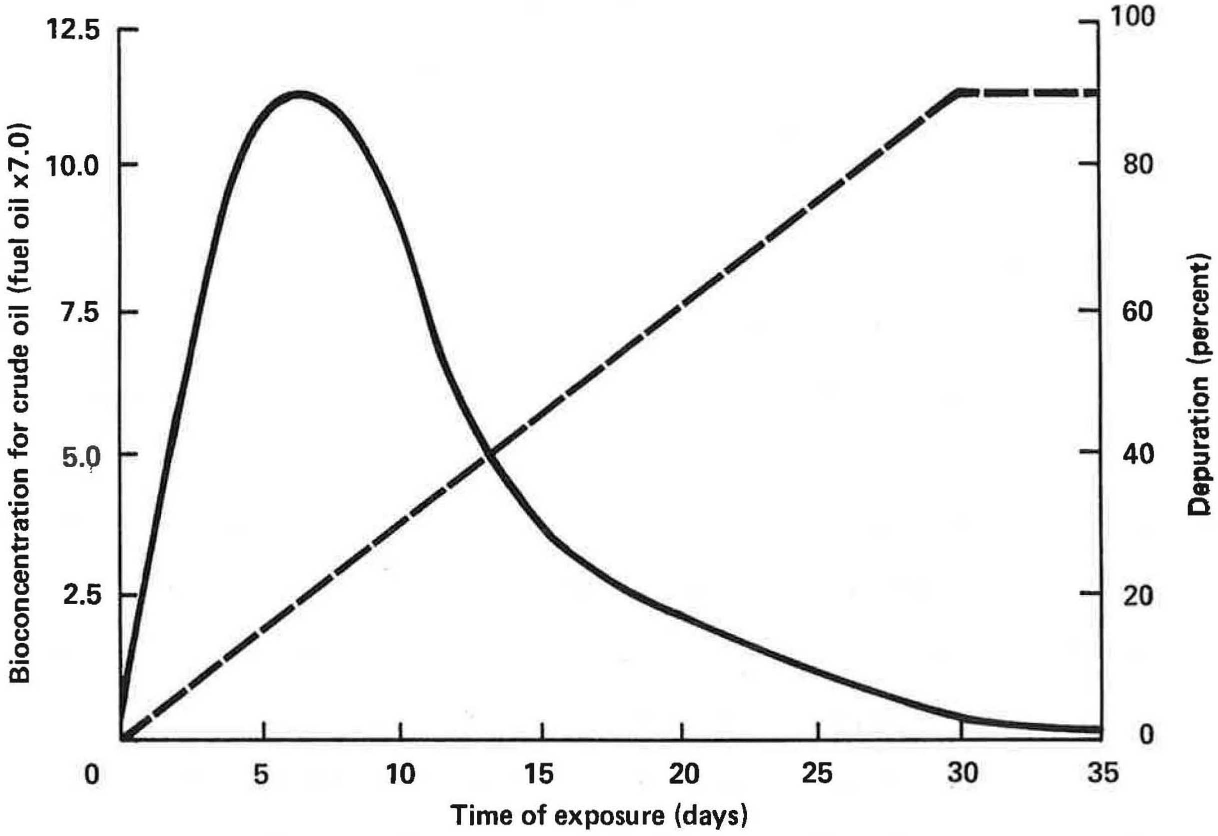


Figure 11.--Functions describing the bioconcentration and depuration of crude oil in the tissues of salmon exposed to a naturally weathering oil spill. (Bioconcentration factors for No. 2 diesel oil are 7 times higher than those for crude oil.)

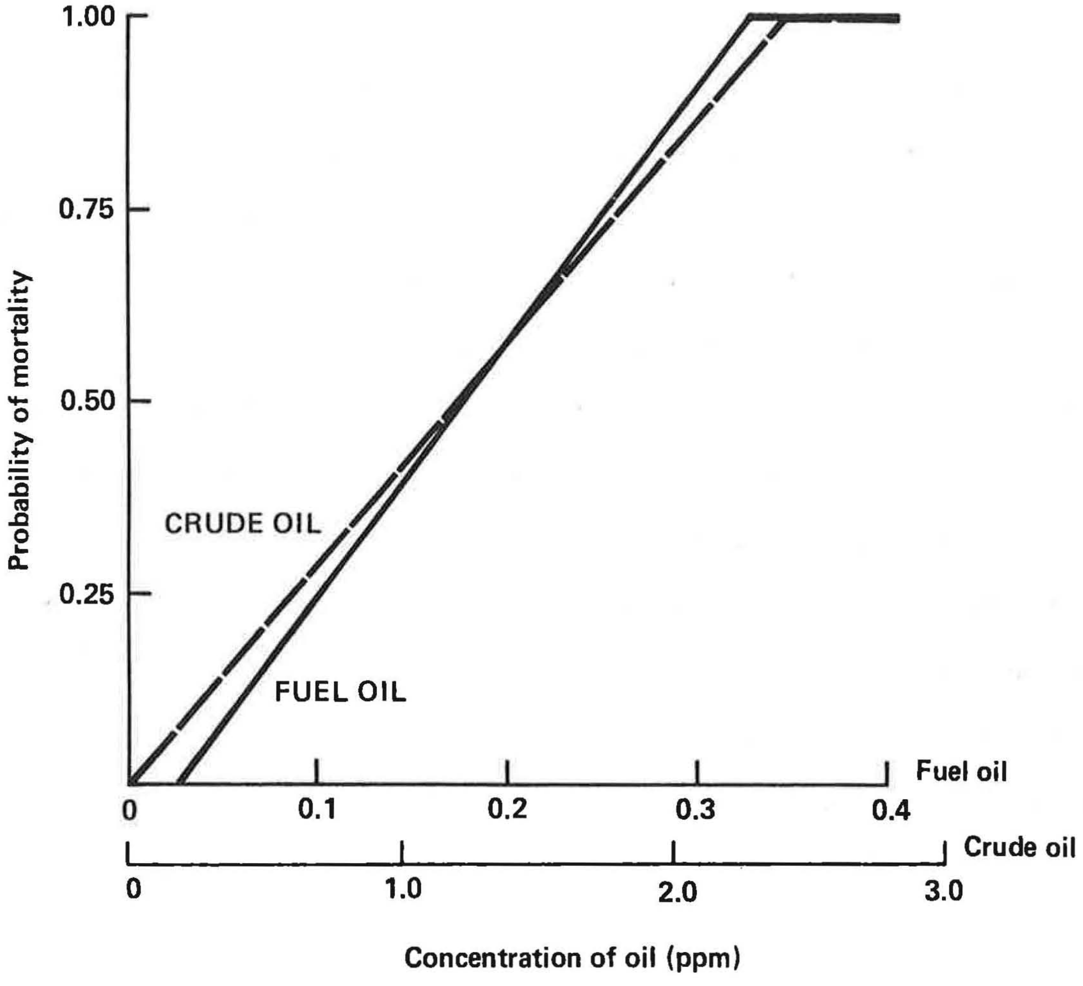


Figure 12.--Functions describing the probabilities of mortality of sockeye salmon exposed to oil in the marine environment over a 24 hr period.

calculations to preserve independence in the mortality probabilities between 24 hr periods. Firstly, the probability of mortality is computed only at the end of distinct 24 hr periods (i.e., at 24, 48, ..., 240 hrs), and secondly the probability of mortality is computed only if the current mean exposure concentration is greater than that immediately predating it. In conjunction with this calculated probability of mortality, a random number generator is used to decide the fate of the fish. When a fish is considered dead, it is removed from future timesteps and is excluded from all summary statistics (e.g., for tainting).

This simulation of oil-induced mortalities completes the simulation for the individual fish in this timestep, and the simulation now proceeds to the next fish (Fig. 8). When the migration simulations take a fish outside of the main grid, the fish is flagged and in future timesteps undergoes depuration only. At the end of each timestep some fish will have left the grid and must be replaced. At the same time fish will have moved away from the squares on the perimeter(s) at which they would enter the grid. Accordingly these perimeter squares are replenished at the end of each time step so as to equal the maximum number indicated by their associated historical probability of occurrence (where the maximum probabilities in the grid would receive 5 fish). The simulation is now ready to proceed to the next timestep.

RESULTS

PRESENCE IN AREA

Juveniles

Simulations were run for all years from 1955 to 1982 for which there were data, and the maximum percentage of the juveniles from each river occurring in the area of the spill over a 10 d period determined. No diffusion of the juveniles was allowed for in the simulation since although data from Straty (1974) indicates a threefold spread in distribution upon saltwater entry, no information is available on the final shape of the distribution. Data on pink salmon migrating from the Bella Coola River system suggest a saltatory movement (Healey 1967); data from chum salmon migrating through Hood Canal indicate a multimodal distribution (Bax 1983). The spread of the juveniles will be the same through both the Port Heiden and Port Moller spill areas with the above simplification; absolute numbers will be higher in the Port Heiden spill area which is closer to shore than the Port Moller spill area.

Results are presented in Table 9. For individual rivers a higher percentage of the one year olds than the two year olds were at the latitude of the spill areas over a 10 d period due to the greater length and thus migration speed of the two year olds leading to a more diffuse distribution. Juveniles from the Kvichak River had the most clumped distribution, and thus the greatest percentages at the latitude of the spill areas at any one time (65 or 59%; Table 9). Those from the Naknek River were the most diffuse (40 or 37%). This variability in distribution of the migrants between rivers is caused by differences in the number of independent rearing areas on each river.

Only 10 years of data were available when the model was run to simulate the maximum weighted proportion of the summed juveniles from the four rivers at the

Table 9. Mean and standard errors of the maximum percent of the outmigration present within the east and west boundaries of the simulation grids over a 10 d period. Data encompasses 1955 to 1982 as available.

River	Age	n	% (SE) in latitude of grid over 10 d Tanker accident	Blowout
Kvichak	1.	24	65 (11)	87 (8)
	2.	24	59 (12)	77 (9)
Naknek	1.	21	40 (8)	60 (12)
	2.	21	37 (10)	56 (13)
Ugashik	1.	14	62 (14)	79 (8)
	2.	14	44 (7)	71 (11)
Wood	1.	13	49 (12)	68 (11)
	2.	13	41 (10)	62 (13)
Combined Rivers	1.	10	36 (7)	54 (11)
	2.	10	47 (16)	65 (16)

latitude of the spill areas. Between 36 and 65% of the juveniles from four of the principal sockeye salmon rivers in Bristol Bay were simulated to be at the spill areas latitudes over a 10 d period (Table 9).

Median simulated times and the range in simulated timing of the maximum proportions at the latitude of the spill areas are given in Table 10. The range in timings was approximately 2 months in both areas; early June to early August at Port Heiden, and mid June to mid August at Port Moller.

Actual proportions of the outmigration present in a spill area over a 10 d period are determined by the proportion at the latitude of the spill area (a function of the longshore distribution) and the inshore/offshore distribution. In all four scenarios (tanker spill/blowout at Port Heiden/Port Moller) the outer boundary of the grid is at a greater distance from shore than the reported extent of the juveniles distribution. Thus all passing juveniles would be within the offshore boundary of the spill area. The inshore boundaries of the spill areas encompass the shoreline with the exception of the tanker accident grid at Port Moller where the first 5.6 km representing 11.9% of the distribution of juveniles is excluded (Table 11). Taken together with the temporal distribution of juveniles in the latitude of the spill areas these data provide an estimate of the maximum proportion of the juveniles within each of the spill areas over a 10 d period (Table 12).

Adults

No simulation was run for the adults; historical data indicate that the return of the adults to Bristol Bay as monitored by the fisheries is well defined. From 1956 to 1976 80% of the run passed the fishery over a 12.9 day period (S.D. 1.58 days (Burgner 1980). The mean time of return over the same period was July 4

Table 10. Median and range of the time of the maximum percent of the outmigration present within the east and west boundaries of the simulation grids over a 10 d period. Data encompasses 1952 to 1982 as available.

River	Age	n	Date of maximum proportion in area			
			Port Heiden		Port Moller	
Kvichak	1.	24	6/25	(6/15-7/5)	7/15	(7/5-8/4)
	2.	24	6/15	(6/15-7/5)	7/5	(6/25-7/25)
Naknek	1.	21	6/25	(6/15-8/4)	7/15	(7/5-8/24)
	2.	21	6/25	(6/15-8/4)	7/15	(6/25-8/14)
Ugashik	1.	14	6/5	(6/5-6/25)	6/25	(6/15-7/15)
	2.	14	6/5	(6/5-6/25)	6/15	(6/15-7/5)
Wood	1.	13	7/5	(6/25-7/15)	7/25	(7/15-8/14)
	2.	13	7/5	(6/25-7/15)	7/25	(7/5-8/4)
Combined Rivers	1.	10	6/25	(6/25-7/25)	7/15	(7/15-8/14)
	2.	10	6/15	(6/15-6/25)	7/5	(6/25-7/15)

Table 11. Dimensions and boundaries of simulation grids and estimates of the percent of the migrants passing within the inshore and offshore boundaries of these grids.

Area	Spill scenario	Grid size (km)	Inshore boundary (km)	Offshore boundary (km)	Estimated % of population passing through grid	
					Juveniles	Adults
Pt. Heiden	Tanker Accident	64 x 68	0	74	100	36
	Blowout	100 x 100	0	95	100	51
Pt. Moller	Tanker Accident	64 x 68	6	94	88	55
	Blowout	100 x 100	0	117	100	71

Table 12. Mean of the maximum percent of the outmigration present within all boundaries of the simulation grids over a 10 d period. Data encompasses 1955 to 1982 as available.

River	Age	n	Maximum % of population passing through spill area over a 10 d period			
			Port Heiden		Port Moller	
			Tanker	Blowout	Tanker	Blowout
Kvichak	1.	24	65	87	57	87
	2.	24	59	77	52	77
Naknek	1.	21	40	60	35	60
	2.	21	37	56	33	56
Ugashik	1.	14	62	79	55	79
	2.	14	44	71	39	71
Wood	1.	13	49	68	43	68
	2.	13	41	62	36	62
Combined Rivers	1.	10	36	54	32	54
	2.	10	47	65	41	65

(extremes June 28-July 10). Mean migration rate of the returning adults is approximately 50 km/d, thus essentially all of the returning run would pass the latitude of the blowout grid (100 km long) over the 20 d simulation. Over the 10 d simulation for the tanker accident, adults spread out over 568 km (10 d at 50 km/d, plus the length of the grid) would pass the latitude of the 68 km long grid. Since 80% of the run is spread out over 12.9 d, or at 50 km/d, over 45 km, and assuming a normal distribution (SE = 508 km), 74% of the run would pass the latitude of the 68 km long grid in a 10 d period.

As with the juveniles the actual proportion of the adults passing through the grids will be determined by the offshore distribution in addition to the longshore distribution. Offshore boundaries of all four grids are inside the estimated offshore extent of the adults distribution and thus only a proportion of the total population will pass through the grids (Table 11). Maximum proportions of adults passing through the grids in a 10 d period (Table 13) are correspondingly reduced from the proportions passing the latitude of the grid.

Avoidance of Oil

To illustrate the effect of the avoidance algorithm on the migrations of salmon in the presence of oil and their resulting contamination, I simulated the movement of a concise distribution of juveniles through a stationary oil spill. Movements were simulated for migration with avoidance and direct migration. The stationary oil spill is that simulated for 48 hours after the tanker spill of 240,000 bbl of fuel oil at Port Heiden.

Juveniles migrating at an effective rate of 0.9 BL/sec towards the lower left corner of the grid come into contact with the oil spill at 48 hours. At this time the distributions of the juveniles under the two migration conditions are similar (Fig. 13a). At 72 hrs the juveniles able to avoid start moving around the spill

Table 13. Maximum percentage of adults passing within all four boundaries of the oil simulation grids over a 10 d period. Data encompasses 1956 to 1976.

Area	Spill scenario	Estimated % of adult population passing through grid in a 10 d period
Pt. Heiden	Tanker accident	27
	Blowout	51
Pt. Moller	Tanker accident	41
	Blowout	71

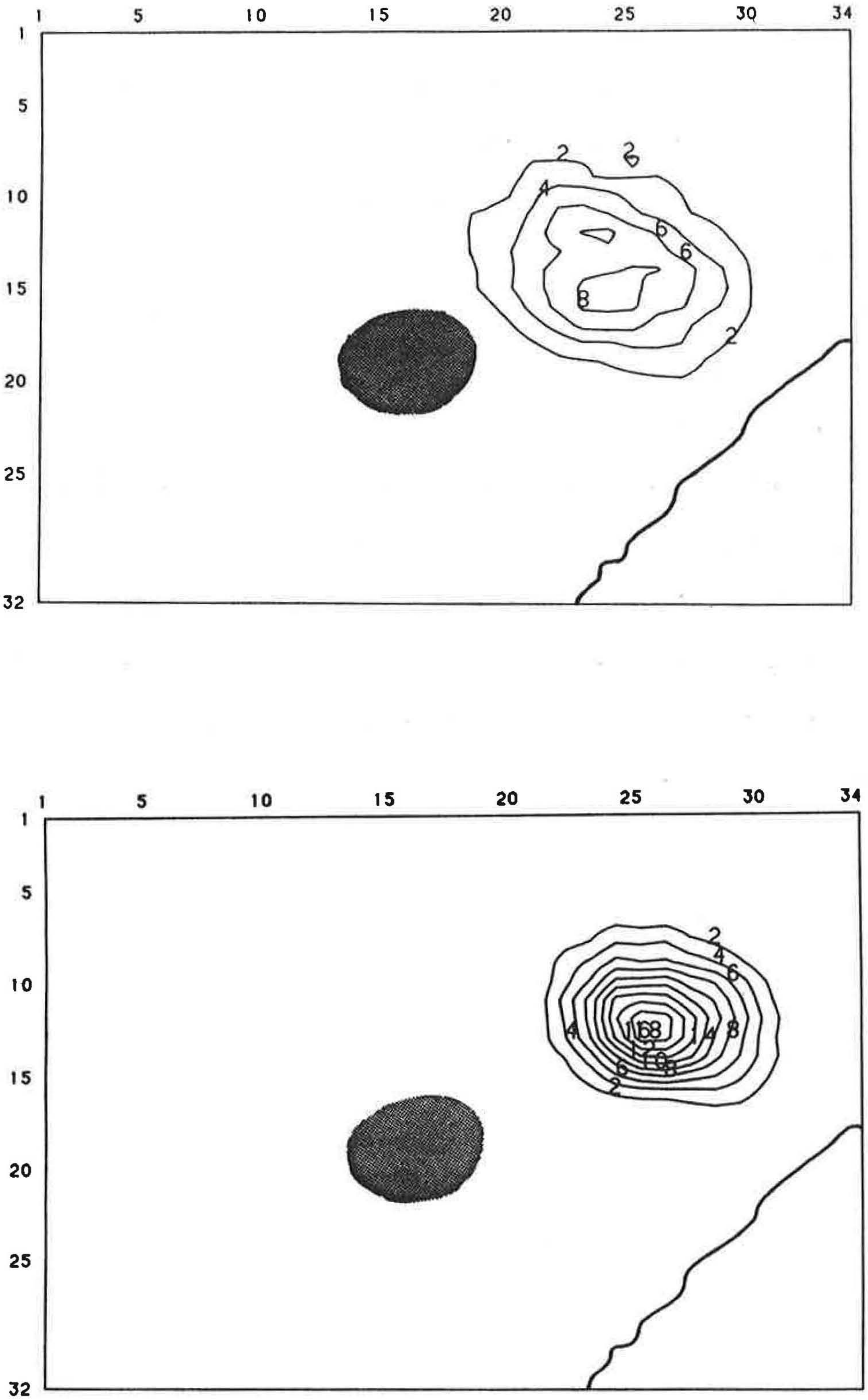


Figure 13.--Simulation of the movement of a concise distribution of juvenile sockeye salmon towards and through a stationary spill of fuel oil in Bristol Bay under the assumptions of avoidance (top) and direct migration (bottom).

a) 48 hrs.

and by 96 hrs have divided into two separate distributions (Figs. 13b and 13c). The distribution of the juveniles migrating directly is unchanged at this point (Figs. 13b and 13c). After 120 hrs the distribution of the directly migrating juveniles is reduced as a result of oil-induced mortalities, and at 144 hrs these remaining juveniles migrate from the grid (Fig. 13d). The juveniles migrating with avoidance continue with their more diffuse distribution indicating both mortalities and individually variable migration pathways, until they too migrate out of the grid at 144 hrs (Fig. 13e).

In this simulation mortalities were far higher when no avoidance occurred (69.8%) than with avoidance (14.8%). Tainting of the surviving population was also higher for the direct migrants (41.4% tainted above 0.6 ppm, compared to 21.3%), although the mean level of hydrocarbon in the muscle was lower (0.79 ± 0.14 ppm, compared to 1.28 ± 0.24 ppm) indicating that the statistical distribution of tainting levels was non-normal.

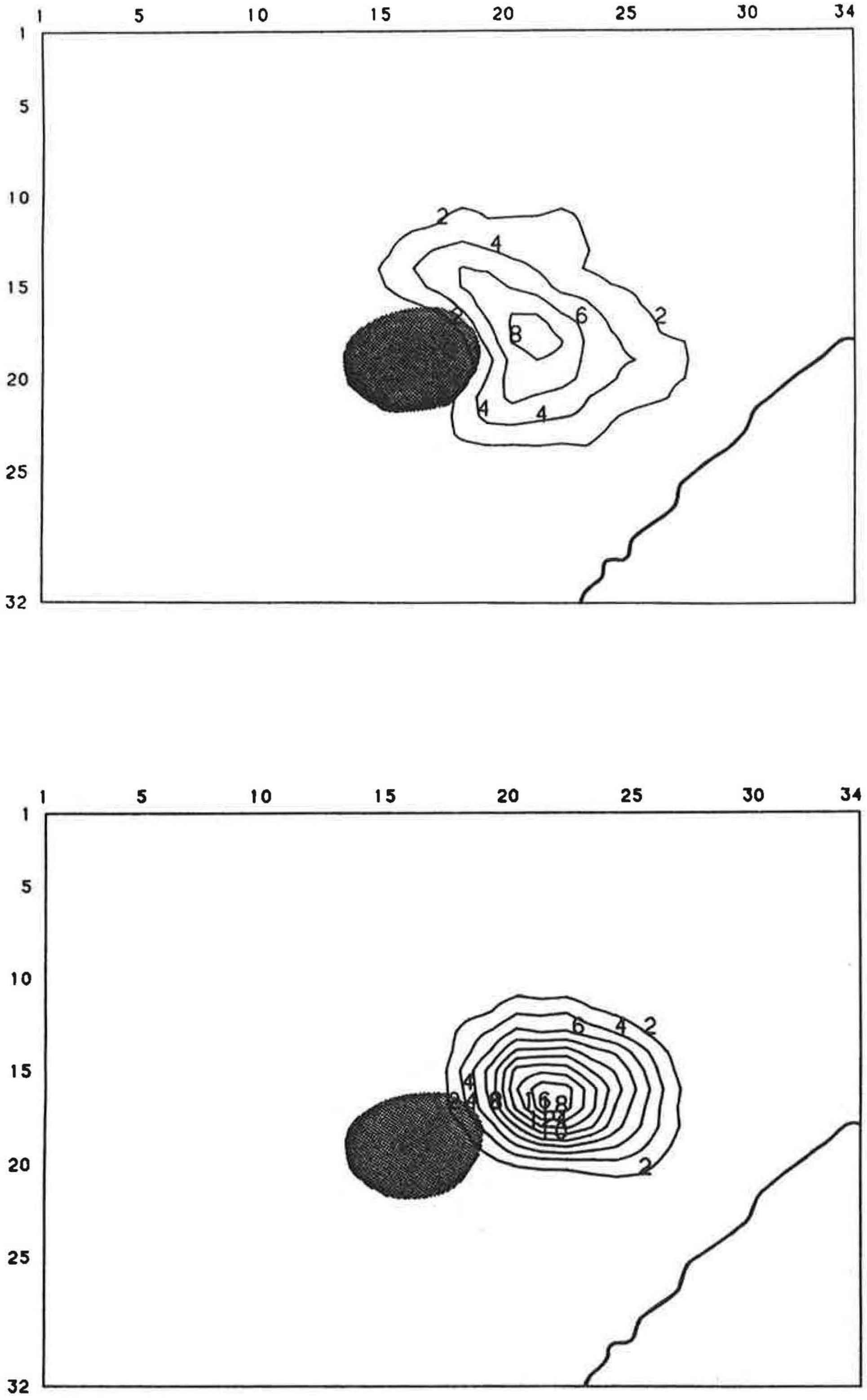


Figure 13b: 72 hrs.

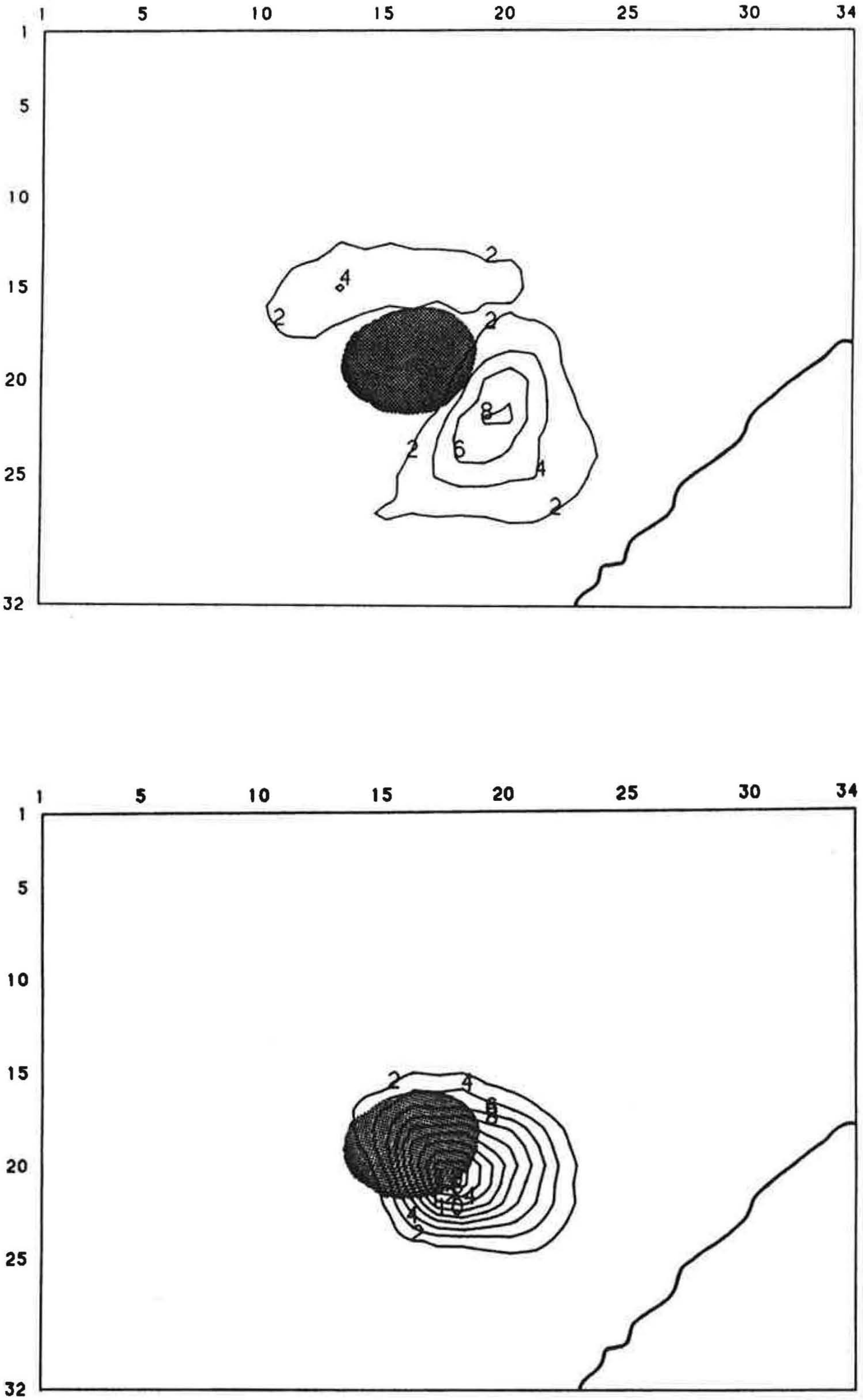
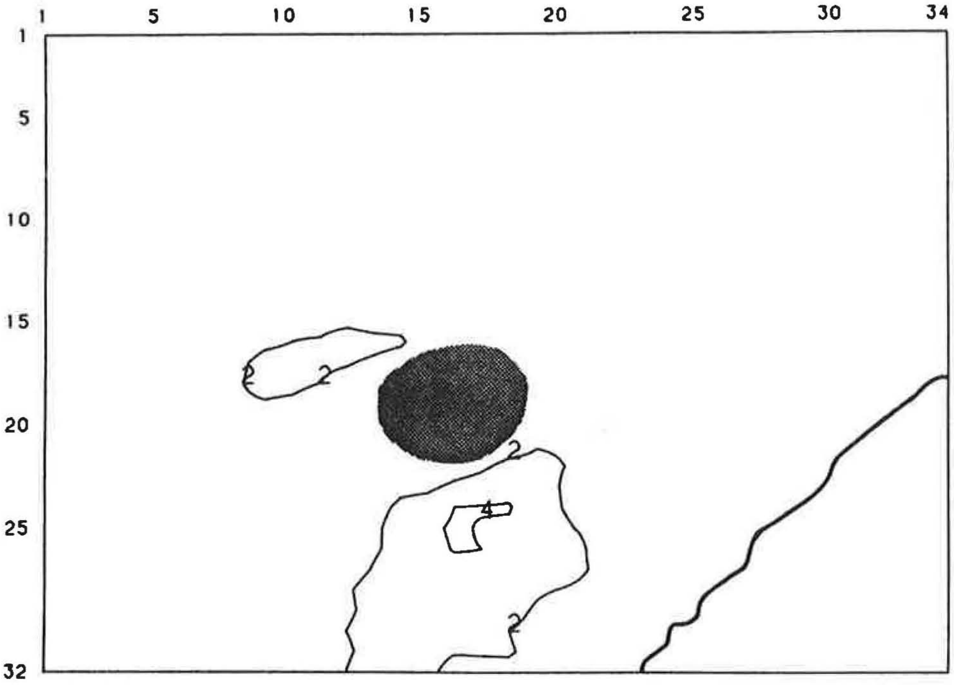


Figure 13c: 96 hrs.



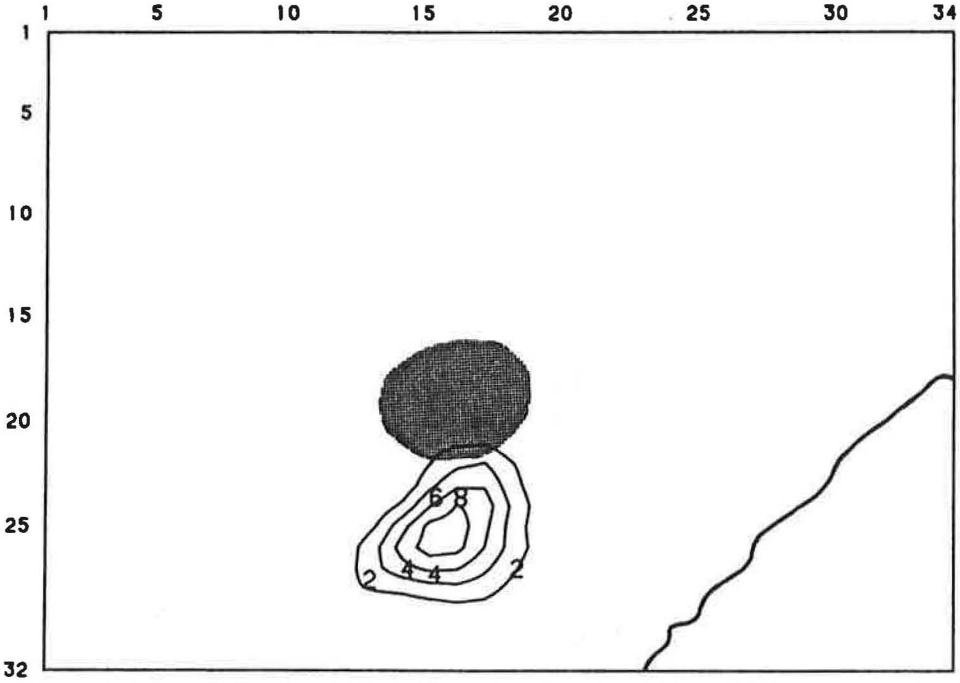


Figure 13d: 120 hrs.

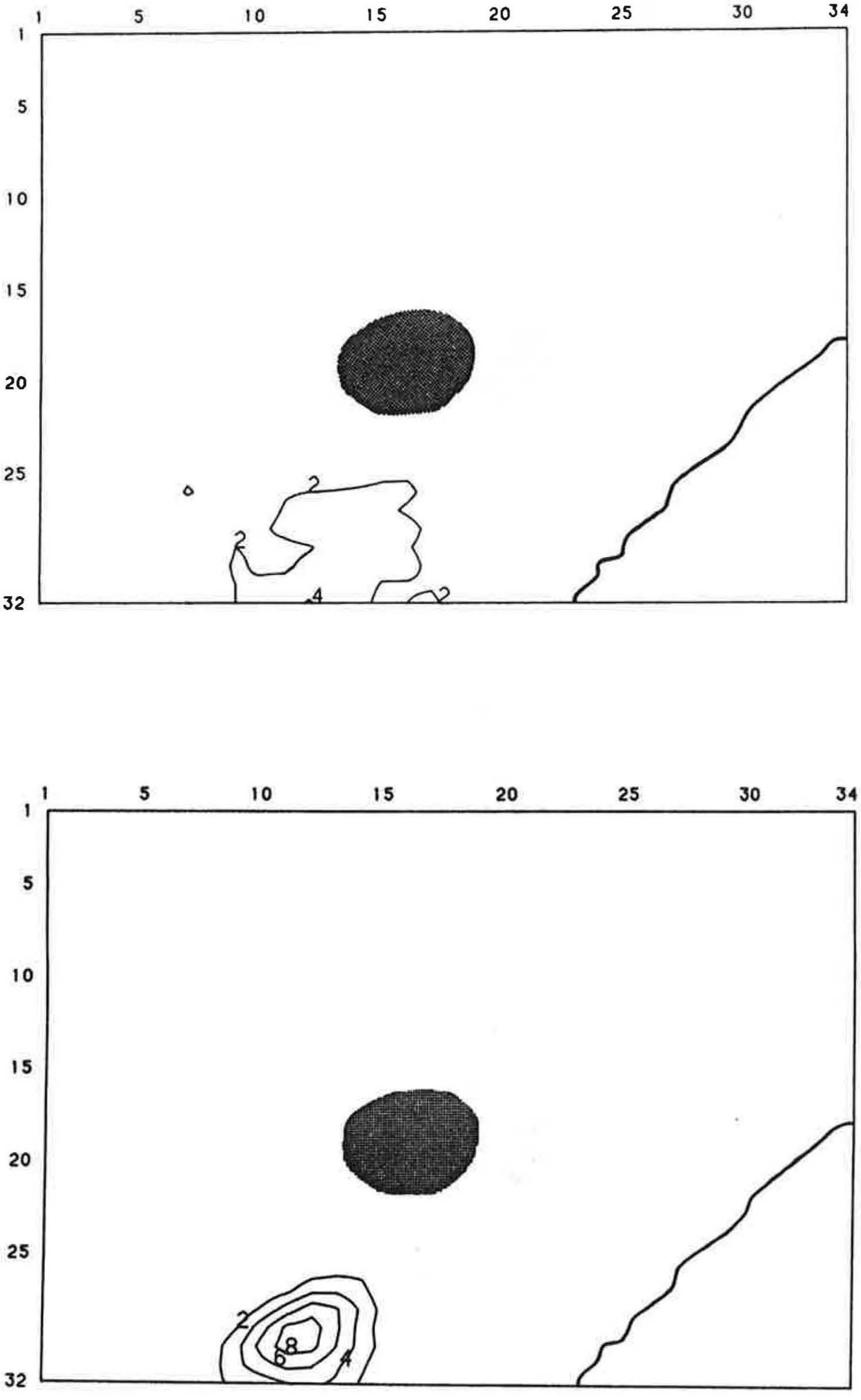


Figure 13e: 144 hrs.

OIL SPILL SCENARIOS

Juveniles

Four oil spill scenarios were investigated: a spill of 240,000 bbl of No. 2 fuel oil off Port Heiden and off Port Moller (data available for 240 hrs), and a blowout of crude oil at a rate of 20,000 bbl/day at the same two locations (data available for first 480 hrs).

Intense avoidance of the spills of fuel oil by the juveniles occurred and the changes in their abundance distribution over time off Port Heiden illustrate this (Figs. 14a - 14c). Avoidance is indicated by the increased abundance around the perimeter of the contaminated area; avoidance and mortality are illustrated by the lack of any fish within most of the contaminated area. As the juveniles first avoid the spill they are forced around it, either closer to, or further from, shore, but towards the end of the 240 hr time series those juveniles forced inshore are funnelled into an ever decreasing area and the chances of completely avoiding contamination are reduced. An important point to note is that the areal coverage of the fuel oil at concentrations above 0.45 ppm (the concentration which caused 100% mortality of juvenile sockeye salmon held in seawater within 24 hrs) increases steadily throughout this scenario (Fig. 15). No allowance has been made in the oil spill scenarios for the changing composition of the oil as the low molecular weight (and more toxic) compounds boil off; Maurin (1981) estimates that 40% of the oil (primarily the lighter fractions) from the Amoco Cadiz evaporated within 48 hrs of the spill, suggesting that this oversight in the analysis will cause overestimates of the effects of the oil spill given that other parameters are estimated within reasonable bounds. This overestimation will be partly offset in the tainting algorithm where the parameters for uptake and depuration were calculated from an experimental design which included weathering of the oil.

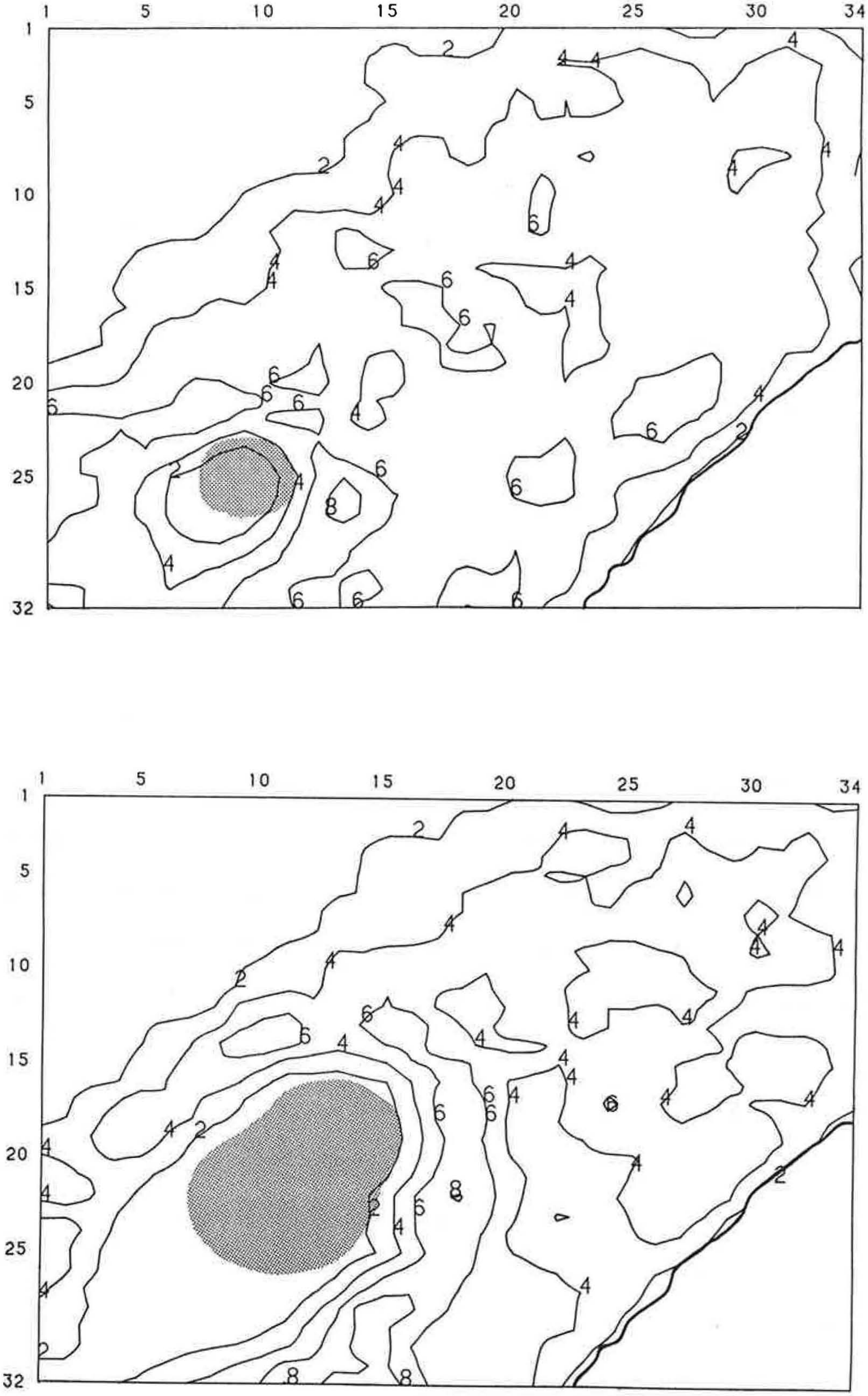


Figure 14.--Simulation of migration of juvenile sockeye salmon and their avoidance of a spill of 240,000 bbl of No. 2 diesel oil off of Port Heiden, Bristol Bay. (Shaded area depicts concentrations of oil greater than 1 ppm.) a) top: 48 hrs from start of spill; b) bottom: 120 hrs from start of spill.

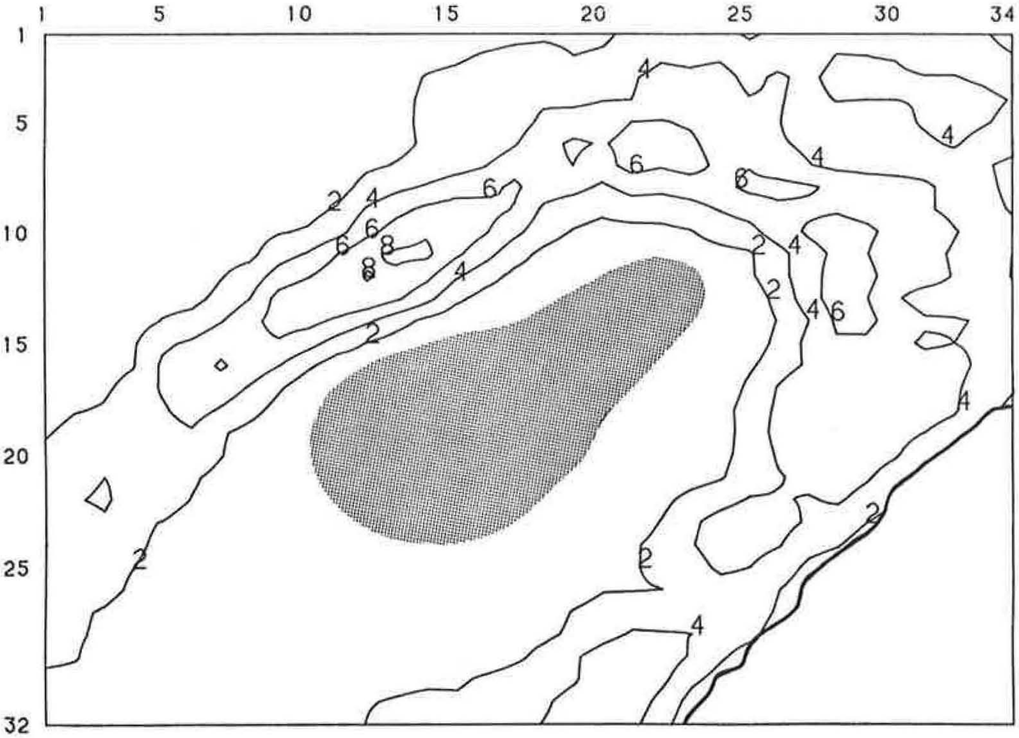


Figure 14c: 240 hrs from start of spill.

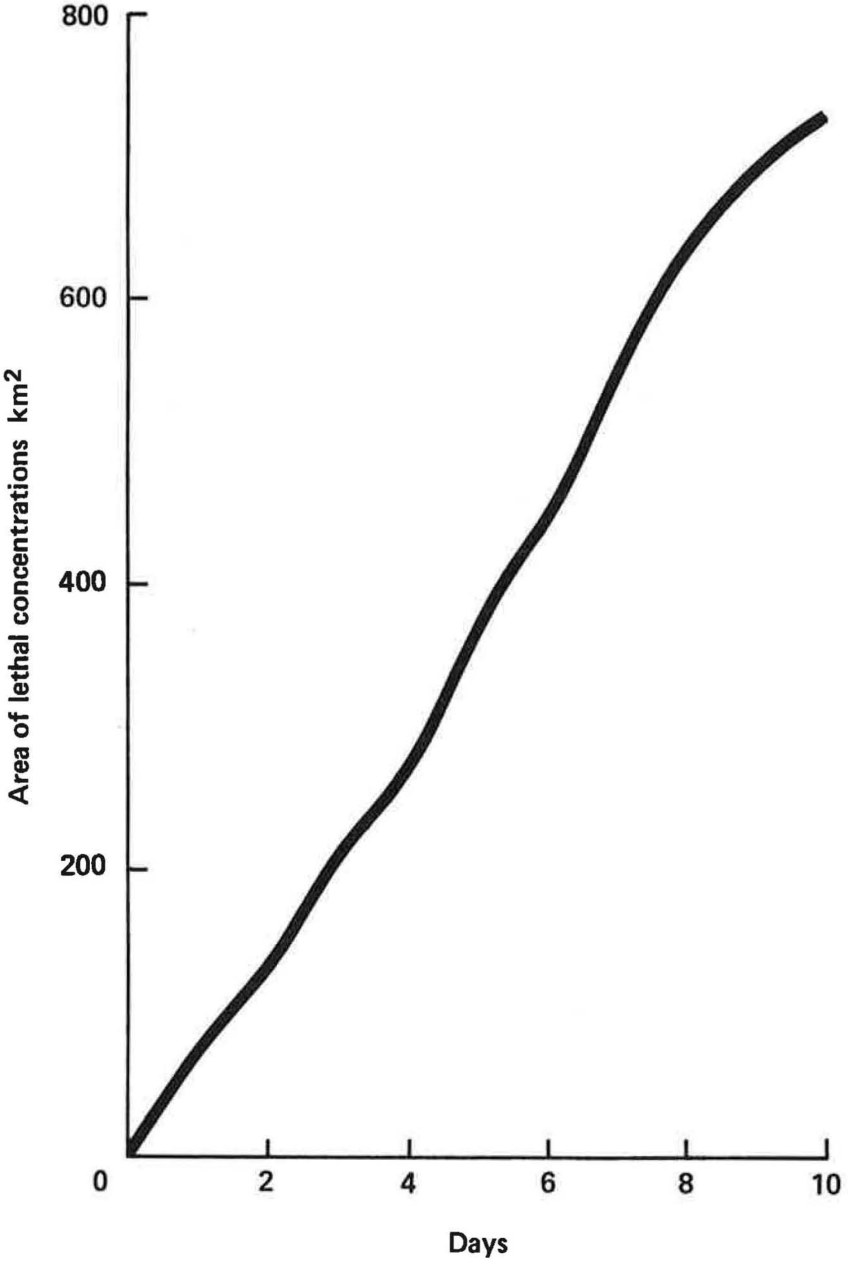


Figure 15.--Areal coverage of concentrations of fuel oil lethal to sockeye salmon after 24 hrs (>0.45 ppm) resulting from a simulated spill of 240,000 bbl of No. 2 diesel oil off Port Heiden or Port Moller in Bristol Bay. (Data from Rand Corporation.)

Areal coverage of the fuel oil spills off of Port Heiden and off of Port Moller are identical over time and thus any differences in their effects on the migrating juveniles will be a result of different migration patterns in the vicinity of the spill in the two areas, principally as a result of the changing probability of occurrence with distance from shore (Fig. 10). In these simulations between 15 and 35% of the juveniles migrating through the grid at Port Heiden and between 7 and 14% at Port Moller died from toxic levels of oil (Table 14). The ranges result from the assumption of either a direct migration (higher value) or of migration with avoidance (lower level). Effects are greater at Port Heiden where the spill is located closer to shore. From the same simulations tainting of the surviving population ranged from 11 to 18% at Port Heiden and from 3 to 5% at Port Moller (Table 15).

In contrast to the strong avoidance of the fuel oil spill, little avoidance of the crude oil blowout was found (Figs. 16a - 16d). This results from the much lower concentrations of the crude oil when compared with those from the fuel oil in the preceding scenarios, and also the greater toxicity of fuel oil compared to an equal concentration of crude oil. Thus at no point during this 20 d scenario does the concentration of crude oil reach the level found to cause 100% mortalities within 24 hours (2.5 ppm); in fact concentrations do not exceed 0.3 ppm. Consequently the simulated mortalities range from only 0.4 to 1.2% and tainting occurs in only 0.1% of the remaining population.

Adults

Adult salmon move through the simulation grid and oil spill more rapidly than the juveniles and were therefore expected to be less contaminated by the oil. Avoidance of the fuel oil spill does occur (Figs. 17a - 17 c) with the adults

Table 14. Simulated percent mortalities of sockeye salmon migrating through the oil spill grids either directly or with avoidance of the spill.

Spill scenario	Run time (hrs)	Percent mortalities	
		Direct migration	Migration with avoidance
Juveniles			
Port Heiden			
Tanker spill/fuel oil	240	35.5	15.4
Blowout/crude oil	480	0.4	0.5
Port Moller			
Tanker spill/fuel oil	240	14.2	7.0
Blowout/crude oil	480	1.2	0.5
Adults			
Port Heiden			
Tanker spill/fuel oil	240	17.6	3.2
Blowout/crude oil	480	0.2	0.1
Port Moller			
Tanker spill/fuel oil	240	11.6	2.1
Blowout/crude oil	480	0.2	0.1

Table 15. Simulated percent taintings of sockeye salmon migrating through the oil spill grids either directly or with avoidance of the spill.

Spill scenario	Run time (hrs)	Percent tainted above 0.6 ppm	
		Direct migration	Migration with avoidance
Juveniles			
Port Heiden			
Tanker spill/fuel oil	240	17.7	10.6
Blowout/crude oil	480	0.0	0.0
Port Moller			
Tanker spill/fuel oil	240	5.2	3.1
Blowout/crude oil	480	0.1	0.0
Adults			
Port Heiden			
Tanker spill/fuel oil	240	7.1	3.1
Blowout/crude oil	480	0.0	0.0
Port Moller			
Tanker spill/fuel oil	240	5.0	2.6
Blowout/crude oil	480	0.0	0.0

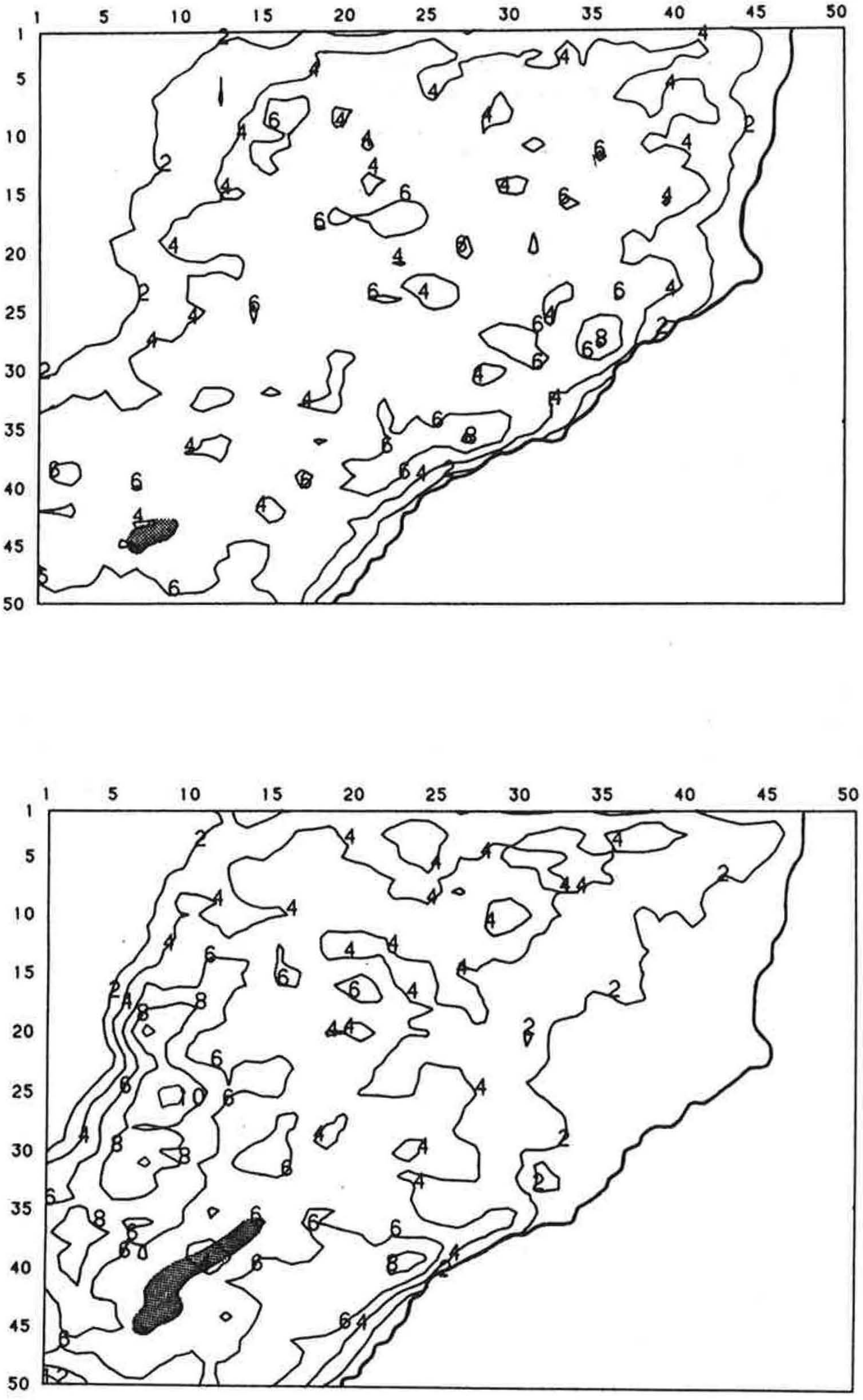


Figure 16.--Simulation of migration of juvenile sockeye salmon and their avoidance of a blowout of 20,000 bbl/day off of Port Heiden, Bristol Bay. (Shaded area depicts concentrations of oil greater than 0.1 ppm.) a) top: 48 hrs from start of spill; b) bottom: 120 hrs from start of spill.

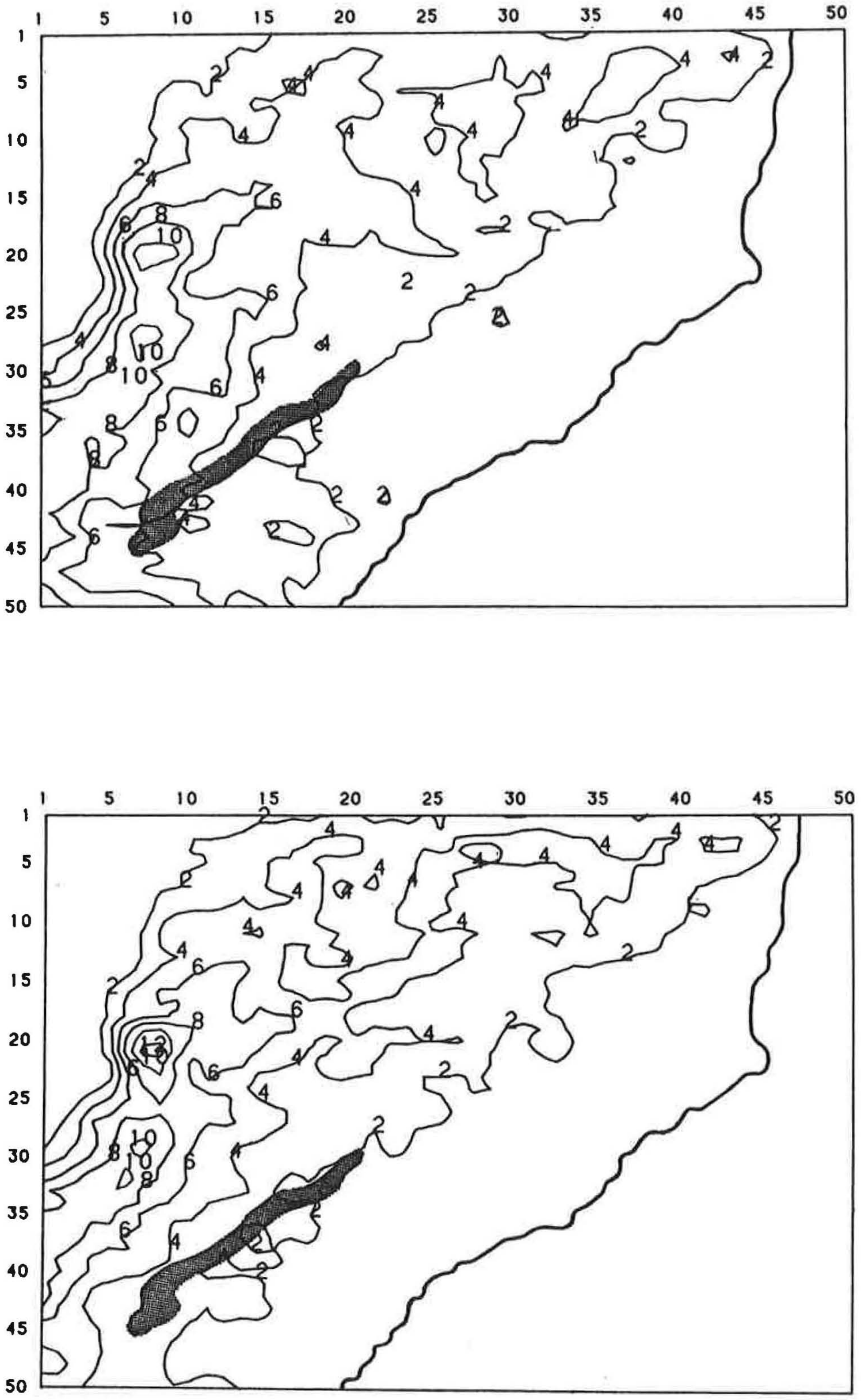


Figure 16 c) top: 240 hrs from start of spill.
d) bottom: 480 hrs from start of spill.

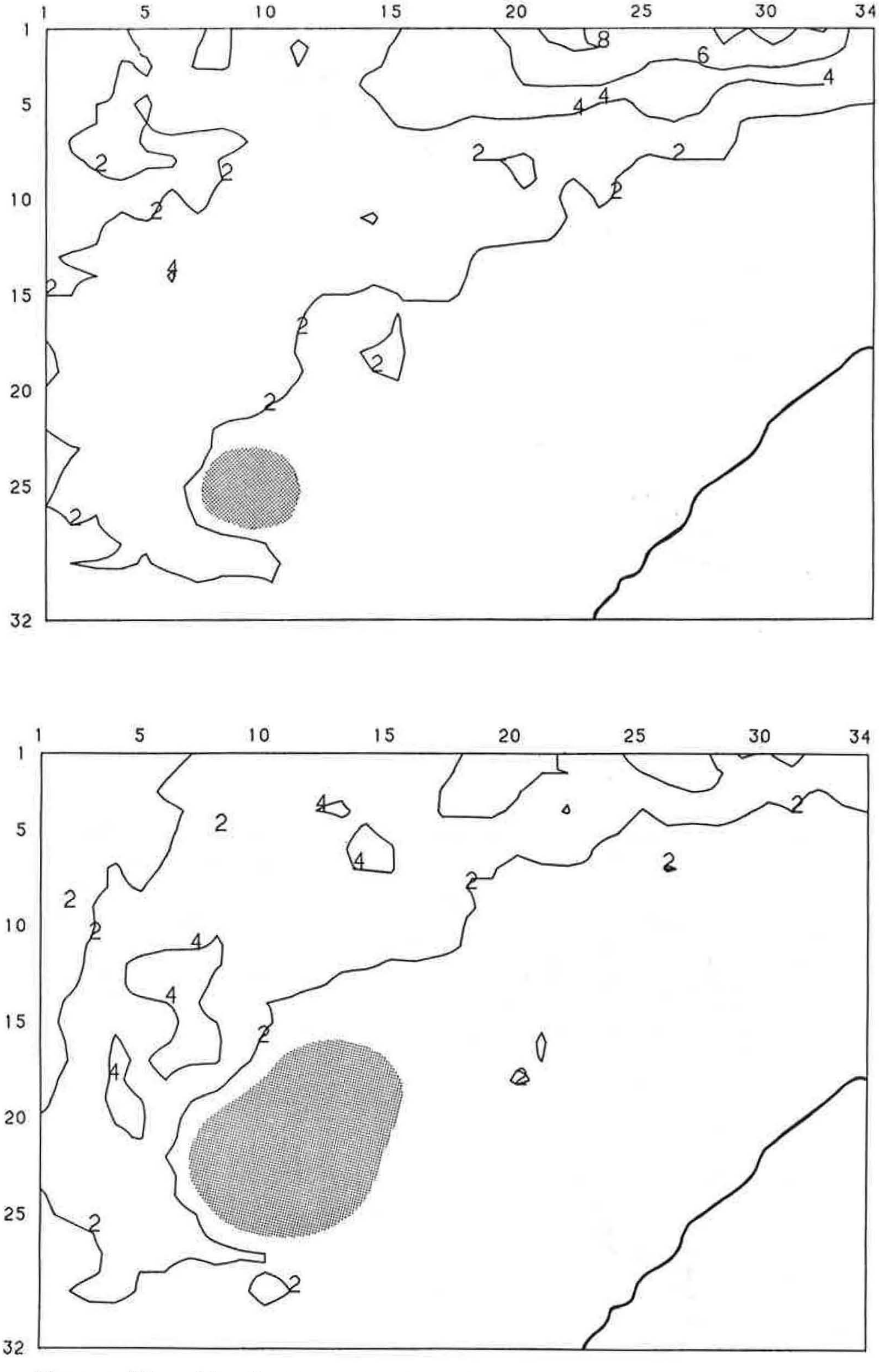


Figure 17.--Simulation of migration of adult sockeye salmon and their avoidance of a spill of 240,000 bbl of No. 2 diesel oil off of Port Heiden, Bristol Bay. (Shaded area depicts concentrations of oil greater than 1.0 ppm). a) top: 48 hrs from start of spill; b) bottom: 120 hrs from start of spill.

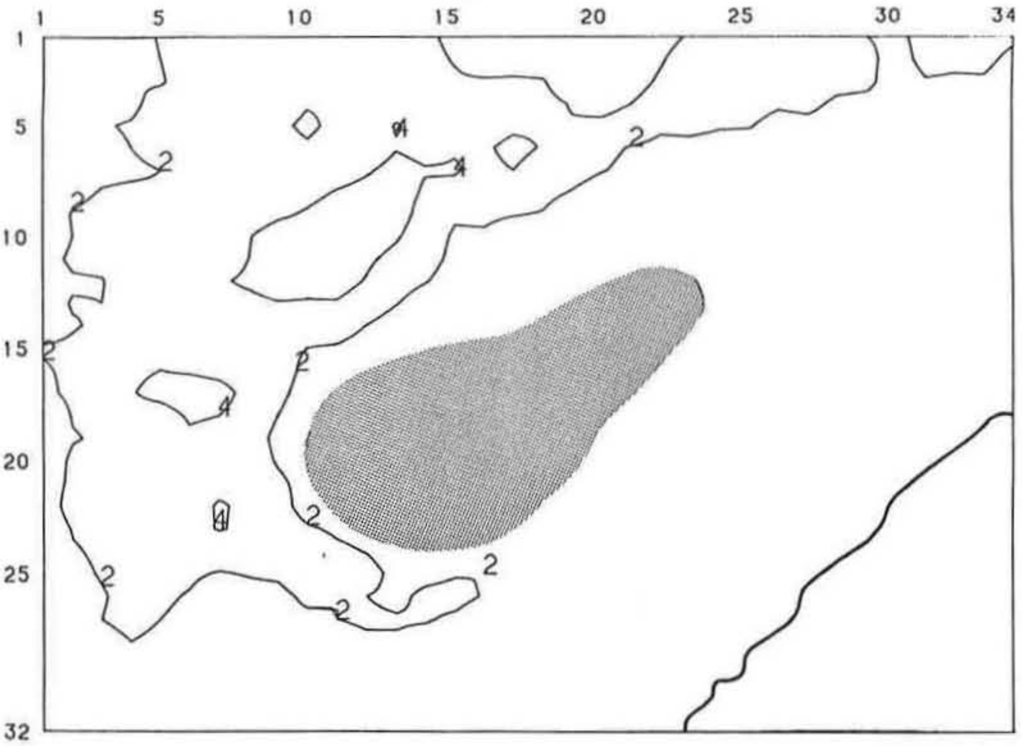


Figure 17c: 240 hrs from start of spill.

mostly moving around the offshore perimeter of the spill; however, their migration is so rapid that little (if any) concentration of numbers occurs. As with the juveniles, avoidance of the crude oil blowout was virtually undetectable (Figs. 18a-18c). Mortalities for the adults were highest for the fuel oil spill scenario at Port Heiden (3 to 18%) and lowest for the two crude oil blowout scenarios (0.1 to 0.2%) (Table 14). Mortalities were reduced when it was assumed that avoidance of the contamination could occur. The percent of the surviving pdpulation that was tainted was lower than that for the juveniles, ranging from 3 to 6% for the Port Heiden fuel oil spill to 0% for the two crude oil blowouts (Table 15). These estimates of tainting are based on the assumption that a level of 0.6 ppm hydrocarbons in salmon flesh is the threshold at which tainting is detected. Percentages of the adults that were tainted would be different if other tainting thresholds were assumed (Fig. 19).

EXTRAPOLATION TO WHOLE POPULATION

As discussed in a previous section and summarized in Tables 11 and 12, only a proportion of the total migration of juveniles or adults passing through Bristol Bay would pass through the grids used for the oil spill scenarios. Thus the percent mortalities and percents tainted need to be reduced if they are to apply to the whole population. If it is assumed that by the end of the 10 day tanker spill scenario the composition of the fuel oil has changed (through evaporation) to such that negligible mortalities or tainting would continue to occur, then this reduction provides the best estimates from this simulation for the maximum effects of the tanker spill scenarios on the sockeye salmon migrating through Bristol Bay. The scenarios describing the effects of a blowout of crude oil will not be discussed further since simulated mortalities were in general less than one percent and the proportion tainted was not greater than one part per thousand.

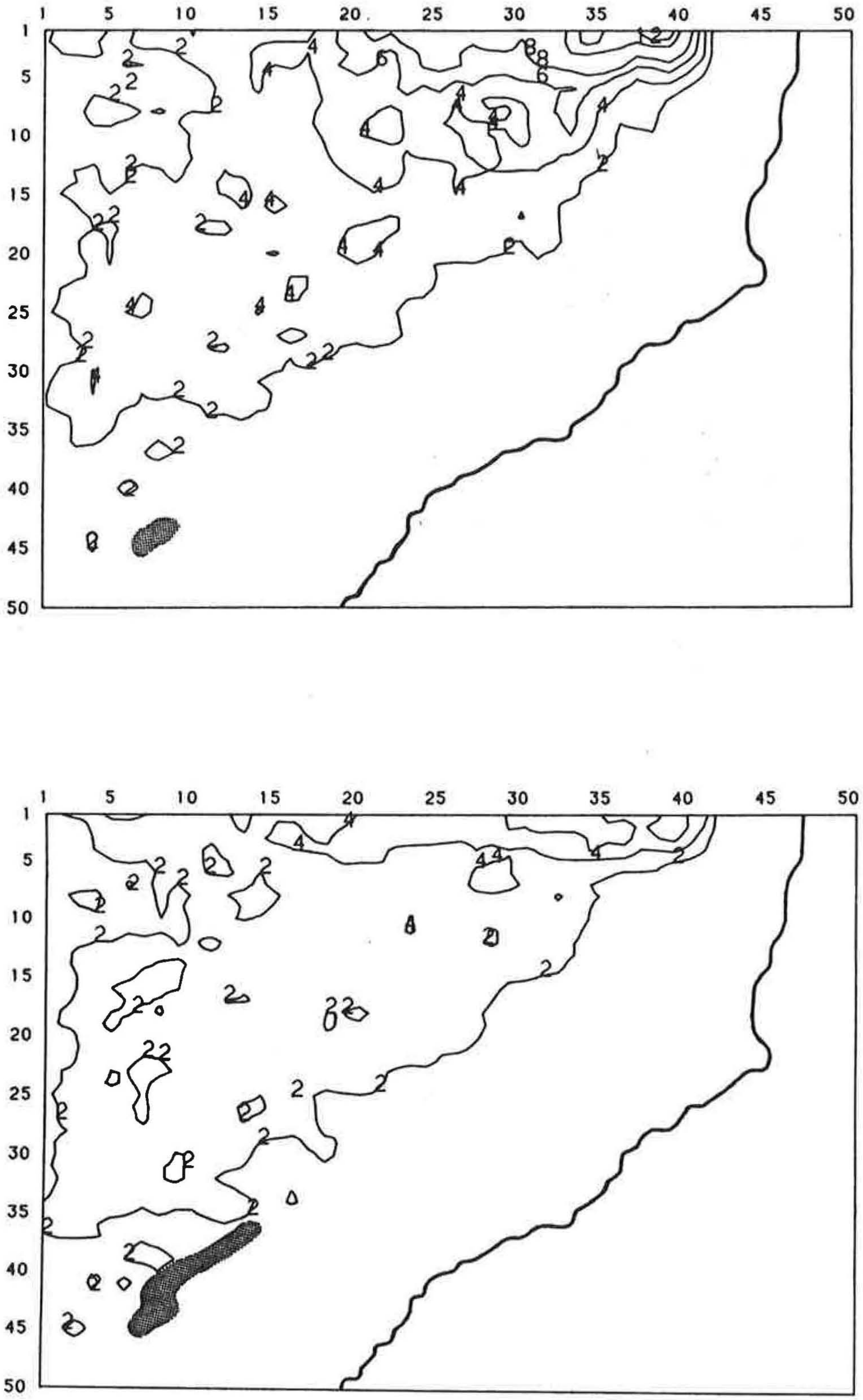


Figure 18.--Simulation of migration of adult sockeye salmon and their avoidance of a blowout of 20,000 bbl/d off of Port Heiden, Bristol Bay. (Shaded area depicts concentrations greater than 0.1 ppm). a) top: 24 hrs from start of spill; b) bottom: 120 hrs from start of spill.

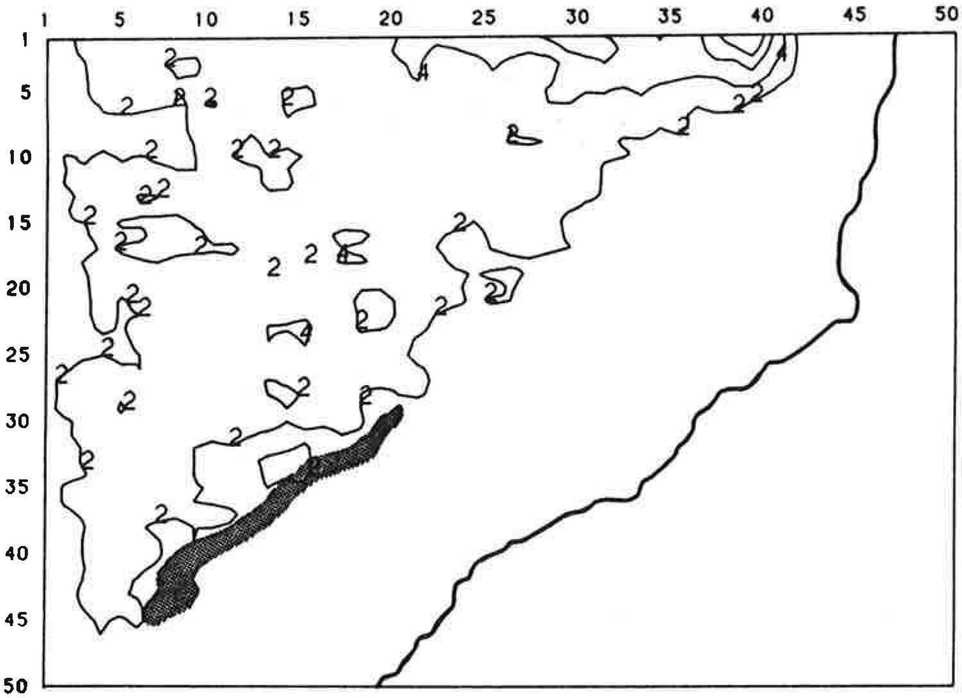


Figure 18c: 480 hrs from start of spill.

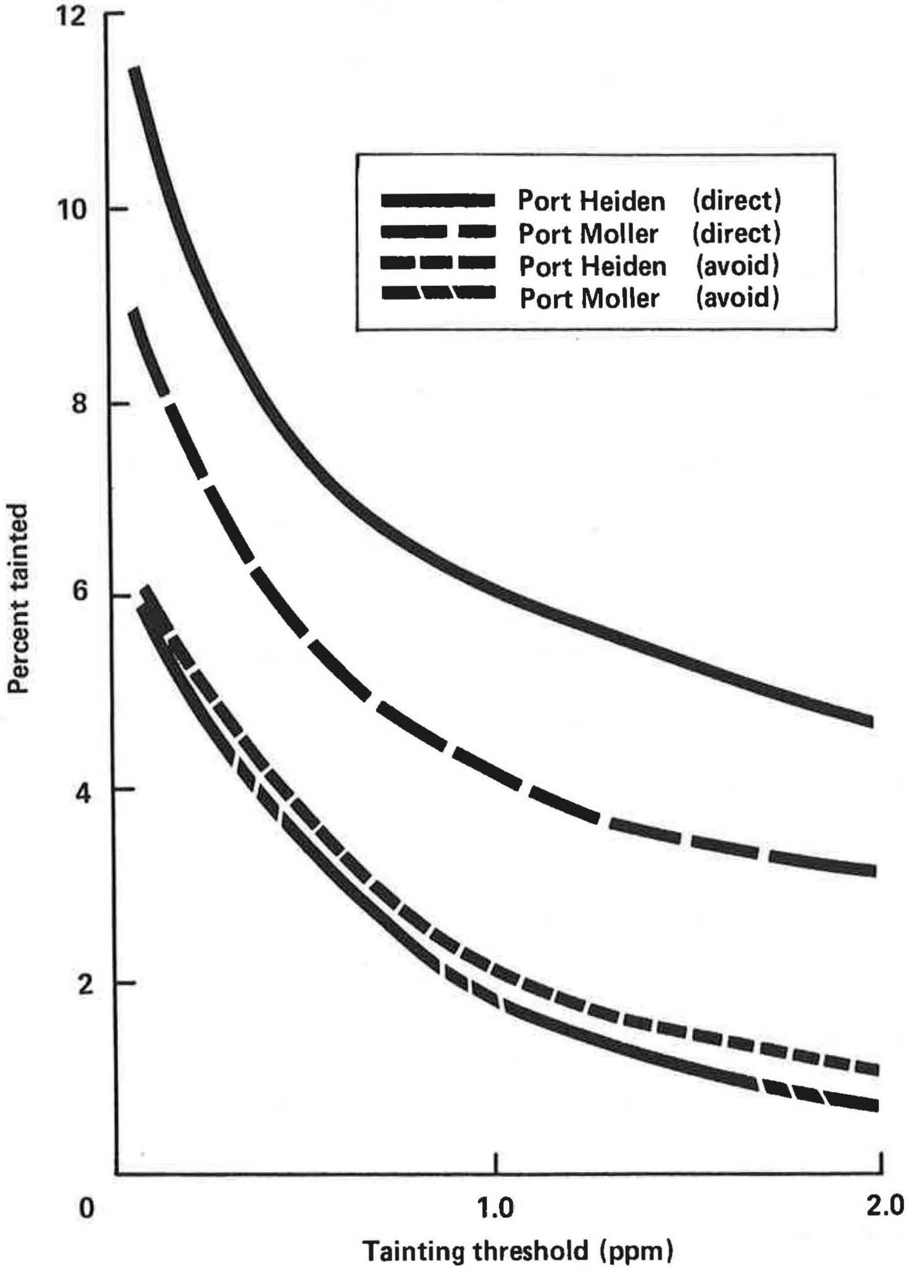


Figure 19.--Percentages of adult sockeye salmon migrating through the fuel oil spill grids that would be tainted at different tainting thresholds and under the assumptions of avoidance or direct migration.

Mortalities for the populations of juveniles migrating out of Bristol Bay (4 major rivers combined) were simulated at 3 to 13% for the one winter fish and 3 to 10% for the two winter fish (Table 16). These estimates do not include any mortalities which sublethal stress might elicit subsequent to the 10 d scenarios. For the same fish the percent tainted above 0.6 ppm ranged from 2 to 6% for the one winter fish and from 1 to 5% for the two winter fish.

Mortalities and tainting were less for the adult population because of their greater rate of migration through the area of the spills. Mortalities were simulated to range from 1 to 5% and tainting from 1 to 2%.

An important consideration for the adults is to what extent tainted fish might arrive on the fishing grounds. During the passage of fish from the area of the oil spill to the fishing areas depuration will occur; the degree of depuration depending on the distance (or time) travelled. Adults tainted in the Port Heiden spill area would arrive at the fishing grounds of the Ugashik River the same day, but would (at 50 km/day) take 2 - 3 days to reach the Kvichak-Naknek fishing area. Those adults tainted off of Port Moller would reach the Ugashik fishing area in about 3 days but take 5 days to reach that off the Kvichak-Naknek rivers. To investigate the degree of tainting remaining in the migrants through the grid over time the simulation was rerun allowing one day's migrants through the grid at a time and following their depuration over the subsequent 5 days. Results from these simulations are tabulated in Table 17, and those from the simulation of adults migrating directly through the Port Heiden spill are graphed in Fig. 20.

Tainting drops off rapidly once the adults enter uncontaminated water, with zero tainting being reached on days 4 to 6, or 3 to 5 days after the fish have

Table 16. Percent mortalities and tainting from tanker spill scenarios extrapolated to whole population.

Age group	Location of spill	Reduction factor	Percent mortalities		Percent tainted	
			Direct	Avoid	Direct	Avoid
Juveniles 1. (combined rivers)	Pt. Heiden	0.36	12.8	5.5	6.4	3.8
	Pt. Moller	0.47	6.7	3.3	2.4	1.5
Juveniles 2. (combined rivers)	Pt. Heiden	0.28	9.9	4.3	5.0	3.0
	Pt. Moller	0.36	5.1	2.5	1.9	1.1
Adults	Pt. Heiden	0.27	4.8	0.9	1.9	0.8
	Pt. Moller	0.41	4.8	0.9	2.1	1.1

Table 17. Percentages of adults passing through the spill simulation areas under two migration assumptions that are tainted at the time and in the following 5 days.

Day of spill	Percent tainted above 0.6 ppm[1]					
	Day of passage	Day 2	Day 3	Day 4	Day 5	Day 6
Pt. Heiden Direct						
1	12.3	10.4	9.6	7.2	0.5	0.0
2	15.4	13.9	11.1	8.6	0.4	0.0
3	19.6	15.6	14.4	10.8	0.7	0.0
4	23.0	21.0	18.0	12.9	0.7	0.0
5	26.8	24.1	20.5	13.9	0.0	0.0
6	27.3	23.0	19.6	14.1	0.0	0.0
7	27.4	23.3	19.5	13.5	0.0	0.0
8	24.7	20.8	18.0	11.4	0.0	0.0
9	25.3	22.2	18.8	11.9	0.0	0.0
10	30.6	26.3	20.8	13.1	0.0	0.0
Pt. Moller Direct						
1	8.3	7.1	6.6	5.0	0.4	0.0
2	10.1	9.1	7.2	5.6	0.2	0.0
3	13.1	10.4	9.5	7.0	0.5	0.0
4	16.0	14.5	12.1	8.3	0.5	0.0
5	18.7	16.8	14.1	9.2	0.0	0.0
6	19.3	16.2	13.7	9.2	0.0	0.0
7	19.6	16.6	13.8	8.9	0.0	0.0
8	18.8	15.9	13.0	7.8	0.0	0.0
9	19.6	17.2	13.9	8.2	0.0	0.0
10	22.7	19.1	14.9	8.8	0.0	0.0
Pt. Heiden Avoid						
1	2.9	2.2	1.1	0.0	0.0	0.0
2	3.0	1.7	0.6	0.0	0.0	0.0
3	6.0	3.3	0.8	0.0	0.0	0.0
4	6.5	3.4	0.7	0.0	0.0	0.0
5	10.7	5.2	1.0	0.0	0.0	0.0
6	10.9	5.4	0.8	0.0	0.0	0.0
7	12.2	6.7	1.2	0.0	0.0	0.0
8	9.7	5.9	1.2	0.0	0.0	0.0
9	11.0	6.5	1.0	0.0	0.0	0.0
10	13.9	8.5	0.7	0.0	0.0	0.0
Pt. Moller Avoid						
1	2.1	1.5	0.7	0.0	0.0	0.0
2	2.2	1.2	0.4	0.0	0.0	0.0
3	3.8	1.9	0.4	0.0	0.0	0.0
4	5.7	3.3	0.6	0.0	0.0	0.0
5	8.6	4.4	0.7	0.0	0.0	0.0
6	8.8	4.4	0.8	0.0	0.0	0.0
7	10.0	5.5	1.0	0.0	0.0	0.0
8	10.0	5.1	0.7	0.0	0.0	0.0
9	12.1	6.4	0.8	0.0	0.0	0.0
10	13.8	6.9	0.6	0.0	0.0	0.0

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[1] For extrapolation to whole population use 0.27 of reported values for spill at Port Heiden and 0.41 for a spill at Port Moller.

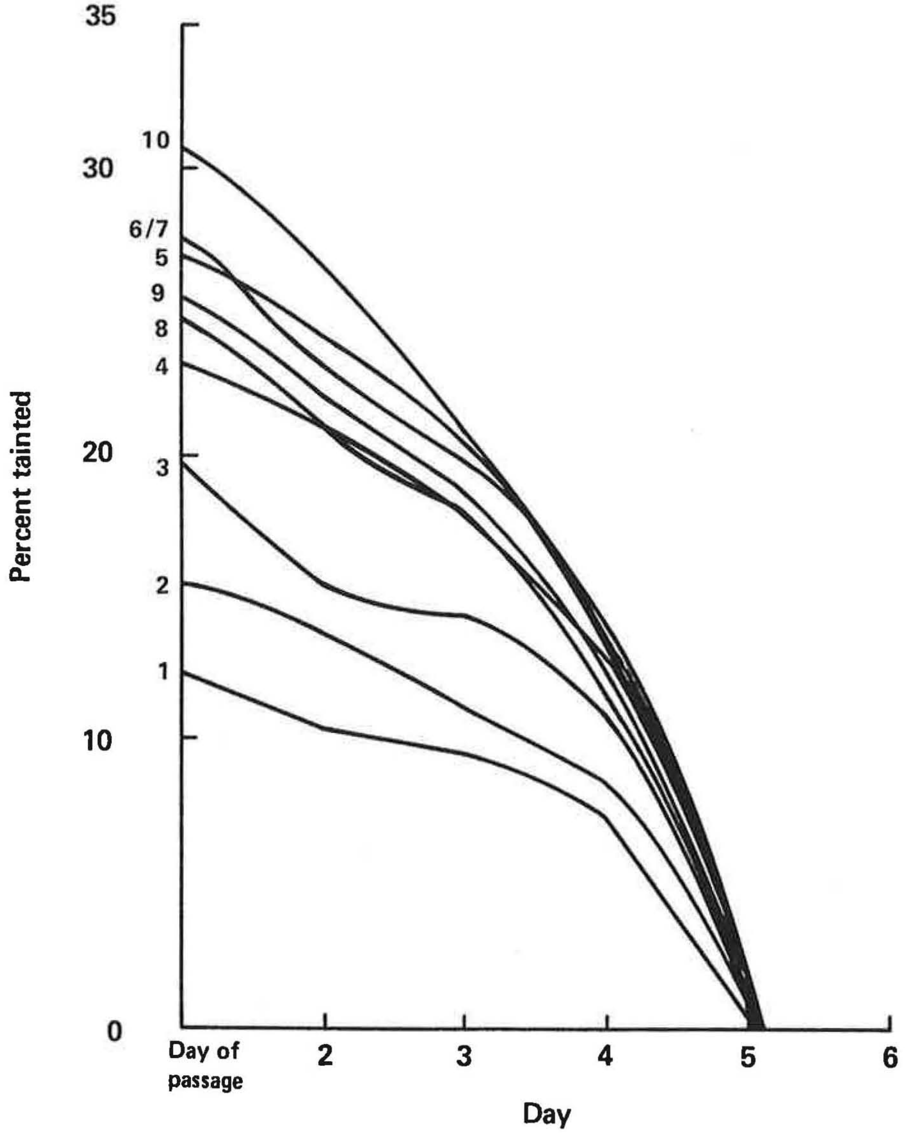

Figure 20.--Percent of adult sockeye salmon tainted after passing through the fuel oil spill grid off of Port Heiden from days 1-10 of the spill, and the percentages remaining tainted over the next 5 days.

left the contaminated area. Percents tainted decrease at an increasing rate over time (Fig. 20). This at first appears contradictory to the exponential decrease found experimentally and reproduced in this simulation as shown in Fig. 11, however this is due to threshold for tainting being set at 0.6 ppm not at 0 ppm.

The second major factor affecting the proportions of adults reaching the fishing areas that are tainted is the proportion of the run from individual rivers that actually passes within the boundaries of the oil spill simulation grid. Earlier I proposed factors by which to reduce the simulation results to account for this; however, these factors assumed no differential offshore distribution of adults from the four rivers. Data on the returns of 27 adults tagged and released at sites between Port Moller and Port Heiden (collated by Straty 1975) were presented earlier in "Stock Characteristic - Adult Migrations", and they suggest ($0.10 > p > 0.05$) that the adults returning to different rivers are to be found at different distances from shore. The proportion of fish tagged at each location and returning to each river was calculated, normalized over river and plotted in Fig. 21; correlation coefficients for the regressions ranged from 0.76 to 0.90 ($n=4$) and as expected with such small sample sizes were not statistically significant ($p > 0.10$). At the risk of imparting too great a significance to too few data I estimated from Fig. 21 the proportions of the adults from each river that would have passed within the boundaries of the oil spill simulation grids and obtained the following reduction factors:

	Ugashik	Egegik	Nak-Kvi	Ushagak
Port Heiden	1.0	0.8	0.3	0.3
Port Moller	1.0	0.9	0.4	0.4

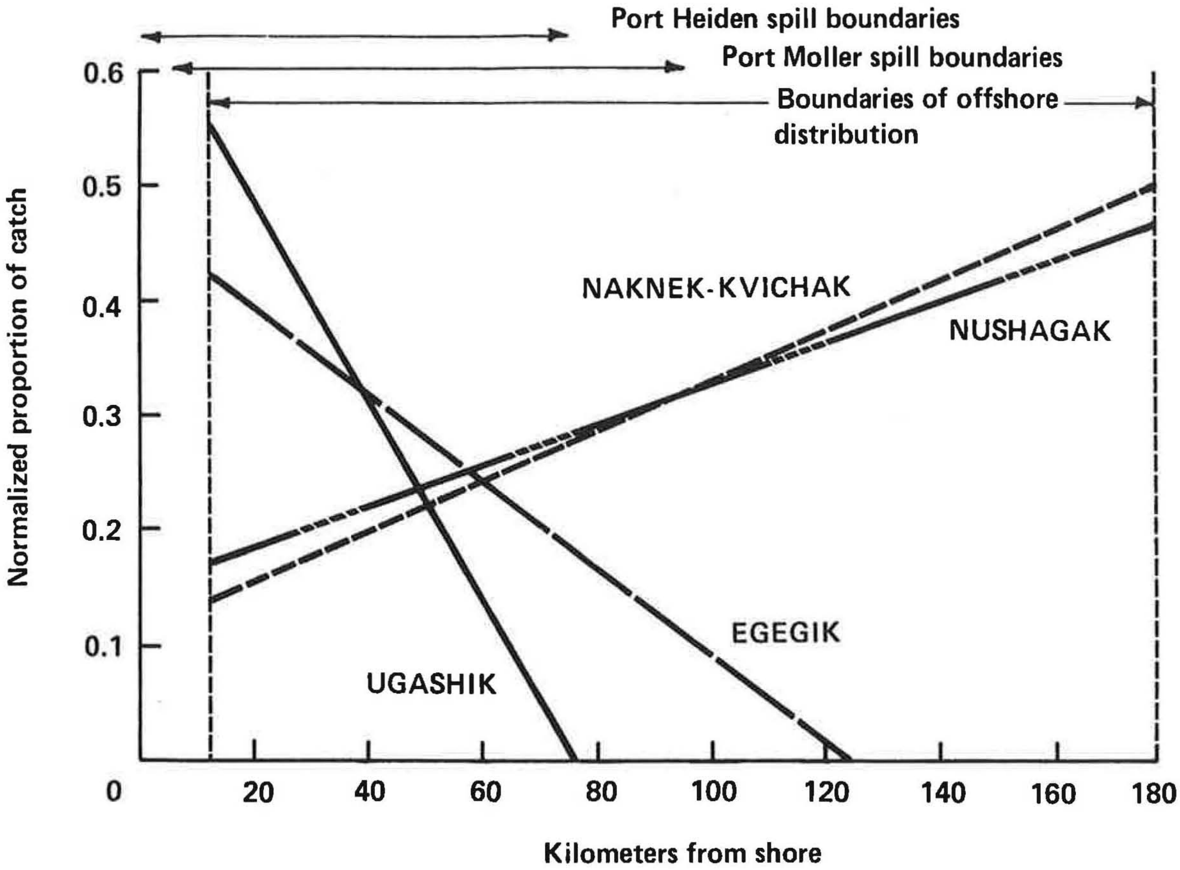

Figure 21.--Proportions of total catches of adult sockeye salmon between Port Moller and Port Heiden caught at varying distances from shore, and delineated by river of return. (Data from Straty 1975.)

These reduction factors can be used to replace the global factors used in Table 17 and to estimate the proportion of the fish arriving daily on the fishing areas in Bristol Bay that would be tainted above 0.6 ppm. Results from their application are plotted in Fig. 22.

The greatest percentage of tainted fish will be found at the Ugashik River fishing areas, and are caused by a spill off of Port Heiden. Percents tainted might reach about 30%, although the higher values occur towards the end of the 10 d period and will be overestimated as no allowance has been made for evaporation of the lighter aromatics which would cause some of the strongest taints.

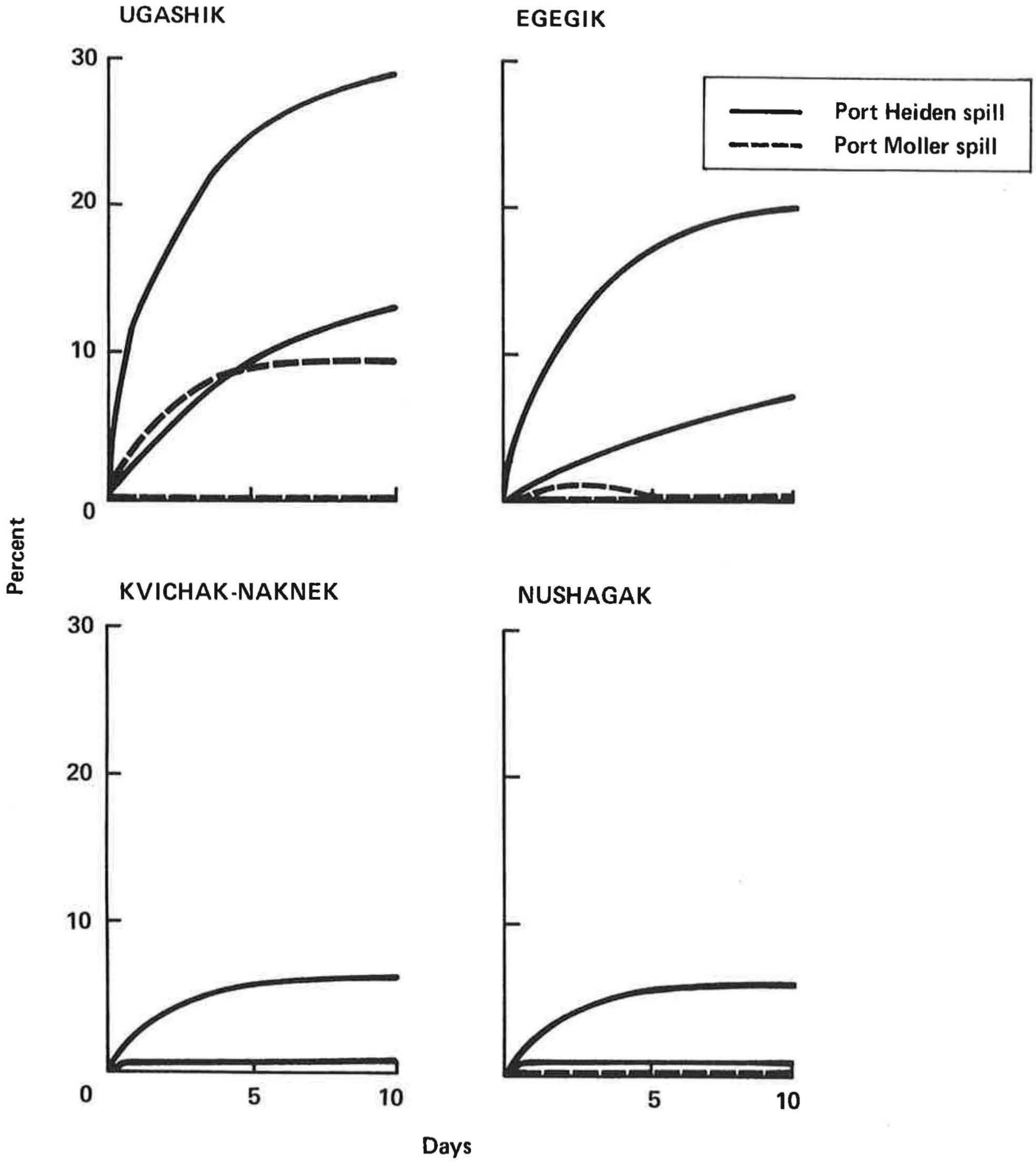

Figure 22.--Simulated proportions of adult sockeye salmon arriving on the fishing grounds in Bristol Bay that would be tainted after passing through a spill of No. 2 fuel oil from day 1 to day 10 of that spill.

DISCUSSION

The simulation presented here is of necessity an abstraction of the real events following an oil spill; further constraints were caused by a general lack of relevant data to estimate the required parameters. For this reason the simulation was kept as simple as was consistent with the objectives of the study and the effects of the various parameters on the results can be qualitatively determined from the provided outputs. A formal sensitivity analysis was not considered appropriate to the goals of the study because error bounds for the parameters could not be reasonably determined from the literature. Instead parameters and algorithms were chosen to maximize any effects of oil on the fish; the lower bounds of effects provided by including a probability of avoidance of the oil are again conservative as the avoidance algorithm operates only once in a timestep (1-2 hours for adults, 3-6 hours for juveniles) and not continuously as would be expected in the natural environment; the avoidance response does, however, assume that the fish can detect a gradient in oil concentrations at sea. Because no sensitivity analysis was performed it is important to emphasize that the results from this simulation should not be considered in isolation from the preceding literature review and the conclusions of this study should be updated as better data become available.

Although a sensitivity analysis was not approriate to the goals of this study, a formal sensitivity analysis could be used to determine the parameters in the simulation, and their natural analogues, that have the greatest effect on the percentages of fish tainted or dying following the described oil spill scenarios. This would indicate where future research efforts could be most profitably applied.

Several features of the simulation require final emphasis. Lack of any consideration for a change in the composition of the oil over the duration of the spill will cause progressively larger overestimates of the effects of the spill. These overestimates are less serious in the simulation of tainting where an allowance is made for the natural weathering of the oil once the fish has come into contact with it. Avoidance, tainting, and death are the only effects of oil contamination considered - no simulation is made of other potential sublethal effects, for example reductions in visual acuity or chemoreception that would affect subsequent schooling, homing, and spawning. Loss of energy caused by cessation of feeding in contaminated waters, by avoidance of the spill, or by a loss of directed migration on first contact with the spill could reduce the probability of survival of future stresses. This could be of critical importance to the adults which have large energy requirements during upstream migration and subsequent activity on the spawning grounds that are not supplemented by an energy intake in freshwater.

Throughout these simulations the sockeye salmon have been considered in isolation from other species which might interact with them as prey or predators. In the short term, response of prey populations did not appear important because the evidence suggested that the salmon would stop feeding at oil concentrations below those at which prey populations would be expected to change. Long term effects on prey populations could be more significant (e.g., Michael 1977). A more significant effect may be the response of predators to the polluted waters. The sockeye salmon smolts from the Kvichak and Nushagak rivers have no choice on descending the rivers but to eventually exit through the Bays where Frost et al. (1983) estimated 1,100 Belukha whales to be present during smolt outmigration in

1983. The belukhas major prey from late May to early June is the sockeye salmon smolt, and from mid-June to mid-August the adult sockeye salmon. The above authors estimated that in Kvichak Bay in 1983, the belukhas consumed about 6 million smolts, about 5% of the average smolt run, and 280,000 adults which comprises 1% of commercial sockeye salmon catch and 9% of the catch of other salmon species. Substantial numbers of other marine mammals and marine birds can also be expected to prey on outmigrating and returning salmon. Oil pollution could affect in the short term feeding behaviour and distribution and over the long term can be expected to adversely affect resident populations.

Another predatory pressure likely to be affected by an oil spill is that of the commercial salmon fishing fleet. It would require tainting in only a small proportion of the overall returns to make fishing unlikely due to adverse consumer perceptions. Following the 'Drupa' oil spill, saithe in seine nets contaminated with crude oil were wanted for neither animal nor human consumption, even though organoleptic analysis indicated an absence of tainting in the flesh (Grahl-Nielsen et al. 1976). Initially any reduction in mortality of the returning adults or emigrating juveniles would appear beneficial to the salmon population, however it has been suggested (Solomon and Mills 1982) that such a lack of fishing could cause an overescapement of salmon to the spawning areas with consequent redd superimposition leading to eggs being lost, or damaged and providing sites for the growth of infectious diseases. If overescapement is to apply it needs be demonstrated that a reduction in juvenile production occurs at higher spawner densities. Rogers (1984, Fig. 14) provides data on the mean spawner density and the resulting adult returns per spawner for six Bristol Bay sockeye salmon stocks (Kvichak, Naknek, Egegik, Ugashik, Wood, and Igushik) from

1952 to 1983. Means in spawner numbers ranged from approximately 270 to 5400 spawner/km^2 of lake area and the ratio of returning adults to spawners ranged from approximately 0.45 to 7.4. Taking the data from this Figure 14 and plotting the natural log of returning adults (R) against the natural log of spawners/km^2 (S) gives a slope of greater than 1.0 ($R = 47.35\ S^{1.74}$; $n = 32$, $0.01>p>0.005$) indicating that over this range of spawner densities there was no indication of declining returns, or even declining percentage returns, at the higher spawner densities. It is possible that at greater spawner densities reductions in recruitment would be observed (i.e., the above data fall only on the ascending limb of a 'Richer type' recruitment curve), however, exploitation rates of the returning Bristol Bay sockeye salmon (47% from 1951 to 1960; 48% from 1961 to 1970; 21% from 1971 to 1976 (Rogers 1977, Tables 1 and 2)) are such that even a complete cessation of fishing would lead to at maximum a doubling of escapement which for most years (27 out of 32) from 1952 to 1983 would still have produced an escapement less than the maximum recorded escapement over the same period. Thus it appears unlikely that on a system-wide basis "overescapement" is of concern in the Bristol Bay rivers; individual rivers could of course show a different trend than the mean.

A final point to emphasize in this study is that the results, especially for the juveniles, are dependent on the distance of the oil spill from shore as demonstrated by the difference between the two spill scenarios at Port Heiden and Port Moller. An inshore spill would have a greater effect on the outmigrating juveniles, especially if the oil entered the inlets where it is to be expected large numbers of juvenile salmon would rear.

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