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Final Reports of Principal Investigators

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Bax, N.J. 1985. Northwest and Alaska Fisheries Center Processed Report 85-03: Simulations of the effects of potential oil spill scenarios on juvenile and adult sockeye salmon (*Oncorhynchus nerka*) migrating through Bristol Bay, Alaska. U.S. Dep. Commer., NOAA, OCSEAP Final Rep 36 Part 1 (1986): 455-595

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OUTER CONTINENTAL SHELF
ENVIRONMENTAL ASSESSMENT PROGRAM

FINAL REPORTS OF PRINCIPAL INVESTIGATORS

Volume 36, Part 1

December 1985

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National Oceanic and Atmospheric Administration
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Final Reports of Principal Investigators

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EVALUATION OF THE EFFECTS OF OIL DEVELOPMENT ON THE COMMERCIAL FISHERIES IN THE EASTERN BERING SEA

by

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Final Report
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Research Unit 643

October 1985

PREFACE

Statements made in this report must be considered in light of the assumptions and caveats associated with the study. Any attempt to develop generalizations from the results of this study must be approached with caution.

This report summarizes findings presented in a series of reports that describe the results of a research project that was undertaken to provide OCSEAP estimates of the impact of two specific oil spill accidents at three sites on red king crab, yellowfin sole, Pacific cod, and sockeye salmon (Table 1). The study addressed three major areas of possible impacts of oil on fisheries: 1) effects of oil on fish and shellfish eggs and larvae and the projection of these effects over subsequent years, 2) effects of oil on adult fish and the possible uptake of hydrocarbons by fish, and 3) effects on the benthic ecosystem of weathered oil on the bottom of the ocean. Evidence based upon the best environmental and biological data available suggests that spills of the magnitude and at the locations indicated would not seriously impact the productivity of these stocks. Tainting was considered to be a potential problem. The impact on the fishery of the loss of fishing area and/or the fouling of vessels or gear were considered to the extent allowed by the data. The impact of oil accidents under various ice conditions, the effects of oil on the beach and inshore areas, and the effects of oil on marine mammals and birds were not considered. It is important to recognize that study findings must be evaluated within the context of the validity of assumptions made. Further, any change in the scenario, location of spills, or changes in the distribution of animals relative to the spill can be expected to produce a different impact. Such an evaluation would require a new study, as would the assessment of the impact of the original scenarios on other species of plants or animals and the impact if, in the simulation, significant quantities of oil were permitted to invade the beaches and estuaries of Bristol Bay.

This report is from a series of processed reports and program documentation produced by the Northwest and Alaska Fisheries Center, National Marine Fisheries Service, NOAA, in Seattle, Washington, and is individually available as Processed Report 85-19 from that source.

This study was funded by Minerals Management Service through an interagency agreement with NOAA.

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1. THE POSING OF THE PROBLEMS OF OIL DEVELOPMENTS AND FISHERIES INTERACTIONS

1.1 The posing of the problems pertaining to the effects of oil developments on the fisheries.

Some past oil spills from grounded tankers have caused extensive damage to beaches and have damaged local inter- and subtidal marine ecosystems. These coastal spills have received considerable attention in the news media and from the scientific community. However, no evidence has been found documenting noticeable detrimental effects of past oil developments on fishery resources (excluding minor local impacts), despite many Draconian forecasts of the possible impacts of oil developments on marine fisheries and ecosystems. Many of these sinister forecasts appear to have resulted from incorrect extrapolations of selective laboratory observations on the effects of hydrocarbons on the physiology, genetics, and mortality of fish (Payne 1982). As a result of misconceptions of possible effects of oil developments on marine ecosystems, an antagonistic attitude between oil development and fisheries (mainly presented by "environmentalists") prevails in the United States, whereas in Europe and in eastern Canada a cooperative attitude exists which is based on multiple use concepts of natural resources.

To clarify the possible effects of offshore oil development on fisheries, it is necessary to investigate this complex of problems quantitatively (numerically) using all available pertinent knowledge. A contract to this effect was given from Mineral Management Service via National Ocean Service to the Northwest and Alaska Fisheries Center. The present report presents the summary of the studies of the possible effects of oil developments on the fishery resources in the eastern Bering Sea, mainly in Bristol Bay.

1.2 Hypotheses on the probable effects of oil development on fisheries and fishery resources.

A main detrimental effect of local oil development on a fishery and its resources might be caused by an oil spill from a well blow-out or from a pipeline rupture. On the other hand, an oil spill from a tanker accident may occur anywhere in the world where oil is transported.

Oil spills at sea spread at the surface, from where the greatest part of it evaporates, and the remainder dissipates through the water column by dissolution and emulsification. Weathered oil settles to the bottom, and if the accident happens near the coast some of the oil might be blown to the shore. (This latter aspect is not considered in this study.)

If considerable concentrations of oil were to be found in the water column (dissolved and/or emulsified), it might have some lethal and sublethal effects on organisms (e.g. fish), before the natural purification restores the environment to pre-spill conditions (a matter of weeks). Sedimentized weathered oil on the bottom will, however, persist longer than in the water column, and may have some effect on benthic animals (including demersal fish) for a longer period.

It has been assumed in the past that some direct (and immediate) effects of an oil spill on fishing might be:

1) Loss of fishing area, due to presumption by the spill or cleanup activity (see Section 4.3).

2) Possibility of fouling of vessels or gear (a discounted possibility of extreme rarity).

3) Inability to sell catch due to tainting (see Sections 4.2 and 4.3). (Possible consumer avoidance, often intensified by journalistic sensationalism.)

4) Possible loss of catch, due to toxic mortality of exploitable stock, or of eggs and larvae affecting future exploitable stock (see Sections 4.1, 4.2, and 4.4).

5) Acute but latent mortality to eggs, larvae, juveniles, and adults (see Sections 4.1, 4.2, and 4.4).

6) Effects on habitat and alteration of prey population and food chain (see Section 4.2).

Although possible genetic mutations are mentioned in some literature, no serious scientific evidence can be found to elaborate on this very remote hypothesis.

Many other factors besides possible oil spill effects operate on fishery resources, such as year to year differences in availability of fish in given locations, natural fluctuations of stocks, effects of fishing on stocks, and market conditions. The effects of all factors affecting fish stocks can be evaluated on a comparative basis (i.e. comparing the oil spill effects to natural fluctuations and to the local effects of resource changes on the fishery as a whole).

1.3 Objectives of present study.

The potential impacts of oil development on fisheries are assessable with the present state of knowledge of complex dynamic, biological processes of stock production and ecosystem interactions, which can be attacked with complex marine fish ecosystem simulations.

The present study addressed three major areas of possible impacts of oil on fisheries:

Table 1.--Hypothetical oil-spill scenarios.

Scenario	Oil type	Volume	Duration	Computation grid size (mesh 2 km)
Blowout	Prudhoe Bay crude	20,000 bbl/day	15 days	50 x 50
Accident	Automotive diesel (refined)	240,000 bbl (10,000 bbl/hr)	10 days	32 x 34

Table 1.--Hypothetical oil-spill scenarios.

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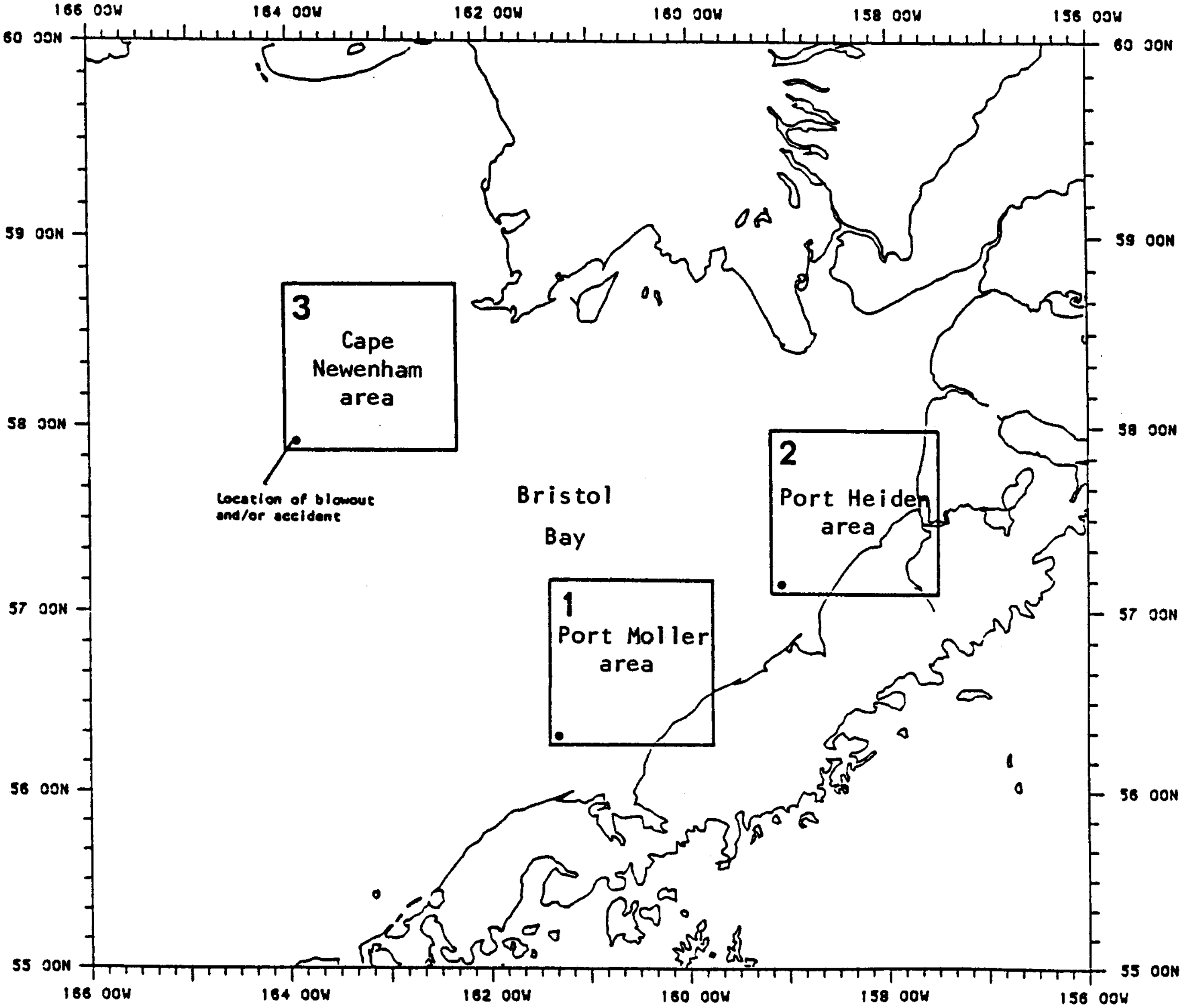

Figure 1.--Locations of hypothetical oil spill scenarios.

of fish per unit area, and/or peak migration time of anadromous fish).

4) The prevailing conditions affecting the sedimentation of the oil to the bottom were such that highest possible quantity of oil accumulated on the bottom in the shortest possible time.

Detailed results of the study are found in fifteen technical reports (see Section 5); this report presents the summary of the essential results.

2. REVIEW OF PAST RESEARCH AND ITS APPLICABILITY.

2.1 Interpretation of laboratory research.

A voluminous amount of literature is available on the laboratory studies of oil effects on fish and other aquatic biota. The corresponding reports on field studies are few, and are mostly qualitative, inconclusive descriptions of past accidents. Some summary works on the subject are also available (e.g. Connell and Miller 1980, and U.S. National Academy of Science (numerous authors) 1984). Objective quantitative evaluation of the laboratory studies and their applicability to the "real world" is difficult indeed, and one has to agree with the conclusion of the National Academy of Science report (1984):

"The single most significant gap existing to date is our difficulty in transferring the information obtained from laboratory studies to predicting and/or evaluating potential impact of petroleum on living marine resources in the field, especially in the case of spill impact on such commercially important stocks as fish and shellfish."

Principal problems with the evaluation of the past effect studies are:

1) Most of the laboratory studies have been carried out with WSF concentrations two to four orders of magnitudes (100 to 10,000 times) higher than would occur in the ocean with the greatest plausible accident.

2) Numerous different components of hydrocarbons have been used in these studies, with very different methods of exposure of fish and other marine organisms.

Only rarely does some report state honestly the applicability of their results, as has been done by Duval and Fink 1981:

"hydrocarbon levels in water following oil spills would rarely persist at the concentrations required to cause many of the physiological and behavioral effects observed during this investigation."

Studies of sublethal effects of petroleum hydrocarbons have also been summarized by Connell and Miller, op. cit., Nat. Acad. Sci. (op. cit.) and by Malins et al. 1982.

The essential applicable conclusion from the numerous past studies is that WSF concentrations in excess of 100 ppb are lethal to fish eggs and larvae within a few days, and that adult fish tolerate concentrations in excess of 1 ppm. The latter concentration can be taken as lowest limit of WSF concentrations which cause mortalities in fish within a few days.

The same concentrations (1 ppm) can be taken as the lower limit which causes sublethal effects in adult fish. The latter are often ill-defined; pathological changes in the liver of flatfish, for example, occur both in oil-exposed and non-exposed fish (Malins et al. 1982).

Most marine animals (including fish) are capable of metabolizing hydrocarbons. Metabolic products are usually retained longer in the bodies than parent hydrocarbons. Most of the hydrocarbons are taken up with food (especially benthos). It was concluded from the literature review that fish can be considered tainted if the concentrations of hydrocarbons in the body is >5 ppm. Hydrocarbons can be present in fish even when no tainting is detected (Grahl-Nielsen, Neppelberg, Palmork, Westrheim, and Wilhelmsen 1976).

2.2 Past experiences with oil spills as pertaining to fisheries.

Frequent remarks on possible effects of oil spills on fish and fisheries can be found in existing "oil spill literature". These unquantified remarks are, however, unsubstantiated in the majority of cases. Only five reports (summarized below) attempt to evaluate quantitatively the possible effects of oil developments and oil spills on fisheries. In addition, there exists a few good local studies on the subject which cover (and emphasize) the socio-economic aspects of oil developments on local fishing communities (e.g. Canadian studies from Newfoundland and Nova Scotia).

An earlier study by Johnston (1977) concludes that losses reckoned as fish production or its approximate cash equivalent are very small even for a catastrophic oil spill. Another study by Norwegian scientists (Norges Offentlige Utredninger NOU 1980:25) points out that the main effects of an oil spill on fish resources is via the effects of oil on fish eggs and larvae. These effects would be delayed several years and entirely masked by natural fluctuations of recruitment, and compensated by the presence of several year classes of fish in exploitable parts of the stocks.

Davenport (1982) reported that field studies have revealed no lasting damage to the planktonic ecosystem (one of the food sources for fish) caused by oil. Conan (1982) described that in case of catastrophic oil spills reaching estuaries (Amoco Cadiz spill), the estuarine benthos was affected by oil (see further details in Laevastu and Fukuhara 1985, ref. 7 in Section 5), whereas the resident fishes (flatfishes and mullets) were affected to a minor degree (possible reduced growth and fecundity, and some fin rot).

A thorough examination of the oil pollution and fisheries by McIntyre (1982) concludes that no long-term adverse effects on fish stocks can be attributed to oil. There might be, however, some local impacts, such as in estuaries as reported by Conan (*op. cit.*).

The results reported in this summary are the very first attempts to comprehensively estimate the possible adverse effects to the eastern Bering Sea environment and biota caused by spills of petroleum of specified composition and volume at designated spill sites. The list of the reports resulting from this study is presented in Section 5.

3. METHODS AND DATA USED IN PRESENT STUDY

3.1 Numerical methods.

3.1.1 *Oil in the water.*

The computations of the distribution of oil from the three sites of hypothetical well blowouts and tanker accidents (see Section 1.2 and Fig. 1) were carried out by Rand Corporation (Liu and Pelton 1984MS, Mannen and Pelto 1984). The dissolution and dispersion of oil in the water was based on studies by Payne, Kirsten, McNabb, Lambach, de Olivera, Jordan, and Hom 1983; and Payne and Kirsten 1985MS.

The presence and distribution of oil on the surface in offshore areas has no consequences to fish or fisheries. Any area closure for fishing will be determined by the area where contaminated fish can be found, which is considerably larger than the oil distribution area on the surface (see

Section 4.3). Obviously in some conditions oil on the surface could be beached, where it will be of local concern. Although some marine birds and mammals could be affected (and killed) by surface oil, these kills are relatively small in offshore waters (most birds and mammals have avoidance reactions), compared to the great amounts of birds and mammals present in the Bering Sea. Some fisheries interests consider it beneficial for fisheries if the birds and mammals get decimated.

The maximum concentrations of oil in water (WSF, including soluble and emulsified oil) was <0.34 ppm from the blowout scenarios. These low concentrations correspond well to observed concentrations from IXTOC blowout. Grahl-Nielsen et al. (1976) also observed low concentrations of oil under the oil slick (0.450 ppm 1 m under oil slick after 8 to 9 hours; 0.01 ppm after 24 hours). An example of distribution of oil from a blowout scenario is shown in Figure 2.

The maximum concentrations from the "tanker accident" were higher (ca 9 ppm), mainly because refined diesel oil was considered to be involved. The areas covered by different concentrations are reported by Pola, Miyahara, and Gallagher 1985 (see ref. 10, Section 5).

3.1.2 Oil on the bottom.

After "weathering" in the water much of the residual oil precipitates to the bottom. Gearing and Gearing (1983) found that about 50% of aromatics with three or more rings and saturates with 10 or more carbon atoms were rapidly transported to the sediments where their half lives ranged from 33 to 80 days. In shallow water the concentration of oil in muddy bottoms might reach 100 ppm (Marchand, Capris 1982).

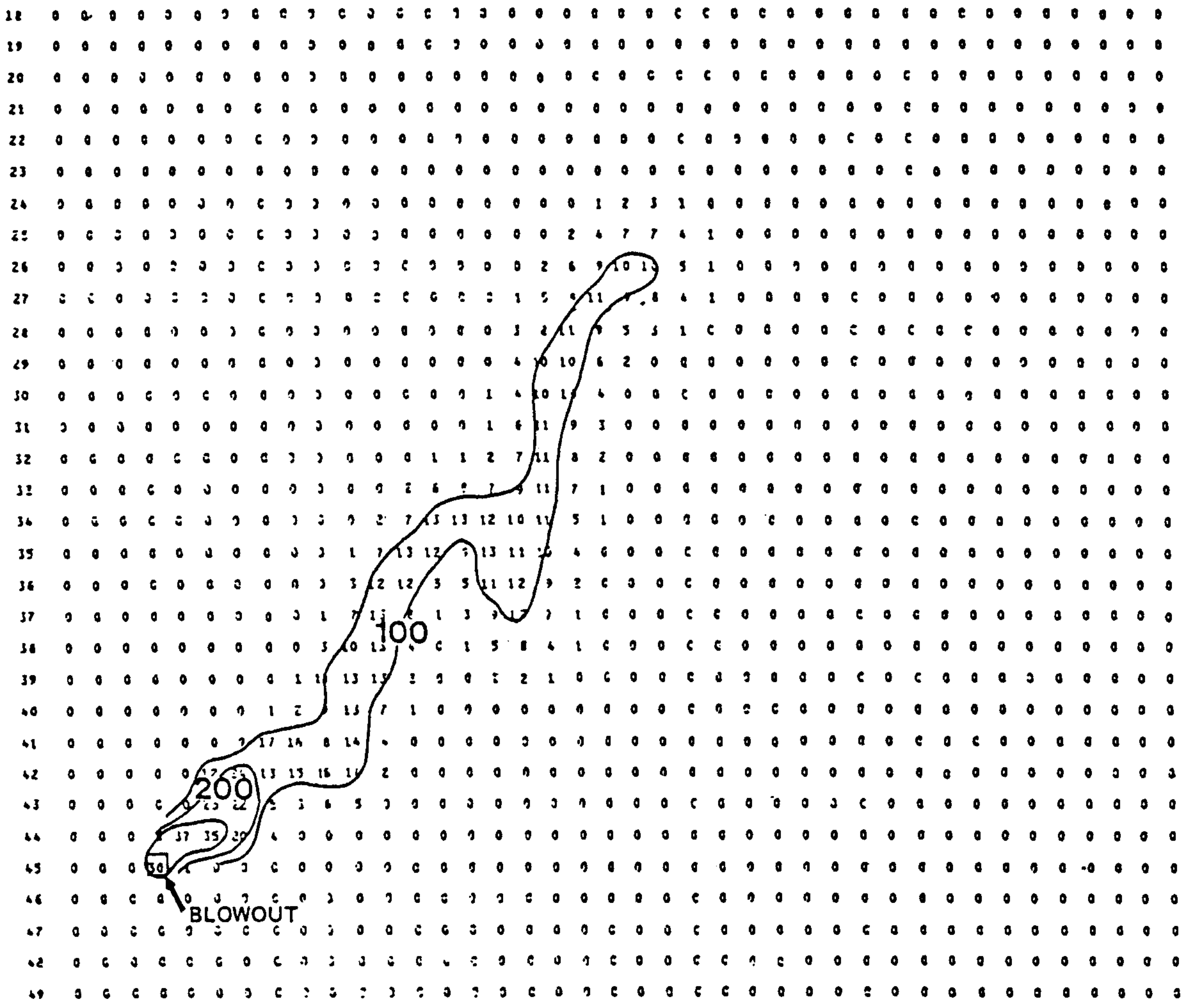

Figure 2.--Distribution of oil from a blow-out (20,000 bbl/day) after 10 days (concentrations in ppb in water - from surface to 15 m; grid mesh size 2 km).

The available literature on the sedimentation of oil and the effects of oil on the bottom on the benthos and demersal fish was reviewed and a numerical model for sedimentation of oil was designed (Laevastu and Fukuhara 1985). This model accounts quantitatively for all factors affecting the oil sedimentation (see example in Figure 3).

Initially the weathered sedimentized oil accumulates in the near-bottom nepheloid layer. The existence and thickness of this layer is dependent on several environmental factors, such as water depth, nature of the bottom, and water movement over the bottom.

Weathered oil is no longer directly poisonous to organisms and is taken up by benthos and via benthic food also by fish, causing tainting in fish. These tainting effects by sedimentized oil are considerably larger than the tainting from WSF of oil. Tainting is a temporary condition, as most petroleum hydrocarbons are disseminated from the body by various means (see Gallagher and Pola 1984, ref. 5, Section 5). The main effects of tainting would be a necessary area closure for fisheries (see Section 4.3).

3.1.3 Uptake and dissemination of petroleum hydrocarbons by fish.

After an extensive review of literature on uptake and dissemination of petroleum hydrocarbons a numerical model was designed which accounts for uptake, bioaccumulation, and dissemination of petroleum hydrocarbons (Gallagher and Pola 1984, Pola 1984, and Gallagher 1984 refs. 5, 3, 4 in Section 5). This model accounts for species differences due to e.g. feeding habits by assigning different uptake and depuration rate constants to different species. The model was tested via sensitivity analyses with the best available empirical data.

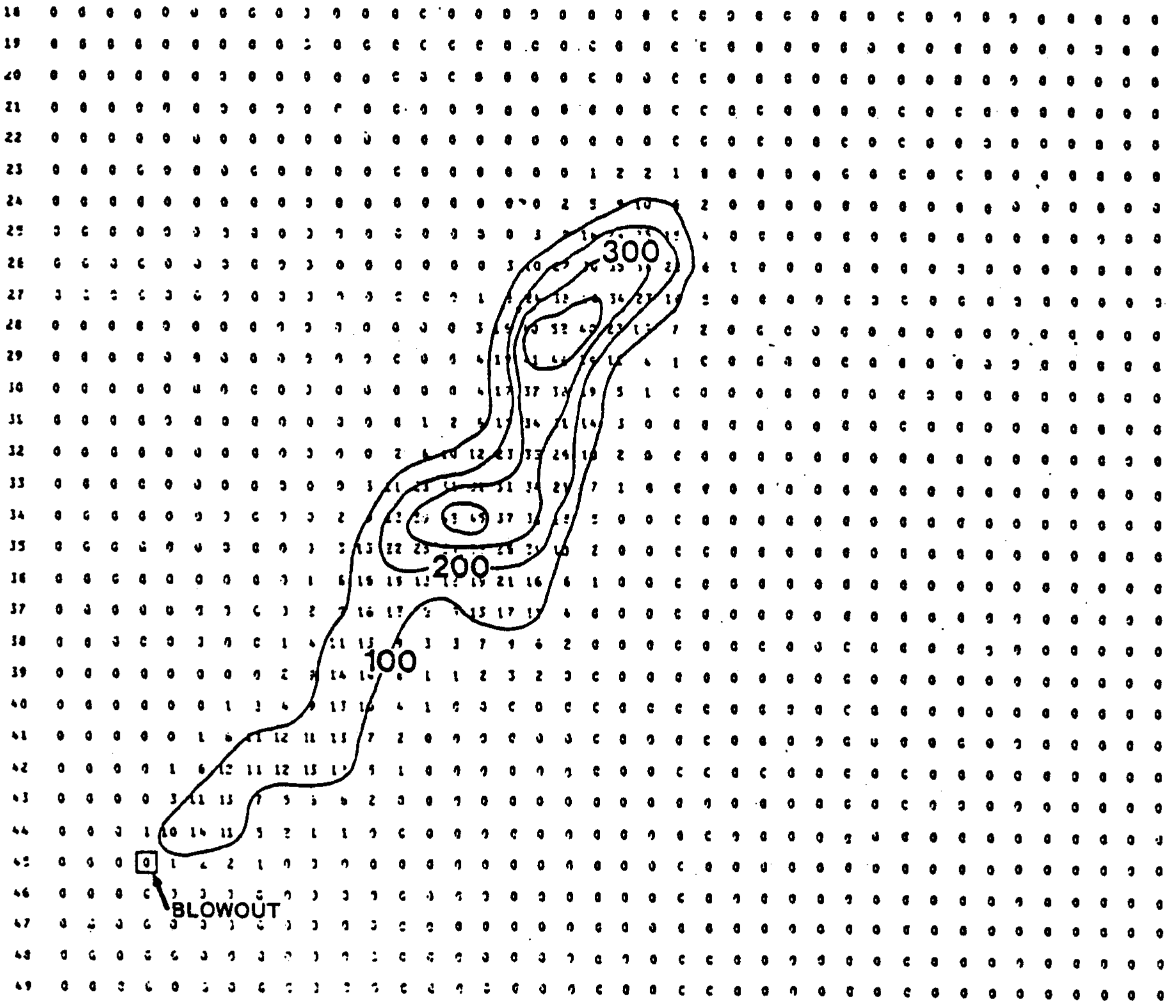

Figure 3--Distribution of oil in the bottom nepheloid layer (10 cm) in ppb 10 days after a well blowout (see Figure 2); grid size 2 km).

Another companion model moves the fish through the oil-contaminated area in various directions and with selected plausible fish migration speeds. During the migrations the uptake and dissemination model computes the contamination of fish by hydrocarbons. Thus the areas and times of possible fishery closure resulting from a given accident can be assessed (see Section 4.3).

As the anadromous fish (salmon) pose somewhat different problems, a special model was devised for computing possible oil contamination effects on migrating salmon (smolts and adults) (Bax 1985, ref. 8 in Section 5) (results see Section 4.4).

3.2 Data.

3.2.1 Environmental data pertaining to oil development - fisheries interactions.

Few environmental data are required for the evaluation of the effects of oil development on fisheries. The distribution of oil in the water and on the bottom was computed with wind and tide conditions which gave maximum concentrations of oil in the water and on the bottom. For computation of oil on the bottom optimum suspended matter load, bottom type, and mixed layer depth was assumed which would give MEC conditions. Some other environmental data is location and season dependent. The essential environmental data were summarized by Miyahara and Ingraham 1984 (ref. 6 in Section 5).

3.2.2 Fishery resources, their fluctuations, and fish species which might be affected by an oil spill.

The fishery resources in the eastern Bering Sea are mobile, with extensive seasonal and life cycle migrations. Thus the total Bering Sea resources and their seasonal distributions must be considered while investigating the effects of oil development. Furthermore, the natural fluctuations of the stocks must be taken into consideration, together with many species specific behaviours.

The resource estimates, using presently existing survey methods, are rather inaccurate, usually gross underestimates, but also overestimates in case of some flatfishes (i.e. "herding" effects of trawls). Resource evaluation with ecosystem simulation models, which account for a number of resource determinants, produces considerably more accurate results and has been used in the present study.

A list of species and their densities (kg/km^2) used in the three oil spill scenario areas (Figure 1) is given in Table 2. Table 3 gives the species present in the three computation areas as percentage of total Bering Sea biomasses of corresponding species. The feeding habits of the species under consideration have been described by Livingston 1985 (ref. 12, Section 5). In addition the biology and ecology of the most important commercial species in the area have been summarized by Fredin 1985, Fukuhara 1985a and 1985b (refs. 9, 13, and 15, Section 5). Pertinent biology, ecology, and resource fluctuations data on sockeye salmon have been summarized by Bax 1985 (ref. 8, Section 5).

Of some pertinence to the evaluation of the oil development impacts might be the following generalized data. The Bristol Bay area (where oil development

Table 2--List of species and input biomass data (by location) used in BIOS[1] model.

No.	Species Name	Input Biomass Data (kg/km^2)[2]		
		Port Moller	Port Heiden	Cape Newenham
1	Herring juveniles	1409	521	1551
2	Herring adults	1121	414	1234
3	Pollock juveniles	3708	2322	3261
4	Pollock adults	11007	6893	9679
5	Pacific cod juveniles	424	279	307
6	Halibut juveniles	730	330	240
7	Yellowfin sole juveniles	722	482	711
8	Other flatfish juveniles	2004	1472	1650
9	Yellowfin sole adults	800	534	789
10	Other flatfish adults	2004	1472	1650
11	Pacific cod adults	861	461	681
12	King and Bairdi crab juveniles	664	222	432
13	King and Bairdi crab adults	1654	553	1078
14	Mobile epifauna	5970	4995	6075
15	Sessile epifauna	13930	11655	14175
16	Infauna	19150	13750	19250

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[1] The DYNUMES model (Laevastu and Larkins, 1981) was used to get initial estimates of input biomass data for the three model locations of the BIOS model.

[2] The following assumptions were used to convert the data obtained from the DYNUMES model to biomass fields for use in the BIOS model.

a) Unless noted differently below, the breakdown of species biomass data into juvenile and adult fractions was based on Niggol (1982).

b) DYNUMES species group 5 (halibut) was assumed to be 100% juvenile (i.e., in these shallow waters during this season).

c) Yellowfin sole data were assumed to comprise 75% of DYNUMES species group 7 (yellowfin and rock sole).

d) DYNUMES species group 13 (Pacific and saffron cod) was assumed to be 100% Pacific cod.

e) DYNUMES species groups 7 (rock sole-25%), 6 (flathead sole, flounder), and 8 (other flatfish) were combined to make up the other flatfish group (species 8 and 9) for the BIOS model. These groups were assumed to be equally divided between juveniles and adults.

f) DYNUMES species groups 19 (king crab) and 20 (Tanner crab) were combined, and using available survey data, assumed to be comprised of 71.4% adults and 28.6% juveniles.

g) DYNUMES species group 24 (epifauna) was assumed to be 30% mobile and 70% sessile.

Table 3.--Percent of Bering Sea biomass (from DYNUMES model) in blowout and accident scenario study areas.

Species (group)		Location		
		Pt. Moller	Pt. Heiden	C. Newenham
1	Herring, juveniles	0.505	0.187	0.556
2	Herring, adults	0.505	0.187	0.556
3	Pollock, juveniles	0.471	0.295	0.414
4	Pollock, adults	0.471	0.295	0.414
5	Pacific cod, juveniles	0.577	0.379	0.418
6	Halibut, juveniles	1.220	0.551	0.401
7	Yellowfin sole, juveniles	0.902	0.602	0.888
8	Other flatfish, juveniles	1.141	0.838	0.939
9	Yellowfin sole, adults	0.900	0.601	0.888
10	Other flatfish, adults	1.141	0.838	0.939
11	Pacific cod, adults	0.577	0.309	0.456
12	King and Bairdi crab, juveniles	0.806	0.269	0.524
13	King and Bairdi crab, adults	0.804	0.268	0.524
14	Mobile epifauna	0.416	0.348	0.424
15	Sessile epifauna	0.416	0.348	0.424
16	Infauna	0.604	0.433	0.607

might occur) is ca 250,000 km and the rest of the fishery area in the Bering Sea is about 400,000 km^2. However, at some defined seasons Bristol Bay might contain about 80% of crab resources, 70% of herring, 70% of yellowfin and halibut, 60% of cod, and 50% of other fish resources of the Bering Sea. These high percentages do not, however, occur at the same time.

4. RESULTS OF QUANTITATIVE EVALUATION OF FISHERIES - OIL DEVELOPMENT INTERACTIONS.

4.1 Possible effects on eggs and larvae.

Eggs and larvae of marine animals are most sensitive to dissolved and emulsified oil (WSF) in the water. The mortalities and serious sublethal effects start at concentration of ca 100 ppb.

The areas covered with WSF >100 ppb are relatively small in case of a substantial blowout lasting 15 days (<150 km^2, Table 4). Even in case of such an unlikely event as 200,000 t tanker accident with diesel fuel (released almost instantaneously), the area covered by this concentration is <1200 km^2 (Table 4).

Most marine fish spawn over relatively large areas, and the pelagic eggs and larvae are distributed with currents and turbulence over very large areas. Furthermore, the spawning of most marine fish lasts three to six months, with peak spawning lasting also in excess of three weeks.

Of the species studied, the spawning of yellowfin sole and its eggs and larvae were found most affected by the simulated blowout and tanker accidents in Bristol Bay. (Coastal spawning of herring and capelin was not considered

Table 4.--Maximum spatial coverage (km^2) and maximum duration (days) of various levels of oil in water (WSF) and in bottom nepheloid layer (TARS) at different concentrations at Port Heiden.

Oil conc. (ppm)	Accident				Blowout			
	WSF		TARS		WSF		TARS	
	area	duration	area	duration	area	duration	area	duration
>1.0	380	13	752	33	0	0	0	0
>0.1	1160	21	1548	>50	132	12	248	24
>0.01	1844	28	2140	>50	444	20	460	43
>0.001	2480	36	2560	>50	616	27	652	>50

in this study and salmon is described in Section 4.4). If all yellowfin sole would spawn within two weeks and this spawning would coincide with the very unlikely tanker accident, only 1.2 percent of yellowfin eggs and larvae would be killed (Table 5). However, the yellowfin sole spawning period is about five times longer than that used for the simulated accident--thus less than 0.3 percent of yellowfin eggs and larvae would be affected. The fraction of eggs and larvae of other fish species that would be killed is less than this fraction.

The natural mortality of fish eggs and larvae is very large (the reduction in numbers from eggs to spawning adults is in general from between 2,000,000 to 2, to 50,000 to 2). Furthermore, if considerable mortality would occur due to an extensive oil spill, this would not affect the fishery resources, as the exploitable stocks are "buffered" by the presence of several yearclasses (Honkalehto 1985, ref. 11, Section 5). Consequently the possible oil developments in Bristol Bay would have no effects on fishery resources in this area via effects on eggs and larvae. Similar conclusion was reached by Järvela, Thorsteinson, and Pelto (1984) in respect to Navarin Basin. Further detailed considerations on this subject are given in reports by Fredin (1985) and Fukuhara (1985a, b, ref. 9, 13 and 15 in Section 5).

4.2 Exposure and contamination of fish by hydrocarbons.

The lethal effects of WSF of oil on fish commence in the 1 to 10 ppm range. In present studies we have used the lower value (1 ppm) to achieve MEC (Maximum Effect Condition). The maximum areas covered with different ranges of concentrations (blowout and the unrealistically large tanker accident) are

Table 5.-- Estimated percentage of mortality from acute toxicity in yellowfin sole in the accident scenarios at Port Moller, Port Heiden and Cape Newenham by life history group and quarter.

A. Percentage Mortality at Port Moller or Port Heiden Spill Sites

QUARTERS	1		2		3		4	
STAGE	WSF	TARS	WSF	TARS	WSF	TARS	WSF	TARS
EGGS & LARVAE	0	0	0	0	0	0	0	0
JUVENILES	.03	.15	.03	.15	.03	.15	.03	.15
ADULTS	0	0	.03	.15	.03	.15	0	0
B. Percentage Mortality at Cape Newenham Spill Site								
EGGS & LARVAE	0	0	0	0	1.2	0	0	0
JUVENILES	.03	.15	.03	.15	.03	.15	.03	.15
ADULTS	0	0	.03	.15	.03	.15	0	0

given in Table 4. In evaluating the effects (lethal and serious sublethal) we have also assumed that concentrations of weathered oil on the bottom (tars) in excess of 5 ppm affect the juvenile adult fish. This assumption is somewhat excessive according to available literature, but would give an absolute MEC.

Detailed computations were made with the models and results given in technical reports (see Section 5). For summary considerations we can use a simplified approach by considering data in Table 4 and Figure 4 with the data in Table 2 which gives the amounts of species present in the three computation areas (Figure 1) and the fraction of this biomass of the total species biomass in the eastern Bering Sea (Table 3) (most species have only one stock in this sea).

Of the species considered in this study, yellowfin sole and king crab were found to be most affected by the hypothetical oil spill (salmon see Section 4.4). A summary of the possible lethal effects of the spills on yellowfin sole are given in Table 5.

The extensive well blowout would kill and/or seriously affect only 0.03 percent of yellowfin (and crab) population in the eastern Bering Sea, which is nearly three orders of magnitude less than the accuracy of resource estimates. Thus an extensive blowout would have no quantifiable effect on offshore fishery resources in the eastern Bering Sea.

An unnaturally large tanker accident as used in our scenarios might kill or otherwise seriously affect 0.15 percent of the adult yellowfin population. This amount is about two orders of magnitude less than the accuracy of resource estimate, and at present less than 2 percent of the catch--i.e. about an

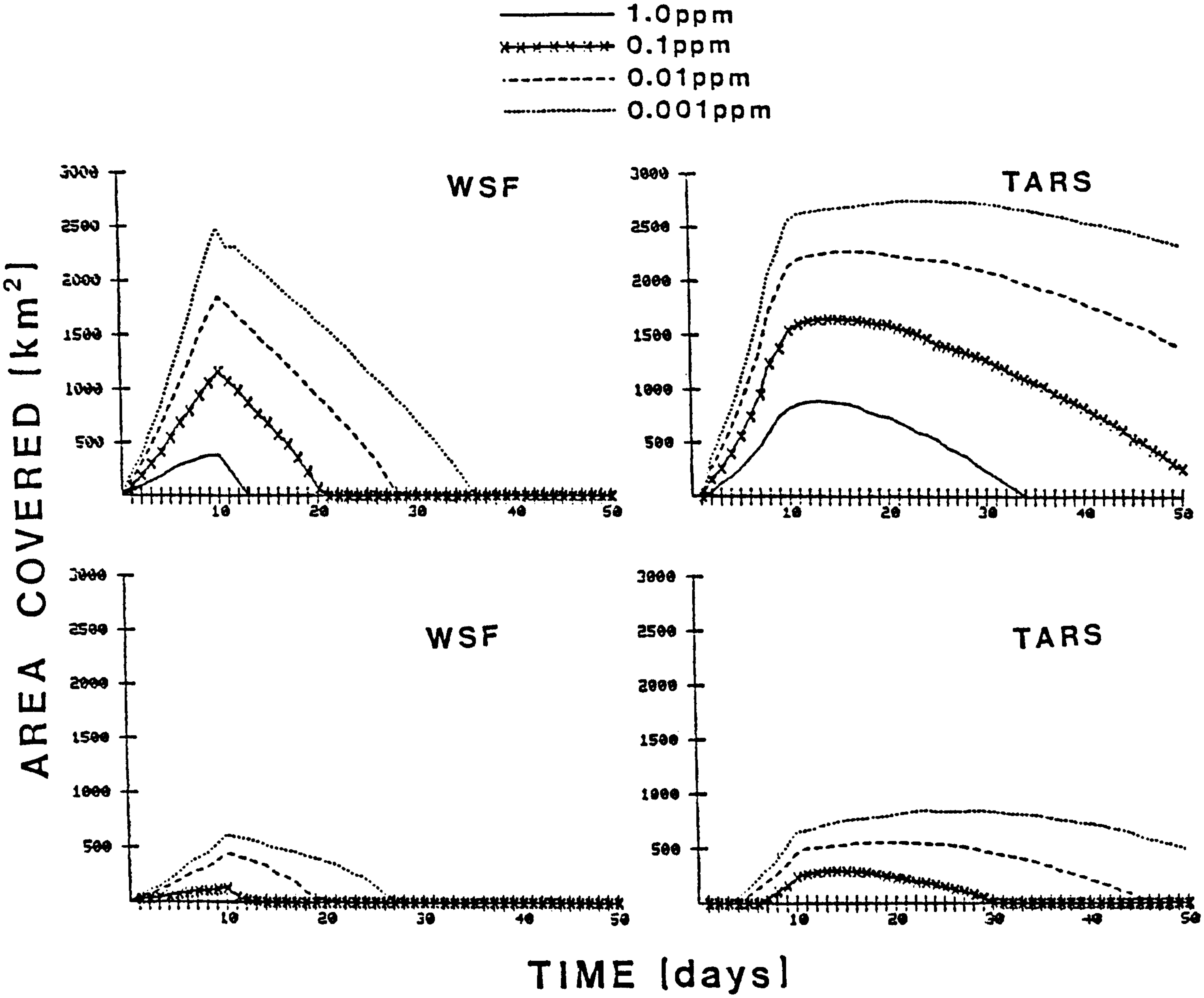

Figure 4.--Time series of total area covered (km^2) by WSF and by TARS at concentrations greater than 1.0 ppm, 0.1 ppm, 0.01 ppm, and 0.001 ppm for the accident (upper) and blowout (lower) scenarios.

order of magnitude less than the error in the estimation of catch. However, a 0.15 percent fluctuation of resource would have no effect on catch whatsoever. Thus even a large, unnatural tanker accident would have no quantifiable effect on the offshore fishery resources in the eastern Bering Sea.

Fish can, however, be temporarily tainted with petroleum hydrocarbons by direct exposure as well as by food uptake of contaminated food, mainly benthos. The uptake of petroleum hydrocarbons and their dissemination with time was computed in detail with numerical models (Gallagher and Pola 1985; Pola, Miyahara, and Gallagher 1985; refs. 5 and 10 in Section 5). The percentage of some biomasses in the computation area with internal contamination of >5 ppm (lower level of tainting) is given in Figure 5. These values have meaning to fisheries in terms of areas covered, which are given in Table 6. These areas are significant in the case of the blowout and/or accident when they should be temporarily closed for fishing to prevent tainted fish from being caught and marketed.

4.3 Effects of possible precautionary measures in offshore areas during an accident.

The possibilities of contaminating fishing gear with oil is often mentioned when listing the possible effects of oil developments on fisheries. We cannot see this ever happening in Bristol Bay. There is very little set gear (e.g. traps, longlines) used in this area. If some gear would be in the vicinity of the accident, there would be ample time to remove it. Mobile fishing gear (e.g. trawls) cannot be contaminated with oil, unless it is done willfully.

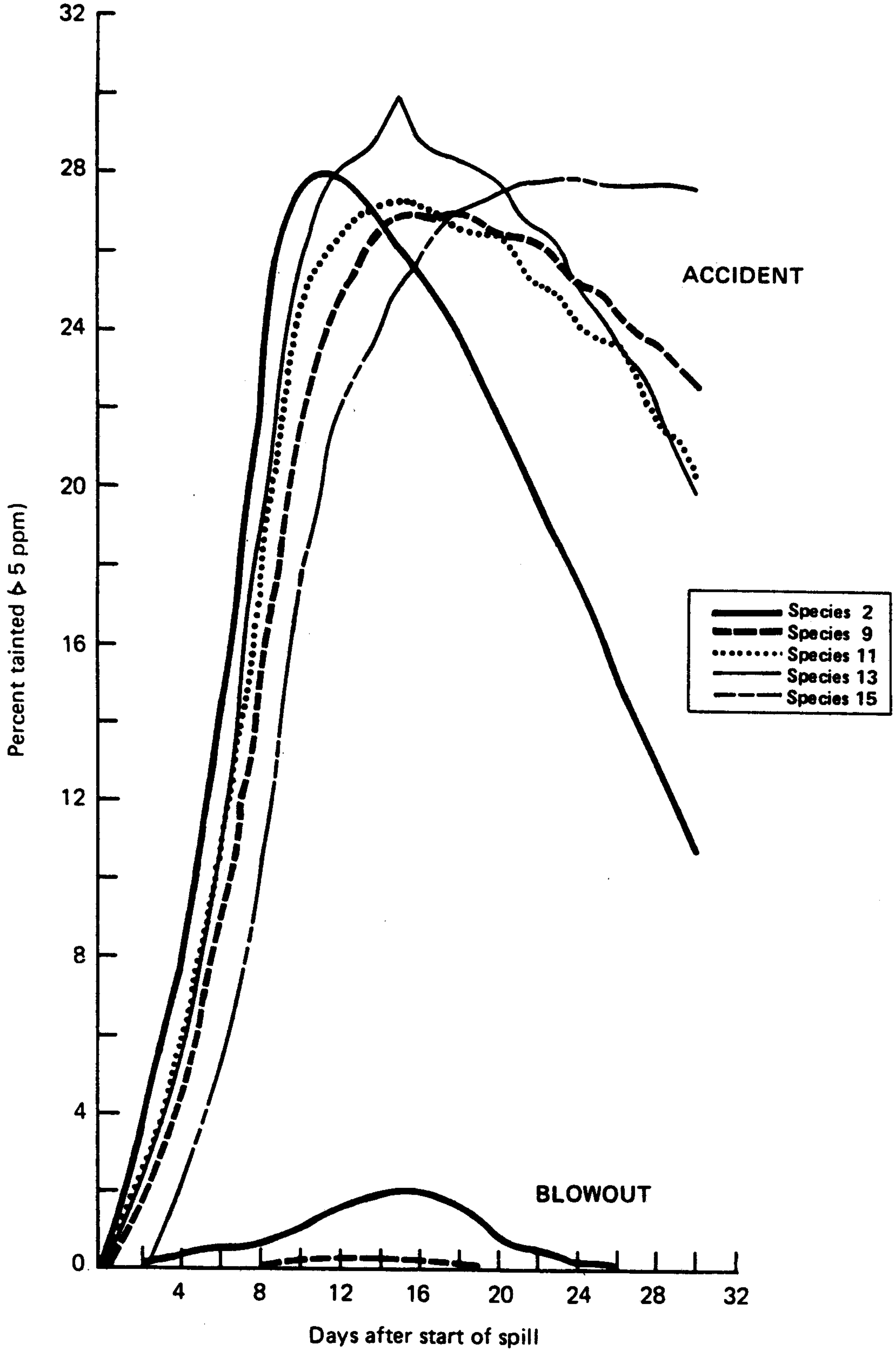

Figure 5.--Percent of biomass of selected species within the BIOS model grid tainted in the accident and blowout scenarios.

Table 6.--Areas covered with herring juveniles (Species 1) and adult King and Bairdi crabs (Species 13) contaminated with petroleum hydrocarbons at various levels. (Time sequence in days; area covered in km^2.) No migrations. Port Heiden accident scenario.

	Species 1				Species 13			
	Contamination (ppm)				Contamination (ppm)			
Day	>5	>10	>50	>100	>5	>10	>50	>100
1	72	56	24	4	24	8	0	0
2	144	120	60	32	80	52	0	0
3	240	208	96	48	144	96	0	0
4	356	292	140	80	228	160	8	0
5	468	392	192	96	320	232	40	0
6	616	512	248	116	424	324	68	0
7	764	632	304	144	564	428	92	0
8	968	792	360	148	736	544	124	0
9	1080	900	412	152	904	684	156	4
10	1212	988	452	168	1020	788	216	20
11	1224	1020	472	152	1108	868	268	24
12	1232	1036	452	136	1144	920	292	32
13	1216	1012	424	104	1192	960	328	32
14	1188	976	376	56	1228	976	356	32
15	1152	936	328	20	1236	1000	364	28
16	1100	880	268	0	1240	1008	368	20
17	1068	856	192	0	1240	1004	352	16
18	1036	792	116	0	1244	996	336	12
19	972	744	52	0	1240	976	320	8
20	924	684	4	0	1220	960	292	4
21	868	604	0	0	1188	952	272	0
22	820	552	0	0	1168	932	220	0
23	744	476	0	0	1152	916	196	0
24	688	400	0	0	1148	884	156	0
25	604	340	0	0	1128	860	112	0
26	540	248	0	0	1104	832	80	0
27	472	152	0	0	1068	784	24	0
28	388	76	0	0	1040	760	0	0
29	324	8	0	0	1016	732	0	0
30	224	0	0	0	968	696	0	0
31	128	0	0	0	940	640	0	0
32	48	0	0	0	904	592	0	0
33	0	0	0	0	864	568	0	0
34	0	0	0	0	824	516	0	0
35	0	0	0	0	780	448	0	0
36	0	0	0	0	748	412	0	0
37	0	0	0	0	692	372	0	0
38	0	0	0	0	648	304	0	0
39	0	0	0	0	588	248	0	0
40	0	0	0	0	548	180	0	0
41	0	0	0	0	488	116	0	0
42	0	0	0	0	440	64	0	0
43	0	0	0	0	384	16	0	0
44	0	0	0	0	324	0	0	0
45	0	0	0	0	268	0	0	0
46	0	0	0	0	192	0	0	0
47	0	0	0	0	136	0	0	0
48	0	0	0	0	92	0	0	0
49	0	0	0	0	28	0	0	0
50	0	0	0	0	0	0	0	0

If an accident should happen (i.e. oil spilled in the water in considerable quantities), fishing in the affected area must stop for awhile in order to prevent the capture and marketing of fish tainted with oil. The tainting of fish and the area covered, and time period of tainting, was computed with our simulations (Pola, Miyahara, and Gallagher 1985, ref. 10 in Section 5). The maximum areas covered in the cases of well blowout and tanker accident are given in Table 6 for two typical species (juvenile herring and adult crabs). Figure 6 shows the development of these areas with time and the subsequent depuration. Both Table 6 and Figure 6 refer to the tanker accident which produces the largest effect.

The maximum area covered with tainted crabs is <1300 km^2. After 30 days the area has decreased to <1000 km^2 and after 50 days all fish and crab would be depurated below detectable level. The tainting from a well blowout was considerably less, covering less than a quarter of the abovementioned areas.

In case of a very unlikely tanker accident (which might happen anywhere in the world), an area of about 2000 km^2 should be closed for fishing for about 45 days. Whether and how much such a closure can affect fisheries is meaningless to evaluate quantitatively. First, the event is extremely rare. Secondly, it might happen in an area which is not a traditional fishing ground. Thirdly, the fishing areas (grounds) are of considerable extent (species and season dependent) and fishing might continue in other nearby areas with same profitability as it would have done in closed area. (It could be noted that 2000 km^2 is less than 1% of the area of Bristol Bay, and equally less than 1% of the "prime" fishing grounds in the Bering Sea.)

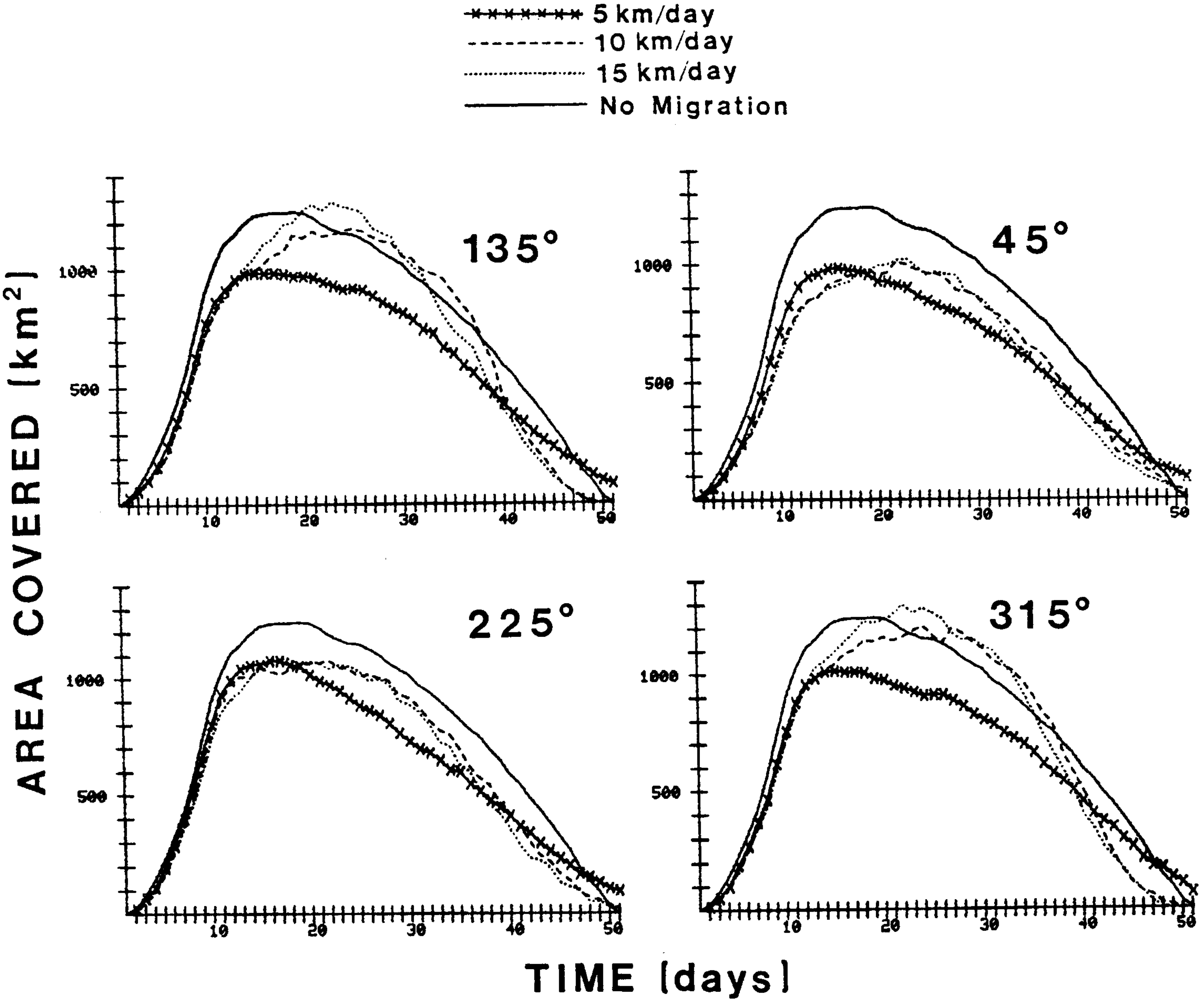

Figure 6.--Area covered by tainting (contamination >5 ppm) of a pelagic fish species from a model run with no migrations (solid line) and with migrations of 5, 10, and 15 km/day. Migration directions are shown.

4.4 Special considerations with anadromous fish

Special, careful consideration of the possible effects of oil development on salmon is required due to its importance in Alaskan fisheries and especially because of the possibility of the presence (and/or passage) of a great portion of outmigrating juveniles (smolt) and returning adults in possible oil spill sites. A thorough numerical study of possible effects of oil on sockeye salmon (the main species in Bristol Bay) was conducted within the three blowout and accident sites (Bax 1985, ref. 8, Section 5). The assumptions of oil effects in this study were more conservative than with marine fish to achieve MEC (100% mortality at 450 ppb of fuel oil in 24 h; 100% mortality at 2.5 ppm of crude oil in 24 h; tainting level in salmon flesh 600 ppb).

The computations of migrations of smolt (juveniles) and adults was carried out with no-avoidance and with avoidance reaction. The results of the effects of blowout and tanker accident happening during the most unfavorable periods in Port Heiden and Port Moller areas are summarized in Tables 7 and 8 (considering only that portion of the populations passing through these areas at the accident time). The mortalities and tainting extrapolated to whole Bristol Bay sockeye population for the tanker accident is given in Table 9.

A maximum of 13% mortality of outmigrating smolt could be caused by unlikely tanker accident. This does not mean that the returning year class would be affected by the same amount, as the natural mortality of smolt is variable from year to year (on average 90%). It is unrealistic to quantify the minor effect on smolt in terms of future (2 or 3 years later) fishing on returning adults.

Table 7.--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 8.-- 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

Table 9.--Percent mortalities and tainting from tanker spill scenarios extrapolated to whole population.

Age group	Location of spill	Reduction factor xx	Percent mortalities		Percent tainted	
			Direct	Avoid	Direct	Avoid
Juveniles 1.[x] (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.[x] (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

x Juveniles which spend 1 resp. 2 years in fresh water.

xx Fraction of the population passing through the three oil spill scenario areas.

The adults of total Bristol Bay sockeye salmon population might sustain a 5% mortality and an additional 2% tainting. Local disruption of salmon fishery might occur if a tanker accident of the unreal magnitude would occur during the peak salmon run (within about a month), especially if this occurred close to the fishing grounds.

5. LIST OF TECHNICAL REPORTS RESULTING FROM THE STUDY

(listed in chronological order of reproduction)

1. Kim, S. and A.W. Kendall.

   1983 (December). The numbers and distribution of walleye pollock eggs and larvae in the southeastern Bering Sea. NWAFC Proc. Rpt. 83-22, 35 pp.

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THE NUMBERS AND DISTRIBUTION OF WALLEYE POLLOCK EGGS AND LARVAE IN SOUTHEASTERN BERING SEA

by

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Final Report
Outer Continental Shelf Environmental Assessment Program
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Figure 1. Research area in southeastern Bering Sea. The size of each subarea is 0.5° latitude by 1° longitude. 100 m isobath is included.

Figure 2. Spatial distribution of pollock eggs and larvae. The three subareas with the greatest densities during each survey are indicated. 100 m isobath is included. (A) Eggs (B) Larvae.

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INTRODUCTION

Our interest and knowledge in walleye pollock, Theragra chalcogramma, the most abundant commercial fish in North Pacific Ocean has increased through last decade. At the same time, interest in petroleum development in the area has increased. Our present research is prepared to exam some possible effects of oil pollution on fish in southeastern Bering Sea. In walleye pollock the planktonic egg and larva is the life history stage that would most likely be severly impacted by oil pollution. Therefore we have examined the amount of spawning and its distribution in time and space in southeastern Bering Sea.

The main spawning ground of walleye pollock is located between Unimak Pass and the Probilof Islands. Spawning occurs during a fairly long season, primarily in spring.

METHOD AND DATA

The spatial and temporal distribution of spawning is inferred from the distribution of the planktonic eggs. For the spatial scales used in our analysis, and considering the generally low velocity of currents in the southeastern Bering Sea, differences in the spatial distribution of eggs and of spawning cannot be detected. The eggs require between 2-3 weeks to hatch at the temperatures at which they occur. Thus, the temporal distribution of spawning would be displaced 2-3 weeks earlier than the occurrence of eggs. Considering egg mortality the actual displacement would probably be less.

We reviewed information about walleye pollock egg and larval distribution from various ichthyoplankton surveys, and collected available data from them.

Our area of concern, the southeastern Bering Sea, has been examined by research teams from several different countries (U.S., U.S.S.R., Japan). However, because each team had different sampling strategies as well as different sampling gear, comparing their data is very difficult and dangerous. Therefore, we chose U.S. scientists' ichthyoplankton surveys (Waldron, 1978; Waldron and Vinter, 1978; and Walline, 1981) that used same gear and same sampling methods for surveys during 1977, 1978, and 1979.

From 1977 to 1979, the sea surface temperatures near Pribilof Islands were warmer than the average (Niebauer, 1980) while from 1973 to 1976 they were colder than the average. However, the average sea surface temperatures during March, which is assumed to be important time for pollock spawning, were nearly the same from 1977 to 1979. Therefore we feel it is reasonable to assume that there was no difference in pollock spawning pattern among the years 1977-79 (the temperature during April 1977 was below normal, although it was above normal in March and May).

In order to investigate the pattern of temporal and spatial spawning and egg and larval distribution, we divided our research area to 37 rectangles of 0.5 degree of latitude by 1 degree of longitude each (Fig. 1). We calculated the areas of the small rectangles based on Lafond (1957). For better temporal resolution, we divided the 1977 cruise into four surveys. We then arranged the U.S. ichthyoplankton surveys in seasonal sequence and assigned them Roman numbers as follows:

Survey I ; March 10-26, 1978; Waldron (1978)
Survey II : April 16-22, 1977; Waldron and Vinter (1978)
Survey III ; April 23-27, 1977; Waldron and Vinter (1978)
Survey IV ; May 3-9, 1977; Waldron and Vinter (1978)
Survey V ; May 10-15, 1977; Waldron and Vinter (1978)
Survey VI ; June 1-23, 1979; Walline (1981)

As metioned above, we divided our research area to 37 subareas. The total number of eggs or larvae in a subarea was calculated by multiplying their average number per square meter by the area of the subarea in square meters. Appendix III includes results of eggs and larval abundance calculations for each subarea.

To arrive at the temporal distribution of spawning, we assumed uniform spawning activity within the total sampling area. Therefore the total amount of eggs present in the entire samping area during each specific survey was calculated by multiplying the area occupied by each station by sum of numbers of eggs present per square meter through all stations.

Generally spawning activity with time approximates a normal distribution. Therefore we could make a normal distribution curve of pollock egg abundance with time from a polynomial equation (Tanaka, 1962) based on the above data sources:

The normal distribution curve is

$$F(X) = \frac{N}{S\sqrt{2\pi}} e^{-\frac{(x-m)^2}{2S^2}}$$

where N = total number of individuals,
S = standard deviation of distribution
m = mean of distribution,
X = abscissa, Julian day,
π = 3.14159

Taking natural logarithms, we get

$$\ln (F(X)) = (\ln \frac{N}{S\sqrt{2\pi}} - \frac{m^2}{2S^2}) + (\frac{m}{S^2})\, X - \frac{1}{2S^2}X^2$$

This is the parabola, also called the second degree polynomial which has form of $Y=A + BX + CX^2$. Because we can get the values of the constants (A, B, and C) by use of a computer, N, m, and S can be calculated;

$$A = \ln (\frac{N}{S\sqrt{2\pi}}) - (\frac{m^2}{2S^2})$$

$$B = \frac{m}{S^2}$$

$$C = - \frac{1}{2S^2}$$

For comparison with our study based on U.S. data, we made another normal distribution curve of pollock egg abundance with time from Moiseev and Bulatov's (1979) data. Because Moiseev and Bulatov (1979) did not include information about egg production during June, we used a digitizer to calculate egg production during June from Kendall (NWAFC, unpublished graph which was based on Bulatov (1979)).

Based on U.S. scientists' ichthyoplankton survey data, total egg production can be estimated as in Houde (1977), assuming there were no significant differences of spawning pattern during 1977-79, i.e., we combined data by month ignoring the year of the survey.

$$N = \Sigma((N_i \cdot D_i/a)$$

where N is the total eggs spawned during one spawning season,

N_i is the number of eggs present in survey i

D_i is the duration of survey i

a is egg hatching time

Fecundity and age composition data of pollock enable us to make an egg production estimation.

There are several data sources for age composition, fish fecundity-length or weight relationship, and von Bertalanffy parameters. After considering all of the data sources, we used data of Smith (1978), Shew (1978), Niggol (1982), and Niggol (1982), respectively because they appeared most reasonable.

In order to calculate the total biomass of eggs spawned, we used egg density data from Kanoh (1954) and egg diameter data from Nishiyama and Haryu (1981).

In general, fecundity of fish is expressed by

$$\text{Fecundity} = A \cdot L^B \qquad (1)$$

where A and B are constants, and L is the length of fish.

$$\text{Fecundity} = E \cdot W^F \qquad (2)$$

where E and F are constants, and W is the weight of fish.

And weight of fish is

$$W = C \cdot L^D \qquad (3)$$

where C and D are constants.

Because the length and weight of fish is the function of time, we can express them with time from von Bertalanffy equation;

$$L_t = L\infty(1-e^{-K(t-t_o)}) \qquad (4)$$

$$W(t) = W\infty(1-e^{-K(t-t_o)})^D \qquad (5)$$

where L_t and Wt = the length and weight at age t

$L\infty$ and $W\infty$ = the asymptotic value of length and weight

K = a relative growth completion rate

t_o = a hypothetical age of zero size

d = a dimensionless exponent reflecting absolute growth rate.

If we combine (1) and (4), and (2), (3) and (5) with each other, we get two fecundity equations with time as follow:

$$\text{Formula I} : \text{Fecundity } (t) = A\ L\infty(1-e^{-K(t-t_o)})^B$$

$$\text{Formula II: Fecundity } (t) = E\,(W\infty(1-e^{-K(t-t_o)})^D)^F$$

From these we can get the total numbers of egg produced by multiplying fecundity by the total number of females spawning;

$$N = \Sigma(F_t \cdot N_t)$$

where N is the total number of eggs sapwned

F_t is the fecundity of age t fish

N_t is the number of fish at age t.

RESULTS

As mentioned before, we divided our research area to 37 subareas and calculated the number of eggs and larvae present in each area during each survey (Table 1 and Fig. 2). It appears that spawning is active on upper slope area during March, it moves to middle and outer shelf during April, and moves to northwestward later in the spawning season. Roughly speaking, spawning is active at around the 100 m isobath in the southeastern Bering Sea although spawning activity probably depends on several abiotic factors in addition to depth. The larval distribution in Fig. 2 shows seaward movement of larvae after hatching. This pattern of larval distribution was also observed by Serobaba (1974).

After the numbers of egg present at each station during each survey were calculated (Appendix I), the total number of eggs present in our research area during each survey was calculated (Table 2). Also egg numbers present in eastern Bering Sea from Moiseev and Bulatov (1979) and Kendall (NWAFC, pers. comm.) during 1978 spawning season were calculated (Table 3).

When we assumed that the spawning started on the middle of February, the normal distribution based on the above data indicates that 104.4th Julian day (14 April) is the peak of spawning with a standard deviation of 13.8 days (Fig. 3: $F(X) = 27.08e^{-\frac{(X-104.41)^2}{382.16}}$). Also under the same assumption, the result derived from Moiseev and Bulatov (1979) and Kendall (pers. comm.) shows 107.9th Julian day (18 April) as the peak of spawning and 26 days as one standard deviation (Fig. 3: $F(X) = 15.56e^{-\frac{(X-107.94)^2}{1428.82}}$), although smaller egg production was indicated by the Soviets' data than by U.S. scientists'. These results are not significantly different from others which indicate that along the slope and outer shelf the peak spawning occurs in March and April (Lynde, 1983).

Larvae were found beginning in the middle of March in southeastern Bering Sea. Table 1 shows that the maximum larval abundance occurred at around second survey, which was conducted just after peak spawning, and that the number of larvae decreases with time. Walline (1983) calculated the egg hatching dates for pollock from 1979 survey data; hatching dates were distributed from 1 April to 15 July; the hatch was most pronounced during the last 2 weeks in April and the last 2 weeks in May.

Based on Table 2, (ichthyoplankton survey data), the total number of pollock eggs produced during the spawning season in southeastern Bering Sea was estimated. If development time from spawning to hatching is assumed to be

17.3 days which corresponds to duration of the pollock egg stage in 4°C seawater (Richard Bates, NWAFC pers. comm.), the total number becomes 3.6918 E+13 eggs (282.7 eggs/m^2) (see Appendix II).

Also we calculated pollock egg production by knowing the fecundity-age composition relationship of the adult population, although more research is needed to estimate the parameters more precisely for pollock in the eastern Bering Sea. For calculations of number of eggs produced the following constants were used: von Bertalanffy constants of K=0.209, to = -0.315 year, L_∞ = 65.01 cm (Niggol, 1981); Shew's fecundity-length relationship data (1978); and Smith's age compositon data (1981). If we multiply fecundity by number of females at each age, the caculation resulted in 6.17 · E+13 eggs in 159,100 km^2 which equals a density of 387.95 eggs/m^2. For transforming this number to total weight of eggs produced, we multiplied the total number of eggs by the average density of egg (Kanoh, 1954), 1.021, and mean egg volume of 0.0027 cm^3 which came from Nishiyama and Haryu (1981). The total weight of eggs produced in the spawning area of 159,100 km^2 is about 170,000 mt in one spawning season.

DISCUSSION

The egg density derived from fecundity data (387.95 eggs/m^2) is higher than that of the ichthyoplankton surveys (282.7 eggs/m^2) and that of Serobaba (1968) (293.44 eggs/m^2). This indicates that spawning probably occurs outside the areas sampled during these ichthyoplankton surveys. Also catchability of eggs may be less than 100% with the methods used in these surveys.

In performing this study, the biggest problem is in data variation. Because small numbers of ichthyoplankton surveys have been conducted in Bering Sea and data collecting methods have varied, we needed to use many assumptions, which should be reconsidered when more data becomes available. Also the basic biology of pollock is not well understood. We do not know the exact spawning time of pollock in southeastern Bering Sea nor do we understand the variation in size of pollock eggs, which influences our estimation of pollock egg biomass. In order to understand pollock in Bering Sea more exactly, we need to increase research activities to study their biology in relation to oceanographic conditions.

CONCLUSIONS

Spawning occurs over a large area of the southeastern Bering Sea, mainly between the Pribilof Islands and Unimak Pass, between the 100-200 m isobaths. Present data are insufficient to resolve the pattern of spawning and larval distribution adequately. The total spawning area was probably not sampled during the surveys considered in this report.

Spawning peaks on about Julian day 104-108 (14-18 April) with a standard deviation of between 14 and 26 days, based on two sets of ichthyoplankton surveys. Some spawning occurs from about 15 February through June.

Based on fecundity and size of the adult population, 6.17 · E+13 eggs are produced during the spawning season. Based on ichthyoplankton surveys 3.69 · E+13 eggs are produced during the spawning season. The difference between these two estimates may partially be a result of the plankton surveys not covering the entire spawning area. The density of egg spawned based on adult population parameters was 388 eggs/m^2 while that based on plankton surveys was 283 eggs/m^2. This difference may be due in part to problems in plankton sampling and egg mortality.

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Table 1.--The total numbers of eggs (E+11) and larvae (E+11) present in each sub area during each survey. Larval data are in parentheses.

	Survey					
Subarea	I	II	III	IV	V	VI
A(1,1)		0.0000			1.3560	
		(0.000)			(0.428)	
A(1,2)		0.0381			1.8716	0.0000
		(0.114)			(0.496)	(0.000)
A(1,3)						0.0000
						(0.024)
A(1,4)						0.0221
						(0.092)
A(1,5)						0.1158
						(0.719)
A(2,1)						0.0000
						(0.000)
A(2,2)			1.0005		0.0867	0.0000
			(1.221)		(0.737)	(0.024)
A(2,3)			1.2447		0.8949	0.0000
			(0.181)		(0.291)	(0.000)
A(2,4)			0.5220		0.6674	0.0079
			(0.019)		(0.000)	(0.075)
A(2,5)			12.9470	3.7147		
			(0.000)	(0.000)		
A(3,1)					0.0485	
					(0.388)	
A(3,2)	0.0193	0.0397			0.0413	0.0000
		(0.218)			(0.496)	(0.000)

Subarea	Survey I	 II	 III	 IV	 V	 VI
A(3,3)	0.0182	0.1532	0.1871		0.4421	0.0000
		(0.153)	(0.122)	(1.118)		(0.103)
A(3,4)	0.0000	0.1547	0.1793		0.2623	0.0000
		(0.000)	(0.021)		(0.031)	(0.123)
A(3,5)	0.0000	0.6378	0.3298	0.7144		
		(0.000)	(0.000)	(0.675)		
A(3,6)		25.8220		0.7017		
		(0.000)		(0.000)		
A(4,1)	0.1158					
A(4,2)	2.2444	0.0867			0.0213	0.0000
		(12.113)			(0.255)	(0.000)
A(4,3)	0.0550	0.1239	0.3672	0.1437	0.2008	0.0000
		(0.137)	(0.516)	(1.265)	(3.414)	(0.015)
A(4,4)	0.0000	0.0186	0.0102	0.1648		
		(0.008)	(0.051)	(0.991)		
A(4,5)	0.0309	1.3577	3.9712	2.2624		
		(0.071)	(0.135)	(0.822)		
A(4,6)	0.0000	2.5091	11.9090	7.6239		
		(0.000)	(0.000)	(0.590)		
A(4,7)				0.4143		
				(0.000)		
A(5,1)	0.0000					
						(0.025)

Subarea	Survey I	 II	 III	 IV	 V	 VI
A(5,2)	0.2358	0.5192 (16.817)		0.0677 (0.293)		0.0000 (0.000)
A(5,3)	34.3490	0.0160 (1.495)	0.0483 (6.884)	0.0990 (2.299)		0.0000 (0.012)
A(5,4)		0.1257 (3.839)	0.5852 (3.162)	0.4044 (0.929)		0.0000
A(5,5)	0.0184	0.6430 (0.047)	0.6932 (0.158)	0.6790 (0.980)		
A(5,6)		0.0559 (0.000)	0.0000 (0.000)	0.1966		
A(6,1)						0.0000
A(6,2)		0.1382 (0.737)	0.1100 (2.134)	0.2351 (2.962)	0.0882 (0.639)	
A(6,3)	1.0763	0.0223 (2.053)	0.0206 (5.615)		0.0758 (0.162)	
A(6,4)	0.0000	0.4393 (4.675)	5.3085 (2.628)	0.7404 (0.912)	0.0468 (0.937)	
A(6,5)	0.0000	2.2694 (5.266)	13.9810 (8.063)	2.5505 (0.548)		
A(7,1)				0.2068 (3.392)		
A(7,2)			0.2745 (5.108)	0.1067 (1.328)		

	Survey					
Subarea	I	II	III	IV	V	VI
A(7,3)		0.4877 (20.950)			0.0843 (0.139)	
Total (E+11)	38.163	35.620 (68.686)	53.689 (36.018)	21.027 (19.104)	6.188 (8.413)	0.146 (1.212)
Mean (E+11)	2.245	1.781 (3.122)	2.557 (1.801)	1.168 (1.061)	0.413 (0.601)	0.009 (0.076)
No. animals/ survey (E+11)	87.55	69.46 (109.27)	99.71 (63.03)	45.56 (37.14)	16.09 (21.04)	0.34 (2.65)

Table 2. Summary table of egg for U.S. scientists' data.

Survey duration	Mid-day of survey (Julian day (X))	No. of station sampled	Total eggs during survey (E+11 eggs)	Number of survey days	No. of eggs sampled/day (E+11)
Feb. 14	45			1	0.001
Mar. 10-16	71	21	70.100	7	10.010
Apr. 16-22	108	35	119.832	7	17.119
Apr. 23-27	114	31	90.430	5	18.086
May 3-9	125	28	40.505	7	5.786
May 10-15	131.5	17	14.564	6	2.427
June 1-23	162	39	0.170	22	0.008

Table 3. The number of eggs present in spawning areas from Soviet survey.

Survey duration	Mid-day of survey (Julian day (X))	Total eggs during survey (E+8)	No. of eggs sampled per day (E+8)
April 10 - May 3	110.5	294.7	12.28
May 10-20	134.0	151.0	13.73
June 1-30	166.0	40.2	1.34

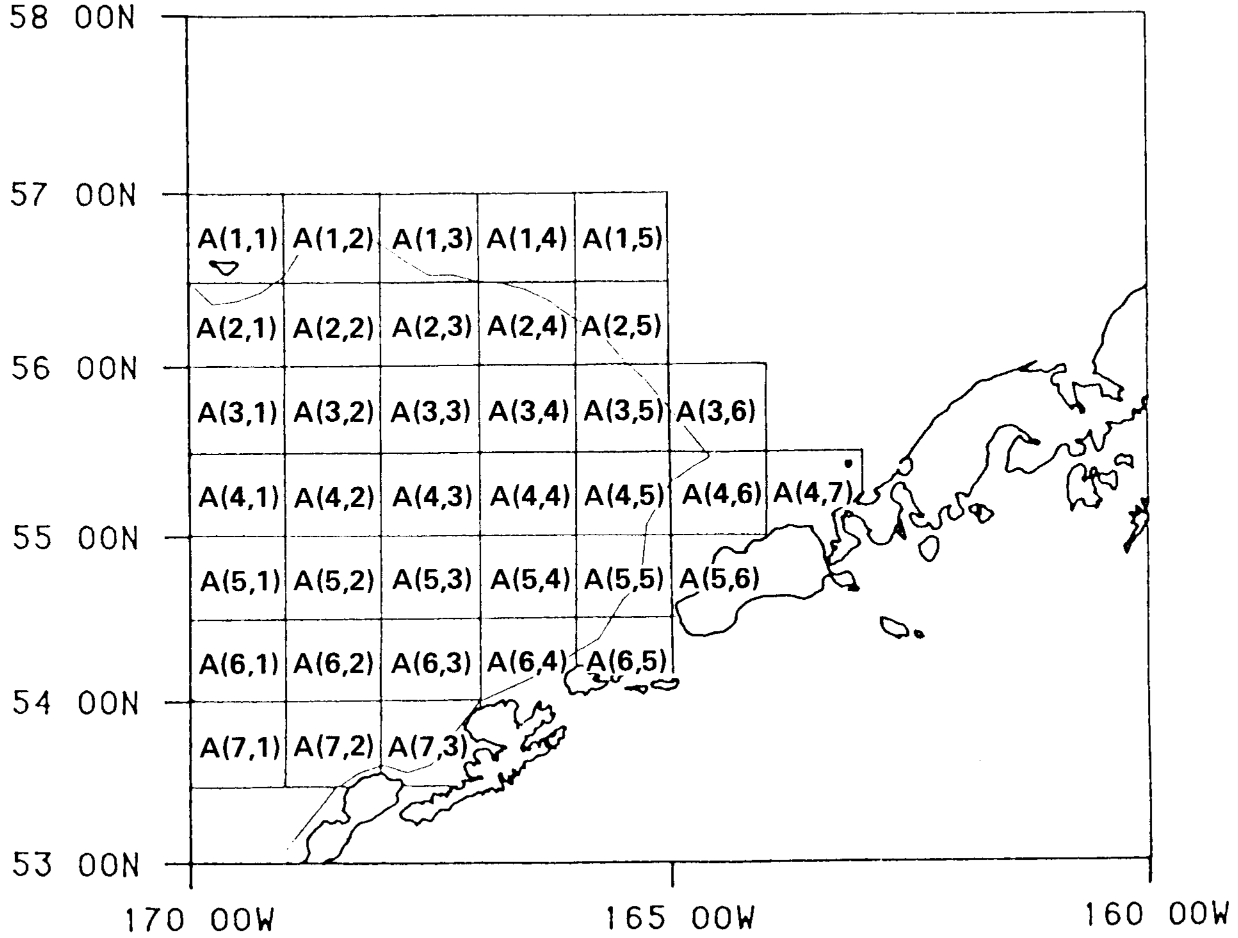

Figure 1. Research area in southeastern Bering Sea. The size of each subarea is 0.5° latitude by 1° longitude. 100 m isobath is included.

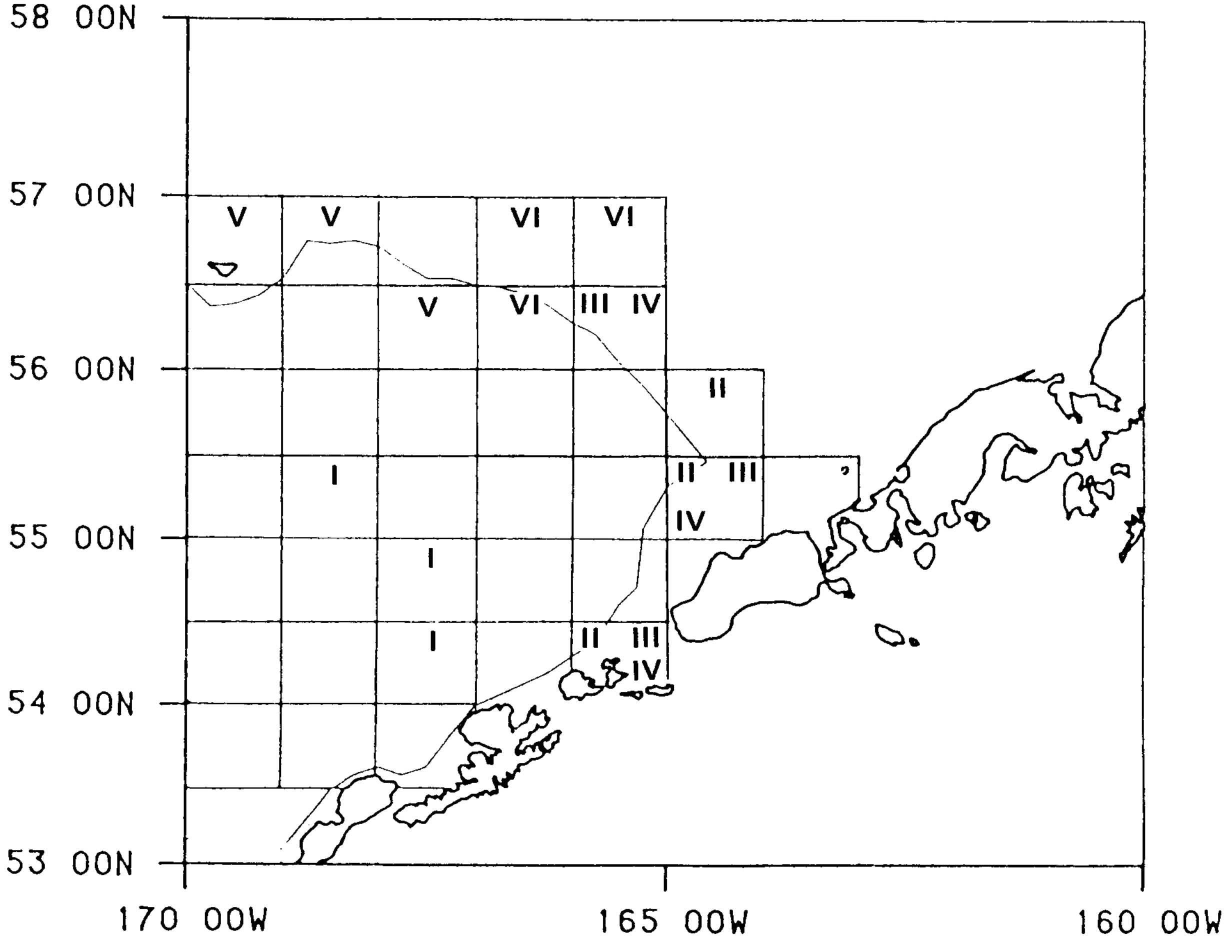

Figure 2. Spatial distribution of pollock eggs and larvae. The three subareas with the greatest densities during each survey are indicated. 100 m isobath is included. (A) Eggs.

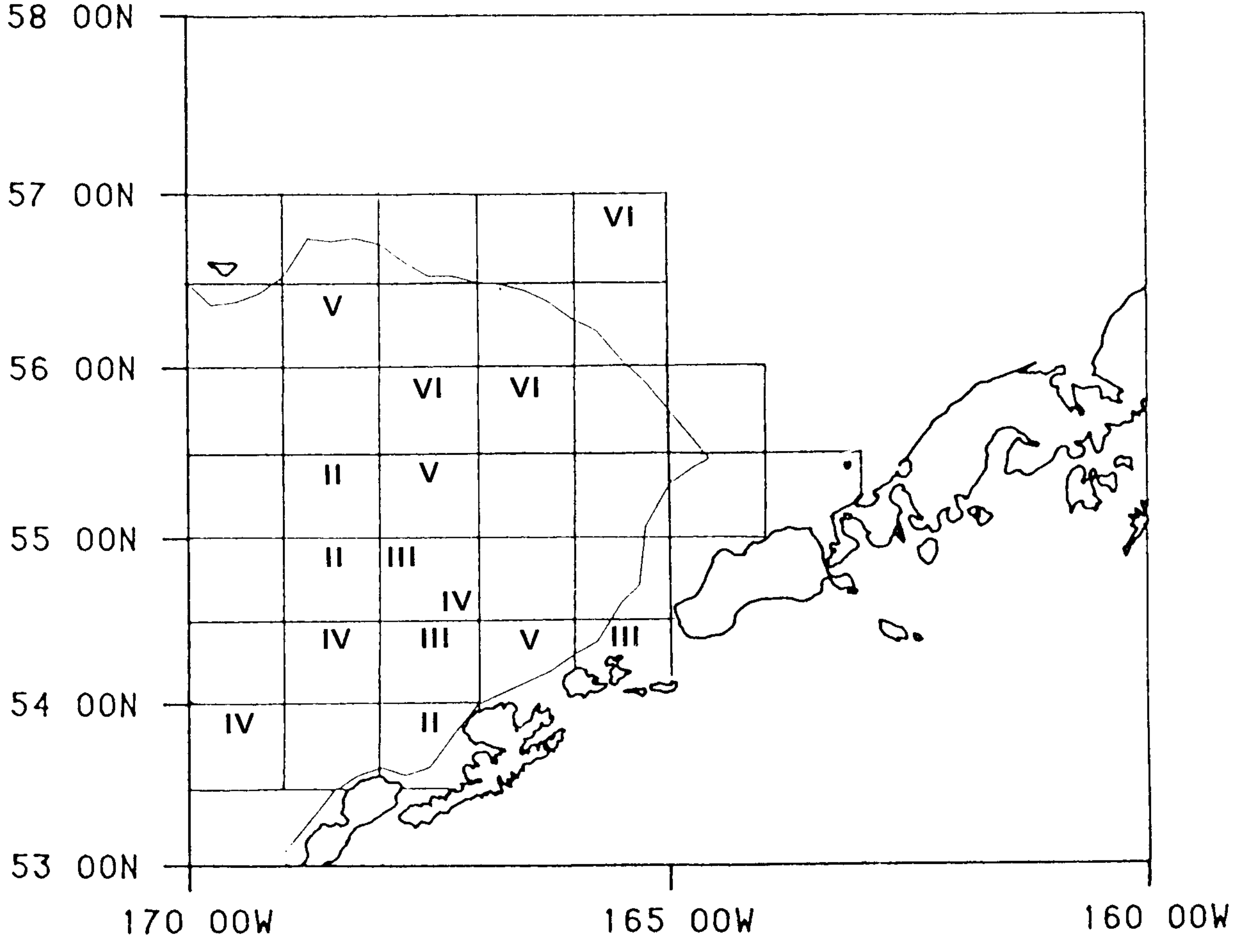

Figure 2. Spatial distribution of pollock eggs and larvae. The three subareas with the greatest densities during each survey are indicated. 100 m isobath is included. (B) Larvae.

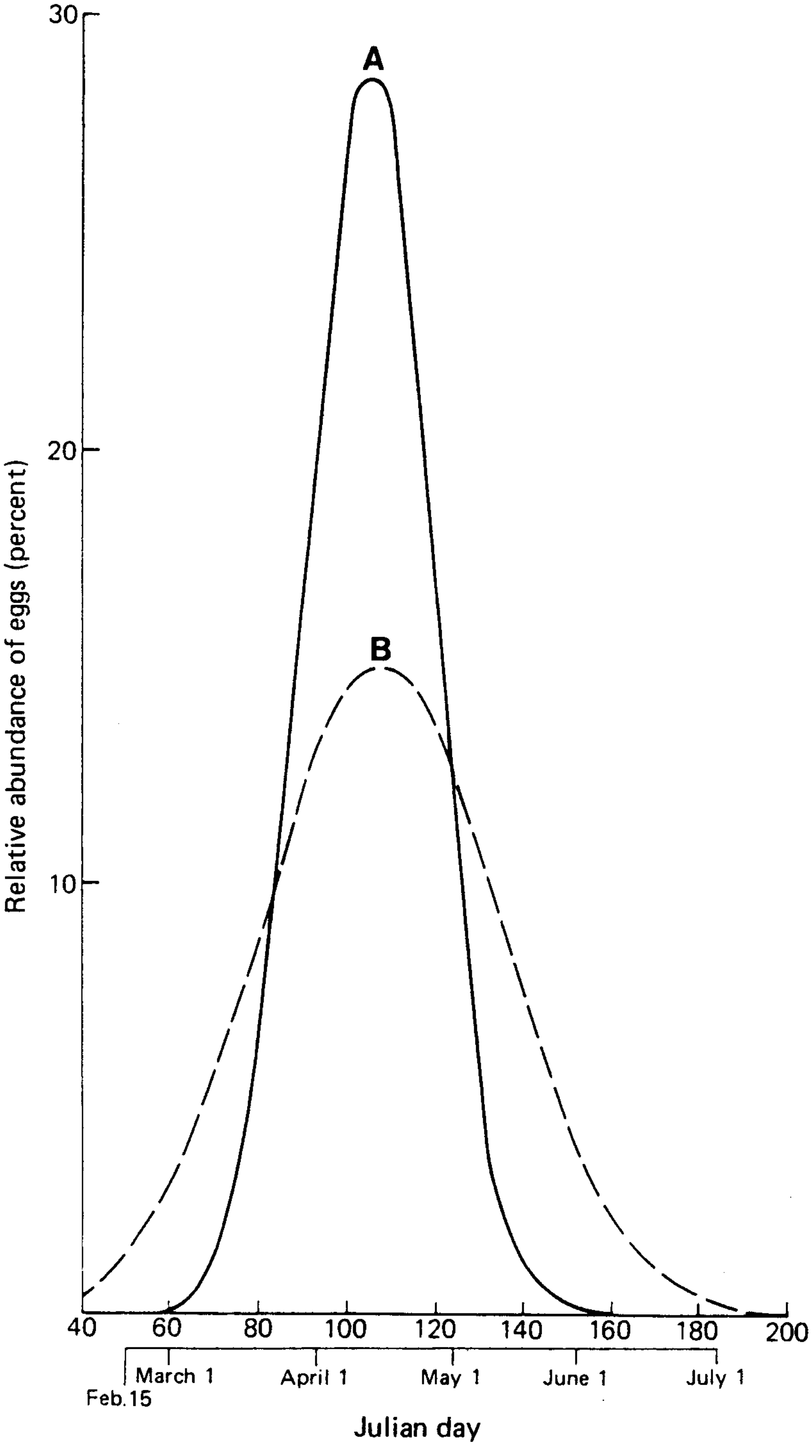

Figure 3. The relative abundance of pollock egg with time. (A) Based on U.S. scientists' data. (B) Based on Moiseev and Bulatov (1978) and Kendall (pers. comm.).

Appendix I: The number of eggs present at each station during each survey.

1) The total number of eggs present at each station during March 10-16, 1978. (Survey I)

Station	No. of eggs sampled during survey	SHF	No. of eggs/m^2
16	203	4.252	86.3156
17	5	5.379	2.6895
18	11	5.980	6.5780
19	1858	5.158	958.3564
20	100	6.340	63.4000
21	6	5.450	3.2700
22	1	5.494	0.5494
23	4	4.876	1.9504
24	1	5.181	0.5181
28	2	5.791	1.1582
29	3	5.817	1.7451
32	1	5.147	0.5147
		Total	1127.0454

SHF (Standard Haul Factor) converts actual catch to numbers beneath 10 m^2 of sea surface.

Total number of egg present in area of concern = (130613.6·1000000/21)·1127.045
= 7.0099·E+12

2) The number of eggs present at each station during April 16-May 15, 1977.*

Station	Survey II			Survey III			Survey IV			Survey V		
		No. of eggs			No. of eggs			No. of eggs			No. of eggs	
	SHF	total	/m²	SHF	total	/m²	SHF	total	/m²	SHF	total	/m²
1	6.55	30	19.635	6.69	427	285.834	6.62	58	38.367			
2	5.73	8	4.584	6.19	11	6.810	6.12	4	2.448	6.46	2	1.219
3	4.99	1	0.499	6.53	1	0.653	5.42	6	3.253			
4	3.92	1	0.392	6.80	3	2.041	5.68	4	2.272			
5	3.04	11	3.347	6.77	6	4.059				5.67	10	5.672
6	4.06	9	3.653	5.96	28	16.685	4.06	10	4.059			
7	5.53	2	1.106	6.54	24	15.698	4.86	19	9.234			
8	5.91	10	5.910	6.28	27	16.956	5.80	23	13.333	Bongo tow aborted		
9	4.73	131	62.553	5.68	678	385.375	5.81	121	70.301			
10	4.77	12	5.725	5.49	9	4.945	4.78	55	26.301			
11	5.20	58	30.154	5.84	58	33.855	5.04	23	11.587			
12	4.34	-	-	6.38	-	-	4.81	15	7.221			
13	5.25	2	1.050	5.78	1	0.578	4.18	5	2.091			
14	5.50	3	1.650	5.96	4	2.385				5.75	16	9.197
15	5.04	14	7.062	5.50	15	8.252				5.50	29	15.936
16	6.00	7	4.202	5.59	14	7.829				5.99	22	13.174
17	5.10	9	4.594	5.91	4	2.363	5.79	3	1.736			
18	5.00	6	2.998	5.87	-	-	5.15	4	2.059			
19	5.38	137	73.706	6.03	372	224.353	5.54	227	125.758			
20	5.20	3	1.561	5.73	-	-	5.49	10	5.485			
21	4.34	2	0.868	4.54	121	54.922	4.52	-	-			
22	5.73	246	140.884	5.50	1123	617.875	4.90	879	430.710			
23	5.82	36	20.941	5.58	25	13.958	5.18	68	35.251			
24	6.38	24	15.314	5.99	8	4.790	5.36	10	5.360			
25	5.98	1741	1041.466				5.08	16	8.120			
26	6.96	682	474.536				5.12	62	31.769			
27	Bongo tow aborted						4.07	54	21.967			
39	5.60	2	1.120							4.70	117	54.943
40	4.86	-	-							5.24	76	39.809
49	6.35	6	3.808				6.48	10	6.479	6.08	4	2.430
50	6.99	19	13.283							7.65	3	2.296
51	6.15	1	0.615							6.85	3	2.055
52	6.30	23	14.485				6.29	3	1.888			
53	6.12	4	2.448							6.01	1	0.601
54	5.64	2	1.128							5.87	2	1.174
55	Bongo tow aborted									6.89	2	1.379
58				5.87	50	29.370				6.37	4	2.546
59				5.89	62	36.537				6.11	43	26.269
60				5.68	27	15.323				6.32	31	19.592
61				5.24	726	380.061	5.65	193	109.045			
62							4.19	142	59.427			
64				6.48	1	0.648				7.07	3	2.122
65				6.06	5	3.032	5.81	5	2.906			
66				6.23	12	7.476	5.63	10	5.633			
		Total =	1965.277			2182.660			984.634			200.486

Total number of eggs present in area of concern: subarea 1; Station No. 1-24
subarea 2; Station No. 25-66

Survey II
subarea 1: $(4.404\text{ E}+10/24)\cdot(19.6350+4.5840+,,,+15.3144) = 7.5673\cdot\text{E}+11$
+) subarea 2: $(5.166\text{ E}+10/10)\cdot(1041.4662+474.5356+,,,+1.1282) = \underline{80.222\cdot\text{E}+11}$
$87.789\cdot\text{E}+11$

Survey III
subarea 1: $(4.404\text{ E}+10/24)\cdot(285.8338+6.8101+,,,+4.7896) = 31.382\cdot\text{E}+11$
+) subarea 2: $(5.166\text{ E}+10/8)\cdot(29.370+36.537+,,,+7.476) = \underline{34.866\cdot\text{E}+11}$
$66.248\cdot\text{E}+11$

Survey IV
subarea 1: $(4.404\text{ E}+10/20)\cdot(38.367+2.448+,,,+5.360) = 17.546\cdot\text{E}+11$
+) subarea 2: $(5.166\text{ E}+10/8)\cdot(8.120+31.769+,,,+5.633) = \underline{12.128\cdot\text{E}+11}$
$29.674\cdot\text{E}+11$

Survey V
subarea 1: $(4.404\text{ E}+10/5)\cdot(1.2912+5.672+,,,+13.174) = 3.9873\cdot\text{E}+11$
+) subarea 2: $(5.166\text{ E}+10/12)\cdot(54.943+39.809+,,,+2.122) = \underline{6.6821\cdot\text{E}+11}$
$10.6694\cdot\text{E}+11$

*They mentioned that the total area of subarea 1 and 2 is 95700 km^2. Because we assumed equal production rate outside this area, we can corrected that number to 130613.6 km^2 by multiplying 1.365 to total egg number present in subarea 1 and 2;

	Survey			
	II	III	IV	V
Amount of egg present (E+11)	119.832	90.430	40.505	14.564

3) Total number of eggs present at each station during June 1-23, 1979 (Survey VI).

Station	No. of eggs/m^2
V01-8	0.7
S 46A	1.3
S 12A	3.4
Total =	5.4

Total number of egg present in area of concern $= (130613.6\cdot 1000000/40)\cdot 5.4$
$= 1.763\cdot\text{E}+10$

Appendix II: The estimation of total egg number produced.

Survey (i)	N_i (E+11) eggs	D_i days
I	70.10	34.5
II	119.83	22.5
III	90.43	7.5
IV	40.51	9.5
V	14.56	14.0
VI	0.17	31.0

$$N = \Sigma((N_i \cdot D_i/a)$$
$$= (6386.805/17.3)E+11$$
$$= 369.18 \cdot E+11 \text{ (eggs)}$$

Appendix III: Total number of pollock eggs and larvae in subarea A(i,j) during specific survey.

EGGS

Subarea	No. of times sampled	No. of times egg caught	Survey (station number)	No. of eggs/ m^2	Total No. of eggs present /subarea
A(1,1)	2	1	II (40)	0	0
			V (40)	39.8088	1.3560 E+11
A(1,2)	5	3	II (39)	1.1196	3.8140 E+9
			V (39)	54.9432	1.8716 E+11
			VI (S47A, S48A)	0	0
A(1,3)	1	1	VI (S45A)	0	0
A(1,4)	2	1	VI (S40A, S46A)	1.3000	2.2142 E+9
A(1,5)	1	1	VI (S12A)	3.4000	1.1582 E+10
A(2,1)	2	2	VI (S50A, S53A)	0	0
A(2,2)	4	2	III (58)	29.3700	1.0005 E+11
			V (58)	2.5460	8.6732 E+9
			VI (S41A, S44A)	0	0
A(2,3)	4	2	III (59)	36.5366	1.2447 E+11
			V (59)	26.2687	8.9487 E+10
			VI (S38A, S39A)	0	0
A(2,4)	5	3	III (60)	15.3225	5.2198 E+10
			V (60)	19.5920	6.6742 E+10
			VII (V01-8, S33A, S34A)	0.2333	0.7949 E+9
A(2,5)	2	2	III (61)	380.0610	1.2947 E+12
			IV (61)	109.0450	3.7147 E+11
A(3,1)	2	1	II (55)	Bongo tow aborted	
			V (55)	1.3788	4.8509 E+9
A(3,2)	5	3	I (22)	0.5494	1.9329 E+9
			II (54)	1.1282	3.9692 E+9
			V (54)	1.1740	4.1304 E+9
			VI (S42A, S43A)	0	0

Subarea	No. of times sampled	No. of times egg caught	Survey (station number)	No. of eggs/m^2	Total No. of eggs present /subarea
A(3,3)	11	7	I (24)	0.5181	1.8228 E+9
			II (14, 15)	4.3557	1.5324 E+10
			III (14, 15)	5.3184	1.8711 E+10
			IV (14, 15)	12.5662	4.4210 E+10
			VI (S08A, S08B S29A, S35A)	0	0
A(3,4)	11	6	I (25)	0	0
			II (16,17)	4.3979	1.5473 E+10
			III (16, 17)	5.0958	1.7928 E+10
			V (16, 17)	7.4546	2.6227 E+10
			VI (S09A, S30A, S31A, S32A)	0	0
A(3,5)	7	6	I (26)	0	0
			II (23, 24)	18.1278	6.3777 E+10
			III (23, 24)	9.3736	3.2978 E+10
			IV (23, 24)	20.3056	7.1439 E+10
A(3,6)	4	4	II (25, 26)	758.0009	2.5822 E+12
			IV (25, 26)	19.9444	7.0168 E+10
A(4,1)	1	1	I (21)	3.2700	1.1576 E+10
A(4,2)	4	3	I (20)	63.4000	2.2444 E+11
			II (53)	2.4484	8.6676 E+9
			V (53)	0.6014	2.1290 E+99
			VI (S13A)	0	0
A(4,3)	13	8	I (23, 28)	1.5543	5.5024 E+9
			II (5, 6)	3.5002	1.2391 E+10
			III (5, 6)	10.3721	3.6719 E+10
			IV (6)	4.0590	1.4369 E+10
			V (5)	5.6720	2.0080 E+10
			VI (S06A, S07A, S28A, S36A, S37A)	0	0
A(4,4)	7	4	I (27)	0	0
			II (12, 13)	0.5251	1.8589 E+9
			III (12, 13)	0.2890	1.0231 E+9
			IV (12, 13)	4.6560	1.6483 E+10
A(4,5)	8	6	I (29, 30)	0.8726	3.0890 E+9
			II (18, 19)	38.3521	1.3577 E+11

Subarea	No. of times sampled	No. of times egg caught	Survey (station number)	No. of eggs/ m^2	Total No. of eggs present /subarea
			III (18, 19)	112.1766	3.9712 E+11
			IV (18, 19)	63.9084	2.2624 E+11
A(4,6)	7	5	I (31)	0	0
			II (21, 22)	70.8759	2.5091 E+11
			III (21, 22)	336.3983	1.1909 E+12
			IV (21, 22)	215.3550	7.6239 E+11
A(4,7)	3	2	II (27)	Bongo tow aborted	
			IV (27)	21.9672	7.7767 E+10
A(5,1)	1	0	VI (S14A)	0	0
A(5,2)	6	3	I (18)	6.5780	2.3577 E+10
			II (52)	14.4854	5.1918 E+10
			IV (52)	1.8882	6.7676 E+9
			VI (S04A, S05A, S27A)	0	0
A(5,3)	10	7	I (19)	958.3564	3.4349 E+12
			II (3, 4)	0.4456	1.5971 E+9
			III (3, 4)	1.3471	4.8282 E+9
			IV (3, 4)	2.7623	9.9005 E+9
			VI (S01A, S02A S03A)	0	0
A(5,4)	8	6	II (7, 8)	3.5078	1.2572 E+10
			III (7, 8)	16.3272	5.8519 E+10
			IV (7, 8)	11.2836	4.0442 E+10
			V (8)	Bongo tow aborted	
			VI (S14A)	0	0
A(5,5)	7	7	I (32)	0.5147	1.8448 E+9
			II (10, 11)	17.9397	6.4298 E+10
			III (10, 11)	19.3996	6.9317 E+10
			IV(10, 11)	18.9442	6.7899 E+10
A(5,6)	3	2	II (20)	1.5609	5.5945 E+9
			III (20)	0	0
			IV (20)	55.4850	1.9659 E+10
A(6,1)	1	0	VI (S26A)	0	0

Subarea	No. of times sampled	No. of times egg caught	Survey (station number)	No. of eggs/m^2	Total No. of eggs present /subarea
A(6,2)	4	4	II (49)	3.8082	1.3816 E+10
			III (65)	3.0320	1.1000 E+10
			IV (49)	6.4790	2.3506 E+10
			V (49)	2.4304	8.8174 E+9
A(6,3)	7	6	I (15, 16, 17)	29.6684	1.0763 E+11
			II (51)	0.6151	2.2316 E+9
			III (64)	0.6476	2.2061 E+9
			V (51, 64)	2.0885	7.5770 E+9
A(6,4)	8	7	I (12)	0	0
			II (1, 2)	12.1095	4.3933 E+10
			III (1, 2)	146.3220	5.3085 E+11
			IV (1, 2)	20.4077	7.4038 E+10
			V (2)	1.2912	4.6844 E+9
A(6,5)	4	3	I (13)	0	0
			II (9)	62.5525	2.2694 E+11
			III (9)	385.3752	1.3981 E+12
			IV (9)	70.3010	2.5505 E+11
A(7,1)	1	1	IV (66)	5.6330	2.0681 E+10
A(7,2)	2	2	III (66)	7.4760	2.7447 E+10
			IV (65)	2.9060	1.0669 E+10
A(7,3)	2	2	II (50)	13.2829	4.8767 E+10
			V (50)	2.2962	8.4303 E+9

LARVAE

Subarea	No. of times sampled	No. of times larvae caught	Survey (station number)	No. of larvae/m^2	Total No. of larvae present (E+10)/subarea
A(1,1)	2	1	II (40)	0	0
			V (40)	12.5712	4.2822
A(1,2)	4	2	II (39)	3.3588	1.1441
			V(39)	14.5576	4.9588
			VI (S47A,S48A)	0	0

Subarea	No. of times sampled	No. of times larvae caught	Survey (station number)	No. of larvae/ m^2	Total No. of larvae present (E+10)/subarea
A(1,3)	1	1	VI (S45A)	0.7	0.2384
A(1,4)	1	1	VI (S40A)	2.7	0.9197
A(1,5)	1	1	VI (S12A)	21.1	7.1874
A(2,1)	2	0	VI (S50A,S53A)	0	0
A(2,2)	4	4	III (58)	35.8314	12.2060
			V (58)	21.6410	7.3722
			VI (S41A,S44A)	0.7	0.2385
A(2,3)	4	2	III (59)	5.3037	1.8068
			V (59)	8.5526	2.9135
			VI (S38A,S39A)	0	0
A(2,4)	5	3	III (60)	0.5675	0.1933
			V (60)	0	0
			VI (V01-8, S33A,S34A)	2.2	0.7495
A(2,5)	2	0	III (61)	0	0
			V (61)	0	0
A(3,1)	1	1	V (55)	11.0304	3.8807
A(3,2)	4	2	II (54)	6.2051	2.1831
			V (54)	14.0880	4.9564
			VI (S42A, S43A)	0	0
A(3,3)	9	7	II (14, 15)	4.3557	1.5324
			III (14, 15)	3.4623	1.2181
			IV (14, 15)	31.7734	11.1780
			VI (S08A, S29A, S35A)	2.93	1.0308
A(3,4)	9	5	II (16, 17)	0	0
			III (16, 17)	0.5907	0.2079
			V (16, 17)	0.8779	0.3089
			VI (S09A, S31A, S32A)	3.5	1.2314
A(3,5)	6	1	II (23, 24)	0	0
			III (23, 24)	0	0
			IV (23, 24)	19.1808	6.7482

Subarea	No. of times sampled	No. of times larvae caught	Survey (station number)	No. of larvae/m^2	Total No. of larvae present (E+10)/subarea
A(3,6)	4	0	II (25, 26)	0	0
			IV (25, 26)	0	0
A(4,2)	3	2	II (53)	342.1639	121.13
			V (53)	7.2168	2.5548
			VI (S13A)	0	0
A(4,3)	11	7	II (5, 6)	3.8558	1.3650
			III (5, 6)	14.5679	5.1572
			IV (6)	35.7192	12.6450
			V (5)	96.4240	34.1350
			VI (S06A, S07A, S28A, S36A, S37A)	0.42	0.1487
A(4,4)	6	4	II (12,13)	0.2171	0.0769
			III (12, 13)	1.4448	0.5115
			IV (12, 13)	28.0048	9.9141
A(4,5)	6	4	II (18, 19)	1.9988	0.7076
			III (18, 19)	3.8155	1.3507
			IV (18, 19)	23.2091	8.2163
A(4,6)	6	1	II (21, 22)	0	0
			III (21, 22)	0	0
			IV (21, 22)	16.66	5.8979
A(4,7)	1	0	IV (27)	0	0
A(5,1)	1	1	VI (S14A)	0.7	0.2509
A(5,2)	5	2	II (52)	469.2010	168.1700
			IV (52)	8.1822	2.9326
			VI (S04A, S05A, S27A)	0	0
A(5,3)	9	7	II (3, 4)	41.7090	14.9490
			III (3, 4)	192.0720	68.8410
			IV (3, 4)	64.1443	22.9900
			VI (S01A, S02A, S03A)	0.3333	0.1195
A(5,4)	6	5	II (7, 8)	107.1016	38.3870
			III (7, 8)	88.2340	31.6240
			IV (7, 8)	25.9135	9.2878

Subarea	No. of times sampled	No. of times larvae caught	Survey (station number)	No. of larvae/ m^2	Total No. of larvae present (E+10)/subarea
A(5,5)	6	5	II (10, 11)	1.2998	0.4659
			III (10, 11)	4.4124	1.5815
			IV (10, 11)	27.3296	9.7953
A(5,6)	3	1	II (20)	0	0
			III (20)	0	0
			IV (20)	2.7425	0.9830
A(6,2)	4	4	II (49)	20.3104	7.3685
			III (65)	58.8208	21.3400
			IV (49)	81.6354	29.6170
			V (49)	17.6204	6.3926
A(6,3)	4	4	II (51)	56.5984	20.5340
			III (64)	154.7764	56.1520
			V (51, 64)	4.4744	1.6233
A(6,4)	7	7	II (1, 2)	128.8466	46.7450
			III (1,2)	72.4442	26.2820
			IV (1, 2)	25.1509	9.1246
			V (2)	25.8240	9.3688
A(6,5)	3	3	II (9)	145.1600	52.6630
			III (9)	222.2444	80.6290
			IV (9)	15.1060	5.4804
A(7,1)	1	1	IV (66)	93.5078	33.9240
A(7,2)	2	2	III (66)	140.7980	51.0810
			IV (65)	36.6156	13.2840
A(7,3)	2	2	II (50)	577.4566	209.5000
			V (50)	3.8270	1.3884

QUANTITATIVE DETERMINATION OF THE EFFECTS OF OIL DEVELOPMENT IN THE BRISTOL BAY REGION ON THE COMMERCIAL FISHERIES IN THE BERING SEA

Edited by

Taivo Laevastu and Frances Fukuhara

Final Report
Outer Continental Shelf Environmental Assessment Program
Research Unit 643

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1. PURPOSE OF THE PROJECT

1.1 The Problem

The objective of this project is to quantitatively analyze the expected impact which might occur as a consequence of hypothetical spill scenarios for PU 3010 (see end of Chapter 1.2). This evaluation is the very first attempt to comprehensively estimate the possible adverse effects to the eastern Bering Sea environment and biota from spills of specified petroleum and volume at designated spill sites. Many data bases and studies on eastern Bering Sea environment and resources are available. Although much of this literature is relevant, since no information was specifically obtained to satisfy the subject, time, and area dimensions of the present study, we can anticipate that our analyses and results may not be as complete or as definitive as we would like. Regardless, the results of these studies may be the only basis for formulating some sensitive operational or socio-economic decisions regarding petroleum exploration and developoment. At the least, it will form a basis for further refinements and extensions of oil impact analyses. Consequently, these studies must be very carefully formulated and executed and apply current and relevant knowledge concerning the effects of oil on ecosystems from past oil spills, model and laboratory studies, and environmental studies of eastern Bering Sea. The findings must be presented in a form understandable to any interested party.

The very copious, marine oil spill literature records a number of instances of the esthetic damage and in some cases the catastrophic annihilation of virtually all intertidal organisms inundated by beached petroleum or fuel oil (e.g., Linden et al., 1983). The tainting of shellfish (e.g., oysters in Brittany from the Amoco Cadiz spill) as a consequence of oil spills is also well documented (Laubier, 1983). We have not, however, seen any documented evidence of the direct detrimental effects of oil on fish or shellfish production (excluding clams) in the marine environment, either from well blowouts or tanker accidents. Some Draconian forecasts of the possible impacts of oil developments on marine fisheries and ecosystems, however, have been presented. These views appear to have resulted from extrapolations of selective laboratory observations and experiments on the effects of hydrocarbons on the physiology, genetics, and mortality of fish (Payne, 1982). The results of past studies have failed to provide any empirical evidence to indicate that any oil spill was responsible for substantial direct mortality to juveniles and adults. Preliminary analyses in our study leads us also to conclude that the spill scenarios under study will have little if any impact on mature fish of eastern Bering Sea.

The preceding highlights one main difficulty in the study of oil spill impacts. As pointed out in the concluding words of the report on Petroleum in the Marine Environment by the National Academy of Science (1983), "The single most significant gap existing to date is our difficulty in transferring the information obtained from laboratory studies to predicting and/or evaluating potential impact of petroleum on living marine resources in the field, especially in the case of spill impact on such commercially important stocks as fish and shellfish."

The possible effects of oil on fishery resources will be evaluated here with numerical ecosystem simulations (models) that are based on validated quantitative laboratory observations as well as on the knowledge of fishery resources in the region. These simulations address four major areas of possible impacts (<u>oil in this report means petroleum hydrocarbons</u>):

1) Effect of oil (from accidents) on fish and shellfish eggs and larvae, and the projection of these effects on exploitable populations years later.

2) Possible effect of oil on adult fish, including effects on crabs and on migrating salmon.

3) Possible tainting of fish by direct exposure and by uptake of oil-contaminated food, and the possible need and extent of area closures in case of accidents.

4) Possible short- and long-term effects of oil on the bottom of the sea on the benthic ecosystem (including demersal fish).

All effects will be evaluated in the perspective of total fishery resources in the region of concern, taking into account the magnitude of errors in estimating the population sizes, the natural fluctuations and mortality of populations, and their seasonal migrations (by life history stages).

Two important subjects will receive somewhat perfunctory consideration in our study, not because we think they are unimportant but because the terms of the study exclude them.

The possible effects of oil on the beaches will be evaluated only in a semiquantitative manner. The emotion-laden problems of possible effects of oil on mammals and birds will be only qualitatively described. With the exclusion of these topics, the impact analysis will be incomplete.

1.2 Principles of evaluation of the possible effects of oil developments

A noticeable quantity of oil in water which might affect marine biota can originate from either a well blowout or from a tanker accident. First order calculations of the concentrations of oil in water originating from large accidents and its distribution in space and time indicate that in order to obtain the concentration of oil in the water or on the bottom which might affect the postlarval fish, and be of noticeable areal extent, the blowout and/or accident must be of considerable magnitude. (Note: The oil on the surface has very negligible effect on marine biota unless it is carried to very shallow water and to the beach.) Therefore, the computations in this project were done with <u>Maximum</u> <u>Effect</u> <u>Conditions</u> (MEC) which are defined as follows:

1) Accident scenarios include the largest plausible well blowout lasting 15 days (and longer) and a large tanker accident releasing most of the oil cargo within 24 hours.

2) The spreading of oil in the water occurs in such conditions of wind, tides, mixed layer depth, and temperature which produce largest possible area of highest possible concentration of water soluble fraction (WSF) of oil in the water (>1 ppm).

3) The blowout/accident occurs during the most unfavorable time with respect to the fishery resources (peak spawning time with maximum aggregation of fish per unit area, and/or peak migration time of anadromous fish).

4) The prevailing conditions affecting the sedimentation of the oil to the bottom are such that highest possible quantity of oil accumulates on the bottom in the shortest possible time.

The effect of oil on the fishery resources (e.g., mortalities, diminished growth, etc.) must be evaluated against the <u>size of the stocks</u> of the fish in the region, taking into account:

a) magnitudes of the errors of the assessment of the stocks;

b) magnitudes of the natural fluctuations of the stocks;

c) size of the fishing and other "natural mortalities" of the stocks.

The degree of possible tainting of the exploitable part of the stock by external exposure and by uptake of oil contaminated food must be quantitatively determined and the area and duration of possible closure of fishing after the accident must be determined, taking into account:

a) the speed of oil depuration by the fish;

b) plausible distances of migration of tainted fish.

The weathered oil which accumulates on the bottom will affect the benthic ecosystem (including demersal fish and crab) in a variety of ways. These effects might be long lasting (up to a few years). The sedimentation of the oil to the bottom is quantitatively simulated (modelled), using the parameters known to affect this process (see Chapter 3). The effects of oil on the bottom will be evaluated with analogs to known and investigated effects, especially from Tsesis spill.

There are other possible effects of accidents which are excluded from this project, such as oil on the beaches and effects of oil on birds and mammals. These possible effects will be estimated, however, at least to the order of magnitude and compared to the oil effects on direct fishery resources.

The following accident scenario has been selected for the project:

1) Well blowout (in 3 locations), 20,000 bbl/day for 5 days. Prudhoe Bay crude oil.

2) Tanker accident (instantaneous spill at 3 locations), 240,000 bbl automotive diesel (refined), released from the tanker at rate of 10,000 bbl/hr.

3) Three locations (Figure 1), 1) off Port Moller, 56°20'N, 161°20'W; 49 m depth; 2) off Port Heiden, 57°10'N, 159°W; 43 m depth; 3) off Cape Newenham, 58°N, 164°W, 43 m depth.

4) Wind direction will be the most frequent for the location. Wind speed and tides will be such that they produce Maximum Effect Conditions (MEC).

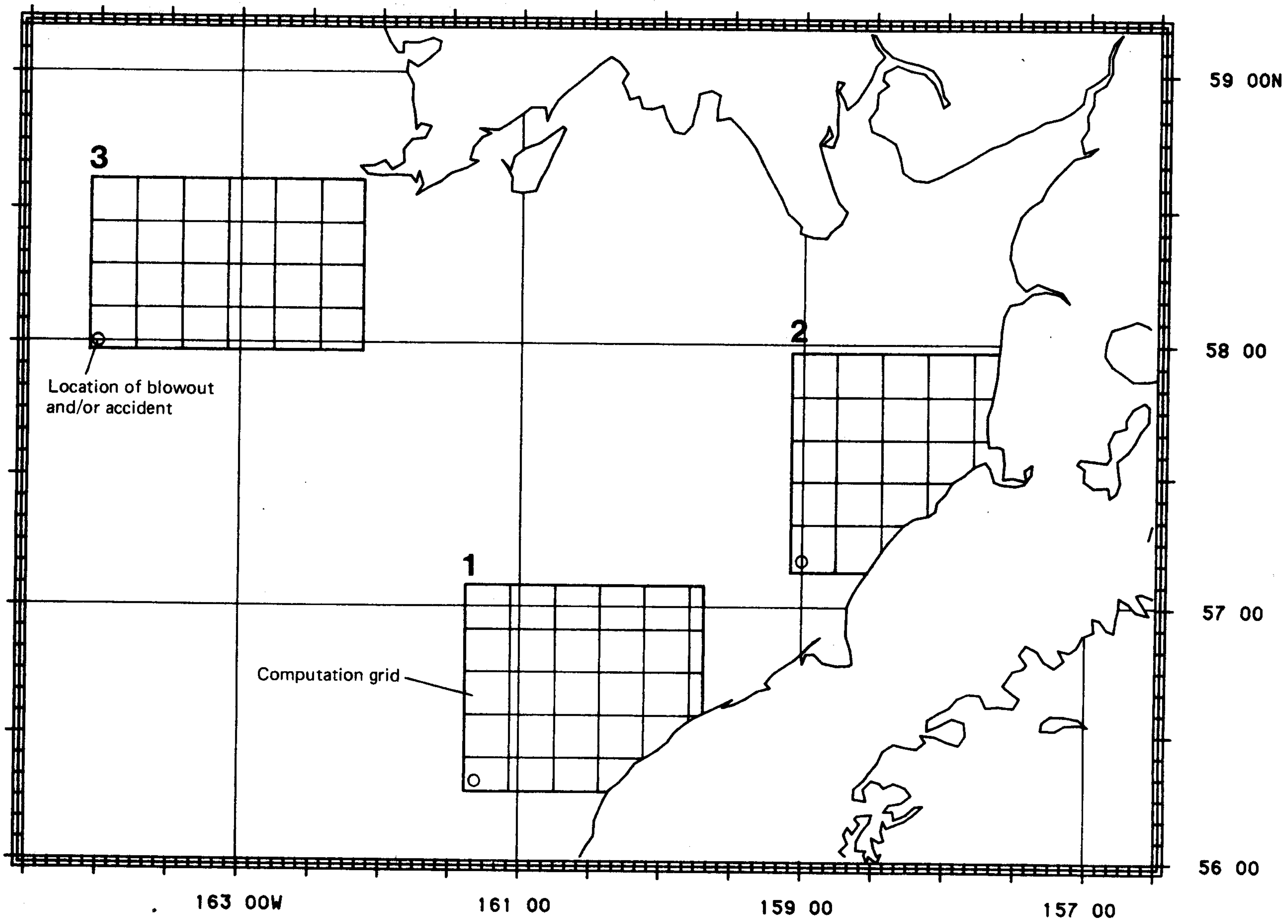

Figure 1.--Locations of hypothetical oil spills, and computational grids in Bristol Bay.

The computation of oil in the water (Water Soluble Fraction, WSF, which includes the oil in solution in molecular form, as well as oil dispersed in the water as very fine droplets) is being computed by Rand Corporation in cooperation with Science Applications Inc. (Liu, 1983).

Other details of the project scenario are given in next chapter under Numerical Methods.

2. METHODS OF THE STUDY AND SOURCES AND NATURE OF DATA

2.1 Numerical methods

Oil in water. The distribution of oil in water from the well blowout and tanker accidents are computed by Rand Corp. in cooperation with SAI*. Some examples of computed distributions are shown in Chapter 2.3. The methods of computations are reported elsewhere (Liu, 1983).

In the present project, we are not concerned with oil on the water surface, as it does not affect fish and/or fisheries to any noticeable degree. The effects of oil in the intertidal zone are estimated in this project only to the order of magnitude, because the Rand models have not as yet quantified the beaching of oil.

We are considering and operating with Water Soluble Fraction (WSF) (see definition above). This may constitute about 10% of the total crude oil present at the surface. In estimating the effects of WSF on marine biota (including fish and crabs), several peculiarities of the oil are taken into account, such as the knowledge that the more toxic volatile (aromatic) hydrocarbons reach their maximum concentration in water about 3 hours after release, but after about 12 hours only about half of their concentration is left (Payne, 1981).

The oil from the tanker accident scenario is considered to be "refined oil". According to Anderson et al. (1974), No. 2 fuel oil and Bunker C residual oil have the same effect on biota when their concentration is 3 to 5 times less than the concentration of WSF crude oil.

Oil on the bottom. About 20 to 50 percent of crude oil released in and/or on the water will ultimately reach the bottom of the sea as "weathered oil". This sedimentation of the oil is affected by the type of oil, temperature, turbidity, turbulence, depth of water, and other factors. The initial

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* Science Applications Inc., La Jolla, CA.

sedimentation (in about the first 15 days) is relatively rapid. However, the sedimentation of all the oil which will ultimately end up on the bottom takes considerable time; thus, it will meanwhile be distributed by currents over a large area, resulting in low concentrations on the bottom of the sea. Higher concentrations of oil on the bottom in shallower water (say <50 m depth) originate from the WSF within about 15 days of the oil addition to the water. Oil will first accumulate in a nepheloid layer near the bottom and will be slowly carried into the sediment, especially by the activity of infauna. This nepheloid layer is moved around with currents near the bottom, effecting the accumulation of it in deepenings (irregularities) in the bottom topography.

The numerical method for simulation of oil on the bottom is described in Chapter 3 of this report.

Oil on the bottom affects the behavior of mobile epifauna (e.g., emigration) and is taken up by sessile epifauna and infauna. The latter two are used as food by demersal fish and crabs. Weathered oil can last in the bottom for relatively long periods (a few years).

Contamination of eggs and larvae by oil. In the case of a well blowout, the eggs and larvae are passively carried into the oiled area by currents and become contaminated (Figure 2). The initial exposure is thus with high concentrations near the blowout. The same current speed and direction, which is used for computation of distribution of oil in the surface mixed layer, is also used for transport of eggs and larvae. Additional eggs and larvae are carried into the oiled area by eddy diffusion. This addition is computed from the area (lateral) extension of the oil concentrations.

The mechanisms effecting the spreading of oil from a tanker accident, complemented with eddy diffusion, are mainly active in exposing the eggs and

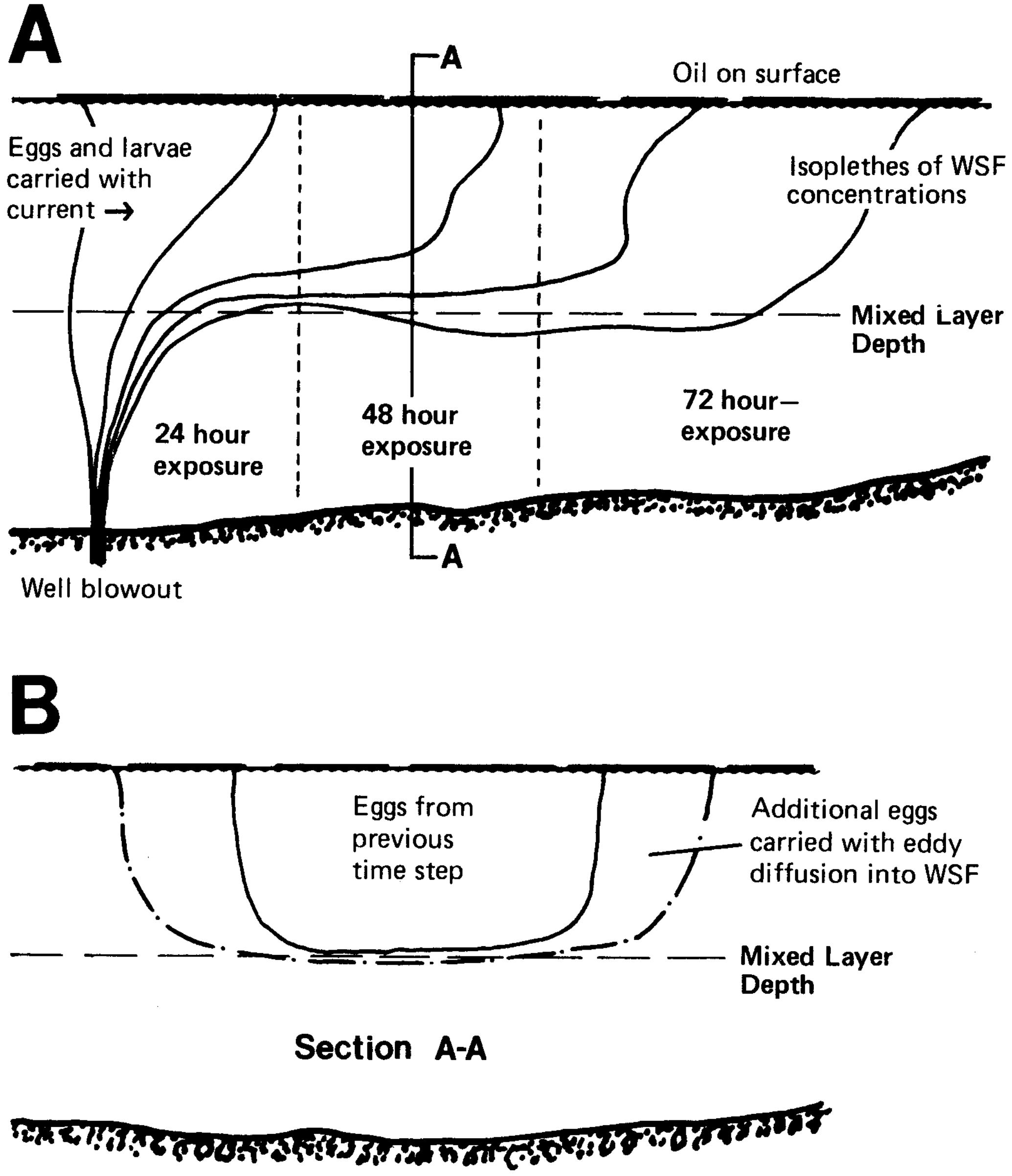


Figure 2.--Schematic presentation of contamination of eggs and larvae by oil. A-Section through an "oil plume". B-Cross-section through above plume (also indicative of contamination by eddy diffusion from an instantaneous source).

larvae to oil originating as instantaneous ("point") source from tanker accidents. These exposures are computed from the expansion of the oiled area with time (Figure 2B).

Empirical data on the spawning and distribution of eggs and larvae in space and time are very limited in the Bering Sea. The MEC conditions for eggs and larvae are computed with the following procedure (schematic presentation see Figure 3): The peak spawning season is estimated for each species. Estimates are also made of the distribution of spawning biomass during peak spawning. Using the data on fecundity of the species and the spawning concentration of biomass during peak spawning, the maximum concentrations of eggs and larvae resulting from 5 day spawning is estimated. These eggs and larvae are assumed to be found in the upper mixed layer. The above assumptions give MEC for the computation of egg and larvae mortality. Examples of the estimated amounts of eggs exposed to different concentrations of oil are given in Table 1.

Egg and larvae mortalities are estimated using data from effect studies (see Table 4 in Chapter 2.4).

The effect of possible egg and larvae mortality on the exploitable population years later is estimated with a linear assumption (see Figure 4), which gives MEC (i.e., the density dependent larval mortalities are neglected). No spawner-recruitment relations have been demonstrated for marine fish. The effects via possible, but uncertain spawner-recruitment relations on coming generations are, therefore, neglected.

Exposure of adult fish to oil. Fish can be affected by external exposure to oil (e.g., uptake of oil through gills). During exposure, fish either can be stationary in respect to general location, while oil moves into the area, or they can migrate through the oiled area (avoidance reaction of fish

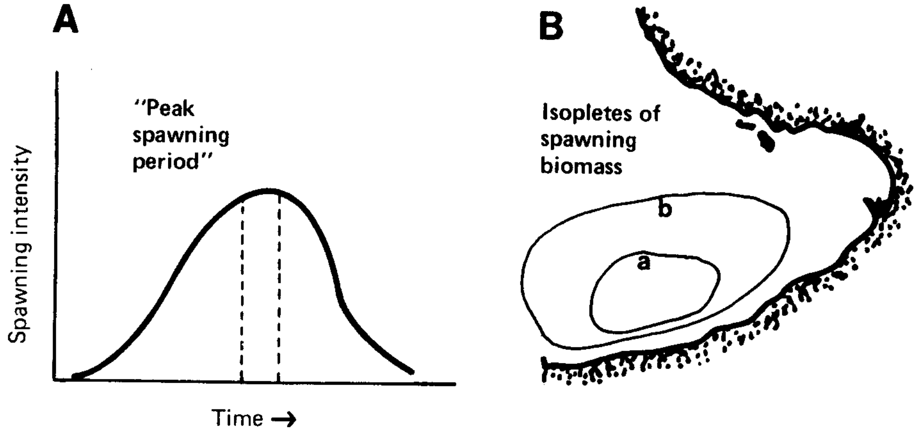


Figure 3.--A-Relative intensity of spawning with time;
B-Distribution of spawning biomass during peak spawning.

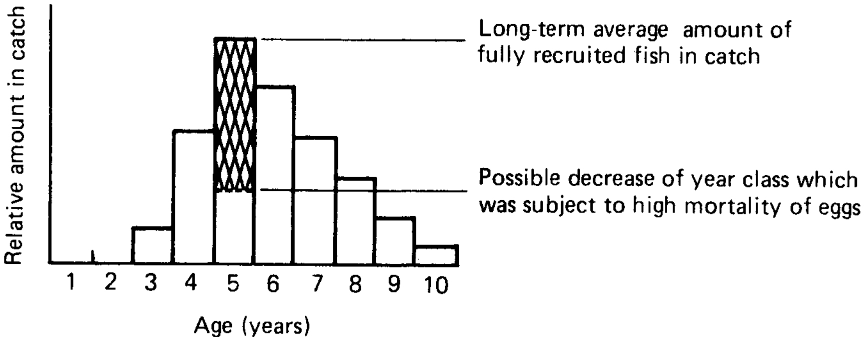


Figure 4.--Schematic presentation of the effect of egg mortality on the exploitable biomass 5 years later.

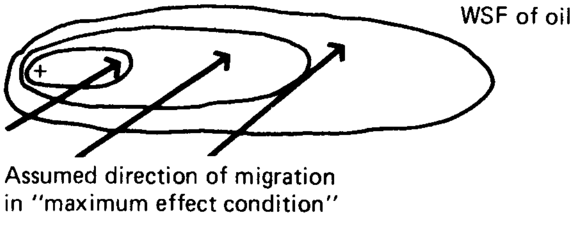


Figure 5.--Migration of fish through oiled area.

Table 1.--Example of computed quantities of eggs exposed to different concentrations of oil.

Area - Port Moller
Species No. 4 850 egg/larvae m^2
Continuous source (well blowout)
Time, days 7

	Number of eggs/larvae exposed (in billions)						
	Exposure time, days						
Oil concentration ppb	1	2	3	4	5	6	7
>100	6.2	-	-	-	-	-	-
50-100	9.6	12.1	11.7	10.6	9.1	6.3	5.2
10-50	3.3	6.0	8.2	7.2	5.5	3.8	2.8
1-10	1.8	11.7	24.6	48.3	28.1	12.4	1.8

is neglected here to achieve MEC). The direction of fish in relation to oil contamination and fish migration speed determine the time of exposure. The selected MEC migration direction is shown on Figure 5. Furthermore, all fish are assumed to migrate between the surface and the MLD.

Examples of the computed exposure of stationary and migrating fish is given in Table 2. Depuration of contaminated fish is a function of time and temperature. The migrating biomass is assumed to be the exploitable stock. The resulting contamination effects are computed by using data in Table 5 (Chapter 2.4).

Adult salmon, as well as smolt, are assumed to migrate through the oil although they may avoid it. The effects of oil on salmon are computed with a special model adapted for this purpose (see Appendix 5).

Uptake of oil with contaminated food. Fish, either "stationary" or moving through an oiled area, will take up pelagic and/or demersal food that may be contaminated with oil (feeding habits see Appendix 6). The magnitude of food uptake can be reduced, however, if the WSF is above a given level of concentration (see Table 6 in Chapter 2.4). For the purpose of MEC, food in the oiled area is assumed to have a bioaccumulation ratio of 50 times the concentration in the environment.

The uptake of oil is computed as mg of oil per kg of biomass. When the fish enter clean water (defined here as < 10 ppb of oil), a time and temperature dependent depuration is computed (see Table 7). Examples of the contamination of fish by food uptake is given in Table 3.

The quantities of fish affected by contaminated food (killed, tainted, etc.) are evaluated on the bases of data in Table 7. Since fish that move through the oiled area can be tainted through the uptake of contaminated food, it is important to know the distance and time that a fish travels before it has depurated below the tainting level. This information can

Table 2.--Example of computed exposure of stationary and migrating fish to different concentrations of oil.

Species 1 - 300 kg/km^2, migrating
Species 2 - 400 kg/km^2, stationary
Time, days 13

Total biomass (kg) exposed to WSF

Oil concentration ppb	Total biomass exposed (kg) Species 1	Species 2
>200	1200	1600
150-200	1200	1600
100-150	8400	11200
50-100	26400	35200
10-50	25200	996800

Table 3.--Example of computed "contamination index" of fish (caused by uptake of contaminated food).

Location: Port Moller, Time Step: Day 15
Species No. 1 - A migrating species (biomass 400 kg/km)
Species No. 2 - A non-migrating species (Biomass 300 kg/km)
(Contaminated biomass is summed over the grid and given in tonnes)

Concentrations in ppb (µg/kg)	Contaminated biomass in tonnes Migrating	Non-migrating
Cont. index greater than 100.000	0.000	0.000
Cont. index 50.000 to 100.000	0.000	0.000
Cont. index 10.000 to 50.000	0.000	0.323
Cont. index 1.000 to 10.000	0.562	2.684
Cont. index 0.100 to 1.000	1.034	11.434
Cont. index less than 0.100	2.678	16.162

be used for estimating the area of a closure to the fishery during an acute oil spill.

2.2 Evaluation of the fishery resources in the eastern Bering Sea and the Resource Thesaurus

The significance of an oil spill on commercial fishery resources cannot be evaluated simply from direct or indirect mortalities. Mortality from petroleum hydrocarbons is a population decrement which is superimposed upon ongoing natural and fishing mortality. Our analysis, therefore, requires not only isolating the component of total mortality which is attributable to petroleum hydrocarbon toxicity, but more importantly, evaluating whether mortality had any effect on population productivity. Such an evaluation requires considerable knowledge of the distribution, migration, life history, fisheries and population dynamics as well as the hydrocarbon toxicology of the affected species and stocks. We must know such things as the magnitude of the resources, their maximum and equilibrium yields, natural fluctuations and projected biomasses, the proportion of biomass distributed in the area of the oil spill.

In addition to the direct data on the magnitudes of the resources, a variety of other information and data are used in the models and in the evaluation of the results. Since the models, as well as results of this project, will be subject to scrutiny by several interested parties, it is imperative that the data used in the computations be available in readily usable form. Therefore, the data on the fishery resources and their environment will be summarized in a Resource Thesaurus. The outline of this thesaurus is given in Appendix 2.

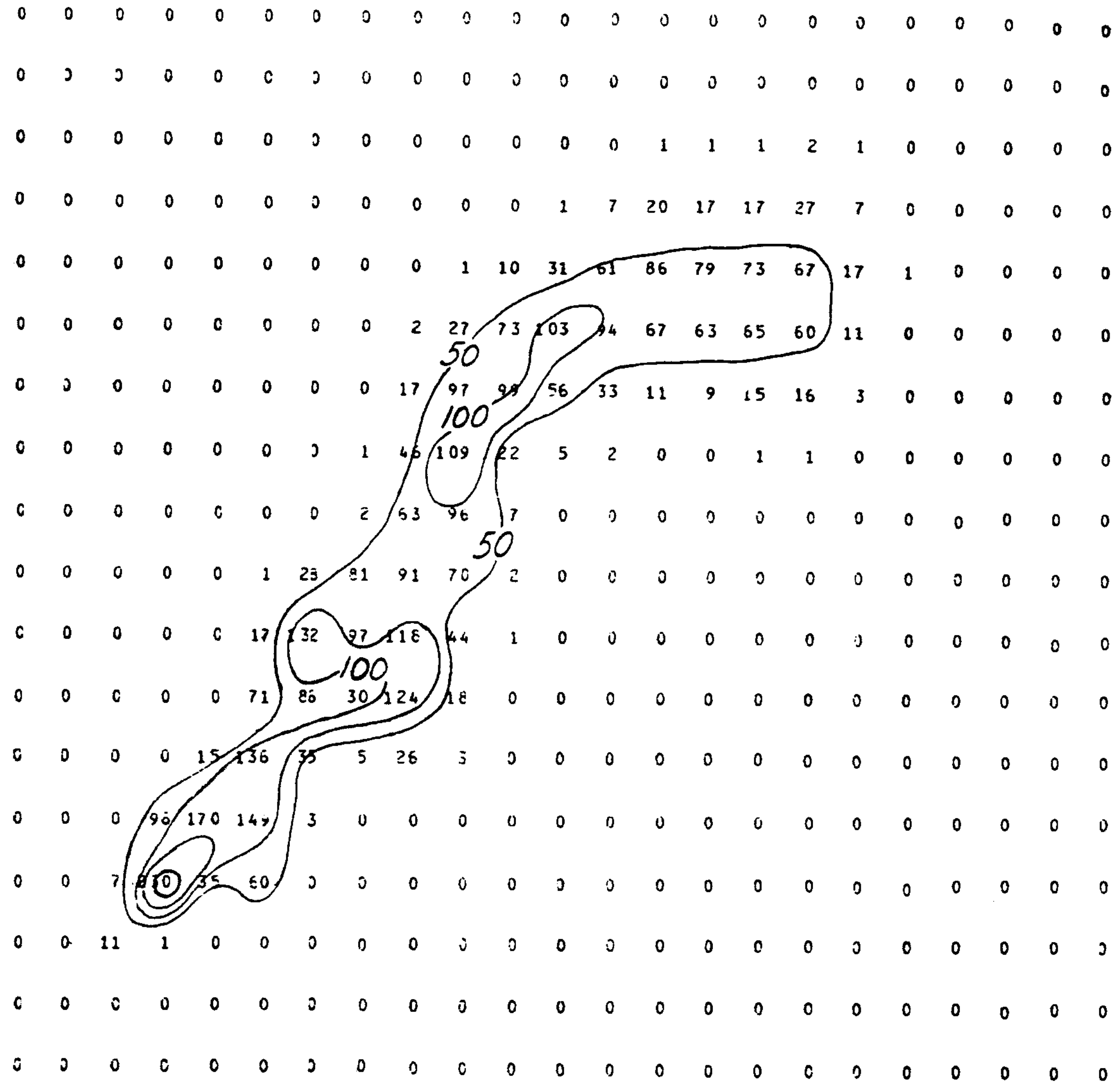
Many data for the project were also derived from the large ecosystem simulations DYNUMES and PROBUB.

2.3 Distribution of oil from blowout and tanker accident.

The computed concentrations of the WSF from a blowout off Port Moller in Bristol Bay were provided by Rand Corporation. The distribution of these concentrations in the fifteenth day after blowout are given in Figure 6. These concentrations are low indeed; only in one grid point (grid size 2 km) the concentration is above 0.2 ppm and at additional seven grid points it is >0.1 ppm. Only 32 grids (128 km^2) have concentrations in excess of 50 ppb. The latter concentration is the lowest at which some mortalities of eggs and larvae have been observed (<50% mortality). No grid point has a concentration above 1 ppm--i.e., the lowest concentration causing any mortality of adult fish.

For the tanker accident, only a qualitative picture of the concentrations has been provided so far (Figure 7). Using this figure and the graphical data provided by Thorsteinson and Thorsteinson (1983), it can be estimated that the resulting initial concentrations of WSF would be 10 to 20 times higher from a tanker accident than from a blowout. It would also cover a more noticeable area, due mainly to the release of a greater quantity of oil in a shorter time period (ca 10 times higher release per unit time than blowout). Even in this case, however, the area of concentrations of 1 ppm and higher will be relatively small, and these concentrations will decay relatively fast.

The hydrocarbon concentrations in water resulting from substantial spills such as in hypothetical spill scenarios PU 3010 are several orders of magnitude less than the concentrations at which toxic effects were seen in laboratory experiments (see next chapter). We would, therefore, expect that the direct effects to the adult fish populations would be small which is contrary to the prevailing popular conception of oil spill damage.



0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	2	1	0	0	0	0	0
0	0	0	0	0	0	0	0	0	0	0	1	7	20	17	17	27	7	0	0	0	0	0
0	0	0	0	0	0	0	0	0	1	10	31	61	86	79	73	67	17	1	0	0	0	0
0	0	0	0	0	0	0	0	2	27	73	103	94	67	63	65	60	11	0	0	0	0	0
0	0	0	0	0	0	0	0	17	97	99	56	33	11	9	15	16	3	0	0	0	0	0
0	0	0	0	0	0	0	1	46	109	22	5	2	0	0	1	1	0	0	0	0	0	0
0	0	0	0	0	0	0	2	63	96	7	0	0	0	0	0	0	0	0	0	0	0	0
0	0	0	0	0	1	25	81	91	70	2	0	0	0	0	0	0	0	0	0	0	0	0
0	0	0	0	0	17	132	97	118	44	1	0	0	0	0	0	0	0	0	0	0	0	0
0	0	0	0	0	71	86	30	124	18	0	0	0	0	0	0	0	0	0	0	0	0	0
0	0	0	0	15	136	35	5	26	3	0	0	0	0	0	0	0	0	0	0	0	0	0
0	0	0	96	170	149	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
0	0	7	810	35	60	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
0	0	11	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Figure 6.--Concentrations of WSF of crude oil in ppb in upper 15 m after 15 days from a well blowout. (Grid size 2 km.)

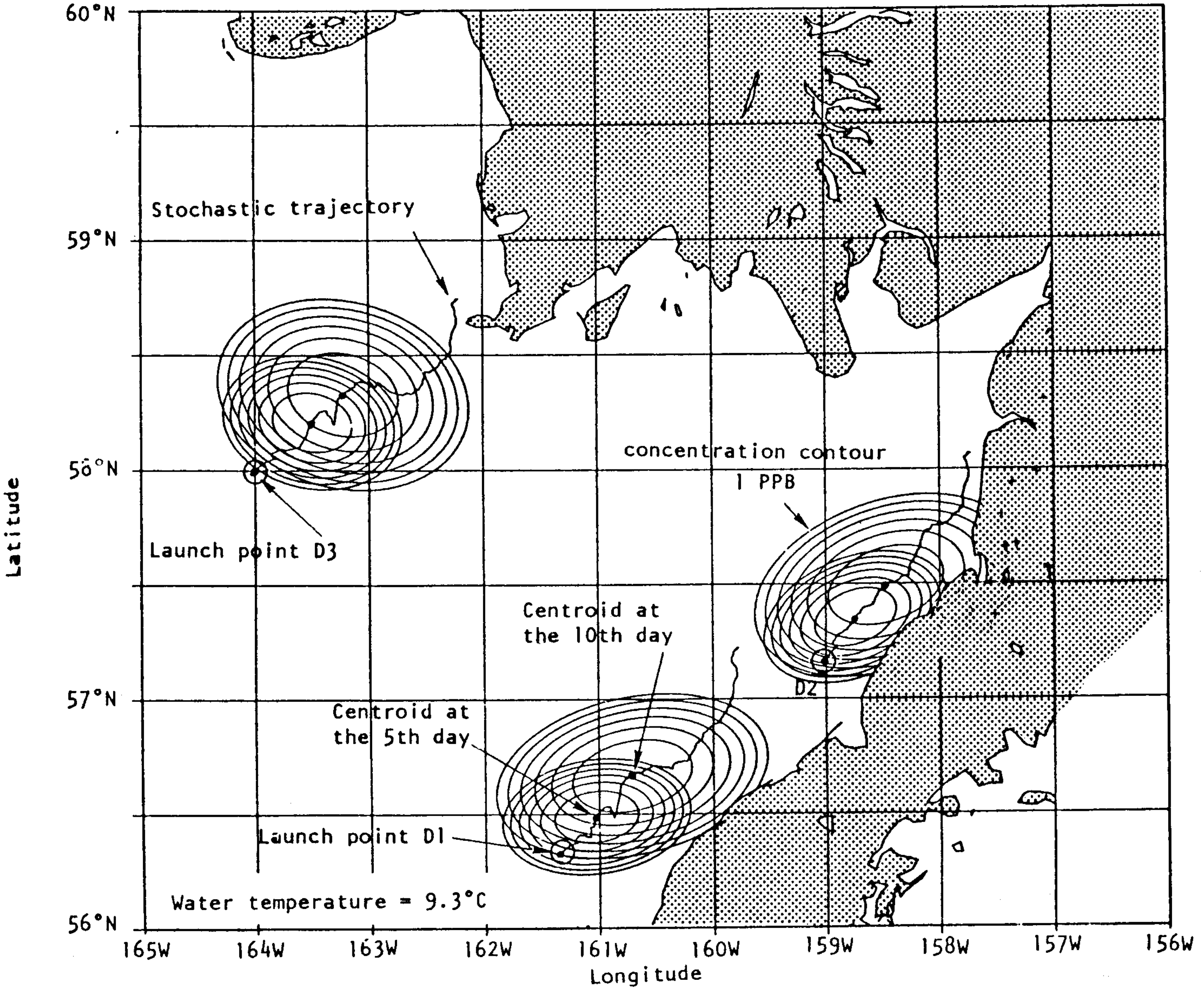


Fig. 7-- Surface oil concentration for the 5th and the 10th day after the instantaneous release of 200000 bbls of Prudhoe Bay crude oil from three hypothetical spill sites. The movements and the shape of the dispersed oil is governed by the cumulated effects of local residual circulation, tidal excursion, vorticity, factors related to viscosity and surface tension (Liu 1983).

2.4 Review of the results of past effect studies.

A voluminous amount of literature are available on the laboratory studies of oil effects on fish and other aquatic biota. The corresponding reports on field studies are few, and their results often inconclusive.

The evaluation of the very variable results of the effect studies is difficult indeed; and their application to real field conditions even more difficult, as has been pointed out in the recent report from the National Academy of Science (see Chapter 1.1).

Main problems with the effect studies are:

1) Most of the studies have been carried out with WSF concentrations two to four orders of magnitude (100 to 10,000 times) higher than would occur with the greatest plausible accident. (Obviously lower concentrations showed little or no effects).

2) Different components of hydrocarbons have been used in these studies.

3) Very different methods of exposure have been used.

4) Treatment of species has been variable, and predominantly small and juvenile fish have been used.

5) Reporting of experimental conditions has often been incomplete.

6) Interpretations of results have often been qualitative.

After review of numerous effect studies, some quasi-quantitative criteria were developed for summarization of the results. The evaluated essentials of the oil exposure results pertaining to the present project are summarized in Tables 4 to 7. Some interpolation of these tables is necessary, and details will be provided in the final report when the results will be presented and evaluated.

Table 4.--Percent of fish eggs and larvae affected by exposure to different concentrations of WSF of crude oil.

WSF concentration, ppm	0.01 to 0.1			0.1 to 1			1 to 10		
Exposure time, h	24	48	96[1]	24	48	96	24	48	96
Lethal		40	60	50	80	100	80	100	100
Hatching and development impaired, other sublethal effects[2]	20	60	80	80	100	100	100	100	100

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[1] Effect after 96 hours remains unchanged (oil is becoming "weathered", etc.

[2] In computation of the effects these amounts are included in mortalities (there will be slower growth and higher vulnerability to predation). Reduced molting in crab larvae are included as sublethatl effects.

[3] The effects of different concentrations of oil on zooplankton are comparable to the effects on eggs and larvae.

Table 5.--Percent of population of marine fish and other fauna affected by exposure to different concentrations of WSF of crude oil. (Obs. in estimating the effects using laboratory experiments it is assumed that "total aromatics" constitute 30% of WSF.)

WSF concentration, ppm[1]	0.1 to 1			1 to 10			10 to 100[4]		
Exposure time, h	24	48	96[5]	24[6]	48	96	24	48	96
Pelagic fish									
Lethal			10	20	40	70	60	80	100
Sublethal[2]		10	30	50	70	100	80	100	100
Demersal fish									
Lethal					20	40	30	60	100
Sublethal				30	50	80	50	80	100
Crustaceans, (epifauna)[3]									
Lethal						30	20	60	100
Sublethal					20	40	20	50	80
Infauna[3]									
Lethal							20	50	80
Sublethal						10	30	80	100

[1] Concentration on the bottom refers to concentration of "weathered oil" in the nepheloid layer near bottom.

[2] Sublethal effects include impairment of growth , fin rot, etc. (delayed effects). (Sublethal effects start at ca 25% concentration of lethal concentrations.) (Sublethal effects do not result in mortality.)

[3] Pelagic crustaceans (incl. larval stages) are included in zooplankton. Epifauna includes some molluscs whereas burrowing molluscs are included in infauna.

[4] Most lethal effects are in this range of concentrations. The effects are often species specific. Values given in this column refer to ca 25 ppm.

[5] Effects after 96 h can be considered as those at 96 h exposure.

[6] Concentration of ca 2 ppm can cause fish to avoid polluted areas (2-3 ppm inhibits salmon migrations).

Table 6.--Effects of various concentrations of WSF of crude oil on the food uptake and tainting of marine fish and depuration times of these fish.

WSF concentration, ppm[1/]	0.1 to 1			1 to 10			10 to 100		
Exposure time, h	24	48	96[6/]	24	48	96	24	48	96
Pelagic fish									
Food uptake impaired[2/]		10	30	20	40	80	100	100	100
Tainting[3/]					20	40	60	80	100
Depuration, days[4/]					2	4	8	10	15
Demersal fish[5/]									
Food uptake impaired			10	10	30	70	40	70	100
Tainting				10	30	50	80	100	100
Depurtation, days				2	4	7	12	15	20

1/ Concentrations on the bottom refer to "weathered oil" in the nepheloid layer.

2/ Values given are percentage of reduction of food uptake.

3/ Tainting refers both to degree and to percentage of population with unacceptable taste and odor.

4/ Depuration is dependent on temperature (longer time required at lower temperatures). Depuration refers to "detainting" of flesh.

5/ Demersal fish includes also clams and crabs.

6/ After 96 hours the effects are considered to remain the same as at 96 hours.

Table 7.--Effects of the uptake of contaminated food by fish (including crabs)

Contamination index[1/]	0.5 to 5	5 to 10	10 to 50	50 to 100	<100
Lethal			10	30	50
Sublethal[2/]	10	30	60	100	100
Tainting[3/]		10	30	60	100
Depuration, days[4/]		4	8	12	16

1/ Contamination index = food uptake (BWD) * concentration of oil in environment (ppm) * 50 (mean bioaccumulation factor) * 0.75 ("retention factor") = mg/kg. The contamination is accumulative. Depuration is computed linearily on contamination index.

2/ Sublethal effects include: slower growth, fin rot, tumors, etc.

3/ Tainting is dependent on accumulation of hydrocarbons in the body (50 ppm and higher).

4/ Depuration is dependent on tempenature (slower in lower temperatures).

3. WEATHERED OIL ON THE BOTTOM

The task concerned with the effects of oil on the bottom has become one of the more important parts of this project. It has also demanded considerable effort for completion, since there exists a scarcity of both data and past investigations on this subject. The notes below present a tentative review of the progress in this subject.

3.1 Sedimentation of oil.

PONAs (paraffins, olefins, and napthas) constitute about 70% of crude oil; the rest (30%) are aromatics. Of the total oil, about 10% is WSF. It is estimated that more than half of the PONAs will ultimately end up as "weathered oil" on the bottom of the sea. The settling of oil is greatly affected by fine silt and clay in the water (suspended mineral matter). In addition, the settling of oiled phytoplankton (especially diatoms) and zooplankton fecal pellets also deposit oil on the sea bed. Although these pathways have been described, documentation of the principal processes of sedimentation and its quantification is generally lacking, particularly in subarctic waters.

Sediment loading in Bristol Bay can be about 40 g $m^{-2}day^{-1}$. According to Baker (1983), this sedimentation can bring down about 10 mg $m^{-2}day^{-1}$ oil, if oil concentration is 1 ppb. (Maximum value in shallow water - 30 mg $m^{-2}day^{-1}$). Oil sedimentation rates of 3 mg $m^{-2}day^{-1}$ have been observed elsewhere.

Besides the amount of mineral suspension in the water, sedimentation of the oil is also dependent on temperature, depth, presence or absence of thermocline, vertical circulation, and obviously on the concentration of oil in the water.

Only part of the oil sedimentizes within about the first 15 days after the accident. The remaining oil sedimentation takes place slowly and is thus distributed over large areas, resulting in low concentrations on the bottom.

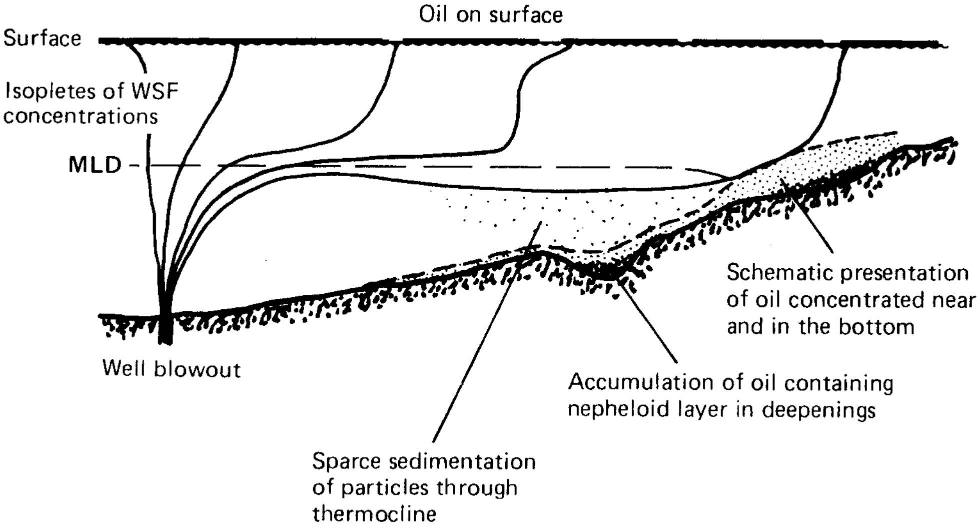

Figure 8.--Schematic presentation of sedimentation of oil.

The following tentative procedure and formulas have been empirically derived for computation of oil on the bottom:

The program simulates the distribution of oil on the sea bottom from the concentrations of WSF (emulsified and dissolved oil in the water) (S).

Deposition and accumulation of oil is made a function of turbulences (waves) which is approximated with wind speed (W, m/sec), depth (D, m), time (presented with the count of 12-h time steps, K), and the presence or absence of the thermocline Potential Layer Depth (PLD) in any given grid point.

1. Instantaneous source ("tanker accident")

   a) PLD not present (shallow water, mixed from surface to bottom). No oil sedimentation to the bottom in first 12 hours. The amount of oil (A0) on the bottom in time step t is:

   $$A0 = A0_{t-1} + pS_t * F_s$$

   p is a tuning factor, dependent of the units of S

   $$F_s = (0.0013W + 0.062/D^{0.7}) * TK_s$$

   TK is time factor. K = 2 to 6 (incl.):

   $$TK_s = K/(6.5 - 0.7K)$$

   K = 6 and larger:

   $$TK_s = K/(3 + 0.2K)$$

   b) PLD (Thermocline) present at the grid point (no oil to the bottom in first 24 hours).

   $$A0_t = A0_{t-1} + pS_t * F_d$$

   p is a tuning factor, dependent of units of S

   $$F_d = (0.00065W + 0.038/D^{0.7}) * TK_d$$

   $$TK_d = K/(6 + K)$$

2. Continuous source ("blowout")

c) Depth is shallower than PLD (shallow water). (No oil to the bottom in first 12 h.)

Formula same as 1a, except distance from the source factor (DF) is added:

$$A0_t = A0_{t-1} + S_t * F_s * DF$$

$DF = DIS(km)/15$

d) Depth is greater than PLD (thermocline present). (No computation in first 24 hours.)

Formula same as 1b, with the same distance factor as 2c added.

3. Oil on the bottom is assumed to form loose, floculous deposit. This deposit is moved 45° to the right of the mean current in the water, which has a speed 5% of that in the water.

4. The time step in the computation is 12 hours. The grid size is the same as used in the simulation of oil in the water (2 km).

5. The decay (degradation) of oil on the bottom from the previous time step is computed from the fourth 12 h time step on, before computing the sedimentation. The decay is a function of depth (D,m) and temperature (T,°C): (if $K-3 \geq 0$, bypass).

$$A0_{t(12)} = A0_{t-1}\, e^{-(t+d)}$$

$$t = T^{2.7} * (2 \times 10^{-5})$$

$$d = 0.015\sqrt{D}$$

$A0$ is computed in a 12-h time step. If a 24-h time step is used, the computation is repeated.

3.2 Fate of oil on the bottom.

Initially the weathered oil accumulates near the bottom in a nepheloid layer. It is nearly impossible to adequately characterize this layer in the ocean with conventional sampling tools. The nepheloid layer moves around with currents near the bottom and will accumulate in depressions. The burrowing animals carry the oil from the nepheloid layer into the sediment, where there is an accumulation of concentration of oil in the sediment. Some laboratory tests show a 1000 fold increase of concentration over that in water.

The weathered oil can persist in the sediment for a year or more. The loss of hydrocarbons from the sediment is a function of sediment type. Sand is found to lose ca 70% of oil during a month, whereas silt loses only ca 10% during the same time period. Biodegration of the oil in sediment is dependent on microbial fauna and temperature, with its rate known to more than double (ca 2.5 times) with a 10°C rise of temperature.

3.3 Effects of oil on the bottom on demersal fish and benthic ecosystem.

It is generally accepted that oil on bottom represents the most serious long term effects of oil spills. The main concern with oil on the bottom is its effects on demersal fish and the benthic ecosystem.

Linden et al. (1979) found that the effects of oil from the Tsesis spill were small and short lived on the pelagic ecosystem. However, the changes caused by oil in the benthic ecosystem were large; mobile macrofauna was drastically reduced, there was increased mortality in ostracods, and the biomass of bivalves (*Macoma baltica*) increased greatly. Amphipods, a very important fish food, were depressed and still are far below prespill levels (Elmgren, personal comm.). Flatfishes who fed on *Macoma* had 50 ppm concentration of hydrocarbons one year later.

Fletcher et al. (1981) have shown that heavily oiled sediment (2300 to 4500 ppm) remained toxic to winter flounder during the summer one year later, whereas during the winter there were no mortalities of the flounder (much reduced feeding during winter). Aged, oiled sediments had no effect on the feeding rate of the flounder.

Oil on the bottom may not directly kill a given predator, but may indirectly reduce the growth and productivity of the population by selectively reducing an important prey population. Conversely, there has been some evidence of an increase in a clam population due perhaps to the oil imposed reduction of predators (Conan 1977, Amoco Cadiz). Ecosystem impact not involving prey-predator relationships have also been documented. For example, reproduction in Baltic herring was significantly reduced as an aftermath of the Tsesis spill. The mortalities were thought not to be attributable to the direct effects of oil on eggs. Rather, it was the consequence of a large mortality and decrease in amphipods which ordinarily graze on fungi growing on fish eggs, thus preventing fungal damage.

Summary of opinions on the effect of oil on the bottom by the world's foremost experts on this subject are found in Appendix 7.

The effects of oil on the bottom will be quantified in the final report of this project.

4. PRELIMINARY QUALITATIVE RESULTS AND OTHER OBSERVATIONS

4.1 Contamination and tainting of migratory and sedentary adult fish.

1. The concentrations of WSF of the oil resulting from the largest possible accidents is several orders of magnitude lower than the concentrations used in laboratory effect studies. The possible effects of oil in the marine environment are, therefore, evaluated in terms of Maximum Effect Conditions (MEC) as defined in this report.

2. The effects of major accidents during the oil development on the Bering Sea continental shelf on adult fish appear to be small indeed, mainly because the concentrations of oil in the water would be low. The effect of a major tanker accident will be larger (but still below the error in a resource estimate), because higher concentrations might result and refined oil is more toxic than crude oil.

3. There may be some tainting of more sedentary fish under a part of the oil trajectory, but the area of possible closure seems to be relatively small and the closure time relatively short.

4. Of the salmon species chinook, pink, and chum salmon smolt may be affected to some degree if the accident occurs during the most unfavorable time.

5. The largest effect of accidental release of oil might be on the spawning and spawn of herring if the accident occurs near the major herring spawning beaches and during the short spawning time. Such a coincidence might be rare.

6. There may be some impact on the egg bearing females of king and Tanner crab since they carry eggs and remain in northern Aleutian shelf area almost all year round.

4.2 Effects of oil on eggs and larvae, and on the recruitment to exploitable stock.

1. The effects of oil on eggs and larvae will start at oil concentrations of 50 ppb and above. The maximum amount of annual egg production killed in MEC is, in general, less than 10% (and smaller).

2. The effect of egg and larval mortality will show up as a possible reduction in the exploitable stock 3 to 8 years later when the affected year class is recruited to the fishery. The reduction of exploitable stock would then be only a few percent, as there are several year classes in the exploitable stock. Thus, this reduction is about an order of magnitude smaller than the plausible error in stock assessment.

3. In extremely rare circumstances the oil may affect a year class of herring eggs (see 4.3).

4. Some quantity of egg-carrying female crabs and instar stage of crab juveniles might be affected by the oil near or on the bottom.

4.3 Environmental and local conditions for maximum impacts.

1. More than 30% of the oil on and in the water will ultimately sedimentize to the bottom. In the area of the "oil plume" during the first 15 days, the sedimented oil will form concentrations high enough to affect the species composition of the benthos and may cause some tainting of more sedentary demersal fish and crabs.

2. The oil effect on the bottom can last longer than a year. The decay of the oil on the bottom is determined by the type of sediment and by temperature.

3. The largest effects of oil could be caused by beaching oil during the spawning of herring. It is possible that more than half of a year class can be killed and the annual harvesting of roe might fail. Such an event can be considered quite rare.

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APPENDIX 1

Personnel of the Project

Principal Investigators:

Marasco, Richard; NWAFC - Fisheries economics and management.

Laevastu, Taivo; NWAFC - Ecosystem simulation (modeling), sedimentation of oil.

Researchers and Other Personnel:

Bax, Nicholas; CS (contr.) - Salmon migrations, effects on smolt and adult salmon.

Fukuhara, Francis; U of W (contr.) - Fishery resources, effects of oil on fish and benthos.

Gallagher, Arthur; NWAFC - Simulation of feeding and uptake of oil as contaminant.

Goiney, Bernard; NWAFC - Laboratory technician.

Gregory, Marjorie; U of W (contr.) - Typist.

Hayes, Maureen; U of W (contr.) - Research assistant (computers, fisheries biology).

Ingraham, James; NWAFC - Environmental data; computations.

Kendall, Arthur; NWAFC - Fish eggs and larvae.

Kim, Suam; U of W (contr.) - Fish eggs, spawning.

Livingston, Patricia; NWAFC - Fish food and feeding habits.

Miyahara, Robert; NWAFC - Environmental and biological data.

Pola-Swan, Nancy; NWAFC - Simulation of migrations, contamination of fish; computation of egg/larvae mortalities.

Rabe, Kevin; CS (contr.) - Computer (production).

APPENDIX 2

THESAURUS OF THE ANALYZED AND DIGESTED DATA ON THE ENVIRONMENT AND BIOLOGICAL RESOURCES OF EASTERN BERING SEA PERTINENT TO HYPOTHETICAL OIL SPILL

F. Fukuhara

Evaluating the impacts of hypothetical oil spill scenarios for PU3010 requires the development and application of a series of predictive models. First, the direction, dimensions and fate of the simulated spills must be estimated. Hydrocarbon concentrations in the water column, their sedimentation and temporal persistence must be estimated. The communities of plants and animals inhabiting the polluted time-space must then be determined. Finally, the lethal and sublethal effects on the productivity and marketability of the commercially important species of fish and shellfish must be estimated.

The credibility of the assessment from this complex sequence of estimation procedures (which are described only in broad and simplistic terms above), depends not only upon the rationality of the simulation models applied, but also on the quality of data bases for certain essential aspects of the environment and resources and on the applicability of experimental results on the physiological effects of hydrocarbons on the eastern Bering Sea biota. Current knowledge regarding the life history, productivity, and dynamics of the principal resources in the study area is also required because the significance of oil imposed mortality must be evaluated in the context of ongoing natural and fishing mortalities to these stocks.

In short, the impact evaluation of PU3010 requires the synthesis of a considerable volume of environmental, biological, and fisheries information collected over the last two decades or more in the eastern Bering Sea. These data, and in most cases their analyses, are copiously documented in data files and in unpublished and published reports of several agencies and institutions concerned with Bering Sea fisheries management or environmental research. Also, although it is difficult to extract many generalizations from past, large oil spill accidents, there are some analogies which have some interpretive value to the present study. Also, for lack of an alternative for directly estimating or assessing certain oil spill related phenomenon such as sedimentation and biological effects of hydrocarbons in the eastern Bering Sea setting, it is necessary to refer to certain empirical observations from past oil spill studies and laboratory studies for insights into formulation of predictive models and conclusions therefrom.

PURPOSE OF THE "RESOURCE THESAURUS": The purpose of this analyzed and digested report is to consolidate and summarize from these diverse sources, information specifically relevant to the following aspects of these oil impact studies:

1. the environmental factors which will determine the transportation, dimensions, dispersal, and persistence of oil spilled at Port Moller, Port Heiden, and off Cape Newenham;

2. the fisheries, species and stocks, life history, and quantity or biomass which might inhabit the spill area, and;

3. the lethal and sublethal impacts of oil on certain commercially important resources and important components of their food webs.

OUTLINE OF THESAURUS

1. Physical Factors Affecting the Transportation, Evaporation, Dissolution, Emulsification, Sedimentation, and Weathering of Crude Petroleum and Automotive Diesel Fuel Spilled off Port Heiden.
    A. Characteristics of Prudhoe Bay crude oil and automotive diesel fuel and the climatological, geological, meteorological, and oceanographic features which affect the dispersal, persistence, and effect on the Bering Sea benthos.
        1. Characteristics of Prudhoe Bay crude
            a. Chemical composition (hydrocarbons and polar compounds)

- b. Physical characteristics (boiling point, viscosity, sp. gravity, etc.)

2. Characteristics of automotive diesel fuel (outline similar to above).
3. Transportation and weathering process of petroleum spill
    - a. Slick formation and transportation (oil viscosity, sea state, wind force and direction, surface and tidal currents).
    - b. Weathering Process
        - i. Evaporation
        - ii. Emulsification
        - iii. Dissolution
        - iv. Photo oxidation
        - v. Microbial degradation
        - vi. Residue
    - c. Sinking and sedimentation
        - i. Physical processes
            - a. entrainment of particles in vertical currents
            - b. adsorption of oil particles to suspended mineral matter (e.g., clay)
        - ii. Bio-physical processes
            - a. oiling and killing of plankton, sinking of carcasses
            - b. ingestion of oil and sinking in fecal pellets of zooplankton
4. Climatological, geological, meteorological, and oceanographic characteristics of E. Bering Sea which modify the dispersal, persistence, and biological effects of spilled petroleum.

a. Climate and weather

  i. Subarctic seasonal characteristics

    a. seasonal duration and variation of sunlight (relevant to photo-oxidation and phyto-plankton production)

    b. monthly mean air temperatures (and range) and seasonal and annual variations

    c. seasonal and annual variation in sea ice cover

    d. monthly ranges and means of precipitation

    e. monthly or seasonal wind duration and speed

b. Geology

  i. Major embayments, lagoons, and shoreline characteristics of the north side of the Alaska Peninsula and Bristol Bay rim.

  ii. Major drainage systems of southeastern Bering Sea (say, Kuskokwin R. and eastward-southward to False Pass).

  iii. Bathymetry and substrate type of eastern Bering Sea.

c. Oceanographic characteristics

  i. Tidal currents (height and period), specifically in eastern Aleutians and Bristol Bay areas.

  ii. Surface current patterns (monthly, seasonal, and annual variability).

  iii. Monthly mean surface temperatures.

  iv. Monthly mean salinities.

  v. Vertical circulation and seasonal turnover or mixing of water column.

    a. thermocline (stability and depth)

    b. speed and depth of vertical currents

vi. Water chemistry
  a. nutrient salts
  b. suspended inorganic matter
  c. naturally occurring, biogenic and petrogenic hydrocarbon concentrations on bottom and in water column.

II. Eastern Bering Sea Biota (commercially important species, prey, predators, and symbiotic organisms) most likely to be impacted by Spill Scenarios PU3010.

A. Food Web, Feeding Rates and Ecosystem Associations
  1. Basic productivity of southeastern Bering Sea
  2. Important herbivores (decapod larvae, copepods, amphipods, mysids, etc.)
  3. Epifauna and infauna (amphipods, echinoderms, clams, worms, etc.)

B. Fish and shellfish of the southeastern Bering Sea benthos
  1. Demersal fishes
    a. General discussion
      i. Commercially important species and their distribution
      ii. Overall fisheries - production and value (products, fleets, processing facilities, work force)
    b. Walleye pollock
      i. Distribution and life history
        a. life history groups
        b. spawning season and migrations
        c. feeding migrations
      ii. The commercial fishery
        a. catch
        b. effort
        c. biomass estimate (virtual pop. est.; fish. data)
        d. annual and inst. fishing mortality
        e. status of stock (MSY, EY, etc.)

iii. Biomass and population estimates from trawl surveys
    a. number and proportion of total population in subareas 1 and 45 as estimated from 1975 and 1979 surveys and seasonal variations
iv. Recent and projected biomass estimates, status of stocks, and trends in productivity and abundance
v. Sensitivity of egg, larvae, juveniles, and adults of pollock to water soluble hydrocarbon fractions
    a. deformities and debilitation
    b. mortality

c. Yellowfin sole
(outline similar to that for pollock)

d. Pacific cod
(outline similar to that for pollock)

e. Rock sole
(outline similar to that for pollock)

f. Alaska plaice
(outline similar to that for pollock)

g. Longhead dab
(outline similar to that for pollock)

h. Arrowtooth flounder
(outline similar to that for pollock)

i. Pacific halibut
(outline similar to that for pollock)

2. Anadromous and transient species (salmon and herring)
    a. General discussion of salmon and herring

- b. Salmon
    - i. Species, distribution, abundance of spawning stocks, and life history
        - a. smolt and juvenile distribution and ecology
        - b. route and timing of spawnings migration
        - c. food habits and predators
    - ii. Fisheries and natural mortality
        - a. targeted and incidental high seas catch
        - b. estimated growth and mortality
    - iii. Sensitivity to oil pollution
        - a. avoidance
        - b. debilitation and delay of migration
        - c. mortality to smolts and adults
- c. Herring
    - i. Distribution and abundance of spawning stocks and life history
    - ii. Spawning, feeding, and juvenile migration routes and timing
    - iii. Fisheries and natural mortality
    - iv. Food habits and predation
    - v. Sensitivity to oil pollution
        - a. direct effects on herring
        - b. indirect effects on food items
        - c. pollution of spawning environment or mortality to symbiotic organisms

3. Crabs
    - a. General discussion
        - i. Species and their distribution
        - ii. Fisheries and their values

b. Red king crab

  i. Distribution and life history

    a. life history groups

    b. spawning season and migrations

    c. feeding migrations

  ii. Numbers and proportion of total population in Subarea 1 as estimated from 1975 and 1979 surveys and seasonal variations

    a. eggs and larvae (fecundity, development of pre-benthic stages)

    b. pre-recruits

    c. exploitable population

  iii. Recent and projected biomass estimates and trends in productivity

  iv. Sensitivity to water soluble fractions of hydrocarbons

    a. deformities and debilitation

    b. mortality

c. *Chionecoetes bairdi*

  i. Distribution and life history

    a. life history groups

    b. spawning season and migrations

    c. feeding migrations

  ii. Numbers and proportions of total population in Subareas 1 and 4 as estimated from 1975 and 1979 surveys and seasonal variations

    a. eggs and larvae

    b. pre-recruits

    c. exploitable population

  iii. Extant and projected biomass estimates and trends in productivity

  iv. Sensitivity to water soluble fractions of hydrocarbons

    a. deformities and debilitation

    b. mortality

4. Clams
    a. General discussion of clams as latent resource
    b. Species, distribution and abundance
    c. Significance in food chain
    d. Reaction to benthic poil pollution
        i. Mortality
        ii. Collectors, concentrators, and transmitters of benthic, hydrocarbon residue
5. Food Chain Organisms
    a. Phytoplankton
    b. Zooplankton (amphipods, copepods)
    c. Hydroids
    d. Worms
    e. Barnacles
    f. Non-commercial crabs and molluscs
    g. Echinoderms

III. Summary Conclusions Relevant to Simulation Analyses for Estimating Impacts of Oil Spill Scenarios PU3010
    A. Formulation of operational hypotheses regarding the transportation, dissolution, evaporation, weathering, and sedimentation of petroleum from spill scenarios PU3010.
        1. lateral transport
        2. chemical changes with time
        3. sedimentation
        4. seasonal variations

    (above discussion based upon literature from Canyon, Cadiz, and Tsesis episodes as well as intuitive and simple "common sense" derivations of oil composition and environmental data in Section I - A through C.

B. Estimates of the exploitable biomass of species of commercial value which can be expected to occur in Subareas 1 and 4S based on past survey and commercial fishery data.

    1. maximum and minimum biomass
    2. seasonal variation
    3. estimated rates of migration
    4. spawning aggregations?

C. Estimates of the expected quantities of pre-adults to occur in Subareas 1 and 4S.

    1. eggs
    2. larvae
    3. juveniles

D. Discussion of oil pollution induced perturbations to the food web.

E. Anticipated effects on various organisms of different concentrations of volatile hydrocarbons based upon past spill and experimental observations.

IV. Effect of oil on coastal spawners.

V. Literature Cited.

VI. Literature Reviewed but not Cited.

APPENDIX 3

Special Computer Programmes and their Subroutines

1) PROBUB - Prognostic Bulk Biomass model

An extensive simulation model for the dynamics of the whole ecosystem in the Bering Sea and Gulf of Alaska. Model area is divided into 9 regions. The model contains numerous subroutines. It is used for a variety of purposes, such as computing the fishery resources (equilibrium biomasses), determination of the effects of fishing, natural fluctuation studies, etc.

2) DYNUMES - Dynamical Numerical Ecosystem simulation model for the Bering Sea

This model is similar to PROBUB (above), except it uses a computational grid (i.e., resolution in space). It allows, therefore, to compute, in addition to the computations with PROBUB, also migrations and any spatial effects of fishing pollution and other location specific phenomena.

3) DEMOIL - "Demersal Oil"

A programme for quantitative testing of the effects of accidents during oil developments.

Control programme sets input parameters, counts time steps, and calls all subroutines. The distribution of oil in the water can be read from tape or created with the subroutine CUROIL.

Subroutine CUROIL can create different currents fields (tidal and wind driven). It computes the distribution of the oil from either a continuous or a point source. The same subroutine is used to move and disperse oil near the bottom with prescribed currents.

Subroutine OILBOT computes the oil to the bottom in various conditions affecting the sedimentation. The oil is also allowed to decay with time, depending on the temperature.

Subroutine EGGLAR moves the eggs and larvae into the oil field (either with currents or with eddy diffusion) and computes the amounts of eggs exposed to different concentrations, as well as their exposure times.

Subroutine FIEXPO computes the quantities of either stationary or migratory fish exposed to different concentrations of oil.

Subroutine CONFOO computes the uptake of contaminated food by fish, and the depuration of contaminated fish.

Subroutine PRIMFS is for printing of various output fields in two dimensions.

4) WFL/OCSEAP - A programme for detailed computation of exposure of eggs, larvae, and fish to oil. Oil distribution in the water and on the bottom is read from the tape.

Subroutine FEDOIL is for detailed computation of uptake of oil hydrocarbons with contaminated pelagic and demersal food. Depuration (decay of hydrocarbons) of fish is also computed. Outputs give the "contamination index" which is the amount of hydrocarbons in the biomass (mg/kg).

Subroutine MIGR is for migrating the fish with different speed and direction through oiled areas and for computation of the exposure.

APPENDIX 4

Commercially Important Species Potentially Impacted at Sites in Hypothetical Spill Scenarios

Port Heiden (57°10'N 159°W)

Herring (eggs, juveniles and adults)
Sockeye (juveniles and adults)
Chinook (juveniles and adults)
Pinks and chum (juveniles and adults)
Yellowfin sole (juveniles and adults)
Pollock (eggs and larvae)
Halibut (juveniles)
Pacific cod (adults and larvae)
Pacific plaice
Greenland turbot } slight
Flathead sole
Rock sole
King crab (*P. camchatica*)
Tanner crab (*C. bairdi*)
Clams
    Surf clams (Spisula)
    Alaska tellin
    Cockles
    Razor clams

Cape Newenham (58°N 164°W)

Herring (juveniles and adults)
Sockeye (juveniles and adults)
Chinook (juveniles and adults
Pinks and chum (juveniles and adults)
Yellowfin sole
Pollock (slight)
Halibut (juveniles)
Pacific cod (larvae, juveniles)
Pacific plaice
Greenland turbot (slight)
Flathead sole (slight)
C. opilio
C. bairdi
Korean hair crab
Surf clams

Appendix 4 (cont'd.)

Port Moller (56°26'N 161°20'W)

Herring (eggs, juveniles, and adults)
Sockeye (juveniles and adults)
Chinook (juveniles and adults)
Pinks and chum (juveniles and adults)
Yellowfin sole (juveniles and adults)
Pollock (eggs and larvae)
Halibut (juveniles)
Pacific cod (larvae and adults)
Pacific plaice
Greenland turbot
Flathead sole } slight
Rock sole
Arrowtooth flounder
King crab (*P. camchatica*)
Tanner crab (*C. bairdi*)
Clams
  Surf clams (Spisula)
  Alaska tellin
  Cockles
  Razor clams

APPENDIX 5

Review of the model for estimation of the effects of oil developments on salmon

Nicholas Bax

This modelling exercise is limited to sockeye salmon (*Oncorhynchus nerka*), one of the four species of Pacific salmon utilizing the Bristol Bay area. The three remaining species (*O. gorbuscha*, *O. keta*, and *O. tshawytscha*) are strongly associated with nearshore habitats during early marine residence but this period could not be studied as the effects of the oil spill scenario on these habitats was not included in the oil trajectory model.

Rogers (1978) summarizes, and Eggers and Rogers (1978) present, the available data on the numbers (or indices) of sockeye salmon emigrations from the four principal river systems in Bristol Bay, and present the temporal distribution of these emigrations. Rogers (1978) lists the estimated adult returns to the coastal area by brood year to those four rivers, and Burgner (1980) illustrates its temporal variability. These data sets are the basis for this modelling exercise.

The model for the simulation of emigration of the smolts through Bristol Bay is complete, although the data sets require updating to include data not available to Rogers in 1978. The model is an adaptation of one proposed by Eggers and Rogers (1978), and simply moves the smolts through Bristol Bay at a rate dependent on their length. The inputs of smolts to this model are the stream counts or indices on the four major rivers, adjusted by river system to total 10 times the total adult coastal return from that brood year. The parameters and assumptions for this model were extrapolated from mark and recapture data presented by Straty (1974) and include the following:

1) Juvenile sockeye grow very little while migrating through the inner Bay (out to Port Heiden); growth begins when they leave the inner Bay.

2) Marked Ugashik sockeye salmon smolts doubled in size from a mean of 107 mm in 8 weeks (2 mm/d). Marked Wood River smolts increased by 50% from a mean of 85 mm in 4 weeks (1.5 mm/d). A value of 1.75 mm/d was used for the growth rate of smolts in outer Bristol Bay in this model.

3) Straty (1974) estimates the amount of time taken by smolts from each of the four river systems to move through the inner Bay. These estimates correspond to an average rate of travel of 0.7 body lengths/sec. This value was used in the model.

The results for this model for the years 1967-1970 are presented as the numbers of smolts from each river system passing Port Heiden (Figs. 1-4). The values presented include no allowance for daily mortality between the time the smolt count was taken and their arrival at Port Heiden, and thus are valid for their relative values only.

It is apparent that there are considerable variations in abundance at Port Heiden within and between years. The available data sets will be analyzed to estimate the maximum proportion of any smolt emigration that would be subject to the presented oil spill scenario at Port Heiden. With the assumptions in the model this will be an overestimate, because no allowance is made for temporal diffusion; a comparison of the temporal spread of the smolts as they leave the rivers (Rogers 1978) with that of the temporal spread of marked smolts in the Bay (Straty 1974) indicates a threefold increase in temporal spread over this period.

The model evaluating the effects of the presented oil spill scenario on the adult salmon will be formulated in the same manner as presented above, but using the annual time of arrival of the adults at the Bristol Bay fishing grounds as the principal data base and Straty's (1975) estimates of adult migration rates.

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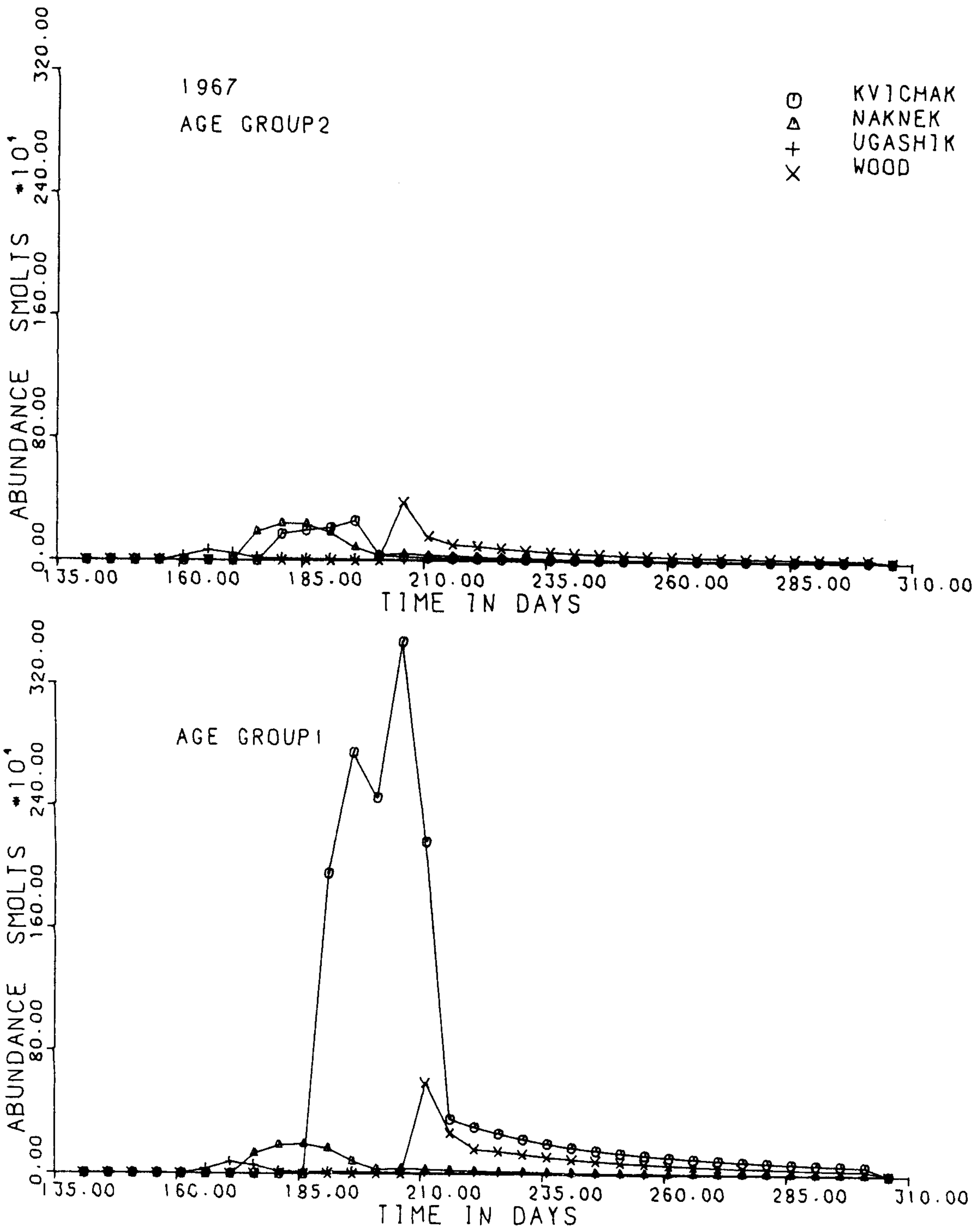

Figure 1.--Projected relative abundance of sockeye salmon smolts at Port Heiden in 1967.

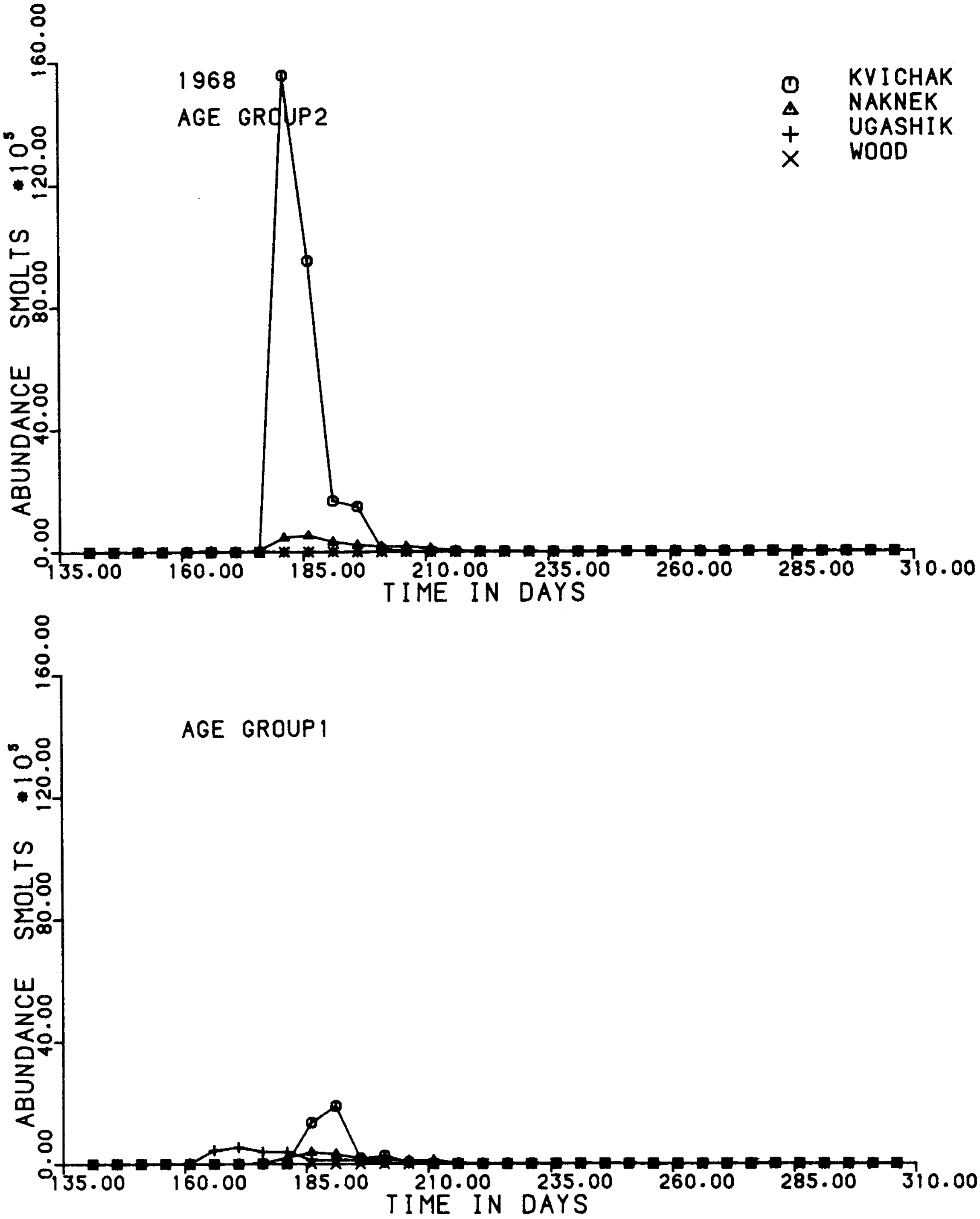

Figure 2.--Projected relative abundance of sockeye salmon smolts at Port Heiden in 1968.

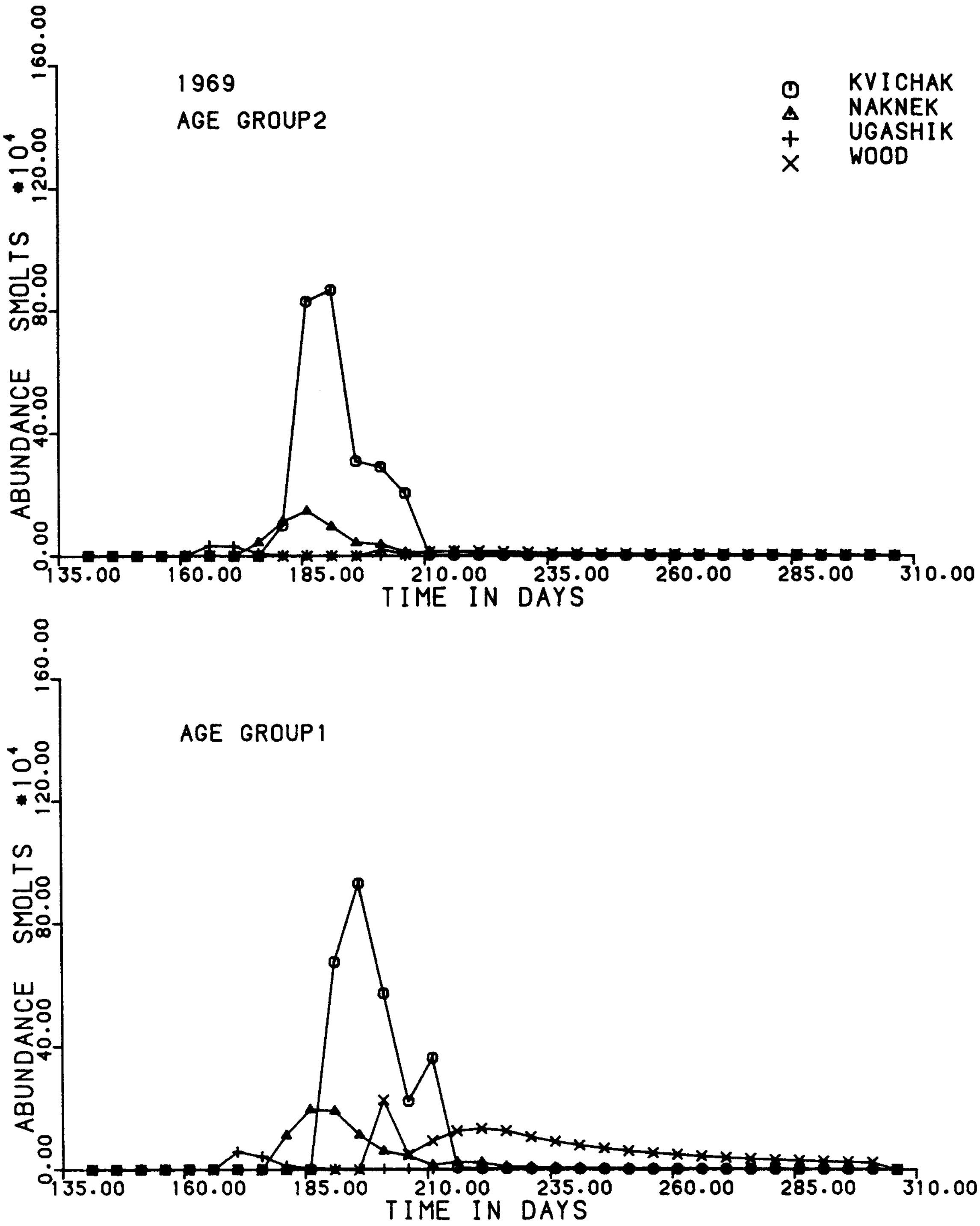

Figure 3.--Projected relative abundance of sockeye salmon smolts at Port Heiden in 1969.

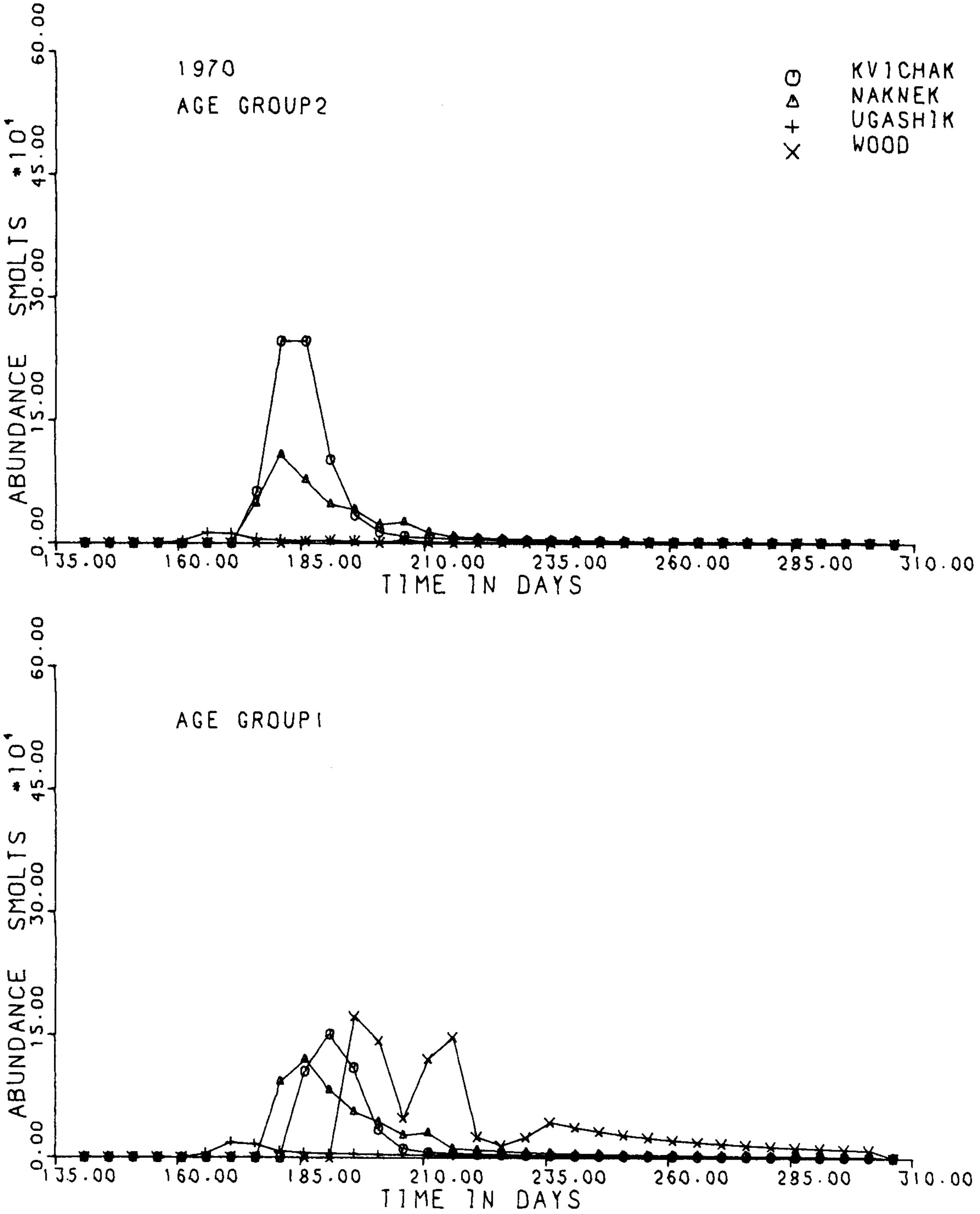

Figure 4.--Projected relative abundance of sockeye salmon smolts at Port Heiden in 1970.

APPENDIX 6

Selected food habits and life history information of four commercial fish and invertebrate stocks of the eastern Bering Sea

P. Livingston

In order to quantify the effects of oil spills on marine fauna in the Bristol Bay area of the Bering Sea, the population dynamics of four species of major commercial importance will be modelled. Some baseline information on the food habits and distribution of these species in the area needs to be established to assist in parameterizing the model. The following is a short summary overview of the food habits of sockeye salmon, *Oncorhynchus nerka*; yellowfin sole, *Limanda aspera*; Pacific herring, *Clupea pallasii*; and red king crab, *Paralithoides camtschatica*. For the most part, an attempt is made to outline only details directly pertinent to those species during that portion of their life history which they spend in Bristol Bay. However, some information is available only for other areas and hopefully should be representative of the species in Bristol Bay.

Sockeye Salmon, *Oncorhynchus nerka*

General Information

Adults begin arriving at river mouths in Bristol Bay in late June and early July. This spawning migration route from their western feeding grounds usually is along the southern portion of Bristol Bay in surface waters (Nishiyama 1974). Outmigrating smolts, which may be 57-122 mm long in British Columbia (Manzer 1969), are usually in the upper 2 m of water (Straty 1974) in Bristol Bay. This migration occurs between May 15-July 15. Smolts move westward alongshore past Port Heiden and Port Moller. They have sometimes been found as late as mid-August off Port Moller. Inshore and in the inner portion of Bristol Bay, most smolts have been found with empty stomachs.

Straty (1964) correlates this with extremely low zooplankton abundance in waters less than 20 fathoms deep. Smolts reaching Port Moller have been found with full stomachs, however.

Food habits

Life history stage	Reference	Season	Area	Food items (% by weight)
Smolts	Manzer, 1969	Su	British Columbia	48% copepods, 24% fish, 14% larvaceans, 5% decapods, 4% insecta, 3% amphipods.
Smolts	Straty, 1974	Su	Bristol Bay	Sandlance larvae, euphausiid larvae, copepods, cladocera, pteropods, decapod larvae, other fish larvae, invert. eggs, insects.
Adults	Kanno and Hamai, 1971	Su	E. Bering Sea shelf	43% euphausiids, 27.7% fish, 25.1% amphipods.
Adults	Nishiyama, 1974	Su	Bristol Bay	70% euphausiids, 20% fish larvae, 10% crab zoea, amphipods and pterapods.

Yellowfin sole, *Limanda aspera*

General Information

Spawning of yellowfin sole has been observed in the Bering Sea at depths of 15-75 m. Eggs have been found off Cape Newenham between June to September and larvae have been observed in the second half of July (Musienko 1968). Seasonal movements of yellowfin sole are not known in great detail but there does seem to be an inshore movement in spring and a movement towards the slope in fall. Some smaller fish (age 5-7) remain in Bristol Bay during winter (Fadeev 1972). It is not known when the pelagic larvae settle to the bottom but presumably they settle inshore. English sole larvae are pelagic for 6 to 10 weeks (Hart 1973) and possibly yellowfin sole larvae are pelagic for a similar length of time. Adults cease feeding in the winter although occasionally in some areas a few fish have been found with some food in their stomachs (Fadeev 1972). Fadeev (1972) also claims that the Cape Newenham area is poor in benthos, therefore yellowfin sole eat more pelagic prey in that area like euphausiids and nektobenthic prey like pandalid shrimp. Simenstad (1979) classified the yellowfin sole in Cook Inlet as a facultative piscivore because they were eating predominantly osmerids but supplementing their diet with other items which in this case were benthic and epibenthic items. In inshore areas of Bristol Bay, the yellowfin sole has been observed feeding on herring eggs attached to vegetation.

Food Habits

Life history stage	Reference	Season	Area	Food Items (% by weight
Larvae (2-10 mm) Limanda limanda	Last, 1980	Su	North Sea	90% copepodites, 10% decapod zoea.
Juveniles	Rogers et al. 1979	Sp, Su	Kodiak I.	22% fish (cottids), 20% poly chaetes, 18% crab, 14% clams.
Adults 100-200 mm	Wakabayashi, 1974	Su	Bering Sea	polychaetes, amphipods, echiuroids.
201-300 mm				polychaetes, bivalves, echiuroids, gadids, osmerids, amphipods.
301+ mm				Mostly bivalves and echiuroids.
Adults	Skalkin, 1963	Sp, Su	SW of Cape Newenham	mysids, euphausiids (30-50 m depth) polychaetes, molluscs (50-65 m depth).

Herring, Clupea harengus pallasii

General Information

Herring leave their wintering grounds in the southeast Bering Sea in March and April. These wintering herring rarely eat (except sometimes euphausiids) according to Dudnik and Usol'tsev (1968). Adults spawn in late April to May in Bristol Bay. Herring caught near Cape Newenham during a spawning survey were all >200 mm long. Adult herring stay inshore in summer for their main foraging period and may stay at least until August. Depths of spawning in British Columbia range between high tide and 11 m (Hart 1973). Eggs hatch in about 10 days and the larval yolk sac lasts at the longest for 2 weeks. After about 2 months the post-larvae begin forming schools which come to the surface in the evening as do the schools of adult herring.

Food habits

Life history stage	Reference	Season	Area	Food items (% by weight)
Larvae 9-20 mm	Wailes, 1963	Sp	British Columbia	40% invertebrate eggs, 40% diatoms, 20% copepods nauplii.
Larvae 9-20 mm	Barraclough, 1967	Sp, Su	British Columbia	90% copepod nauplii, 10% eggs and algae.
Juveniles 20-100 mm	Barraclough, 1967	Su	British Columbia	phytoplankton, copepod eggs, copepods, amphipods, larvaceans.
Adults 100+ mm	Wailes, 1963	Su, F, Sp	British Columbia	euphausiids, copepods.
Adults	Dudnik and U~~s~~sol'tsev 1968	Su	Bering Sea	euphausiids, calanoid copepods, Sagitta.
Adults	Barraclough,	Sp, Su	British Columbia	90% copepods, 10% amphipods, euphausiids, brachyura larvae, and invert. eggs.

King Crab, Paralithodes camtschatica

General Information

During late winter and early spring adult males move inshore where females reside just off of Port Moller in the eastern Bering Sea for breeding purposes. Eggs carried by the females for the past year hatch about April 1 and are found all the way to Port Heiden. Females then molt and males mate with them just after the molt. A female must be inseminated within 5 days after the molt to produce viable eggs. Larvae pass through four pelagic zoeal stages. Each of these stages take about 2-3 weeks to complete. The larvae then spend a month as pelagic megalopae whereupon they metamorphose to first instars about mid-July to August and presumably settle on the bottom. These juveniles presumably stay very nearshore on the shelf north of the Aleutians until they are about 3 yrs. old (60 mm carapace length) (Armstrong et al. 1981).

Food habits

Life history stage	Reference	Season	Area	Food items (% by weight)
Pelagic larvae	Incze, pers. comm.	Sp, Su	Bristol Bay	copepod nauplii, copepodites, cirripedia larvae.
Benthic juveniles	Takeuchi, 1968	Su		polychaetes, seaweed.
Adults	Cunningham, 1969	Su	Bristol Bay	49% echinoderms, 37% molluscs, 10% crustaceans, polychaetes.

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Appendix 7

Views of Dr's. Elmgren (University of Stockholm), Lehtinen and Linden (Swedish Environmental Research Institute) regarding oil in marine environment.

(Recorded by F. Fukuhara)

Dr. R. Elmgren, Zoological Institute, University of Stockholm. Principal Investigator of Tsesis spill.

Elmgren had no specific comments regarding our general approach and agreed that sedimentation and the effects of oil on organisms were the most difficult to predict. He points out that the recognition that some of the oil in spills eventually reaches the bottom is relatively recent and since the Tsesis spill there has been a growing concern that the amount which gets to the bottom exceeds earlier estimates. We also discussed the difficulties of extrapolating laboratory results on the effects of hydrocarbons to actual field conditions. Elmgren recognizes this to be a prevalent problem in almost all toxicological evaluations but he feels that many biological organisms are far more sensitive to hydrocarbons that is normally, experimentally demonstrable.

Some of his views, although not particularly unique, seem useful to consider as general operational principles. One is that, like in the study of all natural phenomenon, there are no certainties in conclusions relating to oil spills and their effects on the ecosystem and organisms. He thought it useful to reiterate this thought to preclude the apiori discarding of speculations or hypotheses simply on the basis of inability to present rigorous proofs.

Like most European scientists in oil spill evaluations, Elmgren considers the impacts of oil on any species to be significant only if long term productivity is substantially reduced or the ecosystem proceeds to some long term secondary succession. Accordingly, he considers oil in the water column to be of interest mostly in terms of determining the quantity, direction, and ultimate deposition or disposition of the spilled oil. He does not consider the oiling of phytoplankton and zooplankton in the water to be anything but of very short term importance. This is partially because the life cycle of these organisms is short (therefore, quick regeneration time), but also because in the Tsesis, MERL, and other oil spill observations, phytoplankton and bacterioplankton increase in abundance and pelagic zooplankton seem not to be visibly impacted (although there is some thought that their feeding may become inhibited). The higher concentrations of sub-surface oil in the RAND analyses can be lethal or debilitating to eggs and larvae (of invertebrates and fish) and may cause significant impact under certain circumstances. Sedimented oil will, of course, affect benthic zooplankton; however, the persistence of unfavorable concentrations will depend upon the hydrocarbon composition of the oil, the relative energy of the benthos and substrate type.

Also as a general consideration, in the real world of a Bering Sea spill he wondered if we should not give some thought and mention to the possible overriding political concern for the protection of birds and mammals. This would argue for the use of dispersants which would minimize the fouling of birds and mammals. This would, of course, increase the problem for fish and shellfish by assuring that the hydrocarbon concentrations in the water would increase from ppb to ppm.

Dr. Elmgren agrees that sedimentation of oil presents the most significant long term hazard of oil spills, but the persistence of oil in his view is dependent upon the hydrocarbon composition of the oil and the bottom sediment type. Oil on rock, gravel, and sand bottom is rather rapidly removed or weathered and the effects of spilled oil on these surfaces would be more immediate and short term. In the case of the Tsesis, the spill occurred in an archipelago of the Baltic Sea which has no measurable tide and very little flushing from the North Sea. Since it is totally surrounded by land, fetch is limited and agitation by wind is comparatively limited. The bottom is muddy or silty, providing the circumstances for long term persistence of oil. Also, the Baltic has little ecological and species diversity.

By contrast, the Amoco Cadiz spill occurred on the relatively high energy coast of Brittany with tides of 5 to 9 meters and very strong coastwise currents and winds generated over the fetch of the North Atlantic Ocean (at about 49°N. lat.). The exposed coast suffered relatively short impacts from the spill; however, some of the bays, inlets, mud flats, and salt marshes were still affected more than 2 years after the spill.

As a rule of thumb, Elmgren considered 2 gm/m^2 of bottom as a threshhold concentration for damage to the benthos. This concentration will cause total mortality to ostracods and amphipods, both of which seem relatively sensitive to petroleum hydrocarbons.

Concerning pathways of oil sedimentation, Elmgren had no real data. His feeling was that much more oil sinks to the bottom than intuition would lead one to believe. In the MERL experiments he now believes 50 - 60% of the introduced oil sank to the bottom. It is his view that most of the oil is transported to the bottom by vertical currents and adsorption of oil droplets to suspended particulate matter. Elmgren considers the oiling of phytoplankton carcasses and consolidation of oil droplets in zooplankton fecal pellets as not important pathways for oil sedimentation. The basis for his views are the MERL experiments in which 50 - 60% sedimentation occurred in tanks where very few zooplankton were present.

His offhand opinion of oil sedimentation in the northern Aleutian area: Given the present information on bottom sediment types, tidal, and boundary currents, and vertical circulation and suspended particulate matter, considerable oil may be sedimented. Because of the relatively active vertical transport of water and the coarse sandy nature of the substrate, he would expect the impacts to the benthos to be relatively short-lived in the P. Heiden and P. Moller spill sites. This would be more comparable to the Amoco Cadiz spill off the Brittany, France coast which has active surface turbulence, boundary and tidal currents so that oil was rapidly dissipated or weathered in the offshore benthos and intertidal areas, except in the heads of sheltered bays (Aber Wrach and Aber Benoil and certain areas of the Bays of Morlaix and Laurion).

In the MERL experiments the oil on the benthos was suspended about 1 cm from the bottom in a flocculence which drifted about and which was seen to escape bottom grabs or flow-through core sampling devices. Therefore, oil on the benthos is apt to impact more severely certain merofauna and infauna which are filter feeders rather than sediment feeders.

In the Tsesis investigations it was evident that oil on the bottom was patchy but according to Elmgren it was difficult to determine if the patchiness was associated with rather small scale variations in bottom topography. There was evidence that oil tended to collect in depressions.

In the MERL tank experiments, oil concentrations were always high in the depressions and fill of core sampling. Thought to occur as a consequence of the way in which sediments were replaced into the hole left by the corer.

As a general rule, Elmgren felt that the effects of an oil spill would last as long as the liftime of the longest living species which was significantly impacted by oil hydrocarbon toxicity. In the Tsesis spill that happened to be the Macoma baltica which may live to be about 15 years old. The M. baltica, however, seemed not to be negatively impacted, their biomass increasing over pre-spill levels. Bering Sea has much more species diversity and a number of species which may live 15 years and considerably longer (e.g. flatfish).

Although he could offer no real hard evidence, Elmgren felt that oil concentrations of themselves may not be nearly as important in biological impacts as the hydrocarbon composition, in particular the aromatic content of the oil. He felt that the multimolecular aromatics such as the tri-methyl benzenes were particularly harmful.

Hydrocarbon in the Tsesis spill area has now returned to pre-spill background levels. These background levels are evidently increasing in the Baltic Sea and Gulf of Bothnia in general and in particular along the major transportation routes of tankers and other ship transportation.

Herring which were reduced in abundance and appeared to avoid certain areas in spawning the year after the spill, have now returned to pre-spill levels of abundance with no apparent aversion to feeding or spawning in areas affected by the Tsesis spill.

Macoma populations no longer carry large concentrations of Tsesis hydrocarbon and now exceed pre-spill abundance levels (roughly 10x pre-spill abundance). The amphipod <u>Pontoporeia affinis</u>, however, the abundance of which was directly related to the abundance of Macoma prior to the spill, remains severely depressed (1% of pre-spill abundance). The failure of Pontoporeia to recolonize the spill area is not easily explained. One thought is that the increasing population of Macoma in the absence of the Pontoporeia which normally prey upon Macoma spat, have so severely agitated the bottom in such a way that Pontoporeia cannot reestablish themselves. Another hypothesis is that unmeasured residues of Tsesis spill (perhaps polar compounds) remain in the sediments, effectively repelling or otherwise preventing the recolonization by <u>P. affinis</u>.

With regard to fish tainting, herring which normally occur in abundance in the Tsesis spill area, a month after the spill showed no particular aversion to oiled areas for feeding but showed no evidence of flesh tainted by petroleum hydrocarbons. Flatfish evidently had hydrocarbon concentation in the muscle of about 50 ppb. Although they were not tested for flavor, they were marketed without adverse consumer reactions. This did not surprise Elmgren since oysters tainted by Amoco Cadiz oil were marketed with tainting levels in the ppm. Although the marketability of oysters from the Bordeaux and Mediterranean, as well as the Brittany region, suffered from customer aversion after the Cadiz spill, levels of hydrocarbon in the tens of ppm were considered to be suitable for human consumption.

There is no evidence that hydrocarbons become progressively concentrated in the food chain. Although certain molluscs may be particularly resistant to high hydrocarbon concentration levels and may themselves be rather inefficient in hydrocarbon metabolization, and therefore, concentrate them, flatfish which eat Macoma may very efficiently dispose of petroleum hydrocarbons. More on this from Linden and Lehtinen.

Elmgren stated that experience with seabirds in the Baltic Sea indicated that most have the reproductive resilience and potential to recover rather rapidly from oil pollution. Animals such as birds which have very high metabolic rates can be expected to suffer more acute damage from the effects of oil spills. His conjecture was that in the Bering Sea, vertebrates such as birds and the sea otter (which lacks the conservative quality of lots of blubber) may be very sensitive to oil, whereas cetaceans and pinnipeds which have comparatively low metabolic rates may be able to better withstand the effects of oil.

In all cases, he suggests we consider such sublethal effects as reduced ability to find mates, capture prey, or elude predators.

Summary of Elmgren's Views

Regarding sedimentation, he agrees that oil on the bottom presents the more serious of oil spill problems. He could, however, present no further evidence relative to quantifying the pathways of oil to the benthos. He is, however, of the strong opinion that most of the sedimented oil results from circulation and adsorption of oil particles to suspended particulate matter. He has reiterated the generally prevalent view that the persistence of oil is related inversely with the rate of water circulation and replacement and in the relatively high energy circumstance of P. Moller and P. Heiden, sedimentation of oil is not expected to be a long term problem.

Concerning the biological impacts of oil, he reiterated what is in the literature relative to the Tsesis, Cadiz, etc. spills, although he brought us up to date on the Tsesis situation. He has added an interesting note regarding the tainting of fish on which Lehtinen and Linden have elaborated.

Elmgren asked that he be kept abreast of our progress and wanted copies of our reports.

References he suggested:

W. E. Haensly, et al. 1982. Histopathology of Pleuronectes platessa, L from Aber Wrach and Aber Benoit, Brittany, France, long term effects of the Amoco Cadiz spill.

Journal of Fish Diseases 1982, 5, 365-391.

Dr. Karl-Johan Lehtinen and Dr. Olaf Linden, Swedish Environmental Research Institute (IVL), Baltic Sea Laboratory, Utovagen, Karlskrona, Sweden.

Dr. Olaf Linden is the director of about 15 scientists studying the impacts of pulp mill effluents, oil, and other pollutants on the marine environment and organisms. They have ongoing studies on the weathering of certain oils on test beaches, they are testing the effectiveness and consequences of various dispersants on oiled beaches (test plots), and they retain a set of outdoor tanks with circulating sea water in which they are testing the effects of pulp mill effluents in various concentrations on the community and organisms of the area. They also have facilities and expertise for bioassays of various pollutants. My impression was that they are a very competent group which are only partially financed by the government but exist on private contracts and grants as a research foundation. Dr. Linden felt personally that he could probably help us most by focusing our reading to the better, more credible and relevant studies of oil spills and related studies. He felt that the oil spill literature was now so voluminous that with our limited involvement with the oil pollution problem it would be virtually impossible to weed out the "wheat from the chaff". He was convinced that a thorough canvasing of the extant oil pollution literature would be a many, many year task.

As with Elmgren, I reviewed very briefly the E. Bering Sea spill scenarios, data bases, background oceanography, biology, etc. and a sketch of our strategy for solution of the simulation exercise.

Linden's view was that the higher concentrations in the Rand outputs were fairly high for actual spill conditions. High enough to impact seriously eggs and larvae and invertebrates and animals carrying exposed eggs.

Linden had nothing to add concerning sedimentation of oil beyond those reports associated with the Argo Merchant, Ixtoc I, Cadiz, Tsesis, and Sefir spills. These reports give estimates of the amounts of oil sedimented but Linden could not elaborate further on the quantities sedimented through different pathways.

(To digress slightly, Dr. Linden is an acknowledged expert on oil spills. He is a member of a U.N. group which responds to world-wide spill problems. In this capacity he has been involved in the investigations of the Ixtoc I spill in which he recalculated the estimates of the amount of fate of the oil spilled (originally calculated by the Mexican government to show least impact on environment). He has also been at the site of at least 2 spills near the Arabian Peninsula. He has a number of publications on the effects of oil on marine organisms, including fish, but he has also published on the fate of oil in spills (Jernelöv and Linden, 1981)).

Linden thought it worthwhile to pay some special attention to the effects of oil washed ashore as it well may in any of the spill scenarios for southeastern Bering Sea. Even if spills occur offshore, if oil washes onshore the effects on the littoral and intertidal fauna can be dramatic and disastrous. In the "Sefir" spill (similar to the "Florida" spill), Linden estimated that more than 90% mortality occurred to the shore and benthic fauna from the acute toxic effects of oil. In the Sefir spill, millions of animals were washed ashore and deposited in windrows on the beaches.

Linden shared Elmgren's view regarding the short term scale of impacts of oil spills on the pelagic community. He was not aware of any documented evidence of major and long term impacts of oil pollution on fish populations. Even in the Tsesis situation, he does not believe there is sufficient evidence to demonstrate that the oil had any direct impact on the behavior or productivity of herring or flatfish.

Also spoke with Dr. Karl-Johan Lehtinen (Carola's husband), a physiologist. On the matter of fish tainting, he believes experimental evidence indicates that oil taken through the food chain would first concentrate in the liver before appearance in circulatory system and flesh. Fish have enzyme systems which metabolize hydrocarbons. Both Lehtinen and Linden felt that high concentrations of hydrocarbon in the liver would probably manifest in debilitating effects before high enough concentrations could be built up in the muscles. For these reasons they felt that tainting of fish loins sufficient to affect human consumption might be a minor consideration. Lehtinen also felt that deposition of oil was more likely in lipids; therefore, might be concentrated in livers, gonads, gills, and perhaps in the flesh of very fatty fish.

Linden referred me to a very good summary report on the Cadiz spill consequences by Lucien Laubier. Laubier states, "oysters containing 20 - 30 ppm dry weight are considered to be slightly polluted by fossil fuels. On a practical basis, an average value of 60 ± 20 ppm, wet weight, was considered as the upper limit for human consumption".

Linden cautioned that much of the information on the Ixtoc I blowout (1979-80) is very unreliable.

Linden suggested the following as excellent references.

1. In the Wake of the Argo Merchant
   Proc. of a Symp., Jan. 11-13, 1978
   Univ. of Rhode Island

   (According to Linden this is one of the best scientific studies of an oil spill.)

2. K. J. Lehtinen, et al.
   1982 - Physiological Effects of Fish Chronically Exposed to Low Levels of Hydrocarbons
   Proc. 5th Arctic Marine Oil Spill Program Technical Seminar
   June 15-17, 1982, Edmonton, Albert, pp 77-92.

3. Neff, Jerry M. and Jack W. Anderson
   1981. Response of Marine Animals to Petroleum and Specific Hydrocarbons. Applied Science Publishers, Ltd., London, pp 177.

   In Linden's view, Dr. Jerry Neff is an authority on the impact of the Amoco Cadiz spill on fish and oysters.

He also considers Boehm as the best oil chemist around (Cadiz expert, also).

Agrees that Van der Mueller of the Bedford Ocean. Inst. is a top-flight sedimentologist and authority on Cadiz oil sedimentation.

He also suggested reading reports related to the grounding of the "Florida" off Falmouth.

APPENDIX 8

Possible Effects of Oil Development in the Bering Sea on the Fishery Resources

(A quantitative numerical evaluation)

Summary

by

Northwest and Alaska Fisheries Center
Resource Ecology and Fisheries Management Division
Seattle, WA 98112

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3. Migrations of fish in the eastern Bering Sea and effect of oil on migratory and stationary adult fish.
4. Food and feeding of fish in the eastern Bering Sea and uptake of contaminated food (including tainting of fish by oil).
5. Sedimentation of oil and effects of oil on the bottom on demersal fish and benthic ecosystem.

BIOLOGICAL IMPACT OF AN OIL SPILL (BIOS)
MODEL DOCUMENTATION
PART 1: FISH MIGRATIONS AND EXPOSURE TO CONTAMINATION

by

Nancy Pola Swan

Final Report
Outer Continental Shelf Environmental Assessment Program
Research Unit 643

April 1984

This report is from a series of processed reports and program documentation produced by the Northwest and Alaska Fisheries Center, National Marine Fisheries Service, NOAA, in Seattle, Washington, and is individually available as Program Documentation No. 21 from that source.

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Figure 2.--Sequence of BIOS model calculations.

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1. Introduction

The BIOS (Biological Impact of an Oil Spill) model, a multispecies ecosystem simulation, was developed at the request of the Outer Continental Shelf Environmental Assessment Program (OCSEAP), as part of their eastern Bering Sea oil impact study. This document is intended as a technical reference for the migration subroutines of BIOS; descriptions of the overall model structure are included for clarification when needed.

BIOS simulates uptake of oil contaminants from exposure to contaminated water and sediments as well as from the consumption of contaminated food. Sixteen species groups are included in the model (Table 1). Salmon appears seasonally in the Bering Sea; the impact of an oil spill on salmon has been modelled by Dr. Nicholas Bax of Compass Systems, Inc. (pers. comm.). BIOS includes depuration of contaminants and effects of fish migrations on the ecosystem.

The model has been applied to three locations in the Bering Sea (Figure 1): offshore of Port Moller, Port Heiden, and Cape Newenham. Computed oil concentrations at each model grid point for 15 days following a hypothetical oil spill were provided by Rand Corporation. Two scenarios were simulated at each location: a "blowout" occurring during a period of high winds and strong tidal mixing, and a tanker deisel spill occurring during a time of weak winds and tidal mixing.

We have attempted to determine the maximum impact of an oil spill on the ecosystem. Wind and tide conditions which would maximize the amount of oil entering the water column were determined by Science Applications, Inc. Avoidance of oil by mobile fish species is not included, allowing maximum exposure to oil. Finally, the migrations of selected fish species across

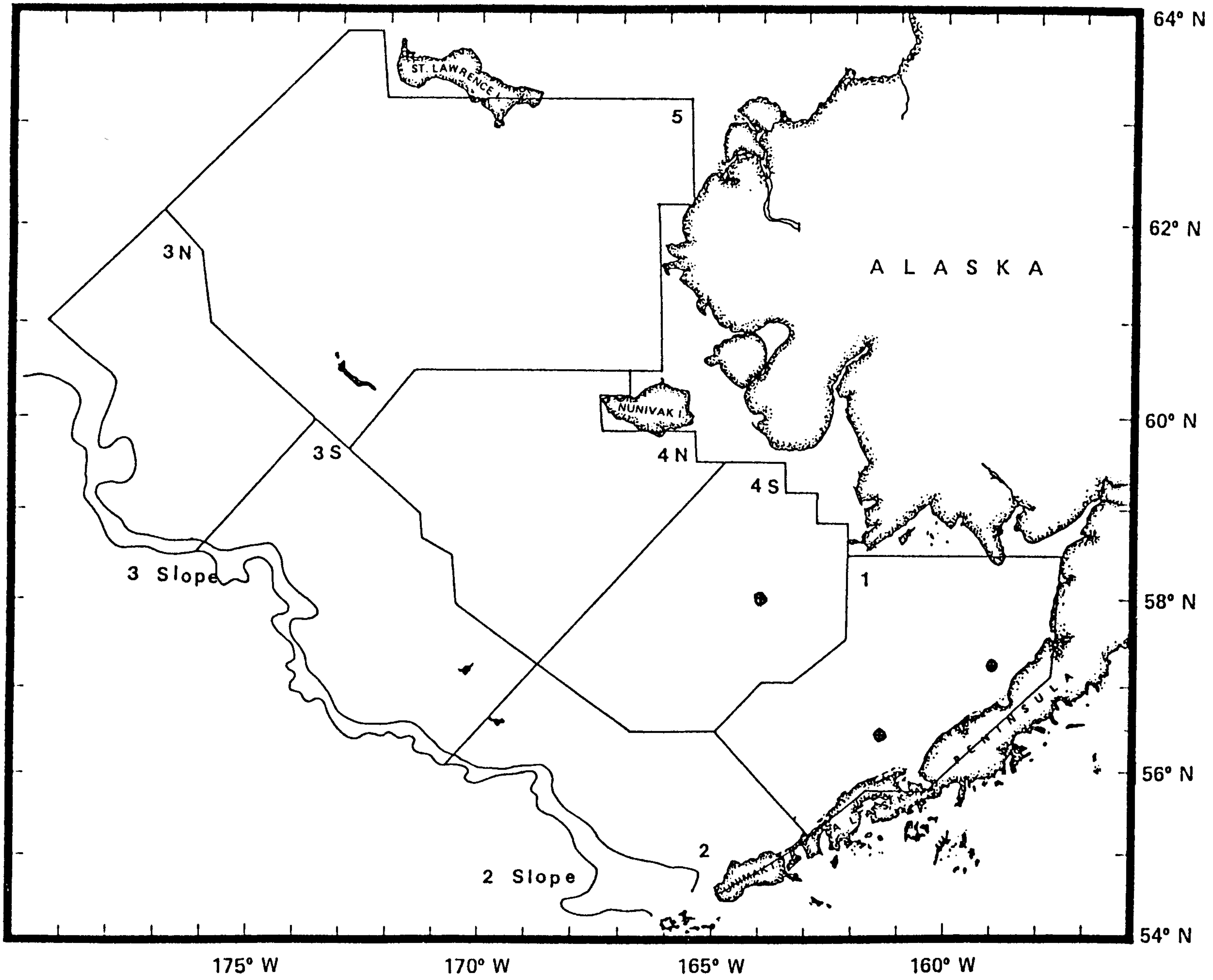


Figure 1.--Locations of hypothetical oil spills in the eastern Bering Sea.

Table 1.--BIOS model species groups

Species number	Species name
1	Herring juveniles
2	Herring adults
3	Pollock juveniles
4	Pollock adults
5	Pacific cod juveniles
6	Halibut juveniles
7	Yellowfin sole juveniles
8	Other flatfish juveniles
9	Yellowfin sole adults
10	Other flatfish adults
11	Pacific cod adults
12	King and Bairdi crab juveniles
13	King and Bairdi crab adults
14	Mobile epifauna
15	Sessile epifauna
16	Infauna

the model grid maximizes the spatial extent of contaminated food. Contaminated fish may also migrate beyond the model grid (i.e., beyond the location of the oil spill), and their effect on the entire eastern Bering Sea ecosystem may be examined by tracing their possible migration routes until depuration is complete.

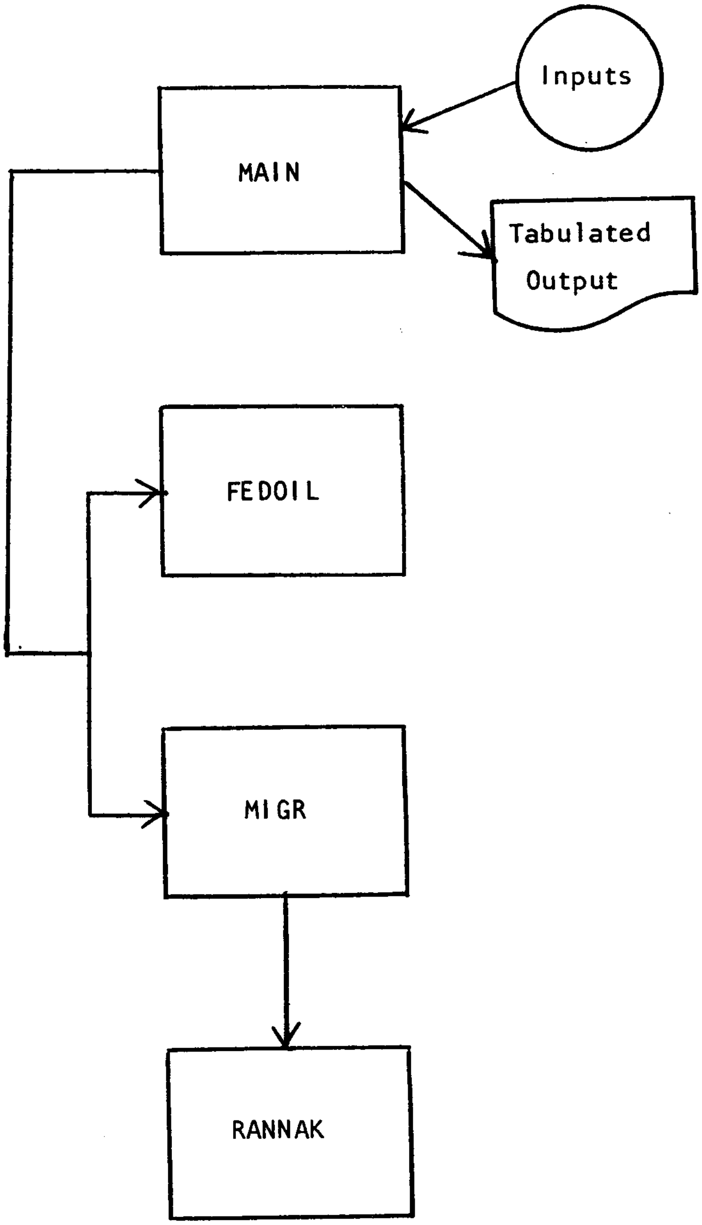
2. Sequence of Model Calculations

The BIOS model is comprised of three sections, as shown in Figure 2. The main program controls the model flow, the feeding subroutine computes uptake of contaminants through consumption, and the migration subroutines simulate fish migrations and uptake of contaminants by exposure to oil in the water or sediments.

At the start of each daily model time step, the main program reads in the concentrations of oil (in parts per billion). Two values are read at each gridpoint: the "water soluable fraction", including oil dissolved or in suspension in the water column, and the "tars" (weathered oil), which concentrate at the sediment-water interface and within the sediments. The tars are computed in a separate model (Laevastu & Fukahara, 1984).

After reading the oil concentrations, the main program calls the feeding subroutine, FEDOIL, which is described in detail in Gallagher, 1984. The feeding subroutine calculates the uptake of contaminated food, updates the fish contamination values, and returns to the main program, which then calls the first migration subroutine, MIGR.

Subroutine MIGR loops through the species list and assigns various parameters. Species-specific choices include: whether or not the species migrates (a function of season), the migrating fraction of the biomass, the migration velocity



Main Program

Directs sequence of model calculations, reads input and prints output.

Feeding Subroutine

Computes uptake of contaminants through consumption of contaminated food.

Main Migration Subroutine

Directs sequence of migration computations. Sets species-specific parameters and velocities, calculates uptake from exposure to oil and depuration.

Migration Calculation Subroutine

Calculates actual migration and redistributes contamination over model grid. Calculates amount of contaminated biomass leaving the model region.

Figure 2.--Sequence of BIOS model calculations.

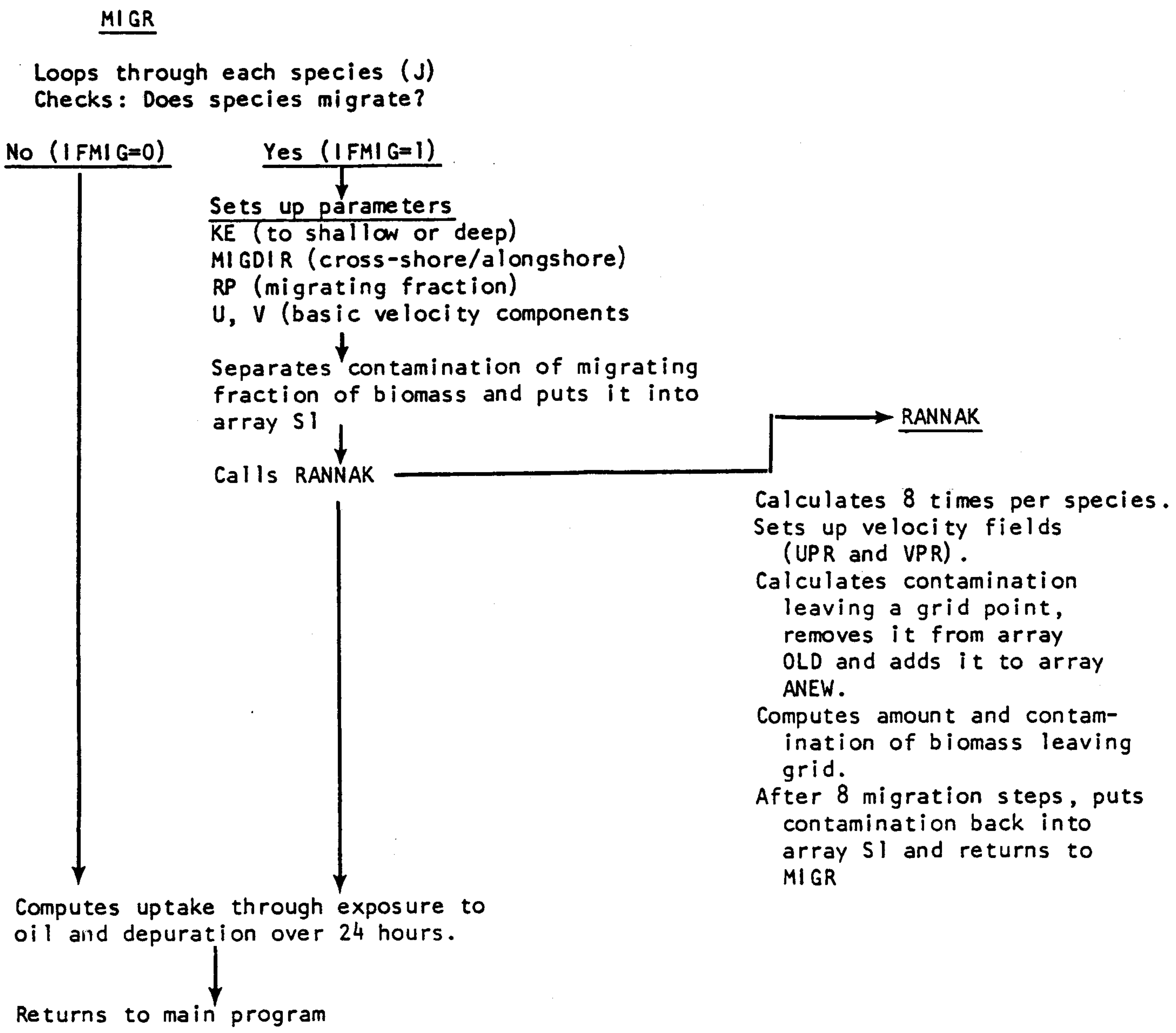
and direction, and uptake and depuration rates. If the species is migrating, MIGR calls RANNAK, which performs the actual migration calculations. On return to MIGR, uptake and/or depuration are calculated, species contamination arrays are updated and selected results are printed. After all species have been considered, MIGR returns to the main program, which prints selected outputs, increments the time step and continues through the next day's calculations.

3. Technical Specifications of Migration Subroutines

The model calculations for the two migration subroutines, MIGR and RANNAK, are diagrammed in Figure 3. MIGR is only called once each model time step (LL); RANNAK is called once for each migrating species (J) during each time step. If a species does not migrate (IFMIG = 0), MIGR calculates the uptake and/or depuration of oil over one day (as described below) and continues on to the next species.

If a species does migrate, MIGR assigns the species-specific migrating fraction and migration speed and direction. The model is presently designed to simulate either alongshore or cross-shore migration (MIGDIR = 1 for cross-shore migration and = 0 for alongshore migration). In addition, migration can either be toward shallow (KE = 1) or deep (KE = 2) water. The migration velocity is assumed constant over the model grid for any one species. Mean velocities (in km/day) are calculated from biomass distributions estimated from fisheries survey data. The model can be easily modified to include spatial variations of migration velocities.

The areal extent of each modelled region is on the order of the station spacing in survey cruises. However, the finer model grid spacing (2 km) is



MIGR

Loops through each species (J)
Checks: Does species migrate?

No (IFMIG=0)

Yes (IFMIG=1)

Sets up parameters
KE (to shallow or deep)
MIGDIR (cross-shore/alongshore)
RP (migrating fraction)
U, V (basic velocity components

Separates contamination of migrating fraction of biomass and puts it into array S1

Calls RANNAK

RANNAK

Calculates 8 times per species.
Sets up velocity fields (UPR and VPR).
Calculates contamination leaving a grid point, removes it from array OLD and adds it to array ANEW.
Computes amount and contamination of biomass leaving grid.
After 8 migration steps, puts contamination back into array S1 and returns to MIGR

Computes uptake through exposure to oil and depuration over 24 hours.

Returns to main program

Figure 3.--Sequence of calculations in BIOS migration subroutines.

necessary to resolve maximum oil concentrations. The biomass for each species is therefore assumed to be constant over the grid. Initial biomass estimates were computed using the DYNUMES eastern Bering Sea ecosystem simulation model (Laevastu and Larkins, 1981). The simulated migrations do not redistribute the biomass over the grid. Instead, since the total duration of a model run is only ten days, it is assumed that biomass is conserved at each grid point, but that the fish contamination is redistributed over the grid. The contamination of each species at each grid point is stored in array OILCON in parts per billion (μg oil per kilogram biomass). As MIGR cycles through each species, the contamination for that species is put into array S1. If a species migrates, the amount of contamination in the migrating fraction of the biomass (RP) is stored in array S3, which is then sent as an argument to RANNAK, along with J, MIGDIR and KE.

RANNAK determines the velocity components from the values of MIGDIR and KE sent from MIGR and sets up arrays of the u-component (UPR) and v-component (VPR) of velocity at each grid point. The u- and v-components of velocity for each of the four possible migration patterns (U1, V1 for cross-shore migration to shallow water; U2, V2 for cross-shore migration to deep water; U3, V3 for alongshore migration to shallow water; and U4, V4 for alongshore migration to deep water) are stored in common block BLKV. If a constant velocity is used over the entire model grid, the appropriate u-component is assigned to each element of array UPR, and the appropriate v-component is assigned to each element of VPR. Specific velocities may, on the other hand, be assigned at any grid point if spatial variation is desired.

The migration calculations in RANNAK are performed 8 times, with a migration time step of 3 hours (TD = 0.125 days) for stability. The stability criterion is:

$$U_m t_d < L \tag{1}$$

where U_m is the maximum migration speed (in km/day), t_d is the migration time step in days (TD in FORTRAN code), and L is the grid spacing in km (AL = 2.0 in the model).

During each of the eight migration periods, the following calculations are performed at all grid points. First, the nearest grid point in the direction of migration is determined. That is, the contamination is moved from point (N, M) to point (N+IN, M+IM), where IN and IM each has a value of either -1, 0, or 1. The amount of contamination leaving a grid point in the x direction (G_x) is:

$$G_x = (C_{n,m} t_d |U|_{n,m})/L \tag{2}$$

and in the y direction (G_y) is:

$$G_y = (C_{n,m} t_d |V|_{n,m})/L \tag{3}$$

$C_{n,m}$ is the contamination of a species at grid point (n,m) before a migration (array S3 in the FORTRAN code), U and V are the velocity components at the grid point (arrays UPR and VPR in the FORTRAN code), and t_d and L are as described for equation (1).

At the start of each migration time step, the values in array S3 are put into array OLD and the elements in S3 are set to zero. The amount of contamination leaving a grid point is taken out of array OLD and put into array ANEW. The FORTRAN code for this operation is:

```
OLD(N,M) = OLD(N,M) - GX - GY
ANEW (N,M+IM) = ANEW (N,M+IM) + GX
ANEW (N+IN,M) = ANEW (N+IN,M) + GY
```

where GX and GY correspond to G_x and G_y in equations (2) and (3), respectively. After the calculations have been completed at all model grid points, the resulting contamination field (the sum of arrays OLD and ANEW) is put into array S3. When the calculations have been performed 8 times (24 hours) the array S3, containing the redistributed contamination, is returned to Subroutine MIGR.

The uptake of contaminants from the water or sediments is simulated by a logistic formula (Wilson, 1975):

$$C_t = C_\infty (1 - e^{-rt}) \qquad (4)$$

where C_t is the contamination of a species at a grid point, r is a species-specific, temperature-dependent uptake rate (on the order of 0.025) and C_∞ is the lethal contamination of the fish. Rates are estimated from available empirical data.

Depuration is simulated by an exponential decay of contamination:

$$C_t = C_{t-1} e^{-R} \qquad (5)$$

where R is the species-specific expiration rate.

Contamination of the fish is stored as parts per billion (µg oil per kg biomass), and is tabulated as a contamination index (contamination in ppm multiplied by time of exposure).

In order to conserve biomass in the model, two boundary conditions were enforced. First, fish were allowed to leave the grid at the boundaries in the direction of migration. The amount of a species' biomass leaving the grid and the contamination of that biomass are calculated and stored for future use. Second, fish may leave the boundaries in the "downstream" region and an equal amount of non-contaminated fish biomass is input to those boundaries.

4. References

Gallagher, A.F.

1984. Biological Impact of an Oil Spill (BIOS) model documentation. Part 2: Contamination through consumption, Nat'l. Mar. Fish. Serv., Northwest and Alaska Fisheries Center, Seattle, WA., MS Program Documentation.

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Appendix A: Subroutine Specifications

SUBROUTINE MIGR

Calling sequence: CALL MIGR

Common blocks used:

COMMON/BLK1/NE, ME, K, LL, ISL

COMMON/OIL/WSF, TARS

COMMON/BLKOIL/OILCON

COMMON/BLKV/RATMG, U1, U2, U3, U4, V1, V2, V3, V4

Variables (in alphabetical order)

A1: Angle (in degrees) of direction of cross-shore migration toward deep water (= ANG1 + 180).

A2: Angle (in degrees) of direction of alongshore migration toward deep water (= ANG2 + 180).

ANG1: Angle (in degrees) of direction of cross-shore migration toward shallow water.

ANG2: Angle (in degrees) of alongshore migration toward shallow water.

CLEFT: Total biomass of a species leaving the model grid during one time step.

IFMIG: Species-specific parameter equal to 1 if the species is migrating or 0 if not.

J: Species number (see Table 1).

KE: Indicator of migration season, equal to 1 for migration to shallow water or 0 for migration to deep water.

MIGDIR: Indictor of type of migration. MIGDIR = 1 for cross-shore migration and MIGDIR = 2 for alongshore migration.

OLDSUM: Total biomass of a species before migration, summed over the model grid.

RATEXP: Expiration rate over 24 hours.

RATMG: Expiration rate over 3 hours.

RP: Migrating fraction of a species' biomass.

SUMNEW: Total biomass of a species after migration, summed over the model grid.

VEL1: Magnitude of cross-shore migration velocity (in km/day).

VEL2: Magnitude of alongshore migration velocity (in km/day).

Arrays (in alphabetical order)[1/]

S1: Contamination of a species at each grid point before migration.

S2: Contamination of the non-migrating fraction of a species' biomass at each grid point.

S3: Contamination of the migrating fraction of a species' biomass at each grid point.

1/ Arrays are dimensioned (NE,ME) unless otherwise specified.

SUBROUTINE RANNAK

Calling sequence: CALL RANNAK (J, MIGDIR, KE, S3)

Arguments:

J: Species number (see Table 1).

MIGDIR: Indicator of type of migration, equal to 1 for cross-shore or 2 for alongshore migration.

KE: Indicator of migration season, equal to 1 for migration toward shallow water or 2 for migration toward deep water.

S3: Array containing contamination of the migration fraction of species J at each grid point.

Common blocks used:

COMMON/BLK1/NE, ME, K, LL, ISL

COMMON/OIL/WSF, TARS

COMMON/BLKV/RATMG, U1, U2, U3, U4, V1, V2, V3, V4

Variables (in alphabetical order):

AL: Model grid spacing, in kilometers.

GONE: Biomass leaving a grid point.

GX: Biomass leaving a grid point in the x-direction.

GY: Biomass leaving a grid point in the y-direction.

IM: Indicator of migration between columns in the model grid. Biomass moves from point (N,M) to point (N,M+IM). If IM=1, biomass moves one column to the right; if IM=-1, biomass moves one column to the left. If IM=0 there is no x-component of migration from point (N,M).

IN: Indicator of migration between rows in the model grid. Biomass moves from point (N,M) to point (N+IN,M). If IN=1, biomass moves one row down; if IN=-1, biomass moves one row up. If IN=0, there is no y-component of migration from point (N,M).

KRC: Number of migration time steps per day.

RATMG: Depuration over one migration time step.

TD: Length of migration time step (in days).

U: U-component of migration velocity.

V: V-component of migration velocity.

Arrays (in alphabetical order)

ANEW: Contamination of migrating fraction of the biomass at each grid point during migration.

OLD: Contamination of migrating fraction of the biomass at each grid point before migration.

S3: Biomass contamination at each grid point before migration, updated after each migration time step, when S3(N,M) = ANEW(N,M)+OLD(N,M).

UPR: U-component of migration velocity at each grid point.

VPR: V-component of migration velocity at each grid point.

Appendix B: Common Blocks

COMMON BLK1:

Statement: COMMON/BLK1/NE, ME, K, LL, ISL

NE: Number of rows in the model grid.

ME: Number of columns in the model grid.

K: Month of model run.

LL: Model time step (days).

ISL: Land-sea array (dimensioned NE x ME), where 0 designates a grid point over land and 1 designates a grid point over water.

COMMON OIL:

Statement: COMMON/OIL/WSF, TARS

WSF: Array dimensioned NE x ME containing the subsurface oil concentration (in ppm) at each model grid point.

TARS: Array dimensioned NE x ME containing the concentration of oil on the bottom (in ppm) at each model grid point.

COMMON BLKOIL:

Statement: COMMON/BLKOIL/OILCON

OILCON: Array dimensioned J x NE x ME containing the contamination of each species (in ppb: µg oil per kg biomass) at each model grid point.

COMMON BLKV

Statement: COMMON/BLKV/RATMG, U1, U2, U3, U4, V1, V2, V3, V4

RATMG: Species-specific depuration over a 3-hour migration time step.

U1: U-component of migration velocity for cross-shore migration toward shallow water.

U2: U-component of migration velocity for cross-shore migration toward deep water.

U3: U-component of migration velocity for alongshore migration toward shallow water.

U4: U-component of migration velocity for alongshore migration toward deep water.

V1: V-component of migration velocity for cross-shore migration toward shallow water.

V2: V-component of migration velocity for cross-shore migration toward deep water.

V3: V-component of migration velocity for alongshore migration toward shallow water.

V4: V-component of migration velocity for alongshore migration toward deep water.

```
      SUBROUTINE MIGR
      DIMENSION ISL(32,34),P1(32,34),OILCON(15,32,34),
     2S1(32,34),S2(32,34),S3(32,34)
      COMMON/OIL/WSF(32,34),TARS(32,34)
      COMMON/BLKOIL/OILCON
      DATA RE,RP,CINF,ANG1,ANG2,VEL1,VEL2/.025,.95,10.,20.,45.,5.,10./
C
      COMMON/BLK1/NE,ME,K,LL,ISL
      COMMON/BLKV/RATMG,U1,U2,U3,U4,V1,V2,V3,V4
C
C
C  PRINT OIL CONCENTRATIONS
C
C  WATER SOLUBLE FRACTION
C
      IONE=1
      ITWO=2
      PRINT 805,LL,IONE
  805 FORMAT("1TIME STEP="I3"  WSF (PPM), PAGE"I2" OF 2"/)
      PRINT 803,(I,I=2,17)
      PRINT 804,((WSF(N,M),M=2,17),N=1,NE)
      PRINT 805,LL,ITWO
      PRINT 803,(I,I=18,33)
      PRINT 804,((WSF(N,M),M=18,33),N=1,NE)
C
C  WEATHERED OIL ON THE BOTTOM
C
      PRINT 806,LL,IONE
  806 FORMAT("1TIME STEP="I3"  TARS (PPM), PAGE"I2" OF 2"/)
      PRINT 803,(I,I=2,17)
      PRINT 804,((TARS(N,M),M=2,17),N=1,NE)
      PRINT 806,LL,ITWO
      PRINT 803,(I,I=18,33)
      PRINT 804,((TARS(N,M),M=18,33),N=1,NE)
C
C  COMPUTE POSSIBLE MIGRATION DIRECTIONS
C
C                        CROSS-SHORE          ALONGSHORE
C TOWARD SHALLOW           U1,V1                U3,V3
C TOWARD DEEP              U2,V2                U4,V4
C
      RAD=0.0174533
      A1=ANG1+180.
      A2=ANG2+180.
      A1=A1*RAD
      A2=A2*RAD
      ANG1=ANG1*RAD
      ANG2=ANG2*RAD
      U1=COS(ANG1)*VEL1
      U2=COS(A1)*VEL1
      U3=COS(ANG2)*VEL2
      U4=COS(A2)*VEL2
      V1=SIN(ANG1)*VEL1
      V2=SIN(A1)*VEL1
      V3=SIN(ANG2)*VEL2
      V4=SIN(A2)*VEL2
C
```

```
C   FOR TEST RUN, USE EXPIRATION RATE (RE)=0.025
C
      RATEXP=EXP(-RE*24.)
      RATMG=EXP(-RE*3.)
C
C   LOOP THROUGH EACH SPECIES
C   S1=CONTAMINATION ARRAY
C
      DO 999 J=1,16
      DO 801 N=1,NE
      DO 801 M=1,ME
  801 S1(N,M)=OILCON(J,N,M)
      PRINT 802,J,LL,IONE
  802 FORMAT("1INITIAL CONTAMINATION, SPECIES"I3", TIME STEP"
     &I3,10X"PAGE"I2" OF 2"/)
      PRINT 803,(I,I=2,17)
  803 FORMAT(16I8/)
      PRINT 804,((S1(N,M),M=2,17),N=1,NE)
  804 FORMAT(16F8.2)
      PRINT 802,J,LL,ITWO
      PRINT 803,(I,I=18,33)
      PRINT 804,((S1(N,M),M=18,33),N=1,NE)
      IF(J.GT.13)GO TO 70
      IF(J.GT.4 .AND. J.LT.8)GO TO 70
      IF(J.GT.10 .AND. J.LT.13)GO TO 70
      IF(K.EQ.1 .OR. K.GT.10)GO TO 70
      MIGDIR=1
C
C   CHECK FOR MIGRATION MONTH
C   KE----INDICATOR OF SEASON.
C   KE=1 (SPRING)        KE=2 (AUTUMN)
C
  800 GO TO(1,2,3,4,70,70,70,8,9,8,70,70,8,70,70,70)J
C
C   HERRING JUVENILES (SPECIES 1)
C
    1 IF(K.LT.6 .OR. K.GT.9)GO TO 70
      KE=2
      GO TO 58
C
C   HERRING ADULTS (SPECIES 2)
C
    2 KE=1
      IF(K.EQ.2 .OR. K.EQ.3)GO TO 58
      KE=2
      IF(K.EQ.7 .OR. K.EQ.6)GO TO 58
      GO TO 70
C
C   POLLOCK JUVENILES (SPECIES 3)
C
    3 KE=1
      IF(K.EQ.4 .OR. K.EQ.5)GO TO 58
      KE=2
      IF(K.EQ.9 .OR. K.EQ.10)GO TO 58
      GO TO 70
C
C   POLLOCK ADULTS (SPECIES 4)
C
    4 KE=1
      IF(K.GT.2 .AND. K.LT.6)GO TO 58
```

```
      KE=2
      IF(K.LT.7 .OR. K.GT.9)GO TO 70
      GO TO 58
C
C  OTHER FLATFISH JUVENILES,ADULTS (SP.8,10) AND CRAB ADULTS(SP.13)
C
    8 KE=1
      IF(K.GT.2 .AND. K.LT.6)GO TO 58
      KE=2
      IF(K.LT.8 .OR. K.GT.10)GO TO 70
      GO TO 58
C
C  YELLOWFIN SOLE ADULTS (SPECIES 9)
C
    9 KE=1
      IF(K.GT.2 .AND. K.LT.7)GO TO 58
      KE=2
      IF(K.LT.8 .OR. K.GT.10)GO TO 70
C
C  SEPARATE MIGRATING AND NON-MIGRATING BIOMASS
C  RP MIGRATING FRACTION
C  FOR TEST RUN, SET RP=0.95, ALL MIGRATING SPECIES
C
   58 OLDSUM=0.$SUMNEW=0.
      DO 65 N=1,NE
      DO 65 M=1,ME
      IF(ISL(N,M).NE.0 .AND. S1(N,M).GT.0.)GO TO 64
      S1(N,M)=0.
      S2(N,M)=0.
      S3(N,M)=0.
      GO TO 65
   64 S2(N,M)=S1(N,M)*RP
      S3(N,M)=S1(N,M)-S2(N,M)
      OLDSUM=OLDSUM+S1(N,M)
   65 CONTINUE
C
C  CALCULATE MIGRATIONS
C
      CALL RANNAK (J,MIGDIR,KE,S2)
C
C     S2 - SPECIES (PORTION WHICH MIGRATED)
C     ISL-SEA-LAND TABLE
C
C     ADDING NONMIGRATING PORTION
      SUMNEW=0.
      DO 63 N=1,NE
      DO 63 M=1,ME
      S3(N,M)=S3(N,M)*RATEXP
      IF(ISL(N,M).EQ. 0)GO TO 63
      S1(N,M)=S3(N,M)+S2(N,M)
      SUMNEW=SUMNEW+S1(N,M)
   63 CONTINUE
      IFMIG=1
C
C     OUTPUT OF MIGRATION RESULTS
C
      CLEFT=OLDSUM-SUMNEW
      PRINT 1007,OLDSUM,SUMNEW,CLEFT
 1007 FORMAT("  PPM BEFORE MIGRATION="F15.6,4X"PPM AFTER "
     2"MIGRATION="F15.6,4X" PPM LEAVING GRID="F15.6/)
```

```
      GO TO 777
C** ** ** **
C  WORK WITH MONTH'S INITIAL BIOMASS IF NO MIGRATION
C
   70 DO 66 N=1,NE
      DO 66 M=1,ME
      S1(N,M)=S1(N,M)*RATEXP
   66 CONTINUE
      IFMIG=0
C
C  UPTAKE CONTAMINATION. FOR TEST, USE UPTAKE RATE=0.025
C
  777 DO 778 N=1,NE
      DO 778 M=1,ME
      S1(N,M)=S1(N,M)+CINF*(1.-RATEXP)
  778 OILCON(J,N,M)=S1(N,M)
C
      IF(IFMIG.NE.1)GO TO 779
      PRINT 1009,J,LL
 1009 FORMAT("1CONTAMINATION (PPM) FOR MIGRATING SPECIES #"I2,2X,
     &"TIME STEP #"I2,10X"PAGE 1 OF 2"/)
      GO TO 780
  779 PRINT 1005,J,LL
 1005 FORMAT("1CONTAMINATION (PPM) FOR SPECIES #"I2,2X,"TIME STEP #"
     &I2,10X"PAGE 1 OF 2"/)
  780 PRINT 803,(I,I=2,17)
      PRINT 804,((S1(N,M),M=2,17),N=1,NE)
 1006 FORMAT("1  PAGE 2 OF 2"/)
      PRINT 1006
      PRINT 803,(I,I=18,33)
      PRINT 804,((S1(N,M),M=18,33),N=1,NE)
  999  CONTINUE
 9999 RETURN
      END
      SUBROUTINE RANMAK(J,MO,KE,S8)
      DIMENSION UPR(32,34),VPR(32,34),S8(32,34),ISL(32,34)
     2,CONTAM(32,34),CNEW(32,34),ANEW(32,34),OLD(32,34)
      COMMON/OIL/WSF(32,34),TARS(32,34)
      COMMON/ELK1/NE,ME,K,LL,ISL
      COMMON/ELKV/RATMG,U1,U2,U3,U4,V1,V2,V3,V4
C     ISL- SEA-LAND TABLE
C     S8 - (SPECIES)
C     KRC - NUMBER OF 2-DAY MIGRATIONS
      AL=2.0
C
C  SET U VELOCITY FIELD
C
C  U,V VELOCITY COMPONENTS
C
C  KE IS INDICATOR OF SEASON
C  KE=1 IF MIGRATION IS TO SHALLOWER WATER; KE=2 FOR MIGRATION TO DEEP
C
      IF(MO.EQ.2)GO TO 10
      IF(KE.EQ.2)GO TO 5
      U=U1
      V=V1
      GO TO 20
    5 U=U2
      V=V2
      GO TO 20
```

```
   10 IF(KE.EQ.2)GO TO 15
      U=U3
      V=V3
      GO TO 20
   15 U=U4
      V=V4
   20 TD=.125
      NEH=NE-1
      MEH=ME-1
C
C  MIGRATE 8 TIMES PER DAY (TD=.125 DAYS)
C
C  S8 IS MIGRATING CONTAMINATION
C
      DO 254 KRC=1,8
C
C  CALCULATE MIGRATION AT EACH GRID POINT
C
      DO 133 N=1,NE
      DO 133 M=1,ME
      OLD(N,M)=S8(N,M)
      S8(N,M)=0.
      IF(ISL(N,M).EQ.0)GO TO 133
C
C  DETERMINE GRID POINT TO GO TO
C
      IF(KRC.GT.1)GO TO 1136
      IF(LL.GT.1)GO TO 1136
      UPR(N,M)=U
      VPR(N,M)=V
 1136 CONTINUE
      IF(U)230,231,232
  230 IM=-1
      GO TO 233
  231 IM=0
      GO TO 233
  232 IM=1
  233 IF(V)234,235,236
  234 IN=1
      GO TO 237
  235 IN=0
      IF(IM.EQ.0)GO TO 133
      GO TO 237
  236 IN=-1
C
C  GX,GY ARE AMTS LEAVING IN X,Y DIRECTION
C  OLD IS ORIGINAL CONTAMINATION LEFT AT GRIDPOINT N,M
C  ANEW IS FIELD OF MIGRATED CONTAMINATION
C
  237 GONE=(OLD(N,M)*TD)/AL
      GX=GONE*ABS(U)
      GY=GONE*ABS(V)
      IF(M.EQ.ME .AND. IM.GT.0)GO TO 991
      IF(M.EQ.1 .AND. IM.LT.0)GO TO 991
      IF(ISL(N,M+IM).EQ.0)GO TO 991
      ANEW(N,M+IM)=ANEW(N,M+IM)+GX
  991 IF(N.EQ.1 .AND. IN.LT.0)GO TO 992
      IF(N.EQ.NE .AND. IN.GT.0)GO TO 992
      IF(ISL(N+IN,M).EQ.0)GO TO 992
      ANEW(N+IN,M)=ANEW(N+IN,M)+GY
```

```
992  OLD(N,M)=OLD(N,M)-GY
     OLD(N,M)=OLD(N,M)-GX
133  CONTINUE
591  DO 751  N=1,NE
     DO 751  M=1,ME
     IF(ISL(N,M).EQ.0)GO TO 751
     S8(N,M)=OLD(N,M)+ANEW(N,M)
     S8(N,M)=S8(N,M)*RATMG
     OLD(N,M)=0.
     ANEW(N,M)=0.
751  CONTINUE
254  CONTINUE
     RETURN
     END
```

DOCUMENTATION OF THE BIOLOGICAL IMPACT OF AN OIL SPILL MODEL, BIOS

PART 2: FISH FEEDING AND CONTAMINATION THROUGH CONSUMPTION - SUBROUTINE FEDOIL

by

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Final Report
Outer Continental Shelf Environmental Assessment Program
Research Unit 643

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1. Introduction

The Biological Impact of an Oil Spill model, BIOS, is a multispecies ecosystem simulation that quantitatively analyzes the expected impact of hypothetical oil spill scenarios on fishery resources in the eastern Bering Sea. It was developed at the request of the Outer Continental Shelf Environmental Assessment Program (OCSEAP), and is a part of their eastern Bering Sea oil impact study. This program documentation is intended to serve as a technical reference for the BIOS computations of the subroutine FEDOIL, which simulates the uptake of oil contaminants through feeding and the consumption of oil contaminated food.

A full description of the OCSEAP study and its relation to the BIOS model can be found in Laevastu and Fukuhara (1984). Details of the BIOS computations for simulating fish migrations, uptake of oil contaminants from exposure to oil contaminated water and sediments, and depuration of oil contaminants is given in Swan (1984). The theory and underlying assumptions of the subroutine FEDOIL, with examples of results, is given in Gallagher (1984).

As general background, BIOS is a gridded model that simulates uptake of oil contaminants in selected marine species resulting from exposure to oil contaminated water and sediments and the consumption of oil contaminated food. The model includes sixteen marine species or species groups (Appendix Table 1), simulates the expected impacts of two hypothetical oil spill scenarios (see Laevastu and Fukuhara, 1984, for details), and has been applied to three locations (Port Moller, Port Heiden, and Cape Newenham) in the Bristol Bay region of the eastern Bering Sea (Figure 1).

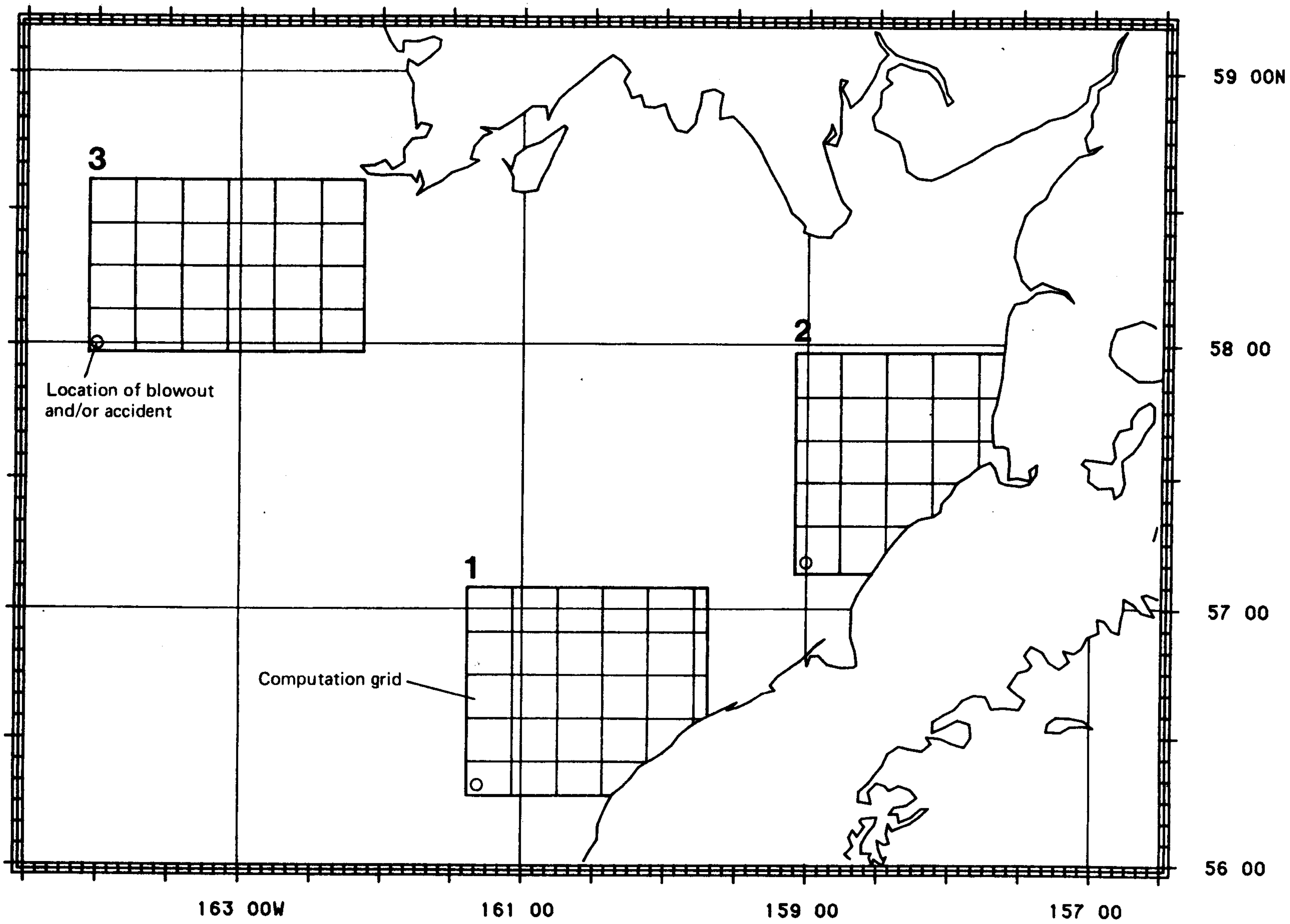


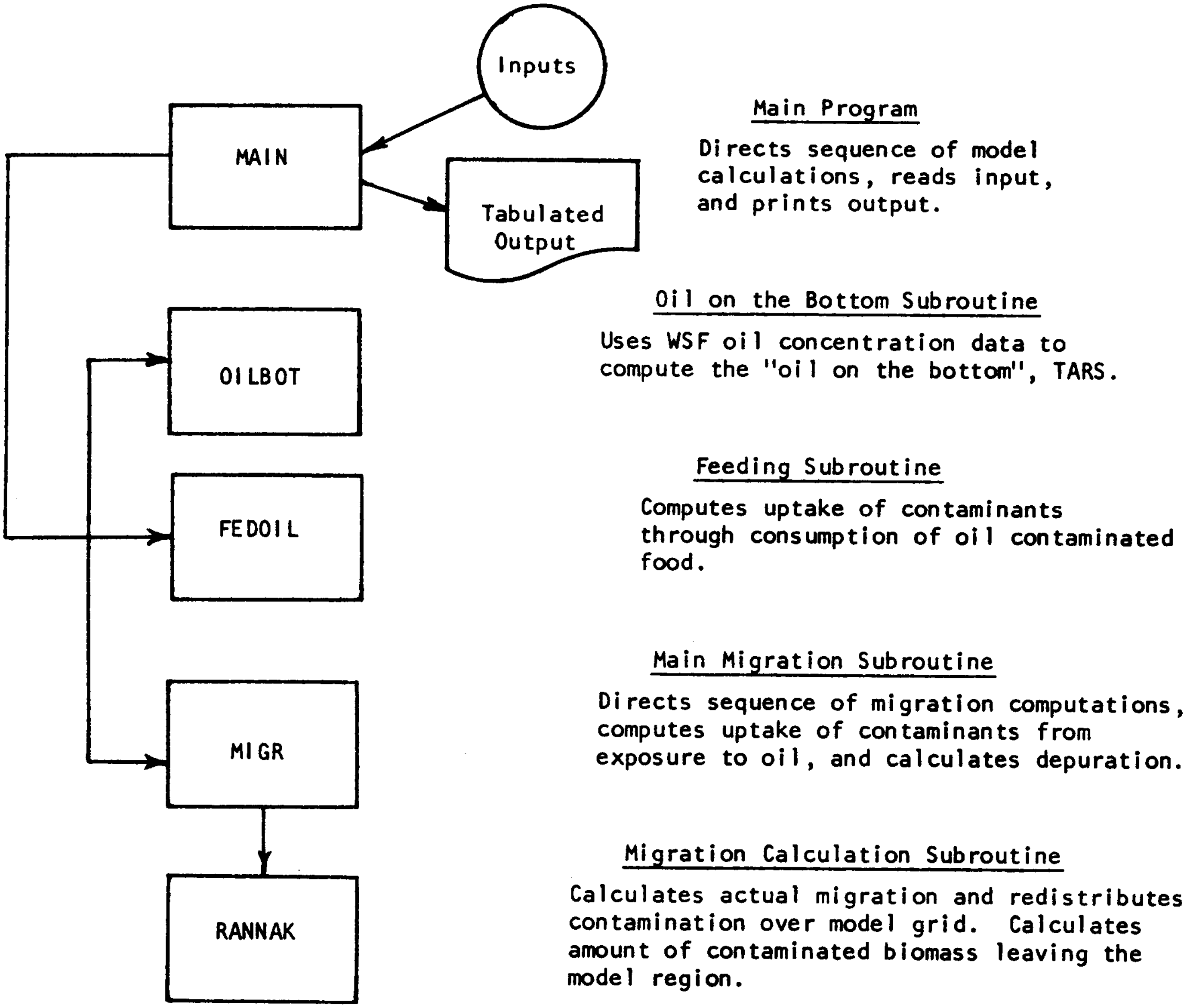
Figure 1.--Locations of hypothetical oil spills, and computational grids in Bristol Bay.

Input data on oil spill concentrations at each model grid point for each scenario and location were provided by Rand Coporation in conjunction with Science Applications, Inc. (SAI); details are provided elsewhere (see Laevastu and Fukuhara, 1984, for references). Input biomass data for each species or species group for the three oil spill locations are given in Appendix Table 1. A list of the parameters and values used in the FEDOIL computations is provided in Appendix Table 2. Appendix Table 4 gives a list of symbols and abbreviations.

2. Sequence of BIOS Model Computations

The BIOS model is comprised of four sections as shown in Figure 2 (this is an update of the model as described in Swan, 1984). The main program controls the model flow; the subroutine OILBOT computes the "oil on the bottom"; the subroutine FEDOIL computes uptake of oil contaminants due to consumption of oil contaminated food; and the subroutines MIGR and RANNAK simulate fish migrations, uptake of oil contaminants due to exposure to oil contaminated water and sediments, and depuration.

At the start of each daily model time step, the main program reads in the appropriate oil concentrations (in parts per billion (ppb)) for the selected scenario and location. These oil concentration data are read in for each grid point ((N,M), location specific and defined in the computer code), and are the "water soluable fraction" (WSF) that includes the dissolved and emulsified oil in the water. The main program then calls the subroutine OILBOT to compute the "oil on the bottom" (TARS) that includes the weathered and sedimentized oil that accumulates in a nepheloid layer at the sediment-water interface and in the sediments (details are given in Laevastu and Fukuhara, 1984). The model then returns to the main program and calls the subroutine FEDOIL.



Main Program

Directs sequence of model calculations, reads input, and prints output.

Oil on the Bottom Subroutine

Uses WSF oil concentration data to compute the "oil on the bottom", TARS.

Feeding Subroutine

Computes uptake of contaminants through consumption of oil contaminated food.

Main Migration Subroutine

Directs sequence of migration computations, computes uptake of contaminants from exposure to oil, and calculates depuration.

Migration Calculation Subroutine

Calculates actual migration and redistributes contamination over model grid. Calculates amount of contaminated biomass leaving the model region.

Figure 2.--Sequence of BIOS model computations.

Upon completion of the FEDOIL computations described in detail below, the model returns to the main program, which then calls the subroutine MIGR. Details of subroutine MIGR and its associated subroutine RANNAK are given in Swan (1984), and will not be repeated here. After completing the MIGR computations, the model returns again to the main program. The main program then prints selected outputs, increments the model time step, and repeats the sequence of subroutine calls for the new time step.

3. Details and Technical Specifications of Subroutine FEDOIL

Subroutine FEDOIL is called once during each time step (LL) of the simulation and computes the uptake of oil contaminants due to consumption (CONOIL) for each species (J). It then adds this value to the current level of oil contaminants (OILCON) in the given species. After completing the computations for all species, FEDOIL returns to the main program. A general flow diagram of the subroutine is given in Figure 3.

FEDOIL first sets general constants for use in subsequent subroutine equations, and then begins to loop through the species specific computations for feeding. The first step in computing the uptake of contaminants is to determine the species specific food coefficient ($TOH_{LL,J}$) of the given time step (LL).

The food coefficient ($TOH_{LL,J}$) for each species group (J) is computed as a function of percent body weight daily. The basic rates of percent body weight daily (TJ_J, given as a fraction), are prescribed by month (t) and adjusted for seasonal variation via the harmonic function:

$$TOH_{LL,J} = TJ_J + [0.35(TJ_J) \cos (ALP(t) - GKAP)] \qquad (1)$$

where ALP is the phase speed and equals 30°; t is the month of the simulation; LL is the daily time step of the simulation (i.e., TOH is held constant over a month); and GKAP is the phase lag and equals 175°. This equation is taken from Laevastu and Larkins (1981), and is discussed in detail in Gallagher (1984).

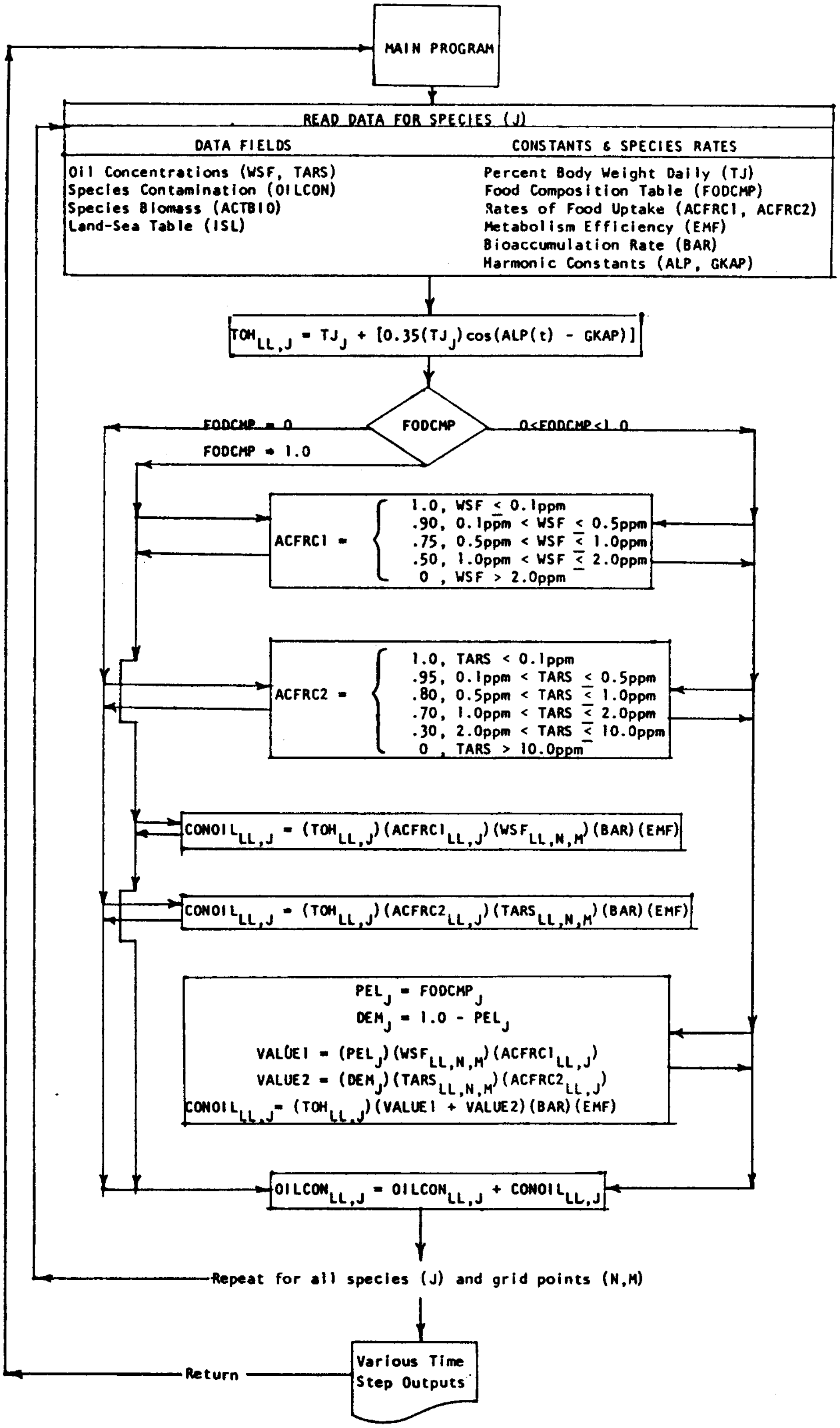


Figure 3.--Flow diagram of subroutine FEDOIL

After computing $TOH_{LL,J}$, the subroutine next checks the percent composition of pelagic food items ($FODCMP_J$) in the diet of the given species in order to determine which of several equations to use in computing the value of CONOIL. As a first approximation, the general food composition data for each species or species group are assumed to be comprised of either general pelagic food, general demersal food, or a combination of both.

If the diet of a given species (J) is composed only of pelagic food items (i.e., FODCMP=1.0), then the amount of oil contaminants taken up through feeding during time step LL, $CONOIL_{LL,J}$, is given by:

$$CONOIL_{LL,J} = (TOH_{LL,J})(ACFRC1_{LL,J})(WSF_{LL,N,M})(BAR)(EMF) \qquad (2)$$

where $ACFRC1_{LL,J}$ is the actual fraction of a species diet that may be consumed and is a function of oil concentration, $WSF_{LL,N,M}$ (see Appendix Table 3); BAR is the bioaccumulation ratio between oil concentration in the environment and oil concentration in the food items in a species diet; EMF is the efficiency of metabolism of oil contaminated food items; and other parameters are as previously given. (The preliminary values for parameters BAR and EMF are 50.0 and 0.75, respectively. See Laevastu and Fukuhara (1984) and Gallagher (1984) for a full discussion.)

Equation 2 is a reasonable approximation given the additional assumptions that 1) a species biomass is constant across the grid and for the duration of the simulation due to the limited temporal and spatial scales of the study; and 2) that in computing the amount of oil contaminants taken up during feeding, no attempt is made to try to estimate species specific growth or consumption rates; instead, each species (J) is assumed to get its full food requirement (food ration) for each time step of the simulation (i.e., there is no starvation). This topic is discussed in detail in Gallagher (1984).

If the diet of a given species (J) is composed only of demeral food items (i.e., FODCMP=0), then the amount of oil contaminants taken up through feeding during time step LL, $\mathrm{CONOIL}_{LL,J}$, is given by:

$$\mathrm{CONOIL}_{LL,J} = (\mathrm{TOH}_{LL,J})(\mathrm{ACFRC2}_{LL,J})(\mathrm{TARS}_{LL,N,M})(\mathrm{BAR})(\mathrm{EMF}) \tag{3}$$

where $\mathrm{ACFRC2}_{LL,J}$ is the actual fraction of a species diet that may be consumed and is a function of oil concentration, $\mathrm{TARS}_{LL,N,M}$ (see Appendix Table 3), and other parameters are as previously given.

If the diet of a given species (J) is composed of both pelagic and demersal food items (i.e., 0<FODCMP<1.0), then the amount of oil contaminants taken up through feeding during time step LL, $\mathrm{CONOIL}_{LL,J}$, is given by:

$$\mathrm{CONOIL}_{LL,J} = (\mathrm{TOH}_{LL,J})(\mathrm{VALUE1} + \mathrm{VALUE2})(\mathrm{BAR})(\mathrm{EMF}) \tag{4}$$

where

$$\mathrm{VALUE1} = (\mathrm{PEL}_J)(\mathrm{WSF}_{LL,N,M})(\mathrm{ACFRC1}_{LL,J}) \tag{5}$$

and

$$\mathrm{VALUE2} = (\mathrm{DEM}_J)(\mathrm{TARS}_{LL,N,M})(\mathrm{ACFRC2}_{LL,J}) \tag{6}$$

and where PEL_J is the fraction of pelagic food items in a species diet and is equal to FODCMP; DEM_J is the fraction of demersal food items in a species diet and is equal to $(1.0 - \mathrm{PEL}_J)$; and other parameters are as previously given.

The subroutine now adds the computed amount of oil contaminants taken up through feeding ($\mathrm{CONOIL}_{LL,J}$) to the current amount of oil contaminants already existing in the species (OILCON_J). These data are then stored as parts per million (ppm; i.e., milligrams (mg) of oil per kilogram (kg) of biomass), for later use in other subroutines. These procedures are then repeated for all species and all grid points for the given location and scenario. When all computations have been completed, subroutine FEDOIL prints selected outputs for the given time step (e.g., Appendix Table 5) and then returns to the main program.

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5. Appendix Tables

Appendix Table 1.--List of species and input biomass data (by location) used in BIOS[1/].

No.	Species Name	Input Biomass Data[2/] Port Moller	Port Heiden	Cape Newenham
1	Herring juveniles	1409	521	1551
2	Herring adults	1121	414	1234
3	Pollock juveniles	3708	2322	3261
4	Pollock adults	11007	6893	9679
5	Pacific cod juveniles	424	279	307
6	Halibut juveniles	730	330	240
7	Yellowfin sole juveniles	722	482	711
8	Other flatfish juveniles	2004	1472	1650
9	Yellowfin sole adults	800	534	789
10	Other flatfish adults	2004	1472	1650
11	Pacific cod adults	861	461	681
12	King and Bairdi crab juveniles	664	222	432
13	King and Bairdi crab adults	1654	553	1078
14	Mobile epifauna	5970	4995	6075
15	Sessile epifauna	13930	11655	14175
16	Infauna	19150	13750	19250

[1/] The DYNUMES model (Laevastu and Larkins, 1981) was used to get initial estimates of input biomass data for the three model locations of the BIOS model. These data were in kilograms per square kilometer (kg/km^2). Complete details on methods used will be provided in the final report.

[2/] The following assumptions were used to convert the data obtained from the DYNUMES model to biomass fields for use in the BIOS model.

a) Unless noted differently below, the breakdown of species biomass data into juvenile and adult fractions was based on Niggol (1982).

b) DYNUMES species group 5 (halibut) was assumed to be 100% juvenile (i.e., in these shallow waters during this season).

c) Yellowfin sole data were assumed to comprise 75% of DYNUMES species group 7 (yellowfin and rock sole).

d) DYNUMES species group 13 (Pacific and saffron cod) was assumed to be 100% Pacific cod.

e) DYNUMES species groups 7 (rock sole-25%), 6 (flathead sole, flounder), and 8 (other flatfish) were combined to make up the other flatfish group (species 8 and 9) for the BIOS model. These groups were assumed to be equally divided between juveniles and adults.

f) DYNUMES species groups 19 (king crab) and 20 (Tanner crab) were combined, and using available survey data, assumed to be comprised of 71.4% adults and 28.6% juveniles.

g) DYNUMES species group 24 (epifauna) was assumed to be 30% mobile and 70% sessile.

Appendix Table 2.--List of parameters and associated values used in the equations of subroutine FEDOIL.

No.	Species (J)	TJ[1/]	Food composition[2/] %PEL	%DEM[3/]
1	Herring juveniles	.016	100	-
2	Herring adults	.010	95	5
3	Pollock juveniles	.012	95	5
4	Pollock adults	.007	72	28
5	Pacific cod juveniles	.015	81	19
6	Halibut juveniles	.012	43	57
7	Yellowfin sole juveniles	.012	20	80
8	Other flatfish juveniles	.012	20	80
9	Yellowfin sole adults	.006	15	85
10	Other flatfish adults	.006	25	75
11	Pacific cod adults	.007	30	70
12	King and Bairdi crab juvenile	.012	30	70
13	King and Bairdi crab adults	.006	10	90
14	Mobile epifauna	.019	0	100
15	Sessile epifauna	.006	0	100
16	Infauna	.006	0	100

[1/] Data taken from DYNUMES model (Laevastu and Larkins, 1981).

[2/] The basic food composition data is given as the percent pelagic food in a given species' diet; i.e., parameter FODCMP. Details on the estimation of FODCMP are given in Gallagher (1984).

[3/] [PEL = FODCMP]; [DEM = 1.0 - PEL].

Appendix Table 3.--Effects of various concentrations of WSF and TARS of crude oil on the actual food uptake of selected marine species.[1]

Oil Concentration (CONC) (in ppm)	CONC≤0.1	0.1< CONC≤0.5	0.5<CONC≤1.0	1.0<CONC≤2.0	2.0<CONC≤10.0	CONC>10.0
Pelagic feeders (WSF)[2] ACFRC1	1.0	.90	.75	.50	0	0
Demersal feeders (TARS) ACFRC2	1.0	.95	.80	.70	.30	0

[1] For a more detailed analysis see Laevastu and Fukuhara (1984).

[2] Values shown are the actual fraction of a species food requirement that would be eaten under the given level of oil concentration.

Appendix Table 4.--List of symbols and abbreviations.

ACFRC1 - Fraction of species diet that may be consumed given the existing oil concentrations of WSF.

ACFRC2 - Fraction of species diet that may be consumed given the existing oil concentration of TARS.

ACTBIO - Species biomass.

ACTCON - Total amount of food items consumed by a species - since there is no starvation, it is a function of percent body weight daily (TOH) and biomass (ACTBIO).

ALP - Phase speed (30° to reflect monthly adjustment).

BAR - Bioaccumulation ratio.

CONOIL - Amount of oil contaminants taken up during feeding.

DEM - Fraction of species diet that is demersal food - equal to (1.0-PEL).

EMF - Efficiency of metabolism of oil contaminated food items.

FODCMP - Fraction of species diet that is pelagic food (input food composition data).

GKAP - Phase lag (175° to prescribe time when function is maximum).

ISL - Land-sea table - defines land and sea areas of computational grids.

K - Same as t (symbol used in computer code).

LL - Time step of the simulation (i.e., daily).

LOC - Index defining location of current simulation (symbol used in computer code).

ME - Columns used in grid array - location specific.

Appendix Table 4 (cont'd)

NE - Rows used in grid array - location specific.

OILCON - Current level of oil contaminants in a given species.

PEL - Fraction of species diet that is pelagic food - equal to FODCMP.

RAD - Radians.

t - Month of simulation.

TARS - Oil concentration "on the bottom".

TJ - Basic species specific rate of percent body weight daily - defined on a monthly basis.

TOH - Species specific food coefficient as a function of percent body weight daily.

WSF - Oil concentration as "water soluable fraction".

Appendix Table 5.--Example of computed 'contamination index' of a species caused by uptake of oil contaminated food[1].

Species No. 1 Biomass: 1409.00 kg/km^2	Location: Port Moller Time Step: Day 1	
	Total contaminated biomass[2]	
Concentrations in ppb (µg/kg)	Kilograms (kg)	Area (km^2)
Cont. Index greater than 1000.00	0.00	0.00
Cont. Index 500.00 to 1000.00	5636.00	4.00
Cont. Index 100.00 to 500.00	73268.00	52.00
Cont. Index 50.00 to 100.00	28180.00	20.00
Cont. Index 10.00 to 50.00	39452.00	28.00
Cont. Index 1.00 to 10.00	78904.00	56.00
Cont. Index 0.10 to 1.00	39452.00	28.00
Cont. Index less than 0.10	5867076.00	4164.00

[1] A full discussion of these and other output data will be given in the final report.

[2] Each grid point is a square of 4 km^2. Thus the total contaminated biomass at a grid point is equal to (ACTBIO x 4).

6. Subroutine Listing in FORTRAN[1]

---

[1] Computer code for the printing of selected outputs has been deleted.

```
C
C**************************************************************************
C                                     MAIN PROGRAM
C**************************************************************************
C
$RESET FREE
$SET OWN OWNARRAYS
$SET LIST LINEINFO STACK
FILE  1(KIND=DISK,TITLE="OCSEAP/OILCON/LOC1",FILETYPE=7)
FILE  2(KIND=DISK,TITLE="OCSEAP/OILCON/LOC2",FILETYPE=7)
FILE  3(KIND=DISK,TITLE="OCSEAP/OILCON/LOC3",FILETYPE=7)
FILE  4(KIND=DISK,TITLE="OCSEAP2/LOC2/LANDSEA",FILETYPE=7)
FILE  6(KIND=PRINTER)
      COMMON/BLK1/NE,ME,K,LL,ISL
      COMMON/BLKBIO/LOC,ACTBIO(16)
      COMMON/INPBIO/BIOLC(3,16)
      COMMON/BLKOIL/OILCON(16,32,34)
      COMMON/OIL/WSF(32,34),TARS(32,34)
      DIMENSION ISL(32,34),D(32,34),TB(4)
      NE=32;ME=34;K=8;LL=1;LLMAX=10;LOC=1
C
C--------------------------------------------------------------------------
C     USING DEFINED LOCATION (I.E., LOC=1),SET ACTBIO EQUAL TO BIOLC.
C     BIOLC IS A BLOCK DATA ARRAY CONTAINING SPECIES BIOMASS DATA FOR
C     ALL THREE OIL SPILL SCENARIO LOCATIONS.
C--------------------------------------------------------------------------
C
      DO 25 J=1,16
      ACTBIO(J)=BIOLC(LOC,J)
   25 CONTINUE
C
C--------------------------------------------------------------------------
C     READ INPUT DATA FOR ISL AND WSF
C     NHRS IS THE TIME STEP OF THE WSF DATA (I.E., 1 DAY = 24HRS)
C--------------------------------------------------------------------------
C
      IF(LOC.NE.2)GO TO 51
      DO 50 N=1,NE
   50 READ(4,100)ME,(ISL(N,M),M=1,ME)
      GO TO 30
   51 DO 52 N=1,NE
      DO 52 M=1,ME
   52 ISL(N,M)=1
  100 FORMAT(*I2)
   30 READ(LOC,/)NHRS
      READ(LOC,/)((WSF(N,M),M=1,ME),N=1,NE)
C
C--------------------------------------------------------------------------
C  CHANGE OIL CONCENTRATIONS FROM PPB TO PPM
C--------------------------------------------------------------------------
C
      DO 32 N=1,NE
      DO 32 M=1,ME
   32 WSF(N,M)=WSF(N,M)/1000.
C
C--------------------------------------------------------------------------
C     COMPUTE OIL ON THE BOTTOM
```

```
C      SEE LAEVASTU AND FUKUHARA (1984) FOR DETAILS
C----------------------------------------------------------------------
C
       BLO=2.
C      BLO=2 CONTINUOUS SOURCE, BLO=1 INSTANTANEOUS SOURCE.
       DL=2000.
C      TAT TIME STEP IN HOURS
       TAT=24.
       TD=20.
       T=LL*1440.
       KAL=1
C      KAL=0 - NO OIL MOVEMENT ON THE BOTTOM, 1 OIL ADVECTED ON BOTTOM
C      KU - CURRENT INDEX, SEE CURCIL; KA - TURBULENCE INDEX(NOT USED);
C      LU - PRINT SCALING INDEX
       KU=3
       KA=1
       LU=0
       UI=0.
       VI=0.
       CALL OILBCT(WSF,LL,TD,DL,D,TARS,TB,BLO,UI,VI,KU,KAL,T,KA,TAT)
       UI=60.
       VI=8.
       IF(KAL.NE.1)GO TO 31
       CALL CURCIL(TARS,KU,UI,VI,DL,LL,BLO,T,KAL)
  31   CONTINUE
C
C----------------------------------------------------------------------
C      CALL FECCIL AND MIGR
C----------------------------------------------------------------------
C
       CALL FECCIL
       CALL MIGR
C
C----------------------------------------------------------------------
C      INCREMENT TIME STEP
C----------------------------------------------------------------------
C
   99  LL=LL+1
       IF(LL.LE.LLMAX)GO TO 30
C
C----------------------------------------------------------------------
C                              END MAIN PROGRAM
C----------------------------------------------------------------------
C
       STOP
       END
C
**********************************************************************
C                                 BLOCK DATA
**********************************************************************
C
       COMMON/INFEID/BIOLC(3,16)
       COMMON/VALUES/FCDCMP(16),TJ(16)
       DATA BIOLC/1409.,521.,1551., 1121.,414.,1234.,
      &3708.,2322.,3261., 11007.,6893.,9679., 424.,279.,307.,
      &730.,330.,240., 831.,555.,819., 2004.,1472.,1650.,
      &922.,615.,908., 2004.,1472.,1650., 861.,461.,681.,
      &664.,222.,432., 1656.,553.,1078., 5970.,4995.,6075.,
      &13930.,11655.,14175., 19150.,13750.,19250./
       DATA FCDCMP/1.00,.95,.95,.72,.81,.43,.20,.20,.15,.25,.30,
```

```
     *.30,.10,0.0,0.0,0.0/
      DATA TJ/.016,.010,.012,.007,.015,.012,.012,.012,.006,.006,
     *.007,.012,.006,.019,.006,.006/
      END
C
C***********************************************************************
C                                SUBROUTINE FEDOIL
C***********************************************************************
C
      COMMON/OIL/WSF(32,34),TARS(32,34)
      COMMON/BLKBIO/LOC,ACTBIO(16)
      COMMON/BLKOIL/OILCON(16,32,34)
      COMMON/BLK1/NE,ME,K,LL,ISL(32,34)
      COMMON/VALUES/FODCMP(16),TJ(16)
C
C-----------------------------------------------------------------------
C     SET CONSTANTS
C-----------------------------------------------------------------------
C
      RAD=0.01745329
      ALP=30.*RAD
      GKAP=175.*RAD
      VALCA=COS(ALP*K-GKAP)
      BAR=50.
      EMF=0.75
C
C-----------------------------------------------------------------------
C     COMPUTE UPTAKE OF OIL CONCENTRATION DUE TO CONTAMINATED FOOD
C     CONCENTRATIONS ARE IN PPM AND ACTBIO IS IN KG
C     FODCMP IS THE % OF PELAGIC FOOD ITEMS IN A SPECIES DIET
C-----------------------------------------------------------------------
C
      DO 99 J=1,16
C
      TCH(J)=TJ(J)+(0.35*TJ(J)*VALCA)
C
      DO 10 N=1,NE
      DO 10 M=1,ME
      IF(ISL(N,M).EQ.0) GO TO 10
      ACFRC1=1.
      ACFRC2=1.
      IF(FODCMP(J).GT.0..AND.FODCMP(J).LT.1.0) GO TO 20
      IF(FODCMP(J).EQ.1.0) GO TO 30
      IF(TARS(N,M).GT.10.0) ACFRC2=0.
      IF(TARS(N,M).GT.2.0.AND.TARS(N,M).LE.10.0) ACFRC2=.30
      IF(TARS(N,M).GT.1.0.AND.TARS(N,M).LE.2.0) ACFRC2=.70
      IF(TARS(N,M).GT.0.5.AND.TARS(N,M).LE.1.0) ACFRC2=.80
      IF(TARS(N,M).GT.0.1.AND.TARS(N,M).LE.0.5) ACFRC2=.95
      ACTCON=TCH(J)*ACTBIO(J)
      CONOIL=TCH(J)*ACFRC2*TARS(N,M)*BAR*EMF
      OILCON(J,N,M)=OILCON(J,N,M)+CONOIL
      GO TO 10
   20 IF(WSF(N,M).GT.2.0) ACFRC1=0.
      IF(WSF(N,M).GT.1.0.AND.WSF(N,M).LE.2.0) ACFRC1=.50
      IF(WSF(N,M).GT.0.5.AND.WSF(N,M).LE.1.0) ACFRC1=.75
      IF(WSF(N,M).GT.0.1.AND.WSF(N,M).LE.0.5) ACFRC1=.90
      IF(TARS(N,M).GT.10.0) ACFRC2=0.
      IF(TARS(N,M).GT.2.0.AND.TARS(N,M).LE.10.0) ACFRC2=.30
      IF(TARS(N,M).GT.1.0.AND.TARS(N,M).LE.2.0) ACFRC2=.70
      IF(TARS(N,M).GT.0.5.AND.TARS(N,M).LE.1.0) ACFRC2=.80
```

```
      IF(TARS(N,M).GT.0.1.AND.TARS(N,M).LE.0.5) ACFRC2=.95
      ACTCON=TCH(J)*ACTBIO(J)
      PEL=FOCCMF(J)
      DEM=1.0-PEL
      CONOIL=TCH(J)*(PEL*WSF(N,M)*ACFRC1+DEM*TARS(N,M)*ACFRC2)*BAR*EMF
      OILCON(J,N,M)=OILCON(J,N,M)+CONOIL
      GO TO 10
   30 IF(WSF(N,M).GT.2.0) ACFRC1=0.
      IF(WSF(N,M).GT.1.0.AND.WSF(N,M).LE.2.0) ACFRC1=.50
      IF(WSF(N,M).GT.0.5.AND.WSF(N,M).LE.1.0) ACFRC1=.75
      IF(WSF(N,M).GT.0.1.AND.WSF(N,M).LE.0.5) ACFRC1=.90
      ACTCON=TCH(J)*ACTBIO(J)
      CONOIL=TCH(J)*ACFRC1*WSF(N,M)*BAR*EMF
      OILCON(J,N,M)=OILCON(J,N,M)+CONOIL
   10 CONTINUE
C
   99 CONTINUE
C-----------------------------------------------------------------------
C                              END SUBROUTINE
C-----------------------------------------------------------------------
C
  999 RETURN
      END
```

THE UPTAKE AND DEPURATION OF PETROLEUM HYDROCARBONS IN MARINE SPECIES

A SIMULATION STUDY OF THE UPTAKE AND DEPURATION OF PETROLEUM HYDROCARBONS AND ITS EFFECT ON SELECTED MARINE SPECIES IN THE BRISTOL BAY ECOSYSTEM

by

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Figure 7.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, BCFPEL. External concentration as described in text. The vertical line at day 10 indicates the boundary between uptake and depuration and depuration only.

Figure 8.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, BCFDEM. External concentration as described in text. The vertical line at day 10 indicates the boundary between uptake and depuration and depuration only.

Figure 9.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, FODCMP. External concentration as described in text. The vertical line at day 10 indicates the boundary between uptake and depuration and depuration only.

Figure 10.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, k_2. External concentration as described in text and shown in Figure 5.

Figure 11.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, BCFPEL. External concentration as described in text and shown in Figure 5.

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1. INTRODUCTION

1.1 Purpose

The purpose of this report is to describe the basic theory and underlying assumptions and provide results from the uptake and depuration algorithm (FEDOIL) of the Biological Impact of an Oil Spill model, BIOS. The BIOS model is a multispecies ecosystem simulation that analyzes the expected impact of hypothetical oil spill scenarios on fishery resources in the eastern Bering Sea. It was developed at the request of the Outer Continental Shelf Environmental Assessment Program (OCSEAP), and is a part of their eastern Bering Sea oil impact study. A full description of the OCSEAP study of which this report is a part is given in Laevastu and Fukuhara (1984a).

As general background, BIOS is a gridded model that simulates the uptake and depuration of oil contaminants in selected marine species (Table 1) resulting from exposure to oil contaminated water and sediments and the consumption of oil contaminated food (submodel FEDOIL). BIOS also simulates the migration of these species over time and space (Swan 1984a, 1984b), studies the expected impact of two hypothetical scenarios (Table 2) (see Laevastu and Fukuhara 1984a, for details), and is applied to three locations in the Bristol Bay area of the eastern Bering Sea: Port Moller, Port Heiden, and Cape Newenham (Figure 1). (The results from Pt. Heiden are emphasized in this report.) Figure 2 provides a diagram of the general sequence of BIOS model computations. Although details are given in Gallagher (1984) and Swan (1984a), the theory and methods described here combine and update the uptake and depuration algorithms described in those preliminary formulations.

Input data for the hydrocarbon concentrations of the water soluable fraction (WSF) of each oil spill scenario were provided by the Rand Corporation in conjunction with Science Applications, Inc. (SAI) (details are given in Laevastu and Fukuhara (1984a)). Hydrocarbon concentration data for the fraction of oil

Table 1.--List of species and input biomass data (by location) used in BIOS[1].

No.	Species Name	Input Biomass Data (kg/km^2) [2] Port Moller	 Port Heiden	 Cape Newenham
1	Herring juveniles	1409	521	1551
2	Herring adults	1121	414	1234
3	Pollock juveniles	3708	2322	3261
4	Pollock adults	11007	6893	9679
5	Pacific cod juveniles	424	279	307
6	Halibut juveniles	730	330	240
7	Yellowfin sole juveniles	722	482	711
8	Other flatfish juveniles	2004	1472	1650
9	Yellowfin sole adults	800	534	789
10	Other flatfish adults	2004	1472	1650
11	Pacific cod adults	861	461	681
12	King and Bairdi crab juveniles	664	222	432
13	King and Bairdi crab adults	1654	553	1078
14	Mobile epifauna	5970	4995	6075
15	Sessile epifauna	13930	11655	14175
16	Infauna	19150	13750	19250

[1] The DYNUMES model (Laevastu and Larkins, 1981) was used to get initial estimates of input biomass data for the three model locations of the BIOS model.

[2] The following assumptions were used to convert the data obtained from the DYNUMES model to biomass fields for use in the BIOS model.

a) Unless noted differently below, the breakdown of species biomass data into juvenile and adult fractions was based on Niggol (1982).

b) DYNUMES species group 5 (halibut) was assumed to be 100% juvenile (i.e., in these shallow waters during this season).

c) Yellowfin sole data were assumed to comprise 75% of DYNUMES species group 7 (yellowfin and rock sole).

d) DYNUMES species group 13 (Pacific and saffron cod) was assumed to be 100% Pacific cod.

e) DYNUMES species groups 7 (rock sole-25%), 6 (flathead sole, flounder), and 8 (other flatfish) were combined to make up the other flatfish group (species 8 and 9) for the BIOS model. These groups were assumed to be equally divided between juveniles and adults.

f) DYNUMES species groups 19 (king crab) and 20 (Tanner crab) were combined, and using available survey data, assumed to be comprised of 71.4% adults and 28.6% juveniles.

g) DYNUMES species group 24 (epifauna) was assumed to be 30% mobile and 70% sessile.

Table 2.--Hypothetical oil spill scenarios.

Scenario	Oil type	Volume	Duration	Temperature	Simulation grid	Locations in Bristol Bay
Well blowout	Prudhoe Bay crude	20,000 bbl/day	15 days	9.3°C	(50 x 50)	Port Moller Port Heiden Cape Newenham
Tanker accident	Automotive diesel	200,000 bbl (instantaneous)	10 days	9.3°C	(32 x 34)	Port Moller Port Heiden Cape Newenham

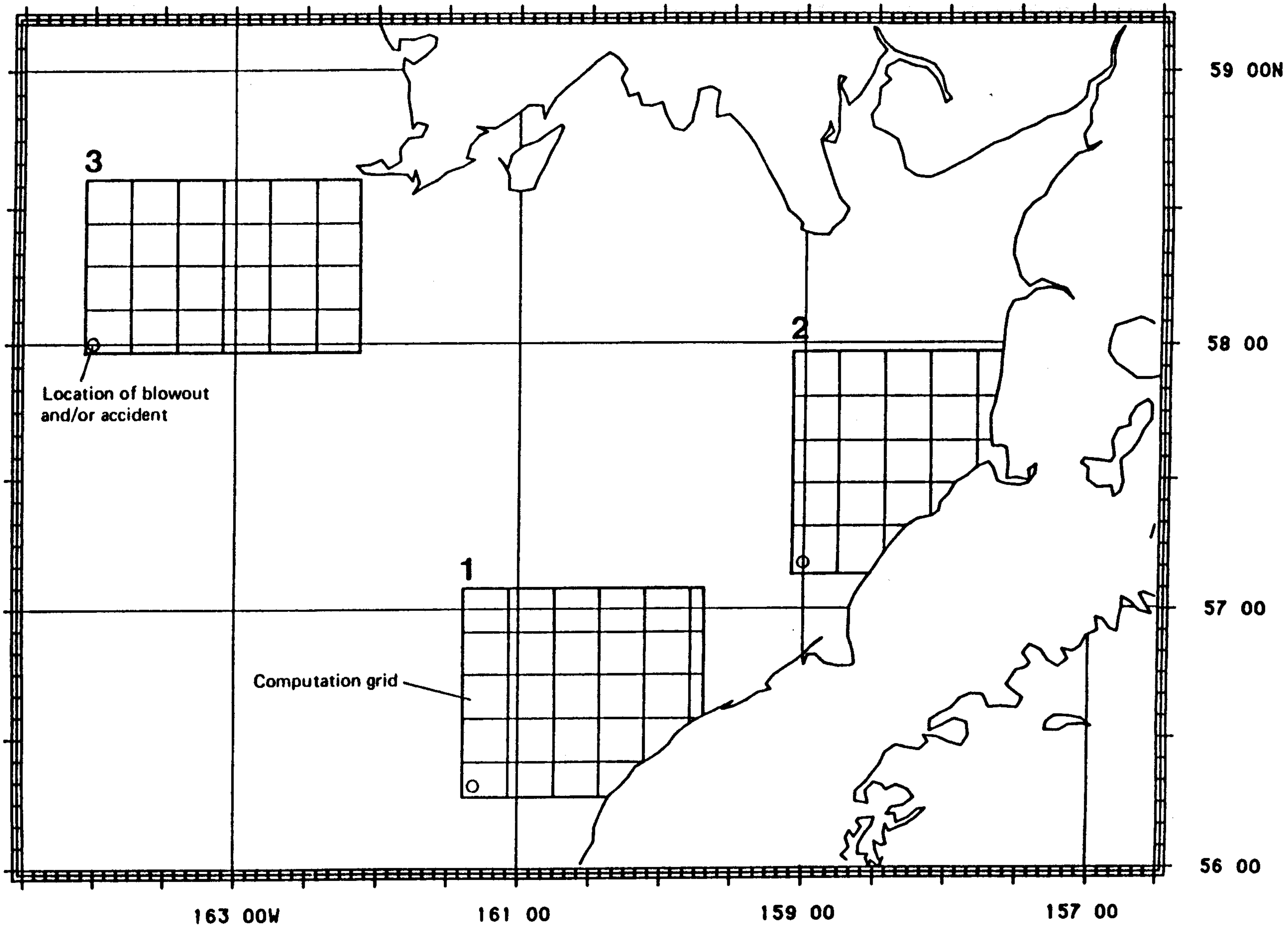
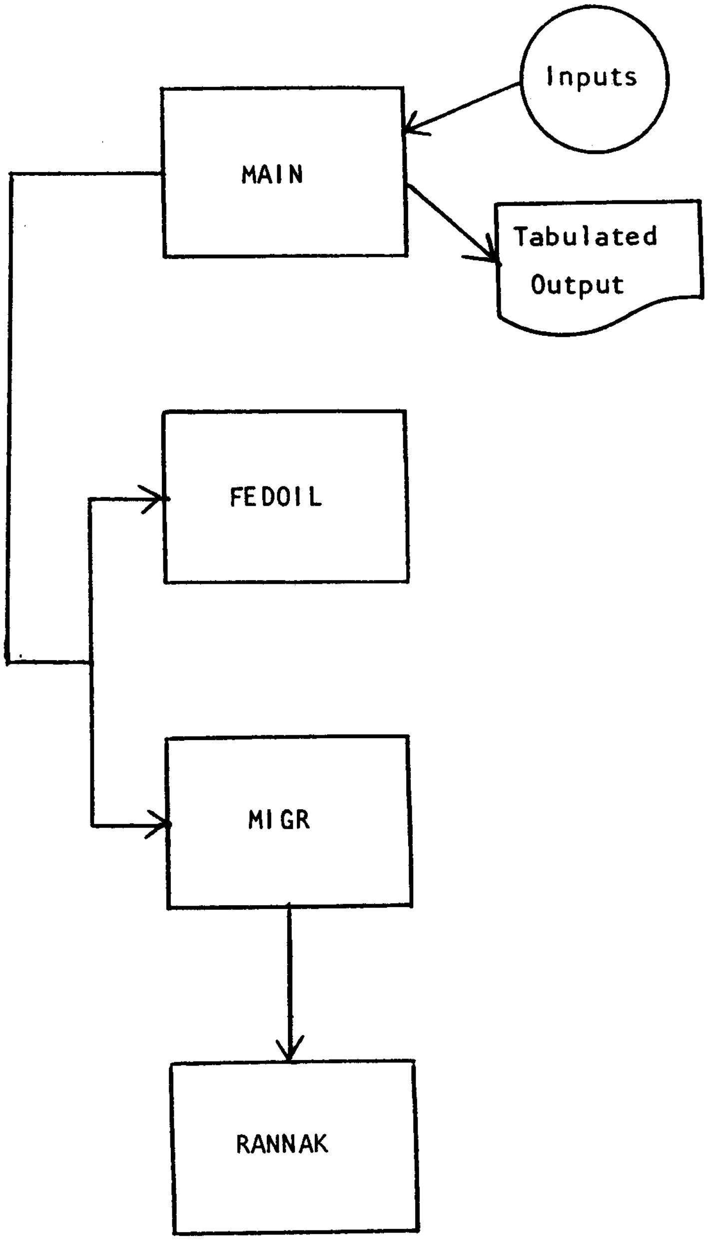


Figure 1.--Locations of hypothetical oil spills, and computational grids in Bristol Bay.



Main Program

Directs sequence of model calculations, reads input and prints output.

Feeding Subroutine

Computes uptake of contaminants through consumption of contaminated food.

Main Migration Subroutine

Directs sequence of migration computations. Sets species-specific parameters and velocities, calculates uptake from exposure to oil and depuration.

Migration Calculation Subroutine

Calculates actual migration and redistributes contamination over model grid. Calculates amount of contaminated biomass leaving the model region.

Figure 2.--Sequence of BIOS model calculations.

reaching the bottom and entering the sediments (referred to here as TARS), were obtained from a simulation model developed by Laevastu and Fukuhara (1984b).

1.2 Selected review of the literature on uptake and depuration of petroleum hydrocarbons.

An extensive literature exists on the fate and effects of petroleum hydrocarbons on marine organisms. Since a variety of authors have recently reviewed this literature (Malins 1977; Wolfe 1977; Connell and Miller 1981a, 1981b; National Academy of Science 1982), this discussion will not attempt to repeat those earlier works. Instead, it will confine itself to reviewing those studies pertinent to the modelling approach used in the BIOS model to simulate the processes of uptake and depuration.

For purposes of this discussion, uptake is defined as the acquisition of petroleum hydrocarbons by an organism either from exposure to oil contaminated water and sediments or from consumption of oil contaminated food. Depuration is defined as the purging of those hydrocarbons from the organisms, both during the uptake process and when the organism is no longer exposed to petroleum contaminants. For a variety of reasons discussed below, no attempt has been made to simulate the disposition of petroleum compounds after uptake; disposition being "what the organism does with a compound (e.g., their conversion to various metabolities)" (Malins and Hodgins 1981).

1.2.1 Uptake

Petroleum hydrocarbons have been shown to accumulate in the tissues and body fluids of many, if not all, marine organisms (Moore and Dwyer 1974, Malins and Hodgins 1981). Although the routes through which hydrocarbons enter marine organisms vary depending on species, life-history stage, and environmental conditions, they can be grouped into two general categories: 1) uptake directly from contaminated water and sediments; and 2) accumulation through consumption of contaminated food (Connell and Miller 1981a; Thomann and Connolly 1984). The relative importance of each route also varies considerably, both by species group and by the actual bioavailability of the petroleum hydrocarbons involved; e.g., chemical compound, concentration, length of exposure, and medium (i.e., whether the compound is dissolved in the water column, adsorbed on particulate sediments, or bound up in food).

1.2.1.1 Benthic Invertebrates

Benthic invertebrates have been shown to readily uptake petroleum hydrocarbons. Bivalves, which filter large volumes of water when feeding, can uptake and concentrate petroleum hydrocarbons from water, whether in solution or absorbed on suspended particles (see Lee 1977, for review). They have also been shown to bioaccumulate hydrocarbons to a level several orders of magnitude above the external concentration (Stegman and Teal 1973, Fossato and Canzonier 1976). Although bivalves tend to accumulate petroleum hydrocarbons more slowly than fish or crustacea (Neff et al., 1976), several studies show that they continue to do so for as long as they are exposed to oil-contaminated seawater (Stegman and Teal 1973; Neff et al., 1976).

As reviewed by both Connell and Miller (1981a) and the National Academy of Science Report (1982), several uptake experiments with the oyster, Crassostrea virginica demonstrate that oysters tend to accumulate higher concentrations of aromatic hydrocarbons than saturated hydrocarbons relative to their respective concentrations in exposure water during the initial uptake phase. Although similar results have been reported for the clam, Rangia cuneata (Neff et al., 1976), rates of uptake differ between species and appear to be related to differences in filtering rates and amounts of lipids in the organisms (Lee 1977), and the water solubilities and molecular weights of the specific hydrocarbon pollutants (Lee 1977; Varanasi and Malins 1977). As will be discussed later, however, it is rather difficult to compare data obtained from different studies because of the considerable variability in experimental technique and type and composition of petroleum compounds used. In fact, the review by Varanasi and Malins (1977) is one of the few studies that divides the experiments reviewed into categories reflecting field studies, laboratory studies using oil-in-water dispersions (OWD) and water-soluble fractions (WSF) of oil, and feeding studies involving petroleum contaminated food.

Benthic crustaceans have been shown to rapidly take up petroleum hydrocarbons from either their food or water (Lee et al., 1976, Neff et al., 1976, Rice et al., 1976, Rice et al., 1983). As with bivalves, the rate and amount of petroleum hydrocarbons accumulated appears to be related to internal lipid content and the different solubilities of the individual petroleum constituents (see Connell and Miller 1981a, for review). The present data, however, do not allow for a clear quantitative partitioning of the uptake process between the routes of feeding and exposure to oil-contaminated water or sediments. For example, Rossi et al. (1978), as reported in Connell and Miller (1981a), indicated that it was impossible to establish whether sand crabs, Emerita analoga, incorporated petroleum hydrocarbons into their tissue or superficially entrained contaminated particulate matter. In addition, Lee et al. (1976) have shown that in the case of the blue crab, Callinectes sapidus, most of the hydrocarbons in the food were not assimilated by the tissues, but instead were immediately eliminated from the animal.

The data for benthic worms are no less confusing. Although benthic worms have clearly been shown to uptake petroleum hydrocarbons, the amount and rate of uptake can vary depending on hydrocarbon constituent and sediment type (Lee 1977, for review). In addition, the actual route of uptake of the hydrocarbons is unclear. Rossi (1977) has reported that most of the aromatic hydrocarbons accumulated by the polycheate, Neanthes arenaceodentata, were derived from water and not sediments, while Prouse and Gordon (1976) indicated that the burrowing activities of the deposit feeding polycheate, Arenicola marina, in sediments may result in uptake from either ingestion of contaminated sediments or through absorption from solution. A variety of other studies indicate that polycheate annelids also vary in sensitivity to fuel-oil soluble fractions at different life stages according to lipid content (e.g., Rossi and Anderson 1976). Moreover, and depending on the study, certain aromatic hydrocarbons (e.g., naphthalenes), have been shown both to accumulate rapidly (Rossi 1977) and not to accumulate to significant levels at all (Anderson et al., 1977).

1.2.1.2 Fish

The principal processes for the uptake of hydrocarbons in fish appear to involve either direct absorption of dissolved and particulate forms via gills or drinking water, or indirect uptake through the ingestion of contaminated food (Connell and Miller 1981a, for review). As in the case of benthic invertebrates, however, the data on uptake in fish are rather contradictory. For example, uptake has been shown to be selective within and between hydrocarbon classes (Connell and Miller 1981a, for review), and within and between species depending on life history stage and ecological niche (i.e., pelagic or demersal) (Korn et al., 1976, Lee 1977; Connell and Miller 1981a, and National Academy of Science 1982, for reviews). In addition, although a variety of authors have concluded that there is a greater storage and persistence of aromatics and polynuclear aromatic hydrocarbons in lipid-rich than in lipid-poor fish species (Whittle et al., 1977, Connell and Miller 1981a, for review), a study by Roubal et al., (1978) indicates that, for aromatic hydrocarbons, factors other than lipid content may be more influential in determining hydrocarbon accumulation in certain species. Roubal et al., 1978 also indicate that because of the great differences in bioconcentration factors observed for individual aromatic hydrocarbons in both of the species they studied (coho salmon, Oncorhynchus kisutch, and starry flounder, Platichthys stellatus),"these differences may complicate attempts to relate tissue hydrocarbon profiles to hydrocarbon profiles of specific sources of petroleum pollution".

The problem of relating tissue hydrocarbon profiles to sources of hydrocarbon contamination in fish is further complicated by the conflicting reports regarding the relative importance of the uptake routes of feeding and exposure to oil (see Lee 1977 and Connell and Miller 1981a, for reviews). For example, feeding behavior and the presence of oil may be interdependent, as shown by the enhanced weight loss and distinct reduction in food intake by oil exposed flatfish (McCain et al., 1978, Fletcher et al., 1981). Additionally, and with respect to specific feeding studies, Mehrle et al. (1977) have shown that the type and quality of diet fed during chronic toxicity testing can strongly influence the results of the biological parameters being measured (e.g., mortality, growth, development, etc.). Finally, not only is it impossible to compare oil toxicities and animal sensitivities in different studies done prior to 1973 because of the lack of data on the chemical analyses of oil-water solutions (Rice et al., 1979), but results from many of the effects studies have been obtained from experiments using relatively high concentrations that probably would not be encountered in the marine environment (Malins and Hodgins 1981).

1.2.1.3 Summary of uptake studies

The available data on uptake rates and accumulation of petroleum hydrocarbons in marine organisms are confusing, contradictory, and in the case of some studies, provide results that may not be representative of events that occur in the natural, multifaceted conditions found in the marine environment (Malins and Hodgins 1981). Consistent data have been presented, however, that demonstrate the importance of lipid content and petroleum water solubilities in the bioaccumulation of hydrocarbons in both benthic invertebrates and fish. These topics and the general subject of estimating uptake rates will be considered in more detail in Section 2.

1.2.2 Depuration

Depuration of petroleum hydrocarbons from marine organisms is a complex process that varies within and between species and hydrocarbon compounds and with environmental conditions. The actual pathways of depuration are unclear, but seem to be related to the mode of uptake (e.g., absorption from solution, feeding, etc.). Any understanding of the depuration processes is considerably confounded, however, by the degree to which acquired hydrocarbons are accumulated and retained as conversion byproducts. In addition, as in the case of uptake rates, conflicting information on depuration rates seems, oftentimes, to be as much a function of differences in experimental design as it is a function of differences in either hydrocarbon or species specific biochemical processes.

1.2.2.1 Benthic invertebrates

As reviewed by Lee (1977) and Connell and Miller (1981a), most depuration studies indicate that bivalves release accumulated petroleum hydrocarbons when placed in clean or oil-free seawater. After an initial phase of rapid discharge, there is an extended period of residual hydrocarbon retention. The initial rapid discharge usually results in the calculated short half-lives for accumulated hydrocarbons (Lee 1977). For example, Stegman and Teal (1973) report a 90% loss of petroleum hydrocarbons from high-fat-content oysters (_C. virginica_) after 14 days of depuration in clean seawater. Stored petroleum hydrocarbon concentration levels, however, were still above the background levels of 1 ppm after 4 weeks. Although several other studies reviewed by Connell and Miller (1981a) also report depuration clearance after 14 days in clean seawater, Fossato and Canzonier's (1976) study of the mussel, _Mytilus edulis_, indicated that mussels still retained petroleum hydrocarbon concentrations of 30 ppm after 56 days of depuration.

The major difficulty in using depuration rates of petroleum hydrocarbons from bivalves obtained under experimental conditions is the fact that bivalves in oil spill areas generally depurate more slowly. This is due, in part, to the continued input of oil from the sediment. Lee (1977) reports that for oysters, the longer the period of uptake, the slower the depuration of the accumulated petroleum hydrocarbons. In addition, while many calculated biological half-lives from laboratory experiments range between 1 and 7 days, results from field experiments suggest considerably longer half-lives (i.e., 48-60 days; DiSalvo et al., 1975) for aromatic hydrocarbons in particular. Although this increased retention time for aromatic hydrocarbons may be related to passive diffusion between lipids and the aqueous phase, as expressed by lipid/water partition coefficients (Stegman and Teal 1973, Neff et al., 1976), an additional hypothesis has been proposed by Stegman and Teal (1973) that suggests that for chronically exposed bivalves the same accumulated hydrocarbons enter a stable tissue compartment where they are retained and released slowly during depuration in clean seawater.

Connell and Miller (1981a) reviewed studies by several other workers (e.g., Neff et al., 1976) that also suggest this latter explanation for the rapid initial loss of hydrocarbons and retention of a small persistent fraction in depuration studies.

The more important factor in the storage of aromatic hydrocarbons in bivalves, however, is probably the absence of detectable aryl hydrocarbon hydroxylases (AHH) activity. As reviewed in Varanasi and Malins (1977), it is generally accepted that the metabolism of aromatic hydrocarbons is mediated by cytochrome P_{450}-dependent enzyme systems (mixed-function oxidases; MFO), and that these oxygenases, or drug-metabolizing enzymes, are believed to account for the formation of virtually all of the primary metabolic products of aromatic hydrocarbon degradation. Since it appears that mollusks do not possess the systems necessary for the metabolism of aromatic hydrocarbons and their subsequent excretion as the more water-soluble hydroxylation products, the ability of bivalves to store and retain petroleum hydrocarbons for considerable periods of time is probably directly related to this apparent lack of MFO activity. As discussed below, such biological and biochemical complexity only further complicates the already difficult task of modelling the uptake and depuration of petroleum hydrocarbons in marine organisms.

Benthic crustaceans have been generally shown to depurate petroleum hydrocarbons rather rapidly when placed in clean seawater (i.e., in 2 to 10 days). The information is not as clear, however, with respect to the depuration of petroleum hydrocarbons in an oil-spill area. Lee et al. (1976) have suggested that crabs should not retain petroleum hydrocarbons in an oil-spilled area, except for very recent uptake, due to their high metabolic and excretion rates. This position is supported by results from their experiments with the blue crab, Callinectes sapidus, in which they found no evidence of storage of hydrocarbons by any crab tissue. Rice et al. (1983), however, report preliminary results from their studies with king crab, Paralithodes camtschatica, exposed to water soluble fractions (WSF) of crude oil that indicate site specific uptake and retention of petroleum hydrocarbons; i.e., although the crabs had virtually no naphthalene in their gill tissues, viscera concentrations of naphthalene were 1200 times the naphthalene concentrations in the WSF. In addition, Burns (1976), as reported in Lee (1977), noted that the fuel-oil hydrocarbon body burden in intertidal fiddler crabs, Uca pugnax, lasted for up to four years in an area where sediments were contaminated by an actual oil spill. This suggests that the crabs continued to take up oil from either the contaminated sediments or from oil released from the sediments. In either case, the complex nature of hydrocarbon retention and depuration in crabs in the natural environment makes it difficult to directly extrapolate experimental findings on depuration rates to field situations.

The depuration of petroleum hydrocarbons in benthic worms is generally rapid. Depending on species and hydrocarbon compound, tissue body burdens of petroleum hydrocarbons have been shown to drop to background levels in 14 to 24 days when benthic worms were placed in clean seawater (Lee 1977, Connell and Miller 1981a, for reviews). Although neither reviewer provided information on depuration rates in the presence of oil contaminated sediments, each indicated that benthic worms have well developed enzyme systems that rapidly metabolize petroleum hydrocarbons. One study by Anderson et al. (1977), however, reports that tissue concentrations of naphthalenes in sediment-exposed sipunculid worms, Phascolosonia agassizii, were comparable to those found in the contaminated sediments. Thus, despite the fact that both the water- and sediment-exposed worms from the Anderson et al. (1978) study released accumulated naphthalenes to background levels after 14 days depuration, the long term effects of continued hydrocarbon exposure on depuration rates is left unclear.

1.2.2.2 Fish

The depuration of petroleum hydrocarbons from fish usually takes between 7 to 14 days when organisms are placed in clean seawater (Lee 1977). As in the case of uptake, however, depuration has been shown to be selective within and between species and hydrocarbon classes (Korn et al., 1976, Roubal et al., 1978). Korn et al. (1976), for example, reported that when fish were placed in clean seawater substantial depuration occurred within 7 to 14 days but, for some naphthalenes and higher-molecular-weight aromatics, a significant residual fraction (about 1 to 10%) was retained for longer periods (see Connell and Miller 1981a, for a review of this topic).

Fish have active enzyme systems (MFO) that can metabolize aromatic hydrocarbons rather rapidly to water-soluble compounds. This process facilitates the removal of toxic hydrocarbons from the body, and as Rice (1981) points out, these already active enzyme systems have been shown to increase after exposure to petroleum hydrocarbons. Several studies, however, have shown that some of the resulting metabolities persist in tissues longer than the parent hydrocarbons (Roubal et al., 1977, Varanasi et al., 1979). Varanasi et al. (1979) has shown also that the extent of biotransformation of naphthalene and the types of metabolities remaining in tissues of flatfish are greatly influenced by both mode of exposure and the time elapsed after the exposure is initiated.

In a follow-up study, Varanasi et al. (1981) further indicated that, in general, lower water temperature increased tissue concentrations of both the parent hydrocarbon (naphthalene) and its metabolities. They pointed out, however, that the actual magnitude of the increase was dependent upon the hydrocarbon compound, the tissue, and the time after the initiation of the exposure. Clearly, the complex nature of the process of retention of petroleum hydrocarbons and their conversion byproducts only further complicates attempts at understanding the depuration process in marine fish.

1.2.2.3 Summary of Depuration Studies

The complex nature of the depuration process and the variability in reported depuration rates, particularly between field and laboratory data, makes any simulation of the depuration of petroleum hydrocarbons a fundamentally qualitative undertaking. This is particularly apparent when one considers the facts upon which most investigators agree; i.e., that depuration rates under actual oil spill conditions are most likely altered and determined by complex interactions between the size of the spill, type of oil, the species and its physiological state, and the existing environmental and hydrodynamic regimes (Lee 1977, Connell and Miller 1981a). As discussed in Section 2, such a complex of factors considerably limits the set of reasonable approaches available for modelling the depuration process.

2. METHODS

2.1 Modelling approaches to the uptake and depuration of petroleum

The various approaches taken in modelling the uptake and depuration of organic compounds in marine species have ranged from simple and direct methods based on first-order kinetics (e.g., Branson et al., 1975), to more complex methods based on the coupling of pollutant biokinetics with fish bioenergetics (e.g., Norstrom et al., 1976). Although each of these approaches has a certain elegance in theory (the latter models in particular), each has been "frought with difficulty because of the paucity of some parameter values" (Hallam and deLuma 1984). In addition, the confusing and oftentimes conflicting results of laboratory and field investigations with respect to the relative importance of uptake from feeding and uptake from exposure to oil-contaminated water or sediments (see Section 1 above), has further complicated the problem of modelling the marine system.

In order to simplify the modelling approach taken here, the uptake of an oil pollutant is assumed to represent the uptake from both feeding and exposure to oil-contaminated water and sediments. Although this approach ignores the predator-prey dynamics of the ecosystem, it circumvents the problem of estimating the many bioenergetic rate parameters needed for the model, recognizing that these rate constants may vary with environmental conditions. In addition, since "we have more gaps than knowledge about the foodweb transfer of hydrocarbons in the ocean" (Teal 1977), the approach taken here further avoids the problem of trying to partition pollutant uptake between feeding and exposure to oil contaminants, a process already complicated by the fact that marine organisms have been shown to have decreased feeding rates when exposed to sublethal concentrations of petroleum.

The model used in this analysis, submodel FEDOIL, will study the total bioaccumulation of a pollutant in an organism. Bioaccumulation is defined to occur when the rates of uptake and redistribution exceed the rates of metabolism and elimination. The modelling approach is based on simple first-order kinetics (Atkins 1969, Moriarity 1975, and Wilson 1975), and can be described by a simple two-compartment (water and organism) reversible reaction model (Branson et al., 1975, Eberhardt 1975, Blanchard et al., 1978). (Banerjee (1984) uses the same approach but refers to it as a one-compartment pharmacokinetic model.) It is given as:

$$C_w \underset{k_2}{\overset{k_1}{\rightleftharpoons}} C_f \qquad (1)$$

where C_w is the concentration in the water, C_f is the concentration in the fish (or other marine organism), and k_1 and k_2 are rate constants for the movement of the pollutant into and out of the fish, respectively (see Figures 3 and 4).

As Moriarity (1975) points out, this approach, although mathematically convenient, is unrealistic in that it assumes a whole organism can be considered as a single compartment. This criticism has been voiced also by Atkins (1969), Wilson (1975), and a variety of field workers such as Stegman and Teal (1973), as reviewed in Connell and Miller (1981a). Most of the available data, however, can only be fitted to an equation with a single exponential (Moriarity 1975), a point borne out by Eberhardt's (1975) inability to fit the "more complex models thus required to data of the kind reported" in the studies he reviewed. MacKay and Hughes (1984) also found that "model complexity greatly exceeds the detail of the experimental information", and thus found it necessary to

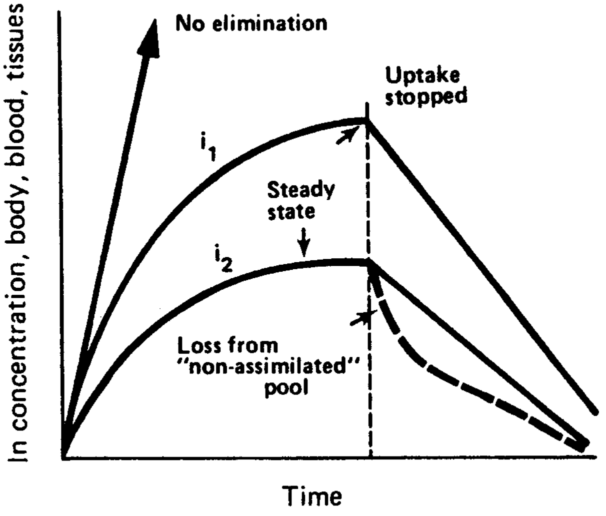


Figure 3.--Generalized relations of uptake and depuration with time at different levels of exposure. $i_1 > i_2$. Broken line indicates depuration from two (or more) compartments (from Moriarity 1975).

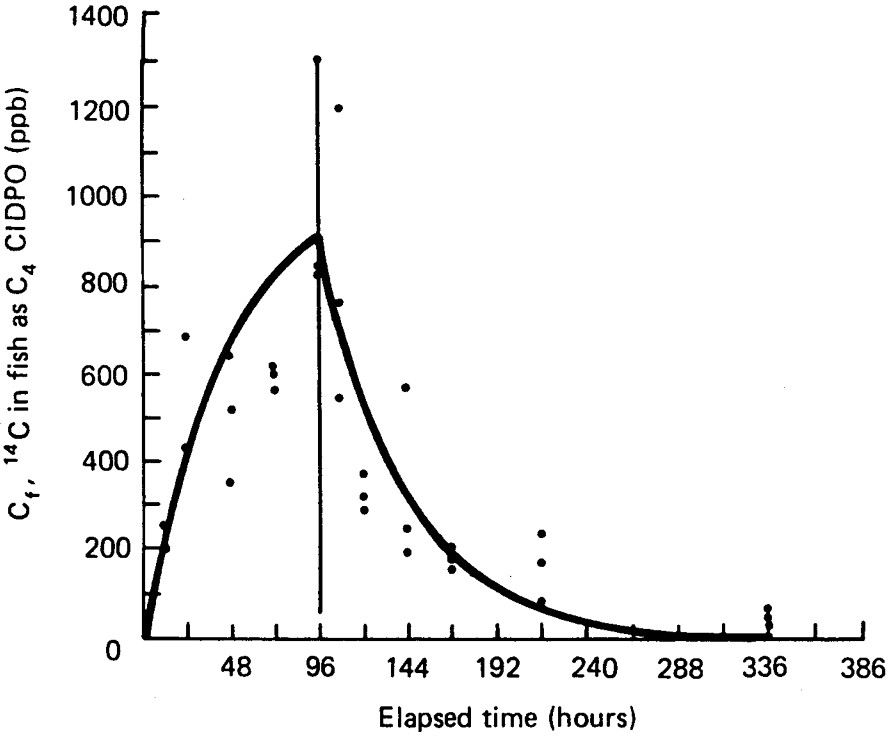


Figure 4.--Uptake and clearance curve of high exposure of C_4ClDPO in trout as linear plot with rate constants from nonlinear regression analysis. Points are concentrations in individual fish. The average concentration in water was 3.6 ppb. The vertical line at 96 hrs indicates the boundary between uptake and clearance. $k_1 = 6.05 \pm 0.98$; $k_2 = 0.0207 \pm 0.0041$. Bioconcentration factor, BCF = 292 ± 75 (from Blanchard et al., 1977).

"introduce simplifying assumptions to reduce the number of parameters". Given the lack of data available for parameterizing the more complex multi-compartment models, the one-compartment model used here seems reasonable as a first order approximation.

The change over time of the internal concentration, C_f, is given by:

$$dC_f/dt = (k_1)(C_w) - (k_2)(C_f) \tag{2}$$

with the solution (for C_w constant):

$$C_f(t) = (k_1/k_2)(C_w)(1-\exp(-k_2t));\ C_f(0) = 0 \tag{3}$$

If the initial concentration of the organism, $C_f(0)$, is not zero, then we have:

$$C_f(t) = (k_1/k_2)(C_w)(1-\exp(-k_2t)) + C_f(0)\exp(-k_2t) \tag{4}$$

This equation assumes a steady state condition as t approaches infinity, such that when $dC_f/dt = 0$, Equation 2 becomes:

$$\frac{C_f}{C_w} = \frac{k_1}{k_2} = BCF \tag{5}$$

where BCF defines the bioconcentration factor (bioaccumulation rate) at steady state.

One of the disadvantages of using this steady state approach is the assumption of a constant external concentration, C_w. Since the simulated external oil concentration data used in this study (as supplied by the Rand corporation; see Laevastu and Fukuhara 1984a, for a discussion) change over time and space, Equation 4 was adjusted to better reflect the dynamic nature of the system. The form used in the BIOS model is given as:

$$C_f(t) = (k_1/k_2(C_w(t))(1-\exp(-k_2)) + C_f(t-1)\exp(-k_2) \tag{6}$$

$$C_f(0) = 0 \tag{7}$$

By replacing the initial concentration, $C_f(0)$, of Equation 4 by the internal concentration of the previous time step, $C_f(t-1)$, and then removing the variable of time, t, in the exponent of the exponential, the Equation Set 6-7 gives a reasonable finite difference approximation of the uptake and depuration of oil contaminants when the external concentration, C_w, is not constant. Test runs of the BIOS model comparing simulation outputs from Equation 4 with those of the Equation Set 6-7 give identical results for the case $C_w(t) = C_w$, constant. Since Equation 4 is clearly not applicable to the case where C_w is changing over time ($C_w = C_w(t)$), the Equation Set 6-7 seems appropriate for the case when the external concentration is time dependent.

External concentration data, $C_w(t)$, are given for both the water soluable fraction (WSF), $C_w(t)_{WSF}$, and for the fraction of the oil that sedimentizes to the bottom (TARS), $C_w(t)_{TARS}$. Since marine organisms may be affected by either one or both of these pollutant levels depending on life history, it was necessary to compute a composite value of external concentration that reflected the relative exposure of a species to the two types of external concentration data. Since a species' feeding behavior can be generalized into the fraction of pelagic and demersal food in its diet, this composite value was also adapted to address the differential feeding behavior of an individual species. Before discussing this topic further, however, it is necessary to make some additional comments regarding the rate constants k_1 and k_2.

Methods for obtaining realistic parameter values for the uptake and depuration rate constants, k_1 and k_2, were complicated by a variety of factors. First, species-specific rates often are lacking and, when available, are usually limited to the specific experimental situation (i.e., time of exposure, experimental system design, temperature), making it difficult to transfer the results to field situations (Malins and Hodgins 1981). Second, most studies work with only very small fish when studying uptake and depuration rates (Eberhardt 1974, Hamelink 1977). Several studies, however, suggest that experiments with larger fish will give substantially different results (Hamelink and Waybrant 1976, Anderson and Weber 1975, Thomann 1981, Thomann and Connolly 1984). Third, although lower water temperature has been shown to increase tissue concentrations of both parent hydrocarbons and their metabolities (Varanasi et al., 1981), no direct function has been developed relating the magnitude of accumulation with temperature (Fossato and Canzonier 1976, Rice et al., 1977). In addition, several studies have shown that the lowering of water temperature significantly influences the rate of elimination of individual hydrocarbons such as naphthalene (Collier et al., 1978, Varanasi et al., 1981). Fourth, the conversion of accumulated hydrocarbons to byproducts that may also accumulate but go undetected limits any attempts to simulate the depuration process. Finally, and most importantly, the considerable differences in bioconcentration factors observed for individual aromatic hydrocarbons seriously complicate attempts to relate tissue hydrocarbon profiles to hydrocarbon profiles of specific sources of petroleum pollution (Roubal et al., 1978).

In order to address these problems, particularly the latter, it was necessary to make several simplifying assumptions in estimating the values of k_1 and k_2. As shown in Equation 5, the bioconcentration factor, BCF, can be estimated from the ratio of k_1 to k_2. Similarly, $k_1(k_2)$ can be estimated if values for BCF and $k_2(k_1)$ are available. Since k_1 values were the most difficult to obtain from the literature, it was decided not to use explicit uptake rates in this analysis but instead to rewrite equation (6) as:

$$C_f(t) = \text{VALUE}\ (1-\exp(-k_2)) + C_f(t-1)\exp(-k_2) \tag{8}$$

where the variable VALUE is calculated according to the pelagic or demersal nature of the species. For the general case, VALUE is given as:

$$\text{VALUE} = (\text{PEL})(\text{BCFPEL})(C_w(t)_{WSF}) + (\text{DEM})(\text{BCFDEM})(C_w(t)_{TARS}) \tag{9}$$

where PEL and DEM are the fraction of pelagic and demersal food, respectively, in a species diet (PEL is set equal to FODCMP, the fraction of pelagic food, and DEM = 1.0 - PEL), and BCFPEL and BCFDEM are the pelagic and demersal bioconcentration factors, respectively (see discussion below).

The depuration parameter, k_2, can now be estimated from either the reported total depuration time of all hydrocarbons from an organism after being placed in clean water via the equation:

$$k_2 = [-\ln(C_f(0)/C_f(t)_c)]/t \tag{10}$$

where $C_f(t)_c$ is the total concentration in the organism just prior to being placed in clean water; or from data on the biological half-life of the hydrocarbon contaminant via the equation:

$$k_2 = \frac{\ln 2}{t(1/2)} \tag{11}$$

where t(1/2) is the biological half-life (Wilson 1975, Connell and Miller 1981a). (See Table 3).

With regards to the bioconcentration factor, BCF, and its pelagic and demersal components, BCFPEL and BCFDEM, a variety of investigators have shown that BCF can be estimated from either the n-octanol water partition coefficient (Neely et al., 1974, Veith et al., 1979), or from the water solubility (Chiou et al., 1977,

Table 3.--Depuration rate (k_2) data used in submodel FEDOIL.

Species type	Depuration half-life or total time in days	Estimated k_2 value used in FEDOIL	Source of data
Pelagic juvenile	2-7 a)	.1980	a) total time - Korn et al. 1976
Pelagic adult	7-14 a)	.1320	a) total time - Lee 1977
Semi-pelagic juvenile	2-7 a)	.1980	a) total time - Korn et al. 1976
Semi-pelagic adult	7-14 a)	.1320	a) total time - Lee 1977
Flatfish juvenile	4.2 a) ≤ 51 b)	.1664	a) half-life - Roubal et al. 1978 b) total time - McCain et al. 1978
Flatfish adult	≤ 51 a)	.1109	a) total time - McCain et al. 1978
King crab juvenile	2.1 a) 2-10 b)	.3342	a) half-life - Lee et al. 1978 b) total time - Lee 1977
King crab adult	2-10 a)	.2228	a) total time - Lee 1977
Mobile epifauna	3-4 a)	.1980	a) total time - Anderson 1977
Sessile epifauna	16 a) 28-35 b)	.0346	a) half-life - Lee 1977 b) total time - Lee 1977
Infauna	10 a) 12-14 b)	.06930	a) half-life - Lee 1977 b) total time - Lee 1977

Spacie et al., 1979) of the hydrocarbon. Since "water solubility is usually the most available measured parameter and probably the most practical for early assessment of potential bioconcentration hazard" (Kenaga and Goring 1980), the BCF values used in this analysis are estimated according to Kenaga and Goring (1980) via the equation:

$$\log BCF = 2.791 - 0.564(\log WS) \tag{12}$$

where WS is the water solubility in parts per million (ppm) of the specific hydrocarbon in question (for a review of the relevant theory of partition coefficients and water solubility, see Chiou 1981). The BCFPEL and BCFDEM values are then set equal to the calculated BCF of Equation 12. Each value could, of course, be set individually if the data so indicated; for example, BCFDEM is set equal to twice BCF for mobile and sessile epifauna, species 14 and 15, due to their high bioconcentration rates.

Since different hydrocarbon compounds have order of magnitude differences in their water solubilities (see Tables 4 and 5), a water solubility index (WS) was used to compute the BCF from Equation 12 (Table 6). This water solubility index represents those hydrocarbon compounds that are the most significant oil contaminant fractions resulting from an oil spill and that have been demonstrated to be most toxic to, and accumulated by, marine organisms (i.e., naphthalenes). Using data from several sources (Clark and Brown 1977, Payne et al., 1984), the naphthalene fraction of the total hydrocarbons reported in the WSF external concentration data supplied by the Rand Corporation (a breakdown of hydrocarbon components was not provided), was assumed to be approximately 50% of the total for both scenarios. The naphthalene fraction of total hydrocarbons simulated for the TARS external concentration data (Laevastu and Fukuhara 1984b), was

Table 4.--Solubility of selected aromatic petroleum hydrocarbon in water [a].

Compound	Carbon number	Solubility[b] (ppm)
Benzene	6	1,780
Toluene	7	515
O-Xylene	8	175
Ethylbenzene	8	152
Naphthalene	10	31.3 22.0 (SW)
1 - Methylnaphthalene	11	25.8
2 - Methylnaphthalene	11	24.6
2 - Ethylnaphthalene	12	8.00
1,5 - Dimethylnaphthalene	12	2.74
2,3 - Dimethylnaphthalene	12	1.99
2,6 - Dimethylnaphthalene	12	1.30

a) - Adapted from Clark and McLeod (1977).

b) - In distilled water, except where noted by (SW), indicating filtered seawater, usually corrected to a salinity of 35 $^{o}/oo$ (parts per thousand); ppm=parts per million - micrograms per gram.

Table 5.--Hydrocarbon content of water-soluble fractions of four test oils [a].

Compound	Hydrocarbon content of water-soluble fraction (ppm)			
	S. Louisiana crude oil	Kuwait crude oil	No. 2 fuel oil	Bunker C residual oil
Alkanes				
Ethane	0.54	0.23	-[b]	-
Propane	3.01	3.30	-	-
Butane	2.36	3.66	-	-
Isobutane	1.69	0.90	0.39	0.05
Pentane	0.49	1.31	-	-
Isopentane	0.70	0.98	-	-
Cyclopentane + 2-methylpentane	0.38	0.59	0.02	0.005
Methylcyclopentane	0.23	0.19	0.019	0.004
Hexane	0.09	0.29	0.014	0.004
Methylcyclohexane	0.22	0.08	0.03	0.002
Heptane	0.06	0.09	0.02	0.004
C_{16} n-Paraffin	0.012	0.0006	0.008	0.0012
C_{17} n-Paraffin	0.009	0.0008	0.006	0.0019
Total C_{12}-C_{24} n-paraffins	0.089	0.004	0.047	0.012
Aromatics				
Benzene	6.75	3.36	0.55	0.04
Toluene	4.13	3.62	1.04	0.08
Ethylbenzene + m-, p-xylenes	1.56	1.58	0.95	0.09
O-Xylene	0.40	0.67	0.32	0.03
Trimethylbenzenes	0.76	0.73	0.97	0.11
Naphthalene	0.12	0.02	0.84	0.21
1-Methylnaphthalene	0.06	0.02	0.34	0.19
2-Methylnaphthalene	0.05	0.008	0.48	0.20
Dimethylnaphthalenes	0.06	0.02	0.24	0.20
Trimethylnaphthalenes	0.008	0.003	0.03	0.10
Biphenyl	0.001	0.001	0.011	0.001
Methylbiphenyls	0.001	0.001	0.014	0.001
Dimethylbiphenyls	0.001	0.001	0.003	0.001
Fluorene	0.001	0.001	0.009	0.005
Methylfluorenes	0.001	0.001	0.009	0.004
Dimethylfluorenes	0.001	0.001	0.002	0.002
Dibenzothiophene	0.001	0.001	0.004	0.001
Phenanthrene	0.001	0.001	0.010	0.009
Methylphenanthrenes	0.002	0.001	0.007	0.011
Dimethylphenanthrenes	0.001	0.001	0.003	0.003
Total saturates	9.86	11.62	0.54	0.081
Total aromatics	13.90	10.03	5.74	1.28
Total dissolved hydrocarbons measured	23.76	21.65	6.28	1.36

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[a] Adapted from Varanasi and Malins (1977).

[b] Showed unresolved GC peaks, probably includes some olefins.

Table 6.--Water solubility index (WS) of total naphthalenes used in computing BCF in Equation 12 [a].

Hydrocarbon	Water solubility (ppm) [b]
Naphthalene	22.0
1 - Methylnaphthalene	17.23
2 - Methylnaphthalene	16.43
1,5 - Dimethylnaphthalene	1.83
2,3 - Dimethylnaphthalene	1.33
2,6 - Dimethylnaphthalene	.868
Mean	9.949

Mean water solubility index (WS) = 9.949

log (WS) = .9978

log BCF = 2.228 (from Equation 12)

BCF = 170

---

a) Concept of total naphthalenes taken from Anderson et al., 1977.

b) Estimated to represent water solubility in filtered seawater; i.e., see Table 4.

assumed to be 10% of the total for both scenarios. Although this use of a water solubility index for naphthalenes further underscores the qualitative nature of this analysis, the lack of data available on the specific hydrocarbon composition of each oil spill scenario made a finer analysis impossible and, if attempted, would have contributed little to making the results more precise.

Although the methods used in Equations 10-12 for estimating the parameters k_2 and BCF are approximations, they do lend themselves to addressing many of the difficulties discussed previously. In addition to making use of the best available, if somewhat limited, data, they also circumvent the need to directly address such factors as metabolic rate, fat content, body size, and dietary intake. In addition, although Laevastu and Fukuhara (1984b) have developed a method of relating temperature to depuration rate, the approach taken here avoids the accompanying problem of estimating both species-specific and temperature-specific depuration rates. This seems appropriate given the facts that 1) there is only "about a 25% change in either the (n-octanol/water) partition coefficient or the aqueous solubility for every 1° variation in temperature" (Chiou et al., 1977); and 2) temperature is assumed constant in this analysis (i.e., 9.3°C).

In general, the methods described here, particularly the necessity of using the naphthalenes component of total hydrocarbons as a water solubility index, seem useful as a first order approximation and qualitative measure of "bioaccumulation potential" for the oil concentration data used in this analysis.

2.2 Sensitivity analysis

An important aspect of model evaluation is validation. Since the submodel FEDOIL is the major component of the BIOS model that is used in simulating uptake and depuration, it seemed appropriate to provide information on the validity of submodel results. Although a model's results normally can be validated by comparison with field data, such an analysis limits the comparison to a specific set of field conditions, which, in the case of oil spill impacts on marine organisms, are usually only available in very broad and qualitative terms. A more general and less restrictive method is a sensitivity analysis where model input parameters are perturbed one at a time and model response to the changes is compared with a base model run which contains best estimates of input parameters. If the input parameters are perturbed within their range of uncertainty, then the sensitivity analysis should give an indication of the amount of uncertainty in model output estimates. The sensitivity analysis can also indicate those particular input parameters that cause the most change in model output, and relatedly, the degree to which model structure (i.e., specific model equations) contribute to model output.

Such a sensitivity analysis was conducted for the submodel FEDOIL. The analysis only considers the non-migration case (i.e., there is no spatial resolution in the submodel FEDOIL), and uses specified external concentration levels (i.e., for both constant and time dependent concentration data) in place of the actual oil spill scenario data used in the larger BIOS model.

The sensitivity analysis involved estimating the absolute error, E, in model input parameters, k_2, BCFPEL, BCFDEM, and FODCMP from a survey of the literature (Table 7). A series of model runs were then made in which each set of parameters were increased and then decreased by the amount of the error. Base parameter values are as given previously. Following the discussion of Livingston (1980), the perturbed value, P_i' of a parameter P_i is given as:

$$P_i' = P_i\ (1 \pm E') \qquad (13)$$

Table 7.--Model input parameters, P_i, and their estimated error, E, used in FEDOIL sensitivity tests a).

	Parameter - P_i	Error - E
P_1	k_2	± 50%
P_2	BCFPEL	± 50%
P_3	BCFDEM	± 50%
P_4	FODCMP	± 20%

a) Errors estimated from general review of literature.

where E' is the fractional error ($E'=E/100$) of the relevant input parameter. The model output measured for sensitivity to parameter changes was the maximum internal hydrocarbon concentration, C_f, for each species.

The sensitivity of a dependent variable X to a small change in a parameter P_i is usually expressed as:

$$S_i = \frac{\partial X}{\partial P_i} \tag{14}$$

which can be approximated as:

$$S_i = \frac{X_i - X_B}{\Delta P_i} \tag{15}$$

where X_B is the value of the dependent variable X from a base model run and X_i is the value of the dependent variable X when the i^{th} parameter, P_i is perturbed (Livingston 1980).

Following Rivard and Doubleday (1979) and Wiens and Innis (1974), Livingston (1980) uses relative sensitivity, R_i, to denote the change in the dependent variable due to a parameter perturbation. Relative sensitivity relates a percent change in the dependent variable to a percent change in the parameter value and is calculated as:

$$R_i = \frac{\Delta X}{X \cdot E'} = \frac{X_i - X_B}{X_B \cdot E'} \tag{16}$$

or in simpler terms,

$$R_i = \frac{\text{\% change in dependent variable}}{\text{\% change in parameter value}} \tag{17}$$

As Livingston (1980) points out, the advantage of a relative sensitivity measurement is that it is less influenced by the orders of magnitude of the dependent variable and the input parameters. The relative sensitivity, R_i, is used to represent the results from the submodel FEDOIL sensitivity tests.

Rivard and Doubleday (1979) describe the following way to interpret relative sensitivity values:

1.) a negative R_i means that a decrease (increase) of the parameter P_i causes an increase (decrease) of the dependent variable X_i;

2.) a positive R_i means that an increase (decrease) of P_i causes an increase (decrease) in X_i;

3.) $R_i = 0$ means that the change in P_i does not affect X_i;

4.) $0 < |R_i| < 1$ the amount of change in P_i causes a lesser amount of change in X_i (i.e., a 10% change in P_i causes a 5% change in X_i);

5.) $|R_i| = 1$ implies that a change in P_i causes a corresponding change in X_i. (The degree of nonlinearity in the model may affect the exactness of this relationship for large parameter changes.)

6.) $|R_i| > 1$ the amount of change in P_i causes a greater amount of change in X_i.

3. RESULTS

3.1 Sensitivity results of submodel FEDOIL

A summary of the relative sensitivity values, R_i, of the submodel output (maximum internal concentration) is given in Tables 8 and 9. The data in Table 8 reflect the case of a constant external concentration of 1 ppm for 10 days, followed by 100% depuration (i.e., organisms assumed in oil free water). Table 9 reflects the case of an initial external concentration of 1 ppm that is decreased exponentially at a rate of approximately 55% per day (Figure 5). The simulation results from the submodel for both concentrations and for various parameter perturbations are illustrated graphically for a semi-demersal species (e.g., Pacific cod) in Figures 6 to 13.

The relative sensitivities, R_i, were generally less than unity for all species studied and independent of the parameter perturbed. The only exceptions were for changes in the bioconcentration factors for pelagic (BCFPEL) and demersal (BCFDEM) species when the species under study were either a 100% pelagic feeder (i.e., Species 1) or a 100% demersal feeder (i.e., Species 14, 15, and 16). In each of these cases the sensitivity of model output was approximately proportional to the changes in the relevant parameter; i.e., $|R_i| = 1$.

The relative sensitivities of changes in the bioconcentration factors (BCFPEL, BCFDEM) and the fraction of pelagic food in the diet (FODCMP) were the same, independent of either the external concentration or the positive or negative perturbation in the given parameter. The specific values varied by species, however, and seem related to their relative pelagic or demersal nature; e.g., the more pelagic (demersal) a species, the greater the relative sensitivity of submodel output given a percentage change in the pelagic (demersal) bioconcentration factor.

Table 8.--Relative sensitivity, R_i, of maximum internal concentration index to parameter perturbations in submodel FEDOIL. (Constant external concentration of 1 ppm.)

	Parameter varied							
	k_2		BCFPEL		BCFDEM		FODCMP	
Species group	-50	+50	-50	+50	-50	+50	-20	+20
Pelagic adults	.681	.352	.990	.990	.010	.010	.792	.208
Semipelagic adults	.681	.352	.682	.682	.318	.318	.545	.545
Flatfish adults	.730	.419	.469	.469	.531	.531	.375	.375
Crab adults	.494	.162	.357	.357	.643	.643	.286	.286
Sessile epifauna	.912	.768	-	-	1.00	1.00	-	-

Table 9.--Relative sensitivity, R_i, of maximum internal concentration index to parameter perturbations in submodel FEDOIL. (Decreasing external concentration starting at 1 ppm.)

	Parameter varied							
	k_2		BCFPEL		BCFDEM		FODCMP	
Species group	-50	+50	-50	+50	-50	+50	-20	+20
Pelagic adults	.835	.670	.990	.990	.010	.010	.792	.208
Semipelagic adults	.835	.670	.682	.682	.318	.318	.545	.545
Flatfish adults	.847	.717	.469	.469	.531	.531	.375	.375
Crab adults	.782	.606	.357	.357	.643	.643	.286	.286
Sessile epifauna	.929	.825	-	-	1.00	1.00	-	-

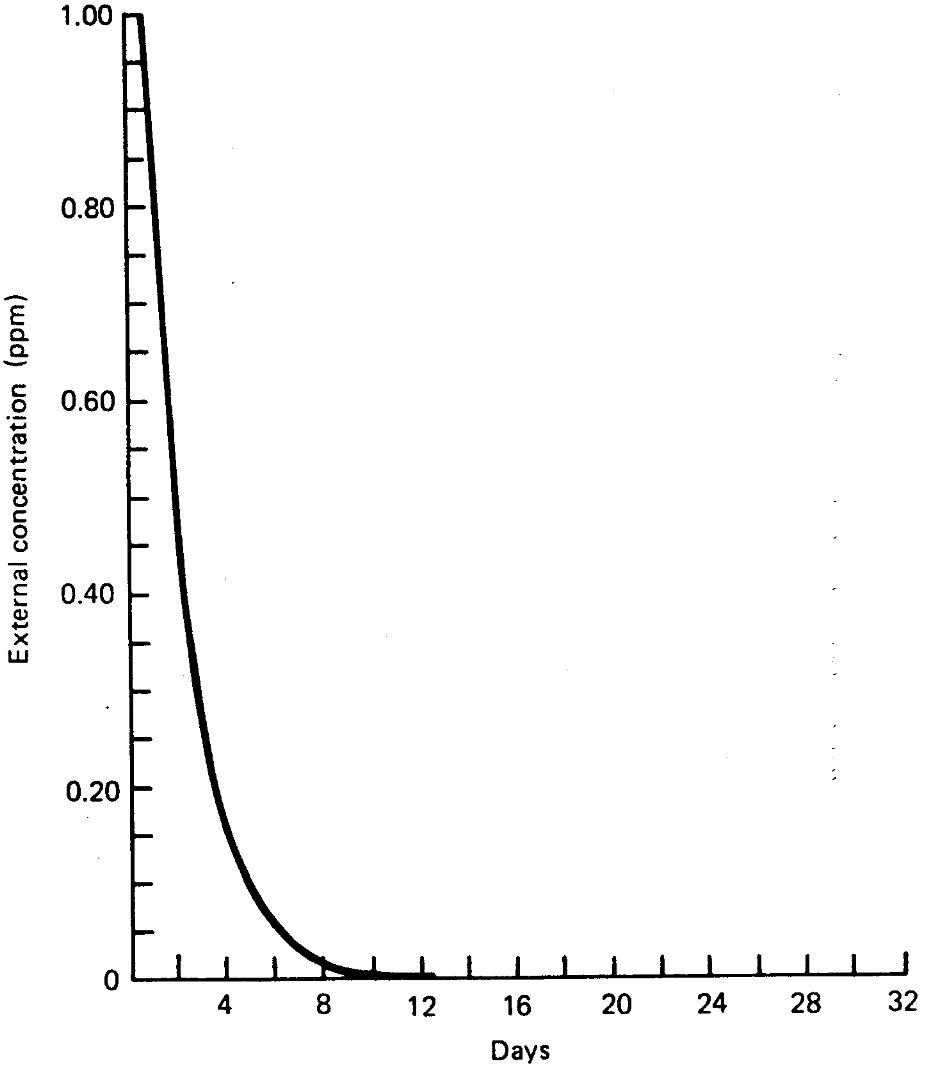

Figure 5.--External concentration (ppm) with time. Data used in sensitivity analysis of submodel FEDOIL.

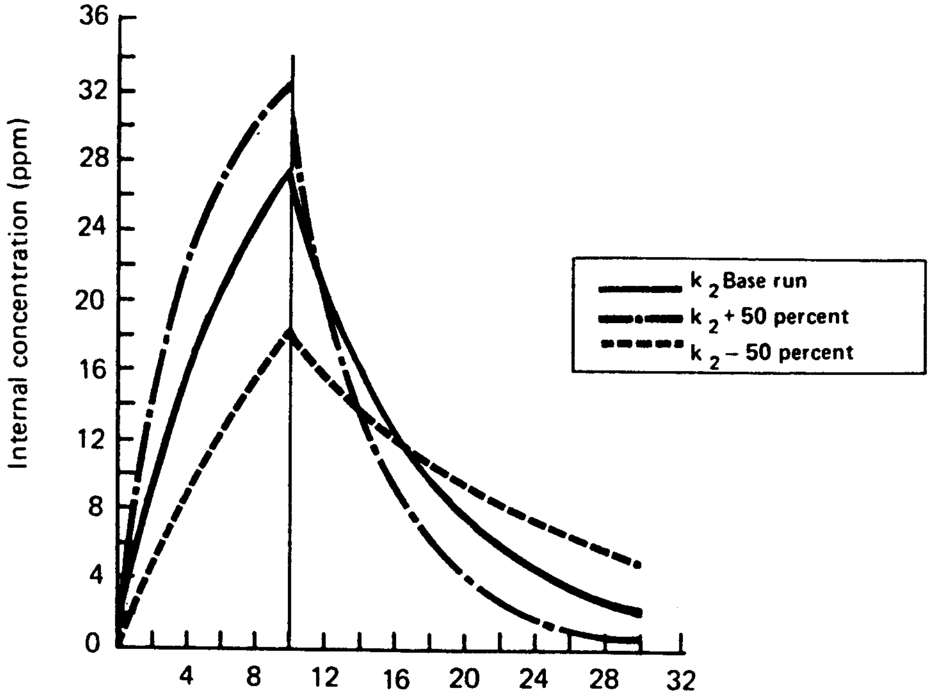

Figure 6.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, k_2. External concentration as described in text. The vertical line at day 10 indicates the boundary between uptake and depuration and depuration only.

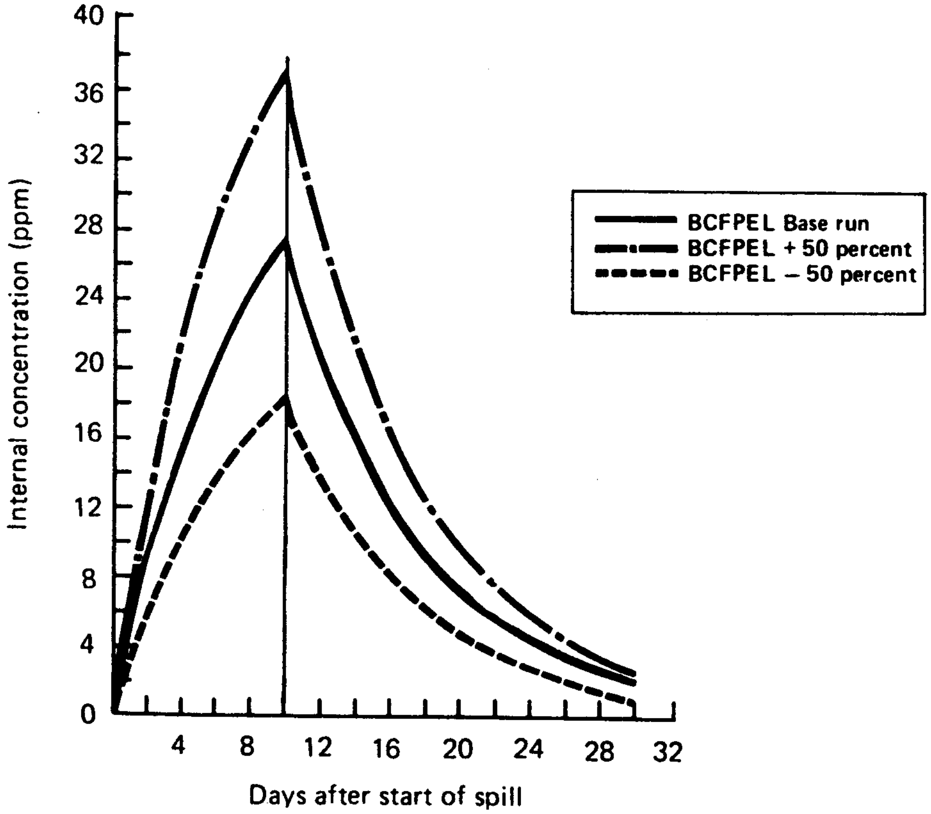

Figure 7.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, BCFPEL. External concentration as described in text. The vertical line at day 10 indicates the boundary between uptake and depuration and depuration only.

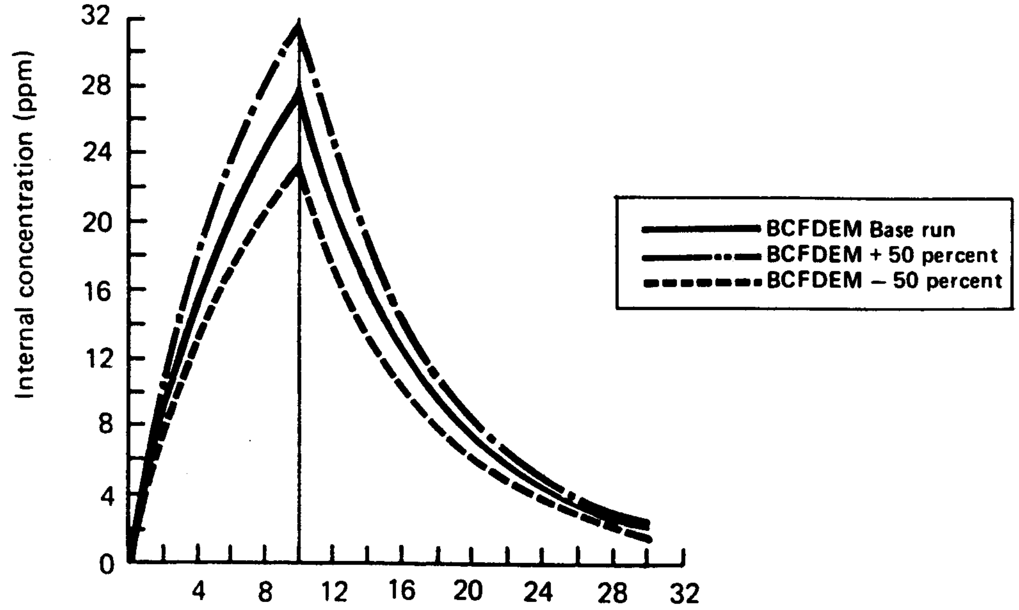

Figure 8.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, BCFDEM. External concentration as described in text. The vertical line at day 10 indicates the boundary between uptake and depuration and depuration only.

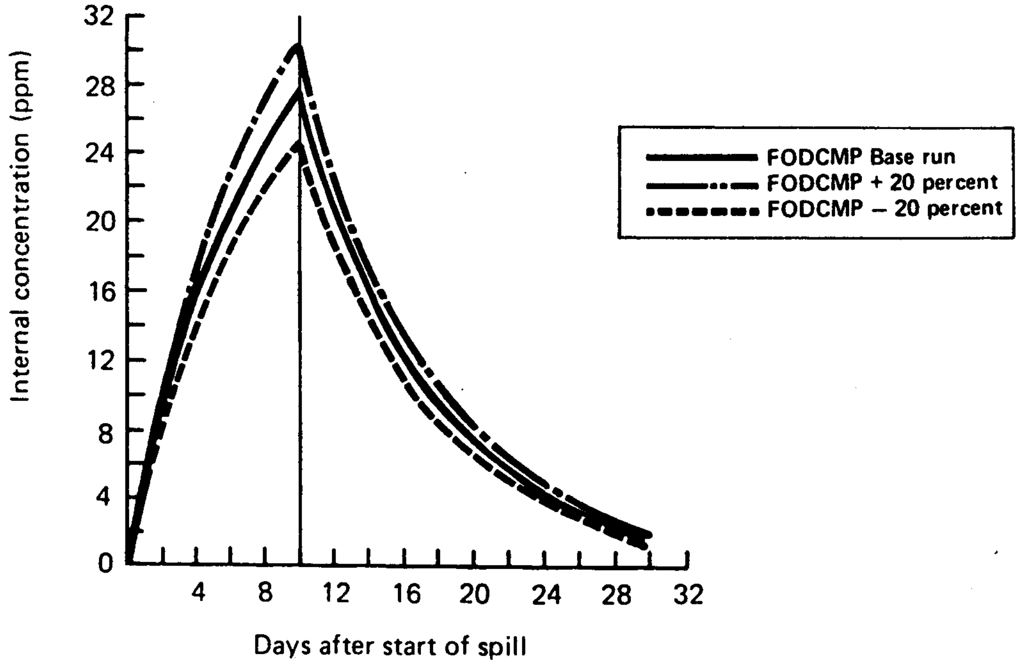

Figure 9.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, FODCMP. External concentration as described in text. The vertical line at day 10 indicates the boundary between uptake and depuration and depuration only.

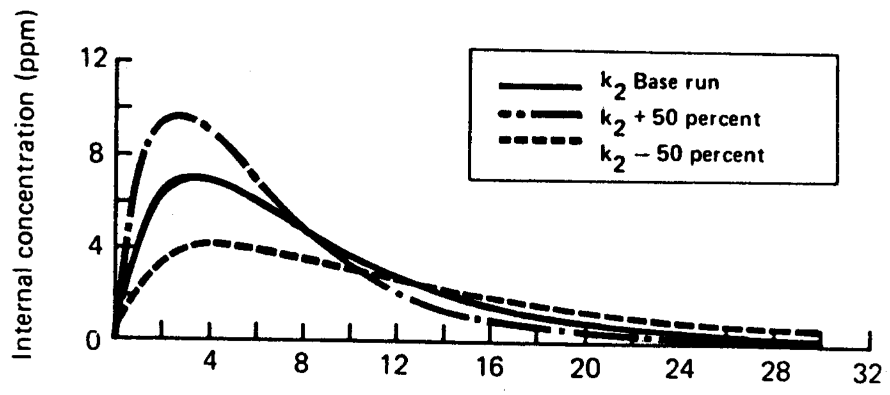

Figure 10.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, k_2. External concentration as described in text and shown in Figure 5.

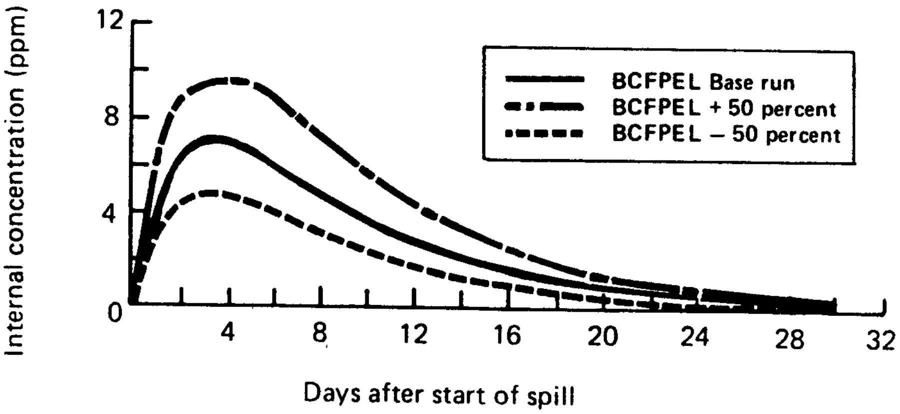

Figure 11.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, BCFPEL. External concentration as described in text and shown in Figure 5.

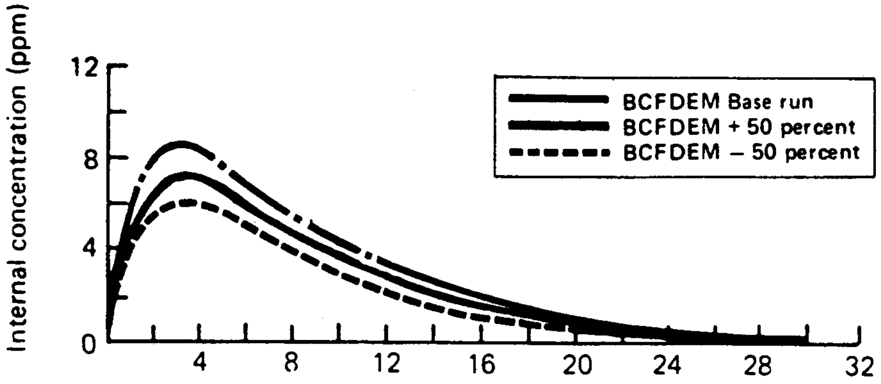

Figure 12.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, BCFDEM. External concentration as described in text and shown in Figure 5.

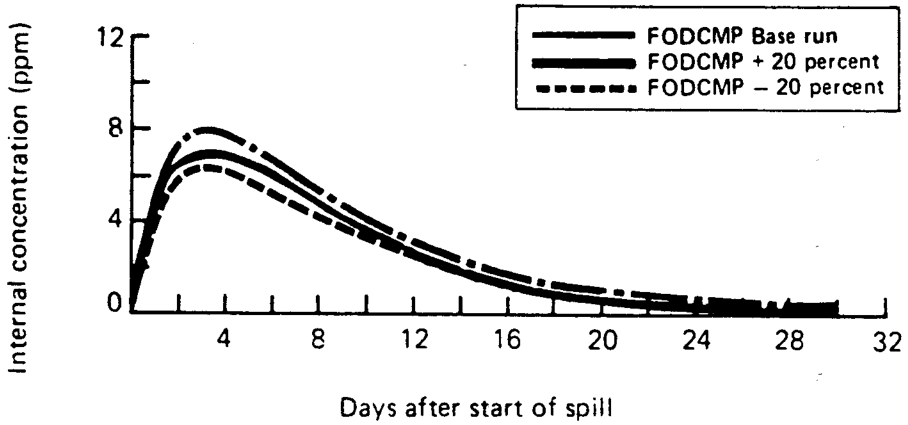

Figure 13.--Variation in internal concentration (ppm) of semi-pelagic species (Species No. 11) given perturbation of model parameter, FODCMP. External concentration as described in text and shown in Figure 5.

Changes in the depuration rate, k_2, produced a mixture of sensitivity results. Although all sensitivity values, $|R_i|$, were less than unity, they varied among species and between external concentration levels, and were dependent on the positive or negative perturbation of the parameter. The greatest effects were on 100% demersal species (e.g., sessile epifauna), independent of external concentration. The results further suggest that species specific sensitivity values following a percentage change in the depuration rate also are related to the relative pelagic or demersal nature of the species. In addition, changes in the depuration rate produced higher $|R_i|$ values for all species for the case of a time dependent (and decreasing) external concentration.

Although the submodel FEDOIL is necessarily qualitative given the limits to the available data and to our knowledge of the uptake and depuration processes, the results of the sensitivity analysis suggest that the submodel is fairly robust with respect to the relative errors associated with the various parameter values. Simulation results of the internal concentrations of five representative species for both the constant and time dependent external concentration data are shown in Figures 14 and 15, respectively.

3.2 Results from BIOS

Time dependent changes in the external concentration data used in this study are illustrated graphically in Figures 16 to 19. These data represent the percentages of the total area at the Pt. Heiden location covered by various levels of the water-soluble fraction (WSF) and oil on the bottom fractions (TARS) of external contamination. Since the external concentration data (WAF) provided by the Rand Corporation were only available for a maximum of 15 days, these data were decreased exponentially at 55% per day from day 10 (accident) and day 15 (blowout), respectively, in order to provide external concentration data to day 30.

In the case of the "blowout scenario", neither the WSF nor the TARS concentrations (Figures 16 and 17) exceeded 1 ppm during the time period of the simulation. Approximately 24 hours following the blowout the WSF concentration (Figure 16)

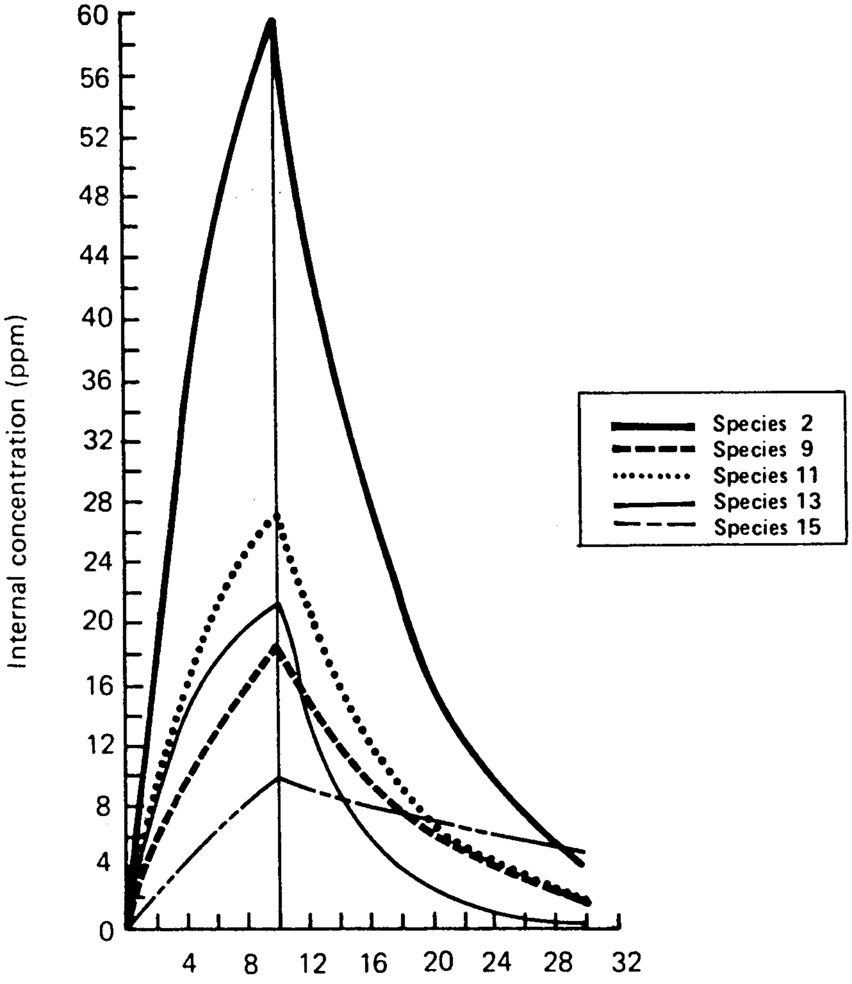

Figure 14.--Variation in internal concentration (ppm) of selected marine species (see text). External concentration as described in text. The vertical line at day 10 indicates the boundary between uptake and depuration and depuration only.

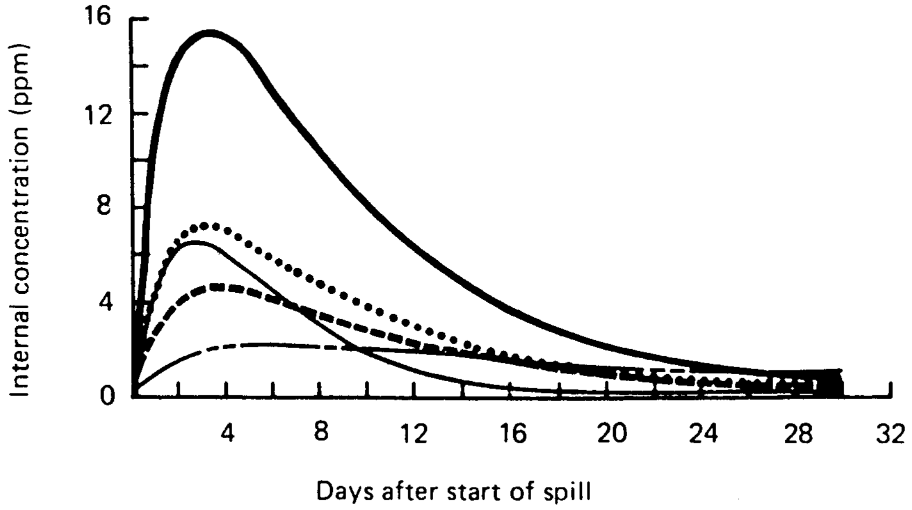

Figure 15.--Variation in internal concentration (ppm) of selected marine species (see text). External concentration as described in text and shown in Figure 5.

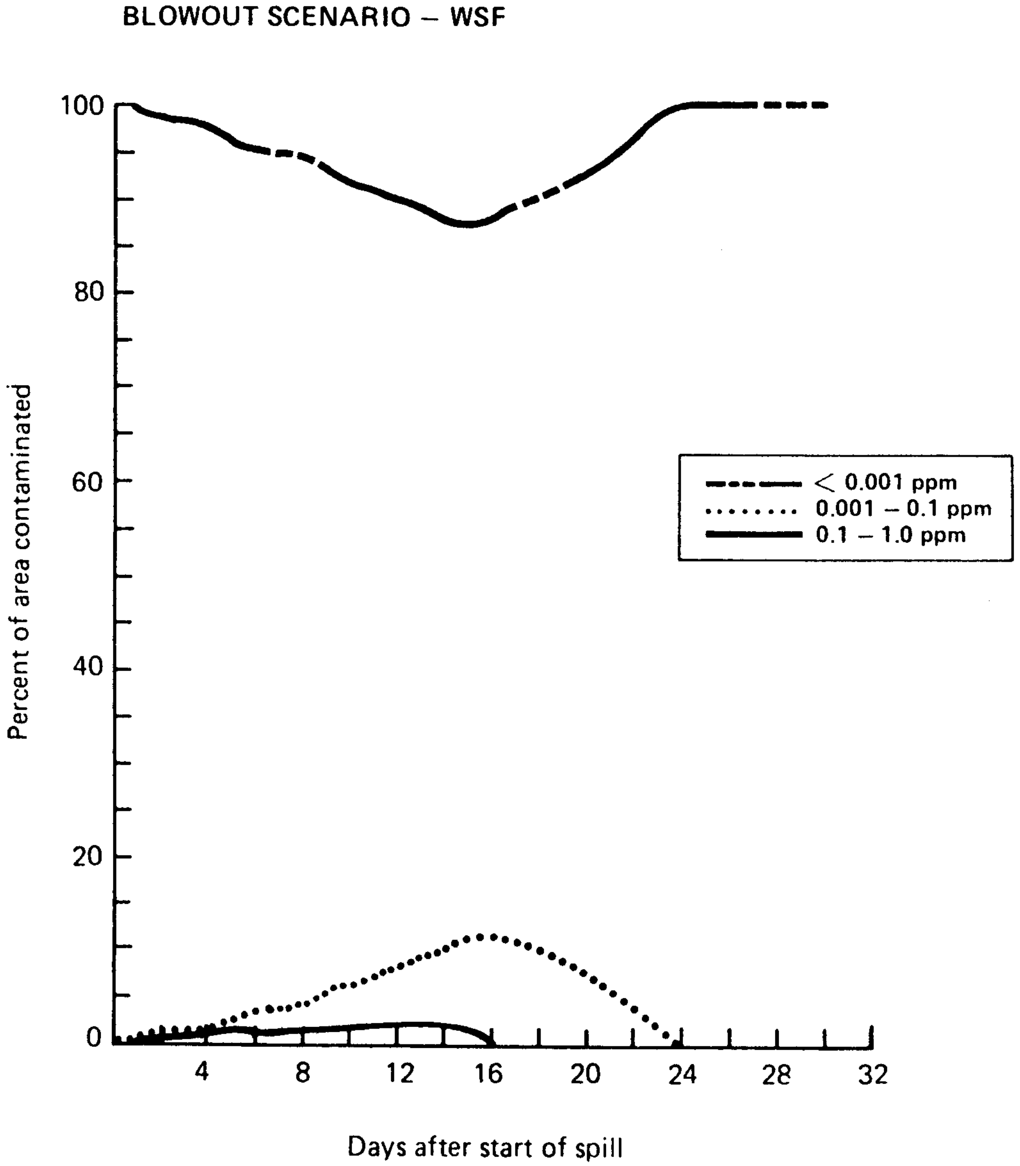

Figure 16.--External concentration of WSF from "blowout scenario" at Pt. Heiden as percent of total area contaminated. Data as used in BIOS model.

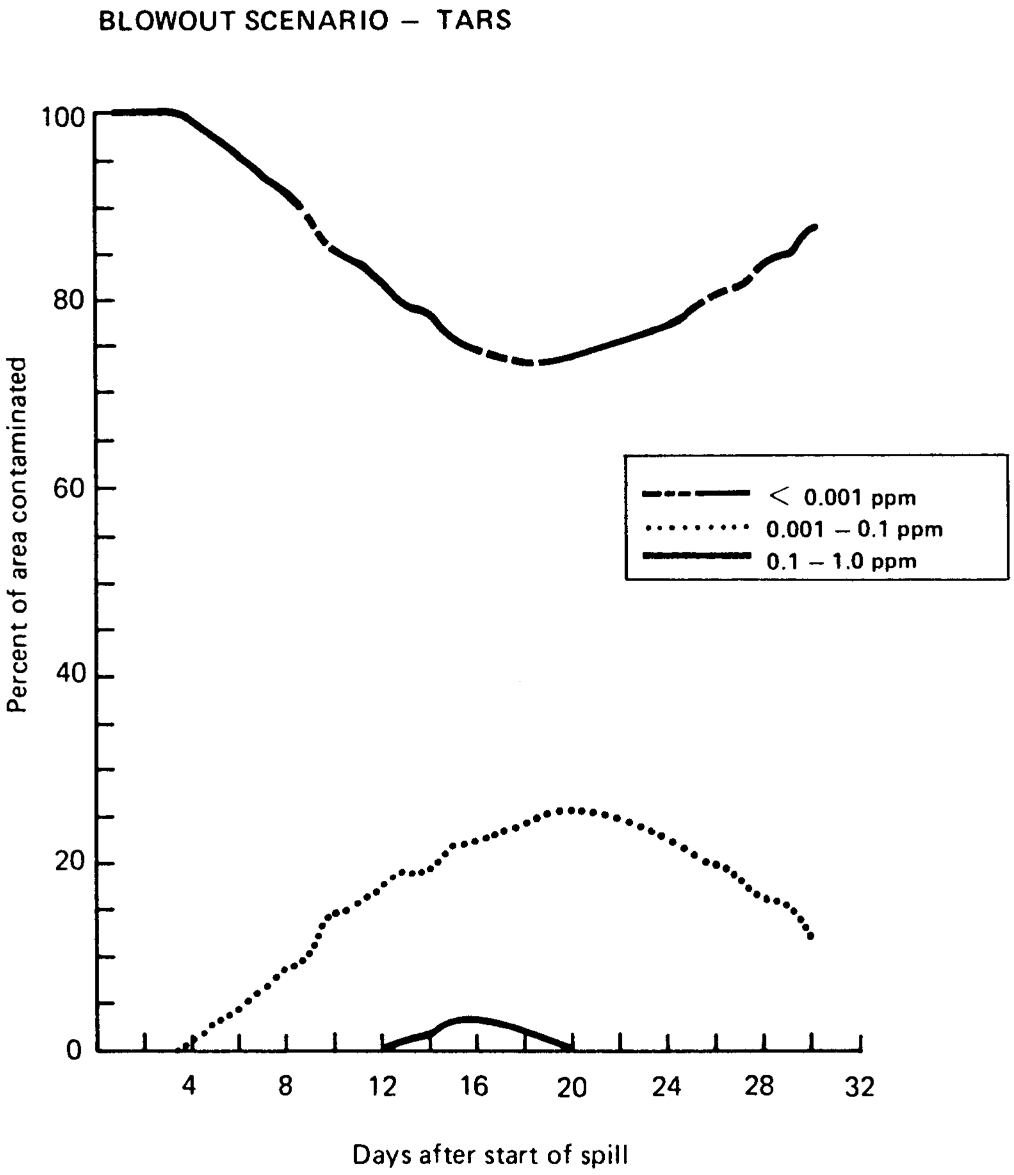

Figure 17.--External concentration of TARS from "blowout scenario" at Pt. Heiden as percent of total area contaminated. Data as used in BIOS model.

is less than .001 ppm over 100% of the spill area. Thirty (30) days after the blowout the TARS concentration (Figure 17) is between .001 and .1 ppm in less than 14% of the spill area.

In the case of the "accident scenario", both WSF and TARS concentrations (Figures 18 and 19) exceeded 5 ppm, although for only 4 days and covering less than 2% of the spill area for the WSF concentration, and for only 12 days and covering less than 5% of the spill area for the TARS concentration. After 23 days the WSF concentration from the accident (Figure 18) is less than .001 ppm over 100% of the spill area, and after 30 days less than 28% of the area has a TARS concentration between .01 and 1 ppm (Figure 19).

These data on the percentage of the total area that is contaminated can be compared to the data on soluble aromatic derivatives (SAD) given in Table 10. to roughly assess the mortality caused by the oil spill scenarios analyzed in this study. The results from the blowout scenario suggest that external concentration data are too low to cause sufficient direct mortalities in either larval or adult life-history stages. The concentrations would be sufficient, however, to disrupt both feeding and reproduction behavior (i.e., effects have been noticed at concentration levels as low as 10-100 ppb; Moore and Dwyer 1974). In addition, since SAD concentrations lower than 0.1-1 ppm may cause sub-lethal toxic effects (Moore and Dwyer 1974), there is a potential for limited but uncertain sub-lethal toxic effects to occur in about 2 to 5% of the available biomasses in the first 20 days following the initial blowout. The effects would, of course, be species specific with demersal species being affected to a greater degree than pelagic species.

The results from the accident scenario suggest the potential for more serious impacts on the marine environment, benthic organisms in particular. As Figure 19

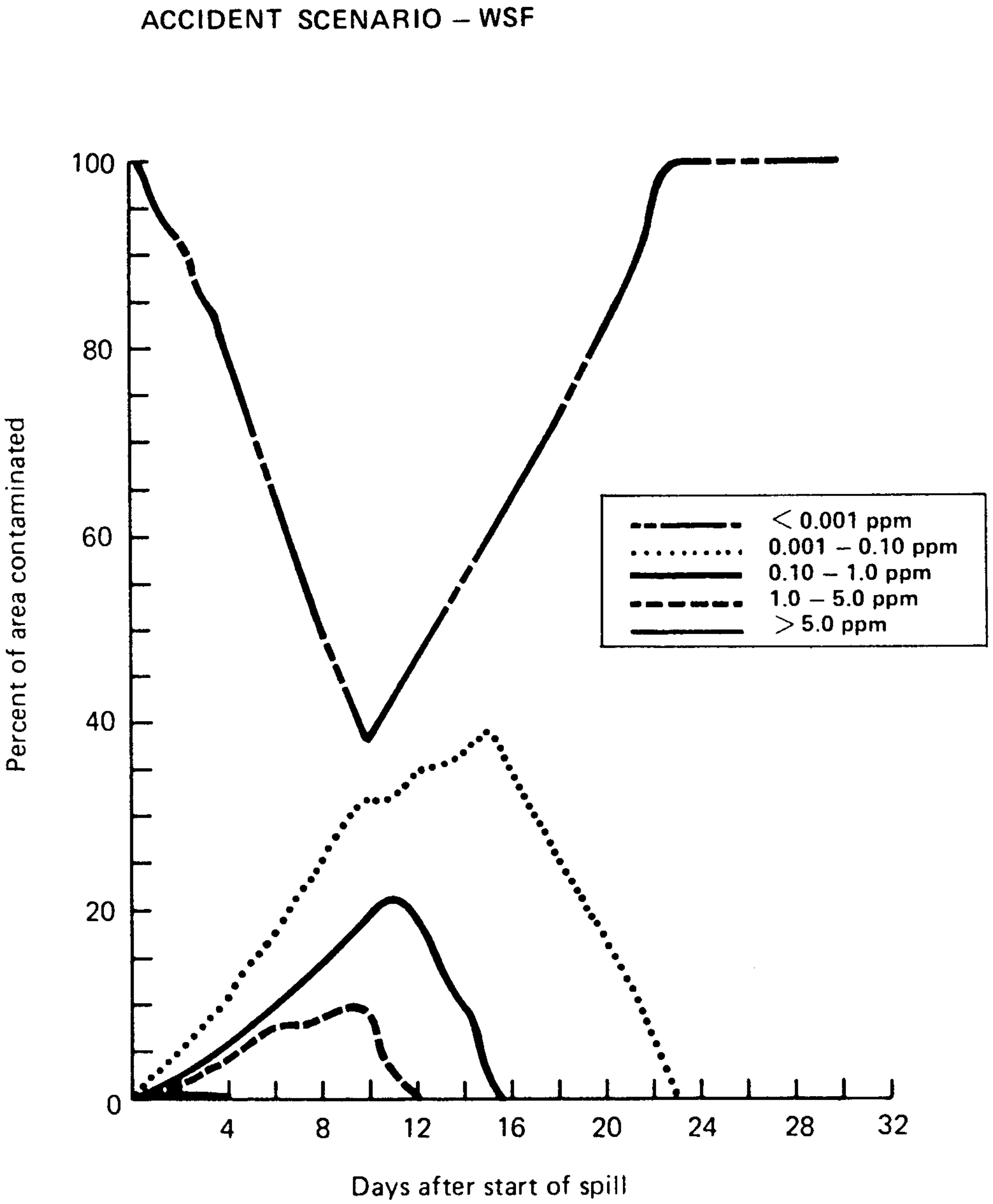

Figure 18.--External concentration of WSF from "accident scenario" at Pt. Heiden as percent of total area contaminated. Data as used in BIOS model.

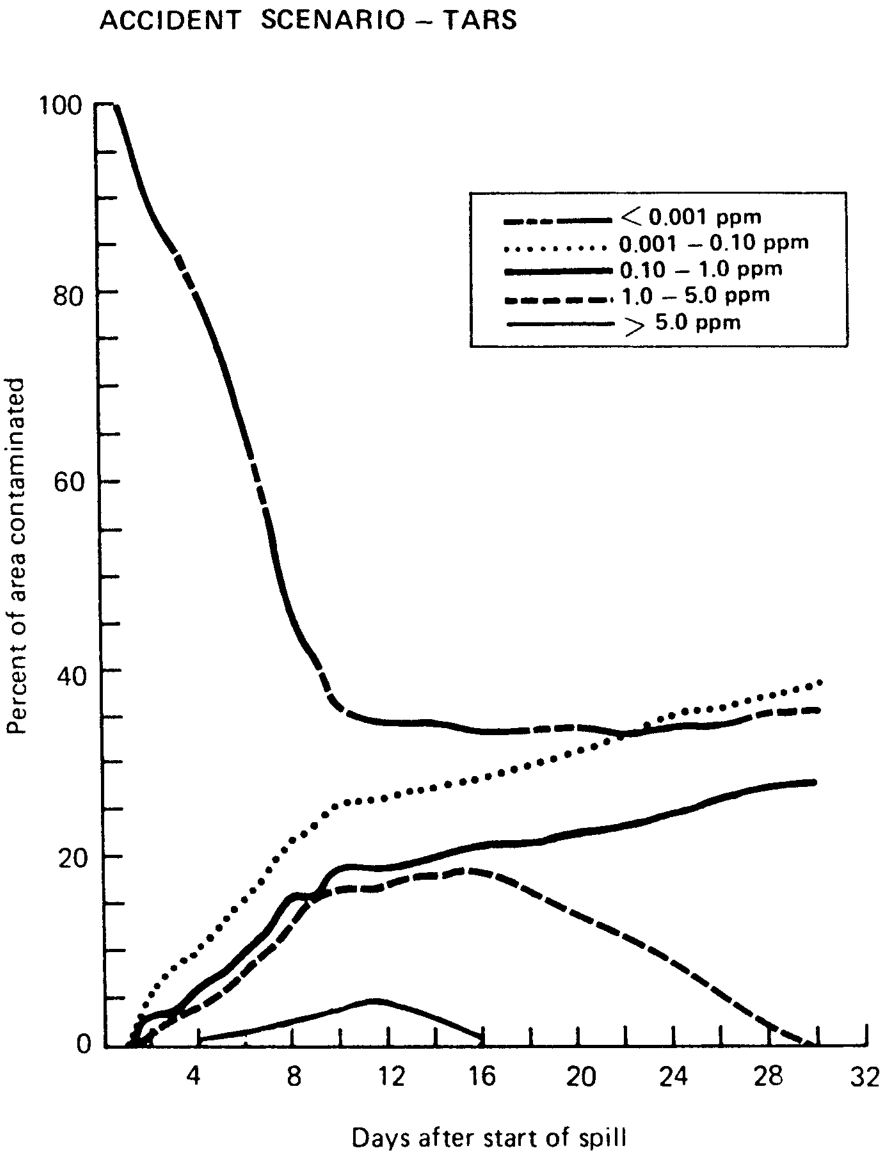

Figure 19.--External concentration of TARS from "accident scenario" at Pt. Heiden as percent of total area contaminated. Data as used in BIOS model.

Table 10.--Summary of toxicity data.[a)]

Class of Organism	Estimated concentrations (ppm) of soluble aromatics causing toxicity
Finfish	5 - 50
Larvae (all species)	0.1 - 1.0
Pelagic crustacens	1 - 10
Bivalves	5 - 50
Benthic crustaceans (e.g., crabs)	1 - 10
Other benthic invertebrates (e.g., worms)	1 - 10

a) Adapted from Moore and Dwyer 1974

illustrates, the external concentrations of TARS are between 1 to 5 ppm for almost 30 days and cover a maximum of approximately 19% of the total area. In addition, the potential for sub-lethal toxic effects and the disruption of feeding and reproduction is also considerably higher in the "accident scenario" than from the "blowout scenario". Since the subject of oil-induced mortalities and their resultant effects on year class strength will be discussed in detail in the final report (Laevastu and Fukuhara, in preparation), the topic will not be considered further in this report. It should be pointed out, however, that the area and species biomasses affected by the oil contamination referred to in this study are only a small fraction of the total area and biomasses of the eastern Bering Sea (see Figure 1 and Table 11).

Figure 20 shows the percentage of the total biomass of 5 representative species that is tainted (internal concentration >5 ppm) from both the blowout and accident scenarios. For the "blowout scenario" only 2 species showed internal concentrations greater than 5 ppm, and then for only a maximum of 2% of the total biomass (e.g., a pelagic species, herring). For the "accident scenario" all species showed tainting, although the maximum percentage of the total biomass tainted did not exceed 30%.

The maximum levels of tainting were reached between 11 and 23 days after the start of the spill (accident scenario). The pelagic species (i.e., herring) was contaminated most quickly (maximum in 11 days) and depurated rapidly from a maximum of 28% of the biomass tainted to less than 11% in 19 days. The slowest uptake was in the benthic invertebrates (i.e., sessile epifauna), with a maximum (less than 28% of the total biomass tainted) reached in approximately 23 days. Depuration (for the benthic invertebrates) is slow and from the data in Figure 20 would appear to be long-lasting.

Table 11.--Species biomass in study areas as percent of total biomass in eastern Bering Sea.[1]

	Species biomass as percent of total biomass (kg) in Eastern Bering Sea		
Species	Pt. Moller	Pt. Heiden	Cape Newenham
Herring juveniles	.505	.187	.556
Herring adults	.505	.187	.556
Pollock juvenile	.471	.295	.414
Pollock adults	.471	.295	.414
Pacific cod juveniles	.577	.379	.418
Halibut juveniles	1.220	.551	.401
Yellowfin sole juveniles	.902	.602	.888
Other flatfish juveniles	1.141	.838	.939
Yellowfin sole adults	.900	.601	.888
Other flatfish adults	1.141	.838	.939
Pacific cod adults	.577	.309	.456
King and Bairdi crab juveniles	.806	.269	.524
King and Bairdi crab adults	.804	.268	.524
Mobile epifauna	.416	.348	.424
Sessile epifauna	.416	.348	.424
Infauna	.604	.433	.607

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[1] Total biomass in eastern Bering Sea taken from DYNUMES model (Laevastu and Larkins 1981).

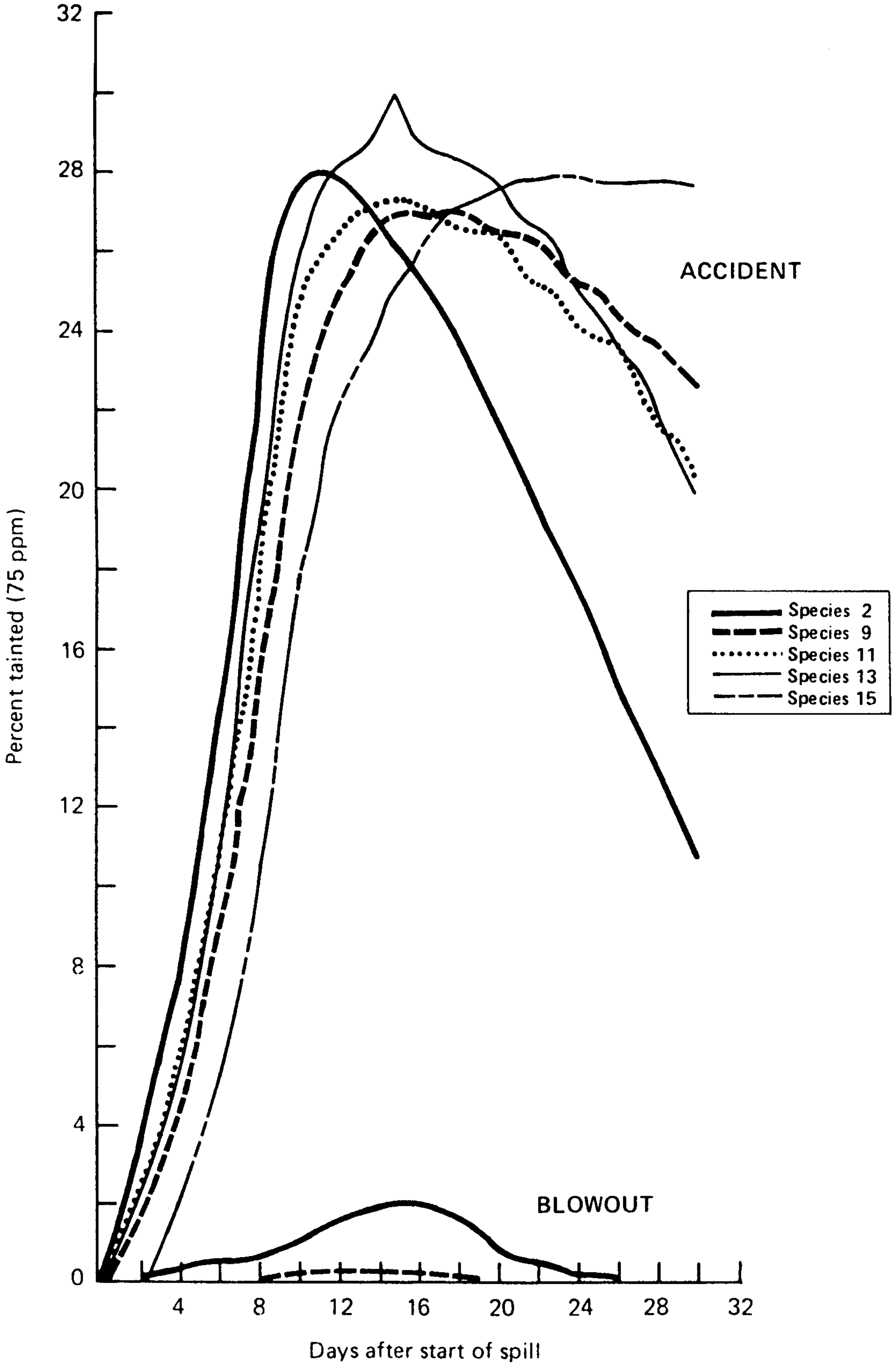

Figure 20.--Percent of biomass tainted (internal concentration >5ppm) for selected marine species at Pt. Heiden. Results from BIOS model for both "blowout" and "accident" scenarios.

As demonstrated previously in the results from the sensitivity analysis (Figures 6 to 13), the percentage of a species' biomass that is tainted appears to be a function of the relative pelagic or demersal nature of the species. In addition, all species, with the exception of benthic invertebrates (i.e., sessile epifauna), depurate rather rapidly within 24 days following the spill, and after 30 days have a maximum percentage biomass tainted of less than 23%.

4. DISCUSSION AND CONCLUSIONS

The submodel FEDOIL proved most sensitive to changes in the depuration rate (k_2), with the absolute value of the relative sensitivity values, $|R_i|$ dependent on the pelagic or demersal nature of the species and the level of external concentration. Since the depuration rate directly determines the amount and retention time of hydrocarbons accumulated (and also indirectly through Equation 5, i.e., an increase (decrease) in the depuration rate k_2 causes a concomitant increase (decrease) in the uptake rate k_1), the relative magnitude of each species group's maximum internal concentration level (i.e., tainting) depends primarily on the submodel's definition of the uptake and depuration processes.

Although all sensitivity values, R_i, were related to the relative pelagic or demersal nature of the species, only changes in the depuration rate showed any direct correlation between relative sensitivity values and the level of external concentration. For example, the relative sensitivity of submodel output to changes in the bioconcentration factors (BCFPEL, BCFDEM) is constant over external concentration values. This suggests that although the external concentration data do determine the type of uptake and depuration curve generated by the model (see Figures 6 and 10), for an individual curve, the bioconcentration values only affect the absolute values of the internal concentration. The actual shape of a specific curve (see Figure 6) is determined almost solely by the depuration rate value, and the larger the rate constant, the sooner any percentage of the asymptotic value (under constant external concentration; see Figure 6) or maximum value (under time-dependent external concentration; see Figure 10) of the submodel is reached. These results not only indicate general model sensitivity but they also highlight how the model structure affects model behavior. As discussed in Livingston (1980), to evaluate a simulation model as a whole, its structure and behavior should be appraised on the basis of generality, resolution, realism, and precision (Orth 1979).

Generality refers to the applicability of the model to other areas and species communities. The submodel FEDOIL was developed from existing and well accepted approaches that have been used to simulate the uptake and depuration processes. The submodel should be fairly transferable to other marine areas but for each area it would require a careful analysis of the external hydrocarbon concentrations in order to define the hydrocarbon specific bioconcentration factors to be used.

Resolution is defined by the number of characteristics of the real system that are included in the model. The submodel FEDOIL has a low resolution. It does not address multi-species predator-prey behavior, size specific effects of uptake and depuration, temperature effects, or hydrocarbon specific bioconcentration rates. Most of these processes are poorly understood and, in almost all cases, are difficult if not impossible to simulate due to lack of available data. As Livingston (1980) points out, higher resolution does not necessarily produce more accurate results. Higher resolution is clearly needed in this study but full utilization of this model, or any other, as an effective and predictive management tool "will only become possible when laboratory (and field) techniques to measure the critical parameters are formulated" (Hamelink 1977).

Realism is the closeness of the model's equations to the actual biological processes. As discussed previously, the submodel FEDOIL is almost by definition a simplification; no attempt has been made to accurately describe the specific biological processes of uptake and depuration. The model is thus useful as a conservative and qualitative measure of bioconcentration potential but must await the results of further laboratory studies before it can attempt to simulate the actual biological processes involved; in particular, the disposition of accumulated hydrocarbons.

Precision is the degree of correspondence of model outputs to observed values. There are few specific data values with which to compare submodel results. In a qualitative sense, however, the low levels of contamination and tainting and the relative differences among species in internal concentration levels and retention times of accumulated hydrocarbons are in general agreement with the findings from actual oil spill events (see Laevastu and Fukuhara 1984a, 1984b, for reviews).

Thus the submodel FEDOIL is a general qualitative estimator of internal hydrocarbon concentration potential that has some limited value in assessing the impact of oil spill scenarios on marine species in the eastern Bering Sea. Although its sensitivity to changes in input parameters suggests the model is somewhat robust with respect to the error associated with those parameters, the low resolution of the submodel severely limits its present use as a predictive management tool.

When viewed in conjunction with the full BIOS model, the results from this study indicate that distinct but very limited tainting and mortality effects will result from the accident scenario in the Port Heiden area. Almost no direct effects will occur under the external concentration conditions of the blowout scenario. Although sub-lethal toxic effects could result from either scenario, they are almost impossible to assess quantitatively. Considered in light of available total biomass estimates for the associated stocks in the eastern Bering Sea, only a small percentage (i.e., less than 2%) of the total species biomasses would be affected directly by the oil spill scenarios analyzed in this study. (This is exclusive of mortality and resultant year class effects, which are considered in detail in the final report.) Finally, the potential impacts from the accident scenario appear to be most pronounced and will be longest lasting in demersal species.

In closing, the limited and qualitative results of this study support the findings of earlier workers who concluded that "relatively little generic information has been generated that can be applied to understanding processes or the dynamics governing petroleum-related perturbations in marine organisms and ecosystems" (Malins and Hodgins 1981). A more detailed quantitative analysis must await the results of future laboratory and field experiments.

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PHYSICAL FACTORS AFFECTING THE FATE OF A PETROLEUM SPILL IN THE SOUTHEASTERN BERING SEA

by

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Final Report
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Introduction

In examining the potential environmental impact of oil development on marine ecosystems of the southeastern Bering Sea, two important factors which determine the distribution, weathering, persistence and toxicity of inadvertantly induced oil are (1) the physical/chemical properties of oil itself, and (2) the physical characteristics of this particular regional environment. Because of the great number of possible combinations of elements within these factors, no two oil contamination events are the same in behavior and environmental impact. Properties differ from various crude oils and their distillated products. Also, crude oil properties vary from region to region, well to well, and even according to the depth of a well. The physical factors of the local environment include the regional climatology, meteorology, oceanography, hydrology, and geology. These are discussed here to illustrate the background information available about these primary agents which play a major roll in determining the ultimate fate of any oil induced into this ecosystem.

This summary information is compiled for three primary areas--offshore of Port Moller, Port Heiden, and Cape Newenham. Its inherent nature includes diurnal, monthly, seasonal, annual, and inter-annual variations. Therefore, annual, seasonal, and monthly data averages are presented where possible as well as extreme conditions. Much of the data, however, does not exist or is unavailable for all seasons. In some cases most of the data were taken only during the summer months, permitting no description of seasonal change. A purposeful attempt was made to present the information in self-explanatory form in the following tables, illustrations, and graphs. This material will hopefully serve as background for evaluation of the effects of oil development and as background information for possible future monitoring of changes.

Climate and Weather

Consideration of the seasonal climatology of the eastern Bering Sea gives us expected values and extremes for the major physical phenomenon and external forces which affect spilled petroleum from above--sunlight, air temperature, precipitation, and wind.

Sunlight measured at King Salmon and Bethel in terms of insolation (Incident Solar Radiation) with both direct and diffuse radiation, varies considerably from the winter minimum (<1 KJ/M^2) in December to the summer maximum (17 KJ/M^2) in June (Figure 1). Although the annual change is large due to the high northerly latitude, the monthly change is smooth. The slight asymetry in the curve may be explained by examining the monthly mean cloud cover (Figure 2). Somewhat contrary to what might be expected, the stormy winter months average the clearest sky with minimum cloud cover from January to March. Maximum cloud cover occurs in summer from July to August. Thus, the solar insolation which reaches the surface is slightly lower during the more cloudy cooling period from June to December than during the corresponding month of the less cloudy warming period from December to June. As expected, a large monthly change in air temperatures follows the large monthly changes in insolation.

About a half century of weather records (air temperature, precipitation, and wind) from coastal reporting stations were summarized by Brower et. al. (1977). Although the four stations in the Bristol Bay area show very similar data patterns, slight differences are apparent in the overall annual averages of weather data (Table 1). Cape Newenham and King Salmon farther north, have more northerly winds from the land, and are about one degree colder than Port Moller and Port Heiden on the south side of Bristol Bay. Southerly winds off the ocean at speeds of about 5.1 m/sec dominate at the southern

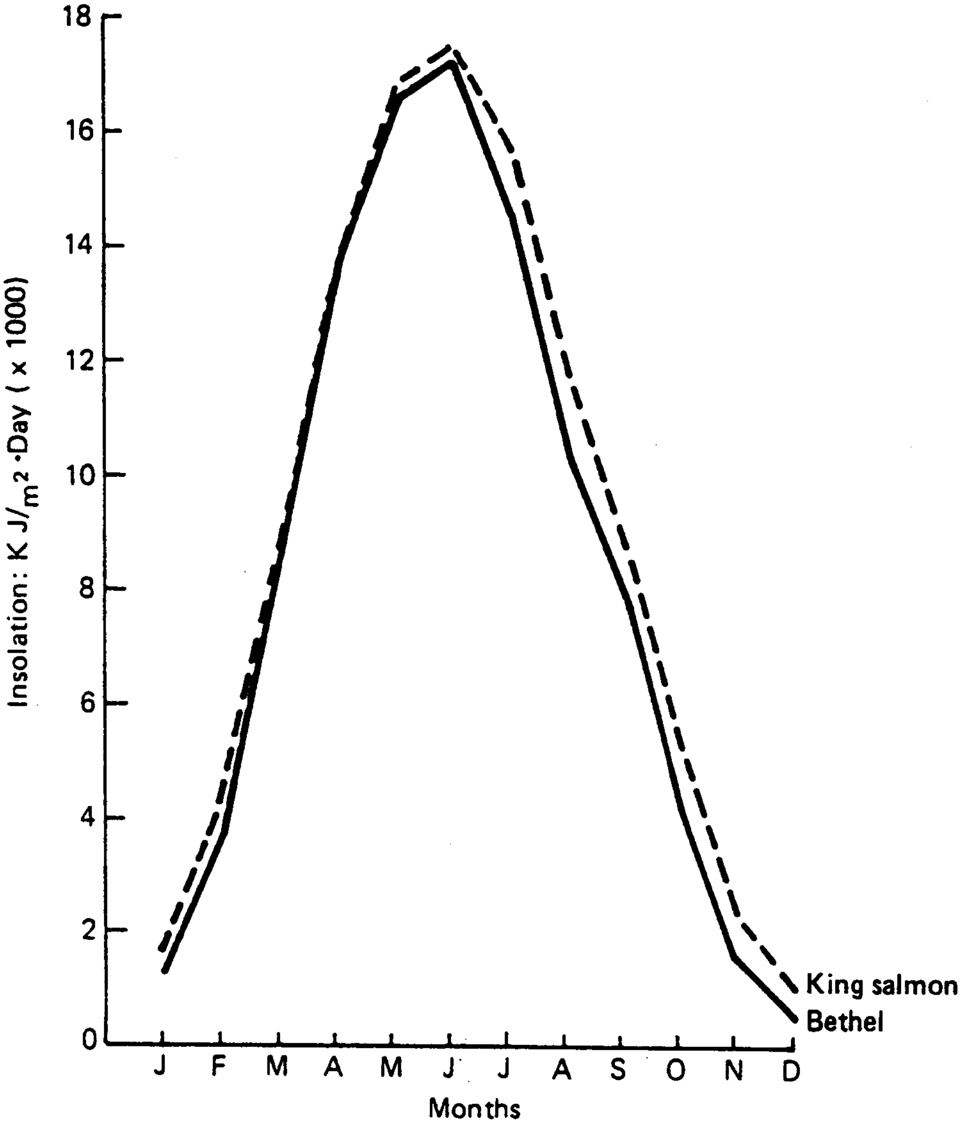


Figure 1.--Monthly mean insolation (KJ/M^2·Day) for each month (Durrenberger, R.W., 1980).

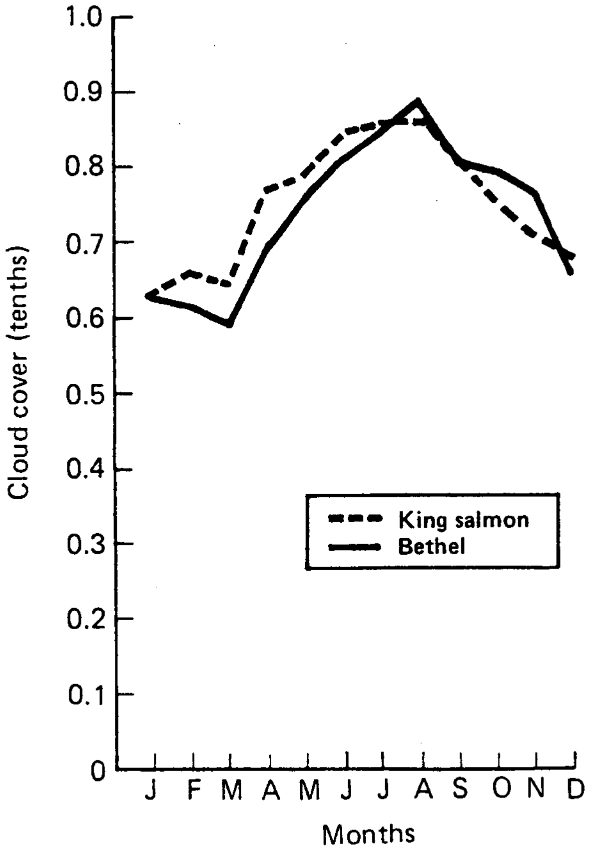


Figure 2.--Monthly mean cloud cover for King Salmon and Bethel, Alaska (Ruffner,1978).

Table 1.--Annual average weather data.

	Cape Newenham	King Salmon	Port Moller	Port Heiden
Maximum temperature (°C)	2.8	5.0	5.5	5.5
Average temperature (°C)	0.6	0.7	1.3	1.9
Minimum temperature (°C)	-1.7	-3.7	-2.9	-1.7
Total precipitation (cm)	94.6	50.2	110.5	37.2
Snowfall (cm)	206.2	113.3	233.9	158.2
Wind direction	N	N	S	ESE
Wind speed (m/sec)	5.0	4.7	4.5	6.6

stations suggesting a net annual air convergence over Bristol Bay. The seasonal data must be examined to see how this convergence relates to the precipitation maximum which occurs at the two westernmost stations.

The seasonal changes of weather from month to month can be clearly seen in the mean data from King Salmon (Table 2). Monthly mean data from Bethel which is just north of Bristol Bay is included for comparison to show the general similarity in station data along the southwestern Alaskan coast. The mean air temperatures are cyclic during the year from about -11° to 12°C, alternating between below freezing then above freezing about six months at a time. This appears related to two distinct wind seasons, cold, relatively dry northerly (from the north) winds off the land from October to April, and warm, moist southerly winds off the ocean from May to September. Wind speeds average about 4.6 m/sec most of the year, then are slightly higher to 5.1 m/sec from February to May. Precipitation is low and consistent (about 25 mm) from December to May then increases about three fold during the peak in August.

More details about the consistency of the wind between stations may be derived from monthly vector averages (Figure 3). Vector mean speeds range from 0.4 to 2.4 m/sec, noticeably lower than the scalar averages of 4.6 to 5.1 m/sec discussed above. Mean wind vectors are lined up in the same direction at all stations only in the peak of winter and summer, from the northeast in December and from the southwest in July-August. In other months wind directions may vary between stations with the northern stations maintaining their northerly or easterly components well into the transition periods, and similarly the southern stations maintaining their southerly or westerly components. Therefore, the apparent convergence suggested in the annual averages is only developed during the transition periods between summer and winter.

King Salmon Lat: 58°41'N Long: 156°39'W

	Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.	Sep.	Oct.	Nov.	Dec.	Ave.
Average Temp.[°C]													
Monthly	-10.3	-8.6	-6.4	-0.3	5.9	10.4	12.5	12.1	8.5	0.9	-5.5	-11.3	0.7
Daily Max.	-6.1	-4.1	-1.9	3.9	10.7	15.4	16.9	16.1	12.7	4.9	-1.7	-6.9	5.0
Daily Min.	-14.6	-13.0	-11.1	-4.6	1.0	5.4	8.0	8.2	4.3	-3.2	-9.4	-15.7	-3.7
Record High	8.3	7.8	12.2	15.6	23.9	26.7	29.4	28.9	23.3	16.7	8.9	8.9	
Record Low	-36.7	-40.6	-41.1	-20.0	-10.6	-1.7	0.6	0.0	-7.8	-18.3	-25.6	-34.4	
Wind													
Direction	N	N	N	NNW	S	SW	S	S	S	N	N	N	N
Speed [m/sec]	4.7	5.1	5.3	5.1	5.1	4.9	4.6	4.6	4.8	4.7	4.9	4.7	4.9
Precipitation [mm]													
Normal	23.9	25.1	29.5	22.9	28.7	36.6	55.4	87.9	78.0	50.8	36.3	26.7	501.7
Monthly Max.	76.7	76.2	61.2	75.9	61.0	96.0	100.0	163.6	185.4	161.3	75.2	86.9	
Monthly Min.	4.1	2.8	1.0	t	2.8	0.0	8.1	49.5	25.4	5.1	t	3.0	

Bethel Lat: 60°47'N Long: 161°48'W

Average Temp.[°C]													
Monthly	-14.9	-13.2	-11.4	-4.2	4.5	10.9	12.6	11.3	7.2	-1.0	-8.2	-15.3	-1.8
Daily Max.	-10.9	-9.0	-6.5	0.4	9.2	15.7	16.6	14.7	11.0	2.3	-4.7	-11.5	2.3
Daily Min.	-19.0	-17.5	-16.4	-8.7	-0.2	6.1	8.6	7.8	3.4	-4.3	-11.8	-19.2	-5.9
Record High	8.9	7.8	7.8	11.7	26.1	30.0	28.3	27.8	21.1	13.3	7.2	7.2	
Record Low	-43.3	-37.8	-39.4	-30.0	-15.6	-2.2	-0.6	0.6	-7.8	-20.6	-30.0	-37.2	
Wind													
Direction	NE	NNE	NNE	NW	S	NW	SSW	SSW	SSW	NNE	NE	NNE	NNE
Speed [m/sec]	6.2	6.6	6.0	6.0	5.3	5.1	5.1	5.1	5.0	5.6	6.0	6.2	5.7
Precipitation [mm]													
Normal	13.7	18.8	20.1	10.9	21.1	31.5	50.3	100.8	61.5	33.5	24.4	15.7	402.3
Monthly Max.	59.7	53.8	39.6	39.1	42.2	84.3	97.5	147.6	90.9	63.8	63.3	65.5	
Monthly Min.	1.5	0.8	0.5	1.5	2.5	6.4	16.8	43.4	10.7	2.8	1.0	2.8	

Table 2.--Monthly weather conditions for King Salmon and Bethel, Alaska (Brower et. al. 1977 and Ruffner et al. 1978)

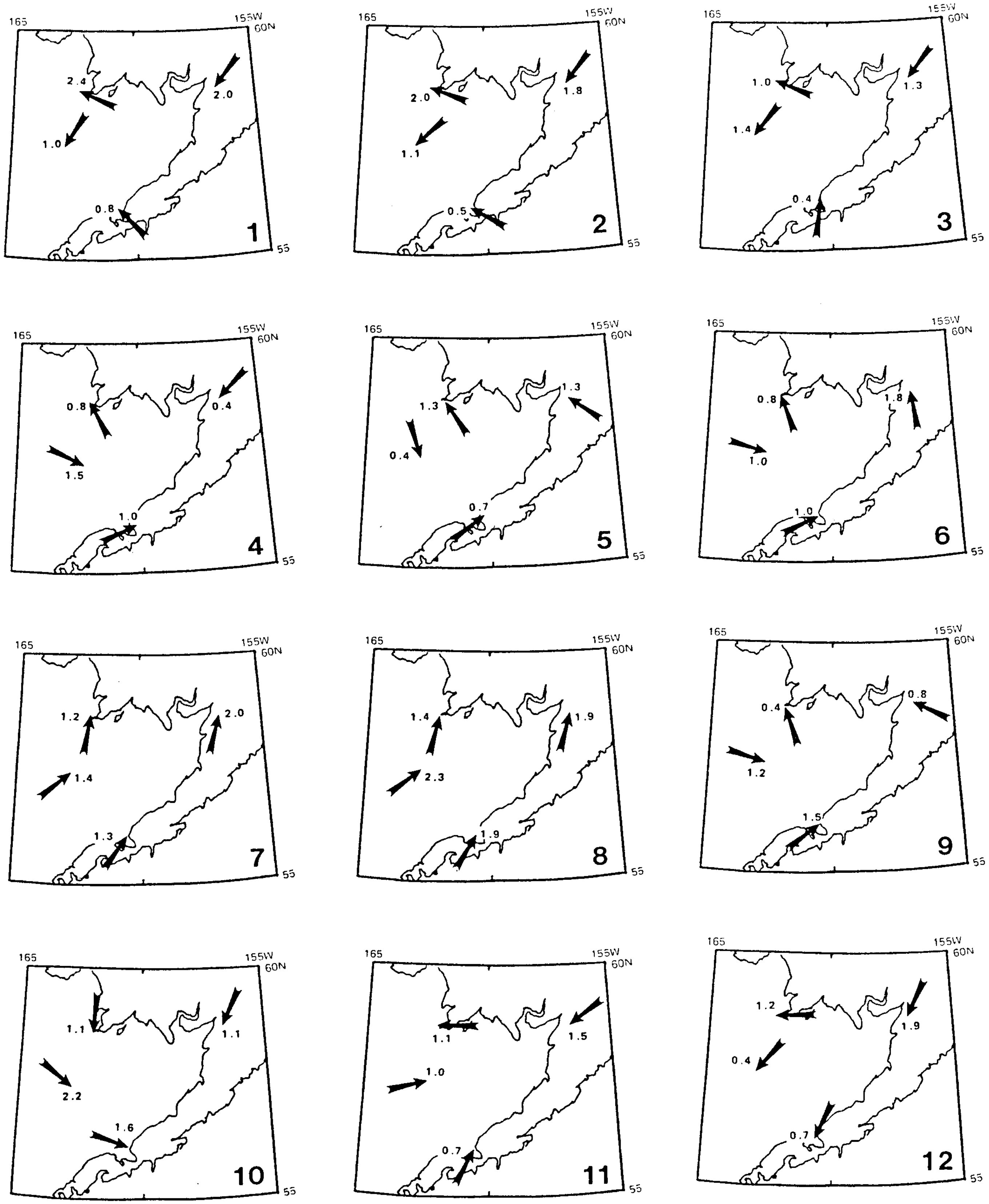
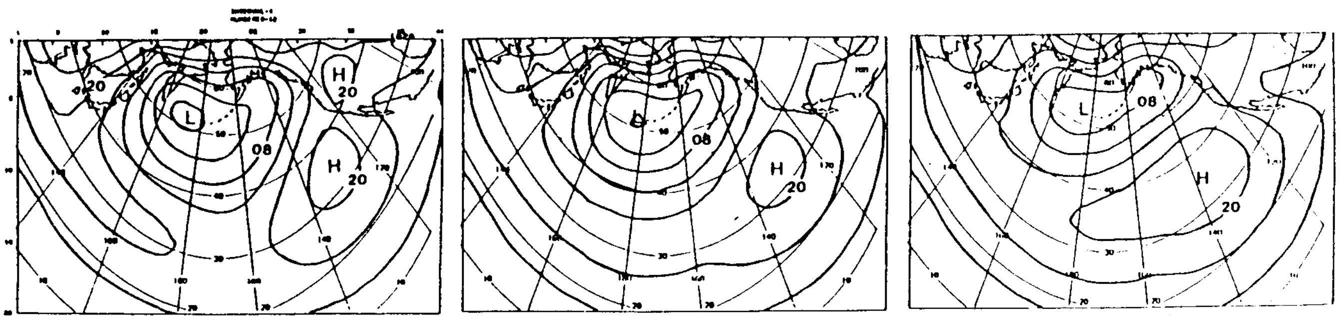
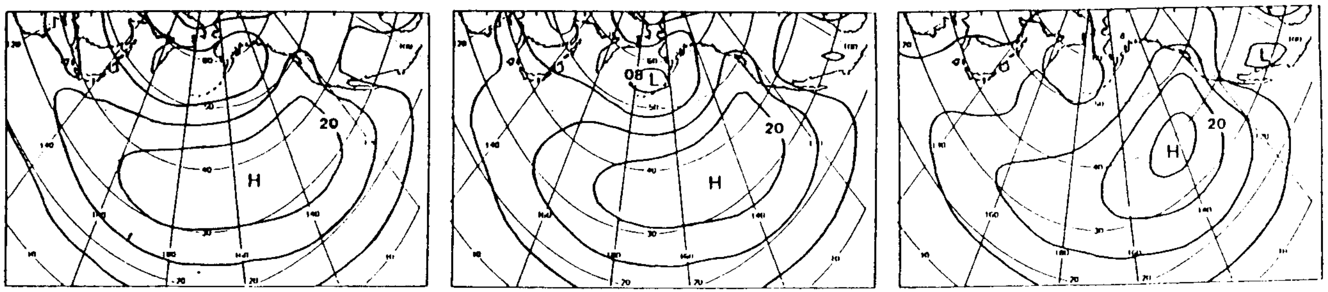


Figure 3.--Vector mean wind speeds (m/sec) and directions for each month (Brower et. al., 1977).

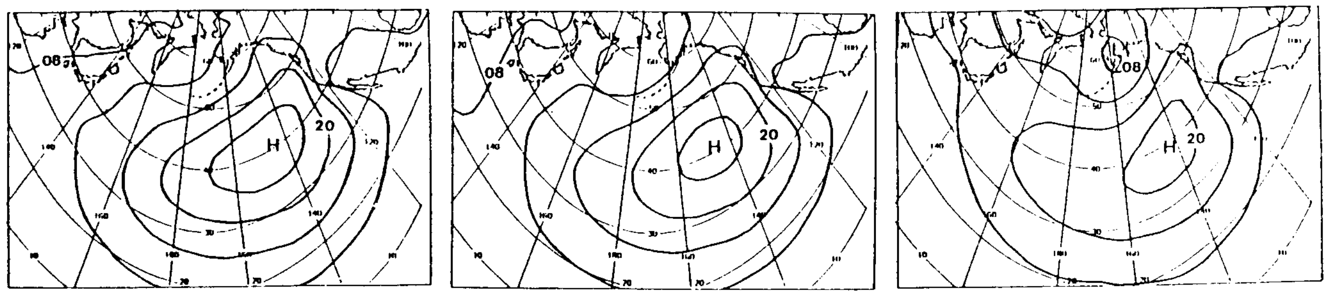
The relationship of Bristol Bay winds to the larger scheme of Pacific northern hemisphere air circulation can be seen in the long term (36 year) monthly averages of sea level pressure (Figure 4). Cyclogenesis starts the winter cycle in September in the western Gulf of Alaska, forming the Aleutian Low which rapidly deepens during November and December. This is the time when northeast winds are most probable in Bristol Bay. Northeast winds continue to dominate but are interrupted occasionally by southerly winds as the main low center migrates southwestward and enlarges in January-February. Then, as the Northeast Pacific High enlarges and pushes the Aleutian Low into the central Bering Sea by Spring, the tendency for southwest winds in Bristol Bay increases, becoming fully developed in July-August coincident with the summer precipitation maximum.



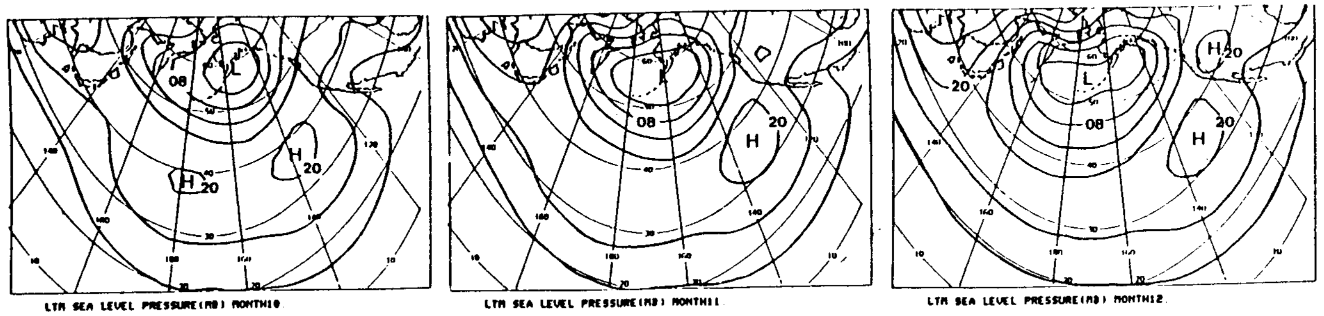
LTM SEA LEVEL PRESSURE(MB) MONTH1 LTM SEA LEVEL PRESSURE(MB) MONTH2 LTM SEA LEVEL PRESSURE(MB) MONTH3



LTM SEA LEVEL PRESSURE(MB) MONTH4 LTM SEA LEVEL PRESSURE(MB) MONTH5 LTM SEA LEVEL PRESSURE(MB) MONTH6



LTM SEA LEVEL PRESSURE(MB) MONTH7 LTM SEA LEVEL PRESSURE(MB) MONTH8 LTM SEA LEVEL PRESSURE(MB) MONTH9



LTM SEA LEVEL PRESSURE(MB) MONTH10 LTM SEA LEVEL PRESSURE(MB) MONTH11 LTM SEA LEVEL PRESSURE(MB) MONTH12

Figure 4.--Long term mean (LTM) sea level pressure −1000(mb) fields by month.

Bathymetry

The bathymetry of the Southeastern Bering Sea is characterized by a few highly irregular channels and ridges superimposed on an extremely broad gently sloping continental shelf (Figure 5). The most distinct features are the two troughs originating from the bays of the Kuskokwim and Kvichak rivers. The Kvichak trough, the most prominent one, lies alongside the Alaskan Peninsula while the Kuskokwim trough extends southward alongside Cape Newenham.

Along the Alaskan Peninsula depths of 20 m occur very close to shore, the exception being in the bays of Port Heiden and Port Moller where the depths do not exceed 15 m.

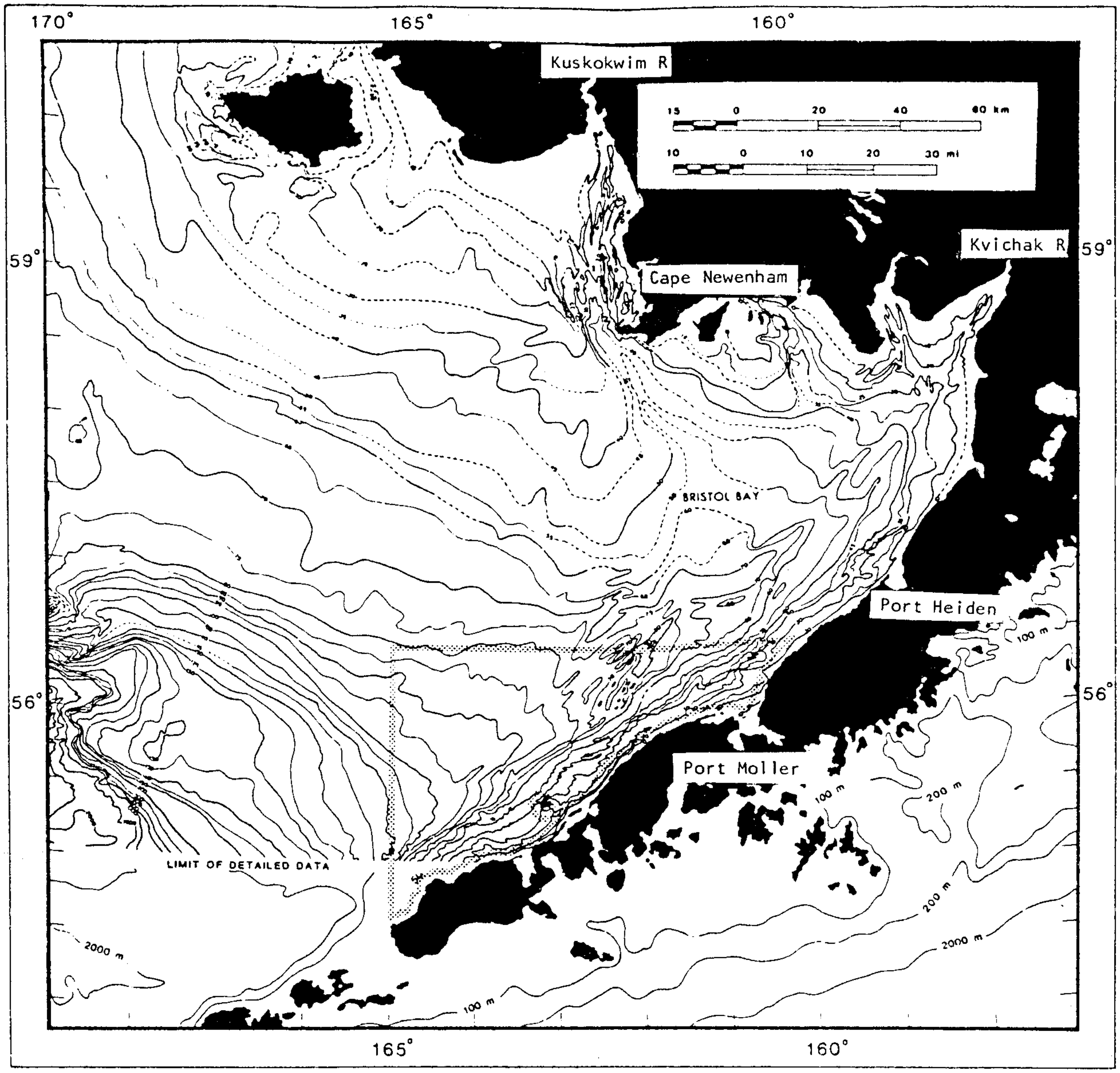


Figure 5.--Bathymetry (m) of the southern Bering Sea shelf.

Sediments

The great majority of the surficial sediments on the Bristol Bay shelf are sand with varying mixtures of silt, clay, and gravel (Sharma, 1979). Mean grain size decreases offshore (Figure 6) as water depth increases. The coarse to very coarse sands near shore are extremely poorly sorted, the medium and fine sands of the middle shelf are moderately well sorted, and as silt and clay components increase in the outer shelf sorting becomes poor again (Figure 7). The dominance of sand is quantitatively illustrated in the weight percent distributions of silt and clay (Figure 8) which are minimal (1 to 3%) on the middle shelf and increase to 40% and 10%, respectively on the outer shelf. Organic carbon content of sediments (0.05 to 0.55%) increases with increasing clay content (Figure 9).

In examining plots of sediment mean size versus water depth Sharma (1979) showed the inner and outer shelf could be distinguished based on sediment size (Figure 10). The inner shelf < 60 m depth was covered with sediments coarser than 3 Ø while the outer shelf > 75 m depth was mantled only with sediments finer than 3 Ø. The transition zone (60 to 75 m depth) contained both coarser and finer sediments as well significant changes in sorting and skewness. Although there appears to be a general directional agreement, this zonation of sediment size on the shelf does not strictly adhere to the well established hydrographic fronts at 50 and 100 m.

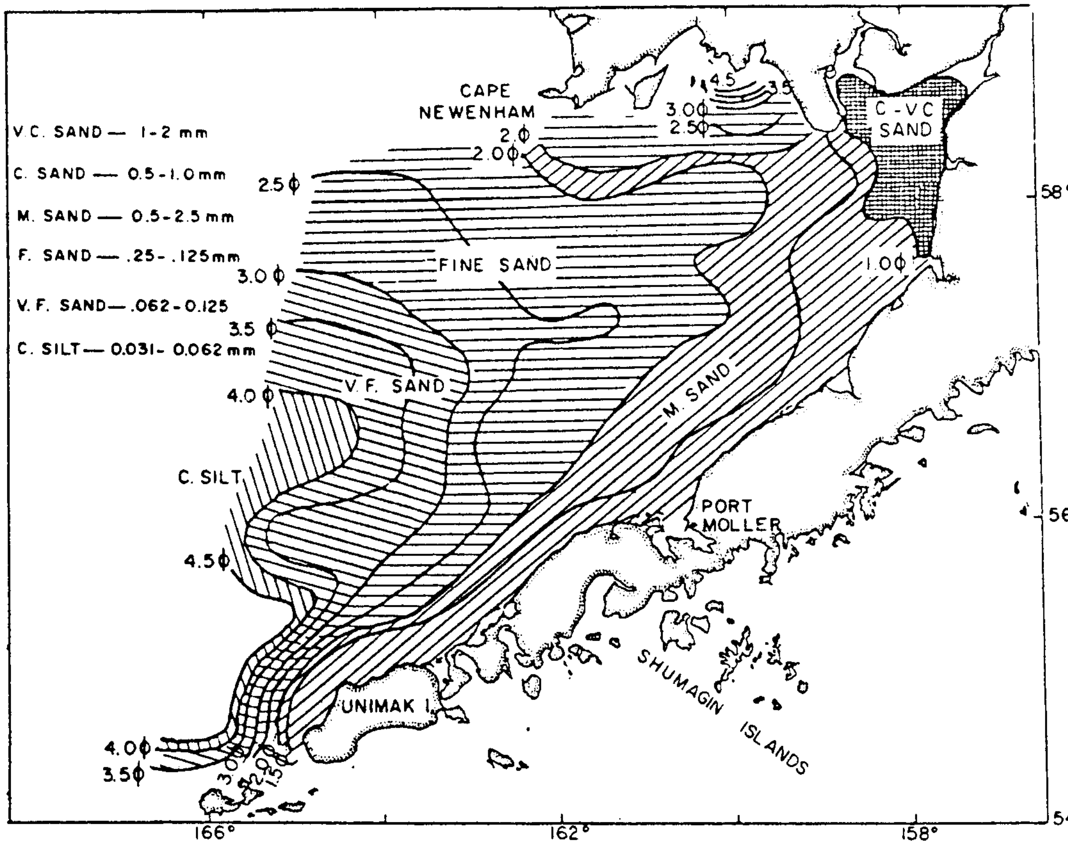


Figure 6.--Sediment mean size distribution (Sharma, 1979).

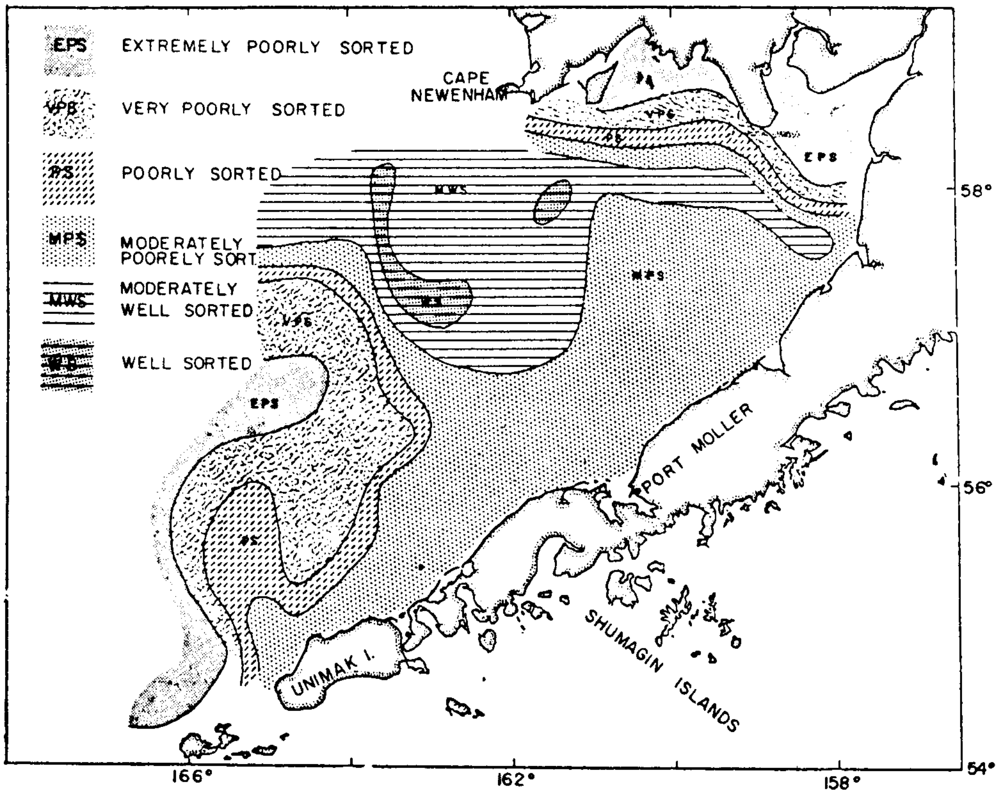


Figure 7.--Grain size sorting (Sharma, 1979).

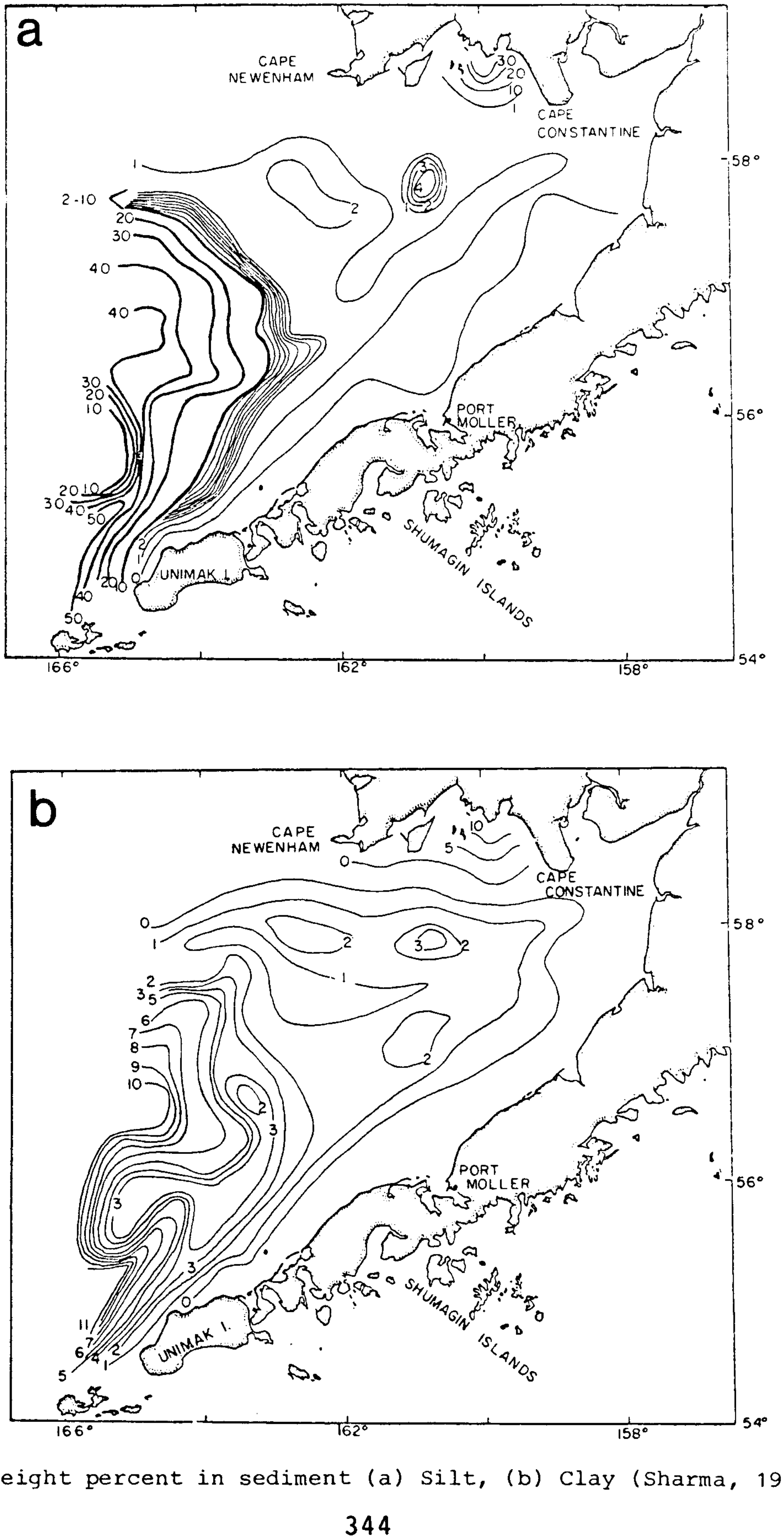


Figure 8.--Weight percent in sediment (a) Silt, (b) Clay (Sharma, 1979).

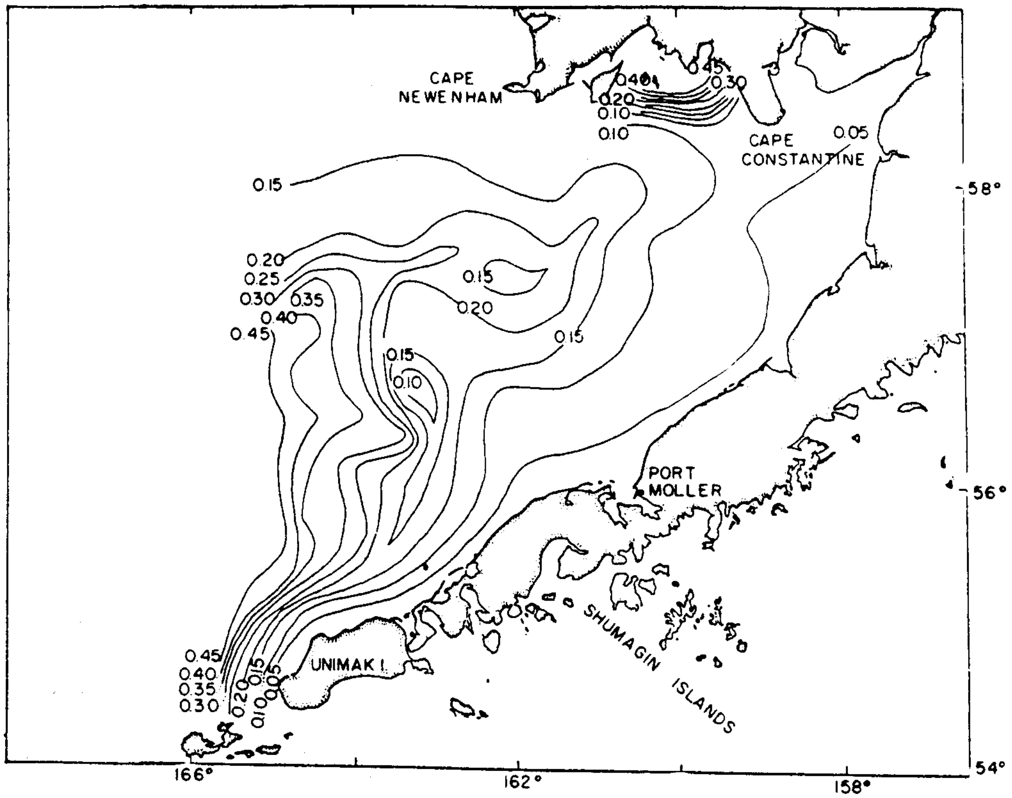


Figure 9.--Weight percent organic carbon in sediments (Sharma, 1979).

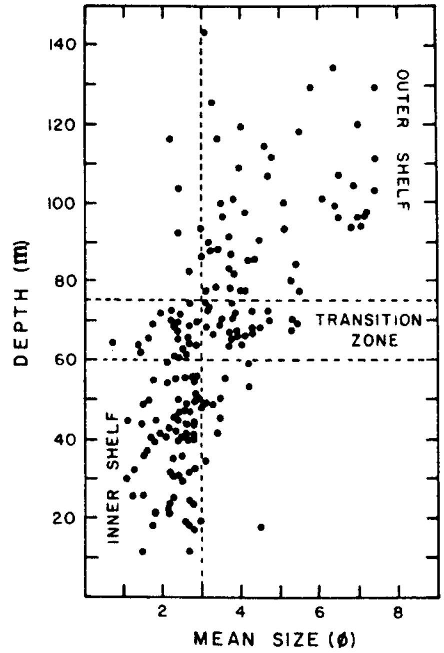


Figure 10.--Water depth (m) versus sediment size (0) for the Bering Shelf (Sharma, 1979)

Suspended Particulate Matter

The major source of suspended sediments in the Bristol Bay area is the Kuskokwim River which supplies four million metric tons per year; but contributions from lesser sources such as the Kvichak, Nushagak, or the Wood rivers are unknown, and no estimate has been made for coastal erosion as a secondary sediment source. Concentrations in any of these rivers can reach 500-2000 mg/l in suspensions (Sharma 1979). Although early studies have shown that these high concentrations in rivers decrease sharply in the coastal zone to <10 mg/l and decrease again to <1 mg/l about 75 km seaward of the Alaska Peninsula (Sharma 1979 and Feeley et al. 1981), their quantitative samples have been of limited extent or too broadly spaced to show the important details in the vertical distribution of suspended particulate matter that were later shown by Baker (1983).

Relative importance of Bristol Bay rivers was semi-quantitatively evaluated by Sharma (1979), using the horizontal distribution of surface sediment plumes on density sliced ERTS-1 imagery after the time of the secondary river runoff peak of August and September (Figure 15). Imagery (isodensity distribution of reflectance) on 15 October 1973 shows three prominent sediment sources in inner Bristol Bay (Figure 11a): (1) Port Heiden where the sediment plume from the harbor extends northeastward along the shore, (2) Kvichak Bay where the sediment plume from the Kvichak River extends westward along the north coast, and (3) Nushagak Bay where the plume fills most of the bay, particularly the western side, then disperses westward into Togiak Bay. Imagery on 5 November 1973 (Figure 11b) shows the plume from Togiak Bay extending westward around Cape Newenham, two plumes from the mouth of the Kuskokwim River extending southwestward and southeastward along the shorelines on each side of Kuskokwim Bay, and the most striking feature is a third and

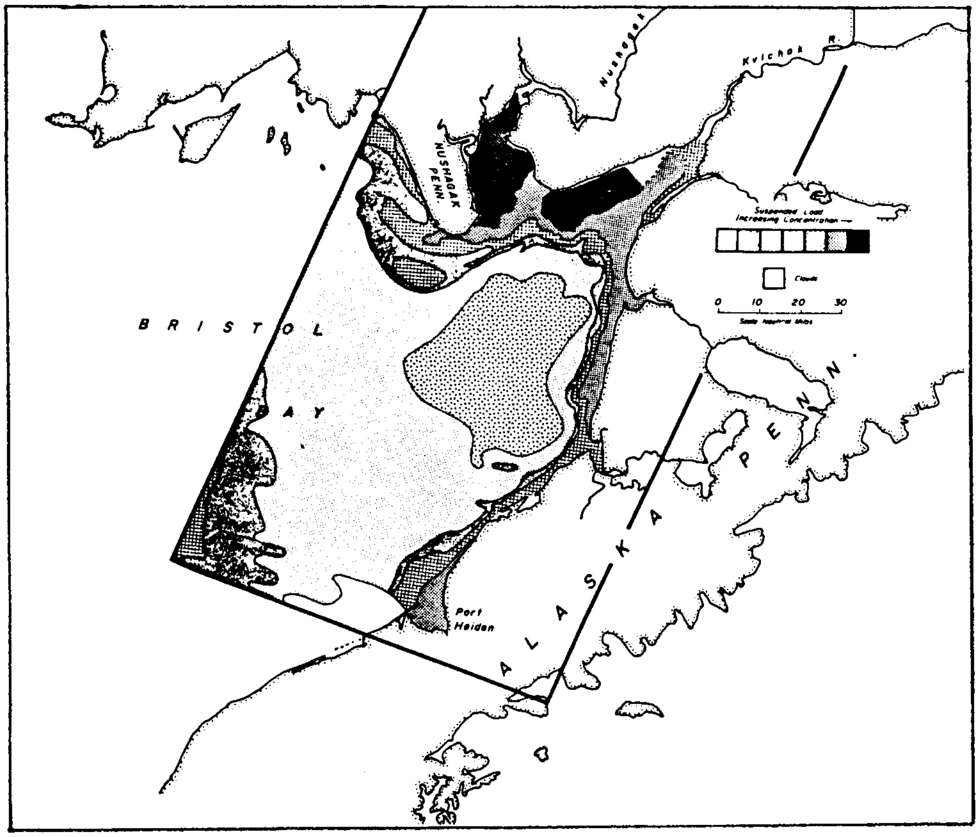


Figure 11a.--Satellite imagery showing relative suspended load in near-surface waters on October 15, 1973 (Sharma, 1979).

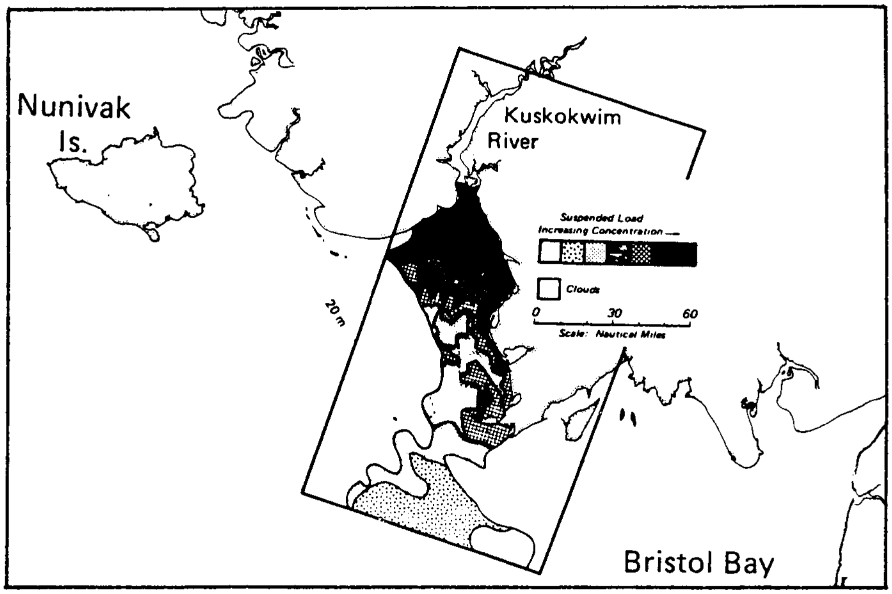


Figure 11b.--Satellite imagery showing relative suspended load in near-surface waters on November 5, 1983 (Sharma, 1979).

broader plume from Kuskokwim Bay extending seaward beyond the coastal zone. Direct measurements, although they were too widely spaced to give details, generally confirmed the features seen in the imagery. Concentrations in the offshore Kuskokwim plume reached 2.2 mg/l in a background of samples generally <1 mg/l outside the plume. Measurements at depth (50-75 m) showed slightly higher concentrations of 2.5 mg/l further westward under the plume. Measurements made near bottom in the coastal zone indicated a two to three fold increase in suspended sediment load during storms due to resuspension.

Additional quantitative measurements of total suspended matter were obtained in Bristol Bay during September-October, 1975 and June-July 1976 by Feeley et al. (1981). These fall surface data (Figure 12a) showed a large plume of suspended matter extending southward from Kuskokwim Bay and a rapid decrease in concentration away from the coast along the Alaska Peninsula confirming the general features discussed by Sharma (1979). The more extensive and detailed data taken near bottom (Figure 12b), however, showed higher than surface concentrations (>1.0 mg/l) throughout the area except for a narrow tongue of less turbid (<1.0 mg/l) water extending into Bristol Bay from offshore just seaward of and parallel to the band of very high concentrations (>2.0 mg/l) in the coastal zone along the Alaska Peninsula.

The latest study (August-September, 1980; January-February, 1981; and May-June, 1981) sampled in great detail both horizontally and vertically along the north coast of the Alaska Peninsula using continuous vertical profiling instruments to obtain both water and suspended particulate structure (temperature, salinity, and light attenuation versus depth) at each station (Baker, 1983). This is the first time the vertical distribution of fine-grained particles has been shown directly related to the hydrographic structural domains (defined based on vertical density structure), a relationship reasonable

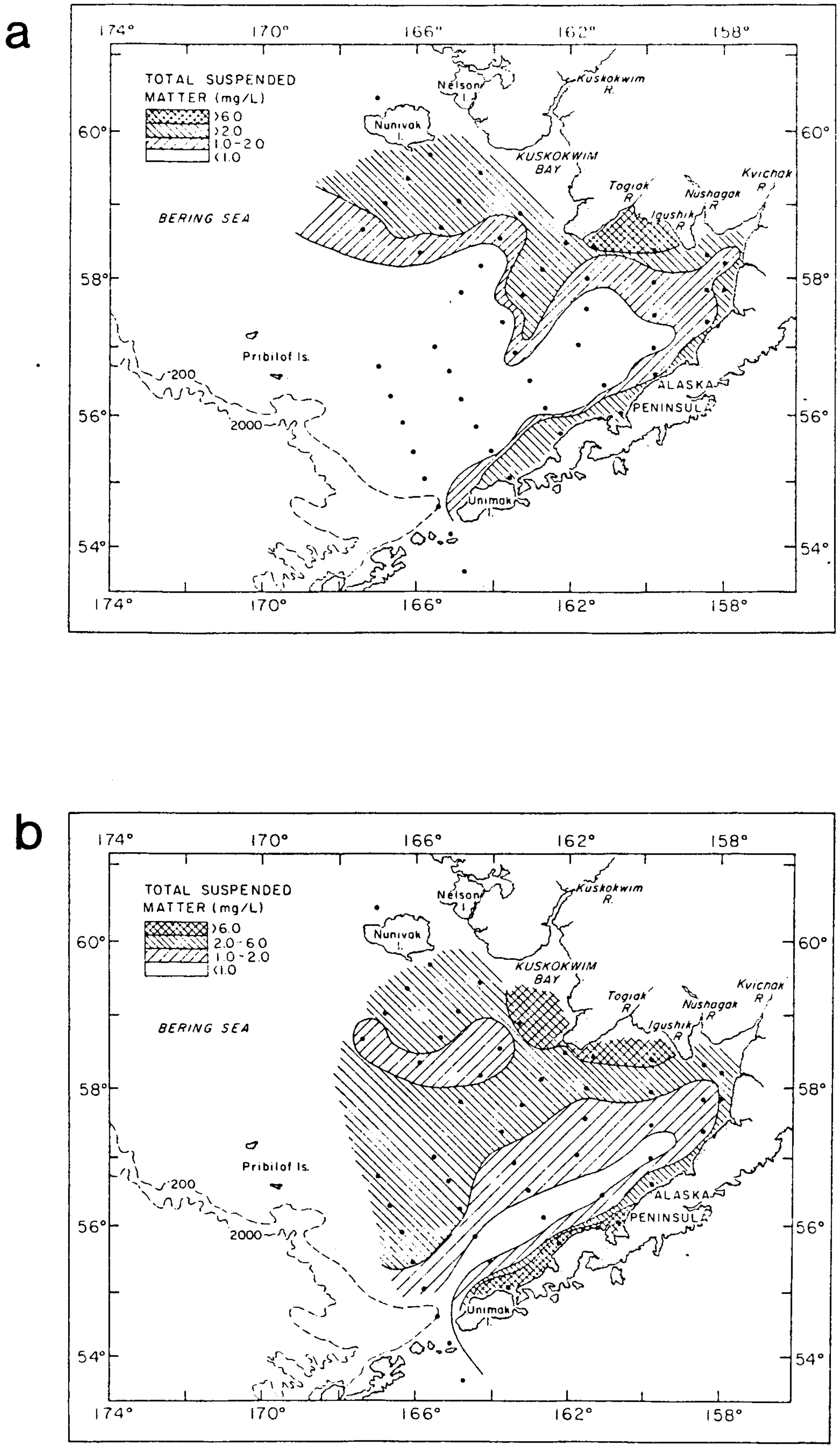


Figure 12.--Distribution of total suspended matter at the surface (a) and 5 meters above the bottom (b) in the southeastern Bering Shelf (cruise RP-4-D: 75-B-III, 12 September - 5 October 1975) (Feely et al. 1981).

to expect because the vertical density structure controls the vertical distribution of turbulence which in turn governs the distribution of particles whose settling velocity is much less than the mean horizontal current. In the coastal domain (about 0-50 m depth) the water properties and particle concentrations are well mixed, as shown in figure 13a by representative vertical profiles of attenuation, Brunt-Vaisala frequency (stability), and sigma-t (density). Although the vertical profiles are consistently well mixed, a strong horizontal gradient is present in the coastal domain with particle concentrations decreasing rapidly off shore. In contrast the mid-shelf domain (about 50-100 m depth) has very little horizontal change and a strong vertical structure, two layers in density and three layers in particle concentration (Figure 13b). In the upper, low density layer suspended particle concentrations are a maximum at the surface then decrease to a minimum layer between 30 and 45 m depth. The strong pycnocline near 50 m puts a diffusive lid on the lower, high density layer thus limiting the upward spread of the higher concentrations of resuspended particles near the bottom due to the consistently strong tidal currents. These same features are seen in the outer-shelf domain (Figure 13c) except the pycnocline has expanded over a greater depth range (50 to 70 m) creating a relatively high stability third layer in the water structure.

Seasonal changes and local variability in these major structural features can be seen in vertical cross-sections normal to the coastline seaward of Port Moller during August, 1980, January, 1981, and May, 1981 (Figure 14a, b, and c). Conditions during summer (Figure 14a) showed a weak two-layer density structure with a pycnocline separating the shallow (<about 40 m), turbid, well mixed coastal water from the deeper (> about 50 m) water with the typical three-layer structure of particulate matter. A distinct minimum concentration

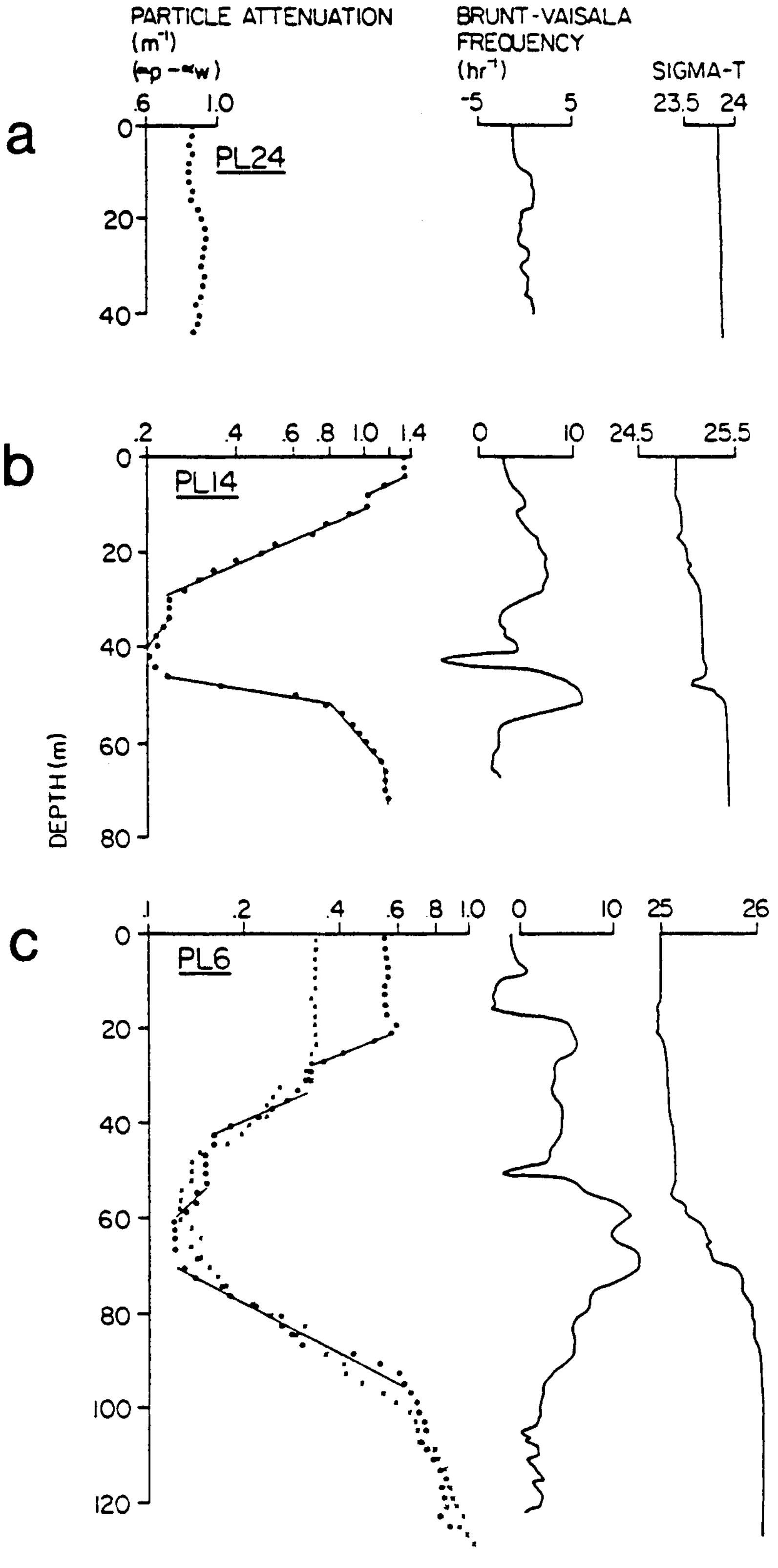


Figure 13.--Net particulate attenuation profiles, stability profiles, and density profiles from the (a) coastal (b) middle and (c) outer -shelf domains (Baker, 1983).

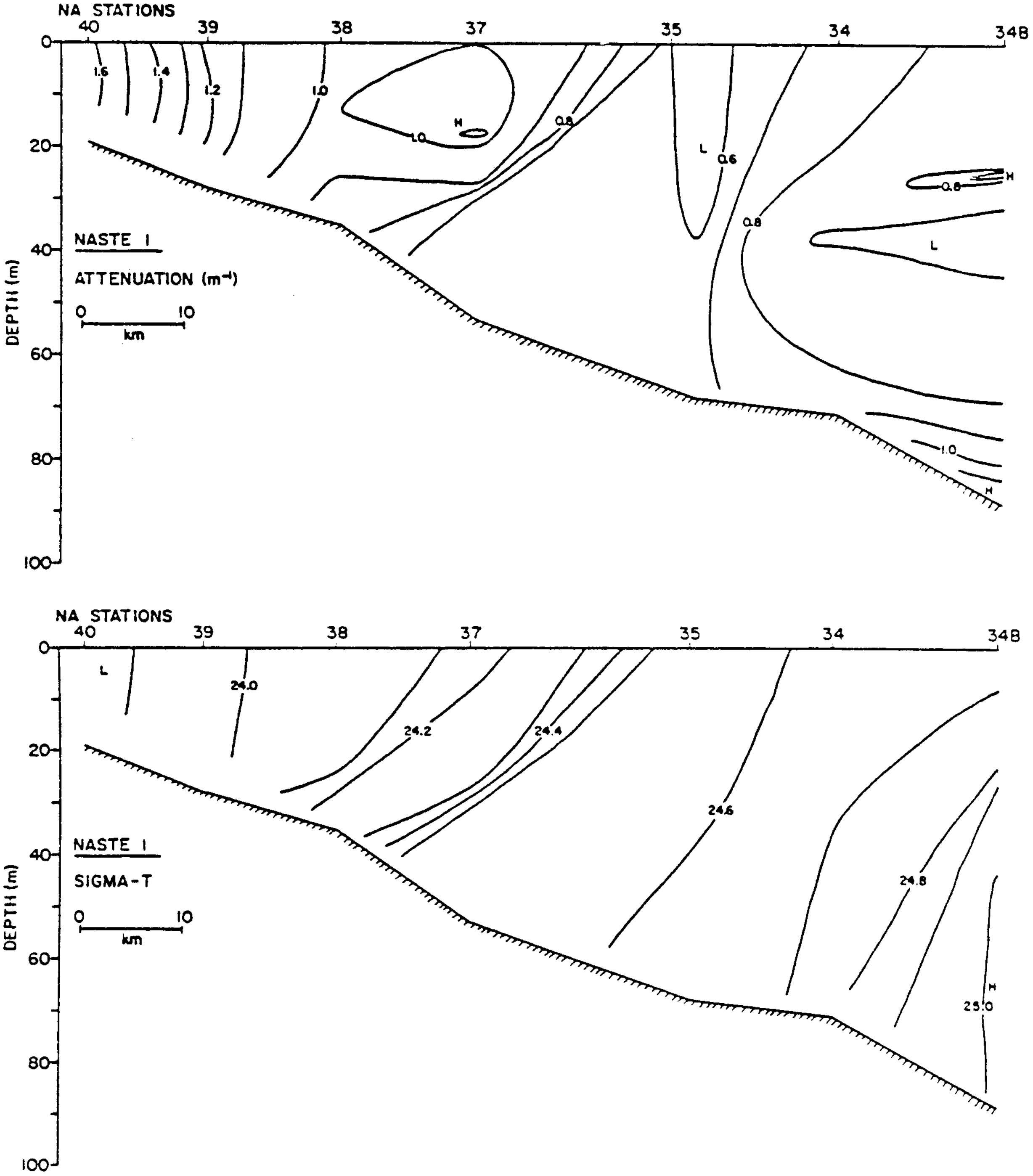


Figure 14a.--Attenuation (top) and density (bottom) cross-sections for line 2 (station NA 34-40), August 1980.

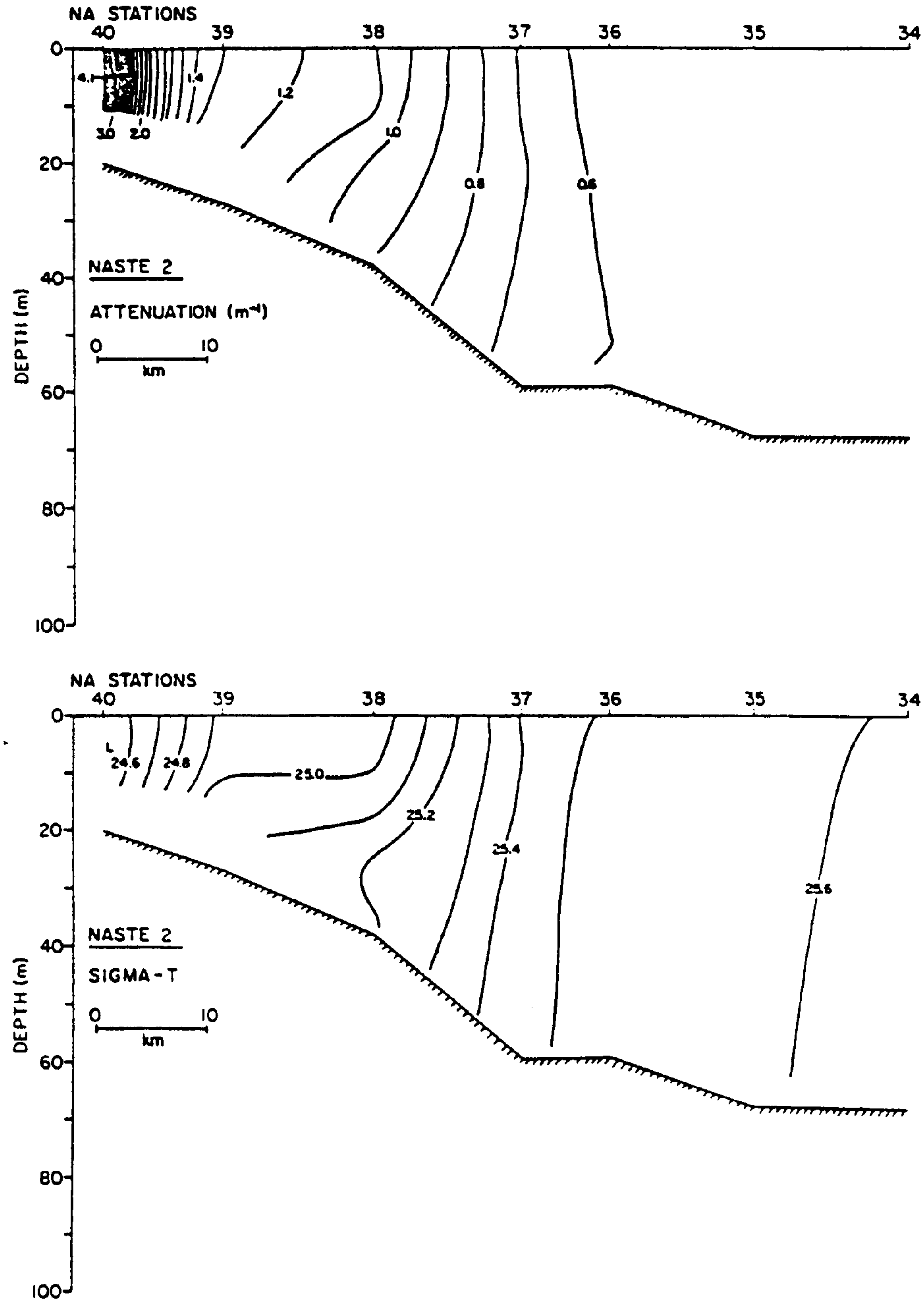


Figure 14b.--Attenuation (top) and density (bottom) cross-sections for line 2 (station NA 34-40), January 1981.

of particulates was evident under the pycnocline along the bottom between about 50 and 60 m. During winter (Figure 14b) conditions changed significantly. Concentrations of particulate matter decreased and density increased seaward in their typical pattern, but mid-shelf domain stratification was essentially absent to at least 80 m depth. Stratification returned during spring (Figure 14c) with isolated maxima in particulates occurring just below the pycnocline and near bottom in the mid-shelf domain. Commensurate with the normal seasonal cycle in biological activity, the organic portion of the suspended particulates ranged from as low as 25% for inshore stations during winter to >50% for offshore surface water samples during spring.

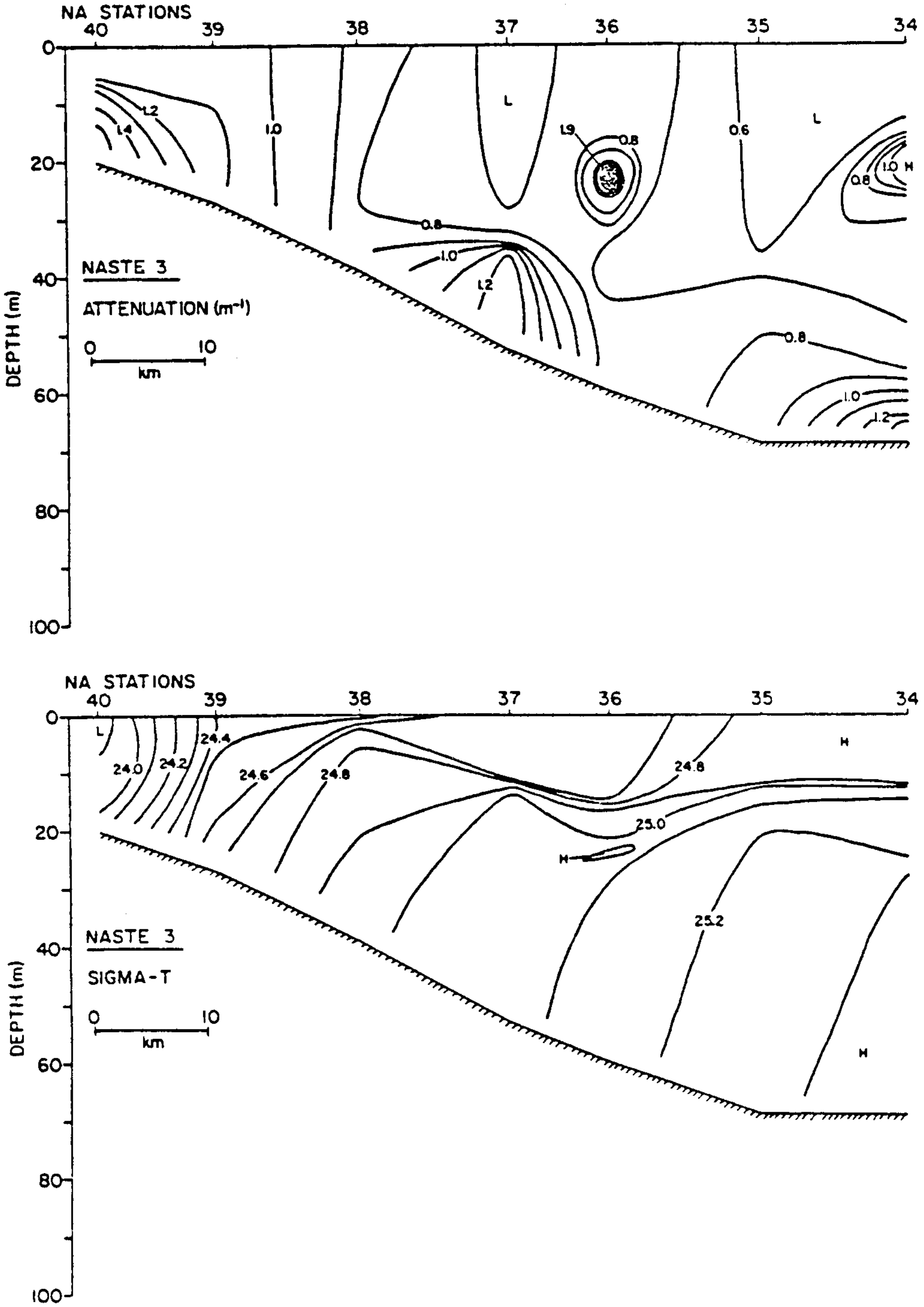


Figure 14c.--Attenuation (top) and density (bottom) cross-sections for line 2 (station NA 34-40), May 1981.

River Runoff

The largest southeastern Bering Sea river, the Kuskokwim, ranks third with 990 m^3/sec in mean annual fresh water discharge into the Bering Sea (Roden, 1967) after the Yukon River with 6220 m^3/sec and the Anadyr River with 1660 m^3/sec. The following seasonal changes of runoff in the Kuskokwim River also reflect the characteristics of its smaller neighbors around Bristol Bay, the Togiak, Wood, Nushagak, Kvichak, Naknek, Egegik, and Ugashik Rivers. River discharge is consistently low (<500 m^3/sec) during the winter ice season from December to April. It is followed by a sharp one month rise to about 2000 m^3/sec during May, then the annual maximum (about 2700 m^3/sec) in June (Figure 15). This sharp rise is associated with the spring melt during these months. In July a slight relaxation occurs, followed by a secondary maximum during August (about 2400 m^3/sec) associated with the annual precipitation maximum. September values remain relatively high, but the river discharge declines sharply during October and November, returning the cycle again to its winter low.

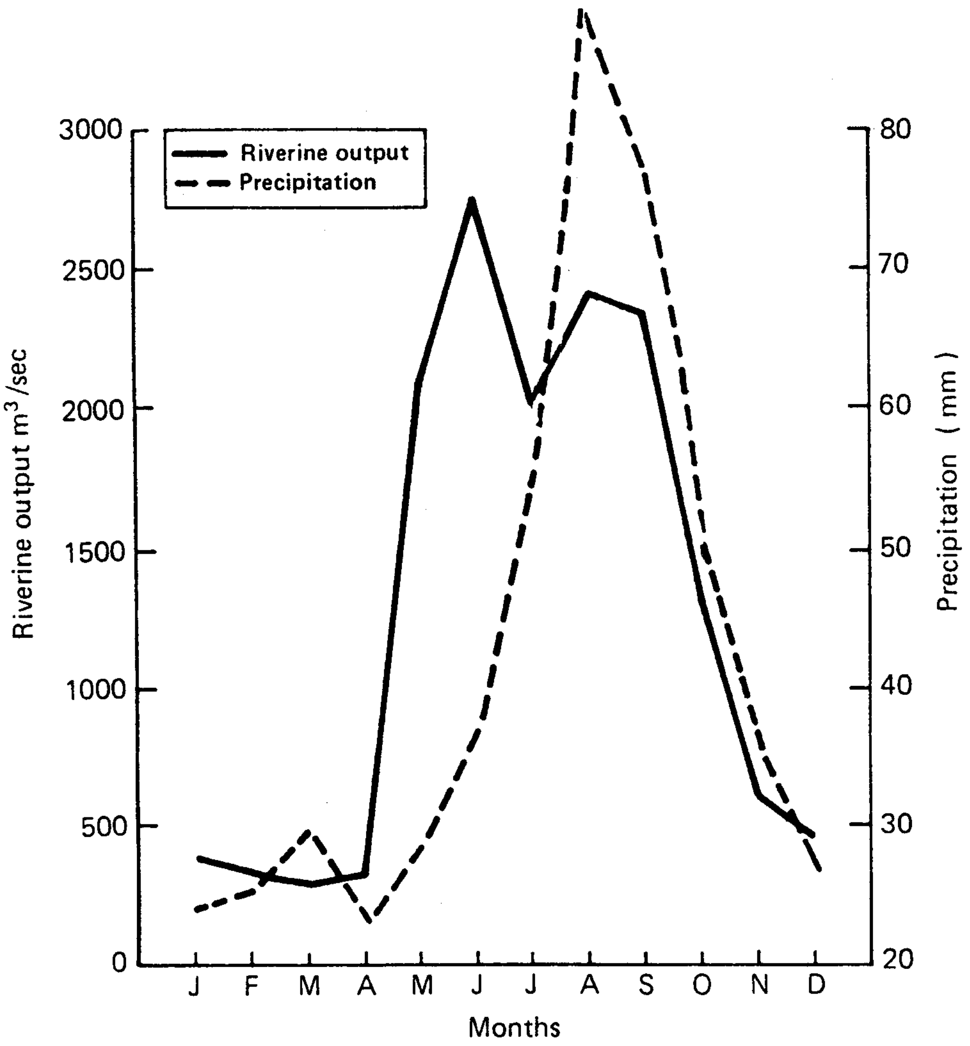


Figure 15.--Average monthly precipitation and riverine runoff for the Kuskokwim River. (Data from Ruffner et. al., 1978 and Roden, 1967).

Tides

Pressure and current meter data taken during 1975-78 provide new information about tidal heights and tidal currents in the Bristol Bay area (Pearson, Mofjeld, and Tripp, 1981). Cotidal charts prepared from these data show the distribution of tidal coamplitudes (the range in height in cm, dashed lines) and cophase (0° to 360° referred to Greenwich, solid lines) for the four major tidal components-M_2, N_2, K_1, and O_1 (Figure 16a, b, c, d). Because the sum of the amplitudes of the semidiurnal components (M_2 + N_2) is about twice the sum of the diurnal components (K_1 + O_1), the tide type is classified as mixed, predominantly semidiurnal, having two highs and two lows per day and large diurnal inequalities. Offshore in outer Bristol Bay near midshelf amplitudes of all components are small, about 20 to 40 cm. The tidal range then increases greatly toward shore as the water depth shallows. Maximum amplitudes greater than 200 cm occur in inner Bristol Bay embayments where M_2 is about three times larger than K_1, and K_1 is about twice the nearly equal O_1 and N_2 components. At coastal stations, average predicted diurnal ranges also increase toward inner Bristol Bay, being 330 cm at Port Moller and 689 cm at the Naknek River entrance where the maximum range is expected to be as high as 915 cm (Brower et al., 1977). Cophase lines (Figure 16a, b, c, d) indicate the tide enters the Bering Sea through the Aleutian Island passes and progresses as a free wave along the north side of the Alaska Peninsula into Bristol Bay where it slows and reflects. As the wave passes any one spot, currents change with time with the instantaneous current vector appearing to rotate either clockwise or counterclockwise as viewed from above. This rotation is seen in the ellipse representation of the major components M_2 and K_1 (Figure 17a,b). Ellipses in the figure are centered at the observing station and

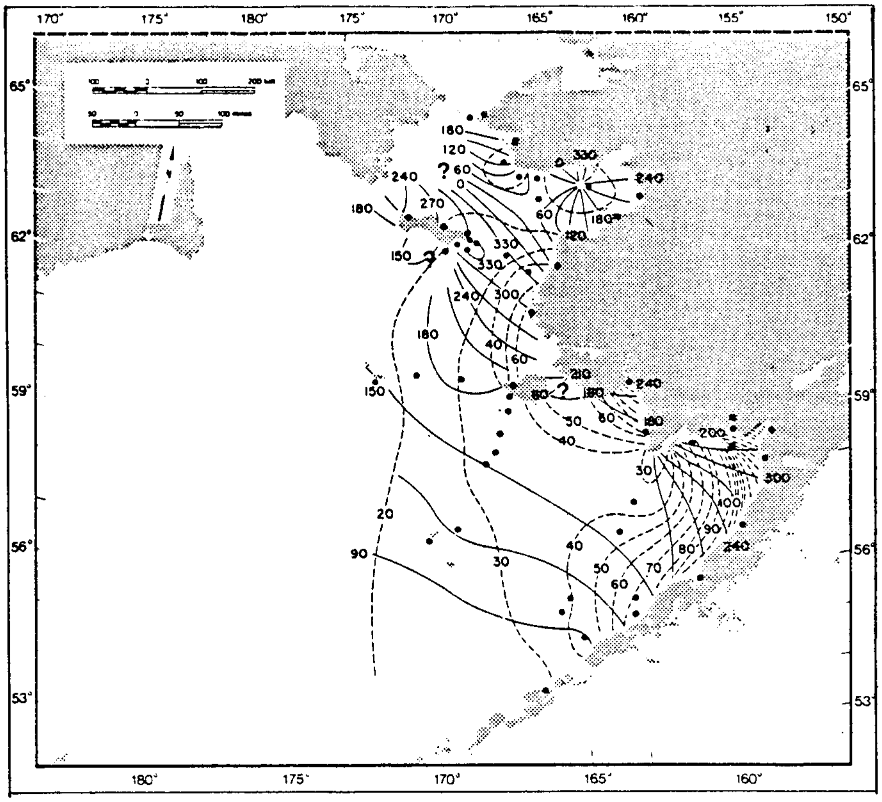


Figure 16a.--Co-tidal chart of the semi-diurnal component M_2. Dots refer to the stations (Pearson et al., 1981).

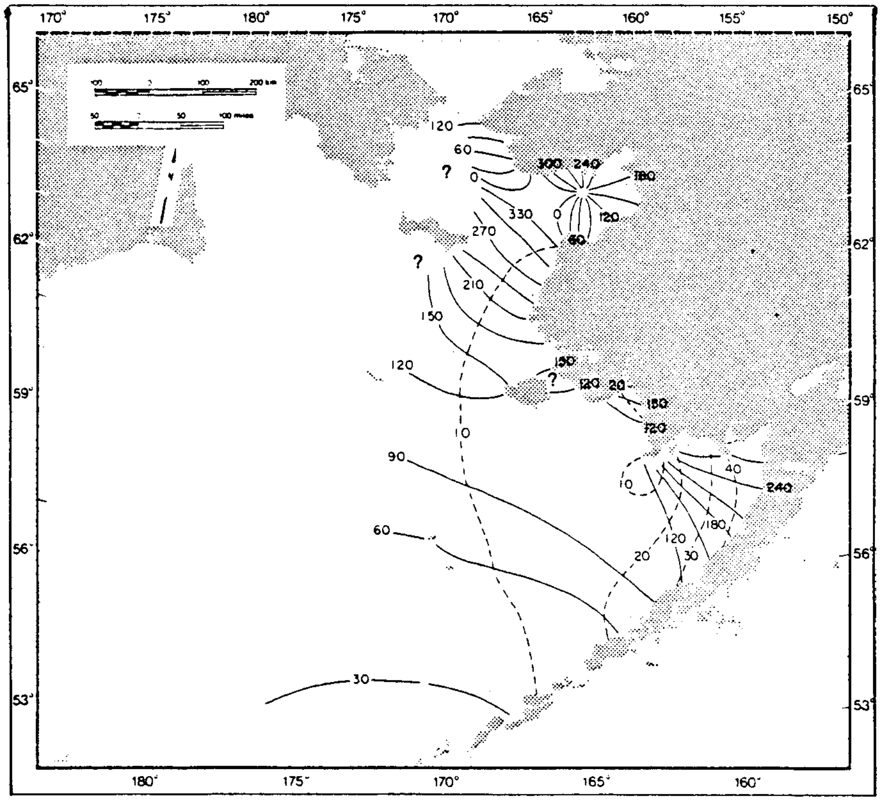


Figure 16b.--Co-tidal chart of the semi-diurnal component N_2 (Pearson et al., 1981).

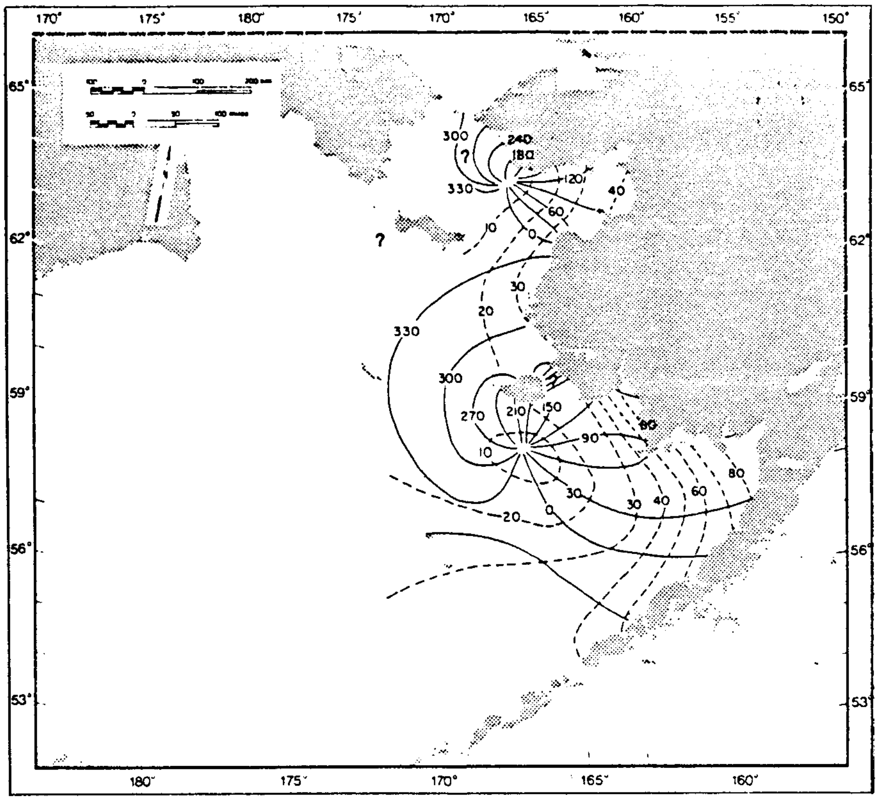


Figure 16c.--Co-tidal chart of the diurnal component K_1 (Pearson et al., 1981).

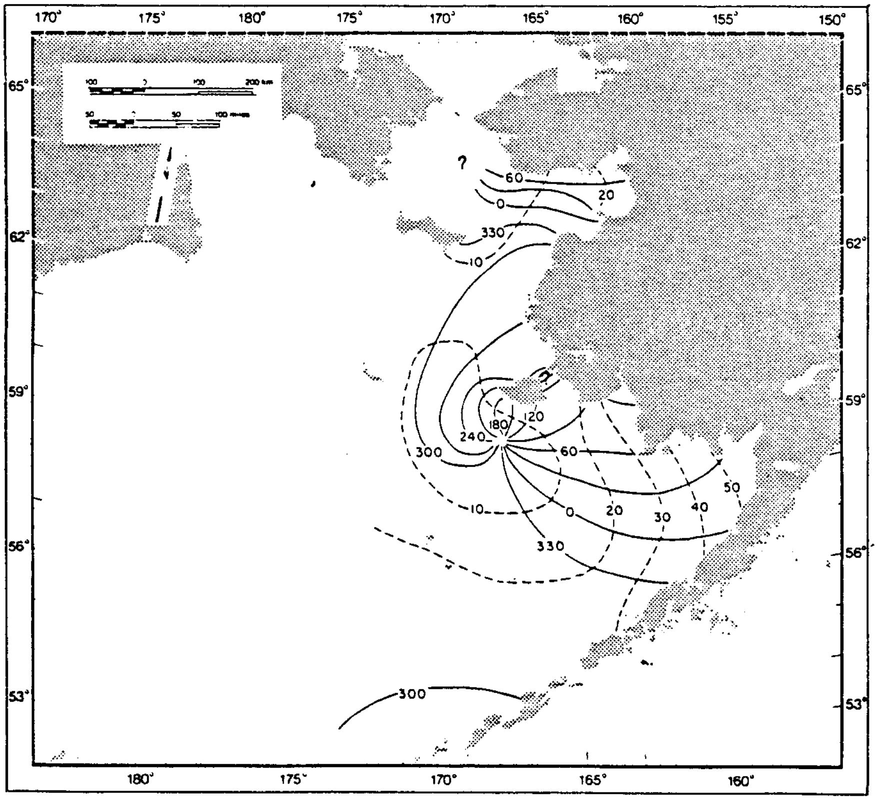


Figure 16d.--Co-tidal chart of the diurnal component O_1 (Pearson et al., 1981).

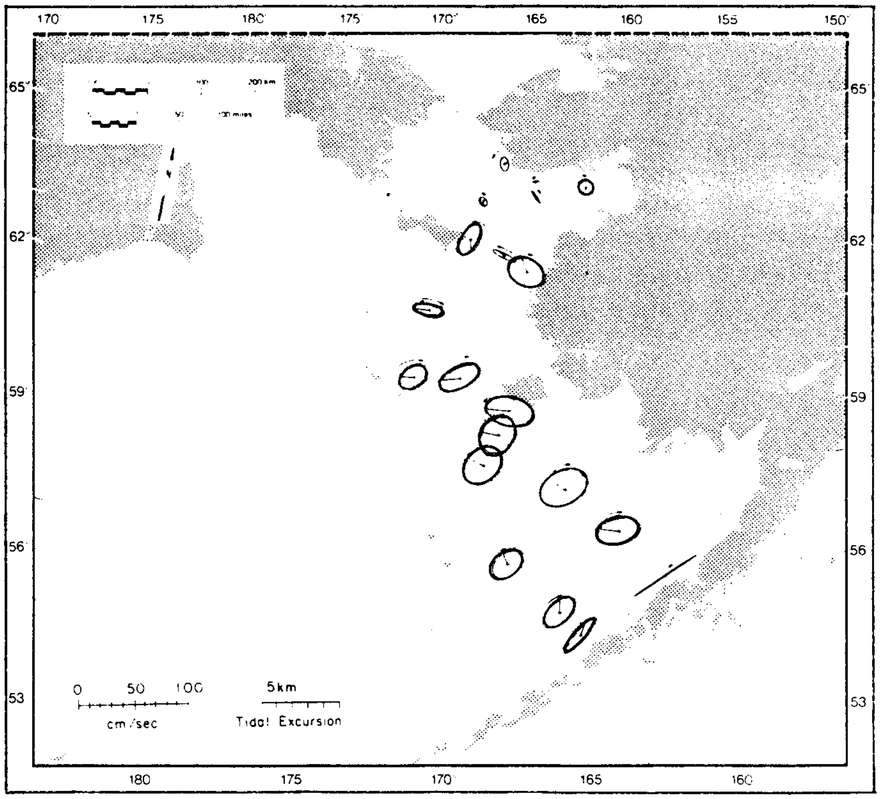


Figure 17a.--M_2 current ellipses (Pearson et al., 1981).

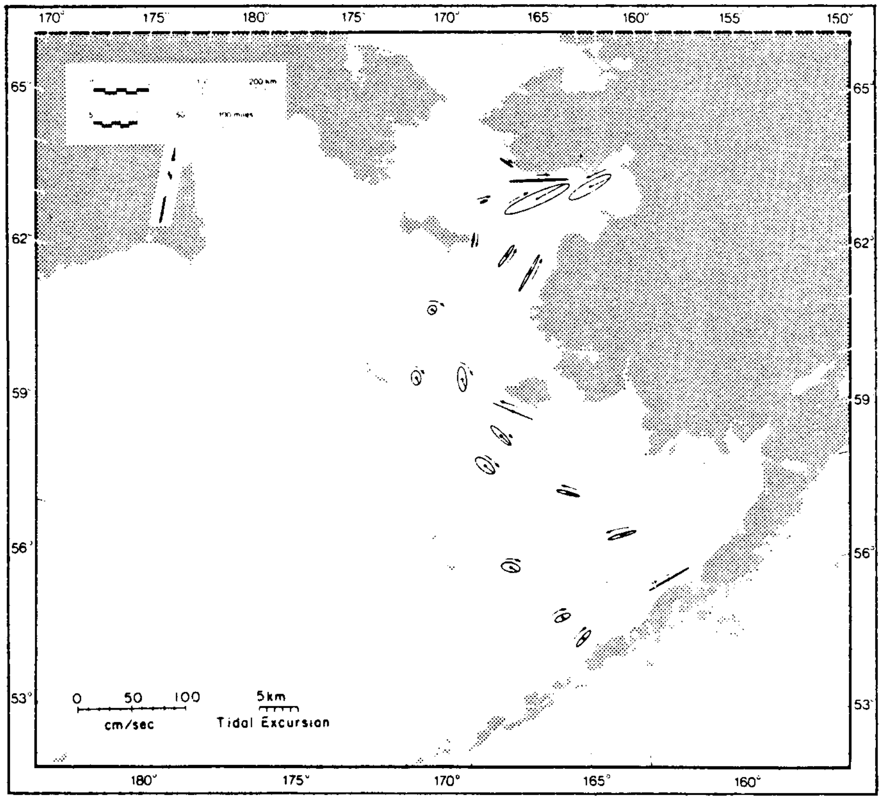


Figure 17b.--K_1 current ellipses (Pearson et al., 1981).

the current vector from the center to the ellipse traces out the amplitude and direction of the current with time. They tend to be more circular away from land with the major axis aligned with the direction of the tidal wave propagation. M_2 rotates clockwise with typical speeds of 15-30 cm/sec throughout Bristol Bay and shows a strong topographic influence, becoming nearly rectilinear north of the Alaska Peninsula. The other semidiurnal component, N_2, shows similar features but is about 25-40% of M_2. The K_1 ellipses are narrower than the M_2 ellipses and rotation is counterclockwise, opposite to that of the M_2 ellipses. K_1 flow speeds of 10-20 cm/sec are one third slower and more directionally aligned into and out of Bristol Bay. The secondary diurnal component, 0_1, is similar to K_1 but small in amplitude by 60-75 percent. Although this discussion of the major periodic components gives good insight into the dominant features which are possible in the flow regime, it must be remembered that the actual flow is forced by the sum of all the interactions of these components plus the added effects of wind, bottom topography, and internal density structure. Instantaneous pictures of the flow field can only be simulated by sophisticated numerical models such as those developed by Liu and Leendertse (1979).

Non-tidal Currents

After the semidiurnal (twice daily) and diurnal (daily) astronomically forced tidal flow which dominated 60 to 90 percent of the total flow energy was filtered out of the current meter data, a low-frequency flow, quasi-periodic at about one week, accounted for the next 3-20 percent and the long term mean flow made up only 1 percent of the total flow energy (Kinder and Schumacher, 1981). Although the mean flows are calculated from vector averages over the individual current meter record lengths which are relatively short (3-7 months) compared to the desired but not available continuous multi-year measurements, there was sufficient data from 1975-78 and enough in each season to compile a general summary.

Three distinct regimes were defined for the Bristol Bay area based upon their mean and wind-driven flow characteristics (Figure 18). The coastal regime, from the shoreline to the 50 m isobath, has a mean current flowing northeastward at 2-5 cm/sec following the local bathymetry of the Alaska Peninsula into Bristol Bay then turns northwestward at 1-3 cm/sec out of Bristol Bay along the Alaskan coast. This flow is augmented by occasional wind-driven pulses of a few days duration. The middle regime, between the 50 and 100 m isobaths, has an extremely weak flow (<1 cm/sec) about the same order of magnitude as the error estimates in the data. Wind-driven pulses appear stronger here than in the coastal regime. The third, or outer regime, beyond the 100 m isobath to the shelf break, has a northwestward mean flow of 1-5 cm/sec parallel to the shelf break and low frequency events associated with variations in the Bering Slope Current rather than related to local winds.

Seasonal changes in flow are expected because winter mean wind speeds are higher, mean wind direction reverses compared to summer, and storms

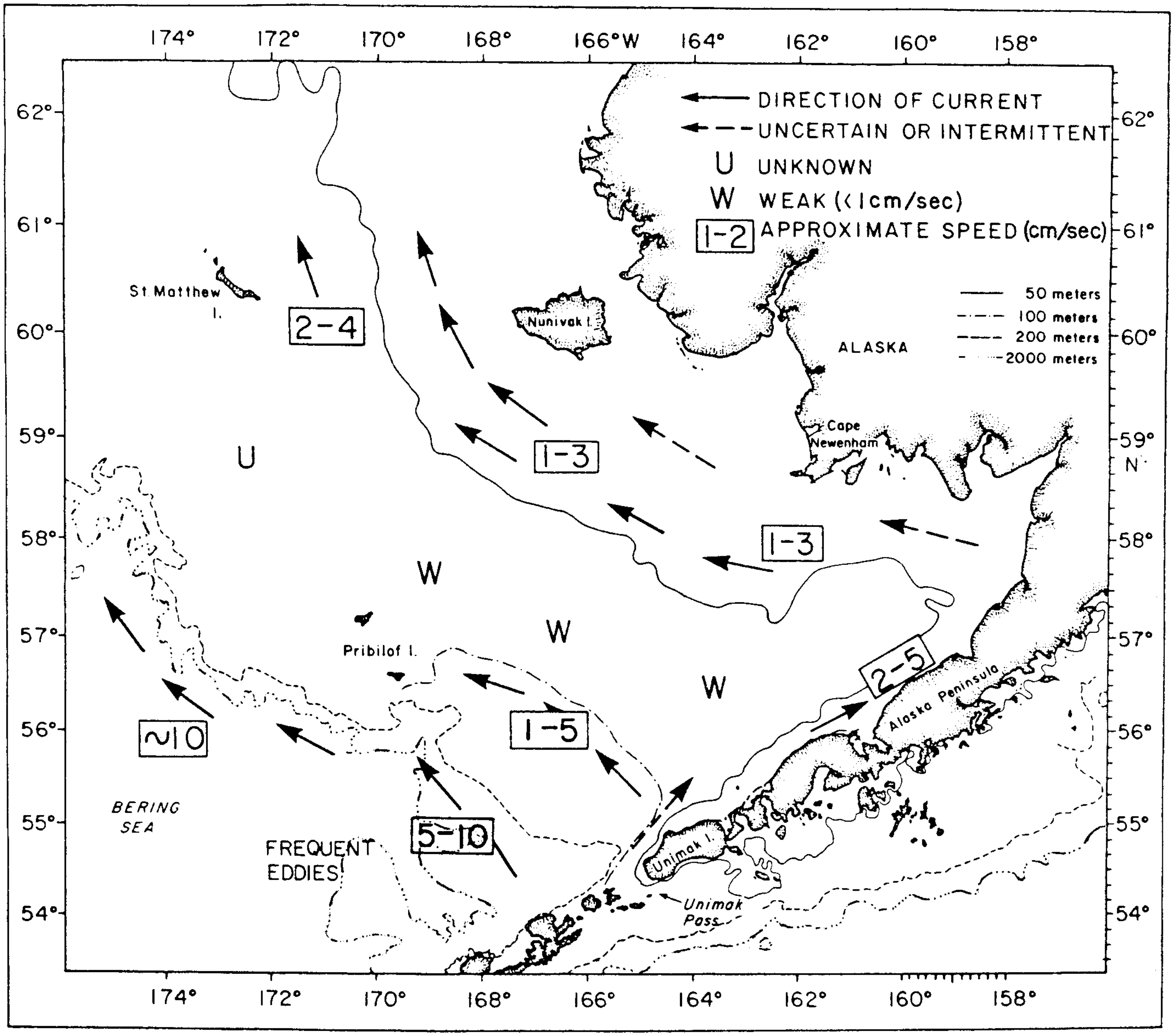


Figure 18.--Mean circulation of Southeastern Bering Sea (Kinder and Shumacher, 1981).

more frequent. Recalling that the above mean flows were measured for periods shorter than one year, their summer-winter change was small but significant. Some data from the coastal regime indicated up to a three-fold increase in mean speed without a change in direction. A two-fold increase in mean speed also occurred in the middle regime of winter, but direction reversed from westward in summer to eastward in winter.

These changes in seasonal mean flow appear related to the cumulative effect of wind forcing from two types of low frequency events. The first type is the eastward-traveling low atmospheric pressure centers which slowly cross the Bristol Bay area mainly during the summer and also occasionally throughout the year. The second type, from large winter low pressure centers in the Gulf of Alaska, is a southward outbreak of cold continental air which frequently produces southerly pulses of surface current at 15-30 cm/sec, lasting 1-5 days. It is suggested that the more complete, meterological data set be studied for the frequency of these events, thus potentially indicating an index of past interannual variations.

Water Temperature

Variations of surface temperature are primarily affected by insolation with secondary effects being air temperature, wind speed, river runoff, cloud cover, and precipitation. Monthly means (Figures 19a and 19b) show the effect of insolation, indicating a warming trend starting during the months of April to May, reaching a maximum temperature of over 10°C during August. The cooling period starts during September to October with coastal ice beginning to form during November and ice expansion continuing until March or April (Ingraham, 1981).

Bottom temperatures are affected by vertical circulation and diffusion and the mean horizontal flow. Mean temperatures (Figures 20a and 20b) show that during winter months the thermocline is nonexistent and the isothermal water column approaches the bottom to a depth of about 75 m.

With all temperature measurements there are some interannual variations. Anomalies (Figures 22 and 23) averaged over DYNUMES Areas 1 to 3 (Figure 21), can be over 6°C during any one year, especially in shallow water (less than 30 m).

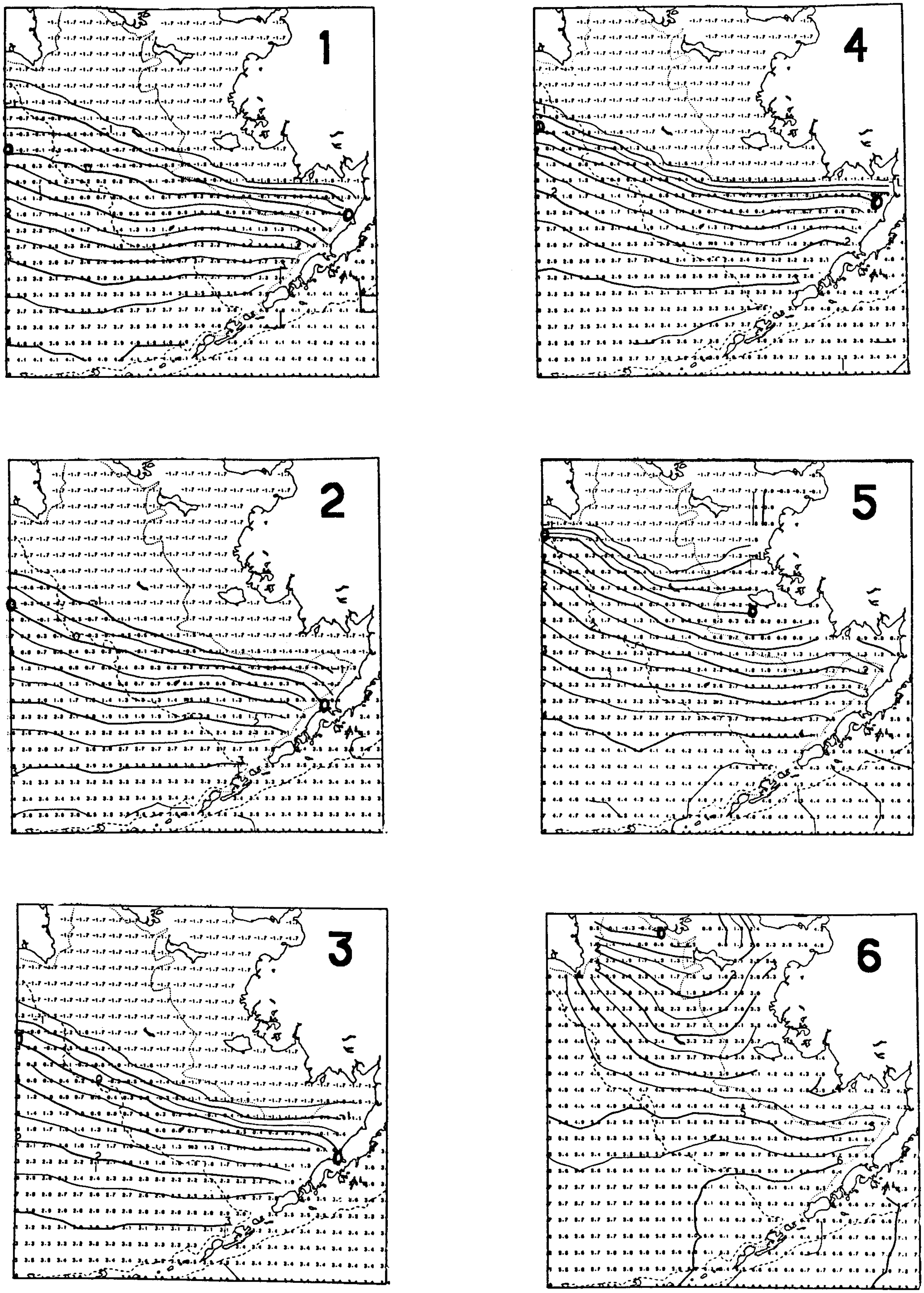


Figure 19a.--Average surface temperatures (°C) months 1 to 6.

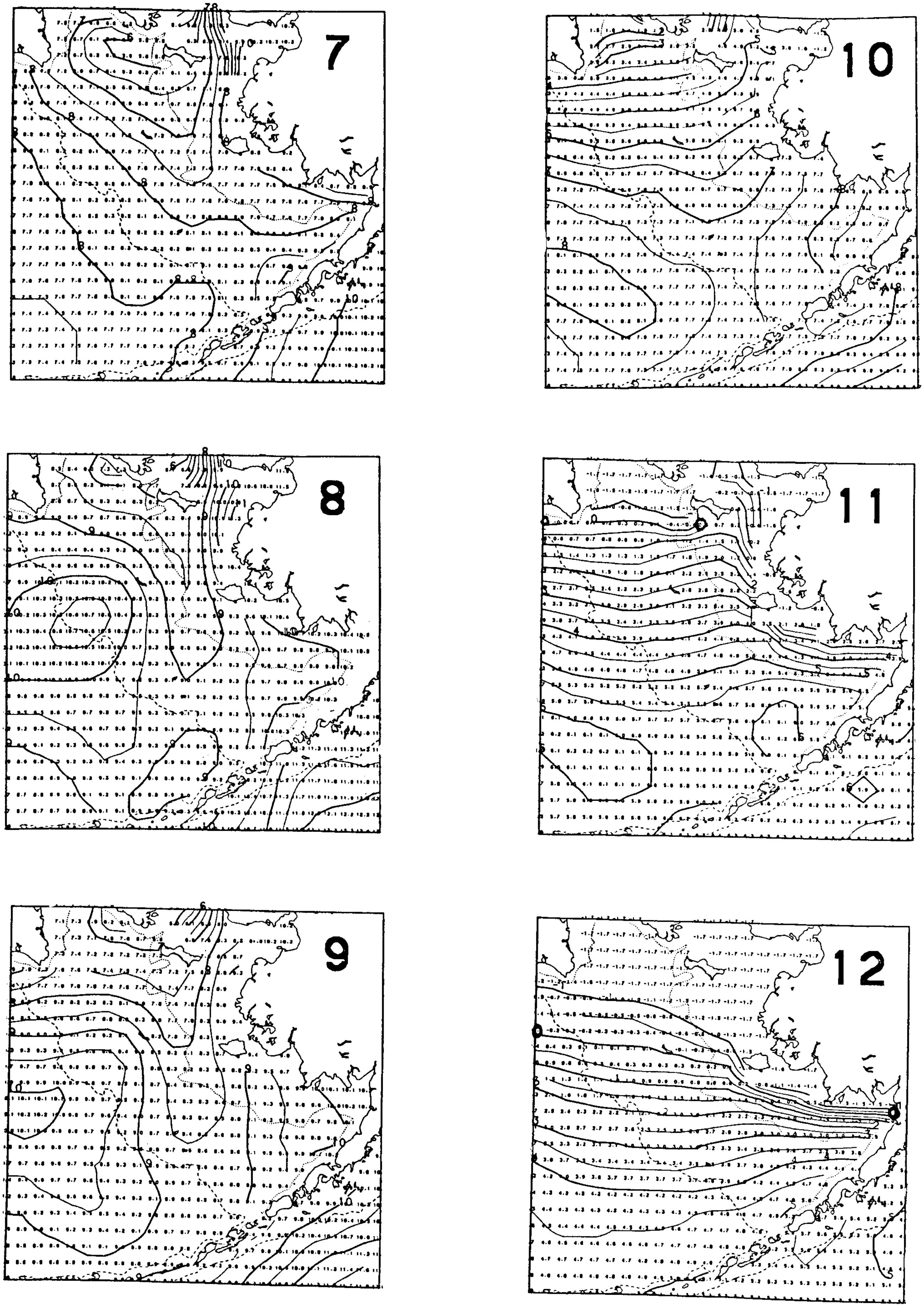


Figure 19b.--Average surface temperatures (°C) months 7 to 12.

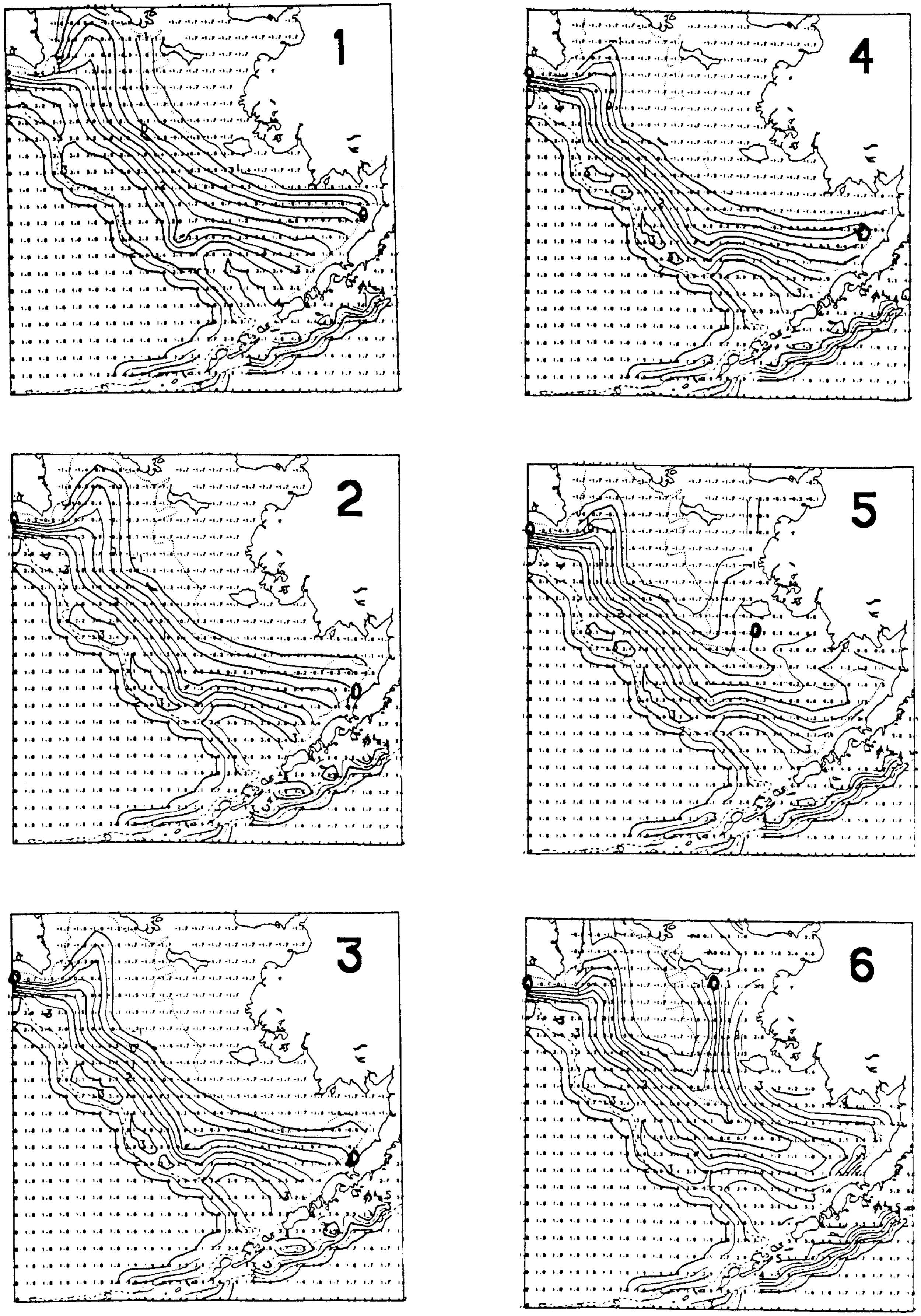


Figure 20a.--Average bottom temperatures (°C) months 1 to 6.

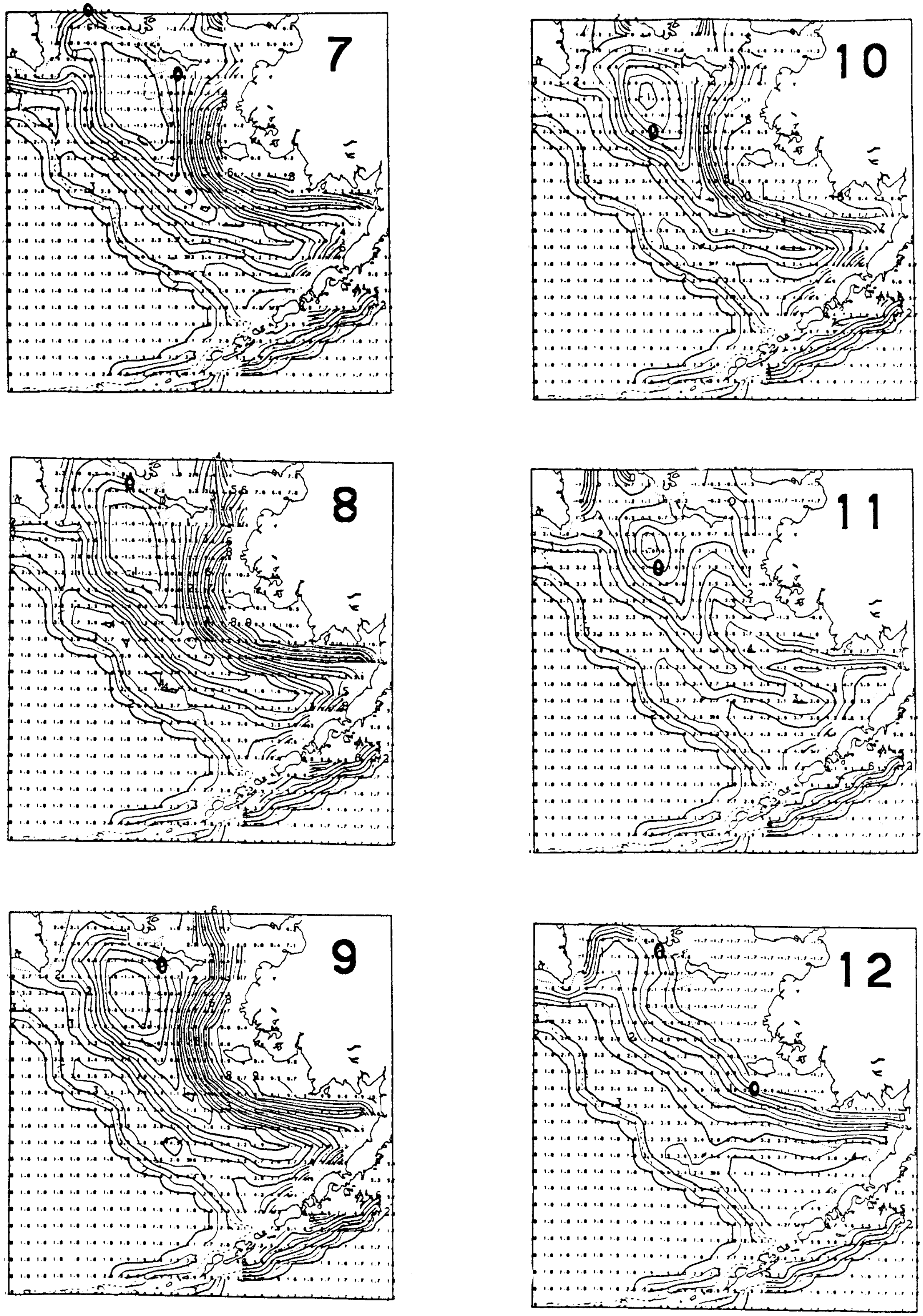


Figure 20b.--Average bottom temperatures (°C) months 7 to 12.

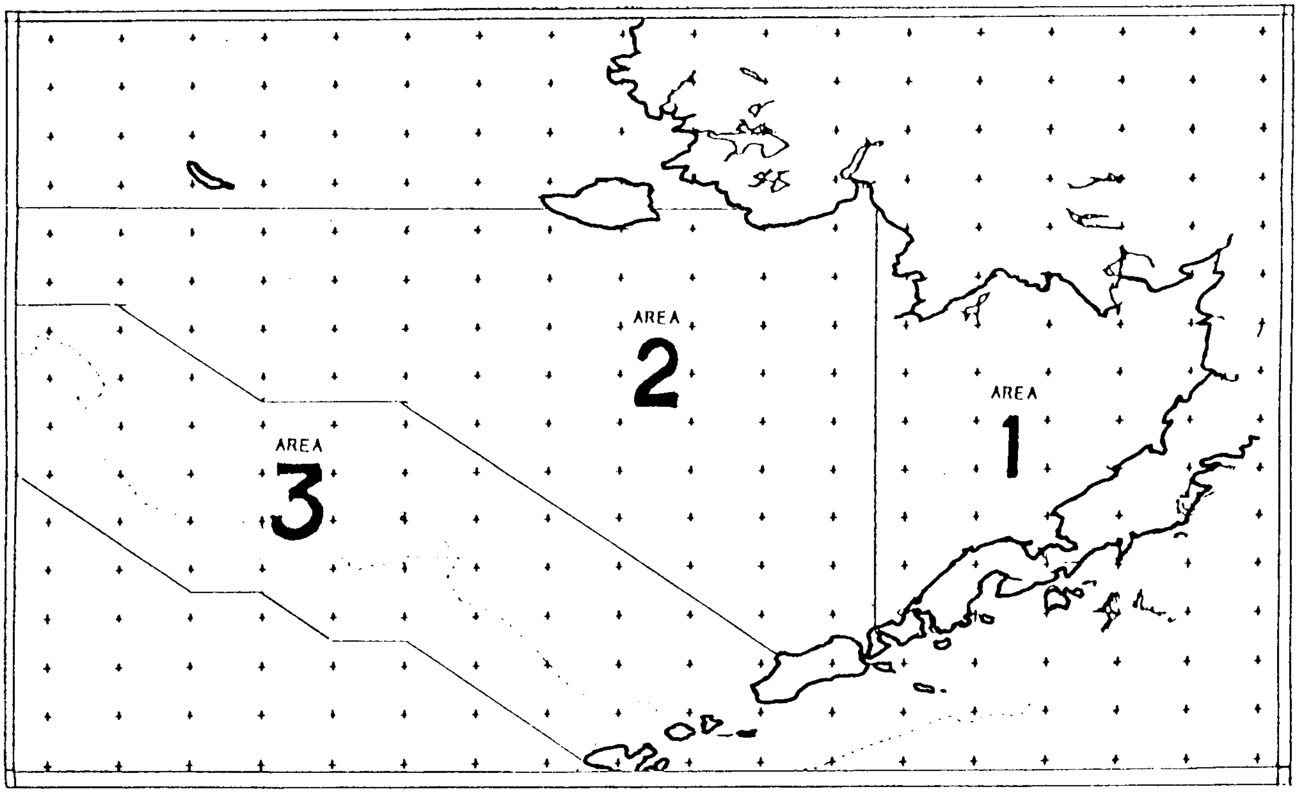


Figure 21.--DYNUMES areas 1 to 3.

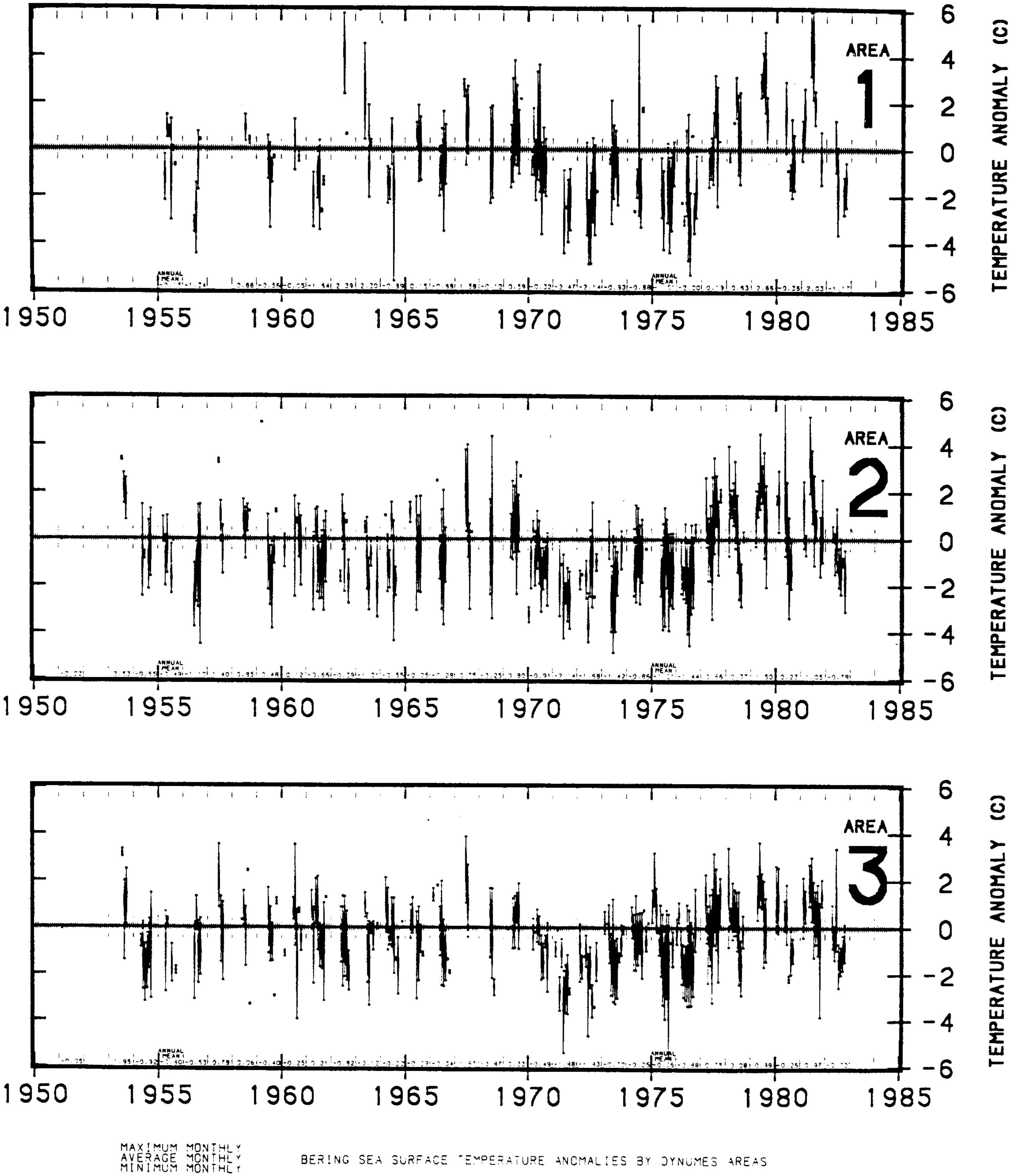


Figure 22.--Surface temperature anomalies - DYNUMES areas 1 to 3 (Pola Swan and Ingraham, 1984.

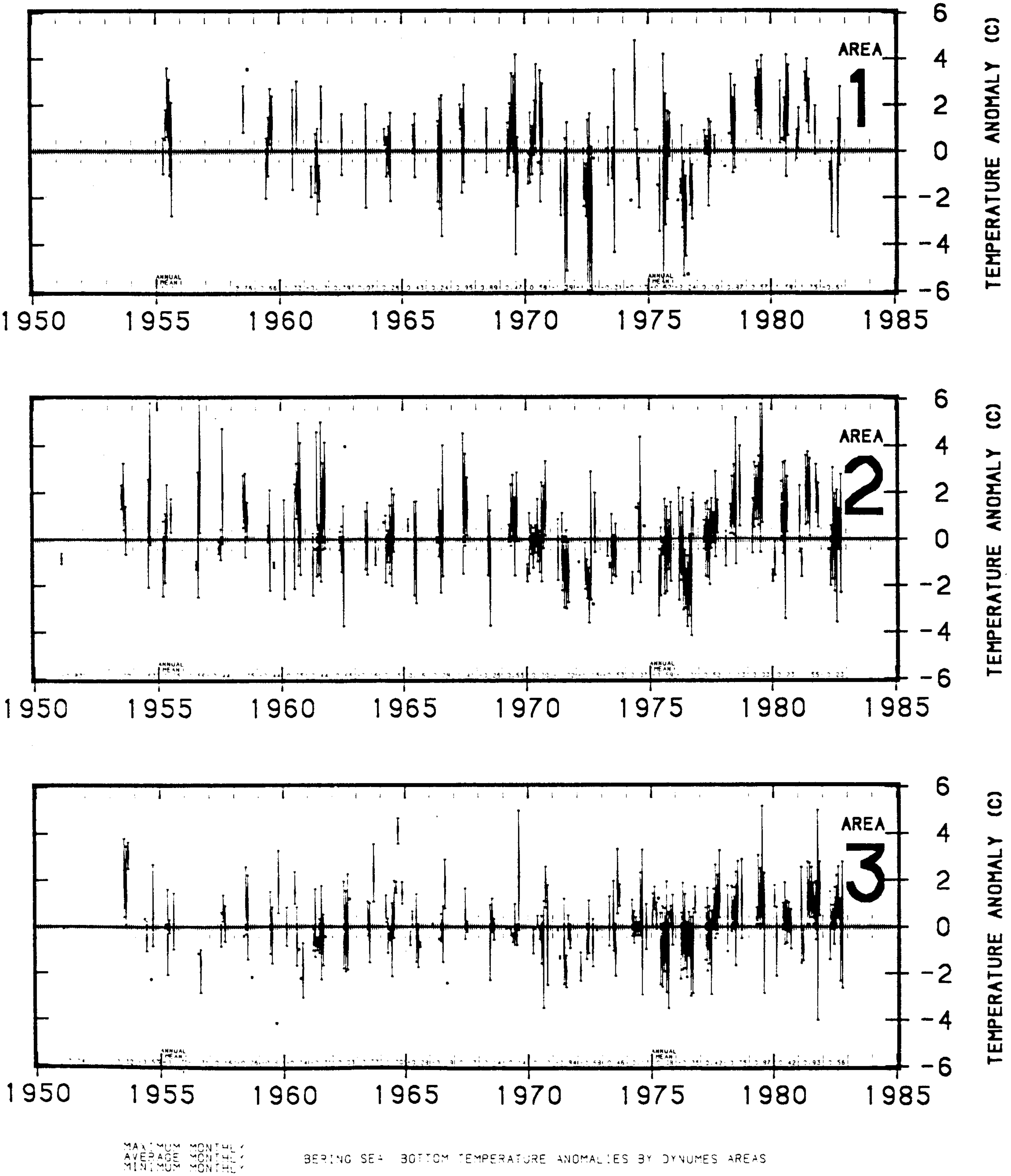


Figure 23.--Bottom temperature anomalies. DYNUMES areas 1 to 3 (Pola Swan and Ingraham, 1984.

Ice

Ice in the Southeastern Bering Sea occurs when the surface water temperatures reach about -1.7°C. Because of the differing latitudes (colder toward north) of Cape Newenham, Port Heiden, and Port Moller, the formation of ice occurs at different times. Ice around Cape Newenham forms during November and dissipates during May. Port Heiden and Port Moller ice forms during January and melts during April (Brower et.al.) (Figure 24).

In anomolousy warm years, the ice around Ports Heiden and Moller are nonexistent because these ports are usually the southern extent of fast ice. In an extremely cold year (1976), fast ice closes both Ports and extends out along the Alaska Peninsula to half way between Port Moller and Cold Bay.

Sea ice in the Bering Sea is renewed and melts each year. Ice over mid-shelf in the northern part of the Bering Sea appears to be advected southward as winter progresses.

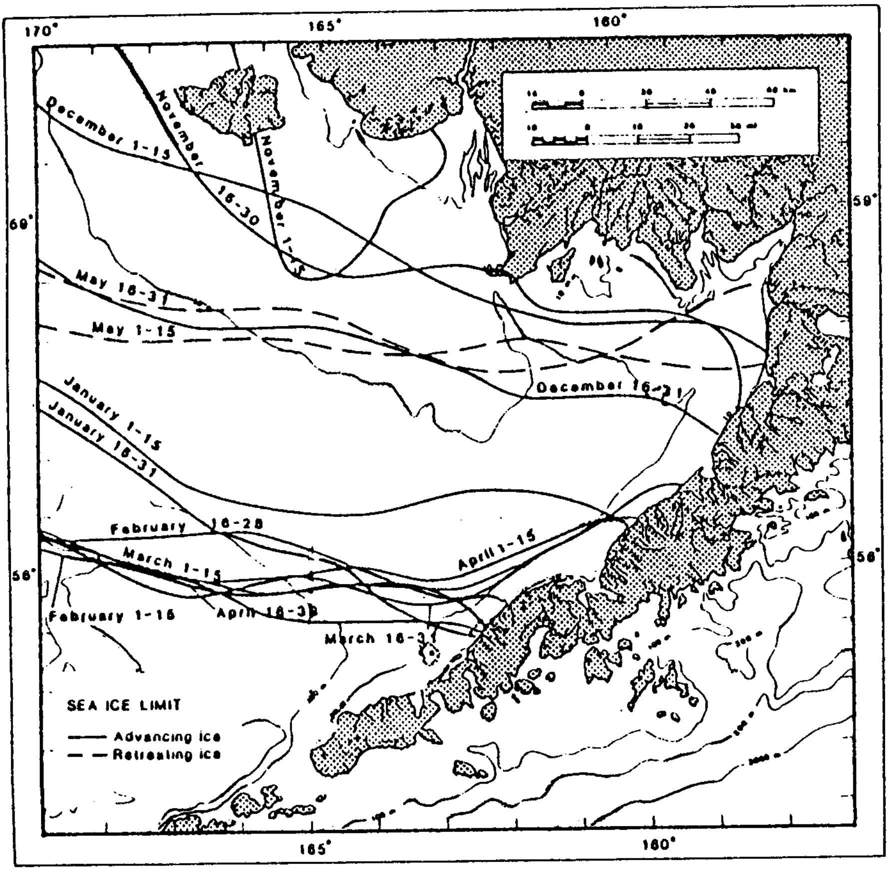


Figure 24.--Approximate ice edge for each month (Brower et al. 1977).

Salinities

Salinities of the Bering Sea are fairly consistent from season to season with the $33^{o}/oo$ isohaline following the 200 m isobath and the $32^{o}/oo$ isohaline following the 100 m isobath (Ingraham, 1981a) (Figures 25 and 26). There are, however, some local effects which change the salinity by $1^{o}/oo$ or $2^{o}/oo$. These effects are most prominent in the winter where the increased salinity is due to the lack of riverine outfall, decreased precipitation, and freezing. The most striking effect is salt exclusion during freezing.

During May the spring thaw begins, the riverine runoff increases, and coastal ice melts. These factors will lower the surface salinity during the spring and early summer. Another factor is that the precipitation dramatically increases during August, further diluting the salinity.

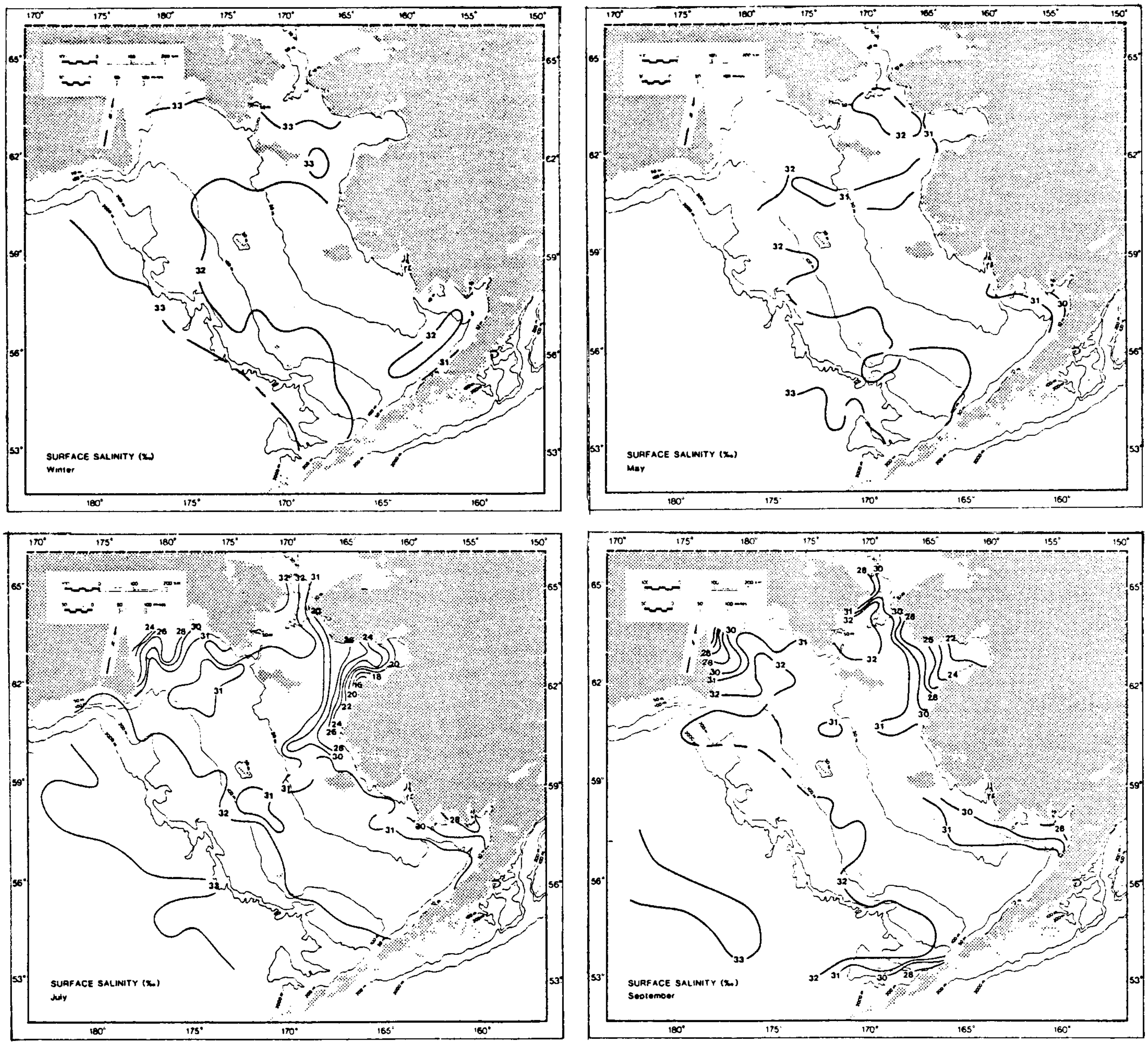


Figure 25.--Long-term mean sea surface salinity (°/oo) for January to March, May, July, and September (Ingraham, 1981).

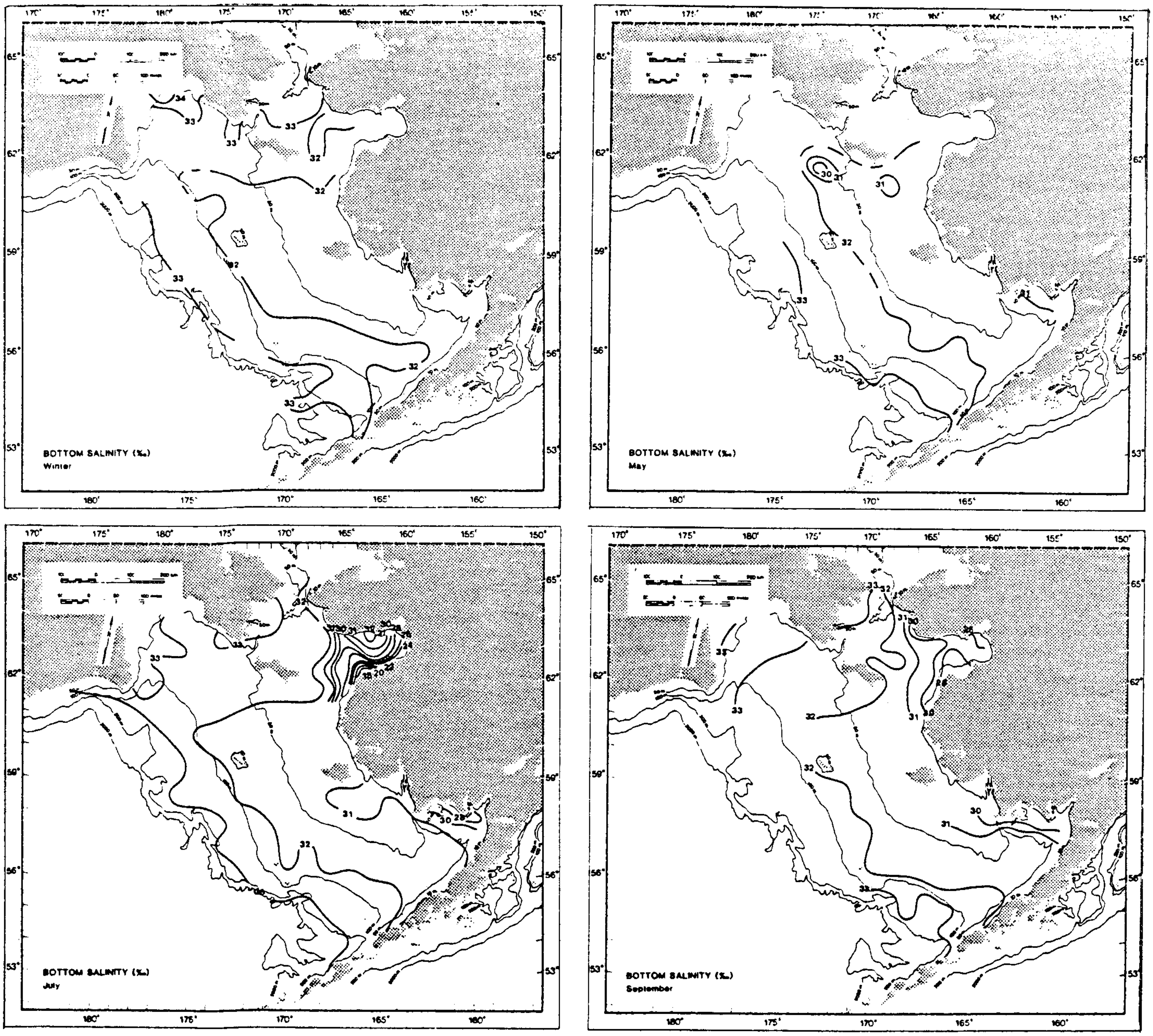


Figure 26.--Long-term mean bottom salinity (°/oo) for January to March, May, July, and September (Ingraham, 1981).

Vertical water structure and seasonal changes

In a synthesis of recent hydrographic data from the southeastern Bering Sea, Kinder and Schumacher (1981) characterized the shelf as having three distinct temperature, salinity, and water structure domains separated by fronts which lie near the 50 and 100 m depth contours (Figure 27). Inshore, the coastal domain (0 to 50 m depth range) has no significant vertical structure, for it is well mixed by consistently strong tidal currents except in local areas which are under the direct influence of river discharge. The middle shelf domain (50 to 100 m depth range) varies from being well mixed during winter when surface cooling and more frequent and stronger storms assist tidal mixing, to being two-layered in summer when buoyancy additions from melting ice and solar insolation exceed tidal and wind mixing. The inner front is a narrow (about 10 km) transition zone at the balance between tidal mixing and buoyant energy input (Schumacher, et al., 1979). The outer domain (100 to 170 m depth range) is three-layered throughout the year, a well-mixed upper layer, a well-mixed higher temperature and higher salinity lower layer, and a stratified interior layer between about 40 and 70 m depth containing a considerable amount of finestructure from interleaving of water masses on vertical scales from 1 to 25 m (Coachman and Charnell, 1977, 1979). The middle front, between the middle and the outer domain, is broader (about 50 km) than the inner front (about 10 km) and it occurs coincidentally with the steepening (five fold) of the bathymetry around the 100 m depth contour. Present data do not show great changes in the location of these fronts on diurnal, fortnightly, or interannual time scales.

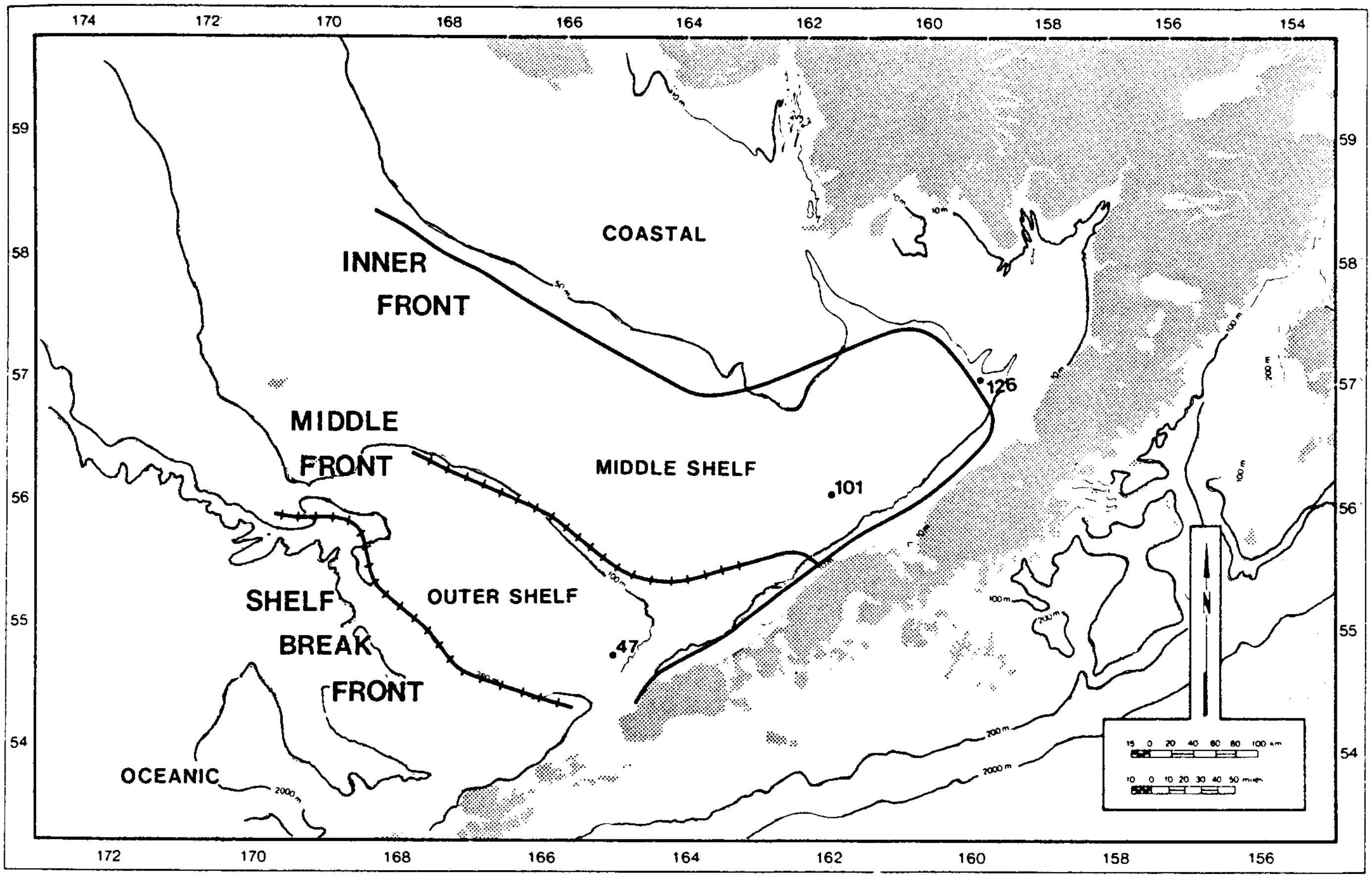


Figure 27.--Locations of the domains (coastal, middle, and oceanic) and fronts (inner, middle, and shelf break) over the eastern Bering Sea shelf (Kinder and Schumacher, 1981).

Nutrient salts

The distributions and cycling of chemical nutrients (phosphate-P, nitrate-N, silicic acid-Si, nitrite-N, and ammonium-N) provide fundamental insight into the nature of biological productivity on the southeastern Bering Sea shelf which is supplied at depth by a nearly unlimited source of nutrients from the Bering Sea proper. In the annual cycle, the major feature is the sharp increase with depth which forms at the seasonal thermocline as nutrients are extracted from the euphotic layer (upper 10 to 50 m) by phytoplankton during spring and summer; this is followed by a gradual replenishment of nutrients during autumn and winter due to vertical mixing and biological-chemical regeneration. Hattori and Goering (1981) reviewed available data and showed that the horizontal features of the nutrient distributions conformed closely to the hydrographic domains (coastal, mid-shelf, outer-shelf, and oceanic) discussed earlier. By comparing temperature, phosphate, nitrate, and silicic acid versus salinity diagrams (Figure 28) they found the four distinct water masses illustrated as separate envelopes of data in the temperature versus salinity diagram and also four corresponding envelopes of data in each of the three nutrient-salinity diagrams. Because the data covered several years, they concluded that the zonation of nutrients is a general feature of the southeastern Bering Sea in summer. Thus, two general trends in the distribution of nutrient concentrations are apparent: (1) a gradual offshore increase which changes in steps at fronts between hydrographic domains, and (2) an increase with depth in one large step in the seasonal thermocline which exists seaward of the constantly well stirred coastal domain. A notable exception is in the ammonium distribution which drops off to a very low (<0.5 g at N/1) concentration in the outer-shelf and oceanic domains below 75 m, leaving a distinct mid-shelf maximum near bottom and a weaker maximum in the outer-shelf thermocline (30-50 m) (Figure 29). The nitrite distribution is similar to that of ammonium.

Ratios of concentrations between major nutrients indicate that the eastern Bering Sea is similar to other ocean areas in terms of global averages with atoms of nitrate-N/phosphate-P about 15:1 and atoms of silicic acid-Si/phosphate-P about 33:1, but deviations from these established ratios locally indicate significant variation in nutrient cycling which is apparently caused by a major separation in food webs at the middle front. The lowest ratios in nitrate-N/phosphate-P (about 5:1) are found in the mid-shelf bottom water, and intermediately lower ratios (about 10:1) occur in the outer-shelf euphotic layer. Both coincide with maxima in the vertical ammonium distributions, indicating areas where phosphate is regenerated faster than nitrate. The lowest ratio in silicic acid-Si/phosphate-P also occurs in the mid-shelf bottom water, again phosphate apparently regenerating more rapidly than silicic acid. In contrast, Si/P ratios higher than the norm do occur randomly over the entire area but only in the euphotic layer when diatoms which require silicic acid are low in numbers, then silicic acid is underutilized relative to phosphate. Reasons for these nutrient distributions, although not varified extensively by large field data sets, become clearer when the outer-shelf and inner-shelf food webs are considered.

Physical, hydrographic processes which restrict the advective exchange of water and its contents across the middle front appear to be the chief cause for the horizontal structuring of nutrient distributions and food

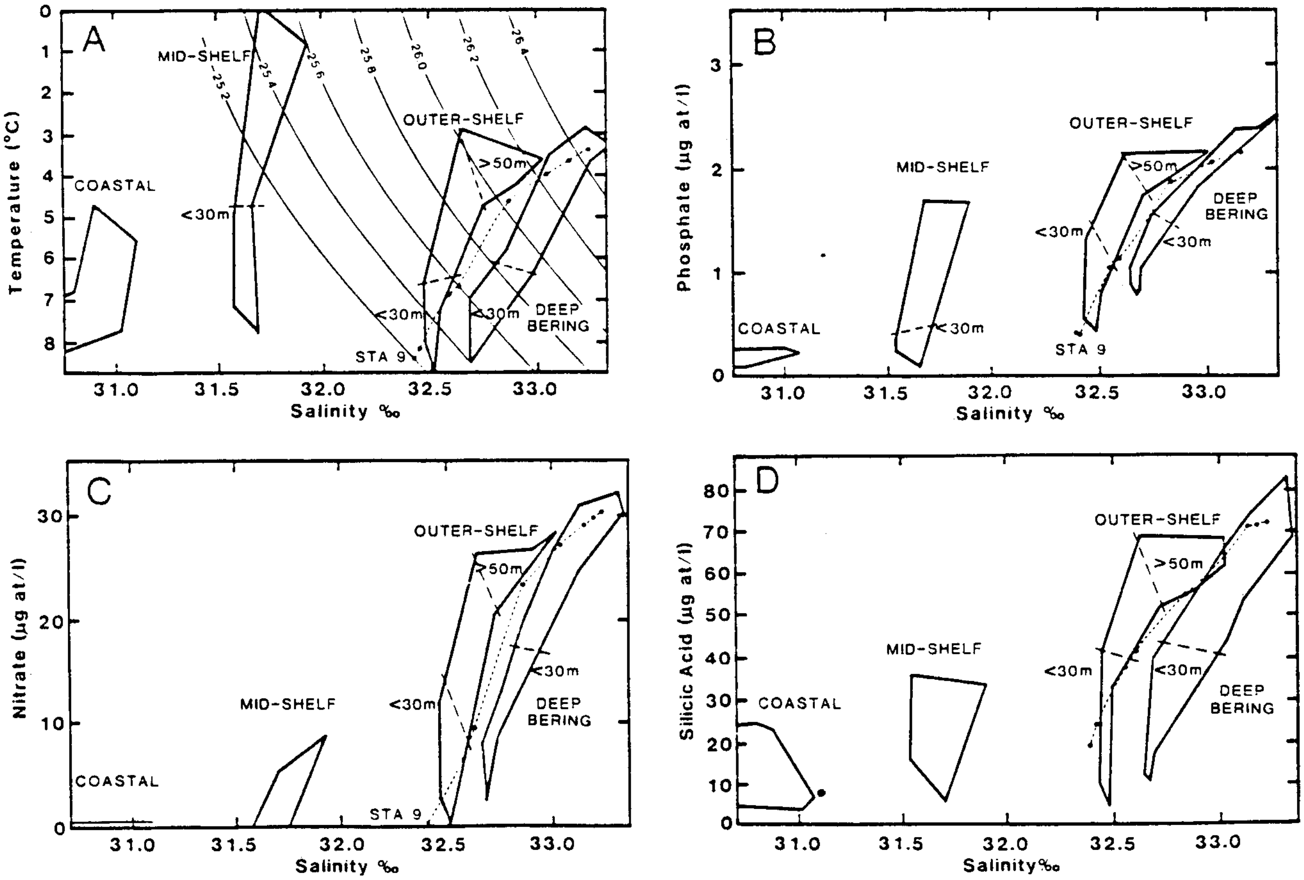


Figure 28.--Temperature-salinity and nutrient-salinity envelops in the eastern Bering Sea in July 1978 (Hattori and Goering, 1981).

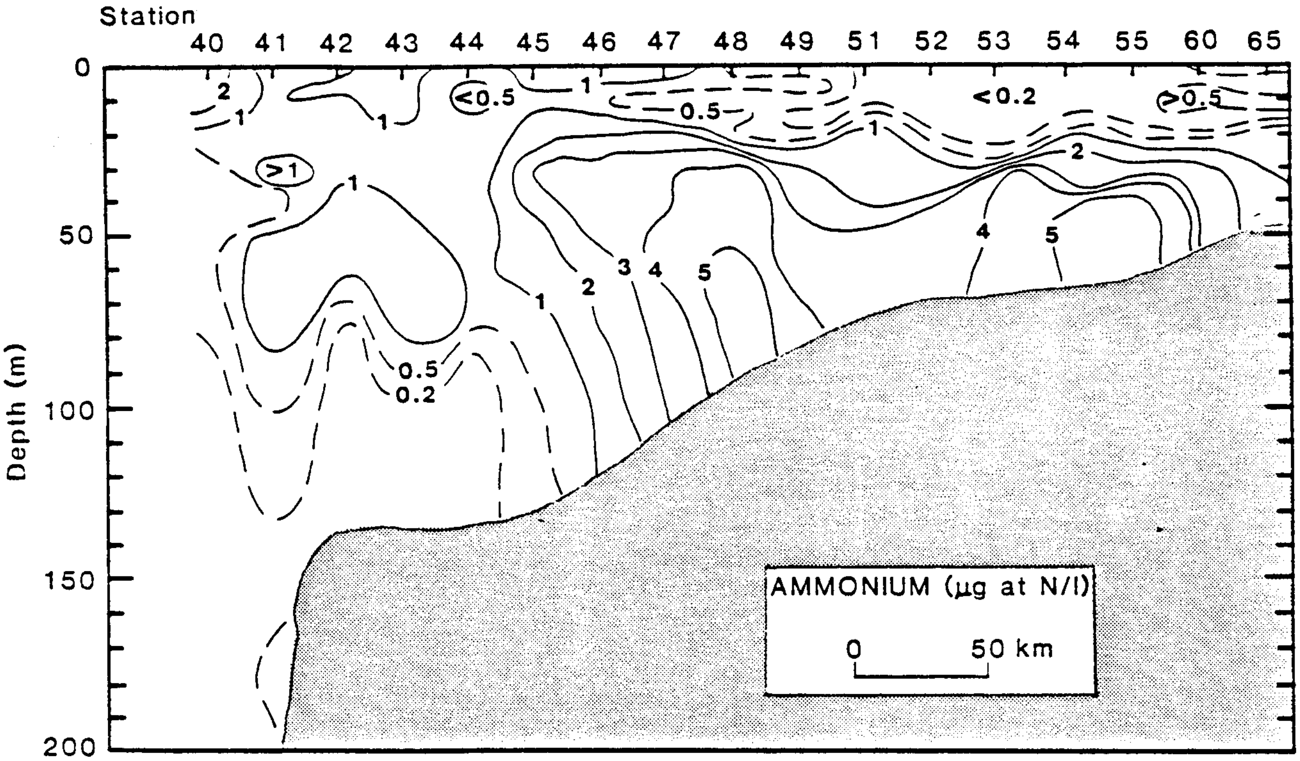


Figure 29.--Ammonium (mg at N/1) cross section in outer Bristol Bay south of Pribolif Islands during 29-31 May 1979 (Hattori and Goering, 1981).

webs on this shelf. The phytoplankton consist mainly of diatoms and microflagellates. The flagellates being more numerous during winter, give way to a major diatom bloom in April which sets the diatom dominance for the rest of the summer. The herbivores which graze on these phytoplankton during spring and summer are segregated by the minimal exchange at the middle front. On the outer-shelf the large standing stock of large sized calanoid copepods and euphausiids effectively graze even the large summer diatoms. The results are: 1) a small standing stock of phytoplankton flagellates and dinoflagellates, 2) an extended supply of nutrients to the euphotic layer by regeneration, and 3) a subsurface maximum concentration of ammonium most likely from the excretion of zooplankton. Thus, the outer-shelf has a pelagic food web. In the mid-shelf domain contrasting conditions occur because the zooplankton are small and unable to effectively graze the large diatoms. This results in: 1) a small standing stock of zooplankton, 2) a flourishing large standing stock of large diatoms with large nutrient demands, 3) a continuous rain of unconsumed diatoms to the sea bed, and 4) a benthic nutrient regeneration cycle with large concentrations of near-bottom chlorophyll-a and ammonium. Thus, the mid-shelf has a benthic food web.

Dissolved hydrocarbons

The distributions of dissolved low molecular weight hydrocarbons in Bristol Bay were summarized by Cline (1981) who described their usefulness as a diagonstic indicator of petroleum hydrocarbons. Although the alkanes (methane, ethane, propane and butane) which are abundant constituents of crude oil and natural gas were present in Bristol Bay waters, their low concentrations, relatively high methane/ethane plus propane ratio of 30 to 500, and low ethane/ethene ratio of less than 1.0 all suggest a biological source. Cline, therefore, concluded that Bristol Bay is "pristine" and discussed his base line data on the naturally occurring biogenic hydrocarbons.

Methane is the dominant low molecular weight hydrocarbon appearing in concentrations of about two orders of magnitude greater than ethane, ethene, propane, and propene (Table 3). Despite the July maximum compared to September-October data, the major features of the methane distribution generally conform to the hydrographic domains with highest concentrations occuring (from lagoons) in the coastal domain, low concentrations near saturation in the mid-shelf domain, and intermediate values in the outer-shelf domain (Figure 30a). Higher values up to 12 fold of saturation near bottom in the outer-shelf domain (Figure 30b) suggest an additional source in the organic-rich sediments of the outer-shelf.

In contrast to methane, concentrations of the higher carbon number hydrocarbons (ethane, ethene, propane, and propene) were governed more by seasonal biological processes than frontal dynamics. For example, the surface distribution of ethene in summer (Figure 31) shows highest values in eastern Bristol Bay, then values gradually decrease westward with no well defined source and little, if any, influence by the 50 m front. Higher values at

CRUISE	DOMAIN		METHANE		ETHANE		ETHENE[1]		PROPANE		PROPENE[1]	
			Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range
Sept.-Oct., 1975	Coastal (<50 m)	a	64	45-94	--	--	0.9	0.3-17	--	--	0.5	0.2-11
		b	59	45-98	--	--	1.0	0.7-1.8	--	--	0.4	0.1-0.6
	Middle Shelf (50-100 m)	a	60	42-83	--	--	0.8	0.3-1.6	--	--	0.1	0.1-1.4
		b	99	65-163	--	--	1.7	1.2-2.7	--	--	0.6	0.3-1.3
	Outer Shelf (100-200 m)	a	76	40-200	--	--	0.5	0.2-0.8	--	--	0.3	0.3-0.4
		b	380	100-615	--	--	1.1	0.7-1.6	--	--	0.3	0.2-0.4
June-July, 1976	Coastal (<50 m)	a	112	74-153	0.9	0.6-1.5	3.8	3.0-4.7	0.4	0.3-0.6	1.4	1.0-2.5
		b	114	73-153	1.0	0.5-2.5	3.4	2.3-4.4	0.4	0.2-0.6	1.2	0.7-1.6
	Middle Shelf (50-100 m)	a	85	52-134	0.6	0.3-1.5	2.9	1.9-4.7	0.3	0.2-0.6	1.1	0.6-1.7
		b	115	62-165	1.3	0.5-2.5	2.2	1.1-4.0	0.5	0.3-0.6	0.5	0.2-1.0
	Outer Shelf (100-200 m)	a	140	53-276	1.1	0.4-2.1	2.3	1.8-2.8	0.4	0.2-0.7	0.7	0.5-1.1
		b	269	164-440	0.9	0.6-1.1	1.2	0.8-1.8	0.3	0.2-0.4	0.3	0.1-0.9

[1] Due to analytical difficulties encountered during the Sept.-Oct. 1975 cruise, concentrations of ethene and propene include ethane and propane respectively.

Table 3.--Average (a) surface and (b) near-bottom concentrations (nl/l, STP) of methane, ethane, ethene, propane, and propene for various water depth intervals (Cline, 1981).

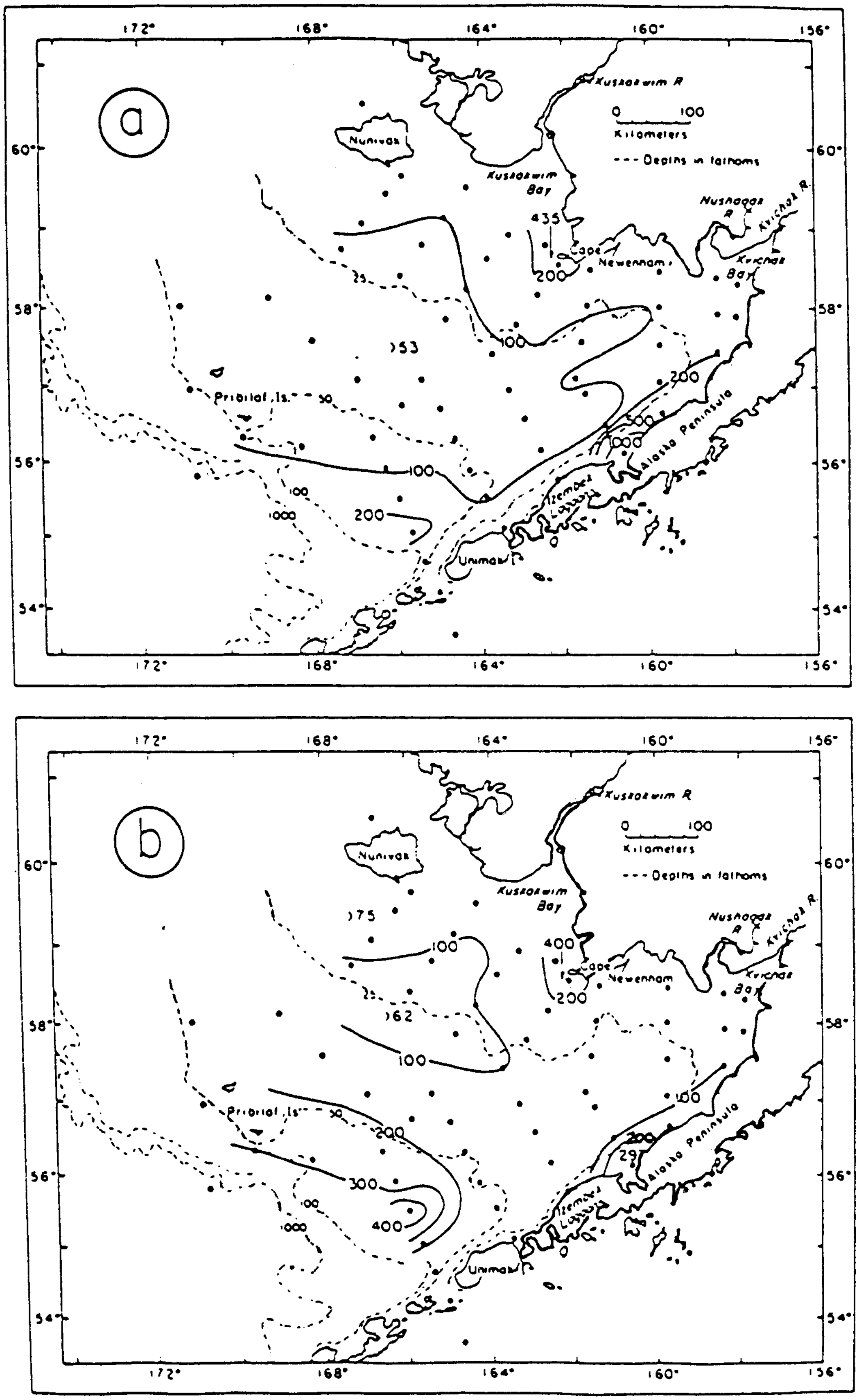


Figure 30.--(a) Surface and (b) near-bottom distributions of dissolved methane during July 1976 (Cline, 1981).

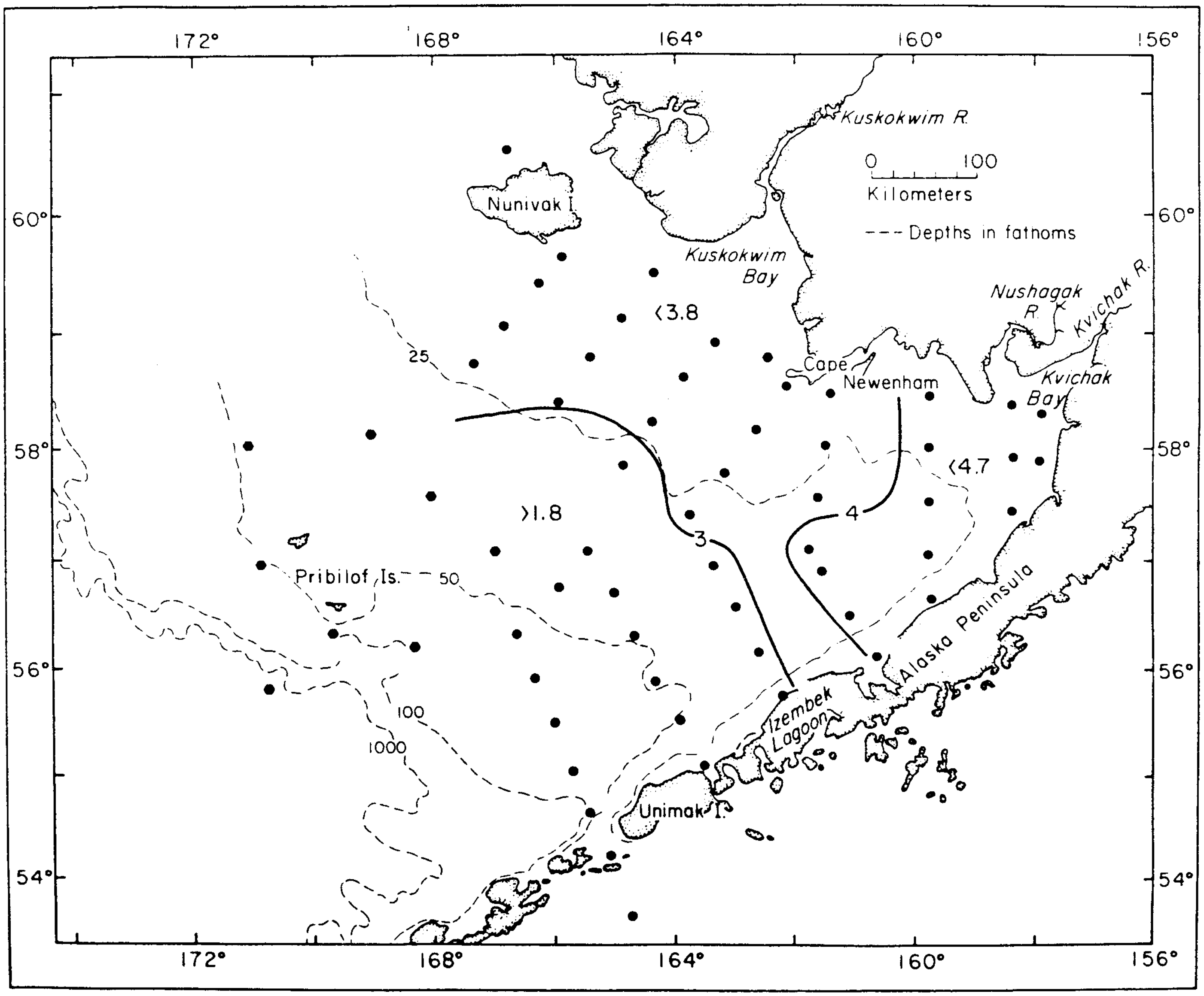


Figure 31.--Surface distribution of dissolved ethene (nl/l, STP) in July 1976 (Cline, 1981).

depth in the fall, higher values at the surface in summer, and the nearly doubling of all values in summer may be attributed to seasonally increased biological or induced photo-chemical production in the surface layers. Ethene is consistently related to ethane at a 3 to 1 ratio. Similar trends are shown in the propene and propane distributions and ratios; but because of their lower concentrations, seasonal and spatial variations are more obscure. Butane concentrations were always below the detection threshold of 0.05 nl/l.

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OIL ON THE BOTTOM OF THE SEA
A SIMULATION STUDY OF OIL SEDIMENTATION AND ITS EFFECTS ON THE BRISTOL BAY ECOSYSTEM

by

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ABSTRACT

Oil budget studies of some recent oil spills show that a considerable amount of oil sedimentizes to the bottom, where it has some immediate as well as long term effects on the benthos and demersal fish.

The relatively meager available quantitative data on the sedimentation of oil and of factors affecting it, are summarized. Based on this knowledge, a numerical model was designed to estimate the quantity and rate of oil sedimentation, including the decay (weathering) of the oil. The computer programme is given in FORTRAN.

The possible effects of oil on demersal fish and on benthic ecosystem, as deduced from laboratory experiments and from a few field observations, are evaluated and summarized.

1. INTRODUCTION

1.1 Purpose of this study

The sinking of oil and its pollution of sea bottom sediments has been demonstrated in a number of oil transportation accidents (FLORIDA; Blumer et al., 1971: ARROW; Keizer et al., 1978: AMOCO CADIZ; D'Ozouville et al., 1979: ARGO MERCHANT; Hoffman and Quinn, 1978: TSESIS; Linden et al., 1979: SEFIR; Linden et al., 1983). Estimates of sedimented oil were also obtained for the well blowouts at IXTOC I (Jernelöv and Linden, 1981) and Platform Bravo (Ekofisk) in the North Sea (Mackie et al., 1978). Studies done in large, controlled mesocosms have also demonstrated the sedimentation of oil in sea water (Elmgren et al., 1980; Grassle et al., 1980; Elmgren and Frithsen, 1982). Whereas oil polluted pelagic environments have been observed to recover relatively rapidly (e.g., about a month after the TSESIS spill and within 4 months after the AMOCO CADIZ spill), oil residues on or in the bottom have been shown to persist for many years (Linden et al., 1979; Laubier, 1980). Oil incorporated into bottom sediments is now generally recognized as presenting the single greatest and long term threat to the environment from oil spill accidents (Linden et al., 1979; D'Ozouville et al., 1979; Conan, 1982; Elmgren et al., 1983; Gundlach et al., 1983; Linden et al., 1983).

In spite of the long term threat to the epipelagic and benthic biota from oil residues on and in the sediment, there is comparatively little knowledge regarding the sedimentation of oil. Direct measurements of oil sedimentation from past spills are very few and even when done, they were incomplete in sampling the temporal and areal dimensions of the spills. Therefore, estimates obtained from past spills which were extrapolations of data from sediment traps or bottom grab samples, generally underestimated the amount of oil sedimentation (e.g., <u>TSESIS</u> and <u>AMOCO CADIZ</u> spills).

The pathways and processes of oil sedimentation have been discussed by many authors and summarized by Clark and McLeod (1977). We have, therefore, very positive evidence for the sedimentation of spilled oil and some approximations of the quantities sedimented. We also have some knowledge regarding the pathways and processes of oil sedimentation. We are not aware, however, of any generalized models for quantifying the rate or amount of oil sedimentation from surface slicks.

The purposes of our study are to: 1) develop models for quantifying the amount and rate of oil sedimentation; 2) simulate the fate of oil on the bottom using available information; 3) summarize and analyze the effects of oil on and in the bottom on demersal fish and benthic ecosystems. The results of these studies will be incorporated together with other analyses into a report which will assess the possible effects of two hypothetical oil spill scenarios at three locations on several commercially valuable fishery resources of southeastern Bering Sea.

The three hypothetical spill sites are seaward of Port Moller, Port Heiden and Cape Newenham in Bristol Bay. One spill scenario is an instantaneous spill of 200,000 bbls of automotive diesel fuel and the other a well blowout of 300,000 bbl of Prudhoe Bay crude oil discharged at a rate of 20,000 bbl/day for 15 days. The volume of oil spilled in the hypothetical tanker accident is exceeded only by the spill of the AMOCO CADIZ (about 1.6 million bbl) and the IXTOC I well blowout (about 3.5 million bbl). Both of these accidents involved crude petroleum. The hypothetical spill of automotive diesel fuel exceeds by far any past spills of middle or heavy distillate petroleum fuels. The total volume (300,000 bbl) of the well blowout scenario is also considerably less than that discharged from the IXTOC I well blowout or the spill from the AMOCO CADIZ. The volume is comparable to the Ekofisk Bravo blowout (146,000 - 219,000 bbl, Mackie et al., 1978). The discharge rate in the hypothetical scenario (20,000 bbl/day) is less than the maximum daily loss from IXTOC I (about 32,000 bbl/day) but somewhat comparable to the estimated rate of discharge in the Ekofisk blowout (19,500 - 29,200 bbl/day).

Initial calculations indicate that the oil concentrations in the water column (both soluble and emulsified) above the thermocline from a blowout of 300,000 bbl of Prudhoe Bay crude oil will be less than 1 ppm in most areas. The rather low concentrations seen in the hypothetical spill are not unlike those estimated for past oil spills and blowouts at sea.

These low concentrations may result in tainting but should inflict little if any mortality to adult fish nor affect the long term productivity of fish stocks. Certain invertebrates and ichthyoplankton, however, may be drastically affected. As previously mentioned, however, the damage to these pelagic communities can be expected to be acute but relatively short term. Even with these low concentrations of oil in the water column, empirical observations have demonstrated that concentrations in deeper soft bottoms can be considerably higher and of much longer duration. The assessment of the effects of possible oil spills in eastern Bering Sea must, therefore, focus upon the extent and duration of oil on the bottom and its short and long term consequences to the abundant and valuable fish and shellfish resources.

1.2 Existing knowledge on sedimentation of oil and its effects.

An oil slick is dynamic, changing not only in physical dimensions but also in chemical composition primarily due to the loss of certain components through evaporation into the atmosphere and dissolution into the sea. The rate of processes is dependent upon such local environmental factors as air and sea temperature, wind strength, surface agitation and currents as well as physical (e.g., viscosity) and chemical characteristics (e.g., hydrocarbon composition) of the oil. The lower molecular weight components will immediately begin to vaporize or leach into the water. Virtually all hydrocarbons C_{15} and shorter will volatilize from the sea surface within 10 days, many of the lighter, volatile materials disappearing within hours. Most components in the C_{15} to C_{25} range and all hydrocarbons longer than C_{25} will be retained in the slick. Evaporation alone will remove about 30 to 50% of the hydrocarbons from a typical crude petroleum slick. About 75% of the hydrocarbons from No. 2 fuel (automotive diesel) and 100% of the hydrocarbons from kerosene or gasoline will vaporize (Clark and McLeod, 1977).

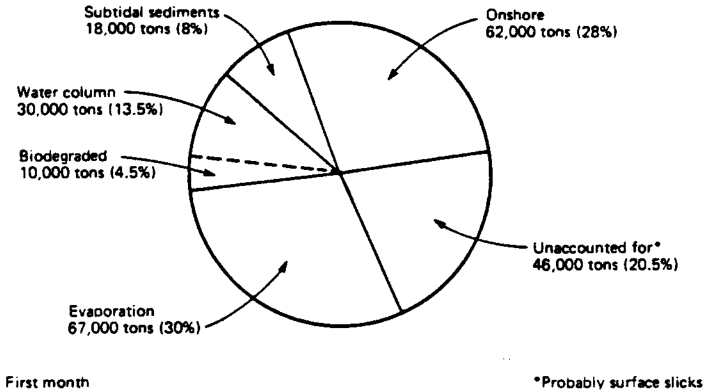
Surface oil enters the sea as dissolved fractions, oil droplets or emulsions (oil-in-water or water-in-oil), the dominant processes being the latter two. In order for the petroleum in the water to sink, processes must intervene to disrupt its positive or neutral buoyancy. The specific gravity of oil may be increased by evaporation and dissolution of low molecular weight hydrocarbons, degradation and oxidation of oil components, formation and agglomeration of dispersed particles and the uptake of sea water during emulsification (Clark and McLeod, 1977). Fresh and weathered oil may be vertically transported through the water column, however, the particles cannot remain near the bottom or be incorporated into bottom sediments unless they adhere to suspended particulate matter which is heavier than sea water. Pathways by which oil is sedimented include the adsorption of oil droplets on suspended mineral matter such as clay, incorporation of oil droplets in the fecal pellets of zooplankton and the oiling of dead siliceous phytoplankton or zooplankton. The relative importance of these pathways of oil sedimentation will depend to some extent upon the area, timing and environmental circumstances of a spill. In most nearshore and estuarine spills in subarctic environment, particularly during the late fall through early spring months when seas are most turbulent, adherence of oil droplets to particulate, mineral matter would seem the most substantial process of oil sedimentation. In our study we will assume that sedimentation is entirely attributable to adsorption of oil onto particulate mineral matter.

It should be noted that Prudhoe Bay crude oil is relatively viscous. Rice et al. (1976) observed that under identical conditions of mixing, the yield of water soluble fractions from Prudhoe Bay crude were about half the concentrations from Cook Inlet crude.

There is no well-substantiated data available on the relative quantities of oil reaching the bottom; the few reported data are indirect estimates. Elmgren (pers. comm.) estimates that 10 to 30% of the spilled TSESIS oil reached the bottom. Of the AMOCO CADIZ spill, 8% is estimated to have gone into subtidal sediment, 28% went on shore, and 20.5% is unaccounted for (Grundlach, et al., 1983) (Figure 1). If the oil had not reached the shore, it could be assumed that the greatest portion of the two last components might have ultimately sedimentized (i.e., 30 to 50% of total oil). Some direct quantitative data on sedimentation of the oil has been obtained in large experimental tanks (5 m. deep), where Elmgren and Frithsen, 1982, found that 40 to 50% of the oil added to the water in the tanks reached bottom (Figure 2). Boehm and Fiest (1980) concluded that only 1 to 3% of IXTOC I oil was to be found in offshore sediments, although near the well blowout high concentrations of oil in the sediment (100 ppm) were detected. Jernelöv and Linden (1981) estimated that 25% (120,000 mt) of the IXTOC I blowout sank to the bottom.

Elmgren et al., 1983, found that the oil from TSESIS spill sedimentized (sank) to the bottom relatively rapidly. There was at least 0.5 g oil per m^2, and in heavily oiled areas possibly considerably more.

If we assume that the oil was accumulating initially in a nepheloid layer near the bottom, say 15 cm thick (the thickness of this layer is variable indeed), the resulting concentration from 0.5 g/m^2 on the bottom would give an oil concentration in this relatively thin nepheloid layer of 3.3 ppm, which is about ten times higher concentration than normally found in the water in oil spill areas. This simple calculation thus demonstrates the importance of the consideration of oil on the bottom.



First month
Total spilled: 223,000 tons

*Probably surface slicks and tar balls

Figure 1.--Quantitative estimate of Amoco Cadiz oil dispersal components for the first month of the spill (Gundlach, et. al., 1983).

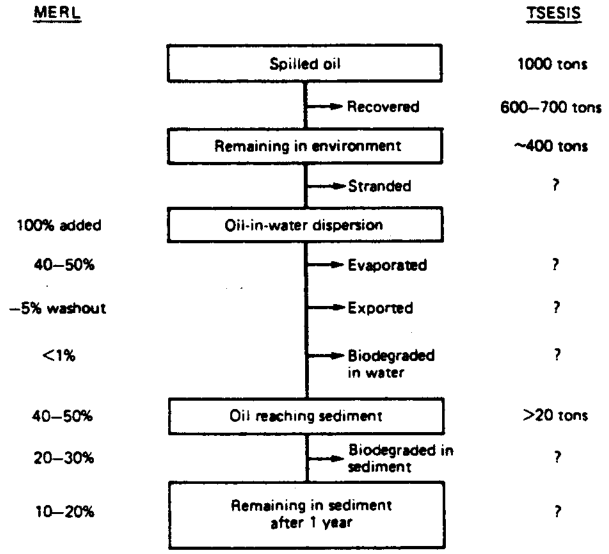


Figure 2.--Fate of oil (Elmgren and Frithsen, 1982).

Some aspects of the oil sedimentation process has been studied in the laboratory. Gearing et al., 1979, found that minerogen (e.g., silt and clay) particulate matter absorbed ca 15% of oil from the tank and carried it to the bottom. Low molecular weight aromatic compounds were not found in this sedimented oil.

The oil on the bottom accumulates first in a flocculent (nepheloid) layer, which floats immediately above the bottom and is difficult to sample. This flocculent layer has a tendency to accumulate in small deepenings in the bottom (Elmgren, pers. comm.) where near-bottom current is absent. The newly sedimented oil contains little toxic aromatic components (Elmgren and Frithsen, 1982). These components decay relatively quickly in the water and near and on the bottom. Therefore, the sedimented oil can be considered as weathered oil. Moore and Dwyer, 1974, also found that oil in water weathers by losing its toxic fraction very rapidly, mostly by evaporation. However, Falk-Petersen and Loenning (MS) have found that sea water extracts of photo-oxidized (weathered) oil is more toxic than extract of unweathered oil.

Oil will penetrate the sediments to 5 to 7 cm depth (and occasionally deeper, depending on the type of the sediment). This penetration of oil into sediment is assumed to be caused by "reworking" of the sediment by burrowing animals (infauna). Higher amounts of oil are found in fine-grained sediments (where the infauna biomass is also expected to be higher) and lower amounts in coarse-grained sediments (sand and gravel) (D'Ozouville et al., 1979).

The absorption and agglomeration of the oil in sediment is accompanied by further fractionation of the original oil mixture. Zürcher and Thüerer, 1978, found that 200 ppm of oil in dry clay is close to "saturation absorption" of this material. In addition to oil adhering to sediment, there is also oil in interstitial water. Vandermeulen and Gordon, 1976, found 10 mg oil per gram natural sediment. Hayes et al., 1979, also found that interstitial water gets heavily oiled, from where it can reenter the water above.

The longevity of oil in sediment is not known. According to Vandermeulen and Gordon, 1976, flow experiments (of interstitial water) indicate that stranded oil could remain in sediment in excess of 150 years (by which time it is fully buried).

In tank tests 10 to 20% of the total oil added to tanks (of which 40 to 50% sedimentized) remained in sediment after 1 year (Elmgren and Frithsen, 1982). In AMOCO CADIZ oil spill area, some oil remained in fine-grained sediments 3 years after spill (Grundlach et al., 1983). Oil degraded (weathered) slower in muddy sediments than in sandy sediments. Whether the more rapid degradation in sandy sediment is due to more intensive microbial action, is unknown at present.

Biodegradation of oil might be one of the main factors for "depuration" of oily bottoms. Biodegradation is known to increase with increased temperature (Gearing et al., 1979).

Laboratory research on the short term effects of weathered oil on benthos seems to be difficult to interpret (Kalko, Duke, and Flint, 1982). The best observations on the effect of oil on bottom on benthos originate from the studies of the TSESIS spill (e.g., Elmgren et al., 1983).

Among initial effects of the TSESIS spill were the disappearance of amphipods (especially *Pontoporeia affinis*) and polychaetes. Bivalves (e.g., *Macoma balthica*) contained high amounts of hydrocarbons (obs. these animals accumulate hydrocarbons from water while filtering food). Their biomass increased rapidly a year after the spill and continue at above prespill levels. Full recovery of the benthos community (in respect to species composition) had not yet occurred 5 years after the TSESIS spill; Pontoporeia have still not returned to prespill levels.

The response of benthos to AMOCO CADIZ oil spill was similar (Conan, 1982). Immediate mortalities of bivalves, periwinkles, limpets, peracarid crustaceans, and heart urchins were observed in heavily oiled shallow water. Populations of clams and nematodes in the meiofauna declined after the spill, and for several clam populations recruitment remained unstable. Benthic species with short life cycle tended to replace long-lived species.

The effects of oil on the bottom on the demersal fish species is difficult to observe in nature. In the TSESIS spill area some flounders (*Pleuronectes flesus*) showed 50 ppm hydrocarbons in liver and muscle one year after the spill (obs. flounders are feeding on *Macoma* sp.) (Linden et al., 1979). In the AMOCO CADIZ oil spill area, estuarine flatfishes and mullets had reduced growth, fecundity, and recruitment; and were affected by fin rot (Conan, 1982). An absence of young sole in shallow water a year after the spill was noticed (Grundlach et al., 1983). Changes in the availability of flatfish (sole) in shallow water were noted, however, no changes were noticed in fish populations in deep water. A taste panel detected tainting in haddock, plaice, gurnard, and lemon sole after the Ekofisk blowout, however, no oil derived hydrocarbons could be found in the muscles (Mackie, 1978). This may be confirmation that some of the major flavor components of oil are not hydrocarbons (Howgate et al., 1977) and are, therefore, not measured.

Any reduction in "worst case" spill on fish stocks is difficult to detect against the background of normal variability in the sea. It is, in general, agreed that commercial stocks in the open parts of the shelf are not at risk from oil (McIntyre, 1982).

The large experimental ecosystems (in tanks) offer some possibility to test the sensitivity of benthic organisms to weathered oil on the bottom (e.g., Elmgren and Frithsen, 1982; Grassle, Elmgren, and Grassle, 1981). So far long-term tests of the toxicity of oil on demersal fish have been more the exception than the rule. The toxicity tests on fish have mostly been done in small laboratory tanks and the duration of which were measured in hours and days rather than in weeks or months (see Chapter 4). The translation of these results to field conditions is often questionable.

2. SEDIMENTATION OF OIL AND FACTORS AFFECTING IT

2.1 Factors affecting sedimentation

The oil from a well blowout or from a tanker accident rises to the surface, where gravity and surface tension promote spreading on calm water while inertia and viscosity retard spreading. The transportation, dissolution, and weathering of the surface oil slick depends upon the characteristics of the oil, and such environmental factors as air and water temperature, wind velocity and direction, surface turbulence, and surface and subsurface currents. The oil which sedimentizes (sinks) to the bottom originates from this oil slick on the surface. (Note: Beached oil which has been shown to cause catastrophic mortalities to intertidal and subtidal fauna is not considered in this paper.) The sedimentizing oil must pass the water mass between the surface and the bottom. The processes of the solution and dispersion of oil from the surface slick into the water have been summarized by Clark and McLeod (1977). Payne, Kirstein, McNabb, Lambach, de Olivera, Jordan, and Hom (1982) had a more recent summary with procedures for quantifying the weathering of oil.

The amount of dissolved and emulsified oil in the water is about 10% (and slightly more) of the oil on the surface at any given time and location. One of the main factors "forcing" emulsified oil into the water is turbulence caused by waves (and currents). The latter are a function of wind (wind energy). The turbulence caused by wind-generated waves determines also the thickness of the near-surface turbulent mixed layer (depth of the thermocline). Obviously there are other factors besides wind waves contributing to space and time variable mixed layer depth and turbulent mixing, such as convective turnover, tidal currents, etc. (for a summary on mixed layer processes see Laevastu, 1976). In the 9 months plus duration of the IXTOC I blowout, a release of 475,000 metric tons of oil escaped, of which 120,000 mt (or 25%) was estimated to have sunk to the bottom (Jernelöv and Linden, 1981). (Some empirical data on the quantitative distribution of oil in the water from IXTOC I blowout is given by Boehm and Fiest, 1982.) Grundlach et al., 1983, found that 13.5% of AMOCO CADIZ oil got into the water, and this amount is considered to present a maximum, due to heavy wave action in the location and time of the AMOCO CADIZ accident.

Only very few crude oils have a specific gravity higher than sea water and can sink (e.g., Michel, 1984). In most cases the oil is lighter than water and rises to the surface, from which it must pass through water column and must be made heavier than water by various processes, in order to sedimentize to the bottom.

The minute oil droplets present in water as oil-in-water may be transported toward the bottom by entrainment in vertical currents. However, unless they become heavier than water, emulsion or droplets cannot remain near or become incorporated into bottom sediments. It has been observed that oil absorbs to minerogen suspension (clay) present in the water. The amount of oil which sediments can carry down is inversely proportional to grain size (Poirier and Thiele, 1941). The clay particles, which are heavier than water, can agglomerate and accelerate sedimentation. Thus, the sedimentation rate depends not only upon the quantity and characteristics of oil, but also on the amount and nature of suspended minerogen particles present. The coagulation of the particles is faster in salt water than in fresh water due to electrolytic action (Bassin and Ichiye, 1977). The collision of the clay particles (and/or oil particles containing minerogen particles) due to differential settling rates are the governing nonbiological processes in formation of natural aggregates (Hawley, 1982). These aggregates fall significantly faster than Stokes Law predicts (Hawley, op. cit.).

It has also been postulated (but not experimentally proven) that fecal pellets of zooplankton will facilitate the sedimentation of oil. This mechanism might work if these pellets were made heavier, e.g., by incorporation of diatom shells in fecal pellets.

The amount of suspended minerogen matter present is a function of depth, bottom type, turbulent mixing (e.g., by tidal currents), and specific locations (e.g., estuaries where suspended matter is carried by river runoff) (Baker, 1983). Baker (1983) measured sedimentation rates of suspended matter <2 to >9 g m^{-2} day^{-1}. Forty to fifty percent of the suspended matter was organic. Furthermore, Baker found experimentally that the sedimentation rate of oil was 0.5 to 32 mg m^{-2} day^{-1}. There is a turbid boundary layer near the sediment surface. This layer and its dynamics was extensively studied in the 1950's (re. Kuenen's turbidity currents). Some later studies of the turbid bottom boundary layer (or nepheloid layer) have been empirical (e.g., Baker, 1983) as well as theoretical (Adams and Weatherly, 1981).

There would obviously be some direct absorption of oil to sediments if and when the mixed layer reaches the bottom.

The sedimentation of oil is a function of time. Ultimately 30 to 50% of the oil residue may reach the bottom (Elmgren and Frithsen, 1982). However, much of the sedimentation of the oil occurs after the surface slick is broken up and transported long distances. Thus, expectedly the sedimented oil will cover large areas and the resulting concentrations of oil on the bottom would be low over most of these areas. In our study we are interested in the sedimentation in the first 15 days (to maximum 30 days for a long-lasting blowout) before the surface slick is broken up and disappears as a semicontinuous layer.

2.2 Quantitative formulation of oil sedimentation.

The distribution of oil on and in the water is computed and given in model grids (about 2.3 km grid size) either in 12-hour or daily time steps (Liu, 1983). The oil in the water column is converted to concentrations (e.g., ppb) to facilitate the evaluation of its effects to biota. Thus, we need to give the quantities of oil on or in the bottom also in terms of concentrations.

Sedimented oil accumulates initially in a flocculus nepheloid layer near the bottom (Elmgren, pers. comm.). For our present purpose we assume that the thickness of this layer is 10 cm, with the concentration of the oil in this layer expressed in the same units as in the water (ppb). The thickness of this nepheloid layer is not uniform and might even be absent in many locations. Further research is required in this matter.

The following formulas for time-dependent computation of the sedimentation of oil have been derived on the basis of the available meager information, most of which is summarized in Chapter 1.2 and 2.1. It is neither possible, nor justifiable to devise theoretical formulas for which necessary parameters are not available, nor verification/validation possible. The various earlier theories on sedimentation are not valid, mainly due to complex flocculation processes as shown in earlier chapters. The following proposed empirical (or, rather, rational) formulas are derived on the premises that the parameters, which can be estimated, are related to the processes of sedimentation of oil. For example, the turbulence in the water, which enhances the collision between minerogen suspended particles and oil droplets, is a function of wind speed. Furthermore, the higher the wind speeds the deeper the surface mixed layer, which might reach bottom in shallower water. In this case the turbulence will bring oil emulsion into contact with the bottom and enhance adsorption of oil to bottom sediments. Furthermore, higher turbulence (equated here with wind) might suspend (erode) more sediment, thus enhance oil sedimentation.

The rate of deposition of oil is made a function of turbulence, which is approximated with wind speed (W), depth of water (D), and concentration of oil in the water (S). The time step is selected either as 12 or 24 hours. Computations are made at each grid point at each time step. The balance of oil is not preserved in the following formulation. The reason for this is that there is an excess of oil on the surface which might go in emulsion into the water (or might be transported away with surface wind and currents).

In order to simulate known differences in sedimentation rate, slightly different constants are used in the continuous source (blowout) and instantaneous source (e.g., tanker accident) cases. Some constants also differ, depending upon the presence or absence of a thermocline (re. suspended oil coming into direct contact with sediment).

<u>Instantaneous source without thermocline</u>:

$$AO = AO_{t-1} + S_t * F_s * P * R * B \qquad (1)$$

where: $F_s = (0.0015W + 015/D^{0.7}) * TK_s$ (2)

and: $TK_s = K/(3 + 0.2K)$ (3)

AO_t is the concentration of oil in "nepheloid layer" at time t; AO_{t-1} is the same concentration in previous time step after decay (see Chapter 3 below);

S_t is the concentration of oil in the water in the surface mixed layer;

P is the zooplankton abundance index (relative values from 1.0 to 2.0, estimated on the basis of expected zooplankton abundance in the location and season);

R is the minerogen suspension index (abundance of minerogen matter) and is made a function of depth: $R+0.2D/\sqrt{D})$, whereby R is selected between 20 and 50 (Note: the amount of minerogen suspended matter is seldom measured, thus a relative abundance index (turbidity index) must be estimated);

B is the bottom type index (0.3-rocky; 0.6-coarse sand and gravel; 1.5-fine silt and clay); (this index simulates the adherence of oil to the bottom);

F_s is the sedimentation rate factor;

TK_s is the time factor;

W is wind speed (in m/sec);

D is depth in meters;

K is number of time steps (in days).

No computation of oil sedimentation is made for first 12-hour period.

<u>Instantaneous source with thermocline</u>:

$$AO_t = AO_{t-1} + S_t * F_d * P * R \tag{4}$$

where: $F_d = (0.001W + 0.20/D^{0.7}) * TK_d$ (5)

and: $TK_d = 6/(3 + 0.5K)$ (6)

F_d is the sedimentation rate factor;

TK_d is the time factor.

All other symbols (and parameters) are the same as in Formulas 1 to 3. No computation of oil sedimentation is made for the first 24-hour period as sedimentation through the thermocline is a time-dependent process.

The relationship of sedimentation factor to depth is shown in Figure 3 and the increase (growth) of time factor with time is given graphically in Figure 4.

<u>Continuous source, no thermocline present</u>:

$$AO_t = AO_{t-1} + S_t * F_{cs} * DF * P * R * B \tag{7}$$

where: $F_{cs} = (0.000iW + 0.25/D^{0.74}) * TK_s$ (8)

and: $DF = (Dis + 4)/20 + 0.1Dis$ (9)

DF is the "distance from source" factor;

Dis is distance (of the grid point) from source in km.

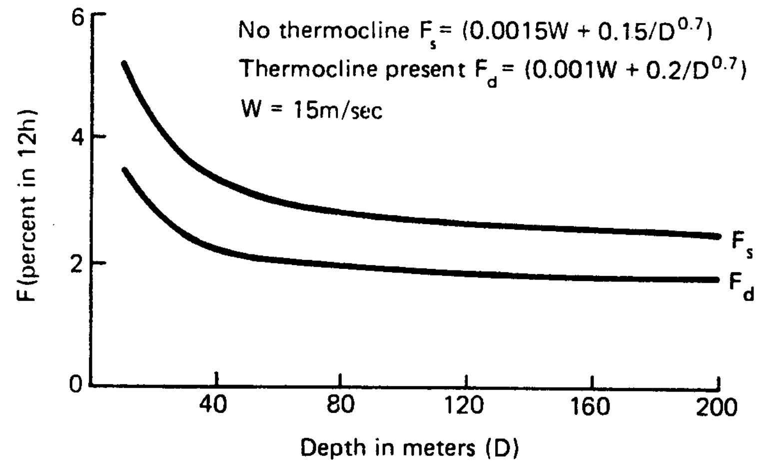


Figure 3.--Oil sedimentation factor F, Instantaneous source.

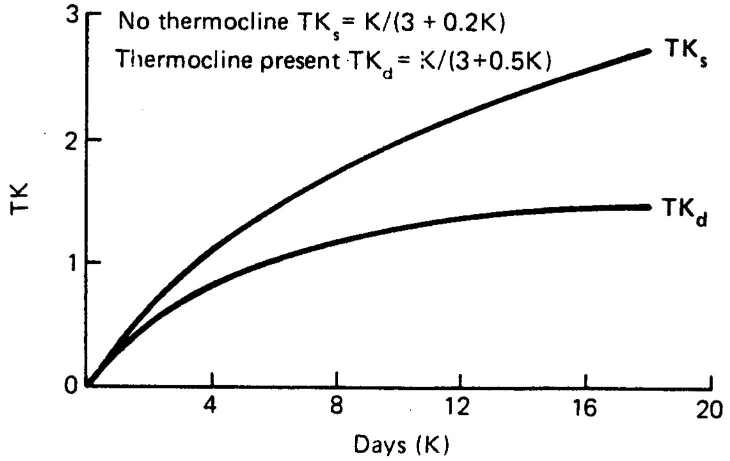


Figure 4.--Time factor for oil sedimentation.

All other symbols correspond to the symbols in Formulas 1 to 3. No computation is made for the first 12-hour period. The dependence of sedimentation rate factor (F_{cs}) on depth is shown in Figure 5, and the distance factor is shown in Figure 6.

Continuous source, thermocline present:

$$AO_t = AO_{t-1} + S_t * F_{cd} * DF * P * R \quad (10)$$

where: $F_{cd} = (0.0008W + 0.035/D^{0.74}) * TK_d$ (11)

No computation is made in the first 24-hour period as sedimentation through the thermocline is a time-dependent process. All symbols correspond to those in Formulas 1 to 3 and 7 to 9.

3. FATE OF OIL ON THE BOTTOM

3.1 Some observations of the fate of oil on the bottom.

The initial accumulation of oil in the bottom nepheloid layer is difficult to observe and sample. These flocculous accumulations are not retained by conventional grabs and other bottom sampling devices. Some conclusions about its existence can be drawn from laboratory tests and from uptake of hydrocarbons by sessile filering organisms, such as clams and polychaetes.

The oil-containing nepheloid layer is expected to move around along the bottom with currents near the bottom and may accumulate in deeper holes (deepenings) in the bottom. Linden, et. al., 1979, found ten months after the TSESIS spill that hydrocarbon concentrations in Macoma balthica increased unexpectedly at a given sampling station. Such an event may be the result of the exposure of the clams to recontamination from oil in the drifting bottom

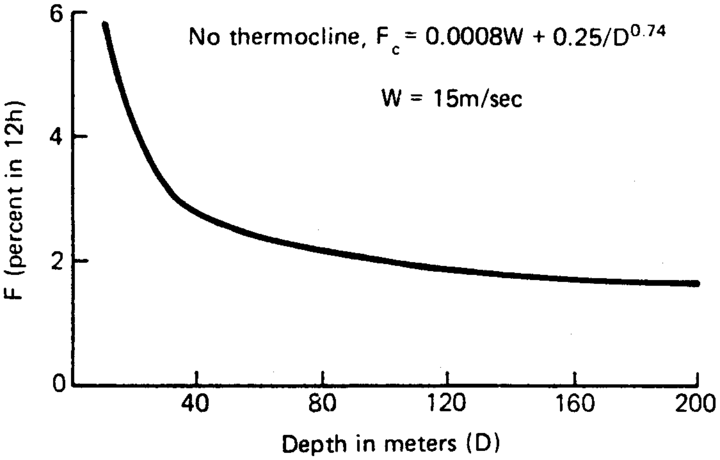


Figure 5.--Oil sedimentation factor F, Continuous source.

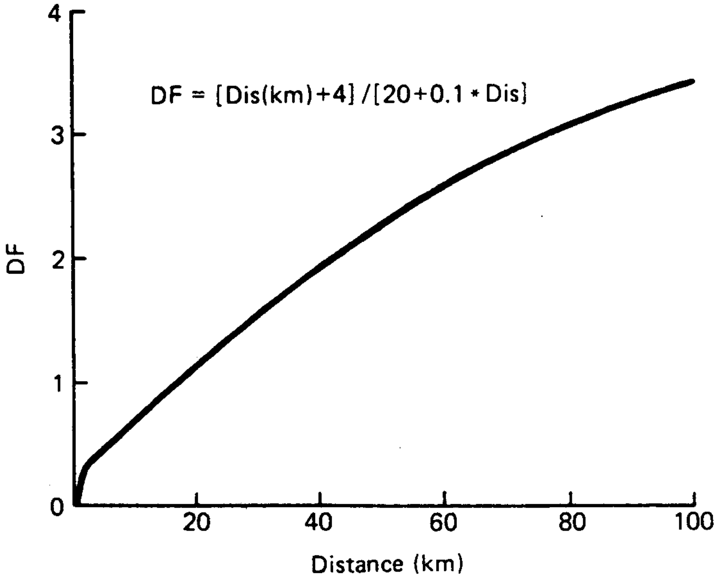


Figure 6.--Distance factor, Continuous source.

nepheloid layer. However, the oil concentrations associated with the nepheloid layer and with sediments is in ppb range and cannot be a major pathway for dispersal of oil (Malinky and Shaw, 1979).

The oil from the nepheloid layer gets absorbed into the sediment, and is carried deeper into it by burrowing animals. In experimental tanks, where the oil concentration in water was kept about 190 ppb for 25 weeks, the top 2 cm of sediment had a hydrocarbon concentration of 109 ppb after 20 weeks (Grassle, et al., 1981). In the area of AMOCO CADIZ spill, oil was found to 5 to 7 cm depth in the sediment five months after the spill. Higher concentrations were found in fine sediments (D'Ozouville, et al., 1979).

The oil in the sediment undergoes decay (weathering); biodegradation being probably the most important decay process. Biodegradation is known to increase with temperature (Gearing, et al., 1979). Furthermore, the decay is assumed to be a function of depth (the "aeration" of sediments and the amounts of biota in them are both in general functions of depth). After concentrations are reduced to some tolerable range, the weathering rate of sedimented oil may be accelerated by the activities of deposit feeders such as polychaetes (Gordon et al., 1978).

Some of the oil gets back into the water above via interstitial water (Vandermeulen and Gordon, 1976). In experimental tanks, 10 to 20% of the oil remained in the sediments after one year (Elmgren and Frithsen, 1982), and in AMOCO CADIZ oil spill area some oil remained in fine-grained sediments three years after the spill (Gundlach, et al., 1983). Residues of Bunker C were identifiable in some locations off Nova Scotia 6 years after the spill from the ARROW (Keizer et al., 1978).

3.2 Computation of the decay of oil on the bottom.

In the oil-on-bottom simulation model (Chapter 5) the "decay" of oil from previous time step is decayed before new oil is added. The "decay" signifies the photo-oxidative degradation of aromatic more toxic components, biodegradation, as well as oil being buried into the sediment. The following formula (12) gives the decay in 12-hour time step which is repeated for the 24-hour time step.

$$AO_{to} = AO_{t-1}\, e^{-(t+d)} \qquad (12)$$

where: $t = T^{2.7} * 10^{-4}$ (13)

and: $d = 0.15/\sqrt{D}$ (14)

t is temperature factor;

T is temperature in °C;

d is depth factor;

D is depth in meters;

The relations between t and T, and d and D are given in Figures 7 and 8, respectively.

Examples of computed distribution of oil in the water and in the bottom are given in Figures 9 and 10. Figure 9 gives the distribution of oil in the water 10 days after a blowout. Corresponding to the same event, the distribution of oil on the bottom is given in Figure 10. The bottom slopes up from the blowout to the north, causing the higher values in the northern part of the field. Figure 10 shows that the concentrations of oil in the bottom nepheloid layer can be considerably higher than the concentrations of oil in the water, thus demonstrating the greater importance of oil in the bottom in respect to its effects on marine biota.

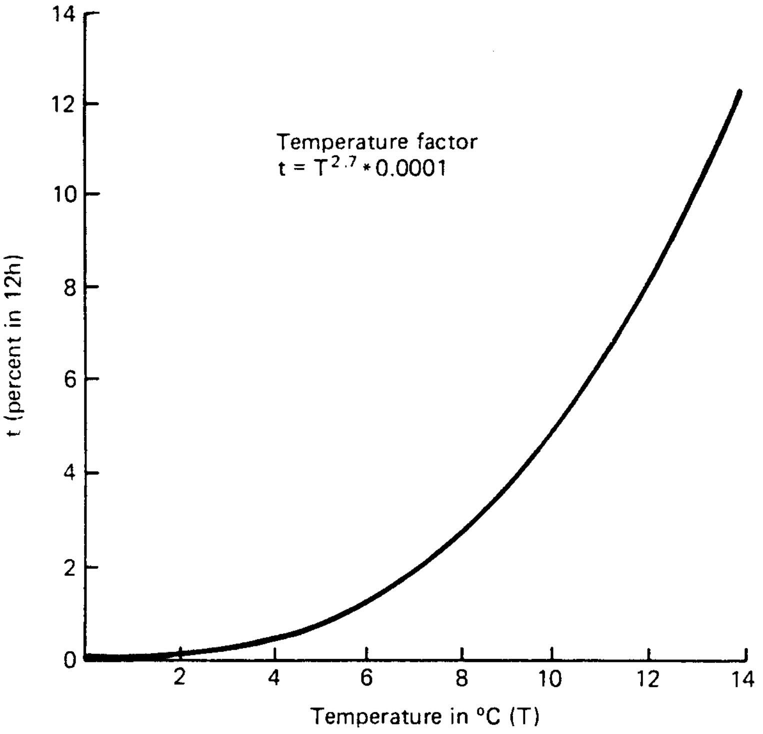


Figure 7.--Effect of temperature on the "decay" of oil on the bottom (time step 12 hours).

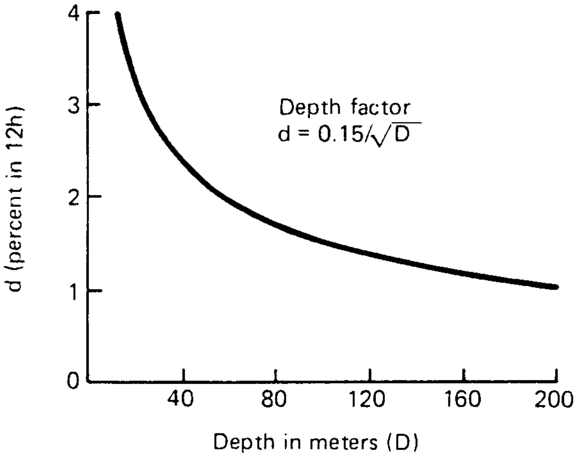


Figure 8.--Effect of depth on the "decay" of oil on the bottom (time step 12 hours).

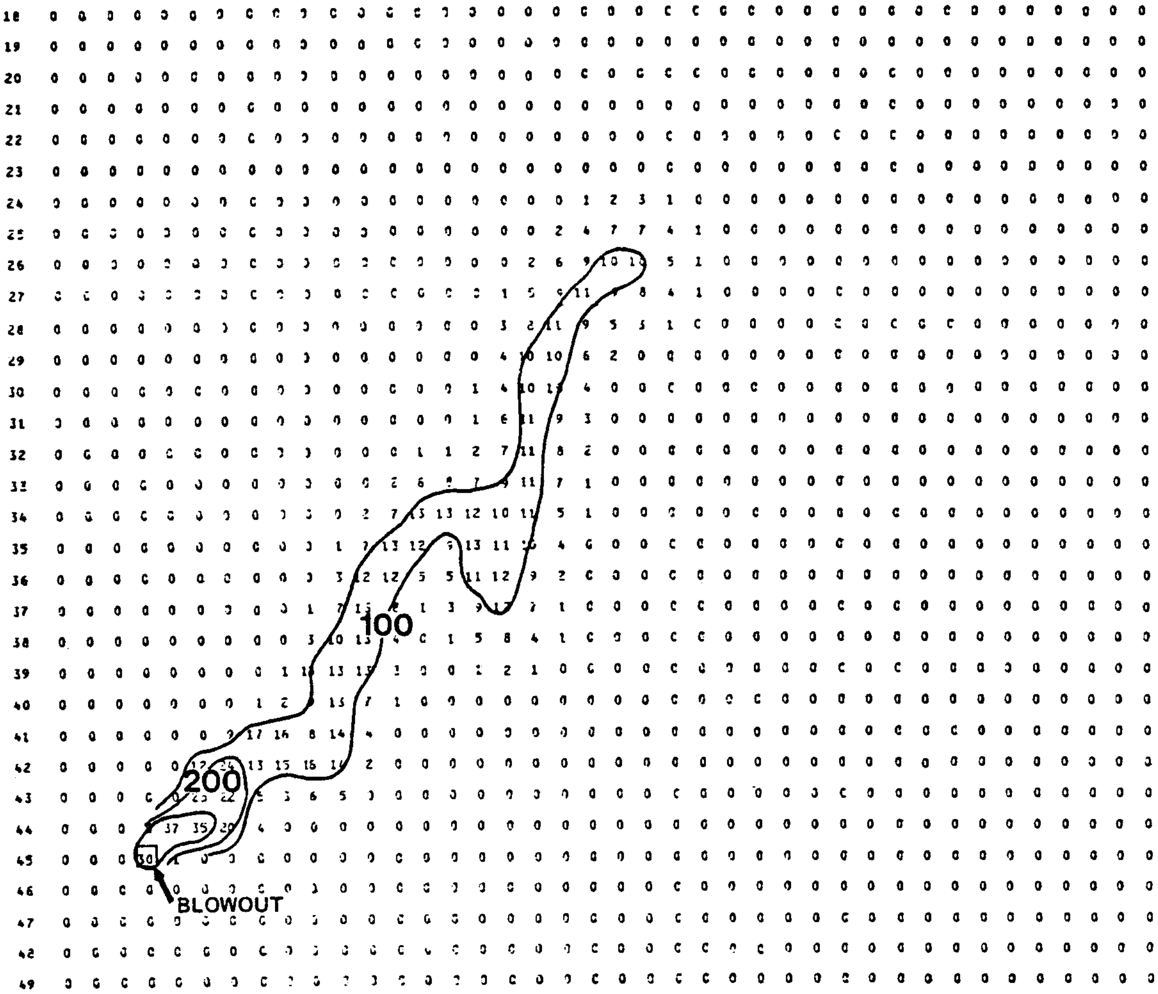


Figure 9.--Distribution of oil from a blow-out (20,000 bbl/day) after 10 days (concentrations in ppb in water - from surface to 15 m; grid size 2.3 km).

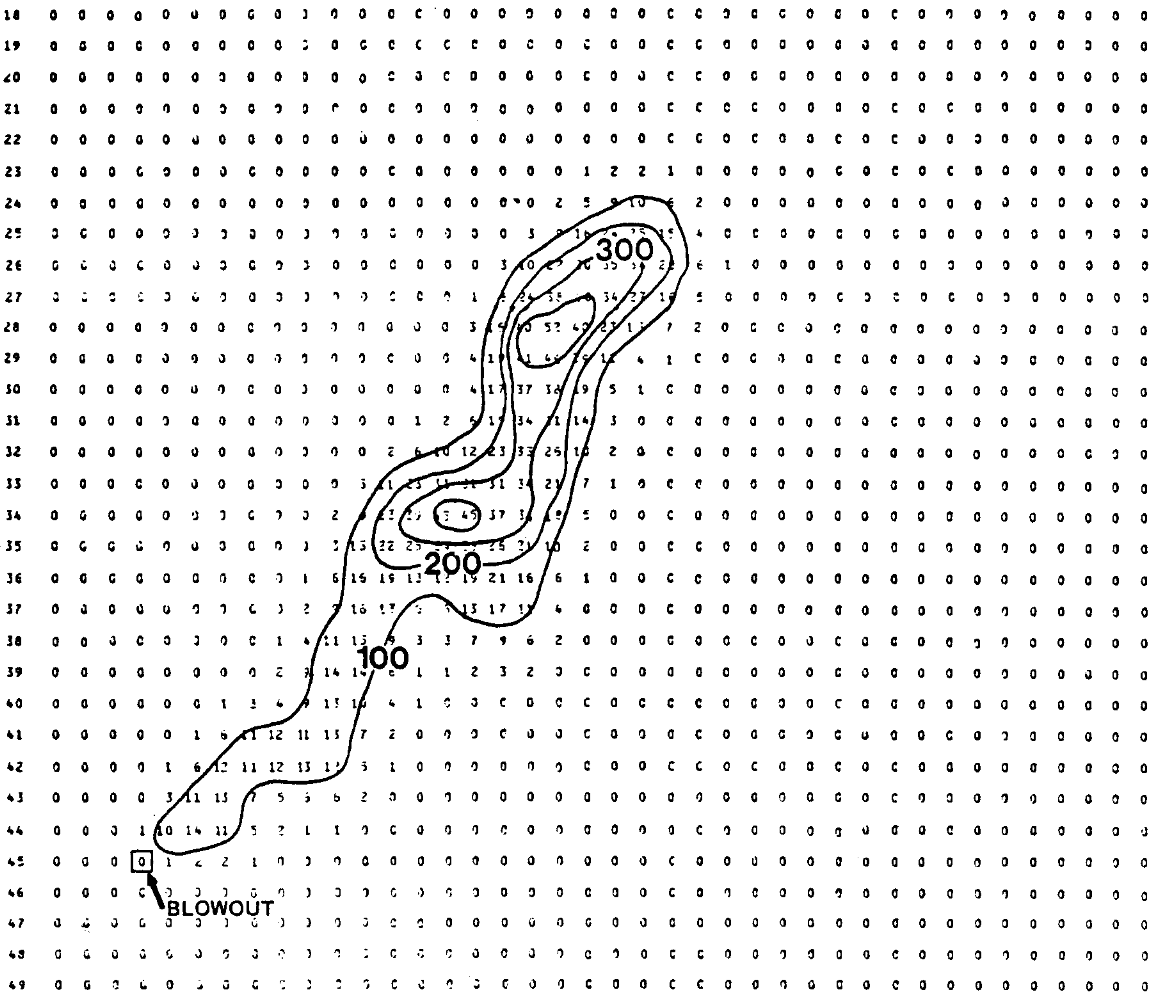


Figure 10.--Distribution of oil in the bottom nepheloid layer (10 cm) in ppb 10 days after a well blowout (see Figure 9); grid size 2.3 km.

The empirical formulae for the time-dependent simulation of the sedimentation of oil, given in this chapter, are based on meager semi-quantitative information (mostly estimates) available in this subject. Further quantitative experimental studies are needed to improve the provisional values for the parameters and coefficients proposed in this paper, and to validate the numerical model in general.

4. EFFECTS OF OIL ON THE BOTTOM ON DEMERSAL FISH AND BENTHIC ECOSYSTEMS

4.1 Avoidance of oiled bottoms by fish and other marine animals.

Some laboratory tests show that fish (e.g., cod) can detect very low concentrations of hydrocarbons, indicating this detection by snapping, darting, coughing, and restless swimming (Hellstrom and Doving, 1983). It is thus possible that some fish (especially semi-demersal species) might avoid oiled bottoms by vertical (upwards) movement into the water mass above the oiled nepheloid layer. The changes of availability of flatfish (sole) in shallow abers after the AMOCO CADIZ spill might be an indication of avoidance of these oiled areas by fish (Gundlach, et al., 1983). On the other hand, laboratory experiments with oiled and clean sediments do not indicate a definite choice of clean sediments by flatfish (Fletcher et al., 1981).

Some epibenthic crustaceans might also use the escape from oiled sediments by movement into water mass above, which might partly explain the disappearance of amphipods from TSESIS spill area.

Burrowing clams do not burrow deep in oiled sediments. This behavior might also be considered as an escape behavior (Olla and Bejda, 1983). Many animals remain, however, on and in oiled bottoms and get contaminated by hydrocarbons by direct adsorption as well as *via* food chain. Other known effects of oiled bottoms on animals are given in Chapters 4.3 and 4.5.

4.2 Uptake of hydrocarbons from oiled bottom.

Hydrocarbons are taken up by biota with different processes, such as adsorption and absorption (especially through gills) and through food chain. Many filtering animals (such as bivalves) will take up hydrocarbons from the nepheloid layer in their filtering process.

Considerable bioaccumulation of hydrocarbons in the benthic animals in oiled areas has been observed in numerous studies. These studies on the uptake and bioaccumulation of hydrocarbons from sediments are reviewed by Connell and Miller (1981). The food chain transfer predominates the hydrocarbon transfer processes (Fowler, 1982). For the purpose of computation of hydrocarbon transfer through the food chain, a conservative bioaccumulation ratio of 50 is assumed. The uptake and decay (depuration) of hydrocarbons by fish and its effects (e.g., tainting) are described in another report in this project report series.

4.3 Effects of oil on the bottom on benthic organisms and demersal fish.

The effect of oil studies have been mostly toxicity studies, using high oil concentrations in laboratory tanks which cannot occur in any accidental release of oil in nature. The concentrations of oil on the bottom, though higher than in water, rarely reach 1 ppm (except in case of beaching of oil) (see Figure 10). Usually less than 10% of the oil initially reaching the bottom is soluble aromatic derivatives (SAD), which are more toxic. Furthermore, SAD disappear quickly from the "weathered" oil on the bottom. Moore and Dwyer, 1974, give the following tables of toxic concentrations of SAD.

5 to 50 ppm	fish
0.1 to 1 ppm	larvae
1 to 10 ppm	crustaceans
5 to 50 ppm	bivalves

Feeding and reproduction can be "disrupted" with lower concentrations (10 to 100 ppb). One recent study by Kanter et al. (1983) has, however, used low levels of petroleum hydrocarbons (6 to 760 ppb) and longer exposure times (about a month) in the studies of the effects of oil on larval and adult stages of California halibut, northern anchovy, and mussels. Results show that larval stages are more sensitive to the exposure to hydrocarbons than previously expected. However, these results are in conformity with Norwegian investigations on the effects of hydrocarbons on eggs and larvae (50 ppb and up), where the effects occur years later as lower exploitable biomasses. However, these later effects are difficult to qualify and separate from changes of natural mortality, effects of fishing, and other natural fluctuations.

Benthic animals are considered to be less sensitive to the toxicity of oil than the pelagic animals (Rice et al., 1979). On the other hand, filtering animals can accumulate hydrocarbons rapidly from relatively low concentrations in bottom nepheloid layer. Oysters can get tainted from 10 ppb of hydrocarbons in water if exposure is of sufficient duration. The tainting levels for fish, crustaceans, and clams is between 4 to 300 ppm (Connell and Miller, 1981; see also summary of various sublethal effects by these authors).

Oil on the bottom can affect the reproductive capacity and embryonic development of benthic and demersal animals. Linden et al., 1979, found that the amphipods Pontoporeia affinis and P. femorata had abnormal eggs 5 months after the TSESIS spill. After the AMOCO CADIZ spill, low percentage of egg-carrying female oysters were observed in 1978/79 (Gundlach et al., 1983).

Augenfeld (1980) found that very high levels of oil concentration in sediment (500 to 1000 ppm) caused some reduction in feeding of Abarenicola pacifica. Reduced feeding by winter flounder on heavily oiled sediments

(2300 to 4500 ppm) were also reported by Fletcher et al., 1981. Such heavy concentrations of oil can be found only in shallow water in case of beaching of oil slicks. On the other hand, Payne et al., 1983, found that the sublethal effects of hydrocarbons on American lobster were minor indeed, only gill browning might have been considered pathological in nature. In similar studies with fish by Payne et al., 1978, no histopathological changes were observed after 6 months and no serious differences in growth and reproduction between oil exposed and control experiments were observed.

Eggs and larvae might be most susceptible to exposure to oil. McIntyre, 1982, states that growth and buoyancy in cod eggs and larvae were affected by oil concentration of 50 ppb, and at 250 ppb malformation of larvae occurred. There are relatively few species with demersal eggs (e.g., herring, egg-carrying females of crabs). The problems of pelagic eggs are dealt with elsewhere in this report series (see REEST, 1983).

4.4 Decay of hydrocarbons in marine organisms.

The knowledge on the metabolism of hydrocarbons in marine organisms has been summarized by Connell and Miller, 1981. Numerical studies of the decay of hydrocarbons is described in another report in this series pertaining to the effects of oil on fish (see REEST, 1983).

The decay of hydrocarbons in demersal fish and benthic organisms is complicated by the continuous uptake of oil from sediments. Filtering and burrowing animals effect the uptake of the weathered oil, which is transferred to fish feeding on them. Linden et al., 1979, found that flounders (*Pleuronectes flesus*, which feed on *Macoma balthica*, showed 50 ppm of hydrocarbons in liver and muscles one year after *TSESIS* spill.

The accumulation, as well as decay of hydrocarbons in fish, is a function of temperature (Varanasi, Gmur, and Reichert, 1981). Retention is higher and decay slower at lower temperatures. In general, the hydrocarbons are lost at a slower rate than they are accumulated (Fowler, 1982).

The computations of decay of hydrocarbons in fish was done in this study with the following general exponential formula, corresponding to the findings of Fowler, 1982:

$$C_t = C_{t-1}\, e^{-b} \qquad (15$$

where: $b = 0.0015T^2$ for demersal fish (16)

and: $b = 0.002T^2$ for pelagic fish (17)

t is time step (12 hours);

C is concentration of hydrocarbons in fish (mainly muscle);

b is decay factor;

T is temperature in °C.

This formula gives about 8% decay in 12 hours at about 10°C. The dependence of the decay from temperature is shown in Figure 11.

4.5 The effect of oil on the bottom on the benthic ecosystems.

Most of the knowledge of the effect of oil on benthic ecosystems originates from tank experiments and field research in TSESIS spill area. Elmgren et al., 1980, found in tank experiments that benthic macrofaunal and metazoan meiofaunal populations declined drastically in "oiled sediments", whereas benthic diatoms and protozoa increased considerably. Benthos biomass in oiled tanks was only about 10% of that in control tanks. Amphipods were sensitive to oil, harpacticoids were not (Elmgren and Frithsen, 1982).

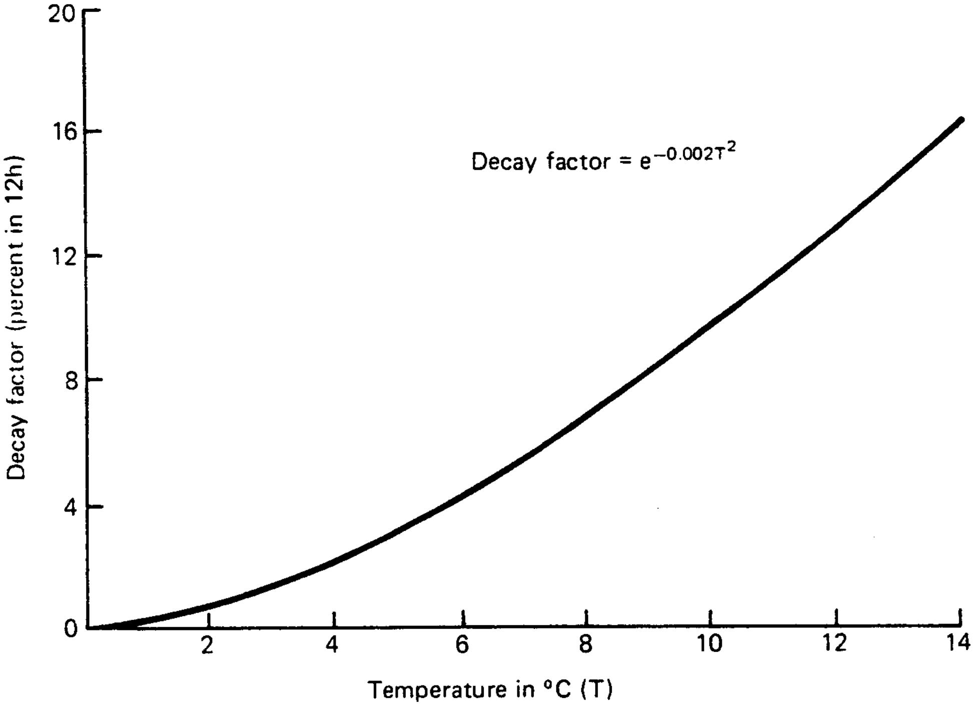


Figure 11.--The effect of temperature on the decay of hydrocarbons in fish.

Middleditch et al., 1982, found that shrimp populations in Buccaneer oil field were not affected by oil developments. On the other hand, changes in benthos in shallow water were rather profound in AMOCO CADIZ spill area, and after three years benthos communities had not reached their former nor new equilibria (Conan, 1982). Species with short life cycles tend to replace long-lived species.

In TSESIS spill area, mobile epibenthic macrofauna was drastically reduced. However, bivalves (*Macoma balthica*) increased greatly (Linden et al., 1979). Small bivalves serve as food source for many demersal fish species. Thus, it cannot be assumed that the changes in benthic ecosystem are always negative from the fisheries production point of view.

5. NUMERICAL SIMULATION OF THE SEDIMENTATION OF OIL

5.1 Overview of the computer programme.

General

The subroutine OILBOT for sedimentation of oil is a part of a larger programme for numerical computations of the effects of oil on marine fisheries ecosystem (DEMOIL). Only the subroutine OILBOT and a few other subroutines essential to it are described and documented herein.

The control programme DEMOIL sets various parameters and calls other subroutines. The computations in the enclosed model are done in a 49 x 54 grid, with a grid size of 2.3 km.

The index BLO (input in control programme) determines whether the oil source is continuous (well blowout) or instantaneous (tanker accident). There are several indices to select for the mode of transport of oil on the bottom with currents. A current subroutine (CUROIL) is used for computation of movement of oil on the bottom, which is essentially the same as that used for advection of smell from baits, and is documented by Olson and Laevastu, 1983.

The oil distribution in the water is computed by Rand Corporation (Liu, 1983) and provided to this project in a grid in 24-hour time step. This oil concentration field (S) in water is read in every time step and converted to concentrations of ppb. The field is printed out with printing subroutine PRIMFS (output see Figure 9). The field is scaled with scaling index LU for convenient printing of the array.

The subroutine EGGLAR is for computation of the exposure of eggs and larvae in water to different concentrations of oil. Subroutine STAFIE computes the corresponding exposure of fish, both to oil in the water as well as oil on the bottom. Subroutine CONFOOD computes the contamination (and tainting) of stationary as well as migrating fish through the food chain. The last-mentioned three subroutines will be documented in NWAFC/REEST Programme Documentation series.

Subroutine SILITA, included in Chapter 5.3, is a 5-point Laplacian type smoother.

Subroutine OILBOT

This subroutine, reproduced in Chapter 5.3, includes a simulation of depth in first time step. In the operational mode, depth should be read in from a prepared data statement or from tape or cards.

Four different bottom temperatures, two mixed layer depths, and three wind speeds are introduced with statements (see Input Parameters) which can be selected for the runs by the "selection parameters" (KT, KP, and KW). Indices for plankton concentrations, suspended minerogen matter, and type of bottom are also introduced in the first time step.

In all other time steps, except the first, the decay of the oil on the bottom left from previous time step, is computed before adding new oil (formula - see Chapter 3.2).

The computation of the sedimentation of the oil is done in 12-hour time steps (repeated if 24-hour time step for calling of the subroutine is used). The selection of the computation formula (see Chapter 2.3) depends on the nature of the spill (continuous or instantaneous) and whether thermocline is present at the grid point or not.

After time step computations, the field is smoothed (subroutine SILITA) and printed (subroutine PRIMFS).

5.2 Symbols and abbreviations used.

Note: Symbols marked with * are input parameters.

*ALPHA - Smoothing parameter (0.78)

AO(N,M) - Concentration of oil in the bottom nepheloid layer (ppb)

*APD - Minimum distance from blowout where sedimentation is computed (2.5 km)

*BB - Bottom type index (0.3-rocky, 0.6-coarse sand and gravel, 1.5-fine silt and clay)

*BCF - Wind speed coefficient (0.0015, 0.001)

*BLO - Index of the mode of computation; 2-continuous source, 1-instantaneous source

*BWF - Wind speed coefficient (0.0016, 0.001)

*CCF - Depth coefficient (0.15, 0.2)

*CDF - Depth coefficient (0.15, 0.2) (Possibility to select different values with continuous source)

*D(N,M) - Depth in meters

DDP - Intermediate (depth factor)

DFA - Intermediate (depth exponent)

DIFAC - Intermediate (distance factor)

DIS - Distance factor (from blowout)

*DL - Grid size (m)

EFA - Intermediate (decay exponent)

FDD - Intermediate (turbulence factor) (F_d)

FS - Intermediate (turbulence factor) (F_s)

K - Counter of 24 h time steps

*KA - Index for type of bottom current; 1 - laminar (used in this programme) (2 layer thickness increasing with distance from "source" - used in computation of distribution of smell from baits)

*KAL - Index for computation of oil advection on the bottom; 0-no advection, 1-compute advection

*KP - Index for potential mixed layer depth value

*KT - Index for bottom temperature value

*KU - "Type of current" indicator; (1 - uni-directional in u direction, 2 - uni-directional in v direction), 3 - current in both components (u and v) - used in this programme

*KW - Index for wind speed value

*LU - Printing and scaling index (see listing in the beginning of subroutine OILBOT)

*ME - Total number of grid points in x direction

*MO - m coordinate of blowout location

*NE - Number of grid points in y direction

N - Grid point counter (y axes)

*PLD(i) - Potential mixed layer depth (m) (2 values given)

*PP - Relative concentration of plankton (1.0 to 1.8)

*R - Relative amount of minerogen suspended matter in the water (20 to 30)

RR - Intermediate (minerogen suspension coefficient)

*S(N,M) - Oil concentration in water in ppb

SK - K, time step counter

STK - Intermediate (time step coefficient) (TK_s)

T - Time counter in minutes

*TAT - Time step in hours

*TB(i) - Bottom temperatures (°C) (4 values given)

*TD - Time step in minutes, for computation of advection of oil (subroutine CUROIL)

TDK - Intermediate (time step coefficient) (TK_d)

TFA - Intermediate (temperature exponent)

*UI - u component of the current on the bottom (in m/min)

*VI - v component of the current on the bottom (in m/min)

*W(i) - Wind speed (m/sec) (3 values given)

5.3 Programme DEMOIL and subroutines OILBOT, SILITA and PRIMFS.

```
$RESET FREE
$SET LINEINFO OWN LIST
FILE  6(KIND=PRINTER)
FILE  66(KIND=PRINTER)
FILE  5(TITLE="PERM/RAND/DATSUM/SUBSURFACE/D1",KIND=DISK,FILETYPE=7)
C     PROGRAM DEMOIL
      DIMENSION S(49,54),PF(49,54),D(49,54),AO(49,54),TB(4),PLD(2),W(3
     2,E(5,2),SE(5,14),FE(5),DIF(5,2)
      COMMON S,PF,D,AO,TB,PLD,W,E,SE,FE,DIF,
     2K,T,TD,DL,UI,VI,BLO,KAL,KU,KA,TAT
      PRINT 30
   30 FORMAT(1H1,5X,20HWIND SPEED 10 M/SEC //)
      PRINT 31
   31 FORMAT(/5X,20HBOTTOM TEMP.  8 DEG.C//)
      PRINT 32
   32 FORMAT(/5X,22HTHERMOCLINE DEPTH 20M//)
      NE=49
      ME=54
      K=1
      BLO=2.
C     BLO=2 CONTINUOUS SOURCE,  BLO=1 INSTANTANEOUS SOURCE.
      DL=2300.
C     TAT TIME STEP IN HOURS
      TAT=24.
      TD=20.
   10 T=K*1440.
C     TIME IN MINUTES
      KAL=1
C     KAL=0 - NO OIL MOVEMENT ON THE BOTTOM, 1 OIL ADVECTED ON BOTTOM
C     KU - CURRENT INDEX, SEE CUROIL; KA - TURBULENCE INDEX(NOT USED);
C     LU - PRINT SCALING INDEX
      KU=3
      KA=1
      LU=0
      UI=0.
      VI=0.
      READ(5,12)((S(N,M),M=1,54),N=1,49)
   12 FORMAT(9F8.0)
C     CONCENTRATIONS IN PPM, CONVERTED TO PPB
      DO 11 N=1,NE
      DO 11 M=1,ME
      S(N,M)=S(N,M)/1500000.
```

```
   11 CONTINUE
      CALL PRIMFS(S,T,UI,VI,DL,K,KA,KAL,BLO,LU)
      CALL OILBOT(S,K,TD,DL,D,AO,TB,BLO,UI,VI,KU,KAL,T,KA,TAT)
      CALL EGGLAR(S,DL,K,SE,E,FE,DIF)
      CALL STAFIE(S,AO,K,DL)
C     MOVE THE OIL ON THE BOTTOM TO THE RIGHT OF SFC FLOW
      UI=60.
      VI=8.
C     KAL=1  COMPUTE OIL MOVEMENT ON BOTTOM
      IF(K-1)15,15,14
  14  CONTINUE
      CALL CUROIL(AO,KU,UI,VI,DL,K,BLO,T,KAL)
      CALL CONFOO(S,AO,K,DL,BLO,TAT)
   15 K=K+1
      IF(K-15)10,10,20
   20 STOP
      END
```

```
      SUBROUTINE OILBOT(S,K,TD,DL,D,AO,TB,BLO,UI,VI,KU,KAL,T,KA,TAT)
      DIMENSION S(49,54),D(49,54),AO(49,54),TB(4),PLD(2),W(3)
C     D-DEPTH
C     AO-OIL ON THE BOTTOM
C     TB-BOTTOM TEMPERATURE, FOUR VALUES GIVEN
C     PLD-THERMOCLINE DEPTH, TWO VALUES
C     W-WIND SPEED, THREE VALUES
C     KT-INDEX OF TB VALUE CHOSEN FOR THE RUN
C     KP-INDEX OF PLD VALUE
C     KW-INDEX OF WIND VALUE
C     BLO=1 INSTANTANEOUS SOURCE, =2 CONTINUOUS SOURCE
C     UI-SURFACE CURRENT SPEED
C     KAL=1 COMPUTATION OF OIL MOVEMENT ON BOTTOM
C     LU=1 DEPTH DATA
C     LU=2 DECAY OF OIL ON THE BOTTOM
C     LU=3 OIL ON THE BOTTOM BEFORE ADVECTION
C     LU=4 OIL ON THE BOTTOM,LAYER THICKNESS DECREASING, ADVECTED
C     LU=5 ADVECTED OIL ON THE BOTTOM
C     LU=6 CONTAMINATION INDEX, PELAGIC FOOD
C     LU=7 CONTAMINATION INDEX, DEMERSAL FOOD
      NE=49
      ME=54
      MO=3
C     MO IS THE M LOCATION OF BLOWOUT
C     SIMULATION OF DEPTH, SLOPING TOWARDS HIGHER N
C     DEPTH CAN BE READ IN
      IF(K-1)16,16,20
   16 DO 11 N=1,NE
      DO 11 M=1,ME
      IF(43-N)12,12,13
   12 D(N,M)=50.
      GO TO 11
   13 IF(36-N)14,14,120
   14 D(N,M)=D(N-1,M)+4.
      GO TO 11
  120 IF(29-N)122,122,15
  122 D(N,M)=D(N-1,M)+2.
      GO TO 11
   15 D(N,M)=8.
   11 CONTINUE
CXXXXXXXXXXXXXXXXXXXXXX
      LU=1
      CALL PRIMFS(D,T,UI,VI,DL,K,KA,KAL,BLO,LU)
CXXXXXXXXXXXXXXXXXXXXXXX
   17 DO 18 N=1,NE
      DO 18 M=1,ME
      AO(N,M)=0.
   18 CONTINUE
C     INPUT PARAMETERS
   20 TB(1)=1.
      TB(2)=4.
      TB(3)=8.
      TB(4)=12.
      PLD(1)=20.
      PLD(2)=40.
      W(1)=5.
      W(2)=10.
      W(3)=15.
C     PP - RELATIVE CONC. OF PLANKTON
C     R - INDEX OF SUSPENDED MATTER
```

```
C       BB - BOTTOM TYPE INDEX
        PP=1.5
        R=20.
        BB=0.8
C       SETTING OF INDICES FOR INPUT PARAMETERS
        KT=3
        KP=1
        KW=2
CXXXXXXXXXXXXXXXXXXXXXXX
C       DECAY OF OIL ON THE BOTTOM
        IF(K-2)30,25,25
     25 DO 29 N=1,NE
        DO 29 M=1,ME
        IF(AO(N,M))29,29,26
     26 TFA=(TB(KT)**2.7)*0.0001
        DFA=0.15/SQRT(D(N,M))
        EFA=-(TFA+DFA)
        AO(N,M)=AO(N,M)*EXP(EFA)
        IF(TAT-12.)29,29,27
     27 AO(N,M)=AO(N,M)*EXP(EFA)
     29 CONTINUE
CXXXXXXXXXXXXXXXXXXXXXXXX
        LU=2
C       CALL PRIMFS(AO,T,UI,VI,DL,K,KA,KAL,BLO,LU)
CXXXXXXXXXXXXXXXXXXXXXXXXX
     30 IF(BLO-1)31,31,51
C       INSTANTANEOUS SOURCE (TANKER ACCIDENT)
     31 DO 45 N=1,NE
        DO 45 M=1,ME
        IF(PLD(KP)-D(N,M))40,33,33
C        NO PYCNOCLINE
     33 SK=K
     35 STK=SK/(3.+0.2*SK)
     56 BCF=0.0015
        CCF=0.15
        RR=(R+0.1*D(N,M))/SQRT(D(N,M))
        FS=(BCF*W(KW)+CCF/(D(N,M)**0.7))*STK
        AO(N,M)=AO(N,M)+S(N,M)*FS*PP*RR*BB
        IF(K-1)45,45,131
    131 IF(TAT-12.)45,45,37
     37 AO(N,M)=AO(N,M)+S(N,M)*FS*PP*RR*BB
        GO TO 45
C       THERMOCLINE PRESENT
     40 IF(K-1)45,45,38
     38 SK=K
        TDK=SK/(3.+0.5*SK)
        BCF=0.001
        CCF=0.20
        RR=(R+0.1*D(N,M))/SQRT(D(N,M))
        FDD=(BCF*W(KW)+CCF/(D(N,M)**0.7))*TDK
        AO(N,M)=AO(N,M)+S(N,M)*FDD*PP*RR*BB
        IF(K-1)45,45,132
   132  IF(TAT-12.)45,45,44
     44 AO(N,M)=AO(N,M)+S(N,M)*FDD*PP*RR*BB
     45 CONTINUE
        GO TO 70
C       CONTINUOUS SOURCE (BLOWOUT)
     51 DO 65 N=1,NE
        DO 65 M=1,ME
        DIS=((M-MO)*0.001*DL)
```

```
      IF(DIS)53,53,54
   53 DIS=0.001
   54 APD=2.5
C     NO COMPUTATION IN IMMEDIATE AREA OF BLOWOUT
C     I.E.  2.5KM FROM THE SOURCE
      IF(DIS-APD)65,59,59
   59 IF(PLD(KP)-D(N,M))60,55,55
C     NO PYCNOCLINE
   55 SK=K
   57 STK=SK/(3.+0.2*SK)
   58 BWF=0.0016
      CDF=0.15
      RR=(R+0.1*D(N,M))/SQRT(D(N,M))
      DIFAC=(DIS+4.)/(20.+0.1*DIS)
      FS=(BWF*W(KW)+CDF/(D(N,M)**0.7))*STK*(DIFAC)
      AO(N,M)=AO(N,M)+S(N,M)*FS*PP*RR*BB
      IF(K-1)65,65,69
   69 IF(TAT-12.)65,65,71
   71 AO(N,M)=AO(N,M)+S(N,M)*FS*PP*RR*BB
      GO TO 65
C     COMPUTATION WITH THERMOCLINE PRESENT
   60 APD=2.5
      IF(DIS-APD)65,61,61
   61 SK=K
   62 STK=SK/(3.+0.5*SK)
   64 BWF=0.001
      CDF=0.20
      DDP=D(N,M)**0.74
      RR=(R+0.1*D(N,M))/SQRT(D(N,M))
      DIFAC=(DIS+4.)/(20.+0.1*DIS)
      FS=(BWF*W(KW)+CDF/DDP)*STK*(DIFAC)
      AO(N,M)=AO(N,M)+S(N,M)*FS*PP*RR*BB
      IF(K-1)65,65,66
   66 IF(TAT-12.)65,65,67
   67 AO(N,M)=AO(N,M)+S(N,M)*FS*PP*RR*BB
   65 CONTINUE
CXXXXXXXXXXXXXXXXXXX
   70 ALPHA=0.78
      CALL SILITA(AO,ALPHA)
CXXXXXXXXXXXXXXXXXXXXXXXXXX
      LU=3
      CALL PRIMFS(AO,T,UI,VI,DL,K,KA,KAL,BLO,LU)
CXXXXXXXXXXXXXXXXXXXXXXXXXX
  100 RETURN
      END
```

```
      SUBROUTINE SILITA (S,ALPHA)
      DIMENSION S(49,54)
      NE=49
      ME=54
      NEH=NE-1
      MEH=ME-1
      BET=(1.-ALPHA)/4
      DO 123 N=2,NEH
      DO 123 M=2,MEH
  103 IF(1-N)105,107,105
  105 VAUP=S(N-1,M)
      GO TO 108
  107 VAUP=S(N,M)
  108 IF(NE-N)110,112,110
  110 VALO=S(N+1,M)
      GO TO 113
  112 VALO=S(N,M)
  113 IF(1-M)115,116,115
  115 VALE=S(N,M-1)
      GO TO 117
  116 VALE=S(N,M)
  117 IF(ME-M)119,121,119
  119 VARI=S(N,M+1)
      GO TO 122
  121 VARI=S(N,M)
  122 S(N,M)=ALPHA*S(N,M)+BET*(VAUP+VALO+VALE+VARI)
  123 CONTINUE
      RETURN
      END
```

```
      SUBROUTINE PRIMFS(S,T,UI,VI,DL,K,KA,KAL,BLO,LU)
      DIMENSION S(49,54),IS(49,54)
      NE=49
      ME=54
C     IF(LU-1)202,401,420
      IF(LU-1)270,401,420
  202 PRINT 201,K,T,UI,VI,DL,KA,KAL
  201 FORMAT(1H1,5X,18HOIL CONCENTRATIONS,2X,2HK=,I5,3X,2HT=,F6.0,3X,3HU
     2I=,F6.4,3X,3HVI=,F6.4,3X,3HDL=,F6.0,3X,3HKA=,I3,3X,4HKAL=,I3)
  270 PRINT 271,K,DL
  271 FORMAT(1H1,5X,18HOIL CONCENTRATIONS,2X,2HK=,I5,3X,3HDL=,F6.0)
C     PRINT 203
      PRINT 504
  203 FORMAT(/5X,12HCONC.  IN PPB/)
  504 FORMAT(5X,19HPRINT FACTOR = 0.1,4X,7HPPB/10./)
CXXXXXXXXXXXXXXXXXXXXXXXX
      GO TO 212
  401 PRINT 402
  402 FORMAT(1H1,5X,16HDEPTHS IN METERS,)
      GO TO 320
  420 IF(LU-3)421,425,430
  421 PRINT 422,K
  422 FORMAT(1H1,5X,34HDECAY OF OIL ON THE BOTTOM, PERIOD,I5)
      GO TO 212
  425 PRINT 426,K
  426 FORMAT(1H1,5X,41HNEW OIL ON BOTTOM BEFORE ADVECTION,PERIOD,I5)
      GO TO 212
  430 IF(KAL-1)202,431,431
  431 PRINT 432,K
  432 FORMAT(1H1,5X,34HADVECTED OIL ON THE BOTTOM, PERIOD,I5)
      PRINT 272,UI,VI
  272 FORMAT(5X,3HUI=,F5.2,3X,3HVI=,F5.2)
      GO TO 212
CXXXXXXXXXXXXXXXXXXXXXXXX
      IF(KA-1)210,210,215
  210 PRINT 211
  211 FORMAT(5X,12HLAMINAR,FLOW/)
      GO TO 212
  215 PRINT 216
  216 FORMAT(5X,26HLAYER THICKNESS INCREASING/)
CXXXXXXXXXXXXXXXXXXXXXXXX
C 212 IF(KAL-1)230,220,220
  212 IF(KAL-1)530,220,220
  220 IF(BLO-1)250,250,252
  250 DO 225 N=1,NE
      DO 225 M=1,ME
      IS(N,M)=S(N,M)*1000.
  225 CONTINUE
      PRINT 260
  260 FORMAT(5X,16HPRINT FACTOR = 1/)
      GO TO 240
  252 DO 253 N=1,NE
      DO 253 M=1,ME
      IS(N,M)=S(N,M)*100.
  253 CONTINUE
      PRINT 261
  261 FORMAT(5X,18HPRINT FACTOR = 0.1,4X,7HPPB/10./)
      GO TO 240
```

```
  320 DO 321 N=1,NE
      DO 321 M=1,ME
      IS(N,M)=S(N,M)
  321 CONTINUE
      GO TO 240
  230 DO 205 N=1,NE
      DO 205 M=1,ME
      IS(N,M)=S(N,M)*1000.
  205 CONTINUE
  530 DO 531 N=1,NE
      DO 531 M=1,ME
      IS(N,M)=S(N,M)*100.
  531 CONTINUE
  240 PRINT 206,(N,N=1,40)
  206 FORMAT(/4X,40I3)
      PRINT 207,(N,(IS(N,M),M=1,40),N=1,49)
  207 FORMAT(/1X,I2,1X,40I3)
C    XXXXXXXXXXXXXXXXXXX
      GO TO 300
C    XXXXXXXXXXXXXXXXXXXX
      PRINT 208,(N,N=41,54)
  208 FORMAT(1H1,//4X,14I3)
      PRINT 209,(N,(IS(N,M),M=41,54),N=1,49)
  209 FORMAT(/1X,I2,1X,14I3)
C     XXXXXX
      GO TO 300
C     XXXXXX
      PRINT 208,(N,N=81,120)
      PRINT 207,(N,(IS(N,M),M=81,120),N=1,100)
  300 RETURN
      END
```

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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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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.)

Figure 16a. 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.) 48 hrs from start of spill;
Figure 16b. 120 hrs from start of spill.
Figure 16c. 240 hrs from start of spill.
Figure 16d. 480 hrs from start of spill.

Figure 17a. 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.) 48 hrs from start of spill.
Figure 17b. 120 hrs from start of spill.
Figure 17c. 240 hrs from start of spill.

Figure 18a. 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.) 24 hrs from start of spill.
Figure 18b. 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.

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.

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).

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.

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	

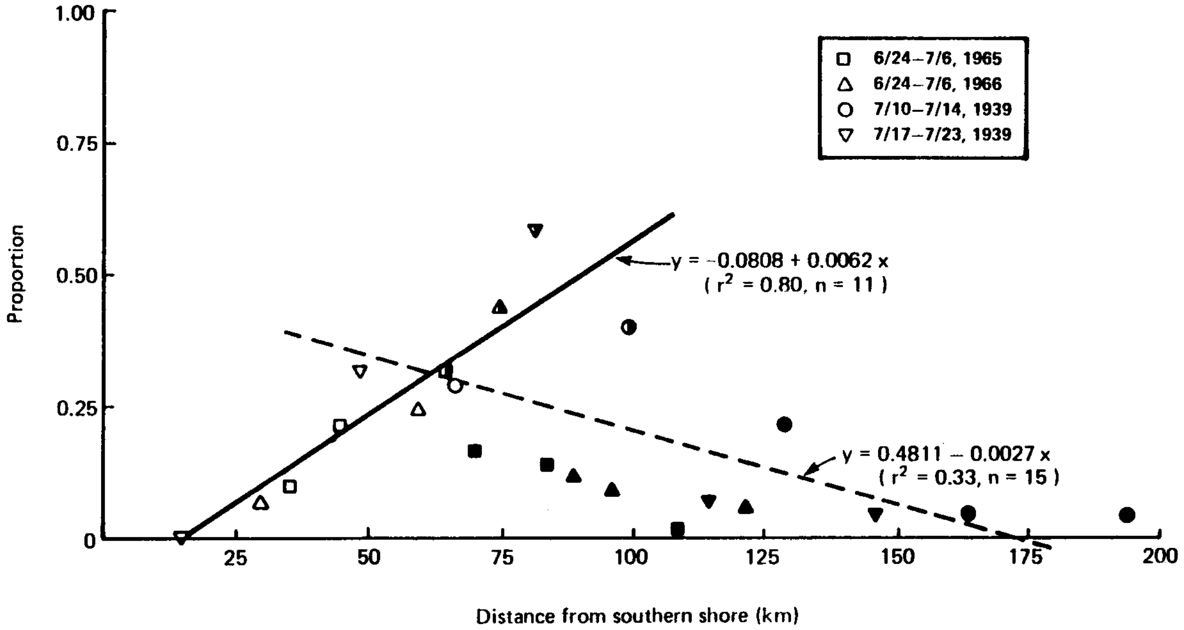

Figure 1.--Proportions of total sockeye salmon caught in exploratory gill net surveys in Bristol Bay, that were caught at varying distances from shore. Data from Straty (1975).

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

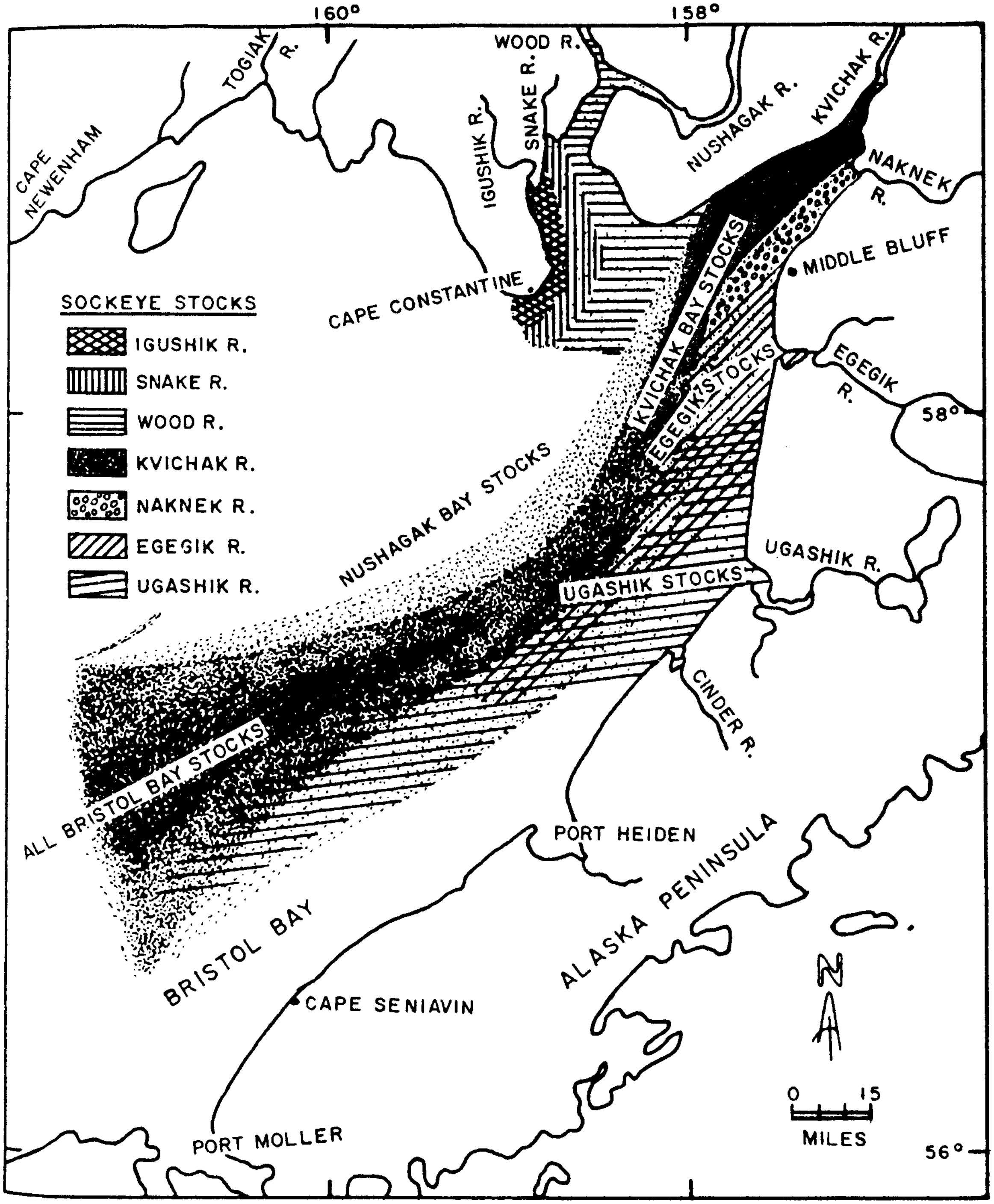

Figure 2.--Migration pathways of adult sockeye salmon returning to Bristol Bay. (Figure 22 from Straty (1975), reproduced with permission of author.)

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

			Intercept	Slope	
River	Age	Number	A	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 recapture[1]	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		
		P. Moller	290	5/27-30	7/19	52	5.6	6.5	2	11.5	0.6	0.7	
		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		
		P. Heiden	225	6/14-7/1	8/11	50	4.5	5.2	1	8.3	0.6		0.9
		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		
		P. Moller	290	5/17-6/5	6/23	27	10.7	12.4	2	12.5	1.0	0.9	
		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		
		P. Moller	360	6/6-6/22	8/16	63	5.7	6.6	1	8.3	0.8	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.

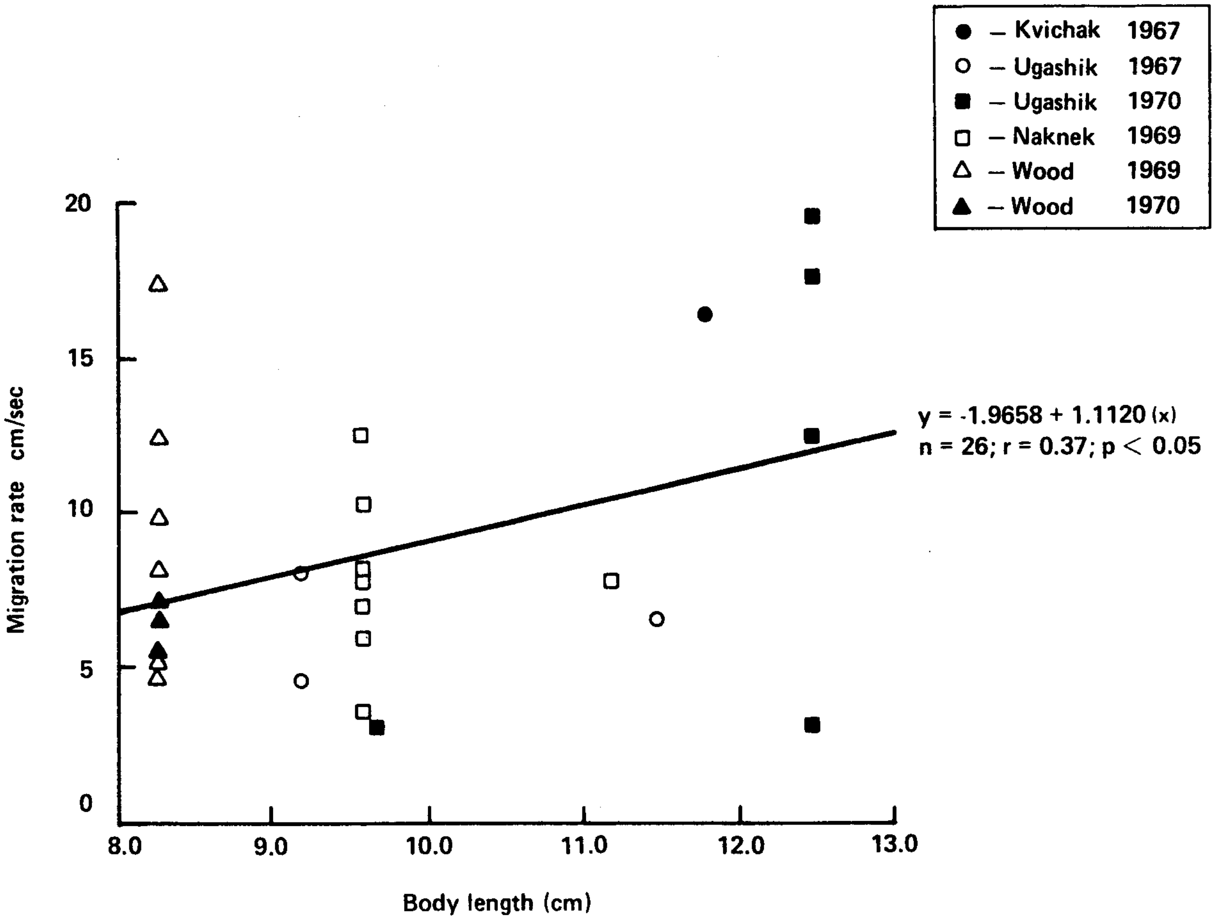

Figure 3.--Average migration rates of marked juvenile sockeye salmon from river to recapture in Bristol Bay. (Data from Straty 1974.)

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

[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.

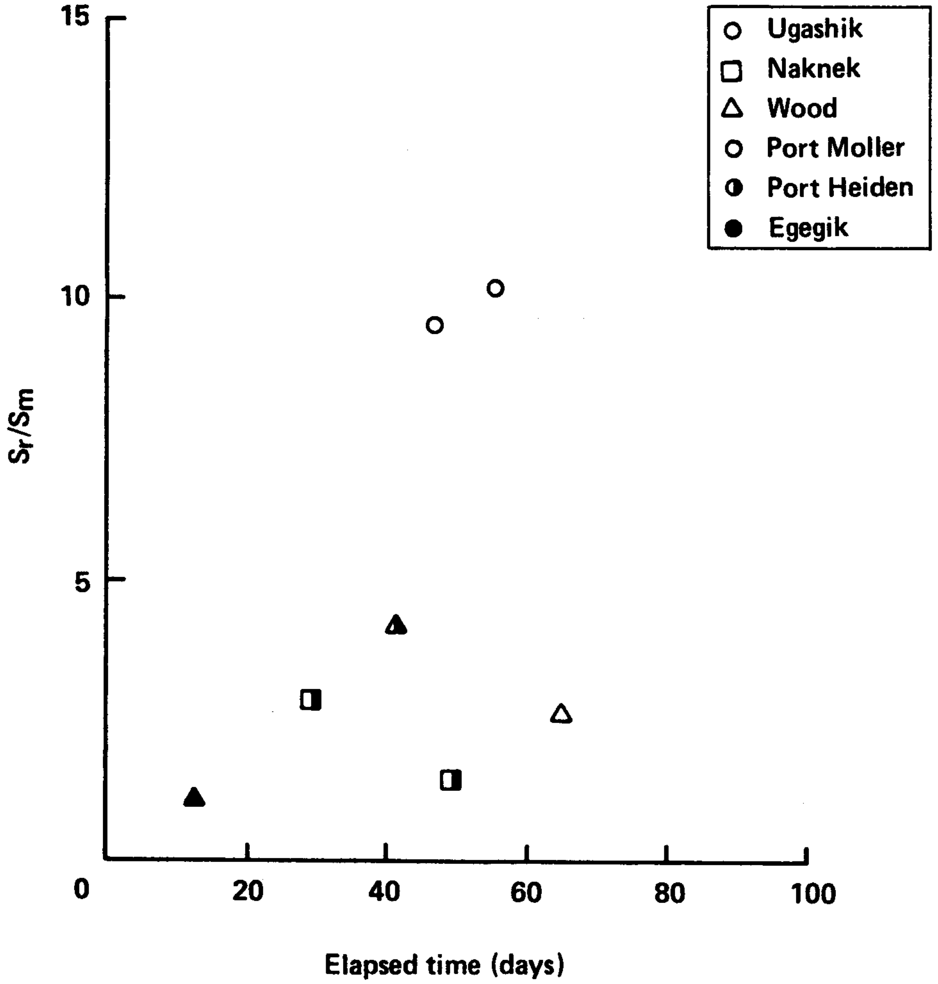

Figure 4.--Ratio of standard deviation of recaptures to standard deviation of releases for marked juvenile sockeye salmon released into rivers leading to Bristol Bay. (Data from Straty 1974. Symbols denote river of release; shading point of recapture.)

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**

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[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.

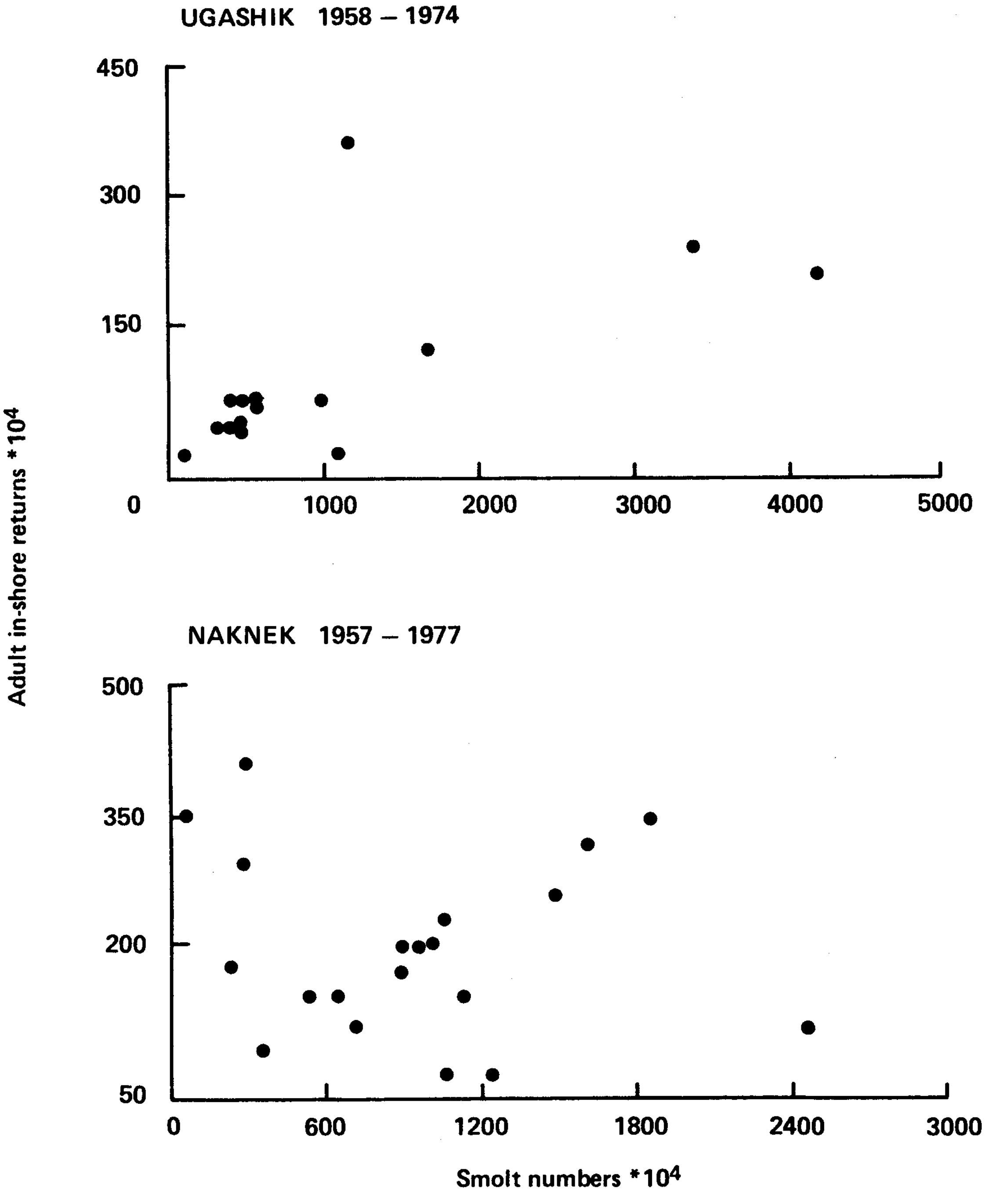
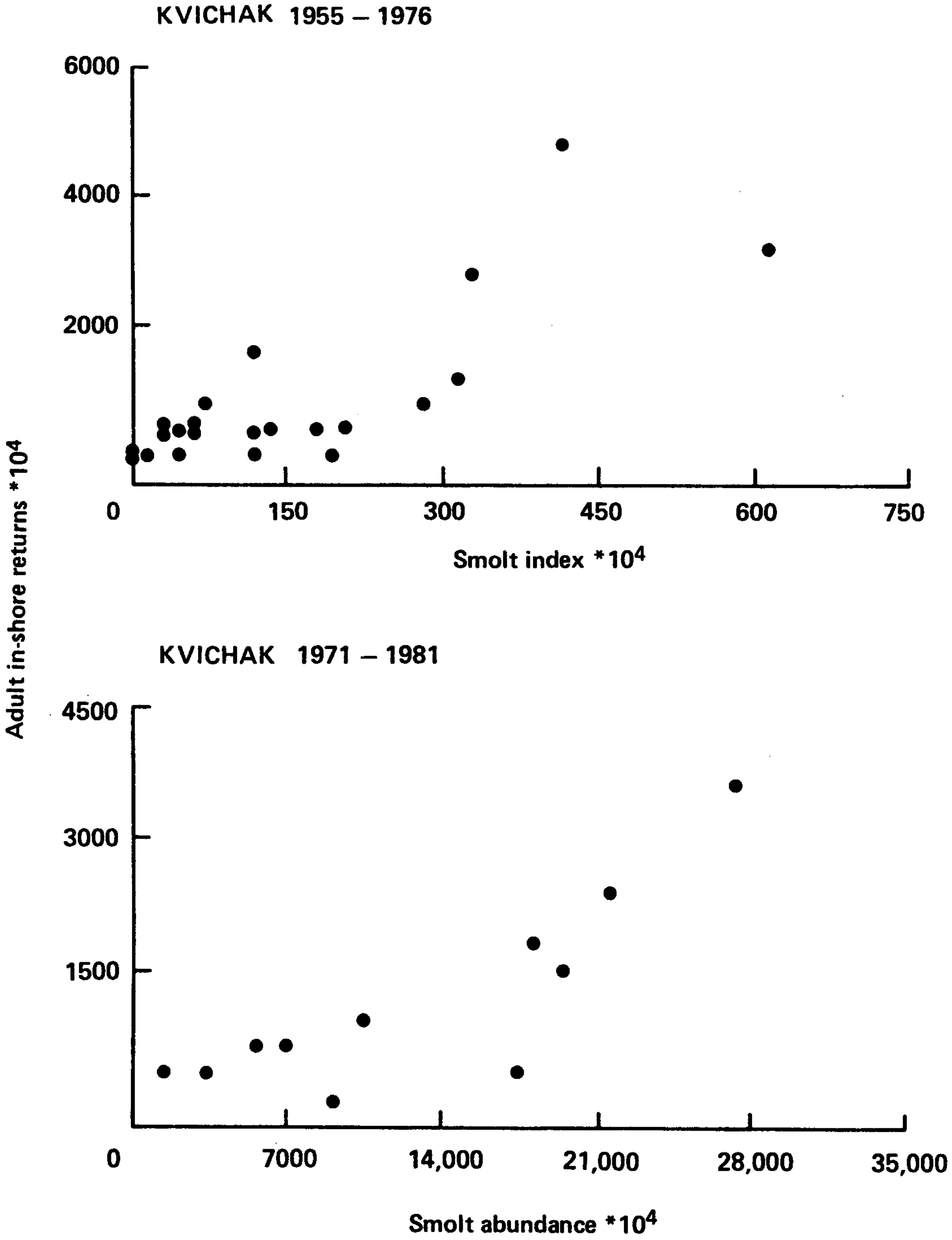


Figure 5.--Returns of adult Bristol Bay sockeye salmon and their estimated numbers (or index) as smolts outmigrating from the rivers. (Data sources detailed in text.)

Figure 5 continued



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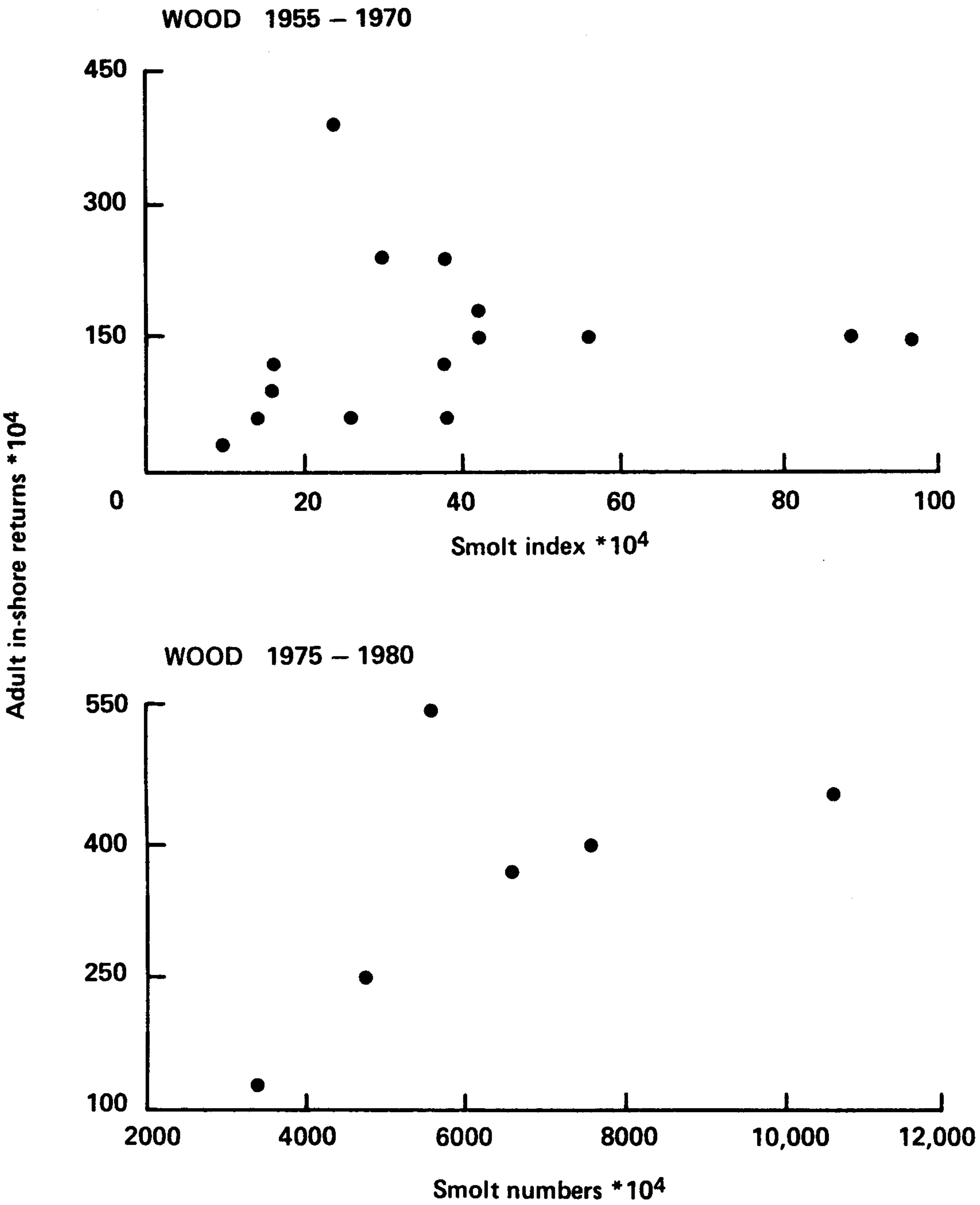

Table 6. Relation between total abundance (or index of abundance) of smolts migrating for the Wood and Naknek river systems and the resulting total abundance of adults, taking the abundance of Kvichak River smolts or adults into consideration.

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	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
		SW	GC				1.0 (0.8-1.4)	8	
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

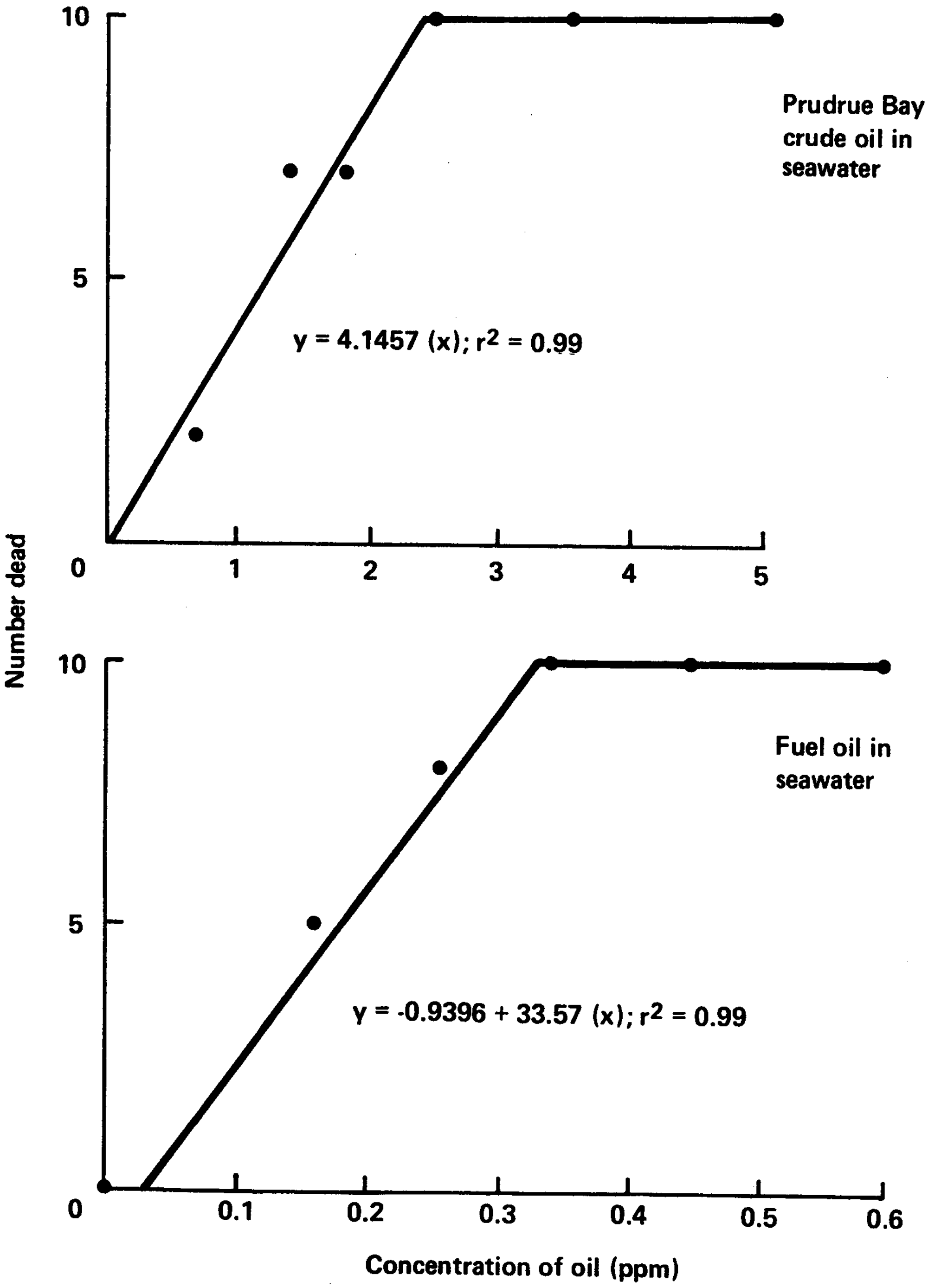

Figure 6.--Number of sockeye salmon smolts dying (out of 10) following exposure to oil in static seawater tests for 96 hr. Data from Adam Moles (NOAA).

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 ét 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 (<u>Mugil</u> <u>cephalus</u>) 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).

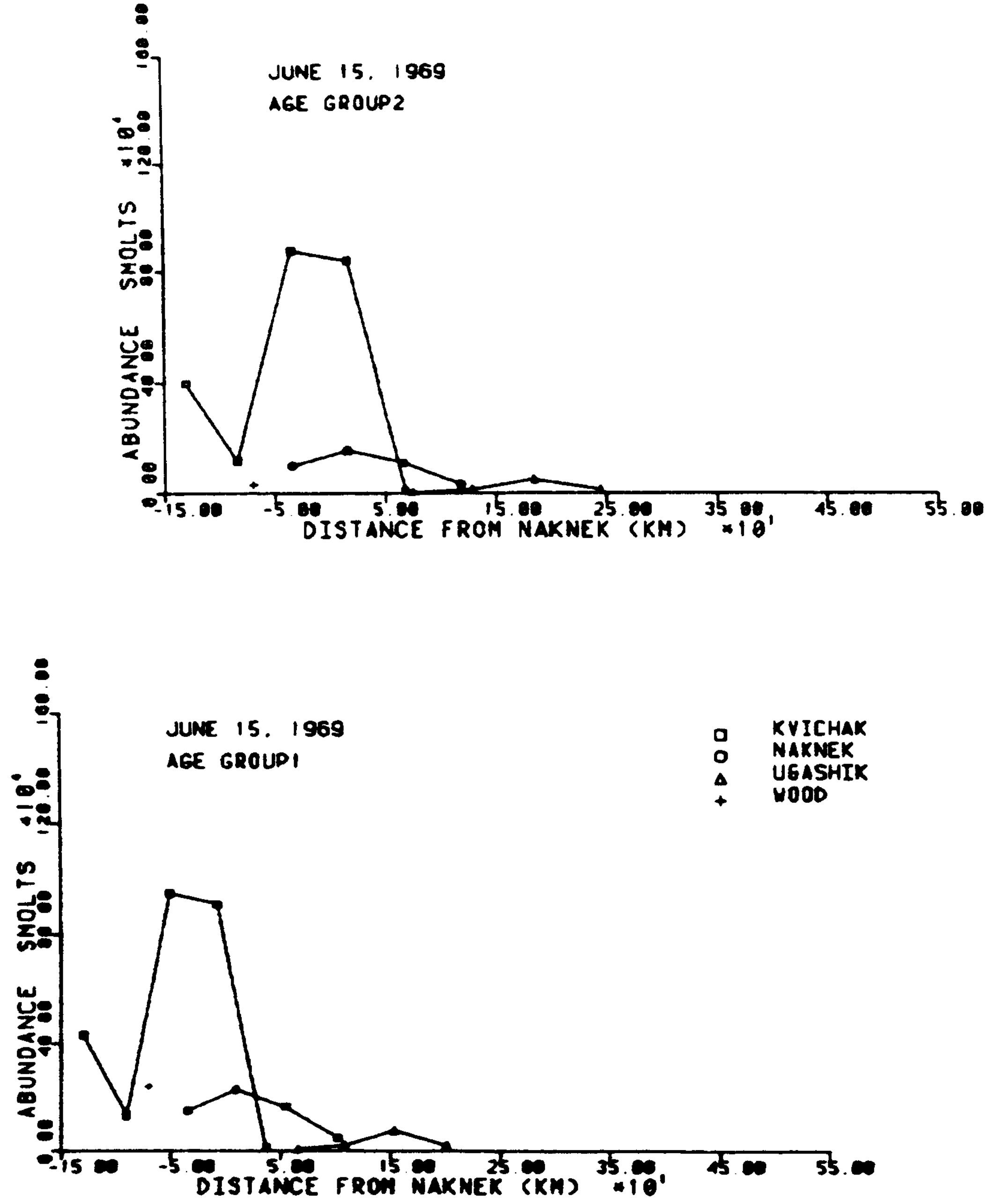

Figure 7.--Simulated abundances of juvenile sockeye salmon migrating through Bristol Bay. (No allowance made for mortality.)

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.

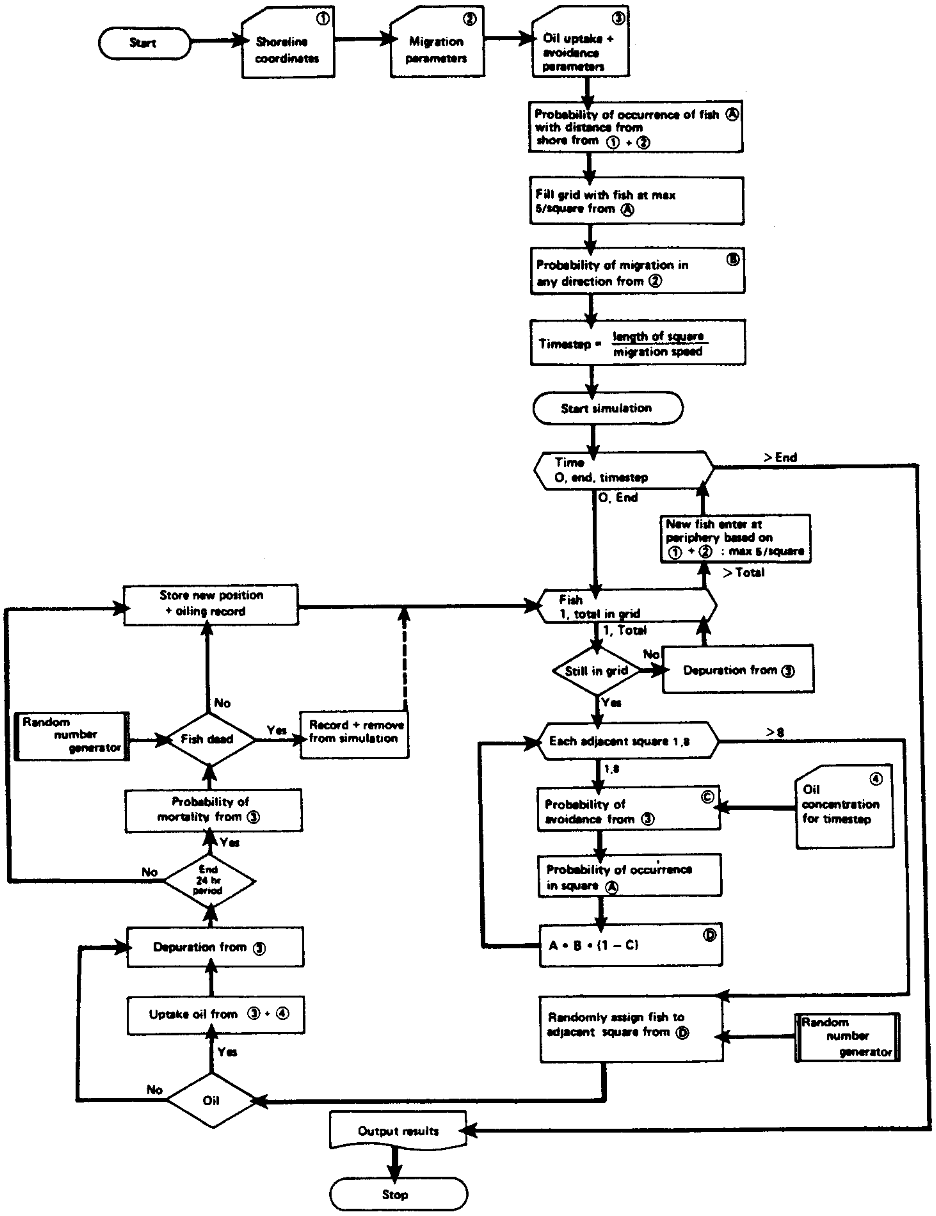
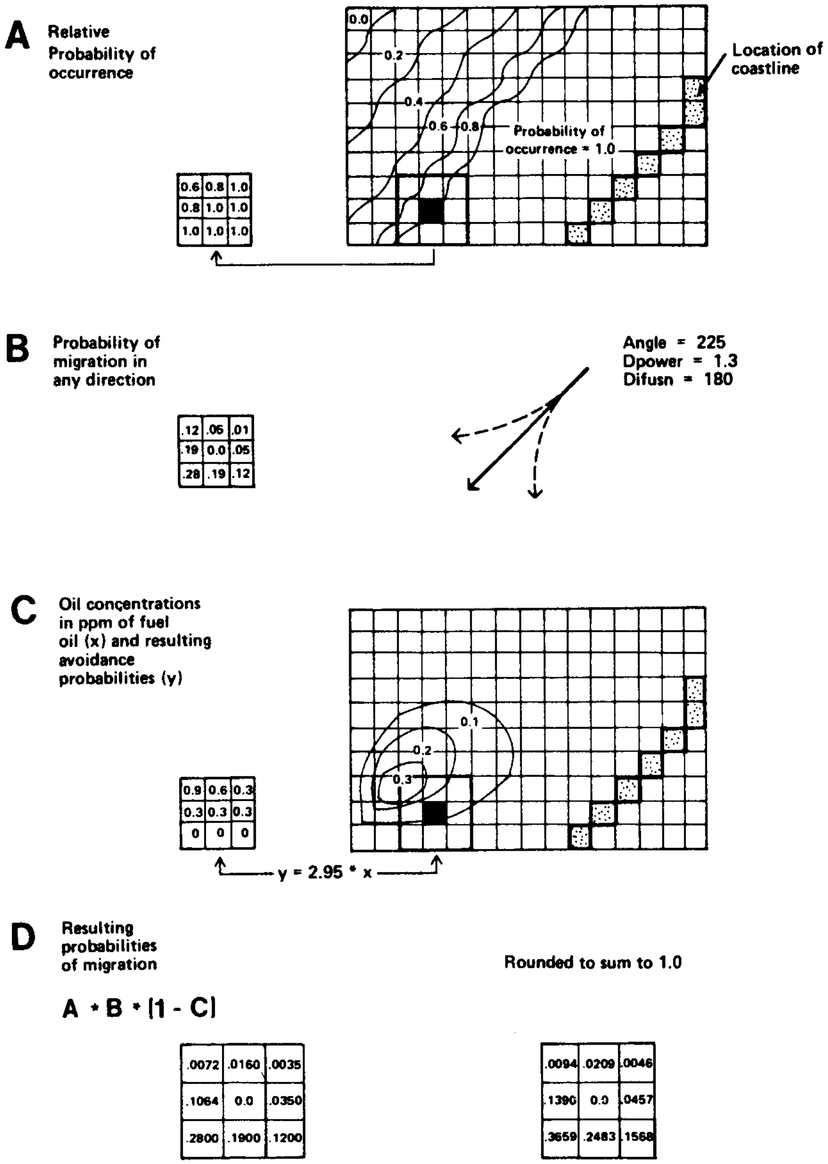


Figure 8.--Flow chart of processes involved in simulating the movement of sockeye salmon through an oil spill in Bristol Bay.



0.6	0.8	1.0
0.8	1.0	1.0
1.0	1.0	1.0

.12	.05	.01
.19	0.0	.05
.28	.19	.12

0.9	0.6	0.3
0.3	0.3	0.3
0	0	0

.0072	.0160	.0035
.1064	0.0	.0350
.2800	.1900	.1200

.0094	.0209	.0046
.1390	0.0	.0457
.3659	.2483	.1568

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:

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

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

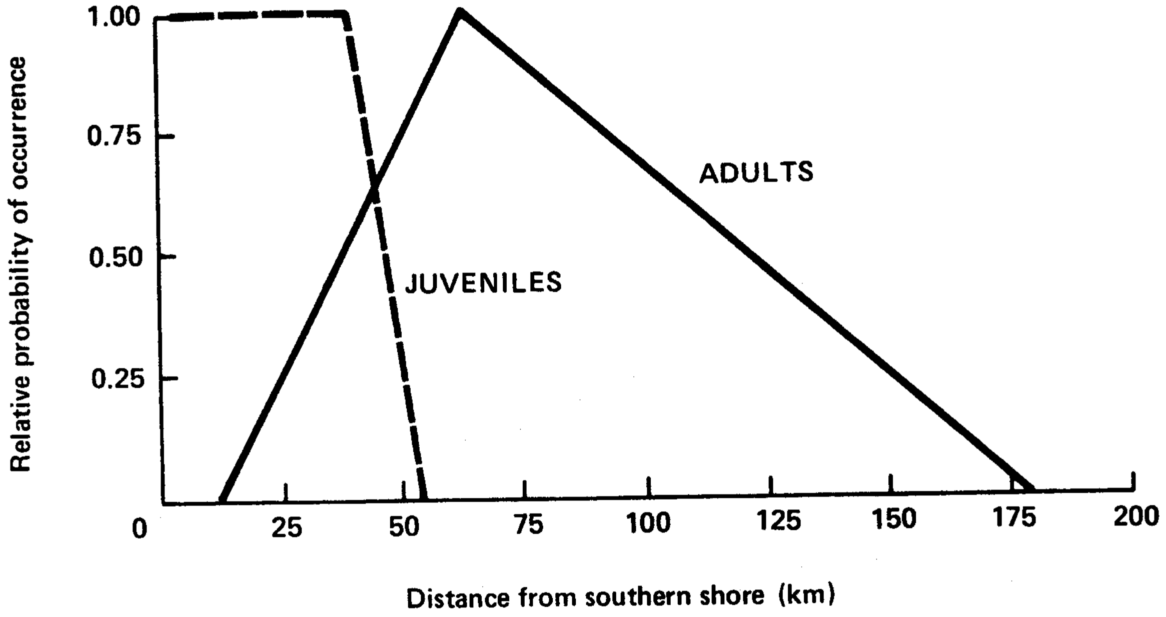


Figure 10.--Functions describing the relative possibilities of occurrence of sockeye salmon with distance from shore between Port Heiden and Port Moller in Bristol Bay.

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

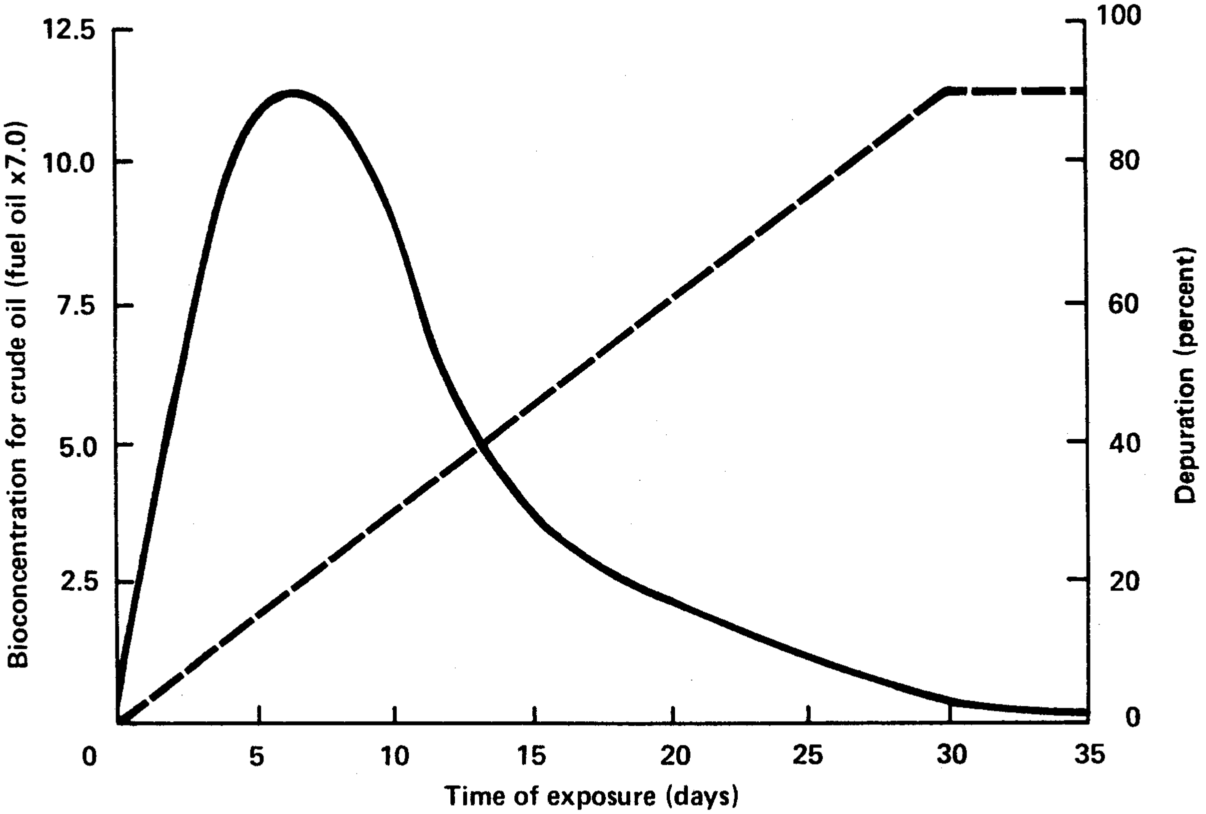


Figure 11.--Functions describing the bioconcentration and depuration of crude oil in the tissues of salmon exposed to a naturally weathering oil spill. (Bioconcentration factors for No. 2 diesel oil are 7 times higher than those for crude oil.)

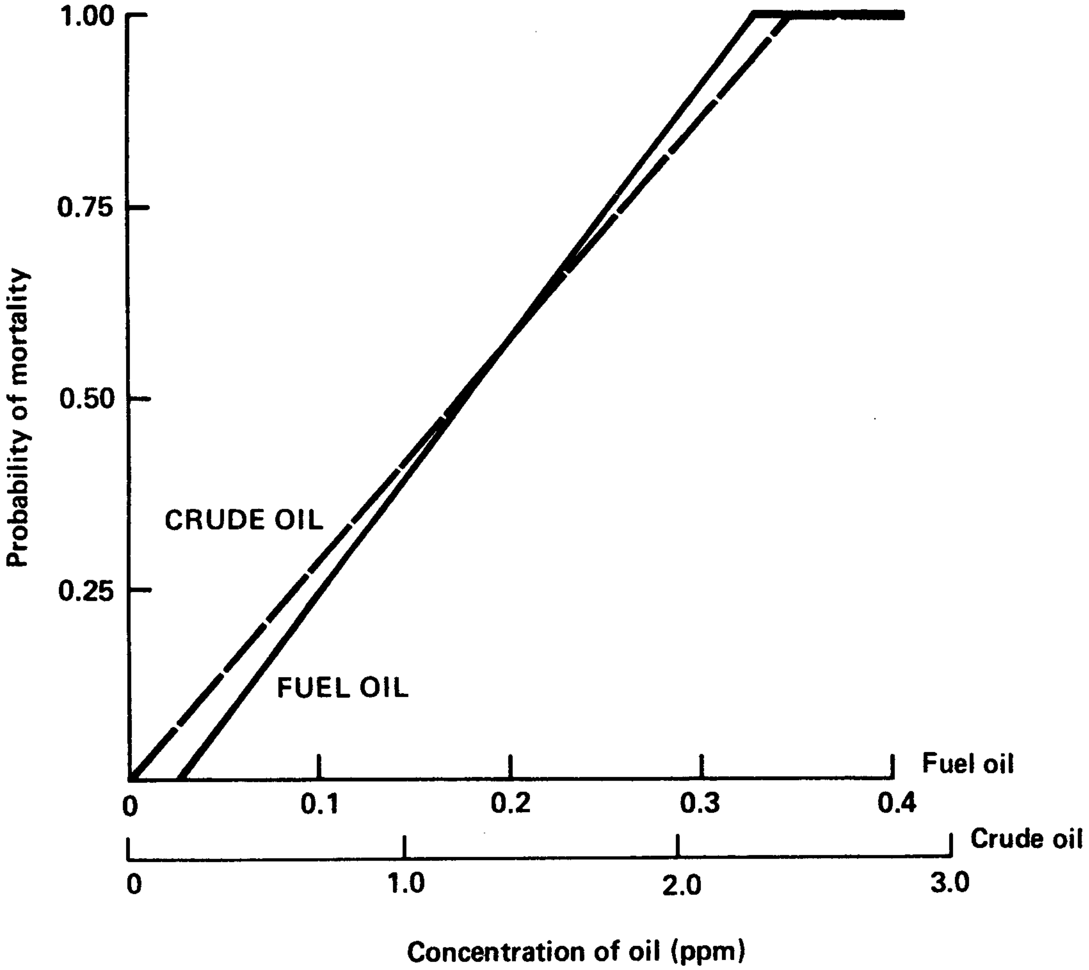


Figure 12.--Functions describing the probabilities of mortality of sockeye salmon exposed to oil in the marine environment over a 24 hr period.

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

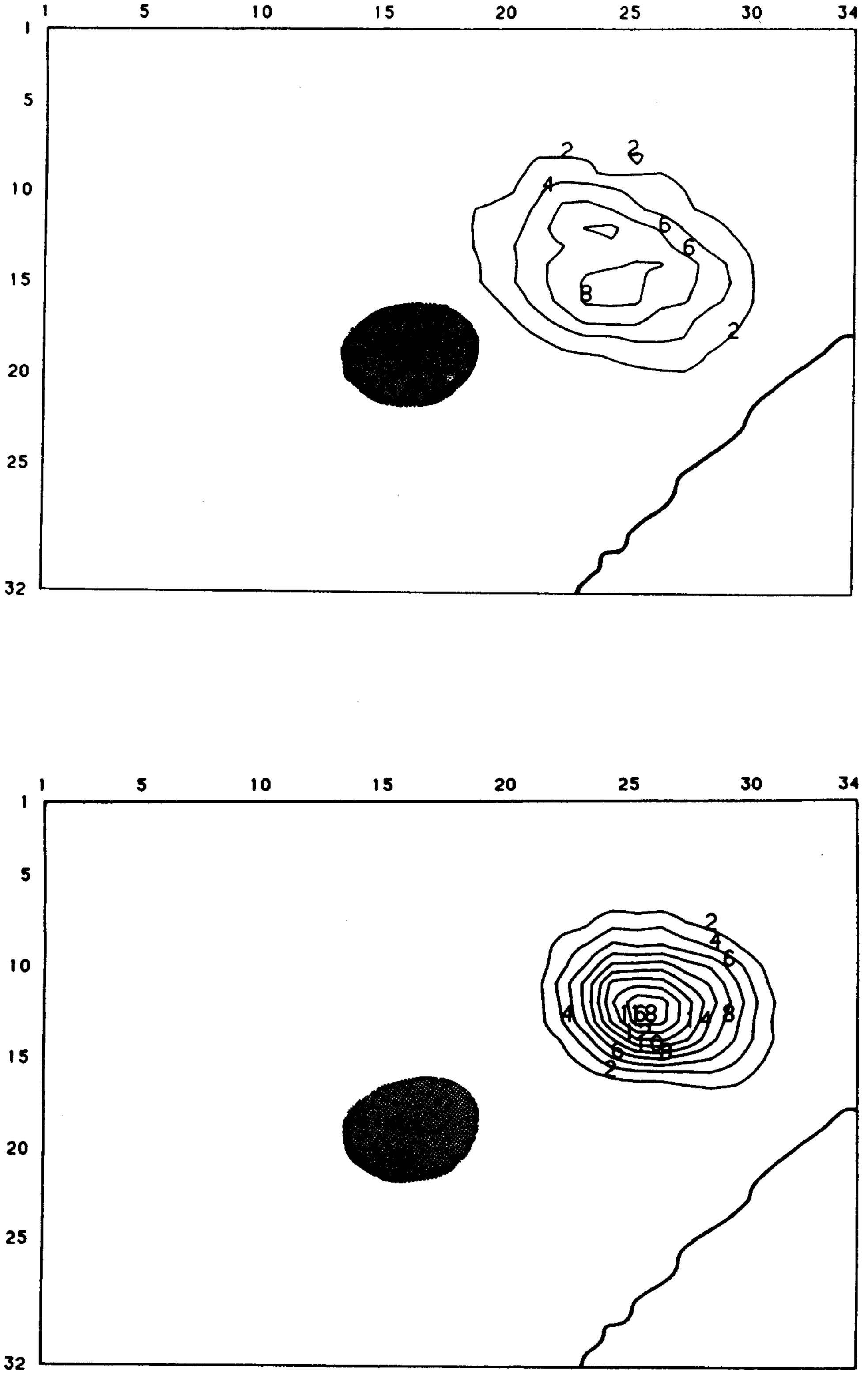


Figure 13.--Simulation of the movement of a concise distribution of juvenile sockeye salmon towards and through a stationary spill of fuel oil in Bristol Bay under the assumptions of avoidance (top) and direct migration (bottom).

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.

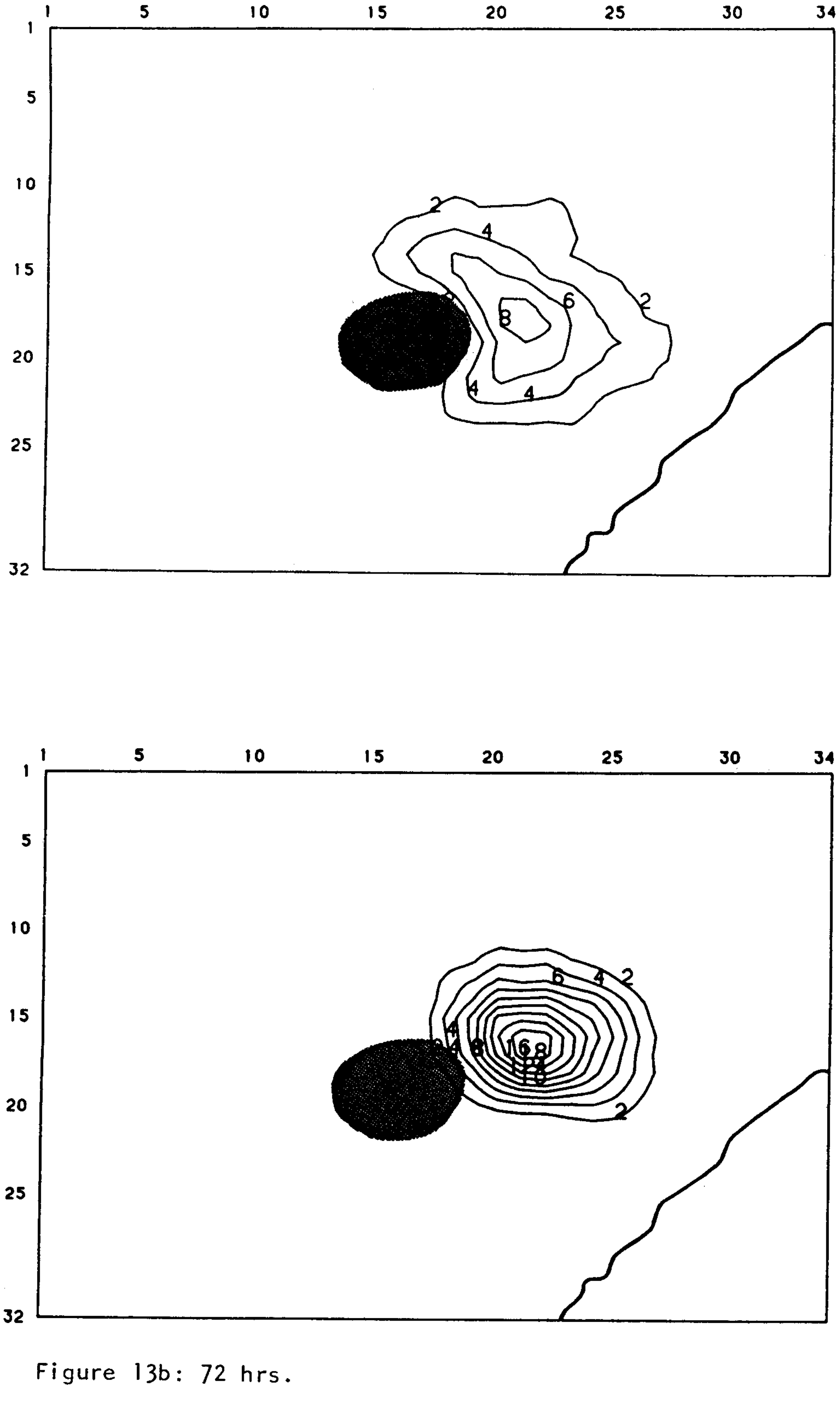


Figure 13b: 72 hrs.

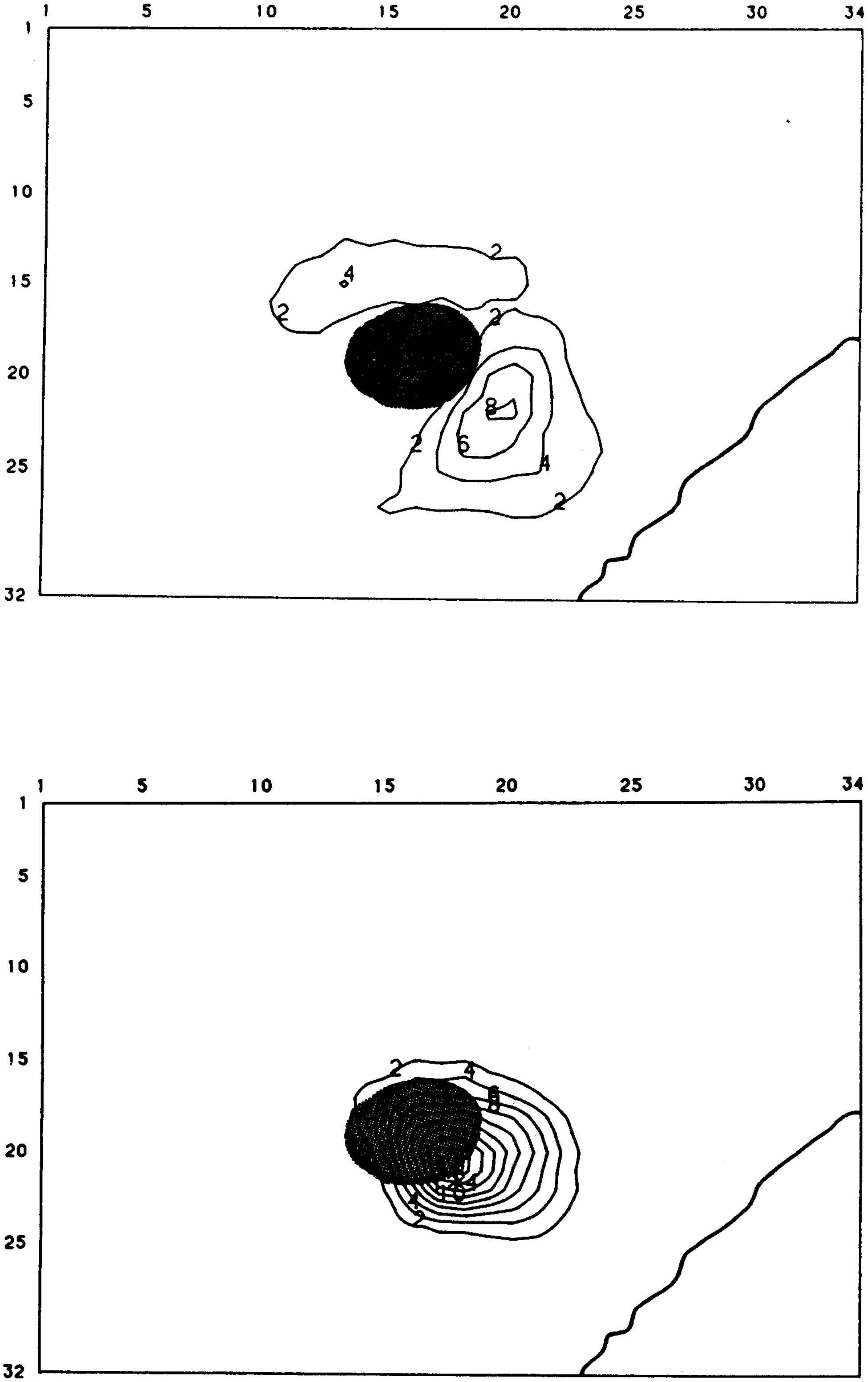


Figure 13c: 96 hrs.

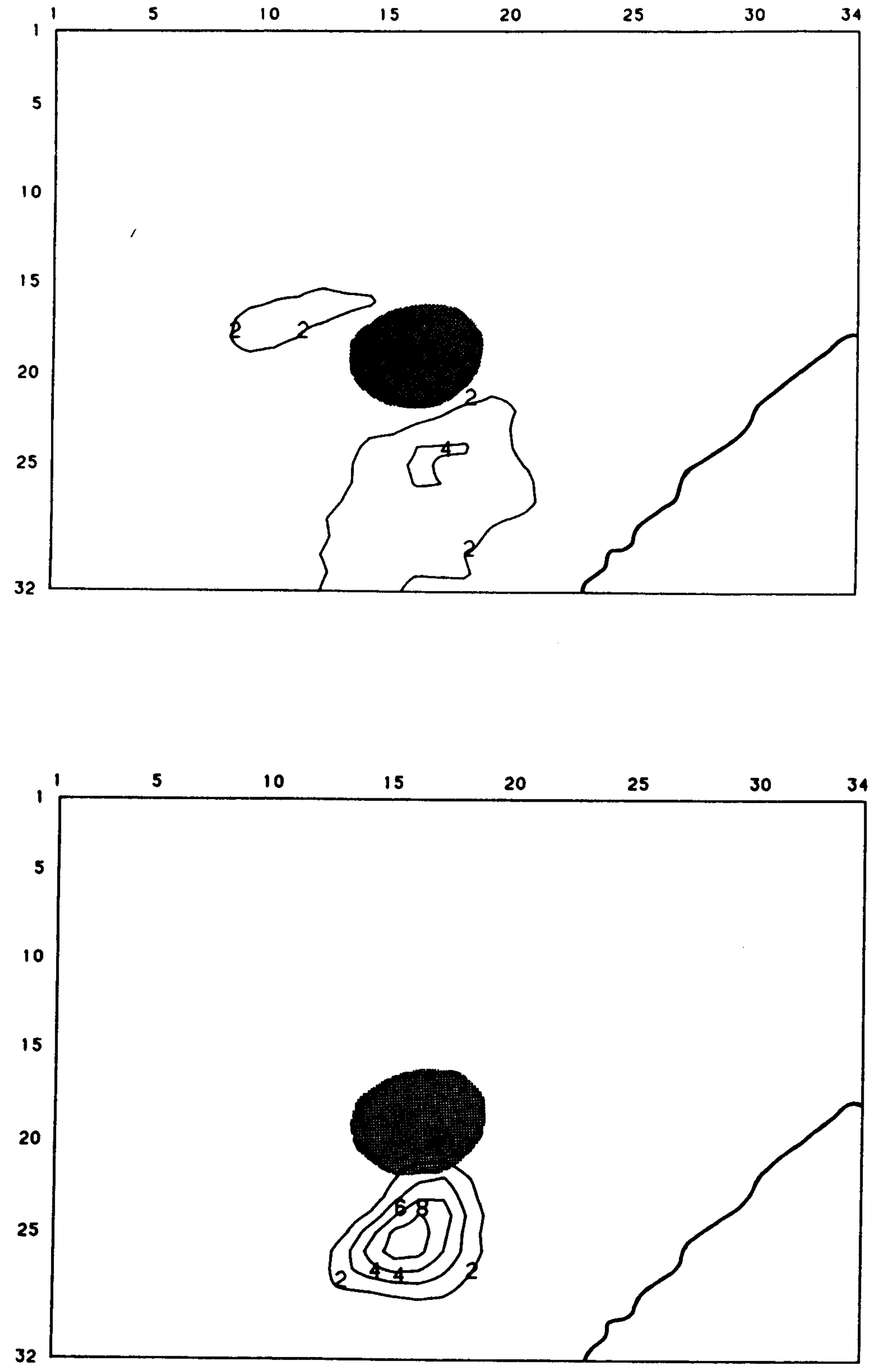


Figure 13d: 120 hrs.

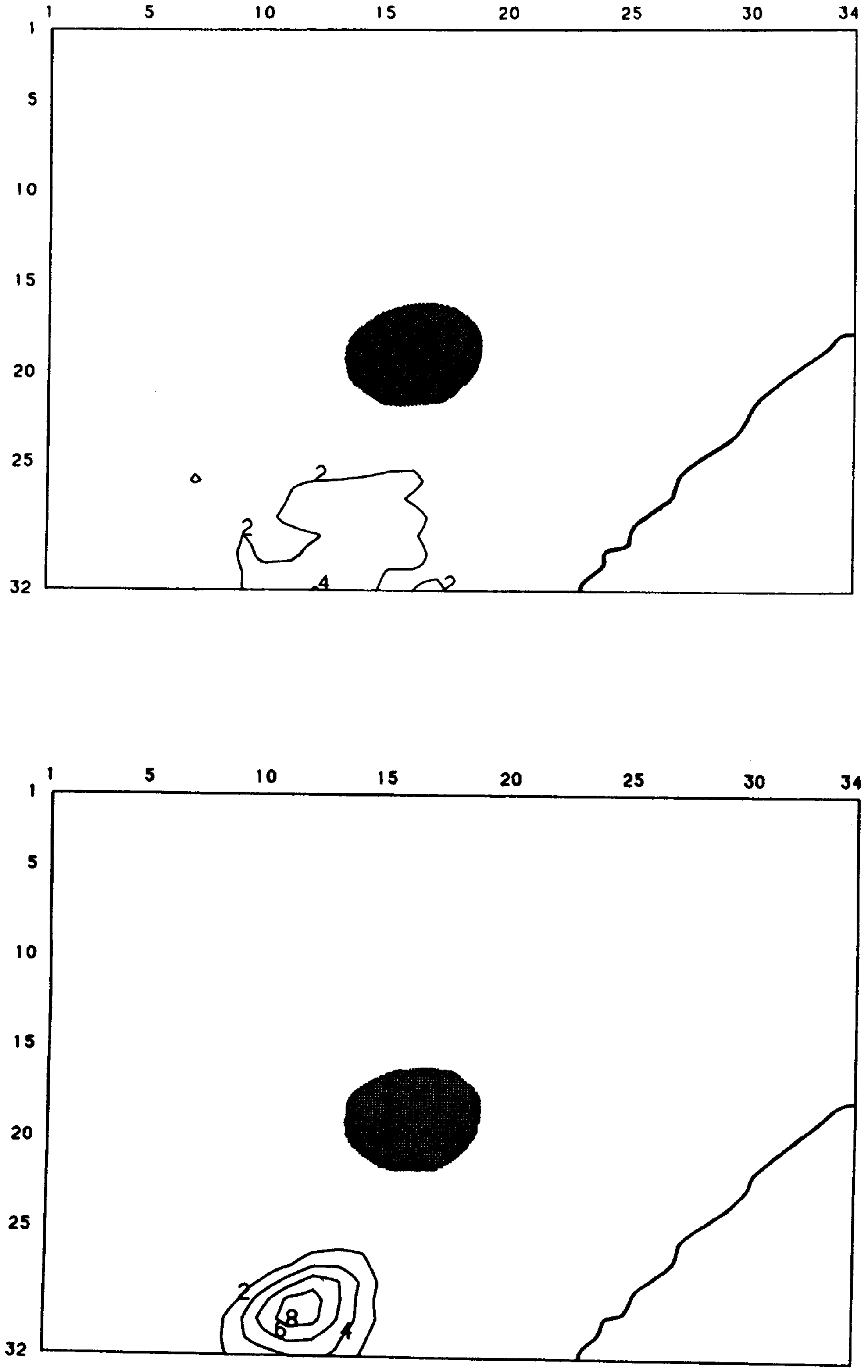


Figure 13e: 144 hrs.

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.

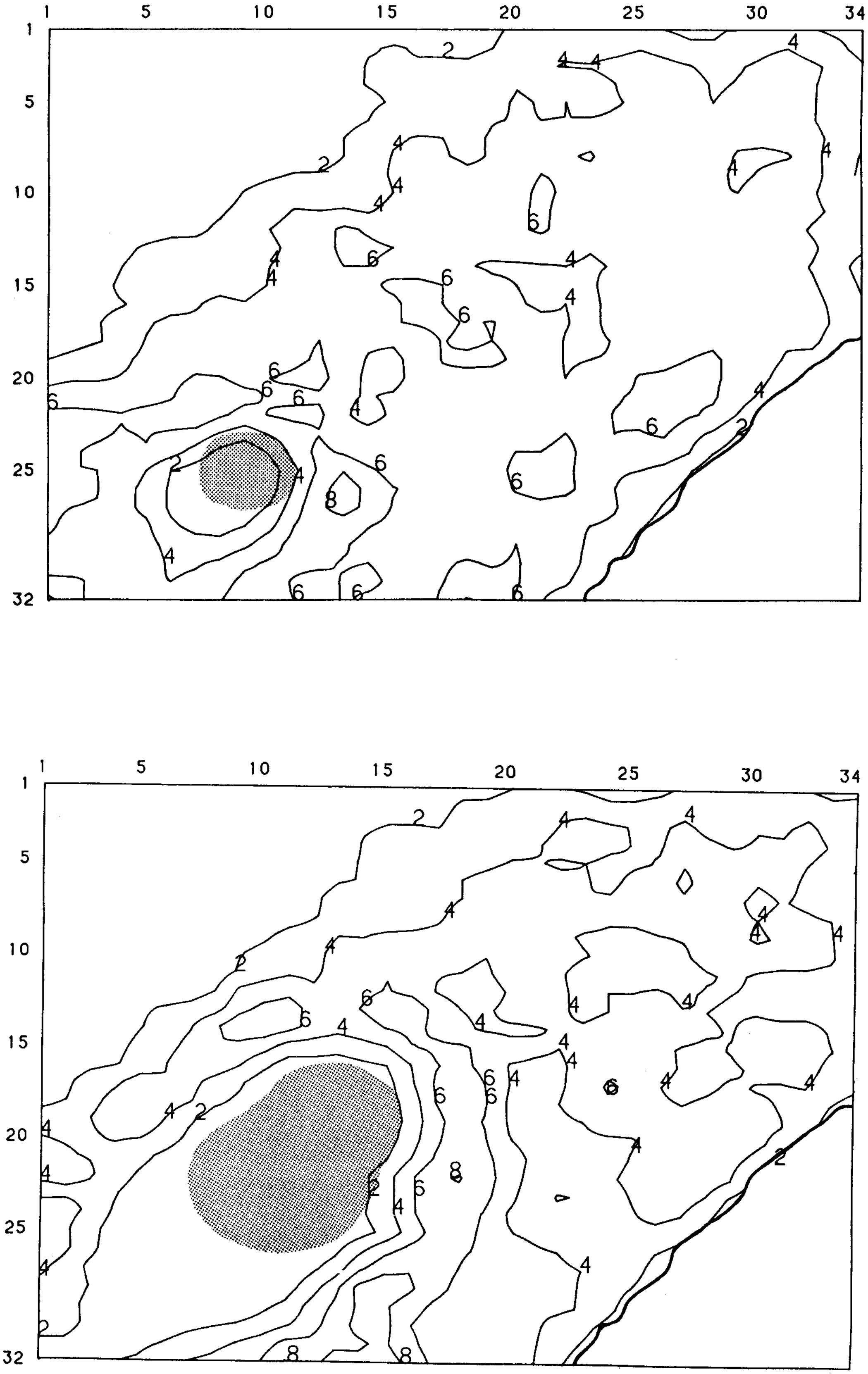


Figure 14.--Simulation of migration of juvenile sockeye salmon and their avoidance of a spill of 240,000 bbl of No. 2 diesel oil off of Port Heiden, Bristol Bay. (Shaded area depicts concentrations of oil greater than 1 ppm.) a) top: 48 hrs from start of spill; b) bottom: 120 hrs from start of spill.

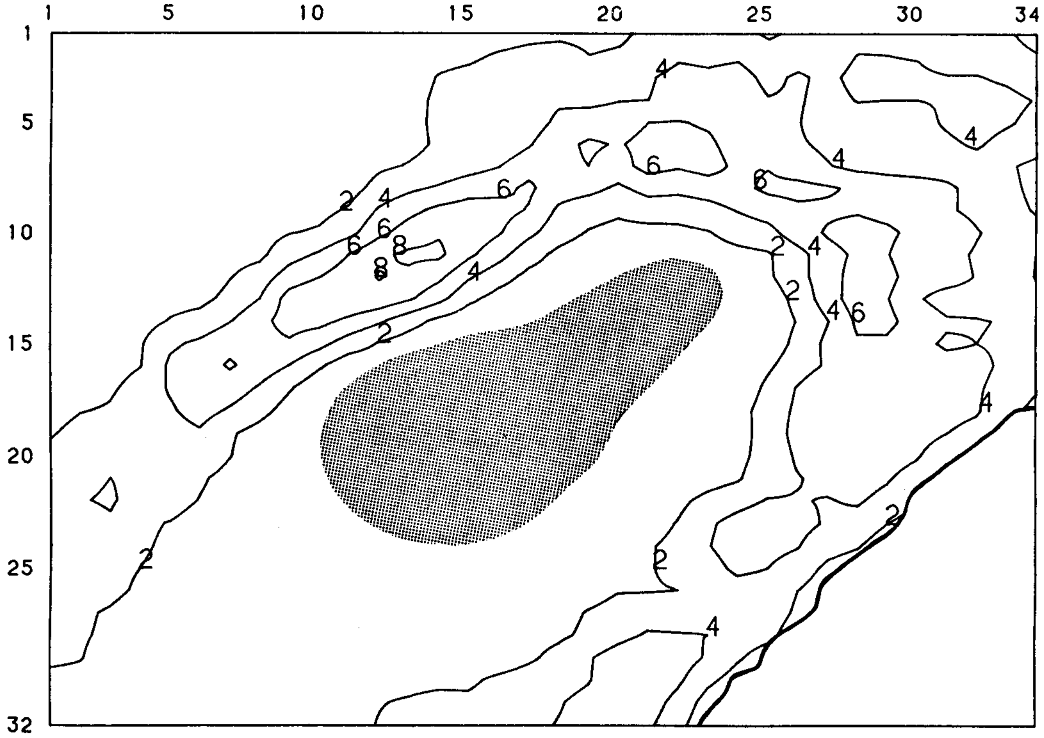


Figure 14c: 240 hrs from start of spill.

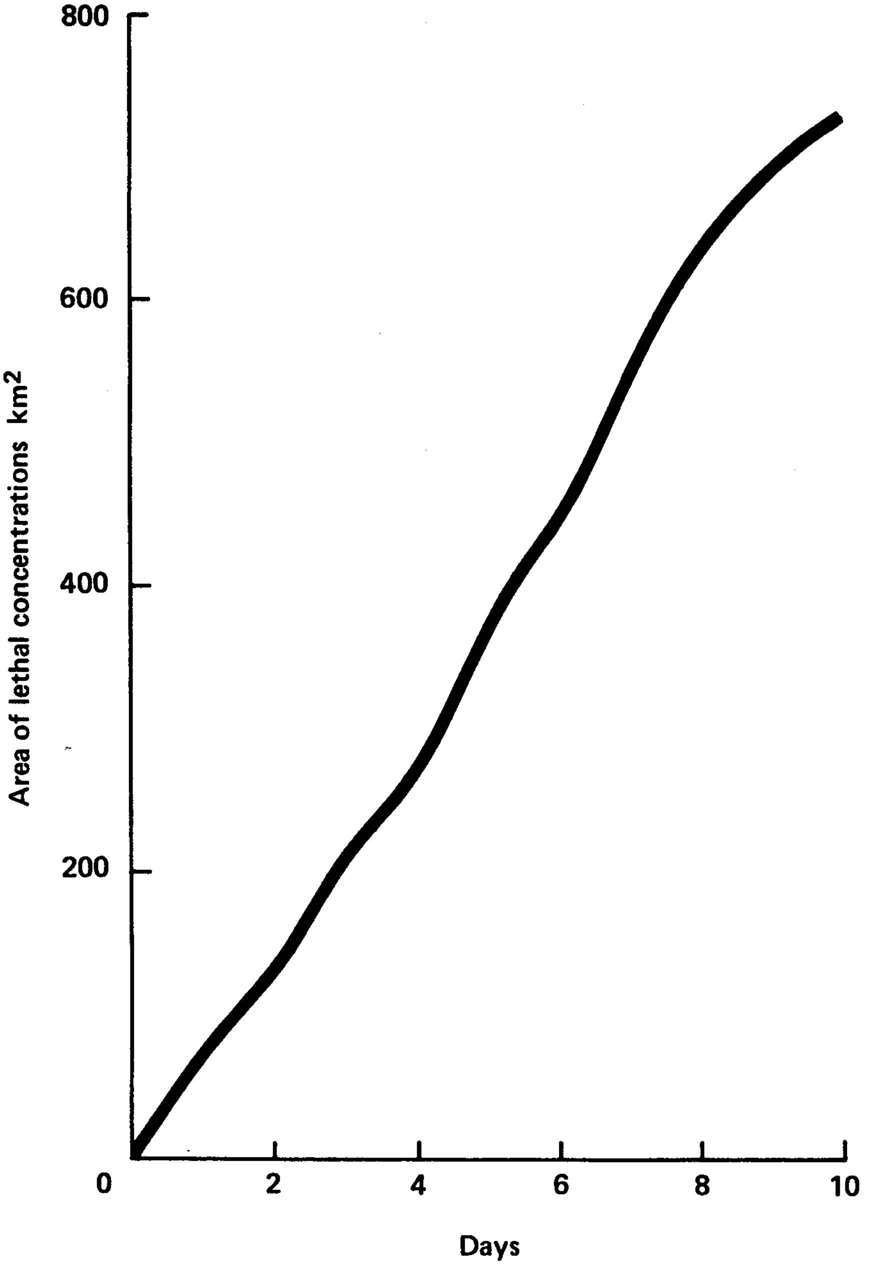


Figure 15.--Areal coverage of concentrations of fuel oil lethal to sockeye salmon after 24 hrs (>0.45 ppm) resulting from a simulated spill of 240,000 bbl of No. 2 diesel oil off Port Heiden or Port Moller in Bristol Bay. (Data from Rand Corporation.)

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

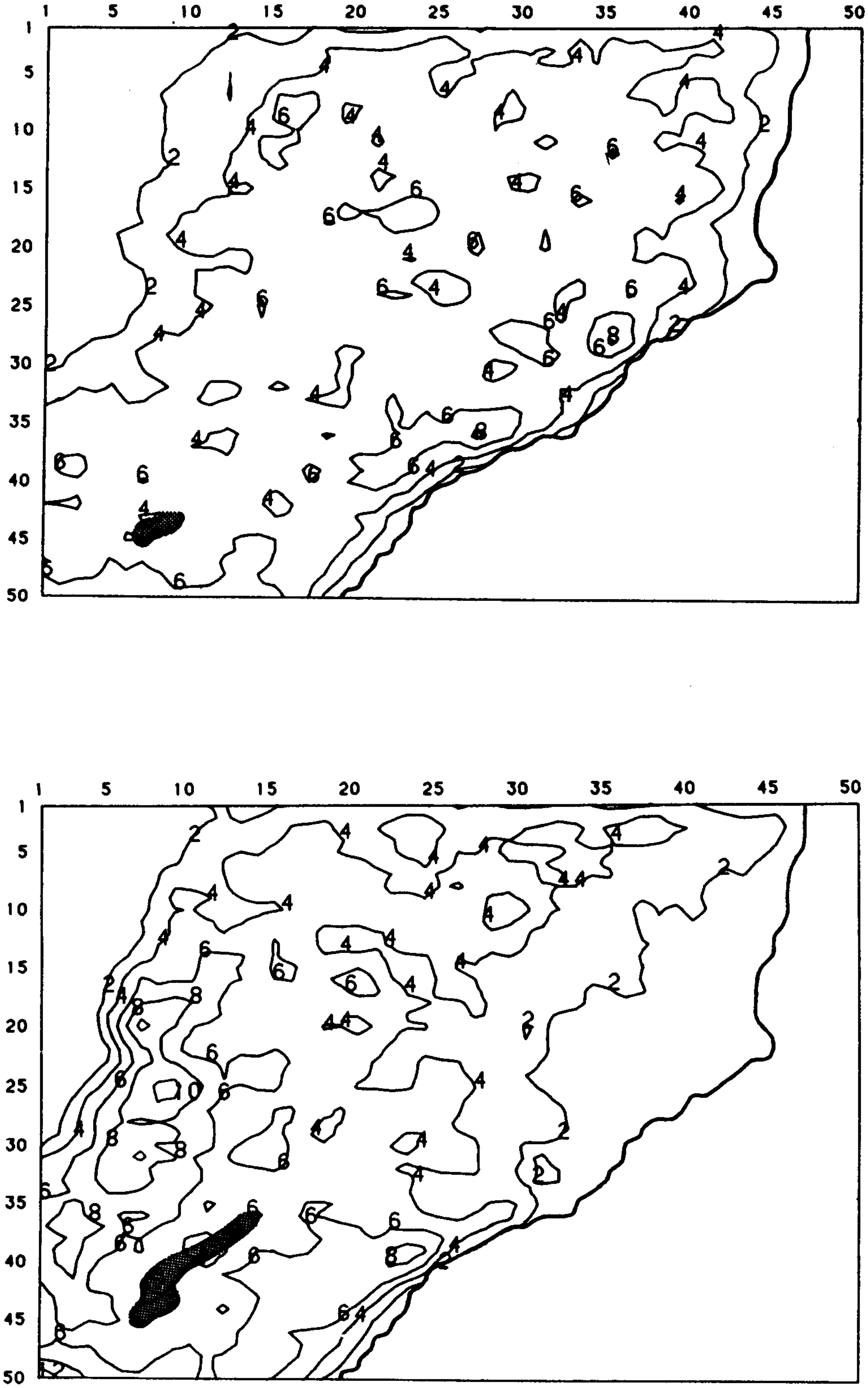


Figure 16.--Simulation of migration of juvenile sockeye salmon and their avoidance of a blowout of 20,000 bbl/day off of Port Heiden, Bristol Bay. (Shaded area depicts concentrations of oil greater than 0.1 ppm.) a) top: 48 hrs from start of spill; b) bottom: 120 hrs from start of spill.

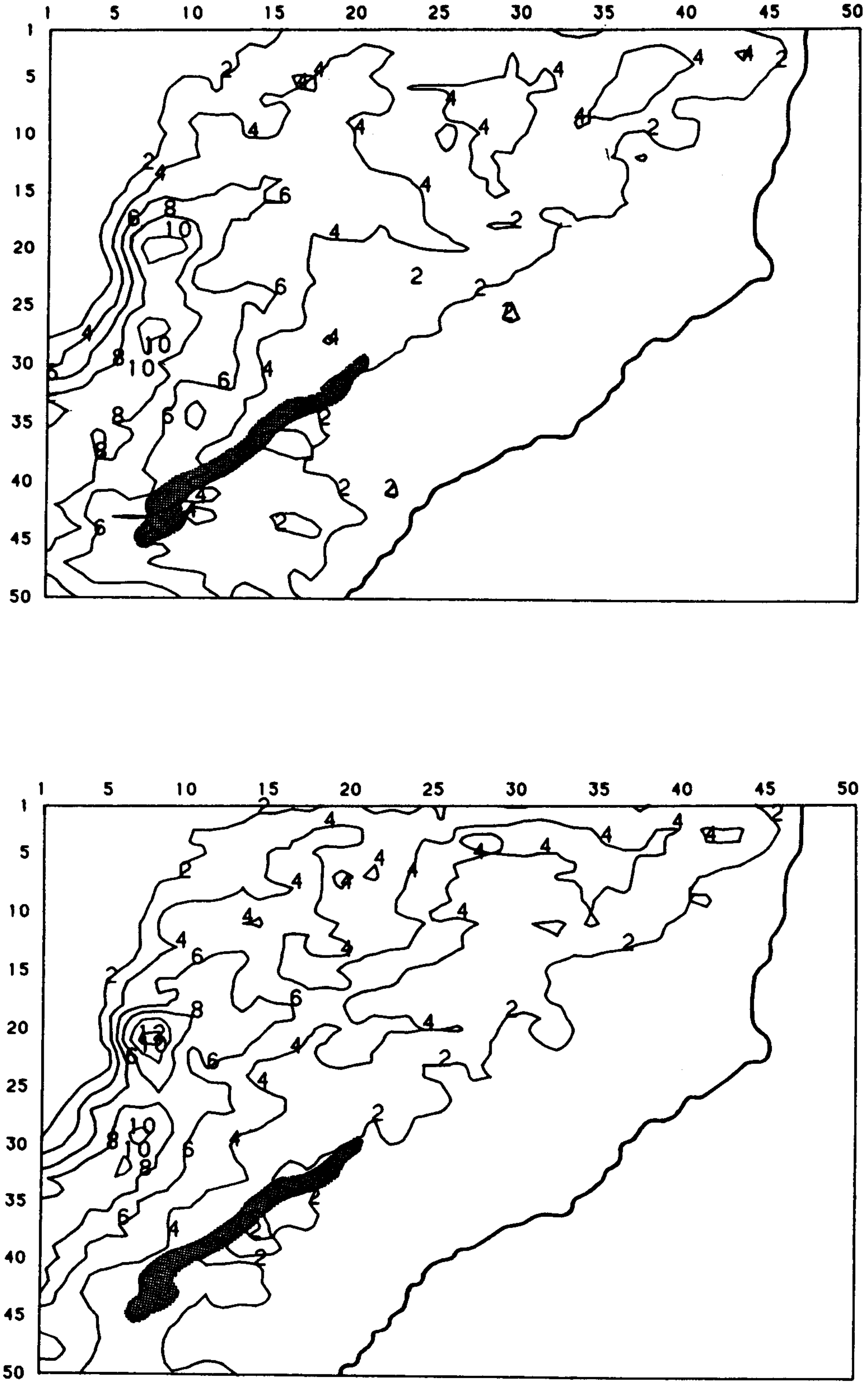


Figure 16 c) top: 240 hrs from start of spill.
d) bottom: 480 hrs from start of spill.

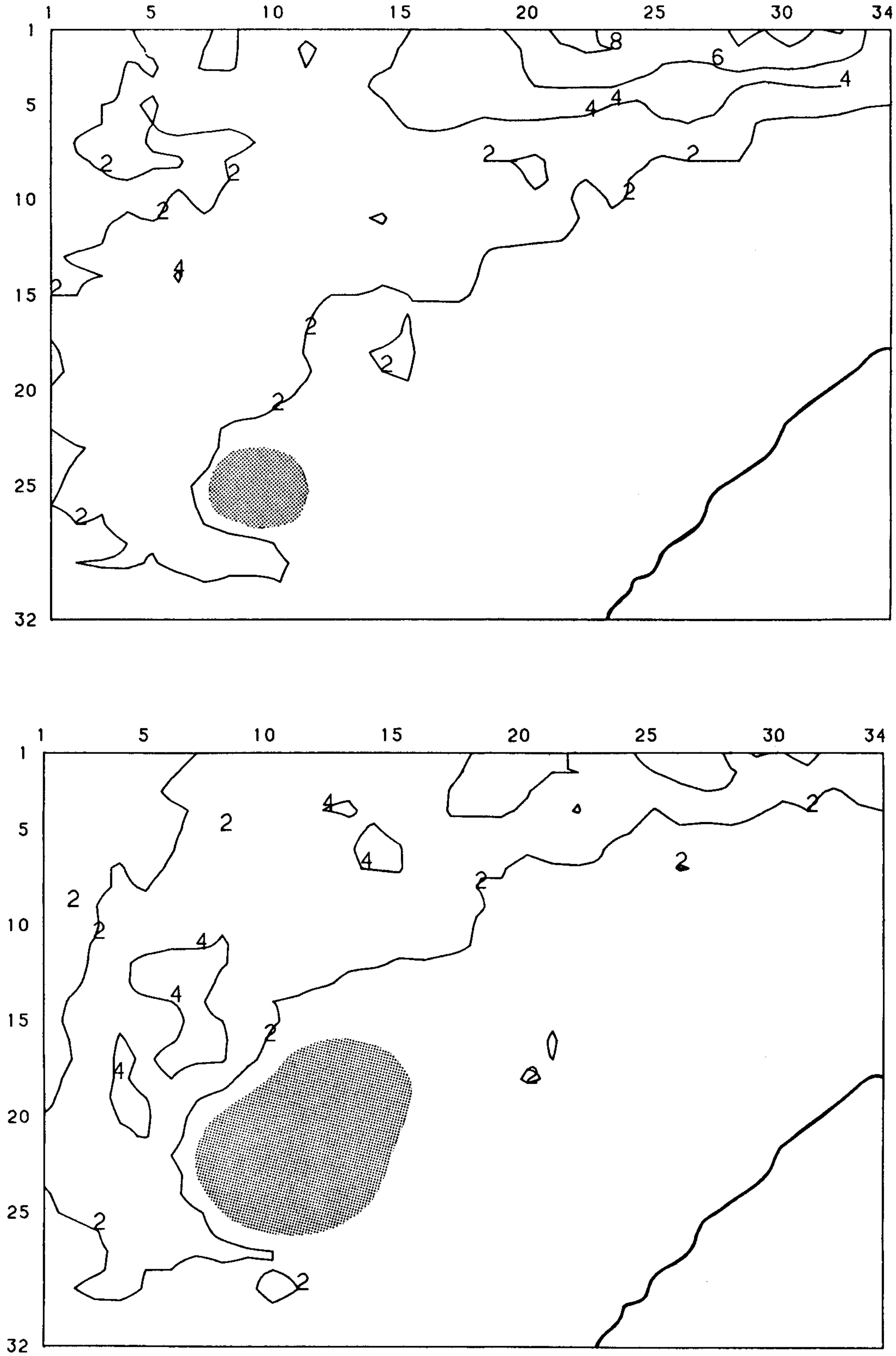


Figure 17.--Simulation of migration of adult sockeye salmon and their avoidance of a spill of 240,000 bbl of No. 2 diesel oil off of Port Heiden, Bristol Bay. (Shaded area depicts concentrations of oil greater than 1.0 ppm). a) top: 48 hrs from start of spill; b) bottom: 120 hrs from start of spill.

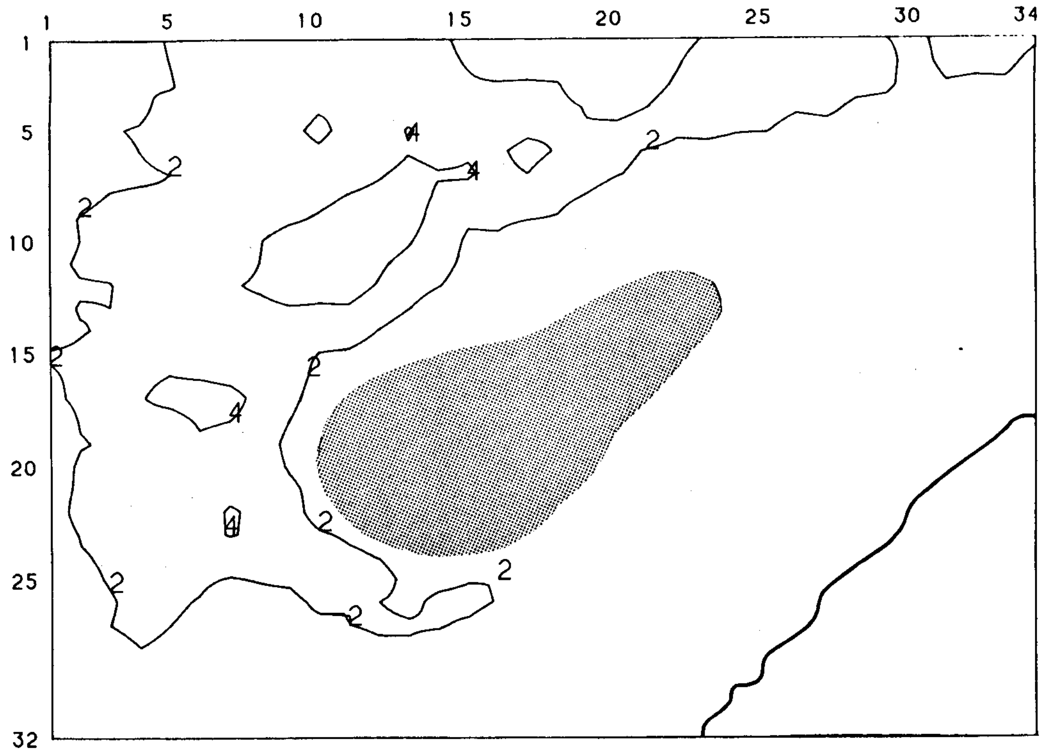


Figure 17c: 240 hrs from start of spill.

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 population 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.

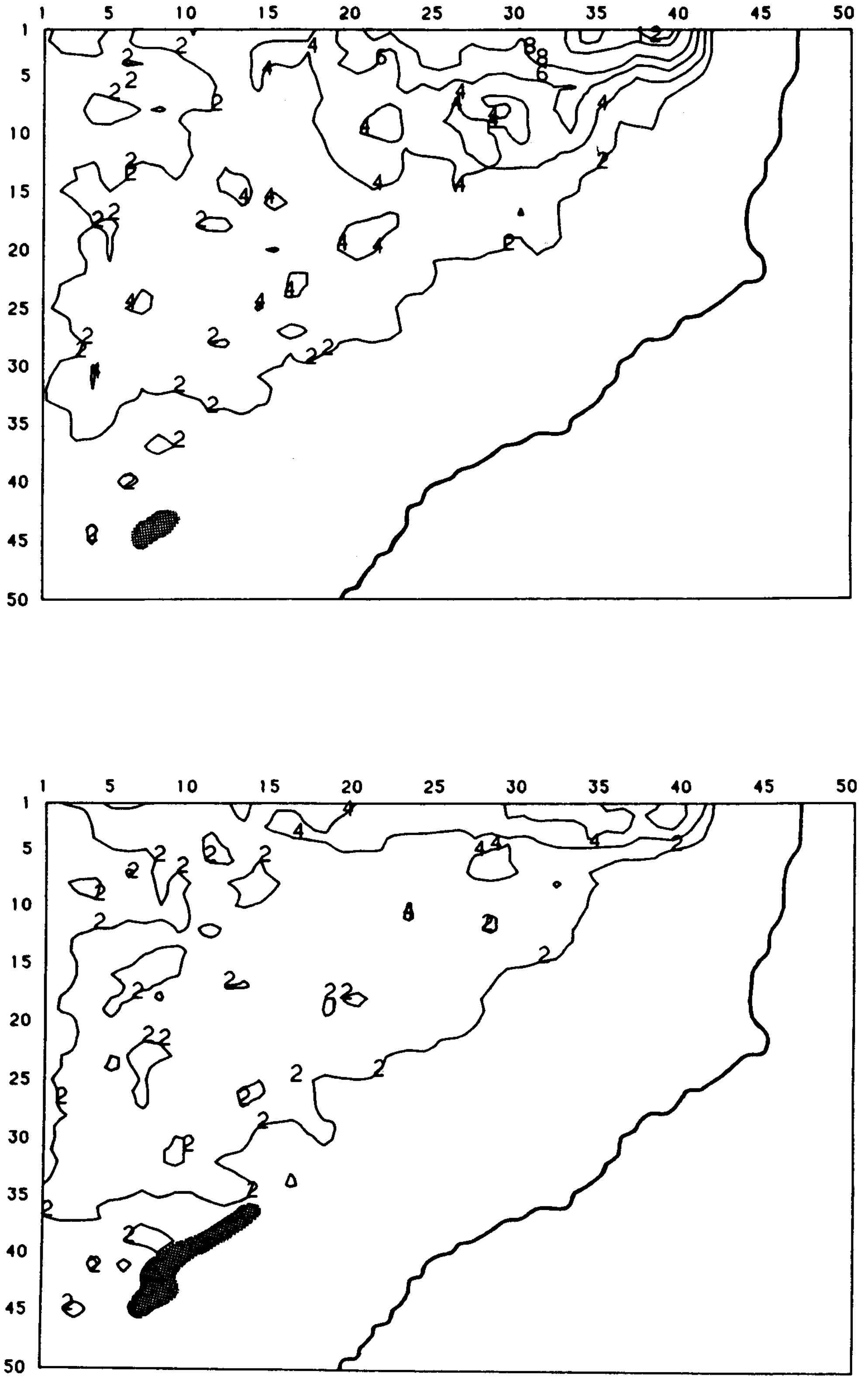


Figure 18.--Simulation of migration of adult sockeye salmon and their avoidance of a blowout of 20,000 bbl/d off of Port Heiden, Bristol Bay. (Shaded area depicts concentrations greater than 0.1 ppm). a) top: 24 hrs from start of spill; b) bottom: 120 hrs from start of spill.

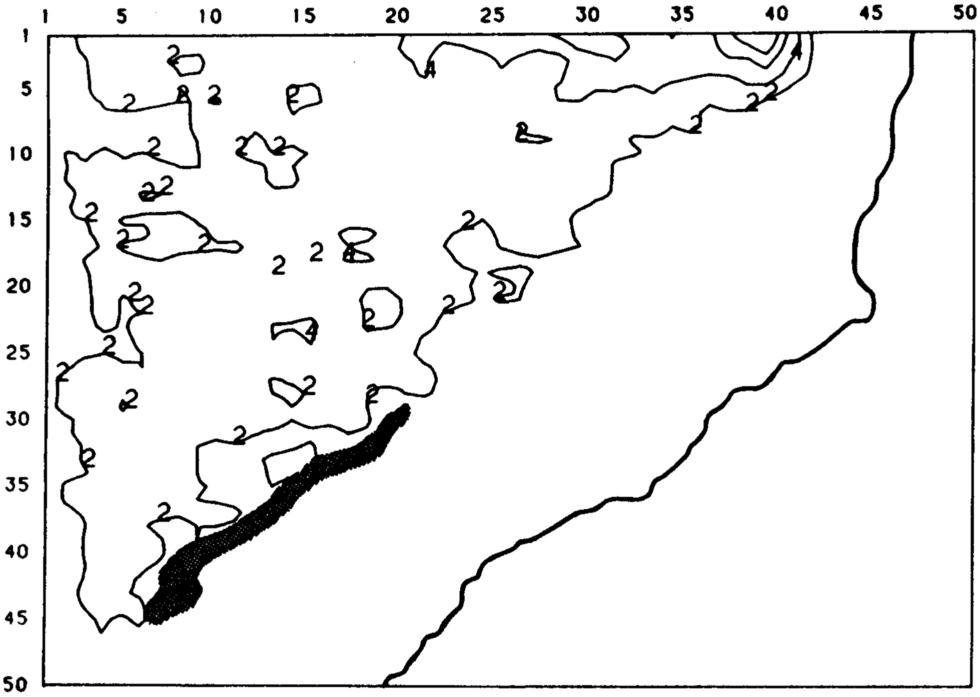


Figure 18c: 480 hrs from start of spill.

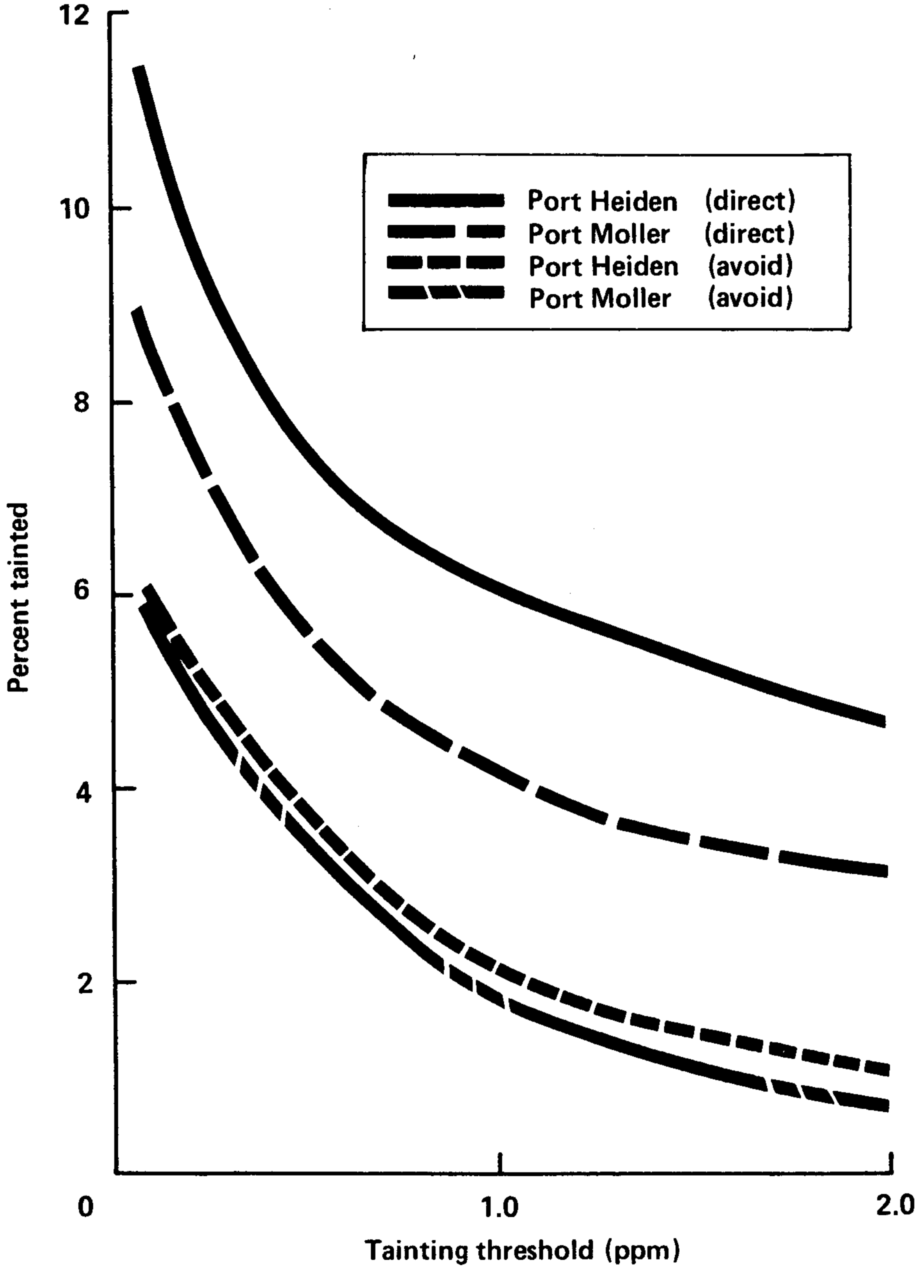


Figure 19.--Percentages of adult sockeye salmon migrating through the fuel oil spill grids that would be tainted at different tainting thresholds and under the assumptions of avoidance or direct migration.

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.

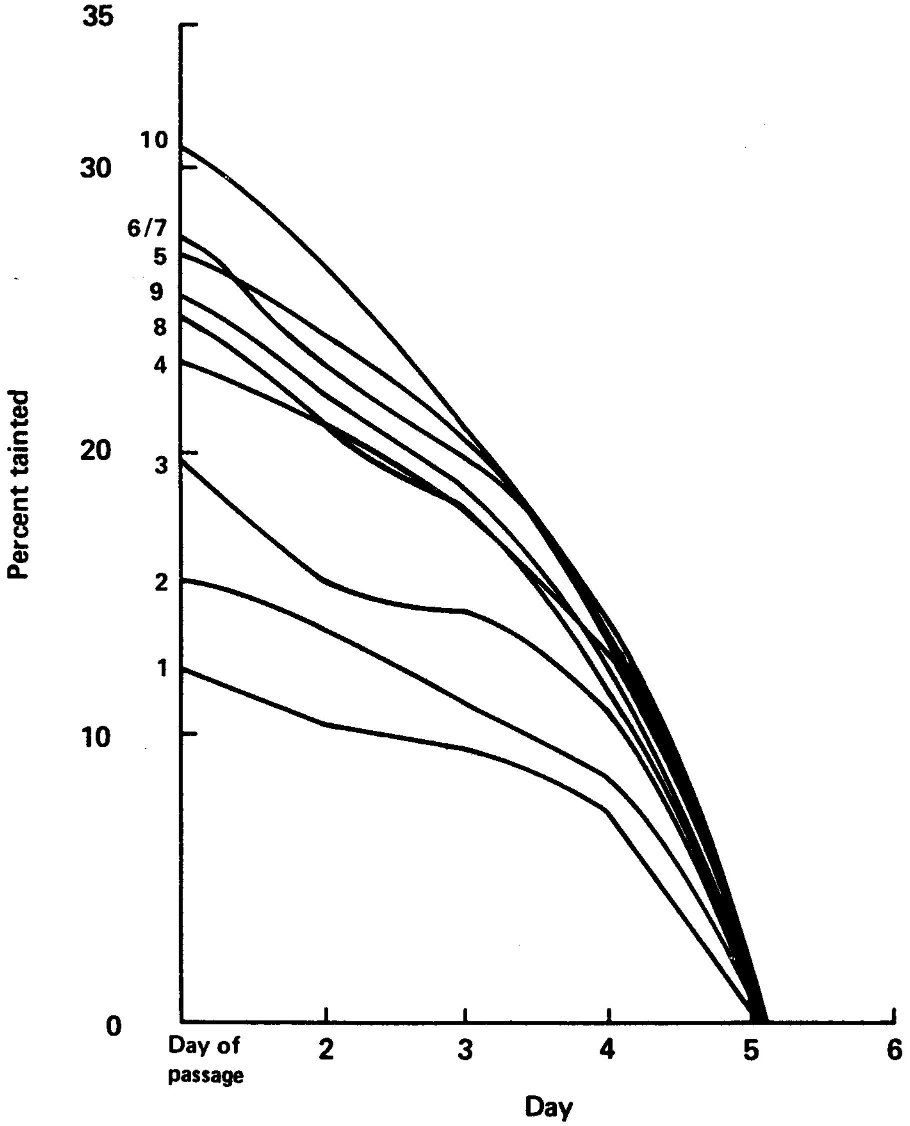

Figure 20.--Percent of adult sockeye salmon tainted after passing through the fuel oil spill grid off of Port Heiden from days 1-10 of the spill, and the percentages remaining tainted over the next 5 days.

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

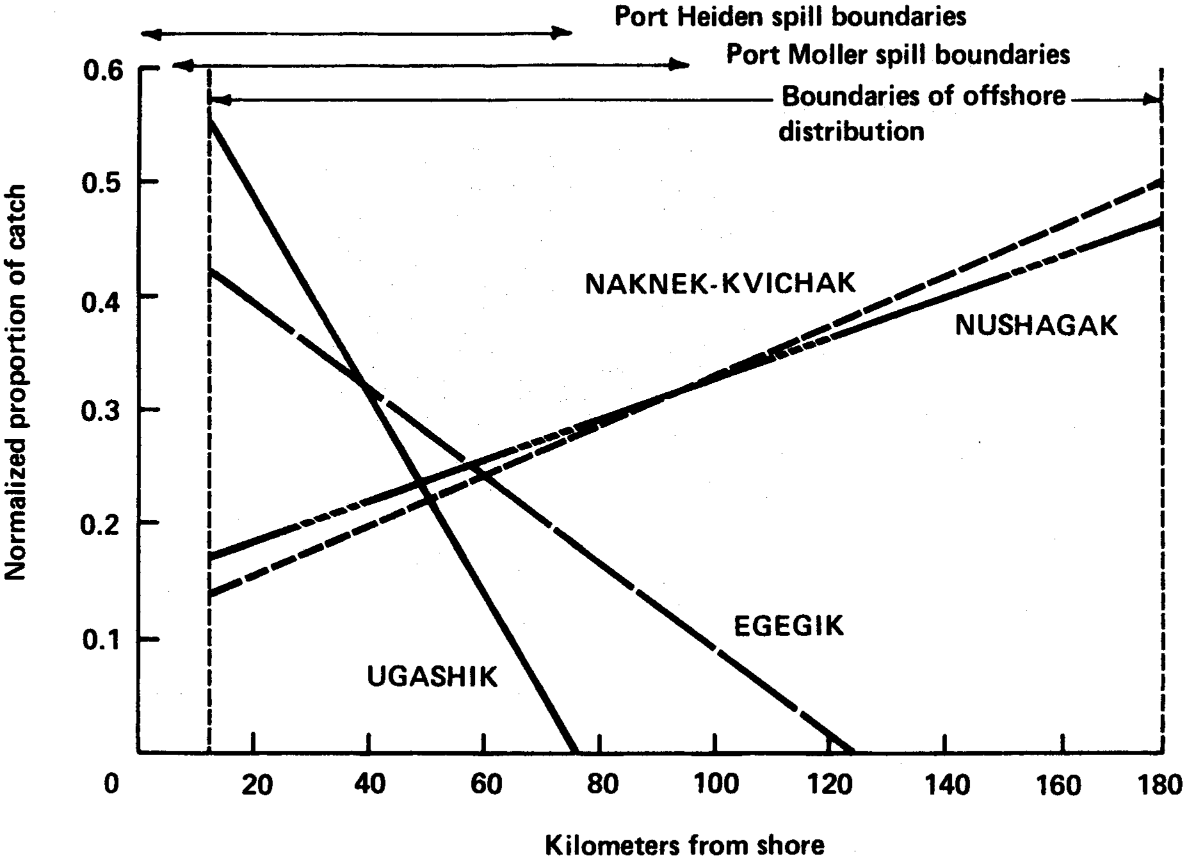

Figure 21.--Proportions of total catches of adult sockeye salmon between Port Moller and Port Heiden caught at varying distances from shore, and delineated by river of return. (Data from Straty 1975.)

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.

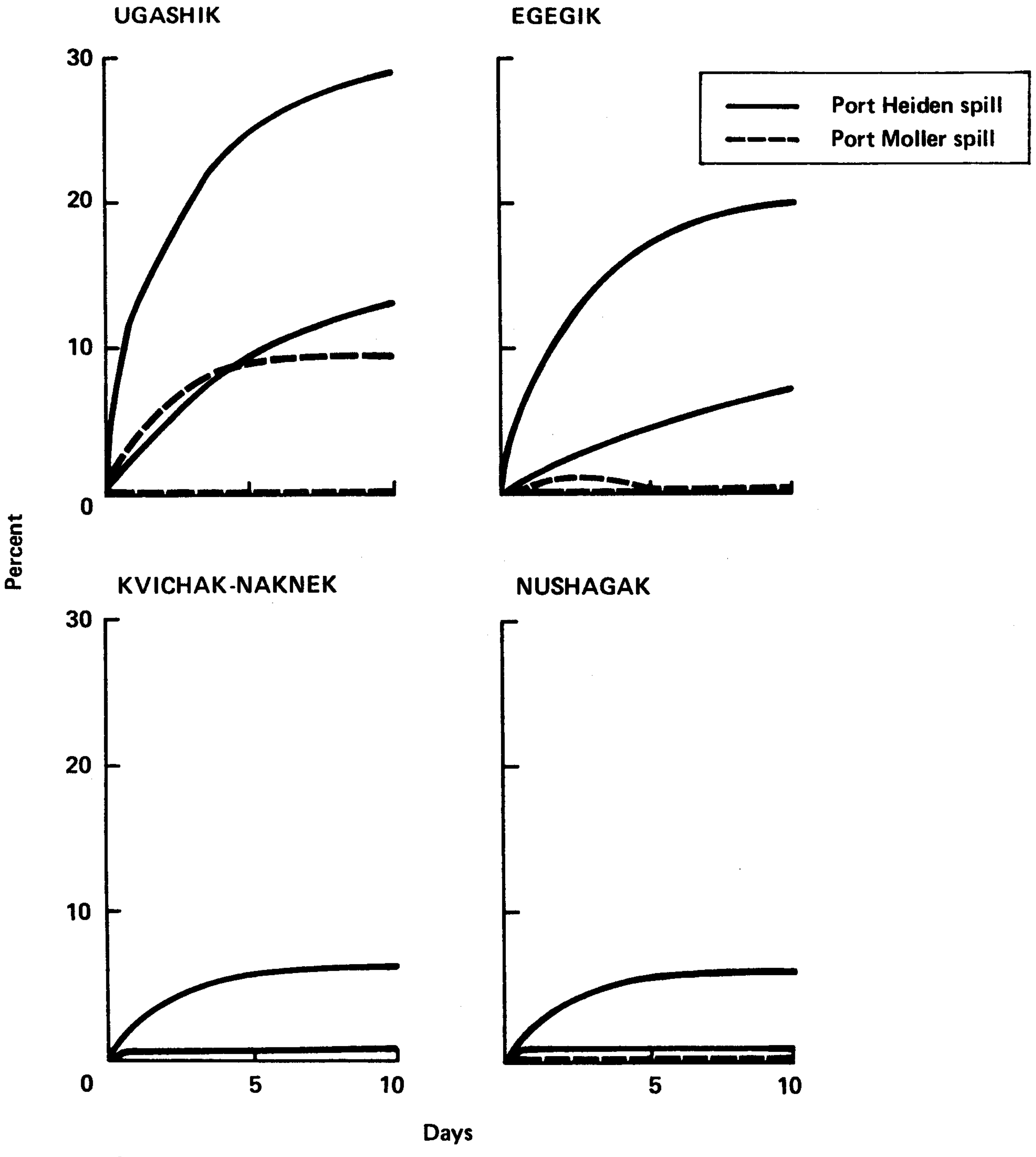

Figure 22.--Simulated proportions of adult sockeye salmon arriving on the fishing grounds in Bristol Bay that would be tainted after passing through a spill of No. 2 fuel oil from day 1 to day 10 of that spill.

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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