Managing Seafood Safety after an Oil Spill

Courtesy of PhotoDisc

Ruth Yender
Office of Response and Restoration
National Oceanic and Atmospheric Administration
Seattle, Washington

Jacqueline Michel and Christine Lord
Research Planning, Inc.
Columbia, South Carolina

National Oceanic and Atmospheric Administration • NOAA's National Ocean Service • Office of Response and Restoration

Managing Seafood Safety after an Oil Spill

Ruth Yender
Office of Response and Restoration
National Oceanic and Atmospheric Administration
Seattle, Washington

Jacqueline Michel and Christine Lord
Research Planning, Inc.
Columbia, South Carolina

National Oceanic and Atmospheric Administration • NOAA's National Ocean Service • Office of Response and Restoration

November 2002

FOR INFORMATION OR COPIES OF THIS DOCUMENT, PLEASE CONTACT:
ruth.yender@noaa.gov

PLEASE CITE AS:
Yender, R., J. Michel, and C. Lord. 2002. Managing Seafood Safety after an Oil Spill. Seattle: Hazardous Materials Response Division, Office of Response and Restoration, National Oceanic and Atmospheric Administration. 72 pp.

Table of Contents

Figures

Tables

I. INTRODUCTION AND BACKGROUND

Seafood safety is a concern raised at nearly every oil spill incident of any significance. Both actual and potential contamination of seafood can substantially affect commercial and recreational fishing and subsistence seafood use. Loss of confidence in seafood safety and quality can impact seafood markets long after any actual risk to seafood from a spill has subsided, resulting in serious economic consequences. Protecting consumers from unpalatable and unsafe seafood is a primary objective of federal and state public health agencies after a spill occurs. Seafood managers may be faced with making many urgent decisions after an oil spill, often based on limited data:

Should seafood harvest in the spill area be closed or restricted?

If closed, what criteria should be applied to re-open a fishery?

How should seafood safety and palatability be evaluated?

How can health risks best be communicated to the public?

Public health officials and other seafood managers do not routinely deal with oil spills as part of their day-to-day responsibilities. Consequently, they typically have little experience with risks to seafood from oil spills when they suddenly are faced with determining appropriate seafood management actions in response to a spill.

The objective of this guide is to provide seafood managers and other spill responders with information to help them evaluate the likelihood that an oil spill will contaminate seafood, determine whether seafood actually has been contaminated, and assess and communicate human health risk from eating contaminated seafood. The guide is divided into the following sections:

I. Introduction and Background

II. Assessing the Likelihood of Seafood Exposure and Contamination

Describes the factors that influence exposure, uptake, and elimination in aquatic organisms.

III. Monitoring Seafood for Contamination

Provides guidance on chemical and sensory testing methods, sampling strategies, and monitoring.

IV. Seafood Risk Assessment

Describes carcinogenic risk assessment methods, assumptions, and interpretation of chemical results.

V: Risk Communication

Provides guidance on communicating risks associated with contaminated seafood and gives examples of advisories.

A glossary of terms used in this guide is included in the appendix.

Decision Process for Managing Seafood Safety

The guide generally follows the flow chart shown in Figure I-1, which suggests a decision process for managing seafood safety after oil spills. Throughout this process, the default is no closure or other restrictions on seafood harvest. In some cases there may be an initial, temporary de facto closure if the U.S. Coast Guard establishes a safety zone restricting access in areas of active oil recovery. Fishermen also may voluntarily avoid working in oiled areas to prevent oiling their gear and catch. This initial period after a spill can provide an opportunity to evaluate spill conditions and conduct limited testing to determine whether a precautionary closure or other immediate restrictions on seafood harvest are warranted.

As indicated on the flowchart, the first step for seafood managers after an oil spill has occurred is to collect and evaluate information on the nature of the spill. The spill response organization should be able to provide the following information almost immediately after the spill occurs:

- overflight maps and trajectory analyses showing the present and predicted spread of surface slicks;
- forecasts of weather and sea conditions that may affect the potential for oil to mix into the water column;
- results of oil weathering models;
- details about the oil type and expected behavior;
- predictions of oil fate and persistence; and
- in some cases, chemical results for water and sediment samples collected in the spill area.

Fishery management agencies and associations should be able to provide information on:

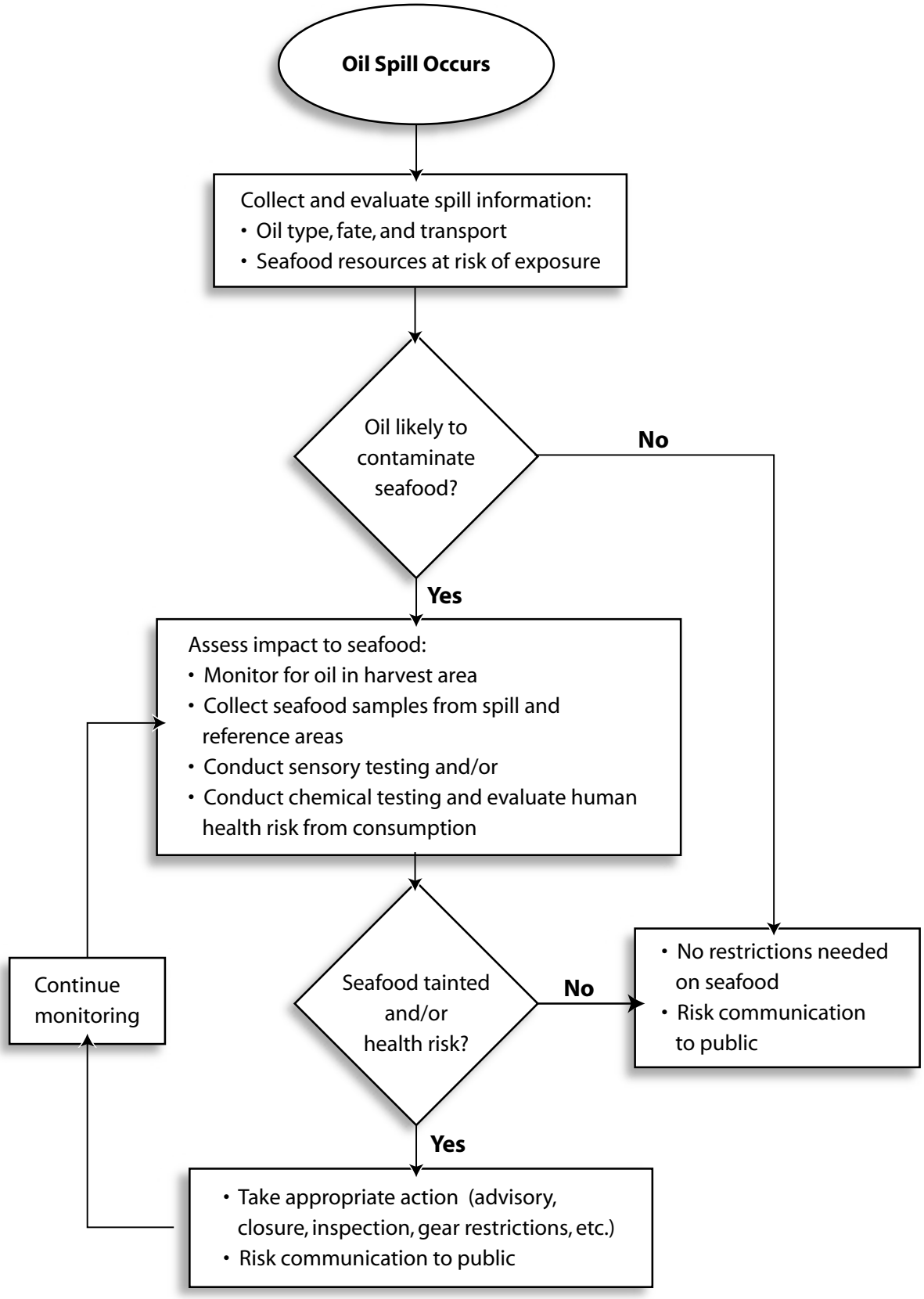
- species being harvested now or in the near future;
- geographical extent of the harvest areas;
- harvest gear types in use; and
- data on background levels of PAH contamination in the spill area (from NOAA Mussel Watch and other monitoring programs).

Based on this type of information, seafood managers can assess whether the oil spill is likely to expose and contaminate seafood. If seafood is not at significant risk, then no harvest closures or other seafood restrictions are needed, and this determination is communicated to the public. Because spills are dynamic, conditions are monitored and risks to seafood re-evaluated until the threat abates.

If managers determine that seafood may be affected, the next step is to assess whether seafood is tainted or contaminated to levels that pose a risk to human health through consumption. Information that can help determine the impacts includes:

- overflights and ground surveys identifying visible oil in seafood harvesting areas;
- chemical analysis of water and/or sediment samples from the harvest area;
- sensory testing of seafood samples from representative species and areas (both spill and reference areas);
- chemical analysis of tissue samples from representative species and areas (both spill and reference areas); and
- data on background levels of oil-related contaminants.

Figure 1.1 Decision process for managing seafood safety after an oil spill



Determining whether seafood has been contaminated can take substantial time. Developing and implementing sampling plans, conducting sensory and/or chemical testing, and evaluating results may require weeks or longer. Monitoring continues and the risk assessment process is repeated as necessary.

If seafood is tainted or is contaminated to a level posing a potential health risk, the next step is to select the most appropriate seafood management action(s). Examples of management actions include seafood advisories, increased inspections of harvested seafood or fishing gear, harvest closures, and fishing gear restrcitions. If a fishery is closed or otherwise restricted, seafood managers must establish criteria for determining that the seafood is palatable and safe for human consumption and that restrictions can, therefore, be lifted. No accepted international or federal criteria have been established for oil-related contaminants in seafood. State seafood managers generally have developed their own criteria for each spill, resulting in some inconsistencies among spills. Varying levels of background contamination also have contributed to inconsistencies in criteria applied.

Several papers summarize some of the difficult seafood management issues encountered after recent oil spills (Mearns and Yender 1997; Mauseth and Challenger 2001; Moller et al. 1989; Moller et al. 1999; Mauseth et al. 1997; Challenger and Mauseth 1998). Table I-1 also summarizes information on a few recent spills at which seafood safety was an issue of concern.

Seafood Safety Management Authority

Typically, authority to manage seafood to protect human health resides with state health agencies. Many states routinely chemically analyze finfish and shellfish tissues for contamination as part of their water-quality monitoring programs. If a state concludes that eating contaminated finfish or shellfish collected from state waters poses an unacceptable human health risk, it may issue local fish consumption advisories or harvest closures for specific water bodies or parts of water bodies and specific species.

The Food, Drug, and Cosmetic Act authorizes the U.S. Food and Drug Administration (USFDA) to protect and promote public health. The USFDA's responsibilities include keeping "adulterated" food off the market. The USFDA has jurisdiction over seafood that crosses state lines in interstate commerce.

The Magnuson Act, 16 U.S.C. 1801 *et seq.*, authorizes NOAA's National Marine Fisheries Service (NMFS) to regulate fishing in federal waters (generally from 3-200 miles from shore). The act is targeted toward fishery conservation rather than protection of public health or economic concerns. Fishery management plans, developed under the authority of the Magnuson Act, specify any limitations imposed on fishing for federally regulated species. Limits on fishing are enforced by means of regulations published in the Federal Register, in compliance with the Administrative Procedures Act. In the event of an oil or chemical spill, publication of an emergency rule in the Federal Register is required to put an enforceable, official fishery closure in place and to make any modifications to the closure once it is put into effect. The Magnuson Act was recently amended to allow emergency action fisheries closures to remain in effect indefinitely. Previously, such closures were limited to two 90-day periods.

Table I-1. Recent oil spills where seafood monitoring was conducted

Spill Name/ Location	Oil Type/ Volume	Spill Conditions	Species Monitored	Closures*	References
M/V New Carissa Near Coos Bay, OR 4 Feb 1999	Two bunker oils and two marine diesels/ 70,000 gallons	Oil released in the surf zone (>5m waves) over several weeks	Oyster, shrimp, crab	Bivalves: 21 days, longer adjacent to the vessel	Gilroy (2000), Michel (2000) Mauseth and Challenger (2001)
M/V Kure Humboldt Bay, CA 5 Nov 1997	Intermediate fuel oil (IFO 180)/ 4,537 gallons	4 days of sheens in bay; light shoreline oiling	Mariculture oyster, rock crab	Mariculture oyster, crabs: 49 days	Challenger and Mauseth (1998)
T/V Julie N Portland, ME 27 Sept 1996	IFO 380 and No. 2 fuel oil/180,000 gallons total	Heavy shoreline oiling in Fore River & Casco Bay	Lobster, scallop, clam, mussel	Shellfish: 15 days	Mauseth et al. (1997)
M/T Provence Piscataqua River, NH 2 July 1996	Heavy fuel oil No. 6 (API 6.2)/ ~880 gallons	Released in Piscataqua River, most of the oil sank	Lobster	None	Mauseth et al. (1997)
T/V Sea Empress Milford Haven, Wales 15 Feb 1996	Forties light crude/ Heavy fuel oil #6/ 21,274,000 gallons total	Severe weather; extensive use of dispersants	Cockle, mussel, crab, lobster, whelk, wild salmon, and other finfish	Marine finfish: 82 days; whelk & crustaceans: 183 days; cockles: 125 days; mussel: 8-19 months	Law et al. (1997); Coates (1998)
T/B North Cape Block Island Sound, RI 19 Jan 1996	Home heating oil No. 2 828,000 gallons	Gale-force winds, release in surf zone, 6-7 m waves, naturally dispersed	Lobster, finfish, bivalves (coastal ponds)	Finfish and bivalves: 73 days; lobsters: 75-155 days	Mauseth et al. (1997)
T/V Braer Shetland Islands 5 Jan 1993	Gullfaks light crude/ 25,000,000 gallons	Hurricane-force winds; release in surf zone, naturally dispersed	Haddock, dab, farmed salmon, cod, sole, ling, lobster, scallop, edible crab	Wild finfish: 2 months; farmed salmon: 12 mo; burrowing lobster: >6 yrs	Kingston (1999) Topping et al. (1997) Whittle et al. (1997)
T/V Exxon Valdez Prince William Sound, AK 24 Mar 1989	Prudhoe Bay crude/ 11,000,000 gallons	Over 700 km of shoreline oiled	Finfish, bivalves from subsistence harvest areas	Herring and salmon: entire season; Advisories on bivalves in 4 subsistence harvest areas	Fall and Field (1996) Field et al. (1999)

*Closure does not necessarily indicate that either tissue contamination or taint was detected or persisted for as long a period as the closure remained in place.

II. ASSESSING THE LIKELIHOOD OF SEAFOOD EXPOSURE AND CONTAMINATION

Each oil spill is a unique combination of conditions and events. Seafood is only at risk of contamination from a spill if it is exposed to the oil. Once exposed to oil, an organism becomes contaminated only to the extent it takes up and retains petroleum compounds. Factors that influence the potential for spilled oil to expose and contaminate seafood are discussed in this section.

Oil Types and Properties

Oil type and properties strongly influence whether seafood is exposed and contaminated. Crude oils and the refined products derived from them are complex and variable mixtures of hydrocarbons of different molecular weights and structures. They can contain hundreds of different compounds. All crude oils contain lighter fractions similar to gasoline, as well as heavier tar or wax fractions. Because of these differences in composition, different oils vary considerably in their physical and chemical properties. For example, consistencies of different crude oils vary, ranging from a light volatile fluid to a viscous semi-solid. Such differences in properties influence behavior of spilled oil and subsequent cleanup operations.

The petroleum hydrocarbons that comprise oil are composed primarily of hydrogen and carbon, but also can contain varying amounts of sulfur, nitrogen, oxygen, and trace metals. The three main fractions of hydrocarbon compounds in oils are saturates, aromatics, and polar compounds. The properties and relative abundance of each fraction in different types of oil products are summarized in Table II-1. Note that toxicity differs among different hydrocarbons and, therefore, different oils.

Table II-1. Components in oil and selected characteristics (modified from NRC 2002).

Group	Sub-groups (alternate name)	Selected Characteristics	Typical Content in Oil (%)
Saturates	1. Alkanes (aliphatics): n-alkanes (paraffins) are straight- chained; isoalkanes are branching 2. Cyclo-alkanes (cyclo-paraffins or naphthenes): saturated ring structures 3. Waxes: larger saturate compounds	High rate of microbial degradation up to C22; Low water solubility; Low aquatic toxicity	Gasoline: 50-60 Diesel: 65-95 Light crude: 55-90 Heavy crude: 25-80 Heavy fuel oil: 20-30
Aromatics	1. Monoaromatics (BTEX): single benzene ring 2. Polycyclic aromatic hydrocarbons (PAH): 2-6 benzene rings	Slower rate of microbial degradation than saturates; Higher water solubility; High aquatic toxicity	Gasoline: 25-40 Diesel: 5-25 Light crude: 10-35 Heavy crude: 15-40 Heavy fuel oil: 30-50
Polar Compounds	1. Resins: smaller compounds that bond with S, N, or O 2. Asphaltenes: very large compounds	Very slow microbial/ physical degradation; Very low water solubility/aquatic toxicity	Gasoline: 0 Diesel: 0-2 Light crude: 1-15 Heavy crude: 5-40 Heavy fuel oil: 10-30

Oils have been grouped into types with similar properties to help predict their behavior at spills (NOAA and API 1994). This same approach can be used to characterize the relative risk of contamination of seafood by oil type. Table II-2 summarizes the properties and risk of seafood contamination for the five oil groups commonly encountered by spill responders. These generalizations can be used when initially screening an incident to evaluate the potential for seafood contamination.

Table II-2. Characteristics of oil types affecting the potential for seafood contamination (modified from NOAA and API 1994).

Gasoline Products	**Diesel-like Products and Light Crude Oils**	**Medium-grade Crude Oils and Intermediate Products**	**Heavy Crude Oils and Residual Products**	**Non-Floating Oils**
Examples – Gasoline	Examples – No. 2 fuel oil, jet fuels, kerosene, West Texas crude, Alberta crude	Examples – North Slope crude, South Louisiana crude, IFO 180, lube oils	Examples – San Joaquin Valley crude, Venezuelan crude, No. 6 fuel oil	Examples – Very heavy No. 6 fuel oil, residual oils, vacuum bottoms, heavy slurry oils
Specific gravity of <0.80; Floats on surface	Specific gravity of <0.85; API gravity of 35-45* Usually floats on surface; although can contaminate suspended sediments that are then deposited on the bottom	Specific gravity of 0.85-0.95; API gravity of 17.5-35* Usually floats on surface, although can mix with sand by stranding on beaches or in the surf zone, and be deposited in the nearshore	Specific gravity of 0.95-1.00; API gravity of 10-17.5* Usually floats on surface but can sink in fresh water or in seawater if they emulsify or mix with sand (in the surf zone or after stranding on beaches) and deposit in the nearshore	Specific gravity greater than 1.00; API gravity < 10* Will sink in fresh water; may sink in seawater if they emulsify or mix with sand (in the surf zone or after stranding on beaches) and deposit in the nearshore
High evaporation rates; narrow cut fraction with no residues	Refined products can evaporate to no residue; crude oils do leave residues	Up to one-third will evaporate in the first 24 hours; will form persistent residues	Very little product loss by evaporation; will form persistent residues	Very little evaporation when submerged; also very slow weathering overall when submerged
Low viscosity; spread rapidly to a thin sheen; readily dispersed; will not emulsify	Low to moderate viscosity; spread rapidly into thin slicks; readily dispersed by natural processes; may form unstable emulsions	Moderate to high viscosity; dispersed by natural processes only very early in the spill; readily emulsifies	Very viscous to semisolid; will not readily disperse or mix into the water column; can form stable emulsions	Very viscous to semisolid; will not readily disperse or mix into the water column; can form stable emulsions
Low risk of seafood contamination because of rapid and complete loss via evaporation; potential contamination for spills in confined areas with high mixing, such as small rivers; no reported cases of tainting for marine spills	Moderate to high risk of seafood contamination because relatively high content of low molecular weight, water-soluble aromatic hydrocarbons, which are semi-volatile and so evaporate slowly; dispersed droplets are also bio-available	Moderate to high risk of seafood contamination because of high percentage of low-molecular weight aromatic hydrocarbons; coating of gear and intertidal species can be significant	Low risk of finfish contamination because of low water-soluble fraction and little natural mixing in the water; moderate to high risk of shellfish contamination where shoreline oiling is heavy; can coat gear and intertidal species	Low risk of finfish contamination because of high viscosity; where thick oil accumulates on the bottom, could become a chronic source; moderate to high risk of contamination of benthic species because of coating and persistence of submerged oil

*API Gravity is used by the petroleum industry rather than density. It is determined by the following equation: API at 60°F = 141.5/oil density -131.5.

Seafood contamination can result from exposure to the dissolved fraction of oil, dispersed oil droplets, or an oil coating. With regard to the dissolved fraction, the aromatic fraction of the oil poses the greatest exposure risk because aromatics are relatively more soluble than the other components in oil. Saturates are a major component of oil, but they have lower solubility and higher volatility compared to aromatics of the same molecular weight. Furthermore, Heras et al. (1992) has concluded that saturates are virtually odorless and tasteless, and do not contribute to tainting.

Of the aromatic hydrocarbons, the mono-aromatic hydrocarbons, such as benzene, toluene, ethyl benzene, xylene (known collectively as BTEX), other substituted benzenes, and the 2- to 3-ringed PAHs (naphthalene, fluorene, dibenzothiophene, anthracene and their substituted homologues, referred to as low-molecular weight PAHs) comprise over 99 percent of the water-soluble fraction (McAuliffe 1987). The distribution of these compounds in the spilled oil is one measure of the potential for contamination of seafood from water exposure. Figure II-1 shows the PAH composition for typical crude oils and refined products. Table II-3 lists the abbreviations used for PAHs, groups the PAHs into low- and high-molecular weight categories, and shows the number of benzene rings. Most crude oils are composed of a wide range of compounds, including the PAHs of concern.

Note that compounds in petroleum-derived oils have a general pattern of increasing abundance with higher level of substitution of a benzene ring (e.g., unsubstituted parent naphthalene is less abundant than C1-naphthalene, which is less abundant than C2-naphthalene). This pattern indicates that the PAHs are "petrogenic," that is, they are from petroleum oils. The PAH pattern is very different for hydrocarbons produced from the combustion of fossil fuels ("pyrogenic" hydrocarbons), in that the parent PAHs are by far the dominant compounds in hydrocarbons of pyrogenic origin. Also, it is important to note that crude oils contain very low concentrations of the high-molecular weight PAHs (e.g., 4- and 5-ringed compounds such as pyrene, chrysene, and benzo[a]pyrene) that are associated with combustion by-products. These differences in relative PAH abundance are key components of fingerprinting analysis.

Refined products have characteristic ranges of PAHs representative of the distillation fraction in the product. In Figure II-1, note that the PAHs in the No. 2 fuel oil are dominated by the 2- and 3-ringed compounds. Heavy fuel oils are sometimes cut or blended with lighter fractions to meet customer specifications, as is the case with the intermediate fuel oil (IFO-180) in Figure II-1D, and so can contain some low-molecular weight PAHs.

For exposure via ingestion of whole oil droplets or contaminated sediments, the high-molecular weight PAHs pose greater risk of contamination. These compounds have low water solubility and are more lipophilic. In organisms with relatively limited capability to metabolize PAHs, such as bivalve mollusks, the high-molecular weight compounds are more likely to accumulate in tissues and persist for longer periods, compared to the low-molecular weight PAHs, which are more rapidly eliminated (Meador et al. 1995). Finfish and some crustaceans, however, readily metabolize and eliminate all of these compounds rapidly.

Figure II-1. Pattern of PAH distribution for different oil types: A) No. 2 fuel oil; B) South Louisiana crude, a medium crude oil; C) No. 6 fuel oil, a heavy oil; and D) an intermediate fuel oil that is a mixture. Note that high-molecular weight PAHs such as benzo[a]pyrene (BAP) have very low concentrations in petroleum oils.

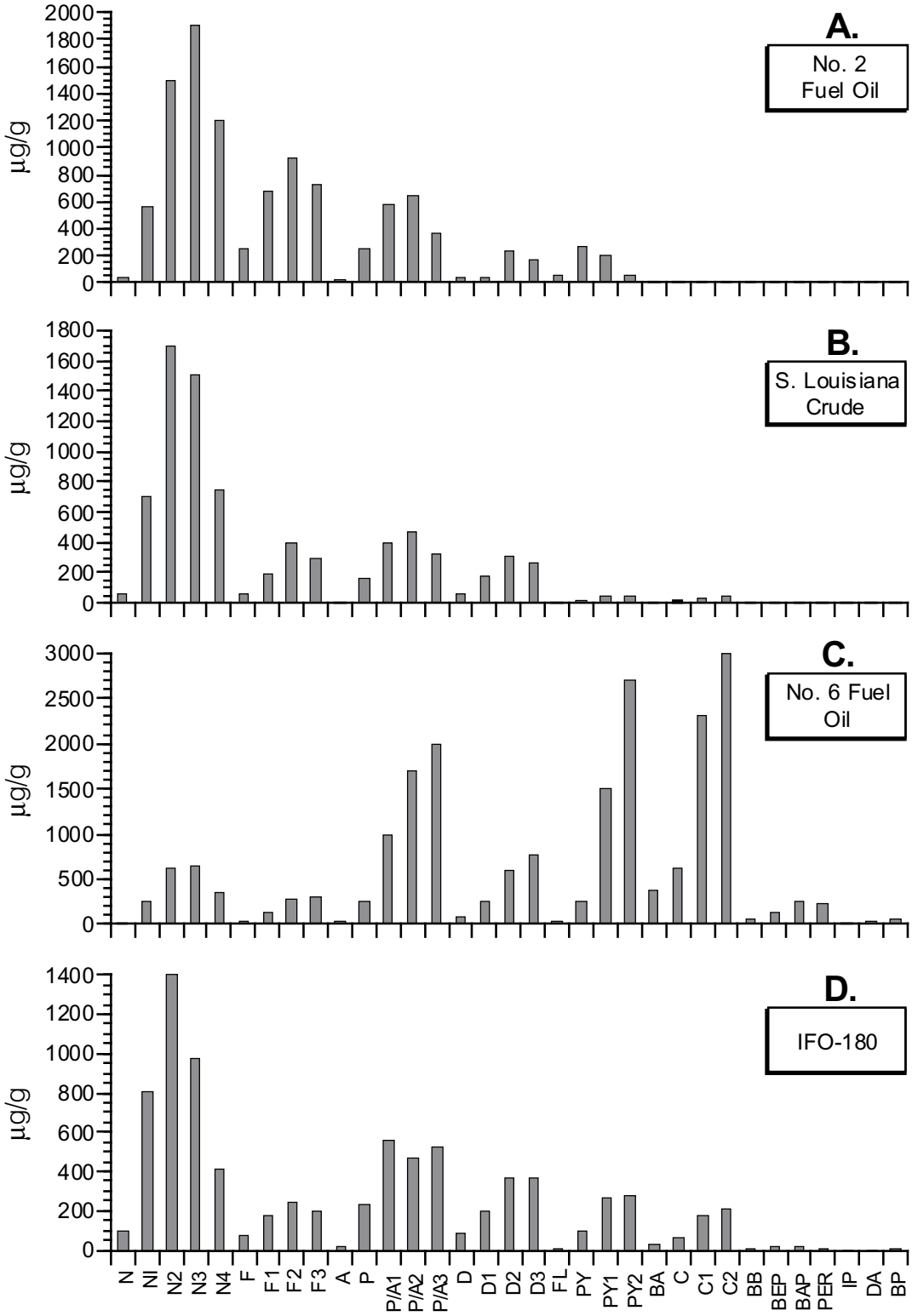


Table II-3. PAHs normally reported in chemical analyses for petroleum compounds (after Sauer and Boehm 1995).

PAH	Abbreviation	No. of Benzene Rings	Molecular Weight
Naphthalene	N	2	Low
C1Naphthalene	N1	2	Low
C2Naphthalene	N2	2	Low
C3Naphthalene	N3	2	Low
C4Naphthalene	N4	2	Low
Biphenyl	BI	2	Low
Fluorene	F	2	Low
C1Fluorene	F1	2	Low
C2Fluorene	F2	2	Low
C3Fluorene	F3	2	Low
Acenaphthylene	AC	3	Low
Acenaphthene	CE	3	Low
Dibenzothiophene	D	3	Low
C3Dibenzothiophene	D3	3	Low
Anthracene	A	3	Low
Phenanthrene	P	3	Low
C1Phenanthrene/Anthracene	P/A1	3	Low
C2Phenanthrene/Anthracene	P/A2	3	Low
C3Phenanthrene/Anthracene	P/A3	3	Low
Napththobenzothiophene	NBT	3	Low
C1Napththobenzothiophene	NBT1	3	Low
C3Napththobenzothiophene	NBT3	3	Low
Fluoranthene	FL	4	High
Pyrene	PY	4	High

PAH	Abbreviation	No. of Benzene Rings	Molecular Weight
C1Pyrene	PY1	4	High
C2Pyrene	PY2	4	High
Benzo[a]Anthracene	BA	4	High
Chrysene	C	4	High
C1Chrysene	C1	4	High
C2Chrysene	C2	4	High
C3Chrysene	C3	4	High
C4Chrysene	C4	4	High
Benzo[b]Fluoranthene	BB	5	High
Benzo[k]Fluoranthene	BK	5	High
Benzo[e]Pyrene	BEP	5	High
Benzo[a]Pyrene	BAP	5	High
Dibenzo[a,h]anthracene	DA	5	High
Indeno[1,2,3-cd]Pyrene	IP	6	High
Benzo[g,h,i]perylene	DP	6	High

Oil Fate and Pathways of Exposure

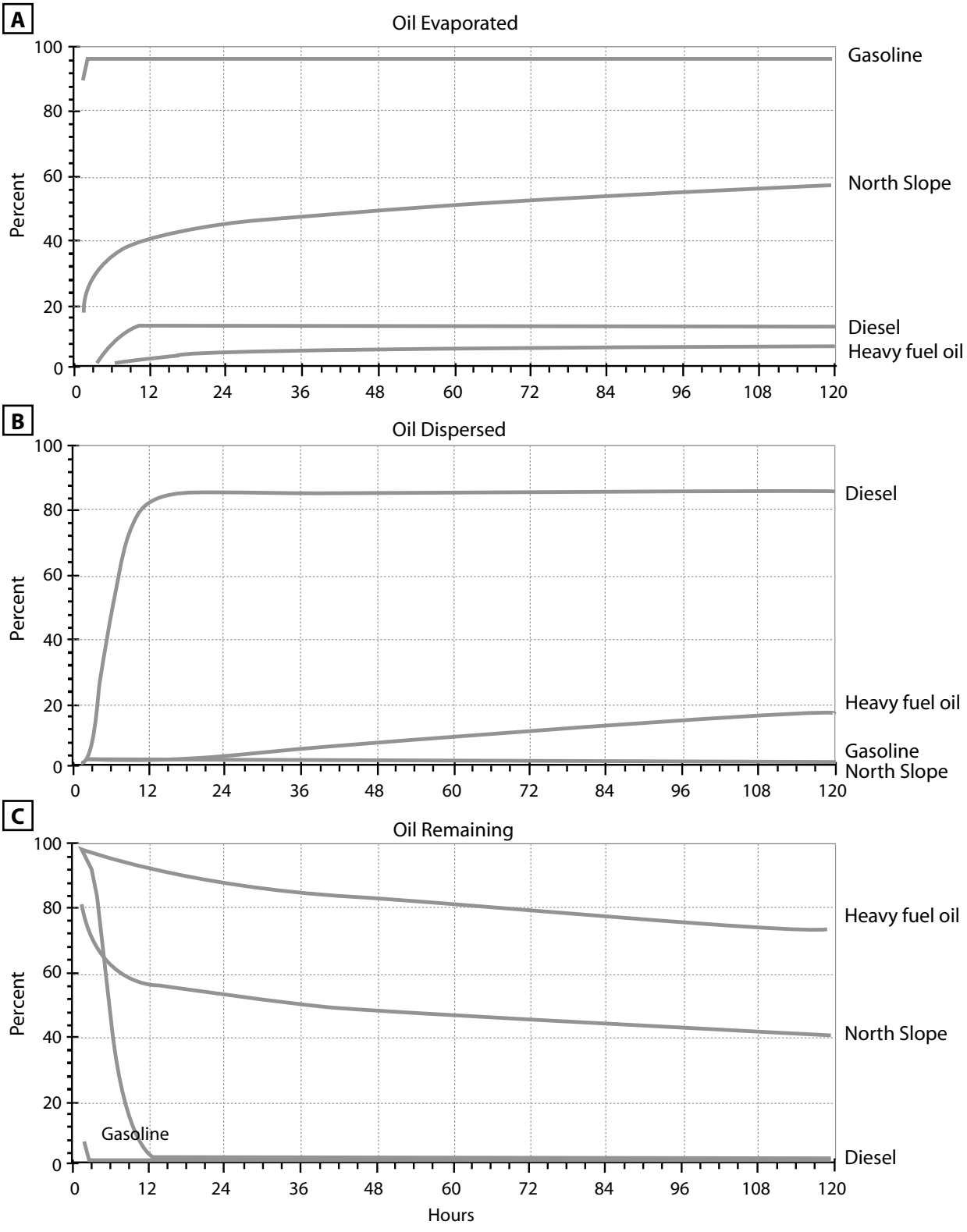
Oil behavior after release determines whether seafood is at risk of exposure. Oil behavior is a function of the processes described below.

Early weathering processes that change oil properties

Evaporation

Evaporation is the transfer of the volatile fractions in oil from the liquid phase to the vapor phase. The rate of evaporation depends on the composition of the oil, surface area of the slick, wind velocity, sea state, water temperature, and solar radiation. Most evaporation occurs in the first 24 hours after release, though it continues at a much lower rate for up to two weeks (NOAA and API 2001). During the first 24-48 hours after a spill, evaporation is the most important weathering process. The amount of oil that evaporates depends primarily on the oil's composition. For light crude oils and refined products, evaporation can account for up to 75 percent loss within a few days. Figure II-2-A shows plots of the loss by evaporation over time for representative oils of the first four oil groups listed in Table II-2. The plots are output from the NOAA oil fate model ADIOS 2 for the same spill scenario for all oil types (Lehr et al. 2000). The lighter the oil, the higher will be the loss by evaporation.

Figure II-2. Plots of predicted evaporation and dispersion for oils representative of four oil types: gasoline, diesel, medium-grade crude oil (North Slope Crude), and heavy fuel oil generated using NOAA's oil weathering model ADIOS 2 (Lehr et al. 2000). The same spill conditions were used for each oil: spill volume 10,000 gallons instantaneously released; wind speed 10 mph; water temperature 60° F.



Evaporation is also important in that the more volatile fractions are also more water-soluble and thus contribute significantly to the oil's uptake and toxicity. Evaporation dominates over dissolution in most spill conditions, so it is a key process that reduces the risk of aquatic exposure to the more soluble, toxic compounds, such as benzene, toluene, ethylbenzene, and xylene (BTEX) and low-molecular weight polycyclic aromatic hydrocarbons (PAHs).

Dissolution

Dissolution is the transfer of water-soluble components in oil to the water. It begins immediately after oil is released and is likely to continue throughout the weathering process. The loss of oil due to dissolution, however, is minor when compared to the other weathering processes. It is not an important process affecting the fate or mass of the spilled oil, since only a small amount dissolves. Less than 0.1 % (very heavy oil) to 2% (gasoline) of the spilled oil volume actually dissolves into the water column. As shown in Table II-4, light refined products, such as gasoline, are more soluble than heavier oils, such as crude oil.

The most water-soluble components in oil are the low-molecular weight aromatic hydrocarbons: the mono-aromatics such as benzene through xylene, and the 2- and 3-ring PAHs, such as naphthalene and phenanthrene (McAuliffe 1987). These components are also the most volatile, and they rapidly evaporate from solution. The rate of dissolution depends on the oil's chemical composition and the surface area of the oil and water.

Though only a small percentage of the spilled oil volume dissolves into the water column, the components that do dissolve are often the most toxic and may also taint seafood at low concentrations. Concentrations of 450 micrograms per liter (µg/L, equal to parts per billion, or ppb) of the water-soluble fraction of a light crude oil have been reported to cause taint in salmon after six hours in laboratory tests (Heras et al. 1992). Davis et al. (1992) reported the tainting threshold for trout exposed to diesel fuel to be 0.08 nanograms per liter (ng/L, equal to parts per trillion). Actual dissolved oil concentrations at spills vary widely, depending on the oil type and environmental conditions. For example, during the *North Cape* spill of approximately 800,000 gallons of home heating oil under conditions of very high natural dispersion, concentrations of dissolved PAHs in water samples were measured to be 3-167 ppb within a few km of the release site (French 1998). These dissolved PAH concentrations are considered to be unusually high for oil spills. During the *New Carissa* release of 70,000 gallons of both marine diesel and bunker oils into the surf zone off Oregon, total dissolved PAHs in water sampled 2-5 km from the release site were reported to be in the range of 0.5-5 ppb (Payne and Driskell 1999).

Table II-4. Example of solubilities of different oil types (Jokuty et al. 1999).

Oil Type	Aqueous Solubility (mg/L or ppm)
Unleaded gasoline	260.9
Diesel	60.4
Prudhoe Bay crude	20.5
Lagomedio	10.0

Dispersion

Wind and waves can break oil slicks into small droplets that mix or disperse into the water column. Under calm conditions, the oil droplets can re-coalesce and resurface as slicks because they are lighter than water. These droplets are composed of the whole oil. Thin slicks of lighter, low-viscosity oils (such as diesel) readily disperse naturally. Heavier, more viscous oils or oils that have become more viscous due to weathering are more resistant to natural dispersion. Applying chemical dispersants, which reduce the oil's surface tension, can enhance natural dispersion. Dispersion is an important mechanism that enhances oil degradation by increasing the exposed surface area. Dispersed oil droplets can be ingested directly (such as by plankton or filter-feeding bivalves) or secondarily by eating oil-contaminated prey. Most past spills that contaminated seafood involved conditions of high natural dispersion (e.g., *Braer*, *North Cape*, and *Amoco Cadiz*).

Emulsification

Emulsification is the process by which one liquid disperses into another in the form of small droplets. This process is most important at oil spills where water droplets mix into the oil and form a stable emulsion (called a "mousse") that does not easily break up. Emulsification causes several response problems: 1) a mousse often contains 50-80 percent water, thus the volume of oily material to be recovered is increased several-fold; 2) emulsified oil is very viscous and difficult to remove or pump (Fingas et al. 1994); and 3) emulsified oil degrades more slowly (NRC 1985).

Comparison of evaporation and dispersion for different oil types

Figure II-2 shows the predicted fate of the first four oil types listed in Table II-2 using the NOAA oil-weathering model ADIOS™2 under the same spill conditions. Note the differences among oil types in the amounts lost due to each of the dominant weathering processes. Light, refined products such as gasoline and diesel evaporate and disperse rapidly, generally within six hours of release. Evaporation can be a dominant weathering process for crude oils, depending on the type of crude. North Slope crude is relatively persistent, particularly if it emulsifies, as in this scenario. Natural dispersion is an important process for low-viscosity oils that are readily broken into droplets by wave action. More viscous oils do not normally disperse naturally. Heavy oils are resistant to weathering and highly persistent.

Gasoline: A light, refined product like gasoline can quickly dissipate when spilled in open-ocean environments. In this particular scenario, strong winds in the model evaporated and dispersed the entire product in the first three hours after the release. The "Oil Remaining" graph shows none of the gasoline remaining three hours after the spill.

Diesel: The diesel selected for this scenario is a light, refined product and, under light wind conditions, the oil will likely remain on the surface with much of the product evaporating. However, strong winds in the scenario (15 knots) will generate breaking waves that tear the surface slick into small droplets. The oil droplets are driven into the water column and, if the droplets are small enough, natural turbulence will prevent the oil from resurfacing. The "Percent Oil Dispersed" graph shows that over 85% of the diesel has dispersed about 12 hours after the spill. Because very little of the oil was available at the surface, a much smaller amount, less than 15%, has evaporated. The "Percent Oil Remaining" graph shows that no product remains after 12 hours.

It is important to note that the terminology for refined products is not standardized, and heavier intermediate fuel oils are sometimes referred to as "marine diesel." These heavier products are much less volatile than normal diesel or Fuel Oil No. 2 and form a more persistent slick than shown in Figure II-2.

North Slope Crude: North Slope crude oil is known to entrain water droplets and form an emulsion if there is sufficient energy in the environment and if a sufficient amount has evaporated. This scenario uses a 15-knot wind so that about 40% of the oil has evaporated in the first 12 hours. After this time, the oil begins to entrain water droplets, eventually forming a stable emulsion containing 70 to 90% water. This process increases the viscosity of the product, making it more difficult for turbulent energy to tear the oil into small droplets and disperse it. Note that the "Percent Oil Dispersed" graph shows that none of the product has dispersed.

Because the *North Slope* crude has emulsified and persisted, the "Percent Oil Remaining" graph shows about 40% remaining 120 hours after the initial release.

Heavy Fuel Oil: Heavy refined products, such as heavy fuel oil, have been refined to remove the lighter components and, as a result, are somewhat pre-weathered. Under strong winds, the "Oil Evaporated" graph shows less than 10% of the product evaporating over the first 120 hours after the release. Heavy products are known to be viscous and, therefore, less likely to be torn into small droplets and dispersed. The "Percent Oil Dispersed" graph shows that less than 20% of the heavy fuel oil disperses over the first 120 hours. Finally, the "Percent Oil Remaining" graph indicates that about 70 to 80% of the oil remains after 120 hours, suggesting that heavy fuel oil is persistent.

During a spill, oceanographers and modelers will generate spill-specific data on the spilled oil's weathering, behavior, trajectory, and fate. They can estimate the present and future spread of surface slicks, extent and persistence of dispersed and dissolved oil plumes, and the risk of oil sedimentation. This information can help seafood managers assess the risk of spilled oil exposing seafood.

Long-term weathering processes that change oil properties

Biodegradation

Biodegradation is the process by which hydrocarbon-degrading organisms such as bacteria, fungi, and yeasts break down petroleum hydrocarbons ultimately into carbon dioxide and water. Oil degradation rates depend on the oil type and may be further limited by oxygen, nutrients, and/or the surface area available to microorganisms. Small droplets of dispersed oil biodegrade more rapidly than tarballs or surface slicks. Light crude oils and light refined products readily biodegrade within weeks to months. Heavier oils can require years to decades to biodegrade. Biodegradation is a very important removal mechanism for persistent oil residues remaining after shoreline cleanup efforts have concluded.

Photo-oxidation

In the presence of oxygen, natural sunlight can cause petroleum hydrocarbons to undergo chemical reactions, a process known as photolysis (NRC 1985). Although the toxicity of photo-oxidation products is a concern because they are more water-soluble and reactive, the rates of photo-oxidation of liquid or solid fractions of the oil are too slow to significantly affect the mass balance of a spill within the first few months (Jordan and Payne 1980).

Sedimentation

Sedimentation is the process by which particles of floating oil sink to the bottom of the water column and become part of the bottom sediments. Sedimentation of oil can occur when oil droplets sorb onto particulate matter, such as sand and clay. Sorption onto suspended sediments in the water column is likely only under very high wave and wind conditions. For example, during the *Braer* spill, 25,000,000 gallons of a light Gullfaks crude oil were released from the grounded vessel during hur-

ricane-force winds, and an estimated 35 percent of the oil was deposited on the seabed in an area of 4,000 km2 (Kingston 1999). The sedimented oil provided a long-term pathway for exposure to benthic organisms. However, this degree of fine-grained, subtidal sediment contamination is highly unusual. More frequently, sedimentation occurs when stranded oil on sandy beaches adheres to the sediment, then is eroded and deposited in small quantities in the nearshore environment (NRC 1999). Sedimentation can also occur through deposition as fecal pellets after ingestion by marine organisms. During the *Arrow* spill in Chedabucto Bay, Canada, zooplankton ingested naturally dispersed Bunker C oil and later excreted it in their fecal pellets (Conover 1971).

Weathering processes that change the location of oil

Spreading

Oil quickly spreads into a very thin layer on the water surface. The rate of spreading is determined by the surface tension of the oil, water currents, and wind. Spreading enhances the rate and effect of other weathering processes by increasing the oil's exposure to sunlight and air.

Advection

Oil moves on the water's surface due to forces generated by winds and currents in a process known as advection. The speed and direction of wind can vary rapidly over time, so weather forecasts must be closely monitored to correctly predict oil spill trajectories.

Submersion

Most oils float on the water surface because they are less dense than water. If oil is denser than water, or becomes denser as the lighter components evaporate, the oil may submerge. If it attaches to suspended sediments, the oil may sink to the bottom (NRC 1999). Once oil is deposited on the bottom, weathering processes are very slow. Submerged oil can be a chronic source of contamination both from slowly dissolving water-soluble fractions and from physical coating of seafood and fishing gear.

Shoreline Stranding

For most oil spills, the oil floats on the water surface, transported by wind and currents until it strands on the shoreline. Stranded oil can directly coat intertidal organisms, habitats, and fishing and aquaculture equipment. Oil stranded on shorelines adjacent to a fishery can be a source of chronic contamination, particularly where shoreline cleanup is not effective or not attempted due to concerns of causing greater harm to the oiled habitat. Even the most effective shoreline cleanups rarely remove all of the stranded oil. Remaining oil is removed or degraded by natural processes. Natural removal processes usually include physical breakup and dispersal of persistent oil residues over a period of months to years (Shigenaka 1997; Hayes and Michel 1999). This remobilized oil, either as whole oil droplets or attached to suspended sediments, can become available to filter feeders, particularly intertidal and shallow subtidal beds of mussels, oysters, and clams (Shigenaka and Henry 1995).

Shoreline type and degree of exposure influence how long oil persists as a secondary source of seafood contamination. Large volumes of oil can penetrate permeable substrates, such as sand beaches, gravel beaches, and rocky rubble shores. Once oil has penetrated into the substrate, weathering rates are slowed and there can be episodic releases of relatively fresh oil. If the oiled shorelines also are sheltered from direct wave energy, the potential for long-term persistence of oil greatly increases. Sheltering can be large-scale, such as in bays and estuaries; it can also be localized, such as in the lee of a large boulder on an otherwise exposed shoreline. For example, during the extensive monitoring of subsistence seafood following the *Exxon Valdez* oil spill, an oil spill health task force determined that finfish from all areas were safe to consume, but that intertidal shellfish from specific areas should not be eaten (Fall and Field 1996; Field et al. 1999). These specific areas were a small

number of sheltered, sedimentary beaches with high levels of oil contamination in the intertidal sediments. Another example is the 1996 *Sea Empress* oil spill in Milford Haven, Wales. Six months after the *Sea Empress* spill, the only seafood harvest activities still restricted outside of Milford Haven were the exploitation of bivalves where heavy shoreline oiling had occurred in sheltered areas (Law et al. 1997).

Seafood Contamination Terminology

Adulteration: According to the U.S. Food and Drug Administration (FDA), a food is considered adulterated if it bears or contains any poisonous or deleterious substance that may render it injurious to health, if it contains any filthy, putrid, or decomposed substances, or if it is otherwise unfit for food (Federal Food, Drug, and Cosmetic Act, Section 402).

Taint: Taint is commonly defined as an odor or flavor that is foreign to a food product, including seafood (ISO 1992). According to this definition, the presence of a taint simply indicates that flavor or odor is altered; it does not characterize the nature of the off-flavor or off-odor, quantify the degree of taint, or imply health hazard.

Body Burden: The concentration of a contaminant in an organism, reported for the whole animal, or for individual tissues such as gonads, muscle, and liver, is referred to as the body burden. It can be reported on the basis of either wet or dry weight of the organism or tissue.

Uptake: Uptake is the process of contaminant accumulation in an organism. Uptake of oil can occur via the following mechanisms:

- adsorption (adhesion) of oil on the skin
- absorption of dissolved components from the water through the skin (including interstitial water exposures for infauna)
- absorption of dissolved components through the gills
- adsorption of dispersed oil droplets to the lipid surfaces in the gills
- ingestion of whole oil droplets directly or of food contaminated with oil, followed by sorption in the gut

Many factors influence uptake, including the exposure concentration and duration, pathway of exposure, lipid content, and feeding and metabolic rates. Uptake from water generally occurs more quickly than dietary uptake or uptake from sediments.

Bioaccumulation: The net accumulation of a substance by an organism as a result of uptake from all environmental sources and possible routes of exposure (contact, respiration, ingestion, etc.) is termed bioaccumulation (ASTM 1994).

Bioconcentration: The net accumulation of a substance as a result of uptake directly from aqueous solution (ASTM 1994).

Biomagnification: The increase in body burden of a contaminant with trophic level is called biomagnification. PAHs generally do not biomagnify in finfish and shellfish because of their low dietary uptake efficiencies, on the order of 1 to 30%, reflecting slow kinetics and short residence time in the gut (Meador et al. 1995).

Elimination: All of the processes that can decrease tissue concentrations of a contaminant, including metabolism, excretion, and diffusive loss are collectively termed elimination (Meador et al. 1995). *Metabolism* is an active physiological process whereby a contaminant is biotransformed into metabolites. For PAHs, the metabolites are more water-soluble, which facilitates *excretion*, another

active physiological process that eliminates contaminants (both parent compounds and metabolites) through bile, urine, or feces. *Diffusive loss* refers to a decrease in tissue burden caused by simple diffusion out of the organism, which is controlled by partitioning between tissue and water. Meador et al. (1995) recommend that *depuration* be used for the mechanism of diffusive loss, and *elimination* be used for the combined process of metabolism, excretion, and diffusive loss. These definitions are slightly different than those used by ASTM (1994), which defines depuration as "the loss of a substance from an organism as a result of any active or passive process" and provides no definition for elimination. However, the definitions by Meador et al. (1995) are more precise and will be followed in this document. Elimination can also include release of PAHs in lipid-rich eggs or gametes during spawning.

Elimination processes begin as soon as uptake occurs. In constant exposure experiments, body burdens tend to reach a "steady state" in which fluxes of the contaminant moving bidirectionally across a membrane or boundary between compartments or phases have reached a balance, not necessarily equilibrium (Meador et al. 1995). When the exposure decreases, elimination rates depend, in part, on the hydrophobic properties of the compound (Spacie and Hamelink 1982). The half-lives of individual compounds vary (see discussion below).

Growth Dilution: Growth dilution occurs when the rate of tissue growth exceeds the rate of accumulation, such that it appears as though elimination is occurring because the tissue concentration is decreasing (Salazar and Salazar 2001). This process may be important when monitoring bivalves during the growing season.

Biological and Ecological Factors Affecting PAH Contamination of Seafood

Petroleum contamination of finfish and shellfish depends upon a variety of biological and ecological factors. Understanding how different feeding strategies, habitat utilization, and physiology influence the likelihood of petroleum contamination of particular species is critical when managing seafood after spills. Table II-5 summarizes several of these factors for different types of seafood organisms.

Metabolic Capacity

Both vertebrates and invertebrates have mixed-function oxygenase (MFO) enzyme systems that enable them to metabolize petroleum substances (Meador et al. 1995). Enzymatic activity is low in invertebrates compared to vertebrates, and therefore induction of metabolism occurs at a higher contamination level in invertebrates (Marsh et al. 1992). Finfish are able to rapidly and efficiently biotransform or metabolize PAHs and excrete the resulting metabolites into bile (Varanasi et al. 1989). These metabolites do not pose a health risk to human consumers of the finfish. Marine invertebrates, including most shellfish, metabolize petroleum compounds slowly and inefficiently; consequently, they tend to accumulate high concentrations and wide ranges of PAHs (Law and Hellou 1999).

Metabolic capacity of organisms is important from a seafood safety standpoint because some PAHs have carcinogenic potential for human consumers, due to the highly chemically reactive oxidation products that form during the first stage of metabolism in vertebrates (ATSDR 1995; Hellou 1996). Human consumers often eat invertebrates in their entirety, and, therefore, may ingest all of the hydrocarbons that have accumulated in the organism and may be present in the organism's gut. Because finfish, like other vertebrates, rapidly and efficiently metabolize petroleum hydrocarbons, they generally pose little or no health risk to human consumers. Exceptions to this may occur for consumers for whom the edible portion of finfish includes tissues such as liver and gall bladder, which tend to accumulate higher levels of PAHs than muscle tissue.

Table II-5. Habitat utilization, feeding strategies, and risk of exposure to oil of different seafood groups (adapted from RPI 1987, 1989).

Seafood Groups	Examples	Metabolic Capacity	Habitat Utilization	Feeding Strategies	Risk of Exposure
Finfish					
anadromous fish	sturgeon, herring, salmon	high capacity	nearshore and shallow water during spawning	predatory	moderate to high in nearshore/shallow water during spawning
marine pelagic and bottomfish	mackerel, jacks, cod, flounder	high capacity	highly mobile, most species prefer depths of > 10 m	predatory	low
reef fish	sea basses, snappers, porgies	high capacity	relatively deep waters (10 - 200 m)	predatory	low to moderate; higher risk in shallow water
estuarine fish	bluefish, mullet, anchovies	high capacity	spawning in intertidal or subtidal habitats; offshore winter migrations	predatory	moderate to high in nearshore/shallow water during spawning
Crustaceans					
lobster, crabs, shrimp	American lobster, pink shrimp, blue crab	reduced capacity	may migrate seasonally; range of depths between estuarine and deep waters	predatory, omnivorous, scavengers	benthic burrowing, estuarine/shallow water species at higher risk than deep water species
Mollusks					
oysters, mussels	American oyster, Pacific oyster, blue mussel	very limited capacity	shallow subtidal and intertidal regions, estuaries; attached to substrates	filter-feeders	high
clams, scallops	hard clam, soft-shell clam, bay scallop, sea scallop	very limited capacity	intertidal and shallow subtidal areas; benthic or buried in the sediment; some mobility	filter/deposit feeders	high
gastropods	abalone, conch, snails, whelk, limpet, top shell	very limited capacity	intertidal and shallow to deep subtidal areas; epibenthic; some mobility	grazers and predatory	moderate to high

Feeding Strategies and PAH Exposure

The feeding strategies of different marine organisms affect their likelihood of exposure to PAHs:

- Finfish and crustaceans are predatory or omnivorous. They are exposed to oil by ingesting contaminated food items or sediments, and by absorbing water-soluble petroleum compounds through the gills.
- Filter feeding bivalves may ingest dispersed oil droplets and absorb water- and lipid-soluble petroleum compounds as they filter plankton and detritus suspended in the water column.
- Deposit-feeding bivalves may be exposed to oil through contaminated sediments as they feed on benthic detritus, and as they absorb water-soluble compounds from the interstitial water in sediments.

Uptake from the water tends to be more rapid than uptake through the diet for both vertebrates and invertebrates. Studies of dietary uptake of PAHs in finfish indicate low uptake efficiencies, on the order of 1 to 30%, reflecting slow kinetics and short residence time in the gut (Meador et al. 1995). Recent studies have shown that the rate of uptake by sediment contact and ingestion varies, yet it tends to be lower than from the water (Meador et al. 1995). How PAHs partition among water, sediment, and prey items in different aquatic environments may impact the bioavailability of the contaminant. In general, both filter-feeding and deposit-feeding bivalves are considered to be at a higher risk of exposure than predatory or omnivorous finfish and crustaceans due to the persistence of oil in contaminated sediments.

Habitat Utilization and Behavior

A species' habitat utilization and behavior affect the likelihood it will be exposed to oil during a spill (Table II-5).

Finfish

- Most pelagic and benthic finfish that occur in relatively deep waters have a low exposure risk to spilled oil because they are highly mobile and often are able to avoid oiled areas (Moller et al. 1989; Law et al. 1997; Law and Hellou 1999). Also, oil concentrations in the water column are usually low and decline very rapidly, minimizing exposure. Exceptions may occur if a large amount of fresh, light oil is mixed into the water column (as occurred at the *North Cape* and *Braer* oil spills) or if bottom sediments become contaminated.
- Finfish that spawn or occur in nearshore, shallow water areas in intertidal and subtidal zones (e.g., salt, brackish, or freshwater marshes, creeks, or rivers) and in shallow reef zones have a greater risk of exposure than offshore finfish, due to shoreline oiling.
- Penned finfish have a greater risk of exposure than wild finfish because they cannot avoid oil in the water column. Most cases of finfish contamination at oil spills have involved penned finfish at spills where a significant quantity of oil was mixed into the water column.

Crustaceans

- Crustaceans (lobsters, crabs, shrimp) have a moderate risk of exposure because they have some mobility, but utilize benthic habitats in shallow nearshore and estuarine areas.
- Some species of lobsters and shrimp migrate seasonally between estuaries and offshore areas, and are at a higher risk of exposure when they are in nearshore, shallow waters.

- When subtidal sediments are significantly contaminated, species that burrow into soft sediments are at higher risk of exposure. For example, during the *Braer* spill, the burrowing Norway lobster remained contaminated for over five years, whereas epibenthic lobsters eliminated petroleum contaminants to background levels of PAHs in one month (Kingston 1999).

Mollusks

- Most mollusks, especially bivalves, are at high risk of contamination because they are sessile and unable to avoid exposure. They generally occur in substrates in shallow subtidal and intertidal areas where exposures are likely to be most persistent if sediment is contaminated. Filter feeding mollusks can ingest dispersed oil and oil attached to suspended sediments. Deposit feeders can ingest oil-contaminated sediments. The longest seafood closure periods associated with oil spills have been for bivalves in areas where adjacent sediments remained heavily contaminated (Law et al. 1997).
- Some bivalve species use defense mechanisms during oil spills, including closing their shells or shutting down their pumping systems, thereby eliminating the uptake route for the contaminants (RPI 1989). Some species can remain closed for several weeks without adverse effects, whereas others start to degrade a few days after closure.

Temperature

It is generally accepted that uptake and elimination rates both tend to increase with increasing temperature, though there is some contradiction among reported study results for PAHs (Fucik and Neff 1977; Landrum 1982; Jovanovich and Marion 1987; Meador et al. 1995).

The rate of reaction in chemical and biological processes generally increases 2- to 4-fold for a 10°C increase in temperature (Kennedy et al. 1989; French 2000). Uptake, metabolic, and elimination rates typically increase with temperature, but at different rates, making it difficult to predict body burdens under the constantly changing oil concentrations that occur at spills. However, at high temperatures and increased respiration and filtration rates, it is expected that uptake will occur quickly, to relatively high concentration, followed by rapid declines (Meador et al. 1995). At low temperatures, body burdens are likely to be lower, but elimination rates will also be slower. At very low temperatures, some species stop feeding and thus are at lower risk of exposure. For example, elevated levels of PAHs from the *North Cape* oil spill were detected in soft shell clams, oysters, and mussels, but not in quahogs because they stop feeding at 6°C and the water temperature during the spill was 4°C (NOAA et al. 1999).

Physiology

Lipid, carbohydrate, and protein levels are known to vary seasonally in certain aquatic invertebrate species, often associated with reproductive changes (Jovanovich and Marion 1987). Some of these changes in biochemical composition may affect uptake and elimination rates seasonally. Seasonal variation may also result from differences in feeding rates, microbial activity, and various environmental factors (Meador et al. 1995).

Organisms with higher overall lipid content generally exhibit higher levels of uptake or retention of petroleum compounds (NRC 1983). For example, Heras et al. (1992) found that salmon (muscle lipid content of 4.0% wet weight) accumulated higher hydrocarbon concentrations than cod (muscle lipid content of 0.75% wet weight). Jovanovich and Marion (1987) have reported that uptake rates of PAHs in clams peaked when gametogenesis was near completion and decreased during spawning, while elimination rates peaked during spawning. Bender et al. (1986) found that oysters and clams

sampled at the high point of lipid and glycogen reserves during their spawning cycles (the fall) had PAH tissue levels that were 2 to 3 times higher than they were when sampled during the spring. High elimination rates during the loss of lipid-rich eggs are consistent with findings that finfish and shellfish tend to accumulate PAHs in tissues with high lipid content because PAHs are strongly hydrophobic (Meador et al. 1995).

Potential variations in PAH uptake and elimination rates in seafood species due to seasonal and physiological variation should be taken into account during spill response. These differences should be considered when designing seafood sampling plans and when comparing analytical results from samples from different species, collected at different times of year, or collected during different stages in the life cycle of the organisms.

Chronic Exposure Stress

Bioaccumulation levels and elimination rates of hydrocarbons for finfish and shellfish may depend on the type and duration of exposure to petroleum products, and the extent to which the organisms have been chronically exposed to other contaminants. Chronic exposure appears to reduce elimination capacity. In fact, there may be two phases of elimination: an initial rapid phase followed by a second slower phase for PAHs that are sequestered in stable compartments of the organism, such as storage lipids (Meador et al. 1995). Some chronic hydrocarbon pollution studies have indicated no significant reductions in PAH levels in tissues over 2-4 months for clams and mussels, even when the animals were moved to cleaner habitats (DiSalvo et al. 1975; Boehm and Quinn 1977). The ratio of liver/muscle concentrations in finfish sometimes can be used as an indicator of the level of chronic PAH contamination at a site. Liver levels represent shorter-term exposure to oil, while muscle levels represent longer-term bioaccumulation. Therefore, lower liver/muscle ratios may indicate decreased efficiency in an organism's ability to biotransform absorbed or ingested oil into compounds that are easily excreted (Hellou 1996).

Other subsistence and recreational seafood organisms

Some organisms that are collected and consumed for subsistence and recreation were not discussed in this section. Examples are octopus, squid, seals, whales, seaweed, and algae. There isn't enough information on these organisms to thoroughly discuss the level of risk they may pose to consumers following an oil spill. It should be noted, however, that if these organisms occur in a spill area and are exposed, restrictions on harvest or consumption advisories might be warranted, depending on contamination and consumption levels.

Summary

- Wild finfish are unlikely to become contaminated or tainted because they typically are either not exposed or are exposed only briefly to the spilled oil and because they rapidly eliminate petroleum compounds taken up. Exceptions may occur if a large amount of fresh, light oil is mixed into the water column or if bottom sediments become contaminated. If nearshore sediments are contaminated, species that spawn in nearshore and shallow waters are more likely to be exposed to spilled oil than pelagic and benthic species.
- Penned finfish are more susceptible to tainting and contamination because they are not able to escape exposure. They are especially at risk if large amounts of oil mix into the water column.

- Shellfish are more likely than finfish to become contaminated from spilled oil because they are more vulnerable to exposure and less efficient at metabolizing petroleum compounds once exposed. Shellfish are generally less mobile and have more contact with sediments, which can become contaminated and serve as a long-term source of exposure.
- Among crustaceans, species that burrow are at the highest risk of exposure at spills where bottom sediments are contaminated, followed by species that utilize nearshore and estuarine benthic habitats.
- Bivalves are at high risk of contamination because they are sessile, filter- and deposit- feed, and occur in substrates in shallow subtidal and intertidal areas that are more likely to become contaminated.
- It is generally accepted that uptake and elimination rates both increase with temperature, though study results are somewhat contradictory.
- PAHs tend to accumulate to higher concentrations in lipid-rich tissues and organisms. Seasonal differences in tissue lipid content associated with spawning may influence uptake and elimination rates of PAHs in some marine species.
- Chronic exposure to hydrocarbons in water and sediments may reduce elimination capacity.

Summary of Literature on Uptake and Elimination

Most of the literature on oil and PAH uptake and elimination by marine organisms is based on laboratory studies using the water-soluble fraction or dispersed oil in aqueous exposures, or contaminated sediments. The organisms are typically exposed to a constant concentration for a period of time (often 24 hours for aqueous exposures; 28 days for sediment exposures) and then placed in clean water and monitored for tissue concentrations over time. The rate of elimination is often reported in terms of half-life, that is, the time it takes for the concentration of a compound to decrease by half.

Laboratory aqueous exposure concentrations are often an order of magnitude or two higher than expected at oil spills. At actual oil spills, organisms are more likely to experience spiked exposures in the water: concentrations that are initially high (for a few hours or less) and then rapidly decline as the oil disperses in three dimensions and degrades. Although laboratory exposure conditions often differ from those at actual spills, laboratory tests can be useful indicators of the relative rates of uptake and elimination among different oil compounds and concentrations, species, routes of exposure, and environmental conditions.

Laboratory study results indicate that PAH uptake from water is rapid, especially for finfish and crustaceans, which may be related, in part, to high ventilatory rates (Meador et al. 1995). For example, laboratory experiments have reported tainting after eight hours of exposure of salmon to 0.4 ppm of the water-soluble fraction of a crude oil (Ackman and Heras 1992) and after 4 hours of exposure of Arctic char to 50 ppm of a crude oil (Lockhart and Danell 1992). Dietary uptake from sediments is slower. Studies indicate that PAH uptake rates decrease with increasing molecular weight (Meador et al. 1995).

Elimination rates vary widely, by organism type, species, size, uptake pathway, oil type, temperature, and season. However, some generalizations can be derived from the literature. First, the half-lives of PAHs in organisms increase with molecular weight (Meador et al. 1995). Table II-6 shows this trend for PAHs in bivalves, which have limited ability to metabolize PAHs (the PAHs are listed in order of increasing molecular weight). It is important to note that the more persistent PAHs (with more than three benzene rings) are present in petroleum at very low levels. Elimination rates for

finfish, which metabolize PAHs more readily, would be faster than the rates shown in Table II-6. Second, passive release and metabolism of PAHs are slower in chronically exposed animals, as discussed earlier (Meador et al. 1995).

Table II-6. Half-lives of PAHs in bivalves based on laboratory tests of both water and sediment exposures (modified after Meador et al. 1995).

Compound	No. of Tests	Half-life, in days mean (range)
Naphthalene	3	1.6 (0.9-2)
Phenanthrene	6	3.3 (1.7-6.1)
Fluoranthene	6	9.9 (2.0-29.8)
Benzo(a)pyrene	6	12.3 (4.8-16)

Field data on the duration of taint and body burdens is limited to a few, well-studied spills. Table II-7 summarizes the available data by spill and organism type. These case studies show that wild finfish are seldom tainted, and the duration of taint is short (less than one month). Caged salmon, however, are more vulnerable to exposure, and taint may persist longer. At the *Braer* spill, in which a very large amount of a light crude oil was released over 12 days and elevated oil concentrations in water persisted in the vicinity of salmon farms for up to 50 days, the salmon closest to the spill reportedly remained tainted for nearly 200 days after the spill (Whittle et al. 1997).

Tainting of crustaceans has been reported for spills at which a light oil was naturally dispersed into the water column immediately after release. Some of the dispersed oil can mix with suspended sediments and accumulate on the seafloor surface, where lobsters, for example, can come into contact with the oil. It appears that epibenthic crustaceans readily uptake oil from sediments and are tainted at low PAH levels. Petroleum hydrocarbons tend to persist longer in crustaceans than finfish, perhaps partly because they are exposed by both water and sediment pathways. The sediment-associated oil has more of the higher-molecular weight PAHs that are more persistent and are eliminated more slowly (Meador et al. 1995).

Bivalves, particularly filter feeders, are more likely to have elevated levels of PAHs when the oil strands on intertidal beds or mixes into the water column over subtidal beds. Heavily oiled sediments can provide a source of chronic exposure, as at the *Sea Empress* spill where intertidal mussels remained contaminated in one heavily oiled bay for 19 months after the spill (Law et al. 1999). Once exposure ceases, elimination can be completed as rapidly as less than one month. Because bivalves accumulate oil compounds and eliminate them very slowly, they sometimes can be used as to indicate the extent and degree of oil exposure after an oil spill.

Table II-7. Presence and duration of taint and tissue contamination with petroleum compounds reported at various oil spills. Refer to Table I-1 for the details on spill location, date, oil type and volume, environmental conditions, and references.

Spill Name	Tissue PAH Concentration (µg/kg or ppb wet weight) and Persistence	Taint Persistence
Finfish		
T/V Sea Empress	Wild salmon: 12-186 Declined "rapidly"	Wild salmon: No taint
T/V Braer	Cod: 1.3-74 Haddock: 8-262 Plaice: 15-184 Whiting: 9-2,650 Lemon sole: 6-1,240 Dab: 25-2,160 All but dab reached background in 1 month; dab in 2 months Caged salmon: up to 14,000; rapid loss to 1,000 in 25 days, reached background in 5 months	Cod: No taint Haddock: 1 month Plaice: Suspect taint 2 months Whiting: No data Lemon sole: No taint Dab: 1 month Caged salmon: 7 months
T/B North Cape	Finfish: 5-1,100; 0 months because no increase over background was observed	All finfish: No taint in 416 samples
Crustaceans		
M/V Kure	Rock crab: 5-350; 0.5 months	Crab: No taint
M/V New Carissa	Dungeness crab: < 15	No sensory testing conducted
T/V Braer	Lobster: 112-1,060; 1 month Velvet crab: 94-308; 2 months Edible crab white meat: 19-281; brown meat: 104-1,390; 12 months for crabs	Lobster: 1 month Edible crab: No taint
T/B North Cape	Lobster: 0-33,150; 2.5-5 months	Lobster: 2.5-5 months
Bivalves		
M/V Kure	Oyster: 264-4,467; 0.5 months	Oyster: No taint
M/V New Carissa	Oyster: 70-1,200; 3 weeks	Oyster: No taint
T/V Sea Empress	Whelk: 50-3,800; 4 months Mussel: up to 19,500; 2.5-5 months Cockle: similar to mussels	Whelk: No taint Mussel: No data
T/V Braer	Whelk: 45-1,130; 12 months Scallop: 223-3,580; 17 months	Whelk: No data Scallop: Suspect taint 2 months
T/B North Cape	Steamer clam: 8,500-18,400; 3 months Oyster: 1,400-13,500; 3 months Mussel: 4,200-24,300; 3 months	Steamer clam: No taint Oyster: No taint Mussel: No taint
Refinery Spill, El Salvador	Oysters: 30,000; <1 month	Oysters: No data
T/V Exxon Valdez	Bivalves from four small areas were above 100; 1 year All other areas < 100	Bivalves: No data

Correlation between Taint and Body Burden

The specific compounds responsible for petroleum taint in seafood have not been unequivocally determined. Consequently, results of chemical analysis cannot yet be used to predict presence or absence of taint. Nevertheless, results from recent spills where both chemical and sensory testing have been conducted indicate a high degree correlation between presence of taint and presence of measured petroleum contaminants, or conversely, absence of both. The relationship, as well as tainting threshold, may vary somewhat depending on species, oil type, exposure pathway, and other unknown factors. Within a series of experiments using the same oil type and species, sensory panels can correctly rank the degree of taint with both tissue concentrations and exposure water concentrations. Some reported minimum concentrations of measured oil compounds in tissues that were determined by sensory testing to be tainted include 0.6 ppm for cod (Ernst et al. 1989b), 5 ppm for salmon (Heras et al. 1993), 9 ppm for plaice (Howgate et al. 1977), and 100 ppm in scallops (Motohiro and Iseya 1976). Sometimes it is possible to develop correlations for specific spills once a large enough data set is generated. For example, during the *Braer* spill, taint in caged salmon was readily perceived if the PAH concentration in the flesh was 1,000 ppb or greater (Whittle et al. 1997).

Laboratory studies have reported tainting thresholds in salmon, rainbow trout, scallops, and mussels (Ernst et al. 1989a; Ackman and Heras 1992; Davis et al. 1995; Heras et al. 1992, 1993; Jacques Whitford Environment 1992). The data are difficult to interpret because tissue levels are seldom measured, or they are reported as "ppm oil" rather than specific compounds, such as PAHs. More often, the studies correlate taint with the amount of oil in the exposure water, again usually reported as "ppm oil." These studies might provide some basis for predicting the potential for tainting for the combination of species and oil tested. However, it is not yet possible to make general predictions.

Conceptual Models of Exposure, Uptake, and Elimination

Because conditions change rapidly at oil spills, it is helpful to have conceptual models of the exposure pathways for a range of spill conditions. These conceptual models may help seafood managers in evaluating the risk of significant contamination of seafood and making decisions based on limited on-scene data. Table II-8 summarizes five conceptual models for exposure, uptake, and elimination at oil spills, applied to seafood. These models are based on actual spill data and supported by laboratory research, as cited in the previous sections. Please refer to these sections to find the citations supporting each of the conceptual models. It is important to note that during some spills more than one of the models will apply. Each of these models is briefly discussed.

Table II-8. Conceptual framework for seafood exposure to, uptake, and elimination of oil at spills.

Exposure Pathway	Exposure Conditions	Seafood at Risk	Tissue Contaminants	Elimination Rates
Dissolved oil fraction only via the water column	-Relatively calm, so slick does not disperse -Separation of dissolved plume from surface slick -Viscous oils that do not readily disperse -Exposure time is short	-Finfish in the area affected by the dissolved plume -Epibiota and filter feeding infauna where the dissolved plume contacts the bottom	-The more water-soluble compounds will dominate, i.e., the 2- and 3-ringed PAHs	-Most rapid because the compounds are more water-soluble and are quickly lost by diffusion and/or metabolism
Dissolved and particulate oil fractions via the water column and water surface	-Turbulence that mixes the oil as droplets into the water column -Light oils that are readily dispersed -Exposure time is short	-All biota in the water column -Epibiota and filter feeding infauna where the dissolved/ dispersed plume contacts the bottom -Intertidal biota (e.g. oyster/mussel beds)	-Same as the whole oil because particulate oil will dominate over dissolved -Over time heavier fractions will predominate as the more soluble fractions are depleted from the slick	- The range of PAHs in the whole oil is present in tissues; -Elimination slower for 4-5 ringed PAHs than for 2-3 ringed PAHs
Resuspension of contaminated sediments into the water column	-Resuspension of heavily oiled sediments from the shoreline or nearshore sediments -Exposure time is likely to be episodic and related to storms	-Nearshore filter feeders (epibiota and infauna)	-Will follow weathering pattern in the stranded oil -Over time, the less soluble, less degradable compounds will dominate	-Relatively slower rates because of the wide range of PAHs in the oiled sediments; chronic exposures may result in longer persistence even after exposure ends
Contaminated intertidal and subtidal sediments	-Oiled intertidal or subtidal sediments -Chronic exposure	- Infauna and some epibiota that are closely associated with bottom sediments, especially deposit feeders	-Same as above	-Same as above
Ingestion of contaminated food	-Usually occurs where sediments are contaminated -Exposure often chronic	-Predators, scavengers, and omnivorous feeders	-Highly variable and poorly understood	-Highly variable and poorly understood

1. Exposure to the dissolved oil fraction only

This exposure model assumes little or no dispersion of the whole oil into the water column, or that the dispersed oil re-coalesced into surface slicks, leaving behind a dissolved oil plume. Alternately, winds may transport the surface slick in one direction, whereas tidal currents can carry the dissolved plume in another direction. Under most conditions, exposure time to the water-soluble fraction of oil is short (in the range of hours to days) due to rapid dilution, evaporation, etc. Exposure concentrations are usually low (ppb range). Uptake by finfish and shellfish will be rapid and dominated by the most water-soluble compounds. However, elimination will also be rapid. Confounding factors can include longer exposure due to multiple or chronic releases, very slow dilution or flushing rates, and very cold temperatures that reduce metabolic activity of animals. Though many laboratory studies have shown rapid uptake of the water-soluble fraction, there are few examples of seafood harvest closures attributed to this pathway during oil spills, probably because the exposure concentrations are too low or rapidly diluted and do not result in persistent contamination.

2. Exposure to dissolved and particulate oil

This exposure model includes dispersed oil droplets that mix into the water column. This behavior could occur with light (low-viscosity) oils, turbulent conditions, or chemically dispersed oil. The total (both dissolved and dispersed) oil concentrations in the water column can be relatively higher (total oil concentrations up to low ppm) than with model 1. Exposure time to such high concentrations, however, is usually very short (in the range of hours to days), as oil concentrations rapidly decline with mixing in three dimensions. Tissue residues may include the full suite of PAHs in the whole oil, not just the water-soluble fraction. Thus, elimination rates are expected to be relatively slower, with the higher molecular weight PAHs having relatively longer half-lives. An example of this type of exposure is the *North Cape* oil spill.

3. Exposure to contaminated sediments re-suspended into the water column

Often, complete cleanup of oiled intertidal or subtidal sediments is not feasible and oil is left to weather and degrade naturally. The oiled sediments (or in some cases, free oil droplets) can be re-suspended during storm events, exposing nearby biota. Filter-feeders are at the greatest risk of exposure. Decline in tissue concentrations of contaminants from this pathway of exposure is likely to be delayed because of repeated exposures, presence of persistent, high-molecular-weight PAHs, and possibly slower overall elimination rates for organisms that are repeatedly exposed. This pathway of exposure has been documented for crude and heavy refined oils stranded on more sheltered shorelines (e.g., *Exxon Valdez* oil spill, *Sea Empress* oil spill).

4. Exposure to contaminated sediments.

Oiled intertidal and subtidal sediments can provide pathways of oil exposure via sediment ingestion to invertebrate deposit feeders, such as bivalves, and sediment grazers, such as shrimp and gastropods. Also, infauna can be exposed to dissolved oil in the sediment porewater, potentially contaminating tissues with the more soluble, lighter-molecular weight compounds. Decline in tissue concentrations will be delayed for organisms that are chronically exposed, and may be slow for the same reasons described in model 3 above. Intertidal sediments are more likely than subtidal sediments to be contaminated. Subtidal sediments are seldom contaminated, and if they are contaminated they are generally at lower concentrations than intertidal sediments. Sorbed oil might be more likely to dissolve, compared to pyrogenic PAHs derived from combustion of fossil fuels that are tightly bound to the sediments. This pathway of exposure has been documented at very few spills (most notably, the *Braer* oil spill). It is primarily associated with chronic pollution.

5. Ingestion of contaminated food.

This exposure model assumes that organisms uptake oil by eating contaminated food, not sediments ingested while feeding. Examples are oil droplet ingestion by copepods that are then eaten by finfish, or crabs feeding on oiled bivalves. Dietary uptake of PAHs is not very efficient, and decreases with increasing molecular weight.

III. MONITORING SEAFOOD FOR CONTAMINATION

Section II described information that can help determine the likelihood that spilled oil will expose and contaminate seafood. If it is decided that seafood is at significant risk, the next step is monitoring to determine whether seafood actually is contaminated, and to characterize the extent and degree of contamination. This section provides general guidelines for developing seafood sampling plans and conducting sensory and chemical testing of seafood samples for petroleum contamination.

Developing Seafood Sampling Plans

The first step in developing a sampling plan is defining the questions to be answered. Sampling should not begin before study objectives have been clearly established. Because every oil spill is a unique combination of conditions and the objectives of seafood sampling may vary from spill to spill, there is no standard sampling plan that can be applied to all seafood contamination monitoring studies. Generally, though, any sampling plan to monitor for potential seafood contamination from an oil spill should specify the study area, sampling locations, target species, number of samples to be collected, timing of initial and repeat sampling, sample collection methods and handling procedures, and analyses to be conducted. The statistical design must ensure sufficient statistical power to provide the information needed at the desired level of confidence to support seafood management decisions.

We suggest some general guidelines for designing a seafood-sampling plan below. For more detailed guidelines, see *Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories Volume 1: Fish Sampling and Analysis* by the U.S. Environmental Protection Agency (2000a). For more detailed sampling guidelines for sensory testing, see *Guidance on Sensory Testing and Monitoring of Seafood for Presence of Petroleum Taint Following an Oil Spill* (Reilly and York 2001). For general sampling guidance related to oil spills, see Mearns (1995).

Selecting sampling locations

In selecting sampling locations, all likely pathways of oil exposure should be identified (e.g., surface slicks, dispersed or dissolved oil in the water column, submerged oil associated with bottom sediments), as discussed in Section II, so that risks to specific fisheries can be evaluated. Inclusion of commercial, recreational, and subsistence harvest areas should be considered.

Collection of pre-exposure samples from the spill area or samples from appropriate unexposed reference areas is extremely important because they can provide information on background levels of contamination in the spill area. Petroleum hydrocarbons are ubiquitous in environmental samples, so we cannot assume that all petroleum hydrocarbons measured in a sample or all increases over time are a result of an oil spill. Furthermore, monitoring often continues until the level of contamination returns to "background." Reference samples are key to determining the range of background concentrations and the baseline against which changes over time will be evaluated.

The best reference samples are pre-spill samples taken in areas not yet oiled but in the potential path of the oil ("before" can be compared with "after" exposure). If pre-spill sampling is not possible, unexposed reference sites comparable to exposed sites can be selected for sampling. However, site histories and differences in the characteristics of the sites should be carefully evaluated to determine whether there are significant differences between the exposed and reference areas. Often, areas that escape oiling do so because they differ fundamentally from exposed areas (for example, bays that face different directions), and so would not be expected to exhibit the same "background" conditions.

Any differences between reference and exposed sites must be considered when analyzing and interpreting results.

National monitoring programs such as NOAA's National Mussel Watch Program can provide valuable pre-spill data for determining historical ranges of background concentrations of PAHs in shellfish at several locations around the country (Mearns et al. 1998, 1999). When available for an area, PAH data from the NOAA Status and Trends Program (including the National Mussel Watch Program) or other monitoring programs may help determine normal background levels and seasonal patterns in contaminant levels.

Selecting target species to be sampled

Evaluating risk to human health from seafood consumption usually is a primary purpose of seafood sampling, so including species harvested commercially, recreationally, and for subsistence use may be important. Species that are present throughout the area of concern may be most appropriate for sampling if results are to be compared spatially or if the results are to be used to make statistical inferences to the entire area.

Hydrocarbon uptake and elimination rates vary widely among species, as described in Section II. Finfish, for example, quickly metabolize and eliminate PAHs. Bivalves generally tend to bioaccumulate most contaminants and often serve as good indicators of the potential extent, degree, and persistence of contamination. On the other hand, some shellfish species stop feeding or passing water over their gills at extreme temperatures and, consequently, may exhibit low uptake rates under certain conditions. Consider such differences when selecting species for monitoring and comparing results among species.

Sampling frequency and duration

Monitoring generally should continue until contaminant levels reach background levels or pre-determined acceptable levels. Periodic sampling before those levels are reached can reveal trends in contaminant levels. Appropriate monitoring frequency and duration will depend on spill conditions, such as oil type and volume spilled, flushing rates of affected water bodies, and the degree of exposure to wave action of contaminated shorelines. Appropriate monitoring frequency and duration will also depend on the species exposed and exposure duration. Finfish generally eliminate hydrocarbons within days or weeks, whereas bivalves may require several weeks or months. Elevated levels of petroleum compounds in bivalves have been detected for years at some sites where high levels of oil persist in adjacent sediments. Time of year should also be considered in some climates because elimination rates may be slower in cold temperatures. Other factors to consider with regard to monitoring frequency are the turnaround time for sample analysis and time required for the evaluation team to meet, interpret the results, and decide on the need for further sampling. Sampling plans may need to be adjusted over time as conditions change and as monitoring results provide new information on the fate of the oil and on which pathways of exposure are significant.

Sample collection and handling

The seafood-sampling plan should specify all details about sample collection. This includes the areas to be sampled, number of samples to be collected from an area (to meet statistical objectives), number of organisms or quantity of tissue to be composited (to meet analytical requirements), size of organisms to be collected, tidal elevations for collection (in the case of intertidal invertebrates), method of marking or recording exact sampling locations, and field notes to be recorded.

The sampling plan should also specify how seafood samples should be handled. This includes any field preparation, packaging and temperature requirements (for example, wrapping in foil, keeping in a cooler at 4°C or below, and freezing within a specified period of time), labeling, and any chain-of-custody requirements during transport to the analytical laboratory. (An example chain-of-custody form is included in the appendix). Only live animals should be collected for seafood analysis. The edible portion, which may vary culturally, is usually the portion of interest. Seafood samples collected for sensory testing generally should be handled as they would be during commercial, recreational, or subsistence harvest and transport.

Procedures should be followed to prevent cross-contamination in the field (such as preventing exposure of samples or sampling equipment to exhaust fumes and engine cooling systems on vessels) and to maintain the integrity of the samples. Likewise, good laboratory practices should be employed to prevent contamination of samples during preparation and analysis.

Testing Seafood for Contamination and Tainting

Generally, two different types of evaluations can be conducted after oil spills to determine whether seafood is contaminated. Sensory testing determines whether seafood is tainted, i.e., if it has an off-odor or off-flavor. Chemical analysis determines whether tissues are contaminated with targeted compounds. Detailed methods of chemical analysis can indicate the presence as well as the quantity of specific contaminants in tissues. These results can be used to evaluate risk to human health through consumption of contaminated seafood (as described in Section 5). Summaries of these types of seafood testing are described below.

Sensory evaluation of seafood for presence of petroleum taint

When an oil spill occurs, local seafood resources may be exposed to petroleum compounds that affect their sensory qualities; that is, smell, taste, and appearance. Even when seafood from a spill area is considered acceptable with regard to food-safety, flavor and odor may still be affected, negatively impacting the seafood's palatability, marketability, and economic value. Furthermore, tainted seafood is considered by the U.S. Food and Drug Administration to be adulterated and, therefore, is restricted from trade in interstate commerce.

Overview of sensory testing of seafood

Tainted seafood is defined as containing abnormal odor or flavor not typical of the seafood itself (ISO 1992). Under this definition, the odor or flavor is introduced into the seafood from external sources and excludes any natural by-products from deterioration due to aging during storage, decomposition of fats, proteins, or other components, or due to microbial contamination normally found in seafood. Taint is detected through sensory evaluation, which has been defined as "the scientific discipline used to evoke, measure, analyze and interpret those reactions to characteristics of foods and materials as perceived through the senses of sight, smell, taste, touch and hearing" (Food Technology Sensory Evaluation Division 1981). Humans have relied for centuries on the complex sensations that result from the interaction of our senses to evaluate quality of food, water, and other materials. In more recent times, sensory testing has developed into a formalized, structured, and codified methodology for characterizing and evaluating food, beverages, cosmetics, perfumes, and other commercial products. Sensory evaluation techniques are routinely used commercially in quality control, product development, and research. Sensory testing can be either subjective or objective. Subjective testing measures feelings and biases toward a product rather than the product's attributes. For objective

testing, highly trained assessors use the senses to measure product attributes. Testing of seafood for petroleum taint should be completely objective and should be conducted by highly trained analysts.

Objective sensory testing serves as a practical, reliable, and sensitive method for assessing seafood quality. Only human testers can measure most sensory characteristics of food practically, completely, and meaningfully. Though advances continue to be made in developing instrument-based analysis, human senses remain unmatched in their sensitivity for detecting and evaluating organoleptic characteristics of food. The U.S. Food and Drug Administration and NOAA's National Marine Fisheries Service routinely employ sensory evaluation in inspecting seafood quality. Seafood inspectors are essentially sensory analysts, or assessors, who work as expert evaluators in the application of product standards. A major objective of seafood sensory inspection is to evaluate quality with regard to decomposition of fisheries products. Sensory analysis can also provide information on presence of taint from external sources, such as spilled oil and chemicals.

Sensory panels

Objective sensory evaluation of seafood is usually conducted using a panel of trained and experienced analysts. Sensory analysts must be screened for sensitivity and then trained in applying established sensory science methodology. Participation in calibration or "harmonization" workshops ensures uniform application of sensory evaluation criteria for particular types of contaminants, including standard terminology and consensus on levels of intensity of sensory characteristics. Descriptive analyses and references are used to yield results that are consistently accurate and precise.

There are different types of sensory analysts, which function differently and have specific selection, training, and validation requirements. *Trained assessors* are sensory analysts selected and trained to perform a specific task. *Expert assessors* are the most highly trained and experienced category of sensory analyst. Expert assessors generally evaluate product full-time, function independently, and often are used in quality control and product development. Examples of products evaluated by expert sensory assessors include wine, tea, coffee, and seafood. Through extensive standardized training and experience with sensory methodology, these expert assessors have become extremely objective and evaluate quality with a high degree of accuracy and precision. Seafood inspectors fall into the category of expert assessors, and can make consistent and repeatable sensory assessments of quality characteristics of seafood as they relate to grade level or decisions to accept or reject product.

The number of panelists needed depends on the level of expertise and experience of the analysts used. For panels of expert assessors, such as NMFS and FDA seafood inspectors, usually only three to five analysts are needed. If less experienced analysts are used, a larger number of panelists is recommended. Whenever possible, use of expert seafood assessors, such as seafood inspectors, is recommended for evaluation of seafood for presence of petroleum taint. Extensive product knowledge and experience enable seafood inspectors to very accurately distinguish variations related to product processing, storage, deterioration, etc. from taint due to external sources. Some seafood inspectors for NMFS and FDA have had specialized training for detecting petroleum taint in seafood and experience evaluating seafood samples at oil spills. If called upon, these specialized inspectors are available to conduct sensory evaluation of seafood during spill events.

Sensory evaluation procedures

Applied as a science, sensory evaluation should be conducted under specific, highly controlled conditions in order to prevent extraneous influences in the testing environment from affecting panelists' sensory responses. Accordingly, sensory testing is best conducted in facilities specifically designed for sensory testing. The NMFS Seafood Inspection Branch maintains several such laboratories around the country. Seafood samples collected during a spill event can be shipped to these laboratories for sensory evaluation. In most cases, NMFS and FDA recommend that samples be shipped and evaluated in the same manner as they normally are shipped and sold (i.e., fresh, live, frozen). When this is not pos-

sible, as may be the case for oil spills in very remote areas, sensory analysts can conduct evaluations at the scene of an incident.

All sensory testing should be conducted under the supervision of a sensory professional, who designs and implements the sensory testing procedure. A trained "facilitator" should coordinate sensory analysis . The facilitator conducts the testing, including receiving, preparing, and presenting samples to the expert sensory panel, and collecting the resulting data in a scientific and unbiased manner. All of these steps should be conducted according to standardized procedures under highly controlled conditions. Suspect samples are presented to assessors in blind tests, along with control or reference samples. Samples are first smelled raw, then smelled cooked, and finally tasted by each panelist independently to determine whether petroleum taint is present. A sensory professional statistically analyzes panelist's responses to determine whether samples pass or fail with regard to presence of petroleum taint. These results, in turn, help seafood managers determine whether restrictions are needed on seafood harvest or marketing from the spill area due to tainting.

In that we are not certain which compounds in petroleum are responsible for taint perceived by humans, chemical analysis cannot yet substitute for sensory testing in determining whether a taint is present. It has been suggested that the principal components of crude and refined oils responsible for tainting include the phenols, dibenzothiophenes, naphthenic acids, mercaptans, tetradecanes, and methylated naphthalenes (GESAMP 1977). The human olfactory system generally is very sensitive to phenolic and sulfur compounds, even though they are minor components of oil.

In 2001, NOAA published a technical guidance document on appropriate sensory methodology to objectively assess seafood for the presence of petroleum taint. Written by sensory scientists with NOAA's National Marine Fisheries Service Seafood Inspection Program and Canada's Food Inspection Agency, in cooperation with the U.S. Food and Drug Administration, *Guidance on Testing and Monitoring of Seafood for Presence of Petroleum Taint Following an Oil Spill* comprehensively describes recommended standard procedures, including collection, preservation, and transport of seafood samples, for sensory evaluation. The guidance is intended to assist in conducting scientifically sound and legally defensible sensory tests on seafood during oil spill response, with adequate and appropriate quality control.

Chemical testing techniques for petroleum contaminants in seafood

Chemical testing of seafood often is conducted after an oil spill to determine whether seafood tissues are contaminated with petroleum compounds. Both detailed and screening methods of analysis can be employed. Below, we summarize methods typically used after past oil spills, including some of their advantages and disadvantages.

Detailed methods of chemical analysis: gas chromatography/mass spectrometry

Detailed chemical analysis of seafood after oil spills typically is conducted using gas chromatography and mass spectrometry (GC/MS), which measures individual PAHs at very low detection levels and provides a PAH pattern (or fingerprint) to compare to that of the source oil. Prior to analysis, hydrocarbons are extracted from seafood tissue samples and the extract is split into three fractions: 1) the saturated hydrocarbons fraction (f_1), containing the n-alkanes, isoprenoids, steranes and triterpanes; 2) the aromatic hydrocarbon fraction (f_2), containing the PAHs and sulfur heterocyclics; and 3) the polar hydrocarbon fraction (f_3), containing the nitrogen heterocyclic compounds. Recovery standards appropriate to each fraction are added (Lauenstein and Cantillo 1993).

The PAHs in the f_2 fraction generally are of greatest concern with regard to risk to human health. The gas chromatograph separates targeted PAH compounds yielding a retention time that, in combination with the mass spectra from the mass spectrometer, enable detailed identification of individual compounds by their ion masses. The method often used is usually referred to as "Modi-

fied" EPA Method 8270, which is EPA Method 8270 for semi-volatile compounds modified to include quantification of the alkyl-substituted PAH homologues, in addition to the standard PAH "priority pollutants." Table II-3 lists the PAHs and their alkyl homologues usually included in this analysis. In oil, alkylated homologues of PAHs are more predominant than parent PAH compounds, often by an order of magnitude. This is in contrast to pyrogenic (combustion) and other potential PAH sources. The detailed chemical fingerprint provided by GC/MS analysis enables differentiation among sources of PAHs found in the sample. Contamination from a specific spill can be distinguished from background sources of contamination, such as PAHs derived from combustion sources. GC/MS can also measure analytes other than PAHs to help with fingerprint analysis of oil or to track oil weathering. The GC/MS can be run in the selected ion monitoring (SIM) mode, rather than the full-scan mode, to increase the minimum detection levels (MDL) of the individual parent and selected homologue PAHs by a factor of 10 to 40. Minimum detection levels for individual PAHs are very low, in the range of parts per billion (ng/g) in tissue. The quantitative results for specific, targeted PAHs can be used to assess whether levels detected pose a risk to human health through seafood consumption.

Normal turnaround time for analysis of tissue samples for PAHs is approximately two weeks. Fast turnaround time is approximately three days for a batch of samples. Costs for GC/MS-SIM analysis of tissues are relatively high, starting from about $750 per sample, plus premiums of 50-100% for fast turnaround. The sample-processing rate depends on the throughput capabilities of the laboratory and the degree of quality control (QC) of the data before the results are released, ranging from approximately 20 to a maximum of 100 samples per week.

Data Reporting and Interpretation

The importance of data reporting and interpretation should not be underestimated in planning seafood safety monitoring programs after oil spills. Some simple steps can be taken to help avoid confusion and prevent incorrect conclusions. For example, the analytical laboratory should include at least the following information for all analytical data reported:

Sample "Header" Information

- *Sample Name or Field ID: the sample name or number assigned by the sampler*
- *Sample Type: e.g., sample, field blank, trip blank, procedural blank, QC*
- *Batch No.: analytical batch number (so samples run as a batch can be identified, particularly if problems are found with a batch run)*
- *Matrix: e.g., water, sediment, tissue, oil*
- *Percent Moisture: for tissue and sediment samples*
- *Sample Size: weight or volume of sample used for analysis*
- *Collection Date: date the sample was collected*
- *Extraction Date: date the sample was extracted*
- *Analysis Date: date the sample was analyzed*
- *Analysis Method: EPA Method or other description*
- *Surrogate Corrected?: Are the reported concentrations corrected for surrogate recovery?*
- *Method Detection Limit: the minimum detection level*
- *Units: units in which the concentration is reported, including whether concentrations are wet weight or dry weight (for tissue)*

Analyte Data

- *Individual and Total PAH concentrations*
- *Surrogate Recovery (%): for every sample*
- *Key to Data Qualifiers: The lab should include a key to any qualifiers used to flag reported values that have some kind of data accuracy issue. For example, two standard qualifiers used under the USEPA Contract Laboratory Program guidelines (USEPA 1994) are:*
- *U = the analyte was analyzed for, but was not detected above the reported sample quantitation limit*
- *J = the analyte was positively identified; the associated numerical value is the approximate concentration of the analyte in the sample*

Analysis of the source oil, if available, is needed to enable fingerprint comparisons. Only expert petroleum hydrocarbon chemists should interpret fingerprints because the complex processes of oil weathering and uptake result in variable PAH patterns in organisms (Sauer and Boehm 1995). Also, patterns can be difficult to interpret in samples collected from areas with high background levels of contamination.

Caution is advised when comparing analytical results for samples of different types, or samples collected from different areas or at different times. Before drawing conclusions, consider any differences in the analyses conducted or the way the data are reported. Examples of differences to watch for include:

- the units in which results are reported, and whether reported concentrations are dry or wet weight;
- whether the lists of analytes and minimum detection limits for individual PAHs are the same;
- whether reported concentrations have been corrected for surrogate recovery; and
- whether reported concentrations have been lipid-normalized. As described in Section II, PAH uptake and retention tend to increase with the increasing lipid content of tissues. Consequently, differences in lipid content may need to be considered when comparing and interpreting analytical results over time or among different organisms.

Rapid screening methods of analysis

Rapid, low-cost analytical methods, generally known as screening methods, can be employed to identify contaminated samples and prioritize them for detailed analysis. Detailed methods of analysis for PAHs in tissue are time-consuming and expensive. The large number of samples often collected after an oil spill can quickly overwhelm laboratory capacity and strain resources. Screening methods of analysis can rapidly process large numbers of samples to yield semi-quantitative estimates of contaminant concentrations and allow ranking of samples by degree of contamination. Used in a tiered approach, screening methods can identify the most contaminated samples, prioritizing or reducing the number of samples that need to be processed by detailed analytical techniques, such as GC/MS.

For example, in response to the need to analyze large numbers of subsistence seafood samples collected after the *Exxon Valdez* oil spill in Prince William Sound, Alaska, NOAA's Northwest Fisheries Science Center used reverse-phase, high performance liquid chromatography (HPLC) with fluorescence detection to screen for metabolites of aromatic compounds in finfish bile (Krahn et al. 1982, 1984, 1986, 1992, 1993a, 1993b, 1993d). Finfish rapidly metabolize aromatic compounds and concentrate the resulting metabolites in bile for excretion, often at concentrations that are orders of magnitude greater than those in edible tissue. Using this rapid, low-cost method, hundreds of finfish tissue samples were screened for indication of exposure to petroleum contaminants, enabling GC/MS analy-

ses to be focused on selected samples to confirm presence and quantities of individual contaminants. Hufnagle et al. (1999) has developed an HPLC/UV fluorescence screening method for rapidly measuring aromatic compounds in invertebrate tissues. This screening method was used successfully on lobster samples collected after the *North Cape* oil spill off the coast of Rhode Island in 1996. For details on a rapid screening method for parent aromatic compounds in sediments see Krahn et al. (1991, 1993c).

Screening analyses, such as the HPLC/fluorescence method described above, generally can be completed in rapid turnaround time (within 24 hours) and can be conducted on a research vessel or onshore lab. Rapid availability of results enables sampling modifications based on indications of exposure. This can be very helpful during the critical early phases of an oil spill response, when decisions regarding closing or otherwise restricting seafood harvest may be made.

The utility of HPLC/fluorescence and other screening methods, however, is more limited than detailed methods of analysis. For example, though it may be possible to recognize chromatographic patterns associated with characteristic classes of petroleum products, HPLC/fluorescence screening does not produce a detailed "fingerprint" similar to the results acquired from GC/MS. Consequently, HPLC/fluorescence usually will not enable differentiation between background contamination sources and the spilled oil, especially in very polluted areas. Since HPLC/fluorescence screening does not quantify individual aromatic compounds, the results cannot be used to assess risk to human health from consumption of contaminated seafood. Furthermore, measurement of fluorescent aromatic compounds in bile is not a standard analysis, limiting temporal and spatial comparisons using historical data sets. Lastly, HPLC/fluorescence screening for fluorescent aromatic compounds in bile is a specialized technique, and laboratory availability and expertise needed to conduct the analyses reliably may be limited.

Water Monitoring

Water samples often are collected and analyzed as part of the initial spill response and assessment. Seafood safety managers can use these results to help estimate the extent and duration of seafood exposure to oil in the water column. Monitoring of water concentrations may also be important if water-quality criteria are applied as a condition for re-opening a closed fishery or removing other harvest restrictions.

Oil concentrations in the water column generally peak early after an oil spill and, in most cases, rapidly decline to background levels within days to a week, as was the case for example at the *New Carissa* oil spill (Payne and Driskell 1999). Accordingly, if water sampling is to be conducted, initial sampling should commence very soon after an oil spill occurs. Oil may persist longer than usual in the water column if there are multiple or ongoing oil releases, if the released volume is extraordinarily large, or if large volumes of oil are physically dispersed. After the *Braer* oil spill, for example, elevated oil concentrations were detected in the water column as long as 50 days after release (Davies et al. 1997). Dissolved and dispersed oil plumes in the water column are driven by currents and so may have a very different spatial distribution than surface slicks, which are driven primarily by wind.

Under the authority of the Clean Water Act (63 FR 68354-68364), EPA has issued national recommended water-quality criteria for priority toxic pollutants to be used by states and tribes in adopting water quality standards. EPA has issued water-quality criteria for protection against human health effects for three mono-aromatic hydrocarbons and eight PAHs (listed in Table III-1). These particular compounds, however, are present in crude oils and refined products at very low levels and constitute a tiny percentage of the PAHs normally detected in water samples after an oil spill. None of the water quality criteria to protect aquatic communities (both freshwater and saltwater) issued by EPA are for PAHs. EPA has issued recommended water quality criteria for organoleptic effects for 23 chemicals, though not for any of the compounds present in petroleum products. Some states have established state water quality standards for PAHs in their coastal waters.

Table III-1. National recommended water quality criteria for priority toxic pollutants for protection against human health effects (63 FR 68354).

PAH Priority Pollutant	Human health criteria for consumption of Water + Organism (μg/L)	Human health criteria for consumption of Organism Only (μg/L)
Benzo(a)anthracene	0.0044	0.049
Benzo(a)pyrene	0.0044	0.049
Benzo(b)fluoranthene	0.0044	0.049
Benzo(k)fluoranthene	0.0044	0.049
Dibenzo(a)anthracene	0.0044	0.049
Fluoranthene	300	370
Fluorene	1,300	14,000

Sediment Monitoring

Sediment monitoring can be included as part of a post-spill monitoring program to determine whether sediments may be a potential chronic source of oil exposure to adjacent seafood collection sites, particularly at intertidal sites where bivalves are harvested. Sediment sampling also may facilitate fingerprint analysis of PAHs in tissues by providing the PAH pattern in contaminated sediments, which may be different than the PAH pattern in the fresh source oil. It is important to recognize, however, that sediments often contain high levels of background PAH contamination, particularly in urban areas and harbors. PAHs and other contaminants detected may not be related to a particular oil spill. Also, characterization of sediment contamination can be difficult because of the inherent heterogeneity of intertidal sediments over space, depth, and time.

There are no national sediment quality criteria for PAHs in marine or freshwater sediments. Some states have established sediment quality standards and cleanup screening levels to prevent adverse biological effects. How these standards would relate to seafood adulteration or safety issues is unclear.

IV. SEAFOOD RISK ASSESSMENT

Several different endpoints can be considered when assessing risks posed to human health from consuming contaminated seafood. These include both carcinogenic and non-carcinogenic effects to the general population, as well as to particularly susceptible segments of the population such as children, pregnant women, and subsistence seafood consumers. Human epidemiological studies, when available, and laboratory studies involving animals are used to assess the likely effects of contaminants at various exposure levels.

As discussed in Section II, petroleum oils are composed of complex and variable mixtures of hundreds of different hydrocarbon compounds. Of these, polycyclic aromatic hydrocarbons (PAHs) are typically of greatest concern with regard to health effects because of their relative persistence and carcinogenicity. Evidence from occupational studies of workers exposed to mixtures of PAHs indicates that many of these compounds may be carcinogenic to humans. Individual PAHs that are considered to be probable human carcinogens include benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[k]fluoranthene, chrysene, dibenz[a,h]anthracene, and indeno[1,2,3-cd]pyrene (IRIS 1994). Most of the data gathered from laboratory studies provides information on carcinogenic effects of lifetime exposure to PAHs. Information on non-carcinogenic effects is limited. Consequently, cancer generally is the primary endpoint considered when assessing potential risks to human health from consumption of seafood from an oil spill area.

Overview of Cancer Risk Calculations for PAHs in Seafood

Most seafood risk assessments conducted after oil spills in the U.S. have followed an approach used by the U.S. Food and Drug Administration (USFDA) in 1990 after the *Exxon Valdez* oil spill in Prince William Sound, Alaska. At the request of the Alaska Oil Spill Health Task Force, a group established after the spill to conduct a survey and assess the impact of the spill on subsistence food supplies, USFDA conducted a risk assessment and provided an advisory opinion on the safety of aromatic hydrocarbon residues in subsistence seafood in the spill area (Bolger et al. 1996; Bolger and Carrington 1999). This approach uses a set of calculations to determine finfish or shellfish PAH tissue concentrations, expressed in benzo[a]pyrene (BaP) equivalents (µg/kg), above which an acceptable risk level for cancer is exceeded. The values for several variables in these calculations can be adjusted on a case-by-case basis, depending on local seafood consumption levels of the exposed population, average body weight of the exposed population, estimates of exposure time for a particular spill, and the cancer risk level deemed acceptable. This approach to calculating seafood advisory or action levels has since been used after several other oil spills, including the *North Cape* spill in Rhode Island, the *Julie N* spill in Maine, the *Kure* spill in California, and the *New Carissa* spill in Oregon.

The basic equation and input parameters are described below:

$$\text{Advisory or Action level (µg/kg) BaP equivalents} = \frac{(RL)(BW)(AT)}{(SF)(ED)(CR)}$$

Acceptable Risk Level (RL): The acceptable risk level is the maximum level of individual lifetime carcinogenic risk that is considered "acceptable" by risk managers. The typical RL used in cancer risk calculations is 1×10^{-6}. In the case of PAHs, this implies that exposure to PAHs in seafood below a specified tissue concentration level at a defined consumption rate over the defined exposure period would yield a lifetime cancer risk of no greater than 1 in 1,000,000. Some states consider higher risk levels, such as 1×10^{-5} (a lifetime cancer risk of no greater than 1 in 100,000) to be acceptable.

A risk level of 1×10^{-6} was used in the risk calculations done by USFDA for the Exxon Valdez oil spill, as well as those done by the State of Rhode Island for the *North Cape* oil spill, the State of California for the *Kure* oil spill, and the State of Oregon for the *New Carissa* oil spill. A risk level of 1×10^{-5} was used in the risk assessment conducted by the State of Maine for the Julie N oil spill and the State of Alaska for the *Kuroshima* oil spill.

Body Weight (BW): The value for body weight used in risk calculations is intended to represent the body weight of an individual consumer (kg). An average body weight of 60-70 kg (132-154 lb) is often used for adults in the general U.S. population. If a particular group of at-risk consumers is considered in a risk calculation, alternative body weights may be used. For instance, children or subsistence harvesters may have lower average body weights than 60-70 kg. Because allowable consumption limits at a certain seafood tissue concentration are linearly related to body weight, risk assessors should consider the actual body weights of the targeted population.

Averaging Time (AT): A typical averaging time value used in cancer risk calculations is 70 years. This value represents the average length of a human lifetime, which is the time period of interest for examining cancer as an endpoint.

BaP Cancer Slope Factor (SF): The cancer slope factor, or cancer potency (q^{1*}), is derived from dose-response data obtained from human epidemiological and animal toxicity studies (USEPA 2000b). High doses of the contaminant of interest are often used in dose-response studies, and extrapolation of the data to lower doses that may be encountered by the general population is often necessary. Cancer potency is estimated as the 95-percent upper confidence limit of the slope of the dose-response curve in the low-dose region. This method provides a conservative estimate of the potential cancer risk of a contaminant. The actual risk may be significantly lower. The USEPA (2000b) has used a cancer potency factor of 7.3 per mg/kg/day to calculate monthly consumption limits for the general population over a range of PAH tissue concentrations in finfish. This same potency value was used in cancer risk calculations for the *New Carissa* and *Julie N* oil spills. A cancer potency factor of 9.5 mg/kg/day, established by the State of California EPA, was used to calculate carcinogenic risk associated with consuming contaminated shellfish following the *Kure* spill in California.

Exposure Duration (ED): The exposure duration is the time period over which an individual is exposed to a contaminant. When calculating risks associated with seafood consumption following an oil spill, the exposure duration is equivalent to the time interval over which an individual consumes contaminated seafood harvested from the spill zone. Exposure duration varies depending on spill conditions. The default assumption for risk assessments generally is 70 years, the average time for a lifetime exposure. Unlike some other contaminants, however, PAH concentrations in contaminated finfish and shellfish decrease over time and exposure levels will decline, eventually dropping to background concentrations. Consequently, exposure periods much shorter than a 70-year lifetime exposure assumption are more realistic and appropriate for PAHs, particularly for oil spills because they are typically very short-term, pulsed contamination events.

An exposure duration of two years was assumed for the risk calculations for the *New Carissa* and *Kure* oil spills. An exposure duration of five years was used for the *North Cape* oil spill calculations (Mauseth et al. 1997). More conservative exposure assumptions have been made at other spills. Both ten- and 30-year exposure durations were used in risk calculations for the *Julie N* oil spill. Consumption risks for the *Exxon Valdez* spill were calculated for both ten and 70-year (lifetime) exposure durations.

Seafood Consumption Rate (CR): Typically, consumption rates are calculated for average and upper-end consumers and correspond to the quantity of seafood (units expressed as grams) that an individual may consume per day. The values used for serving sizes and frequency of seafood meals are often adjusted, due to the significant variability in seafood consumption among individuals and particular groups.

Data from national surveys, such as the Continuing Survey of Food Intake by Individuals (CSFII) conducted by USDA, can be used to help estimate national seafood consumption rates. The consumption rate typically used for the average U.S. seafood consumer is 7.5 grams/person/day. This value is derived from the assumption that an average seafood consumer eats one 8-ounce (227 grams) seafood meal (such as a fish fillet) once a month (per 70 kg consumer body weight for adults) (USEPA 2000b).

The carcinogenic risk assessment conducted after the *Exxon Valdez* oil spill used seafood consumption rates calculated from subsistence harvest survey data (Bolger et al. 1996; Bolger and Carrington 1999). Residents of Alaska Native communities rely on local finfish and shellfish resources for significant portions of their diets. The Alaska Department of Fish and Game Division of Subsistence Consumption had conducted household harvest studies before the spill (Fall 1999; Scott et al. 1992). Subsistence consumption rates were estimated to be 89 grams/person/day for salmon, 52 grams/person/day for other finfish, 21 grams/person/day for crustaceans, and 2 grams/person/day for bivalve mollusks. Note that these consumption levels are much higher than those derived for the general U.S. population from national survey data, described above.

The *New Carissa* and *Kure* risk assessments used shellfish consumption rates for the average commercial product consumer of 7.5 g/day (Challenger and Mauseth 1998; Gilroy 2000). An upper-end consumption rate of 32.5 g/day (one meal/week) for the *New Carissa* risk assessment was based on a reasonable estimate for local recreational harvesters/consumers (Gilroy 2000). Upper-end consumption rates of 50g/day and 30g/day were used for the *Kure* and *North Cape* risk assessments, respectively (Mauseth et al. 1997). For the *Julie N* oil spill, average consumption rates of lobster were assumed to be 13.6 g/day.

Seafood Advisory and Action Levels from Previous U.S. Oil Spills

The action or advisory levels resulting from cancer risk calculations differ among spills, depending on the assumptions made and input values selected. At the *New Carissa* oil spill, the Oregon Health Division calculated action levels for average and upper-end shellfish consumers of 45 ppb BaP equivalents (BaPE) and 10 ppb BaPE, respectively (Gilroy 2000). Action levels derived by the California Department of Health Services for average and upper-end shellfish consumers following the *Kure* spill were 34 ppb BaPE and 5 ppb BaPE, respectively. At the *North Cape* oil spill, the Rhode Island Department of Health essentially applied a BaPE criterion of 20 ppb for the maximally exposed lobster consumer over the five-year exposure duration. Action levels calculated by the Maine Bureau of Health for lobster consumption after the *Julie N* oil spill for ten and 30- year exposure durations were 50 ppb and 16 ppb BaPE, respectively. Advisory levels for subsistence consumers after the *Exxon Valdez* oil spill, assuming a ten-year exposure period, were three ppb BaPE for salmon, five ppb BaPE for finfish, 11 ppb BaPE for crustaceans, and 120 ppb BaPE for bivalve mollusks. Advisory levels based on a lifetime exposure assumption were approximately an order of magnitude lower. None of the finfish or shellfish samples collected from harvesting areas near Prince William Sound exceeded these advisory levels. Interestingly, the upper-bound lifetime cancer risk for Alaskan subsistence seafood consumers eating the most contaminated bivalve mollusks from the spill area was calculated to be two orders of magnitude lower than the lifetime risk calculated for consumers of locally smoked salmon (Bolger et al. 1996).

At several of these spills, the calculated action levels were used as recommended levels for reopening harvest of closed seafood fisheries. For example, at the *New Carissa* oil spill, shellfish were considered safe if all samples contained less than 10 ppb BaP equivalents. If any shellfish tissue levels were above 45 ppb BaP equivalents, shellfish in those areas would be considered unsafe, and further

monitoring considered necessary. If samples contained more than 10 ppb but less than 45 ppb BaP equivalents, the need for further monitoring would be assessed on a case-by-case basis. A similar tiered approach was used at the *Kure* oil spill. If all samples contained less than 5 ppb BaP equivalents, shellfish beds could be reopened. If any samples contained between 5 and 34 ppb BaP equivalents, the need for further action before reopening would be assessed. If any samples contained more than 34 ppb BaP equivalents, additional sampling and environmental monitoring prior to reopening would be considered.

The Equivalency Approach for Risk Assessment

The equivalency approach used in relative cancer risk assessment is a method used for assessing the risk of exposure to a mixture of several different compounds that are related in terms of chemical and biological activity. Rather than calculating individual risks for each compound, one component of known potency is used as a standard. Concentrations of each of the other compounds are adjusted based on their estimated potency relative to the standard, to calculate an equivalent concentration for the standard. Summing the equivalent concentrations yields a single number from which the cancer risk can be estimated (ICF-Clements 1988; Bolger and Carrington 1999).

This toxicity equivalency approach has been widely used for mixtures of dioxins and furans, for example. The relative potencies of individual dioxin and furan compounds are expressed in terms of 2,3,7,8-tetra-chlorodibenzo-*p*-dioxin (2,3,7,8-TCDD) equivalents. 2,3,7,8-TCDD was chosen as the standard by which the potency of individual dioxin and furan compounds are estimated because most laboratory studies on the effects of dioxins have been conducted using 2,3,7,8-TCDD. Data are more limited on the effects of other congeners. The same approach can be used with petroleum compounds, which also occur in complex mixtures.

BaP equivalency approach for PAH contamination

Bolger and Carrington (1999) provide a good summary of the rationale for using an equivalency approach to risk assessment for PAHs. Toxicological data available for BaP are much better than data available for any of the other PAHs. Though there is not adequate data to assess risks for individual PAHs, there is sufficient study data for several compounds to enable approximation of cancer potencies relative to BaP. The equivalency approach thereby relies most heavily on data considered to be the most sound and least likely to need revision. Though the cancer risk calculated by this method is an estimate, it is more reasonable than estimates obtained either by ignoring all but a few well-studied compounds or by assuming all congeners have equivalent potencies. On the other hand, compounds for which there isn't enough toxicity data to calculate a cancer potency relative to BaP are omitted from the total, even though some of these compounds may contribute to carcinogenicity. As can be seen from the lists in Table IV-1, few of the PAH compounds typically measured (see Table II-3) are included in the BaP equivalency total. Furthermore, the PAHs for which cancer potencies relative to BaP have been calculated occur predominately in pyrogenic rather than petrogenic sources.

The potencies relative to BaP of other PAHs are based primarily on animal bioassay studies. Estimates of the potencies can differ depending on the studies selected to derive them. For instance, ICF-Clements (1988) incorporated data into their potency model only if BaP was tested in the same bioassay system as the other PAHs, in the same laboratory, and at the same time. Different mathematical models also may yield different potencies. Examples of potencies for PAHs relative to BaP used or suggested by various agencies and researchers are listed in Table IV-1. Most of these estimates are similar, though some differ by as much as an order of magnitude.

Table IV-1. Relative PAH potency estimates derived from various sources.

	Relative PAH Potency				
Compound	**ICF/EPA[a]**	**USEPA[b]**	**FDA[c]**	**CA EPA[d]**	**Nisbet & Lagoy[e]**
Benzo[a]pyrene	1.0	1.0	1.00	1.00	1
Dibenzo[a,h]anthracene	1.11	1.0	1.05	0.36	5
Indeno[1,2,3-c,d]pyrene	0.232	0.1	0.25	0.10	0.1
Pyrene	0.081		0.13*		0.001
Benzo[b]fluoranthene	0.140	0.1	0.11	0.10	0.1
Benzo[k]fluoranthene	0.066	0.01	0.07	0.10	0.1
Benzo[g,h,i]perylene	0.022		0.03		0.01
Fluoranthene			0.02*		0.001
Benz[a]anthracene	0.145	0.1	0.014	0.10	0.1
Chrysene	0.0044	0.001	0.013	0.01	0.01
Anthanthrene	0.320**				
Benzo[j]fluoranthene	0.061				
Benzo[e]pyrene	0.004				
Cyclopentadieno[c,d]-pyrene	0.023				
Anthracene					0.01
Acenaphthene					0.001
Acenaphthylene					0.001
Fluorene					0.001
2-Methylnaphthalene					0.001
Naphthalene					0.001
Phenanthrene					0.001

a ICF-Clements Associates (1988).

** Identified in Nisbet and LaGoy (1992) as anthracene.

b U.S. Environmental Protection Agency (1993).

c U.S. Food and Drug Administration, Contaminants Standards Monitoring and Programs Branch, Center for Food Safety and Applied Nutrition (Bolger et al. 1996)

* Division of Mathematics, Center for Food Safety and Applied Nutrition.

d California Environmental Protection Agency (1997).

e Nisbet and LaGoy (1992).

Equivalency calculations

To estimate the total amount of PAHs in a sample, it is first necessary to calculate the weighted potency for each compound by multiplying the relative potency (see Table IV-1) of the compound by the concentration (wet weight) of that compound in the tissue sample. The products of these calculations can then be summed and added to the total amount of BaP in the sample (the product of the tissue concentration of BaP multiplied by a potency of 1.0) to estimate the total concentration of BaP equivalents.

The equation is shown below:

$$T_{PAH} = \sum_{j=1}^{n} R_j y_j + x$$

The variables are defined as follows:

T_{PAH} = total PAH exposure
n = the total number of indicator PAHs exclusive of BaP
y_j = exposure to the jth indicator PAH
R_j = relative potency of the jth indicator PAH compared to BaP
x = exposure to BaP

The assumption that exposure to several carcinogenic PAHs in a mixture will have the same carcinogenic effect as exposure to each compound separately at the same dose ("dose additivity assumption") is reasonable because most PAHs appear to metabolize to similar reactive derivatives that produce similar histological effects (ICF-Clements 1988).

Human Consumption Rate Assumptions

Ideally, risk assessments should be based on actual seafood consumption levels for the exposed population rather than default values, such as national averages for consumption rates. Unfortunately, data on seafood consumption levels may not be readily available for all consumer groups. Because seafood advisories or harvest restrictions often are based on cancer risk calculations, it is important to understand how consumption rate assumptions affect cancer risk calculations and, therefore, may affect seafood management decisions after a spill.

Groups of consumers that may be impacted by contaminated seafood include:

- Consumers of commercially harvested seafood;
- Consumers of recreationally harvested seafood; and
- Subsistence fishers and harvesters and their families and communities.

Consumption estimates for consumers of commercially harvested seafood

Consumers of commercially sold products often are not members of the local population in the spill region where the seafood is harvested, therefore national seafood consumption data may be appropriate for deriving consumption estimates to use in cancer risk calculations for these consumers. As summarized by USEPA (2000b), various surveys have reported mean seafood consumption rates for the general U.S. population ranging from 6.5 - 20.1 g/day, and 95th percentile consumption rates ranging from 41.7 – 102 g/day. Rates were based on consumption of commercial and recreational freshwater, saltwater, and estuarine seafood. Before using rates within these ranges for any actual risk assessment calculations, it is important to refer to the original data sources. Closures of commercial fisheries and aquaculture have occurred following several recent oil spills, including the *Exxon Valdez*, *Kure*, *North Cape*, *Julie N*, and *New Carissa*.

Consumption estimates for consumers of seafood harvested recreationally or for subsistence use

Consumers of seafood harvested recreationally or for subsistence use are generally of greater concern than the general population when estimating risk because they tend to have higher seafood consumption rates and rely more heavily on local seafood resources for sources of protein. Consequently, these seafood consumers may be at greater risk of health effects than the general population. National average consumption rates may underestimate their exposure. On the other hand, overestimates of their consumption rates may result in unnecessarily conservative advisories or harvest restrictions, limiting use of an important food source, with concomitant detrimental health, economic and cultural consequences.

For these reasons, we do not recommend using national survey data to develop local risk assessments if more accurate local seafood consumption information is available or can be collected and analyzed in a reasonable time frame. Data sources that can provide useful information on community consumption habits include:

Creel surveys: Creel surveys are conducted by state fish and wildlife management agencies, and consist of on-site interviews of fishers. Information is collected on species, sizes, and quantities of fish caught and taken home.

Fishing license surveys: Although demographic information on the licenses is limited, a record of names, addresses, license purchase locations, and duration of fishing seasons may be available, enabling consumption surveys to be conducted through the mail.

Subsistence surveys: Some state agencies conduct periodic subsistence surveys, such as the baseline research conducted by the Division of Subsistence of the Alaska Department of Fish and Game on subsistence fish and wildlife use by Alaska Native communities.

Anecdotal information: Useful anecdotal information on consumption habits of non-fishers, especially people from minority and low-income populations who may be sold or given fish privately, can be gathered by speaking with local community groups in an informal setting.

Behavioral risk surveillance surveys (BRSS): These are random telephone surveys funded by the Agency for Toxic Substances and Disease Registry (ATSDR). Some states have added questions on fisher demographics and consumption.

If it is not possible to use local, community-specific information on seafood consumption by recreational or subsistence fishers, it may be feasible to use survey data generated from a previously studied representative population that may have similar consumption patterns to the group of interest. Summaries of seafood consumption data obtained from sport and subsistence fisher surveys are shown in Tables IV-2 and IV-3, from USEPA (2000b).

Table IV-2. Sport fishers[a] consumption data (from USEPA 2000b).

Fisher Group	Seafood Consumption Rates (grams/day)					
	Mean	Median	80th Percentile	90th Percentile	95th Percentile	Fish Type
Alabama	45.8				50.7	F+S, F+C
Louisiana (coastal)		65				F+S, R+C
New York	28.1					F+S, F+C
New York (Hudson River)	40.9					F+S,R
Michigan	14.5		30	62	80	F+S,R
Michigan	18.3			50		F+S, F+C
Michigan	44.7					F, R
Wisconsin (10 counties)	12.3				37.3	F, R
Wisconsin (10 counties)	26.1				63.4	F, R+C
Ontario	22.5					F, R
Los Angeles Harbor		37		225		S, R
Washington State (Commencement Bay)		23		54		S, R
Washington State (Columbia River)	7.7					F+S, R+C
Maine (inland waters)	6.4	2.0		13	26	F, R

F = freshwater, S = saltwater, R = recreationally caught, C = commercially caught.

a Sport fishers may include individuals who eat sport-caught fish as a large portion of their diets.

Table IV-3. Subsistence fishers[a] consumption data (from USEPA 2000b).

Fisher Group	Seafood Consumption Rates (grams/day)			
	Mean	95th Percentile	Max	Fish Type
Great Lakes Tribes	351		1,426	F
Columbia River Tribes	58.7	170		F
High-end Caucasian consumers on Lake Michigan	48[b] 27[c]		144 132	F
Native Alaskan adults	109			F+S

F = fish, S = shellfish.

a Subsistence fishers include individuals who eat sport-caught fish at high rates but do not subsist on fish as a large part of their diets.

b Data from 1982 survey of fish eaters.

c Data from 1989 survey of fish eaters.

Consumption estimates for other potentially at-risk groups

Other factors that should be considered when estimating risk are age, reproductive status, general health, and additional occupational or life style exposure potential. For instance, though young children may eat smaller portions than adults, they may consume significantly more seafood per unit body weight. Therefore, a typical risk estimate for a 60-70 kg adult consuming an 8-ounce portion of seafood over a specified time period may underestimate a child's potential exposure level. When children are considered in risk assessment calculations, the USEPA uses an average body weight of 14.5 kg for children under 6 years old. Risks to developing children over a large range of body weights, however, may not be estimated accurately using this value (USEPA 2000b).

Fetuses may be susceptible to maternal PAH exposure because their enzymatic systems are too immature to eliminate toxic metabolites that readily pass through the embryonic and fetal blood-brain barrier. Therefore, it is important to inform women of reproductive age if action levels and consumption limits for PAHs are generated for a carcinogenic endpoint. The elderly, people with certain diseases, and people who may be exposed to PAHs through smoking or at high levels occupationally also may be more susceptible to the effects of PAH exposure from seafood consumption than the general population. Consequently, it may be advisable for people in these groups to limit their consumption of contaminated seafood to levels below those considered safe for the general population.

Considering that many local seafood consumers may fall into these potentially higher-risk groups, risk estimates based on average body weights, meal sizes, and consumption estimates for the general population may not accurately reflect actual risk levels of the exposed population. Therefore, it is important to communicate to the public the assumptions (i.e., body weights, meal sizes, meal frequencies) used to generate risk estimates and action or advisory levels.

For further information on calculating risk-based consumption limits for finfish and shellfish, see the third edition of the USEPA *Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories Volume 2: Risk Assessment and Fish Consumption Limits* (2000b).

V. RISK COMMUNICATION

General Considerations

Risk communication is defined as "an interactive process of exchange of information and opinions concerning risk and risk-related factors among risk assessors, risk managers, consumers, and other interested parties" (FAO/WHO 1998). The definition of risk is essential to the discussion. Risk has been defined as "a combination of the probability, or frequency, of occurrence to a defined hazard and the magnitude of the consequences of the occurrence" (Warner 1992 cited in Jones and Hood 1996). Both technical and social factors should be considered when communicating information on the health and safety of seafood following an oil spill, particularly when dealing with different groups. The risks and consequences have different meanings for the subsistence user, sport fisher, average consumer, commercial fisher, elected official, regulator, and responsible party representative. Regulators and scientists measure risk quantitatively and accept the uncertainty inherent in the risk-assessment process. The public perceives risk more qualitatively and subjectively, and is influenced by prior experience with similar risks and information made available to them. The public wants to know whether the seafood is safe to eat; yet the answers given are typically posed in terms of "acceptable risk" or "not a significant risk." Risk communicators should be aware of and try to overcome: 1) gaps in knowledge, 2) obstacles inherent in the uncertainties of scientific risk assessment, and 3) barriers to effective risk communication (Nighswander and Peacock 1999).

General recommendations for risk communication during oil spills include:

General recommendations for risk communication during oil spills include:

- Be proactive. Acknowledge and discuss the potential impacts to seafood safety from an oil spill as soon as possible. Establish a group responsible for assessing the risks to seafood early and review the risks as necessary as the spill evolves and new information is made available.
- Keep the public informed. Tell the public what you are doing to determine whether seafood safety is at risk. Release information quickly. Publish maps showing where and what type of seafood samples are being collected, and how they are being tested. Identify a Point of Contact for further information, and make sure that the public can reach the Point of Contact without delay. Make sure that the Point of Contact has the most current information and is prepared to answer questions, or knows how to get answers quickly. Response to all requests for information is important. Consider a web-based strategy for distributing seafood safety information, where individuals can check to see whether seafood in their area has been tested and to obtain test results.
- Meet directly with affected groups to discuss the issues and process. Direct meetings with groups such as commercial fishing associations, recreational users, subsistence users, seafood vendors, etc. providing opportunities to ask questions can be very effective. However, meetings can fail if the risk communicators are not prepared or knowledgeable, or appear to be withholding information. Specialized bulletins or communication methods may be necessary for special groups, such as Native American subsistence users and non-English-speaking users.
- Use unambiguous terms whenever possible. Health risks are commonly described in terms of probabilities of cancer based on assumed consumption rates and periods. It is assumed that carcinogens do not have safe thresholds for exposures; that is, any exposure to a carcinogen

may pose some cancer risk (USEPA 2000b). However, it is both useful and appropriate to define "safe" and "unsafe" levels of PAHs in seafood based on risk rates that are commonly considered to be acceptable. For example, water-quality criteria for carcinogenic contaminants in water usually use risk rates in the range of 10^{-5} to 10^{-6}. The general public understands the concepts of acceptable risks, although there may be components of society where these risks conflict with local cultures, such as the Alaska Native subsistence users during the *Exxon Valdez* oil spill (Field et al. 1999). As long as the risk communicators clearly define what is meant by "safe" and "unsafe," these terms are appropriate.

Lessons Learned from Previous Oil Spills

The *Exxon Valdez* and *New Carissa* oil spills provide examples of the range of issues faced in dealing with seafood safety at oil spills and the lessons learned in terms of risk communication. Each is summarized briefly below.

The *Exxon Valdez* oil spill impacted subsistence seafood users over a distance of nearly 800 kilometers, affecting 1,750 kilometers of shoreline and the harvest areas of 15 predominantly Alaska Native villages (Field et al. 1999). It was perceived that seafood safety for subsistence users was addressed relatively late in the spill response, and the active role of the responsible party in the seafood safety studies was a constant source of suspicion on the part of the village residents. Furthermore, there were conflicts in terms of the technical guidance for seafood safety ("use your own sensory tests and avoid collecting seafood in areas that showed evidence of oil") and the subsistence users' expectations that chemical testing would provide definitive answers to the questions about whether it was safe to eat the seafood. An Oil Spill Health Task Force, formed after the spill to deal with subsistence seafood issues, had to deal with the complex cultural issues of Native Alaskan subsistence users without any guidance or health criteria. In fact, much of the guidance in use today with regard to seafood risk from petroleum contamination is based on the approach developed by the task force for this spill. Fall et al. (1999) provided a ten-year perspective on the lessons learned for this spill with a significant impact to Native subsistence users:

- The active role of the responsible party was met with considerable skepticism and resulted in perceived conflict of interests that affected all phases of data collection, interpretation, and recommendations.
- There were significant cultural conflicts in defining seafood safety and edibility. A spill that impacted so many animals and habitats was perceived to also have significant impacts to human health, regardless of the information provided on actual health risks to consumers in the impact area.
- There was a perceived "double standard" for subsistence users, compared with commercial fisheries. Some commercial fisheries were closed within the first year after the spill, applying a "zero-tolerance policy" in order to protect the market for Alaskan salmon, which was not based on concerns about consumer safety. In contrast, subsistence users were told to avoid oiled areas and not eat food that smelled or tasted like oil.
- There was a need for direct communication with village residents, especially during the first year when concerns were greatest. Individual community members will not necessarily receive health-safety information distributed to community representatives. Formal mechanisms are needed for soliciting feedback and evaluating how well the risk communication efforts are being received.

In contrast to the *Exxon Valdez* oil spill, the *New Carissa* oil spill outside Coos Bay, Oregon occurred in a region of commercial and recreational fisheries where health advisories are routine. The Oregon Department of Agriculture (ODA) regulates commercial shellfish harvest under a strict water quality standard set by the U.S. Food and Drug Administration, which assumes there may be raw consumption of the product. Commercial fisheries are routinely closed depending on the amount of rainfall within specific watersheds, based on established correlations between rainfall and coliform counts. "Rainfall" closures are a common occurrence, and there are established communication mechanisms for notification of rainfall closures and openings. With regard to recreational fisheries, clamming and mussel harvesting are often closed due to domoic acid or amnesic shellfish poisoning. Figures V-1 and V-2 show official notifications for closure and opening of shellfish harvests during the *New Carissa* oil spill. Commercial and recreational users are accustomed to notifications of closures and openings based on accepted criteria for seafood safety. The closure of both commercial and recreational shellfish harvests during the *New Carissa* oil spill was met with limited resistance and confusion because of this established relationship between the regulator and user communities.

Figure V-1. Commercial shellfish harvest closure notice issued during the New Carissa oil spill.

To: Interested Parties

From: Oregon Department of Agriculture, Shellfish Program

Date: **February 17, 1999 (corrected update)**

Subject: Status of Commercial Shellfish "Rainfall' Closure

Commercial shellfish harvest is regulated by the Department of Agriculture (ODA) under a strict water quality standard set by the U.S. Food and Drug Administration (FDA), which assumes there may be raw consumption of the product. ODA does not close recreational shellfish areas without the cooperation of Oregon Department of Fish and Wildlife (ODF&W). When sewage or biotoxin contamination is evident, this agency will confer with ODF&W, DEQ and local county health departments to determine whether recreational shellfish harvesters are at risk and if they should be notified that shellfish harvest is closed. Call (503) 986-4720.

Nehalem Bay remains closed. Nehalem R did not fall below 7' since it peaked on 2/8. Nehalem closes if rainfall at Tillamook over 1" in 24 hrs (new plan using river stage in works).

Tillamook Bay, Main Bay closed today, February 17, 1999. The Wilson rose above 7' about 1 am today. The Main Bay is closed when Wilson R. exceeds 7.0'.

Cape Meares Area of Tillamook Bay remains closed. Cape Meares is closed for 7 days if 24 hrs rainfall exceeds 1"or when Wilson R. exceeds 7.0'.

Netarts Bay is open. This bay is closed for shellfish toxin events or flooding catastrophes.

Yaquina Bay, Main River, is **open.** This area closes for 5 days when Toledo rainfall exceeds 1.5"/24 hrs or if 3 days accumulative rain exceeds 3".

Winchester Bay and the Umpqua River to Big Bend, remains closed for rainfall; and harvest restrictions are ongoing due to potential for contamination from the New Carissa oil spill. This area closes for 7 days when the river exceeds 7.5' or > 1.5"/24 hrs.

Umpqua R. Triangle, So Jetty, closed today February 17, 1999 for rainfall/river ht; and harvest restrictions are ongoing due to potential for contamination from the New Carissa oil spill. The Umpqua went over 12' at around 4pm today. This area closes for 5 days if Umpqua R. @ Elkton exceeds 12' or rainfall > 2.0"/24 hrs.

Lower Coos Bay is closed; harvest restrictions are ongoing due to potential for contamination from the New Carissa oil spill. (down bay from No. Bend airport) is not closed for rainfall events.

Upper Coos Bay, opened February 12, 1999 from rainfall closure; but harvest restrictions are ongoing due to potential for contamination from the New Carissa oil spill. Upper Coos is closed 5 days if 24 hr rainfall exceeds 1.5" or 3 day accumulative rainfall exceeds 3"

South & Joe Ney Sloughs opened, February 12, 1999 from rainfall closure; but harvest restrictions are ongoing due to potential for contamination from the New Carissa oil spill. So Slough is closed 5 days if 24 hr rainfall exceeds 1.5" or 3 day accumulative rainfall exceeds 3" In addition to rainfall criteria, Upper So. Slough (area above Younker Pt) closes when tidal exchange exceeds 7.5'. During tidal closures growers may tend but not move shellstock.

Figure V-2. Shellfish harvest closure notice issued during the New Carissa oil spill.

Lower Coos Bay and the Charleston Boat Basin Area Added to Clamming and Mussel Closure Due to Oil Leaks From the *New Carissa*

Oyster Harvesting on Hold

February 12, 1999. The Oregon Department of Agriculture is adding Lower Coos Bay and the Charleston Boat Basin to the areas closed to shellfish harvesting as result of the oil spilling from the New Carissa. Surveys of the area made today indicate that oil sheen and oil globules are visible in these areas. The upper boundary for the Lower Coos Bay closure is the railroad bridge above North Bend; the upper boundary for the boat basin area closure is the Charleston Bridge.

Mussel and clam harvesting was prohibited on the beaches in Coos and Douglas counties yesterday due to possible contamination from the New Carissa oil spill. The extent of contamination on the beaches and bays is being surveyed today. These areas remain closed at this time. The public should take heed of any signs on Coos and Douglas County beaches and bays that alert them to shellfish closures.

The Department has required oyster growers to limit harvesting to areas that have been surveyed and confirmed to be unaffected by the spread of oil. This is an ongoing process due to the changing tides and the survey reporting process. At this time no oil has been seen in the oyster growing areas. Commercial oyster harvest will be prohibited from any areas contaminated by oil. There are inspectors on the scene to inspect shellfish and assure commercial shellfish safety.

The Department is in contact with natural resource advisors at the incident command and will keep the public and the commercial industry advised of shellfish safety information.

For more information call the Department of Agriculture's shellfish information line at (503) 986-4728 or Ron McKay at (503) 986-4720.

Communicating Relative Risks

Risk communicators commonly compare the relative risk of a specific activity to known risks of other activities. For example, the public is accustomed to hearing the risks of death by automobile accident or airplane crash. These are considered voluntary risks taken by people who decide to drive or fly after considering the risks and benefits associated with these activities, whether or not their perceptions are realistic. The public generally will accept risks from voluntary activities that are roughly 1,000 times greater than involuntary risks that provide the same level of benefits (Starr 1996).

Because the potential human-health risks from eating seafood contaminated by an oil spill are associated with PAHs, it is tempting to compare the PAH levels in seafood samples with those found in other food sources. PAHs are ubiquitous contaminants, measurable in many foods. Table V-1 summarizes the levels of PAHs in some commonly consumed foods. Based on information from previous spills, PAH levels in seafood from oil-spill-contaminated waters generally are considerably lower than PAH levels found in smoked foods. During the *Exxon Valdez* oil spill, however, village community residents became upset when it was pointed out that samples of smoked fish from the villages contained carcinogenic hydrocarbon levels hundreds of times higher than any shellfish samples collected from oiled beaches, and nearly 10,000 times higher than wild salmon (Nighswander and Peacock 1999). The residents considered eating smoked salmon to be an acceptable, voluntary risk, and eating oil-contaminated seafood to be an involuntary, unacceptable risk. Guidelines for risk communication include being sensitive to the distinction between voluntary and involuntary risk, and avoiding risk comparisons that equate the two (Chess et al. 1994). Risk comparisons should be made carefully.

Table V-1. PAHs in foods (Bolger and Carrington 1999).

Source	PAH (ppb or μg/kg)	B[a]P (ppb or μg/kg)
Corn oil	2-10	0.4-1.0
Smoked meat and fish	10-20	0.3-60
Bakers yeast	10-350	2-40
Kale	60-500	13-48

VI. Literature Cited

Ackman, R.G. and H. Heras. 1992. Tainting by short-term exposure of Atlantic salmon to water soluble petroleum hydrocarbons. In *Proceedings of the 15th Arctic and Marine Oil Spill Program Technical Seminar*. Environment Canada, Ottawa. 2:757-762.

Agency for Toxic Substances and Disease Registry (ATSDR). 1995. Toxicological Profile for Polycyclic Aromatic Hydrocarbons (PAHs). Atlanta: U.S. Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry.

American Society for Testing and Materials (ASTM). 1994. *Compilation of ASTM Standard Definitions, Eighth Edition*. Philadelphia: ASTM.

Bender, M.E., P.O. DeFur, and R.J. Huggett. 1986. Polynuclear aromatic hydrocarbon monitoring in estuaries utilizing: oysters, brackish water clams and sediments. In *Oceans '86 Conference Record, Monitoring Strategies Symposium, Vol. 3*. Piscataway, New Jersey and Washington, D.C.: Institute of Electrical and Electronics Engineers and Marine Technology Society. pp. 791-796.

Boehm, P.D. and J.G. Quinn. 1977. The persistence of chronically accumulated hydrocarbons in the hard shell clam *Mercenaria mercenaria. Marine Biology 44*: 227-233.

Bolger, M. and C. Carrington. 1999. Hazard and risk assessment of crude oil in subsistence seafood samples from Prince William Sound: lessons learned from the *Exxon Valdez*. In L. Jay Field et al. (eds.). *Evaluating and Communicating Subsistence Seafood Safety in a Cross-Cultural Context: Lessons Learned from the Exxon Valdez Oil Spill*. Pensacola: Society of Environmental Toxicology and Chemistry. pp. 195-204.

Bolger, M., S.H. Henry, and C.D. Carrington. 1996. Hazard and risk assessment of crude oil contaminants in subsistence seafood samples from Prince William Sound. *American Fisheries Symposium 18*:837-843.

California Environmental Protection Agency. 1997. Public health goals for benzo[a]pyrene in drinking water. Sacramento: Pesticide and Environmental Toxicology Section, Office of Environmental Health Hazard Assessment.

Challenger, G.E. and G.S. Mauseth. 1998. Closing and opening fisheries following oil spills; a case study in Humboldt Bay, California. In *Proceedings of the 21st Arctic and Marine Oilspill Program Technical Seminar,* Edmonton, Alberta, Canada, June 10-12, 1998, 1:167-179.

Chess, C., B.J. Hance, and P.M. Sandman. 1994. Improving dialogue with communities: a short guide for government risk communication. New Brunswick, New Jersey: New Jersey Department of Environmental Protection, Division of Science and Research and Environmental Communication Research Program; New Jersey Agricultural Experiment Station; and Cook College, Rutgers University.

Coates, P.J. 1998. The *Sea Empress* oil spill and its effect on the fishermen and fisheries of South West Wales. In R. Edwards and H. Sime (eds.). *The Sea Empress Oil Spill*. London: The Chartered Institution of Water and Environmental Management. pp. 137-151.

Conover, R.J. 1971. Some relations between zooplankton and bunker C oil in Chedabucto Bay following the wreck of the tanker *Arrow*. *Journal of the Fisheries Research Board of Canada 28*:1327-1330.

Davies, J.M., A.D. McIntosh, R. Stagg, G. Topping, and J. Rees. 1997. The fate of the *Braer* oil in the marine and terrestrial environments. In J.M. Davies and G. Topping (eds.). *The Impact of an Oil Spill in Turbulent Waters: The Braer*. Edinburgh: The Stationery Office LTD. pp. 26-41.

Davis, H.K., E.N. Geelhoed, A.W. MacCrae, and P. Howgate. 1992. Sensory analysis of trout tainted by diesel fuel in ambient water. *Water Science Technology 25*:11-18.

Davis, H.K., N. Shepherd, and C.F. Moffat. 1995. Uptake and depuration of oil taint from fish. In *Proceedings of the Second International Research and Development Conference*, International Maritime Organization, London, pp. 353-361.

DiSalvo, L.H., H.E. Guard, and L. Hunter. 1975. Tissue hydrocarbon burden of mussels as potential monitor of environmental hydrocarbon insult. *Environ Science & Technology 9*:247-251.

Ernst, R., J. Carter, and N. Ratnayake. 1989a. Tainting and toxicity in sea scallops (*Placopecten magellancius*) exposed to the water-soluble fraction of Scotian Shelf natural gas condensate. Dartmouth, Nova Scotia: Marine Environment Protection Branch, Environment Canada.

Ernst, R.J., W.M.N. Ratnayake, T.E. Farquharson, R.G. Ackman, W.G. Tidmarsh, and J.A. Carter. 1989b. Tainting of Atlantic cod (*Gadus morhua*) by petroleum hydrocarbons. In F.R. Engelhardt et al. (eds.) *Drilling Wastes*. New York: Elsevier Applied Science. pp. 827-839.

Fall, James A. 1999. Changes in subsistence uses of fish and wildlife resources following the Exxon Valdez oil spill. In: L. Jay Field et al. (eds). *Evaluating and Communicating Subsistence Seafood Safety in a Cross-Cultural Context: Lessons Learned from the Exxon Valdez Oil Spill*. Pensacola: Society of Environmental Toxicology and Chemistry. pp. 51-104.

Fall, J.A. and L.J. Field. 1996. Subsistence uses of fish and wildlife before and after the *Exxon Valdez* oil spill. *American Fisheries Society Symposium 18*:819-836.

Fall, J.A., L.J. Field, T.S. Nighswander, J.E. Stein, and M. Bolger. 1999. Overview of lessons learned from the *Exxon Valdez* oil spill: a 10-year retrospective. In L.J. Field et al. (eds.). *Evaluating and Communicating Subsistence Seafood Safety in a Cross-Cultural Context: Lessons Learned from the Exxon Valdez Oil Spill*. Pensacola: Society for Environmental Toxicology and Chemistry. pp. 237-269.

Field, L.J., J.A. Fall, T.S. Nighswander, N. Peacock, and U. Varanasi (eds.). 1999. *Evaluating and Communicating Subsistence Seafood Safety in a Cross-Cultural Context: Lessons Learned from the Exxon Valdez Oil Spill*. Pensacola: Society for Environmental Toxicology and Chemistry, 338 pp.

Fingas, M., B. Fieldhouse, and J. Mullin. 1994. Studies of water-in-oil emulsions and techniques to measure emulsion treating agents. In *Proceeding of the 17th Arctic and Marine Oilspill Program Technical Seminar*, Vancouver, British Columbia, June, 8-10, 1998, 1:213-244.

Food and Agriculture Organization/World Health Organization (FAO/WHO). 1998. Joint FAP/WHO expert consultation on the application of risk communication to food standards and safety matters, February 2-6, 1998, Italian Ministry of Health, Rome.

Food Technology Sensory Evaluation Division. 1981. Sensory evaluation guide for testing food and beverage products. *Food Technology 35*(11):50-59.

French, D.P. 1998. Modeling the impacts of the *North Cape* oil spill. In *Proceedings of the 21st Arctic Marine Oil Spill Program Technical Seminar*, Environment Canada, Ottawa, 1: 387-430.

French, D.P. 2000. Estimation of oil toxicity using an additive toxicity model. In *Proceedings of the 23rd Arctic Marine Oil Spill Program Technical Seminar*, Environment Canada, Ottawa, 1: 561-600.

Fucik, K.W. and J.M. Neff. 1977. Effects of temperature and salinity on naphthalenes uptake in the temperate clam *Rangia cuneata* and the boreal clam *Protothaca staminea*. In W.D. Wolfe (ed.) *Fate and Effects of Petroleum Hydrocarbons in Marine Organisms and Ecosystems*. New York: Pergamon Press. pp. 305-312.

GESAMP (IMO/FAO/UNESCO/WMO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Pollution). 1977. Impact of oil on the marine environment. Reports and Studies No. 6. Rome: Food and Agriculture Organization of the United Nations. 250 pp.

Gilroy, D.J. 2000. Derivation of shellfish harvest reopening criteria following the *New Carissa* oil spill in Coos Bay, Oregon. *Journal of Toxicology and Environmental Health, Part A, 60*:317-329.

Hayes, M.O. and J. Michel. 1999. Factors determining the long-term persistence of *Exxon Valdez* oil in gravel beaches. *Marine Pollution Bulletin 38*:92-101.

Hellou J. 1996. Polycyclic aromatic hydrocarbons in marine mammals, finfish, and mollusks. In G.H. Heinz and A.W. Redmon-Norwood (eds.). *Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations*. Boca Raton, Florida: Lewis Publishers. pp. 229-250.

Heras, H., R.G. Ackman, and E.J. MacPherson. 1992. Tainting of Atlantic salmon (*Salmo salar*) by petroleum hydrocarbons during a short-term exposure. *Marine Pollution Bulletin 24*:310-315.

Heras, H., S. Zhou, and R.G. Ackman. 1993. Uptake and depuration of petroleum hydrocarbons by Atlantic salmon: effect of different lipid levels. In *Proceedings of the 16th Arctic Marine Oil Spill Program Technical Seminar* 1:343-351.

Howgate, P., P.R. Mackie, K.J. Whittel, J. Farmer, A.D. McIntyre, and A. Eleftheriou. 1977. Petroleum tainting in fish. Rapports et proces-verbaux des reunions. In *Conseil permanent international pour l'exploration de la Mer 171*:143-146.

Hufnagle, L.C. Jr., S.E. Camarata, D. Ernest, C.A. Krone, S.-L. Chan, and M.M. Krahn. 1999. Development and application of a high-performance liquid chromatography screening method for aromatic compounds in invertebrate tissues. *Archives of Environmental Contamination and Toxicology 37*:220-226.

ICF-Clements Associates. 1988. Comparative potency approach for estimating the cancer risk associated with exposure to mixtures of polycyclic aromatic hydrocarbons. Interim final report EPA 68/02/4403. Fairfax, Virginia: U.S. Environmental Protection Agency.

Integrated Risk Information System (IRIS). 1994. EPA's carcinogenicity risk assessment verification endeavor work group. Cincinnati: U.S. Environmental Protection Agency, Environmental Criteria and Assessment Office.

International Standards Organization (ISO). 1992. Sensory Analysis – Methodology – Vocabulary. Report ISO 5942. Geneva: ISO.

Jacques Whitford Environment Limited. 1992. Characteristics of tainting in farmed Atlantic salmon and cultured blue mussels, exposed to water soluble fractions of Brent crude oil and Scotian Shelf condensate. Dartmouth, Nova Scotia: Marine Environment Protection Branch, Environment Canada. 30 pp.

Javitz, H. 1980. Seafood consumption data analysis. Final Report. EPA 68/01/3887. Washington, D.C.: Office of Water Regulations and Standards. U.S. Environmental Protection Agency.

Jokuty, P, S. Whiticar, Z. Wang, M. Fingas, B. Fieldhouse, P. Lambert, and J. Mullin. 1999. Properties of crude oils and oil products. EE-165. Ottawa, Ontario: Environment Canada.

Jones, D. and C. Hood. 1996. Introduction. In: C. Hood and D.K.C Jones (eds.). *Accident and Design: Contemporary Debates in Risk Management*. London: UCL Press.

Jordan, R.E. and J.R. Payne. 1980. *Fate and Weathering of Petroleum Spills in the Marine Environment*. Ann Arbor: Ann Arbor Science. 174 pp.

Jovanovich, M.C. and K.R. Marion. 1987. Seasonal variation in uptake and depuration of anthracene by the brackish water clam *Rangia cuneata*. *Marine Biology 95*:395-403.

Kennedy, C.J., K.A.Gill, and P.J. Walsh. 1989. Thermal modulation of benzo[a]pyrene uptake in the gull toadfish, *Opsanus beta*. *Environmental Toxicology and Chemistry 8*:863-869.

Kingston, P. 1999. Recovery of the marine environment following the *Braer* spill, Shetland. In *Proceedings 1999 Oil Spill Conference*, Seattle, Washington, March 8-11, 1999, pp. 103-109.

Krahn, M.M., D.G. Burrows, G.M. Ylitalo, D.W. Brown, C.A. Wigren, T.K. Collier, S-L. Chan, and U. Varanasi. 1992. Mass spectrometric analysis for aromatic compounds in bile of fish sampled after the *Exxon Valdez* oil spill. *Environmental Science & Technology 26*(1): 116-126.

Krahn, M.M., T.K. Collier, and D.C. Malins. 1982. Aromatic hydrocarbon metabolites in fish: automated extraction and high-performance liquid chromatographic separation into conjugate and non-conjugate fractions. *Journal of Chromatography 236*:441-452.

Krahn, M.M., L.K. Moore, and W.D. MacLeod, Jr. 1986. Standard Analytical Procedures of the NOAA National Analytical Facility, 1986: Metabolites of aromatic compounds in fish bile. NOAA Technical Memorandum NMFS F/NWC-102. Seattle: National Marine Fisheries Service, National Oceanic and Atmospheric Administration.

Krahn, M.M., M.S. Myers, D.G. Burrows, and D.C. Malins. 1984. Determination of metabolites of xenobiotics in the bile of fish from polluted waterways. *Xenobiotica 14*(8):633-646.

Krahn, M.M, G.M. Ylitalo, J. Buzitis, J.L. Bolton, C.A. Wigren, S-L. Chan, and U. Varanashi. 1993a. Analyses for petroleum-related contaminants in marine fish and sediments following the Gulf oil spill. *Marine Pollution Bulletin 27*:285-292.

Krahn, M.M., G.M. Ylitalo, J. Buzitis, S.-L.-Chan, and U. Varanasi. 1993b. Rapid high-performance liquid chromatographic methods that screen for aromatic compounds in environmental samples. *Journal of Chromatography 642*:15-32.

Krahn, M.M., G. M. Ylitalo, J. Buzitis, S.-L. Chan, U. Varanasi, T.L. Wade, T.J. Jackson, J.M. Brooks, D.A. Wolfe, and C-A. Manen. 1993c. Comparison of high-performance liquid chromatography/fluorescence screening and gas chromatography/mass spectrometry analysis for aromatic compounds in sediments sampled after the *Exxon Valdez* oil spill. *Environmental Science & Technology 27*(4):699-708.

Krahn, M.M., G.M. Ylitalo, J. Buzitis, C.A. Krone, J.E. Stein, S.-L.-Chan, and U. Varanasi. 1993d. Screening methods for assessing damage to natural resources following the *Exxon Valdez* oil spill. In *Proceedings of the 1993 Oil Spill Conference*, Tampa, Florida, March 29-April 1, 1993. pp. 872-873.

Krahn, M.M., G.M. Ylitalo, J. Joss, and S.-L.-Chan. 1991. Rapid, semi-quantitative screening of sediments for aromatic compounds using sonic extraction and HPLC/fluorescence analysis. *Marine Environmental Research 31*:175-196.

Landrum, P.F. 1982. Uptake, depuration, and biotransformation of anthracene by the scud *Pontoporeia hoyi*. *Chemosphere 11*:1049-1057.

Lauenstein, G.G. and A.Y. Cantillo (eds.). 1993. Sampling and analytical methods of the National Status and Trends Program, National Benthic Surveillance and Mussel Watch Projects 1984-1992, Vol. IV, Comprehensive descriptions of trace organic analytical methods. NOAA Technical Memorandum NOAA ORCA 71. Silver Spring, Maryland: Office of Ocean Resources Conservation and Assessment, NOAA. 181 pp.

Law, R.J. and J. Hellou. 1999. Contamination of fish and shellfish following oil spill incidents. *Environmental Geoscience 6*:90-98.

Law, R.J., C.A. Kelly, K.L. Graham, R.J. Woodhead, P.E.J. Dyrynda, and E.A. Dyrynda. 1997. Hydrocarbons and PAHs in fish and shellfish from southwest Wales following the *Sea Empress* oil spill in 1996. In *Proceedings 1997 International Oil Spill Conference*, Fort Lauderdale, Florida, April 7-10, 1997, pp. 205-211.

Lehr, W, D. Wesley, D. Simecek-Beatty, R. Jones, G. Kachook and J. Lankford. 2000. Algorithm and Interface Modifications of the NOAA Oil Spill Behavior Model. In *Proceedings of the 23rd Arctic and Marine Oil Spill Technical Seminar*, Vancouver, British Columbia, June, 14-16, 2000, 2:525-540.

Lockhart, W.L. and R.W. Danell. 1992. Field and experimental tainting of arctic freshwater fish by crude and refined petroleum products. In *Proceedings of the 15th Arctic and Marine Oil Spill Program Technical Seminar*, Edmonton, Alberta, pp. 763-771.

Mackenzie, K.M. and D.M. Angevine. 1981. Infertility in mice exposed in utero to benzo[a]pyrene. *Biology of Reproduction 24*:183-191.

Marsh, J.W., J.K. Chipman, and D.R. Livingstone. 1992. Activation of xenobiotics to reactive and mutagenic products by the marine invertebrates *Mytilus edulis*, *Carcinus maenus*, and *Asterias rubens*. *Aquatic Toxicology 22*:115-128.

Mauseth, G.S. and G.E. Challenger. 2001. Trends in Rescinding Seafood Harvest Closures Following Oil Spills. In *Proceeding of the 2001 International Oil Spill Conference*, Tampa, Florida, March 26-29, 2001, pp. 679-684.

Mauseth, G..S., C.A. Martin, and K. Whittle. 1997. Closing and reopening fisheries following oil spills; three different cases with similar problems. In *Proceedings of the 21st Arctic and Marine Oil Spill Program Technical Seminar*, Vancouver, British Columbia, Canada, June 11-13, 1997, 2:1283-1303.

McAuliffe, C.D. 1987. Organism exposure to volatile/soluble hydrocarbons from crude oil spills-a field and laboratory comparison. In *Proceedings 1987 Oil Spill Conference*, Baltimore, Maryland, April 6-9, 1987, pp. 275-288.

Meador, J.P., R. Stein, and U. Varanasi. 1995. Bioaccumulation of polycyclic aromatic hydrocarbons by marine organisms. *Reviews of Environmental Contamination and Toxicology 143*:79-165.

Mearns, A.J. 1995. Elements to be considered in assessing the effectiveness and effects of shoreline countermeasures. *Spill Science & Technology Bulletin 2*:5-10.

Mearns, A.J., G. Lauenstein, and T. O'Connor. 1998. The U.S. Mussel Watch: Nationwide geographic and longterm trends of PAHs in coastal mussels and oysters. In *Proceedings of the 21st Arctic and Marine Oil Spill Technical Seminar 1*:465-472.

Mearns, A.J., T.P. O'Connor, and G.G. Lauenstein. 1999. Relevance of the national mussel watch program to seafood fisheries management issues during oil spill response. In *Proceedings 1999 International Oil Spill Conference*, Seattle, Washington, March 8-11, 1999, pp. 701-708.

Mearns, A.J. and R. Yender. 1997. Workshop on managing seafood problems during the response phase of an oil spill. In *Proceedings of the 20th Arctic and Marine Oil Spill Technical Seminar 1*:203-217.

Michel, J. 2000. Interim Preassessment Report, *M/V New Carissa* Oil Spill, Coos Bay and Waldport, Oregon. Silver Spring, Maryland: Damage Assessment Center, National Oceanic and Atmospheric Administration. 84 pp. and appendices.

Moller, T.H., B. Dicks, and C.N. Goodman. 1989. Fisheries and mariculture affected by oil spills. In *Proceedings 1989 Oil Spill Conference*, San Antonio, Texas, February 13-16, 1989, pp. 389-394.

Moller, T. H., B. Dicks, K.J. Whittle, and M. Girin. 1999. Fishing and harvesting bans in oil spill response. In *Proceedings 1999 International Oil Spill Conference*, Seattle, Washington, March 8-11, 1999. pp. 693-699.

Motohiro, T. and Z. Iseya. 1976. Effects of water polluted by oil on aquatic organisms. II. n-paraffins, aromatic hydrocarbons and crude oil concentration on taint in scallop (*Pecten yessoensis*). *Bulletin of the Hokkaido University Faculty of Fisheries 26*:367-371.

National Oceanic and Atmospheric Administration (NOAA) and American Petroleum Institute (API). 1994. Inland oil spills: Options for minimizing environmental impacts of freshwater spill response. American Petroleum Institute Publ. No. 4558. Seattle and Washington, D.C.: NOAA and API. 130 pp.

National Oceanic and Atmospheric Administration (NOAA) and American Petroleum Institute (API). 2001. Environmental considerations for marine oil spill response. American Petroleum Institute Publication No. 4706. Seattle and Washington, D.C.: NOAA and API.

National Oceanic and Atmospheric Administration (NOAA), Rhode Island Department of Environmental Management, U.S. Department of the Interior. 1999. Restoration plan and environmental assessment for the January 19, 1996 *North Cape* oil spill. Silver Spring, Maryland: Damage Assessment Center, Office of Response and Restoration.

National Research Council (NRC). 1983. Polycyclic aromatic hydrocarbons in the aquatic environment: Formation, sources, fate and effects on aquatic biota. NRCC 18981. Washington, D.C: National Academy Press. pp. 106-107.

National Research Council (NRC). 1985. *Oil in the Sea: Inputs, Fates, and Effects*. Washington, D.C.: National Academy Press. 601 pp.

National Research Council (NRC). 1999. *Spills of Nonfloating Oils: Risk and Response*. Washington, D.C.: National Academy Press. 75 pp.

National Research Council (NRC). 2002. *Oil in the Sea III: Inputs*. Washington, D.C.: National Academy Press. 260 pp. + appendices.

Nighswander, T. and N. Peacock. 1999. The communication of health risk from subsistence seafood in a cross-cultural setting: Lessons learned from the *Exxon Valdez*. In L.J. Field et al. (eds). *Evaluating and Communicating Subsistence Seafood Safety in a Cross-Cultural Context: Lessons Learned from the Exxon Valdez Oil Spill*. Pensacola: Society for Environmental Toxicology and Chemistry. pp. 205-236.

Nisbet, I.C.T. and P.K. LaGoy. 1992. Toxic Equivalency Factors (TEFs) for polycyclic aromatic hydrocarbons (PAHs). *Regulatory Toxicology and Pharmacology 16*:290-300.

Payne, J.R. and W.B. Driskell. 1999. Analysis of water samples collected in support of the *M/V New Carissa* oil spill natural resource damage assessment. Silver Spring, Maryland: Damage Assessment Center, National Oceanic and Atmospheric Administration. 69 pp.

Reilly, T. I. and R.K. York. 2001. Guidance on sensory testing and monitoring of seafood for presence of petroleum taint following an oil spill. NOAA Technical Memorandum NOS OR&R 9. Seattle: Office of Response and Restoration, National Oceanic and Atmospheric Administration. 109 pp.

RPI International, Inc. 1987. Natural resource response guide: Marine fish. Seattle: Ocean Assessments Division, National Oceanic and Atmospheric Administration. 95 pp.

RPI International, Inc. 1989. Natural resource response guide: Marine shellfish. Seattle: Ocean Assessments Division, National Oceanic and Atmospheric Administration. 95 pp.

Salazar, M.H. and S. M. Salazar. 2001. Standard guide for conducting in-situ field bioassays with marine, estuarine and freshwater bivalves. In *2001 Annual Book of ASTM Standards*. Philadelphia: American Society for Testing and Materials.

Sauer, T.C., and P.D. Boehm. 1995. Hydrocarbon chemistry analytical methods for oil spill assessments. MSRC Technical Report Series 95-032. Washington, D.C.: Marine Spill Response Corporation 114 pp.

Scott, C.L., A.W. Paige, G. Jennings, and L. Brown. 1992. Community profile database catalog. Technical Paper 104. Juneau: Alaska Department of Fish and Game, Division of Subsistence.

Shigenaka, G. and C.B. Henry, Jr. 1995. Use of mussels and semipermeable membrane devices to assess bioavailability of residual polynuclear aromatic hydrocarbons three years after the *Exxon Valdez* oil spill. In G. Wells et al. (eds.). *Exxon Valdez Oil Spill: Fate and Effects in Alaskan Waters*. ASTM STP 1219. Philadelphia: American Society for Testing and Materials. pp. 239-260.

Shigenaka, G. (ed.). 1997. Integrating physical and biological studies of recovery from the *Exxon Valdez* oil spill. NOAA Technical Memorandum NOS ORCA 114. Seattle: Office of Ocean Resources Conservation and Assessment, National Oceanic and Atmospheric Administration. 206 pp.

Spacie, A. and J.L. Hamelink. 1982. Alternative models for describing the bioconcentration of organics in fish. *Environmental Toxicology and Chemistry 1*:309-320.

Starr, C. 1996. Social benefit versus technological risk. *Science 165*:1232-1238.

Topping, G., J.M. Davies, P.R. Mackie, and C.F. Moffat. 1997. The impact of the *Braer* spill on commercial fish and shellfish. In J.M. Davies and G. Topping (eds.). *The Impact of an Oil Spill in Turbulent Waters: The Braer*. Edinburgh: The Stationery Office LTD. pp. 121-143.

U.S. Environmental Protection Agency (USEPA). 1993. Provisional guidance for quantitative risk assessment of polycyclic aromatic hydrocarbons. EPA/600/R-93/089. Final Draft. Cincinnati: Environmental Criteria and Assessment Office, U.S. Environmental Protection Agency, 20 pp.

U.S. Environmental Protection Agency (USEPA). 1994. USEPA contract laboratory program national functional guidelines for organic data review. Publication 9240.1-05. Washington, D.C.: Office of Emergency and Remedial Response, U.S. Environmental Protection Agency.

U.S. Environmental Protection Agency (USEPA). 2000a. Guidance for assessing chemical contaminant data for use in fish advisories, Volume 1: Fish sampling and analysis, Third Edition. EPA 823/B/00/007. Washington, D.C.: Office of Science and Technology, U.S. Environmental Protection Agency.

U.S. Environmental Protection Agency (USEPA). 2000b. Guidance for assessing chemical contaminant data for use in fish advisories, Volume 2: Risk assessment and fish consumption limits, Third Edition. EPA 823/B/00/008. Washington, D.C.: Office of Science and Technology, U.S. Environmental Protection Agency.

Varanasi, U., J.E. Stein, M. Nishimoto. 1989. Biotransformation and disposition of polycyclic aromatic hydrocarbons (PAH) in fish. In: U. Varanasi (ed.). *Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment*. Boca Raton, Florida: CRC Press. pp. 94-149.

Whittle, K.J., D.A. Anderson, P.R. Mackie, C.F. Moffat, N.J. Shephard, and A.H. McVicar. 1997. The impact of the *Braer* oil on caged salmon. In J.M. Davies and G. Topping (eds.). *The Impact of an Oil Spill in Turbulent Waters: The Braer*. Edinburgh: The Stationery Office LTD. pp. 144-160.

VII. Glossary of Terms

API gravity: An arbitrary scale expressing the gravity or density of liquid petroleum products. The petroleum industry uses API gravity rather than density because the API scale provides greater distinction between different kinds of oils than does specific gravity. The measuring scale is calibrated in terms of degrees API. API gravity is determined by the equation API at 60°F = 141.5/oil density -131.5.

API gravity is based on the density of pure water with an arbitrary API gravity value of 10. The higher the API gravity, the lighter the product. Light crude oils generally exceed 38 degrees API and heavy crude oils are commonly labeled as all crude oils with an API gravity of 22 degrees or below. Intermediate crude oils fall in the range of 22 degrees to 38 degrees API gravity. Most oils have densities that are less than water and will generally float on the water surface. Oils with a specific gravity greater than 1.0 (API gravity of less than 10) will sink in fresh water (which has a specific gravity of 1.0 and an API gravity of 10). Non-floating oils in seawater have a specific gravity greater than 1.02 or an API gravity less than 7.

Adulteration: A food is deemed to be adulterated if it bears or contains any poisonous or deleterious substance that may render it injurious to health, or if it contains any filthy, putrid, or decomposed substances, or if it is otherwise unfit for food.

Advection: The transport of oil by water currents.

Aerial observation: Trained experts fly in helicopters or airplanes to make systematic observations on the position of oil slicks and stranded oil, oceanographic features that might influence oil behavior (such as eddies, rip currents, river outflow plumes, current speeds), distribution of wildlife (birds, turtles, marine mammals), or the effectiveness of response operations (dispersant applications, skimming).

Aliphatics: Hydrocarbon compounds composed of straight or branched chains of hydrogen and carbon. They have low water solubility and low aquatic toxicity. The low molecular weight compounds have high rates of microbial degradation.

Aromatics: Hydrocarbon compounds that contain one or more benzene rings. Mono-aromatics include benzene, toluene, ethylbenzene, and xylenes. Polycyclic aromatic hydrocarbons (PAH) (also sometimes referred to as polynuclear aromatic hydrocarbons) contain two or more benzene rings. Most of the toxicity of oil to water-column organisms results from the low molecular weight aromatic compounds.

Asphaltenes: Large, heavy compounds in oil that weather extremely slowly. Not present in light, refined products such as gasoline and diesel. Can be the dominant group of compounds in heavy refined oils.

Barrel: A volume measure of oil = 42 U.S. gallons.

Benthos/Benthic: Animals associated with the bottom of a body of water. If the animals are on the surface, they are called epifauna; if they live in the sediment, they are called infauna.

Bioaccumulation: The net accumulation of a substance by an organism as a result of uptake from all environmental sources and all possible routes of exposure, including contact, respiration, and ingestion.

Bioconcentration: The net accumulation of a substance as a result of uptake directly from aqueous solution.

Biodegradation: The breakdown of substances such as oil by microbes (bacteria, fungi, yeast) as they use it as a food source. Intermediate products are formed during the process, but the final products are carbon dioxide and water. This process is limited to a great extent by temperature, nutrient and oxygen availability, and the amount of oil present.

Biomagnification: The increase in body burden of a contaminant with trophic level.

Density of oil (specific and API gravity): Mass of a given volume of oil (in grams/cm^3) used to define "light" and "heavy" oils. Also measured in specific gravity (the oil's relative density compared with that of water at 15°C). The higher the specific gravity, the heavier the product. API gravity is based on the density of pure water with an arbitrary API gravity value of 10. The higher the API gravity, the lighter the product. Most oils have densities that are less than water and generally will float on water. Non-floating oils in seawater have a specific gravity greater than 1.02 or an API gravity less than 7.

Dispersants: Specially designed products composed of detergent-like solvents and agents applied directly from planes, helicopters, or vessels to help break oil slicks into small droplets that disperse into the water column and spread in three dimensions through natural water movement.

Dispersion: The process of breaking oil into very small particles or droplets (ranging in size from less than 0.5 microns to several mm) that mix into the water column. The smaller droplets will not refloat to the surface, but rather will move with the currents; larger droplets may refloat under calm conditions and reform slicks or sheens.

Dissolution: Loss of water-soluble components of oil into water. Compounds in oil are only very slightly soluble (maximum water-soluble fraction for crude oils in salt water is usually 10 to 30 ppm).

Distillation Fractions: The fraction (generally measured by volume) of oil that is boiled off at a given temperature. Used in models to predict the amount of oil loss via evaporation.

Elimination: All of the processes that can decrease tissue concentrations of a contaminant, including metabolism, excretion, and diffusive loss.

Emulsification (mousse formation): The process whereby small water droplets are incorporated into the oil, changing many of the oil's properties. Often has the consistency of chocolate mousse. Water content can be as high as 80%, increasing the volume of oily material for recovery and disposal. Greatly affects the efficiency of skimmers and pumps.

Evaporation: Transfer of the volatile fractions in oil from the liquid phase to the vapor phase. It is the single most important weathering process for the first several days of an oil spill.

Fingerprinting: Chemical analyses and interpretations used to compare an oil (usually the spilled oil) with other oils to determine whether they are from the same source. It is a critical process when the spill source is unknown. It is also important to determine the source of oil in environmental samples, such as seafood, compared to background contamination.

Growth Dilution: The process whereby the rate of accumulation is exceeded by the rate of tissue growth so that when the concentration is expressed on mass of chemical per mass of tissue over time, it appears as though elimination is occurring because the tissue concentration is decreasing.

Half-life: The time it takes for the concentration of a compound to decrease by half.

HAZMAT: NOAA Hazardous Material Response Division. Coordinates scientific support to the U.S. Coast Guard for oil and chemical spills. Has information for oil spill response at Web sites: http://response.restoration.noaa.gov and http://www.IncidentNews.gov

High-molecular weight PAHs: PAHs with 4-6 benzene rings.

Hydrophobic: "Water-fearing," a substance that is attracted to oil, lipids, and fats and repelled by water.

Lipophilic: "Lipid-loving," a substance that is attracted to oil, lipids, and fats.

Low-molecular weight PAHs: PAHs with 2-3 benzene rings.

Metabolism: Enzymatic process that converts insoluble petroleum hydrocarbons into more soluble breakdown products (metabolites) that can be more readily excreted by animals that have a kidney or kidney-like organ.

Microbes: At oil spills, the focus is on bacteria, fungi, and yeast that are able to degrade petroleum hydrocarbons.

Pelagic: Marine animals that live free from direct dependence on the sea bottom or shore. Free-swimming forms are nektonic; floating forms are planktonic.

Petrogenic: Hydrocarbons derived from petroleum oils, in contrast to pyrogenic hydrocarbons, derived from the combustion of fossil fuels.

Photo-oxidation: The process by which the components in oil are chemically transformed through a photochemical reaction, in the presence of oxygen.

Polar compounds: Very heavy, persistent compounds in oil, including asphaltenes (very large compounds) and resins (smaller compounds that bond with sulfur, nitrogen, or oxygen). Slowest to biodegrade.

Pour point: The temperature to which a substance must be heated to make it flow. Oils with a high pour point can congeal into semi-solid masses when spilled.

Pyrogenic: Hydrocarbons derived from the combustion of fossil fuels.

Salinity: A measure of how much salt is dissolved in water. Full strength seawater is about 35 parts per thousand (ppt). Freshwater is 0 ppt. The water in estuaries is a mixture of these two.

Saturates: Group of petroleum hydrocarbons consisting primarily of alkanes, but also cyclo-alkanes and waxes (large saturates).

Scientific Support Coordinator (SSC): Provides liaison with the scientific research and response community to the U.S. Coast Guard for oil and chemical spills.

Sedimentation: When particles suspended in the water column settle to the bottom. Can include settling of silt and clay in calm water and oil and sand mixtures in the surf zone and in rivers.

Sheen: A very thin layer of oil on water. Color indicates the thickness and volume per area:

Silver sheen	0.00007 mm	75 gallons/square nautical mile
First color trace	0.0001 mm	150 gallons/square nautical mile
Rainbow colors	0.0003 mm	300 gallons/square nautical mile
Dull colors	0.001 mm	1,000 gallons/square nautical mile
Dark colors	0.003 mm	3,000 gallons/square nautical mile

Solubility: How much of an oil will enter the water column on a molecular basis. Solubility of oil in water is generally <100 parts per million (ppm); thus it not a significant loss mechanism for oil.

Taint: An off-flavor or off-odor in seafood that is not typical of the flavor or odor of the seafood itself.

Tonnes (metric): a weight measure for oil, approximately = 300 gallons.

Toxicity (acute and chronic): An adverse affect on a living organism caused by exposure to a contaminant, such as oil. Acute toxicity occurs over a very short exposure period (hours to days) and usually results in death. Chronic toxicity occurs from long-term exposure (weeks or more) and causes impacts to reproduction, growth, and behavior.

Trajectory: A prediction of where the oil will be transported by wind and currents over time.

Uptake: Acquisition of a substance from the environment by an organism as a result of any active or passive process. Uptake is controlled externally by the partitioning behavior of the contaminant (between sediment, water, and food) and internally by the organism's behavior and physiology.

Viscosity: Resistance to flow in a liquid. Determines whether dispersants will be effective on an oil slick. Viscosity increases as it gets colder and as the oil weathers. Low viscosity is like water, medium viscosity is like molasses, and high viscosity is like tar.

Weathering: Changes in the physical and chemical properties of oil due to natural processes that begin when the discharge occurs and continue until the oil is removed. Major weathering processes include evaporation, emulsification, dissolution, photo-oxidation, and biodegradation.

Appendix

NOAA/NMFS NATIONAL SENSORY SECTION
CHAIN OF CUSTODY FORM

7600 Sand Point Way NE, Seattle, WA
For more information contact Michael DiLiberti
978-281-9123 or FAX 978-281-9125

Project_____ **Sampler**_____

Sample I.D. #	Date Collected	Location	Sample Type (Tissue, oil, water. Include species name and tissue type)	Comments

Collected by	Received by:(signature)	Condition:	Date/Time
Relinquished by: (signature)	Received by:(signature)	Condition:	Date/Time
Relinquished by: (signature)	Received by:(signature)	Condition:	Date/Time
Relinquished by: (signature)	Received by:(signature)	Condition:	Date/Time
Relinquished by: (signature)	Received by:(signature)	Condition:	Date/Time
Relinquished by: (signature)	Received by:(signature)	Condition:	Date/Time

* If shipped, include carrier name and copy of shipping invoice

Donald L. Evans
Secretary, U.S. Department of Commerce

Vice Admiral Conrad C. Lautenbacher, Jr., USN (Ret.)
Under Secretary for Oceans and Atmosphere and NOAA Administrator

Jamison S. Hawkins
Acting Assistant Administrator for
Ocean Services and Coastal Zone Management,
NOAA Ocean Service

November 2002