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Oil Impacts on Cold-water Marine Resources: A Review Relevant to Parks Canada's Evolving Marine Mandate

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Oil Impacts on Cold-water Marine Resources: A Review Relevant to Parks Canada's Evolving Marine Mandate

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EXECUTIVE SUMMARY

This review provides a brief current account of the effects of oil pollution relevant to the new stewardship mandate of Parks Canada for representative Canadian cold-water ocean and estuarine ecosystems. It is timely because of recent contributions from the 1989 *Exxon Valdez* oil spill (EVOS), important long-term oil fate and effects studies and the commitment in Parks Canada to Geographic Information System (GIS)-based shoreline inventory and oil spill response planning where GIS information plays a key role.

The acute, short-term effects of oil spills are reasonably well described. There remains, however, uncertainty and controversy over the chronic, long-term and sublethal effects of oil at population and ecosystem levels.

The EVOS experience has shown, again, that spills can have sociopolitical dimensions that, in a crisis situation, will overwhelm science considerations. Each spill event is a unique blend of place, nature and human influences among which science is a facet.

Effects on pelagic (open water) and deeper subtidal benthic (sea bottom) systems are relatively small. Effects at interfaces can be great, such as on groups contacting the sea surface (e.g., seabirds and marine mammals) and on intertidal ecosystems at the land-sea interface. Pelagic species, which complete part of their life cycle at interfaces by having floating eggs at the sea surface or by spawning intertidally, are also vulnerable to oil. Substrate and exposure to wave energy are the critical variables to impacts of intertidal oil. Sheltered and sedimentary habitats retain oil, while exposed rocky shores are more quickly cleaned by nature.

The following key weaknesses in our understanding of the biology of oil impacts remain:

- poor pre-oiling environmental baseline data for comparison with post-oiling status;
- defining and quantifying exposure to oil hydrocarbons remains speculative;
- few integrated population or ecosystem studies compared to single species studies;
- inability to differentiate natural ecosystem changes from oil pollution effects; and
- poor understanding of long-term, chronic sublethal impacts of oil pollution.

Despite these problems, recent experiences such as lessons learned from the EVOS, will lead to a more prominent role for science. There now is a body of knowledge on oil fate and effects that can assist non-specialists to understand and anticipate likely impacts, cooperate in damage assessment and remediation, and predict recovery. The utility of this information is enhanced by Parks Canada's use of GIS for coastal inventory that provides a framework for meaningful cooperation in a spill situation.

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INTRODUCTION

"Parks Canada must have a scientific capability, in the sense of access, contribution to, and use of a systematic, current body of information in the natural, physical, and social sciences. This capability is essential to the understanding and the sound decisions directed at the protection of parks, sites, and canals." Lopoukhine et al. (1998)

This review provides Parks Canada Service Centre and Field Unit (Park) personnel with a current overview of the effects of spilled oil (crude and refined) and offshore petroleum development applicable to Canadian cold-water (temperate to Arctic) ocean species, ecosystems and resources. It is also a guide to the source literature if more detailed information is needed. Science from the *Exxon Valdez* oil spill (EVOS) of March 1989, by far the largest and most integrated contribution to the cold-water literature, and recent publication of long-term oil fate and effects studies enhance the review's timeliness.

Marine oil pollution is of direct concern to Parks Canada. In the most recent "State of the Parks" Report to Parliament, "petrochemical pollution" was recognized as a stressor in 15 Parks (Parks Canada 1998 a). Of these, five were National Parks with ocean shorelines. Further, the high environmental quality associated with National Parks is expected, given the agency's aspiration to meet or exceed the letter and spirit of Canadian environmental legislation, such as the *Canadian Environment Protection Act* , *Canadian Environmental Assessment Act* and *Arctic Waters Pollution Prevention Act,* in its management planning,

Parks Canada's specific national marine park policy is influenced by the recognition that marine ecosystems are fundamentally different than terrestrial ecosystems (Mondor 1992; Parks Canada 1994). One great difference influencing Parks Canada's policy is the increased susceptibility of marine ecosystems to "downstream" pollution effects. A second marine policy point is that of managing in a broad, regional context (Mondor 1992). Given the prime Parks Canada mandate of maintaining marine ecosystem integrity as specified in the *National Parks Act* and the forthcoming *Marine Conservation Areas Act*, oil pollution within or nearby marine conservation areas is a direct concern. An example would be the international rights for innocent passage of vessels in Canadian waters. A third marine policy issue is that science has been given relatively more importance in management because of the poorer understanding of Canada's marine ecosystems compared to terrestrial ecosystems (Mondor 1992). Parks Canada has a low profile in marine science and this should change if the agency wants to partner with other agencies that have traditional marine environmental mandates such as Department of Fisheries and Oceans and Environment Canada. Finally, oil spills have already been identified as a major stressor in the Ecological Integrity Statement of Pacific Rim National Park Reserve and in the Management Plan of Saguenay-St. Lawrence Marine Park (Parks Canada 1995). As Parks Canada's representative marine conservation area system grows, the agency's national marine profile and mission to protect its expanding marine assets from oil pollution will grow.

The review begins with generalizations concerning spilled oil impacts and then focuses on marine ecosystems and organism groups relevant to the cold ocean waters of Canada's distinct National Marine Conservation Areas Natural Regions (Mercier and Mondor 1995). Examples of the range of potential threats include tanker traffic passing through Sagueney-St.Lawrence Marine Park (Parks Canada 1995), the prospect of oil and gas development offshore of Gwaii Haanas (Dietrich 1995, 1998) and, most likely, smaller vessel fuel spills along ocean and estuary shorelines of all parks.

The review's focus is on ocean oil pollution "science", but scientific uncertainties abound, especially concerning long-term, chronic sublethal impacts. Further, there is the reality that spills have overarching sociopolitical attributes that can obscure science.

Table 1 is an overview of the sources of oil hydrocarbons in the sea. The largest source is "marine transportation", which includes shipping accidents, washing of vessel tanks and loading/unloading of tankers at oil terminals. The second largest source is from land-based activities. The term "natural sources" includes seeps from the sea bottom and leakage of naturally formed hydrocarbons from the metabolism of marine organisms.

There are many physical, chemical and biological conditions that affect the degree of oil impacts on marine organisms and their habitats. Oil floats and acute impacts tend to occur to the species and ecosystems at interfaces such as the sea surface and the intertidal. If oil remains on open water, it is usually of less environmental concern than when it comes ashore (Baker 1991; Thorhaug 1992).

Table 1. Sources of oil hydrocarbons in the sea (Anonymous 1984).

Source Type	%
Marine transportation	45.5
Runoff from land-based activities	29.0
Natural sources	9.0
Atmospheric deposition	9.0
Refinery discharges	6.0
Offshore oil production	1.5
TOTAL	100

What is meant by the term "oil"? Crude oil is a complex mixture of usually > 90% hydrocarbons ranging widely in structure, molecular weight and toxicity. Table 2 provides a summary of the major products from refined crude oil, starting with the lighter fractions, with fewer carbon atoms, at the top. The chemical composition of different refined products can vary considerably within the same product group, depending on the crude oil type, refining process and market specifications for a product such as unleaded gasoline. The Canadian Coast Guard (CCG 1998) uses the definition of oil from *Canada Shipping Act* as "*…. oil of any kind and in any form and, without limiting the generality of the forgoing, includes petroleum, fuel oil, sludge, oil refuse and oil mixed with wastes but does not include dredged spoil.*"

Table 2. The major products of oil refining (Anonymous 1984).

Fraction	No. of Carbon Atoms	Product Name
Natural gas	1-6	Natural gas
Petrol & Naptha	4-12	Gasoline
Medium distillates	10-20	Kerosene, Light gas oil, Aviation fuel, Diesel fuel
Heavy distillates	18-45	Heavy gas oil, Feed for "cracking"* process, Wax, Lubricating oil
Residue	> 40	Heavy fuel oil, Asphalt, Coke

* degeneration of high molecular weight components at high temperatures or by means of catalysts

Once oil enters the sea, it is immediately subjected to physical, chemical and biological processes that alter its characteristics (USOTA 1991; Owens et al. 1994; O'Clair et al. 1996). Abiotic or "weathering" processes include the following:

- evaporation, dissolution and dispersion;
- photochemical oxidation;
- water-in-oil emulsification ("mousse" formation); and
- adsorption on to particulates (e.g., clay-oil flocculation, sinking and sedimentation).

Biotic degradation processes include microbial degradation and ingestion by organisms. Oil's organic matrix is a potential food source for some species throughout the weathering and degradation processes. Organisms can be stressed by exposure to oil through the following:

- toxic chemical interference with life processes;
- physical inhibition by smothering; and
- disruption of sensory perception and behavioral alteration.

As the aqueous solubility of oil is low, the estimation of actual concentrations of soluble oil is difficult and interpretation of toxicity to organisms must be cautious (Hawker and Connell 1992). Moreover, whether oil hydrocarbons are noncumulative remains unresolved. Different ecosystems, different seasons, different species and different life stages of the same species can vary in their responses to oil (Howarth 1991; Thorhaug 1992).

SUMMARY OF CONDITIONS AFFECTING OIL IMPACTS

Many conditions affect oil impacts, often in complex, cumulative or synergistic ways that are still not well understood. The net result is that each spill event, and its environmental impacts, is unique. The following is a summary of major conditions affecting oil impacts:

- type of habitats contacted by oil;

- type and quantity of oil - refined oils are more volatile and toxic than heavier crude;
- species of organisms contacted directly or indirectly by oil;
- the time the oil has weathered at sea before beaching which allows for the more toxic low molecular weight fractions to evaporate - within days oil volume can decrease up to 50% while density and viscosity increases (USOTA 1991);
- spill timing such as season and stage in species' life cycles influences their vulnerability (e.g., eggs and larvae are usually more vulnerable than adults);
- hydrographic (tides/currents) - meteorological (storm) conditions which disperse oil;
- climate (e.g., oil weathers slower in colder areas and the presence of sea ice);
- frequency and duration of exposure to the oil; and
- effectiveness of spill countermeasures (e.g., amount of oil recovered [usually <30%], dispersants used , bioremediation, extent of mechanical intervention, etc.).

ROLE OF BIOLOGY IN SPILL PREPAREDNESS AND RESPONSE

"*Expert advice should be sought from marine biologists and similar experts before a decision is made on the means to be used and the extent of the planned clean-up operation.*" (CONCAWE 1992)

Oil pollution preparedness and response is a large technical and logistical field beyond the scope of this biologically-based review. Accordingly, the focus here is on the role of biological knowledge in supporting response implementation such as the biological bases of identification and ranking areas for protection and treatment. Given the prime spill response directive is minimizing ecosystem impacts, effective planning has three general requirements as follows (Adams et al. 1983):

- identification and ranking of coastal areas sensitive to oil;
- development of strategies for protection of these areas; and
- formulation of cleanup guidelines to minimize ecosystem damage.

In the event of a spill, a Field Unit will have to function cooperatively with other agencies in implementing specified response options within a previously agreed command and control structure (IPIECA 1991). Best local knowledge about the area's shores will likely come from Field Unit staff - the *de facto* local experts with attendant expectations from other agency personnel.

For example, if a spill threatens Gwaii Haanas National Park Reserve, a Park representative would automatically have a seat at the Regional Environmental Emergency Team (REET) table. The *Canada Shipping Act* mandates REET the consolidated science and technical advisory role, chaired by Environment Canada (EC), to advise the lead agency, the Canadian Coast Guard (CCG). In B.C., the province (BC Ministry of Environment, Lands and Parks) is mandated by the *Emergency Program Act* to be fully involved with REET and to partner with CCG in overall spill management. Under ideal conditions, the polluter would be capable of its legal responsibility as "On-scene Commander" for adequate response with the CCG functioning as the federal "On-scene Monitor". A private sector consortium would be standing by to service the polluter with professional assistance. If the polluter cannot take responsibility, the CCG (and the province) would take over, engage the contractors and later seek legal redress from the polluter. Gwaii Haanas' role would be to advise REET in a science and environment capacity regardless of who (polluter or agency) implements the response. The on-water oil is seen by the federal government as a federal responsibility and a provincial responsibility when it is on shore or on the sea bottom, with exceptions such as National Parks and National Marine Conservation Areas. The province makes no such federal-provincial distinction; a complexity the REET process is intended to manage.

As a parallel example in response planning, the Great Barrier Reef Marine Park Authority perceives oil pollution as a major regional threat (Craik 1991). The Authority provides a "Scientific Support Co-ordinator" to contribute accurate local environmental information and advice to the overall on-site coordinator appointed from another agency (Craik 1989, 1991). The Scientific Support Co-ordinator advises on wildlife rescue, monitoring and restoration.

The key to effective response is sensitivity mapping based on dividing shorelines into a series of individual segments of homogeneous geological makeup. The process began in the Atlantic and the Arctic in the late 1970s and early 1980s such as the Beaufort Sea protection and cleanup strategy of Worbets (1979). However, it was not until the 1990s that a national shore sensitivity mapping protocol was developed by EC. Mapping for oil sensitivity has been the foundation of Canadian coastal resource mapping schemes as they share the same geological segmentation concept.

In B.C., shoreline sensitivity mapping has been lead by the province since the 1970s with minor EC involvement. The first modern era spill response atlas was for the south west Vancouver Island coast (Dickins et al. (1990). Mapping was fully developed in the Howes et al. (1993) atlas of contiguous 1:40,000 "Operational" maps for the southern Strait of Georgia. Electronic atlases have now been developed by the B.C. Land Use Co-ordination Office. Environmental and human-use attributes are attached to the coastal segments as distinct and directly comparable layers in a GIS system described by Harper et al. (1991) for the southern Strait of Georgia. Generic types of information and sensitivity rankings used for the maps are listed in Table 3.

The three sets of Operational maps use coloured shoreline segmentation. The Sensitivity to Oiling maps provide an aggregate sensitivity ranking of segments as well as their monthly sensitivity according to assumed seasonal changes. Countermeasures and Logistics maps provide segmentation according to geological shore types and physical attributes affecting response. Resource Operational maps illustrate segments according to habitat type, provide offshore polygons of similar resource values, overlay symbols illustrating noteworthy attributes such as seabirds and provide changing vulnerability of segments' resource values on a monthly basis. These temporal vulnerability changes could be based on, for example, seasonal seal haulout sites, herring spawn locations and varying recreational beach uses.

Table 3. Generic information used in the mapping system of B.C. oil spill response atlases.

Information Type	Shoreline sensitivity	Biological resources / nearshore or intertidal harvesting / tourism & recreation / industry & commercial fishing / communities & heritage sites / duration of biological activity & resource use / environmental descriptions
	Physical environment & countermeasures	Shore morphology / shore access / anchorages / helipads & landing sites / possible countermeasures / access limitations
Sensitivity Ranking	Human uses	Communities / Aboriginal harvesting / commercial fishing / tourist facilities / recreation uses / industry uses

	Biological resources	Species groups (e.g., sea lions) / species' abundance-distribution-seasonal occurrence / vulnerability to oil / long-term sensitivity to oil
	Shore zone oil residence	Wave exposure index / coastal processes / shore morphology
	Special status areas	Protected areas-parks / Indian Reserves / heritage sites

Legacies of this B.C. mapping system include applications such as the coastal biophysical resource inventory of Gwaii Haanas National Park Reserve (Harper at al. 1994). The entire B.C. coast will be mapped in this manner by 2001. There are attendant physical shore zone mapping protocols (Howes et al. 1994) that include, but are not limited to, oil pollution response and biological mapping protocols (Searing et al. 1995). These have since been refined further into the British Columbia Marine Ecosystem Classification (Zacharias et al. 1998) for marine conservation, resource management and planning. There also is a shoreline cleanup manual for assessing shore oil conditions and cleanup options (Environment Canada 1992) and complimentary provincial manual (Owens et al. 1992).

On the Atlantic coast, EC has led the approach to standardized mapping and preparedness. Pre-spill data collection and mapping for shoreline protection and cleanup rely on a database compatible with EC's Shoreline Sensitivity Mapping Program (Owens and Dewis 1995). The thrust is to use standard terms and definitions and of standard objective and strategy statements. There are complimentary field guides for documentation and description (Owens and Sergy 1994) and protection and cleanup (Owens 1995) of oiled shorelines. Individual areas such as the west coast of Newfoundland have their own oil spill sensitivity atlases (Dempsey et al. 1995) as planning tools for preparation, prioritization and implementation of countermeasures.

For Saguenay-St.Lawrence Marine Park, there is a spill response plan and a shoreline sensitivity atlas on GIS; both based on the EC system (Parks Canada 1995; Nadia Menard, personal communication). Fathom Five National Marine Park does not have a spill contingency plan, but they do have a shoreline sensitivity atlas on GIS, based on the EC system and provided by the CCG/EC for the Great Lakes region (Parks Canada 1998 b; Scott Parker, personal communication).

In summary, local environmental knowledge combined with improved understanding of the fate and effects of oil in Canadian ocean and estuarine waters provided by this review serves Parks Canada's ecosystem integrity mandate. Specifically, this review contributes to Parks Canada's spill response effectiveness for the following reasons:

- fills an information gap not fully covered in spill response planning and training;
- provides staff with critical background information that will foster more effective interaction with other agency (REET) personnel and the media; and
- helps position staff in a leadership science support role (e.g., the Field Unit's local knowledge and shore zone mapping on GIS may be the best baseline information available) during a spill threatening resources under Parks Canada's jurisdiction.

CAVEAT ON THE SCIENTIFIC LITERATURE

"*Catastrophic events are likely to be characterized by missed data collection opportunities. Decisions often must be made hastily, and despite the best intentions of scientists, studies may be constrained by pressures from attorneys, politicians, government officials, and various impacted or concerned parties, each with agendas that tend to compromise good science.*" (Garshelis 1997)

"*Ideally, data would show that oil reached individuals, that oil killed individuals, and that abundances in oiled sites were lower than in either non-oiled sites or in the same sites before the spill.*" (Hilborn 1996)

It is important to realize the limitations of science in environmental disasters (Cormick and Knaster 1986; Strickland 1990). Although additional data help with specific issues, they will likely never fully answer some questions (Strickland 1990) and managers can expect to be working with incomplete technical information. A complex of socioeconomic, environmental, legal and political variables will influence decision-making during a spill crisis or confronting the effects of cumulative, chronic pollution after a spill. Major spills become political in which "*scientific logic plays but a small role*" (Strickland 1990). Further, imperfect organizational response during spill crises remains a serious concern because of an emphasis on control instead of decision-making (Herrald et al. 1992).

Scientific opinion varies considerably concerning many of the impacts of oil in the sea. Further, the inability to separate natural from pollution-induced variability strongly affects conclusions from marine pollution studies generally (Underwood and Peterson 1988) and oil pollution studies in particular (Wells et al. 1995). Howarth (1989) concluded that "*current knowledge does not allow precise estimation of ecological damage resulting from oil pollution*". Controversy remains in oil spill fate and effects science as exemplified by the conflicting conclusions from similar studies in post-EVOS research published in volumes directly funded by EXXON Co. (Wells et al. 1995) and by agency-related entities such as the EVOS Trustee Council (EVOSTC 1993; Rice et al. 1996). These studies concentrated in the enclosed Prince William Sound location of the spill and adjacent areas in the more open Gulf of Alaska south and west of the Sound and some will be discussed in detail later. The net effect of the adversarial, litigious environment was to create a pair of science camps working on the same topics but forbidden to communicate with each other. In retrospect, the schism, the emotion and the politics of the moment likely induced flawed objectivity and failed to serve the course of best possible science.

Much of the early literature on effects of oil comes from laboratory studies on individual species (Moore and Dwyer 1974) or ecosystem components conducted under controlled conditions. Other studies include the following:

- experimental field studies using large enclosures called "micro or mesocosms" (Spies 1987; Howarth 1989);
- monitoring effects of controlled oil spills in the field (Ballou et al. 1987; Snow et al. 1987; Feder et al. 1990); and
- post-oil pollution (mostly short-term acute but a few long-term chronic) field assessments (Baker et al. 1990).

The following problems are common in the marine oil pollution literature:

- base-line data are usually lacking (e.g., natural variation in marine communities) yet critical when comparing pre- with post-spill conditions – considered by Hilborn (1996) as the biggest lesson learned from the EVOS;
- some post-spill work has flawed field sampling and experimental design because it was hastily planned and executed under crisis conditions (Howarth 1989; Foster et al. 1990);

- the applicability of the many laboratory studies to nature is questionable (Neff and Anderson 1981; Fuick et al. 1984; NAS 1985; Jackson et al. 1989; Guzman et al. 1991) because:
  - field conditions of each spill are unique
  - unnatural spatial scale in the laboratory
  - inability to account for complex natural variations in laboratory stations
  - short duration of laboratory manipulations; and
- long-term (chronic sublethal) effects of oil at the marine ecosystem level are less well understood than short-term acute effects (Nelson-Smith 1982; Howarth 1991; Suchanek 1993; Owens et al. 1994; Vandermeulen and Singh 1994; Spies et al. 1996).

Hilborn (1996) discussed the widely held perception that the EVOS studies were not as conclusive in detecting population-level impacts as many had hoped and, therefore, casted doubt on the ability of current methods to detect such impacts. Most studies were on single species, there were some intertidal and shallow subtidal community studies and no ecosystem-level studies (i.e., studies on changes in ecosystem structure). Hilborn (1996) listed five methods for detecting spill impacts at the population level:

- dead organism counts;
- pre- and post-spill comparisons of abundance;
- oiled versus unoiled comparisons of abundance;
- oiled versus unoiled comparisons of vital rates (e.g., a life history parameter such as growth); and
- direct experimental oiling.

Hilborn (1996) pointed out that no single source of evidence would be absolutely convincing and that a linked "*chain of evidence*" consisting of different methods is needed.

COLD CLIMATES AND SEA ICE

"....the rate of (oil) input required to exceed the rate of natural removal will be considerably slower than in temperate and tropical seas." (Percy and Mullin 1975).

In cold waters, oil tends to degrade more slowly and oiled biota recover more slowly (AMAP 1997). Serious concerns arise when spills occur during continuous darkness, sea ice conditions or seasonal migratory activities of bird and mammals. The influence of sea ice on the fate and effects of spilled oil warrants special attention. For example, from approximately November to May northern Labrador inshore waters are covered by landfast ice. On the one hand, onshore areas are protected by ice from contact with spilled oil (McLaren 1980; Owens 1995). On the other hand, cold weather and sea ice retains and perpetuates oil (Percy and Mullin 1975) and hampers spill countermeasures (Webb 1995). Even from a temperate shore, where a freeze event occurred six weeks after a spill, Dethier (1991) stated that "*oil effects at all habitat types were very difficult to distinguish from freeze effects*".

Oil with sea ice presents the following problems according to Clark and Finley (1982) and Nelson-Smith (1982):

- ice interferes physically with spilled oil containment/cleanup/treatment activities;
- behavior of oil dispersal is much less predictable in sea ice conditions;
- there is virtually no operational experience with large Subarctic/Arctic spills;
- cold oil is viscous (congeals) and is less likely to form a treatable slick;
- viscous oil-water-air emulsions mix vertically in broken ice and are hard to treat;
- sea ice protects oil when it accumulates sub-ice on the water surface;
- sub-ice oil tends to weather more slowly and retain its toxicity longer;
- ice can entrap and transport oil from the spill site; and
- melting ice reintroduces oil at spring breakup during intense biological activity.

Canada is a leader on the environmental impacts of oil on cold-climate marine systems. The first major Canadian contribution was the Beaufort Sea Project sponsored by the Department of Environment in the 1970s. This initiative was stimulated by concerns over potential impacts of Arctic offshore petroleum exploration, production and transport. The resulting FEARO (1984) and Beaufort Sea Technical report series provided a research benchmark such as the review by Percy and Mullin (1975) on effects of oil on Arctic marine invertebrates. There were other studies such as the 1978 to 1981 Offshore Labrador Biological Studies (OLABS) Program. OLABS was an industry, government and communities' initiative to gather marine environmental baseline data. These were augmented with other agency-supported projects including offshore (LeDrew and Gustajtis 1979) and inshore areas (Barrie et al. 1980; McLaren 1980; Rosen 1979; Gilbert et al. 1982, 1984). A series of GIS-based oil spill shoreline sensitivity atlases were developed Arctic, Atlantic and Pacific shorelines as previously discussed. Environment Canada also developed the Arctic and Marine Oilspill Program (AMOP) in the 1970s to research spill countermeasures, and it is still producing conference proceedings.

The most important individual cold-water research program worldwide was the Baffin Island Oil Spill (BIOS) Project (Sergy 1986; Humphrey et al. 1992; Owens et al. 1994). In BIOS, the Canadian government and Canadian oil industry collaborated with US, UK and Norwegian entities to monitor the effects of experimentally spilled oil (treated and untreated) in the Arctic for over a decade. Publications from this work are cited throughout this review.

IMPORTANCE OF HABITAT

Habitat is the single most important influence on impacts of oil in marine ecosystems (API 1985; NAS 1985; Neff 1987; Baker et al. 1990; Baker 1991; Rolan and Gallagher 1991). Intertidal habitats are exposed to much higher concentrations of oil than subtidal habitats (Ballou et al. 1987). Benthic (sea bottom) habitats are generally more affected by contact with oil than pelagic (open water) habitats. The two key intertidal habitat variables are exposure to wave action and substrate (Baker 1991). Table 4 lists sensitivity rankings of marine habitats (excluding tropical) to oil; augmented from the American Petroleum Institute (API 1985).

Table 4.   Relative Sensitivity of Marine Habitats to Oil.

Sensitivity Ranking	**Habitat Type**
High	Saltmarsh
	Sheltered Rocky Intertidal
	Sheltered Intertidal Flat (mud to cobble/boulder barricade)
	Special Use (endangered species/marine protected areas)
Medium – High	Seagrass Meadow (low intertidal to shallow subtidal)
Medium	Open Water Enclosed Bays and Harbours
Low – Medium	Exposed Sand/Gravel/Cobble Intertidal

Low	Exposed Rocky Intertidal
	Kelp Forest Subtidal
	Open Water, Non-enclosed Nearshore and Offshore
	Soft Bottom to Rocky Subtidal

Sheltered habitats with fine-grained sediments are highly sensitive whereas exposed rocky shores have a relatively low sensitivity to oil pollution. In sheltered, fine-grained habitats oil tends to linger whereas on exposed rocky shores oil is subject to rapid removal by wave action. Moreover, rocky intertidal species are adapted to counteracting the stressful effects of desiccation and these adaptations can help them against oil. The importance of habitat is also reflected in the Vulnerability Index and habitat recovery generalizations (excluding tropical habitats) listed in Table 5.

Table 5. Vulnerability Indices of Shoreline Types to Oil Damage (Gundlack and Hayes 1978; Baker 1991; Teal et al. 1992; Owens et al. 1994 and others).

Vulnerability Index[1]	Shoreline Type	Comments
10	Marine Wetlands	Very productive aquatic ecosystems; oil can persist for decades
9	Sheltered Tidal Flat Boulder Barricade Beach	Areas of low wave energy-high biological productivity; oil may persist for decades
8	Sheltered Rocky Coast	Areas of reduced wave action; oil may persist for over a decade
7	Gravel Beach	Same as Index 6; if asphalt pavement forms at high spring tide level it will persist for decades
6	Mixed Sand/Gravel Beach	Oil may undergo rapid penetration/burial under moderate to low-energy conditions; oil may persist for decades
5	Exposed Compacted Tidal Flat	Most oil not likely to adhere to or penetrate the compacted sediments
4	Course Sand Beach	Oil may sink and/or be buried rapidly;

		under moderate to high-energy conditions oil likely removed naturally within months
3	Fine Sand Beach	Oil does not usually penetrate far into the sediment; oil may persist several months
2	Eroding Wave-cut Platform	Wave-swept; most oil removed by natural processes within weeks
1	Exposed Rocky Headland	Wave reflection keeps most oil offshore

[1] 10 = most vulnerable, 1= least vulnerable; index is a qualitative rank order.

On the high seas, the belief that oil is "*cleansed from open waters within weeks*" (Strickland 1990) has been challenged by Howarth (1989, 1991) - but it is relatively rapid. On shores exposed to strong wave action, oil persists for up to a year, whereas oil in sheltered muddy areas can persist over 20 years (Baker 1991; Vandermeulen and Singh 1994) and likely considerably longer if buried in anoxic sediments (Teal et al. 1992).

Two long-term (over 12 years) Arctic post-BIOS assessments have been reported. They were from sheltered, low wave energy, low permeability sedimentary intertidal sites that experienced an annual average of 63 open-water days (Humphrey et al. 1992; Owens et al. 1994). These authors claimed "*clear evidence of natural cleaning*" by processes such as clay-oil flocculation, because <10% of surface oil remained after 12 years. Asphalt formed in the high intertidal after two years, locking up ~50% of the residual oil. It was removed "*slowly but continuously*". Asphalt in the supra tidal backshore was not subject to dynamic marine processes and was expected to "*…. persist for a very long time*" (Humphrey et al. 1992). Subsurface oil is also expected to persist for many years but its fate is poorly understood in Arctic environments.

Observations over 22 years from the *Arrow* spill on temperate Nova Scotia shores yielded similar results. Vandermuelen and Singh (1994) found that even low molecular weight components of subsurface oil in the sheltered porous intertidal sediments were virtually unchanged after 20 years, despite regular tidal inundation. The transfer of oil from undisturbed sediments was in the parts per billion range. They proposed that oil which penetrates sediments before much weathering forms the major reservoir of residual oil. Persistence of oil was a direct function of beach permeability and depth of penetration. Owens et al. (1994) found the surfaces of the heavily oiled, most sheltered sites receiving no wave action to be "*virtually oil free*", which they attributed to "*natural self-cleaning*". Isolated patches of asphalt, to 10 cm thick, were found as well as unweathered subsurface oil. The persistence of asphalt was related to rapid formation of weathered crust in high spring tide deposition sites and a lack of fines for the clay-oil flocculation process (Owens et al. 1994).

The recovery times for oiled marine communities range from a presumed few weeks in open waters, to a year in the most exposed habitats. It may take decades for ecosystems to fully recover to their original pre-spill states (Ballou et al. 1987; Baker et al. 1990; Baker 1991; Suchanek 1993; Owens et al. 1994). Fuick et al. (1984) caution, however, that recovery is no easier to define or measure than is damage; and difficult to assess if there are no pre-spill baseline data. Jackson et al. (1989) speculated that ecological changes induced by oil on slow-recovering (e.g. cold climate) communities will likely continue long after all traces of the oil are gone.

SUBLETHAL EFFECTS OF OIL

The long-term, chronic sublethal effects of oil pollution are perhaps the most contentious post-spill environmental issue. These effects have been difficult to demonstrate in nature. Further, they remain relatively speculative with excessive reliance on extrapolation from laboratory experiments. After reviewing laboratory studies, Moore and Dwyer (1974) proposed that low levels of soluble oil components could effect fish and invertebrate behaviour just as minute amounts of natural pheromones do, but this remains speculative. Moreover, scientific opinion is divided on cumulative environmental degradation due to chronic, low-level oil pollution (Strickland 1990).

Acute, short-term toxic effects of fresh oil tend to be caused by low molecular weight fractions that weather relatively quickly. Chronic sublethal toxic effects are due mostly to high molecular weight (polynuclear aromatic hydrocarbon - PAH) fractions, which have a relatively minor role in acute effects, but persist to cause long-term effects (Spies et al. 1996).

Howarth (1991) suggested that most analyses underestimate the potential of oil to cause harm by not taking into account the importance of sublethal effects. He, along with Nelson-Smith (1982), speculated that even very low levels of oil pollution can have significant long-term deleterious effects on species. Clark and Finley (1982) contended that in cold waters, *"...sublethal physiological and behavioural effects on organisms are likely to be of more lasting ecological significance than immediate lethal effects.*" Suchanek (1993) speculated that sublethal effects of marine invertebrates to concentrations in the range of 1 to 10 parts per billion (ppb) "*.... likely produce dramatic population level changes*." He proposed that physiological, carcinogenic and cytogenetic effects can occur, among which the most common physiological effects involve reproduction, growth, respiration, excretion, chemoreception, feeding, movement, responses to stimuli and susceptibility to disease.

Bienert and Pearson (1995), Spies et al. (1996) and Bue et al. (1998) summarized the difficulties for demonstrating sublethal effects:

- an accurate and precise estimate of oil exposure is difficult to obtain, e.g., hydrocarbon levels in tissues are often inconclusive for measuring exposure;
- massive initial injury and the dynamics of early recovery can also mask chronic effects;
- other indicators or biomarkers such as enzyme induction, histopathology and cytogenetic markers are still contested in the literature;
- many recent sublethal studies are yet to be published;
- potential exposure to oil through food pathways needs more research; and
- studies need to consider synergistic effects of natural variability and exposure to non-chemical stressors.

To this could be added the often inadequate commitment to long-term post-spill field monitoring.

Studies from the EVOS are providing important contributions because funding permitted longer-term research. Nonetheless, significant problems were suggested by Spies et al. (1996):

"Distinguishing initial acute effects of oil from chronic sublethal effects is further obscured by the occurrence of secondary, density-dependent after-effects of the initial acute mortality, and by substantial delays, or lags, between the initial exposure and its ultimate expression at the population level."

Further, there remain disagreements among specialists on EVOS results and Bienert and Pearson (1995) concluded:

"*Questions concerning potential sublethal exposure to oiling may, in fact, never be adequately resolved with existing data. In situations where data are lacking and uncertainty exists, the environmental risk assessment process* (of the U.S. Environmental Protection Service [EPA 1992]) *calls for the use of best scientific judgement, and cautions that conclusions must be presented with the appropriate qualifications. An inevitable consequence of this process, however, is that consensus among scientists may not always be achieved."*

BIODEGREDATION OF OIL

" *…. bacteria are the gateway by which some of the carbon and energy contained in the oil finds its way into the upper echelons of the food web by way of protozoa, zooplankton and meiofauna.*" Floodgate (1995)

"*An increase in the biodegredation activity of naturally occurring populations of microorganisms can lead to substantial removal of petroleum from the environment.*" (Braddock et al. 1996)

The omnipresent marine oil-degrading bacteria, representing a wide range of genera, are the ultimate biological agents of breakdown of weathered oil (Floodgate 1995). Despite their importance to ecosystem restoration, studies on role of marine bacteria in post-spill situations has been minimal. Very fine silty intertidal sediments tend to support low bacterial populations because their small grain size, low organic content and small interstitial spaces renders them relatively unresponsive to the organic enrichment of oil (Norrell and Johnston 1976). In their three-year post-EVOS study, Braddock et al. (1996) reported that populations of hydrocarbon degraders were always higher in oiled intertidal sediment than at unoiled reference sites - diminishing steadily towards background levels within two years. In contrast, sediment bacteria numbers at 20 m to 100 m depths peaked in the first post-spill year, indicating oil transport into offshore sediments.

Anaerobic conditions of buried oil in marsh sediments may be responsible, in part, for oil longevity exceeding 20 years (Teal et al. 1992). Whether persistence of buried oil in such conditions is due to the anoxia, low nutrients, or both is unknown (Floodgate 1995). Bacterial attack on oil under anaerobic conditions has been observed in the laboratory, but not in nature (Floodgate 1995).

The EVOS fostered the widest implementation of bioremediation yet undertaken. Bioremediation was defined by Floodgate (1995) as: "*a field procedure designed to increase the rate of natural degradative processes by changing the rate-limiting factor to a faster one*." Bioremediation seeks to promote the presence of oil-consuming bacteria mostly through adding environmental nutrients of nitrogen and phosphorous to supplement the nutrition of bacterial populations that consume oil. Although government agencies promoted the widespread use of bioremediation after the EVOS (USOTA 1991), scientists remain sceptical as to its utility beyond experimental applications (Floodgate 1995).

MARINE PLANTS AND ASSOCIATED COMMUNITIES

Seaweeds or macro algae are usually epibenthic (grow on hard substrates) and seagrasses root in sedimentary substrates. Both constitute productive nearshore marine ecosystems. Other algae species are wholly planktonic, single-celled micro algae comprising phytoplankton and benthic micro algae comprising epontic ice flora.

Phytoplankton

The acute and chronic (sublethal) effects of oil on phytoplankton in nature remain poorly understood (Howarth 1991). Toxicity is closely related to the amount of dissolved, non-volatile components of the oil (Ostgaard 1994). There are laboratory studies and field studies using large enclosures containing a portion of the water column and possibly underlying sea bottom (Spies 1987). Howarth (1989) concluded that enclosure experiments yielded the "best information of the ecological effects of oil", particularly for low-level chronic contamination of species composition of plankton communities.

Diatoms, perhaps the dominant phytoplanktonic algal group, are well reported in laboratory studies on temperate species. In a review of laboratory studies, Capuzzo (1987) stated that oil could be lethal or reduce photosynthesis and growth in phytoplankton, and, at low concentrations, stimulate phytoplankton growth. Depending on the amount and type of oil, and the species, effects in experiments vary from death to growth stimulation (Hsiao 1978; Dahl et al. 1983; Ostgaard et al. 1984; Morales-Loos and Goutx 1990). Laboratory studies are, however, of limited use when applying their results to predicting plankton well-being, abundance and distribution in response to oil in nature (Foster et al. 1990). For example, season must be considered as the time of blooming represents a period of greatest vulnerability to impacts of oil. Finally, natural planktonic systems have characteristically large spatial and temporal fluctuations among which effects of oil would be hard to identify (Howarth 1989).

There is little evidence of long-term oil damage to phytoplankton in open waters (Teal and Howarth 1984; Wells and Percy 1985; Reid 1987; Shin 1988; Strickland and Chasan 1989; Foster et al. 1990; Baker et al. 1990; Baker 1991; Ostgaard 1994). It seems likely that phytoplankton in enclosed inshore waters are likely more vulnerable to oil than open-water populations. Spies (1987) speculated on "*transitory effects*" of oil on plankton in nature in stating that the "*effects are measurable for only several days, if at all*". Johansson et al. (1980) found that phytoplankton abundance near a spill showed a short-term increase because of the suspected mortalities of zooplankton grazers. On the other hand, Howarth (1989, 1991) has warned of longer residence time of oil in the water column than is widely accepted and challenged the assumptions that dissolved oil is rapidly dissipated in the water column and that evaporation removes most oil toxins within days.

Benthic Algae

Multicellular algae (seaweed) have no vascular system, roots, flowers or seeds. Many species are characterized by alternating life phases. The benthic life phase of seaweed is the diploid (two sets of chromosomes) "sporophyte". Surface structures produce the short-lived haploid (one set of chromosomes) planktonic life phase of spores that eventually settle on the bottom to develop into tiny male and female "gametophytes". Male gametophytes release planktonic gametes that seek out female gametophytes. The diploid plant created by their fusion grows into seaweed.

It is important to differentiate between intertidal and subtidal seaweed. Because of the possibility of oil deposition, oil residence and the greater ease of observations, oil impacts on intertidal algae are better recorded than those on subtidal algae. Overall, the literature on oil and algae is limited (Strickland and Chasan 1989).

Epontic Microalgae

Single-celled "epontic" micro algae grow on the undersurface of sea ice. A rich, ephemeral bloom, dominated by shade-tolerant diatoms, occupies a few centimeters of ice in spring; before the open-water phytoplankton bloom. Epontic ice flora can account for up to 30% of the annual productivity in the water column (Clark and Finley 1982). Being at the water-ice interface, epontic algae are extremely vulnerable to oil pollution and their destruction would have significant ecological effects (Nelson-Smith 1982).

Intertidal Seaweed

Reports on the effects of oil on intertidal algae from higher latitudes are reviewed by Cross et al. (1987a). Results range from widespread death and disappearance, to decrease in vertical distribution of seaweed on shores, to necroses and bleaching, to no measurable effects. Within the same spill, effects can vary from mortalities of some species, to replacement by others, to no observable effects. Also, increases in vertical distribution of algae due to oil-related mortalities of herbivores are known. There is little effect on seaweed growth from weathered residual oil as its toxicity is low (Topinka and Tucker 1981; Foster et al. 1990).

Overall, damage and death can occur after physical contact with oil, growth in highly oiled areas may decrease and growth in lightly oiled areas may increase due to increased nitrogen availability (Topinka and Tucker 1981). The highest intertidal populations of rockweed (*Fucus* sp.) recovered more slowly after EVOS than mid and lower intertidal populations (VanTamelen and Stekoll 1996). Topinka and Tucker (1981) reported recovery and return to normal growth rates of *Fucus* 14 to 16 months after a spill. VanTamelen and Stekoll (1996) suggested a population recovery time of 36 months for untreated (not cleaned) shores. DeVogelaere and Foster (1994) and VanTamelen and Stekoll (1996) found low *Fucus* cover on intensively treated (high pressure - hot seawater) shores 30 months after the EVOS. The post-EVOS treatment slowed rockweed recovery (Duncan and Hooten 1996; Lees et al. 1996) and was considered "*not environmentally justified*" (DeVogelaere and Foster 1994).

Likely as important as the temporary damage to seaweeds is the damage to biologically structured communities associate with seaweeds including nursery habitat for some animals (Boesch and Rabalais 1987). Crothers (1983) and Teal and Howarth (1984) suggested that animals associated with intertidal algae were more sensitive to oil than the seaweeds themselves.

Subtidal Seaweed

There are very few studies on the effects of oil on subtidal seaweeds. Overall, subtidal seaweed is considered less vulnerable to oil than intertidal seaweed because greater contact with oil and cleanup impacts is experienced by the latter (Baker et al. 1990). Subtidal seaweeds in higher latitudes suffer little direct impact in oil spills, and may even show increased growth due to herbivore mortalities (Conan 1982; Maurin 1984). In an Arctic study, Cross et al. (1987a) found no detectable impacts on biomass, density or reproduction of shallow (3 m depth) species. Jewett et al. (1995) and Dean et al (1996a) reported no significant differences in plant density, biomass or percent cover between post-EVOS oiled and reference sites at 2 to 20 m depth. They speculated that an increase in small plant density at oiled sites could have represented a recruitment event in response to an unknown post-spill impact.

The impacts of oil on subtidal kelp (large brown seaweed) forests are likely to be less than effects on intertidal seaweeds because oil does not contact submerged plants and the surface mucus of moist seaweed decreases adherence of oil compared to exposed, dried seaweed (Capuzzo 1987; Baker et al. 1990). There could be an indirect positive effect if oil kills herbivores, (e.g. sea urchins which are very sensitive to oil) and plant biomass can increase (North et al. 1965). The concern for biologically structured communities associated with plants is relevant for subtidal seaweeds as they shelter important inshore communities and supply organic detritus for inshore food webs.

The aspects of sublethal effects of oil on seaweeds remains unclear (Strickland and Chasan 1989). Interference of the chemically-mediated orientation behaviour of their flagellated gametes could be important. Thus, the season, reproductive cycle, and alternate generations of seaweeds are relevant when considering effects of oil.

Seagrass

Seagrasses are rooted, flowering vascular plants that have adapted to the sea from land ancestors. In lower intertidal and shallow subtidal sand/mud shores they can form dense, productive meadows that provide nursery areas for young animals, full-time habitat for others and be a source of organic detritus important to inshore food webs (Zeiman et al. 1984). Seagrasses spread vegetatively by horizontal underground stems (rhizomes) which, with the roots, form dense mats in the substrate. Broad-scale dispersal occurs by sexual reproduction involving seed-release from pollinated flowers.

Most information on oil impacts on seagrasses comes from the tropics. This information is reviewed for reference and comparable observations need to be completed in cold water areas. Regeneration of oiled seagrass is common and relatively rapid, provided that the root/rhizome mat is intact (Jacobs 1980; Zeiman et al 1984; Fortes 1988; Strickland and Chasan 1989). Oil adheres to seagrass more readily than to algae (Harding and Elgar 1989). On an oiled reef flat, Jackson et al. (1989) reported that intertidal seagrasses (leaves, root mats and associated infauna) perished while subtidal portions of the meadow survived. Jewett et al. (1995) found that mean shoot densities were lower at oiled subtidal sites than unoiled sites in the first year after the EVOS only.

As with seaweeds, there has been concern that oil affects the biologically structured community associated with seagrasses (Boesch and Rabalais 1987). Phillips (1984) speculated that seagrasses may be less sensitive to oiling than their associated fauna. On the other hand, some species may flourish and Jewett et al. (1995) reported that oiled seagrass sites yielded more epifauna than unoiled sites a year after the spill.

The most useful approach to assessing oil pollution effects on seagrasses is to consider meadow ecosystems as a whole. Zeiman et al. (1984) listed potential ecosystem impacts while stressing that little work is being done on this field. Potential damage to seagrass ecosystems includes:

- direct mortality due to smothering and poisoning, or absorption of dissolved toxins;
- indirect mortality of associated species due to loss of food sources or habitat destruction;
- mortality of juvenile fauna using seagrasses as nurseries;
- incorporation of sublethal oil fractions into tissues and attendant lowering of tolerance to other potential stresses; and
- incorporation of oil toxins into seagrass-associated food webs.

WETLANDS

Coastal wetlands are highly productive ecosystems important to biodiversity, fisheries, subsistence, coastal protection, and wildlife (Getter et al. 1984). These are sheltered habitats which filter out nutrients from land runoff and where fine sediments rich in organic matter accumulate.

In temperate and Subarctic areas, marine wetlands are saltmarshes dominated by grass and herb species. Chronic oil pollution severely reduces marsh grasses and can destroy salt marsh ecosystems (Baker et al. 1990).

Although weathered oil is less toxic to marsh plants, it still suffocates them. Oil can penetrate soils and adversely affect root systems, microbial populations and oxygen content. Oiled surface vegetation threatens roots needing oxygen diffusion from leaves. As plant recovery is dependent on surviving roots, perennial species can survive by regeneration from healthy roots. Maximum oil damage is to seedlings and annual species, especially in the spring and summer growth season.

Oil adheres to most salt marsh plants and is not readily washed off by the marsh's irregular tidal regime (Baker et al. 1990). Temperate saltmarshes recover from single spills within months to a few years (Baker et al. 1990). Clarke and Ward (1994) found that oiled plants failed to resprout or recruit seedlings 17 months after oiling. Fauna associated with salt marshes suffer from oil coating because of the infrequent tidal inundations, but there is some indication that population recovery is more rapid than for plants (Clarke and Ward 1994). Baker et al. (1993) reported that thick, untreated oil stranded during a high spring tide in highly sheltered saltmarsh areas remained in large amounts 17 years later and continued to exert negative impacts to vegetation. Teal et al. (1992) found that oil impacted a marsh ecosystem for up to 6 post-spill years but after 20 years residual effects were "*extremely small*" provided the oil remained undisturbed in sediments. They suggested that an important question remains as to at what point between six and 20 post-spill years could oil be expected to no longer exert effects on saltmarshes.

INVERTEBRATES

Different life modes greatly influence the vulnerability and response of invertebrates to oil pollution (Strickland and Chasan 1989). The literature has been dominated by short-term mortality studies of single species and needs to progress to longer-term impact studies of ecosystems (Suchanek 1993). When considering invertebrates, it is useful to differentiate between:

- planktonic and benthic life modes;
- intertidal and subtidal species; and
- habitat types occupied.

Pelagic Invertebrates - Zooplankton

"T*he pelagic environment is the first to be impacted by an oil spill, but apparently recovers quickly, depending on the time of year, zooplankton life cycle stages, and the amount of contact with oil*". (Horner 1981)

Zooplankton includes species which spend their entire lives in the water column (holoplankton) and those benthic species which have early planktonic life stages for dispersal (meroplankton). It is generally held that plankton are unlikely to suffer long-term impacts from spilled oil due to rapid dilution and dispersion of oil in the water column (Suchanek 1993). Johansson et al. (1980) found that densities declined significantly near a spill, but biomass reestablished within five days. Overall, effects of oil on zooplankton are greater at the water surface than at depth and greater in enclosed inshore waters than in open seas.

Field research carried out by use of large enclosures suggests that zooplankton community structure may be susceptible to low oil concentrations (Davies et al. 1980; Teal and Howarth 1984; Reid 1987). Spies (1987) listed some sublethal effects of oil on feeding and other behaviors, reproduction and development in enclosed zooplankton. Many species contain high levels of natural oils in their eggs or overwintering cysts as energy stores, so it is possible that they could incorporate petroleum (Reid 1987). Zooplankton may be important in transporting oil into sediments. For example, copepods will ingest oil droplets, which they apparently do not metabolize (Teal and Howarth

1984), and release them in their fecal pellets which settle to the bottom (Capuzzo 1987). Alternatively, the ingested oil could be transferred to copepod predators.

Benthic Invertebrates

Benthic studies dominate the literature on the effects of oil on marine invertebrates. Howarth (1989) generalized these effects as follows:

- sensitive species die the soonest;
- oil-resistant opportunistic species flourish; and
- overall species diversity and biomass decrease.

Suchanek (1993) described a generalized oiled invertebrate community response as starting with an acute decrease in abundance and diversity, then a slow reestablishment of diversity while individual species fluctuate with decreasing amplitude over time until comparable pre-spill communities are established. Suchanek (1993) warned that some invertebrates have a disproportionately high influence of local community structure such that oil impacts on them could significantly influence species at another level (i.e., a "*trophic cascading*" effect). An example would be grazing of sea urchins, which are highly sensitive to oil, whose herbivory can structure nearshore communities. Suchanek (1993) stressed the need to focus on critical species-to-species interactions for improved invertebrate community impact studies and underscored the importance of a sound baseline such as species composition, measures of relative community health and those important species-species interactions.

Habitats are central when discussing effects of oil on invertebrates. For example, the longest-term effects of oil occur in sheltered areas with fine sediments, whereas species on exposed rocky shores tend to recover from oiling much quicker (Gundlach et al. 1983; Teal and Howarth 1984; Shin 1988; Baker 1991). Invertebrate community recovery usually occurs over a range of one to ten years. Recovery rate depends on substrate, exposure to waves, amount and type of oil, season of initial oiling and species (Boucher 1985; Baker et al. 1990; Baker 1991). Baker et al. (1990) concluded from recovery studies that more than ten years was unusual whereas Suchanek (1993) suggested a general range of two to 15 years. Special cases such as the formation of asphalt pavements and the entrapment of oil in fine, anaerobic sediments can lengthen recovery longer than ten years (Baca et al. 1987). Heavily oiled nearshore tidal channels with standing water and isolated populations can take a decade to fully recover (Dauvin and Gentil 1990). Baker et al. (1990) stated that: "*once the damaging effect of oil is sufficiently reduced, the continuous nature of the marine environment ensures that eventual community recovery is inevitable. This usually takes* (in the subtidal) *one to five years*".

The acute effects of dispersed oil are greater, although shorter-lived, than undispersed oil on benthic invertebrates (Crothers 1983; Cross and Thompson 1987; Cross et al. 1987b; Mageau et al. 1987; Neff et al. 1987). This is also true for the greater narcotizing effect of dispersed oil compared to untreated oil on invertebrates (Wells and Percy 1985; Mageau et al. 1987). In an Arctic field experiment, dispersed oil affected invertebrates at greater depths (three to seven meters), than undispersed oil (Cross and Thompson 1987).

Intertidal Invertebrates

Oiling of intertidal species has been more studied than for subtidal species because:

- intertidal oil is more visible and receives more public interest;
- intertidal species experience relatively more direct contact with oil; and
- the greater ease of intertidal surveys compared to subtidal surveys.

Spies (1987) summarized the following three stages of effects of oil on intertidal invertebrates, which are similar to Suchanek's (1993) generalizations:

- brief period of high moralities of sensitive species such as small crustaceans and meiofauna (infauna that pass through a 0.5 mm mesh sieve but are retained on 0.05 mm mesh) (Elmgren et al. 1983; Bodin 1988) and sea urchins (Cross et al. 1987 b) - generally, meiofauna recolonize oiled sediments relatively quickly, often within a month (Fleeger et al. 1996);
- medium-term period during which some organisms prevail at normal expected levels, with proliferation of opportunistic species, such as polychaetes (Ibanez and Dauvin 1988); and
- eventual return of the original community (Dauvin and Gentil 1990).

Natural patchiness of intertidal species in areas of similar habitat type, interannual variability and other natural site-to-site differences (tidal elevation, exposure, substrate, and distance from spill) complicate oil impact assessments (Dethier 1991; Stekoll et al. 1996). These complexities, coupled with usually poor baselines, present large uncertainties in the assessment of oil impacts to the intertidal. Post-EVOS shoreline impact assessments were further complicated by four types of shore treatment (cleanup), among which high pressure – hot water was the most destructive (Lees et al. 1996). Many authors have since been critical of the highly intrusive EVOS cleanup practices, but the politics of the moment demanded the perception of cleanup during the crisis regardless of science-based advice (Foster et al. 1990; DeVogelaere and Foster 1994; Spies et al. 1996).

In an Exxon-funded study of Prince William Sound, Gilfillan et al. (1995a) suggested that by summer 1990 (~16 months later), 70 to 90% of shorelines examined had "*largely recovered*", being indistinguishable from reference sites. This was reflected from findings of parallel chemical and toxicological studies in which hydrocarbon toxicity was very low in intertidal sediments by 1990. Oil effects were, however, actually the combined oil and cleanup effects and difficult to separate. In a similar study in more open Gulf of Alaska shorelines, Gilfillan et al. (1995b) found that oiling of mid and high intertidal communities was also highly patchy, although less affected overall than those of Prince William Sound. There were few significant differences between oiled and reference sites by 1990 and mussel tissue hydrocarbon levels were "*near background levels*".

Agency-funded studies, also dealing with the combined effects of oiling and treatment, reached different conclusions. In a general survey of intertidal impacts, Highsmith et al. (1996) and Stekoll et al. (1996) concluded that injuries had occurred to a wide range of organisms and their communities, particularly in the mid and upper intertidal. Driskell et al. (1996) found that lower intertidal infauna of oiled but untreated sediment shores "*rebounded quickly*", however, the recovery of treated sites was lagging significantly behind that of untreated sites by 1992. Houghton et al. (1996) reported that the epibiota of rocky sites (oiled, unoiled reference and treated) could survive the oiling, but not the most rigorous treatment type. By summer 1991, there were no significant differences between reference and oiled-but-untreated sites. Although there were some visible impacts by summer 1992, but there were "*few differences*" remaining between reference and treated beach communities. An exception was untreated intertidal mussel beds in which their underlying sediments retained oil and continued to contaminate mussels, and perhaps their predators, four years later (Babcock et al. 1996). Contamination was greatest in mid intertidal mussels compared to lower intertidal populations and occurred by uptake of whole particulate oil rather than dissolved fractions of oil (Short and Babcock 1996). Mussels were identified as a hydrocarbon vector into nearshore predator populations such as young sea otters (Loughlin 1994). Overall, treated sites were slower to recover and Lees et al. (1996) reported some high pressure-hot water treated areas as showing damage in 1995 - six years post-EVOS.

Subtidal Invertebrates

The subtidal is protected from contact with oil, which usually floats, but oil can enter subtidal sediments in sufficient concentrations to affect infauna species (Teal and Howarth 1984). Oil enters the subtidal after initial intertidal contact by being adsorbed onto particulates transported seaward (Baker et al. 1990). There can also be natural sinking of oil particles, the enhanced introduction of oil into the water column by use of chemical dispersants and biological processes of oil transport by plankton (Howarth 1989).

The acute effects of oil spills on offshore benthos have been little studied, because oil contamination there tends to be low and to "*cause little harm*" (Howarth 1989). From what little has been researched, deeper offshore infauna may recover more quickly than shallower inshore benthos (Sanders et al. 1980) and disturbance effects can reasonably be expected to decrease with depth (Feder and Blanchard 1998). Sixteen months after the EVOS, Feder and Blanchard (1998) reported that deep (40 to 100 m) benthic infauna communities showed no difference between reference and under-oil trajectory stations. O'Clair et al. (1996) found oil in shallow subtidal sediments at 3 to 20 m depth seaward of oiled shores, but little evidence of oil in sediments at >40 m depth. Armstrong et al. (1995) examined effects on pelagic, epibenthic and infauna invertebrates from 20 to 230 m depth in Prince William Sound after the EVOS and found no significant effects at individual or population levels for any life stage. Five months after the *Braer* spill, Kingston et al. (1995) found little evidence of impact on infauna macrobenthos at 74 to 124 m depth, although Moore and Stevenson (1997) reported that the abundance of certain meiofauna copepods from the same sample sites demonstrated a negative correlation with oil content. For other groups of meiofauna, the effect of natural sediment type on species diversity and abundance obscured possible oil impacts.

In the shallow subtidal (<20 m depth) post-EVOS studies, impacts were discussed in the context of epibenthic invertebrates associated with seagrass or kelp communities (Jewett et al. 1995; Dean et al. 1996b). The shallower sites were seagrass meadows and they showed greater impacts than the deeper, kelp-associated sites (Jewett et al. 1995). Some groups such as amphipods and snails demonstrate acute impacts while others such as polychaetes showed positive, opportunistic responses in abundance and biomass. Within 4 years, oiled and unoiled sites showed little differences although 2 larger epibenthic species (a sea star and a crab) continued to be less common at oiled sites.

Howarth (1989) was more concerned about oil entering offshore sediments from chronic sources such as oil production platforms. By far, the major route is through deposited drill cuttings impregnated with oil lubricants (Kingston 1992) and heavy metals (Montagna and Harper 1996). Gray et al. (1990) found that at varying distances from an oil source, some species increased in abundance and there were changes in presence/absence patterns of rare species. As well, the platform itself acts as a reef attracting higher biological productivity and enriching nearby sediments with organic detritus (Montagna and Harper 1996). Kingston (1992) identified the following infauna community responses detected near North Sea platforms:

- "smothering-toxic effect" of low numbers among few species; or an
- "organic enrichment effect" of a few abundant species at the most polluted sites.

This contrasting response pattern was also reported from Gulf of Mexico platforms (Montagna and Harper 1996). As oil weathers, its toxic components dissipate and some species use petroleum hydrocarbons as an energy source. Spies et al. (1988) concluded that the response of benthos to crude oil in sediments was similar to that expected from general organic enrichment.

FISHES

Adult fish kills attributed to oil have been reported for shallow-water, near-shore species (Teal and Howarth 1984), but not for offshore species. The study of both acute and sublethal effects of oil on young and adult fish has, until recently, relied mostly on laboratory results rather than experimental studies from nature. Further, until EVOS, long-term or multigenerational studies did not exist.

The key issue for fish is the extent to which a species' life history stages put them at risk. If a species' life cycle is wholly pelagic and not associated with the sea surface or the intertidal, then little direct impacts could be expected. Baker (1991) found no evidence of an oil spill significantly affecting adult fish populations in the open sea. Moreover, she found that no subsequent adult population decreases have yet been linked to larval fish kills from oil pollution. If any part of a fish's life cycle occurs near-surface or in intertidal habitats, however, they can be directly exposed to oil or oiled habitat.

Adult fishes can directly ingest oil or oiled food, take up dissolved oil compounds through their body surfaces, or their eggs and larvae can become contaminated. Direct impacts of oil on fishes are immediate acute toxic effects, physical (mechanical) effects and chronic contamination. Acute effects include death or debilitation due to central nervous system disruption, osmoregulatory dysfunction, metabolic dysfunction or tissue (histological) damage. Central nervous system disruption can be directly lethal or indirectly lethal through behavioural modification that renders the fish unable to avoid predators or perform vital functions. Stressed fish are less able to escape predation, resist disease, feed or adapt to changing environmental conditions.

Histological studies have documented oil damage to a wide range of fish organ systems such as the liver, gills, gut, brain and ovaries. Physiological responses to oil include increased heart beat, changes in respiration and blood parameters, ionic and osmotic imbalances and decreased energy reserves.

Physical or mechanical effects occur when oil adheres to fishes. Indirect physical impacts of oil occur when fish habitats entrap oil. Residual, heavy fractions of oil have the greatest potential for bioaccumulation in fish, particularly when they exceed their ability to metabolize those compounds.

The early lives of fishes are more vulnerable to oil due either to their greater sensitivity, or to their greater proximity to the water surface, or both. In general, eggs and larvae are more sensitive than juvenile stages which, in turn, are more sensitive than adults. Oil can affect developing fish by retarding growth, causing premature hatching and causing developmental or genetic changes (Carls and Rice 1990). Many marine fish species have floating eggs or larvae which inhabit near-surface waters; putting them in close proximity to slicks at vulnerable times in their lives. Moreover, if the oil is treated with dispersant, this will likely increase the toxic effects on the young fish and eggs (Baker

1991). On the other hand, while floating eggs and larvae are vulnerable, many are produced and they tend to disperse widely which decreases their likelihood of contact with oil.

Oil can indirectly affect fishes through contamination and/or loss of habitat and food sources, and disruption of food webs supporting fish populations. Habitat loss can occur after oil coats the bottom and the time required for habitat recovery will vary according to habitat type. Food sources are damaged through direct mortalities of prey or food contamination by oil and possible bioaccumulation of oil compounds in prey. Oil impacts on fish food resources could cause the loss of sensitive prey species and an increase in abundance of more oil-tolerant species.

In an EVOS study of fish that spend their most of their life cycle in the intertidal, Barber et al. (1995) found that species density and biomass were significantly higher in unoiled reference sites one year after the spill, but no different another year later – suggesting "*strong evidence*" of recovery. In a shallow subtidal (<20 m depth) EVOS studies Jewett et al. (1995) and Laur and Haldorson (1996) found that one year later, oiled kelp and seagrass sites had more fish than reference sites because of higher young-of-the-year abundance. They speculated that this could have been due to more prey, less predators or more recruitment at oiled sites one year after the spill.

Pink salmon (*Oncorhynchus gorbuscha*) and Pacific herring (*Clupea pallasi*) studies arising from the EVOS provide case studies underscoring the importance of life history characteristics of pelagic fish that put them at risk to oil pollution. As well, these studies also illustrate the effects of research being done in a litigious environment. There were parallel, non-cooperating studies on pink salmon and herring funded separately by Exxon and government agencies that consistently yielded different conclusions. When reading the literature, therefore, the context of who supported the research can be important – despite scientists' and editors' best efforts at objectivity.

Case Study 1: Pink Salmon

Approximately 75% of the pink salmon in Prince William Sound spawn in the upper intertidal near creeks, where oil tended to be deposited, rendering the species particularly vulnerable to oil contamination. All life stages were potentially exposed to oil during the first six post-spill months (April onward) including emigrating fry in the spring and spawning adults from July through October (Marty et al. 1997). Further, in-gravel egg and larval stages were exposed to oiled habitat for up to eight months. Post-EVOS funding permitted long-term observations and, because pink salmon have a generation time of approximately two years, this work was the first multigenerational oil effects study for any vertebrate (Marty et al. 1997).

Exxon-funded studies by Brannon et al. (1995), concluded *"…no substantial effects of critical early life stages."* and that no impacts on wildstock (non hatchery production) adults could be detected (Maki et al. 1995). Agency-funded studies, however, came to very different conclusions. Concerning early life history stages, Willette (1996) reported reduced growth of juveniles from oiled areas and proposed a ~2% reduced survival to adulthood. Geiger et al. (1996) speculated that "*direct poisoning*" of embryos led to 60, 000 to 70,000 less adults returning in 1991 and 1992. In a laboratory study covering fertilization through to emergence, Marty et al. (1997) found persistent significant differences between oiled and unoiled stream fish through 1993. They claimed that long-term, low-level exposure to oil significantly altered early development over two generations. Bue et al. (1998) reported from a protracted series of controlled breeding experiments that a significantly higher mortality of embryos from oiled streams occurred through to 1993, but not in 1994 and 1995. Even though there was evidence of "*dramatic*" declines in intertidal oil contamination by 1991, elevated embryo mortalities persisted; implying either genetic damage or persistence of oil toxicity through 1991 (Bue at al. 1998). In a life history approach to damage estimation, Geiger et al. (1996) claimed that 31% of spawning streams were oiled; whereas Maki et al. (1995) claimed 14% of spawning streams were on oiled shores. Geiger et al. (1996) contended that 1.9 million adult pinks did not return to Prince William Sound in 1990 because of lack of growth in the critical nearshore life stage when they entered seawater in the spring of 1989 during the spill. Maki et al. (1995), on the other hand, reported that the 1990 returns in Prince William Sound were the highest during the four years of 1989 to 1992. The true impacts to adult pink salmon stocks are likely at some point on the spectrum between these two camps and more time for stock status work is needed to find that point with greater certainty.

Case Study 2 : Pacific Herring

In Prince William Sound, large schools of herring congregate in shallow water usually around early April and lay their eggs on lower intertidal and shallow subtidal vegetation.

Although not the area of maximum likely oil deposition, the lower intertidal is still vulnerable. Attached eggs incubate for approximately three weeks before hatching. In 1989, spawning and incubation occurred just weeks after the spill when exposure to unweathered beached oil occurred. Most of the science has focused on young herring.

A similar pattern of opposing views shown for pink salmon applies to reported impacts on Pacific herring. From Exxon-funded studies Pearson et al. (1995 a,b) stated that ~4% of spawning in 1989 occurred along oiled shores. They reported effects from a few areas where oil was seen on eggs and concluded that ~2% of the total 1989 spawn experienced effects of oil contamination. No oil was seen on eggs in 1990 and no negative effects were recorded according to 10 biological response variables for young herring. Pearson et al. (1995 a) concluded; "*Effects on herring eggs were minor in 1989 even in oiled areas.*" In an agency study, however, Brown et al. (1996) claimed that 50% of egg biomass were within the spill trajectory and that >40% "*sustained oil exposure during early development*". Hose et al. (1996) reported that herring larvae hatched in 1989, but not in 1990 and 1991, showed sublethal lesions and morphological and cytogenetic abnormalities and various other authors reported sublethal effects as well (Kokan et al. 1996). McGurk and Brown (1996) found that egg mortalities were greater from oiled sites than unoiled sites. They suggested that oiling led to a decrease in survival and hatching success of late-stage herring embryos, although this could not be conclusive as natural processes of desiccation, predation and wave action also effect mortality rates between sites. Kokan et al. (1996) focused on 1-year olds that could have been contaminated by the EVOS and were returning as 4-year old first-time spawners. Adults collected from known oiled sites in 1992 yielded a lower percentage hatch and significantly fewer larvae than those collected from unoiled sites. Kokan et al. (1996) speculated that possible reasons for this included homing (unverified in Prince William Sound) by herring contaminated in 1989 and contamination from residual oil in beach sediments. Although much of the population level effects on herring remain speculative, the prospect of long-term sublethal effects on young herring cannot be discounted despite great uncertainties with the region's herring natural history.

Concerning adult herring, the 1989 fishery was cancelled, the 1990 harvest was above average and record harvests occurred in 1991 and 1992. The 1993 spawn, the first year that 1989 recruits would begin to spawn, was very poor due to low physiological condition from a parasite (epizooic) outbreak (Brown et al. 1996). There is so much uncertainty about natural factors effecting herring recruitment that spawning success remains unpredictable for Prince William Sound (Brown et al. 1996). Pearson et al. (1995a) claimed that the years of high post-EVOS landings reflected no population-level effects on herring, whereas Brown et al. (1996) underscored that poor understanding of natural recruitment obscures attributing any specific causes at this time.

FISHERIES

Sedentary fishery species such as scallops are more vulnerable to oil than mobile species such as halibut. Intertidal species are more vulnerable than subtidal species and open-water fisheries are not as vulnerable as inshore subtidal fisheries. An exception could be with pelagic, near-surface eggs and larvae of some commercial species (McIntyre 1982). For example, the reproductive success of a flatfish with floating eggs was decreased in the year after a spill (Conan 1982). Salmon from the Gulf of Alaska and other fish and shrimp species commercially harvested from oiled areas of Prince William Sound after the EVOS were deemed safe for human consumption by the U.S. Food and Drug Administration (Saxton et al. 1993).

The U.S. National Academy of Science (NAS 1985) concluded that transferring laboratory study results on effects of oil to predicting and evaluating impacts on commercial finfish and shellfish stocks was exceptionally difficult. Howarth (1991) concluded that despite much research, the effects of oil exploration and production on fisheries, and the marine ecosystems nurturing them, remains largely unknown. There is no conclusive evidence that offshore oil pollution decreases commercial fish stocks (Howarth 1989).

Natural fluctuations in fish recruitment are so great that even in well studied fisheries, oil-induced mortalities of juvenile and pre-recruit fish of less than an order of magnitude would unlikely be detectable (Howarth 1989). Boesh and Rabalais (1987) proposed that experimental studies on effects of oil on pelagic eggs and larvae of commercial species in the field were unlikely to succeed. They recommended that existing data on distribution and abundance of eggs and larvae be mapped and analyzed to identify locations and timing important to stock recruitment. These data combined with oil trajectory models could yield vulnerability models for commercial stocks. Howarth (1991) provided a warning about chronic, low-level oil pollution when he suggested that larval fish could experience sublethal effects with low concentrations of oil in the water column.

Commercial shellfish species more commonly show contamination from oil than finfish species (McIntyre 1982, Teal and Howarth 1984). Much has been written on the effects of oil on shellfish from European spills (McIntyre 1982, Berthou et al. 1987; Dyrynda et al. 1997). Effects range from mortality to flesh tainting, reproductive disruption and compromised immunity lasting up to three months. Possible effects on eggs and larvae are poorly understood and remain speculative.

Strickland and Chasan (1989) speculated that oil penetrating crab nurseries in estuaries could damage stocks. Moreover, the persistence of oil in fine sediment, low-energy sheltered habitats such as estuaries could affect nursery habitat quality for years. Specific effects could be lethal/sublethal impacts on eggs and larvae, damage to nursery habitat quality and bioaccumulation of oil compounds in rapidly-growing animals. Strickland and Chasan (1989) suggested that oysters were somewhat tolerant of oil, but contamination could continue for years with the long-term presence of oil in their sheltered, inshore habitats. After the *Amoco Cadiz* spill in Brittany, oysters readily accumulated oil compounds (Neff et al. 1985) and seven years after the spill they still demonstrated contamination (Berthou et al. 1987).

In summary, oil pollution has a potential negative impact on fisheries in the following ways (McIntyre 1982, Davis et al. 1984, Teal and Howarth 1984, NAS 1985; Hetrick and Daisey 1997):

Interference

- oil-contaminated gear interferes with harvesting;
- temporary closure of oiled areas; and
- long-term closure of areas containing oil production equipment and pipelines.

Tainting

- oil on product or incorporated into tissues (in the absence of more oil, depuration is estimated to occur quickly, but in sheltered habitats shellfish can be chronically contaminated by long-term resident oil); and
- the loss of consumer confidence in seafood quality can induce a marketing disaster extending past the point when stocks are technically no longer tainted.

Effects on Species (extrinsic/intrinsic and lethal/sublethal)

- extrinsic factors include amount and type of oil, duration of exposure (continuous versus intermittent, acute versus chronic), actual oil bioavailability and season; and
- intrinsic factors include type and stage of species' life cycle, species' feeding mode/diet, relative ability to metabolize and detoxify oil compounds and ability to avoid oil (e.g., closing shells or swimming away).

Subsistence fisheries of coastal communities can also be effected. In the first year after the EVOS, 10 Aboriginal communities showed a >70% decrease in traditional food harvesting, and a decrease in traditional sharing practices (Fall and Field 1996). This was attributed to a loss of confidence in the quality of country food that continued to have a measurable impact on harvest rates three years after the spill. Seafood analyses revealed that harvested and processed product from the Gulf of Alaska was safe for consumption in 1989 and 1990 (Saxton et al. 1993).

SEABIRDS

"*…. great variability in seabird numbers and breeding success due to natural factors is a common natural event, making detection of the degree of oil effects difficult if not impossible*." (Wells et al. 1985)

Marine oil pollution is a serious threat to seabirds with many reports of mortalities (NAS 1985; Clark 1987; Dunnet 1987, Hunt 1987, Baker 1991, Berger 1993 a,b; Wells et al. 1995; Ford et al. 1996). Chronic, low-level pollution from ship operations may have a greater effect on bird populations that episodic spills (Camphuysen 1989; Wiens et al. 1996). Wiens (1995) identified the main avenues of spill impacts on seabirds as being on:

- population size and structure;
- reproduction; and
- habitat occupancy and use.

Birds coated with oil are physically impaired to the extent that they can become unable to fly or forage, and often die (Koeth and Vauk-Hentzelt 1988). Oil clogs the feathers, leading to ingestion of oil from attempting to preen, hypothermia stress and drowning from reduced buoyancy. Stress is worsened when oiled birds increase their metabolic rate to counteract decreasing body temperature.

Physiological responses are greatest after seabirds have ingested oil. Most physiological studies have been done in the laboratory. Greatly lowered red blood cell counts (anemia) occur in oiled seabirds (Fry et al. 1986) and this decreases their ability to recover from stress. Reduced body weight and liver damage have also been reported from oiled seabirds (Koeth and Vauk-Hentzelt 1988). Laboratory studies also show that different crude oils have differing toxic effects and responses of birds to oral oil doses and external body oiling vary.

Oil is particularly threatening at locations where seabirds are attracted such as continental shelf and upwelling areas and areas of other ocean processes that concentrate fish and plankton feed (Berger 1993 b). These processes can also concentrate the oil itself. Other danger areas are near rookeries and in gathering sites such as spring-time Arctic open-water leads and polynyas (Alexander et al. 1997). Many authors admit, however, that population-level impacts are difficult to establish given the significant natural interannual fluctuations those seabird populations undergo.

The behaviour of species influences the likelihood of their being oiled. Characteristics which make birds more susceptible to oiling include: spending large periods of time on the water, weak fliers that dive often, species with flightless feather-moulting stages, species that dive to feed and those that roost at night on water (Speich et al. 1991). For example, diving alcids such as auklets and murres are more vulnerable than surface feeders such as fulmars and gulls in offshore spills (Berger 1993 a). These concepts are incorporated into a Bird Oil Index (Speich et al. 1991) which quantifies qualitative attributes for assessing a species' vulnerability and for estimating the importance of particular marine habitats for birds.

The data that best quantify oil spill effects on seabirds are counts of dead oiled birds on beaches (despite complications over estimation of lost carcasses) and live oiled bird counts. Various reviews have addressed lethal and sublethal effects of oil on birds (NAS 1985; Clark 1987; Hunt 1987) and the need of models to evaluate these effects (Weins et al. 1984; Speich et al. 1991). There is no evidence so far that an oil spill has permanently damaged an entire seabird population, although localized populations can be severely damaged (Baker 1991).

Again, the EVOS experience illustrates the controversial nature of post-spill assessments. Spies et al. (1996) suggested a best estimate of 250,000 (range: 100,000 to 645,000) seabirds killed. Mortality estimates varied widely (Ford et al. 1996) as did opinions on long-term, sublethal effects on breeding (Hartung 1995; Wiens 1995; Spies et al. 1996; Wiens et al. 1996). This was exacerbated by incomplete baseline data for comparison with post-spill population status. Although common murres accounted for ~74% of all mortalities, investigators could not differentiate between spill effects and natural marine environmental effects on murre populations (Wiens 1995; Piatt and Anderson 1996). Wiens et al. (1996) concluded from community-level research that effects on marine bird populations diminished rapidly (by 1991) and that early concerns about colony devastation and decadal recovery periods were premature and overstated.

The worst effect of oil on breeding seabirds is nesting failure (Eppley and Rubega 1990). Field studies on oiled seabirds have shown reduction and delay in egg laying, reduced hatching success, reduction in fledgling success and slowed growth rates (Boersma et al. 1988). All these effects have serious implications because time is important to successful nesting in seabirds. Breeding adults oiled at sea can transfer that oil to their eggs. Oil on eggs can cause death of the embryo during the incubation stage (Albers 1980). Oiling can change the breeding behavior of seabirds. Exposed birds can abandon breeding attempts, decrease pair-bonding or neglect nestlings (Eppley and Rubega 1990). Timing of oiling during different stages of incubation can also be important (Eppley and Rubega 1990).

Treating oiled seabirds is costly, yet little is known about the subsequent viability (success) of rehabilitated birds returned to nature. There are few observations of oiled birds returning to breeding colonies (Clark 1987). The survival of different species of treated and released seabirds appears to vary considerably (Clark 1987).

MARINE MAMMALS

Until recently, the impacts of oil on marine mammals were not well understood (Engelhardt 1987; Geraci and St. Aubin 1990). Generally, populations of cetaceans (whales, dolphins) and pinnipeds (seals, sealions, elephant seals) were considered to be relatively unaffected by oil spills (Baker 1991). The major exception was sea otters (*Enhydra lutris*) for which it is well known that death by hypothermia results from oiled fur (Williams et al. 1988, Geraci and St. Aubin 1990). Pinnipeds and cetaceans rely on a fat layer under their skin for insulation, so their body temperature is relatively unaffected by surface oil. Pinnipeds readily become oiled, but there are few observations of oil on free-ranging cetaceans, likely because of their smooth skin (Geraci and St. Aubin 1990). There have been two (speculative) reports associating cetacean deaths with oil (Duguy 1978).

The literature on this field can be divided into pre-EVOS, with its many gaps as reviewed by Geraci and St. Aubin (1990), and post-EVOS. One of the scientific legacies of the EVOS has been the most extensive series of studies on marine mammals ever reported from a spill (Loughlin 1994; Wells et al. 1995). These collectively represent a new era for the field and provide a sound overview on marine mammals and oil. For example, the detailed sea otter observations have been used in oil spill risk assessments for sea otter populations elsewhere (Brody et al. 1996; Ralls et al. 1996).

Extensive EVOS studies on sea otters and harbor seals, and others on Steller sea lions, killer whales and humpback whales covered issues of population impacts, rehabilitation, animal behaviour, pathology and toxicology (Loughlin 1994). The tissue pathology and toxicology samples for sea otters and harbour seals are the most detailed in existence, and essentially form the genesis of their research fields. As well, there were general population and behaviour studies including gray whales, porpoises and dolphins.

Sea otters suffered the most acute effects with ~1,000 confirmed dead (871 beached carcasses and 123 rehabilitation centre mortalities). With limited pre-EVOS data, total mortality estimates ranged widely according to different models used, with the range of 3,500 to 5,500 being the most widely cited (Loughlin 1994; Spies et al. 1996). Oil-related death was implicated in 71% of the necropsies focusing on kidney, liver and lung, but also including gut, muscle, fat, brain and testes histopathology. All oiled carcasses had tissue lesions and lesion occurrence was two to eight times higher in oiled carcasses compared to unoiled carcasses.

Garshelis (1997) reviewed the assumptions behind the mortality estimates from pre- and post-EVOS counts and carcass counts. He focused on the recovery data derived from beach counts of tagged carcasses previously released at sea, and found shortcomings that led to a revised mortality estimate of ~750 (versus ~2,650) for Prince William Sound and a total spill area mortality of ~1,500. Clearly, field methodologies for mortality estimates require improvement.

Chronic, long-term lethal and/or sublethal, impacts were reported from post-spill surveys extending to 1993, at which time populations were speculated to be recovering. Until 1993, sea otter population recovery was suggested as retarded through abnormal mortality patterns, including lowered pup survival rates. Chronic damage was speculated to arise from tissue pathology (especially liver and kidney), continued exposure in oiled areas with oiled prey and spill-related impacts on prey populations. Subtidal clam prey preferred by adults showed no oil contamination whereas the intertidal mussels preferred by juveniles were highly contaminated at least until 1992; thus placing juveniles at risk of long-term contamination from their food. Rehabilitation centre success was limited with 35.8% mortality and up to 21 of 45 radio-tagged released animals may have died. Costs were high and Garshelis (1997) estimated that each otter mortality at a centre represented ~$80,000 US expended. Low success was attributed, in part, to oil toxicity, handling stress and relocation in non-home areas.

For those oiled sea otters that did not die quickly, the following course of events was constructed:

- hypothermic otters reduce feeding while grooming frantically;
- energy stores deplete rapidly and oil may be ingested;
- pulmonary emphysema sets in;
- an acute stress reaction takes place with attendant gastric erosion; and
- profound shock leads to death.

Harbour seal (*Phoca vitulina*) was the other species besides sea otters examined in detail. Pre-EVOS harbour seal estimates revealed a regional population in appreciable decline. Nonetheless, populations were concluded to have undergone significant post-EVOS decline in Prince William Sound. Losses of 135 seals were inferred from reduced counts, not carcasses, at seven oiled haulout sites known from past surveys and over 300 were suggested to be missing because of the spill from the whole Prince William Sound region. Seals did not avoid oil and continued to use oiled haulouts, including for birth and nursing of pups and summer moulting. There was an estimated 25% decrease in pups recruited in 1989 and evidence of oil ingestion while nursing. Noticeable eye damage was recorded among oiled seals. Oiled seals behaved lethargically, this was attributed to brain damage from inhalation of volatile fumes as they breath just above the water surface. This was suggested as being especially threatening with less weathered oil on the calm waters and on haulouts early in the spill. Tissue work revealed that oiled seals commonly had brain lesions. Although seals efficiently metabolize hydrocarbons and most tissue levels are low, high concentrations of aromatic compounds were found in bile over a year after EVOS. Detailed laboratory studies revealed that long-term chronic effects were not traceable. Short-term acute effects may have occurred through inhalation of fumes early on in the spill. Rehabilitation success appeared good with 15 of 18 seals surviving. Stellar sea lions, as with harbor seals, were already in pre-EVOS population decline and no population impacts were attributable to the EVOS.

Killer whales had been last recorded six months before EVOS. Although no carcasses were found, 14 individuals from one pod were not accounted for in a post-EVOS census and two individuals were noted with dorsal fin collapses. Both observations were linked to the spill, for example, Loughlin et al. (1996) stated that "*coincidental evidence supports*" the spill as causing the resident killer whale pod decline. The humpback whale census found no anomalies that could be attributed to the EVOS. Examination of beached gray whale, minke whale and harbour porpoise carcasses yielded no clues as to their deaths. One of 80 Dall's porpoises seen had oil on its back and "*appeared stressed because of laboured breathing*". Overall, cetacean observations were inconclusive.

Much more work needs to be done on the linkage between tissue hydrocarbon levels, oil-related tissue damage and health status. Acute responses of marine mammals are now better understood, but long-term chronic health and population status indicators remain uncertain with a heavy reliance on circumstantial evidence. Field methodologies for mortality estimates require better verification as do the speculative circumstantial linkages between pre- and post-spill field counts of survivors. One thing is certain, marine mammals show no avoidance of oiled areas and will always be at risk in a spill.

The EVOS studies yielded some of the key needs for future oil and marine mammal investigations as follows (Loughlin 1994):

- sound pre-spill baseline census data for reliable population impact assessments;
- wise allocation of research resources between acute and chronic impact studies;
- study design and attendant logistics arrangements on standby;
- standard field sampling protocols for timely, reliable post-spill execution; and

- link the rehabilitation centre better to supporting pathology/toxicology studies.

CULTURAL RESOURCES

The impacts of spills on coastal cultural resources have been largely overlooked in the literature. The EVOS had a relatively large archaeological program (Wooley and Haggarty 1995; Bittner 1996). Wooley and Haggarty (1995) suggested that pre-spill erosion damage to objects and their context minimized spill impacts. Positive outcomes were a doubling of the number of confirmed sites in the study area. Negative outcomes were a few instances of site vandalism, damage by shore treatment crews and the concern that oil contamination could impede the radiocarbon dating process by a large addition of carbon to artifacts (Bittner 1996).

RECOMMENDATIONS

Parks Canada needs to be duly diligent concerning the threat of oil to ocean and estuary shorelines under its marine ecosystem integrity stewardship mandate. Recommendations arising from this review are as follows:

- increase awareness of Service Centre and Field Unit personnel on oil fate and effects including a healthy skepticism about the current state of spill-related science;
- examine ways of combining this awareness with local habitat knowledge into a meaningful Parks Canada support role within the multi-agency (EC/CCG) response framework;
- if the increased awareness reveals important data gaps in the Field Unit's GIS (e.g., inadequate inventory of habitats particularly sensitive to oil), plan to fill these gaps within a Field Unit's Ecosystem Conservation Strategy or Monitoring Plan;
- implement training to enable staff to function as technical environmental support within the EC/CCG REET response framework; and
- ensure that Field Units' spill contingency plans are up to date.

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