THE BIOLOGICAL EFFECTS OF HYDROCARBON EXPLORATION AND PRODUCTION RELATED ACTIVITIES, DISTURBANCES AND WASTES ON MARINE FLORA AND FAUNA OF THE BEAUFORT SEA REGION

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Prepared for

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by

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PREFACE

This report has been prepared as a supporting document to the Environmental Impact Statement for Beaufort Sea Hydrocarbon Production. Its primary purpose is to provide a detailed review of the literature describing various activities, wastes and disturbances which may be associated with future hydrocarbon exploration and production activities in this region. The report also identifies the degree of potential concern associated with various aspects of this proposed development. Specific details regarding quantities of wastes and levels of activity provided in this report are not necessarily consistent with figures shown in the EIS since the latter document reflects the most current estimates of the petroleum industry.

Wayne S. Duval
Project Director

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All sections of this report, particularly statements regarding the degree of concern associated with various resource-activity interactions have been technically reviewed and edited by Wayne Duval (Project Director).

THE BIOLOGICAL EFFECTS OF HYDROCARBON EXPLORATION AND PRODUCTION RELATED ACTIVITIES, DISTURBANCES AND WASTES ON MARINE FLORA AND FAUNA OF THE BEAUFORT SEA REGION

1.0 INTRODUCTION

This report summarizes existing information on the potential biological effects of activities, disturbances and wastes associated with petroleum hydrocarbon exploration and production. Since this is a supporting document to the Dome/Esso/Gulf Environmental Impact Statement for Beaufort Sea Hydrocarbon Production, primary emphasis has been placed on the biological resources of this region as well as the activities, wastes and disturbances which may be associated with this development. Major sections of the report discuss the biological effects of (1) common disturbances, activities and wastes, (2) wastes and disturbances associated with both exploration and production drilling, (3) production, storage and transportation-related sources of disturbance, and (4) environmental emergencies including gas blowouts, crude oil spills or blowouts, and refined fuel spills. A separate section describing the biological effects of chemically dispersed oil has been included within the discussion of environmental emergencies since this cleanup measure, if approved and undertaken, could result in significantly different biological effects than those which may be associated with crude or refined oils alone.

Significant interactions between development components and specific marine biological resources were identified through the use of a matrix (Figure 1-1). The level of available information describing the nature of these interactions is highly variable, and the biological effects of some wastes and sources of disturbances such as drilling fluids, aircraft and dredging are reasonably well documented. On the other hand, the effects of some materials such as cement slurries, completion fluids and water/glycol blowout-preventer control fluid are less extensively discussed in the literature. In addition, there is considerable difference in the level of existing information for some biological communities, with the biological effects of many activities and wastes on micro-organisms and epontic communities being virtually unknown.

As a supporting document to Volume 4 of the Beaufort Production EIS, this report also identifies the degree of potential concern (negligible to major) which may be associated with each of the interactions shown in Figure 1-1. The following definitions were used during this assessment:

A Major Concern exists when a regional population or species may be affected to a sufficient degree to cause a decline in abundance and/or a change in distribution beyond which natural recruitment (reproduction and immigration from unaffected areas) would not likely return that regional population or species, or any population or species dependent upon it, to its former level within several generations.

A Moderate Concern exists when a portion of a regional population may be affected to a sufficient degree to result in a change in abundance and/or distribution over more than one generation of that portion of the population or any population dependent upon it, but is unlikely to affect the integrity of any regional population as a whole.

A Minor Concern exists when a specific group of individuals of a population at a localized area and/or over a short time period (one generation or less) may be affected, but other trophic levels are not likely to be affected in a manner which is considered regionally significant, or the integrity of the population itself is not significantly affected.

A Negligible Concern exists when the degree of the anticipated biological effects are expected to be less than minor.

The region is defined as the Canadian portion of the Beaufort Sea south of the permanent polar ice pack, including western Amundsen Gulf (to 123°45'W longitude) and all coastal marine and estuarine habitats adjacent to these waterbodies.

While the EIS focuses greatest attention on the most significant biological concerns, mitigation of potential impacts as well as potential cumulative impacts of the proposed development scenario, this report discusses all interactions to the extent that is possible with the available literature. However, a more detailed treatment of the observed effects of oil spills is provided in another supporting document titled "A prospectus on the biological effects of oil spills in marine environments" (Duval et al. 1981).

Figure I-1 Interaction matrix of petroleum hydrocarbon exploration and production related activities, wastes and disturbances and marine biological resources of the Beaufort Sea

ACTIVITY, WASTE OR SOURCE OF DISTURBANCE	Marine Mammals	Birds	Fish	Bacteria	Phytoplankton	Zooplankton	Benthic Communities	Epontic Communities
COMMON DISTURBANCES & WASTES								
Presence of Artificial Structures	●	●	●				●	
Human Presence	●	●	●					
Icebreaking	●	●	●		●	●	●	●
Dredging	●	●	●	●	●	●	●	●
Sewage	●	●	●	●	●	●	●	●
Solid Wastes	●	●	●	●			●	
Underwater Sound	●	●	●					
Aircraft /Other Surface Noise	●	●						
Engine Exhausts /Air Emissions	●	●						
NORMAL DRILLING OPERATIONS								
Drilling and Completion Fluids	●	●	●	●	●	●	●	●
Formation Cuttings	●	●	●	●	●	●	●	
Produced Water	●	●	●	●	●	●	●	●
Water /Glycol BOP Fluid	●	●	●	●	●	●	●	
Underwater Shock	●	●	●					
PRODUCTION, STORAGE, TRANSPORT								
Gas Flares	●	●						
Heated Water	●	●	●	●	●	●	●	●
Ballast Water/ Exotic Organisms			●	●	●	●	●	●
ENVIRONMENTAL EMERGENCIES								
Natural Gas	●		●	●	●	●	●	●
Crude Oil	●	●	●	●	●	●	●	●
Refined Fuels	●	●	●	●	●	●	●	●
Chemically Dispersed Oil	●	●	●	●	●	●	●	●

2.0 COMMON WASTES AND DISTURBANCES

2.1 PRESENCE OF ARTIFICIAL STRUCTURES

2.1.1 Introduction

The physical presence of an artificial offshore structure may have at least four general effects on local environments: 1) loss of a small portion of the existing bottom habitat; 2) creation of a small area of hard substrate, a type of underwater habitat that is uncommon in the Beaufort Sea; 3) alteration of local oceanographic conditions, and 4) potential effects of the superstructures of some facilities on birds and mammals. These effects would vary in magnitude according to the relative size, mobility and permanence of each structure, and would be most significant for relatively large structures such as a production island or Arctic Production and Loading Atoll (APLA) which may exist for 20-30 years. The majority of artificial structures proposed for the Beaufort Sea will be caissons built of steel or reinforced concrete and placed on subsea berms of dredged sand. The following discussion focuses primarily on these large, relatively immobile structures including artificial islands, causeways and APLA's. Drillships and other mobile vessels are not discussed in this section since they do not result in habitat loss, and the organisms which colonize their surfaces are frequently removed.

2.1.2 Types of Effects

In general, the presence of artificial structures causes a localized increase in the diversity and abundance of marine biota on and around the structure, particularly where the adjacent seabottom topography is consistently flat (Carlisle et al. 1964; Turner et al. 1969; Gallaway et al. 1979). To date, there have been no published studies on the biological effects of artificial structures in Arctic waters, although such an investigation was initiated by Dome Petroleum at the Tarsiut exploration island in 1981 (R. Hoos, Dome Petroleum Ltd. pers. comm.). Underwater structures usually attract fish because they create areas of reduced currents, refuge from predators, and enhanced food production (Tarbox and Spight 1979), and may also elicit a general thigmotropic reaction (attraction to objects) in some species (Breder and Nigrelli 1938). Subsea water intake screens may attract and subsequently result in the impingement of some species of mobile benthic invertebrates, while steel, rock or concrete structures can provide habitat for the attachment of hard-substrate colonizers such as kelp, barnacles and tunicates (Chin et al. 1979). The presence of offshore artificial structures in the Beaufort Sea could subsequently attract mobile benthos and atypical fish species, and lead to the establishment of "island communities" which are different from the communities in surrounding soft-bottom areas. These communities could be similar to those associated with the boulder patches off Prudhoe Bay.

2.1.2.1 Habitat Loss

The presence of permanent offshore structures will result in the complete loss of benthic communities beneath subsea berms, although the areas affected would represent a very small portion of the available offshore benthic habitat. Offshore artifical islands constructed to date have covered between 0.005 km^2 (Adgo) and 0.2 km^2 (Isserk) of benthic habitat, depending on water depth and construction method. Future APLA's constructed in water depths up to 60 m could each occupy as much as 1.8 km^2 of sea bottom (EIS Volume 2). Nevertheless, this will represent a very small percentage of the total "marine zone" habitat (depths of 30 to 200 m; Wacasey 1975) in the southeast Beaufort Sea. Other effects of artificial island construction associated with dredging are discussed in Section 2.4.

2.1.2.2 Artificial Substrate Effects

The construction of several artificial structures in the Beaufort Sea may cause significant but localized changes in the composition of biological communities due to the rarity of natural hard substrates in this region. The most extensive area of natural hard substrate in the Canadian Beaufort is along the west coast of Banks Island.

The potential form and sequence of artificial substrate colonization in the Beaufort Sea cannot be accurately predicted but may be estimated from available information on the biology and occurrence of benthos in this region, in conjunction with the results of studies of hard substrate colonization in temperate and tropical waters. However, the results of investigations initiated at Tarsiut during 1981 will provide more site-specific information on recolonization processes. The pioneer stages of colonization usually begin with the accumulation of bacteria and benthic diatoms on the barren surfaces of recently-immersed structures (O'Neill and Wilcox 1971). This is followed by settlement of algal spores and invertebrate larvae, and their subsequent growth and proliferation. The later stages of colonization involve the immigration of motile fauna which become residents and the eventual attraction of visiting pelagic species.

Several factors will likely influence the degree and rate of colonization of artificial substrates in the Beaufort Sea. The species and number of invertebrates which could colonize these substrates would depend on the number of species having planktonic larval stages (meroplankton), the abundance of meroplankton and the growth rates of colonizing species. Limiting factors which could subsequently affect the growth and survival of species which colonize offshore structures in the Beaufort Sea may include ice formation and scouring of substrates, particularly within 3 to 4 m of the surface, and possible adverse effects associated with the release of heat, oil and other potentially toxic substances from or near the artificial structures.

Annual growth rates of arctic invertebrates are typically low (Dunbar 1968), and as a result, establishment of communities on artificial structures in the Beaufort Sea may be considerably slower than that documented for structures in temperate or tropic latitudes. Nevertheless, recent observations of a blowout preventer stack in the Beaufort Sea which was located at a water depth of 60 m for approximately 1 year indicated that colonizing fauna (hydroids, tube worms and tunicates) were already more diverse and abundant than in adjacent undisturbed soft substrate habitats (Can Dive Ltd. videotape). The slow growth rate of arctic flora and fauna may be increased if the structure itself or the surrounding water is artificially heated (Section 4.2), or if any exotic organisms present in ballast water released from tankers are able to survive and colonize the structure (Section 4.3). On the other hand, concurrent release of drilling muds and formation cuttings could reduce the growth of some less motile colonizing species (Section 3.1.8). However, there is limited information regarding the nature of these primary and synergistic effects in Arctic waters, or the physical conditions that may prevail around such artificial structures. As a result, the types and rates of colonization on artificial structures in the Beaufort Sea remain largely unknown.

2.1.2.3 Altered Oceanographic Regimes

The presence of an artificial structure may cause local changes in existing currents, wave action against the structure, ice rubble fields, areas of thin ice and open water in the lee of a structure, and altered sediment deposition patterns. These effects would only be expected to occur in a very small portion of the Beaufort Sea, and would tend to diversify the existing offshore oceanographic environment. Regionally significant alterations in oceanographic patterns due to artificial structures are considered unlikely, although concern has been raised by local hunters and trappers that artificial islands built beyond the outer edge of the landfast ice may cause an extension of this zone and delay breakup.

A review of ERTS/Landsat satellite photographs taken each spring since 1973 has indicated that the greatest normal variation in the extent of shorefast ice occurs off Mackenzie Bay, and since this date, the location of the landfast ice edge has fluctuated over a distance of 70 km (Joint Tuk-Industry Task Force 1982). In 1973, the ice edge was located at about the 20 m water depth contour. During the other years, the sheltered nature of Mackenzie Bay has resulted in an extension of the ice edge northwards into the deeper parts of the Bay. The least annual variability in the extent of the landfast ice is found northwest of Herschel Island where the edge is typically located about 20 km from the shore. An exception to this trend was observed in 1975, when unusually cold and severe conditions in this part of the Beaufort Sea caused the ice edge to be almost twice this distance from shore. Monitoring programs conducted near two offshore artificial islands in the Beaufort Sea (Issungnak and Kaglulik) indicate that these structures did not influence the location of the landfast ice edge (Joint Tuk-Industry Task Force 1982).

Artificial islands within the landfast ice zone create rubble fields as a result of the relatively slow movement (usually less than 10 m over one winter) of the ice sheet against islands (EIS Volume 2). Islands constructed in the transition zone where the ice is moving at a greater rate (3 to 13 km/day) would likely cause the formation of ice rubble fields in a roughly tear-shaped pattern downstream of the island as ice is deflected by the structures, although this shape may differ as the rate and direction of ice movement in this zone change throughout the winter (EIS Volume 2). However, neither of these ice anomalies is likely to significantly delay the timing or extent of ice breakup in the Beaufort Sea, which are naturally variable (Burns 1974) and depend on the annual variability in winds, temperature and the discharge of the Mackenzie River.

2.1.3 Effects of Artificial Structures on Marine Mammals

The biological effects of the physical presence of artificial structures on marine and marine-associated mammals would probably be largely indistinguishable from effects associated with composite industrial activities in both offshore and terrestrial areas. Although terrestrial mammals may be directly affected by artificial structures through habitat loss, the relatively limited size of these structures in relation to available habitat would make this loss insignificant, provided that structures were not located in critical habitats.

Artificial structures in offshore areas which may affect marine mammals include shallow and deep water production islands, Arctic Production and Loading Atolls (APLA's) and temporary exploration islands. However, noise generated at these structures will probably have greater effects on marine mammals than the actual physical presence of exploration and production facilities (Sections 2.6.5 and 2.7.2). To date, eighteen exploration islands have been constructed off the Mackenzie Delta without apparent adverse effects on marine mammal populations. Short-term disturbance of white whales occurred during the construction phase of some islands, but this disturbance was believed to be related to the supporting logistics traffic. On the other hand, white whales were frequently observed within 100 m of the Netserk South artificial island during a period when the island was operational in 1975. Kannerk artificial island is situated within the travel corridor of white whales along the Tuktoyaktuk Peninsula, and has had no detectable effect on the movement of whales in this area (Fraker 1977a).

Similarly, large numbers of bowhead whales have been observed in the vicinity of artificial island (Issungnak O-61) construction activities in the southeastern Beaufort Sea, and these individuals were apparently not disturbed by the presence of the artificial structure and associated noise (Section 2.6.5) (Fraker et al. 1981). One bowhead came within 16 m of the barge camp, and a total of 64 whales were observed within 10 km of Issungnak during 5 days of survey effort from August 5 to 22, 1980 (Fraker et al. 1981).

Artificial structures within the 100 m isobath could result in a very localized reduction in the food supply of bearded seals due to loss of habitat for benthic infauna (Section 2.1.6). However, these indirect effects would undoubtedly be negligible in view of the extremely limited area affected. Since ringed seals feed primarily on pelagic invertebrates and arctic cod, artificial structures would not have similar indirect effects on this species.

The greatest potential concern related to artificial structures is the attraction of polar bears and Arctic foxes to offshore sites during periods of ice cover and to onshore sites year-round. This attraction has been documented during existing operations in the Beaufort, and increased utilization of terrestrial, and landfast and transition zone ice areas for hydrocarbon production-related activities would almost certainly increase the frequency of encounters with these species. During operations at Tarsiut from November to April 1982, industrial personnel sighted numerous Arctic foxes and a total of 23 polar bears at distances ranging from 0.6 to 19 km of the island (Ward 1982). The potential attraction of these species is the subject of further discussion in relation to human presence (Section 2.2.2).

2.1.4 Effects of Artificial Structures on Birds

As with marine and marine-associated mammals, the biological effects of the physical presence of artificial structures on Arctic birds will also be largely indistinguishable from the effects of composite industrial activities (e.g. airborne noise, aircraft, human presence, etc.). However, some collisions of birds with offshore platforms and ships may be attributable to the actual physical presence of these structures. Unenclosed gas flares on production islands may also attract some birds during periods of low visibility (Section 4.1.3), resulting in collisions with permanent facilities. In the southeastern Beaufort Sea region, the birds most likely to collide with offshore structures include loons, eiders, oldsquaws, thick-billed murres and black guillemots since these species migrate offshore at low altitudes and are relatively unmaneuverable.

Migrant birds could collide with man-made structures during periods of low light and foggy weather (cf. Weir 1976; Avery et al. 1978). The potential for collisions is likely to be greatest during fog, rain or snow, since birds are often attracted to lighted structures under poor visibility conditions (Weir 1976; Avery et al. 1978). Avery et al. (1977) reported that on overcast nights, more nocturnal migrants were seen near a 366 m navigational tower in South Dakota than at a site 305 m away, while the opposite trend was observed on clear nights. This investigation suggested that migrant birds avoided the tower under good visibility conditions.

Bourne et al. (1979) report that some marine birds linger around offshore structures in the North Sea, and this increases the probability for contact with oil or other industrial discharges normally or accidentally associated with these facilities. In the Beaufort Sea region, birds most likely to linger near artificial structures may include gulls, jaegers and terns since these species typically forage offshore during the open water period and are attracted to sites of human activity elsewhere in the world.

2.1.5 Effects of Artificial Structures on Fish

Artificial structures in aquatic environments attract most age classes of a variety of pelagic and mid-water fish species. There is a relatively extensive literature describing colonization of subsurface artificial structures (reefs) constructed of materials ranging from rip-rap and automobile tires to modular cement structures. These installations have encouraged fish colonization in otherwise 'barren' environments or have further increased the carrying capacity of already productive areas such as estuaries. Artificial structures may provide refuge from predators (Wickham and Russel 1974) or habitat which allows establishment of various cryptic fauna and epifauna, and this in turn may attract juvenile fish. Hunter and Mitchell (1967) suggest that submerged artificial structures may increase the schooling behaviour of some species by acting as a visual stimulus. In a similar manner, Klima and Wickham (1971) report that artificial habitats function as a spatial reference which aides in the orientation of pelagic species.

Information describing the effects of artificial structures on fish in Arctic marine environments is largely in the form of anecdotal observations. Tarbox and Thorne (1979) reported that resident marine species were attracted to a simple PVC structure placed beneath the ice, while Tarbox and Spight (1979) observed Arctic cod in association with artificial habitats in Prudhoe Bay. Olmsted (1977a) reported a significantly greater abundance of epifaunal invertebrates in the lee of an artificial island (Arnak L-30) in the Beaufort Sea, and since many species in this region feed almost exclusively on epifauna (isopods, mysids, cumaceans, amphipods), some fish may be attracted to artificial structures which support higher standing stocks of epifauna than adjacent undisturbed areas.

Investigations conducted at subsurface structures in temperate and tropical latitudes suggest that fish utilization of environments created by artificial structures is both more rapid (Wickham and Russell 1974) and intensive (Russell 1975) than adjacent natural habitats. Immigration to artificial structures tends to be immediate (Klima and Wickham 1971), but rapidly stabilizes as the carrying capacity of the substrate and adjacent waters is reached (Fast and Pagan 1974). For example, Wickham and Russell (1974) observed colonization of an artificial structure by pelagic fishes the day following its placement. Although both pelagic and demersal species utilize artificial habitats, the relative abundance of colonizing surface fishes invariably exceeds that of bottom fishes (Klima and Wickham 1971; Chihiro and Takayoshi 1978).

Several studies have been directed at examination of the characteristics of underwater artificial structures which lead to attraction of fish. For example, Wickham and Russell (1974) were unable to significantly increase the rate or number of fish attracted to an artificial structure either by increasing its size or painting it with a fluorescent colour.

However, Wickham et al. (1973) found that larger schools of bait fishes were attracted to multiple structures. This investigation also indicated that artificial structures attracted more fish than comparable control areas, irrespective of the depth of the structures, although the greatest attraction to underwater structures was observed at a water depth of 26 m. Wickham and Russell (1974) subsequently demonstrated that larger numbers of fish were attracted to offshore than to inshore artificial habitats. They also noted that drifting structures (i.e. boats, barges, floating platforms) were accompanied by fish even when towed. Russell (1975) suggests that wave action is the singlemost important factor affecting colonization. Severe wave action and resultant substrate scour limit biotic succession, and affect colonization through the transport of mobile fauna and invertebrates.

Available data suggest that pelagic and mid-water fishes will be attracted to production platforms, vessels and barges, and other subsurface structures in the Beaufort Sea. Although attraction of fish to structures (or underwater sound) may not be directly harmful, congregations of fish are more likely to be exposed to other wastes and sources of disturbance (e.g. heated water, drilling wastes) than naturally distributed stocks in Arctic regions.

2.1.6 Effects of Artificial Structures on Benthic Communities

There is an extensive data base describing benthic recolonization in marine areas of the world other than the Arctic, and although different species are involved, the basic ecological requirements of marine flora and fauna are often common. In California, Carlisle et al. (1964) described colonization of submerged streetcars by barnacles (mostly *Balanus tintinnabulum*) within a few days after their placement in an artificial reef, while tunicates appeared within 6 months. Other common encrusting invertebrates on offshore oil production facilities have included scallops, mussels, hydroids, sea anemones, sponges and tube worms. Large macrophytic algae have also been shown to colonize solid surfaces.

The effects of artificial structures on benthic flora and fauna have been a subject of limited investigation in the Beaufort Sea region, although the following observations provide some basis for assessment of possible effects. Thomas et al. (1980) reported that barnacles and attached algae colonized oceanographic instruments moored in Tuktoyaktuk Harbour within several weeks of installation, while Slaney (1973) noted colonization of filamentous green algae on submerged cobble and driftwood in shallow waters off the Mackenzie Delta. The best documented example of a hard-bottom benthic community in the Beaufort Sea region is the boulder patches near Prudhoe Bay. Divers removed encrusting flora and fauna from 0.05 m^2 plots on 14 boulders between August and May to observe the recolonization process (Dunton and Schonberg 1980). Hydroids were the only organisms to tolerate the heavy siltation on the denuded surfaces and many plots were not recolonized at all during the period of study. However, as indicated earlier (Section 2.1.2.2), diverse and extensive recolonization of a blowout preventer stack has also been documented in the Beaufort Sea region.

Generally, after the initial establishment of a bacterial film on a newly submerged surface, subsequent stages of hard-substrate colonization involve various members of the benthic community. In the Chukchi-Beaufort Sea region, the larval component of the plankton is apparently dominated by barnacle larvae (Johnson 1956; Horner 1978). Other relatively dominant groups include meroplanktonic stages of hydrozoans (Carey 1978), bivalves, gastropods, annelids, echinoderms, and decapods, while relatively few larvae of ascidians and bryozoans are present in the planktonic community (Johnson 1958). Annual variability in the abundance and composition of meroplankton in the Chukchi Sea was described by Johnson (1956). The abundance of larvae in the eastern and southern Beaufort Sea was found to be low relative to that in the Chukchi Sea and western coastal Beaufort Sea (Johnson 1956; Grainger 1965). More recently, Horner (1978) has reported that larvae of barnacles, polychaetes and echinoderms were more abundant in the Chukchi Sea than in the Beaufort Sea. The presence of some meroplanktonic forms in the southeast Beaufort Sea indicates that submerged concrete and steel caissons would be colonized to a certain extent. Consequently, although there is a clear potential for and some documentation of colonization of hard artificial substrates in the offshore Beaufort, it is not possible to predict the rate of colonization or the initial community composition without more detailed information regarding the meroplankton in the region.

Artificial structures may also affect benthic organisms by creating physical obstructions to the local wind, waves, ice and currents. Production and exploration islands will likely provide sheltered and exposed environments similar to those associated with natural barrier islands and spits in the region. Windward and leeward habitats exist primarily during the open water season, and the limited available data suggest that densities of mobile invertebrates such as amphipods and euphausiids could be much higher in the lee of artificial islands (Olmsted 1977b). These alterations in oceanographic regimes would increase the diversity of offshore habitats and as such, will probably represent positive but localized effects of artificial structures.

The elimination of natural soft-bottom habitat by an artificial island would likely affect benthic organisms to varying degrees, depending on the depth zones where exploration and production facilities are constructed. Wacasey (1975) characterized benthic faunal associations off the Mackenzie Delta according to water depth. Using his data for average benthic biomass in different depth zones, an artificial island constructed in 2 m of water could result in the direct loss of approximately 2400 kg of benthic organisms, whereas the same size island in 60 m of water would require a larger base and could eliminate 14,000 kg of benthic biomass (Section 2.4.8). However, the amount of benthic habitat affected would also depend on the design of artificial drilling platforms. At a given water depth, caisson-retained structures require a smaller base (and resultant habitat loss) than sacrificial beach designs since the former do not reach the surface where erosion takes place, and therefore can have steeper slopes. Benthic recolonization of sacrificial beach islands and underwater slopes resulting from dredging programs are also discussed in Section 2.4.8.

2.1.7 Summary of Concerns Related to Artificial Structures

Artificial structures in coastal and offshore waters of the Beaufort Sea may result in both positive and negative impacts on marine resources. Shelter produced by production platforms and hard surfaces associated with various types of underwater facilities may increase the abundance and diversity of some communities, particularly benthic epifauna and pelagic fish species. On the other hand, construction of production islands will result in loss of benthic infauna and associated habitat, and during operation, these facilities may attract some species of birds and mammals. The most significant concern regarding artificial structures in offshore waters is the possible attraction of polar bears and the element of human safety (Section 2.1.3), although collisions of birds with offshore facilities and the loss of habitat for some benthic organisms also may be locally significant. As indicated in Table 2.1-1, most concerns associated with artificial structures are considered NEGLIGIBLE from a regional perspective. The degree of regional concern regarding attraction of polar bears, Arctic foxes and some species of migrant birds is greater, and considered MINOR according to the criteria previously described in Section 1.

TABLE 2.1-1

SUMMARY OF POTENTIAL CONCERNS RELATED TO THE PRESENCE OF ARTIFICIAL STRUCTURES IN THE BEAUFORT SEA REGION

Environmental Component or Resource	Potential or Probable Effect	Degree of Potential Regional Concern
Currents and Sediment Deposition	Creation of exposed and sheltered environments	See individual resources
Ice Environment	Creation of local rubble fields	See individual resources
Terrestrial Mammals	Local habitat loss	NEGLIGIBLE
White Whale, Bowhead Whale, Ringed Seal	Disturbance	NEGLIGIBLE
Bearded Seal	Localized loss of infaunal food resources; disturbance	NEGLIGIBLE
Polar Bear	Attraction to shorebased or offshore structures	MINOR
Arctic Fox	Attraction to shorebased or offshore structures	MINOR
Migrant Birds	Attraction to and possible collision with offshore structures, particularly loons, eiders, oldsquaws, thick-billed murres and black guillemots	MINOR
Fish (primarily pelagic species)	Attraction to subsurface structures and habitats created by offshore platforms	NEGLIGIBLE
Benthic Infauna	Mortality and temporary habitat loss at artificial island sites	NEGLIGIBLE
Benthic Epifauna	Colonization of hard surfaces, increasing diversity and abundance relative to adjacent soft-sediment habitats. Utilization of sheltered habitats in the lee of artificial islands.	NEGLIGIBLE

2.2 HUMAN PRESENCE

2.2.1 Introduction

The numbers of personnel at offshore exploration sites and shorebases will substantially increase with continued exploration and the eventual production of oil and gas reserves in the southern Beaufort Sea. Personnel may be located on a variety of offshore facilities including production platforms, Arctic Production and Loading Atolls (APLA's), drillships, support vessels and barge camps, as well as at shorebased facilities such as service centres, harbours and borrow sites. It has been estimated that the total work force (including off-shift personnel) could range from 16,400 in 1990 to 23,600 by the year 2000 (EIS Volume 2). In addition, an estimated 45,000 non-project personnel could move into the region as part of the supporting infrastructure at shorebased facilities and administrative centres. Access of both project and non-project personnel to previously inaccessible inland areas and sites along the southern Beaufort Sea coast could increase substantially.

Additional or existing personnel located on offshore facilities are not likely to create environmental concerns that are distinguishable from other types of disturbance, including the physical presence of artificial islands and drillships, and the potential effects associated with the use of helicopters, fixed-wing aircraft and various support vessels (see Sections 2.1 and 2.7). Personnel located offshore would have little or no contact with fish or wildlife, and their off-shift activities are presently and will continue to be regulated by the petroleum industry.

2.2.2 Effects of Human Presence on Mammals

The biological effects of human presence on arctic marine and marine-associated mammals will probably be indistinguishable from the effects of composite offshore activities associated with hydrocarbon exploration and production, while the presence of industry personnel at shorebased facilities may have potential adverse effects on some terrestrial mammal populations. The only marine-associated mammals which occur in offshore areas and are likely to detect humans are polar bears and Arctic foxes foraging on the landfast and transition zone ice during winter and spring (Stirling et al. 1981a). In general, polar bears are not afraid of humans (Schweinsburg and Stirling 1976, cited in Polar Gas 1977), and past encounters between bears and people have often resulted in serious injury or death of both (Perry 1966; Jonkel 1970, both cited in Polar Gas 1977). For this reason, a polar bear monitoring program is presently maintained to alert industry personnel of bears in the vicinity of both shorebases and offshore operations. Although mitigative measures include sedation and removal of live bears, some individuals may have to be destroyed for reasons of human safety. The latter are included in the community quotas administered by the N.W.T. government.

Mammals which may be affected by human presence in terrestrial areas include polar and grizzly bears, caribou, reindeer, Arctic and red foxes, timber wolves, muskrats and other small furbearers. These species may be subject to potential adverse effects which result from increased hunting pressure, habitat loss, and harrassment. The significance of these potential sources of disturbance is not known, but would probably be negligible when recreational hunting by industry personnel at base camps is controlled, access to biologically sensitive areas is restricted, camp/settlement garbage is disposed of in an acceptable manner, and access of wildlife to food storage areas is prevented. However, as indicated earlier, increased numbers of non-industry personnel would also enter the region as development of oil and gas reserves proceeds. It will be more difficult to regulate the recreational hunting activities of this segment of the population, and overharvest of some mammal species could occur unless government regulatory agencies establish appropriate guidelines and restrictions.

The only marine mammal species which occurs in nearshore areas of the Beaufort Sea in substantial numbers is the white whale. Approximately 7000 whales concentrate in the Mackenzie River estuary each July, probably for calving and for calf rearing (Fraker and Fraker 1979). Populations of white whales from other regions (e.g. Churchill, St. Lawrence River) have been successfully observed from terrestrial vantage points without apparent disturbance (Sergeant and Brodie 1975), and there is little reason to expect that this form of human presence would have different effects on the Mackenzie Estuary population. However, the latter population is subject to intense human disturbance associated with the annual harvest by the Inuit during its residence in the Estuary. Short-term changes in the distribution of the whales result during the hunt, and there is also evidence that the concentration areas in the Estuary are vacated earlier than usual after repeated chases (Fraker 1978; Fraker and Fraker 1979, 1981). The noise and disturbance associated with this hunt undoubtedly exceeds disturbances of humans per se, and probably also exceeds disturbances associated with the present level of industrial logistics traffic in the Estuary.

2.2.3 Effects of Human Presence on Birds

The effects of human presence on bird populations of the southeastern Beaufort Sea region are a relatively important area of potential concern with respect to hydrocarbon exploration and production. Species-specific sensitivity, type and duration of disturbance, and stage of the life cycle are all factors which would determine the degree of potential concern. The effects of human presence in offshore areas would be largely indistinguishable from the effects of other activities and facilities. However, shorebased industry personnel and non-industry people may have adverse effects on some bird populations in terrestrial and nearshore areas.

Nesting and incubating birds are probably most vulnerable to human disturbance, although staging migrants may also be affected to a certain degree. Studies conducted in several areas have shown that human disturbance reduced nesting success and/or survival of young Canada geese (MacInnes and Misra 1972), Arctic and red-throated loons (Davis 1972; Booth 1978), common eiders (Cooch 1965), herring gulls (Hunt 1972), glaucous-winged gulls (Gillett et al. 1975), western gulls (Robert and Ralph 1975), American robins, red-winged blackbirds, eastern bluebirds and mourning doves (Bart 1977). However, the reasons for reduced reproductive success varied between these species. The following discussion summarizes the results of studies on the effects of human disturbance on bird species that are present in the Beaufort Sea region, and describes the possible manners through which such disturbances may affect nesting birds. The susceptibility of selected groups of birds in the Mackenzie Delta region to camps and men on foot are summarized in Table 2.2-1, and were determined following 3 years of unstructured observations in this region (Slaney 1974a).

Gollop et al. (1974a) examined the effects of human activity on incubation and reproductive success of three colonial species at Nunaluk Spit. The results of this study indicated that incubating black brant left their nests when observers approached within 30 m, and would not return while people remained in the area. Incubating common eiders flushed when approached by ground observers at distances ranging from 1 to 6 m, but returned within 35 minutes if the observer had left. Similar disturbance responses by common eiders were observed in the Cape Dorset area by Cooch (1965), who also reported that frequent flushing of incubating common eiders resulted in above average egg mortality. All incubating glaucous gulls took flight when the ground observer moved to within 30 m of the nesting colony, but returned to their nests within 10 min when the observer moved to a distance of 45 m and remained still (Gollop et al. 1974a). Observers walking to a blind positioned 20 m from the colony also disturbed the incubating gulls for a period of 10 min. In addition, Gollop et al. (1974a) reported that Arctic terns were highly sensitive to the placement of a blind near the colony after commencement of nesting, but the authors did not examine the sensitivity of this species to observers.

Gollop et al. (1974b) later investigated the effects of human disturbances associated with a 6 to 20 man camp (9 tents, 2.5 hp generator) on breeding densities and breeding success of birds on the Yukon North Slope in 1972, although only the Lapland longspur was present in sufficient numbers to permit meaningful comparisons. There were no statistically significant differences in population density between control and human disturbance sites, but the reproductive success of disturbed birds may have been reduced since it was suggested that nesting densities were lower during the year after disturbance by human activity (Gunn et al. 1974).

TABLE 2.2-1

SUSCEPTIBILITY OF SELECTED AVIAN GROUPS TO HUMAN DISTURBANCES
(adapted from Owens 1977)

Avian Group	Susceptibility to Disturbance	Estimated Distances (m) at which Disturbance Ceased to Affect Birds*	
		Camps	Men on Foot
Loons	low to moderate		
	- during migration	200	200
	- at nesting site	800	200
	- general	400	200
Whistling Swans	moderate to high		
	- during migration	200	800
	- at nesting site	600	1200
	- general	200	800
Geese	high**		
	- during migration	400	400
	- at nesting site	1600	200
	- general	800	400
Ducks	low to moderate		
	- during migration	200	100
	- at nesting site	400	100
	- general	200	100
Shorebirds	low		
	- during migration	<50	<50
	- at nesting site	<50	<50
	- general	<50	<50
Passerines	less than ten meters in all circumstances at all seasons		

* based on 3 years of unstructured observations in the Mackenzie Delta (Slaney 1974a)

** susceptibility to men on foot: black brant - low
snow goose - high
Canada goose - moderate
White-fronted goose - moderate

There are several ways in which human presence may have potential adverse effects on the reproductive success of incubating or nesting birds. Northern geese and ducks experience a critical period of energy demand during incubation, and must rely on fat reserves to sustain themselves for several weeks during this period (Godfrey 1966). Additional energy demands required to respond to repeated disturbances (e.g. flush) could result in mortality of weakened birds or cause them to abandon the clutch and nest; both effects would lead to egg mortality. In addition, the temporary absence of a female from the nest would increase the susceptibility of the eggs to predation. MacInnes and Misra (1972) reported that some predators such as jaegers and gulls learn to follow humans and will prey on eggs of nesting arctic birds flushed by human presence. Disturbance of nesting gulls, however, may result in chick mortality due to conspecific strife. For example, Gillett et al. (1975) and Robert and Ralph (1975) reported an increase in chick mortality at gull colonies disturbed by humans. Mortality resulted when chicks moved from home territories into adjacent territories in response to the disturbance and were subsequently attacked by neighbouring adults.

Windsor (1977) studied the effects of human presence on nesting peregrine falcons in the Campbell Hills. In some instances, observers were able to walk to the base of the nest cliff without causing overtly stressed behaviour, while in other cases, birds exhibited stressed behaviour when the observer approached to within 1500 m of the cliff. The peregrines were more susceptible to disturbance during the nesting stage than during the incubation period. Platt and Tull (1977) reported that gyrfalcons became habituated to a blind 300 m from nest sites during the egg-laying and incubation periods, while Nelson (1978) reported that two gyrfalcons near fledging died as a result of several days intensive geological sampling near the nest. The adults successfully fledged young from the nest during the following year when there was no disturbance. Fyfe (1969) reported that the activities of trappers in early May may have caused the desertion of four gyrfalcon nests along the Anderson River.

Although the effects of human presence per se on staging and migrant waterfowl have not been documented, several authors have examined the sensitivity of geese to airborne noise disturbances associated with industrial activity (Section 2.7.3). Staging snow geese and white-fronted geese were particularly susceptible to airborne noise and would probably be adversely affected by human disturbance when concentrated on the staging grounds. Potential disturbance responses (e.g. flushing and/or relocation to less favourable areas) may place additional energy demands on geese during the period when they accumulate fat reserves for several weeks prior to their long migrations. However, the ecological significance of this and other effects of disturbance on these species remain poorly documented.

2.2.4 Effects of Human Presence on Fish

Relatively large human populations associated with industrial projects in previously unsettled areas can result in over-exploitation of easily accessible local fish stocks. Some species could be vulnerable to human exploitation when and where they are concentrated in small watercourses during migration, spawning or overwintering. In addition, areas where recruitment from adjacent populations is restricted, or where growth and reproductive rates are low (e.g. arctic lakes; Dunbar 1973) may also be susceptible to overfishing. However, in the coastal and offshore marine habitats in the Beaufort Sea region, fish populations are well dispersed and mobile, and virtually no recreational fishing occurs in this region. Consequently, no significant effects of human presence associated with future offshore development would be expected. Over-exploitation of inland fishery resources by project and non-project personnel would also likely be limited to the extent possible by the industry and government regulatory agencies, respectively.

2.2.5 Summary of Concerns Related to Human Presence

As indicated in Table 2.2-2, the degree of regional concern related to the presence of industry and non-industry personnel in the Beaufort Sea region varies from NEGLIGIBLE, in the case of terrestrial mammals and fish resources, to MINOR or MODERATE with some species of birds during critical life history stages. Snow geese would be particularly susceptible to human disturbance during nesting and staging. Since the regional population could be affected to a sufficient degree (in the worst case) to result in a change in abundance and/or distribution for more than one generation, the degree of regional concern with respect to this species could be considered MODERATE.

TABLE 2.2-2

SUMMARY OF POTENTIAL CONCERNS RELATED TO HUMAN PRESENCE IN THE BEAUFORT SEA REGION

Resource	Potential or Probable Effect	Degree of Potential Regional Concern
Terrestrial Mammals	Disturbance, habitat loss and increased hunting pressures, attraction of some species	NEGLIGIBLE to MINOR, depending on industry and government controls
Seals, Whales	Probably indistinguishable from effects of composite industrial activities	NEGLIGIBLE
Polar Bear and Arctic Fox	Attraction to areas of human (and industrial) activity	NEGLIGIBLE to MINOR
Nesting and brood-rearing birds	Decreased reproductive success, nest abandonment, increased susceptibility to egg predation, conspecific strife (dependent on species), increased energy expenditures associated with flushing, increased hunting pressures	NEGLIGIBLE to MODERATE (dependent on species and nesting habits)
Staging and nesting snow geese	Disturbance and increased energy demands	MINOR to MODERATE, depending on frequency of disturbances and location
Coastal and offshore fish resources	Over-exploitation	NEGLIGIBLE
Fish resources in freshwater habitats	Over-exploitation	NEGLIGIBLE to MINOR, depending on industry and government controls

2.3 ICEBREAKING

2.3.1 Introduction

The development of offshore petroleum hydrocarbon resources in the Beaufort Sea region will involve icebreaking activity throughout exploration, production and transportation phases of the project. Icebreaking vessels proposed for operation in this region are described in Table 2.3-1.

TABLE 2.3-1

ESTIMATED NUMBER OF ICEBREAKING VESSELS PROPOSED FOR OR IN CURRENT USE IN THE BEAUFORT SEA REGION

(Source: EIS Volume 2)

Vessel	Existing	1982-1990	1991-2000	Shaft Horsepower
Supply vessels (ice-strengthened)	9	19	29	-
AML X-3*	1	9	9	12,000
AML X-6	0	13	13	27,000
AML X-10	0	1	3	60,000
Class 6 Superdredge	0	4	4	60,000
Class 10 Tanker	0	11	24	75-150,000

* Arctic Marine Locomotive - Experimental Class 3

The ice-strengthened supply vessels and AML X-3 and AML X-6 vessels would operate primarily in the transition zone, although some activity would also occur in restricted areas of landfast ice. They would operate year-round to break up ice around the offshore platforms, and to maintain access channels through the fast ice between the transition zone and McKinley Bay and King Point during spring 'break-out' and moorage in fall. The larger AML X-10's would also be used to break up ice around offshore platforms, and would assist the icebreaking tankers. At the present time, the largest icebreaking vessel operating in the Beaufort Sea is the MV CANMAR KIGORIAK (AML X-4). An estimated 54 icebreaking support vessels may be required in this region by the year 2000 (EIS Volume 2).

The Class 10 icebreaking tankers are proposed for the transport of crude oil to southern markets, and would probably operate along a route known as the Northwest Passage through the southeastern Beaufort to Prince of Wales Strait, and through Viscount Melville Sound, Lancaster Sound, Baffin Bay and Davis Strait. After 1990, a small proportion of the fleet may move westward via the Alaskan coast and south through the Bering Sea. Although details regarding the proposed timing and exact geographic location of the tanker corridors have not yet been finalized, the fleet would probably operate year-round, but with more frequent trips during open water and 'thin ice' periods. Oil would be transferred from offshore storage and loading terminals (APLA's) in the Beaufort to the Class 10 icebreaking tankers. Each tanker could have a capacity to transport 2.4 x 10^7 m^3 (200,000 tons) of oil per trip and the tanker fleet is expected to accommodate 8000 m^3 (50,000 bbls) per day of oil production, based on transportation to Canada's east coast. An average round trip will require approximately 28 to 30 days including 8 to 15 hours for loading cargo in the Beaufort. The maximum number of icebreaking tankers which may be required in the Beaufort Sea by the year 2000 is expected to be 24, given a peak production rate of 193,980 m^3/day (1.22 x 10^6 BOPD) (EIS Volume 2).

There are several sources of potential disturbance associated with icebreakers operating in the Beaufort Sea region. Subsequent sections describe the potential effects of: (1) the physical breaking of ice and movement of vessels; (2) changes in the ice regime, stability and timing of freeze-up and breakup, and (3) the creation of a track and artificial leads in the ship's wake. In addition, the effects of fish entrainment in propellors of marine vessels will be briefly described. Other effects of operating icebreakers on marine flora and fauna are associated with disturbances that are common to various other industrial activities and are discussed in Sections 2.2 (Human Presence), 2.5 (Sewage), 2.6 (Underwater Sound), 2.8 (Airborne Emissions), 2.9 (Disposal of Solid Waste), 4.2 (Discharge of Heated Water) and 4.4 (Disposal of Ballast Water).

There is concern that icebreaking activity in the Beaufort Sea may influence the ice regime and affect the stability of the landfast ice-edge. This region is characterized by marked year to year variability in the timing of ice formation and breakup (EIS Volume 3A, Section 1.1) due to wind, surface currents, regional circulation and influences of the Mackenzie River. Consequently, limited icebreaking activity in the landfast ice zone is not expected to cause changes in the ice regime beyond the range of the natural ice variability (EIS Volume 4, Chapter 2, Section 4.4). The integrity of the artificial lead created in the track of the ship is also greatly influenced by prevailing winds, currents and ice pressures, and its characteristics may vary according to the purpose of the icebreaking activity. For example, the single passage of a vessel would probably have different effects on the surrounding ice than repeated icebreaking activity around a stationary drillship or offshore facility.

Repeated breaking and refreezing of ice in corridors through the landfast ice may augment ice growth, subsequently make the tracks more difficult to maneuver through, and delay local breakup (Martec 1981). For example, measurements in the track left by the MV CANMAR KIGORIAK in McKinley Bay indicated average ice thicknesses were 25 percent greater than in areas where no icebreaking had occurred (Danielewicz 1982). Although the effects of augmented ice growth on breakup patterns may be locally significant, potential changes in the regional ice regime are expected to be within the range of natural variability.

Ice in a ship's path may be overturned (exposing the under-ice surface), broken and forced under the ice sheet on either side of the ship track, or surface behind the ship. In the case of vessels making a single passage through the ice cover, the spatial extent of these disturbances is normally confined to a track approximately equal to the beam of the vessel. However, substantially larger areas would be disturbed on a regular basis when icebreakers are utilized to maintain ice-free areas around offshore platforms.

There is concern that the passage of an icebreaker through an ice field may create a temporary open-water track that could represent a barrier to Inuit hunters or Arctic foxes. However, under most conditions, the passage of a small icebreaking vessel through thick ice leaves a rubble-filled track that quickly refreezes. For example, the MV CANMAR KIGORIAK (beam 17 m, draft 8.5 m) produced a clean channel in ice thicknesses up to approximately 75 cm, while relatively little ice was cleared out of the track at greater ice thicknesses (MacLaren Marex 1979a). In ice approaching 120 cm thick, approximately 10/10 ice remained in the channel, and travel across the track was possible within 200 m of the passing vessel. It is expected that larger icebreakers would maintain a clearer channel in ice thicknesses approaching 120 cm, while tracks created in ice thicker than 120 cm would likely contain relatively large amounts of ice (MacLaren Marex 1979a).

Trials with the MV CANMAR KIGORIAK in McKinley Bay during late November 1981 demonstrated that the slush between the ice blocks in the track created by the icebreaker and 4 supply vessels had refrozen to a thickness of 2 cm within 1 hour of their passage. Within 2 hours, this slush had refrozen to a thickness of 5 cm, and a skidoo and driver (total 300 kg) was taken across the track with no indication of ice fracture. A 570 kg loaded komatik was successfully transported across the track 2.5 hours after the icebreaking activity (Danielewicz 1981).

2.3.2 Effects of Icebreaking on Mammals

The effects of icebreaking on marine mammals of the Beaufort Sea would vary with species life history activities of affected individuals during exposure, and the timing and location of the icebreaking activity. Potential effects could include direct mortality through collisions, behavioural disturbances associated with the physical presence and movement of the vessel, habitat loss, and disturbance by underwater noise generated by the vessel itself and by the physical breaking of ice. The first three potential areas of concern are discussed in the following sections, while the effects of underwater noise generated by icebreaking vessels are discussed in Section 2.6.5.

2.3.2.1 Direct Mortality

Direct mortality of ringed and bearded seal pups may result from collisions with icebreakers and/or crushing by displaced ice. Adult seals and other marine mammals (e.g. whales, polar bears, Arctic foxes) in the Beaufort region would probably not suffer direct mortality from icebreaking activity because of their ability to detect and avoid the vessels.

Breeding adult ringed seals occupy the landfast ice zone during the winter, while bearded seals and subadult and non-breeding ringed seals occur primarily in the transition zone or at the edge of the pack-ice (Stirling et al. 1977). The passage of icebreaking vessels through the landfast ice may directly affect ringed seals, although mortality of adults would be unlikely because seals should be able to avoid the vessels and occupy the zone of ice rubble in the wake of the ship. For example, recent studies in the Beaufort Sea showed that ringed seals reoccupied and maintained breathing holes in the track of the experimental Class 4 icebreaker MV CANMAR KIGORIAK (Alliston 1980). However, mortality of newborne ringed seal pups may occur during spring if icebreakers operate in landfast ice areas. Ringed seal pups are born in subnivean birth lairs on stable landfast ice in late March or early April (Smith and Stirling 1975). Primary pupping habitat for this species in the southeastern Beaufort Sea region includes the large bays in Amundsen Gulf and the west coast of Banks Island (Stirling et al. 1977, 1981b). Lactation lasts for 6-8 weeks, although females may abandon the pups if the ice breaks up early (McLaren 1958). During the nursing period, the pups would be unable to avoid an approaching vessel and could therefore be crushed by vessels passing through any birth lairs. In addition, exposure of pups to cold water early during the nursing period would probably be fatal. The extent of pup mortality would depend on the frequency and location of icebreaking activity and the density of birth lairs in the area of icebreaking.

Bearded seal pups are typically born on moving pack ice during late April or early May, and lactation lasts for 12 to 18 days before the pups are abandoned (Burns and Frost 1979). Mortality of bearded seal pups may also occur if icebreaking vessels pass through bearded seal pupping habitat, although the number of individuals affected should be relatively low because this species is widely distributed, the pups are able to enter the water and feed themselves soon after birth, and most individuals should be able to avoid a vessel.

2.3.2.2 Behavioural/Disturbance Responses

During the September 1979 maiden voyage of the MV CANMAR KIGORIAK from St. John, N.B. to Tuktoyaktuk, N.W.T., direct behavioral observations of marine mammals along the route through pack ice areas of Viscount Melville Sound were recorded (MacLaren Marex 1979a). All 5 polar bears observed during that portion of the trip retreated in response to the presence of the icebreaker, while most ringed and bearded seals observed within 1.6 km (some as close as 500 m) of the bow responded by submerging and vacating the area. Similar avoidance reactions by whales and Arctic foxes have not been documented, although 'fright/flight' behavioural responses may also occur with these species. The bioenergetic implications of icebreaker-induced avoidance responses by marine mammals in the Beaufort Sea are unknown, but would presumably vary in significance depending on species, general health of individuals, time of year, and frequency of disturbance.

2.3.2.3 Habitat Loss

Icebreaking activity in the landfast ice zone or at the landfast ice edge may alter the timing of local breakup, although regional effects will probably be inconsequential (EIS Volume 2). Local habitats for breeding ringed seals and polar bears with cubs could be created if breakup is delayed, while habitat for whales, bearded seals, non-breeding ringed seals, and male and non-breeding polar bears may be reduced. Nevertheless, these species are probably well adapted to the marked annual variability in ice conditions which occur naturally in the Beaufort Sea. In addition, a localized enhancement or reduction of food sources for some species of mammals may occur in the icebreaker's track, although the degree of concern associated with these changes is expected to be NEGLIGIBLE due to their limited spatial extent and short-term nature (Sections 2.3.4, 2.3.7 and 2.3.9).

Alliston (1980) studied the effects of winter icebreaking by the MV CANMAR KIGORIAK in landfast ice areas off McKinley Bay on the distribution of ringed and bearded seals during winter and during spring haul-out. The vessel completed three icebreaking excursions through the experimental portion of the study area during the winter and spring of 1980. The total length of icebreaker tracks was approximately 250 km, and about 4 km^2 of landfast ice (or 0.5 percent of ice in the experimental area) was directly affected by the icebreaking. Comparison of breathing hole densities in the experimental and control areas did not indicate any significant differences in use of these areas by ringed seals during winter, while comparison of densities of hauled-out seals during the first day of the survey suggested that there were higher ringed seal densities in the experimental area. During the second day of the survey, the densities of ringed seals in the experimental and control areas were similar, and the author concluded that use of the experimental area was as high (or higher) than the control area during haul-out. In addition, both wintering and hauled-out ringed seals within the experimental area exhibited an apparent preference for areas in which some icebreaking had occurred. This observation is consistent with the hypothesis that seals are attracted to late freezing cracks in the ice (Smith and Stirling 1975). Although the number of bearded seals recorded during these surveys was not sufficient to determine any statistically significant trends, movement of at least 30 seals into the breakout track was observed during one survey (Alliston 1980).

There has been recent concern that white (and possibly bowhead) whales may follow artificial leads created by icebreaking vessels and become entrapped when the leads refreeze. Natural entrapment of white whales and narwhals has been reported in west Greenland (Vibe 1967; Kapel 1977) and in the Canadian Arctic (Degerbøl and Freuchen 1935; Freeman 1968; Finley and Johnston 1977), although there are no records of entrapment in the Beaufort Sea. The potential for entry and entrapment of white and bowhead whales in artificial leads in the Beaufort Sea during spring is probably remote since these mammals are well adapted to orientation and navigation in areas with extensive ice cover, and because tracks left by vessels moving through thick ice (e.g. >6 m) would be filled with heavy ice rubble and refreeze rapidly under most conditions.

The degree of potential regional concern regarding the overall effects (direct mortality, disturbance and habitat loss) of icebreaking on marine and marine-associated mammals will likely vary from <u>NEGLIGIBLE</u> in the case of ringed and bearded seal adults and subadults, white and bowhead whales (given the present data base), polar bears and Arctic foxes, to <u>MINOR</u> and <u>MODERATE</u> in the case of bearded seal pups and ringed seal pups, respectively.

2.3.3 Effects of Icebreaking on Birds

In general, the potential effects of normal shipping and icebreaking activity on marine-associated birds in the Beaufort Sea region will probably be limited because these species are highly mobile and can readily avoid vessels. Energy expenditures involved in local movements of most species would likely be inconsequential. One possible exception to this generalization may occur during the passage of marine traffic near nesting colonies where birds are highly concentrated during the breeding season. Continual disturbance may result in reduced reproductive success, nest destruction or abandonment of the colony (Petro-Canada Ltd. 1981), although the extent and regional significance of this disturbance would vary with the location and timing of the vessel traffic.

The creation of 'clean' artificial leads or pools by icebreaking vessels operating in thin-ice areas may provide additional staging habitat for spring migrants (e.g. oldsquaws, eiders, loons, glaucous gulls). However, any effects associated with creation of artificial leads or alteration of the timing of breakup would probably be insignificant in relation to the marked annual variability in ice conditions observed in the Beaufort Sea (EIS Volume 3A, Section 1.1).

Birds have adapted to the presence of ship traffic throughout the world, and some species (e.g. gulls, fulmars) are frequently attracted to ships and follow in their wakes (Wahl and Heinemann 1979). During the voyage of the MV CANMAR KIGORIAK through nearshore ice areas in Viscount Melville Sound in September 1979, glaucous gulls and ivory gulls were frequently attracted to the vessel in the same manner as gulls anywhere react to the presence of ships (MacLaren Marex 1979a). Several species of birds would probably feed within the ice rubble in the track behind the ship because of increased availability of fish and amphipods (Sections 2.3.4 and 2.3.8).

The degree of regional concern regarding potential effects of icebreaking and icebreaker traffic on birds in the Beaufort Sea region is expected to vary from NEGLIGIBLE for spring migrant species to MODERATE in the case of colonial nesting birds if vessels frequently operate near the colonies.

2.3.4 The Effects of Icebreaking and Marine Vessel Traffic on Fish

Although the effects of icebreaking on fish remain poorly documented, observed and potential effects include disturbance and avoidance responses associated with underwater noise, direct mortality through entrainment by propellers, and stranding during displacement of ice. The effects of underwater noise on fish are discussed in Section 2.6.6, while available information regarding the latter two potential effects is briefly summarized in the present section.

The entrainment of fish by propellers could occur with all forms of vessels including icebreakers, although the extremely limited evidence of mortality suggests that most fish avoid or are forced away from ship propellers. In the Arctic, the only documented case of fish mortality occurred when Arctic cod were apparently congregated under a moored vessel in Prudhoe Bay, and were killed when the ship began to move (Tarbox and Spight 1979). Although similar incidents may also occur under certain circumstances in the Beaufort Sea, the regional significance of fish mortality due to entrainment by propellers would undoubtedly be NEGLIGIBLE.

Andriashev (1970) reported that Arctic cod were stranded on ice overturned by icebreaking, and subsequently taken by kittiwakes. Since the association of Arctic cod with irregularities in the under-ice surface has been documented (Emery 1973), the increased structural complexity resulting from icebreaking may attract or provide additional habitat for this species. However, in view of the relatively small area of fish habitat affected by icebreaking, the mobility of these species and the period when most icebreaking occurs, the degree of regional concern associated with the effects of this activity on fish communities of the Beaufort Sea is considered NEGLIGIBLE.

2.3.5 Effects of Icebreaking on Phytoplankton

There are no available data describing the effects of icebreaking on planktonic communities. However, some possible effects of this activity on phytoplankton may include the stimulation of primary production due to an increase in light intensity and nutrient availability, or the physical disturbance of the epontic community (Section 2.3.8) which may be important for the initiation of the summer phytoplankton bloom.

The physical effects of icebreaking activities may be similar to the formation of natural leads and polynyas in the spring, although icebreaker tracks would refreeze within minutes during most of the winter. In a recent review on polynyas, Stirling (1980) emphasized the lack of information on the biological importance of these features, but suggested that they were likely one of the more ecologically important areas in the Arctic. Considerable biological activity occurs around polynyas, as evidenced by the large populations of birds and mammals which frequent these areas (EIS Volume 3A; Sections 3.2 and 3.3). However, it is not known whether this utilization is a result of increased productivity at the lower trophic levels, or simply due to the availability of open water and new food supplies. There is some evidence that wind-induced upwelling occurs along ice edges (Buckley et al. 1979), and this may act as a mechanism for nutrient regeneration. The combination of increased nutrient availability and higher light intensities in these open water areas may stimulate primary production and phytoplankton growth since ice cover in the Arctic is one of the major factors limiting the development of phytoplankton populations (Bursa 1963; Allen 1971). However, icebreaking would only lead to a potential increase in primary productivity during spring and early summer, when sufficient light is available at these high latitudes and temperatures are warm enough to prevent immediate refreezing of icebreaker tracks.

Increased primary production would generally be considered a positive impact of icebreaking activity, although its spatial extent would be regionally insignificant in comparison to open-water productivity during the late spring and summer. As indicated in the following section, the only possible concern related to localized early increases in primary production would be the potential for premature stimulation of zooplankton breeding during a period of only temporary increased food availability. Nevertheless, the degree of potential concern regarding effects of icebreaking on regional or local phytoplankton communities is considered NEGLIGIBLE.

2.3.6 Effects of Icebreaking on Zooplankton

The available information on the possible effects of icebreaking on zooplankton is limited to those species (primarily amphipods) that are associated with the epontic community (Section 2.3.8). In general, any effects of icebreaking would be indirect, and related to any increase in food availability resulting from stimulated phytoplankton productivity (Section 2.3.5).

In the fall, the majority of the zooplankton species descend to depths well below the surface water layer and the euphotic zone. They remain in deep water throughout the winter and begin their ascent in the spring. Even those species which continue to move into surface waters during diel vertical migrations are only present near the surface in low numbers. Those species which tend to remain in the upper levels of the water column during the winter (e.g. the copepods Oithona and Pseudocalanus) are not concentrated in a narrow surface layer as they are known to be in the late spring and summer (Vinogradov 1970). Therefore, if localized increases in phytoplankton productivity did occur during early spring, the subsequent indirect effects on the zooplankton community would be NEGLIGIBLE.

The timing of the ascent and reproduction of zooplankton in the spring is coincident with the occurrence of the phytoplankton bloom. Therefore, if a localized premature bloom resulted from icebreaking activities, it could lead to the earlier production of eggs and larvae by zooplankton. This would be detrimental to the zooplankton community if the phytoplankton bloom was only temporary and renewed ice cover resulted in a reduction of available food for the larvae. Since larval fish (ichthyoplankton) also feed extensively on phytoplankton and zooplankton, these short-term increases in primary production could have similar local negative effects on some fish populations. However, it is anticipated that regional effects of icebreaking on zooplankton would be NEGLIGIBLE since only relatively small and temporary areas of open water would result from this activity.

2.3.7 Effects of Icebreaking on Benthic Communities

There is no information on the effects of icebreaking activity on benthic communities, presumably because icebreaking generally occurs in waters deeper than where benthic flora and fauna would be affected. Icebreaking activities could influence benthic communities by increasing the amount of incident light available to benthic micro-algae during a period when the ice cover normally prevents development of these populations. However, these flora normally only occur at depths considerably shallower than those used by marine vessels, particularly in the nearshore Beaufort Sea where relatively high suspended sediment levels further restrict light transmissibility. Since areas of open water would also be relatively localized and would generally refreeze except near breakup, any positive effects of temporary increases in light levels at the seafloor would likely be NEGLIGIBLE in terms of the seasonal productivity of the micro-algal community.

2.3.8 Effects of Icebreaking on Epontic Communities

Physical disruption of the ice sheet by icebreaking vessels during the spring bloom of epontic flora would likely result in mortality of exposed epontic organisms on overturned ice blocks, and also of epontic organisms which become trapped above or below ice fragments pushed into lateral rubble fields. This latter effect of icebreaking on epontic organisms has been observed from the decks of icebreakers in both Arctic and Antarctic waters (Andriashev 1970; MacLaren Marex 1979a). Direct mortality of some exposed epontic fauna may also be associated with predation by birds (Section 2.3.3). Other investigators have suggested that rough under-ice surfaces created in the icebreaker's track may protect invertebrates grazing on epontic flora from under-ice currents (Lewis and Milne 1977; Divoky 1978), and could also provide habitat for cod which feed on these invertebrates (Emery 1973). There is an increasing volume of evidence which suggests that ice-edge areas of polynyas and recurrent leads may be more important biologically than either ice-covered areas or areas of open water, particularly during the winter and spring (Dunbar 1981). Observations of birds, fish and marine mammals concentrating in these areas lend support to this hypothesis, although little is presently known regarding the role of the epontic commmunity in this food web. During the spring, icebreaking may lead to an increase in the primary and secondary production of these regions when and where vessel traffic is frequent enough to maintain open areas in the ice.

The lack of data describing other potential lethal and sublethal effects of icebreaking on epontic organisms, and the fact that most available information regarding the epontic community is site-specific and limited to a brief period during the spring bloom (March to June), hamper assessment of the overall impacts which may be associated with this activity. For example, the light requirements of epontic algae are not well understood, and potential effects which may result from the more variable under-ice light intensities in areas disturbed by icebreaking are therefore difficult to assess. Available data indicate that very low light intensities are capable of initiating photosynthesis in the Arctic (54-107 lux; Grainger 1977), and it has been suggested that high light intensities may bleach the photosynthetic pigments of epontic algae, reduce carbon fixation and ultimately lead to the deterioration of the community (Apolloni 1965; Grainger 1977). Grainger (1977) suggested that light and annual differences in the quantity and duration of snow cover, rather than available nutrients, were the most important factors causing the variability in the timing and quantity of the annual sea-ice algal bloom. However, other researchers have expressed conflicting opinions regarding the factors that limit spring epontic production. If the epontic flora are light limited, then the community which developed under areas containing thicker rafted ice rubble could have an overall lower biomass and productivity than communities in adjacent undisturbed areas where light penetration through the ice may be greater. There is only circumstantial available information regarding the effects of varying light intensities on epontic fauna. Observations of amphipods preferentially remaining in areas of higher light intensity (Grainger 1977) and near cracks in the ice (Green and Steele 1975) suggest that positive phototropism could result in higher concentrations of amphipods in areas fractured by icebreaking.

The potential effects of icebreaking prior to the spring bloom of epontic flora are impossible to predict because of lack of information regarding the distribution and physiology of the overwintering epontic organisms. However, since the areal extent of any effects of icebreaking on epontic organisms would be minimal in comparison to the available epontic habitat in the Beaufort Sea region, the degree of potential concern regarding this activity-resource interaction is considered NEGLIGIBLE.

2.3.9 Summary of Concerns Related to Icebreaking

As indicated by the summary data provided in Table 2.3-2, the degree of potential regional concern associated with icebreaking is NEGLIGIBLE for most flora and fauna of the Beaufort Sea. This is primarily due to the small area of disturbance in relation to available habitat for most resources. The most significant area of concern with respect to icebreaking in the Beaufort region is the potential for mortality of ringed seal pups during the spring lactation period (March to May) if icebreaking occurs in landfast ice areas, particularly off the Yukon coast, in Amundsen Gulf and along the west coast of Banks Island. Depending on the frequency and location of icebreaking activity, and the density of birth lairs in these areas, the degree of regional concern could be considered MODERATE. There is a lesser degree of concern (MINOR) for mortality of bearded seal pups, which are typically born on moving pack ice, since this species is widely distributed and the pups are able to enter the water soon after birth and could therefore potentially avoid an icebreaker. There would also be a MODERATE degree of regional concern if icebreaker routes were located too close to nesting habitats of colonial bird species since disturbance during the nesting period could lead to reduced reproductive success or nest abandonment.

TABLE 2.3-2

SUMMARY OF POTENTIAL CONCERNS RELATED TO ICEBREAKING ACTIVITY IN THE BEAUFORT SEA REGION

Environmental Component or Resource	Potential or Probable Effects	Degree of Potential Regional Concern
Ice edge	Decreased integrity when tracks intersect to produce continuous broken ice region	Within range of natural ice variability when routes controlled
Artificial leads and rafting	Tracks and ice rubble would quickly refreeze in most months; rafted ice and/or open leads may affect some mammals	See specific resources for biological concerns
Ringed seal pups	Direct mortality through collision or crushing by rafted ice (spring; landfast ice zone)	MODERATE
Bearded seal pups	Direct mortality as above (spring; pack ice zone); avoidance responses (see text)	MINOR
Ringed and bearded seal adults and subadults	Avoidance responses with subsequent attraction to broken ice areas; habitat loss for breeding ringed seals; habitat creation for bearded seals and non-breeding ringed seals	NEGLIGIBLE
Polar bears	Avoidance responses; habitat loss (bears with cubs) and creation (males and non-breeding adults)	NEGLIGIBLE
Arctic foxes	Avoidance responses	NEGLIGIBLE
White and Bowhead Whales	Avoidance responses; habitat creation; entrapment in artificial leads	NEGLIGIBLE, given present data base
Colonial, nesting bird species	Disturbance leading to reduced reproductive success or nest abandonment	MODERATE, depending on routes, frequency, and timing

TABLE 2.3-2 (Cont'd)

Environmental Component or Resource	Potential of Probable Effects	Degree of Potential Regional Concern
Spring migrant bird species	Creation of staging habitat for oldsquaws, eiders, loons and gulls; increased availability of food; attraction of some species (e.g. gulls, fulmars)	NEGLIGIBLE
Pelagic Fish	Entrainment by propellers; stranding on overturned ice and attraction to under-ice irregularities	NEGLIGIBLE
Phytoplankton	Localized and temporary increases in primary production and growth due to increased light and nutrient availability (ice-edge upwelling)	NEGLIGIBLE
Zooplankton and ichthyo-plankton	Indirect local increases in food availability if primary production is increased	NEGLIGIBLE
Benthic micro-algae	Increased light availability in shallow areas may increase production after icebreaker-induced increases in water turbidity disappear	NEGLIGIBLE
Epontic flora	Mortality on overturned ice; potential decreased production in areas with thicker or rafted ice where light intensity is reduced	NEGLIGIBLE
Epontic fauna	Direct mortality on overturned ice and increased predation by birds; phototropic attraction to fractures where light intensity is higher	NEGLIGIBLE

2.4 DREDGING

2.4.1 Introduction

Dredging and subsequent disposal or placement of spoil will be major activities associated with almost all phases of future oil and gas exploration and production in the Beaufort Sea region. This section describes the various types of dredging activities, while subsequent sections provide detailed discussions of the potential physical effects of these operations and biological effects of dredging on mammals, birds, fish and members of lower trophic levels.

Dredging is presently required or may be necessary in the future for the following purposes:

1. To provide materials used in the construction of artificial islands and foundation pads or berms for offshore platforms, and ballast for caisson platforms. These materials are generally taken in areas where water depths range from 10 to 50 m.

2. To excavate glory holes required for drilling from floating platforms (conventional drillships and conical drilling units) in less than 54 m of water.

3. To provide ice-scour protection required for production trees, manifold templates, subsea pipelines and other possible seafloor systems.

4. To construct and maintain harbours of appropriate depth and configuration to accommodate present and future marine fleet requirements.

5. To maintain artificial islands damaged by ice or wave activity, and possibly to assist in the removal of abandoned islands.

Although the amount of material dredged for each purpose would vary depending upon site characteristics, changing development requirements and environmental factors (such as storm frequency), volumes of material which may be required per facility are presented in Table 2.4-1. The types and sizes of dredging equipment employed in the Beaufort Sea region would also vary with the location, volume and type of material to be removed, but could include conventional cutter suction, clamshell, dragline, suction bucket and/or trailing suction hopper dredges. Dredging activities are presently restricted to the open water season, although after delivery of the first Arctic class dredge in 1983-84, dredging could occur throughout most of the year. The duration of dredging activities would vary with the type of operation and

TABLE 2.4-1

ESTIMATED DREDGING REQUIREMENTS FOR VARIOUS PROPOSED FACILITIES IN THE BEAUFORT SEA

Type of Facility	Time Required for Construction (yrs)	Volume of Dredged Material per Facility (million m^3)
Tanker Loading Facility		
Shallow water (<20m)	3	30
Production Platform		
Shallow water (<20m)	1 - 2	5
Deep water (>20-60m)	3 - 4	40
Exploration Island		
Shallow water	1	1
Deep water	2	7
Caisson Island Ballast		0.70-0.80
Pad and Berms		0.76-3.8
New Harbour		2 - 6
Pipeline		0.05/km
Glory Hole		0.04-0.05

could range from short-term dredging programs required for maintenance of islands and harbours to an estimated three year intensive dredging program for construction of an APLA. In situations where granular materials for artificial island construction are available in the immediate vicinity of the site, a standard suction dredge (e.g. BEAVER MACKENZIE) equipped with a floating or submerged pipeline may be used. In other instances, approved borrow locations distant from construction sites may also be utilized; in this case, dredged materials would be barged or carried in hopper dredges to the construction location. The new trailing hoppper suction dredges will be capable of operating in water depths of 80 m and would generally remove a 1.0 to 1.5 m deep surface layer of granular materials.

2.4.2 Physical and Chemical Effects of Dredging

Dredging operations and artificial island construction began in the southern Beaufort Sea during 1972, and since this date several studies and monitoring programs have been conducted in conjunction with these activities. The results of most of these investigations were summarized in an annotated bibliography by ESL (1980). In addition to these actual measurements of the effects of dredging in the Beaufort Sea, there are numerous literature reviews describing the effects of dredging on temperate aquatic environments and resources (e.g. Morton 1977; Wright 1978). The primary effects of dredging and spoil disposal activities are alterations in the physical and chemical properties of the water column and substrate. Potential or documented physical and chemical effects which have been described in the above sources include:

1. increased suspended sediment levels and turbidity downstream of operating dredges;
2. reductions in dissolved oxygen and increases in nutrient concentrations downstream of operating dredges;
3. localized changes in vertical salinity/temperature structure of the water column;
4. altered bottom contours and sediment composition;
5. changes in water circulation patterns;
6. thicker ice and delayed ice breakup in spring;
7. erosion and/or re-deposition of sediments in areas adjacent to dredging or deposition sites;
8. resuspension of toxic substances such as trace metals from contaminated sediments or from drilling muds and formation cuttings near exploration and production platforms;
9. introduction of treated sewage, noise and grease into the water column;
10. icebreaking (when winter dredging occurs).

The spatial extent, duration and magnitude of physical and chemical effects which have been documented following past dredging operations are discussed below, while the biological effects of dredging are the subject of subsequent subsections.

2.4.2.1 Turbidity Plumes

The most visible and immediate effect of dredging operations is the creation of a turbidity plume which extends downstream of the activity. Sediment plumes have been monitored during dredging and artificial island construction operations in the Beaufort Sea (Slaney 1974b; Envirocon 1977; McDonald and Cambers 1977a; Thomas 1979; Erickson and Pett 1981), as well as during numerous dredging projects in southern latitudes (see Morton 1977 and Wright 1978 for reviews). These studies indicate that the size and character of the plume are affected by the type and volume of materials being removed, the existing background water quality and current regime, local weather conditions and the type of dredge involved in the operation. For example, McDonald and Cambers (1977a) monitored the effects of the hydraulic suction dredge, BEAVER MACKENZIE, during an uncontained fill operation in approximately 10 m of water outside of Kugmallit Bay. The results of the study indicated that the dredge-created plume was measurable (transects of percent transmissibility) for a maximum distance of 4.8 km from the outfall when the Mackenzie River plume did not extend into the area of dredging activities. However, the turbidity plume was only detectable for 1.6 km from the outfall when deposition of dredged material was within the Mackenzie River plume.

Dredging activities in McKinley Bay and Tuktoyaktuk Harbour caused substantially smaller (500-1200 m in length) turbidity plumes than those observed in Mackenzie Bay (1600-4800 m), possibly due to the more restricted water circulation patterns and coarser sands in these sheltered habitats (Thomas 1979; Erickson and Pett 1981). Envirocon (1977) monitored the construction of an artificial island, and reported a turbidity plume which was visible for up to 1 km downstream of the island when background turbidities were low. However, no dredge plume was detectable after an August storm which created high background turbidities.

The width of the plume usually varies from a very narrow (a few metres) and well defined area near the source to greater than 1 km as it disperses with distance from the dredging activity. While the shape of turbidity plumes appears to be quite variable, the maximum surface area affected is probably approximately 5 km from the outfall and approximately 2 km wide, or a total areal extent of 10 km^2 for a stationary dredging operation in water with relatively low background turbidity and moderate surface currents.

In some situations, the turbidity plume may be wider in mid-depth to bottom waters than at the surface. For example, McDonald and Cambers (1977a) measured percent transmissibility depth profiles 20 m outside a visible surface plume, and found that transmissibility declined from 85 percent (background) at the surface to 50 percent at a depth of 6 to 7 m. This study also indicated that transmissibility profiles within the turbidity plume were variable, with a uniform zero percent transmissibility from surface to bottom occurring in some instances, and a relatively unaffected mid-depth layer being found at other sites McDonald and Cambers (1977a). Envirocon (1977) also reported subsurface sinking and spreading of a turbidity plume at an artificial island construction site in approximately 12 m of water north of Pullen Island. The plume was not visible at the surface 2 km from that island, but suspended solids concentrations measured at depths of 1 and 5 m indicated that it was still dispersing in subsurface waters. Gordon's (1974, cited in Morton 1977) study of the disposal of dredged clay and silt in Long Island Sound indicated that these spoils fell as a density current rather than as individual particles, and that upon reaching the bottom, the density current spread laterally to produce a bottom turbidity cloud.

Morton (1977) suggests that a second process called flocculation has an important influence on the settling velocities of suspended sediments. In seawater which has a high electrolytic content, clay particles are attracted to and cohere with inorganic salts and organic compounds, forming larger sized flocs whose settling velocities are many times faster than the individual clay particles (Meade 1972, cited in Morton 1977). This phenomenon would be accentuated when sediments were dredged from a less saline area and released in a more saline region, since increasing salinity enhances the cohesion between particles, and increasing suspended concentrations enhances the probability of collision, both of which promote flocculation. Therefore, the vertical extent of the plume is determined by a number of factors (local current regimes, depth of the water column, particle size distribution of the plume, vertical water density characteristics, and local weather conditions which could cause mixing of the water column), and while in some cases the whole water column may be affected, in other areas the plume may be present only in the bottom waters. In a similar manner, only mid-depth waters may be affected if water depth is sufficient, a strong pycnocline is present and the plume entrains enough ambient water to become neutrally buoyant (Wright 1978).

Percent transmissibility, turbidity, and concentrations of suspended solids are parameters frequently used to determine the area affected by dredging operations, although levels of suspended sediment are not directly comparable to turbidity and must therefore be considered independently. The maximum surface concentration of suspended solids within turbidity plumes monitored by McDonald and Cambers (1977a) was 187 mg/L, but a much higher level (333 mg/L) was recorded in a near-bottom sample collected from within the plume. During this dredging program, background concentrations of suspended solids ranged from 8.8 to 16.6 mg/L at the surface and from 44.5 to 66 mg/L at the bottom (McDonald and Cambers 1977a). Envirocon (1977) reported a similar maximum concentration of suspended sediments (388.8 mg/L) in a dredge-created turbidity plume, but levels were reduced to 14.5-99.8 mg/L within 500 m of the artificial island construction site. In a more recent study, Thomas (1979) reported maximum suspended solid levels of 410-586 mg/L in surface samples at a dredge discharge site in McKinley Bay, with concentrations decreasing rapidly with distance from the outflow point.

Suspended sediment levels measured at distances of 160, 320 and 400 m were 83, 61, and 21 mg/L, respectively. Gordon (1974, cited in Morton 1977) conducted an examination at a spoil ground in Long Island Sound to investigate the disposal of unconsolidated spoil with a high silt-clay fraction which was being discharged as a jet with 70-75 percent water content. Using turbidity measurements to calculate the lateral spread of descending spoils, he estimated that 80 and 90 percent of the spoil reached the bottom (water depth 20 m) within a 30 m and 120 m radius of the discharge point, respectively. The results of these studies indicate that although a turbidity plume may be visible for several kilometres from the dredging activity, concentrations of suspended solids generally decrease rapidly with distance.

The duration of the physical and chemical effects associated with dredging-created turbidity plumes is perhaps the most important factor affecting subsequent impacts on aquatic organisms. Studies of dredging in the Beaufort Sea have clearly documented the spatial extent of the turbidity plumes, but few studies have described the attenuation of turbidity plumes with time. Nevertheless, investigators that have examined dredging in the Beaufort Sea agree that turbidity plumes are "short-term", "temporary" (McDonald and Cambers 1977a) or "short-lived" (Thomas 1979). Post-dredging sampling in Tuktoyaktuk Harbour indicated that excess suspended particulate material decreased to pre-dredging levels 10 hours after the termination of a dredging operation (Erickson and Pett 1981). Turbidity plumes have been monitored in southern latitudes following the cessation of dredging, and have been reported to last for 1 to 2 hours (Chesapeake Biological Laboratory 1970, cited in Morton 1977) and as little as 30 minutes (Wright 1978). Wright (1978), in a review of several dredge monitoring programs conducted for the U.S. Army Corp of Engineers, reports that the duration of the turbidity plume depends on particle size, currents, turbulent mixing and other related factors. The attenuation of turbidity plumes following the termination of dredging activity may be relatively rapid (in the order of hours) when coarse bottom materials are involved, but when fine-grained sediments (e.g. clays and silts) are suspended in quiescent waters, longer periods may be required before turbidities return to normal levels (Slaney 1975). Depth may also be a factor affecting persistence of a plume; where bottom waters have a higher density than surface waters, a plume may persist longer in intermediate or bottom waters due to slower particle settling rates (Wright 1978). These investigations all indicate that the duration of turbidity plumes beyond the completion of dredging appears to be insignificant in relation to the usual duration of the operation.

2.4.2.2 Dissolved Oxygen and Nutrient Concentrations

Other chemical water quality characteristics which have been temporarily affected by dredging activities include dissolved oxygen and nutrient concentrations. Changes in dissolved oxygen and nutrient levels are primarily the result of the suspension of bottom sediments with a high organic content. Nutrients are released when organic compounds which originate from low-oxygen content sediments are introduced into oxygen-saturated waters. These nutrients, in conjunction with available oxygen, stimulate the growth of micro-organisms, which then consume oxygen. Unlike turbidity plumes, changes in nutrient and oxygen levels have not always been observed during dredging activities. Thomas (1979) recorded slight reductions in surface dissolved oxygen levels during the operation of a cutter suction dredge in McKinley Bay. The lowest oxygen concentration measured was 7.4 mg/L 200 m from the outfall, while other concentrations ranged from 8.9 to 10.1 mg/L at stations located 20 m to 1700 m from the outfall (Thomas 1979). The background dissolved oxygen concentrations during this study were 10.9 to 11.6 mg/L. Thomas (1979) calculated that oxygen saturation was depressed from a minimum of 3 percent to a maximum of 30 percent below background. Slightly depressed oxygen concentrations were also reported in the turbidity plume created by a cutter suction dredge at Tuft Point; dissolved oxygen levels decreased from 7.34-8.07 mg/L at a control station to 6.05 - 6.31 mg/L within the plume (McDonald and Cambers 1977a). Other investigators monitoring dredging operations have found no differences in surface dissolved oxygen concentrations within and outside dredge-created turbidity plumes. For example, McDonald and Cambers (1977a) monitored a suction dredging operation offshore of Kugmallit Bay and reported that surface concentrations of oxygen within and outside the plume were 10-13 mg/L and 9-14.5 mg/L, respectively. Envirocon (1977) also found no differences in dissolved oxygen profiles at stations within and beyond the influence of artificial island construction activities in the Beaufort Sea, while data collected by Slaney (1974b) during open water gravel removal operations in Mackenzie Bay indicated that dissolved oxygen concentrations were not significantly affected by these activities. Erickson and Pett's (1981) study in Tuktoyaktuk Harbour also indicated that dissolved oxygen levels were not affected by dredging activities. However, even when minor reductions in oxygen levels occur as a result of dredging, such changes in water quality are unlikely to persist due to natural mixing caused by currents, wind and waves.

Nutrient concentrations have only been occasionally monitored during investigations of the effects of dredging operations in the Beaufort Sea, and as in the case of dissolved oxygen, they have not always been affected by dredging. McDonald and Cambers (1977a) measured phosphate, nitrate, silica and total carbon concentrations at control stations and within turbidity plumes created by two dredging operations located just outside Kugmallit Bay. During July, elevated levels of total organic carbon were recorded within the turbidity plume (8-33 mg/L compared to 5-17 mg/L at control stations), although concentrations of phosphate, nitrate and silica were not affected by the dredging. However, in September, there were no significant differences in any of the nutrient concentrations measured within and outside the plume. Silica, ortho-phosphate and nitrate-nitrogen levels were also unaffected during an August 1974 gravel removal operation in Mackenzie Bay (Slaney 1974b). On the other hand, Envirocon (1977) reported slight increases in the concentration of nitrate-nitrogen at stations near island construction activities (mean 0.37 μg-at/L compared to 0.11 μg-at/L). Investigators monitoring dredging activities in southern latitudes have reported much greater changes in nutrient concentrations. For example, total phosphate concentrations 1000 times ambient levels and total nitrogen concentrations 50 times ambient levels were recorded in Chesapeake Bay following spoil disposal (Chesapeake Biological Lab. 1970, cited in Morton 1977). Changes in nutrient levels of this order of magnitude are likely associated with the suspension of bottom sediments contaminated with high concentrations of organics. Thomas (1979) reported that total organic carbon content of sediments in McKinley Bay ranged from 1 to 4 percent, while Slaney (1974b) found 1.2 to 12.8 percent organic content in Mackenzie Bay sediments. Since nearshore Beaufort Sea sediments appear to have relatively low levels of total organic carbon, and have not been contaminated by disposal of nutrient-rich wastes, increases in nutrient concentrations in these waters as a result of dredging are likely to be minimal.

2.4.2.3 Resuspension of Toxic Chemicals

Other potential changes in water quality associated with dredging operations may result from the resuspension of toxic substances from previously contaminated sediments. These effects have been particularly observed in industrialized harbours where long-term release of effluents in confined waters have resulted in increased metal levels in the water column (Keeley and Engler 1974, cited in Morton 1977). However, this is probably not a significant area of concern in the Beaufort Sea since sediments in this region do not contain the magnitude or diversity of contaminants found in sediments adjacent to large industrial and population centers in temperate latitudes (EIS Volume 3A, Section 1.5). In addition, monitoring programs completed to date in the Beaufort Sea have not demonstrated any changes in trace metal concentrations, pH or other indicators of sediment contamination during dredging operations (McDonald and Cambers 1977a,b; Thomas 1979). The only potential area of significant concern would be if dredging released any toxic materials from localized areas where drilling or oily wastes or untreated sewage had previously accumulated, although it is anticipated that most dredging operations would avoid areas where these wastes are released.

2.4.2.4 Temperature and Salinity

The vertical temperature and salinity stratification of the water column has occasionally been affected by mixing at a dredge spoil disposal site. During July 1976, a strong halocline and thermocline were measured outside the dredging-created plume at Arnak L-30 artificial island site (McDonald and Cambers 1977a). Monitoring within the plume indicated that hydraulic dredging caused localized mixing of warmer and less saline surface water with cooler saline water from greater depths, resulting in a homogeneous vertical profile within the plume. However, the authors reported that the normal stratification was only broken down for a short period, and that temperature and salinity profiles measured within the plume were within the normal seasonal range expected at the site. On the other hand, dredging in McKinley Bay during 1979 did not result in a disruption of the normal vertical profile of temperature and salinity (Thomas 1979). Dredging in nearshore areas may also allow colder and more saline bottom water to intrude into otherwise warm and less saline areas, rendering these habitats unsuitable in terms of the habitat requirements of resident or migratory fish and benthic invertebrate species (Sections 2.4.5 and 2.4.9). This effect has not been documented following dredging operations in the Beaufort Sea to date, although long-term changes in the temperature and salinity regime in important fisheries habitat would clearly be of greater potential concern than temporary and localized alterations in water column stratification due to mixing.

2.4.2.5 Altered Bottom Contours

Dredging results in deep troughs and holes at borrow locations, and shallower water at island or spoil disposal sites. Changes in nearshore bottom contours may subsequently alter circulation patterns and wave refraction processes, which could result in sediment erosion or deposition along adjacent shorelines, as well as changes in the salinity and temperature regimes. Erosion and sediment transport are major processes in the Beaufort and Chukchi seas, and much of the shoreline has been classified as unstable (EIS Volume 3A, Section 1.5). Ice-rich shorelines have been reported to retreat 1 to 2 m/yr, while in some very unstable areas, as much as 10 m/yr of shoreline has been eroded. In addition, the Mackenzie River contributes approximately 15 million tonnes of sediment to the Beaufort Sea annually (Bornhold 1975). Consequently, future nearshore dredging activities would have to be assessed on a site-specific basis to determine their potential for altering adjacent shorelines in view of these concurrent erosional and depositional processes.

Changes to the sea bottom in offshore areas are unlikely to affect either circulation or wave patterns, although the active processes in this area remain poorly documented. Ice scouring is common at depths from 25 m to 50 m in the Beaufort Sea, and can naturally cause trenches up to 7 m deep, tens of metres wide and hundreds of metres long (EIS Volume 3A, Section 1.4). Studies of ice scouring have indicated that existing scours in the Beaufort Sea range from recent to very old, suggesting that the energy levels at these depths are low (EIS Volume 3A, Section 1.4). Therefore, dredge-created trenches and disposal sites may remain as distinct bottom features for extended periods, although they would likely be difficult to distinguish from natural ice scours or areas of bottom slumping. Another natural feature of the seabed which has recently been identified in the Beaufort Sea is the "pingo-like feature" (EIS Volume 3A, Sections 1.4.5 and 1.4.7). In the future, it may be difficult to distinguish these natural mounds from a dredge disposal site. Thus, while offshore dredging may locally change sea bottom contours, natural processes are also altering the configuration of the bottom on a more extensive scale.

2.4.2.6 Altered Sediment Composition

Dredging involves removal of surface and/or subsurface (depending on the type of dredge) sediment from one location, possibly exposing a different particle size substrate, and deposition in another area, potentially creating a different bottom type than previously existed at that site. For example, recent investigations by the petroleum industry have confirmed the existence of gravel beds overlain with 1 to 5 m of clay in many parts of the Beaufort Sea; in this case, dredges would remove the clay overburden to access the gravel. This is an important concern related to dredging where suitable benthic habitat is lost and recreated in both borrow and disposal areas. The implications of these physical changes in substrate composition to benthic invertebrates and fish are discussed in detail in Sections 2.4.5 and 2.4.9. Bottom substrates in the Beaufort/Chukchi Sea generally range from soft to firm clays and silts to medium-grained sand (EIS Volume 3A, Section 1.4.4). Areas near dredge disposal sites are likely to have a higher relative proportion of very fined-grained sediments due to the settlement of suspended fines. However, in nearshore areas, the active processes of ice scouring, erosion, slumping and sediment transport could cause dredged substrates to resemble adjacent areas relatively rapidly.

2.4.2.7 Changes in Ice Thickness and Breakup Patterns

It has been suggested that the construction of artificial islands and deepening of the water column at borrow areas may cause thicker winter ice cover and delay spring breakup. The effect of artificial islands on the ice regime was previously discussed in Section 2.1. This section briefly addresses the possibility of thicker ice forming in nearshore borrow areas as a result of dredging activities. In waters less than approximately 2.5 m deep, the landfast ice sheet freezes to the bottom. If dredging creates deeper areas at borrow sites, the ice sheet may not ground in these areas. This could have positive rather than negative biological effects since benthic species which could not otherwise colonize the area may survive throughout the winter. It seems unlikely that thicker ice in these localized areas could significantly affect breakup, since thicker areas of ice would be transported offshore by winds and currents along with other ice fragments during breakup. In addition, many of these areas could be sites of year-round activity by the petroleum industry, and any areas of persistent or thicker ice would normally be broken up by icebreakers during the spring.

2.4.3 Effects of Dredging on Marine Mammals

Although dredging activities *per se* may potentially affect marine mammals, most disturbance effects would be largely indistinguishable from effects of composite activities associated with dredging operations. The primary area of concern is that underwater noise caused by dredging activities may disturb whales. The effects of underwater noise on marine mammals are discussed in detail in Section 2.6.5, while other components of the dredging operation that may affect marine mammals are described in sections dealing with sewage, icebreaking, heated water and fuel spills.

In general, bowhead and white whales have been observed to tolerate stationary dredging activity, but occasionally react to supporting logistics traffic (e.g. tugs, barges, crew boats). For example, white whales have avoided dredging operations at distances of up to 4 km, but approached to within 400 m of active dredges on other occasions (Fraker 1977a, b, 1978). Bowhead whales have also been observed in the vicinity of artificial island construction activities where dredges were operating (Fraker et al. 1981). Between August 6 and 10, 1981, industry personnel reported at least 9 sightings of from 1 to 6 bowheads approaching, passing or circling the dredge BEAVER MACKENZIE during operations at Issungnak (Fraker et al. 1981). These authors reported a total of 20 bowheads within 5 km of the island on 4 days of survey effort in August 1981.

Ward (1981) reported that dredging activities in McKinley Bay during July and August 1981 had no detectable effect on the use of the area by seals. For example, ringed seals were regularly observed in the vicinity of the dredge, with some individuals as close as 50 m. The largest group of ringed seals observed near the dredging operation was a group of 5 present for several hours on August 26, while industry personnel reported 12 seals near the dredge about 5 days earlier. Densities of ringed seals recorded by industry personnel in the dredged channel during late August ranged from 1.3 to 21.4 seals/km^2. Bearded seals are not abundant in McKinley Bay, although Ward (1981) observed a single bearded seal on several occasions near the barge camp and industry personnel recorded another in the dredged channel during late August.

The removal or covering of benthic fauna in the dredged area or smothering of fauna in adjacent areas by settling of suspended solids may eliminate or cause a localized reduction of food sources for some species of marine mammals. For example, bearded seals feed on benthic fauna within the 100 m isobath, and bowhead whales also obtain an unknown portion of their diet from benthic habitats (Vibe 1950; Stirling et al. 1977; Wursig et al. 1981). However, since the extent of the dredging activity and concomitant destruction of benthic habitat will be spatially limited, and recolonization by benthic fauna would occur over a period of several years (Section 2.4.8), the degree of concern regarding potential effects of dredging on these species is considered NEGLIGIBLE.

Marine mammals that feed on pelagic fish and/or invertebrates may be affected by the highly localized increase in water turbidity and reduction in prey detectability within the dredge-created plume. Marine mammals that may be affected by temporary interference with foraging capabilities include ringed seals, white whales and bowhead whales (Stirling et al. 1977; Fraker et al. 1978; Wursig et al. 1981). In addition, bearded seals may be temporarily affected by turbidity plumes since they are also known to forage on pelagic species in some areas (Vibe 1950; Kosygin 1971). Although the interference with foraging capabilities of marine mammals by a dredge plume has not been documented, the significance of this potential effect would probably vary according to species, stage of life cycle, spatial and temporal extent of the plume and degree of turbidity. In most cases, however, the effects would be highly localized. White whales may be the least susceptible of the aforementioned species to temporary interference with foraging abilities, because they have a well-developed capacity for echolocation (Ford 1977) and are known to frequent areas of naturally high turbidity within the Mackenzie Delta (Fraker 1977a, 1978; Fraker and Fraker 1979, 1981). Consequently, the degree of concern regarding potential effects of dredge-created turbidity plumes on marine mammals in the Beaufort Sea region will likely be NEGLIGIBLE.

2.4.4 Effects of Dredging on Birds

Dredging activities in the southeastern Beaufort Sea may have several direct and indirect effects on birds in this region, including disturbance, localized loss of food sources through removal or smothering of benthic fauna in shallow waters (i.e. <40 m), temporary disruption of foraging capabilities within the dredge plume due to increased turbidity and reduced visibility of prey items, and attraction of some species to industrial sites by food items brought to the surface with the dredge spoil. Other potential effects of dredging on birds may result from common disturbances associated with the dredging operation such as airborne noise from aircraft and machinery (Section 2.7.3), presence of artificial structures (Section 2.1.4), icebreaking necessary for dredging during early spring or late fall (Section 2.3.3), discharge of heated cooling water (Section 4.2.5) and fuel spills (Section 5.2.5). In addition, there is potential for contamination in the vicinity of routine discharges such as sewage, drill muds, etc. if these materials are resuspended during dredging or discharged in areas frequented by birds.

Disturbance of birds by dredging activities in the Beaufort Sea region is expected to be of MINOR regional concern. Ward (1981) concluded that dredging activities in McKinley Bay during 1980 did not affect the abundance of birds using the bay. The numbers of diving ducks recorded were as great or greater than numbers recorded in years prior to dredging. In addition, geese migrating across McKinley Bay in late August did not react adversely to the operating dredge. Although some gulls and shorebirds were attracted to the island, presumably as a result of increased accessibility of invertebrates, the potential effects of dredging activities in McKinley Bay on local bird populations were considered minor (Ward 1981).

Loss of benthic food sources will occur in the immediate area of dredging and in adjacent areas where benthic fauna are smothered by settling of suspended materials. Nevertheless, the loss of food items will be relatively localized and recolonization by epibenthic invertebrates would occur over a period of several months after dredging has ceased (Section 2.4.8). The loss of benthic food sources may be a significant concern during certain periods of the year and in critical feeding areas. Birds known to forage in shallow, offshore benthic habitats during spring and/or fall migration, and therefore most likely to be affected by a reduction in the benthic community, include oldsquaws, eiders, loons, and alcids. Birds which forage in shallow nearshore benthic habitats include these species and several other species of diving ducks. Depending on the habitats affected by dredging programs, the degree of potential concern regarding loss of benthic food sources of birds would likely vary from NEGLIGIBLE to MINOR.

Increased turbidity and reduced visibility of prey items within the dredge plume may also interfere with the foraging capability of some species. Birds that forage under the water surface are most likely to be affected in this manner, and primarily include loons, diving ducks and alcids. Although the potential significance of reduced prey visibility would be dependent on the location, duration and timing of dredging activities, the short-term and localized nature of turbidity plumes suggests the degree of regional concern regarding this resource-activity interaction is probably NEGLIGIBLE.

As described earlier, some species of birds may be attracted to dredging operations to feed on invertebrates brought to the surface with the dredge spoil. For example, Thayer's and glaucous gulls were observed feeding at the edge of the turbidity plume created by a dredge operating in McKinley Bay (Ward 1981). They may have been attracted to the McKinley Bay dredging operation because of the availability of invertebrates and/or the same reasons they are attracted to sites of industrial or human activity throughout the world. The only potential concerns related to the attraction of these or other bird species to dredging operations are the increased probability for collisions with artificial structures during periods of poor visibility (Section 2.1.4) and contamination in the event of a minor fuel spill (Section 5.2.5).

2.4.5 Effects of Dredging on Fish

The potential effects of dredging activities on fish populations have been discussed in several overview documents and site-specific studies. These effects include direct physiological or pathological responses to high levels of suspended sediment, loss of benthic food sources, decreased feeding efficiency resulting from high turbidity, mortality resulting from entrainment in suction dredge equipment, and alterations in behaviour (including migration), as well as the direct loss of fish habitats (particularly rearing or spawning habitats) in localized areas of material removal or deposition (Poulin 1975; Morton 1977; Hirsch et al. 1978; ESL 1980). Although various disturbances associated with vessel and machinery movements, noise, and human presence may also contribute to adverse effects on fish populations, these are separately discussed elsewhere in this report (Sections 2.1.5, 2.6.6, 2.2.4).

2.4.5.1 Physiological and Pathological Effects

The direct effects of suspended sediments on fish have been primarily examined (largely in freshwater environments) in relation to logging or mining activities (Cordone and Kelly 1961; Phillips 1971; Ripley et al. 1978; Miles et al. 1979), with a few studies being directly applicable to dredging (Poulin 1975; Morton 1977; Hirsch et al. 1978; Pelletier and Wilson 1981). In general, adult fish appear to be tolerant of relatively high levels of suspended solids. In the laboratory, lethal or serious sublethal effects of suspended sediments on marine or estuarine fish usually only occur at concentrations above several thousand ppm or after chronic exposure to suspended solid levels of several hundred ppm. The seriousness of these effects varies with the nature of the sediment. For example, angular particles can cause significantly more damage to fish than smooth rounded particles, and contaminated sediments may also contribute to other acute or sublethal effects on fish (Sherk et al. 1974; Hirsch et al. 1978). Exposure of fish to high concentrations of inert suspended sediments in laboratory experiments usually results in the coating of fish gills by fine particles, or in larger particles lodging in the gill lamellae, blocking the passage of water through the gills and eventually affecting respiratory exchange. Asphyxiation occurs in severe cases of gill blockage, although various sublethal pathological conditions (e.g. hematological stress, abrasion of the body epithelium, reductions in metabolic reserves) may result in other instances. It is important to note, however, that most sublethal effects of suspended sediments on fish also appear to be reversible once fish are removed from water containing high sediment levels (Morton 1977).

Despite the results of these laboratory investigations, there have been no documented situations where high levels of suspended solids from dredging activities have contributed to significant fish mortality or observable physiological stress. At dredge sites monitored in the Beaufort Sea and elsewhere, concentrations of suspended solids in the immediate vicinity of the dredge normally reached only a few hundred ppm, and often these concentrations have been within the normal seasonal range of background levels (McDonald 1975).

Monitoring of fish near dredge sites has also revealed few serious effects of this activity. Many fish have been found to remain, apparently unaffected, near dredging sites examined in the Beaufort Sea (Byers and Kashino 1980) and at numerous locations in southern latitudes (Morton 1977; Hirsch et al. 1978). Juvenile and young-of-the-year whitefish were also present near dredging activities in Mackenzie Bay. It has been suggested that the lack of any observed effects of dredging on fish is probably the result of their ability to avoid severely disturbed areas, as well as the tolerance of estuarine fish (including arctic species) to wide fluctuations in suspended sediment (Poulin 1975; Hirsch et al. 1978; Craig and Haldorson 1980). It has also been suggested that demersal fish species are generally more tolerant of suspended particulates than pelagic species, while juveniles of all species are generally more sensitive to suspended sediments (Sherk et al. 1974).

Overall, most authors agree that the direct effects of suspended sediments on fish are usually not as important as other indirect effects such as the loss of important habitats or food sources in areas where dredging has occurred (Poulin 1975; Hirsch et al. 1978). Mortality or serious sublethal effects resulting from suspended sediments would only be anticipated when unusually large geographic areas were affected, when extreme long term or chronic levels of suspended solids resulted from dredging, or if contaminated sediments were resuspended in the water column. Consequently, the degree of concern related to potential effects of increased suspended sediment levels on regional fish populations is expected to be NEGLIGIBLE.

2.4.5.2 Loss of Food Sources and Reduced Feeding Efficiency

The indirect effects of food source depletion or impairment of fish feeding activities resulting from dredging have not been thoroughly investigated. Although the local abundance of invertebrate food sources for fish in the Beaufort Sea coastal environments (particularly epibenthic crustaceans) would likely be temporarily decreased in areas of borrow removal or spoil deposition (Section 2.4.8), the relatively small geographic areas affected, the likelihood of relatively rapid recolonization, and the observed abundance of prey in most coastal waters (EIS Volume 3A, Section 3.4) suggest that food losses resulting from dredging operations in the Beaufort Sea would not cause regionally significant effects on fish populations. In some instances, a temporary increase in the availability of benthic invertebrates has attracted fish to disposal areas after dredging was completed in temperate marine environments (Morton 1977).

Increased water turbidity may cause reduced feeding efficiency of opportunistic-feeding species of fish. Vinyard and O'Brian (1976) and O'Brian (1977) examined the feeding of both visual and non-visual feeders in turbid Kansas reservoirs. They reported that with visual feeders, such as the most common Beaufort Sea species, increased turbidity decreased the reaction distance of fish to all prey sizes. Therefore, increased turbidity may limit feeding efficiency in the immediate areas where dredging or spoil deposition occurs. However, in habitats where wide fluctuations in turbidity are normal, these effects would presumably be less important. For example, in nearshore Beaufort Sea habitats such as Simpson Lagoon and along the Tuktoyaktuk Peninsula, where most anadromous and many marine species concentrate, turbidities normally fluctuate between values of 1 and 146 NTU and 0 to 82 percent transmittance, respectively (McDonald and Cambers 1977a; Craig and Haldorson 1980). These naturally occurring turbidities often reach the levels which have been observed to result from dredging activities (Slaney 1975).

The degree of regional concern regarding loss of food sources and reduced feeding efficiency as a result of dredging operations is expected to vary with the type and location of habitats affected. Both of these potential indirect effects of dredging are considered of NEGLIGIBLE concern in offshore waters, despite the fact that reduced feeding efficiency may be more pronounced in areas normally outside the influence of the Mackenzie plume. However, loss of food sources could represent a MINOR area of potential concern in some nearshore habitats that support relatively high densities of anadromous and marine fish species.

2.4.5.3 Entrainment

Entrainment of fish by a suction dredge can result in mortality by direct physical trauma or by burial in discharged spoil. Dutta and Sookachoff (1975) assessed the survival of salmon smolts introduced into a suction dredge pipeline behind the cutter head and found that approximately 95 percent of the introduced fish were buried in the discharged sediment, and that most (70 percent) of the fish retrieved from the runoff water died within 96 hours.

The likelihood of fish being entrained by dredges has been less clearly demonstrated. Although fish may be attracted to subsurface structures and vessels (Section 2.1.5 and 2.3.4), some authors have assumed that fish capable of avoiding a water intake would do so (Tarbox and Spight 1979), while others have suggested that fish are attracted to a dredge inlet itself (Dutta and Sookachoff 1975).

Attempts to monitor the incidence of fish entrainment during actual suction dredging programs have had some success, but the small capacity of sampling equipment relative to the dredge output has resulted in such a small proportion of spoil being monitored that extrapolation of results has been extremely difficult and potentially misleading. For example, an observation of only 12 salmon fry in dredge spoil at a site on the Fraser River (British Columbia) led to a calculated daily mortality estimate of over 26,000 fish (Dutta and Sookachoff 1975). Recent attempts to determine the number of fish entrained by dredging in Tuktoyaktuk Harbour and McKinley Bay were not able to relate the number of fish collected to the total number which may have been affected, or the significance of the number of entrained fish to the local fish population (Pelletier and Wilson 1981). However, it was established that in McKinley Bay, cod (thought to be saffron cod) 7-10 cm in length and fourhorn sculpins 3-4 cm in length were entrained by the 90 cm suction dredge, while in Tuktoyaktuk Harbour, least ciscos 5-24 cm, Arctic ciscos 6-20 cm, inconnu 7-24 cm, fourhorn sculpins 5-10 cm, lake whitefish 11-16 cm, and one saffron cod 34 cm in length were entrained.

This demonstrates that in nearshore areas of the Beaufort region, a broad size range of several species, including those of importance to domestic fisheries, may be entrained by suction dredging equipment. The degree of concern related to entrainment of a few fish from widely dispersed populations is expected to be NEGLIGIBLE. However, this level of potential concern would be increased to MODERATE if dredging and subsequent entrainment occurred in some coastal areas where fish congregate for migration, spawning or feeding, since populations could be affected to a sufficient degree to result in a change in abundance and/or distribution which persists for more than one generation. It should also be noted, however, that most borrow sites will be located offshore and not in areas which are known to be critical fish habitat.

2.4.5.4 Migratory or Behavioural Effects

The potential that increased suspended solids levels and other disturbances (e.g. noise) associated with dredging may alter migratory patterns of fish (particularly anadromous species in the Beaufort Sea) has not been directly examined in any field investigations. However, there is some indirect evidence which suggests that dredging does not seriously interfere with normal behaviour patterns. For example, the capture of fish near active dredge sites in the Beaufort Sea and elsewhere suggest that fish may not avoid the entire area of disturbance (Poulin 1975; Morton 1977; Hirsch et al. 1978; Byers and Kashino 1980). As indicated earlier, the levels of suspended solids and turbidities near dredge sites are also often within the normal range of background concentrations (Poulin 1975; Hirsch et al. 1978; Thomas 1979). Consequently, it seems unlikely that dredging would interfere with fish migrations unless equipment or spoil deposits in shallow waters prevent or delay fish passage. In addition, there is also no information which indicates that such obstructions have or could occur in the Beaufort Sea.

2.4.5.5 Loss of Habitat

The loss of fish habitat through borrow removal and deposition or siltation would occur to some degree during most dredging operations. The potential effects of these disturbances would be most significant with demersal species and others which spawn, have incubating eggs, or rear in nearshore benthic environments. However, the magnitude of these effects would depend upon the location of the dredge, the size of the disturbed area, and the time of year. For example, the nearshore "edge" habitats on the lee side of barrier islands and along the mainland shores of the Beaufort Sea are extensively utilized by adults and juveniles of anadromous and marine species during the summer. It has been suggested that dredging in these areas (e.g. removal of barrier spits) could alter habitat conditions sufficiently to reduce fish utilization (Poulin 1975). Similarly, during periods of ice cover, some species such as sculpins, herring, flounders, or cod may spawn or have incubating eggs in coastal habitats which are sensitive to material removal or excessive siltation. Although the likelihood of such dredging-related effects on fish of the Beaufort Sea region remains unknown, the vulnerability of egg and larval stages of some fish populations would almost certainly necessitate the avoidance of potentially sensitive habitats during dredging programs. However, since most borrow sites for artificial island construction are located in offshore waters (EIS Volume 2) and limited nearshore dredging will be required, the degree of potential concern regarding loss of fish habitat is considered MINOR.

2.4.6 Effects of Dredging on Phytoplankton

The effects of dredging on local phytoplankton communities could include: 1) a decrease in photosynthesis and/or change in species composition due to light-related effects within the turbidity plume; 2) an increase in photosynthesis due to nutrient regeneration from the sediments; and, 3) a change in photosynthesis and/or species composition due to mixing of various layers of the water column.

The presence of a dredge-created turbidity plume could limit the amount of light available for photosynthesis in an area up to about 10 km^2 (Section 2.4.2.1). The effect of this light reduction on local primary production would depend on the amount of material in suspension, water circulation, background turbidity, and the rate at which the plume disappears after dredging. Grainger (1975) reported a considerable natural reduction in primary production within the plume of the Mackenzie River, and concluded that much of the primary production in the nearshore area is light-limited. Duval (1977) also found a reduction in photosynthetic production within a dredge-created turbidity plume compared to adjacent control areas. However, since such decreases in primary production would be very localized and would only persist until the dredging operation was complete, the degree of regional concern related to decreased productivity is considered NEGLIGIBLE.

Increased turbidity can also result in a slight shift in the spectral composition of incident light (Parsons and Takahashi 1973), and this may favour the growth of certain species. However, a turbidity plume would probably have to persist for several days to weeks to result in any locally significant change in community structure, particularly since phytoplankton would be continuously transported into and out of any areas with a different light regime.

Nutrients present in the sediments may be released to the water column during dredging, and this may stimulate primary production by shade-tolerant species during the dredging and light-limited forms once the turbidity plume is dispersed. This phenomenon has been observed in temperate waters (e.g., Morton 1977; Copeland and Dickens 1974, cited in Hirsch et al. 1978), although only in areas which had considerable silt and organic deposits. Nevertheless, since primary production in at least offshore areas of the Beaufort Sea outside the influence of the Mackenzie River plume may be nitrogen-limited (Grainger 1975), increased nutrient availability may favour the growth of some phytoplankton species. For example, Envirocon (1977) reported an increase in nitrate concentrations within a dredge-created turbidity plume, while both Envirocon (1977) and Duval (1977) measured increases in the standing stock of phytoplankton inside a turbidity plume compared to control sites. Although these observed increases in phytoplankton abundance may have been the result of nutrient enrichment, they may also reflect the presence of benthic micro-algae which could have been introduced to the water column during the dredging. Nutrient enrichment would be considered a positive effect of dredging, although it would be too brief and localized to be of any long-term or regional significance (Buchanan et al. 1977).

At some dredging sites in southern latitudes, resuspension of bottom material has been shown to release contaminants such as trace metals or hydrocarbons (Morton 1977), although Hirsch et al. (1978) concluded that sediment-bound metals are not particularly toxic. Furthermore, analysis of the trace metal and hydrocarbon content of sediments in the Beaufort Sea has indicated levels which are within the range of unpolluted areas elsewhere in the world (EIS Volume 3A; Section 1.6). Consequently, this is not considered a significant area of potential concern in relation to dredging activities within this region.

As indicated in Section 2.4.2.3, dredging operations may also locally affect the vertical stratification of the water column. The potential effects of this mixing on phytoplankton may include a possible decrease in productivity if the population is transported into deeper light-limited waters. Alternatively, an increase in production may result if nutrients from deeper layers are transported to the surface, especially if the phytoplankton in the upper portion of the water column are nutrient-limited. However, it should be emphasized that wind-induced mixing would produce similar distributions, and any resultant changes in the structure or productivity of phytoplankton communities would be regionally insignificant due to their short-term and localized nature.

Overall, the degree of concern regarding potential effects of dredging on regional phytoplankton populations is expected to be NEGLIGIBLE since both positive and negative effects would be localized and unlikely to alter regional productivity.

2.4.7 Effects of Dredging on Zooplankton

Dredging activity in the Beaufort Sea could affect the spatial distribution of some zooplankton species through modification of local salinity and temperature profiles and possibly through alteration of local circulation patterns (Reed 1975; Sullivan and Hancock 1973, cited in Morton 1977). Depending on the nature of the suspended sediment, localized mortality or sublethal effects may also result from the abrasive contact of particulates with sensitive membranes (Davis 1960; Davis and Hidu 1969; Krenkel et al. 1976; Conklin et al. 1980), while respiratory impairment can occur from coating of gills and other gas-exchange tissues (Robinson 1957). In addition, Sherk et al. (1974) demonstrated reduced ingestion of phytoplankton by copepods exposed to suspended sediments, and suggested that dilution of food with sediment particles was one of the potential effects of suspended solids on zooplankton.

In addition to these potential direct effects, several indirect effects of dredging on Beaufort Sea zooplankton communities are possible. Potential changes in phytoplankton species composition or abundance in the vicinity of turbidity plumes (Section 2.4.6) may reduce or increase the availability and nutritional quality of food for zooplankton. The resuspension of detrital matter may also provide additional food sources in the form of bacteria-coated particles and partly-decayed organic matter, while ingestion of mineral particles may trigger secondary behavioural responses in zooplankton (Syvitski and Lewis 1980). In addition, low visibility in the turbidity plume may enhance the survival of prey species while decreasing the success of predators (Reed 1975; Vinyard and O'Brian 1976).

Syvitski and Lewis (1980) suggested that filter-feeding zooplankton probably play an important role in increasing the sedimentation rate of small particles of clay and other minerals in the water column by compacting them into faecal pellets which settle out much faster than the original suspended matter. When present in relatively high densities, zooplankton could potentially increase the rate of removal of suspended sediments from the water column.

The direct and indirect effects of dredging on zooplankton populations under actual field conditions remain largely unknown. The results of some field studies have indicated either no measurable effects on zooplankton communities (Wright 1978), or effects on local zooplankton density and species composition that could not be attributed to dredging (Chesapeake Biological Laboratory 1970, cited in Morton 1977). On the other hand, Duval (1977) found that surface zooplankton (primarily the copepod Limnocalanus macrurus) occurring within a dredge-created plume in Mackenzie Bay were not only more numerous and significantly larger in size, but also had higher feeding rates than zooplankton from nearby control areas. However, the author emphasized that the observed differences in feeding and standing stock probably did not reflect a general increase in secondary production throughout the entire water column since only surface measurements were undertaken. Nevertheless, in view of the potential areas which could be affected by dredging activities, the regional significance of both potential positive and negative effects of this activity on zooplankton populations is expected to be NEGLIGIBLE.

2.4.8 Effects of Dredging on Benthic Communities

Potential effects of dredging and spoil disposal on benthic communities include: 1) physical disruption of the sea bottom, including removal or burial of benthos; 2) habitat alterations such as local changes in particle size distributions, bottom topography, water flow regimes, salinities and sedimentation patterns; 3) resuspension of sediments causing physiological stress; 4) altered dissolved oxygen and nutrient concentrations; and in some environments, 5) release of sediment-bound toxicants and bioaccumulation of some elements or compounds within local food webs (Hirsch *et al*. 1978).

2.4.8.1 Physical Disruption of the Sea Bottom

Physical disruption of the bottom during excavation and deposition of spoil materials is the most visible effect of dredging operations. Mortality of benthic invertebrates may occur at various stages of the operation, including entrainment and physical damage during excavation, and suffocation during transport or deposition of the dredged material. The potential effects of excavation operations on benthic communities would vary with the type of dredge used and the diversity and abundance of benthic populations in the area dredged, although sedentary or slow-moving infauna would usually be more seriously affected than the mobile epibenthic species. Immobile or attached organisms such as benthic algae, oysters, mussels, sponges and barnacles have been directly smothered by past dredging operations (Lunz 1938, 1942; Wilson 1950; Brehmer 1965; Carriker 1967; Saila *et al*. 1972; Rose 1973, all cited in Morton 1977). However, these forms are relatively rare in fine-grained sediment habitats such as those which characterize most of the Beaufort Sea.

Very mobile benthic species may be able to vacate areas of high sediment deposition or escape entrainment. Other invertebrates have various capabilities for migrating vertically through newly-deposited sediments such as dredge spoil, and to re-occupy similar positions relative to the sediment-water interface (Hirsch *et al*. 1978). The ability of Beaufort Sea invertebrates to migrate through dredge spoil has not been documented, although information is available for several non-arctic species. For example, mud crabs and amphipods which have morphological and physiological adaptations for crawling through sediments have been shown to migrate vertically through deposits up to 32 cm thick (Maurer *et al*. 1978, cited in Hirsch *et al*. 1978). On the other hand, Chang and Levings (1978) reported that cockles (*Clinocardium nuttalli*) and Dungeness crabs (*Cancer magister*) were unable to escape when quickly covered by 20 cm of sand, but were able to avoid burial in 10 cm of deposits. Saila *et al*. (1972, cited in Morton 1977) also found that the large polychaete *Nephtys incisa* and the small bivalve *Mulinia lateralis* surfaced through 21 cm of sediments, while a 1 cm-long tube-dwelling polychaete *Streblospio benedicti* surfaced through 6 cm of deposits. Shulenberger (1970) reported that the small (2-5 mm) clam species *Gemma gemma* was able to survive instantaneous burial under 23 cm of sand or 5.7 cm of silt. However, Turk and Risk (1981) examined the ability of the soft-shelled clam *Mya arenaria* to migrate through sediments, and concluded that even moderate rates and depths of deposition (1 to 3.5 cm of sediment at rates of 2 to 10 cm/month) could severely affect clam populations. In general, the ability of benthic invertebrates to vertically migrate through deposits appears to decrease with a corresponding decrease in the sediment particle size. The observed difference in the rate of movement through various substrates are apparently related to the greater amount of interstitial water in sand than in silt or clay. Consequently, benthic invertebrates in the Beaufort Sea may be most susceptible to burial when clay overburden is removed and discarded from offshore borrow sites.

Smaller invertebrates are often more vulnerable to burial when sediments are anoxic, since they are less able to reach the surface prior to suffocation. However, Nicol (1967) reported that at least some molluscs and polychaetes can temporarily reduce their respiratory requirements under low oxygen conditions. Overall, these studies suggest that the degree of mortality resulting from burial will depend on the particle size composition of the dredge spoil as well as the escape abilities of individual species.

In the Beaufort Sea region, sand is the most common borrow material used for artificial island construction, but is usually located in deposits which are covered by a 3-5 m thick overburden of clay and silt which must be removed to expose the sand layer. This overburden supports various populations of benthic invertebrates and in some cases microalgae, although only the upper 5 to 15 cm of substrate represents actual benthic habitat. Most stationary dredging programs also attempt to remove the maximum volume of material from a relatively small area by excavating deeper layers which are naturally devoid of organisms, and this would tend to minimize the total amount of benthic habitat disturbed by dredging activities.

Most of the potential borrow sites identified to date in the Canadian Beaufort Sea are located offshore in water depths between 15 and 40 m. This depth zone approximately coincides with the Transition Zone (15-30 m) which Wacasey (1975) characterized as receiving intense scouring by ice keels and having fluctuating water temperatures and salinities. On the basis of data provided by this author, an average benthic biomass of 5 g/m^2 (dry wt.) may be expected in this region, with the polychaetes *Artacama proboscidea* and *Trochochaeta carica* and the clam *Portlandia arctica* representing the dominant species.

In the Beaufort Sea, the mobile epibenthic fauna such as isopods, mysids and amphipods may be able to avoid burial by sediments which settle out of turbidity plumes, although they would remain relatively susceptible to entrainment by dredging equipment. Infauna in this region are dominated by polychaetes and molluscs (EIS Volume 3A; Section 3.5) which generally have low mobility and will be less likely to avoid heavy sedimentation or escape entrainment. Benthic microalgal cells which colonize the surface of the seafloor in certain areas may also be ruptured or resuspended in the water column during dredging programs.

2.4.8.2 Benthic Habitat Alterations

Post-dredging alterations in water depths, particle size composition of exposed sediments, food availability and possibly temperature and salinity regimes may occur in both excavation and dredge spoil deposition areas. These habitat changes can result in differential rates of recolonization, survival and reproduction by various invertebrate populations, and therefore lead to shifts in the interspecific relationships within the benthic community. Post-dredging studies completed in temperate marine waters indicate that numbers and species of organisms in dredged and spoil deposit areas can be markedly different from those in adjacent undisturbed areas (Morton 1977). Excavated areas also can contain different dominant species than spoil deposition areas, which is evidence that different species have variable abilities to adapt to specific environmental disturbances (Copeland and Dickens 1974; Kaplan *et al.* 1974).

Carricker (1967) reviewed many investigations dealing with the specific nature of organism-sediment adaptations and interactions, as well as the sensitivities of fauna to other physical habitat parameters such as temperature, salinity and currents, which could be affected by dredging. In general, greater densities of estuarine macrobenthos can be supported in "muddy sand" habitats with sufficient organic matter than in either clean, coarse unstable sands and gravels or soft slurry muds. These investigations often emphasize the important role of the sediment as a source of food, as well as the rather close relationship between infaunal feeding habits, gross organic content and the mechanical nature of the sediments.

Altered abilities of sediments to support specific invertebrate populations due to dredging-induced changes in interstitial water content or particle size composition have been the subject of extensive investigation. Sykes and Hall (1970) reported fewer numbers and less species diversity among molluscs in the silt/clay sediments of dredged canals than in the sand and shell sediments in the undisturbed bottom of a Florida bay, although other factors such as depth, currents and food availability may have been partially responsible for this differential distribution. Turk *et al.* (1980) found that sediment introduced behind a causeway in an Nova Scotia intertidal mudflat had not compacted significantly over a 10 year period, and that high water:mud ratios likely prevented populations of the clam *Macoma balthica* and the tube-dwelling amphipod *Corophium volutator* from recolonizing the area. On the other hand, *Heteromastus filiformis*, a deposit-feeding polychaete which has a high body surface/mass ratio, was unusually abundant compared to nearby unaltered mudflats. Other reviews on the effects of dredging have emphasized the non-cohesive nature of recently-dredged silt and clay spoil, particularly where suction methods have been used to fluidize the sediments for transport (Morton 1977; Hirsch *et al.* 1978).

In the Beaufort Sea region, excavation of subsea glory holes, harbours and borrow pits would create localized basins or depressions in the seafloor. These new habitats may also be characterized by a different oceanographic regime including higher salinities, colder temperatures and, in some cases, reduced current velocities due to a depression effect. The potential for creation of these new habitats may be greatest within the Mackenzie River estuary due to the marked vertical temperature and salinity stratification which exists in this area throughout the year (EIS Volume 3A, Section 1.3). With time, the relatively high sediment load of the Mackenzie River and bottom-scouring currents during wind storms could eventually fill some of these newly-created basin habitats with settled fines, depending on their location. However, ice gouges in deeper waters having very low hydrodynamic energy levels have been estimated to be 100 years old and up to 5.5 m deep (Reimnitz and Barnes 1974). Sedimentation rates have been estimated at between 0.3 and 1.0 m/1000 yr off the Mackenzie Delta in 60 and 350 m depths, respectively (Shearer 1972, cited in Barnes and Reimnitz 1974). Consequently, glory holes and borrow pits dredged in offshore areas are likely to remain as relatively permanent alterations to the seafloor topography.

It should also be emphasized, however, that most proposed offshore dredging and island construction in the Beaufort Sea will be located within the benthic Transition Zone (Wacasey 1975) which is a dynamic zone of frequent ice scouring (0.1 to 0.4 scours/km/yr) and fluctuating oceanographic conditions. Ice keels mechanically disturb the seafloor, producing "a mosaic of unscoured areas, recently scoured areas, and scoured areas in various stages of [benthic] recovery" (Wacasey 1975). For example, divers observed a wide diversity of benthic species in the Alaskan Beaufort Sea, but only a small number of ice gouges were populated with a community comparable in maturity to that of the undisturbed bottom (Reimnitz and Barnes 1974). Therefore, it is likely that the benthic community of the proposed offshore production zone contains at least some species adapted to recolonizing recently disturbed sediments, either by planktonic dispersal of larvae or direct relocation of adults.

2.4.8.3 Suspended Sediments and Turbidity

As indicated earlier in Section 2.4.2.1, turbidity plumes and relatively high suspended sediment levels are a characteristic of virtually all dredging programs. However, benthic organisms normally associated with mud or silt substrates are highly tolerant of most suspended sediment conditions created in the water column by dredging and construction activities (Hirsch et al. 1978). The prevalence of mud and silt substrates in the Beaufort Sea region and the existence of relatively high background turbidities in areas influenced by the Mackenzie River plume suggest that indigenous benthic forms would be reasonably well adapted to temporary suspended sediment increases created by dredging. Turbidity levels created during dredging may also be within the range of natural background variability in the Beaufort Sea (Thomas 1979). However, other factors such as the location, season, frequency and duration of dredge-created turbidity plumes would also influence the effect of suspended sediments on benthic flora and fauna.

The most serious form of turbidity is the condition known as "fluid mud" or fluff, which can blanket the bottom and does not physically allow the upward movement of macrofauna to the clearer overlying water (Hirsch et al. 1978). Another possible short-term effect of increased siltation on filter-feeding infaunal species is a reduction in their feeding efficiency due to dilution of food cells by sediment particles or physical clogging of filtration appendages. Oysters and clams in temperate regions have been shown to reduce their average water pumping rates by up to 94 percent when exposed to high concentrations (3-4 g/L) of suspended silt (Loosanoff and Tommers 1948; Loosanoff 1961). Suspended clay also greatly affects the normal development of clam (*Venus mercenaria*) eggs and larvae, and in some cases, causes mortality by clogging of the digestive tract (Davis 1960).

Sublethal effects of high suspended sediment concentrations on benthic flora and fauna of the Beaufort Sea have not been investigated, although high turbidity levels in shallow waters where light normally reaches the bottom could cause short-term reductions in the rate of primary production by benthic microalgae.

2.4.8.4 Water Quality Changes

Undisturbed sediments are typically characterized by a vertical gradient from oxidized surface deposits down to increasingly reduced sediments in deeper layers. The latter create a chemical oxygen demand when they are exposed to the overlying water body and undergo oxidation. Sediment grain size and burrowing activity by benthic invertebrates have major roles in determining the amount and circulation of interstitial water within the sediments, and hence the depth of the anaerobic layer. Hydrogen sulfide also creates an oxygen demand and is toxic to marine invertebrates at certain concentrations (Theede et al. 1969). The Environmental Studies Board (1972) suggests that concentrations of sulfide equal to or exceeding 0.01 mg/L constitute a hazard in the marine environment between pH 6.5 to 8.5. In addition, a biological oxygen demand may be created if the sediment suspension contains high concentrations of nutrients which stimulate active bacterial metabolism requiring oxygen (Section 2.4.10). Infaunal benthic invertebrates may be relatively well adapted to low oxygen concentrations due to their burrowing behaviour, whereas epibenthic forms are expected to be generally more sensitive to anaerobic conditions. The primary concerns related to low oxygen levels associated with dredging are: 1) the decreased probability of survival of benthic organisms if smothered by anaerobic spoil, and 2) the creation of localized anoxic bottom waters.

Nutrient concentrations are usually higher in fine, highly organic sediments than in overlying waters, due largely to microbial decomposition of allochthonous plant debris. For example, Windom (1973) reported significant releases of ammonia during dredging operations within five warm temperate estuaries. However, the potential stimulation of local primary production by ammonia and other nutrients released during dredging may be offset by the shading effect associated with increased turbidity (Morton 1977). In addition, surface sediments of the Beaufort Sea shelf are highly oxidized and generally contain less organic matter than sediments at similar depths elsewhere in tropic and temperate zones (Carsola 1954; Naidu and Mowatt 1974). It is possible that the year-round low temperatures near the seafloor limit microbial utilization and transformation of organic matter into nutrients.

2.4.8.5 Release of Sediment-Bound Toxicants

Although the release of toxic materials such as trace metals during dredging has been identified as a potential area of concern in some polluted regions in temperate and tropical latitudes (Keeley and Engler 1974; Morton 1977), this is not a significant area of concern at present in the Beaufort Sea region. Sediments dredged in the Beaufort Sea would not contain the magnitude or diversity of contaminants found in sediments adjacent to large industrial and population centers. For example, Thomas (1979) compared the concentrations of 10 different trace metals measured in sediments collected from McKinley Bay, Tuktoyaktuk Channel, and the Mackenzie Delta with levels reported for uncontaminated coastal sediments elsewhere in the world, and found that trace metal levels in the Beaufort Sea were well within the range of values which are considered typical of unpolluted sediments.

The only potential area of concern would be the release of any toxic materials (particularly H_2S) generated during the degradation of untreated sewage which presently enters Tuktoyaktuk Harbour from the hamlet of Tuktoyaktuk. Maintenance dredging in this harbour could release H_2S and other unknown compounds until such time as a sewage treatment plant is installed for this community.

2.4.8.6 Benthic Recolonization

A environmental concern in many areas has been the period which may be required for recolonization of bottom habitats where dredging or spoil deposition has occurred. The suitability of newly-deposited or excavated substrate for the attraction and support of benthic fauna is dependent on a large number of interrelated factors. Recolonization of borrow sites and spoil deposits in the Beaufort Sea and other non-polluted waters is reasonably well-documented. Information describing recolonization in the Beaufort Sea is reviewed in the following subsection, while the major factors affecting this process are summarized below.

The general factors affecting the re-establishment of benthic communities on an azoic (uninhabited) bottom were discussed by McCave (1974) and include:

1. Appropriate physical properties of sediments, particularly sediment grain size, packing or density, degree of flocculation, interstitial water content and permeability, and stability/resistance to erosion;

2. Acceptable chemical properties of sediments including levels of oxygen and decomposition gases, concentrations of organic material and nutrients, pH, redox potential, and acceptable levels of toxic compounds or elements;

3. Certain biological conditions including dispersal capabilities for adults and larvae, relatively low predation pressures, and feeding opportunities; and

4. Suitable hydrographic conditions in the overlying water column including currents, oxygen content, temperature, light and salinity.

Oliver et al. (1977) conducted a four year field evaluation of spatial and temporal variations in recolonization of several dredging sites in Monterey Bay. Their study showed that marine organisms recolonizing dredged material were not the same species as the original residents, and these new colonizers were generally opportunistic species whose environmental requirements were adaptable enough to permit occupation of the disturbed areas. Re-establishment of the original community at the various locations followed no predictable sequence, although some original species were noted within several months of dredging and complete recovery was approached within one year. Vertical escape of existing organisms through the dredged spoil was not considered a factor in the recolonization process, and Oliver et al. (1977) suggested that the adjacent undisturbed communities were the primary source of replacement organisms through either adult migration or larval dispersal and settlement.

Other authors have reported generally similar findings in different temperate regions. However, recolonization patterns and rates are affected by a host of site-specific variables, particularly the size of the dredging-affected area in relation to the nature and area of the surrounding coastal habitat. For example, Leathem et al. (1973) reported that the naturally low density of benthos (<100 individuals/m^2) before dredging in Delaware Bay hampered detection of post-dredging changes in density. Successful recolonization in temperate regions has occurred in periods ranging from as little as 2 weeks in Coos Bay, Oregon (Slotta et al. 1973) to 18 months in Chesapeake Bay (Pfitzenmeyer 1970; cited in Morton 1977), while Turk et al. (1980) reported that a benthic community in Nova Scotia had not returned to normal 10 years after induced siltation.

The colonization of disturbed areas by sedentary infauna, sessile epifauna, and many benthic plants usually involves planktonic dispersal of juveniles, while re-establishment of mobile epifauna is usually rapid and may occur within months of dredging. Some larvae of benthic fauna exhibit various degrees of substrate discrimination, and do not settle or develop on unfavourable substrates. Due to these differences in recolonization modes and rates, the former two classes of benthic fauna represent better indicators of successful recolonization of a disturbed substrate.

2.4.8.7 Beaufort Sea Dredging and Benthic Recolonization Studies

Recolonization of borrow pits and spoil deposits in the Beaufort Sea is reasonably well-documented, and to date, environmental studies have indicated relatively few potential concerns related to the interaction between benthic organisms and dredging in this region. Wacasey (1975) described the infaunal benthos in four depth zones within the southern Beaufort Sea, three of which overlap the depth range where dredging and artificial island construction is expected during future hydrocarbon production activities. These are the Estuarine (0-15 m), Transitional (15-30 m) and Marine (30-200 m) zones, where there is a general trend of increasing numbers of species, abundance and biomass with increased depth. The early artificial islands off the Mackenzie Delta were constructed in shallow waters (<5 m) where natural densities of infaunal benthic invertebrates were low, probably due to the low salinities off the river mouth and annual ice scour in this area (Slaney 1973, 1974c; Bengeyfield 1976). In recent years, islands have been constructed in deeper waters, and this has required a concomitant increase in the basal area of the islands and hence more borrow material. These islands have also been situated in a richer benthic zone (Wacasey 1975; Envirocon 1977; Beak 1981). Of the conventional artificial islands having at least a 1:15 slope and wide sacrificial beach, Issungnak 0-61 is located in deepest water at 19 m. However, the recently completed Tarsiut island, built on a 1:5 slope with concrete caissons extending up through and above the water surface, is the deepest island (23 m) constructed in the Beaufort region to date.

Beak Consultants Ltd. (1981) described the distribution of benthic invertebrates adjacent to the Issungnak artificial island, and concluded that post-dredging alterations in sediment particle size were primarily limited to the area encompassed by the 0.53 km^2 island base and the two borrow pits. Sand sediments, located at the outer edge of the island base 300 m away from the shoreline, formed a transition zone with some mixture of sand and silt-clay components, while areas 900-1800 m from the site had natural silt-clay substrates. The effects of construction of Issungnak on benthic fauna did not extend far beyond the underwater slopes of the island base or the principal borrow pits. The density and number of species found during post-construction studies were inversely related to sediment particle size. Recolonization of the construction zone began immediately, with 7 to 23 species per m^2 occurring in the spoil disposal zone compared to 29 to 47 species per m^2 in the background zone. Complete denudation of the substrate was not observed at any station. Species colonizing the construction zone included some species from the background zone, as well as three polychaetes found only in the construction zone, which probably colonized the area through widespread dispersal of planktonic larvae (Beak Consultants Ltd. 1981).

The Isserk F-27 artificial island was built in the summer of 1977 in 12.8 m of water, and a baseline study was conducted during its construction by Envirocon Ltd. (1977). As in the case of Issungnak, sand was distributed adjacent to the island base, and this was undoubtedly dredged or barged material used during construction since the natural sediments in the area were primarily silt and clay. Erosion of the island during a severe storm in late August may have also caused the dispersal of sand to areas beyond the island base. The species diversity and biomass of benthic invertebrates did not show any statistically significant trends related to island construction, although the abundance of benthos tended to increase with increasing distance away from the island. The authors concluded, however, that there were no apparent adverse effects of construction on the bottom fauna surrounding the Isserk site. Several earlier studies near other artificial islands in the region had resulted in similar conclusions (Slaney 1973, 1974c; Bengeyfield 1976). Envirocon (1977) calculated that the Isserk island base occupied approximately 0.2 km^2 of benthic habitat, while the adjacent borrow area extended over an additional 0.3 km^2. An estimated 6000 kg (wet weight) of bottom fauna were lost during the construction of Isserk, based on a mean biomass of 12 g/m^2 (wet wt) in this part of the Beaufort Sea.

Olmsted (1977a) reported that the indirect sedimentation resulting from the construction of Arnak L-30 in 7 m of water did not significantly alter either infaunal biomass or abundance at two stations 400 and 500 m from the site when compared with a control station, although the extent to which these two stations had been affected by the turbidity plume was not known. The base of Arnak L-30 covered about 0.11 km^2 of substrate, and the adjacent borrow pits accounted for an additional loss of 0.087 km^2 of benthic habitat. Olmsted (1977a) calculated that about 1500 kg (wet weight) of benthos were lost during construction of Arnak L-30, using average biomass figures determined for adjacent areas.

Overall, the results of studies conducted following artificial island construction in the Beaufort Sea indicate that: (1) direct and total loss of benthic fauna has been primarily limited to the basal area of exploration islands and adjacent borrow pits; (2) changes in the particle size composition of sediments have only been evident within 1 km of construction sites; (3) the diversity and abundance of benthic fauna may or may not decrease in areas immediately adjacent to island bases; and, (4) recolonization of disturbed areas begins relatively quickly. It should also be emphasized that most exploration islands constructed to date have been sacrificial beach designs with very gradual slopes, whereas many of the future exploration and production platforms will employ concrete or steel caissons placed on subsea berms with much steeper slopes. As a result, the borrow requirements and basal areas of islands, and therefore the direct loss of benthic fauna, will each be minimized.

2.4.8.8 Summary of Potential Concerns

Future offshore and coastal dredging operations in the Beaufort Sea will be necessary for artificial island and causeway construction, harbour maintenance, dredging of access channels, construction of glory holes and subsea pipeline ditching. The primary effects of these activities on benthic communities will be: (1) direct mortality caused by physical burial of organisms under spoil deposits and island bases, and (2) loss of fauna in borrow pits and other excavation sites. Less severe effects of dredging on benthic flora and fauna may be associated with turbidity plumes and localized changes in water quality, although these sources of potential disturbance would be relatively short-term.

The magnitude of the potential and probable effects of dredging on benthic communities would be generally proportional to the size of dredged and spoil deposition areas in relation to the total unaffected area which supports similar benthic populations. However, this proportional relationship would not apply in some localized inshore areas along the Beaufort Sea coastline, which are unusually productive for this region due to specific combinations of oceanographic conditions. Some of these productive areas include the lee of Pelly Island spit and Hooper Island (Slaney 1973), Simpson Lagoon and similar barrier island environments (Griffiths and Dillinger 1980) and Mason Bay (Wacasey 1975). Since these areas may be considered sensitive habitats in terms of their annual contribution to the benthic biomass of the Beaufort Sea region, dredging operations which affect these and other productive habitats would likely be of far greater regional concern, particularly when they also support important fish and bird populations. However, the only dredging programs which are presently anticipated in inshore areas are the continued maintenance dredging in Tuktoyaktuk Harbour and McKinley Bay, the continued use of the Tuft Point borrow site, and dredging requirements associated with construction of a breakwater at King Point and the subsea pipeline system near North Head on Richards Island (EIS Volume 2).

According to the criteria described in Section 1 of this document, the degree of regional concern regarding the effects of dredging on benthic commununities in the Beaufort Sea would range from NEGLIGIBLE, in many nearshore areas where benthic biomass is low, to MINOR in offshore waters characterized by a more diverse and abundant infauna and epifauna, to MODERATE in the isolated high productivity areas previously described. The degree of potential concern is higher in the latter two areas, not only because of the higher benthic standing crop in these environments, but also because other indirect effects of dredging may be of greater significance in waters which do not normally contain relatively high suspended sediments throughout much of the year. The potential concern related to indirect effects of dredging may also be greater during the winter when background turbidities are relatively low, and benthic larval stages, which may be more sensitive than adults to suspended fines (Davis 1960; Loosanoff 1961), are present in the water column.

2.4.9 Effects of Dredging on Epontic Communities

Since the technical feasibility of efficient dredging in ice covered waters is still in the conceptual stage, there are no data describing potential physical and chemical effects of this activity on the under-ice surface and water column, or the biological effects of dredging on epontic organisms. However, there is little reason to expect that physical and chemical changes within the water column observed during open water dredging operations would be substantially different during winter operations, although background turbidity/suspended sediment levels in the nearshore Beaufort Sea would be much lower due to the lower discharge of the Mackenzie River. The effects of winter dredging on epontic communities would likely be associated with potential changes in dissolved oxygen and nutrient concentrations, increased concentrations of suspended sediment, alterations of the temperature and salinity structure of the water column, and icebreaking. The potential effects of icebreaking on epontic communities were previously discussed in Section 2.3.8, while the following sections address potential effects resulting from other physical and chemical disturbances.

2.4.9.1 Increased Suspended Sediment

Mangarella et al. (1979) studied under-ice water circulation and chemistry in the Beaufort Sea near Prudhoe Bay, and utilized these measurements of background water conditions to model the dispersion and settling characteristics of an under-water discharge containing 182 ppm of suspended solids. Their calculations indicate that rough under-ice surfaces caused faster sedimentation than smooth under-ice surfaces. Under smooth ice, the sediment plume extended approximately 650 m and eventually affected about 20 acres (8.1 ha). Under rough ice, the horizontal extent of the plume was reduced to approximately 450 m, but since the lateral spread of the plume was greater, the total area affected was about equivalent to that calculated under smooth ice (approximately 8.1 ha). These are the only available data describing the possible extent of increased turbidity and suspended solid levels under an ice cover. During the open-water season, dredge-created turbidity plumes have been occasionally masked by high background suspended sediment levels and turbidity. However, under-ice water sampling in the Arctic indicates that suspended sediment concentrations and turbidity are very low, and therefore turbidity could have greater effects on marine biota during this period. For example, Mangarella et al. (1979) measured water transmissibility at stations up to a depth of 6.8 m near Prudhoe Bay and found that the average levels approached the in-air calibration standard for the instrument. Waters off the Mackenzie Delta, although relatively clear compared to high summer turbidities, are still influenced by Mackenzie River outflow during the winter and have slightly higher turbidity than areas not affected by river discharge. Slaney (1974c) measured suspended solids and turbidities at 16 stations off the Mackenzie Delta during March and April, 1974 and reported ranges of 1.6 to 104 ppm and 0.7 ppm to 22.0 ppm, respectively.

There is no available literature describing the effects of increased under-ice turbidities and suspended solids on epontic communities. Reduced transmissibility of the water column is unlikely to directly affect epontic flora, since particulate material would progressively settle with increased distance from the dredging site. However, if suspended sediments adhered to the under-ice surface and became trapped in growing ice in the immediate vicinity of such operations, light penetration could be reduced and this may decrease primary productivity of epontic flora. It is unlikely that light could be blocked by ice-trapped sediment over extensive areas since modelling studies indicate that suspended sediment settles relatively rapidly before it could be encapsuled by growing ice. Mangarella et al. (1979) estimated that in 1 hr and over a 0.7 acre (0.28 ha) area, concentrations of suspended solids were reduced from 16 ppm to 1 ppm. In addition, the degree of concern regarding potential effects of reduced light intensity on epontic flora are likely to be NEGLIGIBLE because this community is already adapted to low light levels, as well as fluctuating light intensities associated with varying snow and ice thicknesses (EIS Volume 3A, Section 3.5.8).

2.4.9.2 Alteration of the Temperature and Salinity Structure

During late winter and early spring, the water column under the ice cover is relatively homogeneous with respect to temperature and salinity, and offshore areas of the Beaufort Sea are generally characterized by the presence of oceanic water. However, nearshore areas with significant river inflow such as off the Colville River and the Mackenzie Delta may still have a layer of fresh water beneath the ice. Since the water column is not stratified in most areas during this period, mixing as a result of dredging would have little effect on the temperature and salinity structure. In fact, the homogeneous temperature and salinity structure would favour the more rapid settlement of suspended solids since there are no marked density gradients which would trap and horizontally transport mid-water or bottom plumes (Section 2.4.2.1). The more rapid sinking of suspended particulates would also decrease the period of exposure of epontic organisms to turbid water.

As the ice melts in spring and early summer, water beneath the ice warms slightly and salinities decrease due to the inflow of fresh melt water. Turbulence created by dredging activities at this time of year could break down this stratification for short periods in areas adjacent to the operation. Saline water from near the bottom could be a source of nutrients which would be in short supply in ice melt water due to desaltation of the ice sheet (Meguro et al. 1967). The epontic organisms should rapidly adjust to the increased salinity associated with dredging because this community begins to develop earlier during spring when salinities are relatively homogenous and is likely adapted to changes in salinity. Temperatures under the ice could be decreased slightly due to mixing with colder bottom waters during dredging. The potential effects of changes in temperature and salinity on regional populations of under-ice organisms are expected to be NEGLIGIBLE, but may actually be positive due to the concurrent upward transport of nutrients. On the other hand, icebreaking activities in the vicinity of the dredging and spoil deposition sites would probably result in localized adverse effects on epontic flora and fauna. These potential effects were described earlier in Section 2.3.8.

2.4.9.3 Changes in Nutrient and Dissolved Oxygen Concentrations

Nutrient enrichment as a result of dredging activities in the Beaufort Sea has been limited to slight increases in total organic carbon and nitrate-nitrogen concentrations (Section 2.4.2.2). Phosphate and silicates, which are also important nutrients for diatom growth, have not been reported to increase during open water dredging operations in this region. Meguro et al. (1967) suggest that the major mechanism for nutrient supply to the growing epontic flora is from nutrient salts which penetrate to the bottom ice surface with the brine drainage as the ice begins to melt (EIS Volume 3A, Section 3.5.3). Bacterial conversion of organic compounds and upward transport of nutrients from the seawater under the ice are two other minor pathways of nutrient supply to this community. However, slow bacterial metabolism at temperatures less than 0°C and the fact that nutrients in the sea water would not be accessible to the most densely populated diatom zone (within the ice crystals) limit the nutrient input from the latter two sources. Although there are no available data describing the effects of nutrient enrichment on the growth of epontic flora, any increase in the rate of growth would likely be very limited in space and time, and thus the overall significance in terms of regional epontic communities would be considered NEGLIGIBLE.

Short-term reductions in dissolved oxygen concentrations have been reported several hundred metres from dredge outfalls in the Beaufort Sea during the open water season (Section 2.4.2.2). However, during winter when water temperatures under the ice are less than 0°C, biological oxygen demand would likely be reduced due to lower levels of bacterial activity. Measurements of under-ice dissolved oxygen concentrations off Prudhoe Bay and in Mackenzie Bay indicate that oxygen levels may be below saturation for the ambient salinity and temperature regime, but probably do not restrict biological activity (Slaney 1974c; Mangarella et al. 1979). Although slight reductions in dissolved oxygen could result from dredging activity, icebreaking in the vicinity of the dredging operation would likely be the more significant source of effects of this activity on epontic fauna. Consequently, the degree of concern regarding the potential effects of dredging-related oxygen reductions on regional epontic communities is considered NEGLIGIBLE.

2.4.10 Effects of Dredging on Micro-organisms

The effects of dredging on micro-organisms have not been investigated during previous dredging studies in the Beaufort Sea. Significant increases in the number and variety of bacteria in the water column, and a subsequent decrease in dissolved oxygen due to biochemical oxygen demand and photo-oxidation of reduced sediments have been reported at dredge sites in temperate latitudes (Morton 1977). These effects have usually been associated with the suspension of sediments with high organic content and/or sewage and waste contaminated sediments, particularly at dredge sites with poor water circulation. There is some indirect evidence of potentially increased numbers of bacteria in turbidity plumes at dredging sites in McKinley Bay and Tuft Point (Section 2.4.2.2), where minor reductions in dissolved oxygen concentrations were reported (McDonald and Cambers 1977a; Thomas 1979). However, dissolved oxygen concentrations have not been affected during other dredging and artificial island construction operations in this region. In general, Beaufort Sea sediments do not have a high organic content and are relatively free of contaminated wastes (Section 2.4.2.2), and as a result, the degree of concern associated with increases in numbers or activity of marine bacteria due to dredging is likely to be NEGLIGIBLE.

2.4.11 Summary of Concerns Related to Dredging

The most significant biological concerns associated with dredging activity in the Beaufort Sea are related to losses of benthic infauna and epifauna in areas where these fauna contribute substantially to the biomass and productivity of isolated rich coastal areas such as Pelly Island spit, Hooper Island, Simpson Lagoon and Mason Bay. The degree of potential concern regarding the effects of dredging on benthic fauna will be highly dependent on the location and size of these operations, particularly where the aforementioned sensitive habitats could be affected, or where dredging occurs in more offshore waters where the biomass of benthos is relatively high. In a similar manner, the degree of regional concern may be considered MODERATE when the benthic community is important as a food source for local breeding bird populations and the juvenile stages of anadromous fish.

As indicated in Table 2.4-2, the degree of potential concern regarding the effects of dredging on the majority of resources in the Beaufort Sea region is considered NEGLIGIBLE. This is at least in part due to the localized and short-term nature of most physical and chemical disturbances associated with dredging operations, but also because of the extensive research which has examined effects of dredging in the Beaufort Sea and elsewhere. In addition to the moderate concerns associated with potential effects on some benthic communities, there are also MINOR concerns related to the possible effects on some bird species if dredging operations were located near 'critical' feeding areas during staging and brood rearing, as well as dredging operations located in nursery areas for some marine and anadromous fish species.

TABLE 2.4-2

SUMMARY OF POTENTIAL CONCERNS RELATED TO DREDGING ACTIVITIES IN THE BEAUFORT SEA REGION

Environmental Component or Resource	Potential or Probable Effects	Degree of Potential Regional Concern
Water Quality	Short-term turbidity plumes extending up to 5 km downstream of dredging activities; potential short-term decreases in dissolved oxygen concentrations and increases in nutrient levels	See specific resources for biological concerns
Water Column Structure	Temporary mixing of water column layers of different density, reducing degree of temperature and salinity stratification; transport of cold, saline waters into habitats normally characterized by relatively warm temperatures and low salinities	See specific resources for biological concerns
Substrate Character	Localized alteration of bottom contours and sediment composition.	See specific resources for biological concerns
Ice Regime	Localized thickening of ice in nearshore areas.	See specific resources for biological concerns
Bowhead Whales	Avoidance or attraction responses depending on degree and type of activity (see Underwater Sound, Section 2.6.2); localized loss of benthic food sources, and temporary interference with foraging abilities in turbidity plumes	NEGLIGIBLE for dredging *per se*
White Whales	Avoidance or attraction responses depending on degree and type of activity (see Underwater Sound, Section 2.6.2); possible temporary interference with foraging capabilities within turbidity plumes	NEGLIGIBLE for dredging *per se*

TABLE 2.4-2 (Page 2)

Environmental Component or Resource	Potential or Probable Effects	Degree of Potential Regional Concern
Ringed seal	Potential avoidance or attraction responses, and temporary interference with foraging capabilities within turbidity plumes	NEGLIGIBLE
Bearded seal	As for ringed seal, and localized loss of benthic food sources	NEGLIGIBLE
Oldsquaws, eiders, loons, other diving ducks and alcids	Localized and relatively short-term loss of benthic food resources, temporary reductions in prey visibility within turbidity plumes, disturbance.	NEGLIGIBLE to MINOR; the degree of concern could be considered MINOR if dredging operations were located near 'critical' feeding areas during staging and brood rearing.
Gulls, jaegers, shorebirds	Attraction to dredging activities due to increased availability of invertebrate food sources and/or human presence	NEGLIGIBLE
Marine fish	Temporary avoidance or sublethal effects (e.g. decreased feeding efficiency) resulting from exposure to suspended solids; localized loss of benthic food sources and habitat; potential entrainment of some less mobile or juvenile forms	NEGLIGIBLE to MINOR
Anadromous fish	Effects as with marine fish, but degree of concern is considerably greater where populations are concentrated	NEGLIGIBLE to MODERATE

TABLE 2.4-2 (Page 3)

Environmental Component or Resource	Potential or Probable Effects	Degree of Potential Regional Concern
Phytoplankton	Localized and short-term increases or decreases in growth and production associated with altered nutrient and light regimes within turbidity plumes; potential changes in local community structure during chronic or long-term disturbances	NEGLIGIBLE
Zooplankton	Potential mortality and sublethal effects resulting from exposure to relatively high suspended sediment levels. Increased or decreased feeding depending on concentration and nature of suspended solids	NEGLIGIBLE
Benthic Infauna	Direct mortality due to excavation and spoil deposition in benthic habitats; habitat changes and creation of some new or unique habitats resulting in recolonization by different species; sublethal effects on feeding efficiency associated with turbidity plumes; localized effects associated with changes in water quality and sediment chemistry (see text)	Variable; the degree of concern ranges from MINOR in nearshore (estuarine) zone where biomass is low to MODERATE in local sensitive and highly productive habitats (see text). Degree of concern in offshore areas ranges from MINOR to MODERATE depending on location and size of dredging operation.

TABLE 2.4-2 (Page 4)

Environmental Component or Resource	Potential or Probable Effects	Degree of Potential Regional Concern
Benthic Epifauna	As above except for reduced degree of direct mortality due to greater mobility of some epifaunal species. Recolonization of disturbed habitat or new habitat more rapid	Variable; degree of concern ranges from NEGLIGIBLE to MODERATE depending on the location and size of dredging operations. A MODERATE concern exists in some highly productive coastal environments where epifauna are important for the support of local breeding bird populations and rearing anadromous fish species.
Epontic Flora	Localized incorporation of suspended sediments in lower ice surface decreasing light penetration and potential production; possible short-term and localized increases in nutrient availability due to water column mixing. See also Icebreaking, Section 2.3.8	NEGLIGIBLE for dredging <u>per se</u>
Epontic Fauna	Altered temperature and salinity structure requiring physiological adjustment. Localized short-term decreases in oxygen availability. See also Icebreaking, Section 2.3.8	NEGLIGIBLE for dredging <u>per se</u>
Micro-organisms	Possible increased numbers and and metabolism of bacteria in areas where relatively high organic content or nutrient-rich sediments are excavated	NEGLIGIBLE

2.5 TREATED SEWAGE

2.5.1 Introduction

Sewage can contain a variety of pollutants such as trace metals, oils, greases, toxic chemicals, and in the case of untreated sewage, large amounts of organic material and solids. A discussion of all possible components in sewage and their potential effects on marine flora and fauna is beyond the scope of this document since relatively limited quantities of treated sewage would be released from marine vessels, production facilities and shorebases. Consequently, the following discussion is limited, in general terms, to potential effects of organic compounds and solids (= sludges) and nutrient enrichment, and specifically with potential concerns associated with discharge of treated sewage.

Treated sewage from proposed offshore exploration production facilities would normally be disposed of at depth via a submarine outfall, although in some cases, the effluent could be discharged on top of ice with mixing taking place during breakup in the spring. Submarine outfalls would always be used at shorebases, and all sewage from marine vessels and both offshore and onshore facilities would undergo secondary treatment. All outfalls would also be located in areas allowing acceptable dispersion and dilution of the effluent. Sludge produced from these treatment plants would either be incinerated or deposited in landfills.

Expected characteristics for secondary treated effluent from these facilities are 45-130 mg/L BOD, 60-130 mg/L suspended solids, 10-20 mg/L nitrogen and 3-6 mg/L phosphorus (Cain and Swain 1980), and dilutions in excess of 100:1 should further reduce nutrient concentrations to ambient levels found in the Beaufort Sea.

Potential environmental concerns associated with sewage disposal in the Beaufort Sea include nutrient enrichment, oxygen depletion in receiving waters, and smothering of benthic organisms by solids, although all of these potential effects would be very localized in the case of effluent which has received secondary treatment. The degree of concern related to sewage disposal is a function of the volume and circulation of the receiving water in relation to the volume of sewage discharged. On a global basis, the open ocean and most small seas have not been seriously affected by sewage discharge (Topping 1976). The effects of sewage discharges, when documented, have generally been restricted to shoreline areas, bays and fjords near large cities, as well as to the estuaries of rivers that have received sewage discharges from major population centres. For example, it has been calculated that the total annual amount of biodegradable material discharged into the North Sea (about 545,740 tonnes) consumes only about 0.13 percent of the dissolved oxygen in the North Sea at any one time (I.C.E.S. 1974). However, on a smaller scale, local marine waters have been adversely affected by sewage discharges from major centres of human population; these effects have included algal blooms (Ryther and Dunstan 1971), oxygen depletion and shellfish contamination (Berg 1975).

One of the most pronounced effects of sewage disposal in aquatic environments is the settling of solid materials. With time, solids accumulate and eventually cover the bottom, especially in regions of restricted water circulation. The degradation of these solids can lead to severe oxygen depletion, and in some instances, they may also contain high levels of trace metals and other potentially harmful materials. In an extreme case, Pearce (1970a, b) reported that indigenous benthic and epibenthic organisms were virtually eliminated over an area of about 26 km^2 off New York harbour where massive quantities of sewage sludge had been dumped over a period of about 45 years. Adjacent to this area, clams were contaminated with enteric bacteria, and fin rot disease of fish was also quite common. In the Beaufort Sea, the potential detrimental effects of such solids would be considered NEGLIGIBLE since the current plans for waste management indicate that sludge will not be discharged to the ocean.

Another potential adverse effect of sewage disposal in natural waters is the transmission of diseases. Disinfection with chlorine is successful against most bacteria but less so with viruses. Outbreaks of infectious hepatitis and acute gastroenteritis have been traced to sewage-contaminated shellfish (Liu 1970). In some cases, contamination of water by sewage has also necessitated local closures of commercial shellfish fisheries, recreational beaches and sports fisheries.

The greatest potential area of concern with respect to sewage discharge in the Beaufort Sea would be the release of effluent into ice-covered nearshore waters. In shallow areas, little free water remains beneath the landfast ice sheet in mid to late winter, and circulation may also be greatly reduced in these areas. In addition, if the sewage discharge is of lesser density than the receiving water, it could form a frozen layer between the undersurface of the sea-ice and the seawater, thereby exposing epontic flora and fauna to relatively concentrated sewage effluent when this layer melts in the late spring. This and other potential effects of sewage on marine resources of the Beaufort Sea are discussed in the following sections.

2.5.2 Effects of Sewage on Mammals

The degree of concern regarding potential effects of domestic sewage discharged from offshore production facilities and marine vessels on marine mammals of the Beaufort Sea is expected to be NEGLIGIBLE. All wastes would receive secondary treatment, dilution of the effluent in surrounding waters would be rapid, and sludges would be either incinerated or landfilled. Similarly, the discharge of treated sewage in nearshore areas is also not considered a significant area of concern with respect to marine mammals. A highly localized nutrient enrichment and oxygen depletion zone may be present near some outfalls, but it is unlikely that marine mammals would approach these shallow areas for extended periods due to the level of activity associated with offshore and shorebased activities (Sections 2.6.2 and 2.6.3).

The release of treated sewage in ice-covered nearshore waters may result in more pronounced nutrient enrichment and oxygen depletion when water circulation is restricted. Nevertheless, the effects of sewage discharge on marine mammals and their food sources under the ice would be highly localized and of NEGLIGIBLE regional concern.

2.5.3 Effects of Sewage on Birds

As with marine mammals, the release of domestic sewage from offshore and nearshore facilities and marine vessels would have no regionally or locally significant effects on marine birds because the effluent would receive secondary treatment and should be rapidly diluted. Outfalls may attract some species (e.g. gulls) and thereby increase their susceptibility to disturbances associated with other activities at shorebases (e.g. potential fuel spills, airborne noise), although the attraction of birds to outfall sites would be largely indistinguishable from the attraction of some species to this (and any) area of human activity (Section 2.2.3). Consequently, the degree of concern regarding effects of sewage on birds is considered NEGLIGIBLE.

2.5.4 Effects of Sewage on Fish

There are few documented effects of domestic sewage on fish. Normally, the dilution of sewage in receiving waters is sufficient to prevent bacterial infections or other kinds of contamination disorders associated with sewage outfalls. Only in extreme situations, such as in New York Harbour (Section 2.5.1) or in southern California, have infections such as fin rot disease been reported in local fish populations (Pearce 1970a,b; Sherwood and McCain 1976), and these unusual cases are primarily due to contamination of the sediments with sewage.

The addition of disinfectants (e.g. chlorine or bromine) during secondary treatment of sewage may have toxic or sublethal effects on fish depending on the concentrations of these chemicals near discharge sites. Several toxicity studies suggest that residual chlorine is more toxic than bromine, and concentrations less than 0.5 ppm have been shown to be acutely toxic to several temperate water fish species in 96-h bioassays (Ward and DeGraeve 1977; Roberts 1980). The current recommended level of residual chlorine for the protection of aquatic life in fresh water is 0.002 ppm (EPA 1976). Ammonia (non-ionized) and surfactants also contribute to the toxicity of sewage effluents.

The oxygen demand of domestic wastes can reduce dissolved oxygen concentrations to levels which are not acceptable for fish habitat in confined areas. Davis (1975) suggests a level A requirement of 7.75 to 9.75 mg/L of dissolved oxygen, depending on salinity and temperature, while the EPA criteria is a minimum dissolved oxygen concentration of 5 mg/L (EPA 1976).

In the Beaufort Sea, the relatively small volumes of effluent involved and the anticipated dilution would likely ensure sufficient dispersion of wastes. Dissolved oxygen concentrations are unlikely to drop more than a few percent below ambient levels, and if disinfectants are used during secondary treatment, dilution should rapidly reduce residual chlorine to acceptable levels. Consequently, the degree of concern regarding the potential effects of treated sewage on regional or local fish populations is expected to be NEGLIGIBLE.

2.5.5 Effects of Sewage on Phytoplankton

The discharge of sewage to the marine environment can have either an inhibitory or stimulatory effect on phytoplankton populations, depending on the amount and type of contaminants contained in the effluent (North *et al.* 1972). However, most contaminants in sewage which adversely affect phytoplankton are of industrial origin, and only secondary treated domestic wastes would be discharged to the Beaufort Sea.

The primary effects of domestic sewage on phytoplankton are associated with the addition of nutrients, particularly nitrogen and phosphorous. Both of these nutrients are required for phytoplankton growth, and growth will be restricted once either is present in limiting concentrations. In freshwater environments, phosphorous is generally the limiting nutrient (Schindler 1974), while nitrogen usually limits phytoplankton growth in marine waters (Ryther and Dunstan 1971; Reed 1975; Topping 1976). Nutrient measurements in the Beaufort have confirmed that primary productivity can be nitrogen-limited during periods when the growth of phytoplankton is not already light-limited (Grainger 1975). In some cases where solids are not removed from sewage, a localized reduction of light and hence productivity may result from increased water turbidity, although this would not occur in the Beaufort Sea because sludges would either be incinerated or landfilled.

The effects of sewage are related to the capacity of the receiving waters to accept, dilute and disperse the effluent. Most investigations have indicated little or no overall impacts of marine sewage disposal on phytoplankton, with higher primary production only occurring in localized areas near the discharge sites despite the fact that the volumes of effluent were often very large and included industrial wastes (Pike and Gameson 1970; Chen and Orlob 1972; Eppley *et al.* 1972; Reed 1975; McIntyre and Johnston 1975; Kleppel and Manzanilla 1980). On the other hand, Saunders and Kuenzler (1979) reported that the productivity and species diversity of a phytoplankton community were affected in an enclosed estuary receiving only domestic wastes.

The waste management plan for the Beaufort region indicates that secondary treated wastes will be discharged via submarine outfalls into areas which offer considerable mixing capabilities, and enclosed bays and inlets will be avoided (EIS Volume 2). Volumes of sewage discharged would also be small in relation to the size and assimilation capacity of receiving waters. The greatest potential for effects on phytoplankton is likely to occur during the period of initial (spring) development of the community under ice cover, when water circulation and dilution capacity are reduced. However, since the phytoplankton are light-limited at this time (Alexander 1974) nutrient enrichment should not have a significant influence on the community. Consequently, the potential effects of treated sewage disposal on the primary productivity and species composition of phytoplankton communities are likely to be very localized and of NEGLIGIBLE regional concern.

2.5.6 Effects of Sewage on Zooplankton

If nutrient-rich sewage is discharged into the euphotic zone of open marine areas, local enhancement in primary production may occur under some circumstances (Bascom *et al.* 1980), and this can result in concomitant localized increases in zooplankton production and biomass when the degree of enrichment is sufficient (Welch 1980). The most pronounced effects of sewage on phytoplankton and zooplankton can occur in enclosed coastal regions, especially in the vicinity of major river estuaries (Topping 1976). In these areas, considerable stress may be exerted on the marine environment if nutrient levels are increased more than threefold. Under such circumstances, some phytoplankton species may be inherently better adapted to take advantage of the extremely high nutrient levels, and therefore become the dominant species in the community. When these species cannot be ingested by herbivorous zooplankton or are detrimental to predaceous forms, the species diversity of the zooplankton community may be affected (Topping 1976), and this can be reflected elsewhere in the food web. However, it should be emphasized that a threefold increase in nutrient levels would rarely be achieved in a marine environment with adequate water circulation, except possibly in the immediate vicinity of outfalls. In addition, any changes in the species composition of the planktonic community would be relatively localized, and since phytoplankton and zooplankton are continuously transported into and out of well circulated areas, the period of exposure to sewage effluent would generally be insufficient to cause regionally significant alterations in community structure.

As in the case of phytoplankton, the greatest potential effects of sewage on zooplankton in the Beaufort Sea would probably occur in nearshore areas when ice cover is present and water circulation reduced. Nutrient levels could be locally increased and the sewage components which are less dense than seawater may also tend to accumulate beneath the ice. Those zooplankton species associated with the epontic community may, under such circumstances, be exposed to relatively concentrated effluent (Section

2.5.8). However, the majority of the zooplankton usually inhabits deeper waters at this time, and would not be exposed to concentrated sewage effluent. In addition, since sewage would be discharged from submarine outfalls into well-mixed receiving waters located well away from enclosed coastal inlets, potential effects on zooplankton in open waters would be localized and of NEGLIGIBLE regional concern. When ice cover is present, the effects may be more significant, but would still be considered NEGLIGIBLE and restricted to a localized area in the vicinity of the outfall.

2.5.7 Effects of Sewage on Micro-organisms

Although secondary treatment of sewage would greatly reduce the organic content of effluent released in the Beaufort Sea region, some biodegradable matter would still enter the water column. Organic matter in sewage is normally 40 to 50 percent protein, which contains amino acids that are metabolized by bacteria. The carbohydrates present in sewage are mainly readily-degradable starches and sugars and more resistant cellulose, while fats in sewage are degraded very slowly (Glayna 1971, cited in Zain-ul-Abedin 1978). Relatively large quantities of phosphate and nitrate also remain in secondarily-treated sewage (Nester *et al.* 1978), and together with the added organic matter, can stimulate growth of bacteria around outfalls. This may in turn be beneficial to bactivorous invertebrates, although high rates of bacterial degradation can also deplete dissolved oxygen levels in the water column, particularly in shallow nearshore areas during winter when ice cover greatly restricts oxygen exchange with the atmosphere (Fonselius 1978). In the Beaufort Sea, however, the low winter temperatures would limit bacterial activity and likely prevent serious oxygen depletion in the water. Low oxygen levels could also develop during the open water season if nutrients released both with the sewage and as metabolites of bacterial degradation stimulated a localized phytoplankton bloom. These phytoplankton cells would eventually die and also be degraded, contributing to an increased biological oxygen demand (Fonselius 1978). Extensive microbial activity during the summer months can lead to the creation of anaerobic conditions below the thermocline in highly stratified waters, although this situation has not been documented in either nearshore or offshore waters of the Beaufort Sea.

Sewage can contribute both intestinal pathogenic and beneficial autotrophic and heterotrophic micro-organisms to marine and freshwater environments (Zain-ul-Abedin 1978). Raw sewage has been found to contain average maximum concentrations of 3.4×10^{10} aerobic and 2.8×10^{7} anaerobic bacteria per 100 ml (Coetzee and Fourie 1965, cited in Zain-ul-Abedin 1978). Pathogenic bacteria commonly include strains of *Salmonella*, *Vibrio* and *Mycobacterium* (Rao 1978). However, as indicated earlier, disinfection with chlorine during secondary treatment would minimize concerns related to the introduction of pathogens to the Beaufort Sea. As a result, the overall degree of potential concern regarding sewage-related increases in bacterial activity in the Beaufort Sea is considered NEGLIGIBLE.

2.5.8 Effects of Sewage on Benthic Communities

Sewage disposal can affect benthic communities by: 1) modifying the physical and chemical characteristics of the substrate, 2) reducing dissolved oxygen levels near the substrate, and 3) smothering some flora and fauna by the deposition of material. These effects can individually or in combination alter benthic species diversity, biomass, distribution or community structure (Botton 1979). Modifications to temperature, transparency, pH, dissolved oxygen content of the water column and salinity associated with sewage discharge have usually been negligible and their effects on benthic communities insignificant due to rapid dilution of effluent in surrounding waters. However, the nature and magnitude of the effects of sewage discharge on substrate characteristics and the degree of smothering of benthos by sewage solids has varied with the physical and chemical characteristics of the discharge as well as with various oceanographic characteristics of the receiving environment (substrate composition, current velocity, water depth).

In the Beaufort Sea, increased nutrient levels and deposition of some solids on the substrate could result in localized effects on benthic flora. Grainger (1975) reported that phytoplankton primary productivity of the southern Beaufort Sea was usually limited by water transparency within the Mackenzie River plume and by nitrate concentrations outside the plume. This suggests that outside of the Mackenzie River plume, nutrient addition by sewage discharge could have a minor but localized positive effect on benthic marine flora, although the abundance of benthic flora in these waters is not known.

Grigg and Kiwala (1970) examined the effects of sewage discharged at an outfall in southern California on marine life in that area. They reported a complete absence of benthic algae near the outfall, and suggested that this appeared to be due to deposits on the bottom which modified or covered an otherwise suitable substrate. The deposition of material originating from sewage discharge may smother benthic microalgae and reduce substrate suitability for benthic macrophytes. However, in the Beaufort Sea, amounts of settleable solids would be considerably reduced in the secondary treatment process, and the area affected by remaining material would be spatially limited.

Sewage may also indirectly affect benthic fauna due to modification of the physical and chemical characteristics of the substrate. Armstrong et al. (1980) reported that several trends in the distribution and abundance of benthic fauna have been observed in marine areas near sewage outfalls. The amount of organic matter in the sediment is often high near these discharges and benthic macroinvertebrate populations differ with distance from the outfall. Near the outfall, there is usually a zone containing very few species and numbers of benthic invertebrates, while a zone with large numbers of a few tolerant species often occurs at a greater distance from the outfall. With additional increases in distance from the outfall, the number of species increases and the number of individuals decreases until a community structure more characteristic of the area occurs. Anger (1975) and Otte and Levings (1975) observed the same trend in benthic macroinvertebrate populations exposed to raw or primary treated sewage in coastal areas of the Baltic Sea and Strait of Georgia, respectively.

The organic content of sediment appears to be the most important factor affecting the qualitative distribution of benthic macroinvertebrates (Watling et al. 1974) because organic detritus is utilized as a food source by many benthic species (Otte and Levings 1975). Reish (1973) reported that carnivorous species are eliminated before filter feeders as the organic content of sediments increases. The remaining detrivores often dominate in organically polluted conditions, but even these species may be eliminated in severely polluted environments. For example, Watling et al. (1974) found that approximately 73 percent of the benthic fauna at a sludge disposal site were infaunal species and almost half of these were polychaetes. Armstrong et al. (1980) also found that burrowing deposit feeders predominated in enriched areas near a combined sewer outfall. It appears, therefore, that the deposition of material originating from sewage can have a minor negative effect on benthic macroinvertebrate communities by reducing overall species diversity and the abundance of carnivorous and herbivorous forms. At the same time, in areas with low current velocities and poor circulation, the waters overlying the sediments may have a reduced dissolved oxygen content due to the BOD of the organic matter (Otte and Levings 1975). This can also lead to differences in the species composition of benthic communities.

Deposition of materials originating from sewage discharge can also affect the diversity and abundance of the epibenthic component of benthic communities. The deposition of fine organic matter may reduce substrate stability, and therefore the availability of attachment sites for sessile epibenthic invertebrates. The survival of the pelagic larvae of both infaunal and epifaunal macroinvertebrates also can be reduced if larvae are smothered by organic material originating from sewage discharge. For example, Grigg and Kiwala (1970) suggested that a reduction in the number of benthic species near a sewage outfall off the southern California coast was due to decreased settlement and survival of larvae caused by the presence of fine-grained sediments which covered the bottom.

In the Beaufort Sea, the effects of sewage disposal on benthic communities are expected to be associated with modification of the physical and chemical characteristics of the substrate, transient reductions of dissolved oxygen levels near the substrate and smothering. However, these potential effects are also expected to be very localized because of the rapid dilution of sewage, as well as the limited amounts of solids which would remain in the effluent following secondary treatment. Smothering of benthic flora and fauna would probably be restricted to very localized areas in the vicinity of the outfalls. Similarly, any effects related to decreased oxygen availability are likely to be spatially limited. Increased mortality of benthic larvae at the settling stage is possible, but only in the immediate vicinity of the outfalls, while localized increases in the abundance of detrivores may occur due to the increased availability of organics. Consequently, from a regional point of view, the degree of concern associated with all of these potential effects of treated sewage on benthic communities is considered NEGLIGIBLE.

2.5.9 Effects of Sewage on Epontic Communities

The epontic algal community may contribute as much as 25-30 percent of the total annual primary production in the Arctic (Alexander 1974, cited in Campbell 1981), but its main importance is probably associated with the timing of this production, which occurs 6-8 weeks prior to significant phytoplankton blooms. Assessment of the potential effects of sewage on this community is hampered by the lack of direct research, although the effects of sewage on planktonic algae are well-documented and probably similar. Depending on location, epontic algae may or may not be nutrient-limited (Alexander 1975; Grainger 1977), and there is also some evidence of heterotrophy (utilization of organic compounds through non-photosynthetic pathways) by some species (Horner and Alexander 1972). Whether the presence of nutrients in sewage would stimulate further growth of the algae, or the community is light-limited (Bunt 1964; Campbell 1981) is not known.

Sewage discharged beneath the ice cover in winter could rise to the ice-water interface and almost immediately freeze, because its specific gravity would generally be less than that of seawater. During the spring melt, this layer of ice would begin to melt, reducing the normally high salinity levels, and in some cases, entering brine channels in the sea-ice. However, this may not be detrimental since growth of a number of species of ice algae has been shown to increase when salinities naturally decrease (Horner 1977). This freshwater layer may also affect the epontic fauna, although these organisms are likewise exposed to a wide range of salinities during the spring melt. Nutrient-related increases in the productivity of the epontic flora could increase food availability for herbivorous invertebrates grazing at the under-ice surface. However, any positive effects of sewage on epontic flora and fauna would be very localized and insignificant in terms of regional primary and secondary production. Consequently, the degree of concern regarding potential effects of sewage discharge on epontic communities in the Beaufort Sea is expected to be NEGLIGIBLE.

2.5.10 Summary of Concerns Related to Sewage

Proposed waste management plans for the Beaufort Sea indicate that domestic sewage from all sources (shorebases, exploration and production platforms, and marine vessels) will be treated in secondary treatment plants, with effluents from shorebases likely receiving chlorine disinfection prior to disposal in marine waters via a submarine outfall. Solids (sludge) from the treatment plants would be either incinerated or disposed of in an approved landfill. The location of the outfall will be particularly important since the adverse effects of sewage effluent are primarily related to the degree of dilution and dispersion in the receiving waters. As a general waste management strategy, only receiving waters which will allow sufficient dilution would be considered and enclosed bays, lagoons or estuaries would be avoided (EIS Volume 2).

Unlike previous sections dealing with other common wastes and disturbances which will be associated with future hydrocarbon exploration and production in the Beaufort Sea, the degree of regional concern regarding sewage disposal is NEGLIGIBLE for all resources. For this reason, a summary table identifying significant potential effects and degree of regional concern for each biological resource has not been provided in this section. In all cases where the above guidelines are followed, potential effects of sewage effluent would be limited to the immediate vicinity of the discharge sites. The most significant effects would result from localized nutrient enrichment, and could include minor changes in the species composition and abundance of phytoplankton and benthic microalgae, as well as indirect effects on members of higher trophic levels (herbivorous zooplankton and some benthic fauna) which feed on these algal communities.

2.6 UNDERWATER SOUND

2.6.1 Introduction

Most industrial activities associated with petroleum hydrocarbon exploration and production in marine environments produce waterborne sounds of various frequencies and intensities. There is considerable concern regarding the possible effects of underwater industrial sound on some marine fauna since underwater noise will be associated with virtually all phases of development. Sources of underwater noise which have been identified as those most likely to result in effects on marine fauna include icebreaking tankers and support vessels (e.g. icebreakers, ships, barges and tugs), aircraft, vehicles on ice, dredges, drilling activities, and oil and gas production and processing activities (EIS Volume 4, Chapter 2, Section 3.6). Potential areas of concern include direct disturbance, temporary masking of marine mammal communication or navigational signals, and at intense noise levels, physiological damage.

Although virtually every activity involving the operation of machinery in or near the marine environment may generate underwater sound, disturbance effects on marine fauna are only possible if the species can detect the noise. It is assumed that no disturbance or masking effects will occur at distances where the noise has attenuated to background levels. The range of sound detection depends on: (1) the source level ("loudness"), frequency, bandwidth and directional characteristics of the noise source; (2) transmission losses between the source and the potential receiver (propagation), and (3) the level and characteristics of ambient noise at the receiver. In addition, sound detection and the potential for communication signal masking depend on the hearing sensitivity, directional characteristics and the lowest acceptable signal-to-noise ratio of the receiver (i.e. "critical ratio").

The range within which industrial underwater noise could affect marine mammals or fish depends on the interaction of all of these factors. The following sections discuss source levels of industrial underwater noise (Section 2.6.2), propagation characteristics (Section 2.6.3) and ambient noise levels (Section 2.6.4) in the Beaufort Sea, and the hearing and vocal abilities (Section 2.6.5) of marine mammals and fish in the region. Section 2.6.6 estimates the potential zone of influence of underwater noise and assesses the potential degree of concern with respect to effects of underwater noise on marine mammals and fish in the Beaufort Sea region.

2.6.2 Characteristics of Industrial Underwater Noise

Mobile sources of industrial underwater noise primarily include support vessels (ranging from small boats to icebreaking tankers) and aircraft, while stationary sources of underwater noise include dredges, drillships, and facilities for production and processing of petroleum hydrocarbons.

2.6.2.1 Composite Artificial Island Construction Activities

Underwater industrial sounds were measured by Ford (1977) at the Tuft Point borrow pit and the Arnak L-30 artificial island construction site in the eastern Beaufort Sea, and discussed in relation to their probable ranges of audibility by white whales. At Tuft Point, a suction dredge (ARCTIC NORTHERN), one or two small crew boats and several tugs were typically operating during the period of study. Normal activities resulted in estimated peak source levels between 152 and 157 dB re 1 $\mu Pa/Hz^{1/2}$ in the 500-1000 Hz range [unless otherwise indicated, all sound levels in this section are expressed in dB (decibels) re 1 μPa (microPascal)/$Hz^{1/2}$ (also termed 'spectrum level'), and source levels are referred to a standard distance of 1 m]. Occasional transient sounds had estimated source levels as high as 175 dB and contained substantial energy at frequencies as high as 15 kHz. Sound transmission losses were also high in the Tuft Point area, likely as a result of the very shallow depth (about 5 m), and most industrial sound in the 250-2000 Hz range had reached ambient noise levels (50-60 dB) by a distance of 3.6 km.

Composite underwater sound generated by the suction dredge BEAVER MACKENZIE, a small tending tug, a clamshell shovel and at least one small crew boat operating at the Arnak L-30 artificial island construction site had an estimated peak source level of 164 dB at 1390 Hz (Ford 1977). Transmission losses at Arnak (depth 11 m) were lower than at Tuft Point, and significant energy (82 dB at 480 Hz) was still present at the maximum range tested (4 km). It was expected that sound energy from this site would reach ambient at a distance of 5-6 km.

Sound pressure levels and frequency characteristics of noise generated by each of 3 tugs and a crew boat were also measured near the Tuft Point borrow site by Ford (1977). Results from these recordings indicated that a tug pushing a full barge generated estimated peak source levels of 164 dB at 200-1150 Hz, while a small crew boat (ARCTIC EXPEDITOR) and a tug pushing an empty barge produced peak source levels of approximately 154 and 151 dB, respectively.

Fraker et al. (1981) characterized sound produced by a 16 m crew boat (IMPERIAL ADGO) and a supply vessel (CANMAR SUPPLIER VIII) in the southeastern Beaufort Sea during August 1980. Most of the sound energy transmitted into the water from the former vessel was at frequencies <2 kHz, although appreciable energy was recorded up to 4 kHz. The peak tone recorded from the IMPERIAL ADGO was at 90 Hz (107.5 dB), measured as it moved at full speed past the recording device at 200 m. Sound levels were 20-30 dB above background levels (quiet ambient) at frequencies from 1 to 4 kHz. The CANMAR SUPPLIER VIII, also recorded from a distance of about 200 m, produced its loudest tone at 56 Hz (116 dB). This vessel produced sounds 30-40 dB above quiet ambient levels throughout the spectrum up to at least 8 kHz.

Fraker et al. (1981) reported that composite sounds from island construction activities at Issungnak O-61 (dredge, tugs, barge camps) were above ambient noise levels at a distance 4.6 km north of the artificial island site. Sound levels measured 1.2 km from the site were 20-50 dB above quiet ambient levels at frequencies up to 8 kHz during a period when a suction dredge (BEAVER MACKENZIE) was operating. At another island construction site (Alerk), the same dredge plus attending equipment produced sound levels of 90-100 dB at frequencies of 1 kHz and below, while most of the remaining energy was at frequencies between 1 and 2 kHz.

2.6.2.2 Icebreaking Tankers

Sources of underwater noise that would be produced by icebreaking tankers are sounds associated with the cracking of ice, ice rubbing against the vessel and the noise generated by the ship's machinery. Arctic Pilot Project (APP) (1981) suggests that the noises produced by LNG carriers as a result of the physical breaking of ice will be similar to noises produced during natural processes of ice fracturing and cracking. Arctic marine fauna in pack-ice and transition zone areas are presumably adapted to these natural sounds, while inhabitants of shorefast ice areas may not be as familiar with natural sounds associated with ice fracturing (APP 1981). Icebreaking tankers proposed for use in the Beaufort Sea have not been constructed to date, and consequently measurements of underwater sound from the physical breaking of ice by these vessels are not available. Brown (1982) states that noises produced during the physical breaking of ice by LNG carriers would be insignficant in comparison to noise produced by propeller cavitation (the depressurization of bubbles against the propeller). The free-field source levels produced by LNG carriers are given on Figure 2.6-1, and can be defined as the sound pressure level which would be measured in an infinite body of water, at a distance of one metre from a point source. The actual source is neither a point nor is it located in an infinite fluid. Taking these facts into account, estimates of received noise levels for LNG carriers travelling through Baffin Bay at various ship speeds and frequencies are provided in Table 2.6-1 (APP 1981). Data from the world's largest icebreaker, ARKTIKA (75,000 SHP), during trials in the Antarctic indicated the presence of two spectral peaks. The first peak at 150-200 Hz was due to breaking of ice (and ship's machinery), while the second peak at 1-2 kHz was associated with rubbing of ice against the hull of the vessel (Bogorodskii and Gavrilo 1979).

In contrast to noise resulting from the breaking or fracturing of ice, underwater sound generated by the ship itself is unnatural and could affect marine mammals and fish. APP (1981) examines the potential underwater sound produced by Class 7 icebreaking LNG carriers (75,000 SHP) proposed for the Arctic Pilot Project. These estimates will be taken in the present discussion as representative of noise levels produced by icebreaking tankers, since both the tankers and LNG carriers are of similar design and specifications (EIS Volume 2).

TABLE 2.6-1

ESTIMATED RECEIVED SOUND PRESSURE LEVELS FROM CLASS 7 LNG CARRIERS TRAVELLING THROUGH BAFFIN BAY OVER WATERS 2000 m DEEP (Receiver Depth 20 m) (after APP 1981)

Operating Conditions	FFSL*	Sound Pressure Level (dB//μPa²/Hz) Range (km): 1	4	10	30	60
		25 Hz				
Full power, ice (22.2 km/h)	177	104	102	94	84	78
Full power, open water (40.7 km/h)	172	91	89	81	69	63
Half power, open water (31.5 km/h)	165	84	82	74	62	56
		100 Hz				
Full power, ice	172	102	101	97	92	88
Full power, open water	169	95	95	86	79	77
Half power, open water	163	89	89	80	73	71
		1 kHz				
Full power, ice	152	92	87	84	77	71
Full power, open water	149	85	77	74	67	61
Half power, open water	143	79	71	68	61	55
		5 kHz				
Full power, ice	138	81	74	62	51	37
Full power, open water	135	80	63	55	44	29
Half power, open water	129	74	57	49	38	23

*FFSL = Free Field Source Level, dB//(μPa x m)²/Hz
- the sound pressure level which would be measured in an infinite body of water, at a distance of 1 metre from a point source of propeller cavitation.

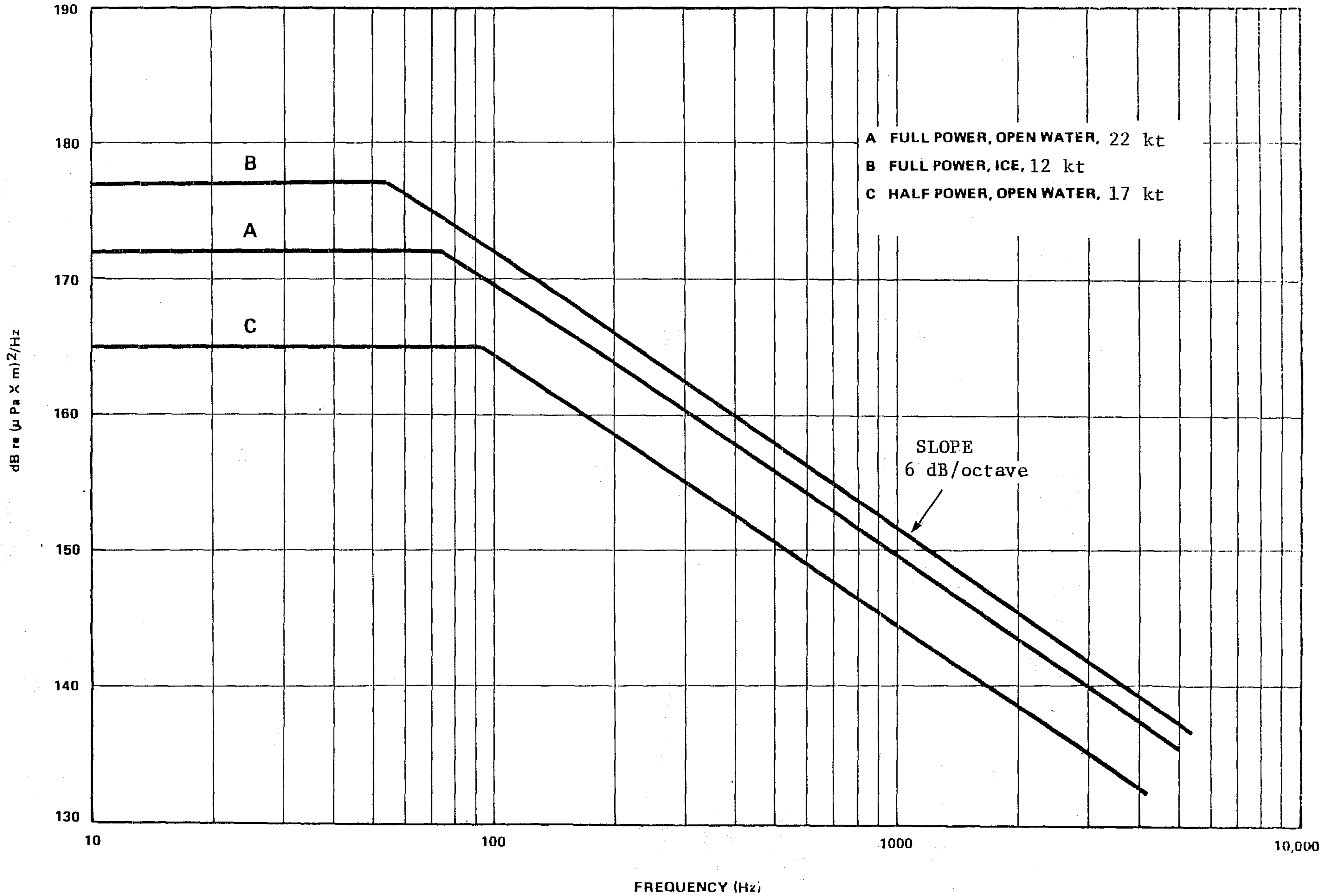


FIGURE 2.6-1 ESTIMATED "FREE-FIELD" SOURCE LEVELS OF LNG SHIP

(After APP 1981)

Surface ship-radiated noise would be generated primarily by propeller cavitation. At full power in thick ice, the carriers would produce a maximum free-field broadband spectrum level of 172 dB at 100 Hz. While travelling through open water, the noise level is expected to be reduced to about 169 and 163 dB at full and 40 percent power, respectively (APP 1981). Sound levels are expected to decline at the rate of 6 dB per octave (Figure 2.6-1).

In addition to these broadband sounds, high intensity, narrowband tones are generated at specific frequencies by propellers. These blade-rate sounds are a function of the propeller revolution rate and number of blades. For the APP LNG carriers, the fundamental blade-rate frequency is expected to be 5.33 Hz, with higher harmonics. Although it is not yet possible to determine the sound levels for each harmonic or the number of harmonics which may be generated by the proposed LNG carriers, the expected maximum level for a carrier operating at full power in heavy ice is 191 dB. However, because the noise source would be poorly coupled to the water column due to the shallow depth of the propeller, the tonal would be reduced to 164 dB.

2.6.2.3 Aircraft

The underwater reception of turbo prop and helicopter noise have been discussed by Urick (1972) and Medwin and Helbig (1972), respectively. In shallow water with good transmission conditions, waterborne sound from a passing aircraft may persist detectably longer than airborne sound (Fraker et al. 1981). Sounds from aircraft flying near sonobuoys are often received at the sonobuoy hydrophones (Ljungblad and Thompson 1979; Fraker et al. 1981). Fraker et al. (1981) reported that noise from a Britten-Norman Islander aircraft was received at a sonobuoy hydrophone located at a depth of 14.5 m (sea floor) during a period of calm seas in the Canadian Beaufort. Tonal sounds at frequencies corresponding to the revolution rate of the propeller blades and the cylinder firing rate were prominent in the received spectrum. When the aircraft was flown over the sonobuoy at an altitude of 610 m, sound levels received at the hydrophone were as loud as 97 dB at 70 Hz. This noise and sounds at other low frequencies (<1 kHz) were above quiet ambient levels (Fraker and Richardson 1980). Greene (1982) recorded underwater noise produced by a Bell 212 helicopter at an altitude of 150 m in the Beaufort Sea during August. Levels received ranged from 80-105 dB at frequencies below 250 Hz, and 50 to 70 dB between 1 and 8 kHz.

2.6.2.4 Drilling Noise

Waterborne drilling noise may emanate from drillships, platforms or artificial islands. The various activities associated with drilling may generate different types of underwater sound and these noises may originate from above or below the water. Malme and Mlawski (1979) recorded noise from drilling rigs on a natural and artificial island in shallow (depths 2-12 m) areas of Prudhoe Bay during a period with ice cover. Most sound energy was at frequencies below 200 Hz, with tonal components predominating below 100 Hz.

The broadband noise level was highest when the rotary table was turning, while the diesel engines and other rotating machinery produced the tonal components of the noise spectrum. Low frequency tones were still recorded at a distance of 6.4 to 9.6 km from the drilling rigs under quiet ambient noise conditions, although this distance was decreased to 1.6 km under noisy ambient levels. It should be noted, however, that distances at which drilling noise in the Beaufort reaches ambient may be greater because sound propagation would be better due to greater depths.

Fraker et al. (1981) recorded noise from the drillship EXPLORER 1 in the Beaufort Sea during August 1980. Spectrum noise levels were measured at distances of 0.6 km, 1.9 km and >1.9 km from the drillship and a nearby supply ship. The strongest tone had a level of 97 dB at 147 Hz, and was recorded at a distance of 1.9 km. Broadband noise levels were quite variable, apparently as a result of changes in on-board activity.

2.6.2.5 Dredging

The only studies of noise produced by operational dredges in the Beaufort Sea were those of Ford (1977) and Fraker et al. (1981) who described composite noise associated with various combinations of dredges, tugs and crew boats. The results of these investigations were previously discussed in Section 2.6.2.1.

2.6.3 Propagation of Sound

Sound is more efficiently transmitted through seawater than through air, and its velocity is also greater in water than in air. There are numerous factors which influence the propagation characteristics of underwater sound. In the marine environment, sound is attenuated by spreading, absorption and scattering losses associated with reflection, with the latter two phenomena depending on frequency.

Two generalized types of spreading occur: spherical and cylindrical. The former type of spreading occurs when sound radiates from the source in all directions, and the signal strength decreases at a rate of 6 dB per doubled distance. On the other hand, cylindrical spreading occurs when sound waves are channeled and propagated in only two dimensions. In this case, the signal strength decreases at a lower rate of only 3 dB per doubled distance. The distance from the source at which the transition from spherical to cylindrical spreading occurs is a complex function of water characteristics, source depth and bathymetry (Milne 1960, 1967; Mellen and Marsh 1965; Buck 1968). In the shallow Beaufort Sea, propagation loss due to spreading is probably between the amounts expected from spherical (greatest propagation losses) and cylindrical (least propagation losses) spreading (Fraker et al. 1981). In deep arctic waters, spreading is often cylindrical since sound may be channelled into the low salinity, near-surface zone, particularly when there is smooth, annual ice cover (Verrall 1981).

Losses of sound energy due to absorption are generally low in arctic waters, with losses being on the order of 0.0062 f^2 dB/km where f is the frequency in kHz (Milne 1967). As evident by this expression, absorption losses are insignificant at low frequencies. Scattering losses are also insignificant at low frequencies, with the amount of scattering depending on roughness of the under-ice or water surface in relation to the length of the sound waves. Scattering may be important in the attenuation of sound at frequencies >40 Hz (Verrall 1981).

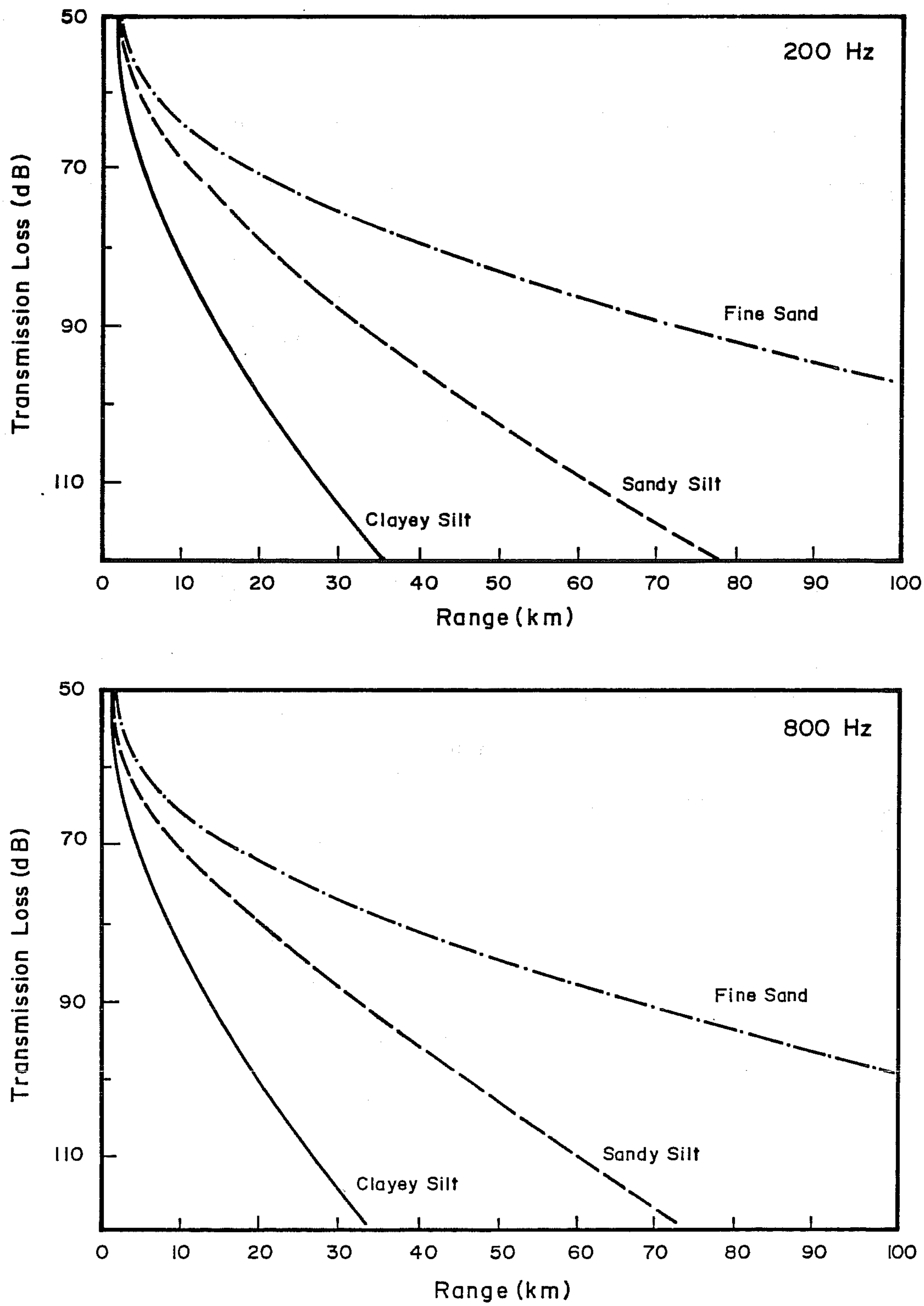
Propagation loss also varies with water depth, with losses being considerably greater in shallow water. For example, Leggat _et al._ (1981) found that propagation losses of up to 25 dB (at 63 Hz) above deep water loss levels occurred at receiving stations over the shelf in Melville Bay when the noise source was 185 km away. Bottom material and structure will strongly influence sound propagation. Transmission losses are particularly pronounced in shallow areas with sand or mud sediments, which tend to absorb sound energy. An exception to this generalization may occur at very low frequencies, where the sound waves can travel in the sediment and re-enter the water column at considerable distances from the source (Fraker _et al._ 1981).

During a period when Parry Channel (Viscount Melville Sound, Barrow Strait and Lancaster Sound) was covered by shorefast annual ice, Verrall (1981) observed greater attenuation of sound in water with a depth of 150 m, than in depths of 200 to 300 m, particularly at very low frequencies (10-40 Hz) and at frequencies >300 Hz. At the shallower depth, propagation losses at ranges of >10 to 20 km were generally greater than would be expected from spherical spreading. Losses were lower in water >500 m deep under smooth annual ice, where propagation corresponded closely to cylindrical spreading.

Sound propagation in the Beaufort Sea will vary markedly with season. Since lowest temperatures occur at the surface during periods of solid ice cover, a positive sound velocity profile (increasing speeds at greater depths) and a resultant upward refraction of sound rays will occur during winter. In summer, the sound velocity profile may show a sharp negative gradient (decreasing speeds at greater depths) (Fraker _et al._ 1981) because solar heating and wave action leads to the formation of a warm isovelocity layer in the upper portion of the water column.

Rogers (1981) discusses the extreme difficulty in predicting propagation losses in shallow waters with a negative velocity gradient. Losses can be dependent on at least 24 factors (e.g. water depth, velocity profile, sediment characteristics, etc.). Figure 2.6-2 illustrates how extreme and variable attenuation can be with different substrates for water with a negative velocity gradient. Substrates in the Beaufort Sea are largely of the 'clayey-silt' category, and as a result, propagation losses would undoubtedly be great.

Figure 2.6-2 Depth averaged propagation loss vs. range for water with a 0.2 s^{-1} negative velocity gradient at 200 Hz and 800 Hz, 100 m water depth and three different sediment types. (after Rogers 1981)



Greene (1981) examined sound transmission characteristics of shallow water (about 50 m) in the Chukchi Sea during both winter (100 percent ice cover) and summer (about 50 percent ice cover). During the winter, low frequencies (5 to 20 Hz) were propagated well, and tones were lower than that predicted by spherical spreading. Propagation at frequencies of 75 to 500 Hz was better than that predicted by spherical spreading to ranges of about 5-10 km, beyond which transmission loss increased. In summer, propagation losses at all frequencies between 15 and 400 Hz were less than spherical spreading over ranges of <15 km, while at longer distances, losses were greater than spherical spreading except at the lowest frequencies (~15 Hz). For example, transmission losses at 250 Hz were approximately 110 dB at a range of 60 km, 14 dB greater than that predicted by spherical spreading and 62 dB more than cylindrical spreading. Propagation of most frequencies was better in summer than winter.

Estimates of propagation losses for an LNG carrier travelling over deep (2000 m) waters of Baffin Bay are given on Table 2.6-1, and will be used in the present assessment in the absence of empirical data on transmission losses in the Beaufort Sea. The expected noise levels at distances of 1, 4, 10, 30 and 60 km at frequencies of 25 Hz, 100 Hz, 1 kHz and 5 kHz and a receiver depth of 20 m are also shown on Table 2.6-1. At low frequencies (e.g. <500 Hz), propagation losses would be greater when the receiver was nearer than 20 m of the surface due to the Lloyd Mirror Effect. For example, at a receiver depth of 20 m, noise at 1 km produced by a carrier operating at half power in open water would be 89 dB at 100 Hz. At a receiver depth of 1 m, however, the noise level would be 84 dB.

2.6.4 Ambient Noise

Ambient noise levels are important since they frequently limit the hearing sensitivity of marine animals. As a result, an increase in ambient noise, from either natural or abnormal sources, could decrease the effectiveness of vocalizations as communication or orientation signals. The level of natural ambient noise also serves to define the zone of possible influence of industrial noise, since unusual sound sources can not be detected below ambient noise levels.

Major reviews of literature describing ambient noise in ice-free waters have been presented by Knudsen *et al.* (1948) and Wenz (1962). These authors identify three main sources of ambient noise: water motion caused by winds, tides, surf, rain and hail; soniferous marine organisms; and noise produced by ships. Geographic location, season, depth, temperature and the presence of ice are all factors which contribute to the marked variability in ambient noise levels (Wenz 1962; Milne and Ganton 1964; Piggot 1964; Myrberg 1978; Greene and Buck 1979).

During the open water season, wind-dependent sea noises and biological noise are the predominant sources of ambient noise. In areas with little industrial activity, noise spectra are relatively flat from 20 to 500 Hz, and decrease above this frequency at a rate of about 5 dB per octave (Ross 1976). Increased wind speed and sea state result in increased noise levels throughout the spectral range. Shipping noise is also a major component of low-frequency ambient noise, with its peak energy being below 100 Hz (Wenz 1962; Ross 1976). However, the shipping component of background noise levels in the Beaufort Sea is probably relatively small at the present time because of its distance from major ports and shipping routes (Fraker et al. 1981).

Although ambient noise levels in shallow, open water are highly variable (Myrberg 1978), data from the Chukchi Sea indicated that sound levels were generally lower in shallow than in deeper water (Polar Research Lab unpublish. data, cited in Fraker et al. 1981). Noise levels at the pack-ice edge are generally high, but decrease with distance from the ice edge (Diachok 1980). In general, the noise levels under shorefast ice are lower than those under pack-ice since the latter is dominated by noise produced by floes grinding under the force of the wind.

Fraker et al. (1981) measured ambient noise in August under calm sea conditions in about 25 m of water off Tuktoyaktuk Peninsula. The received spectrum level at 100 Hz was approximately 52 dB, and decreased at a level of about 5 dB per octave at higher frequencies. The ambient noise levels reported by these authors are summarized in Table 2.6-2.

TABLE 2.6-2

QUIET (SUMMER) AMBIENT NOISE LEVELS RECORDED OFF THE TUKTOYAKTUK PENINSULA (after Fraker et al. 1981)

Frequency (Hz)	dB/1μPa^2/Hz
100	52
1000	40
2000	36
4000	37
8000	24

Comparable ambient noise measurements are not available for the Beaufort Sea in winter, although Holliday et al. (1980) made a number of ambient noise recordings in shallow-water areas (8 to 45 m) under 100 percent ice cover during May at several locations near Point Barrow and Prudhoe Bay, Alaska. Spectrum levels averaged approximately 50 dB at 100 Hz and generally decreased at higher frequencies at a rate of 3 dB per octave. During brief quiet periods, the levels went as low as about 30 dB at 100 Hz. In a small, open water lead, ambient noise was as high as 68 dB at 100 Hz and 50 dB at 1 kHz, while at the edge of the shorefast ice under calm conditions, levels were approximately 54 dB and 57 dB at 100 Hz and 1 kHz, respectively.

2.6.5 Effects of Underwater Sound on Mammals

2.6.5.1 Marine Mammal Vocalizations

Many marine mammals rely on sound for transfer of social information and in echolocation (or sonar) for perception of spatial information regarding their environment. Odontocetes (toothed whales and dolphins) have developed an efficient underwater acoustical system, and probably also use environmental sounds for additional navigational cues. Although toothed whales produce sounds as low as 100 Hz, most of their vocalizations are at frequencies between 2 kHz and 75 kHz. This range corresponds well with the range of frequencies to which their hearing is most sensitive.

The extent to which odontocetes must echolocate is undoubtedly dependent on the degree of underwater visibility. Inhabitants of oceanic areas where visibility is typically good probably rely to some extent on vision for orientation. On the other hand, the Ganges River dolphin (Platanista gangetica) is totally blind and relies entirely on acoustic social signalling and echolocation for survival (Nishiwaki and Mizue 1970).

The only odontocete species which occurs regularly in the eastern Beaufort Sea is the white whale, Delphinapterus leucas. The population of white whales that migrate to the Beaufort Sea each spring has been estimated to number at least 7000 (Fraker and Fraker 1979). During July, these whales are concentrated in the highly turbid waters of the Mackenzie River estuary (EIS Volume 3A; Section 3.2) where underwater vision is undoubtedly limited. In these waters, white whales probably rely extensively on underwater acoustics for orientation as well as communication (Ford 1977).

White whales produce a variety of sounds for communication and two types of echolocation clicks. Ford (1977) describes both pure tone and relatively wide bandwidth social signals from 400 Hz to 12 kHz. Echolocation clicks were categorized as those used for general orientation, and those used for discrimination. The former consisted of click series with repetition rates of 30 to 80 clicks/sec, sweeping through frequencies from 2 kHz to 75 kHz. These click series were expected to be effective for orientation at ranges of 2 to 10 m. The 'discrimination' clicks were more rapid (about 250 per second), occurred within the range from 100 Hz to 75 kHz, and were probably effective at a target range of about 0.5 m. Peak energy of the latter sounds is in the range of 38 to 47 kHz (Ford 1977).

The underwater sounds of baleen whales (mysticetes) consist of a wide variety of low frequency (10 Hz to 2 kHz) calls, with higher frequency (3 to 30 kHz) sounds being reported for a few species (Thompson et al. 1979; Herman and Tavolga 1980). Mysticete sounds are thought to serve primarily as social or communication signals. Although there is no evidence of a well-developed echolocation system in baleen whales, it has been suggested that the echos of loud calls may be employed in general orientation to the sea bottom topography or other large targets (Herman and Tavolga 1980). There is evidence that mysticetes possess highly sensitive hearing (Herman and Tavolga 1980), and it has been postulated that these marine mammals use environmental sounds such as surf noise and vocalization of other species as an aid to navigation (Norris 1967).

The only species of mysticete that occurs regularly in the southeastern Beaufort is the bowhead whale, Balaena mysticetus. The western Arctic population of bowheads winter in the Bering Sea, and range to the Chukchi and Beaufort seas during spring, summer and fall. This population has been recently estimated to number at least 2300 whales (Braham et al. 1979).

Bowhead whales produce sounds primarily at frequencies between 50 and 600 Hz, with greatest intensities between 100 and 300 Hz (Ljungblad et al. 1980a; C. Clark, pers. comm., cited in APP 1981; Wursig et al. 1981). The phonations produced by bowheads can be classified as tonal and pulsive calls. The sound energy of tonal calls is concentrated in the 100 to 300 Hz frequency band, while pulsive calls are distributed throughout the 50 to 600 Hz band (Johnson and Clark in prep., cited in APP 1981). Ljungblad and Thompson (in prep., cited in APP 1981) also recorded a few simple moans which contained some energy at frequencies as low as 25 Hz, and occasionally energy up to 2 kHz, either as harmonics or as broad-band noise. Sound intensity levels for bowhead phonations have not been documented.

There is also no direct evidence of a well developed echolocatory system in pinnipeds, although they are probably passive listeners of environmental sounds. Most species produce underwater sounds which are likely used for social communication. In general, pinniped vocalization is predominately at frequencies below 10 kHz, although transient pulses produced by some species have higher frequency components (Møhl et al. 1975). The most important region of sound detection is between 500 Hz and 30-45 kHz (Myrberg 1978).

The two species of pinnipeds (both phocid or hair seals) which are common residents of the southeastern Beaufort Sea are the ringed seal (Phoca hispida) and bearded seal (Erignathus barbatus) (EIS Volume 3A; Section 3.2). In 1978, a year of high seal abundance, the ringed seal population in western Amundsen Gulf to 123°45'W, the Canadian Beaufort and the west coast of Banks Island (to 160 km offshore) was estimated at 61,260. The estimated size of the bearded seal population during the same year was 3109 (Stirling et al. 1981b).

Stirling (1973) describes four types of sounds produced by ringed seals during all times of the day and night and in all seasons. These sounds probably facilitate communication and social organization and include high- and low-pitched barks, yelps and chirps in the 500 Hz to 6 kHz range. The bearded seal produces a descending song that typically starts at 2 to 3 kHz and ends with a moan at 200 to 300 Hz, although most of the song occurs at frequencies above 1 kHz (Ray et al. 1969).

2.6.5.2 Hearing Sensitivity of Marine Mammals

As discussed previously (Section 2.6.1), the zone of influence of industrial underwater noise depends on numerous factors which include hearing sensitivity of the receiver, characteristics of the noise such as the source level, frequency, bandwidth and directional characteristics of the sound source, propagation and the ambient noise level at the receiver. The hearing sensitivity or hearing thresholds of several species of marine mammals have been determined, and represent the lowest intensity at a particular frequency that the test subject can hear a pure tone under quiet experimental conditions.

The auditory thresholds of two captive white whales were determined by White et al. (1978) who reported that the upper hearing sensitivity limit was 122 kHz, while the lower limit was at least as low as 1 kHz. [Background noise in the tank prevented measurements at lower frequencies.] Peak hearing sensitivity was between 20 and 85 kHz, and within this band, sound levels as low as 40 to 50 dB could be detected (Figure 2.6-3). Maximum hearing sensitivity was measured at 30 kHz where sounds as low as 36 dB were detected, while thresholds at low frequencies such as 1 kHz were within the range from 95 to 100 dB (White et al. 1978). Although hearing sensitivity at frequencies <1 kHz has not been tested for the white whale, lower frequency thresholds have been determined for the bottlenose porpoise by Johnson (1967). He reported a sensitivity of 98 dB (re 1 μPa) at 1 kHz, and this diminished steadily to 132 dB at 75 Hz, which was the lowest frequency tested. Since the audiogram of the white whale is similar to the bottlenose porpoise at higher frequencies (White et al. 1978), these figures may be representative of white whale sensitivity to low frequency sound.

Terhune and Ronald (1975a) tested ringed seal hearing sensitivities at frequencies from 1 kHz to 90 kHz (Figure 2.6-4). The lowest hearing threshold was 68 dB at 16 kHz, but relatively uniform sensitivity (68 to 81 dB) was found between 1 and 45 kHz. Above 45 kHz, the hearing threshold increased at a rate of 60 dB/octave. In a subsequent study, Terhune and Ronald (1976) reported that the upper frequency hearing limit for ringed seals was 60 kHz. Above 60 kHz, the animals tested could detect sounds, but were unable to distinguish different frequencies. The hearing abilities of phocid seals below 1 kHz have not been examined.

Figure 2.6-3 Underwater hearing threshold of white whales (adapted from White *et al*. 1978)

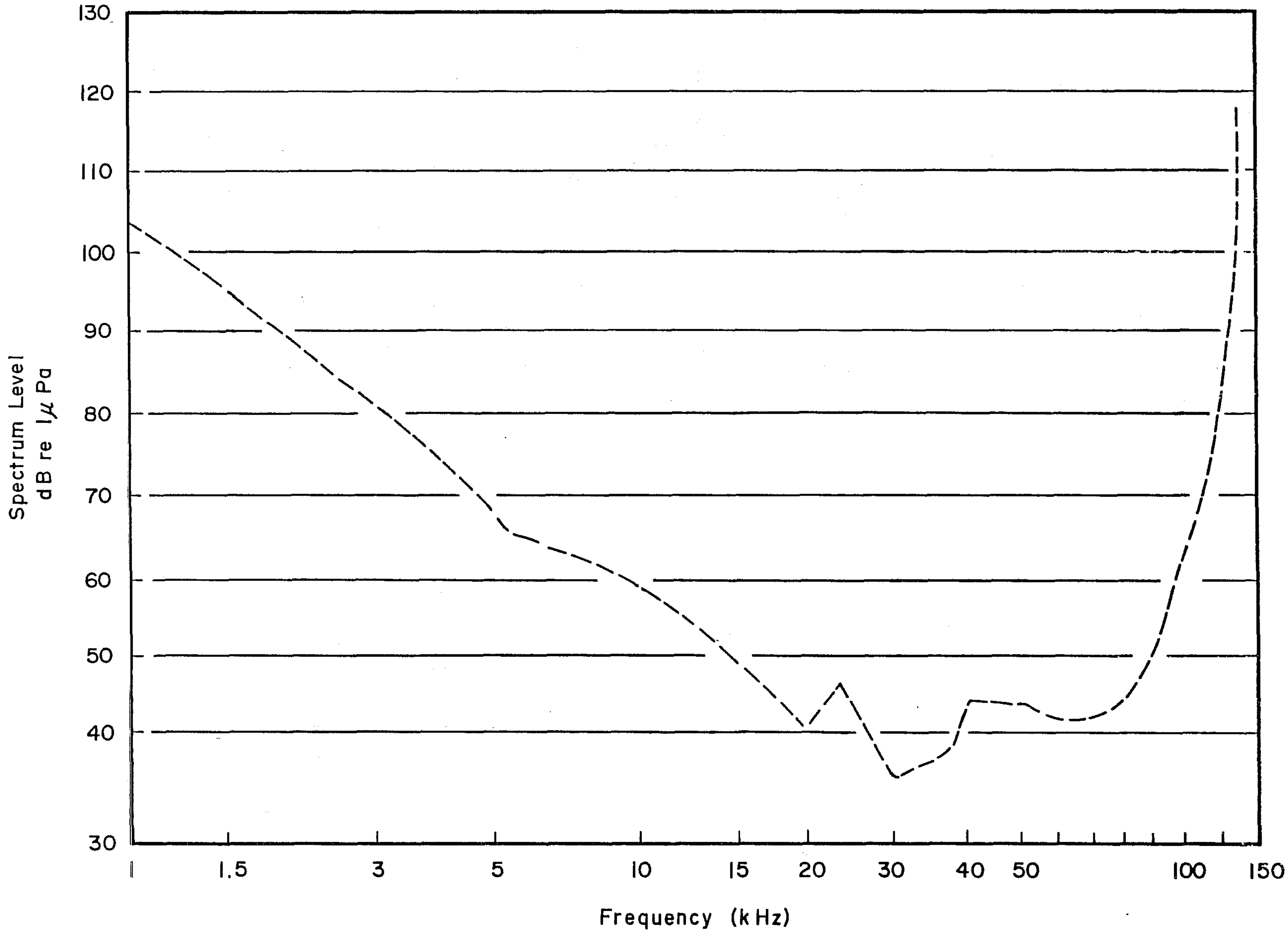
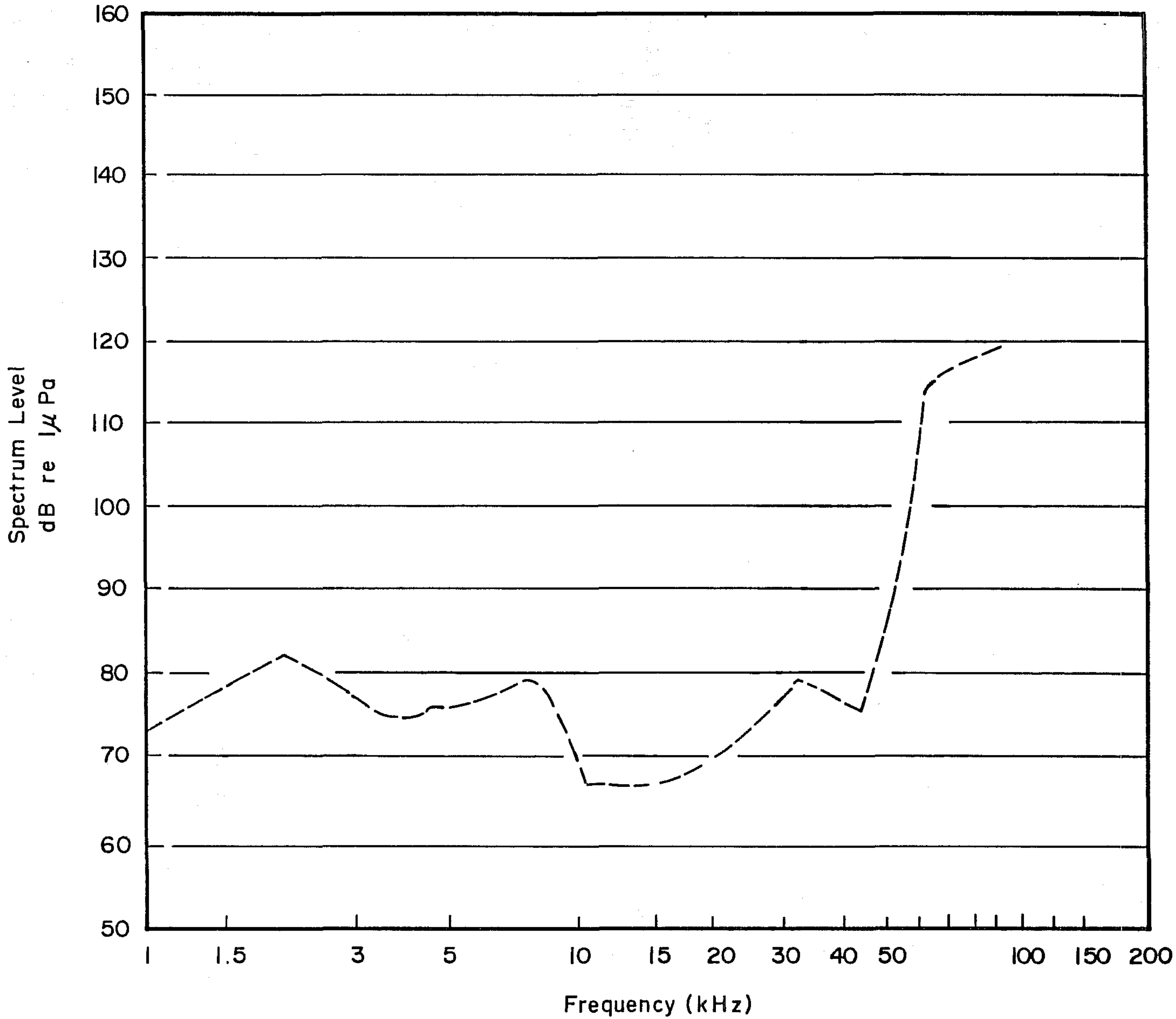


Figure 2.6-4 Underwater hearing threshold of two ringed seals (adapted from Terhune and Ronald 1975b)



The hearing sensitivity of the bowhead whale has not been determined, although it has generally been assumed that maximum sensitivity occurs in the frequency range of their vocalizations (e.g. 50 Hz to 600 Hz; Ljungblad et al. 1980; Wursig et al. 1981). Similarly, the hearing sensitivity of bearded seals has not been determined, but audiograms of five species of pinnipeds discussed by Myrberg (1978) suggest a marked similarity between each of the species tested, particularly the phocid species. Therefore, the hearing sensitivity of bearded seals is probably not unlike that previously described for ringed seals.

The ability of a marine mammal to detect a signal against a noise background is not determined by hearing sensitivity alone since there must be an adequate signal-to-noise ratio or "critical ratio". In order for a marine mammal to hear and discriminate a signal, it must be louder than background noise at that frequency. In general, the critical ratio increases with increasing frequency. For example, the critical ratio for ringed seals is between 30 and 35 dB in the 4 to 32 kHz range (Terhune and Ronald 1975a). Consequently, if the ambient noise level is 90 dB at 4 kHz, the threshold of detection of a pure tone at this frequency will be 120-125 dB. Terhune and Ronald (1975a) also discuss critical ratios determined for a bottlenose porpoise (Johnson 1968), a harp seal (Terhune and Ronald 1972), and humans (Hawkins and Stevens 1950)(Figure 2.6-5). They suggested that critical ratios of all marine mammals are probably of the same order of magnitude. When the audiogram of a particular species is available, it should then be possible to estimate its pure-tone thresholds in the presence of a specific background noise at similar frequencies.

Payne and Webb (1971) speculated that the large brains of baleen whales may have sophisticated signal processing capabilities, allowing them to detect pure tone calls of conspecifics at critical ratios of 0 dB (or at similar intensities as background noise). Although this ability has not been demonstrated experimentally in marine mammals, Payne and Webb (1971) provide a review of studies demonstrating detection of signals by humans at signal-to-noise ratios of 0 dB.

As a result of the ability of the mammalian auditory system to process sounds at different frequencies independently, vessel-induced underwater noise at low frequencies will theoretically not affect hearing at higher frequencies. This concept is related to the critical ratio and is known as the "critical band". For example, the reception of a pure tone may be affected by noise at frequencies adjacent to the frequency of the tone (Johnson 1968; Terhune and Ronald 1972; Popper 1980). Noise outside this "critical band" has little effect on tone discrimination. Scharf (1970, cited in Popper 1980) found 24 non-overlapping critical bands within the frequency range from 50 Hz to 16 kHz in humans, while Johnson (1968) found evidence of up to 40 in the bottlenosed porpoise. Terhune and Ronald (1972) estimated that the critical band of the harp seal was within 10 and 30 percent of the test frequencies. However, it should be emphasized that the concept of

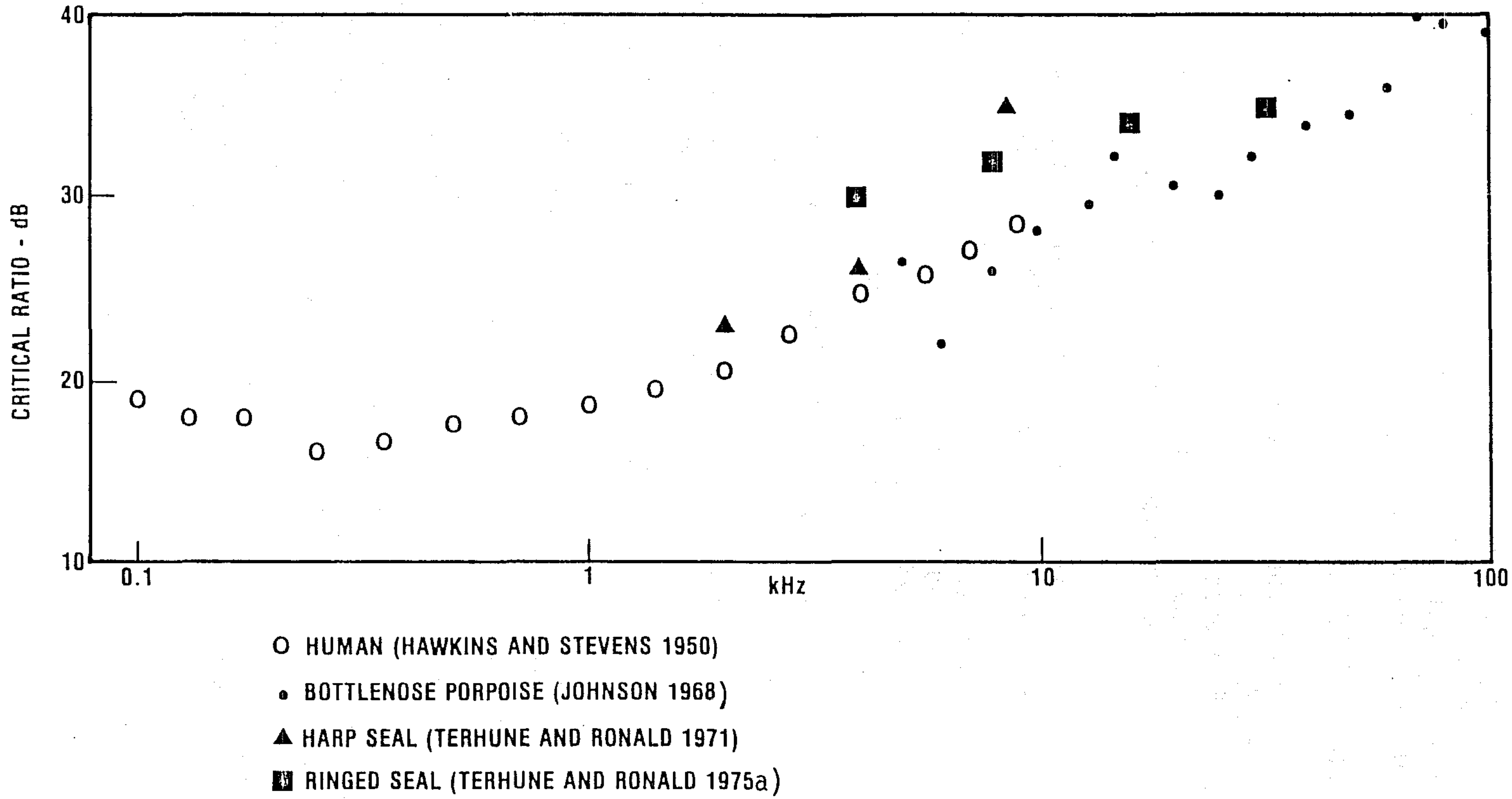


FIGURE 2.6-5 **Critical ratio of humans, bottlenose porpoise, harp seal and ringed seal**
(After Terhune 1981)

critical bands in marine mammal hearing has not been experimentally confirmed. Critical bands described to date have been derived from critical ratio data obtained during experiments using wide-band (or "white") noise sources. The masking effect of low-frequency noise on high-frequency sound reception (or high-frequency noise on low-frequency reception) has not been directly investigated with marine mammals.

2.6.5.3 Potential Biological Effects

In view of the importance of underwater sound for communication and navigation in marine mammals and the efficiency of sound transmission in water, there is a MODERATE degree of concern regarding the potential effects of industrial noise on the marine mammal populations of the Beaufort Sea. The potential effects of detectable industrial underwater noise on marine mammals may include:

(1) the presence of unfamiliar sounds which may disturb/alarm mammals and cause a startle response (Fraker 1977 a,b; Ford 1977);

(2) noise which may interfere with or mask reception of marine mammal communication or echolocation signals, or interfere with environmental sounds used by marine mammals for navigation (Penner and Kadane 1980; APP 1981; Terhune 1981); and,

(3) intense noise that could damage the hearing of marine mammals or cause other physical or physiological harm (Norris 1981).

Although the startle reflex or "fright/flight" response has not been studied *per se*, this phenomenon has been documented in pinnipeds and cetaceans by several observers and reported in numerous anecdotal accounts. The most vulnerable pinnipeds to noise disturbance may be nursing females and pups/calves, moulting individuals and animals already stressed by parasitism or disease (Geraci and Smith 1976). Repeated disturbance by industrial underwater noise may cause abandonment of preferred habitats and relocation to potentially less favourable areas (Terhune *et al*. 1979).

Some cetaceans, particularly gregarious odontocetes, respond to sudden disturbances by sounding, aggregating or dispersing, and subsequently reorganizing their group structure (Leatherwood 1977, cited in Geraci and St. Aubin 1980). Although "fright/flight" responses in cetaceans have been reported following encounters with fishing vessels (Norris *et al*. 1978) and tagging operations (Ray *et al*. 1978), there are only a few examples (Section 2.6.5.4) of this response in white and bowhead whales following exposure to mobile sources of underwater sound from offshore petroleum development activities in the Beaufort Sea (Ford 1977; Fraker 1977a,b; Fraker *et al*. 1981).

A second area of potential concern is that underwater noise associated with offshore industrial activities may interfere with or mask important sounds, particularly communication signals of whales and seals and the high frequency echolocatory sounds produced by toothed whales (Penner and Kadane 1979). In addition, masking of natural environmental sounds used by cetaceans and pinnipeds for navigation and orientation may occur in some instances. The long term significance of these potential effects can only be assessed under field conditions, but may include population decline through stress-mediated disease, decreased productivity, and displacement from favourable habitats (Geraci and St. Aubin 1980).

The third area of concern is that very intense noise, such as source levels which could be produced by proposed icebreaking oil tankers or LNG carriers, may temporarily or permanently affect the hearing abilities of marine mammals or cause other physical or physiologial harm. Terrestrial, and most likely marine, mammals have a "middle ear reflex" which de-sensitizes the animals' hearing in the presence of loud noise through the action of middle ear muscles (Norris 1981). Norris (1981) suggests that while this reflex is effective in protecting the animals' hearing from possible damage caused by loud, impulsive sounds, unnaturally long exposure to intense noise may surpass the protective capacity of this reflex.

Geraci and St. Aubin (1980), in a summary of the effects of noise on laboratory animals, state that high frequency sounds can cause noticeable physical damage to the middle and inner ear, but low frequency noise is generally less destructive and its effects are more difficult to detect. Among the possible non-auditory physiological effects of noise on marine mammals, Geraci and St. Aubin (1980) mention potential stress-mediated effects which may result in lowered resistance to disease, increased vulnerability to environmental disturbances, and endocrine imbalances which may in turn affect reproduction. In addition, Møller (1981) described a number of physiological effects of very low frequency (<20 Hz) sound (or infrasound) on humans, such as changes in blood pressure, heart rate, EEG and production of certain hormones, and suggests that similar effects are possible among marine mammals.

Numerous factors may influence the type or magnitude of the potential effects of underwater industrial noise on marine mammals. Different species use acoustic signalling for different purposes, while a variety of vocalizations at various source levels and frequencies may occur within the same species (Section 2.6.5.1). In addition, the stage in the life history of affected individuals may increase the vulnerability of some species to noise disturbances during certain periods such as breeding, feeding, moulting and migration. The duration, frequency and source level of the sound source are also important in assessment of the degree of concern regarding the potential biological effects of waterborne noise. For example, loud noise at frequencies <500 Hz may affect the ability of bowheads to communicate, but not white whales. The temporal, spatial and cumulative aspects of the industrial noise are also important considerations, as are the distribution and abundance of marine mammals within the ensonified zone. Finally, mobile and stationary noise sources may elicit different responses from the same species (see Fraker et al. 1981).

An additional factor which must be considered during assessment of the degree of concern associated with underwater sound is the rate of recovery following a short-term disturbance, or the acclimation period in response to a chronic source of noise. Some marine mammals become habituated to low-level background noises such as those which may be associated with ship traffic and offshore petroleum activities (Geraci and St. Aubin 1980; Davis 1982; Evans 1982). For example, humpback and gray whales, harbour and elephant seals, bottlenose dolphins, walruses and sea lions apparently coexist with human activities (Geraci and St. Aubin 1980). However, Nishiwaki and Sasao (1977) suggest that the increased ship traffic near Tokyo Bay has displaced migrant minke and Baird's beaked whales, while Norris and Reeves (1977) documented a short-term displacement of gray whales from a lagoon in Baja California as a result of a temporary increase in industrial activity.

2.6.5.4 Documented Responses of Marine Mammals to Industrial Underwater Noise

The generalized frequency ranges of marine mammal vocalizations, hearing sensitivities and underwater industrial noise are summarized in Figure 2.6-6. The majority of sound energy emanating from industrial activities occurs at frequencies <2 kHz, although Ford (1977) documented occasional energy up to 15 kHz and Fraker et al. (1981) reported appreciable energy above ambient levels up to 8 kHz. These data clearly indicate that the vocalizations (and therefore probably hearing sensitivity) of the bowhead whale closely correspond to the frequencies of industrial noise. There is also some overlap between the hearing sensitivities of ringed seals, white whales and probably bearded seals with low frequency industrial sounds. It should be emphasized, however, that the hearing sensitivities of these species at frequencies below 1 kHz are still largely unknown. The zone of influence depends on the hearing sensitivity of the species at that frequency, the critical ratio assumed for that species, and ambient noise levels.

There has been limited direct study of the effects of underwater industrial noise disturbances on marine mammals. In addition, the long-term or cumulative impacts of underwater industrial noise on these species have not been documented since effects of this type can only be assessed over several years through intensive field investigations. As indicated by the data provided in Table 2.6-3, noise from industrial activity in the Beaufort Sea has resulted in undetectable or only short-term disturbance of white and bowhead whales. Comparable information on the potential disturbance of ringed and bearded seals as a result of underwater industrial noise in the Beaufort Sea is not available, although Ward (1981) reported seals in McKinley Bay were undisturbed by dredging activities therein during July and August 1981.

TABLE 2.6-3

OBSERVED RESPONSES OF BOWHEAD AND WHITE WHALES TO UNDERWATER INDUSTRIAL NOISE IN THE BEAUFORT SEA

Underwater Sound Source	Source Levels/ Frequencies	Location/Date	Response
<u>White Whales</u>			
Moving barge tow (a number of barges towed as a unit)	Waterborne sounds from logistics traffic at Tuft Point (Ford 1977): tugs pushing full barges: peak levels 164 dB re 1μPa (at 1 m); most energy between 250-3000 Hz. Ford calculated white whale range of audibility of the barge to be 3.3 km at 3 m depths under quiet ambient conditions (50-60 dB)	'Niakunak' Bay July 11, 1976	The few white whales within 0.8 km of the tow and whales within 2.4 km radius of the tow moved away. Beyond 2.4 km, they were apparently undisturbed. The disrupted distribution was still evident after 3 h, but not after 30 h (Fraker 1977a). <u>Short-term disturbance</u>
Concentrated barge traffic (eg. 25/day for 2 week period)		Along Tuktoyaktuk Peninsula near Tuft Point July 20, 1976	Approx. 100-150 white whales remained in Beluga Bay (4 km east of Tuft Point) for a 2 week period 5 days after the heavy barge traffic began, but left when the barge movements temporarily ceased. No significant interference or concentration of whales occurred in Beluga Bay in 1978 and 1979 when barge traffic was only about 6/day (Fraker 1977a,b). <u>Short-term disturbance</u>

TABLE 2.6-3 (cont'd)

Underwater Sound Source	Source Levels/ Frequencies	Location/Date	Response
Stationary dredge	Composite noise from dredge, tugs and barges at Tuft Point: peak source levels 155 dB re 1 μPa (at 1 m); most energy between 250 Hz and 2 kHz; occasionally to 5 kHz (Ford 1977).	Tuft Point July 1976	White whales moved within 400 m of dredge, apparently undisturbed. However, in one case when a barge tow began to move toward the whales, they moved from the track (Fraker 1977a). Negligible disturbance
Drilling at Adgo J-37; helicopter traffic	n/a	SW of Garry Island Summer 1979	White whales observed within 400 m of drilling activity and were apparently undisturbed. In addition, helicopter traffic had no observable effects on whales (Fraker and Fraker 1979). Negligible disturbance
Artificial island Netserk B-44	n/a	NW of Garry Island Summer 1975	White whales frequently observed within 100 m of artifical island (Fraker 1977b). Negligible disturbance
Presence of artificial island Kannerk G-42 along whale travel route	n/a	Off McKinley Bay on Tuktoyaktuk Peninsula	No detectable effect on whale movements (Fraker 1977b). Negligible disturbance

TABLE 2.6-3 (cont'd)

Underwater Sound Source	Source Levels/ Frequencies	Location/Date	Response
Single piston aircraft	n/a	Mackenzie Estuary	Repeated flights at 305 m ASL produced avoidance; altitudes of 457 m ASL used during observation periods with no apparent disturbance (LGL Ltd., unpubl. data, cited in Fraker *et al.* 1981). Short-term disturbance
Bowhead Whale			
Construction of Issungnak 0-61	Well above ambient to 4.6 km north; at 1-2 km sounds received were 20-50 dB//($1\mu Pa^2$)/Hz above ambient up to 8 kHz (dredge was operating). At another island construction site, dredge and support vessels producing 90-100 dB//($1\mu Pa^2$)/Hz at 1 kHz and frequencies below at a range of 550 m (Fraker *et al.* 1981)	32 km north of Pullen Island in Mackenzie Delta (Surveys August 5,9,11,12,22,1980)	During 6 aerial surveys, a total of 20 bowheads were seen within 5 km of the island (as close as 800 m) and a total of 64 were seen within 20 km. Industry personnel reported 18 sightings of one or more whales (some within 0.5 km) from dredges, vessels or barge camps. One whale was reported 16 m from the barge camp. Whales appear to tolerate the presence of the island, boats, dredges, etc. and the associated noises (Fraker *et al.* 1981). Negligible disturbance

TABLE 2.6-3 (cont'd)

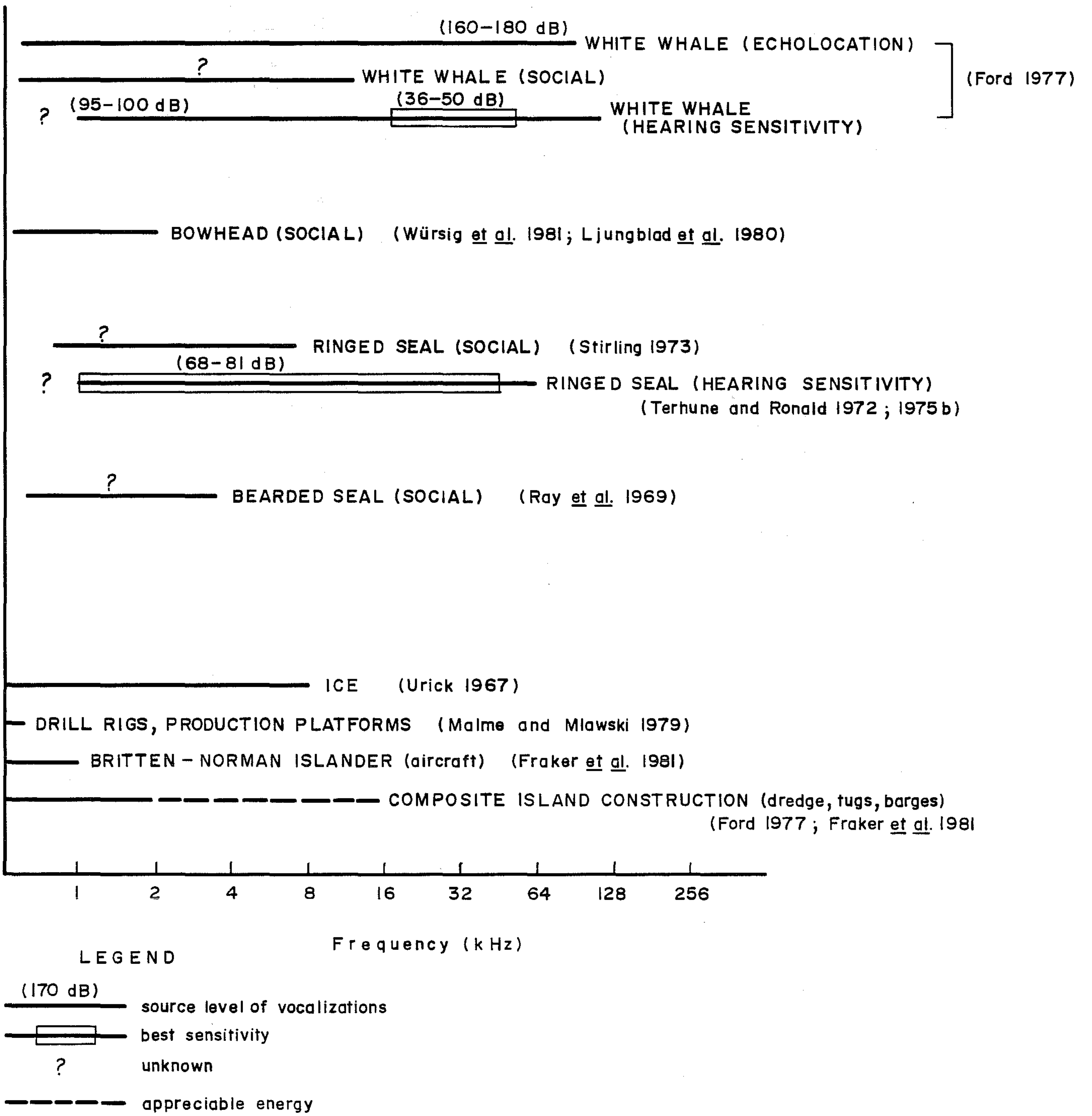
Underwater Sound Source	Source Levels/ Frequencies	Location/Date	Response
IMPERIAL ADGO (16 m crew boat)	Most energy below 2 kHz, occasionally to 4000 Hz. Strongest tone at 90 Hz [107.5 dB//($1\mu Pa^2$)/Hz] at full speed at 200 m. From 1 to 4 kHz, sound pressure levels about 20-30 dB above quiet ambient.	off Tuktoyaktuk Peninsula August 23,24,26, 27, 1980	Bowheads orientated themselves away from ADGO when it was moving and at a distance within 900 m less intense response when ADGO was idling or more than 900 m away; negligible response when stationary with engines off (Fraker et al. 1981). Short-term disturbance
			Four bowheads responded to the boat by spending less time at the surface and scattering when boat was 3.7 km away; greater effects when boat approached and passed whales. Recovery was observed in post-disturbance period (Fraker et al. 1981). Short-term disturbance
CANMAR SUPPLIER Supply Ship - Class 2 ice-breaker (56-62 m class)	Recorded at 200 m: strongest tone at 56 Hz [116 dB//($1\mu Pa^2$)/Hz] Sounds produced 30-40 dB above quiet ambient throughout spectrum up to at least 8 kHz	18 km E of Pullen Island August 19, 1980	Fifteen bowheads did not react until vessel was within 800 m; initial response to 'outrun' it and to scatter as it came closer. Three hours after the disturbance, the bowheads were still present in the area (Fraker et al. 1981). Short-term disturbance

TABLE 2.6-3 (cont'd)

Underwater Sound Source	Source Levels/ Frequencies	Location/Date	Response
BRITTEN-NORMAN ISLANDER aircraft (2 piston engines)	Aircraft at an altitude of 610 m, sound levels at hydrophone as loud as 97 dB//1μPa^2/Hz at 70 Hz: 10-20 dB above ambient at frequencies <1 kHz	Conducted opportunistically during aerial surveys (Wursig *et al*. 1981)	Whales dove when circled by aircraft ≤305 m ASL, but no response was observed in bowheads circled at an altitude of 457 m ASL or greater. (Fraker *et al*. 1981). Short-term disturbance
TWIN OTTER aircraft (2 turbo prop engine)	n/a	Altitude 305 m (Beaufort Sea)	Nearly all bowheads circled by a Twin Otter at 305 m or less dove (Fraker unpubl., cited in Fraker *et al*. 1981).
		Altitude 90 m (eastern Arctic)	Always dove (Koski, pers. comm., cited in Fraker *et al*. 1981)
		Altitude 150 m (eastern Arctic)	Usually did not dive in first pass (Koski, pers. comm., cited in Fraker *et al*. 1981).
		Altitude 91-152 m (north of Alaska)	Always dove (Ljungblad *et al*. 1980) Short-term disturbance

n/a = not available

Figure 2.6–6 Frequency ranges of most waterborne sounds produced by marine mammals and industrial activities, and hearing sensitivities of white whales and ringed seals.



(a) White Whales

Although lack of quantitative data and systematic surveys limits interpretation of the anecdotal information provided in Table 2.6-3, one study indicated that white whales in waters <2 m deep in the Mackenzie Estuary reacted to 2 barges and a tug at distances within 2.4 km (Fraker 1977a). This distance is considerably less than the 3.3 km audible range for white whales and barge traffic estimated by Ford (1977). Fraker and Fraker (1981) suggested that the whales may therefore have tolerated a certain level of disturbance produced by marine logistics traffic. Artificial islands per se have had no apparent effects on white whales, since they have approached and migrated past stationary sound sources in the Beaufort Sea (islands, drilling rigs, dredges) on several occasions (Fraker 1977b). In addition, fixed wing survey aircraft flying at altitudes of 457 m and higher have caused no apparent disturbance of white whales in the southeastern Beaufort Sea.

(b) Bowhead Whales

As indicated in Figure 2.6-6, there is considerable overlap between the frequencies of bowhead vocalizations and the noise produced from all phases of industrial activity. Since low frequency sounds are propagated well and it is assumed that bowhead hearing sensitivity is within the range of their vocalizations, this species is probably the most sensitive marine mammal to underwater industrial noise. Industrial noise will result in an increase in ambient noise levels, although certain transitory natural events are louder. Depending on assumptions made regarding the signal-to-noise ratio required by bowheads, the increased noise levels associated with industrial activity in the Beaufort Sea could reduce conspecific communication distance through masking. The extent and biological significance of potential masking effects are unknown.

The existing information on the responses of bowhead whales to various noise sources (Table 2.6-3) suggests that this species is not disturbed by stationary activities such as artificial island construction and dredging, but may react to mobile logistics traffic under certain conditions. However, marked responses have only been observed when the sound source approaches within about 1 km of the whale(s). The lack of systematic studies prevents accurate assessment of the results summarized in Table 2.6-3, although the available data suggest that this species is probably less easily startled or frightened by marine logistics traffic than white whales. Aircraft flying at altitudes of 100 to 150 m ASL probably also produce a disturbance or avoidance response in bowheads (diving), although flights at altitudes higher than 150 m lead to variable responses which probably depend on numerous factors such as ambient noise, propagation of the underwater sound and the specific activities of the whales at the time of disturbance.

(c) Seals

Low frequency underwater industrial noise may reduce communication distances between seals, elicit a startle response, or result in exclusion of individuals from their territories. Although these effects have not been observed in the Beaufort Sea, Terhune et al. (1979) reported reduced harp seal vocalization in the Gulf of St. Lawrence in response to vessel activity. The motor noise from an operational vessel masked the seal calls within a 2 km radius, but the authors were not certain if the decreased vocalizations may have reflected changes in behaviour or movement of seals from the area. Stirling (1973) reported increased amounts of ringed seal vocalizations during periods of ice cover, and suggested this may reflect a necessity for maintenance of social order at breathing holes and leads in comparison to open water periods when individuals are widely dispersed. Consequently, the potential effects of masking seal vocalizations may be more pronounced during periods of ice-cover, particularly if industrial noise is propagated under the landfast ice where ambient levels are typically lower and have fewer components than at the ice edge or within the transition zone.

Underwater industrial noise may also cause short-term disturbance or masking of communication sounds in bearded seals. However, because transition zone and pack-ice habitats occupied by bearded seals are naturally noisy, low frequency sounds would have to be proportionately louder than noise in landfast areas to affect seals. The biological significance of startle responses and masking of vocalizations by both ringed and bearded seals are unknown.

2.6.6 Effects of Underwater Sound on Fish

Hearing in fish involves an inner ear mechanism coupled to the pressure-sensitive lateral line system. In many species, including cod and herring, the gas bladder also contributes to sound reception via the inner ear (Tavolga 1971). Although the significance of hearing in fishes is not well understood, intraspecific communication in temperate areas has been documented in several of the relatively few species investigated (Myrberg 1978). In general, hearing in most fishes is thought to be associated with sensing and locating approaching obstructions, prey or predators, and maintenance of orientation in the water column (Tavolga 1971).

The sensitivity of fish hearing depends on a complex relationship between the frequencies and intensities of sounds. Most fish are sensitive to a range of frequencies, usually up to 2 kHz, and a few species (including herring) can distinguish sounds over 5 kHz (Myrberg 1978). The ability of fish to distinguish sounds also depends on the intensity of the sound and the ambient noise level. For example, under naturally occurring conditions in temperate environments, the noise of background sea states appears to often mask fish hearing abilities (Myrberg 1978). An increase in background noise resulting from industrial underwater sound could therefore reduce the perception of other sounds by fish.

The effects of industrial or vessel associated noises on fish have rarely been studied, although it has been postulated that these sounds could affect fish movements, distribution and behaviour. Available information on ambient noise and the industrial sounds produced in Arctic enviroments was summarized earlier in Section 2.6.1. Sounds generated as a result of drilling, dredging, icebreaking and other activities in the range of frequencies heard by fish may be up to 80 dB above background levels near the noise source. These sounds may not be fully attenuated to levels below fish hearing thresholds for several kilometres from the source in shallow environments and even further in deeper water.

The ecological significance of industrial sound perception by fish is not clear since variable responses and habituation to underwater sound have been observed in a number of instances. The presence of fish within the immediate vicinity of active harbours and fishing grounds has also been clearly documented, while fish have been reported near active dredges in the Beaufort Sea (Byers and Kashino 1980). On the other hand, some authors have suggested that fish avoid noise associated with dredging operations and large vessels (Neproshin 1978; Konagaya 1980). In a review of literature describing the reaction of fish to sound, Chapman and Hawkins (1969) concluded that intermittent high amplitude sounds at low frequencies generate avoidance responses. For example, they indicated that whiting (Merluccius sp.) reacted quickly by diving after the firing of an airgun, but habituated to its continuous firing in less than 1 hour. Olsen (1975) reported that herring will locate and avoid similar noise sources, but also habituate to noise relatively rapidly if the signals occur less than several minutes apart. Popper and Clarke (1976) reported that goldfish exposed to intense sound (149 dB) for 4 hours experienced a 24 hour hearing loss, while Chapman (1975) suggested that fish may be generally tolerant of high intensity sounds (130 to 140 dB) within their hearing range.

Less obvious effects of sound were postulated by Banner and Hyatt (1973) who investigated the viability and growth of eggs and larvae of two species of estuarine fishes (genus Cyprinodon, Fundulus) in aquaria under two levels of underwater noise. They reported that the higher noise level resulted in significantly reduced viability of eggs and larvae of one species, and reduced growth of fry of both species. However, these results cannot be extrapolated to ocean conditions since the noise levels, even in the quiet tank, were well above ambient noise in the ocean under heavy traffic conditions.

Overall, the available information describing the effects of noises generated by industrial activities on fish is ambiguous, and this hampers assessment of the potential concern associated with the effects of underwater sound on species present in the Beaufort Sea. It is likely that fish will hear noise from drilling, vessels and other sources over distances of several kilometres, and while some fish may avoid the immediate areas of chronic or intermittent high amplitude sounds, very few individuals are likely to be affected in a regional context. In addition, it appears probable that many species will become habituated to stationary and relatively continuous noise sources.

2.6.7 Summary of Concerns Related to Underwater Sound

Most marine industrial activities create unnatural underwater noise. The frequencies and levels of noise vary with the type of activity, while the area affected by underwater noise depends on the sound propagation characteristics of the region. The degree of concern regarding the potential effects of industrial underwater noise on marine resources of the Beaufort Sea varies with species, and in most cases, also reflects the general lack of information regarding the biological significance of noise-related disturbances. Nevertheless, the degree of concern associated with adverse effects of underwater sound on fish is expected to be NEGLIGIBLE to MINOR (Table 2.6-4). In the case of marine mammals, the degree of potential concern is greater since these species have sensitive underwater hearing and make use of waterborne acoustic signals for social communication and, in the white whale, navigation and orientation (Table 2.6-3). Underwater industrial noise may have three types of potential effects on marine mammals: (1) unfamiliar sounds may cause alarm or startle reactions; (2) noise may mask or interfere with communication and/or echolocation, and (3) prolonged exposure to intense noise may damage the hearing of marine mammals or result in other physical/physiological effects. These three effects are likely inter-related and may result in similar disruption and avoidance responses.

Observed interactions between marine mammals and industrial noise sources in the Beaufort Sea indicate that marked behavioural responses generally involve mobile sound sources, with stationary sources appearing to have little or no effect on either seals or whales. In addition, reactions to mobile sources have been localized and short-term, and have varied with the proximity of source and level of noise. Noise sources which have been observed to cause avoidance responses by marine mammals in the Beaufort Sea include crew and supply vessels, tugs and aircraft at low altitudes (<300 m).

Although industrial activity in the Beaufort Sea has resulted in only MINOR and localized noise-related disruption of marine mammals, an increase in the amount and duration of activity during future development may result in greater impacts over the long-term. Marine mammals are able to tolerate or habituate to noise of certain levels and characteristics, but it is possible that there is some tolerance limit which, once surpassed, results in avoidance of high-noise areas. For example, if underwater noise becomes widespread in critical areas (e.g. concentration, feeding or migration areas), significant disruption and displacement of portions of regional seal and whale populations may occur.

TABLE 2.6-4

SUMMARY OF POTENTIAL REGIONAL CONCERNS OF UNDERWATER SOUND IN THE BEAUFORT SEA PRODUCTION ZONE ON MARINE MAMMALS AND FISH

Species	Tanker Activities		Marine Logistics Traffic		Regulated Aircraft Activity and Stationary Sources	
	Early Production	Production after 1990	Early Production	Production after 1990	Early Production	Production after 1990
White whales*	MINOR	MODERATE	MINOR	MODERATE	NEGLIGIBLE	MINOR
Bowhead whales*	MINOR	MODERATE	MINOR	MODERATE	NEGLIGIBLE	MINOR
Ringed and Bearded Seals**	MINOR	MODERATE	MINOR	MODERATE	NEGLIGIBLE	NEGLIGIBLE
Fish**	NEGLIGIBLE-MINOR	NEGLIGIBLE-MINOR	NEGLIGIBLE-MINOR	NEGLIGIBLE-MINOR	NEGLIGIBLE	NEGLIGIBLE

* All seasons but winter.
** All seasons.

The potential for disturbance of marine mammals by underwater noise will depend on the type and level of noise, as well as the number and distribution of sources. In addition, each species likely has a different susceptibility to industrial underwater noise. White whales are highly social and undoubtedly rely extensively on underwater acoustics for communication and navigation, particularly in turbid areas such as the Mackenzie Estuary, and noise masking could reduce the effectiveness of their acoustic signals. This species is most vulnerable to disturbance in their concentration areas and along migration routes during the spring and summer months. Bowhead whales are less gregarious, but do concentrate on offshore feeding grounds in the eastern Beaufort Sea during summer months. The low-frequency communication signals of bowheads, which are susceptible to masking by underwater noise, may be used to locate concentrations of conspecifics in prime feeding habitats. Avoidance reactions to vessels have also been observed in this species, and concentrated traffic in feeding areas may result in disruption of normal activities.

Unlike whales, ringed and bearded seals are present in the Beaufort Sea throughout the year and could be exposed to industrial noise over longer periods. These species also produce vocalizations which may be masked by industrial noise. However, due to the widespread distribution of seals throughout the region, smaller proportions of the populations would be susceptible to disturbance at any given time.

The degree of concern regarding adverse effects of the present level of industrial underwater noise on marine mammals in the Beaufort Sea region is probably NEGLIGIBLE to MINOR. However, industrial underwater noise associated with the long-term proposal for hydrocarbon development in this region is considered an area of MODERATE concern since the long-term and cumulative effects of noise-related disturbance of marine mammals remain unknown.

2.7 AIRBORNE NOISE

2.7.1 Introduction

Aircraft and other sources of airborne noise associated with offshore hydrocarbon exploration and production may cause localized disturbances of marine mammal and bird populations in the southeastern Beaufort Sea. The types of aircraft used by the petroleum industry include turbojets (Boeing 737, Cessna Citation and Lear jets), turboprops (Twin Otter, Hercules and Grummen aircraft and most helicopters including Bell 206 and 212 and Sikorsky S-76 and S-61), and piston-engined aircraft (Cessna 150 and Douglas DC-3). Airstrips would be established at each shorebase, and most runways would be capable of handling Boeing 737 and 767's. Composite noise sources produced at an airstrip facility are described in Volume 2 of the EIS.

Other sources of mobile airborne noise include on-ice vehicular traffic, and seasonal and icebreaking marine vessels, while major stationary sources of airborne noise include artificial island construction activities, dredges and drilling rigs. The potential effects of atmospheric emissions from these noise sources are separately discussed in Section 2.8.

2.7.2 Effects of Airborne Noise on Mammals

Mammals that occur in terrestrial areas or on the ice surface may be affected by noise produced by aircraft and other industrial activities. In the Beaufort Sea region, marine mammals or marine-associated mammals which are potentially susceptible to these disturbances include ringed seals, bearded seals, polar bears and Arctic foxes. The potential effects of underwater sound generated from the above and other sources on marine mammals were previously discussed in Section 2.6.

Ringed and bearded seals may be susceptible to disturbance from airborne noise when they are hauled-out on the sea-ice during the annual moulting period. Hauled-out ringed seals occur on the landfast ice in large bays of Amundsen Gulf and off the west coast of Banks Island (Stirling et al. 1981a), while bearded seals typically haul-out on the transition zone ice or on nearshore pack-ice during late June. Both species have been observed to dive when approached by low-flying (e.g. 100 m ASL) survey aircraft (LGL Ltd., unpubl. data). This diving response occurred more frequently during helicopter flights (G. Alliston, pers. comm.). During five years of aerial surveys flown in Cessna 337's flying at an altitude of 152 m (91 m in fog) and an airspeed of 220 km/hr, Stirling et al. (1981a) found that a proportion of the hauled-out seals in mid-transect always dove in response to the aircraft. The percentage of seals responding has not been quantified, but appears to be dependent on airspeed, altitude and type of aircraft (I. Stirling, pers. comm.).

The potential effects of repeated immersion of moulting seals in response to aircraft disturbance during the haul-out period are unknown, but may include thermoregulatory stress. The annual moult is known to be a period of natural stress in seal populations. The decrease in the amount of blubber during spring can reduce the weight of seals by 23 to 40 percent, and much of their behavior during this period is associated with thermoregulatory adjustments (McLaren 1958; Smith 1973; Finley 1979). Consequently, the potential degree of concern with respect to effects of aircraft noise on moulting seals is considered MINOR.

The potential effects of other mobile sources of airborne noise on seals are not known, but a diving response may also be induced if the noise is sudden and the source is at close range. Stationary sources of airborne noise may temporarily alter the distribution or abundance of these species within the immediate area of disturbance, although the degree of concern regarding potential effects of these noise sources on regional populations of ringed and bearded seals is probably NEGLIGIBLE.

Polar bears and Arctic foxes are most susceptible to noise disturbances from low-flying aircraft during winter and spring when they forage on transition zone and landfast ice. During the open water period, most polar bears move north with the retreating pack-ice, while Arctic foxes move ashore to den. Polar bears (and probably Arctic foxes) usually run from low-flying aircraft, although bears will occasionally react aggressively (G. Alliston, pers. comm.). Other mobile sources of airborne noise may produce similar responses in foxes and bears, while stationary noise sources (e.g. drill rigs) may actually alert and attract these species to sites of human activity. The biological significance of these responses is unknown, but the degree of concern for the regional population is considered MINOR because the number of individuals affected would be small.

2.7.3 Effects of Airborne Noise on Birds

Disturbance of birds by airborne noise produced by helicopters and STOL aircraft required in support of the Beaufort Sea development is a MAJOR area of concern due to the extensive projected use of these aircraft and the documented susceptibility of some species to airborne disturbances. Noise produced by large passenger jets and executive jets is only expected to be a MINOR and localized concern because they would fly at high altitudes except near the airstrips.

The reaction of birds to aircraft depends on species, previous exposures, stage of their annual cycle, type of aircraft and the vertical and horizontal distance of the aircraft from the birds. Species most likely to be affected by aircraft activity are those that nest colonially or in restricted areas, or occur in large concentrations during staging or moulting periods either offshore or along the mainland coast. These groups include several species of waterfowl, glaucous gulls, Arctic terns and thick-billed murres. Since the murres only nest at one colony in the region (Cape Parry), they are only likely to be affected by aircraft disturbance if Wise Bay is developed as a contingency fuel storage site (EIS Volume 2). Aircraft disturbance may cause: (1) habitat loss through exclusion of birds from frequently disturbed areas; (2) increased energy expenditures, and (3) behavioural reactions that may increase mortality rates of young.

Gollop et al. (1974a) described disturbances of the nesting activities of common eiders and glaucous gulls at Nunaluk Spit in the Yukon as a result of overflights by fixed-wing aircraft and a helicopter. None of the 34 incubating eiders under observation during this study were disturbed by either the helicopter or fixed-wing aircraft overflights at altitudes as low as 40 m. However, an increasing proportion of the gulls incubating the 39 observed nests flushed during progressively lower helicopter overflights at altitudes below 600 m. All incubating gulls were flushed by the helicopter at and below altitudes of 150 m, but overflights by the fixed-wing aircraft had virtually no effect on the incubating birds. During the pre-nesting and nesting seasons, Arctic terns and snow geese left their nesting areas adjacent to the Beaufort Sea when fixed-wing aircraft or helicopters flew at altitudes less than 300 m AGL and at horizontal distances less than 2.4 km from these birds (Barry and Spencer 1971; Gollop et al. 1974a). Repeated flights of a Bell 206 helicopter over tundra nesting areas resulted in a delay in nest initiation and reduced reproductive success of Lapland longspurs (Gollop et al. 1974b). There was also evidence of reduced nesting density in this local population during the year following the disturbance (Gunn et al. 1974). The potential short- and long-term effects of flushing incubating or nesting birds are unknown, but may include reduced reproductive success through increased nest predation or destruction of young, increased energy requirements, and delays or prevention of hatching. Populations of birds nesting in colonies are particularly susceptible to this form of disturbance because large numbers of birds may be repeatedly affected.

Dunnet (1977) studied the effects of aircraft disturbance on a colony of various cliff-nesting seabirds in Scotland. Species at this colony included fulmars, shags, herring gulls, kittiwakes, guillemots, razorbills and puffins, and aircraft flying at altitudes as low as 100 m above the cliff-top did not affect the attendance of incubating and brooding birds. Non-nesting birds resting on nearby cliffs flushed in response to the aircraft, but also did so when no observable stimulus was present. Ward (1979) reported that thick-billed murres from the Cape Parry colony did not flush when a helicopter (Bell 206) landed 0.8 km away, although some birds left the colony when a DC-3 passed about 500 m seaward of the colony at an altitude of 300-500 m ASL. W. Renaud (LGL Ltd., pers. comm.) found that thick-billed murres at the Cape Hay (Bylot Island) colony did not flush when a de Havilland Twin Otter flew by the cliff-top nests at a horizontal distance of about 2 km.

Brood rearing geese are usually more sensitive to aircraft disturbances than brood rearing ducks. For example, several small lakes in the Parsons Lake area were overflown several times daily at altitudes ranging from 30 to 150 m to determine the effects of helicopter (Bell 3B1 and/or Bell 204) disturbance on various species (RRCS 1972). The helicopter flights had no detectable effects on the number of Arctic loons, whistling swans, pintails, American wigeon, scaup, oldsquaws and white-winged scoters frequenting lakes in the area. However, the number of non-breeding white-fronted geese in the experimental area decreased markedly within two days after the disturbance began, although too few observations were made in the control area to permit statistical evaluation of this trend. Nevertheless, white-fronted geese that remained in the experimental area included adults with young. Overflights by fixed-wing aircraft at an altitude of 90 m (1.5 h apart) only caused a short-term (2 min) interruption of normal activities of about 100 American wigeon present on a lake in the Mackenzie Delta (RRCS 1972). Similarly, brood rearing ducks did not abandon a lake subjected to hourly landings of a fixed wing aircraft (Cessna 185) over five hours each day for a period of several days (Salter and Davis 1974).

The reaction of moulting ducks without broods to aircraft is variable. For example, moulting ducks left the lake which was subjected to disturbance by the Cessna 185 landings described above (Salter and Davis 1974). Hourly helicopter flights along the southwest shore of Herschel Island at altitudes ranging from 30 to 230 m did not cause an overall reduction in the number of moulting sea ducks (primarily oldsquaws and surf scoters) using the area in 1972 (Gollop et al. 1974c). However, even the highest overflight caused most of the ducks on shore to move into the water. Ducks began to dive in response to the overflights when flight altitudes decreased to 150 m, while aircraft at altitudes of 30 m caused entire flocks to dive. Gollop et al. (1974c) reported that surf scoters were generally more sensitive than oldsquaws to disturbance from helicopters. Ward and Sharp (1974) also reported temporary disturbance of moulting oldsquaw and scoters at Herschel Island by helicopters at altitudes of 100 m ASL and 45 m from shore. However, it was believed that the ducks did not leave the area since their numbers increased as the study progressed, irrespective of the helicopter flights (Ward and Sharp 1974).

Several studies have shown that migrating and staging snow geese are particularly sensitive to aircraft disturbances. Campbell and Shepard (1973) reported that spring migrants flushed when aircraft passed at altitudes of 300 m AGL and horizontal distances of 3.2 km. Schweinsburg (1974) also found that all resting snow geese in his study area along the North Slope were repeatedly disturbed by a Cessna 185 at altitudes ranging from 90 to 3050 m. Aircraft overflights at altitudes of 90 to 120 m and distances of 1.6 to 8 km flushed all flocks, while aircraft at 210 m flushed all flocks at distances from 3.2 to 14.4 km. Overflights at 305 m and horizontal distances of 3.2 to 5 km flushed all flocks, while flights at altitudes from 1800 to 3050 m caused flocks to flush either as the aircraft approached or when it was directly overhead (Schweinsburg 1974). Davis and Wiseley (1974) studied the effects of fixed-wing aircraft and helicopter disturbance on the behavior of fall staging snow geese on the Yukon North Slope. Geese flushed at greater distances from helicopters than from fixed-wing aircraft, but their normal behavior was disrupted for longer periods following overflights by fixed-wing aircraft. Koski and Gollop (1974), Alliston et al. (1976) and McLaren et al. (1977) have also reported that fall staging or migrating snow geese flush far ahead of survey aircraft.

Other species of waterfowl may react to aircraft disturbance during migration. For example, king and common eiders flush far ahead of survey aircraft (Davis et al. 1974; Alliston et al. 1976), while Campbell and Shephard (1973) found that migrant whistling swans, Canada geese, white-fronted geese and several species of ducks were relatively tolerant of survey aircraft during the spring. The responses of some species (e.g. oldsquaws, eiders, gulls, loons) to airborne noise when they are staging in the open leads and polynias of the southeastern Beaufort Sea during spring remain unknown. The susceptibility of different species of birds to aircraft disturbances during specific times in their life histories were summarized by Slaney (1974a) following 3 years of unstructured observations in the Mackenzie Delta and are indicated in Table 2.7-1.

The response of birds in the Beaufort Sea region to airborne noise associated with industrial activities other than aircraft remains largely unknown. Gollop and Davis (1974) studied the effects of a gas compressor station simulator on fall staging snow geese along the North Slope of the Yukon and eastern Alaska. Flocks of geese feeding within 5 km of the simulator flushed and relocated, although some of these birds returned to feeding areas within 2.5 km of the simulator. Nevertheless, this noise source resulted in a net loss of staging habitat within a 2.5 km radius of the simulator. However, Gollop et al. (1974d) did not observe changes in breeding densities or reproductive success of Lapland longspurs exposed to the simulated high noise levels of a gas compressor station. The effects of other stationary sound sources on birds in the Beaufort Sea region have not been documented.

TABLE 2.7-1

SUSCEPTIBILITY OF SELECTED BIRDS TO AIRCRAFT DISTURBANCES
(adapted from Slaney 1977a)

	Susceptibility to Disturbance	Estimated Distance Range (m)
Loons	low to moderate	
	- during migration	200
	- at nesting site	200
	- general	200
Whistling Swans	moderate to high	
	- during migration	<100
	- at nesting site	<100
	- general	<100
Geese	high**	
	- during migration	3200
	- at nesting site	1600
	- general	800
Ducks	low to moderate	
	- during migration	200
	- at nesting site	200
	- general	200
Small Shorebirds	low	
	- during migration	100
	- at nesting site	100
	- general	100
Passerines	- less than ten metres in all circumstances at all seasons	

* based on 3 years of unstructured observations (Slaney 1974a).

** snow geese - high
brant - moderate
Canada geese - moderate
white-fronted geese - high

2.7.4 Summary of Concerns Related to Airborne Noise

As indicated in Table 2.7-2, the degree of potential regional concern associated with the effects of airborne noise on marine mammals in the Beaufort Sea region is NEGLIGIBLE or MINOR, but ranges from NEGLIGIBLE to MAJOR for various species of birds. Snow geese are considered the most vulnerable species to disturbance by mobile airborne noise sources because of their documented susceptibility to noise, and the fact that they nest colonially and occur in large concentrations on staging grounds. In general, geese and swans are more vulnerable to airborne noise than ducks, loons, gulls, alcids, and shorebirds. The impacts of aircraft and other noise sources on all species of birds would depend on a number of factors including the life cycle stage of affected individuals and the altitude, frequency and route of flights, and type of aircraft. Adherence to altitude guidelines (>305 m AGL) will minimize the effects of aircraft on most species, although snow geese and possibly other species of geese will probably react to aircraft at higher elevations under most circumstances. Stationary sources of airborne noise in terrestrial areas may result in habitat loss for some species of birds, depending on the location and type of noise produced by these facilities or activities. The species most likely to be affected by stationary noise sources are snow geese and white-fronted geese in nesting or staging areas, although unlike mobile sound sources, effects on regional populations are only of MINOR concern due to the limited amount of habitat that would be affected.

The degree of concern related to the effects of airborne noise from mobile sources on marine mammals is NEGLIGIBLE to MINOR. Hauled-out seals may dive, and polar bears and Arctic foxes on the sea-ice may retreat from approaching aircraft. As in the case of birds, the response of these species and its possible significance would depend on the frequency, altitude, routes, and type of aircraft. Stationary sources of airborne noise may alert and attract polar bears or Arctic foxes to sites of human activity, although the degree of regional concern regarding attraction of these species is expected to be NEGLIGIBLE.

TABLE 2.7-2

SUMMARY OF POTENTIAL CONCERNS RELATED TO AIRBORNE NOISE IN THE BEAUFORT SEA REGION

Environmental Componment or Resource	Potential or Probable Effects	Degree of Potential Regional Concern
Ringed and Bearded Seals	Disturbance resulting in diving response of hauled-out seals during annual moult; possible thermoregulatory stress	NEGLIGIBLE for stationary noise sources; MINOR for mobile noise sources
Polar Bears and Arctic Foxes	Attraction to stationary noise sources; retreat from mobile noise sources	NEGLIGIBLE and MINOR for stationary and mobile noise sources, respectively
Snow Geese	Disturbance at nesting colonies or on fall staging areas resulting in reduced breeding success or less than optimum weight gain on staging grounds	MINOR, stationary sources MODERATE to major for mobile sources depending on flight frequency, routes, altitudes, type of aircraft
Other Geese, Whistling Swans, Ducks	Disturbance at nesting areas, or during moulting, brood-rearing and/or staging resulting in reduced breeding or fledgling success, increased energy expenditures associated with repeated flushing, habit loss, less than optimum weight gain	MINOR to MODERATE for mobile sources depending on flight frequency, routes, altitudes, type of aircraft NEGLIGIBLE for stationary noise sources (possibly MINOR for white-fronted geese)

TABLE 2.7-2 (cont'd)

Environmental Componment or Resource	Potential or Probable Effects	Degree of Potential Regional Concern
Colonial nesting species (thick-billed murres, black guillemots, glaucous gulls, Arctic terns, snow geese, common eiders)	Disturbance at nesting colonies resulting in nest abandonment, reduced breeding success	MINOR to MAJOR depending on species, source and location of noise, frequency of noise, altitude and routing of aircraft
Other marine-associated birds (loons, shore-birds, cranes and other gulls)	Disturbances resulting in reduced breeding, increased energy expenditures associated with repeated flushing	NEGLIGIBLE to MINOR, depending on factors listed above

2.8 ENGINE EXHAUSTS/ATMOSPHERIC EMISSIONS

2.8.1 Introduction

Activities associated with oil and gas production in the Beaufort Sea region would involve the operation of large numbers of exhaust-producing machines such as electrical generators, engines which power drill rigs, heating plants, incinerators, construction equipment, vessels and aircraft. These exhausts could contain varying quantities of gases, particulates, water vapour and partially combusted hydrocarbons. In addition, the continuous operation of large engines on production platforms may contribute to the formation of "heat islands". Atmospheric emissions would also occur during gas flaring, while noise would also be associated with many of these emissions. The potential effects of gas flares and airborne noise on marine resources of the Beaufort Sea are discussed separately in Sections 4.1 and 2.7, respectively.

Development plans include the design of incinerators that will produce minimal particulates and assure that federal guidelines for ambient air quality are not exceeded (Table 2.8-1). Most emissions would be rapidly dispersed to the surrounding atmosphere. Ice crystals are formed when warm exhausts are vented into cold air and the presence of these crystals can lead

TABLE 2.8-1

SUMMARY OF FEDERAL AMBIENT AIR QUALITY STANDARDS ($\mu g/m^3$)
(Source: ECO-LOG 1981)

Pollutant	Time Base	Maximum Desirable	Maximum Acceptable	Maximum Tolerable
Total Suspended Particulate	24 hr	-	120	400
	1 yr	60	70	-
Sulphur Dioxide	1 hr	450	900	-
	24 hr	150	300	800
	1 yr	30	60	-
Carbon Monoxide	1 hr	15,000	35,000	-
	8 hr	6,000	15,000	20,000
Nitrogen Dioxide	1 hr	-	400	1,000
	24 hr	-	200	300
	1 yr	60	100	-

to the formation of ice fog when air temperatures are less than -30°C and when temperature inversions or periods of calm reduce mixing and dispersion. The height of the exhaust stack, emission temperature and velocity also affect the formation of ice fogs. However, since offshore and shoreline areas of the Beaufort Sea tend to be relatively windy and prolonged periods of calm and temperatures less than -30°C are not as common in these areas as at inland locations (EIS Volume 3A, Section 2.1), significant ice fogs are unlikely to be a frequent occurrence and should not affect biological resources of the region.

2.8.2 Effects of Atmospheric Emissions on Birds and Mammals

The degree of concern regarding the effects of engine exhaust and atmospheric emissions on marine mammals and birds in the Beaufort Sea region is expected to be NEGLIGIBLE since the emissions would be rapidly dissipated to the surrounding atmosphere, and would usually be present in relatively low concentrations. As indicated earlier, atmospheric emissions are also expected to conform to federal regulations under normal operating conditions. The only potential concern related to atmospheric emissions is the possible attraction of birds and mammals to localized "heat islands" created by the continuous operation of machinery during cold periods. The attraction of mammals and birds to "heat islands" surrounding production and shorebased facilities would be largely indistinguishable from the attraction of these groups to composite activities at these sites (e.g. artificial illumination, human presence), particularly during periods with both low temperatures and light levels. Nevertheless, warm temperatures *per se* may attract certain species of birds and mammals to shorebased and offshore structures, although other factors such as airborne noise (Section 2.7.3) and gas flares (Section 4.1.3) may offset the tendency for "heat islands" to attract some fauna. Species which may be attracted to sites of industrial activity could include pelagic birds such as gulls, terns and jaegers and small mammals such as Arctic and red foxes.

2.8.3 Summary of Concerns Related to Engine Exhaust/Atmospheric Emissions

Since emissions from all operating equipment at offshore exploration and production facilities or shorebases are expected to conform to appropriate regulatory guidelines, the only residual concerns related to engine exhausts are short-term and localized reductions in visibility associated with ice fog formation, and the possible attraction of some mammals and birds to localized "heat islands". The latter effects are expected to be indistinguishable from other factors which may cause attraction of birds and mammals, and as indicated in Table 2.8-2, the degree of regional concern regarding all potential effects of atmospheric emissions is considered NEGLIGIBLE.

TABLE 2.8-2

SUMMARY OF POTENTIAL CONCERNS RELATED TO ENGINE EXHAUSTS AND OTHER ATMOSPHERIC EMISSIONS IN THE BEAUFORT SEA REGION

Environmental Component or Resource	Potential or Probable Effects	Degree of Potential Concern
Visibility	Localized formation of ice fog (reduced visibility) at temperatures less than -30°C during calms or temperature inversions	See Section 2.8.1
Air quality	Atmospheric emissions, primarily engine exhausts; subsequent effects on mammals and birds	See Section 2.8.1
Birds and Mammals	Attraction to "heat islands" during cold periods	NEGLIGIBLE

2.9 SOLID WASTES

2.9.1 Introduction

Combustible and non-combustible solids from offshore platforms or drillships in the Beaufort Sea would usually be transported to shorebased landfill sites or incineration facilities, although the latter will also be installed on permanent offshore production platforms. The ash from the incineration of combustible solids and the non-combustible waste would be stored and eventually transported to shore for disposal. One offshore platform in the Beaufort Sea is expected to produce an estimated 900 kg of solid waste per day (Bercha and Associates Ltd. 1979), and of this total, approximately 85 percent would be combustible and 15 percent non-combustible. Although solid wastes would not be disposed of in the marine environment during normal operations at exploration and production facilities, some materials could occasionally enter the water and accumulate on the seafloor.

These materials would be primarily steel and could include used drill bits, pipe, mild steel pieces, drilling tubulars and wire rope. With the exception of drill bits which contain small percentages of chromium (0.3 to 1.6 percent) and vanadium (0.5 to 2.0 percent), most metal present on offshore facilities would be regular carbon steels which are highly susceptible to oxidation and corrosion in seawater. The following sections briefly summarize the documented and potential effects of steel and other solid wastes on marine resources.

2.9.2 Effects of Solid Wastes on Mammals

The accidental introduction of steel from offshore production facilities or drillships into the surrounding waters may result in highly localized mortality of benthic organisms (Section 2.9.5) and a concomitant loss or reduction in the feeding habitat for bearded seals and bowhead whales when disposal occurs within the 100 m isobath. However, the spatial extent of the affected areas would be so small in relation to the available benthic habitat within the region that the degree of potential concern related to possible effects on marine mammals is considered NEGLIGIBLE.

Other non-combustible solid wastes produced by offshore and shorebased activities and facilities would be disposed of at approved landfill sites. Some terrestrial and marine-associated mammals may be indirectly affected by this disposal if they are attracted to and scavenge at the landfills. Species in the Beaufort Sea region that may be attracted to these sites include polar bear, grizzly bear, Arctic fox, red fox, tundra wolf and other furbearers. The degree of regional concern associated with the attraction of most species to landfills is expected to be NEGLIGIBLE because the numbers affected would be regionally insignificant. However, the degree of concern related to potential attraction of polar and grizzly bears is considered MINOR since the human safety hazard may necessitate removal of nuisance animals.

2.9.3 Effects of Solid Wastes on Birds

As with marine mammals, the accidental loss of steel from offshore exploration and production facilities or drillships may result in a very localized loss of benthic feeding habitat for some species of birds in relatively shallow waters (e.g. <40 m). Birds in the Beaufort Sea region which feed in benthic habitats primarily include diving ducks, loons and alcids. However, the degree of concern associated with potential indirect effects of steel loss on birds would be considered NEGLIGIBLE because the affected areas would be insignificant in relation to available feeding habitat and benthic food sources.

Small numbers of birds (primarily shorebirds, gulls and ravens) in the Beaufort Sea region may be attracted to landfill sites. The degree of concern related to this attraction in terms of the regional populations of these species is considered NEGLIGIBLE because relatively few individuals would be involved and such effects would be largely indistinguishable from attraction of some birds to shorebased physical structures and sites of human activity (Sections 2.1.4 and 2.2.3).

2.9.4 Effects of Solid Wastes on Fish

The effects of steel objects from hydrocarbon exploration and production facilities on fish would be primarily related to the physical presence of these objects. Although the metals would corrode readily, the resultant ferric oxides are relatively non-soluble and non-toxic (Hann and Jensen 1974), and would not be expected to result in any adverse effects on local fish populations.

The potential effects of the presence of steel on the seafloor would be similar to the general effects of artificial structures on fish populations (Section 2.1.5). Some species of marine fish may be attracted to micro-habitats created by drilling tubulars and other steel objects, particularly when these solid surfaces are colonized by benthic fauna. Nevertheless, since steel would not be routinely disposed of in marine environments and any effects on fish would be extremely localized, the degree of potential concern regarding effects of solid wastes on regional fish populations would be NEGLIGIBLE.

2.9.5 Effects of Solid Wastes on Benthic Communities

The potential effects of steel lost from exploration and production platforms and drillships on benthic organisms would include localized mortality and the creation of new habitats, since iron oxides are relatively non-toxic (Section 2.9.4). Highly localized crushing, suffocation or physical damage of benthic flora and fauna would occur immediately under steel accidentally entering marine environments, although such losses would be NEGLIGIBLE from a regional prospective.

Steel would also provide a substrate for eventual colonization by other benthic fauna. Hard substrates suitable for attachment of macrophytic algae and epifaunal invertebrates are uncommon in the Beaufort Sea, and the few areas with hard substrates support an atypical and more diverse benthic community than is usually found in this region. It is possible that similar benthic communities could develop on steel accidentally lost near drilling sites and production platforms in the Beaufort Sea since artificial reefs and breakwaters constructed in temperate waters from derelict ships or wrecked automobile bodies have been rapidly colonized by benthic organisms. However, as in the case of fish, creation of new habitats for benthic flora and fauna would be extremely localized and NEGLIGIBLE in terms of regional benthic communities.

2.9.6 Summary of Concerns Related to Solid Wastes

Most solid wastes produced by offshore and shorebased facilities in the Beaufort Sea region will be combustible and would be incinerated. Remaining ash and any non-combustible solids will be disposed of at approved landfills. However, in some circumstances, steel objects, such as drill bits and tubulars, may be accidentally lost from offshore exploration and production platforms, and would settle to the seafloor surrounding these facilities. As indicated in Table 2.9-1, the degree of regional concern related to the effects of solid wastes on most coastal and offshore biological resources is expected to be NEGLIGIBLE. However, the potential attraction of polar bears and grizzly bears to landfills is considered a MINOR area of potential regional concern since nuisance animals may have to be removed and occasionally destroyed in the interests of human safety.

TABLE 2.9-1

SUMMARY OF POTENTIAL CONCERNS RELATED TO SOLID WASTE DISPOSAL IN THE BEAUFORT SEA REGION

Environmental Component or Resource	Potential or Probable Effects	Degree of Potential Regional Concern
Bearded Seal	Localized reduction of feeding habitat and benthic food sources	NEGLIGIBLE
Arctic fox, red fox, tundra wolf, other furbearers	Attraction to landfills	NEGLIGIBLE
Polar bear, grizzly bear	Attraction to landfills, and occasional destruction of nuisance animals	MINOR
Diving ducks, loons and alcids	Localized reduction of feeding habitat and benthic food sources	NEGLIGIBLE
Gulls, shorebirds and ravens	Attraction to landfills	NEGLIGIBLE
Fish	Attraction to microhabitats created by steel objects near exploration and production platforms	NEGLIGIBLE
Benthic Communities	Localized mortality of flora and fauna beneath steel accidently entering marine environments; creation of new hard substrate habitats and subsequent colonization by epibenthic organisms	NEGLIGIBLE

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3.0 DISTURBANCES AND WASTES ASSOCIATED WITH NORMAL DRILLING OPERATIONS

3.1 DRILLING FLUIDS, FORMATION CUTTINGS AND PRODUCED WATER

3.1.1 Introduction

During both exploratory and production drilling for oil and natural gas, a drilling fluid (commonly referred to as 'drilling mud') is circulated to remove rock fragments loosened by the drill bit. The drilling fluid is transported from a surface reservoir by mud pumps and forced down the centre of the steel drill pipe as the drilling progresses. It enters the bore hole through nozzles in the bit, collects formation cuttings, and returns to the surface between the drill pipe and the walls of the bore hole and/or the casing. When this material reaches the surface, it is diverted through a shale shaker screen that removes the larger formation cuttings which may reach diameters of 4 mm. These cuttings are sprayed with water as they move down and off the slanted, vibrating shale shaker into the surrounding water. The larger drill cuttings typically settle close to the point of discharge and form cuttings piles 10 to 100 cm high and 50 m in diameter (Zingula 1975). These accumulations may be resuspended and dispersed over time, particularly in high wave energy environments (Meek and Ray 1980). Drilled solids too fine to be separated from the drilling fluid by screening are generally removed by gravity segregation and centrifugation, and then discharged with solids removed by the shale shaker. When oil-based drilling muds are required for a specific drilling program (e.g. when formation temperatures are very high or when damage to the formation must be minimized), additional equipment (cuttings washer) is used to remove hydrocarbons from the cuttings prior to discharge. In addition, free hydrocarbons are gravity separated and then returned to the mud system.

After separation and disposal of the cuttings, the drilling mud is returned to the reservoirs ('mud tanks') for recirculation down the bore hole. Relatively small volumes of the drilling mud are continuously lost through disposal with the cuttings, while larger amounts are discharged when water or additional solids are added to adjust mud properties as the drilling program progresses. When a completely different mud system is needed, such as after installation of casings, or when the properties of the mud must be modified for deeper formations, the old drilling fluid is released to the surrounding water. The drilling mud (except oil-base systems) is also typically discarded at the conclusion of a drilling program. Cement slurry may be discharged to the water column at certain points during the drilling program, particularly during cementing of conductor and surface casing strings at floating exploration platforms (drillships). The biological effects of cement slurries are separately discussed in Section 3.4.

In some instances, the turbidity plume resulting from a drilling operation has been visible for more than 3 km, although direct measurements and theoretical calculations indicate that the fine muds discharged during offshore drilling rapidly mix with seawater and can be diluted by 1000-fold between 0.1 and 1.0 km down-current of the disposal site (Dames and Moore Inc. 1978; Monaghan et al. 1980). Dispersion model calculations indicate that bulk mud discharged at high rates (250 bbl/hr) at the completion of a well can still be diluted as much as 100-fold 300 m down-current of the discharge site in less than 1 hr (Monaghan et al. 1980). The wash-down seawater combined with reversing wave surge within the discharge pipe is likely responsible for much of this high initial dilution. These dilution rates, however, have been calculated for offshore environments other than the Beaufort Sea, where ice cover would reduce oceanographic circulation for at least six months each year. Miller et al. (1980) suggest that above-ice disposal of drilling fluids often results in greater dilution and wider dispersal than below-ice discharge due to spring overflooding from rivers.

The volume of drilling fluids used during the drilling of exploratory and production wells varies with the depth of the gas or oil-bearing formation and the need to change the formulation of the mud system as the drilling progresses through strata of different geological origin, pressure and permeability. Monaghan et al. (1976) reported that approximately 525 m^3 of drilling fluid were required to drill a 6100 m well in the Gulf of Mexico which started with a bore hole size of 91 cm and ended with a hole diameter of 16.5 cm, while from 1500 to 2900 m^3 of mud were discharged during the drilling of three wells in the Beaufort Sea ranging in depth from 2700 m to 3900 m (Friesen 1980). Under normal operational conditions, approximately 44 m^3 of water-based drilling mud are discharged per day for each rig operating in the Canadian Beaufort (Hopkins and Taylor 1979), although this volume may increase to 200 m^3/day under abnormal circumstances such as when a mud system has become contaminated. On the average, approximately 1500 m^3 of drilling mud and 200 to 400 m^3 of formation cuttings will be discharged during the drilling of a typical 4000 m well in the Beaufort Sea.

The drill mud formulation is often of greater significance than the total volume of wastes in terms of potential biological effects since it contains certain inorganic and organic constituents which are slightly to moderately toxic to marine flora and fauna. Drilling muds are basically suspensions of clays in water, but include barium sulphate (barite) to control density, and a variety of additives to control flow and filtration properties of the fluid and minimize corrosion of the steel drill pipe and casing. A number of fibrous and bulky solid materials are also added to the mud system to prevent loss of circulation ability. The most important functions of drilling fluids were summarized by Monaghan et al. (1976) who state that 'They must remove the cuttings from under the bit and transport them to the surface, promote continuous drilling by cooling and lubricating the bit and drill string, maximize the penetration rate, have sufficient density to control subsurface pressure and prevent inflow of formation fluids, coat the exposed walls of the hole with filter cake to minimize fluid loss into the permeable formations, and suspend the cuttings and the mud solids if circulation is stopped but release them at the surface. The mud must also have the properties that ensure maximum information from the well. That is, it must transport cuttings large enough for geologic evaluation and must have properties that permit accurate logging of the formation and pore characteristics.' Consequently, the formulation of drilling mud systems must not only have some common constituents to produce general properties, but also specific compounds to meet the individual requirements of a given well and rock formation.

Drilling in the Beaufort Sea poses special requirements due to the presence of a permafrost layer ranging from a few meters to over 600 m in thickness, as well as the prevalence of gas hydrates and abnormal formation pressures (Friesen 1980). KCl muds are generally used when drilling through permafrost because their high chloride content acts as a freezing point depressant, allowing use of a cold mud to prevent permafrost damage. The basic components of KCl mud systems are bentonite, KCl and XC polymer to provide viscosity and caustic soda to control the pH (Friesen 1980). In the Beaufort Sea, a fresh water gel mud system is typically used for drilling beneath the permafrost layer due to its suitability in abnormally pressured wells. Table 3.1-1 shows the average concentration of components in waste drilling fluids from three artificial island wells drilled by Esso in the Beaufort Sea. These wastes were diluted 25:1 and the pH adjusted with caustic soda, lime or soda ash to between 6 and 8.5 prior to disposal in the marine environment.

Formation or produced water, completion fluids and tritiated water may also be released to the water column during certain phases of drilling in the Beaufort Sea. The salinity of formation water in this region is similar to that of water in the drilling muds. Since it is necessary to differentiate between formation water and mud filtrate in drill stem tests, low concentrations (0.003 μCi/ml) of tritiated water are used as radioactive tracer from the surface casing to the total depth of the well (Friesen 1980). However, the maximum permissible throw-away level for tritiated water (0.003 μCi/ml; 6600 DPM) is not exceeded, and drill muds containing this radioisotope can be disposed of in a normal manner. The use of tritiated water as a drill mud tracer has also been endorsed, subject to prescribed guidelines, by the Atomic Energy Control Board of Canada and the Radiation Protection Division of the Department of National Health and Welfare (Millar and Buckles 1974). The only potential area of concern related to the use of drill muds containing tritiated water is the human health hazard associated with inhalation of water vapour over the mud tanks, although only trained personnel handle the tracer on drilling platforms.

TABLE 3.1-1

AVERAGE CONCENTRATION OF COMPONENTS IN DRILLING WASTE FLUIDS FROM IMMERK B-48, ADGO F-28 AND PULLEN E-17 ARTIFICIAL ISLANDS IN THE SOUTHEAST BEAUFORT SEA
(Source: Friesen 1980)

Component	Concentration (Percent by Weight)	Description
Drilled Solids	22.8	Sand and Mudstone
Barium Sulphate (Barite)	5.8	$BaSO_4$
Bentonite	3.5	Montmorillonite Clay
Sub-Total	32.1	Suspended Inorganic Solids
Potassium Chloride (Potash)	0.9	KCl
Sodium Hydroxide (Caustic Soda)	0.1	NaOH
Sodium Bicarbonate	0.1	$NaHCO_3$
Sub-Total	1.1	Dissolved Inorganic Chemicals
"DFE-506"	0.1	Ferric Lignosulponate
"XC Polymer"	0.2	Xanthan Gum
"Dakolite"	0.04	Lignite
Sub-Total	0.3	Dissolved Organic Chemicals
Water (By Difference)	66.5	Water Base of Drilling Fluid
TOTAL	100.0	

The majority of the formation or produced water could be reinjected into the producing formation as part of recovery enhancement programs, with the remainder being discharged into the marine environment after oil concentrations are reduced to <50 mg/L, as required by the Canadian Oil and Gas Productivity Regulations. Water in excess of that required for reinjection may result in discharges approximating 100,000 barrels (19,100 m^3) per day per production platform (EIS Volume 2). Other than the oil content of formation water, two remaining concerns associated with the discharge of produced water are its temperature (10 to 55°C) and trace metal content. The potential concerns associated with various sources of trace metals present in drilling wastes are addressed separately at the end of this section (Section 3.1.10), while heated water is discussed in Section 4.2. It should also be emphasized that the trace metals, increased temperature and oil content of drilling wastes (as well as other wastes released at exploration and production facilities) could act synergistically or antagonistically in some resources, although there is virtually no information on the cumulative impacts of multiple waste sources in oil production regions throughout the world.

Oil-bearing sand formations in the Beaufort Sea are unconsolidated and may produce sand during production, as well as during extended testing. This sand decreases the flow of hydrocarbons and also increases the probability of blowouts. One successful method of stopping the flow of formation sand is the use of gravel packing in conjunction with a high density completion fluid which contains $ZnBr_2$, $CaBr_2$ and traces of oxygen scavenger and corrosion inhibitor. This fluid is recirculated in the same manner as drilling fluid, except that it is not discharged to the marine environment. During the well completion operation, acidic solutions (HCl and HF) are also used to eliminate any wellbore damage created by the drilling fluid. These acid solutions may also contain trace amounts of surfactant, clay stabilizer, acid inhibitor, de-emulsifier, fluid loss additive and iron control agents, and are followed by treatment with a NH_4Cl solution. Once placed into the well, most of the acids and some completion fluids migrate into the subterranean geological structures and are lost. When a well is re-entered for testing or production purposes, a proportion of the acid and completion fluid which migrated into the formation is drawn back out and "produced" along with formation water and hydrocarbons. Approximately 500 barrels of fluid containing $ZnBr_2$, $CaBr_2$, acid and other minor constituents may be produced at this time, and are released to the marine environment following separation of hydrocarbons. The acids are rapidly buffered, neutralized and diluted in the surrounding water, while the dissolved zinc tends to chelate with organic matter and anions normally present in seawater and precipitates out of solution.

The principal environmental concerns associated with the disposal of drilling fluids, completion fluids, formation waters and formation cuttings are potential toxicity of certain chemical additives, increased turbidity in the water column, smothering of benthic infauna, and the possible localized accumulation of trace metals in sediments and marine biota. These concerns have been the subject of considerable research by both government and industry, as well as a recent major symposium on the environmental fate and effects of drilling fluids and cuttings (Lake Buena Vista, Florida, January 21 - 24, 1980). As indicated in Table 3.1-2, the major components of drilling fluids approved for use in the Canadian North are virtually non-toxic with 96-h LC_{50} values ranging from 1000 to greater than 100,000 ppm. Some minor additives such as metal chlorides, lignosulphonates, biocides, rust inhibitors and defoamers are more toxic. Nevertheless, laboratory bioassays of whole muds used in Arctic drilling programs indicate relatively low toxicity values (96-h LC_{50} values from 0.4 to 13 percent) with marine invertebrates and fish (McLeay 1975). In addition, the normally rapid dilution in receiving waters further reduces potential toxic effects except in shallow waters (<5 m) or when ice cover limits effective water depth and/or restricts adequate dispersal of wastes. The following sections discuss available information describing the biological effects of drilling fluids, formation cuttings and produced water on marine mammals, birds, fish and members of lower trophic levels, as well as the level of potential environmental concern regarding these effects during offshore drilling operations in the Beaufort Sea. A separate section (3.1.10) discusses the trace metals which may be associated with Beaufort Sea drilling operations, bioavailability and their documented effects on marine organisms.

TABLE 3.1-2

DESCRIPTION, RATE OF USE AND TOXICITY
(RAINBOW TROUT) OF DRILLING MUD COMPONENTS
APPROVED FOR USE IN THE CANADIAN NORTH
(Source: Adapted from Industry/Government Working Group "A" 1976)

Product	Description Chemical Composition	Estimated Use Range (lbs/bbl)	Acute Toxicity 96-h LC_{50} (ppm)
WEIGHTING AGENTS			
Barite	Barium Sulphate	30-700	>100,000
Calcium Carbonate		0-320	>56,000
Galena	Lead Sulphide	240-1180	
VISCOSIFIERS			
Wyoming Bentonite	Montmorillonite Clay	5-40	>50,000
Attapulgite	Salt Gel	5-30	23,500
Kelzan CX Polymer	Xanthan Gum	0.5-1.5	1900
FLR-100	Organic Polymer	0.5-1.5	2300
Visbestos	Asbestos	5-10	2750
EXTENDERS FOR BENTONITE			
Benex	Copolymer of Polyvinyl Acetate and Malic Anhydride	0.05-0.25	1300
Rapidril	Organic Polymer	0.025-0.125	550
Kwik-Vis	Polymer	0.05-0.25	1600
Drillaid 421	Synthetic Organic Polymer	0.025-0.125	280-550
THINNERS			
Peltex	Ferro-chrome Lignosulphonate	1-10	3000
Q-Broxin	Ferro-chrome Lignosulphonate	1-10	1750

TABLE 3.1-2 (Continued)

Product	Description Chemical Composition	Estimated Use Range (lbs/bbl)	Acute Toxicity 96-h LC_{50} (ppm)
Spersen	Chrome Lignosulphonate	1-10	3750
Uni-Cal	Chrome Modified Sodium Lignosulphonate	1-10	860
DFE - 506	Ferrous Lignosulphonate	1-10	3500
Carbonox	Lignite	1-8	6500
Calcium Lignosulphate		1-10	>2000
Mil-Flo 600	Hemlock Bark Extract		1-5
Desco	Organic Extract	1-5	1200
ALKALINITY - pH CONTROL			
Caustic Soda	Sodium Hydroxide	0.25-2	105
Caustic Potash	Potassium Hydroxide	0.25-2	105
Lime	Calcium Hydroxide	0.1-1	50
LOST CIRCULATION MATERIALS			
Sawdust		25-100	
Quik Seal	Wood Fibre + Cellophane	25-50	Random
Mica		25-100	na
Walnut Shells		25-100	Stirred 2800 Settled 5500
Cane Fibre		Variable	na
Cellophane		Variable	na
PRESERVATIVES			
Paraformaldehyde		0.25-0.5	60
Dowicide B	Sodium Trichlorophenate	0.1-0.2	<1.0

TABLE 3.1-2 (Continued)

Product	Description Chemical Composition	Estimated Use Range (lbs/bbl)	Acute Toxicity 96-h LC_{50} (ppm)
SALTS			
Sodium Chloride		10-125	10,000
Muriate of Potash		10-20	2100
Calcium Chloride	For Cementing Only		10,000
CORROSION INHIBITORS			
Sodium Sulphite		Variable	na
Sodium Chromate		Variable	na
DEFOAMERS			
Aluminum Stearate		0.1-0.25	1100
Capryl Alcohol		0.1-0.25	83
CALCIUM REMOVERS			
Bicarbonate of Soda	Sodium Bicarbonate	0.5-1.5	7500
S.A.P.P.	Sodium Acid Pyrophosphate	0.1-0.5	1700
Soda Ash	Sodium Carbonate	0.5-1.5	75
FLUID LOSS REDUCERS			
C.M.C. 3000	Sodium Carboxymethyl Cellulose		0.25-1.5
Cypan	Sodium Polyacrylate	0.25-1.5	1250
Drispac	Polyanionic Cellulose	0.25-1.5	2750
Dextride	Organic Polymer (Bactericide Treated)	1-2	<1
SHALE INHIBITOR			
D.A.P. 14,500-58,000	Diammonium Phosphate		5-20
Protectomagic	Blown Asphalt	4.6	50,000-75,000

TABLE 3.1-2 (Continued)

Product	Description Chemical Composition	Estimated Use Range (lbs/bbl)	Acute Toxicity 96-h LC_{50} (ppm)
SPECIAL PRODUCTS			
Plaster of Paris	Gypsum	0.1-0.5	>56,000
Potassium Chrome Alum	Chromalit	0.1-0.25	>730
Pipe Lax	Surfactant	Variable	na
B-Free	Surfactant	Variable	na
Skot-Free	Surfactant	Variable	na
BaraFloc	Clay Flocculant	0.25-0.5	800
Cyanamid RC-326	Encapsulator	0.5-1.0	na
Separan AP-273	Encapsulator	0.5-1.0	na
Percol 155	Encapsulator	0.5-1.0	na
Swift's Rig Wash	Detergent	Variable	22
Dominion Rig Wash	Detergent	Variable	14
Amway Rig Wash	Detergent	Variable	0.8
V-Wis Rig Wash	Detergent	Variable	58
Graco	Detergent	Variable	100-200
Ivory Snow	Detergent	Variable	154

3.1.2 Effects of Drilling Wastes on Marine Mammals

Formation cuttings and drilling fluids have not been shown to be hazarous to marine mammals. However, several of the individual components of drilling fluids are known to be acutely toxic (Table 3.1-2) and may affect marine mammals in the vicinity of waste disposal sites. The susceptibility of marine mammals in the Beaufort Sea to the toxic effects of some drilling fluid additives would likely vary with species and area of drilling waste disposal. Ringed and bearded seals could be most susceptible to drilling waste exposure since these species have been shown to be attracted to drilling operations in the region (Section 2.1.3). Any toxic effects of these compounds would be extremely localized around drilling platforms since waste drilling fluids would be rapidly diluted to non-toxic concentrations, particularly in offshore areas with unrestricted water circulation. In a similar manner, any indirect effects of drilling waste disposal associated with reductions in the abundance of the benthic food organisms of some marine mammals would also be highly localized. As a result, the overall degree of concern regarding adverse effects of drilling fluid and formation cutting disposal on regional marine mammal populations of the Beaufort Sea is expected to be NEGLIGIBLE.

3.1.3 Effects of Drilling Wastes on Birds

The primary areas of concern related to drilling waste disposal and birds are the localized loss of benthic food sources of some species and the potential uptake of trace metals through ingestion of contaminated food items (Section 3.1.10). Most potential effects of drilling wastes on birds would be restricted to relatively shallow areas (<30 m) which provide important feeding habitat for diving ducks, alcids and loons (LGL and ESL 1981). To date, however, drilling fluids and formation cuttings have not been shown to be hazardous to birds, and the overall degree of concern regarding adverse effects of these wastes on birds in the Beaufort Sea region is expected to be NEGLIGIBLE.

3.1.4 Effects of Drilling Wastes on Fish

Numerous laboratory and field studies have examined the toxicity of drilling wastes to fish species from Arctic waters and elsewhere. Drilling fluids, formation water and cuttings, and completion fluids may have similar potential effects on fish, including direct acute toxicity of certain elements or compounds, sublethal physiological and behavioural effects, and bioaccumulation of certain constituents (e.g. trace metals). The effects of whole drilling fluids and suspended solids, as well as other potential concerns associated with the disposal of drilling wastes and fish populations in the Beaufort Sea, are discussed in the following subsections. Trace metal toxicity and bioaccumulation is separately addressed later in Section 3.1.10.

3.1.4.1 Effects of Whole Drilling Fluids

The toxicity of drilling fluids varies with fish species, mud composition, well location and well depth (Tables 3.1-3 and 3.1-4). In addition, the sensitivity of a particular species varies with age and stage of development (Houghton et al. 1980), while acute lethal concentrations (LC_{50}) determined also vary with bioassay methods (e.g. static versus flow through). Both the physical and chemical composition of drilling mud systems can contribute to their toxicity. Physical effects result from suffocation or abrasion of gill epithelia by suspended solids (e.g. bentonite and barite additives) and additives used to increase the viscosity of drilling fluids (e.g. organic polymers). Chemical effects may result from the direct toxicity of certain constituents such as trace metals (e.g. cadmium, chromium) or osmotic stress (e.g. KCl additive; Beckett et al. 1975).

In general, drilling muds used in the Canadian Arctic are most toxic at the shallowest (1500 m) and deepest (3000 m) well depths (Weir et al. 1974a; Moore et al. 1975; Industry/ Government Working Group "A" 1976). At the shallower depths, a large amount of KCl is added to the mud system to facilitate drilling through the permafrost layer. Weir et al. (1974a) report that 96-h LC_{50} values of KCl-based drilling muds with rainbow trout ranged from 1.73 to 2.20 percent. In deeper portions of the well, the mud system requires an increased weighting. This is achieved by the addition of large amounts of bentonite and barite, which increases the concentration of suspended solids in the drilling fluid formulation. Acute toxic concentrations of these weighted mud systems with rainbow trout may range from 0.32 to 2.00 percent (Weir et al. 1974a).

Most of the available literature on the toxicity of drilling fluids to fish is for freshwater rather than marine species, primarily as a result of the popularity of rainbow trout as a bioassay test organism. However, McLeay (1975) found that 96-h LC_{50} values (1.5 to 19 percent v/v) for seven drilling fluids with seawater acclimated coho salmon (Oncorhynchus kisutch) did not vary appreciably from LC_{50} values determined with freshwater acclimated coho (0.4 to 13 percent v/v). McLeay (1975) also found that LC_{50} ranges for rainbow trout were very similar to ranges observed with coho salmon. More recently, Tornberg et al. (1980) found that 96-h LC_{50} values (v/v) for drilling fluids and fish species from the Beaufort Sea ranged from 5 to greater than 35 percent (broad whitefish 6 to 37 percent; fourhorn sculpin 4 to 35+ percent; Arctic cod 20 to 25 percent; saffron cod 17 to 30 percent), and attributed the differences in toxicity levels for each species to variations in drilling fluid composition. Dames and Moore Inc. (1978) reported that the 48-hr LC_{50} value for drilling fluids and staghorn sculpins (Leptocottus armatus) collected from Lower Cook Inlet, Alaska ranged from 10 to 20 percent (v/v). They also investigated in situ sublethal effects of drilling fluids on pink salmon fry (Oncorhynchus gorbuscha). Cages containing the juvenile fish were placed in an area of maximum influence of the plume from a drilling rig, but no deleterious effects were observed after 4 days exposure Dames and Moore Inc. (1978).

TABLE 3.1-3

SUMMARY OF REPRESENTATIVE INVESTIGATIONS DESCRIBING THE ACUTE LETHAL TOXICITY OF WHOLE DRILLING FLUIDS TO FISH

Drilling Fluid	Fish Species	Medium[1]	96-h LC_{50} (ppm)	Reference
Rig 51	lake chub	FW	120,000	Falk and Lawrence 1973
	rainbow trout	FW	8,300	Logan *et al.* 1973
	9-spine stickleback	FW	103,000	Falk and Lawrence 1973
	rainbow trout	FW	112,000	Falk and Lawrence 1973
	rainbow trout	FW	53,000	Logan *et al.* 1973
	lake chub	FW	35,500	Falk and Lawrence 1973
	rainbow trout	FW	42,000	Logan *et al.* 1973
Immerk B-48 (Imperial Oil)	lake whitefish	FW	25,000	Lawrence and Scherer 1974
	rainbow trout	FW	75,000	Lawrence and Scherer 1974
Shell Kipnik 0-20	lake witefish	FW	25,000	Lawrence and Scherer 1974
	rainbow trout	FW	42,000	Lawrence and Scherer 1974
	rainbow trout	FW	120,000	Moore *et al.* 1975
	rainbow trout	FW	42,000	McLeay 1975
	rainbow trout	SW	24,000	McLeay 1975
	coho salmon	SW	29,000	McLeay 1975
	chum salmon	SW	24,000	McLeay 1975
	pink salmon	SW	41,000	McLeay 1975
Gulf Imperial	rainbow trout	FW	390,000	Weir *et al.* 1974b
Union Aklavik	rainbow trout	FW	837,000	Weir *et al.* 1974b
Shell Kugpik	rainbow trout	FW	86,000	Weir *et al.* 1974b
IOL Wagnark	rainbow trout	FW	770,000	Weir *et al.* 1974b
IOL Taglu	rainbow trout	FW	86,000	Weir *et al.* 1974b
IOL Langley	rainbow trout	FW	380,000	Weir *et al.* 1974b
Gulf Mobil Ikhil	rainbow trout	FW	350,000	Weir *et al.* 1974b
Panarctic Drake	rainbow trout	FW	620,000	Weir *et al.* 1974b

[1] FW = freshwater
SW = seawater

TABLE 3.1-3 (cont'd)

Drilling Fluid		Fish Species	Medium[1]	96-h LC_{50} (ppm)	Reference
IOL Pullen	450m	rainbow trout	FW	10,000	Moore *et al.* 1975
	1220m	rainbow trout	FW	550,000	Moore *et al.* 1975
	1525m	rainbow trout	FW	390,000	Moore *et al.* 1975
	1830m	rainbow trout	FW	700,000	Moore *et al.* 1975
	2130m	rainbow trout	FW	500,000	Moore *et al.* 1975
	2440m	rainbow trout	FW	550,000	Moore *et al.* 1975
	2740m	rainbow trout	FW	360,000	Moore *et al.* 1975
	3050m	rainbow trout	FW	160,000	Moore *et al.* 1975
	3350m	rainbow trout	FW	90,000	Moore *et al.* 1975
	3660m	rainbow trout	FW	150,000	Moore *et al.* 1975
	3960m	rainbow trout	FW	90,000	Moore *et al.* 1975
Aquitaine-Polar Bear O-20		rainbow trout	FW	250,000	Moore *et al.* 1975
		rainbow trout	FW	130,000	McLeay 1975
		coho salmon	FW	130,000	McLeay 1975
		coho salmon	SW	130,000	McLeay 1975
Sun-Pelly B-35		rainbow trout	FW	58,500	Moore *et al.* 1975
		rainbow trout	FW	34,000	McLeay 1975
		coho salmon	FW	20,000	McLeay 1975
		coho salmon	SW	23,000	McLeay 1975
Shell Niglintgak M-19		rainbow trout	FW	42,500	Moore *et al.* 1975
		rainbow trout	FW	16,000	McLeay 1975
		coho salmon	FW	4,000	McLeay 1975
		coho salmon	SW	15,000	McLeay 1975
Chevron Upluk M-38		rainbow trout	FW	95,000	Moore *et al.* 1975
Pan Arctic Chad Creek B-64		rainbow trout	FW	550,000	Moore *et al.* 1975
Pan Arctic Pym Point C-44		rainbow trout	FW	2,900	Moore *et al.* 1975
Dome Imperial Imnak J-29		rainbow trout	FW	42,000	McLeay 1975
		coho salmon	FW	23,000	McLeay 1975
		coho salmon	SW	30,000	McLeay 1975

1 FW = Freshwater
SW = Seawater

TABLE 3.1-4

ACUTE TOXICITY OF DRILLING FLUIDS TO FOURHORN SCULPIN (*Myoxocephalus quadricornis*) AND BROAD WHITEFISH (*Coregonus nasus*) FROM THE BEAUFORT SEA (Adapted From Tornberg *et al.* 1980)

Drilling Fluid[1]	Well Depth (m)	Fish Species	96-h LC_{50} (ppm)
Lignosulphonate drilling fluid	2256	fourhorn sculpin	402,500
XC-Polymer	2778	broad whitefish	~423,500
CMC/Gel	2780	broad whitefish	>246,800
CMC/Gel/Resinex	2786	broad whitefish	59,900
CMC/Gel	2786	fourhorn sculpin	148,080
CMC/Gel/Resinex	2786	fourhorn sculpin	59,000-71,880
XC - Polymer	3064	fourhorn sculpin	260,150
XC - Polymer	3082	fourhorn sculpin	>72,600
XC - Polymer	3319	fourhorn sculpin	77,440
XC - Polymer	3319	broad whitefish	242,000
XC - Polymer	3323	broad whitefish	121,000
CMC/Gel/Resinex	3466	fourhorn sculpin	71,880
XC - Polymer	3646	fourhorn sculpin	58,700-117,400

1 Drilling fluids were obtained from well sites in the vicinity of Prudhoe Bay, Alaska.

Although these studies clearly indicate that whole drilling fluids can be toxic to a variety of fish species, the concentrations and exposure conditions necessary to produce these toxic effects would rarely be achieved in natural situations. Waste drilling fluids are normally diluted by a factor of at least 20:1 before disposal in marine waters, and further dilution would rapidly occur due to natural mixing. For example, the most toxic whole mud examined to date (96-h LC_{50} with rainbow trout = 2900 ppm; Table 3.1-3), once diluted prior to disposal, would likely be virtually non-toxic within a very few metres of the drilling site, even assuming that fish would remain in the immediate area of waste disposal for the period necessary for toxic effects to occur.

3.1.4.2 Effects of Suspended Solids

Suspended solids are released to the marine environment with both drilling fluids and formation cuttings. The concentration of suspended solids in these wastes varies with drilling fluid composition (i.e. concentration of barite and bentonite in the mud system), and is generally greatest at the deepest portions of the well. Barite is relatively insoluble, inert and non-toxic to marine fish species, although the potential deleterious effects of suspended solids from these wastes on fish species include physical irritation of gill tissue, suffocation, and decreased resistance to disease. Other effects may include changes in behavioural responses, exposure to more toxic contaminants in the barite and increased oxygen demand (Mortensen et al. 1976, cited in Houghton et al. 1980).

Beckett et al. (1975) examined the toxicity of barite to rainbow trout, and reported that the acute toxic level was 100,000 ppm (10 percent). The only significant adverse effect noted was clogging of the gills. Weir et al. (1974b) observed clogging of gill chambers and hemorrhaging from gill chambers and the eyes of rainbow trout during bioassay experiments with drilling sump wastes. Houghton et al. (1980) also reported an accumulation of mucus on the gills of pink salmon at relatively low concentrations of drilling fluids (1 to 3 percent), and stated that gill irritation and damage contributed to the death of the fish, probably due to effects of suspended solids and/or other drilling fluid components.

Both Dames and Moore Inc. (1978) and McLeay (1975) found that stirring of drilling fluids (thereby maintaining elevated suspended solids levels) increased the apparent toxicity of the solutions. In unstirred tests, Dames and Moore Inc. (1978) found that the 96-h LC_{50} value for pink salmon (Oncorhynchus gorbuscha) fry was 0.3-2.9 percent (v/v), while Monaghan et al. (1977) reported 96-h LC_{50} values of 1.5-19 percent (v/v) for coho salmon (Oncorhynchus kisutch). Tornberg et al. (1980) investigated the contribution of suspended solids to the acute toxicity of XC-polymer drilling fluids, and found that 96-h LC_{50} values for broad whitefish were higher when the supernatant fraction (i.e. excluding most of the settleable solids) was used. The acute toxic concentration of whole drilling fluid was 10.0 percent (v/v), while that of the supernatant fraction was 10.0-17.0 percent (v/v). Lawrence and Scherer (1974) examined the preference responses of whitefish and rainbow trout to clean water and water containing 1000 ppm of drilling fluids from Immerk B-48. The whitefish preferred the water containing drilling fluids, with degree of preference increasing with concentrations up to 1000 ppm. On the other hand, rainbow trout exhibited a neutral response to the mud-containing waters. However, in another experiment, McLeay (1975) found that rainbow trout, coho, chum and pink salmon avoided suspended solids associated with drilling fluids, and remained in the uppermost layers of test solutions where the concentration of suspended solids was lower.

Logan et al. (1973) state that suspended solids may contribute to some of the direct lethal toxicity of drilling fluids at higher concentrations, while soluble toxic constituents are probably more responsible for mortality at concentrations near the 96-hr LC_{50} value. Although suspended solids from drilling fluids are generally thought to be inert in solution, Beckett et al. (1975) found that addition of these components resulted in changes in the pH and conductivity of test solutions.

3.1.4.3 Tritiated Water Disposal

The effects of radiation on organisms are progressive and long term (Bron 1981), with uptake of radionuclides by fish taking place through water, through food or by surface adsorption (Pillai 1978). The harmful effects of beta particles emitted from tritiated water are caused by the ionization produced in living cells. Ionization causes alteration or damage to some of the cell constituents and, in some cases, may destroy the entire cell (Pillai 1978). However, tritiated water is only used in approved low concentrations in drilling fluids, and would be diluted even further with other drilling wastes before release to the marine environment. Consequently, release of tritiated water is not considered a significant area of concern with respect to fish populations of the Beaufort Sea.

3.1.4.4 Habitat Alterations

The abundance and distribution of fish in the vicinity of drilling operations may be affected if food availability or habitat type are altered by the presence of drilling fluids or other drilling wastes. Wohlschlag (1977, cited in Gettleson 1980) indicated that demersal fish numbers and biomass were lower in post-drilling trawls, although the low number of samples collected allowed only tentative conclusions. The abundance of fish was found to increase adjacent to drilling sites off the Californian coast (Allen and Moore 1976) and in the shelf break region off Atlantic City, New Jersey (Menzie et al. 1980). Menzie et al. (1980) attributed the increased fish aggregation to the protection afforded by microhabitats created by accumulations of formation cuttings in the vicinity of the rig.

3.1.4.5 Summary of the Potential Effects on Fish

From the results of the few studies that have been conducted in the vicinity of drilling rigs, it appears that there have been no direct toxic effects of drilling fluids and formation cuttings on fish. Due to the rapid dilution and dispersion of drilling fluids and other drilling wastes following discharge to the marine environment, the period of contact between fish and the wastes is probably on the order of minutes (Gettleson 1980), unless the fish are actively swimming into the most concentrated portion of the turbidity plume (Houghton et al. 1980) or are attracted to the spoil deposits and the associated benthic community. Due to this rapid dilution of drilling fluids, coupled with their usual low toxicity, most effects on fish would be highly localized, and Houghton et al. (1980) suggested that these discharges would result in no detectable environmental impact on pelagic fish species. However, there are two potential residual concerns related to drilling waste disposal in the Beaufort Sea. Drilling wastes released during winter operations into poorly flushed, ice-covered nearshore waters may accumulate near the site of release, particularly near relatively shallow artificial islands where ice ridges may restrict the normal water circulation. Overwintering demersal fish species would be most likely to encounter drilling wastes under such circumstances, although the disposal of wastes on the ice surface would eliminate this potential area of concern. The second potential area of residual concern related to drilling wastes and fish would be accidental spills of some of the more toxic drilling mud additives indicated in Table 3.1-2. Localized mortality of fish would be likely if many of these additives entered marine waters in an undiluted form. However, since all hazardous chemicals are handled and stored in accordance with strict regulatory requirements, the probability of such spills would be low. Overall, the degree of concern regarding adverse effects of drilling fluids and formation cuttings on regional fish populations is expected to be MINOR since any effects would be localized and generally restricted to deeper offshore waters outside the critical habitat of all anadromous and most marine fish species.

3.1.5 Effects of Drilling Wastes on Phytoplankton

Within the plume created by disposal of drilling wastes, decreased light transmittance and increased concentrations of suspended solids are the most likely factors which would affect phytoplankton. Other physical parameters including dissolved oxygen, temperature and salinity levels are not significantly altered by drilling waste discharges (Ayers *et al.* 1980a, b). However, some laboratory experiments have shown that undiluted and unbuffered drilling fluids can have a very high COD and high pH levels (Weir *et al.* 1974a).

Very few studies have examined the toxicity of drilling fluids to phytoplankton or other plants. However, EG and G Environmental Consultants (1976, cited in Gettleson 1980) determined the EC_{50} (median effective concentration or concentration of material required to produce a 50 percent reduction in cell numbers compared to controls) of three drilling fluid formulations with the diatom *Skeletonema costatum*. EC_{50} values ranged from 652 ppm to 18,000 ppm, depending on the drilling fluid formulation. Hardin (1976) found no definitive trends in phytoplankton photosynthesis immediately following exposure to drilling sump wastes. In the case of one fluid, the photosynthetic rate actually appeared to increase as a function of increasing sump fluid concentration. Longer exposure times also led to inconclusive results. After nine days exposure, phytoplankton photosynthesis was higher in one drilling sump waste and lower in two other fluids than in the control.

The release of drilling fluids and formation cuttings results in increased levels of suspended solids in the water column, primarily due to the large proportions of barite and bentonite clays used in mud systems. This suspension of fine clay particles results in increased turbidity levels and localized decreases in light intensities throughout the water column. The effects of increased turbidity on phytoplankton were previously discussed in relation to dredging. In brief, a very localized reduction in photosynthesis and growth rates of some species may be expected (Section 2.4.6), although populations would quickly recover once the discharge was terminated or they were naturally transported out of the associated turbidity plume.

Since most drilling wastes settle to the seafloor relatively close to the discharge site (Section 3.1.1) and the most abundant phytoplankton populations occur in the upper portion of the water column (LGL and ESL 1981), toxic effects of any drilling mud additives would be restricted to the immediate vicinity of exploration or production platforms. In addition, phytoplankton would be continuously transported into and out of waters contaminated by drilling wastes or associated turbidity plumes as a result of natural oceanographic circulation processes. These factors suggest that the degree of concern associated with potential effects of drilling wastes on regional phytoplankton populations of the Beaufort Sea would be MINOR.

3.1.6 Effects of Drilling Wastes on Zooplankton

Since most of the drilling fluids and formation cuttings discharged from offshore rigs rapidly settle to the ocean floor (Ayers *et al.* 1980a,b), the effects of drilling fluids on marine zooplankton have not been extensively examined. Field studies have generally failed to detect any acute effects of whole mud discharges (Ray and Shin 1975; Zingula 1975). Nevertheless, several components of drilling muds including defoamers, pH controllers, and other additives may be acutely toxic to zooplankton when present in sufficiently high concentrations. Neff *et al.* (1980) and Carr *et al.* (1980) examined the acute lethal and sublethal effects of various chrome lignosulphonate drilling muds on the opossum shrimp *Mysidopsis almyra*, and stated that much of the toxicity of drilling fluids is associated with the seawater-soluble components. Any toxic effects of these soluble constituents, however, would be extremely localized since a 1000-fold dilution of wastes would be expected within 0.1 to 1.0 km of the disposal site under favourable dispersal conditions (Dames and Moore Inc. 1978; Monaghan *et al.* 1980).

The physical effects of suspended formation cuttings and particles of barium sulphate and bentonite clays from drilling fluids may result in limited mortality of zooplankton and a range of sublethal effects. For example, high concentrations of fine particles can clog respiratory appendages (Robinson 1957), resulting in reduced filtering efficiency and possible suffocation. Conklin *et al.* (1980) observed ultrastructural damage in the midgut epithelium of the grass shrimp *Palaemonetes pugio* following relatively long-term (30 days) exposure to, and ingestion of, barite particles. Other filter-feeding crustaceans may experience similar effects following ingestion of suspended particles. Additional potential sublethal effects include altered behavioural responses due to altered light regimes (including predator-prey interactions). These effects would be similar in type to those previously discussed in relation to dredging (Section 2.4.7), but more localized due to the smaller size of turbidity plumes resulting from drilling waste discharge. The rapid dispersal and dilution of drilling fluids and fine formation cuttings once they enter the marine environment would also minimize the impacts of drilling-related suspended sediment on zooplankton. As a consequence of this dilution and dispersal of drilling wastes, and the localized nature of drilling activities, the degree of potential concern related to possible effects of these wastes on regional zooplankton communities in the Beaufort Sea is considered MINOR.

3.1.7 Effects of Drilling Wastes on Micro-organisms

Organic bactericides such as aldehydes, quaternary amines, diamine salts and chlorinated phenols are added to drilling fluids to prevent microbial degradation of organic additives and to suppress the formation of hydrogen sulphide by anaerobic sulphate-reducing bacteria (Gettleson 1980). In 1979, the use of chlorinated phenols was prohibited in oil and gas operations on the outer continental shelf of the United States due to the phenol's high environmental persistence. Bacteria in the drilling mud are also controlled by maintaining a high pH through the addition of sodium hydroxide or lime. Nevertheless, bacteria may still proliferate in drilling muds in the absence of bacterial control agents. For example, Page et al. (1980) reported that one mud intended for use in a study became heavily contaminated with bacteria prior to the experiment and had to be discarded.

The use of bactericides in drilling operations was reviewed by Robichaux (1975). As indicated in Table 3.1-2, organic bactericides are not included in the list of drilling fluid additives approved for use in the Canadian Arctic. The inorganic bactericides used in the Beaufort Sea would likely prevent bacterial proliferation in waste disposal areas until the concentrations of these compounds decreased due to natural dilution, although increased concentrations of naturally occurring marine bacteria are eventually likely in areas adjacent to drillships, exploration and production islands because of the other organic constituents of drilling fluids. However, the degree of regional concern regarding this phenomenon would be NEGLIGIBLE due to the localized nature of bacterial proliferation and the likelihood that the abundance of pathogenic species would not be increased if domestic sewage is adequately treated and not released in the same area as drilling wastes. The potential effects of trace metals contained in drilling fluids and formation water on marine bacteria are separately discussed in Section 3.1.10.

3.1.8 Effects of Drilling Wastes on Benthic Communities

Benthic flora and fauna can be affected by drilling wastes as a result of: 1) increased turbidity and suspended sediment concentrations; 2) direct smothering of sessile organisms; 3) changes in the physical nature of the substrate; 4) toxic effects of chemical additives, and 5) toxic effects and bioaccumulation of various trace metals. The latter area of potential effects is separately discussed for all marine organisms in Section 3.1.10.

3.1.8.1 Effects of Suspended Solids and Smothering

The release of drilling fluids and formation cuttings from exploration and production platforms would result in physical-chemical changes to the water column similar to those associated with dredging operations (Section 2.4.2), including increased water turbidity and suspended solid concentrations, reduced light intensities, and in some cases, decreased dissolved oxygen levels. These perturbations could result in reduced primary production of benthic microalgae and decreased feeding efficiency of some species of benthic invertebrates, although these effects would be more localized than those associated with dredging.

Direct burial of sessile benthic flora and fauna by formation cuttings and drilling fluids is the most obvious impact of drilling waste disposal. Menzie et al. (1980) documented burial of sessile epifaunal organisms by patches of drilled cuttings within a 75 m radius of a well site in the shelf break region off New Jersey. Benthic micro-algae and sessile fauna are most vulnerable to smothering by drilling wastes, while relatively mobile epifauna may escape to outlying areas. The degree of impact of drilling waste disposal on benthic communities is partially dependent on the composition of the natural substrate. Hard substrate benthic communities tend to be comprised of organisms which are less tolerant to fine sediments, and these habitats are generally situated where surge and current energies prevent or disperse any sediment accumulations (Meek and Ray 1980). For example, certain scleractinian corals are very sensitive to increased sedimentation and smothering. Thompson and Bright (1980) found that a 1000:1 dilution of one drilling mud formulation introduced into test chambers caused significant mortality of three species of coral within 65 hr, while four other species were unaffected.

Several in situ observations of the apparent effects of sediments from discharged drilling wastes on marine benthos have been summarized by Ray and Shinn (1975), Zingula (1975) and Benech et al. (1980). Ray and Shinn (1975) found encrusting organisms such as barnacles actually living within a downpipe where drilling wastes were released, while Zingula (1975) observed crabs and gastropods actively digging in cuttings and mud piles at a depth of 24 m in the Gulf of Mexico, apparently undisturbed by the material still falling through the water column. Benech et al. (1980) described significant differences in the benthic fouling communities between mud-free and mud-exposed locations, and suggested that this was a result of species-specific sensitivities to sedimentation and reduced light levels.

Some localized mortality of benthic organisms probably also occurs beneath cuttings piles in soft bottom habitats. Menzie et al. (1980) suggested that sessile epifauna (sea pens and sea anemones) as well as mobile epifauna (annelids, molluscs, brittle stars and crustaceans) were either smothered by formation cuttings or were induced to emigrate outside a drilling waste disposal zone at a well site off New Jersey. In addition, Tagatz et al. (1980) found that whole mud additions prevented the planktonic larvae of polychaete, coelenterate, and amphipod species from colonizing aquaria sediments. On the other hand, Lees and Houghton (1980) reported that benthic populations were not measurably affected by drilling waste discharge in Lower Cook Inlet, Alaska, probably as a result of the low rate of cuttings accumulation in this dynamic environment.

The effects of suspended sediments from drilling programs in the Beaufort Sea would probably be less pronounced than those observed in other regions since the area is comprised of soft substrate habitats and has naturally high background turbidities. In addition, many areas where drilling wastes would accumulate will likely already be almost devoid of infauna as a result of dredging programs associated with artificial island construction and preparation of glory holes for drillship operations. For example, Envirocon (1977) reported that construction of Isserk F-27 had no significant adverse effects on benthos in the area surrounding the artificial island. In contrast, data collected by Thomas (1978a) showed that the combination of glory hole dredging and release of drilling wastes during exploratory drilling at Tingmiark K-92 decreased the abundance of benthic invertebrates within distances less than approximately 4 km of the wellhead.

3.1.8.2 Altered Sediment Habitats

Changes in the particle size distribution of the bottom sediments may affect subsequent benthic recolonization after drilling ceases by inhibiting larval settlement and the burrowing or feeding abilities of some species (Carricker 1967; Menzie et al. 1980; Tagatz et al. 1980). Benthic recolonization was previously discussed in Section 2.1 in relation to the presence of artificial structures. Didiuk and Wright (1975) reported that the blanketing effect of small particles from drilling wastes drastically impaired the life cycle of *Chironomus tentans*, a freshwater burrowing chironomid. Release of drill cuttings and fluids during future exploration and production drilling in the Beaufort Sea will result in localized changes in microtopography and sediment size composition until water movements redistribute and level areas of waste accumulation. However, the amount of habitat altered by drilling waste disposal would not be regionally significant.

3.1.8.3 Effects of Whole Drilling Fluids

The toxic effects of some chemical additives used in drilling mud (Table 3.1-2) and completion fluid formations are of potentially greater concern with respect to benthic communities than sediment-related effects, although again any adverse impacts of these compounds would be relatively localized. The most significant impacts could occur in shallow, poorly circulated waters where toxic constituents may build up and adversely affect sessile or slow-moving benthic organisms. However, many of the highly toxic components listed in Table 3.1-2 are only used in small quantities and the majority of the future exploration and production drilling will take place in offshore waters where circulation is not restricted.

In addition to the individual toxicities of each compound used in drilling fluids, some combinations of additives could act synergistically. This is at least part of the rationale for testing the toxicity of whole drilling muds as well as their individual components. McLeay (1975) determined the acute lethal concentrations of five whole drilling muds used in the Beaufort Sea with a polychaete (Nereis vexillosa), clam (Mya arenaria), crab (Hemigrapsus nudus) and amphipod (Orchestia traskiana); the ranges in 96-h LC_{50} values were 23,000 to 220,000, 10,000 to 460,000, 53,000 to 560,000, and 14,000 to 560,000 ppm, respectively. Hardin (1974) also reported variable toxicity of whole drilling fluids from the Mackenzie Delta to freshwater invertebrates. Of four sump fluids tested by the latter author, one was acutely toxic to chironomids and three to amphipods. Various toxic effects of whole drilling muds on invertebrates were discussed during the 1980 Lake Buena Vista symposium. While procedures vary considerably between investigators, the available data indicate that specific mud formulations vary widely in their toxicities under similar test conditions, and also that larval or molting benthic invertebrates are generally more susceptible than adults in intermolt (Conklin et al. 1980; Gerber et al. 1980; Neff et al. 1980). Dames and Moore Inc. (1978) also found different acute toxic concentrations with shrimp, amphipods, isopods and mussels depending on mud type and duration of exposure. Tornberg et al. (1980) studied the acute toxicity of Prudhoe Bay whole drill muds and found that mysids and amphipods were most sensitive (96-h LC_{50} range <6-38 percent mud in seawater by volume), while isopods, gastropods and polychaetes were less affected (40 to 70 percent by volume). For a specific well, toxicity of the whole muds appeared to increase with drilling depth. Reported long-term effects of exposure to drilling muds include reduced growth rates in mussels and oysters, as well as reduced survival of mysids and polychaetes (Gerber et al. 1980; Rubenstein et al. 1980).

Overall, the degree of concern regarding the toxic and physical effects of drilling fluids and formation cuttings on benthic communities of the Beaufort Sea is expected to be MINOR because habitat loss and direct mortality would both be relatively localized, and recolonization would likely begin throughout waste disposal areas shortly after abandonment.

3.1.9 Effects of Drilling Wastes on Epontic Communities

Drilling wastes released in shallow waters beneath the ice or onto the ice surface may directly or indirectly affect epontic communities. Northern Technical Services (1981) documented the physical aspects of drilling effluent disposal both on the ice surface and in shallow water beneath the ice. The results of their study suggest several mechanisms through which epontic organisms could be affected by drilling wastes: (1) effluents deposited on the ice surface may flow through melt channels and cracks in the ice during spring melt; (2) effluents deposited on the ice surface would tend to reduce (or eliminate) light reaching the epontic community, and (3) effluents discharged in shallow water beneath ice may, in some circumstances, affect the bottom of the ice sheet adjacent to the outflow pipe. The Northern Technical Services (1981) study indicated that effects of these shallow discharges were limited to an area approximately 20 m from an outfall 2.5 m below the ice during a period of relatively high ambient currents. However, there have been no field surveys or laboratory experiments which have examined the effects of drilling fluids or their constituents on epontic organisms. Available information regarding the toxic and sublethal effects of drilling fluids and formation cuttings on arctic phytoplankton and zooplankton were summarized in Sections 3.1.5 and 3.1.6, while Section 3.1.10 discusses the effects of trace metals on planktonic organisms. The effects discussed in these sections would probably be similar in the case of epontic species. Dilution of under-ice discharges would likely reduce concentrations of toxic constituents to non-lethal levels within a relatively short distance of discharge sites, and rapid settling of heavier solids in the effluent should further reduce potential effects of drilling wastes on epontic flora and fauna. However, the chemical composition of drilling wastes released onto the ice surface could remain basically unchanged throughout the winter, and epontic organisms may be exposed to relatively undiluted drilling fluids which flow through melt channels in the ice. The duration of exposure of epontic organisms to these contaminants would probably be a major factor determining the nature of potential effects, although localized impacts of surface disposal of drilling wastes appear more probable than from under-ice discharges where effluents would be diluted.

The reduction or elimination of light as a result of ice surface disposal of drilling wastes could affect the development of the epontic algae directly beneath the discharge sites during the spring. Although the minimum light intensities necessary to initiate photosynthesis are apparently in the range of 54-108 lux (Grainger 1977), there is no available information regarding the range and spectrum of light intensities required for maximum productivity of epontic algae. Natural variability in the abundance of epontic organisms is high, even over relatively small distances (Alexander et al. 1974; Horner et al. 1974), and may be related to differences in ice type, thickness or depth of snow cover which would affect light permeability, and/or variability in nutrient concentrations (Alexander et al. 1974; Horner 1972). Northern Technical Services (1981) reported that natural processes in nearshore areas contributed sufficient quantities of sediment to the ice to give it a "dirty" appearance, and may normally limit light penetration in some areas. Although there have been no direct measurements of light intensities beneath drilling waste disposal sites, the amount of epontic habitat affected by disposal of drilling wastes on the ice surface is unlikely to be regionally significant. Consequently, the overall degree of concern regarding both physical and toxic effects of drilling waste disposal on epontic flora and fauna is expected to be MINOR.

3.1.10 Effects of Trace Metals

3.1.10.1 Introduction

Waste discharge associated with offshore exploration and production drilling will result in the introduction of trace metals into the Beaufort Sea. Although these metals are naturally present at low levels in both seawater and sediments of the region, a large number of studies have shown that elevated concentrations of some trace metals can have adverse effects on aquatic organisms. Formation cuttings, drilling fluids and produced water will be the major sources of trace metals. The specific formulation of drilling muds, in combination with the geological formations drilled, will largely determine the type and total concentration of trace metals entering the Beaufort Sea. Total concentrations in seawater and sediment are not necessarily good indicators of the potential biological impacts of a metal, since availability of a metal for biological uptake (bioavailability) is dependant on factors such as chemical form and dissolved concentration. These are influenced, in turn, by a variety of parameters including dissolved oxygen, pH, temperature, the presence of organic and inorganic ligands and the nature and concentration of other metals present in the water column or sediments.

Most formation cuttings removed by a shale shaker are relatively large and would settle rapidly through the water column. In areas of low current velocity, cuttings and their associated metals would accumulate on the seafloor close to the exploration or production platforms. Although trace metal content varies with the geological formation being drilled, all metals are bound within the mineral lattice structure and are not readily available for biological uptake. However, they may become soluble under certain conditions, producing increased concentrations of biologically available metals in the bottom waters (Lee and Mariani 1977; Lu and Chen 1977, both cited in Taylor and Demayo 1980). Trace metal concentrations in the upper portions of the water column are unlikely to be affected by formation cutting disposal.

Drilling fluids are primarily water-based colloidal suspensions with additions of barite (barium sulphate) to increase density, bentonite to increase viscosity, ferro-chrome lignosulphonate to act as a thinner and a variety of minor constituents to control other mud properties. Although barite and bentonite clays are the major components of drilling muds, the exact formulation depends upon drilling requirements. Fluids are generally recycled while drilling conditions remain constant, but changes in the formation and/or termination of drilling activities necessitate drilling mud disposal. Upon discharge, 93 to 95 percent of the solids settle out almost immediately, leaving only 5 to 7 percent of the particulate matter in the liquid phase (Ayers et al. 1980b; Ray and Meek 1980). These suspended materials tend to form a plume that is diluted as it moves away from the discharge pipe (Ray and Shinn 1975).

Although trace metals may be present in most of the drilling fluid constituents, the majority are associated with barite and are not readily available for biological uptake. Metals may be incorporated into the barite structure or they may form insoluble sulphide minerals. Large divalent ions such as lead (Pb) and mercury (Hg) are reported to isomorphously substitute for barium in the barite structure. Zinc (Zn) and iron (Fe) frequently occur in metallic bonding as the sulphides sphalerite (ZnS) and pyrite (FeS_2) (Kramer et al. 1980), as does nickel (Ni) (Goldschmidt 1954, cited in Kramer et al. 1980). In addition to occurring within the barite structure, lead can also be present as the sulphide galena (PbS). Mercury and cadmium (Cd) are commonly associated with sphalerite, arsenic (As) with pyrite arsenopyrite and galena, copper as malachite and with sphalerite, and nickel with pyrite (Applied Earth Science Consultants 1980). Sulfide mineral solubility and conditions affecting metal absorption and desorption on barite and bentonite surfaces are thus important factors in determining the release to solution and bioavailabiliy of these metals (Kramer et al. 1980). The solubility of ferro-chrome lignosulphonate determines the levels of biologically available chromium (Kramer et al. 1980).

Table 3.1-5 summarizes dissolved trace metal concentrations measured in the interstitial waters of a drilling fluid (Thomas 1978a) and compares these levels with those reported for the Beaufort Sea and world coastal oceans, and recommended levels for environmental protection. Only the concentration of mercury in drilling fluid interstitial water exceeds the range for unpolluted seawater, although it does not surpass the level considered hazardous by the Environmental Studies Board (1972). Concentrations of Cd, Fe and Ni exceed minimum risk levels, but do not surpass levels considered a hazard and lie within the observed ranges for either the Beaufort Sea or world coastal oceans. Consequently, even in undiluted form, the degree of concern regarding the effects of trace metals from drilling fluids in the upper water column appears to be minimal.

In a study comparing total trace metal concentration in a drilling mud with background sediment levels in the Beaufort Sea, Crippen et al. (1980) reported that concentrations of Hg, Pb, Zn, Cd and As in the drilling mud exceeded background levels by factors of 185, 35, 15, 9 and 2.4, respectively. A potential for metal accumulation in sediments surrounding drilling operations is suggested by these results, and supported by studies completed at Tingmiark K-91 in the Beaufort Sea (Thomas 1978a). The latter author reported that total Hg, Pb, Zn, Cd and Cr, as well as Cu concentrations decreased with increasing distance from the well head. Arsenic levels were not examined during the Tingmiark K-91 study. In contrast, a study of Pb, Zn, Ni, Vn and barium (Ba) concentrations in sediments near a drilling rig off New Jersey indicated that only Ba levels were directly attributable to drilling mud deposition (Mariani et al. 1980; Menzie et al. 1980).

TABLE 3.1-5

DISSOLVED TRACE METAL LEVELS IN DRILLING FLUID COMPARED WITH SEAWATER AND MARINE WATER QUALITY CRITERIA

Trace Metal	Drilling Fluid Interstitial Water[1]	Beaufort Sea (1977)[1]	World Coastal Oceans[1]	Minimal Risk[2]	Hazardous[2]
Cd (μg/L)	0.6	0.02 - 0.11	0.01 - 0.8	0.2	10
Cr (μg/L)	0.43	0.02 - 0.3	0.1 - 0.8	50	100
Cu (μg/L)	0.80	0.3 - 3.5	0.2 - 3.0	10	50
Fe (μg/L)	67.2	1.0 - 213	1.0 - 10	50	300
Hg (ng/L)	71.0	4.4 - 17.5	10 - 20	-	100
Ni (μg/L)	3.0	-	0.5 - 10	2	100
Pb (μg/L)	0.09	0.02 - 1.7	0.1 - 2.0	10	50
Zn (μg/L)	6.3	0.2 - 3.4	1 - 10	20	100

[1] Thomas (1978a)

[2] Environmental Studies Board (1972)

To estimate the potential impacts of Hg, Pb, Zn and Cu from drilling fluid disposal in the Beaufort Sea, MacDonald (1980) examined discharge levels in relation to the natural budget for each metal. As indicated by comparisons of total barite input per well with yearly inputs from the Mackenzie River and atmosphere, contributions of these metals through discharge of barite are expected to be relatively minor (Table 3.1-6). Therefore, it appears that input of Cu, Pb, Hg and Zn through drilling fluid release does not represent a regionally significant area of concern in the Beaufort Sea.

TABLE 3.1-6

SUMMARY OF METAL INPUTS TO THE BEAUFORT SEA
(from MacDonald 1980)

Metal	Mackenzie River Source (kg/yr, particulate and dissolved)	Atmospheric Source (kg/yr)	Disposal per Well from Barite Source (kg) Daily	Total
Copper	$3.5\text{-}5.5 \times 10^6$	$1\text{-}23 \times 10^4$	1.5	6×10^2
Lead	11.3×10^5	6×10^3	10	4×10^3
Mercury	$13\text{-}24 \times 10^3$	1.1×10^3	.055	22
Zinc	1.6×10^7	$2.6\text{-}28 \times 10^4$	18	7×10^3

As in the case of formation cuttings, the trace metal content in formation (produced) water would vary with the geological formation drilled. However, in contrast to cuttings and drilling fluids, dissolved metal levels in formation waters can be high relative to those naturally present in seawater. This is a result of chemical conditions within the formation (e.g. a strong reducing environment) which can release metals bound within the mineral lattice structure to the groundwater in ionic form. A comparison of the dissolved trace metal content in formation water flows from Kaglulik A-75 (Thomas 1978b) with baseline concentrations in seawater indicates that levels of Fe, Cu, Zn, Cr, Cd and Hg are elevated in formation water (Table 3.1-7).

TABLE 3.1-7

COMPARISON OF DISSOLVED TRACE METAL LEVELS IN KAGLULIK A-75
FORMATION WATER WITH BASELINE SEAWATER CONCENTRATIONS
AND MARINE WATER QUALITY STANDARDS

Trace Metal	Maximum Value of Formation Water (n = 2)[1]	1977 Seawater Baseline[1]	Minimal Risk[2]	Hazardous[2]
Cd (µg/L)	3.11	0.06	0.2	10
Cr (µg/L)	0.99	0.06	50	100
Cu (µg/L)	35	1.6	10	50
Fe (µg/L)	270	103	50	300
Hg (ng/L)	59	6.3	-	100
Ni (µg/L)	86.2	-	2	100
Pb (µg/L)	0.13	0.16	10	50
Zn (µg/L)	57	1.7	20	100

1 Thomas (1978b)

2 Environmental Studies Board (1972)

However, none of these concentrations exceed critical levels which are considered a hazard to the marine environment according to the Environmental Studies Board (1972). Although below suggested hazardous concentrations, Fe and Ni values in formation water from this exploratory well were approaching critical levels. Dissolved metal concentrations were also measured before and after a formation water flow at Tingmiark K-91 (Thomas 1978a), and indicated that formation water contributed to elevated Zn, Cr, Pb, Cd and Hg levels in near bottom waters (Table 3.1-8). With the exception of Hg, concentrations of trace metals were well below critical levels. An increase in total concentrations of Cu, Zn, Ni, Cr and Cd in the sediments was also reported after this formation water flow (Table 3.1-9), but with the exception of Cr and Cd, post-flow levels were within the the reported range for non-polluted world coastal oceans.

TABLE 3.1-8

COMPARISON OF MAXIMUM DISSOLVED TRACE METAL CONCENTRATIONS IN SEAWATER AFTER FORMATION WATER FLOW AT TINGMIARK K-91 IN 1978 WITH A 1977 SITE SURVEY AND MARINE WATER QUALITY STANDARDS

Trace Metal	Site Survey 1977[1]	Tingmiark K-91 Maximum Seawater Levels in 1978[1]	Minimal Risk[2]	Hazardous[2]
Cd (μg/kg)	0.02-0.11	0.01-0.16	0.2	10
Cr (μg/kg)	0.02-0.30	<0.01-1.48	50	100
Cu (μg/kg)	0.3-3.5	0.11-0.81	10	50
Fe (μg/kg)	1-213	2.4-192	50	300
Hg (ng/kg)	4.4-17.5	1-82	-	100
Ni (μg/kg)	-	0.26-3.20	2	100
Pb (μg/kg)	0.02-1.7	<0.2-12.05	10	50
Zn (μg/kg)	0.2-3.4	<0.3-5.3	20	100

1 Thomas (1978a)

2 Environmental Studies Board (1972)

TABLE 3.1-9

A COMPARISON OF TOTAL TRACE METAL CONCENTRATIONS IN TINGMIARK K-91 GLORY HOLE SEDIMENTS BEFORE (1977) AND AFTER (1978) FORMATION WATER FLOWS WITH THOSE FOR WORLD COASTAL OCEAN SEDIMENTS
Source: Thomas (1978a)

Trace Metal	Tingmiark Glory Hole Area 1977	Tingmiark Glory Hole Area 1978	World Coastal Ocean
Cd (ppm)	1.2-3.9	4.8-6.0	0.2-3.0
Cr (ppm)	6.4-31.8	111-241	2-200
Cu (ppm)	3.1-25.0	31.3-38.3	5-40
Fe (percent)	1.22-4.22	2.51-3.05	1-8
Hg (ppb)	<10-196	76-120	-
Ni (ppm)	8.2-18.7	30-110	2-120
Pb (ppm)	8.0-26.4	4-12	10-100
Zn (ppm)	15.4-31.8	130-174	5-200

Mercury therefore appears to be the trace metal of principle concern in relation to formation water release in the Beaufort Sea, although other metals may also be of concern when wells are drilled in deeper formations than Kaglulik and Tingmiark. In addition, under conditions favouring mobilization of metals from the sediments, bottom water Cr and Cd concentrations could increase to levels which may represent an area of potential biological concern. Contributions of different trace metals from multiple sources may also act synergistically to increase the degree of concern associated with release of formation water to the Beaufort Sea, particularly since this produced water will also contain emulsified oil. Subsequent sections briefly discuss the potential fate, and acute and sublethal effects of metals reported in elevated concentrations in formation waters or drilling fluids in the Canadian Arctic.

3.1.10.2 Cadmium

Pure cadmium is not abundant in nature and ranks 43rd in the list of elements present in seawater (Reeder *et al.* 1979a). In the Beaufort Sea, the concentration of dissolved cadmium ranges from 0.02 to 0.11 µg/L (Table 3.1-8), which is lower than that of other trace metals measured by Thomas (1978a). Dissolved cadmium is present in both drilling fluid interstitial water and formation water. Levels in interstitial waters have been reported to exceed background concentrations in the Beaufort Sea by a factor of 5 to 30 (Table 3.1-5), while cadmium in formation water may be up to 50 times background levels in seawater (Table 3.1-7). These data suggest that drilling waste disposal may lead to elevated levels of dissolved cadmium in waters surrounding exploration platforms, and particularly at production islands where produced water could be continuously discharged to the marine environment. Cadmium is also commonly associated with the sphalerite (ZnS) component of drilling mud barite, although this form is relatively insoluble and therefore unlikely to increase dissolved Cd levels. However, elevated concentrations of cadmium in sediments would be expected as a result of drilling mud disposal, while some cadmium may be complexed or adsorbed to organic matter present in the sediments or water column (Gardiner 1974).

Free cadmium ions (Cd^{2+}) and soluble complexes are of greatest biological concern when present in the marine environment. Cd^{2+} tends to form ionic pairs with chloride ions to produce cadmium chloride ($CdCl^{+}$), and may also be removed from solution through the formation of low solubility complexes such as cadmium carbonate, sulphide and hydroxide (Reeder *et al.* 1979a). During maximum summer activity, marine bacteria and their metabolic by-products may play a significant role in removing ionic cadmium and depositing it as CdS in the sediments (McLerran and Holmes 1974). However, Gardiner (1974) reports that cadmium may be mobilized from the sediments and re-distributed in the water column well after deposition through other processes. Although a number of processes can lead to removal of Cd from solution, it exhibits the greatest oceanic residence time of all trace metals found in drilling wastes (Table 3.1-10).

The toxicity of Cd may arise through numerous modes of action (Reeder *et al.* 1979a). For example, cadmium has been shown to inhibit several enzyme systems involved in intermediary metabolism due to its affinity for sulphur and carboxylate sites (Carty *et al.* 1976, cited in Reeder *et al.* 1979a). This element has also been shown to reduce the iron content of liver and blood plasma (Fox and Fry 1970, cited in Reeder *et al.* 1979a). Cadmium has been reported to bind to the tissue proteins of plants and animals (Casterline and Yip 1975, cited in Reeder *et al.* 1979a), although only one such protein type, metallothioneins, have been identified to date (Friberg *et al.* 1975, cited in Reeder *et al.* 1979a). Kohler and Riisgard (1982) found that as the body burden of cadmium increased in the mussel (*Mytilus edulis*), the amount of metallothioneins also increased. They propose that cadmium binding may

TABLE 3.1-10

OCEANIC RESIDENCE TIMES AND PROBABLE DISSOLVED FORMS IN SEAWATER FOR TRACE METALS PRESENT IN DRILLING WASTES (Source: Riley and Chester 1971)

Element	Oceanic Residence Time (Years)	Probable Main Dissolved Forms	Category[1]
Chromium (Cr)	350	Hydroxy complexes	III
Lead (Pb)	2×10^3	Pb^{2+}, $PbOH^+$, PbO^+	II
Nickel (Ni)	1.8×10^4	Ni^{2+}	II
Mercury (Hg)	4.2×10^4	$HgCl_4^{2-}$	II
Copper (Cu)	5×10^4	Cu^{2+}, $CuOH^+$	II
Zinc (Zn)	1.8×10^5	Zn^{2+}	II
Cadmium (Cd)	5×10^5	$CdCl^+$, Cd^{2+}	II

1 Categories defined by Martin (1970) as follows:
- I long residence time ($>10^6$ years), low reactivity
- II intermediate residence times (10^3 - 10^6 years)
- III short residence times ($<10^3$ years), elements typically enter as solids and settle rapidly

denature metallothioneins and new proteins are produced to compensate for the loss. Considerable quantities of unbound cadmium were also present in the mussels. In the most contaminated mussels, only 22 percent of the cadmium was bound to metallothioneins (Kohler and Riisgard 1982). The initial uptake of cadmium by animals is through the gastrointestinal tract (Reeder et al. 1979a), and as a result, accumulated cadmium is most likely to originate in food supplies.

Several factors have been reported to influence the toxicity of cadmium. La Roche (1972, cited in Environmental Studies Board 1972) noted a substantial increase in cadmium toxicity when copper or zinc were present at concentrations of 1 mg/L or more, while other studies have indicated an antagonistic relationship between zinc and cadmium (Reeder et al. 1979a). Salinity has also been shown to affect cadmium toxicity. For example, Von Westernhagen et al. (1975, cited in Leland et al. 1976) found reduced effects of cadmium on the embryonic development of herring (*Clupea harengus*) at higher salinities.

The acute lethal effects and bioaccumulation of cadmium have been extensively examined during the last decade. Some studies examining the acute and sublethal toxicity of cadmium are summarized in Table 3.1-11. Information on the toxicity of cadmium to phytoplankton, zooplankton and adult fishes is only available for freshwater species. The lowest concentration found to produce acute toxic effects was 0.04 mg/L in an amphipod (Lake and Thorpe 1974). Investigations on the sublethal toxicity of cadmium indicate significant effects at 0.05 mg/L in the snail *Australorbis glabratus* and at 1 mg/L in the eggs of both Atlantic and Pacific herring (Table 3.1-11). A comparison of these concentrations with those measured in drilling mud interstitial water (0.6 μg/L) and formation waters (3.1 μg/L) indicates that waste cadmium concentrations are likely to be significantly lower than those known to be toxic, even prior to dilution of drilling muds and formation water.

TABLE 3.1-11

SUMMARY OF ACUTE AND SUBLETHAL CONCENTRATIONS OF CADMIUM FOR MARINE LIFE

Species	Common Name	Concentration (mg/L)	Effect(s)	Reference
BENTHIC INVERTEBRATES				
Austrochiltonia subtenius	amphipod	0.04	*Acute*: 96-h LC_{50}	Lake and Thorpe 1974
Crassostrea virginica	oyster	0.12	*Acute*: 4-8 wk LC_{50}	Shuster and Pringle 1969, cited in ESB 1972
		0.1	*Acute*: 15-wk LC_{50}	Pringle *et al.* 1968
		0.1-0.2	*Sublethal*: little shell growth, lost pigmentation of mantle edge, coloration of digestive diverticulae after 20-wk exposure	Shuster and Pringle 1969, cited in ESB 1972
Australorbis glabratus	snail	0.05-0.1	*Sublethal*: stress reactions	Harry and Aldrich 1958

TABLE 3.1-11 (cont'd)

Species	Common Name	Concentration (mg/L)	Effect(s)	Reference
FISH				
Oncorhynchus gorbuscha (alevin)	pink salmon	3.65	*Acute*: 168-h LC_{50}	Servizi and Martens 1978
Oncorhynchus nerka (alevin)	sockeye salmon	4.5	*Acute*: 168-h LC_{50}	Servizi and Martens 1978
Belone belone (eggs)	needlefish	1.0	*Sublethal*: hatched larvae with frayed fin folds and caudal fins	Von Westernhagen *et al*. 1975, cited in Leland *et al*. 1976
Belone belone (eggs)	needlefish	<5	*Sublethal*: reduced pectoral fin activity in embryo	Von Westernhagen *et al*. 1975, cited in Leland *et al*. 1976
Clupea harengus harengus and *C. h. pallasi* (eggs)	Atlantic and Pacific herring	1.0	*Sublethal*: reduced eye diameter in yolk-sac larval stages	Von Westernhagen *et al*. 1975, cited in Leland *et al*. 1976
Clupea harengus harengus and *C. h. pallasi* (eggs)	Atlantic and Pacific herring	5-10	*Sublethal*: otic capsules absent in newly hatched larvae	Von Westernhagen *et al*. 1975, cited in Leland *et al*. 1976
Clupea harengus harengus (larvae)	Atlantic herring	2-3	*Sublethal*: loss of swimming ability	Von Westernhagen *et al*. 1975, cited in Leland *et al*. 1976

Several authors have examined the effects of cadmium on phytoplankton photosynthesis or growth. Although the cadmium concentration required to obtain a certain effect varies greatly with different species (Burnison *et al*. 1975), cadmium generally has deleterious effects on phytoplankton productivity. Extremely low concentrations (1 to 20 μg/L) of cadmium have been shown to inhibit phytoplankton photosynthesis (Patin *et al.* 1974, cited in Leland *et al.* 1976; Kayser and Spirling 1980; Rabsch and Elbrachter 1980) and growth (Conway 1978). In one study, cadmium was found to have a greater effect on photosynthesis than either zinc or chromium (Nakani and Korsak 1976, cited in Leland *et al.* 1976). Growth of the alga *Scenedesmus quadricauda* was significantly inhibited at concentrations as low as 6 ppb (Klass *et al.* 1974). In a review of metal research, Reish *et al.* (1978) report that residual concentrations of cadmium in phytoplankton and macrophytic algae range from 0.08 to 2.41 ppm.

Acute and chronic toxicities of various metals, including cadmium, to freshwater zooplankton were measured by Baudouin and Scoppa (1974) and Biesinger and Christensen (1972). In water from Lake Monate, the 48-h LC_{50} values for *Cyclops abyssorum*, *Eudiaptomus padanus* and *Daphnia hyalina* were 3800, 550 and 55 ppb, respectively (Baudouin and Scoppa 1974). Biesinger and Christensen (1972) reported that the chronic (3-wk) LC_{50} for *Daphnia magna* was only 5 ppb, and that reproduction was prevented at cadmium concentrations as low as 0.17 ppb. Bohn and McElroy (1976) also report that cadmium is rapidly accumulated by arctic marine zooplankton. Nevertheless, in a study of cadmium- and nickel-contaminated sediments in a cove near Cold Spring, New York, Kneip (1978) reported maximum suspended and dissolved cadmium concentrations up to 80 µg/L, but was unable to demonstrate any effect on zooplankton species composition. It was suggested that tidal action resulted in the exposure of zooplankton to high cadmium concentrations for only an hour or two at a time. Arctic amphipods exposed to 5, 10 and 20 percent concentrations of various drilling fluids initially accumulated cadmium, but the levels declined after 15 days for unknown reasons (Tornberg *et al.* 1980). Mussels (*Mytilus edulis*) have been found to take up cadmium at a linear rate throughout 36 and 124 day experiments, accumulating as much as 300 to 500 ppm cadmium (Kohler and Riisgard 1982). Body parts were found to accumulate the metal at different rates, in the order: body > mantle > muscles.

Bioaccumulation and sublethal and toxic effects of cadmium on benthic invertebrates have also been documented. Reish *et al.* (1978) reported that cadmium concentrations in tissues range from 0 to 3.4 ppm in polychaetes, from 0 to 108 ppm in pelecypods, and from 0.03 to 46 ppm in gastropods. Stenner and Nickless (1975) indicated that cadmium concentrations in benthic molluscs may vary from less than 0.5 ppm to 7.9 ppm, while Zook *et al.* (1976) reported levels of 0.00 to 3.19 ppm in a wide variety of benthic invertebrates. The acute lethal toxicity of cadmium to five invertebrate species was investigated by Lake and Thorpe (1974). As indicated earlier, lethal concentrations (LC_{50} - 96h) varied from 0.04 ppm for the amphipod *Austrochiltonia subtenius* to over 2000 ppm for various insect larvae. Several sublethal effects of cadmium on benthic fauna have also been reported. The gill lamellae of the shrimp *Paratya tasmaniensis* showed considerable evidence of damage by cadmium, including accumulation of granules in mitochondria, mitochondrial degradation, and some dilation of intercellular spaces and rough endoplasmic reticulum (Lake and Thorpe 1974). Cadmium concentrations of 0.05 to 0.1 ppm have also been shown to produce sublethal stress responses in the gastropod *Australorbis glabratus* (Harry and Aldrich 1958).

Cadmium levels in fish tissue have been reported to range from as low as 0.01 ppm to as high as 200 ppm (Reish et al. 1978). Bohn and McElroy (1976) found cadmium levels ranging from 0.26 to 1.6 ppm dry weight in fish from the Canadian Arctic. Naidu (1974, cited in Leland et al. 1976) found that residual cadmium and zinc concentrations in tissues increased with the size of Pacific hake (Merluccius productus), while concentrations of mercury increased with age of the fish. This suggests that cadmium and zinc concentrations may be physiologically regulated in at least this species. The acute toxic effects of cadmium on fish were reviewed by Reish et al. (1978). They reported that LC_{50} (96-h) values for threespine sticklebacks (Gasterosteus aculeatus) ranged from 6.5 to 25 ppm, while the value for rainbow trout (Salmo gairdneri) was only 6.6 ppb of cadmium nitrate. Birge et al. (1977, cited by Reish et al. 1978) found substantial mortality and teratogenesis in goldfish (Carrassius auratus) and rainbow trout (Salmo gairdneri) eggs when they were cultured over natural sediments containing from 0.1 to 1 mg Cd/kg.

Several authors have investigated the sublethal effects of cadmium on fish. Rosenthal and Alderice (1976) provided a review of sublethal effects of a number of toxicants on immature stages of fish species. They suggested that short-term exposure of herring eggs to cadmium can cause changes in the properties of external egg membranes, resulting in subsequent changes in egg diameter, volume, buoyancy and bursting pressure. These effects could cause a significant reduction in the survival of fish during later stages of development. Rosenthal and Alderice (1976) suggested that as a result of exposure to cadmium, decreased activity of enzymes involved in biosynthetic processes results in smaller, inactive larvae at hatching.

The potential effects and accumulation of cadmium in marine mammals and birds have not been well documented. Hunt (1979) provided data on concentrations of several trace metals in four female white whales harvested in the Mackenzie Delta. Cadmium concentrations in liver, muscle and muktuk were 1.47-2.74, 0.03-0.11 and 0.01-0.04 ppm, respectively. Buhler et al. (1975) found cadmium levels up to 12.0 ppm in kidneys of California sea lions, while Heppleston and French (1973) and Anas (1974, cited in Buhler et al. 1975) reported cadmium concentrations of 22 ppm and 15.6 ppm in gray seals and fur seals, respectively. Birds also occasionally contain high levels of cadmium, although its effects on birds are poorly documented (Bourne 1978). Tissues of the ashy petrel (Oceanodroama homochroa) from California coastal waters had cadmium levels twice as high as in Wilson's petrel (Oceanites oceanicus) from Antarctica (Anderlini et al. 1972, cited in Environmental Studies Board 1972).

Overall, the available data on the toxicity of cadmium suggest that concentrations which are likely to occur in areas of drilling waste disposal and formation water discharge in the Beaufort Sea would be considerably lower than levels which are known to be acutely toxic. Localized sublethal effects and accumulation of cadmium in the tissues of some marine flora and fauna may occur near exploration and production platforms, although these effects would be restricted to a relatively small proportion of the regional populations.

3.1.10.3 Chromium

Chromium is rarely found in uncontaminated natural waters (Beak 1978). Background concentrations in the Beaufort Sea have been reported to range between 0.02 and 0.3 μg/L (Table 3.1-5), placing it among the least abundant naturally occurring metals which may be introduced through discharge of drilling wastes. The primary sources of drilling waste chromium are formation water and chrome lignosulphate, a thinning agent used in drilling fluids. In the Beaufort Sea, chromium concentrations from these sources have been reported to exceed background seawater levels by factors of 1.5 to 22 and 16.5, respectively. Consequently, disposal of drilling fluids and formation water could increase dissolved chromium concentrations in the vicinity of exploration or production facilities.

Drilling mud chromium is primarily in the hexavalent form Cr^{6+}, which is most toxic to marine life (Environmental Studies Board 1972). Although Cr^{6+} is not strongly absorbed by soil or particulate matter, it does react strongly with oxidizable substances (e.g. organic molecules) to form Cr^{3+}. This cation has a marked tendency to form stable complexes with negatively charged inorganic or organic species (Taylor et al. 1979a), and can also react with water to form colloidal hydrous oxides (Offshore Drilling Fluid Disposal I/GSC 1982). Although certain conditions such as the presence of only low concentrations of oxidizable substances can favour the persistence of Cr^{6+} in the marine environment, the oceanic residence time is relatively short (Table 3.1-10).

Elevated chromium levels have been reported in sediments adjacent to drill sites in the Beaufort Sea and elsewhere (Bryant and Hrudey 1976; Hrudey and McMullen 1976; Thomas 1978a; Newbury 1979). However, chromium does not appear to be readily released from sediments, even in the presence of chelating agents (Barica et al. 1973, cited in Taylor et al. 1979a).

The toxicity of Cr^{6+} is related to its affinity to react with enzymes containing a sulphydryl group, thus inactivating certain proteins (Schroeder and Lee 1975, cited in Taylor et al. 1979a). The results of several acute and sublethal toxicity studies with chromium and various marine organisms are summarized in Table 3.1-12. Acute toxicity levels reported in the literature range from 10-12 μg/L in the oyster to 31.8 mg/L in coho salmon. Available data on sublethal toxicities of chromium indicate that significant effects occurred at 0.1 mg/L in the polychaete Capitella capitata, a species characteristic of organically enriched sites, and at 5.0 mg/L in the giant kelp. By comparison, dissolved chromium levels in the interstitial water of drilling fluids and formation water examined to date in the Beaufort Sea are 0.43 and 0.99 μg/L, respectively, and therefore considerably lower than concentrations shown to produce acute toxic or sublethal effects.

TABLE 3.1-12

SUMMARY OF ACUTE AND SUBLETHAL CONCENTRATIONS OF CHROMIUM FOR MARINE LIFE

Species	Common Name	Form and Concentration (mg/L)	Effect(s)	Reference
BENTHIC ALGAE				
Macrocystis pyrifera	giant kelp	Cr^{6+} 5.0	Sublethal: 50 percent reduction in photosynthesis	Clendenning and North 1960
BENTHIC INVERTEBRATES				
Capitella capitata	polychaete worm	0.1	Sublethal: decrease in reproductive output	Reish 1977
Neanthes arenaceodentata	polychaete worm	Cr^{6+} 0.3	Sublethal: decrease in reproductive output	Oshida 1977
Nereis virens	polychaete worm	1	Threshold toxicity	Raymont and Shields 1964, cited in ESB 1972
Ophiothrix spiculata	brittle star	1.7	Acute: 7-day LC_{50}	Oshida and Wright 1977
-	sea urchin	2.9-29	Acute: 48-h LC_{50}	Oshida and Wright 1977
Leander squilla	prawn	5	Threshold toxicity	Raymont and Shields 1964, cited in ESB 1972
-	oyster	1.0-1.2 x 10^{-2}	Acute: mortality	Haydu, unpub. data, cited in ESB 1972

TABLE 3.1.12 (cont'd)

Species	Common Name	Form and Concentration (mg/L)	Effect(s)	Reference
FISH				
Citharichthys stigmaeus	speckled sanddab flatfish	Cr^{6+} 31	*Acute*: 96-h LC_{50}	Sherwood 1975
		Cr^{6+} 5.4	*Acute*: 21-day LC_{50}	Sherwood 1975
Oncorhynchus kisutch	coho salmon	Cr^{6+} 31.8	*Acute*: 100 percent mortality	Holland *et al.* 1960

In contrast to cadmium, fewer studies have examined the acute toxic and sublethal effects of chromium, and as in the case of the former metal, many of these investigations have involved freshwater flora and fauna. Hervey (1949) found that the growth of some diatoms was inhibited at a chromium concentration of 320 ppb, but not at 32 ppb. In more recent studies, Wium-Anderson (1974) observed a 50 percent decrease in the photosynthesis of the diatom *Nitzschia palea* at 650 ppb of hexavalent chromium, while Clendenning and North (1960) reported that photosynthetic rates of the kelp *Macrocystis pyrifera* were decreased within 5 days by chromium concentrations of 1.0 ppm. It has been suggested that phytoplankton are more sensitive than fish to chromium (Strick *et al.* 1975, cited in Taylor *et al.* 1979a), although low chromium concentrations have been found to stimulate growth of some aquatic plants (NRCUS 1974). Reish *et al.* (1978) report residual chromium levels from 0.0 to 13.0 ppm in phytoplankton and marine macrophytic algae.

The effects of chromium on zooplankton are not as well documented, although Biesinger and Christensen (1972) found that 0.33 ppm resulted in reproductive impairment of *Daphnia*, and the 48-h LC_{50} was 2.0 ppm. These results are consistent with those of Anderson (1950) who reported that the toxic threshold of chromium with *Daphnia magna* was less than 1.2 ppm.

Young and McDermott (1975) reported that concentrations of chromium in marine molluscs from California ranged from 0.7 to 11 ppm, while Zook *et al.* (1976) found concentrations of 0.00 to 0.69 ppm in a variety of benthic invertebrates. Chromium levels in marine gastropods are reported to vary from

0.00 to 17 ppm, while levels in decapods range from 0.0 to 8 ppm (Reish et al. 1978). McCulloch et al. (1980) concluded that chromium in drilling muds is more available than lead or zinc to marine bivalve molluscs, although during short exposures, the trace metals monitored all have limited bioavailability. Page et al. (1980) monitored chromium uptake by the mussel Mytilus edulis from: (1) an aqueous solution of a medium density lignosulphonate mud; (2) a solution of ferrochrome lignosulphonate, and (3) a solution of Cr^{3+}. All solutions were of approximately equal concentration in total chromium, but mussels gained the most chromium from the Cr^{3+} solution, and the least from the used drilling mud. This study emphasizes the importance of considering the form of chromium available in discharged drilling muds, since this has a major effect on toxicity and uptake by aquatic organisms. Low concentrations of chromium have been shown to be acutely toxic to a variety of marine benthic invertebrates. Concentrations as low as 0.03 ppm caused a reduction in the number of progeny of the polychaete worm Neanthes arenaceodentata (Oshida 1977). A similar type of reproductive impairment was observed by Reish (1977) when Capitella capitata was exposed to 0.1 ppm of chromium. Oshida and Wright (1977) found that the LC_{50} (48-h) for chromium and sea urchins varied from 2.9 to 29 ppm, depending on test conditions. The same authors reported that the 7-day LC_{50} for chromium and the brittle star Ophiothrix spiculata was 1.7 ppm.

Reish et al. (1978) reviewed a number of investigations which show that chromium concentrations in fishes range from 0.01 to 4.9 ppm. McKee and Wolf (1963) suggested that fish are more tolerant of chromium than are members of lower trophic levels, but that concentrations greater than 1 ppm are still acutely toxic to fish. In addition to the studies summarized in Table 3.1-12, Murdock (1953) also reported that chromium concentrations from 1.2 to 2.0 ppm were lethal to sticklebacks.

Residual chromium levels and toxic effects of this metal on birds and mammals are not well documented, although studies with rats have shown that chromium is required for the maintenance of normal glucose tolerance in mammals and is also essential for cholesterol metabolism (Schroeder 1968; Vokal et al. 1975, both cited in Taylor et al. 1979a). Trivalent chromium is not particularly toxic if ingested by mammals, while hexavalent chromium accumulation without any toxic effects has been observed at a concentration of 5 mg/L (Taylor et al. 1979). The latter authors also suggest that there is no evidence of bioconcentration of chromium in aquatic food webs.

The results of the aforementioned studies, in conjunction with observations of chromium levels in drill muds and formation water from the Beaufort Sea, suggest that this metal will not have more than localized effects on marine flora and fauna of the region. Acute lethal effects are considered unlikely because of the rapid dilution of these wastes in the receiving environment, and the most probable effects would be accumulation of chromium in tissues of sessile invertebrates near exploration and production facilities.

3.1.10.4 Copper

Copper is widely distributed in nature, and is present in the Beaufort Sea in concentrations ranging from 0.3 to 3.5 μg/L (Thomas 1978a). Small amounts of copper are not lethal to aquatic organisms, and in fact are essential to the respiratory pigments of some animals (Wilber 1969, cited in Environmental Studies Board 1972). The primary drilling waste source of dissolved copper input to the Beaufort Sea would likely be formation water, since concentrations measured to date have exceeded background levels in seawater by a factor of 22 (Table 3.1-7). On the other hand, copper levels in drilling fluid interstitial waters examined by Thomas (1978a) were only 3 times higher than concentrations in seawater (Table 3.1-5). Copper may also be associated with drilling fluid barite where it occurs as sphalerite or malachite. Since the malachite form of Cu has been shown to be subject to release into the marine environment (Earth Sciences Consultants 1980, cited in MacDonald 1980), it may also contribute to dissolved Cu levels. Sphalerite, in contrast, is relatively insoluble and would only be expected to increase copper levels in sediments surrounding drilling operations.

The form of copper shown to be acutely toxic to marine life at high concentrations is the Cu^{2+} ion. This ion and $CuOH^{+}$ are the primary dissolved species in seawater (Table 3.1-10). Free copper may enter into reactions which will either remove it from solution or produce less toxic forms. When bound to organic molecules copper may be 100 to 1000 times less toxic than free Cu^{2+} (MacDonald 1980). As a result of these and other reactions, effects from the release of dissolved copper to the water column may be both temporary and localized.

Several studies which have examined the acute toxicity and sublethal effects of copper are summarized in Table 3.1-13. The lowest acute toxic level reported in the literature is a 7-day LC_{50} of 0.035 mg/L observed in the clam <u>Mya arenaria</u> (Eisler 1977). Although this is significantly higher than copper concentrations in interstitial water of drilling fluids (0.0008 mg/L), it equals that in formation water examined to date in the Beaufort Sea (0.035 mg/L). Concentrations of copper in formation water exceed levels where sublethal effects have been documented in the diatom <u>Nitzschia palea</u> (0.01 mg/L) and the copepods <u>Calanus plumchrus</u> and <u>Metridia pacifica</u> (0.005 mg/L). However, since copper acts synergistically when present with zinc, cadmium and zinc, and mercury (Environmental Studies Board 1972) the effective acute and sublethal toxic thresholds may be lowered in the presence of these metals.

TABLE 3.1-13

SUMMARY OF ACUTE AND SUBLETHAL CONCENTRATION OF COPPER FOR MARINE LIFE

Species	Common Name	Concentration (mg/L)	Effect(s)	Reference
PHYTOPLANKTON				
Thalassiosira pseudonana	diatom	0.2	*Acute*: lethal concentration	Braek and Jensen 1976
Nitzschia palea	diatom	0.01	*Sublethal*: complete inhibition of growth	Bryan 1976
ZOOPLANKTON				
Calanus plumchrus and *Metridia pacifica*	copepod	0.005 0.01 0.05	*Sublethal*: reduction in feeding rate	Reeve *et al.* 1976
BENTHIC ALGAE				
Macrocystis pyrifera	giant kelp	0.06	*Sublethal*: 30 percent decrease in photosynthesis after 2 days	Clendenning and North 1960
BENTHIC INVERTEBRATES				
Mya arenaria	clam	0.035	*Acute*: 7-day LC_{50}	Eisler 1977
Mytilus edulis	mussel	0.5	*Acute*: 48-h LT_{50}	Davenport 1977
-	Japanese oyster	1.9	*Acute*: 96-h LC_{50}	Fujiya 1960, cited in ESB 1972
FISH				
Clupea harengus (egg)	herring	1.0	*Acute*: high mortality	Blaxter 1977
Clupea harengus (fry)	herring	0.09-0.3	*Sublethal*: impaired activity	Blaxter 1977

Copper is extremely toxic to planktonic organisms, and has been shown to be concentrated from surrounding waters by factors ranging from 1000 to more than 5000 (Krumholz and Foster 1957). Clendenning and North (1960) reported a decrease in the photosynthetic production of the macrophyte Macrocystis at a copper concentration of 100 ppb in seawater, while Overnell (1976), Bentley-Mowat and Reed (1977), and Braek and Jensen (1976) all measured sublethal effects of copper exposure on marine phytoplankton. Each group of researchers noted marked reductions in photosynthesis and growth rates at copper concentrations less than 100 ppb, and Braek and Jensen (1976) found that 200 ppb was lethal to Thalassiosira pseudonana. Copper concentrations between 5 and 50 ppb intially caused a reduction in the standing crop of natural phytoplankton communities in controlled ecosystem experiments (CEPEX), but abundance returned to levels comparable with the control after 20 days. However, significant changes in species diversity of the phytoplankton community were recorded during this period (Thomas et al. 1977, cited in Reish et al. 1978). The susceptibility of zooplankton to copper varies greatly among species. In a comprehensive study of the acute lethal effects of copper on 10 species of marine zooplankton, Reeve et al. (1978, cited in Reish et al. 1978) found that 24-h LC_{50} values varied from 14 ppb to 2.78 ppm, depending on species. However, the mean lethal concentration for the ten species was less than 0.4 ppm. Biesinger and Christensen (1972) demonstrated that a copper concentration of 22 ppb would decrease the reproductive capacity of Daphnia, while 44 ppb was lethal to this freshwater zooplankter. Reduced growth of planktonic echinoderm larvae was also observed at 10 ppb of copper (Bryan 1976).

Bioaccumulation of copper by benthic marine organisms has been reported by several authors. Stenner and Nickless (1975) reported copper concentrations as high as 505 ppm in pelecypods from Spain and Portugal, while Ruddell and Rains (1975) found levels ranging from 10 to 2100 ppm in oysters from California. Reish et al. (1978) reported that copper concentrations in the tissues of a wide selection of benthic invertebrates ranged from 0.0 to 554 ppm. MacDonald (1980) suggests that filter feeders would be most susceptible to copper accumulation. The polychaete Nereis diversicolor has shown high rates of copper uptake from copper-rich sediments and may develop a tolerance to this metal (Bryan and Hummerstone 1971, cited in Environmental Studies Board 1972). However, copper toxicity or bioaccumulation could occur in predators of this species (Environmental Studies Board 1972). It is likely that benthic organisms will accumulate some copper from drilling wastes released to the Beaufort Sea. MacDonald (1980) suggests that this will primarily result in sublethal effects (e.g. community structure changes or histopathological effects), and be confined to relatively small areas. The acute toxic effects of copper on benthic invertebrates, like those of chromium, occur at very low concentrations. Eisler (1977) found that the 7-day LC_{50} of copper with the pelecypod Mya arenaria was 0.035 ppm of copper, and Davenport (1977) reported that the 48-h LT_{50} for the common mussel Mytilus edulis was 0.5 ppm.

Copper is a required element for the growth of fish species (Giesy and Wiener 1977), although it can be acutely toxic at higher concentrations when present as the divalent (Cu^{2+}) ion. With the exception of samples analyzed from Norway, residual concentrations of copper in fish (0.02 to 367 ppm) are considerably lower than those reported for benthic invertebrates (Reish et al. 1978). The toxicity of copper to aquatic and marine organisms not only varies with the species, but also with the physical and chemical characteristics of the water including temperature, salinity, turbidity and carbon dioxide concentration (Trama 1954). For example, Ellis and Ladner (1935) found that the degree of toxicity of copper to fish increased with the simultaneous presence of magnesium and phosphates. Blaxter (1977) reported that eggs of the Pacific herring (Clupea harengus) showed high mortality when exposed to 1 ppm copper, while impaired activity was noted in fry at copper concentrations as low as 0.09 to 0.3 ppm. Numerous sublethal effects of copper on fish have been documented. These effects have included inhibition of spawning by fathead minnows (Pimephales promelas; Mount 1968), depressed olfactory response in rainbow trout (Salmo gairdneri; Hara et al. 1976), loss of osmoregulatory activity in coho salmon (Oncorhynchus kisutch; Lorz and MacPherson 1977), altered plasma volume of striped bass (Roccus saxatilis; Courtois 1974, cited in Leland et al. 1976), increases in copper residues in liver and gill tissue of the brown bullhead (Ictalurus nebulosus; Brungs et al. 1973), and changes in cough frequency, locomotor activity and feeding behaviour of yearling brown trout (Salvelinus fontinalis; Drummond et al. 1973). Fish may avoid areas with elevated copper levels in their natural habitat; in laboratory experiments, Atlantic salmon (Salmo salar) avoided copper concentrations as low as 2.4 ppb (Sprague 1971, cited in Environmental Studies Board 1972).

Unlike cadmium and chromium, there is a more extensive literature describing the toxic effects and bioaccumulation of copper in birds. Copper and zinc are essential elements in nutrition of these vertebrates, and generally are present in relatively high concentrations in body tissues, particularly the liver (Anderlini et al. 1972; Sturges et al. 1974; Vermeer and Peakall 1979). Some differences in residual copper concentrations can occur among species although it appears that a metallothionein binding mechanism may be present in seabirds to allow elimination of excessive quantities of certain trace metals, including copper (Brown et al. 1977; Vermeer and Peakall 1979). Nevertheless, high concentrations of copper can be toxic to waterfowl. Henderson and Winterfield (1974) reported acute copper toxicosis in Canada geese following ingestion of copper sulphate at 600 mg/kg of feed. Necrosis and sloughing of the proventriculus and gizzard, as well as a greenish discolouration of the lungs, were noted by these authors.

The effects of copper and bioaccumulation of this metal in marine mammals are not well documented. Residual copper concentrations in the liver, muscle and muktuk of four white whales harvested in the Beaufort Sea were 4.86-9.88, 0.80-1.04 and 0.34-0.62 ppm, respectively (Hunt 1979). These concentrations are considerably lower than those reported for marine benthic invertebrates and fish in the available literature.

On the basis of the existing data regarding copper toxicity and bioaccumulation in marine flora and fauna, release of formation water from production islands is expected to be one of the most significant areas of potential concern associated with future hydrocarbon exploration and production activities in the Beaufort Sea. Although dissolved copper present in formation water would be rapidly diluted to sublethal concentrations at progressive distances from production islands, the long-term nature and multiple source inputs of this metal may contribute to significant bioaccumulation and sublethal physiological effects in benthic invertebrates and plankton. However, the overall degree of concern associated with copper contamination and these communities cannot be accurately assessed with the limited data available on dissolved copper levels within formation water in the Beaufort region.

3.1.10.5 Lead

Lead is widely distributed in the earth's crust, but is not required in even small quantities for the growth and development of terrestrial or aquatic organisms (Laws 1981). In the Beaufort Sea, dissolved concentrations of Pb in seawater have been reported to range from 0.02 μg/L to 1.7 μg/L (Table 3.1-5), while the probable dissolved species are Pb^{2+}, $PbOH^+$ and $PbCl^+$ (Table 3.1-6). Although both formation waters (Table 3.1-7) and the interstitial waters of drilling fluids (Table 3.1-5) contain dissolved lead, concentrations fall either within or below the observed range of values for the Beaufort Sea. This suggests that the impact of drilling waste disposal on water column Pb levels will be minimal. Lead is also associated with drilling mud barite as the sulphide galena. An increase in sediment Pb levels near offshore drilling platforms might be expected since this mineral is relatively insoluble.

Lead can exist naturally in several valence states: Pb(0)- metal, Pb^+, Pb^{2+} and Pb^{4+}. With the possible exception of Pb^+, all of these forms are of environmental importance (Demayo et al. 1980). However, organo-lead compounds, particularly those containing the tetralkyl group, are considered to be the most toxic forms of this metal (Chan and Wong 1978; Chan et al. 1979, both cited in Demayo et al. 1980). Ionic Pb such as that present in formation waters and interstitial waters of drill muds may be removed from solution by a number of processes. In the presence of a clay suspension (e.g. the bentonite in drilling muds), most of the lead ions are precipitated and adsorbed in forms such as $PbOH^+$ (Farrah and Pickering 1977, cited in Demayo et al. 1980). When anions are available (e.g. sulphate and phosphate), lead may also precipitate as low solubility salts (Demayo et al. 1980). The Pb which remains in solution is usually complexed with organic ligands to produce soluble but non-toxic colloidal and particulate compounds.

Lead sulphide (PbS) may also be introduced to the Beaufort Sea with barite, but it is non-toxic and virtually insoluble. Nevertheless, elevated levels of Pb will likely occur in sediments surrounding drilling rigs due to the contribution of PbS from barite. Benthic organisms could accumulate Pb in these habitats, but MacDonald (1980) considers migration of detectable levels of this element up the food chain unlikely. Although Pb contained in the sediments is non-toxic, certain reactions may lead to the formation of toxic compounds. The ability of clay materials to remove Pb from solution has been previously discussed. This process can be reversed by a decrease in pH or a change in the ionic composition of the interstitial waters within sediments. Other di- and trivalent ions compete and exchange with lead to allow the latter to come into solution (Griffin and Shimp 1976; Scrudato and Estes 1975, cited in Demayo et al. 1980). Certain inorganic and organic Pb compounds in the sediments may also be converted to the highly toxic tetramethyl lead by methylating bacteria (Wong et al. 1975).

The toxicity of lead appears to be attributable to an affinity for sulphydryl groups on proteins, and this results in tissue damage and interference with enzyme functioning (Waldron and Stofen 1974, cited in Laws 1981). Lead concentrations that produce either acute or sublethal effects in various marine organisms are summarized in Table 3.1-14. In the case of zooplankton and fish, data regarding acute effects of lead are only available for freshwater species.

TABLE 3.1-14

SUMMARY OF ACUTE AND SUBLETHAL CONCENTRATIONS OF LEAD FOR MARINE LIFE

Species	Common Name	Concentration (mg/L)	Effect(s)	Reference
PHYTOPLANKTON				
Skeletonema	diatom	1.0×10^{-2}	*Sublethal*: Decrease in growth rate and altered cellular structure	Rivkin 1979
BENTHIC ALGAE				
Macrocystis pyrifera	giant kelp	4.1	No adverse effects on photosynthesis	North and Clendenning 1958; in ESB 1972
BENTHIC INVERTEBRATES				
Crassostrea virginica	oyster	0.5	*Acute*: 12 week LC_{50}	Pringle, unpub. data, in ESB 1972
	oyster	0.3	*Acute*: 18 week LC_{50}	Pringle, unpub. data, in ESB 1972
Mya arenaria	clam	8.8	*Acute*: 7 day LC_{50}	Eisler 1977
Crassostrea virginica	oyster	0.1-0.2	*Sublethal*: Change in gonadal and mantle tissue after 12 weeks	Pringle, unpub. data in ESB 1972
Uca pugilator	crab	0.1	*Sublethal*: No apparent effect on regeneration	Weis 1976

The effects of lead on planktonic communities have been the subject of only a limited number of studies. Biesinger and Christensen (1972) provide data that demonstrate lead toxicity to zooplankton (Daphnia) at 0.3 ppm and significant loss of reproductive capacity at only 30 ppb. On the other hand, Zavodnik (1977) found that 1.0 ppm of lead had no significant effect on the respiratory metabolism of four species of marine macrophytic algae during 3 to 6 day exposures. Stewart (1977) reported that 10 ppm of lead in the water was sufficient to prevent growth and to cause loss of colour of the red alga Platythamnion pectinatum, while decreased growth rates and altered cellular structure were observed in the diatom Skeletonema at a lead concentration of only 0.01 ppm. Reish et al. (1978), in a review of studies regarding uptake of trace metals and other pollutants, indicate that marine macrophytes and phytoplankton contain residual lead concentrations in the range of 0.5 to 170 ppm, while the zooplankton examined contain residual concentrations from 11.6 to 81 ppm.

Concentrations of lead in pelecypods were reported to range from 0.7 to 10 ppm by Reish et al. (1976). Zook et al. (1976) found that residual lead concentrations varied from 0.00 to 1.57 ppm in a wide variety of invertebrate species. Eisler (1977) reported that in the case of Mya arenaria, the 7-day LC_{50} for lead was 8.8 ppm, while Weis (1976) found that a lead concentration of 0.1 ppm had no apparent effect on regeneration in the decapod Uca pugilator. Sublethal and acute toxic effects of lead on the oyster Crassostrea virginica occur at concentrations from 0.1 to 0.5 ppm (Table 3.1-14).

In freshwater fish, the 96-h LC_{50} of lead has been reported to range between 5.58 and 482 mg/L, depending on the species tested and water hardness (Demayo et al. 1980). Residual concentrations of lead in the tissues of fish range from 0.01 to 21.2 ppm (Reish et al. 1978). Sublethal concentrations of lead to fish are also highly variable and dependent on species, life history stage and water hardness. The lowest toxic concentration of lead reported in the literature is 4.0×10^{-3} mg/L for rainbow trout fry (Salmo gairdneri) (Davies et al. 1976, cited in Demayo et al. 1980). Several studies have indicated that suffocation through gill coating is one of the primary causes of lead toxicity to fish.

The effects of lead on birds, particularly waterfowl, have received extensive research in relation to ingestion of lead shot in hunting areas (e.g. Bates et al. 1968; Irwin and Karstad 1972). Symptoms of lead poisoning include weakness, weight loss, and eventually death with sufficient poisoning. Lead accumulates in several tissues, especially bone during laying periods. The significance of lead passed on in eggs is not fully understood (Finley et al. 1976). Lead appeared in lower concentrations in the scaup and scoters examined by Vermeer and Peakall (1979) than it did in the ducks' food, and it has been suggested that a binding agent (metallothionein) allows excretion of excess amounts of some trace metals. Intake of lead and other trace metals may also be related to the intake of grit required to grind food in the gizzard. Vegetation-eating ducks require more grit than ducks whose diet consists largely of fish or invertebrates, and are thus likely to absorb more metals associated with the grit (Vermeer and Peakall 1979).

The effects of lead on marine mammals are poorly documented. Braham (1973) reported that lead concentrations in brain tissue of California sea lions were 3.2 ppm, with concentrations greater than 2 ppm in brain tissue expected to be toxic. On the other hand, Hunt (1979) measured less than 0.1 ppm of lead in the liver and muscle tissue of four female white whales harvested in the Mackenzie Delta, but up to 5.98 ppm of lead in muktuk from these animals.

As indicated earlier, the primary source of lead input to the Beaufort Sea during future exploration and production drilling would be the release of formation water from production islands. The maximum concentration of lead in formation water examined by Thomas (1978b) was 0.13 μg/L, which is well below levels that are likely to cause significant sublethal or toxic effects. In addition, these concentrations would be further reduced through natural mixing and dilution in the receiving environment. The most probable effects of lead contained in formation water or other drilling wastes would be accumulation in members of lower trophic levels, since concentration factors of 1000 to 9000 and 16,000 to 20,000 have been reported in invertebrates and algae, respectively (Demayo et al. 1980). However, such accumulation would also be relatively localized because of the rapid dilution of lead-containing wastes following discharge, and there is no evidence to suggest that this metal would bioconcentrate in higher trophic levels (Demayo et al. 1980).

3.1.10.6 Mercury

Mercury is present throughout freshwater and marine ecosystems, and can occur in the marine environment in metallic (elemental) form or as inorganic and organic compounds. In seawater, the probable dissolved species is the inorganic salt $HgCl_4^{2-}$ (Table 3.1-6), which is present in the Beaufort Sea in concentrations ranging from 4.4 to 17.5 ng/L (Table 3.1-5). In aquatic ecosystems, most (90 to 99 percent) mercury is located in the sediments (Jernelov and Lann 1973, cited in Reeder et al. 1979b), where it may occur as: (1) particles of mercuric sulphide; (2) droplets of metallic mercury, or (3) chemisorbed or adsorbed on either organic or inorganic materials in the form of mercuric ion and methylmercuric ion (Krenkel 1974, cited in Reeder et al. 1979b).

Dissolved mercury is present in both formation waters and drilling fluids, and these sources could lead to localized increases in mercury levels in seawater adjacent to exploration and production platforms. Concentrations found in Beaufort Sea formation waters examined by Thomas (1978b) were 9 times higher than background seawater levels (Table 3.1-7), while mercury concentrations from drilling fluid interstitial waters exceeded these background levels by factors of 4 to 16 (Table 3.1-5). Crippen et al. (1980) found that concentrations of mercury in waste drilling fluid were 185 times higher than in surficial sediments of the Beaufort Sea. The greatest mercury contributions would likely be associated with drilling mud suspensions, where mercury is associated with barite and occurs predominantly as the relatively insoluble mercuric sulphide (MacDonald 1980). Mercuric sulphide occurs naturally in the form of cinnabar deposits. Sediments near these deposits may have mercury levels over 100 times that of background, without observable effects on local surroundings (Gavis and Ferguson 1972, cited in MacDonald 1980).

As in the case of other metals, mercury toxicity depends on its chemical form, which may be broadly classified as being either inorganic or organic. Inorganic forms include metallic mercury (Hg°) and various "salts" of mercury in which the element is in the Hg^{+} or Hg^{2+} oxidation state. Elemental mercury generally goes into the vapour phase and is lost from water (Wood 1974). The major organic compounds are phenyl mercury (e.g. phenyl mercury acetate), methoxy mercury (e.g. methoxy-ethyl mercury acetate) and alkyl mercury (e.g. methylmercuric acetate). Mercury has been found to be most toxic in the organic form, and the methylmercury group is the most dangerous of these compounds (Laws 1981).

Although mercury in drilling wastes is inorganic, methanogenic bacteria in the sediments are capable of converting virtually all forms of mercury into methylmercury (Laws 1981). This is apparently achieved by conversion to ionic Hg^{2+} (if not already present in this state) and then to methyl or dimethylmercury. Methylation can proceed under both aerobic and anaerobic conditions (Bisogni and Lawrence 1975, cited in National Research Council 1978), although the rate of conversion declines if oxygen concentrations are greatly reduced (Wood 1972, cited in Laws 1981). The rate of methylmercury synthesis is also controlled by the concentration of available Hg^{2+}, the composition of the microbial population, pH, temperature, redox potential and synergistic or antagonistic effects of other metabolic processes (Wood 1974, cited in Wood 1974). The methylation rate has been shown to be extremely rapid under optimum conditions (DeSimone et al. 1973, cited in National Research Council 1978). However, Sommers and Floyd (1974, cited in Leland et al. 1976) found that increasing the temperature enhanced the microbial transformation of mercury, and presumably low temperatures in the Beaufort Sea would slow down the reaction rate. In addition, published evidence indicates that mercury in drilling mud is not prone to methylation (MacDonald 1980). Methylation is possible but occurs very slowly since most mercury is in the sulphide state, and the rate controlling step (oxidation of sulphide to sulphate) is slow (Jernelov 1975, cited in MacDonald 1980). If discharged drilling muds are buried by accumulating sediment and maintained in an anoxic condition, mercury should be very effectively immobilized in the Beaufort Sea (MacDonald 1980).

The formation of insoluble sulphides is considered, in fact, to be one of the detoxification processes in natural waters (Reeder et al. 1979b). Mercuric ions in water form mercuric sulphide (HgS) by: 1) exchanging with other cations in sulphidic minerals; and 2) reacting with bioproduced hydrogen sulphide. Mercuric sulphide is the least soluble of all sulphides. Similarly, the methylmercury ion (CH_3Hg^+) reacts with the sulphide ion (S^{2-}) to form CH_3Hg_2S, the second most insoluble sulphide (Clarkson 1972, cited in Reeder et al. 1979b). Mercury compounds can also evaporate from the water surface, and this is another important detoxification mechanism (Johnson and Braman 1974, cited in Reeder et al. 1979b). A third detoxification process involves a bacterial conversion of methylmercury to methane and the volatile mercury metal. Since the rate of demethylation is much slower than methylation, this detoxification process will not prevent the steady-state concentration of methylmercury from building up in an ecosystem (National Research Council 1978).

There has been much interest in the effects of mercury on the environment during the past few decades. Sherbin (1979) recently summarized available data on mercury levels in resources from Canadian environments. It has been suggested that mercury uptake into cells is linked to a process requiring Mg^{2+}, since uptake of mercury by the freshwater diatom Synedra appeared to be enhanced in the presence of magnesium ions (Fugita et al. 1976, cited in Reeder et al. 1979b). Some phosphatase enzymes involved in cellular transport processes are known to require K^+, Na^+ and Mg^{2+} or Mg^{2+} alone for their activity (White et al. 1973, cited in Reeder et al. 1979b). The exact nature of mercury toxicity, particularly that of methyl- mercury, is not fully understood. The underlying reaction is believed to be the attraction between Hg and sulphydryl groups (-SH) on proteins (Goldwater 1971, cited in Laws 1981). Since proteins are essential constituents of cell membranes, mercury binding could readily disrupt numerous cell functions. The National Research Council (1978) reported that the acute toxicity of inorganic mercuric salts to marine invertebrates ranged from 0.05 to 1800 μg/L. For species listed in Table 3.1-15, acute toxicities vary from 50 μg/L in the crustacean Acttia clausi to 1.0 x 10^4 μg/L in the mollusc Clinocardium nuttalli. With the exception of the 0.05 μg/L level, these values are considerably higher than concentrations which are likely to be introduced to the Beaufort Sea in drilling muds (0.071 μg/L) and formation water (0.059 μg/L).

TABLE 3.1-15

SUMMARY OF ACUTE AND SUBLETHAL CONCENTRATIONS OF MERCURY FOR MARINE LIFE

Species	Common Name	Form and Concentration (μg/L)	Effect(s)	Reference
PHYTOPLANKTON				
Variety of species		60 organic: ethyl mercury phosphate	Acute: lethal concentration	Ukeles 1962, cited in ESB 1972
		0.1-0.6 organic: alkyl mercury	Sublethal: inhibition of photosynthesis	Ukeles 1962, cited in ESB 1972
Asterionella	diatom	370	Sublethal: cell division totally inhibited	Tompkins and Blinn 1976
Frasilaria	diatom	37	Sublethal: reduced rate of cell division	Tompkins and Blinn 1976
BENTHIC ALGAE				
Macrocystis pyrifera	giant kelp	50 inorganic: mercuric chloride	Sublethal: 50 percent reduction in photosynthesis	Clendenning and North 1960

TABLE 3.1-15 (cont'd)

Species	Common Name	Form and Concentration (µg/L)	Effect(s)	Reference
BENTHIC INVERTEBRATES				
Acttia clausi	crustacean	50 inorganic: mercury	Acute: 2.5-h LC_{50}	Corner and Sparrow 1956, cited in ESB 1972
Clinocardium nuttalli	mollusc	1.0×10^4 inorganic: mercuric chloride	Acute: 48-h LC_{50}	Portmann 1968, cited in ESB 1972
Hemigrapsus oregonensis	crab	1.0×10^3 inorganic: mercuric chloride	Acute: 48-h LC_{50}	Portmann 1968, cited in ESB 1972
Penaeus aztecus	shrimp	6×10^3 inorganic: mercuric chloride	Acute: 48-h LC_{50}	Portmann 1968, cited in ESB 1972
Penaeus duorarum	shrimp	100 inorganic: mercuric chloride	Acute: 48-h LC_{50}	Portmann 1968, cited in ESB 1972

As indicated earlier, organometallic mercury is much more toxic than metallic mercury, and enters the food chain primarily through uptake by aquatic plants, fish and members of lower trophic levels (Jernelov 1969, cited in Environmental Studies Board 1972). Available information suggests that diffusion controls the bioaccumulation of methylmercury in the tissues of higher organisms (Rakow and Lakowicz 1977, cited in National Research Council 1978), but attractive forces may accelerate the diffusion process. For example, Krauskopf (1956, cited in National Research Council 1978) showed that methylmercury had a greater affinity for living planktonic cells than for suspended clay particles. Once methylmercury enters cells, it is bound in such a way that even very low concentrations in water will rapidly build up in an ecosystem (National Research Council 1978). Young and Mearns (1978) reported that mercury concentration increased by up to a factor of 20 along the food chain, and suggested that the apparent biomagnification may be associated with the relatively long biological half-life of organic mercury. However, accumulation and biomagnification of residual mercury through the aquatic food chain appear to be more common in freshwater than marine ecosystems. If mercury biomagnification does occur, fish-eating birds and mammals at the top of the food chain are most likely to be seriously affected (Environmental Studies Board 1972).

Phytoplankton are adversely affected by even very low concentrations of inorganic or organic mercury. Gaechter (1976, cited in Leland et al. 1976) found that inhibition of photosynthesis decreased in the following order of trace metals: Hg > Cu > Cd > Zn > Pb. Growth of the diatom Phaeodactylum tricornutum was inhibited by inorganic mercury at concentrations in the ppb range (Bryan 1976), while as little as 1 ppb of mercury resulted in a reduction in growth rates of marine dinoflagellates (Kayser 1976). In addition, mercury levels as low as 0.1 ppb were found to reduce algal numbers and species diversity in periphyton communities examined by Sigmon et al. (1977). The same concentration (0.1 ppb) of some alkylmercurial fungicides decreased the photosythesis of the marine diatom Nitzschia delicatissima (Panel on Mercury 1978), while 0.1 to 0.6 ppb of alkyl mercury has been shown to inhibit photosynthesis and growth of marine phytoplankton (Ukeles 1962, cited in Environmental Studies Board 1972). Mercuric chloride and methylmercuric chloride were found to inhibit the lipid biosynthesis in some photoplankton species, and it has been concluded that algae may be highly susceptible to alkylmercurials because of the high lipid content of their cell membranes (Panel on Mercury 1978).

The limited available data suggest that the toxicity of mercury to marine zooplankton is also relatively high, with larval stages appearing to be more susceptible to this metal than adult invertebrates (Connor 1972; Bernhard and Zattera 1975; Vernbert et al. 1973, cited in Bryan 1976). For example, the 24-h LC_{50} values for larvae and adults of the marine copepod Acartia tonsa were 3 ppb and 34 ppb, respectively (Reeve et al. 1976).

Acute mercury toxicity varies widely among marine invertebrates, and lethal concentrations range from 0.05 to 1800 ppm for inorganic mercuric salts, with larval stages being generally most sensitive to mercury exposure (National Research Council 1978). Two-hour LC_{50} values in the range from 13,000 to 18,000 ppb have been documented with mercury and the planktonic larvae of marine benthic fauna (Bernhard and Zattera 1975), and sublethal effects have been observed at concentrations as low as 5 ppb (Bougis 1962, 1965; Soyer 1963, both cited in Bryan 1976). Luoma (1977) found that polychaetes and shrimp can rapidly accumulate dissolved mercury from seawater, and that this is followed by a rapid biochemical transformation to a complexed form within the body and then a slow depuration from the organism. Overall, the retention of mercury by an organism depends on its chemical form and pathway of accumulation. Bioaccumulation of mercury in benthic infauna has been suggested at two stations near drilling fluid discharge sites in the Beaufort Sea (Crippen et al. 1980). Mariani et al. (1980) also found that brittle stars (Amphioplus macilentus), molluscs (Lucinoma filosa) and polychaete worms in the vicinity of a drill rig off New Jersey accumulated significant amounts of mercury, although drilling activities resulted in negligible mercury discharges (Ayers et al. 1980a), and sediment mercury levels were below the detection limit of 0.05 µg/g (Mariani et al. 1980). The degree of mercury accumulation by benthic invertebrates may also depend on their trophic level, and whether they feed on the bottom or from the water column (Holm and Cox 1974, cited in Reeder et al. 1979b). For example, in a freshwater ecosystem, bottom feeders accumulated ten times as much mercury as planktivores (Hamilton 1972, cited in Reeder et al. 1979b).

Several authors (Kusher 1974; Colwell and Nelson 1974; Walker and Colwell 1974, all cited in Leland et al. 1976) have reported the presence of Hg-resistant bacteria in natural waters, sediment and as part of planktonic communities. For example, natural populations of marine bacterioplankton showed enhanced mercury tolerance following exposure to 1 mg/L of mercury (Farooq et al. 1977). Heterotrophic activity in this population initially decreased to 1 percent of control levels, while bacterial biomass decreased to 8-40 percent of the control. However, both parameters were reported to return to 'normal' within 5 days. Walker and Colwell (1973) indicated that some mercury-resistant bacterial populations were also able to degrade oil. Since formation water will also contain some petroleum hydrocarbons, the presence of these strains in the Beaufort Sea, although not presently documented, could reduce the potential synergistic effects of trace metals and oil near production facilities.

The most common form of mercury found in fish is methylmercury. The pathways of mercury accumulation by fish have not yet been fully documented, although it is known to at least enter via the alimentary canal (i.e. from food), the gills, and through the skin (Reeder et al. 1979b). In freshwater fish, significant bioconcentration of mercury can occur as a result of exposure to very low, nontoxic mercury levels in the water (Reeder et al. 1979b). Similar bioaccumulation by fish in marine waters has not been reported, although marine species exposed to methylmercury associated with industrial effluents in Minamata Bay and Niigata in Japan accumulated up to 40 µg/g of mercury.

In a study of the acute toxicity of several trace metals to rainbow trout, Hale (1977) reported that mercury was second only to cadmium in toxicity. Mercury toxicity to freshwater fish appears to be greater at higher temperatures (Rehwoldt *et al.* 1972; MacLeod and Pessah 1973, both cited in Reeder *et al.* 1979b), low levels of dissolved oxygen (Lloyd 1961, cited in Reeder *et al.* 1979b) and low pH (Larsson 1971, cited in Reeder *et al.* 1979b). However, the simultaneous presence of copper has been shown to be antagonistic to the toxicity of mercury to fish (Roales and Parlmutter 1974, cited in Reeder *et al.* 1979b). Documented sublethal effects of mercury on fish include reduced activity of the liver enzyme δ-aminolevulinate (Bryan 1976), decreased olfactory response (Hara *et al.* 1976), decreased swimming fatigue velocity (Alexander 1974, cited in Leland *et al.* 1976) inhibition of sodium uptake (Meyer 1952, cited in Renfro *et al.* 1974), depression of ion transport in osmoregulatory systems (Renfro *et al.* 1974) and impaired swimming performance (Bryan 1976).

As for other trace metals, limited information has been published on the effects of mercury on marine mammals. Hunt (1979) found that like cadmium and copper, mercury was accumulated in the livers of four female white whales harvested from the Mackenzie Delta, but was present in considerably lower concentrations in muscle tissue and muktuk.

Birds are able to tolerate methylmercury levels in their diet approaching 0.5 mg/kg (Reeder *et al.* 1979b). Surface-feeding birds which feed primarily on vegetation and macroinvertebrates have been found to have liver mercury levels which are considerably less (< 10 mg/kg total Hg) than those measured in the liver of fish-eating birds (100 mg/kg total Hg) (Fimreite 1974, cited in Reeder *et al.* 1979b). In Sweden, two species of fish-eating birds (the osprey, *Pandion haliaetus*, and the great-crested grebe, *Pocideps cristatus*) have shown a gradual increase in feather mercury content which parallels the increase in industrial use of mercury in the region (Johnels and Westermark 1969, cited in Environmental Studies Board 1972). The percentage egg hatch has been shown to be lower at mercury levels above 0.5 mg/kg Hg, although mercury does not appear to affect eggshell thickness (Fimreite 1970, cited in Reeder *et al.* 1979b). This has led to the suggestion that mercury bioaccumulation may reduce the reproductive capacity of birds at the top of food chains (Fimreite *et al.* 1970, cited in Environmental Studies Board 1972), including the white-tailed sea eagle (*Haliaetus albicilla*) in regions of Finland where the species feeds on marine birds and fish (Henriksson *et al.* 1966, cited in Environmental Studies Board 1972).

In summary, localized areas of mercury contamination in sediments may result from discharge of drilling wastes and formation water in the Beaufort Sea. The degree to which inorganic and organic mercuric compounds could be accumulated by local flora and fauna is not presently known, nor are the potential synergistic reactions which may result from the presence of other trace metals and petroleum hydrocarbons. However, on the basis of the available information, there is no reason to expect that more than localized sublethal effects would result from the low concentrations of mercury which are likely to enter the Beaufort Sea during future exploration and production activities.

3.1.10.7 Nickel

Nickel is relatively abundant in the earth's crust and is now considered an essential trace element in the metabolic processes of some organisms (Taylor et al. 1979b). In seawater, it ranks 24th in abundance (Taylor et al. 1979b) with concentrations ranging from 0.5 to 1 μg/L in world coastal oceans (Table 3.1-5). Background nickel concentration values are not available for the Beaufort Sea. Nickel in drilling wastes is not as serious a potential concern as some other trace metals (i.e. cadmium and mercury) discussed previously, but it may nevertheless be present in relatively high concentrations in formation water (Table 3.1-7) and to a lesser extent within the interstitial water of drilling fluids (Table 3.1-5). A comparison of the concentrations from each of these sources with the background levels in the Beaufort Sea is not possible, although if concentrations of nickel in world coastal oceans are representative as baseline values, Ni levels in interstitial waters fall within the normal range, while those in formation waters exceed background levels by a factor of 8.6 to 172. Formation water discharge could therefore lead to elevated dissolved nickel levels although concentrations measured by Thomas (1978b) are still considerably less than the 100 ppb of nickel which has been suggested as a lower limit for posing a hazard to marine organisms (Environmental Studies Board 1972). Nickel may also be introduced to the Beaufort Sea in association with the pyrite component of discharged barite. Since this mineral is relatively insoluble, accumulation of nickel in the sediments near drilling waste disposal sites could be expected.

Nickel occurs in a series of valence states from -1 to +4, although the ionic Ni^{2+} state is the probable dissolved and toxic species in seawater (Table 3.1-10). Nickel can also form stable, solid, organometallic compounds, while Ni^{2+} halides form a very large number of complexes with ligands which often have nitrogen or phosphorus donor atoms (Nicholls 1973, cited in Taylor et al. 1979b). The presence of suspended solids removes a large percentage of dissolved free nickel from solution (Taylor et al. 1979b). Thus, the presence of drilling mud bentonite and fine formation cuttings in waters surrounding a drilling rig could reduce locally elevated Ni ion concentrations. There appears to be no available information on the mobility of sediment-bound nickel.

Nickel has been shown to be less soluble, and hence less toxic, at a higher pH (Hutchinson and Stokes 1975, cited in Taylor et al. 1979b). In freshwater environments, water hardness also affects nickel toxicity. For example, the 96-h LC_{50} values for fathead minnows increased from 4.58 and 5.18 ppm in soft water (20 mg/L as $CaCO_3$) to 25.0 and 28.0 ppm in hard water (210 mg/L as $CaCO_3$), and 42.4 ppm in very hard water (360 mg/L as $CaCO_3$) (Pickering and Henderson 1966; Pickering 1974, both cited in Reeder et al. 1979b). Under alkaline conditions, a nickel cyanide complex had no apparent toxic effects on fish at concentrations below 100 mg/L (Doudoroff 1956, cited in Environmental Studies Board 1972).

The vast majority of the studies completed on nickel toxicity and phytoplankton have focused on freshwater species. Stokes et al. (1973, cited in Taylor et al. 1979b) found that the freshwater algae Chlorella and Scenedesmus can develop a tolerance to nickel. The growth of laboratory cultures of phytoplankton was inhibited by a nickel concentration of 0.5 ppm, while members of the same species collected from nickel-contaminated lakes could still grow in water containing 3.0 ppm of nickel. A synergistic effect between nickel and copper has been shown in toxicity studies with freshwater algae (Hutchinson 1973, cited in Taylor et al. 1979b).

The literature describing the sublethal and acute toxic effects of nickel on planktonic organisms and benthic invertebrates is extremely limited, particularly for marine ecosystems (Table 3.1-16). Acute toxicity (48-h LC_{50}) of nickel to three freshwater zooplankton species ranged from 1.9 to 150 mg/L (Baudouin and Scoppa 1974). Nickel ions appeared to affect the cell wall of the marine bacterium Arthrobacter marinus (Cobet et al. 1970). Cell division was interfered with but growth continued, and this resulted in very enlarged cells (Cobet 1968, cited in Jones 1973).

The effects of nickel on marine fish are poorly documented, although Podubsky and Stedronsky (1951) provide evidence that nickel is less toxic to fish than copper, iron or zinc. Brown and Dalton (1970, cited in Taylor et al. 1979b) reported that the 48-h LC_{50} value of nickel with rainbow trout was 32.0 mg/L, while the lethal concentration with sticklebacks has been estimated at 0.8 ppm (Jones 1939). Tillery and Thomas (1980) found that concentrations of nickel in sheepshead (Archosargus probatocephalus) muscle tissue were significantly higher in fish captured in the vicinity of a Gulf of Mexico petroleum production platform than in other areas of the Gulf. However, the authors suggested that the high levels of trace elements introduced into the area by the Mississippi River limited interpretation of any trends in residual nickel levels in this species.

The uptake and toxicity of nickel in birds and marine mammals is very poorly documented, although evidence from freshwater studies indicates an absence of nickel biomagnification through the food web (Hutchinson et al. 1975, cited in Taylor et al. 1979b). Anderlini et al. (1972) attributed concentrations of nickel in liver and bones of ashy petrels (Oceanodroma homochroa) from California to industrial wastes.

Despite the rather limited amount of information which appears to be available regarding the effects of nickel on marine flora and fauna, concentrations of this element in formation water examined to date in the Beaufort region are below the lower limit considered a hazard to marine resources. In addition, there is no evidence of bioconcentration of nickel in either freshwater or marine ecosystems, and the concentrations of Ni^{2+} present in drilling wastes discharged from exploration and production facilities would decrease rapidly in the surrounding waters due to natural mixing.

TABLE 3.1-16

SUMMARY OF ACUTE AND SUBLETHAL CONCENTRATIONS OF NICKEL FOR MARINE LIFE

Species	Common Name	Concentration (mg/L)	Effect(s)	Reference
BENTHIC INVERTEBRATES				
Crassostrea virginica (larvae)	oyster	1.2	Acute: 12-day LC_{50}	Reeve *et al.* 1976
Mercenaria mercenaria (larvae)	clam	5.7	Acute: 8-10 day LC_{50}	Reeve *et al.* 1976
Mya arenaria	clam	>50	Acute: 7-day LC_{50}	Eisler 1977

3.1.10.8 Zinc

Zinc is an essential element at trace levels, and the concentration in the Beaufort Sea is estimated to range between 0.2 and 3.4 µg/L (Table 3.1-5). Dissolved zinc is present in both drilling mud interstitial water (Table 3.1-5) and formation waters (Table 3.1-7). Studies completed in the Beaufort Sea by Thomas (1978a,b) indicated that zinc levels in interstitial waters of drill muds exceed concentrations of this element in seawater by a factor of 2 to 3, while Zn in formation waters may be up to 33 times higher than background values. These data suggest that dissolved Zn concentrations may be locally increased as a result of drilling activities, particularly near production platforms. Zinc may also be discharged as sphalerite (ZnS) in association with drilling mud barite. Although zinc is relatively insoluble in this form and minimal effects on dissolved concentrations would be expected, increased Zn levels in the sediments would likely occur in waste disposal areas (MacDonald 1980).

The toxic form of zinc is the Zn^{2+} ion (Zirino and Yamamoto 1972, cited in Taylor and Demayo 1980), and this is also the most probable dissolved species in seawater (Table 3.1-10). In addition to persisting as Zn^{2+}, zinc may form a range of inorganic compounds and organic complexes, and become adsorbed onto or occluded in inorganic (Zn^{2+} - clay) or organic colloids (Zn^{2+} - humic acids) (Taylor and Demayo 1980). The proportion of zinc in each of these forms varies with factors such as pH, the total amount of zinc in the water, and the presence of organic and inorganic compounds and other metal ions (Florence and Batley 1977, cited in Taylor and Demayo 1980). In seawater, for example, model calculations have shown that the expected proportion of uncomplexed Zn^{2+} would be 51 percent at pH 7 and only 17 percent at pH 8.1 (Zirino and Yamamoto 1972, cited in Taylor and Demayo 1980). The present pre-disposal pH range for drilling fluids is 6 to 8.5 (Friesen 1980). This model suggests that a closer restriction of the pH range of discharged drill muds to a pH closer to 8 could minimize localized increases in free zinc levels. Documented formation water pH values, as exemplified by Kaglulik A-75 values of 8.3 (Thomas 1978b), could simultaneously favour maintenance of low Zn^{2+} levels.

Several processes result in Zn mobilization from the sediments. Dissolved oxygen concentrations at the sediment-water interface have a significant effect on zinc mobilization. Under reducing conditions, zinc will remain in the metallic sulphide form and free zinc ions will precipitate as ZnS. On the other hand, oxidizing conditions will lead to increased Zn^{2+} concentrations in the bottom waters. The mechanism is believed to involve the conversion of metallic sulphide to the more soluble carbonate, hydroxide, oxyhydroxide, oxide or silicate forms (Lu and Chen 1977, cited in Taylor and Demayo 1980). Metal transfer between sediments and bottom waters is also dependent on the nature and concentration of other metals, the organic and inorganic ligands present, pH and particle size composition of the bottom sediments (Lu and Mariani 1977; Lu and Chen 1977, both cited in Taylor and Demayo 1980).

The mechanism of zinc toxicity appears to be related to an interference with gaseous exchange (Taylor and Demayo 1980). Studies which have examined the acute toxic concentrations of zinc for several marine organisms are summarized in Table 3.1-17, although like most trace metals, the vast majority of the studies conducted to date have been directed at freshwater species.

Zinc has been shown to be toxic to phytoplankton, although it is generally considered less toxic to planktonic communities than cadmium, copper, lead or mercury (Gaechter 1976, cited in Leland et al. 1976). Normal growth has been reported at 200 ppb (flagellate) and 800 ppb (diatom), while zinc concentrations from 50 to 200 ppb inhibited or reduced growth of other diatom species (Bernhard and Zattera 1975; Kayser 1977, cited in Rabsch and Elbrachter 1980). The unicellular alga Chlorella has been shown to develop a resistance to zinc after several generations. Resistant cultures showed a marked resistance to zinc uptake, as well as a lower number of cell wall sites available for adsorption of zinc ions (de Filippis and Pallaghy 1976, cited in Taylor and Demayo 1980). Similarly, the filamentous alga Hormidium rivulare from a river with high zinc content was found to be much more tolerant of zinc than the same species collected from a river with a low zinc concentration (Say et al. 1977, cited in Taylor and Demayo 1980).

Zinc is relatively toxic to zooplankton and benthic invertebrates (Table 3.1-17), and may also result in a range of sublethal effects. Reduced growth and structural abnormalities in the planktonic larvae of benthic organisms have been documented at zinc concentrations greater than 0.04 ppm (Bougis 1962, 1965; Soyer 1963; Timourian 1968; Bereton et al. 1973, all cited in Bryan 1976). Acute toxicity of zinc to invertebrates has occurred at concentrations as low as 0.195 mg/L in the clam Mercenaria mercenaria but at 200 mg/L in another clam, Clinocardium nuttalli (Table 3.1-17). Reish et al. (1976, 1978) indicate that residual levels of zinc in benthic invertebrates range from 0.006 to 5000 ppm, which indicates a considerable degree of bioaccumulation, particularly in pelecypods. McDonald (1980) suggests that filter feeders would be most likely to accumulate zinc from contaminated sediments in the vicinity of drill rigs in the Beaufort Sea.

Bacteria have been shown to be a pathway through which zinc can accumulate in higher trophic levels. For example, bacteria cultured in media with sediments contaminated by 2 mg/L Zn for 7 days at 28°C were fed to worms (Tubifex sp. and Limnodrilus sp.), which in turn accumulated 868 mg Zn/kg (dry weight). On the other hand, worms fed bacteria cultured in the absence of contaminating metals had only 262 mg/kg zinc within their tissues (Patrick and Loutit 1976).

TABLE 3.1-17

SUMMARY OF ACUTE AND SUBLETHAL CONCENTRATIONS OF ZINC FOR MARINE LIFE

Species	Common Name	Concentration (mg/L)	Effect(s)	Reference
ZOOPLANKTON				
Artemia salina (larvae)	Brine shrimp	0.1	*Acute*: 150-h LC_{50}	Bernhard and Zattera 1975
(adults)	Brine shrimp	1.0	*Acute*: 12-day LC_{50}	Bernhard and Zattera 1975
(adults)	Brine shrimp	0.1	*Sublethal*: Suppression of growth	Bernhard and Zattera 1975
BENTHIC INVERTEBRATES				
Clinocardium nuttalli	Clam	200	*Acute*: 48-h LC_{50}	Portman 1968, cited in ESB 1972
Hemigrapsus oregonensis	Crab	12	*Acute*: 48-h LC_{50}	Portman 1968, cited in ESB 1972
Mercenaria mercenaria (larvae)	Clam	0.195	*Acute*: 8-10 day LC_{50}	Calabrese *et al.* 1977, cited in Reish *et al*, 1978
Mya arenaria	Clam	1.55	*Acute*: 7-day LC_{50}	Eisler 1977
Penaeus aztecus	Shrimp	100	*Acute*: 48-h LC_{50}	Portman 1968 cited in ESB 1972
Penaeus duorarum	Shrimp	10	*Acute*: 48-h LC_{50}	Portman 1968 cited in ESB 1972

The effects of zinc on fish have been primarily examined in freshwater environments. Rainbow trout (Salmo gairdneri) avoided waters containing a zinc concentration of only 5.6 ppb during 50 percent of the trials conducted by Sprague (1968, cited in Taylor and Demayo 1980). In a similar manner, Atlantic salmon (Salmo salar) have been shown to avoid zinc and copper mixtures at low concentrations (Sprague 1965; Sprague and Saunders 1963, both cited in Environmental Studies Board 1972). It is unknown if this behaviour would also occur in marine environments, although background zinc levels in the Beaufort Sea approach the avoidance thresholds observed with rainbow trout, and it is likely that marine species are adapted to higher zinc concentrations than freshwater species examined to date.

Zinc is toxic to fish at relatively low concentrations. Its toxic action seems to involve interruption of gaseous exchange both at the tissue level, through inhibition of the enzyme carbonic anhydrase, and at the gills, through damage of gill epithelia (Taylor and Demayo 1980). Several studies have suggested that zinc concentrations may be physiologically regulated in some fish species (Eisler and Gardner 1973; Naidu 1974, cited in Leland et àl. 1976). Zinc toxicity to freshwater fishes appears to increase with increased pH (Mount 1966; Holcombe and Andrew 1978, both cited in Taylor and Demayo 1980), and to decrease with increased hardness (Mount 1966; Sinley et al. 1974, both cited in Taylor and Demayo 1980). A rise in water temperature and low dissolved oxygen levels both have been shown to increase zinc toxicity to the bluegill Lepomis macrochirus (Pickering 1968; Burton et al. 1972, both cited in Taylor and Demayo 1980). Atlantic salmon (Salmo salar) have also been shown to be more tolerant of zinc at lower temperatures (Hodson and Sprague 1975, cited in Taylor and Demayo 1980).

Sublethal effects of zinc on fish include reduced reproductive capacity. Female whitefish and northern pike from a mine tailings pond were either barren or had smaller and fewer eggs than fish from less contaminated areas (Falk et al. 1973, cited in Taylor and Demayo 1980). Zinc may also affect behaviour of fish. For example, the stone loach (Noemacheilus barbatulus) lost its instinct to hide during daylight when exposed to sublethal zinc concentrations (Solbe and Flook 1975, cited in Leland et al. 1976).

Freshwater fish can accumulate zinc from their environment, as shown with fish inhabiting a Northwest Territory lake affected by mine wastes. Zinc was accumulated in the liver at concentrations four to eight times as high as in muscle (Falk et al. 1973, cited in Taylor and Demayo 1980). The role of feeding habits in zinc accumulation by fish was demonstrated by DeLisle et al. (1975, cited in Taylor and Demayo 1980) with bottom and mid-water feeding aquarium fishes. Following exposure to zinc-contaminated sediments for 5 months, the mid-water feeder (Brachydanio rerio) accumulated between 2.4 and 10 times less zinc than the bottom feeders (Carassius auratus and Corydores aeneus).

As is the case of other trace metals, little information is available on the effects of zinc on marine mammals. Residual zinc concentrations were approximately equal in the liver, muscle tissue and fat of four white whales harvested in the Mackenzie Delta (Hunt 1979). This is in contrast to mercury, cadmium and copper which were found to be preferentially accumulated in the liver.

Zinc has been shown to be toxic to birds at high concentrations, with high dietary levels resulting in anemia and paralysis in domestic mallards (Gasaway and Buss 1972). Experiments with domestic mallards at 3000 to 12,000 ppm zinc (air-dried weight) resulted in loss of body weight, severe stress (demonstrated by increased adrenal and kidney size and decreased pancreas), and mortality (Gasaway and Buss 1972).

As in the case of other trace metals contained in drilling wastes, some buildup of zinc levels in sediments surrounding exploration and production platforms is expected, but dissolved concentrations in the water column will be rapidly reduced to non-hazardous levels within a few metres of offshore facilities. As a result, the impacts of this metal on marine resources of the Beaufort Sea will likely be restricted to localized sublethal effects in the immediate vicinity of drilling waste disposal, including accumulation of zinc in the tissues of sessile benthic fauna and possibly some species of fish.

3.1.10.9 Summary of Concerns Related to Trace Metals

As discussed earlier, most trace metals present in drilling wastes are not biologically available, and uptake of metals by marine organisms would only be likely if relatively large quantities of formation waters containing dissolved metals are released to offshore waters of the Beaufort Sea. It is not possible to accurately predict the concentrations of trace metals which could be present in waters surrounding production facilities as a result of formation water release, particularly since formation water chemistry will likely differ with well location and depth. In addition, the amount of trace metals remaining in solution or removed from the water column through precipitation and/or adsorption will depend on temperature, dissolved oxygen, pH, salinity, the presence of various organic and inorganic compounds, and the presence and size of suspended clay particles. Measurement of total trace metal concentrations in sediments does not give an accurate estimate of the metals which may be available to organisms, particularly with those metals which are not easily released once they are bound to the sediments.

The levels of dissolved trace metals documented in the interstitial water of drilling fluids and in formation water (Thomas 1978a,b) are generally well below the concentrations at which sublethal and lethal effects have been documented for aquatic organisms. In addition, field studies in offshore areas have shown that dilutions are in the order of 100:1 in the immediate zone of discharge of waste drilling fluids, and under normal conditions, trace metals are diluted to background concentrations within 200 m of the discharge site. McDonald (1980) suggests that serious water column effects would only occur if drilling wastes were discharged in a basin with restricted circulation.

The use of primarily freshwater organisms in trace metal toxicity and sublethal experiments hampers assessment of the possible effects of formation water and drilling waste discharge on Beaufort Sea organisms, although according to Environmental Studies Board (1972), freshwater bioassay data do provide some measure of potential acute toxicity in the marine environment. In addition, the toxic effects of trace metals on various organisms can be affected by factors such as pH, light, water temperature and hardness, food or nutrient availability and dissolved oxygen concentration. Condition of the organism and the simultaneous presence of other toxicants (which may produce synergistic and antagonistic effects) may also affect the toxicity of trace metals. A complex chemical media such as seawater may be expected to result in more synergism or antagonism than a media having few chemical compounds (Environmental Studies Board 1972). In view of the relatively high trace metal levels required to produce adverse biological effects and the dilution which has been shown to occur at a drilling waste disposal site, the discharge of formation water and drilling fluids in the Beaufort Sea is not expected to result in more than localized effects on marine organisms. However, the bioaccumulation of several trace metals may occur in areas of long-term formation water release, particularly in benthic invertebrates, which are a food source for higher trophic levels.

Newbury (1979) suggested that trace metals present in drilling fluids and formation cuttings may have adverse effects on some species of marine mammals. However, the trace metals which are most likely to result in localized effects on marine mammals are those present in dissolved form in formation or "produced" water, and then only if individuals feed or remain in areas relatively close to production platforms where this water is not being reinjected for recovery enhancement. The very limited number of studies which have examined the effects of trace metals on marine mammals indicate that at least some species can accumulate trace metals. For example, Heppleston and French (1973) found higher residual metal levels with increased age of gray seals, suggesting that this species concentrates metals from the surrounding environment or food resources. However, the physiological effects and long-term ecological significance of accumulation of trace metals by marine mammals remain poorly documented.

In the absence of evidence to the contrary, it should be assumed that marine mammals in the Beaufort Sea could indirectly (via ingestion of contaminated prey) or directly accumulate some trace metals present in drilling wastes, particularly in areas where produced water containing dissolved metals is discharged to the marine environment. However, the susceptibility of marine mammals in the Beaufort Sea to trace metal uptake would likely vary with species and area of drilling waste disposal. The degree of concern related to the accumulation of metals by bowhead and white whales is expected to be NEGLIGIBLE since these species are not year-round residents, and would only be temporarily exposed to elevated metal levels during their movement through the production region. On the other hand, seals and polar bears spend a larger proportion of the year in or near the area where exploration and production drilling would primarily occur, and may actually be attracted to these sites under some circumstances (Section 2.1.2). In addition to the greater risk of direct exposure of these species to drilling wastes, seals have also been shown to accumulate trace metals, and the feeding habits of both seals and polar bears are likely to favour indirect transfer of trace metals from contaminated food organisms in the vicinity of drilling platforms (from benthic invertebrates and fish to bearded and ringed seals, respectively, and from seals to polar bears). Arctic foxes feeding on ringed seal carrion could also accumulate trace metals. Nevertheless, the overall degree of concern regarding potential effects of trace metals in drilling wastes on bearded and ringed seals, polar bears and Arctic foxes is only considered MINOR because of the small proportion of the regional populations that would be affected.

To date, trace metals in drilling wastes have not been shown to be hazardous to birds, although this phenomenon is considered possible in habitats where dissolved metals in formation water are available for uptake by dominant prey species of bird populations found in the Beaufort Sea. Some of the trace metals present in significant amounts in drilling fluids and formation water have been shown to adversely affect and/or be accumulated by birds.

Several species of birds have been shown to accumulate trace metals as a result of their diet, although primarily in areas near sewage outfalls (Vermeer and Peakall 1979). Different feeding habits have been shown to cause differences in trace metal concentrations within two species of seaducks feeding in the same area. Vermeer and Peakall (1979) attributed differences in concentrations of some metals between surf scoters (Melanitta perspicillata) and greater scaup (Aythya marila) to the higher dependence of the scoters on mussels (Mytilus edulis), which were shown to concentrate those metals. Uptake of trace metals may also be related to intake of grit required to grind food in the gizzard. Ducks which feed on vegetation require more grit than ducks whose diet consists largely of fish or invertebrates, and are thus likely to absorb more trace metals from the grit (Vermeer and Peakall 1979).

Birds most likely to be affected by trace metals from drilling wastes released to the Beaufort Sea are those which feed on benthic invertebrates, particularly if disposal occurs in shallow areas with limited water circulation that also happen to be used as feeding habitats by birds. However, since any adverse effects would be localized and involve a small number of individuals from regional populations, the overall degree of concern related to birds and trace metals is expected to be MINOR.

Tillery and Thomas (1980) reported higher concentrations of Cr, Fe and Ni in two fish species in the Gulf of Mexico, but were unable to substantiate that the chemicals originated from a drilling operation. Neither Wohlschlag (1977) nor McDermott-Ehrlich et al. (1978, both cited in Gettleson 1980) found elevated levels of trace metals in fish collected near drilling sites in the Gulf of Mexico and off the Californian coast, respectively. Due to the rapid dilution and dispersion of drilling fluids and other drilling wastes following discharge to the marine environment, the period of contact between fish and any wastes is probably in the order of minutes (Gettleson 1980), unless the fish are actively swimming into the most concentrated portion of the turbidity plume (Houghton et al. 1980) or are attracted to the spoil deposits and associated benthic community. Consequently, most effects of trace metals on fish would be highly localized, and Houghton et al. (1980) suggested that metals in drilling wastes would result in no detectable environmental impact on pelagic fish species.

On the other hand, there may be some adverse effects if demersal fishes feed on benthic invertebrates contaminated with trace metals. Overwintering demersal fish species would be most likely to encounter elevated trace metal levels and/or contaminated food sources in poorly flushed, ice-covered nearshore waters, particularly near relatively shallow artificial islands where ice ridges may restrict the normal water circulation. If residual trace metal levels in invertebrate fauna are relatively high, additional concentration in fish tissue could result in sublethal or lethal effects, or represent a source of contamination of members of higher trophic levels, including several species of birds and marine mammals. Therefore, although the degree of concern regarding immediate effects of drilling waste trace metals on fish populations of the Beaufort Sea will likely be MINOR, the long-term effects associated with repeated feeding on contaminated invertebrates could be of MODERATE concern.

Due to rapid dilution of drilling wastes and rapid settling out of heavier components, any acute lethal or sublethal effects on phytoplankton and zooplankton would only occur in the immediate vicinity of drilling waste discharge sites and would almost certainly be insignificant in a regional context. Documented sublethal effects of trace metals include reduced photosynthesis and growth of phytoplankton, and changes in morphology, growth, reproduction, metabolism and behaviour of zooplankton. Plankton may also accumulate some trace metals, and this could contribute to a buildup of trace metals in higher trophic levels. Significant uptake of trace metals would only occur, however, if local currents and/or active swimming maintained plankton in the drilling waste plume for considerable periods of time (i.e. for at least several hours), and this is considered very unlikely. As a result of the extremely localized and short-term nature of the contact between Beaufort Sea planktonic communities and drilling wastes released from exploration and production platforms, the degree of potential concern related to trace metal uptake and toxicity is expected to be NEGLIGIBLE.

Benthic invertebrates are the group of organisms most likely to be chronically exposed to and accumulate trace metals from the sediments, and would also be the dominant pathway for the uptake of trace metals by higher trophic levels. Some classes of invertebrates such as bivalves have a high capacity for accumulation of certain trace metals (Kohler and Riisgard 1982), and may represent sites of metal entry into food webs. Newbury (1979) suggests that epibenthic invertebrates overwintering in shallow areas with restricted under-ice water circulation would be most likely to accumulate trace metals released into an Arctic environment. These nearshore benthic invertebrates are also an important food source for many of the seabirds and fish that inhabit the Beaufort Sea during the open water season (LGL and ESL 1981). McDonald (1980) predicted some very localized uptake of copper, lead and zinc by benthic fauna exposed to drilling fluids, although bioaccumulation or toxicity of trace metals from drilling wastes has not been documented in benthic infauna collected near well sites in the Beaufort Sea or elsewhere (Crippen et al. 1980; Neff 1980). As in the case of other marine resources of the Beaufort Sea, the most serious area of concern with respect to the effects of trace metals in drilling wastes would be the chronic and long-term exposure of benthic fauna to formation water released at production platforms.

Benthic invertebrates can accumulate trace metals through direct adsorption or by ingestion with food, including suspended material, sediments and other organisms (Gettleson 1980). Bacteria are also consumed in large quantities by invertebrate deposit and filter feeders, and may be a mechanism through which trace metals enter food chains. Studies have shown that bacteria concentrate metals from their environment and that these high metal concentrations are then transferred to bacterivores. For example, Loutit et al. (1973, cited in Patrick and Loutit 1976) found that bacteria below effluent entry points in a river had higher metal levels than those above, while mixed cultures of these bacteria were able to concentrate Cr, Cu, Mn, Fe, Pb and Zn. Patrick (1976, cited in Patrick and Loutit 1977) reported that the bacterium Sphaerotilus concentrated heavy metals by a factor of up to 2 percent of its cellular dry weight. The subsequent transfer of these metals to members of higher trophic levels was demonstrated in a study where bacteria grown in media with and without addition of metals were fed to tubificid worms (Patrick and Loutit 1976). Those worms ingesting bacteria grown in media containing 1 μg/mL of each metal had higher levels of Cu, Mn and Fe, and over twice the amount of Cr, Pb and Zn in their tissues than those worms fed bacteria grown in metal-free media. Lee et al. (1975, cited in Leland et al. 1976) found that epiphytic bacteria can transfer all six of these metals to the grazing periwinkle Melarapha. Mercury may also enter the food chain via bacteria. Colwell et al. (1975) reported that a Hg^{203} label on a pseudomonad bacteria species was both taken up and concentrated by a ciliate (Keronopsis sp.).

Benthic infauna could be exposed to trace metal concentrations which become chronically lethal as drilling wastes reach and accumulate within the sediments. Since drilling from production platforms is expected to continue for several years and may be followed by relatively long-term discharge of formation water, trace metal levels in sediments and in benthic animals could build up to toxic levels and result in delayed mortality. The size of the area where noticeable bioaccumulation of trace metals may occur is not known, although from a regional perspective, it is likely to be relatively small. Metal-contaminated benthic fauna could subsequently affect members of higher trophic levels that feed intensively near the drilling sites. Due to the slow growth and longevity of some benthic invertebrates in the Arctic, trace metal accumulation is expected to be a relatively localized but long-term form of impact, and as a result, the degree of potential concern would be considered MODERATE.

The overall degree of concern regarding effects of trace metals from drilling wastes on epontic organisms is expected to be MINOR because (1) most wastes containing trace metals would rapidly sink away from the lower ice surface, and (2) formation water, which would be the primary source of metal exposure to this community, will be diluted at progressive distances beyond production platforms. Localized effects could include accumulation of trace metals by some flora and fauna and indirect contamination of some birds, mammals and fish prior to spring breakup, although these impacts should not be regionally significant.

There is very limited available information on the interactions between bacteria and most of the trace metals which may be present in drilling wastes. Powerful inhibitors of enzyme action such as mercury and lead (Stanier *et al.* 1963) could adversely affect bacteria in localized areas where formation cuttings and expended drilling muds accumulate. If microorganisms are exposed to drilling wastes for a sufficient period, conditions favouring the growth of certain strains may also result in local changes in species composition (Jones 1973). Gonye and Jones (1973) suggest that low amounts of organic matter in the open ocean, in comparison with coastal and estuarine waters, results in less chelation of metal ions, and as a result, open ocean bacteria may have mechanisms to counteract metal toxicity. Consequently, addition of trace metals to offshore areas in the Beaufort Sea may have less of an effect on bacteria than in nearshore environments under the influence of the Mackenzie River.

Marine bacteria have a marked ability to tolerate trace metal ions (Gonye and Jones 1973), and are very important in the dynamic biological cycles of many toxic metals. However, the products of bacterial detoxification may be more or less toxic than the original ions to members of higher trophic levels, and there may be potential concern when the addition of toxic agents to a system upsets the equilibrium of such cycles and affects the concentration of toxic intermediates (Wood 1974). As indicated earlier, bacteria can also accumulate trace metals, and ingestion of these microorganisms by invertebrates can represent a pathway for introduction of trace metals into food chains. Nevertheless, the overall degree of concern associated with the effects of trace metals on bacteria in the Beaufort Sea is considered MINOR due to the relatively small areas which would be affected by drilling wastes and undiluted formation water.

3.1.11 Summary of Concerns Related to Drilling Wastes

Overall, the most significant area of concern related to the discharge of drilling wastes in the Beaufort Sea is the potential trace metal content of formation water, as well as possible synergistic effects associated with the simultaneous presence of trace metals and petroleum hydrocarbons in the formation water. However, it should also be emphasized that all solid, liquid and dissolved wastes released from exploration and production platforms would be rapidly diluted in the receiving environment, and therefore, most adverse impacts would be extremely localized. In addition, areas where drilling wastes are released would already be disturbed to a greater or lesser extent by borrow placement during island construction or dredging of glory holes for placement of B.O.P. stacks for drillship operations. The degree of regional concern associated with the effects of drilling wastes on specific resources of the Beaufort Sea is summarized in Table 3.1-18.

The presence of dissolved trace metals (and emulsified oil) in formation water and relatively large volumes of produced water which may be discharged from offshore facilities in the Beaufort Sea are considered a MODERATE area of potential regional concern with respect to some benthic invertebrates, fish and birds (Table 3.1-18). Both benthic epifauna and demersal fish species feeding in areas near production platforms may accumulate trace metals, although concentrations of metals measured in formation water to date in the region have been generally less than those considered hazardous. The primary reason for this degree of concern with trace metals is the potential long-term and chronic nature of the exposure which would increase the probability of significant bioaccumulation of most metals and biomagnification of others (Section 3.1.10). In addition, formation water is also expected to represent a chronic low-level source of petroleum hydrocarbon input to the Beaufort Sea, and as a result, a range of synergistic effects with trace metals are considered possible.

TABLE 3.1-18

SUMMARY OF POTENTIAL CONCERNS RELATED TO RELEASE OF DRILLING WASTES (DRILL MUDS, FORMATION CUTTINGS AND PRODUCED WATER) IN THE BEAUFORT SEA REGION

Environmental Component or Resource	Potential or Probable Effects	Degree of Potential Regional Concern
Substrate and Sediment	Accumulation of formation cuttings in piles approximately 10-100 cm high and 50 m in diameter; accumulation of fines from drilling fluids and formation cuttings 'downstream' of disposal sites and local increases in the concentrations of some trace metals	See specific resources
Water Quality	Localized increases in water turbidity and the concentrations of dissolved trace metals and various organic and inorganic constituents of drilling muds. Extremely localized and harmless increase in radio-active tracer levels surrounding drill sites	See specific resources
Bowhead and White Whale	Very localized reduction in food availability or detectability. Potential uptake of some trace metals if whales congregate in areas chronically exposed to discharged formation water	NEGLIGIBLE
Bearded and ringed seal; polar bear and Arctic fox	As above except potential direct and indirect effects of drilling wastes are more likely because of probable attraction of these species to exploration and production facilities, and their year-round residence in the region.	MINOR

TABLE 3.1-18 (Cont'd)

Environmental Component or Resource	Potential or Probable Effects	Degree of Potential Regional Concern
Birds	Localized reduction in prey availability of benthic feeders; potential uptake of trace metals through the ingestion of contaminated prey, particularly near production facilities located in shallow waters. Simultaneous presence of emulsified oil in formation water would increase degree of concern when spring migrants (oldsquaws, eiders, glaucous gulls, loons and alcids) are concentrated in leads and other open water areas	NEGLIGIBLE to MODERATE depending on species, time of year and presence of oil in formation water
Fish	Localized direct and indirect effects including possible acute toxicity of certain elements or compounds, reduced prey availability or detectability, sublethal physiological and behavioural responses and bioaccumulation of certain elements or compounds. Toxic effects of drilling muds would be unlikely due to rapid dilution of wastes, although bioaccumulation of some trace metals is possible in instances where contaminated invertebrates near produced water disposal sites are ingested	MINOR to MODERATE depending on amount of ingestion of invertebrates contaminated with trace metals
Phytoplankton	Localized reduction in photosynthesis and growth due to turbidity; significant toxic effects and uptake of trace metals unlikely because of natural transport of organisms into and out of waste disposal areas	MINOR
Zooplankton	Localized toxic effects and uptake of trace metals; reduced feeding efficiency and respiratory effects within turbidity plumes	MINOR

TABLE 3.1-18 (Cont'd)

Environmental Component or Resource	Potential or Probable Effects	Degree of Potential Regional Concern
Micro-organisms	Localized bacterial proliferation near drilling mud waste disposal sites due to organic constituents in mud formulations. Probable uptake of trace metals present in both the water column and sediment	MINOR
Benthic Communities	Direct burial of sessile benthic flora and fauna by formation cuttings and solids in drilling fluids; localized changes in physical character of benthic habitat; other effects of increased suspended solids concentrations. Uptake of dissolved trace metals may occur in areas of long-term formation water release	MINOR to MODERATE depending on degree of trace metal uptake
Epontic	Localized toxic effects if wastes discharged below ice cover; possible 'shading' and reduced spring photosynthesis if wastes discharged on ice surface, as well as exposure of organisms to toxic wastes which migrate through melt channels	MINOR

3.2 WATER/GLYCOL B.O.P. CONTROL FLUID

3.2.1 Introduction

Blowout preventer (B.O.P.) stacks installed on the sea floor (drillships) or on the surface (artificial islands) are actuated hydraulically using a fluid which may consist of various proportions of ethylene glycol, water and occasionally certain hydrocarbon additives. However, for future Beaufort operations, a fluid consisting of 50 percent ethylene glycol and 50 percent water is proposed. Regulatory requirements necessitate testing of the B.O.P. stack on an average of once every 2 weeks, and this results in the release of approximately 1 m^3 of hydraulic fluid. Although this fluid can be recovered and stored on fixed drilling platforms (artificial islands), there is no means of recovering the control fluid from subsea well heads below moored drillships and conical drilling units, and this material is released to the water column. In the Beaufort Sea, B.O.P. stacks are placed approximately 8-10 m below the seafloor in a dredged "glory hole" to prevent damage from ice keels where water depths are less than 54 m. In such locations, the relatively high specific gravity control fluid would tend to sink upon release and remain within the glory hole. Depending on local circulation patterns, all or a portion of the water/glycol solution could accumulate within the glory hole.

3.2.2 Effects of B.O.P. Control Fluid on Mammals

Although the toxicity of B.O.P. control fluid to marine mammals has not been documented, the release of this fluid from seafloor mounted stacks during required testing procedures is expected to have a NEGLIGIBLE effect on marine mammals because most species would not occur within a glory hole (Vol. 3A; Section 2.5.3). The bearded seal is probably the only species which may occur in a glory hole and be potentially affected by B.O.P. fluid or through a highly localized reduction in food availability. However, few (if any) seals are expected to forage within glory holes because (1) the composite disturbances associated with drilling activity would probably cause most seals to avoid the affected areas, and (2) benthic infauna and epifauna (food sources of bearded seals) are removed during construction of the glory hole, and recolonization may not occur while drilling is still in progress (Thomas 1978a). Loss of benthic infauna within the glory hole may last for a period exceeding 1 to 2 years after drilling is complete, but the degree of concern associated with this loss in terms of food availability for bearded seals is considered NEGLIGIBLE because of the limited area affected in a regional context.

3.2.3 Effects of B.O.P. Control Fluid on Fish

There is only limited available information on the toxicity of ethylene glycol to fish, and no studies have been conducted with arctic species. In laboratory bioassay studies, Jank *et al.* (1974) reported relatively low toxicities of ethylene glycol with rainbow trout (96-h LC_{50} = 19,000 ppm), while Hann and Jensen (1974) indicated that this compound was moderately toxic to various aquatic organisms, with 96-h TLm (equivalent to 96-h LC_{50}) values ranging from 100 to 1000 ppm. Consequently, the release of 1 m^3 of B.O.P. control fluid (50 percent ethylene glycol) approximately once every two weeks would not likely have a serious effect on fish populations except in the immediate vicinity of the B.O.P. stack within the dredged glory hole.

Dissolved oxygen concentrations could also be depressed in areas immediately adjacent to the B.O.P. stack due to metabolism of ethylene glycol by certain strains of bacteria (Section 3.2.6), and this could indirectly affect fish or reduce the rate of recolonization by benthic food sources during the period when B.O.P. fluid is routinely released. Although it is unlikely that fish (primarily demersal species) would remain in areas with low oxygen concentrations or relatively high ethylene glycol levels for sufficient periods for these conditions to become acutely toxic, some sublethal effects such as temporary narcosis or avoidance could occur in the immediate vicinity of the stack. The occurrence of these potential effects assumes that fish would be attracted to and remain in glory holes, and this is considered unlikely since these habitats would be virtually devoid of benthic fauna following dredging.

When control fluid is no longer released from B.O.P. stacks, recolonization of glory holes by benthic fauna and the physical attraction of some fish species to underwater structures could have positive but localized effects on fish. For example, videotape records of a B.O.P. stack at Orvilruk in the Beaufort Sea (Can Dive Ltd.) indicated the presence of a colonizing epibenthic community that was considerably more diverse than that observed in adjacent soft substrate areas, while the isopod Saduria spp. was unusually abundant in previously disturbed areas. Fish could be attracted to these habitats and benefit from increased food availability, although from a regional perspective, this effect would not be significant. The small size of the areas affected and the subsequent dilution of the control fluid at increasing distances from the stack suggest that any potential effects on fish will be very localized, relatively short term and of NEGLIGIBLE concern in relation to regional fish populations.

3.2.4 Effects of B.O.P. Control Fluid on Phytoplankton

There is no available information on the toxic or sublethal effects of B.O.P. fluid or ethylene glycol on phytoplankton, although concentrations between 100 ppm and 1000 ppm may be toxic to some marine organisms (Hann and Jensen 1974). However, since the density of ethylene glycol (1.109) is greater than seawater, even at a 1:1 mixture, BOP fluid would probably sink gradually to the seafloor after discharge, likely within the glory hole itself and at depths considerably deeper than where phytoplankton are abundant. B.O.P. control fluid would also be rapidly diluted in surrounding waters, further decreasing the zone within which phytoplankton could be affected. Consequently, the overall degree of concern regarding the potential effects of B.O.P. control fluid on phytoplankton in the Beaufort Sea is considered NEGLIGIBLE.

3.2.5 Effects of B.O.P. Control Fluid on Zooplankton

Very little information describing the toxicity of ethylene glycol to marine zooplankton is currently available. Price *et al.* (1974) found that the 24-h median tolerance limit of brine shrimp (*Artemia salina*) to this compound exceeded 20,000 ppm, while Portman and Wilson (1971) indicated that the 48-h and 96-h LC_{50} values of ethylene glycol with the brown shrimp (*Crangon crangon*) were 100 ppm. These data suggest that ethylene glycol may be relatively toxic to at least some crustaceans. However, due to the specific gravity of the glycol-based fluid, and the fact that it would be discharged below the mean sea floor level within a glory hole, there will be only limited opportunities for zooplankton to encounter toxic concentrations of B.O.P. fluid. Consequently, little if any zooplankton mortality is anticipated as a result of the discharge of B.O.P. fluid, and the degree of anticipated regional concern is expected to be NEGLIGIBLE.

3.2.6 Effects of B.O.P. Control Fluid on Micro-Organisms

Potential toxic effects of ethylene glycol on micro-organisms are not described in the pollution-related literature. On the other hand, a considerable body of information exists on the microbial breakdown of glycols and the ability of certain strains of bacteria, primarily those of the genus *Psuedomonas*, to use ethylene glycol as their sole carbon source (see Miller 1979, for a recent review). The widespread occurrence of bacteria capable of degrading ethylene glycol is undoubtedly related to the simple nature of the molecule and its relationship to components of various metabolic pathways. For example, tracer studies conducted by Child and Willets (1978) indicate that only a single step is required to convert ethylene glycol to glycolate, which is itself readily convertible to glyoxolate, a primary constituent of the Kreb's citric acid cycle.

The rate of degradation of glycol is temperature dependent, and ranges from approximately 2 mg/L/day at 20°C to about 0.2 mg/L/day at 4°C in freshwater (Evans and David 1974). Rates of decomposition may be significantly higher in bottom sediments containing high concentrations of bacteria. Evans and David (1974) also reported that *in vitro* metabolism of ethylene glycol in water obtained from a major watercourse and a tributary which were thought to be enriched with nutrients was initially slow, but increased after an incubation period of seven days. Although synergistic effects of ethylene glycol disposal and relatively high background nutrient levels on bacteria remain poorly documented, metabolism of ethylene glycol by bacteria in the Beaufort Sea could be highest in nearshore environments characterized by higher nutrient levels. Price et al. (1974) reported that the rate of biodegradation of ethylene glycol was about 20 to 25 percent slower in seawater than in fresh water, and that the actual chemical oxygen demand required to convert ethylene glycol to CO_2 and water was 1.29 mg O_2/mg ethylene glycol. Assuming that B.O.P. control fluid is 50 percent ethylene glycol and that these strains of bacteria are present in the Beaufort Sea, the COD of 1 m^3 of discharged fluid would require the dissolved oxygen (at 10 mg/L) in 7.15×10^4 m^3 of seawater/day. However, at a conservative current velocity of 5 cm/s, this figure represents only 0.8 percent of the water expected to pass through a glory hole in a given day. Consequently, only localized oxygen depletion would be expected due to the discharge of B.O.P. control fluid.

3.2.7 Effects of B.O.P. Control Fluid on Benthic Communities

Since B.O.P. stacks would be mounted in a previously dredged glory hole, control fluid would tend to concentrate in an area which is expected to be devoid of benthic infauna and contain only limited epifauna until significant colonization from adjacent areas occurs. If the latter community extends into areas where the B.O.P. control fluid has accumulated, some mortality and/or sublethal effects could result from either the acute toxicity or oxygen demand of ethylene glycol. However, due to dilution of this fluid in surrounding waters, potential effects on benthic communities are expected to occur in a very localized area, and the regional degree of concern would likely be NEGLIGIBLE.

3.2.8 Summary of Concerns Related to B.O.P. Control Fluid

A number of factors are expected to reduce the potential biological concerns associated with B.O.P. control fluid release from seafloor-mounted blowout preventer stacks:

1. The primary constituent, ethylene glycol, has been shown to be only moderately toxic to aquatic life.

2. Its release in limited quantities (1 m^3 twice per month) within a previously dredged glory hole should eliminate the potential for serious effects on benthic communities which might otherwise be an area of biological concern.

3. The presence of numerous bacteria in seawater capable of using ethylene glycol as their sole carbon source would facilitate its rapid degradation.

4. The fact that only a single step is required to convert ethylene glycol to glycolate (a tricarboxylic acid pathway precursor), together with the fact that exposure in micro-organisms results in increases in enzymes involved in its catabolism, demonstrate its basic compatibility with existing metabolic pathways.

However, release of B.O.P. control fluid may still result in several localized effects. Differential growth of bacteria capable of metabolizing ethylene glycol may result in changes in microfloral species composition in the vicinity of blowout preventer stacks. Within the glory hole itself, oxidation of ethylene glycol to CO_2 may result in reduced oxygen levels. Most bacterial degradation studies have been conducted at ethylene glycol concentrations ranging from 3 to 200 mg/L, and it is probable that concentrations in the glory hole following discharge may initially exceed this range. In seawater, the rates of microbial degradation will probably be somewhat slower than in freshwater, and low temperatures will also tend to reduce the degradation. As a result, the introduction of glycol twice per month may exceed its rate of biodegradation in the glory hole.

Of the potential effects of B.O.P. control fluids on marine flora and fauna, indirect effects associated with reduced oxygen concentrations would likely be the most serious area of concern. The rate at which oxygen is removed from seawater will depend on the conditions available for bacterial growth, and on the temperature. Although some depletion of oxygen can be anticipated at the bottom of the glory hole, prevailing ocean currents and diffusion should prevent noticeable reduction of oxygen concentrations in the area surrounding the drilling platforms. In a recent review of environmental contaminants, Miller (1979) also stated that "The glycols are subject to moderately rapid breakdown by both acclimated and unacclimated soil, water, and sewage micro-organisms, thus precluding persistence in the environment. There is no evidence to suggest that the glycols would bioaccumulate". Consequently, the degree of concern regarding the release of small amounts of B.O.P. control fluid is expected to be NEGLIGIBLE. Marine flora and fauna would probably be affected within a very limited area in and around the glory hole itself, primarily due to oxygen depletion rather than the toxicity of ethylene glycol. Changes in benthic community structure and use of this community by members of higher trophic levels would also likely return to normal after the ethylene glycol was no longer being released.

3.3 UNDERWATER SHOCK WAVES

3.3.1 Introduction

Seismic programs are extensively used during the exploration phase of hydrocarbon development, and have been employed in the Beaufort Sea region for the past decade. Although 'high' or detonating-type explosives may be used for managing ice, the use of high explosives for seismic surveys is not normally permitted. The principal seismic devices currently used in the Beaufort Sea by the petroleum industry are compressed air expansion ('air guns') and sleeve exploders. Air at about 200 PSI is released from air gun chambers in arrays of from 10 to 20 air guns varying in size from 164 cm^3 to 1640 cm^3 and generating a pulse with a frequency from 15 to 80 kHz. The pulse is tuned by varying the spacing, depth, pressure and size of guns in the array (Brooks 1980). Shock waves produced by air guns differ from those of explosives in that peak pressures are low and both the rise time of the shock pulse and the time-constant of the pressure decay are comparatively long (Geraci and St. Aubin 1980). A 'sleeve exploder' has a rubber cylinder that receives a charge of propane and oxygen which is ignited by an electrical spark. Although no source level measurements are available for air gun pulses in the Beaufort Sea, Fraker et al. (1981) recorded one seismic signal from a 'sleeve exploder' in August 1980. Frequencies recorded 13 km from the survey vessel ranged from 160 to 500 Hz, although higher frequencies were probably present in the received signal (the tape-recorded signal was lowpass filtered at 500 Hz). The received spectrum level at 300 Hz was between 135 and 146 dB//(1 μPa^2)/Hz at a distance of 13 km.

The proposed development plan for the Beaufort Sea indicates that explosives are unlikely to be used in ice management programs, and a variety of mechanical icebreaking techniques are presently being evaluated by the petroleum industry (R. Hoos, Dome Petroleum Ltd., pers. comm.). Nevertheless, the potential effects of underwater shock waves from high explosives are discussed in this document since their use has not been entirely discarded at this time. Shock waves produced by high explosives are compressional waves having almost instantaneous rise time to a very high peak pressure, followed by a rapid decay to ambient (or more usually below ambient) hydrostatic pressure (Hill 1978). They differ from low intensity sound waves in that peak pressures are so high that significant changes in water density occur with the passage of the wave. Large gradients of pressure, temperature and density exist from the 'front' to the 'rear' of the wave, and much of the energy of the wave is dissipated as heat (Hill 1978).

The primary concern is that shock wave reflections at an interface between tissue and an air-filled cavity (e.g. lungs, hollow viscera, and ears of a marine mammal or swim bladder of a fish) can cause tissue destruction at the interface, leading to damage or death of organisms (Geraci and St. Aubin 1980). On the other hand, shock waves generated by air guns are harmless to fish (Falk and Lawrence 1973) and would not appear to be immediately injurious to marine mammals (Geraci and St. Aubin 1980).

3.3.2 Effects of Underwater Shock Waves on Mammals

Physical damage to marine mammals is only of potential concern when 'high' explosives are used. Although the areal extent of explosive utilization would be relatively limited, the potential effects may be locally significant when large charges are used.

Yelverton et al. (1973; cited in Hill 1978) developed a formula (using fish and small terrestrial mammals) to calculate the lethal range and minimum safe distance from an underwater explosion using target (animal) depth, detonation depth and charge weight. Hill (1978) subsequently calculated that the minimum safe distance for a ringed seal at depths of 25 m or less with a 5 kg charge detonated at a depth of 5 m was approximately 360 m. However, the calculated safe range may be an underestimate if the seal is in shallow water with a rocky bottom or if the charge is detonated under thick ice. Hill (1978) also concluded that even at a distance of 60 m, marine mammals would probably be safe from gross physical damage, with the exception of potential eardrum rupture. This author's calculations predict no eardrum rupture in marine mammals when a 5 kg charge is detonated at a distance of 120 m. In addition, he concluded that marine mammals would be less vulnerable to damage from underwater shock waves than terrestrial mammals of comparable size since the former have pressure adaptations and increased protection from thick body walls. Marine mammals would be further protected by virtue of their large size. Wright and Alton (1971) report that beavers and sea otters were more resistant to instantaneous overpressure than terrestrial mammals (dogs), while pregnant sea otters were considerably more susceptible to underwater shock waves than non-pregnant individuals. As a result, a 5 kg charge may cause mortality of pups and pregnant female seals at distances of 60 m (Wright and Alton 1971; Hill 1978). Wright and Alton (1971) also report that the weight of an aquatic mammal is a factor affecting its survival after exposure to underwater shock. For example, the LD_{50} for a 4.5 kg nutria and a 36 kg sea otter are expected to be 70 and 197 N/cm^2, respectively. Consequently, the earlier data of Wright and Alton (1971) would suggest that the safe distances calculated by Hill (1978) for ringed seals are probably also adequate for the protection of larger marine mammals.

Shock waves from conventional seismic blasts (not proposed for use in the Beaufort Sea) have resulted in marine mammal displacement or mortality. Fitch and Young (1948) reported that California sea lions were killed by underwater explosions used in seismic exploration, while gray whales in the area were apparently unharmed. Although dead harbour seals and sea otters were found near the Amchitka Island nuclear detonation site in the North Pacific, pressures associated with this event were unquestionably higher than those generated during conventional seismic blasting operations (Rausch 1973, cited in Geraci and St. Aubin 1980).

The effects of on-ice seismic exploratory activity on ringed seal densities off the north coast of Alaska were studied by Burns et al. (1981) from 1975 to 1977 and in 1981. Densities of ringed seals on the fast ice were lower in areas where seismic operations had occurred than in adjacent control areas. The authors indicated that the use of seismic devices in the landfast ice zone could result in displacement of newborn pups and females to potentially less favourable habitats, but because of the localized nature of the source of disturbance, effects on the regional populations would probably be inconsequential.

In addition to the physical effects of shock waves at close range, low frequency sound pulses could be propagated for considerable distances. Depending on ambient conditions and source pressure levels, these pulses may be detected by whales and seals within about a 100 km radius (Northrop 1980), and could reduce conspecific communication distances. The effects of underwater sound on marine mammals were previously discussed in detail in Section 2.6.

Air guns and sleeve exploders which would be used for any future seismic programs in the Beaufort Sea are not expected to cause physical damage to marine mammals, but may result in short-term disturbance of some individuals. Fraker et al. (1981) observed a group of at least 7 bowhead whales in 12-13 m of water approximately 13 km from a seismic exploration vessel using 'sleeve exploders' in the southeastern Beaufort Sea. These authors reported no apparent tendency for the whales to make any net movement toward or away from the vessel. The ecological significance of any short-term disturbance of marine mammals is not known, but would presumably vary with the species and life cycle stage. For example, bowhead whales may be most vulnerable to underwater shock waves during spring migration, while white whales could be vulnerable during this migration and their period of residence in the Mackenzie Estuary. In a similar manner, the aforementioned studies of Wright and Alton (1971) and Hill (1978) suggest that ringed and bearded seals would be most seriously affected by underwater shock waves during the pupping period. Nevertheless, since most seismic work in the Beaufort Sea has been completed and any future programs would be extremely localized, potential regional concerns related to disturbance or mortality of marine mammal populations are considered NEGLIGIBLE.

3.3.3 Effects of Underwater Shock Waves on Birds

Underwater shock waves resulting from the use of air guns and sleeve exploders will not have regionally significant direct or indirect effects on diving birds since the charges are not abrupt and have limited energy. On the other hand, underwater shock waves produced by the use of explosives in ice management programs could result in localized mortality of some species of diving birds (Fitch and Young 1948), although the regional concern associated with these losses would be NEGLIGIBLE.

3.3.4 Effects of Underwater Shock Waves on Fish

The effects of non-explosive seismic devices (air guns) on fish were examined by Weaver and Weinhold (1972) and Falk and Lawrence (1973). The former authors found that firing of 129 to 258 cc air guns at various depths and distances had no harmful effects on caged yearling salmon (genus *Oncorhynchus*), while Falk and Lawrence (1973) indicated that the lethal range of a 7 cc air gun was only 0.6-1.5 m with caged coregonids (whitefish). As a result, the use of air guns and sleeve exploders during the limited future seismic programs is therefore of NEGLIGIBLE concern with respect to regional fish populations, since the numbers of affected fish would be small and significant adverse impacts unlikely.

The use of explosives around artificial structures and for ice management is of greater concern, although as indicated earlier, various mechanical methods of ice management are currently under investigation. Shock and explosions in the aquatic environment result in a high initial release and outward acceleration which fish perceive as a pressure increase (Falk and Lawrence 1973). A substantial portion of the total force released to an aquatic medium by an explosion is retained and temporarily stored as kinetic energy. This is later radiated as a shock wave travelling with finite velocity (Falk and Lawrence 1973).

There have been numerous observations of fish injury and mortality following underwater explosions. Damage may be severe and include muscle tissue lesions or rupture of the abdominal cavity, as well as damage to the kidney, liver, heart, spleen, gonads and swimbladder (Falk and Lawrence 1973). Fish possessing open swimbladders are not as sensitive to pressure fluctuations as those species with closed systems since the former group can compensate for changes in hydrostatic pressure. In general, pelagic and mid-water species possess swimbladders, whereas demersal species do not.

Rasmussen (1967) found that the effects of pressure on juvenile fishes was species and age-specific, and also related to the presence and functional morphology of swimbladders. Newly hatched herring and salmon fry were not affected by pressure since they lack a swimbladder on emergence. However, 3 to 6 months after emergence, these species died within 24 h at pressures exceeding 2.7 psi. Post et al. (1974) examined the subgravel survival of rainbow trout eggs (genus Salmo) exposed to physical shocks at various stages of development, and failed to demonstrate a significant difference between non-shocked control and experimental groups.

In a study of more short-term pressure increases, Alpin (1947) found no apparent relationship between water depth and charge size in the biomass of fish killed by underwater detonation of 60 percent Petrogel. On the other hand, Fitch and Young (1948) reported that body shape was a factor affecting the resistance of fish to shock pressure. Fishes with thickwalled swimbladders and cylindrical bodies were more resistant to underwater shock than species which were laterally compressed and had thinwalled swimbladders.

Several studies have also demonstrated spatial differences in the effects of underwater shock waves on fish, both in terms of distance from the shock source and depth in the water column. Coker and Hollis (1950) reported that 114 to 546 kg explosives were lethal to fish within a radius of 100 to 200 m. In addition, Hubbs and Rechnitzer (1952) found that the effects of underwater explosions were intensified at the water surface where positive pressure waves are reflected, since fish are equally susceptible to negative pressure pulses. These authors also noted marked variation in the lethal range of underwater shock waves depending on the shape and nature of the ocean floor. Kearns and Bayd (1965) and Paterson and Turner (1968) recorded extensive mortality of both pelagic and demersal fishes following exposure to underwater shock. The former authors and later Rasmussen (1967) concluded that the lethal effects of shock waves were directly proportional to the quantity of explosive used and increased with the depth of detonation. In shallow waters, the lateral lethal range was greater than in deeper waters. Rasmussen (1967) also reported that burying charges in the sea floor reduced the lethal range of resultant shock waves, while charges located above a solid stratum increased the lethal range by upward reflection of pressure waves.

The overall degree of concern regarding the effects of high explosive-type underwater shock waves on fish is considered MINOR because only localized mortality of fish would be expected in the event that explosives were used during future development in the Beaufort Sea.

3.3.5 Summary of Concerns Related to Underwater Shock Waves

Two potential sources of underwater shock waves during continued development of petroleum resources in the Beaufort Sea are seismic programs which utilize air guns and sleeve exploders, and ice management programs which utilize conventional high explosives. The types of underwater shock produced by these programs would be considerably different in terms of the rise time and peak pressure of shock pulses. The available information on the air guns and sleeve exploders used in seismic programs suggests that underwater shock waves produced by these devices is of NEGLIGIBLE concern for all marine resources of the Beaufort Sea, particularly since only limited seismic work will be required during the future. Underwater shock waves generated by seismic devices may result in localized disturbance of marine mammals and fish, although mortality of individuals from either group is considered extremely unlikely.

Shock waves produced by high explosives (if used) are of potentially greater concern since they have been shown to cause considerable tissue damage and mortality in marine mammals and fish. Various investigations conducted with seals and fish suggest that the lethal range of shock waves produced by detonation of high explosives would vary with the amount of explosives used, the location, water depth and depth of detonation, and the species (including life history stage) present in the area. Mortality of marine mammals and fish would likely be greatest if explosives were detonated in shallow waters, although studies completed to date suggest that mortality of most species and life history stages is unlikely beyond a distance of 200 m. Consequently, the potential effects of high explosives on marine mammal, fish and diving bird populations in the Beaufort Sea would be relatively localized. In the event that explosives are used during ice management programs, the degree of regional concern with respect to potential impacts on marine mammals and fish would be considered MINOR, while the degree of concern with diving birds is expected to be NEGLIGIBLE.

3.4 CEMENT SLURRY AND CEMENT POWDER

3.4.1 Introduction

Cement is used during drilling operations to grout the upper casings to the riser at platform and floating drill rigs. In the case of floating rigs, approximately 25 m^3 of cement are lost from each well during normal grouting operations, while no cement enters the water column when this grouting is completed on artificial islands. In addition, past industry experience suggests that up to 150 m^3 of water-damaged cement or cement-contaminated barite may be discharged into the water column or dumped on the ice 3 or 4 times a year. Several factors are likely to influence the degree to which the discharged cement could affect the surrounding waters: (1) whether the cement is in the form of a mixed slurry and therefore discharged at the seabottom, or a powder which enters the water column at the surface; (2) the depth of water column, which will partly determine the degree of breakup and subsequent dispersal of particulate cement; and (3) the prevailing oceanographic conditions, which will affect the rate at which both dissolved and particulate components are dispersed.

Physical and chemical effects of cement which may have subsequent localized impacts on marine flora and fauna may include: (1) release of toxic compounds to the water column prior to hardening of the cement; (2) increases in the pH of sediments and water near discharge sites; (3) localized increases in water turbidity; and (4) loss of benthic habitat in areas where cement slurries harden (BOP stacks within dredged glory holes) or cement powder settles following release to the water column.

3.4.2 Components and Characteristics of Cement

Portland cements are comprised of four main components: tricalcium silicate, dicalcium silicate, tricalcium aluminate, and a tricalcium aluminoferrite. The composition of cement ranges from 61 to 65 percent lime (CaO), 20 to 25 percent silica and 3 to 6 percent alumina. In addition, small quantities of magnesia, alkalies, and sulphur trioxides are typically present. Cements used for drilling applications are usually modified to withstand high temperatures and pressures, and consist of portland or pozzolanic cements containing various organic retardants to delay the onset of hardening. Pozzolanic cements are mixtures of portland cement and silica-containing materials. Submicroscopic crystals form in the cement during hydration, and this results in a hardening of the mixture over a period from 40 min to 10-12 h, depending on the formulation. Hydration of cement is an exothermic reaction and releases a small quantity of lime.

3.4.3 Effects of Cement on Fish

The discharge of powdered cement or cement slurry could affect fish populations in the Beaufort Sea through short-term and localized increases in water pH and turbidity, exposure to toxic compounds, and localized but relatively long-term changes in substrate character. Increases in water turbidity would only result when powdered cement was released to the water column, and subsequent effects on fish would be similar to those associated with dredging operations (Section 2.4.5), but even more localized and temporary. In a similar manner, any increases in water pH caused by the dissolution of alkaline compounds in either cement slurry or powdered cement would have only localized effects on fish until the material was diluted and buffered in the receiving environment. However, cement can have serious effects on fish populations in areas where rapid dilution does not occur. For example, the U.S. Dept. of Interior (1968) reported complete mortality of downstream fish populations following a concrete spill in a freshwater environment, while calcium oxide and calcium hydroxide which are primary constituents of cement have been shown to be moderately toxic to a range of aquatic fauna, with 96-h LC_{50} values of 100 to 1000 ppm and 10 to 1000 ppm, respectively (Hann and Jensen 1974). Consequently, accidental spills of cement powder in sheltered coastal waters of the Beaufort Sea (e.g. at shorebases) would be of greater potential concern with respect to local fish populations than discharge of either cement slurry or powdered cement in offshore waters. Nevertheless, since any effects of cement on fish populations would be localized, the degree of potential regional concern regarding offshore and coastal fish resources would likely be NEGLIGIBLE and MINOR, respectively.

3.4.4 Effects of Cement on Phytoplankton

The release of cement slurry during the grouting of upper casings to the riser at floating exploration platforms would not be expected to significantly affect phytoplankton populations since this discharge would occur at water depths considerably greater than where these flora are abundant. On the other hand, release of powdered cement or cement-contaminated barite at the surface may affect phytoplankton populations through localized increases in water turbidity and pH, and the presence of toxic compounds in cement formulations. The impacts of increased turbidity on phytoplankton would mainly be the result of a reduction in available light and similar to the effects of dredge-created turbidity plumes. However, unlike the latter type of disturbance, reductions in light intensity associated with cement disposal would be extremely short-term and therefore of NEGLIGIBLE concern with respect to the productivity of regional or local phytoplankton populations.

There are no available data on the toxicity of calcium oxide (lime) or calcium hydroxide to phytoplankton, although the ranges of toxic concentrations reported for aquatic organisms (Section 3.4.3) are probably representative. As in the case of fish, toxic effects of certain constituents in cement would be localized but potentially of greatest concern in some sheltered coastal environments where these compounds are not as rapidly diluted in surrounding waters. Discharge of powdered cement is also expected to cause a slight increase in the pH of water near the site of release, although the effects of this change in water quality on phytoplankton are also expected to be localized due to the high buffering capacity of seawater. Overall, the degree of concern regarding potential adverse impacts of cement on regional phytoplankton populations of the Beaufort Sea is considered NEGLIGIBLE.

3.4.5 Effects of Cement on Zooplankton

The toxicity of cement slurries, suspended cement particles or dissolved cement components to zooplankton has not been documented, although some temporary adverse effects may be expected as a result of the increased pH in the immediate vicinity of the cement discharge. As in the case of phytoplankton, most effects of cement on zooplankton would occur when powdered wastes are released at the surface rather than when slurries are discharged within relatively deep glory holes. Dispersion of cement particles in surface waters may create a zone of increased turbidity, and result in effects on the zooplankton similar to those described for dredging (Section 2.4.7). However, the release of cement powder or cement-contaminated barite would be an intermittent phenomenon of relatively short duration, and would not be expected to have regionally significant effects on zooplankton populations. Consequently, the degree of potential regional concern regarding this resource-disturbance interaction is considered NEGLIGIBLE.

3.4.6 Effects of Cement on Benthic Communities

As indicated earlier, the routine disposal or accidental release of cement slurry or powder may increase local turbidity and suspended solid levels, alter the particle size of seabottom sediments and/or harden on the substrate, and increase water and/or sediment pH. The magnitude and duration of these physical and chemical effects have not been examined, nor have the effects of cement *per se* on benthic flora and fauna been investigated. However, the general type of physical and chemical perturbations which may be associated with cement disposal in the marine environment have been examined and this provides a sufficient basis for assessment of the potential degree of concern regarding impacts of slurries and cement powder on benthic populations.

With the exception of minor substrate loss resulting from hardening of cement slurry within glory holes and/or flocculation of powdered cement, virtually all physical and chemical effects of cement are likely to be extremely localized and short-term since these materials would be rapidly diluted in the surrounding seawater. For example, the buffering capacity of sea water would rapidly minimize any chemical effects related to the alkalinity of cement. In a similar manner, any increases in water turbidity and suspended solid concentrations will be generally insignificant in waters such as the Beaufort Sea which may be naturally turbid.

3.4.6.1 Benthic Flora

The release of cement slurry is unlikely to affect benthic flora in the Beaufort Sea since slurries are only used for grouting the upper casings to the marine riser at floating drill rigs (conventional drillships and conical drilling units). These operations would only take place at water depths considerably greater (> 25 m) than where benthic microalgae are likely to be abundant, particularly in view of the fact that this discharge would occur in a recently dredged glory hole. However, the disposal of powdered cement or cement-contaminated barite could affect benthic microalgae in some shallower coastal waters through (1) direct smothering, (2) modification of substrate character, and (3) reduced light intensities associated with temporary turbidity plumes.

The release of cement would initially smother existing flora within the areas where cement particles settle, although re-establishment of these populations would be expected to occur either directly on the cement surface or on Mackenzie River borne sediments which would likely bury cement in many nearshore habitats. The deposition and subsequent solidification of cement powder could produce a hard substrate in areas where fine sediments normally occur, and these habitats may be colonized by epiphytic flora. However, the amount of habitat modified due to cement deposition is not expected to be significant in relation to the total available habitat for benthic flora in the Beaufort Sea.

The effect of cement powder on water transparency would depend on the natural turbidity levels in areas where cement was released. Grainger (1975) suggested that light limits phytoplankton production within the Mackenzie River plume. Consequently, it is likely that cement powder would have no significant effect on the primary productivity of benthic flora within the Mackenzie River plume. The influence of cement powder on primary productivity of benthic flora outside the plume would depend on the degree of light reduction and the size of the affected area, although the duration and spatial extent of any light-related effects would be considerably less than those which may be associated with dredging operations (Section 2.4.8.3). In view of the small amount of habitat that is likely to be modified by cement deposition, the localized nature of any direct mortality of flora through smothering, and the expected localized and temporary effects on the light environment, the overall degree of concern regarding potential impacts of cement release on benthic flora is considered NEGLIGIBLE.

3.4.6.2 Benthic Infauna

Cement slurry released from floating drill rigs during grouting operations will not affect benthic infauna since this material would all settle in areas which will be devoid of fauna (i.e. recently dredged glory holes). However, cement powder released to the marine environment in both coastal and offshore areas will likely affect benthic infaunal communities by smothering existing infauna and through alteration of the physical characteristics of the substrate. Filling of interstices within the substrate by cement powder would cause suffocation of most infauna in areas where cement deposition is extensive, although these areas would be eventually recolonized once the cement was sufficiently buried by sediments transported into the region by the Mackenzie River. Changes in sediment pH or toxic effects of some cement constituents could also result in localized mortality of some infaunal invertebrates. Nevertheless, since the areas affected would be small in a regional context, the degree of potential concern regarding adverse effects of cement release on benthic infauna is also expected to be NEGLIGIBLE.

3.4.6.3 Benthic Epifauna

Unlike the situation with benthic infauna, both cement slurry and powder could have localized impacts on epifaunal invertebrate species in the Beaufort Sea, since some epibenthic species have been found to rapidly recolonize disturbed areas such as glory holes where slurries will be released. During the period between dredging of glory holes and grouting operations, some sessile epifauna such as the sand-dwelling anemone *Cerianthus* and a larger number of motile epifauna including amphipods, isopods (particularly *Saduria entomon*) and mysids may be expected to colonize dredged habitats. The release of cement slurry during the grouting operations will probably smother sessile fauna in an area extending 2 to 5 m from B.O.P. stacks, while the more mobile amphipod and isopod species may be able to avoid the advancing slurry. Epifauna in areas adjacent to the cement may also be affected by increased pH associated with the CaO and CaOH, although these effects would be both localized and short term (until slurries harden). Nevertheless, since the same habitats affected by cement would be exposed to B.O.P. control fluid released each day during exploratory drilling, complete recolonization by both infauna and epifauna would not likely occur until the sites are abandoned.

Epifauna in areas other than glory holes may also be affected by the occasional release of cement powder and cement-contaminated barite. Potential effects would be similar to those discussed for cement slurries, with most mortality being associated with the direct suffocation of sessile forms. However, these habitats would be expected to recover more rapidly since the powdered cement would usually be present as a discontinuous layer. In addition, the solidification of cement may provide a hard substrate in areas otherwise characterized by silt or sand substrates, and this could eventually result in the establishment of a more diverse epibenthic community. Nevertheless, the degree of potential concern regarding positive or negative impacts of cement disposal on regional epibenthic invertebrate communities is expected to be NEGLIGIBLE to MINOR, with a MINOR degree of concern occurring in some nearshore habitats where epifauna are important in the diet of anadromous fish, marine mammals and birds.

3.4.7. Summary of Concerns Related to Cement Slurry and Powder

The degree of potential regional concern regarding impacts of cement slurries and powder on most marine resources of the Beaufort Sea is expected to be NEGLIGIBLE due to the small amount of habitat likely to be affected and the short-term nature of most potential effects (habitat loss, pH increases and turbidity plumes). However, a MINOR degree of concern would be expected if powdered cement were accidentally or purposely discharged to the marine environment in some nearshore habitats supporting abundant epibenthic invertebrate and fish populations, particularly in waters with poor circulation where lethal effects associated with pH changes could be more pronounced.

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4. DISTURBANCES AND WASTES ASSOCIATED WITH PRODUCTION PROCESSES AND THE STORAGE AND TRANSPORTATION OF PETROLEUM HYDROCARBONS

4.1 GAS FLARES

4.1.1 Introduction

During the initial years of oil production in the Beaufort Sea, all produced gas from each field will be flared. The quantity of flared gas would increase as more wells within a field are brought into production. Subsea pipelines would be utilized to move produced hydrocarbons to a single production platform within a field, where a processing facility would separate and then flare the gas. Given the expected schedule of production platform construction and production drilling (EIS Volume 2), a maximum of 2 or 3 flares would likely be in operation at any one time. Gas may be reinjected to enhance recovery from the reservoir after each field has been in production for two or more years. A flare tip design which promotes complete, continuous combustion, minimizes radiant heat reaching the ground and reduces noise levels would be employed to reduce gas flare emissions (EIS Volume 2). Operators in the Beaufort region are also currently examining a specialized flare which includes a 10 m high stainless steel sheath (R. Hoos, Dome Petroleum Ltd., pers. comm.). In the event that this type of flare is used on production facilities in the Beaufort, the flame would not be visible and most of the potential effects of this activity on mammals and birds described below would not occur. In addition, heat generated by flares at tanker loading facilities may also be used in ice management programs (R. Hoos, Dome Petroleum Ltd., pers. comm.).

Sulphur dioxide is the primary pollutant which may be of potential concern when sour gas (containing H_2S) is flared, although Beaufort Sea gas tested to date (EIS Volume 2) has contained no hydrogen sulphide (sweet). In addition, complete combustion will be promoted by the proposed flare design, and only small quantities of particulates, smoke or hydrocarbons should be emitted. At the same time, if gas temperatures do not exceed 1200°F (650°C), large quantities of nitrogen oxides will not be formed (U.S. Environmental Protection Agency 1977). Gas flares would also be a source of heat, noise and light, although the proposed flare tip would minimize both heat and noise emissions. Nevertheless, ambient temperatures will be increased in a sphere surrounding the flare, particularly during periods when winds are calm. The size of the area affected by increased heat from gas flares at different rates of gas production or under other wind conditions is not documented, although the proposed location of flares on production platforms would prevent any significant melting of ice, heat damage at the platform surface or hazards to personnel (EIS Volume 2). As indicated earlier, a portion of the heat produced by flares could also be used for ice management programs at tanker loading facilities during the winter.

During cold periods, the formation of ice fog will be unavoidable due to the high proportion of warm water vapour in the emissions. Although ice fog is unlikely to have any direct effects on the marine environment, it may be a local visual obstruction to some birds and mammals.

4.1.2 Effects of Gas Flares on Marine and Marine-Associated Mammals

The flaring of produced gas from offshore production platforms and/or exploratory vessels is expected to have NEGLIGIBLE effects on regional populations of marine mammals in the Beaufort Sea. The only concern related to gas flares is the attraction of some species to the bright light (or heat) of the flare during periods of darkness or low visibility, thereby increasing the potential for disturbance by other activities at the site (e.g. airborne noise, human presence). Winter residents of the offshore region which may be attracted to offshore flares include polar bears, Arctic foxes, ringed seals and bearded seals. The potential for attraction of the latter two species is considered relatively remote because of other sources of disturbance at production platforms. The attraction of polar bears and Arctic foxes to unenclosed gas flares *per se* would be largely indistinguishable from the attraction of these species to any site of human or industrial activity. The numbers of bears and foxes which may be attracted to the 2 or 3 offshore gas flares which may be operating at any given time is unknown, although it would likely be an insignificant proportion of the regional populations. Therefore, the degree of regional concern regarding the effects of gas flares on these species is expected to be NEGLIGIBLE.

4.1.2 Effects of Gas Flares on Birds

The unenclosed flaring of gas from offshore production platforms and exploratory vessels in the Beaufort Sea may result in the attraction of certain bird species, although most birds would probably avoid the heat sphere associated with the flare if they approached these facilities. In general, birds do not approach gas flares during periods of daylight or clear weather, but may be attracted on foggy or rainy nights (Bourne 1979; Hope-Jones 1980). Gas flares are least likely to attract birds during spring migration since most migrants travel over the Beaufort Sea during periods of virtually continuous daylight. Attraction to flares is more probable during late summer and autumn, although the routes and numbers of birds which migrate offshore during fall over the Beaufort Sea are not well documented. For example, female oldsquaws and young-of-the-year migrate from the region until late September, and female and young common and king eiders have been recorded until late October (Gabrielson and Lincoln 1959). Light will be emitted from many other sources on the drilling platform, and the attraction to flares *per se* may be indistinguishable from the attraction to the composite sources of light in production fields.

Birds that migrate at low altitudes during periods of darkness through offshore areas are most likely to be affected by gas flares. Loons and flocks of oldsquaw and eiders are known to migrate at relatively low altitudes through the offshore Beaufort Sea during spring and/or fall migration (LGL and ESL 1981). These species are also relatively unmaneuverable during flight, and some individuals may be unable to avoid gas flares. Other birds in the Beaufort region that could also be affected by gas flares as a result of their migration routes and altitudes include thick-billed and common murres and black guillemots since these species also fly at low altitudes and are relatively unmaneuverable. However, the alcid species are not as abundant in offshore areas as the aforementioned species and do not typically flock; consequently, the potential that large numbers of individuals would be affected by flares is lower. In addition, some mortality of certain seabirds that soar in updrafts (e.g. gulls, fulmars, jaegers and terns) is possible if birds attempt to soar in the warm rising air above gas flares. However, these species are not known to concentrate during offshore foraging and migration, are relatively agile, and are therefore not likely to collide with flares.

It is difficult to accurately predict the effects that gas flares may have on birds in the Beaufort Sea region, although bird mortality (starlings and thrushes) has been documented in flares at North Sea production facilities (Bourne 1979; Bourne et al. 1979). On the other hand, Hope-Jones (1980) did not observe any bird mortality in a platform flare within the same region during a 5-week observation period. Migrating birds attracted to light at flares are known to circle and occasionally land on drilling platforms (Bourne 1979; Hope-Jones 1980). Any subsequent mortality usually results from predation by other bird species or exhaustion, and to a lesser extent through starvation (Bourne 1979). Bourne (1979) and Hope-Jones (1980) suggest that birds attracted to platforms are already disoriented and/or exhausted and are unlikely to survive regardless of the presence of offshore structures. Both of these authors also concluded that the number of migratory birds killed by gas flares in the North Sea is insignificant in relation to the total number of birds moving through the area. Consequently, the overall degree of potential concern regarding the effects of gas flares on birds in the Beaufort Sea region is expected to be MINOR, particularly since a maximum of only 2 or 3 (unenclosed) flares would be operational at any one time and the number of birds attracted to these sites would likely be small. The degree of concern regarding the effects of enclosed gas flares on birds in the Beaufort Sea is NEGLIGIBLE.

4.1.4 Summary of Concerns Related to Gas Flares

Enclosed gas flares would probably have NEGLIGIBLE effects on all resources in the region. The most significant potential concern with respect to the effects of unenclosed gas flares in the Beaufort Sea is mortality of migrant birds following their attraction to the heat/light sources. Loons, oldsquaws and eiders are the primary species that could be affected since: (1) they may migrate during periods of darkness; (2) their routes are offshore through the production zone; (3) they fly at relatively low altitudes; (4) they are relatively unmaneuverable during flight, and (5) they may occur in large numbers. Thick-billed and common murres and black guillemots may also be affected by gas flares, but to a lesser extent because these species are not abundant in the production zone and do not occur in large numbers. However, the overall degree of potential concern regarding mortality of birds in gas flares is expected to be MINOR, based on observations at offshore rigs elsewhere in the world and the fact that a maximum of only 2 or 3 unenclosed flares would occur in the production zone at any one time. The degree of potential concern regarding the effects of flares on marine and marine-associated mammals is considered NEGLIGIBLE, and would be generally limited to the attraction of polar bears and occasionally Arctic foxes.

4.2 RELEASE OF HEATED WATER

4.2.1 Introduction

Heated water will be released to the Beaufort Sea during several development activities proposed by the petroleum industry. The major sources of these discharges are formation ("produced") water, drill rig machinery, ship engine manifolds, and possibly gas liquefaction processes. Discharges will vary in quantity, temperature and timing. In protected areas, such as the interior harbour of a tanker loading terminal, heated water may be intentionally discharged to delay the formation and reduce the thickness of ice as part of ice management programs.

Excess formation water, at temperatures ranging from 10° to 55°C, which is not required for recovery enhancement in production fields may be released at rates up to 100,000 barrels/day (approximately 16,000 m^3/day) per platform, and at potentially higher rates under abnormal conditions. The discharged formation water may contain concentrations of dispersed crude oil up to 50 ppm, as permitted by the Canadian Oil and Gas Productivity Regulations. Analyses of formation water flows in the Beaufort Sea have indicated that concentrations of most dissolved metals are within the range of levels normally observed in both the Beaufort Sea and coastal waters elsewhere in the world (Thomas 1978). However, concentrations of chromium, lead, mercury and iron were higher than background levels. The potential biological effects of the trace metals and oil are separately discussed in Sections 3.1.10 and 5.2, respectively.

Produced water is a potential source of heated water for below-ice discharge in ice management programs. Calculations indicate that the daily discharge of 10^5 bbl of formation water at a temperature of 20°C above ambient will add approximately 3 X 10" cal of heat energy per day to the local marine environment. The depth of discharge will largely determine the spatial extent of the thermal plume since colder receiving water will dissipate heat more quickly. Beaufort Sea bottom waters at depths of 20 m or greater are generally isothermal (-1.5°C) throughout the year (Giovando and Herlinveaux 1981). The discharge of formation water into these relatively deep waters will probably only slightly alter the local stratification pattern. However, discharge of produced water into the surface layer above the summer pycnocline would tend to increase the spatial extent of the thermal plume since the cooling capacity would be much less during this season. This formation water is also expected to be saline and probably anoxic (OCSEAP 1979).

Shallow discharges just below the ice surface at an approximate depth of 2 m are expected during ice management programs from October to May (EIS Volume 2). Localized areas of ice fog may develop when air temperatures are less than -30°C, particularly in the fall and when ice formation is delayed by heated water discharges. When thermal discharges are used for ice management, the effluent temperature is expected to be in the range of 4° to 8°C above ambient (EIS Volume 2).

A second source of heated water which may be released to the marine environment is cooling water from generators and other platform machinery. Approximately 10^4 m^3 of heated seawater at temperatures from 10° to 55°C may be discharged per day from each drilling platform as a result of various cooling functions (EIS Volume 2). The third source of heated water discharge is from ship engine manifolds; this water is discharged to the marine environment at an approximate temperature of 15°C (EIS Volume 2). However, heated water discharges from marine vessels are not expected to be of sufficient volume to measurably affect the local temperature of the Beaufort Sea. Heated water discharged from ships in harbours is likely to remain close to the surface, especially under ice cover, while the turbulence of a moving ship will rapidly disperse warm discharges into surrounding colder waters.

4.2.2 General Effects of Temperature Changes on Aquatic Ecosystems

The proliferation of coal and nuclear-powered electric generating plants, with their attendant need for large volumes of cooling water, has resulted in an extensive data base dealing with thermal effects in temperate environments. The following discussion is based, in large part, on a recent literature review by Talmage and Coutant (1978), who examined the effects of thermal discharges on both freshwater and marine communities. It must be stressed, however, that there is currently very little available information regarding the effects of temperature on arctic organisms.

Temperature is a major factor affecting the physiology and behaviour of aquatic organisms, and not only regulates the development, growth and metabolic rates of a wide variety of organisms, but also plays a major role in determining the biochemical structure of cellular membranes and critical regulatory enzymes in both photosynthetic and respiratory pathways. The effects of temperature on key membrane and enzyme-linked pathways are thought to be major factors limiting the geographical distribution of many species. For these reasons, alterations in temperature regimes are considered an important area of potential concern with respect to offshore oil production in the Beaufort Sea.

Most organisms can acclimate to temperatures higher or lower than normal when these changes occur gradually, although in most cases, acclimation only occurs within the normal seasonal temperature range found in the natural habitat of an organism. In general, it appears that the rate of temperature change is more important to survival than the overall magnitude of temperature change. Rapid temperature increases do not allow sufficient time for physiological adjustment and often result in mortality.

Reproduction tends to occur within narrower temperature ranges than other activities. For example, Brett (1970) reported that spawning of marine fish occurs in about one-quarter to one-third of the range of temperatures over which fish can survive. In addition, reproduction is often triggered by attainment of specific environmental temperatures, while survival of eggs and young is greatly influenced by temperature since these early life history stages have narrower tolerance ranges than adults. A natural periodicity in temperature (diel or seasonal) is also required for successful reproduction of some marine species such as the oyster *Crassostrea virginica* (Butler 1965).

In general, rates of metabolism and activity of poikilotherms increase with temperature to a point near the upper limit of temperature tolerance, and above this limit, they rapidly decrease (Kinne 1970a). Depending on the physiological state of a poikilotherm, oxygen consumption, feeding rate and other metabolic processes normally increase with increases in water temperature. Growth rates of aquatic organisms are also affected by temperature. Growth of many temperate organisms stops or is much slower in the winter, while many crustaceans do not moult in winter (Kinne 1970b).

Temperature has pronounced effects on the distribution and behaviour of organisms, limiting the broad geographical distribution of most organisms through its effect on spawning, reproduction, larval stages, growth and activity. In northern aquatic ecosystems, existing thermal plumes commonly attract fish during the winter and these individuals become acclimated to the warmer water. Mortality can occur if the heated water discharge is interrupted or discontinued (Talmage and Coutant 1978), exemplifying the importance of a period of acclimation as well as the effects of unnaturally high temperatures on the microdistribution of organisms.

4.2.3 Temperature Tolerances of Arctic Organisms

Ecosystems which naturally fluctuate within wide temperature ranges are believed to be less sensitive to thermal pollution since flora and fauna in these environments are adapted to survival over a range of temperatures (eurythermal). Organisms in the temperate zone are generally thought to be the least sensitive to temperature changes, while tropical organisms and those living at great depths where temperature is virtually constant are most sensitive to changes in temperature (stenothermal).

Arctic marine environments are characterized by different seasonal temperature regimes. The water temperature in the Beaufort Sea at depths between 20 and 120m is nearly constant at approximately -1.5°C throughout the year (EIS Vol. 3A; Section 1.3). Temperatures in offshore surface waters range from -1.5°C in winter to 9°C during the summer. However, certain coastal areas such as shallow lagoons can become hypersaline in winter and may have an annual temperature range from -3° to 12°C. The southern Beaufort Sea is influenced year round by freshwater input from the Mackenzie River. Winter surface temperatures range from -1.5° to 1.5°C, while summer maxima can reach

17°C off the Delta. As a result, arctic waters contain some organisms that have adapted to a wide thermal regime and others which inhabit environments with very constant temperatures. During the summer when estuaries such as Mackenzie Bay become stratified, eurythermal and stenothermal organisms may be vertically separated by a distance of only 10m.

The discharge of large volumes of heated water may have significant, but relatively localized, impacts on the flora and fauna surrounding offshore platforms in the Beaufort Sea. The magnitude and duration of these impacts will depend on a number of factors including: (1) the volumes, depths and temporal pattern of heated water discharge; (2) the temperature differential between ambient and discharge waters; (3) the prevailing oceanographic conditions, particularily currents, vertical stratification, and ice during the period of discharge; and, (4) the specific temperature tolerances of indigenous flora and fauna. The following sections summarize available information regarding the effects of temperature on various marine resources, as well as the potential degree of regional concern which may be associated with discharge of heated water in the Beaufort Sea.

4.2.4 Effects of Heated Water on Mammals

Heated water discharge into areas of open water in the Beaufort Sea is expected to have insignificant effects on regional marine mammal populations since the heat plumes would be highly localized and mammals would be able to avoid intolerable temperatures in most if not all cases. The only potential area of concern is the attraction of seals to the limited areas of open water caused by thermal discharges under ice. Although the probability of this attraction is remote because of other sources of disturbance at offshore structures, individuals that do move into these open waters may be more susceptible to adverse effects of fuel spills, airborne or waterborne noise and other disturbances. Polar bears may also be attracted to offshore structures, although the attraction to open water created by heated water discharge would probably be indistinguishable from the overall attraction to composite activities at these sites. Consequently, the degree of potential concern regarding the direct effects of thermal discharges on marine or marine-associated mammals is considered NEGLIGIBLE.

4.2.5 Effects of Heated Water on Birds

The only potential concern related to the effects of heated water discharge on birds would occur during spring when oldsquaws, eiders, loons and glaucous gulls migrate offshore and stage at the edge of the landfast ice (Barry *et al.* 1981). Some individuals may be attracted to the open water areas created by the under-ice discharges and would therefore be more susceptible to effects of fuel spills, artificial structures, airborne noise, flares and logistics traffic at these sites. However, since open water areas resulting from discharge of heated water would be extremely small in a regional context and the majority of these species would likely avoid sites of industrial activity, the overall degree of potential concern related to heated water discharge is expected to be NEGLIGIBLE.

4.2.6 Effects of Heated Water on Fish

The role of temperature in the physiology and behaviour of fish and the effects of thermal discharges on a number of species are very well documented. The thermal regime of aquatic environments is a major factor controlling the distribution of most species (Brett 1952; Hart 1952). In addition, temperature governs physiological responses to changes within aquatic environments, regulates metabolism and behavioural processes and modifies dietary preferences. Aquatic environments are characterized by diurnal, seasonal and annual temperature extremes, and the upper and lower temperature tolerance limits vary from species to species and within species (Hickman and Dewey 1973). These thermal tolerance limits are affected by a range of biotic and abiotic factors (Fry 1964). Since temperature markedly affects the composition and behaviour of the aquatic community, modification of thermal regimes may have both detrimental and beneficial effects on different species.

The effects of temperature changes in freshwater environments have received a great deal of attention, largely in response to concerns regarding the thermal effluents associated with various power installations. On the other hand, the effects of thermal discharges on marine environments and resources are not documented as well, and there are virtually no data for arctic environments. Some species of fish appear to be attracted to thermal effluents, while others avoid areas having warmer temperatures (Neill and Magnuson 1974; Gray et al. 1977). For example, Olmsted and McPhail (1978) reported the concentration of various freshwater species in a thermally enhanced area of the Peace River during winter, while Gray et al. (1977) reported that juvenile salmon actively avoided a thermal plume which was 9° to 11°C above ambient temperature. The latter authors also postulated that normal behaviour may prevent some species from being exposed to potentially lethal temperatures, and noted that salinity, photoperiod and feeding responses can alter this behaviour. Neill and Magnuson (1974) also found evidence of thermoreception and thermoregulation in freshwater fishes, and suggested that a species tends to maximize its exposure to a specific temperature range. Diel and seasonal variability in species' responses to temperature were also documented by Neill and Magnuson (1974), who also found that thermoregulation was modified (but not overriden) by behavioural reactions to food abundance, strong currents and predators.

Sylvester (1972) and Coutant (1972, 1973) demonstrated increased vulnerability of thermally stressed fishes to predation. In similar studies, Beitinger (1974) and Hartwell and Hoss (1979) each showed a decrease in fish activity (thought to account for increased predation) following exposure to either heat or cold shock. On the other hand, Deacutis (1978) reported both increased and decreased susceptibility of different species of larval fish to predation following thermal shock.

Numerous researchers have demonstrated that increased temperature markedly accelerates hatching time in fishes, but may ultimately reduce survival (e.g. Hubbs and Bryan 1974; Brooke 1975; Berlin et al. 1977). For example, Berlin et al. (1977) observed advanced embryonic development, reduced incubation time and overall diminished reproductive success in lake whitefish (Coregonus) at higher water temperatures. Brooke (1975) also showed that whitefish eggs subjected to thermal enhancement hatched earlier, but produced smaller and more abnormally formed fry than normal, while Hubbs and Bryan (1974) demonstrated critical upper temperature thresholds in shad (Dorosoma) eggs, and observed severe malformations and behavioural abberations when this threshold was exceeded. In addition, Pokrovskii (1961) found that increased temperature caused male whitefish to prematurely vacate spawning areas.

The frequency and lethality of diseases among fishes is also correlated with temperature. Fryer and Pilcher (1974) and Whelen et al. (1981) both reported that the progress of disease in salmonids was accelerated during periods of temperature increase and retarded by decreasing temperatures.

The effects of temperature on fish may be modified by other abiotic factors, while temperature may act synergistically to alter the lethality of contaminants such as trace metals (Section 3.1.10). For example, Barker et al. (1981) observed a synergistic interaction of salinity with temperature, such that increased salinities increased the thermal stress to larval herring (Clupea), flounder (Liopsetta) and rainbow smelt (Osmerus). Stober and Hanson (1974) reported that the toxicity of chlorine to salmon (Oncorhynchus) increased markedly with temperature, and Burton et al. (1972) observed a similar relationship between temperature and zinc toxicity.

Synergistic effects of thermal discharges may occur at production platforms in the Beaufort Sea, particularly if and when formation water is released to the marine environment, since this produced water (1000 m^3/day/platform) may not only have a temperature of up to 55°C, but may also contain elevated levels of dissolved trace metals and up to 50 mg/L of dispersed oil. However, in many cases, fish may avoid thermal plumes prior to exposure to undiluted trace metals or oil near these production facilities. Potential effects of heated water discharge on fish in offshore areas would largely be restricted to temporary alterations in behaviour and sublethal physiological/metabolic changes since long-term exposure would be unlikely and spawning habitats will not be affected. Consequently, the degree of regional concern regarding potential adverse effects of heated water from production platforms on fish is expected to be NEGLIGIBLE.

The effects of heated cooling water from moving ships on fish will also be NEGLIGIBLE because of the limited volume of these discharges and their rapid dilution in surrounding waters. Discharge of cooling water from moored vessels could also temporarily increase water temperatures in the immediate vicinity of the release sites, although the degree of concern regarding adverse effects on regional fish populations would again be NEGLIGIBLE due to the localized area affected and the short-term nature of the discharges.

4.2.7 Effects of Heated Water on Phytoplankton

Although the effects of temperature on arctic marine phytoplankton have not been directly examined, there is a large body of literature describing thermal effects in temperate regions. Talmage and Coutant (1978, 1979, 1980) reviewed the recent literature discussing the effects of temperature on phytoplankton and other aquatic organisms.

In general, an increase in temperature up to some critical threshold increases enzyme activity, and this is reflected in higher photosynthetic activity, growth rates and respiration. Although each species of phytoplankton has some optimum temperature for growth (Goldman and Ryther 1976; Goldman 1977a, b; Eng-Wilrot et al. 1977, cited in Talmage and Coutant 1978), many phytoplankton are found in sub-optimal environments and this results in differential growth rates within the community. Due to these differing growth rates, species succession in marine environments can also be influenced by temperature, although light intensity and spectral composition, nutrient availability and other factors can modify the effects of temperature (Jones 1977).

Thermal discharges usually result in a slight to moderate increase in rates of primary production by phytoplankton (Mothes et al. 1976; Kemp 1977, both cited in Talmage and Coutant 1978), and occasionally alter the species composition within the community (Goldman 1977a,b; Jones 1977; Goldman and Ryther 1976, cited in Talmage and Coutant 1978; Bourgarde 1977, cited in Talmage and Coutant 1980; Campbell 1978, cited in Talmage and Coutant 1980). However, several studies have indicated that the overall impact of thermal discharges on phytoplankton was minimal, and no detrimental effects on regional communities have been documented (Campbell 1978, cited in Talmage and Coutant 1980; Goldman and Quinby 1979).

Heated cooling water and formation water discharged from production facilities in the Beaufort Sea may approach 55°C and would rise through the water column to form a plume of warm water at the surface. Mortality of phytoplankton in the immediate vicinity of the discharge would likely occur, particularly in the presence of dissolved trace metals and dispersed oil, since Cairns et al. (1978, cited in Talmage and Coutant 1979) found that higher temperatures increased the sensitivity of some phytoplankton species to toxicants. Some enhancement of primary production would probably occur in adjacent areas as the thermal plume spreads and temperatures decrease. Localized alteration of the species composition of the phytoplankton community is possible, but only if the plume is persistent and affects the same water mass for a prolonged period, since several generations would be necessary to result in a change in the community structure. However, it is more likely that the plume would affect plankton only briefly as normal water movement transports the populations through the affected zone.

The discharge of heated cooling water and formation water would have a very localized detrimental effect on phytoplankton in offshore waters of the Beaufort Sea, but the overall degree of concern regarding impacts on regional or local populations is considered NEGLIGIBLE. Large and persistent thermal plumes may, in fact, have a slight positive impact by locally increasing the rate of primary production.

4.2.8 Effects of Heated Water on Zooplankton

The initial interaction between arctic marine zooplankton and the discharge of heated water is likely to be one of involuntary entrainment within a zone or plume of warm water, resulting in subsequent mortality or sublethal effects which result from exposure to temperatures beyond the range to which indigenous fauna would be acclimated. Since the passage of planktonic organisms through an area is governed by the movement of the water masses, acclimation resulting from long-term residence within a thermal plume is expected to be extremely unlikely. Due to their relatively small size, zooplankton will probably undergo rapid fluctuations in body temperature in the transition zones between ambient and plume temperatures, and this is likely to preclude the possibility of adaptive physiological responses. As a result, considerable mortality due to thermal shock may be expected within the immediate vicinity of the thermal discharge.

Thorhaug (1974) determined the upper lethal temperature thresholds for various larval stages of a tropical penaeid shrimp *Penaeus duorarum* and a stone crab *Menippe mercenaria* (Table 4.2-1). These data illustrate the variable response of different crustacean species to temperature. Older life history stages of the shrimp generally had a higher temperature tolerance, while the crab larvae exhibited somewhat decreasing temperature tolerance as they developed. The relationship between developmental stage and temperature tolerance has not been examined in arctic crustacea, although similar species-specific differences may be expected in Beaufort Sea zooplankton populations.

TABLE 4.2-1

UPPER TEMPERATURE LIMITS OF LARVAL STAGES FOR TWO TROPICAL CRUSTACEAN SPECIES (from Thorhaug 1974)

Species/Life History Stage	Exposure Time (hrs)	Upper Lethal Limit (°C)
Penaeid Shrimp (*Penaeus duorarum*)		
nauplii	22	30.5 - 31.5
1st protozoea	18	36.0 - 37.6
3rd protozoea	17	36.8 - 37.8
3rd mysis	72	36.8 - 37.8
1st postlarvae	1	37.9 - 40.7
late juvenile	40	36.3 - 38.5
Stone Crab (*Menippe mercenaria*)		
eggs	40	36.3 - 38.5
1st zoea	24	34.4 - 36.0
2nd zoea	91	33.1 - 34.2
5th zoea	44	34.7 - 35.5
megalopa	16	36.0 - 37.0
zoea/mega.	24	30.5 - 31.4
mega./juv.	24	28.9 - 30.5

Beyond the immediate site of release of heated water, diffusion and turbulence will decrease the temperature of the thermal plume as well as mortality of zooplankton. However, a range of sublethal effects may still be anticipated in this area, including altered respiration, feeding rate (Kremer 1976; Bogdon 1977) and possibly behaviour and reproduction. In addition to these direct effects, entrainment in a warm water plume may modify the rate of zooplankton movement (Nicol 1967) and temporarily alter their ability to avoid predation (Talmage and Coutant 1978). If the effects of temperature on locomotion are sufficient to produce a complete loss of swimming ability, this loss tends to be largely irreversible (Reeve and Casper 1972) and would probably quickly result in the death of the affected individuals.

The many zooplankton species which undergo diurnal vertical migration may also be affected by a thermal discharge since this vertical movement can be inhibited either by a large temperature gradient over a small vertical distance or by a temperature above the preferred temperature of individuals in a particular zooplankton population (Gehrs 1974). Insufficient food may be available for maintenance of normal rates of secondary production if the thermal discharge restricts the movement of zooplankton into the euphotic zone. It has also been suggested that obligate herbivorous zooplankton would have difficulty surviving in waters affected by thermal discharges (Gehrs 1974), although both lethal and sublethal effects of heated water release would be relatively localized.

The presence of dispersed oil within warm formation water discharges may increase both the thermal effects and the toxic action of the oil. In addition, the likely presence of open water in the vicinity of thermal discharges during winter may result in the concentration of vertically migrating zooplankton near the outfall, and this would tend to increase the numbers of zooplankton affected. Although the release of heated cooling water and formation water may result in significant detrimental effects on the zooplankton passing through the zone of influence of the discharge, the overall degree of regional concern regarding adverse impacts of these discharges on zooplankton is expected to be NEGLIGIBLE because of the localized nature of any potential lethal and sublethal effects.

4.2.9 Effects of Heated Water on Micro-organisms

Most bacteria survive and grow within a temperature range of approximately 30°C, with optimal growth temperatures being those at which bacteria have the shortest generation time. Growth rates of aquatic bacteria generally increase as water temperatures increase until an upper thermal limit is reached, which varies according to the individual tolerance of each species or strain. Psychrophilic bacteria grow best in cooler temperatures between -5°C and 20°C; this category encompasses most of the marine species (Morita 1974). For example, a marine bacterium isolated from North Pacific water of 3.2°C had optimal growth at 15 to 16°C (Morita and Haight 1964). When the temperature was raised to 29°C for 1 hr some reversible damage was detected, and viability of this culture was completely destroyed after 6 hr at 29°C. Since psycrophilic bacteria in the Beaufort Sea do not live at their optimum temperature for most of the year, a slight rise in water temperature in the vicinity of offshore structures may promote bacterial growth.

Degradation of petroleum hydrocarbons by oleoclastic bacteria is a temperature-dependent process of much interest to marine scientists and the offshore oil industry. Bunch and Harland (1976) found that mixed heterotrophic cultures from the Beaufort Sea had optimum rates of oil degradation at 20°C or lower, indicating the combined presence of both psychrophilic and oleoclastic characteristics. Oleoclastic degradation occurred at temperatures as low as 0° to -1°C although not at optimal rates. The overall degree of regional concern regarding the stimulatory effects of heated water discharge on bacterial growth is considered <u>NEGLIGIBLE</u> due to the localized and generally transitory nature of the thermal discharges.

4.2.10 Effects of Heated Water on Benthic Communities

Since thermal discharges are expected to initially remain in the upper layer of the water column, with the exception of certain epibenthic invertebrates which may swim up into the water column, benthic communities will generally be well below the depth range affected by thermal plumes. Although the effects of temperature on marine organisms is very well documented, information on arctic benthos is limited to several theoretical treatises and laboratory investigations (e.g. Dunbar 1968).

The known temperature tolerances of Beaufort Sea benthos are restricted to data for a few mobile crustacean species. Robilliard and Busdosh (1979) described the thermal preferences of three species of isopods (genus *Saduria*). The most common species, *S. entomon*, is eurythermal and tolerates experimental temperatures ranging from about -2° to 25°C (George 1977; Percy *et al.* 1978, cited in Robilliard and Busdosh 1979). The other two species, *S. sibirica* and *S. sabini*, are generally collected from deeper water (>7m) and are not often found in areas with low salinity and high temperatures. Busdosh and Atlas (1975) described the temperature and salinity tolerances of two amphipods from the Beaufort Sea, *Gammarus zaddachi* and *Boeckosimus* (*Onisimus*) *affinis*. Both species tolerated a wide range of temperatures and salinities, including abrupt changes in these parameters, such as those experienced in crossing the thermocline. Gammarus zaddachi survived sudden temperature changes from 5° to 25°C, while Boeckosimus affinis could only tolerate sudden changes from 5° to 15°C. However, both species were tolerant of higher temperatures when the change was gradual and increased by 3.0°C every 12 hours (Table 4.2-2).

TABLE 4.2-2

TOLERANCE OF AMPHIPODS TO GRADUAL TEMPERATURE CHANGES AT VARIOUS SALINITIES (from Busdosh and Atlas 1975)

	Upper Survival Temperature (°C)	
Salinity °/oo	Boeckosimus affinis	Gammarus zaddachi
5	28	33
10	28	33
20	28	33
30	28	30
40	23	12
50	15	12

Bayne et al. (1977) emphasized that measurement of thermal tolerance may be almost biologically irrelevant when assessing the potential effects of thermal discharges on a sessile invertebrate population. Their study of two mussel populations indicated that individuals adjacent to a warm water discharge from a power station had a higher thermal tolerance but lower "condition factor" compared to individuals of similar size and reproductive state within the control population. These authors suggest that growth, which reflects ingestion rate, assimilation efficiency and metabolic rate, is a more adequate measure of the extent of thermal stress experienced by a population than simple thermal tolerance limits.

Griffiths and Dillinger (1980) reported that the mysids Mysis litoralis and M. relicta and the amphipod Onisimus glacialis all thrived near Simpson Lagoon where the annual temperature range was -2° to 13.5°C, and the mysids appeared to be equally active in summer and winter. These and other authors have suggested that eurythermal arctic fauna can survive over a wide temperature range, but certain processes such as breeding and growth may occur within a relatively narrow band of temperatures.

The wide thermal tolerances exhibited by the species examined to date indicate that localized temperature increases of 5° to 10°C will not adversely affect these mobile epibenthic invertebrates. This and the fact that most benthic organisms would not be exposed to heated water suggests that the degree of regional concern regarding the adverse impacts of thermal discharges on these populations would be NEGLIGIBLE.

4.2.11 Effects of Heated Water on Epontic Communities

The discharge of heated water into the near-surface waters of the Beaufort Sea during and following the normal period of ice cover development may locally reduce the areal extent and duration of the ice cover and its associated epontic community in some offshore areas. Relatively small areas may remain ice-free throughout the year, while additional adjacent areas would likely freeze later and thaw earlier than normal. However, the effects of heated water on the ice cover would be localized and the area affected considered insignificant in comparison with the amount of available epontic habitat in the Beaufort Sea.

Outside the region where the thermal discharge would prevent development of the epontic community and be lethal to phytoplankton in the upper layer of the water column, some enhancement of primary production may occur when light availability in the upper layer of the water column is no longer a limiting factor in the spring. However, conflicting opinions have been expressed as to whether nutrients may (Grainger 1977) or may not (Alexander 1974) be limiting to phytoplankton growth at this time of year. Increased algal biomass in areas adjacent to thermal plumes may attract larger numbers of epontic fauna. As indicated in Section 4.2.10, some amphipods appear to be tolerant of increases in temperature and would therefore be able to take advantage of localized areas of increased food availability near offshore production facilities. Nevertheless, the degree of concern regarding the impacts of thermal discharges on the epontic community is considered NEGLIGIBLE since any positive or negative effects would be localized and not regionally significant given the extent of this community in the Beaufort Sea.

4.2.12 Summary of Concerns Related to Heated Water Discharge

The release of heated cooling and formation water in the offshore Beaufort Sea is expected to result in localized mortality of planktonic and epontic organisms in areas immediately adjacent to outfalls, and increased productivity of these flora and fauna (including bacterioplankton) in outer portions of thermal plumes. If discharges are continuous, most fish would likely avoid any areas where temperatures exceed tolerance limits and most potential effects of heated water on these species would be sublethal in nature. On the other hand, intermittent discharges of heated water could result in localized fish mortality when some individuals are unable to avoid a thermal plume. Areas of open water created around production facilities could also attract marine mammals (primarily seals) throughout the winter as well as offshore migrant birds during the spring and late fall. Although heated water *per se* is not likely to have adverse impacts on birds and marine mammals, attraction of these species to offshore production facilities may increase the opportunity for indirect and direct effects of other sources of disturbance and wastes, particularly dispersed oil and dissolved trace metals present in formation water. Nevertheless, the degree of regional concern regarding adverse impacts of heated water discharge on all marine resources of the Beaufort Sea is expected to be NEGLIGIBLE because of the rapid dissipation of thermal plumes.

4.3 BALLAST WATER/EXOTIC ORGANISMS

4.3.1 Introduction

In order to maintain stability when not carrying cargo, approximately 200,000 m^3 of ballast water will be pumped aboard tankers bound for loading facilities in the Beaufort Sea. About 50 percent of the ballast requirements would be taken from the southern ports since tankers will maintain minimum draft until additional draft is required for icebreaking in Davis Strait, Baffin Bay and the Northwest Passage. The remainder of the ballast water will therefore be pumped aboard at a location which will vary with seasonal ice conditions (usually Davis Strait or Baffin Bay). The water will be pumped into the tanker through a pipe (approximately 1.2 m in diameter) equipped with a strainer comprised of one coarse screen (5 cm slots) and one fine screen (0.5 cm slots) to prevent the potential entrainment of larger marine invertebrates and fish. A backflushing system will be used to clean off the screen as required. The chemical quality of the ballast water will be monitored to ensure that only clean approved water is loaded. Since the ballast compartments in the proposed tankers will be located outside of the oil compartments and adjacent to the hull, the temperature of this water will be approximately equal to the ambient seawater temperature as the tankers travel from southern terminals to the Beaufort Sea. At the Beaufort Sea offshore tanker loading terminals, this ballast water will be discharged to the marine environment at a rate of between 13,000 and 25,000 m^3/hr simultaneously with the on-loading of crude oil. Since the ballast will be carried in segregated tanks, it will not be contaminated with hydrocarbons under normal circumstances. If accidental leakage occurs between the ballast and crude oil storage compartments, the ballast water would be passed through an oil-water separator to reduce oil concentration to 50 ppm or less prior to discharge, as required by the Canadian Oil and Gas Productivity Regulations (EIS Vol. 2).

It is anticipated that there would be no regionally significant biological concerns related to the normal discharge of ballast water. Although the ballast water will probably have a specific gravity different from that of Beaufort Sea water, it has been estimated that the ballast water would be quickly diluted by a factor of 150x in a 1 km^2 area around the tanker loading platforms (EIS Vol. 2). There is a potential for the introduction to the Beaufort Sea of exotic marine species not filtered out by the on-loading pump system. However, for exotic species from temperate or subarctic waters to successfully colonize waters of the Beaufort Sea, they would not only have to survive in ballast tanks until this water is discharged at the loading facilities, but also survive and reproduce once they are in the Beaufort Sea. Although a few species may be able to initially survive the adverse environmental conditions, the majority would not be able to successfully reproduce.

Elton (1958) described accidental carriage in water ballast tanks or on the hulls of ships as a "powerful and steady agency dispersing marine plants and animals about the world". Rosenthal (1980) reviewed many of the accidental transplantations of marine organisms associated with marine traffic and concluded that there is little likelihood of reducing the continuous and world-wide transfer of organisms on ship hulls and in ballast tanks. Certain fish, benthic and planktonic invertebrates, phytoplankton, benthic algae and presumably micro-organisms have been dispersed beyond their normal geographic range by marine vessels, although only a few species have been able to establish breeding populations outside their native range. Introduction of an exotic species does not automatically result in successful colonization, since many intentional transplants have been failures.

Due to the fact that the degree of colonizing success of a given species is highly unpredictable, the potential for transfer of flora and fauna from temperate and subarctic waters to the Beaufort would depend on their trophic level, niche, competing species, physiological tolerance and reproductive requirements. Phytoplankton, bacterioplankton and zooplankton would likely be the only types of organisms transported in ballast water due to the 0.5 cm screens on the ballast intake pipes. Autotrophic phytoplankton species are unlikely to survive the 2 to 3 week period in darkness during transport to the Beaufort, although species capable of heterotrophic uptake as well as bacteria and zooplankton could remain viable providing they are capable of acclimating to progressively lower temperatures over this period. Once these flora and fauna are released to the Beaufort Sea, continued survival would depend on their ability to adapt relatively quickly to a potentially different physical/chemical environment. Immediate mortality of some of the more fragile zooplankton species would be expected as a result of the turbulence associated with ballast release (discharge velocity $\cong$ 320 to 615 cm/s given 1.2 m pipe). If the transported organisms survive the initial exposure to Beaufort Sea waters, a secondary level of requirements must be met, including suitable substrate, forage and light conditions. Tolerable conditions may only exist during certain seasons and survival of the introduced flora and fauna may not be possible over the entire year.

The final requirement for successful colonization is the ability of the exotic organisms to reproduce and leave viable offspring, and this often requires a special set of environmental conditions. Many marine species have non-breeding populations on the fringes of their natural geographic range where suboptimal conditions permit survival but not reproduction. The marine environment of the Beaufort Sea is characterized by more marked variability in physical parameters such as ice cover and light regime than temperate waters, and this is particularly reflected in the seasonality and relatively low standing crops of primary producers compared to temperate latitudes. The latter biological characteristic of this region would tend to hamper successful colonization by herbivorous species, and generally limit potential invading species to detritivores and predators.

Potential biological concerns related to exotic organisms include the possible introduction of new pathogens or parasites for which there may be little genetic resistance in native populations, and the introduction of pests which proliferate at the expense of indigenous flora and fauna, either by direct competition or by increased predation.

4.3.2 Effects of Ballast Water on Fish

Adult fish and juvenile stages larger than 0.5 cm would not be present in ballast water because of the screens on ballast water intakes. On the other hand, eggs and planktonic larvae of some species could pass through the screens and survive within the ballast tanks until they are released in the Beaufort Sea, while disease organisms and parasites of fish may also be present in ballast water.

Forty-three species of anadromous or marine fish have been identified from the Beaufort Sea (LGL and ESL 1981), with approximately half of these species also occurring on the Atlantic coast of Canada (Leim and Scott 1966). This suggests that certain niches in the marine environment of the two regions are similar enough that some species could survive transplantation in the Beaufort Sea, particularly since the Atlantic coastal area supports approximately 250 species not recorded from the Beaufort Sea. However, as suggested earlier, only marine species with planktonic eggs and larvae could be introduced to the Beaufort Sea during tanker operations. The most significant area of potential concern in this regard would be the successful introduction of a species found to be a pest in the Atlantic region such as the sea lamprey (Petromyzon marinus), although the probability of survival and reproduction of this or other temperate latitude species is unknown.

Another potential area of concern related to release of ballast water in the Beaufort Sea is the spread of communicable parasites and diseases of fish. Several viral and bacterial diseases have sometimes had severe effects when transferred from their indigenous host to a more susceptible fish species (Christensen 1973). In a similar manner, parasites usually restricted (and relatively innocuous) to one fish species can have much more serious effects if transferred to a new host in a new environment (Schulman 1954). The introduction of fish-borne human pathogens to a new environment has also been demonstrated (Rosenthal 1980). It should also be emphasized that fish pathogens can also be transported by fertilized eggs (Fisheries and Environment Canada 1977), which is the most likely path of introduction of Atlantic species to the Beaufort Sea because of their greater chance of survival during entrainment, transport and subsequent release in the Arctic. In addition, since ballast water could be repetitively obtained and released in the same localities (southern port and offshore Beaufort tanker loading facilities) year-round for many years, the probability of successful establishment of Atlantic coast organisms in the Beaufort Sea is increased. At the present time, the degree of potential concern regarding the introduction of exotic fish and fish pathogens or parasites to the Beaufort Sea is considered MINOR to MODERATE since the probability of this phenomenon and its ecological implications remain poorly documented.

4.3.3 Effects of Ballast Water on Phytoplankton

As indicated earlier in Section 4.3.1, survival of most autotrophic phytoplankton species is considered unlikely during or after transport to the Beaufort Sea in ballast water. Although the light reactions and electron transport processes of photosynthesis would probably recover following a period of 2 to 3 weeks in darkness, the enzyme-mediated dark reactions of the photosynthetic process are likely to deteriorate with time and reduce the ability of organisms to survive once they are released in the Beaufort Sea. Mortality of most organisms is considered equally probable during the winter and summer, despite the fact that lower temperatures in winter would normally increase the potential for survival of phytoplankton populations when naturally or experimentally transferred to light-limited environments. However, exposure to some light regime is typically required for re-activation of the light reaction pathways of photosynthesis, and during winter, phytoplankton released to the Beaufort Sea are unlikely to receive this stimulus. In the summer months, the warmer water temperatures in areas where ballast is loaded would result in relatively high energy expenditures by phytoplankton and these energy requirements would likely persist until water temperatures in the ballast tanks decreased with movement of the tankers into colder Arctic waters. Since photosynthetic production would be unable to supply this energy, the survival of phytoplankton in ballast water for 2 to 3 weeks would be unlikely unless some species are capable of heterotrophic uptake and metabolism of organic compounds present in seawater.

It should also be emphasized that even if some phytoplankton from southern latitudes survived transport in ballast water, they may not be able to reproduce under the more extreme environmental conditions which are characteristic of the Beaufort Sea. At the same time, the exchange of waters from the Atlantic and Pacific oceans and the Beaufort Sea is well documented (Pickard 1963; Codispoti and Owens 1975), and this allows normal immigration of organisms from temperate oceans. As a result of this naturally occurring phenomenon and the likelihood that most phytoplankton transported in ballast water would not survive, the degree of concern regarding adverse effects of ballast water discharge on regional populations of phytoplankton in the Beaufort Sea is considered NEGLIGIBLE.

4.3.4 Effects of Ballast Water on Zooplankton

Considerable natural opportunities exist for the colonization of Beaufort Sea waters by marine zooplankton from both the Pacific and Atlantic oceans. This assumption is based partly on the presence of identifiable water masses or currents known to originate in these oceans within the Beaufort (Pickard 1963; Codispoti and Owens 1975), and partly on the presence of both

Atlantic and Pacific zooplankton in samples collected in the Beaufort Sea (Grainger 1965, 1975). For example, of the 34 copepod species found in the Beaufort, three are known to enter from the Pacific Ocean while five are characteristic of the deep Atlantic water layer (Grainger 1975).

Pacific zooplankton species currently found in the Beaufort Sea are those forms which can tolerate the environmental conditions encountered while drifting from the north Pacific ocean through the Bering and Chukchi seas. Horner (1978) reported that planktonic larvae of barnacles, polychaetes and echinoderms (some of which presumably originated from the north Pacific Ocean and Bering Sea) were more abundant in the Chukchi Sea than further east in the Beaufort. However, although these species and the other zooplankters of Pacific or Atlantic origin appear to be able to survive for relatively lengthy periods in Arctic waters, it is unlikely that these exotic species would be able to reproduce under the more extreme environmental conditions.

Historically, ship traffic in the Canadian Arctic has been primarily limited to small or medium-sized vessels operating during relatively ice-free periods. The year-round presence of a fleet of icebreaking tankers operating between the eastern seaports and the oil-producing region north of the Mackenzie Delta may significantly increase the rate of introduction of new marine zooplankton species to the Beaufort Sea. While the vast majority of these species would fail to become established, the chronic reintroduction at all seasons of the year would provide the maximum opportunity for colonization by new species. The potential ecological significance of the introduction of exotic zooplankton species to the Beaufort Sea remains unknown. The most significant concern would be a long-term change in the species composition of the herbivorous component of the planktonic community (secondary producers) since this could affect members of higher (predaceous) trophic levels by decreasing the abundance and/or nutrional suitability of prey species. Although the probability of widespread and successful invasion of exotic zooplankton species to the Beaufort Sea associated with ballast water discharge appears remote, the degree of potential concern regarding adverse effects on regional zooplankton populations and members of higher trophic levels dependent on these species must remain at least MINOR in the absence of evidence to the contrary.

4.3.5 Effects of Ballast Water on Micro-organisms

Bacteria often adapt relatively quickly to new environmental conditions due to their large numbers and short generation time. As a result, some exotic bacteria introduced into the Beaufort Sea through the discharge of tanker ballast could survive and become established in the region. The potential effects of introduction of new bacterial strains on the marine ecosystem cannot be predicted at the present time, although relatively serious outbreaks of disease in fish and marine mammals are considered possible if pathogenic organisms to which indigenous fauna have no previous exposure are introduced and remain viable within the region. However, the proposed monitoring of water quality at southern tanker terminals (Section 4.3.1) will reduce this area of concern to NEGLIGIBLE or MINOR if analyses also include pathogenic bacterial counts in local marine waters and areas of domestic sewage discharge are avoided.

4.3.6 Effects of Ballast Water on Benthic Communities

The concern over the introduction of exotic organisms is the potential establishment and multiplication of any species which would compete with and displace indigenous species or eliminate native fauna through selective predation, and thus disrupt food chains or other community interactions within a region. Although this question is perhaps largely theoretical at the present time, there has been historical evidence of successful invasions of exotic organisms associated with marine transport. Benthic invertebrates which have been dispersed through marine shipping include species of crabs, amphipods, shrimps, barnacles, mussels, hydroids, bryozoans and nudibranchs. For example, *Elminius modestus* is a barnacle which spread from New Zealand and Australian rocky intertidal beaches to England and then to France, Holland and Belgium (Elton 1958). This species has also been detected in South Africa. Other barnacles with ranges expanded via ship traffic are *Balanus eburneus* from North America to the Mediterranean and Britain, and *B. improvisus* from the Northern Hemisphere to Australia (Elton 1958).

In a similar manner, the Chinese mitten crab *Eriocheir sinensis* from estuaries in north China has established itself in several European rivers and Lake Erie, and two large specimens were found in seawater ballast tanks of a German steamer, confirming the mode of dispersion of this species (Rosenthal 1980). The planktonic larvae of a Red Sea prawn *Processa aequimana* also increased in abundance within the southern North Sea for several years after 1946 (Elton 1958), while Ricketts *et al.* (1968) reported the presence of a Korean brackish water shrimp *Palaemon macrodactylus* in San Francisco Bay, which probably originated from ballast water.

Benthic algae, including the genera *Asparagopsis*, *Fucus*, *Codium* and *Sargassum*, have also been transported to other regions by marine vessels. For example, *Asparagopsis armata* was transported by ship from Australia or New Zealand to Mediterranean and European waters (Elton 1958). Consequently, there is considerable evidence of introduction of exotic species in temperate and tropical environments as a result of transport of flora and fauna in the ballast water or on the hulls of marine vessels.

The withdrawal of ballast water from the upper portion of the water column and use of screens on ballast intakes will minimize the risk of introduction of adult benthic vertebrates to the Beaufort Sea during proposed tanker operations. Nevertheless, planktonic larval stages of benthic invertebrates could be present in seawater used for ballast purposes during some months of the year, and could survive transport to the Beaufort Sea within ballast compartments. Subsequent survival in this region would depend not only on their ability to adapt to the more extreme physical environment, but also on the availability of a suitable substrate in an area where silt-clay sediments predominate. Burrowing polychaetes, bivalves, amphipods and isopods would be most likely to locate suitable habitats since these groups already predominate throughout most of the region, while exotic species requiring hard substrates (e.g. barnacles, hydroids, bryozoans, tube-dwelling polychaetes) would be less likely to survive. The degree of concern regarding introduction of exotic benthic organisms to the Beaufort Sea would vary with the type of organisms introduced and their potential for displacement of certain indigenous forms which are important prey species of fish, birds and marine mammals in the region. Unless the abundance and distribution of amphipods and isopods in locally important habitats were significantly altered, the degree of concern regarding the effects of exotic benthic fauna on indigenous fauna would likely be NEGLIGIBLE. If local populations of these species were affected by exotic organisms, the degree of concern could be increased to MINOR or MODERATE.

4.3.7 Effects of Ballast Water on Epontic Communities

The year-round operation of tankers between the Beaufort Sea and east coast ports will increase the possibility of introduction of new species to the epontic community of the former region. However, only one comparative study of southern and arctic epontic communities has been conducted to date. Dunbar and Acreman (1980) found that the epontic flora in the Gulf of St. Lawrence was dominated by centric diatoms, while pennate forms were most common in arctic environments. These authors suggest that differences in the composition of the epontic algae between the Arctic and Gulf of St. Lawrence could be related to the thinner and shorter period of ice cover in the southern region, which would tend to result in higher under-ice light intensities. These conditions favour the growth of centric diatoms rather than the shade-adapted pennate forms which dominate arctic epontic communities. The fact that Dunbar and Acreman (1980) documented similar nutrient concentrations in ice cores from areas ranging from 75°N to 45°N latitude is further evidence that light and ice cover are the limiting factors for this community. Consequently, it appears unlikely that populations of epontic algae from southern areas could successfully become established and compete with arctic species which are adapted to a different set of environmental conditions (i.e. less light and a longer period of ice cover), and the overall degree of concern regarding introduction of exotic flora to the Beaufort Sea is considered NEGLIGIBLE.

There are no available data which compare the species composition of the faunal component of epontic communities in the Beaufort Sea to waters near southern ports, although the epontic fauna in Davis Strait where a portion of the ballast may be loaded (Section 4.3.1) have been examined (MacLaren Marex Inc. 1979). Gammarid amphipods are the most commonly occurring macroinvertebrates on the undersurface of the ice in both the Beaufort Sea and

Davis Strait (MacLaren Marex Inc. 1979; LGL and ESL 1981). The probability of successful establishment of exotic fauna from the Davis Strait/Baffin Bay region within the Beaufort Sea may be greater than for fauna from near southern ports due to the shorter transport period and the fact that the subarctic forms are more likely to be able to adapt to conditions in the Beaufort. Nevertheless, the degree of concern regarding potential introductions of exotic epontic fauna is expected to be NEGLIGIBLE to MINOR since the overall composition and stability of the amphipod-dominated community is unlikely to be adversely affected in a regional context even if breeding populations of a few species were established.

4.3.8 Summary of Concerns Related to Ballast Water Discharge

All concerns related to release of ballast water from icebreaking tankers in the Beaufort Sea are associated with the introduction of exotic organisms. In the event that breeding populations of exotic species were established, these populations could compete with and displace indigenous species or eliminate native fauna through selective predation. The use of screens on ballast intake lines and monitoring of water quality in areas where ballast water is loaded is expected to minimize this risk, although successful invasions of exotic species as a result of marine vessel operations have been documented in at least temperate and tropical environments. In general, the more rigorous physical environment of the Beaufort Sea is likely to further minimize the potential for establishment of temperate and subarctic species. The degree of potential concern regarding introductions of phytoplankton, zooplankton, epontic organisms and non-pathogenic micro-organisms in ballast water is expected to be NEGLIGIBLE to MINOR. On the other hand, the degree of potential concern regarding introduction of some fish pathogens, parasites, larval fish and epibenthic invertebrates is considered MINOR to MODERATE because the probability and ecological implications of successful colonization by these groups remain poorly documented.

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5.0 ABNORMAL OPERATIONS AND ENVIRONMENTAL EMERGENCY SITUATIONS ASSOCIATED WITH HYDROCARBON EXPLORATION AND PRODUCTION

5.1 NATURAL GAS BLOWOUTS AND PIPELINE RUPTURES

5.1.1 Introduction

Subsea pipeline ruptures or well blowouts would result in percolation of natural gas through the water column before it is vented to the atmosphere or trapped under ice. The overall solubility of natural gas in seawater would depend on the relative solubilities of its component gases, which vary somewhat between hydrocarbon production areas. Natural gas is normally composed of a number of light alkanes (methane, ethane, propane, butane, pentane and hexane) as well as nitrogen, carbon dioxide and sulfur, although methane usually comprises over 80 percent of the mixture. In shallow waters where pressures are relatively low, methane solubility in seawater generally varies from 50 to 200 ml/L, which is approximately equal to 100 to 400 ppm. However, at a depth of 100 m, concentrations of methane up to 3000 ppm could exist in the immediate vicinity of a gas leak (Johnston 1976).

Small quantities of methane are naturally present in seawater, with higher concentrations typically occurring in anoxic bottom waters. Lamontagne *et al.* (1973) found average methane concentrations of 6.4×10^{-5} ml/L in the Norwegian Greenland Sea and 8.6×10^{-5} ml/L in Greenland pack ice, while Wong *et al.* (1976) documented dissolved methane concentrations at various stations in the southern Beaufort Sea (Table 5.1-1). Methane concentrations were generally higher in near-bottom waters, probably as a result of biodegradation processes within the sediments.

TABLE 5.1-1

METHANE CONCENTRATIONS MEASURED AT VARIOUS DEPTHS IN THE SOUTH BEAUFORT SEA
(Source: Wong *et al.* 1976)

Depth (m)	Concentration (ml/L)
0-5	$4.3 - 55.3 \times 10^{-5}$
10-35	$6.9 - 115.5 \times 10^{-5}$
48-70	$7.8 - 41.2 \times 10^{-5}$
>100	$1.6 - 24.3 \times 10^{-5}$

5.1.2 Physical-Chemical Aspects of Natural Gas Blowouts

The solubilities in seawater of most alkanes present in natural gas are higher than that of oxygen. For example, the solubility of methane at 10°C and a pressure of 1 atmosphere is 22.6 mg/L, while the solubilities of ethane and propane, which typically comprise the next largest proportions of natural gas, are greater than 100 mg/L. By comparison, solubilities of oxygen in seawater at this temperature vary from 9 to 11 mg/L, depending on salinity. Concern has been expressed that percolation of natural gas through water will indirectly affect biota through a reduction in dissolved oxygen levels (Foothills Pipe Lines (Yukon) Ltd. 1977; Welch et al. 1978), either by stripping oxygen from the water column, or by increasing bacterial respiration during metabolism of natural gas components. A dissolved gas such as oxygen may be entirely removed ("stripped") from solution by bubbling natural gas (or other gases) through the water. This is due to the fact that the oxygen will leave solution and diffuse into an oxygen-free medium (e.g. a bubble) until the concentrations in the two media are equal. Bubbles of natural gas passing through the water column would produce a relatively large surface area for the diffusion of dissolved oxygen out of solution. For example, Foothills Pipe Lines (Yukon) Ltd. (1977) reported that dissolved oxygen levels dropped from 11.8 mg/L to 1.4 mg/L within 60 min as natural gas was bubbled through small test containers at a rate of about 28 L/min. However, when aeration was simultaneously provided with the natural gas input, dissolved oxygen levels ranged from 7.2 and 7.4 mg/L at 10°C.

This type of experimental situation cannot be directly extrapolated to an undersea gas blowout for a number of reasons. The large flow of water entrained with the bubble plume as it rises to the surface would continually bring dissolved oxygen to the system. In addition, the rate of oxygen removal from plume water would be in part dependent on the time that the gas bubbles and water are in contact. During the open water season in the Beaufort Sea, most of the gas will be released to the atmosphere within a short distance of the plume center. If a mean vertical plume velocity of approximately 0.7 m/s is assumed [Topham (1975) found values of about 1.0 m/s for 15 m deep plumes and 0.7 m/s for 60 m deep plumes], then plume water rising from a blowout site at a depth of 60 m would be in contact with gas bubbles for about 1 to 4 minutes. In a semiclosed system at 10°C, oxygen levels decreased by 30 percent in the first 10 minutes when 15 L of freshwater were bubbled with natural gas at a rate of 28 L/min [Foothills Pipelines (Yukon) Ltd. 1977]. Using these figures as above average stripping rates, a decrease in the oxygen content of plume water of from 3 to 6 percent can be estimated for a 60 m depth blowout.

The behaviour of natural gas released beneath an ice cover has been investigated by Topham (1975, 1978) and summarized by Milne (1980). A recent study by Buist et al. (1981) for Dome Petroleum Ltd. examined a simulated oil and gas release beneath the ice in McKinley Bay, where air was substituted for natural gas for safety reasons. The plume dynamics observed by these authors can be summarized as follows:

1. Violent turbulence associated with the discharge of oil and air set up an inflowing current that suspended and entrained sediments from the seafloor;

2. The rising gas plume began spreading outwards approximately 6 m above the discharge pipe, and some of the sediment on the periphery of this plume began falling out, while the oil droplets continued to rise slowly as a result of their buoyancy;

3. The rising gas, oil, sediment and water ascended in a cylindrical plume to within 7 m of the underside of the ice before beginning to spread outward, creating turbulent eddies which changed to laminar flow about 15-20 m away;

4. The gas quickly rose to the ice/water interface and flowed "uphill" to collect in natural pockets, while the remaining entrained sediments fell out of the plume.

Gas trapped in small pockets under fast-ice during freezing conditions could become encapsulated by the formation of new ice. During stable or melting ice conditions, the oxygen content of trapped gas would quickly approach equilibrium with the seawater below it, while the total volume of the gas pocket would steadily diminish as it dissolved in the surrounding water. Based on the general theory discussed by Welch (1952), surface water moving under freshly trapped pockets of gas would lose oxygen (and absorb gas) at a rate proportional to the surface area in contact, the temperature and pressure of the system, and the proportion of oxygen already mixed with the gas. With the possible exception of gas pockets which accumulate under rough multi-year ice, no major changes in dissolved oxygen would be expected as a result of a natural gas blowout.

The accumulation of gas beneath ice can flex and eventually crack the ice sheet, and the gas would then be vented to the atmosphere. The diameter of the gas area required to fracture an ice sheet would depend on the amount of gas, the size of the under-ice irregularities, and the rates of gas delivery and horizontal spreading. Milne (1980) suggested that landfast ice having an average thickness of 1 m would have a vertical variability in the under-ice surface of perhaps 20 cm. These irregularities could retain a trapped gas layer about 10 cm deep which, if spread to a diameter of 70 m, would have sufficient buoyant force to rupture the 1 m thick ice by upward bending. As a general rule, Milne (1980) suggested that an ice sheet 2 m thick will crack when gas has occupied an area approximately 100 m x 100 m beneath it, although rough multi-year ice would tend to spread impinging gas through cracks and channels, thereby preventing the formation of a single bubble large enough to crack the ice. Buist et al. (1981) reported that 950 m^3 of air lifted 65 cm thick landfast ice into a dome 50 m across and 1 m high before cracking and subsequent venting occurred.

Gas hydrates are formed under conditions of cold temperatures in conjunction with increased pressures such as are found in very deep waters, and are approximately 10 percent gas and 90 percent water by weight. For example, methane gas in the water column at 0°C exists in hydrate form below a depth of 250 m, while other natural gases form hydrates at this temperature in waters deeper than 100 m (Milne and Smiley 1978). Gas hydrates would not likely be encountered during a natural gas blowout in the Beaufort Sea region since all proposed drilling programs and subsea pipelines are in water depths of less than 100 m.

5.1.3 General Biological Considerations

There is only limited literature describing the toxicity of natural gas to aquatic flora and fauna. Hann and Jensen (1974) reviewed the average toxicity of natural gas constituents to a range of aquatic flora and fauna, and reported that TL_m (96-h) values for methane, butane, and propane were all >1000 ppm. Foothills Pipe Lines (Yukon) Ltd. (1977) reported no fish mortality from natural gas when holding waters were simultaneously aerated.

The potential impacts of natural gas leaks or blowouts in the Beaufort Sea will be highly dependent on the rate and duration of gas escape and the time of year. During the open water season, gas leaks would not likely affect most marine organisms since this gas would quickly escape to the atmosphere, although the explosion hazard may represent a potential area of concern for birds and mammals in the region.

On the other hand, a natural gas leak or blowout during the winter could cause greater impacts on certain marine organisms, particularly in areas where gas is trapped but not encapsulated beneath the ice cover. Epontic habitat would be temporarily lost in areas where gas is encapsulated in the growing ice cover. In addition, localized mortality of epontic flora and some fauna could result from anoxic conditions at the ice-water interface. It is possible that some of the methane trapped beneath ice may be metabolized by certain strains of bacteria (*Methanomonas* spp.), but in the case of a major gas blowout, the proportion of the trapped gas metabolized would be small.

Possible indirect effects which may be associated with gas leaks and blowouts are discussed elsewhere in this document and include increased sedimentation (Section 2.4), underwater shock (Section 3.3), underwater sound (Section 2.6) and gas flares (Section 4.1).

5.1.4 Effects of Natural Gas on Marine Mammals

Marine mammals in the Beaufort Sea region may be affected by a natural gas blowout as a result of the initial shock wave and subsequent underwater noise associated with the release and upward movement of gas. In addition, seals may be affected if they surface and breathe from pockets under the ice where gas may accumulate, or if gas permeates subnivean birth lairs of the ringed seal in the landfast ice zone. However, in a recent assessment of the potential effects of offshore oil and gas development on marine mammals of the northern Bering, Chukchi and Beaufort seas, Cowles et al. (1981) state that effects of natural gas are "unlikely to occur to an extent significant enough to affect the overall status of non-endangered mammal populations". Consequently, the degree of concern regarding effects of natural gas on regional populations of whales and seals in the Beaufort Sea is considered NEGLIGIBLE.

Underwater shock waves and sound associated with a natural gas blowout may be audible to marine mammals within several kilometres, depending on ambient noise levels and the location of the well. The potential effects of underwater sound on marine mammals of the southeastern Beaufort Sea were discussed in detail in Section 2.6.5. Individuals in the vicinity of a blowout may elicit a "fright/flight" reaction in response to the initial shock wave, while the continual noise associated with natural gas percolating through the water column may cause avoidance of a localized area by marine mammals. Spatial or temporal displacement of seals or whales is of greatest concern during migratory and breeding periods.

There is also a potential for small numbers of seals to be directly affected by inhaling natural gas from a blowout. Milne (1974) observed seals which were apparently inhaling air bubbles that had accumulated under ice, and some individuals may similarly inhale natural gas that collects in unfractured areas prior to break-up. Although the effects of inhalation of natural gas by seals have not been documented, some compounds of natural gas may be acutely toxic (Cowles et al. 1981) or cause temporary narcosis. Nevertheless, as indicated earlier, the overall degree of concern regarding potential effects of natural gas on whales and seals is considered NEGLIGIBLE.

5.1.5 Effects of Natural Gas on Fish

The effects of natural gas and gas well blowouts on fish are not well documented, but may include localized sublethal effects resulting from toxic effects of dissolved alkanes and minor reductions in dissolved oxygen availability, as well as indirect effects of uplifted bottom sediments. Since natural gas from the Beaufort region has an extremely low sulfur content, toxic effects associated with the presence of hydro-sulfates (Welch 1952; Brooks et al. 1978) would not be expected. The effects of natural gas of similar composition to that tested from the Beaufort region were examined by Foothills Pipe Lines (Yukon) Ltd. (1977). This gas had no acute toxic effects on rainbow trout in freshwater during 96-h bioassays when the test media were simultaneously aerated. Although this investigation indicated complete mortality of rainbow trout in non-aerated samples when dissolved oxygen levels decreased from 11.8 mg/L to 1.4 mg/L within 60 min, significant "stripping" of dissolved oxygen would not be expected in the natural environment (Section 5.1.2).

Redistribution of sediments and increased water turbidity would likely result from entrainment of loose bottom materials in the rising gas plume. In a simulated blowout discussed by Buist et al. (1981), most of these materials were transported to the surface where the sediment separated and began to sink to the bottom. Currents radiating outward from the centre of the gas plume at the ice-water interface turned downwards 15 to 20 m beyond the simulated blowout site (Buist et al. 1981). Distribution of fine sediments beyond this zone would depend on prevailing ocean currents, although the spatial extent of the resultant turbidity plume would probably be no larger and likely less than plumes associated with stationary suction dredging operations. The effects of this sediment on fish would also be similar to those discussed in relation to dredging (Section 2.4.5).

There is no available information on the magnitude of the shock waves which may be associated with blowouts in the Beaufort Sea, although calculations completed for a hypothetical rupture of a 1219 mm gas pipeline in a Yukon lake (operating at approximately 8000 kp) suggest probable mortality of fish within a 70 m radius of the rupture site (McDonald and Bengeyfield 1982), based on both spherical and cylindrical spreading and lethal pressure levels of 100 kp (Teleki and Chamberlain 1978). The effects of shock waves on fish were previously discussed in Section 3.3.4, where it was emphasized that the lethal effects are reduced if the shock originates from the sea floor (e.g. well blowout, pipeline rupture) rather than in the water column (Rasmussen 1967). Consequently, only limited mortality of fish would be expected due to underwater shock waves created by a well blowout or subsea pipeline rupture in the Beaufort Sea.

Overall, the degree of concern regarding the potential effects of gas well blowouts and subsea pipeline ruptures on regional fish populations of the Beaufort Sea are expected to be NEGLIGIBLE because: (1) natural gas in this region is unlikely to cause lethal effects; (2) 'stripping' of dissolved oxygen would be limited and would not affect fish; (3) effects of uplifted sediments would be localized, and (4) direct mortality of fish due to the initial underwater shock wave is also expected to be very localized.

5.1.6 Effects of Natural Gas on Phytoplankton

In the case of a subsea gas pipeline rupture or well blowout in the Beaufort Sea, the buoyancy of the uprising gas plume from water depths <100 m would limit the surface area and duration of contact with phytoplankton in the water column. Information on the toxicity of natural gas to phytoplankton is not available, although toxic concentrations would likely lie within the range (10 to 1000 ppm) reported for other aquatic organisms (Hann and Jensen 1974; Johnston 1976). The effects of the rising gas plume on phytoplankton could include mortality and a variety of sublethal physiological changes, but these effects would be primarily limited to those organisms entrained in the rising column of bubbles since water-soluble alkanes from natural gas would be rapidly dissipated in surrounding waters by diffusion and currents. Uplifted sediments entrained by the rising gas plume could also result in light-related responses similar to those previously discussed in relation to dredge-created turbidity plumes (Section 2.4.6), although this may be offset by increased nutrient availability associated with the upwelling of bottom waters. However, all potential effects of natural gas on phytoplankton would be extremely localized and therefore of NEGLIGIBLE regional concern.

5.1.7 Effects of Natural Gas on Zooplankton

The possible effects of natural gas blowouts or subsea pipeline leaks/ruptures on Beaufort Sea zooplankton may result from the toxicity of dissolved gas, entrainment of individuals within a plume of different salinity and temperature characteristics, and increased water turbidity associated with uplifting of sediments.

As indicated earlier, the limited available literature suggests that natural gas (or its principle constituents) is relatively non-toxic to aquatic life, and therefore any lethal or sublethal effects of gas on zooplankton are expected to be restricted to those individuals which may be entrained in the rising gas plume. However, the buildup of dissolved natural gas levels in the water to the concentrations required for toxic effects may be prevented by the entrainment of large volumes of water at the base of a rising gas-bubble plume (Topham 1975; Buist *et al.* 1981). With the possible exception of gas pockets which accumulate under rough multi-year ice, major changes in dissolved oxygen that would affect zooplankton are also considered unlikely.

The induced upwelling of bottom water through entrainment in a rising gas-bubble plume would likely cause fluctuations in near-surface salinity and temperature profiles, particularly during open water periods in areas influenced by the Mackenzie River discharge. Once the gas bubbles no longer provide lift for this dense, saline water, it would sink and draw with it a quantity of surface water (Topham 1975). Zooplankton most likely to be affected by this phenomenon are freshwater forms transported into the estuary by the Mackenzie River (Grainger 1975). These salt-intolerant species (e.g. *Daphnia* and *Diaptomus*) may suffer premature mortality, although this would not be considered a significant concern since these individuals will no longer be part of their respective freshwater breeding populations. The marine zooplankton in offshore areas would be affected to a lesser degree since they are adapted to wider changes in temperature and salinity.

Buist *et al.* (1981) reported that bottom sediments were drawn into the base of a bubble plume and carried with it to the surface. However, due to the short duration of this simulated blowout (approx. 30 min), it is uncertain whether this entrainment was only an initial phenomenon or one that could be expected to continue for the duration of a natural gas blowout. If the latter was true, then a general increase in surface water turbidity near the rising gas plume may result, causing potential effects similar to those discussed in relation to dredging operations in this region (Section 2.4.7).

In addition, depending upon the size and duration of the blowout, the mixing of bottom and surface waters may stimulate higher rates of local primary productivity in those areas where nutrients are the limiting factor, thereby providing additional food for herbivorous zooplankton. However, the magnitude of these effects is likely to be much lower than similar effects associated with dredging.

In summary, apart from the potential for slightly reduced dissolved oxygen levels in surface water located under gas pockets in rough multi-year ice and the premature mortality of some freshwater zooplankton near the rising gas-bubble plume, it is anticipated that the degree of concern related to potential adverse effects of a natural gas blowout or pipeline rupture on Beaufort Sea zooplankton would be NEGLIGIBLE.

5.1.8 Effects of Natural Gas on Micro-organisms

Methane trapped beneath the ice may undergo some degradation by certain strains of bacteria, although most natural gas would escape to the atmosphere due to fracturing of landfast ice or natural cracks in the transition zone (Section 5.1.2). Pseudomonad bacteria that actively convert methane into carbon dioxide in the presence of oxygen were described by Hutton and Zobell (1949). Although these micro-organisms are normally found in marine sediments, growing ice crystals often transport sediment from the substrate to the underside of the sea ice and probably relocate bacteria during the process. In the event of a natural gas leak or blowout, the amount of methane oxidized by bacteria would likely be relatively small. These micro-organisms have been found in environments where the temperature was as low as 3°C, but oxidation rates in the laboratory at this temperature were extremely slow. Certain pseudomonad bacteria may also oxidize ammonia to nitrite (Hutton and Zobell 1953), and as a result, an increase in metabolism by methane-oxidizing bacteria could simultaneously increase nitrite levels in the water column. The photosynthetic bacterium *Rhodopseudomonas gelatinosa* has been reported to utilize methane rather than CO_2 in the photosynthetic pathway (Wertlieb and Vishniac 1967). Nevertheless, methane-metabolizing bacteria are expected to have NEGLIGIBLE effects on the overall quantity of natural gas that may be released during a blowout or major subsea pipeline rupture relative to losses which would occur through venting and natural dispersion processes.

5.1.9 Effects of Natural Gas on Benthic Communities

The effects of natural gas on benthic flora and fauna have not been documented. Most benthic infauna and microalgae in the area of a well blowout or pipeline rupture would not be affected by natural gas because of its buoyancy. On the other hand, epibenthic invertebrates such as mysids and amphipods could be entrained in the rising plume of gas bubbles and experience the various lethal and sublethal effects described earlier for zooplankton

(Section 5.1.7). However, since any impacts would be very localized, the degree of concern in relation to the effects of natural gas on regional benthic populations is expected to be NEGLIGIBLE.

5.1.10 Effects of Natural Gas on Epontic Communities

There is no available literature describing lethal concentrations or sublethal effects of natural gas or its components on epontic flora and fauna. A natural gas blowout or pipeline rupture during the spring epontic algal bloom would likely result in lethal effects due to anoxia or the direct effects of toxic constituents of the gas on both flora and fauna trapped in the ice above gas pockets. The spatial extent of these effects would depend upon the volume of gas which escaped, although fracturing of the ice cover due to the buoyancy of natural gas (Section 5.1.2) would tend to minimize the volume of gas which accumulates and persists under fast ice. In a regional context, decreased abundance and productivity of epontic flora and fauna would likely be insignificant, and as a result, the degree of potential concern regarding adverse effects of gas blowouts and pipeline ruptures on this community is considered NEGLIGIBLE.

5.1.11 Summary of Concerns Related to Natural Gas Release

The degree of concern regarding potential adverse effects of subsea natural gas release (well blowouts and pipeline ruptures) on all marine resources of the Beaufort Sea is expected to be NEGLIGIBLE. This is primarily due to the relatively low toxicity of 'sweet' natural gas, the small area that would be affected by water-soluble constituents, and the fact that most gas would escape to the atmosphere during both summer and winter. The most significant effects of natural gas on fish and perhaps marine mammals would probably be associated with the underwater shock waves created by a subsea pipeline rupture or well blowout, since this shock could result in localized mortality. In addition, some epontic flora and fauna, epibenthic invertebrates and planktonic organisms could be lost due to entrainment in the uprising gas plume or accumulation of gas in pockets below multiyear ice.

5.2 CRUDE OIL SPILLS AND BLOWOUTS AND REFINED FUEL SPILLS

5.2.1 Introduction

Crude oil could enter the marine environment of the Beaufort Sea due to several types of environmental emergencies including subsea pipeline ruptures, valve failure at tanker loading facilities, tanker accidents, storage tank failures or subsurface well blowouts, as well as in low concentrations (<50 mg/L) during normal activities such as discharge of produced waters. Since the fate and behaviour of oil in marine environments have a major influence on its subsequent biological effects, the following section briefly summarizes available information regarding the potential behaviour and fate of oil in arctic waters following spills in the open water season, surface spills in and on ice, subsea oil blowouts during open water, and subsea oil release under ice as well as the behaviour of oil on shorelines and in bottom sediments.

Refined petroleum products will be utilized continuously for the life of the project to fuel and lubricate all types of machinery, vessels and aircraft. Under normal operating conditions, it is possible that small quantities of refined fuels may enter the water column during fueling operations, in treated drill rig wash water (<50 mg/L), or in treated drainage from vessel decks. Accidental spills of larger volumes of refined petroleum products could occur due to abnormal occurrences such as storage tank failures, vessel collisions or groundings, or line or valve failures during oil handling operations.

5.2.2 Fate and Behaviour of Oil in Marine Environments

The physical-chemical characteristics of a crude oil or refined petroleum product will affect both their behaviour in the water and toxicity to marine organisms. The properties of Kopanoar crude oil from the Beaufort Sea were determined by Mackay (1980) and are summarized in Table 5.2-1. This crude tends to be quite light, relatively low in viscosity and would likely pour or flow under most arctic conditions. Consequently, if Kopanoar crude were spilled, it would easily migrate through channels in the ice, remain fluid on water and ice, and retain a non-viscous character on water for some time. Mackay (1980) also indicated that Kopanoar crude tends to form stable water-in-oil emulsions (mousse) when sufficiently agitated.

TABLE 5.2-1

PROPERTIES OF KOPANOAR CRUDE OIL
Source: Mackay (1980)

Percent mass loss due to evaporation	Pour Point (°C)	Flash Point (°C)	Specific Gravity (g/m^3)	Viscosity (cps) (0°C)	(15°C)	(25°C)
0	-37	75	.900	57	33	175
3.7	-28	86	.901	75	41	24
11.6	-19	118	.902	104	54	30

There is considerable evidence that the aromatic hydrocarbon derivatives, including mono- and dicyclic aromatics and naphtheno-aromatics, are the most acutely toxic constituents of petroleum (Anderson et al. 1974a). These toxic compounds are usually contained in higher concentrations in refined petroleum products than in crude oils. Many of these aromatic compounds are also soluble to greater or lesser extents in water, which increases the potential concern regarding adverse effects of refined fuels on some marine flora and fauna. However, it should be emphasized that many of the toxic constituents of refined fuels are volatile and therefore also dissipate rapidly due to evaporation. Typical physical properties of various refined petroleum products are summarized and compared to crude oils in Table 5.2-2.

5.2.2.1 Surface Oil Spills

There is an extensive amount of literature describing the behaviour and fate of oil in open water, and much of this information was summarized in a recent overview by Fingas et al. (1979). Oil spilled on water immediately spreads in response to gravity, currents and wind, although surface water currents are the most prominent force acting on the oil and the slick may be transported at 50 to 100 percent of the current velocity. The effect of wind on the net movement of a surface oil slick depends on wind velocity and direction relative to the surface water currents. When both these forces act in the same direction, the rate of slick movement may exceed either wind or current components. Very strong winds may even move the slick "up-current", while winds in excess of 16 km/h often cause the slick to break up into streaks or windrows (Fingas et al. 1979).

TABLE 5.2-2

COMPARISON OF TYPICAL PHYSICAL CHARACTERISTICS OF SELECTED REFINED PETROLEUM PRODUCTS AND CRUDE OILS (adapted from Fingas et al. 1979)

	Specific Gravity (15°C)	API Gravity (15°C)	Viscosity (38°C)	Pour Point °C	Flash Point °C	Initial Boiling Point °C
Crude Oils	0.8 to 0.95	5 to 40	20 to 1000	-35 to 10	Variable	30 to 500
Kopanoar crude oil	0.900	-	175*	-37	75	-
Gasolines	0.65 to 0.75	60	4 to 10	na	-40	30 to 200
Kerosene	0.8	50	1.5	na	55	160 to 290
Jet Fuel	0.8	48	1.5	-40	55	160 to 290
No. 2 Fuel Oil (Furnace, diesel stove)	0.85	30	15	-20	55	180 to 360
No. 4 Fuel Oil (Plant Heating)	0.9	25	50	-10	60	180 to 360
No. 5 Fuel Oil (Bunker B)	0.95	12	100	-5	65	180 to 360
No. 6 Fuel Oil (Bunker C)	0.98	10	300 to 3000	+2	80	180 to 500

* at 25°C

The following processes alter the physical composition of the oil during the period when it is spreading or being transported by winds and currents: formation of water-in-oil emulsions which increases the viscosity of the oil and volume of the slick; dispersion which reduces the volume of oil on the water surface and introduces oil to the water column; evaporation and dissolution which both result in the loss of the lighter fractions; and oxidation, and chemical and bacterial degradation which cause slow alteration of the oil composition. A relatively large proportion of the oil may be dispersed in the form of an oil-in-water emulsion, depending on the wind speed and sea state. High winds also increase the evaporative loss of volatile fractions by causing the formation of sprays and aerosols at the crests of waves. However, large droplets of oil dispersed into the water column by high winds may eventually coalesce and rise to reform part of the slick. On the other hand, some smaller oil droplets can adhere to planktonic organisms, silt and detritus present in the water column, and then be transported away from the slick by currents or sink to the bottom. Evaporation accounts for the largest proportion of the volumetric loss of crude oil after a spill, with evaporation rates of 25 percent of the total volume of crude spilled within one day being common under certain conditions (Fingas et al. 1979). Gasoline or aviation fuel spilled on water evaporates even faster, and may lose approximately 50 percent of its original volume within 7 to 8 min at 20°C. On the other hand, diesel fuel evaporates slower than most crude oils, and would typically lose 13 percent of its volume within 40 hours at 23°C (Fingas et al. 1979). The role of evaporation in the weathering of oil decreases with time following the rapid initial loss of volatile hydrocarbons. The loss of oil from the slick due to dissolution is relatively small compared to losses associated with other weathering processes since the majority of hydrocarbons present in petroleum are either non-soluble or have a low solubility in seawater. Highly weathered oil may form tar balls, which can have a density as great as or greater than water, and therefore may be suspended throughout the water column or settle to the bottom. Photo-oxidation and chemical degradation are relatively slow processes which continue throughout the period of weathering. The role of microbial degradation in the weathering of oil is described in detail in Section 5.2.13. Weathering generally tends to reduce the toxicity of the oil to fish and planktonic organisms, although the dispersal of oil during this process can result in contamination of other marine habitats.

5.2.2.2 Oil on Shorelines and in Bottom Sediments

When oil is released into the marine environment, a portion of it will usually be stranded on shorelines by waves and currents, while other portions may sink in offshore areas and become incorporated in the bottom sediments. Oil deposited on shorelines is frequently in the form of a water-in-oil emulsion or "mousse". Oil stranded on high energy shorelines with rocky substrates is usually rapidly removed by wave action. However, most high energy beaches in the Beaufort Sea are composed of gravel or sand, and these porous substrates tend to trap and retain oil. In addition, mousse has a tendency to pick up sand and debris on these types of beaches, and once the water evaporates, this mousse forms a tarry substance which resists further weathering. Oil stranded on porous substrates can be buried, exposed and reburied several times by waves, currents or shoreline erosional processes. Oil and adsorbed beach materials may also be transported along the shore or back into the water column. Preliminary analysis of samples of Kopanoar crude indicate that this oil would remain relatively fluid under cold arctic conditions for some time (Mackay 1980). This characteristic, the primarily porous substrates of the Beaufort Sea coastline, and the active erosion and deposition processes common throughout the region suggest that the potential for re-contamination of the marine environment by oil stranded in shoreline substrates would be relatively high.

Oil can reach subsea sediments as a result of several processes including: 1) sinking of weathered oil in the form of tar balls which have a higher specific gravity than water; 2) adsorption of oil droplets (oil dispersed in the form of an oil-in-water emulsion) onto suspended sediment particles which eventually settle to the bottom; 3) direct sinking of highly weathered surface oil masses with a specific gravity greater than that of seawater (particularly at cold temperatures), and 4) sedimentation of oil ingested by zooplankton and deposited in faecal pellets. The tendency of suspended sediments to adsorb oil and subsequently sink may be most pronounced in portions of the Beaufort Sea influenced by the Mackenzie River plume since high suspended sediment levels are generally characteristic of these areas during the open water season. Although the concentrations of suspended sediments in the water column would have a marked effect on oil sedimentation rates, a number of other factors may have a significant influence on this phenomenon. In a review of existing literature describing the mechanisms of oil sedimentation, Duval et al. (1978) reported that the major factors affecting oil sedimentation were the mineral composition and particle size of suspended materials, temperature and salinity, the quantity, type and rate of input of oil, sea state and the presence of ice. The authors emphasize that the number of variables involved in this mechanism and our present limited understanding of their interactions hampers determination of the relative volume of oil which may sink in the Beaufort Sea as a result of sedimentation.

The fate of sedimented oil has been a subject of investigation following several past oil spills. Monitoring studies following spills in temperate latitudes indicate that the chemical changes which occur in sedimented oil include dissolution of lower boiling point aromatics and microbial metabolism of alkanes (e.g. Burns and Teal 1979). The higher molecular weight aromatics tend to persist for longer periods and their relative concentrations increase with time. Chemical and microbial breakdown of sedimented oil both occur relatively slowly, and many years are required for the disappearance of some oil fractions in temperate climates. The low temperatures in the Beaufort Sea would further retard microbial degradation and could result in the persistence of sedimented crude oil in a relatively unaltered state for a decade or longer, although sediments carried by the

Mackenzie River would rapidly bury sedimented oil in many parts of the region. The movement of sedimented oil is not well understood, but lateral spreading as well as migration deeper into sediments has been reported in the oil spill case history literature (Sanders 1977). The persistence of oil as well as the potential for recontamination of previously unaffected habitats is an area of concern with respect to benthic species and their predators in the Beaufort Sea.

5.2.2.3 Open Water Subsea Blowout

While some information regarding the behaviour of oil during an actual subsea blowout is available from monitoring programs conducted at the Ixtoc 1 blowout in the Gulf of Mexico, most of the existing data base is the result of various blowout simulations. This section discusses the probable behaviour of oil in combination with natural gas, since geological investigations in the Beaufort Sea suggest that the rate of oil flow during a blowout would be approximately 400 m^3/day, with a gas to oil ratio of approximately 1500:1.

The release of oil and gas under pressure from the seafloor results in several phenomena that differ significantly from a spill at the surface. In the case of a blowout, the oil is dispersed throughout the water column as small droplets which can range in diameter from only a few microns to several millimeters (Topham 1975). Topham (1975) calculated the natural rise velocities of oil droplets as well as mean densities, and concluded that it was unlikely that any coalescence would take place within the rising oil and gas plume. However, his model predicts that most oil droplets would coalesce to form an outward moving slick once they reach the surface, although droplets $<50\ \mu$ in size (estimated to be ~1 percent) may be swept down in the eddy currents which develop at the surface and then may be carried several kilometers away from the blowout during their rise to the surface. The conical to cylindrical shaped plume of oil and gas bubbles rising to the surface may entrain bottom sediments, as observed at Ixtoc 1 and during the blowout simulation of Buist et al. (1981). Blowout simulations with oil and gas in 60 m of water indicate that a concentric ring of waves will form beyond the central plume (Topham 1975). Surface currents will likely move outwards within the ring and inwards beyond the ring to create a natural containment area for some of the oil. Topham (1975) suggests that the radius of this ring would be approximately 15 m for a 15 m deep well blowout and 40 m for a 60 m deep blowout. In some cases, a well blowout may be accompanied by a fire. For example, the Ixtoc 1 blowout resulted in a 50 m diameter fireball directly above the well with small fires occurring outside the major area of combustion.

Dissolution of the water-soluble hydrocarbons would begin as soon as oil from a blowout enters the water column, and once the remainder of the oil reached the surface, it would undergo the weathering processes previously described (Section 5.2.2.1). The rate of dissolution of the soluble fractions decreases exponentially with time, and inversely with the square of the droplet diameter (Leinonen and Mackay 1977, cited in Milne and Smiley 1978). This means that water soluble fractions are lost more rapidly from smaller droplets, although given the rise times and droplet sizes estimated by Topham (1975), the percent dissolution of hydrocarbons before oil from a blowout reaches the surface would still be relatively small.

5.2.2.4 Subsea Blowout Under Ice

The behaviour of oil beneath an ice cover has been the subject of considerable research, both in the laboratory and under first and multi-year ice in the field. Laboratory experiments have been conducted to examine the horizontal transport and trapping of spilled oil beneath ice. For example, Uzuner et al. (1978) estimated the speed of movement of current-driven oil beneath a smooth laboratory ice sheet, while Moir and Lau (1975) studied the containment of oil by ice ridges using vinyl covered plastic to simulate the ice. More recently, Cox and Schultz (1980) described the results of a laboratory study designed to determine the effect of slick thickness as well as large and small roughness features in the under-ice surface on the movement of oil beneath ice. Field investigations of the behaviour of oil beneath ice are described in NORCOR (1975), Comfort and Purves (1980) and Buist et al. (1981). The behaviour of oil under multi-year ice in Hecla and Gripper bays off Melville Island was examined during two spring melt periods by Comfort and Purves (1980), while Buist et al. (1981) investigated oil and gas behaviour during a simulated 30 min subsea blowout under first-year ice off McKinley Bay in the Beaufort Sea. In addition, Lewis (1976) and Milne (1980) provide reviews of available information regarding oil/ice interactions.

Until oil reaches the water-ice interface, the behavioural characteristics of the plume from a subsea blowout during winter would probably be similar to those described earlier for an open water blowout (Section 5.2.2.3). Underwater monitoring (video and echo sounder) completed during a simulated blowout in the Beaufort Sea indicates that the plume of rising gas bubbles with oil droplets and entrained sediment has a conical to cylindrical shape (Buist et al. 1981). The plume created turbulent eddies near the ice/water interface, and this turbulence changed to an outward laminar flow within approximately 20 m from the plume center. However, a distinct wave ring (as documented during open water simulated blowouts) was not observed. The oil droplets rose at velocities proportional to their diameter, and smaller droplets with low rising velocities and a longer residence time in the water column were carried by ambient currents and contacted the under-ice surface up to 350 m from the plume centerline (Buist et al. 1981). These investigators also report that once these oil droplets contacted the ice, they were not advected by either gas flow or water currents (1-3 cm/s) and were quickly encapsulated by the growing ice sheet. The majority of the ~6 m^3 (~38 bbl) oil released during this simulated blowout remained within a 50 m radius directly above the discharge point. Field experiments on the dispersion of oil under growing first year sea ice (NORCOR

1975) indicate that oil under smooth-bottomed ice will form an equilibrium layer approximately 0.5 to 1.0 cm thick. However, the under-ice surface is typically ridged because of differential freezing associated with the insulating effects of snow drifts, and oil or gas will preferentially fill depressions in the under-ice surface. Cox and Schultz (1980) suggest that the actual volume of oil that can be contained by ice roughness is largely determined by the density of the oil and the ambient currents. These authors also investigated the advection of oil under ice by currents and concluded that a critical water velocity is required for oil movement. In the case of a perfectly flat under-ice surface, this velocity is determined by the properties of the oil and generally ranges between 5 and 10 cm/s. The threshold increases to 15 to 25 cm/s for ice with small scale roughness (up to 20 cm ridges) which is typical of newly formed sea ice. Measurements of under-ice currents in nearshore areas of the Beaufort Sea suggest that current velocities are generally less than the threshold levels required to move oil under ice characterized by small scale roughness.

Prior to ice melt and breakup in spring, some oil and gas from a subsea blowout could reach the ice surface, if sufficient gas is trapped in pockets under the ice sheet to cause it to rupture. Topham (1975) indicated that a bubble thickness of 40-55 mm over a 60 to 100 m radius would be required to fracture 0.5 m thick ice, while 2 m thick ice could fracture when the gas bubble was 150 mm thick and 200 m in diameter (Malcolm and Cammaert 1981). However, some researchers suggest that oil and gas would usually spread out under the ice cover rather than fracture it, although some oil could reach the surface of the ice if the blowout occurred in fall before a thick ice sheet formed.

Oil trapped beneath the ice as it freezes would become encapsulated in either pockets or as droplets and would migrate to the ice surface during spring melt. Several studies have documented the passage of oil through ice sheets ranging from first-year landfast ice to multi-year floes (e.g. Comfort and Purves 1980; Buist et al. 1981). Two mechanisms are involved in the transport of oil to the ice surface during spring melt. The first is the simple melting of the ice to expose oil that has been sandwiched within the sheet. The second mechanism is oil migration through brine channels and has been observed in ice sheets of all ages. Brine pockets are formed throughout the ice sheet due to salt exclusion when seawater freezes. In spring, these brine pockets melt before the remainder of the ice due to the freezing point depression effect of the salt, and then become interconnected to form channels through the ice and provide a pathway for oil to reach the ice surface. The oil moves to the surface of the melt water pools on the ice as a result of its buoyancy in water. Oil can travel through first-year ice 150 cm thick over a period of several months by this mechanism. For example, Buist et al. (1981) reported that 80 percent of the volume of oil released in their experimental blowout surfaced in the spring via the combination of these two processes. Comfort and Purves (1980) found that oil from an experimental spill began to surface through multi-year ice (2.5-2.9 m thick) in approximately three months, while continued monitoring suggested that 10 percent of the oil remained in the ice sheet 15 months after its release. Oil which reaches the ice surface by either of these mechanisms will be relatively unweathered, but evaporation of volatile hydrocarbons and cold temperatures would quickly increase its viscosity. Some of this oil could subsequently re-enter the water column during breakup.

Oil on the ice surface tends to spread readily, and is only retarded by irregularities in the ice surface and the viscosity, specific gravity, pour point and surface tension of the oil. The upper ice surface is generally quite porous and the top few centimeters may absorb up to 25 percent of its volume in oil (Fingas et al. 1979). A greater absorption of solar energy occurs when oil is present on the ice surface, and this may increase the rate of ice melt. In some instances, oil spilled on the upper surface of the ice may eventually find its way into the water column through downward seepage in pores and crevices throughout the ice.

5.2.2.5 Oil Spills on the Surface in Ice-Infested Waters

This section summarizes available information on the behaviour of oil in freezing surface waters and open leads. Stringer and Weller (1980) recently reviewed data describing micro-scale behaviour (individual crystal matrices) and large scale transport processes which were collected during research conducted in Alaska and the Canadian Beaufort Sea.

Martin et al. (1978, cited in Stringer and Weller 1980) examined the behaviour of oil in growing ice stirred by waves in a laboratory test tank. An early phase of ice growth is termed "grease" ice when the ice behaves partially as a liquid and partially as a solid. Oil introduced into ice at this phase is transported onto the ice surface to a "dead zone" which marks the transition between solid and liquid behaviour. Field observations confirm these laboratory results and suggest that some oil would be emulsified by the breaking waves and would circulate within the grease ice as well as at the ice front (Martin 1980, cited in Stringer and Weller 1980). Circular pancakes of ice form during a later stage in ice development. Experiments conducted with this form of ice indicate that the oscillating motion of ice pancakes pumps some oil onto the upper surface, while the remainder becomes encapsulated as droplets under the pancakes (Martin et al. 1978, cited in Stringer and Weller 1980).

Oil entering open leads in the ice cover may be dispersed by processes such as lead pumping or pressure ridge formation. Lead pumping was described by Campbell and Martin (1973, cited in Stringer and Weller 1980) and could occur on a small scale in the Beaufort Sea when leads containing oil begin to close. This would compress the surface slick and ultimately result in lateral flow along the ice edge until the oil reaches an equilibrium thickness or is blocked by the ice. Studies summarized by Stringer and Weller (1980) indicate that if a lead containing oil closed completely to form a pressure ridge, some of the oil would be spilled on the surrounding ice and as the ridge formed, the oiled ice would be broken up and incorporated in the ridge. Oil can also be moved by winds and currents in open lead systems and depending on the draft and freeboard of the ice sheet, some oil may be swept under or onto the ice surface.

Oil on surface waters containing ice floes may be dispersed in droplet form (<1 mm in diameter) throughout the brash ice and on the edge of floes. It may also be transported beneath or to the surface of ice floes as a result of the processes previously described for leads and migration through ice. In addition, Stringer and Weller (1980) suggest that oil may be distributed throughout the water column in small droplets by downwelling forces that are present in grease ice plumes which form downwind of freezing icefronts. In both these situations, the weathering processes described earlier will continually change the characteristics of the oil on the water surface.

5.2.3 Environmental Concerns Associated with Crude Oil and Refined Fuel Spills

The accidental release of crude oil or refined petroleum products into the marine environment is one of the greatest potential areas of concern associated with hydrocarbon exploration and production in the Beaufort Sea. As the previous sections indicate, oil released in a major spill could affect virtually all habitats in the marine environment of this region. Although there is an extremely large data base describing the effects of oil on aquatic flora and fauna, it remains virtually impossible to accurately predict the biological effects of oil spills because of the complicated nature of the interactions between oil, water and sediments, as well as the uniqueness of the circumstances surrounding any spill event. A review of the effects of past oil spills was completed by Duval et al. (1981) and is another supporting document to this EIS. The case histories of 100 oil spills were reviewed in this parallel supporting document, and have been extensively utilized in the following sections describing the potential effects of oil on marine resources of the Beaufort Sea. The results of this literature review clearly indicate that for some biological communities, the circumstances surrounding each spill greatly affect the type and magnitude of impacts. Even with a single resource, the effects of oil spills and the duration of damage have been dependent on the type and volume of oil lost, time of year, success of oil containment, methods of cleanup and restoration, and the types of habitats contaminated. However, it should be emphasized that assessment of the potential impacts of oil spills on arctic organisms is hampered by the limited case history data for this region, and the relatively low number of laboratory toxicological investigations that have been conducted with arctic species. The available data base also suffers from the fact that certain ecological communities have been studied more than others. For example, the impacts of oil spills on visible populations or communities such as birds, saltmarshes and intertidal invertebrates are well documented, while the effects of oil on other less visible communities (e.g. plankton, subtidal invertebrates) have not been examined after the majority of spills. Tables 5.2-3 to 5.2-5 from Duval et al. (1981) identify those crude oil, refined fuel and bunker spills which have resulted in documented or anticipated impacts on marine resources, as well as spills where certain communities were examined and no effects reported. Nevertheless, based on available laboratory results and the case histories of spills which have occurred in sub-arctic and temperate environments, a range of effects can be predicted for most arctic species. In general terms, these effects could include:

1. Death through impairment of respiration, feeding or movement by physical coating with oil;
2. Death due to contact with toxic petroleum fractions;
3. Death from thermoregulatory impairment;
4. Sublethal interference with physiological or biochemical processes;
5. Accumulation of hydrocarbons in tissues, thereby rendering animals unpalatable to their predators or reducing their commercial value;
6. Loss of habitat due to oil contamination; and
7. Increased susceptibility to predation as a result of altered behavioural patterns.

The assessment of potential indirect and cumulative impacts of oil spills is more difficult because of the lack of detailed information regarding trophic interrelationships within some arctic communities. However, where information is available, indirect impacts on other trophic levels are identified to the extent possible.

TABLE 5.2-3

CRUDE OIL SPILLS HAVING DOCUMENTED IMPACTS ON MAJOR CATEGORIES OF MARINE RESOURCES (from Duval *et al.* 1981)

SPILL	ESL FILE	Terrestrial Plants	Mangroves	Marshes	Fringe/Surfgrasses	Intertidal Algae	Benthic macrophytes	Phytoplankton	Zooplankton	Benthic Infauna	Benthic Epifauna	Sedentary Intertidal Fauna	Mobile Intertidal Fauna	Demersal/Inshore Fish	Pelagic Fish	Terrestrial Mammals	Marine Mammals	Waterfowl	Alcids	Unspecified Birds
RICHARD C. SAUER	89			*		*						*	*				*			*
AMOCO CADIZ	2	●		●		●			●	●	●	●	●	●				●	●	●
Bay Marchand Blowout	6														*					○
BETELGEUSE	7					*					○	*	*							○
CHRISTOS BITAS	12																●		●	●
Ekofisk Bravo	16													○	○					
Ixtoc I	24											●	●	●						○
JACOB MAERSK	25					●						●	●							○
METULA	26			●		●	*					●	●	○				●		●
OCEAN EAGLE	29										●			●						○
Platform Charlie	30											○		*	*					
Santa Barbara	33				●	●	●	*	*		○	●	●	*	*		○	●		●
TORREY CANYON	36	●		●		●		●			●	●	●	●						
Tarut Bay	37		●			●					●	●	●	●					●	●
T.T. DRUPA	38											●	●							
URQUIOLA	39			●	●						●	●	●	●						
VENPET-VENOIL	42											●	●	●	*					
UNIVERSE LEADER	44	●				●	●				○	●	●		*			○		
CONOCO BRITANNIA	45											●							●	
PAVLA	46					*				*	*		●					●		
M/S JAWACHTA	57															○				
OLYMPIC GAMES	69			○								*						●		
ANDROS PATRIA	71											●								
OLYMPIC ALLIANCE	73											○	○						●	●
CHRYSSI P. GOULANDRIS	74			●		●														
ZOE COLOCOTRONIS	75		●		●					●	●									
WAFRA	76					●					●	●	●		○					●
PANTHER	77											○	○							○
Norco Louisiana	79																			●
MESSINIAKI BERGEN	81																			○
Corpus Christi	82													●						
CORINTHOS	83			○	○													●		
AMOCO YORKTOWN	84			○														●		
California Bay	85			○								○								
ROLLNES	92			*		*						*	*				*			*
ESSO BAYWAY	93			○														○		
ESSO ESSEN	98	○					*	*	*		*	○	●	●						●
ANDRON	99						●					●	●							●
WORLD GLORY	100														●					●
Harbor Island	104		●	●						●										
ANTONIO GRAMSCI	105											○					●			●
Coryton	107					●						●	●							
Florida Keys	108		●	●		●					●		●	●	●					
ARGEA PRIMA	106		●		●	●	●				●	●	●	●						
Bay Marchand	86			*		*						*	*				*			*

● Mortality
○ Contamination
* Examined but no impacts documented

TABLE 5.2-4

REFINED FUEL SPILLS HAVING DOCUMENTED IMPACTS ON
MAJOR CATEGORIES OF MARINE RESOURCES
(from Duval et al. 1981)

SPILL	ESL FILE	Terrestrial Plants	Mangroves	Marshes	Fringe/Surfgrasses	Intertidal Algae	Benthic macrophytes	Phytoplankton	Zooplankton	Benthic Infauna	Benthic Epifauna	Sedentary Intertidal Fauna	Mobile Intertidal Fauna	Demersal/Inshore Fish	Pelagic Fish	Terrestrial Mammals	Marine Mammals	Waterfowl	Alcids	Unspecified Birds
SEALIFT PACIFIC	88			*		*						*	*				*			*
Anacortes	3									*	*	○	●	○				●		●
BOUCHARD 65	8			●		●				●	●	●	●	●						
DONA MARIKA	13					●	●			●	●	●	●							
Dounreay	14					○						●	○					●	●	
East Lamma Ch.	15								●				●	●	●					
ELENI V	17					*						○	○	*	*		○			●
Firth of Forth	18																	●	●	●
FLORIDA	19			●		○				●	●	●	●						●	○
HAMILTON TRADER	21																	●		
IRINI	22					*						●	●							
USAF Fuel Depot	40									●	○									
S.S. WITWATER	43	●	●			●					*	●	●							●
THUNTANK 6	51											●	●							
DEWDALE	52																			
GEMINAR	53																			*
BRITISH MALLARD	58																	●	●	
Cromarty Firth	61			●														●		●
JOS. SIMARD	62										○				○					○
TAMPICO MARU	64						●			●	●	●	●	●						
Deception Bay	65					●				●	●	●	●			●				●
BARGE STCO 225	87			●							*									
NEW YORK	90											●	●					●		●
IMPERIAL BEDFORD	91					*						*	*				*			*
R.C. STONER	101										●			●						

● Mortality
○ Contamination
* Examined but no impacts documented

TABLE 5.2-5

BUNKER FUEL SPILLS HAVING DOCUMENTED IMPACTS ON
MAJOR CATEGORIES OF MARINE RESOURCES
(from Duval *et al.* 1981)

SPILL	ESL FILE	Terrestrial Plants	Mangroves	Marshes	Fringe/Surfgrasses	Intertidal Algae	Benthic macrophytes	Phytoplankton	Zooplankton	Benthic Infauna	Benthic Epifauna	Sedentary Intertidal Fauna	Mobile Intertidal Fauna	Demersal/Inshore Fish	Pelagic Fish	Terrestrial Mammals	Marine Mammals	Waterfowl	Alcids	Unspecified Birds
ADRIAN MAERSK	1											●	●		●					
ARGO MERCHANT	4								●						○		*	○	●	○
ARROW	5			●		●			○	●	*	●	●	●			●			●
BARGE STC-101	11			●						*	*	●	●					●		●
Hackensack	20			●							●			●				●		●
IRISH STARDUST	23			●		●							●							○
Mizushima	27					●		●			●	●			●		●	●		
NEPCO 140	28			●												●		●		●
San Francisco Bay	32				○							●	●					●	●	●
ESSO BERNICIA	35															●	●	●	●	
VANLENE	41					*				*	*	●	●	*			*			○
ERAWAN	56											○	○	●						○
AFRAN ZODIAC	59									*	*	○	○				*	●	●	●
GOLDEN ROBIN	60			●													○			●
KURDISTAN	63										○						●	●	●	●
GEN. M.C. MEIGS	67				●	●	●					●	●							
Oakland Estuary	68																			●
TAMANO	70					●				*	○	●	●					●		●
TSESIS	72					*			●	●	●	●	●	○	○					
PENNENT	78			●																
ATHOS	80																	●		
SANSENINA	94					●			●	●	●	●	●							○
LEE WANG ZIN	95			●								*	*	*		○				○
York River	96			○						○	○		●							*
IRVING WHALE	97																			●
USNS POTOMAC	102								○								○			*
Biesbosch	103			●														●		
SPARTAN LADY	47																			*
TERNSJOE	55																			*

● Mortality
○ Contamination
* Examined but no impacts documented

5.2.4 Effects of Oil on Marine and Marine-Associated Mammals

Exposure of mammals to crude oil and refined petroleum products may result in various physical, physiological and behavioural effects, although the severity of these effects would vary with the degree of oil contamination and species. The documented and anticipated impacts of past refined fuel, bunker fuel and crude oil spills on marine and marine-associated mammals were summarized by Duval et al. (1981), while the major incidents and laboratory studies demonstrating the biological effects of oil on marine mammals were reviewed by Smiley (1980). The results of numerous studies reviewed by the aforementioned authors suggest that the direct effects of oil may include: (1) mortality; (2) plugging of external body orifices and eye irritation; (3) impairment of mobility; (4) a decrease in the insulative capacity of fur; (5) elevated metabolic rate in compensation for thermoregulatory stress, and (6) inhalation of volatile fractions or ingestion of oil during grooming, suckling or feeding, resulting in physiological stress or systemic disorders. In addition, indirect effects of oil on mammals may include spatial and/or temporal displacement from critical or important breeding, feeding or rearing habitats, reduced reproductive success, increased susceptibility to predators or a local reduction in food availability.

In general, refined fuels such as diesel are more acutely toxic to marine flora and fauna than crude oil or bunker fuels because they contain more water soluble, low boiling point and volatile hydrocarbons (Anderson et al. 1974a). On the other hand, refined fuels are less viscous than crude oil or bunker fuels, and therefore have less tendency to cause significant physical fouling of marine or marine-associated mammals. However, volatile components of refined fuels may cause severe eye irritation to marine mammals under certain exposure conditions. In general, bunker fuels have resulted in more pronounced effects on mammals than exposure to unweathered crude (Duval et al. 1981).

The six major species of marine mammals that could be affected by a crude oil spill or blowout or a refined fuel spill in the Beaufort Sea region are the ringed seal, bearded seal, bowhead whale, white whale, polar bear and Arctic fox. However, the potential effects of these environmental emergencies would be highly dependent on the species and general health, age and sex of individuals, as well as on the circumstances surrounding the spill such as amount and type of oil spilled, time of year, success of the clean-up operation, and the duration and spatial extent of contamination.

It is not known if marine mammals are capable of detection and avoidance of surface slicks or sub-surface oil. Although the reports from investigators of past incidents (e.g. the ARROW and ESSO BERNICIA spills, and the Santa Barbara blowout) are not conclusive, several authors have suggested that seals and sea otters do not avoid oil in some circumstances (Brownell and LeBoeuf 1971; Muller-Willie 1974; Davis and Anderson 1976). It is possible that some species of marine mammals may even be attracted to a spill site to feed on fish and other organisms disabled or killed by the oil (Geraci and St. Aubin 1980). However, Barabash-Nikiforov et al. (1947) reported that Japanese poachers used petroleum products to repel sea otters from shore rocks, and there is also evidence that gray seals actively avoided the bunker C lost from the tanker ARROW in Nova Scotia (Butler et al. 1974). Seals and sea otters are probably the marine mammals most vulnerable to oil spills since they depend on substrates during pupping, moulting and mating, and therefore may repeatedly contact contaminated shores (Davis and Anderson 1976). On the other hand, oil-contaminated cetaceans have not been reported following oil spills. Although this may indicate that whales can detect and avoid crude oil slicks or oil dispersed in the water column, cetacean skin is smooth and does not accumulate oil, and it is therefore difficult to determine if a whale has been exposed to oil (Geraci and Smith 1977). In addition, oiled whales may go unnoticed since they do not come ashore like pinnipeds and sea otters.

5.2.4.1 Mortality

During the last decade, the media and some scientific review articles have reported occasional marine mammal mortality following oil spills, but there is seldom data to unequivocably link deaths with the presence of oil because autopsies were not completed and baseline (pre-spill) data on the abundance and distribution of affected species have not been available. Nevertheless, the case history literature describes seal and sea lion mortality following some crude oil spills, although unlike bunker fuel spills, the reports of mortality are relatively limited. Of 44 crude oil spills reviewed by Duval et al. (1981), mortality of seals was confirmed following only 2 incidents (Table 5.2-3). Seal pups were killed as a result of the CHRISTOS BITAS spill off Wales (Bourne 1979a), while one dead seal was reported after the ANTONIO GRAMSCI incident in Latria (Anon 1979a,b). The latter event was the only spill north of 55°N discussed by Duval et al. (1981) that resulted in detectable effects on marine mammals.

Mortality of northern elephant seals, California sea lions, and northern fur seals was reported following the Santa Barbara blowout in 1969, although critical reviewers were unable to directly attribute this mortality to the presence of oil (Simpson and Gilmartin 1970; Brownell and Le Boeuf 1971; Le Boeuf 1971). Monitoring programs initiated following this event indicated that: (1) there was no significant increase in mortality of oiled seals during the year after the blowout; (2) sea lions gave birth to normal pups after being oiled, and (3) whales migrated through the area with no apparent adverse effects (Holmes 1969; Straughan 1970).

Mortality of seals following bunker spills was reported in the Gulf of St. Lawrence (Warner 1969), in Chedabucto Bay, Nova Scotia following the ARROW spill (Anon 1970) and along the south coast of Nova Scotia after the KURDISTAN spill (Marston, pers. comm., cited in Geraci and St. Aubin 1980), while the ESSO BERNICIA bunker spill resulted in the loss of 20 to 30 sea otters (Bourne 1979b). However, direct evidence linking the deaths with the presence of oil was again lacking in most cases.

As indicated earlier, refined fuels would have less tendency to cause physical fouling of marine mammals, and this is largely substantiated by the case history literature. Of the 26 refined fuel spills examined by Duval et al. (1981), contamination (but not mortality) of marine mammals was reported following only one incident (Table 5.2-4). Wood (1979) observed 20 oiled seals after the collision of the ELENI V and ROSALINE which resulted in the loss of >24,000 barrels of heavy fuel oil.

Mortality of marine mammals following exposure to oil in laboratory studies was described by Smith and Geraci (1975) and Engelhardt (1981). Smith and Geraci (1975) exposed 3 ringed seals to a 1 cm thick layer of Norman Wells crude oil in a laboratory holding pen, and all individuals died within 71 minutes. However, since the test animals showed signs of stress associated with captivity, it was believed that this may have intensified the effects of oil exposure (Geraci and Smith 1976). This and other investigations also suggest that seal mortality may be greater if individuals contact oil during periods of natural stress such as poor feeding conditions, heavy ice conditions, moulting, or when individuals are extensively parasitized (Kenyon 1975; Geraci and Smith 1976; Williams 1978).

In another laboratory study, Engelhardt (1981) lost 2 of 3 polar bears exposed for 15-50 min to a 1 cm slick of Midale crude. The bears died 4-5 weeks after the exposure as a result of severe physiological and behavioural disorders (see Section 5.2.4.4).

5.2.4.2 Physical and Noxious Effects

Physical and noxious effects of oil on seals and sea otters have been described following laboratory investigations and in the oil spill case history literature. These effects include impairment of mobility, eye and nostril irritation, and plugging of respiratory openings. However, the occurrence and extent of these effects vary between species and with the circumstances surrounding the oil exposure.

Warner (1969) reported that harp seals heavily coated with bunker C in the Gulf of St. Lawrence had difficulty swimming and probably died of exhaustion, while Anon (1969) reported that elephant seal pups encrusted with crude oil and debris after the Santa Barbara blowout had difficulty opening their eyes. In addition, about 24 seals may have died of suffocation when their nose and mouth openings were plugged with bunker C after the grounding of the ARROW in Chedabucto Bay, N.S. (Anon 1970). On the other hand, Geraci and Smith (1976) indicated that ringed seals exposed to a 1 cm slick of fresh Norman Wells crude in a field holding pen in the Arctic for 24 h showed no evidence of mechanical damage during exposure, and that their coats showed virtually no visible evidence of oil immersion after 3 or 4 days in a clean-water holding pen. This may be attributed to the fact that Norman Wells crude is relatively light, highly volatile and has a low viscosity, and is therefore unlikely to cause extensive physical fouling in an unweathered state as is bunker C.

However, the 6 seals that were exposed to the crude oil during the above investigation of Geraci and Smith (1976) showed severe eye irritation. Seven or 8 minutes after the addition of the crude oil to the experimental pen, one seal began to lacrimate and blink excessively, while eye irritation became apparent in the other seals shortly thereafter. Within 4 h, all seals were lacrimating and squinting, and at the end of the 24 h oil exposure, all seals had severe conjunctivitis and swollen nictating membranes. In addition, there was evidence of corneal erosions or ulcers in four of the seals. However, within 3 h after being placed in a clean seawater pen, most squinting and lacrimation had subsided, and the eyes showed no signs of irritation by 20 hours. Geraci and Smith (1976) suggest that at least some of the eye damage appeared to have been caused by volatile components of the Norman Wells crude, and continued exposure to oil may have resulted in more severe and possibly permanent eye disorders.

Whales exposed to crude oil or refined fuels may be as sensitive to eye irritation and plugging of body openings as seals (Geraci and Smith 1977), although this has not been documented. Geraci and St. Aubin (1980) suggest that clogging of the cetacean blowhole is unlikely since it has evolved to prevent inhalation of water and would not discriminate oil. On the other hand, the irritating properties of oil on ocular and periocular tissues has been well documented in seals (Geraci and Smith 1976), and the same effects may occur in other species of marine mammals. The outer layers of cetacean epidermis are not keratinized, and consist of live and metabolically active cells. The potential effects of the toxic fractions of oil on the unique physiological and metabolical characteristics of the cetacean skin are unknown, but are currently under study at the University of Guelph.

Floating oil slicks may also foul the baleen feeding mechanism or damage the structural integrity of baleen plates of surface feeding whales. The right whales such as the bowhead whale are particularly susceptible to this form of contamination since they skim the surface waters for planktonic organisms with their baleen plates partially exposed (Watkins and Schevill 1976). Weathered crude oil and heavy refined products (e.g. bunker C) would be most likely to damage baleen plates (Geraci and St. Aubin 1980). In laboratory experiments, crude oil has been shown to reduce the filtration efficiency of baleen by up to 10 percent (Braithwaite 1980).

5.2.4.3 Effects on Thermoregulation and Basal Metabolism

Marine mammals that rely primarily on hair or fur for thermal insulation are considered most susceptible to thermoregulatory stress as a result of oil contact (Geraci and St. Aubin 1980), and include sea otters, fur seals and polar bears. Sea otters and fur seals (Otariids) have a waterproof fur coat that holds a layer of air against the skin for insulation. Oil contamination can result in matting, clumping and derangement of the fur and a subsequent loss of waterproofing and buoyancy, and may eventually lead to chilling, hypothermia, exhaustion and death (Kenyon 1975; Kooyman *et al*. 1977;

Williams 1978). The period that an oiled marine mammal can maintain its core temperature depends on several factors such as the extent of contamination, degree of exposure to cold water and air, life history stage and the general condition of the affected individual.

Experimental oiling of sea otter pup and subadult fur seal pelts doubled the thermal conductance of the coat (Kooyman et al. 1977). Similarly, McEwan et al. (1974) demonstrated that the thermal conductance of oiled muskrat pelts was increased by 122 percent by a heavy coating of oil. Kooyman et al. (1977) also concluded that oil-fouled sea otters and fur seals would be unable to endure prolonged immersion in cold water if oil was not soon removed from the pelts through grooming.

The decrease in insulative capacity of the fur caused by oil-fouling leads to a subsequent increase in basal metabolism in marine mammals that rely on fur for insulation. For example, light oiling of 30 to 40 percent of the coat of fur seals caused a 50 percent increase in basal heat production (Kooyman et al. 1977), while oiled muskrats had metabolic rates 20 percent above normal 3 days after contact with oil (McEwan et al. 1974).

On the other hand, polar bears, earless (phocid) seals and sea lions rely on both guard hairs and/or wooly underfur and subcutaneous blubber for insulation (Scholander et al. 1950; Irving and Hart 1957; Bryden 1964; Frisch et al. 1974). The sensitivity of these species to oil is highly dependent on the significance of the fur to total unregulated insulation. In general, the insulative contribution of the fur is relatively small for the strictly aquatic mammals but this may vary depending on species, age, general health and season (Irving and Hart 1957; Frisch et al. 1974; Geraci and Smith 1976; Kooyman et al. 1976, 1977). For example, oiling of pelts of ringed seals, bearded seals and California sea lions did not change their thermal conductance (Øritsland 1975; Kooyman et al. 1977). In addition, studies conducted by Geraci and Smith (1976) indicated that the body temperatures of ringed seals immersed in Norman Wells crude for 24 h and 3 to 4 wk old harp seal pups coated with crude for 7 days remained stable and within the normal range. In vitro examination of the pelts at the end of the experiments also indicated that thermal conductance had not changed as a result of oil contamination.

Although light oiling of adult seals and sea lions probably does not significantly alter the underwater thermal conductance of the pelt, some species may be very susceptible to oil contamination during certain periods of their life history such as the moult. For example, some newborn phocids (e.g. harp and ringed seals) rely on white lanugo ("whitecoats") for protection against wind chills until a sufficient blubber layer develops and homeothermy stabilizes during the first weeks of life (Øritsland and Ronald 1973). Oiling of pups at this age could result in loss of insulative capacity, chilling and hypothermia (Smiley 1980). Geraci and Smith (1976) found that core temperatures in 2 to 4 week old whitecoat harp seals with a 2.5 to 5.0 cm layer of blubber were not affected by a brushed-on coating of crude oil. However, the effect of oil on thermoregulation in pups between birth and deposition of blubber, when they may be more vulnerable, has not been documented.

Fur also acts as a wind barrier for adults, and may be particularly important during the annual spring moult and obligatory haulout of ringed seals (Øritsland 1975; Geraci and Smith 1976). However, Øritsland (1975) demonstrated that Norman Wells crude did not disrupt the insulative capacity of ringed seal pelts at varying wind speeds. Since any coating of dark material increases the radiation transmittance of light-coloured fur (Øritsland 1975), changes in solar heating of the skin of hauled-out seals may occur, although the biological and ecological significance of such changes have not been documented.

The effects of oil on thermoregulation in polar bears would differ from effects on the more strictly aquatic groups since bears rely more heavily on fur for insulation than seals and sea lions (Frisch et al. 1974). Schweinsburg et al. (1977) suggested that oiling of the fur would cause thermoregulatory stress in polar bears, although this has not been confirmed. However, Engelhardt (1981) exposed 3 polar bears to a simulated oil slick of 1 cm of Midale crude for 15 to 50 min and found that the fur became deeply coated with oil. Vigorous grooming activities of the bears resulted in the spreading of oil over larger areas and deeper into the fur. In addition, ingestion of oil occurred for at least 4 weeks after exposure (Section 5.2.4.4), despite the fact that the bears had been cleaned and the pelts attained a light colour 17 days after the oil exposure. Persistence of oil in the fur resulted in a prolonged exposure to petroleum hydrocarbons and the potential for concomitant thermoregulatory stress.

Cetaceans and walruses are probably the least susceptible marine mammals to thermoregulatory stress as a result of oil contamination because they rely on thick skin and/or subcutaneous fat or blubber for insulation rather than fur or hair. However, it is not known if oil contact affects other thermoregulatory mechanisms in these mammals such as peripheral blood circulation and tissue cooling.

5.2.4.4 Effects of Inhalation and Ingestion

Marine mammals exposed to an oil spill may inhale volatile petroleum hydrocarbons or may incidentally ingest surface oil during feeding (e.g. baleen whales), grooming (e.g. sea otters), suckling, or indirectly through consumption of contaminated prey. Other potential pathways for uptake of petroleum hydrocarbons include the skin and respiratory surfaces.

The effects of ingested petroleum hydrocarbons on phocid seals were examined by Geraci and Smith (1976). During a 24 h exposure to Norman Wells crude oil, ringed seals absorbed petroleum hydrocarbons in blubber, brain, muscle, kidney and liver tissues. However, the highest hydrocarbon levels were detected in the bile and urine, suggesting that the kidney and liver are principle sites of detoxification and excretion (Engelhardt et al. 1977). There was also histological evidence of kidney damage in at least one seal. Harp seal pups fed up to 75 ml of crude oil (far in excess of that calculated to have been swallowed by the above ringed seals) showed no clinical, biochemical or morphological evidence of tissue damage (Geraci and Smith 1976), while adult ringed seals fed 5 mL of Norman Wells crude per day for 5 days were also not irreversibly harmed. The amounts of oil administered in the aforementioned study are considered representative of the upper limit of what a seal would likely ingest from live, oil-tainted prey (Smiley 1980). Although the results of this study cannot be extrapolated to other marine mammals or larger quantities of ingested oil, the authors suggest that relatively large doses of oil ingested over a short period probably do not cause any acute organ or tissue damage in ringed and harp seals (Geraci and St. Aubin 1980). However, longer exposures to oil would probably result in increased hydrocarbon levels in tissues, particularly in blubber (Kooyman et al. 1976).

Geraci and Smith (1976) also calculated that ringed seals exposed to Norman Wells crude for 24 h absorbed some volatile hydrocarbons through the respiratory tract. Nevertheless, lung damage was not apparent even though the experimental holding pen resulted in a more concentrated exposure to volatile fractions than would normally be encountered during an actual marine oil spill. The effects of prolonged inhalation of volatile petroleum hydrocarbons have not been documented, although these fractions are rapidly lost due to evaporation, minimizing the potential risk of long-term inhalation by marine mammals.

The potential uptake and accumulation of petroleum hydrocarbons by whales and other marine mammals that feed in benthic habitats (e.g. bearded seals) have not been documented.

Polar bears are susceptible to adverse effects associated with ingestion of petroleum hydrocarbons either through grooming of oiled fur or consumption of oil-contaminated prey. Polar bears prey primarily on ringed seals throughout their range (Stirling and McEwan 1975) and ringed seals have been shown to accumulate petroleum hydrocarbons (Engelhardt et al. 1977; Engelhardt 1978). Schweinsburg et al. (1977) suggested that bears would prey on the oiled carcasses of seals or other animals, and may be subject to a toxic load associated with the accumulation of petroleum hydrocarbons. In addition, polar bears could contact a floating slick since they travel across the sea-ice and traverse open water leads in pursuit of food (Øritsland 1969).

Engelhardt (1981) exposed 3 polar bears to a 1 cm slick of Midale crude for 15 to 50 min to demonstrate the clinical effects of petroleum hydrocarbon exposure on this species. The bears became heavily contaminated with oil, and vigorous grooming resulted in extensive uptake of hydrocarbons from the gut for a period of at least 4 weeks. Petroleum residues were accumulated in bone marrow, brain and kidney, and were initially at high levels in the blood plasma. The urine and bile were the major excretion routes, and residues continued to occur in the urine for 5 weeks after the exposure. The major clinical toxic effects which led to the death of 2 of the 3 bears included erythropoietic dysfunction and renal abnormalities.

5.2.4.5 Indirect Effects of Oil

Indirect effects of oil exposure on marine mammals may include spatial or temporal displacement from critical or important habitats, reduced reproductive success, increased susceptibility to predators, local reduction in food supplies or changes in normal behavioural responses. In general, these potential effects are poorly documented since they can only be studied under experimental field or actual spill situations. For example, Davis and Anderson (1976) reported reduced growth rates in gray seal pups contaminated by a floating oil slick off southwest Wales, but were not able to detect changes in nursing behaviour. At the same time, cows were able to locate their pups regardless of oiling. Monitoring programs initiated after the 1969 Santa Barbara blowout also indicated that sea lions gave birth to normal pups after being contaminated with oil (Holmes 1969; Straughan 1970).

Smith and Geraci (1976) exposed 6 ringed seals to a 1 cm layer of Norman Wells crude oil in a field holding pen for 24 h. During the oil exposure, all of the seals showed varying degrees of arching of the back when at the surface, a behaviour that was not observed in the control group nor in the experimental group prior to the introduction of the oil. However, there was no behavioural or physical evidence of oil contamination 4 days after the exposure. Although some behavioural responses of pinnipeds to oil exposure have been noted during laboratory investigations, potential changes in feeding, diving ability, cow-pup interactions, herd organization and haul-out behaviour in natural populations exposed to oil remain unknown.

5.2.4.6 Summary of Concerns

Laboratory investigations and the oil spill case history literature both suggest that the degree of potential concern regarding adverse effects of crude oil spills and blowouts or refined fuel spills on marine mammals of the Beaufort Sea region would vary with species and types of habitats contaminated. Since mortality of whales has not been reported following past spills and most sublethal effects would likely be relatively short-term, the degree of concern associated with effects of oil spills on the regional populations of bowhead and white whales is expected to be NEGLIGIBLE to MINOR. On the other hand, the potential adverse impacts of oil on seals has been documented in both laboratory investigations (Smiley 1980) and during studies conducted following actual spills (Duval *et al.* 1981), and as a result, the degree of concern regarding adverse effects of oil spills on regional populations of ringed and bearded seals is considered MODERATE. In a similar manner, mortality of polar bears following ingestion of oil during grooming has been recently observed (Engelhardt 1981), and the potential for this and other direct and indirect effects of oil spills on this species suggests that the degree of concern would vary from MINOR to MODERATE, depending on the habitats and number of individuals contaminated. Although the effects of oil on the Arctic fox have not been documented, the degree of concern regarding potential effects of spills or blowouts on this species is expected to be MINOR because individuals would likely only be exposed to oil through ingestion of contaminated prey.

5.2.5 Effects of Oil on Birds

It is generally agreed that birds are the most frequently and seriously affected resource following oil spills in marine environments, and suffer from a wide range of direct physical and physiological effects following oil exposure. Indirect effects of oil spills which may also have serious implications to marine bird populations include reduced reproductive success, increased susceptibility to predation and local reductions in food availability. The species of birds likely to be most seriously affected by an oil spill or blowout in the marine environment are the highly aquatic species that dive and forage at sea. In the Canadian Arctic, birds in this category include alcids (murres, guillemots, dovekies, puffins), diving ducks (oldsquaws, eiders), loons, fulmars, cormorants, gulls, shearwaters, kittiwakes and phalaropes. In addition, birds which congregate in nearshore marine areas and river deltas during part of their annual cycle such as geese, swans, shorebirds and diving and dabbling ducks may also be exposed to oil in nearshore environments.

The documented and anticipated impacts of past refined fuel, bunker and crude oil spills on marine bird populations were summarized by Duval et al. (1981), while laboratory investigations demonstrating the effects of oil on birds were reviewed by Brown (1980). In general, laboratory and field studies suggest that oil type is not a major factor in determining the degree of impact on bird populations (Duval et al. 1981), even though the chemical composition of crude oil and refined petroleum products are substantially different. The degree of impact of past crude oil and refined fuel spills on marine birds has varied to a large extent with the circumstances surrounding each event, as well as the distribution and abundance of birds within the contaminated area. The cumulative effects of chronic releases of petroleum hydrocarbons on birds in the marine environment remain largely unknown.

In general, the volume of oil lost has also not been a major factor determining the degree of impact of oil spills on marine bird populations. For example, insignificant bird mortality followed some large crude oil spills, such as the 3 million barrels of oil released during the Ixtoc I blowout (Ross et al. 1979), while greater mortality has resulted from smaller incidents such as the 7 barrel fuel oil spill in Firth of Forth, Scotland which killed 740 birds and contaminated another 1400 (Campbell et al. 1978), and the 5900 barrel bunker spill from the BARGE STC-101 in Virginia which resulted in the loss of up to 50,000 birds (Roland et al. 1979). The most serious past spills in terms of mortality and/or long-term impacts on bird populations are indicated in Table 5.2-6. Reported bird losses following these incidents may be gross underestimates of the actual mortality because of: (1) the logistics associated with counting carcasses because some sink (Hope-Jones et al. 1970), are lost to predators, or are carried offshore by prevailing winds and currents; (2) the difficulty estimating the extent of the contaminated area, the distribution of birds therein, and the probable extent of mortality, and (3) problems associated with delayed mortality of birds as a result of the incident (see Duval et al. 1981).

TABLE 5.2-6

PAST OIL SPILLS CAUSING EXTENSIVE MORTALITY AND/OR
LONG-TERM IMPACTS ON BIRD POPULATIONS
(Data summarized from Holmes and Cronshaw 1977; Duval et al. 1981)

Incident	Spillage	Mortality	Species
San Francisco Bay USA March 1937	crude oil 72,000 bbls	10,000	murre grebe scoter
Howacht Bay Baltic Sea January 1953	oil residues 4000 bbls	10,000	eider merganser scoter
GERD MAERSK Germany January 1953	crude oil 64,000 bbls	275,000	scoter
Seagate, Washington USA September 1956	bunker C	6000	scoter guillemot
Gotland Sweden 1962	no record	30,000	long-tail duck
TORREY CANYON England March 1967	crude oil 860,000 bbls	30,000	guillemot* razorbill*
ESSO ESSEN South Africa April 1968	crude oil 30,000 bbls	1250	penguin* gannet
Santa Barbara USA January 1969	crude oil 80,000 bbls	3600	western grebe loon, scoter cormorant pelican merganser
N. Zeeland Denmark February 1969	no record	10,000	eider common scoter

TABLE 5.2-6 (cont'd)

Incident	Spillage	Mortality	Species
Terschelling Holland February 1969	crude oil	30,000-35,000	eider common scoter
HAMILTON TRADER England April 1969	fuel oil 4000-5600 bbls	4400-10,000	guillemot*
PAVLA Finland May 1969	crude oil 1200 bbls	3000-3500	eider long-tail duck
NE Britain January 1970	fuel oil 8000 bbls	50,000	sea duck auk
East Jutland Denmark February 1970	no record	12,000	eider common scoter velvet scoter
DELIAN APOLLON USA February 1970	bunker C 2400-2800 bbls	9000	no record
ARROW Canada February 1970	bunker C 68,000 bbls	12,000-50,000	sea duck, auk alcid, eider
Kodiak Island USA February 1970	tanker ballast	10,000	alcid, sea duck gull kittiwakes
Biesbosch Holland December 1970	heavy fuel oil 77,000 bbls	>2000	swan, geese mallard coot
South Kattegat December 1970	no record	15,000	eider scoter
San Francisco Bay USA January 1971	bunker C 20,000 bbls	7000	western grebe scoters common murre loon

TABLE 5.2-6 (cont'd)

Incident	Spillage	Mortality	Species
Anacortes USA April 1971	No. 2 fuel 4600 bbls	1000-10,000	black brant duck, loon cormorant grebe merganser
Jutland March 1972	no record	30,000	eider scoter
Danish Waddenzee Denmark December 1972	no record	30,000	eider common scoter
AFRAN ZODIAC Ireland January 1975	bunker C 2700 bbls	no report	seagull, auk gannet cormorant swan
Cromarty Firth Scotland February 1974	fuel oil 7 bbls	>30	mute swan* goldeneye tufted duck scaup
METULA Chile August 1974	crude oil 361,000 bbls	3000-4000	penguin cormorant, tern albatross petrel, duck shorebirds
CORINTHOS USA January 1975	crude oil 309,500 bbls	2500	ruddy duck scaup
BARGE STC-101 USA February 1976	bunker C 5900 bbls	10,000-50,000	horned grebe* oldsquaw*
AMOCO CADIZ France March 1977	crude oil 1,940,000 bbls	>1100	puffin razorbill guillemot

TABLE 5.2-6 (cont'd)

Incident	Spillage	Mortality	Species
Firth of Forth Scotland February 1978	fuel oil 7 bbls	740	great crested grebe*, scaup pochard, eider guillemot
CHRISTOS BITAS Wales October 1978	crude oil 20,000 bbls	2400	guillemot razorbill auk, gannet
ESSO BERNICIA Scotland December 1978	bunker C 8800 bbls	800-3000	black guillemot long tail duck great northern diver*
KURDISTAN Canada March 1979	bunker C 63,200 bbls	7000	dovekie, murre eider, gull crow

*Long-term impacts were anticipated or concern was expressed regarding the integrity of regional populations.

The field data from actual spills are also difficult to interpret because the circumstances surrounding events are rarely similar. In addition, crude oils and refined petroleum products differ both in terms of their chemical composition and the severity of their initial biological effects on birds. For example, Kuwait, Louisana and Cook Inlet crudes all produced more severe effects on osmoregulatory function of ducklings in the laboratory than Alaskan North Slope or Utah crudes (Holmes and Cronshaw 1977). These authors suggest that under similar circumstances, refined petroleum products may be somewhat more hazardous to birds than crude oil, while Grau et al. (1977) also noticed this trend during studies of egg production by quail exposed to bunker C, No. 2 fuel oil and crude oil. Holmes and Cronshaw (1977) reported that ducks tolerated a higher concentration of crude oil in their diet than No. 2 fuel oil before feeding was suppressed. Duval et al. (1981) suggest that since the lighter refined fuels contain a relatively high proportion of toxic aromatic hydrocarbons, they may cause greater mortality through ingestion. On the other hand, crude oils or weathered fuel oils may have more pronounced effects on thermoregulation and buoyancy than lighter or unweathered refined fuels (Duval et al. 1981). Comparison of field and laboratory data is further complicated by the fact that most laboratory experiments are completed with unweathered oil, whereas birds may not contact oil until it has substantially weathered following an actual spill. Nevertheless, birds that contact petroleum hydrocarbons usually suffer one or more of the following physical, thermal, systemic, pathological or indirect effects.

5.2.5.1 Physical and Thermal Effects

The feathers of a bird contaminated with oil initially become matted and the feather barbules deranged (Hartung 1967). Subsequently, the feathers lose their waterproofing capacity because individual feathers are wettable (Newcastle, University of 1972), as well as their insulative and buoyancy capacity after air entrapped within the plumage is reduced or eliminated (Holmes and Cronshaw 1977). Water is absorbed by the feathers, and the cold air and water which directly contact the skin cause a marked increase in the rate of heat loss. In addition, the reduced buoyancy associated with waterlogging can impair or prevent foraging flights, hinder swimming and diving (Holmes and Cronshaw 1977) or cause some birds to drown (Clark 1968).

Oiled birds have greater energy requirements than uncontaminated individuals because a higher metabolic rate is necessary to maintain the normal body temperature (Hartung 1967; McEwan and Koelink 1973; Lambert and Peakall 1981). For example, McEwan and Koelink (1973) demonstrated that heavily contaminated mallard and scaup increased their basal heat production by at least 30 percent. Hartung (1967) suggested that a heavily oiled mallard may be under the same temperature stress at +15°C as an unoiled bird is at -26°C. Ducks contaminated with oil may be able to withstand cold air temperatures temporarily, but must have and liberate substantial fat reserves to survive for longer periods (Hartung 1967). Lambert and Peakall (1981) reported that mallard ducks exposed to crude oil, or crude oil dispersed with Corexit 9527, for 1 hr had basal metabolic rates 10.6 percent and 22.3 percent higher than pre-exposure rates, respectively. The increased metabolic activity of both groups persisted for at least 2 weeks after the oil exposure, and may have been the result of direct effects on the plumage and/or metabolism of ingested oil.

Little information is available to determine the probability that an oiled seabird will survive, or to predict the amount of oil required on the feathers to break down the waterproofing and insulation capacities. Under experimental conditions where food is provided and the temperature is moderate, oiled birds often survive for prolonged periods and may gradually recover from the oil exposure. For example, Hartung (1967) found that the metabolic rate of oiled ducks returned to normal a few days after contamination. The plumage of contaminated birds may require a few months to be thoroughly cleaned through preening (Lambert and Peakall 1981), although Birkhead et al. (1973) found that oiled common murres, razorbills and great black-backed gulls in a Welsh breeding colony cleaned themselves in about two weeks. The cleaning process may be accelerated if the contaminated bird undergoes a moult. However, under severe environmental conditions such as those which may occur in the Arctic, the probability of survival will likely be reduced because thermoregulatory breakdown would probably occur sooner than the oil could be adequately preened from the feathers (Brown 1980).

5.2.5.2 Systemic and Pathological Effects

Oiled birds may ingest appreciable amounts of petroleum hydrocarbons during preening or consumption of oil-contaminated water or food, and this may subsequently result in numerous systemic and pathological effects. The combination of direct and indirect effects of oil ingestion which follow the previously discussed physical and thermal effects (Section 5.2.5.1) are usually fatal (Brown 1980), although the susceptibility of different species and age classes of birds to oil probably varies.

Bourne (1968) observed that gulls and murres began to preen their oiled plumage immediately after contamination, while Hartung and Hunt (1966) reported that ducks preened about 50 percent of the oil from their feathers within 8 days of exposure and ingested most of it. Hartung (1965) similarly reported that a mallard ingested 2 g of oil through preening during the first three days following exposure to a total of 6 g of oil. Holmes and Cronshaw (1977) indicated that ducks consumed normal amounts of food when it was contaminated with 3 ml crude oil/100 g, while ducklings ingested normal quantities of food containing up to 20 percent crude oil. The latter study suggests that at least juveniles of these species may be more susceptible to systemic and pathological effects associated with oil ingestion.

It is generally assumed that the ingestion of petroleum hydrocarbons results in the gross abnormalities observed during autopsies of birds killed in past spills, but this is not well documented and difficult to consistently substantiate through oral-dosing experiments in the laboratory (Brown 1980). The systemic and pathological effects of oil ingestion on birds that have been reported following one or more investigations include lipid pneumonia, osmoregulatory impairment, gastrointestinal irritation, diarrhea, anemia, hyperplasia of the adrenal cortex, and renal, pancreatic and liver disorders (Hartung and Hunt 1966; Holmes and Cronshaw 1977; Miller _et al_. 1978a,b).

Miller _et al_. (1978b) found, during autopsies conducted 9 days after oil exposure, that juvenile herring gulls given a single 0.2 ml dose of crude oil showed significant hypertrophy of the nasal gland, liver and in some cases adrenals, as well as changes in intestinal tissue morphology. On the other hand, Holmes and Cronshaw (1977) found no gross abnormalities in adult mallards that received a daily dose of 5 ml crude oil for 3 months, although some adrenal hypertrophy was evident. Other experiments have shown that mallard ducks survived when given a daily dose of crude oil, but died off rapidly when subjected to an abrupt drop in air temperature or change in salinity of drinking water (Brown 1980). Ducks, chickens and herring gulls given daily doses of crude oil or No. 2 fuel oil (up to 20 ml/kg/day) for 8 to 15 days survived and in some cases grew (Holmes and Cronshaw 1977; Gorman and Simms 1978). However, these dosages may only be tolerable under benign environmental conditions since the toxicity of oil increases when ducks are at temperatures ranging from -10^{o}C to 5^{o}C (Hartung and Hunt 1966; Holmes and Cronshaw 1977). Hartung and Hunt (1966) report that the amount of oil that a duck ingests daily while preening oil from its plumage is likely comparable to the lethal daily intakes under stressful conditions.

Other studies have shown that small doses (e.g. 0.5 ml) of crude oil cause osmotic imbalance and dehydration in Leach's storm petrel (Butler et al. 1979), mallard (Crocker et al. 1974, 1975), juvenile herring gulls and black guillemots (Miller et al. 1978 a,b). The transfer of water and sodium ions across the intestinal mucosa and excretion of sodium through the nasal gland are all inhibited by ingested petroleum hydrocarbons. Weathered and chemically dispersed crude oil appear to have a more pronounced effect on osmoregulation by birds than unweathered oil (Brown 1980). In addition, Miller et al. (1978 a,b) report that nutrient transfer in juvenile herring gulls and black guillemots is inhibited by ingested oil, although Gorman and Simms (1978) and Holmes and Cronshaw (1977) were unable to demonstrate this phenomenon in juvenile herring gulls and mallard ducks, respectively.

5.2.5.3 Indirect Effects of Oil

Oil spills may also result in indirect effects on birds which can be as serious as the aforementioned direct physical and systemic effects. Indirect effects of spills on reproductive success of oil-exposed birds are well documented and include reduced egg production, hatching success and growth rates of juveniles. In general, the amount of oil required to reduce the reproductive success of birds under laboratory conditions is in the same order of magnitude as the amount contaminated birds may ingest while preening (Hartung 1965). Other indirect effects of marine oilspills on bird populations can only be assessed in a field situation, but may include increased susceptibility to predators, decreased abundance of food sources, decreased foraging efficiency, and spatial or temporal displacement from important or critical habitats.

Holmes et al. (1978) reported that mallards fed food that was contaminated with 3 percent crude oil showed reduced or no egg laying, reduced fertilization and thinning of eggshells. Grau et al. (1977) found that although oral doses of Kuwait and South Louisiana crude oils had less severe effects on quail egg production than bunker C or No. 2 fuel oil, 800 mg of crude temporarily reduced egg production and hatching success. A similar dose of Prudhoe Bay and Cook Inlet crudes did not reduce hatchability. Eggs produced by female quail, chickens and Canada geese given doses of Kuwait crude, Louisiana crude and No. 2 fuel oil had yolk abnormalities (Grau et al. 1977; Holmes and Cronshaw 1977), and although the biological significance of this response remains unknown, it was most pronounced in the eggs of females treated with No. 2 fuel oil.

Mineral oil (0.005 ml) covering 5 percent of the surface area of mallard eggs caused a significant decline in hatchability (Albers 1977), while eggs incubated by a contaminated great black-backed gull in the field failed to hatch (Birkhead et al. 1973). Patten and Patten (1976, 1977) found that North Slope crude applied to gull eggs caused much greater embryonic mortality than the application of a similar amount of mineral oil coating a larger portion of the egg. The results of this study suggest that egg mortality was probably related to both toxicity of the oil and an interruption of normal gaseous exchange. Quantities of oil comparable to the amounts used during these investigations could be transferred from contaminated feathers of an incubating parent to the eggs (Brown 1980). Other authors indicate that embryos are particularly sensitive to oil early in their development (Szaro and Albers 1977; Albers and Szaro 1978; McGill and Richmond 1979), while Patten and Patten (1976, 1977) suggested that re-nesting is unlikely to occur following failure of a clutch to hatch since non-viable eggs may not be recognized as such by the parents, and will be incubated beyond the normal hatching date.

Some studies also show that crude oil affects the growth of juvenile birds. For example, growth rates of young black guillemots were reduced following a single 0.2 ml dose of crude oil, leaving the birds 20 percent underweight at the normal fledging date (Miller et al. 1978 a,b). Growth rates of juvenile herring gulls may also be reduced by exposure to crude oil (Miller et al. 1978 a, b), although Gorman and Simms (1978) suggest that the cessation of growth reported with these gulls was a natural function of age.

5.2.5.4 Summary of Concerns

The degree of concern regarding potential adverse effects of crude oil spills and blowouts or refined fuel spills on bird populations of the Beaufort Sea region would be highly dependent on: (1) the time of the spill; (2) the extensiveness of the area affected; (3) the type of habitats contaminated and the persistence of oil in these areas, and (4) the species and life history stages exposed to oil. The oil spill case history literature and results of numerous laboratory studies clearly indicate that the most serious and long-term impacts of spills would result if oil reached nearshore and backshore habitats which are important or critical staging and nesting areas. The degree of concern would also be greatest with species that are highly aquatic and have a low reproductive potential (e.g. alcids) or congregate in large numbers during specific activities such as feeding, staging and nesting (e.g. waterfowl). Consequently, depending on the species and habitats affected, the degree of concern regarding impacts of oil spills and blowouts on bird populations of the Beaufort Sea region is expected to range from MINOR to MAJOR.

5.2.6 Effects of Oil on Fish

The toxic and sublethal effects of petroleum hydrocarbons on marine fish have been the subject of relatively intensive laboratory investigations despite the fact that fish mortality has only been rarely observed following actual oil spills (Duval et al. 1981; Penrose 1981). The existence of few records of significant fish kills following oil spills is likely at least partially related to the rapid decrease in the concentrations of soluble petroleum hydrocarbons as oil weathers (Penrose 1981), the mucilaginous coating of fish and avoidance responses documented with some species. Nevertheless, it is generally accepted that larval fish (icthyoplankton) are more sensitive to oil than adults (Section 5.2.7), and mortality of these forms may have gone undetected in a number of past oil spills. Numerous sublethal effects have been observed in the laboratory, and may similarly go undetected following some oil spills. The following sections briefly summarize the results of laboratory investigations into the acute toxic and sublethal effects of petroleum hydrocarbons on fish, impacts actually observed following northern spills, and the degree of potential concern regarding crude oil spills or blowouts and refined fuel spills in the Beaufort Sea.

Laboratory investigations designed to determine the oil concentrations that result in acute or sublethal effects in fish have shown that there is a large degree of variability in toxic thresholds. This is primarily due to the variable sensitivity of different species and life history stages of each species, the variable composition of different crude oils and refined fuels, and particularly the methods used for completion of bioassays with oil and fish. While some investigators have demonstrated acute lethal concentrations (LC_{50} 96-h) of total hydrocarbons or aromatic fractions in the range of 1 to 3 ppm (Rice et al. 1979), other authors have reported no toxic effects at oil concentrations approaching several thousand ppm. The latter very high figures generally result from the unsatisfactory practise of expressing results of bioassays in terms of the amount of oil added to the test media rather than the measured dissolved oil concentration in water, particularly since the concentration of the dissolved oil fraction varies greatly with the method of preparation. However, the more recent toxicity studies have included measurement of hydrocarbon concentrations in the test media, and therefore are more consistent with respect to toxic concentrations. Detailed review of the large number of oil bioassays completed with fish is beyond the intended scope of this document, particularly since comprehensive treatments of this subject have been provided in a number of previous reviews (e.g. Malins 1977, Penrose 1981). Most studies have shown that dissolved oil concentrations less than 10-15 ppm are acutely toxic to marine fish (Table 5.2-7), with refined fuels such as diesel being more toxic than weathered crudes and most unweathered crudes (Craddock 1977).

TABLE 5.2-7

ACUTE TOXICITY OF VARIOUS CRUDE OILS AND REFINED FUELS
TO FISH GENERA FOUND IN THE BEAUFORT SEA
(Adapted from Craddock 1977; Rice *et al.* 1979)

Fish	Oil Type/Form	Acute Lethal (LC_{50}) Concentration (ppm)	Comment
Pink salmon	Prudhoe Bay crude WSF	0.09 (UV) 1.4 (IR)	96-h TLm, static, seawater
	Prudhoe Bay crude OWD	0.08 (UV) 4.5 (IR)	96-h TLm, static, seawater
(0.2-0.5 g)	Prudhoe Bay crude (simulated ballast water)	-	13-95 percent mortality in 48 to 96 h at <1.0-4.5 ppm
(fry)	Prudhoe Bay crude	213 (V/V)	96-h TLm, young fry at 7.5°C; static, seawater
		110 (V/V)	96-h TLm, older fry at 11.5°C; static, seawater
(alevin)	Prudhoe Bay crude OWD	0.6-3.2 (IR)	96-h TLm, static, freshwater
(emergent fry)	Prudhoe Bay crude OWD	0.4-1.3 (IR)	96-h TLm, static, freshwater
(migrant fry)	Prudhoe Bay crude OWD	0.04-0.08 (IR)	96-h TLm, static, seawater
	Cook Inlet crude (treated and untreated) WSF	0.1 (UV) 1.5-2.9 (IR)	96-h TLm, static, seawater
	Cook Inlet crude (treated) OWD	0.05 (UV) 3.4 (IR)	96-h TLm, static, seawater
	No. 2 diesel WSF-OWD	550 (V/V)	48-h LC_{50}, static, seawater
		184 (V/V)	96-h LC_{50}, static, seawater

TABLE 5.2-7 (cont'd)

Fish	Oil Type/Form	Acute Lethal (LC_{50}) Concentration (ppm)	Comment
	No. 2 fuel WSF	0.9 (IR)	24-h TLm, static, seawater
		0.8 (IR)	96-h TLm, static, seawater
Chum salmon (0.4-1.8 g)	Prudhoe Bay crude (simulated ballast water)	-	100 percent mortality in 16 to 70 h at 1.0-7.9 ppm
	No. 2 fuel OWD	1040 (IR)	96-h TLm, flowthrough, seawater
	No. 2 diesel WSF/OWD	538 (V/V)	48-h LC_{50}, static, seawater
		312 (V/V)	96-h LC_{50}, static, seawater
Chinook salmon (fingerling)	Prudhoe Bay crude WSF	-	24-h, bioassay, flowthrough, seawater. Mortality was 0 percent at 0.3 ppm (GC); 0.6 ppm (IR) 70 percent at 0.8 ppm (GC); 1.0 ppm (IR) 100 percent at 1.6 ppm (GC); 1.9 ppm (IR)
Chinook salmon	No. 2 diesel WSF/OWD	1190 (V/V)	48-h LC_{50}, static, freshwater
		349 (V/V)	96-h LC_{50}, static, freshwater
Dolly Varden	Prudhoe Bay crude WSF	0.07 (UV) 1.1 (IR)	96-h TLm, static, seawater
	Cook Inlet crude (treated and untreated) WSF	0.08-0.1 (UV) 1.5-1.9 (IR)	96-h TLm, static, seawater

TABLE 5.2-7 (cont'd)

Fish	Oil Type/Form	Acute Lethal (LC_{50}) Concentration (ppm)	Comment
	Prudhoe Bay crude OWD	0.1 (UV) 16.4 (IR)	96-h TLm, static, seawater
	Cook Inlet crude (treated) OWD	0.2 (UV) 7.3 (IR)	96-h TLm, static, seawater
	No. 2 fuel WSF	0.4 (UV) 2.3 (IR)	96-h TLm, static, seawater
Pacific herring	No. 2 fuel OWD	20 (IR)	96-h TLm, flowthrough, seawater
(larvae)	Cook Inlet crude WSF	0.75 (UV) 3.0 (IR)	96-h TLm, static, seawater
Herring (eggs)	Various crude oils WAF	-	90-100 percent mortality after 140 days in 100-10,000 ppm, static, seawater
Capelin	S. Louisiana crude OWD	150 (IR)	72 h TLm, flowthrough, seawater
Saffron cod	Cook Inlet crude WSF	3.3 (IR) 2.9 (IR)	24-h TLm, static, seawater 96-h TLm, static, seawater
	No. 2 fuel WSF	2.5 (IR) 2.3 (IR)	24-h TLm, static, seawater 96-h TLm, static, seawater
Staghorn sculpin	Kuwait crude OWD	5600	96-h TLm, flowthrough, seawater
Great sculpin	Cook Inlet crude WSF	4.0 (GC; aromatics)	96-h TLm, flowthrough, seawater

TABLE 5.2-7 (cont'd)

Fish	Oil Type/Form	Acute Lethal (LC_{50}) Concentration (ppm)	Comment
	No. 2 fuel WSF	1.3 (GC; aromatics)	96-h TLm, flowthrough, seawater
Sand lance	No. 2 fuel OWD	43 (IR)	96-h TLm, flowthrough, seawater
Starry flounder	S. Louisiana crude OWD	1400 (IR)	72 h TLm, flowthrough, seawater
	Cook Inlet crude WSF	>5.3 (GC; aromatics)	96-h TLm, flowthrough, seawater
	No. 2 fuel WSF	>1.0 (GC; aromatics)	96-h TLm, flowthrough, seawater

Abbreviations:
- WSF - water soluble fraction
- WAF - water accommodated fraction
- OWD - oil/water dispersion
- IR - infrared spectroscopy
- UV - ultraviolet spectroscopy
- GC - gas chromatography
- V/V - volume oil added per volume water

LC_{50} - lethal concentration at which 50 percent of test organisms die within the experimental period.
TLm - median tolerance limit; roughly equivalent to LC_{50}

In a recent review of the case histories of 100 world-wide oil spills, Duval et al. (1981) indicated that fish mortality has only been occasionally documented following oil spills (Table 5.2-3), and has then been largely restricted to individuals present in extensively contaminated shoreline areas, shallow coastal embayments or mariculture pens. Fish kills resulting from crude oil spills were only documented in 8 of the case histories reviewed by these authors, and suggested following 2 other events. The crude spills with unconflicting reports of fish mortality were the URQUIOLA (Gundlach and Hayes 1977), VENPET-VENOIL (Moldan et al. 1979), Tarut Bay (Spooner 1970), OCEAN EAGLE (Cerame-Vivas 1969), ESSO ESSEN (Stander and Venter 1968), WORLD GLORY (Stander and Venter 1968), ARGEA PRIMA (Diaz-Piferrer 1962), and Florida Keys (Chan 1977). In each case, the affected species were shallow water and bottom-dwelling forms. Fisheries surveys have been conducted after several other crude oil spills or blowouts (UNIVERSE LEADER, Santa Barbara, Platform Charlie, Ekofisk Bravo, Ixtoc I) and damage of pelagic species has been limited to occasional tainting of adults and mortality of eggs. In most cases, the impacts of crude oil spills on fish have primarily been the loss of supportive food resources or nearshore habitat. Sublethal effects of oil on fish have been reported (Ixtoc I; URQUIOLA; Florida Keys), but probably go undetected in the large majority of oil spills and blowouts. During the Ixtoc I blowout in the Gulf of Mexico, disturbance of the bottom attracted large numbers of fish to the site, while large schools of fish were also occasionally observed at the edge of the surface slick (Ross et al. 1979).

Although 5 of the 44 crude oil spills examined by Duval et al. (1981) occurred at latitudes north of 55°, effects on fish were only documented following the Ekofisk Bravo blowout in the North Sea. The results of one study showed that alkanes resembling those from Ekofisk oil were present in low concentrations in the gut and stomach of mackerel, haddock and plaice immediately after the blowout, while another investigation indicated that fish (plaice, haddock, gurnard and long rough dab) collected 10 days after the event had concentrations of pristanes in their tissues within the range previously reported for fish from areas of open sea around the U.K. (Law 1978; Mackie et al. 1978). Three months after the Ekofisk Bravo blowout, samples collected from one station contained several dead fish eggs that had oily encrustaceans on the outer membrane, while other abnormal eggs were also observed. However, taste panels did not detect tainting of any of the fish sampled 3 months after the event (Duval et al. 1981).

Fish kills were reported in 6 and thought to be possible in an additional 4 of the 26 refined fuel spills examined by Duval et al. (1981). There were no verified fish kills at latitudes north of 55°, and direct mortality has also only resulted either when refined fuels have been spilled in relatively sheltered marshes, intertidal zones and lagoons or, as indicated earlier, when fish were present in mariculture pens and therefore unable to avoid the oil. Avoidance responses of fish may also be restricted where the extent of shoreline contamination is great, or when suitable uncontaminated habitat is not readily available. These situations may have occurred at spills near isolated coral atolls such as the R.C. STONER grounding near Wake Island in the Pacific Ocean (Gilmore et al. 1970) or at another accident near Kiltan Island in the Indian Ocean (CSLP 1975), where large refined fuel spills (140,000 and 25,000 barrels, respectively) not only contaminated the entire outer shoreline (CSLP 1975), but penetrated extensively within the lagoon and caused relatively high fish mortality.

Local fish communities were investigated after 9 of the 29 bunker fuel spill case histories discussed by Duval et al. (1981) with mortality being reported in five instances (Table 5.2-5). Relatively few dead fish were observed following the ARROW spill, and it was not clearly established whether this mortality was due to the oil or to the physical and chemical methods employed during the shoreline cleanup program (Anon 1970). Since there was no evidence that the commercial fishery yield in Chedabucto Bay was decreased, the ARROW incident apparently had a highly localized impact on fish communities. In a similar manner, only minimal (if any) long-term impacts on cod and pollock populations were expected after the 189,000 barrel ARGO MERCHANT spill in the Atlantic Ocean during December, 1976. Although there was considerable mortality of the pelagic eggs of these species in areas sampled by the investigative team (Longwell 1977), the extensive distribution of these fish and their extremely high fecundity minimized any significant degree of long-term impact. The most dramatic case of fish mortality associated with a bunker spill was the Mizushima refinery incident which killed more than 30,000 fish held in mariculture pens in the Inland Sea of Japan (Nicol 1976). A similar type of damage was reported following the loss of 7000 barrels of bunker fuel from the ADRIAN MAERSK near Hong Kong; in this case, more than 20,000 kg of penned adult fish and 400,000 fry were killed in an area of intensive fish farming (Anon 1978). However, since the penned fish could not avoid oil-contaminated waters and were maintained close to the water surface in areas affected by both spills, mortality of this magnitude must be considered unusual.

As indicated earlier, some of the sublethal effects of petroleum hydrocarbons observed during laboratory investigations may also occur during actual oil spills but are not detected. However, the predictive value of much of this data base is limited to a certain extent by the small number of studies which have been conducted with northern fish species. Considerable research has been directed at various species of Pacific salmon (genus Oncorhynchus), and although these species are only rarely encountered in the Beaufort Sea (LGL and ESL 1981), the sublethal physiological and behavioural responses of anadromous fish species in the latter region may be similar to those observed with other salmonids. The sublethal effects of petroleum hydrocarbons on adult and larval Pacific herring (genus Clupea) are also well documented, and provide an adequate data base for assessment of the potential impacts of oil spills on the abundant herring populations of the eastern

Beaufort Sea. The most important sublethal effects of hydrocarbons which may be of ecological significance to arctic marine fish populations are the uptake of aromatic hydrocarbons in tissues, particularly the brain and liver (Collier et al. 1978; DeMichele and Taylor 1978; Roubal et al. 1978; Duval and Fink 1980), destruction of blood cells and damage to the spleen and kidney (Waluga 1966), induction of developmental abnormalities (Smith and Cameron 1979; Stoss and Haines 1979; Cameron and Smith 1980), development of eye cataracts and reduced hatching success (NMFS 1979), cellular disruption and aberration (Cameron and Smith 1980) and reduced feeding activity (Wang and Nicol 1977; Korn et al. 1979). These potential effects of oil exposure may individually or collectively reduce the survival of some fish populations, although many sublethal physiological and behavioural abnormalities found after exposure of fish to petroleum hydrocarbons are completely reversible. Fish also appear to have a greater ability than invertebrates to metabolize and depurate aromatic hydrocarbons (e.g. Duval and Fink 1980; Penrose 1981).

A factor which could influence the degree of impact of oil spills on arctic marine fish populations is the existence and sensitivity of potential avoidance responses. Although some species avoid extremely low concentrations of oil, other fish apparently will not automatically avoid deleterious levels of petroleum, even if they are able to detect them (Penrose 1981). For example, pink and coho salmon fry show avoidance responses at oil concentrations as low as 1.0 to 1.6 ppm in seawater (Rice 1973; Duval and Fink 1980), while both pink salmon fry and rainbow trout in freshwater do not avoid sublethal oil concentrations (Sprague and Dury 1969; Rice 1973). Avoidance thresholds also vary with species (Syazuki 1964; Patten 1977). Similar variability in avoidance capacities and thresholds of fish may be expected in arctic marine environments, including the Beaufort Sea. One of the greatest potential concerns with respect to Beaufort Sea fish resources would be disruption of the spawning migration of anadromous species, including the ciscos and whitefish. Studies completed by the National Marine Fisheries Service (1978) indicated that upstream migration of adult salmon was inhibited when oil concentrations in nursery streams were 0.7 ppm or greater. Exposure of fish to sublethal oil concentrations prior to their migration caused a delay in their return to natal streams, but did not alter their ability to return.

Both oil spill case history and laboratory literature are of predictive value for assessment of the potential impacts of oil spills on fish species found in the Beaufort Sea. The general tendency for oil to be less toxic in seawater than in freshwater (Anderson and Anderson 1976) suggests that the greatest potential for acute toxic and sublethal effects may occur in coastal areas influenced by discharge of the Mackenzie River and other smaller freshwater drainages. These coastal areas are important rearing and feeding habitats of several anadromous fish species (LGL and ESL 1981), and at the same time have a high oil retention capacity due to the fine-grain composition of most substrates. Consequently, direct and indirect impacts of oil spills and blowouts on fish would likely be greatest in these areas, at least during the open water season when anadromous fish species frequent coastal areas of the Beaufort. The most common anadromous species which could be affected by oil spills in this region are Arctic cisco (Coregonus autumnalis), least cisco (C. sardinella) and Arctic char (Salvelinus alpinus), although boreal smelt, humpback whitefish, broad whitefish and inconnu are also relatively abundant in some coastal areas. Oil reaching coastal areas of the Beaufort Sea east of the Mackenzie Delta and along the Tuktoyaktuk Peninsula to Liverpool Bay could result in considerable damage to Pacific herring populations which spawn in this region. Larval herring would be more susceptible to impacts of oil than the adults. Other marine species such as fourhorn sculpins and flounders could also be affected in coastal habitats, while Arctic cod may be vulnerable to oil spills in the late summer. However, in most cases, damage to coastal fish populations would likely be limited to sublethal effects, including tainting, and short-term behavioural and physiological abnormalities.

The most significant areas of concern with respect to coastal fish populations of the Beaufort Sea are: (1) relatively long-term exposure to low levels of petroleum hydrocarbons as stranded oil is slowly released from the coastal sediments into shallow water, potentially leading to delayed migration of anadromous species or chronic physiological and behavioural effects which may reduce productivity and/or reproductive success; and (2) uptake of hydrocarbons and accumulation of these compounds in other consumers, including birds and marine mammals. Rates of recovery of affected fish populations would depend primarily on the persistence of oil in shallow coastal areas (anadromous species) or offshore sediments (marine species), as well as the opportunity for recruitment from adjacent uncontaminated areas. According to the criteria discussed in the introduction to this report, the degree of concern regarding adverse impacts of oil spills and blowouts on regional fish populations of the Beaufort Sea would likely vary from MINOR to MODERATE because of the potential for chronic exposure of several generations of some local populations to oil released from sediments.

5.2.7 Effects of Oil on Fish Eggs and Icthyoplankton

Both laboratory and oil spill case history literature indicate that petroleum hydrocarbons can result in direct mortality and abnormal development of fish eggs and larvae. Mortality of eggs or larval stages of herring, pilchard, cod, pollock, sandlance and redfish was documented after several oil spills including the ARGO MERCHANT, TSESIS and TORREY CANYON accidents and the recent Ixtoc I blowout in the Gulf of Mexico (Duval et al. 1981).

Laboratory studies conducted to date indicate that ichthyoplankton are generally more sensitive to petroleum hydrocarbons than adult fish (Craddock 1977; Penrose 1981). Wells (1980) reported that fish eggs are usually less sensitive to oil than larvae, with the most vulnerable period being during and immediately after hatching when oil exposure can result in a high proportion of larvae that die soon after hatching or that have morphological deformities. Struhsaker (1977) also found that survival of Pacific herring eggs was reduced after spawning females were exposed to an aromatic hydrocarbon for several weeks. Although it has been suggested that herring are particularly susceptible to oil spills because they generally deposit their spawn in shallow subtidal areas which are most likely to be contaminated with oil (Wells 1980), investigations conducted after the UNIVERSE LEADER spill (CSLP 1975) failed to show any damage to eggs and larvae present in the area affected by oil. This may be partially attributable to the relatively low concentrations of dispersed oil which have been found in the water column after spills (Cormack and Nichols 1977; Law 1978), as well as the decreased acute toxicity of dispersed oil to fish larvae with increased age of the oil dispersion (Wells 1980). Nevertheless, other effects of petroleum hydrocarbons on fish eggs and larvae which have been observed in the laboratory and may occur in some instances following oil spills are slower embryonic growth, changes in heart activity, decreased hatching success, abnormal swimming behaviour, narcosis, tissue damage, reduced feeding, altered respiration rates and various external and internal body deformities (Smith and Cameron 1979; Stoss and Haines 1979; Wells 1980).

An oil spill in arctic waters could result in considerable mortality of the eggs and planktonic larvae of marine fish species such as polar cod, Arctic cod, various flounders and soles as well as larvae of certain anadromous species such as boreal smelt, which are common off the Mackenzie Delta in summer. As indicated earlier, damage to the eggs and larvae of Pacific herring could also occur in the eastern Beaufort Sea. In all cases, the impact of oil spills on fish eggs and planktonic larvae would be highly dependent on the degree of exposure to oil, and the location and time of a spill. In addition to direct mortality of eggs and larvae, sublethal effects of water-soluble components of the oil or dispersed oil could result in physiological, behavioural or morphological abnormalities which decrease the survival of newly hatched larvae. In a local context, these lethal and sublethal effects of oil spills could be relatively serious, affecting an entire year class of a local population of some marine species, and therefore the overall degree of concern is considered MINOR to MODERATE. However, the high fecundity of most marine fish species, together with the opportunity for recruitment and advective transport of eggs and larvae from unaffected areas would minimize the regional impact of oil spills in arctic environments.

5.2.8 Effects of Oil on Phytoplankton

Extensive reviews of the effects of petroleum hydrocarbons on phytoplankton have been provided by Corner (1978), Vandermeulen and Ahern (1976), Johnson (1977) and Snow (1981). Biological observations of changes in structure and abundance of phytoplankton communities were also completed following the Mizushima, Santa Barbara, TSESIS, TORREY CANYON, ARGO MERCHANT and SANSENINA spills (see Duval et al. 1981). These literature sources indicate that there have been considerable differences in the responses of phytoplankton to petroleum hydrocarbons, ranging from stimulated growth to marked reductions in primary productivity and associated changes in the species composition of phytoplankton communities (Table 5.2-8).

Although detailed review of the laboratory-based literature describing the effects of oil on phytoplankton is beyond the scope of the present document, a number of observed trends have important implications in the assessment of the potential degree of concern regarding oil spills in northern environments. Petroleum hydrocarbons generally extend the lag phase of population growth, as well as depress the exponential growth phase (Snow 1981). The water-soluble components (particularly aromatics) of crude oils and refined petroleum products tend to be responsible for most but not all toxic effects. As a result, refined petroleum products such as diesel fuel tend to cause greater depressions of phytoplankton growth and production than crude oils (e.g. Anderson et al. 1974b). However, studies conducted by Hellebust et al. (1975) and Kauss and Hutchinson (1975) indicate that the inhibitory effects of oil on phytoplankton growth diminish with weathering or volatization of some petroleum hydrocarbon constituents.

TABLE 5.2-8

SUMMARY OF REPRESENTATIVE STUDIES REGARDING THE EFFECTS OF PETROLEUM HYDROCARBONS ON PHYTOPLANKTON

Hydrocarbon Concentration (ppm)	Oil Type	Species[1]	Effects	Reference
0.005	unspecified oil	NP (*Nitzschia*)	stimulated photosynthesis	Parsons *et al.* 1976
0.005-0.5	n-alkanes	NP (*Skeletonema*)	depressed photosynthesis	Parsons *et al.* 1976
0.019-0.787	crude and No. 2 fuel	*Dunaliella* and *Fragilaria*	no significant effect on cell division in 24 days	Prouse *et al.* 1976
0.04	No. 2 fuel	NP	decrease in diatoms but inc. in flagellate pop.	Lee *et al.* 1977a
0.04-0.4	No. 2 fuel	*Thalassiosira*	reduced photosynthesis	Pulich *et al.* 1974
<0.05	crude and unspecified fuel	NP	stimulated photosynthesis	Gordon and Prouse 1972
0.05	aromatics	NP (*Skeletonema*)	stimulated photosynthesis	Parsons *et al.* 1976
0.05-0.3	crude and unspecified fuel	NP	inhibited photosynthesis	Gordon and Prouse 1972
0.05	alkanes	NP	inhibited photosynthesis	Parsons *et al.* 1976
0.05-.5	aromatics and alkanes	NP with (*Skeletonema*)	stimulated photosynthesis	Parsons *et al.* 1976
0.1	n-alkanes	NP (*Skeletonema*)	stimulated photosynthesis	Parsons *et al.* 1976

TABLE 5.2-8 (cont'd)

Hydrocarbon Concentration (ppm)	Oil Type	Species[1]	Effects	Reference
0.1-10	unspecified oil	Gymnodinium	delayed or prevented cell division in 5 days	Mironov 1970
0.1-100	volatile aromatics	Dunaliella	stimulated growth rate	Dunstan et al. 1975
<0.1	aromatics, alkanes, alkenes	NP with (Nitzschia)	stimulated photo-synthesis	Parsons et al. 1976
>0.1	aromatics, alkanes, alkenes	(Nitzschia)	inhibited photo-synthesis	Parsons et al. 1976
0.15	No. 2 fuel	NP	reduced photo-synthesis by 50 percent	Gordon and Prouse 1973
0.4	bunker C	Isochrysis	50 percent reduction in cell numbers in 72 h	Anderson et al. 1974b
0.5	No. 2 fuel	Isochrysis	50 percent reduction in cell numbers in 72 h	Anderson et al. 1974b
0.6	bunker C	Glenodinium	50 percent reduction in cell numbers in 72 h	Anderson et al. 1974b
0.6	No. 2 fuel	Melosira	50 percent reduction in cell numbers in 72 h	Anderson et al. 1974b
0.6	No. 2 fuel	Glenodinium	50 percent reduction in cell numbers in 72 h	Anderson et al. 1974b
0.7	bunker C	Cyclotella	50 percent reduction in cell numbers in 72 h	Anderson et al. 1974b

TABLE 5.2-8 (cont'd)

Hydrocarbon Concentration (ppm)	Oil Type	Species[1]	Effects	Reference
<1	refined oil	phytoplankton	toxic effects	Anderson *et al.* 1974b
1	unspecified oil	*Melosira*	inhibited cell division	Mironov and Lanskaya 1966
1	unspecified oil	*Ditylum*	stimulated cell division	Mironov and Lanskaya 1966
3.2	Louisiana crude	*Glenodinium*	50 percent reduction in cell numbers in 72 h	Anderson *et al.* 1974b
3.4	unspecified crude	NP	decreased primary production	Hsiao 1976
4.0	Kuwait crude	*Melosira*	50 percent reduction in cell numbers in 72 h	Anderson *et al.* 1974b
4.4	Louisiana crude	*Isochrysis*	50 percent reduction in cell numbers in 72 h	Anderson *et al.* 1974b
5.5	Louisiana crude	*Melosira*	50 percent reduction in cell numbers in 72 h	Anderson *et al.* 1974b
6.4	Kuwait crude	*Glenodinium*	50 percent reduction in cell numbers in 72 h	Anderson *et al.* 1974b
6.6	Kuwait crude	*Isochrysis*	50 percent reduction in cell numbers in 72 h	Anderson *et al.* 1974b
10	unspecified crude	Diatoms	growth inhibited after 10 days	Hsiao 1978

TABLE 5.2-8 (cont'd)

Hydrocarbon Concentration (ppm)	Oil Type	Species[1]	Effects	Reference
10	unspecified crude	Chlamydomonas	stimulated growth	Hsiao 1978
10	unspecified oil	Chaetoceros	delayed or prevented cell division in 5 days	Mironov 1970
100	unspecified crude	diatoms and flagellates	marked inhibition of growth	Hsiao 1976
100-1000	unspecified oil	Coscinodiscus	delayed or prevented cell division in 5 days	Mironov 1970
100-1000	unspecified oil	Melosira	delayed or prevented cell division in 5 days	Mironov 1970
200	Louisiana crude	Thalassiosira and Chlorella	no growth	Pulich et al. 1974
400	Kuwait crude	Thalassiosira and Chlorella	no growth	Pulich et al. 1974
1000	unspecified oil	Ditylum	100 percent reduction in cell numbers in 24 hr	Mironov and Lanskaya 1966
1600-8000	Kuwait and Louisiana crude	Thalassiosira and Chlorella	no growth	Pulich et al. 1974
1818	unspecified crude	NP	10 fold decrease in primary production	Dickman 1971
10,000	unspecified oil	Melosira	no measurable effect in 10 days	Mironov and Lanskaya 1966

1 Dominant genera indicated in parentheses (where appropriate) for natural phytoplankton (NP) communities

The effects of petroleum hydrocarbons on phytoplankton also vary with species and concentrations of the water-soluble components. Low concentrations (less than 50 ppb) may stimulate or have no effect on growth rates (Gordon and Prouse 1972; Shiels et al. 1973; Boney 1974; Pulich et al. 1974; Prouse et al. 1976; Nunes and Benville 1979), while higher concentrations (50-300 ppb) cause inhibition of growth or a reduction in standing crop (Mironov and Lanskaya 1966; Dickman 1971; Gordon and Prouse 1972; Shiels et al. 1973; Anderson et al. 1974b; Hellebust et al. 1975; Kauss and Hutchinson 1975; Hsiao 1976; Parsons et al. 1976; Hsiao 1978). The primary productivity of natural phytoplankton communities was slightly enhanced following an experimental in situ oil spill (64 m^3) in Balaena Bay near the tip of Cape Parry (NORCOR 1977). A slight increase in the total abundance and diversity of the phytoplankton communities present in oil-contaminated samples was also noted during this investigation. Studies completed with Beaufort Sea phytoplankton suggest that flagellates are more resistant to petroleum hydrocarbons from crude oil than either centric or pennate diatoms (Hsiao 1978). In addition, the effects of petroleum hydrocarbons on phytoplankton may vary with season due to differences in the physiological condition of algal populations (Snow 1981). For example, Gordon and Prouse (1972) found that the effects of crude oils on phytoplankton were more marked in spring than in fall, while Shiels et al. (1973) reported that the toxicity of oil to phytoplankton increased with increasing temperature and was also most toxic at high light intensities. Other reported effects of oil on some phytoplankton species include the accumulation of hydrocarbons (Thompson and Eglinton 1976; Karydis 1980) and destruction of chlorophyll (Soto et al. 1975).

The different responses of phytoplankton to past oil spills (Duval et al. 1981) is not surprising in view of the variable effects observed during laboratory and controlled-ecosystem studies. The impacts of oil spills in arctic marine environments such as the Beaufort Sea are likely to be just as variable. They will be influenced by many factors including the species present at the time of the spill, the type of oil and its rate of weathering, and concentrations of mono- and dicyclic aromatics present in the water column. The initial response of the Beaufort Sea phytoplankton community would probably be a decrease in the growth and productivity of diatoms, and likely a subsequent stimulation of microflagellate and flagellate growth as petroleum hydrocarbon concentrations in the water column gradually decrease. The results of several laboratory, controlled ecosystem and simulated spill studies (e.g. Dickman 1971; Lee et al. 1977a; Hsiao 1978) lend support to this hypothesis. Nevertheless, any changes in the community structure, productivity or abundance of phytoplankton would be relatively short-term due to the weathering and dissipation of petroleum hydrocarbons, as well as the opportunity for replacement with phytoplankton from adjacent unaffected areas through ocean currents. The chronic release of oil trapped in coastal substrates could, however, result in persistent but localized effects on nearshore phytoplankton communities during subsequent growing seasons.

The location, time and type of oil spill would also affect subsequent impacts on phytoplankton communities. Spills of petroleum products having a high aromatic content such as gasoline, JP-4, JP-5 and diesel fuels would likely have more adverse immediate effects on phytoplankton due to their greater initial toxicity. However, these refined fuels would also generally weather more quickly than crude oils, reducing the duration of potential impacts. Impacts would also likely be greatest early in the growing season since any delays in phytoplankton blooms and maximum primary productivity could subsequently affect members of higher trophic levels. In the Beaufort Sea, seasonal differences in the sensitivity of phytoplankton to oil spills could be increased due to the predominance of centric and pennate diatoms during the early part of the growing season (LGL and ESL 1981). The results of laboratory studies suggest that diatoms may be considerably more sensitive to petroleum hydrocarbons than flagellates. Similarly, Beaufort Sea phytoplankton communities in different areas may exhibit different susceptibilities due to the greater proportion of diatoms and flagellates in nearshore and offshore waters, respectively (Hsiao 1976; LGL and ESL 1981). Oil spills in offshore areas of the Beaufort would have a greater tendency to stimulate growth of microflagellates and flagellates, while reductions in diatom growth and production would be more likely in shallow coastal areas.

The overall degree of concern regarding adverse impacts of oil spills on regional phytoplankton communities would also vary with location of the affected areas and time of year. Oil spills or blowouts that affect offshore waters at any time of year are expected to be of <u>MINOR</u> concern with respect to regional phytoplankton populations because: (1) more oil-tolerant species are found in offshore habitats; (2) chronic oil exposure is unlikely (except near production facilities), and (3) advective transport of organisms from uncontaminated areas would be a dominant process due to prevailing current velocities. On the other hand, an oil spill or blowout which affected some sheltered nearshore habitats could be an area of <u>MODERATE</u> concern with respect to phytoplankton populations because of: (1) the presence of diatom species which are more sensitive to oil exposure; (2) the opportunity for chronic oil exposure in shallow habitats with fine-grained substrates, and (3) the more limited water circulation. Diesel fuel spills early in the growing season in these shallow habitats may have more pronounced effects because of the greater toxicity of refined fuels to phytoplankton, and the potential for adverse impacts which persist throughout the growing season and affect members of higher trophic levels.

5.2.9 Effects of Oil on Zooplankton

The effects of petroleum hydrocarbons on zooplankton have been recently reviewed by Kuhnhold (1977), Corner (1978) and Wells (1980). The impacts of oil spills on zooplankton were also examined following the Santa Barbara and Ekofisk Bravo blowouts and the TORREY CANYON, Anacortes, USNS POTOMAC, SANSENINA, TSESIS, ARGO MERCHANT, AMOCO CADIZ and ARROW spills (Duval et al. 1981). As with phytoplankton, laboratory and field studies have indicated that oil may have a range of effects on zooplankton communities. Since crustaceans comprise the dominant proportion of zooplankton communities in the Beaufort Sea, this section focuses on the impacts of oil spills on these pelagic invertebrates, including the larvae of benthic amphipods and mysids. The larval stages of annelids, molluscs and echinoderms, although also planktonic at times during their life cycle, are discussed in Section 5.2.10. The effects of petroleum hydrocarbons on fish eggs and planktonic larval stages were previously discussed in Section 5.2.7.

Laboratory studies of petroleum hydrocarbons and zooplankton (Table 5.2-9) explain much of the variability observed in the case history literature. For example, there are marked differences in the sensitivity of copepods to both the water-soluble fraction of oil and dispersed oil. Wells (1980) reported that water-soluble fractions cause paralysis of copepods at concentrations as low as 0.2 to 0.5 mg/L, while dispersed oil at concentrations from 0.05 to 100 mg/L was lethal to adult and copepodite stages. Dispersed light fuel oils are more toxic to copepods than dispersed crude oils (Wells 1980). Studies initiated after the grounding of the ARROW in Nova Scotia (Conover 1971) and the TSESIS spill in Sweden (Linden et al. 1979) show that some copepod species ingest particles of oil, with the oil forming part of the faeces. These authors also suggest that grazing by herbivorous zooplankton may be a natural method of removal of emulsified oil from the water column, although Spooner and Corkett (1974) found a substantial reduction in the production of faecal pellets by *Calanus* and other species after exposure to 10 ppm of oil for 24 h. In addition, copepods accumulate specific aromatic hydrocarbons such as naphthalenes from the water and from food organisms. Naphthalenes reach an equilibrium within the tissues (40-175 μg/g) within 8 days, but are depurated once the animals are in uncontaminated water (Wells 1980). Sublethal effects of petroleum hydrocarbons on copepods include reduced egg and nauplii production, brood size and longevity (Ott et al. 1978).

TABLE 5.2-9

SUMMARY OF ACUTE LETHAL CONCENTRATIONS OF CRUDE OILS AND REFINED PETROLEUM PRODUCTS ON ZOOPLANKTON IN CANADIAN MARINE WATERS
(Source: adapted from Wells 1980)

Zooplankton Taxonomic Group	Life Cycle Stage	Oil Type/Exposure Duration (h)	Acute Lethal Concentration (LC_{50} ppm)
CTENOPHORES	-	No. 2 fuel/24	0.59 (WE)
MOLLUSCS	embryos	Crude/48	0.23-12.0 (WSF)
	larvae	Crude/48	0.25-25.0 (WSF)
	embryos	No. 2 fuel/48	0.43 (WSF)
	larvae	No. 2 fuel/48	1.3 (WSF)
	embryos	Bunker C/48	1.0 (WSF)
	larvae	Bunker C/48	3.2 (WSF)
COPEPODS	larvae and adults	Crude/ns	0.05-100 (OWD)
	adults	No. 2 fuel/96	1-3 (OWD)
AMPHIPODS	larvae	Crude/48	0.8 (OWD)
	larvae	No. 1 fuel/48	0.3 (OWD)
	larvae	No. 4 fuel/48	6.2 (OWD)
	adults	No. 1 fuel/48	173 (OWD)
	adults	No. 4 fuel/48	1000 (OWD)
SHRIMP	larvae and adults	Crude/96	0.5-7.9 (WSF)
	adults	No. 2 fuel/96	0.8 (OWD)
LOBSTER	larvae	Crude/96	1-5 (OWD)
	larvae	Crude/720	0.14 (OWD)
CRAB	larvae	Crude/96	<5 (OWD)
	larvae	No. 2 fuel/96	4-10 (OWD)

ABBREVIATIONS:

WE - water extract
WSF - water soluble fraction
OWD - oil-in-water dispersion
ns - not specified

Other crustaceans which form major proportions of the zooplankton community during some months are the larval stages of amphipods, mysids and decapods. Larval amphipods are more sensitive to dispersed crude and refined oils than adults; both stages are more sensitive to refined than crude oils, although dispersed crude oil has also been shown to reduce the larval growth and brood size of amphipods (Wells 1980). Laboratory research indicates that juvenile mysids and decapods in the plankton are even more sensitive to dispersed oil than amphipods, and may show a range of sublethal effects in response to low hydrocarbon concentrations, including narcosis, paralysis, uptake of naphthalenes, reduced rates of development, reduced feeding rates, and delayed moulting (Wells 1980).

With the exception of the TORREY CANYON spill where large quantities of toxic first-generation dispersants were used, only localized and short-term impacts on zooplankton have been reported following oil spills (Duval et al. 1981). However, many of the sublethal effects observed in the laboratory probably were undetected, and given the relative sensitivity of many species, particularly the larval stages of amphipods, decapods and mysids, some degree of mortality of zooplankton is probable after most oil spills.

The impacts of oil spills on arctic zooplankton communities would be highly dependent on the time, size, location and duration of the event, as well as the amount and composition of oil which becomes dispersed or dissolved within the water column. In general, a subsea blowout would have far greater effects on zooplankton than a single spill because it would be a continuous and prolonged source of unweathered crude, and the oil would have a greater opportunity to disperse into the water column before reaching the surface. Impacts would also be greater at those locations and during those months when the planktonic stages of mysids, decapods and amphipods were present in the water column. An oil spill or blowout early in the open water season could result in localized impacts which persist for much of the remainder of the year due to residual sublethal effects such as decreased feeding rates, reduced rates of development and delayed moulting. The tendency for zooplankton to accumulate naphthalenes in their tissues could also result in indirect impacts on members of higher trophic levels which prey on zooplankton until these compounds were naturally depurated. Nevertheless, it is anticipated that the impacts of oil spills on arctic zooplankton would be relatively short-term due to the gradual recruitment of individuals from uncontaminated areas. In the Beaufort Sea, impacts of oil spills on zooplankton would tend to be greatest in areas which support relatively high standing stocks during the summer months, such as Mason Bay and the coastal waters along the Tuktoyaktuk Peninsula. The overall degree of concern regarding impacts of oil spills and blowouts and refined fuel spills on zooplankton in this region is considered MINOR to MODERATE, with the latter concern occurring in the aforementioned areas which support relatively high zooplankton standing stocks.

5.2.10 Effects of Oil on Benthic Flora and Fauna

The effects of petroleum hydrocarbons on benthic organisms have been the subject of intensive laboratory research and field investigations following oil spills. Detailed reviews of the literature describing the effects of crude oil on benthic organisms are provided in Craddock (1977) and Johnson (1977), while the documented or anticipated impacts of past spills on this community are also discussed in another supporting document to the Beaufort Production EIS (Duval *et al.* 1981). The following sections describe the potential effects of oil spills on benthic macrophytes and infaunal and epifaunal invertebrates found in the Beaufort Sea, as well as the overall degree of regional concern regarding these potential effects. Although the acute toxic effects of short-term exposure (96-h or less) of benthic organisms to crude oil are well documented, sublethal physiological and behavioural responses and the effects of chronic exposure to oil-contaminated sediments are not as well understood. The latter area of potential impact could be a primary concern in the Beaufort Sea due to the predominance of fine-grain sediments which would tend to retain oil, as well as the slow rate of microbial degradation of oil in low temperature, nutrient-limited environments (Atlas and Bartha 1972; Bunch and Harland 1976).

In a recent review article, Percy (1980) noted that the impacts of past oil spills on benthic and intertidal organisms have ranged from little or no observable biological damage to extensive habitat damage and massive mortalities. The oil spill case history review completed by Duval *et al.* (1981) indicated similar variability in the impacts of oil on benthic flora and fauna. This variability is partially due to the natures of the habitats which have been affected by oil. Since the sublethal and acute toxic effects of oil would vary with species and habitat type, existing information on the effects of oil is summarized by major taxonomic group in subsequent sections. It should also be emphasized that an exhaustive review of this literature is beyond the intended scope of this report because several comprehensive treatments of this subject are available elsewhere (e.g. Malins 1977).

5.2.10.1 Benthic Macrophytes

The effects of oil on benthic plant communities have been documented following several spills, although there are limited data available for arctic and subarctic environments. Generally, intertidal plants have received the most attention but this community is virtually non-existent in the Beaufort Sea. However they are discussed here in relation to general effects of oil on macrophytic algae.

Intertidal plant communities in southern Canadian coastal waters are dominated by brown algae of the genus *Fucus* (Snow 1981). The impacts of various types of oil spills on these species have been relatively well documented in the Northern Hemisphere. The type and extent of damage to *Fucus* and other intertidal algae appears to have varied to a large extent with the season when the spill occurred and the type of shoreline restoration methods used during the cleanup response. For example, only short-term damage to *Pelvetia* and *Fucus* in upper intertidal areas was reported near Dounreay, Scotland after contamination with fuel oil, with oiled areas showing new growth after only six months (Bowman 1978). On the other hand, the IRINI spill in Sweden and the ELENI V spill in England resulted in no visible effects on *Fucus* (Notini 1978; Blackman and Law 1980). The lack of damage in the former event was attributed to the season of the spill and low tidal amplitude in the Baltic region.

The most serious and long-term damage to intertidal plant communities has resulted from large bunker or crude oil spills, particularly when dispersants have been used during the cleanup program. For example, extensive mortality of *Fucus* in all intertidal zones was reported after the grounding of the tanker ARROW in Chedabucto Bay, Nova Scotia, and persistent effects were still apparent after 2 years (Anon. 1970). In the case of the TORREY CANYON and Coryton spills in England, major damage to *Fucus* and other intertidal algae was attributable to the use of chemical dispersants, while *Fucus* in the upper intertidal zone was heavily contaminated with oil from the AMOCO CADIZ and subsequently cropped by the cleanup crews (George 1970; Smith 1970; Chedd 1979).

Impacts of oil spills on subtidal algae have not been as frequently documented. The only instances of visible damage to kelp communities resulted from the TAMPICO MARU, GENERAL M.C. MEIGS, UNIVERSE LEADER and ARGEA PRIMA spills and the Santa Barbara blowout (Diaz-Piferrer 1962; North *et al.* 1965; Foster *et al.* 1971; Clark *et al.* 1973; Cullinane *et al.* 1975). Impacts of these spills included detachment of benthic algae from the substrate, bleaching and direct mortality associated with the toxic or encrusting effects of oil. Increased growth of macroalgae has also occurred following some spills due to a decrease in numbers of herbivorous gastropods or sea urchins (Johnson 1977; Duval *et al.* 1981). Laboratory studies with subtidal algae have shown that relatively thin films of crude oil can reduce gas exchange and adversely affect photosynthesis of some species (Snow 1981).

Benthic macrophytes are not as prevalent in most of the Canadian Arctic as in temperate waters due to the limited tidal amplitude, intensive ice scour and soft substrates found in most areas. The standing crop of macrophytes decreases from east to west, with the north coast of Alaska being almost barren of algae (Mohr *et al.* 1957; Lee 1973). For example, in the Beaufort Sea, benthic macrophytes are usually restricted to the few areas where there is a rocky substrate such as Elson Lagoon, Stefanson Sound and Liverpool Bay. Consequently, the potential impacts of oil spills on marine macrophytes would be very localized in the western Arctic. The dominant algae in this region are *Laminaria* spp. which are highly susceptible to damage from oil spills, as evident after the grounding of the GENERAL M.C. MEIGS off the coast of Washington and the UNIVERSE LEADER in Ireland (Duval *et al.* 1981). Localized damage to the isolated *Laminaria* beds in the Beaufort Sea could include bleaching of photosynthetic pigments, detachment from the substrate and direct mortality. The results of several laboratory studies with marine algae also suggest that other effects of oil may include reduced RNA and DNA synthesis (Davavin et al. 1975), induction of cancerous growths (Boney 1974) and decreased germination and growth of zygotes (Steele 1977). Algae spermatozoa may be very sensitive to petroleum since fertilization of Fucus sp. did not occur even at a crude oil concentration as low as 2 ppb (Steele 1977). Sublethal effects such as these could deleteriously affect the relatively isolated populations of benthic macrophytes of the Beaufort Sea, although the overall degree of concern regarding both sublethal and lethal effects of oil spills and blowouts on these algae is considered MINOR because of their limited distribution and low abundance in the region.

The effects of oil on the mats of filamentous algae and diatoms, which are far more prevalent than benthic macroalgae in the Beaufort Sea, are not well documented. In a laboratory study using benthic freshwater algae, crude oil extracts had less of an effect on community composition than No. 2 fuel oil extracts, with the latter resulting in depressed algal biomass, blue-green algal dominance and decreased diatom occurrence (Bott and Rogenmuser 1978). The effects of crude oil on benthic microalgae would probably be similar to effects observed with phytoplankton (Section 5.2.8), although the duration of exposure would generally be longer and some mortality could occur as a result of direct smothering. Although the distribution of benthic microalgae in the Beaufort Sea is poorly documented, they are likely restricted to relatively shallow areas or some isolated embayments where light can reach the bottom, because the turbid Mackenzie River plume would limit light availability in most coastal benthic habitats. However, since these algae are expected to be an important link in the marine food web of this region (EIS Volume 3A, Section 3.1), the degree of concern regarding impacts of oil spills or blowouts on benthic microalgae is considered MINOR to MODERATE.

5.2.10.2 Annelids

The effects of petroleum hydrocarbons on marine annelids have been relatively well documented as a result of numerous laboratory investigations (Craddock 1977; Johnson 1977) and several oil spill case histories (Duval et al. 1981). Annelids are a major component of the benthic infauna in shallow, sedimentary marine environments. This is particularly the case in arctic environments because of the predominance of soft sediment substrates (Owens 1977). For example, Wacasey (1975) documented 101 species of polychaetes in coastal and offshore areas of the Beaufort Sea. This was nearly half of the combined total of 236 species for all other invertebrate groups.

A number of annelid species are known to be resistant to pollution in general (e.g. Capitella capitata), and to fuel oil in particular (e.g. Cirriformia tentaculata and Cirratulus cirratus) (George 1971; Lee 1976; Carr and Reish 1977). For example, after the grounding of the FLORIDA in West

Falmouth, Massachusetts, the abundance of Capitella capitata increased almost exponentially to more than 200,000 organisms/m^2 in some areas; the highest abundance of this species was observed in sediments where No. 2 fuel oil concentrations were highest (Sanders 1978). Such increases in standing stock may follow a decrease in competition from other less-resistant infaunal species, while some polychaete species actually have a higher reproductive rate when exposed to low concentrations of toxic substances (Emerson 1974, cited in Soule 1980). For example, 1.9 ppm of South Louisiana crude oil was found to stimulate reproduction of Ophryotrocha sp. (Carr and Reish 1977). On the other hand, some polychaete and oligochaete species have been shown to be very sensitive to petroleum hydrocarbons in laboratory investigations or following actual oil spills. Sensitive species include Arenicola marina, juvenile Neanthes arenaceodentata, Nereis diversicolor and Pelescolex benedeni (Wharfe 1975; Prouse and Gordon 1976; Rossi and Anderson 1978). Atlas et al. (1978) reported that three species of Beaufort Sea polychaetes seemed to prefer recolonization in oil-contaminated sediments over clean, control sediments, while Acanthostephieia behrengiensis only colonized uncontaminated sediment trays. The results of most studies conducted to date also suggest that polychaetes are more susceptible to fuel oils that crude oils (Craddock 1977). Nevertheless, localized increases or decreases in the numbers of most infaunal annelid species in the Beaufort Sea would have relatively limited indirect impacts on members of higher trophic levels, with the possible exception of some fish species (e.g. fourhorn sculpin) and the isopod Saduria entomon which feed extensively on benthic polychaetes (LGL and ESL 1981).

Benthic infauna such as polychaetes promote exposure of sediment-bound hydrocarbons to bacterial degradation by overturning sediments during burrowing and feeding activities (Lee 1976; Gordon et al. 1978; Gardner et al. 1979). In addition to promoting bacterial breakdown of sedimented oil, some polychaetes are capable of degrading certain petroleum hydrocarbons themselves. For example, Capitella capitata and two Nereis species have been shown to accumulate and metabolize benz(a)anthracene from sediment, and degradative enzyme activity in these polychaetes can be induced by exposure to oil-contaminated sediments (Lee et al. 1976, cited in Lee 1976). Rossi (1977) reported that Neanthes sp. also accumulates, metabolizes and depurates naphthalene. In addition, the latter author found that petroleum bound to sediment particles or particulate organic matter was less available to the polychaetes than hydrocarbons dissolved in interstitial water.

Sublethal effects of hydrocarbon exposure on marine polychaetes and oligochaetes have also been observed in the laboratory. As in the case of acute toxic effects of hydrocarbons, the oil concentration at which sublethal effects have been documented varies greatly between species. For example, the polychaete Nereis diversicolor developed convulsions followed by proboscis eversion, immobility and flaccidity at crude oil concentrations of 7 to 20 ppm, while the oligochaete Nais elinguis was quickly killed at hydrocarbon concentrations greater than 3 ppm (Kasymov and Aliev 1973). On the other hand, reproduction of Ctenodrillus serratus and Ophryotrocha sp. was not significantly impaired until the concentration of South Louisiana crude oil reached 9.9 ppm (Carr and Reish 1977). The most frequently observed behavioural and physiological effects of oil have been immobilization, narcosis, disorientation, convulsions, eversion of the proboscis, reduced feeding, growth and reproduction, and the uptake and depuration of naphthalenes (Chia 1973; Kasymov and Aliev 1973; Akesson 1975; Prouse and Gordon 1976; Carr and Reish 1977; Rossi and Anderson 1978; Lyles 1979). Annelid genera which have shown one or more of these sublethal effects include Nereis, Serpula, Nais, Arenicola, Cirriformia, Cirratulus, Ophryotrocha, Ctenodrilis and Neanthes.

Oil spills in the Beaufort Sea would likely have greater impacts on benthic polychaetes than effects documented in temperate latitudes due to their greater representation in the benthic infauna, predominance of fine-grained substrates which would tend to retain sedimented oil, and the generally slower growth rates of arctic benthic fauna. The potential for serious and long term impacts on marine annelids of the Beaufort Sea is particularly high due to the shallow nature of most areas; these shallow depths would increase the likelihood of considerable quantities of oil reaching the substrate. Furthermore, it is believed that the high sediment load of the Mackenzie River would increase the amount of oil reaching some nearshore substrates (Duval et al. 1978) due to the tendency for emulsified oil to become adsorbed to and sedimented with inorganic particulate matter. The slow rates of microbial degradation of oil in arctic environments may ultimately retard the rate of recovery of marine polychaetes following oil spills, and this persistence, in conjunction with slow growth rates, may result in long-term impacts. However, it is noteworthy that two of the three polychaete species which have been shown to be resistant to hydrocarbon pollution (Capitella capitata and Cirratulus cirratus) are also found in the Beaufort Sea (Wacasey 1975), and these annelids could increase in abundance as a result of oil spills (e.g. FLORIDA; Sanders 1978). Nevertheless, except for some fish species such as the fourhorn sculpin and the isopod Saduria entomon which feed extensively on benthic polychaetes (LGL and ESL 1981), relatively minor indirect impacts on members of higher trophic levels would result from reductions or increases in the abundance of infaunal annelids, and therefore the degree of concern regarding adverse effects of spills and blowouts on these fauna is considered MINOR to MODERATE.

5.2.10.3 Crustaceans

Crustaceans are the most common epifauna in the Beaufort Sea, and generally tend to be one of the most sensitive groups of benthic invertebrates to oil (Percy 1980). Mortality of marine and estuarine crustaceans has occurred following all sizes of refined fuel, bunker and crude oil spills, although the most serious and long-term damage has resulted from spills of refined petroleum products, in particular No. 2 diesel fuel (Duval et al. 1981). Mortality has been greatest when oil has been stranded in the intertidal zone or in shallow subtidal areas with fine sediments, where the most frequently affected benthic crustaceans have been crabs, shrimp, lobsters and amphipods. Intertidal organisms such as crabs, barnacles, isopods, amphipods and sandy beach meiofauna have suffered the greatest losses following oil spills, and in some cases, up to 100 percent mortality of crustaceans has occurred (e.g. TSESIS). However, there are few intertidal crustaceans in the Beaufort Sea during the open water season and none in winter.

Differences in the sensitivity of benthic crustaceans to petroleum hydrocarbons have been confirmed by both laboratory investigations (Craddock 1977; Percy 1980) and field observations available in the oil spill case history literature. For example, Sanborn and Malins (1977) report that hydrocarbon concentrations as low as 8-12 ppb are lethal to shrimp and crab larvae, while adult shore crabs (Hemigrapsus spp.) have survived more than 1 day after immersion in 100 percent fuel oil for short periods (Cardwell 1973). A number of authors have suggested that different permeabilities of arthropod exoskeletons between species and in a given species at different ontogenetic and moult stages may account for much of the variability in the sensitivity of this group to petroleum hydrocarbons (e.g. Craddock 1977). Several studies have shown that moulting crustaceans are more sensitive to hydrocarbons than those in the intermoult periods. The larvae of Pandalus hypsinotus were most susceptible to the water-soluble fraction of Cook Inlet crude oil during moulting (Rice et al. 1976) and juvenile Tanner crabs (Chionecetes sp.) exposed to crude oil autotomized their legs during or soon after moulting (Karinen and Rice 1974). On the other hand, Percy (1978) found that Mesidotea entomon was not more sensitive to oil exposure during its moult.

The results of numerous laboratory investigations and oil spill case histories from elsewhere suggest that the impacts of spills on arctic benthic crustaceans would also vary with season, oil type and circumstances surrounding the events. Recent work by Schneider (1980) showed that three of the common inshore epifaunal crustacean species (the amphipods Anonyx nugax and Boeckosimus affinis, and the mysid shrimp Mysis littoralis) were very sensitive to dispersions of Prudhoe Bay crude oil, particularly at elevated salinities where the animals are already experiencing stress. He suggested that an oil spill in shallow Beaufort Sea waters during winter when populations are sometimes stressed by hypersaline conditions may have a significant impact on the growth of and energy flow through these epibenthic species. However, sedimentation of oil would be a prerequisite for these effects, and experimental subsea blowouts indicate that most oil rises to the water/ice interface and is encapsuled in the growing ice sheet rather than sedimented. Mysis littoralis also appears to be quite sensitive to medium crude oil dispersions of 100 µl oil in 500 ml seawater at 32 $^{o}/oo$ salinity. As salinity stress increases to 40 and 50 $^{o}/oo$, light oil dispersions of 25 µl oil/500 ml seawater become lethal.

The presence of high suspended sediment concentrations in nearshore waters of the Beaufort Sea during the spring and summer could increase the amount of oil reaching bottom habitats due to sedimentation, and increase the degree of impact of an oil spill on benthic crustaceans in these areas. Several species are known to be indiscriminate particle feeders (Schneider and Koch 1979), and may accidentally ingest oil. Atlas et al. (1978) reported that after 60 days, five species of Beaufort Sea amphipods did not colonize oil-contaminated sediments as frequently as control sediments, but that the isopod Saduria entomon was found with equal frequency in both sediment types. Percy and Mullin (1975) also reported that this species showed little preference for either oil-contaminated or uncontaminated sediments, and was extremely resistant to crude oil dispersions.

Mortality of amphipods and perhaps other crustaceans would probably be highest as a result of refined fuel spills. For example, Lee et al. (1977b) found that No. 2 fuel oil was approximately three times more toxic to amphipods (Gammarus mucronatus and Amphithoe valida) than crude oil, while temperature, degree of weathering and the presence of sand substrates were shown to affect the toxicity of No. 2 fuel oil to another amphipod, Neohaustorius schmitzi (Michael and Brown 1978).

Impacts of oil spills on benthic crustaceans in arctic environments would likely be greatest during periods when larval forms are most abundant since it is generally believed that juvenile crustaceans are more sensitive to petroleum hydrocarbons than adults (Craddock 1977). For example, Linden (1976) found that the amphipod Gammarus oceanicus has larvae which are several hundred times more sensitive to acute oil exposure than adults. Those species whose young develop in shallow portions of the Beaufort Sea could be more susceptible to oil spills and blowouts due to the probable persistence of oil in these habitats. The isopod Mesidotea entomon is included in this group since small juveniles are only found in abundance close to shore (Bray 1962, cited in Percy 1978).

Numerous sublethal effects of oil exposure on benthic amphipods, isopods and mysids may also occur in arctic environments. Common physiological and behavioural effects observed with these organisms have included altered respiration, assimilation and excretion rates, uptake of aromatic hydrocarbons (e.g. naphthalenes), decreased activity and locomotion, decreased frequency of moulting, disruption of osmoregulation, reduced rates of development and brood size, narcosis, decreased burrowing activity, decreased chemoreception during feeding and reproduction, avoidance responses, reduced feeding rates and reduced reproductive success (Anderson et al. 1974a, b, 1979; Milovidova 1974; Percy and Mullin 1975; Johnson 1977; Percy 1977, 1978; Ott et al. 1978; Duval and Fink 1980; Duval et al. 1980 and others). Although many of these sublethal effects have been shown to disappear once oil concentrations in the water are reduced (Duval et al. 1980), relatively long-term sublethal effects may occur when oil persists in sediments. Of greatest concern in this regard are those sublethal effects which may be considered ecologically significant (Percy 1980); these effects include decreased growth, delayed moulting, decreased carbon assimilation and any interference with reproductive processes or success.

Several studies have examined the behavioural response of common amphipods and isopods from the Beaufort Sea to oil. In general, amphipods appear to be capable of sensing and avoiding petroleum hydrocarbons, while isopods do not seem to have this ability. In a laboratory study, the amphipods Onisimus affinis and Gammarus oceanicus generally avoided oil masses and showed a preference for untainted over oiled food, while the isopod Mesidotea (Saduria) entomon did not avoid oil masses and readily consumed both food types (Percy and Mullin 1975; Percy 1976). In addition, O. affinis showed a preference for clean rather than oil-contaminated sediments, while the amphipod Corophium clarencense and the isopods M. entomon and M. sibirica did not react to the presence of oil in sediments (Percy 1977). Atlas et al. (1978) also reported that amphipods (but not isopods) avoided oil-contaminated sediments resulting from a simulated spill of Prudhoe Bay crude. Experiments with amphipods and isopods from temperate environments also suggest that amphipods may be more sensitive than isopods to oil. Oil exposure decreased the feeding rates of an amphipod (Gammarus sp.) and isopod (Idotea sp.) from the Black Sea, but the effect was most pronounced with amphipods (Milovidova 1974). Duval and Fink (1980) also examined the sublethal effects of oil exposure on an intertidal amphipod (Anisogammarus confervicolus) and isopod (Exosphaeroma oregonensis), and found that water-soluble hydrocarbons not only caused a greater number of sublethal physiological and behavioural effects in the amphipod, but also caused them at lower concentrations.

As indicated earlier, the most serious impacts of an oil spill in the Beaufort Sea would result when shallow nearshore habitats are contaminated with oil, particularly areas less than 5 m deep which are important summer feeding and rearing habitats for anadromous fish species (LGL and ESL 1981). The potential effects of oil spills or blowouts on crustaceans in the Beaufort Sea would tend to be greatest if oceanographic conditions promoted the dispersion of oil throughout the water column, since the concentrations of soluble oil fractions and the opportunity for incorporation of petroleum hydrocarbons into sediments would both be higher. Some epibenthic crustaceans in intertidal areas during the summer could also be exposed to floating oil slicks and would be susceptible to mechanical fouling and smothering. An oil spill in this region could result in decreased abundance of the amphipods Boeckosimus affinis, Onisimus glacialis and Pontoporeia affinis, and the mysids Mysis femorata and M. relicta, which are important food items in the diet of species of fish, birds and marine mammals. Depending on the extensiveness of the habitats contaminated with oil, the degree of concern regarding adverse impacts of spills and blowouts on these species could range from MINOR to MODERATE. The case histories of past spills also indicate that recovery of affected communities could require a decade or more, with the rate of recovery being dependent on the dissipation of oil from contaminated substrates. Benthic crustaceans from adjacent uncontaminated areas would gradually recolonize affected areas, with the oil resistant groups such as isopods returning to their normal abundance and distribution before sensitive juvenile stages and adult amphipods and mysids.

5.2.10.4 Molluscs

The effects of petroleum hydrocarbons on gastropods and bivalves have been well documented in both laboratory investigations and oil spill case histories. Since adult bivalves are generally either sessile or sedentary organisms compared to the free-living epibenthic gastropods, they have often been more severely affected by oil spills which contaminate soft sediment habitats (Duval et al. 1981). For example, there was extensive mortality of the clam Mya arenaria following the grounding of the tanker ARROW in Chedabucto Bay, Nova Scotia, while the intertidal gastropod Littorina spp. was still relatively abundant in areas that were not heavily contaminated. In a similar manner, major mortality of bivalves (Mytilus) was reported following the Deception Bay and IRINI refined fuel spills, while only sublethal effects, including uptake of petroleum hydrocarbons by bivalves, were reported after the T.T. DRUPA crude oil spill and TSESIS bunker fuel spill. All of these events except the ARROW incident occurred in waters north of 55°N latitude.

The acute toxic and sublethal effects of petroleum hydrocarbons on bivalves and gastropod molluscs were summarized by Craddock (1977) and Johnson (1977). More recently, several authors have noted that mortality of gastropods and particularly bivalves may be delayed after the period of oil exposure (Nunes and Benville 1978; Straughan and Hadley 1978). Umezawa et al. (1976) reported that mortality of larval mussels (Mytilus spp.) and oysters (Crassostrea gigas) may be closely related to their tendency to ingest oil droplets and subsequent inability to eliminate these droplets from the stomach. In addition, the presence of oil in suitable substrates may prevent settling of bivalve larvae, with the degree of reduction in larval settlement being directly proportional to the amount that the oil has weathered (Umezawa et al. 1976; Ho and Karim 1978).

Many sublethal effects of oil have been noted in gastropods, although not with species found in Beaufort Sea waters. These effects have included inability to maintain pedal attachment due to an anesthetic effect of oil on the foot (Dicks 1973), as well as narcosis and decreased responsiveness to external stimuli (Griffith 1970; Ehrsam et al. 1972). Griffith (1970) also reported that the narcotic effect of oil on pedal attachment was reversible, with the rate of narcotization increasing with temperature, but recovery being largely unaffected by temperature. Additional sublethal effects of petroleum hydrocarbons on gastropods have included increased respiration and crawling rates (Hargrave and Newcombe 1973; Linden 1977), reduced activity (Baker 1973), and alteration of chemotactic responses which affect feeding, predation and reproductive success (Blake 1960; Jacobson and Boylan 1973; Brown et al. 1974; Eisler 1975). Similar sublethal effects may be expected in gastropods inhabiting the Beaufort Sea and other arctic waters.

The sublethal effects of oil have been examined in at least four genera of bivalves known to occur in the Beaufort Sea (Macoma, Mya, Mytilus and Pecten spp.). In addition, a wider range of sublethal effects have been observed in bivalves than in gastropods, and many of these behavioural and physiological aberrations may be considered ecologically critical according to criteria described by Percy (1980). Many physiological functions of bivalves are dependent upon the pumping of seawater through the mantle cavity, and virtually all species are able to modify their ventilation rate in the presence of pollutants such as oil (Johnson 1977). Consequently, oil spills or blowouts could affect a number of activities or processes in bivalves including the rate of feeding, oxygen uptake, reproduction and excretion of metabolic wastes. However, species-specific differences in many responses of bivalves to oil exposure have been documented. For example, Swedmark et al. (1973) reported that shell closure in Mytilus edulis was unaffected by exposure to dispersions of Oman crude oil at concentrations of 1000 ppm, while Cardium edule closed their shell for the 96-h exposure period and Pectin opercularis maintained valve closure for only a portion of the treatment. Increases and decreases in pumping rates of bivalves have been observed following exposure to oil. A decrease in oxygen consumption of bivalves exposed to oil has been attributed both to impaired filtration resulting from shell closure and inhibition of gill ciliary action. For example, the respiratory rates of Brachidontes sp. and Donax sp. were depressed by over 50 percent in some cases by light crude oil concentrations of 10,000 and 25,000 ppm (Avolizi and Nuwayhid 1974, cited in Johnson 1977). A crude oil concentration as low as 1 ppm was sufficient to reduce the net carbon flux of Mytilus edulis, while the respiration rate of this species increased in the presence of small amounts of crude oil (Gilfillan 1975, cited in Johnson 1977). If the amount of carbon assimilated falls below levels required for normal maintenance, this may in turn affect subsequent spawning success, since M. edulis that survived the West Falmouth oil spill did not spawn the following year (Johnson 1977). Increased respiration rates were also reported in the clam Mya arenaria after a 3-week period exposure to crude oil (Fong 1976). An anesthetic effect of petroleum hydrocarbons on ciliary activity of bivalves (Galtsoff 1964, cited in Percy and Mullin 1975) may also be responsible for the cessation of regular feeding activity observed in the arctic bivalve Yoldiella intermedia following exposure to the seawater soluble components of crude oil (Percy 1974, cited in Percy and Mullin 1975). A reduction in filtration rate and increased respiration could have resulted in the reduced growth and survival observed by Dow (1975) in M. arenaria within oiled intertidal sediments, as well as in the lowered abundance, shell growth rate and carbon assimilation reported in the same species in Chedabucto Bay, Nova Scotia six years after the grounding of the ARROW (Gilfillan and Vandermeulen 1978).

Other effects of petroleum on bivalves have included gill tissue degeneration (Clark and Finley 1974), avoidance by closure of the valves which may result in the depletion of energy reserves (Dunning and Major 1974), increases in the rate of mucus secretion (Stainken 1976), in situ decreases in growth and production (Dow 1975), inhibited burrowing of unburied bivalves and surfacing of buried organisms (Taylor et al. 1976; Linden 1977), inhibited byssal attachment by mussels (Cardwell 1973; Swedmark et al. 1973; Linden 1977), impaired fertilization and embryological development (NMFS 1978), and various chemosensory and biochemical disorders (e.g. Moore et al. 1978). In addition, several effects of oil on bivalve gametes and larvae have been documented. The water soluble fractions of three crude oil types were toxic to spermatozoa and caused abnormalities in the larvae of Mulinia sp. and Crassostrea sp. (Renzoni 1975). Umezawa et al. (1976) also reported that the settling of bivalve larvae may be prevented by the presence of oil in substrates. Although many sublethal effects have also been shown to be completely reversible, persistent exposure of bivalves to relatively low levels of petroleum hydrocarbons could result in long-term sublethal behavioural and physiological effects which decrease overall productivity and reproductive success.

Oil spills in the Beaufort Sea could result in both acute lethal and chronic sublethal effects on benthic gastropods and bivalves. As in the case of most benthic invertebrates, the severity and duration of these impacts as well as recovery of affected habitats would vary with the amount of oil reaching the substrate and its subsequent rate of dissipation. The most susceptible molluscs in this region would probably be the bivalves, particularly in waters less than 15 m where Macoma balthica, Cyrtodaria kurriana and Yoldiella intermedia are relatively common in fine-grained substrates (Wacasey 1975). Potential damage to molluscs would also likely be greatest in shallow embayments where the sediments may retain the oil longer. Recovery of gastropod populations would likely begin more quickly than bivalves due to their greater mobility and capacity for recruitment from adjacent uncontaminated areas. Depending on the degree of persistence of oil in the sediments, initial recolonization by more oil-resistant gastropod species could begin within 1 to 5 years, while complete recovery of bivalve populations could require a decade or more. Sublethal effects of petroleum hydrocarbons would also gradually disappear as the stranded oil weathered and was dissipated to the surrounding waters. The indirect effects of loss or contamination of molluscs on members of higher trophic levels in the Beaufort Sea would probably be less than if benthic crustaceans were similarly affected, since crustaceans tend to be dominant prey items for a much larger number of fish, birds and marine mammals. Nevertheless, indirect impacts on whitefish, flounders, eiders, oldsquaw, scaup, scoters, bearded seal and walrus could result from decreased abundance or contamination of molluscs in the Beaufort Sea. Consequently, the overall degree of potential concern regarding the effects of oil spills and blowouts on molluscs in the region could vary from MINOR to MODERATE, depending on the extensiveness of the contaminated area and the long-term persistence of oil in fine-grained substrates.

5.2.10.5 Echinoderms

Unlike other groups of benthic invertebrates, the toxic and sublethal effects of petroleum hydrocarbons on echinoderms have only been the subject of a few laboratory investigations. However, mortality or damage to sea urchins or starfish was documented in 15 spills examined by Duval et al. (1981). None of these spills occurred in waters north of 55°N latitude. Sublethal effects of petroleum hydrocarbons on echinoderms observed in the laboratory or field have included increased oxygen consumption (Yentsch et al. 1973), retraction of tube feet which can result in loss of adherence to the substrate (North et al. 1965), decreased responsiveness to chemicals released by prey and reduced rates of predation (Crapp 1971; Zafiriou et al. 1972), interference with chemosensory defensive behaviour (Johnson 1979), decreased fertilization rates and developmental abnormalities (Lonning and Hagstrom 1975).

An oil spill or blowout in the Beaufort Sea would probably not have as serious an impact on echinoderms as on other major invertebrate groups in this region. Firstly, echinoderms are generally absent from waters less than 15 m deep (Wacasey 1975) where much of the oil would be expected to be sedimented, and secondly, the relatively mobile and epibenthic nature of these fauna would facilitate recolonization from adjacent unaffected habitats. In addition, echinoderms are not dominant prey items for vertebrates in the Beaufort region, thereby minimizing potential indirect impacts on other members of the food web. Consequently, the degree of concern regarding adverse impacts of oil spills or blowouts on this group of benthic invertebrates is considered MINOR.

5.2.11 Effects of Oil on Epontic Organisms

Some effects of petroleum hydrocarbons on the flora and fauna associated with the under-ice surface in arctic waters have been recently documented following simulated oil spills in the field. Acreman et al. (1980) observed abnormal deposits and possibly oil or tar particles inside diatoms as well as extensive diatom mortality in a limited number of core samples taken during a simulated under-ice oil blowout experiment near McKinley Bay, Northwest Territories. These effects could be of considerable significance in view of the apparent importance of this community to the seasonal productivity of areas such as the Beaufort Sea (LGL and ESL 1981), as well as many eastern Arctic waters (LGL 1981). Epontic organisms would be particularly vulnerable in the event of a winter subsea blowout. However, oil trapped in pockets under the ice surface would become rapidly encapsulated and only be in contact with the epontic community for a relatively short period.

It is probable that when initially trapped beneath the ice, the oil would result in virtually complete mortality of epontic flora and fauna. The spatial extent of this mortality would depend on the duration of the blowout, the degree of irregularity of the under-ice surface, under-ice currents and the rate of encapsulation of the oil into the ice sheet. Although the presence of pockets, ridges and other irregularities on the under-ice surface would tend to slow the horizontal spread of trapped oil, a blowout which continued for several weeks or months could eventually contaminate extensive areas and result in widespread mortality of epontic organisms. Unlike the situation with planktonic organisms, there would be little or no opportunity for recruitment of flora and fauna from adjacent uncontaminated areas. The impacts of an oil blowout on epontic communities would be most serious in late spring and early summer prior to breakup since maximum floral and faunal abundance occurs at this time, and encapsulation of oil in the ice would be less likely. In addition, a blowout would have further indirect impacts on members of higher trophic levels which feed on species present in the epontic community. These indirect impacts could be relatively serious depending on the extensiveness of the area contaminated with oil and the total loss of epontic flora and fauna. In addition, any oil remaining trapped beneath the ice at breakup would represent a source of further contamination of pelagic habitats. Consequently, the overall degree of concern regarding the potential effects of oil blowouts on epontic flora and fauna is likely to be MODERATE, while the effects of non-continuous events are considered of MINOR concern.

5.2.12 Effects of Oil on Terrestrial Plant Communities

In the low-lying coastal areas of the Beaufort Sea, the potential for contamination of shoreline and backshore plant communities may be somewhat greater than in southern latitudes because of the occurrence of storm surges and high onshore winds (Owens 1977). For example, driftwood in McKinley Bay (Tuktoyaktuk Peninsula) has been observed 2.4 to 2.7 m above mean sea level and from 90 to 300 m inland (MacKay 1974). Industry videotape records of the entire Beaufort Sea coastline from Herschel Island to Cape Bathurst (Woodward Clyde Consultants 1980) indicate many similar locations where oil could be deposited up to 500 m into low-lying backshore environments under certain conditions. However, the potential for contamination of terrestrial plant communities is generally limited to the brief open water season, which usually occurs between July and October. Owens (1977) suggested that oil deposited in areas above those affected by normal wave activity would be weathered and dispersed very slowly due to the low energy levels in such environments. The slow rates of microbial degradation in the Arctic (Westlake 1981) would also contribute to the persistence of oil in backshore substrates.

The oil spill case history literature reviewed by Duval et al. (1981) is only of limited use for assessment of the potential impacts of petroleum hydrocarbons on arctic backshore vegetation. The TORREY CANYON, AMOCO CADIZ and ESSO ESSEN spills all resulted in damage or mortality of terrestrial plants, with recovery being evident after a few weeks in the latter spill (Stander and Venter 1968) but requiring several years in the case of the TORREY CANYON grounding (Southward and Southward 1978). The biological impacts of oil spills on arctic terrestrial vegetation would undoubtedly be greater than those observed following these temperate latitude spills because of the characteristic low-lying tundra, the predominance of mosses and lichens, and the presence of permafrost. As a result, the accidental and simulated spills which have occurred in terrestrial environments of Alaska and the Yukon are a better predictive tool for assessment of potential impacts of oil on backshore vegetation along the Beaufort Sea coastline.

Hunt et al. (1973) reviewed the biological effects of various terrestrial oil spills in Alaska and the Yukon, and reported extensive mortality of vegetation following several spills from ruptured pipelines.

These authors reported little evidence of recovery of terrestrial vegetation at sites of either JP-4 or diesel fuel spills after 5 to 15 years. The highest recovery occurred on hummocks and in areas where abundant rainfall leached the petroleum hydrocarbons from the soil. Similar contact mortality and slow recovery of terrestrial vegetation was also observed after a diesel spill in the Yukon (Swader 1975). The slow recovery following this event was again related to the persistence of petroleum hydrocarbons in the soil. In another study, Atlas et al. (1976) reported that terrestrial habitats near Prudhoe Bay, Alaska where natural oil seepage occurred were virtually devoid of vascular plants. Experimental applications of crude oil have also caused widespread mortality and subsequent slow recovery of arctic terrestrial plant communities (e.g. Hutchinson and Freedman 1975), as well as emergent plants and rooted macrophytes along the margins of a subarctic lake (Hellebust et al. 1975). A relatively large number of laboratory studies have indicated that petroleum hydrocarbons affect vascular plants in several ways, including decreased root and shoot growth, delayed and reduced flowering, and disruption of respiratory and photosynthetic processes (Baker 1970, 1971; Blankenship and Larson 1978), while van Overbeek and Blondeau (1954) reported that the primary toxic effect of hydrocarbons on plants was due to disruption of the lipid bilayer of the cell membrane.

Another potential concern with respect to arctic backshore plant communities is damage to permafrost integrity by the oil, or more often, cleanup activities. Increased thaw and/or slope instability can result from the decreased albedo associated with oil contamination. Soil types which are especially susceptible to permafrost degradation are ice-rich silty substrates which show very low cohesiveness when thawed (Hunt et al. 1973). However, these authors reported that permafrost was not affected at those oiled sites where the hydrocarbons did not penetrate the surface vegetation (moss) mat or the substrate was not disturbed by the use of heavy cleanup equipment.

Oil spills or blowouts in the Beaufort Sea which subsequently contaminate backshore tundra environments would likely cause mortality of most if not all vegetation contacting the oil. The damage would likely be greater with fresh crude or refined fuels due to their relatively high toxicity (Craddock 1977), particularly if jet fuel (JP-4 or JP-5), gasoline or No. 2 diesel were directly spilled in shoreline environments. The results of studies conducted after simulated or accidental spills in terrestrial environments clearly indicate that damage would be relatively long-term, requiring one or more decades for recovery of heavily contaminated sites. In addition, the area of impact of marine spills on backshore vegetation in this region could extend inland up to 500 m from the shoreline. Impacts on vegetation in this zone would probably be increased if the cleanup response involved the use of heavy equipment since this could increase the degree of penetration of oil into the substrate. Recovery would probably be most rapid in areas where the oil was left to naturally weather and disperse, although this could increase the risk of additional and relatively long-term indirect impacts on birds and mammals utilizing these habitats. However, since the amount of terrestrial vegetation affected by a blowout or oil spill in the marine environment would be relatively small, the overall degree of concern regarding this long-term damage to coastal plant communities is considered MINOR.

5.2.13 Effects of Oil on Micro-organisms and the Role of Oleoclastic Bacteria in Oil Degradation

Oleoclastic bacteria utilize petroleum hydrocarbons as a substrate, and play an important role in the long-term degradation of oil after initial losses due to evaporation, dissolution, photolysis and metabolism by biota (Kallio 1976; Vandermeulen 1981). The rate of oil degradation by oleoclastic bacteria is dependent on a number of factors including the nature and concentration of oil, the availability of oxygen and nutrients, temperature, and the type and abundance of available micro-organisms (Mackay 1981). In addition, the ability of some bacteria to metabolize hydrocarbons is readily lost when petroleum is not present in the environment (Westlake 1981).

Lee (1980) reports that most biodegradation occurs once oil is dispersed into fine droplets in the water column or deposited on the substrate. Dissolved hydrocarbons are apparently adsorbed to detrital particles, and these particles support bacteria which are attached by pads and fibrillar appendages (Lee 1977, cited in Lee 1980). The surfaces of such bacterial-coated particles are expected to be hydrophobic and provide areas for hydrocarbon adsorption (Lee 1980).

Petroleum hydrocarbons present in crude oils and refined fuels vary in their susceptibility to degradation by oleoclastic bacteria. Straight-chain alkanes are rapidly attacked by bacteria, while branched alkanes, cycloalkanes and aromatic hydrocarbons are metabolized at slower rates and high boiling point compounds such as asphaltenes resist degradation (Walker et al. 1973, cited in Lee 1980). Petroleum hydrocarbons containing nitrogen, sulfur or oxygen groups are also more resistant to microbial degradation. This variability in hydrocarbon degradation has also been documented following oil spills. For example, microbial breakdown of bunker C fuel spilled in March from the POTOMAC off western Greenland proceeded very slowly and there was no significant increase in numbers of oleoclastic bacteria in the ice-laden waters within a few weeks after the 100,000 gal spill (Grouse et al. 1979). Cretney et al. (1978) also investigated the long-term fate of 180 tons of heavy fuel oil spilled in a British Columbia coastal bay in 1973. They found almost complete removal of n-alkanes by biological activity during the first year after the spill. Pristane and phytane were biodegraded more slowly, but were virtually absent after 4 years. The non-n-alkanes of the C_{28} to C_{30} range were the most persistent petroleum hydrocarbons. A pipeline rupture in March 1971 caused a small amount of JP-4 jet fuel and No. 2 fuel oil to enter the intertidal zone of a marine cove at Searsport, Maine (Mayo et al. 1978). After five years, investigators found that the average area contained roughly 20 percent less hydrocarbon than immediately after the spill. An analysis of this data by Atlas (1981) suggested that microbial action was greatly suppressed in the fuel residues absorbed on the cold clay substrate, likely as a result of a lack of available oxygen. Pierce et al. (1975) investigated a spill of No. 6 fuel oil (bunker C) on an estuarine beach in Rhode Island, and found that hydrocarbon concentrations in the mid-tide region declined as the abundance of oleoclastic bacteria increased. However, rates of degradation were less than 1 μg of hydrocarbon per g (dry wt.) of sediment per day during the winter months. Blumer and Sass (1972) examined the fate of a major No. 2 fuel oil (diesel) spill in Massachusetts and reported a slow disappearance rate of hydrocarbons, although bacterial degradation contributed to the removal of n-paraffins from contaminated areas. Teal et al. (1978) attributed the disappearance of simple naphthalenes from the sediments of Buzzards Bay, Massachusetts following both major spills in this harbour to microbial degradation, while the more substituted aromatics appeared more resistant to biodegradation.

Bunch and Harland (1976) reported that oleoclastic bacterial species were ubiquitous in the waters and nearshore sediments of the Beaufort Sea, but were not present in offshore sediments. The proportion of oil-degrading bacteria in non-polluted arctic habitats generally appears to be low (about 1 percent) (Atlas et al. 1978; Eimhjellen et al. 1981), although the composition of the bacterial community is expected to differ between sites and at a given site with season (Westlake 1980). Several studies of bacteria in the Arctic have shown that both the absolute abundance and relative proportion of oleoclastic bacteria increase in areas with oil contamination. Eimhjellen et al. (1981) found that the number of oil-degrading bacteria increased from about 10^2 to 10^5 cells per ml of sand and oil mixture 15 days after beach sand was sprayed with crude oil. Numbers of oleoclastic bacteria reached 10^7 cells/ml in another plot sprayed with 50 percent emulsified crude. Similarly, Atlas et al. (1978) found that the number of oil-degrading micro-organisms in open water increased from 1.5 to 50 percent 30 days after simulated crude oil spills, while numbers over and under the ice increased to 0.03 percent and 0.5 percent, respectively.

Low nutrient levels, in particular phosphate or nitrate, may limit the growth of oleoclastic bacteria following oil spills in the Beaufort Sea. Although nutrients used during oil degradation by bacteria in the water column would be replenished from surrounding waters, nutrients in sub-surface sediments may be depleted by bacterial activity. For example, Bunch and Harland (1976) reported that degradation of the aliphatic fraction of 200 mg of Norman Wells crude oil at a depth of 1 m in the water column required up to 2.3 μg-at/L of nitrate and 0.36 μg-at/L of phosphate. Westlake and Cook (unpubl. manuscript) found very little oleoclastic bacterial activity in composite beach and intertidal sediment samples from the Beaufort Sea until nitrates and phosphates were added. In addition, degradation of petroleum hydrocarbons by oleoclastic bacteria in this region would be limited by low temperatures throughout much of the year, although hydrocarbon degradation has been documented at temperatures between 0° and 5°C (Bunch and Harland 1976).

Atlas et al. (1978) suggest that little or no oleoclastic activity would be expected under ice, while oil encapsulated within the ice cover would also not be significantly degraded because bacteria frozen into the ice matrix are unlikely to grow (Kaneko et al. 1977). The major site of petroleum biodegradation in this region would probably be the few centimeters of the bottom sediments where oxygen and nutrients are available, and benthic macrofauna and meiofauna continuously expose underlying sediments during burrowing activities. Natural processes such as ice scour and wave action would also expose oil present within the substrate of relatively shallow areas (<50 m). On the other hand, oil which is buried by sediment transported into coastal environments of the Beaufort Sea by the Mackenzie River would likely persist in relatively unaltered form for decades once it is covered by sufficient sediment to be within an anoxic environment. Consequently, although oleoclastic bacteria do occur in the Beaufort Sea region, the types of habitats where oil degradation could take place and the actual rates of biodegradation would both be limited and therefore inadequate to remove most of the petroleum hydrocarbons which could enter the marine environment due to a major oil spill or blowout (Vandermeulen 1978; Atlas 1981).

The effects of petroleum hydrocarbons on non-oleoclastic bacteria are not well documented, and existing data tend to show variable responses. Atlas et al. (1978) found that petroleum hydrocarbons generally caused an increase in the numbers of both viable heterotrophic and oil-degrading bacteria, while Bunch and Harland (1976) suggest that a moderate input of petroleum to the marine environment may not affect the activity of non-oleoclastic micro-organisms. Kator and Herwig (1977) reported that the abundance of chitinolytic and cellulytic bacteria in oiled and exposed plots within a Virginia tidal mrash were not significantly different. On the other hand, Walker et al. (1974, 1975) concluded that both crude oil and No. 2 fuel were toxic to ecologically important lytic bacteria from a Chesapeake estuary, with the fuel oil being more toxic than the crude oil.

Many species of marine bacteria are motile and locate their substrate by positive chemotaxis (Mitchell et al. 1972). Walsh and Mitchell (1973, cited in Johnson 1977) estimated that the chemotaxis of marine bacteria (primarily Pseudomonas spp.) was reduced 50 percent by only 10 ppm of Kuwait crude oil, and suggested that this inhibition was due to blockage of the chemoreceptors. Although this effect is sublethal and reversible, Johnson (1977) suggests that bacteria unable to detect a suitable substrate would have a limited capacity to degrade material. Nevertheless, there is presently no evidence to suggest that there is more than a NEGLIGIBLE degree of concern regarding adverse effects of oil spills or blowouts on non-oleoclastic bacteria in the Beaufort Sea.

5.2.14 Summary of Concerns Related to Oil Spills and Blowouts

The environmental impacts of oil spills and blowouts are undoubtedly the greatest area of potential concern associated with hydrocarbon exploration and production in the Beaufort Sea region. As in the case of most past spills in marine waters, the severity of these impacts would depend on the types of habitats affected by dissolved, dispersed and surface oil, as well as the species and life history stages of biological resources utilizing these habitats during or following spills. The degree of concern regarding the effects of oil spills on some resources would also depend on the amount of oil which becomes trapped in marine sediments, while the rate of recovery of these resources would depend on the persistence of petroleum hydrocarbons in the environment. Oil well blowouts would also generally pose a greater threat to marine resources of this region than single discontinuous releases of crude oil or refined fuels due to the potential for contamination of more extensive areas during blowouts, particularly during the open water season when oil movement would not be restricted by the presence of an ice cover and more resources (particularly birds, marine mammals and anadromous fish) would be present in the region.

Table 5.2-10 summarizes the nature of the potential impacts and degree of regional concern associated with oil spills and various marine and coastal resources of the Beaufort Sea. In general, the degree of concern would range from MINOR to MODERATE for most resources depending on the season of the spill and the extensiveness and location of areas affected by oil. In other cases, the degree of concern would also vary with the species exposed to surface slicks or oil present in the water column since some populations have been shown to be particularly vulnerable to oil exposure, while others are highly sensitive and exhibit a variety of lethal or sublethal effects once they contact petroleum hydrocarbons. For example, highly aquatic birds that dive and forage at sea, including alcids, diving ducks, loons, fulmars, cormorants, gulls, shearwaters, kittiwakes and phalaropes, are vulnerable to oil exposure, while other marine species such as some crustaceans and benthic molluscs are sensitive to oil exposure since low concentrations of dissolved or dispersed oil have been shown to be acutely toxic. As indicated in Table 5.2-10, the degree of concern regarding adverse impacts of oil spills on some species of birds could be MAJOR under circumstances where a regional population is affected sufficiently to cause a decrease in abundance and/or a change in distribution which persists for several generations. Species which could be most severely affected by oil spills or blowouts in the Beaufort Sea include the various alcids (e.g. black guillemot, thick-billed murre), black brant and common eider (ESL 1982), although depending on the circumstances surrounding any spill and the proportion of the regional populations affected, the degree of concern regarding effects of spills or blowouts could also be MODERATE to MAJOR for the white-fronted goose, scaup and scoters, oldsquaw, king eider, glaucous gull, Arctic tern and several shorebirds.

TABLE 5.2-10

SUMMARY OF POTENTIAL CONCERNS REGARDING OIL SPILLS AND BLOWOUTS AND REFINED FUEL SPILLS IN THE BEAUFORT SEA REGION

Environmental Component or Resource	Potential or Probable Effects[1]					Degree of Potential Concern[2]
	M	S	H	C	F	
Bowhead whale		X		X	X	NEGLIGIBLE to MINOR
White whale	X	X		X	X	NEGLIGIBLE to MINOR
Ringed and bearded seals	X	X	X	X	X	MODERATE
Polar bear	X	X		X	X	MINOR to MODERATE
Arctic fox		X		X	X	MINOR
Birds	X	X	X	X	X	MINOR to MAJOR
Fish	X	X	X	X	X	MINOR to MODERATE
Icthyoplankton	X	X		X	X	MINOR to MODERATE
Phytoplankton	X	X		X		MINOR to MODERATE
Zooplankton	X	X		X	X	MINOR to MODERATE
Benthic macrophytes	X	X	X	X		MINOR
Benthic microalgae	X	X	X	X		MINOR to MODERATE
Annelids	X	X	X	X	X	MINOR to MODERATE
Crustaceans	X	X	X	X	X	MINOR to MODERATE
Molluscs	X	X	X	X	X	MINOR to MODERATE
Echinoderms	X	X	X	X	X	MINOR
Epontic communities	X	X	X	X	X	MINOR to MODERATE
Terrestrial vegetation	X	X	X	X		MINOR
Micro-organisms	X	X	X			NEGLIGIBLE

1 M = mortality
S = sublethal effects
H = habitat loss
C = contamination/fouling
F = loss of food

2 Dependent on species, time of year, location and extent of the area affected by oil (see text).

5.3 CHEMICALLY DISPERSED OIL

5.3.1 Introduction

The primary consideration favouring the use of chemical dispersants following oil spills is the removal or reduction of the threat posed by shoreward movement of floating oil towards environmentally sensitive coastal resources. Chemical dispersants increase the natural rate of oil dispersion by lowering the interfacial tension between oil and water and causing the formation of minute droplets of oil in the water column (Canevari 1978). The reduction in the amount of oil on the water surface as a result of dispersant application tends to decrease the potential vulnerability of waterfowl, mammals and intertidal communities to oil exposure, while increasing, at least initially, the potential risk of impacts on pelagic, planktonic, and benthic communities. For example, Brown et al. (1978) reported that the concentration of oil immediately below the surface of a chemically dispersed oil slick in an experimental water column was 27 times higher than below a non-dispersed slick. It must also be recognized, however, that oil efficiently dispersed throughout the water column is subject to more rapid physico-chemical and biological degradation than oil concentrated in a floating surface layer (Nelson-Smith 1978).

5.3.2 Oil Spill Dispersants

The majority of the oil dispersants currently in use are mixtures of nonionic surface acting agents (surfactants) and either a hydrocarbon or some other solvent base. Historically, chemical dispersants have progressed from highly toxic petroleum-based mixtures which were initially designed as general purpose industrial solvent-emulsifiers to highly specialized "third generation" self-mixing dispersant concentrates of low to medium toxicity. This development has resulted in a shift from dispersants where the aromatic fraction of the solvent was the primary cause of toxicity, to dispersants where most of the toxicity is associated with the surfactant itself (Nelson-Smith 1978).

The principle behind chemical dispersion of floating oil can be summarized as follows. A typical surfactant molecule in a dispersant consists of two functional groups, a hydrophilic or "water seeking" group, and a hydrophobic or lipophilic group. When applied to an oil slick the lipophilic end of the molecule is attracted to, and becomes embedded in the oil, leaving the hydrophilic end exposed to the surrounding water. This effectively lowers the interfacial tension between the oil and the water, and promotes the formation of small dispersant-coated oil droplets (Canevari 1978). The balance of forces between the two functional groups affects the choice of solvent, the degree of solubility of the dispersant in oil or water, and the overall efficiency with which the dispersant causes the oil to break up into droplets.

5.3.3 Dispersant Toxicity

In any dispersant application program, a significant proportion of the mixture is added directly to the water column, either because the oil slick itself is discontinuous or because the application method results in the spread of the chemical beyond the slick margin. Aerial dispersant applications are particularly prone to situations of the latter type. For example, Gill and Ross (1980) report that as much as 50 percent of the dispersant released from an aircraft may fall outside the intended dispersal zone, depending on the size of aircraft used, the height from which spraying takes place, the size range of dispersant droplets, and the prevailing meterological conditions. Dispersant application from boats where forward visibility of the slick is restricted and where the passage of the vessel temporarily disrupts the distribution of surface oil can also lead to a direct loss of dispersant to the water column.

Although modern dispersants are sometimes considered virtually non-toxic to marine life compared to oil or oil/dispersant mixtures (Canevari 1977,1978), there is considerable variability in the toxicity of dispersants to different classes and life cycle stages of marine organisms (Swedmark *et al.* 1973; Wells and Harris 1980; Percy 1980; Wells 1980). Toxic mechanisms of nonionic surfactants probably relate to their interaction with biological membrane systems (Nelson-Smith 1978). For example, much of the published literature describing the effects of dispersants on marine zooplankton suggests that respiration and neuromuscular systems are particularly sensitive to surfactants (Section 5.3.6), while the membrane regulated fertilization of benthic invertebrate and fish eggs is also inhibited by very low concentrations of some dispersants (Sections 5.3.4 and 5.3.7).

Although the lethal effects of dispersants and dispersant/oil emulsions have been extensively studied (see review by Sprague *et al.* 1980), the long-term sublethal effects of short-term exposure of marine organisms to "second" and "third generation" dispersants have yet to be examined. Consequently, the characterization of these dispersants as essentially non-toxic must be considered somewhat premature. In addition to being chemically toxic, a surface coating of dispersant may alter the physical relationship between an organism and its environment to the extent that survival may be adversely affected (Sections 5.3.5, 5.3.6). For example, there is circumstantial evidence suggesting that low concentrations of dispersant may be accumulated on the feathers of seabirds (Section 5.3.6).

5.3.4 Toxicity of Dispersant/Oil Mixtures

The influence of chemical dispersion on the aquatic toxicity of petroleum hydrocarbons has been a subject of constant review, due at least in part to the historical controversy surrounding the use of chemical dispersants following oil spills (Nelson-Smith 1978). Recent reviews on the topic include those of Sprague et al. (1980) and MacKay and Wells (1981), while a bibliography of dispersant-related literature has been compiled by Doe et al. (1978), and a symposium on various aspects of dispersant use was sponsored by the American Society for Testing and Materials (1978).

Although there is a general concensus that dispersant use increases the amount of dispersed oil below a slick, and further that increased stability of the dispersion will probably result in the exposure of pelagic organisms to higher than usual concentrations of dissolved and particulate oil, there is considerable disagreement concerning the potential effect that the dispersant may have on the toxicity of the oil itself. Despite the relatively low toxicity of most modern dispersants (Swedmark et al. 1973; Cowell 1978; Doe and Wells 1978), laboratory investigations have generally indicated that dispersant/oil mixtures are more toxic to aquatic invertebrates than naturally (physically) dispersed oil (Percy 1980; Wells 1980). On the other hand, Penrose (1980) states that the increased toxicity of oil treated with dispersants to fish "appears to be solely due to the increased solubilization of oil by dispersants", and that "chemically dispersed oil is no more specifically toxic than undispersed oil". Similar views have been expressed by representatives of dispersant manufacturers (Canevari 1977,1978), as well as independent laboratories (Battelle Pacific Northwest Laboratories 1973). It has also been suggested that dispersant-coated oil droplets may preferentially release toxic chemicals normally retained by physically dispersed droplets (see for example, Lonning and Hagstrom 1976), although this has not been demonstrated by existing analytical techniques.

Apart from the apparent increase in the toxicity of chemically dispersed oil relative to physically dispersed oil associated with the increased "available concentration" and duration of exposure, there is little evidence to suggest that the actual toxic mechanism is any different between the two dispersion types. On the contrary, there is considerable evidence that various categories of petroleum-related toxicants including oil, dispersants, and oil/dispersant mixtures may have at least broadly similar sites of toxic action (Percy 1980; Wells 1980). Hutchinson et al. (1979) have proposed a unifying concept of toxicity for low boiling point petroleum hydrocarbons based in part on their partition coefficient between water and n-octanol (representing the partition between water and cellular lipids), and this approach has allowed relatively accurate prediction of the toxicity of previously untested hydrocarbons to phytoplankton. Stegeman and Teal (1973) reported that oysters with high lipid content accumulated more hydrocarbons when exposed to No. 2 fuel than did oysters with a lower lipid content. The fact that cellular "leakage" has been reported both with phytoplankton exposed to low boiling point hydrocarbons (Hutchinson et al. 1979) and intertidal macrophytes exposed to dispersants (Percy 1980) is certainly suggestive of a common mode of action involving disruption of cellular membrane processes.

Attempts to relate the toxicity of petroleum toxicants to various taxonomic categories of marine organisms have often been unsatisfactory (Percy 1980). Other approaches, including examination of habitat type (i.e. intertidal vs. subtidal) and developmental state (i.e. adult vs. larval forms), appear to be of greater immediate value in prediction of possible environmental effects of hydrocarbon exposure (Swedmark et al. 1973; Rice et al. 1977). Examination of the surficial extent and molecular composition of exposed membrane structures such as gills, chemoreceptors and osmoregulatory organs in various marine organisms, and their sensitivity to disruption by different surface acting agents and lipophilic hydrocarbons, may also provide useful information.

5.3.5 Effects of Chemically Dispersed Oil on Marine Mammals

With the exception of studies involving the use of detergents to remove oil from the fur of contaminated sea otters (Williams 1978, cited in Smiley 1980), the effects of exposure of marine mammals to oil spill dispersants or dispersant/oil mixtures have not been examined. The natural skin oils coating the fur of polar bears and fur seals could absorb dispersants directly from the water column (see also Section 5.3.6). The residual hydrophilic nature of the dispersant molecules may then increase the wettability of the fur and ultimately alter its thermal conductivity. However, unlike crude oils or heavy bunker fuels, dispersant by itself is unlikely to cause matting of the hairs and considerable insulative capacity may still be retained by dispersant-coated fur. The extent to which modifications of the thermal properties of skin and fur might affect the survival of exposed marine mammals would likely depend on several factors including air and water temperature and the relative contribution of skin, fur and subcutaneous fat to the insulation of the organism (Smiley 1980).

The biochemical toxicity of dispersant/oil mixtures to marine mammals is not documented. External irritation of eyes and nostrils that has been reported with crude and refined oils (Section 5.2.4.2) would probably be less severe with chemically dispersed oil since the dispersant-coated oil droplets have a much lower tendency to stick to foreign materials (Canevari 1977, 1978). The potential effects of ingested oil/dispersant mixtures on marine mammals are also not known, although the probability for ingestion of large quantities of dispersed oil should be remote following successful dispersant application. Consequently, in the absence of evidence to the contrary, the overall degree of concern regarding exposure of marine mammals to free dispersant or chemically dispersed oil is expected to be MINOR.

5.3.6 Effects of Chemically Dispersed Oil on Birds

Despite the potential for contact between swimming and diving birds and chemically dispersed oil, there is very little information regarding the effects of oil/dispersant mixtures or dispersants on birds. For example, none of the 402 dispersant-related papers compiled by Doe et al. (1978) discuss potential effects of chemically dispersed oil on birds. However, a recent paper by Lambert and Peakall (1981) examined thermoregulatory effects of Prudhoe Bay crude oil, crude oil/Corexit 9527 mixtures, and the Corexit alone on mallard ducks (Anas platyrhynchos). The dosage level in these experiments was 12 ml of oil (air-weathered for 1 hour) dispersed in 60 litres of sea water. Both the crude oil and the crude oil/dispersant mixtures were quickly adsorbed onto the breast feathers of the swimming mallards, and both resulted in significant increases in basal metabolic rates. Metabolic rates increased by up to 16 percent for periods up to 14 days after the initial 1 hour exposures. Birds treated with Corexit-dispersed oil (30:1 oil dispersant ratio) quickly became waterlogged and remained wet long after the exposure. Similar matting of the feathers was evident after both crude oil and oil/dispersant exposure. The Corexit 9527 by itself, at a concentration of 10 ppm, resulted in a rapid loss of bouyancy in the ducks and elicited strong escape responses. Dispersant-treated birds were unable to shake off or preen water from their feathers and remained dripping wet after the exposure, although no significant increases in metabolism were detected. These experiments suggest that there may be a strong potential for the uptake and accumulation of free dispersant from seawater onto the feathers of seabirds. Under some circumstances, the subsequent loss of bouyancy and increased wetness of feathers may be sufficient to prevent birds with marginal flight capabilities from lifting off the water surface, and could conceivably lead to mortality through prolonged exposure and exhaustion.

There is no information on how long free dispersants may be expected to remain "active" in seawater, although Corexit 9527 is reported to rapidly lose its effectiveness in freshwater (Hagstrom and Lonning 1977). As indicated in an earlier section, the use of chemical dispersants is generally expected to reduce the risk of exposure of birds to oil following spills and blowouts, at least when they are applied in offshore areas before surface slicks reach more sensitive coastal habitats. Since potential effects resulting from direct contact with free dispersant are also expected to be relatively localized and would only occur in offshore areas, the degree of concern associated with the use of chemical dispersants and effects on regional bird populations would likely be MINOR.

5.3.7 Effects of Chemically Dispersed Oil on Fish

There is an extensive literature describing the acute toxicity of dispersants and oil/dispersant mixtures to fish. This is at least in part due to the fact that criteria for assessing the toxicity of dispersants typically specify freshwater (and more recently marine or estuarine) fish as the preferred test organism (Harris and Doe 1977; Doe and Wells 1978; Wells and Harris 1980).

In addition to the increase in the effective toxicity of oil expected following dispersant applications (Section 5.3.4), there is some evidence that dispersants themselves may be more toxic in seawater than in freshwater. For example, Wells and Harris (1980) examined the toxicity of No. 2B fuel oil, fuel oil/Corexit 9527 (10:1 ratio) and Corexit alone to rainbow trout (freshwater) and threespine stickleback (brackish-marine). The mean 96-h LC_{50} values for rainbow trout were 92.8, 57.1, and 192.2 mg/L, respectively, while corresponding levels for threespine stickleback in seawater of 29-31 $^{o}/oo$ salinity were 25.7, 24.6, and 28 mg/L (initial concentrations). Although there is no reason to expect that the toxicities of the various mixtures to these two species would be the same, the 10:1 oil/dispersant mixture was roughly twice as toxic to sticklebacks, while the toxicity of the dispersant to sticklebacks was nearly seven times that measured with rainbow trout.

Wells and Harris (1980), using a similar approach to Marking (1977), suggested that the toxicities of oil and dispersants may be considered in an additive sense (either positively or negatively), although this approach assumes that the toxicity of the dispersant is at least partially maintained after the dispersant is in close association with an oil droplet. Conceptually, this is difficult to visualize since the toxicity of the dispersant is probably associated with the disruption of cellular membrane processes brought about by the attraction to, and subsequent interference with, phospholipid membrane structures by the lipophilic functional group of the dispersant molecule (Sections 5.3.3 and 5.3.4). Until it can be demonstrated that oil-associated dispersant molecules may be actively or passively exchanged into exposed biological membrane surfaces, or that some factor in the dispersant formulation other than the surfactant is responsible for toxicity, it must be assumed that the lipophilic properties of the dispersant are completely satisfied by the oil. As a result, very little residual toxicity of the dispersant would be expected in an oil/dispersant mixture providing that all of the available dispersant is associated with oil droplets (see Section 5.3.10).

In a recent review of the effects of oil and dispersants on fish, Penrose (1980) states that because of the low toxicities of dispersants, and the stringent regulations on their use in Canada, "it is practical to consider dispersants only as mixtures with oil", and adds that the significance of sublethal effects of dispersants at low concentration (e.g. Lonning and Hagstrom 1976) is open to question. The former statement may not even be completely valid for fish since the previously described work of Wells and Harris (1980) indicated that the 96-h LC_{50} for Corexit 9527-exposed stickleback was 28 mg/L, and this was equivalent to the toxicity of dispersed No. 2 B fuel oil. Toxicity curves derived for rainbow trout and stickleback appear to indicate that while a threshold toxicity level is evident for both oil and oil/dispersant mixtures, the threshold level for Corexit 9527 is less easy to define (Wells and Harris 1980), indicating the potential for delayed mortality at relatively low concentrations (<10 ppm). The long-term stability of dispersant/membrane associations is also virtually unknown, so that a realistic assessment of the potential for accumulation of dispersants by fish cannot be made at the present time.

The potential effects of chemically dispersed oil or free dispersants on fish populations of the Beaufort Sea would vary with a number of factors including season, location and extensiveness of the area treated, prevailing current patterns and surface layer turbulence, and particularly the proximity of the treated area to shallow coastal habitats. Acute toxic and sublethal effects of the dispersed oil on fish would likely be greater than if the oil were naturally (physically) dispersed since the duration of exposure would be longer and deeper habitats would be affected. Consequently, depending on the location and extensiveness of the area treated with dispersants and the subsequent fate of free dispersant and dispersed oil, the degree of concern regarding adverse effects of dispersant application on regional fish populations of the Beaufort Sea is expected to range from MINOR to MODERATE.

5.3.8 Susceptibility of Lower Trophic Levels to Chemically Dispersed Oil

The use of chemical dispersants on floating oil slicks would probably have its greatest immediate effects on the planktonic organisms directly under the dispersal zone, since the concentrated populations in the upper few metres of the water column would not only be exposed to high levels of relatively unweathered particulate oil, but also to considerable amounts of dissolved hydrocarbons and possibly free chemical dispersant. In addition, the factors regulating the spatial distribution of oil dispersed from the surface will also, to a greater or lesser extent, control the distribution of plankton, and this could tend to prolong the period of initial contact. One of the factors influencing the degree of potential impacts of chemically dispersed oil on planktonic communities would likely be stratification of the water column since a density discontinuity layer could affect both the rate and extent of vertical diffusion of dispersed oil.

The size of planktonic organisms may also be a factor affecting their susceptibility to petroleum hydrocarbons. For example, copepods were irreversibly paralysed by more than 15 minutes exposure to only 0.2-1.5 mg/L of dissolved hydrocarbons (Wells 1980), suggesting that a large surface area to volume ratio may lead to much faster toxic effects than experienced by larger organisms. The duration of exposure necessary to produce mortality in planktonic organisms may be in the order of several tens of minutes or perhaps several hours, so that the probable persistence of dissolved and particulate oil in the water column below chemically dispersed slicks (Cormack and Nichols 1978) may be sufficient to result in the mortality of more sensitive species.

5.3.9 Effects of Chemically Dispersed Oil on Phytoplankton

The effects of chemical dispersants and oil/dispersant mixtures on phytoplankton have not been extensively examined. A large proportion of the published literature (recently reviewed by Snow 1980) also describes the effects of highly toxic "first generation" emulsifiers such as those used following the TORREY CANYON spill (Nelson-Smith 1978), and these chemicals were clearly more toxic to marine flora and fauna than those presently approved for use.

Two recent studies have examined the effects of chemical dispersion on the toxicity of oil to phytoplankton. Trudel (1978) found that Lago Medio crude oil dispersed with Corexit 9527 was no more toxic to Newfoundland marine phytoplankton than physically dispersed oil when actual measured concentrations were compared. However, even concentrations of physically and chemically dispersed oil as low as 0.1 ppm resulted in a 10 percent decrease in carbon fixation. Through the use of regression analysis, Trudel (1978) estimated that the "zero effect" concentration of crude oil on this phytoplankton community was approximately 0.07-0.08 ppm.

Scott _et al._ (1979) examined the short and long-term effects of physically and chemically dispersed Norman Wells crude oil on artificial freshwater pond ecosystems. One day after treatment, the oil concentration in the water column of the chemically dispersed oil pond was 14 times higher than in the physically dispersed oil pond and remained higher for 40 days. Following the initial exposure to chemically dispersed oil, chlorophyll _a_ levels decreased, reactive silica and dissolved organic carbon dramatically increased, and there was a pronounced shift in the taxonomic composition of the phytoplankton community towards Chrysophytes and Chlorophytes relative to the control pond and the pond treated with physically dispersed oil. Phytoplankton growth in the pond treated with physically dispersed oil was actually stimulated, while reactive silica decreased relative to the control during the first 28 days after the oil addition.

The use of chemical dispersants would increase the probability of adverse effects of oil spills or blowouts on phytoplankton in the Beaufort Sea since they would increase the opportunity for contact between both dissolved and particulate oil and organisms in the euphotic zone. As a result, the degree of concern regarding the use of these compounds would not differ from the concern already associated with effects of oil spills or blowouts on phytoplankton, which varies from MINOR to MODERATE depending on the season and location of the event (Section 5.2.8).

5.3.10 Effects of Chemically Dispersed Oil On Zooplankton

In contrast to phytoplankton, there is a relatively extensive body of literature describing the effects of dispersants and chemically dispersed oil on zooplankton. This information has been recently reviewed and summarized by Wells (1980). The toxicity of second and third generation oil dispersants to zooplankton varies markedly with species, age, developmental stage, and general health of affected individuals, as well as with factors associated with the dispersant such as chemical stability, the nature of the solvent carrier, and the temperature, oxygen content and salinity of the water (Wells 1980).

Although the acceptance criteria established by the Canadian government for individual dispersants (Environment Canada 1973) specifies a maximum toxicity level to freshwater fish of 1000 mg/L (96-h LC_{50}), toxicities to marine zooplankton often exceed this level by several orders of magnitude. For example, the incipient lethal level (ILL) of Corexit 7664 with the ctenophore Pleurobrachia was 670 mg/L, while the corresponding level with larval shrimp (Crangon sp.) was 1.6 mg/L (Wells 1980). Hagstrom and Lonning (1977) found that at a Corexit 9527 concentration of 0.003 ppm in seawater, the fertilization capacity of sea urchin spermatozoa was reduced by 25 percent following only a 2 minute exposure, and this was increased to 85 percent by a 10 minute exposure. Earlier studies by Lonning and Hagstrom (1976) indicated that the embryogenesis of cod and flatfish eggs was also sensitive to dispersant treatment, although these effects were observed at concentrations (i.e. 10 to 100 ppm) unlikely to be encountered during actual treatment of oil slicks.

The results of studies describing the effects of chemically dispersed oil on zooplankton found in Canadian waters were summarized by Wells (1980). In addition to the general trend of increased probability of exposure of plankton to oil following dispersant use (Section 5.3.4), studies conducted to date also indicate that copepods are relatively resistant to dispersed oil, while shrimp appear particularly sensitive and the general response of embryonic and larval fish to chemically dispersed oil is similar to their response to physically dispersed oil (Wells 1980). Respiratory and nervous systems have been found to be most sensitive to chemically dispersed oil. Maki (1979, cited in Wells 1980) suggests that the "no observed-effect" concentration of chemically dispersed oil for larval polychaetes, larval decapods and newly hatched fish larvae may be much lower than 0.5 mg/L.

As in the case of phytoplankton, the use of chemical dispersants would not decrease the overall degree of concern associated with adverse effects of oil spills and blowouts on zooplankton in the Beaufort Sea, and may actually increase the susceptibility of some species or life history stages. Consequently, the degree of concern associated with oil spills or blowouts involving dispersant application would remain MINOR to MODERATE.

5.3.11 Effects of Chemical Dispersion on the Oil Degrading Capability of Micro-organisms

The primary concern related to the effects of dispersants on marine micro-organisms is whether or not the potential for biodegradation of oil would be decreased by the presence of dispersant. The limited available information suggests that the use of chemical dispersant would enhance the rate of biodegradation by increasing the surface area of petroleum available for bacterial attack (e.g. Traxler and Bhattacharya 1978).

Mulkins-Phillips and Stewart (1974) examined the effect of four dispersants on the growth and oil-degrading capability of marine bacteria indigenous to Halifax harbour, and found that the effectiveness of the dispersant in promoting the formation of oil-in-water emulsions could be correlated with its effect on bacterial growth curves and hydrocarbon utilization. Indigenous bacterial populations were able to use all four of the dispersants as carbon sources, although each caused an initial inhibition of growth that was directly related to the strength of its surfactant properties. Combinations of Arabian crude oil and dispersant/oil mixtures caused a reduction in the importance of some genera within the bacterial community and an increase in the proportion of pseudomonads and/or achromobacters. Mulkins-Phillips and Stewart (1974) also reported that the addition of either dispersant, oil, or some combination of these generally produced bacterial population densities from 2 to 3 orders of magnitude higher than those in control populations after 8 to 16 days, while the presence of the dispersant resulted in densities of bacteria up to 1 order of magnitude higher than those with oil alone. In a similar study, Traxler and Bhattacharya (1978) reported that Corexit 9527 could be utilized as a carbon source by microbial populations from Narragansett Bay, R.I., and found that under simulated natural conditions, little if any biodegradation of petroleum hydrocarbons was initiated unless the oil was first either physically or chemically dispersed from the surface.

The extent that nutrient recycling in the water column under a chemically dispersed oil slick may be affected by shifts in the composition of the marine bacterial community towards groups of oil-degrading or oil-tolerant strains is not known. It must be stressed that several days are required to cause bacterial population changes in the laboratory while high concentrations of hydrocarbons and/or dispersant following an oil spill may rapidly diminish within several hours of dispersant application due to horizontal and vertical diffusion (Cormack and Nichols 1978). Nevertheless, the use of chemical dispersants would be expected to increase the rate of biodegradation of oil to some extent in the Beaufort Sea.

5.3.12 The Effects of Chemically Dispersed Oil on Benthic Communities

The effects of dispersants and chemically dispersed oil on benthic organisms are well documented (see Percy 1980, for a recent review), although the ultimate fate of chemically dispersed oil in near-bottom waters and marine sediments remains virtually unknown. Oil dispersed from the surface by either physical or chemical processes can reach considerable depths in a relatively short period. For example, Cormack and Nichols (1978) found that at wind speeds of 8-10 knots and a sea state of 2-3, the application of dispersant to an experimental floating slick resulted in a rapid vertical transport of oil away from the upper water column. Linden et al. (1979) reported that within one week of the grounding of the tanker TSESIS in a relatively confined waterway in the Swedish Archipelago, physically dispersed oil was being sedimented to the bottom in water 30 to 40 m deep at rates as high as 60 $mg/m^2/day$.

The availability of sedimented oil for biological uptake by benthic organisms is poorly understood. Physical factors which could influence the potential for uptake of hydrocarbons by benthic infauna and epifauna include: (1) the sedimentation characteristics of the affected area, such as the rate of deposition of new sediment and the extent of bottom reworking; (2) the composition of the sediments, including the amount and type of organic material and the oil adhesion properties of the mineral and organic components; and (3) the physical state of the oil reaching the bottom, its particle size, chemical composition and relative "stickiness".

Important biological factors which may affect either the potential for or rate of oil uptake include: (1) normal habitat occupied by the organism (i.e. infaunal or epifaunal); (2) feeding mechanism (i.e. filter feeding, deposit feeding, scavenging or some combination); (3) behavioural factors, such as detection and avoidance of oiled food or substrate, as well as the general mobility of the organism; and (4) the degree of protection imparted by the integument of the organism.

In areas where reworking of benthic surface sediments is common, sedimented oil may be rapidly buried, and the duration of exposure of epifauna to oil correspondingly shortened. For example, many of the nearshore sediment cores showed oil layers covered by as much as 15 cm of new sediment within four months of the grounding of the AMOCO CADIZ off Brittany (D'Ozouville et al. 1979). In other situations where the sediments are relatively undisturbed, particulate oil may only be concentrated in the upper 1 to 2 cm. Gearing et al. (1979) also report that some conditions favour the development of a petroleum-rich flocculant layer at the water-sediment interface.

Depending on the type of oil-sediment interactions which occur, potential impacts of sedimented chemically dispersed oil could be restricted to one or more classes of benthic organisms. Oil buried under layers of uncontaminated sediment could result in chronic exposure of burrowing clams and deposit-feeding polychaete worms to petroleum hydrocarbons, while exerting a proportionally smaller direct effect on epibenthic crustaceans. The opposite may be the case when oil is concentrated in a shallow surface layer.

Chemical dispersion of surface oil slicks may affect sediment-oil interactions and the subsequent degree of impact of a spill on benthic flora and fauna, although these interactions remain poorly documented at the present time. As indicated earlier (Section 5.2.2), oil which is physically or naturally dispersed from the surface may enter the water column gradually over a period of days or weeks depending on prevailing oceanographic conditions, and oil which is sedimented can be expected to be deposited over a relatively wide area. The application of chemical dispersants tends to reduce the horizontal spreading of an oil slick and simultaneously accelerates its vertical movement through the water column, and this could result in exposure of benthic habitats to much higher concentrations of emulsified oil. In addition, chemically dispersed oil may be less weathered than oil that is naturally dispersed when dispersants are applied soon after a spill or blowout, and this may also increase its toxicity to benthic organisms.

5.3.12.1 Effects of Dispersants

The toxicity of second and third generation dispersants to benthic organisms was recently reviewed by Percy (1980). Most of the recent dispersant formulations are essentially non-toxic to adult benthic organisms when examined in short-term bioassays (Cowell 1978; Nelson-Smith 1978). However, as indicated earlier (Section 5.3.2), different life cycle stages may vary in their sensitivity to chemical dispersants. Reproductive processes and larval forms appear to be particularly susceptible to the effects of dispersants. For example, adverse effects of dispersants have been reported with respect to fertilization in sea urchins (Lonning and Hagstrom 1976), fertility and egg maturation in polychaetes (George 1971; Bellan et al. 1972, cited in Percy 1980), and fecundity in gastropods (Eisler 1973, cited in Percy 1980). Other sublethal effects of chemical dispersants on benthic species include narcosis, decreased locomotory activity, loss of equilibrium, and reductions in respiration and other intermediary metabolic processes (Percy 1980). These effects, together with observations of visible tissue necrosis, support the hypothesis that the primary toxic action of dispersants is associated with effects on the structure and integrity of biological membrane systems.

The potential effects of free dispersants on benthic organisms in the natural marine environment would be highly dependent on: (1) the minimum concentration required to produce a measurable effect; (2) water depth, and (3) the uptake and/or neutralization of free dispersant by chemical or biological processes within the water column. Since effects on fertilization have been documented at dispersant concentrations as low as several parts per billion (Lonning and Hagstrom 1976), the "no effect level" may be in the order of 1 ppb or less for some sensitive biological processes. Increasing water depths should decrease the amount of free dispersant which is likely to reach benthic habitats, particularly with dispersants that are completely soluble in seawater. Nevertheless, an initial dispersant concentration of 1 ppm in the upper metre of the water column (Cowell 1978) could still theoretically result in concentrations of 10 and 1 ppb in water depths of 10 and 100 m, respectively, assuming that horizontal diffusion accounts for a further dilution by a factor of 10. Although these concentrations could be toxic to more sensitive species, there have been no confirmed reports of significant ecological impacts of offshore use of dispersant on benthic communities (Cowell 1978). It is possible that the biological uptake and chemical neutralization of dispersant within the water column may ultimately prevent toxic concentrations of dispersant from reaching deep benthic habitats.

5.3.12.2 Effects of Chemically Dispersed Oil

The apparent increase in the toxicity of petroleum products and crude oil to benthic organisms following treatment with dispersants is probably attributable to the increased stability of the resultant oil-in-water emulsions and the greater opportunity for organisms to contact both dissolved and particulate oil (Swedmark et al. 1973; Percy 1980). Several authors (e.g. Rossi 1977; Roesijadi et al. 1978) have shown that uptake of petroleum hydrocarbons by benthic polychaetes occurred primarily from the dissolved fraction rather than from oil bound up in the substrate.

The overall degree of concern regarding the effects of dispersants and chemically dispersed oil in the Beaufort Sea would vary with the location and water depth where dispersants were applied and the time of year of the spill or blowout. For example, benthic flora and fauna in shallow areas would be far more susceptible to contamination by emulsified oil, particularly during early summer (June) when larval invertebrates are most abundant. The degree of potential concern would also be greatest when dispersant is applied to relatively unweathered oil since this could increase the exposure of benthic organisms to dissolved petroleum hydrocarbons. Oil/sediment interactions may also be modified by the presence of dispersants (Abbiss et al. 1981) and this could subsequently increase the susceptibility of certain members of the benthic community. Further information regarding the effects of chemically dispersed oil on arctic benthic invertebrates will be available once the results of the Baffin Island Oil Spill (BIOS) program are published. At the present time, the degree of concern regarding impacts of chemically dispersed oil on benthic populations is expected to range from MINOR to MODERATE, depending on the depth and extensiveness of the habitat affected.

5.3.13 Summary of Concerns Related to Chemically Dispersed Oil and Dispersants

The use of chemical dispersants following oil spills or blowouts in the Beaufort Sea would decrease the degree of potential concern regarding adverse impacts of petroleum hydrocarbons on regional marine mammal and bird populations, largely as a result of removal of oil from the surface where contact with these species is most likely. On the other hand, dispersant application would generally increase the concentrations of dissolved and particulate oil in the water column as well as the duration of exposure of pelagic and benthic organisms to these forms of oil. Consequently, the degree of concern regarding potential adverse impacts of spills or blowouts on pelagic and benthic flora and fauna would remain the same or be increased by the application of dispersants, particularly in shallow coastal areas where relatively large amounts of oil could each the seafloor and the abundance of important fish resources tends to be highest. The degree of concern regarding impacts of chemically dispersed oil and free dispersant on regional populations of birds and marine mammals is considered MINOR, while the level of concern associated with potential effects of chemically dispersed oil on planktonic and benthic communities and regional fish populations is expected to be MINOR to MODERATE depending on the location and extensiveness of the areas treated with dispersants. The use of dispersants in waters less than 20 m is not recommended since adverse impacts on both pelagic and benthic resources would be most probable in such water depths.

TABLE 5.3-1

SUMMARY OF POTENTIAL CONCERNS RELATED TO USE OF CHEMICAL DISPERSANTS IN THE BEAUFORT SEA REGION

Resource or Environmental Component	Potential or Probable Effects	Degree of Potential Regional Concern
Water Quality	Higher concentrations of both particulate (emulsified) and dissolved oil to greater depths in the water column than normally found when dispersants are not used	see individual resources
Benthic Sediments	Greater contamination of sediments with oil, particularly in relatively shallow areas	see individual resources
Marine Mammals	Disruption of thermoregulation, external irritation of eyes and nostrils, and ingestion of emulsified oil, although the latter two effects are expected to be less severe than those which may be associated with non-treated surface slicks	MINOR
Birds	Application of dispersants is generally expected to reduce exposure of birds to surface oil. Diving ducks contacting emulsified oil may experience increased basal metabolic rates and quickly become waterlogged, while contact with free dispersant may result in loss of buoyancy	MINOR

TABLE 5.3-1 (cont'd)

Resource or Environmental Component	Potential or Probable Effects	Degree of Potential Regional Concern
Fish	More pronounced acute toxic and sublethal effects than naturally dispersed oil or surface slicks since the exposure duration would be longer and deeper habitats would be affected. The degree of concern would increase with the size of the area affected by dispersed oil and proximity to the shoreline	MINOR to MODERATE
Phytoplankton	As above	MINOR to MODERATE
Zooplankton	As above	MINOR to MODERATE
Benthic Communities	As above. Degree of potential concern would also be higher when larval forms are present (early summer)	MINOR to MODERATE

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6.0 SUMMARY OF SIGNIFICANT CONCERNS RELATED TO HYDROCARBON DEVELOPMENT IN THE BEAUFORT SEA

Previous sections of this report have discussed the possible effects and degree of regional concern associated with wastes and disturbances resulting from a range of normal activities common to offshore hydrocarbon exploration and production operations, as well as the possible impacts of oil spills and natural gas well blowouts. These effects and concerns have been discussed on an activity-by-activity basis, without consideration of the possibility that several sources of disturbance or wastes could act together to produce synergistic and cumulative impacts. As a result, the following sections summarize the anticipated degree of regional concern related to hydrocarbon development in the Beaufort Sea on a resource-by-resource basis, and identify to the extent possible where potential cumulative impacts may occur. It should be emphasized from the outset that there is virtually no information describing the cumulative impacts which may be associated with hydrocarbon development in offshore waters, although extremely intensive offshore operations such as those which occur in the North Sea and Gulf of Mexico have not resulted in any documented impacts on regional populations of marine resources.

Figures 6-1 to 6-4 summarize the degree of regional concern associated with the individual resource-activity interactions discussed previously in this report. It should be evident from these matrices that the most serious area of concern with respect to most biological resources would be oil spills (Figure 6-4), while the majority of concerns associated with normal activities of the petroleum industry in the Beaufort region are expected to be NEGLIGIBLE or MINOR.

6.1 MARINE AND MARINE-ASSOCIATED MAMMALS

6.1.1 White Whale

The population of white whales that ranges to the Beaufort Sea region has been estimated to number about 7000 individuals. This stock winters in the Bering Sea, and undertakes annual migrations through the Chukchi Sea and offshore Beaufort Sea to the Mackenzie River estuary. The white whales concentrate in the estuary for about 2-3 weeks each year, beginning in late June or early July. Their distribution after they leave the Mackenzie Estuary is not well known, although some probably move far offshore, while others remain in the estuary or move east to areas off the Tuktoyaktuk Peninsula or Amundsen Gulf. The fall migration occurs during September and October, and is believed to be largely offshore (LGL and ESL 1982).

Other than industrial underwater noise or shock waves produced by high explosives, the potential effects of all normal activities, wastes and disturbances associated with hydrocarbon exploration and production are considered of NEGLIGIBLE concern with respect to the regional population of

FIGURE 6-1

SUMMARY OF INTERACTIONS AND DEGREE OF REGIONAL CONCERN REGARDING THE EFFECTS OF DISTURBANCES AND WASTES ASSOCIATED WITH THE PETROLEUM INDUSTRY ON MARINE RESOURCES OF THE BEAUFORT SEA

	BIOLOGICAL RESOURCES																							
	MAMMALS						BIRDS							FISH			LOWER TROPHIC							
	White Whale	Bowhead Whale	Ringed Seal	Bearded Seal	Polar Bear	Arctic Fox	Loons	Ducks	Geese/Swans	Alcids	Jaegers/Gulls/Terns	Shorebirds	Other Marine Birds	Pelagic Marine	Demersal Marine	Anadromous	Phytoplankton	Zooplankton	Benthic Epifauna	Benthic Infauna	Benthic Flora	Epontic Fauna	Epontic Flora	Microorganisms
ARTIFICIAL STRUCTURES																								
Mortality																			○	○	○			○
Habitat/Food Source Loss		○		○	○	○									○					○	○			
Disturbance/Physical Presence	○	○	○	○	○	○	△	△		△	○	○	○											
Attraction			○	○	△	△					△		△	○	○	○								
Habitat/Food Source Gain														○	○	○			○		○			○
HUMAN PRESENCE																								
Over-exploitation					○			○	○					○	○	○								
Disturbance/Encroachment	○	○	○	○	△	○	▲	▲	▲	▲	▲	▲	▲											
Attraction/Avoidance			○	○	△	△																		
ICEBREAKING																								
Mortality			▲	△										○	○							○	○	
Habitat/Food Source Loss			○	○	○	○																○	○	
Other Indirect Effects																		○						
Disturbance	○	○	△	△	○	○	○	▲	▲	▲	▲	○	○											
Habitat/Food Source Gain					○		○	○		○	○		○	○			○	○				○		
Increased Productivity																	○				○			
DREDGING																								
Mortality														△	△	▲	○	○	△	▲	○			○
Habitat/Food Source Loss		○	△				△	△		△					△	▲		○	▲	▲	○			
Other Sublethal Effects	○	○	○	○			△	△		△				△	△	▲		○	▲	▲		○		
Disturbance	○	○	○	○			△	△		△			△	△	△	▲								
Change in Primary Productivity																	○				○		○	○
Change in Species Composition																	○		▲	▲	○		○	○
Habitat/Food Source Gain							○	○		○	○		○					○	▲	▲		○		○
TREATED SEWAGE																								
Mortality																			○	○	○			
Disease	○	○	○	○	○		○	○		○	○	○	○	○	○	○			○	○				
Habitat/Food Source Loss															○	○			○	○	○			
Sublethal/Indirect Effects														○	○	○	○	○	○	○	○	○	○	
Change in Species Composition																	○		○	○	○			○
Attraction to Outfalls			○	○			○	○			○	○	○	○	○	○								
Habitat/Food Source Gain							○	○			○	○	○	○	○	○		○	○			○		○
Increased Productivity																	○				○		○	○
UNDERWATER NOISE																								
Physiological Damage	△	△	△	△										○	○	○								
Masking	▲	▲	▲	▲										○	○	○								
Sublethal/Indirect Effects	▲	▲	▲	▲																				
Disturbance	▲	▲	▲	▲										△	△	△								
AIRBORNE NOISE																								
STATIONARY SOURCE – Indirect Effects			○	○			△	△	△	△	△	△	△											
STATIONARY SOURCE – Disturbance			○	○	○	○	△	△	△	△	△	△	△											
STATIONARY SOURCE – Attraction			○	○	△	△					△		△											
MOBILE SOURCE – Indirect Effects			△	△			△	■	■	■	△	△	△											
MOBILE SOURCE – Disturbance			△	△	△	△	△	▲	■	▲	△	△	△											
ENGINE EXHAUSTS/ AIR EMISSIONS																								
Attraction to "Heat Islands"					○	○	○	○		○	○		○											
SOLID WASTE DISPOSAL																								
Mortality					△	○													○	○	○			○
Habitat/Food Source Loss				○			○	○		○				○	○	○								
Attraction					△	○					○		○	○	○	○								
Habitat/Food Source Gain					△	○					○		○	○	○	○			○		○			○

○ - NEGLIGIBLE
△ - MINOR
▲ - MODERATE
■ - MAJOR

FIGURE 6-2

SUMMARY OF INTERACTIONS AND DEGREE OF REGIONAL CONCERN REGARDING THE EFFECTS OF DISTURBANCES AND WASTES ASSOCIATED WITH NORMAL DRILLING OPERATIONS ON MARINE RESOURCES OF THE BEAUFORT SEA

		BIOLOGICAL RESOURCES																							
		MAMMALS						BIRDS							FISH			LOWER TROPHIC							
		White Whale	Bowhead Whale	Ringed Seal	Bearded Seal	Polar Bear	Arctic Fox	Loons	Ducks	Geese/Swans	Alcids	Jaegers/Gulls/Terns	Shorebirds	Other Marine Birds	Pelagic Marine	Demersal Marine	Anadromous	Phytoplankton	Zooplankton	Benthic Epifauna	Benthic Infauna	Benthic Flora	Epontic Fauna	Epontic Flora	Microorganisms
DRILLING FLUIDS, FORMATION CUTTINGS AND PRODUCED WATER																									
Mortality								▲	▲		▲	△	△	▲	▲	▲	▲	△	△	△	△	△	△	△	△
Sublethal Toxic Effects		○	○	○	○			△	△		△	△		△	▲	▲	▲	△	△	△	△	△	△	△	
Habitat/Food Source Loss			○		○			○	○		○			○	▲	▲	▲		△	△	△	△	△	△	
Uptake of Trace Metals		○	○	△	△	△	△	△	△		△	△	△	△	▲	▲	▲	△	△	▲	▲	△	△	△	△
Decreased Productivity																		△				△		△	
Other Sublethal Effects															▲	▲	▲		△	△	△		△		
Increased Productivity																									△
B.O.P. CONTROL FLUID																									
Mortality															○	○			○	○					○
Habitat/Food Source Loss					○										○	○	○								
Other Sublethal Effects																○			○	○					○
Habitat/Food Source Gain																									○
UNDERWATER SHOCK																									
AIR GUNS/SLEEVE EXPLODERS	Disturbance	○	○	○	○	○	○								○	○	○								
HIGH EXPLOSIVES	Mortality	△	△	△	△			○	○		○			○	△	△	△								
	Sublethal Damage	△	△	△	△			○	○		○			○	△	△	△								
CEMENT SLURRY AND POWDER																									
Mortality															○	○	△	○	○	△	○	○			○
Habitat/Food Source Loss			○		○			○	○		○			○	○	○	△		○	△	○	○			○
Other Sublethal Effects															○	○	△	○	○	△	○	○			

○ - NEGLIGIBLE

△ - MINOR

▲ - MODERATE

■ - MAJOR

FIGURE 6-3

SUMMARY OF INTERACTIONS AND DEGREE OF REGIONAL CONCERN REGARDING THE EFFECTS OF DISTURBANCES AND WASTES ASSOCIATED WITH PRODUCTION PROCESSES AND THE STORAGE AND TRANSPORTATION OF PETROLEUM HYDROCARBONS ON MARINE RESOURCES OF THE BEAUFORT SEA

	BIOLOGICAL RESOURCES																							
	MAMMALS						BIRDS							FISH			LOWER TROPHIC							
	White Whale	Bowhead Whale	Ringed Seal	Bearded Seal	Polar Bear	Arctic Fox	Loons	Ducks	Geese/Swans	Alcids	Jaegers/Gulls/Terns	Shorebirds	Other Marine Birds	Pelagic Marine	Demersal Marine	Anadromous	Phytoplankton	Zooplankton	Benthic Epifauna	Benthic Infauna	Benthic Flora	Epontic Fauna	Epontic Flora	Microorganisms
GAS FLARES																								
Mortality							△	△		△	△	○	△											
Attraction			○	○	△	○	△	△		△	△	○	△											
HEATED WATER																								
Mortality														○		○	○	○				○	○	
Habitat/Food Source Loss																						○	○	
Other Sublethal Effects														○	○	○	○	○	○		○	○	○	
Attraction			○	○			○	○		○	○		○	○	○	○								
Change in Species Composition																	○				○			○
Habitat/Food Source Gain																		○	○			○		
Increased Productivity																	○				○		○	○
BALLAST WATER / EXOTIC ORGANISMS																								
Competitive Species/ Predators														▲	▲	▲	△	△	▲	○		△	△	△
Disease/ Parasites														▲	▲	▲								

○ - NEGLIGIBLE
△ - MINOR
▲ - MODERATE
■ - MAJOR

FIGURE 6-4

SUMMARY OF INTERACTIONS AND DEGREE OF REGIONAL CONCERN REGARDING THE EFFECTS OF ABNORMAL OPERATIONS AND ENVIRONMENTAL EMERGENCY SITUATIONS ASSOCIATED WITH HYDROCARBON EXPLORATION AND PRODUCTION ON MARINE RESOURCES OF THE BEAUFORT SEA

	BIOLOGICAL RESOURCES																							
	MAMMALS						BIRDS							FISH			LOWER TROPHIC							
	White Whale	Bowhead Whale	Ringed Seal	Bearded Seal	Polar Bear	Arctic Fox	Loons	Ducks	Geese/Swans	Alcids	Jaegers/Gulls/Terns	Shorebirds	Other Marine Birds	Pelagic Marine	Demersal Marine	Anadromous	Phytoplankton	Zooplankton	Benthic Epifauna	Benthic Infauna	Benthic Flora	Epontic Fauna	Epontic Flora	Microorganisms
NATURAL GAS BLOWOUTS AND SUBSEA PIPELINE RUPTURES																								
Mortality			○	○										○	○	○	○	○	○	○	○	○	○	
Habitat/Food Source Loss		○	○	○														○				○	○	
Other Sublethal Effects	○	○	○	○										○	○	○	○	○	○			○	○	○
Disturbance	○	○	○	○										○	○	○								
Habitat/Food Source Gain																		○						○
Increased Productivity																	○							○
CRUDE OIL SPILLS AND BLOWOUTS / REFINED FUEL SPILLS																								
Mortality	△	△	▲	▲	▲	○	■	■	■	■	▲	■	■	▲	▲	▲	▲	▲	▲	▲	▲	▲	▲	○
Habitat/Food Source Loss		△	△	▲	△	○	▲	▲	△	▲	△	△	▲	△	▲	▲		▲	▲	▲	▲	▲	▲	○
Hydrocarbon Accumulation	○	△	▲	△	▲	△	▲	▲	▲	▲	▲	▲	▲	△	▲	▲	▲	▲	▲	▲	▲	▲	▲	
Other Sublethal/Indirect Effects	△	▲	▲	▲	▲	△	■	■	■	■	▲	■	■	▲	▲	▲	▲	▲	▲	▲	▲	▲	▲	○
Change in Species Composition																	▲	△	△	▲	▲	△	△	○
Habitat/Food Source Gain																		△	△					○
Increased Productivity																	△				△			
*CHEMICALLY DISPERSED OIL																								
Mortality	△	△	△	△	△	○	▲	▲		▲	△	△	▲	▲	▲	▲	▲	▲	▲	▲	▲	▲	▲	○
Habitat/Food Source Loss		○	○	△	○	○	△	△		△	○	○	△	▲	▲	▲		▲	▲	▲	▲	▲	▲	○
Hydrocarbon Accumulation	○	△	▲	△	▲	△	△	△		△	○	○	△	▲	▲	▲	▲	▲	▲	▲	▲	▲	▲	
Other Sublethal/Indirect Effects	△	△	▲	▲	▲	△	▲	▲		▲	△	△	▲	▲	▲	▲	▲	▲	▲	▲	▲	▲	▲	○
Change in Species Composition																	▲	△	△	▲	▲	△	△	○
Habitat/Food Source Gain																		△	△					○
Increased Productivity																	△				△			

*Interactions for birds assume oil is dispersed offshore and does not reach the coast.

○ - NEGLIGIBLE

△ - MINOR

▲ - MODERATE

■ - MAJOR

white whales which summers in the Beaufort Sea. The question of underwater noise is difficult to resolve on the basis of existing information. There is little doubt that sounds produced by vessel traffic and various stationary noise sources could affect white whales, since some individuals have been shown to react to hydrocarbon exploration-related activities in the Beaufort Sea in the past (Section 2.6.5.4). Although the biological significance of masking or disturbance effects on white whales remains unknown, cumulative sources of underwater noise in the Beaufort region are not likely to cause a decline in abundance and/or change in distribution of the population that would persist for several generations. Consequently, the degree of regional concern regarding the effects of underwater sound on white whales is not expected to be greater than MINOR or MODERATE. The degree of regional concern associated with the effects of high explosives such as those which may be required for ice management programs is considered MINOR, while concerns related to conventional seismic air guns and sleeve exploders are NEGLIGIBLE.

Unlike most marine resources, the degree of concern associated with the effects of oil spills on white whales is only expected to be MINOR (Figure 6-4). Mortality of whales has not been documented in the oil spill case history literature (Duval *et al.* 1981), and potential indirect impacts of spills (e.g. reduced food availability) are unlikely to affect this species in a manner which would be considered regionally significant (Section 5.2.4).

Development of petroleum hydrocarbon resources in the Beaufort Sea could result in cumulative or synergistic impacts on white whales. As indicated in the summary matrices, a number of facilities and activities may cause disturbance of white whales due to their contribution to underwater noise levels. The major sources of underwater noise in the region would be vessel movements (including icebreaking), tanker traffic, artificial island construction activities (including dredging operations) and seismic programs. All of these activities and the level of noise produced would increase as development proceeds in the offshore Beaufort. For example, it is estimated that 69 temporary exploration islands and from 16 to 26 long-term production islands may be constructed in the Beaufort Sea by the year 2000 (Everitt *et al.* 1982), while the number of marine vessels operated by or on behalf of the petroleum industry is expected to increase from the present level of 22 to 136 by the year 2000, with 75 of these being able to operate year-round (EIS Volume 4; Table 3.6-1). Consequently, noise levels may progressively increase with time, and this could lead to cumulative impacts if interactions with white whales become more frequent and are of longer duration.

6.1.2 Bowhead Whale

The western Arctic population of bowhead whales winters in the Bering Sea, and undertakes an annual migration to summer feeding areas in the southeastern Beaufort Sea and Amundsen Gulf during May and June. This stock has been estimated to number at least 2300 whales (LGL and ESL 1982). The distribution of the whales throughout the summer varies during and among years, and may include waters of Amundsen Gulf, as well as nearshore or offshore areas of the southeastern Beaufort Sea. The return fall migration occurs during September and October, and is believed to be largely coastal.

As with white whales, the potential effects of most activities and common wastes and disturbances associated with normal hydrocarbon exploration and production activities are of NEGLIGIBLE concern with respect to the regional population of bowhead whales (Figures 6-1 to 6-3). The only exceptions to this overall assessment are the MODERATE concern regarding the potential effects of underwater industrial noise, and the MINOR concern which may be associated with the effects of underwater shock waves produced by use of high explosives for ice management programs at offshore structures.

Bowhead vocalizations (and presumably their hearing sensitivity) closely correspond to the low frequency noise produced by most industrial machinery (Section 2.5.6). As a result, underwater noise in the Beaufort Sea production zone may affect bowheads through disturbance or masking, although as indicated earlier, the biological significance of these effects remain unknown.

The potential impacts of crude or refined fuel spills on bowheads are an area of MINOR regional concern, and could range from direct sublethal effects such as uptake of hydrocarbons to indirect effects associated with reduced food availability (Section 5.2.4). The most significant impacts of spills or blowouts would likely be displacement of feeding animals, pathological effects from ingestion of petroleum hydrocarbons and fouling of baleen feeding mechanisms. However, mortality of bowheads as a result of oil exposure is not considered a significant area of potential concern (Figure 6-4).

Development of hydrocarbon resources in the Beaufort Sea could result in cumulative or synergistic effects on bowhead whales. As indicated in the previous section, the number of sources producing underwater noise would increase both spatially and temporally as the development proceeds. Nevertheless, the degree of regional concern regarding the effects of multiple noise sources on this species is not expected to be more than MODERATE. Cumulative effects of multiple waste discharges and disturbances other than underwater noise are expected to be a NEGLIGIBLE area of concern with respect to bowheads due to the mobility of whales and because the wastes would be rapidly diluted in the receiving environment.

6.1.3 Ringed Seals

Ringed seals are widely distributed and relatively abundant residents of the Beaufort Sea and adjacent waters. The estimated size of the regional population in areas to 160 km offshore of the mainland coast and the west coast of Banks Island, and western Amundsen Gulf ranged from 23,000 to 62,000 between 1974 and 1979. Pupping occurs in areas of landfast ice, primarily in the large bays of Amundsen Gulf, and to a lesser extent, offshore of the mainland coast (LGL and ESL 1982). As indicated in Figures 6-1 to 6-3, the effects of icebreaking, underwater noise, airborne noise, discharge of formation water and underwater shock waves produced by high explosives are the most significant areas of potential concern with respect to normal operations of the petroleum industry.

There is a MODERATE degree of concern associated with the effects of icebreaking on regional populations of ringed seals because some pups may be lost if vessels pass through landfast ice breeding areas during the period from about late March to early June. STOL aircraft and helicopter activity may cause repeated diving of hauled-out ringed seals during the 2-3 week moulting period each June, although this response is only considered a MINOR area of potential concern. Underwater noise produced by most industrial activities in the region may disturb or mask vocalizations of this species. The potential effects of multiple underwater noise sources in the Beaufort Sea production zone could be a MODERATE concern with respect to the seal population, although this is considered a worst case assessment because of the absence of information on the biological significance of masking and disturbance and the possibility that seals would become habituated to increased underwater noise levels. Underwater shock waves produced by high explosives are a MINOR regional concern since they could cause localized mortality or disturbance of ringed seals.

The potential effects of formation water discharge on ringed seals are also an area of MINOR regional concern because individuals attracted to offshore facilities could be exposed to dissolved trace metals and hydrocarbons or contaminated prey for extended periods, and this may result in a variety of sublethal effects described in Section 3.1.10.9.

The potential effects of crude oil and refined fuel spills on ringed seals are a MINOR to MODERATE concern in the offshore Beaufort Sea development zone (Figure 6-4). Highly viscous (weathered) crude could impair mobility, cause mortality or sublethal stress, or result in localized reductions in food availability for this species. Unweathered crude and diesel fuel could cause similar effects, as well as eye irritation or damage and various physiological disorders discussed in Section 5.2.4 and Duval *et al.* (1981). The use of chemical dispersants would reduce the potential for contact of seals with surface oil slicks, but increase potential effects resulting from exposure to dissolved hydrocarbons and emulsified oil within the water column.

If seals are attracted to offshore exploration or production facilities, combinations of wastes and the physical presence of structures could result in synergistic effects on some individuals. For example, attraction to sites of industrial activity could increase the potential for masking and disturbance by underwater noise, and increase the probability of exposure to various combinations of wastes released from these sites. Nevertheless, only those seals actually attracted to the sites could be affected in this manner, and these individuals would probably represent a very small proportion of the regional population of ringed seals.

6.1.4 Bearded Seals

Bearded seals are widely distributed throughout the Beaufort region, but are not as abundant as ringed seals (LGL and ESL 1982). During 1974 to 1979, the estimated size of the bearded seal population in the Canadian Beaufort Sea and western Amundsen Gulf ranged from a low of 1300 (1977) to a high of 3100 (1978). Pupping occurs primarily on moving pack ice during late April or early May, and pups are precocious at birth.

As in the case of ringed seals and in the absence of evidence to the contrary, the potential effects of underwater noise on regional populations of bearded seals are considered a MODERATE concern, although this species could also become habituated to increased underwater noise levels. The effects of underwater shock waves, formation water discharge and airborne noise are all expected to be MINOR areas of concern (Figures 6-1 to 6-3). In addition, because this species is a benthic feeder, potential effects of dredging and solid waste disposal may be of MINOR but localized concern as a result of indirect effects associated with habitat loss and reduced food availability. Unlike ringed seals, icebreaking is only a MINOR concern with bearded seals because the number of pups lost if icebreakers moved through breeding areas would be small and probably insignificant in relation to the size of the regional population (Section 2.3.2). Potential effects of all other wastes and disturbances associated with normal operations of the petroleum industry are expected to be of NEGLIGIBLE regional concern with this species.

Crude oil or refined fuel spills are of MINOR to MODERATE concern with respect to the bearded seal population of the Beaufort region, and could result in effects similar to those previously described for ringed seals (Section 6.1.3). Reduced food availability for bearded seals may be a more significant area of concern than with ringed seals, particularly in the event of a well blowout which results in contamination of extensive areas of benthic habitat. Bearded seals feeding in such areas may be indirectly affected by reduced food availability, and directly affected through ingestion of sunken oil or contaminated prey (Section 5.2.4). In addition, exposure to surface oil slicks could result in mortality and a range of sublethal behavioural and physiological effects (Figure 6-4).

Cumulative effects of various activities, wastes and disturbances associated with offshore hydrocarbon development could occur if bearded seals are attracted to industrial sites characterized by vessel and air traffic, multiple waste discharges, dredging and icebreaking. The potential for cumulative effects would be greatest in waters surrounding offshore structures, particularly since the number of exploration and production facilities would progressively increase throughout the development. Nevertheless, the total number of seals affected by industrial activities would probably be insignificant in terms of the size and distribution of the regional population.

6.1.5 Polar Bear

The polar bear population in the Canadian Beaufort Sea and Amundsen Gulf was estimated at 1700-1800 in the period from 1972 to 1974 (LGL and ESL 1982). During winter and spring, polar bears mainly occur in transition zone areas off the west coast of Banks Island and in Amundsen Gulf, and to a lesser extent in areas off the Mackenzie Delta and Tuktoyaktuk Peninsula. Pregnant females den from November to April in coastal areas, although primarily along the west coast of Banks Island and therefore outside of the proposed Beaufort development zone. During the open water period, most polar bears move north with the retreating pack-ice, and this would also minimize the degree of concern regarding impacts of future hydrocarbon exploration and production activities or facilities.

As indicated in Figures 6-1 to 6-3, the potential effects of some normal activities, wastes and disturbances of the petroleum industry are expected to be a MINOR concern with respect to the regional population of polar bears. Human presence, solid waste disposal, stationary sources of airborne noise, artificial illumination, gas flares and the physical presence of offshore structures may individually or collectively alert and attract bears to sites of industrial activity. Although most bears attracted to industrial facilities would be sedated and transported away from these sites, some nuisance animals may have to be destroyed. This loss is not expected to be more than a MINOR area of regional concern. Activities, wastes and disturbances associated with development that would not alert or attract polar bears are considered of NEGLIGIBLE regional concern. Polar bears may be indirectly affected by formation water discharge at offshore production platforms and accumulate some trace metals or hydrocarbons if they prey on seals which have been attracted to sites of formation water discharge. However, this indirect effect is expected to be an area of NEGLIGIBLE to MINOR regional concern.

The potential effects of crude oil or refined fuel spills on polar bears are of MODERATE regional concern due to the documented susceptibility of this species to hydrocarbon induced mortality, thermoregulatory stress, and systemic disorders. The effects of oil spills on polar bears may be less severe if chemical dispersants are used to dissipate the slick, and are successful in reducing the potential for contact with surface oil masses (Figure 6-4).

Changes in the distribution and abundance of ringed seals in the Beaufort Sea region due to natural phenomena have been suggested to be the cause of changes in the distribution and abundance of polar bears. However, since normal industrial activities in the region are not likely to cause extensive seal mortality or marked fluctuations in seal abundance, indirect impacts on bears through reduced prey availability are not expected. On the other hand, these indirect effects could occur if a large spill or blowout resulted in marked changes in ringed seal distribution and/or abundance, and would be considered an area of MODERATE concern with respect to the regional population of bears.

6.1.6 Arctic Fox

Arctic foxes from coastal populations in the Beaufort Sea region forage on the landfast ice during winter and spring. Their primary prey items include seal carrion and ringed seal pups, and during the open water period, Arctic foxes go ashore to den (LGL and ESL 1982).

Foxes may be affected by offshore hydrocarbon development activities when they occur on the landfast ice. Like polar bears, various normal activities and disturbances may alert and attract foxes to sites of industrial activity. Attraction may occur as the result of the combination of human presence, presence of artificial structures, airborne noise, gas flares, artificial illumination and solid waste disposal. However, the degree of regional concern associated with the combination of these attractive sources is expected to be MINOR. Mortality of foxes is unlikely in most instances, although animals thought to be rabid would probably be destroyed. Most normal activities, wastes and disturbances that would not result in attraction of foxes are of NEGLIGIBLE regional concern. A possible exception would be the potential for uptake of hydrocarbons and trace metals which may be present in prey (e.g. seals), although this is only considered of MINOR regional concern.

The potential effects of crude oil and refined fuel spills on foxes are of MINOR regional concern (Figure 6-4). Effects of oil on this species would be indirect, and are only possible if individuals ingested contaminated prey or if their fur became oiled during contact with prey. Foxes affected in this manner would probably suffer systemic disorders and thermoregulatory stress similar to polar bears, although the number of individuals affected would likely be insignificant in terms of the size of the regional population. Although speculative, changes in either the distribution or abundance of the ringed seal population as a result of a major oil spill or blowout may indirectly lead to decreased productivity of Arctic foxes. Nevertheless, these effects are only considered of MINOR regional concern because fox populations fluctuate dramatically under natural conditions, and indirect effects of spills or blowouts are not likely to cause abundance changes of this magnitude.

6.2 BIRDS

6.2.1 Loons

The three species of loon which are common summer residents of the Beaufort Sea region are the Arctic loon, red-throated loon and yellow-billed loon. The first two species commonly breed throughout the region, while yellow-billed loons are common migrants but rarely breed in the Beaufort region. Loons could be affected by offshore exploration and production activities primarily during spring and fall migration, which both occur over marine areas (LGL and ESL 1982).

As indicated in Figures 6-1 through 6-3, icebreaking, treated sewage discharge, solid waste disposal, discharge of cement slurry and heated water, and underwater shock waves are all of NEGLIGIBLE regional concern with respect to loons. On the other hand, the presence of artificial structures, dredging activities, airborne noise and gas flares may be of MINOR regional concern because of the potential for these activities to directly disturb loons or result in collisions with offshore facilities.

The potential effects of formation water discharge on loons are considered an area of MODERATE regional concern because (1) of the documented susceptibility of this species to petroleum hydrocarbons, (2) their highly aquatic nature and migration routes over offshore waters where formation water would be discharged, and (3) the relatively large quantities of produced water containing emulsified oil that could be released during the development. Human presence is also a MODERATE concern due to the potential for encroachment onto coastal and inland nesting areas by industry personnel, and the probable vulnerability of loons to these disturbances.

Crude oil or refined fuel spills in the Beaufort Sea are an area of MAJOR concern with respect to the regional populations of loons. As indicated earlier, loons are particularly susceptible to oil because they are highly aquatic, and at the same time, have a low reproductive potential (LGL and ESL 1982). The use of chemical dispersants in offshore waters is expected to reduce this degree of concern to MODERATE if large surface slicks are dispersed and either surface slicks or emulsified oil do not reach coastal areas.

Multiple sources of disturbance and wastes at offshore facilities of the petroleum industry may result in cumulative impacts on loons. For example, loons attracted to heat or light sources at production platforms may then be affected by formation water or airborne noise. Nevertheless, the cumulative effects of multiple wastes and disturbances associated with normal operations of the petroleum industry are still not expected to be more than a MINOR to MODERATE concern with respect to the regional populations of loons.

6.2.2 Ducks

King and common eiders, and oldsquaws are the only species of ducks that migrate offshore and commonly occur in the proposed offshore production zone of the Beaufort Sea. Other ducks including scaup, scoters, mergansers, pintails and American wigeon migrate overland, while all species nest and moult in coastal and backshore environments throughout and to the east of the Beaufort region (LGL and ESL 1982). Consequently, most offshore activities would only potentially affect eiders and oldsquaws during spring and fall migration, but coastal activities could affect several species from about late June to September.

Icebreaking, shock waves, and the discharge of treated sewage, solid wastes, cement slurry and heated water are all of NEGLIGIBLE regional concern with respect to duck populations in the Beaufort Sea region. The presence of offshore artificial structures and gas flares on production islands may have MINOR effects on eiders and oldsquaws in offshore areas through potential attraction and/or collisions. Dredging activities in both coastal and offshore areas may result in localized disturbance or reduced food availability for some species, but these effects are only of MINOR regional concern.

Wastes or disturbances associated with normal operations of the petroleum industry which may be of more significant regional concern include human presence, noise produced by aircraft, and the release of formation water. Like loons, ducks are susceptible to human encroachment onto nesting and brood-rearing areas, and the degree of regional concern is considered MODERATE. Some species of ducks may also be subject to increased hunting pressure. Airborne noise produced by aircraft is an area of MODERATE concern with respect to ducks, due to the potential for disturbance of staging migrants during spring (eiders and oldsquaws), nesting birds during summer, and moulting and brood-rearing birds in the summer and early fall.

The discharge of formation water is of MODERATE concern with respect to regional populations of oldsquaws and eiders. During spring and fall migration, these species occur in offshore waters where formation water would be released, and like loons, ducks are highly aquatic and therefore susceptible to mortality or various sublethal effects following direct contact with oil. Other effects of produced water on ducks may include localized reductions in food availability and uptake of trace metals, which are considered of NEGLIGIBLE and MINOR regional concern, respectively.

Crude oil or refined fuel spills are a MAJOR regional concern with all species of ducks which frequent marine areas in the region during part of their annual cycle (Figure 6-4). Ducks are highly vulnerable to hydrocarbon induced mortality, as well as a range of sublethal and indirect effects associated with oil exposure. Chemically dispersed oil is of lesser concern (MODERATE) with respect to eiders and oldsquaws and of NEGLIGIBLE to MINOR concern for other species of ducks, assuming that the oil is dispersed offshore, does not remain in the upper water column or coalesce to reform surface slicks, and does not reach the coast.

There is a potential for various combinations of wastes and disturbances associated with normal activities of the petroleum industry to have synergistic effects on ducks, particularly eiders and oldsquaws which occur in large numbers during migration throughout the offshore development zone. An example of a potential synergistic effect was discussed in relation to the attraction of loons to offshore facilities (Section 6.2.1), and could also occur with ducks. Although speculative at present, cumulative effects of hydrocarbon development on offshore migrants could occur as a result of formation water discharge, airborne noise and gas flares, while coastal species may be affected in a similar manner by the combination of other sources of disturbance such as airborne noise at airports, human encroachment onto nesting areas and dredging of harbours.

6.2.3 Geese and Swans

Whistling swans and four species of geese (brant, Canada geese, white-fronted geese, snow geese) are common summer residents in the Beaufort Sea region. With the exception of brant, these species migrate overland to reach their nesting areas, and nesting birds are not typically associated with the marine environment during the breeding period (LGL and ESL 1982). The migrations of brant are largely coastal, and they nest colonially or as dispersed pairs, often just above the high tide line. Moulting and brood-rearing brant are known to feed on vegetation in the littoral zone prior to fall migration (LGL and ESL 1982).

Since geese and swans are largely terrestrial, there are few activities, wastes or disturbances associated with normal petroleum industry operations that could affect these species (Figures 6-1 to 6-3). Potential areas of regional concern primarily include airborne noise produced by aircraft and human presence. Depending on flight frequencies, routes, altitudes and types of aircraft, there is a MINOR to MODERATE concern regarding the effects of aircraft on white-fronted geese, brant, Canada geese and swans, and MODERATE to MAJOR concern with respect to the effects of airborne noise on snow geese. The degree of potential concern associated with stationary sources of airborne noise is expected to be MINOR for snow geese, and NEGLIGIBLE for the other species listed above. Aircraft may affect geese or swans through disturbance at nesting locations or at staging areas, potentially resulting in reduced productivity or fitness for migration, or habitat loss through exclusion. There is also a MODERATE degree of regional concern that human encroachment onto nesting or staging areas may disturb, affect the productivity, or cause habitat loss for geese and swans in the Beaufort region. In addition, cumulative effects on these species may occur as a result of the combined influence of aircraft activity and human presence.

The degree of concern regarding the effects of crude oil or refined fuel spills on some geese and swans is considered MAJOR, since the habitats of these species (and therefore nesting birds) could be contaminated for several generations. The species likely to be the most seriously affected by spills are brant because they frequently occur near the marine environment, and snow geese because they nest at only two coastal colonies in the region. Assuming that the application of chemical dispersants prevents oil from reaching coastal areas, there should be no interaction between geese or swans and chemically dispersed oil, and the overall degree of concern regarding the effects of spills or blowouts which occur and are dispersed in offshore waters should be substantially reduced.

6.2.4 Shorebirds

At least 27 species of shorebirds are known to nest in coastal areas adjacent to the Beaufort Sea (LGL and ESL 1982). All species but red and northern phalaropes migrate overland, while the latter two species migrate over the offshore Beaufort and forage in marine areas during summer. During July, August and September, adults and juveniles of many shorebird species stage along coastal barrier beaches throughout the region.

With the exception of phalaropes, most normal activities, wastes and disturbances associated with normal petroleum industry operations would not affect shorebirds (Figures 6-1 to 6-3) or are of NEGLIGIBLE regional concern. Aircraft activity is the only MINOR concern, while encroachment onto nesting areas by industry or non-industry personnel would be of MODERATE concern. Cumulative effects that result from the combination of these sources of disturbance are considered possible in some areas, although the degree of concern associated with cumulative impacts of multiple disturbances is not expected to be more than MODERATE.

The regional concern regarding the potential effects of offshore activities, disturbances and wastes such as presence of exploration or production islands, gas flares, icebreaking, dredging, and the discharge of treated sewage and heated water on phalaropes is NEGLIGIBLE. However, the discharge of formation water is of MINOR concern with respect to the regional population of this species, due to the potential for mortality of individuals contacting any emulsified oil which coalesces and forms surface slicks.

Spills of crude oil or refined fuel are also considered a MAJOR concern with shorebirds, particularly if surface oil slicks were to innundate coastal nesting or staging areas. Phalaropes may contact oil during these periods, as well as during spring migration when they are present in offshore areas. Chemically dispersed oil in the upper portion of the water column may also affect phalaropes in offshore areas, and is considered an area of MINOR regional concern.

6.2.5 Jaegers, Gulls and Terns

Jaegers, gulls and terns are highly aerial birds that occur in the Beaufort Sea region during spring, summer and fall. Jaegers nest in tundra habitats, although migrants and non-nesting birds forage in marine areas. Glaucous gulls and Arctic terns nest at several colonies located along the Beaufort Sea coast, and occur offshore during migration as well as throughout the open water season (LGL and ESL 1982).

Gulls, terns and jaegers are less susceptible to activities of the petroleum industry because they are highly aerial and opportunistic feeders. As indicated in Figures 6-1 to 6-3, the effects of icebreaking, dredging and discharge of sewage and heated water are of NEGLIGIBLE concern with respect to the regional populations of these species. However, potential effects of the presence of artificial structures, airborne noise, solid waste disposal, human presence and gas flaring are considered probable. The combined effects of these activities and disturbances are of MINOR regional concern because they may all contribute to the potential attraction of birds to sites of human and industrial activity. Attraction to such sites may lead to synergistic effects such as exposure to contaminants present in formation water, which are themselves considered an area of MINOR regional concern.

Disturbance of nesting gulls and terns through human encroachment is of MODERATE concern because it could lead to reduced productivity and reduced fledgling success at a colony. Cumulative effects of petroleum industry operations on gulls and terns are also considered possible as the result of several activities that may cause their attraction to industrial sites in conjunction with potential disturbance by aircraft or human activity near nesting colonies.

There is a MODERATE degree of concern regarding the potential effects of crude oil or refined fuel spills on these species. Mortality or sublethal effects of oil contact could occur with both non-breeding birds foraging offshore and adults nesting along the coast. Nevertheless, these species are not considered as vulnerable to oil as the more highly aquatic birds such as ducks or loons. The effects of chemically dispersed oil on jaegers, gulls and terns are of MINOR concern, if oil does not reach coastal nesting areas or reform surface slicks in either coastal or offshore areas.

6.2.6 Alcids

There are only two alcid nesting colonies in the Canadian Beaufort Sea region. A small colony of black guillemots nest at Herschel Island, while a colony of approximately 800 thick-billed murres nest at Cape Parry (LGL and ESL 1982). Although little is known regarding the timing or migration of birds from either colony, it may occur over the offshore Beaufort. Consequently, murres and guillemots from this region are only likely to be affected by hydrocarbon development activities in the vicinity of the nesting colonies and during migrations.

Like loons and ducks, the presence of artificial structures and gas flares may be a MINOR area of concern with alcids during migration. Some individuals of these species may be attracted to and/or collide with offshore structures because they fly at low altitudes and are relatively unmaneuverable in flight.

Other offshore activities such as dredging and disburbance by airborne noise are of MINOR regional concern with respect to alcids, while icebreaking, sewage discharge, heated water release and solid waste disposal are considered of NEGLIGIBLE regional concern (Figures 6-1 to 6-3). On the other hand, the discharge of formation water is a MODERATE area of concern due to the exceptional vulnerability of alcids to the oil which may be present in emulsified form in this waste and subsequently coalesces to form surface slicks near production platforms. This oil would be most likely to affect offshore migrants and could result in mortality of individuals contacting discontinuous surface slicks during these periods. Cumulative effects of offshore activities on alcids may also occur during migration as a result of multiple sources of disturbance and various wastes discharged to the region, although collectively, potential concerns are not expected to be greater than MODERATE since most effects would be localized and unlikely to involve more than a small proportion of the regional populations of alcids. Aircraft overflights and potential encroachment onto colonies by industry personnel are considered another area of MODERATE concern with murres and guillemots during the nesting period.

Due to the vulnerability of alcids to oil, the effects of spills or blowouts is a MAJOR concern with respect to the regional populations of murres and guillemots during migration and nesting (Figure 6-4). Oil dispersed by chemical agents in offshore areas is of lesser concern (MODERATE) if emulsified oil does not remain in the upper portion of the water column, reform surface slicks or reach the two coastal nesting colonies in the region.

6.2.7 Other Marine-Associated Birds

Other marine or marine-associated birds found in the Beaufort region include fulmars, shearwaters, cormorants, grebes, ospreys and sandhill cranes (LGL and ESL 1982). These populations are discussed together because they are not abundant in the region. The first three groups would be affected primarily by offshore activities and facilities, while the latter species could be affected by coastal and shorebased activities.

As described in previous sections, the physical presence of offshore artificial structures, artificial illumination and gas flares at production islands may lead to the attraction of some birds. Although there is a potential for mortality of a few individuals through collisions, the degree of regional concern regarding these losses is expected to be NEGLIGIBLE or MINOR, depending on species. Icebreaking, discharge of sewage and solid wastes are of NEGLIGIBLE concern for all species, while dredging may result in MINOR effects through disturbance or reduced food availability for highly aquatic species such as fulmars, shearwaters, cormorants and grebes. As in the case of virtually all of the marine-associated birds discussed in previous sections, the oil content of formation water is an area of MODERATE regional concern for these species. Noise-related disturbances from aircraft operations is expected to be a MINOR concern for both offshore and coastal species in this category, while encroachment onto nesting areas by industry and non-industry personnel is considered a NEGLIGIBLE to MODERATE concern, depending on species.

The potential effects of crude oil or refined fuel spills are a MAJOR concern for all species of birds in this category (Figure 6-4). Mortality due to contact with chemically dispersed oil in the upper portion of the water column is a MODERATE concern with respect to those birds which occur in offshore areas during some part of their annual cycle.

As indicated in earlier sections, some combinations of facilities, wastes and disturbances associated with normal operations of the petroleum industry may result in cumulative impacts on species of birds included in this group. However, the degree of concern is only considered NEGLIGIBLE to MINOR because few individuals in the relatively small regional populations of these species would be affected by the widely separated activities in offshore and coastal areas.

6.3 FISH

6.3.1 Pelagic Marine Species

Arctic cod and Pacific herring are the most common pelagic marine fish found in the Beaufort Sea (LGL and ESL 1982), although there are only limited data describing the distribution, abundance and life history of all species of fish in offshore waters of the region. During summer, Arctic cod are probably widely distributed throughout offshore areas, and also have been observed feeding in large numbers in some nearshore waters (EIS Volume 3A; Section 3.4), where they feed primarily on mysids, amphipods and copepods. This species spawns during the winter (probably November to March), and available information suggests that spawning areas may occur in both shallow coastal areas and offshore waters (Tarbox and Moulton 1980; Craig and Haldorson 1981). The majority of individuals probably remain in offshore areas throughout most of the winter. Arctic cod have apparently been attracted to physical structures such as ships and docks in the Beaufort Sea (Tarbox and Spight 1979).

In the southeastern Beaufort Sea, Pacific herring are most abundant along the Tuktoyaktuk Peninsula and in Liverpool Bay (LGL and ESL 1982). They are also particularly abundant within Tuktoyaktuk Harbour (Byers and Kashino 1980). Fenco and Slaney (1978) suggest that Pacific herring may spawn in areas with macrophyte growth (e.g. Liverpool Bay, Eskimo Lakes) during June and July.

The probable interactions between these and other pelagic marine fish species and petroleum industry facilities or activities were summarized in Figures 6-1 to 6-3, while the degree of concern associated with the effects of environmental emergencies on marine resources is indicated in Figure 6-4. The degree of concern regarding the impacts of most activities or wastes associated with hydrocarbon exploration/production on pelagic marine fish is expected to be NEGLIGIBLE. However, dredging operations, underwater industrial noise, drilling fluid discharge and underwater shock waves (from ice management programs using explosives) are considered MINOR areas of potential concern with respect to regional populations of these species. The most significant (MODERATE) sources of concern are the trace metal content of formation water discharged to offshore areas (Section 3.1.10), the presence of exotic organisms in ballast water (Section 4.3.2), and refined or crude oil spills (Section 5.2.6), particularly if chemical dispersants are used.

The limited information which is available for the Beaufort Sea indicates that formation waters examined to date contain elevated concentrations of some dissolved trace metals, although not at levels which are generally considered a hazard in the marine environment. However, the reason for the MODERATE degree of concern related to trace metals in formation water is that (1) some species of fish such as Arctic cod may be attracted to and remain in the general vicinity of offshore production islands, (2) relatively large volumes of formation (produced) water may be released to the Beaufort Sea during peak production, and (3) several trace metals have been shown to be accumulated in the tissues of fish. Further investigations of the concentrations of trace metals in Beaufort Sea formation waters and probable rates of produced water discharge from offshore facilities may be required to assess the short- and long-term significance and overall concern related to the effects of trace metals on offshore fish populations.

The potential role of ballast water discharge in the introduction of exotic species to some parts of the world was discussed in Section 4.3, and on the basis of available information, cannot be dismissed as a relatively significant area of possible concern in the Beaufort Sea. At peak production, up to 200,000 m^3 of ballast water loaded in icebreaking tankers from temperate and subarctic regions could be discharged daily at only two locations in the Beaufort Sea. Although adult fish (>2 cm) would not be transported in ballast water due to screens on intake lines, some larval stages of marine fish could be entrained during ballasting and may survive the average 12 day voyage to the Beaufort. These organisms could then pose a risk to the integrity of indigenous fish populations <u>if</u> they are able to survive and successfully reproduce. In addition, the introduction of certain disease organisms or parasites could result in serious impacts on local fish populations. The potential degree of concern related to the transport of exotic organisms to the Beaufort Sea in ballast water could be reduced by (1) undertaking regular microbial analyses of water in areas where ballast water is loaded, (2) avoidance of areas (for ballast intake) where domestic or industrial wastes are released, (3) installation and regular maintenance of fine mesh screens on ballast intake lines, and (4) investigation of potential methods of reducing the survival of entrained organisms such as treatment with short-lived biocides or hypochlorite at ballasting locations.

Although fish mortality has been only occasionally observed following oil spills, the degree of concern regarding impacts of spills or blowouts on regional populations of pelagic marine fish is still considered <u>MODERATE</u> since a major event could cause a change in the abundance or distribution of some regional populations which persists more than one generation. A spill or blowout which contaminated coastal habitats along the Tuktoyaktuk Peninsula could affect the relatively abundant Pacific herring populations in these waters, particularly during June to August when the sensitive larval stages would likely be present in the water column. The effects of a spill or blowout on Arctic cod would likely be less severe than effects on herring since both adult and juvenile cod should be more widely distributed throughout the region. In general, blowouts could cause more serious impacts on pelagic marine fish than single spills because they would continuously introduce toxic water-soluble hydrocarbons to the water column, whereas these compounds would be rapidly dispersed and weathered following a spill. At the same time, a blowout under winter ice cover within the landfast ice zone could have more serious impacts on fish because the normally volatile (and toxic) constituents in crude oil would have less opportunity to dissipate through evaporation at the water surface. Diesel or gasoline spills would also be expected to have greater impacts on pelagic fish (compared to an equivalent volume of crude oil) since these refined petroleum products have a larger proportion of low molecular weight and aromatic hydrocarbons which are considered most acutely toxic to fish (Section 5.2.6).

There is some potential for synergistic effects of combinations of wastes and the physical presence of offshore structures in the Beaufort Sea, particularly with species of fish that may be attracted to these facilities (e.g. Arctic cod). As indicated in previous sections of this report, several types of wastes may be continuously or intermittently released to the marine environment surrounding offshore exploration or production islands and drillships. These wastes could include some combination of treated sewage, heated water, produced water, BOP control fluid, completion and maintenance fluids, formation cuttings, drilling fluids and oily wastes. Although any potential synergistic or antagonistic effects related to multiple waste sources are largely speculative, several cumulative but localized impacts are considered possible. For example, trace metal uptake from formation water, and to a lesser extent drilling fluids, may be increased in areas where thermal effluents are discharged, while the effects of low concentrations (<50 ppm) of emulsified oil from both oily wastes and formation water may be increased in the presence of either higher background concentrations of dissolved trace metals or ambient water temperatures. Similarly, the chance of survival of organisms released during ballast water discharge at combined production and tanker loading facilities may be increased when heated water is discharged for the purposes of ice management or when produced water release creates localized areas with above average water temperatures. Nevertheless, synergistic effects of multiple wastes are not expected to result in regionally significant impacts on populations of pelagic marine fish because of the rapid dilution of all industrial wastes expected in offshore waters of the Beaufort Sea. In addition, there is no evidence to suggest that multiple wastes discharged from production facilities elsewhere in the world have caused observable effects on fish populations.

6.3.2 Demersal Marine Species

As indicated in the previous section, there is only limited information on the distribution, abundance and life history of fish populations in offshore waters of the Beaufort Sea. This information was recently summarized by LGL and ESL (1982). The most common species of demersal marine fish in the region are fourhorn sculpin, snailfish, Arctic flounder, starry flounder and saffron cod. The fourhorn sculpin is usually associated with shallow brackish environments and is not believed to be abundant at depths greater than 15-20 m (Griffiths et al. 1975) where most offshore facilities of the petroleum industry would be located. The species spawns on the bottom during mid-winter in nearshore habitats, and feeds primarily on amphipods, isopods and mysids (Bendock 1977). In a similar manner, snailfish are normally associated with kelp or rocky substrates (Tarbox and Moulton 1980), and are therefore unlikely to be particularly abundant near offshore facilities in the Beaufort Sea.

The most probable demersal marine fish species expected in offshore environments are the flounders, and the life history and distribution of these fish are poorly documented in this region (LGL and ESL 1982). According to Bendock (1977), Arctic flounders tend to be most common in the outer margins of river deltas than in other coastal areas within the Beaufort Sea, and all flounders are believed to spawn and overwinter under ice in coastal areas.

In general, the potential interactions and degree of concern associated with the effects of hydrocarbon development on demersal fish species in the Beaufort Sea are similar to those discussed with pelagic marine species (Section 6.3.1). Consequently, the most significant concerns would be the potential effects of formation water, introduction of exotic organisms including pathogens and parasites (ballast water), and oil spills or blowouts. However, the lack of data describing fish populations in offshore regions where industry facilities would be located hampers, to a certain extent, assessment of the numbers of fish which may be affected by the facilities, sources of disturbance and wastes shown in Figures 6-1 to 6-3. Demersal species of fish could also be attracted to offshore structures and since they have a closer association with the substrate than pelagic species, may be more susceptible to the effects of drilling wastes other than formation water. These species could also be more seriously affected by wastes such as formation cuttings that result in habitat loss, although potential effects of discharges from offshore exploration and production facilities would still be relatively localized.

Demersal species may also be more susceptible than pelagic fish to the effects of oil spills and blowouts since the clay-silt substrates which predominate in the Beaufort Sea would tend to retain sedimented and/or sunken oil. Habitats which support demersal species and do not receive large quantities of sediment transported by the Mackenzie River would likely be most seriously affected by spills or blowouts because oil would not be as rapidly buried. This is the primary reason for the MODERATE degree of concern shown in Figure 6-4 in relation to habitat and food source loss as well as hydrocarbon accumulation. As in the case of pelagic marine fish, blowouts would generally pose a greater risk to regional demersal fish populations than single spills. The effects of blowouts or spills on demersal species would also likely increase with a decrease in water depth since larger quantities of oil would tend to reach the sea floor. This latter trend has been clearly demonstrated in the oil spill case history literature (Duval *et al*. 1981).

6.3.3 Anadromous Species

During the summer months, coastal habitats of the Beaufort Sea are utilized as feeding grounds and migration corridors by anadromous fish. The most abundant species found in this region are Arctic cisco, least cisco, broad and humpback whitefish, boreal smelt and Arctic char, although studies conducted to date in the Beaufort indicate that these species are generally restricted to a narrow nearshore corridor where water depths are less than 5 m (LGL and ESL 1982). As a result, most activities of the petroleum industry in offshore waters are unlikely to affect a significant proportion of the regional populations. The distances which anadromous fish travel along the coast and the timing of their occurrence in coastal habitats vary with species and life history stage. For example, most mature anadromous fish do not spawn each year, and in some species, the spawning segment of the population may either remain in freshwater habitats during the summer, or undertake short coastal migrations in the early summar, returning to spawning rivers early in the open water season (McCart 1980; Craig and Haldorson 1981). Consequently, immature fish and mature non-spawners tend to be more prevalent in coastal waters, and migrate greater distances and remain for longer periods away from their native rivers compared to the season's spawning fish (LGL and ESL 1982).

Anadromous fish almost exclusively utilize coastal habitats for feeding during the open water season, rather than for spawning or overwintering. There is also a large degree of overlap in the diet of most anadromous species, with epibenthic crustaceans (particularly mysids and amphipods) being the dominant food source (LGL and ESL 1982).

Due to the distribution of anadromous fish species in the Beaufort region, most interactions with petroleum industry activities and facilities would occur in nearshore coastal environments rather than offshore waters where most exploration programs and production facilities would be located. Nevertheless, there is a potential for interactions between industry activities and anadromous fish populations in some areas such as near shallow production fields (e.g. Adgo), coastal borrow sites (e.g. Tuft Point) and shorebases (e.g. King Point, Tuktoyaktuk and McKinley Bay). As indicated in Figures 6-1 to 6-3, the degree of concern regarding the effects of most exploration and production-related activities on anadromous fish populations of the Beaufort Sea is expected to be NEGLIGIBLE to MINOR. As in the case of demersal and pelagic marine species, the potential effects of trace metals in produced water, exotic organisms in ballast water, and oil spills or blowouts are considered MODERATE areas of concern. In addition, dredging activities in coastal areas are also an area of MODERATE concern with respect to anadromous fish during the open water season (Figure 6-1).

Since most production facilities would be located in offshore waters, anadromous fish are less likely to be directly exposed to elevated concentrations of trace metals associated with discharge of drilling wastes and formation water. Depending on water circulation patterns and dilution factors, some areas with elevated trace metal concentrations could occur within the 5 m isobath and thereby directly affect anadromous fish species. However, the most probable path of uptake of trace metals by these species would be through the ingestion of contaminated epibenthic crustaceans since Craig and Haldorson (1981) suggest that there is an annual immigration of prey species into coastal waters as a result of the exchange between nearshore and offshore waters. As a result, epibenthic invertebrates feeding in offshore waters near production facilities throughout the winter could accumulate trace metals which are subsequently transferred to anadromous fish when prey organisms migrate into the coastal feeding habitats during summer.

Most potential effects of ballast water discharge on anadromous fish would also be indirect since tanker loading facilities would be located in offshore areas well beyond the coastal habitats occupied by these species during the open water season. Unlike many pelagic and demersal marine species which have planktonic larvae, juvenile stages of anadromous fish in temperate latitudes would not be entrained during ballast water intake. As a result, transfer of competing anadromous species to the Beaufort Sea in icebreaking tankers would be extremely unlikely. On the other hand, anadromous fish in this region rely on relatively few prey species (LGL and ESL 1982), and successful introduction of crustaceans could be an area of potential concern if they eventually displaced or reduced the abundance of indigenous fauna. However, this phenomenon would be extremely unlikely since widespread changes in community structure in offshore habitats would be required for regionally significant indirect impacts on anadromous fish populations. The most significant area of potential concern with respect to ballast water discharge would be the introduction of pathogenic micro-organisms and parasites to the Beaufort Sea. Survival of these organisms in the ballast water of tankers could pose a risk to anadromous fish species in the region because of the documented susceptibility of some anadromous fish to foreign parasites and disease organisms. However, the avoidance of areas where industrial/domestic wastes are discharged for ballast intake and regular monitoring of waters in the region of southern tanker terminals would minimize this area of potential concern. In addition, the treatment of ballast water with biocides is a recommended area of future investigation to further minimize the risk of introduction of pathogens or parasites to the Beaufort Sea.

Only limited dredging in nearshore waters inhabited by anadromous fish should be required for future hydrocarbon exploration and production in the Beaufort Sea since borrow material for offshore island construction would be generally taken from offshore granular deposits (EIS Volume 2). Nearshore dredging will be required for the maintenance of some harbours (e.g. Tuktoyaktuk), expansion of shorebases (e.g. McKinley Bay), construction of new shorebases (e.g. King Point), completion of subsea pipeline systems in nearshore areas (North Head), and construction of artificial islands in shallow waters. As indicated in Figure 6-1, these operations could directly and indirectly affect anadromous fish populations through entrainment, habitat loss, decreased abundance of prey species and various sublethal effects associated with turbidity plumes and changes in water quality, although most effects would be extremely localized and insignificant in terms of regional populations of these species. Nevertheless, the degree of concern associated with the effects of dredging operations on anadromous fish would be considered MODERATE if these activities occurred in habitats which are within the narrow migration corridors or concentrated feeding areas of some species during the open water season. As a result, fisheries investigations are recommended in any nearshore areas where dredging operations are proposed and the open water use of these habitats by fish remains poorly documented (e.g. King Point, Herschel Island).

Oil spills and blowouts are another area of MODERATE concern with respect to regional populations of anadromous fish in the Beaufort Sea. Despite the fact the mortality of fish has been relatively rare following past spills in subarctic, temperate and tropical waters (Duval et al. 1981), an accidental spill or blowout which contaminates nearshore habitats of this region could result in a change in the abundance or distribution of fish populations which persists more than a single generation, particularly due to the fine-grained nature of the substrate which would tend to retain oil and cause persistent indirect and sublethal impacts on fish. It should also be emphasized that a spill or blowout during the winter could also affect anadromous fish, despite the absence of these species in the region at this time of year, due to the high oil retention potential of Beaufort Sea sediments. The most serious impacts of spills or blowouts could occur when these events result in contamination of nearshore and lagoon habitats, and as in the case of marine species, blowouts and refined fuel spills would likely result in more serious short- and long-term impacts on anadromous fish.

6.4 LOWER TROPHIC LEVELS

6.4.1 Phytoplankton

Available information regarding the species composition, distribution and abundance of phytoplankton in the Beaufort Sea and N.E. Chukchi Sea was summarized in LGL and ESL (1982). These organisms are only abundant in the region during the open water season, and generally tend to decrease in abundance with increased distance from the shoreline. The species composition and abundance of phytoplankton communities in arctic waters are highly dependent on seasonal and local differences in the physical-chemical environment, and these differences, in conjunction with biological factors such as grazing pressure, contribute to dynamic seasonal patterns of succession. In the offshore waters where most industrial facilities would be located, flagellates would be the predominate forms during most of the open water season, while diatoms should be most abundant in nearshore waters.

All normal activities, wastes and facilities associated with hydrocarbon development in the Beaufort Sea are expected to be either NEGLIGIBLE or MINOR areas of concern with respect to regional phytoplankton communities, with MINOR concerns being restricted to the potential effects of drilling wastes and ballast water (exotic organisms) on these organisms (Figures 6-1 to 6-3). However, oil spills and blowouts are expected to be a MODERATE area of potential concern, particularly in nearshore habitats where rates of primary production are highest (LGL and ESL 1982).

The degree of concern regarding the effects of drilling wastes (including formation water) on phytoplankton is only considered MINOR since (1) most exploration and production activities would be located in offshore waters where phytoplankton are not particularly abundant, (2) exposure to drilling wastes would be relatively short-term because organisms would be continuously transported into and out of areas affected by discharges, and (3) any adverse effects would be restricted to a single generation of phytoplankton.

Exotic organisms present in ballast water are also expected to be of less concern with phytoplankton than fish because survival of exotic phytoplankton species in ballast water, and subsequent survival and reproduction in the Beaufort Sea are all relatively unlikely as a result of the complete darkness during transport and the relatively harsh physical-chemical environment of the Beaufort Sea.

The impacts of oil spills or blowouts on phytoplankton of the Beaufort region would likely be most severe if oil reached the more productive coastal environments. In these areas, the degree of regional concern is considered MODERATE because of the potential for sedimented/sunken oil to affect several generations of phytoplankton and have subsequent impacts on members of higher trophic levels. The use of chemical dispersants in these shallow environments would probably increase the severity of acute toxic and sublethal effects of oil due to their tendency to increase concentrations of soluble hydrocarbons within the water column.

6.4.2 Zooplankton

As in the case of phytoplankton, there are marked seasonal and spatial differences in the species composition and abundance of zooplankton in the Beaufort Sea (LGL and ESL 1982). The diversity of zooplankton communities tends to be highest in offshore waters where the majority of the petroleum industry activities would be centered, although the abundance of zooplankton is also lower in these areas than in some sheltered coastal embayments and in nearshore waters along the Tuktoyaktuk Peninsula.

The potential interactions between zooplankton communities and future hydrocarbon development activities or environmental emergencies are summarized in Figures 6-1 to 6-4. As with phytoplankton, all activities, wastes and facilities associated with normal operations are expected to be of either NEGLIGIBLE or MINOR concern with respect to regional zooplankton populations and secondary production by these organisms. The degree of concern regarding drilling waste (including produced water) and ballast water discharges are only considered MINOR, since any potential adverse effects would be highly localized and only involve short-term exposure of a small proportion of regional populations to these wastes due to the natural transport of planktonic organisms into and out of waters near discharge sites. For the same reasons, significant synergistic impacts of multiple waste discharges at offshore exploration or production facilities would not be expected.

The potential effects of oil spills or blowouts on regional zooplankton populations are considered a MODERATE area of concern (Figure 6-4), and as with phytoplankton, the use of chemical dispersants would likely increase the severity of impacts. In a similar manner, contamination of nearshore habitats along the coast of the Tuktoyaktuk Peninsula or some productive embayments would likely have more serious effects on regional zooplankton populations because of (1) the greater abundance of zooplankton in these areas, (2) the probable importance of secondary production from these environments to the region as a whole, and (3) the potential for chronic release of hydrocarbons from stranded oil masses, thereby resulting in at least sublethal effects on several generations of zooplankton.

6.4.3 Benthic Epifauna

Epibenthic invertebrates are an important trophic link between primary producers and vertebrate predators in the Beaufort Sea, and mobile crustaceans such as amphipods, mysids and isopods form a substantial portion of the diet of many fish species as well as some birds and marine mammals (LGL and ESL 1982; EIS Volume 3A, Section 3.1). Epifauna are often abundant along the Beaufort coastlne during the open water season, and certain localized habitats such as Mason Bay (Richards Island) and a spit-protected bay on Hooper Island appear particularly productive (Slaney 1973; Wacasey 1975). Available data for the region suggest that there is an active emigration of mobile epifauna to deeper water for overwintering (LGL and ESL 1982). Sessile epifauna such as hydroids, barnacles and anemones are only relatively abundant on hard substrates, and are therefore limited in distribution to relatively few areas in the region (e.g. west coast of Banks Island).

The potential for effects of normal petroleum industry activities and environmental emergencies on benthic epifauna would likely be greatest during the winter since most exploration or production facilities and activities would occur in offshore waters where these species are thought to overwinter. In addition, certain activities such as dredging could affect some species of crustaceans in nearshore habitats during the open water season. The most significant (MODERATE) areas of concern associated with interactions between benthic epifauna and future hydrocarbon exploration and production programs in the Beaufort Sea are the potential effects of nearshore dredging operations, drilling waste disposal, ballast water discharge, and oil spills or blowouts (Figure 6-1 to 6-4). In addition, the routine release of cement slurry or occasional disposal of powdered cement may result in localized MINOR impacts on benthic epifauna.

The majority of the proposed borrow locations and artificial island construction sites are located in offshore waters of the Beaufort Sea where the abundance of epibenthic invertebrates is expected to be relatively low. As a result, dredging operations in these areas are not expected to result in significant effects on regional populations of these organisms. On the other hand, some dredging programs may be required in nearshore environments (e.g. North Head, Tuft Point, King Point) during the open water season. Due to the greater abundance of epifauna in these areas and their documented importance as prey organisms of anadromous fish species in relatively confined feeding habitats and migration corridors, large scale dredging programs could be a MODERATE area of concern due to indirect impacts on members of higher trophic levels.

As indicated earlier in Section 3.1.8.1, mobile epibenthic invertebrates will likely be able to avoid direct burial by formation cuttings and solids present in drilling fluids released during exploratory and production drilling in offshore waters of the Beaufort Sea. However, organisms present in the immediate vicinity of offshore platforms may be exposed to elevated dissolved trace metal levels during periods of continuous or intermittent formation water discharge. The largest numbers of individuals could be exposed to and accumulate trace metals during the winter months when some mobile species apparently emigrate to offshore habitats to overwinter. Species such as *Saduria entomon* which are characteristically attracted to disturbed areas would be most susceptible to impacts from produced water and other drilling wastes, although most effects are expected to be of a sublethal nature and affect only a small proportion of regional populations. Nevertheless, the migration of some species of epibenthic crustaceans to nearshore waters during the open water season could result in indirect effects on coastal anadromous fish populations which prey extensively on these invertebrates during the summer (Section 6.3.3).

Survival of exotic organisms in ballast water collected from southern latitudes could affect the species composition of the epibenthic component of the benthic community, particularly if heated water discharge from combined production/tanker loading facilities increases the chance of survival and reproduction of foreign organisms. The most likely form of colonization at these offshore structures would be the establishment of sessile epifauna such as anemones, barnacles and hydroids on artificial hard substrates including concrete caissons. However, such changes in community structure, if possible, would be highly localized and extremely unlikely to affect the integrity of regional indigenous populations as a whole.

The results of numerous laboratory studies and the case histories of past oil spills (Section 5.2.10; Duval et al. 1981) suggest that epibenthic invertebrates (particularly crustaceans in intermoult) are relatively sensitive to petroleum hydrocarbons. As in the case of most marine resources in the Beaufort region, the impacts of spills or blowouts which affected shallow coastal waters with a high oil retention potential would likely be considerably greater than spills which were confined to deeper offshore areas. The degree of concern related to the effects of oil spills on benthic epifauna is considered MODERATE because (1) the potential for acute toxic and sublethal effects which persist for several generations, (2) the documented sensitivity of some epibenthic invertebrates to oil, (3) the high oil retention potential of most seafloor substrates throughout the region, and (4) the importance of these organisms to the marine food web of the Beaufort Sea.

As in the case of pelagic and demersal fish species, some combinations of wastes discharged from offshore petroleum industry facilities may result in cumulative impacts on local populations of epifauna. The most probable wastes which may lead to synergistic effects are trace metals and emulsified oil in produced water, organic additives to drilling fluids and thermal discharges which locally increase ambient temperatures and therefore probable rates of uptake/metabolism of contaminants. Nevertheless, synergistic effects of multiple waste sources should be restricted to small areas around exploration and production facilities due to the high dilution factor expected in offshore waters of the Beaufort Sea.

6.4.4 Benthic Infauna

Studies conducted in the Beaufort Sea to date have indicated that the diversity and species abundance of benthic infaunal component of benthic communities increases with distance from the shoreline, primarily due to the annual ice scour which occurs in shallow nearshore environments (Reimnitz and Barnes 1974; Wacasey 1975; LGL and ESL 1982). In addition, the distribution of infaunal benthic invertebrates throughout the continental shelf tends to be patchy due to frequent ice scour which causes reworking of the sediments. Polychaetes and bivalve molluscs are the predominate taxonomic groups represented in the infauna of the Beaufort region. Offshore exploration and production activities and facilities proposed by the petroleum industry would be primarily located in the estuarine (0-15 m) and transition (15-30 m) depth zones discussed by Wacasey (1975), with some structures extending into the marine zone (30-200 m) which is characterized by the highest biomass and diversity of benthic infauna.

Although the majority of the significant interactions between petroleum industry activities and benthic infauna in the Beaufort region would be similar to those previously discussed with benthic epifauna (Section 6.3.3), the sessile nature of the infauna and their greater relative abundance in offshore areas would result in differences in the regional significance of potential adverse effects. For example, offshore dredging operations are expected to result in more extensive mortality of infaunal organisms due to their limited mobility and greater abundance in these waters, while nearshore dredging is less likely to affect large numbers of benthic infauna. In addition, available information for the region suggests that the localized loss of benthic infauna due to dredging would have less significance to members of higher trophic levels since fewer species prey on these organisms in comparison to benthic epifauna. The degree of concern related to the effects of dredging on benthic infauna would still be considered MODERATE because several generations would be required for complete recolonization of disturbed areas, despite the fact that mortality would not be significant in a regional context due to the widespread distribution of infauna in the region and small proportion of offshore habitat that would be affected by dredging programs.

Benthic infauna would also be more susceptible than epifauna to direct burial by disposal of formation cuttings and solids in drilling fluids at offshore exploration and production platforms, and would be slower to recolonize areas adjacent to drilling waste disposal following completion of drilling programs. At the same time, their sessile nature would increase the likelihood of trace metal uptake from drilling fluids and produced water, although for the reasons mentioned earlier, this would be less likely (compared to epifauna) to affect members of higher trophic levels. As indicated in Figure 6-2, the degree of concern associated with various effects of drilling waste disposal on benthic infauna is expected to range from MINOR to MODERATE because of the long-term nature of some sources of waste discharge and disturbance. However, due to the small proportion of offshore habitat that would be affected by drilling wastes, mortality and sublethal responses to contaminants (including trace metal uptake) is not expected to be regionally significant.

There is a MODERATE degree of concern regarding the potential effects of oil spills and blowouts on benthic infauna in the Beaufort Sea. At the same time, the chronic exposure of infauna near production platforms to emulsified oil contained in formation water would likely result in a range of sublethal effects including hydrocarbon uptake, and could cause mortality of some organisms in areas where petroleum hydrocarbons become incorporated into the sediments. The recovery of infauna following oil spills or chronic hydrocarbon contamination would probably be substantially slower than that of the epifauna, and would depend on the rate of weathering of sediment-bound oil and/or burial of contaminated areas of the seafloor with sediments transported into the region by the Mackenzie River. The results of investigations conducted after oil spills in temperate latitudes (Duval et al. 1981) also suggest that differences in the sensitivity of infauna to petroleum hydrocarbons could result in relatively dramatic changes in community structure. Unlike the situation with epibenthic invertebrates and many species of fish, the most serious impacts of spills and blowouts on benthic infauna would likely occur if these events affected the more productive offshore habitats. The use of chemical dispersants in offshore areas is also expected to increase the degree of impact of oil spills on these organisms since larger amounts of oil would generally reach the substrate and become incorporated into the sediments.

Since benthic infauna are relatively immobile or sessile, they would be more susceptible to synergistic impacts of multiple waste discharges from offshore facilities of the petroleum industry. Depending on the type of offshore facilities (e.g. drillship, exploration island, production island, tanker loading facility), infauna could be simultaneously exposed to drilling wastes, thermal discharges, treated sewage, BOP control fluid and ballast water containing exotic organisms from southern latitudes. However, as in the case of most marine flora and fauna of this region, the cumulative impacts of multiple sources of disturbance and waste are not expected to pose a risk to regional populations of benthic infauna or result in more than localized changes in abundance, distribution and community structure.

6.4.5 Benthic Flora

Benthic microalgae are important primary producers in the marine food web of the Beaufort Sea, but due to the extremely turbid conditions which exist throughout much of the region during the summer, benthic algal growth is probably restricted to relatively shallow waters where light can reach the seafloor (LGL and ESL 1982). Prominent mats of benthic diatoms and filamentous green algae have been observed in some localized clearwater habitats such as in the lee of spits and offshore islands or in embayments away from the Mackenzie River plume. These habitats have generally also supported higher numbers of epibenthic invertebrates and fish (LGL and ESL 1982), and the importance of microalgae in the diet of shallow water amphipods, mysids and isopods is relatively well documented (Bray 1962; Schneider and Koch 1979).

Offshore activities and facilities of the petroleum industry are not expected to affect large numbers of benthic microalgae because they would be located in waters much deeper than those likely to support significant microalgal growth. This is the primary reason for the NEGLIGIBLE degree of concern indicated for most interactions shown in Figures 6-1 to 6.3. In some cases, dredging operations in nearshore waters could result in localized loss of microalgae, although algal populations are likely to fully recover during the following open water season. Drilling wastes discharged from exploration and production islands in shallow waters could result in mortality, habitat loss and various sublethal effects with microalgae, but the overall degree of concern associated with these effects is considered MINOR because of their probable localized and short-term nature.

Oil spills or blowouts which contaminate shallow nearshore habitats of the Beaufort Sea could seriously affect populations of benthic microalgae, and due to the importance of these organisms in the diet of various crustaceans and the probability that several generations of algae would be affected, the degree of concern is considered MODERATE. The slow release of petroleum hydrocarbons from oil masses stranded in nearshore habitats could lead to chronic sublethal effects which persist for several years, although recovery would be expected shortly after contaminated sediments were buried by fines transported into the region by the Mackenzie River. The application of chemical dispersants would be expected to increase the acute toxic and sublethal effects of oil on benthic microalgae. However, dispersants are unlikely to be used in shallow waters where these flora are relatively abundant.

6.4.6 Epontic Fauna

The distribution and abundance of epontic fauna under the winter ice cover in the Canadian Beaufort Sea have only been the subject of limited studies to date, but based on studies conducted in the Alaskan Beaufort, the community is likely to be dominated by gammarid amphipods. The direct observations of divers have shown that amphipods and other metazoans graze on the epontic flora under fast ice during the spring bloom (see LGL and ESL 1982). The utilization of epontic fauna by members of higher trophic levels is poorly documented, although they are known to be part of the diet of some species of birds, ringed seals and Arctic cod in some areas. The relative abundance and distribution of epontic fauna in landfast and transition zone ice areas where offshore facilities would be located are unknown, and this hampers assessment of the ecological significance of some petroleum industry activities, wastes and disturbances on these fauna, as well as indirect effects on vertebrates which rely to a greater or lesser extent on epontic organisms for a portion of their diet.

The degree of concern regarding the effects of most wastes and disturbances associated with offshore petroleum industry operations in the Beaufort Sea on epontic fauna is expected to be NEGLIGIBLE or MINOR (Figures 6-1 to 6-3). The most significant concern related to normal operations are the effects of some drilling wastes (Figure 6-2), although any impacts of drilling fluid and formation cutting disposal are expected to be very localized because of the rapid sinking of solids and dilution of dissolved compounds and elements. The trace metal and oil content of formation water may cause local sublethal effects, and emulsified oil which coalesces and becomes trapped in depressions in the under-ice surface would likely cause mortality of epontic fauna in these areas. The discharge of heated water during ice management programs at tanker loading facilities and/or production islands could act in a synergistic manner to increase the effects of trace metals and hydrocarbons, but such effects would be restricted to small areas surrounding waste discharge sites.

Epontic fauna would be highly vulnerable to oil spills and blowouts, particularly if relatively unweathered oil accumulated beneath the ice surface. Blowouts are expected to be the most serious area of concern with respect to offshore hydrocarbon development, and would likely cause virtually complete mortality of epontic fauna in areas where oil reaches the under-ice surface. The potential impacts of blowouts on these organisms are discussed in detail in two other supporting documents to this EIS (Duval et al. 1981; ESL 1982). The significance of widespread mortality of epontic flora to fish, bird and seal populations in the Beaufort Sea can not be fully assessed at the present time, although indirect impacts on some species through decreased food availability and hydrocarbon uptake would be anticipated. It has also been suggested that epontic amphipods become part of the benthic community during spring breakup, and as a result, losses of these organisms due to a blowout in the Beaufort Sea during winter could have subsequent impacts on the abundance of epibenthic fauna in the following open water season.

6.4.7 Epontic Flora

Microalgae are present in low abundance throughout sea ice from the time that it forms, but the highest concentrations occur during spring and early summer in the soft, crystalline bottom few centimetres of ice (LGL and ESL 1982). As indicated in the previous section, these flora are consumed by fauna that graze at the under-ice surface, but also may be sloughed off into the water column where they apparently cease photosynthesis and become a potential food source for detritivores. Pennate diatoms are the dominant component of the epontic microalgal community. Spatial and seasonal differences in the abundance of epontic flora in the zone which would be affected by future hydrocarbon exploration and production activities or facilities have not been examined, although some data on the ecology and physiology of these organisms have been collected in the Alaskan Beaufort.

The potential interactions between activities and facilities of the petroleum industry and epontic flora are summarized in Figures 6-1 to 6-3, and are generally similar to the potential effects of hydrocarbon exploration and production on the epontic fauna. Drilling waste disposal and the introduction of exotic organisms with ballast water are expected to be the only areas of more than NEGLIGIBLE regional concern (Figures 6-2 and 6-3), although oil spills and blowouts would also be a MODERATE area of concern with these organisms because of their close association with the under-ice surface. The potential effects of oil spills on epontic flora of the Beaufort Sea were discussed earlier in Section 5.2.11, and are treated in further detail in ESL (1982) and Duval *et al.* (1981).

6.4.8 Micro-organisms

Bacteria play a fundamental role in nutrient recycling processes in the marine environment, and are also an important heterotrophic component of epontic communities. Some species, called oleoclastic bacteria, can also degrade certain petroleum hydrocarbons and can assume a relatively major role in the weathering of oil (Fingas *et al.* 1979). In addition, some bacteria are important food sources for microzooplankton and benthic detritivores, while still others are pathogens to various flora and fauna. Studies directed at isolation and characterization of bacterial populations from the Canadian and Alaskan Beaufort Sea are summarized in LGL and ESL (1982). These investigations have shown that bacteria generally occur throughout the water column and within sediments of the region in concentrations similar to those measured in temperate latitudes. The maximum concentration of bacteria in the water column has been found to coincide with the post spring breakup phytoplankton bloom, and most species present in the summer are psychrotrophs (tolerate cold temperatures). On the other hand, bacterial populations isolated in the winter have been typically psychrophiles (grow better at cold temperatures).

Bacteria will likely play a role in the degradation of many of the wastes routinely discharged by the petroleum industry. Although rates of degradation are not well documented, bacteria are likely to break down many organic compounds contained in treated sewage, drilling fluids and BOP control fluid, as well as small quantities of organic matter uplifted from the sediments during dredging programs. On the other hand, microbial activity in the vicinity of drill rigs may also result in the mobilization of some of the trace metals contained in drilling fluids, although the degree of regional concern regarding this process is expected to be MINOR. Bacterial populations contained in ballast water discharged at tanker loading facilities may also be able to survive if local ambient temperatures are simultaneously increased by heated water discharge. However, this is not expected to threaten the integrity of regional bacterial populations, and the overall degree of concern is therefore considered MINOR. The most significant potential concern related to interactions between petroleum industry activities and bacteria is the introduction of pathogenic species to the Beaufort Sea and subsequent impacts of these disease causing organisms on vertebrate populations. However, the mitigative measures discussed earlier in Section 6.3.1 would minimize this area of concern.

The literature describing the presence of oleoclastic bacteria which could play a role in the degradation of oil chronically (i.e. oily waste water and produced water) or accidentally (i.e. spills and blowouts) released to the Beaufort Sea is inconsistent in terms of the abundance and activity of these micro-organisms. The studies summarized by LGL and ESL (1982) suggest that oleoclastic bacteria may occur in some parts of the region, but are likely present in low numbers due to the absence of petroleum contamination in this ecosystem. It is possible that increasing activity of the petroleum industry and concomitant chronic introduction of hydrocarbons to offshore waters will increase the relative abundance of oleoclastic bacteria. However, because of the low temperatures and predominance of fine-grained sediments which would retain oil, it is unlikely that microbial degradation would play as important a role in the weathering of oil released to the Beaufort Sea as it apparently does in temperate and tropical latitudes.

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