FINNISH INSTITUTE OF MARINE RESEARCH (FIMR)

GROWTH Project GRD2-2000-30112 "ARCOP"

Effects of oil spills on Arctic marine ecosystems

WP4.2.3.   ENVIRONMENTAL PROTECTION: BIOLOGY AND POTENTIAL EFECTS OF OIL SPILLS ON THE ARCTIC SEA ICE

WP4.2.3.2.   Effects of oil spills on Arctic marine ecosystems

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DELIVERABLE D4.2.3.2.

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CONTRACT N°:   GRD2/2000/30112-S07.16174
PROJECT N°:   GRD2/2000/30112-S07.16174-ARCOP
ACRONYM: ARCOP
TITLE: Arctic Operational Platform
PROJECT CO-ORDINATOR: Kvaerner Masa-Yards Inc. (KMY)

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The Norwegian College of Fishery Science	NO
Ministry of Trade and Industry	FIN

REPORTING PERIOD: From 01.12.2002
PROJECT START DATE: 01.12.2002
DURATION: 36 Months
DATE OF ISSUE OF THIS REPORT: 28.4.2005

DELIVERABLE SUMMARY SHEET

Short Description
This report is a literature survey of the effects of oil and oil spills on Arctic marine organisms and ecosystems. The published data of oil effects is based on experiments (laboratory studies, field experiments) and monitoring data during oil spills in different parts of the world's oceans. Particular emphasis is on information gathered from the Arctic seas and experiments made on arctic organisms. Effects of oil on all major systematic groups of organisms, ecological entities (habitats), and their interactions (food web structures) in the Arctic seas are described and discussed in this report.

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Document history					
Revision	Date	Company	Initials	Revised pages	Short description of changes
Gerhard Dieckmann	31.4.2005	AWI		4-41	none
Karl-Ulrich Evers	9.5.2005	HSVA		4-41	editorial

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Effects of oil spills on Arctic marine ecosystems

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1. Introduction

Oil in the Arctic marine ecosystem originates mainly from two sources: drilling activity and oil spills during transportation. Drilling activity causes long-term exposure and thus chronic effects on Arctic marine biota, such as changes in species composition, dominance and biomass, while oil spill effects are acute and can cause severe damage locally. In the scope of the ARCOP project this report concentrates on the latter type of contamination.

Oil in the marine environment affects organisms on all systematic levels: microscopic plankton (phyto- and zooplankton), invertebrates such as crustaceans, molluscs and benthic worms, and vertebrates (e.g. fish, birds, seals, polar bear). To suffer from the negative effects of oil, a given organisms does not need to become directly into contact with the medium: biomagnification is a process where oil is transported within the food web from a lower level to the next one.

Bioaccumulation of oil (concentration of oil into one organism) leads to more pronounced effects than when the organism is in contact with oil only for a short period of time. Petroleum effects on a given organism can vary from mere physical nuisance (such as oil clinging onto the body) to pathological. Oil can affect different organs and physiological functions, and thus lead to changes in e.g. behaviour (feeding, activity and motility, avoidance reactions etc.), growth, and reproduction. The developmental stage of an organism is often crucial; generally, larval and juvenile stages are more vulnerable than adult individuals.

In this report, examples of oil effects on life are given for all Arctic marine habitats: plankton (chapter 2.1.), littoral and benthic communities (chapter 2.2.), and vertebrates (fish, birds, otters, seals, whales and the polar bear; chapter 2.3.). Information is gathered through times mainly by performing experiments in the laboratory and on field (in particular in the 1970's and 1980's), and by sampling the biota during an oil spill.

2. Biological effects of oil, in particular PAHs

Oil consists of a wide variety of compounds that are toxic to organisms, the worst being the polynuclear aromatic hydrocarbons (PAHs). The effect of PAH compounds on a given

marine organism is dependent on numerous factors, which can be both abiotic and biotic (GESAMP 1993). In the arctic environment, important abiotic factors that affect in particular oil spreading and weathering processes include e.g. the presence or absence of ice and snow cover, water temperature, light conditions, and vertical and horizontal water currents (e.g. Mackay et al. 1975, Clark & Finley 1982, Mackay 1985 (and references therein), Payne et al. 1991, Sydnes 1991, Singsaas et al. 1994). Biological events in the Arctic marine ecosystem are strongly linked to specific seasons, thus the possible consequences of a relatively short-lived oil spill depend on the time of the year. Should an oil spill take place after the ice break-up in spring, it would affect the vernal bloom of ice algae and phytoplankton, and thus the rest of the pelagic food chain. Migratory birds would get disposed to oil, which in turn would be transported to coast and thus affect the littoral communities too.

Biochemical processes affecting PAHs in marine ecosystems, and bioavailability of PAH to aquatic organisms are discussed in depth in the works of McElroy et al. (1989) and GESAMP (1993), and are illustrated in Figure 1. Oil is transformed and transported within the marine ecosystem by e.g. metabolism, excretion, incorporation into fecal matter, microbial transformation and degradation, and bioturbation. Numerous laboratory studies have showed that aquatic organisms can accumulate PAH from the water column, from sediments, and from their diet, but that the bioavailability of PAH from these sources is not equivalent (Varanasi & Malins 1977, Neff 1979, McElroy et al. 1989, and references therein). Direct uptake of PAHs from the water column (dissolved PAHs) is the major pathway in bioaccumulation. Trophic transfer of hydrocarbons (biomagnification) is important in aquatic organisms, which have been shown to be capable of accumulating PAH via their diet (McElroy et al. 1989, and references therein). Thus, another important pathway is via food: in cases where uptake from food vs. sediments has been compared, dietary route appears to be more efficient (Corner et al. 1976, Varanasi & Malins 1977 and references therein, McElroy 1985). A good example of this is amphipods, which feed on Arctic phytoplankton in the underside of the ice, and can thus be exposed to toxic components of oil trapped under the ice. Amphipods also consume massive amounts of dead plant and animal material, and in turn serve as an important food source for Arctic cod and other fishes, several species of birds, seals and some whales. Busdosh & Atlas (1977) suggested that oil spills in the Arctic region are likely to cause large-scale local mortality of important amphipod species, resulting in serious ecological changes in detritus decomposing processes and food-web relationships.
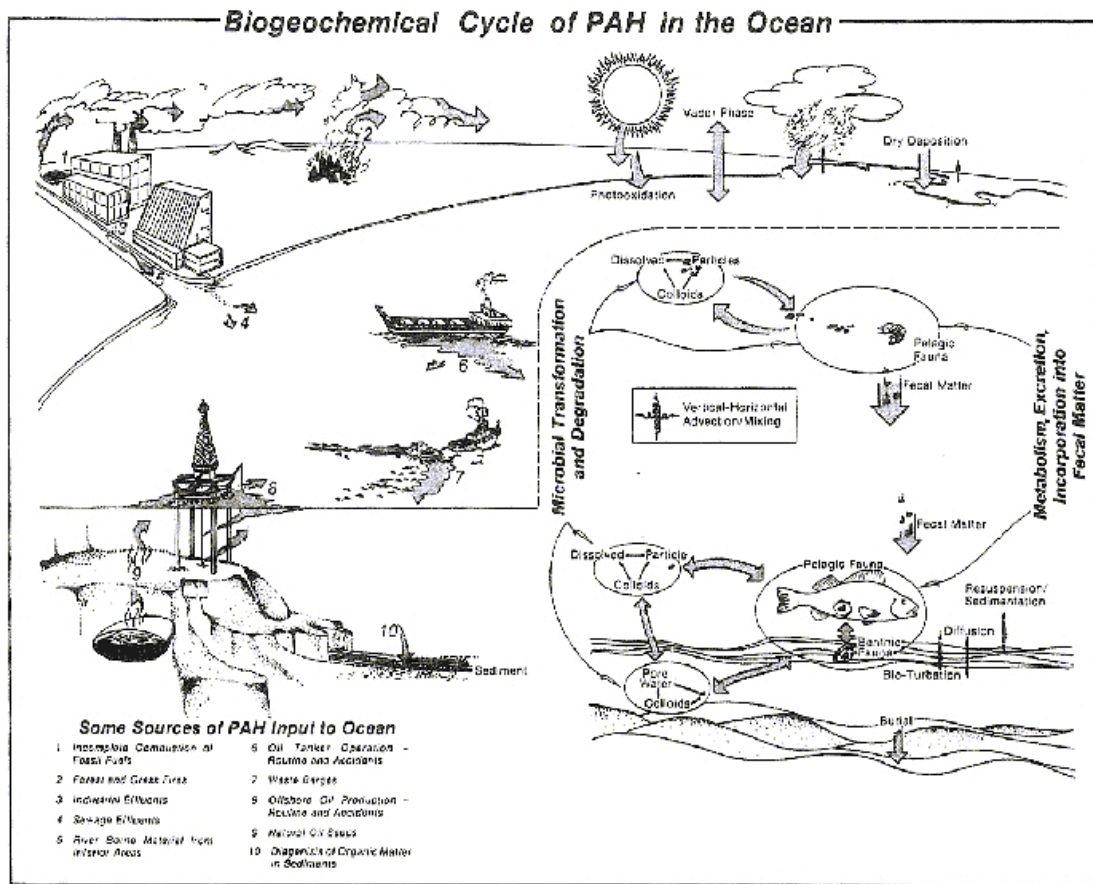
Figure 1. Biochemical processes affecting PAH in marine ecosystems. Source: McElroy et al. (1989).

Bioaccumulation of oil (accumulation of oil in a single specimen) is positively correlated with physical/chemical properties of the PAHs, such as the molecular weight, and octanol/water partition coefficients (McElroy et al. 1989). By knowing the physical/chemical properties of spilled oil, the degree at which the organism would bioaccumulate PAHs can be predicted. The bioaccumulation factor has been studied mainly for fish and tends to increase with increasing molecular weight (McElroy et al. 1989).

In toxicity tests, LD_{50} concentrations of the tested oil have yielded much information about the tolerance of a given organism to oil and/or dispersant disposition. Typically, oil with a high aromatic content (2- and 3-ringed PAHs, such as naphtalenes, fluorenes, phenantrenes and anthracenes) and fresh crude oil have a greater acute toxicity to marine organisms than oils with more aliphatic nature or weathered oil (e.g. Anderson et al. 1977, Lee et. al 1978). High angular configurations are more carcinogenic than linear or highly condensed ring arrangements (Neff 1979).

Petroleum and its products may have either a mechanical effect, as it is able to penetrate, has a tendency to cling to surfaces and form coverings on objects, or a chemical one,

where toxic components (mainly aromatic hydrocarbons of low molecular weight) affect the organism (e.g. Nelson- Smith 1982, Wells and Percy 1985, Robertson 1998). Organisms can become affected by oil by filtration or ingestion, penetration of oil through cell membranes, or through becoming smothered by oil. The sensitivity of a given organism to oil is largely dependent on the severity of contamination, the organism's physiological state, and life cycle. Reproductive and juvenile stages are particularly sensitive to PAH effects. Also, organisms on a higher systematic level generally have a better PAH metabolism, and consequently tolerance, than organisms at lower levels (e.g. Rice et al. 1977a).

In addition to acute toxicity - virtually causing death of an organism, which is tested using LD_{50} concentrations of a given compound - PAHs can also cause a multitude of sublethal effects. These may become visible almost instantly after exposure to oil or during later life cycle stages of the organisms (e.g. during maturation or reproduction). Sublethal effects can be expressed in various ways, such as changes in feeding or other behaviour, growth, reproduction capacity, or in the organism's offspring (such as organ abnormalities in the developing embryo). On a cellular level, PAHs can either bind to lipophilic sites in the cell, or affect the DNA creating covalently bound products called adducts (Robertson 1998). Adducts are thought to be one of the initial steps in tumor development caused by carcinogenic PAHs. Metabolic activation is a prerequisite for carcinogenic effects of PAHs in mammals and fish (Robertson 1998) and is introduced later in this report.

In the following chapters, petroleum effects on marine organisms, with particular emphasis on oil spills and arctic species are presented with examples from literature. The report proceeds from plankton (open water) communities to littoral (shorelines), benthos (sea bottom) and, finally, marine mammals.

2.1. Plankton

Plankton organisms live in the open water, and comprise of bacteria, fungi and viruses, primary producers (photosynthesising microscopic algae), heterotrophic consumers (e.g. ciliates, rotifers, fish eggs, medusae) and mixotrophic organisms (from several systematic ranks) (e.g. Valiela 1984). Arctic zooplankton consumes phytoplankton and ice algae for food (Runge et al. 1990, Werner 1997). Plankton in general is the foundation of the marine open water food web, and is the primary food source for several macroscopic open water organisms, such as fish and whales. Fisheries, for example is very sensitive to changes in plankton quantity and quality.

In many cases, the comparison of results from oil experiments is difficult, questionable or impossible (for discussion on this issue see e.g. Craddock 1977). Particularly older literature with oil experiments shows that tests were made in various ways, and common guidelines for testing oil effects on plankton were generally missing. This is true not only for plankton, but in commonl for littoral and benthic studies as well. OECD guidelines for e.g. testing of chemicals have later been prepared for some aquatic organisms, i.e. alga (growth inhibition test), zooplankton (*Daphnia magna* reproduction test) and several for fish (e.g. acute and prolonged toxicity tests, and juvenile growth test) (OECD 1984a, b, 1992, 1998, 2000). The plankton data available derive from three sources: field studies in

oil-contaminated areas, studies with enclosed systems - such as mesocosms – with test organisms, and laboratory experiments.

Information about the effects of oil on plankton is much sparser than for littoral or benthic organisms. One reason for this is that oil is regarded to affect sea shores and bottoms more than the open water ecosystem (e.g. Robertson 1998). However, as plankton is the food source for a large variety of organisms even on the top level of the marine food chain, it is vital to know petroleum effects on open water communities. Plankton is generally thought to remain rather unaffected by oil, mainly for its capability to escape the contaminated water area. This is partially true for in particular larger zooplankton and in case of a small scale spill. Also, phytoplankton cells reproduce mainly asexually by cell division, therefore enabling a short recovery time. Local and short-term effects on plankton communities (decreased photosynthetic rates, oiled zooplankton) are likely during even a small oil spill, but have only a temporary dampening effect on plankton productivity. In case of a larger spill, the effects last notably longer and are spread to a larger area (e.g. Teal & Howarth 1984). An oil slick in the water column or associated with sea ice has an effect on the organisms therein, and thus the type and composition of algal material (phytoplankton vs. sea ice algae) sinking to the bottom may affect the food quality of benthic animals (Clough 2005).

Hydrocarbon utilising bacteria (such as *Vibrio*, *Pseudomonas*, *Micrococcus*, *Nocardia* and *Acinetobacter* in the Arctic) are long known to be widely distributed in the world's oceans, also in cold marine ecosystems (e.g. Bunch & Harland 1976, Austin et al. 1977, Atlas 1978, Roubal & Atlas 1978, Atlas et al. 1982). Among fungi, *Penicillium* and *Verticillium* spp. from the northern Canada oil-producing areas were capable of growth on one or more crude oils tested by Davies & Westlake (1979), and oil-degrading strains of a few other genera were isolated. Several genera such as the truly marine *Corollospora*, *Dendryphiella*, *Lulworthia* and *Varicosporina* are known (Kirk & Gordon 1984), but their function as hydrocarbon utilisers is still an issue in the Arctic. Microbial degradation of oil by bacteria (bioremediation) and fungi is dependent on several physical, chemical, and biological factors which are extensively studied and discussed by e.g. Leahy & Colwell (1990). A dramatic rise in concentration of planktonic hydrocarbon utilising bacteria in response to acute input of petroleum hydrocarbons has been documented in some cases (Horowitz & Atlas 1978, Johansson 1980, Dahl et al. 1983), but not as a rule. The biomass increase of hydrocarbon degraders can also be extremely slow and take place within months or even years after oil introduction (Haines & Atlas 1982). As for the diversity of bacteria, it may either decrease or increase due to oil exposure (Atlas et al. 1982, Atlas 1983). In sediments, the biomass of oil degraders may be notably higher than in the water column, like in the case of the *Arrow* oil spill in the Chedabucto Bay, Nova Scotia in 1970 (Stewart & Marks 1978). Generally, hydrocarbon utilisers in sediments seem to increase relatively slowly after the actual spill (e.g. Eimhjellen et al. 1982, Bunch 1987). Hydrocarbons can also cause a temporary change in the behaviour (chemotaxis) of bacteria as evidenced by Mitchell et al. (1972) and Walsh & Mitchell (1973).

Good evidence of effects on plankton derives from field studies of the *Tsesis* spill in the Baltic Sea in 1977 (Johansson 1980). Phytoplankton species composition was not changed by the oil (concentration 50-60 µg/l after 2-5 days of the accident): microflagellates dominated the community before and after the spill. Phytoplankton biomass and productivity increased after the accident, which was probably largely due to

depressed zooplankton grazing. Although planktonic bacteria biomass increased notably after the oil spill, the populations of their grazers (ciliates, rotifers) did not (Johansson et al. 1980). Larger zooplankton abundance changed shortly after the accident, probably due to narcosis effects and/or avoidance reactions. No notable change in phytoplankton communities were detected after the platform *Bravo* spill either (concentration of aromatic hydrocarbons up to 8 µg/l) in the North Sea in 1977 (Rey et al. 1977), while an obvious retardation of phytoplankton growth, and a considerable mortality of zooplankton was recorded for several weeks in the vicinity of the *Amoco Cadiz* accident on the NW coast of France in 1978 (Cabioch et al. 1981). Further away, phytoplankton growth was elevated, which was interpreted as a consequence of nutrient release from dead organisms. Zooplankton was contaminated with oil in proportion to their distance from the wreck. Also the groundings of *Arrow* in Nova Scotia in 1970, and *Argo Merchant* on Nantucket shoals in 1976 lead to copepod contamination (Conover 1971, Polak et al 1978): oil droplets had incorporated into their guts. The *Potomac* spill off Western Greenland in 1977 caused external contamination of plankton (primarily the copepod *Calanus hyperboreus* and the amphipod *Themisto libellilula* (syn. *Parathemisto libelillula*) in the vicinity of the spill, but no oil was found in either copepod or amphipod guts (Maurer & Kane 1978), and thus no severe damage to the zooplankton communities was detected. Thus, field observations at numerous accidental spills show that negative biological effects can occur after a spill, but that the consequences appear rather mild and short-lived.

More detailed information of the effects of oil on plankton organisms originates from laboratory and other experimental studies. The older literature is reviewed extensively by O'Brien & Dixon (1976) and Johnson (1977). Microscopic algae from different systematic ranks may show different responses and tolerance to oil. Pulich et al. (1974) used six phytoplankton species to study experimentally the effects of different crude oils on the growth and photosynthetic rate of microscopic algae. All were inhibited either fully or partially, but significant differences between algal groups were discovered: *Thalassiosira pseudonana* (a diatom) showed the least tolerance to oil, while blue-green algae (*Agmenellum quadlupcanum*, *Nostoc* sp.), green algae (*Dunaliella tertiolecta, Chlorella autotrophica*) and the dinoflagellate *Gymnodinium halli* were several times more tolerant to one or all types of oils tested. Hsiao (1978) found that exposure to various crude oils generally inhibited arctic phytoplankton growth, but also some signs of stimulated growth were documented after several days of exposure to 10 ppm of a range of crude oils. The degree of inhibition was temperature dependent: at +15 °C the oils were generally less toxic than at 0 or +10 °C. Temperature dependence of oil effects is important in particular in the case of an oil spill accident in the Arctic. Hsiao (1978) further speculated that a major oil spill could cause a change in phytoplankton species composition (from diatoms to microflagellates) and therefore an alteration in the zooplankton communities (species, biomass) that feed on microalgae. Effects of oil on plankton algae may vary even between clones of an algal species as was evidenced for the diatom *Skeletonema costatum* by Mahoney & Haskin (1980) (an important food source for the eastern oyster *Crassostrea virginica*). In the same experiment, other algae (chrysophytes *Monochrysis lutheri* and *Isochrysis galbana*, chlorophyte *Dunaliella euchlora*, and the eustigmatophyte *Nannochloris oculata*) showed generally better tolerance to oil than the diatom. Evidence of differences in oil effects on different phytoplankton groups is published also by Davenport (1982) and Dahl and co-workers (1983).

Sea ice associated (epontic) communities are characteristic to polar regions. They consist of a wide variety of microscopic organisms (size ranges from picoplankton (0.2-2 µm to mesoplankton (20-200 mm) (e.g. Ikävalko 1997, Werner 1997, Thomas 2004). Primary producers, grazers, predators and, finally, degraders are present within sea ice. Sea ice communities consist largely of the same groups of organisms (but often different species) as the plankton, and live trapped in brine channels within ice. Thus they are not capable of escaping oil contamination, and the effects of oil on ice biota can be much stronger than on free-floating plankton in the open water. Ikävalko and co-workers (2005) made experiments on the effects of Statfjord crude oil on sea ice biota by exposing ice algae to oil for 63 days. Oil was practically lethal for dinoflagellates and chlorophytes, while some diatoms survived by forming thick-walled resting stages which are relatively resistant to environmental changes. Vegetative diatom cells survived better in the interior and close to the underside of the ice than in the ice surface where oil was distributed. No negative effects of oil on ice diatom growth or photosynthesis were detected in in-situ experiments by Cross (1987). Conflicting results from experiments on sea ice algae may be explained by several factors, such as differences in study methods (oil type and concentration, duration of predisposition to oil, laboratory vs. field experiments), microalgal species studied, and possibly the physiological state of algae (not measured).

Studies on zooplankton responses to oil are reviewed e.g. by Wells & Percy (1985) and Robertson (1998). Most zooplankters appear to be very sensitive to in particular dispersed and dissolved oil. The acute lethal toxicity of dispersions and water soluble fraction (WSF), usually expressed as 4-day LD_{50} values using initial measured concentrations, ranges between 0.05-9.4 mg/litre (Wells & Percy 1985). The major routes of contamination are direct uptake from the water, uptake from food (important for in particular copepods), or ingestion of oil particles that may be the size of the food item Wells & Percy 1985). The capability of detoxifying hydrocarbons varies between different organisms. Oil particles taken up alone, or with food items, seem to pass chemically unchanged through the gut of for example copepods and pelagic barnacle larvae, and may become discharged in fecal pellets. This, in turn, may lead to biomagnification of oil in the arctic foodweb: the transfer of ingested oil to higher predators or coprophages (organisms that eat feces).

Low concentrations of hydrocarbons can cause sublethal effects in zooplankton, such as changes in behaviour, physiology, development, growth and reproduction (Wells & Percy 1985). Further studies are however, required as in earlier experiments there has been much variation in e.g. exposure conditions, life stages of test animals, and oil types used. Field observations are made during oil spills and in chronically exposed areas like in the vicinity of oil platforms. Biological effects seem to be detectable but short-lived. Organisms at spills have suffered from direct mortality (copepods, fish eggs, plankton in general), external oil contamination (crustaceans, fish eggs), tissue contamination by aromatic compounds, abnormal development of fish embryos, altered feeding behaviour in copepods, and changes in metabolic rates of zooplankton. Mesocosm experiments by Vargo (1981) showed several negative effects of chronic low concentrations of fuel oil on temperate zooplankton, in particular changes in respiration and excretion rates.

Wide distribution of zooplankters and rapid change of water masses in the open waters promote the recovery of zooplankton communities after oil contamination, while in enclosed water bodies, such as estuaries and bays, the recovery may take notably longer (Wells and Percy 1985, and references therein).

Dahl et al. (1983) studied the effects of Ekofisk crude oil on a planktonic ecosystem using a simplified mesocosm set-up. While diatoms and copepods suffered from the addition of oil, the rapid stimulation in growth of planktonic bacteria (for which low molecular-weight fractions served as energy sources) was observed. Due to increased food availability (i.e. bacteria) their grazers (mainly heterotrophic choanoflagellates and tintinnid ciliates) increased as well. Very low concentrations of petroleum hydrocarbons (470µg/l) were considered to be the toxic to diatoms. Clear range of oil sensitivity of arctic freshwater zooplankton was detected in experimental studies by O'Brien (1978) and Atlas et al. (1978). As branchiopods (fairy shrimp *Branchionecta paludosa*) and amphipods seem particularly sensitive, cladoceran *Daphnia middendorfiana*, the calanoid crustacean *Heterocope septentrionalis* and isopods in general showed better tolerance to oil.

Copepods are good test organisms in oil experiments: they are easy and inexpensive to access and maintain, and show rapid responses to treatments. Petroleum is acutely toxic, but has also numerous sublethal effects (narcosis, paralysis, decreased feeding and defecation rates, disrupted phototaxis and altered swimming activity) on copepods. The effects of petroleum hydrocarbons on copepods have thus been extensively studied in laboratory set-ups, outdoor enclosure experiments, and at various spilled sites, of which only a few examples are given here (for a careful review of older literature see Wells & Percy 1985). Cross & Martin (1987) examined effects of untreated, solidified and dispersed oil on under-ice meiofauna during the Baffin Island Oil Spill (BIOS) project (for a description of the project, please see Sergy & Blackall 1987). Harpacticoid copepods and polychaete worms showed high sensitivity to oil, in particular the dispersed type, while cyclopoid and calanoid copepod nauplii (juvenile stages) were more tolerant to it. Untreated and solidified oil did not affect nematode, polychaete and copepod densities. The growth of adult harpacticoid copepods and their copepodite (juvenile) stages, and cyclopoid nauplii was, in fact, slightly stimulated by untreated and solidified oil (Cross & Martin 1987). Negative effects of oil on arctic copepods have been evidenced by several researchers. Melbye et al. (2001) studied the effect of low oil concentration on the copepod *Calanus finmarchicus*, which is an important species in arctic pelagic food web. Oil with very low water-soluble component (and low content of aromatics) had very weak acute toxicity towards the test organism. Furthermore, another arctic copepod, *Calanus hyperboreus* is considered very resistant to crude oil (Percy & Mullin 1975, Foy 1979) when compared to e.g. amphipods and isopods. Short-term exposures of high concentrations of aromatic hydrocarbons did have profound effects of the copepod *Eurytemora affinis* as documented by Berdugo et al. (1977): significant reduction in subsequent length of life, total number of eggs produced, mean brood size, and the rate of egg production was evident.

The sensitivity of planktonic cod eggs (*Gadus morrhua*), and sea urchin eggs and embryos (*Strongylocentrotus droebachiensis*) to naphatalenes and Ekofisk crude oil was documented by Falk-Petersen et al. (1982, 1983). The locomotory motion and thus swimming of the Arctic medusa *Halitholus cirratus* is negatively affected by crude oil disposition (Percy & Mullin 1975).

2.2. Littoral and benthic communities

The marine benthic habitat can be divided into two areas, the intertidal (here also referred to as the littoral zone) and the sea floor (benthos). Characteristic for the intertidal zone are strong variations in water level, notable stress caused by wave action and, in the Arctic, ice scouring and summertime reduction in surface salinity due to melting of ice and snow. Thus, the marine organisms in the intertidal must be tolerant to e.g. exposition to air and direct sun (desiccation), and fresh/brackish water. The High Arctic intertidal is regarded inhospitable to colonisation (Menzies et al. 1973). In the Eastern Canadian Arctic it is typically colonised by the rough periwinkle *Littorina saxatilis* and the barnacle *Balanus balanoides* (Ellis 1955, Ellis & Wilce 1961). Below the intertidal zone is the benthos, which light never reaches. The upper part is the so called barren zone, that typically extends to a depth 3-5 metres, sometimes even 15 metres), which, due to ice scavenging and low surface water salinities, is devoid of infauna (animals partly or completely buried into the substrate) and sessile epifauna (animals attached onto substrates) (Ellis 1960). In the subarctic, ice scouring effect is infrequent and the intertidal communities become notably more diverse. High intertidal zone is inhabited by a variety on invertebrates and smaller macroalgae, such as green and coralline (red) algae. Where ample light for photosynthesis, macroalgae such as kelps (giant brown algae) flourish in deeper intertidal, and offer habitats for diverse invertebrate communities – amphipods, barnacles, mussels, echinoderms, nematodes et cetera (George 1977, Wells & Percy 1985). Much of the polar basin lies beneath the barren zone and thus the permanent ice cover. These zones are called the shelf zone, slope zone and the abyssal (deep sea) (Figure 2. (from Wells and Percy 1985)). The vulnerability of littoral and benthic communities to spilled oil varies due to the environmental factors introduced above, and thus the vertical and horizontal distribution of biological communities (Figure 2.). Oligomixity (high population densities of a single species) is characteristic to the Arctic Ocean in general and particularly true for the benthic communities (George 1977). The most abundant and diverse groups in the arctic benthos include bivalve molluscs, polychaete worms, amphipods and isopods (Marshall 1982). Benthic populations in the Arctic tend to show less fluctuation in abundance than those of warmer seas (Ellis 1960). This is partially due to slower growth and longer life span, but also their altered reproduction strategies: many benthic species have shortened or eliminated the vulnerable pelagic larval stage, and larvae are produced in brood chambers.
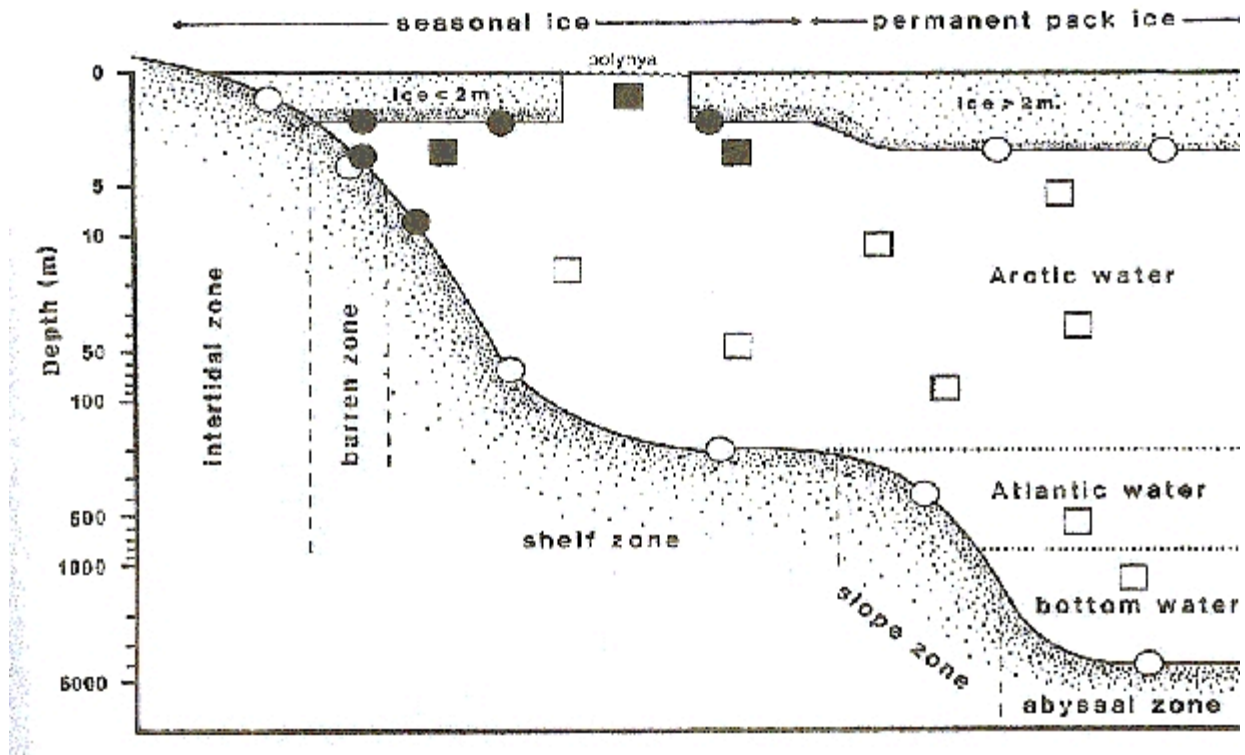
Figure 2. Habitats of Arctic marine invertebrates and their vulnerability to spilled oils (filled circles = possible impacts on benthos, filled squares = possible impacts on zooplankton, open circles = no anticipated impacts on benthos, open squares = no anticipated impacts on zooplankton. Source: Wells & Percy (1985)

Shoreline and shallow subtidal communities are most affected during a coastal oil spill, and oil impacts on sedimentary shorelines have been reported from several accidental and experimental spills at lower latitudes. Also laboratory experiments have been made on various littoral and benthic organisms, and some of them are introduced as examples in this report. In nature, the impacts of an oil spill are dependent largely on the amount of dispersed oil and the type of substratum on the shoreline (e.g. Robertson 1998). On rocky coasts, wave action may remove the oil rather quickly and transport it to the open sea or the benthos, while in sheltered estuaries and on muddy shores the oil effects in the littoral zone are more pronounced. Oil can reach the bottom by various mechanisms: 1) direct mixing of oil with sediments by wave action in shallow water, and consequently transport to deeper water by density currents, 2) sorption onto particulate matter suspended in water column and subsequent sinking, 3) uptake by zooplankton, release in and subsequent sinking of pellets, and 4) take-up of non-volatile aromatic hydrocarbons by phytoplankton, and further sedimentation (Conover 1971, Mackie et al. 1978, Sanders et al. 1980, Teal & Howarth 1984).

Attached algae (seaweeds) and their responses to oil have not been given much attention in the Arctic, although vegetations of macroalgae (in particular green and brown algae) live attached to rocky shores. Production estimates for arctic kelp *Laminaria solidungula* communities, for example, vary between 7-20 gC/m^2/year (Dunton et al. 1982, Chapman & Lindley 1980, 1981). The rate of photosynthesis by marine phytoplankton and macroalgae may be stimulated or repressed depending upon the concentration of hydrocarbons and method of exposure (Johnson 1977, and references therein). Oil can form coatings on algae, and thus decrease CO_2 uptake and water loss. This has been documented for *Laminaria digitata* and *Fucus vesiculosus* (Phaeophyceae, brown algae), *Porphyra umbilicalis* (Rhodophyceae, red alga) and *Enteromorpha* sp. (Chlorophyceae, green alga) (Schramm 1972). Intertidal macroalgae are considered relatively resistant to oil due to mucus production, as evidenced e.g. for the giant kelp (*Macrocystis* sp.). Mucus prevents contact between the petroleum and kelp tissue (Mitchell et al. 1970). During the BIOS project macroalgae were exposed to untreated and dispersed oil (Cross et al. 1987a). Biomass, number of species and reproductive condition of the most dominant algae did not seem to be adversely affected by neither type of oil. The lack of major effects was explained partially by the mortality of herbivores and thus decreased grazing pressure on macroalgae, and the vegetative mode of reproduction, which is not as sensitive to environmental disturbances as sexual reproduction (Cross et al. 1987a). A heavy oil pollution can cause retarded growth and even death of seaweeds as was evidenced after the *Torrey Canyon* accident (Ranwell 1968) and a spill of 1000 tons of bunker oil on the Arctic coast of Norway (Wikander 1982). Even in the latter case, in lightly polluted areas the new sprouts of contaminated algae looked healthy the following spring (Wikander 1982). In mesocosm experiments, oil caused growth reduction in macroalgae *Ascophyllum nodosum* and *Laminaria digitata* (Gray 1987). Hydrocarbons can also affect reproduction of some fucoid macroalgae (*Fucus serratus*, *F. vesiculosus, F. edentatus*) by hindering gamete release, fertilisation, or by altering the behaviour of motile gametes as PAHs produce similar attraction to *Fucus* sperms as do the egg cells (Cook & Elvidge 1951, Steele 1977).

Sublethal biological effects of PAHs on benthic invertebrates in general are numerous, and include changes in behaviour, physiology, growth and reproduction of coelenterates (corals, sea anemones, hydrozoa, medusae), annelid worms, adult and juvenile arthropods (marine crustaceans such as mysids, and amphipods, isopods, shrimps, prawns, crabs), molluscs (gastropods such as limpets, and bivalves such as oysters, mussels and clams), and echinoderms (e.g. starfish, asteroids, sea urchins) (Straughan 1976, Johnson 1977, Wells & Percy 1985). Generally, long-term effects of petroleum include the development of tumours, neoplasms, diseases caused by bacteria and viruses, and mycoses in invertebrates (Hodgins et al. 1977, Wells & Percy 1985 and references in both works). Benthic invertebrates may also suffer from ostial closure, loss of responsiveness to mechanical stimuli and narcosis (Mageau et al. 1987) which are discussed later in this chapter. Chronic oil predisposition and long-term responses of benthic fauna to hydrocarbons has been monitored in the vicinity of oil platforms by e.g. Menzie 1982, Addy (1987), Kingston (1987), Moore and co-workers (1987a), Gray et al. (1990) and Olsgard & Gray (1995). To summarize, the responses of benthic communities to offshore oil exploration are measurable, complex, often irreversible, and results are presented mainly on the community level (species dominance, competition, and succession). Regarding the scope of the ARCOP project and in particular WP 4 (oil

transportation and effects of oil spills on marine biota), oil field monitoring studies will not be discussed here further.

The two classes of marine molluscs which have been studied extensively with respect to petroleum pollution, Gastropoda and Bivalvia, consist of animals with distinctly contrasting adult habits. Most gastropods (e.g. limpets, periwinkles and snails) are free-living epibenthic animals, while most bivalves (mussels, oysters and cockles) are either sessile or sedentary burrowing forms (Fish & Fish 1989).

The literature lists a multitude of oil effects on bivalves and gastropods. They include e.g. mortality, abnormal larval development, and various molecular, biochemical, cellular and physiological responses such as enzymatic hydrocarbon detoxification or elimination, changes in enzymatic activity in general, atrophy of the epithelium, changes in oxygen consumption, feeding, excretion, growth and, finally, ecophysiological consequences like the general reduction in population fitness leading to changes on an ecosystem level, i.e. affecting biological interactions (e.g. Haranghy 1956, Bayne et al. 1982, Lewis 1982, Southward 1982, Stickle et al. 1985, Gray 1987, Moore et al. 1987b, Neff et al. 1987). During the BIOS project Neff and co-workers (1987) reported several histopathological changes in bivalves *Mya arenaria* and *Macoma calcarea* caused by untreated and/or dispersed oil; digestive track necrosis, increase in the number of mucus cells in the digestive track epithelium, granulocytomas, invasive neoplasia (probably cancer), vacuolisation of the digestive tubule epithelium, increased parasitism, and hemocytic infiltration The stress level (indicated by concentrations of glucose, glycogen, trehalose, total lipid and free amino acids) was lower in animals exposed to dispersed oil than for animals receiving oil alone (Neff et al. 1987).

Documented sublethal effects of oil in gastropods only range from simple narcotisation to loss of chemosensitivity, and have been reviewed earlier in depth by (Johnson 1977, and references therein). Narcosis caused symptom is for example the detachment from substrate. The loss of chemosensitivity may change motility and alter the direction of movement, and thus affect food capture. Reduced filtration rates, possibly due to direct inhibition of the cilia by hydrocarbons (Johnson 1977), affect several functions in bivalves and gastropods, including reduced feeding. This has been documented for e.g. the oyster, *Crassostrea virginica* (Stegeman & Teal 1973), blue mussel *Mytilus edulis* (Phelps et al. 1981, Widdows et al. 1982), the hard clam *Mercenaria mercenaria* (Keck et al. 1978), and an arctic bivalve, *Yoldiella arctica* (Percy & Mullin 1975).

High concentrations of oil can cause shell closure and narcotisation of ciliary surfaces in bivalves, and consequently affect respiration and feeding rates negatively (Johnson 1977, Bayne et al. 1982 and references therein, Mageau et al. 1987). At low concentrations of oil, rates of oxygen consumption are first increased in bivalves, such as the soft-shelled clam (*Mya arenaria*), blue mussel (*Mytilus edulis*), Baltic telling (*Macoma balthica*) and the gastropod *Littorina littorea* (edible periwinkle) (Bayne et al. 1982 and references therein). Metabolic rates increase due to hydrocarbon association in the body tissues, and mucus secretion and excretion increase. As a result, energy expenditure increases while less energy (reduced carbon flux) is available for growth and reproduction (Stainken 1978, Widdows et al. 1982, Bayne et al.1982, and references therein). Other effects are manifested in the structure and development of the eggs and embryos, like anomalies in the gonads of the Baltic telling *Macoma balthica* (Stekoll et al. 1980). The BIOS project

showed different uptake dynamics among species (filter-feeding bivalves *Mya truncata* and *Serripes groenlandicus*, and deposit-feeding green sea urchin *Strongylocentrotus droebachiensis*), but in all cases the effect was immediate, short-lived and resulted in temporary accumulation of hydrocarbons (Mageau et al. 1987). As a consequence of oil exposure, the bivalves suffered from ostial closure, retraction of the siphon (decreased filtration rate and growth), the loss of responsiveness to mechanical stimuli, narcosis, increased enzymatic activities and accumulation of hydrocarbons in tissues.

Low levels of petroleum hydrocarbons can also affect the behaviour of molluscs. The production of the bussys by juvenile and adult mussels may be reduced, leading to weakened attachment on substrate. In mesocosm experiments, oil affected negatively the recruitment of the edible periwinkle *Littorina littorea*, a common inhabitant on rocky coasts in subarctic, and consequently populations declined over time (Gray 1987). The burrowing behaviour of infaunal bivalves such as *Macoma balthica* may be impaired, and clams can be stimulated to leave the contaminated area (Lindén 1977, Taylor & Karinen 1977, Stekoll et al. 1980). The crawling rates of gastropods *L. littorea* and *Theodoxus fluviatilis* may change (Hargrave & Newcombe 1973, Lindén 1977). During the BIOS project Cross &Thomson (1987) noted that the use of untreated and dispersed oil had very distinct effects on macrobenthic infauna (bivalves *Mya truncata*, *Macoma calcarea*, *Serripes groenlandicus*, *Astarte borealis*). While untreated oil had practically no effect, dispersed oil caused marked acute effects on infauna, including emergence from the substrate, narcosis and progressive decrease in condition. Neither type of oil release caused any large scale mortality of benthic infauna, neither were significant changes in community structure detected (Cross & Thomson 1987).

Annelids, such as bristle worms (Polychaeta), are common on the shore under stones and rocks, and buried in mud and sand. While the adult worms seem rather resistant to oil pollution (Johnson 1977, and references therein), hydrocarbons can cause narcosis, immobilisation and death of their larvae (Chia 1973, Carr & Reish 1977). Once in sediments, hydrocarbons are taken up by benthic organisms with greater uptake of the heavier relative to the lighter molecular weight aromatic compounds. Uptake from water may occur more readily than from sediments in carnivores and filter feeders, while deposit feeders such as polychaetes with more intimate contact with porewaters could be expected to show a more rapid uptake from substrate (Anderson et al. 1978). Exposure of the polychaete *Nereis succinea* to oil in a laboratory experiment resulted in a decrease in growth rate and an increase in mixed function oxygenase (MFO) activity relative to unexposed individuals (Lee et al. 1981). MFO is responsible for the metabolic modification of foreign organic compounds in verteberates, such as fish and cetaceans and has been detected also in marine crabs and polychaetes (e.g. Johnson 1977, and references therein, Lee et al. 1981, Rice 1985, Lockhart & Metner 1991, George et al. 1995). Studies on annelids and hydrocarbons are few. Some monitoring on the succession of polychaetes has been made during oil spills. Several years after the *Amoco Cadiz* and *Arrow* spills, the lugworm *Arenicola marina* was very common, in case of *Arrow* even more abundant than prior to the accident (Gordon et al. 1978, Gundlach et al. 1981). Hydrozoa, corals, and anemones are typical benthic organisms in lower latitudes, where hydrocarbons are known to affect the behaviour, growth and reproduction of these sessile animals (Johnson 1977, and references therein).

Echinoderms include intertidal and benthic, slowly moving invertebrates such as sea urchins, brittle-stars and starfish. Fuel oil and gasoline interfere with the development of boreoarctic sea urchins (Falk-Petersen 1979). In sea urchins oil may stimulate the oxygen consumption, weaken their adherence to the substrate, cause retarded fertilisation, and interfere with the development of the embryo (Johnson 1977, and references therein). A range of behavioural changes in the green sea urchin *Strongylocentrotus droebachiensis* and the starfish *Leptasterias polaris* was recorded during the BIOS project caused by a short-term exposure to dispersed oil by Cross and co-workers (1987b) and Mageau et co-workers (1987); unnatural postures (animals upside down), narcosis (i.e. loss of responsiveness to mechanical stimuli), and for sea urchins impairment of the tube foot (resulting in substrate detachment), changes in spine attitude, and frequent shedding of gametes was evident. Furthermore, in starfish the chemoreception system may become partially inhibited, thus affecting food particle sensing and capture (predation) (Johnson 1977, and references therein).

The effects of oil on littoral crustaceans are extensively studied, and earlier literature is carefully reviewed by e.g. Johnson (1977) and Wells & Percy (1985). Evidence shows that these organisms may possess of a variety of responses to hydrocarbons. Physiological responses include changes in e.g. respiration rate (increases and decreases), hormone production (thus reproduction), molting and hatching, development of larvae, chemoreception-mediated behaviour (affecting reproduction and feeding) and disruption of osmoregulation. Behavioural changes are mainly due to narcosis, which in turn affect locomotor activity (and thus feeding and escape reactions), burrowing behaviour, and reception of chemical signals (food particle capture), even shell evacuation. Also physical nuisance is caused: when oil is adhered on the locomotory parts of the crustacean, swimming and feeding is affected. Also, oily coatings on hard substrates hinder the settlement of e.g. pelagic motile barnacle larvae (Straughan 1971). Ingestion of oil droplets tends to decrease feeding rates (Blackman 1972). The arctic marine amphipod *Gammarus oceanicus*, for example, has showed several negative physiological responses to crude oil (Aunaas et al. 1990). Water soluble fractions of crude oil increased respiration, sodium in haemolymph, and thus water content of the organism. Water emulsions reduced respiratory rates, causing oxygen deficiency due to oil droplet adherence to gill membranes. The use of dispersants reduced the mortality of the amphipods. Low concentrations of crude oil and oil fractions significantly depressed respiration rates of arctic amphipods *Boeckosimus affinis* and *Anonyx nugax*, while with high concentrations the depression was reversed (Percy 1977, Busdosh 1978). Baden & Hagerman (1981) and Baden (1982 a, b) exposed the shrimp *Palaemon adspersus* to water soluble fraction of North Sea crude oil. As a result, the ventilatory behaviour was disturbed, osmoregulation impaired, and a significant increase in oxygen consumption was detected. A slight stimulation of metabolism at low hydrocarbon concentration followed by a decrease with increasing concentrations was detected for the littoral mysid *Mysis litoralis* (Wells & Percy 1985). In a behavioural study, arctic amphipods *G. oceanicus* and *Onisimus affinis* did not show avoidance of contaminated area in the presence of even high concentrations of crude oil (Percy & Mullin 1975, Percy 1976). Petroleum hydrocarbons interfere also with reproductive processes (e.g. reduced precopulation frequency andnumber of larvae, premature shedding of eggs) of crustaceans such as *Gammarus oceanicus* and *Boeckosimus affinis* (Lindén 1976, Busdosh 1978). Differences in response rates to various oil-water mixtures were documented also by Riebell & Percy (1989), by exposing the arctic littoral mysid, *Mysis oculata*, to oil-in-water dispersions and

water-soluble fractions of crude oil. The latter fraction was significantly more toxic, and the species was regarded as exceptionally sensitive to crude oil. Opposed to sessile and slowly moving animals in the littoral and benthos motile invertebrates, like littoral amphipods, are generally capable of avoiding oil slick by escaping it (Bonsdorff & Nelson 1981, Gulliksen & Taasen 1982).

Massive kills and long-term effects can occur when oil reaches the littoral and the benthos in sufficient quantity, as has been evidenced for several oil spills in the past. Within 12 hrs after the *Florida* spill in West Falmouth, Massachusetts in 1969, the macrobenthos was nearly eradicated at most heavily oiled sites (Sanders et al. 1980). Opportunistic species typically play a vital role in the initial recolonisation of an eradicated area (Teal & Howarth 1984). After the Florida spill the annelid worm *Capitella* sp.and the nematode *Mediomastis* sp. increased greatly in abundance, monopolising the otherwise defaunated sediments for months after the accident. The *Arrow* spill in Nova Scotia in 1970 caused the decline of the bladder wrack, *Fucus vesiculosus* for five years, while the spiral wrack, *Fucus spiralis*, disappeared and had not reappeared even six years after the spill (Thomas 1978). Rocky shore animals such as barnacles and periwinkles, however, did not change in abundance or distribution except in areas where their habitat changed. While the *Tsesis* oil spill in the Baltic Sea in 1977 had virtually no effect on the bladder wrack *Fucus vesiculosus* (dormant at the time of the accident) it caused a dramatic acute reduction in biomasses of the sediment dwelling amphipod *Pontoporeia affinis* and the polychate *Harmothoe sarsi* (Notini 1980, Elmgren et al. 1983). Although heavily contaminated with oil, the Baltic telling, *Macoma balthica*, and nematodes were more tolerant and showed only little mortality. Recovery in the littoral zone began within two months, but the speed depended on the degree of exposure to oil and the species involved. One year later the animals had returned to their pre-spill condition, except at the most heavily contaminated stations. Abundance of amphipods, *H. sarsi* and harpacticoids began to increase, and hydrocarbon concentration in *M. baltica* decrease during the second summer after the spill. Three years after the accident *Pontoporeia* and *M. baltica* biomasses had remained depressed, while *H. sarsi* showed normal abundance. The recovery of species with long life span, such as *M. balthica* in this example, require considerably more time than short-lived species (Elmgren et al. 1983). Biological consequences of also other oil tanker wrecks in the Baltic have been monitored. The accidents of *Eira* in 1984, *Antonio Gramsky* in 1979 and 1987, and *Baltic Carrier* in 2001 caused notable effects in particular in the littoral and benthos (The Baltic oil spill 1979, Hirvi 1989, Pécseli et al. 2004). Increased hydrocarbon concentrations in the tissues of e.g. *Macoma balthica*, *Mytilus edulis*, *Lymnea palustris* and *Gammarus* spp. were recorded, and in some cases decrease in the population sizes of these organisms changed. Furthermore, oiled specimens of zooplankton and dead birds (e.g. black guillemots *Cepphus gylle*, and eiders *Somateria mollissima*) were collected. Some fish suffered from morphological abnormalities after exposure to oil. Also, the growth of fish was dampened and in skin diseases were documented (The 1979 Baltic Oil Spill, Hirvi 1989). In the case of Baltic Carrier spill off the Danish coast, hydrocarbon concentration in flounder remained elevated long after the accident (Pécseli et al. 2004).

Oil from the *Amoco Cadiz* on the northwest coast of France spread over a large area, with highest concentrations found in muddy sediments (Cabioch et al. 1981). Amphipods were virtually eliminated, and in the intertidal massive mortality of e.g. heart urchins and razor clams were observed. The *Amoco Cadiz* spill caused permanent changes in the shallow water eelgrass (*Zostera marina*) community, i.e. the total disappearance of filter feeders,

and the very diverse amphipod community has been replaced by the dominance of only two species, one of the new to the area (den Hartog & Jacobs 1980). The spill had little effect on polychaete annelids (Chasse 1978). In the intertidal, the knotted wrack, *Ascophyllum nodosum* (Phaeophyceae, brown alga) was replaced by the much more tolerant bladder wrack, *Fucus vesiculosus* at sites where it grew in the vicinity (Gundlach et al. 1981). Populations of bivalves, periwinkles, and limpets in the intertidal, heart urchins in the benthos, copepods in the pelagial, and sea birds were most severely affected (Conan 1982). The *Exxon Valdez* oil spill in the Prince William Sound in 1989 had relatively mild effects on the littoral communities (Stoker et al. 1992). The survivors were seaweeds (fucoids), barnacles, mussels and periwinkles, which after two years of the accident had re-established themselves and were in a state of prespill condition. A long-time survey of oil effects on sublittoral fine-sand macrobenthic community was made after the *Aegean Sea* oil spill off the NW coast of Spain in 1992-1996 (Gómez Gesteira & Dauvin, 2005). A short period of high mortality of in particular amphipods was followed by a period of low species diversity and low abundance. Recovery began 3 years after the spill.

2.3. Vertebrates: fish, birds, otters, seals, whales and the polar bear

Fish are generally more sensitive to hydrocarbons than invertebrates (Rice et al. 1977a, 1979). Arctic cod (*Boreogadus saida*), for example, is among the most sensitive fish species to oil that were studied by Rice and co-workers (1979). Most pelagic fish also show response relatively quickly to toxicants, while sedate bottom species react slower. Stress, such as fluctuations in water salinity, temperature, food abundance, disease and parasites depress the fitness of fish and thus reduce its ability to tolerate pollution (e.g. Moles 1980, for a review see Rice 1985). Water temperature affects the toxicity of hydrocarbons; in cold water aromatic hydrocarbons persist longer (decreased biodegradation and evaporation), and extreme temperatures (both low and high) may affect the ability of fish to metabolise or excrete aromatic hydrocarbons and their metabolites (Rice 1985). Fish also possess the ability to "learn" to tolerate hydrocarbons; previous sublethal exposure to hydrocarbons induces higher levels of hydrocarbon metabolising enzymes in fish (Egaas & Varanasi 1982). Fish take up oil through ingestion of contaminated food and directly from water (for a review, see Rice 1985). The rate and quantity of hydrocarbon uptake depends on exposure concentration, the molecular weight of the compounds tested, and the amount of lipid in the fish (which, again, is related to fish species, age, season and reproductive stage). Once hydrocarbons are accumulated in fish, many compounds will be metabolised or excreted. Enzymatic metabolising takes place in liver by the mixed function oxygenase systems MFO and the enzyme CYP1A (Rice 1985, Lockhart & Metner 1991, George et al. 1995). The hepatic enzyme CYP1A can be used as a biomarker for petroleum hydrocarbon exposure in the polar cod (*Boreogadus saida*) (George et al. 1995). Most metabolites are probably less toxic than the parent compound and will be excreted. The effects of oil can be targeted to one or several organs in the fish: liver, gut, pancreas, vertebrae, eye lens, stomach, brain and olfactory (odour sensing) organs (studies are reviewed e.g. by Rice 1985) –all sensed as odd fish flavour or smell by us humans. Tainting of commercial fish by oil has been studied on field and experimentally, and in many cases fish show relatively fast purification after light exposure, i.e. the flavour and odour caused by oil disappear (e.g. Ackman & Heras 1992, Lochart & Danell 1992). Fish exposed to sublethal concentrations of petroleum in

the environment show several behavioural, physiological, biochemical and various long-term effects, which are reviewed in depth by Patten (1977), Rice (1985), and in the extensive text book edited by Varanasi (1989). Fish can detect hydrocarbons at different threshold levels (Patten 1977, and references therein) as they have excellent olfaction, which most fish use for detecting hydrocarbons. However, short-term exposures can damage the olfactory epithelia and render these tissues useless (e.g. Solangi & Overstreet 1982). The detection and avoidance behaviour varies among species and life stages, and fish larvae for example are not always capable of avoiding the contaminated area. There are conflicting records of the avoidance behaviour of fish during oil spills; sometimes fish seem to actively avoid the contaminated area, while in other cases they seemed rather affected to it (discussed by e.g. Rice 1985). Other physiological responses include alteration in metabolism and activity, such as decreased rate of heartbeat, cough responses or convulsive respiratory reactions of fish (possibly due to the aromatic compounds in the petroleum), altered respiration, changes in blood parameters and ion concentrations, and decreased energy reserves (Rice et al. 1977b, reviewed in depth by Rice 1985). Oxygen consumption may either increase or decrease, depending on the type of oil and fish species (Patten 1977, and references therein). Narcosis and consequently cessation of movement and feeding, and changes in activity patterns (swimming movements, gulping at the water surface, erratic motion, hyperactivity) is reported for several fish species and their larvae, and are reviewed by Patten (1977) and Rice (1985). Effects on fish reproduction is little studied, but evidence shows that oil affects the survival (mortality) of in particular fish eggs and larvae, hatching, and the development of the embryo (Patten 1977, and references therein, Whipple et al. 1981, Teal & Howarth 1984). Eggs and larvae are easily affected by temperature, salinity and pollutants because they have fewer structures and organs capable of detoxifying oil, are intimate with the environment, their mobility is restricted, and many develop at or near the water surface where oil spill can be expected (Rosenthal & Alderdice 1976, Rice 1985). The earlier the juvenile is exposed to oil the more severely it is damaged. When the embryo approach hatching it is more sensitive to oil than after hatching. The sensitivity again increases until its yolk is absorbed and it begins to feed on its own (Rice 1985, and references therein). There is also a negative correlation between hydrocarbon concentration in water and growth, food uptake and thus weight, percent fat and caloric content of adult and juvenile fish (Korn et al. 1976, Moles & Rice 1983, and the review by Rice 1985).

The published literature concerning bird mortality due to oil is large, and reviewed in depth by e.g. Bourne (1968, 1976), Vermeer & Vermeer (1975), Holmes & Cronshaw 1977, Leighton et al. (1985) and Robertson (1998). In many cases the mortality has been substantial, but it is not always clear how estimates were made, as discussed by Leighton and co-workers (1985). *Torrey Canyon* accident in 1967 caused death of at least 20 -30 000 birds but the actual number may exceed even the estimated one (Holmes & Cronshaw 1977). Even very small oil spills can cause similar mass mortality of sea birds (Barret 1979, Robertson 1998). Divers are at high risk, because they spend much of their time sitting on water. King & Sanger (1979) ranked sea birds in terms of vulnerability to oil pollution, and identified Atlantic puffin (*Fratercula arctica*), common murre (*Uria aalge*), thick-billed murre (*U. lomvia*), razorbill (*Alca torda*) and northern gannet (*Sula bassanus*) as particularly vulnerable. Sea birds are at considerable risk because of their social behaviour (Robertson 1998). Large aggregations of birds occur in connection with breeding, molting, overwintering, and preparation for migration. Oil kills birds in many ways, but the main way is by breaking down the bird's waterproofing and

thermoregulation. An oiled bird will response by preening itself, and consequently inhale and swallow toxic compounds that damage its liver, lungs, kidneys, intestines and other internal organs. Such poisoning is as lethal as the loss of waterproofing, so hypothermia is the actual cause of death (Holmes and Cronshaw 1977, Leighton et al. 1985).

The embryotoxic effects of oil have been studied experimentally to some extent (e.g. Couillard & Leighton 1989, for review see Leighton et al. 1985). Oil from the feathers may pass through the pores in eggshells and either cause death of embryos, abnormalities or affect hatching success. Ingested petroleum may cause ovarian dysfunction (affecting fecundity) and thus delay the onset and rate of lay (Holmes & Cronshaw 1977, Harvey et al. 1982). Depression of growth rate in young birds ingesting oil is a commonly reported phenomenon for a variety of species (reviewed by Leighton et al. 1985). Marine birds must constantly excrete sodium chloride through nasal glands, as they receive salt in excess through their diet. Such osmoregulation can be affected by oil (e.g. Peakall et al. 1983), lead to ion imbalance in body fluids and thus affect e.g. the function of muscles and the nervous system. Anaemia has been reported by e.g. Leighton and collaborators (1983). Exposed birds are also more likely to suffer from other pathological effects such as irritation of the gastrointestinal mucosa, lipid pneumonia, fatty degeneration of liver, atrophy of pancreas, toxic nephrosis, enteritis, aspergillosis, and infective arthritis (for review see Holmes & Cronshaw 1977). Other physiological changes in birds caused by oil are e.g. increased basal metabolic rate, changes in hormone production, body temperature and water flux (Harvey et al. 1982, Hughes et al. 1990).

The Arctic seas are a habitat for a large proportion of marine mammals in the world, in particular whales, seals, sea otters and polar bears (Engelhardt 1985, GESAMP 1993). All marine mammals need to remain in contact with the air-water interface as they are dependent on air breathing, and the polar bear feeds on fish and seals it catches from the surface water. Thus, they may come in contact with a surface oil slick, in particular in sea ice covered areas where the open surface (such as breathing holes) is limited (Engelhardt 1985, Robertson 1998). Oil causes problems to mammals through coating by oil and inhalation of volatile hydrocarbons. Inhalation can be life threatening in the case of prolonged exposure (Geraci & St. Aubin 1980). Certain marine mammal species may additionally be vulnerable through the food vector, e.g. bivalves and crustaceans that have a potential to bioaccumulate hydrocarbons (Engelhardt 1985). The information of effects of oil on whales is quite limited, but generally whales are anticipated to remain unharmed by contact with oil (Engelhardt 1985, and references therein, Robertson 1998). There are no records of oil fouling of the skin of free-living whales, suggesting either that oil may not stick to the skin surface due to its quality, or that contact with oil is rare because whales avoid slicks. Whether active avoidance occurs remains uncertain, but observations in spill situations suggest that whales do not take notice of oil spills (Engelhardt 1985, and references therein, Robertson 1998). However, in experiments exposure of cetacean skin to oil has caused cell damage in epidermis (Geraci & St. Aubin 1982). The unique structure of cetacean skin and the fact that it contains lots of vitamin C may serve to protect against harmful effects of oil (Geraci & St. Aubin 1980). There is some suggestion that whales may take up petroleum derived hydrocarbons (Engelhardt 1985). The presence of MFO and a hydrocarbon marker enzyme P-450 has been demonstrated in the liver of several cetacean species (Geraci & St. Aubin 1982), indicating that cetaceans should be capable of detoxifying oil. Spilled oil may interfere with feeding behaviour, in particular filtering efficiency, through effect on baleen function as oil becomes trapped onto

baleen hairs (Geraci & St. Aubin 1982, Braithwaite et al. 1983). However, the development of offshore petroleum and gas resources seems to present more threats to marine mammals than accidental oil spills in the Arctic (Geraci & St. Aubin 1980). Seismic activities, noise (affecting physiology and behaviour), and long-term accumulation of petroleum fractions through the food chain in the vicinity of oil platforms are thus more of concern.

Behavioural consequences and thermal effects of oil fouling have been noted for pinnipeds and sea otters (Geraci & St. Aubin 1980). Davis & Anderson (1976) noted reduced growth rate in oiled seal pups, but could not detect changes in nursing behaviour as a result of oiling. Experimentally oiled sea otters spend more time underwater trying to clean them selves, and seals show variable signs of aggression and arching of the back (Geraci & Smith 1976a, Geraci & St. Aubin 1980). After removal from oiled experiment tanks, animals' behavioural and physical signs disappeared rather quickly. Thermal effects are noted for sea otter pups (*Enhydra lutris*), and the Weddell seal (*Leptonychotes weddellii*) (Kooyman et al 1977). Most true and phocid seals, sea lions and walrus, however, have a relatively coarse and short fur and thick blubber. The danger of heat loss due to oil fouling is therefore small, but the fouling may cause a physical hindrance to swimming (Davis & Anderson 1976, Kooyman et al. 1977, Robertson 1998). Sea otters are peripheral in their occurrence in the Arctic and may therefore be more sensitive to oil effects, in particular thermal effects, than the "true" arctic mammals (Kooyman et al. 1977). Oiling and consequent washing of sea otter furs caused oxygen consumption increase and weakened thermoregulation (Costa & Kooyman 1981). The vulnerability of sea otters was documented clearly during the *Exxon Valdez* oil spill in Prince William Sound in 1989, where estimated 2000-3000 animals perished (Waldichuk 1990). Some anticipated effects of surface contact with oil are irritation and inflammation of eyes, skin, and sensitive mucous membranes (Geraci & Smith 1976a). In experiments acute organ damage has not been recorded even after ingestion of relatively large quantities of oil in ringed seals (*Phoca hispida*) (Geraci & Smith 1976a), but such findings cannot be extrapolated to greater quantities of oil and other marine mammals. Ringed seals rapidly absorbed crude oil hydrocarbons to body tissue and fluids, ultimately excreting the compounds via bile and urine (Engelhardt et al. 1977). Experimental exposure of adult ringed seals to hydrocarbons increased the mixed function ogygenase (MFO) activity, indicating the enzymatic break-down of hydrocarbons in these animals (Engelhardt 1981). Furthermore, studies in ringed seals showed that volatile hydrocarbons and likely to become absorbed through respiratory tract (Geraci & Smith 1976b). Kidney and liver lesions were observed, but no associated lung pathology. Effects of prolonged inhalation may cause disturbance of the central nervous system, pneumonia and death (Carpenter et al. 1978). Oiling of grey seal pups (*Halichoerus grypus*) did not affect the body weight development as the oil typically disappeared when the white lanugo fur was moulted (Jenssen et al. 1991).

Regarding polar bears, severe heat loss and elevated compensatory metabolism have been observed after experimental oiling of their fur (Hurst et al. 1982). Extensive grooming causes ingestion of oil leading to tissue elevation of hydrocarbons and gradual development of dysfunction, and lethal damage in several internal organs. Renal failure may be the ultimate cause of death (Øritsland et al. 1981).

3. Discussion

Petroleum hydrocarbons affect organisms in a variety of ways, ranging from death to biomolecular, pathological and cellular effects to merely physical nuisance. The effect on organisms, and whether it is reversible or not, depends on numerous physical and biological factors, that also affect oil spreading, weathering (including biodegradation), and absorption of toxic compounds into organisms. These are for example the volume and type of spilled oil, water temperature and water currents, the presence of sea ice and snow cover, season (in particular in the Arctic), location of the spill (open water vs. shore line) and oil combating measures taken. At least some effects seem temperature dependent and are more pronounced in warmer water, thus not likely in the Arctic. For long-lived organisms, such as kelps and vertebrates, the developmental stage can be crucial; reproductive organs, embryos and juveniles are at most risk. Thus, we cannot nominate only a single species in the Arctic that could be determined as particularly sensitive to oil and consequently an indicator species of the severity of oil pollution. Sublethal effects are numerous and can become manifested in various ways in the organisms' reproduction, behaviour (e.g. feeding, mating) and thus growth, and physiology (general fitness). The apparent complexity of e.g. the metabolic response to petroleum is hardly surprising in view of the broad range of physiologically active compounds present in crude oil and physiological processes that are undoubtedly affected in different aquatic species.

An evaluation of the consequences of the environmental contamination requires an understanding of the extent to which it is responsible for changes in individuals and populations in the affected area (discussed in depth by e.g. Clark 1982, Jones 1982). Population change is not solely related to mortality which may be observed, but depends also on the population dynamics (e.g. migration, age structure and reproduction patterns of a given population), stock size, survival strategy of the species affected, and other possible disturbances in the area. Furthermore, while individual organisms may die to oil effects, on a population level the outcome may not be dramatic. At open sea, plankton is likely to be transported from one area to another by wave action and water currents, and thus the contaminated part of the plankton population may become substituted with "fresh" material from elsewhere. In such an example, the actual sufferers may be found in the benthos; organisms whose food quality and uptake is dependent on the quantity and quality of settling material from above (here dead, oil contaminated plankters).
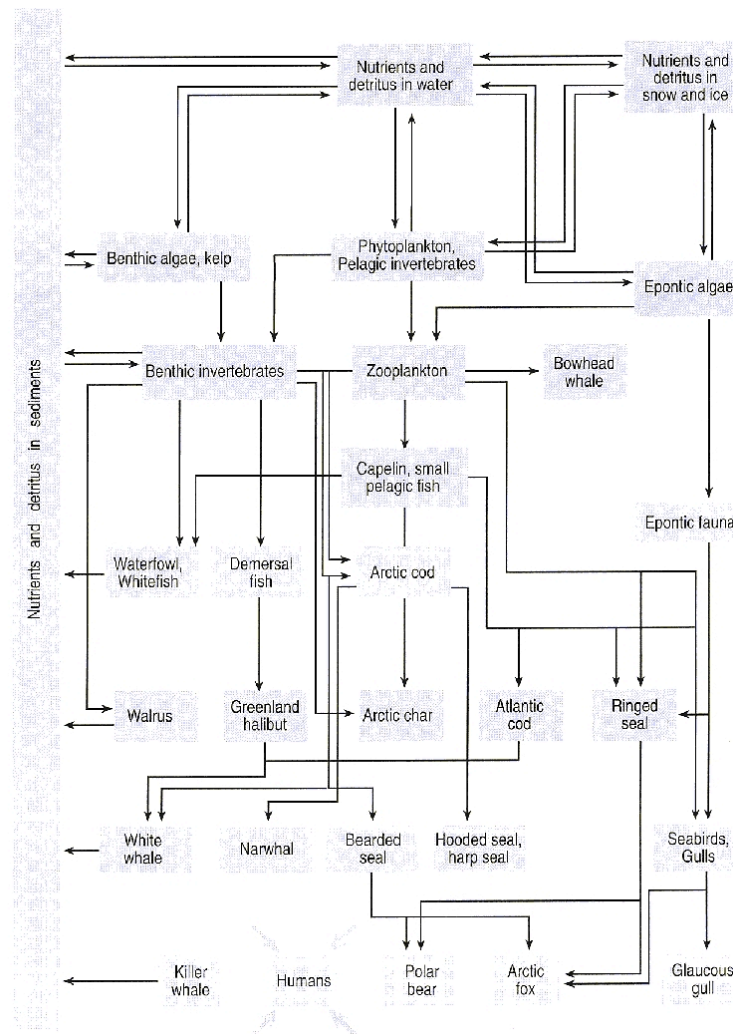
Figure 3. A generalized marine food web in the Arctic. Souce: de March et al. (1998).

In the littoral, most severe effects of oil spills are documented for soft bottoms and sheltered bays. Oil penetrates and readily remains between the tiny pores of mud and silt, where it will have more time to stay in contact with benthic organisms. On hard bottoms, such as rocky shores, wave action is usually stronger, and thus capable of transporting even large masses of oil to the open sea. The clinging of oil onto hard surfaces is much less than oil penetration into soft substrates. In general, field studies of the biological consequences of oil spills show good agreement with the experimental data: intertidal and subtidal benthic communities are affected and can take a long time to recover, undergoing slow and subtle changes.

Oil does not need to affect directly all organisms on every level of the food web, and yet with time the consequences of an oil spill may be discernible in the entire system (Figure 3.). Such is a consequence of biomagnification; when littoral and benthic crustaceans for example are contaminated by oil, the negative effects will soon be observed in their predators, such as polar cod, seals and, finally, the polar bear, i.e. animals than migrate to the contaminated area from elsewhere. Another example is from the open water ecosystem. Divers and whales will bioaccumulate hydrocarbons by consuming pelagic fish

and invertebrates exposed to oil, and consequently develop various hydrocarbon related symptoms (Figure 3.).

Fishery is always one of the main concerns when an oil spill happens. Field studies after oil spills have generally failed to document the widespread effects of oil on fish. In *Argo Merchant*, *Ekofisk Bravo* and *Amoco Cadiz* accidents effects on fish stocks were negligible (e.g. Rice 1985 and references therein, Teal & Howarth 1984). Although adult fish can be killed by oil spills, this probably poses less of a threat to commercial fisheries than do damage to eggs and larvae, or changes in the ecosystem supporting the fishery (Teal & Howarth 1984). Ecosystem changes in the lower levels of the food chain and thus long-term effects on fish are more likely than acute effects. However, such interactions are poorly studied, as is discussed e.g. by Vandermeulen (1982) and Teal and Howarth (1984). One of the reasons for this is that long-term spill effects are generally local phenomena. Fish have been exposed to oil in many studies, but the methods have varied considerably among studies. Thus, direct comparison between studies is not usually valid.

There is a general impression of the exceptional sensitivity of Arctic ecosystems to oil. Sensitivity of Arctic zooplankton and benthic organisms to oil contamination, for example, is discussed by Wells and Percy (1985), and it seems to be a consequence of several biotic and physical factors. In cold water zooplankton has lower uptake, metabolic and excretion rates, possibly leading to lower inputs of oil into organisms, but also to longer detoxification and depuration times. Low temperature can influence the availability of oil by changing its solubility, physical form and the stability of different fractions, or by extending the exposure time by retarding the loss of hydrocarbons from the medium, and extending the predisposition time of organisms to oil. It can also delay the onset of e.g. criteria of death like immobilisation. Based on extensive experimental data set Rice and co-workers (1976, 1977a) concluded, that Alaskan marine invertebrate species may be slightly more sensitive than similar species in lower latitudes. However, our knowledge of the species specific sensitivity of Arctic organisms to oil is still patchy. In the past a number of arctic benthic species have been subjected to acute lethal tests with petroleum hydrocarbons (Wells & Percy 1985) but the taxonomic coverage has been uneven: of the 24 species listed by Wells and Percy (1985) almost 90% are crustaceans and at least 50% are amphipods. Also, comparison of results is complicated by differences in oil types, methods of preparation, exposure regimes and criteria of death. In general, amphipods and decapods seem relatively sensitive to oil, while isopods are more resistant. As noted earlier, the polar cod seems to be particularly sensitive to oil (Rice et al. 1979).

Avoidance is one way of minimising the negative effects of oil. However, avoidance of oil by motile organisms is not a universal type of behaviour, as was evidenced for the isopods, and fish such as the juvenile English sole (*Parophrys vetulus*, syn. *Pleuronectes vetulus*) by Percy (1976) and Weber and co-workers (1979). Isopods did not seem to exhibit any particular attraction or repulsion responses when confronted by oil masses. The effects that the oil in the sediments may have on fish can depend on overall amount, composition (age) of the pollutant, and the season (activity level of the fish) (Fletcher et al. 1981). Whether fish in nature avoid, ignore or are attracted to an area with an oil spill largely remains unanswered. There are several cases where field observations have been conflicting, as evidenced during past oil spills, where both avoidance behaviour and attraction of in particular fish has been recorded (Teal & Howarth 1984). Non-avoidance behaviour of animals when confronted with crude oil may play a crucial role in determining

the subsequent impact upon organisms and thus populations. Burrowing species, like the much studied arctic species, the Baltic telling *Macoma balthica,* rarely venture onto the surface, although most extend their siphons into the overlying water for feeding and respiration. The impact of oil on burrowing species depends largely upon how it is introduced to the habitat. The response of the animal to dispersed oil in the overlying water and to oil within the sediment is to emerge onto the surface, not to burrow deeper into the substrate, and usually in proportion to the dose (Taylor & Karinen 1977, Wells & Percy 1985). Although the sediment depth may affect the surface speed, chronic contamination will force animals to eventually surface (Wells and Percy 1985, and references therein). Not all animals would become killed by oil, at least in the short term, but high concentrations (3 mg/l) and long-term exposure would no doubt be lethal. Reburying is not possible for as long as the oil prevails (Taylor & Karinen 1977, Engelhardt et al. 1983).

Altogether, it seems that long-term effects of oil spills on open water ecosystem, including cetaceans may be less important than generally assumed, and restricted to a relatively small area. However, there is uncertainty about the effect of oil in restricted environments, such as in polynyas and ice leads. In the littoral and benthos, where organisms are often either sessile or slowly moving, and the motility of the spilled oil is weaker than in the open water the effect is likely to be more pronounced, including consequences on an individual and population level, and expanding the time of disturbance. Observed effects of long-term and chronic discharges in marine ecosystem have not been discussed in this report. However, to summarise the existing literature, effects of e.g. oil drilling activities are sooner local than widespread, but the chronic exposure of organisms to oil and drill muds, for example, can cause pronounced, permanent changes in species composition, fitness and diversity of benthic communities (e.g. Sharp & Appan (1982), Addy (1987), Kingston (1987), Moore and co-workers (1987a), Gray et al. (1990) and Olsgard & Gray (1995). The temporal changes seen after oil spills are comparable with the spatial changes observed around chronic discharges, in particular in case of an ecosystem with dominance of only a few species.

4. Summary and conclusions

Oil spill effects on Arctic marine organisms and ecosystems can locally be dramatic. The severity of contamination is not only dependent on the type of organisms exposed to oil, but also on the type and volumes of spilled oil, the speed of oil weathering processes (e.g. evaporation, dispersion, degradation by bacteria), oil combating measures taken, and the location of the spill. Large spill of oil with a high aromatic fraction is worse to marine life than oil with less aromatic compounds. Weathering of large oil spills require more time than that of smaller spills, and the use of oil dispersants (not discussed in depth in this report) has usually more severe consequences on organisms than the mechanical and/or biological oil combating. Organisms in the littoral and benthos, particularly sessile and slowly moving invertebrates such as molluscs, snails, and crustaceans suffer the most from oil exposure. Seaweeds (macroalgae) attached to hard substrates seem to have a fairly good potential for recovery after oil exposure. In the open water, contaminated plankton can be replaced by "fresh material" transported from sea areas outside the spill site. Avoidance of oil is typical for fish and larger vertebrates (seals, whales), but sea birds, in particular divers are at risk in the pelagial. Oil associated with sea ice may have a

notable effect on the ice biota, particularly as oil tends to become trapped between ice platelets and floes, and within brine channels in ice, and thus lengthens the contamination time. Recovery of marine life may be complete, nearly complete or only partial. Complete recovery is more likely in the open water ecosystems where water currents disperse oil both horizontally and vertically. Partial recovery is typical of littoral and benthic ecosystems, and is dependent not only on the factors described above, but also on the type of the original community (species composition, dominance, biomass), the organisms' potential of adjusting to changes (abiotic and biotic) in the habitat, and thus the ability of recolonising the deserted area. Even after several years of an oil accident, the recovery process of a littoral or benthic ecosystem may still be incomplete and, in the worst case, may not ever reach the original state.

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