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Assessing the long-term impact of oil spills: an examination of recent incidents

Following major oil spill incidents there is a considerable focus on environmental impact assessment, particularly in the short-term. These can arise both as a result of toxic impacts (toxicity and bioaccumulation) and physical impacts (smothering and sediment contamination). There may also be impacts as a result of remedial activities, for instance to sand-dune systems caused by heavy vehicle traffic. Different ecosystems also have varying recovery rates. Long-term impacts can be more difficult to assess, particularly against a background of natural variability and change which would occur in the absence of a spill. We consider the impact assessments made following a number of recent incidents (*Braer*, *Sea Empress*, *Erika*, *Prestige* and *Tricolor*), alongside studies following the *Exxon Valdez* spill, to determine whether the current framework of environmental risk assessment and toxicology can adequately address the issues of long-term impacts of oil spills.

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Introduction

This paper has been prepared in response to an OSPAR request to the ICES Working Group on Marine Chemistry at its 2005 meeting, regarding the assessment of the long term impact of oil spills on marine and coastal life. Specifically, whether the current framework of environmental risk assessment and toxicology is sufficient to take account of the long-term effects of oil pollution. In order to address this question, we have reviewed the impact assessments made following a number of recent incidents in Europe (involving the *Braer*, *Sea Empress*, *Erika*, *Prestige* and *Tricolor*). In addition, we have also reviewed the findings of the long-term studies which have followed the 1989 *Exxon Valdez* spill in Alaska, and assessed the ability of these studies to unambiguously assign the effects observed to each incident. Although it is often assumed that oil spills impact essentially clean, even pristine, environments, this is never the case in fact. Other human activities also have effects similar to those caused by spilled oil and often occur in the same coastal waters that are affected by spills. The elucidation of cause and effect presents a considerable challenge to scientists, particularly as the incident progresses and concentrations of oil and its component chemical compounds begin to fall and approach the local background concentrations.

Case-studies

Braer

On 5 January 1993 the tanker MV *Braer* grounded at Garth's Ness, in south Shetland. Over the following twelve days the entire cargo of 84,700 tonnes of Norwegian Gullfaks light crude oil, together with a quantity of bunker fuel, was released to the marine environment. This was the eleventh largest spill in history. Due to the severe wind, wave and current conditions the oil was carried to the north and west. In addition, significant quantity of oil was carried some distance south. Relatively little of the oil impacted the coastal ecology. On 8 January 1993, a Food and Environment Protection Act 1985 (FEPA) Exclusion Zone was put in place to prevent contaminated fish and shellfish reaching the market place (Whittle *et al.*, 1997). A programme of monitoring was put in place to establish that the limits of the Exclusion Zone had been correctly drawn and to provide support for any decision-making process with regards to the possible extension or future lifting of the Zone. The restriction was lifted for fish and most shellfish over the following 2.5 years under the criteria that the fish or shellfish contained no petrogenic taint and the concentrations of polycyclic aromatic hydrocarbons (PAHs) were within the range of reference fish and shellfish (Topping *et al.*, 1997). From 1995 until the final lifting of the FEPA Exclusion Zone in 2000, the Zone remained effective for only Norway lobster (*Nephrops norvegicus*) and mussels (*Mytilus edulis*).

A decision was taken to utilise mussels to monitor the long term hydrocarbon pollution within the Exclusion Zone. As part of this monitoring exercise, mussels from a reference site (Olna Firth) were transplanted to three sites within the Exclusion Zone (Stromness Voe, Sandsound Voe and Merry Holm), suspended in mesh boxes from rafts (Webster *et al.*, 1997). The transfer of reference mussels from a single site to all three sites within the Zone ensured there a commonality for the test matrix. Samples were collected at regular intervals for PAH analysis. Seasonal trends were observed in the PAH concentrations, with concentrations being higher in winter and decreasing in the spring. Total PAH concentrations in the Zone mussels at all three sites were consistently higher than concentrations found in the reference mussels. Despite there being no petrogenic taint detected the Exclusion Zone could not be lifted. Not until March 2000, when further work looking at the PAH profiles and the geochemical biomarker profiles demonstrated that the higher PAH concentrations were not due to contamination from Gullfaks crude oil, could the Zone finally be lifted for all species. The PAH profiles were dominated by the heavier PAHs, and the lack of oil-related geochemical biomarkers confirmed that the higher PAH concentrations within the Zone were not due to Gullfaks crude oil. This work highlighted the importance of establishing background concentrations in the marine environment to ensure the relevant information is available to enable the competent authorities to make appropriate decisions following a marine incident such as an oil spill.

Immediately after the oil spill, sediment samples from over a wide area both east and west of Shetland were screened for aromatic hydrocarbons using ultraviolet fluorescence spectroscopy (UVF) with selected samples being analysed for PAHs by gas chromatography-mass spectrometry (GC-MS) (Davies *et al.*, 1997). It was estimated that approximately 30,000 tonnes (35%) of the Gullfaks crude oil from the MV *Braer* was deposited in the sediments. Much of the oil was deposited in the

sheltered coasts of south-west Shetland and in two offshore areas; to the west of Burra Isle (Burra Haaf), and an area south east of Fair Isle. Most of the sediment around the southern tip of Shetland is coarse sand, however Burra Haaf and south-east Fair Isle have a higher mud content. Long term temporal monitoring programmes were put in place to study the fate of oil in these two sedimentary basins. Between 1993 and 2000, core sediment samples were collected from a small grid of nine stations in each area. In addition, sediment samples were collected from three transects in Burra Haaf (1993 and 1994) and from a reference transect in St. Magnus Bay (1993). In 2003, 10 years after the *Braer* oil spill, sediment samples were collected from the Burra Haaf and St. Magnus Bay transects. Total PAH concentrations were generally low ($<150 \mu\text{g kg}^{-1}$ dry weight for 2- to 6-ring PAHs, parent and alkylated) in both Burra Haaf and St. Magnus Bay. The PAH profiles of the 2003 sediments were dominated by the heavier, 4- to 6- ring, PAHs and contained a high proportion of parent PAH ($>40\%$) consistent with pyrolytic sources of PAHs. Individual PAH concentrations were below OSPAR's proposed Background Assessment Concentrations (BACs) for PAHs in sediment (Moffat *et al.*, 2004). However, all Burra Haaf sediments showed evidence of petrogenic contamination containing unresolved complex mixtures (UCMs) in the GC-FID chromatograms. The triterpane profiles indicated this was North Sea oil containing the North Sea oil marker, bisnorhopane and the sterane profiles were consistent with that of Gullfaks oil. Highest oil equivalent and PAH concentrations in the Burra Haaf cores, sampled in 2003, were found in the 2-5 cm and 5-10 cm sections, indicating there has been some movement of the Gullfaks oil downcore.

In addition to the sediment monitoring, biological effects of oil on flatfish from Burra Haaf and south east Fair Isle were determined by measuring the levels of a detoxifying enzyme, ethoxyresorufin-*O*-deethylase (EROD, produced by the liver in response to the presence of planar molecules, including PAHs). The enzyme was measured in dab from Burra Haaf and south east Fair Isle and from a reference site outwith the zone of impact. In 1993, detoxifying enzyme levels were found to be higher in dab from Burra Haaf and at south east Fair Isle compared to the reference area. By 1996, and on subsequent occasions, the levels at these locations had fallen to those found at sites not impacted by the oil spill.

Sea Empress

The tanker *Sea Empress* grounded on rocks at the entrance to Milford Haven, West Wales, in February 1996. Over a period of a week, 72,000 t of Forties blend crude oil and 480 t of heavy fuel oil were lost (Law & Kelly, 2004). The UK government established the Sea Empress Environmental Evaluation Committee (SEEEC, 1998) to co-ordinate the monitoring which supported the environmental impact assessment of the incident. Within the broad categories of pollutant behaviour, marine impacts and shoreline and terrestrial impacts SEEEC commissioned or received input from almost 100 individual studies, which were conducted in parallel with the fish and shellfish monitoring programme which underpinned the fishery closure implemented shortly after the spill began. Most of these studies were short-term (weeks to months) although a few continued for two years or longer.

A vigorous dispersant spraying operation significantly reduced the quantity of oil which came ashore and entered subtidal sediments (Lunel & Elliott, 1998). Within

Milford Haven itself, an oil sheen was apparent in bottom sediment samples in March 1996, but this had disappeared by October 1996 when the profile of the polycyclic aromatic hydrocarbons (PAH) indicated that the majority source was combustion-related rather than oil-related (Rutt *et al.*, 1998). Both the fish and shellfish monitoring programme and a range of biological effects studies showed that the demonstrable effects of the spill were over within two years (Law *et al.*, 1998). Within little more than a year, the contribution of 2-3 ring PAH from the spilled oil to the contamination of shellfish tissues had returned to the presumed background concentrations prior to the spill, and had been replaced by a seasonal cycle of 4-6 ring PAH derived from combustion sources (Law *et al.*, 1999). Studies of immune function in mussels showed that the capability of the mussels to resist infection was reduced in the aftermath of the spill, but was also reduced in response to the rising concentrations of combustion-derived PAH after the background had been re-attained (Dyrynda *et al.*, 1997; 2000). That is to say, the signal due to the Sea Empress oil spill could no longer be distinguished from that due to the chronic pollution levels in the environment. Subsequent work has shown that DNA-adduct levels in mussels also show a seasonal variation in response to the PAH concentrations in their tissues, with the highest incidence of adducts in the winter when PAH concentrations are also at their highest (Halldóra *et al.*, 2005). Slow recovery of the amphipod populations within the Milford Haven waterway was documented by Nikitik & Robinson (2003), but effects on plankton and fish populations could not be seen during 1996/97 (Law & Kelly, 2004).

Erika

The tanker *Erika* broke in two and sank about 30 nautical miles off the coast of Northern Brittany (Penmarc'h Point, Finistère, France) on 12 December 1999. Within a few hours a total of about 19 000 tonnes (Laubier *et al.*, 2004) of heavy fuel oil was released from the sunken tanker. The oil formed thick slicks which drifted and spread over the sea surface before massively stranding on the coast between 23 and 27 December. The recovery of the heavy fuel oil from the sea was difficult, due partly to the weather conditions and especially to the difficulties of pumping the very dense and viscous oil. Almost 400 km of the coastline were impacted to differing degrees by the stranded fuel-oil. An intensive cleaning of the coastline took place during 2000 and 2001, and an estimated 240 000 tonnes of oily wastes were collected from the beaches and rocky shores.

After the spillage, an important number of actions and programs were implemented by the French government authorities. This included both rescue and response action plans including intensive clean up and restoration of oiled shores, as well as a number of research, technological and monitoring programs for a broad range of environmental and social impact assessments (all together more than 60 projects were launched). The monitoring programme included thirty studies aimed at the assessment of the spatial and temporal extents of the chemical contamination, studies of transformation and bio-availability of contaminants (mostly polycyclic aromatic hydrocarbons PAHs, vanadium and nickel) and studies of effects on living organisms including supralittoral and intertidal species.

The investigation of the chemical contamination of water, suspended particulate matter (SPM) and intertidal marine molluscs showed that all of these compartments of

the Bay of Biscay were significantly and persistently contaminated by PAHs originating from the *Erika* fuel-oil (Tronczynski *et al.*, 2004). In the case of the *Erika* oil spill, this observation was rather straightforward as the French National Monitoring Network (Réseau National d'Observation - RNO) data and sample bank provided for the Bay of Biscay and following the *Erika* oil spill a unique opportunity to compare the results between numerous pre-spill and post-spill sediment and mollusc samples (Tronczynski *et al.*, 2004). Indeed, to have good coverage of pre-spill environmental reference samples in oil spill studies is extremely rare, and very often in order to distinguish chronic contamination of the marine habitats by PAHs from that due to a given oil spill the comparison can be made only with those within of the source oils (Douglas *et al.*, 1996, Christensen *et al.*, 2004). Thus, after the *Erika* oil spill, significant changes in concentrations and compositional patterns of PAHs in post-spill samples were reported for all environmental compartments of the Bay of Biscay. For instance, the compositional patterns of PAHs constantly included alkyl-substituted phenanthrenes, pyrenes, chrysenes and sulfur heterocyclic compounds in higher relative abundances than those in the pre-spill samples from these compartments. However, the assessment of the chemical contamination in subtidal sediments did not show significant contamination by *Erika*-derived PAHs. The contamination of the subtidal sediments by *Erika* fuel oil was limited only to a few isolated locations and intertidal sediments (Tronczynski *et al.*, 2004). This suggests that very little of the *Erika*'s fuel oil reached the seabed by direct deposition. The large floating oil slicks did not tend to break up and were very compact when they came ashore. A low incidence of contamination of the subtidal sediments might be a more general characteristic of heavy fuel oil spills (Bassin & Ichiye, 1977), although in turbid waters the fuel oils can pick up sediments and eventually sink due to the increased density, as happened following the *Eleni V* spill in the UK in 1978 (Blackman & Law, 1980).

The contamination of the molluscs by *Erika*-PAHs was shown by a very rapid and significant increase in the concentrations of PAH in their tissues, as well as by a dramatic change in the distribution pattern of compounds in the shellfish after the spill. The differences observed in PAH concentrations between pre-spill and maximum post-spill levels in February 2000 show that PAH levels rose by a factor of 22 times for the sum of the unsubstituted PAH and of 171 times for the sum of the alkylated PAH. The direct ingestion of the fuel oil was certainly one of the main routes of contamination in molluscs. However, subsequent recontamination via water and SPM intake prolonged the depuration period. Despite a consistent and significant decline of PAH concentrations in all compartments during the months following the spill, three years after the spill the PAH levels in the mollusc samples collected at the sites which were most heavily contaminated by the fuel were still elevated and the composition distinguishable from that observed before the spill. At all of the less impacted sites the concentration levels and compositional patterns of PAH returned more rapidly to the reference pre-spill situation. The results of chemical monitoring after the *Erika* disaster shown that heavily oil contaminated shorelines, including beaches, rocky coasts as well as sandy sediments apparently became reservoirs of spilled fuel and these continued to contaminate seawater, suspended particulate matter and mussels with PAHs (Tronczynski *et al.*, 2004).

Levels of nickel and vanadium have also been monitored in benthic invertebrates following the *Erika* oil spill (Chiffolleau *et al.*, 2004). The concentrations and

temporal trends of these metals were determined and studied in mussel (*Mytilus edulis*) and oyster (*Crassostrea gigas*) soft tissue samples collected after the *Erika* oil spill and in the pre-spill samples obtained from the mollusc specimen sample bank of the RNO. These retrospective analyses provided baseline reference concentrations for vanadium and nickel in mussels from the Bay of Biscay. The daily growth bands of a scallop (*Pecten maximus*) shell sampled by laser ablation were also analysed for nickel and vanadium. These analyses may provide a record of the contamination event by metals incorporated into shell of the mollusk, as this is not subject to the depuration processes of the mollusc (Chiffoleau *et al.*, 2004). The nickel concentrations in mollusks tissues did not show any significant rise following the *Erika* oil spill. The vanadium concentrations show a peak 5 months after the wreck, in May 2000. This peak was probably related to delayed bio-availability of this metal from the oil (Chiffoleau *et al.*, 2004). The analysis of the daily growth bands of the scallop shell similarly showed a peak concentration of vanadium in May 2000, whereas nickel concentrations stayed remarkably stable. The results of this study provided a new monitoring strategy for following chemical contamination by metals related to an oil spill (Chiffoleau *et al.*, 2004).

The ecological effects of the *Erika* oil spill were studied for littoral plants, marine invertebrates, benthic fish, marine birds, and marine mammals (Laubier *et al.*, 2004). The temporal changes in community structure in intertidal rock pools were investigated over a 3 year period (Barillé-Boyer *et al.*, 2004). Initially, in tidal pools at sites heavily contaminated by *Erika* fuel oil, there was a dramatic increase in the abundance of two macroalgae *Ulva sp.* and *Grateloupis doryphora*, followed by a 100% mortality of sea urchin (*Paracentrotus lividus*) about three weeks after the accident (Barillé-Boyer *et al.*, 2004). Only two years later did the first sea urchins returned to these intertidal pools and it took three years to attain sea urchin population densities comparable to the reference state recorded prior to the oil spill (Barillé-Boyer *et al.*, 2004). A multiparametric diagnosis of the immune system response in Pacific oysters (*Crassostrea gigas*) was conducted on individual haemolymph samples to identify structural, immunopathological alternations and functional impairment of immunocompetent cells. The results show that, one year after the spill, severe immunological alternations could be detected in oysters from heavily contaminated sites (Auffret *et al.*, 2004). A 3 year study of a number of biological markers was also conducted in mussels (*Mytilus edulis*) (Bocquené *et al.*, 2004). Most of the biomarkers studied were enzymes involved in detoxification systems such as glutathione-S-transferase (GST), catalase (CAT) and malondialdehyde (MDA). Deoxyribonucleic acid (DNA) adducts were also studied and the level of acetylcholinesterase (AChE) inhibition activity was determined (Bocquené *et al.*, 2004). The results demonstrated a rather short and weak response of these biomarkers in mussels exposed *in situ* to the *Erika* fuel oil. No significant reductions in GST or CAT levels were observed, however, the levels of DNA adducts and of MDA were much higher during the winter and spring following the oil spill. The levels of AChE were also much lower during the first year of the study, suggesting physiological stress (Bocquené *et al.*, 2004). It was suggested also that the potential physiological and immunological impairments on an oil spill on the mussel population require longer-term investigations (Bocquené *et al.*, 2004). Although no direct mortality was reported for the benthic fish population of the sole (*Solea solea*), certain biological markers of functional integrity at an individual scale were validated both *in-situ* and by experimental studies (Claireaux *et al.*, 2004). It appears that, for

this benthic fish species, the most appropriate and easily implemented biological indicators of the oil spill effects are the condition factor and somatic or otolith growth (Claireaux *et al.*, 2004).. 1-hydroxypyrene was determined as the major metabolite in the bile of sole exposed to *Erika* fuel oil both *in-situ* and under experimental conditions (Budzinski *et al.*, 2004). The *Erika* oil spill was a major ecological disaster for seabirds wintering in the Bay of Biscay (Bretagnolle *et al.*, 2004; , Cadiou *et al.*, 2004; , Castège *et al.*, 2004). Among different species oiled, common guillemot (*Uria aalge*) appeared to be the most affected species (nearly 70,000 birds were found dead or oiled on beaches). Among the most affected species, at-sea populations of some declined during the two years following the accident (ex. razorbill, common scoter), whereas others did not (guillemot, gannet) (Castège *et al.*, 2004). Some less abundant seabird species in the northern Bay of Biscay decreased very strongly (ex. *Gavia sp. Fulmarus glacialis*, Castège *et al.*, 2004). Finally, no marine mammal mortalities were attributed to the *Erika* oil spill (Ridoux *et al.*, 2004).

The *Erika* oil spill case has shown a need for a better understanding of the reference state of marine habitats. A better knowledge of the structure and functioning of marine ecosystems prior to accidents such as oil spills is necessary in order to be able to better understand changes induced in the ecosystems as a result of such accidents. An improved knowledge of the effects of low-level, chronic contamination should allow also a better discrimination of the effects related to the oil spills.

Prestige

On the 13th of November 2002, the *Prestige* got into trouble near the Spanish coast. After a week of erratic towing during which about 19,000 t of heavy fuel oil was spilled, it finally broke in two and sank off the Galician coast (NW Spain). During the next few months, some 40,000 t of fuel oil leaked into the sea, forming large slicks. This oil reached the coast over a long period of time. It affected all the Galician coasts (except the inside of the Rías Bajas), the Spanish part of the Bay of Biscay and even part of the French coast. An important part of this fuel remained at sea in the Bay of Biscay until the summer of 2003, when northerly winds took it ashore in the form of small tarballs.

A Scientific Coordination Committee was established to coordinate the initial, urgent actions that should be carried out, and to evaluate the three-year projects that were going to be funded by the Government.

The physicochemical characteristics of the fuel determined its low solubility in the sea water and the studies showed low levels of total dissolved/dispersed hydrocarbons in the area studied (at least 3-4 miles from the coast) (Soriano Sanz, J.A. *et al.*, 2003). The sediments also showed low-moderate hydrocarbon concentrations, probably due to the low dispersibility of this type of fuel oil and because the main slicks reached the shoreline and were, when possible, cleaned from land (Franco Hernández, M.Á. *et al.*, 2004). An important impact was however seen in the biota living on the coast: concentrations of PAH in mussels, sea urchins, goose barnacles and razorshells reached a maximum two to three months after the spill and then started to go down to re-establish “normal” values during the spring-summer 2003 (Soriano Sanz, J.A. *et al.*, 2005, Viñas Diéguez, L. *et al.*, 2005).

With regard to other studies including those on biological impacts, the conclusions are not definitive as the projects will not be completed until December 2006. Meanwhile, a lot of publications are being published on this spill and its assessment (Marine Pollution Bulletin, special issue, in press) and further presentations of their findings can be found in the proceedings of the ICES Annual Science Conference, 2005.

Tricolor

On the 14th of December 2002 the car carrier *Tricolor* collided with the container ship *Kariba* when both vessels were about to enter the north-south shipping route through the English Channel. The *Tricolor* turned on its side and sank in less than half an hour. The position of the shipwreck was in the middle of a very busy shipping route in French waters approximately 35 km north of Dunkirk, and close to the boundaries of the Exclusive Economic Zones of Belgium and the United Kingdom. The *Tricolor* had nearly 2,000 tons of hydrocarbons on board, most of it being heavy fuel oil (Kerckhof *et al.*, in press). On 1 January 2003, the tanker *Vicky*, with a cargo of 70,000 tons of gasoline, and more than 2,000 tons of heavy fuel oil on board, ran into the wreck of the *Tricolor* at full speed.. The *Vicky* sustained serious damage, as did the wreck of the *Tricolor*. The incident resulted in a considerable number of oiled birds, most probably fouled with oil originating from the *Vicky* (Kerckhof *et al.*, in press). Also, during the salvage operations, chronic pollution occurred in the vicinity of the shipwreck for the major part of 2003. Especially after an incident on 22 January 2003, during salvage works, the consequences for seabirds became especially apparent. Although the amount of hydrocarbons released was relatively small in comparison with those released during incidents involving tankers, such as the *Erika* and the *Prestige*, the consequences for the seabirds wintering off the coasts of Northern France, Belgium and The Netherlands proved to be devastating. Many thousands of oiled seabirds washed ashore (Kerckhof *et al.*, in press).

The Management Unit of the North Sea Mathematical Models (MUMM), a department of the Royal Belgian Institute of Natural Sciences (RBINS), is the responsible administration for marine environmental matters in Belgium. It had the task of continuously assessing the environmental impact of the incident. During the *Tricolor* incident, frequent pollution control flights were conducted. Observations of oil pollution were reported to relevant authorities (Coastguard, Flemish Community, coastal Province, French authorities) on a regular basis, and were fed into the mathematical models run by MUMM. Mathematical models describing the movement, spreading, and physical and chemical development of pollutants, particularly hydrocarbons were used, to determine the potential impact of pollution, and to provide support for decisions to be taken in pollution control/combating operations.

During the incident, MUMM, together with other authorities such as the Coastguard, the Flemish Community and local authorities, took samples of the oil on beaches, on birds, at sea, and from the wreck itself. The French authorities provided reference samples of oil from the bunkers of the *Tricolor*, and also reference samples of oil from the *Vicky* and the *Prestige*. These samples were taken to MUMM's laboratory in Oostende for comparative analysis. Given the risks posed for human consumption, fish and shrimp were sampled in the immediate vicinity of the wreck and at reference locations by the Sea Fisheries Department (Cooreman & Raemaekers, 2003). PAH

concentrations were low to undetectable in almost all samples and the risks for human consumption were thought to be negligible. MUMM also has a routine monitoring programme for PAHs both in sediment and mussels (*Mytilus edulis*) along the coast in collaboration with the Sea Fisheries Department. The effects of the *Tricolor* incident will be monitored as part of this routine programme. Although the final conclusions are not available as yet, it does seem that PAH concentrations in sediments along the Belgian coast are not elevated as a result of the disaster. The same appears to be true for the concentrations in mussels (Roose, unpublished results).

Exxon Valdez

Of the oil spills reviewed in this paper, it was the *Exxon Valdez* spill of 1989 which was the most studied in terms of long-term impacts. Unfortunately, obtaining a consensus regarding these impacts has been difficult to achieve. From 1989 to 1991, the results of government-funded studies were considered litigation sensitive, and this polarized the scientific studies being conducted by the federal and Alaska governments and those funded by Exxon (Morris, 1996).

The *Exxon Valdez* spilled approximately 36,000 tonnes of Alaska North Slope crude oil into Prince William Sound, Alaska following the grounding of the tanker. Sublethal and chronic effects were investigated with respect to egg mortality and larval growth of pink salmon; reproductive success, larval deformations, genetic aberrations, and disease of Pacific herring; histopathological alterations of pink salmon, herring, Dolly Varden, cutthroat trout, and sea otters; juvenile mortality and blood chemistry of sea otters; dietary change and acute-phase proteins in the blood of river otters; reproductive success and blood chemistry of bald eagles; reproductive success of pigeon guillemots and black-legged kittiwakes; and growth of, and byssal thread production by, mussels (Spies *et al.*, 1996).

As examples of the difficulties in ascribing long-term effects unequivocally to the oil spilled by the *Exxon Valdez*, we can examine some of these studies and the differing assessments made of them. The effects of oil on pink salmon eggs incubating in streams in Prince William Sound after the spawning season in 1989 were investigated. Higher embryo mortality was observed in oiled rather than non-oiled streams (Craig *et al.*, 1995). Higher mortality was also reported to continue through 1993 (Bue *et al.*, 1996) and to reappear in 1997, well after measurable oil disappeared from the streams, and it was concluded that the reported mortality represented long-term injury to pink salmon in Prince William Sound (*Exxon Valdez* Oil Spill Trustee Council, 2000). A recent re-analysis of the data gathered in these studies has concluded that sampling shock (an artefact of the sampling regime) was a major source of embryo mortality in these studies, and that that source of mortality would have been interpreted as an oiling effect in the original studies (Brannon *et al.*, 2001). Studies of the size of runs of pink salmon back into Prince William Sound suggested that significant numbers of fish (60,000 – 1.9M) failed to return in each of the years 1990, 1991 and 1992 as a result of embryo mortality, but assessing the significance of these estimates was hampered by the high natural year-to-year variability observed in the absence of oiling (Geiger *et al.*, 1996).

Following record harvests of Pacific herring (*Clupea pallasii*) in Prince William Sound, Alaska, in the 3 years after the spill (1990-1992), the fishery failed in 1993. The hypotheses advanced to explain the dramatic decline fell into 3 categories:

- 1 those suggesting a causal connection with the 1989 oil spill
- 2 those suggesting harvesting effects
- 3 those attributing the decline to natural phenomena

(Pearson *et al.*, 1999). Following a detailed examination of each of these hypotheses and the evidence gathered in a broad range of studies, the authors concluded that the evidence was most consistent with the hypothesis that the 1993 herring decline resulted from a combination of natural factors, increasing herring biomass and decreasing food supply, leading to poor nutritional status.

The effects of an oil spill on biota are often obscured by hydrocarbons and PAH arising from confounding sources other than the oil spill. The outcome is that longer-term effects which may result from the spilled oil become difficult to distinguish from those due to other sources with much statistical confidence (Short *et al.*, 1999). This is particularly the case as concentrations of hydrocarbons and PAH in the affected environment fall towards the pre-spill “background” present as a result of historic and chronic, ongoing and often diffuse contamination. Although sophisticated fingerprinting techniques can be deployed in an effort to resolve this problem and have been applied in Alaska (eg. Boehm *et al.*, 2001 ; Hostettler *et al.*, 1999 ; Mudge, 2001; Page *et al.*, 2004) it is much more difficult to separate the effects that compounds from these differing sources are having in biological systems. As in other post-spill studies, mussels proved to be ideal organisms for monitoring the levels of PAH in the local environment due to oil deposited in shoreline sediments (Page *et al.*, 2005).

Conclusions

To directly address the question posed by OSPAR “whether the current framework of environmental risk assessment and toxicology can adequately address the issues of long-term impacts of oil spills” the answer is “no”. Consideration of the impact assessments conducted following recent oil spill incidents in Europe has demonstrated a range of short-term effects, but studies have not been continued for more than a few years. In most cases, the assessment of both short- and long-term impacts of spilled oil is hampered by a lack of pre-spill data on marine resources, and of the pre-existing levels of hydrocarbon and PAH contamination (Shaw & Bader, 1996). The assessment of long-term impacts is further compromised by the difficulty of unambiguously assigning effects observed to the spilled oil rather than to either hydrocarbons and PAH deriving from other local sources and chronic inputs, or to other chemicals present in the local environment. High levels of natural variability in biological systems add further difficulties. As can be seen from the studies reported here, many investigative techniques can be effectively applied in oil spill impact assessment. It would be useful to develop a framework for this process, within which comparable methodologies could be applied to monitoring the most vulnerable and sensitive components of the ecosystem (Wells *et al.*, 1995) and the foodchain. As has been suggested by others (Peterson *et al.*, 2003), the development of an ecosystem-based toxicology is required if we are to be able to understand the chronic, delayed and indirect long-term risks and impacts of oil spills. This is in line with the holistic ecosystem-based approaches to monitoring and assessment currently being developed within ICES.

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