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State-of-art in biological monitoring - Literature review report on the monitoring of biological effects of oil spills and oil spill responses in the northern Atlantic and Baltic Sea

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Executive Summary

The GRACE project is targeted at developing oil spill response methods in cold environments and examining the ecological risks of these spills and the subsequent mitigation activities. As an integral part of the project work, compiling previous information available on these topics is an important requirement for the successful execution of the project, and interpretation of the results obtained during its course.

This deliverable provides a state-of-the-art review on the monitoring of the biological effects of oil spills and oil spill responses in cold and temperate seas. It aims at providing a general view of the results, concepts and strategies concerning past major oil spills both in temperate and cold seas. The compilation will be useful to understand the biological effects of oil spills and oil spill responses in cold environments and to develop ad hoc oil response methods, including mitigation activities, impact assessments, and pre- and post-spill monitoring strategies.

The vast majority of the reported large oil spills have occurred in temperate seas, especially in the Northern Atlantic Ocean; this provides an invaluable reference to understand oil spills and oil spill responses and their biological effects in cold seas of the Arctic Ocean, the North Atlantic, and also the brackish-water, semi-enclosed Baltic Sea.

The review shows that oil spills of highly different magnitude have occurred around the globe and their environmental effects are variable, depending on a large variety of factors. Thus, no simple answers can be given when asked about the effects of oil spills on marine and coastal ecosystems. Some obvious items arise, among them the need for chemical and biological baseline data. It is practically impossible to distinguish the effects of oils spills if adequate pre-spill environmental data is missing from the impacted areas. The sparse application of biological effects methods, including biomarkers, in marine monitoring and assessment is sadly reflected in studies of the exposure and effects of oil spills on organisms. Early-warning biomarkers have prognostic power for effects taking place at higher biological levels. Although significant progress in the application of biological effects methods has been seen during the past couple of decades, a general lack of readiness to use these methods exists in many countries, while regular monitoring campaigns, which produce also the much-needed baseline data, are far too sparse and also limited in regard to the number of parameters measured.

Most of the major accidents have occurred in temperate sea regions, which means that also the oil response actions taken aboard as well as our knowledge of the response of the ecosystem to the spill is largely originating from these areas. Thus, comparing these with the cold seas characterised by markedly different physical conditions and biology is obviously quite difficult. The new information generated within the GRACE project must be therefore seen important and also very timely, taking into account the ongoing and foreseen increases in oil transport in northern sea areas.

Monitoring and assessment of biological effects of oil spills and oil spill responses in temperate and cold seas: a review of past major incidents, actions and observations

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1. Introduction

The GRACE project is targeted at developing oil spill response methods in cold environments and examining the ecological risks of these spills and the subsequent mitigation activities. As an integral part of the project work, compiling previous information available on these topics is an important requirement for the successful execution of the project, and interpretation of the results obtained during its course.

Accordingly, during an early phase of the project, GRACE project partners have produced the present literature review report; this includes a reference list (Annex) with more *than 1500 titles related to:*

- monitoring of pollution and biological effects in North Sea and Baltic Sea (and other polar seas);
- monitoring of oil spills and oil spill responses; and
- ecology and biology of polar seas.

Together with the reference list, which will be continuously updated and completed during the course of the project, we have produced a state-of-the-art review on the monitoring of the biological effects of oil spills and oil spill responses in cold and temperate seas. This compilation of main results, concepts, and strategies will be useful to understand the biological effects of oil spills and oil spill responses in cold environments and to develop ad hoc oil response methods, including mitigation activities, impact assessments, and pre- and post-spill monitoring strategies.

A brief summary of the "Top 20" major oil spills that have occurred since the *Torrey Canyon* accident in 1967 is shown in Table 1; 19 of them occurred before the year 2000. Indeed, the average number of tanker incidents involving large oil spills has progressively reduced during the last decades despite an overall increase in oil trading and transport since the mid-1980s. Nearly 50% of the large spills took place when the vessels were sailing in open water, especially during seasons characterised by heavy weather and rough sea conditions (*e.g.*, winter storms in December-February in the North Atlantic or tropical rainstorms in July-August in tropical regions).

Response to oil spills has been diverse throughout history (Table 2). The main actions observed are different in each of the events analysed. Depending on prevailing environmental conditions during the spill (weather and sea) and the properties of the crude oil spilled (heavy or light), the responses used are different and more or less effective. The vast majority of the reported large oil spills have occurred in temperate seas, especially in the Northern Atlantic Ocean; this provides us with an invaluable reference to understand oil spills and oil spill responses and their biological effects in cold seas of the Arctic Ocean (*e.g.* Norwegian Sea), the North Atlantic (including the North Sea), and also the brackish-water, semi-enclosed Baltic Sea although in the latter area the amount of spilled oil has at its worst been one to two orders of magnitude smaller than that in the other areas. This review aims at providing a general view of the results, concepts and strategies concerning past major oil spills both in temperate and cold seas.

Table 1. Top 20 list of major oil spills since the *Torrey Canyon* accident in 1967. Modified from ITOPF (2016).

RANK	SHIP NAME	YEAR	LOCATION	SPILL SIZE (TM)
1	*Atlantic Empress*	1979	Off Tobago, West Indies	287000
2	*ABT Summer*	1991	Off Angola	260000
3	*Castillo De Bellver*	1983	Off Saldanha Bay, South Africa	252000
4	*Amoco Cadiz*	1978	Off Brittany, France	223000
5	*Haven*	1991	Genoa, Italy	144000
6	*Odyssey*	1988	Off Nova Scotia, Canada	132000
7	*Torrey Canyon*	1967	Scilly Isles, UK	119000
8	*Sea Star*	1972	Gulf of Oman	115000
9	*Irenes Serenade*	1980	Navarino Bay, Greece	100000
10	*Urquiola*	1976	La Coruna, Spain	100000
11	*Hawaiian Patriot*	1977	Off Honolulu	95000
12	*Independenta*	1979	Bosphorus, Turkey	94000
13	*Jacob Maersk*	1975	Oporto, Portugal	88000
14	*Braer*	1993	Shetland Islands, UK	85000
15	*Aegean Sea*	1992	La Coruna, Spain	74000
16	*Sea Empress*	1996	Milford Haven, UK	72000
17	*Khark 5*	1989	Off the Atlantic coast of Morocco	70000
18	*Nova*	1985	Off Kharg Island, Gulf of Iran	70000
19	*Katina P*	1992	Off Maputo, Mozambique	67000
20	*Prestige*	2002	Off Galicia, Spain	63000

Table 2. Oil spills and oil response actions in the North Atlantic Ocean (modified after ITOPF 2016)

SHIP NAME	YEAR	LOCATION	CRUDE OIL TYPE	CARGO TM	SPILL SIZE TM	RESPONSES
Torrey Canyon	1967	Scilly isles, UK	Kuwait	119000	119000	-Dispersants -Bombed (burned)
Jakob Maersk	1975	Porto, Portugal	Iranian light	88000	88000	-Accidental burning -Booms -Ineffective dispersant application (no waves) -Manual-mechanical removal

Urquiola	1976	A Coruña, Spain	Arabian light	100000	100000 25000 (shore)	-Dispersants (2000 Tm) -Mechanical removal -Accidental burning
Ekofisk[1]	1977	Norway	Ekofisk	Platform	27600	-Natural dispersion
Amoco Cadiz	1978	Brittany, France	Arabian and Iranian light	223000	223000	-Dispersants (300 Tm) -Chalk application (sinking) -Mechanical-hand removal -Ineffective skimmers -Fisheries/aquaculture banned
Aegean Sea	1992	A Coruña, Spain	North Sea Brent	80000	74000	-Accidental burning -Trespass -Mechanical-hand removal -Fisheries stopped
Braer	1993	Shetland islands, Scotland	Norwegian Gullfak	85000	85000	-Dispersants (130 Tm)/ fail -Natural dispersion -Aquaculture banned
Sea Empress	1996	Pembrokeshire, Wales	Forties Blend North Sea	130000	72000	-Dispersant -Trespass -Mechanical-manual removal -Protective booms -Fisheries banned
Erika	1999	Brittany, France	Heavy fuel Type II	31000	15000-25000	-Mechanical-manual removal at shore -Trespass -Protective booms -Ineffective skimmers
Prestige	2002	Spain	Heavy fuel Type VI	77000	63000	-Mechanical-manual removal at shore and sea -Trespass -Protective booms -Fisheries banned

2. Temperate seas

2.1. North Atlantic Ocean

The Northeast Atlantic Ocean has suffered many of the most significant oil spills in terms of environmental relevance (Figure 1). These have taken place especially in the Bay of Biscay and the Iberian Coast and the Celtic Seas, and the North Sea (Figure 2; ITOPT, 2016). The present section covers information on the major oil spills that have occurred in this region up to this day.

[1] https://incidentnews.noaa.gov/incident/6237

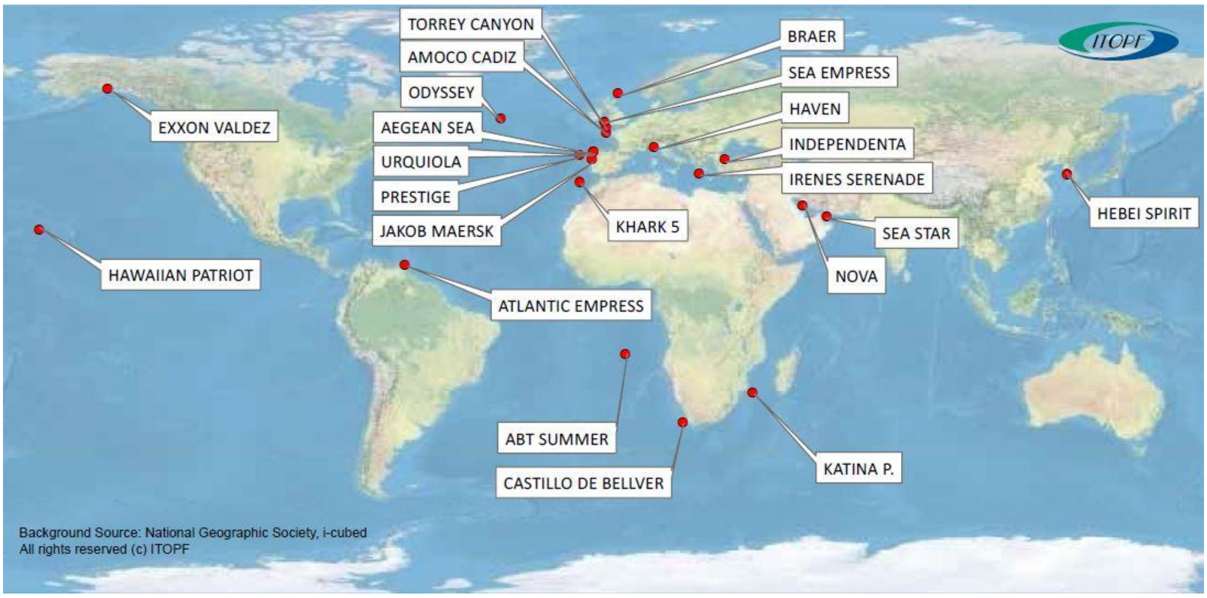

Fig. 1. Global location of past major oil spills (source: ITOPF 2016)

Oil spills in the North Sea. Today, offshore oil and gas industry are the main sources of oil entering the North Sea (Carpenter 2016). Relatively dense drilling activities in the central North Sea are carried out between the northeast UK and southwest Norway and extend into the southern North Sea between the UK and Dutch sectors. The number of offshore oil and gas installations in the region increased during the last decades whilst the quantities of oil discharged in waters declined markedly. In respect of oil from shipping, there had been an increase in the levels of airborne surveillance conducted under the Bonn Agreement, which has resulted in reducing oil releases. In early accidental oils spills in the region, oil response was minimal and the biological effects were not determined. For instance, in 1977 oil release from the Bravo Ekofisk oil platform (North Sea) after an oil and natural gas blowout was declared by the Norwegian State Pollution Control Board to result in no major ecological damage due to the weather and water conditions existing in the area in those particular moments, which contributed to the evaporation and dispersion of the oil[2]. Likewise, oils response and biological effects assessment were not carried out in high seas oil spills such as the Odyssey (1988) and Kharks-5 (1989) oil spills, even though important amounts of oil were spilled; this was probably due to the dispersion of the spilled oil[3].

Torrey Canyon (1967). The first relevant spill, and world's first major supertanker disaster, occurred 50 years ago when the *SS Torrey Canyon* released about 120000 Tm of Kuwait crude oil off the Cornwall coast in the English channel, killing more than 25000 seabirds and other marine organisms and coating beaches in southern England, Channel Islands and north-western France.

The spill was treated with different methods (burning, bombing, physical removal from the shoreline) including the use of dispersants and detergents, which proved to be of limited effectiveness and caused considerable ecological damage, showing for the first time the ecological risks of using dispersants to clean up coastal oil spills (Nelson-Smith, 1968; Wells,

2 https://incidentnews.noaa.gov/incident/6237
3 http://wwz.cedre.fr/en/Ourresources/Spills/Spills/Odyssey; http://wwz.cedre.fr/en/Our-resources/Spills/Spills/Khark-5

2017). The harm produced by these chemicals was also demonstrated in bioassays performed with the barnacle *Elminius modestus* and the same crude oil type and dispersants used in the *Torrey Canyon* event (Corner et al., 1968). Although few studies were performed in relation to the biological effects produced by the oil spill and the dispersants employed in the clean-up, and the recovery of biodiversity (Soudward and Soudward, 1978), the *Torrey Canyon* case has been recently considered as the trigger of the initiation of scientific studies of long-term monitoring the fate and effects of oil spills in the sea, especially for organisms and habitats in intertidal and shallow sub-tidal zones (Wells, 2017).

Jakob Maersk (1975). The tanker *Jakob Maersk*, which was carrying approximately 88000 tonnes of oil comprising mostly of Iranian light crude oil cargo, grounded and exploded at the entrance to the Leixoes Harbour in Porto (Portugal). It was estimated that 40000-50000 Tm of oil were burned, 25000 Tm were dispersed and nearly 15000 Tm were washed up on the beach. A floating boom was installed in the harbour entrance to prevent oil slicks from entering the harbour. Dispersant was sprayed from vessels but it was considered largely ineffective due to minimal mixing as a result of calm seas. Both manual and mechanical shoreline clean up were undertaken with local inhabitants, the army, navy and fire service cleaning some areas. Heavily oiled sand was removed using bulldozers. More lightly oiled sand was pushed into the sea and treated with dispersants. The most affected beach was the shore immediately adjacent to the *Jakob Maersk*.

Hydrocarbon traces were found on beaches situated 50 km from the wreck; however, few impacts on the biota were reported. Very few birds were observed to have been affected as a result of the spill and local fisheries were relatively unaffected. The majority of the ecological damage was observed on the in rocky areas found immediately adjacent to the vessel, where significant mortalities of seaweed and molluscs were recorded but fortunately, they recovered a few months later[4]. Very few scientific studies related with the oil spill have been found and they only included sediment and water analysis (Canelas & Calejo Monteiro 1977). Although the authors of the report stated the lack of catastrophic consequences when compared to other similar oil spills and the need of long-term studies to estimate the possible influence of the different fractions of oil, as well as the dispersants and detergents used on biotic communities, no further publications related to this event have been found in scientific literature.

Urquiola (1976). In A Coruña (Spain) harbour, the supertanker *Urquiola* grounded, exploded and burned while transporting 100000 Tm of Persian Gulf crude oil. Although most of the oil burned, it is estimated that about 25000-30000 Tm washed ashore, affecting approximately 215 km of shoreline. Dispersants were used but then abandoned because they promoted the infiltration of the oil into the sediments (Gundlach et al., 1978). Shoreline clean-up efforts with limited mechanical support were undertaken although problems arose with secondary oiling, with mechanical equipment churning the oil deeper into the sand and the large tidal range in the area[5].

Land-based clean up and control methods resulted to be inadequate to combat the spreading of oil, and ineffective in preventing large scale environmental damage (Gundlach and Hayes, 1977). Although only limited impacts on bird and fish were reported, local shellfish

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[4] http://wwz.cedre.fr/en/Our-resources/Spills/Spills/Jakob-Maersk; http://www.itopf.com/in-action/case-studies/case-study/jakob-maersk-leixoes-portugal-1975/

[5] http://wwz.cedre.fr/en/Our-resources/Spills/Spills/Urquiola

stocks (clams, mussels and oysters) were significantly affected by the spill, taking several years for the affected biota to regain normal growth patterns[6]. Despite the above-mentioned environmental damage, only a report about the a study on the accumulation of crude oils in sediments has been found to be published (Gundlach et al, 1978).

Amoco Cadiz (1978). Over a period of two weeks, the tanker *Amoco Cadiz* released her entire cargo of 223000 Tm of light Iranian and Arabian crude oil offshore the coastline of Brittany (France). Much of the oil quickly formed a viscous water-in-oil emulsion, increasing the volume of the polluting water mass by up to five times. Strong winds and heavy seas prevented an effective offshore recovery operation.

Numerous lessons were learned from the *Amoco Cadiz* incident concerning clean-up and ecological impacts, and it remains one of the most comprehensively studied oil spills in history. Dispersants were applied and chalk was used as a sinking agent, with the consequence of transferring a part of the problem from the water column and shoreline to the sea bed. The at-sea response did little to reduce shoreline oiling; the oil and emulsion contaminated 320 km of the Brittany coastline and extended as far east as the Channel Islands[7]. Removal of bulk free oil was achieved using vacuum trucks and agricultural vacuum units and by hand, by a personnel of more than 7000 (mainly military). Clean-up activities on rocky shores caused biological impacts. Whilst rocky shores recovered relatively quickly, for the salt marshes it took many years. The failure to remove oil from the temporary oil collection pits on some soft sediment shorelines before inundation by the incoming tide also resulted in longer-term contamination.

The *Amoco Cadiz* oil spill became the largest loss of marine life ever recorded after an oil spill with millions of dead molluscs, sea urchins and other benthic species washed ashore, nearly 20,000 dead birds and oyster cultivation, seaweed gathering and other shell and fin fisheries seriously affected[8]. Five years later, the local biota was still far from to be completely recovered (Seip, 1984). By November, a special edition about the *Amoco Cadiz* oil spill was published in Marine Pollution Bulletin, dealing with topics such as lines of study and early observations, chemical analysis of the petroleum hydrocarbons, sediments and toxicity, analysis of zooplankton and sea bed, ecological impacts and birds among others (for more details, see Spooner 1978). In addition, long-term monitoring studies in oysters, that included histopathological analyses and chemical measurements (Berthou et al, 1987), and especially impacts on benthic communities were also carried out (Cabioch et al, 1982; Boucher, 1985; Dauvin, 1998; 2000; Gómez Gesteira et al, 2003), in the latter case promoted by the fact that chalk was used as a sinking agent in the responses.

Aegean Sea (1992). In December, the OBO carrier *Aegean Sea* was carrying 80000 Tm of Brent crude oil when it ran aground off the port of A Coruña, broke in two and burst into flames. About 74000 tonnes spilled out but a large part burned in the fire or was dispersed in the sea. Winds and sea-currents drove the petrogenic hydrocarbons and pyrolytic products from the burned oil offshore towards the north and east. More than 300 km of shoreline was affected. In order to fight pollution, large clean-up operations were set up both on and off

[6] http://www.itopf.com/in-action/case-studies/case-study/urquinola-apin-1976/;http://wwz.cedre.fr/en/Our-resources/Spills/Spills/Urquiola

[7] http://wwz.cedre.fr/en/Our-resources/Spills/Spills/Amoco-Cadiz

[8] http://www.itopf.com/in-action/case-studies/case-study/amoco-cadiz-france-1978; http://wwz.cedre.fr/en/Our-resources/Spills/Spills/Amoco-Cadiz

shore. Floating booms were used to collect some 5000 Tm of oil/water mix, which was then treated. Operations also involved manual cleaning of the shoreline, in which some 1200 m³ of sand and polluted debris were collected and then burned at a local ceramics factory.

A variety of commercially important species, including mussels (*Mytilus edulis*), were tainted, so bans on fishing and the sale of all seafood from the area were imposed[9]. Soon after the accident a monitoring program was established by the Spanish authorities in order to evaluate the presence of petrogenic and combustion compounds in sediments and water (Pastor et al, 2001) and the spatial and temporal evolution of petrogenic and combustion compounds in bivalves mussels, clams (*Tapes semidecussata*), cockles (*Cardium edule*), and oysters (*Ostrea edulis*) from several locations along the coast and inside the estuarine areas where bivalves are commercially produced and oil was less exposed to weathering (Porte et al, 2000). Six months after the accident, biological effects measurements using biomarkers such as ethoxyresorufin-O-deethylase (EROD), DNA damage and oxidative damage were included in the monitoring program (Solé et al, 1996; Porte et al, 1996). In addition, the macrobenthic sandy community was also monitored in order to identify any short-term effects of the oil spill since large quantities of oil might have been washed ashore, with possible contamination of the subtidal sediments (Gómez Gesteira et al, 2003; Gómez Gesteira and Dauvin, 2005). For this purpose, amphipods were used as bioindicators since during the *Amoco Cadiz* oil spill they were established as good bioindicators (Gómez Gesteira and Dauvin, 2000).

Braer (1993). In January, the tanker *Braer*, ran aground in the Shetland Islands (United Kingdom). During 12 days, the entire cargo of 85000 Tm of Norwegian Gullfaks crude oil became released. Adverse weather conditions, with exceptionally strong winds and wave energy, prevented response operations at sea, although about 130 tonnes of a chemical dispersant was applied from aircraft during periods when the wind abated slightly and some oil remained on the surface. At the same time, a combination of the light nature of the oil and the mentioned weather conditions naturally dispersed the oil throughout the water column and no surface slick was produced[10]. Oiling of shorelines was minimal relative to the size of the spill and clean-up involved the collection of oily debris and seaweed by a small workforce. A significant amount of airborne oil was blown on to land adjacent to the wreck site.

A wide range of fish and shellfish over a large area became contaminated with oil, resulting in the imposition of a Fisheries Exclusion Zone. Farmed Atlantic salmon held in sea cages in the surface waters within this zone were especially affected since they could not escape the cloud of dispersed oil (Whittle et al, 1997). Taking advantage of the situation, temporal changes in the levels of CYP1A in the liver of caged salmon in response to dispersed oil in the water column were measured (Stagg et al., 1998). In addition, supporting laboratory experiments investigating the selectivity and dose dependence of salmon liver CYP1A to selected PAHs were also undertaken by measurements of CYP1A mRNA, protein and catalytic activity. They found out that measurements of CYP1A as mRNA, protein or EROD activity provide comparable results in field or experimental exposures and CYP1A levels also correlate well with exposure to oil (Stagg et al., 2000). This research revealed the importance of biomarkers to better understand the biological effect of accidental oil spills in the

[9] http://wwz.cedre.fr/en/Our-resources/Spills/Spills/Aegean-Sea; http://www.itopf.com/in-action/case-studies/case-study/aegan-sea-spain-1992
[10] http://wwz.cedre.fr/en/Our-resources/Spills/Spills/Braer; http://www.itopf.com/in-action/case-studies/case-study/braer-uk-1993/

environment. Moreover, mussels from a reference area were transplanted to different impacted areas in order to monitor the long-term hydrocarbon pollution within the exclusion zone, analysing PAHs at regular intervals (Webster et al., 1997). The Exclusion Zone was progressively lifted as fish and shellfish species were found to be free of contamination by chemical analysis and taste testing. The importance of establishing background data in the marine environment to ensure that relevant information is available for decision makers facing a marine incident such as an oil spill became clearly highlighted.

Although no surface slick was produced, oil droplets were adsorbed onto sediment particles, which eventually sank to the seabed. Sub-surface currents led to this oil being spread over a very wide area. Sediment samples were screened for PAHs, and it was estimated that 30000 tonnes were deposited in sediments with a significant portion eventually ending up in two deep, fine sediment basins. Long-term temporal monitoring programs were established to study the fate of oil in these basins (Davies et al., 1997). In addition, biological effects on the flatfish *Limanda limanda* from these two basins were monitored measuring EROD activity in liver tissue (Stagg et al., 1998). Oiling of shorelines was minimal considering the size of the spill but a significant amount of oil was blown on to land adjacent to the wreck site. The effects of this airborne oil were localised and had no more than a temporary impact on vegetation and livestock (Wolff et al., 1993). Seabird casualties were also relatively low but important population level effects were recorder in the Blacklegged Kittiwake (*Rissa tridactyla*) with evidence of low breeding rates (Walton et al., 1997). Considering the size of the spill, the observed environmental impacts were surprisingly limited.

Sea Empress (1996). In February, the tanker *Sea Empress* ran aground at the entrance to Milford Haven in Wales while transporting 131000 Tm of Forties blend crude oil. During a week, the vessel spilled out 72000 Tm and it is estimated that 15000 Tm of emulsified oil came ashore along 200 km of coastline[11].

The clean-up operations were wide-ranging and effective. At sea, these included dispersant spraying, mechanical recovery and the use of protective booms, which greatly reduced the quantity of oil reaching inshore waters. The Marine Pollution Control Unit sent DC-3 aeroplanes with dispersant spraying equipment and Oil Spill Response Ltd sent trailers with shore response equipment. However, the prime objective was still to keep the vessel afloat and transfer the cargo as quickly as possible. The tanker was towed overnight to a disused oil wharf, berthed and encircled with floating containment booms. The *Sea Empress* incident is considered an example of an efficient use of targeted dispersants in responding to an oil spill (Lunel et al. 1996). Some 200 km of coastline (much of it in a national park) was contaminated and a major shoreline clean-up effort was mounted, including mechanical and manual recovery, trenching, beach washing, and the use of dispersants and sorbents. A temporary ban was imposed on commercial and recreational fishing in the region and there was concern that tourism, important to the local economy, would be badly affected by the heavily oiled beaches. Several thousand oiled birds washed ashore, leading to a major cleaning and rehabilitation operation.

As a landmark in oil spill response and impact assessment, UK government created an independent committee for evaluating the environmental effects of the *Sea Empress* oil spill (Edwards and White, 1999). The impacts on a national park, the characteristics of the fuel oil

[11] http://wwz.cedre.fr/en/Our-resources/Spills/Spills/Sea-Empress; http://www.itopf.com/in-action/case-studies/case-study/sea-empress-milford-haven-wales-uk-1996/

(heavy), and probably also the social concern and pressure were determinants for the monitoring of the biological effects of this oil spill. Several field and laboratory studies were performed to assess biological effects, including biomonitoring and biomarkers. A large amount of marine species, vertebrates and invertebrates, were used as bioindicators and/or sentinel species (Law and Kelly, 2004). For instance, contamination by hydrocarbons including PAH compounds persisted longest in tissues of mussels from the intertidal area to the east of Milford Haven (Freshwater West to Pendine Sands), deriving from bulk oil contamination of the shoreline (Law and Kelly, 2004). Besides the mortalities recorded in birds and other vertebrates like fish, biological effects were described also at sublethal levels using the biomarker approach. The impact of the spill was determined to be much less than expected from the quantity of oil spilled (Law and Kelly, 2004). Limpets were used as bioindicator species (Crump et al, 1999; Glegg et al, 2000). Biomarkers such lysosomal alterations and cytochrome P450 were applied in mussels (Peters et al., 1999, Fernley et al, 2000), and EROD activity in flatfish (Kirby et al., 1999). Plankton communities were apparently not markedly disturbed (Batten et al., 1998).

Erika (1999). In December, the Maltese tanker, *Erika*, laden with 31000 Tm of heavy fuel oil n°6 was faced with structural problems off the Bay of Biscay and split in two in international waters, about thirty miles south of Southern Brittany (France). The two parts of the wreck ended up 10 km apart from each other, at the depth of 120 m. The quantity of oil spilled at that time was estimated between 7000 and 10000 tonnes. The island of Groix, opposite Lorient, was severely affected while the bulk of the oil pollution reached the north and south banks of the Loire River as viscous oil layers (5-30 cm thick). The Polmar Sea Plan was implemented.

Shellfish farming sites were seen to be under critical threat. The French Agency for Food Safety (AFSSA) recommended the shellfish production or harvest zones to be closed and follow-up of shellfish contamination. In Finistère, the levels of oil contamination seldom exceeded the reference value of 0.5 ppm fixed by the AFSSA. Bird protection was seen crucial as well. The *Erika* oil spill had the greatest impact on seabirds ever recorded due to an oil spill. Bird protection associations took care of the operations on saving the oiled birds, an aspect of the spill little accounted for in the Polmar Plans.

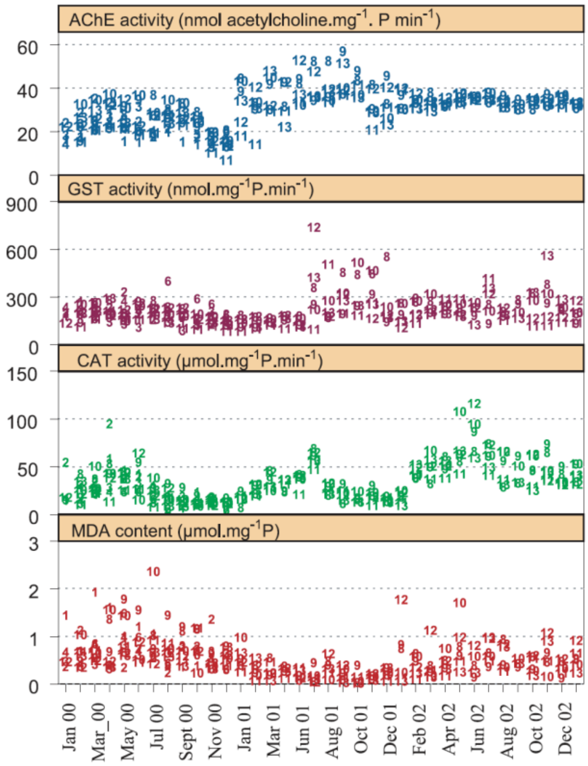


Fig. 2. The *Erika* oil spill. Levels of acetylcholinesterase (AChE), catalase (CAT) and malonaldehyde (MDA) in the common mussel (*Mytilus edulis*) during the three-year survey (Jan 200-Dec 2002). All results measured at each site are plotted using the reference site. From Bocquené at al. (2002).

The French government implemented programmes focusing on the ecological and ecotoxicological consequences of the *Erika* oil. Contamination levels and biological effects were studied at all ecological levels, including (a) detailed determinations of PAH in water, sediments and biota, (b) monitoring of ecotoxicological biomarkers in mussels, and (c) evaluation of the effects on marine and coastal bird populations (Moigne and Laubier 2004). Chemical analyses were extensive, and the effectiveness of the oil spill response, cleaning activities and oil degradation were thoroughly investigated (Geffard et al. 2004, Bordenave et al 2004; Cedou 2004). Biomonitoring of biological effects was accomplished measuring a battery of biomarkers in Pacific oysters (*Crassostrea gigas*; Auffret et al. 2004) and mussels (Boquené et al. 2004). In parallel, several investigations dealt with human health and ecotoxicological risk assessment as well as with laboratory studies aimed at understanding, interpreting and predicting the environmental effects produced by the spill (Frederic et al. 2004, Geffart et al. 2004).

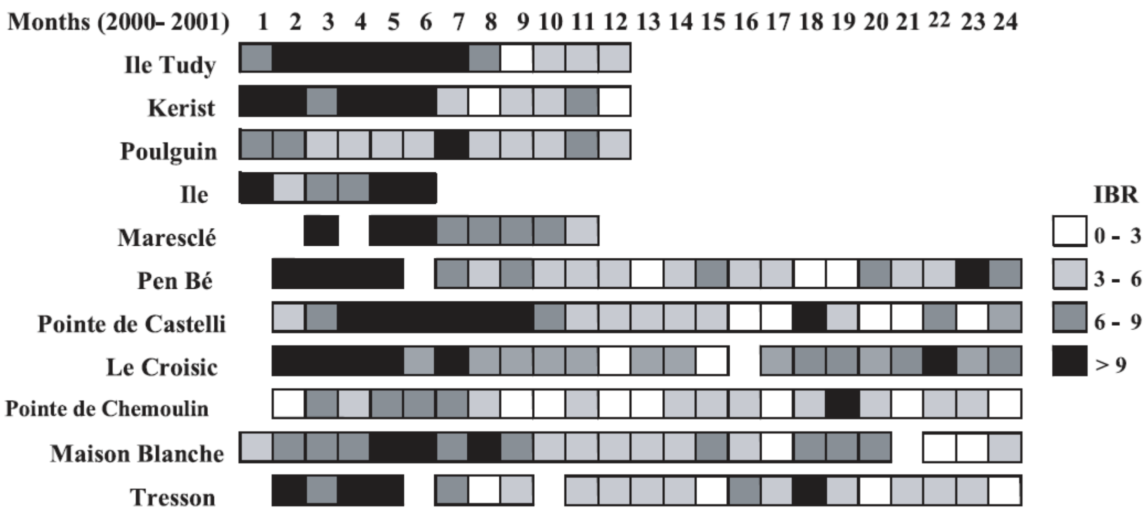


Fig. 3. The *Erika* oil spill. Variation with time of the integrated biomarker response index (IBR) in the common mussel (*Mytilus edulis*) from January 2000 to December 2001. A darker IBR indicates greater effects. From Bocquené at al. (2002).

Prestige (2002). The *Prestige* oil spill constituted a landmark in terms of the length of coastline affected by the spilled oil, the oil spill response, and the biological effects assessment. In November, the single-hulled oil tanker *Prestige* carrying 77000 Tm of heavy fuel oil broke in two about 130 nautical miles off the Spanish coast, west-southwest of Cape Finisterre, and sank into the depth of 3500 m. For the first time an oil spill contaminated shores of six countries, from Galicia in Spain to Aquitaine in France. The impact was heterogeneous with highly impacted and less impacted coastal areas, lasting at least all the year 2003. The oil was also significantly weathered during the occurrence of the spill. In all, it is estimated that some 63,000 tonnes came out from the tanker. Owing to its highly persistent nature, the released heavy fuel oil drifted for extended periods with winds and currents, travelling great distances. According to the report presented by the Spanish Government to the International Oil Pollution Compensation Funds, 70% of the Galician beaches was affected by the oil although significant amounts of the spilled oil did not enter the inner estuarine parts where commercial raft culture of mussels takes place.

Intermittent discharge of oil from the tanker, combined with large-scale sea surface dispersion, created a tracking and recovery problem. In Galicia, conventional oil recovery approaches were adopted close to the wreck and thousands of volunteers were organized to help to clean up the affected coastline. The massive cleaning campaign was a success, oil slicks from most portions of coastline being removed. Cleaning was completed through the use of dispersants and pressure water to detach oil from the rocky shore stones. With time and distance from the source, the oil dispersed dramatically and became less viscous. A unique monitoring, prediction and data dissemination system was established in the Cantabrian coast and the Basque Country, based upon the principles of `operational oceanography'; this utilised in situ tracked buoys and numerical modelling outputs, in combination with remote sensing. Overall, wind effects on the surface waters were found to be the most important mechanism controlling the smaller oil slick movements. The recovery operation involved up to 180 fishing boats (9-30 m in length). Such labour-intensive recovery of the oil (21000 Tm, representing an unprecedented ratio of 6.6 Tm at sea, per Tm recovered on land) continued over a 10 months period. The overall recovery at sea by the fishing vessels represented 63% of the total oil recovered at sea; this compares to only 37% recovered by specialised `counter-pollution' vessels. Recovery at the sea was highly successful and hence the cleaning of the Basque coastal areas was done only in locations that posed a potential risk for human health or in those that received massive amounts of oil. Moreover, dispersants or pressure water systems were only exceptionally used and for most coastal areas the fuel oil removal was left to occur naturally by wave action and monitored as a part of the Orbankosta action (inventory of coastal impact of the *Prestige* oil spill [POS] in the Basque coast, 2003-2004; Marigómez 2012). The oil spill was visibly extended throughout the Bay of Biscay offshore covering tremendously large areas in the open sea, and kept on arriving to the shore for more than one year. The shorelines were largely cleaned up manually. Fisheries exclusion zones were put in place in Galicia for one year shortly after the incident, banning virtually all fishing along about 90% of the coastline.

Since January 2003, Urgent Research Actions and the Scientific Research Strategic Plan were launched by the Spanish Ministry of Science and Technology,[12] designed and managed upon coordination by a Scientific Commission (CCC-VEM-MEC) nominated ad hoc. Urgent research actions were carried out to (a) inter-calibration among the laboratories in charge of the PAH analyses, (b) evaluate the initial impact on biological communities on coastal and continental shelf ecosystems, and (c) determine the geophysical characteristics of the zone where the ship sank. The Scientific Research Strategic Plan was organized in six main subject areas: (a) oil behaviour in the sunken tanks; (b) seismic risks for the wreck; (c) the fate of the oil in the environment; (d) biological effects; (e) socio-economic impact and (f) definition and implementation of contingency plans. As a result of both urgent actions and more extended research activities (2003-2008) over 200 international scientific publications were produced (Marigómez, 2012).

A tremendous research effort was addressed to determine *Prestige* oil spill impact on biological systems (Albaiges et al., 2006). The first impact was physical but, due to its chemical composition (heavy chain hydrocarbons), the fuel oil was not expected to cause major acute toxicity but mainly long-term or chronic effects (Soriano et al., 2007). As the released fuel oil was highly persistent, the effects were expected to remain or to be delayed for many years. Moreover, the impact was heterogeneous and of distinct nature depending on the diverse geographical areas affected.

The impact of the spill was investigated in a great variety of organisms (plankton, benthos, crustaceans, molluscs, fishes and birds), particularly in coastal sentinel species such as mussels. The impact was readily evident in several biological systems and complexity levels, and an overall recovery trend was often envisaged after one to two years after the accident. Nevertheless, a clear relationship between environmental levels of oil-related pollutants and the biological effects measured was rarely found; this was attributed to various confounding factors, including seasonal variability, global stress sources, different response times, adaptability of biological responses, incident of secondary effects that persist beyond abatement of the direct action of pollutants, and, importantly, to the absence of pre-spill baseline data on the measured parameters(Marigómez, 2012).

Most of the results obtained indicated a strong initial impact during the first two years after the spill, mainly on intertidal communities and fishing resources, with a recovery by 2004-2005. The number of physically impacted birds were estimated to be between 115000 and 230000. In addition, long term effects were expected since the levels of PAH in bird eggs were recorded to be elevated (Zabala et al., 2011). Subsequently, it was shown that health of

[12] http://otvm.uvigo.es

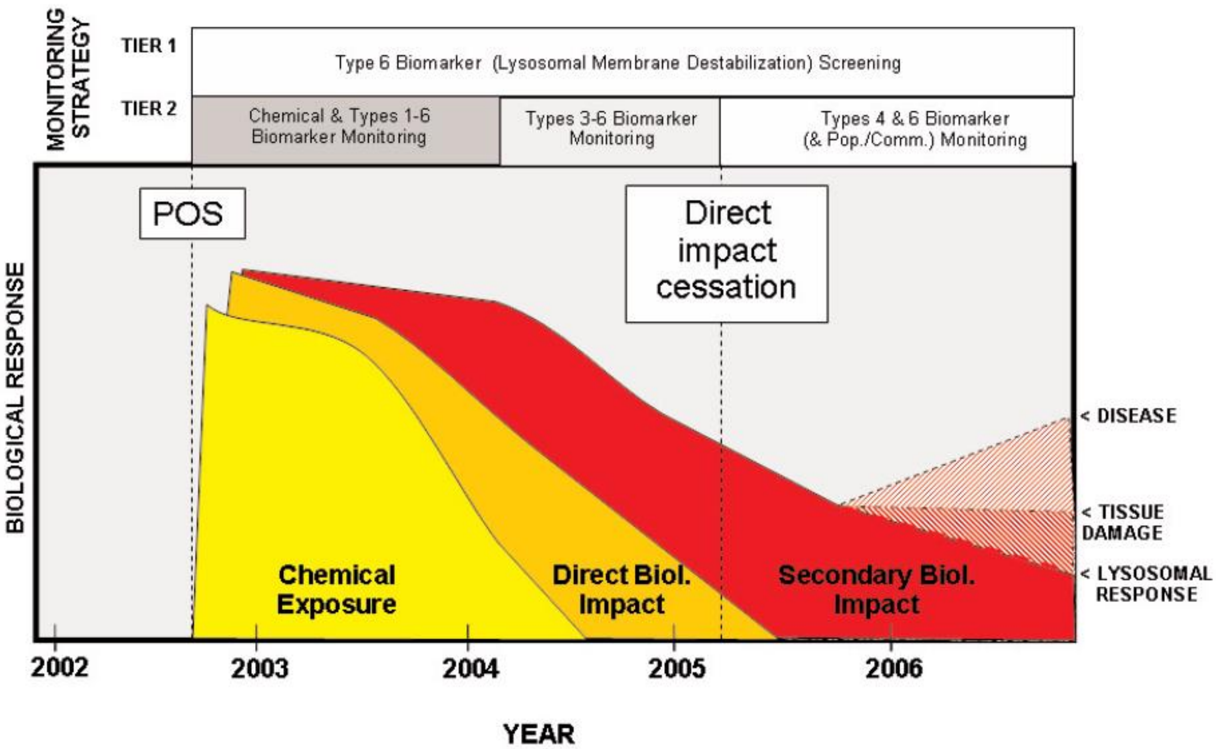
adult yellow-legged gulls breeding in oiled colonies was affected 17 months after the spill with the birds presenting high TPAH blood levels, physiological disorders, damage in liver and kidneys, and elevated hepatic EROD activity levels (Perez et al., 2008). Neurotoxicity was also investigated in common guillemot, razorbill, and Atlantic puffin, and an inhibitory effect on brain AChE activity was observed in the two first (Oropesa et al., 2007).

Coastal habitats, ranging from supralittoral, intertidal and sublittoral levels to oceanic and bathyal environments were heterogeneously affected by the spill (e.g. Serrano et al., 2006; Lobon et al., 2008; Puente et al., 2009; Veiga et al., 2009; 2010; Crego-Prieto et al., 2014; Junoy et al., 2014), alternating from intensely affected zones with most benthic organisms almost extinct at local scale to nearby sites with minimal disturbance. In benthic macroinvertebrate communities, anomalies in temporal trends in abundance were observed for some key species but these could not be related to the *Prestige* oil spill due to the high range of natural variability, confounding effects of other contamination sources, and the absence of previous reference conditions. Substantial alteration in marine food webs were concluded after investigating the follow-up trends in the composition and structure of the parasite communities in the marine sparid fish *Boops* (Perez del Olmo et al., 2007), and the radioisotopic signature of sea bird feathers (Moreno et al., 2013). However, no significant differences in phytoplankton biomass and primary production were detected all along the N-NW Iberian Peninsula during the first spring bloom after the spill, which could be due to the fact that the oil spill occurred in late autumn when plankton biomass is low (Varela et al., 2006).

The health condition of fish and mussels, commonly applied as sentinels indicative of ecosystem health status, was severely affected along 2003-2004 but some of the effects were transient or at least not fully attributed to the oil spill, mainly due to the lack of baseline data and the intricate nature of the parameter used. In addition, despite of all the solid evidence of alterations at cell, tissue and individual level recorded during months and years, most ecological surveys carried out at population or community levels did not show major alterations attributable to the oil spill.

Hepatic biomarker responses (EROD, glutathione *S*-transferase, glutathione reductase, catalase, and DNA integrity) were measured in two demersal fish species (*Lepidorhombus boscii* and *Callionymus lyra*) from the Northern Iberian shelf five months after the oil spill (Martínez-Gómez et al., 2006). Antioxidant activities were significantly elevated in *L. boscii* in the most oil impacted area (Finisterre in Galicia) and positively correlated with the *Prestige* oil spill tar aggregate densities (Martínez-Gómez et al., 2009). A significant decrease in biomarker responses (detoxification and antioxidant enzymes) was recorded 2-3 years after the spill, indicating a recovery to baseline levels. Sublethal adverse effects (increased EROD activities and histopathological lesions) in juvenile seabream (*Sparus aurata*) and Senegalese sole (*Solea senegalensis*) were also associated with increasing Prestige oil spill PAHs in sediments in ex situ bioassays (Morales-Caselles et al., 2006; 2008Hepatocellular nuclear polymorphism and nematode parasitization in the liver of European hake and European anchovy sampled between in 2003 and 2004 were remarkably prominent in some areas of the Bay of Biscay but they could not be related to the *Prestige* oil spill due to the lack of baseline data. (Marigómez et al., 2006). In mussels from Galicia the immune defence system was affected three months after the spill (Novas et al., 2007) and genotoxicity in mussels monitored in Galician localities between August 2003-June 2004 showing DNA damage in oil-exposed mussels was significant (Lafon et al., 2006). PAH levels in mussel tissues in Galicia had decreased to background levels after two years post-spill (Soriano et al., 2007) but biochemical parameters revealed persistent oxidative stress. However, the bioenergetic index scope-for-Growth did not show any evidence of physiological disturbance on mussels 17-24 months after the oil spill (Peteiro et al., 2007).

Fig. 3. Hypothetical working model showing the successive impacts of different nature after the Prestige oil spill and the corresponding strategy for their monitoring following a two tier approach based on analytical chemistry and biomarkers in combination with population and community parameters. It is assumed that some biomarkers may recover at longer times (i.e., lysosomal responses) whereas others may exhibit the inverse trend (i.e., diseases) (Garmendia et al 2011b).



A biological Mussel Watch programme was carried out to assess the long-term effects caused by the Prestige oil spill. The mussels were collected in 22 localities from Portugal to the Basque coast over three years (2003-2006) (Orbea et al., 2006; Apraiz et al., 2009; Garmendia 2011a, 2011b, 2011c; Ortiz-Zarragoitia et al., 2011; Marigómez et al., 2013; Izagirre et al., 2014). Several biomarkers at cell and tissue-level were analysed (lysosomal membrane stability, lysosomal enlargement, cell type composition and atrophy in the digestive gland, histopathology and reproduction and growth alterations) together with chemical analysis and ecological surveys of littoral communities. Overall, the most remarkable effects connected to the *Prestige* oil spill were a drastic lysosomal membrane destabilization, lysosomal enlargement, changes in cell type composition, atrophy of the digestive alveoli and increases in particular parasitic infestations and in cumulative prevalence of inflammatory responses recorded in 2005-2006. Although several parameters had returned to reference values (but not earlier than 2004-2005) some others were not fully recovered even by the end of the study in April 2006 (Cajaraville et al., 2006). PAH tissue levels in mussel decreased drastically after February 2004 but most biomarkers continued to be affected at least until 2005. According to these results, it was interpreted that successive impacts of different nature occurred after the Prestige oil spill (Fig. 2). PAHs (mainly naphthalene) bioaccumulation and associated biological effects in sentinel mussels were evidenced for two years after the *Prestige* oil spill. Sublethal effects in mussels in the absence of bioaccumulation prevailed for one more year, possibly due to the persistence of the oil spill biological impact (i.e., deteriorated ecosystem with reduced food sources, etc.). Finally, secondary effects reducing the health status of mussels (such as lysosomal membrane impairments and augmented chronic disease) seemed to persist in localities where some other biomarkers indicated the absence of any environmental insult (i*.e.,* cessation of the direct impact of the spill), being still evident at least until the end of the study in April 2006.

2.2. Gulf of Mexico

To our knowledge there are not relevant oil spills resulting from vessel incidents in the region. However, according to the US National Response Centre, the oil industry has thousands of minor accidents in the Gulf of Mexico every year.

Ixtoc 1 (1979). In June, the *Ixtoc 1* oil platform[13] in the Bay of Campeche suffered a blowout leading to a catastrophic explosion, which resulted in a massive oil spill that continued for nine months before the well was finally capped. The total quantity of oil spilled at sea will never be known exactly (470000-1500000 Tm); between 30-50% of this oil burned causing a vast atmospheric pollution and the remaining spread over the Gulf of Mexico in the form of drifting slicks. Dispersants and containment booms were applied but oil slicks remarkably reached the coast from Vera Cruz to Texas. Shrimp nurseries, mangroves, beaches and seabirds were oiled. Fishing and tourist activities were affected. No detailed report on the response operations was published. This was ranked as the largest oil spill in the Gulf of Mexico until the Macondo blowout during mid-April to July 2010 (see *Deepwater Horizon* oil spill).

Deepwater Horizon (2010). As a result of the explosion on the *Deepwater Horizon* drilling unit more than 700000 Tm of crude oil and 250000 Tm of gas were released uncontrolled into the Gulf of Mexico over a period of 87 days (Griffiths, 2012; Joye, 2015; McNutt et al., 2012a). This deep sea spill resulted in some unparalleled characteristics compared to previous spills, some of which were not foreseen or anticipated. Besides the expected surface oil slick a deepwater plume formed, affecting subsurface biota. Unexpectedly, huge amounts of oil from the deepwater plume were transported to deepwater sediments via marine snow. Before the *Deepwater Horizon* spill, sedimentation was not considered as an important fate of oil (Joye, 2015). The sedimentation rate was not determined during the incident (Joye et al., 2016). Besides these special distribution scenarios and sinks for oil, also the extensive use of the chemical dispersant Corexit 9500A® injected at the wellhead was unique. Since it is known that these chemicals alter the exposure pathway, the use of dispersants as an oil spill response action remains under critical discussion in the scientific community[14].

Following the *Deepwater Horizon* oil spill several studies focused on the fate and distribution of the spilled oil while biological monitoring of acute and chronic toxicity was also conducted. Indeed, the Gulf of Mexico ecosystem has now become one of the best-monitored regions with biological data across a huge number of taxa. Extreme harm on the coral communities have been reported at the spill affected sites with visible effects such as bare skeleton or broken and missing branches of the corals. The recovery has not been complete

[13] http://wwz.cedre.fr/en/Our-resources/Spills/Spills/Ixtoc-1
[14] http://wwz.cedre.fr/en/Our-resources/Spills/Spills/Deepwater-Horizon

by the last study year in 2014 in comparison with historical data on coral monitoring from previous decades (Etnoyer et al., 2016; Silva et al., 2016). The deep-sea gulf coral communities were also affected in a similar range as described for the mesophotic species; the effects described included the partial covering of corals by a brown flocculent material and tissue damage (Fisher et al., 2014). More than 60 sediment samples with increasing distance to the wellhead were investigated for macro- and meiofaunal abundance 2-3 months after the well was capped (Baguley et al., 2015; Montagna et al., 2013). The abundance was lowest in the wellhead zone, and also the taxonomic diversity was decreased in zones of greater impact. Sediment fauna is suggested as a useful bioindicator for the monitoring of oil spills as it occurs ubiquitous in deep-sea soft sediments and is unable to avoid exposure. Especially, the nematode-to-copepod ratio (N:C) was described as one of the strongest community-related metrics in *Deepwater Horizon* impacts (Baguley et al., 2015). Nematode dominance increased with increasing impact, hence in zones near the wellhead. Regarding deep-sea sediment biomonitoring, a lack of investigations regarding microbial deep-sea sediment communities was noticed (Montagna et al., 2013). Furthermore, arthropod and mollusc communities from estuarine habitats and salt marshes were intensively monitored after the blowout. Because the *Deepwater Horizon* spill coincided with the peak in blue crab spawning season, the megalopae settlement and body weight of the larval stage were investigated during 2010 and 2011 (Grey et al., 2015). When compared to available data from the 1990s and to unaffected reference sites, no significant difference in settlement rates or body weight was found. As a large amount of unexplained daily and yearly variation in settlement rates occurred, the question arises whether the settlement rate is a good bioindicator for contamination impacts with only few baseline data available. During the first years after the oil spill, salt marsh inhabiting crabs and snails were monitored. Even though a recovery of salt marsh fiddler crabs (*Uca* spp.) population was reported in 2011 (McCall and Pennings, 2012) the abundance declined again in 2013 and the recovery was concluded to be incomplete by the end of the study in 2014 (Zengel et al., 2016b). The salt marsh periwinkle population (*Littoraria irrorata*) was found to be unaffected in some monitored oil contaminated marsh areas since the reported abundance was comparable to reference sites (McCall and Pennings, 2012). Nevertheless, a great decrease in snail abundance was reported for seaward marsh zones where oiling was heaviest (Zengel et al., 2016a).

In order to evaluate the habitat structure for the marsh community, the impact of the *Deepwater Horizon* oil on the shoreline vegetation was evaluated. Following the accident in 2010, complete perishing of the vegetation occurred in heavily oiled shorelines (Mendelssohn et al., 2012). The stress on vegetation decreased with increasing distance to the shoreline (Khanna et al., 2013). While in 2011 no or minimal recovery of the heavily contaminated shorelines was reported (Mendelssohn et al., 2012), the analysis of infrared spectrometer data found the marsh vegetation to be recovering (Khanna et al., 2013). However, all studies agreed that *Spartina* sp. as one dominant species in marsh vegetation, is very resistant and tolerant to oil contamination (DeLaune and Wright, 2011; Khanna et al., 2013). The authors suggest that marsh vegetation recovery progresses naturally without the need of intensive remediation (DeLaune and Wright, 2011).

Murawski et al. (2014) assessed the occurrence of skin lesions among fish following the Deep Water Horizon oil spill. They also measured relatively high concentrations of PAH metabolites in the fish and were able to correlate the PAH parent compound composition to the oil collected at the *Deepwater Horizon* well head; it was deemed likely that the spill is the cause of elevated frequencies of skin lesions among the fish. Whitehead et al. (2012) found effects on the transcriptome of killifish sampled at a coastal marsh habitat that was affected by the oil. Transcriptomic effects corresponded with effects described for the exposure to PAHs, such as anti-estrogenic effects and effects on blood vessel morphogenesis. The AHR signalling pathway was also found to be regulated, which corresponded to measured CYP-1A expression levels, which was assessed to be diagnostic of exposure to the toxic constituents of oil. In 2011 adult killifishes from the gulf, already investigated in Whitehead et al. (Whitehead et al., 2012), were sampled again at the same site (Dubansky et al., 2013). Different biomarkers in gill tissue such as CYP expression levels were measured using transcriptomic or immunohistochemical approaches. Compared to reference sites that were not influenced by the Macondo oil the gill epithelia of fishes from exposed sites showed more lesions and increased CYP1A expression. The occurrence of gill lesions in fish species from oil influenced areas was also confirmed for menhaden (*Brevoortia* sp.) samples from the coastal zones of Louisiana sampled in the fall of 2010 (Bentivegna et al., 2015). Besides the biomarker analysis, dynamics in near-coastal fish assemblages following the *Deepwater Horizon* accident were investigated in the Northern Gulf of Mexico in 2011 and 2014 (Schaefer et al., 2016). In contrast to previous expectations, they found a comparatively high abundance of fish assemblages in 2011, but concluded that this can be explained by the few months lasting fishing ban after the *Deepwater Horizon* accident. The abundance of fish species was monitored in the following years and was found to have returned to levels comparable to pre-spill conditions.

Besides the evaluation of biomarker in coastal fish species, oil spill impacts on dynamics in pelagic fish species in the Gulf of Mexico were analysed (Frias-Torres and Bostater Jr, 2011; Rooker et al., 2013a). By combining biological data from literature with satellite data from the literature Frias-Torrez et al. (Frias-Torres and Bostater Jr, 2011) found that oil slick areas overlapped with the spawning areas of Bluefin tuna and blue marlin as well as with habitats of whale shark. The exposure of high quality habitats for different pelagic fish species was later verified by Rooker et al. (Rooker et al., 2013a). Within a biological sampling, they also observed a decline in larval fish densities in 2010 compared to the years before the oil spill. The study of Tarnecki and Patterson (Tarnecki and Patterson III, 2015) found an impact of the *Deepwater Horizon* oil spill on the diet and trophic position of the fish red snapper (*Lutjanus campechanus*) in artificial and natural reefs in the northern Gulf of Mexico.

Table 3. Species in which biomarkers were applied to follow-up exposure and biological effects after the *Deep Water Horizon* oil spill in 1989.

SPECIES	BIOMARKER	STUDY YEAR	REFERENCES
Gulf killifish Fundulus grandis	Immunolocalization of CYP1A in gill, liver, intestine, and head kidneys	2010, 2011	Dubansky et al. (2013)
Bottlenose dolphin *Tursiops truncatus*	Physical examination, pulmonary assessment, hematology, serum chemistry, adrenal hormones, thyroid hormones, reproductive hormones	2011	Schwacke et al. (2014)
Deep water corals	Tissue loss, sclerite enlargement, excess mucous production, bleached commensal ophiuroids	2010	White et al. (2012)

SPECIES	BIOMARKER	STUDY YEAR	REFERENCES
Pelagic and deepwater fish	External lesions, presence of fin rot disease, presence of parasites, presence of tumors	2011, 2012	Murawski et al. (2014)

It was not evident to directly link the observed changes in fish with the *Deepwater Horizon* spill. The exposure history of the fishes is not known and the influences of other contaminant sources could not be excluded as PAHs are ubiquitous and persistent, and because fishes are able to avoid oil contamination to some degree. Chemical analysis of fish tissue, if not below the detection limit, did not find a typical chemical fingerprint of *Deepwater Horizon* crude oil (Bentivegna et al., 2015; Dubansky et al., 2013).

Because some migratory bird populations use the Gulf of Mexico as a habitat during their migration, the impacts of the *Deepwater Horizon* oil slick on bird populations was monitored in some studies. Combining bird carcass sampling results with modelling data Haney et al. (2014) calculated the bird mortality caused by the oil slick. Carcass sampling is expected to clearly underestimate real mortality rates; hence additional information regarding the evaluation of mortality rates is necessary. Because avian mortality is one direct, immediate measure of ecological impacts caused by marine spills (Haney et al., 2014) it should be monitored properly after a spill incident. Besides mortality assessment, the analysis of avian blood samples for biomarkers of long term effects of oil and PAH contamination is gaining increasing attention in biomonitoring. Franci et al. (2014) did not detect elevated PAH contents in blood samples from population of northern gannets (*Morus bassanus*), which overwintered in *Deepwater Horizon* affected areas in 2010 to 2011. However, in another study focusing on the tundra peregrine falcon (*Falcon peregrinus tundrius*) increased PAH concentrations in blood cell samples from 2010 were found (Seegar et al., 2015) which decreased to basal levels in 2011. The non-destructive blood monitoring and PAH analysis is suggested as a useful tool for oil spill biomonitoring (Seegar et al., 2015). Throughout the available literature on biomonitoring, most authors remarked a lack of baseline environmental data for the evaluation of oil spill impacts on biota. Without concrete knowledge on the complex fluctuation dynamics of the ecosystem, it is difficult to detect and evaluate true population impacts. The *Deepwater Horizon* oil spill revealed many unexpected challenges but due to a comprehensive monitoring new knowledge, contributing to the improvement of future oil spill response actions was generated. During the last six years, a large set of biomonitoring data as well as data from supporting field and laboratory studies enabled a detailed insight into ecotoxicological effects of the *Deepwater Horizon* oil spill in the Gulf of Mexico. Additionally, the early imprecise underestimated oil discharge rates hindering the early response efforts showed that a reliable documentation and estimation of oil discharge rates into the environment is crucial for response action assessment and should be of high priority for future response scenarios (Joye, 2015). However, data and knowledge gaps in understanding the consequences of oil spills remain to be filled and monitoring of the Gulf of Mexico ecosystem should continue as the impacts of an oil spill can take years to decades to manifest (Grey et al., 2015).

3. Cold seas

3.1. The Arctic Ocean

According to a comprehensive report on Arctic oil pollution performed by the Arctic Monitoring Assessment Programme (AMAP) in 2007, no large marine oil spills have occurred in the Arctic (AMAP, 2010a; AMAP, 2010b). However, there have been a number of smaller marine oil spills such as following the grounding of M/V *Selendang Ayu* near Unalalska Island, Alaska, and from shipwrecks in the Russian part of the Barents Sea (AMAP, 2010b; Bambulyak and Frantzen, 2005). In these cases, reports of the spills appear to have been in the range from < 50 m^3 and up to 1270 m^3 (M/V *Selendang Ayu*). In addition, there have numerous smaller marine oil spills in harbours, ports and terminals in Alaska, Canada, Greenland, Norway and Russia (AMAP, 2010b). Presumably, a large number of these spills were related to re-supply of fuel oil or diesel to the communities, fishing vessels and small ships, and accidents on land causing leakage to the coastal marine environment, such as a spill of 42 m^3 of Arctic Grade fuel oil in Greenland where some of the oil leaked to the shore-line (Lindgren and Lindblom, 2004). A comprehensive literature search in scientific databases, and reviewing of relevant scientific reviews (Murphy et al., 2016) have not revealed any reported significant marine oil spill accidents in the Arctic following these relatively recent AMAP reports (AMAP, 2010a; AMAP, 2010b). Monitoring of biological effects following marine oil spill incidences in the Arctic are very limited.

Surprisingly, a comprehensive search in scientific literature databases only identify one follow-up study related biological effects of actual crude oil spills or fuel or diesel oil spills in the Arctic. This is in accordance with the bibliographic study published by Murphy et al. (2016). It should, however, be mentioned that there are numerous follow-up studies related to effects of terrestrial oil spills, including spills of crude oil, fuel oil and diesel oil. Although there have been some experimental in situ marine oil spills in the Arctic, such as in the Barents Sea, in sea-ice infested Arctic waters and shorelines (Brandvik and Faksness, 2009; Faksness et al., 2016; Fingas and Hollebone, 2003; Prince et al., 2002; Sergy et al., 2003), the literature search revealed only a few biological effect studies related to such experimental in situ marine oil spills in the Arctic (Cross, 1987; Powell et al., 2005). These two studies were related to effects on ice algae following an in situ crude oil experiment (Cross, 1987), and bacteria communities following in situ long-term exposure to fuel oil and lubricating oil (Powell et al., 2005). However, since the It should, however, be noted that there are numerous experimental laboratory studies on effects of different crude oils, fuel oil, diesel oil, and chemically treated oil, including oil droplets and the water accommodated fractions of these oils. Focus herein is on actual oil spills, it is beyond the scope of the present review to focus on these studies. Moreover, several experimental in situ marine oil spill experiments that have focused on oil spill actions have been performed (Faksness et al., 2016). These have included the use of chemical dispersants and in situ burning in oil spill response practices. However, none of these in situ studies appears to have applied biomarker studies to assess biological effects and consequences of these actions. Thus, there is a clear need to include biomarker studies of biological effects to investigate if the oil spill responses results in adequate and positive effects on biota.

Selendang Ayu (2004). Following the M/V Selendang Ayu oil spill at Unalaskan Island, Alaska, EROD activity was analysed in liver biopsies of harlequin ducks (Histrionicus histrionicus) as an indicator of hydrocarbon exposure in three oiled bays and one reference bay in 2005, 2006, and 2008. The EROD activity in ducks from oiled bays was significantly higher than in the reference bay in seven of nine pairwise comparisons, indicating that harlequin ducks were exposed to lingering hydrocarbons more than three years after the spill (Flint et al., 2012).

3.2. Antarctica

As in the Arctic, there are few reports of large marine oil spills in the Antarctica of sub-Antarctica. The two most known are spills following the grounding of *Bahia Paraiso* in January 1989 close to the US Palmer Station on the Antarctic Peninsula, where an estimated volume of 1000 m^3 of diesel and jet fuel leaked into the sea (Barinaga and Lindley, 1989), and the marine oil spill following the grounding of *Nella Dan* at Macquarie Island in December 1987, which resulted in the release of 270 m^3 of mostly light marine diesel into the sea (Smith and Simpson, 1995).

Paradoxically, there are somewhat more scientific literature available on biological effects of real oil spills in the Antarctica than in the Arctic. Several studies reported effects following the grounding of *Nella Dan* on Macquarie Island (1987) and *Bahia Paraiso* on the Antarctic Peninsula (1989).

Nella Dan (1987). Following the *Nella Dan* oil spill, large mortalities of marine invertebrates were reported within a 2-km stretch coastline the first few days after the spill (Smith and Simpson, 1995). One year after the effect of the spill on rocky substrate benthic communities was restricted to the lower littoral and sublittoral zones, where particularly echinoderms and a limpet (*Nacella macquariensis*) and the isopod *Limnoria stephenseni* were affected (Simpson et al., 1995; Smith and Simpson, 1995). There were differences in cover for some algal species between locations and within the kelp holdfasts; communities were dominated by peracarid crustaceans at control locations and by opportunistic polychaetes at oil-affected locations (Simpson et al., 1995). Six years following the *Nella Dan* oil spill, there were no significant differences between the benthic macrofaunal community structure in oiled and control rocky shore locations, but a moderately oiled location in a bay showed little evidence of recovery due to sediments containing traces of diesel oil (Smith and Simpson, 1998).

Bahia Paraiso (1989). The *Bahia Paraiso* oil spill was suggested to indirectly cause a complete reproductive failure in a population of south polar skuas (*Catharacta maccormicki*) just after the spill (Eppley and Rubega, 1990). Adults were observed to forage in oil slicks and became fouled, but the adult mortality rate was low. There was no transfer of oil to eggs or chicks and chicks showed no evidence of toxicity. However, a short-term disruption of normal parental attendance behaviour exposed the chicks to fatal intraspecific aggression. Within three weeks after the spill incidence, all chicks had died (Eppley and Rubega, 1990). However, it I also possible that the reproductive failure was unrelated to the presence of oil, but caused by natural variation producing food shortages (Eppley, 1992). A study conducted two months after the *Bahia Paraiso* spill concluded that the subtidal soft-bottom macrofauna in the area (depth 30-115 m) not was affected by the spill due to that the sediments were not contaminated by oil (Hyland et al., 1994). Another study reported elevated concentrations of PAH metabolites in the bile and tissues of fish close to the wreckage two years after the incidence (McDonald et al., 1992). However, in a follow-up study (1991-1993), concentrations of biliary PAH metabolites and EROD activity in fish from the *Bahia Paraiso* wreckage area did not differ from that fish caught closer to the Palmer Station (McDonald et al., 1995).

3.3. The Northern Pacific Ocean

***Exxon Valdez* (1989).** Although not in the Arctic, the *Exxon Valdez* oil spill in Prince William Sound, Alaska in March, occurred in an environment that well resembles Arctic conditions. In this accident, approximately 42 000 m^3 of North Slope crude oil was spilled into the Prince William Sound, eventually affecting more than 1 900 km of the pristine Alaskan coastline (Peterson et al., 2003).

Following the *Exxon Valdez* oil spill in Alaska, a large number of scientific publications related to this spill was been published. A search on Web of Science (WOS) (Core collection) using the search terms “oil spill” AND “Exxon Valdez” in “title” revealed 137 research articles and 13 review articles. These articles focused on effects across a number of taxa, ranging from cultural and societal effects in humans to molecular, cellular, organismal and population effects in algae, invertebrates, fish, and wildlife birds and mammals. According to Murphy et al. (2016), the Oil Spill Paper Database[15] contains 91 studies for the *Exxon Valdez* spill.

Biological effects of the *Exxon Valdez* oil spill, from the immediate effects following the spill to the long-term effects and the restoration of the ecosystems have as been reviewed in several review articles. The first two were published in 1996 (Bence et al., 1996; Paine et al., 1996) and the majority of the reviews were published during the first decade of the 21st century (Bowyer et al., 2003; Golet et al., 2002; Harwell and Gentile, 2006; Peterson, 2001; Peterson et al., 2001; Peterson et al., 2003). According to the comprehensive review by Peterson et al. (2003), during the first days after the spill mass mortality was reported among sea otters, harbour seals and killer whales, seabirds, macroalgae and benthic invertebrates (Peterson et al., 2003; and references therein). During the following years, oil in the intertidal zone and oil that was trapped under mussel beds provided long-term exposure causing mortality of pink salmon (*Oncorhynchus gorbuscha*) embryos until 1993, whereafter (1994 and 1995) it appeared to cease (Bue et al., 1998), but mortality and sublethal effects continued in other fish species, sea otters, and sea ducks for years (Peterson et al., 2003; and references therein). Elevated oil residues persisted in the clam *Protothaca staminea*, which is an important prey for sea otters, until at least 1996 (Peterson, 2001), impacting these otter populations negatively and hindering their recovery and causing contact with oil until at least 2008 (Bodkin et al., 2012). In birds, populations of sea ducks, such as harlequin ducks and Barrow’s goldeneye (*Bucephala islandica*), which prey on intertidal benthic organism, were highly affected by long-term effects (Peterson et al., 2003) and references therein). According to a recent study, analysis of EROD activity strongly indicated that wintering harlequin ducks were exposed to oil until 2014, 25 years after the spill occurred, when EROD activity had reached baseline (Esler et al., 2016). In wintering Barrow’s goldeneye, EROD activity was still elevated in 2009 (Esler et al., 2011). Elevated EROD activities were also reported in breeding pigeon guillemots in 1999 (Golet et al., 2002). No follow-up study on that species has been reported. In their review, Harwell and Gentile (2006) concluded that, except for one pod of killer whales and one subpopulation of sea otters, 17 years following the spill there were no detectable effects on producers, filter feeders, fish and bird primary consumers, fish and bird top predators, a bird scavenger, mammalian primary consumers and top predators, biotic communities, ecosystem-level properties of trophodynamics and biogeochemical processes, and landscape-level properties of habitat mosaic and wilderness quality in Prince William Sound, and that the ecosystem had effectively recovered from the *Exxon Valdez* oil spill. In

[15] https://data.gulfresearchinitiative.org

contrast, more recent studies indicate that effects in sea ducks and sea otters persisted until 2014, i.e. 25 years after the spill (Bodkin et al., 2012; Esler et al., 2016). In 2014, exposure to subsurface oil residues appeared to have become low, at least indicating that sea otters did not appear to be exposed (Harwell and Gentile, 2014).

Table 4. Species in which biomarkers were applied to follow-up exposure and biological effects after the *Exxon Valdez* oil spill in 1989.

SPECIES	BIOMARKER	STUDY YEAR	REFERENCES
Pacific herring *Clupea pallasai*	Embryo sensitivity	1992	Kocan et al. (1996)
River otters	Faecal profiles of porhyrines	1990, 1996	Blajeski et al. (1996) Taylor et al. (2000)
River otters	Haptoglobin	1989, 1990	Duffy et al. (1993)
River otters	Blood haptoglobins, interleukin-6, aspartate aminotransferase, alanine aminotransferase, creatine kinase activities	1991, 1992	Duffy et al. (1994a) Duffy et al. (1994b)
Sea otter	DNA cell content in white blood cells	1991	Bickham et al. (1998)
Sea otter	Blood leukocyte gene transcript profiles related to immunomodulation, inflammation, cell protection, tumor suppression, cellular stress-response, xenobiotic metabolizing enzymes, antioxidant enzymes	2008	Miles et al. (2012)
Bivalves *Mya arenaria* *Mytilus trossulus*s	Cu/Zn superoxide dismutase (SOD), cytochrome P450, glutathione peroxidase, glutathione *S*-transferase , 4-hydroxy-2E-nonenal-adducted protein, heat-shock protein (HSP) 60, 70 and90, malondialdahyde-adducted protein, Mn SOD, small HSPs, ubiquitin	1999	Downs et al. (2002).
Pigeon guillemot *Cepphus columba*	Red blood cell count, packed cell volume, mean cell volume, haemoglobin, mean cell haemo- globin content, counts of white blood cells, heterophils, lymphocytes, eosinophils, basophils, activity of creatine phosphokinase, lactate dehydrogenase, aspartate aminotransferase, alkaline phosphatase, gamma-glutamyl transferase, concentration of calcium, uric acid, plasma protein, total protein, alpha-1 macroglobulin, alpha-2 macroglobulin, beta globulin, gamma globulin, albumin, albumin to gamma globulin ratio, bile acid, phosphorus and sodium, haptoglobin, EROD	1997, 1998, 1999	Golet et al. (2002) Seiser et al. (2000)
Masked greenling *Hexagrammos octogrannnus* Crescent gunnel *Pholis laeta*	CYP1A in liver vascular endothelium, liver EROD, biliary fluorescent aromatic compounds (FAC)	1996-1999	Jewett et al. (2002)
Rockfish Rock kelp greenling Pacific halibut	FAC, liver EROD, tissue (liver, heart, gill) immunohistochemistry	1999, 2000	Huggett et al. (2003) Page et al. (2004)

Rockfish *Sebastes* spp.	FAC, microscopic lesions (pigmented macrophage aggregates and hepatic megalocytosis, fibrosis, and lipid accumulation)	1989-1991	Marty et al. (2003)
High cockscomb prickleback *Anoplarchus purpurescens*	FAC, and liver EROD	2004-2005	Huggett et al. (2006)
Caged juvenile Coho salmon *Oncorhynchus kisutch*	CYP1A, SOD, and glutathione peroxidase, gene expression	2004	Roberts et al. (2006)
Wintering harlequin ducks *Histrionicus*	EROD	1998, 2005-2009 2011, 2013, 2014	Esler et al. (2016) Esler et al. (2010)
Barrow's goldeneye *Bucephala islandica*	EROD	1996-1997, 2005, 2009	Esler et al. (2011) Trust et al. (2000)
Pink salmon *Oncorhynchus gorbuscha*	Embryo mortality	1989-1993, 1994, 1995	Bue et al. (1998)

The use of biomarkers to assess the *in situ* biological effects of the *Exxon Valdez* oil spill, appear to have been very limited. Off the relatively few articles related to biological biomarkers, the majority of the studies appear to have been performed on three species of birds, river otters and sea otters, whereas there were few studies on fish and invertebrates (Table 4). The biomarker studies were used to assess if it was likely or not that the investigated organisms were exposed or not at the sampling year, and to investigate time-dependent reductions of exposure to oil, and thus the recovery process of the ecosystems following the spill. According to Harwell and Gentile (2006), in 2006, 17 years after the oil spill, there were no detectable effects on most organisms, and the ecosystem had effectively recovered from the oil spill. However, biomarker studies showed that effects were still detectable in seaducks and sea otters until ca. 2014, when no effects on biomarkers in harlequin ducks were reported (Esler et al., 2016), and when exposure to subsurface oil residues had approached background levels (Harwell and Gentile, 2014).

Conclusively, biological biomarker studies appeared to be were successful in documenting the recovery process following the *Exxon Valdez* oil spill, providing indications or evidence on when central organisms in different ecosystems had recovered. The studies showed that pelagic species and some benthic species recovered first, and that more 20 years passed on before seaducks and sea otters that feed on benthic organisms not were exposed to oil residues from the *Exxon Valdez* oil spill.

In the aftermath of *Exxon Valdez* oil spill, Peterson et al. (2003) focused on the need for including a range of physiological, biochemical, and histopathological evaluations of toxicity following oil spills, *i.e.*, to apply biomarker responses to assess biological effects. Indeed, recently there has been a focus on developing and testing the use of relevant biomarkers in several Arctic organisms, such as blue mussels, polar cod (*Boreogadus saida*), copepods (*Calanus glacialis* and *Calanus finmarchicus*) (Andersen et al., 2015; Hansen et al., 2013; Lysenko et al., 2015; Nahrgang et al., 2010). However, these have not been tested or applied following in situ oil spills.

3.4. The Baltic Sea

Due to its characteristics (brackish waters, climate conditions, closed inland sea with a slow water exchange, fractal coastline, unique biota) the Baltic Sea has been classified as a particularly vulnerable sea, and oil pollution is likely to have a negative impact on its sensitive ecosystem. Furthermore, as the coastline of the Baltic Sea is fragmented and *e.g*. in the northern part consists of tens of islands and skerries, several hundreds of kilometres of coastline may be polluted as a result of a large-scale oil spill unless the spilled oil is recovered in the open sea before it reaches the shoreline.

The Baltic Sea receives frequently many very small oil spills. For instance, according to the statistics reported by the Finnish Environment Institute SYKE, a record number of 107 spills (<180 l each) were reported back in 2001. Nevertheless, the number of oil spills caused by vessels has decreased over the past decades due to tighter requirements regarding vessel condition, strict sanctions on spills and improved overall surveillance – especially from the air – and route planning. Thus, according to the HELCOM report, a total of 82 oil spills were identified in the Baltic Sea by air in 2015 but most of them (78%) were smaller than 100 l (Rousi & Kankaanpää, 2012).

Compared to the major oil spill accidents reported in other parts of the world, those that have occurred in the Baltic Sea have been up to two orders of magnitude smaller in regard to the quantity of the spilled oil. Taken the specific characteristics of the Baltic Sea they have still to be taken as highly significant accidents affecting negatively the local ecosystems. Concerning especially the earlier ones, no environmental impact assessments or ecosystem effect studies were carried out.

Palva (1969). In 1969, the tanker *Palva* ran aground in the Kökär Archipelago in southwest Finland and 120-150 Tm of Russian crude oil was released to the sea, eventually spreading to cover an area of 200 km2 (Leppäkoski, 1973). Due to the oil spill and the chemical recovery operations, some crustacean, fish and common eider population were affected but the ecosystem appeared to have recovered relatively rapidly (Pelkonen and Tulkki, 1972).

Tsesis (1977). In 1977, the tanker *Tsesis* grounded in the archipelago off Södertälje (Sweden) and over the next few days ca. 1100 Tm of medium grade fuel oil were released into the sea (Lindén, 1979). The impacts of the oil spill on the ecosystem were severe with the benthos being heavily affected. The recovery period for the total area was estimated to be 2-3 years (Lindén et al., 1979), although sublethal long-term effects on the organisms in the area were most likely longer lasting. Considerable oil quantities reached the benthos by sedimentation (Elmgren et al., 1983). Within 16 days, benthic amphipods of the genera *Pontoporeia* and *Monoporeia*, as well as the polychaete *Harmothoe sarsi* Kinberg, showed reduction to less than 5% of pre-spill biomasses at the most impacted station. The Baltic clam *Macoma balthica* was more resistant but was heavily contaminated by oil (about 2000 µg/g dry wt total hydrocarbons). The meiofauna was strongly affected. In the winter following the spill, gravid *Monoporeia affinis* females showed a statistically significant increase in the frequency of abnormal or undifferentiated eggs. Food-chain transfer of oil to flounder (*Platichthys flesus*) was indicated. Not until the second summer after the spill were the first signs of recovery noted at the most heavily impacted station. Three years after the spill *Pontoporeia/Monoporeia* biomass was still depressed in the most affected area, while *H. sarsi* showed normal biomass, and *M. balthica* abundance was inflated. The results underlines that different sensitivities of organisms to oil can rapidly cause drastic changes in benthic community structures, leading to alterations in the food web. This is of great importance in areas characterized by an extremely low benthic biodiversity such as the Baltic Sea where only a handful of species are present which of a few (amphipods and *M. balthica*) dominate completely.

Antonio Gramsci (1979). The tanker *Antonio Gramsci* grounded off the Latvian coast in May 1979, releasing ca. 5500 Tm of crude oil, which drifted in the Baltic Sea for 2-3 months before reaching the Stockholm and Åland archipelagos, affecting the littoral benthos (Bonsdorff, 1980), aquatic plants (Suomalainen, 1980) and common eiders. Petroleum hydrocarbons remained in the sea and the sediment layers, and petroleum constituents accumulated in the ecosystem, possibly causing sublethal long-term impacts.

Alambra (2000). The spill, from the Maltese tanker *Alambra*, occurred in the port located about 10 km outside of Tallinn on Sept. 16, 2000. A week after the spill about 240 tons of heavy oil had been collected from the sea. Port authorities claimed that the pollution was contained inside the harbour and there were no major damage to harbour biota. But by Sept. 25, reports were circulating that the oil had spread to outlying areas. Officials confirmed that one dead swan was found covered in oil. However, there were not large quantities of poisoned fish and birds washing ashore.

Between 22 September and early October 2000, persistent oil landed on the shores of Fårö and Gotska Sandön, two islands to the north of Gotland in the Baltic Sea, and thereafter on several islands in the Stockholm archipelago. The Swedish Coastguard, the Swedish Rescue Service Agency and local authorities undertook clean-up operations, which resulted in the collection of some 20 m^3 of oil from the sea and from shore. Investigations by the Swedish authorities indicated that the oil could have been discharged within the Swedish Exclusive Economic Zone to the east of Gotland, possibly from the Maltese tanker *Alambra*, which had passed the area at the assumed time of the oil spill on a ballast voyage to Tallinn (Estonia). According to the Coastguard, analyses of oil samples from the polluted islands matched those of samples taken from the *Alambra*.

Eira (1984). In August 1984, the motor vessel *Eira* grounded in the Quark in the Gulf of Bothnia and ca. 200 Tm of heavy fuel oil were released into the sea, spreading to cover 1500 square kilometres of sea and coast, mainly on the Finnish side of the Quark (Nyman et al., 1987). The impacts of the oil on the ecosystem were detected over a considerably larger area than the visible spill area; these included bioaccumulation of hydrocarbons in clams, deformations in planktonic fry of herring and gobbies, and damage to sea birds. Oil response operations failed due to storm conditions and an insufficient number of booms. Studies show that the environmental impacts of the oil spill were smaller than anticipated, although the long-term impacts could not be determined during the three-year research period (Koivusaari, 1987).

Antonio Gramsci (1987). The tanker *Antonio Gramsci* grounded a second time in February 1987, this time near the Porvoo lighthouse in the Gulf of Finland, and spilled ca. 570 tonnes of crude oil, affecting, for example, local fish catches by polluting salmon hoop nets. Bird communities in the area suffered only minor damage, as the oil drifted towards the opposite shore. The National Board of Waters and Environment organized environmental impact studies in the area that have been published as an extensive report (in Finnish).

Baltic Carrier (2001). In March, the tanker *Baltic Carrier* collided with the bulk carrier *Tern* between 30 nautical miles northeast of Rostock (Germany). Neither vessel sank, but approximately 2700 Tm of heavy fuel oil was lost from *Baltic Carrier*. The spilled oil was driven north west by wind-induced currents towards Denmark, where it began to come ashore on the same day. The affected shorelines were predominantly sandy beaches, although a number of marsh areas received heavy oiling along the water line[16]. The Danish Coast Guard responded to the spill with several of its own vessels and others requested from Germany and Sweden. Due to the nature of the spilled oil, mechanical grabs were used to recover the oil/water mix, rather than conventional skimmers. The shallow nature of many of the oiled areas limited the ability to conduct water-based recovery operations. Booms were used to protect sensitive areas and to contain floating oil for recovery. Approximately a third of the released oil was recovered at sea. Manual recovery was also necessary. More than 2000 bird casualties were reported. A report of the follow-up of the accident is available under the name "The "Baltic Carrier" Oil Spill: Monitoring and Assessment of Environmental Effects in Gronsund (DK)".

Runner 4 (2006). On March 5, 2006, a message was received of a collision in a convoy led by the ice-breaker *Kapitan Sorokin*, Russian flag. The convoy was moving westward (departure St. Petersburg). *SV Apostol Andrey*, Maltese flag, and the Runner 4, flag of the Commonwealth of Dominica, collided and the latter sank. The ships collided during ice-breaking activities while moving in a convoy in Estonian waters. As the ice lane narrowed due to weather conditions, the *Runner 4* stopped but the *SV Apostol Andrey* behind her was unable to stop and hit the *Runner 4* stern with her bow. The ship's engine room was damaged and all 14 crew members (Russian citizens) were taken on board the ice-breaker, after which the *Runner 4* sank within three minutes. The cargo on the *Runner 4* consisted of aluminium; bunker supplies: 102 tonnes of heavy fuel, 35 tonnes of light fuel oil, and 600 litres of lubricating oil. According to the crew of the ice-breaker, no pollution was detected after the collision.

The collision took place because of severe ice conditions in the Gulf of Finland. The Gulf of Finland ice conditions in winter 2005/2006 were normal. According to the ice charts of the Finnish Institute of Marine Research, ice first appeared in the eastern Gulf of Finland in the beginning of December 2005, approximately one week earlier than average; the ice was first forming in the western part after mid-December, which is at the average time. The weather was mild in the beginning of January 2006 and there was little new ice forming in the Gulf of Finland. Then in the end of January the weather became colder and ice spread soon from the eastern part to the west. The ice conditions remained similar in February; but in the beginning of March the cold increased and more ice was rapidly formed. As a result, the Gulf of Finland was soon completely covered by ice. After this the ice conditions began to decrease. In April the weather was cool and the ice melted slowly.

The first evidence of oil pollution was observed 40-50 km to the southwest from the collision site on 12 March. No oil pollution was observed until 15 March, when several polluted areas were detected. Two areas were 60 and 70 km from the collision site, but remained in

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16 http://www.itopf.com/in-action/case-studies/case-study/baltic-carrier-off-denmark-2001/

the middle of the Gulf. The third area at the collision site indicated that oil leakage had been continued (see also Wang et al., 2008). One day later the oil was observed at the coast of the location where oil spill was the first seen on 12 March. On 17 March many oil slicks were observed at the longitude of Tallinn-Helsinki line. The region where oil slicks were detected extended 60 km zonally. In general the westward motion of the oil pollution from the collision site is explained by the East dominated wind from 1–17 March. On 18-19 March the oil pollution was observed in Tallinn Bay. Since 25 March, the oil was observed in numerous places in Tallinn and Muuga Bay and at the entrances to the bays. Still in several occasions, oil was observed some 50–60 km to the southwest from the collision site, while the most distant oil pollution was 120 km away from the collision site. The last survey on 5 April indicated oil slicks near the Estonian coast over the distance of 130 km. The leakage of oil from the wrecked ship was observed even on 9 April, but it was reported that the oil slick was very thin and was composed of light oil (Wang et al. 2008).

In-situ observations showed that the spills contained both light and heavy fuel oil, which stained a large area of the ice field. It was also noticed that the drifting oil tended to accumulate on ice edges. The oil spill was observed to be mixed light and heavy fuel oil during the first combating period, but it was seen to be pure light fuel oil during the second combating period. The oil was continuously leaking from the shipwreck and drifting with the pack ice during the whole event. Due to the difficulties in detecting oil spill under thick ice cover, it is thus far not fully clear how the oil spread to cover a large area of over 500 km. The spatially scattered occurrence of the oil pollution can be attributed to the ice conditions in the Gulf of Finland and temporally irregular oil spill from the wrecked ship.

Since detection of the oil, the oil drift forecasting system STW was used to predict the movement of oil during the oil recovery operations. The forecasts were daily updated according to collected observational data.

A thorough analysis of the quality of the oil drift forecasts compared with observations was investigated in a workshop between MSI, SMHI and RDANH in December 2006. The workshop was sponsored by SIDA East, RDANH and SMHI. The forecasts were quite good, but could be improved. As a result, a cooperation project between SMHI, MSI, SYKE, UH and FIMR was carried out to improve observations, the STW oil drift model as well as the HIROMB ocean circulation model.

Oil pollution on North-West coast of Estonia (2006). An oil spill was detected in the southwestern Gulf of Finland in January 2006. Severe storms hindered the removal of the oil from the sea surface. It was estimated that approximate amount of the heavy oil was all together about 40 tonnes of which 10 tonnes of oil stranded to the shores of the Keibu Bay area, northwestern Estonia, along 35 kilometers of coastline. The pollution source was never identified.

Samples for macrobenthos were collected a few months after the pollution and in 2007 and 2009. Data collected in the area in 1997 were available for reference. Statistical analyses did not show explicit effects of oil on benthos. The noticed decrease in the abundance of *Bathyporeia pilosa* and *Macoma balthica* may have been related to the oil, but it would be speculative to attribute this pattern solely to the spill without a proper before-after-control-impact (BACI) design with several control locations. The study clearly showed that in the case of accidental environmental impacts like oil pollution it is impossible to apply a proper setup of the BACI design. This leads to difficulties in distinguishing between the effects of natural environmental factors and oil on the biota. The study also advocates for needs of alternative methodologies in order to effectively assess the impacts of accidental anthropogenic disturbances on benthic communities (Herküll and Kotta, 2015).

Randel Kreitsberg and Arvo Tuvikene from Estonian University of Life Sciences studied the flounders (*Platichthys flesus trachurus*) of the Nõva region in search of signs of the oil pollution in the sea. The samples taken from flounders indicate that the situation was not very bad and that nature recovers fast – the samples taken a year after the oil spill were as clean as from not-spilled waters.

Estonian Found for Nature reported that thousand of birds were killed by the oil pollution (by some sources about 4000 bird casualties).

4. Conclusions

The current review shows that oil spills of highly different magnitude have occurred around the globe and their environmental effects are variable, depending on a large variety of factors from the nature of the oil to environmental physical conditions, the type of response actions, and from the type of effect ranging from molecular level responses through tainting of individuals to structural and biodiversity alterations in communities. Thus, no simple answers can be given when asked about the effects of oil spills on marine and coastal ecosystems.

Some obvious items arise from the literature review. Among them, the need for chemical and biological baseline data has been stressed in most of the cases. It is practically impossible to distinguish the effects of oils spills if adequate pre-spill environmental data is available from the impacted areas. The sparse application of biological effects methods, including biomarkers, in marine monitoring and assessment is sadly reflected in studies of the exposure and effects of oil spills on organisms. For example, in case of the Baltic Sea no such investigations were carried out, partly due to the lack of available methodologies at the time of the accidents. Most of the past post-spill studies focused on population and community level responses, which naturally are ecologically the most relevant impacts but have by nature a long response time (excluding acute catastrophic exposure events) and are difficult to link to any specific chemicals. Instead, early-warning biomarkers have prognostic power for effects taking place at higher biological levels. Although significant progress in the application of biological effects methods has been seen during the past couple of decades, a general lack of readiness to use these methods exists in many countries, while regular monitoring campaigns, which produce also the much-needed baseline data, are far too sparse and also limited in regard to the number of parameters measured.

Most of the major accidents have occurred in temperate sea regions, which means that also the oil response actions taken aboard as well as our knowledge of the response of the ecosystem to the spill is largely originating from these areas. Thus, comparing these with the cold seas characterised by markedly different physical conditions and biology is obviously quite difficult. Therefore, the new information generated within the GRACE project must be therefore seen important and also very timely taking into account the ongoing and foreseen increases in oil transport in northern sea areas.

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ANNEX

Literature review (Main reference Lists (30/07/2016)

- Part 1. Monitoring of Pollution & Biological Effects in the Arctic Ocean, the Greater North Sea and the Baltic Sea (and other Cold Seas)
- Part 2. Monitoring of Oil Spills and Oil Spill Responses
- Part 3. Ecology and Biology of Polar Seas

Part 1. Monitoring of Pollution & Biological effects in Arctic Ocean, the North Sea and the Baltic Sea (and other cold seas)

(KEYWORDS: Baltic Sea; North Sea; Pollution Monitoring; Oil Spill; Biological Effects)

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Part 2. Monitoring of oil spills and oil spill responses

(KEYWORDS:** Amoco., Exxon., Tsesis., Erika., Prestige., Deepwater., Torrey., Sea Empress., Ekofisk., Odyssey., Bohai., VLCC., Braer., oil spill., oil spill response**)

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Part 3. Ecology and Biology of Polar Seas

(KEYWORDS: marine; mollusc or fish; Arctic or ice)

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