



Environmental impacts of produced water and drilling waste discharges from the Norwegian offshore petroleum industry[☆]



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ABSTRACT

Operational discharges of produced water and drill cuttings from offshore oil and gas platforms are a continuous source of contaminants to continental shelf ecosystems. This paper reviews recent research on the biological effects of such discharges with focus on the Norwegian Continental Shelf. The greatest concern is linked to effects of produced water. Alkylphenols (AP) and polyaromatic hydrocarbons (PAH) from produced water accumulate in cod and blue mussel caged near outlets, but are rapidly metabolized in cod. APs, naphthenic acids, and PAHs may disturb reproductive functions, and affect several chemical, biochemical and genetic biomarkers. Toxic concentrations seem restricted to <2 km distance. At the peak of discharge of oil-contaminated cuttings fauna disturbance was found at more than 5 km from some platforms, but is now seldom detected beyond 500 m. Water-based cuttings may seriously affect biomarkers in filter feeding bivalves, and cause elevated sediment oxygen consumption and mortality in benthic fauna. Effects levels occur within 0.5–1 km distance. The stress is mainly physical. The risk of widespread, long term impact from the operational discharges on populations and the ecosystem is presently considered low, but this cannot be verified from the published literature.

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1. Introduction

Offshore oil and gas activities have been established on the Norwegian Continental Shelf (NCS) over the past 40 years. At present about 65 oil and gas producing fields are in operation and the number is increasing. In 2012 the total Norwegian production of oil and gas was 226 million standard cubic meters of oil equivalents (Sm³oe), 39% of which was oil (Norwegian Oil and Gas, 2013).

Environmental pressures from offshore oil and gas operations are greatest in the North Sea (NS), but there are also high activities in the Norwegian Sea and the Barents Sea. The NS is probably the most studied offshore oil and gas production area in the world. Formation water brought up with the hydrocarbons (produced water, PW) and rock cuttings from drilling (drill cuttings) are the major sources of contaminants entering the sea from regular

operations. Drilling waste and PW are cleaned by various physical means before discharge and regulations put strict limits on levels of contaminants which can be discharged to the sea. Also reinjection has been used to reduce overall discharges for many years. Displacement and drain water are also discharged, but the total amount of contaminants discharged is relatively low compared to the other two sources. Accidental spills of oil and chemicals can arise during operation. In 2012 totally 122 small incidents were reported with a total oil discharge of 16 m³. Acute spills of chemicals have been stable at 100–150 incidents per year on the NCS over the past decade (Norwegian Oil and Gas, 2013). Large chemical spills in 2007, 2009 and 2010 came from leakages from injection wells. No leakage has occurred after that due to technical improvements (Norwegian Oil and Gas, 2013).

Until the mid 1990s the discharge of cuttings with oil based drilling mud (OBM cuttings) was the main source of oil hydrocarbons entering the marine environment from the offshore petroleum industry in the NS. The average annual discharge of oil on cuttings to the NCS for the period 1981–1986 was 1940 tons (Reiersen et al., 1989). This source was gradually eliminated by regulation, in 1993 in Norway and in 1996 and 2000 within the OSPAR region (OSPAR Commission, 2000). Concurrently oil

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discharged with PW on the NCS has increased and amounted to 1535 tons in 2012 (Norwegian Oil and Gas, 2013) i.e. almost at level with the former peak discharges of oil on cuttings. This is primarily due to an increase in overall PW volumes due to well ageing and rising number of producing fields.

One of the main objectives of environmental monitoring is to assess if discharge regulations are sufficiently protective. The history of sediment monitoring on the NCS has demonstrated that detection of unexpected ecological effects alone has led to stricter discharge legislation. The most conspicuous example is the identification in the early 1990's of much larger areas with fauna effects from OBM cuttings discharges than previously known (Gray et al., 1990), leading to the prohibition of such discharges by OSPAR in 1996 (Gray et al., 1999). Extensive experimental and field studies have later been made to assess the ecological effects of the discharges. This review summarizes the findings of a large, Norwegian research program¹ which combines experimental research and *in situ* monitoring on the NCS to address the likelihood of population and ecosystem effects from operational discharges of PW and drill cuttings. The concern and focus of the program is very much on PW since the potential environmental effects are less clearly understood than for drilling waste.

2. Produced water

PW is water from the formation produced along with oil or gas. It may sometimes also contain injection water and condensation water. The composition and characteristics of naturally-occurring chemical substances in PW are closely coupled to the geological characteristics of each reservoir. The composition of PW is complex and can comprise several thousand compounds that vary in concentration between wells and over the lifetime of a well. Dispersed oil, aromatic hydrocarbons and alkylphenols (AP), heavy metals, and naturally occurring radioactive material (NORM) are of particular environmental concern (Neff et al., 2011). PW can also contain large amounts of organic material, particles, inorganic salts, and low molecular weight organic acids like acetic acid and propionic acid, and can have high levels of sulfur and sulphide. Furthermore, injected water following PW can bring traces of added chemicals such as biocides, corrosion inhibitors, scale inhibitors, emulsion breakers, coagulants/flocculants and oxygen scavengers to the surface (Johnsen et al., 2004; Neff, 2002). Sulfate reducing bacteria may also be present in PW (Kaur et al., 2009). The large overall discharge volumes, the complex content of partially hazardous chemicals, and the lack of knowledge on possible long term ecological impact has made PW discharges the strongest target for concern and research in recent years.

2.1. Discharges

Summary reports on emission and discharge data for the NCS are published annually by the Norwegian Oil and Gas Association (<http://www.norskoljeoggass.no/>) based on separate reports from all oil and gas installations. In 2012 about 130 million cubic meters (m³) of PW were discharged to the NCS. The highest average daily discharge from a single field was 76 700 m³. Since 2007 the OSPAR regulation has required that dispersed oil in PW discharges shall not exceed a performance standard of 30 mg L⁻¹ (OSPAR Commission, 2001). In 2012 the average oil concentration in Norwegian PW discharges was 11.7 mg L⁻¹. Currently used cleaning

equipment seems able to reduce the levels to less than 5 mg L⁻¹ (Voldum et al., 2008).

2.2. Composition

Monocyclic aromatic hydrocarbons (BTEX: benzene, toluene, ethyl benzene, xylenes), polycyclic aromatic hydrocarbons (PAH), and related heterocyclic aromatic compounds are considered major toxicants in PW (AMAP, 2010; Neff et al., 2011). A compilation of data from field specific discharge reports for 2012 shows that the average BTEX concentrations in PW on NCS installations varied between 2 and 58 mg L⁻¹ (<http://www.norskoljeoggass.no/no/Publikasjoner/Miljorapporter/Miljorapport-2013/Feltspesifikke-utslippsrapporter-20121/>). The variation in concentrations was 0.4–6.7 mg L⁻¹ for 2- and 3-ring aromatic hydrocarbons (NPD: naphthalene, phenanthrene, dibenzothiophene and their C₁–C₃ alkylated homologs) and 0.4–12 µg L⁻¹ for 4- to 6-ring PAH (sum benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene benzo(ghi)perylene, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, pyrene).

BTEX are rarely included when considering the effects of PW since they evaporate rapidly from seawater (Neff, 2002; Neff et al., 2011; Terrens and Tait, 1996). However, for organisms in close contact with discharge points one cannot totally exclude subtle biological effects caused by chronic exposure to BTEX over a longer period. More concern has been expressed due to discharges of 2–6 ring PAHs. PAH constitute a diverse class of hydrophobic substances that are ubiquitous environmental contaminants (Harvey, 1997). Some PAH are known to be potent carcinogens and this class of contaminants is therefore given high priority for environmental pollution regulation and in risk assessment of industrial discharges. Ecotoxicological issues related to PAH have been investigated in detail for many years and have been reported in a high number of scientific papers and reviews. PAHs may cause e.g. DNA damage (Aas et al., 2000a) oxidative stress (Sturve et al., 2006), cardiac function defects (Incardona et al., 2004), or embryotoxicity (Carls et al., 2008). Fish growth may be affected by aryl hydrocarbon receptor (AhR) agonists such as PAHs (Carls et al., 2005). Some PAHs may form DNA adducts and neoplasia in fish liver through metabolic intermediates (Myers et al., 1991). A recent overview of biological effects of aromatic hydrocarbons and oil hydrocarbons has been published by AMAP (2010).

Alkyl phenols (AP) have created the greatest concern due to their documented hormone-disrupting effects (e.g. Arukwe et al., 2000, 2001; Nimrod and Benson, 1996; Soto et al., 1991). Phenol and AP are both hazardous and toxic and can cause a range of biological effects (Priatna et al., 1994). In 2012 the total amount of phenol and C₁–C₉ AP discharged on the NCS was 206 and 316 tons respectively. Naphthenic acids, another constituent of PW, have been reported to function as xeno-estrogens (Thomas et al., 2009). In 2012 the total amount of naphthenic acids discharged was 96 tons. In 2012 concentration of phenol in PW discharged from different installations on the NCS varied between 0.004 and 41 mg L⁻¹ and for C₁–C₉ AP between 0.1 and 23 mg L⁻¹. C₁–C₃ APs dominate, with lower levels of C₄–C₆ AP and very low levels of C₇–C₉ AP. Other publications have shown that concentrations of total AP typically vary between 0.6 and 10 mg L⁻¹ with phenol plus C₁–C₃ APs constituting more than 95% (Boitsov et al., 2007; Brendehaug et al., 1992; Røe, 1998; Utvik, 1999).

Metals in PW include arsenic, cadmium, copper, chromium, lead, mercury, nickel and zinc. In the 2012 reports for all PW discharges on the NCS (<http://www.norskoljeoggass.no/no/Publikasjoner/Miljorapporter/Miljorapport-2013/Feltspesifikke-utslippsrapporter-20121/>) the highest levels of lead, mercury and zinc were more than a factor 1000, and arsenic and cadmium

¹ PROOF/PROOFNY: Long-term effects of discharges to the sea from petroleum-related activities. A program under The Research Council of Norway (2002–2015).

more than a factor 100 above Norwegian coastal water background levels. The highest concentration reported for arsenic, cadmium, copper, and lead was from one low volume PW source from a gas and condensate field. If these values are excluded the levels of all metals except mercury were a factor <100 above naturally levels in seawater. Barium and iron are also exceeding background concentrations in seawater (by a factor more than 1000). In 2012 the concentration range for barium was 0.0017–1100 mg L⁻¹ and for iron 0.8–75 mg L⁻¹. The highest values are far above the solubility of these elements in seawater. Dilution and chemical processes will reduce the concentration of inorganic elements rapidly when entering the sea. Barium and iron are redox-sensitive and may precipitate upon discharge (Azetsu-Scott et al., 2007; Lee et al., 2005). Barium will precipitate as barium sulfate and iron as oxide/hydroxide. Such processes may also influence the behavior of other metals, e.g. by co-precipitation. The study by Azetsu-Scott et al. (2007) indicated three different pathways for inorganic elements: components that 1) stayed in solution would dilute along with the PW plume, 2) oxidize/precipitate to form insoluble inorganic compounds that would sink, 3) associate with oil droplets that are lighter than seawater and rise to the surface. There are a range of biogeochemical processes affecting the behavior and fate of inorganic elements in seawater, the treatment of which goes beyond this review. Monitoring studies on the NCS have only found elevated levels of trace metals in sediments collected close to the installations. This is primarily due to discharges of drill cuttings. There is no indication that the levels of trace metals in fish and shellfish collected close to offshore installations are significantly above natural background concentrations.

The most abundant NORM elements in PW are radium-226 and radium-228. PW from different installations and areas on the NCS contain low and varying levels of these elements (Gärfvert et al., 2007). Monitoring studies carried out at NCS fields have not seen any evidence for increased environmental concentrations of radium-226 (seawater, sediments, biota) caused by PW discharges.

The chemical composition of PW from the NCS has been described in many scientific papers (e.g. Durell et al., 2006; Johnsen et al., 2004; Lee et al., 2005; Neff et al., 2011; Utvik, 1999; Utvik et al., 1999). These studies show high variability in PW composition from different fields. Utvik et al. (1999) found that there was no correlation between the total hydrocarbon content (THC, present regulatory standard), and the content of aromatic compounds in PW. The toxicity of PW may be influenced by chemical partitioning and kinetics following discharge (Lee et al., 2005). Consequently, the effects of PW discharges cannot be inferred from regulatory compliance of THC alone, but must be based on field-specific and detailed chemical characterization of each PW effluent. This large variability also makes it difficult to generalize about dose-dependent biological effects of particular effluents.

2.3. Measurement of environmental levels of contaminants

It is difficult to quantify environmental concentrations of PW compounds by direct extraction with organic solvents or using absorbents, as the discharge is rapidly diluted in the receiving seawater. Various passive sampling devices have therefore been developed to provide unattended large-volume and time-integrated sampling (see review by e.g. Namiesnik et al., 2005). Back-calculation from sampler to seawater concentrations is uncertain, but reliability of the method has been greatly improved by use of performance reference compounds (PRC) (Booij et al., 2002; Huckins et al., 2002).

Harman et al. (2008a, 2008b) used a flow-through exposure system to test the uptake of APs and PAHs from seawater in SPMDs

(semi-permeable membrane devices) and POCIS (polar organic chemical integrated sampler) spiked with PRCs. SPMDs were found suitable to determine *in situ* seawater concentrations of PAHs, but were not appropriate for extraction of more polar compounds such as APs. The POCIS extracted APs more effectively except for some C₄–C₈ APs. The absence of these compounds was explained by a combination of their hydrophobic nature and rapid degradation of the n-alkylphenols. The POCIS did not provide reproducible results for low concentrations of phenol and C₁-AP due to their volatility and the presence of background contamination. Despite these limitations, the authors concluded that the combined application of SPMD and POCIS samplers improves the detection limits for PAHs and APs in seawater compared to older methods. Harman et al. (2009b) reported levels of total PAH between 32 and 49 ng L⁻¹ (SPMD) and total APs between 20 and 55 ng L⁻¹ (POCIS) out to a distance of 1 km from a NS offshore installation. By use of SPMDs and caged mussels Durell et al. (2006) estimated seawater levels of total PAH in the range 5–37 ng L⁻¹ within 1 km distance from the same NS installation.

Results from field and laboratory studies have shown that levels of APs in fish muscle and liver tissue are very low, often below detection. One reason is that both PAHs and APs are rapidly metabolized by vertebrates. Analysis of tissue concentrations of parent compounds is therefore of limited value when assessing exposure to PW contaminants in fish around rigs. Since the early 1980s analysis of PAH metabolites in fish bile has been used to assess exposure to PAHs (e.g. Aas et al., 2000b; Krahn et al., 1986; McDonald et al., 1995). Sundt et al. (2009) used radio-labeled APs to demonstrate that the concentrations of APs in liver were low whereas AP metabolites were mainly present in the bile. Reviews of methods to determine contaminant metabolites in fish bile have recently been published; for PAH by Beyer et al. (2010) and for APs by Beyer et al. (2011, 2012).

Quantitative analysis of PAH and AP metabolites in bile is useful in integrated monitoring systems as it indicates both chemical contamination and a biological response. The relationship between exposure to PW or oil and levels of PAH and AP metabolites in bile has been studied in several laboratory experiments with Atlantic cod (*Gadus morhua*) (Grung et al., 2009; Skadsheim et al., 2009) and other fish species (Jonsson and Björkblom, 2011). Grung et al. (2009) found a dose and lipophilicity dependent relationship of bile metabolite levels of specific PAHs and APs in Atlantic cod exposed to seawater containing a simulated PW mixture for 2 and 8 months. The relationship was stronger in fish exposed to PW compounds in the water than in the diet, and less clear for the more volatile substances such as phenol and naphthalene. Most of the compounds were detected as monohydroxy-metabolites. Sundt et al. (2009) found that the bioconcentration of four radio-labeled APs in Atlantic cod was ten times higher from water-borne exposure (8 ng L⁻¹) than from absorption through the gut wall following food-borne exposure (5 µg kg⁻¹). Skadsheim et al. (2009) and Jonsson and Björkblom (2011) found that PAH metabolites in different fish species exposed to dispersed crude oil correlated both with exposure parameters (PAHs and THC) and effects (DNA adducts).

Sundt and Björkblom (2011) detected elevated levels of AP metabolites in the bile of Atlantic cod exposed to 0.125% PW. Meier et al. (2010) found that Atlantic cod embryos, larvae up to 3 months of age, and juveniles from 3 to 6 months of age exposed to 0.01, 0.1, and 1% PW accumulated APs dependent on dose and developmental stage. Such dilutions are typically encountered between 50 m and 1 km from a PW outfall (Meier et al., 2010). Sundt et al. (2011) and Brooks et al. (2011b) detected a significant increase in bile metabolite levels of APs in Atlantic cod caged for 6 weeks about 200 m from a NS PW outfall.

Juvenile Atlantic cod are able to effectively metabolize and excrete short chain APs (Meier et al., 2010). Tollefsen et al. (1998) found that heptylphenol (4-n-HEPP) accumulated rapidly in most tissues of juvenile Atlantic cod. Depuration was also rapid with an estimated half-life of 13 h. This corresponds well with the half-lives of 10–20 h observed for APs in Atlantic cod tissue (Sundt et al., 2009), and to earlier studies with other fish species (Arukwe et al., 2000; Pedersen and Hill, 2002). Therefore, elevated levels of AP metabolites in offshore caged fish indicate recent exposure to APs.

2.4. Environmental effects

2.4.1. Field studies

Monitoring surveys focusing on the effects of PW were first performed on the NCS in 1997 and surveys have been repeated almost annually up to present (Bakke et al., 2011; Brooks et al., 2011a,b; Durell et al., 2004, 2006; Hylland et al., 2008; Neff et al., 2006; Nilssen and Bakke, 2011; Sundt et al., 2011). The present strategy is based on the results from the international BECPELAG (Biological Effects of Contaminants in Marine Pelagic Ecosystems) workshop (Hylland et al., 2002). The surveys cover one selected field each year and comprise direct measurements and estimates of levels of PW compounds in the water column (Harman et al., 2009a, 2009b, 2010) as well as analysis of contaminant body burden and biomarkers in Atlantic cod and blue mussel (*Mytilus edulis*) caged for 6 weeks at various distances from the PW outlet (Brooks et al., 2011b; Hylland et al., 2008; Sundt et al., 2011). In mussel the information of PAH body burden has been combined with biological responses such as lysosomal membrane stability (LMS, neutral red retention time assays) DNA-damage (micronuclei frequency) and digestive gland histochemistry (Brooks et al., 2009). In cod the exposure conditions indicated by biliary PAH metabolites have been linked to cytochrome P450 1A protein (CYP1A) responses and formation of DNA adducts (Aas et al., 2000a, 2001; Sanni et al., 2005; Skadsheim et al., 2009). Other parameters analyzed in cod include biliary AP metabolites, vitellogenin, zona radiata protein, glutathione S-transferase and gill histopathology. The surveys have mostly detected exposure to PAH and AP from PW and biomarker responses no further than 0.5–1 km from the discharge points, but in one survey effects out to 1.6 km were detected (Sundt et al., 2008). Corresponding biliary PAH metabolites and biomarker responses in wild fish caught within 100 m from three Australian offshore platforms where the PW comes from a heavy crude oil also suggest that the effects were local as no effects were detected at 5 km distance (Gagnon, 2011). There is, however, a concern that current methods are not sensitive enough to reveal subtle effects further out. Also, few of the biomarker endpoints look beyond the compensatory capacity of the organisms, and the significance of these responses for the fitness and survival of the organisms is still debated. Extrapolation from short-term biomarker effects in individual organisms to long term effects on populations and communities is inherently difficult (Forbes et al., 2006), and the conclusion that impacts are largely local is still unverified (Wells, 2005).

Some fish species seem to be attracted to production platforms. Jørgensen et al. (2002) showed that about half the cod tagged near an NS platform remained there or around neighboring platforms. Gill net catches have been bigger near platforms than further away (Løkkeborg et al., 2002). Monitoring studies on free living fish in the NS have shown interesting results with respect to effects on biomarkers. Samples collected in 2002 from two areas with extensive oil and gas production showed induction on biotransformation enzymes, oxidative stress, altered fatty acid composition, and genotoxicity in natural populations of haddock (*Melanogrammus*

aeglefinus) (Balk et al., 2011; Grøsvik et al., 2010). Genotoxicity was reflected by a hepatic DNA-adduct pattern typical for exposure to a mixture of PAHs. Atlantic cod showed similar, but less pronounced responses. Repeated monitoring in 2005, 2008 and 2011 confirmed this pattern, although with weaker genotoxic signals in haddock from the northern NS (Tampen area). It is still not clear whether the effects are caused by PW contaminants, contaminated drill cuttings, smaller oil spills, or a combination of these sources (Hylland et al., 2006). These findings as well as results from caging experiments have shown that individual fish can be affected sublethally in several ways by PW discharges (Brooks et al., 2011b). Whether these biomarker effects can result in effects at the population level depends on what percentage of the total population is exposed sufficiently, which again depends on PW contaminant levels, the total rate of discharge, plume dilution pattern, and population distribution.

2.4.2. Endocrine and reproductive effects

APs can affect a number of reproductive parameters in fish, including gonadal development (Meier et al., 2007b), induction of plasma vitellogenin (Vtg) in male and juvenile fish (Jobling and Sumpter, 1993; White et al., 1994), inhibition of spermatogenesis (Gimeno et al., 1998; Jobling and Sumpter, 1993; Miles-Richardson et al., 1999; Weber et al., 2002), and oogenesis (Tanaka and Grizzle, 2002; Weber et al., 2003). Tollefsen et al. (2007) and Tollefsen and Nilsen (2008) found that APs were able to bind to plasma sex steroid-binding proteins (rtSBP) in rainbow trout (*Oncorhynchus mykiss*). The highest affinity was seen for mono-substituted APs with 4–8 carbon chain length, but this was still 10^4 – 10^6 times lower than the affinity for the natural sex steroid 17β -estradiol (E2). The results suggested that endocrine disruption may occur after exposure to realistic concentrations of APs and a variety of other PW compounds. Tollefsen et al. (2006) further showed that chemicals in solid phase extracts of PW were able to displace E2 from the rtSBP and induce estrogenic effects. The bioactive chemicals were not identified.

Tollefsen et al. (2011) demonstrated that complex mixtures of oil-related compounds could modulate the endocrine physiology of Atlantic cod. Fish were exposed to either diluted PW (0.5% and 0.1%), dispersed oil (0.2 mg L^{-1}), or artificial PW water mixed with nine low to medium molecular weight APs and PAHs. The total sex-steroid binding capacity was up-regulated in the blood of female cod, indicating interference with blood steroid transport. Induction of plasma Vtg was not found, although the number of males and females with elevated Vtg was higher in certain exposure groups than in the control group. General health parameters such as gonadosomatic, hepatosomatic or fish condition index were not affected, which suggests that the endocrine disrupting effect was too low to elicit clear physiological or growth effects. When exposing late larvae and juveniles of Atlantic cod to PW Meier et al. (2010) found that individuals exposed to 1% PW had significantly higher levels of Vtg and CYP1A in plasma and liver, respectively. No similar effects were seen at exposure to 0.1% and 0.01% PW.

Serious reproductive disturbance was demonstrated by Meier et al. (2007b) in first-time spawning Atlantic cod that were force fed a paste containing C4–C7 APs. Total AP doses during 1 and 5 weeks were 0.02 – 80 mg kg^{-1} body weight. Treatment impaired oocyte development, reduced estrogen levels, and delayed spawning by 17–28 days in female fish. Male fish demonstrated reduced 11-keto-testosterone, a small induction of Vtg levels, and impaired testicular development with increased amount of spermatogonia and reduced amount of spermatozoa. Extended exposure (14 weeks) to the lowest dose ($0.02 \text{ mg AP kg}^{-1}$) gave similar results (Meier et al., 2011). These exposure levels are difficult to compare with real-life exposure to PW plumes, especially since

many endocrine disruptors seem not to produce linear dose–response curves (Vandenberg et al., 2012), but the authors themselves consider the exposure level higher than what is realistic, possibly demonstrating a worst-case disturbance of reproductive fitness in the cod. Also, Sundt and Bjorkblom (2011) recorded impaired oocyte development and reduced estrogen levels in pre-spawning female Atlantic cod, as well as altered testicular development, an increase in the amount of spermatogonia and primary spermatocytes, and a reduction in the amount of mature sperm in males following exposure to realistic concentrations of PW (0.066–0.2%) for twelve weeks. Therefore, one cannot exclude that APs in PW effluents under certain circumstances could cause reproductive disturbance in sensitive stages (e.g. pre-spawning) of wild fish that stay close to offshore platforms for long periods of time. However, it seems unlikely that this could affect a significant fraction of Atlantic cod populations.

Estrogens are involved in many biological processes, including control of gonad maturation in male and female fish. The enzyme cytochrome P450 aromatase converts androgens, like testosterone or androstenedione to estrogen (E2) and estrone. Teleost fish have two aromatase genes; one that is mainly expressed in the gonads (aromatase A or *cyp19a1a*), and one that is mainly expressed in the brain (aromatase B or *cyp19a1b*) (Diotel et al., 2010). Meier et al. (2011) did not find any regulation of *cyp19a1a* in the ovary (mRNA expression or enzyme activity), or of aromatase activity in the brain of female cod exposed to AP or PW. The specific activity of aromatase in the ovary was therefore not affected by the AP-exposure.

Tollefsen et al. (2007) and Thomas et al. (2009) used recombinant yeast estrogen and androgen screens to determine the *in vitro* estrogen receptor (ER) agonist and androgen receptor (AR) antagonist potencies of solid phase extracts (SPE) of PW collected from 20 Norwegian installations. They found estrogenic activities at levels equivalent to <0.1–4 ng L⁻¹ E2 (dependent on PW source), similar to those previously reported for the UK continental shelf (UKCS) (Thomas et al., 2004). No activity was detected after exposure to filtered oil droplets from PW suggesting that ER activity was primarily associated with the dissolved phase. Thomas et al. (2009) identified short-chain petrogenic APs to be responsible for around 35% of estrogen receptor (ER) agonist activity measured *in vitro*. Androgen receptor (AR) antagonists were detected both in the dissolved and oil associated phase. They also reported that naphthenic acids, which occur in significantly higher concentrations than C₄–C₇ APs in PW, were weak ER agonists. Naphthenic acids accounted for much of the 65% of the “unknown” ER agonist potency in the PW samples tested, and further disrupted the binding of AR agonists to the androgen ligand receptor. PAHs were also reported to be AR antagonists. The study indicated that these petrogenic compounds are responsible for most of the ER and AR mediated activity in PWs.

In summary, these studies document that compounds present in PW have a potential to exert endocrine effects in fish. The experimental exposure levels studied cover a range of PW concentrations that are typically found in close proximity to PW discharge points. They might therefore elicit effects on fish standing close to platforms. Meier et al. (2010) still concluded that widespread and long lasting xenoestrogenicity and reproduction effects of PW on the population level in fish are unlikely. This was also supported by Sundt et al. (2011) who compared data from PW-exposed fish in the laboratory to similar data from Atlantic cod caged at the Ekofisk oil field in the NS. No Vtg induction was observed in fish exposed experimentally to PW in the dilution range 0.125%–0.5% PW giving 2.6–11 mg L⁻¹ AP metabolites in the fish bile. Levels of the corresponding APs in the water ranged from 3.0 to 9.7 µg L⁻¹. In fish caged about 200 m from the large Ekofisk PW outfall (average rate 37 000 m³ day⁻¹) the AP metabolite levels were significantly

elevated compared to control cages, but still one order of magnitude lower than in bile from the lowest exposure concentration in the laboratory experiment. It was therefore not possible to determine a LOEC (Lowest Observable Effects Concentration) for AP metabolites from these studies. Since LOEC must be higher than the highest observed NOEC of 11 mg L⁻¹ AP metabolites, and the AP metabolite levels in the caged cod were only a fraction of this, the AP content in the Ekofisk PW discharge was well below a critical level for induction of Vtg. Still, the critical level for induction of Vtg is probably not far above these cited values, which is supported by Tollefsen et al. (2011) who found elevated Vtg levels in 72% of individual male Atlantic cod exposed to 21 µg L⁻¹ of sum C₁–C₅ APs.

Meier et al. (2011) showed that oral exposure to a mixture of 4 APs affected the endocrine system and gonad development in cod through changes in the hypothalamic-pituitary-gonadal (HPG) axis at doses that were much lower than those that resulted in Vtg induction. So, although Vtg is a sensitive parameter for detection of endocrine disruption, lack of response in Vtg alone does not exclude that the endocrine system in fish may be disturbed by PW components.

Compelling evidence thus exists from *in vitro* bioassays that PW contains estrogenic compounds (Thomas et al., 2004, 2009; Tollefsen et al., 2007) and that 0.5–1% dilutions of PW induce Vtg in juvenile cod (Meier et al., 2010; Sundt et al., 2011). Concentrations of APs lower than those that induce Vtg may also cause severe endocrine disruption, as demonstrated for the HPG-axis in fish by Meier et al. (2011). It is further demonstrated that PW contains other compounds that might have estrogenic effects such as naphthenic acids (Thomas et al., 2009). On the other hand, *in vivo* studies showed no effect on gonad maturation or the ratio of juvenile to mature females after long-term exposure of Atlantic cod to low levels of selected PW compounds in the laboratory (Holth et al., 2010). Risk assessment by Beyer et al. (2012) also concluded that the environmental exposure of fish to APs from PW is most probably too low to induce endocrine disruption to an extent that causes significant effects on the reproduction in NS fish stocks. This assessment takes into account that PW discharges offshore are rapidly diluted, which reduces the risk of population effects, and is supported by results from the monitoring of caged fish exposed to PW offshore where no endocrine effects based on Vtg measurements have been detected (Brooks et al., 2009).

2.4.3. Non-endocrine effects

APs are known to induce hydroxyl and oxygen radical generation (Fujisawa et al., 2002; Obata and Kubota, 2000; Okai et al., 2000), but the effects on the redox status in fish are unclear. Hasselberg et al. (2004) studied the oxidative stress response to APs in Atlantic cod by measuring amounts of hepatic glutathione and hepatic activity of glutathione reductase (GR), glutathione S-transferase (GST), and glucose-6-phosphate dehydrogenase (G6PDH). The total glutathione concentration in female cod increased in response to 1-week of feeding with an AP-containing diet, an effect not seen after 4 weeks of feeding. Male fish had higher levels of glutathione than females. Increased GR activity was seen in both males and females after 4 weeks of exposure to a weekly dose of 0.02 mg AP kg⁻¹ body weight. GST activity was affected only in males exposed for 1 week, and G6PDH activity increased only in females after 1 week exposure. The results provide evidence that APs may affect the redox status in Atlantic cod through increased oxidative stress and stimulated GSH dependent detoxification.

When exposing rainbow trout hepatocytes to the water soluble (by SPE) and particulate organic (by glass wool filtering) fractions of PW from 10 different NCS oil producing installations Farmen et al. (2010) recorded a concentration-dependent increase in reactive

oxygen species (ROS) after 1 h exposure, and changes in levels of total glutathione and cell death after 96 h. The water soluble fraction (WSF) apparently contained most of the toxic potential, as was also seen by Tollefsen and Nilsen (2008), but in some cases the particulate fraction, containing mainly oil droplets, was equally toxic. The effects were not correlated to the total oil content in the PW. The levels of PAHs and APs varied by a factor of 10 and 60 respectively among the different PW sources tested, and the exposure concentrations were not clearly stated. The authors concluded that the concentrations needed to elicit oxidative stress responses in fish hepatocytes appeared to be orders of magnitude higher than concentrations found around representative NS oil platforms.

Holth et al. (2010) exposed Atlantic cod for 11 months to artificial PW containing APs, PAHs and phenol at high (PAH $5.4 \mu\text{g L}^{-1}$; AP $11.4 \mu\text{g L}^{-1}$) and low (PAH $0.54 \mu\text{g L}^{-1}$; AP $1.14 \mu\text{g L}^{-1}$) concentrations. Exposure was continuous as well as 2 weeks pulsed mode for the high concentration. A range of toxicologically relevant genes were differentially expressed following exposure, including AhR-responsive genes (CYP1A, UDP-GT) and genes relevant to immune function (complement C3, MHC 1, CYP27B), apoptosis (PERP), and oxidative stress (hepcidin, serotransferrin, glutathione peroxidase). Estimated spawning time was significantly delayed in the exposed females, but not in relation to dose. Gross health parameters (condition factor, liver somatic index, gonadosomatic index, and hematocrit), frequency of micronucleated erythrocytes, oxidative stress in whole blood, and survival were not affected. Holth et al. (2011) reported reduced LMS of head kidney cells after two weeks at the highest concentration. The LMS reduction was dose related over the whole 11 months period and did not adapt to the exposures. No differences in peroxisomal proliferation, measured as acyl-CoA oxidase activity in head kidney, were detected between treatments, although gender differences and change over time were observed in acyl-CoA oxidase activity. In conclusion, LMS in head kidney cells appeared to be a sensitive biomarker for exposure of Atlantic cod to oil related compounds.

Induction of the cytochrome P-450 detoxification enzyme system after exposure to oil and other organic contaminants has been amply documented. Elevated hepatic CYP1A activity was found in Atlantic cod caged for 6 weeks about 200 m from the PW outfall at the Ekofisk oil field both in 2008 (Sundt et al., 2008) and 2009 (Brooks et al., 2009). Hasselberg et al. (2004) showed that force feeding of Atlantic cod for 4 weeks with a paste containing 0.02–80 ppm of a mixture of four different APs induced a slight dose-dependent increase of hepatic CYP1A activity in females, but not in males. The increase was not reflected in the CYP1A-mediated EROD (ethoxyresorufin-O-deethylase) activity, implying that APs inhibited the CYP1A enzyme activity *in vivo*. *In vitro* studies with pooled liver microsomes from Atlantic cod confirmed the inhibition, and that the APs also inhibited CYP3A enzyme activity *in vitro*, but to a lesser extent. Such inhibition complicates the interpretation of cytochrome P-450 detoxification enzyme responses in the monitoring of PW discharges. Increase in hepatic CYP1A activity was also seen by Meier et al. (2010) exposing early juvenile Atlantic cod (3–6 months of age) to 1% PW for 3 months. Sundt et al. (2011) exposed Atlantic cod to PW in laboratory and field experiments and found CYP1A induction after exposure to 0.25% PW in the laboratory. Abrahamson et al. (2008) used an *ex vivo* gill EROD assay in Atlantic cod as a biomarker for CYP1A-inducing compounds in NS crude oil and PW. Exposure of cod to fairly high nominal concentrations of dispersed crude oil (1 and 10 mg L^{-1} THC) for 24 h induced a concentration-dependent EROD activity. The same was found following 14 days of exposure to typical near-zone concentrations of PW (0.5% and 0.1% PW) and dispersed crude oil (0.2 mg L^{-1}). On the other hand, EROD activity was not induced in

cod caged for 6 weeks between 500–10 000 m from two NCS platforms (Abrahamson et al., 2008). Jonsson and Björkblom (2011) compared hepatic CYP1A enzyme activity in Atlantic halibut (*Hippoglossus hippoglossus*), turbot (*Psetta maxima*), long rough dab (*Hippoglossoides platessoides*), Atlantic salmon (*Salmo salar*), and Atlantic cod exposed to dispersed crude oil ($0.3\text{--}9.1 \mu\text{g L}^{-1}$ PAHs) for 4 weeks. CYP1A activity was induced in all species except sprat. The activity level varied with species and concentration level.

Changes in the hepatic lipid composition following exposure to crude oil have been reported for Atlantic cod and winter flounder (*Pseudopleuronectes americanus*) (Dey et al., 1983). Meier et al. (2007a) studied changes in the fatty acid profile and cholesterol content in membrane lipids from liver and brain tissues in Atlantic cod after 5 weeks of force feeding with AP containing paste. APs altered the fatty acid profile of polar lipids in the liver towards more saturated fatty acids (SFA) and less n-3 polyunsaturated fatty acids (n-3 PUFA). A similar effect was found in the brain, although with elevated SFA content in the neutral lipids (mainly cholesterol ester), but not in the polar lipids. The AP exposure also caused a decline in the cholesterol levels in the brain. Changes in hepatic lipid composition were also reported by Grøsvik et al. (2010) in free-living Atlantic cod and haddock caught in the vicinity of the Tampen area, a northern NS region with very high petroleum activity. Haddock from Tampen had lower hepatic lipid content than haddock from other NCS regions. Also, the fatty acid profiles had relatively high levels of arachidonic acid (20:4; n-6), and the ratio between omega-3 and omega-6 polyunsaturated fatty acids was significantly lower in neutral lipids, free fatty acids, phosphatidylcholine and phosphatidylethanolamine compared with haddock from other regions. The lipid alterations may have been caused by exposure to PW, oil, or contaminated drill cuttings. The biological implication, significance and reversibility of these fatty acid alterations are not yet understood. Widdows et al. (1987) found complete recovery within 55 days in blue mussel that had digestive gland lipid changes and heavy digestive disorder (Lowe and Pipe, 1987) after 8 months of exposure to 28 and $125 \mu\text{g L}^{-1}$ dispersed diesel oil. This may suggest that the changes in the lipid composition seen in the haddock from Tampen would also be reversible if exposure ceases.

Tollefsen et al. (2008) studied the cytotoxicity of a range of APs in cultures of primary hepatocytes from rainbow trout. Toxicity measured as metabolic inhibition and loss of membrane integrity increased with the hydrophobicity of the APs for compounds with $\log K_{OW} < 4.9$, but deviated from this for more hydrophobic compounds ($\log K_{OW} > 4.9$). Metabolic inhibition occurred at lower concentrations than loss of membrane integrity for most of the APs, which suggests that effects on cellular metabolic functions were the main causes of the cytotoxicity. The study gives insight into the structure–toxicity relationship of important PW components, but it is difficult to extrapolate to real PW exposure. Still, for chemicals with $\log K_{OW} < 2\text{--}3$ the metabolic inhibition and to a lesser degree also loss of membrane integrity was claimed to correspond to reported *in vivo* acute toxicity in fathead minnow (*Pimephales promelas*) (Schultz et al., 1986). The *in vitro* toxicity of the more hydrophobic compounds underestimated the *in vivo* toxicity in this fish.

Meier et al. (2010) found that exposure of Atlantic cod to PW during the embryonic and early larval stages (up to 3 months of age) and during the early juvenile stage (from 3 to 6 months of age) had no effect on embryo survival or hatching success, but 1% PW interfered with the development of normal larval pigmentation. After hatching most of the larvae exposed to 1% PW failed to begin feeding and died of starvation. This inability to feed may be linked to an increased frequency of jaw deformities in the exposed larvae. No similar effects were seen at exposure to 0.1% and 0.01% PW.

Analysis of DNA adducts in fish tissue has been recommended for assessment of genotoxic effects of contaminants in PW (Balk et al., 2011; Hylland et al., 2006). Similarly, the micronuclei frequency method has been found sensitive and feasible for use as a biomarker of genotoxicity in blue mussel exposed to PW contaminants (Brooks et al., 2009). Holth et al. (2009) found time and dose dependent formation of DNA adducts in Atlantic cod exposed for 44 weeks to APs and a WSF of oil. Elevated DNA adduct values have been measured in wild haddock in the Tampen region in 2002, 2005 and 2008 (Balk et al., 2011; Grøsvik et al., 2010; Hylland et al., 2006). The cause of the effect was unclear, as the DNA adduct signal could possibly stem from recent PW discharges or from fish being in contact with PAHs or other contaminants in deposits of drill cuttings. Monitoring surveys at the Ekofisk field have detected elevated micronuclei frequencies in blue mussel caged up to 1.6 km from the discharge point (Sundt et al., 2008). After implementation of a new PW treatment system elevated micronuclei frequencies were only detected in cages at 500 m distance (Brooks et al., 2009).

Brooks et al. (2011a) studied the biological impact of treated PW under laboratory conditions in the blue mussel. Mussel health status was assessed using an integrated biomarker approach in combination with chemical analysis of both water (with SPMDs), and mussel tissues. Acyl-CoA oxidase activity, neutral lipid accumulation, catalase activity, micronuclei formation, LMS in digestive cells and hemocytes, cell-type composition in digestive gland epithelium, and the integrity of the digestive gland tissue were measured after 5 week exposure to 0.01%–1% PW. Significant sublethal responses were found at 0.01–0.5% PW, even though individual chemical compounds of PW were at extremely low concentrations in both water and mussel tissues.

The studies above show that exposure to PW may cause a range of non-endocrine and partly dose-dependent effects in fish and invertebrates. Several of these responses are compensatory, such as responses to oxidative stress and xenobiotics, and should not necessarily cause biological dysfunction or affect survival unless their capacity is chronically exceeded. Others suggest more profound effects on the individual, such as loss of membrane integrity, cytotoxicity, gene expression changes, DNA adducts, hepatic lipid composition, and reproductive disorder (spawning time shift, larval survival). One common feature seems to be that the effects are triggered only at exposure for weeks to months and at less than 100–1000 times dilution of the PW concentrations. Even large PW plumes will rapidly become more diluted than this, hence damaging exposure is unlikely. Field data also strongly suggest that the biomarker effects are local. An exception is the responses in wild haddock caught away from platforms in areas with high petroleum activity (Balk et al., 2011; Grøsvik et al., 2010). It is more likely that these effects were due to fish migrating after local exposure rather than from low exposure at the distance where the fish were caught. The results do not suggest that a significant part of the fish populations would be affected in this way, but this cannot be verified.

2.4.4. Use of “-omic” approaches

Establishing links between sub-individual responses to contaminants and higher level effects on individuals and populations is an important yet unresolved challenge. To assess if such links exist and are predictable it is necessary to increase the mechanistic understanding of the biological effects related to PW exposures and to develop means to screen large number of wild organisms for effect signals. Techniques have recently been developed to screen cells or tissues for their total fingerprint of selected compounds such as genes (genome), RNAs (transcriptome), proteins (proteome), and total metabolites (metabolome) (see review by Karlsten et al. (2011) on proteome responses to various contaminants). It

is believed that studies of responses in “omic” patterns would not only increase our understanding of the mechanisms involved in the responses of marine organisms to toxicants, but could also lead to identification of biological molecules or molecular patterns that are sensitive, regulate important biological functions, and can be used to screen large numbers of individuals.

Bohne-Kjersem et al. (2009, 2010) studied protein changes in plasma of juvenile Atlantic cod and in fertilized Atlantic cod eggs and fry. The juveniles were exposed to dispersed NS crude oil (0.06–1.0 mg oil L⁻¹) and a surrogate PW (the 1.0 mg L⁻¹ crude oil spiked with APs and PAH). Similarly, eggs and subsequent fry were exposed to 0.01, 0.1, and 1% NS PW for about 90 days. In juvenile cod 137 proteins were differentially expressed due to exposure, and 40 of these at the lowest exposure (Bohne-Kjersem et al., 2009). Twenty-nine proteins were identified, and a total of 14 proteins were considered potential biomarker candidates. These proteins are linked to a wide range of biological systems and processes including fibrinolysis and the complement cascade, the immune system, fertility, bone resorption, fatty acid metabolism, oxidative stress, impaired cell mobility, and apoptosis. Several responses were interlinked, suggesting that an array of biomarkers may give a better indication of the adverse effects in fish than single biomarkers. Also, in exposed cod eggs many of the protein changes occurred at the lowest exposure, including structural, cytoskeletal, and signaling proteins regulating muscle development, rod/retina function, cellular signaling, and tissue integrity of the fry. These are important for swimming and predator escape. The changes indicate that PW can affect liver functions such as cellular integrity, signal transduction and metabolism. This supports earlier indications (e.g. Meier et al., 2010) that effects of PW at low doses on cod fry are mainly non-estrogenic.

Karlsten et al. (2011) compared the proteome changes in cod fry and juveniles based on studies on protein changes in brain, liver, and plasma of juvenile Atlantic cod following exposure to PW and surrogate PW. Proteome changes in fry seemed more linked to morphological changes and disturbances of cod development, whereas the changes in the proteome of juvenile cod seemed to reflect functions important for vitality. This might reflect difference in responses between different developmental stages, but it could also be explained by difference in function between tissues.

In another study with juvenile Atlantic cod exposed to crude oil, 17 β -estradiol (E2) and 4-nonylphenol Nilsen et al. (2011) investigated the suitability of the SELDI-TOF MS (Surface-Enhanced Laser Desorption/Ionization Time-Of-Flight Mass Spectrometry) technique for screening of protein biomarkers in plasma indicating exposure to estrogenic compounds. Protein expression analysis revealed that 13 plasma peaks were significantly altered in response to the E2 treatment, and found reproducible when re-analyzed six months later. Antibody-assisted SELDI-TOF MS identified two possibly E2-responsive peaks. These were identified as fragments of the well-known biomarkers Vtg and/or Zrp. A plasma proteomic profile assumed to be specific for AP exposure was also found in female Atlantic cod.

The recently completed genome sequence of Atlantic cod (www.codgenome.no) has opened up the possibility of a systems biology approach to elucidate the molecular mechanisms of toxicity. Karlsten et al. (2011) attempted to map and understand genomic responses in cod to PW contaminants by combining data generated from proteomics- and transcriptomics analyses to concurrent searchable EST – (expressed sequence tags) and genomic databases. Such an interdisciplinary study may open up new possibilities of gene annotation and pathway analyses.

Gene transcription and other molecular responses relevant to offshore discharges have been studied in the copepods *Calanus finmarchicus* and *Calanus glacialis* kept in multi-generation cultures

(Hansen et al., 2007, 2008a, 2008b, 2009, 2010, 2011). They found that dissolved and dispersed crude oil, naphthalene and copper modulated the expression of genes involved in fundamental biological functions such as feeding, ecdysis, lipid storage and metabolism, amino acid and protein metabolism, cellular detoxification and antioxidant systems. These genomic biomarkers may therefore have a potential for use in oil and gas related effect monitoring of zooplankton.

The application of “omic” techniques is still in its infancy and clearly more research is required to clarify to what extent causative patterns are linked to specific discharge factors and also to assess their applicability as screening tools in practical monitoring.

3. Drilling waste

Waste from borehole drilling consists of crushed rock cuttings from the borehole and remnants of drill mud. The function of the mud is to lubricate and cool the drill bit, stabilize the borehole, control pressure, and bring cuttings to the platform. Drilling waste also comprises used drill mud that has lost its technical properties.

3.1. Composition and discharges

The major components of drill muds are a liquid (water, oil, or another organic fluid) and a weighting material (typically barite, BaSO₄). Various additives are used to improve the technical performance of the mud. Among these are viscosifiers (e.g. polyacrylates, and other organic polymers), emulsifiers (e.g. alkylacrylate sulphonate and polyethylene oxide), pH and shale control agents, and defloculants (Davies and Kingston, 1992). The additives vary between drilling operations and in the course of the drilling. Three main types of drilling mud are recognized based on the type of base liquid, water based muds (WBM) containing usually seawater as the base liquid, oil based muds (OBM) with either diesel oil or low-aromatic mineral oil as the base liquid, and synthetic muds (SM) using other types of “pseudo-oil” organic liquids such as ethers, esters, olefins or vegetable oils. OBM and SM are used to improve lubrication and stabilization in the borehole, especially during non-vertical drilling. The chemicals used as additives in the drill muds today are mostly classified as PLONOR (Pose Little or No Risk to the Environment) by OSPAR Commission (2012).

Cuttings with diesel OBM were discharged extensively from NS drilling operations until 1984, but diesel oil was then replaced by low-aromatic oils being less toxic to both workers and the outer environment. Typical oil content of OBM cuttings at discharge was in the range 5–15% or more (Breuer et al., 2004; Davies et al., 1989). The total amount of oil discharged to the NS with cuttings was 25 000 tons in 1985, decreasing to 13 000 tons in 1990 (North Sea Task Force, 1993). A tightening of the discharge control of OBM cuttings, in Norway in 1993 and in the OSPAR area in 1996 and 2000 (OSPAR Commission, 2000), setting the discharge limit of oil adhered to cuttings at not more than 1%, effectively eliminated this discharge. OBMs were partially replaced by SMs being less toxic and, for ester and olefin SMs, also more biodegradable under aerobic conditions (Schaanning and Bakke, 1997). Since SM cuttings proved not to be environmentally superior to cuttings with OBM and in particular had a negative effect on sediment oxygen conditions, SM was gradually phased out. Due to tightened regulations (OSPAR Commission, 2000) SM cuttings have rarely been discharged to the NS after 2001. Today only WBM cuttings and spent WBM are allowed for discharge in the NS. Total quantities of WBM cuttings discharged on the NCS peaked in 2010 at 200 000 tons (Norwegian Oil and Gas, 2012). In 2012 around 80 exploration and production wells were drilled on the NCS and approximately

172 000 tons of cuttings were discharged at sea (Norwegian Oil and Gas, 2013).

3.2. Environmental effects

3.2.1. Impact of old cuttings piles

Before the regulations in 1993/1996 large volumes of cuttings heavily contaminated with OBM and SBM piled up on the seafloor beneath and around the rigs causing widespread sediment contamination and effects on the benthos. At some NS fields hydrocarbon contamination of the sediments extended out to 5–10 km distance (Davies and Kingston, 1992; Kingston, 1992; Reiersen et al., 1989; Stagg and McIntosh, 1996; Ward et al., 1980) and changes in the benthic macrofauna could be traced out to 2–5 km or more (Bakke et al., 1989; Gray et al., 1999; Olsgard and Gray, 1995; Reiersen et al., 1989). Large cuttings piles are still present in the northern and central part of the NS, and may have volumes of up to 45 000 m³, a height of up to 25 m, and a footprint of more than 20 000 m² (Bell et al., 2000; Breuer et al., 2004; Kjeilen et al., 2001). In the southern NS the cuttings have not formed extensive deposits due to strong tidal and storm driven currents. An inventory of cuttings piles present in the North Sea (Park et al., 2001) identified 79 large (>5000 m³) and 66 small (<5000 m³) piles on the UKCS and the NCS. The total hydrocarbon concentration measured in NS piles is in the range 10 000 to 600 000 mg kg⁻¹ (Bell et al., 2000; Breuer et al., 2004; Park et al., 2001; Westerlund et al., 2001). Aerobic biodegradation of the hydrocarbons occurs only in the upper few millimetres. Anaerobic degradation may take place down to at least 20–50 cm, but only very slowly (Brakstad and Ramstad, 2001; Breuer et al., 2004). The oil in deeper parts of the piles seems to be essentially unchanged (Breuer et al., 2004). Many studies cover toxicity of individual OBM and SM components and of complete mud formulations (see e.g. Altin et al., 2008; Frost et al., 2006; Kingston, 1987; Neff, 1987; Roddie et al., 1999). Toxicity seems to be determined primarily by the hydrocarbon content (Conklin et al., 1983; Grant and Briggs, 2002), but mud chemicals and heavy metals from impurities in the barite may add to this. There is also a concern that biodegradation and other diagenetic processes in the piles over the years may have produced other potentially toxic compounds such as complex esters and organic acids which until recently could not be identified analytically (see Rowland et al., 2011). Little is known of *in situ* toxic effects as toxicity is confounded by other stressors and biological interactions. In a field experiment Bakke et al. (1986a) ranked the main mud types in order of decreasing toxicity in standard bioassays as diesel-OBM, low-aromatic OBM, and WBM. This order was the same after 9 months in trays on the seabed. In the same field experiment Bakke et al. (1986b) found almost no macrofauna recolonization over a 2 year period on defaunated sediments capped with diesel and low-aromatic OBM cuttings, which suggests that also other factors than the aromatic hydrocarbons impaired recolonization. After 5½ years on the seafloor the fauna development was still very much reduced in sediments that had been capped with 10 mm of diesel and low-aromatic OBM cuttings (Bakke et al., 1989). During this time 70% of the total hydrocarbons had disappeared from the caps, but the levels were still high (27 000–30 000 mg kg⁻¹). Besides chemical toxicity factors such as grain size deviation and hydrogen sulphide content may retard fauna recovery, especially close to or on the piles. Bakke et al. (1986b) found that fauna recolonization on sediments capped with 10 mm WBM cuttings differed little in overall diversity from that on natural sediment after 1 year, but the species composition was clearly different, which was thought to be due to the WBM cuttings being classified as ‘very fine sand’ as opposed to the natural sediment being ‘medium sand’.

Cuttings piles seem resistant to chemical change (e.g. [Brakstad and Ramstad, 2001](#); [Breuer et al., 2004](#); [Hartley et al., 2003](#)), and physical disturbance from platform activities, storms, and trawling are thought to be the major causes for dispersion of the material. Such erosion may repeatedly uncover deeper layers of the piles and thus enhance leakage of contaminants. Hence, there is a concern that older cuttings piles may be a source of episodic and continuous contamination for many years to come.

The sediment monitoring around NS installations is focussed on past and present discharges of drill cuttings ([Bakke and Nilssen, 2004](#); [Carroll et al., 2000](#); [Gray et al., 1999](#); [Kingston, 1992](#); [Renaud et al., 2008](#)). In most cases spatial and temporal changes in the benthic fauna around OBM and SM piles follow a pattern typical for organic enrichment as described by [Pearson and Rosenberg \(1978\)](#). Several of the indicator species for eutrophicated sediments are also dominating close to the cuttings piles, e.g. the polychaetes *Capitella capitata* and *Chaetozone setosa* and the bivalve *Thyasira* sp. ([Ugland et al., 2008](#)). Since the discharges of OBM cuttings to the NCS were terminated following new legislation in 1993, the recovery of local sediment fauna has been substantial ([Bakke et al., 2011](#); [Bakke and Nilssen, 2004](#); [Carroll et al., 2000](#); [Renaud et al., 2008](#); [Schaanning and Bakke, 1997](#)). At present, recorded effects on benthic macrofauna are most often confined to within a 250 m radius and seldom detected beyond 500 m, even around the largest piles ([Jarandsen and Fadnes, 2011](#); [Renaud et al., 2008](#)).

[Hartley et al. \(2003\)](#) made a comprehensive assessment of the potential for bioaccumulation, biomagnification, and food chain transfer of organic and inorganic cuttings pile contaminants on the basis of data from the NS and Gulf of Mexico. They concluded that old cuttings piles most likely had no significant food chain effect and did not pose a risk to human health. However, they also emphasized that very little direct information existed on physical and chemical pile structure and on contaminant accumulation in pile surface organisms. Since then very little new information has emerged.

[Olsgard and Gray \(1995\)](#) argued that as hydrocarbons become less of a problem around old cuttings piles, the metals will become the main source of environmental impact. This is yet to be demonstrated. [Grant and Briggs \(2002\)](#) found that metal levels were too low to explain toxicity beyond sites immediately adjacent to a large cuttings pile at the UK “NW Hutton” field. From tests with the amphipode *Corophium* ERT (1999) concluded that metals did not contribute to the toxicity of cuttings from around the “Beryl A” platform. [Leung et al. \(2005\)](#) and [Bjørgesæter \(2009\)](#) determined sediment quality guidelines (SQG) for several metals from field based sensitivity distribution (f-SSD) of more than 600 macrofauna taxa recorded between 1990 and 2001 around petroleum fields on the NCS. A preliminary screening of later monitoring data from 147 stations around other NCS cuttings piles ([Bakke unpublished](#)) showed that, out of 62 stations with metal levels above the SQGs of [Leung et al. \(2005\)](#) and [Bjørgesæter \(2009\)](#) and low levels of hydrocarbons, macrofauna disturbance was only found at 18 stations. These studies support the conclusion that metals dispersed from old piles have little impact on the surrounding benthos.

3.2.2. Impact from discharge of WBM cuttings

In practice, current UK and Norwegian legislation allows only WBM cuttings and used WBM to be discharged offshore. Typically, modern WBMs contain fresh or salt water as the base fluid and barite (BaSO₄) or ilmenite (FeTiO₃) as weighting agent. Clays or organic polymers are incorporated to create a homogenous fluid. Other chemicals (e.g. potassium formate and various glycols) are added to achieve viscosity control, shale stability, cooling and lubrication (c.f. [Hudgins, 1994](#); [Neff, 2005](#)). There is a vast literature

on the acute toxicity of WBM components, the presentation of which goes beyond the scope of this review, but in general the acute toxicity of WBM is low ([Neff, 1987](#)). Monitoring in the NS ([Daan and Mulder, 1993](#); [Olsgard and Gray, 1995](#); [Park et al., 2001](#); [Renaud et al., 2008](#)) has not revealed any *in situ* effects of WBM cuttings on sediment macrofauna community structure, implying that any such effects, if present, will be confined to bottoms inside the innermost stations in these studies, i.e. nearer than 25–250 m from the discharge point. The effects mechanisms of WBM cuttings after sedimentation have been studied in several laboratory and mesocosm experiments. [Dow et al. \(1990\)](#) reported that redox values were depressed for 3 months in sediments mixed with WBM cuttings in an onshore tank system. [Schaanning et al. \(2008\)](#) exposed undisturbed fjord sediment core samples to thinly sedimented layers of ilmenite based WBM cuttings. Iron sulphide precipitated under caps thicker than 10 mm. Sediment oxygen (SOC) and nitrate consumption, and release of silicate increased immediately under a 12–46 mm cap. The SOC peaked after 9 days and, for most treatments, returned to background levels after 3 weeks. The increase was positively correlated with cap thickness. A 3 mm cap on top of undisturbed sediment box cores from 200 m depth gave no increase in SOC, and macrofauna biomass and community structure did not change during a 3 month experiment. In a repeat experiment a 3 mm layer of WBM cuttings caused elevated SOC for more than 3 months and 6–24 mm layers for more than 6 months ([Trannum et al., 2010](#)). After 6 months the macrofauna species richness, abundance, biomass, and diversity were negatively correlated with layer thickness. Corresponding layers with natural sediment did not affect the fauna. [Trannum \(2011\)](#) concluded that the most plausible reason for the fauna effects was sediment oxygen deficiency due to degradation of organic WBM compounds, presumably mud glycol, although chemical toxicity may have played a role as well. It is not likely that glycol degradation will cause the same effects around a cuttings discharge since the glycol most probably will dissipate before the cuttings reach the bottom.

[Trannum et al. \(2011\)](#) found only slight differences in macrofauna recolonization in defaunated trays with coarse and fine sediments capped with 6 and 24 mm ilmenite based WBM cuttings deployed *in situ* at 200 m depth in the Oslofjord, Norway. After six months the species richness was the same in all treatments, but species composition in the 24 mm treatment differed from the others. Sediment grain size had stronger effect on recolonization than exposure to the cuttings. In a similar recolonization experiment at 10 m depth [Bakke et al. \(1989\)](#) found normal fauna diversity in azoic sediment capped for less than 2 years with 10 mm of WBM cuttings.

The experiments described above cover one single capping event, and there is little experimental evidence from repeated sedimentation which is typical around multi-well rigs. [Barlow and Kingston \(2001\)](#) exposed two filter feeding bivalves (*Cerastoderma edule* and *Macoma balthica*) to daily sedimentation for 12 days by drill mud barite equivalent to 1–3 mm coverage at each application. They found exposure dependent damage of gill ctenidia in both species in the 1 mm application, and severe mortality within 12 days following the 2 and 3 mm applications.

The smallest cuttings cap eliciting effects in the experiments by [Schaanning et al. \(2008\)](#), [Trannum et al. \(2010, 2011\)](#), and [Bakke et al. \(1989\)](#) was 3 mm, which is typical for conditions less than 250 m from a drilling rig ([Trannum, 2011](#)). The conditions simulated by [Barlow and Kingston \(2001\)](#) were typical for exposure 100–500 m from a drilling discharge. Other studies of the effects of WBM cuttings on sediment fauna also suggest that the impact is normally restricted to within 100–250 m and recovery seems rapid ([Bakke et al., 1986b](#); [Candler et al., 1995](#); [Carr et al., 1996](#); [Currie and](#)

Isaacs, 2005; Daan and Mulder, 1996; Daan et al., 1994; Montagna and Harper, 1996; Neff, 1987; Netto et al., 2010; Olgard and Gray, 1995; Trannum, 2011; Trannum et al., 2011). Hence there is strong evidence to conclude that sedimentation of WBM cuttings onto the seafloor has only local and short term effects on the sediment fauna.

WBM cuttings in suspension could affect other parts of the marine ecosystem such as pelagic organisms, sponges, corals and other sessile, hard bottom fauna entrained in a discharge plume. Such exposure will in most cases be short term, episodic or pulse-wise depending on plume behaviour. Hyland et al. (1994) found local reduction in hard bottom fauna abundance due to suspended particle loading around a WBM discharge site outside California. Cranford et al. (1999) showed that exposure for 6–70 days to concentrations between 0.5 and 10 mg L⁻¹ of used WBM in suspension had a negative effect on somatic and/or reproductive tissue growth in scallops. The same was seen following exposure to barite and OBM suspensions at less than 5 mg L⁻¹. The effects were linked to physical stress from the mud particles rather than chemical toxicity. Bechmann et al. (2006) found that suspensions of used barite-based WBM caused histopathological gill changes, reduced lysosome membrane stability, oxidative stress, DNA damage, reduced filtration rates, growth, and survival and modified haemolymph protein pattern in blue mussel and scallops (*Pecten maximus*). These effects were dose dependent. The lowest exposure causing effects was 0.5 mg L⁻¹ of WBM for 3 weeks. The same exposure caused histopathological changes in gills and changes in blood plasma in juvenile Atlantic cod. Interestingly, 1–10 mg L⁻¹ suspensions of WBM had a positive effect on feeding efficiency, growth and survival in cod larvae after 14 days exposure. The positive effects were assumed to be from particles of a particular size stimulating feeding activity. Feeding efficiency and growth in blue mussel larvae were reduced after exposure to 4 mg L⁻¹ suspensions of used barite-based WBM, whereas similar exposure to barite alone stimulated growth.

Berland et al. (2006) made a field validation of the results from Bechmann et al. (2006) by exposing caged scallops and blue mussels to an offshore discharge of WBM cuttings for 5 weeks. Scallops caged 250 m from the platform at a depth of 35 m showed increased GST enzyme activity and reduced gonad weight. DNA damage was seen in the mussels from the same cage. Filtration rate was reduced in both species, but shell growth was not affected. The other endpoints measured by Bechmann et al. (2006) were not affected (LMS, tolerance in mussel to air exposure, proteomics, and barium body burden). Exposure levels around the cages were not measured, but the average concentration of suspended cuttings where effects were found was estimated to be 0.15 mg L⁻¹. This corresponds well with the lowest concentration of suspended cuttings eliciting effects in the laboratory studies mentioned above (0.5 mg L⁻¹). From their experiments Bechmann et al. (2006) proposed 0.8 mg L⁻¹ as a chronic PNEC for suspended cuttings. Smit et al. (2008) estimated PNEC values of 7.6 mg L⁻¹ and 17.9 mg L⁻¹ respectively for suspended bentonite and barite clays on basis of SSDs from tests with 12–15 marine species. Although these PNEC estimates were made in somewhat different ways and hence are not directly comparable, the far lower PNEC for whole WBM cuttings proposed by Bechmann et al. (2006) could indicate that there may be other effects factors in play than just physical stress from the clay particles. The proposed PNEC is also within the typical range of natural SPM (suspended particulate matter) levels in the open NS (0.2–1 mg L⁻¹, Eisma and Kalf, 1987) which also indicates that WBM in suspension may elicit stronger effects than physical stress from suspended particles.

Studies on effects of suspended cuttings on sessile filter feeders such as sponges and cold water corals have not been published and

there are only a few published studies on the effects of cuttings particles settling onto these organisms. Larsson and Purser (2011) found that the cold water coral *Lophelia pertusa* was able to survive repeated, slight smothering by natural sediment and drill cuttings, but polyp death occurred when wholly covered by the particles. The response to cuttings and natural sediment did not differ. It was concluded that the current effects level from non-toxic burial of 6.3 mm proposed by Smit et al. (2008), may result in damage to *L. pertusa* colonies. Still, evidence of extensive growth of *L. pertusa* on offshore platform legs even after many years of discharge of OBM cuttings (Bell and Smith, 1999) suggest that the corals must be rather tolerant to drilling waste. Video monitoring carried out during WBM cuttings discharge episodes at the Norwegian Morvin field in 2009 and 2010 revealed no significant behavioural differences between exposed and unexposed *L. pertusa* (Buhl-Mortensen et al., 2010). Polyp retraction responded more systematically to changes in current velocity and direction than to cuttings plumes.

In conclusion, it is evident that discharged WBM cuttings may cause biological effects both during suspension in the water masses and after sedimentation. The studies indicate that the effect mechanism is mainly physical stress, but chemical toxicity cannot be ruled out. The levels of suspended WBM and WBM cuttings causing effects have been above 0.5 mg L⁻¹. Such levels are typically restricted to a radius of less than 1–2 km in the water masses (Neff, 1987). WBM cuttings deposits found to affect the benthos have a thickness of at least 3 mm or more. Such layer thicknesses will normally be confined to a distance of 100–500 m (Carr et al., 1996; Currie and Isaacs, 2005; Daan and Mulder, 1996; Ellis et al., 1996; Montagna and Harper, 1996; Neff, 1987; Trannum, 2011). Still, the WBM cuttings discharges are large and frequent, and the material widely dispersed and one cannot rule out that they in the long run may cause subtle changes to the benthic community structure on a wider geographical scale than this. One must assume that it will be extremely difficult to distinguish such effects from the temporal shift in the benthic community one sees on the NCS (Brattegard, 2011).

4. Population and ecosystem effects

4.1. Produced water

It has not yet been feasible to document effects of PW discharges on the population and community levels. Most of the laboratory and field studies described above support the conclusion that significant biological effects on pelagic organisms will be limited to a distance of less than one km due to rapid effluent dilution and very short exposure time. Knowledge on individual sensitivity is a prerequisite, but not sufficient, for assessing effects on populations and communities. Phyto- and zooplankton populations and most fish species have a much wider distribution than the documented PW impact zones. Hence, for a significant impact to occur either harmful exposure to PW has to be sufficiently wide scale or the population influence from locally affected individuals has to be large enough. None of these are likely. It is also inherently difficult to make reliable extrapolation to the population level since effects on individuals may be masked by other factors acting on populations e.g. distribution patterns, seasonality, species interaction, density dependent functions, other stressors, and the complex and dynamic physical conditions in the offshore pelagic ecosystem (Hjermann et al., 2007).

At present, bridging the organism-population gap seems only feasible through use of population models as demonstrated for Arctic cod, capelin (*Mallotus villosus*), and herring (*Clupea harengus*) by Hjermann et al. (2007) and for northern shrimp (*Pandalus*

borealis) by Ravagnan et al. (2010), or by employing a risk assessment approach. Beyer et al. (2012) performed a risk assessment for effects of C4–C7 APs in PW on three economically important fish populations on the NCS: Atlantic cod, haddock, and saithe (*Pollacius virens*), based on fish distribution data, hazard information of APs in PW, data on PW discharges, and plume dispersion described by the exposure and risk model DREAM (Reed and Hetland, 2002; Reed et al., 2001). Their conclusion was that the environmental exposure to C4–C7 APs from PW is too low to have any significant effect on the reproduction of fish stocks. Neff et al. (2006) and Durell et al. (2006) came to the same conclusion regarding the risk from PAHs in PW to the wider pelagic ecosystem in the NS when combining dispersion modeling by DREAM and PAH measurements in passive samplers (SPMDs) and caged mussels.

Smit et al. (2009) described a systematic relationship between sub-individual and individual sensitivity to oil from SSDs for DNA damage and oxidative stress biomarkers in 6 marine species and similar SSDs for whole-organism chronic fitness in 26 marine species. On average the selected biomarkers were a factor 35–50 more sensitive than the whole-organism response. The results implied that the 95% safety level (the lower 5 percentile or HCs, commonly used as PNEC in risk assessments), for whole-organism exposure to total hydrocarbons would safeguarded only 86% of the species from genotoxic damage and 79% from oxidative stress. The authors stress that their data were insufficient to support this as a general relationship, but data from Bechmann and Taban (2004), Bechmann et al. (2004), Buffagni et al. (2010), Carls et al. (1999), (Hansen et al., 2011), Heintz et al. (2000), Jonsson and Björkblom (2011), Pinturier et al. (2008), Sanni et al. (2005), and Stien et al. (1998) provide supporting evidence from a wider range of sub-tropical to high-arctic species of fish and invertebrates that the whole organism responses are less sensitive to oil than biomarker responses. Smit et al. (2009) present a conceptual model suggesting further reduction in sensitivity as one moves up to the population level. This would concur with the idea that environmental factors governing the health and performance of a population, may override toxic effects on parts of the population. The studies above cover sensitivity to oil, but the authors suggest that the relationship may be valid for PW as well. If that is the case, it is even more unlikely that wide scale population effects should occur when individual effects are only seen locally.

4.2. Drilling waste

It has in general been easier to study population and ecosystem effects from drilling waste than from PW. The main reason is that drilling waste primarily affects the sediment ecosystem for which analysis of community responses to natural and man-made perturbations have a very long tradition in marine environmental monitoring. A large number of harmonized techniques have been developed for such studies (Elliott, 1996; Gray, 2000; Gray et al., 1988). The sessile nature of benthic communities also facilitates repeated studies of the same sites to assess temporal changes and recovery over time. Extensive environmental monitoring both on the NCS and in the Dutch and UK regions of the NS, coupled with the mesocosm and field experiments described earlier, have given a comprehensive and mostly coherent picture of the spatial effects of muds and cuttings on sediment macrofauna community structure and on the rate of community recovery from past OBM and SBM cuttings discharges. Community restitution at previously impacted sites has been complete within 4–10 years (Bakke et al., 2011; Schaanning and Bakke, 1997). Around older multi-well discharge sites on the NCS the areal extent of the fauna effects has in general been reduced from up to 15 km² to less than 1 km² (Bakke et al., 2011). Studies from unimpacted reference sites on the NCS

(Renaud et al., 2008) do not indicate that past and present cuttings discharges are causing accumulative or long-lasting effects on the macrofauna structure on a wider scale. A concern still is that one knows little of possible effects on other elements of the benthic ecosystem. Some studies suggest that meiofauna does not respond fundamentally different from macrofauna to cuttings discharges (Montagna and Harper, 1996; Moore et al., 1987; Netto et al., 2010), but there is very little knowledge on the sensitivity of microfauna, epifauna, hyperfauna and coral and sponge communities to drilling waste. Feral haddock and cod caught in the NS Tampen region have shown biomarker effects (Balk et al., 2011; Grøsvik et al., 2010) which may reflect exposure to cuttings when the fish are foraging on the piles, but this may also stem from exposure to PW. Furthermore, beyond what can be inferred from the functional roles of macrofauna species, there is virtually no information of potential long term effects on population and community functions such as production, reproduction, and trophic interaction.

5. Concluding remarks

Operational discharges from the offshore industry have created public concern because they represent a very large continuous input of contaminants to the sea from many widely dispersed point sources. Furthermore, it is notoriously difficult to study effects of the discharges on populations (e.g. of commercial fish stocks) and the structure and function of marine ecosystems. This review shows a wealth of studies on the effects of produced water on individuals of important species, and on the effects of drilling waste on benthic communities. All evidence suggests that the effects of present discharges are local, and in general confined to within 1–2 km from an outlet both in the waters and on the seabed, and that the risk of widespread impact from the operational discharges is low. However, it is also important to stress that apart from studies on the effects of drilling waste on sediment macrofauna community structure there is nearly no published information on the effects on populations or communities. Hence, one cannot ignore the possibility of subtle, cumulative effects from the operational discharges which we are not able to measure at present, although risk assessments suggest that this will not be the case. It is a discouraging fact that at the moment there seems to be no options other than risk related modelling for assessing potentially significant effects of produced water discharges at the population and ecosystem levels. Published literature has not yet been able to validate with confidence or empirically verify that the effects of the discharges are only local. We believe that research addressing this challenge should emphasize 1) development of effects methods and endpoints that may be used in health screening of organisms on a scale large enough to reflect population health conditions with confidence, and 2) effects studies that encompass natural species interaction on an ecosystem level.

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