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Environmental effects of offshore produced water discharges: A review focused on the Norwegian continental shelf

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ABSTRACT

Produced water (PW), a large byproduct of offshore oil and gas extraction, is reinjected to formations or discharged to the sea after treatment. The discharges contain dispersed crude oil, polycyclic aromatic hydrocarbons (PAHs), alkylphenols (APs), metals, and many other constituents of environmental relevance. Risk-based regulation, greener offshore chemicals and improved cleaning systems have reduced environmental risks of PW discharges, but PW is still the largest operational source of oil pollution to the sea from the offshore petroleum industry. Monitoring surveys find detectable exposures in caged mussel and fish several km downstream from PW outfalls, but biomarkers indicate only mild acute effects in these sentinels. On the other hand, increased concentrations of DNA adducts are found repeatedly in benthic fish populations, especially in haddock. It is uncertain whether increased adducts could be a long-term effect of sediment contamination due to ongoing PW discharges, or earlier discharges of oil-containing drilling waste. Another concern is uncertainty regarding the possible effect of PW discharges in the sub-Arctic Southern Barents Sea. So far, research suggests that sub-arctic species are largely comparable to temperate species in their sensitivity to PW exposure. Larval deformities and cardiac toxicity in fish early life stages are among the biomarkers and adverse outcome pathways that currently receive much attention in PW effect research. Herein, we summarize the accumulated ecotoxicological knowledge of offshore PW discharges and highlight some key remaining knowledge needs.

1. Introduction

Operational discharges of produced water (PW) from offshore oil and gas platforms are a continuous source of contaminants to continental shelf ecosystems (Fig. 1) (Lee and Neff, 2011). The PW is treated to lower the content of unwanted components (Jimenez et al., 2018) and then reinjected to a geological formation or discharged to the sea. Reinjection is considered the Best Environmental Practice for PW management, but these operations are not always technically feasible, making discharge to the sea a very common management solution. In the oceanic area covered by the Oslo-Paris (OSPAR) conventions, about 300 million standard cubic meters (Sm^3) were discharged to sea annually in recent years; of these the Norwegian discharges were about 130 million Sm^3 (OSPAR, 2019). In 2017, the amount of dispersed oil in these discharges was reported to be about 1600 tons for Norwegian installations, and about 4000 tons for the whole OSPAR area (ibid.). Currently, treated PW is the largest operational waste stream discharged to sea from the offshore oil and gas industry worldwide.

Although large amounts of PW are discharged to the water column offshore, many consider the PW issue to be a classic case of "the solution to pollution is dilution" as a more rapid dilution occurs offshore compared to in freshwater systems affected by land-based oil and gas fields. From early on, the strong dilution of PW when discharged to the sea was considered adequate to mitigate risks for harmful ecological effects at offshore production fields, e.g., (Koons et al., 1977; Middleditch, 1984; Somerville et al., 1987; Girling, 1989; Stephenson, 1992). Although it is challenging to characterize all possible biological impacts associated with PW discharges, several earlier summaries of research and monitoring programs have concluded there is little to no evidence that significant impacts occur outside the primary dilution zone several kilometers downstream of PW outfalls (Nilsen et al., 2006; Bakke et al., 2013). But despite these general assessments, some remaining unknowns still create concern. One key concern is the prospect of increasing offshore oil and gas activities in northern regions, such

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as the Barents Sea. As the ice cap of the Arctic diminishes due to global warming, there will be better conditions for year-round offshore oil and gas operations in the Southern Barents Sea, and the polar sailing route north of Russia will most likely be open for large parts of the year. Both researchers and regulators are concerned over how key species, populations and ecosystems in the Barents Sea will respond to the cumulative stress from a warming climate and increased pollution inputs from industrial discharges such as PW, e.g., (Knol, 2011; Blanchard et al., 2014; Zheng et al., 2016).

The objective of the present review is to provide an updated summary of our knowledge on the ecotoxicology of offshore PW discharges. We primarily address research and monitoring reports from Norwegian research groups, and put most emphasis on peer-reviewed publications. Furthermore, we discuss the notion that marine species and systems of the Southern Barents Sea might be particularly sensitive and vulnerable to adverse effects of future PW discharges. Lastly, we look briefly at the adequacy of the present systems of environmental regulation and management of PW on the NCS and highlight some remaining environmental research needs associated with PW discharges.

2. Environmental contamination by offshore PW discharges

PW consist of formation water and previously injected water. A treated PW discharge typically contains dispersed oil (normally 10–100 mg/L range), dissolved hydrocarbon (HC) gases, suspended particles (e.g. clay), inorganic salts, organic acids, aromatic hydrocarbons, ketones, phenol/alkylphenols, heavy metals, and naturally occurring radioactive materials (NORM) (Fig. 2), and often also chemicals added to the production system to aid the extraction processes or to protect against biofouling and corrosion (Røe Utvik, 1999; Ahmadun et al., 2009; Neff et al., 2011). Many PW compounds remain unidentified and in chromatographic analyses they elute in what is called the “hump” or Unresolved Complex Mixture (UCM) (Melbye et al., 2009; Petersen et al., 2017b).

Treated PWs that meet regulatory limits can be discharged to the sea. The PW discharge will disperse and dilute in the receiving water body according to a number of factors, including: the composition of the discharged mixture; the flow rate, depth, direction and speed of the discharge jet; the sea current and tidal and wind conditions at the site; the differences in temperature, salinity, density, and buoyancy between the PW plume and the surrounding seawater; the stratification of the water column; turbulent mixing conditions; and advection-diffusion processes (Rye et al., 1995; Reed and Hetland, 2002; Niu et al., 2016; Premathilake and Khangaonkar, 2019). Computer-modelling tools such as DREAM (Dose-related Risk and Effect Assessment Model) (Reed and Rye, 2011) are used to predict the PW plume behaviour and the ecotoxicological risk of PW constituents (Neff et al., 2006; Durell et al., 2006). Modelling studies show that offshore PW plumes dilute rapidly and may typically reach a 1000-fold dilution at a distance of 1000 m from the PW outfall (Neff, 2002), although the rate of dilution may vary depending on the site-specific factors mentioned above. Subsequently, the far-field dilution process will proceed until the PW plume mixture becomes so diluted that it can no longer be analytically differentiated from background concentrations of the constituents. In addition, the PW mixture will be transformed and reduced by the weathering of its constituents, particularly by volatilization, microbial biodegradation and

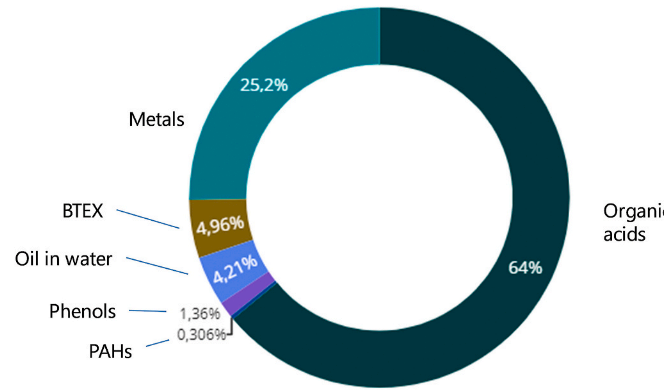

Fig. 2. Relative emissions of naturally occurring substances in PW discharges from Norwegian offshore production platforms in 2019. Data source: Norwegian Oil and Gas and the Norwegian Environment Agency.

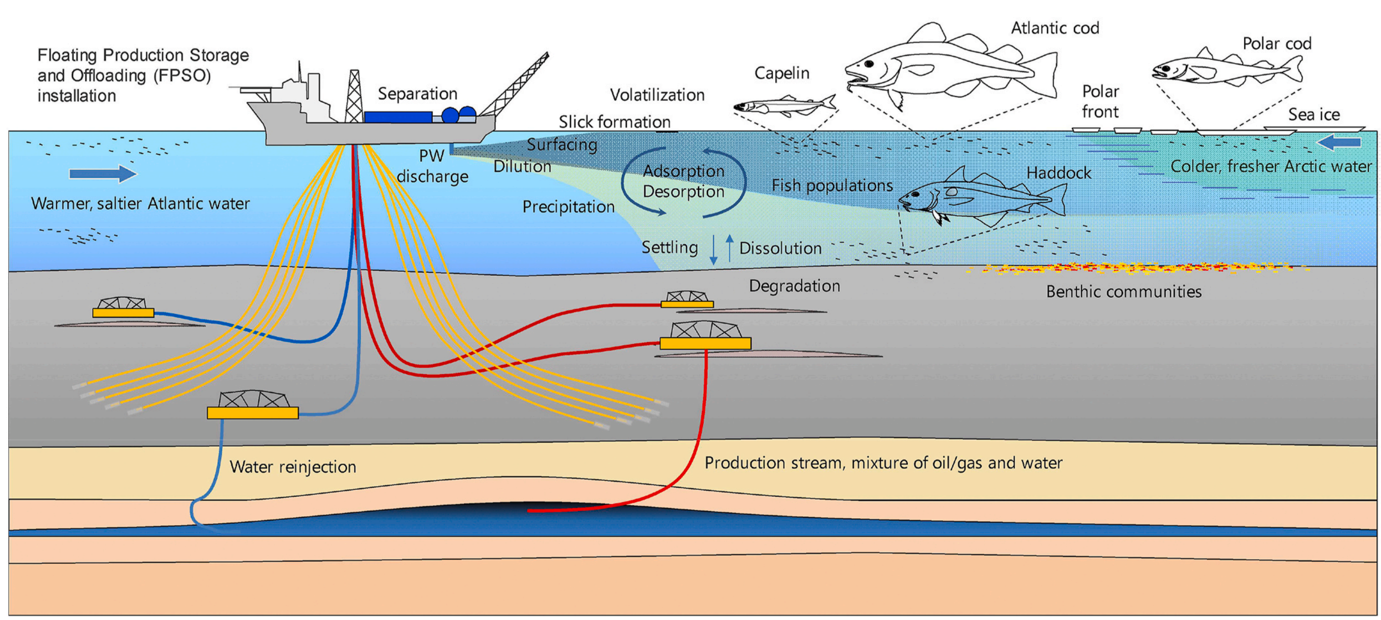

Fig. 1. Produced water (PW) discharged from offshore oil and gas production will spread with oceanic currents, forming a continuously diluting plume that exposes downstream ecosystems (illustrated by a simplified Barents Sea system) to the components of PW. Much research and monitoring have been performed to study possible environmental impacts from PW. Some mild acute effects are found in water column biota some km away from large PW outfalls, but whether there are notable chronic (and ecologically adverse) effects occurring in a larger area downstream, and whether species, populations and ecosystems in the sub-arctic part of the Barents Sea are extra sensitive or vulnerable to PW, remain unknown. These are issues that attract continued research interest.

photodegradation (McFarlin et al., 2018).

Dispersed oil is a key risk component in the PW mixture. Rye et al. (1998a) used PW discharge data from 95 production platforms in the North Sea and the numerical model PROVANN to study the processes of dilution and spreading of dispersed oil from PW discharges on the NCS. They estimated that the highest average concentrations of oil hydrocarbons in the upper 50 m of the water column occur in the platform-dense Tampen area in the northern part of the North Sea. Concentrations as high as 3 μg/L would occur in areas as large as 50–100 km in diameter downstream from major PW discharging platforms, consistent with the estimates of oil concentrations in water reported by Stagg and McIntosh (1996). The study concluded that typical concentrations of oil in water measured in the Tampen area would correspond to an approximate PW dilution rate of 1:10,000 (ibid.).

The concentrations of polycyclic aromatic hydrocarbons (PAH) is a key risk component for PW. According to the monitoring guideline for the NCS (Miljødirektoratet, 2020), the sum of all PAH concentrations (total PAH or TPAH) shall be quantified by the EPA 16 priority PAHs, even though that set of PAHs is not very representative for petrogenic PAH mixtures, e.g. (Richter-Brockmann and Achten, 2018). The EPA 16 PAHs include: naphthalene, acenaphthene, acenaphthylene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benz [a]anthracene, chrysene, benzo [b]fluoranthene, benzo [k]fluoranthene, benzo [a] pyrene, benzo [ghi]perylene, indeno [1,2,3,cd]pyrene, and dibenz [a,h] anthracene. Surface seawaters in offshore production fields of the NCS typically contain TPAH concentrations of 25–350 ng/L within 1 km downstream of "normal size" PW discharges; a background concentration of 4–8 ng/L TPAH is typically reached within 5–10 km of the discharge (Durell et al., 2006). This PAH background represents an approximately 100,000-fold dilution of TPAH concentrations measured in a "typical" offshore PW effluent (ibid.). Stagg and McIntosh (1996) used a towed *in situ* UV fluorimeter to measure the total oil hydrocarbon concentrations in the water column at multiple field sites in the North Sea, both close to and far from oil and gas production platforms. As the fluorimeter was calibrated with the water-soluble fraction of a crude oil, oil hydrocarbons other than the 16 priority PAHs were also measured. They found background concentrations of approximately 0.1 μg/L in areas far from platforms, 1–2 μg/L in areas relatively close to platforms and up to 10 μg/L in the German Bight (ibid.). The concentrations of PAHs in fish fillets and other samples from different North Sea fish populations have been studied for years in the Norwegian Condition Monitoring program. Generally, these surveys find low PAH concentrations (below the limits of quantification, LOQ) in muscle samples for gadoid fish species sampled from all regions. However, fluorometric analyses of bile samples indicated a measurable PAH exposure in haddock (*Melanogrammus aeglefinus*) collected in the platform-dense Tampen area (Grøsvik et al., 2009). Fluorometric PAH metabolites detected in fish bile are often a more sensitive exposure marker of recent PAH exposure compared to PAH parent compound measurements in fish tissues; for a review see Beyer et al. (2010).

The uptake routes for the bioaccumulation of different PW contaminants by marine biota will differ depending on the properties of the discharge mixture, the given contaminant and the organism (and its life-phase). Passive uptake over gill surfaces is a key route/mechanism for uptake of hydrophobic organic contaminants. Other important uptake routes include passive and active uptake over digestive epithelia, and active uptake of contaminated food particles. Realistic PW exposure models can therefore be made quite simply as flow-through dilutions in seawater of major PW constituents or real PW mixtures (Sundt et al., 2009b, 2011, 2012b). However, some of the PW exposure studies that have used PW dilutions have used very high test concentrations, sometimes >1% PW, e.g., (Daniels and Means, 1989), and even as high as 50% (1:1 PW and seawater), e.g., (Caliani et al., 2009). These high-level exposures would occur only in the most concentrated part of a PW plume, typically within 100 m of the PW outfall. Controlled behaviour studies have shown that fish will actively avoid environments contaminated with PAH or oil above certain thresholds, e.g., (Martin, 2017; Claireaux et al., 2018). This means that "reef populations" of fish, i.e. groups of fish congregating around offshore platforms (Jørgensen et al., 2002b; Løkkeborg et al., 2002b), could be less exposed to PW than their close proximity to the PW outfall would suggest, simply because they avoid the PW plume. PW contaminant uptake, and long-term effects, have repeatedly been studied in fish that have received food artificially contaminated with PW-relevant chemical mixtures, e.g., (Meier et al., 2007b, 2011), although the ecological realism of the dietary uptake system used remains uncertain. However, copepods of the genus *Calanus* are key food organisms for fish and can potentially accumulate lipophilic oil components because they have a high lipid content and can filter and ingest oil droplets (Hansen et al., 2017b, 2018b, 2020b). Exposures in fish that occur via contaminated food or due to certain feeding behaviours will be discussed more in connection with measurements of DNA adducts further below. Such contaminant uptake routes could have special relevance for the assessments of possible long-term impact of offshore PW discharges, especially in benthic fish species.

Biomarkers that are used to detect exposures or effects of PW contaminants in marine biota in field surveys must be very sensitive. In fish, the liver-bile-faeces pathway is the major route for elimination of toxic organic aromatic substances such as PAHs and alkylphenols. Bile metabolites of petrogenic PAHs and alkylphenols can act as sensitive markers of PW exposure (Jonsson et al., 2003, 2004, 2008a, 2008b, 2012; Sundt et al., 2009b, 2012b; Beyer et al., 2010, 2011). Recent studies of PW oil droplet toxicity to embryos of Atlantic cod (*Gadus morhua*) and haddock suggest that early life stages of haddock may be more sensitive than cod to dispersed crude oil due to the very sticky chorion (egg surface) of haddock eggs (Sørhus et al., 2015, 2016b, 2017; Sørensen et al., 2017, 2019a; Hansen et al., 2018c, 2019c). Because of their sticky chorion, haddock eggs accumulate far more oil droplets compared to eggs of Atlantic cod which have a non-sticky chorion, hence increasing the exposure to and bioconcentration of hydrophobic petrogenic contaminants. The increased amounts of oil droplets on the chorion reduces the exposure time necessary to cause toxicity, e.g. in post-spawn situations when spawned eggs may drift into a diluted PW plume containing dispersed crude oil. In this scenario, a brief exposure to elevated concentrations of dispersed oil may continue to affect the haddock embryos because the attached oil droplets represent a continued source of exposure even after the embryos are transferred to uncontaminated water (Sørhus et al., 2015).

Field surveys in the NCS offshore monitoring program have repeatedly used transplant caging of fish and mussels to test for accumulation and effects of PW contaminants. The caged sentinels are kept for up to 6 weeks at fixed positions in the likely path of the diluting PW plume, at least for a certain part of the tidal cycle. Such a caging approach provides a semi-controlled *in situ* exposure model. Offshore monitoring surveys on the NCS have repeatedly demonstrated that the caged biota develops exposure signals that are in quite good agreement with the distance they have to the upstream PW outfall. The best exposure markers have been two- and three-rings PAHs, with parent PAHs being detected in caged blue mussels and biliary PAH metabolites being detected in the caged fish. Interestingly, increased concentrations of bile metabolites of petrogenic PAHs and alkylphenols have been detected in fish caged as far as 10 km downstream from offshore production platforms in the assumed direction of the PW plume (Aas et al., 2002; Hylland et al., 2008). The use of offshore caging systems can also be further advanced by applying certain sensitive species or life stages as the caging sentinels. For example, researchers at SINTEF Ocean (Trondheim) have recently developed a new caging rig for use in marine effect monitoring that employs *in situ* fish embryo testing (Hansen et al., 2020a). However, it is uncertain to what extent caged fish are representative for the real-world conditions experienced by wild and free moving fish in offshore waters. For example, in the offshore Condition Monitoring surveys, wild-caught fish species collected in areas relatively

Table 1
Overview of published research or monitoring papers relevant to assessing ecotoxicological effects of offshore PW discharges.

PW study issue	Published research or monitoring studies
Chemical composition of offshore PW discharges and PW mixes in seawater	(Soto et al., 1991; Jacobs et al., 1992; Neff et al., 1992; Priatna et al., 1994; Nimrod and Benson, 1996; Terrens and Tait, 1996; Rye et al., 1998b; Sanni et al., 1998; Vik et al., 1998; Røe Utvik, 1999; Røe Utvik et al., 1999; Neff, 2002; Frost et al., 2002; Røe Utvik and Hasle, 2002; Brakstad et al., 2004; Faksness et al., 2004; Johnsen et al., 2004; Boitsov et al., 2004; Lee et al., 2005; Durell et al., 2006; Brakstad and Bonaunet, 2006; Boitsov et al., 2007; Meier et al., 2007b; Thomas et al., 2009; Ahmadun et al., 2009; Melbye et al., 2009; Balaam et al., 2009; AMAP, 2010b, a, c; Neff et al., 2011; OLF, 2011; Hosseini et al., 2012; Harman et al., 2014; Hale et al., 2016; Godøy et al., 2016; Samanipour et al., 2016; Lofthus et al., 2016; Nepstad et al., 2017; Samanipour et al., 2017a; Samanipour et al., 2017b; Silvani et al., 2017; Dudek et al., 2017; Alyzakis et al., 2018; Samanipour et al., 2018a; Samanipour et al., 2018b; Lofthus et al., 2018a; Lofthus et al., 2018b; McFarlin et al., 2018; Samanipour et al., 2019; Sørensen et al., 2019b; Samanipour et al., 2020; Sørensen et al., 2020)
Monitoring of PW using passive sampling devices and *in vitro* bioassays	(Harman et al., 2009b, 2010, 2011, 2014; Hale et al., 2016, 2019)
Determination of PW contaminant concentrations	(Krahn et al., 1986; Brendehaug et al., 1992; McDonald et al., 1995; Neff and Burns, 1996; Johnsen et al., 1998; Tollefsen et al., 1998; Rye et al., 1998a; Røe, 1998; Røe Utvik, 1999; Aas et al., 2000a; Baussant et al., 2001a; Baussant et al., 2001b; Pedersen and Hill, 2002; Booij et al., 2002; Huckins et al., 2002; Lucarelli et al., 2003; Bagni et al., 2005; Namiesnik et al., 2005; Meier et al., 2005; Aas et al., 2006; Bulukin et al., 2006; Boitsov et al., 2007; Jonsson et al., 2008a; Jonsson et al., 2008b; Harman et al., 2008; Brooks et al., 2009; Sundt et al., 2009a; Sundt et al., 2009b; Grung et al., 2009a; Harman et al., 2009b; Skadsheim et al., 2009; AMAP, 2010c, b, a; Meier et al., 2010; Beyer et al., 2010; Beyer et al., 2011; Sundt et al., 2012a; Jonsson and Björkblom, 2011; Sundt and Björkblom, 2011; Jonsson et al., 2012; Broch et al., 2013; Harman et al., 2014; Hale et al., 2016; Hale et al., 2019)
Bioaccumulation and/or effects of PW or oil HCs and PAHs in fish under temperate or boreal conditions	(Lowe and Pipe, 1987; Myers et al., 1991; Aas et al., 2000b; Stephens et al., 2000; Incardona et al., 2004; Taban et al., 2004; Sturve et al., 2006; Laffon et al., 2006; Thomas et al., 2007; Nahrgang et al., 2008, 2010b, 2010c, 2010d; Carls et al., 2008; Baussant et al., 2009, 2011; Holth et al., 2009; Sørhus et al., 2015, 2016b, 2017; Szczybelski et al., 2016, 2019a; Hansen et al., 2016b, 2018a, 2019a, 2019b, 2019c; Sanni et al., 2017a; Sørensen et al., 2017, 2019a, 2020; Krause et al., 2017; Toxværd et al., 2018; Nepstad et al., in press)
Bioaccumulation and/or effects of PW or oil HCs and PAHs in polar cod (*B. saida*) under sub-arctic or arctic temperature conditions	(Christiansen and George, 1995; George et al., 1995; Nahrgang et al., 2008, 2009, 2010a, 2010b, 2010c, 2010d, 2016, 2019; Christiansen et al., 2010; Jonsson et al., 2010; Gardiner et al., 2013; Dussauze et al., 2014; Geraudie et al., 2014; Andersen et al., 2015; Bakke et al., 2016; Bender et al., 2016, 2018; Vieweg et al., 2017, 2018; Song et al., 2019; Fahd et al., 2019; Laurel et al., 2019)
Exposure of marine biota to PW radioactivity	(Røe Utvik, 1999; Eriksen et al., 2006; Grung et al., 2009b; Olsvik et al., 2012a; Hosseini et al., 2012; Godøy et al., 2016)
Field studies relevant to offshore PW discharges	(Stagg and McIntosh, 1996; Johnsen et al., 1998; Rye et al., 1998a; Røe Utvik and Johnsen, 1999; Reed et al., 2001; Reed and Hetland, 2002; Aas et al., 2002, 2006; Jørgensen et al., 2002a; Løkkeborg et al., 2002a; Wells, 2005; Neff et al., 2006; Durell et al., 2006; Grøsvik et al., 2007, 2009, 2012; Hylland et al., 2008; Brooks et al., 2009, 2011b, 2012; Harman et al., 2009a, 2009b, 2011, 2014; Sundt et al., 2011, 2012b; Bakke et al., 2011; Balk et al., 2011; Smit et al., 2011; Hale et al., 2016)
Non-endocrine effects in fish of PW contaminants and mixtures	(Dey et al., 1983; Schultz et al., 1986; Widdows et al., 1987; Lowe and Pipe, 1987; Daniels and Means, 1989; Strømgren et al., 1995; Stephens et al., 2000; Obata and Kubota, 2000; Okai et al., 2000; Hasselberg et al., 2004a; Hasselberg et al., 2004b; Hurst et al., 2005; Hylland et al., 2006; Olsvik et al., 2007; Meier et al., 2007a; Hylland et al., 2008; Sundt et al., 2008; Tollefsen et al., 2008a; Abrahamson et al., 2008; Holth et al., 2008; Brooks et al., 2009; Holth et al., 2009; Lie et al., 2009; Hannam et al., 2009b; Olsvik et al., 2010; Jonsson et al., 2010; Grøsvik et al., 2010; Farmen et al., 2010; Holth et al., 2010; Perez-Casanova et al., 2010; Holth et al., 2011a; Sundt and Björkblom, 2011; Sundt et al., 2012a; Jonsson and Björkblom, 2011; Balk et al., 2011; Holth et al., 2011b; Olsvik et al., 2011a; Olsvik et al., 2011b; Olsvik et al., 2011c; Perez-Casanova et al., 2012; Grøsvik et al., 2012; Holth and Tollefsen, 2012; Sundt et al., 2012b; Tollefsen et al., 2012; Knag and Taugbøl, 2013; Knag et al., 2013a; Bratberg et al., 2013; Carlsson et al., 2014; Camus et al., 2015; Jensen et al., 2016; Froment et al., 2016; Sanni et al., 2017a; Holth et al., 2017; Petersen et al., 2017b; Petersen et al., 2017c; Petersen et al., 2017a; Hale et al., 2019; Meier et al., accepted)
Endocrine and reproduction related effects in fish of PW contaminants and mixtures	(Jobling and Sumpter, 1993; White et al., 1994b; Gimeno et al., 1998b; Gimeno et al., 1998a; Miles-Richardson et al., 1999; Weber et al., 2002; Tanaka and Grizzle, 2002; Weber et al., 2003; Thomas et al., 2004a, b; Mjøs et al., 2006; Tollefsen et al., 2006; Meier et al., 2007b; Tollefsen et al., 2007; Boitsov et al., 2007; Meier et al., 2008; Tollefsen et al., 2008b; Tollefsen and Nilsen, 2008; Brooks et al., 2009; Thomas et al., 2009; Lie et al., 2009; Meier et al., 2010; Holth et al., 2010; Tollefsen et al., 2011; Meier et al., 2011; Sundt and Björkblom, 2011; Petersen and Tollefsen, 2011; Beyer et al., 2012; Knag et al., 2013b; Knag and Taugbøl, 2013; Knag et al., 2013a; Petersen et al., 2013; Salaberria et al., 2014; Geraudie et al., 2014; Hultman et al., 2015; Sanni et al., 2017a; Petersen et al., 2017c; Petersen et al., 2017a; Hultman et al., 2017)
Benthic invertebrate bioindicators useable for pollution monitoring in the Barents Sea	(Andrade and Renaud, 2011; Jørgensen et al., 2011; Olsen et al., 2011; Wlodarska-Kowalczuk et al., 2012; Nahrgang et al., 2013; Kutti et al., 2013, 2015; Tjensvoll et al., 2013; Larsson et al., 2014; Edge et al., 2016; Zetsche et al., 2016; Dauvin et al., 2016; Szczybelski et al., 2016, 2019b; Baussant et al., 2017, 2018; Leys et al., 2018
Effects of oil and PW associated contaminant exposures in marine crustacean plankton	(Haukas et al., 2007; Camus and Olsen, 2008; Olsen et al., 2008, 2013a; Hatlen et al., 2009; Krapp et al., 2009; Hansen et al., 2011, 2012, 2013a, 2013b, 2014, 2015, 2016a, 2017a, 2017b, 2018b; Broch et al., 2013 Jager and Hansen, 2013; Miljeteig et al., 2013, 2014; Jager and Ravagnan, 2015, 2016; Jager et al., 2015, 2016, 2017; Nepstad et al., 2015; Nordtug et al., 2015; Camus et al., 2015; Farkas et al., 2017; Tollefsen et al., 2017; Krause et al., 2017; Toxværd et al., 2018; Sørensen et al., 2019b; Skottene et al., 2019)
Effects of oil and PW contaminants in krill and shrimps	(Bechmann et al., 2010; Arnberg et al., 2017, 2019; Moodley et al., 2018; Øverjordet et al., 2018)
Sensitivity drivers for oil contamination effect in marine fish	(Sørensen et al., 2014, 2015, 2016a, 2016b, 2017, 2019a; Duan et al., 2015; Vikebø et al., 2015 Sørhus et al., 2016b, 2017; Nepstad et al., 2017; Hansen et al., 2018c; Tørresen et al., 2018; Torvanger et al., 2018; Jawad et al., 2018)

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Table 1 (*continued*)

PW study issue	Published research or monitoring studies
Risk-Based Approach (RBA) tools for PW effect studies (CHARM, EIF, DREAM, and species sensitivity distributions (SSDs).	(Vik et al., 1998; Karman and Reerink, 1998; Scholten et al., 2000; Smit et al., 2005; Brooks et al., 2011a; Radovic et al., 2012; Beyer et al., 2012; Rye et al., 2014; Camus et al., 2015; Jager and Ravagnan, 2015, 2016; Arnberg et al., 2017; Langangen et al., 2017a; Sanni et al., 2017a, 2017b, 2017c; Lofthus et al., 2018a; Parkerton et al., 2018; Karman and Smit, 2019)
Use of "omics" approaches in PW-relevant effect studies	(Bjørnstad et al., 2006; Grøsvik et al., 2006; Olsvik et al., 2007, 2012b; Hansen et al., 2007, 2008a, 2008b, 2011, 2013b; Mæland et al., 2008; Kjersem et al., 2008; Bohne-Kjersem et al., 2009, 2010; Karlsen et al., 2011; Nilsen et al., 2011a, 2011b, 2011c; Sørhus et al., 2016a; Song et al., 2018; Tørresen et al., 2018)
Oil weathering and ecosystem sensitivity to oil pollution under arctic conditions	(Sydnes et al., 1985; Faksness and Brandvik, 2008a, b; Faksness et al., 2008; Brandvik and Faksness, 2009; Sikorski and Pavlova, 2018)
Assessment of mixture toxicity of compounds in PW discharges	(Song et al., 2012, 2014a, 2014b, 2016, 2018; Petersen and Tollefsen, 2011, 2012; Tollefsen et al., 2012; Petersen et al., 2013, 2014; Beyer et al., 2014)
Decision support tools for oil pollution management in the Barents Sea ecoregion	(Sørensen et al., 2014; Nepstad et al., 2015; Stordal et al., 2015a, 2015b; Alver et al., 2016; de Hoop et al., 2016; Carroll et al., 2018; Lofthus et al., 2018b; Christie et al., 2019)

close to oil platforms have generally shown few indications of exposure of contaminants from PW discharges (Grøsvik et al., 2009, 2010, 2012, 2015), although there are reasons to believe haddock may represent an exception (as will be further discussed below).

3. Effect biomarkers in PW exposed marine organisms

Studies of the toxic effects of offshore PW discharges have focused mainly on water column biota, especially fish, but also invertebrates such as calanoid copepods, the arctic ice amphipod (*Apherusa glacialis*), or the northern krill (*Meganyctiphanes norvegica*) (Table 1).

Cytochrome P450 (CYP) enzymes in fish have received much attention as biomarkers for PW and oil-related contaminants. Research on CYP biomarkers goes back to the early 1970s, pioneered by (among others) R.F. Addison, J.F. Payne and J.J. Stegeman. Various methods for determining CYP1A (a subfamily of CYP) were established, harmonised and quality assured during the 1980s and early 1990s (Goksøyr and Solberg, 1987; Goksøyr et al., 1988; Stagg, 1991; Goksøyr and Förlin, 1992; Stagg and Addison, 1995). The ethoxyresorufin-O-deethylase (EROD) assay is often used as a measurement of CYP1A activity. In several field surveys close to North Sea oil and gas production platforms, Stagg and McIntosh (1996) found elevated EROD activities in fish larvae of both sandeel (*Ammodytes marinus*) and gadoids, suggesting that aromatic components of PW discharges, such as PAH, were bioavailable and capable of inducing CYP1A activity. However, a laboratory-based dose-response follow-up study with 50d-old juvenile turbot exposed to 0.001–1% (v/v) dilutions of a PW sample did not confirm EROD responses similar to those seen in the field survey (Stephens et al., 2000). There have been hundreds of other studies describing virtually all sides of CYP responses in fish as sensitive biomarkers to aromatic contaminant exposures, e.g. see review by van der Oost et al. (2003), but the reliability of this biomarker system as a sensitive effect parameter for ecotoxicological monitoring of offshore PW discharge situations seems not to have been firmly established. One problem may be species differences in the sensitivity of the response. For example, recent PW effect studies such as Meier et al. (2010), found consistent CYP induction responses only when the fish specimens (Atlantic cod) were exposed to a PW concentration of 1% (v/v), which realistically would represent a distance of less than 100 m from a PW outfall offshore. Other controlled PW exposure studies have yielded more sensitive CYP1A induction responses, e.g., as observed by Geraudie et al. (2014) with the use of polar cod (*Boreogadus saida*) exposed at a 0.05% (v/v) concentration for 21 days or more, a concentration which can be expected more than 1000 m downstream from offshore PW outfalls.

Endocrine Disrupting Compounds (EDCs) in offshore PW discharges became a concern around the mid-1990s, especially after alkylphenols (AP), a common class of substances in PW, were identified as a likely cause for xenoestrogenic effects in fish in some UK rivers (Jobling and Sumpter, 1993; White et al., 1994a). The concept of EDC effects of chemical mixtures such as PW also gained increased attention after the discovery of cumulative and synergistic effects phenomena (Beyer et al., 2014; Tollefsen et al., 2014; Song et al., 2018). Fortunately, the APs that are dominant in PW mixtures are the "low molecular weight" C1 – C3 alkyl congeners, as shown by Boitsov et al. (2007), who characterized 52 different AP species in PW samples from nine different platforms on the NCS. Both the size and substitution position of alkyl side chains have important effects on the ability of APs to act as estrogen receptor (ER) agonists. Para-substituted, tertiary octyl- and nonylphenols (C8 and C9 APs) are the most estrogenic. These AP species are practically absent from typical offshore PW mixtures. According to Boitsov et al. (2007) the total concentration of tertiary, para-substituted APs in nine different PW mixtures from platforms on the NCS was in the range 0.2–1.7 μg/L, with 4-tert-butylphenol (4TBP) accounting for the highest proportion. Compared to the total amount of APs in these nine PWs, the sum of tertiary, para-substituted APs were always less than 1% (and most often far less than 0.1%) of the total AP concentrations. The most common AP with a clear ER agonist potency was 4TBP, although this potency is much weaker than the more potent C8 and C9 AP isomers. However, even though it is the "small AP structures" with little estrogenic potency that are typically found in PW effluents, the very big volumes of PW that are released make it important to assess the xenoestrogenic risk of these discharges. For example, based on *in vitro* models, the input of *in vitro* ER agonists into the North Sea from offshore platforms is estimated to equal almost 11 kg of 17β-estradiol equivalents per year (Thomas et al., 2004a). The AP components in the PW account for about 35% of the observed ER activity (Balaam et al., 2009), whereas the naphthenic acids, another very common compound class in PW (Samanipour et al., 2020), and which includes multiple weak ER agonists, account for about 65% of the non-AP ER agonist activity of PW mixtures (Thomas et al., 2009). The biomarkers most in focus for assessing xenoestrogenicity of offshore PW effluents have been the induction of the female egg-yolk precursor protein vitellogenin and reduced 11-keto-testosterone concentrations in the plasma of male fish and the impaired oocyte development, reduced estrogen levels, and delayed spawning time in females (Meier et al., 2007b). More than 30 research studies that address the effects of PW discharges on the endocrinology and reproduction of offshore fish and fish populations are listed in Table 1; few have investigated effects on invertebrates. Many of the EDC effect studies have been conducted by the Institute of Marine Research (IMR) in Bergen (Norway). The results from these studies were included in a risk assessment of the potential effects of APs from offshore PW on the reproductive success of the key fish stocks in the North Sea (Meier et al., 2008). The assessment concluded that there was no significant risk of adverse reproductive effects on the cod, saithe (*Pollachius virens*) or haddock populations in the North Sea as a result of exposure to APs from offshore PW discharges.

PAH-related DNA adducts are a form of DNA damage caused by covalent attachments of PAHs to DNA. Many studies have shown that fish will develop high concentrations of adducts in different tissues when exposed to PAH mixtures of pyrogenic or petrogenic origins (Aas et al., 2000b; Aas et al., 2001; Holth et al., 2009; Balk et al., 2011; Sundt et al., 2012b; Pampanin et al., 2017; Meier et al., accepted). Adducts that are not removed by the DNA repair systems of the affected cells can cause mutations that eventually may give rise to development of tumors and other cancer-associated conditions (Baird et al., 2005; Idowu et al., 2019). Adducts are therefore frequently used as biomarkers for hazardous PAH exposure. The potency for forming DNA adducts varies greatly among PAHs and the most common petrogenic PAHs in PW generally have a relative low potency. The most sensitive (but non-specific) method for quantifying PAH-DNA adducts is the ^{32}P-post-labelling assay followed by thin-layer-chromatography (TCL) and autoradiographic determination of PAH-adducted DNA nucleotides (Gupta et al., 1982; Jones and Parry, 1992; Pampanin et al., 2017). Hepatic PAH-DNA adduct formation has been a priority biomarker in the Norwegian offshore monitoring program. Groups of fish (Atlantic cod) have in several surveys been exposed to diluted PW plumes for 5–6 weeks by caging at fixed positions in the water column at increasing distances from different PW outfalls. These offshore caging exposures have generally *NOT* caused increased DNA adduct concentrations, including when caging was performed close to the Statfjord and Gullfaks platforms (i.e. in the Tampen area) in 2001, 2002 and 2004 and at the Troll field in 2003 (Hylland et al., 2008). Likewise, subsequent surveys in 2006, 2008 and 2009 at the Ekofisk field, centrally in the North Sea, showed no increase in DNA adduct concentrations in cod caged for 5–6 weeks, even not in the groups caged in the diluting plume only 200–250 m downstream of the PW discharge point (Brooks et al., 2011b). It is important to emphasize that several other markers in the caged fish, as well as in blue mussels caged at the same positions, confirmed their exposure to the diluted PW plume. These included increased concentrations of petrogenic PAH in the caged mussels and increased concentrations of biliary PAH metabolites in the caged fish.

In contrast to data from caged fish commented above, samples of wild haddock collected in several parts of the North Sea, and particularly in the platform-dense Tampen area, have repeatedly shown significantly increased concentrations of PAH-DNA adducts (in 2002, 2005 and 2008) (Grøsvik et al., 2009, 2010; Balk et al., 2011). Further confirmatory data of increased adducts in haddock from the Tampen area were obtained by a follow-up survey in 2011, and in this survey there were also found increased DNA adduct concentrations in haddock specimens collected in putative reference areas such as the Egersund Bank and the Viking Bank (Grøsvik et al., 2012). Furthermore, more recent field surveys in several other offshore sites in the North Sea, including in anticipated reference areas far from oil and gas platforms, have found increased concentrations of DNA adducts in haddock and other benthic fish species compared to fish from more pristine remote areas near Iceland and in the Barents Sea (Sundt et al., 2012a; Brooks et al., 2013, 2015; Pampanin et al., 2019). The hepatic DNA adduct signal in haddock from Tampen was particularly high in 2002, with a mean concentration of about 20 nmol adducts per mol normal nucleotides (Balk et al., 2011). Interestingly, during that survey it was noted that virtually all haddock individuals sampled had their stomach and intestine totally full with a sediment-like material, suggesting that they shortly before being captured had been actively reworking surface sediments to search for invertebrate prey, such as soft sediment infaunal brittle stars like *Amphiura filiformis*. No similar sediment-filled gut was found in Atlantic cod and saithe (*Pollachius virens*) although they were collected at the same trawling stations as the haddock, nor did the cod and saithe show any increased PAH-DNA-adduct concentrations (Balk et al., 2011). Thus, the haddock specimens that show increased adduct levels could possibly have experienced increased PAH exposure because of an active sediment-reworking behaviour in areas with PAH-contaminated surface sediments, although this hypothesis has not yet been proven. It is also unknown whether the surface sediments in the wider Tampen area are notably PAH contaminated, for example due to the large on-going discharges of PW from the many production platforms in the area, or because of the past extensive discharge of oil-based drilling waste (banned in 1993), or a combination of these sources.

Other PAH-associated biomarker effects that have received much focus lately are the cardiac toxicity and impaired development of fish embryos exposed to crude oil and petrogenic PAHs, even at very low concentrations. The key signs of toxicity include cardiovascular dysfunctions, pericardial and yolk sac edemas, subcutaneous haemorrhages, craniofacial deformities, reduced growth, as well as increased mortality rates of larvae and juvenile fish (Spitsbergen et al., 1991; Walker et al., 1991; Hose et al., 1996; Carls et al., 1999, 2008; Guiney et al., 2000; Incardona et al., 2004, 2014, 2015; Dussauze et al., 2014; Incardona and Scholz, 2016). Norwegian groups have recently become active in this area of effects research and have contributed both to improved mechanistic understanding of this effect phenomenon and effect knowledge in new fish species, such as Atlantic cod and haddock (Meier et al., 2007b, 2010, 2011; Sørhus et al., 2015, 2016b, 2017; Sørensen et al., 2017; Hansen et al., 2018c, 2019b, 2019c; Jawad et al., 2018; Lie et al., 2019). Most reports have linked this syndrome to the presence and toxic actions of alkylated tricyclic PAHs, most likely acting via disruptive modulation of intracellular transmembrane calcium and potassium transport, but research is still ongoing to unravel the exact effect mechanism underlying the syndrome and the contaminants that are capable of causing it (Meador and Nahrgang, 2019). To assess the possible risk of PW discharges to Barents Sea fish populations it would be relevant to know whether a low concentration of cardiotoxic PAHs in a PW plume can possibly induce the cardiac toxicity syndrome in fish larvae if they drift through an offshore production field during an active discharge of PW. Hansen et al. (2018a) conducted cardiac toxicity experiments with developing cod embryos exposed to a microbially degraded petrogenic PAH mixture under cold water conditions during a critical period of their heart development. Unexpectedly, significant effects on embryos were found after the PAH mixture had been subjected to a 21-day biodegradation treatment. The authors suggested two possible causes: either PAH metabolites from biodegradation are equally as toxic as the parent PAHs, or there are toxic components within the large UCM fraction that are not measured and that are resistant to biodegradation. The potential ecotoxicity of unknown organic compounds in the UCM hump will normally be assessed by the use of *in vitro* toxicity screening approaches, such as Toxicity Identification Evaluation (TIE) and/or Toxicity Reduction Evaluation (TRE) methodologies (Tietge et al., 1997; Sauer et al., 1997; Thomas et al., 2004b; Elias-Samlalsingh and Agard, 2004; Melbye et al., 2009; Petersen et al., 2017b; Sørensen et al., 2019b, 2020).

The use of immunological responses as effect biomarkers in PW-exposed marine organisms has been investigated with fish such as Atlantic cod (Perez-Casanova et al., 2010, 2012; Holth et al., 2010, 2017), bivalves such as blue mussels (Grundy et al., 1996; Hannam et al., 2009b) and several species of scallops (Hannam et al., 2009a; Hannam et al., 2010a, b; Hannam et al., 2010c). The studies indicate that both PAHs as well as other organic and inorganic components of PW mixtures may have immune modulating properties. Fish have an advanced cell-mediated and humoral (antibody-mediated) immune system comprising a network of cells capable of rapid proliferation and differentiation, regulated by a variety of factors, and closely integrated with other organ-systems and functions. Although this system is responsive to exposure and stress from PAHs and other ecotoxicants, the responses are largely non-specific. Therefore, it is a challenge to understand the immune-modulating effects that PW pollutants may have on fish, because of the complexity of the immune system and the combined effects on immune function of toxicants and various natural stressors (see van der Oost et al. (2003) for an overview discussion). Blue mussels and other bivalves, on the other hand, have a much less advanced cell-based immune system which is affected by pollutant stress and which has been much used in ecotoxicological research and monitoring studies (see Beyer et al. (2017) for a summary). Indeed, pollutant-induced change to lysosomal membrane stability (LMS) in blue mussel haemocyte cells has been among the most successfully used biomarkers in the Norwegian offshore monitoring program (Brooks et al., 2011b).

Since 2002, high-throughput "omics" technologies involving multi-endpoint methods in genomics, proteomics, or metabolomics, have been used increasingly for ecotoxicological biomarker discovery research, e.g. (Aardema and MacGregor, 2002; Martins et al., 2019). The key principle of an omics approach is to analyse a large number (often thousands) of unidentified analytes per biological sample to assess complex patterns of up- and down-regulations in organisms responding to chemical exposures. In Norway, omics methods have been applied frequently in biomarker discovery studies to search for new measures of effects in fish exposed to offshore PW discharges (Grøsvik et al., 2006; Kjersem et al., 2008; Bohne-Kjersem et al., 2009, 2010; Karlsen et al., 2011) (see also other references in Table 1). These studies suggest that exposures of fish to dispersed crude oil or diluted PW may cause changes in the protein profiles of eggs, larvae and juveniles of Atlantic cod. Systems affected included those regulating the immune system, fertility, bone and muscle development, eye development, lipid metabolism, cell mobility, apoptosis and other vital functions (Bohne-Kjersem et al., 2009; Karlsen et al., 2011, 2012; Nilsen et al., 2011a; Eide et al., 2018). Hence, omics data may provide insights about the systemic and mechanistic effects of PW exposures, such as effects on lipid metabolism related to fitness, and they lead to the discovery of new biomarkers or sets of biomarkers (Karlsen et al., 2011). Although omics and multi-omics approaches represent very powerful tools for biomarker discovery research, more work on verification and validation is needed to bring candidate biomarkers from the discovery phase to their regular use in pollution monitoring (Paulovich et al., 2008; Bohne-Kjersem et al., 2010).

The effects of oil in seawater are relevant for offshore PW discharges simply because these discharges always contain low concentrations of dispersed oil. However, oil concentrations in PW discharges on the NCS are normally lower than regulatory limits (total hydrocarbon content $<$30 mg/L). For risk and effect assessments of crude oil released to sea, it is important to discriminate between large acute spills and operational releases of PW typically with low oil concentrations. Large accidental oil spills have become rare in recent times as a result of systematic work by industry and governmental bodies to reduce risk. Nevertheless, the largest accidental marine oil spill in history is also among the most recent, i.e. the Deepwater Horizon oil spill in 2010 (Beyer et al., 2016), illustrating the necessity for always maintaining a high alert for oil spill accidents. However, although the many operational offshore PW discharges that are active on the NCS clearly lack the pollution drama associated with major marine oil spills, they still represent the largest point sources of marine crude oil contamination on the NCS. The effects of accidental marine oil spills will therefore, at least as a worst-case scenario, have relevance also for effect assessments of PW discharges.

4. Risk-based assessment and management of offshore PW discharges

When assessing and managing the environmental risk of offshore PW discharges, OSPAR recommends adhering to a harmonised, structured procedure as presented below (Fig. 3). In the OSPAR area, environmental risk assessments of PW discharges have traditionally focused mostly on the residues of dispersed oil, assuming they are the key risk components of PW mixtures. As described in OSPAR Recommendation 2001/1, the management required using of Best Available Techniques (BAT) and Best Environmental Practices (BEP) to minimize the oil content of PW prior to discharge. However, a PW mix may also contain a wide range of other potentially harmful production chemicals, such as heavy metals, corrosion inhibitors, biocides, H_2S scavengers, scale inhibitors, wax inhibitors, emulsion breakers, foam inhibitors, and flocculants (Kelland, 2014). Several modelling tools were developed to assess the risks attributed to offshore production chemicals, such as CHARM (Chemical Hazard Assessment and Risk Management), EIF (Environmental Impact Factor) (Johnsen et al., 2000; Reed and Rye, 2011), and DREAM (Reed and Rye, 2011). The offshore industry has continued to expand the risk tools for PW (and other discharges) with a cluster of 'Risk-Based Approach' (RBA) methodologies under the auspices of the OSPAR Offshore Industry Committee (OIC) (OSPAR, 2012). The key rationale for using RBA in PW management is to focus mitigation actions on discharges and substances that represent the greatest risk to the environment. The RBA requires that risks of PW discharges are assessed with either a whole effluent toxicity (WET) approach, or a substance based (SB) approach, or a combination of the two (de Vries and Jak, 2018; Karman and Smit, 2019). The responsible operator must also generate site-specific data that enables prediction of how much risk is associated with each specific discharge (Karman and Reerink, 1998). For substances that cause direct effects, the basis for the risk assessment is a PEC:PNEC (predicted environmental concentration/predicted no effect concentration) and/or a msPAF (multisubstance potentially affected fraction) approach. For bioaccumulative and persistent substances that can induce long-term effects when a threshold body burden is exceeded, other ways of risk assessment are required. The use of toxicokinetics in PW risk models introduces the possibility to relate effect predictions to internal exposure concentrations (body burden) in organisms, and not just to water concentrations. Body burden is then something that can be measured in the field for verification of the model simulations. Research is ongoing to enable modelling of uptake, body burden and associated toxicity of PW PAHs and alkylphenols in cod eggs and to use these data to further predict risks of harmful effects by employing a novel toxicokinetics module in DREAM, i.e. the so-called DREAM-MER setup (Nepstad et al., in press).

The relevance of biodegradation issues in relation to the risk assessment of diluted PW discharges has attracted increasing research attention, e.g., (Brakstad et al., 2008, 2018; Lofthus et al., 2016, 2018a; Liu et al., 2016; McFarlin et al., 2018). According to Brakstad et al. (2008, 2018), oil biodegradation is generally slower in cold arctic seawater than in water from temperate seas, especially for saturates (linear, branched and cyclic alkanes). They suggested that slower biodegradation is caused by lower macronutrient concentrations (both N- and P-compounds) in the arctic seawater. They also pointed out that predictions of oil degradation rates in arctic seawater should be based on experimental data obtained directly from the relevant Arctic environment, rather than adjusting results from temperate seawater

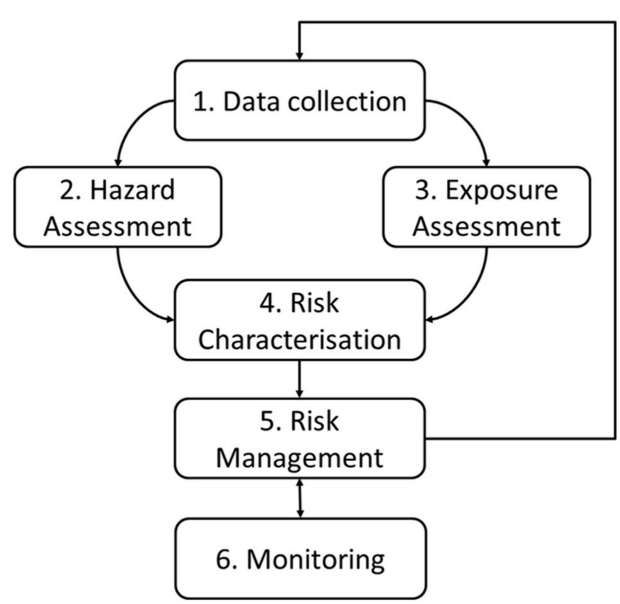

Fig. 3. The overall structure for OSPARs Risk-Based Approach (RBA) used for management of PW and other operational waste discharges in the offshore oil and gas industry, source: (OSPAR, 2012).

environments.

It is an unresolved question whether high dosage effect data can be extrapolated to low-concentration situations to assess possible risks of long-term impacts, or whether results from one chemical can be extrapolated to a situation with multiple (possibly unknown) contaminants with the same (or different) Mode of Action (MOA). Exposure-effect data (e.g. No Observed Effect Concentration NOEC and Lowest Observed Effect Concentration LOEC values) are typically obtained by short-term exposures to high concentrations of PW or PW constituents, e.g., (Caliani et al., 2009). Extrapolating such high concentration effect data to field situations often remains questionable. To be realistic, extrapolations need to consider the toxicokinetic behaviours of PW constituents at high dilutions and the toxicodynamic effect mechanisms and possible adverse outcome pathways that are involved when organisms are exposed to low dosages. There have also been many objections raised regarding the adequacy of risk assessment tools that build on NOEC or LOEC test data, hypothesis testing, and significance testing based on *p*-values, like the PEC/PNEC approach. This relates to the quarter of a century-long controversy between the schools of frequentist statistics (Fisher, 1955; Huggett et al., 2002; Lele and Allen, 2006) and Bayesian statistics (Kooijman, 1996; Landis and Chapman, 2011; Fox and Landis, 2016), a dispute which is outside the scope of this paper.

5. Is the Barents Sea more sensitive to PW discharges?

The Barents Sea is among the most productive oceans in the world, with many commercially important species, particularly the stock of Northeast Atlantic cod (Jakobsen and Ozhigin, 2011; Ottersen et al., 2014). These resources are under threat from several natural and man-made pressures, although overall, their present status appears healthy. The key ecological characteristics of the Barents Sea are: (1) the division between a relatively warm southern and western part mostly influenced by Atlantic water, with the sea surface always warmer than 0 °C, and a significantly colder northern and eastern part influenced by Arctic water, with the sea surface always colder than 0 °C, and ice-covered for large parts of the year. The polar front (a.k.a. the marginal ice zone or the ice edge) being the border between the two; (2) short and simple food webs with a relatively small number of keystone species crucial for the overall ecosystem function; (3) a very strong seasonality of biological production; and (4) a strong pelagic-benthic coupling of energy flow. Scientific information about the Barents region is reported in thousands of papers and tens of thousands of reports and the area is currently under even more scrutiny under the auspices of the large Norwegian “Nansen Legacy” research program (https://arvenetternansen.com/).

The ecosystems located north and east of the polar front are representative of true arctic conditions whereas the systems located south and west of the polar front resembles more sub-arctic to boreal conditions. The warmer southern and western parts of the Barents Sea are dominated by salty water streaming northerly into the area driven by the North Atlantic current and the Norwegian coastal current, whereas the colder northern part is dominated by the less-saline, ice-cold arctic water streaming south from the Arctic Ocean. The polar front fluctuates between a northern extreme during the summer and fall and a southernly extreme during the winter and spring season. The long-term warming of the Barents Sea that is presently associated with global warming will decrease stratification, expand the area with boreal and sub-arctic conditions, and shrink the area that has polar conditions (Wassmann et al., 2011; Fossheim et al., 2015; Kedra et al., 2015; Lind et al., 2018; Møller and Nielsen, 2019). These changes will represent a huge challenge to the populations of cold-water species in the Barents Sea region, and these populations will need to move further north and east to find suitable living conditions.

The possible increase of Norwegian oil and gas activities in the southwestern part of the Barents Sea has raised environmental concerns. The activities started in 1980 but only two fields (Snøhvit and Goliat) have so far been developed and a third field (Johan Castberg) is presently being developed. More developments are expected as resource surveys suggest that the Barents Sea contains about one third of the remaining hydrocarbon resources on the NCS. The increased oil and gas industry activity in the Barents region has been and still is politically controversial. To ensure the best possible protection of the Barents Sea ecosystem against pollution from offshore operations, a *zero-physical-discharge* regulation was implemented in 1998 by the Norwegian authorities for new oil and gas field developments in this region (Marthinsen and Sørgård, 2002; Anonymous, 2003; Hasle et al., 2009; Knol, 2011). This ultrastrict regulation was soon replaced with a zero-*adverse-effect* goal, referring to normal operations, and was implemented at all offshore fields on the NCS, partly to speed up the phase-out of all environmentally hazardous offshore chemicals (ibid.). A consequence of the zero-*adverse-effect* regulation was an increased need for precise knowledge about possible biological effects of offshore discharges, especially regarding the Barents Sea where the ecological resources are particularly large, and the amount of effect data was very limited. It became a question to what degree effect data generated with temperate species and systems from the North Sea were representative for the Barents Sea context. During the 1960s, a general view had risen in the scientific debate that organisms and ecosystems in the Arctic were systematically more sensitive to pollutant exposures than temperate systems. Some authors opposed that view, stating rather that organisms and systems in cold marine regions such as the Barents Sea and the Arctic are probably comparable to organisms and systems elsewhere (Dunbar, 1973, 1992, 1977; 1986). They inferred that the dangers of chemical pollution in the high north are of the same sort and magnitude as elsewhere. A key reasoning was that all species are evolutionarily adapted to their natural environment. Therefore, an assertion that harsh polar conditions in themselves automatically make arctic (or Barents Sea) species, populations, and ecosystems more sensitive and vulnerable to PW or other pollutant stressors, is not scientifically valid. Olsen et al. (2013b) compiled oil exposure and toxicity data for mortality, development, growth, bioaccumulation and reproduction for a selection of cold-water marine fish and plankton associated with the Barents Sea. They concluded that the data were limited to a sub-set of the required endpoints, highlighting the need for more experimental studies, especially for development and bioaccumulation endpoints in key zooplankton, and for growth and development endpoints in larvae and juveniles of key fish species (ibid.).

The strong seasonality of primary and secondary production causes winter starvation and emaciation of high north animals, followed by a period of ‘fattening’ during the summer months, when food typically occurs in excess. Research on arctic charr has demonstrated that seasonal emaciation temporarily increases tissue concentrations of persistent and toxic pollutant chemicals, such as PCBs, leading to extra high sensitivity to toxic effects in May, just prior to the onset of the fattening period (Jørgensen et al., 2006; Letcher et al., 2010). These results suggest that all Arctic species that accumulate large fat deposits during intense feeding periods and tolerate long periods of starvation during winter and spring will also be subjected to seasonal extremes in tissue concentrations of persistent, lipophilic contaminants. The study by Toxværd et al. (2018) found that *Calanus glacialis* showed high sensitivity to the four-ring PAH pyrene during the overwintering phase in the annual cycle. However, other studies found *C. glacialis* to be *less* sensitive compared to *Calanus finmarchicus* to a stress combination of pyrene exposure and increasing water temperature (Hjorth and Nielsen, 2011). Lower sensitivity of *C. glacialis* than *C. finmarchicus* has also been observed by others (Hansen et al., 2011, 2013a), leading to further studies into whether this species sensitivity difference could possibly be due to differences in lipid content and toxicokinetics (Hansen et al., 2016a; Øverjordet et al., 2018). In another study, C5 stage *Calanus* copepodites that were exposed to the water-soluble fraction of a naphthenic crude oil during their diapause termination showed a slower utilization of lipid stores and down-regulation of genes in the beta-oxidation pathway (Skottene et al., 2019). The authors underscored that any significant delay on the triggering of the copepod diapause termination and subsequent migration to the surface could have serious ecological implications. In natural populations of copepods collected at offshore oilfields, Hansen et al. (2020b) recently showed correlations between copepod PAH profiles and proximity to PW outfalls and possible responses of copepod lipid metabolism indicated by lipidomic and metabolomic analyses.

Camus et al. (2000) studied the effect of a combined exposure to low temperature and the common PW hydrocarbon phenanthrene on the stability of lysosomal and cell membranes in haemocytes of the blue mussel (*Mytilus edulis*). Interestingly, they found a destabilizing effect caused by low temperature, but not by the phenanthrene exposure dosages (ibid.). Low temperatures will make it necessary for cell membranes to increase their relative content of unsaturated lipids to maintain membrane fluidity, and such acclimations could possibly have relevance for membrane associated ecotoxicant effect mechanisms. It can be questioned which organisms are the most suitable for investigating species sensitivity differences among temperate, boreal and arctic habitat conditions. Several comparisons have used copepods such as *Calanus finmarchicus, C. hyperboreus,* and *C. glacialis* and *Acartia tonsa* which are all key zooplankton species, e.g. (Hansen et al., 2011, 2013a, 2013b, 2014). In acute toxicity testing, the most sensitive copepod was often the temperate species *A. tonsa*, which thus may provide conservative effect estimates if it is used as a representative copepod in risk assessments (Hansen et al., 2014).

Camus et al. (2015) studied possible sensitivity differences between arctic and temperate species by comparing a species sensitivity distribution (SSD) made from toxicity testing with an artificial PW mixture and six temperate test species versus a SSD that was made based on similar testing with six arctic species. Each species group included algae, molluscs, crustaceans and fish. Overall, the temperate and arctic groups showed a comparable sensitivity towards the exposure mixture. Bejarano et al. (2017) also used an SSD approach to compare the relative sensitivity of arctic and non-arctic species to the toxicity of physically and chemically dispersed oil. They also found that the sensitivities of arctic and non-arctic species were comparable. Yet, even though biological sensitivity to toxicants may be the same in arctic and temperate species, one cannot exclude the influence of over-arching ecosystem factors. The spatiotemporal distribution of PW discharges and their effects could be modified by ecosystem complexity, seasonal environmental characteristics, spatial distribution of populations and communities, and population interactions in such a way that one region becomes more vulnerable than another. However, as pointed out by Hjermann et al. (2007), such factors may be too stochastic to enable useful predictions of the overall impact of a PW discharge. For example, the fraction of the total population of a species that must be impacted by an operational PW discharge to elicit significant population damage may not be a constant figure and will easily become a matter of conjecture.

As earlier noted, PW discharges typically include small amounts of crude oil that could be toxic to fish eggs and larvae, with possible consequences for population recruitment in important spawning areas, see (Hjermann et al., 2007; Stige et al., 2011, 2018; Nordtug et al., 2011; Vikebø et al., 2014; Sørensen et al., 2017; Langangen et al., 2017b). Recently, Carroll et al. (2018) simulated how the Barents Sea cod stock may be impacted by a major spill in its core spawning areas in the Lofoten area. By modelling the life history of individual fish eggs and larvae, they predicted that the most severe oil spills (4500 m^3/day over 90 days) caused population losses of up to 43% for the pelagic stages. Still, the recruitment of juveniles into the adult stock was sufficient for maintaining the reproductive health of the population. Thus, the cod population recruitment appeared rather insensitive to adverse effects from the oil spill, although the authors recommended their results should be applied cautiously. Also, the Lofoten area is key for other cod populations and fish species that may be less resilient (Hjermann et al., 2007).

Pollutant sensitivity studies with testing of fish under low-temperature conditions have frequently focused on polar cod (see numerous citations in Table 1). Polar cod is a keystone species for the Arctic ecosystem on the cold side of the polar front. It is specialized for a life in ice-cold waters and lives further north than any other fish species (Hop and Gjøsæter, 2013). Laurel et al. (2019) found that polar cod are sensitive to the developmental impacts of crude oil exposure, especially during mid-organogenesis when a transient exposure to 0.3 mg/L of oil in water significantly disrupted jaw and heart development in exposed fish. An even lower oil exposure led to a dysregulation of lipid metabolism and growth in otherwise morphologically normal juveniles. Nahrgang et al. (2016) found that early life stages of polar cod were negatively affected (increased incidence and severity of spine curvature, yolk sac alterations and a reduction in spine length) by hydrocarbon and PAH contaminants from water-soluble fractions of crude oil in the ng/L range. Most exposure concentrations were below the limits of detection throughout the experiment for all treatments. Nahrgang et al. concluded that the viability and fitness of polar cod early life stages can be significantly reduced even at extremely low and environmentally realistic concentrations of aqueous hydrocarbons. In maturing polar cod, on the other hand, no notable long-term effects on growth were observed in fish exposed to environmentally realistic concentrations of mechanically dispersed oil, chemically dispersed oil or burned oil residues (Bender et al., 2018). However, Geraudie et al. (2014) observed effects on reproductive function of polar cod exposed long term (up to 28 d) to relatively high concentrations of PW-relevant chemical mixtures, including responses of biomarkers of biotransformation, endocrine disruption and gonad histology. These, and more recent follow-up studies exposing polar and Atlantic cod to WSF of crude oil during gonadal maturation, showed an effect on ripening of oocytes and gamete quality, indicating that petroleum exposure affects the reproductive function of adult fish (Nahrgang & Vieweg, pers. comm.). Taken together, polar cod appears particularly sensitive to petroleum hydrocarbons both during early life stage development and sexual maturation, showing that the timing of exposure is important.

6. Discussion

This review examined our knowledge of the fate and effect of PW discharges from offshore oil and gas production. Some key conclusions are, first: PW discharges are the largest source of crude oil and petrogenic PAH pollution to the seas from the offshore operations on the NCS; second: *acute* effects that have been measured in mussels and fish after they have been exposed *in situ* for several weeks to naturally diluted PW in the proximity of offshore production platforms are relatively mild; third: elevated concentrations of petrogenic bile metabolites (small PAHs and APs) in caged fish have been detected as far as 10 km from the nearest PW discharge; and fourth: haddock tend to show stronger effects of petrogenic exposure compared to Atlantic cod and other gadoids, although effects are not necessarily caused exclusively by PW. The best example of the latter is the increased DNA adduct levels that have been found repeatedly in wild specimens of haddock (and other benthic fish species) in several regions of the NCS, sometimes in proximity to and sometimes relatively far from oil and gas production installations. PAHs are the most likely causal substances for DNA adduct formation, but the exact source(s) of these PAH contaminants (drilling discharges, PW discharges, other) is not yet clarified.

In the second part of this knowledge summary we addressed the ecological state of the Barents Sea region and whether marine resources could be extra sensitive to anthropogenic pollutants, including PW and other operational discharges from oil and gas activities. The total publication base on the Barents Sea is huge, but only a few studies have addressed this issue, e.g. (Olsen et al., 2011; Hansen et al., 2014; Camus et al., 2015; Bejarano et al., 2017; Szczybelski et al., 2019b). Overall, there appears to be a paucity of data showing a systematically higher sensitivity among the Barents Sea cold water species in comparison to their temperate/boreal counterparts. Rather, the available literature seems to suggest there are no systematic differences as both more sensitive and more tolerant species can be found in both systems. Nevertheless, species and populations living in the cold part of the Barents Sea region (and the Arctic generally) could still be more vulnerable as their life histories are tuned to the extreme changes taking place between the dark and cold polar night in winter, and the continuous daylight during summer. The combined effects of global warming and pollutant stressors may shift the homeostatic balance in the wrong direction. Another critical aspect is that there is a finite limit to how far north or east these species could move to avoid rises in habitat temperature.

Another question is whether the ecosystem 'simplicity' of the Barents Sea (few keystone species, simple food web, etc.) determine its resilience to significant external perturbations. For example, will a significant disturbance of one keystone species be more severe in a Barents Sea context than in temperate or boreal systems elsewhere on the NCS? Are northern ecosystems more likely to undergo regime change due to external disturbance? How ecosystem complexity affects ecosystem stability is a long-standing debate, but in general the evidence indicates that more complex ecosystems are more stable than simpler ones (Hooper et al., 2005). Changes that the fish stocks of the Barents Sea have gone through over the last 40 years indicate that significant disturbance to key species may have strong and long-lasting repercussions. For instance, in the years following a successful reproduction of herring, the large amounts of young herring in the Barents Sea induced a collapse in the capelin stock due to high predation on larval capelin. In turn, this led to poor growth and increased cannibalism in subadult and adult cod in the following years, e.g. Hjermann et al. (2004). A recent study by Olsen et al. (2019), using the highly complex Atlantis ecosystem model, showed even more dramatic long-term instabilities. A 10% reduction in recruitment in one single year (e.g. due to an oil spill) may lead to dramatic changes (~40% reduction of herring, >100% increase of haddock) as much as 30 years after the perturbation! However, in that study the herring stock showed a continuous decrease over the next decades even in the control run of the model, casting some doubt on the model's validity.

In the context of the zero-*adverse-effect* regulatory goal for the NCS, the difference between regulations introduced because of precautionary considerations and regulations introduced because of empirical knowledge of effects (or adequate predictions of risks of effects) must be understood. Precautionary considerations are often applied in strategic decision making for industrial operations in regions of special uniqueness and potential vulnerability, particularly when effect knowledge is lacking. This is the key to the precautionary principle: the less information there is on vulnerability, the stricter the means to safeguard and protect. Later, when and if the knowledge is improved and new insight allows, the regulatory regime can be eased. However, during the period when knowledge remains limited, it is important that strict precautionary protection measures are not mistaken as empirical proof of *actual* sensitivity or vulnerability. This problem was discussed in a popular manner by Gray (2002), who highlighted the important difference between a *perceived risk* and a *real risk* for offshore PW discharges. Gray made the point that although PW mixtures may contain a suite of chemicals that are toxic to test organisms in controlled laboratory or field-caging experiments, a PW discharge will not necessarily cause similar impacts in wild organisms exposed in real-world situations. Effects generated in controlled toxicity tests represent a *perceived risk*, whereas effects that materialise in wild organisms/populations are representative of *real risks*. However, one could also argue that effects observed in controlled tests represent potential risks, under worst-case scenario conditions. But also, that real risks are almost impossible to measure until the system has responded, in which case it is too late, and the resources have not been protected.

Most laboratory and field studies of PW impact assessments seem to suggest that significant *acute* effects on water column organisms caused by chemical constituents in PW discharges on the NCS are limited to the first 100 m, and possibly up to 1000 m, of the diluting PW plume. The degree of effects will depend on the volume and composition of the PW discharge and the physicochemical and ecological conditions and properties of the receiving waters. However, the cumulative total of PW effluents released from active production platforms in the whole North Sea will eventually contribute to the total (diffuse) petrogenic contaminant input to the North Sea (Nepstad et al., in press). In such a wider perspective, the discovery of increased DNA adduct concentrations in some regional haddock populations, and other sediment-feeding benthic fish species, over large areas of the North Sea will be relevant to consider. It is widely agreed that PW discharges presently account for the largest operative input of crude oil (and crude oil associated contaminants) to the North Sea. Therefore it will be relevant to know whether the diffuse distribution of elevated DNA adduct signals are due to PW or to other sources of petrogenic PAH pollution, such as previous large-scale discharges of oil-based drill cuttings that took place in many North Sea oil fields, not least in the Tampen region (Gray et al., 1999; Grant and Briggs, 2002).

6.1. Needs for more knowledge

It is inherently difficult to make reliable extrapolations from effects on individuals to effects on populations. Effects on individuals may be masked by factors acting on populations, such as distribution patterns, seasonality, species interactions, density-dependent functions, other stressors, and the complex and dynamic physical conditions of pelagic ecosystems. There is, therefore, a continued need for more knowledge about the ecotoxicological effects of chronic PW exposures at very diluted concentrations, and biomarker discovery research must be focussed on such scenarios. It is a question whether we presently monitor adequate sets of contaminants and possible ecotoxicity drivers in connection with offshore PW discharges. For example, recent effect studies with early life stages of zebrafish, Atlantic cod and haddock suggest that there are other PW substances than PAHs that may contribute significantly to toxicity and also that mixture effects (e.g. synergism) is not sufficiently accounted for (Sørensen et al., 2019b, 2020). Further, more knowledge is needed to explain the main cause(s) of unexpectedly high concentrations of DNA adducts in haddock from Tampen and several other areas of the North Sea, especially to clarify whether PW discharges play a causal role. Further in-depth analyses of other health parameters and/or low-concentration-responsive biomarkers should be conducted with emphasis on fish populations that show a high prevalence of DNA adducts. For example, what is the possible influence on lipid metabolism in such adduct impacted fish populations compared to unstressed populations? The effect-based environmental monitoring of oil and gas operations on the NCS is unprecedented, but the actual relevance and suitability of the parameter set used in this program remain questionable. There is a need for quality assurance and quality control systems for these ecotoxicological biomarkers, as well as a certain minimum of validation data that demonstrate their suitability in effect monitoring of offshore PW discharges. Biomarkers in fish and other marine species must also be developed to evaluate sublethal effects that are relevant to biological fitness, using study conditions that are environmentally realistic. It is of key importance to keep the monitoring methodology as user-friendly and simple as possible. Environmental risk assessment tools provide robust methods for assessing the likelihood of PW-associated impacts in wild fish populations in the North Sea. Nevertheless, there is an ongoing debate regarding the adequacy and suitability of NOEC and LOEC estimates as a basis for risk estimates. There is also a general shortage of information concerning the possible effects and risks of oil-related compounds at all levels of biological organisation in cold-water marine ecosystems, although the research performed to date suggests that the differences in sensitivity to PW and PAH toxicity between species from Arctic or sub-Arctic seas and temperate or boreal seas are smaller than many have expected. It has also become clear that timing of exposure is important, with the early life stages and the sexual maturation in fish being particularly sensitive stages. Further research should aim to broaden and deepen this field of knowledge. The research should particularly address the sub-lethal and long-term impacts of environmentally realistic low-dose exposures to PW and mixtures of associated contaminants.

6.2. Summary

Offshore oil and gas activities on the NCS must comply with zero adverse effect regulations that require operations to not cause significant negative impacts to marine biological and ecological resources. Effects-based research and monitoring are performed by the industry and independent research institutions to investigate the compliance to this regulation. The industry has refined their operations with the objective of minimizing environmental risk and impacts without compromising efficient production. Continuous discharges of PW still represent a concern as they are the largest source of crude oil and PAH contamination to sea from oil and gas operations offshore. Both risk-based modelling and effects-based monitoring indicate that only mild acute effects in populations of water-column biota are caused by these PW discharges. More recently, the attention of effects-based offshore studies has shifted towards possible effects that may result from chronic, low-concentration exposures, especially in populations of benthic fish species. Most importantly, haddock populations from different areas in the North Sea have repeatedly shown signs of exposure to substances causing DNA adducts, most likely petrogenic PAHs. However, the exact sources (drilling discharges, PW discharges, or other) are not clarified. The prospect of future increased oil and gas activity in the ecologically rich Barents Sea is met with concern. There are multiple signs suggesting that major ecosystem changes in these regions are ongoing, driven predominantly by global warming. All species that require arctic or subarctic conditions are vulnerable simply because there is a limit to how far north they can move to avoid increasing ambient temperatures. In that context, increased competition from temperate species migrating northward is expected. Such immigration can be expected to make high north species even more vulnerable. There is a strong rationale for research to improve the knowledge of the effect of anthropogenic pressures and their combinations, on organisms and ecosystems in the Barents Sea and in the Arctic in general. For example, area-wise comparisons show that haddock collected in the Barents Sea are considerably less exposed to substances causing DNA adducts than comparable haddock from the less-exposed parts of the North Sea. Such a low-contaminated condition status for Barents Sea benthic fish species is a valuable ecosystem quality by itself. Toxicity tests have yet to support the notion that subarctic species from the Southern Barents Sea are systematically more sensitive to PW effluents or other offshore industry contaminants than their temperate counterparts in the North Sea. Sensitive species and more tolerant species are found in both places. However, a combination of exposure during sensitive life stages and special life history adaptations to Arctic environments raise concerns about potential effects on reproduction and recruitment in the Barents Sea. Hence, due to the increasing releases and the remaining effect uncertainty, the offshore PW issue should receive continued attention from ecotoxicologists as well as regulators for years ahead.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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