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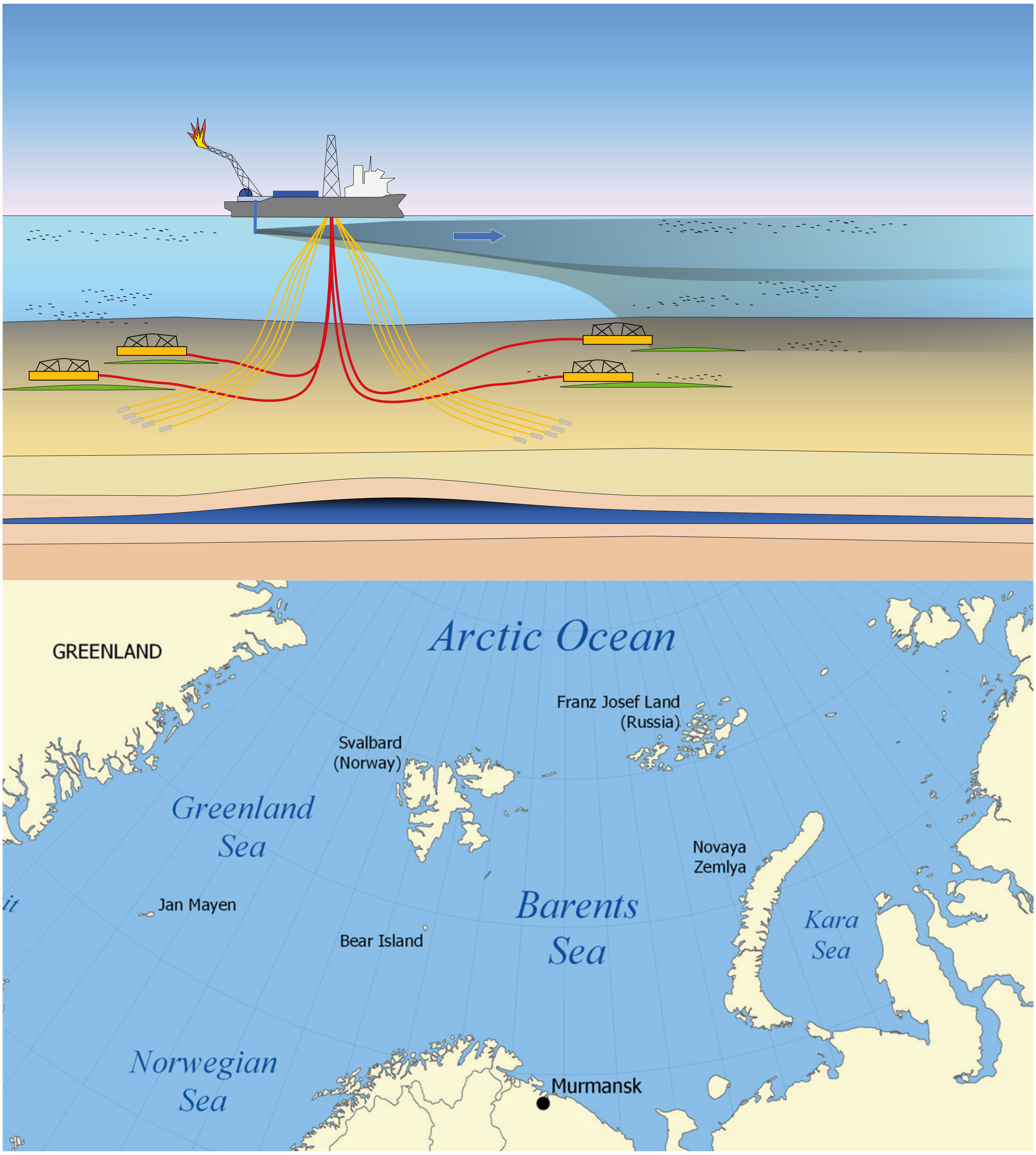
Environmental effects of offshore produced water discharges evaluated for the Barents Sea



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Norwegian Institute for Water Research

REPORT

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Summary

This report presents (1) an overview of the ecotoxicological knowledge regarding offshore produced water (PW) discharges on the Norwegian Continental Shelf (NCS), (2) an overview of the ecological characteristics and values of the Barents Sea, and (3) an overview of the tools that presently are available for assessing the possible environmental risk associated with offshore PW discharges. The overall objective with this work is to provide a basis for discussing the need for a stricter environmental management of PW discharges in the Barents Sea in comparison to elsewhere at the NCS. Based on information presented herein there appear not to be a clear systematic pattern that organisms/systems in Barents Sea are significantly more sensitive to chemical contamination and ecotoxicity of PW compared to organisms/systems elsewhere at the NCS. However, there are still many unknowns in this field of study, as only a relatively small number of PW effect studies provide data for sensitivity comparisons of Barents Sea and non-Barents Sea organisms/systems.

Four keywords	Fire emneord
1. Offshore Produced Water Discharges	1. Offshore produsert vann utslipp
2. Barents Sea	2. Barentshavet
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The Expert Group for Offshore Environmental Monitoring

Environmental effects of offshore produced water discharges evaluated for the Barents Sea

Preface

In this report, the expert group on offshore environmental monitoring appointed by the Norwegian Environment Agency (NEA) provides a broad knowledge summary of the ecotoxicological implications of offshore produced water (PW) discharges, especially viewed in a Barents Sea context. The rationale for making this report is the need for providing an updated input to the knowledge base for the Norwegian governmental management plan for the Barents Sea ecoregion. The main author of the report is Jonny Beyer with co-author contributions from Torgeir Bakke and Rainer Lichtenthaler (both NIVA, presently retired), and Jarle Klungsøyr (Institute of Marine Research (IMR), Bergen, Norway). Expert inputs to the report have been provided by Prof. Gro van der Meeren (IMR, Bergen) and Prof. Anders Goksøyr (IMR and University of Bergen). Quality assurance of the final report draft was provided by Steven Brooks (NIVA).

Oslo, 22. May 2019

Jonny Beyer

Table of contents

Summary

A question has been raised whether the ecotoxicological risks associated with offshore produced water (PW) discharges are significantly and systematically larger in the Barents Sea (and the Arctic seas) compared to oil and gas extraction regions elsewhere on the Norwegian Continental Shelf (NCS). In this report, practically all the Norwegian environmental research and monitoring activities on offshore PW discharges are summarized. In addition, an overview is provided of the key ecological properties of the Barents Sea. On this dual foundation, the question of sensitivity to PW is discussed. The rationale for making this report is a need for providing an updated input to the knowledge base for the Norwegian governmental management plan for the Barents Sea.

The offshore oil and gas industry operating on the NCS uses risk-based management tools for the natural and added substances in PW discharges. These risk simulations suggest that the risk for adverse impact on wild fish populations due to PW discharges is generally negligible. The low risk has even decreased further in recent years because of better PW treatment and other PW management improvements (increased reinjection). Laboratory based PW effect research and repeated field-monitoring surveys in the North Sea show that organisms encountering diluted PW plume water express signs of contaminant exposure and biomarker effects. These responses are, however, generally within a tolerable range, and do not indicate that there are significant and adverse effects of PW occurring at population or community levels in areas downstream from oil and gas installations.

Strict regulations on PW management were initially put in place for offshore developments in the Barents Sea. However, research has yet to find reasons for claiming that Barents Sea ecosystems and organisms are systematically more sensitive to PW associated contamination than comparable ecosystems and organisms at other offshore fields. Certain species within both categories (Arctic and non-Arctic) appear to be more sensitive than others, and research to unravel the reasons for the differences in species sensitivities is ongoing.

The research community share a deep concern for the future survival of the cold-sea ecosystems of the Artic and the Barents Sea region. Multiple signs suggest that major ecosystem changes are ongoing in the whole Arctic region driven by the processes associated with regional and global warming. In that context, Arctic species are particularly vulnerable simply because there is a clear limit to how far north they can move to adapt to rises in air and sea temperatures and declines of sea ice. Increased competition from southern species migrating north is expected to even out the species differences between the western/southern Barents Sea area and shelf areas further south. Such invasions may also make marine species and communities of the high north even more vulnerable.

These factors clearly suggest the rationale for continuing to lessen the anthropogenic pressures on the Barents Sea and the Arctic ecosystems. The progress and success of this process will depend on our ability to identify the most important man-made ecological perturbations and to find efficient management solutions for them.

Based on the relevant scientific literature summarized herein, the possible ecological risk associated with offshore PW discharges is most probably not larger at offshore fields of the Barents Sea than elsewhere on the NCS, and by all practical means this risk is negligible compared to the much bigger threat to these systems from global warming. However, there are still many unknowns in this field of study, as only a relatively small number of PW effect studies provide high-quality data for sensitivity comparisons of Barents Sea and non-Barents Sea organisms/systems.

Sammendrag

Tittel: Miljøeffektvurdering av offshore produsert vann utslipp i Barentshavet

År: 2019

Forfatter(e): Jonny Beyer, Torgeir Bakke, Rainer Lichtenthaler, Jarle Klungsøyr

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Det har vært reist spørsmål om den miljøtoksikologiske risikoen forbundet med utslipp av produsert vann (PV) fra offshore petroleumsutvinning kan være større i Barentshavet (og arktiske havområder generelt) enn på andre olje- og gassfelt på norsk kontinentalsokkel. I denne rapporten gis det en oversikt over den norske miljøforskningen og overvåkingen av PV-utslipp, i tillegg gis en kort økologisk beskrivelse av Barentshavet. Med grunnlag i disse kunnskapsoversiktene blir så spørsmålet om mulig høyere PV sensitivitet for Barentshavet vurdert. Hensikten med arbeidet og rapporten er å bidra til å oppdatere kunnskapsbasen for den norske statlige forvaltningsplanen for Barentshavet.

Olje- og gassindustrien som opererer på norsk sokkel, bruker risikobaserte styringsverktøy for stoffene som fins i PV-utslippene. Disse risikosimuleringene viser ubetydelig risiko for uønskede miljøeffekter av PV-utslipp på villfiskpopulasjoner offshore. Den lave risikoen har til og med blitt ytterligere redusert de siste årene som følge av forbedret teknologi for rensing av PV og reinjeksjonsrutiner. Forskning og overvåking finner målbar eksponering og visse økotoksikologiske responser hos organismer når de eksponeres for fortynnet PV, men effektene vurderes totalt sett å være innenfor et tolerabelt område. Det er ikke gjort funn som tilsier at det forekommer vesentlige miljømessige effekter på ville bestander eller på økosystemnivå i norske farvann.

Ekstra strenge regler for håndtering av PV ble innført for de første feltetableringene i Barentshavet, men bekymringene for om arter og økosystemer i Barentshavet er mer sensitive for PV utslipp enn ellers på norsk sokkel har ikke blitt understøttet av funn i forskning. Enkelte arter (arktiske og ikke-arktiske) kan vise seg å være mer følsomme enn andre, men da basert på spesifikke artsegenskaper.

Det er stor bekymring i forskningsmiljøet for hvordan fremtiden blir for økosystemene i Barentshavet og Arktis forøvrig. Mange tegn tyder på at det skjer spesielt store endringer her drevet frem av global oppvarming. I denne sammenhengen er arktiske spesialiserte arter veldig sårbare rett og slett fordi det er begrenset hvor langt nordover de kan bevege seg som en tilpasning til varmere klima og nedgang i mengde sjøis. Det vil også bli økt konkurranse fra sørlige arter som beveger seg nordover etter som miljøforskjellene mellom Barentshavet og sokkelområdet lengre sør reduseres. Det øker risikoen for at rene arktiske arter kan bli utryddet.

Det er lite trolig, med utgangspunkt i dagens kunnskap, at PV-utslipp vil ha større miljømessig effekt på felt i Barentshavet enn i andre havområder på norsk sokkel. De miljømessige effektene av slike utslipp i Barentshavet vil sannsynligvis være ubetydelige, særlig sett i forhold til den alvorlige påvirkningen som global oppvarming mest sannsynlig vil forårsake i dette området. Men vi har fremdeles behov for økt kunnskap om effektene av menneskeskapt påvirkning, inklusiv PV-utslipp, i Barentshavet og Arktis, og om hvordan vi kan utvikle og iverksette best mulige mottiltak mot de viktigste av disse påvirkningene.

1 Introduction

1.1 Background

The Barents Sea is the large (approx. 1.4 million km^2) and shallow (average depth 230 m) arctic-boreal continental shelf sea that borders to the Norwegian Sea in the west/south-west, to the Arctic Ocean in the north and to Novaja Semlja in the East. It differs from other offshore areas further south on the Norwegian continental shelf (NCS) by its low sea temperatures and strong seasonal fluctuations. The Barents Sea is a productive ocean and includes large populations of fish, seabirds, and marine mammals. The fish species/stocks of greatest commercial importance are the Northeast Arctic cod (*Gadus morhua*), Northeast Arctic haddock (*Melanogrammus aeglefinus*) and Barents Sea capelin (*Mallotus villosus*).

Norway is a major producer of oil and gas and all the production fields are situated offshore on the NCS. Production operations at these fields include continuous and voluminous production of oily produced water (PW) as a by-product, and which for a large part is discharged to sea after treatment. Concern of PW discharges as a potential environmental risk gained much attention in Norway and received renewed attention recently because of offshore activities increasing in the ecologically rich Barents Sea.

1.2 Objectives

The objective of this study is to make an overview of the research literature that concern environmental effect of offshore PW discharges, with emphasis on issues that are relevant for the Barents Sea and Arctic seas, and particularly regarding Norwegian research studies. The work is an update of the review by Bakke et al. (2013) summarising recent research on operational discharges of PW offshore. The scope of the present work is to contribute to the knowledge base underpinning the governmental management plan for the Barents Sea-Lofoten area.

We want to address/answer the following partial questions:

- What new knowledge has emerged after 2010 about the possible ecological effects of PW discharges, and particularly regarding issues relevant for Barents Sea and Arctic waters?
- Are species, populations, and ecosystems in the Barents Sea systematically more sensitive and vulnerable to PW discharges than species, populations, and ecosystems in temperate shelf seas?
- Is there a comparable level of understanding about the ecological risk of offshore PW discharges in Barents Sea / Arctic marine waters as compared to temperate marine waters, both at the organism and ecosystem level?
- Will Barents Sea ecosystems respond differently from temperate ecosystems because of differences in climatic conditions, ecological seasonal variation, distribution of biological resources in time and space, and overall accumulation and magnification of PW contaminants and other environmental pollutants?

Although this review is limited to ecotoxicological implications of PW discharges in a Barents Sea context, it is necessary to also look into several other fields of study; including: PW ecotoxicology studies performed under temperate sea conditions (for comparison to the Arctic conditions); basic oceanology and ecology studies performed in high north seas (for assessing the possible vulnerabilities of different biological resources in certain high sensitive areas, such as the Polar Front, the Ice Edge, and Tromsøflaket); research and knowledge concerning Arctic key species, such as polar cod, and ecologically relevant effect considerations, such as sensitive time frames e.g. during spawning; and research and knowledge that concern possibly negative effects on reproduction of individuals and populations. In this context, it will also be important to attempt to clarify any key knowledge gaps that still exist on these issues.

1.3 Sources of information

The existing research literature base on environmental risk and impact of PW is broad. Peer reviewed items were collected mainly by means of Web of Science (WoS) and Scopus literature search engines whereas Google was used as a search tool for literature searching in the grey literature field. The present review has its primary focus on articles that are published in peer reviewed scientific journals and studies that have been performed by Norwegian groups. Information from grey literature sources (reports, academic theses, books, conference proceedings, regulatory or guideline documents, etc.), is to a limited degree included herein. The present review also highlights studies that have received funding from the Research Council of Norway (RCN). The RCN programs were: Program for Marine Pollution (Program for marin forurensning, PMF, 1992-97), Marine Resources and Environment (Marine ressurser og miljø, MAREMI, 1995-99) and Marine Resource Management (Marin ressursforvaltning, MARRES, 1995-99), PROFO (2000- 2005), PROOF (2002-08), PROOFNY (2008-09, from 2010 continued as a sub-programme under HAVKYST), the Oceans and Coastal Areas programme (HAVKYST, 2006-2015), and the Marine Resources and Environment programme (Marine ressurser og miljø, MARINFORSK, 2016-2025). Also, several of the research programs that primarily were oriented towards petroleum science topics, such as Petromax and Petromax2, have submitted calls that partly have concerned issues relevant to offshore PW discharges. In addition to the research projects funded via the RCN programs noted above, many projects of considerable size have over the years been funded directly by the oil and gas industry, one relevant example being the BioSea Joint Industry project (Pinturier et al., 2008; Buffagni et al., 2010). Information about the research programs and projects is available at the RCN Project Bank website at this website address:

https://www.forskningsradet.no/prosjektbanken/#

Although this research literature summary is limited mostly to Norwegian research studies and to peer-reviewed articles, it includes reference to more than 450 items.

2 Offshore produced water discharges on the Norwegian continental shelf

2.1 Brief about offshore produced water

Since the start in 1971, oil and gas have been produced from a total of 107 fields on the Norwegian Continental Shelf (NCS). At the end of 2017, 85 fields were in production: 66 in the North Sea, 17 in the Norwegian Sea and two in the Barents Sea, according to data from Norwegian Petroleum (https://www.norskpetroleum.no/en/facts/field/). In the Barents Sea, Snøhvit and Goliat fields are in production, whereas the Johan Castberg field now is approved for production by the Norwegian Parliament, with first oil scheduled for 2022. Exploration drilling activity is presently high in the Barents Sea, but a continued growth of offshore petroleum activities in the region is uncertain, partly due to environmental concerns. Offshore Produced Water (PW) discharges are among the key issues in this regard.

PW is the oily water which always is present in the well-stream during offshore petroleum extraction, see various chapters in the PW review edited by Lee and Neff (2011) and Bakke et al. (2011). Together with drilling discharges and accidental oil spills, offshore PW is among the key issues that oil and gas operators on the NCS must handle in a proper manner (Figure 1). PW consists mainly of formation water, injection water, and in the case of gas production, condensed water. PW contains, among other things, dispersed oil, many different dissolved organic components, heavy metals, naturally occurring radioactive isotopes and residues of production chemicals (see section 2.3 for more info about composition of PW). When a well is producing petroleum from a geological formation offshore, there is normally much more water than oil in the well-stream, typically 2-4 times more by volume. The relative water content increases as the well and the production field matures; sometimes reaching as high as 98% before closure of the well.

The hydrocarbon fraction of the well-stream is separated from the water fraction by means of a water treatment system. Advanced oil-water separator systems are often equipped with several treatment steps both for maximising the hydrocarbon yield and minimizing the content of other constituents in the PW before it either is reinjected into the geological formation or discharged to sea. DNV-GL recently reported on the Best Available Techniques (BAT) for PW cleansing in offshore industry applications focussing primarily on content of dispersed oil but also on other potentially hazardous substances (DNV-GL, 2015). Reinjection from an environmental perspective is considered the best treatment/management option for PW, especially if it is focussed on potentially hazardous substances. The following techniques were assessed in the DNV-GL report: hydrocyclones, compact flotation, hydrocyclones in combination with compact flotation, hydrocyclones in combination with CTour PW treatment process, Macro Porous Polymer Extraction (MPPE), and hydrocyclones in combination with Nutshell based filtering. If the right conditions are present, the concentration of dispersed oil in water can be reduced to 5 mg/L (5 ppm). Some fields have a well-stream where oil-water separation is comparatively easy and good cleansing is obtained using simple techniques. Other fields may have far more complex operating conditions, resulting in poorer cleaning effect even with advanced cleaning techniques. In addition, the PW cleaning effect may vary considerably over time due to increased water volume, varying well-flow quality, changed pressure conditions, as well as phase-in of new wells with different types of oil and other chemical additives. Even a cleaned PW stream will typically contain a low level of residual hydrocarbons as well as a variable mixture of naturally occurring substances and/or various chemicals that have been added at some point during production. The quantity of PW varies considerably between fields but according to data from the offshore industry, the total discharges of PW from the oil and gas fields on the Norwegian continental shelf for the period 2003-2016 have varied between approx. 125 million and 160 million standard cubic meters on an annual basis. These discharges have resulted in an annual release of between 1400 and > 1900 tonnes of crude oil to sea (NOROG, 2017). PW is the largest source of operational oil spills from the offshore industry, even though the relative oil content in the discharged PW has been down to 10 ppm on average (NOROG, 2017), which is far lower than the authorities' requirement for <30 ppm. The relative amount of re-injected PW was around 20% of the total PW volume produced during the period 2003-14.

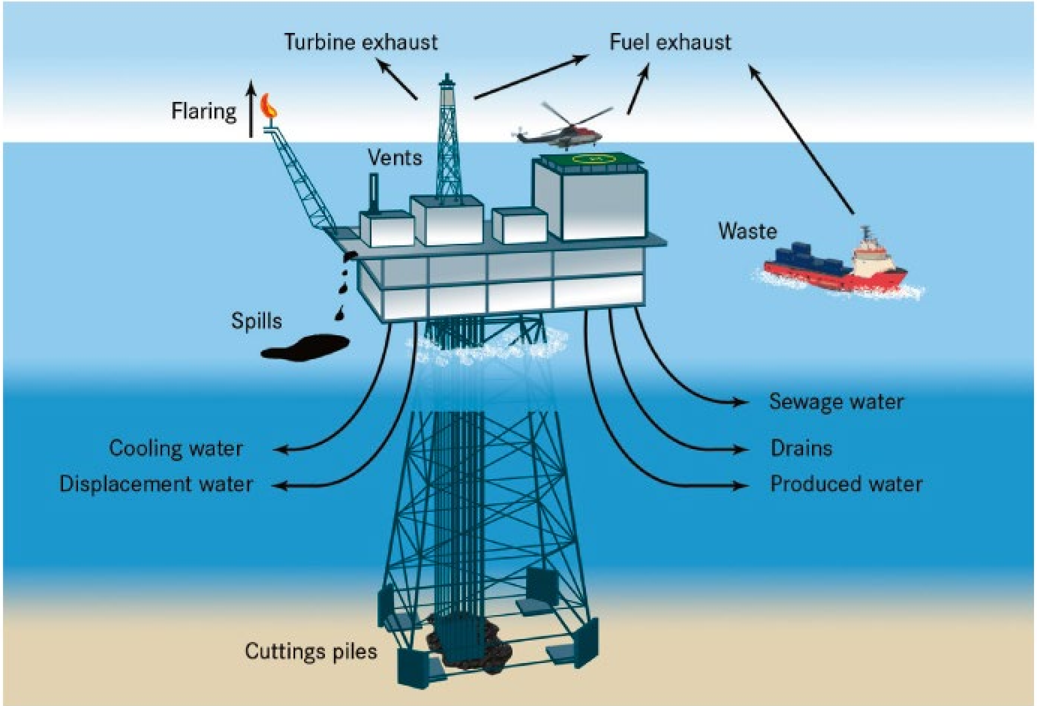

Figure 1: This sketch shows the key sources for chemical contamination of the surrounding offshore environment that are associated with routine operations in offshore oil and gas production. Illustration source: OSPAR: https://qsr2010.ospar.org/en/ch07_01.html.

The first environmental regulations applicable to discharge of PW to sea were initiated in the USA in the 1960s, i.e. operators had to ensure a “no slick/no sheen” discharge. During the 1970-80s, the environmental concern grew related to the increasing discharges of PW to sea (Koons et al., 1977), and this led to considerable efforts by the industry to continuously develop more efficient water treatment systems. In the mid-80s, the introduction of hydrocyclone technology in the produced water treatment led to a new generation of improved water cleaning systems, and to PW discharge routines being judged as ecologically sound based on a combination of predictive and observational studies (Middleditch, 1984). The predictive approach involved determining the composition of the effluents and considering the toxicities of individual components to deduce overall toxicity of the effluent, whereas the observational approach consisted of studies of 'real-world' effects occurring in the sea at offshore sites of operational PW discharges.

However, as our knowledge in environmental science and ecotoxicology further evolved during the 1990s, the overall environmental judgement of offshore PW effluents as benign low-risk was increasingly challenged by new effect studies and new effect-detecting endpoints that addressed novel ecotoxicity endpoints such as endocrine control of embryo development and individual and population reproduction processes. Particularly, in the public discourse in Norway during that period it was questioned and debated whether fish populations living in areas downstream from major offshore production facilities were at risk for being harmed somehow from a more or less continuous but low-concentration exposure to undesirable substances originating from the PW discharges, such as alkylphenols. Particularly there was concern associated to substances which possibly could act as hormonal mimics or disruptors and influence vital biological processes in economically and ecologically important fish populations. At that time there was no research available that could rule out that possible risk. This period of increased concern also coincided with a marked increase in the number of offshore fields that were put into production on the Norwegian shelf in the 90s.

In the British sector of the North Sea, PW discharges were already large, and the British discharges were predicted to stay larger than the Norwegian discharges for many years ahead. Predictions of future large volumes of PW to be discharged, led to demands on the petroleum industry to develop even better PW treatment systems and management routines. This was followed by a mostly policy driven development of a much stricter environmental regulation regime for PW discharges on the Norwegian shelf (see more in paragraph 2.2). Furthermore, a series of national research funding programmes were initiated particularly targeted to studies that could contribute to an improved understanding of key ecotoxicological issues related to offshore PW discharges. Most of these research projects were conducted during the period 1995 – 2010. The petroleum industry has also financed many research and development projects, which often have focussed on the development of environmental risk assessment tools, such as Environmental Impact Factor (EIF), Dose Related Effects Assessment Model (DREAM), or Species Sensitivity Distributions (SSDs), which were suitable for use on PW discharges, drilling discharges and other operative discharges. Important experience and knowledge about PW impacts as well as the suitability of impact detecting tools has come from monitoring activities that are demanded by the authorities (using methods as described in the "Guidelines for environmental monitoring of petroleum activities on the Norwegian continental shelf", current edition M-408).

Improvement of the knowledge and tools for environmental monitoring of offshore PW discharges during the 1995-2010 period has tended to ease the environmental concern for operational PW discharges amongst most stakeholders. However, with the prospects of increasing offshore activities in Arctic seas, the concern for possible ecological risk of PW discharges has returned, partly because of the rich fisheries and other unique ecological resources in these areas, as well as the lack of knowledge on their vulnerability to PW contamination, and because of the harsher environmental conditions in the Arctic areas. These introduce a series of new challenges to the operations as well as to the assessments of environmental risk and environmental impact.

2.2 Produced water regulation in Norway

The Norwegian Government prepared an environmental impact assessment (EIA) for oil development in the Lofoten and Barents Sea region in 2003 (OED, 2003), which was founded on data and information from several underlying assessments, such as the assessment of possible impacts of accidental oil spills on life in the water column in this area (Johansen et al., 2003).

The Zero Discharge Target for the offshore industry was introduced in the Norwegian Parliament Report no. 58 (1996-97) (Miljøpolitikk for en bærekraftig utvikling) stating the aim of "zero discharge" (later "zero harmful discharge") to sea from offshore activities on the NCS before 2005 (Marthinsen and Sørgård, 2002). It is important to note that zero discharge is not a standard or a discharge level, but more a strategy or a philosophy and in line with the precautionary principle. This was a strategy to encourage the operators on the NCS to continue investing in better systems and technology for discharge reductions. For all new fields on the shelf, and especially those situated in the Barents Sea region, the Zero Discharge Target advocated the use of re-injection as the strategy of choice for handling PW. The Zero Discharge Target also paved the ground for the Norwegian government later to apparently move even further, suggesting implementing a “zero physical discharge” policy in new oil and gas fields in the Barents Sea (Norwegian Parliament Report no 38, 2001-2002), although this zero-physical-discharge target in the Barents Sea was in 2011 amended to a general “zero discharge” target. The zero-discharge target is currently the basis for environmental regulation for all fields on the NCS.

To maintain the appropriate focus of the zero discharge work, an advisory cooperation group (the Zero Discharge Group) was established consisting of representatives from the Norwegian Authorities (the Norwegian Pollution Control Authority, the Norwegian Petroleum Directorate), and the Norwegian Oil Industry Association, and with a mandate to implement the zero discharge goal before the end of 2005. The industry's efforts on the zero discharge target is continuously monitored by the Norwegian Environment Agency (Miljødirektoratet, 2016a).

Ever since the term "zero discharges" was introduced in the Norwegian Parliament Report no. 58, the term has been the subject of discussions and interpretations, and a source of confusion. The current understanding of the Zero Discharge Target implies requirement for zero hazardous discharges during normal operation (and not zero-physical discharges). The field operators are required to use best available technology (BAT) to counter pollution of the environment as far as this is technically and economically feasible based on field-specific conditions. Operators on the Norwegian continental shelf increased the discharges of dispersed oil and naturally occurring substances in PW discharges during the period 2010 – 2015. Key reasons for the increase were an increasing number of older production fields and reduced use of re-injection of produced water (Miljødirektoratet, 2016a). The trend has created some concern with the authorities. Stricter requirements for cost reductions seem to render measures to reduce discharges and environmental impacts too expensive. This makes it imperative to ensure that the current environmental standards of the petroleum industry are maintained or improved (Miljødirektoratet, 2016b).

The status of the work towards the Zero Discharge Target for the petroleum activities on the Norwegian continental shelf has been recently described in the report "Work towards zero discharge at sea from petroleum activities offshore” (*Arbeid mot nullutslipp til sjø fra petroleumsvirksomhet offshore*) (Miljødirektoratet, 2016a), and recommendations for future requirements for oil and gas fields in different parts of the shelf have recently been described in the report “The petroleum sector and marine environmental considerations” (*Petroleumssektoren og hensynet til marint miljø*) (Miljødirektoratet, 2016a). One result that follows from the Norwegian zero adverse effect regulation is that it creates a need for the industry to prove that their operation is environmentally safe, i.e. that it has no significant adverse ecological impact. To produce such a no-effect documentation with a 100% certainty is not possible.

Operational PW discharges are one of many petroleum industry activities that are relevant to risk, considered in conjugation with the industries presence in the Arctic. Oil and gas exploration and production in the Arctic dates back to the period around the second world war and exploration activities have since then identified many significant oil or gas resources and geological structures which potentially may contain oil and gas resources located to multiple shelf areas surrounding the Arctic Ocean. Comprehensive overviews of such data are provided in the reports from the Arctic Monitoring and Assessment Programme (AMAP) (AMAP, 1998, 2010a, b, c).

In connection with the Parliamentary Report no 38 (St.meld.nr.38, 2001-2002), the Norwegian authorities proposed an unprecedented strict regulatory policy on future PW discharges (and other discharges from any future offshore petroleum activities) in the Barents Sea (and Lofoten) area, stating a Zero Physical Discharge policy for PW, drilling fluids and cuttings (only with exception of top hole cuttings that were allowed to be deposited on the seabed). This represented a significantly sharpened requirement in comparison to the previous zero-discharge target (i.e., zero-discharge of oil and hazardous chemicals). In 2011, this zero-physical-discharge target in the Barents Sea was amended to the general zero discharge target valid for other parts of the Norwegian continental shelf. The environmental implications of this change were not considered in the scientific basis for the revised Management plan.

2.3 Chemical composition of offshore PW

2.3.1 Naturally occurring substances in PW

Offshore PWs contain a complex mixture of naturally occurring organic and inorganic substances, including suspended particles (e.g., clay), dispersed oil (tiny oil droplets), dissolved organic compounds, dissolved hydrocarbon gases, inorganic salts, heavy metals, and naturally occurring radioactive materials (NORM) (Table 1). According to demands from OSPAR, the content of dispersed oil in PW discharges to sea shall not exceed 30 mg/L (ppm). The chemical composition of PW mixtures varies greatly not only between different offshore fields but also spatially and temporally within the same production field (Røe Utvik, 1999; Neff, 2002; Neff et al., 2011; OSPAR, 2014). Because of the great variability in chemical composition, detailed chemical analysis is required when assessing the environmental risk of a PW discharge (Røe Utvik, 1999).

Data from chemical analyses of PWs over years have yielded a list of substances that are of high relevance to environmental monitoring, and recommended guidelines for sampling and chemical analyses of naturally occurring substances in PWs have been developed (NOROG, 2013). The environmental toxicity of substances (and groups of substances) that are found in PW is normally described by their Predicted No Effect Concentration (PNEC value) (Table 2), i.e. smaller PNEC values indicate more ecotoxic substances. The PNECs of substances are decided by means of standardized toxicity testing. These standardized toxicity tests are generally performed with temperate test organisms. Therefore, there has been uncertainty regarding how relevant these PNECs are with regard to Arctic organisms. A further discussion of PNECs and environmental risk assessment of PW constituents is provided in section 4.

The paper of Røe Utvik (1999) is clearly the most influential Norwegian paper regarding chemical composition data of PW discharges on the NCS. The study was based on chemical characterisation of PW samples from four offshore oil production platforms in the North Sea (Oseberg Feltsenter, Oseberg C, Brage and Troll B). More recently, many other reports containing such data have emerged from Norwegian research groups, see section 2.4. For example, McFarlin et al. (2018) recently reported average composition data of naturally occurring substances in PWs obtained from 11 different fields on the Norwegian Continental Shelf. Internationally, hundreds of reports are available containing chemical composition data of PW discharges, a broad summary of which is provided by Lee and Neff (2011).

The natural substances in PW that are most ecotoxicologically relevant depend on the amount released and combination of their PBT (persistence, bioaccumulation, and toxicity) properties. Most of the attention in the research has been focussed on dispersed oil droplets, PAHs and alkylphenols, whereas in more recent studies also naphthenic acids and certain natural radioactive materials have in an increasing manner been investigated as possible ecotoxicants. Physicochemical and biological degradation of substances released in PW further complicates the question of which substances in PW discharges are the most relevant to study and to risk assess. Knowledge on this issue is important for predicting and estimating the exposure concentrations (PEC) around PW effluents. There are several recent articles and reports that include data of PW constituent degradation include, e.g. (Lofthus et al., 2016; McFarlin et al., 2018; Lofthus et al., 2018a; Brakstad et al., 2018). Presently, there is a shortage of studies on degradation processes in Arctic seawater.

Brakstad et al. performed several field studies to investigate the biodegradation rate of dispersed oil in Arctic seawater (Brakstad et al., 2008; Brakstad et al., 2018). In the latter study compared processes in seawater at Western Greenland with temperate seawater (from a Norwegian fjord) at temperature conditions of 4-5 °C. They observed a slower oil biodegradation in the Greenland seawater, especially for saturates (linear, branched and cyclic alkanes), and suggested the difference was possibly caused by lower macronutrient concentrations (both N- and P-compounds) in the Arctic samples. They also pointed at the relevance of obtaining experimental data directly from the relevant Arctic environment, rather than from temperate seawater environments adjusted to Arctic conditions, when making predictions on oil degradation in Arctic seawater.

Table 1: Offshore produced water discharges contain many natural occurring substances that are potentially harmful to sea organisms. This overview shows major groups of these substances and by which substances they typically are analyzed (OSPAR, 2014), and the total amounts released on the NCS in 2015 (source, Norwegian Oil and Gas).

Substance group	Measured by which substances	Total discharges at NCS in 2015
Produced water		148 million Sm^3
Dispersed oils	C7-C40 aliphatic hydrocarbons	1819 tons
Monoaromatic hydrocarbons (BTEX)	benzene, toluene, ethylbenzene, and xylene	2266 tons
Polycyclic Aromatic Hydrocarbons (PAHs)	The 16 US-EPA PAHs: naphthalene, acenaphthene, acenaphthylene, fluorene, anthracene, phenanthrene, fluoranthene, pyrene, benz(a)anthracene, chrysene, dibenzo(a,h)anthracene, benzo(g,h,i)perylene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, indeno(1,2,3,-cd)pyrene	131 tons
Other PAHs	C1-naphthalenes, C2-naphthalenes, C3-naphthalenes, C1-phenanthrenes, C2-phenanthrenes, C3-phenanthrenes, dibenzothiophene, C1-dibenzothiophenes, C2-dibenzothiophenes, C3-dibenzothiophenes	
Phenol/alkylphenols	phenol, C1-alkylphenols, C2-alkylphenols, C3-alkylphenols, C4-alkylphenols, C5-alkylphenols, C6-alkylphenols, C7-alkylphenols, C8-alkylphenols and C9-alkylphenols	634 tons
Organic acids	formic acid, acetic acid, propionic acid, butyric acid, valeric acid, isobutyric acid and isovaleric acid and naphthenic acids	
Metals	arsenic, lead, cadmium, copper, chromium, mercury, nickel, zinc, iron, and barium	As: 746, Pb: 84, Cd: 5, Cu 128, Cr 99, Hg 9, Ni 1210, Zn: 1523 kg
Radioactive elements	Ra^{226}, Ra^{228}, Pb^{210}, in certain cases, also Th^{228}	

Table 2: Expected concentrations of potentially harmful naturally occurring substances in offshore produced water discharge relating to a 15 mg/L oil in water level (source: Equinor and (Dahl-Hansen et al., 2017)) and PNEC values for the same substances according to the OSPAR Commission Agreement 2014-05 (OSPAR, 2014).

		Expected concentration relating to 15 mg/L oil in water	PNEC µg/L
	Dispersed oil	15	70.5
BTEX	Benzene	8.40045	8
	Toluene	5.08233	7.4
	Ethylbenzene	0.31611	10
	Xylene	0	8
2-3 ring PAHs	Naphthalene	0.92623	2
	Acenaphthene	0.00317	0.38
	Acenaphthylene	0.00111	0.13
	Fluorene	0.01227	0.25
	Anthracene	0.03381	0.1
	Phenanthrene w. substitutes	0.08422	1.3
	Dibenzothiophene w. substitutes	0	0.1
4 ring PAHs	Fluoranthene	0.00034	0.01
	Pyrene	0.00055	0.023
	Benz(a)anthracene	0.00018	0.0012
	Chrysene	0.00099	0.007
5-6 ring PAHs	Dibenzo(a-h)anthracene	0.00001	0.00014
	Benzo(g-h-i)perylene	0.00004	0.00082
	Benzo(a)pyrene	0.0001	0.022
	Benzo(k)fluoranthene	0.0001	0.017
	Indeno(1-2-3-cd)pyrene	0.00001	0.00027
	Benzo(b)fluoranthene	0.00009	0.017
Phenols	Phenol (C0-C3-alkyl-phenol)	6.03395	7.7
	Butylphenol (C4-alkyl-phenols)	0.0616	0.64
	Pentylphenol (C5-alkyl-phenols)	0.02359	0.2
	Octylphenols (C6-C8-alkyl-penols)	0.00117	0.01
	Nonylphenol (C9-alkyl-phenols)	0.00006	0.3
Heavy metals	Arsenic	0.000068	0.6
	Cadmium	0.000013	0.21
	Chromium	0.000438	0.6
	Copper	0.001048	2.6
	Nickel	0.000762	8.6
	Mercury	0.000003	0.048
	Lead	0.000082	1.3
	Zink	0.003583	3.4

2.3.2 Added chemicals in PW (offshore chemicals)

In addition to natural chemical substances, PW may contain chemical substances that have been deliberately added to the process due to technical-operative needs, these are the so-called “oilfield chemicals” or “offshore chemicals” (Vik et al., 1992; Beyer et al., 2001; Dahl-Hansen et al., 2017). Most of these are drilling chemicals, whereas others are added to the production process, these are called production chemicals. Common production chemicals that presently are used on the NCS include: scale inhibitors, emulsion breakers, wax inhibitors, foam inhibitors, flocculants, and biocides.

Because many offshore chemicals are unfriendly to the environment their use on the NCS is strictly regulated based on their PBT (persistence, bioaccumulation, toxicity) properties. Data about the ecotoxicological properties for offshore chemicals can be obtained from various organizations and databases, such as: The European Chemicals Agency (ECHA) (https://echa.europa.eu/home) and NEMS Chemicals® (https://nems.no/services/nems-chemicals/), the latter which is an online chemical management software designed to handle eco-toxicological data in the HOCNF (Harmonized Offshore Chemical Notification, OSPAR Recommendation 2010/3) Format.

The Oslo – Paris convention for the Protection of the Marine Environment of the North-East Atlantic (the 'OSPAR Convention') came into force in 1998. The convention contains among other issues regulations on use of chemicals. The OSPAR commissions developed an international environmental testing and regulatory regime for offshore chemicals, the so-called HOCNF, to stimulate the offshore petroleum industry to replace environmentally hazardous offshore chemicals with less hazardous alternatives. The HOCNF regulation demands that certain data regarding ecotoxicological properties must be available for each substance, including data on:

- Bioaccumulation/bioconcentration potential
- Biodegradability (persistence)
- Aquatic toxicity

In Norway, offshore chemicals are classified in black, red, yellow, and green chemicals based on the HOCNF data, with black chemicals being most environmentally harmful. Chemicals in the black category are not readily biodegradable, show a high potential for bioaccumulation and have a high acute toxicity. In principle, use and discharge of these chemicals is not permitted unless deemed necessary based on safety- and technical reasons, or it has been documented in special cases that application of these will result in the lowest risk for environmental harm. Chemicals in the red category are slowly biodegraded in the marine environment, show potential for bioaccumulation and/or are acutely toxic. Organic chemicals are classified as red when the biodegradation measured as BOD28 (biological oxygen demand after 28 days) is ≤ 20%, or if the chemicals fulfil two of the following three criteria: biodegradation measured as BOD < 60%, log Pow ≥3, and acute toxicity EC50 or LC50 ≤ 10 mg/L. Chemicals in the red category can be harmful to the environment and shall be prioritized for substitution with less harmful alternatives.

The Norwegian legislation and regulation on offshore chemicals expands beyond the HOCNF demands, and details about this stricter regulation is provided by the Norwegian "Activities Regulation" http://www.ptil.no/activities/category399.html#_Toc503938340. Briefly, the Activities Regulation requires that operators are responsible for the environmental evaluation/ranking of the offshore chemicals that they are using, and for choosing the chemicals that give the lowest risk of environmental harm.

A model (CHARM - chemical hazard assessment and risk management) was developed to give operators, chemical suppliers, and environmental authorities a scientific framework for analysing the environmental hazard and risk of offshore chemicals used and discharged to the marine environment (Vik et al., 1998). Weideborg et al. (1997) compared the results of testing the toxicity of a total of 82 offshore chemicals by using different screening toxicity tests, demonstrating good correlations. Sverdrup et al. (2002) conducted a related method comparison study, testing the relative sensitivity of one freshwater and two marine acute toxicity tests to determine the toxicity of 30 offshore chemicals. NIVA recently conducted a review of the available data on biodegradation properties of 21 selected offshore chemicals and groups based on a weight of evidence approach (Wennberg et al., 2017). For most of the assessed compounds, it was not possible to draw a clear conclusion about the biodegradability. Only two of the investigated compounds, benzotriazole and N-methyldiethanolamine, were assessed to be very likely and likely to have a biodegradability of less than 20% in seawater.

2.4 Research on the effects of PW in marine organisms

The research field of biological effects of environmental chemicals is generally known as ecotoxicity biomarker research (Van der Oost et al., 2003). The image of an ideal biomarker assay is an effect parameter that both is: highly sensitive, specific against the stressors investigated, relevant for the ecological fitness of the exposed sentinel organism and population, rapid and uncomplicated to assess, quantitative in relation to the stressor, and possible to quality assure/control in a straightforward manner (e.g., standard or reference material available). Questions related to biomarkers include; what are the most relevant biological impacts to consider/investigate in marine ecosystems exposed to PW contaminants, and are there significant differences between cold-water and temperate-water ecosystems on these issues? Olsen et al. (2013b) evaluated the availability of ecotoxicity data of oil and PW relevant compounds for a selection of cold-water marine species of fish and plankton associated with the Barents Sea ecosystem. They concluded that the amount of data was limited. There was particularly a need for new experimental studies for zooplankton focusing on endpoints that regarded biological development and toxicant bioaccumulation and for larvae and juvenile fish focusing on growth and development.

Many research projects performed in Norway during the last three decades have addressed environmental issues of PW (see the overviews provided in Table 3 and Table 4) and have produced a considerable number of scientific papers and reports extending and improving our knowledge on several environmental aspects of PW discharges. A range of effect issues of PW and petroleum associated contaminations in fish and other organisms have been investigated including endocrine and reproductive effects, oxidative stress, cytotoxicity, larval mortality, induction of detoxification enzymes, change in gene expression, lysosome membrane stability, and hepatic lipid composition, as well as novel assessment approaches based on *in vitro* methods, various “Omics” based analytical principles, and probabilistic environmental risk assessment modelling approaches. Bakke et al. (2013) compiled an overview of the results of these studies, and an update of that summary is provided in the present report. Some of the studies show that PW discharges can contain natural substances and/or added chemicals (offshore chemicals) that have so-called PBT (persistence, bioaccumulative and toxic) properties, and which can, at least in a theoretical worst case scenario, induce adverse effects in fish or other marine organisms in the recipient waters. But, importantly, because of the dispersal and dilution of the PW plume in the recipient sea, any notable effects will most likely be restricted to the first couple of kilometres from the discharge point, and evidence of PW causing any ecologically significant effects (reproduction, life-expectancy, etc.) in pelagic organisms and populations, has not yet been demonstrated in field studies or in laboratory studies when using environmentally realistic exposure regimes. In this connection, it is relevant to mention the offshore field study of several authors (Reed et al., 2001; Reed M and (2002), 2002; Jørgensen et al., 2002; Løkkeborg et al., 2002; Wells, 2005; Forbes et al., 2006; Neff et al., 2006; Durell et al., 2006; Harman et al., 2009c; Hylland et al., 2008; Brooks et al., 2009; Harman et al., 2009b; Sundt et al., 2011b; Harman et al., 2011; Brooks et al., 2011b; Bakke et al., 2011; Balk et al., 2011; Grøsvik et al., 2012) which found significantly increased DNA adducts in livers of haddock in the Statfjord area of the North Sea, a region in which much offshore petroleum activities have been conducted for decades. Although, that effect was more likely to be linked to the presence of contaminated sediments in the study area caused by earlier extensive field disposals of old OBM (oil-based-mud) drill cuttings in the Statfjord area, and less likely causally associated with the large PW discharges that also have been taken place at this offshore field.

Table 3: An overview of research papers relevant to ecotoxicology of offshore PW discharges sorted by study issues used in Bakke et al. (2013). The right column lists studies that were not covered by Bakke et al. (2013).

PW issue	Research studies that were referred to in Bakke et al. (2013) sorted by PW issue	Research studies not examined by Bakke et al. (2013)
Chemical composition of offshore PW discharges and PW mixes in seawater	(Soto et al., 1991; Priatna et al., 1994; Nimrod and Benson, 1996; Terrens and Tait, 1996; Røe Utvik, 1999; Røe Utvik et al., 1999; Arukwe et al., 2000; Arukwe et al., 2001; Neff, 2002; Frost et al., 2002; Johnsen et al., 2004; Boitsov et al., 2004; Lee et al., 2005; Durell et al., 2006; Boitsov et al., 2007; Meier et al., 2007b; Thomas et al., 2009; AMAP, 2010c; Neff et al., 2011; OLF, 2011)	(Jacobs et al., 1992; Neff et al., 1992; Rye et al., 1998; Sanni et al., 1998; Vik et al., 1998; Røe Utvik and Hasle, 2002; Brakstad et al., 2004; Faksness et al., 2004; Brakstad and Bonaunet, 2006; Melbye et al., 2009; Balaam et al., 2009; AMAP, 2010b, a; 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)
Field studies of or relevant to PW effluents	(Reed et al., 2001; Reed M and (2002), 2002; Jørgensen et al., 2002; Løkkeborg et al., 2002; Wells, 2005; Forbes et al., 2006; Neff et al., 2006; Durell et al., 2006; Harman et al., 2009c; Hylland et al., 2008; Brooks et al., 2009; Harman et al., 2009b; Sundt et al., 2011b; Harman et al., 2011; Brooks et al., 2011b; Bakke et al., 2011; Balk et al., 2011; Grøsvik et al., 2012)	(Johnsen et al., 1998; Røe Utvik and Johnsen, 1999; Grøsvik et al., 2007; Brooks et al., 2011a; Smit et al., 2011; Sundt et al., 2012; Brooks et al., 2012; Harman et al., 2014; Hale et al., 2016)
Determination of PW contaminant levels	(Krahn et al., 1986; Brendehaug et al., 1992; McDonald et al., 1995; Neff and Burns, 1996; Tollefsen et al., 1998; Røe, 1998; Røe Utvik, 1999; Aas et al., 2000b; Arukwe et al., 2000; 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; Harman et al., 2008a; Harman et al., 2008b; Brooks et al., 2009; Sundt et al., 2009a; Sundt et al., 2009b; Grung et al., 2009a; Harman et al., 2009c; Skadsheim et al., 2009; AMAP, 2010c; Meier et al., 2010; Beyer et al., 2010; Beyer et al., 2011; Sundt et al., 2011a; Jonsson and Björkblom, 2011; Sundt and Björkblom, 2011)	(Johnsen et al., 1998; Baussant et al., 2001; Jonsson et al., 2008a; Jonsson et al., 2008b; AMAP, 2010b, a; Jonsson et al., 2012; Broch et al., 2013; Harman et al., 2014; Hale et al., 2016; Hale et al., 2019)
Endocrine and reproduction effects of PW	(Jobling and Sumpter, 1993; White et al., 1994; Gimeno et al., 1998; Miles-Richardson et al., 1999; Weber et al., 2002; Tanaka and Grizzle, 2002; Weber et al., 2003; Thomas et al., 2004a; Tollefsen et al., 2006; Meier et al., 2007b; Tollefsen et al., 2007; Tollefsen and Nilsen, 2008; Brooks et al., 2009; Thomas et al., 2009; Meier et al., 2010; Holth et al., 2010; Tollefsen et al., 2011; Meier et al., 2011; Sundt and Björkblom, 2011)	(Thomas et al., 2004b; Mjos et al., 2006; Boitsov et al., 2007; Meier et al., 2008; Lie et al., 2009; Petersen and Tollefsen, 2011; Beyer et al., 2012; Knag et al., 2013b; Knag and Taugbol, 2013; Knag et al., 2013a; Petersen et al., 2013; Geraudie et al., 2014; Hultman et al., 2015; Sanni et al., 2017a; Petersen et al., 2017c; Petersen et al., 2017a; Hultman et al., 2017)
Non-endocrine effects of PW	(Dey et al., 1983; Schultz et al., 1986; Widdows et al., 1987; Lowe and Pipe, 1987; Obata and Kubota, 2000; Okai et al., 2000; Hasselberg et al., 2004a; Hylland et al., 2006; Meier et al., 2007a; Hylland et al., 2008; Sundt et al., 2008a; Tollefsen and Nilsen, 2008; Abrahamson et al., 2008; Brooks et al., 2009; Holth et al., 2009; Grøsvik et al., 2010; Farmen et al., 2010; Holth et al., 2010; Holth et al., 2011a; Sundt and Björkblom, 2011; Sundt et al., 2011a; Jonsson and Björkblom, 2011; Balk et al., 2011; Grøsvik et al., 2012)	(Strømgren et al., 1995; Stephens et al., 2000; Hasselberg et al., 2004b; Hurst et al., 2005; Olsvik et al., 2007; Holth et al., 2008; Holth et al., 2009; Lie et al., 2009; Hannam et al., 2009; Jonsson et al., 2010; Holth et al., 2011a; Holth et al., 2011b; Olsvik et al., 2011a; Olsvik et al., 2011b; Olsvik et al., 2011c; Holth and Tollefsen, 2012; Sundt et al., 2012; Tollefsen et al., 2012; Knag and Taugbol, 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; Petersen et al., 2017b; Petersen et al., 2017c; Petersen et al., 2017a; Hale et al., 2019)
Accumulation and effects of oil hydrocarbons and PAHs	(Lowe and Pipe, 1987; Myers et al., 1991; Aas et al., 2000a; Incardona et al., 2004; Taban et al., 2004; Sturve et al., 2006; Laffon et al., 2006; Thomas et al., 2007; Carls et al., 2008; Baussant et al., 2009; Baussant et al., 2011)	(Stephens et al., 2000; Holth et al., 2009; Sørhus et al., 2015; Sørhus et al., 2016b; Sanni et al., 2017a; Sørensen et al., 2017; Sørhus et al., 2017; Krause et al., 2017; Toxværd et al., 2018)

Use of "omics" approaches	(Hansen et al., 2007; Hansen et al., 2008a; Hansen et al., 2008b; Bohne-Kjersem et al., 2009; Bohne-Kjersem et al., 2010; Hansen et al., 2010; Hansen et al., 2011; Karlsen et al., 2011; Nilsen et al., 2011a)	(Bjørnstad et al., 2006; Grøsvik et al., 2006; Olsvik et al., 2007; Mæland et al., 2008; Kjersem et al., 2008; Nilsen et al., 2011b; Nilsen et al., 2011c; Olsvik et al., 2012b; Hansen et al., 2013b; Sørhus et al., 2016a; Song et al., 2018; Tørresen et al., 2018)

Table 4: Other research papers produced by Norwegian groups and relevant to ecotoxicology assessment of offshore PW discharges by study issues other than those reviewed in Bakke et al. (2013).

PW issue	Published research studies not examined by Bakke et al. (2013)
Effects of oil and PW associated contaminants in marine crustacean plankton	(Hansen et al., 2012; Hansen et al., 2013a; Hansen et al., 2013b; Jager and Hansen, 2013; Olsen et al., 2013a; Hansen et al., 2014; Jager and Ravagnan, 2015; Jager et al., 2015; Hansen et al., 2015; Nepstad et al., 2015; Nordtug et al., 2015; Jager et al., 2016; Jager and Ravagnan, 2016; Jager et al., 2017; Farkas et al., 2017; Hansen et al., 2017a; Hansen et al., 2017b; Tollefsen et al., 2017; Krause et al., 2017; Toxværd et al., 2018)
Effects of oil and PW associated contaminants in krill and shrimps	(Bechmann et al., 2010; Arnberg et al., 2017; Moodley et al., 2018)
Sensitivity drivers for oil contamination effect in marine fish	(Sørensen et al., 2014; Duan et al., 2015; Sørensen et al., 2015; Vikebo et al., 2015; Sørensen et al., 2016a; Sørensen et al., 2016b; Sørhus et al., 2016b; Sørhus et al., 2017; Sørensen et al., 2017; Nepstad et al., 2017; Hansen et al., 2018b; Sørensen et al., 2019; Tørresen et al., 2018; Torvanger et al., 2018; Jawad et al., 2018)
Weathering of marine oil spills and ecosystem sensitivity to petroleum pollution under Arctic conditions	(Faksness and Brandvik, 2008a, b; Faksness et al., 2008; Brandvik and Faksness, 2009; Sikorski and Pavlova, 2018)
Monitoring PW using passive sampling devices and *in vitro* bioassay techniques	(Harman et al., 2009c; Harman et al., 2009a; Harman et al., 2010; Harman et al., 2011)
Radioactivity in PW discharges - concentrations, bioavailability and doses to marine biota	(Grung et al., 2009b; Olsvik et al., 2012a)
Assessment of mixture toxicity of compounds in PW discharges	(Song et al., 2012; Petersen and Tollefsen, 2011, 2012; Tollefsen et al., 2012; Petersen et al., 2013; Song et al., 2016; Song et al., 2014a; Song et al., 2014b; Petersen et al., 2014; Song et al., 2018)
Benthic indicators for pollution monitoring and ecosystem monitoring in the Barents Sea, accentuating sponge and corals	(Andrade and Renaud, 2011; Jorgensen et al., 2011; Olsen et al., 2011; Wlodarska-Kowalczuk et al., 2012; Nahrgang et al., 2013; Kutti et al., 2013; Tjensvoll et al., 2013; Larsson et al., 2014; Kutti et al., 2015; Edge et al., 2016; Zetsche et al., 2016; Dauvin et al., 2016; Baussant et al., 2017; Luter et al., 2017; Strand et al., 2017; Leys et al., 2018; Baussant et al., 2018)
Integration of PW biomonitoring with environmental risk assessment	(Radovic et al., 2012; Rial et al., 2013; Jager and Ravagnan, 2015, 2016; Arnberg et al., 2017; Langangen et al., 2017a; Sanni et al., 2017a; Sanni et al., 2017c; Sanni et al., 2017b)
Impact analysis and decision support tools for oil industry pollution management in Lofoten/Barents Sea ecoregion	(Sørensen et al., 2014; Nepstad et al., 2015; Stordal et al., 2015a; Stordal et al., 2015b; Alver et al., 2016; de Hoop et al., 2016; Carroll et al., 2018; Lofthus et al., 2018b; Christie et al., 2019)

There are two distinct strategies for discerning possible ecological effects of offshore PW effluents: predictive and observational. A *predictive* strategy typically involves first to determine (by predictions or by analysis of real PW samples) the chemical composition of the PW effluent, and to subsequently test the toxicity of these substances (alone and in different combinations) with using (preferably standardised) toxicity tests. Then, by combining this information with information (predictions or measurements) on the behaviour of the PW mixture in the recipient water, the risk for toxic impact on organisms downstream the discharge can be assessed. An *observational* approach typically involves ecotoxicity exposure and/or effect studies carried out in the field, but also controlled exposure/effect studies performed in the laboratory with the use of real PW samples. The latter strategy has great potential in PW ecotoxicological research, although it is generally hampered by PW being a highly variable and unstable mixture. By combining predictive and observational approaches, one may compare observed test data and field data with environmental safety standards and conclude with reasonable confidence whether a given strategy for environmental management of the PW discharge is ecologically sound.

In the last 20 years, much attention has been focused on whether alkylphenols and other hormone disrupting substances in PW, may interfere with the reproductive capacity of downstream fish stocks. Effect-controlled analysis of PW show that the mixture contains compounds that can affect estrogen and androgen control processes. These are hormonal processes that are crucial to sexual development and maturation in fish and other organisms. Certain alkyl phenols (especially the C8- and C9-alkylated phenols) are known to have estrogen-like modes of action (estrogen receptor (ER) agonists or androgen receptor (AR) antagonists). The estrogenic potency varies widely between different alkyl phenol types. The types that give the strongest estrogenic effect are generally very low in produced water. Other common substances in produced water, such as naphthenic acids, may also be considered to cause hormonal effects in fish (Thomas et al., 2009; Knag et al., 2013a; Petersen et al., 2017b). The possibility of hormonal disturbances in fish stocks downstream of oil fields triggered considerable concern and this led to a significant number of research projects and publications on this topic from Norwegian research groups, mostly representing lab-based studies (see Table 3 and Table 4). The performed field studies on this issue have been those on the occurrence of alkylphenols downstream of platforms monitored by means of passive sampler devices. In general, the studies show that several reproductive-relevant changes occur when fish are exposed to PW over time and to exposure concentrations that are higher than what would be the expected exposure levels in the recipient, apart from in the first part of the PW dilution zone. Based on the most sensitive responses in cod, a LOEC (lowest observable effect concentration) level of PW relevant alkylphenols in seawater was estimated at approximately 8 ng/L (Meier et al., 2011). However, spatial fish stock distribution data for the North Sea suggest that only negligible parts of the fish stock will be exposed to PW-alkylphenols above LOEC, even when worst-case scenarios for alkylphenol content, PW discharge dispersal patterns, and LOEC considerations are used (Beyer et al., 2012). Effect studies on Endocrine Disruptive Compounds (EDCs) have sometimes challenged traditional concepts in ecotoxicology, particularly regarding the dogma of "the dose makes the poison," as EDCs can have effects at low doses that are not predicted by extrapolation of effects observed at higher doses (Vandenberg et al., 2012). The possible EDC impacts to the reproductive health of Atlantic cod (*G. morhua*) and other key marine fish species has had high priority in Norwegian PW effect investigations. PW constituents that have received much attention include both natural substances (polyaromatic hydrocarbons, alkylphenols and naphthenic acids) and added chemicals. A thorough examination and discussion of PW EDC effect data from studies done before 2012 is provided in Bakke et al. (2013), and this information is also largely included herein. The EDC effect studies that are published later than 2012 fall generally in line with the previous studies confirming that substances that often are present in PW have a potential to exert endocrine effects (xenoestrogenicity and reproduction-relevant EDC effects) in fish, but these effects are induced only when the PW exposure level and duration are considerably higher than what fish realistically may encounter in areas downstream of PW discharges. It is therefore unlikely that a PW discharge can elicit notable EDC effects in wild fish populations, unless the exposed fish population has a fixed and unfavourable localisation, for example being a “reef population” standing close to the platforms. Offshore installations are known to attract various fish species as artificial reefs providing both shelter and increased food supply. These factors may often lead to increased and quite stable local fish populations around each platform (Jørgensen et al., 2002; Løkkeborg et al., 2002). Interestingly, artificial reef fish populations may therefore represent a natural “worst-case” exposure scenario. This may be utilised to test the hypothesis that PW discharges do not cause notable effects even in wild fish staying close to PW discharges.

The complex composition of PW renders many individual constituents unidentified and unquantified by conventional analytical techniques. They may not be extracted from the PW sample or end up in a "hump" in the chromatograms called the Unresolved Complex Mixture (UCM). The UCM may sometimes contain the dominating set of organic substances in the sample and may contain both nonpolar and polar compounds. Theoretically, some of the UCM substances may have ecotoxic properties and some studies have therefore suggested that UCMs in industrial discharges could represent an unknown but possibly significant ecotoxicity risk to aquatic organisms (e.g. (Melbye et al., 2009; Petersen et al., 2017b). The ecotoxicity of the organic compounds in UCMs will normally be assessed by the use of *in vitro* toxicity screening approaches, such as Toxicity Identification Evaluation (TIE) &/or Toxicity Reduction Evaluation (TRE) methodologies (Tietge et al., 1997; Sauer et al., 1997; Thomas et al., 2004b; Elias-Samlalsingh and Agard, 2004).

There is a concern that current methods for marine biomonitoring of offshore PW discharges are not sensitive enough to reveal subtle effects caused by a low-level exposure to contaminants originating from PW. Oil droplets, PAHs, alkylphenols, and naphthenic acids are identified as natural constituents of concern in the PW discharge mixture. In fish, the markers that many see as the most sensitive tools for discriminating between non-exposed and low-level-exposed organisms are biliary metabolites of polycyclic aromatic hydrocarbons (PAH) and alkylphenols (APs) (Jonsson et al., 2008a; Jonsson et al., 2008b; Beyer et al., 2010; Jonsson et al., 2012). Such metabolites have in several offshore surveys been demonstrated as highly sensitive markers for assessing ongoing or recent PW exposure at low concentrations. While most PW responsive markers typically will provide an effect signal in sentinel species exposed within 0.5-2 km downstream of the PW discharge, the metabolites of petrogenic PAHs and alkyl phenols in fish bile have in some surveys been demonstrated to yield a weak but statistically significant exposure signal up to 10 km from PW discharge outlets (see e.g. Hylland et al. (2008). The detection of pollutant metabolites of petrogenic substances in fish bile can provide very sensitive exposure indicators for assessing whether a PW discharge has led to an increased uptake in the sampled fish population of the parent compounds which are metabolized by the fish detoxification system. These data indicate the degree by which the studied fish specimens have recently been exposed to substances originating from a PW discharge. Hence, this type of information is very valuable for the investigators in monitoring and effect studies of PW, especially in connection with field studies. However, such exposure signals are not automatically indicating that an *adverse* effect has manifested in the exposed organism (or population). Indeed, to extrapolate from an observed short-term *exposure above background* signal to a longer-term *adverse-effect phenomenon* in the organism, and especially at population and community levels, is most often a challenge in ecotoxicological field studies as only few biomarker endpoints look beyond the compensatory capacity of the affected organisms (Forbes et al., 2006).

One effect phenomenon in oil exposed fish that has received much attention in recent years is the development of irreversible cardiac defects and impaired cardiorespiratory function in fish fry after exposure to very low levels of PAH contaminants originating from oil contamination (Incardona et al., 2004; Carls et al., 2008; Dussauze et al., 2014). Controlled laboratory experiments show that the induced cardiac defects impact the individual fitness of fish fry in several critical ways, including reduced swimming performance, prey capture, and prey avoidance, with repercussions for survival and possibly for population recruitment (Incardona et al., 2015). Oil induced cardiotoxicity of developing eggs and larvae of haddock (*M. aeglefinus*) was recently studied by The Institute of Marine Research (IMR) in Bergen (Sørhus et al., 2015; Sørhus et al., 2016b; Sørensen et al., 2017; Hansen et al., 2018b). Haddock are believed to be particularly sensitive to oil during early life stages because the egg/embryo surface (chorion) is very sticky and adsorbs dispersed oil droplets (Figure 2, Figure 3). This probably leads to a stronger and prolonged interaction between oil and embryo in comparison to species without a sticky chorion, as for example Atlantic cod. Increased amount of oil droplets on the chorion may lower the exposure time sufficiently to cause toxicity, e.g. in a diluting PW plume.

Research by IMR suggests that even a short exposure to a high concentration of dispersed oil may continue to affect the haddock embryos even after they have been transferred into non-contaminated water by the carry-over of attached oil droplets as a continued source of exposure (Sørhus et al., 2015). Whether these differences between haddock and Atlantic cod can make the haddock more sensitive/vulnerable to oil contamination, and to develop adverse cardiac conditions after oil exposure, remains to be seen.

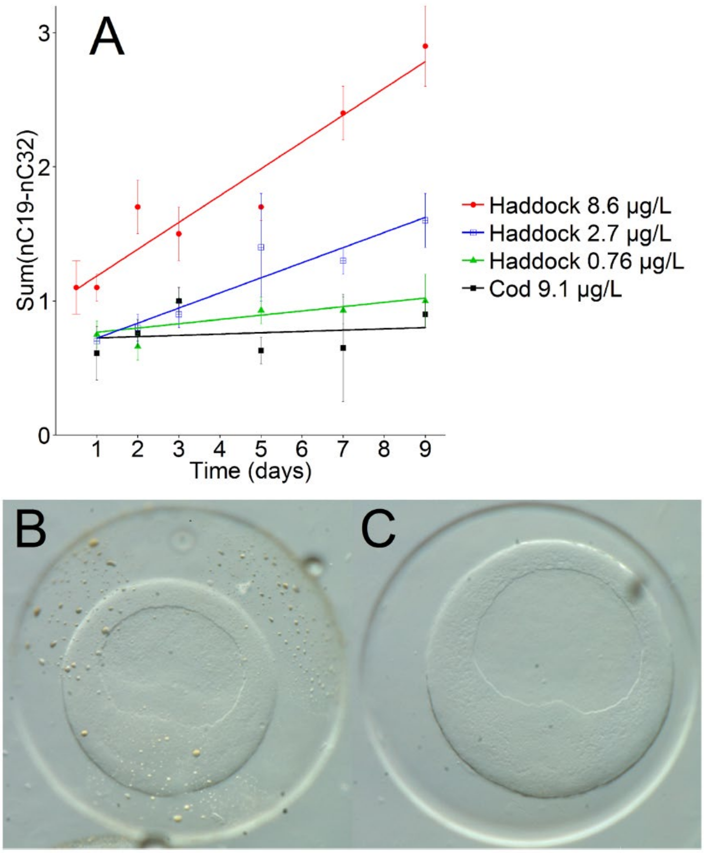

Figure 2: Haddock (B) and cod (C) embryos exposed to 600 µg/L crude oil dispersions for 12 hours, where fouling of oil droplets on the chorion of the haddock egg can be observed. In panel A, the relative response of alkanes (Σ(nC19-nC32)) normalized to the response of internal standard pyrene-d12, during the uptake period for three doses of crude oil exposed haddock and the highest exposure dose for cod. Error bars represent one standard deviation (n = 3). A linear trendline is fitted to each group (R2 = 0.926, 0.921, 0.530 and 0.023 for haddock 8.6, 2.7, 0.76 and cod 9.1, respectively). Source: (Sørensen et al., 2017).

Figure 3: Abnormalities of haddock larvae resulting from embryonic oil exposure. The shown larvae are one day post hatch, after 7 days of exposure. (A) Control. (B) Oil exposed. Abnormalities: Pericardial edema (P), yolk sac edema (Y), spinal curvature (S), craniofacial deformities (CD), jaw deformities (JD). Source: (Sørhus et al., 2015).

It is unknown whether PAH cardiac toxicity can be induced in fish larvae drifting through an offshore production field during active discharge of PW. In a recently published study Hansen et al. (2018a) conducted cardiac toxicity experiments with developing cod (*G. morhua*) embryos exposed to a microbially degraded petrogenic PAH mixture under cold water conditions (Low-energy water accommodated fraction of a weathered crude oil prepared with nutrient amended seawater at 5 °C, kept in the dark, and sampled at 0, 10, 14, and 21 days) during a critical period of their heart development. Survival, hatching, morphometric aberrations, and cardiac function were studied in the fish embryos. Unexpectedly, significant effects were found in the embryos also after the PAH mixture had been subjected to a 21-day biodegradation treatment. The authors suggest that the reason for this may be one or a combination of two 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.

In the North Sea exposure to PW chemicals in caged fish has been detected at distances of up to 10 km downstream large discharge points (Aas et al., 2002; Hylland et al., 2008). The exposure is, however, very low due to rapid natural dilution of the PW plume. The dilution process will vary widely from field to field, depending both on the actual discharge and several key factors of the recipient (current, wind, depth, etc.). Obviously, the parameters that can detect such a low chemical exposure signal from PW in water column organisms (such as fish), must be very sensitive. Research has shown that measurements of degradation products (metabolites) of PAH and alkylphenols in fish bile, act as highly sensitive exposure markers for PW (Beyer et al., 2010; Beyer et al., 2011; Sundt et al., 2012). In laboratory-based studies, both clear absorption and biological effects of produced water in fish are found, but risk assessment studies indicate that the ecological significance of such stock-related disturbances will be extremely small (Beyer et al., 2012). One must nevertheless emphasize that the large chemical complexity of produced water, combined with the longevity of low-concentration contamination in the areas downstream of oil and gas fields where PW discharges take place, makes it impossible to exclude the possibility that yet unstudied substances in the PW mixture may prove relevant for environmental risk and effect assessments.

Mixture-stress effects are another important issue that has gained recent attention in PW effect research; i.e. situations when the contaminants that occur together in a discharge stream can give additive, synergistic or antagonistic effect situations in the recipient sea ecosystem. In additive and synergistic effect situations, a toxic response may be generated in an organism or biotic system although each of the substances/stressors in the mixture are present at concentrations below their No Observable Effect Concentration (NOEC) for the investigated organism/system, e.g. (Beyer et al., 2014; Tollefsen et al., 2014; Song et al., 2018). Research on mixture effects is a complex, but growing field. Developmental, endocrine and reproduction related effects are among the ecotoxicological endpoints that most often are in focus regarding mixture stress situations of PW discharges.

It is important to know whether PW effect knowledge obtained under temperate test conditions is applicable to sub-Arctic and Arctic conditions. Reliable comparability (Arctic vs non-Arctic) would mean that investigators that undertake environmental risk assessments of PW discharges in an Arctic or sub-Arctic context can utilise information from the much broader database of effect data produced under temperate test conditions. Interestingly, the research studies that have assessed *the Arctic - non-Arctic comparability issue* have apparently not found clear evidence that support the postulation that organisms adapted to cold-seas being significantly and systematically more sensitive in comparison to related organisms adapted to temperate environments. Indeed, the concept of "the fragile Arctic" has been quite heavily debated both in the scientific and environmental management society, and this is still a clear matter of conjecture. One might as an introduction phrase M.J. Dunbar (1973, 1977, 1986; Dunbar, 1992), who stated:

> *"I can see little reason to suppose that Arctic ecosystems are any more or any less vulnerable to human interference than other ecosystems; and it seems from present development that a really fragile system can be found in the tropical rainforest. It is true that Arctic systems are usually simpler than others, involving lower diversity of species, so that extinction of a given link in the food web might be serious. But on the other hand, the individual numbers within species tend to be larger, and moreover the same ecosystem extends over very large geographical areas, as on the tundra and in the sea, so that damage in one area can be repaired by immigration from adjacent areas. In fact, the arctic ecosystem appears to be at least as tough as others. Small lakes, permafrost, and the Subarctic forest are examples of terrestrial systems that one has to treat with care and understanding, but the marine system do not show these special regions of concern"* (Dunbar, 1992).

A reasonable interpretation from this statement is simply that the dangers of pollution in the high north are more or less of the same sort and magnitude as elsewhere. In this report, the goal is to seek out to what extent our present knowledge on the marine ecosystem structure and function in the Barents Sea supports this interpretation, or whether the available knowledge on effects of PW on organisms and ecosystems justifies a different discharge regulation regime for PW from offshore installations in the Barents Sea in comparison to offshore fields elsewhere on the Norwegian continental shelf.

2.5 Environmental monitoring of PW discharges on the NCS

The Norwegian authorities have since 1999 required environmental monitoring of the water column by the oil companies operating in the Norwegian Sector of the North Sea, in addition to discharge monitoring. The general requirements to the Water Column Monitoring (WCM) are described in chapter 10 of the "Activities Regulation" (http://www.ptil.no/activities/category399.html#) for the offshore activities on the NCS, as well as in the Norwegian guideline for offshore environmental monitoring (Nilssen, 1999). These guidelines have been revised several times (Iversen et al., 2011; Iversen et al., 2015), and will be revised again in 2019. Rapid development of effects study methodology and sensitivity has called for frequent revisions. In an Arctic perspective it may be necessary to introduce other methods, effects endpoints and target organisms than in the present guidelines, which have evolved for use in temperate waters.

The Norwegian WCM program has been active since 2001. It has consisted of two elements: (1) the environmental effects monitoring and (2) the environmental condition monitoring, until the two elements were merged in the most recent revision of the offshore monitoring guidelines (Iversen et al., 2015).

(1) **The environmental effects monitoring**: Monitoring performed by using controlled deployment of sentinel organisms (fish and blue mussels) and passive samplers (semi-permeable membrane devices (SPMDs and POCIS)) in cages to assess exposure and effects of PW discharges at set distances from the offshore installations, and to validate PW dispersal models. The following reports were produced from these surveys: (Hylland et al., 2002; Børseth and Tollefsen, 2004; Hylland et al., 2005; Sundt et al., 2006; Sundt et al., 2008b; Brooks et al., 2009; Brooks et al., 2011b; Pampanin et al., 2013; Brooks et al., 2013; Brooks et al., 2015).

(2) **The condition monitoring**: Monitoring performed by analyses of wild fish collected in selected regions of the NCS to assess whether chemical and biological effect markers relevant to offshore industrial discharges deviate from reference regions. The following reports were produced from these surveys: (Grøsvik et al., 2007; Grøsvik et al., 2009; Grøsvik et al., 2012).

The funding for the WCM program activities has mainly been provided by the oil industry companies that operate on the NCS. Besides, a number of regular offshore field survey projects which recently have been conducted every three years. The WCM program has also included several laboratory and desk-top projects that have been conducted during the in-between years aiming to develop, test and validate new effect monitoring methodology, e.g. (Sundt et al., 2009a; Sundt et al., 2009b; Sundt and Björkblom, 2011; Sundt et al., 2011b; Sundt et al., 2012).

The offshore monitoring surveys done so far on the Norwegian continental shelf have covered major production sites, preferably in the Tampen, Ekofisk, and Sleipner regions. No Barents Sea sites have yet been covered by this monitoring. The data from the WCM monitoring surveys have mostly been presented in “grey literature” survey reports (see the citations to these reports highlighted in (1) and (2) above), but also in a some papers published in peer reviewed journals, i.e. (Hylland et al., 2008; Brooks et al., 2011b; Nilssen and Bakke, 2011; Bakke et al., 2013; Hale et al., 2016). The WCM program is quite unprecedented by its scope and complexity and especially by its broad use of chemical and biological markers in caged and wild organisms as tools for assessing possible impacts of petroleum industry discharges in offshore waters.

The broadest summary yet of results from WCM program activities and neighbouring environmental research activities was compiled by Bakke et al. (2013). Typically, WCM survey results have in a dominating manner revealed either no-effect-situations or low-effect-situations in the offshore waters that receive PW discharges. The typical worst-case findings have been relatively modest impacts detected in caged fish and mussels after these have been kept within the diluted PW plume and relatively close (some hundred meters to a few kilometres) downstream from major PW discharges. In wild fish, on the other hand, the specimens that have been collected in the neighbouring areas downstream from the PW discharge points have typically not displayed clear signals regarding typical PW associated contaminants. Possibly, the most striking impact results that have been found in wild fish populations studied in connection with WCM surveys are the increased hepatic DNA adduct levels that repeatedly have been found in haddock specimens collected in the Statfjord area (Tampen region) and which first were reported by Balk et al. (2011). These relevant and interesting observations of putative anthropogenic genotoxic stress in an offshore fish population are more likely to be associated to presence of large and old OBM drill cutting deposits in the Tampen area, and less likely to be caused by pollutants originating from the PW discharges in this region, albeit these discharges from the platforms at the Statfjord field have been large for many years.

The typical modest responses that have been detected in the WCM field surveys suggest that the overall risk for PW discharges to induce adverse impact in populations of wild fish and possibly other pelagic organisms is low. A low-risk situation is also widely supported from studies using environmental risk modelling procedures for environmental impact and risk assessment of PW discharges offshore. Increased efficiency of PW cleaning systems, continued phase out of non-green offshore production chemicals, increased use of PW reinjection operations, and development of discharge free field solutions are all processes that tend to increase the environmental safety level of PW management routines on the Norwegian oil and gas fields.

3 Ecological values and vulnerabilities of the Barents Sea ecoregion

The area known as the Barents Sea south, bordered by 74˚30'N, was opened for petroleum activities in 1989. According to the Norwegian Petroleum Directorate (OD), there are currently 71 production licenses in the Barents Sea (as of May 2017). A total of 157 wells have been drilled in the Norwegian sector of the Barents Sea, 126 of which are exploration wells, and 49 discoveries have been made in the period 1980 to 2016. The first Barents Sea field that came into production was Snøhvit (production start 2007), which mainly contains condensate. The next was Goliat (production start 2016), which contains both oil and gas. Both these fields are located in the area Tromsøflaket, in the southernmost part of the Barents Sea. The next field will be the Johan Castberg oilfield, which in 2018 was approved by the Norwegian Parliament, and the first oil production is scheduled for 2022. Other Barents Sea fields that currently are being considered are Alta / Gotha, located north of Snøhvit, and Wisting, located considerably further north and east.

The major fisheries resources and other important ecological values in the Barents Sea made offshore petroleum activities in these areas controversial. Therefore, the Norwegian government launched the zero-discharge policy for the future offshore developments in the Barents Sea ecoregion (which also includes the Lofoten area) (OED, 2003), demanding the establishing of PW re-injection as a standard routine for management of PW, and that the reinjection should have at least a 95% regularity level, and with thorough PW cleaning during the 5% periods when the PW reinjection wasn't active. Later, this zero-physical-discharge target was amended/softened to a "zero harmful discharge" target, which also was established as a regulatory target for all offshore oil and gas fields in Norwegian waters.

3.1 The Barents Sea knowledge base

In 1889, H. Mohn published the earliest Barents Sea study, *The physical conditions of Barents Sea*, in the Scottish Geographical Magazine. After that, the research base on the Barents Sea ecoregion has grown tremendously and today it includes many tens of thousands of items; including oceanography, geosciences, marine biology, ecology, fisheries, and environmental sciences being the most active scientific disciplines. A continued increase in research data from Barents Sea (and Arctic) studies is expected as the process of global warming will represent a particularly serious threat on polar ecosystems (Serreze and Francis, 2006; Screen and Simmonds, 2010). Certain issues are still understudied, such as toxicological test data for polar marine species (Chapman and Riddle, 2003, 2005), although this data-shortage has recently improved considerably. To keep track of the rapidly increasing knowledge about the Barents Sea region and the Arctic is a challenge. In Norway, the advisory group "**Overvåkingsgruppen**" (OVG, the task group for environmental monitoring of the marine ecosystems) has since 2006 had a special responsibility for collecting, systematising and reporting on new information and data regarding the condition status of three key oceanographic areas of Norway; i.e. the Barents Sea ecoregion (including areas off Lofoten), the Norwegian Sea, and the North Sea and Skagerrak. The condition status and key trends of the Barents Sea ecoregion is to be reported on each third year, with the most recent report being submitted in 2017 (Overvåkingsgruppen, 2017). The OVG reports are intended to provide knowledge support to the Norwegian governmental management of these ocean areas.

Internet websites become increasingly important for making research data popular and accessible to public users, also regarding the Barents Sea. An important website for disseminating knowledge and data about the Barents Ecoregion is the MOSJ website (Environmental monitoring of Svalbard and Jan Mayen) (http://www.mosj.no/en/) which is an environmental monitoring system and part of the Norwegian governmental environmental monitoring activities, alongside the national environmental information website environment.no (http://www.environment.no/topics/marine-and-coastal-waters/barents-sealofoten-area/). The Norwegian Polar Institute is the secretariat for MOSJ. Another organisation and web-resource of high relevance is the Arctic Council (https://arctic-council.org/index.php/en/) which is a high-level intergovernmental forum that addresses issues faced by Arctic governments and the indigenous people of the Arctic. There are six working groups associated to the Arctic Council, all which have relevance to the present report and Norway is the chair of AMAP- the Arctic Monitoring and Assessment Programme (https://www.amap.no/). Another web-resource associated to Arctic transnational cooperation is the BarentsInfo.Org website (http://www.barentsinfo.org/) which is related to the so-called Barents cooperation that formally was established in 1993 and which currently has Norway, Denmark, Finland, Iceland, Russia, Sweden, and the European Commission as members.

3.2 Key oceanographic and ecological features of the Barents Sea

The Barents Sea is one of several shelf seas surrounding the Polar basin. The Barents Sea is bordered by the Finnmark county (Norway) and Kola Peninsula (Russia) to the south, the shelf edge towards the Norwegian Sea to the west, the archipelagos of Svalbard to the northwest, Franz Josef Land to the northeast and Novaya Zemlya to the east (Figure 4). The Barents Sea can be considered as a relatively deep shelf sea. It covers an area of approximately 1.4 million km^2 and has an average depth of about 230 m, but with large regions that either are shallower or considerably deeper. More than 50% of the Barents Sea have depths of 200-500 m.

The oceanographic circulation pattern in the Barents Sea is strongly influenced by topography which causes the seawater to flow along the slopes rather than across them. The currents are stronger over the slopes than over the flat areas. In addition, this effect causes the current to generally follow the slopes around the shallow banks in the clockwise direction, and around the deeper basins in the counterclockwise direction. This is true not only for the Barents Sea, but for the entire northern hemisphere. Through the western entrance of the Barents Sea there is a massive inflow of warm and saline Atlantic waters from the North Atlantic drift current (i.e., the northern extension of the Gulf Stream), which is defined by salinity higher than 35‰, as well as a warm and fresher coastal current along the Norwegian coast. The inflowing current divides into two large branches, one northern branch which flows into the Hopen Trench and one southern branch, which follows the coast eastwards in direction of Novaya Zemlya (Figure 4). The relative strength of the two branches depends on the local wind conditions in the Barents Sea. In the northern part of the Barents Sea, cold Arctic water with lower salinity flows from northeast towards southwest.

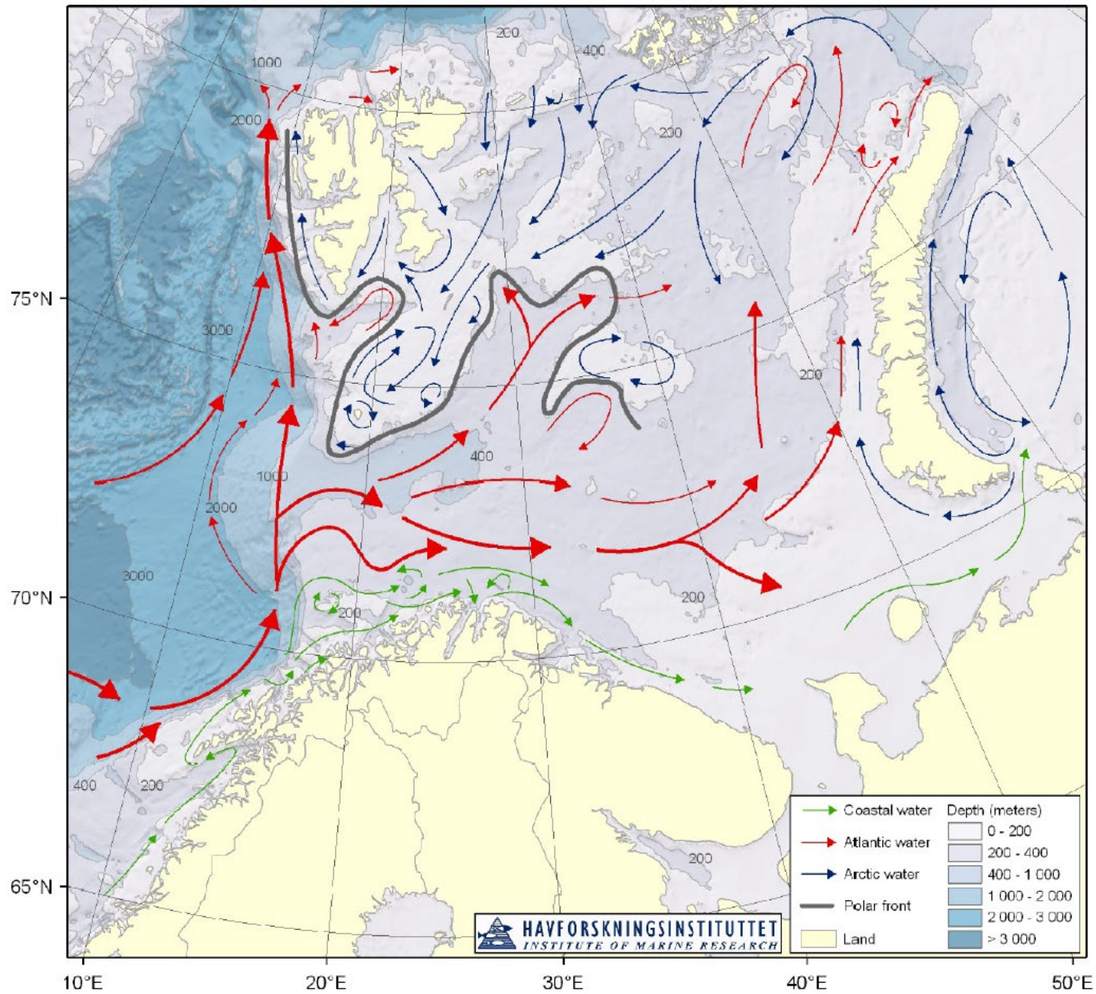

Figure 4: Bottom contours and major current systems in the Barents Sea. Red arrows show Atlantic water currents, blue arrows show Arctic water currents and the green arrows show coastal water currents. The typical position of the polar front is indicated with dark gray line. Map source: Institute of Marine Research and ICES (2008). See also: (Eriksen et al., 2017a).

Two major oceanographic features in the Barents Sea that are very important for the Barents Sea ecosystems are the Polar Front (PF, polarfronten) (Figure 4) and the Marginal Ice Zone (MIZ, often called the ice-edge, iskanten) (Figure 5). The PF is the mixing zone between warmer Atlantic water and colder Arctic water and is characterized by a strong gradient for temperature and salinity. This gradient is typically most pronounced in the upper part of the water column. The MIZ, on the other hand, is the highly dynamic transition zone between open ocean and floating sea ice, which is particularly important for the biological production within the Barents Sea, especially during the spring and summer months (Figure 6). The spatial position of the MIZ is highly variable and depends on factors such as wind-direction and ocean currents. The ice reaches its maximum southern extent in March-April, whereas areas as far south as Bjørnøya, and some years even further south, can be covered with ice. During the summer the ice melts, and in the late summer and early autumn the ice edge is usually north and east of Svalbard. The melting of the ice releases freshwater, which forms a surface layer of 10-50 m thickness with lower salinity along the edge of the ice. The water masses are stabilized, and phytoplankton remains in the upper water layer, where sunlight and nutrient salts provide the basis for a very strong algae bloom. This bloom moves northwards as the ice withdraws.

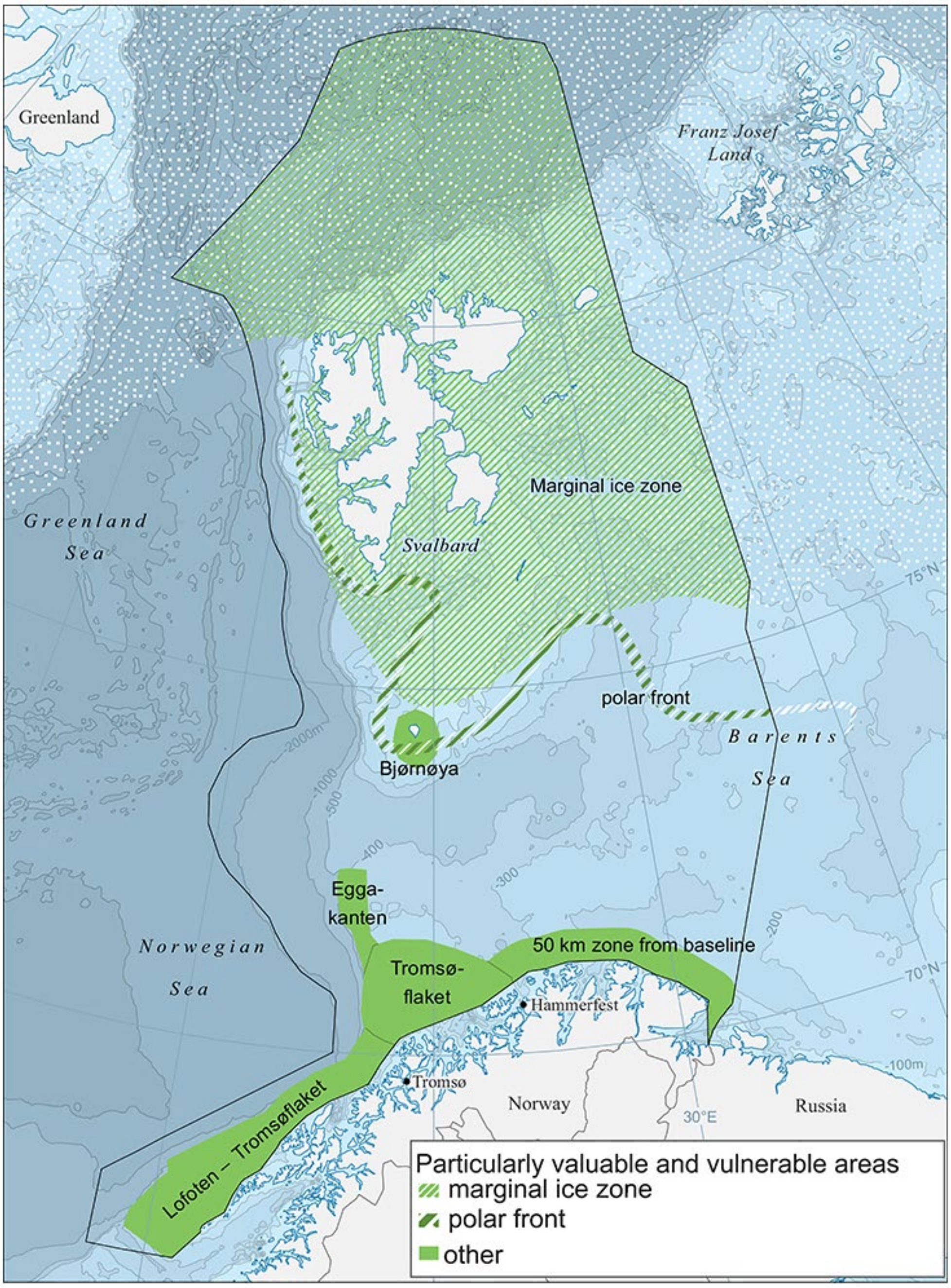

Figure 5: The marginal ice zone (MIZ), the highly dynamic transition zone between open ocean and floating sea ice, is a particularly valuable and vulnerable area within the Barents Sea–Lofoten management plan area. The delimitation of the marginal ice zone has been updated using data on sea ice extent for the period 1985–2014. Map source: Norwegian Polar Institute.

The ecosystem of this Marginal Ice Zone is very variable, but the production of phytoplankton is very high. The plankton attracts both fish, birds, and mammals. The relevance of the PF, MIZ and the ice edge for the ecological production in the Barents Sea during the annual cycle (e.g., Figure 6) has attracted attention from research groups and many good quality research studies are available, e.g. (Falk-Petersen et al., 1999; Wassmann et al., 1999; Arashkevich et al., 2002; Rat'kova and Wassmann, 2002; Reigstad et al., 2002; Riser et al., 2002; Soreide et al., 2003; Wassmann et al., 2006; Reigstad et al., 2008; Carroll et al., 2014; Erga et al., 2014; Kedra et al., 2015; Leu et al., 2015). The Institute of Marine Research (Bergen) and the Norwegian Polar Institute (Tromsø) have recently compiled broad research reviews regarding the current oceanographic and ecologic knowledge of the PF (Lien, 2018) and the MIZ and the ice edge (von Quillfeldt, 2017).

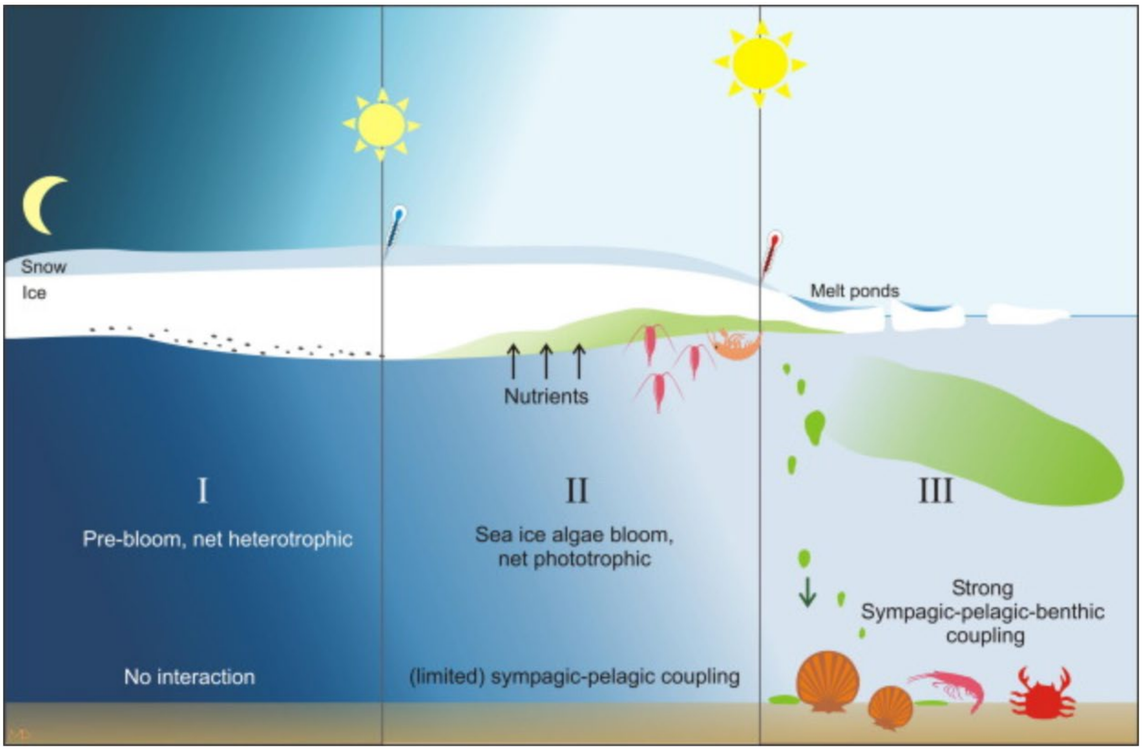

Figure 6: In the transition from winter to spring there are three main stages in the development of algal blooms associated with the Marginal Ice Zone (MIZ) in the Barents Sea. The transition from Phase I to II is controlled by light, while temperature is more important for the transition between Phase II and III. The marginal ice zone is very dynamic due to the influence of the weather and rapid changes. Changes in its extent may take place over hours or days. Ecological vulnerability is greater in the marginal ice zone because of high production in spring and summer, and the high density of vulnerable environmental elements in some parts of the year. Illustration source: Leu et al. (2015).

During the winter, the surface seawater in the Barents Sea becomes cooled by low air temperature. When the seawater becomes cold enough to freeze to ice, most of the salt that is in the water precipitates, so that the salinity in the seawater under the ice increases. Cold water is heavier than warm water, and salty seawater is heavier than less salty seawater. The very cold and salty water is heavy and can sink down to the bottom. Such formation of cold, salty bottom water takes place in certain regions of the Barents Sea. The cold bottom water flows out of the Barents Sea to the east and into the Arctic Ocean, and can reach deep depths, and is probably crucial for the large-scale circulation in the North Atlantic.

Because of the influence from Atlantic waters and coastal waters, the southern part of the Barents Sea is relatively warm. In the southwest, the Atlantic waters are always warmer than 3°C, but Atlantic conditions are normally defined at temperatures above 2°C. The Arctic waters are always below 0°C. The polar front is largely linked to the bottom topography. In the western part of the Barents Sea, the polar front locates to the slopes around the Spitsbergen bank and to the Central Bank, with only minor variations. Further east, the front is wider and the position is more variable. Because the Atlantic waters and coastal waters bring nutrients and animal plankton from the Norwegian Sea, the area in and south of the Polar Front is usually very productive.

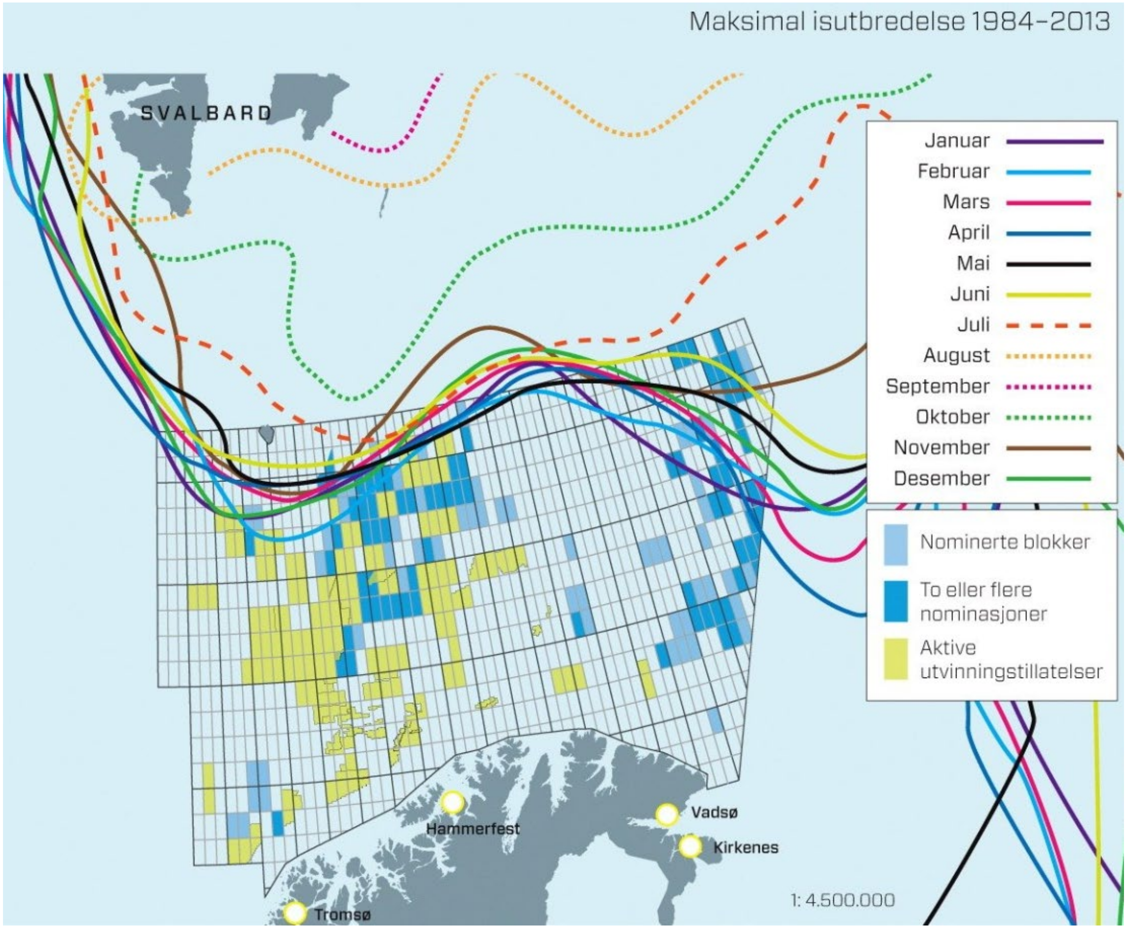

Figure 7: Variability of the ice-edge (marginal ice zone) in the Norwegian part of the Barents Sea during a year cycle. The overview is based on meteorological data from the last 30 years. Several offshore blocks proposed by the Norwegian Ministry of Petroleum and Energy are in areas that are influenced by ice for much of the year. Map source: Norwegian Petroleum Directorate and the Polar Institute.

A characteristic feature of the Barents Sea is the large natural variation within the year cycle and from year to year in sea temperatures and ice conditions (Figure 7, Figure 8). The most important cause of the variation is the variable amount and temperature of the Atlantic water entering the Barents Sea. The key recent development for sea temperatures in the Barents Sea is a general increase that has occurred during the past 40 years, but with significant yearly variations. Currently the temperature is well above the long-term average for the Barents Sea. In parallel with this general rise in sea temperature, the spread of sea ice in the Barents Sea has decreased significantly since satellite measurements started in 1979. The winter of 2015-2016 was characterized by high temperatures, large influx of Atlantic waters, and the smallest ice expansion since the measurements started (Overvåkingsgruppen, 2017).

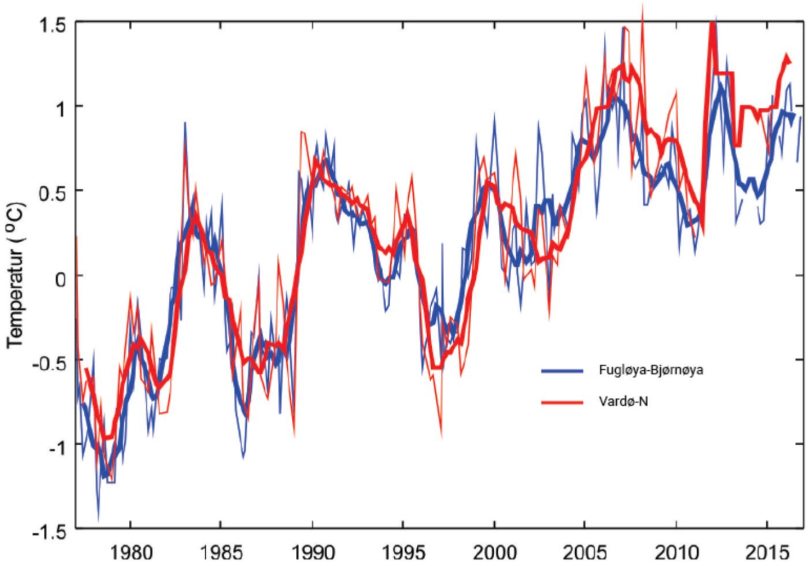

Figure 8: The figure shows the temperature deviation in the inflow Atlantic waters in relation to the long-term average (1977-2006) at between 50 and 200 meters depth on the oceanic intersections Fugløya-Bjørnøya (blue) and Vardø-N (red). Thin lines correspond to measured values whereas thick lines correspond to one-year sliding means. Source of figure: IMR / miljøstatus.no.

Satellite measurements of chlorophyll a suggest that biomass of phytoplankton in the Barents Sea may have been higher in the years 2013-2016 than previously in parts of the Barents Sea. This may mean that primary production may have been higher in these years. It is unclear whether this may represent a trend or only intermittent variation. Zooplankton eat phytoplankton and is itself food for fish and other plankton-eating animals. They are therefore the most important link between the primary production and the other parts of the Barents Sea ecosystem, including the major fish stocks. The biomass of zooplankton varies between years and it has apparently been high in recent years (Overvåkingsgruppen, 2017). However, zooplankton species that prefer warm seawater has increased while the amount of important Arctic species has decreased. This could cause poorer nutritional conditions for arctic plankton eaters that are dependent on fatty Arctic animal plankton.

For benthos, there has been significant variation measured by biomass in different parts of the Barents Sea in recent years. In the eastern areas, the phenomenon is probably mostly due to increased predation from snow crabs (*Chionoecetes opilio*, snøkrabbe), which is an invading north-pacific species which first was detected in the Barents Sea in 1996. Currently, snow crabs are spreading westward and northward, although it still has its main distribution within the Russian part of the Barents Sea. The Barents Sea stock of deepwater shrimps (*Pandalus borealis*, dyphavsreke) has increased slightly and is above the long-term average. The stock of red king crab (*Paralithodes camtschaticus*, kongekrabbe, also called Kamchatka crab) is stable. The current fishing of this species west of 26 ° East seems, for the time being, to be effective in preventing further spreading westwards (Overvåkingsgruppen, 2017).

For fish populations, the monitoring of Barents Sea stocks is thought to provide fairly accurate estimates of stock developments for the main commercially exploited species, although there are more uncertainties concerning the ecological interactions of different species. Generally, the commercial fish stocks of the Barents Sea are in good condition, with the exceptions for polar cod (*Boreogadus saida*, polartorsk) which seems to have been forced northwards to the northern borders of the Barents Sea. This trend is by many taken as a sign of a temperature induced borealization of Barents Sea fish communities (Fossheim et al., 2015). Another major concern is the Atlantic redfish (*Sebastes norvegicus*, vanlig uer) for which there has been a stock collapse in the Barents Sea. More information about the assessments of Barents Sea fish stocks can be found on the website of the Institute of Marine Research (Bergen) and in workgroup reports from the International Council for the Exploration of the Sea, e.g. ICES (2016b). The Barents Sea stock of cod (*G. morhua*) is currently considered to be in very good condition, although there has been a slight dip in recent years. The spawning stock is at a level far above the average of the 70 years period in which systematic registrations have been carried out. In recent years, cod have also adapted a more northern distribution in comparison to previous records. The stock of deepwater redfish (*Sebastes mentella*, snabeluer) has had a positive development, with good recruitment in recent years. The Barents Sea population of Atlantic redfish has shown a failing recruitment since the early 1990s, and despite increasingly stricter protective measures, the stock of this species is still decreasing and is now lower than ever. In the Norwegian Red List from 2015, the stock is characterized as "highly threatened". Also the Polar cod stock has declined for several years in the Barents Sea, but this species showed a sharp rise from 2015 to 2016 (Overvåkingsgruppen, 2017).

For seabirds in the Barents Sea ecoregion, almost all indicators show a decline in the breeding populations. The decline regards both the last ten years and over the total time the populations have been monitored. This applies to common guillemot (*Uria aalge*, lomvi) and black-legged kittiwake (*Rissa tridactyla*, krykkje) along the mainland coast and Brünnich's guillemot (*Uria lomvia*, polarlomvi) and Atlantic puffin (*Fratercula arctica*, lundefugl) in the whole area. A key cause for population effects is thought to be food shortage during the nesting period. However, it is difficult to decide the causes for low food availability, but secondary effects of climate-related changes, lower production of prey, or fishing activities have been proposed (Overvåkingsgruppen, 2017). In addition, some of the breeding sites are under pressure from growing white-tail eagle (*Haliaeetus albicilla*, havørn) populations, preying on the breeding birds. It is not known if ingestion of marine litter occurs at a harmful level for seabirds in the Barents Sea.

For marine mammals, both growth after conservation and climate change are believed to affect the populations of different species in the Barents Sea. Walrus (*Odobenus rosmarus*, hvalross) is an example of a species that has increased significantly in number after several decades of conservation. Ringed seal (*Pusa hispida*, ringsel) is one of several species that is very dependent on sea ice, and studies indicate that this species, due to a decrease in the amount of sea ice, now is spending significantly more time (and energy) on food search in comparison to earlier. Polar bears (*Ursus maritimus*) in the Barents Sea is an example of a stock that is affected by both conservation measures and climate change. Several whale species along the coast of Svalbard currently show stable or increased numbers. We have gained increased knowledge of several ice-dependent whales in the region. Sea mammal species which are harvested have stable or growing populations. The situation is good for seals along the mainland coast of the Barents Sea, unlike seal populations further south along the Norwegian coast (Overvåkingsgruppen, 2017).

3.3 Structural properties of Barents Sea food webs

A key feature of the Barents Sea ecosystem (and for ecosystems elsewhere in the Arctic area) is simple food webs that have a relatively small number of species (i.e., low species richness). This means there is a small number of dominant feeding links and a short distance between primary production level and the top predators (Durant et al., 2014; Murphy et al., 2016; Olivier and Planque, 2017). Such a trophic structure makes ecosystems of the high north more vulnerable for changes in comparison to temperate ecosystems. If one species is changing significantly in abundancy (e.g., decreasing) there may be no other species that are ready or capable of filling in, thereby the change that occur for one species will often propagate further to other species, resulting in so-called cascading effects. Research has been performed to clarify the possible role of zooplankton in modulating ecosystem effects of acute pollution events such as oil spills in the Norwegian and Barents seas (Basedow et al., 2010; Stige et al., 2010; Stige et al., 2011; Hidalgo et al., 2012).

The whole Barents Sea area is in general dominated by pelagic ecological processes, but the area is by no means ecologically uniform, e.g. regarding key physicochemical drivers (salinity, seawater temperature, water depth, etc.) or important biotic parameters such as the spatial distribution of key species and species-species interactions. Within the relative limited scope of this report, most emphasis is devoted to identifying and discussing the most essential factors that make the Barents Sea ecosystems unique, and especially to explain key differences compared to the marine ecosystems further south on the Norwegian continental shelf. For a more detailed description of the Barents Sea ecosystems one may consult comprehensive reviews that address these topics more in depth, such as (Tjelmeland and Bogstad, 1998; Gjosaeter et al., 2002; Ushakov and Prozorkevich, 2002; Olsen et al., 2010; Falk-Petersen et al., 2011; Drinkwater, 2011; Lilly et al., 2013; Michalsen et al., 2013; Ottersen et al., 2014a; Kortsch, 2016; Eriksen et al., 2017b).

Due to the inflow of warm Atlantic water and the mixing with cold polar water, the Barents Sea has a high biological production compared to most other high-latitude marine ecosystems. The spring bloom of phytoplankton throughout the Barents Sea starts when stable stratification is established in the water column top layer (Strass and Nothig, 1996; Wassmann et al., 1999; Olli et al., 2002; Sharples et al., 2006; Hegseth and Tverberg, 2013). This stratification phenomenon can be caused by different factors in different regions of the Barents Sea. Salinity driven stratification occur close to the ice edge, with fresher seawater from melting sea ice stabilising the top layer of the water column. In other areas, lateral spreading of the fresher coastal water from the southern branch give the same effect, and the solar heating of the surface waters in the Atlantic water masses will also cause a temperature driven stratification that may trigger the algae bloom. Diatoms are typically the dominating phytoplankton group in the Barents Sea, especially during the early phase of the spring bloom, with *Chaetoceros socialis* often being the most abundant species. The algae concentrations sometimes reach several million cells per litre (Giraudeau et al., 2016). In later phases of the spring bloom, other algal groups such as flagellates take over. The most important species in this later phase is often *Phaeocyctis pouchetii*, but these patterns often vary considerably between years. The phytoplankton biomass then becomes food for key zooplankton species such as the copepods *Calanus finmarchicus*, *C. glacialis*, *C. hyperboreus*, and *Oithona* spp., as well as krill (Euphausiacea). Subsequently, these primary consumers become food sources for key secondary consumers such as capelin (*Mallotus villosus*, lodde), juvenile herring (*Clupea harengus*, NVG sild), young Arctic cod (*G. morhua*), polar cod (*Boreogadus saida*), as well as for sea birds such as the little auk (alkekonge, *Alle alle*) and species of baleen whales feeding on the krill. Further on, especially the primary consumers and particularly capelin become food sources for a range of secondary consumers such as the north-east Arctic cod (the Barents Sea has the world's largest *G. morhua* population), baleen whales, harp seals, and many seabirds such as common guillemot (lomvi, *Uria aalge*) and Brunnich's guillemot (polarlomvi, *Uria lomvia*).

A key question for research currently is how the ecosystems of the Barents Sea and elsewhere in the Arctic will be influenced by global warming. The Arctic has warmed dramatically in recent decades, with greatest temperature increases observed in the northern Barents Sea, resulting in decreased stratification and increased Atlantic conditions in this area (Lind et al., 2018).

3.4 Key fish resources of the Barents Sea

Eriksen et al. (2017b) recently analysed the spatial and temporal variation in Barents Sea biomass production over the period 1993-2013, involving 25 components of the pelagic community, and including macroplankton, 0-group fish, and juvenile and adult pelagic fish. They estimated that the total biomass of the investigated pelagic compartment, not including mesozooplankton, ranged between about 6 and 30 million tonnes wet weight with an average of 17 million tonnes over the 21-years period. Krill was found to be the dominant biomass component (63%), whereas pelagic fish (capelin, polar cod, and herring) made up 26% and 0-group fish 11% of the biomass on average. The spatial distribution of the biomass showed a broad-scale pattern reflecting differences in distribution of the main pelagic fishes (capelin in the north, polar cod in the east, and herring in the south) and transport of krill and 0-group fish with the Atlantic water flowing into the southern Barents Sea. Dividing the Barents Sea into six regions, the highest average biomass values were found in the Southwestern and South-Central subareas (about 4 million tonnes in each), with krill as the main component (Ibid.). The fisheries of the Barents Sea, particularly the cod fisheries, are of great economic importance for both Norway and Russia. The Northeast-Arctic stock of Atlantic cod is the world's largest cod population (Kjesbu et al., 2014). This population spawn along the west and north coasts of Norway from mid-February to early May and the eggs and larvae drift pelagically north- and eastwards towards the nursery area in the Barents Sea (Olsen et al., 2010; Ottersen et al., 2014b). Also many other species in the region are of great commercial interest, such as capelin, haddock, redfish, Greenland halibut and shrimp. In total, there are more than 200 species of fish, thousands of benthic invertebrate species and diverse communities of plankton, seabirds and marine mammals inhabiting or visiting the Barents Sea, however, only a limited number of these species are of commercial interest currently (Jakobsen and Ozhigin, 2011).

3.5 High-vulnerability areas and indicators in the Barents Sea

There are certain areas within the Norwegian part of the Barents Sea ecoregion which are regarded as being extra valuable and/or potentially extra vulnerable, i.e. so-called SVSO (Særlig Verdifulle og Sårbare Områder), and these areas should be specially protected. According to the Convention on Biological Diversity, the set of criteria used for identifying SVSOs are:

- Uniqueness or rarity
- Special importance for sensitive life-history stages of species
- Importance for threatened, endangered, or declining species and/or habitats
- Biological productivity
- Biological diversity
- Vulnerability, fragility, sensitivity, or slow recovery
- Naturalness

Other synonyms or closely related terms to SVSO are: Particularly Valuable Areas (PVA), Valued Ecosystem Component (VEC), Ecological and Biological Significant Areas (EBSA).

The definition of an area to be SVSO is based on the presence of one (or several) highly important ecosystem element (so-called SVSO indicators) (Table 5). These indicators are subjected to repeated quality status assessments aiming to monitor temporal developments of their condition status.

Table 5: SVSO areas in the Norwegian part of the Barents Sea and the ecological elements which are used as indicators for assessing their condition status. Further data concerning the status and trends of each SVSO and each indicator can be found in the most recent report from Overvåkingsgruppen (2017).

SVSO areas	SVSO Indicators	The following indicators are reported for the ecosystem and not within each SVSO through the Overvåkingsgruppen, but it is still important data for the state of the ecosystem within the SVSOs [1]
The area from Lofoten to Tromsøflaket, including the Eggakanten area.	North-East-Arctic cod, Norwegian spring-spawning herring, redfish (both species), common guillemot, black-legged kittiwake, Atlantic puffin	Zooplankton species diversity; biomass, phytoplankton: biomass; spring bloom; Spawning stocks and/or total stock sizes of cod, haddock, capelin; blue whiting; Coral reefs and sponges Breeding success and spatial distribution of common guillemot, black-legged kittiwake, Atlantic puffin Spatial distribution of whales
Tromsøflaket	North-East-Arctic cod, Norwegian spring-spawning herring	Zooplankton species diversity; biomass, phytoplankton: biomass; spring bloom; Spawning stocks and/or total stock sizes of cod, haddock, capelin; blue whiting Coral reefs and sponges Breeding success and spatial distribution of common guillemot, black-legged kittiwake, Atlantic puffin Spatial distributions of whales
The coast-near areas from Tromsøflaket to the border to Russia	Common guillemot, black-legged kittiwake, Atlantic puffin	Zooplankton species diversity; biomass, phytoplankton: biomass; spring bloom; Spawning stocks and/or total stock sizes of cod, haddock, capelin; blue whiting Coral reefs and sponges Breeding success and spatial distribution of common guillemot, black-legged kittiwake, Atlantic puffin Spatial distribution of whales
The Marginal Ice Zone (MIZ, the Ice Edge, Iskanten)	Polar bear, capelin, common guillemot, black-legged kittiwake,	Zooplankton species diversity; biomass, phytoplankton: biomass; spring bloom; Spawning stocks and/or total stock sizes of cod, haddock, capelin, beaked redfish; Atlantic redfish; young spring-spawning herring; Greenland halibut; blue whiting Red king crabs, coral reefs, and sponges Breeding success and spatial distribution of common guillemot, Brünnichs guillemot; black-legged kittiwake, Atlantic puffin Spatial distribution of whales
The polar front	North-East-Arctic cod, Capelin, common guillemot, black-legged kittiwake,	Zooplankton species diversity; biomass, phytoplankton: biomass; spring bloom; Spawning stocks and/or total stock sizes of cod, haddock, capelin, beaked redfish; Atlantic redfish; young spring-spawning herring; Greenland halibut; blue whiting Red king crabs, coral reefs, and sponges Breeding success and spatial distribution of common guillemot, Brünnichs guillemot; black-legged kittiwake, Atlantic puffin Spatial distribution of whales
The oceanic area around Svalbard and Bear Island (Bjørnøya).	Common guillemot, Brünnich's guillemot, black-legged kittiwake	Zooplankton species diversity; biomass, phytoplankton: biomass; spring bloom; Spawning stocks and/or total stock sizes of cod, haddock, capelin, beaked redfish; Atlantic redfish; young spring-spawning herring; Greenland halibut; blue whiting Red king crabs, coral reefs, and sponges Breeding success and spatial distribution of common guillemot, brünnichs guillemot; black-legged kittiwake, Atlantic puffin Spatial distribution of whales

[1] **pers. info from Dr. G. van der Meeren, IMR (Bergen, Norway)**

3.6 Sources of stress to Barents Sea ecosystems

The commercial fisheries represent the most significant negative pressure against the ecological quality status of Barents Sea ecosystems, as illustrated in Figure 9. The fishing activities, especially with bottom trawls, cause abrasion and damage to benthic habitats and communities, although there has been some recent improvements with fishing gear and fishing practices of Norwegian vessels. According to data from the Norwegian fishing authorities, Norwegian vessels are responsible for about 20% of the total landings by bottom trawls in the Barents Sea.

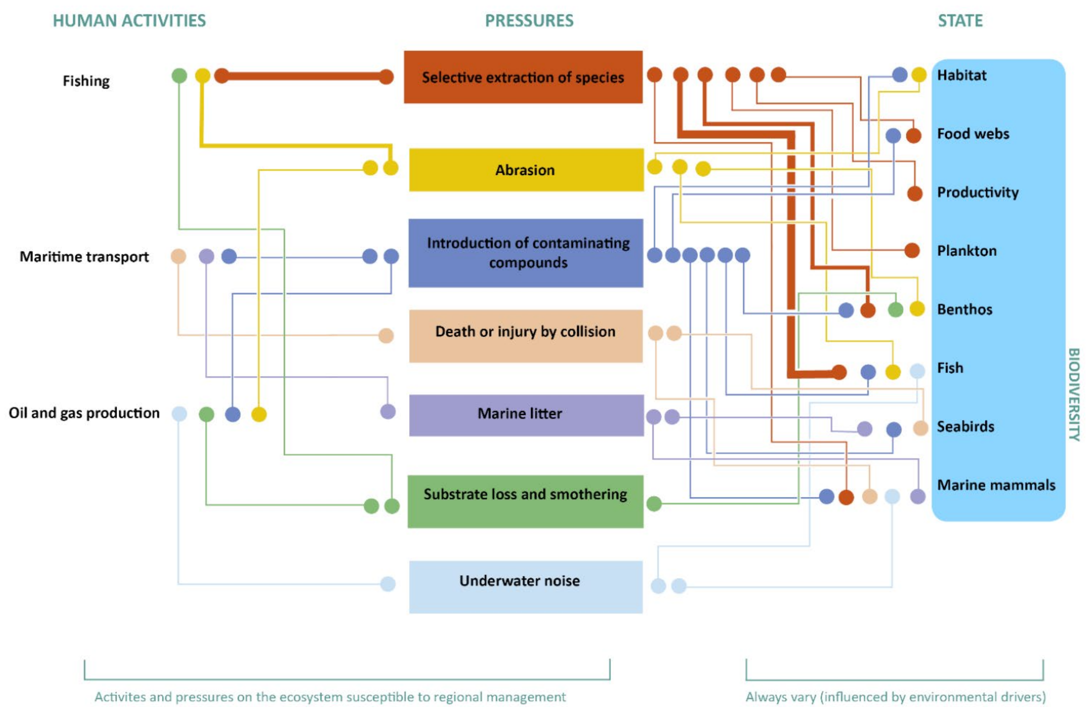

Figure 9: Overview of how key human activities in the Barents Sea area represent pressures to the state of different ecosystem components in the Barents Sea marine ecosystem. The width of lines indicates the relative importance of pressures. Illustration source: ICES (2016a).

Accidental oil spills are most likely a more significant risk factor than PW discharges in relation to potential damage to the quality status of the Barents Sea ecosystem, especially if a spill coincides in space and time with the spawning of an important fish stock, as the period post spawning is when the most sensitive early life stages of a single year's juvenile recruitment cohort are present (Carls et al., 1999; Heintz et al., 1999; McIntosh et al., 2010; Vikebo et al., 2012; Vikebo et al., 2014). Notably, PW discharges always contain dispersed oil, although the levels normally are low and under strict regulation. Operational PW discharges have for years been the largest source of oil released to the marine environment on the NCS. Effect studies of accidental marine oil spills have therefore some relevance also for effect assessments of PW discharges, although accidental oil spills are also very different from PW due to the typical large amount of oil entering the ecosystem within a relative short period. Recently, Carroll et al. (2018) simulated the impacts of a major oil spill occurring in the core spawning areas of Atlantic cod in the Lofoten area. By modelling the life history of individual fish eggs and larvae, they predicted some deviations from the historical pattern of recruitment to the adult population due to toxic oil exposures, but even in the worst case scenario the cod population remained at a full reproductive potential, mostly because of the diverse age distribution and because enough juveniles survived to replenish the population. The cod population in the Barents Sea appears to be rather resilient towards oil spills, at least with regard to the potentials of oil spills to damage the quality of reproduction and recruitment processes. Much research has been performed to characterize the spatiotemporal variability in mortality and growth of fish larvae in the Lofoten-Barents Sea ecosystem and the possible long-term effects of local scale oil pollution on fish Populations and communities, e.g. (Melsom et al., 2012; Kvile et al., 2014; Langangen et al., 2014a; Langangen et al., 2014b; Lien et al., 2014; Ottersen et al., 2014a; Stige et al., 2015; Kvile et al., 2016a; Kvile et al., 2016b; Langangen et al., 2016; Kvile et al., 2017; Langangen et al., 2017b; Langangen et al., 2017a; Stige et al., 2017; Stige et al., 2018b; Stige et al., 2018a; Langangen et al., 2018; Farber et al., 2018).

The maritime transport sector is also a significant source of stress to the quality state of the Barents Sea ecosystems. Together with the oil and gas industry, the maritime sector is expected to increase significantly in coming years, and this increase is met with concern (Blanchard et al., 2014; Hauge et al., 2014; Ottersen et al., 2014b). Salmon farming and other aquaculture industry activities that operate along the coast of Troms and Finnmark are expected to increase significantly in the coming time period, and that trend may possibly add to various pressures on the coastal ecosystems in this part of the southern Barents Sea region unless strict regulations and technological developments at many levels can ensure a greener production.

As illustrated in Figure 9, organisms or populations in the Barents Sea may typically be exposed to several negative pressures at the same time. It is difficult to estimate the risk for negative effects on individuals and populations from a range of stressors/pressures acting together, e.g. see discussion by Beyer et al. (2014), but the relevance of such issues is increasingly recognised. Stige et al. (2018b) discussed the possibility that fishing activities and petroleum activities could act in concert to cause negative effects on fish stocks. They suggested that if high fishing pressure erodes the demographic structure of a fish stock this may impact the stock spawning strategy, so distributions of offspring may become more concentrated in space and time, leading a larger fraction (up to twice as large) of the year-class of offspring to be hit and negatively affected in the event of a large oil spill. Similarly, the study of Christie et al. (2019) suggested the possibility that Arctic benthic communities can be sensitised to events of added stress when already significantly stressed by sea urchin overgrazing.

3.7 Will the vulnerability of the Arctic increase by global warming?

The effect of global warming appears to be particularly strong in the Arctic. Regardless of whether the global warming process is caused by man-made release of greenhouse gases, a significant warming will lead to wide-ranging ecosystem changes within the Barents Sea ecoregion and elsewhere in the Arctic. There are already several alarm signals indicating that large scale ecosystem modifications are ongoing in the Northern Atlantic, Arctic ocean and the Barents Sea ecoregion. According to ICES, the key alarm signals of greatest concern with respect to the Barents Sea are:

- Warming seawater and less sea ice in the Barents Sea (Bentley et al., 2017; Christiansen, 2017; Eriksen et al., 2017b; Lind et al., 2018).
- Decrease of mesozooplankton (in particular copepods and *T. libellula*, leading to possible decline and recruitment failure of polar cod (Siegelman-Charbit and Planque, 2016).
- Abnormal North Atlantic Oscillation variation pattern affecting Barents Sea ice coverage.
- Decreases in fisheries landings in the Barents Sea, after the peak of 2011.
- Northerly movement of the Barents Sea cod stock, impacting the food web, e.g. through predation on polar cod.
- Introduction of non-indigenous species, such as snow crabs and red king crab.

Several recent studies highlight the urgent need to calibrate parameters for risk assessment of oil spills under Arctic conditions because of an increased risk of accidental oil spills due to increased shipping and oil exploration activities, e.g. Øverjordet et al. (2018). As the ice cap of the Arctic diminishes due to global warming, the polar sailing route will most likely be open for larger parts of the year and these changes are expected to increase the maritime traffic intensity as well as the pollution load from specific contaminant groups, such as PAHs. The traffic will be going to and from new Russian terminals in the Barents Region, or sailing the Northern Sea Route (Nordøstpassasjen) that runs along the Russian Arctic coast from Murmansk on the Barents Sea, along Siberia, to the Bering Strait and Far East. To prepare for monitoring of traffic-related contamination trends in these waters. Jorundsdottir et al. (2014) mapped the current background concentrations of PAHs, PAH metabolites and inorganic trace elements in the North-Atlantic Arctic and sub-Arctic coastal environment.

4 Environmental Risk Assessment (ERA) of PW

The zero-adverse discharge policy for petroleum activities on the NCS implies that the operators should describe how different activity-alternatives are likely to affect the environmental risk of the assessed activity. The essential tools for obtaining these impact predictions are Environmental Risk Assessment (ERA) models, which today largely are integrated in the planning, decision-making and management routines. This risk-based management of PW discharges from offshore installations is described by OSPAR (2012), and an overview of the achievements of the Risk-Based PW management on the NCS for the period 2002–2008 is described by Smit et al. (2011).

4.1 ERA tools of relevance to offshore PW discharges

An environmental risk assessment (ERA) is an evaluation to determine whether the discharge/introduction of a substance or mixture causes an unacceptable risk to the recipient environment. Thus, ERA deals with the risk from events that may have not yet happened. ERA encompasses several related and often overlapping subfields, such as: environmental impact assessment, ecological risk assessment, human health risk assessment, strategic environmental assessment, sustainability assessment, the precautionary principle, etc.

The basic structure of an ERA framework is shown in Figure 10. The key element in most ERAs is the PEC:PNEC ratio concept, PEC (predicted environmental concentration) being the measured or predicted exposure concentration of a substance. PNEC, the predicted no effect concentration, is the exposure concentration below which no significant effect in the receptor environment/organism will occur. PNECs are usually calculated by dividing toxicological dose descriptors by an assessment factor, and the endpoints that most frequently are used for deriving PNECs are mortality (LC50), growth (ECx or NOEC) and reproduction effects (ECx or NOEC). The assessment factor should provide confidence that no adverse effects should occur in the environment. The PEC:PNEC ratio is often expressed as the risk quotient (RQ). If the RQ < 1 it means that there is no/limited risk of effect under the conditions used in the ERA prediction. The literature on ERA of PW discharges is broad, with more than 100 journal publications and an unknown number of grey literature items available internationally. Almost 50% of the research papers about ERA of PW are "Norwegian" studies.

ERA modelling tools are widely applied in management of PW discharges and other environmental discharges from petroleum operations on the NCS. Commonly used tools are the Environmental Impact Factor (EIF) and DREAM (Dose related Risk and Effect Assessment Model) models; which both have been developed by SINTEF (Trondheim, Norway) in cooperation with a group of petroleum companies operating in Norwegian waters.

The EIF model, (Johnsen et al., 2000; Rye et al., 2004), is a PEC/PNEC and ERA based indicator for assessing and quantifying the environmental risk from PW and other discharges (drill cuttings and mud releases). The EIF is related to the recipient water volume where the PEC/PNEC ratio > 1, this means the key result of the EIF modelling is the volume of recipient seawater that contains the assessed stressors at a concentration exceeding the assumed threshold for ecotoxicological effects. The use of EIF modelling facilitates the identification and selection of cost-effective risk mitigation measures, e.g. substitution of adverse offshore chemicals with more benign substances. The EIF model was developed to assist the work towards reduction of possible environmental impacts from PW releases to a "zero release" option, or a "zero effects release" option. According to Smit et al. (2011), the active use of EIF tools by operators on the NCS has during the period 2002 – 2008 contributed to about 55% risk reduction of the PW discharges, although the total volume of PW discharged increased by approximately 30% during the same period.

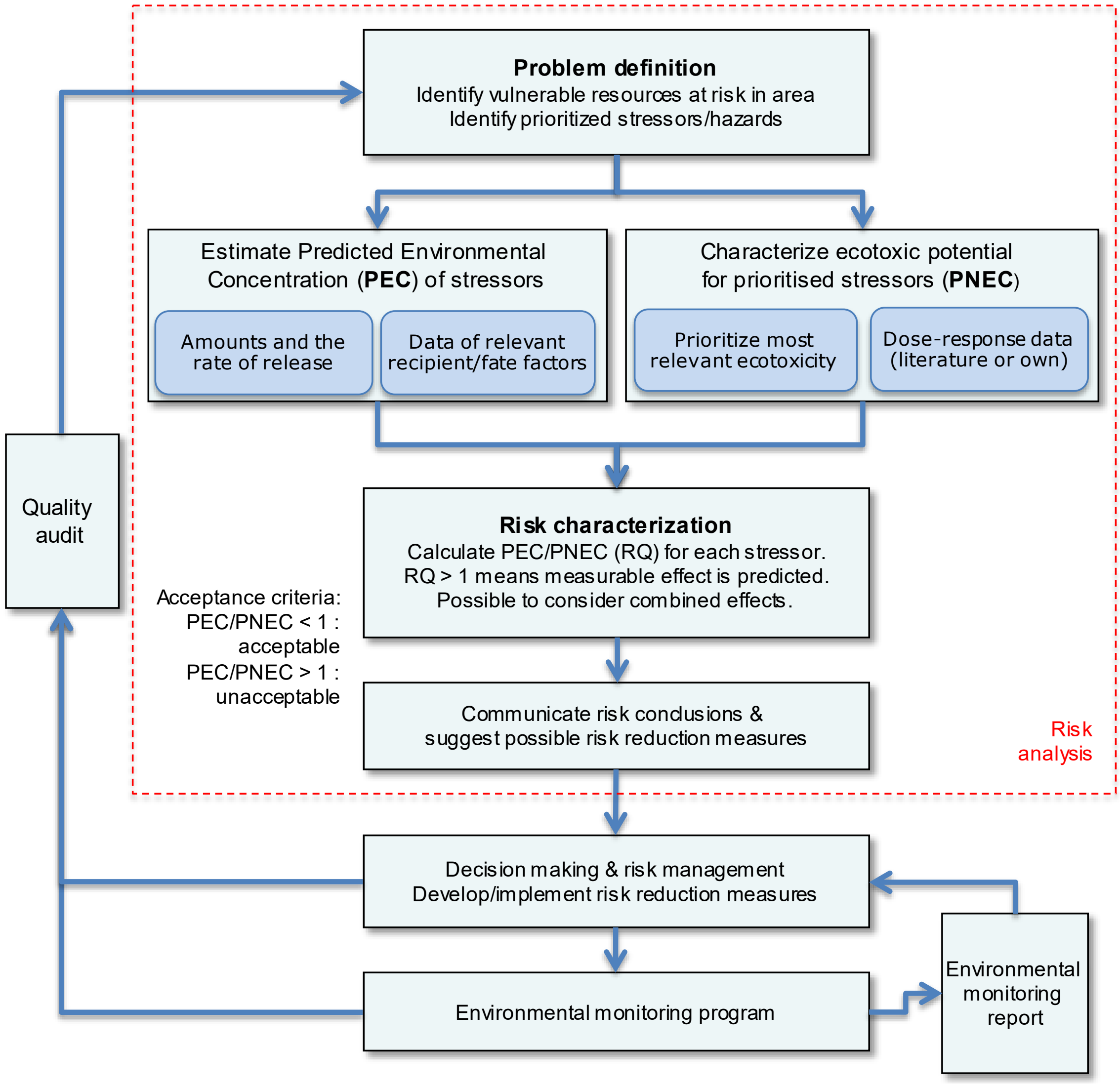

Figure 10: Basic outline of a RQ (PEC/PNEC) based Environmental Risk Assessment (ERA) scheme which is a compatible extended version of the more basic scheme shown by OSPAR (2012).

The second ERA modelling tool, DREAM, was developed for simulating the environmental concentrations of PW substances in the water column (Rye et al., 2008; Reed and Rye, 2011; Rye et al., 2013; Rye et al., 2014). DREAM is a three-dimensional, time-dependent numerical model that computes transport, exposure, dose, and effects in the marine environment (benthic and water column) in connection with management of drilling discharges, produced water and other discharges from offshore operations. According to SINTEF, DREAM can account simultaneously for up to 200 chemical components, with different release profiles for 50 or more different sources with each chemical component in the mixture described by a set of physical, chemical, and toxicological parameters. The model incorporates a complete surface slick model in addition to processes governing pollutant behaviour and fates in the water column (Figure 11). The model can also calculate exposure, uptake, depuration, and effects for fish and zooplankton simultaneously with physical-chemical transport and fates. Beyer et al. (2012) used the DREAM model to assess the exposure level of PW-relevant alkylphenols in the water column in areas downstream PW discharges and to assess the likelihood for these substances to give endocrine disruption (ED) effects in wild fish populations in the North Sea, i.e. measured as induction of female VTG proteins in male fish. The DREAM estimates suggested that even fish populations in the most PW affected region of the North Sea (Tampen area) were highly unlikely to display any measurable ED effect, also when the most conservative exposure data were used.

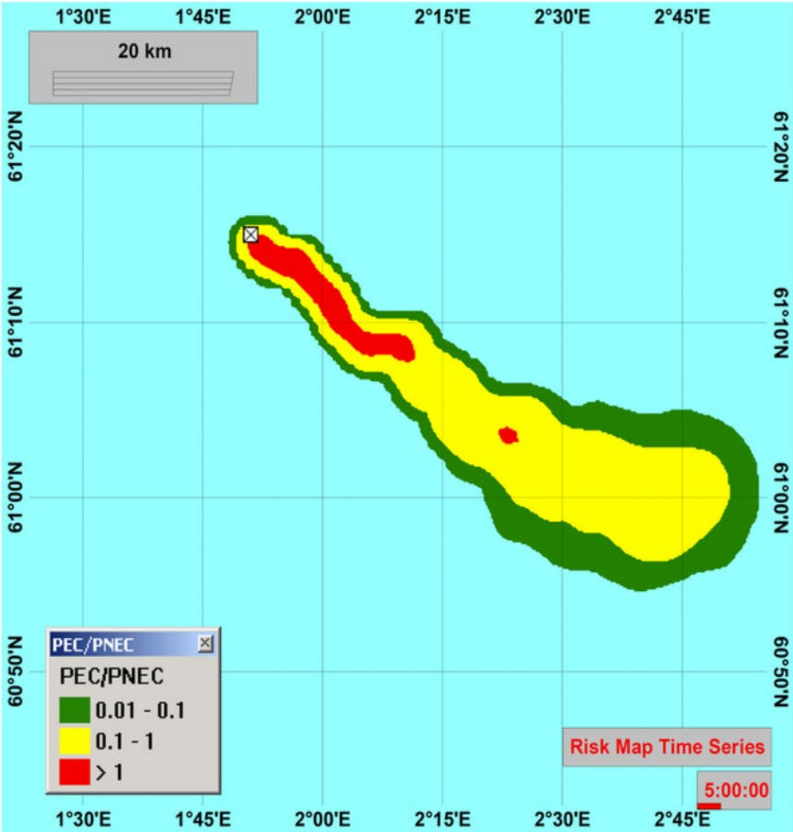

Figure 11: Snapshot of a risk field simulation of a produced water discharge at an offshore petroleum field. The red color indicates expansion of seawater for which the predicted environmental concentration of contaminants yields an estimated risk quotient larger than the value of 1, PEC/PNEC > 1, which in this simulation corresponds to a risk level > 5%. Illustration source: (Rye et al., 2014).

McFarlin et al. (2018) recently conducted a literature review aiming to provide more reliable data on biodegradation rates for various naturally occurring substances in PW discharges, as the DREAM model requires such information. They found that there was a general shortage of good quality data on biodegradation rates. They applied a Q10[1] approach to calculated ultimate biodegradation rates at three relevant temperatures (5, 13, and 20°C) and suggested that these estimates substantially improve the DREAM model results, although most of these rates were extrapolated estimates. The ultimate biodegradation rates found varied considerably either way from what had previously been applied. For example, the biodegradation based half-lives that were calculated for ethylbenzene,

[1] Q10 refers to the Q10 temperature coefficient which is a measure of the rate of change of a biological or chemical system as a consequence of increasing the temperature by 10 °C.

naphthalene, phenanthrene, p-Cresol, and pentylphenol were substantially higher, while the half-lives for chrysene and nonylphenol were lower (McFarlin et al., 2018).

OSPAR has established a list of PNECs for the most common naturally occurring substances in PW discharges (OSPAR, 2014) (see Table 2 previously) to support a risk based approach for management of offshore PW discharges. The list is maintained by OSPAR and updated periodically (e.g. every 5-10 years) or as new relevant scientific data become available. These PNECs can be used in a first-tier risk assessment of PW discharges, i.e. by conducting a direct comparison between the aqueous concentrations and the generic Environmental Quality Standard (EQS), which can be considered as a relatively conservative assessment.

The use of Species Sensitivity Distribution (SSD) probabilistic models (OECD, 1992; Schudoma, 1994; Posthuma et al., 2001), i.e. cumulative probability distributions of toxicity values for multiple species, are more frequently being used in ecological risk assessment, i.e., see Del Signore et al. (2016) for a recent review. In Norway, SSD risk modelling tools have so far almost exclusively been used in the offshore petroleum industry; first for drilling discharge management and for deriving sediment quality guidelines (Leung et al., 2005; Altin et al., 2008; Kwok et al., 2008; Smit et al., 2008a; Smit et al., 2008b), and later for conducting risk assessment of oil in seawater and PW discharges to the water column (Smit et al., 2009; de Hoop et al., 2011; Olsen et al., 2011; Rye et al., 2013; Camus et al., 2015; Sanni et al., 2017a; Sanni et al., 2017c). Briefly explained, an SSD model curve is a constructed representation of the variable toxicity that a certain chemical substance (or a mixture of substances) have across a group of test species, preferably covering several trophic levels. The sensitivity observed within the group is extrapolated to be representative for the sensitivity within the whole ecological community if exposed to the chemical stressor. The aim of an SSD analysis is to determine the chemical concentration level which is protective of most species in the environment, typically for 95% of all species. SSD curves are normally constructed based on data from standard toxicity tests (typically EC_{50} values, LC_{50} values or no-observed-effect concentration (NOECs) values), but more recently also data from non-standardised ecotoxicity biomarker methods are used to construct SSDs. The toxicity test data are visualized as a cumulative distribution function (Figure 12). The reliability of an SSD analysis is highly dependent on the amount of toxicity data that is available hence data shortage can be a limitation. Bejarano et al. (2017) used an SSD approach for comparing the relative sensitivity of Arctic and sub-Arctic species to physically and chemically dispersed oil. They found generally that the Arctic and non-Arctic species had comparable sensitivities to the oil exposures.

It is beyond the scope of the present report to cover in any great details any ongoing scientific debate regarding the adequacy or quality of different ERA tools. Nevertheless, it is important to mention that there have for years been discussions within the ecotoxicology community regarding the adequacy of using no-observed-effect-concentrations (NOEC data) vs the use of ECx values in the calculation of PNEC data. The defenders of NOEC use, e.g. represented by Green et al. (2016) and others, assert that NOEC is a legitimate toxicity metric worthy of continued use by ecotoxicologists, particularly when dealing with “problematic” concentration-response data, whereas the opponents, e.g. as represented by Fox and Landis (2016) and others have the diametral opposite position, that the NOEC statistic is fundamentally flawed as a measure of toxicity, and should therefore not be used in ERA. Herein, we choose not to take any firm position regarding this controversial issue particularly since the use of NOEC is the regulatory accepted approach.

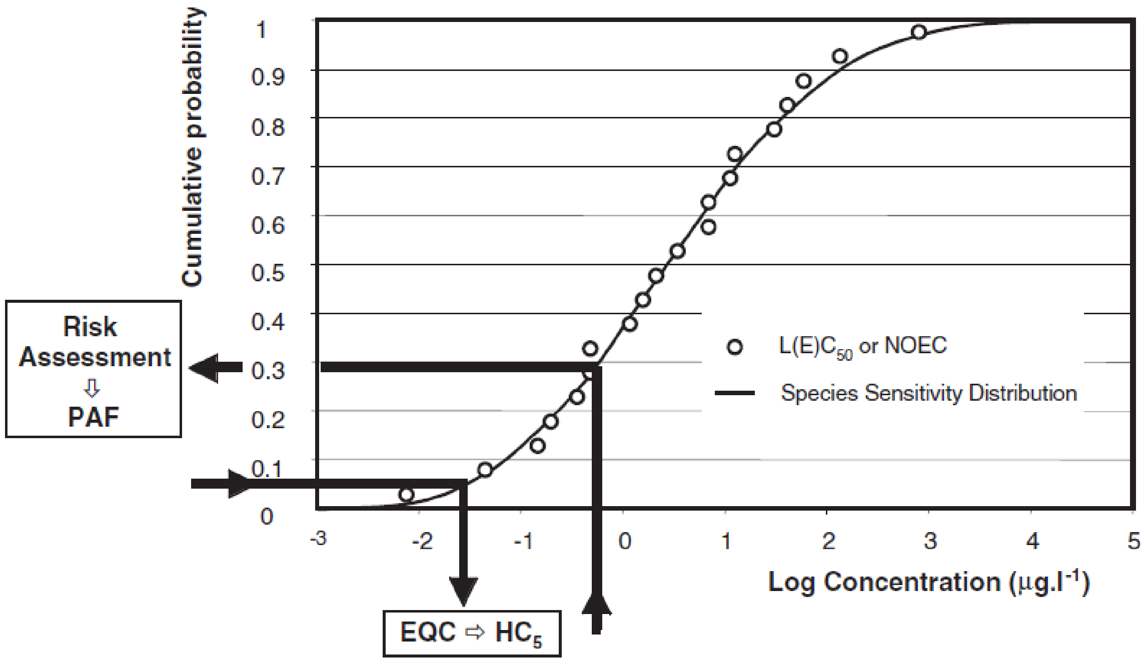

Figure 12: The basic appearance of Species Sensitivity Distributions, expressed as a cumulative distribution function. The dots are toxicity test input data from different test species. The line is the fitted SSD. The units of the Y axis are defined as the potentially affected fraction (PAF), meaning the fraction of test species that are significantly affected at the given stressor exposure concentration. Forward use (arrows from X → Y) yields the PAF as an estimate of risk. Inverse use (arrows from Y → X) yields an environmental quality criteria (EQC) at a certain risk cutoff value, here shown as the hazardous concentration for 5% of the species, HC_5. Illustration source: (Posthuma et al., 2001).

4.2 ERA tools for comparing sensitivity of temperate and Arctic systems

There is little direct data available from ecotoxicity studies on Arctic organisms. However, the SSD approach can be used as a tool for comparing relative sensitivity/responsivity to chemical stressors in Arctic and non-Arctic bioindicator organisms. Olsen et al. (2011), Camus et al. (2015), and a small number of other studies have covered such comparisons. Good comparability of Arctic organisms with temperate organisms would reduce problems of data shortage and thereby greatly facilitate risk assessment studies and environmental risk management efforts under Arctic and sub-Arctic conditions.

Olsen et al. (2011) performed comparative toxicity tests on Arctic and temperate species exposed to 2-methyl naphthalene and used the results to quantify LC_{50} and NOEC data, construct SSDs, and estimate 5% and 50% survival metrics. The tests were performed on 11 Arctic and 6 temperate species encompassing 7 different taxonomic classes. The observed results indicated similar sensitivity to 2-methyl naphthalene for the Arctic and temperate species tested, supporting a conclusion that values of survival metrics for temperate regions are transferrable to the Arctic for the chemical 2-methyl naphthalene, as long as extrapolation techniques are properly applied and uncertainties are taken into consideration.

Camus et al. (2015) performed acute and chronic toxicity testing of an artificial PW mixture to derive effect levels for 6 temperate test species and 6 Arctic species, representing 5 taxonomic groups from low to high trophic levels from Arctic and temperate habitats. They used the experimental results to estimate EC_{50} and NOEC data, which subsequently were used to develop an Arctic species SSD and a temperate species SSD for the artificial PW mixture. By comparing the obtained SSD results, Camus et al. found that the toxicity of the artificial PW mix on Arctic species was more or less comparable to the observed toxicity for the temperate test species. Hence, they concluded that the temperate and Arctic species appeared to be largely comparable under the test conditions studied. However, Camus et al. also advocated to use considerable caution when basing assessments of potential impacts in Arctic species on non-Arctic data, at least until the factual knowledge on these matters improves and the potential differences in the sensitivity of Arctic and non-Arctic systems at higher levels of biological organisation are more thoroughly investigated (ibid.).

Based on acute toxicity data collected from scientific literature, reports, and databases, De Hoop et al. (2011) performed a sensitivity comparison of groups of polar and temperate marine species to crude oil water soluble fractions (WSF), 2-methyl-naphthalene, and parent naphthalene. SSDs that were constructed based on the collected data were used to estimate hazardous concentrations for 5 and 50% of the species. The results of the study suggested that the sensitivity to the chemical stressors did not differ significantly between the groups of polar and temperate species studied.

In ERA, the use of SSDs is considered as a more comprehensive tool than arbitrary Safety Factors to compensate for data-shortage and uncertainty in risk assessment. A safety factor (or assessment factor AF) is a number (e.g. 10, 100, or 1000) which the toxicity test data (e.g. NOEC values) is divided by in order to define a PNEC value that safeguards most of all the species within the ecological community (typically 95% or more). Typically, the selected AF becomes smaller as the amount of relevant NOEC data increases. However, although the use of SSD modelling has gained considerable momentum in ERA of industrial operations and discharges, the approach is often being criticised for being obscure and hard to understand for others than experts in the SSD field e.g. (Forbes and Forbes, 1993; Forbes and Calow, 2002).

4.3 PW issues at offshore fields in the Barents Sea

Zero-discharge has been a key principle for waste management at offshore fields in the Barents Sea, although cleaned PW is not prohibited to be discharged to sea.

The Snøhvit field (operator: Equinor) is developed with only subsea facilities and produces natural gas which is transported to land through a 143 km long pipeline to the LNG plant Melkøya (Hammerfest). The water which follows the pipe stream is separated from the hydrocarbons onshore and cleaned in a special water treatment facility at Melkøya before being released to sea. This discharge was previously investigated and found unlikely to represent any significant risk for the local marine environment under normal operations (Beyer et al., 2013).

The Goliat oil and gas field (operators Eni and Equinor) is located roughly 85 km northwest of Hammerfest in Finnmark and 50 km southeast of the Snøhvit field. It has a production capacity of approximately 110,000 barrels of oil per day, based on 12 production wells. Water injection is used for maintenance of formation pressure and this also enables full reinjection of all the produced water generated during normal operation.

The planned Johan Castberg field (former Skrugard, Havis and Drivis) (Operators: Equinor, Eni and Petoro) is hitherto the northernmost oilfield in the Barents Sea and positioned approximately 100 kilometres north of the Snøhvit-field in the Barents Sea (Figure 13). The field will be developed as an FPSO (floating production storage and offloading) vessel with additional subsea solutions. After the planned production start in 2022, it will generate 30,000 m^3 of produced water per day. The PW will be reinjected into the formation after cleaning. However, according to the field development plan the PW reinjection system will not operate 100% of the time. It is assumed that downtime will be 5% of the time, during which PW will be released (at 20 m depth). The PW discharge will during such periods mainly affect plankton in the water column. The PW will be treated in a 3 stage cleaning process before the release to sea, leading to a predicted dispersed oil content at an average of 15 mg/L (see Table 2, paragraph 2.1) for the predicted content of the cleaned PW). Four different effluent scenarios of this discharge is assessed in the field EIA, all with small predicted impacts on the marine environment (Stephansen et al., 2017; Dahl-Hansen et al., 2017). The worst-case discharge scenario (i.e., discharge scenario O with the PW reinjection system being permanently out of function) will result in a RQ > 1 for the recipient sea stretching a maximum distance of 1300 m from the discharge point (Dahl-Hansen et al., 2017).

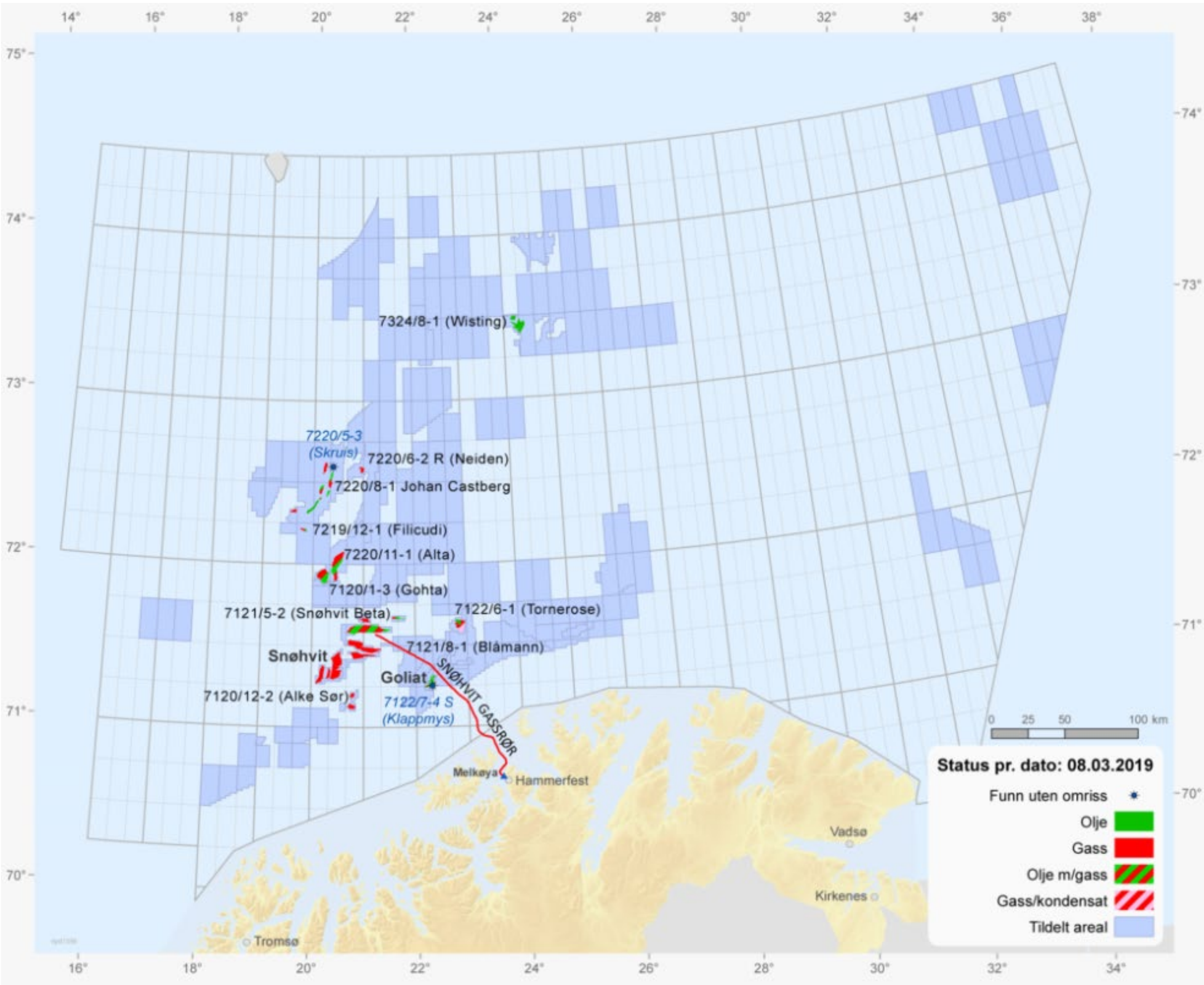

Figure 13: Active production licenses, fields and discoveries in the Barents Sea. Map source: Norwegian Petroleum Directorate

5 Discussion

The key question addressed in this literature review is this: *Are species and biotic systems in the Barents Sea more sensitive to ecotoxic effects of offshore oil and gas PW discharges compared to comparable species and systems in temperate waters?*

To address this issue, we first compiled a review of the Norwegian research about offshore PW ecotoxicology, and second, we made an overview of the ecology characteristics of the Barents Sea area. Next, we examined relevant ecotoxicology effect studies conducted with Barents Sea (and cold water) study organisms, emphasising especially studies that have addressed cold water vs temperate water ecotoxicity sensitivity comparisons. The scientific rationale for this review is to describe the present knowledge status and the remaining uncertainty regarding offshore PW as a possible hazard to the marine ecosystems of the Barents Sea.

In Norway, PW discharges to sea from offshore oil and gas production has been a concern for years due to the view within the research community that these discharges could possibly represent a hazard to the quality status of fishery resources in Norwegian offshore waters. The environmental regulation of the offshore industry operating on the NCS is therefore developed in a way that encourage the industry to establish environmentally safe routines for PW management, e.g. by using water reinjection, optimised water cleaning technologies and avoid using environmentally unfriendly offshore production chemicals. In the Barents Sea area, the fisheries resources are particularly rich and important. Therefore, extra strict regulation that required zero discharge was initially developed for PW management when the first oil and gas developments were launched in the Barents Sea. This strict demand was later eased to a zero adverse discharge policy. However, such a zero adverse discharge policy requires insight into whether the behaviour and ecotoxicity of PW is any different in the cold seas of the north in comparison to on the fields further south in the North Sea and the Norwegian Sea. If offshore PW discharges pose a greater threat to organisms in Barents and Arctic Seas than in temperate seas, then these cold-water systems must somehow be more sensitive and susceptible to PW than temperate systems.

A large body of PW effect studies conducted under laboratory control and with temperate test species have demonstrated that the chemical constituents and mixtures that occur naturally in PW discharges can induce sublethal toxicities in various species of marine organisms, especially when the exposure doses are in the upper end (and above) of the concentration ranges that are generated in the primary plume ultimately downstream from the PW discharges. Furthermore, the performance of ecological risk modelling and field observations from water column monitoring surveys at offshore fields supports a notion that although typical offshore PW discharges on the NCS generate contaminant exposure situations above PNECs (in the closest proximity of discharge points), both the relatively mild nature of the actual toxicological effects that are observed, and the relative small risk volume of the recipient that show concentrations above PNECs for the most sensitive parameters, suggest that there is a negligible risk for offshore PW discharges to cause adverse impacts in wild populations of fish or other pelagic organisms. This perspective seems to be shared by most independent researchers that have investigated the PW issue. However, and as commented in the latter paragraph of chapter 4.1, it can be important to be critical when considering the adequacy and overall quality of the environmental risk data that are developed and used for assessing the environmental risk of chemical substances which are present or are potentially present in offshore PW discharges.

Some studies claim, without providing suitable scientific evidence, that extra strict offshore produced water management is required in Barents Sea/Arctic because these environments are much more sensitive to changes in water quality than more temperate climates, e.g. (Zheng et al., 2016). On the other hand, a relatively small number of research studies from Norway (and elsewhere) have attempted to specifically address the issue of possible sensitivity differences between the cold-water biological systems and temperate-water systems, e.g. (Olsen et al., 2011; Hansen et al., 2014; Camus et al., 2015; Bejarano et al., 2017; Szczybelski et al., 2019). It is interesting to note that these studies seem to reach quite similar conclusions; namely that there seems not to be any clear systematic difference in sensitivity between temperate and cold sea marine species to PW mixtures or key individual PW contaminants. There are tolerant species and more sensitive species in both places. The notion of similar sensitivity across temperate and cold-water systems disagrees with the review study of Zheng et al. (2016) on the current practice and challenges for offshore PW management in harsh/Arctic environments. Zheng et al. (2016) rightly concluded that there still is a shortage of studies relevant for PW management in the Arctic environment, but the authors also claim without apparent documentation that Arctic organisms and ecosystems indeed are much more sensitive and vulnerable than their non-Arctic counterparts against chemical contaminants from PW discharges. Furthermore, they claim that because of this increased sensitivity/vulnerability, oil and gas projects in these areas must adapt to stricter environmental regulations than operations in temperate areas.

The zero-physical-discharge scheme that was implemented for the earliest offshore developments in the Barents Sea was seemingly applied more because of a precautionary principle approach to meet authority requirements and expectations, rather than because of PW effect evidence pointing to the necessity of such extreme protective measures (Hasle et al., 2009). The discussion paper of Knol (2011) who describes and analyzes the application of the precautionary principle in the regulation of offshore discharges in general, and particularly the PW discharges in the Barents Sea, suggests that precautionary action should be scrutinized for its proportionality: how do the benefits of the measure relate to the technological, financial, and environmental costs? She also concludes that the most (cost) effective solutions that could lead to the lowest total environmental harm are not always the solutions that are most politically feasible. In this context, it is key to understand the difference between regulations introduced because of precautionary considerations and regulations introduced because of empirical knowledge of effects (or of risks of effects). Precautionary considerations will often be used in strategic decision making for industrial operations in regions of special uniqueness and potential vulnerability and particularly when appropriate empirical knowledge is lacking. This is the key of 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 the new insight allows this, the regulatory regime can be eased. However, during the period when the 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 his point that although PW discharge mixtures may contain a suite of chemical substances that can be demonstrated to generate ecotoxic responses in test organisms exposed in a controlled manner in the laboratory or by field-caging, the PW discharge will not necessarily cause similar impacts in wild organisms exposed to the real world situation. Ecotoxic effects generated in controlled exposure-effect tests represents a *perceived risk*, whereas ecotoxic effects that materialise in wild organisms/populations are representative of *real risks*.

Variable sensitivity to environmental pollutants for different species and for different life stages of the same species is a well-known phenomenon. But because all species are evolutionarily adapted to their natural environment, an assertion that 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. Research evidence is required for substantiating such assertions. Only a relative limited number of studies have attempted to do this regarding PW in a Barents Sea context, i.e. exposure-effect studies that are relevant for offshore PW discharges under Arctic test conditions and under relevant/realistic exposure scenarios. Realistic exposures are performed via the water phase, at several exposure concentrations, and consisting of key chemical constituents of PWs, artificial mixtures of a few major PW constituents, or real PW mixtures. The Norwegian studies closest to meeting these criteria are: (Geraudie et al., 2014; Camus et al., 2015; Jensen et al., 2016). However, there are some reports that describe interesting findings of unexpected high pollutant sensitivity in Arctic species. For example, Toxværd et al. (2018) recently investigated the effects of pyrene, a normal PAH component in many crude oils, on overwintering *Calanus glacialis*, one of the key species in Arctic shelf areas. Females were exposed from December to March and then transferred to clean water and fed until April. They found that long-term exposure during overwintering conditions reduced the copepod survival and lipid mobilization in a dose-dependent manner at concentrations that previously were considered sublethal for this species. After exposure, strong delayed effects were observed in lipid recovery markers and in production of faecal pellet and eggs. The study concluded that the 50% lethal threshold concentrations were much lower than expected, less than 1/300 of the expected concentration. Effects were found even at the lowest pyrene concentrations of 1 and 10 nM, which have never previously been shown to negatively affect the survival of *C. glacialis*. The 50% effect threshold concentrations for production of pellet and eggs were at less than one tenth of the concentrations previously found. The study provides novel insight into the effects of oil contamination in *Calanus glacialis* during winter, information that has largely been unknown, but it can also be argued that the unexpectedly strong effect-sensitivity observed in the study may warrant conduction of at least one replicate, confirmatory effect study.

Given the assumption of *no systematic differences* between temperate- and cold-water systems, the plans, demands and regulations of management systems for PW discharges at existing and potential production fields in the Arctic region can be done, to a larger degree than expected, based on existing knowledge. However, the Barents Sea is a very heterogenous area regarding several key environmental conditions. The southernmost part of the Barents Sea, south and west of the polar front, resembles much of the temperate ecosystem conditions found further south, whereas the Northern Barents Sea (north and east of the polar front) resembles more the extreme cold and harsh conditions of the Arctic ocean. The Snøhvit and Goliat fields are both located in the southern Barents Sea. Other fields, such as the Johan Castberg and Wisting fields, are located much further north, closer to the polar front. How offshore PW discharges may behave and affect the ecosystem in the polar front zone is still an issue of many unknowns. The ecosystems located north/east of the polar front are representative of true Arctic conditions, with sea surface temperatures always being below 0°C and with ice-cover for large parts of the year. In contrast, the systems located south and west of the polar front resembles more sub-Arctic to boreal conditions, with sea surface temperature always warmer than 0°C, and with no ice-cover during winter. The flora and fauna in the southern/western region also have a high and increasing representation of boreal species. If there were major differences in sensitivity/vulnerability to PW in Barents Sea species, the differences would primarily be expected to occur between high Arctic species and temperate/boreal species. The notion of no systematic sensitivity differences between Arctic/sub-Arctic organisms and temperate organisms would mean that the knowledge that has been accumulated from PW relevant ecotoxicity testing with temperate species could be employed also in an Arctic context. As outlined in Chapter 4.2 several research studies have investigated possible differences between Arctic and non-Arctic species in sensitivity/vulnerability against PW and largely failed to identify any systematic differences. Actually, some studies suggest a slightly higher tolerance in Arctic organisms, but none the opposite. Camus et al. (2015) compared PW toxicity in Arctic and temperate species. Acute and chronic toxicity of artificial PW for six Arctic and six temperate species was experimentally tested and evaluated and the hazardous concentrations affecting 5% and 50% of the species were calculated from SSD curves. They concluded that the hazardous concentrations derived from individual species' toxicity data of temperate and Arctic species were comparable, although responses at higher levels of biological organization should be studied to reveal potential differences in sensitivities to produced water between Arctic and non-Arctic ecosystems.

It is questioned which study organisms are the most suitable for investigating species sensitivity differences of comparable species in temperate, boreal and Arctic habitat conditions. The copepods such as the closely related species *Calanus finmarchicus, C. hyperboreus,* and *C. glacialis* and *Acartia tonsa* are all key zooplankton species, and they have been much used for sensitivity comparison studies by Norwegian researchers, e.g. (Hansen et al., 2011; Hansen et al., 2013a; Hansen et al., 2013b; Hansen et al., 2014). However, these studies have not resulted in the Arctic species being identified as particularly sensitive when compared to temperate species. Rather contrary, the temperate species *Acartia tonsa* has been found to be the most sensitive of these copepods in acute toxicity testing and the one that may be used as to provide conservative effect estimates for pollutant stressors (Hansen et al., 2014).

Other crustacean study species that have been tested for investigating sensitivity of Arctic systems versus boreal and temperate systems linked to offshore industry activity include the gammaridean amphipods, Northern krill (*Meganyctiphanes norvegica*), and deep-water shrimps in the Pandalidae family. Several PROOFNY studies have addressed effects of oil or other anthropogenic contaminants on the Arctic ice amphipod *Gammarus wilkitzkii* (Haukas et al., 2007; Camus and Olsen, 2008; Olsen et al., 2008; Hatlen et al., 2009; Krapp et al., 2009). This amphipod species inhabits the subsurface of sea ice and forms an important link between the lower and higher trophic level in the Arctic ice edge community (Lonne and Gabrielsen, 1992; Werner et al., 2002). Arnberg et al. (2017) studied the effect of chronic oil exposure (0.01 - 0.1 mg/L) on development and feeding of early life stages of the Northern krill, but found no significant effects on egg respiration, hatching success, development, length and larval survival due to the treatments. Moodley et al. (2018) studied the effect of crude oil on adult northern krill collected during three seasons and found increased digestive gland pathologies (enhanced apoptosis and pathology of digestive tubules) in the oil treatments (27-80%) although there was no effect on survival after 2 weeks of exposure. Bechmann et al. (2010) demonstrated the use of adults and larvae of the northern shrimp (*Pandalus borealis*) to test for effects of different oil exposure scenarios at different temperature conditions (5°C and 8°C). The study found a lowered larval mortality in the cold-water test group whereas PAH accumulation and mortality in adults did not differ notably. In summary, there is still much research needed for identifying the most suitable test-species and effect-endpoints within zooplankton and larger crustaceans for future research on PW and other oil industry issues in the Barents Sea. So far, the majority of the research seems to suggest that these species are relatively tolerant to oil hydrocarbons and other PW relevant exposures.

A trait that is typical for the Arctic ecosystem is a trophic energy flux strongly based on lipid transfer. Arctic marine organisms have high lipid reserves that varies with season and accumulate lipids during the feeding period (spring/summer), both to build up a large nutrition reserve necessary for surviving the winter and (for warm blooded animals) as insulation against low air and sea temperatures. Lipophilic toxicants accumulate most strongly in lipid rich tissues. Hence one might expect a more pronounced seasonal variability in accumulation of, and sensitivity to, lipophilic toxicants in Arctic (and other polar) species than in temperate organisms. Earlier studies have demonstrated the influence of lipid content on hydrocarbon uptake and retention, although the coupling is not straightforward. Harris et al. (1977) found by a series of experiments with uptake of 14C naphthalene in seven species of temperate copepods that hydrocarbon uptake and retention rates were positively correlated with several measures of biomass (dry weight, ash free dry weight, body size, body surface area and total lipid content), and the best correlation was with lipid content. Furthermore, there were gender differences in hydrocarbon turnover which indicated that lipid composition might play a significant role in hydrocarbon retention. However, it was also seen that uptake and retention was dependent on e.g. the route of entry (dietary or dissolved hydrocarbon), and the nutritional status (starved or fed) of the test animals. They also measured a rapid loss of hydrocarbons during the first days of depuration in clean water, but sometimes as much as 10% of the initial body burden was retained for up to 34 days after the exposure. In the same experiments it was estimated that for *Calanus helgolandicus* around 40% of the hydrocarbon ingested were expelled in faeces particles. Sinking of copepod faecal pellets has by several authors been suggested as a transport route of oil hydrocarbons to the benthos, although partitioning to sinking particulate organic matter (POM) probably is a far more important route. Nevertheless, the overall impression is that, although a large fraction of ingested hydrocarbon in copepods may be expelled as faeces or be depurated when exposure ceases, a significant amount may be present for a long time and will thus become available to a higher trophic level. It is also likely, although not yet demonstrated, that the trophic "lipid wave" in Arctic waters will enhance the trophic transfer of lipophilic contaminants relative to temperate food chains. Yet, even though biological sensitivity to toxicants may be the same in Arctic and temperate species, one cannot exclude the possibility that differences caused by superior ecosystem factors, such as ecosystem complexity, seasonal characteristics, spatial distribution of populations and communities, population-interaction, and spatiotemporal behaviour of discharges and pollutants, may differ in such a way that one region becomes more vulnerable than the other. Such factors may possibly, as pointed out by Hjermann et al. (2007), be too stochastic to enable useful prediction of the overall impact of a discharge. For example, what fraction of the total population of a species that must be impacted by an operational PW effluent to elicit significant population damage may not be a constant figure and clearly a matter of conjecture.

In Norway, fish, and in particular the Atlantic cod, has always been most in focus for ecotoxicological studies of PW, or other oil industry pollution scenarios. For effect studies with fish under polar conditions, the polar cod (*Boreogadus saida*) has attracted much attention in recent years, but also the use of cold-water adapted Atlantic cod provides good opportunities as test models. The results of this research, which for the most are summarized herein, span most scientific disciplines, and the level of attention is expected to continue due to the growing concern for how fish populations in the Barents Sea and elsewhere on the NCS will manage to cope with increased challenges associated with climatic changes and other anthropogenic pressures. Regarding PW discharges, it has not yet been feasible to prove any significant effects on fish population and community levels. Most of the laboratory and field studies support the conclusion that significant biological effects on pelagic organisms by constituents in PW are limited to a small impact zone represented by the diluting PW plume a few hundred meters to a few km downstream from the PW discharge point, all depending on the size of the discharge, various conditions of the water column and the presence of organisms. Most fish species have a much wider distribution than these PW impact zones and their exposure to PW constituents at noteworthy levels will expectedly be short and transient. Hence, for a significant impact to occur in a fish population either harmful exposure to PW must be sufficiently wide scale or the population influence from locally affected individuals must 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 interactions, density dependent functions, other stressors, and the complex and dynamic physical conditions in the offshore pelagic ecosystem (Hjermann et al., 2007).

Research and monitoring demonstrate that fish can take up contaminants from a diluted PW plume, and respond to this exposure, but there is no empiric backing for claiming that the ecotoxicological effects of PW discharges in offshore fish populations have been underestimated. The technical sophistication of research and monitoring tools (biomarkers, risk assessment models, etc.) that are accessible for effect studies of PW has improved greatly. This has opened new avenues for assessing possible adverse influences of PW contaminants in fish and other offshore organisms at low environmental concentrations. Assumptions of possible extra sensitivity of organisms and ecosystems in the Barents Sea led initially to the implementation of a strict zero-physical-discharge policy for PW discharges, but this has later been edited to a zero-adverse-discharge policy. As more and more ecotoxicity research and test data for cold-water organisms have become available, the research consensus suggests that Arctic organisms are not systematically more ecotoxicologically sensitive than temperate and boreal systems and organisms. Robust/tolerant and sensitive/fragile species are found in all systems. Increased sensitivity of certain species seems rather to be associated with characteristics and properties of the organisms which not necessarily are linked to their Arctic distribution. One illustrative example is the sticky haddock egg which has the capacity to adsorb far more microscopic oil droplets (and hydrophobic contaminants) than the non-sticky Atlantic cod egg. As discussed herein, this can make a haddock embryo to be more exposed to organic ecotoxicants than a cod embryo although they are present in the same water. However, the overall ecology of the Barents Sea region and Arctic seas is special. The Arctic food chains are generally shorter and simpler than in temperate waters, and ecologically structural elements such as the polar front divides the ocean in transition zones which for some are characterised by extreme high production and an obvious ecological vulnerability (for example the ice edge). These special conditions will in various ways influence the behaviours of contaminants, e.g. dispersion, dissolution, and chemical degradation, and secondly have relevance for how organisms are exposed and will respond. Still, such particularities of the cold-water systems seem not to render them to be systematically more sensitive to adverse effects of PW contamination. However, there is an urgent need for more knowledge in this field of study. The amount of ecotoxicity data on Artic and cold-water species is still limited and only a relatively small number of PW effect studies provide data for sensitivity comparisons of Barents Sea and non-Barents Sea organisms/systems.

6 Key research needs

Sets of biomarkers in fish and other marine species have been developed as tools for evaluating possible sublethal effects to PW-relevant exposures in the laboratory or in offshore surveys in various distances from PW discharges at oil and gas production platforms. Yet there is a continued need for more and better method validation data of the existing effect detection methods as well as more sensitive and fitness-relevant methods to assess possible impacts of offshore PW discharges at different levels of ecological organisation (ranging from the molecular levels to the health of ecological communities), and in a manner that is environmentally realistic. It is also important to keep the methodology updated, user-friendly, and as simple as possible. Environmental risk assessment tools have been demonstrated as robust methodology for assessing the likelihood of PW associated impacts in wild fish populations in the North Sea. More knowledge about the ecotoxicological sensitivity of species, populations and ecological communities in the Barents Sea and other Arctic areas must be developed in order to adapt the risk assessment procedures to cold-water environments. There is still a general shortage of information concerning possible effects of oil-related compounds at all levels of biological organisation in cold-water marine ecosystems. However, the relatively limited research that so far have been performed tend to suggest that the differences in ecotoxicological sensitivity between marine species and systems in the Arctic versus in temperate and boreal seas are smaller than many have expected. There are more-sensitive and less-sensitive species in all ecosystems, and the Arctic system has so far not been singled out as significantly and systematically more sensitive. Further research is needed to broaden and deepen this field of knowledge. The research should particularly address sub-lethal and long-term impacts of environmentally realistic low-dose chronic exposures to PW associated contaminant mixtures as well as to continue to increase our basic ecosystem understanding of the various areas within the Barents Sea ecoregion.

7 Summary and conclusions

The marine ecosystems of the Barents Sea and the Arctic in general are, most likely, at great risk due to mounting pressures associated with global warming and increased human activities. To ensure protection of these important ecosystems, minimization of pollution inputs and other manmade perturbations is warranted. In the late 1990s, stricter environmental regulations were suggested for future offshore oil and gas produced at the Norwegian Continental Shelf, and particularly for future field developments in the Barents Sea region. One key measure for improving the environmental safety was to introduce a “zero discharge” goal for PW and other wastes produced during normal operations. However, already from the start the term “zero discharge” became subject of unclarity, discussions and interpretations. When regarding oil and gas activities in Barents Sea a key question is whether biological species and ecosystems that exist in these cold seas are more sensitive to toxic substances in PW discharges compared species and systems from more temperate waters. As discussed herein, ecotoxicological research and monitoring have yet to confirm such an assertion that cold-water species/systems systematically are more sensitive. Some species within both categories (Arctic and non-Arctic) appear to be more sensitive, but this extra sensitivity seems more to be due to certain physiological characteristics and less to an Arctic or cold-water distribution. For example, species that inhabit the cold waters of the high north are often high in lipid content. But this characteristic trait seems more often to increase the tolerance against toxic substances that may occur in a PW discharge. However, at the same time, a high fat content will increase the ability of the species to act as a transport vehicle for lipophilic PW contaminants upwards in the food chain.

The present report attempts to summarize many years of research and monitoring on assessing the possible environmental effect of offshore PW discharges at oil and gas fields on the Norwegian continental shelf. Both laboratory experiments and field surveys suggest that detectable exposure as well as some quite modest impact responses can be induced in fish and mussels when these are confirmable exposed to PW, e.g. when they are kept within the PW plume in the water column relatively close (some hundred meters to a few km) downstream of the PW discharge point. Similar impacts have yet not been demonstrated in wild fish when they are collected in PW influenced areas. Similarly, impacts were modest in laboratory exposed organisms when these where exposed to PW associated contaminations at high field-realistic concentrations. The typical modest responses found suggest that the overall risk for PW discharges to induce adverse impact in populations of wild fish and possibly other pelagic organisms is low. A low-risk situation is also widely supported from studies using environmental risk modelling procedures for environmental impact and risk assessment of PW discharges offshore. Increased efficiency of PW treatment systems, continued phase out of non-green offshore production chemicals, increased use of PW reinjection operations, and development of discharge free field solutions, are all processes that tend to increase the environmental safety level of PW management routines on the Norwegian oil and gas fields.

The research community share a deep concern regarding the future survival of the cold-sea ecosystems in the Barents Sea and Arctic region. There are multiple signs suggesting that major ecosystem changes are ongoing in the whole Arctic region driven predominantly by processes associated to regional and global warming. In that context, all Arctic species are vulnerable simply because there is a clear limit to how far north they can move to avoid rises in air and sea temperatures and declines in sea ice extent. Increased competition from southerly species migrating northward is expected to even out the species diversity composition differences between the western and southern Barents Sea and the shelf areas further south, and such an immigration trend can be expected to make high north species even more vulnerable. These factors clearly suggest the rationale for continuing to seek better knowledge about and to lessen the anthropogenic pressures on the Barents Sea and the Arctic ecosystems. The success of the process will depend on our ability to identify the key man-made ecological perturbations in these regions, and to find efficient management solutions for these problems. Nevertheless, the possible ecological risks associated to offshore PW discharges is most probably not larger at offshore fields of the Barents Sea than elsewhere on the NCS. And by all practical means, this risk is most likely negligible when compared to the much bigger threat to these important ecosystems caused by global warming.

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