

## Environmental Policy & Regulation

# Modeling of Environmental Fate and Effects of Oil Leakages from Abandoned Subsea Wells Using an Environmental Impact Factor Tool

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### ABSTRACT

Potential environmental consequences of oil leakages (i.e., continuous uncontrolled releases at low flow rate over a long period of time) need to be taken into consideration in the ongoing development of plug and abandonment (P&A) activities on the Norwegian continental shelf. Regulations of P&A wells employ a “zero leakage” target; however, environmental risk monitoring strategies for permanent abandonment are not yet in place. Predicting and estimating the consequences of adverse environmental impacts through a modeling approach can play a key role in evaluating and monitoring environmental risk. In this paper, we present a modeling study of the fate and effects of an oil leakage from abandoned wells using a theoretical scenario on the Norwegian continental shelf. Environmental impact factors (EIFs) derived from the Dose related Risk and Effect Assessment Model (DREAM), previously designed to characterize the effects of produced water discharges, were used to assess impacts of leakages from abandoned wells. Exposure assessments for the EIFs were modified to include specific hydrocarbon contributions derived from different sized oil droplets from the leakages. Because DREAM is not generally used for chronic low-rate oil releases, an update of the database with chronic predicted no-effect concentrations, as input data for effects modeling, was conducted. In general, EIFs became stable after simulations of 30 d. The area from the release site and up to a few hundred meters southward had the most locations of high impact. Chronic exposure and effects on organisms potentially occurred as a steady-state effect over a long period. Risks, at which more than 95% of species will be negatively affected, appeared surrounding the release site, indicating a need for mitigation measures. These results show that the EIF tool can be used for risk management and P&A regulation by identifying potentially harmful leakages. *Integr Environ Assess Manag* 2021;17:626–638. © 2021 The Authors. *Integrated Environmental Assessment and Management* published by Wiley Periodicals LLC on behalf of Society of Environmental Toxicology & Chemistry (SETAC)

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### INTRODUCTION

Abandoned offshore wells have uncertain interactions with the seafloor, the water column, and the sea surface in the marine environment (DNV GL 2016). Oil leakages from abandoned wells may be comparable to natural seepages in that the discharge rate is relatively low and the exposure to biota is chronic. If undetected, the persistence is reckoned in terms of years. Certainly, sudden, large releases could occur, but likely under conditions where well barriers are severely degraded or insufficiently constructed. Indicative

average gas (methane) flux rates from plugged onshore wells in the Netherlands; Pennsylvania, USA; and the UK were reported as 443, 264, and 41 g CH<sub>4</sub> per day (Townsend-Small et al. 2016; Schout et al. 2019). Oil rates would require a large buoyancy of gas bubbles to be in the same order of magnitude for equivalent cases.

The leaked oil, when entering the ocean from the seabed in small amounts, can be gradually weathered through a variety of physical, chemical, and biological processes (NRC 2003). This leads to a chronic and/or long-term exposure of marine organisms to the oil and its chemical components. Oil and its components such as polycyclic aromatic hydrocarbons (PAHs) and *n*-alkanes have been shown to adversely affect marine ecosystems (Abha and Singh 2012; Almeda et al. 2013; Pampanin and Sydnes 2013). Natural oil seeps, which have been leaking continuously at very low rates for thousands of years in many

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parts of the ocean, have been studied to understand the potential effects of leaked oil on marine organisms (NRC 2003). However, abandoned wells are potentially an additional disturbance to the oceanic environment, despite efforts of prevention by the industry, and therefore need further attention.

As of 2019, it is estimated that about 800 of 3800 petroleum wells have been abandoned on the Norwegian continental shelf (NCS), and about 360 wells are reaching the end of their productive lifespan and will be abandoned by 2025 (Oil & Gas UK 2016). Approximately 20% to 30% of wells on the NCS have experienced at least 1 documented leak during their lifetime (Randhol and Cerasi 2009), and 38% of the temporarily abandoned wells have had some degree of leakage (Totland 2011). However, it is not known if permanently plugged wells on the NCS continue to leak. Failures of well integrity and barriers, such as a defect in sealing of caps, or active faults and/or fractures can provide potential for slow, localized fluids to move up the well to the seabed (Davies et al. 2014).

The “zero harmful discharge” regime was implemented on the NCS in 1996 with the goal of “zero environmental harmful discharges” for the marine environment (Smit et al. 2011). The plug and abandonment (P&A) activities on the NCS carried out in accordance with the NORSOK (2013) standard D-010 are thus required to reach the goal of zero leakage for permanent abandonment. The standard requires 2 cross-sectional plugs of at least 50 m set in competent formation, with an environmental plug close to the surface. As mentioned, there is no evidence that these requirements are insufficient. Rather, there is a move to take a more risk-based approach to assess the requirements. Because even verified cement may possibly have or develop small fissures leading to leakage, it is worthwhile to consider the environmental consequences if such pathways to the marine environment exist. Companies operating on the NCS are currently conducting environmental monitoring of operating wells every 3 y (Norwegian Environment Agency 2015). However, the NORSOK Standard D-010 does not require monitoring after the well is permanently abandoned. According to the Norwegian Petroleum Act, operators on the NCS have been asked to issue a decommissioning plan, which includes an environmental risk assessment (ERA) of the sites after production has ceased (NORSOK 2013; Oil & Gas UK 2016). Regulatory requirements for the ERA include the International Organization for Standardization 31000 standard (ISO 2018), NORSOK standard Z-013 (2010), PSA regulations, as well as regulations from the Oslo and Paris Conventions (OSPAR) (Vinnem and Røed 2014). A comprehensive ERA requires an integration of laboratory and monitoring data with fate and exposure model predictions (EC 2003).

A general modeling system includes a physical fate or oil weathering model module, a biological exposure and effects module, and a series of environmental scenarios to simulate future conditions at one or more abandoned subsea well locations. The physical fate module describes drift trajectories or advection of the spilled or leaked oil. It

includes dynamic weathering processes, such as spreading, evaporation, dispersion, emulsification, biodegradation, and sedimentation in the marine environment by wind, currents, and waves (Fingas 2011). The biological exposure and effects module estimates the water volume or areas exposed above a selected effects threshold and is based upon estimated concentrations of the chemical constituents of the oil that may be available to aquatic biota. Essentially, biological exposure and effect modules rely on acute or chronic effect threshold data and compare those values to concentrations derived from the physical fate module (Xiong et al. 2000). The predictions based on these modules and used in different environmental scenarios typically include the physicochemical state of the oil and the potential effects of the oil on exposed organisms at particular distances from the point of oil release. Oil spill models commonly used at present in different parts of the world such as SIMAP, OilMap, SEATRACK WEB, Oil Spill Contingency and Response (OSCAR), Dose related Risk and Effect Assessment Model (DREAM), MOHID, and Delft3D generally follow this approach to oil spill modeling. However, these models are not generally used to assess chronic low-rate oil leakages such as are possibly encountered after P&A of subsea production or exploration oil well.

With regard to evaluating environmental risks, oil spill models are often linked to human or ecological risk assessment with the aim to understand the consequences of spilled oil on a particular species or more broadly on the ecosystem in which the spill occurs (Rausand and Utne 2009). Such work typically includes quantification of the likelihood and severity of biological effects and comparison with risk criteria to evaluate whether the consequences of the spilled oil to a species or ecosystem is within acceptable limits (Smit et al. 2005).

Risk assessments for petroleum activities conducted on the NCS are guided by OSPAR (2012) guidelines that provide methods for data collection, hazard assessment, exposure assessment, and risk characterization, which are then linked to further steps involving risk management and monitoring plans. Regarding the exposure assessment, the predicted environmental concentration (PEC) is an indication of the expected or actual concentration of a compound in the environment (Von der Ohe and De Zwart 2013). In terms of the effect assessment, the predicted no-effect concentration (PNEC) is the threshold value for environmental compartments or an environmentally safe concentration of a compound, at which no effects on the biota are expected and below which adverse effects will most likely not occur (Von der Ohe and De Zwart 2013; Aven 2015).

A PEC/PNEC approach (also known as a hazard quotient) indicated in current OSPAR guidelines is based on worst-case assumptions for sensitivity and exposure. The exposure represented by the PEC can be obtained by monitoring data or by estimations using environmental fate models. The toxicity threshold (i.e., PNEC) represents the sensitivity of the ecosystem and is usually derived from standardized toxicity tests (EC 2003). A chronic PNEC can be calculated

by dividing the lowest no observed effect concentration (NOEC) value by an appropriate assessment factor (AF), according to the European technical guidance document (EU-TGD; EC 2003). An AF refers to a numerical adjustment used to estimate the actual substance exposure from experimentally determined dose–response relationships.

The risk assessment steps involved in the PEC/PNEC approach regarding leakages from abandoned wells are presented in Supplemental Data Figure S1. Briefly, PNECs are compared to PEC, and if the ratio PEC/PNEC is below or equal to 1, then the risk associated with oil leakage is assumed to be acceptable (EC 2003). On the other hand, the risk associated with oil leakage may be assumed to be significant if the ratio PEC/PNEC is above 1, at which more than 5% of species will be affected (Rye et al. 2006). The PEC/PNEC approach, using scientific information relevant to the ecosystem at hand, provides valuable information regarding the potential environmental risk of damage to marine organisms caused by any release or discharge of oil.

The present paper evaluates an approach for modeling biological exposure and effects of leaked crude oil from abandoned wells. The results may be utilized by regulators and stakeholders for petroleum management decisions. Our model employed a theoretical oil leak in the North Sea on the NCS as a case study. The objective was to estimate chronic biological exposure and effects of continuous low-rate oil discharges simulating a leak. The case study is based upon oil releases that are too small to be detected or controlled. Metrics that are usually considered for oil spills, such as oil ashore and area of oil on the surface, were not considered in this case. Because of the slow release scenario, the exposure scenario focused on oil-derived organic hydrocarbons in the aqueous phase and did not consider sediment deposits or droplet sizes that would be trapped in the sediment.

## METHODS

With respect to continuous low-rate oil leakages in NCS conditions, environmental impact factors (EIFs) were calculated using DREAM version 10.0.1 (Rye et al. 2008), considering a theoretical oil leak case study at an oil field, with a light crude oil type from the North Sea.

The DREAM has both a physical fate module and a biological effect module, and an oil database based on laboratory experiments (Kilesso et al. 2014). Table 1 shows DREAM's simulation capacity in different release conditions and includes physical fate, biological exposure, and effects. The DREAM has been previously used for EIF calculations, where the EIF represents the risk in a volume unit equal to 100 m (length) × 100 m (width) × 10 m (height), and the EIF water volume is also adjusted upward by a factor of 2 for those compounds that are subjected to small biodegradation combined with a large bioaccumulation (Rye et al. 2006, 2008). In compliance with the OSPAR guideline for an ERA (OSPAR 2012), this model was applied to assess potential chemical hazards, as well as exposure to hazardous substances, and to characterize environmental risks of discharges.

**Table 1.** Overview of simulation capacity of DREAM<sup>a</sup> by SINTEF

	Open-source code	DREAM
Release modes	Instantaneous release	X
	Continuous release	X
	Surface spill	X
	Subsurface spill	X
Physical fate	Advection	X
	Diffusion	X
	Backtracking	—
	Spreading	X
	Evaporation	X
	Emulsification	X
	Dispersion	X
	Dissolution	X
	Sedimentation	X
	Biodegradation	X
Biological exposure and effects	Interaction with shoreline and coast	X
	Areas or volume exposed above threshold	X
	Dose that aquatic biota exposed to	X
	Exposure of algae, fish eggs, or larvae	X
	Contact to wildlife (birds, mammals)	—
	Acute effects or expected mortality	X
	Chronic effects	X

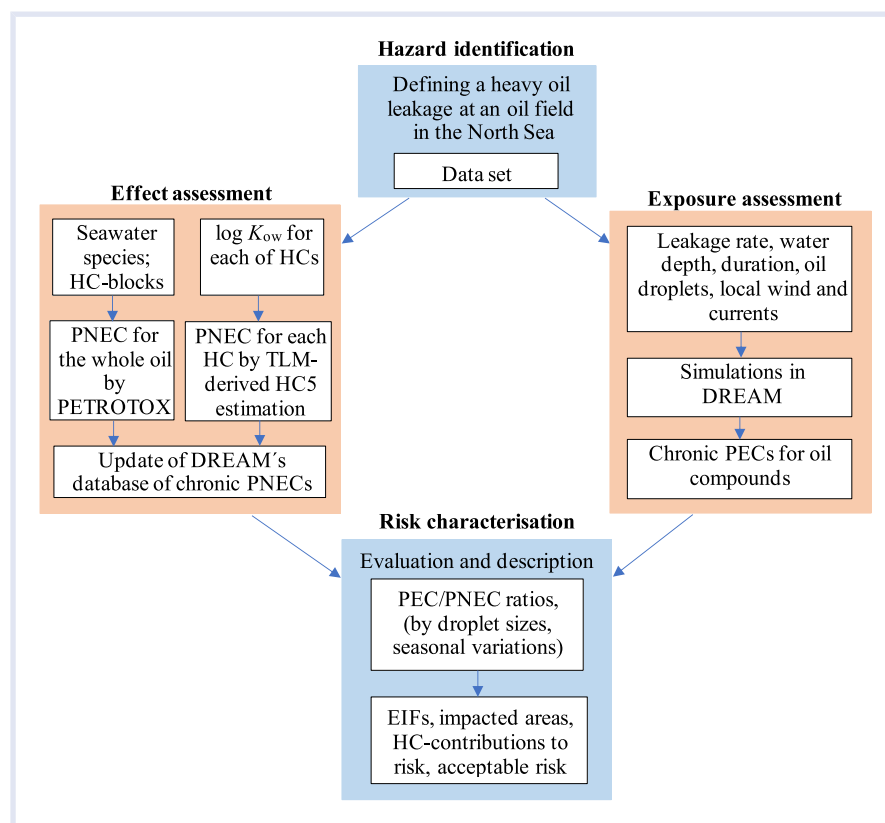
<sup>a</sup> Rye et al. 2008.

DREAM = Dose related Risk and Effect Assessment Model; X = the model can simulate the process; — = no simulation or unknown information.

### Update of DREAM's database of chronic PNEC values

An update of the database of PNEC values in DREAM was needed because the model was originally developed to calculate only PECs for produced water discharges into the sea. There had not been a focus on providing chronic PNEC values for the oil components.

Chronic PNECs are usually derived by toxicological dose descriptors (e.g., NOEC) and an AF (EC 2003). An AF refers to a numerical adjustment that is used to extrapolate from laboratory single-species toxicity data to multispecies ecosystem effects. When a large data set for different taxonomic groups is available, PNECs can be derived by a hazardous concentration threshold for 5% of species (HC5) derived from statistical extrapolation of a species sensitivity distribution (SSD) to estimate fitness of all species (EC 2003).



**Figure 1.** The modeling and environmental impact factor (EIF) workflow in environmental risk assessment for oil leakages. DREAM = Dose related Risk and Effect Assessment Model; HCs = hydrocarbon compounds;  $K_{ow}$  = the octanol–water partition coefficient for the substance; PEC = predicted environmental concentration; PNEC = predicted no-effect concentration; TLM-derived HC5 = Target Lipid Model derived the 5<sup>th</sup> percentile hazardous concentration.

Thus, in the present study, a chronic PNEC value for crude oil was derived using chronic toxicity data NOECs from the model PETROTOX (version 3.06), which is recommended by the European Petroleum Refiners Association as an effective tool in the derivation of PNECs (EC 2003; Concawe 2011). Chronic PNECs for each oil component were obtained by an estimation of hazard concentration affecting 95% of targeted species (HC5) based on the Target Lipid Model (or TLM-derived HC5 estimation) (Di Toro et al. 2000). These PNECs then became a source of data input for running DREAM to produce values of EIF and establish risk maps for the corresponding location (Figure 1).

#### PETROTOX estimates

The PNEC value for the whole oil was derived from hydrocarbon groups, where similar oil components with similar properties were allocated in the same hydrocarbon blocks. Initial and ending boiling point ranges and fraction of aliphatic or aromatic weights of each hydrocarbon block were used as input data for the solubility calculation (Supplemental Data Table 3). The computed dissolved hydrocarbon concentrations were then used to calculate aquatic toxicity. Certain species were selected and coupled with an arbitrary particulate organic carbon (POC) concentration of 2.0 mg/L to drive the bioavailability model (Concawe 2011). The toxicity of a mixture of chemicals was driven by the sum of the toxicity of individual components.

In our study, 25 oil hydrocarbon blocks (Supplemental Data Table 3) were designed at different boiling point ranges and fraction of weights (wt %). In addition, 16 seawater species from 4 trophic levels (fish, alga, crustacean, and polychaetes) were theoretically tested in PETROTOX to obtain chronic ecotoxicological values (Supplemental Data Table 4).

Predicted chronic ecotoxicological values of NOEC calculated by PETROTOX for 16 available seawater species showed that *Rhepoxyinus abronius* was the most sensitive species among the tested species in different trophic levels in the water column. Ecotoxicological data (0.14 mg/L) from the most sensitive species, *R. abronius*, were used for calculating  $PNEC_{pelagic}$  (or PNEC in the water column, as chronic  $PNEC = NOEC/AF = 14$  ppb). An AF of 10 was used in accordance with the European Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) regulation and the EU-TGD (EC 2003). For simulation modeling in DREAM, the chronic  $PNEC_{pelagic}$  of 14 ppb was selected as the threshold value to compute impacted area or polluted volume based on PEC values.

#### Target Lipid Model-derived HC5 estimates

Chronic PNEC values for each oil component were derived by calculating an HC5 value for each hydrocarbon component and using an AF,  $PNEC = HC5/AF$ . The derivation of PNEC was based on the TLM (Di Toro et al. 2000), given that

hydrocarbons have high solubility in lipid tissues of organisms, where toxic responses occur (Redman et al. 2014).

Oil hydrocarbons contain nonpolar (hydrophobic) organic compounds that have a low aqueous solubility and a high lipid solubility. If the concentration of nonpolar organic chemicals into an aquatic animal's tissue lipids reaches a critical concentration, toxic actions occur in the organism (NRC 2003). The equilibrium partition or hydrophobicity of hydrocarbon compounds can be displayed by the octanol–water partition coefficient for the substance ( $K_{ow}$ ). The  $\log K_{ow}$  describes the tendency of a chemical to solve in an artificial biotic lipid (i.e., octanol) compared to the solubility in water, and it is an indicator of the tendency of an organic compound to be adsorbed into organisms (Neff 2002).

The TLM is a quantitative structure–activity relationship model (QSAR) that is based on the  $K_{ow}$  and the individual sensitivity of the tested organism. In the derivation from the  $\log K_{ow}$ -based QSAR, a critical target lipid body burden (CTLBB) is a representative of a toxicity endpoint of the challenge organism. By the use of the CTLBB, the inherent toxicity of individual hydrocarbons can be estimated (Redman et al. 2014).

Previous studies suggest that an additional AF is not required to establish a protective PNEC for each of the oil components (McGrath and Di Toro 2009; Redman et al. 2014). Therefore, the AF was 1 and a chronic PNEC (ppb) was based on HC5 (ppb). Thus, the TLM-derived HC5 was calculated as

$$\begin{aligned} \log \text{HC5} &= -0.936 \times \log K_{ow} - \log \text{ACR} + \Delta c + \log \text{CTLBB} \\ &\quad - k_z \sqrt{(V_{\text{slope}} \log K_{ow}^2 + V_{\log \text{CTLBB}} + V_{\log \text{ACR}})}, \end{aligned}$$

in which CTLBB = mean critical target lipid body burden ( $\mu\text{mol/g}$  octanol), ACR = mean acute to chronic ratio,

$k_z$  = 95% confidence extrapolation factor,  $V_{\text{slope}}$  = variance in the universal target lipid slope,  $V_{\log \text{CTLBB}}$  = variance of  $\log$  CTLBB,  $V_{\log \text{ACR}}$  = variance in the ACR, and  $\Delta c$  = chemical class correction.

Values of  $\log K_{ow}$  of each component are presented in Supplemental Data Table 5. Parameters and their values for the calculation of  $\log \text{HC5}$  are presented in Supplemental Data Table 6 (McGrath et al. 2004; McGrath and Di Toro 2009). Calculated PNEC values for each oil component by TLM-derived HC5 estimation, which are presented in Supplemental Data Table 7, were used as input data for DREAM.

#### Case study and model parameters

Our theoretical scenario represented an oil leakage from abandoned wells in an oil field with a water depth of 70 m in the southern part of the Norwegian sector in the North Sea (Table 2).

The crude oil type in the North Sea had an American Petroleum Institute (API) gravity of 36.4 and a density of  $840 \text{ kg/m}^3$ . Hydrocarbons of the oil were divided into different groups based on physical, chemical, and biological properties of 1 compound or associated representative compounds of the group (Supplemental Data Table 2).

The oil droplet size is a primary determinant of both subsurface and surface oil transport. The oil droplet sizes for our case study were taken from oil droplets of natural seeps, which were measured and ranged from 1 to 10 mm in diameter (Garcia-Pineda et al. 2016) with an average of 5 mm at the release point (Johansen et al. 2017). Substantial differences in biodegradable rate and evaporation rates have been observed between the oil droplet sizes of 1 mm to 5 mm (Tveit 2018). Because the oil leakage occurred at continuous low rate, oil droplets at the leakage point with diameters of 1 mm, 3 mm, and 5 mm were therefore chosen to

Table 2. Scenario parameters, assumptions, and values for running DREAM<sup>a</sup>

Assumptions		Values
Release rate	Very low	1 L/h
Release depth	Water depth reference	80 m
Release duration	Simulation lengths change between scenarios	1 d, 5 d, 30 d, 60 d, 90 d
Oil droplet size	Small	1 mm, 3 mm, and 5 mm
Environmental parameter	Province	North Sea
Seasonal current and wind data	In winter	Simulations start from 1 January 2014
	In summer	Simulations start from 1 July 2014
Model parameter	Small grids	2 km × 2 km
	A great number of grid cells	1000 × 1000 × 8
	Small time steps	5 min for 1 leakage day 15 min for 30 d

<sup>a</sup>Rye et al. 2008.

DREAM = Dose related Risk and Effect Assessment Model.

investigate the importance of the droplet size. The rise velocity of droplets in DREAM was calculated from Stokes' law using lower Reynolds numbers, with a harmonic transition to constant drag at higher Reynolds numbers (Nordam et al. 2019; Johansen 2000).

Current and wind data in the regions of the Norwegian North Sea were required as input for DREAM modeling and were provided by SINTEF Ocean. The current data were generated by the SINMOD ocean model developed by SINTEF (Slagstad and McClimans 2005) at a resolution of 1.3 km and a time step of 2 h. The model used boundary conditions from a larger scale model and atmospheric data from ERA-Interim, which is a data set that shows the latest global climate analysis produced by the European Centre for Medium-Range Weather Forecasts (Dee et al. 2011).

The release rate from the abandoned well was set to 1 L/h. Due to limited leakage rates associated with abandoned wells, the model was constructed using small grids, cell sizes, and time steps, where a large number of grid cells was applied (Table 2).

## RESULTS AND DISCUSSION

Potential biological exposure and chronic effects of continuous low-rate oil leakages from abandoned wells can be assessed by an estimation of chronic exposure and effect values, that is, PEC and PNEC, respectively. These data provide a calculation of the risk as an EIF utilized within an ERA.

### *Distribution of impacted areas and variation of EIFs by different oil droplet sizes*

Calculations of impacted areas by EIFs under the influence of oil droplet sizes (i.e., 1 mm, 3 mm, and 5 mm) were executed through different release scenarios of 1 d, 5 d, and

30 d starting from 1 January 2014. In order to see differences between 1-mm, 3-mm, and 5-mm droplets, dimensions of grids and time steps were constant for 3 droplet sizes with all settings and dimensions of grids and cell sizes being the same for all of 3 simulation durations. Model parameters for simulation of 1-mm, 3-mm, and 5-mm droplets are presented in Supplemental Data Table 8.

Table 3 shows that impacted areas and EIF values varied considerably among 3 droplet sizes. Directions of impacted areas were mainly to the southwest (SW) and northeast (NE) during the first days and then around all sides in the following period (i.e., 30 d). Environmental impact factor values were inversely proportional to oil droplet sizes. Environmental impact factors of smaller droplets (i.e., 1 mm) increased significantly with time and were larger than EIFs of larger droplets (i.e., 3 mm and 5 mm).

Simulations of 1 leakage day and 30 leakage days for 1-mm and 5-mm oil droplets are reported in Figure 2. Distributions of impacted areas varied between different oil droplets after 1 d. For the 1-mm droplets, black areas or volumes, where more than 5% of species would be affected, appeared more clearly and densely than for the 5-mm droplets. Moreover, the 5-mm droplets could scatter and disperse far away from the release site, whereas the 1-mm droplets moved slowly within areas near the release site. Also, in Table 3, after 1 d, impacted areas in cases of 5-mm droplets could spread out to 226 m SW and 382 m NE in distance, whereas for 1-mm droplets the distances were shorter, at 180 m SW and 60 m NE.

After 30 d, impacted areas for the 5-mm oil droplet would disperse around the release site in all directions (Figure 2). In addition, impacted areas of the 1-mm droplets tended to be concentrated near the release site and increased only in certain areas. In general, those cases reached out to 1-km

**Table 3.** Impacted areas and EIF by different oil droplet sizes of 1 mm, 3 mm, and 5 mm

Duration (from 1 January 2014)	Oil droplet (mm)	Impacted volume (km <sup>3</sup> )	EIF	Distance from a release site to areas where >5% of species would be affected (for main directions)
1 d	5	$3.66 \times 10^{-4}$	0.0051	22–226 m SW; 10–382 m NE
	3	$5.36 \times 10^{-4}$	0.0083	14–247 SW; 10–340 m NE
	1	$5.46 \times 10^{-4}$	0.015	75–180 m SW; 40–60 m NE
5 d	5	$4.33 \times 10^{-3}$	0.0086	45–460 m SW; 60–590 m SE; denser areas in 45–90 m S
	3	$4.99 \times 10^{-3}$	0.0093	25 SW–380 m SW; 50–345 m E; denser areas around 40–120 m S
	1	$6.26 \times 10^{-3}$	0.0227	Denser area in between 15–280 m SW, 0–245 m S, 0–150 m E, and 0–250 m SE
30 d	5	$4.03 \times 10^{-3}$	0.0118	Scattering around to 1-km radius; dense areas in 150 m to the south
	3	$4.42 \times 10^{-3}$	0.0173	Scattering around to 1-km radius; dense areas in 150 m to the south
	1	$5.93 \times 10^{-3}$	0.0309	Dense areas within 300-m radius to the S–SE, whereas some reach out to 1-km radius

E = east; EIF = environmental impact factor; NE = northeast; S = south; SE = southeast; SW = southwest.

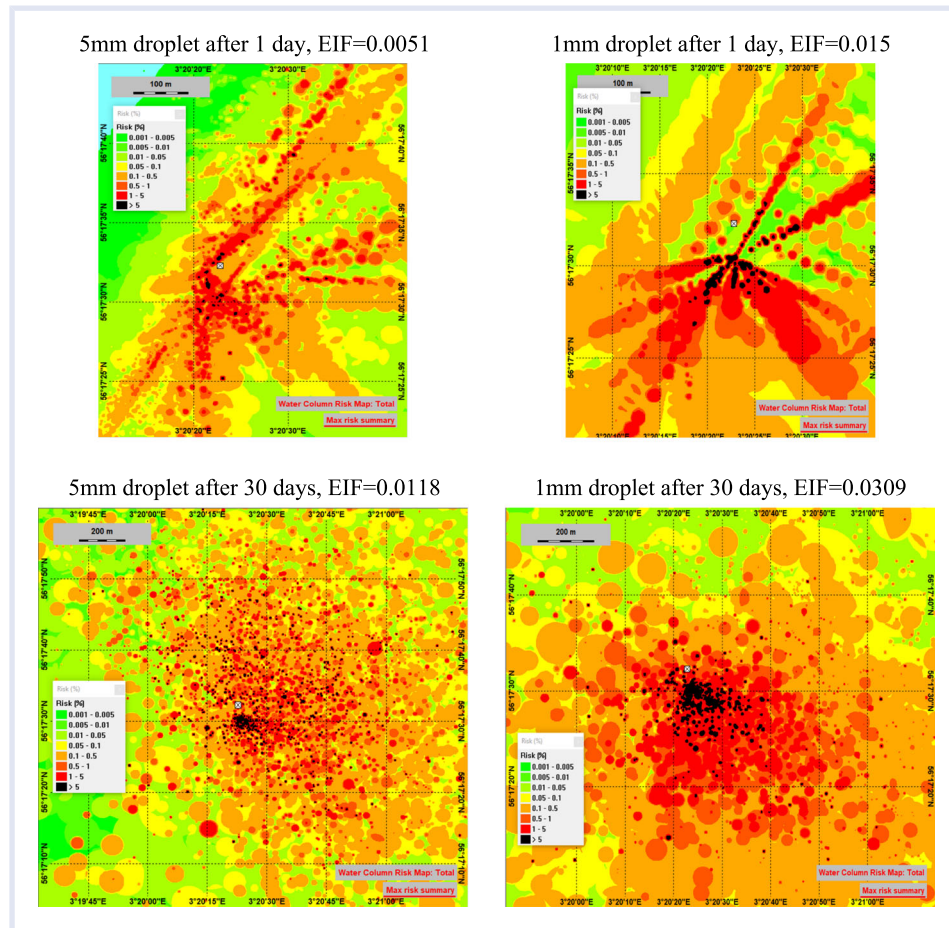


Figure 2. Environmental impact factor (EIF) and impacted areas for 5-mm and 1-mm droplets after 1 d of leakage and 30 d of leakage.

radius, including a location of denser areas located about 150 m from the site to the south and southeast (S–SE) for the 3-mm and 5-mm droplets and within 300-m radius mostly to the west, south, and southeast (W–S–SE) for the 1-mm ones.

Overall, different droplets had different distribution patterns of impacted areas. Bigger droplets were more scattered or dispersed than were smaller droplets. Small droplets had a tendency to be more concentrated over time in certain areas near the release site. These results show that small droplets of 1 mm may be kept suspended in the water column by turbulent diffusion and, following the Stokes' law, they rise up to the surface more slowly than the larger ones.

When evaluating water column dispersion, impacted areas for the 5-mm droplets were located on the surface, whereas the 1-mm droplets were suspended throughout the water column showing a higher EIF value (Figure 3). The model predicts that droplets will rise to the surface, whereas the dissolved chemicals will be neutrally buoyant and will move horizontally with the current. The vertical cross sections confirmed that more hydrocarbons dissolved and remained in the water column in the case of 1-mm droplets, whereas most were found on the surface in the case of 5-mm droplets.

Being suspended in the water column, smaller droplets would more easily be absorbed and degraded by

oil-degrading bacteria than would the larger droplets, due to their higher surface area relative to volume of oil and their more effective oil–water contact time (Almeda et al. 2013). Therefore, small suspended oil droplets could easily interact with planktonic organisms and, for example, be absorbed by zooplankton (protozoan and metazoans) or be adsorbed to phytoplankton, leading to chronic effects in the organisms (Almeda et al. 2013). Thus, smaller oil droplet sizes may have a greater impact on the ecosystem.

#### Hydrocarbon contribution to risks

Modeling simulations for different oil droplets predicted contributions of specific hydrocarbon fractions from oil at different leakage times. A scenario was set up for 1 L/h leakage of the 1-mm, 3-mm, and 5-mm droplets for 30 d starting from 1 January 2014.

The PAH-1 was predicted to be the most toxic fraction of oil (Figure 4), emphasizing that PAHs are much less biodegradable in comparison with saturated hydrocarbons and have been shown to be the most toxic components of crude oil (NRC 2003; Stout and Wang 2016).

Higher molecular weight hydrocarbons, including C11–C12 total saturates + aromates to C25+ (C26–C36) total saturates + aromates, showed limited contributions to overall predictions of the risk (Figure 4). According to Neff

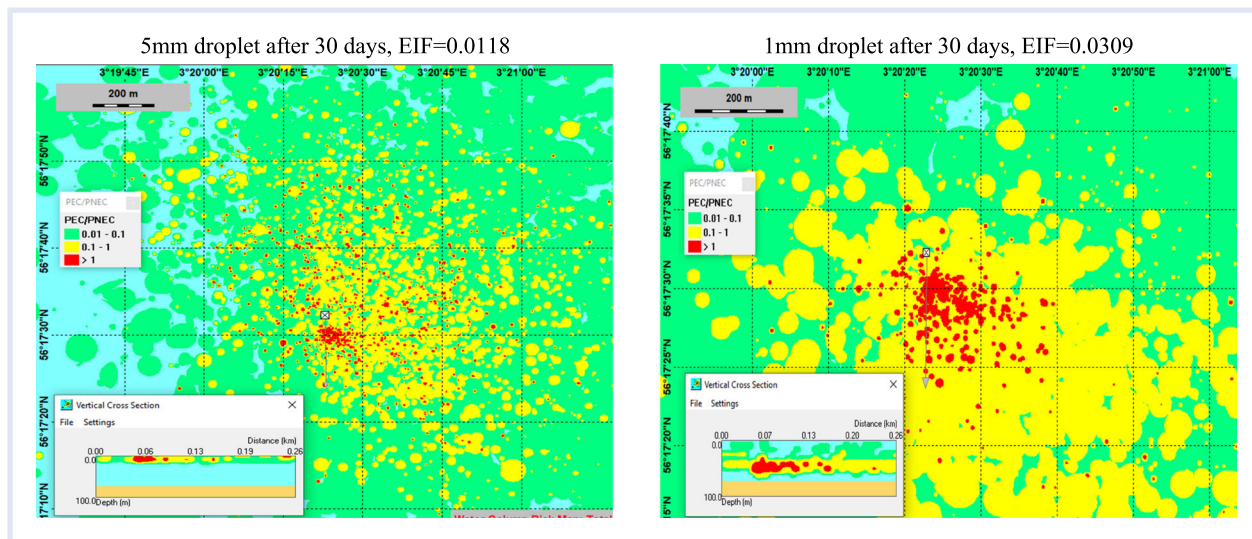


Figure 3. Environmental impact factor (EIF) and locations of 5-mm and 1-mm droplets after 30 d of leakage; related to predicted environmental concentration/predicted no-effect concentration (PEC/PNEC).

(2002), high molecular weight hydrocarbons have low bio-availability due to limitations of uptake by organisms, low solubility in tissue lipids, and rapid metabolism in some marine organisms. Chemicals with  $\log K_{OW}$  values from 4.5 to 6.0 are of greater concern because they have the potential

for uptake in organisms, whereas chemicals with  $\log K_{OW}$  above 6.0 have diminished uptake (Dimitrov et al. 2012). These theoretical views suggest that compounds of C9-saturates, C10-saturates, cycloparaffins, and PAH-1 (having  $\log K_{OW}$  values  $>4.7$  and  $<6.0$ ) may have more

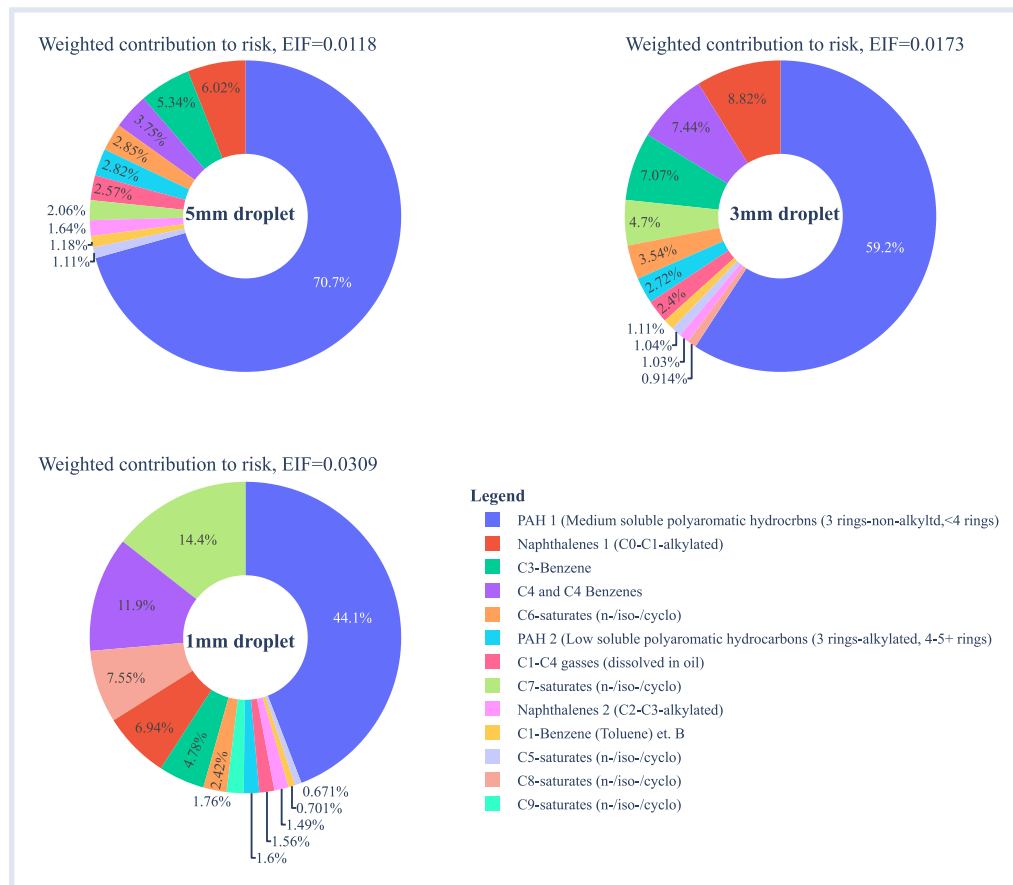


Figure 4. Hydrocarbon contributions to risks of 5-mm, 3-mm, and 1-mm droplets by after a 30 d simulation. EIF = environmental impact factor.



**Table 4.** Environmental impact factors (EIFs) in summer and in winter

Droplet	Summer (from 1 January 2014)		Winter (from 1 July 2014)	
	5 d	30 d	5 d	30 d
5 mm	0.0074	0.0085	0.0086	0.0118
1 mm	0.023	0.0336	0.0227	0.0309

potential for toxicity due to greater uptake. On the contrary, high molecular weight hydrocarbons and PAH-2, which had log  $K_{ow}$  values reaching up to 6.0 or nearly 6.0, may be less soluble in tissue lipids of organisms and have lower toxicity.

Figure 4 also shows that fractions of other toxic components, such as naphthalenes 1 (C0- to C1-alkylated, C3-benzene, C4-benzene, C7-saturates, and C8-saturates (*n*-*iso*/*cyclo*), increased in smaller droplets. This increase could be related to the fact that bigger droplets rise more quickly in the water column to the surface where the evaporation and oxidation occur, whereas smaller ones likely dissolve in the water due to their large surface area to volume ratio. Thereby, monocyclic aromatic hydrocarbons (e.g., benzene, toluene, ethylbenzene) or low molecular weight hydrocarbons (C1 to C6) would undergo evaporative loss given their higher vapor pressure values, as well as photooxidation at the surface. Therefore, concentrations were lower in fractions from larger droplets than from smaller droplets. In addition to compounds in the PAH-1 fraction, toxic hydrocarbon components of naphthalenes 1, C3- to C4-benzenes, and C7- to C8-saturates should be of

concern when it comes to evaluating effects of the oil in this type of scenario.

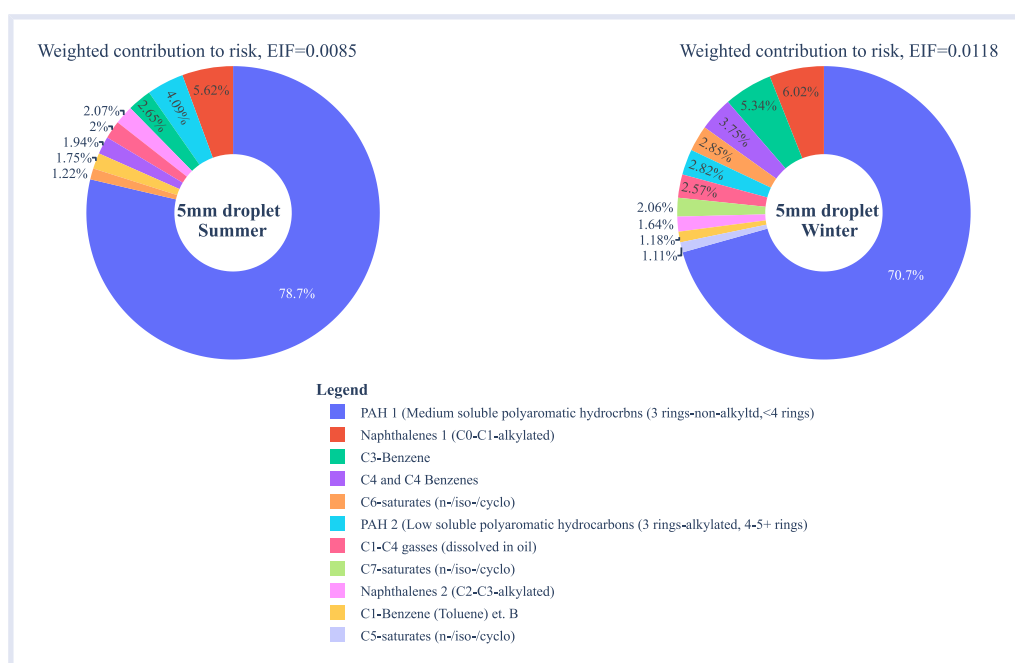
#### Seasonal variation of EIFs

Seasonal variation of EIF was evaluated, considering oil droplet sizes of 1 mm and 5 mm and simulation lengths of 5 d and 30 d in the winter period, starting from January 2014 and in the summer period (from July 2014).

Environmental impact factor values varied between the 2 droplet sizes during 2 seasons (Table 4). Environmental impact factors for the 1-mm droplets were slightly higher in summer (e.g., EIF = 0.0336 after 30 d) than in winter (e.g., EIF = 0.0309 after 30 d). In contrast, EIFs for the larger 5-mm droplets were lower in summer (e.g., EIF = 0.0085 after 30 d) than in winter (e.g., EIF = 0.0118 after 30 d).

Because larger (5-mm) droplets move up to the surface faster than smaller (1-mm) droplets, higher evaporation and oxidation rates in summer than in winter would likely eliminate *n*-alkanes, monocyclic hydrocarbons, benzenes, and other volatile toxic hydrocarbons more rapidly (NRC 2003), resulting in a reduced oil toxicity in summer compared to winter (Figure 5). This hypothesis was consistent with the EIFs of the larger droplets being lower in summer than in winter.

Moreover, thermoclines are commonly observed in the North Sea (Heessen et al. 2015; Quante and Colijn 2016). Small droplets in summer would be kept suspended in the water column (see Figure 3) and trapped under the thermocline, where greater oil–water contacts and water-in-oil emulsion could occur, resulting in an increased potential for biological exposure to oil compounds (Wolfe 2013). Also, a larger number of species (e.g., phytoplankton, herring, blue whiting, cod) would extend their habitats, growing and



**Figure 5.** Environmental impact factors (EIFs) and hydrocarbon contribution to risks by 5-mm droplets in summer and in winter.

spawning more rapidly and successfully in the warmer water column in summer (IMR 2019). These views explain simulation results with higher EIFs for smaller droplets in summer than in winter. Smaller oil droplet sizes might have a greater impact on the ecosystem. Thus, oil leakage could have greater impacts on the pelagic organisms in summer than in winter.

Calculations of EIFs in DREAM are determined by the local current and wind in different seasons; EIFs of the oil leakage could likely be influenced by processes of dilution, dispersion, evaporation, and oxidation under variation of the ocean current and wind regimes in the water column and on the surface.

Acceptable risks

To consider long-term effects and chronic risks of relatively small magnitudes of oil leakage with a constant release rate over a long period, a scenario was set up for 1 L/h leakage of the 1-mm droplets for 1, 5, 30, 60, and 90 d starting from 1 January 2014. Grids and other model parameters were used for these scenarios, in which parameters for 60-d and 90-d simulations were maintained the same as those for the 30-d simulations (Supplemental Data Table 8).

Simulations showed that impacted areas or pollution volumes are continuously enlarging over time in potential leakages from abandoned wells, and apparently they reach an equilibrium at 30 d (Table 5). During the first 30 d, EIFs increased considerably from 0.015 to 0.0309, whereas between 30 and 90 d there were no significant variations in EIF values (from 0.0309 to 0.0329). An increase in the impacted area indicates a reduction in the exposure concentration after 30 d, which could imply the influence of physical spreading or dispersion of oil leakage and chemical or biological alterations of the oil components over time.

Results also indicated that no matter how long simulation durations were, distances from the release site to the impacted areas above the acceptable risk were within a 1-km radius from the release point. Moreover, impacted areas became more dense and compact over time near the release site (Figure 6). The impacted areas were distributed within a 300-m radius mostly to the W–S–SE after 30 d to 60 d. They were further concentrated within a 300-m radius on all sides after 90 d.

Formation of a diverse benthic community and adaptation of marine organisms over generations have been observed from areas of active seepages (NRC 2003; Sahling et al. 2016). For example, at only a few centimeters from the active sources of seepage, nematodes, worms, and copepods increase with high abundance, but *Rhepoxiniid* amphipods that are sensitive to oil are found only outside the seep area (Spies and Davis 1979). Similarly, asphalt on the sea floor is associated with a unique community of benthic organisms (Sahling et al. 2016). Over a period of thousands of years, animals living near seeps have unique adaptations (Sahling et al. 2016). It could be argued that leakages from abandoned wells, with long-term exposure to the continuous low-rate oil leakage, may cause chronic or long-term

Table 5. Results of 1-d, 5-d, 30-d, 60-d, and 90-d simulations

	Simulation duration (from 1 January 2014)				
	1 d	5 d	30 d	60 d	90 d
Impacted volume (km <sup>3</sup> )	5.46 × 10 <sup>-4</sup>	6.26 × 10 <sup>-3</sup>	5.93 × 10 <sup>-3</sup>	9.18 × 10 <sup>-3</sup>	1.73 × 10 <sup>-2</sup>
EIF	0.015	0.0227	0.0309	0.0336	0.0329
Distance from a release site to areas where more than 5% of species would be affected	75–180 m SW and 40–60 m NE	Dense areas in distances of 15–280 m SW, 0–245 m S, 0–150 m E, and 0–250 m SE	Dense areas within 300-m radius mostly to the west, south, and southeast, whereas some reach out to 1-km radius	Dense areas within 300-m radius mostly to the west, south, and southeast, whereas some reach out to 1-km radius	Dense areas within 300-m radius on all sides and reaching out to 1-km radius

E = east; EIF = environmental impact factor; NE = northeast; S = south; SE = southeast; SW southwest.

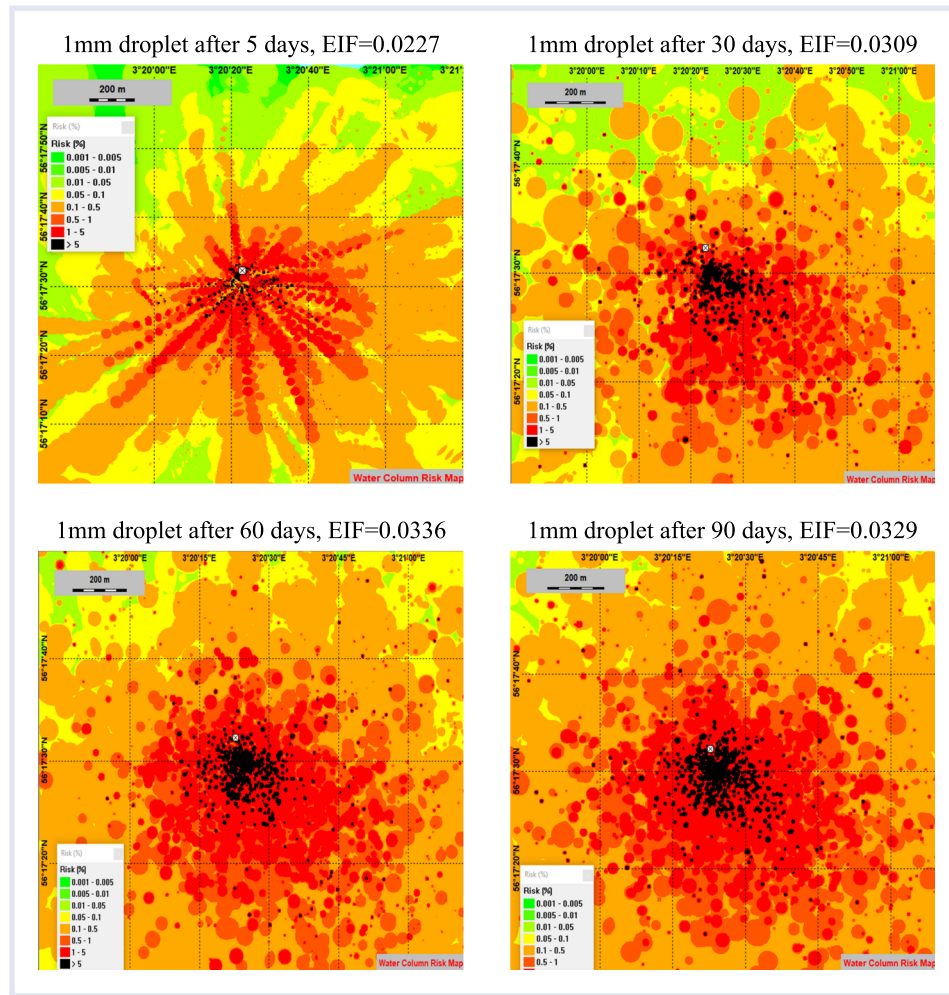


Figure 6. Environmental impact factors (EIFs) and risk maps for 5-d (top left), 30-d (top right), 60-d (bottom left), and 90-d (bottom right) simulations.

effects on organisms and would potentially lead to steady-state effects that remain over a long period.

Oil released to marine systems may be oxidized by microbes and serve as a supplementary food source for food webs. However, at sufficiently high concentrations of aromatic components, the oil becomes toxic to marine organisms (NRC 2003). Chronic exposure to hydrocarbons derived from oil could result in alterations in reproduction and development of sensitive populations of marine organisms. Through food chains, alterations may affect consumers of zooplankton and eventually top predators of fishes (NRC 2003). As in Figure 6, the impacted areas (in black), where the risk is not acceptable, were increasing from 5 to 90 d around the release site. Environmental consequences from chronic exposure by continuous low-rate oil leakage from abandoned wells must somehow be considered with respect to the zero-leakage target. Based upon our results, additional risk assessments are needed at specific wells to evaluate the impacts of leakage from abandoned wells to determine whether interventions and/or reparations of the leakage are needed.

## CONCLUSION

Aiming to provide an approach for modeling fate and effects of oil leakages from abandoned wells, the present study has shown that the estimation of biological exposure and chronic effects by continuous low-rate oil leakages can be achieved through simulation modeling. The novelty of the present study was the use of chronic ecotoxicological thresholds derived from the PETROTOX database, allowing DREAM to be used in the leakage scenario for specific oil components. Thus, impacted areas and chronic risks of oil leakage can be predicted by EIF estimates through DREAM. Even at a very low leakage rate, the calculated risk emphasizes the necessity for mitigation measures and more consideration regarding the goal of “zero discharges” when it comes to abandoned wells.

The implementation of the EIF assessments for oil leakages from abandoned wells could provide necessary information to improve risk management and regulation for P&A activities. However, it should be noted that this EIF-based approach provides only an initial screen for environmental impact for management decisions, and site-specific

parameterization may be required for more sensitive locations, for example, coral reefs or nursery areas during spawning seasons.

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**Disclaimer**—The authors declare no conflicts of interest.

**Data Availability Statement**—All data are publicly available.

## SUPPLEMENTAL DATA

**Figure 1.** General workflow for modeling and environmental impact factor (EIF), related to predicted environmental concentration/predicted no effect concentration (PEC/PNEC) in an environmental risk assessment process.

**Figure 2.** An example of scenario parameters used in the Dose related Risk and Effect Assessment Model (DREAM).

**Table 1.** Hydrocarbon groups used for modeling

**Table 2.** Physical, chemical, and biological properties of hydrocarbon groups (from Dose related Risk and Effect Assessment Model [DREAM] database)

**Table 3.** Hydrocarbon blocks with boiling point ranges and fraction of weight % for PETROTOX v3.06 calculations

**Table 4.** Predicted no observed effect concentration (NOEC) for available seawater species calculated by PETROTOX 3.06

**Table 5.** Log  $K_{ow}$  (octanol–water partition coefficient) calculated for each hydrocarbon component

**Table 6.** Parameters for the calculation of log HC5 by TLM-derived HC5 estimation

**Table 7.** Chronic predicted no-effect concentration (PNEC) for the each of oil components by the 5<sup>th</sup> percentile hazardous concentration (HC5) estimation

**Table 8.** Model parameters for simulations of 1-mm, 3-mm, and 5-mm droplets.

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