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Assessing impacts of simulated oil spills on the Northeast Arctic cod fishery

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ABSTRACT

We simulate oil spills of 1500 and 4500 m^3/day lasting 14, 45, and 90 days in the spawning grounds of the commercial fish species, Northeast Arctic cod. Modeling the life history of individual fish eggs and larvae, we predict deviations from the historical pattern of recruitment to the adult population due to toxic oil exposures. Reductions in survival for pelagic stages of cod were 0–10%, up to a maximum of 43%. These reductions resulted in a decrease in adult cod biomass of < 3% for most scenarios, up to a maximum of 12%. In all simulations, the adult population remained at full reproductive potential with a sufficient number of juveniles surviving to replenish the population. The diverse age distribution helps protect the adult cod population from reductions in a single year's recruitment after a major oil spill. These results provide insights to assist in managing oil spill impacts on fisheries.

1. Introduction

Fisheries and petroleum resource development are concomitant in some of the world's most productive continental shelf seas (Halpern et al., 2008). In the Barents Sea, commercial fishing has been carried out for at least 1000 years. This cod fishery is arguably among the world's healthiest commercial fish stocks (Kjesbu et al., 2014). The fishery is sustained through the recruitment of 3-year olds originating from spawning grounds along the Norwegian coast in the North Atlantic but with the main spawning grounds in the Lofoten-Vesterålen region (Ottersen et al., 2014). The Lofoten-Vesterålen region is currently under consideration for petroleum development posing a potential for oil spills to damage harvestable fish populations. Much of our understanding of the impacts of oil in the environment on fisheries and the general health of ecosystems has been acquired from detailed post-spill investigations of a few major events (French-McCay, 2004; Incardona et al., 2014; Peterson et al., 2003). The current probability for a blowout during drilling of an exploration well according to Gulf of Mexico and North Sea standards is estimated to be 1.4×10^{-4}, or once per every 7000 wells drilled (Lloyd's's, 2016). The low frequency, unique circumstances, and complexity of past major oil spills limits their applicability toward predicting the probability of new spills and developing prescriptive knowledge that would aid in the prevention or at least, minimization of damages to fishery resources in future spills (Eckle et al., 2012).

Scientific assessments of the Exxon Valdez oil spill and Deep-water Horizon oil spill call for the development of predictive models to assist in determining how best to respond to oil spills in order to minimize their impacts on the environment (Buskey et al., 2016; Peterson et al., 2003). Models provide quantitative capacity to examine processes on relevant spatial and temporal scales (French-McCay, 2003) and to conduct numerical experiments when real experimentation, namely controlled field studies on oil spills, is infeasible. Elucidation of the responses of fish populations to oil spills requires integrative understanding of 1) the fate and transport of oil, and 2) the transport, behavior, and interactions of biota in the environment and 3) the biological effects linked to the toxicity of oil compounds.

The perceived highest risk to fish populations is from oil spills that coincide in space and time with spawning events, when the most sensitive life stages of a single year's juvenile recruitment cohort are present (Carls et al., 1999; Heintz et al., 1999). The current exposure-response paradigm assumes that bio-available compounds induce effects to organisms when concentrations exceed a certain concentration in the organism, or threshold level (Jager et al., 2011; Jager and Kooijman, 2008). Threshold levels of sensitivity to compounds are organism- and life stage-specific. Their values are widely debated, particularly in connection to damage assessments in the aftermath of major oil spill events (Incardona et al., 2014; Peterson et al., 2003). Traditional

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environmental risk assessment procedures quantify immediate toxic responses of organisms to petroleum compounds using thresholds derived from standardized laboratory bioassays, such as a four-day survival test in the case of juvenile fish (OECD, 2002). Yet, controlled laboratory experiments show that delayed developmental defects such as abnormal hearts and reduced cardiorespiratory function affect individual fitness in a number of critical ways, including reduced swimming performance, prey capture, and prey avoidance, with repercussions for survival and population recruitment (Incardona et al., 2015). Reduced cardiorespiratory fitness is a potentially significant factor in the slow recovery of Prince William Sound fish populations; chronic exposure of early life stages (ELSs) of herring were linked to trace levels of oil and an observed decrease in the adult population of herring 4 years after the spill (Incardona et al., 2015). Delayed developmental effects may also affect the recovery of some species in the Gulf of Mexico after the Deep-water Horizon spill (Fodrie et al., 2014; Incardona et al., 2014).

In the present study, we simulate impacts of hypothetically severe oil spills in the core spawning areas of Northeast Arctic cod (*Gadus morhua*) hypothesizing that these severe oil spills will lead to transient reductions in harvestable cod (Fig. 1). For this population, the development stages from egg through larvae are at greatest risk because juveniles inhabit waters both deeper and farther north than the core spawning areas and are generally more widely distributed than earlier stages (Ottersen et al., 2014). Despite the previously described developments, there is a paucity of experimental data available for deriving reliable toxicity parameters for mechanistic modeling of toxicity processes for fish (De Laender et al., 2011; Klok et al., 2014). This has hindered the establishment of well-constrained threshold levels for organisms exposed to petroleum-related compounds (Olsen et al., 2013). To assure that effects are not underestimated, in the present study we simulate oil spill impacts based on four sets of threshold levels for petroleum compounds to cover a wide range of uncertainty. We evaluate and discuss the implications of our simulation results for understanding the resiliency of the Northeast Arctic cod population to oil spill events in the Lofoten-Vesterålen spawning areas. We conclude with a demonstration of how the long-term impact of an oil spill may be reduced by lowering fish quotas in the aftermath of a spill. These results provide valuable insights into pre-spill planning and spill response that may assist in managing impacts on fisheries.

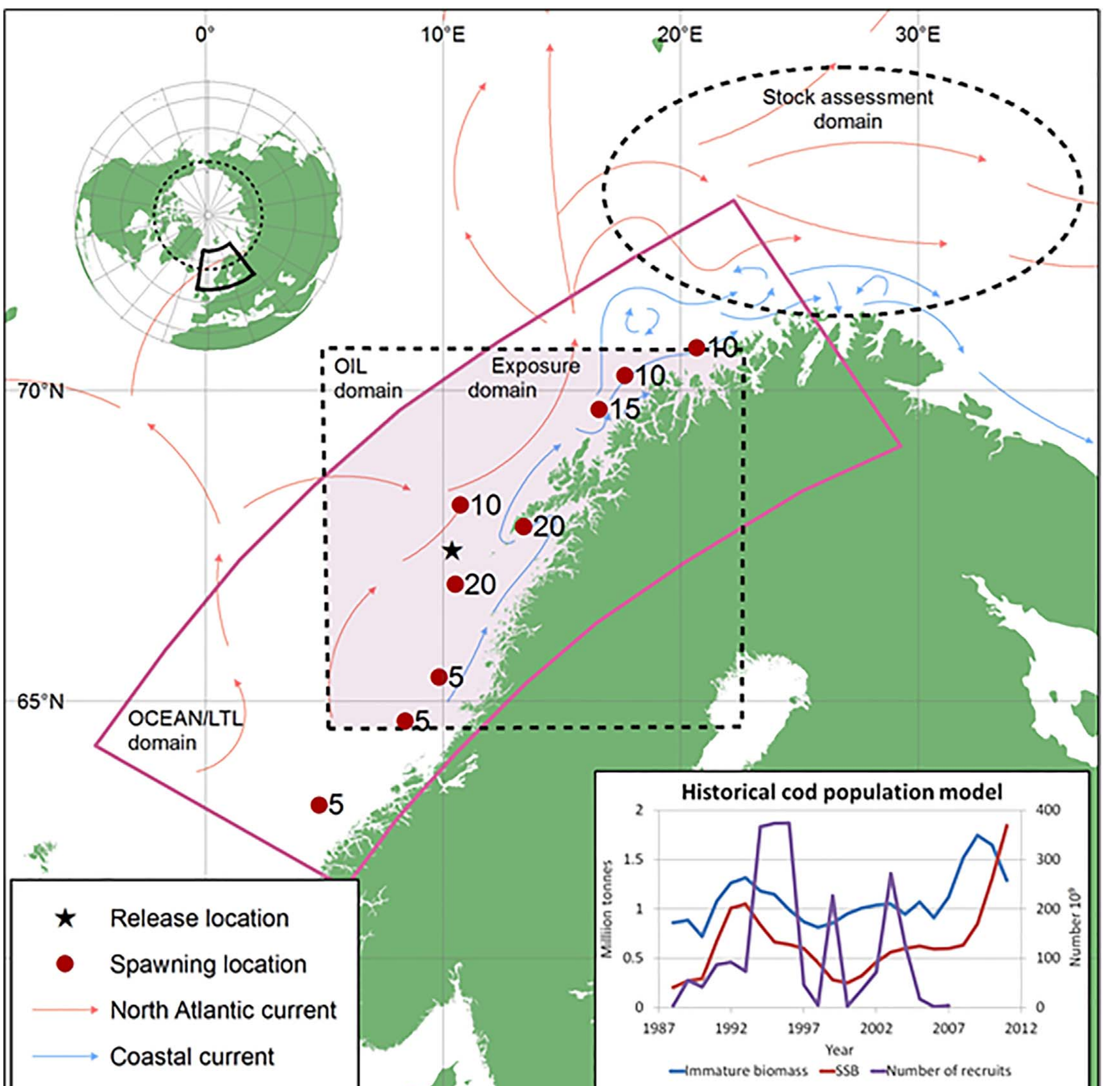

Fig. 1. The model domain for Lofoten-Vesterålen in the North Atlantic and for cod stock assessments in the Barents Sea. The location of the oil releases is 67.700 N 10.841E (black star). Nine cod spawning locations are also highlighted (red dots). The associated numbers refer to the percentage of eggs spawned assigned to each spawning location. The inset graph shows the predicted historical changes in cod population from 1988 to 2011 based on the annual Barents Sea cod stock assessment. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Table 1
Percent survival of early life stages (to the end of the pelagic stage) and spawning stock biomass of Northeast Arctic cod for oil release rates of 1500 or 4500 m^3/day for 14, 45, and 90 days at a single location on the Lofoten-Vesterålen shelf (75.2 N 32.4E). Four toxicity parameter sets were applied (P1–P4) representing different threshold levels of effects from petroleum compounds.

Scenario	Year	Release depth	Release rate (m^3/d)	Release duration (days)	Oil released (mt)	Oil type	P1	P2	P3	P4	P1	P2	P3	P4
1	1995	Topside	1500	14	18,095	Balder	100	100	100	99	100	100	100	100
2	1995	Topside	1500	45	58,152	Balder	100	100	98	87	100	100	99	97
3	1995	Topside	4500	45	174,469	Balder	100	100	96	81	100	100	99	95
4	1995	Topside	4500	90	348,917	Balder	100	100	86	66	100	100	97	91
5	1995	Topside	4500	90	335,698	Draugen	100	100	90	71	100	100	98	93
6	2001	Topside	1500	14	18,092	Balder	100	100	100	99	100	100	100	100
7	2001	Topside	1500	45	58,152	Balder	100	100	98	89	100	100	99	98
8	2001	Topside	1500	90	116,308	Balder	100	100	94	73	100	100	99	93
9	2001	Topside	1500	14	17,407	Draugen	100	100	100	99	100	100	100	100
10	2001	Topside	1500	45	55,948	Draugen	100	100	99	91	100	100	100	98
11	2001	Topside	1500	90	111,902	Draugen	100	100	97	79	100	100	99	94
12	2001	Topside	4500	14	54,281	Balder	100	100	100	98	100	100	100	99
13	2001	Topside	4500	45	174,471	Balder	100	100	98	82	100	100	99	95
14	2001	Topside	4500	90	348,915	Balder	100	100	86	57	100	100	97	88
15	2001	Topside	4500	14	52,229	Draugen	100	100	100	99	100	100	99	96
16	2001	Topside	4500	45	167,851	Draugen	100	100	97	82	100	100	98	90
17	2001	Topside	4500	90	335,766	Draugen	100	100	91	62	100	100	98	90
18	2002	Topside	4500	45	174,465	Balder	100	100	97	88	100	100	99	97
19	2002	Topside	4500	90	348,976	Balder	100	100	86	65	100	100	97	91
20	2002	Topside	4500	90	335,678	Draugen	100	100	90	73	100	100	98	93
21	1995	Topside	4500	45	174,461	Balder	100	100	96	77	100	100	99	94
22	1995	Subsea	4500	90	348,968	Balder	100	100	89	66	100	100	97	91
23	1995	Subsea	4500	90	335,738	Draugen	100	100	92	72	100	100	98	93
24	2001	Subsea	1500	14	18,093	Balder	100	100	100	98	100	100	100	99
25	2001	Subsea	1500	45	58,155	Balder	100	100	98	88	100	100	99	97
26	2001	Subsea	1500	90	116,317	Balder	100	100	95	74	100	100	99	93
27	2001	Subsea	1500	14	17,407	Draugen	100	100	100	99	100	100	100	100
28	2001	Subsea	1500	45	55,958	Draugen	100	100	99	92	100	100	100	98
29	2001	Subsea	1500	90	111,904	Draugen	100	100	98	82	100	100	99	96
30	2001	Subsea	4500	14	54,270	Balder	100	100	99	97	100	100	100	99
31	2001	Subsea	4500	45	174,460	Balder	100	100	96	78	100	100	99	95
32	2001	Subsea	4500	90	348,901	Balder	100	100	90	58	100	100	98	89
33	2001	Subsea	4500	14	52,218	Draugen	100	100	99	97	100	100	100	99
34	2001	Subsea	4500	45	167,854	Draugen	100	100	98	82	100	100	99	96
35	2001	Subsea	4500	90	335,717	Draugen	100	100	94	66	100	100	99	91
36	2002	Subsea	4500	45	174,475	Balder	100	100	96	82	100	100	99	96
37	2002	Subsea	4500	90	348,924	Balder	100	100	90	68	100	100	98	92
38	2002	Subsea	4500	90	335,700	Draugen	100	100	93	73	100	100	98	93

2. Materials and methods

We simulate oil spill scenarios with release rates of 1500 or 4500 m3/day lasting 14, 45, and 90 days (Table 1) at a single location on the Lofoten-Vesterålen shelf (67.700 N 10.841E) (Fig. 1) and the temporal and spatial life history events for cod early life stages (ELSs) and its primary prey species, *Calanus finmarchicus*. Major oil spills simulated for this location in an oil spill simulation study commissioned to support the update of the Integrated Management Plan for the Marine Environment of the Barents Sea–Lofoten Area, resulted in the largest areal impact on the marine environment (Det Norske Veritas, 2010). Based on a review of the historical record of fish stock abundances, we selected three years to simulate oil releases into the environment. The selected years, 1995, 2001, and 2002 represent years in which the stock exhibited high, low and average abundance respectively. Each simulation started on March 1 to correspond with the beginning of the spawning season.

We obtain information on the population properties and the effects on cod ELSs of toxic petroleum compounds introduced into the environment during an oil spill. Our longest release rate (90-day) is comparable to the duration of the Deepwater Horizon (DWH) spill release. However, oil release rates are controlled by reservoir characteristics (Dimmen and Syversen, 2010). Based on expert knowledge of our study location, we selected a release rate of approximately half the volume of the DWH oil spill. We compute survival probabilities for cod ELSs through to the end of the pelagic stage (when the juveniles descend to the seabed and become relatively stationary) for two cases: with and without oil (Pelagic Stages: Section 2.1). We adjust these survival probabilities for the influence of density dependent mortality from the end of the pelagic stage (Section 2.2: After settlement < 1 year). We then transfer these estimates to an area-based fish population model that describes changes in fish stock dynamics and fishing effort calibrated with 26 years of fishery monitoring data (Section 2.3: Juvenile and mature cod). We apply the additional oil-induced mortalities to each year of recruitment in the fish population model. This was done to increase the range of possible stock states that an oil spill could affect. Based on this 3-step process, we obtain a predicted fish population structure and catches for the two cases (with/without oil). We discuss the population impact and post-spill recovery based on changes in the aggregated weight of mature fish in the cod stock, termed the spawning stock biomass (SSB). We consider the limitations in our approach as a consequence of the limits of contemporary knowledge (Limitations: Section 2.4). All simulations are stored in a permanent electronic library (https://symbioses.no).

2.1. Pelagic stages of cod ELSs

The life history of cod eggs spawned on the Lofoten-Vesterålen shelf is as follows (Olsen et al., 2010; Ottersen et al., 2014). Mature cod originating in the Barents Sea spawn consistently from early March to late April at patchy spawning areas along the Norwegian coast, but mainly in the Lofoten-Vesterålen region (Fig. 1). Developing eggs,

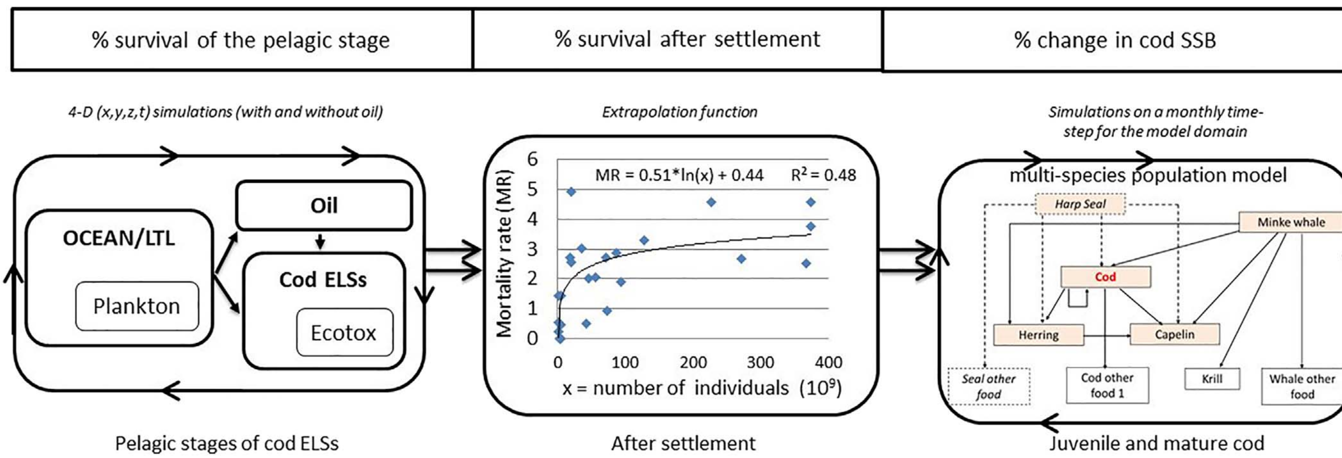

Fig. 2. The model framework predicts the difference in cod population structure (with/without oil) for selected climate, petroleum, predation, and harvesting scenarios. Larvae surviving the effects of petroleum compounds at the end of the pelagic stage are determined from 4D simulations (left). We apply a density dependent mortality function to extrapolate from the number of surviving cod from the end of the pelagic stage to age = 1 year (center). Each point in the plot represents a single year's recruitment and the line represents the overall trend. Mortality increases as population increases, flattening out at large stock sizes. Using a multi-species compartmental model (right), we simulate changes in the spawning stock biomass (SSB) of the juvenile and mature cod population (> 1 year old) for > 10 years post-spill. ELS = early life stage.

larvae, and juveniles inhabiting the pelagic zone are transported north and east toward the Barents Sea by near-surface currents. In late summer, ~August, the juveniles descend to deeper water layers and at age 3 they recruit to the fishable population. In any given year, only a tiny fraction of juvenile cod survive in this environment (Bogstad et al., 2015), which is characterized by spatially and temporally varying temperatures, advection and dispersal, light conditions, turbulence, prey availability, and predation. There is a 3–4 year period between juvenile recruitment to the population at age 1 and maturation. Mature cod live for > 10 years.

Early life stages of fish are vulnerable to the toxic effects of oil spills, from the egg stage until they become juveniles with sufficient mobility to avoid contaminated waters (Carls et al., 1999; Heintz et al., 1999). To predict the survival of drifting cod ELSs exposed to petroleum compounds after an oil spill, we coupled three models: a model to simulate oceanographic processes and a lower trophic level ecosystem model (Ocean/LTL) (Alver et al., 2016; Slagstad and McClimans, 2005; Wassmann et al., 2006); an ecological model for the pelagic stages of cod ELSs (Vikebø et al., 2007); and an oil transport and fate model (Oil) (Reed, 2004); (Fig. 2). These models are four-dimensional (three spatial dimensions and time). These are mature models that have undergone individual upgrades and validation exercises over time. The models are calibrated with regional data supported by long-term monitoring data sets on the Northeast Arctic cod fishery and wider ecosystem in the Barents Sea (Bogstad et al., 2015; Jakobsen and Ozhigin, 2011). The cod ELS model includes an ecotoxicology subroutine (Ecotox ELS) to predict the toxic effects of exposure to petroleum compounds (Klok et al., 2014). This is a new development and is described in a subsequent subsection.

2.1.1. Ocean dynamics and lower trophic level ecology model (Ocean/LTL)

This model (www.sinmod.com) is a 3D free-surface, *z*-coordinate ocean model based on the primitive Navier-Stokes equations and solved by a finite difference scheme (Slagstad and McClimans, 2005; Støle-Hansen and Slagstad, 1991). It simulates ocean currents, temperature, salinity, and turbulence and the lower trophic level ecosystem and has been validated against field data (Alver et al., 2016). The spatial domain of the ocean is 1340 × 460 km (Fig. 1), with a fixed 4 × 4 km horizontal grid resolution and 35 vertical layers ranging from 5 m close to the surface to 500 m at depths > 1000 m. Boundary conditions are obtained from a larger 4-km resolution model domain for the North Atlantic and Nordic Seas (Alver et al., 2016). Atmospheric forcing is from ECMWF ERA-Interim (Dee et al., 2011) and freshwater discharges from rivers and land are described in Slagstad et al. (2015). This model has been shown to realistically resolve ocean circulation and ice dynamics in the Barents Sea (Ellingsen et al., 2009; Slagstad and McClimans, 2005) and on the Norwegian shelf off Northern Norway (Anon, 2011; Skarðhamar and Svendsen, 2005). The lower trophic level ecological component, which is fully coupled to the ocean component, simulates the state of the lower trophic level ecosystem from nutrients up to mesozooplankton (*C. finmarchicus*) in an Eulerian framework (Slagstad et al., 2015). It includes egg and nauplii stages, development, spawning, and distribution as functions of food availability and temperature. The Ocean/LTL model supplies the cod ELS model with current, temperature, salinity, and food concentration fields (*C. finmarchicus* nauplii and copepodite).

2.1.2. Cod eggs and larvae model (Cod ELS)

This model simulates dispersal, diel migration, feeding and growth for different early life stages of Northeast Arctic cod using a particle-tracking model with built-in modules for individual physiological and behavioral responses of eggs and larvae to ambient forcing (Sandvik et al., 2016; Vikebø et al., 2007). The model has been validated empirically using observations from the Norwegian and Barents Sea (Vikebø et al., 2015).

The Cod ELS model includes an Ecotox ELS subroutine based on DEBtox equations (Dynamic Energy Budget Theory with ecotoxicology) described in Klok et al. (Klok et al., 2014). It predicts the effects on individual fish eggs and larvae as a function of the concentration of chemical groups in seawater and temperature (Jager et al., 2011). DEBtox assumes a hazard model with stochastic death (SD) and scaled internal concentration (SIC). Mortality effects are determined by a combination of three parameters: the no effect concentration (NEC, mmol/L), the elimination rate (k_e, 1/day), and the killing rate (b, L/mmol/day). The NEC represents the environmental concentration below which no effect occurs. The intensity of the effect is a function of b; the higher the b value the more toxic the compound. The elimination rate integrates both uptake and elimination processes, driving the speed of the effects; lower values imply a longer time to surpass the NEC threshold value. Uptake occurs only through water and not through food. We describe the toxicity of the mixture by assuming the independent NEC model (Baas et al., 2009). We corrected all rates (k_e and b) for temperature according to the Arrhenius relationship. As the DEBtox subroutine is part of the Cod ELS model, it also incorporates dilution by growth.

Given the lack of sufficient experimental data for deriving highly

Table 2
Parameter values established by Klok et al. (2014) at the reference temperature of 298 K to model the impact of bioavailable chemical groups on fish early life stages. These parameter values are extrapolated from parameters for fathead minnow and naphthalene (Baas et al., 2009; Jager et al., 2011) and the extrapolation equations of Jager and Kooijman (2008). We based our four parameter sets (P1–P4) on these parameter values. All four parameter sets include assessment factors that result in effects on fish early life stages at lower concentrations. In parameter sets P3 and P4, the groups shown in **purple,** were artificially set to induce immediate mortality at concentrations (P3: 1.0 ppb; P4: 0.1 ppb) associated with delayed developmental defects.

Group	Identification	k_e (/day)				NEC (mmol/L)				$\dot{b}$ (L/mmol/day)			
	Naphthalene	5.53				4.40x10-02				1.00x10+02			
	Saturates	P1	P2	P3	P4	P1	P2	P3	P4	P1	P2	P3	P4
1	C1-C4 saturates	4.31	4.31	4.31	4.31	8.19×10^{-04}	8.19×10^{-04}	8.19×10^{-04}	8.19×10^{-04}	$4.11\times10^{+02}$	$4.11\times10^{+02}$	$4.11\times10^{+02}$	$4.11\times10^{+02}$
2	C5-saturates	6.50	6.50	6.50	6.50	1.52×10^{-03}	1.52×10^{-03}	1.52×10^{-03}	1.52×10^{-03}	$2.53\times10^{+02}$	$2.53\times10^{+02}$	$2.53\times10^{+02}$	$2.53\times10^{+02}$
3	C6-saturates	4.69	4.69	4.69	4.69	9.29×10^{-04}	9.29×10^{-04}	9.29×10^{-04}	9.29×10^{-04}	$3.72\times10^{+02}$	$3.72\times10^{+02}$	$3.72\times10^{+02}$	$3.72\times10^{+02}$
4	C7-saturates	1.90	1.90	1.90	1.90	2.39×10^{-04}	2.39×10^{-04}	2.39×10^{-04}	2.39×10^{-04}	$1.09\times10^{+03}$	$1.09\times10^{+03}$	$1.09\times10^{+03}$	$1.09\times10^{+03}$
5	C8-saturates	1.70	1.70	1.70	1.70	2.02×10^{-04}	2.02×10^{-04}	2.02×10^{-04}	2.02×10^{-04}	$1.24\times10^{+03}$	$1.24\times10^{+03}$	$1.24\times10^{+03}$	$1.24\times10^{+03}$
6	C9-saturates	0.80	0.80	0.80	0.80	6.56×10^{-05}	6.56×10^{-05}	6.56×10^{-05}	6.56×10^{-05}	$3.01\times10^{+03}$	$3.01\times10^{+03}$	$3.01\times10^{+03}$	$3.01\times10^{+03}$
	Monoaromatics												
7	Benzene	30.3	30.3	30.3	30.3	1.53×10^{-02}	1.53×10^{-02}	1.53×10^{-02}	1.53×10^{-02}	$4.09\times10^{+01}$	$4.09\times10^{+01}$	$4.09\times10^{+01}$	$4.09\times10^{+01}$
8	C1-benzenes	13.2	13.2	13.2	13.2	4.41×10^{-03}	4.41×10^{-03}	4.41×10^{-03}	4.41×10^{-03}	$1.09\times10^{+02}$	$1.09\times10^{+02}$	$1.09\times10^{+02}$	$1.09\times10^{+02}$
9	C2-benzenes	7.67	7.67	7.67	7.67	1.94×10^{-03}	1.94×10^{-03}	1.94×10^{-03}	1.94×10^{-03}	$2.08\times10^{+02}$	$2.08\times10^{+02}$	$2.08\times10^{+02}$	$2.08\times10^{+02}$
10	C3-benzenes	3.88	3.88	3.88	3.88	6.98×10^{-04}	6.98×10^{-04}	6.98×10^{-04}	6.98×10^{-04}	$4.66\times10^{+02}$	$4.66\times10^{+02}$	$4.66\times10^{+02}$	$4.66\times10^{+02}$
11	C4/C5-benzenes	1.74	1.74	1.74	1.74	2.09×10^{-04}	2.09×10^{-04}	2.09×10^{-04}	2.09×10^{-04}	$1.20\times10^{+03}$	$1.20\times10^{+03}$	$1.20\times10^{+03}$	$1.20\times10^{+03}$
12	C10-sat (GC/FID)	0.57	0.57	0.57	0.57	3.91×10^{-05}	3.91×10^{-05}	3.91×10^{-05}	3.91×10^{-05}	$4.53\times10^{+03}$	$4.53\times10^{+03}$	$4.53\times10^{+03}$	$4.53\times10^{+03}$
	Polyaromatics												
20	Naphthalenes 1	3.82	3.82	∞	∞	6.82×10^{-04}	6.82×10^{-05}	7.41×10^{-06}	7.41×10^{-07}	$4.75\times10^{+02}$	$4.75\times10^{+03}$	∞	∞
21	Naphthalenes 2	1.14	1.14	∞	∞	1.11×10^{-04}	1.11×10^{-05}	6.13×10^{-06}	6.13×10^{-07}	$1.99\times10^{+03}$	$1.99\times10^{+04}$	∞	∞
22	PAH-1	0.87	0.87	∞	∞	7.42×10^{-05}	7.42×10^{-06}	5.65×10^{-06}	5.65×10^{-07}	$2.73\times10^{+03}$	$2.73\times10^{+04}$	∞	∞
23	PAH-2	0.23	0.23	∞	∞	1.01×10^{-05}	1.01×10^{-06}	4.49×10^{-06}	4.49×10^{-07}	$1.32\times10^{+04}$	$1.32\times10^{+05}$	∞	∞

reliable toxicity parameters for mechanistic modeling, we constructed four DEBtox parameter sets to establish the toxic levels of effects for the bioavailable chemical groups (Table 2). These parameter sets were designed to cover both immediate and delayed effects, narcotic, physiological, and behavioral effects and benchmarks for total polyaromatic hydrocarbon (PAH) exposure (McGrath and Di Toro, 2009; Trustees, 2016).

Parameter set P1 is based on empirically supported linear relationships between log Kow (Kow = n-octanol-water partition coefficient) and the no effect concentration (NEC), k_e, and b for juvenile fathead minnow (Eqs. 11–13, Klok et al. (Klok et al., 2014) and references therein), i.e.

$$\log(\mathrm{NEC}) = -0.90\ \mathrm{logKow} + 1.80 \quad (11)$$

$$\log(b) = 0.71\ \mathrm{logKow}{-}1.60 \quad (12)$$

$$\log(k_e) = -0.6\ \mathrm{logKow} + 2.76 \quad (13)$$

However, observed lethal and sublethal effects derived from experimental studies on fish ELSs are underestimated by this parameter set. Thus, Klok et al. (Klok et al., 2014) applied an assessment factor (AF) of 50 to include ELS sensitivity whereby NEC = NEC/AF and b = b × AF. The application of AF = 50 reduces toxicity levels to below the lowest known levels as considered by the Target Lipid Model which predicts sub-lethal effects of exposure to PAHs in fish early life stages (McGrath and Di Toro, 2009). Next, three additional parameter sets are created, P2–P4, to ensure coverage of uncertainty in the sensitivity of fish ELSs to bio-available petroleum compounds (based on published and unpublished data, including Incardona et al., 2015) and to incorporate principles of benchmarks for total polycyclic aromatic hydrocarbons applied in the Gulf of Mexico Natural Resource Damage Assessment (Trustees, 2016). To account for additional concerns that exposure to PAHs interferes with the development of fish eggs and larvae, though the mechanism is not yet fully known (Incardona et al., 2015), for parameter set P2 we applied a safety factor of 500 for polyaromatics (including naphthalenes). Parameter sets P3 and P4, based on the concept of a benchmark for total PAHs, assumes instantaneous mortality when thresholds exceed concentrations of 1 ppb (P3) and 0.1 ppb (P4) for each of the polyaromatics (including naphthalenes). Together, these parameter sets encompass a wide range of uncertainty in threshold levels for petroleum compounds, an acknowledgment of the limitations of current knowledge and the potential for further reducing threshold values in the future.

2.1.3. Oil transport and behavior model (Oil)

The Oil model consists of a three-dimensional multi-phase Lagrangian particle model for predicting the fate and transport of marine oil spills (Reed, 2004) (www.sintef/oscar). The behavior and fate of oil during a spill is governed by the physico-chemical properties of the oil, environmental weathering processes, and hydrodynamic conditions. A chemical group approach (Reed et al., 1999) tracks oil throughout different phases (surface, entrained droplets, and dissolved) using 25 chemical groups (Table 3). The approach assumes that individual hydrocarbon constituents in a given group behave similarly (i.e., have similar distributions and fates in the environment). The model predicts spatiotemporal dispersal along with buoyancy effects on vertical displacements, hydrocarbon dissolution, and hydrate formation (Vikebo et al., 2013). The model has been applied in historical oil spill events and compared against predictions (Broström, 2011). The Oil model supplies concentration fields for oil chemical groups (Table 3) to the Ecotox ELS subroutine within the Cod ELS model. In the present work, we simulate discharges for two different oil types, Balder and Draugen with properties shown in Table 3.

At the end of a simulation, we quantify individual cumulative survival probability through pelagic stages to ensure that the influence of the oil spills have become negligible. For poorly understood reasons, natural survival during the pelagic stages is highly variable (Houde, 2008). Therefore, we quantify the cumulative difference in survival for identical simulations with and without oil. We do not transfer the number of larvae directly to the fish population model, rather we transfer the reduction in survival, and use this to modify the number of recruits estimated within the fish population model.

2.2. After settlement

The fish population model for cod from age 1 + computes its own annual estimates of recruitment, based on annual deviations from a Beverton and Holt SSB-recruitment function. The recruitment function parameters are estimated, along with all other model parameters, by tuning the model to the available fisheries and survey data. The fish population model then modifies these recruitment estimates by

Table 3
The classification system for hydrocarbon compounds and concentrations of chemical groups (% by weight) in Draugen and Balder oil. The oil transport and behavior model simulates the distribution of chemical groups rather than individual compounds. The approach assumes that hydrocarbons in a given group behave similarly, i.e., have similar distributions and fates in the environment.

Group	Identification	Compound	Draugen	Balder
1	C1–C4-saturates	C1 to C4 gases	4.50	0.00
2	C5-saturates	n-pentane, isopentane, cyclopentane	3.50	1.37
3	C6-saturates	n-hexane, 2-methylpentane, 3-methylpentane, methylcyclopentane, cyclohexane	3.47	1.35
4	Benzene	Benzene	0.03	0.02
5	C7-saturates	n-heptane, 3-methylhexane, 2,3-dimethylpentane, methylcyclohexane	3.50	1.34
6	C1-benzenes	Toluene	0.35	0.10
7	C8-saturates	n-octane	6.15	2.09
8	C2-benzenes	Ethylbenzene; o-, m-, p-xylene	0.90	0.20
9	C9-saturates	n-nonane	4.66	3.65
10	C3-benzenes	Propylbenzene, 1-methyl-3-ethylbenzene, 1-methyl-4-ethylbenzene, 1-methyl-2-ethylbenzene, 1,3,5-trimethylbenzene, 1,2,4-trimethylbenzene, 1,2,3- trimethylbenzene	0.94	0.26
11	C10-saturates	n-decane	3.50	2.74
12	C4–C5-benzene	n-butylbenzene, 1,2,3,4,5-tetrametylbenzene, n-pentylbenzene	0.02	0.01
13	C11–C12	C11-C12 total saturates + aromatics	9.43	4.53
14	C0–C4-Phenols	C0- to C4-phenols	0.05	0.11
15	Naphthalenes 1	C0- to C1-naphthalenes	0.16	0.54
16	C13–C14	C13-C14 total saturates + aromatics	8.34	5.94
17	Naphthalenes 2	C2- to C3-naphthalenes	0.32	1.20
18	C15–C16	C15-C16 total saturates + aromatics	4.68	6.34
19	PAH-1	C4-naphthalenes, biphenyl, acenaphthylene, acenaphthene, dibezofurane, C0- to C1-fluorenes, C0- to C1-phenanthrenes/ anthracenes, C0- to C1-dibenzothiophenes	0.21	0.70
20	C17–C18	C17-C18 total saturates + aromatics	3.29	8.83
21	C19–C20	C19-C20 total saturates + aromatics	5.50	7.12
22	UCM	Unresolved chromatographic materials	0.00	0.00
23	C21–C25	C21-C25 total saturates + aromatics	7.24	9.58
24	PAH-2	C2- to C3-fluorenes, C2- to C4-phenanthrenes/anthracenes, C2- to C4-dibenzothiophenes, fluoranthrene, pyrene, C1-to C3-fluoranthrenes/pyrenes, benzo(a)anthracene, C0- to C4-crysenes, benzo(b,k)fluoranthene, benzo(e,a)pyrene, perylene, dibenzo(a,h)anthracene, benzo(g,h,i)perylene, indeno(1,2,3-c,d)pyrene	0.26	1.10
25	C25 +	Longer alkanes	29.00	40.88

applying the reduction in survival computed by the Cod ELS model to a range of possible recruitment years, allowing a wide range of different situations to be examined.

Cod ELS includes fixed mortality rates for eggs and only mortality as a function of prey availability until July. After July the diversity in prey preferences increases and we no longer consider natural mortality in Cod ELS. By around June–July, the cod ELS have mostly gone through metamorphosis becoming pelagic juveniles–young fish transported toward nursery grounds in the Barents Sea. Around August pelagic juveniles settle close to the seabed and become rather stationary. At this stage they are called 0-group.

There is a gap between the two models that must be bridged. The Cod ELS model (Section 2.1) simulates the development and distribution of pelagic stages until juveniles descend to the seabed and settle while the juvenile and mature cod model (Section 2.3) simulates fish starting from the end of their first year of life (January 1). Hence, the time between settlement and until January 1st the following year is not covered by either model. We therefore applied a density-dependent mortality function to adjust our abundance estimates from the Cod ELS model during this period. Density-dependent mortality includes cannibalism as a mediator of population size for juvenile cod. Without such density dependence, the mature cod model would over-estimate the abundance of juveniles recruited to the fish population.

We derived the density-dependent mortality function from annual abundance data collected during field surveys (Bogstad et al., 2015). The density-dependent mortality function is of the form $M = a * \ln(x) + b$, where x is the abundance at the start of the 0 group stage, and a and b are estimated internally through an optimization process. The final optimized density-dependent mortality function is $M = 0.51 * \ln(x) + 0.44$. This accounts for approximately 50% of the variation in the data ($R^2 = 0.48$). The remaining 50% is annual variability based on other factors in the environment, such as food availability, predation, temperature, and currents. These annual variations are addressed by estimating an annual recruitment variation value that gives the best fit to the data over the life span of the recruiting fish.

After adjusting the obtained survival percentages using the density dependent mortality function, these estimates are calibrated using the historical record (1985–2012) of the abundance of juvenile recruits in the age-structured Barents Sea cod population. By calibrating differential abundance estimates across all stock survey years (1985–2012), we obtain a more robust estimate of the potential changes in the abundance of juvenile recruits for a given year. These differential abundance estimates represent the predicted juvenile recruitment stock that transfers to the Barents Sea cod population after an oil spill. These estimates are then used to assess the impact and post-spill recovery for the population (Section 2.3).

2.3. Juvenile and mature cod

We used a forward-simulation model of fish population dynamics, Globally applicable Area-Disaggregated General Ecosystem Toolbox (GADGET), to simulate adults and juveniles from age 1 (Howell and Bogstad, 2010; Lindstrøm et al., 2009) (Fig. 2). This model simulates whole populations of a given species rather than the movement and distribution of individual organisms (http://www.hafro.is/gadget). GADGET performs forward simulations of biological processes (growth, predation, maturation, etc.), including both bottom-up and top-down effects and spatial and temporal variations in species interactions (Guldbrandsen Frøysa et al., 2002). The main state variables are the number and mean weight of individuals in each age/length group for a given population and area. GADGET models are the approved ICES stock assessment methodology (ICES, 2015). The version we use is calibrated with 26 years of fishery monitoring data and has been through several rounds of peer review in academic publications and in an assessment context (i.e., ICES stock assessments). Using this model, we assess the impact and track recovery for the harvestable cod population over more than 10 years post-spill, by which time an affected cohort of juvenile cod is reproductively mature. We assessed the population impact and post-spill recovery based on the change in the aggregated weight of mature fish in the cod stock, termed the spawning stock biomass (SSB). The population structure is modulated by the Harvest Rule (regulated fishing quotas) and predation (loss of fish due to predator/prey interactions) for each year. We assume that the fishing intensity (i.e., the fraction of the stock caught) is the same for identical scenarios, with and without oil. Note that we apply the additional oil-induced mortalities arising from the Cod ELS model to each year of recruitment in the fish population model. This was done to increase the range of possible stock states that an oil spill could affect.

In addition, we use this model to examine the role of reduced annual fishery quotas as a mitigation action after an oil spill. In this hypothetical exercise, we assessed the case of a 100% loss of recruits in 1995, systematically applying reduced fishery quotas in order to determine the optimal level of reduced fishing required to protect the SSB from a single catastrophic loss. This exercise demonstrates the utility of reduced fishing as a mitigation action.

2.4. Limitations

All models represent contemporary knowledge translated into simplified mathematical expressions. We acknowledge that current knowledge remains incomplete on the life history of the early life stages of fish and their primary prey and predators, as well as the toxico-kinetics and toxico-dynamics of petroleum compounds for these organisms. In the current version of the model, we only consider chemical uptake through water and not through food and describe the toxicity of the mixture by assuming the independent NEC model (Baas et al., 2009). We do not explicitly take into consideration the effect of photo-enhanced toxicity (Barron et al., 2004; Buskey et al., 2016). In addition, the population size of fish larvae is assumed to be density-dependent; given a finite amount of prey, larval mortality increases as a function of population size (Houde, 2008). Few studies and limited data exist to inform models on the vertical behavior of prey and fish larvae. And the causes of spatio-temporal variation in the natural mortality of early life stages are not fully understood. It would also seem plausible that some components of the spawning contribute disproportionately to the recruiting yearclass, and thus that the oil spills affecting different parts of the larval distribution may have non-linear impacts on the overall fish population. However, the data to model this is lacking, and therefore this spatial structure in survival is not included in the current modeling.

The simulations performed in this investigation are limited to the assessment of effects of oil exposure on cod stocks based on a single year's decrease in recruitment. It does not include other impact factors, such as other contaminant sources. In all simulations, oil is assumed to not be present after one year and all surviving cod are healthy. Effects of oil spills on other fish species and ecosystem components around Lofoten-Vesterålen are also not included. Despite both the recognized and currently un-recognized limitations, this modeling approach provides the opportunity to address ecological processes operating on relevant spatial and temporal scales and to conduct numerical experiments where real experimentation has heretofore been infeasible.

3. Results

3.1. Oil spill simulations

For oil spills with release rates of 1500 or 4500 m^3/day for 14, 45, and 90 days, the total volume of discharged oil was 18–349 × 10^3 metric tons (Table 1). A subsea release is at the sea bed (183 m water depth). This generates a positively buoyant plume rising to the surface. A topside release is at the water surface. Between 43 and 62% of discharged oil is lost through the processes of biodegradation and evaporation combined (Table 1 and Fig. 3). Biodegradation removes more oil for subsea discharges compared to topside, while the opposite is true for evaporation. For the two oil types, Draugen and Balder, Draugen contains a larger fraction of lighter oil components, resulting in a higher percentage of oil removal by evaporation (Fig. 3). Draugen scenarios also have more dissolved oil than the corresponding Balder scenarios, particularly for subsea discharges. Subsea discharge scenarios exhibit significantly more dissolved oil as well as higher maximum and mean concentrations (Fig. 3). While oil concentrations are similar for comparable topside and subsea scenarios, the composition of dissolved oil differs. For topside releases, PAH-type compounds dominate while for subsea releases, light/medium saturates dominate (data not shown).

3.2. Survival of cod ELSs linked to oil exposure

We simulated the uptake by drifting cod ELSs of temporally varying ambient concentrations of bioavailable oil compounds in the environment. There was no impact on cod ELS survival relative to a simulation without oil for any of the modelled oil discharge scenarios when applying toxicity parameter sets P1 and P2 (Table 1). Survival was reduced when assuming immediate mortality from exposure to low concentrations of polyaromatics (P3 and P4). Not surprisingly, the highest discharge rate (4500 m^3/day) combined with the longest duration (90 days) resulted in the highest reduction in survival (43%). For these simulations, average (± standard deviation) survival reductions were 4.6 ± 3.6% (P3) and 19 ± 11% (P4).

3.3. Recruitment losses for the mature cod stock

Recruitment losses due to major oil spills are an additional risk factor in maintaining healthy fish populations and fisheries. Following the cod population as it developed in response to reduced juvenile survival after oil spills in Lofoten-Vesterålen, we found no discernible reduction in SSB for toxicity sets P1 and P2. A minor reduction in SSB occurred for P3 (≤ 3%) whereas the most conservative toxicity parameter set (P4) had an impact on SSB of ≤ 12% (Table 1). In all cases, the stock remained at full reproductive potential. Despite the loss of ELSs due to exposure to oil, a sufficient number of juveniles survived to replenish the adult stock. The SSB returned to within 1% of the baseline SSB (without oil) within ~12 years of a spill, by which time the affected juvenile recruits entering the adult stock no longer comprised a significant fraction of the adult population. Thus, none of these oil spill scenarios and toxicity sets reduced the SSB to a point where the adult stock was impaired due to insufficient numbers of juveniles entering the adult stock in subsequent years.

3.4. Mitigating the impacts on fisheries

Reduced fishing is a mitigation action that may help ensure that a sufficient number of recruits mature so that the SSB does not drop below a sustainable limit. For cod, the 3–4 years between juvenile recruitment (age 3 years) and maturation (age 6–7 years) provides time to apply such mitigation measures. While the loss of 100% of recruits in a single year is beyond that expected after an oil spill, we considered this hypothetical case to determine the level of reduced fishing that would be needed to protect the SSB from a single catastrophic loss. A 100% loss would eliminate the contribution of the effected year's juveniles to the age structure of the SSB (Ohlberger and Langangen, 2015). Taking 1995 as an example, such a loss reduces the SSB by up to 16% (2001) and recovers to 99% of the default value by 2005 (Fig. 4). Over the 10 years, following the oil spill, there is a loss of 300,000 tonnes of catch (out of a total of 6,007,000 tonnes). Applying a 10% reduction in fishing effort for eight years post-spill, the stock recovers by 2003 (Fig. 4). There is an initial reduction in catch, but a shallower and

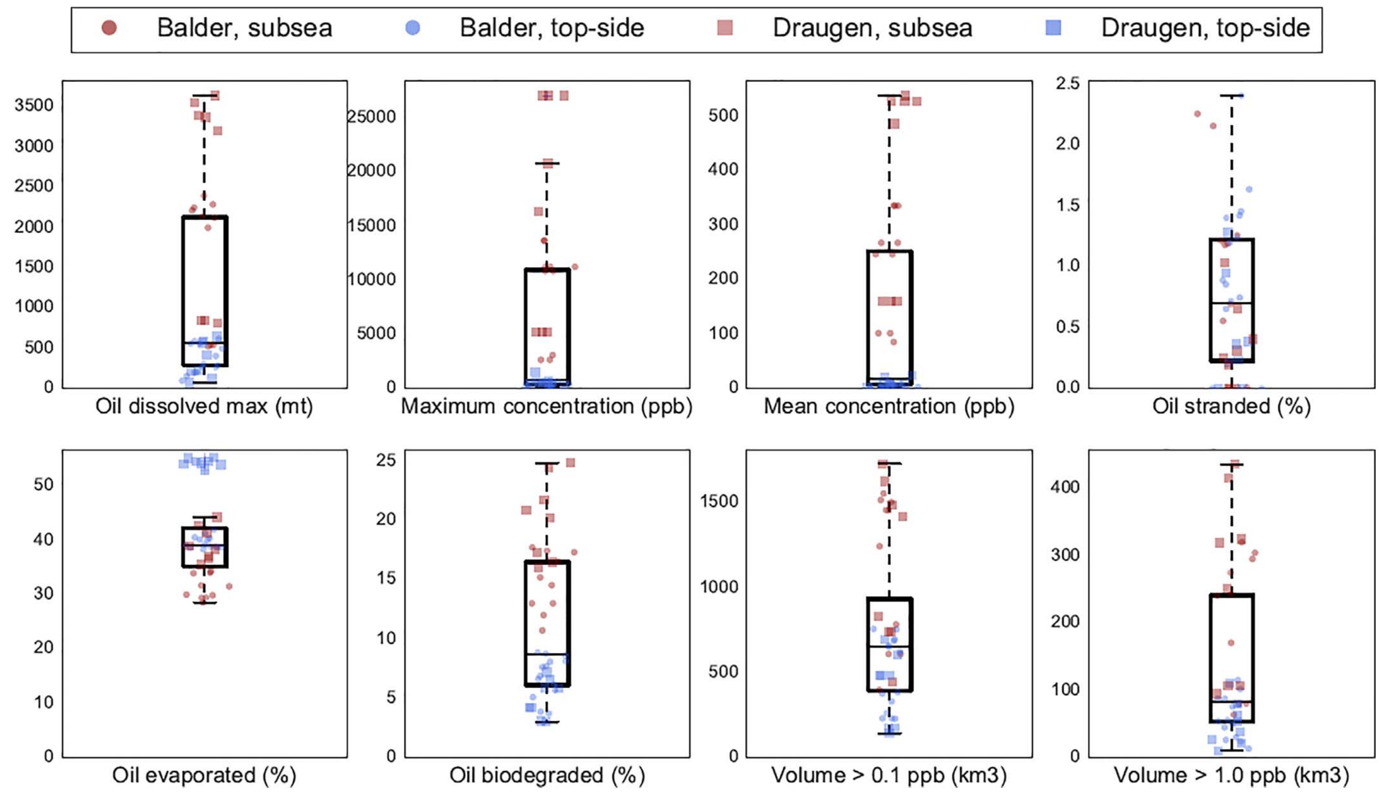

Fig. 3. Box-plot distribution of oil-related endpoints for all scenarios, colored by release type (subsea or topside), and marked by oil type (Draugen or Balder). The boxes show median and 25/75 percentile of values. The whiskers show extend to the smallest and largest observations that are not outliers (https://stats.stackexchange.com/a/149178). Abbreviations: mt = metric tonnes; ppb = parts per billion. Oil dissolved max = the maximum amount (in metric tonnes) of dissolved oil for each simulation. Maximum concentration = spatial and temporal maximum dissolved oil concentration in each simulation. Mean concentration = spatial mean dissolved oil concentration in each simulation. To obtain volume > 0.1 ppb/1.0 ppb, we calculate, for each time step, the volume of water where the dissolved oil concentration exceeded 0.1 ppb/1.0 ppb. The maximum of these values for all time steps in a given simulation is presented in the box plot.

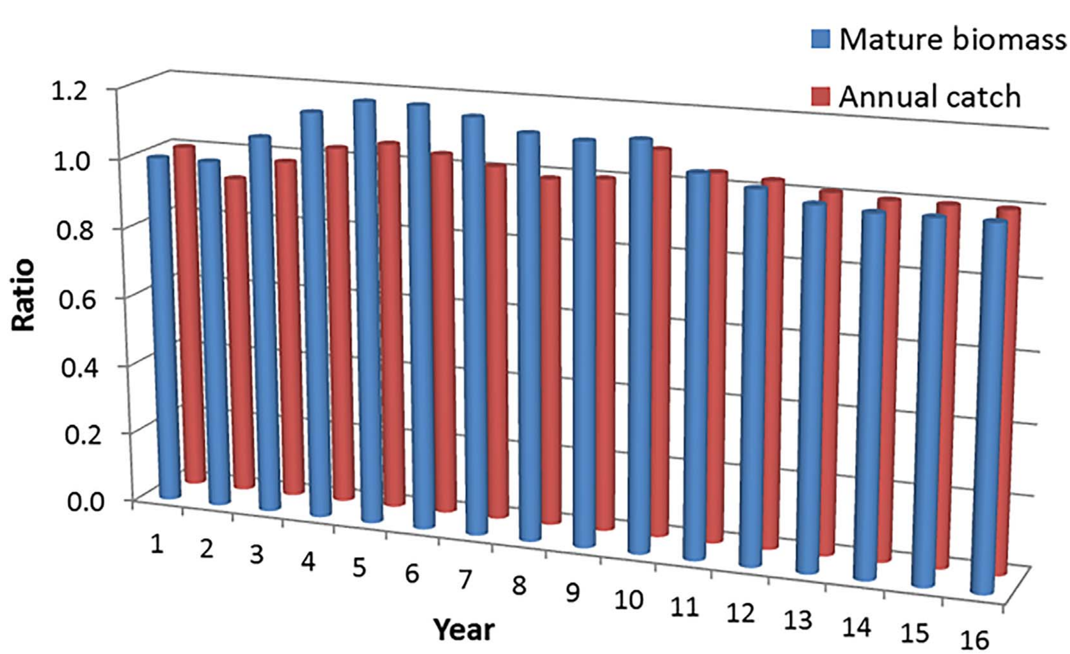

Fig. 4. Ratios of cod mature biomass and annual catch for the two cases: after an oil spill only and with a 10% reduction in fishing effort for 8 years post-spill (1996–2003). Application of a reduction in fishing effort decreases the overall loss of catch from 300,000 to 230,000 t. The year of the oil spill (Year 1) is 1995.

shorter dip in the stock more than compensates for the initial reduction. The loss of catch is reduced from 300,000 t (no mitigation) to 230,000 t (mitigation).

4. Discussion

We utilize the predictive capacity of an ecosystem-based modeling framework to extrapolate the effects of oil spills on individual fish eggs and larvae to recruitment losses in an adult fish population. The development of this framework was motivated by the need to increase current scientific understanding of the potential impacts of a major oil spill in a region with multi-sectoral interests and a government seeking to balance industrial development and environmental resource protection. We have simulated variations in the transport and dispersal of individual cod eggs and larvae and its primary prey, the zooplankton species, *Calanus finmarchicus,* together with changing environmental concentrations of oil compounds. We also apply a mechanistic toxicity model that accounts for time-varying oil exposures in the assessment of biological effects. Through modeling, we are able to examine relevant processes over their characteristic spatial and temporal scales. In most scenarios, the resulting impact on survival of cod ELSs, relative to a corresponding no-oil scenario, is low even when applying threshold levels to account for delayed toxic effects (P3 and P4). These reductions translate to reductions in adult stock biomass of $\leq 12\%$. Herein we discuss the main reasons for these findings.

4.1.1. Oil entering the sea evaporates and degrades

Our simulations indicate that cod encountering an oil spill in Lofoten-Vesterålen are exposed to transient and generally low concentrations of oil across a large geographic region of open marine waters. In the immediate aftermath of a spill, oil concentrates at the sea surface, facilitating evaporation of the most volatile compounds (French-McCay, 2004). In subsequent days, oil disperses through wind, mixing, and currents, and is broken down via photo-oxidation and biodegradation processes (French-McCay, 2002; Reddy, 2012). In our simulations, evaporation and biodegradation accounted for the loss of 43–61% of discharged oil (Fig. 3). These percentages are comparable to estimates obtained in several previous marine accidents (Buskey et al., 2016; French-McCay, 2003, 2004). Most of the remaining oil in our simulations became widely dispersed, with only a minor fraction of oil ($< 3\%$) reaching the shoreline. Thus, for these scenarios, unlike Prince William Sound where organisms were chronically exposed to oil stranded on the shoreline, we would not expect multi-generational effects linked to the persistence of weathering oil in shoreline habitats.

4.1.2. The population of Northeast Arctic cod is healthy

Time series of fish stock abundance exhibit complex oscillatory patterns (Bjørnstad and Grenfell, 2001; Bjørnstad et al., 2004). High frequency oscillations are attributed to environmental variability and species interactions, whereas low frequency oscillations are associated with external forcing factors, such as overfishing and climate variability. Considerable evidence has demonstrated that increased mortality makes a population more susceptible to stochastic environmental forcing (Botsford et al., 2014; Rouyer et al., 2012), rendering the population more vulnerable to increased mortality levels from intensive exploitation. This leads to low proportions of old and large individuals, truncating the age and size distributions of the population. This impact on the population is expected to be greatest for populations at the boundaries of their geographic distribution (Rouyer et al., 2014), as is the case for Northeast Arctic cod.

For a healthy stock, the typical recruitment pattern is a majority of poor to average years, with only a few highly successful years (Fig. 1 inset). Healthy fish stocks produce so many more eggs than can survive that recruitment success depending on the influence of environmental conditions on the survival of young fish rather than on the size of the adult population. Although cod in general is characterized as having high variability in recruitment, it is still less than for many other stocks (Yaragina et al., 2011). The Northeast Arctic cod stock in particular is characterized as having good recruitment years sporadically, meaning that some year classes represent an above average fraction of the spawning stock. The stock has also been subjected to low fishing pressure in recent years. Low fishing pressure combined with strong recruitment in 2004–2005 has resulted in strong growth and age diversity in the Northeast Arctic cod stock (Kjesbu et al., 2014). The stock size is above that required for full reproductive capacity, and the population is currently at least as resilient, and probably more resilient, to reductions in a single year's recruitment as in the 1990s and early 2000s. The current health of the Northeast Arctic cod population is considered a buffer from catastrophic declines in population such as a major oil spill event. We found that, under recent stock conditions, any single year of reduced recruitment would not drive the population below the SSB limit required to ensure full reproductive potential. However, it should be noted that multiple years of reduced juvenile recruitment may have a more severe impact due to an increase in the chance of pushing the population below this level.

4.1.3. A diverse age distribution protects the population from recruitment loss

Oil spills impact a single year-class of cod, while the adult population of cod consists of fish from many year-classes. The biomass of the adult cod stock in the Barents Sea consists of fish ranging in age up to > 13 years. Species that have several year-classes contributing to the biomass of the population are weakly impacted by severe mortality events affecting a single generation of individuals (e.g. Ohlberger and Langangen, 2015). As a result, major losses to a single year-class do not directly translate into equivalent changes in the abundance of the SSB. In the present examination of the impact of oil spills, the maximum observed reduction in the recruitment stock in parameter set P3 ($\leq 14\%$) translated to a reduction in SSB of $\leq 3\%$, a ratio of ~ 5. Similarly, for P4, the maximum reduction in recruitment stock ($\leq 43\%$) translated to a reduction in SSB of $\leq 12\%$, a ratio of ~ 4. These losses, obtained for a single year-class, do not impact the population biomass or the future reproduction of the Northeast Arctic cod population.

Fish with shorter lifespans (e.g., capelin) or with more variable recruitment (e.g., herring or haddock) are expected to be more vulnerable to losses in a single recruitment event (Botsford et al., 2014) as demonstrated with the post-spill investigations of the Prince William Sound herring population (Incardona et al., 2015). An overfished stock also lacks the buffer capacity of a large and diverse adult population which makes it more vulnerable to impaired recruitment (Petersen, 1903; Rouyer et al., 2014). Though we have investigated a specific perturbation factor in detail, namely toxic exposure of cod ELSs to oil, our findings correspond well with previous investigations of the ecological drivers behind species resilience to generic catastrophic mortality events (Ohlberger and Langangen, 2015; Vincenzi et al., 2014).

4.1.4. Reduced fishing may reduce losses to the adult population after an oil spill

Reduced fishing is a mitigation action to ensure that, as impacted recruits become mature adults, the adult biomass does not drop below a sustainable limit. In this hypothetical example of a 100% loss of recruits in 1995, 6 years after the event there was a maximum reduction in SSB of 16%. The SSB recovered four years later to within 99% of the default (no oil spill) value. Applying a fishing reduction of 10% for eight years post-spill, the stock recovered 2 years after the peak reduction and the biomass loss was reduced. This example illustrates that prompt application of reduced fish quotas after an oil spill by fishery managers can help mitigate the damage to both the stock and the fishery.

5. Conclusions

The development of an integrated modeling framework to predict the impact of oil spills of varying discharge volumes and durations on juvenile recruitment to the adult population of Northeast Arctic cod provides opportunities to explore questions that cannot be feasibly addressed through other scientific approaches. Fish eggs and larvae are sensitive to the toxic effects of oil, but the impact on individuals depends on the conditions of oil exposure in the environment. Most of the simulated oil spills in Lofoten-Vesterålen resulted in generally low concentrations of bioavailable oil compounds in open sea areas that resulted in minor reductions on the survival of pelagic stages of cod even with the application of toxic threshold values designed to cover narcotic, physiological, and behavioral effects. However, the most severe oil spills, i.e. highest discharge rates and longest durations, resulted in population losses up to 43% for the pelagic stages. For the various spill scenarios presented in this study, recruitment of juveniles into the adult stock remained sufficient to maintain the reproductive health of the population. Most oil spill scenarios reduced the biomass of the adult stock by < 3% with a worst-case scenario leading to a 12% reduction in the adult stock biomass. We discuss several factors that protect the Northeast Arctic cod population from severe recruitment impairment from a major oil spill in Lofoten-Vesterålen: significant oil evaporation and biodegradation which limits exposure of drifting eggs and larvae to toxic levels of petroleum compounds in the sea; the population of Northeast Arctic cod is considered healthy and thereby more resilient to recruitment losses; and the diverse age distribution of the mature population protects the stock from recruitment losses. These model results must be applied cautiously, particularly in a management context as they are inherently simplifications and incorporate the limitations in our current knowledge of complex natural processes.

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