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The mussel path – Using the contaminant tracer, Ecotracer, in Ecopath to model the spread of pollutants in an Arctic marine food web

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ABSTRACT

As the polar ice cap is receding, shipping in the Arctic seas becomes easier, and both destination and Atlantic–Pacific transit shipping is expected to increase. Thereby, the risk of accidents increase. Immediate negative impacts are expected from oil spills through the acute mortality for marine organisms, especially from heavy fuel oil (HFO). Marine Diesel oil (MDO) is therefore suggested as a preferable fuel for ships operating in Arctic waters. However, Polycyclic Aromatic Hydrocarbons (PAHs) are toxic components in both types of fuel, are highly bioavailable and can transfer up the food chain. A spill of MDO following a shipwreck could therefore have impacts beyond the spill site and long after the diesel has spread and evaporated. We model the spread of PAHs from a fictitious spill of MDO in the Pechora Sea (South Eastern Barents Sea) using the contaminant tracer module Ecotracer, in the Ecopath modelling software. We address the effects on the food-web including long term effects by combining toxicology and food-web modelling. Ecotracer assumes that pollutants follow the biomass passively through the system, and degradation of pollutants is following user specified rates. By combining in natura measurements of PAHs in seawater and in blue mussels (*Mytilus edulis*) recorded at an accidental MDO spill site, with experiments conducted on the red king crab (*Paralithodes camtschaticus*) and blue mussels, we derived values as inputs into the model. The Ecotracer predicted that the pollution in the mussels will spread throughout the food-web, especially to the top predators of mussels, king eider (*Somateria spectabilis*) and Atlantic walrus (*Odobenus rosmarus rosmarus*) and also from snow crab (*Chionoecetes opilio*) to seals and toothed whales.

1. Introduction

The Pechora Sea (67–71 °N, 44–60 °E) in the Russian Arctic is situated in the south-eastern Barents Sea (Fig. 1) and is considered to be a separate sea area because of marked differences in environmental conditions compared to the rest of the Barents Sea. The area is an important spawning ground for Arctic fish and is rich in seabirds and mammals that feed on benthic invertebrates (Boltunov et al., 2010). The Atlantic walrus (*Odobenus rosmarus rosmarus*) occur in the Pechora Sea, and one of very few population estimates indicates a summer population of 3943 animals (Lydersen et al., 2012). Walrus feed on benthic organisms in shallow waters, and haul out on either low elevation beaches, or on the sea ice. The Pechora sea is identified as an area of high ecological importance by the Arctic Monitoring and Assessment Programme (AMAP/CAFF/SDWG, 2013) based on criteria set by the International Maritime Organization (IMO, 2006). The coastline includes many low-level marshes and wetlands which are exposed to frequent and long-term flooding during spring and summer, as well the abrasive effects of sea ice. Muddy, shallow water coasts are characterized by high abundances of mussels. In the case of an oil-spill, shallow, soft sediment areas are known to accumulate oil, as seen, for example, after a spill on the coast of Massachusetts, USA in 1969 (Culbertson et al., 2008).

As the polar sea ice cap recedes, shipping and industrial activities become easier in the Arctic seas. Increased activities mean increased risk of accidents. The Pechora Sea is one of the areas expected to hold large sub-seabed deposits of hydrocarbons. The first offshore oilfield in the Pechora Sea, Prirazlomnoye, 60 km off the Siberian coast (Fig. 1) at a water depth of 20 metres started production in 2014. The oil from the Prirazlomnaya installation is exported by ice strengthened tankers.

This modelling exercise is based on a fictitious ship wreck (Larsen et al., in press) where the container vessel *MV "Oleum"* suffers an engine malfunction and runs aground on the north western corner of Vaygach (Fig. 1), thereby releasing approximating 200 tonnes of MDO. To assess the environmental impact of an oil

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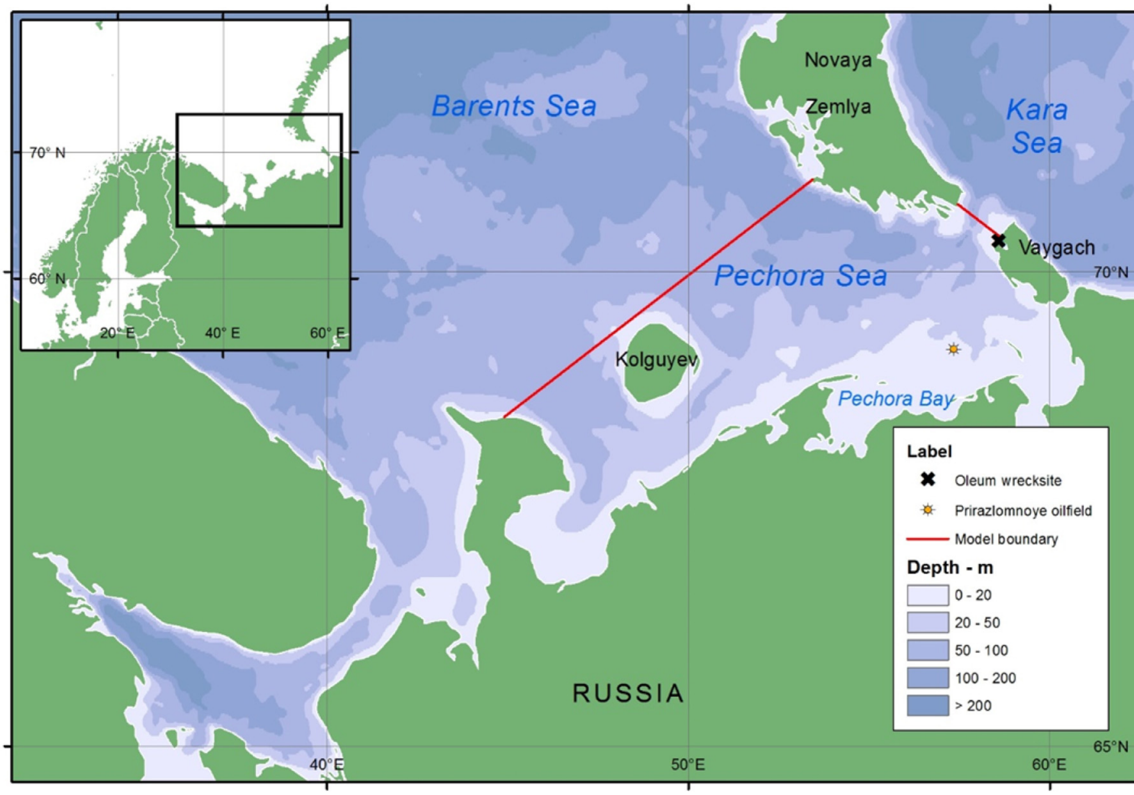

Fig. 1. Location and seabed topography of the Pechora Sea, the south-eastern part of the Barents Sea. Red lines indicate model boundary. Model area is approximately 125 000 km^2. Fictional wreck site of "Oleum" is indicated.

spill we are combining ecotoxicology and ecological modelling. We apply results from ecotoxicology studies done in our laboratory (MDO exposure of blue mussels (*Mytilus edulis*) and red king crab (*Paralithodes camtschaticus*) (Sagerup et al., unpublished data), and verify its realism by applying measurements performed upon an accidental release of 180 tonnes of MDO in Skjervøy harbour, Northern Norway (70 $^{\circ}$N, 20 $^{\circ}$E) in December 2013 (Sagerup and Geraudie, 2014).

1.1. Toxicology of PAHs and food web modelling

Polycyclic aromatic hydrocarbons (PAHs) are lipophilic hydrocarbon components and may therefore be susceptible to bioaccumulation (NB not biomagnification). PAH exposure is associated with narcosis, mutagenicity, carcinogenicity, embryotoxicity, genotoxicity, cellular damage, endocrine disruption and reduced survival of larval fish (Moore et al., 1989; Baussant et al., 2009; Bechmann et al., 2010; Nahrgang et al., 2010). PAHs are suspected to be responsible for several of the biological impacts recorded after the 1989 Exxon Valdez spill in Alaska, USA, such as increased egg mortality, and reduced survival rates and growth in pink salmon (*Oncorhynchus gorbuscha*) (Peterson et al., 2003). In molluscs, population effects such as reduced recruitment, increased mortality and reduced production have been shown in the field following exposure to petroleum hydrocarbons in the sand gaper (*Mya arenaria*) (Gilfillan and Vandermeulen, 1978) and in mesocosm studies for blue mussels (Bakke and Sørensen, 1985; Widdows et al., 1985).

Toxicological models generally focus on the dynamics of the chemical, the kinetics and the model may be limited to the fate within one organism. The physiological responses are extremely complex and therefore the food-web must be simplified (e.g., Thomann, 1989). However, combining ecotoxicology and food-web modelling is important to be able to address effects of pollutants at an ecosystem level. Ecopath has been used to model the spread and accumulation of pollutants or toxins, e.g., the fate of dioxins (Carrer et al., 2000), and to compare how ecosystem structure dictates mercury concentration (Ferriss and Essington, 2014), however both of these studies chose to not use the Ecotracer module. Booth and Zeller (2005) assessed the implications of mercury accumulation for human health using the Ecotracer module. Our work focuses on testing whether we can use Ecotracer for a major single discharge of pollutants, exemplified by a MDO spill from a ship wreck.

The main interest of this work is to test and evaluate how Ecotracer works. Is Ecotracer able to model the spread of PAHs in an Arctic marine food web at levels likely to occur after an accidental spill? What are methodological challenges and despite challenges, what can we learn from using Ecotracer? To answer these questions, we investigate the effects of MDO that contain bioavailable PAHs on the marine environment, and assess the spread in the food-web.

2. Materials and methods

A fictitious scenario describing a cargo ship voyage from Hamburg (Germany) to Yamburg (Russia) forms the basis for our modelling exercise. The case study (Larsen et al, in press) describes the rescue operation and outlines potential environmental effects from the loss of MDO and cargo from rupturing containers.

To investigate the sensitivities of organisms representing ecosystems along current and future Arctic shipping routes, laboratory experiments with exposure to MDO were performed (Sagerup et al., unpublished data). Two species of mussel, the Icelandic scallop (*Chlamys islandica*) and blue mussel, were exposed to dispersed MDO. As part of the experiment the exposure, trophic transfer and recovery were studied in red king crab. Impacts of discharges of MDO were also studied in natura, after the accidental release of 180 000 l of MDO in the harbour of Skjervøy, 14th December 2013 (Sagerup and Geraudie, 2014). Blue mussels were used to assess uptake after the spill by analysing local mussels and by placing uncontaminated specimens in cages in the harbour five days after the spill. Total PAH levels were measured in the mussels after five days, one month, two months, three months and one year (Sagerup et al., unpublished data).

2.1. Input to Ecopath model for the Pechora Sea

The Ecopath model (Polovina, 1984; Christensen and Walters, 2004) is a mass balance modelling approach based on a set of linear equations representing flow of biomasses between groups in the ecosystem. The "mass balance" term refers to the physical constraint of the model parameters describing the system to be in "balance". This occurs when the flows into a group equal the flows out of the group and mortality for a prey equals consumption by a predator. Ecopath is the base model representing a snapshot of the system. The Ecopath model balances losses and gains for each functional group using Eq. (1).

$$Bi * \left(\frac{P}{B}\right) i * EEi = BAi + Yi + \sum_{(j=1 \quad \text{to} \quad n)} Bj * \left(\frac{Q}{B}\right) j * DCij \tag{1}$$

where B is biomass, P/B is production per biomass and EE (ecotrophic efficiency) is the fraction of production transferred within the model, BA is biomass accumulation and Y is mortalities due to fisheries and hunting. Q/B is consumption per biomass ratio and DC is the fraction of group i in the diet of group j.

Our model area (the Pechora Sea) covers an area of approximately 125 000 km^2 (Fig. 1). We have defined 27 functional groups in the model, including four groups of mammals, two groups of birds, seven groups of fish, 10 invertebrate groups, two groups of primary producers and two detritus groups (Appendix A).

Most of the production and consumption values are derived from, or compared to, an Ecopath model made for the Norwegian and Barents Seas in 2002 (Dommasnes et al., 2002). Recordings from the Pechora Sea were used to calculate the biomass of whales (Boltunov et al., 2014), seals (Boltunov et al., 2014), walrus (Lydersen et al., 2012), seabirds (Spiridonov et al., 2011) and ducks and eiders (Strøm et al., 2000; Spiridonov et al., 2011). Most data from the Pechora Sea have been collected during summer and autumn, and with the lack of seasonal data, we had to assume year-round validity for the biomass data.

For the fish groups, data on biomass were calculated based on Prokhorova (2013). For Atlantic cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*) and long rough dab (*Hippoglossoides platessoides*) biomass estimates were supplied by the Polar Research Institute of Marine Fisheries and Oceanography, Murmansk, Russia (PINRO) (pers. comm., Dmitri Prozorkevich). For the benthic groups, data from Denisenko et al. (2003) were used and the calculated P/B and Q/B values from Ullsfjord (69–70 °N, 20 °E) in Northern Norway (Nilsen et al., 2006). Snow crab (*Chionoecetes opilio*) is an invasive species with an increasing biomass in the Barents Sea (Sundet, 2015). The biomass of snow crab was estimated by the model, but diet (Kolts et al., 2013), production and consumption are estimated based on values from the Eastern Bering Strait were the species occurs naturally (Aydin et al., 2007). For zooplankton, data from Dvoretsky and Dvoretsky (2009, 2015) were used for biomass calculations. Detritus groups were adjusted to sustain the large benthic production in the Pechora with food. A detailed description of the assembly of the Ecopath model is given in Appendix A.

2.1.1. Quality of model and ecological robustness

Link (2010) outlined a set of comparisons of input data, ratios and information to be performed in advance of any fitting of the model (PREBAL analysis). These are described as a set of "rules of thumb" to apply in an early search for outliers (unrealistically high or low values), and identify needs to reevaluate any of the data attached to the functional groups in the model. For our Ecopath model, a PREBAL diagnostic identified a relatively fair set of biomass input values. Both in absolute values and related to primary production (Appendix B).

2.2. Ecospace

Ecospace is the spatial module of Ecopath, which dynamically allocates biomass across user defined grid cells (Christensen and Walters, 2004). We used a 20 × 30 grid cell map (Fig. 2) of the model area with 23 km × 23 km cells. Four habitats were defined, <20 m, 20–50 m, 50–100 m, >100 m water depth. The spatial model is two-dimensional, so the depth is merely a name for the type of habitat. All groups in the model were assigned in proportion of the population to the different habitats. The group "nearshore bivalves" was fully assigned to the habitat <20 m, while offshore bivalves were assigned with 0.7 (70%) of the population in 20–50 m and 0.3 (30%) in 50–100 m. Seaweed were assigned as being all in <20 m, and cod and haddock both 50% to 50–100 m and 50% to >100 m. All other functional groups are by default "everywhere" (25% per habitat).

2.3. Ecotracer

The contaminant module of Ecopath, Ecotracer, uses the flow of biomass between the functional groups and the environment and predicts concentrations of contaminants that flow passively with the biomass in the food-web. The contaminants are assumed to follow the biomass passively and instantaneously. The model also allows for direct uptake from the environment, for example across the gills. Decay rate of the pollutant is also specified for each functional group in the model and in the surrounding environment (water).

We investigated how different inputs would influence the results in Ecotracer by testing three different input combinations (Table 1).

1. Case 1: The level of PAHs in mussels and red king crab after exposure in our laboratory experiments were used as the initial concentration assuming similar uptake rates in the field as in the laboratory. Looking at the spread only through diet but using no direct uptake, e.g., over gills.
2. Case 2: The same initial concentration of PAHs as in case 1 but including direct uptake from the environment as calculated from the concentrations in water.
3. Case 3: The scenario of a sudden release of MDO. Zero initial concentration of PAHs. PAHs uptake from water and food. A maximum concentration of PAHs was defined for the groups of nearshore and offshore bivalves and snow crab so that the output from Ecotracer was similar to the input values in cases 1 and 2.

The following parameters were entered as a basis for the calculations:

a. Initial environmental concentration *Co*: we used 0.0104 tonnes/km^2, a value recorded in the harbour at Skjervøy 5 days after the spill.
b. Decay rate (/year): we used 10 tonnes/year, a generic high value as we expect the MDO with the PAHs to disperse, evaporate and degrade rapidly unless taken up by biota.

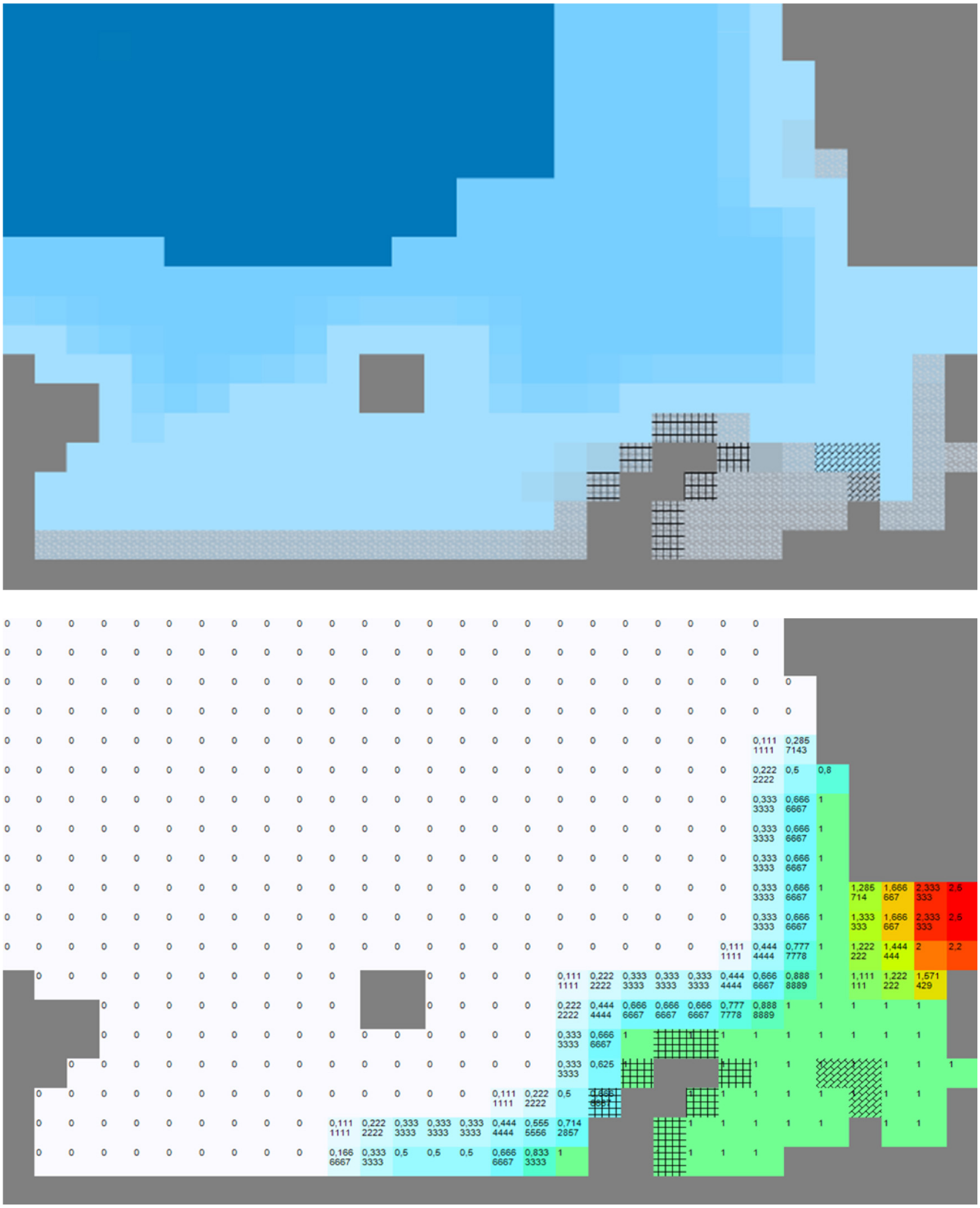

Fig. 2. *Above*: Habitats layer in Ecospace for the Pechora Sea. Dark blue area are >100 m water depth, lighter blue, 100–50 m, blue/grey 50–20, light grey <20 m water depth. Dark grey: land. *Below*: The contaminants layer used in Ecotracer, red is high concentration of PAHs, and gradually decreasing until the white area, containing no PAHs. Grey is land. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

c. Base inflow rate of the contaminant to the ecosystem (tonnes/km^2/year): we used 0 as we are looking at a one-time event.
d. Base volume exchange loss of contaminant (/year): we used 0, and this is not relevant when there is no inflow.

For each functional group, we specified:

e. Initial concentration in tonnes pollutant per tonne biomass (t/t): in case 1 and 2 we used observed concentrations from the experiments and in case 3 we used 0.
f. Concentration in immigrating biomass (t/t): We used 0 for every group as we haven't estimated any immigrating biomass.
g. Direct absorption rate (tonnes/tonnes/year), for example uptake over the gills: we used 0 for every group in the first run, in the second run we used uptake rates calculated from the exposure experiments for the benthic groups including crustaceans. In the third run we used the rates that gave the observed concentrations in the experiments.
h. Decay rate (tonnes/year): we used 10 tonnes per year for warm-blooded groups, mammals and seabirds, 1 tonne/year for all other groups except the detritus and phytoplankton groups where we used 0.

Table 1
Measured values used as input to Ecotracer. Accumulation ratios obtained after a one week study. The polycyclic aromatic hydrocarbons (PAHs) were measured in soft tissues of blue mussels and Islandic scallops and the hepatopancreas of the red king crab. All tissue concentration in μg/kg wet weight (wet wt) and water concentration (μg/L).

Group/environment	Ecopath group	Data	Water concentration	Accumulation ratio
Blue mussel (*Mytilus edulis*)	Nearshore bivalvia	4466 (field)4482 (lab)	18	249 (one week)
Icelandic scallop (*Chlamys islandica*)	Offshore bivalvia	2957 (lab)	18	169 (one week)
Red king crab, (*Paralithodes camtschaticus*)	Snow crab	22 254 (lab)	18	1236 (one week, accumulation from feed and water)
Skjervøy harbour seawater 5 days after spill	Initial concentration, Environment		10.4[a]	

[a] Assumed to stay mostly at top 1 m of the water column.

i. Proportion of contaminant assimilated [0–1]: we used 0.7 for all groups.

All the equations are described in the Ecopath user manual (Christensen et al., 2008) and described in Appendix A.

Our initially measured PAHs concentration in water ($\sum PAH_{16}$) from the harbour of Skjervøy, following the accident was 10.4 μg/L. 1000 L per m^3 gives 0.0104 g/m^2 of PAHs, assuming that the diesel stays mostly at the surface, top 1 m. This was used as the initial environmental concentration.

From the experiments in the laboratory we started by using 0.000004466 tonnes contaminant/tonne biomass (4466 μg/kg wet weight (wet wt)) for nearshore bivalves. From the experiments with blue mussels, 0.000002957 tonnes/tonne (2957 μg/kg wet wt) for offshore bivalves, taken from the experiments with Icelandic scallop. Finally we used 0.0000222548 tonnes/tonne (22 258 μg/kg wet wt) from red king crab, used for the snow crab. These were used as the initial concentrations for the corresponding groups in cases 1 and 2 and the upper limit for these groups in case 3.

The direct uptake ratio was calculated from the experiments. In the high exposure group, the level of PAHs was 17.94 μg/L and after one week of exposure, the blue mussels had a concentration of 4482 μg/kg wet wt. This means there is a high ratio of accumulation from water to mussels. The Icelandic scallop also had a high ratio of uptake from the water with the same level of PAHs, reaching a soft tissue concentration of 2957 μg/kg wet wt. Red king crab were exposed both through feed and water and accumulated much higher tissue concentrations than mussels (22 254 μg/kg wet wt). From this we assumed a high degree of trophic transfer and used 0.7 as assimilation efficiency in Ecotracer.

We compared the model using zero as the direct uptake for every group in the first case study to eliminate this as a variable, while in the second case we ran with the high uptake rates measured in the laboratory: a ratio of 249 for nearshore bivalves and 150 for offshore bivalves. For snow crab, we used 250 as well even though the observed concentration was much higher than in the bivalves.

The third case study used zero initial concentration for all the groups and the direct uptake was adjusted to get comparable levels of PAHs. By using a direct uptake ratio of one and an assimilation rate of 0.7 a similar initial concentration, as measured in the experiments (Sagerup et al., unpublished data), was achieved for nearshore and offshore bivalves. For the snow crab, we reduced the direct uptake ratio to 0.2 as they are not filter feeders.

Our three model runs were designed to investigate a momentary release of MDO from the wreckage of a ship. The PAHs was added as a layer in Ecospace with high concentration near the wreck site at Vaygach, and with rapidly decreasing concentrations with distance to the spill (Fig. 2). Ecotracer was run with Ecospace as a spatial model with PAHs as a contaminant layer.

3. Results

3.1. Ecopath

Building an Ecopath model is an informative exercise that generates a knowledge base on biological components of an area and identifies any existing knowledge gaps. Even though mussels are the preferred prey of the estimated almost 4000 walruses, 24 000 eiders, snow crab and many species of fish, the ecotrophic efficiency (EE) for the offshore bivalve was estimated by the model to be 0.126 and nearshore bivalves to be 0.150. The EE is estimated on a scale from 0 to 1, where 1 would mean all production is consumed. The EE is low for several of the benthic groups. This means there is a lot of benthic production not being consumed by predators.

3.2. Ecotracer

PAHs levels spread fast in the Pechora food web and especially to top predators such as seals, eiders and walruses (Table 2). Table 2 also shows resulting values after 0.5 years and 5 years for all groups. Fig. 3 summarizes the concentration per biomass for the first 5 years after the spill for nearshore bivalves and walruses for all three cases. All the results from the Ecotracer runs are provided in Appendix A.

3.2.1. Ecospace

Since the contaminants came from one point source and were not distributed throughout the whole ecosystem, we ran Ecotracer as a spatio-temporal model (Ecospace). The contaminant concentration gradually decreases with distance from the spill site (Fig. 2).

The spatial overlap between the functional groups of animals and the spill layer means that many animals are not exposed at all to the pollutant. The spatio-temporal model estimated the biomass concentrations to be reached after about 0.5–3 years (Appendix A).

For case 1, the PAHs levels of 3713 μg/kg wet wt in eiders, 1290 μg/kg wet wt in seals and walrus were estimated to be 753 μg/kg wet wt (Appendix A).

In case 2, the concentration levels per biomass were similar to case 1 for the top predator groups (mammals, seabirds and fish). However, the benthos and crustacean groups accumulated slightly higher levels as a result of direct uptake from the water. This was the only model run where concentration in bivalves continued to increase for 2 years (Appendix A).

There was no correlation between maximum levels of PAHs according to Ecotracer and the variables production/biomass, consumption/biomass or the consumption of bivalves (Pairwise correlation test, Pearsons, $p > 0.05$).

4. Discussion

Ecological modelling systems are valuable support tools for managing human influence on the marine ecosystems. Using

Table 2
Results from Ecotracer for all three cases. Concentrations in μg/kg wet weight.

	Case 1		Case 2		Case 3	
Time (years)	0.5	5	0.5	5	0.5	5
Water	1294.86	44.28	0.41	0.08	0.38	0.09
Toothed whales	288.87	73.27	289.00	81.60	0.05	0.01
Baleen whales	8.27	15.97	8.63	25.90	0.01	0.01
Seals	984.65	228.16	985.00	244.00	0.05	0.02
Walruses	553.40	173.86	553.00	215.00	0.69	0.07
Seabirds-pelagic	6.12	4.27	8.98	38.40	0.23	0.02
Ducks and eiders	2685.53	722.63	2690.00	1000.00	1.59	0.36
Cod	24.15	1.23	24.30	3.18	0.05	0.02
Haddock	168.31	9.62	169.00	12.90	0.18	0.05
Long rough dab	0.91	0.51	1.16	5.94	0.17	0.09
Polar cod	0.16	0.10	0.63	1.65	0.00	0.00
Other demersal fish	0.68	0.40	0.90	3.95	0.12	0.07
Pelagic fish	0.13	0.10	0.62	1.93	0.00	0.00
Sandeel (Ammodytidae)	0.28	0.11	0.71	1.87	0.00	0.01
Snow crab	6813.47	38.67	6830.00	54.00	0.13	0.07
Echinoderms	0.78	0.57	6.96	12.90	0.10	0.17
Polychaetes	1.14	0.21	6.25	4.99	0.05	0.05
Nearshore bivalves	7.32	1.18	23.40	43.60	1.60	0.33
Offshore Bivalvia	15.60	1.37	21.00	18.60	2.90	0.24
Other benthos	0.94	0.27	6.42	6.46	0.05	0.07
Shrimp	2.57	1.15	12.20	26.40	0.41	0.14
Krill	1.12	–	8.93	–	0.02	10.14
Zooplankton	2.34	0.10	6.93	2.07	0.03	0.01
Jellyfish	0.24	0.03	0.30	0.25	0.00	0.00
Detritus	9.37	0.23	10.30	2.15	0.10	0.02

ecosystem modelling combined with ecotoxicology is interesting and combining the two methodologies to quantitatively assess expected impacts throughout the food web is a valuable tool for environmental management. Ecological impacts of pollutants, such as PAHs, are only significant if the impair survival, growth, reproduction, cause genetic disruption or seriously affect energy flow through the ecosystem.

The Ecotracer module in Ecopath has not been widely applied before, but our current application indicates good potential for being a tool for combining toxicology and food-web modelling. A key challenge for modelling ecotoxicology is the availability of data on the same species and toxicological compounds on a comparative scale as the ecosystem components. MDO dissolves and disperses rapidly in seawater and from field measurements at Skjervøy one month after of the spill, the water in the immediate surroundings had non-detectable levels of PAHs. However, clean blue mussels set out 2.5 months after the spill accumulated PAHs from the surroundings (Sagerup and Geraudie, 2014). This indicate that blue mussels are extremely efficient in accumulating PAHs from seawater, and supports our use of high accumulation and direct uptake rates as input to Ecotracer. Mussels are suspension-feeding organisms that retain particles on their gills, including oil droplets. The blue mussels accumulate PAHs from both food and water indiscriminately (Baussant et al., 2001), indicating that the accumulation of PAHs in blue mussels are independent of hydrocarbons water solubility.

The bioavailability of PAHs after a spill depends on evaporation, dissolution, dispersion of oil droplets into the water column, water-in-oil emulsification, sinking and sedimentation (Fingas, 2011). MDO is relatively easily dispersed in water. Therefore, a pool of hydrocarbon may quickly be available for accumulation in the organisms after a spill. The solubility depends on the structure and decreases approximately log-linear with molecular weight (Miller et al., 1985). The heavy molecules are bound in the dispersed oil droplets, but as these droplets are filtered by the mussels, accumulation occurs (Baussant et al., 2001).

The assimilation efficiencies for PAHs (concentration in prey/concentration in predator) are poorly known for the Arctic. From our experiment on trophic transfer using red king crab and mussels, we can conclude that there are high assimilation efficiencies as the concentration per biomass in red king crab was much higher than in mussels (Table 1). The red king crab efficiently accumulates PAHs from mussels as well as directly from the water. PAHs have a different molecular structure, stability and bioavailability and the assimilation efficiencies may vary greatly. Our generic value for assimilation efficiencies of 0.7 was chosen to reflect the variation in accumulation of the different PAHs. Baussant et al. (2001) show that the bioconcentration factor in fish, calculated as the ratio between uptake and elimination, varied from 22 to 1495 for different PAHs. Our assimilation rate of 0.7 will not be the same for all groups in the model. As no experimentally verified assimilation rates exist for individual PAHs we applied a value of 0.7 for all groups, except for case three were 0.2 was used for snow crab.

Using 3 cases, or model runs with different input combinations, let us explore how the model responds to different input variables. In case 1, using 0 for the uptake rate from water for biological groups, the resulting values are reached only through consumption and the links of the food-web. In cases 1 and 2, the model predicted very high assimilation efficiency in top predators. Investigating the diet proportions of each functional group will help track the PAHs. We used 0.114 as diet proportion of snow crab in seals and this probably explains why seals were predicted to accumulate high levels of PAHs. For walruses we applied diet proportions of 0.5 nearshore and 0.35 offshore bivalves and 0.05 snow crab, while ducks and eiders have a diet proportion of 0.5 nearshore and 0.06 offshore bivalves. Toothed whales have a 0.1 diet proportion of snow crab, indicating the reasons why all these groups accumulate high maximum levels (Table 2 and Appendix A). Therefore, contribution of PAHs in fish groups does not contribute to PAHs in the top predators.

The two processes of advection and spreading determine the movement behaviour of an oil spill. MDO is a low viscous oil forming a thin film on the surface of the water (Fingas, 2011). As the dissolution and dispersion of MDO the concentration gradually decreases with increasing distance from the spill site. Therefore the spatial model (Ecospace) of the initial concentration in the environment is needed and was integrated in our cases. Further, PAHs are relatively

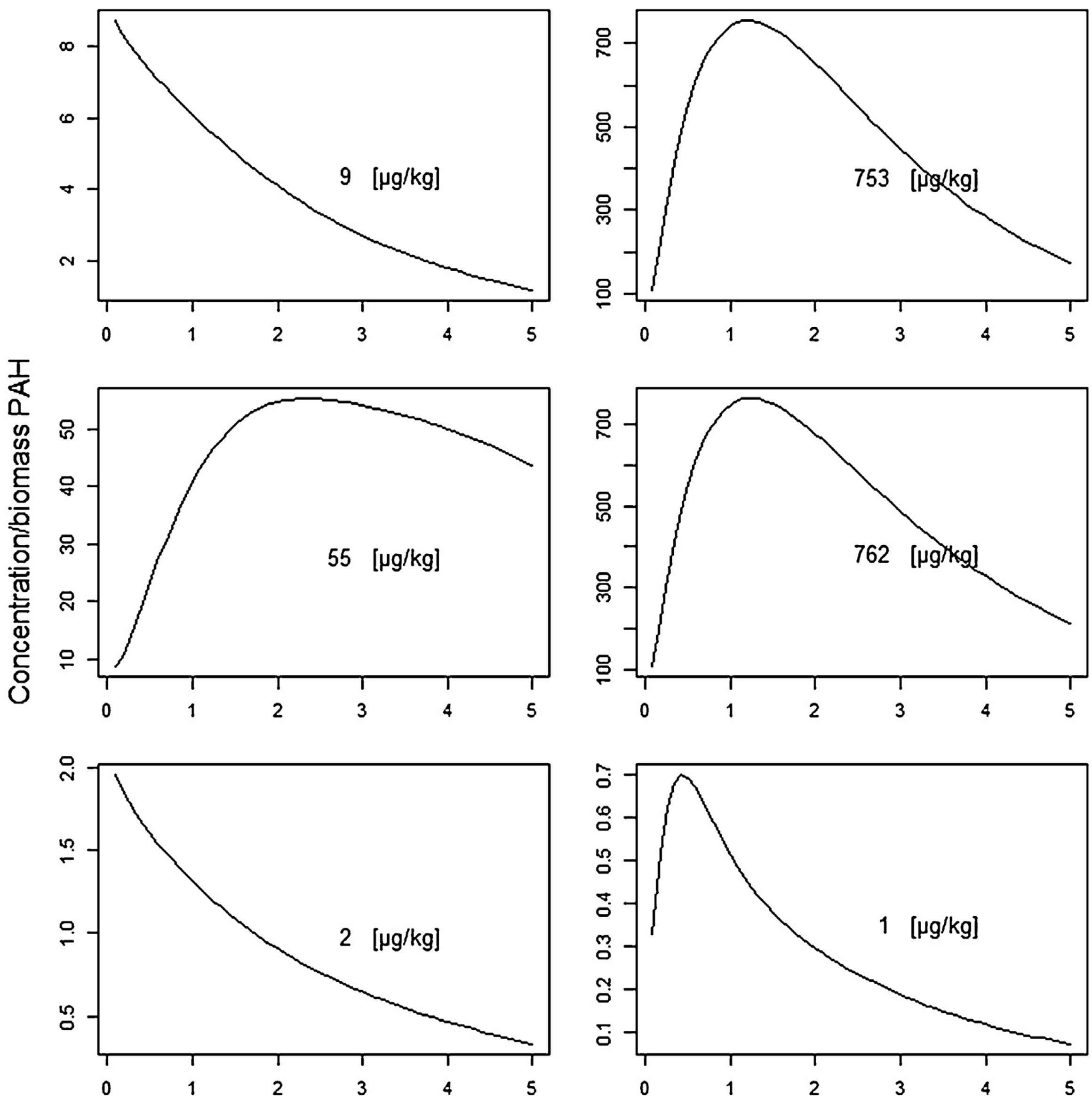

Fig. 3. Ecotracer output for the groups "nearshore bivalves" (left) and "walruses" (right) for case 1 (upper), case 2 (middle) and case 3 (lower). *y*-Axis shows PAHs concentration per biomass in μg/kg wet weight (note very different scales) and *x*-axis shows years after spill. Maximum value given in text on each figure.

biodegradable and the model may be overestimating the degree of bioaccumulation, as PAHs do not biomagnify (Neff, 2002). PAHs also decompose by photochemical oxidation (Dutta and Harayama, 2000; King et al., 2014) and microbial degradation in seawater (Harayama et al., 2004). These processes are of high importance for the degradation of oil spilled at sea, but for simplicity of the model, the pollutants are only handled as tracer molecules with a set decay rate.

When running Ecotracer, one needs a balanced time series model as changes in biomass influence the concentration of pollutants per biomass. For the group sandeel (Ammodytidae) we see the result of the group dying out, thereby producing a high concentration as an artefact of low biomass (see figure of cases 1 and 2 in Appendix A). We can observe similar peaks in PAHs concentrations per biomass for krill. However krill is not predicted to die out, so this may be a result of fluctuating population sizes.

An accidental, momentary oil spill usually spreads from the discharge site, and has a spatial and temporal component different from that of a continuous pollution situation, e.g., mercury levels in the oceans (Booth and Zeller, 2005). To model the spatial component with Ecospace, is unique as it gives the possibility to model a range of concentrations from one spill site. After the Prestige oil spill in Galicia, Spain, November 2002, PAHs were measured in blood of yellow-legged gulls (*Larus michaelis*). The value of total PAHs in blood in gulls from the oil exposed colony at Lobeiras, was a maximum of 228 μg/kg wet wt 17 months after the spill, while gulls from unexposed colonies had a total PAHs concentrations in the blood of about 100 μg/kg wet wt (Pérez et al., 2008). In May 2009 a cargo vessel, MV "*Petrozavodsk*" ran aground at Bjørnøya (74 °N 19 °E) in the Barents Sea, and leaked MDO. In June 2009, one month after the grounding, PAHs levels in blood from glaucous gull (*Larus hyperboreus*) reached 214 μg/kg, but the average of 28 birds sampled was 42.7 μg/kg (Strøm et al., 2011). In 2010, only 3 of 14 gulls from Bjørnøya had detectable levels of PAHs in their blood. This agrees with the spatial model prediction where maximum value for pelagic seabirds was 24 μg/kg wet wt after about 20 months.

Within ecotoxicology, measured and calculated concentrations of contaminants are usually very low (μg/kg or even ng/kg). But the input into the Ecotracer is on tonnes/tonnes level and visual representations of the concentrations per biomass are therefore difficult to interpret. Research on the link between toxicology and ecology applying modelling on an ecosystem level is close to non-existent. Carrer et al. (2000) state that "most toxic substance models focus on the dynamics of the chemical, and therefore simplify the problem of assessing the rate of consumption of contaminated food using empirical equations based on the dimensions of organisms." Predicting the actual outcome of an oil spill is virtually impossible, as there will be unforeseen consequences and interactions. In general, mussels, such as blue mussel, are sensitive to increased levels of hydrocarbons. However, mussels have the possibility to close their shells for long periods and even months of zero food intake. This ability may prove advantageous to the mussels in a spill situation. O'Clair and Rice (1985) found that mussels were less sensitive to hydrocarbons in the water than their predator, the starfish *Evasterias troschenii*, and they suggest that chronic exposure from an oil spill would increase mortality in the starfish and thus give the mussels the possibility to expand due to reduced predation pressure.

The aim of this paper was to test whether Ecotracer can be used to simulate contaminant spread in an Arctic food web. Using one assimilation efficiency for all groups made it easier to compare the spread instead of adding uncertainty by using different assimilation efficiencies. It may be claimed that Ecotracer oversimplifies the kinetics of spread of pollutants by only taking a few variables into consideration. However, we argue that simplification is necessary as the problem is infinitely complex just like an ecosystem.

Applying the Ecotracer module in a sea area with limited background data has been a challenging exercise. The Pechora Sea holds limited fisheries resources, and thus data on human removal of biomass from the model area are poor. The Ecopath with Ecosim modelling system has broadly been developed and applied in areas where time series of recordings of landings are available as input data. However the lack of such data made it impossible to satisfactorily apply the Ecosim module.

The Pechora Sea has strong seasonality, and by only considering the limited data, mostly collected during summer, and using it to represent a full year only added uncertainty to the model predictions. Internally in the model area, migrations occur. Also immigration to and emigration from the model area take place as part of the life cycle of several of the functional groups (e.g., whales and birds). Immigration and emigration were not addressed in our study.

5. Conclusions

Ecotracer is a valuable tool to combine food-web modelling and ecotoxicology. Bridging these two branches of biology is of importance to lift the focus of environmental pollution to an ecosystem level. The modelled concentrations seemed unrealistically high in some functional groups, especially top predators. Providing data that can be used as input, from the same species or functional groups, prey types and pollutants was a major challenge. As well, there are elements of physiological character and kinetics that are not taken into consideration in Ecotracer. However, to be able to model the spread of pollutants at ecosystem level, using many functional groups, simplification is also very important and Ecotracer proves to be a comprehensive modelling tool.

Ecopath with Ecosim and Ecotracer is a comprehensive model to study ecosystem complexity. Attempts to link models for the spread of pollutants in the food web are essential to identify gaps of knowledge. However, in the current application of the model package, we did not manage to get hold of sufficiently reliable time series except for no more than a few of the 27 functional groups, and we were thus unable to make use of the time integrating properties of the Ecosim module. The scarcity of data over time for combined ecotoxicology – ecology modelling in Arctic seas thus becomes evident here. Despite these shortcomings and potential sources of error, our exercise has shown that a food-web influenced by a single accidental event can be modelled, and spread of contaminants addressed in a satisfactory way applying the Ecotracer module.

We suggest further work to include data on tropic transfer to top predators and spread of PAHs in the ecosystem. Also, comparisons of modelling results from experiments with heavy fuel oils (HFO) are encouraged.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolmodel.2015.10.011.

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