Hazard/Risk Assessment

Bioaccumulation of Polycyclic Aromatic Hydrocarbons by Arctic and Temperate Benthic Species

Ariadna S. Szczybelski,a,b,* Noël J. Diepens,a Martine J. van den Heuvel-Greve,c Nico W. van den Brink,d and Albert A. Koelmansa,c

aAquatic Ecology and Water Quality Management Group, Department of Environmental Sciences, Wageningen University, Wageningen, The Netherlands
bDepartment of Animal Ecology, Wageningen Environmental Research (Alterra), Wageningen, The Netherlands
cWageningen Marine Research, Yerseke, The Netherlands
dSubdepartment of Toxicology, Department of Agrotechnology and Food Sciences, Wageningen University, Wageningen, The Netherlands

Abstract: Increasing oil and gas activities may substantially increase chemical stress to benthic ecosystems in the Arctic, and it is necessary to evaluate such environmental risks in these systems. Risk assessment procedures for oil-related compounds (e.g., polycyclic aromatic hydrocarbons [PAHs]) should address differences in exposure between Arctic and temperate benthos. We compare for the first time the bioaccumulation of PAHs by Arctic benthic invertebrate species with that of temperate species, based on their biota–sediment accumulation factors (BSAFs). Measured PAH BSAFs were generally higher in temperate bivalves (Limecola balthica) than in Arctic bivalves (Macoma calcarea), whereas BSAFs in Arctic polychaetes (Nephtys ciliata) were higher than in temperate polychaetes (Alitta virens). Differences in measured BSAFs were explained by species-specific feeding modes and traits. However, modeled BSAFs revealed that steady state was not likely to be reached in the 28-d tests for all PAHs and organisms. Due to the low numbers of individuals, most species-specific parameters were too uncertain to reveal differences between Arctic and temperate species. The results of the present study suggest that data from temperate species could be used as a surrogate for Arctic species in risk assessment. Environ Toxicol Chem 2019;38:883–895. © 2019 The Authors. Environmental Toxicology and Chemistry published by Wiley Periodicals, Inc. on behalf of SETAC.

Keywords: Arctic; Benthic macroinvertebrates; Bioaccumulation; Biota-sediment accumulation factors; Polycyclic aromatic hydrocarbons; Modeling

INTRODUCTION

The Arctic has a high sensitivity to oil spill impacts and has limited capacity for natural recovery due to a profound seasonality, mainly dictated by the reduction in sea ice. Sensitivity may be increased by expanding shipping and oil and gas activities, as well as a lack of appropriate oil spill response methods for this area (Protection of the Arctic Marine Environment, Arctic Council 2009). Therefore, biological targets (bioindicators) for priority monitoring during all phases of oil and gas activities should be used, to assess, minimize, and mitigate adverse effects (Protection of the Arctic Marine Environment, Arctic Council 2009; Szczybelski et al. 2016).

Petrogenic polycyclic aromatic hydrocarbons (PAHs) constitute a large group of hydrophobic contaminants that have been the focus of previous environmental assessments because of their potential toxicity and bioaccumulation (OSPAR Commission 2002; Bakke et al. 2013). Such chemical properties may be used to select appropriate bioindicators of acute and chronic effects of oil and gas production.

Bioindicators provide knowledge about chemical concentrations in the abiotic environment, and they are also considered particularly useful to monitor trends in oil and gas-related pollution, because they integrate chemical and nonchemical stress, and account for in situ ecological conditions (Martínez-Gómez et al. 2010; Montagna et al. 2013). Arctic benthic systems are characterized by a high trophic diversity, relatively long lifespan, and sedentary lifestyles of species (Renaud et al. 2011), which makes them applicable for monitoring purposes, particularly in areas of oil and gas production. A particularly useful exposure metric to define the bioindicator potential of bioaccumulative chemicals in benthic invertebrates is the biota–sediment accumulation
factor (BSAF). The BSAFs of oil and gas-related PAHs can adequately mirror the pollution state of the local Arctic ecosystem near oil and gas activities (DeBlois et al. 2014; Szczybelski et al. 2016). Compared with bioaccumulation (i.e., net result of influx [uptake] and efflux of contaminants) by temperate species, little is known about the bioaccumulation of PAHs from sediment by Arctic benthic species. Cross-chemical extrapolation techniques such as read-across may help to predict PAH bioaccumulation by Arctic species in the absence of experimental data (European Centre for Ecotoxicology and Toxicology of Chemicals 2010). If bioaccumulation was found to be comparable between Arctic and temperate species with similar traits, risk assessment for the Arctic might be simplified by using temperate species data as a surrogate for Arctic species. Therefore, it is useful to explore the potential for reading across 1) Arctic and temperate species and 2) species with different feeding traits and from the same region, with respect to bioaccumulation. Similarity in exposure routes between Arctic and temperate species may not be self-evident, however. For instance, exposure times to contaminants may be longer in Arctic systems, because Arctic species have a longer life-span or biological reaction times are generally slower in polar than in temperate biota (Chapman and Riddle 2005). Clearance of PAHs and their metabolites may also be slower in Arctic invertebrates, because chemical elimination rates are temperature dependent and are affected by seasonal variability in lipid content (Arctic Monitoring and Assessment Programme 2010). Finally, seasonality in the environmental conditions in the Arctic may have a major impact on the physiology of local species (Kędra et al. 2015).

The BSAF has traditionally been used as a metric to assess the bioaccumulation of contaminants from sediment and has been compared across a wide range of aquatic ecosystems (Burkhard et al. 2005; Weisbrod et al. 2009; Selck et al. 2012). If equilibrium partitioning theory applies, BSAF values can be expected to range between 1 and 2 (Boese et al. 1996). This is based on the assumption that chemicals partition between biota lipids and sediment organic carbon. However, equilibrium partitioning theory does not consider the possibility that feeding as a route of uptake may lead to higher than equilibrium steady-state concentrations. To accommodate the situations in which equilibrium partitioning theory does not apply, a kinetic BSAF model can be used (Diepens et al. 2015; Sidney et al. 2016). Potentially, the BSAF may be a useful metric to compare effects of species traits on bioaccumulation, because it can correct for differences in chemical concentration, sediment organic matter, and organism lipid content between the sites or species that are compared. For benthic invertebrates, whose geographic distribution can span to (sub-)Arctic areas, we are aware of only 2 studies reporting BSAFs (Lake et al. 1990; Szczybelski et al. 2016).

The objective of the present study was to compare bioaccumulation between 4 Arctic and 2 temperate species (with different species traits), using 28-d whole-sediment tests (Diepens et al. 2015; Sidney et al. 2016). An additional objective was to assess whether the results can be generalized by using a time-dependent BSAF model, previously used for describing bioaccumulation in temperate marine and freshwater benthic invertebrates (Diepens et al. 2015; Sidney et al. 2016). For the first time, we explore to what extent BSAF is useful as a metric for bioaccumulation in Arctic benthic invertebrate species so that Arctic and temperate species can be compared with respect to bioaccumulation.

**MATERIALS AND METHODS**

We performed a 28-d sediment bioaccumulation experiment with Arctic species at Kings Bay (Ny-Ålesund, Svalbard, Norway) between September and October 2014 (termed the “Arctic experiment”) and another investigation with temperate species at Wageningen Marine Research (Yerseke, the Netherlands) between July and August 2014 (termed the “temperate experiment”). Sediments in the Arctic and temperate experiments were the same for the same phylum. Emphasis was on simulating environmentally realistic PAH exposure levels, which is achieved by the use of representative species of the macrobenthic community of each climate region, and using field-contaminated sediments. Although ecologically relevant, such a strategy led to a low level of replication during the actual tests due to a low availability of field individuals. This was particularly the case for the Arctic experiment.

**Sediment collection and preparation**

Because some of the selected Arctic bivalves (Astarte borealis and Macoma calcarea) usually reside in sandy sediment, and the selected Arctic polychaete (Nephtys ciliata) is generally found in sandy mud (Jørgensen et al. 2014), 2 different batches of sediment were manually prepared after sediment collection to ensure an optimal habitat for the test species. Each contained different ratios of muddy and coarse sediment, respectively defined as <0.50- and 0.50- to 1-mm grain size. Bivalve sediment contained two-thirds coarse and one-third muddy sediment, whereas polychaete sediment contained two-thirds muddy and one-third coarse sediment, based on volume. Sediment was collected in the Oosterschelde estuary (The Netherlands; 51°36′13.5″N, 3°47′49.3″E) on 24 to 25 April 2014 and 2 May 2014, and was used to create the sediment treatments for both the Arctic and the temperate experiments. This was done to maximize the homogeneity of treatment exposures among experiments. These sediment mixtures contained background PAH levels and were therefore termed the “low” treatments. In the low bivalve and polychaete treatments, the PAH concentrations differed (Supplemental Data, Tables S1 and S2) due to differences in their mud content and other variables creating sediment heterogeneity; the PAH concentrations were 1 order of magnitude higher than in previously collected sediments at the end of the Arctic species sampling transect (Blomstrandhalvøya Island, Kongsfjorden Bay, Svalbard, Norway; Szczybelski et al. 2016).

From each of these 2 sediment mixtures with low chemical concentration, another 2 sediment treatments were prepared with a higher chemical concentration, termed the “medium” and “high” treatments. The medium treatment contained 5% (vol) harbor sediment (Rotterdam, The Netherlands), and...
the high treatment contained 10% (vol) harbor sediment. These percentages were used 1) on a precautionary basis because neither the harbor sediment chemical concentrations nor the effect threshold for each of the species were known, and 2) to be able to link the nonlethal effects of chemical concentration to the endpoints studied (Szczybelski et al., 2019). These preparations thus resulted in 6 sediments: 3 for bivalves (bivalve sediment low [BSL], bivalve sediment medium [BSM], and bivalve sediment high [BSH]) and 3 for polychaetes (polychaete sediment low [PSL], polychaete sediment medium [PSM], and polychaete sediment high [PSH]); one of them (BSM) was finally excluded from further analysis due to low numbers of individuals. All sediment treatments were thoroughly mixed with a turbine mixer for approximately 15 min before storage/transport from The Netherlands to Svalbard, and again mixed before use in the exposure experiments. In terms of guidelines for sediment spiking, a short mixing time is acceptable for estuarine sediments because of their homogeneous composition. In all sediment treatments, chemical concentrations and other characteristics (dry wt and organic matter content) were assessed (Supplemental Data, Tables S1 and S2). Due to logistical restraints, the storage time (at 3–7 °C) for sediment treatments was 9 wk longer in the Arctic than in the temperate experiment. It was hypothesized that such a difference in storage time would not significantly affect the partitioning of PAHs between porewater and sediment particles because sediments had been aged for a long time and thus the PAHs were already fully equilibrated (Poot et al. 2014). Prior to the start of exposure, sediment treatments were allowed to stay in contact with filtered seawater at a 1:6 sediment-to-water volume ratio without aeration for 3 d, and with aeration for the following 4 d. Some experimental units in the Arctic experiment were aerated for 12 to 17 d because biota field sampling took place for a longer period than initially expected (the first 3 wk after aeration began).

**Collection of test organisms**

Based on their feeding habits, sessility, and relative abundance (Word et al. 2014), 4 Arctic benthic species were selected; 3 bivalve species—A. borealis (Schumacher, 1817), M. calcarea (Gmelin, 1791), N. pernula (O.F. Müller, 1771)—and a polychaete species—N. ciliata (O.F. Müller, 1776). These species are primarily abundant in Arctic climate zones, although they are not restricted to this area (Table 1). In addition, 2 temperate benthic species (a bivalve, Limecola balthica [Linnaeus, 1758], and a polychaete, Alitta virens [Sars, 1835], formerly known as Macoma balthica and Nereis virens, respectively) were selected based on their comparable feeding habits, sessility, availability of chronic test protocols, and thus the potential for a comparison with the Arctic species. This was also ensured by using the same sediment type among species with similar feeding traits. We limited the present study to 2 temperate species because we prioritized the comparison of bioaccumulation among Arctic and temperate deposit-feeding bivalves and polychaetes. Because M. calcarea individuals were scarce at the sampling area, a second Arctic deposit feeder (N. pernula) was included. Macoma calcarea may feed both on suspended and sediment organic matter, so an obligate suspension feeder (A. borealis) was also included to estimate the contribution of suspended organic matter to the bioaccumulation of PAHs by M. calcarea.

Permission for sampling of Arctic and temperate species was issued by The Governor of Svalbard and the Province of Zeeland (The Netherlands), respectively. Sampling of Arctic species was performed along a transect from Tønsneset peninsula (79° 02′ N, 11° 57′ E) to Blomstrandhalvaya (78° 59′ 14″ N, 11° 57′ 28″ E; Svalbard, Norway; Supplemental Data, Figure S1). Limecola balthica were collected at low tide at the Oesterdam (Oosterschelde National Park, The Netherlands, 51° 26′ 24″ N, 4° 13′ 16″ E), and A. virens were obtained from a professional bait farm, Topsy baits (Wilhelminadorp, The Netherlands).

**Experimental design**

Arctic and temperate species were acclimatized under test conditions. Arctic species were kept in filtered natural seawater (20 μm) for 2 to 6 d without food; in the temperate experiment, animals were kept in filtered natural seawater (0.20 μm) for 5 d, and were fed once at the start of the acclimatization. Feeding of Arctic species prior to exposure would have notably impacted their feeding behavior and metabolic rate because field sampling took place over 3 wk at the end of the productive season. This means that individuals of the same species were not acclimatized for the same length of time and that their nutritional requirements may have been altered between consecutive weeks. For further details of the acclimatization of test species, see the Supplemental Data.

At the initiation of the Arctic experiment, only subsamples of A. borealis and N. pernula were collected, to assess the initial conditions (termed the “background”). This was not feasible for M. calcarea and N. ciliata due to the low number of individuals. Organisms were allowed to depurate their guts for 24 h in filtered seawater. Samples were stored at −20 °C for later determination of wet weight, lipid fraction (% wet wt), and chemical concentrations (μg/kg wet wt). Sediment samples were also taken at the beginning of the Arctic experiment (Supplemental Data, Table S1) and stored at −20 °C for later determination of dry weight, organic matter content, and chemical concentrations (μg/kg dry wt).

Experimental designs were similar for the Arctic and temperate experiments except for the ambient temperature and photoperiod used. The Arctic experiment was a 28-d test in a temperature-controlled room at 3 °C under a 12:12-h light: dark photoperiod. All planned sediment treatments were run with N. ciliata (PSL, PSM, and PSH; n = 1–3; Table 2). Two treatments (BSL and BSH) were tested with M. calcarea (n = 1–2), and only one treatment (BSH) was tested with A. borealis and N. pernula (n = 2–3; Table 2). The temperate experiment was a 28-d test in a temperature-controlled room at 18 °C under a 16:8-h light: dark photoperiod. All sediment treatments were tested with A. virens (PSL, PSM, and PSH; n = 2–3), and 2 treatments were tested with L. balthica (BSL and BSH; n = 1–3).

Individuals were randomly divided into groups of 16 to 50 individuals/experimental unit (Table 2). Animals were not fed
During exposure, although sediment organic matter content (2–6%), and in some cases food supply during acclimatization (for temperate species), prevented body weight (Diepens et al. 2015) and lipid weight loss (Tables 3 and 4) during the experiment. Experimental units were checked daily for mortality (animals at the surface and immobile after poked), and dead organisms were removed daily.

At the conclusion of the experiments, organisms were allowed to depurate their guts for 24 h in filtered seawater. Animals were weighed, measured for body or shell length, dissected (the bivalves), and pooled by species. Biota samples and samples from each sediment treatment were stored (at –20°C) and analyzed for the same parameters as in archived samples. Biota and sediment samples from both experiments were shipped frozen in dry ice to Wageningen University (Wageningen, The Netherlands) for chemical analysis. Due to either a low level of replication of BSL and PSM treatments (Table 2) or to the heterogeneity among PSL replicates in their organic matter content (Supplemental Data, Tables S1 and S2), statistical analyses were only performed with data measured for BSH and PSH treatments. However, all treatments could be used for bioaccumulation modeling.

### Chemical analysis

Chemical analysis was conducted according to methods used by Kupryianchyk et al. (2011). The following PAHs were analyzed: phenanthrene, anthracene, fluoranthene, pyrene, ben[a]
TABLE 2: Overview of the number of experimental units (no.) for Arctic bivalves\(^a\), Arctic polychaetes\(^b\), temperate bivalves\(^c\), and temperate polychaetes\(^d\) per sediment treatment

| Experiment | Species                   | Ind./EU\(^f\) | Background | BSL | BSH | PSL | PSM | PSH |
|------------|---------------------------|---------------|------------|-----|-----|-----|-----|-----|
| Arctic     | Astarte borealis          | 25            | 3          | n.t.| n.t.|     |     |     |
|            | Macoma calcarea           | 40–50         | n.t.       |     | 1   | 2   |     |     |
|            | Nuculana pernula          | 40–50         | 1          | n.t.|     |     |     |     |
|            | Nephtys ciliata           | 20            | n.t.       |     |     |     |     |     |
| Temperate  | Limicola balthica         | 50            | 3          | 1   | 3   | 2   | 2   | 3   |
|            | Alitta virens             | 16            | 3          | 2   |     |     |     |     |

\(^a\)A. borealis, M. calcarea, N. pernula
\(^b\)N. ciliata
\(^c\)L. balthica
\(^d\)A. virens

Sediment treatments with different levels of polycyclic aromatic hydrocarbon (PAH) contamination are referred to as follows: BSL = low bivalve sediment; BSH = high bivalve sediment; PSL = low polychaete sediment; PSM = medium polychaete sediment; PSH = high polychaete sediment. Background refers to pooled field samples in which individuals were allowed to depurate for 24 h in filtered seawater before storage.

\(^e\)Experimental unit (EU) refers to a sediment treatment replicate or aquarium in which a specific number of individuals of a single species are exposed for 28 d to the corresponding sediment treatment, allowed to depurate for 24 h in filtered seawater, and pooled for analytical treatment.

\(^f\)n.t. = not tested.

TABLE 3: Polycyclic aromatic hydrocarbons (mean ± SD; µg/kg lipid wt) in bivalves (Astarte borealis, Macoma calcarea, Nuculana pernula) and polychaetes (Nephtys ciliata) from the Arctic experiment

|        | M. calcarea | N. ciliata | A. borealis | N. pernula |
|--------|-------------|------------|-------------|------------|
|        | BSL         | BSH        | PSL         | BSH        | Background | BSH |
| No.    | 1           | 2          | 1           | 1          | 3          | 3   |
| Lipid wt (% wet wt) | 0.82   | 0.84       | 0.39        | 0.41       | 0.45       | 0.42 |
| PHE    | 3566 ± 393  | 2821 ± 287 | 1402 ± 681  | 729        | 2462 ± 941 | 9697 ± 1482 |
| ANT    | 890 ± 89    | 718 ± 48   | 336 ± 143   | 149        | 297 ± 120  | 2384 ± 442  |
| FLT    | 1612 ± 178  | 3329 ± 1174| 2353 ± 788  | 1771       | 4468 ± 969 | 1376 ± 280  |
| PYR    | 1314 ± 87   | 2623 ± 930 | 1753 ± 530  | 1384       | 3758 ± 920 | 1119 ± 120  |
| BaA    | n.d.        | 673 ± 240  | 113 ± 70    | 77         | 218 ± 104  | n.d.        |
| CHR    | 7434 ± 6614 | 470 ± 113  | 1327 ± 1709 | 0.15       | 6425 ± 664 | 6164 ± 1276 |
| BeP    | 316 ± 0     | 957 ± 374  | 427 ± 178   | 298        | 650 ± 129  | 884 ± 169   |
| BbF    | 175 ± 30    | 658 ± 305  | 117 ± 78    | 100        | 228 ± 69   | 223 ± 56    |
| BghP   | 1162 ± 745  | 50 ± 65    | 344 ± 454   | n.d.       | 1106 ± 955 | 1409 ± 2154 |
| dBhA   | 275 ± 2     | 506 ± 133  | 166 ± 76    | 40         | n.d.       | 479 ± 233   |
| Ind123P| 69 ± 40     | 298 ± 102  | 34 ± 37     | n.d.       | 205 ± 144  | 421 ± 187   |
| Σ\(_1\)PAH | 16 933 ± 6714 | 13 104 ± 3552 | 8333 ± 267 | 4550       | 20 242 ± 9762 | 34 245 ± 20 336 |

SD = standard deviation; BS = bivalve sediment; PS = polychaetes sediment; L = low treatment; M = medium treatment; H = high treatment; n.d. = at least one sample shows a concentration below the limit of detection; PHE = phenanthrene; ANT = anthracene; FLT = fluoranthene; PYR = pyrene; BaA = benz[a]anthracene; CHR = chrysene; BeP = benz[e]pyrene; BbF = benz[b]fluoranthene; BghP = benz[ghi]perylene; dBhA = dibenz[a,h]anthracene; Ind123P = indeno[1,2,3-cd]pyrene; PAH = polycyclic aromatic hydrocarbon.

Data analysis

Lipid-normalized concentrations after 28 d were calculated in biota samples for all available sediment treatments. The BSAFs were calculated as follows:

\[
BSAF = \frac{C_{org}}{C_{sed}/F_{OC}}
\]

with \(C_{org}\) being the chemical concentration in the organism (µg/kg wet wt), \(C_{sed}\) the chemical concentration in sediment (µg/kg dry wt), \(f_{lip}\) the fraction of lipids in the organism based on wet weight, and \(F_{OC}\) the fraction of sediment organic carbon based on loss of ignition and an organic carbon/organic matter conversion ratio of 0.40 (Diepens et al. 2015; Sidney et al. 2016).

Lipid-normalized biota concentrations and BSAFs of BSH-exposed bivalves and PSH-exposed polychaetes were checked for normality with Q–Q plots and Shapiro–Wilk tests, and equality of variances with Levene’s test. If data were normally distributed, lipid-normalized biota concentrations and BSAFs were tested for species and climate region effect with a one-way analysis of variance or an independent \(t\) test, respectively, for each PAH compound. If data were non-normally distributed, they were log-transformed, and when a normal distribution still was not reached, lipid-normalized biota concentrations and BSAFs were tested for species and climate
were performed using SPSS Ver 22. Corrections were applied, respectively. All statistical calculations at \( p \) 0.05; for pairwise comparisons among species or chemicals, Bonferroni’s and Holm’s sequential Bonferroni corrections were applied, respectively. All statistical calculations were performed using SPSS Ver 22.

**Bioaccumulation modeling**

The usefulness of modeling bioaccumulation in invertebrate lipids was explored according to methods described by Diepens et al. (2015), following previously published models (e.g., Thomann and Komlos 1999). In the absence of background data for all test species, we used background data for *N. pernula* to model *M. calcarea* experimental data, and background data for *A. borealis* to model *N. ciliata* experimental data. Uncertainties in the correctness of such assumption will only have a relatively small effect on the first term of Supplemental Data, Equation S1. For an explanation of modeling, calculation of 90% confidence intervals (CIs), percentage of uptake through water, and fraction of steady state reached in the bioaccumulation test, the reader is referred to the Supplemental Data.

**RESULTS AND DISCUSSION**

**PAH background concentrations**

In the Arctic filter-feeding bivalve *A. borealis*, average \( \Sigma_{11} \) PAH concentrations when the animals were collected were 9.44 mg/kg lipid weight. Changes of one order of magnitude in *A. borealis* PAH concentrations were observed for phenanthrene, anthracene, chrysene, and BghiP between September 2014, when samples were collected, and July 2013 (Szczybelski et al. 2016). From these compounds, concentrations of phenanthrene and chrysene were highest (46% of the \( \Sigma_{11} \) PAH) in *A. borealis*. A significant increase in *A. borealis* chrysene concentration would comply with generally high 4-ring PAH concentration in sediment surface layers from the inner areas of Kongsfjorden Bay (Pouch et al. 2017). Such an increase was higher for the deposit-feeding bivalve *N. pernula* than for *A. borealis* (Table 3) and may be explained by *N. pernula*’s higher capacity to select particles with higher organic matter content in sediments with a low organic content (Wodarska-Kowalczuk and Pearson 2004) such as Bloemstrandhalsvåya Island sediments (Szczybelski et al. 2016).

In the temperate bivalve *L. balthica*, average \( \Sigma_{11} \) PAH concentrations in individuals collected at Oesterdam were 35.51 mg/kg lipid weight (Table 4). Concentrations of pyrene and BaA were below concentrations found in *L. balthica* collected in the Westerschelde estuary (The Netherlands; Steur et al. 1996). In the temperate polychaete *A. virens*, \( \Sigma_{11} \) PAH concentrations were lower than in *L. balthica* (10.36 mg/kg lipid wt; Table 4). This was expected, because *A. virens* were obtained from an aquaculture farm, and *L. balthica* were collected in the field.

**Effects of Arctic species traits on PAH bioaccumulation**

**Bioaccumulation.** Lipid-normalized \( \Sigma_{11} \) PAH concentrations in Arctic invertebrates in the high treatment were 13.10 mg/kg in *M. calcarea*, 20.24 mg/kg in *N. ciliata*, 37.67 mg/kg in *A. borealis*, and 14.48E1 mg/kg in *N. pernula* (Table 3). Statistically higher concentrations were observed for anthracene in the filter feeder *A. borealis* compared with the other Arctic species (\( p < 0.05 \); Supplemental Data, Table S9). In addition, the 3- to 4-ring PAHs (phenanthrene, fluoranthene, pyrene, and chrysene) were mainly accumulated by *A. borealis*, which also
agreed with a higher fraction of these compounds in high sediment. The 5- to 6-ring PAHs BeP, BbF, and Ind123P were strongly accumulated (i.e., up to 1 order of magnitude higher than background concentrations) by the deposit feeder N. pernula, which differed from the composition of these compounds in high sediment. This disagreement might be explained by a higher PAH accumulation by N. pernula than the other 2 Arctic bivalves prior to the start of the experiment (Weems et al. 2012), slow PAH excretion (Neff et al. 1987), and nonequilibrium between N. pernula lipid tissue and sediment during the Arctic experiment.

The 3- to 4-ring PAH (phenanthrene, anthracene, fluoranthene, and pyrene) concentrations were on average 2.5 to 4 times higher in the high-exposed filter feeder A. borealis compared with the deposit-feeding bivalves (M. calcarea and N. pernula). An increase in fluoranthene and pyrene concentrations can be explained by the fact that the concentrations of these PAHs in high sediment were also high (Supplemental Data, Table S1). Concentrations of chrysene were on average 42 times higher in high-exposed N. pernula than A. borealis (Table 3), although the chrysene concentration in high sediment was very low (Supplemental Data, Table S1). In this case, considerably high chrysene concentrations in N. pernula might be explained by the species’s ability to ingest sediment to a larger extent than the other Arctic bivalves (Weems et al. 2012) and to retain PAHs, as observed by Neff et al. (1987). In the polychaete N. ciliata, differences in the concentration of phenanthrene, fluoranthene, pyrene, BaA, BbF, and BghiP between the low- and high-exposed individuals reflected the difference in concentrations measured in the low and high sediments (Table 3 and Supplemental Data, Table S1).

No statistical differences in PAH concentrations were found between the deposit-feeding bivalves M. calcarea and N. pernula in the high exposures. However, this conclusion should be viewed with caution because the number of samples available per species was low, resulting in a low statistical power ($p \leq 0.20$) for chrysene, BghiP, and dBahA, and the 2 species have different feeding behaviors (Table 1; Sun et al. 2009; Weems et al. 2012; Keđra et al. 2012). Variability in the metabolism of PAHs by some of our target species (A. virens, L. balthica) is well described in the literature (Rust et al. 2004a), with generally higher metabolism rates in temperate polychaetes than in bivalves. Although not evaluated in the present study, an increase in general metabolic activity (Jensen et al. 2012; Carrasco-Navarro et al. 2015) and a moderate production of reactive oxygen in Arctic species at the end of summer (Tschischka et al. 2000; Nahrgang et al. 2013) are suggested as a means to increase efficiency and to prevent the inhibition of PAH biotransformation, respectively. The metabolism of PAHs in Arctic species is expected to be lower than in temperate species due to low temperature and generally low food availability in their environment (Olsen et al. 2007; Chapman 2016).

**BSAFs.** The BSAFs of PAHs were generally low (i.e., BSAF $< 1$; Figure 1), and were higher in high-exposed A. borealis than high-exposed N. ciliata (Supplemental Data, Tables S7, S8, and S11). This may be linked to a higher black carbon and amorphous organic matter content in the high sediment of polychaetes compared with bivalves. The strong sorption of the planar PAHs to black carbon (Moermond et al. 2005; Cornelissen et al. 2006) most likely decreased PAH bioavailability in polychaetes, and a higher organic matter content may result in a higher nutritional value of the high sediment of polychaetes and a reduction in polychaete feeding. Harbor sediments are known to contain more black carbon (Cornelissen et al. 2005; Koelmans et al. 2010) than sediments from the collection site in the Oosterschelde National Park, The Netherlands. In particular, the Rotterdam harbor area has been subject to considerable black carbon deposition (Keuken et al. 2015). This is supported by a similar pyrogenic PAH component of bivalves’ and polychaetes’ high-sediment treatments, which may be indicative of black carbon presence in both sediment treatments (Tobiszewski and Namieśnik 2012).

**Effects of temperate species traits on PAH bioaccumulation**

**Bioaccumulation.** Lipid-normalized $\Sigma_{11}$PAH concentrations in the high-exposed invertebrates were 32.93 mg/kg in L. balthica and 9.37 mg/kg in A. virens (Table 4). Statistically higher concentrations were observed for BaA and BbF in the high-exposed L. balthica compared with the high-exposed A. virens ($p < 10^{-3}$; Supplemental Data, Table S13). This may be related to factors such as higher PAH background concentrations, higher bioavailability, lower elimination rates, longer exposure times, and higher food selectivity in L. balthica (Rust et al. 2004b; Diepens et al. 2015).

In L. balthica, PAH concentrations were 2-fold (fluoranthene, BbF, dBahA, Ind123P), 3-fold (pyrene, BeP), or 4-fold (BaA) higher in the high- than in the low-exposed individuals, whereas concentrations were 4 to 10 times higher in the high-than in the low-sediment treatment (Table 4 and Supplemental Data, Table S2). The reasons for such a discrepancy may be either a low PAH uptake from sediment (Beukema et al. 2014; Diepens et al. 2015) or differences in congener-specific absorption efficiencies from sediment in L. balthica (McLeod et al. 2004). In contrast, a 2-fold increase in fluoranthene, pyrene, BeP, and dBahA concentrations was observed for both A. virens and sediment samples between the low and the high treatment.

**BSAFs.** Polycyclic aromatic hydrocarbons BSAFs were higher in high-exposed L. balthica than in high-exposed A. virens (Figure 1 and Supplemental Data, Tables S7, S8, and S14), which is likely due to differences in biota PAH background concentrations and bioavailability between sediment treatments. The BSAFs were on average 1.5 to 24 times higher in the high-exposed L. balthica than in the high-exposed A. virens. This may be explained by a combination of different factors such as lower PAH bioavailability in A. virens due to a possibly higher black carbon and other organic material content in the high sediment of polychaetes compared with bivalves (Yates et al. 2011), lower sediment ingestion by A. virens compared with L. balthica (Diepens et al.
PAH bioaccumulation among Arctic and temperate species

Bioaccumulation. Good water quality was attained in both experiments (Supplemental Data, Tables S3 and S4), and mortality was negligible for all species except for the N. ciliata exposed to the high treatment (24%), which may have been related to the reproductive and feeding activity of the species during the experiment. Lipid-normalized PAH concentrations were generally higher in the temperate deposit-feeding bivalve (L. balthica) than in the Arctic bivalve (M. calcarea), after exposure to the same sediment treatment (e.g., high). Differences were generally nonsignificant, which in some cases resulted from low statistical power (i.e., fluoranthene, pyrene, chrysene, and BghiP). Only anthracene concentrations were found to be 2 times higher in M. calcarea compared with L. balthica (Figure 2 and Tables 3 and 4). This may be explained...
by the fact that temperature affects organic matter–water and lipid–water partition coefficients, and thus bioaccumulation and BSAF values if other conditions are the same. Lower temperature has been found to increase PAH affinity to organic matter (Tremblay et al. 2005), and also to decrease lipid partitioning (Koelmans and Jiménez 1994). In the case of M. calcarea, these processes probably played a role, but it was not possible to unambiguously identify the main reason for the apparent difference between the species.

In addition, differences between field-sampled L. balthica and M. calcarea, and differences in food selectivity may have led to differences in their general performance during the temperate and Arctic experiments. For instance, in the case of M. calcarea, strong temperature gradients (0–4 °C) described at sampling depth (20 m) within a 2-mo period in late summer (Hanelt et al. 2001; Svendsen et al. 2002) could have increased energy allocation to physiological maintenance in the Arctic bivalves, decreasing their energy budget (Abele et al. 1998; Pörtner et al. 2007), whereas the absence of phytoplankton input in the experiment could have caused a rapid onset of lowered metabolic rate in M. calcarea adults (Prevodnik et al. 2007; Kędra et al. 2012).

The lack of significant differences between the Arctic (N. ciliata) and temperate (A. virens) polychaetes may also be partly due to an insufficient statistical power (i.e., phenanthrene, chrysene, and BghiP). Anthracene, pyrene, BaA, and BbF concentrations in N. ciliata were on average 2 to 3 times higher than in A. virens (Figure 2 and Tables 3 and 4). This may be related to the fact that food conversion can be more effective in N. ciliata under low temperature compared with A. virens (Neuhoff 1979), thus lowering energy loss and improving the Arctic polychaete’s fitness. Reproductive and morphological differences between the species can also affect their feeding activity rates and PAH bioaccumulation. A <12-h photophase is known to cause sexual maturity in polychaetes and to reduce their feeding rate (Olive 1978; Lawrence and Soame 2009). It is possible that the feeding rate for the Arctic species N. ciliata was reduced during the period of sampling (8–26 September), when the natural photophase rapidly reaches <12 h (Olive 1978; Graf et al. 1982). In the case of the temperate species A. virens, both farm growing and experimental conditions (photophase >12 h, 18 °C) inhibited gametogenic development in mature females (Peter et al. 1998).

Under this last assumption, stable feeding activity would have made a PAH assimilation peak possible, which in this case was assumed to have been reached at an earlier exposure time in the temperate polychaete A. virens than in the Arctic polychaete N. ciliata. This can be due to continuous feeding by A. virens, which may not only increase the contact time of the species’ intestinal epithelium and coelomic fluids with PAHs (Christensen et al. 2002), and thus increase PAH solubilization, but may also lead to oxidative stress and ultimately to the induction of biotransformation enzymes (Jørgensen et al. 2008; Catalano et al. 2012). Nephthidae species such as N. ciliata usually have a much smaller gut volume than other deposit-feeding polychaetes and take discrete meals as part of their carnivore diet (Jumars et al. 2015). This may to some extent restrict the species’ PAH absorption (Ahrens et al. 2001), although stable pre-}

**BSAFs.** For 2- to 3-ring PAHs, the BSAFs were higher in the high-exposed M. calcarea than in L. balthica, which, similarly to the differences between PAH lipid-normalized concentrations, could be explained by higher waterborne PAH uptake in M. calcarea. The BSAFs for PAHs were generally higher in the high-exposed N. ciliata than in A. virens. Ranges of BSAFs for all PAHs were <1E-4 to 18, in the order M. calcarea ≈ A. virens < A. borealis < N. ciliata < L. balthica < N. pernula. However, only species exposed to the same sediment treatment (BSH or PSH) under different climatic conditions (Arctic vs temperate) and with the same feeding modes can be directly compared (i.e., M. calcarea vs L. balthica and N. ciliata vs A. virens). The phenanthrene, anthracene, and fluoranthene BSAFs were 2 to 3 times higher in the Arctic bivalve compared with the temperate bivalve (Supplemental Data, Table S7), whereas PAH BSAFs were generally 2 to 17 times higher in the Arctic polychaete compared with the temperate polychaete (Supplemental Data, Tables S8 and S16), which indicates a generally higher PAH uptake from water in the Arctic compared with the temperate species. Similarly to PAH lipid-normalized concentrations, an absence of significant differences in PAH BSAFs between species with the same feeding mode was partly due to low statistical power (Supplemental Data, Table S16).

**Modeling PAH bioaccumulation by Arctic and temperate species**

The modeled BSAFs matched well with the measured BSAF data. However, the CIs for modeled BSAFs were generally wide for most parameters and species, reflecting the variability in the biological data (Figure 1 and Supplemental Data, Table S17). Complete data sets, with all treatments, were available for N. ciliata and A. virens, and only data for the high treatment were present for all species. This also defines the cases for which parameters were estimated (Supplemental Data, Table S17).

The intercept b in the relation log $K_{\text{SCD}}^{\text{OC}} = \log K_{\text{OW}} + b$, determining the affinity of chemical partitioning to organic carbon, was optimized to a value of 1.07 (0.81–1.33; 90% CI). This is higher than the well-established value of –0.21 for natural sediment (Karickhoff et al. 1979), and can be explained by the efficient binding of PAHs to black carbon present in the sediment (Ruus et al. 2010; Poot et al. 2014). This is consistent with the aforementioned PAH diagnostic ratios indicating pyrogenic PAH sources, and with the fact that many measured BSAF values are smaller than 1. The sorption affinity ratio γ was fitted and appeared to be indistinguishable from 1, whereas the fractions of ingested sediment (parameter β) had 90% CIs extending beyond parameter constraints and were overlapping.
among species. This finding implies that the present experiments did not identify a difference in sorption to suspended organic matter and sediment organic matter (i.e., $K_{OC}^{SS} = K_{OC}^{SED}$), and thus that it does not matter what type of organic matter is ingested. Accordingly, the fraction of ingested sediment (parameter $\beta$), was set to 1. This reduced the number of parameters fitted ($p = 19$; Supplemental Data, Table S17), which yielded narrower CIs for the remaining parameters. Hence, the present model analysis was less rigorous than those provided by Diepens et al. (2015) and Sidney et al. (2016), who detected significant values for $\beta$ based on larger data sets.

In general, BSAs decreased or remained at a constant value with increasing octanol–water partition coefficient ($\log K_{OW}$; Figure 1 and Supplemental Data, Figure S2), which agrees with earlier findings (Diepens et al. 2015; Sidney et al. 2016). One explanation, as discussed by Diepens et al. (2015), may be that steady state was only reached for phenanthrene, anthracene, fluoranthene, and pyrene in $M$. calcarea and $L$. balthica during the 28-d experiment (Supplemental Data, Table S18).

However, for 2 chemicals (chrysene and BghiP), high BSAs were measured and predicted, which was not in line with the other values and expected trends with $\log K_{OW}$. This can be explained by the high chrysene and BghiP background concentrations in the organisms, at the start of the experiment (Tables 3 and 4), as well as low clearance rates.

Because particle ingestion may be a dominant PAH uptake route in benthic organisms (Selck et al. 2012), ingestion rates ($I$) were fitted and expressed on an organic matter basis. In general, ingestion rates were low or even 0 (Figure 3 and Supplemental Data, Table S17). For Arctic species, a low ingestion rate would be expected, because metabolic rates are reduced and growth rates are constrained in cold-adapted stenotherms (Pörtner et al. 2007). However, comparison of our data for the temperate species $L$. balthica and $A$. virens with previously published data shows that these values are below the lower 90% CI boundary and below the range of 0.13 to 0.62 kgOC × kgLipids$^{-1}$ × d$^{-1}$ reported by Thomann et al. (1992). We speculate that the low apparent ingestion rates may be caused by a high nutritional value of the sediment, resulting in the dominance of dermal uptake.

For $M$. calcarea and $A$. borealis, the values for the proportionality parameter $a$ in $\log k_a = -\log K_{OW} + a$, required to calculate the elimination rate constant ($k_a$), were either (respectively) overlapping or higher than previously published CIs for $L$. balthica (Supplemental Data, Table S17). For $N$. ciliata, the a values were higher than previously published CIs for $A$. virens (Diepens et al. 2015). The magnitudes of $a$ for the temperate species $L$. balthica were higher than previously published CIs for polychlorinated biphenyls (PCBs; McLeod et al. 2007, 2008; Diepens et al. 2015). For $A$. virens, magnitudes of $a$ were lower than previously published CIs for PCBs (Diepens et al. 2015), although it should be noted that PAHs are often considered to be metabolized more easily than PCBs (Rust et al. 2004a; Jørgensen et al. 2008).

The relative importance of chemical uptake pathways depends on the species, the chemical, the treatment, and the estimated value of the ingestion parameter (Supplemental Data, Table S17). In all cases in which the ingestion parameter fitted rates were 0, the contribution of the organic matter ingestion pathway was consequently modeled as 0%. In the high treatment, $N$. pernula and $A$. virens showed 100% uptake from organic matter ingestion, whereas for $A$. borealis, $M$. calcarea, and $N$. ciliata, the model suggested 100% uptake from water (Supplemental Data, Table S19). For $L$. balthica, the chemical uptake from sediment increased with increasing log $K_{OW}$. These patterns and values agree with earlier published data (McLeod et al. 2007, 2008; Diepens et al. 2015). Although lower elimination and ingestion rates for Arctic species compared with their temperate counterparts may be expected, this cannot be seen from the present data and parameters. Because the present bioaccumulation data set was not large enough for model validation purposes, as it was in previous modeling studies (Diepens et al. 2015; Sidney et al. 2016), we ascribe these lower rates to the higher variability and uncertainty in the present bioaccumulation data.

**CONCLUSIONS AND IMPLICATIONS**

The present study showed significant differences in PAH bioaccumulation among temperate species with different feeding traits, and for anthracene among Arctic species. Due to the low availability of field individuals, there was a general lack...
of significant differences partly related to low statistical power (e.g., chrysene and 
BghiP). However, differences between Arctic and temperate species with similar feeding traits 
were significant for some of the PAHs among polychaetes. Bioaccumulation of 
PAHs from sediment was generally higher in the Arctic polychaete 
N. ciliata than in the temperate A. virina. On the other hand, the 
temperate deposit-feeding bivalve L. balthica accumulated 
PAHs to a greater extent from sediment than its Arctic 
counterpart M. calcarea. Consequently, bioaccumulation 
metabolism experimentally determined in temperate species might be 
too conservative in the risk assessment for similar Arctic species 
and in other cases might be too moderate.

The time-dependent BSAF model shows that model parameters 
did not significantly differ among species and climate 
regions, which is supported by the generally nonsignificant differences 
in PAH bioaccumulation between Arctic and 
temperate species. This finding implies that although modeling 
had been shown to be a valid tool in earlier work (Diepens et al. 
2015; Sidney et al. 2016), insufficient data for an appropriate 
parametrization limited the insights provided by modeling in 
the present study. Field validation of bioaccumulation models, for 
which data assumptions are built on temperate species data, 
will, however, require Arctic standard single-species tests for 
further characterization of sediment bioaccumulation mecha-
nisms, such as contaminant uptake and elimination routes, under 
local conditions.

The present study used sediments and conditions mimicking 
Arctic and temperate environmental conditions, and used Arctic 
and temperate species with pairwise matching functional 
groups. Differences in bioaccumulation in a direct comparison 
of the 2 climate regions’ characteristics were observed, which, 
however, were not statistically significant due to considerable 
variability in the biological control data, and similarity in 
exposure routes. This means that a cautionary note should be 
sounded on the use of temperate benthic species as a surrogate 
for Arctic benthic species in bioaccumulation assessment. It 
could also be a reasonable approach to consider the use of 
safety factors (e.g., 10) to extrapolate from temperate to Arctic 
species.

Supplemental Data—The Supplemental Data are available on 
the Wiley Online Library at DOI: 10.1002/etc.4366.

Acknowledgement—The authors thank A. Meldahl (who was 
captain of the research vessel MS Teisten in Kings 
Bay [Ny-Ålesund, Svalbard, Norway] in the summer of 2014), 
D. Loonen, B. Frederiks, L. Sidney, V. Mohaupt, T. Dupeyron, 
K. Binder, A. Roman, and W. van Duin for their assistance with 
laboratory and sampling. M. Wodarska-Kowalczuk, M. Kędra, 
H. Hillewaert, and A. Semenov are acknowledged for granting us 
permission to use the image files in Table 1. Equinor (grant SAP 
4502687550) and the Wageningen UR TripleP@Sea innovation 
program (grant KB-14-007) are acknowledged for funding the 
Development of Arctic Biological Indicators (ARCIND) Project.

Data Accessibility—For access to data, please contact 
the corresponding author (ariadnaszc@gmail.com).

REFERENCES

Abele D, Burlando B, Vairenga A, Pörtner H-O. 1998. Exposure to elevated temperatures and hydrogen peroxide elicits oxidative stress and 
antioxidant response in the Antarctic intertidal limpet Nacella concinna. 
Comp Biochem Physiol B Biochem Mol Biol 120:425–435.

Ahrens MJ, Hertz J, Lamoureux EM, Lopez GR, McElroy AE, Brownwell BJ. 2001. The effect of body size on digestive chemistry and absorption 
efficiencies of food and sediment-bound organic contaminants in Nereis 
succinea (Polychaeta). J Exp Mar Biol Ecol 263:185–209.

Arctic Monitoring and Assessment Programme. 2010. Assessment 2007: Oil 
and gas activities in the Arctic—Effects and potential effects. Tromsø, 
Norway.

Anisimova NA, Jørgensen LL, Lyubin PE, Manushin IE. 2010. Mapping and 
monitoring of benthos in the Barents Sea and Svalbard waters: Results from 
the joint Russian–Norwegian benthic programme 2006–2008. 
Institute of Marine Research, Tromsø, Norway/Polar Research Institute 
of Marine Fisheries and Oceanography, Murmansk, Russia.

Bakke T, Klungsøy J, Sanni S. 2013. Environmental impacts of produced 
water and drilling waste discharges from the Norwegian offshore 
petroleum industry. Mar Environ Res 92:154–169.

Beukema JJ, Cadée GC, Dekker R, Philippart CJM. 2014. Annual and spatial 
variability in gains of body weight in Macoma balthica (L): Relationships 
with food supply and water temperature. J Exp Mar Biol Ecol 457:105–112.

Blanchard AL, Parris CL, Knowlton AL, Wade NR. 2013. Benthic ecology of the 
northeastern Chukchi Sea. Part I. Environmental characteristics and 
macrofaunal community structure, 2008–2010. Continental Shelf Res 
67:52–66.

Boeke BL, Lee Ii H, Specht DT, Randall R, Pelletier J. 1996. Evaluation of PCB 
and hexachlorobenzene biota-sediment accumulation factors based on 
ingested sediment in a deposit-feeding clam. Environ Toxicol Chem 
15:1584–1589.

Burkhard LP, Cook PM, Lukasewycz MT. 2005. Comparison of biota–sediment 
accumulation factors across ecosystems. Environ Sci Technol 39:5716–5721.

Carrasco-Navarro V, Jæger I, Honkanen JO, Kuukkonen JVK, Carroll J, 
Camus L. 2015. Bioconcentration, biotransformation and elimination of 
pyrene in the arctic crustacean Gammarus setosus (Amphipoda) at two 
temperatures. Mar Environ Res 110:101–109.

Catalano B, Molteo G, Martuccio G, Gastaldi L, Virno-Lamberti C, Lauria A, 
Aulisii A. 2012. Can Hediste diversicolor (Nereidae, Polychaeta) be 
considered a good candidate in evaluating PAH contamination? A 
multimarker approach. Chemosphere 86:875–882.

Chapman PM. 2016. Toxicity delayed in cold freshwaters? J Great Lakes Res 
42:286–289.

Chapman PM, Riddle MJ. 2005. Toxic effects of contaminants in polar marine 
environments. Environ Sci Technol 39:200A–206A.

Christensen M, Andersen O, Banta GT. 2002. Metabolism of pyrene by the 
polychaetes Nereis diversicolor and Arenicola marina. Aquat Toxicol 
58:15–25.

Cornelissen G, Gustafsson O, Bucheli TD,Jonker MTO, Koelemans AA, 
Van Noort PCM. 2005. Extensive sorption of organic compounds to black 
carbon, coal, and kerogen in sediments and soils: Mechanisms and 
consequences for distribution, bioaccumulation, and biodegradation. 
Environ Sci Technol 39:6881–6895.

Cornelissen G, Breedveld GD, Nes K, Oen AMP, Ruus A. 2006. Bioaccumu-
lation of native polycyclic aromatic hydrocarbons from sediment by 
a polychaete and a gastropod: Freely dissolved concentrations and activated 
carbon amendment. Environ Toxicol Chem 25:2349–2355.

DeBlois EM, Kiceniuk JW, Paine MD, Kilgour BW, Tracy E, Crowley RD, 
Williams JP, Gregory Janes G. 2014. Examination of body burden and 
taint for Iceland scallop (Chlamys islandica) and American plaice 
(hippoglossoides platessoides) near the Terra Nova offshore oil development 
over ten years of drilling on the Grand Banks of Newfoundland, Canada. 
Deep Sea Res II Top Stud Oceanogr 110:65–83.

Decho AW, Luoma SN. 1991. Time-courses in the retention of food material in 
the bivalves Potamocorbula amurensis and Macoma balitica: Signifi-
cance to the absorption of carbon and chromium. Mar Ecol Prog Ser 
78:303–314.

Denisenko NV, Rachor E, Denisenko SG. 2003. Benthic fauna of the Southern 
Kara Sea. In Stein R, Fahlik K, Futterer DK, Galimov EM, Stepanets OV, eds,
Siberian River Run-off in the Kara Sea. Elsevier Science, Amsterdam, The Netherlands.

Diepens NJ, Van den Heuvel-Greve MJ, Koelmans AA. 2015. Modeling of bioaccumulation in marine benthic invertebrates using a multispecies experimental approach. Environ Sci Technol 49:13575–13585.

European Centre for Ecotoxicology and Toxicology of Chemicals. 2010. High information content technologies in support of read-across in chemical risk assessment. No. 109. Brussels, Belgium.

Gogina M, Nygård H, Blomqvist M, Daunys D, Josefson AB, Kotta J, Maximov G, Marselli L, Delong K, Delong K, Delong K, Delong K, Delong K. 2010. Estimation of in situ sediment-to-water fluxes of polycyclic aromatic hydrocarbons, polychlorobiphenyls and polybrominated diphenylethers. Mar Environ Sci Technol 65:69–75.

Environ Monit Assess 189:175–198.

Neuhoff H-G. 1979. Influence of temperature and salinity on food conversion and growth of different Nepis species (Polychaeta, Nephtyeidae). Mar Ecol Prog Ser 1:255–264.

Peter JWO, Simon WR, Ali D. 1998. Influence of photoperiod and temperature on oocyte growth in the semelparous polychaete Nereis (Neanthes) virens. Mar Ecol Prog Ser 172:169–183.

Poot A, Jonker MTO, Gillissen F, Koelmans AA. 2014. Explaining PAH desorption from sediments using Rock Eval analysis. Environ Pollut 193:247–253.

Peter JWO, Simon WR, Ali D. 1998. Influence of photoperiod and temperature on oocyte growth in the semelparous polychaete Nereis (Neanthes) virens. Mar Ecol Prog Ser 172:169–183.

Poot A, Jonker MTO, Gillissen F, Koelmans AA. 2014. Explaining PAH desorption from sediments using Rock Eval analysis. Environ Pollut 193:247–253.

Portner HO, Peck L, Somero G. 2007. Thermal limits and adaptation in marine Antarctic ectotherms: An integrative view. Philos Trans R Soc B Biol Sci 362:2233–2258.

Pouch A, Zaborska A, Pazdroot K. 2017. Concentrations and origin of polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) in sediments of western Spitsbergen fjords (Kongsfjorden, Hornsund, and Adventfjorden). Environ Monit Assess 189:175.

Previdnik A, Gardeström J, Lilia K, Elfving T, McDonagh B, Petrović N, Tedengren M, Sheehan D, Bollner T. 2007. Oxidative stress in response to xenobiotics in the blue mussel Mytilus edulis L.: Evidence for variation along a natural salinity gradient of the Baltic Sea. Aquat Toxicol 82:63–71.

© 2019 The Authors

wileyonlinelibrary.com/ETC
Bioaccumulation of PAHs by Arctic and temperate benthic species—Environmental Toxicology and Chemistry, 2019;38:883–895

Renaud PE, Tessmann M, Evesen A, Christensen GN. 2011. Benthic food-web structure of an Arctic fjord (Kongsfjorden, Svalbard). Mar Biol Res 7:13–26.

Rust AJ, Burgess RM, Brownawell BJ, McElroy AE. 2004a. Relationship between metabolism and bioaccumulation of benzo[α]pyrene in benthic invertebrates. Environ Toxicol Chem 23:2587–2593.

Rust AJ, Burgess RM, McElroy AE, Cantwell MG, Brownawell BJ. 2004b. Influence of soot carbon on the bioaccumulation of sediment-bound polycyclic aromatic hydrocarbons by marine benthic invertebrates: An interspecies comparison. Environ Toxicol Chem 23:2594–2603.

Ruus A, Bøyum O, Grung M, Næs K. 2010. Bioavailability of PAHs in aluminium smelter affected sediments: Evaluation through assessment of pore water concentrations and in vivo bioaccumulation. Environ Sci Technol 44:9291–9297.

Selck H, Drouillard K, Eisenreich K, Koelmans AA, Palmqvist A, Ruus A, Salvito D, Schultz I, Stewart R, Wesbrod A, van den Brink LW, van den Heuvel-Greve M. 2012. Explaining differences between bioaccumulation measurements in laboratory and field data through use of a probabilistic modeling approach. Integr Environ Assess Manag 8:42–63.

Sidney LA, Diepens NJ, Guo X, Koelmans AA. 2016. Trait-based modelling of bioaccumulation by freshwater benthic invertebrates. Aquat Toxicol 176:88–96.

Steur C, Seys J, Eppinga J. 1996. Ecologisch profiel van het Nonnetje (Macoma balthica). Ministerie van Verkeer en Waterstaat, Rijkswaterstaat, Rijksinstituut voor Kust en Zee, Den Haag, The Netherlands.

Strand J, Jacobsen JA, Pedersen B, Grannmo A. 2003. Butyltin compounds in sediment and molluscs from the shipping strait between Denmark and Sweden. Environ Pollut 124:7–15.

Sun M-Y, Clough LM, Carroll ML, Dai J, Ambrose Jr WG, Lopez GR. 2009. Different responses of two common Arctic macrobenthic species (Macoma balthica and Monoporeia affinis) to phytoplankton and ice algae: Will climate change impacts be species specific? J Exp Mar Biol Ecol 376:110–121.

Swendsen H, Beszczyńska-Möller A, Hagen JO, Lefaauconnier B, Tverberg V, Gerland S, Berre Ørbak J, Bischof K, Papucci C, Zająckowski M, Azzolini R, Bruland O, Wiencke C, Thorsen E, Thorsen W 2003. The physical environment of Kongsfjorden–Krossfjorden, an Arctic fjord system in Svalbard. Polar Res 21:34.

Szczybski AS, van den Heuvel-Greve MJ, Koelmans AA, van den Brink NW, Koelmans AA. 2016. Bioaccumulation of polycyclic aromatic hydrocarbons, polychlorinated biphenyls and hexachlorobenzene by three Arctic benthic species from Kongsfjorden (Svalbard, Norway). Mar Pollut Bull 112:65–74.

Szczybski AS, van den Heuvel-Greve MJ, Koelmans AA, van den Brink NW. 2019. Biomarker responses and biotransformation capacity in Arctic and temperate benthic species exposed to polycyclic aromatic hydrocarbons. Sci Total Environ 662:631–638.

Thomann RV, Komlos J. 1999. Model of biota-sediment accumulation factor for polycyclic aromatic hydrocarbons. Environ Toxicol Chem 18:1060–1068.

Thomann RV, Connolly JP, Parkerton TF. 1992. An equilibrium model of organic chemical accumulation in aquatic food webs with sediment interaction. Environ Toxicol Chem 11:615–629.

Tobiszewski M, Namiesnik J. 2012. PAH diagnostic ratios for the identification of pollution emission sources. Environ Pollut 162:110–119.

Tremblay L, Kohl SD, Rice JA, Gagné J-P. 2005. Effects of temperature, salinity, and dissolved humic substances on the sorption of polycyclic aromatic hydrocarbons to estuarine particles. Mar Chem 96:21–34.

Tschischka K, Abele D, Portner HO. 2000. Mitochondrial oxyconformity and cold adaptation in the polychaete Nereis pelagica and the bivalve Arctica islandica from the Baltic and White Seas. J Exp Biol 203:3355.

Weems J, Iken K, Gradinger R, Woolier MJ. 2012. Carbon and nitrogen assimilation in the Bering Sea clams Nuculana radiata and Macoma moesta. J Exp Mar Biol Ecol 430:431–32–42.

Weisbrod AV, Woodburn KB, Koelmans AA, Parkerton TF, McElroy AE, Borgå K. 2009. Evaluation of bioaccumulation using in vivo laboratory and field studies. Integr Environ Assess Manag 5:598–623.

Wlodarska-Kowalczuk M, Pearson TH. 2004. Soft-bottom macrobenthic faunal associations and factors affecting species distributions in an Arctic glacial fjord (Kongsfjord, Spitsbergen). Polar Biol 27:155–167.

Word JQ, Gardiner W, Melbye A, Merlin F, Brakstad O, Camus L, Galloway B, Hansen BH, Coelho G. 2014. Environmental effects of spilled oil and response technologies in the Arctic: Literature review and recommendations. Arctic Oil Spill Response Technology Joint Industry Programme, International Association of Oil and Gas Producers, London, UK.

Yates K, Pollard P, Davies IM, Webster L, Moffat CF. 2011. Application of silicone rubber passive samplers to investigate the bioaccumulation of PAHs by Nereis vires from marine sediments. Environ Pollut 159:3351–3356.