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Tissue distribution of ultraviolet absorbents and industrial antioxidants in Atlantic walruses (*Odobenus rosmarus rosmarus*) and ringed seals (*Pusa hispida*) from the Canadian Arctic: Influence of sex, body size, and spatial variation

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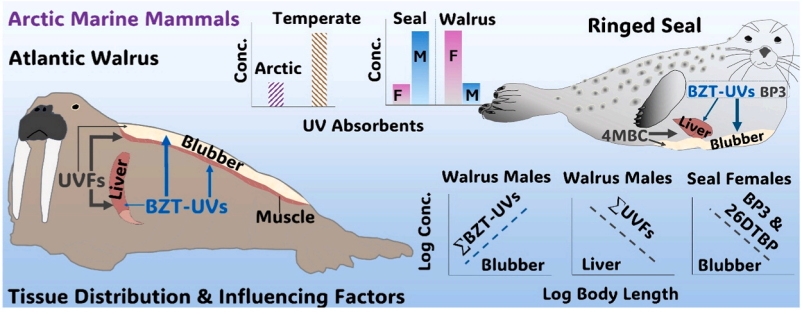
HIGHLIGHTS

- Arctic walruses and seals had lower UV absorbent levels than temperate mammals.
- Blubber had more BZT-UVs, while some UVFs accumulated more in muscle and liver.
- Sub-Arctic ringed seals had higher levels of UV absorbents and 26DTBP than high-Arctic.
- Walrus and seal showed different sex-specific accumulations of UV absorbents.
- BZT-UV levels increased with body length, but UVFs and 26DTBP showed growth dilution.

GRAPHICAL ABSTRACT



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ABSTRACT

UV absorbents and industrial antioxidants are contaminants of emerging concern. However, their tissue distribution in marine mammals remains poorly characterized. This study investigated the tissue distribution of benzotriazole UV stabilizers (BZT-UVs), organic UV filters (UVFs), 2,6-di-*tert*-butylphenol (26DTBP), and aromatic secondary amines (Ar-SAs) in Atlantic walrus (*Odobenus rosmarus rosmarus*) (blubber, muscle, and liver) and ringed seal (*Pusa hispida*) (blubber and liver) tissues collected around several communities in the Canadian Arctic. In both species, blubber accumulated higher levels of BZT-UVs than other tissues, whereas some UVFs, such as benzophenone and 2-hydroxy-4-methoxybenzophenone (BP3), accumulated more in the muscle or liver of walruses. ∑BZT-UVs in male walrus blubber correlated positively with body length, demonstrating a bioaccumulation trend as individuals grew larger. This contrasts with the evidence of growth dilution observed for

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∑UVFs in male walrus liver, and for BP3 and 26DTBP in female ringed seal blubber. Both species exhibited spatial variation in UV absorbent concentrations. This study provides the first data on the tissue distribution of these contaminants in Arctic marine mammals, revealing tissue-, spatial-, sex-, and compound-specific distribution patterns. The findings offer current reference data to support environmental monitoring, risk assessment, and management efforts, particularly for species important to northern Indigenous communities.

1. Introduction

Ultraviolet (UV) absorbents and industrial antioxidants (Figure S1) are two groups of additives used in commercial and industrial products (Table S1) [1–4]. UV absorbents are divided broadly into two groups: benzotriazole UV stabilizers (BZT-UVs) and organic UV filters (UVFs). They are incorporated into various products, such as plastics, paints, coatings, personal care items, cosmetics, and sunscreens (Table S1), to reduce UV-induced material degradation, prolong product durability, and/or protect human skin from sunburn [1,2]. Industrial antioxidants, including synthetic phenolic antioxidants (SPAs) and aromatic secondary amines (Ar-SAs), are employed to inhibit oxidative degradation in products such as plastics, rubbers, fuels, lubricants, adhesives, and sealants (Table S1) [3,4]. These additives are not covalently bound but are physically mixed into materials, allowing them to be released into the environment via diffusive emissions during manufacturing [5], leaching from landfills or wastewater [6–10], and gradual migration during product use [11].

As summarized in Table S1, several of these chemicals are produced and consumed in high volumes, exceeding the thresholds defined by the U.S. Environmental Protection Agency (>450 tonnes/year) and REACH regulations in Europe (>100 tonnes/year) for high-volume chemicals. For example, the production and import of 2-(2H-benzotriazol-2-yl)-4,6-di-*tert*-pentylphenol (UV328) ranged between 450 and < 4500 tonnes in 2019 in the U.S. and from 100 to 1000 tonnes/year in Europe (Table S1). The high production and widespread use (Table S1) of UV absorbents and industrial antioxidants have led to increasing environmental concerns about their persistence, bioaccumulation, and toxicity. These compounds have been frequently detected in environmental matrices and may pose toxicological risks to ecosystems and human health, as documented in prior reviews [1–4]. They have been detected in Arctic biota, but direct measurements in the surrounding abiotic environment are scarce [12,13]. Possible sources include consumer product use and waste in northern communities, as well as long-range transport [13]. To mitigate risks, some international and regional regulations have been introduced. For example, the Stockholm Convention on Persistent Organic Pollutants (POPs) has listed UV328 (Figure S1), a BZT-UV, in Annex A (Elimination) with specific exemptions for use in certain products until 2044 [14]. In addition, the European Chemical Agency has added UV328, 2-benzotriazole-2-yl-4,6-di-*tert*-butylphenol (UV320), 2-*tert*-butyl-6-(5-chloro-2H-benzotriazol-2-yl)-4-methylphenol (UV326), 2, 4-di-*tert*-butyl-6-(5-chloro-2H-benzotriazol-2-yl) phenol (UV327), 2-(2H-benzotriazol-2-yl)-4-(1,1,3,3-tetramethyl butyl)phenol (UV329), and 4-methylbenzylidene camphor (4MBC) (Figure S1), all UV absorbents, to the Candidate List of Substances of Very High Concern (SVHC) for Authorization due to their persistent, bioaccumulative and toxic (PBT), very persistent and very bioaccumulative (vPvB) or endocrine disrupting properties (Table S1) [15], despite ongoing debate and conflicting evidence [16,17]. These chemical management efforts may reduce the future environmental concentrations of these contaminants in remote regions, including the Arctic. However, initial environmental reference values must be established to track the effectiveness of these regulations by comparing them with future monitoring results.

UV absorbents and industrial antioxidants have been detected in tissues of cetaceans (e.g., whales, porpoises, dolphins) and pinnipeds (e.g., seals) from temperate areas [18–22]. In Arctic ecosystems, previous studies have similarly identified these contaminants in the blood and/or liver of apex predators, including polar bear (*Ursus maritimus*), beluga whale (*Delphinapterus leucas*), and American mink (*Neogale vison*) [12,13,23]. However, existing studies often focus on a single tissue (e.g., liver, blood, or blubber) and analyze limited analyte types (e.g., only analyzing a few UV absorbents) with small sample sizes (e.g., a few stranded animals), which hinders a comprehensive understanding of exposure and toxicity. In addition, a recent non-target screening study detected 2,6-di-*tert*-butylphenol (26DTBP), a SPA, in the blubber of East Greenland marine mammals, including polar bears, killer whales (*Orcinus orca*), narwhals (*Monodon monoceros*), and long-finned pilot whales (*Globicephala melas*) [24]. While Pedersen et al. [24] confirmed the presence of 26DTBP, they did not quantify its concentrations, leaving critical data gaps on exposure levels. Overall, these results underscore two key findings: (1) UV absorbents and industrial antioxidants accumulate across different marine mammal taxa, and (2) even remote Arctic wildlife are not isolated from exposure to these contaminants of emerging concern [12,13,23,24]. The presence of these compounds in Arctic biota raises concerns about potential ecological risks in regions with limited local sources of contamination.

In the Arctic, Atlantic walruses (*Odobenus rosmarus rosmarus*) feed on benthic invertebrates [25], making them valuable indicators of the ecological health of the benthic environment. This is particularly important for UV absorbents and industrial antioxidants, as sediments are known to be sinks for these contaminants [26]. The benthic food web is therefore expected to be a primary pathway for the exposure and accumulation of these contaminants in higher trophic level organisms [27,28]. Some walruses hunt or scavenge on pinnipeds, cetaceans, or seabirds, which may increase their trophic level, change exposure pathways, and alter contaminant concentrations in their tissues [29,30]. The trophodynamics of each contaminant also influence this change in contaminant levels, as some biomagnify (e.g., BZT-UVs and SPAs), while others may biodilute (e.g., Ar-SAs) in the food web [31–33]. Ringed seals (*Pusa hispida*) are widely distributed in the Arctic and have been used as sentinel species to reflect ecosystem health [34]. Ringed seals have a diverse diet that varies seasonally and regionally but primarily consists of small fish and invertebrates [35,36]. They commonly feed on Arctic cod (*Boreogadus saida*), capelin (*Mallotus villosus*), sand lance (*Ammodytes spp.*), and various species of shrimp, amphipods, and krill [35,36]. The tissues of both walruses and ringed seals are also part of the traditional diet of some Indigenous and northern communities. Thus, the distribution of emerging contaminants, such as UV absorbents and industrial antioxidants, in walruses and ringed seals provides critical insights into the exposure of key Arctic species to these emerging contaminants and the potential implications for human dietary exposure in the Arctic.

Knowledge of the tissue distribution of contaminants is essential for understanding their toxicity in organisms [37,38]. Such information is also important for human health, given that humans may consume tissues (e.g., liver, muscle, and blubber) and consequently be exposed to various contaminants. Lipophilic compounds are expected to preferentially accumulate in marine mammal blubber if not significantly eliminated [39]. Protein-binding contaminants are more likely to accumulate in the liver [40,41], particularly when introduced through oral ingestion. Muscle analysis is relevant for evaluating human exposure risks, as muscle is a direct pathway of exposure for Indigenous and northern communities of the Arctic, who regularly consume these species [42]. In addition, the accumulation patterns of contaminants in organisms vary spatially due to variation in the sources, distribution, and fate of contaminants in the surrounding environment [43,44]. However, there is a lack of understanding of the tissue and spatial distribution of UV

absorbents and industrial antioxidants in wildlife, particularly for marine mammals in the Arctic. Given these limitations in existing research, there is an urgent need for comprehensive, multi-tissue studies that can provide foundational reference data for regulatory assessment and guide future monitoring efforts.

To address these critical knowledge gaps, the objectives of this study were to analyze the tissue-specific distribution of 16 UV absorbents and seven industrial antioxidants in the blubber, liver, and muscle of Atlantic walruses across five eastern Canadian Arctic sites, as well as in ringed seal blubber and liver from ecologically contrasting sub-Arctic and high Arctic locations. Statistical associations between contaminant concentrations and biological variables (e.g., body length and sex) for walruses and ringed seals were assessed to identify factors affecting the accumulation patterns. We predicted that (1) contaminant concentrations in these Arctic species would be lower than those reported for marine mammals from temperate regions; (2) blubber would accumulate more lipophilic contaminants than liver and muscle; (3) contaminant levels would vary spatially, with samples from lower latitudes (closer to anthropogenic emission sources) showing higher concentrations relative to high Arctic populations; and (4) males would accumulate more contaminants than females due to the absence of maternal transfer as a route of excretion, as well as differences in metabolism and feeding behavior, as observed for POPs.

To our knowledge, this is the first study to examine the tissue-specific distribution of UV absorbents and industrial antioxidants in Arctic marine mammals, focusing on Atlantic walruses and ringed seals. While previous research has addressed single-tissue accumulation in other species, our study presents a comprehensive multi-tissue profile across a broader suite of contaminants. These data provide novel insights into tissue partitioning, exposure pathways, and the potential toxicological relevance of these chemicals in Arctic marine mammals. The findings of this work provide guidelines for future toxicity exposure studies in mammals. It also contributes to understanding contaminant exposure in species of cultural and nutritional importance to Indigenous Arctic communities. Additionally, by comparing multiple tissues, this study supports the selection of appropriate tissues for long-term biomonitoring of these emerging contaminants.

2. Materials and methods

2.1. Study sites and sample collection

Walrus blubber ($n = 40$), muscle ($n = 24$), and liver ($n = 36$) samples were collected from the same individuals in 2020 and 2021 by Inuit hunters in collaboration with Fisheries and Oceans Canada (DFO), with some walrus individuals providing all three tissues and others providing only one or two tissues. The samples were obtained near five communities in Nunavut, Canada: Sanirajak (blubber $n = 27$, muscle $n = 17$, liver $n = 26$), Naujaat (blubber $n = 3$, muscle $n = 2$, liver $n = 3$), Salliq (Coral Harbour) (blubber $n = 2$, muscle $n = 2$, liver $n = 2$), Pangnirtung (blubber $n = 5$, muscle $n = 3$, liver $n = 5$), and Iqaluit (blubber $n = 3$, muscle $n = 0$, liver $n = 0$) (Fig. 1 and Table S2). The collection was part of a long-standing community-based sampling program [45]. The collected samples originate from two walrus management stocks: the Foxe Basin (FB) stock (i.e., samples from Sanirajak) and the Hudson Bay-Davis Strait (HBDS) stock (all other sites) [46]. According to Stewart [46], "stocks of walruses" are defined as distinct aggregations within a population that are geographically or behaviorally isolated and impacted by human activity in a way that affects their productivity. These aggregations are often identified by breeding groups during the winter and are influenced by patterns such as age and sex segregation, as well as genetic changes [46].

Ringed seal tissues were collected in 2017 by local hunters, with blubber samples from Arviat, Nunavut ($n = 10$) and blubber ($n = 14$) and liver ($n = 12$) from Sachs Harbour, Northwest Territories (Fig. 1 and Table S2). The morphometric and sex variables of walrus and ringed seal samples are summarized in Table S3. Age class comparisons were not conducted for walruses because most individuals sampled were adults, with only 1–3 juveniles per stock. For ringed seals, age information was unavailable. This limitation prevented the present study from assessing

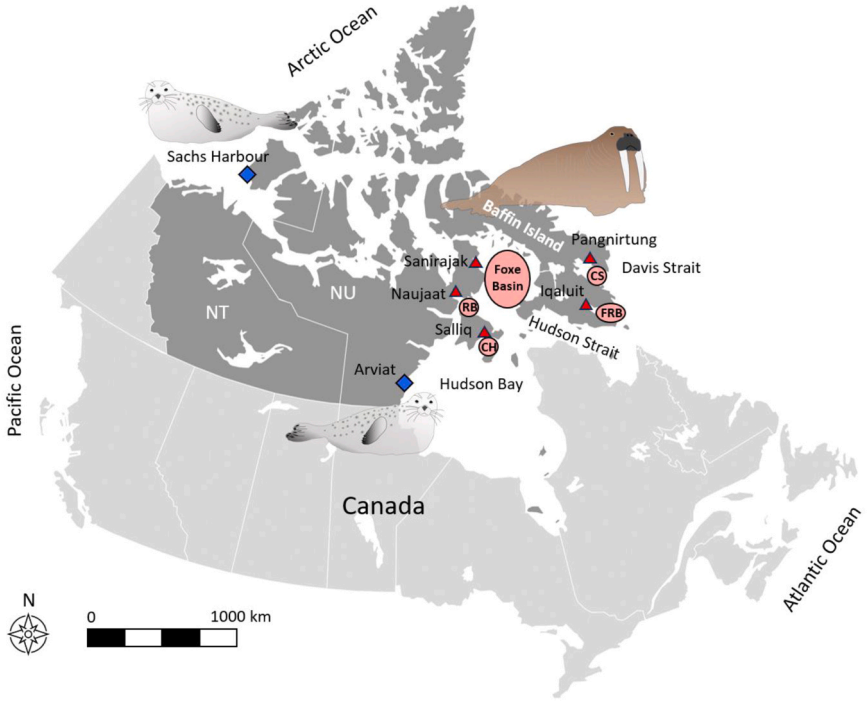

Fig. 1. Sampling locations of walrus (circle) and ringed seal (diamond) in the Canadian Arctic. Walrus tissues were collected from Sanirajak (Foxe Basin), Naujaat (Repulse Bay (RB)), Salliq (Coral Harbour (CH)), Pangnirtung (Cumberland Sound (CS)) and Iqaluit (Frobisher Bay (FRB)). Red triangles represent communities close to walrus sampling locations. The sampling site symbols (diamond) of ringed seals overlap the communities of Arviat and Ikaahuk (Sachs Harbour). The map was created with MapChart.

age-related differences in contaminant concentrations in walruses and ringed seals.

2.2. Chemicals and reagents, sample preparation, instrumental analysis and QA/QC

This study analyzed 10 BZT-UVs, 6 UVFs, 1 SPA, and 6 Ar-SAs. The structures, full names, acronyms, and CAS registry numbers of these target contaminants are presented in Figure S1, and their physicochemical properties are summarized in Table S4. Sample preparation and instrumental analysis followed published methods [13]. In brief, homogenized biota samples were extracted using an ultrasound bath with a solvent mixture of *n*-hexane and dichloromethane, followed by clean-up via gel permeation chromatography. The final extracts were analyzed using gas chromatography-mass spectrometry. Additional details and QA/QC results are provided in Text S1, Tables S5 and S6.

2.3. Data analysis

Data analysis was conducted using GraphPad Prism 10.0 (La Jolla, CA) and R 4.4.1 (RStudio v2024.04.2). Descriptive statistics were calculated for analytes with a detection frequency (DF) of 30 % or higher using the Robust Regression on Order Statistics (ROS) method, which is suitable for censored data when the DF is greater than 20 % [47]. The 30 % DF threshold was chosen to ensure that sufficient detected values are available for reliable estimates. This method reliably estimates medians, quartiles, and percentiles for censored datasets [47]. Unlike simple substitution methods (e.g., replacing non-detects with half of the limit of detection (LOD)), which can introduce bias and distort measures of central tendency when $n > 3$, ROS provides a statistically sound alternative [47]. All descriptive statistics, including box plots, were generated using ROS where applicable. The NDExpo tool (https://expostats.ca) was used to estimate descriptive statistics when $n \geq 5$ and detects ≥ 3 individuals based on the ROS method. This is because the NDExpo tool requires a minimum of five total observations and at least three detections for reliable output. When $n = 4$–6 and target contaminants were detected in more than 30 % of samples, descriptive statistics were estimated using the *cenros* function in the Nondetects and Data Analysis (NADA) package (version 1.6–1.1) in R, if the dataset did not meet the requirements of the NDExpo method. Half LOD was used to replace non-detects when $n \leq 3$ and the contaminant was detected in at least one sample [47]. This is because, at very small sample sizes (e.g., when $n = 3$), the ROS method yields results equivalent to substituting non-detects with LOD/2, so that ROS is not designed for application to such limited datasets [47].

Contaminant concentrations can be corrected for lipid content for better comparisons between tissues or species if the wet weight (ww)-based concentration correlates with the lipid content in biota samples [48]. In the present study, concentrations were not corrected for lipid content because there was no correlation between target contaminant concentrations (ww) and lipid content in walrus or ringed seal tissues in most cases (Table S7). Therefore, concentrations (mean ± standard deviation (SD) and median) were reported on a ww basis. Data were log-transformed to approximate a normal distribution before statistical comparisons. The ROS-based estimation method assumes lognormality [47], which supports the use of parametric tests following transformation. To compare tissue distribution, we initially analyzed paired samples from individuals with three tissues using repeated-measures Analysis of Variance (ANOVA) to assess differences across tissues. For pairwise comparisons between two tissues, we applied paired *t*-tests. Then, Welch's ANOVA, followed by Dunnett's T3 multiple comparisons, was used to compare contaminant levels among all samples of the three walrus tissues. The unpaired *t*-test with Welch's correction was used to compare contaminant concentrations between the two groups of all samples. For contaminants detected in ≥ 30 % of walrus or ringed seal samples, the maximum likelihood method in R (v2022.07.2; *clikcorr* package, v1.0) [49], a method for testing correlations involving censored concentration data (i.e., concentration < LOD), was used to analyze associations between $\log_{10}$-transformed contaminant concentrations and the body length of walruses or ringed seals. Multivariate analyses were not pursued further due to the lack of robust statistical methods capable of adequately handling censored data with non-detects. The significance level was set as $p \leq 0.05$. When comparing with data reported in the literature, concentrations were presented on a lipid-weight (lw) or ww basis, or as mean, median, or range values, depending on the format and availability of the data in the literature. An overview of the research workflow is presented in Figure S2.

3. Results and discussion

3.1. BZT-UVs

3.1.1. Tissue distribution

Nine of 10 target BZT-UVs were detected in walruses, excluding 2-(2H-benzotriazol-2-yl)-4,6-bis(1-methyl-1-phenylethyl)phenol (UV234), whereas six out of 10 congeners were found in ringed seal tissues, including UV328, 2-(2H-benzotriazol-2-yl)-4-(*tert*-butyl)-6-(*sec*-butyl)phenol (UV350), 2-(2H-benzotriazol-2-yl)-p-cresol (UVP), 2-[3-(2H-benzotriazol-2-yl)-4-hydroxyphenyl]ethyl methacrylate (UV090), UV234 and UV329 (Table S8). UV328 and UV090 were the dominant congeners in walrus blubber, while UV328 and UV327 dominated the walrus muscle and liver, respectively (Figure S3). UV328 was also a dominant congener in ringed seal blubber, followed by UVP and UV350 (Figure S3). The detection frequency of BZT-UVs in ringed seal liver was very low (0 %–8 %) (Table S8), so the composition profiles are not discussed further.

When all samples are considered, the ΣBZT-UVs concentrations were significantly different among tissues in walruses ($F_{2,75.05} = 31.26$, $p < 0.0001$). Post-hoc Dunnett's T3 tests showed that concentrations were significantly higher in blubber than muscle ($p < 0.0001$) and liver ($p < 0.0001$), and higher in muscle than liver ($p = 0.0185$), supporting the trend of blubber > muscle > liver (Table 1 and Fig. 2). Ringed seals had a similar pattern, with BZT-UVs more frequently detected in blubber than in liver; however, no statistical comparison was conducted due to liver detection frequencies being below the threshold of 30 % (Table 1 and Fig. 2). Tissue distribution comparisons of ΣBZT-UVs using paired samples from individuals with all tissues or all available samples produced consistent patterns (Fig. 2).

For individual BZT-UV congeners with detection frequency ≥ 30 % (all samples considered together), UV090 (1.59 ± 1.22 ng/g ww; median: 1.15 ng/g ww in blubber; not detected (ND) in muscle and liver) was more frequently detected in walrus blubber (42.5 % detection frequency) than in muscle (0 %) and liver (0 %) (Fig. 3). Comparisons using paired walrus tissue samples showed the same pattern for UV090 (Fig. 3). Different from comparing all samples, paired walrus samples ($n = 23$) showed significantly higher levels of UV328 in the blubber (0.16 ± 0.19 ng/g ww; median: 0.09 ng/g ww) than in the muscle (0.06 ± 0.08 ng/g ww; median: 0.04 ng/g ww) ($p = 0.0098$) (Fig. 3). Walruses had significantly higher blubber ΣBZT-UVs than ringed seals (2.26 ± 3.49 ng/g ww *vs.* 0.41 ± 0.79 ng/g ww; $p = 0.0037$) (Fig. 2). However, the compositions of individual BZT-UV congeners in the blubber of the two species differed, as discussed above (Table S8 & Figure S3). These results suggest that blubber is the most appropriate tissue for long-term monitoring of these compounds in marine mammals. This is relevant for trend studies and passive surveillance programs where blubber biopsies are more feasible than liver sampling. Although significant differences were observed among tissues or species, these findings should be interpreted with caution due to the high variability in concentrations (with SDs exceeding 100 % of the mean in some cases). This level of variability may influence the robustness of group comparisons, even with moderately sized sample groups ($n = 23$–40).

The higher accumulation of BZT-UVs in blubber is consistent with

Table 1
Mean ± SD, median, and concentration ranges (ng/g, ww) of ΣBZT-UVs and ΣUVFs in the Canadian walrus and ringed seal tissues. The concentrations were quantified via GC-MS as described in Section 2.2. NA: not calculated due to detection frequency < 30 %; LOD: limit of detection.

Species	Tissues	Total BZT-UVs			Total UVFs		
		Mean ± SD	Median	Range	Mean ± SD	Median	Range
Walrus	**Blubber (*n*=40)**	2.26 ± 3.49	0.91	<LOD−17.8	2.86 ± 3.92	1.32	<LOD−17.1
	Muscle (*n*=24)	0.14 ± 0.22	0.04	<LOD−0.96	1.65 ± 1.65	1.19	<LOD−7.27
	Liver (*n*=36)	0.31 ± 0.89	0.01	<LOD−4.53	2.85 ± 5.93	0.59	<LOD−31.5
Ringed Seal	**Blubber (*n*=24)**	0.41 ± 0.79	0.07	<LOD−27.2	4.28 ± 5.85	1.43	<LOD−19.1
	Liver (*n*=12)	NA	NA	<LOD−8.41	2.07 ± 1.56	1.24	<LOD−5.06

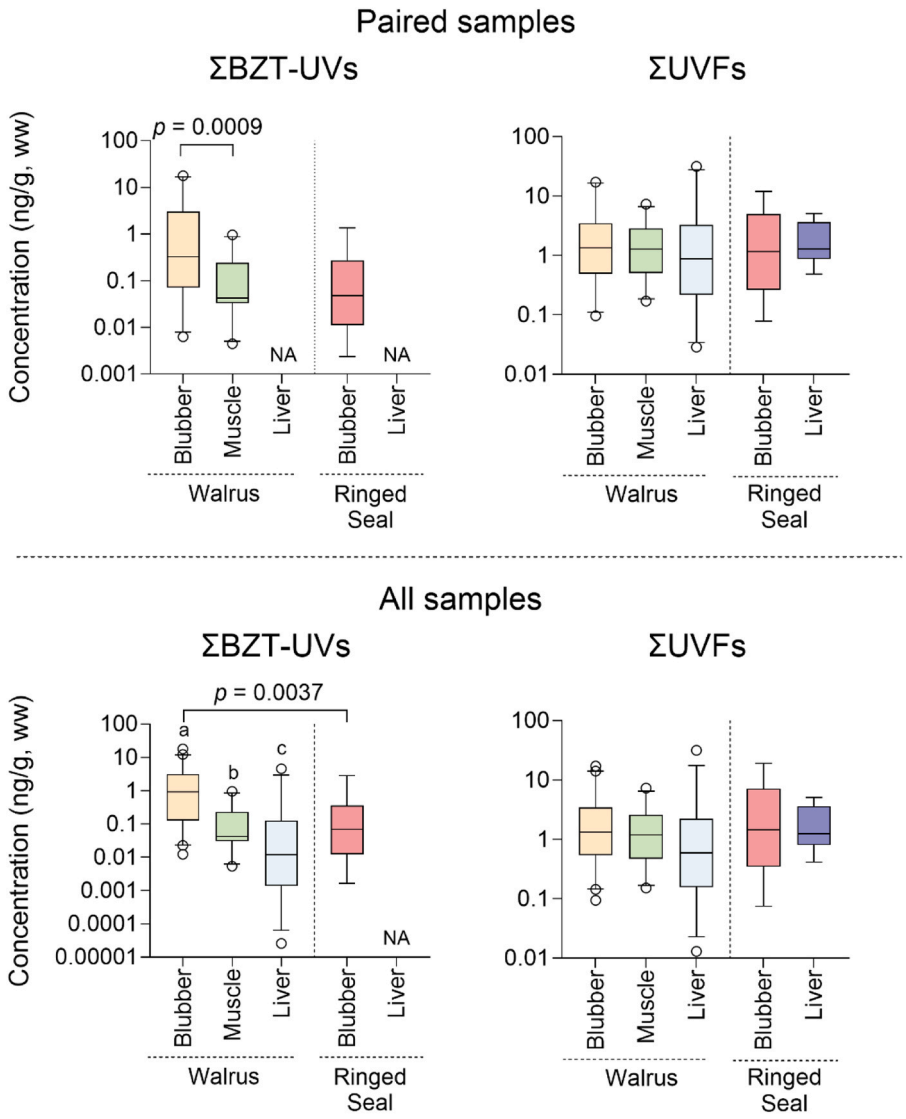

Fig. 2. Tissue distribution of total ΣBZT-UVs and ΣUVFs in walruses and ringed seals. The paired sample sizes are $n = 23$ for walrus and $n = 11$ for ringed seals. Boxplots are defined as follows: center line, median; box plot edges, 25th and 75th percentile; whiskers, 5th and 95th percentile of the distribution. NA: detection frequency < 30 %; Different letters indicate significant differences, and p values show statistical differences between species. The circles represent outliers that are below the 5th percentile or above the 95th percentile.

their high lipophilicity (e.g., log K_{ow} = 3.00–7.67) (Table S1). Human toxicokinetic studies suggest that compounds such as UV328 and UV327 tend to store in adipose tissue, which enables their recycling via enterohepatic circulation and a temporary bypass of liver metabolism [50,51]. This pattern is consistent with our observation of elevated BZT-UV levels in the blubber of both walruses and ringed seals. Also, we found that the lipid content in the muscle of the walrus from the FB stock positively correlates with UV328 concentrations (likelihood $r_{17} = 0.48$, $p = 0.04$) (Table S7), suggesting that lipid content may influence the partitioning of UV328 into the muscle of this walrus stock. In contrast, toxicokinetic studies report conflicting information about UVP [52,53]. While UVP is rapidly metabolized in humans and much faster than UV328 and UV327, it has the longest half-life among BZT-UVs in the blood of Sprague-Dawley rats (*Rattus norvegicus*) [52,53]. These differences emphasize the species-specific nature of toxicokinetics and highlight the uncertainty in extrapolating such findings to marine mammals.

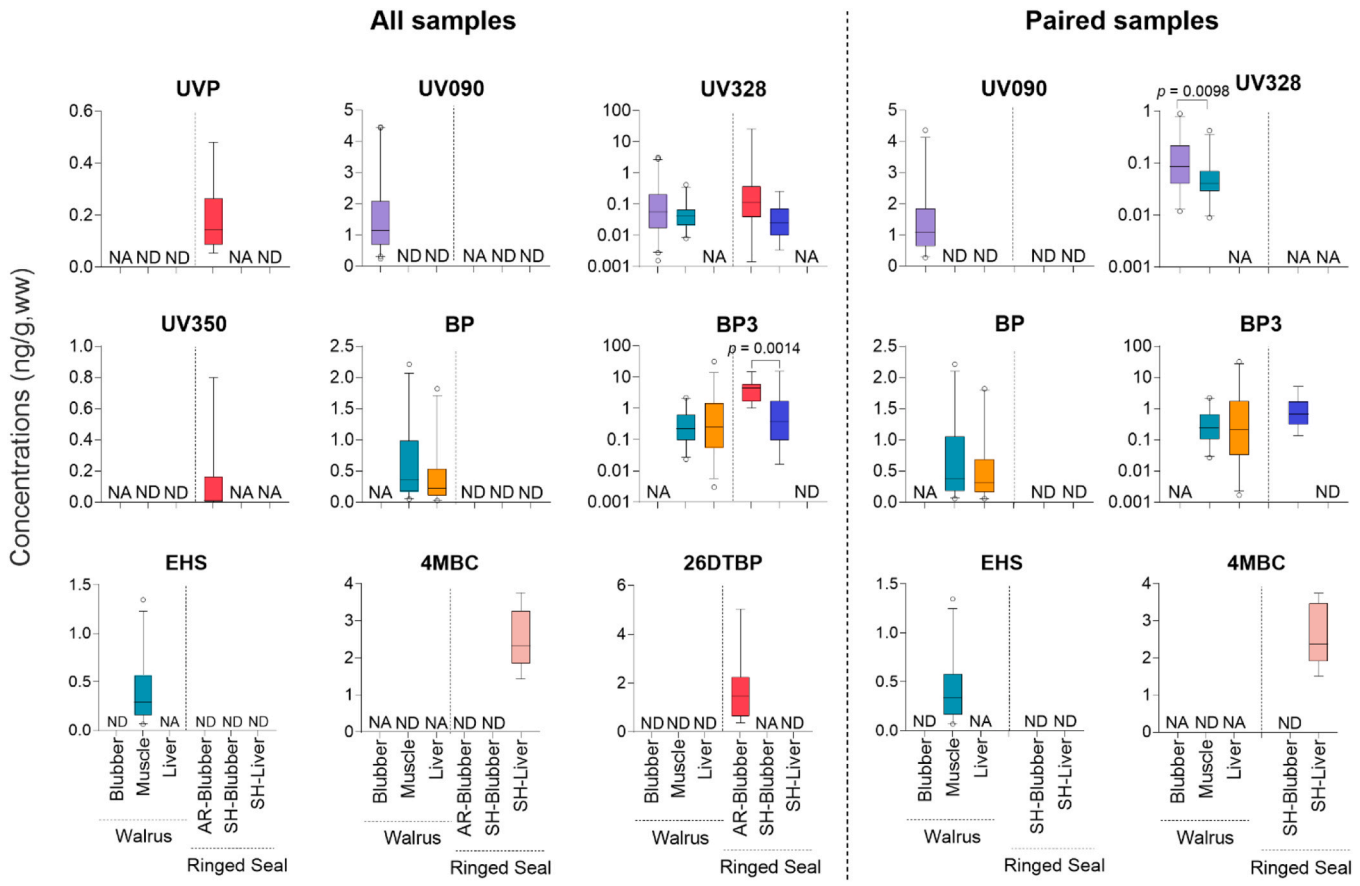

Fig. 3. Tissue distribution of BZT-UVs (UVP, UV090, UV328, UV350), UVFs (BP, BP3, EHS, 4MBC), and 26DTBP in walruses and ringed seals. Box plots are defined as follows: center line, median; box plot edges, 25th and 75th percentile; whiskers, 5th and 95th percentile of the distribution. NA: detection frequency < 30 %; ND: not detected. The circles represent outliers that are below the 5th percentile or above the 95th percentile. AR: Arviat. SH: Sachs Harbour.

Differences in absorption, distribution, metabolism, and excretion physiology among species such as walruses and seals may further influence distribution and elimination patterns.

3.1.2. Comparison with concentrations in marine mammal blubber and liver reported in the literature

The following comparisons with previously reported data are descriptive and should be interpreted cautiously, as differences in contaminant sources, sampling time, environmental distribution, and species-specific physiology and ecology may influence contaminant profiles. The tissue distribution pattern of BZT-UVs in walruses and ringed seals was, in general, similar to that of St. Lawrence Estuary (Canada) belugas (blubber>liver) [18]. However, BZT-UV levels in the tissues of walruses and ringed seals in the present study were generally lower than concentrations in marine mammals from the Arctic or temperate regions (Tables S8 and S9). For example, concentrations of UV327 and UV328 in the blubber of finless porpoises (*Neophocaena phocaenoides*) collected from the Ariake Sea in Japan were in the range of 4.5 – 31 ng/g (median: 9.5 ng/g ww) and 11 – 64 ng/g (median: 20 ng/g ww), respectively (Table S9) [19]. Additionally, concentrations of UV329 in the blubber of St. Lawrence Estuary beluga were in the range of < 0.5–370 ng/g ww (median: 6.8 ng/g ww) (Table S9) [18]. The generally lower BZT-UV concentrations in walrus and ringed seal tissues compared to those of marine mammals from temperate regions may reflect a combination of factors. Species in temperate regions are typically closer to major sources of these contaminants, such as wastewater discharges, urban runoff, and industrial activities. This leads to higher environmental exposure. In contrast, the remote habitats of the Canadian Arctic likely experience lower direct inputs, and long-range transport processes may deliver contaminants at more diluted concentrations. Additionally, differences in prey composition, trophic position, and species-specific metabolism may influence bioaccumulation of these contaminants. However, data on these factors are currently lacking.

Moreover, the same BZT-UV congeners were previously analyzed in the livers of polar bears (including samples from Arviat) and belugas (from Hendrickson Island, in relative proximity to Sachs Harbour) in the Canadian Arctic [13]. Therefore, the results of ΣBZT-UVs can be compared with the present study. The median concentrations of ΣBZT-UVs were higher in the livers of the polar bears (1.31 ng/g ww) and beluga (0.44 ng/g ww) (Table S10) than in the walruses (0.01 ng/g ww) (median not available (NA) for ringed seals due to low detection frequency) [13]. Further analysis of additional species in the Arctic would better elucidate the trophodynamics of BZT-UVs in the Arctic food web. In the Norwegian Arctic, UV326 (range: 0.31 – 0.51 ng/g ww; mean: 0.37 ng/g ww) and UV328 (range: 0.08 – 0.36 ng/g ww; mean: 0.18 ng/g ww) were detected in the livers ($n = 10$) of American mink collected from the islands of Sommarøy and Hillerøy in 2013–2014 (Table S10) [23]. Since UV326 was not detected in any of the walrus and ringed seal liver samples in the present study, and UV328 was found in only 8 % (ringed seals) and 11 % (walruses) of the liver samples, these results suggest different sources, transportation and/or transformation of UV326 and UV328 in the Arctic environments. In addition, UV320 and UV329 were not detected in the liver samples of the American mink from the Norwegian Arctic (Table S10) [23], which is consistent with the results of the walruses and ringed seals in the present study. No studies reporting BZT-UVs in the muscle of marine mammals could be found in the literature.

3.2. UVFs

3.2.1. Tissue distribution

In walruses, all target UVFs were detected in at least two tissue types, except for ethylhexyl methoxycinnamate (EHMC), which was not detected in any tissues. The concentrations of ΣUVFs are summarized in Table 1. Unlike ΣBZT-UVs, no significant differences in the concentrations of ΣUVFs were found among tissues of walruses (Fig. 2). 2-Hydroxy-4-methoxybenzophenone (BP3) and benzophenone (BP) were the most prevalent UVFs in walrus tissues (Figure S3), while their detection frequencies (31 % – 50 %) in the muscle and liver were higher than in blubber (15 % – 20 %) (Fig. 3 and Table S10). Another UVF frequently found in walrus muscle was 2-ethylhexyl salicylate (EHS) (detection frequency: 58 %; 0.39 ± 0.31 ng/g; median: 0.29 ng/g ww), with a higher detection frequency than in blubber (0 %) and liver (6 %), where descriptive statistics were not available. This suggests greater accumulation potential of EHS in walrus muscle (Fig. 3 and Table S11). Both lipid content and protein binding may contribute to this distribution pattern. For the walrus muscle samples from the FB stock, lipid content was positively correlated with BP (likelihood $r_{17} = 0.60$, $p = 0.009$) and EHS concentrations (likelihood $r_{17} = 0.47$, $p = 0.048$) (Table S1), suggesting that lipid content may influence the partitioning of these two UVFs into the muscle of the FB walruses. Some UVFs are also known to bind with proteins. For example, studies have demonstrated the ability of BP, BP3, and EHS to bind to proteins, including human serum albumin and skin proteins [54,55]. This may also help explain their relatively higher concentrations in the muscle and liver tissues of walruses. However, data on protein-binding affinities of these compounds in marine mammals are currently lacking, and further research is needed to confirm these associations.

BP3, 4MBC, 3,3,5-trimethylcyclohexyl salicylate (HMS), and EHMC were detected in ringed seal tissues with the median ΣUVFs of 1.43 ng/g ww in blubber (4.28 ± 5.85 ng/g; range: < LOD – 19.1 ng/g ww) and 1.24 ng/g ww in the liver (2.07 ± 1.56 ng/g; range: < LOD - 5.06 ng/g ww) (Table 1 and S11). The ΣUVFs were not different between the two tissues in ringed seals (Fig. 2). The dominant UVFs in the blubber and liver of ringed seals were BP3 and 4MBC, respectively (Fig. 3 and S3). This tissue-specific distribution underscores the importance of including walrus muscle in future monitoring efforts for BP, BP3, and EHS, as muscle is commonly consumed and may better reflect internal exposure relevant to human health. In contrast, for BP3 in ringed seals, blubber appears to be the more suitable tissue for monitoring due to its higher accumulation.

In ringed seals, BP3 in the blubber was 3.39 ± 4.34 ng/g ww (median 1.58 ng/g ww), whereas it was not detected (ND) in the liver, which differs from the results in walruses (muscle and liver > blubber) (Fig. 3 and Table S11). Previous *in vivo* studies in Sprague-Dawley and Fischer rats have demonstrated that the tissue distribution of BP3 in mammals is dependent on the route of exposure [56–58]. After oral administration, the rat liver could accumulate more BP3 than other tissues, including muscle and fat [56,57]. However, after dermal administration, some tissues (e.g., intestine, fat, and muscle) showed higher levels of BP3 than the liver, which was also affected by exposure time [56–58]. Orally ingested BP3 can undergo first-pass metabolism in the liver, which may result in lower levels of BP3 in the blood plasma, thereby affecting its circulation and deposition in rat bodies [56,57]. In contrast, the metabolism of BP3 absorbed through dermal exposure will be much slower, contributing to relatively higher levels of BP3 in blood plasma and deposition in other tissues in rats [56,58]. Thus, factors such as exposure and metabolism may contribute to the differences in tissue distribution of BP3 in different mammal species.

Ringed seal livers tended to accumulate more 4MBC than their blubber (2.47 ± 0.78 ng/g ww *vs.* <LOD) (Fig. 3 and Table S11). It has been reported that 4MBC can undergo extensive first-pass metabolism in Sprague-Dawley rats after oral ingestion, resulting in very low levels of parent 4MBC in the blood for circulation and deposition in various body tissues [59]. Similar to BP3, the blood concentration of parent 4MBC was much higher in rats after dermal exposure than after oral exposure [60]. In addition, Wistar rats showed faster metabolism of 4MBC after dermal exposure than humans, suggesting species-specific metabolism of 4MBC in mammals [60]. The differences in the tissue distribution patterns of 4-MBC and BP3 in ringed seals suggest that these contaminants may be taken up by different routes of exposure and/or may be metabolized in different ways.

3.2.2. Comparison with concentrations in marine mammals reported in the literature

Comparisons with other marine mammals are presented to highlight general trends rather than definitive assessments, since contaminant levels in tissues can be influenced by diverse anthropogenic, environmental, and biological factors. The distribution of 4MBC in ringed seals in the present study differs from that of a deceased ringed seal from the German coasts of the Baltic Sea [21], which had higher levels of 4MBC in its blubber (70 ng/g lw) than liver (ND) [21]. The dissimilarities may be due to the different exposure processes of ringed seals to 4MBC (e.g., via water and/or diet) between regions. It should also be noted that the previous result was based on a single sample, and the cause of death of the individual was unknown [21]. Stranded or sick animals can be more likely to accumulate different levels of contaminants than healthy animals [61], although the reasons for this can vary. The unknown degradation of 4MBC between the time of death and the time of sampling of the ringed seal may also affect the results. In addition, Reiter et al. [21] used silicone polydimethylsiloxane passive samplers (PDMS) to accumulate target contaminants from biota samples instead of direct extraction. Since the partition coefficient of these target contaminants between the biota sample and the PDMS is unknown, the estimated final concentrations in tissues may be highly uncertain [21]. Comparisons to passive sampling collected data (e.g., PDMS-based samplers used by Reiter et al. [21]) are subject to limitations due to differences between methodologies. These include uncertainty in biota-PDMS distribution coefficients, differences in extraction efficiencies, and variations in calibration approaches, thereby reducing the reliability of direct concentration comparisons between methods. Despite these methodological differences, we included this comparison to provide context, as data on these compounds in marine mammals remain extremely limited in the published literature.

The concentrations of target UVFs measured in walrus and ringed seal blubber and muscle in the present study were generally lower than those of other marine mammals from Arctic and temperate regions (Table S9 and S12). EHMC and 4MBC, which were not detected or detected with low frequencies in walrus or ringed seal blubber and muscle (Table S9 and S12), were detected in the blubber and muscle of Franciscana dolphins (*Pontoporia blainvillei*) (blubber-median: EHMC 66.8 ng/g, lw, 4MBC 7.3 ng/g lw; muscle-median: EHMC 69 ng/g, lw, 4MBC 110 ng/g lw) and Guiana dolphins (*Sotalia guianensis*) (blubber-median: EHMC up to 205 ng/g, lw, 4MBC 25.8 ng/g lw; muscle-median EHMC 84.3 ng/g lw, 4MBC 155 ng/g lw) from the Brazilian coast (Table S9 and S12) [20]. Much higher levels of 4MBC and HMS were also found in the blubber of harbor porpoises (*Phocoena phocoena*) (median: 4MBC 65.5 ng/g lw; HMS 41.1 ng/g lw), harbor seals (*Phoca vitulina*) (median: 4MBC 71 ng/g lw; HMS 534 ng/g lw), ringed seal ($n = 1$; 4MBC 70 ng/g lw; HMS 121 ng/g lw), and orca (*Orcinus orca*) ($n = 1$; 4MBC 23 ng/g lw; HMS not detected) from the German coasts of the North and Baltic Seas [21] than in the blubber of Canadian Arctic walruses (median: NA) and ringed seals (4MBC: ND; median of HMS in Arviat ringed seal: 1.05 ng/g, lw and NA for other samples) from the present study (Table S9). The median BP3 in St. Lawrence Estuary beluga blubber (3.15 ng/g ww) [18] was also higher than that of walrus (NA) and Sachs Harbour ringed seals (0.39 ng/g ww) but comparable to that in Arviat ringed seal blubber (4.50 ng/g ww) (Table S9). Similar to BZT-UVs, the lower UVF concentrations observed in Arctic marine mammals compared to those in temperate regions are likely influenced

by fewer nearby sources, as well as ecological and physiological factors discussed earlier.

Comparisons of UVF levels in the livers of different marine mammals were more varied than the pattern observed in blubber and muscle. EHMC was detected only in the liver of American mink (mean 4.0 ng/g, range <3 – 4.9 ng/g ww) from the Norwegian Arctic but not in any Canadian Arctic species, including walruses and ringed seals from the present study, as well as polar bears and beluga whales (Table S10) [13,23]. In contrast, average BP3 concentrations in walrus liver (2.01 ± 5.52 ng/g; median: 0.25 ng/g ww) were higher than those measured in ringed seal (ND) or American mink livers (mean 0.65 ng/g, range <0.2 – 2.3 ng/g ww), though much lower than levels observed in beluga whale liver from the St. Lawrence Estuary (mean 115 ng/g; median 10.9 ng/g ww) (Table S10) [13,23]. For 4MBC, polar bear (median 4.22 ng/g ww) showed higher median concentrations than ringed seal livers (2.47 ± 0.78 ng/g; median 2.32 ng/g ww) (Table S10) [13]. Additionally, HMS was detected sporadically across all marine mammal liver samples (Table S10), suggesting limited accumulation of this UVF in the liver of these species.

3.3. Industrial antioxidants

Three industrial antioxidants, including 26DTBP, bis(4-(2,4,4-trimethylpenta-2-yl)phenyl)amine (C8C8) and bis[4-(2-phenyl-2-propyl)phenyl]amine (diAMS), were detected in walrus and ringed seal tissues (Tables S13 and S14). Overall, the detection frequency and levels of industrial antioxidants were lower than those of UV absorbents, suggesting that they may have less long-range transport potential and/or be less accumulative in marine mammals, even though their production volume is greater [16] (Table S1). This pattern is consistent with results in the liver of various Canadian Arctic seabird, mammal, and fish species [13].

3.3.1. 26DTBP

The SPA 26DTBP was detected in 29 % of ringed seal blubber with concentrations in the range of <LOD – 5.03 ng/g (ww) (Table S13). This finding supports the use of blubber as a preferred matrix for monitoring this antioxidant in Arctic marine mammals. A non-target study qualitatively found that 26DTBP was prevalent in the blubber of polar bears (detection frequency 67 %), killer whales (40 %), narwhals (27 %), and long-finned pilot whales (53 %) from East Greenland [24]. The referenced study [24] did not confirm the identity of 26DTBP using an authentic standard, leaving uncertainty as to whether the detected compound was 26DTBP or an isomer, such as 2,4-di-*tert*-butylphenol. In addition, the 26DTBP median level in the blubber of ringed seals from Arviat was 1.45 ng/g (ww), whereas the median concentration in livers of polar bears from the same area was 1.91 ng/g (ww) [13]. Cui et al. [62,63] have identified potential carcinogenic risks of 26DTBP and its metabolite 2,6-di-*tert*-butyl-1,4-benzoquinone in mice and rats (e.g., the lowest tested dose of 0.001 μM ≈ 0.2 ng/g to H4IIE rat hepatoma cells), suggesting that measured levels of these contaminants may pose potential health risks to mammals. These results highlight the importance of further quantifying the concentrations of 26DTBP and its metabolites and assessing the risks of this SPA to Arctic mammals.

3.3.2. Ar-SAs

For the Ar-SAs, walruses and ringed seals showed different patterns (Table S14). C8C8 was found in the blubber (< LOD – 0.31 ng/g ww) and muscle (< LOD – 0.03 ng/g ww) of walruses but was not detectable in their liver samples or any ringed seal tissues (Table S14). In contrast, diAMS was found in the liver of ringed seals (< LOD – 1.56 ng/g ww) but not in the liver and blubber of walruses and was sporadically detected in the muscle of walruses (< LOD – 0.11 ng/g ww) (Table S14). Given that the detection limits for Ar-SAs are comparable to or lower than those of other target contaminants (Table S6), the low detection frequencies may reflect environmental transformation or biodilution [31,32], rather than analytical limitations.

3.4. The concentrations of target contaminants in relation to sex and body size

Biological and ecological factors can affect tissue distribution patterns. This study examined the effects of sex, body size, and spatial variation on the accumulation of these emerging contaminants in walruses and ringed seals.

3.4.1. Walrus

A distinct sex difference was observed in walrus livers of the FB stock, with females accumulating higher concentrations of ∑BZT-UVs (female 0.66 ± 0.92 ng/g ww *vs.* male 0.38 ± 1.20 ng/g ww; $p = 0.0078$) and ∑UVFs (female 1.61 ± 0.88 ng/g ww *vs.* male 1.04 ± 1.82 ng/g ww; $p = 0.0089$) compared to males (Figure S4). UV327, BP3, and BP from the FB stock, as well as ∑UVFs, BP3, and BP from the HBDS stock, were more frequently detected in female livers than in males, suggesting greater accumulation potential in females (Figure S4). However, statistical comparisons were not conducted due to low detection frequencies in males (Figure S4). This disparity may be attributed to a slower metabolic rate in female walruses [64], which allows for greater accumulation of these contaminants. However, the biotransformation kinetics of UV absorbents in walruses remain unknown, warranting further investigation. In FB during summer, female walruses may consume larger amounts of prey than males, although this difference was not statistically significant [65]. Both sexes fed predominantly on the bivalve *Mya truncata*, but females obtained a significantly higher proportion of their energy intake from the clam *Hiatella arctica* (9.9 % *vs.* 2.2 % for males) [65]. These sex-specific dietary patterns may influence the elevated concentrations of certain contaminants in female livers.

Unlike liver samples, walrus muscle showed no significant sex-specific differences in target contaminant levels (Figure S5). The pattern in blubber showed some variability between walrus stocks (Figure S6). BZT-UVs demonstrated no sex-based differences in the blubber of walruses of either stock (Figure S6). Although UV328 was more frequently detected in the female blubber from the HBDS stock (40 % *vs.* 20 %) (Figure S6), statistical comparisons could not be performed due to the detection frequency in male blubber below the threshold of 30 %. For UVFs, detection patterns of BP3 differed between the two stocks. In the FB stock, BP3 was detected more frequently in the blubber of females (38 % *vs.* 0 %), whereas in the HBDS stock, it was more frequently detected in male blubber (40 % *vs.* 0 %) (Figure S6). Due to zero detection frequencies in one sex, these observations of BP3 are descriptive and not supported by statistical comparisons.

These sex-based differences likely reflect a combination of physiological factors (e.g., metabolism, diet) and compound-specific properties such as persistence and lipophilicity. To further explore individual variability in contaminant accumulation, we also examined body size effects in walrus males and females, which can provide insight into bioaccumulation trends independent of reproductive or sex-specific behaviors. Combined analysis of males from both stocks revealed a positive correlation between body length and ∑BZT-UVs levels in blubber (likelihood $r_{19} = 0.67$, $p = 0.02$), while ∑UVFs concentrations in their livers showed an inverse relationship with body length (likelihood $r_{17} = -0.52$, $p = 0.03$). This suggests that larger males exhibit elevated BZT-UV accumulation in blubber but reduced UVF retention in the liver. These different trends may reflect differences in the biotransformation potential of these chemical classes, with BZT-UVs potentially being more persistent and bioaccumulative than UVFs; however, there is currently limited empirical evidence in the literature to confirm these mechanisms in marine mammals.

3.4.2. Ringed seal

The higher detection frequency of UV350 (44 % *vs.* 0 %), ∑UVFs

(78 % *vs.* 20 %), and BP3 (56 % *vs.* 20 %) in blubber and ∑BZT-UVs (43 % *vs.* 0 %) in the liver of male ringed seals from Sachs Harbour compared to females indicates that these contaminants may be more accumulated in males (Figure S7). This finding is consistent with sex differences in the accumulation of organohalogen compounds in ringed seals, which have been attributed to the loss of contaminants in females during reproduction (e.g., depuration to placenta and milk and elimination from the body) [43,44]. Although the biotransformation process of UV absorbents in ringed seals is unknown, adult ringed seals do not exhibit sex differences in the phase I and II enzyme activity [66]. Thus, the variation in biotransformation between the sexes may not significantly affect this pattern within a population of ringed seals. Comparisons between males and females were not made for the Arviat samples due to the limited sample size, as only one male was collected.

Building on these sex-based differences, we next examined the role of the body size of ringed seals as a potential explanatory variable. BP3 (all: likelihood $r_{10} = -0.66$, $p = 0.019$; female: likelihood $r_9 = -0.66$, $p = 0.027$) and 26DTBP (all: likelihood $r_{10} = -0.74$, $p = 0.007$; female: likelihood $r_9 = -0.74$, $p = 0.01$) concentrations in ringed seal blubber samples from Arviat were negatively correlated with body length, suggesting a growth dilution effect in the blubber of this population. This pattern is consistent with the growth dilution of BP3 in wild freshwater and marine fish [67,68]. However, since 90 % of the Arviat blubber samples were from females, this effect may also be related to the increased reproductive and nursing activities of larger individuals. This growth dilution pattern may also be due to higher biotransformation potential in larger and older individuals, as adults are known to have higher phase I enzyme activity than juveniles [66]. In contrast, no growth dilution was found in male ringed seals from Sachs Harbour, while the relationship in females was not analyzed due to low detection frequency.

3.5. Spatial variation

In addition to biological factors, spatial variation in contaminant concentrations may reflect differences in local environmental conditions, contaminant sources, and ecological exposure pathways. We compared the levels of these contaminants in walruses and ringed seals from different locations to evaluate these geographic influences.

Although walrus tissues were collected from different sites, sample sizes of each site were small ($n = 0$–5, in most cases $n = 1$–2 for each sex) (Table S3), except for Sanirajak. Therefore, spatial variations of target contaminants are discussed between the two stocks instead of each site.

Walruses from the FB and HBDS stocks showed different accumulation patterns of UV absorbents across tissues. Both stocks had comparable BZT-UV concentrations in blubber, though FB males displayed a higher UV328 detection frequency than HBDS males (47 % *vs.* 20 %) (Figure S8). For UVFs, there was a difference in BP3 levels based on sex. In blubber, BP3 was more frequently detected in HBDS males than in FB males (40 % *vs.* 0 %), while in females, detection was higher in FB than in HBDS (38 % *vs.* 0 %) (Figure S8). In liver tissues, FB walruses showed higher detection frequencies of BZT-UVs than HBDS walruses in both sexes (males: 36 % *vs.* 0 %; females: 75 % *vs.* 20 %) (Figure S9). In muscle, FB females showed greater concentrations of ∑UVFs (3.33 ± 2.72 ng/g ww *vs.* 0.48 ± 0.47 ng/g ww; $p = 0.02$) and higher detection frequencies of BP (75 % *vs.* 25 %), EHS (100 % *vs.* 0 %), and HMS (75 % *vs.* 0 %) compared to HBDS females (Figure S10). These findings suggest spatial differences in UV absorbent accumulation for walruses, with the FB stock tending to accumulate more of these contaminants overall. However, these comparisons should be interpreted with caution because, although the FB stock is known to be largely isolated from other stocks [46], it is still possible that some males move to other nearby stocks and then return later [46], reducing the power of this comparison between two stocks.

Ringed seal blubber samples from Arviat showed higher concentrations of BP3 than those from Sachs Harbour (5.18 ± 4.23 ng/g ww *vs.* 2.01 ± 4.15 ng/g ww; $p = 0.0014$) (Fig. 3). UV328 concentrations in Arviat seal blubber averaged 2.80 ± 8.29 (median 0.12) ng/g ww, compared to 0.05 ± 0.07 (median 0.02) ng/g ww in Sachs Harbour seals; however, this difference was not statistically significant ($p = 0.15$). Detection frequencies of UVP (40 % *vs.* 14 %) and 26DTBP (60 % *vs.* 7 %) were also higher in Arviat seal blubber samples. Given the influence of sex on contaminant accumulation (as discussed above), we conducted a separate analysis of female seals from both locations. Contaminant detection frequencies were consistently higher in the blubber of Arviat females than in Sachs Harbour females (UVP: 33 % *vs.* 20 %; UV328: 78 % *vs.* 40 %; UV350: 33 % *vs.* 0 %; BP3: 78 % *vs.* 20 %; HMS: 44 % *vs.* 20 %; 26DTBP: 56 % *vs.* 20 %) (Figure S11). Since body length did not differ significantly between the two populations or the female groups, length normalization was deemed unnecessary for this comparison. This pattern is consistent with contamination observed for other compounds in seals, such as polybrominated diphenyl ethers (PBDEs) in samples collected between 2003 and 2010 [43]. It is also similar to the distribution pattern of polychlorinated biphenyls (PCBs) in the seal blubber collected before 2000 but different from those collected between 2003 and 2016 [43,44,69]. Local sources may influence the higher contaminant levels in Arviat, as Arviat has a larger human population than Sachs Harbour (2864 *vs.* 104) [70]. In addition, this pattern may reflect the relatively early stage of environmental loading and/or the lower persistence of the emerging contaminants studied in the Arctic compared to legacy contaminants [43]. As a result, higher concentrations are generally observed in southern ringed seal populations, which are closer to human activities and contaminant sources than in more remote northern regions. Moreover, dietary differences between ringed seals from these two regions may influence contaminant accumulation [71]. Seals in Sachs Harbour consume a combination of native Arctic prey, such as Arctic cod and amphipods, and sub-Arctic species, such as capelin [71]. In contrast, the Arviat population relies primarily on sub-Arctic prey, including capelin and sand lance, with little consumption of Arctic cod [71]. These dietary differences are known to be directly linked to variations in various contaminant profiles in seal tissues, such as mercury, legacy POPs, and perfluoroalkyl substances [71].

Although we hypothesize that proximity to human activities and dietary differences may contribute to the elevated levels of UV absorbents and industrial antioxidants observed in Arviat seals, based on established patterns seen with legacy POPs, we recognize that direct environmental monitoring data (e.g., from seawater, sediments, or prey species) are not available. Confirming these potential pathways would require targeted environmental sampling and source tracking, which are beyond the scope of this study and not currently supported by existing literature. More data on these contaminants in the surrounding environment and the food web of walruses and ringed seals and larger sample sizes from each site are needed to better confirm this spatial distribution and to elucidate the mechanisms behind these patterns.

Animal movements may influence their exposure and accumulation of these contaminants; however, the walrus and ringed seal populations studied here are regionally resident. Atlantic walruses in the Canadian Arctic are confined to Arctic and sub-Arctic waters [46]. The FB stock is considered largely isolated to Foxe Basin, and while some walruses from the HBDS stock move seasonally between Canada and West Greenland, their distribution is limited to the Arctic and sub-Arctic [46,72]. Similarly, ringed seals show regional movements within Arctic systems: Arviat populations are genetically distinct and remain within Hudson Bay [73], while western Arctic populations (e.g., near Sachs Harbour) may travel considerable distances depending on season but remain regional migrants [74]. Therefore, the tissue burdens measured here reflect exposures within Arctic ecosystems. Moreover, given the relatively short biological half-lives of UV absorbents [17,75,76], concentrations in tissues are expected to reflect exposures over short timescales (days to weeks), further supporting their interpretation as indicators of local or regional exposure.

3.6. Uncertainty, limitations, and outlook

Differences in detection limits across tissues and between species, primarily due to variations in sample size, matrix effects, and variability in blank levels across analytical batches, may have influenced the comparability of concentrations. Future studies should aim to minimize this issue by standardizing detection limits across matrices whenever feasible.

Comparisons of contaminant concentrations between walruses and ringed seals, and previously reported values in other marine mammals, should be interpreted cautiously, as differences in species-specific physiology, diet, sampling year, and local contaminant sources were not accounted for due to the limited availability of data. These comparisons are presented descriptively to provide general context rather than to draw definitive conclusions about relative bioaccumulation.

We acknowledge that the geographic distribution of walrus samples was uneven, with a majority collected from Sanirajak and fewer individuals sampled from other communities. This reflects logistical constraints and variability in tissue availability through community-based harvest programs. As such, findings related to spatial comparisons should be interpreted with caution, and future studies would benefit from more balanced sampling across multiple regions to better represent the full geographic range of walrus populations in Nunavut.

Although some UV absorbents and industrial antioxidants were detected in tissues that humans may consume, this study does not include an exposure or intake assessment, as key data, such as consumption rates and formal toxicological thresholds, are unavailable.

Future research should investigate sources of these contaminants and additional environmental factors, such as the distribution of these contaminants in water, ice melt, runoff, sediment, and the food webs of walrus and ringed seal habitats, as well as biological factors, including age-related differences and sex-specific metabolic processes that affect contaminant accumulation in these species. These studies are necessary to address the limitations of the current work and evaluate potential adverse effects on Arctic marine mammals.

4. Conclusion

This study provides the first multi-tissue assessment of emerging contaminants, including BZT-UVs, UVFs, 26DTBP, and Ar-SAs, in Atlantic walruses and ringed seals from the Canadian Arctic. BZT-UVs were predominantly accumulated in blubber for both species, consistent with their high lipophilicity and partitioning into lipid-rich tissues. In contrast, $\sum$UVFs did not differ significantly between tissues, with BP, BP3, EHS, and 4MBC being more frequently detected in liver and muscle, suggesting possible affinity for protein-rich matrices and/or active biotransformation. These results highlight distinct tissue distribution patterns across compound classes, reflecting differences in chemical properties, bioaccumulation pathways, and potential metabolic processes. The phenolic antioxidant 26DTBP was frequently detected in ringed seal blubber from Arviat, indicating its widespread presence in this population. Target amine antioxidants Ar-SAs were detected sporadically, suggesting limited accumulation in Arctic marine mammals. Overall, concentrations of these contaminants were lower than those reported in temperate marine mammals, possibly reflecting regional differences in contaminant sources, exposure pathways, or metabolism. Additionally, factors such as sex, body size, and habitat were found to be associated with variations in contaminant levels. These findings provide important current reference data to support long-term monitoring of emerging contaminants in Arctic marine ecosystems. The observed tissue-specific patterns provide insight into the mechanisms of contaminant storage and processing in marine mammals, helping to identify which tissues are most relevant for assessing exposure and toxicological risk. Therefore, these results contribute to a broader understanding of contaminant fate in Arctic food webs and provide a foundation for evaluating potential health impacts on both wildlife and Indigenous communities that rely on these species.

Environmental implication

This study reveals tissue distribution patterns and influencing factors of UV absorbents and industrial antioxidants in Arctic marine mammals, raising concerns about their persistence and potential adverse effects in remote ecosystems. For substances regulated as POPs or listed as Substances of Very High Concern, these data offer a benchmark for tracking the effectiveness of regulatory actions over time and highlight the need for long-term monitoring. As marine mammals are important to the diet and culture of northern Indigenous communities, understanding their emerging contaminant burdens is also critical for assessing human exposure and informing community health and conservation strategies.

CRediT authorship contribution statement

Ingrid-Alejandra Granados-Galvan: Writing – review & editing, Writing – original draft, Visualization, Investigation, Funding acquisition, Formal analysis. **Jennifer F. Provencher:** Writing – review & editing, Resources, Methodology, Investigation, Funding acquisition, Conceptualization. **Mary Gamberg:** Writing – review & editing, Resources, Methodology, Investigation, Funding acquisition, Conceptualization. **Magali Houde:** Writing – review & editing, Resources, Methodology, Investigation, Funding acquisition, Conceptualization. **Steven H. Ferguson:** Writing – review & editing, Resources, Investigation, Funding acquisition. **Mark L. Mallory:** Writing – review & editing, Resources, Investigation, Funding acquisition. **Cory J.D. Matthews:** Writing – review & editing, Resources, Investigation, Funding acquisition. **Zhe Lu:** Writing – review & editing, Writing – original draft, Visualization, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.jhazmat.2025.140121.

Data availability

Data will be made available on request.

References

[1] Chen, H., Hu, X., Yin, D., 2024. Benzotriazole ultraviolet stabilizers in the environment: a review of occurrence, partitioning and transformation. Sci Total Environ 954, 176362. https://doi.org/10.1016/j.scitotenv.2024.176362.

[2] Huang, Y., Law, J.C.-F., Lam, T.-K., Leung, K.S.-Y., 2021. Risks of organic UV filters: a review of environmental and human health concern studies. Sci Total Environ 755, 142486. https://doi.org/10.1016/j.scitotenv.2020.142486.

[3] Liu, R., Mabury, S.A., 2020. Synthetic phenolic antioxidants: a review of environmental occurrence, fate, human exposure, and toxicity. Environ Sci Technol 54, 11706–11719. https://doi.org/10.1021/acs.est.0c05077.

[4] Jin, R., Venier, M., Chen, Q., Yang, J., Liu, M., Wu, Y., 2023. Amino antioxidants: a review of their environmental behavior, human exposure, and aquatic toxicity. Chemosphere 317, 137913. https://doi.org/10.1016/j.chemosphere.2023.137913.

[5] Cantwell, M.G., Sullivan, J.C., Katz, D.R., Burgess, R.M., Bradford Hubeny, J., King, J., 2015. Source determination of benzotriazoles in sediment cores from two urban estuaries on the atlantic coast of the United States. Mar Pollut Bull 101, 208–218. https://doi.org/10.1016/j.marpolbul.2015.10.075.

[6] Langford, K.H., Reid, M.J., Fjeld, E., Øxnevad, S., Thomas, K.V., 2015. Environmental occurrence and risk of organic UV filters and stabilizers in multiple matrices in Norway. Environ Int 80, 1–7. https://doi.org/10.1016/j.envint.2015.03.012.

[7] Lu, Z., Smyth, S.A., Peart, T.E., De Silva, A.O., 2017. Occurrence and fate of substituted diphenylamine antioxidants and benzotriazole UV stabilizers in various Canadian wastewater treatment processes. Water Res 124, 158–166. https://doi.org/10.1016/j.watres.2017.07.055.

[8] Lu, Z., Smyth, S.A., De Silva, A.O., 2019. Distribution and fate of synthetic phenolic antioxidants in various wastewater treatment processes in Canada. Chemosphere 219, 826–835. https://doi.org/10.1016/j.chemosphere.2018.12.068.

[9] Ramos, S., Homem, V., Alves, A., Santos, L., 2016. A review of organic UV-filters in wastewater treatment plants. Environ Int 86, 24–44. https://doi.org/10.1016/j.envint.2015.10.004.

[10] Schutten, K., Morrill, A., Lu, Z., Chandrashekar, A., Cunningham, J.T., Robertson, G.J., et al., 2025. Accumulation of benzotriazole UV-stabilizers in relation to ingested plastics and associated health metrics in larus gulls feeding at a landfill in atlantic Canada. J Hazard Mater 488, 137498. https://doi.org/10.1016/j.jhazmat.2025.137498.

[11] Giokas, D.L., Salvador, A., Chisvert, A., 2007. UV filters: from sunscreens to human body and the environment. TrAC Trends Anal Chem 26, 360–374. https://doi.org/10.1016/j.trac.2007.02.012.

[12] Lu, Z., De Silva, A.O., Provencher, J.F., Mallory, M.L., Kirk, J.L., Houde, M., et al., 2019. Occurrence of substituted diphenylamine antioxidants and benzotriazole UV stabilizers in Arctic seabirds and seals. Sci Total Environ 663, 950–957. https://doi.org/10.1016/j.scitotenv.2019.01.354.

[13] Granados-Galvan, I.A., Provencher, J.F., Mallory, M.L., De Silva, A.O., Muir, D.C.G., Kirk, J.L., et al., 2024. Ultraviolet absorbents and industrial antioxidants in seabirds, mammals, and fish from the Canadian Arctic. Sci Total Environ 951, 175693. https://doi.org/10.1016/j.scitotenv.2024.175693.

[14] United Nations Environment Programme (UNEP). 2023. Recommendation by the persistent organic pollutants review committee to list UV-328 in annex A to the convention and draft text of the proposed amendment. UNEP/POPS/COP.11/14. pp.1–6.

[15] European Chemical Agency., 2025. https://echa.europa.eu. (accessed 8 May 2025).

[16] Castilloux, A.D., Houde, M., Gendron, A., De Silva, A., Soubaneh, Y.D., Lu, Z., 2022. Distribution and fate of ultraviolet absorbents and industrial antioxidants in the St. Lawrence river, quebec, Canada. Environ Sci Technol 56, 5009–5019. https://doi.org/10.1021/acs.est.1c07932.

[17] Zhang, S., Wang, Z., Chen, J., Xie, Q., Zhu, M., Han, W., 2021. Tissue-specific accumulation, biotransformation, and physiologically based toxicokinetic modeling of benzotriazole ultraviolet stabilizers in zebrafish (*danio rerio*). Environ Sci Technol 55, 11874–11884. https://doi.org/10.1021/acs.est.1c02861.

[18] Blouin, K., Malaisé, F., Verreault, J., Lair, S., Lu, Z., 2022. Occurrence and temporal trends of industrial antioxidants and UV absorbents in the endangered St. Lawrence estuary beluga whale (Delphinapterus leucas). Sci Total Environ 842, 156635. https://doi.org/10.1016/j.scitotenv.2022.156635.

[19] Nakata, H., Shinohara, R., Murata, S., Watanabe, M., 2010. Detection of benzotriazole UV stabilizers in the blubber of marine mammals by gas chromatography-high resolution mass spectrometry (GC-HRMS). J Environ Monit 12, 2088–2092. https://doi.org/10.1039/C0EM00170H.

[20] Alonso, M.B., Feo, M.L., Corcellas, C., Gago-Ferrero, P., Bertozzi, C.P., Marigo, J., et al., 2015. Toxic heritage: maternal transfer of pyrethroid insecticides and sunscreen agents in dolphins from Brazil. Environ Pollut 207, 391–402. https://doi.org/10.1016/j.envpol.2015.09.039.

[21] Reiter, E., Escher, I., Rojo-Nieto, B., Nolte, E., Siebert, H., Jahnke, A, U., 2023. Characterizing the marine mammal exposome by iceberg modeling, linking chemical analysis and in vitro bioassays. Environ Sci Process Impacts 25, 1802–1816. https://doi.org/10.1039/D3EM00033H.

[22] Lu, Z., De Silva, A.O., Zhou, W., Tetreault, G.R., de Solla, S.R., Fair, P.A., et al., 2019. Substituted diphenylamine antioxidants and benzotriazole UV stabilizers in blood plasma of fish, turtles, birds and dolphins from North America. Sci Total Environ 647, 182–190. https://doi.org/10.1016/j.scitotenv.2018.07.405.

[23] Schlabach, M., van Bavel, B., Lomba, J.A.B., Borgen, A., Gabrielsen, G.W., Götsch, A., , 2018. Screening Programme 2017-AMAP Assessment Compounds. Norwegian Institute for Air Research (NILU) (report 21/2018).

[24] Pedersen, A.F., Bayen, S., Liu, L., Dietz, R., Sonne, C., Rosing-Asvid, A., et al., 2024. Nontarget and suspect screening reveals the presence of multiple plastic related compounds in polar bear, killer whale, narwhal and long-finned pilot whale blubber from east Greenland. Environ Pollut 357, 124417. https://doi.org/10.1016/j.envpol.2024.124417.

[25] Committee on the Status of Endangered Wildlife in Canada (COSEWIC). 2006. COSEWIC assessment and update status report on the Atlantic walrus Odobenus rosmarus rosmarus in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. ix + 65 pp.

[26] Lu, Z., De Silva, A.O., Peart, T.E., Cook, C.J., Tetreault, G.R., Servos, M.R., et al., 2016. Distribution, partitioning and bioaccumulation of substituted diphenylamine antioxidants and benzotriazole UV stabilizers in an urban creek in Canada. Environ Sci Technol 50, 9089–9097. https://doi.org/10.1021/acs.est.6b01796.

[27] Peng, X., Fan, Y., Jin, J., Xiong, S., Liu, J., Tang, C., 2017. Bioaccumulation and biomagnification of ultraviolet absorbents in marine wildlife of the pearl river estuarine, South China Sea. Environ Pollut 225, 55–65. https://doi.org/10.1016/j.envpol.2017.03.035.

[28] Kim, J.W., Isobe, T., Ramaswamy, B.R., Chang, K.H., Amano, A., Miller, T.M., et al., 2011. Contamination and bioaccumulation of benzotriazole ultraviolet stabilizers in fish from Manila bay, the Philippines using an ultra-fast liquid chromatography–tandem mass spectrometry. Chemosphere 85, 751–758. https://doi.org/10.1016/j.chemosphere.2011.06.054.

[29] Muir, D.C.G., Segstro, M.D., Hobson, K.A., Ford, C.A., Stewart, R.E.A., Olpinski, S., 1995. Can seal eating explain elevated levels of PCBs and organochlorine pesticides in walrus blubber from eastern Hudson Bay (Canada)? Environ Pollut 90, 335–348. https://doi.org/10.1016/0269-7491(95)00019-N.

[30] Scotter, S.E., Tryland, M., Nymo, I.H., Hanssen, L., Harju, M., Lydersen, C., et al., 2019. Contaminants in atlantic walruses in svalbard part 1: relationships between exposure, diet and pathogen prevalence. Environ Pollut 244, 9–18. https://doi.org/10.1016/j.envpol.2018.10.001.

[31] Lu, Z., De Silva, A.O., Spencer, C., Tetreault, G.R., Shane, D., 2024. Distribution and trophodynamics of substituted diphenylamine antioxidants and benzotriazole UV stabilizers in a freshwater ecosystem and the adjacent riparian environment. Environ Sci Process Impacts 26, 1031–1041. https://doi.org/10.1039/D4EM00193A.

[32] Lu, Z., De Silva, A.O., McGoldrick, D.J., Zhou, W., Peart, T.E., Cook, C., et al., 2018. Substituted diphenylamine antioxidants and benzotriazole UV stabilizers in aquatic organisms in the great lakes of North America: terrestrial exposure and biodilution. Environ Sci Technol 52, 1280–1289. https://doi.org/10.1021/acs.est.7b05214.

[33] Wang, X., Hou, X., Hu, Y., Zhou, Q., Liao, C., Jiang, G., 2018. Synthetic phenolic antioxidants and their metabolites in mollusks from the Chinese bohai sea: occurrence, temporal trend, and human exposure. Environ Sci Technol 52, 10124–10133. https://doi.org/10.1021/acs.est.8b03322.

[34] O'Brien, D.J., Kaneene, J.B., Poppenga, R.H., 1993. The use of mammals as sentinels for human exposure to toxic contaminants in the environment. Environ Health Perspect 99, 351–368. https://doi.org/10.1177/0300985810388525.

[35] Insley, S.J., Tauzer, L.M., Halliday, W.D., Illasiak, J., Green, R., Kudlak, A., et al., 2021. Ringed seal diet and body condition in the amundsen gulf region, eastern Beaufort Sea. Arctic 74, 127–138. https://doi.org/10.14430/arctic72447.

[36] Chambellant, M., Stirling, I., Ferguson, S., 2013. Temporal variation in Western Hudson Bay ringed seal phoca hispida diet in relation to environment. Mar Ecol Prog Ser 481, 269–287. https://doi.org/10.3354/meps10134.

[37] Escher, B.I., Ashauer, R., Dyer, S.D., Hermens, M., Lee, S.K., Leslie, H.A., et al., 2010. Crucial role of mechanisms and modes of toxic action for understanding tissue residue toxicity and internal effect concentrations of organic chemicals. Integr Environ Asses Manag 7, 28–49. https://doi.org/10.1002/ieam.100.

[38] Nyman, A.M., Schirmer, K., Ashauer, R., 2014. Importance of toxicokinetics for interspecies variation in sensitivity to chemicals. Environ Sci Technol 48, 5946–5954. https://doi.org/10.1021/es5005126.

[39] Tanabe, S., Iwata, H., Tatsukawa, R., 1994. Global contamination by persistent organochlorines and their ecotoxicological impact on marine mammals. Sci Total Environ 154, 163–177. https://doi.org/10.1016/0048-9697(94)90086-8.

[40] Gebbink, W.A., Sonne, C., Dietz, R., Kirkegaard, M., Born, E.W., Muir, D.C.G., et al., 2007. Target tissue selectivity and burdens of diverse classes of brominated and chlorinated contaminants in polar bears (*ursus maritimus*) from east Greenland. Environ Sci Technol 42, 752–759. https://doi.org/10.1021/es071941f.

[41] Cao, H., Zhou, Z., Wang, L., Liu, G., Sun, Y., Wang, Y., et al., 2019. Screening of potential PFOS alternatives to decrease liver bioaccumulation: experimental and computational approaches. Environ Sci Technol 53, 2811–2819. https://doi.org/10.1021/acs.est.8b05564.

[42] Kinloch, D., Kuhnlein, H., Muir, D.C.G., 1992. Inuit foods and diet: a preliminary assessment of benefits and risks. Sci Total Environ 122, 247–278. https://doi.org/10.1016/0048-9697(92)90249-R.

[43] Brown, T.M., Macdonald, R.W., Muir, D.C.G., Letcher, R.J., 2018. The distribution and trends of persistent organic pollutants and Mercury in marine mammals from

Canada's eastern Arctic. Sci Total Environ 618, 500–517. https://doi.org/10.1016/j.scitotenv.2017.11.052.
[44] Houde, M., Wang, X., Colson, T.L.-L., Gagnon, P., Ferguson, S.H., Ikonomou, M.G., et al., 2019. Trends of persistent organic pollutants in ringed seals (*phoca hispida*) from the Canadian Arctic. Sci Total Environ 665, 1135–1146. https://doi.org/10.1016/j.scitotenv.2019.02.138.
[45] Jardine, A., Cory, Provencher, J.F., Hornby, C., Gamberg, M., Bourdages, M.P.T., et al., 2023. No microplastics detected in the first assessment of atlantic walrus stomachs from nunavut. Canada Arctic Sci 9, 769–780. https://doi.org/10.1139/as-2023-0002.
[46] Stewart, R.E.A., 2008. Redefining walrus stocks in Canada. Arctic 61, 292–308. https://doi.org/10.14430/arctic26.
[47] Helsel, D.R., 2012. Statistics for censored environmental data using minitab and r (second ed). John Wiley & Sons Inc, USA.
[48] Hebert, C.E., Keenleyside, K.A., 1995. To normalize or not to normalize? Fat is the question. Environ Toxicol Chem 14, 801–807. https://doi.org/10.1002/etc.5620140509.
[49] Li, Y., Gillespie, B., Shedden, K., Gillespie, J., 2019. Profile likelihood estimation of the correlation coefficient in the presence of left, right or interval censoring and missing data. R J 10, 159–179. https://doi.org/10.32614/RJ-2018-040.
[50] Fischer, C., Leibold, E., Hiller, J., Göen, T., 2022. Human metabolism and excretion kinetics of benzotriazole UV stabilizer UV-327 after single oral administration. Arch Toxicol 97, 165–176. https://doi.org/10.1007/s00204-022-03401-3.
[51] Denghel, H., Hiller, J., Leibold, E., Göen, T., 2021. Human metabolism and kinetics of the UV absorber 2-(2H-benzotriazol-2-yl)-4,6-di-*tert*-pentylphenol (UV 328) after oral administration. Arch Toxicol 95, 2677–2690. https://doi.org/10.1007/s00204-021-03093-1.
[52] Fischer, C., Hiller, J., Leibold, E., Göen, T., 2024. Toxicokinetics of benzotriazole UV stabilizer UV-P in humans after single oral administration. Arch Toxicol 99, 623–631. https://doi.org/10.1007/s00204-024-03907-y.
[53] Waidyanatha, S., Mutlu, E., Gibbs, S., Pierfelice, J., Smith, J.P., Burback, B., et al., 2021. Phenolic benzotriazoles: a class comparison of toxicokinetics of ultraviolet-light absorbers in Male rats. Xenobiotica 51, 831–841. https://doi.org/10.1080/00498254.2021.1927239.
[54] Stiefel, C., Schwack, W., 2014. Reactivity of cosmetic UV filters towards skin proteins: model studies with Boc-lysine, Boc-Gly-Phe-Gly-Lys-OH, BSA and gelatin. Int J Cosmet Sci 36, 561–570. https://doi.org/10.1111/ics.12157.
[55] Ma, J., Yang, B., Hu, X., Gao, Y., Qin, C., 2023. The binding mechanism of benzophenone-type UV filters and human serum albumin: the role of site, number, and type of functional group substitutions. Environ Pollut 324, 121342. https://doi.org/10.1016/j.envpol.2023.121342.
[56] El Dareer, S.M., Kalin, J.R., Tillery, K.F., Hill, D.L., 1986. Disposition of 2-hydroxy-4-methoxybenzophenone in rats dosed orally, intravenously, or topically. J Toxicol Environ Health 19, 491–502. https://doi.org/10.1080/15287398609530947.
[57] Kadry, A.M., Okereke, C.S., Abdel-Rahman, M.S., Friedman, M.A., Davis, R.A., 1995. Pharmacokinetics of benzophenone-3 after oral exposure in Male rats. J Appl Toxicol 15, 97–102. https://doi.org/10.1002/jat.2550150207.
[58] Okereke, C.S., Abdel-Rhaman, M.S., Friedman, M.A., 1994. Disposition of benzophenone-3 after dermal administration in Male rats. Toxicol Lett 73, 113–122. https://doi.org/10.1016/0378-4274(94)90101-5.
[59] Völkel, W., Colnot, T., Schauer, U.M.D., Broschard, T.H., Dekant, W., 2006. Toxicokinetics and biotransformation of 3-(4-methylbenzylidene)camphor in rats after oral administration. Toxicol Appl Pharm 216, 331–338. https://doi.org/10.1016/j.taap.2006.05.012.
[60] Schauer, U.M.D., Völkel, W., Heusener, A., Colnot, T., Broschard, T.H., von Landenberg, F., et al., 2006. Kinetics of 3-(4-methylbenzylidene)camphor in rats and humans after dermal application. Toxicol Appl Pharm 216, 339–346. https://doi.org/10.1016/j.taap.2006.05.011.
[61] Jepson, P.D., Deaville, R., Barber, J.L., Aguilar, À., Borrell, A., Murphy, S., et al., 2016. PCB pollution continues to impact populations of orcas and other dolphins in european waters. Sci Rep 6. https://doi.org/10.1038/srep18573.
[62] Cui, S., Yu, Y., Zhan, T., Zhang, C., Zhuang, S., 2021. 2,6-Di-*tert*-butylphenol and its quinone metabolite trigger aberrant transcriptional responses in C57BL/6 mice liver. Sci Total Environ 778, 146322. https://doi.org/10.1021/acs.est.1c06866.
[63] Cui, S., Yu, Y., Zhan, T., Gao, Y., Zhang, J., Zhang, L., et al., 2021. Carcinogenic risk of 2,6-di-tert-butylphenol and its quinone metabolite 2,6-DTBQ through their interruption of RARβ: in vivo, in vitro, and in silico investigations. Environ Sci Technol 56, 480–490. https://doi.org/10.1021/acs.est.1c06866.
[64] Rosen, D., 2020. Resting and swimming metabolic rates in juvenile walruses (*odobenus rosmarus*). Mar Mam Sci 37, 162–172. https://doi.org/10.1111/mms.12743.
[65] Fisher, K.I., Stewart, R.E.A., 1997. Summer foods of atlantic walrus, odobenus rosmarus rosmarus, in Northern foxe basin, northwest territories. Can J Zool 75, 1166–1175. https://doi.org/10.1139/z97-139.
[66] Wolkers, J., Witkamp, R.F., Nijmeijer, S.M., Burkow, I.C., de Groene, E.M., Lydersen, C., et al., 1998. Phase I and phase II enzyme activities in ringed seals (*phoca hispida*): characterization of hepatic cytochrome P450 by activity patterns, inhibition studies, mRNA analyses, and Western blotting. Aquat Toxicol 44, 103–115. https://doi.org/10.1016/S0166-445X(98)00063-0.
[67] Peng, X., Zhu, Z., Xiong, S., Fan, Y., Chen, G., Tang, C., 2019. Tissue distribution, growth dilution, and species-specific bioaccumulation of organic ultraviolet absorbents in wildlife freshwater fish in the pearl river catchment, China. Environ Toxicol Chem 39, 343–351. https://doi.org/10.1002/etc.4616.
[68] Moualek, F., Babin, M., Parent, G.J., Ponton, D.E., Senay, C., Amyot, M., et al., 2024. Organic UV absorbents in the deepwater redfish (*sebastes mentella*) from the St. Lawrence estuary and gulf: distribution and human health risk assessment. Sci Total Environ 906, 167515. https://doi.org/10.1016/j.scitotenv.2023.167515.
[69] Braune, B.M., Outridge, P.M., Fisk, A.T., Muir, D.C.G., Helm, P.A., Hobbs, K., et al., 2005. Persistent organic pollutants and Mercury in marine biota of the Canadian Arctic: an overview of spatial and temporal trends. Sci Total Environ 351-352, 4–56. https://doi.org/10.1016/j.scitotenv.2004.10.034.
[70] Statistics Canada. 2023. 2021 Census of Population. Statistics Canada Catalogue no. 98-316-X2021001. Ottawa. ⟨https://www12.statcan.gc.ca/census-recensement/2021/dp-pd/prof/index.cfm?Lang=E⟩ (accessed 10 February, 2025).
[71] Facciola, N., Houde, M., Muir, D.C.G., Ferguson, S.H., McKinney, M.A., 2022. Feeding and contaminant patterns of sub-Arctic and Arctic ringed seals: potential insight into climate change-contaminant interactions. Environ Pollut 313, 120108. https://doi.org/10.1016/j.envpol.2022.120108.
[72] Dietz, R., Born, E.W., Stewart, R.E.A., Heide-Jørgensen, M.P., Stern, H., Rigét, F., et al., 2014. Movements of walruses (*odobenus rosmarus*) between central west Greenland and southeast baffin island 2005-2008. NAMMCO Sci Publ 9, 53–74. https://doi.org/10.7557/3.2605.
[73] North Atlantic Marine Mammal Commission, 2023. Report of the scientific committee working group on ringed seals.
[74] Harwood, L.A., Smith, T.G., Auld, J.C., Melling, H., Yurkowski, D.J., 2015. Seasonal movements and diving of ringed seals, *pusa hispida*, in the Western Canadian Arctic, 1999–2001 and 2010–11. Arctic 68, 193–209.
[75] Provencher, J.F., Malaisé, F., Mallory, M.L., Braune, B.M., Pirie-Dominix, L., Lu, Z., 2022. 44-Year retrospective analysis of ultraviolet absorbents and industrial antioxidants in seabird eggs from the Canadian Arctic (1975–2019). Environ Sci Technol 56, 14562–14573.
[76] Giraudo, M., Colson, T.-L.L., De Silva, A.O., Lu, Z., Gagnon, P., Brown, L., et al., 2020. Food-borne exposure of juvenile rainbow trout (*oncorhynchus mykiss*) to benzotriazole ultraviolet stabilizers alone and in mixture induces specific transcriptional changes. Environ Toxicol Chem 39, 852–862.