Science of the Total Environment 951 (2024) 175693

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Ultraviolet absorbents and industrial antioxidants in seabirds, mammals, and fish from the Canadian Arctic

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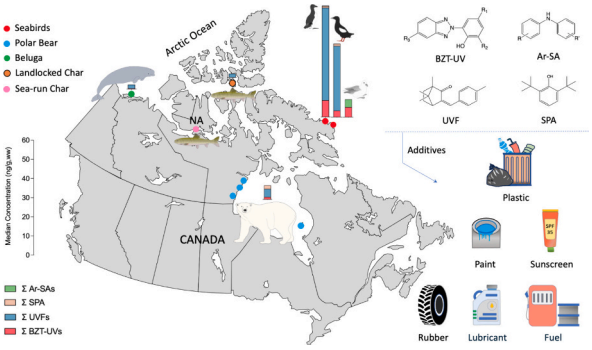
HIGHLIGHTS

- Arctic species accumulated more UV absorbents than industrial antioxidants.
- Seabirds had more UV absorbents and industrial antioxidants than other species.
- First report of SPA and UVFs in the liver of Canadian Arctic species.
- Spatial variations of UV absorbents and industrial antioxidants were found in some species.
- Arctic fish had less UV absorbents and antioxidants than fish in temperate regions.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Lingyan Zhu

Keywords:
Ultraviolet absorbents
Industrial antioxidants
Contaminants of emerging concern
Wildlife
Fish
Canadian Arctic

ABSTRACT

Ultraviolet (UV) absorbents and industrial antioxidants are two groups of plastic-derived contaminants of emerging environmental concern. However, their distribution and fate are poorly understood in Arctic wildlife. In the present study, 16 UV absorbents (10 benzotriazole UV stabilizers (BZT-UVs) and 6 organic UV filters (UVFs)) and 7 industrial antioxidants (6 aromatic secondary amines (Ar-SAs) and 2,6-di-*tert*-butylphenol (26DTBP)) were analyzed in the livers of thick-billed murre (*Uria lomvia*; $n = 28$), northern fulmar (*Fulmarus glacialis*; $n = 4$), black guillemot (*Cepphus grylle*; $n = 11$), polar bear (*Ursus maritimus*; $n = 18$), beluga whale (*Delphinapterus leucas*; $n = 10$), landlocked ($n = 25$) and sea-run ($n = 10$) Arctic char (*Salvelinus alpinus*) from the Canadian Arctic collected between 2017 and 2021. Compared to industrial antioxidants (median range: ΣAr-SAs: not calculated due to detection frequency < 30 % (NA)-4.06 ng/g, wet weight (ww); 26DTBP: NA-1.91 ng/g ww), UV absorbents (median range: ΣBZT-UVs: NA-8.71 ng/g ww; ΣUVFs: NA-48.3 ng/g ww) generally showed greater concentrations in the liver of these species. Seabirds accumulated higher levels of these contaminants

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https://doi.org/10.1016/j.scitotenv.2024.175693
Received 22 May 2024; Received in revised form 30 July 2024; Accepted 20 August 2024
Available online 22 August 2024

(median range: ΣBZT-UVs: 3.38–8.71 ng/g ww; ΣUVFs: NA-48.3 ng/g ww; ΣAr-SAs: 0.07–4.06 ng/g ww; 26DTBP: NA-1.14 ng/g ww)) than the other groups (median range: ΣBZT-UVs: NA-1.31 ng/g ww; ΣUVFs: NA-4.22 ng/g ww; ΣAr-SAs: NA; 26DTBP: NA-1.91 ng/g ww), suggesting that seabirds may be useful indicator species for future long-term monitoring. The livers of Arctic char in the Canadian Arctic generally contain lower levels of these contaminants than those of freshwater fish in temperate regions. Spatial variations were found in the liver of black guillemots, Hudson Bay polar bears, and landlocked char for some target contaminants, indicating differences in the levels of these contaminants in their surrounding environment or diet. Consumption of liver tissues from these species may expose humans to varying levels of UV absorbents and industrial antioxidants. This study establishes a baseline for future research of the spatial and temporal trends of these contaminants in Arctic species. It provides the basis for elucidating the fate of these contaminants and assessing their adverse effects at environmental-relevant concentrations in the Arctic. Factors influencing the accumulation patterns of these contaminants in Arctic biota and their potential health risks require further investigation.

1. Introduction

Ultraviolet (UV) absorbents and industrial antioxidants are additives used in various products to prevent materials degradation and many of these chemicals are produced in high quantities (Castilloux et al., 2022; Wiesinger et al., 2021). Benzotriazole UV stabilizers (BZT-UVs) and organic UV filters (UVFs) are two groups of UV absorbents used in plastics, paints, coatings, and personal care products (Castilloux et al., 2022; Lu et al., 2016). Industrial antioxidants, including synthetic phenolic antioxidants (SPAs) and aromatic secondary amines (Ar-SAs), are used primarily in plastics, rubbers, fuels, lubricants, building materials and electronics (Ji et al., 2023; Liu et al., 2019; Liu and Mabury, 2020). UV absorbents and industrial antioxidants can enter the environment during manufacturing (Cantwell et al., 2015), use (Giokas et al., 2007), and waste disposal processes (Liu et al., 2015; Lu et al., 2017b, 2019b; Ramos et al., 2016) because they are not covalently bonded with the materials to which they are added. In addition, as plastics-derived compounds, some of these contaminants have been detected in plastic debris in the environment (Karlsson et al., 2021; Liu and Mabury, 2021; Rani et al., 2015) and may adsorb or absorb to plastics under environmental conditions (Kapelewska et al., 2021). Thus, exposure of wildlife and humans to plastic-derived contaminants, including UV absorbents and industrial antioxidants, may occur not only through direct uptake *via* the surrounding environment (*e.g.*, air, water, and prey), but also by ingestion of plastic debris (Hamilton et al., 2023).

Some Canadian Arctic species (*e.g.*, beluga whale (*Delphinapterus leucas*) and seabirds) are known to ingest microplastics (Baak et al., 2020; Bourdages et al., 2021; Moore et al., 2020). They have been proposed as vectors of plastic-derived contaminants in the environment (Hamilton et al., 2023; Kühn et al., 2020; Tanaka et al., 2019). Therefore, those individuals that ingest or accumulate more microplastics may have higher levels of plastic-derived contaminants in their tissues (Sühring et al., 2022). However, our current understanding of the distribution, fate, and impacts of plastic-derived contaminants in the Arctic is limited to certain types of plasticizers and flame retardants (*e.g.*, organophosphate esters (OPEs) and polybrominated diphenyl ethers (PBDEs)). Data for other types of additives, such as UV stabilizers and antioxidants, is scarce. These knowledge gaps impede the risk assessment of plastic pollution in this region and the development of conservation strategies that will reduce the risk of these contaminants to biodiversity.

Some of UV absorbents and industrial antioxidants may result in multiple negative health impacts in exposed organisms, such as hepatic toxicity, endocrine disruption, and carcinogenicity, as demonstrated in several laboratory *in vitro* and *in vivo* exposure studies (Fent et al., 2014; Liang et al., 2017; Nagayoshi et al., 2015; Kim and Choi, 2014; Cui et al., 2021a, 2021b). However, the ecological risks posed by these contaminants are uncertain, partially because of insufficient data on their concentrations in the tissues of wildlife and surrounding environments. Therefore, their distribution, fate, and environmental impacts are of emerging scientific and public concern. For example, the Stockholm Convention on Persistent Organic Pollutants (POPs) has recommended the phase-out of 2-(2H-benzotriazol-2-yl)-4,6-di-*tert*-pentylphenol (UV328), a BZT-UV, by 2026 and banned globally by 2044 (United Nations Environment Programme UNEP, 2023). The European Chemical Agency (https://echa.europa.eu) has also classified UV328, 2-benzotriazole-2-yl-4,6-di-*tert*-butylphenol (UV320), 2,4-di-*tert*-butyl-6-(5-chloro-2H-benzotriazol-2-yl) phenol (UV327), and 2-(2H-benzotriazol-2-yl)-4-(*tert*-butyl)-6-(sec-butyl)phenol (UV350) as Substances of Very High Concern in the environment (Castilloux et al., 2022).

Some of these additives have been detected in previous studies on seabirds (eggs and livers) and ringed seals (*Pusa hispida*) (liver) in the Canadian Arctic (Lu et al., 2019a; Provencher et al., 2022), in preen gland oil of seabirds from the islands of Bering Sea (Yamashita et al., 2021), as well as in seabird eggs, polar bear (*Ursus maritimus*) blood, American mink (*Neovison vison*) liver, and landlocked Arctic char (*Salvelinus alpinus*) muscle in the Norwegian Arctic (Lucia et al., 2016; Schlabach et al., 2018). These findings demonstrate the presence of these plastic-derived contaminants in the Arctic food web. However, the spatial and species coverage has been very limited, and the distribution in key species used as important food sources for local communities is unknown. In addition, different tissues were used for previous studies, making spatial and species comparisons difficult. Furthermore, there is currently no information on the occurrence and distribution of UV absorbents and industrial antioxidants in other mammals and fish from the Canadian Arctic.

This study aimed to investigate species differences in the concentrations and compositions of UV absorbents and industrial antioxidants in Arctic seabirds, mammals, and fish livers. The liver was selected for analysis because previous studies have reported greater accumulation of BZT-UVs in the livers of fish and seabirds than other analyzed tissues (Lu et al., 2017a; Peng et al., 2020; Tanaka et al., 2020; Tang et al., 2019). Some of the analyzed species are consumed as a traditional food (*e.g.*, Arctic char and beluga) by Indigenous communities. Although the liver of many of the species we look at is not consumed as part of the traditional diet, the data obtained from the liver may be useful as an indicator of contamination in these species. Additionally, we aimed to compare spatial differences for some of these species and examine the variation of these contaminants between different life history types of Arctic char (anadromous *vs.* landlocked). Understanding the occurrence and distribution of UV absorbents and industrial antioxidants in Arctic organisms is the first step in elucidating the fate (*e.g.*, trophodynamics), temporal and spatial trends, as well as potential ecological risks of these contaminants in polar regions. It will also provide the basis for further toxicity analysis in different species at environmentally relevant concentrations.

2. Materials and methods

2.1. Sample collection

The information on studied species, sampling locations, and sample sizes are summarized in Table 1 and Fig. 1. All sampling was conducted under territorial and/or federal permits.

Table 1
Summary of species, location, sampling year and size of analyzed Arctic fish and wildlife livers.

Species			Location		Year	*n*
Seabirds	Thick-billed Murre	*Uria lomvia*	Nunavut	Qaqulluit	2018	28
	Northern Fulmar	*Fulmarus glacialis*		Qaqulluit	2018	4
	Black Guillemot	*Cepphus grylle*		Qaqulluit	2018	5
				Akpait	2018	6
Mammals	Polar Bear	*Ursus maritimus*		Sanikiluaq	2020	10
				Southern Hudson Bay	2019–2020	2
				Western Hudson Bay	Whale Cove	4
					Rankin Inlet	2
					Arviat	10
					2017–2018	
Fish	Beluga	*Delphinapterus leucas*	Northwest Territories	Hendrickson Island		5
	Landlocked Char	*Salvelinus alpinus*	Nunavut	Cornwallis Island		
				Small Lake	2021	5
				North Lake	2021	5
				Resolute Lake	2021	5
				Amituk Lake	2021	5
				Meretta Lake	2021	5
	Sea-run Char		Nunavut	Victoria Island		
				Halokvik River	2019	10
Total						106

2.1.1. Seabirds

Thick-billed murres (*Uria lomvia*) ($n = 28$), black guillemots (*Cepphus grylle*) ($n = 5$), and northern fulmars ($n = 4$) (hereafter murres, guillemots and fulmars, respectively) were collected in collaboration with local Inuit hunters in July 2018 at Qaqulluit (also known as Cape Searle) National Wildlife Area (Nunavut), while guillemots ($n = 6$) were also collected from Akpait (also known as Reid Bay) National Wildlife Area (Nunavut). Details on the collection can be found in Baak et al. (2020). Liver samples (28 murres and 4 fulmars) were matched to some individual birds analyzed for microplastics as described previously (Bourdages et al., 2021).

2.1.2. Mammals

The harvest and sampling of polar bears (all males) were carried out by local hunters and the Government of Nunavut (Igloolik, Nunavut) from southern ($n = 10$; 8 adult and 2 sub-adult) and western ($n = 8$; 3 adult and 5 sub-adult) Hudson Bay in 2019–2020 as detailed elsewhere, including the details of sample processing for contaminant analysis (Letcher et al., 2018). Beluga liver samples ($n = 10$) were obtained from Hendrickson Island, near Tuktoyaktuk, Northwest Territories (more details are shown in the Supplementary Data) (Moore et al., 2020). Seven liver samples were matched to the individual beluga whales analyzed for microplastics described in Moore et al. (2020).

2.1.3. Fish

Landlocked Arctic char samples were collected from 5 lakes on Cornwallis Island ($n = 25$) (Nunavut), including Small Lake, North Lake, Resolute Lake, Amituk Lake, and Meretta Lake ($n = 5$ for each lake) using the methods described in Hudelson et al. (2019). Biological parameters (length, fork length, weight, $\delta^{13}C$, $\delta^{15}N$, age and condition factor (weight/length^3 (hectogram/cm^3) of landlocked char are listed in Table S1. Sea-run char samples were collected by local partners during the commercial fish season at Halokvik River ($n = 10$) on Victoria Island (Nunavut).

All tissues were collected without contacting plastics and stored under <-20 °C until further analysis. Samples were dissected with solvent-cleaned tools and stored in solvent-cleaned glass jars.

2.2. Chemicals and reagents

Target contaminants included 16 UV absorbents (10 BZT-UVs and 6 UVFs) and 7 industrial antioxidants (1 SPA and 6 Ar-SAs). Detailed information on the target contaminants, including the CAS numbers, molecular weights, and log K_{ow}, are shown in Table 2. The structures of the target contaminants are shown in Fig. S1. The suppliers of the chemicals and reagents used in the analyses are presented in the Supplementary Data.

2.3. Sample preparation and instrumental analysis

The sample preparation and instrumental analysis procedures were adapted from previously reported methods (Blouin et al., 2022; Castilloux et al., 2022). Details are described in the Supplementary Data and Table S2. Briefly, liver samples (1–2 g, wet weight (ww)) were mixed with Na_2SO_4 and extracted using 7 mL 1:1 (*v:v*) hexane/dichloromethane in an ultrasonic bath three times. The extracts were combined, concentrated, and purified using gel permeation chromatography. Final samples were analyzed using gas chromatography–mass spectrometry (GC–MS) in single-ion monitoring mode.

2.4. QA/QC

Glass materials were used in the experiment whenever possible to avoid contact with plastics, and they were rinsed with acetone (free of detectable target contaminants) before use. Aluminum foil (450 °C overnight) was used to cover the top of test tubes, beakers, and flasks to

prevent samples, extracts, and solvents from contact with plastic caps or air particles. One procedure blank and one spike-recovery sample (spike 20 ng of target contaminants in each matrix) were prepared for each experiment batch (10 to 15 samples). Target contaminants were not detected in procedure blanks. As quality controls for GC–MS analysis, one standard (20 ng/mL) and two *n*-hexane solvent blanks were evaluated for every five samples during the GC–MS analysis. Calibration curves contained nine concentrations of standards (0, 0.1, 0.5, 1, 5, 10, 20, 50, and 100 ng/mL). Some samples with target contaminants that exceeded the concentration range of the calibration curve were diluted and reanalyzed. The limit of detection (LOD) and limit of quantification (LOQ) was defined as the concentration of target analyte in the matrix that generated a peak with a signal-to-noise (S/N) ratio of 3 and 10, respectively. Recoveries ranged from 88 ± 32 % to 107 ± 17 % (mean ± SD) at the spike level of 20 ng (Table S3). LOD and LOQ were in the ranges of 0.02–9.0 ng/g ww and 0.1–30 ng/g ww, respectively (Table S3).

2.5. Data analysis

GraphPad Prism 10.0 (La Jolla, CA) and R 4.1.2 (with RStudio v2023.09.0) were used to analyze the data. The robust regression on order (ROS) method (Helsel, 2012) in the NDexpo (https://expostats.ca) was used to estimate descriptive statistics of the data with censored values of ≤70 % when sample size >7. When the sample size was 5 or 6, the ROS method was used if the detection frequency was >50 %, as this method requires three detections. For fulmar ($n = 4$), half LOD was used to replace non-detects for estimating descriptive statistics when detection frequency ≥ 50 %. Concentration measurements were reported as wet weight (ww) based because the concentration data do not have correlations with lipid content (Hebert and Keenleyside, 1995). The data were logarithmically transformed to approximate normal distributions based on the Shapiro-Wilk test before comparisons. Due to the absence of equal variances in the data (Brown-Forsythe test), Welch Analysis of Variance (ANOVA) followed by Dunnett's T3 multiple comparisons were used to compare the levels of contaminants among more than three groups. Using the unpaired *t*-test with Welch's correction, the differences in the concentrations of contaminants between two groups were compared based on log-transformed data. Although the sex information for fulmars and guillemot was available (Baak et al., 2020), concentrations of target contaminants were not compared between males and females because small sample size at each sampling site. The maximum likelihood approach in R (v2022.07.2; clikcorr package, v1.0) (Li et al., 2018), which tests correlations with censored data, was used to examine the relationships between logarithmically transformed UV234 concentrations in the liver and previously reported ingested microplastics in the stomach, intestine, and entire gastrointestinal tract in 7 matched belugas. Relationships between ΣBZT-UVs or ΣUVFs and various biological variables (length, fork length, weight, $\delta^{13}C$, $\delta^{15}N$, age and condition factor) of landlocked char were tested using the Data Analysis for Censored Environmental Data (NADA2) package (v1.1.3) in R (maximum likelihood estimation). The statistically significant level was set as $p < 0.05$.

3. Results and discussion

3.1. Benzotriazole UV stabilizers (BZT-UVs)

Considerable variation of concentrations and compositions of BZT-UVs in the liver samples were found among the Arctic species. Each target BZT-UV, except for UV326, was detectable in the liver of at least two species of the analyzed Arctic wildlife (Table S4). UVP, UV234, UV328, and UV329 showed relatively higher detection frequencies than other congeners (Table S4). Seabirds from Qaqulluit National Wildlife

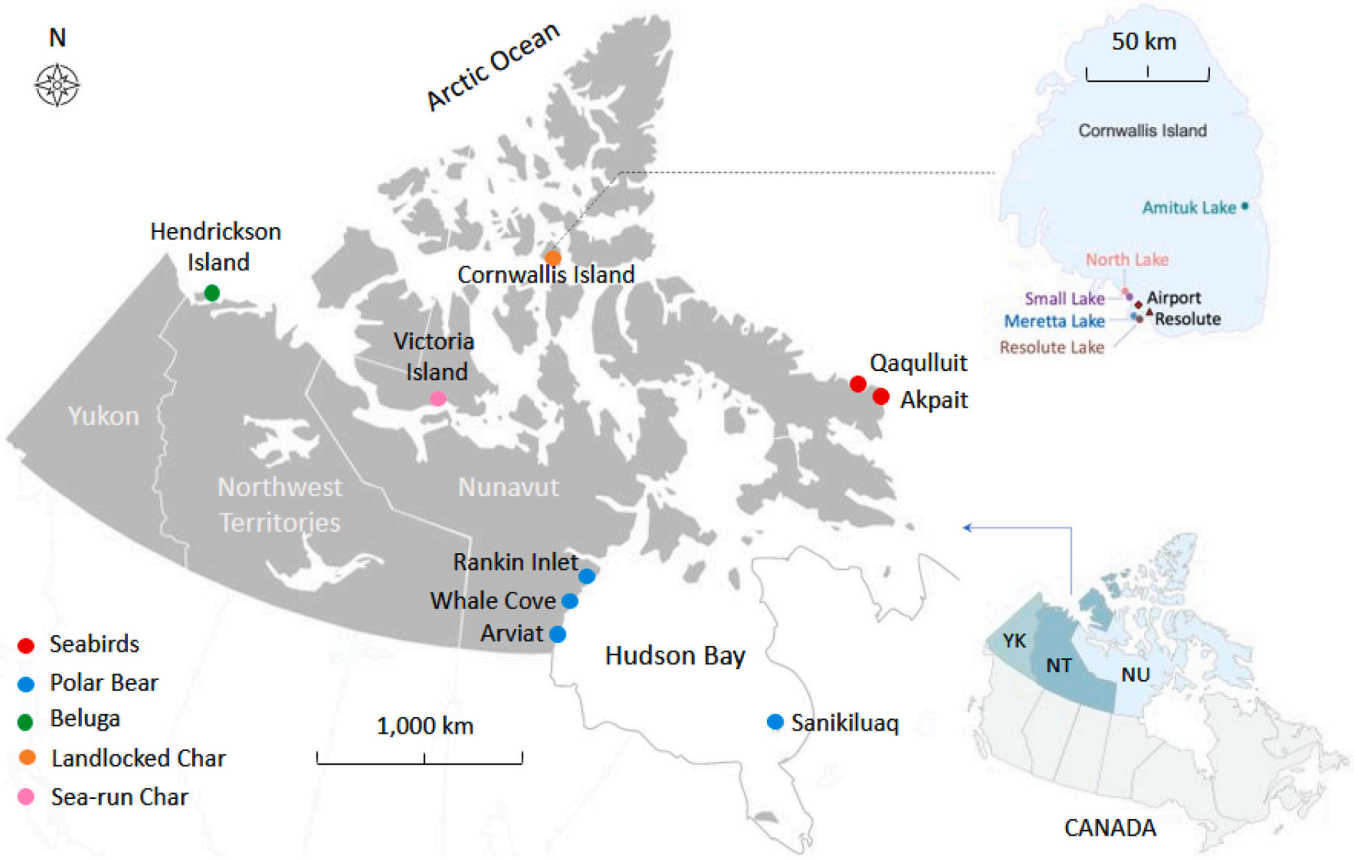

Fig. 1. Sampling locations of seabirds, marine mammals, and fish in the Canadian Arctic. YK: Yukon; NT: Northwest Territories; NU: Nunavut.

Table 2
Names, CAS registry numbers, acronyms, and physicochemical properties of target contaminants. Log *K*ow was estimated using the Estimation Program Interface (EPI) Suite V 4.11.

Chemical Group	Chemical Name	CAS No.	Acronyms	Molecular weight ($g\ mol^{-1}$)	Log K_{ow} (25 °C)
UV Absorbents					
BZT-UVs	2-(2H-benzotriazol-2-yl)-p-cresol	2440-22-4	UVP	225.3	3.0
	2-[3-(2H-benzotriazol-2-yl)-4-hydroxyphenyl]ethyl methacrylate	96,478–09-0	UV090	323.4	3.9
	2-(2H-benzotriazol-2-yl)-4-methyl-6-(2-propenyl)phenol	2170-39-0	UV9	265.3	4.4
	2-(2H-benzotriazol-2-yl)-4,6-bis(1-methyl-1-phenylethyl)phenol	70,321–86-7	UV234	447.6	7.7
	2-Benzotriazole-2-yl-4,6-di-*tert*-butylphenol	3846-71-7	UV320	323.4	6.3
	2-*tert*-Butyl-6-(5-chloro-2H-benzotriazol-2-yl)-4-methylphenol	3896-11-5	UV326	315.8	5.6
	2,4-Di-*tert*-butyl-6-(5-chloro-2H-benzotriazol-2-yl) phenol	3864-99-1	UV327	357.9	6.9
	2-(2H-benzotriazol-2-yl)-4,6-di-*tert*-pentylphenol	25,973–55-1	UV328	351.5	7.3
	2-(2H-benzotriazol-2-yl)-4-(1,1,3,3-tetramethyl butyl)phenol	3147-75-9	UV329	323.4	6.2
	2-(2H-benzotriazol-2-yl)-4-(*tert*-butyl)-6-(*sec*-butyl)phenol	36,437–37-3	UV350	323.4	6.3
UVFs	Benzophenone	119–61-9	BP	182.2	3.2
	2-Hydroxy-4-methoxybenzophenone	131–57-7	BP3	228.3	3.5
	Ethylhexyl methoxycinnamate	5466-77-3	EHMC	290.4	5.8
	4-Methylbenzylidene camphor	36,861–47-9	4MBC	254.4	5.9
	2-Ethylhexyl salicylate	118–60-5	EHS	250.3	6.0
	3,3,5-Trimethylcyclohexyl salicylate	118–56-9	HMS	262.4	6.2
Industrial Antioxidants					
SPA	2,6-Di-*tert*-butylphenol	128–39-2	26DTBP	206.3	4.5
Ar-SAs	Diphenylamine	122–39-4	DPA	169.2	3.3
	N-Phenyl-1-naphthylamine	90–30-2	AOA	219.3	4.5
	N-Phenyl-2-naphthylamine	135–88-6	AOD	219.3	4.5
	Bis(4-*tert*-butylphenyl)amine	4627-22-9	C4C4	281.4	7.1
	Bis[4-(2-phenyl-2-propyl)phenyl]amine	10,081–67-1	diAMS	405.6	8.5
	Bis(4-(2,4,4-trimethylpenta-2-yl)phenyl)amine	15,721–78-5	C8C8	393.7	10.8

Area (Nunavut) generally showed higher concentrations of ΣBZT-UVs in the liver (median: murre 8.71 ng/g ww; fulmar 5.25 ng/g ww; guillemot 3.38 ng/g ww) than was detected in mammals (median: polar bear 1.31 ng/g ww; beluga 0.44 ng/g ww) and fish (landlocked char 0.15 ng/g ww; sea-run char: not calculated due to detection frequency < 30 % (NA)) from various locations across Canadian Arctic (Fig. 2), suggesting greater exposure to and/or accumulation of BZT-UVs in seabirds from these colonies than other studied species.

The absence of detection of UV326 is consistent with previous findings of the low or no detection of this contaminant in organisms, surface water, wastewater, and sediment in Canada, including those samples from the temperate zone (Castilloux et al., 2022; Lu et al., 2016; 2017ab; 2019a). Peng et al. (2017) reported that UV326 was ubiquitous in the water and sediment of the Pearl River Estuary (South China Sea), but only found in 33 % of food web samples. In seabirds, UV326 showed about 5–10 times lower concentrations compared to UV328 and UV327 in the liver of streaked shearwater (*Calonectris leucomelas*) when feeding with plastics containing BZT-UVs (Tanaka et al., 2020). These findings suggest that UV326 may have limited bioaccumulation and biomagnification potential in the aquatic environment. Indeed, both *in vivo* exposure study (Zhang et al., 2021) and modeling results (Provencher et al., 2022) demonstrated the faster elimination of UV326 (half-life in zebrafish (*Danio rerio*) liver: 2.6–3.2 days; modelled half-life in fish 2.8 days) in fish compared to UV328 (half-life in zebrafish liver: 6.4–7.4 days; modelled half-life in fish 14.3 days), UV234 (half-life in zebrafish liver: 6.8–7.5 days; modelled half-life in fish 12.6 days), and UV329 (half-life in zebrafish liver: 3.6–8.8 days; modelled half-life in fish 8.0 days), which were more frequently detected in the present study. This indicates a possible rapid biotransformation of UV326, even if it is taken up by organisms. Furthermore, the modeling results showed UV326 (1377 km) has a lower potential for long-range transport than UV328 (2801 km), UV234 (2838 km), and UV329 (2315 km) (Lu et al., 2017b), which may also contribute to the no detection of this contaminant in the present study.

3.1.1. Seabirds

UV234 was a prominent BZT-UV in seabird livers in general (median: 2.18 ng/g ww in murre; 3.72 ng/g ww in fulmar; 0.80 ng/g ww in guillemot), whereas UV328 was more significant in murre livers (median: 0.59 ng/g ww in murre; NA in fulmar and guillemot) (Table S4 and Fig. S2). We note that the livers of murres were more contaminated by UV328, than other species examined in this study (Table S4 and Fig. S2). The frequent detection of UV234 and UV328 in seabird livers in the present study is consistent with the results found for the preen gland oil of seabirds from the sub-Arctic St. Lawrence Island and Pribilof Island at the Bering Sea (various species, including murre) (Yamashita et al., 2021). At the Pribilof Island, UV328 was the dominant BZT-UV in the preen gland oil of murres, which was not detectable in the red-legged kittiwake (*Rissa brevirostris*) from the same island (Yamashita et al., 2021). These results suggest that BZT-UVs are accumulated in a species-specific manner, despite the seabirds occupying similar habitats.

In the present study, the concentrations of UV328 in murre livers were as high as 423 ng/g ww (median: 0.59 ng/g ww), and 25 % (7/28) of murre samples had UV328 levels above 45 ng/g ww. In addition, the murre livers tended to accumulate more UVP than other studied Arctic species (median 0.54 ng/g ww; range < 0.3–18.5 ng/g ww) (Table S4 and Fig. S2). These data suggested that certain murres in this population may have been exposed to high levels of UV328 and UVP. Braune et al. (2014) reported that murres from Akpait, the nearby breeding colony on Baffin Island, which is the source of the murres collected near Qikiqtarjuaq, accumulated higher levels of PBDEs than other colonies outside the island in the Canadian Arctic. This difference was attributed to the different overwinter migration routes of murres from Akpait (farther south over the eastern Grand Banks and southern Labrador Sea) compared to other murre colonies (Braune et al., 2014; Gaston et al., 2011; McFarlane Tranquilla et al., 2013). Thus, this factor may affect the relatively high levels of UV328 and UVP accumulation in this murre population.

Another factor that needs to be considered is microplastic ingestion. Among the murres analyzed in the present study ($n = 28$), microplastics (1–4 particles per sample; shape included fiber, fragment, and foam)

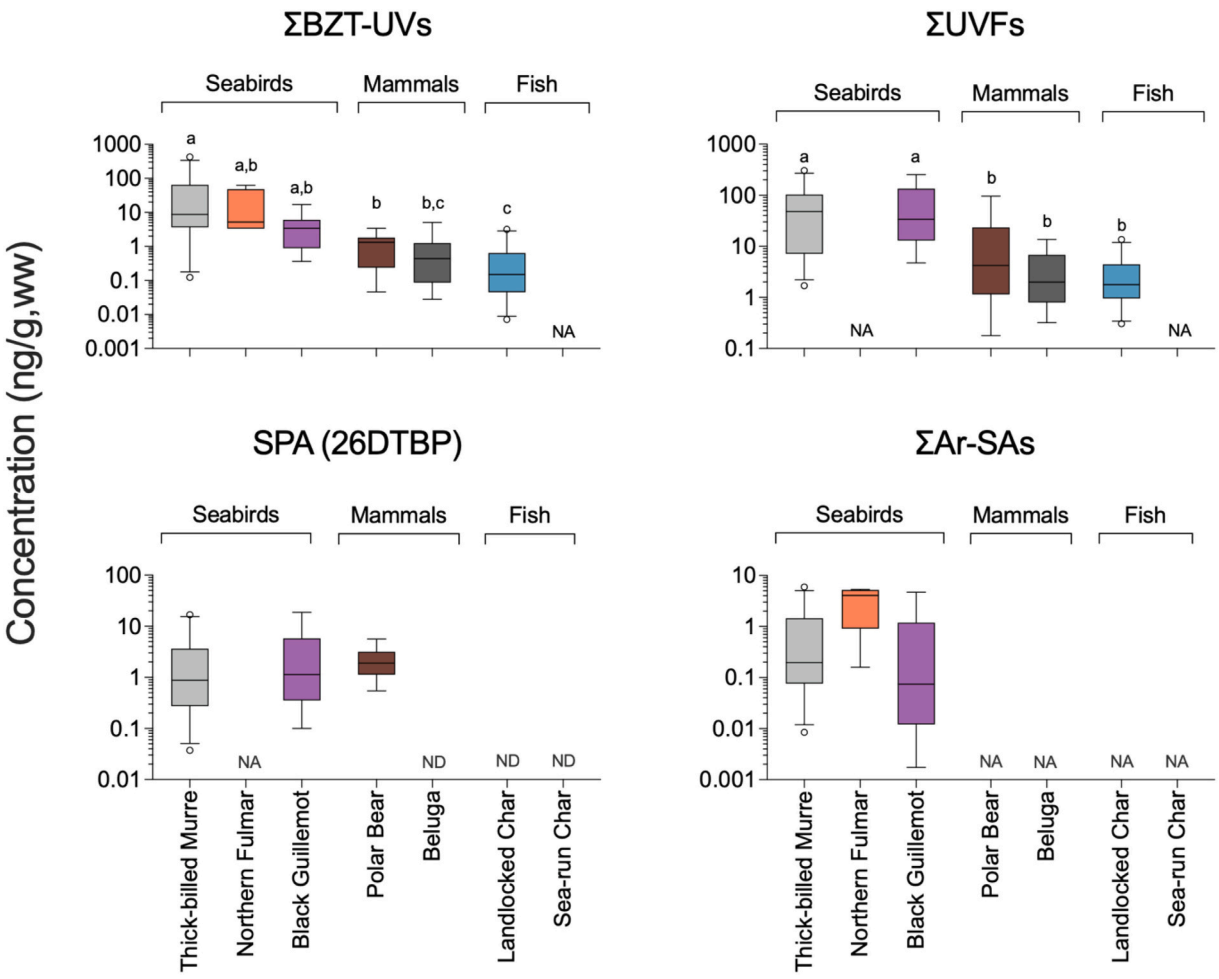

Fig. 2. Concentrations of ΣBZT-UVs, ΣUVFs, SPA (26DTBP), and ΣAr-SAs (see Table 2 for the full chemical names of the abbreviations) in the Canadian Arctic biota livers. Box plots are defined as follows: center line, median; box plot edges, 25th and 75th percentile; whiskers, 5th and 95th percentile of the distribution. NA: detection frequency <50% for fulmar or < 30% for other species; ND: not detected. Different letters indicate significant differences. The circles represent outliers.

were found in the faecal precursors of 5 individuals (Table S5) (Bourdages et al., 2021). Interestingly, the detection frequency and concentrations of UVP (80 % *vs.* 26 %; median 2.58 ng/g ww *vs.* NA) and UV328 (80 % *vs* 65 %; median 21.1 ng/g ww *vs.* 0.33 ng/g ww; $p = 0.0097$) was higher in the livers of the murres that detected microplastics in the faecal precursor than in those that did not, while the detection frequency of these BZT-UVs increased to 100 % in the 3 individuals that had detectable plastic fibers (Table S5). However, there was no clear pattern of how polymer types may be related to the detection of these BZT-UVs due to the large variation or unidentified polymer types. To our knowledge, although UV326 is known to transfer from marine plastic debris to the stomach oil of fulmars and UV328 can be transferred from plastics to the liver, abdominal adipose and preen gland oil of streaked shearwater (Kühn et al., 2020; Tanaka et al., 2020), there is currently no data on the leaching of UVP from microplastics into seabird tissues.

Sex differences were found in the concentrations of BZT-UVs in the livers of murres (Fig. S3). Female murres had higher levels of UV328 in the liver (median: females 7.72 ng/g ww; males 0.15 ng/g ww), whereas males accumulated more UVP (median: males 0.73 ng/g ww; NA in females) (Fig. S3). There was no significant difference between male and female murres for UV234 concentrations (median: 2.45 ng/g ww and 0.80 ng/g ww, respectively) (Fig. S3; $p = 0.12$). Contaminant accumulations have shown sex differences in many avian species (Robinson et al., 2012). During the breeding season, female birds may exhibit decreased contaminants in their tissues as these substances are depurated to eggs (Robinson et al., 2012). This phenomenon may account for the lower UVP (and other detectable UVF (4MBC) and antioxidants (diAMS and 26DTBP), as discussed below) concentration in female murre liver. UV234 and UV328 did not follow this pattern, indicating that these two BZT-UVs may be less transferrable to eggs, or the birds are continually exposed even after egg formation. In a previous study, female murres from the Northwater Polynya accumulated more *trans*-nonachlor and MC7 ($C_{10}H_6Cl_8$) in the liver than males (Fisk et al., 2001), similar to UV328. These discrepancies could be attributed to male and female murres' differing foraging behaviors (*e.g.*, females have significantly more foraging trips than males and forage in patches of higher quality) during the breeding season (Mallory et al., 2019; Paredes et al., 2006). In addition, the UV234 result is consistent with most organohalogen pollutants, which did not show significant sex differences in murres (Braune et al., 2014). Differences in metabolism between male and female murres, if present, may also influence the observed patterns. However, no information is available on sex differences in metabolism in murres.

UV234, but not UV328, was previously found in the livers of fulmars collected in the southern Labrador Sea in 2015 (Lu et al., 2019a) with a lower detection frequency and concentrations (10 %; <0.2–0.6 ng/g ww) than the fulmars in the current study (50 %; <0.5–27.8 ng/g ww) (Table S4). In comparison to the present study (25 %; <0.1–2.20 ng/g) (Table S4), a similar contamination pattern of UV328 was detected in the liver of fulmars obtained in 2013 from Prince Leopold Island (Nunavut; a high Arctic site) (11 %; <1.0–3.8 ng/g ww), however,

UV234 was not found in the 2013 samples from Prince Leopold Island (Lu et al., 2019a). Due to the limited sample size ($n = 4$) of fulmar livers in the present study, these comparisons should be interpreted cautiously. It has been reported that fulmars from different colonies in the Arctic had varying prey composition (Mallory et al., 2010), which may affect the accumulation of contaminants *via* trophic transfer. For example, fulmars collected near Qaqulluit, which is the present study's sampling site, had significantly more invertebrates (*e.g.*, copepods 62 items per bird) and fewer fish (0.3 items per bird) in their diet than fulmars collected from Prince Leopold Island (copepods 0.1 item per bird; fish 2.2 item per bird) (Mallory et al., 2010). In addition, although fulmars are known to ingest more microplastics than other seabird species in the Canadian Arctic (Baak et al., 2020), the relationships between microplastics and target contaminants in the present study were not analyzed due to the small sample size ($n = 4$).

3.1.2. Mammals

Polar bears (median: 1.31 ng/g ww; range < LOD-3.41 ng/g ww) and belugas (median: 0.44 ng/g ww; range < LOD-5.10 ng/g ww) had similar concentrations of ΣBZT-UVs in their livers, but the compositions were different (Fig. 2 and Table S4). UV234 (60 %; median 0.36 ng/g ww; range < 0.2–0.62 ng/g) and UV329 (67 %; median 1.27 ng/g ww; range < 0.3–3.14 ng/g ww) were the most prevalent BZT-UV in the liver of beluga and polar bears, respectively (Table S4 and Fig. S2). In a previous study (Blouin et al., 2022), UV234 was not detected in the liver of belugas from the St. Lawrence Estuary (Canada) population, which may be related to the higher detection limit of UV234 (1.6 ng/g ww) compared to the present study (0.2 ng/g ww; concentration detected: <0.2 ng/g-0.62 ng/g ww) (Table S4). For 7 of the beluga samples analyzed in this study, the presence and characteristics of microplastics ≥ 20 μm were examined and detected (45 % > 1 mm) (Table S6) (Moore et al., 2020). No correlation was found between the concentrations of the most commonly detected UV234 and microplastic (≥ 20 μm) abundances (based on total microplastics or different polymer types) in the stomach, intestine, and total gastrointestinal tract of 7 paired beluga samples. This does not necessarily mean that plastic ingestion is not a factor influencing the bioaccumulation of UV234 in these belugas, as the available plastic ingestion data were limited to particle counts with small sample sizes and specific size ranges, and the mass of plastics detected was unknown (Moore et al., 2020). Larger sample sizes and standardized plastic contamination evaluation methods are needed to elucidate better the relationships between plastic ingestion and the accumulation of plastic-associated contaminants in wildlife.

For polar bears, elevated levels of UV329 were found in southern Hudson Bay samples compared to western Hudson Bay samples (median 1.52 ng/g ww and 0.47 ng/g ww, respectively) (Fig. 3). This pattern is similar to the previously observed distribution of PBDEs and *per*- and polyfluorinated substances (PFAS), which may be attributable to the closer proximity of southern Hudson Bay to more human activities and potential sources (Letcher et al., 2018). Different diet composition is another possible factor affecting the accumulation of contaminants, as polar bears from western Hudson Bay tended to have more prey items foraging near or within freshwater than those of southern Hudson Bay (McKinney et al., 2011b). However, the occurrence and fate of BZT-UVs in the prey of these polar bear populations are unknown. For the samples from western Hudson Bay, adult polar bears showed more frequent detection of UV329 than subadults (100 % *vs* 20 %; median 0.86 ng/g ww *vs.* NA), suggesting that older individuals from this population may have higher levels of UV329 contamination. In contrast, this pattern was not found for the southern Hudson Bay polar bears. UV329 was previously detected in 10 % of polar bear blood plasma samples from the northeastern part of Svalbard in the Norwegian Arctic (< 0.6–2.3 ng/g ww) (Schlabach et al., 2018). This concentration range is similar to the results of Canadian Arctic polar bear liver samples of the present study (< 0.3–3.14 ng/g ww) (Table S4).

3.1.3. Fish

Landlocked char (median: 0.15 ng/g ww; range: <LOD-3.17 ng/g ww) showed higher detection frequencies and concentrations of ∑BZT-UVs than sea-run char (median: NA; range: <LOD-0.78 ng/g ww)

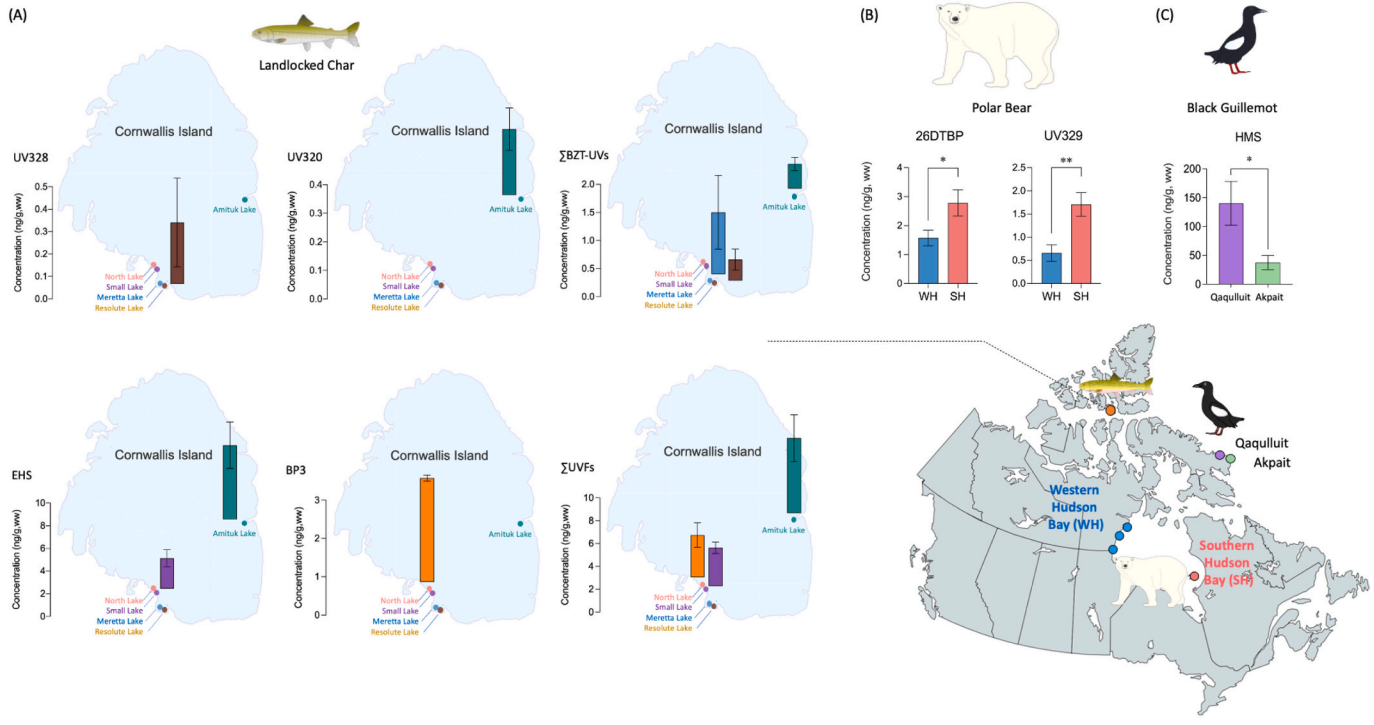

Fig. 3. Spatial variation (mean ± SE) of the most frequently (> 50 %) detected target contaminants in the liver of (A) landlocked char from North Lake, Small Lake, Meretta Lake, Resolute Lake, and Amituk Lake of the Cornwallis Island (Nunavut), (B) polar bear from Western Hudson (WH) and Southern Hudson (SH) Bay of Nunavut (adults and sub-adults are considered together), and (C) black guillemot from Qaqulluit and Akpait of Nunavut. * $p < 0.05$; ** $p < 0.01$.

(Table S4 and Fig. 2), suggesting that freshwater fish in the Arctic accumulate more BZT-UVs than marine-accessible conspecifics. This result is consistent with previous research comparing other contaminants, such as mercury and PFAS, between the two types of Arctic char (Muir et al., 2019; Swanson et al., 2011). Possible explanations include fewer BZT-UV sources near sea-run char sampling regions, greater marine water dilution effects, and faster growth rates of sea-run compared to landlocked Arctic char. Four of the 5 landlocked lakes investigated in the present study were close (3–12 km away) to the nearby community of Resolute Bay (population approximately 150), and one of them (Meretta Lake) is known to have received wastewater from the airport and former military base until the late 1990s (Antoniades et al., 2011), which has resulted in higher levels of PFAS and polychlorinated biphenyls in fish and catchment soils of lakes near the airport (Cabrerizo et al., 2018a, 2018b; Lescord et al., 2015). The lakes also receive plastic debris from the solid waste handling site of the community. Thus, the surrounding environment of landlocked char may be more contaminated than that of the sea-run char. Although no data on the distribution of BZT-UVs in the water of the sampling areas are available, previous research found that the concentrations of PFAS and OPEs in the water of the lakes on Cornwallis Island were higher than in the seawater in nearby Barrow Strait and also in Dease Strait close to the sea-run char sampling area (*i.e.*, near Victoria Island) (Benskin et al., 2012; Lescord et al., 2015; McDonough et al., 2018; Stock et al., 2007). Another possible reason is that the growth rate of landlocked char is slower than that of sea-run char (Muir et al., 2019), which could lead to higher levels of UV absorbents in the tissues of landlocked char due to the growth dilution effect of some of these contaminants in fish (Peng et al., 2020). However, no correlation was found between ΣBZT-UVs and any biological variables (including the length, weight and age) when considering all landlocked char samples from the 5 lakes on Cornwallis Island. In addition, landlocked char livers from Cornwallis Island had a higher detection frequency of UV320 (20 %), UV327 (12 %), and UV328 (36 %) (Table S4) than the muscle of conspecifics from Lake Erlingvatnet and Lake Ellasjøen in Svalbard in the Norwegian Arctic (not detected), whereas UV326 and UV329 were not detectable in the landlocked char from both studies (Lucia et al., 2016).

UV328 was the most abundant BZT-UV in landlocked char livers, with a detection frequency of 36 % and concentrations in the range of <0.03–2.17 ng/g. This concentration is lower compared to liver samples of other fish species from temperate locations in Canada or worldwide. For example, UV328 levels in the liver of white sucker (*Catostomus commersonii*) from an urban creek in Canada and northern pike (*Esox lucius*) in the St. Lawrence River (Canada) were in the ranges of 0.64–20.7 ng/g (median: 1.44–3.61 ng/g ww) and < 0.3–4.2 ng/g (median: 0.31 ng/g, ww), respectively (Giraudo et al., 2020; Lu et al., 2017a).

Among the 5 sampling lakes in Cornwallis Island, landlocked char livers from the Resolute Lake showed the highest detection frequency (60 %) and concentrations of UV328 (median: 0.13 ng/g, ww; range: <0.03–1.06 ng/g) (Fig. 3), followed by Meretta Lake (40 %; <0.03–2.17 ng/g, ww; median not estimated because $n = 5$ and detection frequency < 50 %). Interestingly, UV320 detection frequency and concentrations were higher in landlocked char liver samples from one of the most remote study lakes, Amituk, than in samples from other lakes (Fig. 3). Amituk Lake is roughly 48 km northeast away from the community of Resolute Bay and less impacted by human activities than other studied lakes located on the southwestern edge of the island and closer to (3–12 km) Resolute Bay (Cabrerizo et al., 2018b). The higher levels of UV320 in Amituk Lake char livers is similar to the trend for mercury, dichlorodiphenyltrichloroethane (DDTs), and toxaphene in char muscle (with skin in some cases) in this lake compared to the other lakes (Cabrerizo et al., 2018b; Hudelson et al., 2019). Greater atmospheric deposition may be a factor affecting higher levels of some contaminants in this lake (Cabrerizo et al., 2018b; Hudelson et al., 2019). Amituk Lake and its catchment has been shown to receive greater precipitation due to its topography and higher elevation (140–240 m above sea level) compared to the other lakes (catchments of 20–160 m above sea level) (Semkin et al., 2005). Atmospheric deposition has been suggested as a dominant contributor to emerging organic contaminants in the Arctic (Brown et al., 2018; Xie et al., 2022). However, no information is currently available on the atmospheric deposition of BZT-UVs in the Arctic.

Contaminant accumulation in biota tissues varies depending on exposure and toxicokinetics. The results above discussed various exposure-related factors that influence contaminant levels in species/individuals, such as habitat, migration, diet composition, ingestion of microplastics, and concentrations in the surrounding environment. In addition to these factors, absorption, distribution, metabolism, and excretion (toxicokinetics) also influence contamination patterns among species. *In vivo* studies have shown that BZT-UVs have varying half-lives in seabirds, fish, and mammals. Tanaka et al. (2020) found that the half-lives of UV328, UV326, and UV327 in the liver of streaked shearwater chicks, *via* feeding plastics containing these additives, were approximately 16 days. In the liver of juvenile rainbow trout (*Oncorhynchus mykiss*) (diet exposure) and zebrafish (water exposure), the half-life of UV328 was 3 days and 6–7 days, respectively (Giraudo et al., 2020; Zhang et al., 2021). No data are available regarding the half-life of BZT-UVs in wild mammals, such as polar bears and beluga, analyzed in the present study. However, a study on human volunteers found that the half-life of UV328 in blood was approximately 5–16 h after food exposure (Denghel et al., 2021). In these *in vivo* studies, various metabolites of BZT-UVs have been found in fish and humans (Denghel et al., 2021; Zhang et al., 2021), indicating that metabolism plays an important role in affecting their accumulation in biological tissues. Compared to fish and mammals, the higher accumulation of BZT-UVs in seabird livers is consistent with the longer half-life of these contaminants in seabirds.

For the species examined in this study, varying biotransformation activities may impact the observed levels of contaminants. While no data were available for the biotransformation of contaminants investigated in this study, previous research has indicated that murre, fulmar, and guillemot have lower hexachlorocyclohexane biotransformation activity compared to other Arctic seabird species, such as black-legged kittiwakes (*Rissa tridactyla*) and glaucous gulls (*Larus hyperboreus*) (Braune et al., 2019; Borgå et al., 2001; Moisey et al., 2001). Furthermore, studies have suggested that the biotransformation activities of chlordane follow the order of murre> fulmar> guillemot (Fisk et al., 2001). *In vitro* studies have shown that while the liver microsomes of polar bears and beluga cannot biotransform some organic contaminants, polar bear liver microsomes exhibit faster biotransformation of N-ethyl-perfluorooctane sulfonamide and some PBDE congeners than beluga (Letcher et al., 2014; McKinney et al., 2011a). Consequently, the potential differences in the biotransformation of target contaminants among species in the present study may influence their species-specific accumulation in the Arctic biota.

3.2. UV filters (UVFs)

Akin to BZT-UVs, seabirds (except for fulmar) (median: murre 48.3 ng/g ww; guillemot 33.8 ng/g ww) showed higher levels of ΣUVFs than other studied Arctic species (median: NA-6.23 ng/g ww) (Fig. 2 and Table S7). The dominant UVFs in the liver of guillemots and murres were HMS (55 %; median: 33.8 ng/g; range: < 2.9–258 ng/g ww) and 4MBC (36 %; median:16.5 ng/g; range: < 4.2–292 ng/g ww), respectively (Table S7 and Fig. S4). Guillemots from Qaqulluit accumulated more HMS than those from Akpait (median: 124 *vs.* 26.7 ng/g, ww; Fig. 3), demonstrating the variation of the accumulation of this contaminant in seabirds even between close colonies in the Arctic (< 50 km apart). This is consistent with the distinguishable gene expression signatures related to stressors of the two populations (Zahaby et al., 2021). The higher detection of 4MBC in murre livers (36 %; median 16.5 ng/g; range: <4.2–292 ng/g ww) than other seabird species (guillemot <5.3 ng/g ww; fulmar: <5.8 ng/g ww) in the present study is consistent with

previously reported more frequent detection of this contaminant in the eggs of murres than fulmars and black-legged kittiwakes from Prince Leopold Island (Lu et al., 2019a). Murres and guillemots forage at higher trophic levels than fulmars in most Canadian Arctic colonies (Braune et al., 2019; Pratte et al., 2019), while HMS and 4MBC can biomagnify in aquatic environments (Lyu et al., 2022; Yang et al., 2020), which may explain the greater enrichment of these two UVFs in murres and guillemots than in fulmars. For murres, the liver of males (median: 24.5 ng/g ww) exhibited greater accumulation of 4MBC than females (median: NA) (Fig. S3), which may be related to the egg depuration mechanism.

In the liver of polar bears, 4MBC (39 % detection frequency; estimated median: 4.22 ng/g, ww) was determined as the major UVF (Table S7 and Fig. S4). There was no significant difference in 4MBC concentrations between the southern and western Hudson Bay polar bear livers. For beluga, HMS (<2.7–12.5 ng/g ww), BP3 (<3.0–4.96 ng/g ww), and 4MBC (<2.4–13.6 ng/g, ww) were only found in 10 % of the liver samples, implying low contamination of these UVFs in this population.

In addition, landlocked char from Amituk Lake (60 %; median: 6.23 ng/g ww; range: <1.1–13.6 ng/g ww) and Small Lake (80 %; median: 1.92 ng/g ww; range: <1.1–4.48 ng/g ww) on Cornwallis Island revealed higher levels of EHS than the char from the other 3 sampling lakes (Fig. 3), while landlocked char livers overall showed higher detection and concentrations of EHS (36 %; median: 1.0 ng/g ww; range: <1.1–13.6 ng/g ww) than other species (Fig. S4 and Table S7). Interestingly, ΣUVFs levels in landlocked char from 5 lakes were negatively correlated with $\delta^{13}C$ (likelihood $r = -0.58$, $p = 0.008$), but not other biological variables. This finding suggests that char that feed greater pelagic food sources, *e.g.* zooplankton, may accumulate more UVFs. This relationship is consistent with previously reported results for the relationships between BP3 and $\delta^{13}C$ in deepwater redfish (*Sebastes mentella*) from the St. Lawrence Gulf in Canada (Moualek et al., 2024). However, this result should be interpreted with caution because all fish from 5 lakes were considered together for analysis due to the small number of samples analyzed from each lake, which may introduce some uncertainties.

3.3. Industrial antioxidants

In general, industrial antioxidant detection frequency and concentrations in Arctic biota samples were much lower than UV absorbents, consistent with previous findings (Lu et al., 2019a). These results suggest that these antioxidants are less persistent and/or less bioaccumulative compared to UV absorbents in Arctic wildlife, despite the higher production levels of industrial antioxidants (1–2 orders of magnitude) over UV absorbents (Lu et al., 2019a).

The SPA 26DTBP was detected in the liver of seabirds (detection frequency: 25 % (fulmar) - 45 % (guillemot); median: NA (fulmar) - 1.14 ng/g ww (guillemot)) and polar bears (detection frequency: 89 %; median: 1.91 ng/g ww) (Table S8). The frequency of detecting 26DTBP was higher (80 % *vs.* 22 %; median 6.01 ng/g ww *vs.* NA) in the livers of the murres, which had microplastics in the faecal precursor, than in those that did not (Bourdages et al., 2021), similar to UVP and UV328 (Table S5). The detection frequency of 26DTBP increased to 100 % in the 3 individuals in which we detected plastic fibers (Table S5). There was no clear pattern linking polymer types to the detection of 26DTBP due to the large variation or unknown polymer types in some samples (Bourdages et al., 2021). It is unknown if 26DTBP could be transferred from microplastics into seabird tissues. However, Kühn et al. (2020) demonstrated that 2,4-di-*tert*-butylphenol (24DTBP), an isomer of 26DTBP, can leach from marine plastic debris into seabirds' stomach oil. Given the very similar physicochemical properties of these two isomers of butylphenol, it is possible that 26DTBP could also be transferred from plastics to avian tissues.

Higher levels of 26DTBP were found in the livers of polar bears from southern Hudson Bay (median: 2.73 ng/g ww), compared to western Hudson Bay (median:1.34 ng/g ww), which is consistent with the distribution pattern of UV329 (Fig. 3). Given the recently identified potential carcinogenic risks of 26DTBP and its metabolite 2,6-di-*tert*-butyl-1,4-benzoquinon in mice, the bioaccumulation of this contaminant and its metabolites in Arctic mammals raising concerns about its health effects (Cui et al., 2021a, 2021b).

For ∑Ar-SAs, seabird livers accumulated higher concentrations than other species (murre: median 0.20 ng/g ww, range < LOD-5.94 ng/g ww; fulmar: median 4.06 ng/g ww, range 0.16–5.31 ng/g ww; guillemot: median 0.07 ng/g ww, range < LOD to 4.71 ng/g ww) (Table S9). The livers of fulmars and murres accumulated more C8C8 (median 0.73 ng/g ww, range < 0.1–3.16 ng/g ww) and diAMS (median 0.19 ng/g ww, range < 0.2–5.94 ng/g ww), respectively than other species (Fig. S5 and Table S9). C8C8 was previously detected in fulmar livers from Prince Leopold Island and the Labrador Sea (median 0.02 ng/g ww) (Lu et al., 2019a), but the concentrations were lower compared to the levels detected in the present study (median 0.73 ng/g, ww). In addition, sex differences in the concentrations of 26DTBP (median: males 1.06 ng/g ww, females NA) and diAMS (median: males 0.14 ng/g ww, females NA) in murre livers were found in the present study, with higher detection frequency and levels in the liver of males (Fig. S3).

4. Conclusion

This is the first report to compare the contamination of UV absorbents and industrial antioxidants in the liver of various Canadian Arctic wildlife species. Understanding the occurrence and distribution of these contaminants of emerging concern in Arctic wildlife provides a baseline for elucidating their fate and impacts in polar regions. Seabirds tended to accumulate more UV absorbents and industrial antioxidants than mammals and fish, suggesting that seabirds can be used as indicator species for long-term monitoring of these contaminants in the Arctic. Bird species, sex, colony location, and ingestion of microplastics may influence the concentrations of UV absorbents and industrial antioxidants in their tissues during the breeding season, which should be considered for future monitoring programs. The SPA 26DTBP was more frequently detected in the livers of seabirds and polar bears, raising concerns about the potential health effects of this contaminant and its metabolites on Arctic wildlife. Although these contaminants were detectable in Arctic char, concentrations were generally lower than previously reported in fish from temperate regions. Spatial variation was found for the contamination of some target compounds in guillemot, polar bear, and landlocked char, suggesting that the levels of these contaminants may differ in their environment or prey and that human exposure risks may differ when consuming them from distinct locations. The mechanisms underlying the greater accumulation of these contaminants in seabirds than in other species and the factors driving the differences between sexes and colony variation remain unresolved. As well, the species-specific toxicokinetics, toxicodynamics, and health effects of these contaminants on wildlife and humans in the Arctic need to be further investigated.

CRediT authorship contribution statement

Ingrid-Alejandra Granados-Galvan: Visualization, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization, Writing – review & editing, Writing – original draft. **Jennifer F. Provencher:** Resources, Methodology, Investigation, Funding acquisition, Conceptualization, Writing – review & editing. **Mark L. Mallory:** Resources, Methodology, Investigation, Conceptualization, Writing – review & editing. **Amila De Silva:** Resources, Methodology, Investigation, Funding acquisition, Conceptualization, Writing – review & editing. **Derek C.G. Muir:** Resources, Methodology, Investigation, Funding acquisition, Conceptualization, Writing – review & editing. **Jane L. Kirk:** Resources, Investigation, Writing – review & editing. **Xiaowa Wang:** Resources, Writing – review & editing. **Robert J. Letcher:**

Resources, Methodology, Investigation, Funding acquisition, Conceptualization, Writing – review & editing. **Lisa L. Loseto:** Resources, Methodology, Investigation, Funding acquisition, Conceptualization, Writing – review & editing. **Bonnie M. Hamilton:** Resources, Methodology, Investigation, Funding acquisition, Writing – review & editing. **Zhe Lu:** Visualization, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization, Writing – review & editing, Writing – original draft.

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.

Data availability

Data will be made available on request.

Acknowledgments

This research was primarily funded by Crown-Indigenous Relations and Northern Affairs Canada (Northern Contaminants Program). Supplemental funding was provided by the Natural Sciences and Engineering Research Council of Canada (Discovery Grants to ZL) and the Fonds de recherche du Québec–Nature et Technologies (Research Support for New Academics to ZL, and Scholarships: PBEEE 334561 and B2X 331680 to AG). We appreciate the scholarships provided by Centre De Recherche En Écotoxicologie Du Québec (EcotoQ) and UQAR-ISMER (to AG). We thank the Sululiit ACMC and Nattivak Hunter and Trapper Organization in Qikiqtarjuaq for supporting the seabird work at Qaqulluit and Akpait National Wildlife Areas. We thank the Hunters' and Trappers' Association (HTA) in Resolute Bay and Resolute Bay team member Debbie Iqaluk for the ongoing support and collaboration which enabled all the work on landlocked Arctic char. We thank Polar Continental Shelf Program for logistical support for all char collections. We thank the Ekaluktutiak Hunters' and Trappers Organization (EHTO) for their continued support of the work on sea-run char. We acknowledge the support from Fisheries and Oceans Canada, including staff support of S. MacPhee, J. Brewster, A. Elliot, D. Neumann, L. Murray, M. Wardekker for sample collection and management, the Fisheries Joint Management Committee, in particular, E. Way-Nee and K. Campbell, the Tuktoyaktuk Hunters and Trappers Committee and beluga monitors R. Noksana, L. Kikoak, and R.Green, and beluga harvesters from Tuktoyaktuk for supporting the beluga sample collection. We are thankful for our partnerships under NWRPs 2019-001 and 2020-001 with the Nunavut Department of Environment and the HTAs in Sanikiluaq, Rankin Inlet, Arviat and Whale Cove for polar bear sample collection and provision. We also thank Amelie Roberto-Charron and Alyssa Bohart at the Nunavut Department of Environment and the Nunavut Hunters Trappers Organizations partners in the participating Hudson Bay communities with respect to the polar bear annual harvests, sample collections, and Indigenous knowledge. We also appreciate the technical assistance provided by Florentine Malaisé, Winny Pan, Alice Guillot, and Mathieu Babin at UQAR. Finally, we thank the anonymous referees whose suggestions helped improve our manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2024.175693.

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