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Toxic elements in arctic and sub-arctic brown bears: Blood concentrations of As, Cd, Hg and Pb in relation to diet, age, and human footprint

Boris Fuchs [a,*], Kyle Joly [b,**], Grant V. Hilderbrand [c], Alina L. Evans [a], Ilia Rodushkin [d,e], Lindsey S. Mangipane [f], Buck A. Mangipane [g], David D. Gustine [f], Andreas Zedrosser [h,i], Ludovick Brown [j], Jon M. Arnemo [a,k]

[a] Department of Forestry and Wildlife Management, Inland Norway University of Applied Sciences, Campus Evenstad, 2480, Koppang, Norway
[b] National Park Service, Gates of the Arctic National Park and Preserve, 99709, Fairbanks, Alaska, USA
[c] National Park Service, Alaska Regional Office, 99501, Anchorage, Alaska, USA
[d] Division of Geosciences, Luleå University of Technology, 97187, Luleå, Sweden
[e] ALS Scandinavia AB, 97187, Luleå, Sweden
[f] U.S. Fish and Wildlife Service, Marine Mammals Management, 99503, Anchorage, Alaska, USA
[g] Lake Clark National Park and Preserve, National Park Service, 99501, Anchorage, Alaska, USA
[h] Department of Natural Science and Environmental Health, University of South-Eastern Norway, 3800, Bø in Telemark, Norway
[i] Institute for Wildlife Biology and Game Management, University of Natural Resources and Life Sciences, 1180, Vienna, Austria
[j] Département de Biologie, Université de Sherbrooke, J1K 2R1, Sherbrooke, Québec, Canada
[k] Department of Wildlife, Fish and Environmental Studies, Swedish University of Agricultural Sciences, 901 83, Umeå, Sweden

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ABSTRACT

Contamination with arsenic (As), cadmium (Cd), mercury (Hg) and lead (Pb) is a global concern impairing resilience of organisms and ecosystems. Proximity to emission sources increases exposure risk but remoteness does not alleviate it. These toxic elements are transported in atmospheric and oceanic pathways and accumulate in organisms. Mercury accumulates in higher trophic levels. Brown bears (*Ursus arctos*), which often live in remote areas, are long-lived omnivores, feeding on salmon (*Oncorhynchus* spp.) and berries (*Vaccinium* spp.), resources also consumed by humans.

We measured blood concentrations of As, Cd, Hg and Pb in bears (n = 72) four years and older in Scandinavia and three national parks in Alaska, USA (Lake Clark, Katmai and Gates of the Arctic) using high-resolution, inductively-coupled plasma sector field mass spectrometry. Age and sex of the bears, as well as the typical population level diet was associated with blood element concentrations using generalized linear regression models.

Alaskan bears consuming salmon had higher Hg blood concentrations compared to Scandinavian bears feeding on berries, ants (*Formica* spp.) and moose (*Alces*). Cadmium and Pb blood concentrations were higher in Scandinavian bears than in Alaskan bears. Bears using marine food sources, in addition to salmon in Katmai, had higher As blood concentrations than bears in Scandinavia. Blood concentrations of Cd and Pb, as well as for As in female bears increased with age. Arsenic in males and Hg concentrations decreased with age.

We detected elevated levels of toxic elements in bears from landscapes that are among the most pristine on the planet. Sources are unknown but anthropogenic emissions are most likely involved. All study areas face upcoming change: Increasing tourism and mining in Alaska and more intensive forestry in Scandinavia, combined with global climate change in both regions. Baseline contaminant concentrations as presented here are important knowledge in our changing world.

* Corresponding author.
** Corresponding author.
E-mail addresses: boris.fuchs@inn.no (B. Fuchs), kyle_joly@nps.gov (K. Joly).

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1. Introduction

Heavy metals/metaloids (HM), such as arsenic (As), cadmium (Cd), mercury (Hg), and lead (Pb), are non-essential toxic elements and exposure to these elements in the body from both natural and anthropogenic sources is a global One Health issue (Buttke et al., 2015). Even in areas with a minimal cumulative human impact on the ecosystem, measured as the human footprint (Williams et al., 2020), anthropogenic contaminant levels of these elements increase natural background levels substantially (AMAP, 2005; Dastoor et al., 2022; Macdonald et al., 2005; Renberg et al., 2001). Heavy metal and metalloid emissions mainly originate from fossil fuel combustion, non-ferrous metal production, waste incineration, or changes in the environment, such as the draining of wetlands, and are spread globally through atmospheric deposition (AMAP, 2005; Bjerregaard et al., 2022). In addition, contamination with Pb causing toxicity in vertebrates occurs when humans hunt with Pb-based ammunition (Arnemo et al., 2022; Bjerregaard et al., 2022).

Arsenic, Cd, Hg and Pb are known to readily bioaccumulate (i.e., concentrate in bodies over time), whereas biomagnification (i.e., increases at higher trophic levels) is more pronounced for Hg (Ali and Khan, 2019; Atwell et al., 1998; Le Bourg et al., 2019). Arctic and sub-arctic regions are susceptible to Hg contamination because snow and ice in combination with polar light conditions facilitate the formation and deposition of Hg oxides. Dietary uptake of methyl-Hg^+ (MeHg) is the major exposure pathway in terrestrial and aquatic wildlife (Scheuhammer et al., 2007).

Wildlife is especially prone to HM exposure; for example, the California condor (*Gymnogyps californianus*) was on the brink of extinction with Pb poisoning being an important accelerating factor (Finkelstein et al., 2012). Effects of exposure are often sublethal, for example, golden eagles (*Aquila chrysaetos*) are documented to change movement behavior with increasing Pb exposure (Ecke et al., 2017). Feeding ecology and habitat use are important factors of pollutant exposure. For example, coastal Arctic foxes (*Vulpes lagopus*) have higher Hg concentrations than their conspecifics further inland due to different diets (Bocharova et al., 2013), and omnivores have been found with higher concentrations of Cd and Pb than sympatric carnivores (Lazarus et al., 2017). Black bears (*Ursus americanus*) in areas with higher ungulate harvest density have higher concentrations of Pb measured in their teeth likely due to ingestion of fragments from spent Pb – based ammunition (Brown et al., 2022).

Brown bears (*U. arctos*) inhabit the lower arctic and sub-arctic regions and are susceptible to HM exposure, especially Pb, in areas with a relatively large human footprint (Boesen et al., 2019; Fuchs et al., 2021; Lazarus et al., 2017; Rogers et al., 2012). No direct links to any sources of exposure have been established yet, but there is general agreement on the dietary intake of either environmental Pb as well as the ingestion of Pb fragments from Pb-based ammunition used for hunting ungulates as major exposure routes (Fuchs et al., 2021; Lazarus et al., 2017; Rogers et al., 2012). During hibernation, bears do not defecate or urinate, both of which are major pathways to excrete HM from the body. Individual bears exhibit highly adaptive feeding behaviors, ranging from nearly herbivorous to highly carnivorous, and food sources from mainly marine to mainly terrestrial depending on the region (Hertel et al., 2018; Hilderbrand et al., 1999; Koike et al., 2012). During hyperphagia bears within a region typically facilitate the same major food source: In Scandinavia, bears mainly feed on berries (*Vaccinium* spp.) (Hertel et al., 2018), in southern and eastern Europe mainly on hard mast (i.e. nuts and seeds) (Bojarska and Selva, 2012; Kavčič et al., 2015), and in North America, some populations rely on salmon (*Oncorhynchus* spp.) (Hilderbrand et al., 1999). The enrichment of ^{13}C and ^{15}N isotopes has been used to reconstruct and link diet to HM exposure in vertebrates and invertebrates in both terrestrial and maritime systems (Larsson et al., 2007; Le Croizier et al., 2019; Noël et al., 2014; Ramos et al., 2009). For example, polar bears (*U. maritimus*) and brown bears that use maritime food sources, as measured by the enrichment of the ^{13}C isotope, have higher Hg levels compared to bears using terrestrial food sources (Cardona-Marek et al., 2009; Noël et al., 2014).

Studies on HM contamination in sub-arctic terrestrial wildlife commonly report concentrations for internal organs, such as the liver, kidneys, or bones from dead specimens (AMAP, 2005). These tissues are well suited to study exposure and accumulation of As, Cd, Hg and Pb (Tan et al., 2009); however, they often involve destructive sampling. Tissues such as blood or hair, which can be collected from live animals, are more suitable for specimens from protected areas or endangered species. Blood is often used as an indicator for recent exposure to one of these four elements. The gastrointestinal tract takes up As, Cd, Hg and Pb which are distributed by the blood stream and stored in different body compartments. Half-life of elimination in blood is relatively short, for example, in humans; As has a half-life in blood of three to 4 h, and blood is not used as a tissue to detect low exposure (i.e., population means < 1 μg/L) (ATSDR, 2020), however, blood As concentration accurately reflects levels of chronic exposure (Hall et al., 2006). For Cd, Hg and Pb, whole blood concentrations are used to detect exposure. Half-life of Cd in blood is three to four months (ATSDR, 2015). Total Hg half-life in blood is one to three weeks and in contrast to the other discussed elements, also half-time of elimination from the human body is with estimated three months much shorter (ATSDR, 2022; Yaginuma-Sakurai et al., 2012). The half-life of Pb in blood is four to five weeks in humans (Rabinowitz et al., 1976), and probably similar in bears (Arnemo et al., 2022). However, elements accumulated in organs and bones remobilize into the blood stream and concentrations reach equilibrium between the different compartments (Mattisson et al., 2010; Rabinowitz et al., 1976; Yaginuma-Sakurai et al., 2012). Thus, blood HM concentrations reflect a combination of recent uptake and total body burden. Different physiological states, such as lactation, hibernation, malnutrition, or death, might change the concentrations at which these elements are at equilibrium between the different compartments (Fuchs et al., 2021; Silbergeld, 1991; Söderberg et al., 2023).

The goal of our study was to compare total concentrations of As, Cd, Hg and Pb in whole blood samples from four brown bear populations living on different continents and subsisting on different diets, as well as to identify potential sources of exposure. Three of these populations were located in Alaska, USA, and include bears with an almost entirely marine protein-based diet consisting mainly of salmon (Hilderbrand et al., 2018), as well as a population in Scandinavia with a diet based on terrestrial proteins and vegetation (Stenset et al., 2016). To help identify potential sources for As, Cd, Hg and Pb, we paired blood metal concentrations with stable $\delta^{13}C$ and $\delta^{15}N$ isotopes in hair samples in one population (Gates of the Arctic National Park and Preserve, Alaska) where individual bears subsist on either a terrestrial vegetation-based diet or a maritime protein-based diet.

We hypothesized that *i)* HM concentrations is highest in Scandinavia associated with the magnitude of the human footprint index at the sample location; *ii)* HM concentrations reflect the major food resource of the population, and *iii*) bioaccumulation of As, Cd and Pb in target tissues with age is reflected in whole blood samples. Based on these hypotheses, we predicted that: I) HM concentrations are greater in areas with a larger human footprint; II) bears acting as apex predators (i.e., with a high proportion of salmon in their diet) have higher concentrations of HM compared to bears acting as primary consumers (i.e., with high vegetation levels in their diet); and III) older bears have higher whole blood concentrations of As, Cd and Pb than younger bears.

2. Material and Methods

2.1. Study populations

The study area for the Scandinavian brown bear population is in south-central Sweden (~61°N, 15°E). The area (~13,000 km^2) is predominantly covered with intensively managed coniferous forest (*Pinus sylvestris* and *Picea abies*). Forest stands are typically planted, thinned, and clear-cut at an age of 80–100 years, followed by scarification and replanting (Martin et al., 2010; Swenson et al., 1999). The hilly landscape is interspersed with bogs and only few agricultural fields in the east. Small settlements occur in the entire study area with a human density <9/km^2. Low-traffic roads used for forestry and recreational access are spread throughout the study area at a density of 0.3 km/km^2 and 0.7 km/km^2 for paved and gravel roads, respectively. Between 2010 and 2020, the annual moose (*Alces alces*) harvest rate was 0.23/km^2 (Länsstyrelserna, 2020); 98% of the moose were shot with Pb-based ammunition and their digestive tracts, organs, and slaughter remains were mostly left in the forest (Stokke et al., 2017). There are indications for, but no hard data, that more moose hunters are using Pb-free bullets in recent years. In 2022, brown bears were hunted at a rate of 0.002/km^2 within management areas 2 and 6 in Dalarna County, representing the study area (Länsstyrelserna, 2022). Based on scat sampling during the hyperphagic period, berries (*Vaccinium* spp.*, Empetrum hermaphroditum*) make up 68% of the estimated dietary energy content, insects (mainly ants of the genus *Formica*) 14%, and ungulates (moose) 14%. In spring (April/May), ungulates are the most important food source (61%), followed by insects (20%), and berries (9%, mainly *V. vitis-idaea* from the previous fall) (Stenset et al., 2016). No anadromous fish species occurs in the study area and there are no indications of fish in the bears' diet. Aerial depositions are the primary source of environmental As, Cd and Pb in south-central Scandinavia (Renberg et al., 2001; von Storch et al., 2003). Blood samples were collected during the period 2010–2021, between mid-April and mid-June (Table 1). Pb concentrations from 14 of these samples have been included in Fuchs et al. (2021).

Gates of the Arctic National Park and Preserve (Gates NP) is a remote and undeveloped 34,400 km^2 protected area in the Brooks Range in northern Alaska (~68°N, 154°E), USA. This interior park is > 200 km from the coast and is covered by arctic tundra, boreal forest (with main tree species *Picea* spp., *Betula neoalaskana*, P*opulus tremuloides*), lowland riparian communities, and high alpine terrain. There are no roads within the park boundary. Bears in Gates NP show high individual and moderate interannual variation in primary food intake. Based on stable carbon and nitrogen isotope analysis, protein contributes 2–96% to the late summer and fall diet in female bears (Mangipane et al., 2020). Of the protein intake, 77% is of marine origin (i.e., chum salmon (*Oncorhynchus keta*)) (Mangipane et al., 2020). Other sources of protein are terrestrial mammals, such as moose, caribou (*Rangifer tarandus*), Dall's sheep (*Ovis dalli*), and Arctic ground squirrel (*Urocitellus parryii*). Bears relying on vegetation during hyperphagia consume mainly berries (*Vaccinium* ssp.*, Empetrum nigrum, Shepherdia canadensis*). Approximately 10,000 visitors travel to Gates of the Arctic NP each summer, and most visitors access the park by small aircraft (IRMA, 2022). Blood and hair samples were collected in May and June 2016 (Table 1).

Lake Clark National Park and Preserve (Clark NP) is situated along the coast of southwestern Alaska (~60°N, 153°E) and covers approximately 16,300 km^2. The area for this study was on the western side of the Chigmit Mountain Range and none of the bears collared as part of this study accessed the coastal (east) side of the park (Mangipane et al., 2018). Deciduous and coniferous forest, as well as different shrub and grass communities, cover large parts of this study area (Hilderbrand et al., 1999). Salmon originating in Bristol Bay were the primary resource for Clark NP bears, but berries and other ungulates were available. Clark NP is visited by approximately 20,000 visitors each summer and is not connected to any road system but primarily accessed by small aircraft (IRMA, 2022). Blood samples were collected in May 2016 (Table 1).

The coastal Katmai National Park and Preserve (Katmai NP) covers about 16,500 km^2 in south-western Alaska (58°N, 155°E) and has one of the highest brown bear densities in the world (Ferguson and McLoughlin, 2000). Salmon and other marine resources, such as clams (*Mya arenaria, Siliqua patula*), flounder (*Platichthys stellatus*), and marine mammals were important food resources for bears (Erlenbach, J.A., 2020). However, diets have shifted during recent decades and vegetation is now the dominant food component at the population level. Bear habitat included extensive areas of shrubs (*Alnus* spp., *Salix* spp., *Betula* spp.), as well as boreal forest and alpine terrain. All bears sampled in Katmai NP generally remained on the coast and did not venture inland and thus mainly relied on marine resources from Cook Inlet (Erlenbach, J.A., 2020). Katmai NP, visited by approximately 40,000 visitors annually, is not connected to any road system and small aircraft is used to access the park (IRMA, 2022). Blood samples were collected in May and July in 2016 (Table 1). Recreational hunting is not permitted within all three Alaskan parks, but occasional subsistence hunting occurs at a very low level.

We used a human footprint index based on the 2013 data by Atkinson and Williams (2020) that indicates the cumulative anthropogenic pressure on an ecosystem to quantify the human impact in the different study areas. The index compiles survey and remotely sensed data on land use, represents the degree of wilderness, and has been linked to biodiversity and species extinction risk (Williams et al., 2020). We created a 2-km radius buffer around each sample location and extracted the highest index value within the buffer. Brown et al. (2023) found that background Pb contamination and the probability of hunter killed moose as a potential Pb source within a 2-km radius around the sampling location best correlated with the blood Pb concentration of bears in Scandinavia.

2.2. Sample collection

All bears, in all study areas, were chemically immobilized from a helicopter between mid-April and beginning of July, had their sex-determined, and were weighed. Capture and handling procedures in Alaska were approved by the US National Park Service (AKR_KATM_Hilderbrand_BrownBear_2014, AKR_LACL_Mangipane_BrownBear_2014, AKR_GAAR_Gustine_GrizzlyBear_2014) and the US Geological Survey (2014–01, 2015–04, 2015–06) Animal Care and Use Committee. In Scandinavia, captures were approved by the Swedish Ethical Committee on Animal Research, Uppsala, Sweden (C18/15) and the Swedish Environmental Protection Agency, Stockholm, Sweden (NV-00741-18). Age was determined by the cementum analysis of a vestigial first premolar (Hilderbrand et al., 1999; Mattson, 1993) or estimated by tooth wear (Hilderbrand et al., 2018). All Alaskan bears were four years or older. In Scandinavia, most bears had been captured as yearlings together with their mother, thus their ages were known. Blood for HM analysis was collected from the jugular vein in Scandinavia or the cephalic vein in Alaska in either 8 ml evacuated heparin trace element tubes or 4 ml evacuated K3EDTA tubes and frozen at −20 °C the same day. All samples were analyzed in the same laboratory (ALS Scandinavia AB, Luleå, Sweden). Shipping from the USA to Sweden followed the CITES regulations (export permit #19US15593D/9, import permit #4.10.18–12752/2019). Samples were prepared by closed vessel MicroWave-assisted acid digestion and HM concentration measured by high-resolution, inductively-coupled plasma sector field mass spectrometry. Concentrations, in μg/L, were for the total amount of each element. Lower level of quantification was 0.02 μg/L for As, 0.006 μg/L for Cd, 0.4 μg/L for Hg and 0.2 μg/L for Pb. Further details on the analytical methods can be found in the supplementary material as well as in Rodushkin et al. (2000) and Söderberg et al. (2023).

Table 1
Sample size (N), range and mean of body mass (kg), age (years), and date of blood sampling (month of the year) of brown bears (*Ursus arctos*) in Scandinavia and national parks and preserves (NP) in Alaska, USA.

Sample area	N	Body mass	Age	Sample Date
Scandinavia	30	58–197, 90	4–13, 6	April–June 2010–2021
Gates NP	18	80–251, 114	8–25, 13	May–June 2016
Clark NP	12	100–202, 159	9–30, 15	May 2016
Katmai NP	12	108–184, 146	5–19, 10	May & July 2017
Total	72	58–251, 116	4–30, 10	

2.3. Sample selection

We analyzed a total of 72 blood samples in this study. From Alaska, we included 18 samples from Gates NP, 12 from Katmai NP, and 12 from Clark NP. In Scandinavia, over 300 blood samples from 71 individual bears have been analyzed for HM. To obtain a balanced data set, we subsampled the Scandinavian data set to contain only individuals ≥ four years, and with the variables age, sex, and body mass available. We then randomly selected only one sample per individual, and from this pool we again randomly selected 30 samples for statistical comparison with the Alaskan data.

2.4. Statistical analysis

We used generalized linear models (glm) with a gamma distribution and a log-link function to estimate differences in blood HM concentration of individual bears between the study areas in R 4.2.1 (R Core Team, 2022). We formulated the same set of candidate model for each element tested (Table SM 1), and selected top models using the second order Akaike Information Criterion (AICc) in the AICmodavg package (Mazerolle, 2019). Models within ΔAICc <2 were considered equivalent and estimates averaged using the *modavg* function. For all candidate models, we fitted the blood HM concentration as response variable and study area as explanatory variable. To this base model, we added either sex or age of the bear, both variables combined, as well as a model with an interaction of these variables. We did not include body mass due to the correlation with age (Spearman's rho = 0.65). Sex was included to control for behavioral and physiological differences due to reproduction between males and females (Hertel et al., 2018; Swenson et al., 2007; Tan et al., 2009), and age as indication of bioaccumulation. We included the interaction age × sex to test for different accumulation trends between males and females at different ages. We considered differences between the study areas significant if the 95% confidence intervals did not overlap. To facilitate model performance we kept values below level of quantification as provided by the mass spectrometry. Pointwise back transformation was used when interpreting summary tables and for predictions using the exp function in R.

2.5. Heavy metal (loids) and diet in Gates NP

Both the relative abundance of stable isotopes of carbon (δ^{13}C) and nitrogen (δ^{15}N) in guard hair and derived estimates of proportions for terrestrial protein, marine protein, and vegetation in the diet of individual bears have been previously analyzed and were available for Gates NP (Mangipane et al., 2020). In brief: Hair sections 4 cm from the root, representing growth from July through October, were selected for analysis to evaluate assimilated diet within that period (Mangipane et al., 2020). Isotopic values are reported as the relative difference from the sample isotopic ratios (^{13}C/^{12}C or ^{15}N/^{14}N) to the standard isotopic ratios; i.e., $[(R_{Sample}/R_{Standard})-1]*1000$. PeeDee Belemnite limestone was standard for carbon and atmospheric N_2 for nitrogen. To estimate dietary proportions, δ^{13}C and δ^{15}N values were compared to source data using the bayesian isotopic mixing model MixSIAR (version 3.1.10), with sex as a fixed effect and bear ID and a process error term as random effects. Previously estimated δ^{13}C/δ^{15}N for chum salmon (Johnson and Schindler, 2009), a sub-arctic plant baseline (Mowat and Heard, 2006), and terrestrial meat was used as source data (Mowat and Heard, 2006). This model was run on a data set including 80 samples from 58 individual bears. The proportional contribution of the analyzed dietary sources was estimated for each bear-year ID. We combined the model estimates and 95% credible intervals for individual bears where blood samples for HM were available. Hair and blood were collected at the same capture event, hair represents diet the previous year and blood HM concentration the combination of recent and long-term exposure. We assume limited annual variation on the individual level in the dominant diet, however, salmon abundance varies between years. We used glm's to test if the blood concentrations of As, Cd, Hg and Pb in bears with >50% salmon in the diet the previous year differed compared to the bears with >50% vegetation in their diet the previous year. The difference between the groups was considered significant if $P < 0.05$.

3. Results

Summary statistics for each element in each area are presented in Table 2. Arsenic concentrations in six blood samples from Scandinavian bears and one from the Gates NP were below the limit of quantification (<0.15 μg/L). The human footprint index within the 2-km buffer of each sample location was 0 for all samples from the Alaskan study areas; we therefore decided to not include the human footprint index in further modelling. In Scandinavia, the maximal human footprint index within the buffer around each sample location ranged from 1 to 18, with a mean of 9.2. None of the HM concentrations in the Scandinavian samples were correlated with the human footprint index (correlation factors ranged between $r = 0.01$ for Hg and $r = 0.17$ for As with all p-values >0.05 (0.29–0.94)).

3.1. Arsenic

Three models were considered equivalent in explaining As in blood samples: the top model contained sample area as the only predictor, another model included sample area and the interaction between sex and age, and third model included sample area and sex (Table SM 1). We based interpretation on model-averaged estimates of these three models (Table 3). Predicted As blood concentrations for 10-year-old (mean age all bears) female bears were lowest in Scandinavia (0.4 μg/L; 0.2–0.9 μg/L) but confidence intervals overlapped with Gates NP (1.0 μg/L; 0.5–1.9 μg/L) and Clark NP (1.1 μg/L; 0.5–2.4 μg/L) (Fig. 1a). Katmai NP had the highest predicted blood As concentrations (3.3 μg/L; 1.2–6.7 μg/L) but had overlapping confidence intervals with the other Alaskan areas (Fig. 1). We found that As blood concentration decreased with increasing age in males and increased with increasing age in females. In Gates NP, a five-year-old (25% percentile all bears) female had a predicted As blood concentration of 0.8 μg/L (0.5–1.2 μg/L) and a male had a concentration of 1.0 μg/L (0.8–1.5 μg/L). At 13-years of age (75% percentile), the model predicted 1.1 μg/L (0.4–3.2 μg/L) for females and 0.7 μg/L (0.2–2.0 μg/L) for males, respectively.

3.2. Cadmium

The best model contained sample area and age (Table SM 1). The

Table 2
Mean, standard deviation (SD), range and sample size (N) of blood concentrations for arsenic (As), cadmium (Cd), mercury (Hg) and lead (Pb) in μg/L of 72 brown bears (*Ursus arctos*) sampled between 2010 and 2021 in Scandinavia and national parks and preserves (NP) in Alaska, USA.

Element	Study Area	Mean	SD	Range	N
As	Gates NP	1.1	0.9	<0.15–3.1	18
	Katmai NP	3.6	2.9	0.7–10.6	12
	Clark NP	1.2	1.7	0.3–6.2	12
	Scandinavia	0.4	0.5	<0.15–2.7	30
Cd	Gates NP	0.4	0.3	0.1–1.0	18
	Katmai NP	0.1	0.1	0.1–0.3	12
	Clark NP	0.2	0.1	0.1–0.4	12
	Scandinavia	0.3	0.2	0.1–0.8	30
Hg	Gates NP	15.9	19.5	1.5–43.5	18
	Katmai NP	19.7	10.5	4.5–39.9	12
	Clark NP	40.1	32.6	5.6–119.2	12
	Scandinavia	1.4	0.7	0.5–4.0	30
Pb	Gates NP	38.4	32.3	4.7–138.6	18
	Katmai NP	10.0	8.9	3.4–37.1	12
	Clark NP	16.7	13.2	2.7–52.5	12
	Scandinavia	83.9	33.1	24.6–167.0	30

Table 3
Model averaged estimates (log scale) of a generalized linear model predicting blood concentrations for arsenic (As) and lead (Pb) in blood samples from brown bears *(Ursus arctos)* based on a sampling area in Scandinavia (reference level) and three national park and preserves (NP) in Alaska, sex of the bear and age at sampling between 2010 and 2021. All values are on log scale.

	Variable	Estimate	95% Confidence Interval	
			Lower	Upper
As	Scandinavia	−0.646	−1.471	0.180
	Gates NP	0.905	0.256	1.555
	Clark NP	1.014	0.237	1.791
	Katmai NP	2.111	1.389	2.833
	Males	−0.692	−1.954	0.570
	Age	−0.048	−0.130	0.035
	Males × Age	0.090	−0.005	0.184
Pb	Scandinavia	4.350	4.001	4.698
	Gates NP	−1.123	−1.612	−0.635
	Clark NP	−1.990	−2.571	−1.408
	Katmai NP	−2.430	−2.929	−1.930
	Sex Males	−0.157	−0.528	0.214
	Age	0.033	−0.005	0.070

model predicted that the Cd blood concentration of a 10-year-old bear (mean age all bears) was significantly higher in Scandinavia (0.4 μg/L; 0.3–0.5 μg/L) and Gates NP (0.3 μg/L; 0.2–0.4 μg/L) compared to Clark NP (0.2 μg/L; 0.1–0.2 μg/L) and Katmai NP (0.1 μg/L; 0.1–0.2 μg/L) (Fig. 1b). Predictions of Cd concentrations increased with increasing age (Table 4). For example, the model predicted a Cd blood concentration of 0.2 μg/L (0.2–0.3 μg/L) for a five-year-old bear and 0.3 μg/L (0.3–0.4 μg/L) for a 13-year-old bear in Gates NP. Models with sex of the bear were not selected (Table SM 1).

3.3. Mercury

The best model for Hg contained sample area, body mass, and sex (Table SM 1). Predicted blood Hg concentrations for 10-year-old bears were lowest in Scandinavia (1.5 μg/L; 1.1–2.0 μg/L). Within the Alaskan study areas, predictions for Clark NP where highest and confidence intervals did not overlap with any other area (47.8 μg/L; 31.2–73.5 μg/L) (Fig. 1c). Blood Hg concentration did not differ significantly between Gates NP (18.2 μg/L; 13.0–25.4 μg/L) and Katmai NP (20.5 μg/L; 13.9–30.2 μg/L). Mercury blood concentrations decreased with age (Table 4): For a five-year-old bear in Gates NP the model predicted a blood Hg concentration of 22.7 μg/L (14.7–34.9 μg/L) and for a 13-year-old 15.9 μg/L (11.6–21.8 μg/L).

3.4. Lead

For Pb, two models were equivalent based on AICc scores: The top model contained sample area and age and the second-best model also included sex (Table SM1). The model averaged predictions (Table 4) revealed a significantly higher Pb blood concentration of a 10-year-old female bear in Scandinavia (107.3 μg/L; 75.7–152.0 μg/L) than in the Alaskan areas. Within the Alaskan areas, Gates NP bears had significantly higher Pb concentration (34.9 μ/L; 21.4 μ/L – 56.9 μg/L) than Clark NP (14.7 μg/L; 8.2–26.2 μg/L) and Katmai NP (9.5 μg/L; 5.7–15.5 μg/L). Estimated 95% CI's for the Clark NP samples overlapped with the Katmai NP samples (Fig. 1d). Predictions of Pb concentrations increased with increasing age (Table 3) from 29.6 μg/L (23.8–35.7 μg/L) in a five-

Table 4
Model estimates (log scale) of a generalized linear model predicting blood concentrations for cadmium (Cd) and mercury (Hg) in blood samples from brown bears *(Ursus arctos)* based on sampling area in Scandinavia (reference level) and three national park and preserves (NP) in Alaska and age at sampling between 2010 and 2021.

	Variable	Estimate	Standard Error	T-value	P-value
Cd	Scandinavia	−1.449	0.113	−12.769	<0.001
	Gates NP	−0.267	0.162	−1.653	0.103
	Clark NP	−0.806	0.191	−4.212	<0.001
	Katmai NP	−1.146	0.166	−6.925	<0.001
	Age	0.050	0.012	4.055	<0.001
Hg	Scandinavia	0.833	0.169	4.926	<0.001
	Gates NP	2.511	0.241	10.415	<0.001
	Clark NP	3.479	0.285	12.194	<0.001
	Katmai NP	2.631	0.247	10.664	<0.001
	Age	−0.044	0.018	−2.426	0.018

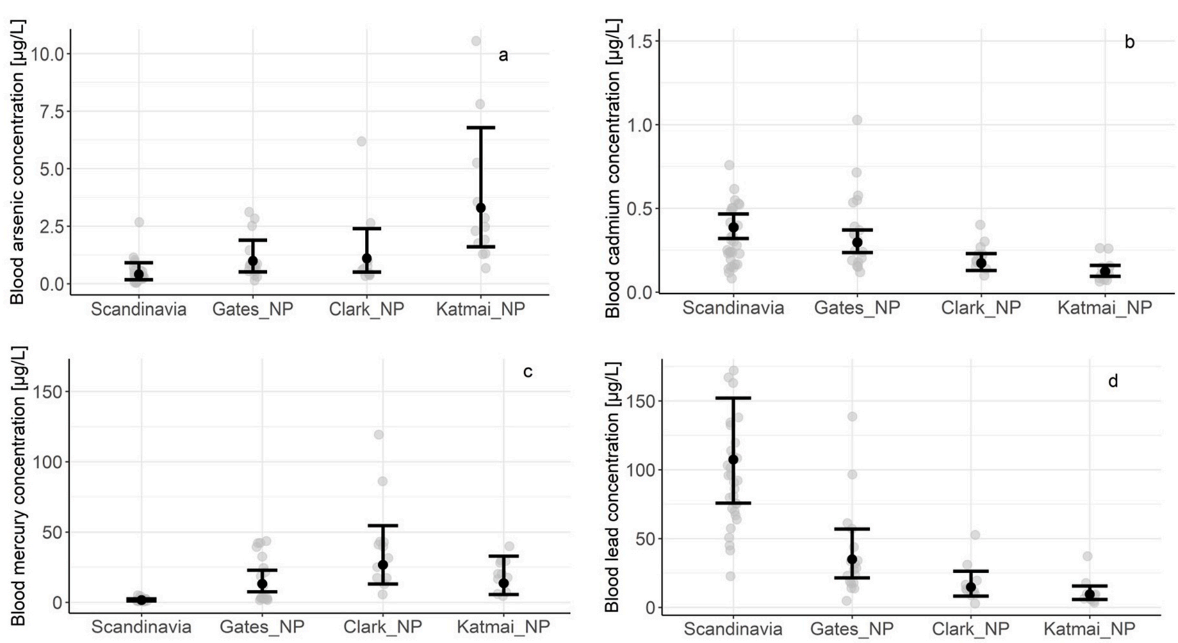

Fig. 1. Concentrations of arsenic (a), cadmium (b), mercury (c), and lead (d) in blood from brown bears *(Ursus arctos)* (μg/L, black dots) in south-central Scandinavia and three national park and preserves (NP) in Alaska, USA, predicted by generalized linear models including sample area, sex, and age of the bears as fixed factors. Error bars indicate 95% confidence intervals for the predictions. The raw data, collected between 2010 and 2021, are displayed as grey dots.

year-old to 38.5 (27.9–62.3 μg/L) in a 13-year-old female bear from Gates NP. Females had higher estimates than males; for example, in the Gates NP, a 10-year-old male had a predicted Pb blood concentration of 29.8 μg/L (20.6–43.2 μg/L), 5.1 μg/L lower than a female (34.9 μg/L; 24.1–50.6 μg/L).

3.5. Heavy metal (loids) and diet

Dietary proportion estimates in combination with blood HM concentrations were available for 17 Gates NP bears. The MixSIAR outputs suggests two distinct groups: Bears using predominantly vegetation-based resources and bears focusing on salmon, represented as marine protein. Nine bears used vegetation as their main dietary resource, and the proportional dietary estimates were >75% vegetation, with 95% credible intervals entirely >50% (Fig. 2). In the salmon group, for five bears, marine protein was estimated to comprise >75% of the diet, with 95% credible intervals entirely >50%, and one bear with an estimate of 68% (44–82%) marine protein in the diet. For the remaining two bears, the estimates where close to 50%, likely combining both sources. For all bears, the model estimated the terrestrial protein source contribution to <4% with upper 95% credible intervals <25%. Bears primarily feeding on vegetation had significantly higher As blood concentrations (1.6 μg/L, SE: 0.37 μg/L, $P < 0.05$) than bears feeding on salmon (0.6 μg/L, SE: 0.17 μg/L). No significant differences in Cd blood concentration were found between bears that fed primarily on either salmon or vegetation. The estimated mean Hg blood concentration of bears that had >50% vegetation in their diet was 2.7 μg/L (SE: 0.2 μg/L) but blood Hg concentration were 11 times higher (31.7 μg/L, SE: 3.5 μg/L, $P < 0.001$) for bears with >50% salmon in their diet. In comparison, bears with >50% vegetation in their diet had triple the blood Pb concentration (56.3 μg/L, SE: 13.5 μg/L, $P = 0.016$) than bears that primarily fed on salmon (18.6 μg/L, SE: 5.4 μg/L).

4. Discussion

We report blood concentrations of four non-essential elements in four different brown bear populations in Alaska and Scandinavia. While Scandinavian bears were exposed to a larger human footprint and had higher concentrations of Cd and Pb, they had lower concentrations of As and Hg. Bears in the Gates NP, the most remote of the Alaskan areas, had the highest levels of Cd and Pb. Contrary to hypothesis *i*), it suggests that local human footprint is a poor predictor of HM exposure and intake. Bears with a marine protein-based diet had higher Hg but lower Pb concentrations compared to bears with high vegetation levels in their diet, supporting our hypothesis *ii*) connecting diet to heavy metal exposure. Bears in areas with a high level of marine proteins in the diet had higher As concentrations. However, in Gates NP, bears feeding primarily on salmon had lower As concentrations than bears feeding primarily on vegetation. No relationship between diet and Cd blood concentration was found in Gates NP. Hypothesis *iii*), suggesting increasing whole blood HM concentrations with age, was supported for Cd, Pb and As in females, whereas concentrations of As in males and Hg decreased with age.

HM are distributed into our study areas via atmospheric distribution, river run-off, and ocean currents, as well as via upstream migrating anadromous salmon. Arsenic, Cd, Hg and Pb are contaminants that cycle in the environment for long periods, before they finally are deposited in lake or ocean sediments (AMAP, 2005). For example, 60% of the annual emitted Hg are re-emissions, depositions from anthropogenic sources that are mobilized primarily due to natural processes such as wildfires (Dastoor et al., 2022). There is strong evidence that the rapid environmental changes, due to global warming, will alter contaminant pathways and cycles in unexpected and abrupt ways in the Arctic and Sub-Arctic (Macdonald et al., 2005). Increased wildfire activity, erosion, melting ice and thawing permafrost are processes that increase with global warming and have high potential for contaminant re-emission (AMAP, 2005; Chételat et al., 2022b). From a One Health perspective, we found that bears were exposed to HM likely by food

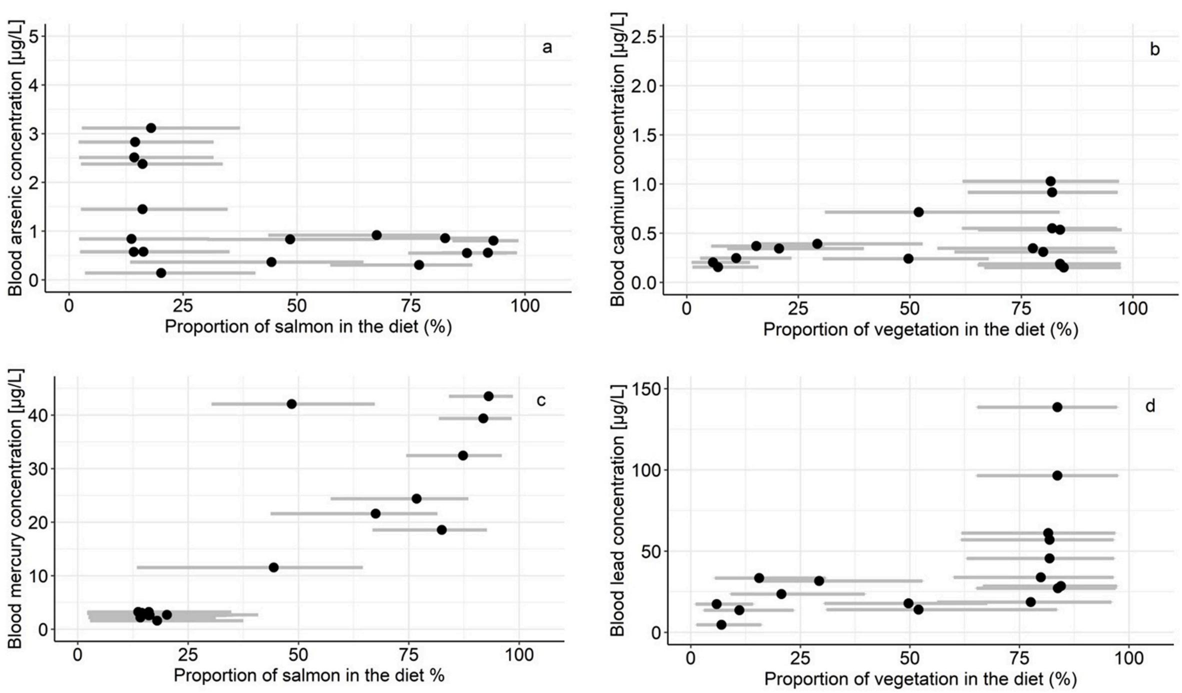

Fig. 2. Proportion of salmon (a and c) and vegetation (b and d) in the diet (black dots and grey 95% credible intervals) of 17 brown bears from the Gates of the Arctic National Park and Preserve, Alaska, USA, in relation to blood concentrations of arsenic (As), cadmium (Cd) mercury (Hg) and lead (Pb). Dietary estimates by (Mangipane et al., 2020) were based on C/N isotopes in hair; both hair and blood samples were collected in 2016. The proportion of terrestrial protein intake was estimated to be <4% for all individuals and is not shown.

items such as salmon and berries, that also are highly appreciated by humans. Protected areas, such as the national parks in Alaska, are supporting a healthy, diverse and resilient environment and although they are protected from direct human impact, they can still be vulnerable to pollution. We suggest extending contaminant monitoring in remote terrestrial ecosystems and to increase efforts to reduce both direct and re-emissions of HM.

Bears in Scandinavia feed exclusively on terrestrial resources (Stenset et al., 2016) and had the highest Pb concentrations but the lowest Hg blood concentrations. Bears in Katmai NP feed predominantly on a marine diet, consisting of migrating salmon in fresh water and flounders and clams along the seashore. Similarly, bears from Clark NP feed primarily on salmon (Rogers et al., 2020). Both Alaskan populations with a marine based diet had low Pb blood concentrations and high Hg concentrations. In comparison, bears from Gates NP either feed on spawning salmon and have high Hg and low Pb, or feed on upland terrestrial vegetation (Mangipane et al., 2020) and have low Hg with some individuals having elevated Pb blood concentrations. Besides in Gates NP, Cd concentrations were higher in Scandinavia than in Alaska, generally, reflecting the higher reported background levels there (AMAP, 2005). We suggest that HM exposure on brown bears is connected to the population typical food sources and the regional background concentrations, except for Hg, for which we suggest that background concentrations are less relevant and that migrating salmon are the dominant source of exposure.

In general, there are two ways to contextualize HM blood concentrations: The comparison to benchmark dose limits (BML) for the onset of toxicological effects or to compare individuals to the population exposure to detect deviating concentrations. Knowledge about the onset of sublethal toxicological effects of HM on bears, and for terrestrial top predators in general, is very limited (Rodríguez-Jorquera et al., 2017). We chose here to compare four populations to each other and discuss deviating results in context of each population's typical diet and human impact on the landscape level.

In bears, As concentrations were higher in Alaska than Scandinavia and highest in Katmai NP. Bears in Katmai NP also consume marine sources other than salmon. The intake of seafood has been linked to high As concentrations in human urinary samples, however in the organic form, which is less toxic and readily excreted (ATSDR, 2020). We quantified the total As blood concentration and were thus unable to distinguish between different forms of As. Only blood As concentrations in bears from Katmai NP reached a predicted mean (3.3 μg/L) which is above the lower detection limit (2.37 μg/L) of the study by Lazarus et al. (2020) on bears in Poland and Croatia, where most bears tested below that concentration. We found decreasing concentrations in females but not in males; however, effect sizes were small and must be interpreted with care. The same care should be taken when comparing between species: For example, brown rats *(Rattus norvegicus)* can bind 15–30 times more As (in the form of inorganic arsenite) to red blood cells than humans which in turn are more sensitive to As compared to other mammals (Lu et al., 2004).

Predicted Cd blood concentrations of Scandinavian bears (0.4 μg/L) were higher than in south-eastern European bears (0.29 μg/L) (Lazarus et al., 2020), which, in turn, were similar to bears from Gates NP (0.3 μg/L). This is counterintuitive, because there are higher Cd (and Pb) background concentrations in top soils in Croatia and Poland (0.10–0.38 mg/kg Cd) compared to Scandinavia (0.03–0.08 mg/kg Cd) (Salminen et al., 2005) and most likely also compared to the very remote Gates NP. This suggest that bears in Scandinavia, and to some extent Gates NP, accumulate more Cd and Pb from their environment compared to bears in Croatia and Poland. There are large differences in many aspects between these populations, but the Scandinavian and the Gates NP bears share a hyperphagic diet that is based on berries, which is different compared to bears from south-eastern Europe where hard mast dominates the fall diet (Lazarus et al., 2017; Mangipane et al., 2020; Stenset et al., 2016). Also, the predicted concentrations in samples from Clark NP and Katmai NP are lower (0.2 μg/L and 0.1 μg/L) and those bears have more marine food resources. For all four elements studied here, diet is the most likely exposure route. Cd is assumed to bioaccumulate in organisms (Järup, 2003; Zhang and Reynolds, 2019), which is supported by our result of higher Cd blood concentrations in older bears, similar to bears from Croatia where renal tissue Cd concentrations increased with age (Lazarus et al., 2017).

Mercury has biomagnification properties, increasing exposure risk in top level consumers, especially in the arctic marine system. The top 100 m of the Earth's oceans are a significant Hg reservoir and gaseous evasion of elemental Hg is restricted by arctic sea ice while advection of Hg from the oceans from the south is ongoing (AMAP, 2005). Although bears in Katmai NP also feed on other marine sources, migrating salmon are the main marine food resource for brown bears in Alaska. All five Pacific salmon species occurring in Alaska are fast growing, not top-trophic level consumers, and are considered the "best choice" for human consumption based on their low Hg concentrations (Bridges et al., 2020). However, the relationship between increased Hg blood concentrations in brown bears from the Gates NP and their $\delta^{13}C$ values indicating marine dominated diet, suggests salmon as the main source of Hg. The Hg blood concentrations we found in brown bears are also closer to polar bears than we expected. Knott et al. (2011, 2012) reported blood Hg mean concentrations of polar bears from the Bering Sea grouped by year, cohort, and sex, ranging from 39.05 μg/kg wet weight (ww) (males in 2007) to 74.62 μg/kg ww (both sexes, <4 years old in 2005). The mean Hg blood concentration from Clark NP brown bears (this study) was 40.1 μg/L (consider 1 L blood ~ 0.9 kg blood). Maximum measured concentrations in polar bears are two-fold higher than in brown bears and only in two samples from Clark NP blood Hg concentrations of >50 μg/L were measured. Our model estimated a tendency of decreasing Hg concentration with age. Visual interpretation of the measured Hg blood concentrations in relation of age of the Alaskan brown bears (not shown), indicate that the highest concentrations were measured in individuals between 7 and 13 years of age. This coincides with the age bears in these populations reach asymptotic body size (Hilderbrand et al., 2018). Such young but full-sized bears might be more successful in competing for good fishing spots, getting exposed to more Hg from salmon, while younger and older bears need to feed more on alternative source such as vegetation in addition to salmon.

Previous studies on Pb exposure in brown and American black bears proposed that Pb in their diet either come from a mixture of environmental concentrations and Pb-based ammunition in ungulate carcasses discarded by hunters (Brown et al., 2022; Fuchs et al., 2021; Lazarus et al., 2018; Rogers et al., 2012). In the Scandinavian study area, bears are likely exposed to both sources (Brown et al., 2023), but recreational hunting is prohibited and subsistence hunting occurs only at very low levels in the Alaska national parks. This suggests that environmental Pb is likely the primary source in the Alaskan study areas. The minimum blood Pb concentrations found in bears from national parks in Alaska (<5 μg/L) are below current published values for the species. Brown bears in the greater Yellowstone area have blood Pb concentrations of ≥11 μg/L (Rogers et al., 2012). In mammals, Pb bioaccumulates in bones with a half-life of 10–30 years (Andreani et al., 2019; Hryhorczuk et al., 1985; Rabinowitz et al., 1976). Lead stored in bones becomes an endogenous source during periods of nutritional stress and increased bone turnover (Bellinger et al., 2013), for example, during hibernation, pregnancy and lactation. In previous work in Scandinavia, we found all bears to have increased blood Pb concentrations and had concerns that hibernation alters Pb kinetics in bears such that the comparison of blood Pb concentrations to other mammals might not be valid (Boesen et al., 2019; Fuchs et al., 2021). However, based on the Pb blood concentrations found in the Alaskan samples, especially the high variation within Gates NP, suggests that blood is an adequate tissue to quantify Pb in brown bears. The bears with a plant-based diet in Gates NP had blood Pb concentrations >50 μg/L, which is consistent with the higher Pb concentrations reported in bears from Scandinavia where most bears also rely on vegetation during the hyperphagic period (Stenset et al., 2016). We found females to have higher Pb blood concentrations compared to males which is in line with previous work, where blood Pb concentrations significantly increased with lactation, but not age (Fuchs et al., 2021). Sex differences were also found in American black bears where Pb concentration in teeth was slightly higher in females and increased with age in both sexes (Brown et al., 2022). This highlights that blood Pb concentrations need to be interpreted with care; lower blood Pb in males is in part a result of a different physiology and not necessarily based on a different exposure.

Scandinavia has been exposed to extensive aerial Pb depositions emitted from Pb-enriched gasoline, (von Storch et al., 2003). At peak use of Pb in gasoline at the end of the 1970's, background Pb concentrations in Scandinavia were increased by an order of magnitude (Renberg et al., 2001). Since then, both deposition and concentrations of Pb in humans have decreased by > 90% (Pacyna et al., 2009; Skerfving et al., 2015). The atmospheric deposition of Cd in Scandinavia follows a similar trend. Aerial transport distance of Cd might be shorter compared to Pb, thus deposition is closer to sources resulting in a decreasing south to north exposure gradient of Cd in Scandinavia (AMAP, 2005). The Scandinavian Peninsula is closer to large combustions than Alaska and the weather conditions facilitate the aerial transport from central Europe to the north, unlike in Alaska (AMAP, 2005; von Storch et al., 2003). Nevertheless, a recent study from western Canada found that organisms, including plants, fish and mammals. Accumulate Pb emitted by industrial activities (Chételat et al., 2022a).

4.1. Conclusion

The human footprint was a poor predictor for HM exposure. The human influence on the boreal forest and tundra is considered low or very low in more than 50% of these biomes' global extent, however, indices of the human footprint are calculated based on local human infrastructure and land use change but do not consider pollution from distant sources (Keys et al., 2021; Riggio et al., 2020). Sub-arctic regions are susceptible to HM pollution from aerial depositions from entire Europe and marine currents in the Pacific, but also ongoing natural processes such as erosion by glaciers and rivers mobilize HM (Chételat et al., 2022a, 2022b). Natural processes releasing HM are altered by human activities, for example, draining of wetland mobilizes Hg in Scandinavia (AMAP, 2005). Global climate warming increases glacier run off, erosion and thawing permafrost potentially increasing HM mobilization whereas declining sea ice might allow increased Hg escape from the arctic marine system (AMAP, 2005; Macdonald et al., 2005). Establishing baseline HM exposure and continuous monitoring of HM exposure are needed, even in wilderness areas.

Author contributions

Boris Fuchs: Conceptualization, Methodology, Formal analysis, Investigation, Data Curation, Writing - Original Draft. **Kyle Joly:** Investigation, Resources, Data Curation, Writing - Review & Editing. **Grant V. Hilderbrand:** Investigation, Resources, Writing - Review & Editing, Supervision. **Alina L. Evans:** Conceptualization, Investigation, Writing - Review & Editing. **Ilia Rhoduskin:** Investigation, Resources, Writing - Review & Editing. **Lindsey S. Mangipane** Methodology, Formal analysis, Investigation, Resources, Data Curation. **Buck A. Mangipane:** Methodology, Formal analysis, Investigation, Resources, Data Curation. **David D. Gustine:** Investigation, Resources, Data Curation**. Andreas Zedrosser:** Writing - Review & Editing, Supervision. **Ludovick Brown:** Validation, Writing - Review & Editing. **Jon M. Arnemo:** Conceptualization, Investigation, Resources, Writing - Review & Editing, Supervision, Funding acquisition.

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Ethics and animal welfare

Capture and handling procedures in Alaska were approved by the US National Park Service (AKR_KATM_Hilderbrand_BrownBear_2014, AKR_LACL_Mangipane_BrownBear_2014, AKR_GAAR_Gustine_GrizzlyBear_2014) and the US Geological Survey (2014–01, 2015–04, 2015–06) Animal Care and Use Committee. In Scandinavia, captures were approved by the Swedish Ethical Committee on Animal Research, Uppsala, Sweden (C18/15) and the Swedish Environmental Protection Agency, Stockholm, Sweden (NV-00741-18). Shipping of whole blood samples from brown bears *(Ursus arctos)* from the USA to Sweden followed the CITES regulations (export permit #19US15593D/9, import permit #4.10.18–12752/2019).

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: I. Rodushkin is employed by ALS Global AB in Luleå, Sweden.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envres.2023.115952.

References

Ali, H., Khan, E., 2019. Trophic transfer, bioaccumulation, and biomagnification of non-essential hazardous heavy metals and metalloids in food chains/webs—concepts and implications for wildlife and human health. Hum. Ecol. Risk Assess. 25, 1353–1376. https://doi.org/10.1080/10807039.2018.1469398.

Amap, 2005. AMAP Assessment 2002: Heavy Metals in the Arctic, Arctic Monitoring and Assessment Programme (AMAP). Oslo, Norway.

Andreani, G., Cannavacciuolo, A., Menotta, S., Spallucci, V., Fedrizzi, G., Carpenè, E., Isani, G., 2019. Environmental exposure to non-essential trace elements in two bat species from urbanised (Tadarida teniotis) and open land (Miniopterus schreibersii) areas in Italy. Environ. Pollut. 254, 113034 https://doi.org/10.1016/j.envpol.2019.113034.

Arnemo, J.M., Fuchs, B., Sonne, C., Stokke, S., 2022. Hunting with lead ammunition: a one health perspective. In: Tryland, M. (Ed.), Arctic One Health: Challenges for Northern Animals and People. Springer International Publishing, Cham, pp. 439–468. https://doi.org/10.1007/978-3-030-87853-5_21.

Atkinson, S.C., Williams, B.A., 2020. Human Footprint 2000-2013 [WWW Document]. URL. https://github.com/scabecks/humanfootprint_2000-2013.

ATSDR, 2022. Toxicological Profile for Mercury (Draft for Public Comment). Agency for Toxic Substances and Disease Registry (ATSDR), U.S. Department of Health and Human Services, Public Health Service, Atlanta.

ATSDR, 2020. Toxicological Profile for Lead. Agency for Toxic Substances and Disease Registry (ATSDR), U.S. Department of Health and Human Services, Public Health Service, Atlanta. https://doi.org/10.15620/cdc:95222.

ATSDR, 2015. Toxicological Profile for Cadmium. Agency for Toxic Substances and Disease Registry (ATSDR), U.S. Department of Health and Human Services, Public Health Service, Atlanta.

Atwell, L., Hobson, K.A., Welch, H.E., 1998. Biomagnification and bioaccumulation of mercury in an arctic marine food web: insights from stable nitrogen isotope analysis. Can. J. Fish. Aquat. Sci. 55, 1114–1121. https://doi.org/10.1139/f98-001.

Bakka, M., 2020. Blood concentrations of lead (Pb), mercury (Hg) and cadmium (Cd) in Scandinavian and Alaskan brown bears (Ursus arctos) (Master Thesis). Evenstad, Norway. https://brage.inn.no/inn-xmlui/bitstream/handle/11250/2660562/Haugen.pdf?sequence=1&isAllowed=y.

Bellinger, D.C., Burger, J., Cade, T.J., Cory-Slechta, D.A., Finkelstein, M., Hu, H., Kosnett, M., Landrigan, P.J., Lanphear, B., Pokras, M.A., Redig, P.T., Rideout, B.A., Silbergeld, E., Wright, R., Smith, D.R., 2013. Health risks from lead-based ammunition in the environment. Environ. Health Perspect. 121, a178–a179. https://doi.org/10.1289/ehp.1306945.

Bjerregaard, P., Andersen, C.B.I., Andersen, O., 2022. Chapter 26 - ecotoxicology of metals—sources, transport, and effects on the ecosystem. In: Nordberg, G.F., Costa, M. (Eds.), Handbook on the Toxicology of Metals, fifth ed. Academic Press, pp. 593–627. https://doi.org/10.1016/B978-0-12-823292-7.00016-4.

Bocharova, N., Treu, G., Czirják, G.Á., Krone, O., Stefanski, V., Wibbelt, G., Unnsteinsdóttir, E.R., Hersteinsson, P., Schares, G., Doronina, L., Goltsman, M., Greenwood, A.D., 2013. Correlates between feeding ecology and mercury levels in historical and modern arctic foxes (Vulpes lagopus). PLoS One 8, e60879. https://doi.org/10.1371/journal.pone.0060879.

Boesen, A.H., Thiel, A., Fuchs, B., Evans, A.L., Bertelsen, M.F., Rodushkin, I., Arnemo, J. M., 2019. Assessment of the LeadCare® plus for use on scandinavian Brown bears (Ursus arctos). Front. Vet. Sci. 6 https://doi.org/10.3389/fvets.2019.00285.

Bojarska, K., Selva, N., 2012. Spatial patterns in brown bear Ursus arctos diet: the role of geographical and environmental factors. Mamm Rev. 42, 120–143. https://doi.org/10.1111/j.1365-2907.2011.00192.x.

Bridges, K.N., Furin, C.G., Gerlach, R.F., 2020. Subsistence fish consumption in rural Alaska: using regional monitoring data to evaluate risk and bioavailability of dietary methylmercury. Sci. Total Environ. 736, 139676 https://doi.org/10.1016/j.scitotenv.2020.139676.

Brown, L., Fuchs, B., Arnemo, J.M., Kindberg, J., Rodushkin, I., Zedrosser, A., Pelletier, F., 2023. Lead exposure in brown bears is linked to environmental levels and the distribution of moose kills. Sci. Total Environ., 162099 https://doi.org/10.1016/j.scitotenv.2023.162099.

Brown, L., Rosabal, M., Dussault, C., Arnemo, J.M., Fuchs, B., Zedrosser, A., Pelletier, F., 2022. Lead exposure in American black bears increases with age and big game harvest density. Environ. Pollut., 120427 https://doi.org/10.1016/j.envpol.2022.120427.

Buttke, D.E., Decker, D.J., Wild, M.A., 2015. The role of One Health in wildlife conservation: a challange and opportunity. J. Wildl. Dis. 51, 1–8. https://doi.org/10.7589/2014-01-004.

Cardona-Marek, T., Knott, K.K., Meyer, B.E., O'Hara, T.M., 2009. Mercury concentrations in Southern Beaufort Sea polar bears: variation based on stable isotopes of carbon and nitrogen. Environ. Toxicol. Chem. 28, 1416–1424. https://doi.org/10.1897/08-557.1.

Chételat, J., Cousens, B., Hebert, C.E., Jung, T.S., Mundy, L., Thomas, P.J., Zhang, S., 2022a. Isotopic evidence for bioaccumulation of aerosol lead in fish and wildlife of western Canada. Environ. Pollut. 302, 119074 https://doi.org/10.1016/j.envpol.2022.119074.

Chételat, J., McKinney, M.A., Amyot, M., Dastoor, A., Douglas, T.A., Heimbürger-Boavida, L.-E., Kirk, J., Kahilainen, K.K., Outridge, P.M., Pelletier, N., Skov, H., St Pierre, K., Vuorenmaa, J., Wang, F., 2022b. Climate change and mercury in the Arctic: abiotic interactions. Sci. Total Environ. 824, 153715 https://doi.org/10.1016/j.scitotenv.2022.153715.

Dastoor, A., Wilson, S.J., Travnikov, O., Ryjkov, A., Angot, H., Christensen, J.H., Steenhuisen, F., Muntean, M., 2022. Arctic atmospheric mercury: sources and changes. Sci. Total Environ. 839, 156213 https://doi.org/10.1016/j.scitotenv.2022.156213.

Ecke, F., Singh, N.J., Arnemo, J.M., Bignert, A., Helander, B., Berglund, Å.M.M., Borg, H., Bröjer, C., Holm, K., Lanzone, M., Miller, T., Nordström, Å., Räikkönen, J., Rodushkin, I., Ågren, E., Hörnfeldt, B., 2017. Sublethal lead exposure alters movement behavior in free-ranging golden eagles. Environ. Sci. Technol. 51, 5729–5736. https://doi.org/10.1021/acs.est.6b06024.

Erlenbach, J.A., 2020. Nutritional and Landscape Ecology of Brown Bears (Ursus arctos). Ph.D. Thesis). Washington State University, Pullman WA, USA.

Ferguson, S.H., McLoughlin, P.D., 2000. Effect of energy availability, seasonality, and geographic range on brown bear life history. Ecography 23, 193–200. https://doi.org/10.1111/j.1600-0587.2000.tb00275.x.

Finkelstein, M.E., Doak, D.F., George, D., Burnett, J., Brandt, J., Church, M., Grantham, J., Smith, D.R., 2012. Lead poisoning and the deceptive recovery of the critically endangered California condor. Proc. Natl. Acad. Sci. USA 109, 11449–11454. https://doi.org/10.1073/pnas.1203141109.

Fuchs, B., Thiel, A., Zedrosser, A., Brown, L., Hydeskov, H.B., Rodushkin, I., Evans, A.L., Boesen, A.H., Græsli, A.R., Kindberg, J., Arnemo, J.M., 2021. High concentrations of lead (Pb) in blood and milk of free-ranging brown bears (Ursus arctos) in Scandinavia. Environ. Pollut. 287, 117595 https://doi.org/10.1016/j.envpol.2021.117595.

Hall, M., Chen, Y., Ahsan, H., Slavkovich, V., van Geen, A., Parvez, F., Graziano, J., 2006. Blood arsenic as a biomarker of arsenic exposure: results from a prospective study. Toxicology 225, 225–233. https://doi.org/10.1016/j.tox.2006.06.010.

Hertel, A.G., Bischof, R., Langval, O., Mysterud, A., Kindberg, J., Swenson, J.E., Zedrosser, A., 2018. Berry production drives bottom–up effects on body mass and reproductive success in an omnivore. Oikos 127, 197–207. https://doi.org/10.1111/oik.04515.

Hilderbrand, G.V., Gustine, D.D., Mangipane, B.A., Joly, K., Leacock, W., Mangipane, L. S., Erlenbach, J.A., Sorum, M.S., Cameron, M.D., Belant, J.L., Cambier, T., 2018. Body size and lean mass of brown bears across and within four diverse ecosystems. J. Zool. 305, 53–62. https://doi.org/10.1111/jzo.12536.

Hilderbrand, G.V., Schwartz, C.C., Robbins, C.T., Jacoby, M.E., Hanley, T.A., Arthur, S. M., Servheen, C., 1999. The importance of meat, particularly salmon, to body size, population productivity, and conservation of North American brown bears. Can. J. Zool. 77, 132–138. https://doi.org/10.1139/z98-195.

Hryhorczuk, D.O., Rabinowitz, M.B., Hessl, S.M., Hoffman, D., Hogan, M.M., Mallin, K., Finch, H., Orris, P., Berman, E., 1985. Elimination kinetics of blood lead in workers with chronic lead intoxication. Am. J. Ind. Med. 8, 33–42. https://doi.org/10.1002/ajim.4700080105.

IRMA, 2022. National Park Service IRMA Portal (Integrated Resource Management Applications) [WWW Document]. URL. https://irma.nps.gov. accessed 8.March.22.

Järup, L., 2003. Hazards of heavy metal contamination. Br. Med. Bull. 68, 167–182. https://doi.org/10.1093/bmb/ldg032.

Johnson, S.P., Schindler, D.E., 2009. Trophic ecology of Pacific salmon (Oncorhynchus spp.) in the ocean: a synthesis of stable isotope research. Ecol. Res. 24, 855–863. https://doi.org/10.1007/s11284-008-0559-0.

Kavčič, I., Adamič, M., Kaczensky, P., Krofel, M., Kobal, M., Jerina, K., 2015. Fast food bears: brown bear diet in a human-dominated landscape with intensive supplemental feeding. Wildl. Biol. 21, 00855 https://doi.org/10.2981/wlb.00013.

Keys, P.W., Barnes, E.A., Carter, N.H., 2021. A machine-learning approach to human footprint index estimation with applications to sustainable development. Environ. Res. Lett. 16, 044061 https://doi.org/10.1088/1748-9326/abe00a.

Knott K., K., Boyd D., Ylitalo G., M., O'Hara T., M., 2011. Concentrations of mercury and polychlorinated biphenyls in blood of Southern Beaufort Sea polar bears (*Ursus maritimus*) during spring: variations with lipids and stable isotope ($\delta^{15}N$, $\delta^{13}C$) values. Can. J. Zool. 89 (11), 999–1012. https://doi.org/10.1139/z11-071.

Knott, K.K., Boyd, D., Ylitalo, G.M., O'Hara, T.M., 2012. Lactational transfer of mercury and polychlorinated biphenyls in polar bears. Chemosphere 88 (4), 395–402. https://doi.org/10.1016/j.chemosphere.2012.02.053. ISSN 0045-6535.

Koike, S., Kozakai, C., Nemoto, Y., Masaki, T., Yamazaki, K., Abe, S., Nakajima, A., Umemura, Y., Kaji, K., 2012. Effect of hard mast production on foraging and sex-specific behavior of the Asiatic black bear (Ursus thibetanus). Mamm. Stud. 37, 21–28. https://doi.org/10.3106/041.037.0103.

Länsstyrelserna, 2022. License hunting bears [WWW Document]. URL https://www.lansstyrelsen.se/dalarna/djur/jakt-och-vilt/licensjakt-efter-stora-rovdjur/licensjakt-bjorn.html. . accessed 10.3.22.

Länsstyrelserna, 2020. Algdata [WWW Document]. URL http://www.algdata.se/sv/pages/default.aspx. . accessed 8.19.20.

Larsson, P., Holmqvist, N., Stenroth, P., Berglund, O., Nyström, P., Granéli, W., 2007. Heavy metals and stable isotopes in a benthic omnivore in a trophic gradient of lakes. Environ. Sci. Technol. 41, 5973–5979. https://doi.org/10.1021/es0704838.

Lazarus, M., Orct, T., Reljić, S., Sedak, M., Bilandžić, N., Jurasović, J., Huber, Đ., 2018. Trace and macro elements in the femoral bone as indicators of long-term environmental exposure to toxic metals in European brown bear (Ursus arctos) from Croatia. Environ. Sci. Pollut. Res. 25, 21656–21670. https://doi.org/10.1007/s11356-018-2296-4.

Lazarus, M., Orct, T., Sergiel, A., Vranković, L., Marijić, V.F., Rašić, D., Reljić, S., Aladrović, J., Zwijacz-Kozica, T., Zięba, F., Jurasović, J., Erk, M., Maślak, R., Selva, N., Huber, Đ., 2020. Metal(loid) exposure assessment and biomarker responses in captive and free-ranging European brown bear (Ursus arctos). Environ. Res. 183, 109166 https://doi.org/10.1016/j.envres.2020.109166.

Lazarus, M., Sekovanić, A., Orct, T., Reljić, S., Kusak, J., Jurasović, J., Huber, Đ., 2017. Apex predatory mammals as bioindicator species in environmental monitoring of elements in Dinaric Alps (Croatia). Environ. Sci. Pollut. Res. 24, 23977–23991. https://doi.org/10.1007/s11356-017-0008-0.

Le Bourg, B., Kiszka, J.J., Bustamante, P., Heithaus, M.R., Jaquemet, S., Humber, F., 2019. Effect of body length, trophic position and habitat use on mercury concentrations of sharks from contrasted ecosystems in the southwestern Indian Ocean. Environ. Res. 169, 387–395. https://doi.org/10.1016/j.envres.2018.11.024.

Le Croizier, G., Schaal, G., Point, D., Le Loc'h, F., Machu, E., Fall, M., Munaron, J.-M., Boyé, A., Walter, P., Laë, R., Tito De Morais, L., 2019. Stable isotope analyses revealed the influence of foraging habitat on mercury accumulation in tropical coastal marine fish. Sci. Total Environ. 650, 2129–2140. https://doi.org/10.1016/j.scitotenv.2018.09.330.

Lu, M., Wang, H., Li, X.-F., Lu, X., Cullen, W.R., Arnold, L.L., Cohen, S.M., Le, X.C., 2004. Evidence of hemoglobin binding to arsenic as a basis for the accumulation of arsenic in rat blood. Chem. Res. Toxicol. 17, 1733–1742. https://doi.org/10.1021/tx049756s.

Macdonald, R.W., Harner, T., Fyfe, J., 2005. Recent climate change in the Arctic and its impact on contaminant pathways and interpretation of temporal trend data. Sources Occur. Trends Pathw. Contam. Arct. 342, 5–86. https://doi.org/10.1016/j.scitotenv.2004.12.059.

Mangipane, L.S., Belant, J.L., Hiller, T.L., Colvin, M.E., Gustine, D., Mangipane, B.A., Hilderbrand, G.V., 2018. Influences of landscape heterogeneity on home-range sizes of brown bears. Mamm. Biol. 88, 1–7. https://doi.org/10.1016/j.mambio.2017.09.002.

Mangipane, L.S., Lafferty, D.J.R., Joly, K., Sorum, M.S., Cameron, M.D., Belant, J.L., Hilderbrand, G.V., Gustine, D.D., 2020. Dietary plasticity and the importance of salmon to brown bear (Ursus arctos) body size and condition in a low Arctic ecosystem. Polar Biol. 43, 825–833. https://doi.org/10.1007/s00300-020-02690-7.

Martin, J., Basille, M., Van Moorter, B., Kindberg, J., Allainé, D., Swenson, J.E., 2010. Coping with human disturbance: spatial and temporal tactics of the brown bear (Ursus arctos). Can. J. Zool. 88, 875–883. https://doi.org/10.1139/Z10-053.

Mattisson, J., AndrÉN, H., Persson, J., SegerstrÖM, P., 2010. Effects of species behavior on global positioning system collar fix rates. J. Wildl. Manag. 74, 557–563. https://doi.org/10.2193/2009-157.

Mattson, G., 1993. A Laboratory Manual for Cementum Age Determination of Alaska Brown Bear First Premolar Teeth. Alaska Department of Fish and Game and Matson's Laboratory, Milltown, Montana.

Mazerolle, M.J., 2019. AICcmodavg: Model Selection and Multimodel Inference Based on (Q)AIC(c).

Mowat, G., Heard, D.C., 2006. Major components of grizzly bear diet across North America. Can. J. Zool. 84, 473–489. https://doi.org/10.1139/z06-016.

Noël, M., Spence, J., Harris, K.A., Robbins, C.T., Fortin, J.K., Ross, P.S., Christensen, J.R., 2014. Grizzly bear hair reveals toxic exposure to mercury through salmon consumption. Environ. Sci. Technol. 48, 7560–7567. https://doi.org/10.1021/es500631g.

Pacyna, J.M., Pacyna, E.G., Aas, W., 2009. Changes of emissions and atmospheric deposition of mercury, lead, and cadmium. Atmospheric Environ. - Fifty Years Endeav. 43, 117–127. https://doi.org/10.1016/j.atmosenv.2008.09.066.

R Core Team, 2022. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing.

Rabinowitz, M.B., Wetherill, G.W., Kopple, J.D., 1976. Kinetic analysis of lead metabolism in healthy humans 58, 260–270. https://doi.org/10.1172/jci108467.

Ramos, R., González-Solís, J., Forero, M.G., Moreno, R., Gómez-Díaz, E., Ruiz, X., Hobson, K.A., 2009. The influence of breeding colony and sex on mercury, selenium and lead levels and carbon and nitrogen stable isotope signatures in summer and winter feathers of Calonectris shearwaters. Oecologia 159, 345–354. https://doi.org/10.1007/s00442-008-1215-7.

Renberg, I., Bindler, R., Brännvall, M.-L., 2001. Using the historical atmospheric lead-deposition record as a chronological marker in sediment deposits in Europe. Holocene 11, 511–516. https://doi.org/10.1191/095968301680223468.

Riggio, J., Baillie, J.E.M., Brumby, S., Ellis, E., Kennedy, C.M., Oakleaf, J.R., Tait, A., Tepe, T., Theobald, D.M., Venter, O., Watson, J.E.M., Jacobson, A.P., 2020. Global human influence maps reveal clear opportunities in conserving Earth's remaining intact terrestrial ecosystems. Global Change Biol. 26, 4344–4356. https://doi.org/10.1111/gcb.15109.

Rodríguez-Jorquera, I.A., Vitale, N., Garner, L., Perez-Venegas, D.J., Galbán-Malagón, C. J., Duque-Wilckens, N., Toor, G.S., 2017. Contamination of the upper class: occurrence and effects of chemical pollutants in terrestrial top predators. Curr. Pollut. Rep. 3, 206–219. https://doi.org/10.1007/s40726-017-0061-9.

Rodushkin, I., Ödman, F., Olofsson, R., Axelsson, M.D., 2000. Determination of 60 elements in whole blood by sector field inductively coupled plasma mass spectrometry. J. Anal. At. Spectrom. 15, 937–944. https://doi.org/10.1039/B003561K.

Rogers, M.C., Hilderbrand, G.V., Gustine, D.D., Joly, K., Leacock, W.B., Mangipane, B.A., Welker, J.M., 2020. Splitting hairs: dietary niche breadth modelling using stable isotope analysis of a sequentially grown tissue. Isot. Environ. Health Stud. 1–12. https://doi.org/10.1080/10256016.2020.1787404.

Rogers, T.A., Bedrosian, B., Graham, J., Foresman, K.R., 2012. Lead exposure in large carnivores in the greater Yellowstone ecosystem: lead exposure in large carnivores. J. Wildl. Manag. 76, 575–582. https://doi.org/10.1002/jwmg.277.

Salminen, R., Batista, M., Bidovec, M., Demetriades, A., De Vivo, B., De Vos, W., Duris, M., Gilucis, A., Gregorauskiene, V., Halamić, J., others, 2005. Geochemical Atlas of Europe, Part 1, Background Information, Methodology and Maps.

Scheuhammer, A.M., Meyer, M.W., Sandheinrich, Mark B., Murray, Michael W., 2007. Effects of environmental methylmercury on the health of wild birds, mammals, and fish. Ambio 36, 12–18.

Silbergeld, E.K., 1991. Lead in Bone: Implications for Toxicology during Pregnancy and Lactation, vol. 91, pp. 63–70. https://doi.org/10.1289/ehp.919163.

Skerfving, S., Löfmark, L., Lundh, T., Mikoczy, Z., Strömberg, U., 2015. Late effects of low blood lead concentrations in children on school performance and cognitive functions. Neurotoxicology 49, 114–120. https://doi.org/10.1016/j.neuro.2015.05.009.

Söderberg, C., Rodushkin, I., Johansson, A., Kugelberg, F.C., 2023. Postmortem reference concentrations of 68 elements in blood and urine. Int. J. Leg. Med. https://doi.org/10.1007/s00414-023-02952-z.

Stenset, N.E., Lutnæs, P.N., Bjarnadóttir, V., Dahle, B., Fossum, K.H., Jigsved, P., Johansen, T., Neumann, W., Opseth, O., Rønning, O., Steyaert, S.M.J.G., Zedrosser, A., Brunberg, S., Swenson, J.E., 2016. Seasonal and annual variation in the diet of brown bears (Ursus arctos) in the boreal forest of southcentral Sweden. Wildl. Biol. 22, 107–116. https://doi.org/10.2981/wlb.00194, 10.

Stokke, S., Brainerd, S., Arnemo, J., 2017. Metal deposition of copper and lead bullets in moose harvested in fennoscandia. Wildl. Soc. Bull. https://doi.org/10.1002/wsb.731.

Swenson, J.E., Adamič, M., Huber, D., Stokke, S., 2007. Brown bear body mass and growth in northern and southern Europe. Oecologia 153, 37–47. https://doi.org/10.1007/s00442-007-0715-1.

Swenson, J.E., Jansson, A., Riig, R., Sandegren, F., 1999. Bears and ants: myrmecophagy by brown bears in central Scandinavia. Can. J. Zool. 77, 551–561. https://doi.org/10.1139/z99-004.

Tan, S.W., Meiller, J.C., Mahaffey, K.R., 2009. The endocrine effects of mercury in humans and wildlife. Crit. Rev. Toxicol. 39, 228–269.

von Storch, H., Costa-Cabral, M., Hagner, C., Feser, F., Pacyna, J., Pacyna, E., Kolb, S., 2003. Four decades of gasoline lead emissions and control policies in Europe: a retrospective assessment. Sci. Total Environ. 311, 151–176. https://doi.org/10.1016/S0048-9697(03)00051-2.

Williams, B.A., Venter, O., Allan, J.R., Atkinson, S.C., Rehbein, J.A., Ward, M., Di Marco, M., Grantham, H.S., Ervin, J., Goetz, S.J., Hansen, A.J., Jantz, P., Pillay, R., Rodríguez-Buriticá, S., Supples, C., Virnig, A.L.S., Watson, J.E.M., 2020. Change in terrestrial human footprint drives continued loss of intact ecosystems. One Earth 3, 371–382. https://doi.org/10.1016/j.oneear.2020.08.009.

Yaginuma-Sakurai, K., Murata, K., Iwai-Shimada, M., Nakai, K., Kurokawa, N., Tatsuta, N., Satoh, H., 2012. Hair-to-blood ratio and biological half-life of mercury: experimental study of methylmercury exposure through fish consumption in humans. J. Toxicol. Sci. 37, 123–130. https://doi.org/10.2131/jts.37.123.

Zhang, H., Reynolds, M., 2019. Cadmium exposure in living organisms: a short review. Sci. Total Environ. 678, 761–767. https://doi.org/10.1016/j.scitotenv.2019.04.395.