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# Mercury contamination and potential health risks to Arctic seabirds and shorebirds

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## **Abstract**

Since the last Arctic Monitoring and Assessment Programme (AMAP) effort to review biological effects of mercury (Hg) on Arctic biota in 2011 and 2018, there has been a considerable number of new Arctic bird studies. This review article provides contemporary Hg exposure and potential health risk for 36 Arctic seabird and shorebird species, representing a larger portion of the Arctic than during previous AMAP assessments now also including parts of the Russian Arctic. To assess risk to birds, we used Hg toxicity benchmarks established for blood and converted to egg, liver, and feather tissues. Several Arctic seabird populations showed Hg concentrations that exceeded toxicity benchmarks, with 50% of individual birds exceeding the “no adverse health effect” level. In particular, 5% of all studied birds were considered to be at moderate or higher risk to Hg toxicity. However, most seabirds (95%) were generally at lower risk to Hg toxicity. The highest Hg contamination was observed in seabirds breeding in the western Atlantic and Pacific Oceans. Most Arctic shorebirds exhibited low Hg concentrations, with approximately 45% of individuals categorized at no risk, 2.5% at high risk category, and no individual at severe risk. Although the majority Arctic-breeding seabirds and shorebirds appeared at lower risk to Hg toxicity, recent studies have reported deleterious effects of Hg on some pituitary hormones, genotoxicity, and reproductive performance. Adult survival appeared unaffected by Hg exposure, although long-term banding studies incorporating Hg are still limited. Although Hg contamination across the Arctic is considered low for most bird species, Hg in combination with other stressors, including other contaminants, diseases, parasites, and climate change, may still cause adverse effects. Future investigations on the global impact of Hg on Arctic birds should be conducted within a multi-stressor framework. This information helps to address Article 22 (Effectiveness Evaluation) of the Minamata Convention on Mercury as a global pollutant.

109 **Keywords**

110 Mercury, Arctic, birds, toxicity benchmarks, toxicological effects

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115 **1. Introduction**

116 Among contaminants of concern, mercury (Hg) is a non-essential trace element from both natural  
117 (e.g., volcanic eruptions, forest fires, biomass burning) and anthropogenic sources (e.g., fossil fuel  
118 combustion, mining, waste disposal and chemical production). Due to long-range transport by atmos-  
119 pheric, oceanic, and riverine pathways (AMAP 2021), the Arctic is considered a sink to atmospheric  
120 Hg deposition (Dastoor et al., 2022; Nerentorp et al., 2022; AMAP 2021). Mercury contamination has  
121 increased globally from anthropogenic inputs and has become a major concern in the Arctic (AMAP  
122 2021).

123 The toxic form of Hg, methylmercury (MeHg), bioaccumulates in organisms, biomagnifies throughout  
124 trophic food webs, and can have numerous detrimental effects on Arctic wildlife (Scheuhammer et al.,  
125 2007; Ackerman et al., 2016). Avian reproduction is especially sensitive to MeHg toxicity, with even  
126 low levels of exposure potentially leading to adverse effects (Wiener et al., 2003; Heinz et al., 2009).  
127 Aquatic birds typically have the highest exposures to environmental Hg contamination (Ackerman et  
128 al. 2016), although terrestrial birds, like riparian songbirds, may also bioaccumulate MeHg to poten-  
129 tially harmful levels (Cristol et al., 2008; Ackerman et al., 2019; Cristol and Evers, 2020). Within the  
130 Arctic, aquatic birds are primarily exposed to elevated levels of MeHg in pelagic environments  
131 (Provencher et al., 2014a; Braune et al., 2015; Peck et al., 2016; Burnham et al., 2018; Albert et al.,  
132 2019, 2021), coastal shorelines, and wetland foraging habitats (Hargreaves et al., 2011; McCloskey et  
133 al., 2013; Perkins et al., 2016; Sun et al., 2019).

134 Among aquatic birds, seabirds are long-lived species often at the top of food web chains, both leading  
135 them to exhibit some of the highest Hg concentrations observed in wildlife, making them particularly  
136 sensitive to the harmful effects of MeHg exposure. Hence, seabirds are commonly and efficiently used

137 as bio-indicators of the health of their environment (Elliott and Elliott, 2013). Originally centered on  
138 the Canadian Arctic (e.g., Braune et al. 2014a), the characterization of MeHg contamination in  
139 seabirds has been recently expanded to many other localities and species of the Arctic regions (e.g.,  
140 Albert et al., 2021).

141 Recently Hg data have also become available for another group of aquatic birds - shorebirds (Perkins  
142 et al., 2016; 2018; Pratte et al., 2020, Burnham et al., 2018). As long-lived species with widespread  
143 distributions across the Arctic during the breeding season, shorebirds may represent an ideal group for  
144 Arctic Hg exposure research. Shorebirds nest and forage within wetland habitats in the Arctic tundra  
145 and can occupy relatively high trophic levels. Because Hg research on terrestrial avian invertivores in  
146 the Arctic is limited (Scheuhammer et al., 2015), recent investigations of shorebird Hg concentrations  
147 fills an important knowledge gap (Perkins et al. 2016, Burnham et al. 2018).

148 The first objective of the present study was to provide contemporary (post-2000) information on Hg  
149 contamination and potential health risks for Arctic seabirds and shorebirds, updating the AMAP 2011  
150 Hg Assessment and the AMAP 2018 Effect Assessment (AMAP 2011, AMAP 2018, Dietz et al. 2013,  
151 2019). Therefore, we provide the most up-to-date risk assessment of Hg exposure on Arctic seabirds  
152 and shorebirds, including more species, tissues (blood, feather, eggs, liver), regions, and larger sample  
153 sizes than done in previous assessments. This review was meant to be an update to the last AMAP  
154 effects assessment (AMAP, 2018; Dietz et al., 2019a), and, as such, does not constitute a complete  
155 synthesis of raw data extracted from the literature that has been previously published. In particular,  
156 Burnham et al. (2019) provides an additional resource, particularly for Hg concentrations in birds from  
157 Greenland during 2010-2012. We also address some of the knowledge gaps identified in previous  
158 AMAP (2011) assessments. Some of these gaps include geographical data gaps in the Russian Arctic,  
159 where new seabird data have become available, including a substantial amount from the ARCTOX  
160 project (<https://arctox.cnrs.fr/en/work-area/>, e.g., Albert et al. 2021). In addition, we include Hg data  
161 on shorebirds (Perkins 2018, in collaboration with the Arctic Shorebird Demographics Network  
162 (ASDN); Lanctot et al. 2016) which have recently become available. This paper includes more Hg  
163 data, a greater number of species, and a wider coverage of the Arctic than was reported previously in

164 the assessment of the risk of Hg to birds in western North America first carried out by Ackerman et al.  
165 (2016). The generation of such a knowledgebase is also important for Article 22 of the Minamata  
166 Convention, which calls for an effectiveness evaluation program, within which Hg levels in Arctic  
167 avian species are an important component  
168 ([https://www.mercuryconvention.org/sites/default/files/documents/information\\_document/4\\_INF12\\_MonitoringGuidance.English.pdf](https://www.mercuryconvention.org/sites/default/files/documents/information_document/4_INF12_MonitoringGuidance.English.pdf)).

170 The second objective of this study is to update the previous AMAP (2011) assessment by reviewing  
171 the effects of Hg contamination on Arctic seabirds and shorebirds. Establishing links between  
172 contaminant exposure and health is difficult (Rodríguez-Estival and Mateo, 2019). However, this  
173 information is extremely important for managing and conserving wildlife populations including those  
174 of globally declining seabirds and shorebirds (Colwell 2010, Paleczny et al. 2015). Herein, we report  
175 on post-2011 studies, mostly on seabirds, that have examined the relationships between Hg exposure  
176 and behavioral and physiological mechanisms that may explain the links between Hg exposure and  
177 Arctic seabird and shorebird demography (reproduction and survival). Studies conducted on Arctic  
178 seabirds and shorebirds are viewed with respect to research conducted on Antarctic birds, which face  
179 ecological constraints comparable to those of Arctic birds. Studies based on long-term mark-capture-  
180 recapture monitoring, have made it possible to explore the effects of Hg exposure on demographic  
181 parameters. Hene, in recent years there have been some studies on the effects of Hg in both Arctic and  
182 Antarctic birds (e.g., Bårdsen et al. 2018, Amélineau et al. 2019, Carravieri et al. 2021). As this type  
183 of study is still rare in the Arctic, we believe it is relevant to place Arctic studies in the more general  
184 context of the effects of Hg in polar birds. This section is intended to be comprehensive, including  
185 potential mechanisms that have been little studied.

186

187 In this article we:

188 1) Provide Hg exposure and potential health risk assessments for 36 Arctic seabird and shorebird  
189 species, by using toxicity benchmarks established for blood and converting them into their toxicity  
190 equivalents for egg, liver, and feather tissues.



191 2) Discuss Hg exposure and potential health risks. To do so, we will review recent studies that  
192 have investigated relationships between Hg exposure, behavior, physiology, and fitness in Arctic  
193 seabirds and shorebirds.

194 3) Make suggestions for future research on the impact of Hg exposure in Arctic seabirds and  
195 shorebirds.

196

## 197 **2. Methods**

198 In birds, most of the total Hg (THg) consists of MeHg in many tissues including blood, feather  
199 and eggs (Bond and Diamond 2009, Rimmer et al. 2005; Ackerman et al. 2013, Renedo et al. 2017). In  
200 the present study, Hg refers to THg and is reported as a proxy for MeHg, except for liver where a sig-  
201 nificant proportion of Hg is in its inorganic form (Eagles-Smith et al. 2009). Adult seabird blood, body  
202 feathers, eggs, and liver Hg data were collected in 24 species from several areas of the Arctic (see  
203 Supplementary Material, Tables S1-S3). Data presented here are based on published data (Ackerman  
204 et al., 2016; Tartu et al. 2013, 2016; 2017, 2018; Goutte et al. 2015; Blévin et al. 2018, 2019; Fleish-  
205 man et al. 2019; Braune 2015; Braune et al. 2014, 2016; Jæger et al. 2009; Helgason et al. 2008, 2011;  
206 Miljeteig et al. 2009; Hoydal and Dam 2005, 2009; Nielsen et al. 2014; Saunes 2011) and, where pos-  
207 sible, raw data as cited in the Supplementary Material, Tables S1-S3 (particularly from ARCTOX:  
208 <https://arctox.cnrs.fr/en/work-area/>).

209 For shorebirds (see Supplementary Material, Tables S4 and S5), a study of the period 2012 to  
210 2013 led by Perkins (Perkins et al. 2016; Perkins 2018), in collaboration with the Arctic Shorebird De-  
211 mographics Network (ASDN; Lanctot et al. 2016) and several other partners, sampled 12 breeding  
212 species from five sites in Alaska located near Nome, Cape Krusenstern, Barrow, the Ikpikpuk River  
213 and the Colville River (Alaska), and three sites in Canada near the Mackenzie River Delta, Bylot Is-  
214 land and East Bay (Nunavut). An additional Canadian study site, Igloodik (Nunavut), was included  
215 with the previously sampled sites in 2013. The ASDN biologists collected blood and feather samples  
216 from adult shorebirds captured while conducting routine fieldwork during the breeding season.

217 We reviewed and assessed the potential for MeHg toxicity in Arctic seabirds and shorebirds  
218 using the available data (see Supplementary Material, Tables S1-S5). To assess risk, we used MeHg  
219 toxicity benchmarks previously established for bird blood (Ackerman et al., 2016) and converted these  
220 values into equivalent total Hg concentrations in other bird tissues that are also commonly sampled in  
221 the Arctic, such as eggs, liver, and body feathers.

222 Blood-equivalent Hg concentrations  $<0.2 \mu\text{g/g}$  (wet weight, ww) are below the lowest-ob-  
223 served effect levels, whereas birds are generally considered to be at low risk when blood Hg concen-  
224 trations are  $0.2\text{--}1.0 \mu\text{g/g}$  ww, moderate risk at  $1.0\text{--}3.0 \mu\text{g/g}$  ww, high risk at  $3.0\text{--}4.0 \mu\text{g/g}$  ww, and se-  
225 vere risk at  $>4.0 \mu\text{g/g}$  ww (Ackerman et al., 2016). We converted these toxicity benchmarks in bird  
226 blood into equivalent concentrations in eggs, based on a review paper that established a general bird  
227 maternal transfer equation of Hg from females to their eggs (Ackerman et al., 2020). Similarly, we  
228 converted blood to equivalent liver Hg concentrations using an inter-tissue correlation equation built  
229 for four species of seabirds and shorebirds (Eagles-Smith et al., 2008). Because many of the Arctic  
230 bird data for Hg contamination have been sampled using bird feathers (Albert et al., 2019), we also  
231 converted these toxicity benchmarks for bird blood into equivalent body feather Hg concentrations us-  
232 ing an unpublished dataset ( $n=16$  species,  $n=2077$  measurements during 2015 to 2017; Equation:  $\ln$   
233  $(\text{Hg body feather } \mu\text{g/g dw}) = 0.64 \times \ln(\text{Hg Blood } \mu\text{g/g ww}) + 1.51$ ; Fort, unpublished). Unlike the  
234 other equations, inter-tissue correlations for feathers and internal tissues, such as blood or eggs, tend to  
235 be poor (Evers et al., 1998; Eagles-Smith et al., 2008; Ackerman et al., 2016). Feather molt represents  
236 a major excretion pathway in birds during which 60% to 90% of accumulated Hg is excreted (Honda  
237 et al., 1986; Braune, 1987; Braune and Gaskin, 1987; Lewis et al., 1993; Agusa et al., 2005). Mercury  
238 in feathers becomes stable once they have been fully grown to the adult stage (Appelquist et al., 1984),  
239 even though they are changed on a regular basis during annual or bi-annual molt. Since feathers have  
240 often grown months before they are collected, the existing temporal and spatial mismatch in Hg con-  
241 centrations between feathers and other tissues is exacerbated. In addition, feather Hg concentrations  
242 can be difficult to interpret for several reasons: a) the complex timing and location of feather molt  
243 (Pyle, 2008; Pyle et al., 2018); b) differences among feather tracts (such as head vs. body feathers;

244 Braune and Gaskin, 1987; Ackerman et al., 2016, Fort et al., 2016); c) the large scale movements of  
245 birds that expose migratory species to Hg over different regions (Fleishman et al., 2019); and d) the  
246 extreme variability in Hg concentrations in some species both within and among individual feathers  
247 from the same individual bird (Peterson et al., 2019).

248 Because of these complexities, feathers of adult birds are typically not recommended for Hg biomon-  
249 itoring programs if detailed information about bird-species biology is lacking (Ackerman et al. 2016;  
250 Chételat et al., 2020) such as the species-specific precise feather molt timing or distribution over their  
251 annual cycle (Ackerman et al., 2012; Albert et al., 2019). Feathers can be a useful sampling tool to  
252 represent Hg contamination in specific cases, such as in remote oceanic locations where birds are diffi-  
253 cult to sample. For example, nape feathers of red-legged kittiwakes (*Rissa brevirostris*) are thought to  
254 be grown at the end of the wintering period and were sampled from birds on their breeding grounds,  
255 where these kittiwakes can be more easily captured on their nests (Fleishman et al., 2019). This sam-  
256 pling strategy coupled with tracking dataloggers demonstrated that red-legged kittiwakes wintering at  
257 more southern latitudes within the North Pacific Ocean had higher Hg concentrations than birds win-  
258 tering at more northern latitudes (Fleishman et al., 2019). Similarly, head feathers of little auks (*Alle  
259 alle*), which grow on the wintering grounds, were used to demonstrate that the bird's feathers were  
260 3.5 times more contaminated when outside of their Arctic breeding locations, indicating that MeHg  
261 acquired at non-Arctic wintering areas in the northwest Atlantic Ocean can be transported to Arctic  
262 breeding areas by migratory birds and has the potential to affect reproductive success (Fort et al.,  
263 2014). Therefore, feathers were useful for demonstrating that non-Arctic regions that were used by the  
264 Arctic avian community for several months per year, during which birds travel thousands of kilome-  
265 ters between their Arctic breeding sites and non-Arctic non-breeding grounds (e.g., Egevang et al.,  
266 2010), are of high concern due to higher Hg contamination experienced during winter (Albert et al.,  
267 2021).

268

### 269 **3. Results and Discussion**

### 270 3.1. Hg exposure and potential health risk for Arctic seabirds and shorebirds

#### 271 3.1.1. Seabirds

272 Based on toxicity benchmarks, we found 50% of individual seabirds ( $n=5000$ ) showed tissue Hg con-  
273 centrations that were above the no risk level for adverse health effects (i.e., a blood-equivalent Hg con-  
274 centration  $>0.2 \mu\text{g/g ww}$ ) and that 1% of seabirds were either in the high or severe risk categories (Fig-  
275 ures 1.1, 1.2, and 1.3. Supplementary Material, Table S1, S2 and S3). In particular, northern fulmar  
276 (*Fulmarus glacialis*), ivory gull (*Pagophila eburnea*), glaucous-winged gull (*Larus glaucescens*),  
277 glaucous gull (*Larus hyperboreus*), lesser black-backed gull (*Larus fuscus*), black-legged kittiwake  
278 (*Rissa tridactyla*), red-legged kittiwake, Atlantic puffin (*Fratercula arctica*), thick-billed murre  
279 (Brünnich's guillemot, *Uria lomvia*), black guillemot (*Cepphus grylle*), pigeon guillemot (*Cepphus co-*  
280 *lumba*), rhinoceros auklet (*Cerorhinca monocerata*), and double-crested cormorant (*Phalacrocorax*  
281 *auritus*) had at least 5% of the individuals sampled with blood-equivalent Hg concentrations at levels  
282 considered to be at moderate, high, or severe risk to toxicity (Figures 1.1, 1.2 and 1.3.; Supplementary  
283 Material, Table S1-S3). Mercury concentrations in seabirds tended to increase through time within the  
284 Arctic, but trends have flattened recently in several Arctic regions (Braune et al., 2001, 2006, 2016;  
285 Braune, 2007; Bond et al., 2015; but c.f. Fort et al., 2016 in East Greenland and Tartu et al. 2022 in  
286 Svalbard). As is common, seabird Hg concentrations differed widely among sites in the Arctic (Figure  
287 2) due to differences in bioaccumulation pathways and processes (Braune et al., 2002, 2014b). Braune  
288 et al. (2014a) found that thick-billed murre breeding at two High Arctic colonies (above  $66^{\circ}30' \text{ N}$ )  
289 tended to have higher Hg concentrations than murre breeding at three Low Arctic locations (below  
290  $66^{\circ}30' \text{ N}$ ). In contrast, seabirds wintering at more southern latitudes generally had higher Hg exposure  
291 (Fort et al., 2014; Fleishman et al., 2019). In general, seabirds tended to have higher Hg concentrations  
292 in the Canadian Arctic, western Canada, and western Greenland than in the European Arctic or Rus-  
293 sian Arctic (Figure 2; Provencher et al., 2014a; Albert et al., 2021).

294

#### 295 3.1.2. Shorebirds

296 Blood Hg concentrations in individual shorebirds ranged from 0.01 to 3.52  $\mu\text{g/g ww}$ , with an overall  
297 mean  $\pm$  standard deviation of  $0.30 \pm 0.27 \mu\text{g/g ww}$  (Figure 3.1., Supplementary Material Table S4).  
298 Among the shorebird species reviewed, the mean blood Hg concentration for long-billed dowitchers  
299 (*Limnodromus scolopaceus*;  $0.74 \pm 0.25 \mu\text{g/g ww}$ ) was over 4.9 times greater than for American  
300 golden plovers (*Pluvialis dominica*;  $0.15 \pm 0.07 \mu\text{g/g ww}$ ). For feathers of shorebird species, the mean  
301 Hg concentration was  $1.14 \pm 1.18 \mu\text{g/g dw}$  for all samples analyzed and ranged from 0.07 to 12.14  
302  $\mu\text{g/g dw}$  (Supplementary Material, Table S5). Feather Hg concentrations also differed by species, with  
303 the mean feather Hg concentration for pectoral sandpipers (*Calidris melanotos*;  $2.58 \pm 1.76 \mu\text{g/g dw}$ )  
304 over 4.3 times greater than for red phalaropes (*Phalaropus fulicarius*;  $0.60 \pm 0.44 \mu\text{g/g dw}$ ). Most Arc-  
305 tic-breeding shorebirds had blood Hg concentrations which placed individuals in the no risk or low  
306 risk categories (Figure 3.1) and below the level at which there can be adverse effects of Hg exposure,  
307 with approximately 45% of individuals in the no risk category and 47% in the low risk category (Fig-  
308 ure 3.1). We found no individuals in the severe risk category, and a low proportion of individuals with  
309 blood Hg concentrations in the moderate risk (4.5%) and high risk (2.5%) categories. The greatest pro-  
310 portion of individuals in the moderate risk and high-risk categories were sampled at the Barrow  
311 (Alaska) study site while the Cape Krusenstern, Ikpikpuk River, Mackenzie River Delta, and Bylot Is-  
312 land sites did not have any individual shorebirds with blood Hg concentrations in these moderate and  
313 high-risk categories. Long-billed dowitchers had the greatest proportion of individuals in the moderate  
314 risk (22%) and high-risk categories (17%). Pectoral sandpiper also had 17% and 9% of individuals  
315 within the moderate and high-risk categories, respectively. Individual American golden plover, Baird's  
316 sandpiper (*Calidris bairdii*), grey plover (*Pluvialis squatarola*), and black turnstone (*Arenaria melao-*  
317 *cephala*) had blood Hg concentrations only in the no effect and low risk categories. Because no tem-  
318 poral trend data were available from shorebirds, no further information on highly Hg exposed species  
319 and regions with temporal trend information could be conducted.

320

## 321 **3.2. Review of the consequences of Hg exposure in Arctic seabird and shorebirds**

### 322 **3.2.1 Demographic consequences**

323 Impaired reproductive success is the most widely investigated and reported consequence of Hg expo-  
324 sure in wildlife (Evers et al. 2008, Scheuhammer et al. 2012, Whitney and Cristol 2017). Chronic ex-  
325 posure to MeHg might also compromise survival rates and long-term reproductive output, potentially  
326 leading to population declines, although very few studies have investigated such demographic conse-  
327 quences (Tartu et al. 2013).

328 *3.2.1.1. Reproductive performance.* Exposure to MeHg can alter avian sexual and mating behaviors  
329 (Frederick and Jayasena 2011 for waterbirds, Spickler et al. 2020 for a domesticated songbird model  
330 species, the zebra finch *Taeniopygia guttata*). For Arctic birds, no information is available. In contrast  
331 to legacy chlorinated Persistent Organic Pollutants (POPs; Blévin et al. 2014), Hg concentrations in  
332 Svalbard black-legged kittiwakes were not associated with carotenoid-based sexual ornamentations,  
333 carotenoid concentrations in plasma, nor to pairing success (Blévin, pers. comm.). Further, the preva-  
334 lence of abnormal sperm cells observed in Svalbard kittiwakes (Humann-Guillemot et al., 2018) was  
335 unrelated to Hg blood concentrations (Blévin et al., pers. comm.). Amélineau et al. (2019) reported  
336 that adult little auks with high Hg concentrations had reduced body condition. Skipping a breeding  
337 event is a common phenomenon in long-lived birds (Charlesworth et al. 1980). Investigations con-  
338 ducted in Svalbard have shown that foregoing breeding was associated with high Hg concentrations in  
339 pre-laying kittiwakes, whereas laying date and clutch size were not related to Hg concentrations (Tartu  
340 et al. 2013). Similarly, Hg levels were unrelated to clutch size and timing of breeding in common ei-  
341 ders (*Somateria mollissima*; Provencher et al., 2017) and Leach's storm petrel (*Hydrobates leu-*  
342 *corhous*; Pollet et al. 2017). In another study of Svalbard kittiwakes, breeding success (probability to  
343 raise at least one chick) in males was negatively related to Hg concentration (Tartu et al. 2016). In a  
344 study of three shorebird species (ruddy turnstones, grey plover, and semipalmated plover *Charadrius*  
345 *semipalmatus*), hatching success was not influenced by egg Hg concentrations (Hargreaves et al.,  
346 2010). In contrast, hatching success was negatively related to paternal Hg concentrations in feathers  
347 (Hargreaves et al., 2010).

348 Impaired reproductive performance may also originate from eggshell thinning (Olivero-Verbel et al.,  
349 2013), although evidence is lacking for seabirds (Peterson et al. 2020). Low hatching success may also

350 be the consequence of a reduced egg size. For example, female little auks with greater feather Hg con-  
351 centrations laid smaller eggs in Greenland (Fort et al., 2014), whereas egg volume was not related to  
352 Hg in Leach's storm petrels (Pollet et al., 2017). The negative effect of Hg exposure on egg hatchabil-  
353 ity was experimentally demonstrated in thick-billed murres and Arctic terns (*Sterna paradisica*) by in-  
354 jecting birds with a range of environmentally relevant concentrations (0–6.4 µg/g ww) of MeHg chlo-  
355 ride (MeHgCl). This study by Braune et al. (2012) showed the relative sensitivity of the developing  
356 embryos to MeHg in these two Arctic seabird species. Finally, in Greenland, chicks of little auks with  
357 the highest Hg concentrations hatched with a body mass reduced by approximately 30% compared to  
358 those with the lowest concentrations, although no impact was further observed on their growth and  
359 fledging success (Kerric, pers. comm). Nevertheless, Amélineau et al. (2019) found that the long-term  
360 increase in Hg contamination of this same population of little auks was associated with a decreased  
361 chick growth rate during the last decade.

362 Previous studies therefore demonstrated that reproductive performances can be impaired by MeHg in a  
363 number of Arctic species. However, those effects were not observed in all study species and further  
364 meta-analysis approaches (i.e. Carravieri et al. 2022) are now required to fully understand how Hg,  
365 alone or in interaction with other environmental stressors, is impacting the reproduction of Arctic sea-  
366 birds. Additionally, most of those studies focused on breeding birds, thus excluding individuals which  
367 did not breed and potentially hiding some Hg effects on seabird body condition or breeding probabil-  
368 ity. Considering this part of the population is challenging but important when investigating effects of  
369 contaminants to avoid biased interpretations (Tartu et al. 2013).

370 *3.2.1.2. Long-term demographic consequences.* Our understanding of the ultimate consequences of Hg  
371 exposure on long-term fitness is still limited in free-living Arctic birds because of the paucity of long-  
372 term data sets that would be required to address this topic. Few long-term capture-mark-recapture  
373 studies on Antarctic (wandering albatross *Diomedea exulans*; Goutte et al. 2014a, Bustamante et al.  
374 2016, grey-headed albatrosses *Thalassarche chrysostoma*, Mills et al. 2020; subantarctic and south po-  
375 lar skuas *Catharacta lonnbergi*, *C. maccormicki* Goutte et al. 2014b) and Arctic seabirds (Svalbard:  
376 glaucous gull and black-legged kittiwake Erikstad et al. 2013; Goutte et al. 2015; Greenland: little auk,

377 Amélineau et al., 2019; Northern Norway: common eider Bårdsen et al., 2018) have estimated the im-  
378 pact of contaminants on long-term breeding probability, reproductive success, and/or adult survival.  
379 These studies, based on long-term ringing programs have mainly focused on legacy chlorinated POPs,  
380 but some of them have included blood and feather Hg concentrations in demographic models (Goutte  
381 et al., 2014 a,b; 2015; Bustamante et al., 2016; Pollet et al., 2017; Bårdsen et al., 2018; Amélineau et  
382 al., 2019). Regarding Arctic seabirds, a long-term study on Svalbard kittiwakes found reduced breed-  
383 ing probability with higher Hg concentrations, but the overall impact of Hg on demographic parame-  
384 ters was modest compared to that of some legacy chlorinated POPs (Goutte et al., 2015). Importantly,  
385 all these long-term studies (reviewed in Whitney and Cristol, 2017) revealed no effect of Hg on adult  
386 survival, a key parameter for seabird population dynamics, despite an order of magnitude range in  
387 blood Hg concentrations across species (from 0.89 to  $8.22 \pm 0.24 \mu\text{g/g dw}$ ). In summary, it appears  
388 that Hg exposure affects reproduction more than adult survival in Arctic birds. Further studies are  
389 nonetheless needed to confirm this pattern.

390

### 391 **3.2.2 Behavioral and physiological mechanisms potentially involved in the demographic conse-** 392 **quences of Hg exposure**

#### 393 *3.2.2.1. Parental behavior*

394 In birds, incubation-related behaviors are influenced by hormonal regulation. In some Antarctic  
395 seabirds, Tartu et al., (2015a) showed that Hg concentrations were associated with a lower  
396 commitment to incubate eggs. Nonetheless, relatively little is known about the effect of contaminants  
397 on incubation temperature for wild birds (Hartman et al., 2019; Taylor et al., 2018). By using loggers  
398 placed into artificial eggs, Blévin et al., (2018) investigated relationships between three groups of  
399 contaminants (organochlorine pesticides (OCs), poly-/per-fluoroalkyl substances (PFAS), and Hg)  
400 with incubation temperature and brood patch in Svalbard black-legged kittiwakes. This study revealed  
401 that, contrary to OCs, Hg concentrations in blood ( $2.00 \pm 0.59 \mu\text{g/g dw}$  in males;  $1.43 \pm 0.38 \mu\text{g/g dw}$   
402 in females) were not related to the minimum incubation temperature, nor the size of the brood patch.  
403 However, incubation does not solely imply the active warming of the eggs but also the active turning



404 of eggs to facilitate albumen absorption by the embryo, reduce the likelihood of an embryo being  
405 mispositioned for hatching (Herring et al., 2010), and prevent the embryo from adhering to the inner  
406 shell membrane. Using egg-loggers, Blévin et al. (2020) found that, unlike polychlorinated biphenyls  
407 (PCBs) and PFAS, blood Hg concentrations were unrelated to egg-turning behavior in Svalbard black-  
408 legged kittiwakes, similar to studies on egg turning in seabirds at temperate latitudes (Taylor et al.,  
409 2018).

#### 410 3.2.2.2. *Endocrine system*

411 To maximize fitness, individuals must make behavioral decisions on their reproduction depending on  
412 environmental conditions (e.g., whether to breed or not, when to breed, what level of parental invest-  
413 ment). These behavioral decisions are mediated by hormones, including luteinizing hormone, a pitui-  
414 tary hormone involved in the onset of breeding (Dawson et al. 2001); stress hormones (corticosterone,  
415 Wingfield and Sapolski, 2003); and prolactin, a pituitary hormone involved in the expression of paren-  
416 tal care (Angelier and Chastel, 2009). Because Hg is a known endocrine disruptor (Tan et al., 2009),  
417 Hg may impair breeding decisions (Hartman et al., 2019) and could alter the ability of Arctic seabirds  
418 to adequately respond to ongoing environmental changes (Jenssen, 2006).

419 Research conducted on Svalbard black-legged kittiwakes has shown that Hg (0.91–3.08  $\mu\text{g/g dw}$ ) in-  
420 fluences pituitary hormones. For example, high Hg concentrations in blood were related to a decreased  
421 secretion of luteinizing hormone (Tartu et al., 2013). Additionally, experimental challenges with exog-  
422 enous GnRH (gonadotropin-releasing hormone) were conducted to test the ability of the pituitary to  
423 release luteinizing hormone in relation to Hg concentrations. These investigations suggested that Hg  
424 disrupted luteinizing hormone secretion by suppressing GnRH input to the pituitary and that elevated  
425 Hg concentrations were linked to years where birds did not reproduce (Tartu et al., 2013). A similar  
426 pattern was observed for Antarctic seabirds (Tartu et al., 2014). As for luteinizing hormone, Hg seems  
427 to impact another pituitary hormone, prolactin, which is known to play a key role in the expression of  
428 avian parental care (Angelier and Chastel 2009). In black-legged kittiwakes from Svalbard as well as  
429 in several Antarctic seabirds, high Hg exposure appeared to be associated with lower plasma prolactin  
430 levels and poor incubation behavior (Tartu et al., 2015a, 2016, Smith et al. pers comm). The effect of

431 Hg on stress hormones secretion is less clear (see Herring et al., 2012 for temperate seabird nestlings  
432 and Provencher et al., 2016a for common eider ducks). In Svalbard kittiwakes, baseline and stress-in-  
433 duced corticosterone levels were unrelated to Hg concentrations (range: 0.82-2.96  $\mu\text{g/g dw}$ ). In this  
434 population, exacerbated baseline and stress-induced corticosterone levels appeared to be triggered by  
435 PCBs, possibly via a stimulation of adrenocorticotrophic hormone (ACTH) receptors (Tartu et al.,  
436 2015b). Further research is thus required on a wider diversity of species to fully grasp the relationships  
437 between Hg contamination and endocrine system in Arctic seabirds.

438

#### 439 *3.2.2.3. Bioenergetics-energy expenditure and thyroids hormones*

440 Individual variation in energy metabolism may influence fitness because of the trade-off in allocating  
441 energy toward self-maintenance (survival), activity, growth, and reproduction (Stearns, 1992). The  
442 minimal energetic cost of living in endotherms, the basal metabolic rate (BMR), is known to be influ-  
443 enced by thyroid hormones (THs) which can stimulate *in vitro* oxygen consumption of tissues in birds  
444 and mammals (Merryman and Buckles 1998). A disruption of THs by environmental contaminants  
445 could act on energy expenditure, yet the effect of Hg on BMR is still poorly documented for wildlife  
446 (see Alexander et al. 2019 on a lab passerine model). Blévin et al. (2017) investigated the relationships  
447 between OCs, PFAS, and Hg with metabolic rate and circulating total THs (thyroxine (TT4) and triio-  
448 dothyronine (TT3)) in adult black-legged kittiwakes from Svalbard. This study indicated that, contrary  
449 to some OCs and PFAS (Blévin et al., 2017; Melnes et al., 2017 for glaucous gull), metabolic rate and  
450 some thyroid hormones (T3) were not associated with Hg blood levels in Svalbard kittiwakes. Further  
451 investigation on the link between Hg exposure, THs, and energy expenditure (basal and field meta-  
452 bolic rate) could be helpful, especially for the most Hg-contaminated species.

453

#### 454 *3.2.2.4. Oxidative stress and telomeres*

455 One potential important biochemical mechanism for Hg to influence wildlife is how it may affect oxi-  
456 dative stress, because of its possible detrimental effects on fitness traits (e.g., reproduction, susceptibil-  
457 ity to disease, survival; Costantini, 2014; Sebastiano et al., 2016). Investigations on temperate and  
458 Antarctic seabirds have reported some associations between Hg and oxidative stress (Costantini et al.,  
459 2014; Gibson et al., 2014; Hoffman et al., 2011). For Arctic seabirds, Wayland et al. (2010) investi-  
460 gated glaucous gulls from the Canadian Arctic and found associations between some oxidative mark-  
461 ers (thiols, lipid peroxidation) and Hg burdens. In Svalbard, there was no association between blood  
462 Hg concentrations (1.96-4.82  $\mu\text{g/g dw}$ ) and several oxidative status markers for kittiwakes (Chastel,  
463 unpublished). Similarly, Fenstad et al. (2016) found no association between Hg exposure and total an-  
464 tioxidant capacity in Baltic and Svalbard common eiders. Oxidative stress can be considered as one of  
465 the mechanisms involved in telomere shortening. Telomeres are repeated sequences of non-coding  
466 DNA located at the terminal ends of chromosomes (Blackburn, 2005). Following their discovery and  
467 implications for maintaining chromosome stability, health, and ageing, there has been a growing inter-  
468 est for studies relating telomere dynamics to contaminant exposure (Angelier et al., 2018). Because  
469 they are associated with longevity and survival in vertebrates, telomeres represent a physiological  
470 marker that may be useful to estimate the toxicological consequences of contaminant exposure (Sebas-  
471 tiano et al., 2020). A recent study has reported that higher feather Hg concentration was associated  
472 with shorter te-lomeres in Cory's Shearwater *Calonectris borealis* (Bauch et al., 2022). However, to  
473 date only a few studies have explored telomere-contaminant relationships in Arctic free-living birds,  
474 mainly in relation to organic pollutants (Sletten et al., 2016; Blévin et al., 2016, 2017; Eckbo et al.,  
475 2019; Sebastiano et al., 2020). Regarding Hg, only one study has been conducted, and they found that  
476 absolute telomere length was positively but weakly associated with blood Hg concentrations in Sval-  
477 bard kittiwakes (Angelier et al., 2018). Thus, there is a current data gap for our understanding of the  
478 relationship between Hg and telomeres.

#### 479 3.2.2.5. *Genotoxicity*

480 Alterations in genetic material may have severe consequences on the survival of individuals and ulti-  
481 mately on the fate of populations. Since 2010, several studies have investigated the genotoxic effects

482 of environmental exposure to pollutants in Arctic seabirds (e.g., Fenstad et al. 2014, Haarr et al.,  
483 2018). Fenstad et al. (2016b) assessed the impact of blood Hg concentrations on DNA double-strand  
484 break (DSB) frequency, in blood cells of a higher exposed Baltic (Hg: 0.43–1.71 nmol/gww), and  
485 lower exposed Arctic population (Svalbard, Hg: 0.31–0.98 nmol/gww) of common eiders. Significant  
486 positive relationships between Hg and DNA DSB frequency were found in Baltic, but not in Svalbard  
487 eiders.

#### 488 3.2.2.6. *Neurology*

489 To understand the effects of Hg exposure on developing thick-billed murre and arctic tern embryos,  
490 Braune et al. (2012) investigated the concentrations of receptors in the brain, a biomarker of MeHg ef-  
491 fects in wildlife (Basu et al., 2006; Scheuhammer et al., 2015). However, no relationship was found  
492 between Hg concentration and density of specific neuroreceptors in brain tissue in either species. De-  
493 spite strong evidence that Hg poses neurotoxic risks to a diverse range of taxa (including birds), there  
494 is limited information from the Arctic.

#### 495 3.2.2.7. *Immune system*

496 Exposure to Hg can be associated with depressed avian immune responses (Fallacara et al., 2011;  
497 Lewis et al., 2013), and this may interfere with reproduction and survival in some contaminated indi-  
498 viduals. Furthermore, such impairment of the immune system may pose an additional threat to Arctic  
499 birds because climate change could increase the emergence of new infectious diseases or a higher  
500 prevalence of parasites (Eagles-Smith et al., 2018; Lee et al., 2020). Provencher et al. (2016a) did not  
501 find an association between Hg blood levels and immunoglobulinY (IgY) in female eider ducks from  
502 the Canadian Arctic. Similarly, in an experimental study of Svalbard barnacle geese (*Branta leu-*  
503 *copsis*), de Jong et al. (2017) found that exposure to Hg from a historic coal mine area had little impact  
504 on four innate immune parameters (haemolysis, haemagglutination, haptoglobin-like activity, and ni-  
505 tric oxide) in goslings. Though studies have failed to detect strong effects of Hg on immune response  
506 in Arctic birds, contaminants and parasites may negatively affect wildlife health and reproduction ei-  
507 ther additively or synergistically (Marcogliese and Pietrock, 2011). To date, only one study on com-  
508 mon eiders from the Canadian Arctic indicated that Hg (breast muscle levels:  $0.63 \pm 0.24 \mu\text{g/g dw}$

509 [mean  $\pm$ SD]) and gastrointestinal parasites potentially influence each other (Provencher et al., 2016b).  
510 Because of their connection with the immune system, changes in vitamins A, D, and E have been in-  
511 vestigated as biomarkers of contaminant exposure and effects in Arctic wildlife (for POPs see Hel-  
512 gason et al., 2010; Braune et al., 2011; Verreault et al., 2013). Since 2010, few studies addressing the  
513 relationships between Hg and vitamins in Arctic seabirds have been conducted. In the ivory gull, a  
514 year-round and contaminated resident of the Arctic (Bond et al., 2015; Lucia et al., 2015), eggs from  
515 Svalbard and the Russian Arctic populations were sampled to investigate relationships between whole  
516 egg Hg content (0.06 - 0.30  $\mu$ g/g ww), eggshell thinning, vitamin A, and vitamin E (Miljeteig et al.,  
517 2012). No association between Hg concentration, eggshell thinning and the two vitamins were found  
518 in this study. Additional research on Hg-vitamin relationships could be helpful for interpreting Hg ef-  
519 fects on birds in the arctic, especially in the context of thiamine (vitamin B1) deficiency observed in  
520 the Baltic Sea (Sonne et al., 2012).

521

#### 522 **4. Discussion and suggestions for future research on the potential impact of Hg in arctic seabirds** 523 **and shorebirds**

524 Overall, the ability of the AMAP monitoring program to provide Hg concentrations in Arctic seabirds  
525 and shorebirds has greatly improved since 2011 with the addition of 24 Arctic seabird and 12 Arctic  
526 shorebird species, and include more tissues (e.g., blood, feathers, eggs, embryo, liver), regions  
527 (Alaska, Nunavut, Northwest Territories, Greenland, Svalbard, Scandinavia), and sample sizes than  
528 prior work. This manuscript also addresses some of the knowledge gaps identified in previous AMAP  
529 assessments. These prior data gaps include geographical data gaps in the Russian Arctic, where new  
530 seabird data have become available and allowed the risk analysis first carried out by Ackerman et al.  
531 (2016) of North American birds to be extended to other parts of the circumarctic region. Continuous  
532 efforts to further fill some still existing spatial gaps (especially in the Russian Arctic) is nonetheless  
533 needed. The knowledge on Hg exposure of Arctic shorebirds has considerably progressed but is still  
534 limited to North America and there is a need to collect additional data on Hg exposure in birds from  
535 the European Arctic and Russia which support considerable shorebird populations (Colwell, 2010).

536 The present study is particularly timely given the recent entry into force of the Minamata Convention  
537 on Mercury (<https://www.mercuryconvention.org/en>), which makes special note of the Arctic's vul-  
538 nerability to Hg. In particular, there is specific interest in monitoring Hg levels in birds in this region  
539 as part of the Minamata Convention's plans for effectiveness evaluation.

540

#### 541 **4.1. Incorporating Hg monitoring into long-term banding studies**

542 Despite recent important advances in assessing Hg exposure in Arctic seabirds and shorebirds, our un-  
543 derstanding of the ultimate consequences of Hg exposure in Arctic seabirds and shorebirds is still lim-  
544 ited by the availability of long-term demographic studies. Seabirds and shorebirds are long-lived ani-  
545 mals and, as such, their populations are especially sensitive to any decrease in adult survival (Sæther  
546 and Bakke 2000). To date, long-term ringing (banding) studies investigating the demographic conse-  
547 quences of Hg and other contaminants in Arctic seabirds are limited to a handful of species and loca-  
548 tions (Svalbard black-legged kittiwakes, Svalbard glaucous gulls, Greenland little auks, Northern Nor-  
549 way common eider; Erikstad et al., 2013; Goutte et al., 2015; Bårdsen et al., 2018; Amélineau et al.,  
550 2019; Sebastiano et al., 2020). For shorebirds, this type of long-term ringing study incorporating Hg  
551 measurements is not currently available, but it would be useful given the current, large-scale declines  
552 in many Arctic shorebird populations (e.g., Kubelka et al., 2018). Thus, investigating whether Hg con-  
553 tamination is linked to adult survival and reproduction, especially for at-risk species, may be particu-  
554 larly useful. These mark-recapture studies on individuals that have been marked and subjected to de-  
555 mographic monitoring over several years would help our understanding of the effects of Hg on bird  
556 demography.

557

#### 558 **4.2. Considering a multi-stressor perspective**

559 Overall, our review indicates that most individual Arctic seabirds (93%) and shorebirds (95%) were  
560 considered at lower risk to potential Hg impairment ( $<1.0 \mu\text{g/g}$  ww in blood). For instance, according  
561 to Hg toxicity benchmarks observed for blood, most Svalbard kittiwakes and 100% of Greenland little

562 auks were considered at no or low risk, yet behavioral and physiological disruption together with im-  
563 paired breeding performances have been reported in these species (Tartu et al., 2013, 2016; Goutte et  
564 al., 2015; Amélineau et al., 2019). Arctic birds are exposed to multiple stressors and the impacts of Hg  
565 probably act in concert with both natural and other anthropogenic stressors (e.g., other contaminants,  
566 diseases, parasites, climate-related environmental changes; see Provencher et al., 2016b). Thus, even  
567 Hg concentrations considered as posing low or moderate risks to birds, may cause adverse effects if  
568 they co-occur with other stressors (Goutte et al., 2014b; Fort et al., 2015; Tartu et al., 2016; Amélineau  
569 et al., 2019).

570

#### 571 *4.2.1. Interactions between Hg and other contaminants*

572 The Arctic is a sink for a mixture of various pollutants (Dietz et al., 2019), thus future investigations  
573 could also incorporate other types of contaminants (e.g., legacy chlorinated and brominated POPs,  
574 PFAS, and other non-essential trace elements) known to interact with behavior, physiology, and fit-  
575 ness into demographic models to better assess the specific impacts of Hg. For instance, little is known  
576 about levels and effects of trace elements such as copper (Cu), iron (Fe), or selenium (Se) (but see An-  
577 derson et al., 2010; Hargreaves et al., 2010, 2011; Borgå et al., 2006; Fromant et al., 2016). These  
578 trace elements are essential for biological processes within a narrow range of concentrations, but can  
579 lead to deleterious effects outside the range, and are able to interact with other contaminant uptake,  
580 storage, and toxic effects (Walker et al., 2012). Specifically, Se may have a protecting role against Hg  
581 toxicity (Khan and Wang, 2009), yet only a few studies have quantified their co-exposure and interac-  
582 tion in seabirds (e.g., Carravieri et al., 2017; 2020; Carvalho et al., 2013; Cipro et al., 2014; Gonzalez-  
583 Solís et al., 2002; Provencher et al., 2014b). Demethylation of MeHg and subsequent sequestration of  
584 inorganic Hg with Se has often been suggested as a probable detoxification mechanism for vertebrates  
585 (Eagles-Smith et al. 2009, Renedo et al. 2021, Manceau et al. 2021). Ralston et al. (2008) reported that  
586 the molar ratio of Hg:Se was critical to the expression of MeHg toxicity. Specifically, molar excesses  
587 of Se over Hg may be important in the potential to protect cells from Hg toxicity, such as by detoxify-  
588 ing Hg by forming tiemannite complexes (Dietz et al., 2013, 2019). However, the process is complex,

589 and recent studies indicate that demethylation may require four Se rather than one per atom of Hg  
590 (Manceau et al. 2021). Incorporating Se measurements into Hg assays might allow refinement of our  
591 understanding of Hg toxicity and, more generally, allow for better assessment of the overall impact of  
592 Hg on wildlife (Goutte et al. 2014a, Carravieri et al. 2017).

593

#### 594 *4.2.2. Interplay between Hg and parasites*

595 Hg and parasites are ubiquitous stressors that can affect animal physiology and derive from similar di-  
596 etary sources (co-exposure). Parasites could modulate the kinetics of Hg in its organism (assimilation,  
597 distribution in tissues, accumulation) and effects on health (Bustnes et al., 2006). Understanding Hg  
598 concentrations in bird tissues and their parasites (Morrill et al. 2015) and the interaction of Hg with  
599 parasites could be useful for biomonitoring pollution, and to realistically quantify the health risks for  
600 Arctic birds (Provencher et al. 2016). Finally, quantifying the effects of Hg and parasites, alone or in  
601 combination, on markers representative of the health status of the organism (Carravieri et al., 2020)  
602 could test the hypothesis that parasites may act as a "contaminant sink" and thus relieve the host of  
603 some of its Hg contamination.

604

#### 605 *4.2.3. Possible carry-over effects*

606 Most seabirds and shorebirds leave the Arctic after the breeding period and some are long distance mi-  
607 grants, spending the winter in sub-Arctic, temperate, tropical, or Antarctic areas (e.g., Egevang et al.,  
608 2010; Battley et al., 2012; Gilg et al., 2013). Environmental stressors and Hg uptake experienced out-  
609 side of the breeding season can result in sub-lethal to lethal effects and can synergistically contribute  
610 to high bird mortality by impacting their body condition (Fort et al., 2015). Environmental stressors  
611 can also result in non-lethal effects that will be carried on to the next breeding season (carry-over ef-  
612 fects; Norris, 2005) and which could impact fitness and population dynamics. Combining miniaturized  
613 tracking systems (e.g., geolocators to document migratory movements and wintering areas) with meas-  
614 urements of Hg levels in tissue archives (e.g., feathers molted during winter; Albert et al., 2021;



615 Fleishman et al., 2019), environmental stressors, and detailed demographic surveys could provide rele-  
616 vant information on the global impact of Hg on Arctic birds. Moreover, recent advances in Hg isotope  
617 analyses and studies suggest that Hg isotopes may be used to differentiate between specific environ-  
618 mental Hg sources and processes (Tsui et al., 2020), which again may provide insight into the sources,  
619 source areas, and biogeochemical processes involved in Hg uptake and exposure in migrating birds.

620

#### 621 *4.2.4. Climate change and assessments of Hg risk for Arctic seabirds and shorebirds*

622 The Arctic is warming two to three times faster than any other region on Earth (AMAP 2021) with  
623 impacts on precipitation, snow cover, permafrost levels, and sea-ice thickness and extent. These  
624 changes are causing fundamental alterations in ecosystems that affect biogeochemical fluxes, bottom-  
625 up processes, ecosystems, and food webs, which may lead to modifications in Hg exposure in Arctic  
626 biota (Stern et al., 2012; Braune et al., 2014b; McKinney et al., 2015; Tartu et al., 2022). Of particular  
627 concern is that global warming, which will lead to an earlier onset of thawing and a later start of  
628 freezing, will likely extend the period of Hg methylation (Stern et al., 2012) and thus may increase  
629 exposure of the toxic form of Hg, MeHg, to seabirds and shorebirds. The permafrost stores large  
630 amounts of Hg, nearly twice as much as other soils, the ocean, and the atmosphere combined, which  
631 may become mobilized and released during thawing and therefore may represent a significant source  
632 of Hg (Schuster et al., 2018). A recent study on Hg contamination of polar bears (*Ursus maritimus*) in  
633 the Barents Sea (northern coasts of Norway and Russia) showed that the increased Hg in polar bears  
634 during the last two decades was attributed to re-emissions of previously stored Hg from thawing sea-  
635 ice, glaciers, and permafrost, with this Hg then becoming bioavailable and biomagnifying in the Arctic  
636 marine food webs (Lippold et al., 2020). These results indicate that climate-induced re-emission of  
637 legacy Hg may already be happening in the Arctic.

638

639 Climate change in the Arctic may challenge physiological processes of individuals (water balance, ther-  
640 moregulation, nutrition, immune, endocrine, and neurological systems) critical for coping with the ex-  
641 ternal environment, causing Arctic birds to become more sensitive to Hg contamination because they  
642 may be pushed to the limits of their physiological tolerance (Hooper et al., 2013). Alternatively, in-  
643 creased exposure to Hg could make birds more sensitive to stressors (heat waves, increased precipita-  
644 tion, diseases, changes in food web, and nesting habitats) induced by climate change (Hooper et al.,  
645 2013). In this multi-stressor context, the challenge will be to identify potential interactions between non-  
646 chemical and chemical stressors affecting key physiological processes in Arctic seabirds and shorebirds.  
647 Understanding Hg exposure and climate change interactions could facilitate the assessment of the po-  
648 tential health risks for Arctic birds.

649

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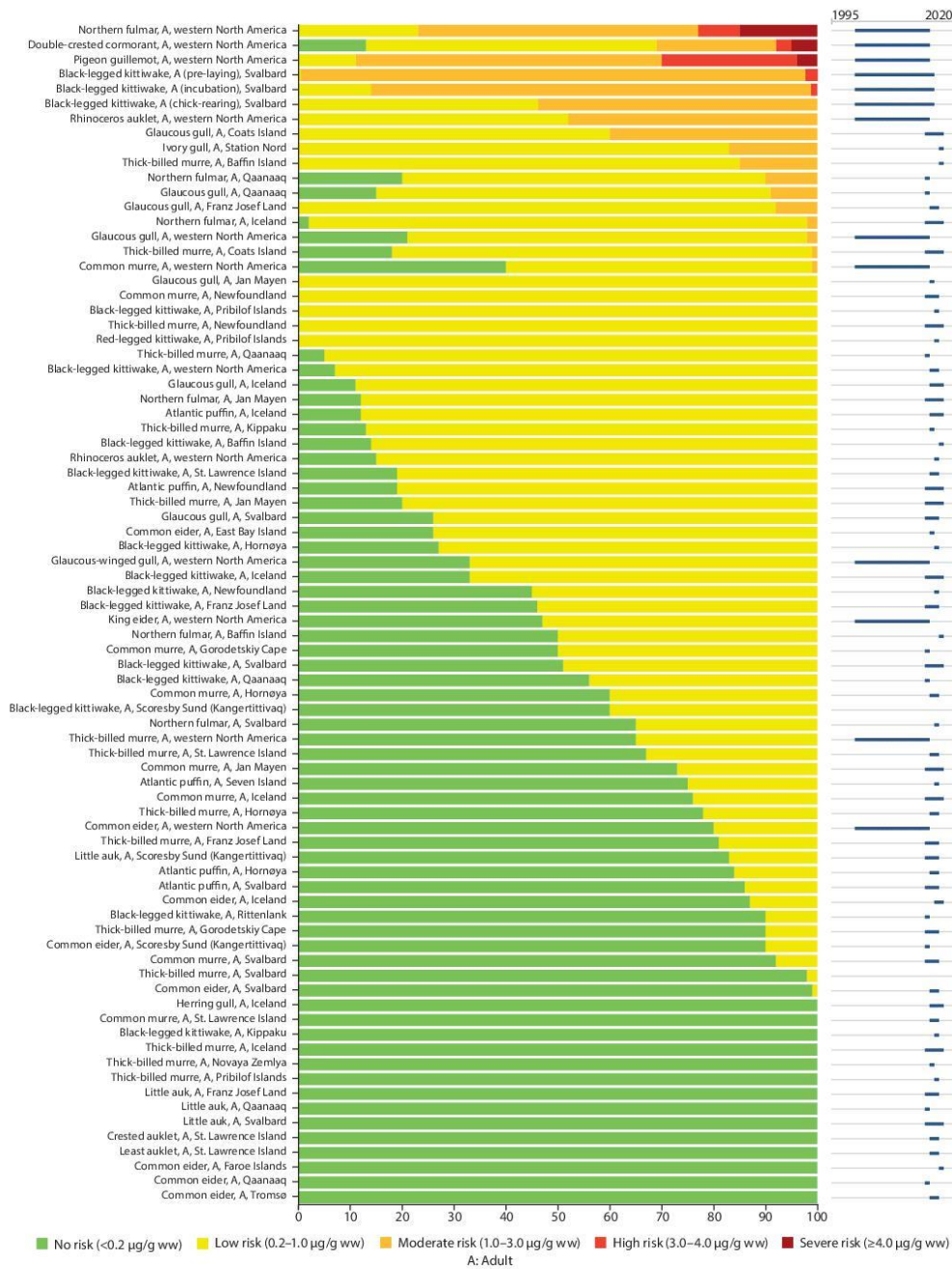
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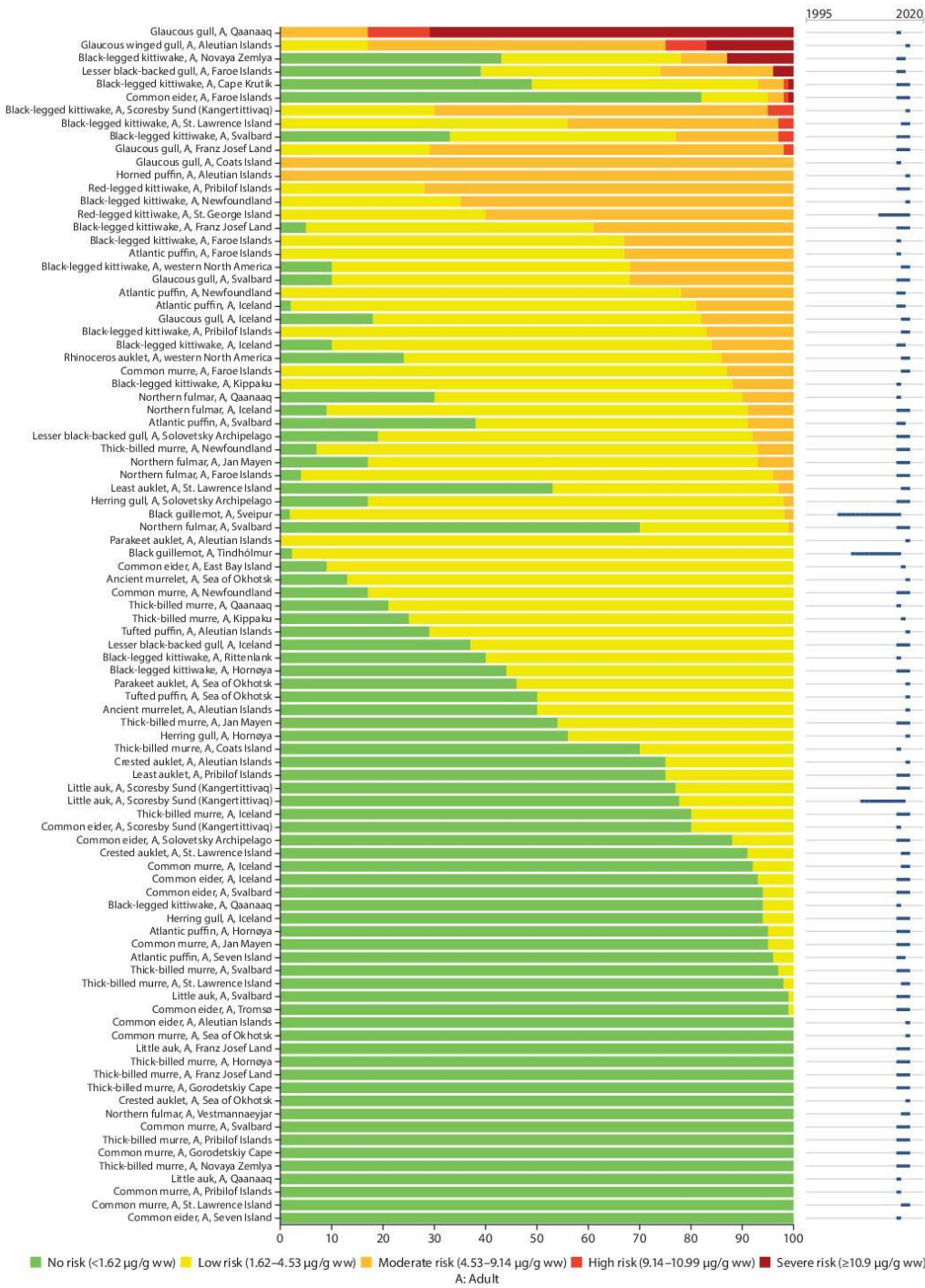
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1133 **Figure 1.1.** Ranked overview (from highest to lowest risk) of the proportion of seabird blood per re-  
 1134 gion from the Arctic, which are at risk for Hg-mediated health effects (categorized in five risk catego-  
 1135 ries based upon blood Hg effect thresholds). Blue bars on the right side indicate the date range of the  
 1136 samples. Please see Supplementary Material Table S1 for detailed information upon which this sum-  
 1137 mary graphic is based, including scientific names for all species.

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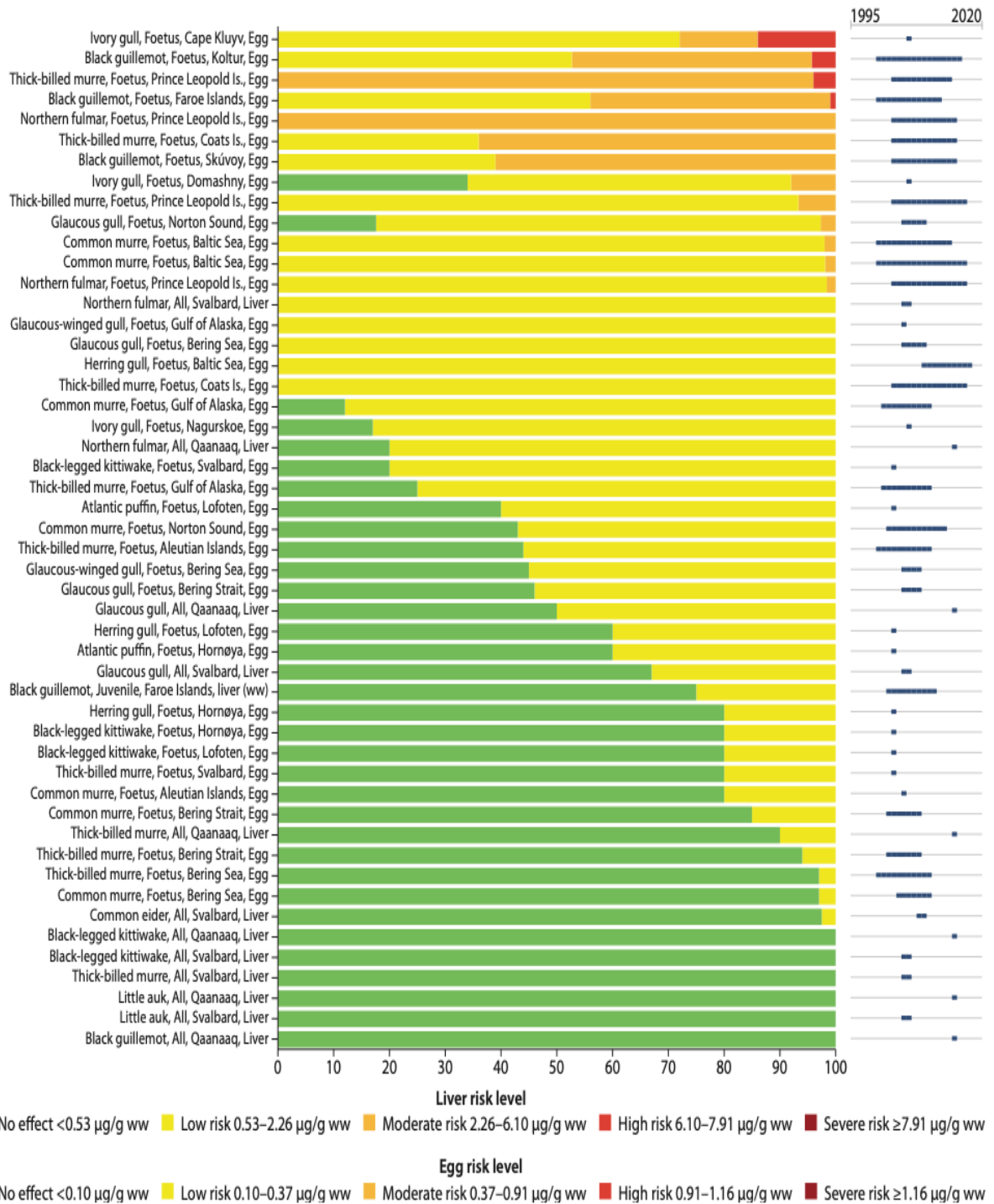
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1142 **Figure 1.2.** Ranked overview (from highest to lowest risk) of the proportion of seabird feathers per  
 1143 region from the Arctic, that are at risk for Hg-mediated health effects (categorized in five risk catego-  
 1144 ries based upon feather Hg effect thresholds). Blue bars on the right side indicate the date range of the  
 1145 samples. Please see Supplementary Material Table S2 for detailed information upon which this sum-  
 1146 mary graphic is based, including scientific names for all species.

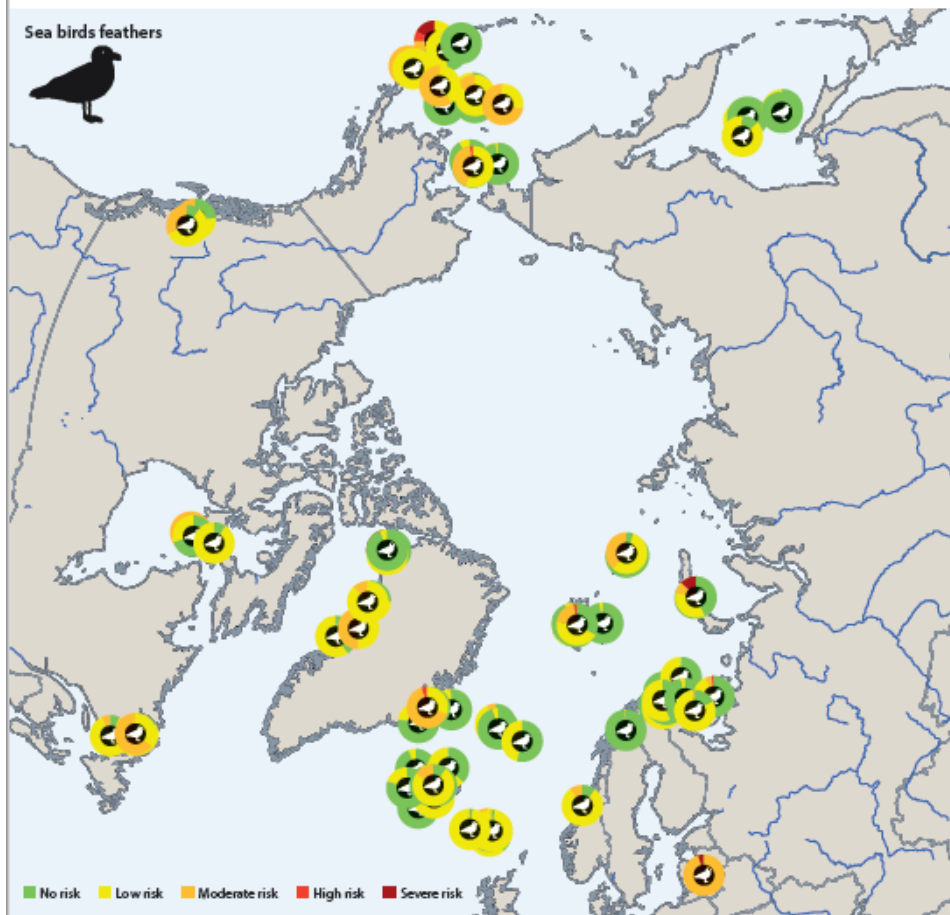
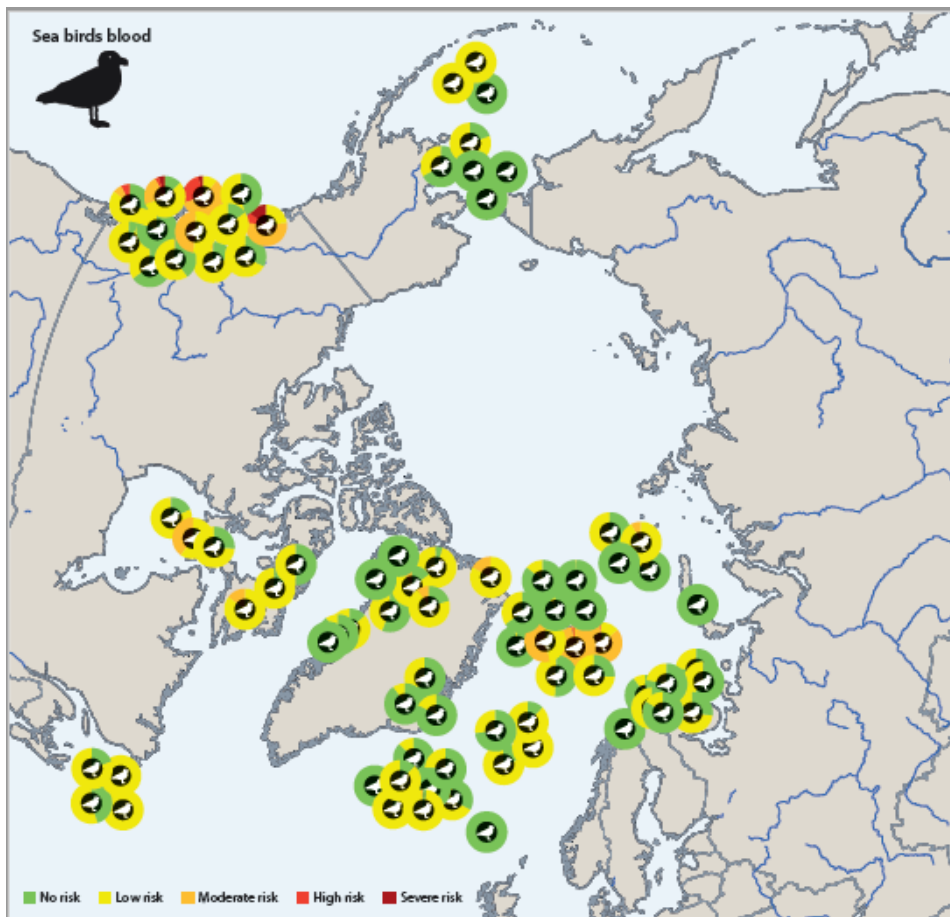


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1148 **Figure 1.3.** Ranked overview (from highest to lowest risk) of the proportion of seabird livers and eggs  
 1149 per region from the Arctic, that are at risk for Hg-mediated health effects (categorized in five risk cate-  
 1150 gories based upon liver and egg Hg effect thresholds). Blue bars on the right side indicate the date  
 1151 range of the samples. Please see Supplementary Material Table S3 for detailed information upon  
 1152 which this summary graphic is based, including scientific names for all species.

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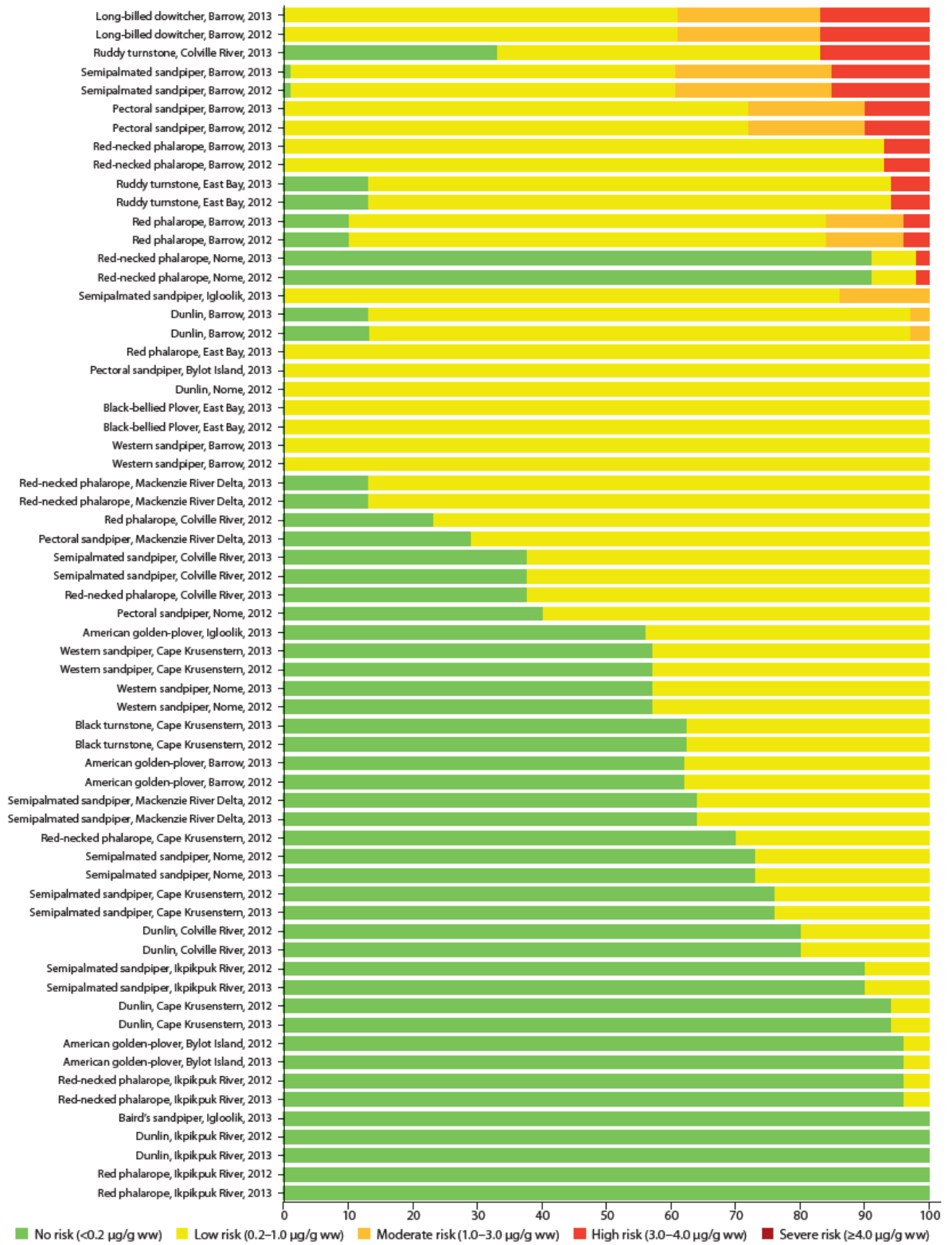




1156 **Figure 2.** Geographical overview of the proportion of adult seabirds that are at risk of Hg-mediated  
1157 health effects based on data for blood (upper) and feathers (lower). Please see Supplementary Material  
1158 Table S1 and S2, and Figures 1.1 (blood) and 1.2 (feathers), for detailed information upon which this  
1159 summary graphic is based.

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**Figure 3.1.** Ranked overview (from highest to lowest risk) of the proportion of shorebirds blood, per region from the Arctic, that are at risk for Hg-mediated health effects (categorized in five risk categories based upon liver Hg effect thresholds). Please see Supplementary Material Table S4 for detailed information upon which this summary graphic is based, including scientific names for all species.





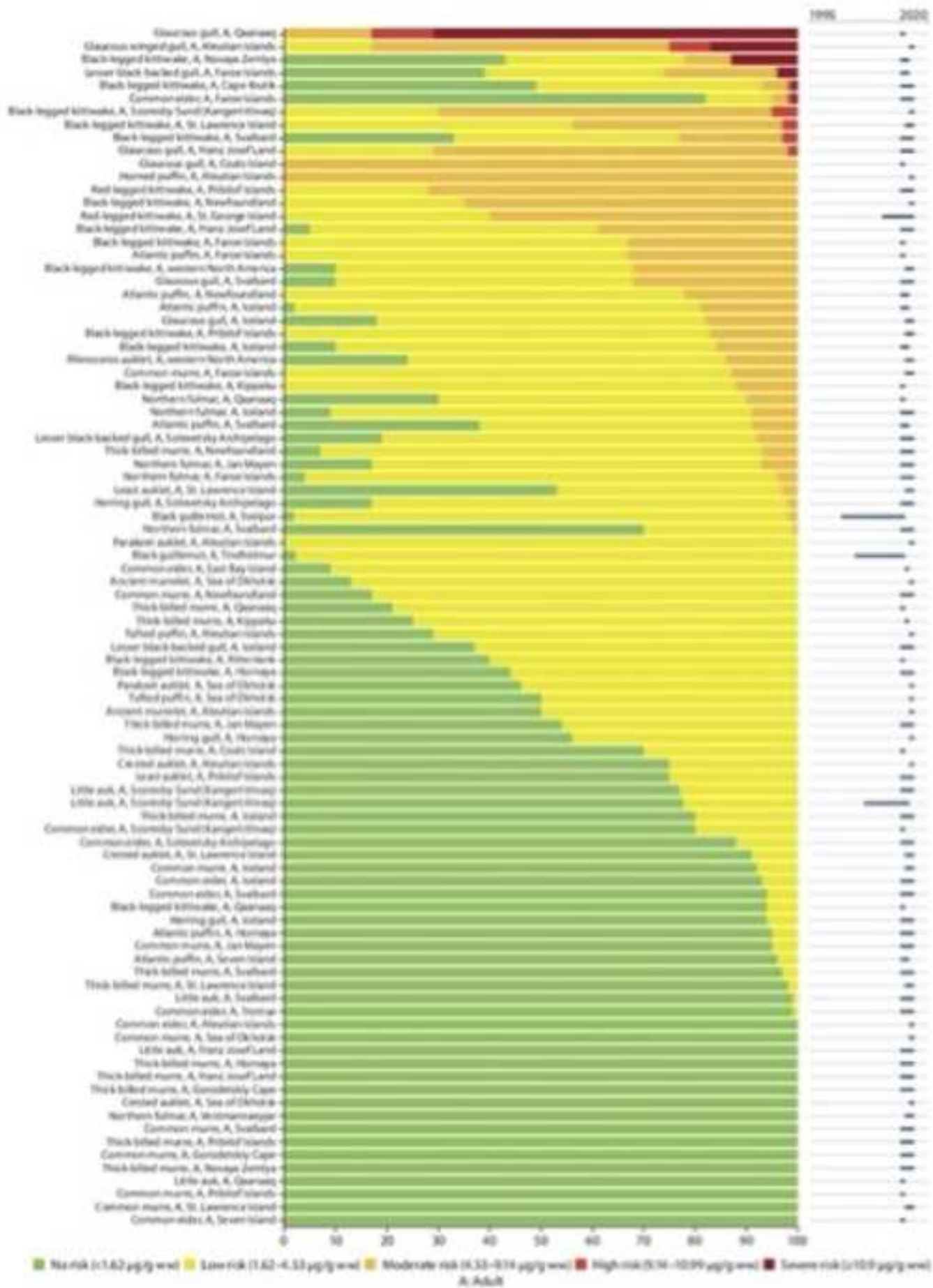


Figure 1.3

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