Mercury biomagnification in a Southern Ocean food web

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Abstract

Biomagnification of mercury (Hg) in the Scotia Sea food web of the Southern Ocean was examined using the stable isotope ratios of nitrogen (δ^{15}N) and carbon (δ^{13}C) as proxies for trophic level and feeding habitat, respectively. Total Hg and stable isotopes were measured in samples of particulate organic matter (POM), zooplankton, squid, myctophid fish, notothenioid fish and seabird tissues collected in two years (austral summers 2007/08 and 2016/17). Overall, there was extensive overlap in δ^{13}C values across taxonomic groups suggesting similarities in habitats, with the exception of the seabirds, which showed some differences, possibly due to the type of tissue analyzed (feathers instead of muscle). δ^{15}N showed increasing enrichment across groups in the order POM to zooplankton to squid to myctophid fish to notothenioid fish to seabirds. There were significant differences in δ^{13}C and δ^{15}N values among species within taxonomic groups, reflecting inter-specific variation in diet. Hg concentrations increased with trophic level, with the lowest values in POM (0.0005 ± 0.0002 µg g^{-1} dw) and highest values in seabirds (3.88 ± 2.41 µg g^{-1} in chicks of brown skuas *Stercorarius antarcticus*). Hg concentrations tended to be lower in 2016/17 than in 2007/08 for mid-trophic level species (squid and fish), but the opposite was found for top predators (i.e. seabirds), which had higher levels in the 2016/17 samples. This may reflect an interannual shift in the Scotia Sea marine food web, caused by the reduced availability of a key prey species, Antarctic krill *Euphausia superba*. In 2016/17, seabirds would have been forced to feed on higher trophic-level prey, such as myctophids, that have higher Hg burdens. These results suggest that changes in

the food web are likely to affect the pathway of mercury to Southern Ocean top predators.

Capsule: Changes in foodweb dynamics influence Hg bioaccumulation in top predators.

Keywords: Trophic magnification slope; Stable isotopes; Contaminants; Antarctica; Polar.

Introduction

Antarctica and the Southern Ocean comprise a globally unique ecosystem characterised by cold environmental conditions. Due to its isolation and often-inhospitable weather, much of the region has remained relatively untouched by direct human activity, such that it is often considered by the scientific community as a natural laboratory (Walton, 2013). Given the absence of any local manufacturing industry, anthropogenic mercury (Hg) emissions from within Antarctica are negligible. Nevertheless, there is long-range dispersal of Hg from regions outside Antarctica, and relatively high concentrations can be found in Southern Ocean waters (Cossa et al., 2011). The global distillation process (Wania and Mackay, 1996) leads to atmospheric transportation of a range of volatile and semi-volatile pollutants in the form of vapour. Hg is transported by atmospheric currents in the form of Hg^0 to the polar regions where it condenses and precipitates in rain or snow (O'Driscoll et al., 2005). Indeed, Antarctica can be considered a gigantic fridge where atmospherically transported pollutants are stored in the ice fields (Eisele et al., 2008), becoming bioavailable when glaciers and icebergs melt (Mastromonaco et al., 2017). Moreover, recent studies have shown that vegetation in the polar environments can play an important role in the uptake of Hg0. Plus, Atmospheric Mercury Depletion Events (AMDE) occurring during springtime at polar sunrise (Ebinghaus et al., 2002) are known to increase the deposition fluxes of atmospheric Hg (Brooks et al., 2008; Elizalde, 2017), and

74   katabatic winds can redistribute Hg within Antarctica, carrying it from the high Antarctic

75   plateau towards coastal areas (Bargagli, 2016; Bromwich1989, 1989).

76   Among pollutants, Hg is one of the most toxic elements, particularly its organic

77   form (methyl-Hg, $[CH_3Hg]^+$) (Clarkson, 1992). Due to its high affinity for proteins and

78   hence retention within tissues (Bloom, 1992), Hg is highly bioaccumulative in

79   organisms over the course of their lives. It also biomagnifies along food webs from

80   plankton up to top predators ((Ackerman et al., 2014; Coelho et al., 2013; Dehn et al.,

81   2006). Methyl-Hg is assimilated more efficiently by organisms than inorganic Hg, and

82   is accumulated rather than excreted (Monteiro et al., 1996). Hg concentrations are

83   therefore higher in the upper trophic levels of food webs, especially those in aquatic

84   systems which tend be highly size-structured (Heneghan et al., 2019), and may

85   become very toxic for large, long-lived top predators (Goutte et al., 2014; Tartu et al.,

86   2014; Tavares et al., 2013). Indeed, some Southern Ocean predators have particularly

87   high Hg concentrations. For example, wandering albatross, *Diomedea exulans,* which

88   can live to over 50 years (Lecomte et al., 2010) have amongst the highest reported

89   feather Hg concentrations (up to 73.42 µg g^{-1}) of any seabird, reflecting a high degree

90   of exposure to this metal (Anderson et al., 2009; Cherel et al., 2018; Tavares et al.,

91   2013; Thompson et al., 1998).

92   Indices of the enrichment of stable isotopes of nitrogen (δ^{15}N) and carbon (δ^{13}C)

93   provide proxies for the trophic level and carbon source (habitat) of consumers,

94   respectively (Cherel and Hobson, 2007; Stowasser et al., 2012). ^{15}N becomes

95   enriched in tissues in a consistent way in food webs, on average by +3.4 ‰ per trophic

96   level (Minagawa and Wada, 1984). ^{13}C is enriched at a lower rate (on average by +1

97   ‰ per trophic level) but gives a useful indicator of feeding habitat; this indicates, for

98   example, the relative contributions of feeding inshore *versus* offshore, in shallow water

99   *versus* deep water, or in particular water masses given the broad latitudinal gradients

100   in the Southern Ocean (Hobson et al., 1994). Although some studies have reported

101   Hg concentrations both in low trophic level organisms and top predators in the

102   Southern Ocean (e.g. Anderson et al. 2009), none have evaluated Hg biomagnification

103   across multiple levels in the food web, from particulate organic matter (POM) to apex

104   predators, using $\delta^{15}N$ to determine trophic position.

105   Antarctic krill (*Euphausia superba*; hereafter krill) is a key component of the

106   food web in the Southern Ocean (Murphy et al., 2007), and can be the main trophic

107   link between primary producers and predators (Everson, 2000; Xavier and Peck, 2015;

108   Xavier et al., 2018). However, recent studies have shown that there are alternative

109   trophic pathways in periods and regions of low Antarctic krill abundance. These

110   include, for example, copepods to mesopelagic fish to high predators (Ballerini et al.,

111   2014; Murphy et al., 2007; 2013; Saunders et al., 2019). These alternative pathways

112   are unlikely to support the same biomass of predators given the greater number of

113   steps in the food chain over which energy can be lost (Barnes et al., 2010). Decreases

114   in the reproductive performance of top predators, including seabirds and pinnipeds, in

115   low-krill years in the Scotia Sea region suggests that these alternative pathways

116   cannot entirely replace those involving krill in maintaining very large predator

117   populations (Croxall et al., 1999; Xavier et al., 2003; 2017).

118   In the present study, we measured total Hg concentrations, $\delta^{13}C$ and $\delta^{15}N$

119   values in a suite of species from different trophic levels to elucidate Hg

120   biomagnification in components of the food web in the Scotia Sea, one of the most

121   productive regions of the Southern Ocean (Atkinson et al., 2001). POM and tissues

122   from key functional organisms from multiple trophic levels were analysed, including

123   zooplankton, squid, mesopelagic fish (myctophids), necto-benthic fish (notothenioids),

124   and seabirds. The field sampling was undertaken in two austral summers, 9 years

125   apart (December 2007 to February 2008 and December 2016 to January 2017). The

126   main objectives were to 1) describe Hg dynamics in the Scotia Sea food web, 2)

127   evaluate Hg biomagnification-rates, and, 3) evaluate possible differences in the Hg

128   pathway from POM to top predators in different sampling years.

129

130   **Materials and Methods**

131

132   *Field sampling*

133   Samples were collected during oceanographic research cruises on board the

134   RRS *James Clark Ross* around the islands of South Georgia (54°17′S, 36°30′W)

135   during the austral summers of 2007/08 and 2016/17 (cruises JR177 and JR16003

136   respectively). Background concentrations of Hg in POM were determined from water

137   samples collected in Niskin bottles deployed on a CTD (conductivity, temperature,

138   depth) rosette fired at the depth of the *chlorophyll* a maximum (which ranged from 30

139   to 76 m) and at 500 m. The depths were chosen to enable comparison of the likely

140   highest concentrations of POM with the lower values expected below the euphotic

141   zone. POM was obtained by vacuum-filtering 5 L of water through glass fibre filters

142   (GF/F Whatman, 47 mm) during JR16003. Zooplankton, squid and myctophid fish

143   species were collected using either an 8 or 25 m² mouth-opening Rectangular

144   Midwater Trawl [RMT8 - mesh size reducing from 4.5 mm to 2.5 mm in the cod end to

145   collect zooplankton; RMT25 - mesh size reducing from 8 mm to 4.5 mm in the cod end

146   to collect mesopelagic fish and squid (Roe and Shale, 1979)]. Both systems

147   incorporated two separate nets that could be opened and closed remotely on

148   command at different depths. Myctophid fish and squid were identified using

149   appropriate taxonomic keys (Gon and Heemstra, 1990; Hulley, 1981; Nesis, 1987;

150 Xavier and Cherel, 2009). Sub-samples were then frozen individually in separate

151 plastic bags at -20ºC for later laboratory analyses. Zooplankton species were identified

152 following Boltovskoy (1999). Sub-samples of these species were either preserved in

153 plastic bags at -20ºC (JR16003) or in glass vials in 70% ethanol (JR177).

154 Notothenioid fish were also obtained from South Georgia waters in the austral

155 summer of 2016/17. They were caught by the Fishing Vessel (FV) *Sil* during research

156 survey SG17. Samples were obtained from bottom trawls using a FP120 trawl net with

157 a standard steel bobbin rig. Whenever possible, samples were identified on board but,

158 in some cases, identification was not possible at sea and was performed later at the

159 laboratory. Individuals were frozen at sea at -20ºC for later laboratory processing.

160 Feathers from seabird chicks were collected at Bird Island (54°00' S, 38°03'

161 W), South Georgia, in austral summers 2007/08 and 2016/17. Chicks were sampled

162 rather than adults because Hg and stable isotope ratios in their tissues reflect those

163 of food consumed during the chick-rearing period (Blévin et al., 2013; Moreno et al.,

164 2016). Furthermore, all of these feathers are grown by chicks over the same period,

165 so Hg concentrations and stable isotope ratios reflect similar time periods and can be

166 compared directly (Carravieri et al., 2014a).

167

168 *Laboratory procedures*

169 POM filters were digested with HNO_3 4M for determination of Hg concentrations

170 (for details, see Pato et al., 2010). Analyses were performed at the University of Aveiro

171 by cold-vapour atomic fluorescence spectrometry (CV-AFS) using a PSA model Merlin

172 10.023 equipped with a PSA model 10.003 detector, with tin chloride as a reducing

173 agent (2% in 10% HCl) following Pato et al., (2010). The limit of quantification of this

174 technique was 0.02 μg L^{-3}.

175   Zooplankton were analysed as whole individuals. Specimens that had been

176   preserved in 70% ethanol were first dried for 72 h at ambient temperature to remove

177   ethanol, and then freeze-dried for 48 h (Fort et al., 2014). Samples of squid and fish

178   muscle tissue (freeze-dried for > 24 h) were chosen for analysis since muscle is the

179   most important tissue in terms of transfer of Hg to predators (Bustamante et al., 2006;

180   Cipro et al., 2018). Feathers were first cleaned to remove surface contaminants using

181   a 2:1 chloroform:methanol solution followed by two methanol rinses, and then oven

182   dried for 48 h at 50 ºC.

183   Dried individual zooplankton, muscle samples and feathers were homogenized

184   to powder and analysed for total Hg by thermal decomposition atomic absorption

185   spectrometry with gold amalgamation, using a LECO AMA-254 (Advanced Mercury

186   Analyser) at the University of Aveiro, following Coelho et al., (2008). Analytical quality

187   control was performed using certified reference materials (CRMs): for zooplankton we

188   used TORT-2 and TORT-3 with recoveries of 87 ± 3% and 90 ± 8%, respectively; for

189   squid we used NIST 2976, ERM-CE278K and TORT-3 with recoveries of 85 ± 7%, 92

190   ± 5% and 93 ± 8%, respectively; for myctophids we used DORM-4 and ERM-BB422

191   with recoveries of 96 ± 13% and 100 ± 4%, respectively; for notothenioid fish we used

192   ERM-BB422 with recovery of 98 ± 7%, and for seabirds we used TORT-3 with

193   recovery of 99 ± 3%. Sample analyses were repeated in duplicate or triplicate until the

194   relative standard deviation was <10% for multiple aliquots. The limit of detection for

195   this analytical method was 0.01 ng of absolute mercury.

196   As lipid is enriched in ^{13}C relative to other tissue components (DeNiro and

197   Epstein, 1977), lipids were extracted from fish muscle. An aliquot of approximately 10

198   mg of fine muscle-tissue powder was agitated with 4 ml of cyclohexane for 1 h. Next,

199   the sample was centrifuged for 5 min at 4000 g, and the supernatant containing lipids

200 was discarded. Due to small masses and low sample numbers it was not possible to

201 remove lipids from zooplankton, nor to pool samples, so stable isotope ratios were

202 corrected according to Post et al., (2007). As squid muscle and bird feathers have low

203 lipid content, no delipidation was applied. Low lipid content was checked by

204 considering C/N mass ratios, which were always below 4.0 (squid – 3.91 and seabirds

205 – 3.46).

206   Approximately 0.4 mg of dry subsample homogenates were weighed out into

207 tin cups. A continuous flow mass spectrometer (Thermo Scientific Delta V Advantage)

208 was coupled to an elemental analyzer (Thermo Scientific Flash EA 1112), at either the

209 LIENSs or MAREFOZ laboratories, to measure $\delta^{13}C$ and $\delta^{15}N$ values. Stable isotope

210 ratios are expressed using standard δ notation relative to carbonate Vienna PeeDee

211 Belemnite and atmospheric nitrogen. The internal laboratory standard is acetanilide.

212 Observed analytical errors were <0.10 ‰ for both $\delta^{13}C$ and $\delta^{15}N$ values at both

213 facilities.

214

215 *Statistical analysis*

216   All analyses were performed using R software v. 3.4.2 (R Core Team, 2013).

217 Distributions of Hg concentrations and $\delta^{13}C$ and $\delta^{15}N$ values within samples were

218 tested for normality using Shapiro-Wilk normality test, and homogeneity of variance

219 was tested using Bartlett's test. Wilcoxon rank and Kruskal–Wallis tests were used to

220 compare Hg, $\delta^{13}C$ and $\delta^{15}N$ values among trophic groups (zooplankton, squid,

221 myctophid fish, notothenioid fish, and seabirds) and species, followed by a Dunn's

222 multiple comparisons test. T-test or Mann-Whitney were used to compare Hg, $\delta^{13}C$

223 and $\delta^{15}N$ values between 2007/08 and 2016/17. Linear regressions were examined

224 between $Log_{10}Hg$ and $\delta^{15}N$ values as a tool to evaluate the trophic magnification slope

225 (TMS) (Lavoie et al., 2013). All values are presented as means ± SD. The significance

226 level for statistical analyses was α = 0.05.

227

228 **Results**

229 *$\delta^{13}C$ as a proxy of habitat*

230      $\delta^{13}C$ values for samples collected in 2007/08 and 2016/17 are given in Tables

231 1 and 2 respectively. Among whole zooplankton, $\delta^{13}C$ values ranged from -25.64 ‰

232 in krill (*Euphausia superba*) to -20.48 ‰ in the amphipod *Parandania boecki*. For

233 secondary consumers (myctophid fish and squid), $\delta^{13}C$ values in muscle tissue ranged

234 from -25.67 ‰ (*Electrona antarctica*) to -20.15 ‰ (*Slosarczykovia circumantarctica*),

235 while for notothenioid fish, it was from -23.80 ‰ in *Champsocephalus gunnari* to -

236 20.60 ‰ in *Dissostichus eleginoides.* Values of $\delta^{13}C$ in seabird feathers tended to be

237 higher (i.e. more enriched in ^{13}C), ranging from -22.82 ‰ in Antarctic prions

238 (*Pachyptila desolata*) to -15.24 ‰ in brown skuas (*Stercorarius antarcticus*) (Table 1)*.*

239 There were only significant differences in $\delta^{13}C$ values between seabirds and the other

240 trophic groups within each sampling year (Kruskal-Wallis test, H = 50.23, p<0.0001,

241 in 2007/08; H = 63.69, p < 0.0001 in 2016/17). There were no differences between the

242 other groups.

243   Within the seabird group, brown skuas had significantly higher $\delta^{13}C$ values than

244 northern giant petrels (*Macronectes halli*) in 2007/08 (Kruskal-Wallis test, H = 15.932,

245 p = 0.007; Dunn's multiple comparisons test, p = 0.0028; Table 1). In 2016/17,

246 significant differences among seabirds were also detected (Kruskal-Wallis test, H =

247 22.552, p < 0.0001; Table 2) and these were mainly driven by the higher value in

248 brown skuas compared with Antarctic prions (Dunn's multiple comparisons test, p =

249 0.0013), and blue petrels *Halobaena caerulea* (Dunn's multiple comparisons test, p =

250 0.0144).

251   Within the notothenioid fish (Kruskal-Wallis test, H = 24.794, p = 0.0002; Table

252 2), *N. gibberifrons* had significant higher $\delta^{13}C$ than *C. gunnari* (Dunn's multiple

253 comparisons test, p = 0.0002) and *P. guntheri* (Dunn's multiple comparisons test, p =

254 0.00013). No significant differences in $\delta^{13}C$ were observed between any myctophid

255 fish in 2007/08 (Kruskal-Wallis test, H = 16.141, p = 0.061; Table 1) whereas, in

256 2016/17, $\delta^{13}C$ values were significantly lower in *E. antarctica* than in *Gymnoscopelus*

257 *nicholsi* (Kruskal-Wallis test, H = 16. 751, p = 0.0049; Dunn's multiple comparisons

258 test, p = 0.0115; Table 2).

259       $\delta^{13}C$ values were not significantly different between sampling years either in

260 squid (Wilcoxon rank test, W=3, p = 0.25, in 2007/08; W=0, p = 0.2 in 2017) or

261 zooplankton (Kruskal-Wallis test, H = 5.1538, p = 0.741, in 2007; H = 19.19, p = 0.1388

262 in 2016/17; Tables 1 & 2).

263

264 *$\delta^{15}N$ as a proxy of trophic structure*

265       $\delta^{15}N$ values for each species in 2007/08 and 2016/17 are given in Tables 1 and

266 2, respectively. $\delta^{15}N$ was significantly higher in seabirds (7.69 ‰ in Antarctic prions to

267 15.50 ‰ in wandering albatrosses (*Diomedea exulans*) and notothenioid fish (only in

268 2016/17; 7.71 ‰ in *Patagonotothen guntheri* to 14.41 ‰ in *N. rossii*) than in the other

269 taxonomic groups (Kruskal-Wallis test, H = 59.184, p < 0.0001, H = 80.284, p < 0.0001

270 for birds and fish respectively). Moreover, values decreased from myctophid fish (6.69

271 ‰ in *E. antarctica* to 10.91‰ in *G. opisthopterus*), to squid (5.68 ‰ in *S.*

272 *circumantarctica* to 9.38 ‰ in *Galiteuthis glacialis*) and then zooplankton (1.73 ‰ in

273 *E. superba* to 8.81 ‰ in *Gigantocypris* sp.).

274   In 2007/08, there were significant differences in δ^{15}N values among seabird

275   species (Kruskal-Wallis test, H = 20.979, p = 0.0008, Table 1). Wandering albatrosses

276   had significantly higher δ^{15}N than black-browed albatrosses (*Thalassarche*

277   *melanophris*; Dunn's multiple comparisons test, p = 0.0043), grey-headed albatrosses

278   (*Thalassarche chrysostoma;* Dunn's multiple comparisons test, p = 0.0340) and

279   southern giant petrels (*M. giganteus;* Dunn's multiple comparisons test, p = 0.0049).

280   In 2016/17, there were also differences among seabird species (Kruskal-Wallis test,

281   H = 21.756, p = 0.001 Table 2) with Antarctic prions having significantly lower δ^{15}N

282   values than brown skuas (Dunn's multiple comparisons test, p = 0.0251) and grey-

283   headed albatrosses (Dunn's multiple comparisons test, p = 0.0202).

284   δ^{15}N values also differed significantly among species of notothenioid fish

285   (Kruskal-Wallis test, H = 25.847, p < 0.001, Table 2) with *C. gunnari* and *P. guntheri*

286   having significantly lower values than *N. gibberifrons* (Dunn's multiple comparisons

287   test, p = 0.0045; p = 0.0273) and *D. eleginoides* (Dunn's multiple comparisons test, p

288   = 0.0029; p = 0.0124). No significant differences in δ^{15}N were detected between

289   species of myctophid fish (Kruskal-Wallis test, H = 7.996, p = 0.238, in 2007/08; H =

290   12.888, p = 0.244; Tables 1 & 2) and squid (Wilcoxon rank test, W=11, p = 0.393, in

291   2007/08; W=6, p = 0.200 in 2016/17; Tables 1 & 2) in either of the sampling years.

292   Significant differences in δ^{15}N were only observed between two species of

293   zooplankton (Kruskal-Wallis test, H = 20.03, p = 0.010; Table 2), *E. superba* and *P.*

294   *boecki* in 2007/08 (Dunn's multiple comparisons test, p = 0.0032).

295

296   *Mercury concentrations*

297   In both sampling years, seabirds had the highest Hg concentrations among the

298   analysed species. In seabirds, Hg concentrations ranged from 0.12 µg g^{-1} in Antarctic

299   prions to 7.17 µg g^{-1} in brown skuas; the next highest values were in myctophid fish

300   (0.025 µg g^{-1} in *Krefftichthys anderssoni* to 0.352 µg g^{-1} in *G. nicholsi*) and notothenioid

301   fish (0.007 µg g^{-1} in *C. gunnari* to 0.343 µg g^{-1} in *N. rossii*), squid (0.012 µg g^{-1} in *S.*

302   *circumantarctica* to 0.066 µg g^{-1} in *G. glacialis*) and zooplankton (0.006 µg g^{-1} in *E.*

303   *superba* to 0.141 µg g^{-1} in *P. boecki*). As expected, POM had the lowest Hg

304   concentrations (0.0005 ± 0.0002 µg g^{-1}). Highly significant differences were found in

305   total Hg concentrations between taxonomic groups (including POM), within both

306   sampling years (Kruskal-Wallis test, H = 59.75, p < 0.0001, in 2007/08; H = 82.42, p

307   < 0.0001 in 2016/17).

308   Significant differences in Hg concentrations were observed among seabird

309   species both in 2007/08 (Kruskal-Wallis test, H = 22.621, p = 0.0003; Table 1) and

310   2016/17 (Kruskal-Wallis test, H = 24.175, p = 0.0005; Table 2). In 2007/08, wandering

311   albatrosses showed the highest concentrations, followed by brown skuas, grey-

312   headed albatrosses, black-browed albatrosses, southern giant petrels and northern

313   giant petrels. In 2016/17, brown skuas had the highest concentrations followed by

314   northern giant petrels, southern giant petrels, black-browed albatrosses, grey-headed

315   albatrosses, blue petrels and Antarctic prions.

316   In notothenioid fish, there were highly significant differences between species

317   (Kruskal-Wallis test, H = 27.795, p < 0.0001). In particular, *C. gunnari* had significantly

318   lower Hg concentrations than *N. gibberifrons, N. rossii* and *D. eleginoides* (Dunn's

319   multiple comparisons test, p = 0.0004; p = 0.0009, p = 0.0028; Table 2). Myctophids

320   showed differences between the species in both sampling years (Kruskal-Wallis test,

321   H = 15.317, p = 0.018, in 2007/08; H = 13.52, p =0.019 in 2016/17; Tables 1 & 2), with

322   *G. nicholsi* having a higher Hg concentration than *P. bolini* (Dunn's multiple

323   comparisons test, p = 0.0368 in 2007/08; p = 0.0375 in 2016/17) and *K. anderssoni*

324 (Dunn's multiple comparisons test, p = 0.0325 in 2007/08; p = 0.0381 in 2016/17).

325 There were no significant differences in Hg concentrations between squid species in

326 either year (Wilcoxon rank test, W=15, p = 0.057, in 2007/08; W=4, p = 0.8000 in

327 2016/17) nor among zooplankton species in 2007/8 (Kruskal-Wallis test, H = 7.4231,

328 p = 0.492). However, in 2016/17, there were significant differences between species

329 in 2016/2017 (Kruskal-Wallis test, H = 20.445, p =0.0088).

330 Significant positive linear regressions were found between $\log_{10}$Hg

331 concentrations and δ^{15}N values across all species in both years (Y = 0.2112*X −

332 2.8088, r^2 = 0.7904, p < 0.0001 in 2007/8; Y = 0.2451*X − 3.0363, r^2 = 0.8419, p <

333 0.0001 in 2016/17).

334

335 *Comparison between years*

336 Five of the eight seabird species were sampled in both study years (brown

337 skuas, grey-headed and black-browed albatrosses, northern and southern giant

338 petrels). With regards to δ^{15}N, significant differences between 2007/08 and 2016/17

339 values were only found for northern giant petrels (t-test, t_8 = 3.907, p = 0.0045),

340 whereas δ^{13}C, values differed in most seabirds with the exception of northern giant

341 petrels (t-test, t_9 = 1.693, p = 0.1246). Hg concentrations were generally higher in

342 2016/17 than in 2007/08, but this was significant only for northern giant petrels (Mann

343 Whitney test, U=0, p = 0.0079) and southern giant petrels (t_8 = 6.322, p < 0.005).

344 In myctophids, five out of the seven species were caught in both sampling years

345 (*E. antarctica, G. nicholsi, G. braueri, K. anderssoni* and *P. bolini*). δ^{15}N and δ^{13}C were

346 similar in all species between the two periods. Unlike seabirds, Hg concentrations

347 were lower in 2016/17 than in 2007/08, and significantly so in the case of *E. antarctica*

348 (Mann Whitney test U=45, p = 0.002). Six species of zooplankton were caught in both

349 sampling years (*E. triacantha, Parandania boecki, Gigantocypris* sp., *E. superba,*

350 *Thysanoessa* sp*., Themisto gaudichaudii*). No significant differences in δ^{15}N, δ^{13}C or

351 Hg were detected between any zooplankton species between 2007/08 and 2016/17.

352   There were significant positive correlations between Log$_{10}$Hg concentrations

353 and δ^{15}N values in the species that were sampled in both years (Y = 0. 2028*X -

354 2.6008, r^2 = 0. 8138, p < 0.0001 in 2007/8; Y = 0.2782*X - 3.0960, r^2 = 0.9314, p <

355 0.0001 in 2016/17) (Figure 1). The slope of the relationship was significantly higher in

356 2016/17 (0.2782) than 2007/08 (0.2028) (ANOVA, $F_{1, 110}$ = 9.716, P=0.0023.)

357

358 **Discussion**

359   A number of studies have used nitrogen stable isotope values (δ^{15}N) to

360 contextualise the biomagnification of Hg within food webs in different ecosystems

361 (Cabana and Rasmussen, 1994; Lavoie et al., 2013). However, to the best of our

362 knowledge, this study is the first to evaluate Hg bioaccumulation from POM to top

363 predators in a Southern Ocean food web.

364

365 *Influence of feeding habitat*

366   The first assumption when evaluating food web structure and links using stable

367 isotope ratios is that all the analysed species share broadly the same environment.

368 This assumption is required to rule out the potential for any spatial variation in baseline

369 δ^{15}N in apparent trophic level within geographically distinct ecosystems (Chouvelon et

370 al., 2012). Ecosystem connectedness can be determined using δ^{13}C as a proxy for

371 habitat (Kelly, 2000), and all taxonomic groups in our study had broadly similar δ^{13}C

372 values (Tables 1 and 2). The greatest differences in δ^{13}C values were observed in

373 seabirds, which is to be expected as feathers are more enriched in ^{13}C than blood,

374 muscle and internal organs (Cherel et al., 2014; Kelly, 2000), and we assert that the

375 distinct seabird $\delta^{13}C$ values are a consequence of the use of different tissues rather

376 than any spatial separation. This assertion is backed up by tracking data during chick-

377 rearing, which shows that all the sampled seabird species forage broadly in the same

378 areas where the myctophids, squid and zooplankton were caught [wandering

379 albatrosses (Jiménez et al., 2015); southern giant petrels and northern giant petrels

380 (Granroth-Wilding and Phillips, 2018a); brown skuas (Carneiro et al., 2014); Antarctic

381 prions and blue petrel (Navarro et al., 2013); black-browed albatrosses and grey-

382 headed albatrosses (Phillips et al., 2004b)]. As expected, there were small differences

383 in $\delta^{13}C$ values among some species within taxonomic groups, indicating a limited

384 degree of divergence in foraging habits (e.g. feeding depth), but this has no material

385 effect on our conclusions about trophic level based on $\delta^{15}N$. Hg concentrations

386 throughout the sampled food web were not influenced by differences in habitat

387 utilization (inferred from the $\delta^{13}C$ values), as our analysis reflects a single regional

388 system in the Scotia Sea in the vicinity of South Georgia.

389

390 *Trophic structure in relation to mercury concentrations*

391 Our combined field sampling provided two snapshots, 9 years apart, of Hg

392 concentrations in the Scotia Sea food web. Values of Hg were lowest in POM, followed

393 by zooplankton, which as a group exhibited some degree of variation. This is not

394 surprising since the zooplankton species examined range from predominantly

395 herbivorous Antarctic krill (Quetin and Ross, 1991) to predators such as *Themisto*

396 *gaudichaudii* (Havermans et al., 2019). Squid and myctophid fish link zooplankton to

397 the top predators (seabirds) and, in terms of Hg dynamics, exhibit some degree of

398 variation between individuals and species in Hg concentrations [for more details see

399 (Seco et al., 2020a; 2020b)]. Generally, the highest Hg concentrations were found in

400 predators higher in the food web (notothenioid fish and seabirds). Our first clear result

401 is therefore that total Hg concentration increases with trophic level (reflected by δ^{15}N).

402 This relationship was expected as a result of Hg biomagnification through food webs,

403 and has previously been reported by other studies in the Antarctic region (Anderson

404 et al., 2009).

405   Trophic links to the two groups of top predators analysed here, seabirds and

406 notothenioid fish (the latter taxa are sometimes considered to be mesopredators, as

407 they can themselves be predated by larger species), can be considered as two parallel

408 paths in the Scotia Sea food web, because both groups of top predators occupy similar

409 trophic positions (see Tables 1 and 2). However, our results reveal that the seabirds

410 can be differentiated into three distinct groups with characteristic Hg levels: wandering

411 albatrosses and brown skuas had the highest Hg concentrations of all seabird species

412 (1.556 – 7.173 µg g^{-1}; grey-headed albatrosses, black-browed albatrosses, southern

413 giant petrels and northern giant petrels, had intermediate Hg levels (0.1216 – 2.952

414 µg g^{-1}), and Antarctic prions and blue petrel had the lowest Hg levels (0.1198 – 0.9591

415 µg g^{-1}). This is a classic effect of diet on Hg levels in seabirds: seabirds that feed on

416 zooplankton have lower concentrations than fish- and squid- eating species (Bocher

417 et al., 2003; Carravieri et al., 2014b). Wandering albatrosses feed on higher trophic

418 level prey, mainly large fish and squid (Moreno et al., 2016), which our study shows

419 had relatively high Hg concentrations. Brown skuas have a diverse diet, feeding on

420 Antarctic fur seal *Arctocephalus gazella* carrion (including placentae), other seabirds

421 and occasionally fish or squid which they obtain via kleptoparasitism (Phillips et al.

422 2004a). Carravieri et al., (2014b) showed that brown skua chicks at Kerguelen have

423 higher Hg feather concentrations than the adults, probably because their diet consists

424 mainly of blue petrels, whereas adults during the non-breeding period consume a more

425 diverse diet, including marine prey. High diet diversity probably also explains the high

426 variability in Hg and $\delta^{15}N$ in grey-headed albatrosses and black-browed albatrosses,

427 which feed on fish, squid and crustaceans (Prince, 1980a). Northern giant petrels and

428 southern giant petrels are more generalist, feeding both on carrion on land (from

429 Antarctic fur seals and penguins) and Antarctic krill, squid and other seabirds (Hunter,

430 1983). Additionally, both species present sex differences in feeding habits, with males

431 predominantly scavenging during the early-mid breeding season, whereas females

432 mostly forage at sea (Gonzáles-Solís et al., 2000; Granroth-Wilding and Phillips,

433 2018b). The smaller seabirds in our study, Antarctic prions and blue petrel, feed mainly

434 on zooplankton, including Antarctic krill (Prince, 1980b). They are therefore exposed

435 to lower Hg concentrations than seabirds that are piscivorous or teuthophageous

436 seabirds, as reported in other breeding areas (Blévin et al. 2013, Carravieri et al.

437 2014). Thus, the smaller bird species have Hg concentrations in the same range as

438 some fish and squid species, probably also because chicks have had only a short

439 exposure period, which reduces the potential for Hg bioaccumulation too.

440 Differences in Hg concentrations were also found between the notothenioid fish

441 species. Hg concentrations were lower in *C. gunnari* than in all the other species,

442 which is consistent with the reported results for these fish at the Kerguelen Islands

443 (Bustamante et al., 2003; Cipro et al., 2018). These differences only partly reflect

444 trophic level, as $\delta^{15}N$ was similar in *C. gunnari* and *P. guntheri*. Nor does the

445 discrepancy appear to mirror what is known about diet, as both species are considered

446 to feed mostly on euphausiids and amphipods (Collins et al., 2008; di Prisco et al.,

447 1991). Furthermore, the influence of body size can be discounted, as *C. gunnari* were

448 on average 67 mm bigger than *P. guntheri*, and larger fish tend to have higher Hg

449 concentrations in species with similar growth rates (Dang and Wang, 2012; Gewurtz

450 et al., 2011). Instead, the low Hg concentrations seem more likely to relate to the highly

451 specialized physiological characteristics of *C. gunnari*, including the absence of

452 haemoglobin (Sidell and O'Brien, 2006), which is associated with lower metabolic

453 rates (Johnston and Camm, 1987). However, this would need to be confirmed by

454 further studies.

455   As in seabirds, the notothenioid fish species at higher trophic levels had higher

456 Hg concentrations. *D. eleginoides* feeds mostly on fish, squid and crustaceans (Collins

457 et al., 2010; 2007; Seco et al., 2015). In contrast, *N. gibberifrons* and *N. rossii* feed on

458 diverse prey from algae to amphipods, euphausiids and other fish (Casaux et al.,

459 1990), which would also explain the large variability in their $\delta^{15}N$ values. The diet of *C.*

460 *aceratus* is dominated by crustaceans and fish (Reid et al., 2007). Together, these

461 guild-specific results highlight the important role of diet in Hg bioaccumulation.

462   The slope of the linear regression between $\log_{10}(Hg)$ concentration and $\delta^{15}N$

463 values, also known as the trophic magnification slope (TMS), is an indicator of Hg

464 biomagnification potential in a food web (Lavoie et al., 2013; Yoshinaga et al., 1992).

465 Hg TMS can be influenced not only by food web dynamics but also by habitat

466 characteristics (Lavoie et al., 2013). At lower latitude regions, slower growth rates and

467 slower Hg excretion rates, due to colder temperatures, could lead to greater

468 biomagnification of Hg (0.21 ± 0.07; Lavoie et al., 2013) that the at higher latitudes,

469 where higher primary productivity and growth rates (Gross et al., 1988; Pauly 1998)

470 may lead to lower TMS (0.16 ± 0.08; Lavoie et al., 2013). TMS values for the Scotia

471 Sea ecosystem were 0.267 for 2007/08 and 0.200 for 2016/17, which are both within

472 the range of those previously reported for polar regions (Lavoie et al., 2013).

473

Interannual variation in mercury concentration

When comparing species collected in both sampling years, Hg concentrations

in the mid trophic-level groups (squid and myctophid) were lower in 2016/17 than in

2007/08, as reported in detail by Seco et al. (2020b; 2020a). In contrast, Hg

concentrations were higher in the seabirds sampled in 2016/17 than in 2007/08, which

is the opposite pattern to the mid trophic-level species (Seco et al., 2020a). This was

unexpected as the Hg body burdens of predators should, in theory, reduce if there is

less bioavailable Hg in their prey (Atwell et al., 1998). A plausible explanation is that

there was a change in the main trophic pathway between years (Ward et al., 2010).

Differences in Hg levels in producers as well as changes in the productivity and food

web structure can directly influence Hg concentration in top predators (Ward et al.,

2010). The Scotia Sea food web is centred on Antarctic krill (Murphy et al., 2007), the

abundance of which can determine the reproductive success and survival of

dependent predators (Grünbaum and Veit, 2003; Lynnes et al., 2004; Seyboth et al.,

2016). In years with low Antarctic krill abundance, predators switch to alternative food

sources, particularly myctophid fish or squid (Murphy et al., 2007; Mills et al., 2020).

In our study, Antarctic krill had amongst the lowest Hg concentrations, suggesting that

in situations when less krill is available predators, a switch to alternative prey with

higher Hg burdens (Figure. 2B) will result in predators being subjected to greater levels

of Hg bioaccumulation. Abundance of Antarctic krill based on acoustic surveys to the

northwest of South Georgia was relatively low in 2016/17 (BAS, unpublished data)

when compared with previous years (Fielding et al., 2014). Our Hg data possibly

illustrates the knock-on effect of variable Antarctic krill abundance on food web

dynamics, particularly the effect on higher predators that cascade through the food

web. This is of particular interest in light of studies that suggest there is a long-term

499 decline in abundance of Antarctic krill in the Scotia Sea (Atkinson et al., 2019; Hill et

500 al., 2019; Rintoul et al., 2018). In years of low Antarctic krill abundance, predators not

501 only have to cope with the stress of reduced prey availability, but with a concomitant

502 increase in Hg exposure. Our results highlight that changes in food web dynamics,

503 particularly temporal switches between krill-based and non-krill-based trophic

504 pathways, are likely to be important in the transport of Hg between mid-trophic levels

505 and apex predators in the Southern Ocean.

506

521

522 **References**

523

524 Ackerman, J.T., Eagles-Smith, C.A., Heinz, G., La Cruz, De, S.E.W., Takekawa,
525   J.Y., Miles, A.K., Adelsbach, T.L., Herzog, M.P., Bluso-Demers, J.D., Demers,
526   S.A., Herring, G., Hoffman, D.J., Hartman, C.A., Willacker, J.J., Suchanek, T.H.,
527   Schwarzbach, S.E., Maurer, T.C., 2014. Mercury in birds of San Francisco Bay-
528   Delta, California—Trophic pathways, bioaccumulation, and ecotoxicological risk
529   to avian reproduction: U.S. Geological Survey Open-File Report.
530   doi:10.3133/ofr20141251
531 Anderson, O.R.J., Phillips, R.A., McDonald, R.A., Shore, R.F., Mcgill, R.A.R.,
532   Bearhop, S., 2009. Influence of trophic position and foraging range on mercury
533   levels within a seabird community. Mar. Ecol. Prog. Ser. 375, 277–288.
534   doi:10.3354/meps07784
535 Atkinson, A., Hill, S.L., Pakhomov, E.A., Siegel, V., Reiss, C.S., Loeb, V.J.,
536   Steinberg, D.K., Schmidt, K., Tarling, G.A., Gerrish, L., Sailley, S.X.V.F., 2019.
537   Krill (*Euphausia superba*) distribution contracts southward during rapid regional
538   warming. Nature Climate Change 9, 142–147. doi:10.1038/s41558-018-0370-z
539 Atkinson, A., Whitehouse, M.J., Priddle, J., Cripps, G.C., Ward, P., Brandon, M.A.,
540   2001. South Georgia, Antarctica: a productive, cold water, pelagic ecosystem†.
541   Mar. Ecol. Prog. Ser. 216, 279–308. doi:10.3354/meps216279
542 Atwell, L., Hobson, K.A., Weich, H.E., 1998. Biomagnification and bioaccumulation of
543   mercury in an arctic marine food web: insights from stable nitrogen isotope
544   analysis. Can. J. Fish. Aquat. Sci. 55, 1114–1121. doi:10.1139/f98-001
545 Ballerini, T., Hofmann, E.E., Ainley, D.G., Daly, K., Marrari, M., Ribic, C.A., Smith,
546   W.O., Jr, Steele, J.H., 2014. Productivity and linkages of the food web of the
547   southern region of the western Antarctic Peninsula continental shelf. Prog.
548   Oceanogr 122, 10–29. doi:10.1016/j.pocean.2013.11.007
549 Bargagli, R., 2016. Atmospheric chemistry of mercury in Antarctica and the role of
550   cryptogams to assess deposition patterns in coastal ice-free areas.
551   Chemosphere 163, 202–208. doi:10.1016/j.chemosphere.2016.08.007
552 Barnes, C., Maxwell, D., Reuman, D.C., Jennings, S., 2010. Global patterns in
553   predator-prey size relationships reveal size dependency of trophic transfer
554   efficiency. Ecology 91, 222–232. doi:10.1890/08-2061.1
555 Blévin, P., Carravieri, A., Jaeger, A., Chastel, O., Bustamante, P., Cherel, Y., 2013.
556   Wide Range of Mercury Contamination in Chicks of Southern Ocean Seabirds.
557   PLoS ONE 8, e54508–11. doi:10.1371/journal.pone.0054508
558 Bloom, N.S., 1992. On the chemical form of mercury in edible fish and marine
559   invertebrate tissue. Can. J. Fish. Aquat. Sci. 49, 1010–1017. doi:10.1139/f92-113
560 Bocher, P., Caurant, F., Miramand, P., Cherel, Y., Bustamante, P., 2003. Influence
561   of the diet on the bioaccumulation of heavy metals in zooplankton-eating petrels
562   at Kerguelen archipelago, Southern Indian Ocean. Polar Biol. 26, 759–767.
563   doi:10.1007/s00300-003-0552-6
564 Boltovskoy, D., 1999. South Atlantic Zooplankton. Backhuys Publishers, Leide,
565   Netherland.
566 Bromwich, D.H., 1989, 1989. An extraordinary katabatic wind regime at Terra Nova
567   Bay, Antarctica. Mon. Wea. Rev. 117, 688–695. doi:10.1175/1520-0493
568 Brooks, S., LINDBERG, S., SOUTHWORTH, G., ARIMOTO, R., 2008. Springtime
569   atmospheric mercury speciation in the McMurdo, Antarctica coastal region.
570   Atmos Environ. 42, 2885–2893. doi:10.1016/j.atmosenv.2007.06.038
571 Bustamante, P., Bocher, P., Cherel, Y., Miramand, P., Caurant, F., 2003. Distribution
572   of trace elements in the tissues of benthic and pelagic fish from the Kerguelen
573   Islands. Sci. Total Environ. 313, 25–39. doi:10.1016/S0048-9697(03)00265-1

Bustamante, P., Lahaye, V., Durnez, C., Churlaud, C., Caurant, F., 2006. Total and organic Hg concentrations in cephalopods from the North Eastern Atlantic waters: Influence of geographical origin and feeding ecology. Sci. Total Environ. 368, 585–596. doi:10.1016/j.scitotenv.2006.01.038

Cabana, G., Rasmussen, J.B., 1994. Modelling food chain structure and contaminant bioaccumulation using stable nitrogen isotopes. Nature 372, 255–257. doi:10.1038/372255a0

Carneiro, A.P.B., Manica, A., Phillips, R.A., 2014. Foraging behaviour and habitat use by brown skuas *Stercorarius lonnbergi* breeding at South Georgia. Mar Biol 161, 1755–1764. doi:10.1007/s00227-014-2457-z

Carravieri, A., Bustamante, P., Churlaud, C., Fromant, A., Cherel, Y., 2014a. Moulting patterns drive within-individual variations of stable isotopes and mercury in seabird body feathers: implications for monitoring of the marine environment. Mar Biol 161, 963–968. doi:10.1007/s00227-014-2394-x

Carravieri, A., Cherel, Y., Blévin, P., Brault-Favrou, M., Chastel, O., Bustamante, P., 2014b. Mercury exposure in a large subantarctic avian community. Environ. Pollut. 190, 51–57. doi:10.1016/j.envpol.2014.03.017

Casaux, R.J., Mazzotta, A.S., Barrera-Oro, E.R., 1990. Seasonal aspects of the biology and diet of nearshore nototheniid fish at Potter Cove, South Shetland Islands, Antarctica. Polar Biol. 11, 63–72. doi:10.1007/BF00236523

Cherel, Y., Barbraud, C., Lahournat, M., Jaeger, A., Jaquemet, S., Wanless, R.M., Phillips, R.A., Thompson, D.R., Bustamante, P., 2018. Accumulate or eliminate? Seasonal mercury dynamics in albatrosses, the most contaminated family of birds. Environ. Pollut. 241, 124–135. doi:10.1016/j.envpol.2018.05.048

Cherel, Y., Hobson, K.A., 2007. Geographical variation in carbon stable isotope signatures of marine predators: a tool to investigate their foraging areas in the Southern Ocean. Mar. Ecol. Prog. Ser. 329, 281–287.

Cherel, Y., Jaquemet, S., Maglio, A., Jaeger, A., 2014. Differences in $\delta^{13}C$ and $\delta^{15}N$ values between feathers and blood of seabird chicks: implications for non-invasive isotopic investigations. Mar Biol 161, 229–237. doi:10.1007/s00227-013-2314-5

Chouvelon, T., Spitz, J., Caurant, F., Mèndez-Fernandez, P., Chappuis, A., Laugier, F., Le Goff, E., Bustamante, P., 2012. Revisiting the use of $\delta^{15}N$ in meso-scale studies of marine food webs by considering spatio-temporal variations in stable isotopic signatures. The case of an open ecosystem: the Bay of Biscay (north-east Atlantic). Prog. Oceanogr 101, 92–105. doi:10.1016/j.pocean.2012.01.004

Cipro, C.V.Z., Cherel, Y., Bocher, P., Caurant, F., Miramand, P., Bustamante, P., 2018. Trace elements in invertebrates and fish from Kerguelen waters, southern Indian Ocean. Polar Biol. 41, 175–191. doi:10.1007/s00300-017-2180-6

Clarkson, T.W., 1992. Mercury: Major Issues in Environmental Health. Environ. Health Perspect. 100, 31–38. doi:10.1289/ehp.9310031

Coelho, J.P., Mieiro, C.L., Pereira, M.E., Duarte, A.C., Pardal, M.A., 2013. Mercury biomagnification in a contaminated estuary food web: Effects of age and trophic position using stable isotope analyses. Mar. Pollut. Bull. 69, 110–115. doi:10.1016/j.marpolbul.2013.01.021

Coelho, J.P., Reis, A.T., Ventura, S., Pereira, M.E., Duarte, A.C., Pardal, M.A., 2008. Pattern and pathways for mercury lifespan bioaccumulation in *Carcinus maenas*. Mar. Pollut. Bull. 56, 1104–1110. doi:10.1016/j.marpolbul.2008.03.020

Collins, M.A., Brickle, P., Brown, J., Belchier, M., 2010. Provided for non-commercial research and educational use only. Not for reproduction, distribution or

commercial use., 1st ed, Advances in Marine Biology. Elsevier Ltd. doi:10.1016/S0065-2881(10)58004-0

Collins, M.A., Ross, K.A., Belchier, M., Reid, K., 2007. Distribution and diet of juvenile Patagonian toothfish on the South Georgia and Shag Rocks shelves (Southern Ocean). Mar Biol 152, 135–147.

Collins, M.A., Shreeve, R.S., Fielding, S., Thurston, M.H., 2008. Distribution, growth, diet and foraging behaviour of the yellow-fin notothen *Patagonotothen guntheri* (Norman) on the Shag Rocks shelf (Southern Ocean). J Fish Biol 72, 271–286. doi:10.1111/j.1095-8649.2007.01711.x

Cossa, D., Heimbürger, L.-E., Lannuzel, D., Rintoul, S.R., Butler, E.C.V., Bowie, A.R., Averty, B., Watson, R.J., Remenyi, T., 2011. Mercury in the Southern Ocean. Geochim. Cosmochim. Acta 75, 4037–4052. doi:10.1016/j.gca.2011.05.001

Croxall, J.P., Reid, K., Prince, P.A., 1999. Diet, provisioning and productivity responses of marine predators to differences in availability of Antarctic krill. Mar. Ecol. Prog. Ser.

Dang, F., Wang, W.-X., 2012. Why mercury concentration increases with fish size? Biokinetic explanation. Environ. Pollut. 163, 192–198. doi:10.1016/j.envpol.2011.12.026

Dehn, L.-A., Follmann, E.H., Thomas, D.L., Sheffield, G.G., Rosa, C., Duffy, L.K., O'Hara, T.M., 2006. Trophic relationships in an Arctic food web and implications for trace metal transfer. Sci. Total Environ. 362, 103–123. doi:10.1016/j.scitotenv.2005.11.012

DeNiro, M.J., Epstein, S., 1977. Mechanism of carbon isotope fractionation associated with lipid synthesis. Sci 197, 261–263. doi:10.1126/science.327543

di Prisco, G., Maresca, B., Tota, B., 1991. Biology of Antarctic fish. Springer, Berlin.

Ebinghaus, R., Kock, H.H., Temme, C., Einax, J.W., Löwe, A.G., Richter, A., Burrows, J.P., Schroeder, W.H., 2002. Antarctic Springtime Depletion of Atmospheric Mercury. Environ. Sci. Technol. 36, 1238–1244. doi:10.1021/es015710z

Eisele, F., Davis, D.D., Helmig, D., Oltmans, S.J., Neff, W., Huey, G., Tanner, D., Chen, G., Crawford, J., Arimoto, R., Buhr, M., Mauldin, L., Hutterli, M., Dibb, J., Blake, D., Brooks, S.B., Johnson, B., Roberts, J.M., Wang, Y., Tan, D., Flocke, F., 2008. Antarctic Tropospheric Chemistry Investigation (ANTCI) 2003 overview. Atmos Environ. 42, 2749–2761. doi:10.1016/j.atmosenv.2007.04.013

Elizalde, M.R., 2017. Sources and fate of methylmercury in the Southern Ocean : use of model seabirds and mercury stable isotopes. Université de La Rochelle.

Everson, I., 2000. Krill: Biology, Ecology and Fisheries. Blackwell Science, London.

Fielding, S., Watkins, J.L., Trathan, P.N., Enderlein, P., Waluda, C.M., Stowasser, G., Tarling, G.A., Murphy, E.J., 2014. Interannual variability in Antarctic krill (*Euphausia superba*) density at South Georgia, Southern Ocean: 1997–2013. ICES J Mar Sci 71, 2578–2588. doi:10.1093/icesjms/fsu104

Fort, J., Robertson, G.J., Grémillet, D., Traisnel, G., Bustamante, P., 2014. Spatial Ecotoxicology: Migratory Arctic Seabirds Are Exposed to Mercury Contamination While Overwintering in the Northwest Atlantic. Environ. Sci. Technol. 48, 11560–11567. doi:10.1021/es504045g

Gewurtz, S.B., Bhavsar, S.P., Fletcher, R., 2011. Influence of fish size and sex on mercury/PCB concentration: Importance for fish consumption advisories. Environ Int 37, 425–434. doi:10.1016/j.envint.2010.11.005

Gonzáles-Solís, J., Croxall, J.P., Wood, A.G., 2000. Foraging partitioning between giant petrels *Macronectes spp.* and its relationship with breeding population changes at Bird Island, South Georgia. Mar. Ecol. Prog. Ser. 204, 279–288. doi:10.3354/meps204279

Goutte, A., Barbraud, C., Meillère, A., Carravieri, A., Bustamante, P., Labadie, P., Budzinski, H., Delord, K., Cherel, Y., Weimerskirch, H., Chastel, O., 2014. Demographic consequences of heavy metals and persistent organic pollutants in a vulnerable long-lived bird, the wandering albatross. Proc. Biol. Sci. 281, 20133313. doi:10.1098/rspb.2013.3313

Granroth-Wilding, H.M.V., Phillips, R.A., 2018a. Segregation in space and time explains the coexistence of two sympatric sub-Antarctic petrels. Ibis 161, 101–116. doi:10.1111/ibi.12584

Granroth-Wilding, H.M.V., Phillips, R.A., 2018b. Segregation in space and time explains the coexistence of two sympatric sub-Antarctic petrels. Ibis 161, 101–116. doi:10.1111/ibi.12584

Grünbaum, D., Veit, R.R., 2003. Black-browed albatrosses foraging on Antarctic krill: density-dependence through local enhancement? Ecology 84, 3265–3275. doi:10.1890/01-4098

Havermans, C., Auel, H., Hagen, W., Held, C., Ensor, N.S., Tarling, G.A., 2019. Predatory zooplankton on the move: Themisto amphipods in high-latitude marine pelagic food webs, 1st ed, Advances in Marine Biology. Elsevier Ltd. doi:10.1016/bs.amb.2019.02.002

Heneghan, R.F., Hatton, I.A., Galbraith, E.D., 2019. Climate change impacts on marine ecosystems through the lens of the size spectrum. Emerging Topics in Life Sciences 3, 233–243. doi:10.1042/ETLS20190042

Hill, S.L., Atkinson, A., Pakhomov, E.A., Siegel, V., 2019. Evidence for a decline in the population density of Antarctic krill Euphausia superba Dana, 1850 still stands. A comment on Cox et al. J. Crustac. Biol. 39, 316–322. doi:10.1093/jcbiol/ruz004

Hobson, K.A., Piatt, J.F., Pitocchelli, J., 1994. Using stable isotopes to determine seabird trophic relationships. J Anim Ecol 786–798.

Hunter, S., 1983. The food and feeding ecology of the giant petrels *Macronectes halli* and *M. giganteus* at South Georgia. J Zool 200, 521–538. doi:10.1111/j.1469-7998.1983.tb02813.x

Jiménez, S., Domingo, A., Brazeiro, A., Defeo, O., Wood, A.G., Froy, H., Xavier, J.C., Phillips, R.A., 2015. Sex-related variation in the vulnerability of wandering albatrosses to pelagic longline fleets. Anim Conserv 19, 281–295. doi:10.1111/acv.12245

Johnston, I.A., Camm, J.P., 1987. Muscle structure and differentiation in pelagic and demersal stages of the Antarctic teleost *Notothenia neglecta*. Mar Biol 94, 183–190. doi:10.1007/BF00392930

Kelly, J.F., 2000. Stable isotopes of carbon and nitrogen in the study of avian and mammalian trophic ecology. Can. J. Zool. 78, 1–27. doi:10.1139/z99-165

Lavoie, R.A., Jardine, T.D., Chumchal, M.M., Kidd, K.A., Campbell, L.M., 2013. Biomagnification of Mercury in Aquatic Food Webs: A Worldwide Meta-Analysis. Environ. Sci. Technol. 47, 13385–13394. doi:10.1021/es403103t

Lecomte, V.J., Sorci, G., Cornet, S., Jaeger, A., Faivre, B., Arnoux, E., Gaillard, M., Trouvé, C., Besson, D., Chastel, O., Weimerskirch, H., 2010. Patterns of aging in the long-lived wandering albatross. Proc. Natl. Acad. Sci. U.S.A. 107, 6370–6375. doi:10.1073/pnas.0911181107

Lynnes, A.S., Reid, K., Croxall, J., 2004. Diet and reproductive success of Adélie and chinstrap penguins: linking response of predators to prey population dynamics. Polar Biol. 27, 544–554. doi:10.1007/s00300-004-0617-1

Mastromonaco, M.G.N., Gårdfeldt, K., Assmann, K.M., Langer, S., Delali, T., Shlyapnikov, Y.M., Zivkovic, I., Horvat, M., 2017. Speciation of mercury in the waters of the Weddell, Amundsen and Ross Seas (Southern Ocean). Mar. Chem. 193, 20–33. doi:10.1016/j.marchem.2017.03.001

Minagawa, M., Wada, E., 1984. Stepwise enrichment of 15N along food chains: further evidence and the relation between δ15N and animal age. Geochim. Cosmochim. Acta 48, 1135–1140. doi:10.1016/0016-7037(84)90204-7

Moreno, R., Stowasser, G., Mcgill, R.A.R., Bearhop, S., Phillips, R.A., 2016. Assessing the structure and temporal dynamics of seabird communities: the challenge of capturing marine ecosystem complexity. J Anim Ecol 85, 199–212. doi:10.1111/1365-2656.12434

Murphy, E.J., Hofmann, E.E., Watkins, J.L., Johnston, N.M., Piñones, A., Ballerini, T., Hill, S.L., Trathan, P.N., Tarling, G.A., Cavanagh, R.A., Young, E.F., Thorpe, S.E., Fretwell, P., 2013. Comparison of the structure and function of Southern Ocean regional ecosystems: The Antarctic Peninsula and South Georgia. J Marine Syst 109-110, 22–42.

Murphy, E.J., Watkins, J.L., Trathan, P.N., Reid, K., Meredith, M.P., Thorpe, S.E., Johnston, N.M., Clarke, A., Tarling, G.A., Collins, M.A., Forcada, J., Shreeve, R.S., Atkinson, A., Korb, R., Whitehouse, M.J., Ward, P., Rodhouse, P.G., Enderlein, P., Hirst, A.G., Martin, A.R., Hill, S.L., Staniland, I.J., Pond, D.W., Briggs, D.R., Cunningham, N.J., Fleming, A.H., 2007. Spatial and temporal operation of the Scotia Sea ecosystem: a review of large-scale links in a krill centred food web. Phil. Trans. R. Soc. B 362, 113–148. doi:10.1098/rstb.2006.1957

Navarro, J., Votier, S.C., Aguzzi, J., Chiesa, J.J., Forero, M.G., Phillips, R.A., 2013. Ecological Segregation in Space, Time and Trophic Niche of Sympatric Planktivorous Petrels. PLoS ONE 8, e62897–12. doi:10.1371/journal.pone.0062897

O'Driscoll, N.J., Rencz, A., Lean, D., 2005. The biogeochemistry and fate of mercury in the environment. Met Ions Biol Syst 43, 221–238. doi:10.1201/9780824751999.ch9

Pato, P., Otero, M., Válega, M., Lopes, C.B., Pereira, M.E., Duarte, A.C., 2010. Mercury partition in the interface between a contaminated lagoon and the ocean: The role of particulate load and composition. Mar. Pollut. Bull. 60, 1658–1666. doi:10.1016/j.marpolbul.2010.07.004

Phillips, R.A., Phalan, B., Forster, I.P., 2004a. Diet and long-term changes in population size and productivity of brown skuas *Catharacta antarctica* lonnbergi at Bird Island, South Georgia. Polar Biol. 27, 1–7. doi:10.1007/s00300-004-0633-1

Phillips, R.A., Silk, J.R.D., Phalan, B., Catry, P., Croxall, J.P., 2004b. Seasonal sexual segregation in two *Thalassarche* albatross species: competitive exclusion, reproductive role specialization or foraging niche divergence? Proc. R. Soc. Lond. B 271, 1283–1291. doi:10.1098/rspb.2004.2718

Post, D.M., Layman, C.A., Arrington, D.A., Takimoto, G., Quattrochi, J., Montaña, C.G., 2007. Getting to the fat of the matter: models, methods and assumptions for dealing with lipids in stable isotope analyses. Oecologia 152, 179–189. doi:10.1007/s00442-006-0630-x

773 Prince, P.A., 1980a. The food and feeding ecology of Grey-headed Albatross
774   *Diomedea chrysostoma* and Black-browed Albatross *D. melanophris*. Ibis 122,
775   476–488. doi:10.1111/j.1474-919X.1980.tb00902.x
776 Prince, P.A., 1980b. The food and feeding ecology of Blue petrel (*Halobaena*
777   *caerulea*) and Dove prion (*Pachyptila desolata*). J Zool 190, 59–76.
778   doi:10.1111/j.1469-7998.1980.tb01423.x
779 Quetin, L.B., Ross, R.M., 1991. Behavioral and Physiological Characteristics of the
780   Antarctic Krill, Euphausia superba. Amer. Zool. 31, 49–63.
781 Reid, W.D.K., Clarke, S., Collins, M.A., Belchier, M., 2007. Distribution and ecology
782   of *Chaenocephalus aceratus* (Channichthyidae) around South Georgia and Shag
783   Rocks (Southern Ocean). Polar Biol. 30, 1523–1533. doi:10.1007/s00300-007-
784   0313-z
785 Rintoul, S.R., Chown, S.L., DeConto, R.M., England, M.H., Fricker, H.A., Masson-
786   Delmotte, V., Naish, T.R., Siegert, M.J., Xavier, J.C., 2018. Choosing the future
787   of Antarctica. Nature 558, 233–241. doi:10.1038/s41586-018-0173-4
788 Roe, H.S.J., Shale, D.M., 1979. A new multiple rectangular midwater trawl (RMT
789   1+8M) and some modifications to the institute of oceanographic sciences' RMT
790   1+8. Mar Biol 50, 283–288. doi:10.1007/BF00394210
791 Saunders, R.A., Hill, S.L., Tarling, G.A., Murphy, E.J., 2019. Myctophid Fish (Family
792   Myctophidae) Are Central Consumers in the Food Web of the Scotia Sea
793   (Southern Ocean). Front. Mar. Sci. 6, 142. doi:10.3389/fmars.2019.00530
794 Seco, J., Roberts, J., Ceia, F.R., Baeta, A., Ramos, J.A., Paiva, V.H., Xavier, J.C.,
795   2015. Distribution, habitat and trophic ecology of Antarctic squid *Kondakovia*
796   *longimana* and *Moroteuthis knipovitchi*: inferences from predators and stable
797   isotopes. Polar Biol. doi:10.1007/s00300-015-1675-2
798 Seco, J., Xavier, J.C., Brierley, A.S., Bustamante, P., Coelho, J.P., Gregory, S.,
799   Fielding, S., Pardal, M.A., Pereira, B., Stowasser, G., Tarling, G.A., Pereira,
800   M.E., 2020a. Mercury levels in Southern Ocean squid: Variability over the last
801   decade. Chemosphere 239, 124785. doi:10.1016/j.chemosphere.2019.124785
802 Seco, J., Xavier, J.C., Bustamante, P., Coelho, J.P., Saunders, R.A., Ferreira, N.,
803   Fielding, S., Pardal, M.A., Stowasser, G., Viana, T., Tarling, G.A., Pereira, M.E.,
804   Brierley, A.S., 2020b. Main drivers of mercury levels in Southern Ocean lantern
805   fish Myctophidae. Environ. Pollut. 264, 114711–10.
806   doi:10.1016/j.envpol.2020.114711
807 Seyboth, E., Groch, K.R., Rosa, L.D., Reid, K., Flores, P.A.C., Secchi, E.R., 2016.
808   Southern Right Whale (*Eubalaena australis*) Reproductive Success is Influenced
809   by Krill (*Euphausia superba*) Density and Climate. Scient. Rep. 1–8.
810   doi:10.1038/srep28205
811 Sidell, B.D., O'Brien, K.M., 2006. When bad things happen to good fish: the loss of
812   hemoglobin and myoglobin expression in Antarctic icefishes. J. Exp. Biol. 209,
813   1791–1802. doi:10.1242/jeb.02091
814 Stowasser, G., Atkinson, A., Mcgill, R.A.R., Phillips, R.A., Collins, M.A., Pond, D.W.,
815   2012. Food web dynamics in the Scotia Sea in summer A stable isotope study.
816   Deep Sea Research Part II: Topical Studies in Oceanography 59-60, 208–221.
817 Tartu, S., Bustamante, P., Goutte, A., Cherel, Y., Weimerskirch, H., Bustnes, J.O.,
818   Chastel, O., 2014. Age-Related Mercury Contamination and Relationship with
819   Luteinizing Hormone in a Long-Lived Antarctic Bird. PLoS ONE 9, e103642.
820   doi:10.1371/journal.pone.0103642.t004
821 Tavares, S., Xavier, J.C., Phillips, R.A., Pereira, M.E., Pardal, M.A., 2013. Influence
822   of age, sex and breeding status on mercury accumulation patterns in the

823   wandering albatross *Diomedea exulans*. Environ. Pollut. 181, 315–320.
824   doi:10.1016/j.envpol.2013.06.032
825 Thompson, D.R., Bearhop, S., Speakman, J.R., Furness, R.W., 1998. Feathers as a
826   means of monitoring mercury in seabirds: insights from stable isotope analysis.
827   Environ. Pollut. 101, 193–200. doi:10.1016/s0269-7491(98)00078-5
828 Walton, D., 2013. Antarctica: global science from a Frozen Continent.
829 Wania, F., Mackay, D., 1996. The global fractionation of persistent organic
830   pollutants. Environ. Sci. Technol. 30, 390.
831 Ward, D.M., Nislow, K.H., Folt, C.L., 2010. Bioaccumulation syndrome: identifying
832   factors that make some stream food webs prone to elevated mercury
833   bioaccumulation. Ann. N.Y. Acad. Sci. 1195, 62–83. doi:10.1111/j.1749-
834   6632.2010.05456.x
835 Xavier, J.C., Croxall, J.P., Reid, K., 2003. Interannual variation in the diets of two
836   albatross species breeding at South Georgia: implications for breeding
837   performance. Ibis 145, 593–610.
838 Xavier, J.C., Peck, L.S., 2015. Life Beyond the Ice, in: Exploring the Last Continent:
839   an Introduction to Antarctica, Marine Ecosystems in the Southern Ocean.
840   Springer International Publishing, Cham, pp. 229–252. doi:10.1007/978-3-319-
841   18947-5_12
842 Xavier, J.C., Trathan, P.N., Ceia, F.R., Tarling, G.A., Adlard, S., Fox, D., Edwards,
843   E.W.J., Vieira, R.P., Medeiros, R., De Broyer, C., Cherel, Y., 2017. Sexual and
844   individual foraging segregation in Gentoo penguins *Pygoscelis papua* from the
845   Southern Ocean during an abnormal winter. PLoS ONE 12, e0174850.
846   doi:10.1371/journal.pone.0174850
847 Xavier, J.C., Velez, N., Trathan, P.N., Cherel, Y., De Broyer, C., Cánovas, F., Seco,
848   J., Ratcliffe, N., Tarling, G.A., 2018. Seasonal prey switching in non-breeding
849   gentoo penguins related to a wintertime environmental anomaly around South
850   Georgia. Polar Biol. 41, 2323–2335. doi:10.1007/s00300-018-2372-8
851 Yoshinaga, J., Suzuki, T., Hongo, T., Minagawa, M., Ohtsuka, R., Kawabe, T.,
852   Inaoka, T., Akimichi, T., 1992. Mercury concentration correlates with the nitrogen
853   stable isotope ratio in the animal food of Papuans. Ecotoxicol. Environ. Saf. 24,
854   37–45. doi:10.1016/0147-6513(92)90033-Y
855

Table 1- Mercury (Hg) concentration, δ^{13}C and δ^{15}N values of different species sampled on land or caught in the waters around South Georgia in the 2007/08 austral summer (mean ± standard deviation). The superscript numbers ([1]) and letters ([a]) denotes statistical significance from within the sampling year and comparison between years, respectively.

Species	n	Hg ($\mu g\ g^{-1}$)	$\delta^{15}N$ (‰)	$\delta^{13}C$ (‰)
Zooplankton				
Parandania boecki	15	0.02 ± 0.01[1,a]	8.34 ± 0.42[1,a]	-21.90 ± 1.70[1,a]
Euphausia triacantha	20	0.03 ± 0.01[1,a]	6.71 ± 0.55[1,2,a]	-22.15 ± 1.47[1,a]
Gigantocypris sp.	15	0.03 ± 0.01[1,a]	8.81 ± 0.98[1,2,a]	-22.27 ± 0.41[1,a]
Salpa thompsoni	10	0.03 ± 0.01[1]	4.40 ± 0.22[1,2,]	-23.75 ± 1.32[1]
Tomopteris sp.	6	0.03 ± 0.01[1]	7.25 ± 0.73[1,2,]	-22.64 ± 0.65[1]
Euphausia superba	20	0.04 ± 0.02[1,a]	3.47 ± 2.45[2,a]	-23.17 ± 2.86[1,a]
Themisto gaudichaudii	20	0.04 ± 0.02[1,a]	5.86 ± 0.56[1,2,a]	-24.10 ± 0.45[1,a]
Thysanoessa sp.	20	0.05 ± 0.01[1,a]	6.94 ± 0.32[1,2,a]	-23.99 ± 0.71[1,a]
Sagitta sp.	3	0.06 ± 0.01[1]	8.21 ± 0.84[1,2,]	-22.97 ± 1.03[1]
Squid				
Slosarczykovia circumantarctica	5	0.02 ± 0.01[1,a]	6.72 ± 0.25[1,a]	-21.64 ± 1.88[1,b]
Galiteuthis glacialis	5	0.09 ± 0.01[1,a]	7.66 ± 1.49[1,a]	-23.58 ± 1.05[1,a]
Myctophid fish				
Krefftichthys anderssoni	5	0.04 ± 0.01[2,a]	8.21 ± 0.95[1,a]	-20.96 ± 0.19[1,a]
Protomyctophum bolini	5	0.09 ± 0.01[2,a]	8.87 ± 0.88[1,a]	-21.82 ± 2.13[1,a]
Electrona carlsbergi	5	0.14 ± 0.02[2,3]	8.31 ± 1.12[1]	-20.91 ± 0.83[1,]
Gymnoscopelus braueri	5	0.16 ± 0.03[2,3,a]	9.77 ± 0.56[1,a]	-23.63 ± 0.44[1,a]
Gymnoscopelus opisthopterus	5	0.16 ± 0.06[2,3]	10.81 ± 0.15[1]	-23.12 ± 0.96[1]
Electrona antarctica	5	0.18 ± 0.09[2,3,b]	8.64 ± 0.96[1,a]	-24.91 ± 0.66[1,a]
Gymnoscopelus nicholsi	5	0.29 ± 0.12[3,a]	8.70 ± 0.94[1,a]	-22.00 ± 1.14[1,a]
Seabirds				
Macronectes halli	5	0.47 ± 0.33[4,b]	13.07 ± 0.56[3,4,b]	-18.68 ± 0.87[2,a]
Macronectes giganteus	5	0.85 ± 0.12[5,b]	11.80 ± 0.33[3,a]	-18.23 ± 0.66[2,3,b]
Thalassarche melanophris	5	1.18 ± 0.46[5,a]	11.83 ± 0.19[3,a]	-17.30 ± 0.93[2,3,b]
Thalassarche chrysostoma	5	1.25 ± 0.45[5,a]	12.06 ± 0.56[3,a]	-18.22 ± 1.47[2,3,b]
Stercorarius antarcticus	5	2.44 ± 1.08[6,a]	12.63 ± 0.54[3,4,a]	-15.59 ± 0.32[3,b]
Diomedea exulans	5	3.28 ± 0.63[6,a]	14.83 ± 0.61[4,a]	-18.29 ± 0.42[2,3,b]

Table 2- Mercury (Hg) concentration, δ^{13}C and δ^{15}N values of different species sampled on land or caught in the waters around South Georgia in the 2016/17 austral summer (mean ± standard deviation) The superscript numbers ([1]) and letters ([a]) denotes statistical significance from within the sampling year and comparison between years, respectively.

Species	n	Hg ($\mu g\ g^{-1}$)	$\delta^{15}N$ (‰)	$\delta^{13}C$ (‰)
POM	12	0.0005 ± 0.0002	n.a.	n.a.
Zooplankton				
Euphausia superba	30	0.01 ± 0.003[1,a]	3.38 ± 0.38[1,a]	-24.98 ± 0.66[1,a]
Euphausia vallentini	30	0.01 ± 0.03[2]	3.00 ± 0.78[1]	-21.60 ± 0.18[1]
Thysanoessa sp.	40	0.02 ± 0.01[2,a]	5.62 ± 0.29[2,a]	-23.99 ± 0.64[1,a]
Euphausia triacantha	30	0.02 ± 0.03[2,a]	7.08 ± 0.62[2,a]	-22.72 ± 0.17[1,a]
Euphausia frigida	35	0.05 ± 0.01[2]	5.94 ± 0.04[2]	-21.16 ± 0.48[1]
Themisto gaudichaudii	40	0.06 ± 0.02[2,a]	5.61 ± 0.79[2,a]	-22.46 ± 0.52[1,a]
Gigantocypris sp.	20	0.07 ± 0.01[2,a]	7.07 ± 1.09[2,a]	-24.71 ± 0.35[1,a]
Euphausia spinifera	20	0.07 ± 0.02[2]	5.05 ± 0.66[2]	-22.79 ± 0.09[1]
Parandania boecki	30	0.12 ± 0.03[2,a]	7.55 ± 0.28[2,a]	-21.82 ± 1.90[1,a]
Squid				
Slosarczykovia circumantarctica	5	0.01 ± 0.01[2,a]	6.23 ± 0.78[2,a]	-23.13 ± 0.43[1,b]
Galiteuthis glacialis	5	0.02 ± 0.01[2,a]	7.40 ± 0.09[2,a]	-24.68 ± 0.49[1,a]
Myctophid fish				
Krefftichthys anderssoni	5	0.05 ± 0.01[2,a]	8.48 ± 0.50[2,a]	-23.02 ± 0.10[1,a]
Protomyctophum bolini	5	0.10 ± 0.03[2,a]	7.98 ± 0.75[2,a]	-23.43 ± 0.70[1,a]
Gymnoscopelus braueri	5	0.12 ± 0.06[2,3,a]	9.58 ± 0.97[2,a]	-24.37 ± 0.61[1,a]
Electrona antarctica	5	0.12 ± 0.07[2,3,b]	7.41 ± 0.72[2,a]	-24.54 ± 0.34[1,a]
Gymnoscopelus nicholsi	5	0.30 ± 0.17[3,a]	9.73 ± 0.03[2,a]	-20.74 ± 0.34[1,a]
Nototheniod fish				
Champsocephalus gunnari	11	0.02 ± 0.01[2]	9.18 ± 0.56[2]	-22.95 ± 0.44[1]
Patagonotothen guntheri	5	0.10 ± 0.03[2,3]	8.34 ± 0.56[2]	-23.43 ± 0.32[1,2]
Chaenocephalus aceratus	5	0.11 ± 0.02[2,3]	10.79 ± 1.12[2,3]	-21.66 ± 0.50[1,2]
Notothenia gibberifrons	8	0.18 ± 0.07[3]	11.30 ± 0.54[3]	-19.93 ± 0.78[2]
Notothenia rossii	8	0.18 ± 0.08[3]	10.78 ± 1.67[2,3]	-21.39 ± 0.58[1,2]
Dissostichus eleginoides	5	0.20 ± 0.06[3]	11.86 ± 0.75[3]	-21.76 ± 0.77[1]
Seabirds				
Pachyptila desolata	5	0.22 ± 0.14[3]	8.59 ± 0.78[2]	-21.58 ± 0.73[3]
Halobaena caerulea	5	0.62 ± 0.23[3]	9.25 ± 0.50[2,3]	-21.14 ± 0.67[3]
Thalassarche chrysostoma	5	1.43 ± 0.50[4,a]	11.78 ± 0.67[3,a]	-19.91 ± 0.40[3,4,b]
Thalassarche melanophris	5	1.51 ± 0.46[4,a]	11.52 ± 0.35[2,3,a]	-20.32 ± 0.63[3,4,b]
Macronectes giganteus	5	1.68 ± 0.27[4,b]	11.52 ± 0.41[2,3,a]	-20.46 ± 0.38[3,4,b]
Macronectes halli	5	2.05 ± 0.80[4,b]	11.58 ± 0.65[2,3,b]	-19.69 ± 1.12[3,4,a]
Stercorarius antarcticus	5	3.88 ± 2.41[5,a]	11.88 ± 1.13[3,a]	-18.79 ± 0.90[4,b]

Figure Captions

Figure 1 – Mercury concentration on a log 10 scale versus δ^{15}N for zooplankton (whole individuals), myctophid fish (muscle) and seabirds (feathers) that were caught in 2007/08 (Y = 0.2028*X - 2.6008) and in 2016/17 (Y = 0.2782*X - 3.0960)

Figure 2 – Illustrations of the South Georgia food web in years with high Antarctic krill abundance (left – A) and in years with low Antarctic krill abundance (right – B). Shaded/clear areas represent different trophic levels, boxes represent mercury concentrations (more Hg – higher concentration).
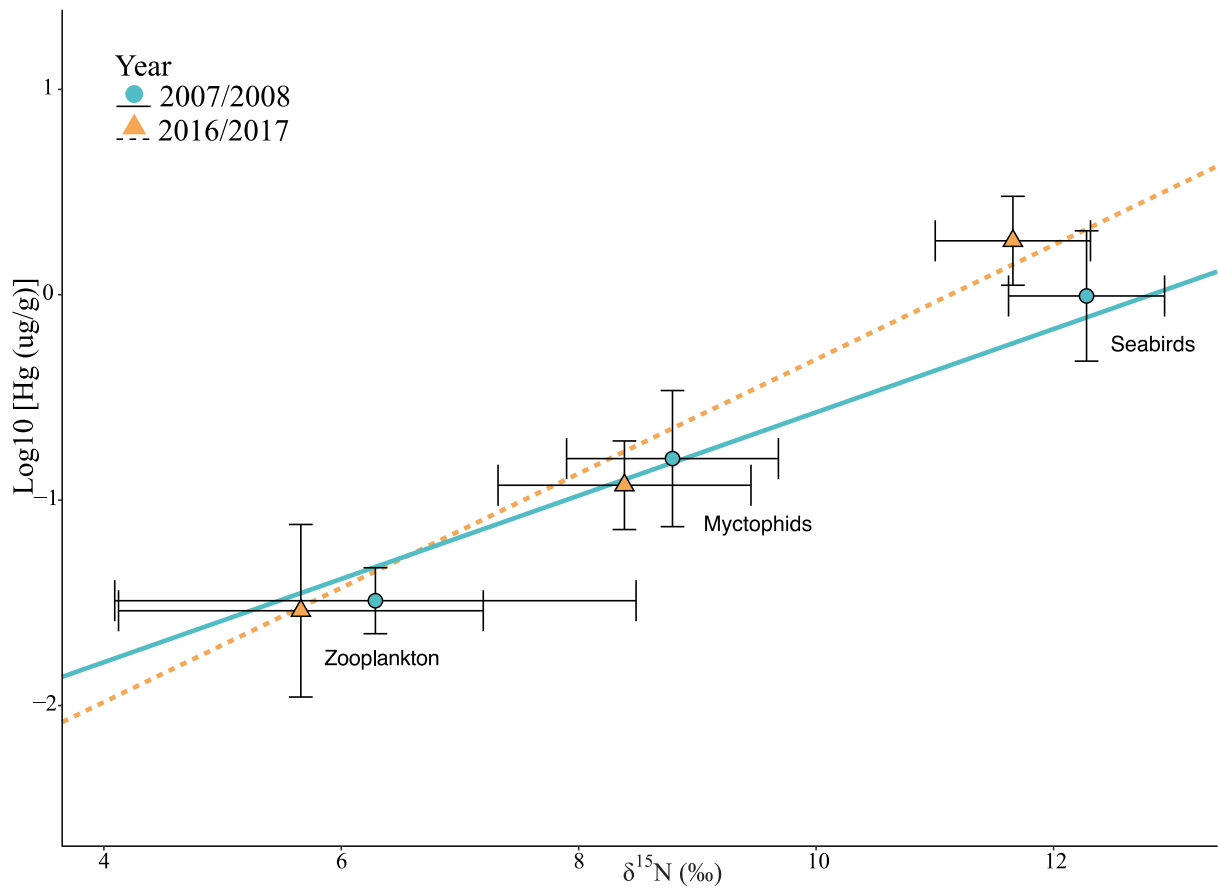
Figure 1

Figure 2