Title:

Using epibenthic fauna as biomonitors of local marine contamination adjacent to McMurdo Station, Antarctica

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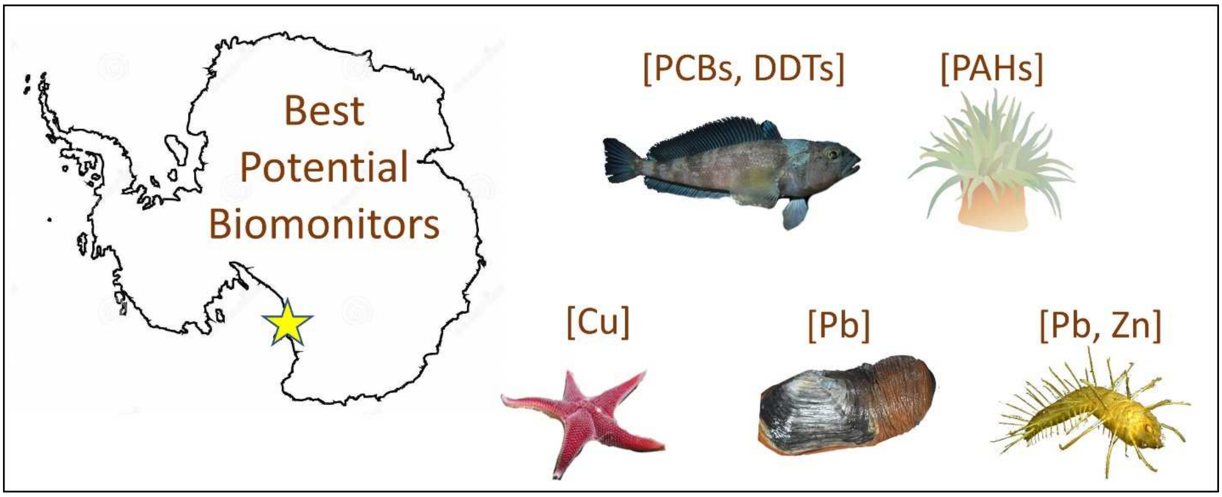
Highlights:

- The sea floor adjacent to McMurdo Station, Antarctica is contaminated.
- Six contaminants were all found in higher concentrations in benthic fauna.
- Bioaccumulation is widespread among benthic taxa (10 of 12 taxa sampled).
- Most whole-body concentrations are below human consumption standards.

Abstract

Ten benthic fauna taxa in a polluted marine area adjacent to McMurdo Station, Antarctica were deemed to be potential biomonitors because PCBs, DDTs, PAHs, copper, lead and/or zinc in their tissues were significantly higher than in tissues of taxa living in reference areas ($p < 0.05$). Concentrations of PCBs and DDT were highest in *Trematomus* (fish). Total PAH concentrations were highest in *Alcyonium antarcticum* (soft coral), *Isotealia antarctica* (anemone) and *L. elliptica*. Copper and lead concentrations were highest in *Laternula elliptica* (bivalve) and *Flabegraviera mundata* (polychaete), and lowest in *Trematomus* and *Parbolasia corrugatus* (nemertean). However, copper concentrations were even higher in the asteroids *Perknaster fuscus antarcticus, Odontaster validus* and *Psilaster charcoti*. Bioaccumulation factors for different species were highest for PCBs and DDT, and lowest for lead. Bioaccumulation of some contaminants are likely prevalent in benthic taxa at McMurdo Station, but concentrations are usually low relative to human consumption standards.

Graphical Abstract



1. Introduction

Antarctica's coastal marine environment is among the least anthropogenically disturbed in the world because access to the continent by humans is restricted by sea ice (Halpern et al., 2008) and its distance from populated land masses. Despite this isolation, Antarctica is still impacted by humans on both global and local scales. Causes of global effects include changes such as warming temperatures (Turner et al., 2014), ocean acidification (Orr et al., 2005), and the deposition of long-range atmospheric persistent organic pollutants (POPs; Risebrough et al., 1976). Some obvious local effects are caused by legacy chemical contamination around some past and present Antarctic research stations (see Tin et al., 2009; Aronson et al., 2011; Palmer et al., 2020). Chemical pollutants including hydrocarbons, metals, organic matter, polychlorinated biphenyls (PCBs) and dichlorodiphenyltrichloroethane (DDT) have accumulated in marine sediments and have caused changes in benthic community composition adjacent to some of these research stations (e.g., Dayton and Robilliard, 1971; Thompson et al., 2003; Conlan et al., 2004; Kennicutt et al., 2010). In addition to community composition effects, both locally and globally derived contaminants have been detected in tissues of many marine fauna species (see Bargagli, 2005).

The bioaccumulation of contaminants into faunal tissues indicates both the presence and bioavailability of contaminants (Kennicutt et al., 1995). Contaminants in the marine environment are often adsorbed to sediments, which then accumulate on the sea floor. Contaminants often accumulate in faunal tissues from direct contact with these benthic sediments and/ or through the ingestion of contaminated food and suspended particulate material (Kennish, 1997). Benthic invertebrate and demersal fish species are therefore more likely than pelagic species to be exposed to contamination in the marine environment. Several benthic species, especially molluscs (e.g., oysters and mussels), have been successfully used as biomonitors, species that accumulate, and can be analyzed to monitor the bioavailability of, contaminants in their tissues (Rainbow and Phillips, 1993; Sericano et al., 1995; Webb et al. 2020). Although extensive bioaccumulation studies have been undertaken, and cosmopolitan biomonitors identified in populated parts of the world, studies occurring in remote parts of the world such as Antarctica have been limited in number of species and/or number of contaminants (e.g., Negri et al., 2006; Trevizani et al., 2016; Webb et al., 2020).

The Antarctic benthic faunal community comprises a diverse group of long-lived (decades), slow growing invertebrates that have adapted to the consistently cold sea temperatures and seasonally limited food supply (Arnaud, 1977; Peck et al., 2006). The most species rich Antarctic benthic taxonomic groups are polychaetes, gastropods, amphipods, bryozoans, isopods and true sponges, although several groups are underrepresented (e.g., gastropods, isopods, reptant decapods) and some overrepresented (e.g., pycnogonids) relative to the global average (Clarke et al., 2004). Decreases in detoxification mechanisms attributed to low temperatures, peculiar feeding strategies (e.g., many filter and opportunistic feeders; Arnaud 1977), and their aforementioned longevity is suggested to cause an increase in bioaccumulation of contaminants in Antarctic relative to other parts of the world (Grotti et al., 2008). Elsewhere in the world (including the Arctic), durophagous (exoskeleton-breaking) predators of benthic invertebrates are abundant and dominated by bony fishes, decapods, and elasmobranchs (Aronson et al., 2011). However, a lack of durophagous predators in the Antarctic has resulted in slow-moving invertebrates, such as asteroids, nemerteans and pycnogonids being most of the top predators of Antarctic shallow benthic fauna (Aronson and Blake, 2001). This means that a diverse range of long-lived Antarctic marine fauna from different trophic levels and taxa groups live on the sea floor, in a place that is most susceptible to contaminant accumulation.

This study aims to quantify the bioaccumulation of contaminants into several different fauna species. Results of this study have implications for understanding the effects of contamination on the Antarctic food web, and which species can potentially be used as Antarctic biomonitors for contamination. It is important to be able to use a suite of biomonitors for contamination monitoring because different taxa have different contaminant-accumulation mechanisms (e.g., feeding mode, assimilation efficiency) and can bioaccumulate contaminants in different ways. This study is located adjacent to McMurdo Station, Ross Island, a location with a well-documented history of marine sediment contamination from locally-sourced waste dumping and sewage discharge (e.g., Dayton and Robilliard, 1971, Lenihan, 1992; Kennicutt et al. 1995; Kennicutt et al., 2010; Palmer et al., 2020). We hypothesize that contaminants have bioaccumulated into epibenthic fauna and that the extent of bioaccumulation varies among species.

2. Methods

2.1 Study Design

Sampling occurred from 2000 to 2015 at two anthropogenically disturbed (hereinafter "disturbed") and two reference transects adjacent to McMurdo Station, and at Turtle Rock (TR), a reference area approximately 12 km northeast of McMurdo Station (Figure 1). One disturbance transect is in Winter Quarters Bay (WQB), a historic dumping area (and adjacent to the primary Station dump until the 1980s; Lenihan et al., 1990) and the location of the floating ice pier that is used to transfer materials on and off McMurdo Station during its annual resupply in January/February. The second disturbance transect is adjacent to a sewage outfall (Outfall or O), which discharged raw sewage into McMurdo Sound until 2003 (Egger, 2003) and is in the vicinity of another historic dumping area. Although the exact historic quantities of sewage are unknown, volumes prior to sewage treatment are expected to be on the same order of magnitude as seasonal volumes occurring in 1997-1998: 53,000 L d^{-1} to 272,000 L d^{-1} during the winter and summer respectively (Conlan at al., 2006). WQB and Outfall bottom sediments are known to have high concentrations of polycyclic aromatic hydrocarbons (PAHs), total petroleum hydrocarbons (TPHs), polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DDT), and several metals, among other contaminants (Palmer et al., 2020). The two local reference transects were at Cape Armitage (CA-E and CA-F), while a distant reference transect was sampled at TR (Transect T). Each transect comprised three sampling stations at water depths of 12, 24 and 36 m. Tissue samples from each transect were pooled because there were not enough replicates (individuals of each taxon) to compare concentrations among depths and transects.

Systematic sampling of sediment contamination occurred in November/December at the two disturbed transects adjacent to McMurdo Station (WQB, O) in 2000 and from 2003 to 2013, at CA-E in 2000 and 2008 to 2013, CA-F in 2000 and 2003 to 2009 (as A, D, E, F in Palmer et al., 2020), and at TR in 2008 and 2011 to 2013. Opportunistic sampling of twelve epibenthic and benthic fauna taxa occurred simultaneously with sediment sampling from 2000 to 2013 and in 2015 (Table 1, Table S1). Therefore, the location and years that each species was sampled varied. Although some decreases in sediment contamination at the disturbed transects has been documented over the study period, contamination was always distinctly higher than background concentrations during the study period (PAHs, TPHs, DDTs, PCBs, Pb, Cu, Ba, Cr, Hg, Cd; Palmer et al. 2020) and species sampled were long-lived (decades; Peck et al., 2006). Therefore, any confounding effects of the timing of sampling on fauna contamination was thought to be minor.

All taxa were identified to, and investigated at, the species level except for the Nototheniidae family of fish *Trematomus*, which were grouped together to increase the sample size. *Trematomus* consisted of 30 *T. bernacchii*, 5 *T. pennelli*, 3 *T. hansoni*, and 6 unidentified *Trematomus* sp. The taxa collected were taxonomically and trophically diverse, and included two filter feeders (bivalve *Laternula elliptica* and soft coral *Alcyonium antarcticum*), a deposit feeder (polychaete *Flabegraviera mundata*), an herbivore (echinoid, *Sterechinus neumayeri*), and eight scavenger/predators (Table 1).

2.2 Sampling and Laboratory Analysis

Marine sediments at each sampling location were sampled by SCUBA divers using hand-held 6.3-cm diameter cores (35.3 cm^2) to a sediment depth of 10 cm. Triplicate sediment samples were taken for chemistry each year at each sampling station. Fauna were collected opportunistically if time permitted at the end of each sediment-sampling dive directly by hand or with a hand-held net. After immediate clean processing, all samples were stored in pre-cleaned glass jars.

Fauna tissue and sediment chemistry samples were frozen at -20 °C and shipped to the Geochemical Environmental Research Group (GERG) at Texas A&M University to be analyzed for hydrocarbons, organochlorines, and trace metals using methods from the National Oceanic and Atmospheric Administration 'Status and Trends Program' (NOAA, 1993) and the United States Environmental Protection Agency (Telliard, 1989; Creek et al., 1994; see Morehead et al., 2008; Kennicutt et al., 2010; Klein et al., 2012). Polycyclic Aromatic Hydrocarbon (PAH), PCB and pesticide concentrations were determined using gas chromatography/mass spectrometry (GC-MS), metal concentrations (except for mercury) were determined using inductively coupled plasma mass spectrometry (ICP-MS), and mercury concentrations were determined by cold vapor atomic absorption spectroscopy (CVAAS). Tissue analysis was of the whole organism, which is 1) more useful for integrating concentrations into bioaccumulation studies involving higher trophic levels (because consumers eat an entire organism and not just a certain tissue), and

2) easier to process, which is more useful should a species be used in future biomonitoring studies. Fish processed in this study (*Trematomus* spp.) were all ~< 20 cm long. All sediment and tissue concentrations are reported as a dry weight basis.

2.3 Statistical Analyses

A one-way general linear model (GLM) analysis of variance (ANOVA) test was used to test for differences in each species' individual contaminant concentrations among sampling locations: Winter Quarters Bay (WQB), the outfall transect (Outfall), reference transects E and F adjacent to Cape Armitage (CA), and the reference transect at Turtle Rock (TR). The CA reference transects were combined because they were not always sampled in the same years and their sediment chemistry was similar. Normality of the residuals of the ANOVA models were assessed graphically and using Shapiro-Wilk tests. Differences among locations were highlighted using Tukey's Studentized Range (HSD) Test ($p < F$ of 0.05). Chemicals were considered to be local sediment contaminant indicators if they were significantly enriched in disturbed (WQB and Outfall transects) relative to reference areas (CA transects E and F, TR transect). Similarly, species were deemed to be potential biomonitors for a contaminant if tissue concentrations of a sediment contaminant indicator were significantly higher in individuals located in disturbed than reference locations. Differences in faunal contaminant composition among species were compared using Principal Components Analysis (PCA) using only contaminants that were significantly higher in disturbed than reference locations for both sediments and fauna. Data were $\log_e$-transformed prior to ANOVA and PCA analysis. Bioaccumulation factors (BFs, = $\text{Concentration}_{\text{organism}}$ / $\text{Concentration}_{\text{sediment}}$) were calculated for fauna in disturbed locations to approximate bioaccumulation from the sediment to each organism (Negri et al., 2006; Trevizani et al., 2016). A BF > 1 indicates that bioaccumulation in an organism is greater than the concentration of the sediment. Size, age, sex, and condition index data were unavailable for each individual sampled so were not incorporated into statistical analyses. All data management and statistics were conducted using SAS 9.4 software (SAS Institute Inc., 2017).

<u>3. Results</u>

A total of 315 fauna samples were analysed for tissue contaminants from 2000 to 2015. Most of these samples (292, 93%) were taken in the eight-year period from 2004 to 2012. Five samples (four *F. mundata*, one *Tritoniella belli* sampled in 2010) were analyzed for metals only.

Chemicals were considered to be local sediment contaminant indicators if they were significantly enriched in disturbed relative to background concentrations in reference areas ($p < 0.05$). These indicators include total DDTs (disturbed was 19x greater than CA, 803x greater than TR), PCBs (43x > CA, 41x > TR), PAHs (26x > CA, 164x > TR), cadmium (2.3x > CA, 3.4x > TR), copper (3.5x > CA, 1.8x > TR), lead (6.4x > CA, 40x > TR), mercury(4.5x > CA, 18x > TR), and zinc (1.4x > CA, 1.4x > TR) ($p < 0.001$ to $p \leq 0.01$, F values from 3.9 to 217.3, n= 117 to 129; Table 3, Table S2, Table S3). All these contaminants had higher or similar mean concentrations at the Outfall compared to WQB (Figure S1 to Figure S13). Arsenic concentrations in sediments were similar in the disturbed area to the CA reference transects but were still higher than at the TR reference transect (2.4x > TR). In contrast to the other metals, sediment barium, cobalt, iron and nickel concentrations were highest at the TR reference transect (Mean Ba: 174 $\mu g\ g^{-1}$, Co: 18 $\mu g\ g^{-1}$, Fe: 55,669 $\mu g\ g^{-1}$, Ni: 74 $\mu g\ g^{-1}$) and lowest at the CA reference transects (Ba: 41 $\mu g\ g^{-1}$, Co: 8 $\mu g\ g^{-1}$, Fe: 17,505 $\mu g\ g^{-1}$, Ni: 37 $\mu g\ g^{-1}$) ($p < 0.0001$).

Total PCB concentrations were higher at the disturbed than reference locations in tissues of all species except the soft coral, *A. antarcticum* ($p \leq 0.11$), the asteroid, *Diplasterias brucei* ($p \leq 0.14$), and the nudibranch, *T. belli* ($p \leq 0.32$). Total PCB concentrations in tissues from the disturbed WQB and Outfall transects were similar to each other, and greater than tissues from CA and TR reference transects in the polychaete *F. mundata* (23x > CA, undetected at TR), the bivalve *L. elliptica* (8.3x > CA, 162x > TR), the echinoid *S. neumayeri* (18x > CA, undetected at TR), the actinarian *I. antarctica* (39x > CA, not sampled at TR), the teleost *Trematomus* (25x > CA, 350x > TR), the nemertean *Parbolasia. corrugatus* (32x > CA, 796x > TR), and the asteroid *O. validus* (8.8x > CA, 139x > TR), *Perknaster fuscus antarcticus* (40x > CA, 586x > TR) and *Psilaster charcoti* (12x > CA, not sampled at TR). Tissue PCB concentrations in the disturbed WQB and Outfall transects were greatest in *Trematomus* (WQB: mean = 11,048 $ng\ g^{-1}$, Outfall: 7081 $ng\ g^{-1}$), followed by *I. antarctica* (WQB: 5191 $ng\ g^{-1}$, Outfall: 4463 $ng\ g^{-1}$), *S. neumayeri* (WQB: 3271 $ng\ g^{-1}$, Outfall: 1917 $ng\ g^{-1}$) and *Parbolasia corrugatus* (WQB: 3513 $ng\ g^{-1}$, Outfall: 1614 $ng\ g^{-1}$).

Although maximum total DDT concentrations for most species were higher at the disturbed than the reference locations, large intraspecies variability meant that there were only differences in means among locations for five species. Total DDT concentrations in *F. mundata* were higher in the disturbed WQB (17.9 ng g^{-1}) transect than all other locations (Outfall: 3.7 ng g^{-1}, CA and TR: 0 ng g^{-1}, $p \leq 0.03$). Total DDT was higher in the anemone *I. antarctica* at the disturbed Outfall (11.1 ng g^{-1}) transect than the disturbed WQB (2.6 ng g^{-1}) transect and the reference CA (1.3 ng g^{-1}, $p \leq 0.0002$) transect. *Trematomus* DDT concentrations were higher at the disturbed WQB (21.6) transect than at the disturbed Outfall (16.4 ng g^{-1}) transect, which in turn was higher than the reference locations (CA: 4.0 ng g^{-1}, TR: 1.2 ng g^{-1}, $p \leq 0.001$). *Parbolasia corrugatus* DDT concentrations were highest in the disturbed WQB (3.1 ng g^{-1}) transect and lowest in the reference TR (0.2 ng g^{-1}) transect, and similar between the disturbed Outfall (1.5 ng g^{-1}) and reference CA (1.0 ng g^{-1}, $p \leq 0.001$) transects. Total DDT concentrations in *T. belli* differed from the other species in that concentrations were higher at the reference CA (14.6 ng g^{-1}), than the disturbed Outfall (1.3 ng g^{-1}, $p \leq 0.03$).

Total PAHs in *L. elliptica* were greater in the disturbed WQB (538 ng g^{-1}) than the other three locations (30 to 260 ng g^{-1}, $p \leq 0.003$). Total PAHs were higher at both disturbed locations, WQB and Outfall, as well as the reference CA, than the reference TR for both *Trematomus* (CA: 9.3 ng g^{-1}, else: 69 to 89 ng g^{-1}, $p \leq 0.01$) and *Parbolasia corrugatus* (CA: 20.6 ng g^{-1}, else: 115 to 247 ng g^{-1}, $p \leq 0.001$). Total PAHs were higher in *I. antarctica* at the disturbed Outfall (486 ng g^{-1}) than the disturbed WQB (383 ng g^{-1}) and reference CA (93 ng g^{-1}, $p \leq 0.02$). Total PAHs in *S. neumayeri* were higher at the reference TR (382 ng g^{-1}) than all three other locations (52 to 237 ng g^{-1}, $p \leq 0.002$)

Pb concentrations were higher at the disturbed Outfall than all other locations sampled for *S. neumayeri* (Outfall: 7.1 µg g^{-1}, [TR, CA, WQB]: 0.3 to 2.3 µg g^{-1}, $p \leq 0.001$), *I. antarctica* (Outfall: 7.9 µg g^{-1}, [CA, WQB]: 0.9 to 1.2 µg g^{-1}, $p < 0.0001$), *O. validus* (Outfall: 3.6 µg g^{-1}, [TR, CA, WQB]: 0.9 to 2.4 µg g^{-1}, $p \leq 0.06$), and *A. antarcticum* (Outfall: 3.4 µg g^{-1}, CA: 0.8 µg g^{-1}, $p \leq 0.02$). There were no differences in Pb concentrations among locations for all other species.

Hg concentrations were also highest at the disturbed Outfall in *F. mundata* (Outfall: 0.08 µg g^{-1}, [TR, CA, WQB]: 0.03 to 0.07 µg g^{-1}, $p \leq 0.01$) and *S. neumayeri* (Outfall: 0.16 µg g^{-1}, [TR, WQB, CA]: 0.08 to 0.12 µg g^{-1}, $p \leq 0.03$). In contrast, Hg concentrations were highest at reference CA in *T. belli* (CA: 0.22 µg g^{-1}, Outfall: 0.08 µg g^{-1}, $p \leq 0.04$), *O. validus* (CA: 0.26 µg g^{-1}, [WQB, TR, Outfall]: 0.15 to 0.22 µg g^{-1}, $p \leq 0.007$), and *Perknaster fuscus antarcticus* (CA: 0.42 µg g^{-1}, [WQB, TR]: 0.14 to 0.28 µg g^{-1}, $p \leq 0.005$).

Conversely, Cd concentrations in fauna were highest at reference TR in *Parbolasia corrugatus* (TR: 39 µg g^{-1}, [Outfall, CA]: 21 to 22 µg g^{-1}, WQB: 9 ± 6 µg g^{-1}, $p < 0.0001$) and *S. neumayeri* (TR: 4.7 µg g^{-1}, [WQB, Outfall, CA]: 1.6 to 2.2 µg g^{-1}, $p \leq 0.009$). There were no spatial differences in Cd concentrations in any other species.

Cu concentrations were highest at the disturbed Outfall in *A. antarcticum* (Outfall: 10 µg g^{-1}, CA: 3 µg g^{-1}, $p \leq 0.04$), *S. neumayeri* (Outfall: 12 µg g^{-1}, [CA, WQB, TR]: 2 to 4 µg g^{-1}, $p \leq 0.003$), *I. antarctica* (Outfall: 18 µg g^{-1}, [CA, WQB]: 7 to 10 µg g^{-1}, $p \leq 0.008$), and *Trematomus* (Outfall: 7 µg g^{-1}, WQB: 6 µg g^{-1}, [TR, CA]: 1 to 7 µg g^{-1}, $p \leq 0.05$), and highest at disturbed WQB in *L. elliptica* (WQB: 25 µg g^{-1}, [CA, TR, WQB]: 13 to 23 µg g^{-1}, $p \leq 0.01$) and *Perknaster fuscus antarcticus* (WQB: 77 µg g^{-1}, [TR, CA]: 9 to 11 µg g^{-1}, $p \leq 0.0006$).

There were no differences in Zn concentrations among locations for any species ($p \geq 0.14$) except for in *L. elliptica* and *Trematomus.* Zn concentrations in *Trematomus* were higher at the transects closer to McMurdo Station ([disturbed WQB and Outfall, reference CA]: 67 to 88 µg g^{-1}) than at reference TR (41 µg g^{-1}, $p \leq 0.004$). Zn concentrations in *L. elliptica* were lower at disturbed WQB (57 µg g^{-1}) than the other three transects ([TR, Outfall, WQB]: 145 to 169 µg g^{-1}, $p \leq 0.004$).

Potential biomonitor species were identified as those that had concentrations of contaminants that were significantly greater in disturbed than reference areas as determined using Tukey's HSD tests ($p < 0.05$). These included all species sampled except for the asteroid *D. brucei* and the nudibranch *Tritoniella belli*. These biomonitor species were incorporated into a PCA using contaminants that were commonly higher in fauna at disturbed than reference locations (PCB, PAH, DDT, Pb, Cu, Zn; Figure 2). Fauna at only the disturbed locations (WQB and Outfall) were included in the PCA to determine how bioaccumulation of contaminants varied among bioindicator species rather than how species compared between disturbed and reference locations. The first two principal components (PC1 and PC2) accounted for 29.8 and 22.0% of the variability within the tissue contamination data. Fauna at the disturbed sites that had high total PCB concentrations (low PC1 scores) generally had high DDT concentrations. Fauna at the disturbed sites that had high Pb concentrations (high PC1 scores) generally also had high Cu concentrations. *Trematomus* had higher total PCBs and DDT concentrations (mean concentration in WQB and Outfall: 9063 and 19 ng g^{-1}) than all other fauna (402 to 4827, and 1 to 11 ng g^{-1}, Figure 3) at the disturbed sites. Total PAH concentrations were highest in *A. antarcticum* (1355 ng g^{-1}), *I. antarctica* (434 ng g^{-1}) and *L. elliptica* (398 ng g^{-1}). Cu and Pb concentrations were both high in *L. elliptica* (24 and 10.6 µg g^{-1}) and *F. mundata* (24 and 14.1 µg g^{-1}), and lowest in *Trematomus* (6.3 and 0.7 µg g^{-1}) and *Parbolasia corrugatus* (5.8 and 0.8 µg g^{-1}). However, Cu concentrations were highest in the asteroids *Perknaster fuscus antarcticus* (77 µg g^{-1})*, O. validus* (50 µg g^{-1}) and *Psilaster charcoti* (35 µg g^{-1}; Figure 4).

Bioaccumulation factors (BFs) of contaminants ([fauna]/[sediment]) were inconsistent among species (Table 2). Total PCB BFs ranged from 0.5 in *D. brucei* and 0.6 in *A. antarcticum* to 6.8 in *I. antarctica* and 12.7 in *Trematomus* Total DDT BFs ranged from < 1 in *T. belli*, *D. brucei* and *Perknaster fuscus antarcticus* to 3.4 in *F. mundata* and 5.9 in *Trematomus.* Zn BFs were highest in *F. mundata* (4.3), *S. neumayeri* (3.9) and *I. antarctica* (3.6). Cu BFs were all < 1.0 except in *Perknaster fuscus antarcticus* (1.8) and *O. validus* (1.2). Pb BFs were < 0.2 in all fauna except for *F. mundata* (0.3) and *L. elliptica* (0.2). Total PAH BFs were ≤ 0.5 in all fauna except for *A. antarcticum* and *T. belli* (both 1.4).

4. Discussion

Sediments

Determining the type and intensity of local contamination in Antarctica is dependent on adequate characterization of background concentrations of trace metals and organic compounds because it is uncommon for historic waste-generating activities to be documented and dumping to be quantified. It is difficult to use a continental baseline for trace metals because metal concentrations are influenced by local geological sources, e.g., high Hg concentrations occur at Deception Island, South Shetland Islands (Bento et al., 2021). The presence of naturally occurring hydrocarbons also makes it difficult to determine low concentrations of anthropogenic hydrocarbons (Cripps, 1994). A further complication is that persistent organic compounds (POPs), which have no natural origins, are transported to the Antarctic via long-range atmospheric transport (Risebrough et al., 1976). These problems are avoided by determining potential contaminant concentrations in at least one local reference location, as occurred in this study.

Total DDTs, total PCBs, total PAHs, Cd, Cu, Pb, Hg, and Zn were all identified as sediment contamination indicators for this study because the bottom sediments of the disturbed area adjacent to McMurdo Station (WQB and the Outfall transects) had greater concentrations than both reference sites, suggesting local, anthropogenic input to the disturbed area (Figure S1 to Figure S13). Ba, Co, Fe and Ni all likely had anthropogenic sources because their concentrations were greater in the disturbed area than the reference CA. However, the high concentrations of these metals at the reference TR indicates that natural geological sources can overshadow any anthropogenic inputs further away from McMurdo Station. Therefore, these four metals could only be used as indicators of anthropogenic contamination within limited spatial scales in this study (< 10 km from McMurdo Station) and were excluded from further statistical analyses. These differences in natural enrichment of metals underlines the importance of using local rather than continental reference values for assessments of anthropogenic contamination.

The largest enrichment in sediment contamination between the disturbed and reference locations occurred in DDTs (mean was 19 to 803x > means of CA and TR), total PCBs (41 to 43x >TR and CA), and PAHs (26 to 164 greater than CA and TR). Mean Cd, Cu, Pb, Hg, and Zn concentrations at disturbed locations were usually less than ten times the concentration at reference locations CA and TR than WQB and the Outfall.

Mean sediment PAH concentrations at disturbed locations WQB (1095 ng g^{-1}) and the Outfall (779 ng g^{-1}) were similar to the higher concentrations occurring at Carlini Station, King George Island (KGI; 228 to 1908 ng g^{-1}; Curtosi et al., 2007) but higher than those occurring adjacent to stations in Admiralty Bay, KGI (9-271 ng g^{-1}; Martins et al., 2004). Sediment PAH concentrations at reference locations CA (36 ng g^{-1}) and TR (6 ng g^{-1}) were similar to those from reference locations at Palmer Station (6 – 30 ng g^{-1}, Palmer et al., in submission) and those 500 m from Signy Station, South Orkney Islands (14 ng g^{-1}; Cripps, 1992).

Mean total DDT concentrations at disturbed locations WQB (2.8 ng g^{-1}) and the Outfall (3.6 ng g^{-1}) are lower or similar to two sediment samples analyzed at Palmer Station, Anvers Island (3.2 and 25.3 ng g^{-1}, Palmer et al. *in submission*) but higher than all other Palmer Station sediment concentrations (none detected). Disturbed McMurdo Station sediments were higher than the North American Effects Range Low (ERL, 10th percentile of effects, 1.58 ng g^{-1}) but much lower than the Effects Range Median (ERM, 50th percentile of effects, 46.1 ng g^{-1}, Long et al., 1995), which indicates that there is a low possibility of biological or ecological effects from total DDT contamination in the sediments (ignoring compounding effects of multiple contaminants). The origin of DDT in marine sediments adjacent to McMurdo Station is suspected to be from US Naval vessels, which used DDT as an insecticide and rodenticide until 1972 (USEPA 1975, Palmer et al., 2021). The US military began permanent occupation of the area that is now occupied by McMurdo Station in 1955 and has maintained a presence at McMurdo Station until today, although their role in operations is now limited to logistical support (Klein et al., 2008).

Historic dumping in the early days of McMurdo Station occupation is also the likely cause of elevated PCB concentrations in sediments at McMurdo Station. The PCB mixture in Murdo Station resembles a mixture used in fluid-filled semiconductors (Aroclor 1260) until 1971 (USEPA 1976, Kennicutt et al., 2010), a mixture that has also been found in former military bases in the Arctic (Poland et al., 2011, Kuzyk et al., 2005). Mean total PCB concentrations in disturbed McMurdo Station locations (WQB: 440 n g^{-1} and the Outfall: 985 ng g^{-1}) are higher than those occurring at reference locations in this study (17 ng g^{-1}), at Palmer Station (0 to 353 ng g^{-1}; Palmer et al., 2021), in Admiralty Bay, KGI (2 to 6 ng g^{-1}; Montone et al., 2001), Zucchelli Station, Terra Nova Bay (0.05 to 0.36 ng g^{-1}; Fuoco et al., 1994), the Effects Range Median (180 n g^{-1}; Long et al., 1995), and the CCME probable effect levels (PEL; 189 ng g^{-1}; CMME, 2009). The PCB concentrations in WQB and the Outfall are large relative to other documented locations in the Antarctic and are likely having probable biological and ecological effects on marine benthic fauna.

Biomonitors

As with sediments, the comparison of bioaccumulated trace metals in fauna among locations is difficult due to differing background metal concentrations. However, spatial comparisons of reported bioaccumulated contaminants is even more difficult because different studies report different measurement units (e.g., per wet or dry weight, per lipid content) and analyze different body parts (particular organs or tissues, whole body; Bargagli, 2005). Comparing different species is a further complication because different taxa vary in factors such as detoxification and excretion mechanisms, ages, growth rates, contaminant tolerance, feeding modes, and diet, which can depend on the time and location given that many Antarctic fauna are opportunistic feeders. Tissue contamination by PCBs, DDTs and some PAHs determined in this study can be compared with the few similar studies conducted in other Antarctic studies because these types of contaminants only have anthropogenic sources. However, this current study focuses on identifying and using potential biomonitors to assess the effects of localized contamination around McMurdo Station because natural background concentrations vary spatially, the ecophysiology of many species is not well known (or not well documented), and laboratory analyses are inconsistent among the few published studies.

All but two taxa sampled were deemed potential biomonitor species in this study because contamination-indicator organic compounds and metals were higher in tissues from contaminated versus reference locations (Figure 3 and Figure 4). The species with the most potential for biomonitoring for specific contaminants in this study area are those with the highest bioaccumulation factors and have higher tissue concentrations in disturbed than reference areas. These species include *Trematomus* for PCBs and DDT, *A. antarcticum* for PAHs, *O. validus* and *Perknaster fuscus antarcticus* for Cu, *L. elliptica* and *F. mundata* for Pb, and *F. mundata* for Zn. Contaminant concentrations in *Tritoniella belli* (nudibranch) were not different among locations except for total DDT and Hg, which were higher at the reference CA (DDT: 14.6 ± 11.1 ng g^{-1}, Hg: 0.22 ± 0.10 µg g^{-1}) than at the disturbed Outfall (DDT: 1.3 ± 1.6 ng g^{-1}, Hg: 0.08 ± 0.03 µg g^{-1}). However, there is some uncertainty in concentrations calculated for *T. belli* because the body is mostly made up of liquid (~93%) and any measurement errors (false positives or interference errors) based on the small amount of dry mass would have been magnified when converting to concentration to a dry weight basis. Contaminants in *D. brucei* (asteroid) tissues were not different for any contaminant indicator ($p \geq 0.13$) among locations. In contrast, all other taxa, including three other asteroids (*O. validus*, *Perknaster fuscus antarcticus* and *Psilaster charcoti*) are suitable biomonitors. It is probable that an increase in sample size of all species would allow us to determine the performance of each species as a biomonitor more accurately. We are not aware of any further studies at McMurdo Station that would allow the sample size for each species to be increased. The monitoring activity that we conducted to sample fauna and generate these tissue contaminant data was discontinued in 2015.

Although this research identified ten species that are potentially useful Antarctic biomonitors, verification of the utility of using a species as a biomonitor to determine temporal changes in environmental contamination requires knowing a relationship between contaminant concentrations in tissues and the ambient environment (Rainbow 1995). These relationships could be determined from further studies of dose-dependence over a spatial gradient of ambient concentrations (Ahn et al. 1996), or possibly from dosing experiments in the laboratory. These types of studies are rare or undocumented in the Antarctic. However, dose-dependence relationships derived from such studies will allow biomonitoring to become a useful environmental management tool in Antarctica. Comparisons of organism contaminant concentrations at McMurdo Station with those occurring at other Antarctic locations will help determine the spatial extent at which a species may be used as a biomonitor.

Asteroids

As with other predator/scavengers, asteroids consume a range of organisms within the food web and are useful as biomonitors because they integrate a wide range of biomagnification pathways (Webb et al. 2020). Of the four asteroids sampled, *O. validus*, is possibly the most well-studied because of its ubiquitous, circumpolar abundance in shallow (most commonly 15 to 200 m) rocky and soft-sediment bottoms (Dearborn, 1977; McClintock et al., 1988). *O. validus* was considered a biomonitor of metals, at Rothera Research Station, Adelaide Island, although for natural, rather than anthropogenic metal bioavailability because of their wide range of food items and feeding behaviour (Arnaud, 1977; Webb et al., 2020). Asteroids, especially *O. validus* and *Perknaster fuscus antarcticus*, were good monitors for anthropogenic bioavailability of PCBs and Cu in our current study. Cu concentrations of *O. validus* at disturbed locations in this study (43 to 58 $\mu g\ g^{-1}$) were similar to those occurring adjacent to Arctowski Station, KGI (35 to 63 $\mu g\ g^{-1}$, Trevizani et al., 2016). Soft tissues of *O. validus* were characterized by having high Cu, Zn and Cd concentrations in Terra Nova Bay, Ross Sea (Grotti et al. 2008). It is probable that *D. brucei* was not identified as a biomonitor species in this study because its small sample size (7 specimens from two transects) did not allow for the statistical power and spatial coverage necessary to determine differences among disturbed and reference locations as occurs with the other three asteroid species that were identified as biomonitors (13-59 specimens from three to five transects).

The three potential biomonitor asteroids identified in our current study also had greater Cd concentrations at each sampling location (6 and 80 to 117 $\mu g\ g^{-1}$) than all other taxa sampled (< 40 $\mu g\ g^{-1}$; Figure S6). De Moreno et al. (1997) documented a similar finding when comparing Cd concentrations in *O. validus* tissues with eleven non-asteroid Antarctic invertebrates in islands and locations along and north of the Antarctic Peninsula (10 to 20 µg $g^{-1}{}_{\text{wet weight}}$ ≈ 50 to 100 µg $g^{-1}{}_{\text{dry weight}}$). Interestingly, there was no difference in Cd concentrations in asteroid tissues among locations in our current study ($p > 0.90$, except in *Perknaster fuscus antarcticus* [$p < 0.09$]), despite there being less Cd in the reference (0.2 µg g^{-1}) compared to the anthropogenically disturbed sediments (0.6 µg g^{-1}). This lack of difference, as well as the two to three order of magnitude apparent BF from the sediments to the tissues indicates a probable natural or non-sediment source. Indeed, several studies have indicated that Cd is not a reliable bioindicator of anthropogenic contamination in the Antarctic because Cd bioaccumulation coincides with upwelling of Cd-rich waters and the resulting increased phytoplankton blooms (e.g., Honda et al., 1987; Bargagli, 1996; Sanchez-Hernandez, 2000). High Cd concentrations in *O. validus* have also been suggested as being attributed to their long lifespan of potentially more than 100 years (Pearse, 1969) rather than Cd availability in their diet (De Moreno et al., 1997). There is evidence in our study that there is a natural source of Cd at TR because of the greater Cd tissue concentrations in *S. neumayeri*, *L. elliptica* and *Parbolasia corrugatus* at TR than those adjacent to McMurdo Station (WQB, Outfall and CA). Our Turtle Rock sampling site is further north and much steeper than the other sampling locations, which could allow Cd-rich upwelling and greater phytoplankton productivity. Weddell Seals (*Leptonychotes weddellii*), who have a colony at Turtle Rock, could also be importing cadmium because they accumulate Cd through their diet, especially in their kidneys and liver, and excrete high concentrations in their urine (Yamamoto et al., 1987; Bargagli 2005). Regardless of the pathway that Cd is bioaccumulated, our study gives no evidence that anthropogenic Cd contamination in the sediments is related to Cd concentrations in Antarctic invertebrate tissues.

Teleost fish (Trematomus spp.)

Trematomus were high in arsenic, DDT and total PCB concentrations but low in Cu, Pb, Fe, Cd, and total PAH concentrations relative to the invertebrates sampled in this study (Figure 3 and Figure 4). However, *Trematomus* is still a potential biomonitor for anthropogenic total PAHs, total PCBs, DDT, and Cu. It has been suggested that organic pollutants (PAHs, PCBs) and/or pathogens from the sewage outfall, rather than metal exposure, caused pathological conditions in *Trematomus bernacchii* occurring in WQB (Evans et al., 2000). Total PCBs and DDT in *Trematomus* in disturbed locations WQB (Total PCBs: 11,044 ng g^{-1}, DDT: 21.6 ng g^{-1}) and the

Outfall (Total PCBs: 7081 ng g^{-1}, DDT: 16.4 ng g^{-1}) were 21 to 1135% (DDTs) and 59 to 2410% (PCBs) greater than in any invertebrate species in this study. These PCB and DDT concentrations at McMurdo Station were much greater than PCBs but similar to DDTs (*p,p*'-DDE + *p,p*'-DDT) in *Trematomus* at Syowa Station, East Ongul Island in 1981 (0.08 to 0.59 ng g^{-1} w.w. ≈ 0.3 to 2.5 ng g^{-1} d.w.; DDTs: 0.4 to 1.5 ng $g^{-1}_{\text{wet weight}}$ ≈ 1.7 to 6.3 ng $g^{-1}_{\text{dry weight}}$; Subramanian et al,. 1983). Half of the whole-body concentrations of bioaccumulated PCBs in *Trematomus* tissues in the disturbed areas (7/11 in WQB, 6/16 at Outfall) are above the U.S. Food and Drug Administration (FDA) action level of 2000 ng $g^{-1}_{\text{wet weight}}$ (≈ 8500 ng $g^{-1}_{\text{dry weight}}$) for total PCBs, but below the action level of 5000 ng $g^{-1}_{\text{wet weight}}$ (≈ 20,800 ng $g^{-1}_{\text{dry weight}}$) for DDT, for the edible parts of finfish and shellfish (FDA, 2020). The FDA will take legal action to remove foods from the market in the US that exceed action limit levels for substances that are deemed poisonous or deleterious to humans, such as PCBs and DDTs. The BF for DDT and PCBs are higher in *Trematomus* than in any other species sampled, which means that *Trematomus* bioaccumulate more of these compounds than benthic invertebrate species. These large BFs mean that PCBs and DDT can likely be passed up the food chain into higher trophic level taxa more readily. This pathway could potentially allow the biomagnification of PCBs and DDTs in Weddell seals (*Leptonychotes weddellii*) in McMurdo Sound because *Trematomus* are part of the seal diet, especially for juveniles (Burns et al., 1998). However, this biomagnification into seals is likely to be minor because the small spatial extent of contamination (≤ 1 km^2; Lenihan et al. 1990) and restricted home range of *Trematomus* (Evans et al., 2000; Bargagli, 2005).

The low PAH concentrations in *Trematomus* relative to invertebrates are probably because PAHs aren't readily accumulated in fish tissues (Varanasi and Gmur, 1981; Kennicutt et al., 1991). It is possible that the higher concentrations of PAHs in whole-body tissues close to McMurdo Station than at the reference TR could be attributed to recently ingested and metabolizing anthropogenic PAHs (McDonald et al., 1992) rather than long-term storage in its tissues.

Other Predator/Scavengers (Isotealia antarctica, Parborlasia corrugatus)

The predatory actinarian *I. antarctica* and predatory scavenger nemertean *P. corrugatus* were both identified as potential biomonitors of PCB, DDT, and PAHs, while *I. antarctica* was also a potential biomonitor of Pb and Cu. Although not much information was found concerning PAHs and PCBs in *Antarctic actinaria*, bioaccumulation of PAHs and PCBs have been documented in deep-sea Actinaria in the Gulf of Mexico (Lawson et al., 2021). The second largest BF for PCBs occurred in *I. antarctica* (6.8). Only two of the whole-body concentrations of bioaccumulated PCBs in *I. antarctica* tissues in the disturbed areas (1/10 in WQB, 1/13 at Outfall) are above the FDA action level of 2000 ng g^{-1} $_{\text{wet weight}}$ (≈ 13300 ng g^{-1} $_{\text{dry weight}}$) for total PCBs.

Cu, Zn and Cd concentrations in *P. corrugatus* measured in WQB and near the sewage outfall and a reference site (Cinder Cones) in 1988 (Lenihan et al., 1990) are similar to those occurring in similar locations in the current study. Background Pb concentrations in *P. corrugatus* were ten times greater in this study (0.5 to 1.3 µg g^{-1}) than in Terra Nova Bay (0.05 µg g^{-1}), however background Cu and Zn concentrations were similar (Grotti et al., 2008). Bioaccumulated concentrations of PCBs, DDT and PAHs are unlikely to be biomagnified further up the food chain from *P. corrugatus* because natural chemical defenses cause ingestion avoidance by potential predators (Heine *et al.* 1991).

Herbivore/Omnivore (Sterechinus neumayeri)

The ubiquitous, circumpolar echinoid *S. neumayeri* is a potential biomonitor of PCBs, PAHs, Pb, Cu and Hg. An Arctic echinoid (*Strongylocentrotus droebachiensis*) was also enriched in Pb, Cu and Hg near a former lead-zinc mine relative to a reference site in Greenland (Søndergaard et al., 2019). Total PCB concentrations in *S. neumayeri* near the Dumont D'Urville Station, Adelie Land, were ~12 ng g^{-1} (Goutte et al., 2013), which is higher than PCB concentrations at reference location TR (0 ng g^{-1}) but lower than PCB concentrations at reference location CA (141 ng g^{-1}). A temperate echinoid *Paracentrotus lividus* was also found to be an efficient bioaccumulator of PCBs in France (Danis et al., 2005). As also observed by de Moreno (1997), *S. neumayeri* had high Zn concentrations (in this study 199 to 282 µg g^{-1}) relative to other invertebrates. There was no difference in Zn among locations in this study, which means *S. neumayeri* is not suitable as a biomonitor for sediment Zn contamination.

Deposit feeder (Flabegraviera mundata)

The surface deposit feeding polychaete *F. mundata* is potentially a good biomonitor for PCBs, DDTs and lead. Total PCB, DDT and Pb concentrations were greater in disturbed (PCB: 1633 to 2138 ng g^{-1}, DDT: 4 to 18 ng g^{-1}, Pb: 10 to 18 µg g^{-1}) than reference locations (PCB: 0 to 83 ng g^{-1}, DDT: 0 ng g^{-1}, Pb: 2 to 5 µg g^{-1}). *F. mundata* has the highest BF of Zn (4.3) and Pb (0.3), and second highest BF for DDT (3.4). Mean *F. mundata* Pb concentrations in disturbed locations (10 to 18 µg $g^{-1}{}_{\text{dry weight}}$ ≈ 1.0 to 1.8 µg $g^{-1}{}_{\text{wet weight}}$) are similar to the maximum level allowable for any food in the European Union (bivalves 1.5 µg $g^{-1}{}_{\text{wet weight}}$; EU, 2020). Mean *F. mundata* Zn concentrations in disturbed locations (29 to 31 µg $g^{-1}{}_{\text{dry weight}}$ ≈ 2.9 to 3.1 µg $g^{-1}{}_{\text{wet weight}}$) are below international criteria allowable for edible fish (60 µg $g^{-1}{}_{\text{wet weight}}$; Summers et al. 1995). Although *F. mundata* is not likely to be consumed by humans, these comparison gives an indication of potential toxicity to other animals. Exported sympagic (sea ice) algae, the main food items of *F. mundata* (Wing et al., 2012; Michel et al., 2019) is not expected to have high contaminant concentrations. However, increased bioaccumulation relative to other species for DDT and Zn could be aided by the direct ingestion of contaminated sediments, or the mucous membrane surrounding *F. mundata*. Zn and DDT could be biomagnified up the food chain more readily into predators of *F. mundata*, such as the giant Antarctic isopod *Glyptonotus antarcticus* (Brueggeman, 2021).

Filter feeders (Laternula elliptica and *Alcyonium antarcticum*)

The burrowing bivalve *L. elliptica* is a ubiquitous, circumpolar filter feeder that is frequently used or proposed as a suitable biomonitor species (e.g., Kennicutt et al., 1991; Ahn et al., 1996; Vodopivez et al., 2015; Webb et al., 2020). Trace metals accumulated in *L. elliptica* are similar to those occurring in oysters and mussels, which are used for large-area (1000s of km) biomonitoring programs (Sericano et al., 1995; Ahn et al., 1996). In our study, *L. elliptica* is a suitable anthropogenic biomonitor for total PCBs, PAHs, Pb and Cu. Total PAH concentrations in *L. elliptica* tissues at disturbed locations WQB (538 ng g^{-1}) and the Outfall (258 ng g^{-1}) in this current study were much lower than those occurring close to Palmer Station immediately after the *Bahía Paraíso* shipwreck and subsequent fuel spill in 1989 (1222 and 17,498 ng g^{-1}; Kennicutt et al., 1991) but higher than a 2015 sample immediately adjacent to Palmer Station (26.2 ng g^{-1}, unpublished data). Cu concentrations were 1.5 orders of magnitude lower in *L. elliptica* at reference location CA in this current study (13 µg g^{-1}) than in a study that sampled in 2002 (275 µg g^{-1}) but similar between studies at reference location TR (current study: 19 µg g^{-1}, 2002: 20 µg g^{-1}; Negri et al., 2006). Cu concentrations in *L. elliptica* tissues in disturbed locations WQB (25 µg g^{-1}) and the Outfall (275 µg g^{-1}) were also lower than those sampled adjacent to Scott Base, Ross Island in 2002 (82 to 170 µg g^{-1}, Negri et al. 2006). Pb concentrations in *L. elliptica* at reference TR were lower in the current study (1.0 µg g^{-1}) than in the 2002 specimens (5.4 µg g^{-1}), but similar at reference CA (current study: 2.4 µg g^{-1}; 2002: 3.1 µg g^{-1}; Negri et al. 2006).

The soft (alcyonacean) coral *A. antarcticum* differs from *L. elliptica* in that it feeds on microscopic particles high in the water column rather than particles at the sediment surface (Dayton and Oliver, 1977; Conlan et al., 2006; Michel et al., 2019). Although documentation about bioaccumulation and biomonitoring of soft corals in Antarctica is sparse, *A. antarcticum* is a biomonitor for Cu, Pb and PAHs in this current study. Cu, Pb and total PAH concentrations were higher in *A. antarcticum* tissues at the disturbed Outfall (Cu: 10 µg g^{-1}, Pb: 3 µg g^{-1}, PAHs: 1354 ng g^{-1}) than at the reference CA (Cu: 3 µg g^{-1}, 1 Pb: µg g^{-1}, PAHs: 295 ng g^{-1}), the only other location that the species was sampled. Soft coral *Leptogorgia setacea* contained total PAHs, including those from pyrogenic sources, at higher concentrations than in sediments in the Gulf of Mexico (Sabourin et al., 2013). The highest BF for total PAHs occurs in *A. antarcticum.* In comparison, a BF of approximately 3.9 occurred in a hard (scleractinian) coral (Acropora), which was higher than two fish and a bivalve in the Great Barrier Reef, Australia (Coates et al., 1986). However, it is speculated that PAHs bioaccumulate in hard corals from the water column rather than sediments (Ko et al., 2014). The soft coral *A. antarcticum* is the only water filter feeder in this study, so may bioaccumulate PAHs better than other species. Natural biochemical deterrents occurring in *A. antarcticum* tissues result in the coral having few known predators aside from a pycnogonid (*Colossendeis megalonyx*, Slattery and McClintock 1985), which means there is minimal risk of biomagnification from *A. antarcticum* to higher trophic levels.

5. Summary

In addition to PCBs and DDT, which have no natural origins, this study identified total PAHs, Cd, Cu, Pb, Hg, and Zn as locally sourced anthropogenic contaminants in marine sediments adjacent to McMurdo Station. All of these organic chemicals and metals, aside from Hg and Cd, can be used in localized sediment contamination biomonitoring because they are concentrated in benthic teleost and invertebrate fauna tissues in greater concentrations where greater sediment concentrations occur. Ten diverse and relatively easy to collect (and analyze) taxa were deemed to be potential biomonitor species of local anthropogenic contamination including a deposit feeding polychaete (*F. mundata*), filter feeding bivalve (*L. elliptica*) and soft coral (*A. antarcticum*), a predatory anemone (*I. antarctica*), an herbivore/omnivore echinoid (*S.*

neumayeri), and predatory/scavenger teleost (*Trematomus*), nemertean (*Parborlasia corrugatus*) and asteroids (*O. validus*, *Perknaster fuscus antarcticus*, *Psilaster charcoti*). Given that ten out of twelve species sampled proved to bioaccumulate localized contaminants in greater quantities in contaminated compared to reference locations, it is likely that bioaccumulation of local contaminants is prevalent throughout the diverse marine food web in the small (≤ 1 km^2) contaminated area adjacent to McMurdo Station, and other research stations where intense sediment contamination may exist. It is possible that bioaccumulated contaminants are biomagnified into larger organisms such as *Leptonychotes weddellii*, whose diet directly and indirectly partially comprises benthic fauna. However, much of the benthic tissue contamination is of low levels relative to human consumption standards, except for Pb in the polychaete *Flabegraviera mundata* and PCBs in the fish *Trematomus*.

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CRediT authorship contribution statement

Terence A. Palmer: Conceptualization, Investigation, Methodology, Software, Formal analysis, Investigation, Data curation, Writing - original draft, Visualization, Supervision, Project administration. Andrew G. Klein: Conceptualization, Methodology, Investigation, Data curation, Writing - review and editing, Supervision, Project administration, Funding acquisition. Stephen T. Sweet: Methodology, Investigation, Writing – review and editing, Supervision, Project administration. Amanda J. Frazier: Formal Analysis, Methodology, Writing - review and editing. Paul A. Montagna: Writing - review and editing, Supervision, Project administration, Funding acquisition. Terry L. Wade: Investigation. Jennifer Beseres Pollack: Investigation, Writing - review and editing.

Declaration of competing interest

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

References

Ahn, I.-Y., Lee, S.H., Kim, K.T., Shim, J.H., Kim, D.-Y., 1996. Baseline heavy metal concentrations in the Antarctic clam, *Laternula elliptica* in Maxwell Bay, King George Island, Antarctica. Mar. Pollut. Bull. 32, 592–598. https://doi.org/10.1016/0025-326X(95) 00247-K.

Arnaud, P.M., 1977. Adaptations within the Antarctic marine benthic ecosystem. *In:* Llano GA (ed) Adaptations within Antarctic ecosystems: Proceedings of the third SCAR Symposium on Antarctic biology Gulf Publishing Company, Houston, pp 135-157

Aronson, R.B. Blake, D.B., 2001. Global climate change and the origin of modern benthic communities in Antarctica. Am. Zool. 41: 27–39. https://doi.org/10.1093/icb/41.1.27

Aronson, R.B., Thatje, S., McClintock, J.B., Hughes, K.A., 2011. Anthropogenic impacts on marine ecosystems in Antarctica. Ann. N. Y. Acad. Sci. 1223, 82–107. https://doi.org/10.1111/j.1749-6632.2010.05926.x

Bargagli, R., Nelli, L., Ancora, S., Focardi, S., 1996. Elevated cadmium accumulation in marine organisms from Terra Nova Bay (Antarctica). Polar Biol. 16, 513-520. https://doi.org/10.1007/BF02329071

Bargagli, R., 2005. *Antarctic Ecosystems. Environmental Contamination, Climate Change, and Human Impact*. Ecological Studies 175, Springer.

Bento, B., Hintelmann, H., dos Santos, M.C., Cesário, R., Canárioa, J., 2021. Mercury methylation rates in Deception Island (Maritime Antarctica) waters and pyroclastic gravel impacted by volcanic mercury. Mar. Pollut. Bull., 164, 112023. https://doi.org/10.1016/j.marpolbul.2021.112023.

Brueggeman, P., 2021. Underwater Field Guide to Ross Island & McMurdo Sound, Antarctica: Annelida. http://peterbrueggeman.com/nsf/fguide/index.html Visited 01 May 2021.

Burns, J.M., Trumble, S.J., Castellini, M.A., Testa., J.W., 1998. The diet of Weddell seals in McMurdo Sound, Antarctica as determined from scat collections and stable isotope analysis. *Polar Biology* 19, 272–282. https://doi.org/10.1007/s003000050245

Canadian Council of Ministers of the Environment, 2009. Canadian Sediment Quality Guidelines for the Protection of Aquatic Life: Polychlorinated Biphenyls (PCBs). Canadian Council of Ministers of the Environment, Winnipeg.

Clarke, A., Aronson, R., Crame, J., Gili, J., Blake, D., 2004. Evolution and diversity of the benthic fauna of the Southern Ocean continental shelf. Antarct. Sci. 16(4), 559-568. https://doi.org/10.1017/S0954102004002329

Coates, M., Connell, D.W., Bodero, J., Miller, G.J., Back. R., 1986. Aliphatic hydrocarbons in Great Barrier Reef organisms and environment. Est. Coast. Shelf Sci. 23, 99-113.

Conlan, K.E., Kim, S.L., Lenihan, H.S., Oliver, J.S., 2004. Benthic changes during 10 years of organic enrichment by McMurdo Station, Antarctica. Mar. Pollut. Bull., 49 (1–2), 43–60. https://doi.org/10.1016/j.marpolbul.2004.01.007.

Conlan, K.E., Rau, G.H., Kvitek, R.G., 2006. δ13 C and δ15 N shifts in benthic invertebrates exposed to sewage from McMurdo Station, Antarctica. Mar. Pollut. Bull. 52, 1695–1707. https://doi.org/10.1016/j.marpolbul.2006.06.010

Creek, J.T., Brockhoff, C.A., Martin, T.D., 1994. Method 200.8 Determination of Trace Elements in Waters and Wastes by Inductively Coupled Plasma—Mass Spectrometry. US EPA, Cincinnati, OH (42 pp.).

Cripps, G.C., 1992. The extent of hydrocarbon contamination in the marine environment from a research station in the Antarctic. Mar. Pollut. Bull. 25:288–92.

Cripps, G.C., 1994. Hydrocarbons in the Antarctic Marine Environment: Monitoring and Background, Int. J. Environ. Anal. Chem. 55(1-4), 3-13, https://doi.org/10.1080/03067319408026204

Curtosi, A., Pelletier, E., Vodopivez, C.L., Mac Cormack, W.P., 2007. Polycyclic aromatic hydrocarbons in soil and surface marine sediment near Jubany Station (Antarctica). Role of permafrost as a low-permeability barrier. Sci. Total Environ. 383(1-3), 193-204. https://doi.org/10.1016/j.scitotenv.2007.04.025.

Danis, B., Cotret, O. Tyssié, J., Bustamante, P., Fowler, S., Warnau, M., 2005. Bioaccumulation of PCBs in the sea urchin *Paracentrotus lividus*: seawater and food exposures to a ^{14}C-radiolabelled congener (PCB#153). Environ. Pollut. 135: 11-16. https://doi.org/10.1016/j.envpol.2004.10.011

Dayton, P.K., Robilliard, G.A., 1971. Implications of pollution to the McMurdo Sound benthos. *Antarctic Journal of the United States*, 6, 53–56.

Dayton, P.K., Oliver, J.S., 1977. Antarctic soft-bottom benthos in oligotrophic and eutrophic environments. Science 197, 55–58.

De Moreno, J.E.A., Gerpe, M.S., Moreno, V.J., Vodopivez, C., 1997. Heavy metals in Antarctic organisms. Polar Biol. 17(2), 131-140.

Dearborn, J. H. (1977). Foods and feeding characteristics of Antarctic asteroids and ophiuroids. In: Llano, G. A. (ed.) Adaptations within Antarctic ecosystems. Gulf Publishing Co., Texas, p. 293-326

Egger, F., 2003. Antarctica research station adds sewage treatment plant. Water and Wastewater Intl. 18, 34.

Evans, C.W.E., Hills, J.M., Dickson, J.M.J., 2000. Heavy metal pollution in Antarctica: a molecular ecotoxicological approach to exposure assessment. J. Fish Biol. 57, 8-19. https://doi.org//10.1111/j.1095-8649.2000.tb02241.x

European Union. 2020. Commission Regulation (EC) No 1881/2006 of 19 December 2006 setting maximum levels for certain contaminants in foodstuffs. Official Journal of the European Union. 02006R1881 — EN — 14.10.2020 — 027.001 — 1. http://data.europa.eu/eli/reg/2006/1881/2020-10-14

Food and Drug Administration (FDA). 2020. Guidance for Industry: Action Levels for Poisonous or Deleterious Substances in Human Food and Animal Feed. U.S. Department of Health and Human Services, Public Health Service, Washington, D.C . https://www.fda.gov/regulatory-information/search-fda-guidance-documents/guidance-industry-action-levels-poisonous-or-deleterious-substances-human-food-and-animal-feed

Fuoco, R., Colombini M.P., Abete C., 1994. Determination of polychlorobiphenyls in environmental samples from Antarctica. Int. J. Environ. Anal Chem. 55, 15-25

Goutte, A., Chevreuil, M., Alliot, F., Chastel, O., Cherel, Y., Eléaume, M., Massé, G., 2013. Persistent organic pollutants in benthic and pelagic organisms off Adélie Land, Antarctica. Mar. Pollut. Bull. 77, 82–89. https://doi.org/10.1016/j.marpolbul.2013.10.027

Grotti, M., Soggia, F., Lagomarsino, C., Riva, S.D., Goessler, W., Francesconi, K.A., 2008. Natural variability and distribution of trace elements in marine organisms from Antarctic coastal environments. Antarct. Sci. 20(1), 39-52. http://dx.doi.org/10.1017/S0954102007000831

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa C. et al., 2008. A global map of human impact on marine ecosystems. Science 319: 948–952. https://doi.org/10.1126/science.1149345

Heine, J,N., McClintock, J.B., Slattery, M., Weston, J., 1991. Biochemical composition, energy content, and chemical defense in the common Antarctic nemertean *Parborlasia corrugatus*. J. Exp. Mar. Biol. Ecol. 153, 15-25.

Honda, K., Yamamoto, Y., Tatsukawa, R., 1987. Distribution of heavy metals in Antarctic marine ecosystems. Proc NIPR Symp Polar Biol 1:184-197.

Kennicutt, M.C., Klein, A., Montagna, P., Sweet, S., Wade, T., Palmer, T., Denoux, G., 2010. Temporal and spatial patterns of anthropogenic disturbance at McMurdo Station, Antarctica. Environ. Res. Lett. 5, 34010. https://doi.org/10.1088/1748-9326/5/3/034010

Kennicutt II, M.C., McDonald, S.J., Sericano, J.L., Boothe, P., Oliver, J., Safe, S., Presley, B.J., Liu, H., Wolfe, D. Wade, T.L., Crockett, A., Bockus, D., 1995. Human Contamination of the Marine Environment-Arthur Harbor and McMurdo Sound, Antarctica. Environ. Sci. Technol. 29 (5), 1279-1287, https://doi.org/10.1021/es00005a600

Kennicutt, M.C. II, Sweet, S.T., Fraser, W.R., Stockton, W.L., Culver, M., 1991. Grounding of the Bahia Paraiso, Arthur Harbor, Antarctica--I. Distribution and fate of oil spill related hydrocarbons. Environ. Sci. and Tech. 25, 509-518.

Kennish, M.J., ed. 1997. Practical Handbook of Estuarine and Marine Pollution. Boca Raton, USA: CRC Press: 524 pp.

Klein, A.G., Kennicutt, M.C. II, Wolff, G.A., Sweet, S.T., Bloxom, T., Gielstra D.A., Cleckley, M., 2008 The historical development of McMurdo station, Antarctica, an environmental perspective, Polar Geogr. 31(3-4), 119-144, https://doi.org/10.1080/10889370802579856

Klein, A.G., Sweet, S.T., Wade, T.L., Sericano, J.L., Kennicutt, M.C. II, 2012. Spatial patterns of total petroleum hydrocarbons in the terrestrial environment at McMurdo Station, Antarctica. Antarct. Sci. 24 (5), 450–466. https://doi.org/10.1017/S0954102012000429.

Ko, F.-C., Chang, C.-W., Cheng, J.-O., 2014. Comparative study of polycyclic aromatic hydrocarbons in coral tissues and the ambient sediments from Kenting National Park, Taiwan. Environ. Pollut. 185, 35–43. https://doi.org/10.1016/j.envpol.2013.10.025

Kuzyk, Z.A., Stow, J.P., Burgess, N.M., Solomon, S.M., Reimer, K.J., 2005. PCBs in sediments and the coastal food web near a local contaminant source in Saglek Bay, Labrador. Sci. Total Environ. 351–352. https://doi.org/10.1016/j.scitotenv.2005.04.050.

Lawson, M.C., Cullen, J.A., Nunnally, C.C., Rowe, G.T., Hala, D.N., 2021. PAH and PCB body-burdens in epibenthic deep-sea invertebrates from the northern Gulf of Mexico. Mar. Pollut. Bull. 162, 111825. https://doi.org/10.1016/j.marpolbul.2020.111825.

Lenihan, H.S., 1992. Benthic marine pollution around McMurdo Station, Antarctica: a summary of findings. Mar. Pollut. Bull. 25, 318–323. https://doi.org/10.1016/0025-326X(92)90689-4.

Lenihan, H.L., Oliver, J.S., Oakden, J.M., Stephenson, M.A., 1990. Intense and localized benthic marine pollution at McMurdo Station, Antarctica. Mar. Pollut. Bull. 21, 264-269. https://doi.org/10.1016/0025-326X(90)90761-V

Long, E.R., MacDonald, D.D., Smith, S.L., Calder, F.D., 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Environ. Manag. 19, 81–97.

Martins, C.C., Bícego, M.C., Taniguchi, S, Montone, R.C., 2004. Aliphatic and polycyclic aromatic hydrocarbons in surface sediments in Admiralty Bay, King George Island, Antarctica. Antarct. Sci. 16, 117–22. https://doi.org/10.1017/S0954102004001932

McClintock, J.B., Pearse, J.S., Bosch, I., 1988. Population structure and energetics of the shallow-water Antarctic sea star *Odontaster validus* in contrasting habitats. Marine Biol. 99, 235–246. https://doi.org/10.1007/BF00391986.

McDonald S.J., Kennicutt M.C., Brooks J.M., 1992. Evidence of Polycyclic Aromatic Hydrocarbons (PAH) exposure in fish from the Antarctic Peninsula. Mar. Pollut. Bull. 25, 313-317.

Michel, L.N., Danis, B., Dubois, P., Eleaume, M., Fournier, J., Gallut, C., Jane, P., Lepoint, G., 2019. Increased sea ice cover alters food web structure in East Antarctica. Sci. Rep. 9, 8062. https://doi.org/10.1038/s41598-019-44605-5

Montone, R.G., Taniguehi, S., Weber, R.R., 2001. Polychlorinated biphenyls in marine sediments of Admiralty Bay, King George Island, Antarctica. Mar. Pollut. Bull. 42, 611-614

Morehead, S., Montagna, P.A., Kennicutt II, M.C., 2008. Comparing fixed-point and probabilistic sampling designs for monitoring the marine ecosystem near McMurdo Station, Ross Sea, Antarctica. Antarct. Sci. 20 (5), 471–484. https://doi.org/10.1017/S0954102008001326.

Negri, A., Burns, K., Boyle, S., Brinkman, D., Webster, N., 2006. Contamination in sediments, bivalves and sponges of McMurdo Sound, Antarctica. Environ. Pollut. 143(3), 456–467. https://doi.org/10.1016/j.envpol.2005.12.005.

National Oceanic and Atmospheric Administration, 1993. NOAA Technical Memorandum NOS ORCA 71 1 (182 pp.).

Orr, J.C., Fabry, V.J., Aumont, O., Bopp, L., Doney, S.C., Feely, R.A., Gnanadesikan, A., Gruber, N., Ishida, A., Joos, F., Key, R.M., Lindsay, K., Maier-Reimer, E., Matear, R., Monfray, P., Mouchet, A., Najjar, R.G., Plattner, G.K., Rodgers, K.B., Sabine, C.L., Sarmiento, J.L., Schlitzer, R., Slater, R.D., Totterdell, I.J., Weirig, M.F., Yamanaka, Y., Yool, A., 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. Nature 437, 681–686. https://doi.org/10.1038/nature04095

Palmer, T.A., Klein, A.G., Sweet, S.T. Montagna, P.A., Hyde, L.J., Beseres Pollack, J., 2021. Anthropogenic effects on the marine environment adjacent to Palmer Station, Antarctica. Antarct. Sci. 1-18. https://doi.org/10.1017/S0954102021000535

Palmer, T.A., Klein, A.G., Sweet, S.T., Montagna, P.A., Serciano, J., Hyde, L.J., Wade, T., Kennicutt II, M.C., Beseres Pollack, J., 2020. Long-term changes in contamination and macrobenthic communities adjacent to McMurdo Station, Antarctica. Sci. Total Environ. 142798. https://doi.org/10.1016/j.scitotenv.2020.142798

Pearse J.S., 1969, Antarctic sea star, Australian Nat. Hist. 16, 234–238.

Peck, L.S., Convey, P., Barnes, D.K.A., 2006. Environmental constraints on life histories in Antarctic ecosystems: tempos, timings and predictability. Biol. Rev. Camb. Philos. Soc. 81, 75–109. https://doi.org/10.1017/S1464793105006871.

Poland, J.S., Mitchel, S., Rutter, A., 2001. Remediation of former military bases in the Canadian Arctic. Cold Reg. Sci. Technol., 32(2-3), 93-105. https://doi.org/10.1016/S0165-232X(00)00022-7

Rainbow, P.S., Phillips, D.J.H., 1993. Cosmopolitan biomonitors of trace metals. Mar. Pollut. Bull. 26, 593–601. https://doi.org/10.1016/0025-326X(93)90497-8

Rainbow, P.S., 1995. Biomonitoring of heavy metal availability in the marine environment. Mar. Pollut. Bull. 31, 183–192. https://doi.org/10.1016/0025-326X(95)00116-5Risebrough, R.W., Walker, W., II, Schmidt, T.T., de Lappe, B.W., Connors, C.W., 1976. Transfer of chlorinated biphenyls to Antarctica. Nature 264, 738–739.

Sabourin, D.T., Silliman, J.E., Strychar, K.B., 2013. Polycyclic aromatic hydrocarbon contents of coral and surface sediments off the South Texas coast of the Gulf of Mexico. Int. J. Biol. 5, 1-12. https://doi.org/10.5539/ijb.v5n1p1

Sanchez-Hernandez, J.C., 2000. Trace element contamination in Antarctic ecosystems. Rev. Environ. Contam. Toxicol. 166, 83-127.

SAS Institute Inc, 2017. SAS/STAT® 14.3 User's Guide. SAS Institute Inc., Cary, NC.

Sericano, J.L., Wade, T.L., Jackson, T.J., Brooks, J.M., Tripp, B.W., Farrington, J.W., 1995. Trace organic contamination in the Americas: an overview of the us national status & trends and the international "mussel watch" programs. Mar. Pollut. Bull. 31, 214–225.

Slattery, M., McClintock, J.B., 1995. Population structure and feeding deterrence in three shallow-water Antarctic soft corals. Mar. Biol. 122, 461–470. https://doi.org/10.1007/BF00350880

Søndergaard, J., Hansson, S.V., Mosbech, A., Bach, L., 2019. Green sea urchins (*Strongylocentrotus droebachiensis*) as potential biomonitors of metal pollution near a former lead-zinc mine in West Greenland. Environ. Monit. Assess. 191(9), 538. https://doi.org/10.1007/s10661-019-7637-3.

Subramanian, B.R., Tanabe, S., Hidaka, H., Tatsukawa, R., 1983. DDTs and PCB Isomers and Congeners in Antarctic Fish. Arch. Environ. Contam. Toxicol. 12, 621-626

Summers, J. K., Paul, J. F., Robertson, A.. 1995. Monitoring the ecological condition of estuaries in the United States. Toxicol. Environ. Chem. 49: 93–108. https://doi.org/10.1080/02772249509358180

Telliard,W.A., 1989. Method 1620 Metals by Inductively Coupled Plasma Atomic Emission Spectroscopy and Atomic Absorption Spectroscopy. US EPA, Alexandria, VA.

Thompson, B.W., Riddle, M.J., Stark, J.S., 2003. Cost-efficient methods for marine pollution monitoring at Casey Station, East Antarctica: the choice of sieve mesh-size and taxonomic resolution. Mar. Pollut. Bull. 46 (2), 232–243, https://doi.org/10.1016/S0025-326X(02)00366-1.

Tin, T., Fleming, Z., Hughes, K.A., Ainley, D., Convey, P., Moreno, C., Pfeiffer, S., Scott, J., Snape, I., 2009. Impacts of local human activities on the Antarctic environment: a review. Antarct. Sci. 21, 3–33. https://doi.org/10.1017/S0954102009001722

Trevizani, T.H., Figueira, R.C., Ribeiro, A.P., Theophilo, C.Y., Majer, A.P., Petti, M.A., Corbisier, T.N., Montone, R.C., 2016. Bioaccumulation of heavy metals in marine organisms and sediments from Admiralty Bay, King George Island, Antarctica. Mar. Pollut. Bull. 106(1-2), 366-71. https://doi.org/10.1016/j.marpolbul.2016.02.056

Turner, J., Barrand, N.E., Bracegirdle, T.J., Convey, P., Hodgson, D.A., Jarvis, M., et al., 2014. Antarctic climate change and the environment: An update. Polar Rec., 50(3), 237-259. https://doi.org/10.1017/S0032247413000296

United States Environmental Protection Agency. 1975. DDT: a review of scientific and economic aspects of the decision to ban its use as a pesticide. Washington, DC: US EPA, 307 pp.

United States Environmental Protection Agency. 1976. PCBs in the United States: industrial use and environmental distribution. EPA report 560/6-76-005. Washington, DC: US EPA, 485 pp.

Varanasi, U., Gmur, D.J., 1981. Hydrocarbons and metabolites in English sole (*Parophyrys vetulus*) exposed simultaneously to [^{3}H] benz[a]pyrene and [^{14}C] naphthalene in oil-contaminated sediment. Aquat. ToxicoL 1, 40-67.

Vodopivez, C., Curtosi, A., Villaamil, E., Smichowski, P., Pelletier, E., Mac Cormack, W.P., 2015. Heavy metals in sediments and soft tissues of the Antarctic clam *Laternula elliptica*: more evidence as a possible biomonitor of coastal marine pollution at high latitudes? Sci. Total Environ. 502, 375-384, https://doi.org/10.1016/j.scitotenv.2014.09.031

Webb, A.L., Hughes, K.A., Grand, M.M., Lohan, M.C., Peck, L.S., 2020. Sources of elevated heavy metal concentrations in sediments and benthic marine invertebrates of the western Antarctic Peninsula. Sci. Total Environ. 698, 134268. https://doi.org/10.1016/j.scitotenv.2019.134268

Wing, S.R., McLeod, R.J., Leichter, J.J., Frew, R.D., Lamare, M.D., 2012. Sea ice microbial production supports Ross Sea benthic communities: influence of a small but stable subsidy. Ecology 93, 314–323. https://doi.org/10.1890/11-0996.1

Yamamoto, Y., Honda, K., Hidaka, H., Tatsukawa, R., 1987.Tissue distribution of heavy metals in Weddell seals (*Leptonychotes weddellii*). Mar. Pollut. Bull. 18, 164–169

Tables

*Table 1. Number of samples and range of years of each taxa were analyzed from each transect. Shaded cells indicate disturbed transects. The numbers in parentheses represent additional samples that were analyzed for metals only. *includes those listed for Trematomus. WQB = Winter Quarters Bay, O = Outfall, CA-E = Cape Armitage transect E, CA-F = Cape Armitage transect F, TR = Turtle Rock*

Dominant Feeding Type	Taxa	Species	n					Year range				
			WQB	O	CA-E	CA-F	TR	WQB	O	CA-E	CA-F	TR
Deposit Feeder	Polychaete	*Flabelligera mundata*	$2^{(1)}$	$3^{(2)}$	$2^{(1)}$		3	2004-2010	2007-2011	2010-2015		2011-2015
Filter Feeder	Bivalve	*Laternula elliptica*	3	3	5	3	4	2004-2011	2000-2012	2000-2012	2004-2007	2008-2012
Filter Feeder	Soft coral	*Alcyonium antarcticum*		4	4	1			2009-2010	2009-2010	2009	
Herbivore	Echinoid	*Sterechinus neumayeri*	17	17	3	4	3	2000-2010	2000-2010	2009-2010	2004-2009	2011
Predator	Actinarian	*Isotealia antarctica*	10	13	8	8		2005-2012	2000-2012	2009-2012	2005-2009	
Predator/Scavenger	Teleost	*Trematomus*	11	16	5	8	2	2004-2012	2004-2012	2008-2012	2004-2009	2013-2015
		*Trematomus bernacchii**	7	11	3	6	2	2004-2012	2004-2012	2008-2012	2004-2007	2013-2015
Predator/Scavenger	Nemertean	*Parborlasia corrugatus*	11	15	6	10	7	2005-2012	2005-2012	2008-2012	2005-2009	2008-2012
Predator/Scavenger	Nudibranch	*Tritoniella belli*		4	$1^{(1)}$	2			2011-2015	2010-2013	2004	
Predator/Scavenger	Asteroid	*Diplasterias brucei*		3	4				2010-2012	2010-2012		
Predator/Scavenger	Asteroid	*Odontaster validus*	17	15	10	12	5	2000-2012	2000-2012	2000-2012	2000-2008	2008-2012
Predator/Scavenger	Asteroid	*Perknaster fuscus antarcticus*	2		2	6	3	2004-2015		2010	2006-2008	2011-2012
Predator/Scavenger	Asteroid	*Psilaster charcoti*	3	4		6		2004	2004-2005		2000-2005	
		Sum	$76^{(1)}$	$97^{(2)}$	$50^{(2)}$	60	27					

Table 2. Mean concentrations of contaminants in sediments and fauna in disturbed locations relative to the Cape Armitage and Turtle Rock reference locations. Inf. = infinity because concentrations were zero at the reference station

Taxa Type	Species	Ar	Cd	Cu	Fe	Pb	Hg	Zn	PAHs	PCBs	DDT
Relative to Cape Armitage reference location											
	Sediment	1.2	2.3	3.5	1.6	6.4	5.8	1.4	238.3	42.9	19.4
Polychaete	*Flabegraviera mundata*	1.1	4.3	2.6	2.1	3.1	2.3	5.3	0.8	22.7	inf.
Bivalve	*Laternula elliptica*	0.5	0.6	1.9	3.9	4.4	0.6	0.8	7.5	8.3	3.7
Soft coral	*Alcyonium antarcticum*	2.3	1.4	2.8	4.0	4.1	0.9	2.8	4.6	4.0	1.5
Echinoid	*Sterechinus neumayeri*	1.1	0.9	3.8	2.0	4.3	1.1	1.4	3.9	18.5	3.8
Actinarian	*Isotealia antarctica*	1.2	0.9	2.2	5.7	4.8	1.1	1.0	4.7	39.3	5.2
Teleost	*Trematomus*	0.7	0.7	0.9	1.2	0.6	1.3	0.8	1.2	25.4	4.7
Nemertean	*Parborlasia corrugatus*	1.9	0.7	1.3	1.9	0.6	0.8	1.0	1.9	32.1	2.4
Nudibranch	*Tritoniella belli*		0.5	1.0	0.4	1.9	0.4	0.5	2.4	1.5	0.1
Asteroid	*Diplasterias brucei*	0.7	0.1	1.3	2.0	4.1	0.8	0.6	0.7	2.3	2.4
Asteroid	*Odontaster validus*	0.9	1.1	1.4	3.8	1.3	0.7	0.9	1.2	8.8	1.9
Asteroid	*Perknaster fuscus antarcticus*	0.3	0.8	7.1	4.7	0.5	0.3	0.6	1.0	40.1	3.6
Asteroid	*Psilaster charcoti*	0.6	0.7	1.1	0.9	2.4	0.8	0.9	1.8	11.8	2.4
Relative to Turtle Rock reference location											
	Sediment	2.4	3.4	1.8	0.5	39.6	18.8	1.4	95.4	41.0	838.2
Polychaete	*Flabegraviera mundata*		2.8	1.9	2.1	8.8	2.7	5.4	0.8	inf.	inf.
Bivalve	*Laternula elliptica*	0.5	0.4	1.3	1.2	9.9	0.6	0.8	13.1	161.8	69.0
Echinoid	*Sterechinus neumayeri*		0.4	2.0	1.5	14.4	1.7	1.0	0.5	inf.	inf.
Teleost	*Trematomus*		6.0	6.1	2.0	4.4	2.1	1.8	9.2	350.3	15.8
Nemertean	*Parborlasia corrugatus*	1.9	0.4	0.8	0.4	1.5	1.2	1.0	10.5	795.7	14.1
Asteroid	*Odontaster validus*	1.1	1.1	2.5	2.2	3.4	1.1	0.9	2.3	138.8	1.8
Asteroid	*Perknaster fuscus antarcticus*	0.4	14.6	8.9	1.7	5.6	0.5	1.6	1.5	586.1	55.2

Table 3. Bioaccumulation factors ([organism]/[sediment]) at Winter Quarters Bay and the Outfall. Values of each variable are color-coded from green to red (small to large).

Taxa	Species	Copper	Lead	Zinc	DDT	PAHs	PCBs
Polychaete	*Flabegraviera mundata*	0.5	0.3	4.3	3.4	0.3	2.6
Bivalve	*Laternula elliptica*	0.5	0.2	1.6	1.4	0.4	1.3
soft coral	*Alcyonium antarcticum*	0.2	0.1	2.0	1.4	1.4	0.6
Urchin	*Sterechinus neumayeri*	0.2	0.1	3.9	1.2	0.2	3.6
Actiniarian	*Isotealia antarctica*	0.3	0.1	3.6	2.1	0.5	6.8
Teleost	*Trematomus*	0.1	0.0	1.0	5.9	0.1	12.7
Nemertean	*Parborlasia corrugatus*	0.1	0.0	2.3	0.7	0.2	3.6
Nudibranch	*Tritoniella belli*	0.1	0.0	1.1	0.4	1.4	1.9
Asteroid	*Diplasterias brucei*	0.2	0.0	0.7	0.5	0.1	0.5
Asteroid	*Odontaster validus*	1.2	0.1	1.4	1.1	0.1	1.2
Asteroid	*Perknaster fuscus antarcticus*	1.8	0.0	1.1	0.9	0.1	2.7
Asteroid	*Psilaster charcoti*	0.8	0.1	1.0	1.7	0.1	1.5

Figures

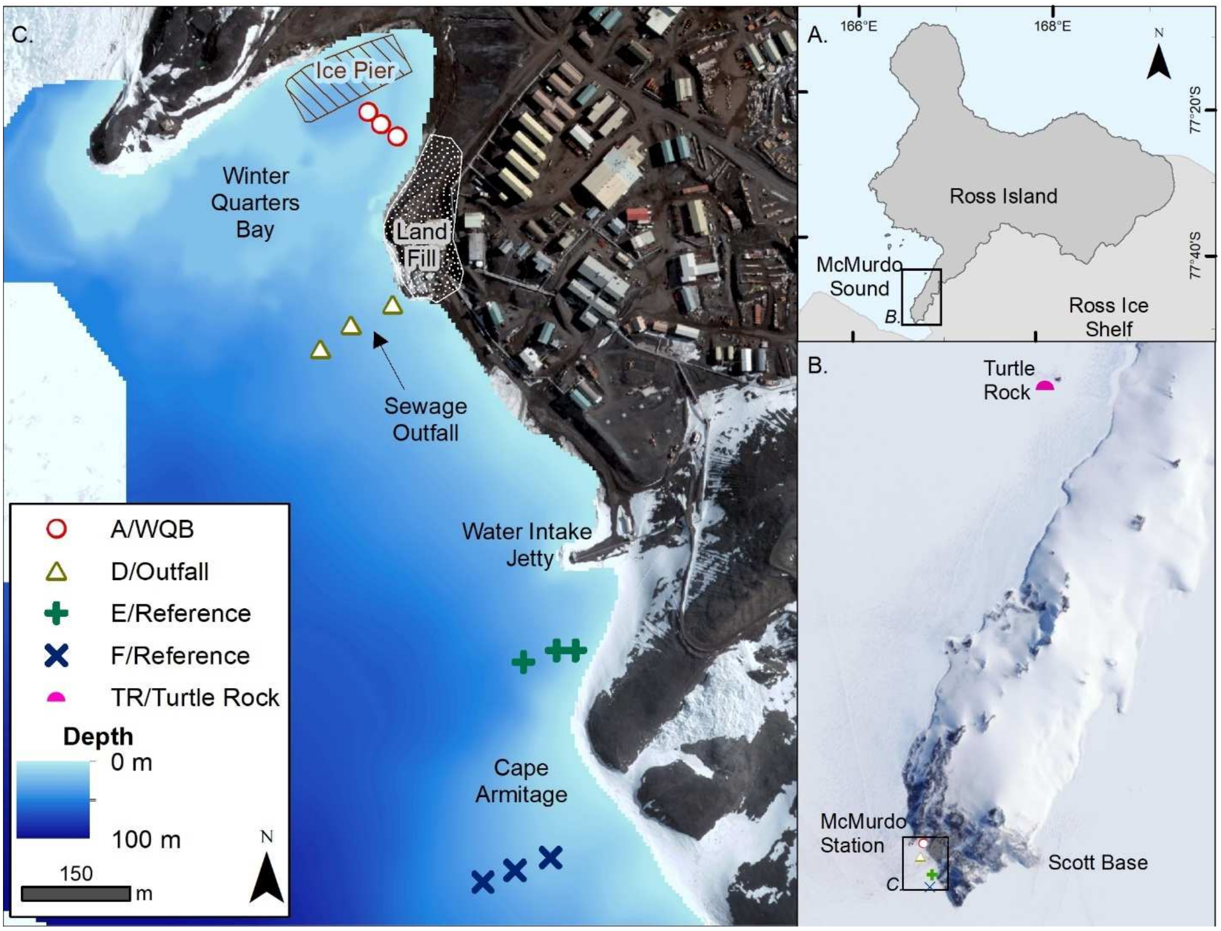

Figure 1. Sampling transects and stations. Disturbed transects = open symbols. Depths of sampling stations within each transect increase with distance away from land (12, 24,36 m).WQB=Winter Quarters Bay. Area of landfill approximated from Lenihan et al., 1990.

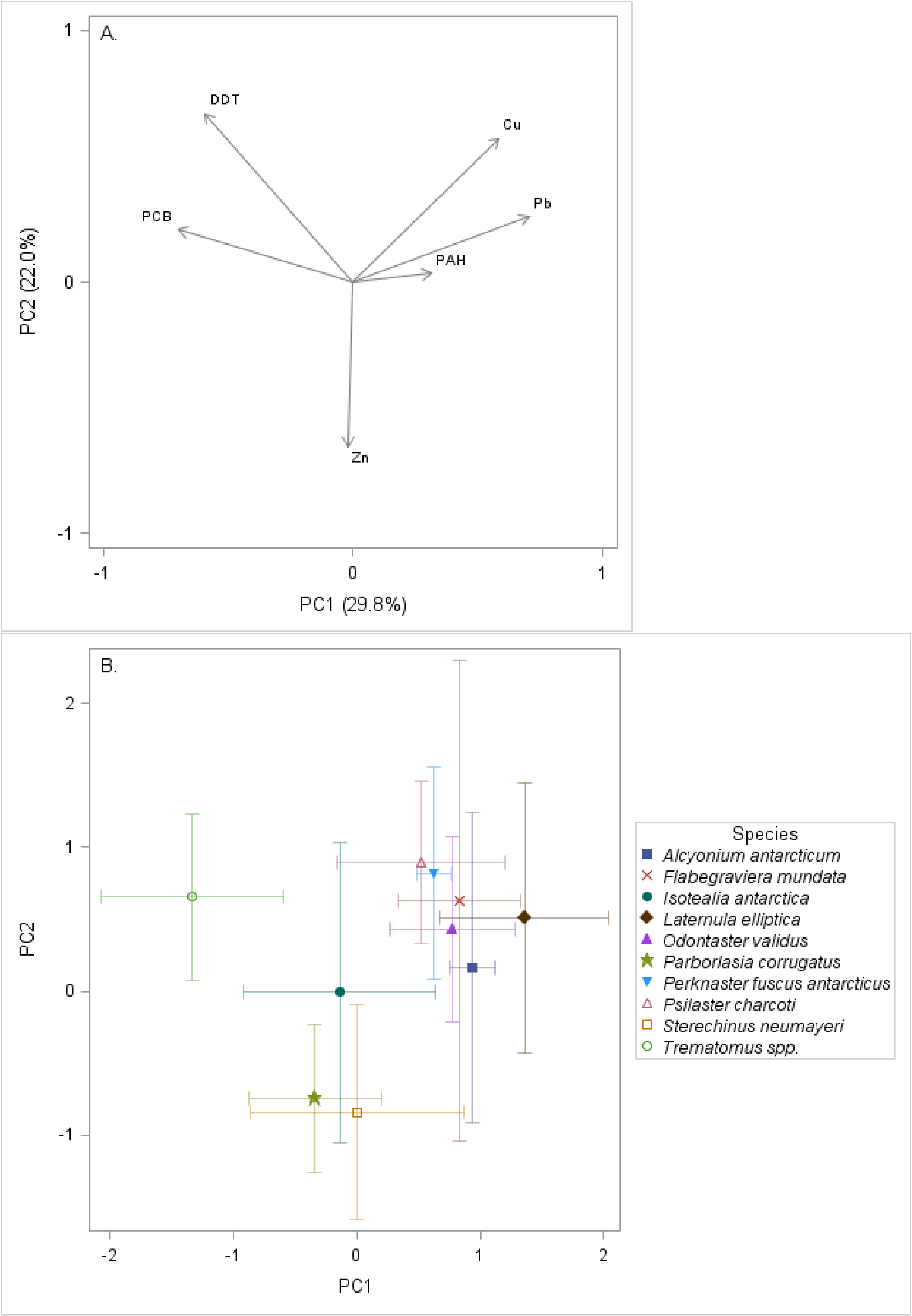

Figure 2. Chemical loads (A) and species scores (B) from Principal components analysis of bioaccumulating contaminants in species occurring at Winter Quarters Bay and the Outfall. Symbols and error bars represent the mean and standard deviation of species scores.

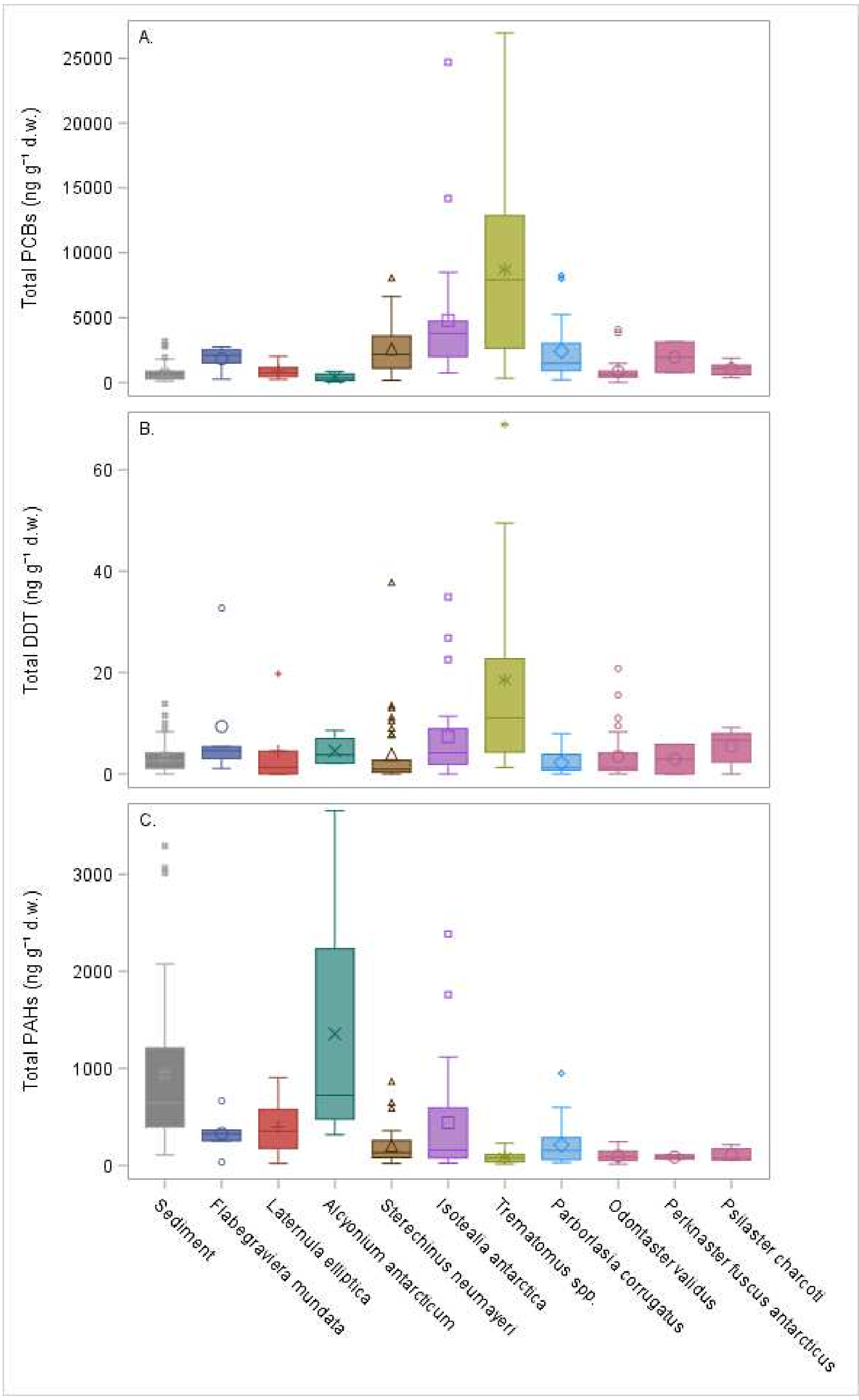

Figure 3. Total PCBs, DDT and PAHs concentrations in sediment and fauna in Winter Quarters Bays and the Outfall. One sediment PAH concentration from WQB (5314 ng g^{-1}) is above the scale of the plot.

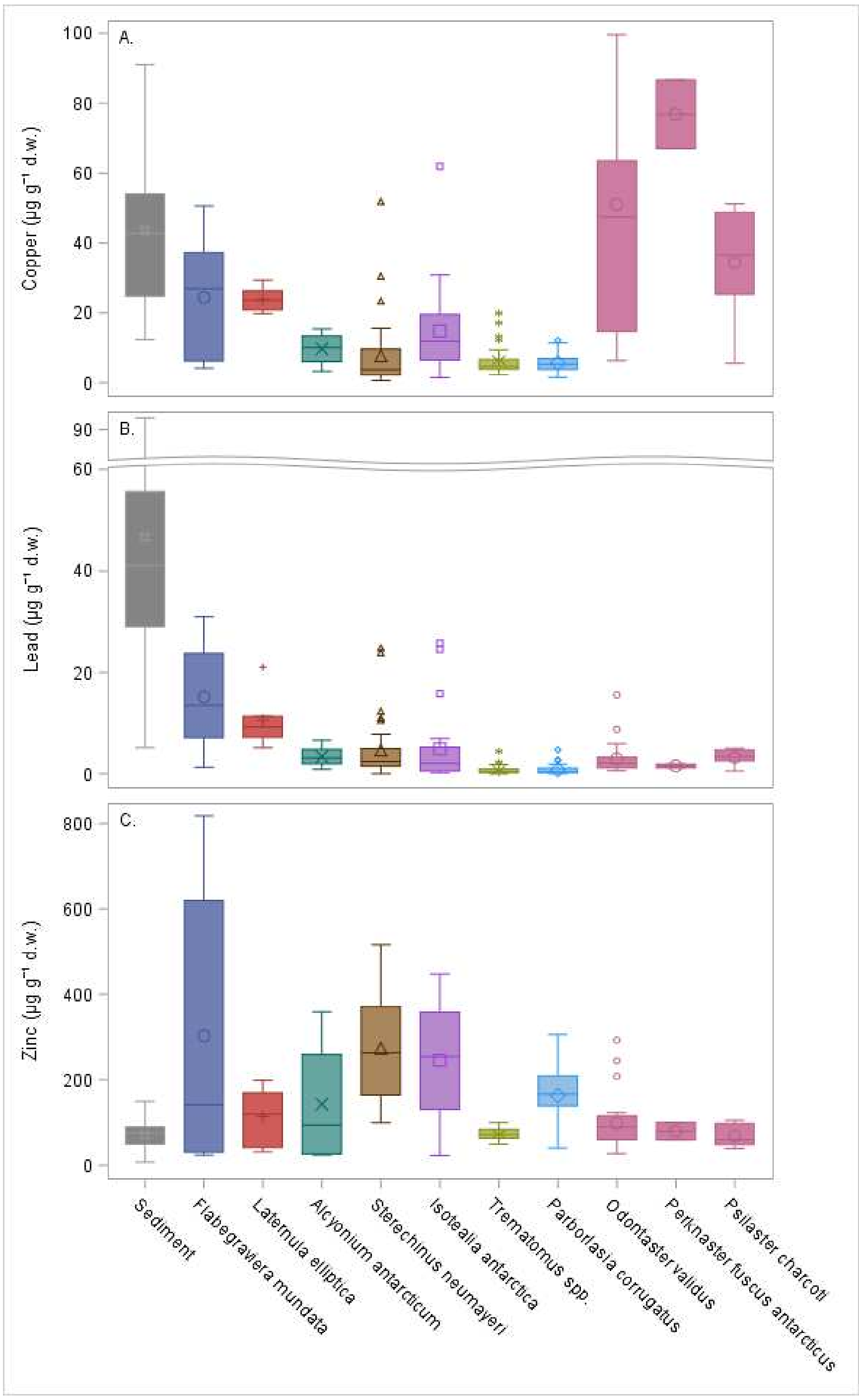

Figure 4. Copper, zinc and lead concentrations in sediment and fauna in Winter Quarters Bays and the Outfall. Three outliers of sediment zinc concentrations (99, 165, 182 µg g^{-1}) are above the scale of the plot of lead concentrations.