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Long-term changes in contamination and macrobenthic communities adjacent to McMurdo Station, Antarctica

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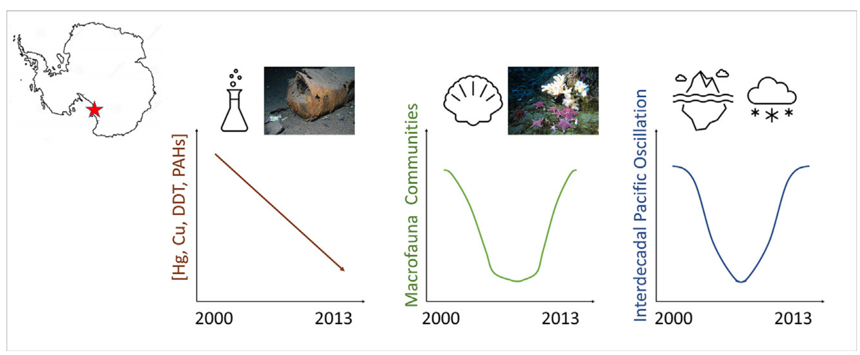
HIGHLIGHTS

- Contamination and benthic communities were sampled at McMurdo Station, Antarctica.
- Some contaminants decreased in concentration over time in polluted marine sediments.
- Macrobenthic communities did not show any signs of recovery.
- Changes in macrobenthic communities had higher correlations with climate cycles.

GRAPHICAL ABSTRACT



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ABSTRACT

Improved waste management at McMurdo Station, Antarctica beginning in the 1980s has been followed by decreases in polycyclic aromatic hydrocarbon (PAH) and metal contamination in the adjacent marine sediments. However, determining the effect of the decreased contamination on marine ecological indicators (macrobenthic fauna) is confounded by concurrent changes in climate cycles and other physical forces. Between 2000 and 2013, there was a decrease in concentrations of some contaminants including mercury, copper, organochlorines, and PAHs in marine sediments adjacent to McMurdo Station. PAH concentrations in Winter Quarters Bay decreased an order of magnitude from 2000/2003 to 2012/2013 and were within an order of magnitude of reference area concentrations by 2013. Macrobenthic communities did not indicate any sign of recovery and have not become more similar to reference communities over this same period of time. Temporal changes in macrobenthic community composition during the study period had higher correlations with climatic and sea ice dynamics than with changes in contaminant concentrations. The Interdecadal Pacific Oscillation climatic index had the highest correlation with macrobenthic community composition. The Antarctic Oscillation climatic index, maximum ice extent and other natural environmental factors also appear to influence macrobenthic community composition. Despite large improvements in environmental management at McMurdo Station, continuing environmental vigilance is necessary before any noticeable improvement in ecological systems is likely to occur. The effects of climate must be considered when determining temporal changes in anthropogenic effects in Antarctica. Maintaining long-term monitoring of both contaminants and ecological indicators is important for determining the localized and global influences of humans on Antarctica, which will have implications for the whole planet.

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1. Introduction

The recognition and mitigation of human influences in Antarctica, including forecasts of human activities and their impacts, has been identified as one of the six top priorities for Antarctic science (Kennicutt et al., 2014). Forecasting of human activities is essential for effective Antarctic governance and regulation. However, before scientists predict the future, they must first understand the present and the past. While the continent of Antarctica has 12 territorial claims, the Antarctic Treaty restricts the continent for nonmilitary scientific pursuits. Thus, there are few, if any, permanent residents of Antarctica and outside of tourism, most individuals residing in Antarctica are there to support science. An ideal location for studying the influence of humans on the Antarctic marine environment is adjacent to McMurdo Station, the most populated (summer population of >1000, Morehead et al., 2008), which is likely one of the most contaminated places on the continent (Aronson et al., 2011). Compared to less populated areas of the Antarctic, any temporal changes of the contaminant effects on the environment should theoretically be more obvious because of the higher concentration of contaminants and greater degree of ecological disturbance at McMurdo Station. In addition, the implications for environmental management decisions will likely be greater because of the higher population density in McMurdo Station.

The sea floor adjacent to McMurdo Station contains hydrocarbon concentrations similar to that of polluted harbors in temperate latitudes (Lenihan, 1992), as well as high concentrations of polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DDT) and its metabolites (DDE and DDD), polybrominated diphenyl ethers (PBDEs), pharmaceuticals and personal care products (PPCPs), and organic carbon and metals (Lenihan et al., 1990; Hale et al., 2008; Kennicutt et al., 2010). Cold temperatures and relatively stable hydrology (Dayton, 1990) mean that the breakdown of contaminants is generally slower in Antarctica than in warmer climes (Matsumura, 1989). Because of this slow breakdown rate, legacy contamination persists for long periods despite dramatic improvements in waste disposal practices at many Antarctica research stations. The Antarctic Treaty's Protocol on Environmental Protection, currently accepted by 53 countries (www.ats.aq), recognizes that “managing the impact of humans in Antarctica requires that disturbances be quantified through long-term, sustained observations, i.e., monitoring” (Kennicutt et al., 2010). Unfortunately, monitoring of contamination alone does not provide information on whether these contaminants have adverse effects on the biota, i.e. whether these contaminants are pollutants (Chapman, 2007).

Marine macrobenthic communities are recognized indicators of environmental change, and therefore are ideal for environmental monitoring, because many infauna species are sedentary, long-lived (relative to plankton), widespread, and sensitive to changes in water and sediment qualities. Changes in macrobenthic communities in Antarctica have been related to both natural environmental and local anthropogenic disturbances. Natural environmental disturbances of the benthos include variations in sea ice cover (Norkko et al., 2007; Lohrer et al., 2013), ice scour disturbance (Lenihan and Oliver, 1995; Gutt et al., 1996), anchor ice formation (Dayton et al., 1969, 1970; Lenihan and Oliver, 1995) and climate change (Aronson et al., 2011). It is important to recognize that global climate change has an indirect anthropogenic influence on some of these natural disturbances, and this influence will become increasingly important in the future (Aronson et al., 2011). The two most common local anthropogenic disturbances directly affecting the benthos in Antarctica include sewage disposal and other chemical contamination (e.g., metals and petrochemicals; Dayton and Robilliard, 1971, Conlan et al., 2004, 2006, Morehead et al., 2008, Tin et al., 2008, Kim et al., 2010, Stark et al., 2014). Intense, historic contamination of marine sediments adjacent to some Antarctic research stations has also caused historic changes to marine macrobenthic communities (e.g., Lenihan and Oliver, 1995; Stark et al., 2014). However, long-term changes in contaminants and their effects (>10 years) are necessary to understand and distinguish between natural environmental and local anthropogenic drivers of benthic community change.

The goal of this current study is to assess long-term (>10 year) changes in contaminants and macrobenthic communities in the nearshore marine sediments at McMurdo Station. Contamination was evident prior to the initiation of this study, which led to increased environmental awareness and more careful waste management beginning in 1987 (Chiang et al., 1997). Improved waste management procedures included implementing a waste management program, cleanup of dump sites, and the development of spill plans. It is hypothesized that contaminants have decreased in concentration over the study period because of improved waste management processes. It is also hypothesized that any decrease in contamination has allowed macrobenthic communities in the nearshore marine sediments at McMurdo Station to become more similar to unaffected (reference) communities. Variations in natural environmental factors previously shown to influence macrobenthos (e.g., ice dynamics, water temperatures, climate cycles) are examined to highlight effects that are unrelated to localized contamination.

2. Methods

2.1. Study design

Marine sediments for this study were sampled annually during Austral summers from 2000 to 2013 (except for 2001 and 2002). Sediment samples were taken along three or four fixed transects each sampling year at water depths of 12, 24, and 36 m (Fig. 1). Two transects in locally disturbed areas were sampled every sampling year. One disturbed transect is in Winter Quarters Bay (WQB, Transect A), a historic dumping area and the location of the floating ice pier that is used to transfer materials on and off the Station during its annual resupply in January/February. The second disturbance transect is adjacent to a sewage outfall (Outfall, Transect D), which discharged raw sewage into McMurdo Sound until 2003, when a secondary wastewater treatment plant was completed (Egger, 2003). Within the vicinity of the sewage outfall is also a historic dumping area but with a lesser impact than WQB (Lenihan and Oliver, 1995; Morehead et al., 2008). Either one or two reference transects were sampled every year at Cape Armitage (Reference, Transects E and F). Transect F (sampled 2000 to 2008) was discontinued in 2009 after it was determined that the natural substrate at Transect E (sampled 2000, 2008–2013) resembled the areas in the vicinity of transects A and D before anthropogenic activities started better than they resembled transect F (Rob Robbins, US Antarctic Program, 2008, *pers. comm.*). The area surrounding transect F has a dense, siliceous spicule mat that lies on top of the sediment proper (especially at depths >~20 m), whereas the spicule mat is less pronounced at transect E. The physical marine environment prior to human occupation in WQB may have differed from the surrounding areas because of its different geomorphology (Lenihan et al., 1990; Crockett and White, 1997, 2003). For example, lower current speeds within WQB may have allowed the accumulation of finer sediments within the bay. However, there is little evidence that these natural spatial differences have had any important effects on community structure within WQB, and any natural differences are speculative, despite their likelihood (Lenihan and Oliver, 1995; Crockett and White, 1997, 2003). Note that the transect letters A, D, E and F correspond with W, O, J and C in the study that initiated the sampling design for this current study (Morehead et al., 2008).

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. Divers accessed the water through holes drilled in sea ice 2–6 m thick. Triplicate sediment samples were taken

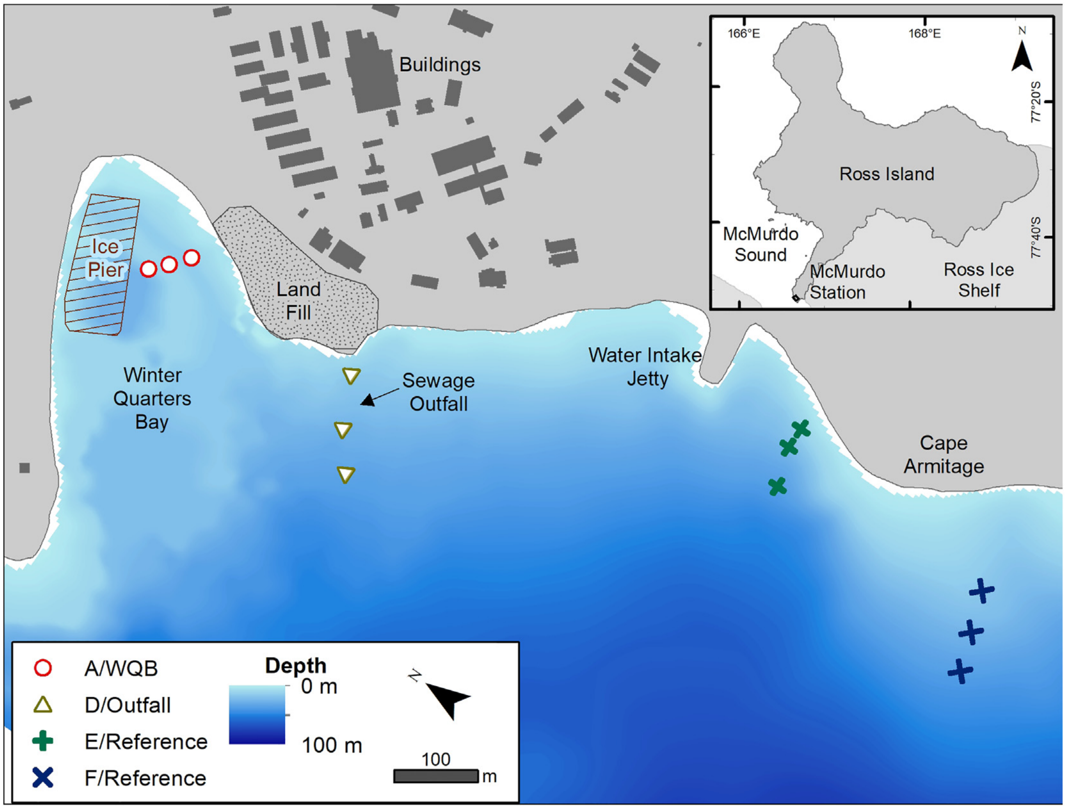

Fig. 1. Sediment sampling transects and stations. Disturbed transects = open symbols. Depths of sampling stations within each transect increase with distance away from land (12, 24, 36 m). WQB = Winter Quarters Bay. Area of landfill approximated from Lenihan et al., 1990.

for chemistry, grain size, and macrobenthic communities each year at each sampling station.

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, trace metals, and carbon concentration 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 and Klein et al., 2012). Sediment grain size samples were stored at 4 °C until analysis. Grain size was determined using the methods of Folk (1980) using a combination of sieve and pipette analysis.

Macrobenthic samples were split into 0–3 and 3–10 cm vertical sections, fixed with 5% buffered formalin and sent to The University of Texas Marine Science Institute (2000–2005) or Texas A&M University-Corpus Christi (2006–2015). The macrobenthos were then washed and extracted on a 0.5 mm sieve, identified to the lowest practical taxonomic level (usually species or genus), enumerated (for abundance), and weighed (for biomass) following the methods of Morehead et al. (2008). Biomasses were measured exclusively as wet weights from 2000 to 2003, exclusively as dry weights from 2006 to 2013 (except when reference taxa were kept), and both wet and dry weights from 2004 to 2005. Dry weight biomasses were converted to wet weight biomasses using ratios developed from times when wet and dry weight biomasses were both measured (Table S1).

2.3. Climate, sea ice and water temperature data

Annual means of Antarctic Oscillation (AAO, also known as Southern Annular Mode, Mo, 2000) and Interdecadal Pacific Oscillation (IPO, Parker et al., 2007) climatic indices were calculated from monthly data. AAO data were sourced from NOAA Climate Prediction Center (NOAA CPC, 2020). IPO data were obtained from Foland (2017). AAO is characterized by fluctuations in the strength of the circumpolar vortex and has been linked to regional temperature changes in the Antarctic (Thompson and Solomon, 2002). IPO is characterized by fluctuations in sea surface temperature anomaly patterns across the Pacific Ocean and has been linked to changes in sea ice cover, especially in the Ross Sea (Meehl et al., 2016). Annual mean, minimum and maximum sea ice area, sea ice area anomaly, sea ice extent, and sea ice extent anomaly values for the Ross Sea were calculated on monthly data sourced from Stroeve and Meier (2018). The sea ice area and extent anomalies are calculated from monthly deviations from the long-term (1978–2018) monthly mean. Sea ice thickness and temperature adjacent to McMurdo Station (Kim et al., 2017a), and date and distance of minimum sea ice edge from McMurdo Station (Kim et al., 2017b) are described in Kim et al. (2018). The sea ice thickness data is indicative of large-scale changes in sea ice thickness and not of spatial variation within the study area. The annual minimum, median, mean, maximum and the upper quartile of daily water temperatures 25 to 40 m deep on the sea floor adjacent to McMurdo Station were obtained from Cziko et al. (2014a) and described in more detail in Cziko et al. (2014b). These climate, sea ice and water temperature variables are hereinafter termed "natural environmental" variables.

2.4. Statistical analysis

Sediment chemical concentrations and macrobenthic communities were compared among disturbed and reference transects over time using univariate and multivariate statistics. Sediment chemistry characteristics of each station over time were compared using Principal Components Analysis (PCA) on $\log_e(x + 1)$ transformed data. An initial PCA was performed on the trace metals to reduce this set of variables (Fig. S1). The purpose of PCA is to discover the underlying structure in a data set by reducing the number of variables to a smaller set of orthogonal variables. If one class of variables has too many auto-correlated variables, then that class can drive the PCA, which is not desirable when the goal is to obtain orthogonal axes. For example, 13 heavy metals were measured during the current study, but many are auto-correlated, so it is desirable to reduce these to a few variables that are representative of the natural background and the potentially toxicity-inducing pollutants. In the PCA for metals, PC 1 is most highly loaded by Ba, Pb, and Mg. Although with lower loadings, Cu, Hg, Cd are known to be highly toxic to marine life. Thus, these six metals are chosen to represent the potential for sediment toxicity. PC2 is most highly loaded by Al and Mn, and these appear to represent natural background characteristics. Monotonic temporal trends of individual contaminants and the first two principal components (PCs) from PCA were determined using Spearman-rank correlations.

Non-metric Multi-Dimensional Scaling (nMDS) was used to characterize macrobenthic community composition at the species level among stations over time. NMDS was also used among transects at the family level to reduce noise and better discriminate between disturbed and reference communities (Warwick, 1988a, 1988b). Determining the effects of large disturbances on macrobenthic communities using family-level taxonomic resolution has been used successfully in other studies in Antarctica (Thompson et al., 2003; Conlan et al., 2004). For this study, the two *Nototanais* species (*N. dimorphus* and *N. antarcticus*) that occur adjacent to McMurdo Station (Rhodes et al., 2015) were treated as the numerically dominant species *N. dimorphus* because the two species were identified as the same species for the first several years of this project.

Monotonic temporal trends of macrobenthic abundance, biomass, Shannon diversity (H′), Hill's N1 diversity and a benthic index of biotic integrity (B-IBI) developed for McMurdo Sound (Morehead et al., 2008) were determined using Spearman-rank correlations with year. The B-IBI was deemed to be a better indicator of contamination adjacent to McMurdo Station than other univariate metrics such as diversity, abundance, and biomass in a previous study (Morehead et al., 2008).

The relationships among contaminants and macrobenthic communities were compared by correlating the first two PCs from PCA with macrobenthic abundance, biomass, and N1 diversity. Bio-Env analysis was used to match the best combination of sediment chemical components with spatiotemporal community assemblage data (Clarke and Ainsworth, 1993). Bio-Env analysis was also used to correlate community assemblage data with natural environmental variables (climatic indices, sea ice variables, water temperature). The number of days between sediment sampling and minimum ice distance date were used when comparing with macrofauna community variables to account for differences in sampling date. Bio-Env and nMDS analyses were conducted using PRIMER 7 software (Clarke et al., 2014). Community composition data were root-transformed prior to multivariate analyses. Sediment chemistry and natural environmental variable data were log-transformed prior to Bio-Env analysis. All univariate analyses, PCA, and data management were completed using SAS 9.4 software (SAS Institute Inc., 2013).

3. Results

3.1. Sediment chemistry

Spatio-temporal patterns of sediment chemistry were summarized using Principal Components Analysis (PCA), where the first two Principal Components (PC1 and PC2) accounted for 48.3 and 15.4% of variation within the contaminant dataset (total = 63.7%, Fig. 2). Station-year combinations with high PC1 scores had high loadings of copper, barium, lead, total dichloro-diphenyl-trichloroethane (DDT), polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), total petroleum hydrocarbons (TPHs), and several metals (Fig. 2A). The stations with high PC1 scores, and the associated high concentrations of the selected contaminants, are all within the contaminated transects (A and D), (Fig. 2B). Chemicals (loadings) with high

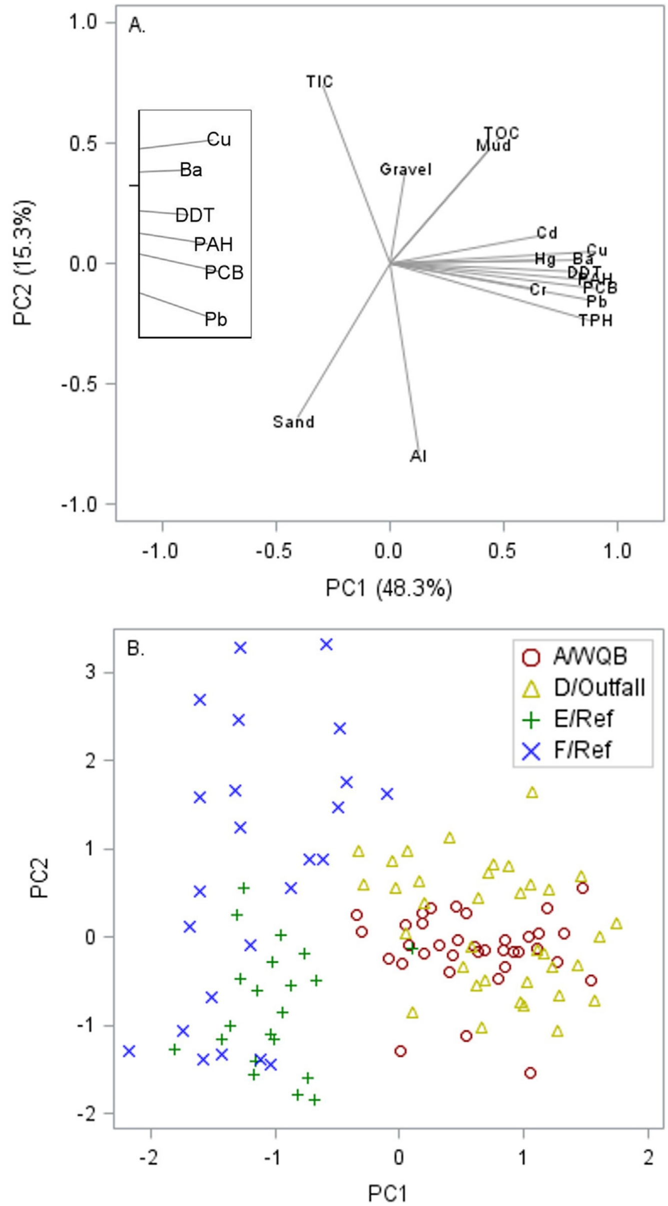

Fig. 2. Principal Components Analysis of sediment chemistry for each station from 2000 to 2013. a. Chemical loads, b. station scores. Abbreviations for transects are as in Fig. 1.

PC1 scores (Ba, Cd, Cr, Cu, Hg, Mg, Pb, DDT, TPH, PAH, and PCB) are good indicators of contamination because their concentrations are higher at the disturbed than the reference transects. In addition, DDT and PCB have no natural sources. Spatio-temporal differences in sediment chemistry within transects were highlighted along PC2. Station-year combinations with a high PC2 score had high concentrations of total organic and inorganic carbon (TIC and TOC), mud (silt + clay) and gravel, and low concentrations of sand and Al.

Spearman-rank correlations were used to compare each sediment chemistry and grain size variable, and the first two PCs, with year for each station to determine any increase or decrease in concentrations over time (Tables 1, S2). Correlations involving the stations from transects A and D had 12 non-consecutive years of data over a 13-year time span (2000, 2003−2013), whereas those involving stations from E and F only had 7 and 8 years (2000, 2008–2013, and 2000, 2003–2008) of data respectively. Overall contamination, as indicated by PC1, decreased at all stations (12, 24, and 36 m) in WQB, and the deep station (36 m) in the outfall transect (all r ≤ −0.65, p ≤ 0.011;

Table 1
Spearman-rank correlations between PCs, indicator chemicals, and year at each station. The mean is the mean over the study period and the slope is the change in concentration per year from linear regression. The number of years of data used in transects A, D, E, and F were 12, 12, 7, and 8. Abbreviations: r = correlation, p = probability. Relationships where $p < 0.05$ are bolded and where $p < 0.10$ are italicized.

	Transect	A (Winter Quarters Bay)			D (Outfall)			E (Reference 1)			F (Reference 2)		
Variable	Statistic	A1	A2	A3	D1	D2	D3	E1	E2	E3	F1	F2	F3
	r	**−0.70**	**−0.81**	**−0.66**	−0.41	0.25	**−0.88**	−0.57	−0.36	−0.32	−0.55	−0.31	−0.43
	p	**0.011**	**0.001**	**0.020**	0.191	0.430	**0.000**	0.180	0.432	0.482	0.160	0.456	0.289
	Mean	**0.76**	**0.84**	**0.20**	1.16	0.76	**0.37**	−1.21	−1.00	−0.84	−1.38	−1.03	−1.05
PC1	Slope	**−0.07**	**−0.10**	**−0.06**	−0.05	0.03	**−0.12**	−0.02	−0.03	−0.09	−0.08	−0.07	−0.11
	r	−0.38	−0.44	−0.05	*−0.57*	−0.01	−0.48	0.36	0.29	−0.39	−0.31	−0.05	−0.10
	p	0.217	0.152	0.880	*0.055*	0.983	0.112	0.432	0.535	0.383	0.456	0.911	0.823
Ba	Mean	116.94	119.42	115.91	*100.30*	83.85	86.17	42.18	49.97	50.05	22.12	36.88	46.62
(ppm)	Slope	−1.75	−1.73	−0.73	*−1.85*	−0.18	−1.66	0.51	0.17	−1.14	−0.82	0.03	−0.80
	r	−0.38	**−0.77**	−0.16	−0.17	0.14	−0.38	*0.75*	*0.68*	0.11	0.10	−0.40	−0.05
	p	0.226	**0.003**	0.618	0.602	0.665	0.217	*0.052*	*0.094*	0.818	0.823	0.320	0.911
Cd	Mean	0.59	**0.72**	0.39	0.94	0.83	0.36	*0.18*	*0.16*	0.18	0.30	0.39	0.43
(ppm)	Slope	−0.02	**−0.06**	−0.01	0.00	0.02	−0.02	*0.02*	*0.02*	0.00	0.02	0.00	0.00
	r	*−0.56*	*−0.51*	**−0.73**	**−0.66**	−0.04	**−0.72**	−0.04	−0.46	−0.50	−0.19	−0.05	−0.38
	p	*0.059*	*0.090*	**0.007**	**0.020**	0.897	**0.008**	0.939	0.294	0.253	0.651	0.911	0.352
Cu	Mean	*39.74*	*58.60*	**29.71**	**55.68**	47.51	**30.73**	9.77	11.66	13.73	9.10	14.05	16.17
(ppm)	Slope	*−2.38*	*−1.12*	**−2.50**	**−3.83**	−0.46	**−2.08**	−0.21	−0.47	−0.99	−0.19	0.06	−0.86
	r	**−0.80**	**−0.60**	−0.50	*−0.55*	−0.46	**−0.80**	−0.41	−0.45	**−0.80**	−0.55	−0.55	−0.30
	p	**0.002**	**0.039**	0.101	*0.067*	0.131	**0.002**	0.364	0.307	**0.030**	0.160	0.157	0.471
DDT	Mean	**2.93**	**3.75**	1.81	*5.21*	2.62	**2.65**	0.05	0.07	**0.37**	0.10	0.18	0.23
(ng g^{-1})	Slope	**−0.35**	**−0.36**	−0.33	*−0.29*	−0.16	**−0.58**	−0.01	−0.02	**−0.20**	−0.03	−0.06	−0.12
	r	*−0.53*	**−0.76**	**−0.64**	−0.33	0.09	**−0.86**	−0.61	−0.34	−0.14	*−0.68*	**−0.71**	**−0.71**
	p	*0.075*	**0.005**	**0.024**	0.297	0.779	**0.000**	0.148	0.452	0.760	*0.062*	**0.047**	**0.047**
Hg	Mean	*0.05*	**0.07**	**0.02**	0.19	0.15	**0.09**	0.01	0.01	0.02	*0.01*	**0.02**	**0.03**
(ppm)	Slope	*0.00*	**−0.01**	**0.00**	−0.01	0.00	**−0.01**	0.00	0.00	0.00	*0.00*	**0.00**	**0.00**
	r	0.01	0.15	0.30	−0.21	−0.20	−0.08	0.39	0.36	−0.50	0.50	−0.05	−0.38
	p	0.966	0.649	0.342	0.513	0.542	0.812	0.383	0.432	0.253	0.207	0.911	0.352
Mg	Mean	19,871	19,833	19,818	19,305	19,626	16,939	14,422	15,249	13,353	10,547	11,722	11,677
(ppm)	Slope	−10.69	186.24	143.19	−162.66	−176.20	−186.99	343.28	36.37	−264.15	706.53	−227.99	−277.80
	r	**−0.76**	**−0.64**	−0.28	−0.11	0.16	**−0.59**	−0.36	−0.25	−0.39	−0.62	−0.62	−0.55
	p	**0.005**	**0.026**	0.379	0.729	0.618	**0.042**	0.432	0.589	0.383	0.102	0.102	0.160
PAH	Mean	**894.41**	**1884.32**	638.37	1186.44	517.25	**644.42**	17.55	50.69	54.07	26.32	36.81	19.42
(ng g^{-1})	Slope	**−98.61**	**−224.84**	−48.94	−21.11	1.64	**−55.78**	−0.83	0.26	−11.25	−3.82	−5.61	−1.89
	r	**−0.78**	−0.36	−0.38	0.24	0.36	−0.45	0.04	0.54	0.21	−0.60	−0.40	**−0.71**
	p	**0.003**	0.245	0.217	0.457	0.245	0.138	0.939	0.215	0.645	0.120	0.320	**0.047**
PCB	Mean	**426**	630	263	1561	709	657	9	27	35	9	11	**11**
(ng g^{-1})	Slope	**−42.72**	−34.14	12.12	80.84	50.54	−79.44	−0.02	0.36	−8.86	−1.93	−3.05	**−6.31**
	r	*−0.51*	*−0.53*	−0.40	−0.06	*0.55*	−0.57	*−0.68*	**−0.79**	*−0.75*	−0.14	−0.05	−0.24
	p	*0.090*	*0.075*	0.199	0.846	*0.067*	0.051	*0.094*	**0.036**	*0.052*	0.736	0.911	0.570
Pb	Mean	*40.24*	*48.54*	18.08	66.92	*48.37*	39.06	*6.31*	**8.31**	*9.02*	9.39	5.86	5.17
(ppm)	Slope	*−1.14*	*−2.33*	−1.11	−0.90	*2.73*	−2.03	*−0.11*	**−0.24**	*−0.67*	−0.40	0.11	−1.25
	r	*−0.53*	**−0.78**	−0.36	−0.32	−0.06	−0.48	*−0.68*	−0.21	−0.21	−0.52	−0.12	−0.05
	p	*0.075*	**0.003**	0.255	0.308	0.863	0.118	*0.094*	0.645	0.645	0.183	0.779	0.911
TPH	Mean	*298*	**305**	113	317	168	55	*11*	17	16	5	3	3
(ng g^{-1})	Slope	*−17.39*	**−28.27**	−11.43	−14.60	−1.31	−8.91	*−1.17*	−1.25	−6.19	−1.49	−0.81	−1.47
	r	*−0.56*	**−0.59**	−0.46	−0.45	−0.46	**−0.60**	−0.04	−0.14	−0.36	−0.24	0.07	−0.60
	p	*0.059*	**0.045**	0.131	0.138	0.131	**0.039**	0.939	0.760	0.432	0.570	0.867	0.120
V	Mean	*53.75*	**52.75**	49.56	53.64	50.30	**45.88**	29.75	34.89	32.91	24.20	26.32	22.96
(ppm)	Slope	*−2.48*	**−2.51**	−1.90	−2.50	−1.67	**−2.05**	−0.48	−0.95	−1.33	−0.83	0.37	−1.95

Fig. 3). Most temporal trends of individual contaminants, including all where $p < 0.10$, were negative, meaning most concentrations of chemicals decreased over time, while the rest had no monotonic (overall increasing or decreasing) trend. The stations with the highest number of decreasing indicator chemicals were A1 and A2 (WQB; 4 and 7 variables with $p < 0.05$), and D3 (Outfall; 6 variables). The contaminants that decreased at the highest number of stations were mercury (five stations), total DDT (four stations), copper and PAHs (three stations). The highest rates of decrease were 22% (-0.58 ng g^{-1} yr^{-1}) for DDT at the deep sewage outfall, 55% (-0.20 ng g^{-1} yr^{-1}) for total DDT at the deep reference E, and 55% (-6.3 ng g^{-1} yr^{-1}) for PCBs at the deep reference F. In these instances where the percentage rate decrease is >20%, concentrations of the particular contaminant in the first or second sampling year (2000 or 2003) were at least three times that of concentrations in any other year. Temporal changes in chemical concentrations are plotted in Figs. S2 to S7.

3.2. Infaunal communities

Each transect had distinctly different mean community composition from each other (Figs. 4 and S8). There is a progressive shift in community composition moving from WQB to the Outfall transect to the reference area (left to right in nMDS plots). This shift is consistent with overall increases in the number of species and families occurring in each transect from WQB (111 species, 62 families) to the Outfall (177 species, 88 families) to the reference area (206 species, 103 families), as well as changes in species composition (Tables 2, 3, S5). WQB and the Outfall transects had higher abundances of the polychaetes *Aphelochaeta* sp. and *Leitoscoloplos kergulensis*, but lower abundances of the taniad *Nototanais dimorphus*, podocopid ostracods, the anemone *Edwardsia meridionalis*, the cumacean *Eudorella splendida*, and the polychaetes *Galathowenia scotiae, Spiophanes tcherniai* and *S. bombyx* than the reference transects. The Outfall had higher abundances of polychaetes *Ophryotrocha claparedei* and *Capitella perarmata* and the amphipod *Heterophoxus videns* than the other transects. At the family taxonomic level, there was an increase in Nototanaidae (Tanaidacea), Paramunnidae (Isopoda), Spionidae (Polychaeta), Oligochaeta, and several other groups moving from WQB to the reference area but decreases in Cirratulidae (Polychaeta) and Orbiniidae (Polychaeta). Mean macrobenthic abundance also increased from WQB (13,622 n m^{-2}) to the Outfall (24,294 n m^{-2}) to the reference area (40,688 n m^{-2}).

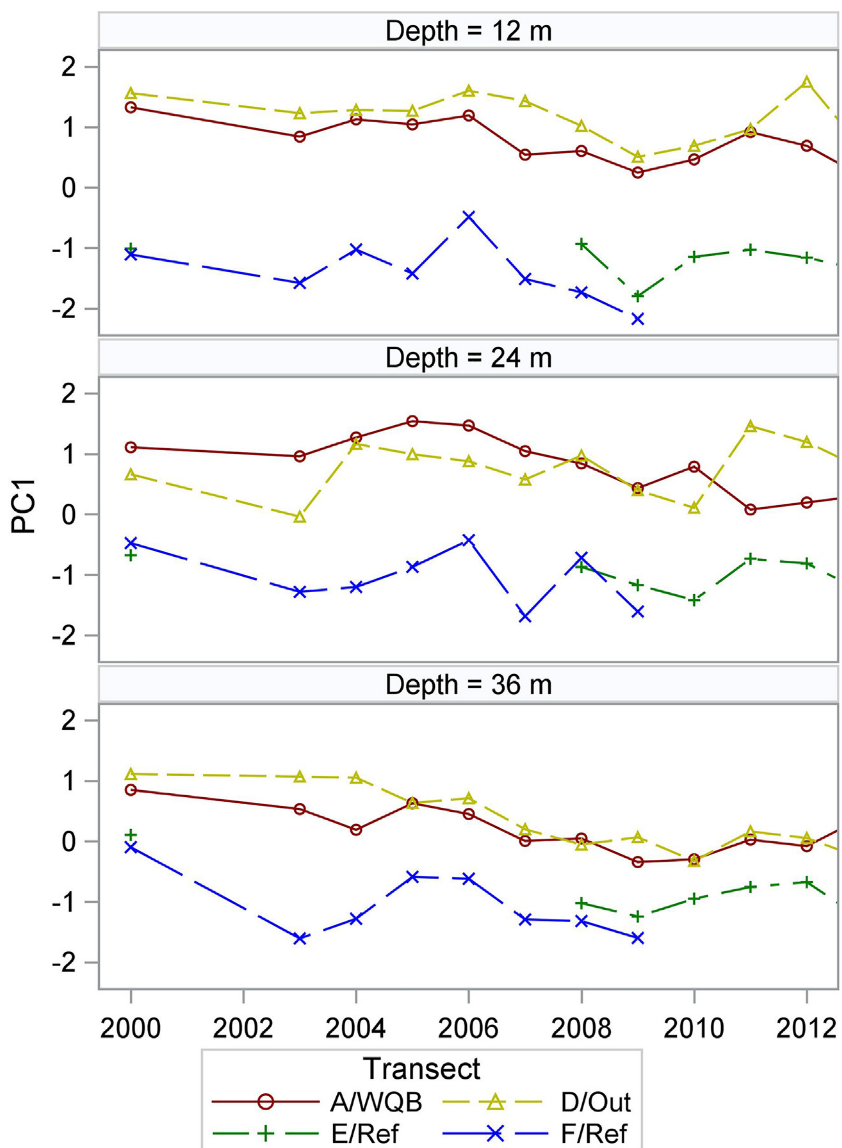

Fig. 3. Time series of chemical contamination as represented by PC1 at each station (transect and depth). WQB = Winter Quarters Bay, Out = outfall, Ref = reference.

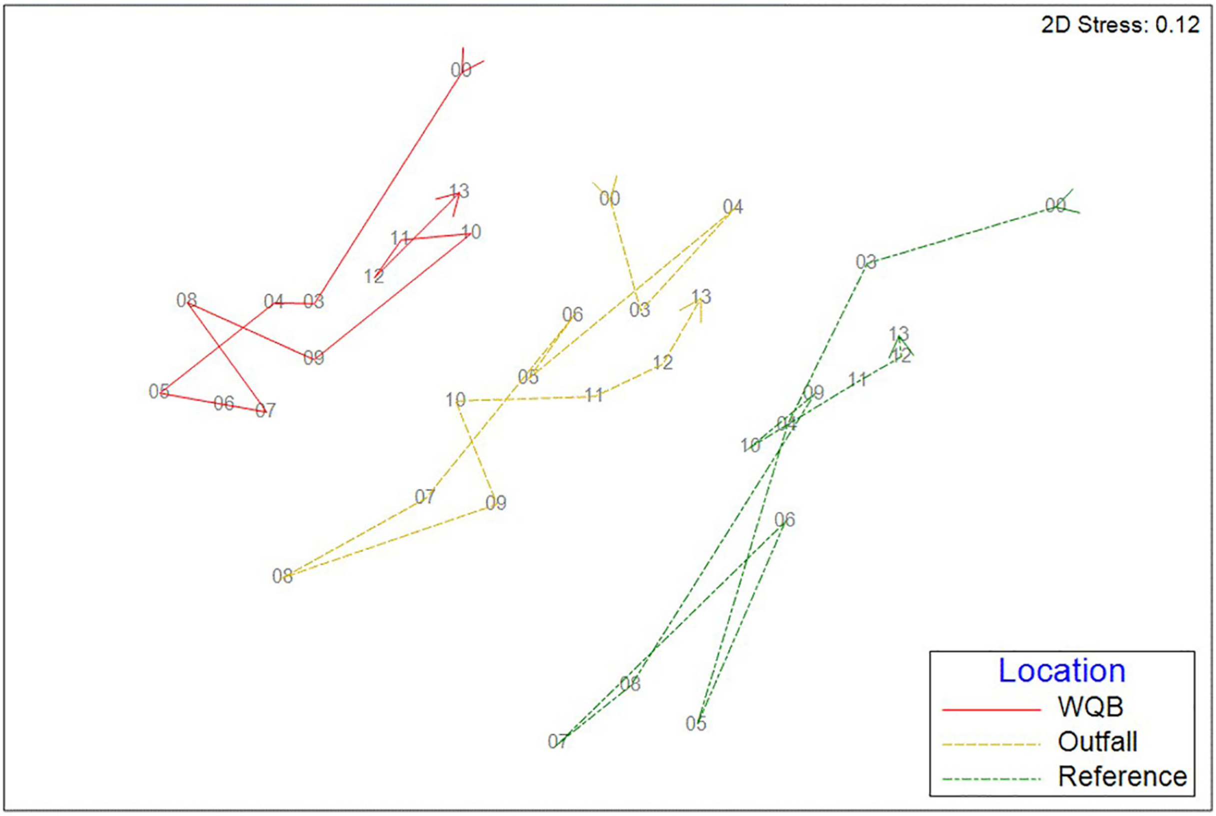

Fig. 4. Nonmetric multidimensional scaling plot of infauna communities change at the Winter Quarters Bay, sewage outfall, and reference transects. The taxonomic resolution of the communities represented in this plot was at the family level.

Macrobenthic community composition at the family level had a Spearman correlation of 0.96 (ρ) with community composition at the species level (Fig. S9). Family-level macrobenthic communities at each transect followed a roughly cyclical pattern of seriation (Fig. 4). Although the pattern of seriation varied slightly among transects, communities became increasingly dissimilar to those occurring in 2000 until around 2007 at each transect, and then became increasingly similar again. The inter-annual similarity in community composition within

Table 2
List of most abundant (top 80%) macrobenthic species occurring at each transect. Abundances are in n m^{-2}. Cum. = cumulative. A = WQB, D = Outfall, EF = Reference transects combined.

Species	Family	Class	A	D	EF	Mean	Mean (%)	Cum. %
Nototanais dimorphus	Nototanaidae	Malacostraca	499	2794	9276	4190	16.0	16.0
Aphelochaeta sp.	Cirratulidae	Polychaeta	4328	1922	75	2108	8.0	24.0
Oligochaeta (unidentified)	Oligochaeta	Oligochaeta	467	1754	2251	1491	5.7	29.7
Austrosignum glaciale	Paramunnidae	Malacostraca	528	1644	2178	1450	5.5	35.3
Philomedes sp.	Philomedidae	Ostracoda	68	1153	3029	1417	5.4	40.7
Spiophanes tcherniai	Spionidae	Polychaeta	29	578	3314	1307	5.0	45.7
Ophryotrocha claparedei	Dorvilleidae	Polychaeta	827	2539	30	1132	4.3	50.0
Capitella perarmata	Capitellidae	Polychaeta	580	2306	30	972	3.7	53.7
Podocopida (unidentified)	Podocopida	Ostracoda	97	205	2256	853	3.3	56.9
Heterophoxus videns	Phoxocephalidae	Malacostraca	562	1287	546	798	3.0	60.0
Edwardsia meridionalis	Edwardsiidae	Anthozoa	0	24	2020	681	2.6	62.6
Hesionidae (unidentified)	Hesionidae	Polychaeta	777	664	479	640	2.4	65.0
Halacaridae (unidentified)	Halacaridae	Arachnida	938	144	754	612	2.3	67.4
Leitoscoloplos kerguelensis	Orbiniidae	Polychaeta	1132	604	98	611	2.3	69.7
Austronanus glacialis	Paramunnidae	Malacostraca	441	407	796	548	2.1	71.8
Gastropoda (unidentified)	Gastropoda	Gastropoda	189	239	1096	508	1.9	73.7
Galathowenia scotiae	Oweniidae	Polychaeta	0	155	1216	457	1.7	75.5
Actiniaria (unidentified)	Actiniaria	Anthozoa	102	609	545	419	1.6	77.1
Spiophanes bombyx	Spionidae	Polychaeta	18	131	982	377	1.4	78.5
Eudorella splendida	Leuconidae	Malacostraca	13	11	902	309	1.2	79.7
Nemertea (unidentified)	Nemertea	Nemertea	215	328	349	298	1.1	80.8
Total abund.: top 80%			11,813	19,499	32,223	21,178	80.8	
Total abund.: 100%			13,622	24,294	40,688	26,202	100.0	
S: top 80%			19	21	21	21		
S: 100%			111	177	206	246		

Table 3
List of most abundant (top 80%) macrobenthic families occurring at each transect. Abundances are in n m^{-2}. Cum. = cumulative. A = WQB, D = Outfall, EF = Reference transects combined.

Family	Class	# sp.	A	D	EF	Mean	Mean (%)	Cum. %
Nototanaidae	Malacostraca	1	499	2794	9276	4190	16.0	16.0
Paramunnidae	Malacostraca	5	990	2195	3813	2333	8.9	24.9
Cirratulidae	Polychaeta	8	4346	1957	116	2139	8.2	33.1
Spionidae	Polychaeta	11	87	1247	4518	1951	7.4	40.5
Oligochaeta	Oligochaeta	1	467	1754	2251	1491	5.7	46.2
Philomedidae	Ostracoda	1	68	1153	3029	1417	5.4	51.6
Dorvilleidae	Polychaeta	4	835	2802	109	1249	4.8	56.4
Capitellidae	Polychaeta	3	591	2311	56	986	3.8	60.1
Podocopida	Ostracoda	1	97	205	2256	853	3.3	63.4
Phoxocephalidae	Malacostraca	2	580	1300	561	814	3.1	66.5
Hesionidae	Polychaeta	4	801	775	681	752	2.9	69.4
Edwardsiidae	Anthozoa	1	0	24	2020	681	2.6	72.0
Orbiniidae	Polychaeta	3	1135	607	102	615	2.3	74.3
Halacaridae	Arachnida	1	938	144	754	612	2.3	76.6
Leuconidae	Malacostraca	4	13	60	1581	551	2.1	78.7
Gastropoda	Gastropoda	1	189	239	1096	508	1.9	80.7
Total abund.: top 80%		51	11,637	19,567	32,219	21,141	80.7	
Total abund.: 100%		246	13,622	24,294	40,688	26,202	100.0	
	S: top 80%		15	16	16	16		
	S: 100%		62	88	103	119		

transects was highest between the years 2000 and 2013 in the reference and WQB transects, and second highest between the years 2000 and 2013 in the outfall transect. Species richness and total abundance was higher in 2000 and 2013 than around 2007 at each transect (Fig. 5). The higher abundance in 2000 and 2013 than ~2007 can be attributed to higher abundances of Paramunnidae (Isopoda: particularly *Austrosignum glaciale*, *Austronanus glacialis* and *Munna* sp.), Nototanaidae (Tanaidacea: particularly *Nototanais dimorphus*) and Oligochaeta.

There were no significant temporal correlations between B-IBI rank, abundance, biomass, Hill's diversity (N1) or Shannon diversity (H′), and time within each station, except for the deep station at Reference F, which had significant decreases in B-IBI rank, abundance, and H′ diversity (Table S6). There were also no significant temporal trends in the same univariate macrobenthic metrics over time within each transect (Table S6, Fig. S10). However, B-IBI score and diversity were significantly correlated with PC1, the composite indicator of chemical contamination ($-0.47 \le r \le -0.35$, $p < 0.0001$, Table S8).

Macrobenthic community composition at the species and family levels were both most highly correlated with TPH ($r = 0.419$ and $r = 0.391$, $p = 0.0001$) of all single sediment chemistry variables (Tables S9 and S10). Community composition at the species and family levels were both most highly correlated with the combination of TPH, barium, and

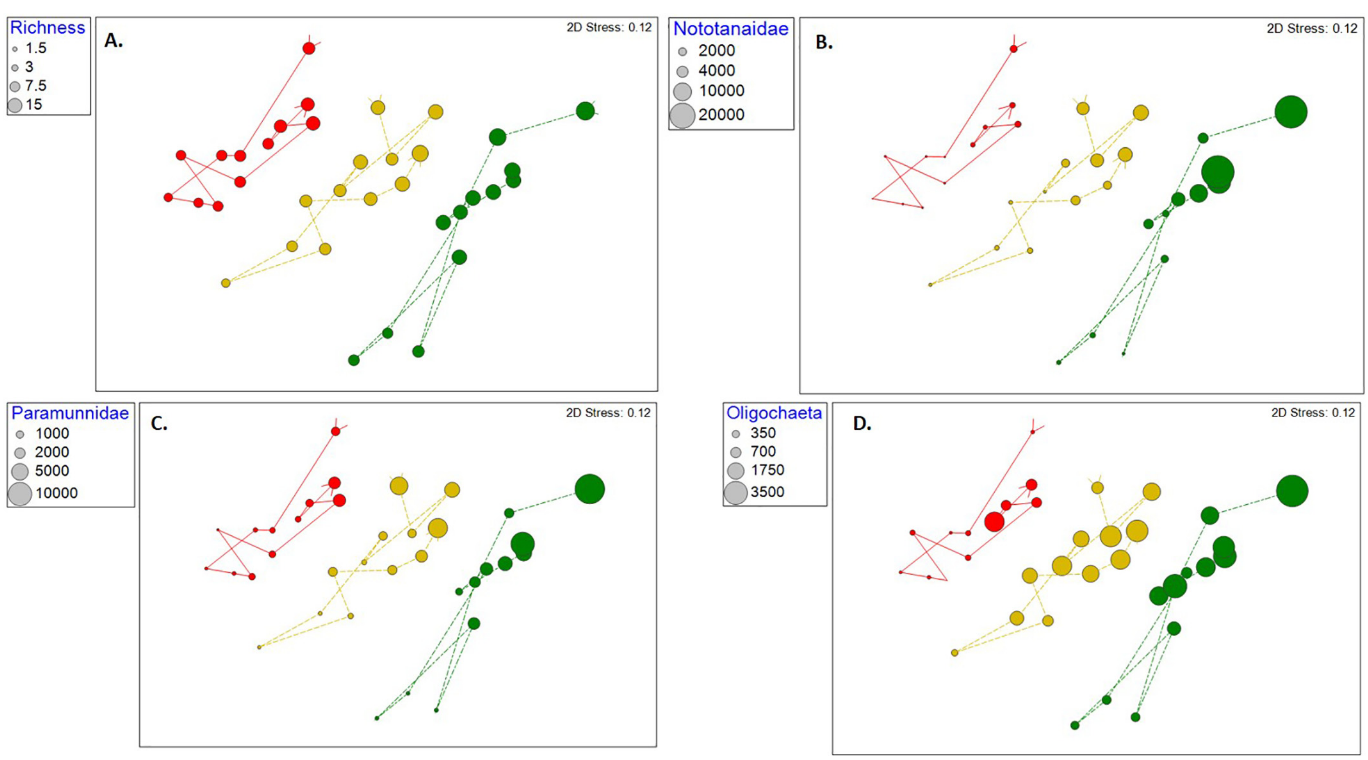

Fig. 5. nMDS plot of macrobenthic communities overlayed with proportionally sized bubbles depicting A. Species Richness, and abundances (n m^{-2}) of three higher taxa groups that had similar temporal changes in abundance within each transect: B. Nototanaidae, C. Paramunnidae, and D. Oligochaeta.

total inorganic carbon (TIC; r = 0.457 and r = 0.414, p = 0.0001) of all variable combinations (that included up to five variables).

3.3. Climate, sea ice and water temperature

Median ice thickness increased from 2.6 to 4.4 m thick from 1999 until 2004, before sharply decreasing and remained between 2.0 and 2.6 m thereafter (Fig. 6). The mean ice temperature for the sediment sampling days ranged from −20 to −7 °C with local minima occurring every three to five years. The minimum ice distance decreased between 2003 and 2013, however, there was no consistent trend in the day of minimum ice cover. The mean annual IPO decreased from −0.29 to −0.97 from 2000 to 2009 and then sharply increased to −0.28 in 2013. The AAO oscillated between positive and negative values almost every year of the study period. Ranges of annual minimum and maximum water temperatures were −1.94 to −1.91 °C and −1.48 to −0.37 °C, with intra-annual maxima occurring in January and February each year (Figs. S12 and S13).

Spearman-rank correlation tests among univariate macrobenthic and natural environmental variables were conducted within each transect. No natural environmental variable was correlated with any univariate macrobenthic variable within any transects (Table S11). The IPO was significantly correlated with the B-IBI, abundance and biomass at the Outfall transect (all r ≥ 0.68), the minimum ice edge distance was correlated with H′ diversity and abundance in WQB (all r ≤ −0.62), and ice thickness was correlated with H′ and N1 diversity at the combined reference transects (r = 0.77). The minimum ice edge distance was significantly correlated with abundance at both WQB and the reference transects (r ≤ −0.60). The single variable that was most highly correlated with community composition within each transect was the IPO (r = 0.36, p ≤ 0.007 Tables 4, S12 and S13). The highest correlation with community composition was the combination of IPO, AAO, maximum

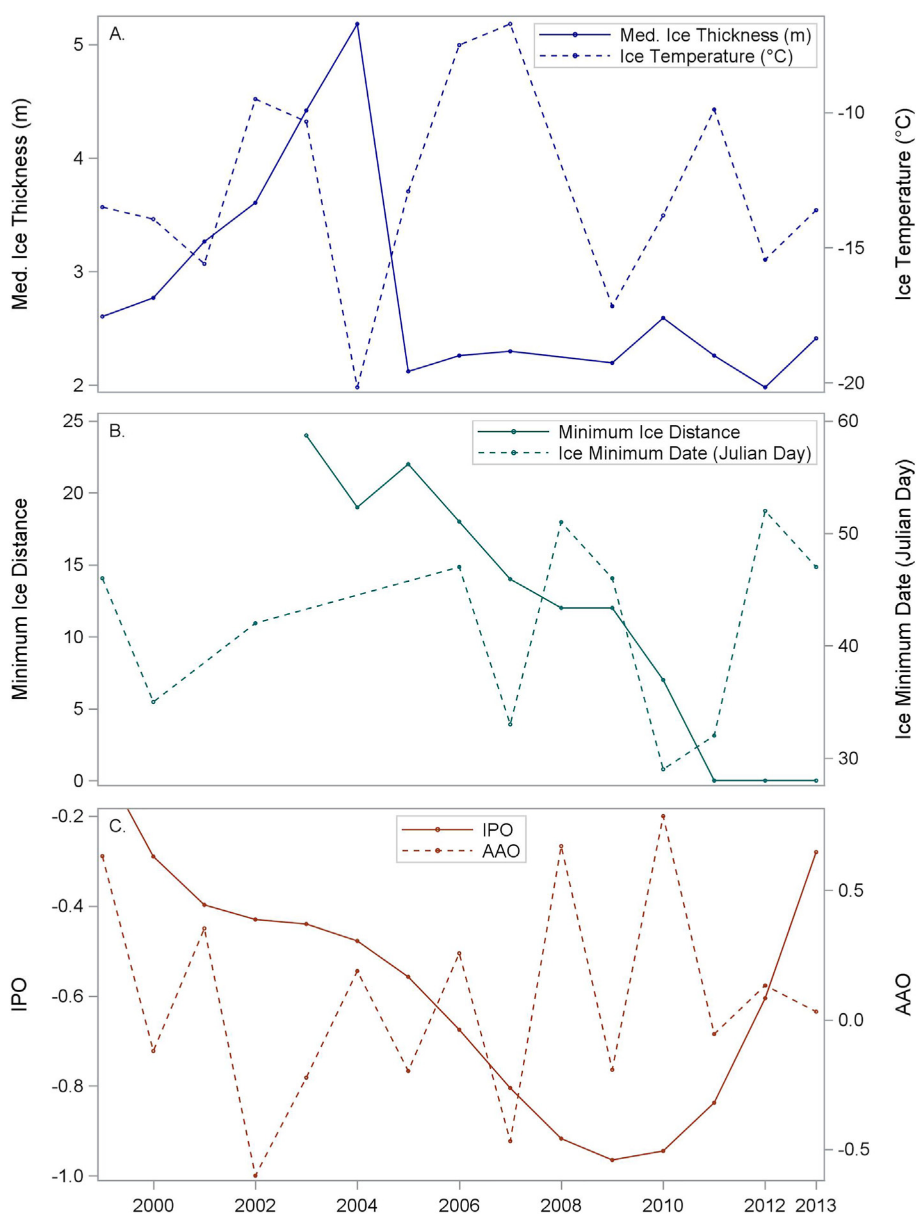

Fig. 6. Ice dynamic descriptors and climatic indices over the sampling period. A. Median Ice temperature and thickness at the sea ice runway adjacent to McMurdo Station (data from Kim et al., 2017a), B. minimum sea ice distance (MID) and Julian day of MID (data from Kim et al., 2017b), and C. Interdecadal Pacific Oscillation and Antarctic Oscillation climate indices (from NOAA CPC, 2020).

Table 4
Highest correlations with macrobenthic community composition for combinations of one to five trial variables. IPO = Interdecadal Pacific Oscillation, AAO = Antarctic Oscillation, TOC = total organic content.

No. Vars	Mean rho	p	Best variable selections
1	0.358	0.007	IPO
2	0.489	<0.001	IPO, Med. water temp.
3	0.530	<0.001	IPO, Max. ice extent, Med. water temp.
4	0.578	<0.001	IPO, AAO, Max. ice extent, Min. ice edge distance
5	0.632	<0.001	IPO, AAO, Max. ice extent, Min. ice edge distance, TOC

ice extent, the minimum ice edge distance and sediment TOC ($r = 0.632$, $p \leq 0.007$). IPO was the most common variable within the variable combinations with the highest correlations with community composition, followed by AAO, maximum annual ice extent, minimum distance of the ice edge from McMurdo Station, and TOC concentration.

4. Discussion

4.1. Sediment chemistry

Historically, most contaminants entered the marine environment adjacent to McMurdo Station at WQB and the area adjacent to the sewage outfall. The majority of contamination is thought to have been introduced by direct dumping, burning, and subsequently sinking of debris along the shoreline and sea ice of WQB prior to 1981, from historically emptied ships bilges, and residual contamination from the WQB landfill (Dayton and Robilliard, 1971; Lenihan et al., 1990; Risebrough et al., 1990; Crockett and White, 2003). These historic practices likely caused elevated concentrations of metals, hydrocarbons (including PAH), DDT, PCB and other organic contaminants to occur in WQB and the area adjacent to the outfall.

Marine sediment contamination has decreased in concentration over the 15-year study period, although these decreases varied spatially. Contamination as a whole, as represented by PC1 from the PCA of sediment chemistry (Fig. 2), decreased at all WQB stations (depths of 12, 24, and 36 m) and the deep Outfall station (36 m), but no other location during the study period. This decrease in PC1 scores indicates a reduction in contamination over time. The greatest number of contaminants decreased in concentration at the two shallowest WQB stations (12 and 24 m), and at the deep Outfall station (36 m). WQB appears visually to be more disturbed (e.g., fewer large epibenthic fauna, more scattered debris) than the area adjacent to the outfall (aside from the mound of organic matter directly below the sewage outfall; Lenihan et al., 1990, *pers obs*). The decrease in contamination in WQB and overall stability of contamination adjacent to the outfall may be attributed to greater runoff, and therefore continued sediment deposition, that likely occurs in WQB (Affleck et al., 2014a), covering the more contaminated WQB sediments with terrestrial sediment. The gradual burial of some large pieces of waste debris (barrels, machinery) over several years has been observed by United States Antarctic Program divers, including a Caterpillar D8H bulldozer has been buried ~1 m since it sank to the bottom adjacent to the ice pier in 1979 (Rob Robbins and Steve Rupp *pers comm.*). Terrestrial runoff entering WQB during melting events often contains elevated trace metal (Cd, Cr, Cu, Pb, Ni, Zn) and selected PAH concentrations (Affleck et al., 2014b). However, the potential for wide dispersal of these contaminants combined with the sediment load likely means that there is a net decrease in contaminant concentrations on the sea floor. It is possible that the Outfall transect has more continuous (recent) contamination from the sewage outfall itself and from the adjacent historic landfill (Crockett and White, 2003), which is located closer to the outfall than the WQB transect (Fig. 1). The decrease in contamination in the deep outfall station could also be related to an anecdotal increase in surficial, mostly dead, sponge spicules that has accumulated on top of the sediment there, although this has not been quantified. Overall decreases in contamination in WQB and the deep outfall station are promising for an eventual recovery of the benthic environment; however the identification of individual contaminants with the most widespread change is also important.

The contaminants whose concentrations decreased in the highest number of locations were DDT, mercury, copper, and PAHs (Table 1). DDT is a persistent organic pollutant that is usually linked to global transportation of historic pesticides in remote regions. However, some marine sediments adjacent to McMurdo Station are dominated by a more intense, local source (Risebrough et al., 1990). Total DDT (both p,p′- and o,p′-isomers) decreased during the current study period at the two most contaminated stations in WQB (12 and 24 m), and the deep outfall station (36 m). The mean total concentration of the p,p′-isomers of DDT and its metabolites (DDE and DDD) occurring at each station within WQB (1.1–2.2 ng g^{-1}) and the outfall transect (1.1–2.8 ng g^{-1}) in the current study were similar to concentrations observed in 1988 adjacent to the sewage outfall (1.3–1.8 ng g^{-1}) but lower than samples from WQB in 1988 (>4.0–8.9, Risebrough et al., 1990). To put this into an environmental disturbance perspective, mean concentrations from 2000 to 2013 are similar to the Effects Range Low (ERL, 10th percentile of effects) of 1.58 ng g^{-1}, but less than the Effects Range Median (ERM, 50th percentile of effects) of 46.1 ng g^{-1} (Long et al., 1995). Mean concentrations have decreased from exceeding the Canadian Council of Ministers of the Environments' (CCME) probable effect levels (PEL) for DDT (4.77 ng g^{-1}) in 1988 to be closer to the CCME Sediment Quality Guidelines (SQM, of 1.19 ng g^{-1}) from 2000 to 2013 (CMME, 1999).

Another group of historical contaminants of biological concern at McMurdo Station are PCBs (Risebrough et al., 1990; Lenihan, 1992). The PCB fingerprint adjacent to McMurdo Station most closely matches the commercial PCB mixture Aroclor 1260, a mixture commonly used in fluid-filled electrical transformers up until 1971 (USEPA, 1976; Kennicutt et al., 2010). Decreases in total PCB concentrations were observed in the shallow WQB station A1 and deep Cape Armitage reference station F3 (Table 1, Fig. S7). Physical, rather than biological, factors are likely to have caused the decrease in PCB contamination at the shallow WQB station because the nature (dechlorination) of PCBs is not altered as would occur from microbial degradation (Kennicutt et al., 2010). A possible physical cause of decrease could be from deposition of runoff-derived sediments. These sediments could be slowly burying the contaminated sediments.

Decreases of PCB at the deep reference station F3, and DDT at the deep reference station E3 only occur when considering high concentrations occurring in 2000. Initial (2000) PCB concentrations at F3 (72 ng g^{-1}) and DDT at E3 (2.5 ng g^{-1}) were high relative to subsequent years (1 to 6 and 0 to 0.06 ng g^{-1}), but low relative to those at the WQB and Outfall stations (means of 708 and 3.2 ng g^{-1}). These concentrations at E3 and F3 are closer to the method detection limits, which results in increased analytical uncertainty. Organochlorine compound concentrations were determined by gas chromatography with an electron capture detector (GC/ECD) in 2000, whereas subsequent years utilized gas chromatography/mass spectrometry (GC/MS), which provides greater specificity and improved compound identification. There is also the likelihood (albeit small) that spatial heterogeneity of contamination contributed to the higher concentrations in 2000 (sampling of hot spots). It is difficult to determine if the relatively higher concentrations of PCB and DDTs in 2000 are the result of spatial heterogeneity and/or analytical interferences. Regardless of the cause, the Cape Armitage stations are appropriate reference locations because these concentrations are low relative to known impacted areas, and probably have minor, if any, ecological impacts. The relatively close proximity of these reference sites to the disturbed area (500–750 m) also minimizes potential differences in substrate type (Lenihan et al., 1990, Klein unpublished data) and latitudinal variation (e.g., ice cover, algal production; Dayton, 1990, Norkko et al., 2007) that may affect macrobenthic communities further from McMurdo Station more than the low concentrations of PCB and DDT reported in 2000.

Mean PCB concentrations in WQB, the Outfall and the Cape Armitage reference transects in the current study (263–629, 656–1561, and 9–35 ng g^{-1}, Fig. S7) are within the ranges of those observed in 1988 (410–1400 and 2–16 ng g^{-1}, Risebrough et al., 1990) and from 1991 to 1993 (250–4300 and 18–28 ng g^{-1}, Kennicutt et al., 1995). The largest PCB concentrations in this study occurred along the outfall transect including samples with a concentration of 7000 ng g^{-1} at the deep outfall station in 2003 and concentrations of 4400 and 4600 ng g^{-1} at the shallow outfall station in 2012 and 2013. PCB concentrations at the sewage outfall were similar at ~20 m depth in 1988 (280 to 410 ng g^{-1}, Risebrough et al., 1990) and 2002 (220–373 ng g^{-1}, Negri et al., 2006). However, annual mean PCB concentrations from the current study were higher than those ranges at the sewage outfall transect at 12 m deep (minimum = 991 ng g^{-1}) and often above the range at 24 and 36 m deep (higher 7 and 8 out of 12 years sampled). The recent maxima at the shallow outfall station come after a general decrease in PCB concentrations from 2000 to 2009 and a general increase thereafter. This increase since 2009 indicates a recent introduction of higher concentrations of PCBs to the marine environment adjacent to the outfall. This introduction is likely due to surface run off, or leachate from the former dump site adjacent to the outfall, because the increase only occurs at the shallow station, which is shallower than another obvious potential source, the sewage outfall. Total PCB concentrations adjacent to McMurdo Station are well above the ERM of 180 ng g^{-1} (Long et al., 1995). Even though there have been decreases in total PCBs in Winter Quarters Bay, PCB contamination remain a concern in marine sediments adjacent to McMurdo Station given that Aroclor 1260 has a lower dichlorination rate then less-chlorinated Aroclor mixtures (Alder et al., 1993) and the elevated concentrations in both WQB and adjacent to the outfall.

Similar to PCBs, TPH concentrations generally decreased until 2009 and subsequently increased at both the shallow outfall and the WQB station. Although in both cases, a second decrease occurred from 2012 to 2013. TPH concentrations decreased over the entire study period at the 24-m outfall station. TPH concentrations sampled in 2002 (81–134 ppm, Negri et al., 2006) are similar to annual means of the closely located shallow (81–690 ppm) and mid-depth (63–409 ppm) outfall stations sampled in this study. TPH concentrations in this study are generally similar or lower than observed in disturbed sediments of Brown Bay, Casey Station, Antarctica in 1998 (318–698 ppm, Stark et al., 2014), but much lower than the total purgeable hydrocarbons occurring in the shallow depths of WQB (in back of the bay) in 1998 (200–4500 ppm).

Total PAHs (likely originating from oil and gasoline) decreased at the two shallowest stations within Winter Quarters Bay and the deep outfall station. Concentrations from all samples from all transects (aside from one sample of 11,686 ng $g^{-1}$from the 24 m deep in WQB in 2003) were less than the maximum PAH concentrations observed adjacent to McMurdo Station in the austral summers of 1990–1991 (6267 ng g^{-1}) and 1992–1993 (6339 ng g^{-1}), and much less than the maximum value observed in late 1993 (13,000 ng g^{-1}, Kennicutt et al., 1995). All of the historic maximum values were observed in WQB. The decreases in PAH concentrations in WQB have decreased an order of magnitude from 2000 and 2003 to 2012 and 2013 and by the end of the study were within an order of magnitude of the reference transect E. While still elevated relative to reference areas, all recent (2012−2013) concentrations of total PAHs are less than the ERL of 4400 ng g^{-1} (Long et al., 1995), meaning that the environmental disturbances of PAHs alone are not of great concern in marine sediments adjacent to McMurdo Station.

Mercury concentrations decreased at 5 of the 12 stations sampled (A2, A3, D3, F2, and F3). Larger decreases occurred at the middle WQB (A2, mean = 0.07, rate = 0.01 ppm y^{-1}) and deep outfall (D3, mean = 0.09, rate = 0.014 ppm y^{-1}, Table 1) stations. However, concentrations and decrease rates were small at the deep WQB (A3) and Reference F (F2, F3) transects (mean = 0.02–0.03 ppm, rate = −0.002 to −0.003 ppm y^{-1}). The highest concentrations occurred along the Outfall transect (station means of 0.09 to 0.19 ppm), with larger temporal fluctuations occurring at the shallow two stations (range of annual means was: 0.01 to ≥0.32 at both stations). Mean mercury concentrations along the outfall transect in this study were similar to maximum values sampled from 1991 to 1993 (0.17 ppm, Kennicutt et al., 1995), higher than those sampled at the outfall in 2002 (0.07 ppm, Negri et al., 2006), but lower than that sampled in 1998 in WQB (0 to 0.9 ppm, Lenihan et al., 1990) and the outfall (0.4 ppm). Comparing mercury concentrations in this study with the previous studies suggest decreased concentrations since the early 1990s. The mercury concentration is now less than ERL and ERM values of 0.15 and 0.71 ppm (Long et al., 1995), indicating partial recovery and minimal likely environmental disturbance.

Copper concentrations decreased at all stations within WQB and two on the outfall transect. The deepest of these stations (A3 and D3) decreased from annuals means of 52 and 43 ppm to 15 and 12 ppm (−2.5 and −2.1 ppm y^{-1}), while the shallow outfall station decreased from 91 to 24 ppm (−3.8 ppm y^{-1}). The mean concentrations for stations in Winter Quarters Bay and outfall are similar, or lower than those recorded for the same areas in 1988 and 2002 (Lenihan et al., 1990; Negri et al., 2006). However, caution must be given to attributing all copper in the sediments to human activity because all concentrations are within the “normal” range of basaltic debris in McMurdo Sound and elsewhere in the Antarctic (Kennicutt et al., 1995; Trevizani et al., 2016). Copper concentrations have either decreased or stabilized at concentrations around or below the ERL of 34 ppm, and are much lower than the ERM of 270 ppm (Long et al., 1995).

4.2. Infaunal communities

Determining that contamination is still elevated in WQB and adjacent to the Outfall relative to reference areas is important. However, it is also important to determine if ecological indicators of contamination in the Antarctic marine environment, in this case macrobenthic communities, have changed in response to any contamination changes. The hypothesis at the beginning of this study was that a potential decrease in contamination has allowed affected macrobenthic communities to become more similar to unaffected (reference) communities. To test this hypothesis, it is necessary to determine whether macrobenthic communities have changed over time, and if these changes correlate with changes in contamination. It is also important to understand potential causes of natural variation in benthic communities that may confound the interpretation of any community changes.

As with several previous studies (e.g., Dayton and Robilliard, 1971; Conlan et al., 2004; Morehead et al., 2008), macrobenthic communities in WQB and adjacent to the outfall continue to differ from reference communities. The current study identified and reaffirmed several indicator taxa at the species and family levels that are indicative of disturbed rather than reference communities over the 13-year study period (Tables 2 and 3). This study also confirmed that spatio-temporal community relationships are similarly represented by family-level and species-level taxonomic resolution, as in other Antarctic analyses (Thompson et al., 2003; Conlan et al., 2004). It has been suggested that determining anthropogenic or large-scale effects on macrobenthic communities is improved when using a higher taxonomic resolution than species-level because there is less sensitivity to small natural environmental changes, e.g. grain size and water depth that may confound spatiotemporal changes (Warwick, 1988a, 1988b).

The univariate macrobenthic variables B-IBI, N1, and H′ diversity were correlated with PC1, the indicator of overall contamination (Table S8). However, there was no evidence of change over time among these macrobenthic variables (Tables S6, S7) within any station or transect despite there being temporal changes in PC1. Spatio-temporal changes in community composition throughout the study area were most highly correlated with concentrations of the contamination indicators TPH and barium, and TIC (Tables S9 and S10). However, macrobenthic communities of the two disturbed transects did not become more similar to those of the reference areas (Fig. 4), indicating a lack of recovery in the disturbed areas. While the original pre-human communities of the disturbed areas may have not been identical to the reference areas, it is likely that they were more similar than the current disturbed communities (Lenihan et al., 1990; Crockett and White, 1997, 2003) and should have become increasingly similar to the reference communities if any ecological recovery occurred. Instead, the WQB, Outfall and reference communities all followed a similar seriation whereby communities became increasingly dissimilar to those occurring in 2000 until approximately 2007 at each transect, and then became increasingly similar to the 2000 communities. This cyclical seriation of macrobenthic communities cannot be solely explained by any sediment contaminant measured in this study because no contaminant exhibits a similar cyclical variation across the entire study area. The temporal changes can be partially explained by the combination of climate (IPO and AAO), sea ice variables (maximum ice extent, the minimum ice edge distance) and sediment TOC (Table S13), which had the highest correlation with community composition within transects. IPO was the most important natural environmental variable in this correlation.

The climatic indices have varied over time within the study period. The IPO decreased from 2000 until 2009, before sharply increasing, but remained negative. Positive mean annual IPO values only occur outside the study period (from 1977 to 1998 and 2014 to at least 2017). On a local scale, the IPO has been linked to regional changes in circulation and expansion of sea ice in the Ross Sea (Meehl et al., 2016). More specifically, negative values have been associated with the expansion of sea ice, particularly in the Ross Sea region. The AAO is the dominant index explaining climate variability in the high-latitude Southern Hemisphere and has been linked to changes in air flow, and subsequent regional heating and cooling within Antarctica (Thompson and Solomon, 2002). These two large-scale climatic indices have measurable effects on hydrodynamics throughout Antarctica, including in the Ross Sea. Although not proven in this study, marine and benthic productivity are affected by hydrodynamic changes including sea ice cover and extent, snow cover of the sea ice, and supply of nutrient-rich currents (Norkko et al., 2007; Lohrer et al., 2013; Kim et al., 2018). In this study, higher IPO values corresponded with higher total abundance and species richness of macrobenthic communities, and higher abundances of Paramunnidae (particularly *Austrosignum glaciale*, *Austronanus glacialis* and *Munna* sp.), Nototanaidae (particularly *Nototanais dimorphus*) and Oligochaeta. It is possible that further changes in macrobenthic communities might occur when the IPO index is in a positive phase and there is a further retreat in sea ice in the Ross Sea region.

Other less predictable factors affecting marine productivity are events such as the mega icebergs (B-15 and C-19) that partially blocked the mouth of McMurdo Sound from 2000 to 2005 and altered the movement of fast- and sea-ice within the Sound (Brunt et al., 2006; Conlan et al., 2010). These mega icebergs coincided with a decrease in macrobenthic abundance, species richness, and community assemblages at Cape Evans, ~25 km north of the study site (Thrush and Cummings, 2011). At Cape Evans, some of the same taxa (*Nototanais* sp. [Nototanaidae], *Austrosignum glaciale* [Paramunnidae]) that were characteristic of the community composition seriation in the current study followed similar decreases after 2002. The recovery responses of abundances of some fauna at Cape Evans until 2009 varied from remaining low (e.g., total abundance, *Nototanais* sp.) to subsequent increases (e.g., species richness). However, the Cape Evans study (2001 to 2009) may have been too short to see the full recovery of taxa as seen in the current study (2000 to 2013).The decreasing abundances of macrobenthos at Cape Evans were linked to an iceberg-related decrease in food supply/primary productivity as supported by decreases in sediment chlorophyll *a* and sediment organic matter. The presence and characteristics of sea ice (e.g., extent, thickness, opacity, snow cover) can have large effects on photosynthetically active radiation and subsequent algal production that stimulate benthic habitats (Lohrer et al., 2013), which in turn were affected by the mega icebergs (Brunt et al., 2006; Kim et al., 2018). However, earlier described climatic and local factors also play a role in changing large- and fine-scale sea ice dynamics.

Fine spatial-scale differences in sea ice characteristics within the study area were caused by the ice breaker and supply vessel that visited McMurdo Station every year throughout the study period (January or February). This vessel movement breaks up sea ice in WQB and sometimes close to the outfall, but not in the reference area. This has allowed maximum sea ice thickness occurring in November/December in WQB to be maintained at close to 2 m throughout the study period, whereas the reference and outfall areas ranged from 2 to 6 m (*pers. obs.*). Despite these differences in in situ sea ice cover among the study locations (transects), benthic communities follow the same seriation pattern throughout the study area. This implies that large scale hydrodynamic processes, as represented by IPO, AAO and sea ice extent have a greater influence on benthic communities than local-scale hydrodynamic processes.

The high correlation between the benthic community and large-scale natural processes on a temporal scale suggests the observed decreases in sediment contamination are not great enough to cause obvious changes in macrobenthic communities. Therefore, contamination represents the dominant controller of spatial variation, and natural environmental factors are the dominant forces controlling temporal variations in benthic community composition adjacent to McMurdo Station over this 13-year study period. Although large-scale indicators of natural environmental changes, largely derived from satellite data, are useful in predicting community changes, the actual mechanisms and causes of community change are difficult to understand without baseline environmental data. Aside from valuable efforts such as those by Cziko et al. (2014a) and Kim et al. (2018), there has only been a small amount of coordinated work to conduct long-term monitoring of the physical marine environment at McMurdo Station, the largest research hub in Antarctica. Continuing long-term studies, such as this current study, are important in determining how long the marine environment will take to recover from intense historic contamination, as well as how this recovery is influenced by long-term climatic cycles and other factors. A coordinated effort to monitor basic physical variables, such as water temperature, salinity, ice thickness, and chlorophyll concentration may help us to understand the impacts of natural environmental variables on marine communities. However, the lack of recovery of benthic communities despite some decreases in contaminant concentrations informs us that contamination has a larger influence on benthic communities adjacent to McMurdo Station than natural environmental factors.

5. Conclusions

Several contaminants including DDT, PAHs, mercury, copper and PCBs have decreased in concentration in some of the more contaminated areas of the sea floor adjacent to McMurdo Station over this 13-year study period. The decrease in some contaminants has likely been occurring since the late 1980s in some locations. Although the concentrations of some contaminants have decreased below ERL values, the combination of contaminants and/or relatively greater influence of some contaminants on the benthos is causing the sediment to remain obviously polluted and unsuitable for native communities. The higher correlation of benthic community composition with large-scale natural environmental processes may be more important than changes in sediment chemistry on a temporal scale. Therefore, it can be speculated that contamination is the dominant controller of spatial variation, and natural environmental factors are the dominant forces controlling temporal variations in benthic community composition adjacent to McMurdo Station over this 13-year study period.

Disposal and contamination practices have greatly improved at McMurdo Station since the 1980s. These improved practices, combined with possible gradual dilution by sediments deposited from runoff events, have resulted in the decrease of some contaminants in surficial sediments. However, the effects of current practices are difficult to determine given the remaining legacy contamination. Vigilance in environmental practices should be continued to facilitate the fastest possible recovery of the marine environment at McMurdo Station. It is also important to maintain some form of long-term monitoring of the marine environment at all Antarctic stations at an appropriate frequency to ensure any future impacts of humans can be managed, as is required by the Antarctic Treaty. The frequency of monitoring required to detect environmental changes in Antarctica could be increased from every one to every two years because of the high cost of monitoring, slow changes of contaminant concentrations, and slow rates of ecological response.

CRediT authorship contribution statement

Terence A. Palmer: Methodology, Software, Formal analysis, Investigation, Data curation, Writing - original draft, Visualization, Supervision, Project administration. **Andrew G. Klein:** Conceptualization, Methodology, Investigation, Data curation, Writing - review & editing, Supervision, Project administration, Funding acquisition. **Stephen T. Sweet:** Methodology, Investigation, Writing - review & editing, Supervision, Project administration. **Paul A. Montagna:** Conceptualization, Methodology, Formal analysis, Investigation, Data curation, Writing - review & editing, Supervision, Project administration, Funding acquisition. **Larry J. Hyde:** Investigation. **Jose Sericano:** Investigation. **Terry L. Wade:** Investigation, Writing - review & editing. **Mahlon C. Kennicutt:** Conceptualization, Methodology, Investigation, Writing - review & editing, Supervision, Project administration, Funding acquisition. **Jennifer Beseres Pollack:** Investigation, Writing - review & 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.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found in the online version, at https://doi.org/10.1016/j.scitotenv.2020.142798. These data include the Google map of the most important areas described in this article.

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