Marine Pollution Bulletin 60 (2010) 419–432

Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Benthic changes at McMurdo Station, Antarctica following local sewage treatment and regional iceberg-mediated productivity decline

K.E. Conlan [a,*], S.L. Kim [b], A.R. Thurber [b,c], E. Hendrycks [a]

[a] Canadian Museum of Nature, P.O. Box 3443, Station D, Ottawa, Ontario, Canada K1P 6P4
[b] Moss Landing Marine Laboratories, 8272 Moss Landing Road, Moss Landing, CA 95039, USA
[c] Integrative Oceanography Division, Scripps Institution of Oceanography, La Jolla, CA 92093-0218, USA

ARTICLE INFO

Keywords:
Macrofauna
Recolonization
Sewage
Iceberg
McMurdo Station
Antarctica

ABSTRACT

McMurdo Station, the largest research station in Antarctica, ceased on-site garbage dumping in 1988 and initiated sewage treatment in 2003. In 2003–2004 its sea-ice regime was altered by the massive B-15A and C-19 iceberg groundings in the Ross Sea, approximately 100 km distant. Here we follow macrofaunal response to these changes relative to a baseline sampled since 1988. In the submarine garbage dump, surface contaminants levels have declined but associated macrofaunal recolonization is not yet evident. Although sewage-associated macrofauna were still abundant around the outfall nearly 2 yr after initiation of treatment, small changes downcurrent as far as 434 m from the outfall suggest some community recovery. Widespread community changes in 2003–2004, not seen in the decade previously, suggests that the benthos collectively responded to major changes in sea-ice regime and phytoplankton production caused by the iceberg groundings.

1. Introduction

In what is considered the otherwise pristine marine environment of Antarctica, research stations provide a unique opportunity to follow the rate and trajectory of recovery from anthropogenic disturbance in high latitudes. The nearshore benthos of McMurdo Station, Ross Sea, Antarctica (77°50.88′S, 166°40.10′E) has been monitored since 1988 in connection with organic enrichment and contamination by sewage, heavy metals and persistent organic compounds (e.g., Risebrough et al., 1990; Howington et al., 1992; Lenihan, 1992; Venkatesan and Mirsadeghi, 1992; McFeters et al., 1993; Smith et al., 1994; Kennicutt et al., 1995; Lenihan et al., 1995; Crockett, 1997; Edwards et al., 1998; Lisle et al., 2004). Conlan et al. (2004) reported on macrofaunal changes related to submarine garbage dumping, which ceased in 1988, and to sewage release over 1988–1998. In 2003, secondary sewage treatment was initiated (Egger, 2003). Herein we identify the magnitude of nearshore benthic recovery over 2002–2004 as a result of this improved treatment during a time period that coincided with large scale reduced primary productivity. This allowed contrast between reduced anthropogenic inputs and larger-scale perturbations in this unique environment.

An overarching influence in McMurdo Sound is the pattern of current flow which dictates whether the pelagic food supply from the Ross Sea is periodically ample or meager and analogous to the deep sea (Dayton and Oliver, 1977). The Ross Sea is the most productive region of the Southern Ocean, producing > 200 g C m^{-2} annually (Smith et al., 2006). The McMurdo Sound benthos is nutritionally dependent on primary production in the Ross Sea (Barry and Dayton, 1988) with the annual phytoplankton bloom reaching McMurdo Station by local seeding and advection under the sea-ice. During the monitoring period, two massive icebergs calved off the Ross Ice Shelf and impacted the sources and magnitude of primary production in what is otherwise the seasonally eutrophic side of McMurdo Sound. In 2002–2003, the offshore oceanographic conditions were modified by the C-19 iceberg (Arrigo and van Dijken, 2003). A similar disruption occurred in 2000–2001 following calving of the B-15 iceberg and part of this iceberg, B-15A, was grounded approximately 100 km northeast of McMurdo Station during the entire 2002–2004 monitoring period. The effect of these icebergs was a 40–70% reduction in phytoplankton production in 2000–2001 and 2002–2003 in the Ross Sea to the north of McMurdo Station which resulted in changes in zooplankton composition and penguin diets (Arrigo et al., 2002; Arrigo and van Dijken, 2003, 2004; Seibel and Dierssen, 2003). At McMurdo Station, the annual summer sea-ice break-up did not reach the monitoring sites over 2002–2004, resulting in a massive thickening of the sea-ice (up to 7 m in 2004) with associated light reduction. This phenomenon had not been seen during the previous 10 yr monitoring period in austral spring 1988–1998, during which the sea-ice over the monitoring sites broke out most summers,

* Corresponding author. Fax: +1 613 364 4027.
E-mail address: kconlan@mus-nature.ca (K.E. Conlan).

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doi:10.1016/j.marpolbul.2009.10.009

providing greater light penetration for primary production and a maximum 2 m sea-ice thickness at the time of sampling. The result was that the benthos sampled during austral spring 2003–2004 had massively thicker sea-ice, reduced light penetration and reduced access to the previous summer's primary production (either local or advected) than that sampled before.

In this paper we quantify the combined effect of sewage abatement, persistent heavy metal and hydrocarbon pollutants and a changing sea-ice regime on macrofauna in one of the few areas of the Antarctic where long-term data exist. The reference community, which occurs inshore of the dense sponge community described by Dayton et al. (1974), is dominated by the spionid polychaete *Spiophanes tcherniae*, which is intolerant of these pollutants (Conlan et al., 2004), and provides a model organism for monitoring recolonization. Conversely, reductions in the abundance and distribution of the sewage-associated polychaetes *Capitella perarmata* and *Ophryotrocha notialis* will identify the extent of sewage treatment effects. We use these assemblages to answer the questions:

1. Has increased sewage treatment facilitated recovery of a high-latitude infaunal system?
2. To what extent have broader, nonpoint-source environmental changes, such as may have been effected by the B-15 and C-19 icebergs, influenced long-term faunal trends in the Antarctic?

2. Materials and methods

Sampling followed the protocol in Lenihan et al. (1990) at three sites in Winter Quarters Bay, seven sites along the station-front at increasing distance downcurrent (Barry and Dayton, 1988; McFeters et al., 1993; but see Thurber, 2007) of the sewage outfall and at two reference sites 13,250 m upcurrent and 1000 m downcurrent of the outfall (Fig. 1). All samples were collected at 18 m depth by scuba in austral spring 1988–2004. Six replicate cores ($0.0075\ m^2$, 9.8 cm dia) per site were taken for macrofauna (sieved live $\geqslant 0.5$ mm) and two for chemical and grain size analysis. The outfall monitoring site was 62 m from sewage release in 1988–1991 but in early 1992 the outfall was extended offshore to 18 m depth. Samples taken in 1992 and following were about 2 m from the outfall, just outside the mat of the bacterium *Beggiatoa* sp. that spread immediately below the outfall mouth. All other time-series replicates were within 4 m of each other and haphazardly placed. Three sites along the station-front were new in 2002–2004 to increase resolution between 166 and 434 m downcurrent of the outfall.

Variables measured were distance from the outfall, the surface sediment content of 23 different heavy metals (Al was excluded due to levels often exceeding upper detection limits), total detectable polychlorinated biphenyls and Aroclor (PCBs) (2003 only) and polynuclear aromatic hydrocarbons (PAHs) (2003 and 2004), %C and N and 7 sediment grain size parameters (2002 and 2003) (% gravel, sand and silt and mean, median, s.d., skewness, and kurtosis). Percentages were square root transformed while heavy metals, PCB and PAH values were log transformed and then all variables were normalized (x-mean sd^{-1}). Patterns of site similarities based on these variables (calculated by Euclidean distance) were correlated against site similarities generated by species composition (calculated by Spearman rank correlation after square root transformation).

Sediment metals were analyzed by ICPMS using EPA Method 6020. Sediment PAHs and PCBs were analyzed by GCMS using EPA method 8270. Analyses were done by CRG Marine Laboratories, Torrance, CA. Sediment grain size distribution was measured by sieving and Coulter Counter at Moss Landing Marine Laboratories (MLML). %C and %N were determined at MLML using the methods of Froelich (1980), Franson (1981), Hedges and Stern (1983) and Verardo et al. (1990,1992).

Macrofauna $\geqslant 0.5$ mm with the exclusion of large meiofauna (nematodes and harpacticoid copepods) were identified to 93 taxa: 37% to species, 31% to genus, 11% to family and 21% to higher levels. Primary feeding designations were determined from Jepps (1926), Fauchald and Jumars (1979), Oliver and Slattery (1985), Ponder (1985), Gambi et al. (1997) and from personal observation.

For multivariate analysis, the Bray–Curtis similarity index was applied to square root transformed abundances following Conlan et al. (2004) using the BEST, SIMPER and one-way ANOSIM procedures in PRIMER 6.1.5 (Clarke and Green, 1988; Clarke, 1993; Clarke and Gorley, 2006). Canonical analysis of principal coordinates (CAP), a constrained ordination procedure, was applied with vector overlay of Spearman rank correlations of individual species to visualize species changes over time (PERMANOVA+, Anderson and Willis, 2003). Univariate analyses were not applied because these measures underestimate recovery relative to multivariate measures (Warwick and Clarke, 1991; Hewitt et al., 2005; Kröger et al., 2006). We followed convention and defined our alpha as 0.05 with variation given as 1 s.e.

Recolonization was determined by comparing macrofaunal composition within each 6-replicate group (each site, each year) to the macrofaunal composition of a baseline using ANOSIM. There were three baselines: Reference (all 36 samples collected over 1988–2004 at the reference site 13,250 m upcurrent of the outfall and the 36 from the reference site 1000 m downcurrent of the outfall); Outfall (all 24 outfall samples collected at the time of outfall operation (1992–2002); and Jetty (all 48 samples collected over 1988–2004 at the station's water intake jetty, which is 434 m downcurrent of the outfall). The jetty is added for comparison because Conlan et al. (2004) found that the community here was uniquely different from other station-front and reference sites. Therefore it is possible that the jetty community is more representative of the pre-impact community at the outfall than is the reference baseline because it is situated closer to the outfall than either reference. ANOSIM R was calculated as the average of all rank dissimilarities between the test and baseline groups minus that within each of the test and baseline groups and scaled between -1 and 1. $R = 1$ if all replicates within the group were more similar to each other than any replicates from different groups; $R \approx 0$ if similarities between and within groups were the same on average (Clarke and Green, 1988). Significance of the difference was determined by comparison of the observed value to 999 re-calculations of ANOSIM R after random label changes of the samples.

3. Results

3.1. Effluent remediation

Secondary sewage treatment began in January 2003. Fig. 2 shows the annual fluctuation in sewage effluent flow volume and the effectiveness of secondary treatment in 2003 and 2004. Due to a large increase in summer population (from about 200 winter to 1000 summer residents), inflow fluctuated from about 1.7–$6.2 \times 10^9\ l\ mo^{-1}$. Despite this large variation, the new sewage treatment system consistently reduced biological oxygen demand (BOD) by 99.1 ± 0.1%, suspended solids by 95.5 ± 0.7% and ammonia levels by 93.1 ± 5.6%. Although there are no comparable data for pre-treatment effluent levels, it is likely that the influent levels reflect that which occurred prior to the initiation of secondary treatment in early 2003.