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Department of **Water**

Benthic macroinvertebrate survey in the Swan Estuary at Claisebrook

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Cover photograph: Aerial view of the Swan Estuary, Perth, Western Australia, showing Claisebrook Cove top centre, D. Tracey, 2002.

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Context of this report

This report presents one of a series of investigations conducted within the Swan-Canning river system, Perth, Western Australia. All reports pertaining to the Swan River near Claisebrook are listed below:

1. *A baseline study of contaminants in the sediments of the Swan and Canning estuaries*, Water Science Technical Series, report no. 6, Department of Water, Western Australia, Nice HE 2009.
2. *Ecotoxicological and bioaccumulation investigations of the Swan Estuary in the vicinity of Claisebrook*, Water Science Technical Series, report no. 28, Department of Water, Western Australia, Nice HE & Fisher SJ 2011.
3. *Benthic macroinvertebrate survey in the Swan Estuary at Claisebrook*, Water Science Technical Series, report no. 63, Department of Water, Western Australia, Nice HE 2013. **[This report]**
4. *Ecotoxicological investigation of the Groundwater Interception Drain outfall at Claisebrook in the Swan Estuary*, Water Science Technical Series, report no. 64, Department of Water, Western Australia, Nice HE 2013.
5. *Investigation of polychlorinated biphenyls and other contaminants in the waters of the Swan Canning Estuary using passive sampler technology*. Report for the Swan River Trust, Fisher SJ in prep.
6. *The current status at Claisebrook in the Swan Estuary, Western Australia – a synthesis of environmental information,* Water Science Technical Series, report no. 65, Department of Water, Western Australia, Nice HE in prep.

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Summary

This study, 'A Benthic Macroinvertebrate Survey in the Swan Estuary at Claisebrook' was conducted to determine whether biotic assemblages adjacent to the Groundwater Interception Drain (GID) outfall at Claisebrook were different to those at other sites in the upper Swan Estuary, and whether differences in biotic assemblages could be explained by the presence of contaminants. A secondary component of this study was to compare the current dataset (2011) with data from a similar study in 1997, to determine whether biotic assemblages in a remediated area have become more similar to reference sites over time.

Recent studies have identified a range of organic and metal contaminants at concentrations exceeding environmental guidelines in the area of the Swan Estuary adjacent to Claisebrook Cove and Mardalup Park (Nice 2009; Nice & Fisher 2011). Sediments were toxic to a range of aquatic organisms representative of those found in the Swan Estuary and some contaminants were shown to have bioaccumulated in aquatic biota (Nice & Fisher 2011). Two drains discharging to the Claisebrook area of the Swan Estuary (Claisebrook Drain and Claisebrook Diversion Drain) appeared to be current sources of contaminants; and the spatial distribution of contaminants indicated that an additional source(s) was likely (Nice & Fisher 2011). A third drain in the area, the GID, has an outfall that discharges to the estuary at the northern boundary of the historic contaminated site, Mardalup Park[1]. The GID was constructed as part of the remediation process associated with the site in the 1990s and has recently been shown to be discharging contaminants including polycyclic aromatic hydrocarbons (PAHs) directly to the Swan Estuary (ENV 2009).

In response to these findings, the Swan River Trust commissioned a comprehensive investigation of the Swan Estuary at Claisebrook, focusing primarily on the GID outfall to the estuary. This investigation comprised three components: i) a benthic macroinvertebrate survey (with supporting sediment chemistry), ii) a sediment toxicity assessment (with supporting sediment chemistry) and iii) a water chemistry assessment using passive sampling technology. These were designed to assist the Swan River Trust in the development of management options for the Swan Estuary at Claisebrook.

This report presents the first component (the benthic macroinvertebrate survey), in which sediment samples were collected from seven sites within the Swan Estuary at Claisebrook (including a site within Claisebrook Cove) and the benthic macroinvertebrate fauna identified and quantified. Sediment chemistry was also assessed at each site, targeting contaminant groups previously demonstrated to be present at levels of concern in the Claisebrook area (Nice 2009): PAHs, organochlorine (OC) pesticides and metals. Polychlorinated biphenyls (PCBs) were also assessed in this study as this is a contaminant group often associated with historic contaminated sites such as those in this area of the Swan Estuary.

[1] The GID also has an outfall that discharges to Claisebrook Cove. A previous study (Nice & Fisher 2011) showed that sediments collected near the GID outfall to Claisebrook Cove were toxic to fish larvae.

Constituents from each of the contaminant groups assessed have been shown to be toxic to aquatic biota (e.g. ANZECC & ARMCANZ 2000).

In summary, this study found that:

- Differences in biotic assemblages existed between certain sites and 16 key contaminants were identified as contributing to the separation of sites (in conjunction with organic carbon concentration and sediment composition).
- Multivariate analyses showed no clear distinction in biotic assemblage at the GID site (CBI03) compared with two sites upstream (CBI01 and CBI02). Simple community measures such as abundance and richness also showed no difference at the GID site (CBI03) when compared with other sites in the estuary. Additionally, sediment contaminant concentrations were generally relatively low at this site – despite the sediments having similar binding capacity to those at sites with higher contaminant concentrations. As such, based on the data presented here, there was no measurable impact on benthic macroinvertebrate communities that could be attributed to the GID.
- Of all the sites examined in this study, site CBI07 adjacent to Claisebrook Main Drain outfall (within Claisebrook Cove) was most different from the others in terms of biotic assemblage, and also generally had among the highest concentrations of contaminants and the largest proportion of silt. Concentrations of several contaminants at this site were above guidelines.
- Of the six sites in the estuary, site CBI06 adjacent to Point Fraser wetland and Heirisson Island was most different from the others in terms of biotic assemblage, and generally had the highest PAH concentrations and the highest concentrations of some metals. Concentrations of several contaminants were above guidelines at site CBI06. The PAH acenapthene showed the strongest correlation in terms of the separation of this site from the others, but was likely acting in conjunction with the other PAHs, pesticides, PCBs and metals measured (and proportion of organic carbon and fine sediments present).
- The PAH concentrations at site CBI06 were also higher than those recorded at site CBI07 within the cove for 12 of the 13 PAHs detected.
- PCBs (not previously detected in sediments of the Swan Estuary at the current limits of reporting to the author's knowledge) were detected at three sites including CBI06.
- Comparison of 1997 and 2011 biotic assemblages at four sites showed that remediation and reference sites were more similar to each other in 2011 than they appeared in 1997. However, factors other than general 'recovery over time' alone appear to have influenced biotic assemblage.

Based on the evidence presented in this report, recommendations have been provided for future management action. As a priority, the site adjacent to Point

Fraser and Heirisson Island (CBI06) should be investigated to assess the extent of the contaminated zone and potential toxicity in this area and to establish likely sources for the contaminants. Other recommendations are provided in Chapter 6.

1 Introduction

1.1 Background

A range of organic and metal contaminants have previously been found in concentrations of concern in the sediments of the Swan Estuary adjacent to Claisebrook Cove and Mardalup Park, Perth, Western Australia (Nice 2009). Sediments were toxic to a range of aquatic organisms representative of those found in the Swan Estuary when tested in the laboratory, and some contaminants were shown to have bioaccumulated in sessile aquatic biota (Nice & Fisher 2011). While a proportion of the contamination in this area of the estuary is likely to be historic, two drains currently discharging to the system (Claisebrook Drain and Claisebrook Diversion Drain) were shown to be ongoing sources of contaminants. Furthermore, spatial information indicated that an additional source was likely to be contributing significant contaminant loads, particularly to the area of the estuary adjacent to Mardalup Park (Nice & Fisher 2011).

Mardalup Park is located on the site of the former East Perth Gasworks which was built on the banks of the Claise Brook and the Swan Estuary (Figure 1). The gasworks operated between 1922 and 1971 and post-decommissioning, the site became a services depot for the State Energy Commission of Western Australia (SECWA). In 1989 SECWA commenced a contaminant assessment at the site and in 1992 it was reported that the site and the adjacent Claisebrook Drain (formerly Claise Brook) and Swan Estuary were extensively contaminated by coal tar and coal tar derivatives including a broad range of carcinogenic and toxic compounds such as polycyclic aromatic hydrocarbons (PAHs) (Bowman Bishaw Gorham 1992). Given this, the site was regarded as a seriously contaminated industrial site (EPA 1992).

The contaminated zone extended from approximately 50 m north to 250 m south of the gasworks site, including the western half of the Swan Estuary, to at least 0.5 m sediment depth in the centre of the contaminated zone (Bowman Bishaw Gorham 1992). The East Perth Redevelopment Authority subsequently redeveloped the site, whereupon an artificial canal-type waterway (Claisebrook Cove) was created at the outlet of the Claisebrook Drain (Figure 2). The resulting waterway is surrounded by both domestic (1 450 homes) and retail properties (EPRA 2009). Extensive remediation was conducted between 1994 and 1996 in accordance with the Minister's Conditions of Approval for the redevelopment of the site. This included replacing approximately 13 000 m^3 of PAH-contaminated sediment from the Swan Estuary (to a depth of 1 m below the estuary bed) with 12 200 m^3 of clean fill between April and October 1994. A further 12 000 m^3 (approximately) of sediment was removed to create the entrance channel for Claisebrook Cove (CMPS & F Pty Ltd 1996). A permanent cut-off curtain was constructed along the western boundary of the foreshore zone and a sheet pile wall was installed along the southern boundary with the estuary and cove to prevent offsite migration of contaminants into the estuary and cove. A drain – the Groundwater Interception Drain (GID) – was

constructed along the western boundary of Mardalup Park to intercept the groundwater and maintain the level under this public open space zone at or below estuary level in order prevent offsite migration of contaminated groundwater to the estuary (Axis Environmental 1996).

Despite extensive remediation, sediments collected from several sites in the remediated zone in 2009 and 2010 exhibited relatively high levels of PAHs (Nice 2009; Nice & Fisher 2011). These were present in forms that may indicate a recent or ongoing source of contamination to the area. In addition, compliance monitoring of the GID suggested contaminants including ammonia and PAHs were being discharged directly to the Swan Estuary (ENV 2009), at concentrations likely to be causing ecological impact.

In response, a comprehensive investigation of the Swan Estuary at Claisebrook was conducted in 2011, focusing primarily on the GID outfall to the estuary and following the multiple-lines-of-evidence approach proposed by Chapman et al. (1997). This investigation incorporated sediment chemistry assessment, whole-sediment toxicity assessment, water chemistry assessment and a benthic macroinvertebrate survey.

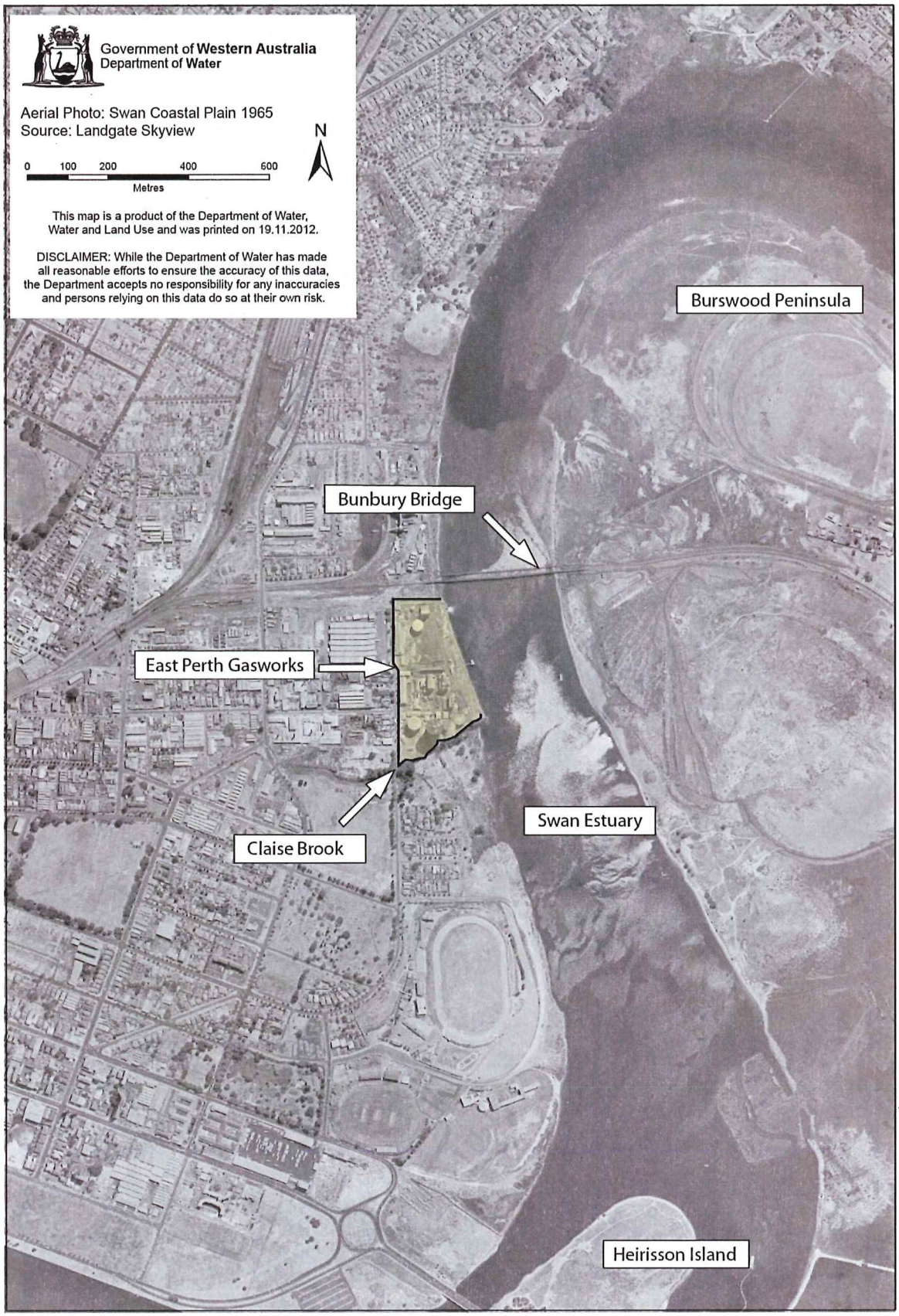


Figure 1 East Perth Gasworks and surrounding area – 1965



Figure 2 Historic East Perth Gasworks site and surrounding area – 2011

1.2 Scope

This report presents the results of the benthic macroinvertebrate survey and supporting sediment chemistry. The whole-sediment toxicity study (Nice 2013) and water chemistry assessment (Fisher 2013 in prep.) are reported separately.

1.3 Objectives

1. To compare biotic assemblages in 2011 at seven sites in the Swan Estuary to determine whether differences existed between sites. Specifically to determine whether:
   - biotic assemblages adjacent to the GID were different from other sites
   - differences in biotic assemblage could be explained by the sediment contaminants surveyed.
2. To compare biotic assemblages in 2011 at four sites in the Swan Estuary with biotic assemblages in 1997 at the same four sites. Specifically to determine whether:
   - formerly remediated sites have recovered; that is, to establish whether biotic assemblages at these sites have become more similar to those at reference sites.

2 Methods

Sediment samples were collected for the analyses of associated benthic macroinvertebrate fauna, chemical and physical characterisation.

2.1 Field sampling

Sampling was conducted in the Swan Estuary during autumn 2011 (29 and 30 March), when saltwater conditions in the estuary are typically most stable (prior to the onset of river discharge resulting from winter rainfall). The sampling period and methodology for macroinvertebrate collection followed that of Trayler and McKernan (1997) to enable the datasets to be compared.

Benthic macroinvertebrates were sampled at seven sites in the Swan Estuary (Figure 3). One site was located within Claisebrook Cove adjacent to the Claisebrook Drain outfall and six sites were located in the estuary spaced 500 m apart (three sites north and three sites south of Claisebrook Cove). One of the sites in the estuary was located adjacent to the GID outfall (CBI03). Four of the seven sites (CBI02 – CBI05) were the same sites previously sampled by Trayler and McKernan (1997).

At each site, a Petite Ponar grab sampler (grab dimensions: 150 mm x 150 mm x 150 mm) was used to collect five replicate grab samples for macroinvertebrate analysis and one grab sample for sediment chemistry and particle size analysis. Each site comprised a rectangular area approximately 25 m (adjacent to the shoreline) by 10 m. All samples were collected randomly within this area at a water depth of approximately 1 m.

Macroinvertebrates were separated from the sediment by elutriation into a 500 µm mesh sieve and preserved in 70% ethanol for laboratory identification by Ocean Vision Environmental Research Pty Ltd, Western Australia. Sediment chemistry samples were transferred immediately to amber glass jars and placed in the dark on ice for laboratory analysis by the National Measurement Institute, Western Australia. Samples for particle size analysis were placed in zip lock low-density polyethylene bags and placed in the dark on ice for laboratory analysis by CSIRO Minerals, Western Australia. All samples comprised sediment through the depth range 0 to 15 cm.

Temperature, salinity, pH and dissolved oxygen were measured in the water column 5 to 20 cm above the sediment surface (according to Simpson et al. 2005) at each sample location every second for two minutes before the sediment was disturbed (Yellow Springs Instruments hand-held meter model: 6600).

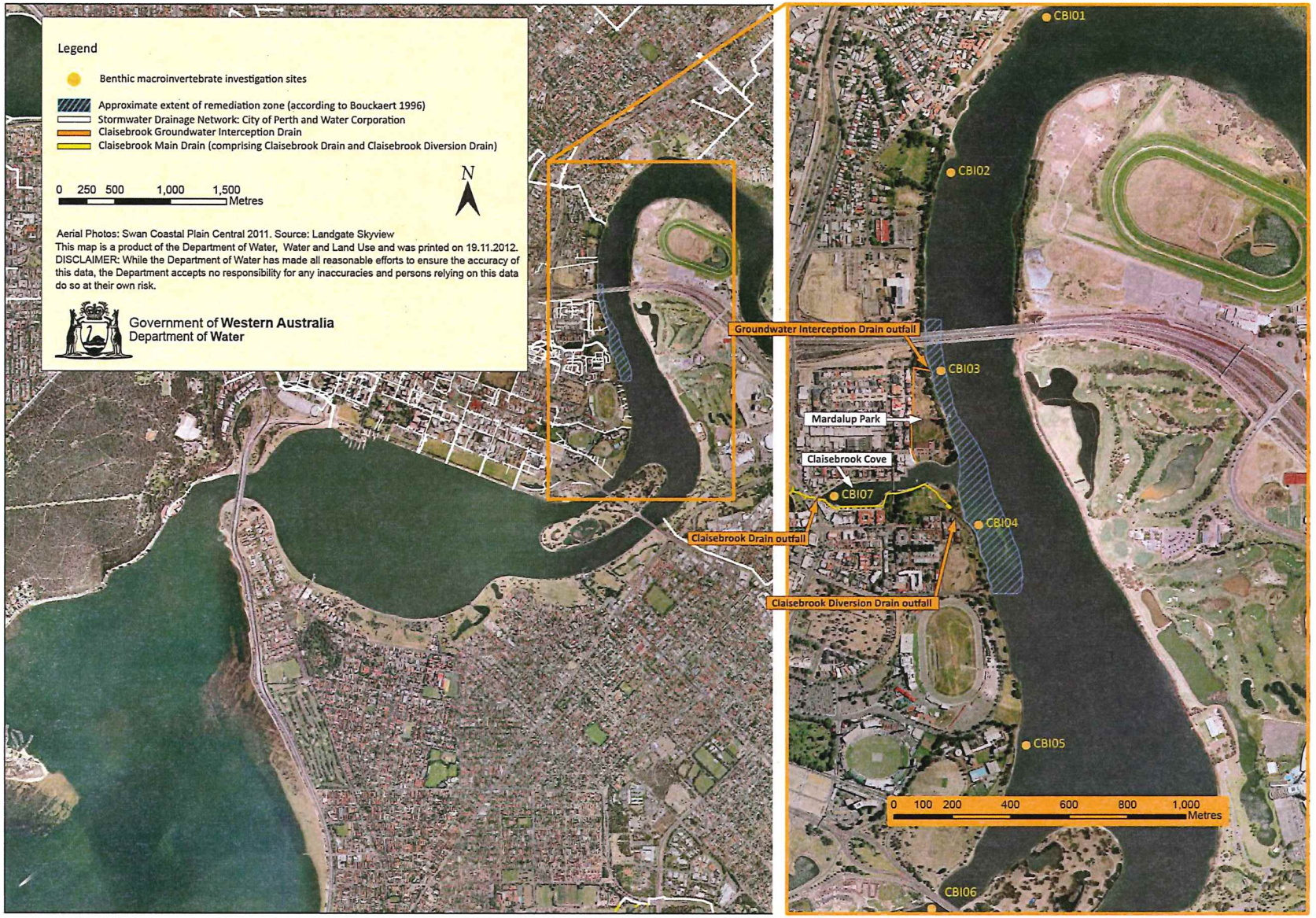


Figure 3 Location of sites in Claisebrook Cove and the Swan Estuary.

2.2 Laboratory processing and analyses

Macroinvertebrates were identified to the lowest taxonomic resolution possible using a dissecting microscope[2]. Sediment chemistry samples were homogenised within a controlled laboratory environment according to method AS 4482.1-1997 (Standards Australia 1997). Contaminants were quantified to the lowest-available limit of reporting using methods accredited by the National Association of Testing Authorities (NATA). Particle size and sediment chemistry analytical methods are provided in Table 1.

Table 1 Sediment chemistry and particle size methodology

Parameter	Limit of reporting (mg/kg)	Description	Analysis method
Bioavailable metals* Arsenic Cadmium Cobalt Chromium Copper Lead Mercury Manganese Nickel Selenium Zinc	0.5 for mercury 0.1 for other metals	Determination of bioavailable metal concentrations in sediments. Samples are tumbled with 1M hydrochloric acid in a sediment:acid ratio of 1:50 for one hour at room temperature (cold dilute acid extraction). Metal concentrations are determined in the extract using inductively coupled plasma mass spectrometry (ICP-MS) and/or inductively coupled plasma atomic emission spectrometry (ICP/AES). Units: mg/kg dry sediment.	ANZECC & ARMCANZ 2000
Polycyclic aromatic hydrocarbons (PAHs) Naphthalene Acenaphthylene Acenaphthene Fluorene Phenanthrene Anthracene Fluoranthene Pyrene Benz[a]anthracene Chrysene Benzo[b]and[k]fluoranthene Benzo[a]pyrene Indeno[1,2,3-cd]pyrene Dibenz[a,h]anthracene Benzo[ghi]perylene	0.01	Determination of PAH concentrations in sediments. PAH concentrations are determined using gas chromatography mass spectrometry (GC-MS) and gas chromatography flame ionization detection (GC-FID) analysis. Units: mg/kg dry sediment.	APHA 1998
Organochlorine (OC) pesticides HCB HCH(BHC) Lindane (gamma-BHC) Heptachlor Heptachlor epoxide Chlordane Alpha endosulphan Beta endosulphan	0.001	Determination of OC pesticide concentrations in sediments. OC pesticide concentrations are determined using GC-MS and gas chromatography electron capture detector GC-ECD analysis. Units: mg/kg dry sediment.	APHA 1998

[2] A reference collection was created with samples stored in 70% ethanol.

Parameter	Limit of reporting (mg/kg)	Description	Analysis method
Endosulphan sulphate Aldrin Dieldrin Endrin p,p'-DDE p,p'-DDD p,p'-DDT Methoxychlor			
Polychlorinated biphenyls (PCBs) Aroclor 1016 Aroclor 1221 Aroclor 1232 Aroclor 1242 Aroclor 1248 Aroclor 1254 Aroclor 1260 Total PCBs	0.01	Determination of PCB concentrations in sediments. PCB concentrations are determined using GC-MS and GC-ECD analysis. Units: mg/kg dry sediment.	APHA 1998
Total organic carbon (TOC)	100	Determination of TOC concentration within the sediments. Units: mg/kg dry sediment.	ANZECC & ARMCANZ 2000
Particle size analysis	n/a	Determination of the particle size distribution of sediments. Particles are separated by wet sieving followed by laser diffraction. Particles grouped into the following size classes according to the Wentworth scale (Wentworth 1922): < 4 µm (clay) >4 - 62 µm (silt) >62 - 250 µm (fine sand) >250 - 500 µm (medium sand) >500 - 2000 µm (coarse sand) >2000 - 10 000 µm (gravel)	Mudroch et al. 1997

** Bioavailable metals are extracted from sediment using a cold dilute acid extraction. This method extracts only metals loosely bound to the surface of sediment particles, leaving behind those tightly bound in the mineral matrix (ANZECC & ARMCANZ 2000). This is considered to provide an approximation of the metals that are biologically available.*

2.3 Application of guidelines

Sediment chemistry data were compared with the Interim Sediment Quality Guideline trigger values (ISQGs) from the Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand (ANZECC & ARMCANZ 2000). The *low ISQG* represents the concentration below which the frequency of adverse biological effects is expected to be low. The *high ISQG* represents the concentration above which adverse biological effects are expected to occur frequently. The ISQGs are typically applied to sediment contaminant concentrations in the top 2 cm of the sediment profile when measuring contaminants in surficial sediment. However, the ISQGs can be applied to sediment contaminant concentrations from a range of depths (Simpson et al. 2005). Since most epifaunal and infaunal organisms occupy the upper 10 cm of sediments, it was deemed appropriate for this study to target at least the top 10 cm.

In this case, the sediment sample was a composite of the top 15 cm so that the macroinvertebrate data could be directly comparable with an earlier dataset (Trayler & McKernan 1997).

Concentrations of organic contaminants such as PAHs, PCBs and OC pesticides measured here are typically normalised to 1% organic carbon for comparison with the ISQGs (Simpson et al. 2005). There is some conjecture as to whether normalising to 1% organic carbon is appropriate where organic carbon concentrations are considered to be high. That is, in instances where total organic carbon concentrations are suspected to have been increased above normal concentrations due to organic contamination (such as petroleum compounds as seen in this study), the organic carbon normalised values may be inappropriately low and may not exceed ISQGs even though adverse biological effects may occur (Michelsen 1992). As such, both normalised and non-normalised PAH, PCB and OC pesticide data are presented here.

2.4 Statistical analyses

Prior to analyses the distributions of biological data were tested for normality (Kolmogorov-Smirnov test) and homogeneity of variance (Levene test). Data were transformed where required and the appropriate tests selected. Simple community measures such as abundance and species richness were analysed using univariate techniques to determine differences between sites using Analysis of Variance (ANOVA) and Tukey HSD post-hoc.

The relationship between biotic assemblages in response to a range of environmental contaminants was examined by non-parametric multivariate analyses performed using the PRIMER (Plymouth Routines in Multivariate Ecological Research) v6 statistical package (Clarke & Warwick 2001). Biotic assemblage data (species abundance) were log transformed (x+1) before the analyses[3]. Environmental data (sediment chemistry concentrations) were range standardised prior to analysis because datasets had different scales.

Species abundance data were ordinated by Multi-Dimensional Scaling (MDS). Sites were clustered according to similarities in biotic assemblage using the Bray-Curtis similarity matrix. One-Way Multivariate Analysis of Similarity (ANOSIM) was conducted to assess overall community compositional differences between sites (statistical significance was set at $\alpha = 0.05$). Similarity Percentage (SIMPER) analysis was conducted to determine the species most responsible for contributing to any separation between sites.

Trends in environmental data were explored with the similarity matrix based on Euclidean distance. The PRIMER procedure Biota and Environmental Matching (BIOENV) was conducted to determine which of the sediment chemistry parameters best explained the patterns in biotic assemblage (using Spearman's rank correlation). Vectors showing sediment chemistry parameters identified by BIOENV

[3] One outlier was identified by visual analysis of MDS and removed from the dataset (one replicate of five at one site).

as best explaining the community assemblage pattern (i.e. with correlations $\geq$ 0.8) were superimposed onto the MDS plot. To further demonstrate the relationships between biotic community assemblages and sediment chemistry parameters, bubble plots were constructed for each of the sediment chemistry parameters identified by BIOENV as best explaining the community assemblage pattern.

For comparisons between the 1997 and 2011 datasets, species abundance data for four of the seven sites tested in 2011 (CBI02, CBI03, CBI04 and CBI05) were divided into two categories: reference sites and remediation sites according to classification in Trayler and McKernan (1997). The 1997 study (Trayler & McKernan 1997) did not incorporate sediment chemistry assessment, thus sediment chemistry data were not available for 1997. Simple community measures such as abundance and richness were compared between sampling years. Species abundance data from 1997 and 2011 (sites CBI02 – CBI05) were ordinated by MDS. Sites were clustered according to similarities in biotic assemblage using the Bray-Curtis similarity matrix. ANOSIM was conducted to assess overall community compositional differences between sites and years. Statistical significance was set at $\alpha = 0.05$. SIMPER analysis was conducted to determine the species most responsible for contributing to any separation between groups.

3 Results

Results are presented in two sections:

1 Analysis of 2011 dataset comprising seven sites within the Swan Estuary and Claisebrook Cove

2 Analysis of a subset of four sites from the 2011 dataset compared with the same four sites in 1997

Summary

Comparison of biotic assemblages - 2011

- There were differences in biotic assemblages between sites.
- Of all the sites examined, CBI07 (Claisebrook Main Drain, within the cove) was most different from the others in terms of biotic assemblage and typically displayed among the highest concentrations of contaminants (concentrations were higher than guidelines for several contaminants) and the highest proportion of silt.
- Of the sites in the estuary (CBI01 – CBI06), CBI06 was most different from the others in terms of biotic assemblage and generally had the highest PAH concentrations and the highest concentrations of some metal contaminants (concentrations were higher than guidelines for several contaminants).
- Sixteen key contaminants were identified as contributing to the separation of sites (in conjunction with the proportion of organic carbon and fine sediments present). The PAH acenapthene showed the strongest correlation in terms of the separation of site CBI06 from the others. Other key contaminants included several other PAHs, one OC pesticide and several metals.
- Multivariate analyses showed no clear distinction in biotic assemblage at site CBI03 (GID) from the two sites upstream (CBI01 and CBI02); and simple community measures such as abundance and richness also showed no difference at site CBI03 when compared with other sites in the estuary.
- Sediment contaminant concentrations were generally relatively low at site CBI03 (GID) despite the sediment having a similar profile (and hence binding capacity) as the other estuary sites, which had higher contaminant concentrations.
- PCBs were detected at three sites: CBI04 and CBI06 in the estuary and CBI07 within the cove.

Comparison of 1997 and 2011 biotic assemblages at remediation and reference sites

- Remediation and reference sites were more similar to each other in 2011 than they appeared in 1997. However, biotic assemblages at both reference and remediation sites in 2011 were strongly departed from both reference and remediation assemblages in 1997. That is, there was no evidence that sites in 2011 had become more like the reference sites as expressed in 1997. Rather, the entire assemblage set was departed from the 1997 status.
- Richness was higher in 2011 than 1997.

3.1 2011 dataset

Biotic assemblages

Thirty-seven taxa were identified in this study (mostly to species level). The fauna comprised species of molluscs, crustaceans, annelids and chordates, with composition and abundance being variable across sites (Figure 4). In terms of composition at phylum level (outer circles Figure 4), sites CBI06 and CBI07 appeared to be most distinct from other sites. Arthropods (sub-phylum crustaceans) dominated composition at CBI06 compared with molluscs for all other sites. CBI06 was the only site where chordates were represented. CBI07 was notably different from other sites by the absence of arthropods (sub-phylum crustaceans).

Univariate analyses of simple community measures showed there were significant differences in mean total abundance ($F_{(6, 28)} = 6.04$, $p < 0.001$) and mean species richness ($F_{(6, 28)} = 14.44$, $p < 0.001$) across sites (Figure 5). Mean total abundance at site CBI07 was significantly lower than at sites CBI01 ($p < 0.001$), CBI02 ($p < 0.01$), CBI03 ($p < 0.05$) and CBI04 ($p < 0.01$). Mean total abundance at site CBI06 was significantly lower than that at site CBI01 ($p < 0.05$). Mean species richness was significantly lower at site CBI07 than all other sites ($p < 0.001$).

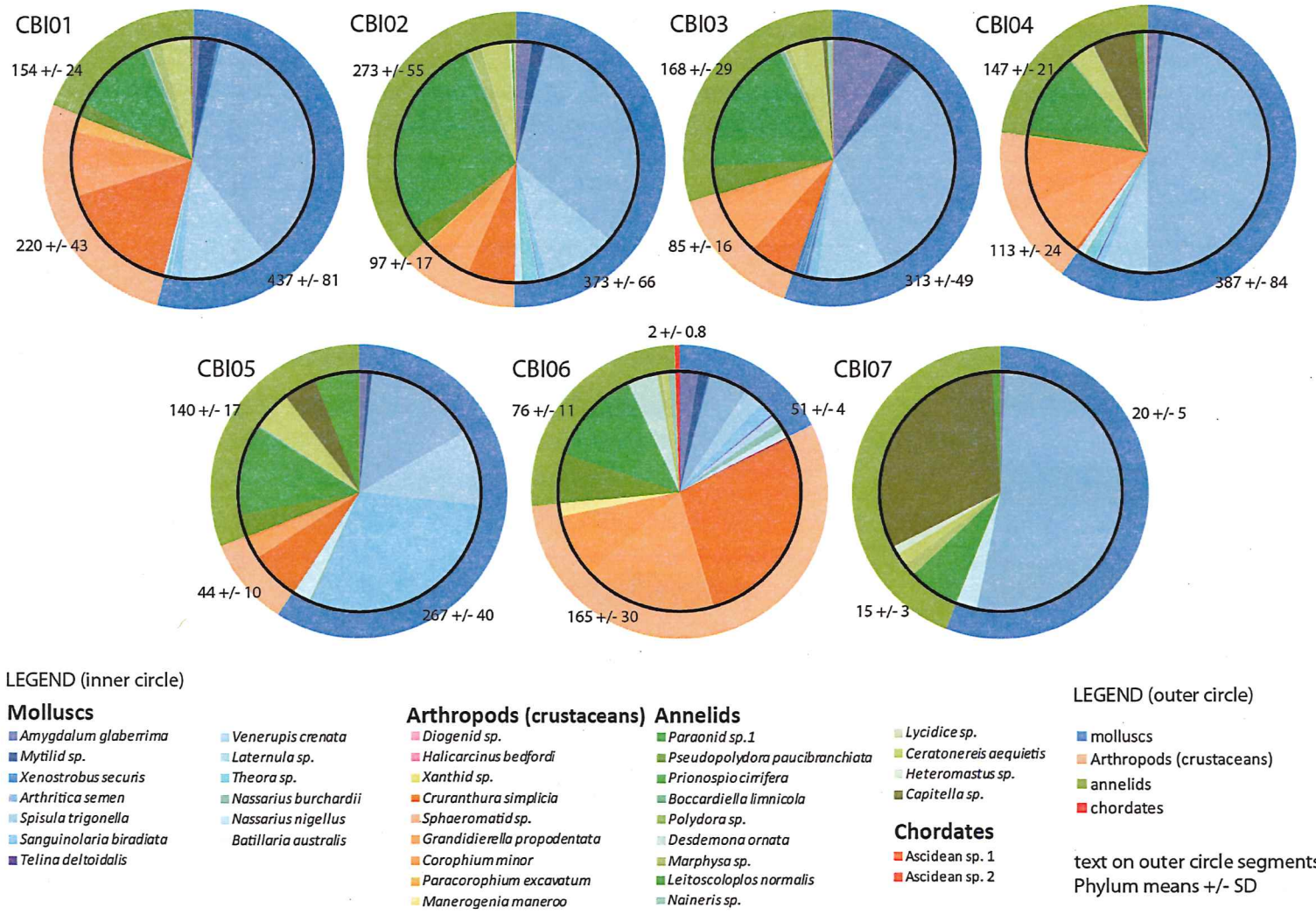


Figure 4 Community composition at each site. Slices represent species; outer circles represent phyla.

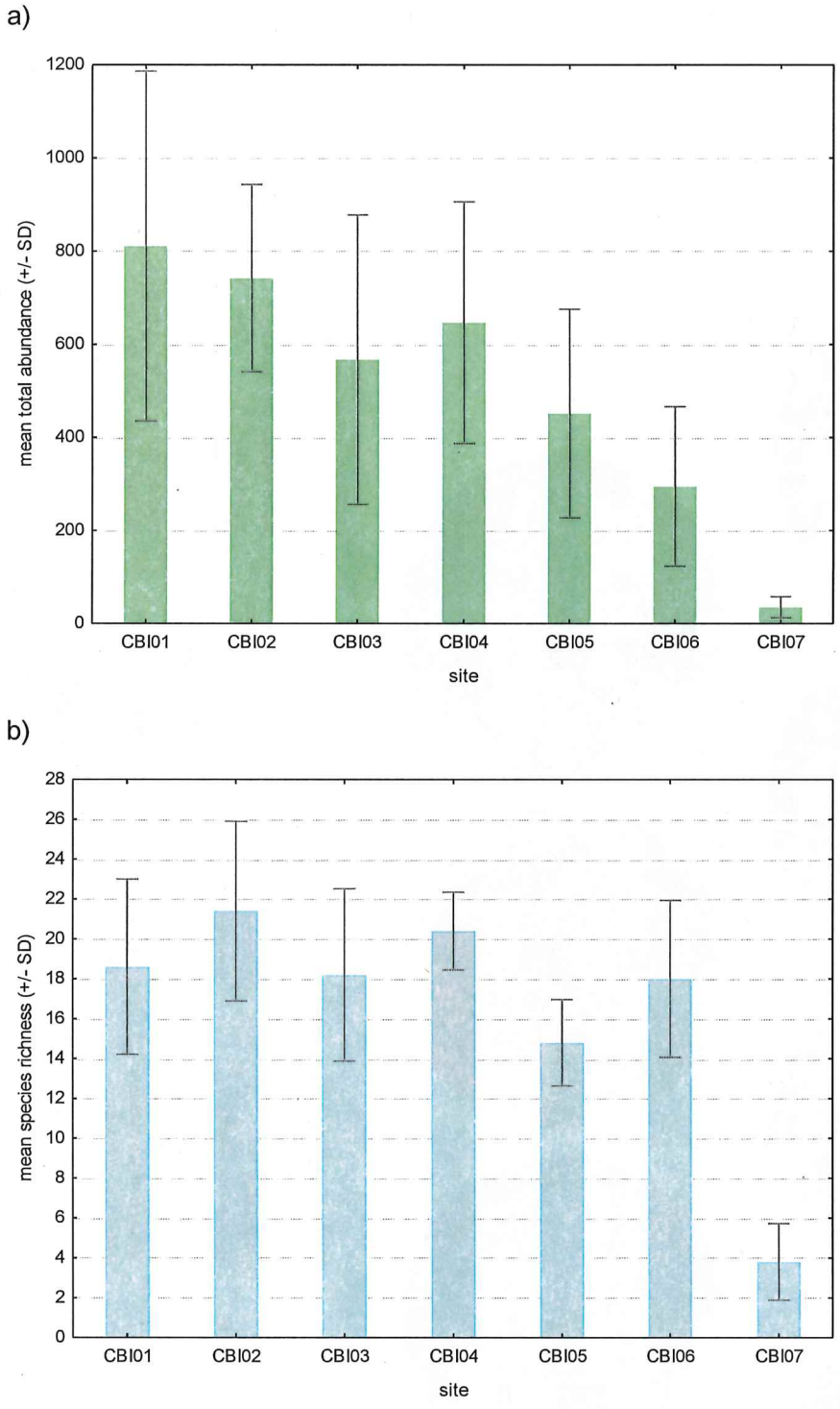


Figure 5 Community measures across sites: a) mean total abundance and b) mean species richness.

MDS ordination[4] (Figure 6) of species assemblages across all seven sites showed site CBI07 to be strongly separated from the other groups. There was a relatively high variability in species assemblage between replicate samples at CBI07 compared with the other sites. Sites CBI04, CBI05 and CBI06 each formed distinct groups, while CBI01, CBI02 and CBI03 were clustered together with no distinct separation of replicates into groups according to site.

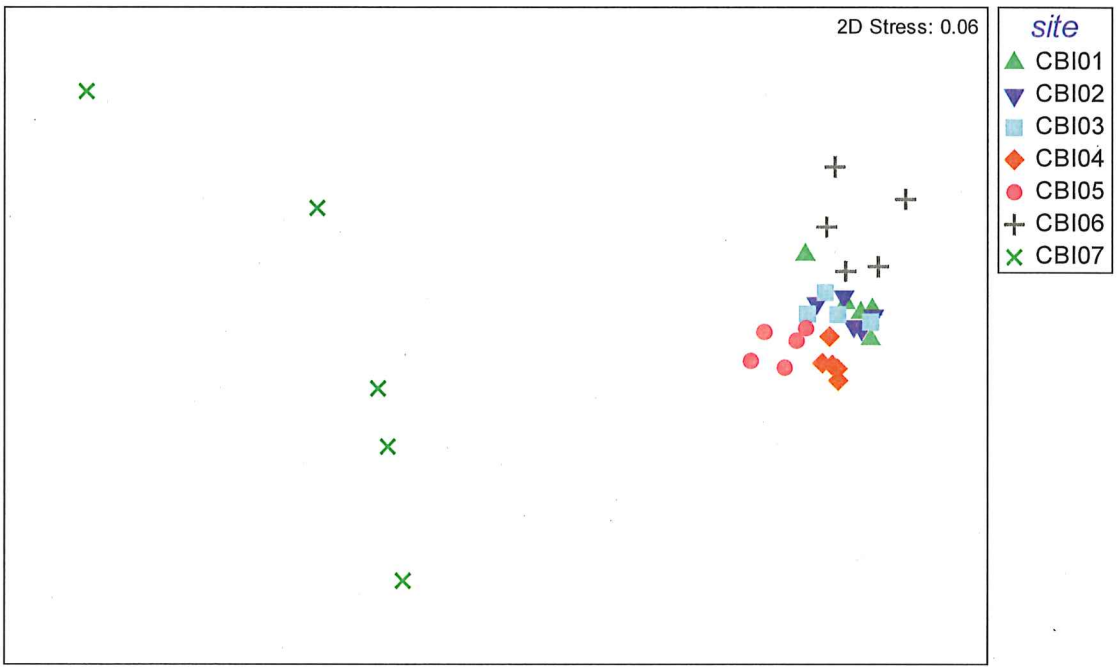


Figure 6 MDS ordination for species abundance (Log(x+1) transformed) at seven sites.

These observations were confirmed by ANOSIM, which showed significant differences between groups (Global R = 0.58; p = 0.001). Pairwise comparisons of sites are shown in Table 2 to illustrate which groups were statistically different from each other. There were statistical differences between many of the sites. In particular, the biotic assemblage at CBI05 was distinctly different from that at CBI02 and CBI04; and that at CBI06 was distinctly different from that at CBI07 – indicated by R-statistics of almost 1 for these cases (R = 0.99, 0.97 and 0.96 respectively) (Table 2).

[4] Note: the stress level of 0.06 provides a good to excellent representation of the data in two dimensions (Clarke & Warwick 2001).

Table 2 Pairwise comparisons of sites by ANOSIM

Pairwise comparison of sites	R-statistic	P
CBI01 x CBI02	0.20	> 0.05
CBI01 x CBI03	0.11	> 0.05
CBI01 x CBI04	0.78	< 0.01**
CBI01 x CBI05	0.90	< 0.01**
CBI01 x CBI06	0.50	< 0.05*
CBI01 x CBI07	0.89	< 0.01**
CBI02 x CBI03	0.11	> 0.05
CBI02 x CBI04	0.78	< 0.01**
CBI02 x CBI05	0.99	< 0.01**
CBI02 x CBI06	0.67	< 0.01**
CBI02 x CBI07	0.92	< 0.01**
CBI03 x CBI04	0.79	< 0.01**
CBI03 x CBI05	0.83	< 0.01**
CBI03 x CBI06	0.48	< 0.05*
CBI03 x CBI07	0.79	< 0.01**
CBI04 x CBI05	0.97	< 0.01**
CBI04 x CBI06	0.84	< 0.01**
CBI04 x CBI07	0.76	< 0.01**
CBI05 x CBI06	0.91	< 0.01**
CBI05 x CBI07	0.75	< 0.01**
CBI06 x CBI07	0.96	< 0.01**

*Significance levels: *significantly different, p < 0.05; ** highly significantly different, p < 0.01.*

To determine whether the apparent grouping (Figure 6) was real, the data from sites CBI01, CBI02 and CBI03 (which were not statistically different from one another and whose pairwise comparisons yielded comparatively low R-statistics – Table 2), were pooled into a combined site (CBI123) and ANOSIM conducted between this new combined site and other sites. The resulting difference between groups was more distinct as indicated by the higher Global R-statistic (Global R = 0.76; p = 0.001). Pairwise comparisons of the combined site CBI123 with the remaining sites and significance levels are shown in Table 3.

Table 3 Pairwise comparisons of sites by ANOSIM (sites CBI01, CBI02 and CBI03 combined)

Pairwise comparison of sites	R-statistic	P
CBI123 x CBI04	0.59	= 0.001**
CBI123 x CBI05	0.83	= 0.001**
CBI123 x CBI06	0.80	= 0.001**
CBI123 x CBI07	0.98	= 0.001**

*Significance levels: ** highly significantly different, p < 0.01.*

Differences in biotic assemblage between each of CBI04, CBI05, CBI06 and CBI07, when compared with combined site CBI123, were all significant (p = 0.001): with that at site CBI07 being distinctly different from that at combined site CBI123 – indicated by an R-statistic of almost 1 (Table 3).

The most influential taxa in separating groups were determined through SIMPER analyses and are shown in Table 4. Sites CBI01, CBI02 and CBI03 were combined to reduce 'noise' for SIMPER analyses.

Table 4 Most influential taxa in separating pairs of sites as determined by SIMPER analyses

Pairwise comparison of site groups	Average % dissimilarity between groups	Taxa most responsible for separation of groups	% contribution towards separation of groups	Observation (based on abundance)
CBI123 x CBI04	29.88	*Cruranthura simplicia* (AC)	11.4	CBI123 > CBI04
		Capitella sp. (A)	9.84	CBI123 < CBI04
CBI123 x CBI05	35.81	*Sanguinolaria biradiata* (M)	8.81	CBI123 < CBI05
		Leitoscoloplos normalis (A)	8.80	CBI123 < CBI05
		Corophium minor (AC)	7.22	CBI123 > CBI05
CBI123 x CBI06	38.25	*Arthritica semen* (M)	11.23	CBI123 > CBI06
		Spisula trigonella (M)	7.22	CBI123 > CBI06
CBI04 x CBI05	32.13	*Corophium minor* (AC)	12.12	CBI04 < CBI05
		Sanguinolaria biradiata (M)	10.22	CBI04 < CBI05
		Cruranthura simplicia (AC)	8.61	CBI04 < CBI05
CBI04 x CBI06	45.75	*Arthritica semen* (M)	10.12	CBI04 > CBI06
		Cruranthura simplicia (AC)	9.08	CBI04 < CBI06
		Capitella sp. (A)	8.91	CBI04 > CBI06
CBI05 x CBI06	47.53	*Leitoscoloplos normalis* (A)	9.21	CBI05 > CBI06
		Sanguinolaria biradiata (M)	8.62	CBI05 > CBI06
		Capitella sp. (A)	7.25	CBI05 > CBI06
CBI123 x CBI07	82.84	*Spisula trigonella* (M)	8.94	CBI123 > CBI07
		Cruranthura simplicia (AC)	8.58	CBI123 > CBI07
		Prionospio cirrifera (A)	8.39	CBI123 > CBI07
		Grandidierella propodentata (AC)	7.01	CBI123 > CBI07
CBI04 x CBI07	77.30	*Corophium minor* (AC)	9.56	CBI04 > CBI07
		Grandidierella propodentata (AC)	9.16	CBI04 > CBI07
		Spisula trigonella (M)	8.61	CBI04 > CBI07
		Prionospio cirrifera (A)	8.23	CBI04 > CBI07
		Arthritica semen (M)	7.20	CBI04 > CBI07
CBI05 x CBI07	75.42	*Sanguinolaria biradiata* (M)	13.70	CBI05 > CBI07

Pairwise comparison of site groups	Average % dissimilarity between groups	Taxa most responsible for separation of groups	% contribution towards separation of groups	Observation (based on abundance)
		Spisula trigonella (M)	10.87	CBI05 > CBI07
		Leitoscoloplos normalis (A)	8.83	CBI05 > CBI07
		Cruranthura simplicia (AC)	8.65	CBI05 > CBI07
		Ceratonereis aequietis (A)	8.06	CBI05 > CBI07
		Prionospio cirrifera (A)	7.90	CBI05 > CBI07
		Grandidierella propodentata (AC)	7.33	CBI05 > CBI07
CBI06 x CBI07	86.74	*Cruranthura simplicia* (AC)	11.64	CBI06 > CBI07
		Grandidierella propodentata (AC)	11.04	CBI06 > CBI07
		Corophium minor (AC)	7.87	CBI06 > CBI07
		Prionospio cirrifera (A)	7.08	CBI06 > CBI07

Key: M: molluscs; AC: arthropod crustaceans; A: annelids; C: chordates.

Note: only species likely to be consistent discriminators of groups are listed (according to Clarke & Warwick 2001). The dissimilarity:standard deviation ratio $\leq$ 1.9 in all cases above.

Environmental data

Sediment metal (bioavailable), PAH, pesticide, PCB and organic carbon concentrations are presented in Figure 7 to Figure 9 and Table 5 to Table 8. Particle size and *in situ* water quality data are presented in Table 9 and Table 10 respectively.

Most metals were present in concentrations above the limits of reporting at all sites with the following exceptions: mercury, which was not detected at any site; and cadmium and selenium, which were both only detected at site CBI07 (Figure 7 and Table 5). Guidelines were exceeded for lead and zinc at site CBI07.

Organic carbon data are notably high (Table 8) and thus the following contaminant data were not normalised to 1% organic carbon as recommended in Michelsen (1992). Normalised contaminant data are available in Appendix A and indicate the same general trends as non-normalised data – with similar numbers of ISQGs being exceeded for pesticides and PCBs and fewer guidelines exceeded for PAHs.

Most PAHs assessed in this study were present in concentrations above the limit of reporting at all sites and a peak in the concentrations of individual PAHs was observed for sites CBI06 (*low* guideline exceeded for six PAHs), CBI07 (*low* guideline exceeded for three PAHs) and, to a lesser extent, CBI04 (*low* guideline exceeded for one PAH) (Figure 8 and Table 6). Of the pesticides assessed in this study, aldrin was present at concentrations above the limit of reporting at sites CBI04 and CBI07; trans-chlordane and p,p'-DDT at sites CBI01, CBI04 and CBI07; dieldrin at sites CBI01, CBI02 and CBI06; p,p'-DDE at all sites except CBI03; and p,p'-DDD at all sites except CBI03 and CBI05. *Low* ISQGs were exceeded at all sites except CBI03 (where no pesticide was detected) and CBI05. A peak in concentrations was

evident at site CBI07 where two high ISQGs were also exceeded (Figure 9 and Table 7). The PCB mixture Arochlor 1254 was present in concentrations above the limit of reporting at sites CBI04, CBI06 and CBI07, exceeding the *low* ISQG for sites CBI04 and CBI07 (Figure 9 and Table 7).

Sediments collected from each site consisted of particles from the spectrum of size categories according to the Wentworth scale (Wentworth 1922). The dominant fraction was medium sand for CBI01, CBI02, CBI03 and CBI04; fine sand for CBI05 and CBI06; and silt for CBI07 (Table 9).

Temperature, specific conductivity, salinity and pH were consistent across sites. Dissolved oxygen concentration varied between sites but the water was generally classed as moderately oxygenated (according to dissolved oxygen classifications for the Swan Estuary by Robb & Evans 2008), with the following exceptions: CBI05 was classed as well oxygenated and CBI07 was classed as poorly oxygenated (Table 10).

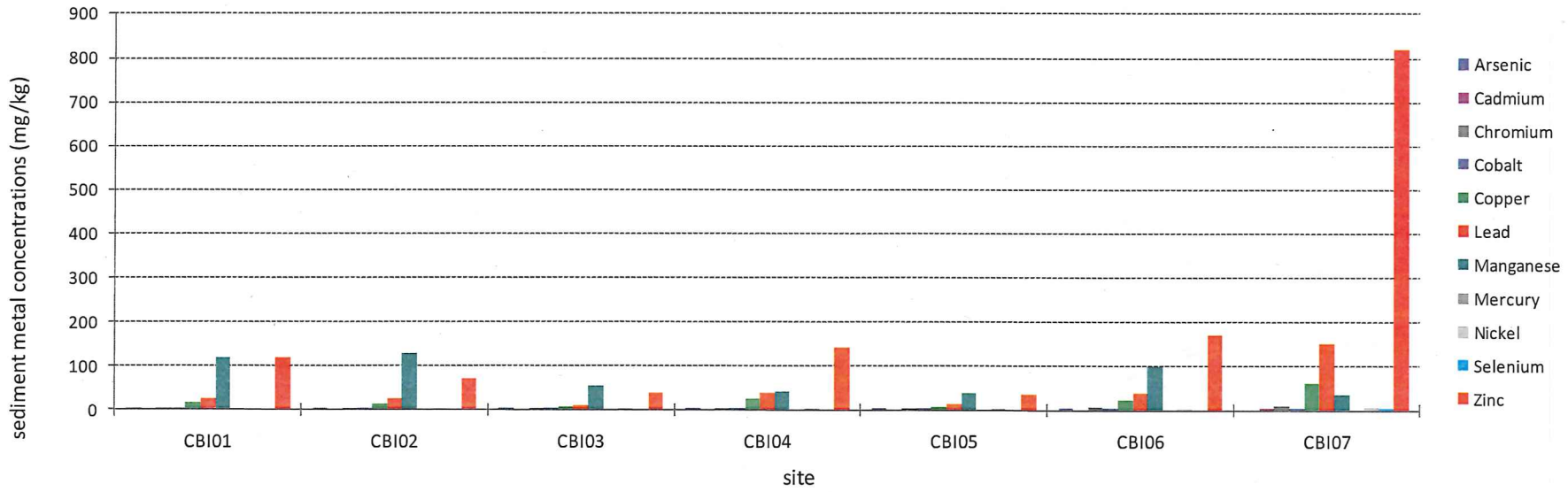


Figure 7 Sediment metal concentrations (bioavailable)

Table 5 Sediment metal concentrations (bioavailable) and corresponding guidelines

	Sediment metal concentrations (bioavailable) mg/kg dry weight										
Site	Arsenic	Cadmium	Chromium	Cobalt *	Copper	Lead	Manganese *	Mercury	Nickel	Selenium *	Zinc
CBI01	1.3	n.d.	2.8	2.6	15	26	120	n.d.	1.7	n.d.	120
CBI02	1.7	n.d.	3.2	2.4	13	26	130	n.d.	1.5	n.d.	71
CBI03	0.55	n.d.	1.8	1	5.6	10	56	n.d.	0.8	n.d.	37
CBI04	1.6	n.d.	4.1	1.2	26	39	43	n.d.	1.3	n.d.	140
CBI05	0.94	n.d.	2.1	0.95	5.2	13	38	n.d.	0.62	n.d.	36
CBI06	2.5	n.d.	5.8	3.2	23	38.5	100	n.d.	2.2	n.d.	170
CBI07	n.d.	0.89	11	2.1	61	150	34	n.d.	5.5	1.1	820
ISQG low	20	1.5	80	n.a.	65	50	n.a.	0.15	21	n.a.	200
ISQG high	70	10	370	n.a.	270	220	n.a.	1	52	n.a.	410

*ISQG = Interim Sediment Quality Guideline (ANZECC & ARMCANZ 2000); limit of reporting for mercury: 0.5 mg/kg; limit of reporting for other metals: 0.1 mg/kg; blue indicates low ISQG exceeded; red indicates high ISQG exceeded; n.a. = no ANZECC & ARMCANZ guideline available; * alternative guidelines for cobalt, manganese and selenium of 50, 1100 and 2 mg/kg respectively (Ontario Sediment Quality Guidelines 1993; Lemly 1996) were also not exceeded. Samples comprised the top 15 cm of the sediment.*

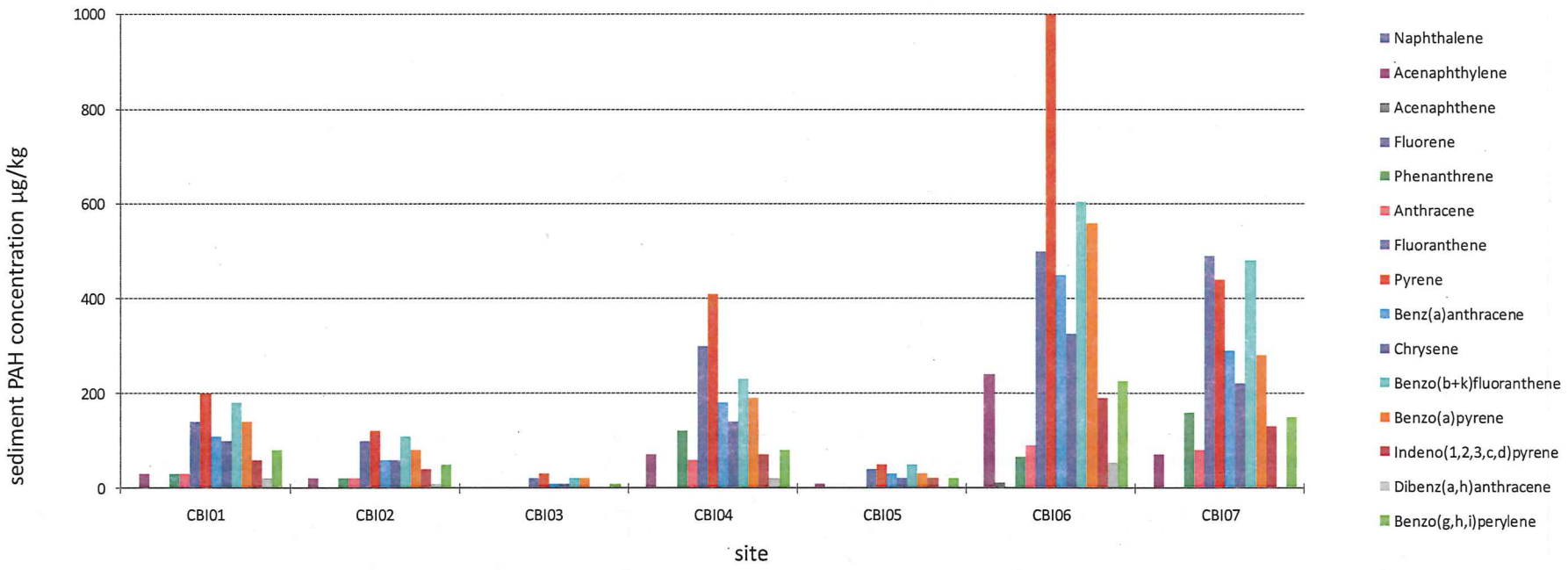


Figure 8 Sediment polycyclic aromatic hydrocarbon (PAH) concentrations

Table 6 Sediment polycyclic aromatic hydrocarbon (PAH) concentrations and corresponding guidelines

	Sediment PAH concentrations (µg/kg) dry weight														
Site	Naphthalene	Acenaphthylene	Acenaphthene	Fluorene	Phenanthrene	Anthracene	Fluoranthene	Pyrene	Benz[a]anthracene	Chrysene	Benzo[b+k]fluoranthene *	Benzo[a]pyrene	Indeno[1,2,3,c,d] Pyrene	Dibenz[a,h]anthracene	Benzo[g,h,i]perylene
CBI01	n.d.	30	n.d.	n.d.	30	30	140	200	110	100	180	140	60	20	80
CBI02	n.d.	20	n.d.	n.d.	20	20	100	120	60	60	110	80	40	10	50
CBI03	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	20	30	10	10	20	20	n.d.	n.d.	10
CBI04	n.d.	70	n.d.	n.d.	120	60	300	410	180	140	230	190	70	20	80
CBI05	n.d.	10	n.d.	n.d.	n.d.	n.d.	40	50	30	20	50	30	20	n.d.	20
CBI06	n.d.	240	12.5	n.d.	65	90	500	1005	450	325	605	560	190	55	225
CBI07	n.d.	70	n.d.	n.d.	160	80	490	440	290	220	480	280	130	n.d.	150
ISQG low	160	44	16	19	240	85	600	665	261	384	n.a.	430	n.a.	63	n.a.
ISQG high	2100	640	500	540	1500	1100	5100	2600	1600	2800	n.a.	1600	n.a.	260	n.a.

*ISQG = Interim Sediment Quality Guideline (ANZECC & ARMCANZ 2000); blue indicates low ISQG exceeded; red indicates high ISQG exceeded; n.a. = no ANZECC & ARCANZ guideline available; * alternative guidelines for benzo[b+k]fluoranthene of 240 and 1340000 µg/kg (Ontario Sediment Quality Guidelines 1993 lowest effect level and severe effect level respectively). N.d. = not detected; limit of reporting: 10 µg/kg. Samples comprised the top 15 cm of the sediment. Data not normalised to 1% OC.*

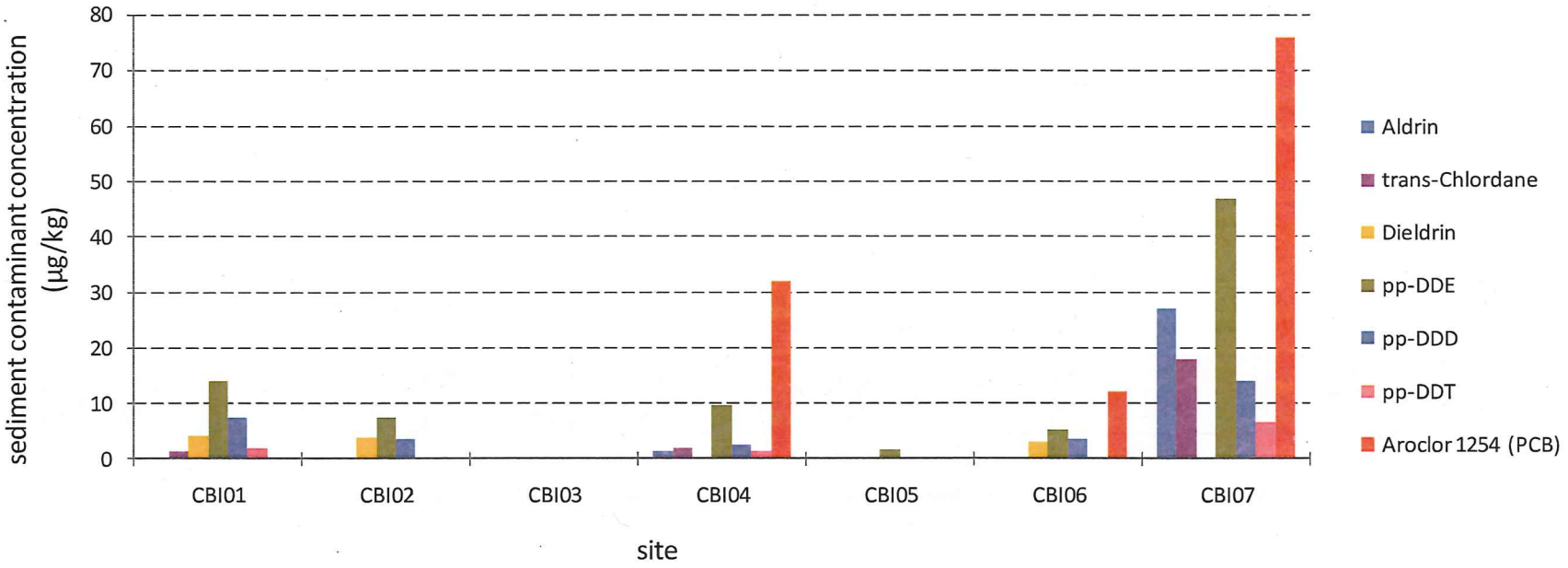


Figure 9 Sediment pesticide and polychlorinated biphenyl (PCB) concentrations

Table 7 Sediment pesticide and polychlorinated biphenyl (PCB) concentrations and corresponding guidelines

Sediment pesticide and PCB concentrations (µg/kg) dry weight							
Site	Aldrin *	trans-Chlordane**	Dieldrin	p,p'-DDT^{+}	p,p'-DDE	p,p'-DDD^{++}	Aroclor 1254^
CBI01	n.d.	1.2	4.2	1.8	14	7.5	n.d.
CBI02	n.d.	n.d.	3.9	n.d.	7.4	3.5	n.d.
CBI03	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
CBI04	1.2	2.0	n.d.	1.3	9.6	2.4	32
CBI05	n.d.	n.d.	n.d.	n.d.	1.5	n.d.	n.d.
CBI06	n.d.	n.d.	3.1	n.d.	5.3	3.4	12
CBI07	27	18	n.d.	6.5	47	14	76
ISQG low	n.a.	0.5	0.02	1.6	2.2	2	23
ISQG high	n.a	6.0	8	46	27	20	n.a.

*Note: Only those parameters detected are shown in this table. See Table 1 for the full list of contaminants. ISQG = Interim Sediment Quality Guideline (ANZECC & ARMCANZ 2000); blue indicates low ISQG exceeded; red indicates high ISQG exceeded; n.a. = no ANZECC & ARMCANZ guideline available. * alternative guideline for aldrin of 2 µg/kg (Canadian SQG 2002). ** trigger values quoted for trans-chlordane are for chlordane. $^{+}$trigger values quoted for p,p'DDT are for total DDT (only measured p,p'-DDT in this study). $^{++}$ trigger values quoted for p,p'-DDD are for total p,p'-DDD and op-DDD (only p,p'-DDD was measured in this study). ^trigger value quoted for Aroclor 1254 is for total PCBs. N.d. = not detected; limit of reporting: 1 µg/kg for pesticides and 10 µg/kg for Aroclor mixtures. Samples comprised the top 15 cm of the sediment. Data not normalised to 1% organic carbon.*

Table 8 Total organic carbon concentrations

Site	Total organic carbon (mg/kg)
CBI01	25 000
CBI02	16 000
CBI03	6 100
CBI04	14 000
CBI05	3000
CBI06	19 000
CBI07	110 000

Table 9 Sediment particle size

Site	Proportion of sediments (% by weight)					
	Fine sediment			Sand		Gravel
	Clay	Silt	Fine sand	Medium sand	Coarse sand	Gravel
	0.02–4 µm	4–62 µm	62–250 µm	250–500 µm	500–2000 µm	2000–10 000 µm
CBI01	4.9	7.0	19.2	48.2	16.7	4.0
CBI02	5.4	11.9	12.9	39.8	22.7	7.4
CBI03	4.3	9.2	7.3	36.4	31.9	10.9
CBI04	3.7	9.1	15.8	32.0	27.9	11.5
CBI05	2.2	6.7	45.2	33.9	2.2	9.8
CBI06	9.9	25.2	46.5	14.7	1.4	2.3
CBI07	8.7	35.1	24.6	17.3	5.7	8.6

Blue text indicates dominant fraction

Table 10 In situ water quality parameters

Site	Temperature (° C)	Specific conductivity (m^S/cm)	Salinity (ppt)	pH	Dissolved oxygen (mg/L)
CBI01	25.18 (+/- 0.05)	56.56 (+/- 0.10)	37.61(+/- 0.07)	7.42–7.44	4.92 (+/- 0.07)
CBI02	25.43 (+/- 0.09)	56.85 (+/- 0.09)	37.81 (+/- 0.07)	7.52–7.56	5.65 (+/- 0.17)
CBI03	25.41 (+/- 0.25)	56.85 (+/- 0.09)	37.82 (+/- 0.07)	7.52–7.58	5.04 (+/- 0.31)
CBI04	26.14 (+/- 0.19)	56.95 (+/- 0.08)	37.87 (+/- 0.07)	7.63–7.68	5.71 (+/- 0.30)
CBI05	26.48 (+/-0.15)	57.37 (+/- 0.03)	38.18 (+/- 0.02)	7.71– 7.77	6.67 (+/- 0.26)
CBI06	25.00 (+/- 0.24)	57.50 (+/- 0.05)	38.31 (+/- 0.04)	7.66–7.71	5.26 (+/- 0.27)
CBI07	25.79 (+/- 0.18)	56.64 (+/- 0.91)	37.64 (+/-0.68)	7.42–7.55	3.64 (+/- 0.53)

Measured 5 to 20cm above sediment surface according to Simpson et al. (2005). Temperature, conductivity, salinity and dissolved oxygen data expressed as means (+/- SD). pH data expressed as range.

Relationship between biotic assemblages and environmental factors: sites CBI01 to CBI06

The following section explores the relationship between biotic assemblages and the range of environmental contaminants quantified in this study. As it was not possible to separate *cove* effect[5] from *contaminant* effect for site CBI07, data from this site have not been subject to further detailed statistical analyses. Multivariate analyses exploring relationships between biotic assemblages and environmental factors are presented here for estuary sites (CBI01 to CBI06) only. The relationships between species assemblages and contaminants at site CBI07 are summarised on page 33.

MDS ordination of species assemblages (based on mean abundance data) showed sites CBI05 and CBI06 to be most strongly separated from the other groups (Figure 10). The BIOENV procedure identified 16 contaminants (in conjunction with proportion of organic carbon and fine sediments[6]) under various combinations within 18 tests as most influential to the separation of sites, where all permutations returned a correlation $\geq$ 0.8. These are each displayed as vectors overlying the MDS to illustrate their relative relationship with the separation of sites (Figure 10). The 16 contaminants are also displayed as bubble plots indicating their relative concentrations (Figure 11). These contaminants are referred to as 'key contaminants' from this point.

Site CBI06 generally had the highest or equal highest concentrations of key contaminants across both organic and metal contaminant groups and CBI05 consistently had among the lowest (Figure 11). The trend in relative concentrations across sites was similar for each of the key metals (Figure 11a). For the key organic contaminants, similar trends were observed across sites for all the PAHs (Figure 11b). The OC pesticide dieldrin was the exception in terms of relative concentrations of organic contaminants across sites (Figure 11b), showing lower concentrations than other contaminants at CBI06 and relatively higher concentrations at sites CBI01 and CBI02. Acenapthene appeared to provide the greatest differentiation between site CBI06 and the other sites – demonstrated by the associated vector (Figure 10) and the relative concentrations of this contaminant (Figure 11b).

[5] Site CBI07 has some distinctly different physical characteristics from the other sites (in addition to the characteristics explored in Tables 5 – 10), for which there are no data to enter into further analyses. That is, CBI07 is located within a sheltered man-made cove that branches off the main estuary channel and is subject to less wave energy and flow than the other sites. Therefore it is not possible to separate cove effect from contaminant effect for this site.

[6] Fine sediments: clay and silt – particles ranging from 0.02 to 62 µm.

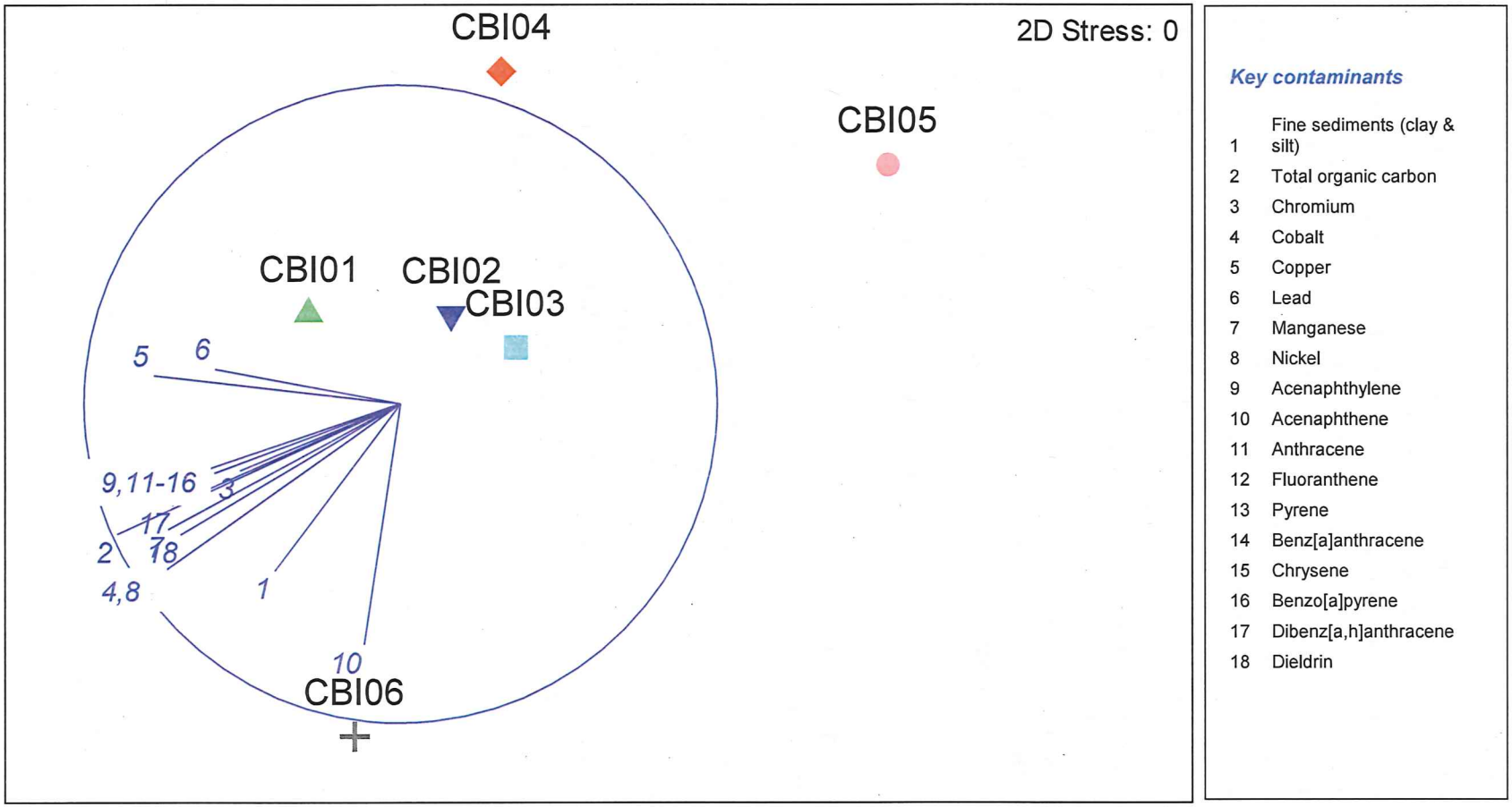


Figure 10 MDS ordination for species abundance (log (x+1) transformed) for sites CBI01 to CBI06 (averaged for site). Vectors show sediment contaminants as identified by BIOENV analyses to best explain the biotic assemblage patterns separating the sites (Spearman's rank correlation ≥ 0.8).

a)

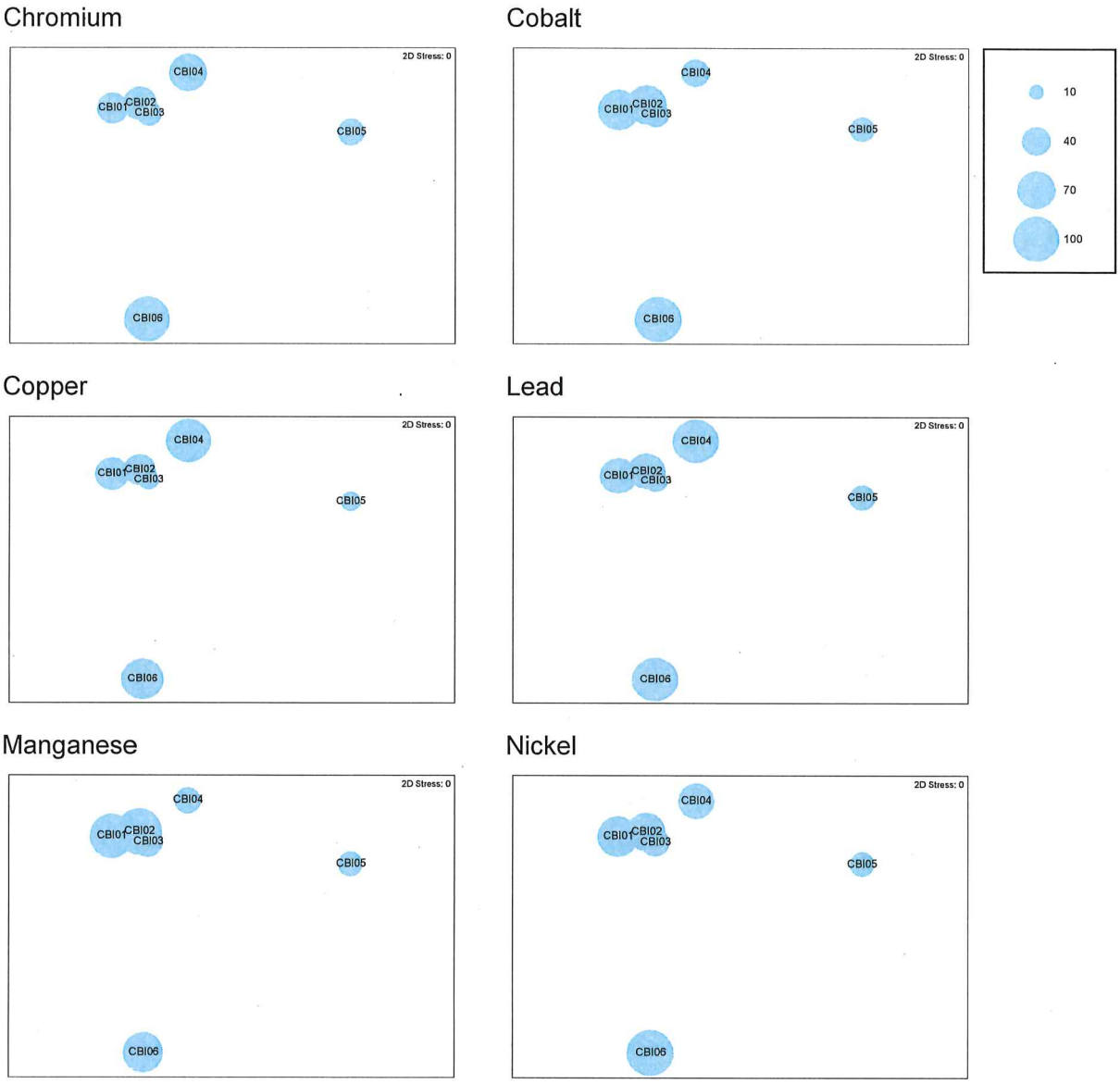
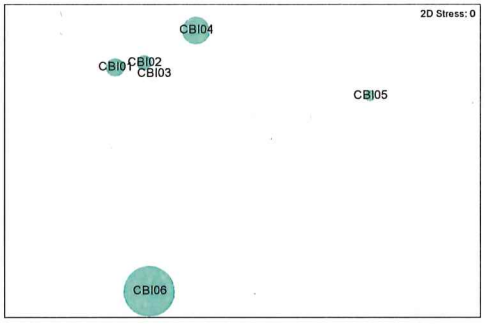


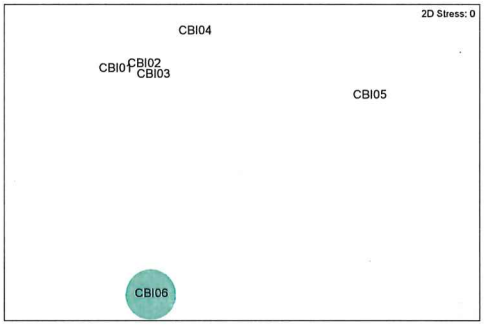
Figure 11 Bubble plots indicating relative concentrations (range standardised) of the contaminants that best explain the separation of sites (determined by BIOENV procedure using Spearman's rank correlation method – $R \geq 0.8$ for all contaminants shown). a) Bioavailable metals [this page]; b) Organic contaminants [following page].

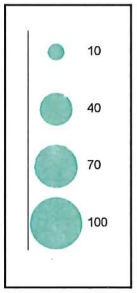
b)

Acenapthylene (PAH)

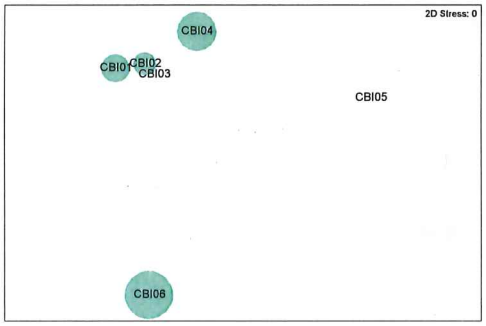


Acenapthene (PAH)

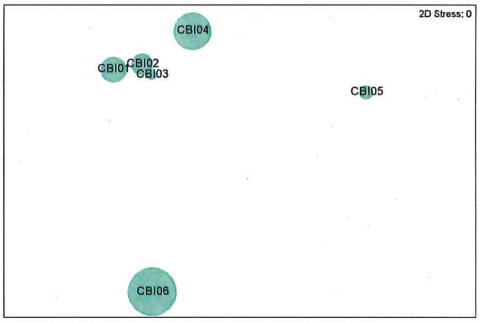




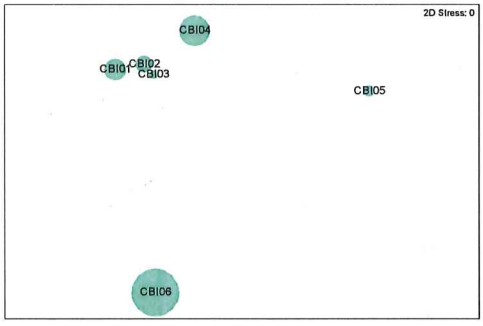
Anthracene (PAH)



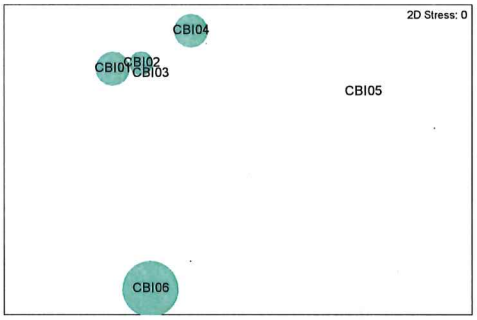
Fluoranthene, Chrysene, Benzo[a]pyrene and Benz[a]anthracene (PAHs)



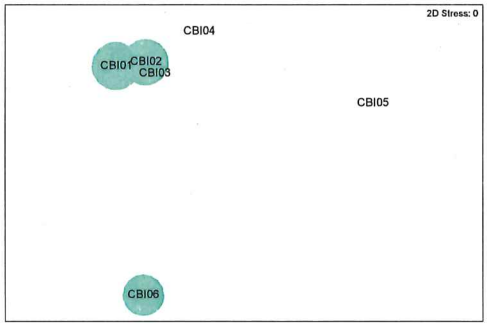
Pyrene (PAH)



Dibenz[a,h]anthracene (PAH)



Dieldrin (OC pesticide)



Relationship between biotic assemblages and environmental factors: site CBI07

Site CBI07 had a distinctly different biotic assemblage to the other sites (Figure 4 to Figure 6) and distinctly different contaminant composition (Table 5, Table 7; Figure 7 and Figure 9). In particular, the concentration of zinc was almost five times as high at site CBI07 compared with the others. Various pesticide and PCB contaminants showed similar trends. Furthermore, this site had the highest total organic carbon and the dominant sediment fraction was silt compared with fine or medium sand for all other sites (Table 9).

3.2 Comparison of current data (2011) with historic data (1997)

The 1997 dataset has been provided by K. Trayler from the *Swan River recolonization study 1997* (Trayler & McKernan 1997). Four sites were investigated in 1997 and the following section presents only 2011 data from the same four sites. Sites from the 2011 dataset have been classified as *reference* (REF) or *remediation* (REM) according to the 1997 study to enable comparisons between the two datasets.

There was an increase in mean total abundance and mean species richness across all sites from 1997 to 2011, except for mean total abundance at CBI02REF, where the opposite was seen (Table 11).

Table 11 Mean total abundance and mean species richness across sites from 1997 and 2011 datasets.

	Mean total abundance (SD)		**Mean species richness (SD)**	
Site	1997	2011	1997	2011
CBI02REF	1876 (345.5)	741.8 (201.5)	11.25 (1.0)	21.2 (4.7)
CBI03REM	134 (33.3)	567.2 (223.4)	7.25 (1.3)	18.4 (1.8)
CBI04REM	225.5 (57.5)	646.6 (259.8)	8 (1.4)	20.4 (1.9)
CBI05REF	283(62.1)	451 (223.4)	10 (0.8)	14.6 (1.8)

Assemblage composition in 1997 was dominated by annelids at the two remediated sites and crustaceans at the two reference sites. In 2011, the data were generally less variable and all four sites were dominated by molluscs. The relative contribution of each group to overall assemblage composition can be seen in Figure 12.

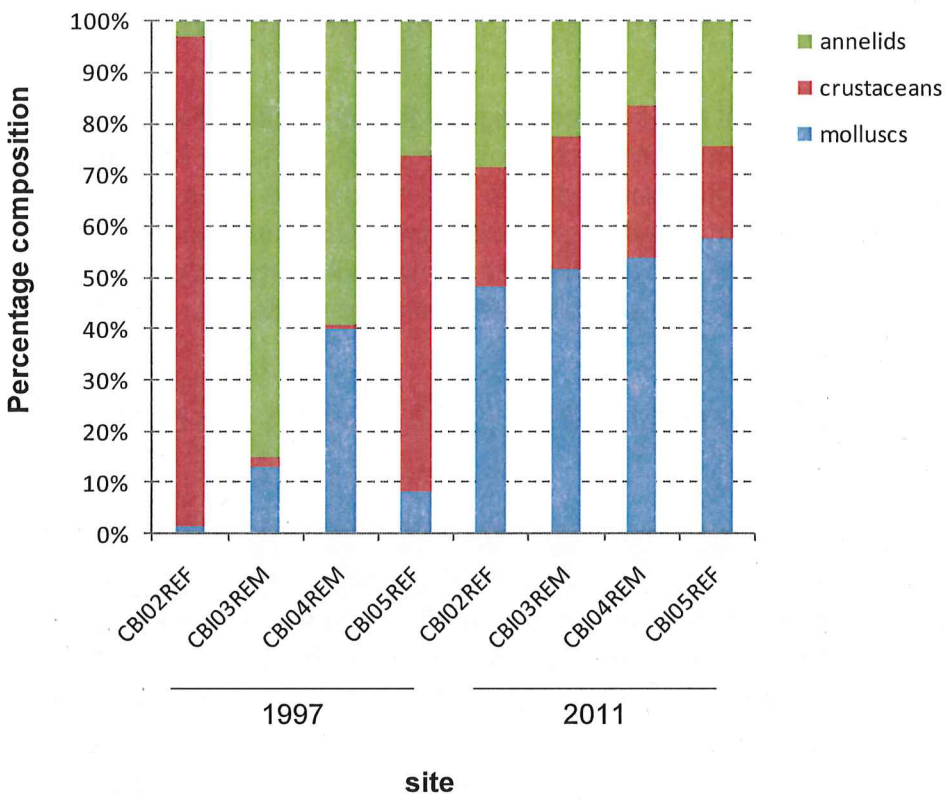


Figure 12 Percentage composition at four sites in 1997 and 2011. REF: reference site; REM: remediation site.

MDS ordination of species assemblages across all four sites in 1997 and 2011 showed three distinct clusters (Figure 13). ANOSIM showed the differences in species assemblage between sampling years to be highly significant ($R = 1$; $p = 0.001$). There were also significant differences between reference and remediation sites in both sampling years. However, the extent of the difference in 2011 was less than in 1997 ($R = 0.87$ and 0.66 for 1997 and 2011 respectively; $p = 0.001$).

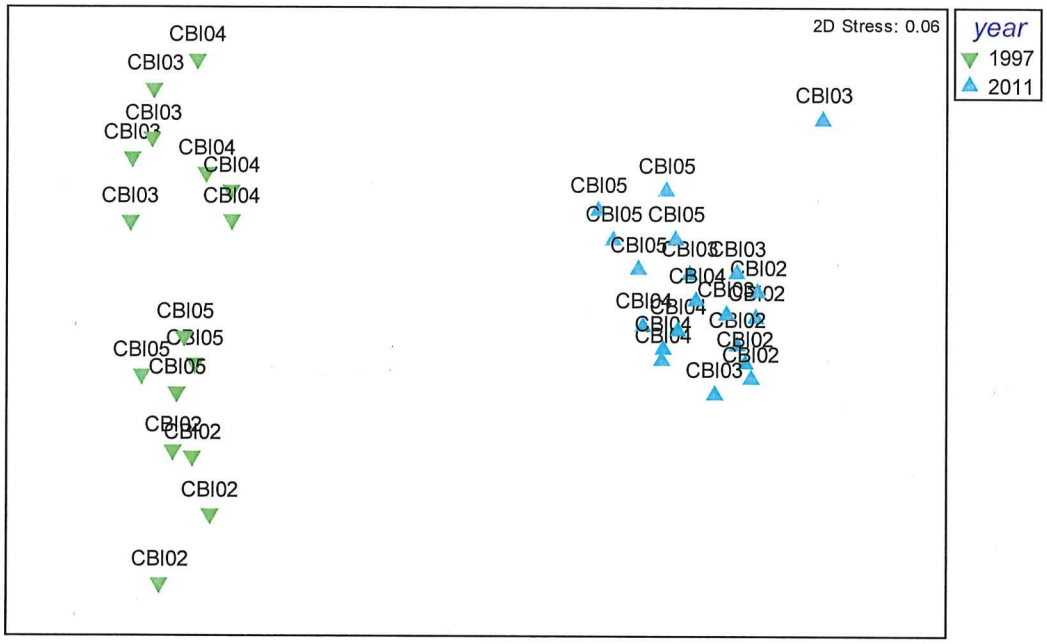


Figure 13 MDS ordination of species abundance. Reference sites: CBI02 and CBI05; remediation sites: CBI03 and CBI04.

The average dissimilarity between the 1997 and 2011 datasets was 72% (SIMPER). The most influential taxa separating the 1997 and 2011 datasets are shown in Table 12.

Table 12 Most influential taxa in separating 1997 and 2011 species assemblages as determined by SIMPER analyses

Taxa most responsible for separation of groups	Average % dissimilarity between groups	% contribution towards separation of groups	Observation (based on abundance)
Prionospio cirrifera (A)	6.29	8.77	1997 < 2011
Spisula trigonella (M)	5.51	7.69	1997 < 2011

Key: M: molluscs; A: annelids. Note: only species likely to be consistent discriminators of groups are listed (according to Clarke & Warwick 2001). The dissimilarity:standard deviation ratios are 5.95 and 6.53 for P. cirrifera and S. trigonella respectively.

Other key species responsible for the separation of the 1997 and 2011 datasets were the crustaceans *Cruranthura simplicia* and *Corophium minor,* and the molluscs *Sanguinolaria biradiata, Amygdalum glaberrima, Nassarius nigellus, Mytilid* sp. and *Theora* sp. – all of which were present in 2011 but absent from all sites in 1997. Conversely, the annelid *Melita matilda* was present in 1997 and absent from all sites in 2011.

4 Discussion

4.1 Comparison of biotic assemblages in 2011

All sites

When all sites were compared, the biotic assemblage at CBI07 (adjacent to Claisebrook Drain within Claisebrook Cove) was most different from all other sites. The separation of sites was demonstrated by MDS, highly significant differences and high R-statistics reported (ranging from 0.75 – 0.96, ANOSIM). Average dissimilarities between CBI07 and other sites ranged between 75 and 87% (SIMPER). This is not surprising given this site is subject to different physical characteristics such as flow regime (being the only site located within the cove) and substrate (having the highest proportion of fine sediments), as well as the significant contaminant burden (pesticides, PCBs and a range of metals higher than at other sites and at levels that exceeded guidelines indicating adverse biological effects) reported in this study and previously (Nice & Fisher 2011). Mean total abundance and mean species richness was lower at this site than all other sites. Comparatively low abundance and richness is not unusual for environments that have received urban and industrial contamination (e.g. Edgar & Barrett 2000). Abundance was orders of magnitude lower at CBI07 than all other sites investigated.

The most obvious difference in biotic assemblage at CBI07 was the absence of all crustaceans, a group generally considered to be sensitive to anthropogenic disturbance (Wildsmith et al. 2011). A comparatively large proportion (approximately half) of the fauna at CBI07 was composed of polychaete annelids, generally a very resilient group renowned for its tolerance to a range of environmental disturbances (Warwick & Clarke 1993; Gray et al. 2002) and many of which have a preference for fine sediments (e.g. *Heteromastus* sp., Fauchald & Bellan 2012). In particular, members of the *Capitella* genus – widely acknowledged as having high tolerance to contaminants in their application as pollution indicators (e.g. Kanandjembo et al. 2001; Mendez et al. 1998) – was the dominant annelid taxon at CBI07 (mean abundance of 11 per sample compared with all other annelids present in numbers ranging from <1 – 2.4).

The molluscs at CBI07 were dominated by *Arthritica semen* (approximately 18 per sample). Only two other mollusc species were represented (*Amygdalum glaberrima* and *Nassarius nigellus*), both with mean sample abundance values of one or less. While many molluscs are relatively sensitive, *Arthritica semen* is tolerant of variable environmental conditions and is particularly well adapted to tolerate rapid changes (Kanandjembo et al. 2001) and broad ranges (Wells & Threlfall 1982a) in salinity. Although salinity at the time of sampling was fairly constant between sites, CBI07 would likely be subject to freshwater flushing more often than the other sites due to its location adjacent to Claisebrook Drain, which discharges fresh water after significant rainfall. Another adaptive feature of *Arthritica semen* is its short lifecycle with continuous reproduction (voltinism) and rapid growth (Wells & Threlfall 1982b),

contributing to its success in challenging environments. Further, while most mollusc species produce pelagic larvae (Beesley et al. 1998), *Arthritica semen* females brood their eggs and larvae within the mantle cavity (Wells & Threlfall 1982b). This may provide protection from direct exposure to contaminants until a more mature and less sensitive developmental stage is reached. Larval molluscs have been shown to have specific windows in their development when exposure to organic contaminants is particularly critical (e.g. a brief exposure to a common organic contaminant at the pediveliger larval stage resulted in inhibition of settlement and metamorphosis and subsequent inability to develop beyond the pelagic larval stage – Nice et al. 2001). Thus species with a protective strategy, such as that of *Arthritica semen,* may be able to avoid such critical exposures to environmental contaminants. The sediments at site CBI07 were previously shown to be toxic to the pelagic larvae of the mussel *Mytilus edulis planulatus* (Nice & Fisher 2011), which may explain why mussels from the *Mytilus* genus were absent at this site yet present at all others investigated in this study.

In summary, fauna composition at site CBI07 was predominantly limited to two taxa: the annelid *Capitella* sp. and the mollusc *Arthritica semen*, both renowned for their tolerance to compromised environmental conditions. Other species found to be influential (SIMPER analyses) in separating CBI07 from other sites due to their absence or presence only in very low numbers at CBI07 were the molluscs *Spisula trigonella* and *Sanguinolaria biradiata*; the crustaceans *Cruranthura simplicia, Grandidierella propodentata* and *Corophium minor*; and the polychaetes *Prionospio cirrifera, Leitoscoloplos normalis* and *Ceratonereis aequietis*. A possible reason for the absence of *Ceratonereis aequietis* at CBI07 is it belongs to a genus with a lifecycle that requires the construction of tubes from sand particles to brood embryos and young (Hutchings & Glasby 1985). The sediment at this site had a dominant fraction of silt and comparatively fewer coarser grain particles, the materials typically required for this process.

As well as an extensive range of contaminants and a predominance of silt, CBI07 had the lowest dissolved oxygen concentration (3.6 mg/L) of all the sites (measured in the water column 5 – 20 cm above the sediment surface according to Simpson et al. 2005). This is considered poorly oxygenated for the Swan Estuary (Robb & Evans 2008) and it is likely that the sediment below (although not measured) was near anoxic. A low oxygen environment is an additional environmental pressure that may help explain the low species abundance and richness at this site.

Estuary sites

When only estuary sites were compared (i.e. CBI01 – CBI06), biotic assemblage at CBI06 was the most different from all other sites (MDS) with significant differences reported between CBI06 and all other sites (ANOSIM). Average dissimilarities between site CBI06 and other sites ranged between 38 and 50% (SIMPER). This separation of sites appeared to be driven by 16 key contaminants (in conjunction with the proportion of fine sediments and total organic carbon). The key contaminants

included several PAHs, with acenapthene showing the strongest correlation (BIOENV) due to its presence at site CBI06 and absence from all others. Furthermore, a peak in all the PAHs measured was recorded at CBI06 compared with all other sites (including CBI07 within the cove, discussed previously). Thus, it is not surprising that biotic assemblages are different at CBI06, given the toxicity of the various PAH compounds (many of which exceeded guidelines) which is widely reported in the scientific literature (e.g. Laughlin & Neff 1979; Kukkonen & Landrum 1994; Landrum et al. 1994; Fleeger & Lotufo 1999).

BIOENV also determined the pesticide dieldrin and a range of metals among those contaminants that best explained the separation of the estuary sites, with concentrations typically higher at site CBI06 (bubble plots). The generally low (or absent) concentrations of these contaminants at site CBI05 appeared to be responsible for its separation from the other estuary sites.

Species richness was not significantly different between any estuary sites although composition was markedly different at CBI06 when compared with the others – indicated *inter alia* by the larger proportion of arthropod crustaceans compared with all other sites and the presence of chordates (albeit only one or two individuals), which were absent at other sites.

While not the dominant sediment fraction for any of the estuary sites, the proportion of fine sediment (clay and silt) was highest at CBI06, providing a greater binding potential for contaminants such as PAHs due to the greater surface area and higher number of binding sites (Simpson et al. 2005). Fine sediments can also offer a greater potential for certain organisms to be smothered (Kerr 1995), although this is unlikely to be the case at any of the estuary sites CBI01 to CBI06, given the relatively high proportion of sand and, to a lesser extent, gravel mixed in with the fine sediments at these sites. Interestingly, while polychaete annelids are generally considered to be fairly resilient to environmental disturbance (Reise 1982; Warwick & Clarke 1993) and generally have a preference for fine sediments as discussed previously, a smaller proportion of these organisms occurred at CBI06 than might be expected. This may be due to the PAH contaminants, given that a sensitivity to the PAH fluoranthene, for example, has been demonstrated (Weinstein & Sanger 2003). This site had the highest concentrations of each of the PAHs (except phenanthrene) for all the sites investigated (including CBI07 within Claisebrook Cove). The PCB mixture Arochlor 1254 was also reported at CBI06 and may also be influencing the biotic composition given its well-reported toxicity (e.g. Nimmo et al. 1975).

With regard to biota, the crustacean *Cruranthura simplicia* was among the species most influential in the separation of sites. Densities were greater at site CBI06 than all other sites. In fact, this is the only species that was greater at site CBI06 from those identified by SIMPER as the key species in the separation of site CBI06 from the other sites. *Cruranthura simplicia* is an isopod crustacean, a group recognised as pollution tolerant, the species within it often used as indicators of aquatic pollution (Rinderhagen et al. 2000). The only other isopod crustacean recorded in this study (*Sphaeromatid* sp.) was also found at CBI06 and was not present at any other site.

Other species found to be influential (SIMPER analyses) in separating site CBI06 from other estuary sites due to their absence or presence only in very low numbers at CBI06 were the molluscs *Spisula trigonella, Arthritica semen* and *Sanguinolaria biradiata*; and the annelid polychaetes *Capitella* sp. and *Leitoscoloplos normalis.*

The separation of estuary sites (CBI01 – CBI06) did not appear to be driven by temperature, salinity, pH or dissolved oxygen, which were relatively consistent between sites at the time of sampling.

The Groundwater Interception Drain

One of the specific aims of this investigation was to determine whether the biotic assemblage adjacent to the GID outfall (site CBI03) was different from the other sites investigated, and whether any observed differences could be linked to the presence of contaminants being discharged from the GID. Multivariate analyses showed no clear distinction in biotic assemblage between site CBI03 and both CBI01 and CBI02 upstream (MDS, ANOSIM). Simple community measures such as abundance and species richness also showed no difference at site CBI03 when compared with other sites in the estuary and the three major taxa (molluscs, crustaceans and annelids) were represented in approximately the same proportions as at CBI01 through CBI05.

The GID discharges to the estuary intermittently and it was not possible to determine whether it was flowing at the time of sampling because the outfall was inundated with estuarine water. The potential absence of an acute stressor may explain the lack of observed impact. However, regardless of flow at the time of sampling, if the contaminants reported in the GID water (ENV 2009) are affecting benthic macroinvertebrate communities at this site on an ongoing basis, it is likely a contaminant signature would be evident in the benthic sediments. Contaminants were present in the sediments (top 15 cm) at the GID site (CBI03) but were relatively low in concentration – which was not necessarily attributable to sediment type at this site, given that other sites with a similar sediment profile (and hence binding capacity for contaminants) such as CBI04 had markedly higher contaminant levels. Compared with other sites examined, no pesticides or PCBs were detected at CBI03 and while PAHs and metals were present, no guidelines were exceeded; and the concentrations were among the lowest for all sites investigated in this study.

Relatively low contaminant concentrations at this site were also observed in the parallel ecotoxicological investigation (Nice 2013), which assessed toxicity and contaminant concentrations in the surficial (top 2 cm) sediments. While metals, PAHs and OC pesticides were detected in the parallel study, only one guideline was exceeded (OC pesticide, dieldrin) and toxicity was relatively low – only being reported for a fish test and not for the three macroinvertebrate tests employed. Thus, it is suggested that contaminants discharged from the GID are not likely to be accumulating in the sediment at this site to concentrations that significantly affect macroinvertebrate assemblages. This does not negate the possibility that contaminant spikes may be discharged intermittently (relatively high concentrations

of PAHs have historically been reported in the discharge water of the GID – ENV 2009), which may result in an acute impact on the local benthic macroinvertebrate fauna, or indeed, the pelagic species in the area. However, if such contaminant spikes have occurred it may be assumed that either enough time had passed to allow the macroinvertebrate communities in the immediate vicinity to the outfall to re-establish before the sediments were sampled for this investigation in autumn 2011, or that any contaminants discharged from the GID are accumulating at a potential deposition site further downstream. Nevertheless, no data presented here suggest the GID outfall was shaping the population at CBI03 at the time of sampling (especially since assemblages were statistically 'the same' as at sites CBI01 and CBI02 located 500 m and 1 km upstream). This does not suggest that populations present in the vicinity of the GID outfall are 'healthy' (e.g. compared with a near-pristine national park type of environment), rather that the assemblages present at the GID site are typical of an urbanised estuary.

A baseline dataset now exists from this GID site (this study), such that future comparisons can be made in the event that the flow regime of the GID (quality and/or quantity) were to change.

4.2 Comparison of 1997 and 2011 biotic assemblages at remediated and reference sites

Abundance and richness were higher at remediation sites in 2011 (this study) than in 1997 (Trayler & McKernan 1997). Abundance and richness at the same remediation sites in 1997 (Trayler & McKernan 1997) were higher than in 1996 (Bouckaert 1996). This would indicate succession in the area since remediation in 1994 when the remediation sites were considered to be devoid of sediment macroinvertebrates[7]. However, richness has also increased at the reference sites over time, suggesting the biotic assemblages at these sites were not particularly stable in 1996 and 1997. This is not surprising considering the level of disturbance due to extensive foreshore development in the general area around this time. Given that increased complexity is typically indicative of a healthier system, and that low diversity systems are often a function of environmental degradation (Hughes 2010), this may indicate improving health (to a certain degree) at all four sites since 1996.

Biotic assemblages at remediation and reference sites were more similar to each other in 2011 than they were in 1997. However, biotic assemblages at both reference and remediation sites in 2011 were strongly departed from both reference and remediation assemblages in 1997. That is, there was no evidence that biotic assemblages at sites in 2011 had become more like those of the reference sites as expressed in 1997 (according to Trayler & McKernan 1997). Rather, the entire assemblage set was departed from the 1997 status. The separation between 1997 and 2011 biotic assemblages was strong (ANOSIM), with an average dissimilarity of

[7] No biotic assemblage data exist either pre or immediately post remediation. However, sediment (to a depth of up to 1m in places) was replaced with cleanfill as part of the remediation process in 1994 (CMPS & F Pty Ltd 1996).

72% (SIMPER) – largely due to 16 species out of 31 present in 2011 being absent in 1997. Conversely, three species were present in 1997 but absent in 2011. The mollusc *Spisula trigonella* was one of the most influential species in separating the 1997 and 2011 datasets (SIMPER). It was present in large numbers in 2011 compared with only one or two individuals in 1997, yet present in large numbers at one of the remediation sites during the same season in 1996 (Bouckaert 1996). Conversely, the annelid polychaete *Prionospio cirrifera*, another key species in the separation of 1997 and 2011 datasets due to its absence in 1997, was also absent in 1996 (Bouckaert 1996). This strong departure for the whole dataset (reference and remediation sites) from the 1997 status indicates that factors other than general 'recovery over time' appear to be influencing the separation of sites. This is not surprising given that 14 years have passed since the previous dataset was collected and considering the spatial and temporal variability of macroinvertebrate data. The current (2011) biotic assemblages at all four sites is likely a product of a complex interplay of factors including but not limited to changes in catchment land use, dredging and development, introduction or reduction of contaminants, increasing eutrophication, introduction of invasive species to the foodweb and La Nina events <www.bom.gov.au/lam/climate/> that have altered rainfall, tidal and water temperature patterns in recent years.

5 Conclusions

This study found no measurable impact on benthic macroinvertebrate communities that could be attributed to the GID, based on information collected in March 2011. A baseline dataset now exists for this and surrounding sites that may assist in determining future changes to macroinvertebrate communities in relation to alterations in GID discharge regime (quality and quantity).

Biotic assemblages at site CBI07 (Claisebrook Drain discharging within the cove) were very different to biotic assemblages at all other sites. This was likely attributable to different physical characteristics of this site (e.g. flow regime and substrate) in conjunction with the presence of sediment contaminants (PAHs, OC pesticides, PCBs and metals) known to exist at this site.

An unexpected finding of this study was the difference in biotic assemblages at site CBI06 compared with the other estuary sites. This was strongly attributable to the sediment contaminant concentrations at this site, particularly the PAHs. These PAH contaminants would likely be acting in conjunction with the sediment type and the pesticides, PCBs and metals measured here (and other contaminants that may be present but not targeted by this study). In particular the PAH contaminant levels at this site were significantly higher than those reported at sites within Claisebrook Cove and adjacent to the GID outfall, which may suggest that site CBI06 represents a downstream deposition site for contaminants from the Claisebrook area; or that an additional contaminant source(s) to this region of the upper Swan Estuary exists. Such potential sources may include any one of a number of stormwater drains or historic contaminated sites in the vicinity. Furthermore, the contaminants present at site CBI06 may represent an historic contaminant signature in the sediments rather than an ongoing source (or a combination of the two). Whatever the source or combination thereof, contaminants such as PAHs may remain bioactive in estuarine sediments for extended periods (e.g. Homebush Bay, Parramatta River, NSW, GHD 2009).

Finally, benthic macroinvertebrate communities at sites CBI02, CBI03, CBI04 and CBI05 had changed significantly between 1997 (after remediation occurred) and 2011. While increasing richness, for example, experienced across all sites may suggest increasing health in the area, it is not possible to link the changes explicitly to recovery due to remediation. This is particularly so, given the absence of a baseline dataset collected prior to remediation; and the same trends (e.g. increasing richness) being seen at the reference sites. While there is no evidence of declining health at sites CBI02, CBI03, CBI04 and CBI05, considering the significant timeframe that has passed since previous monitoring, it is concluded that the change reported at these sites (from 1997 to 2011) is the product of a complex interplay of factors. Such factors include changes in land use adjacent to the estuary, introduction of invasive species, changes in contaminant levels and eutrophication of the system, and the effects of La Nina and its impact on weather and tidal patterns between the two sampling periods.

6 Recommendations

It is recommended that:

1. As a priority, site CBI06 (adjacent to Point Fraser wetland and Heirisson Island) should be investigated in relation to the peak in contaminants reported here and the associated different macroinvertebrate assemblage. The investigation should assess the extent of contamination in this area and attempt to establish likely sources. Where guidelines are exceeded, toxicity assessment (following the methods of Nice 2013) should be conducted to determine whether the sediments are toxic to aquatic organisms and therefore likely to affect ecosystem health. Investigation of this site is particularly important given the planned development and subsequent disturbance to this area of the Swan Estuary.
2. Should there be any significant change in GID flow regime (quality and/or quantity), benthic macroinvertebrate investigations (with supporting sediment chemistry) should be repeated to determine any departure from the current (baseline) position.
3. If conclusions are required for recovery over time in relation to specific management interventions, benthic macroinvertebrate investigations should include sufficient temporal replication (ideally more frequently than 14 years in the last instance) so that confounding factors may be reduced.

7 Appendices

7.1 Appendix A – Organic datasets normalised to 1% organic carbon

Table 13 Sediment polycyclic aromatic hydrocarbon (PAH) concentrations (normalised to 1% organic carbon)

	Sediment PAH concentrations (µg/kg) dry weight														
Site	Naphthalene	Acenaphthylene	Acenaphthene	Fluorene	Phenanthrene	Anthracene	Fluoranthene	Pyrene	Benz[a]anthracene	Chrysene	Benzo[b+k]fluoranthene *	Benzo[a]pyrene	Indeno[1,2,3,c,d]pyrene	Dibenz[a,h]anthracene	Benzo[g,h,i]perylene
CBI01	n.d.	12	n.d.	n.d.	12	12	56	80	44	40	72	56	24	8	32
CBI02	n.d.	13	n.d.	n.d.	13	13	63	75	38	38	69	50	25	6	31
CBI03	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	33	49	16	16	33	33	n.d.	n.d.	16
CBI04	n.d.	50	n.d.	n.d.	86	43	214	293	129	100	164	136	50	14	57
CBI05	n.d.	33	n.d.	n.d.	n.d.	n.d.	133	167	100	67	167	100	67	n.d.	67
CBI06	n.d.	126	7	n.d.	34	47	263	529	237	171	318	295	100	29	118
CBI07	n.d.	6	n.d.	n.d.	15	7	45	40	26	20	44	25	12	n.d.	14
ISQG Low	160	44	16	19	240	85	600	665	261	384	n.a.	430	n.a.	63	n.a.
ISQG High	2100	640	500	540	1500	1100	5100	2600	1600	2800	n.a.	1600	n.a.	260	n.a.

*ISQG = Interim Sediment Quality Guideline (ANZECC & ARMCANZ 2000); blue indicates low ISQG exceeded; red indicates high ISQG exceeded; n.a. = no ANZECC & ARCANZ guideline available; * alternative guidelines for benzo[b+k]fluoranthene of 240 and 1340000 µg/kg (Ontario Sediment Quality Guidelines 1993 lowest effect level and severe effect level respectively). N.d. = not detected; limit of reporting: 10 µg/kg. Samples comprised the top 15 cm of the sediment.*

Table 14 Sediment pesticide and polychlorinated biphenyl (PCB) concentrations (normalised to 1% organic carbon)

	Sediment pesticide and PCB concentrations (µg/kg) dry weight						
Site	Aldrin*	trans-Chlordane**	Dieldrin	p,p'-DDT^{+}	p,p'-DDE	p,p'-DDD^{++}	Aroclor 1254^
CBI01	n.d.	0.5	1.68	0.7	5.6	3.0	n.d.
CBI02	n.d.	n.d.	2.44	n.d.	4.6	2.2	n.d.
CBI03	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
CBI04	0.9	1.4	n.d.	0.9	6.9	1.7	23.0
CBI05	n.d.	n.d.	n.d.	n.d.	5.0	n.d.	n.d.
CBI06	n.d.	n.d.	1.63	n.d.	2.8	1.8	6.3
CBI07	2.5	1.64	n.d.	0.6	4.3	1.3	6.9
ISQG Low	n.a.	0.5**	0.02	1.6^{+}	2.2	2***	23^{++}
ISQG High	n.a	6.0**	8	46^{+}	27	20***	n.a.$^{++}$

*Note: only those parameters detected are shown in this table. Refer to Table 1 for full list of contaminants. ISQG = Interim Sediment Quality Guideline (ANZECC & ARMCANZ 2000); blue indicates low ISQG exceeded; red indicates high ISQG exceeded; n.a. = no ANZECC & ARMCANZ guideline available. * alternative guideline for aldrin of 2 µg/kg (Canadian SQG 2002). ** trigger values quoted for trans-chlordane are for chlordane. $^{+}$trigger values quoted for p,p'DDT are for total DDT (only measured p,p'-DDT in this study). $^{++}$ trigger values quoted for p,p'-DDD are for total p,p'-DDD and op-DDD (only p,p'-DDD was measured in this study). ^ trigger value quoted for Aroclor 1254 is for total PCBs. N.d. = not detected; limit of reporting: 1 µg/kg for pesticides and 10 µg/kg for Aroclor mixtures. Samples comprised the top 15 cm of the sediment.*

8 Shortened forms

AHPA	American Public Health Association
ANZECC	Australia and New Zealand Environment and Conservation Council
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
DDD	Dichlorodiphenyldichloroethane
DDE	Dichlorodiphenyldichloroethylene
DDT	Dichlorodiphenyltrichloroethane
EPA	Environmental Protection Authority
EPRA	East Perth Redevelopment Authority
ESA	Ecotox Services Australasia
GC-MS	Gas chromatography-mass spectrometry
GID	Groundwater Interception Drain
MAFF	Ministry of Agriculture Fisheries and Food (UK)
OC	Organochlorine
PAH	Polycyclic aromatic hydrocarbon
PCB	Polychlorinated biphenyl
PSEP	Puget Sound Estuary Program
SRRC	Swan River Reference Committee
SRT	Swan River Trust
WFPHA	World Federation of Public Health Associations
WHO	World Health Organization

References

ANZECC & ARMCANZ 2000, *Australian and New Zealand guidelines for fresh and marine water quality*, Australia and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand.

American Public Health Association 1998, *Standard methods for the examination of water and wastewater, 20th Edition*, American Public Health Association, Water Environment Federation, American Water Works Association, Washington, US.

Axis Environmental 1996, *Quality assurance report, open space and foreshore zones, East Perth Gasworks remediation*, prepared for East Perth Redevelopment Authority, March 1996, Axis Environmental Consultants Pty Ltd, 7 pp.

Beesley, PL, Ross, GJB & Wells, A (eds) 1998, *Mollusca: the southern synthesis, Fauna of Australia, Vol 5,* CSIRO Publishing, Melbourne. Australia.

Bouckaert, F 1996*, Swan River recolonization study,* 15 pp.

Bowman Bishaw Gorham 1992, *Environmental assessment of polyaromatic hydrocarbon contamination of river sediments near the East Perth Gasworks*, Technical report RI1140.

Canadian Sediment Quality Guidelines 2002, *Canadian Council of Ministers of the Environment (2002) Canadian Sediment Quality guidelines for the protection of aquatic life*, retrieved October 2012. <www.ccme.ca/assets/pdf/sedqg_summary_table.pdf>

Chapman, PM, Anderson, BS, Carr, RS, Engle, V, Haverland, P, Hameedi, J, Hyland, J, Ingersool, C, Long, E, Rodgers, J, Sibley, S, Smith, P, Swartz, R, Thompson, B & Windom, H 1997, 'General guidelines for using the sediment quality triad', *Marine Pollution Bulletin*, 34: 368–72.

Clarke, KR & Warwick, RM 2001, *Change in marine communities: an approach to statistical analysis and interpretation*, 2nd edition PRIMER-E: Plymouth.

CMPS & F Pty Ltd 1996, *East Perth Gasworks Swan River Remediation Environmental Management Program*, report for Department of Environmental Protection and Swan River Trust, April 1996, 9 pp.

East Perth Redevelopment Authority 2009, *Claisebrook Cove, East Perth, Fact sheet no. 1. quick facts.*

Edgar, GJ & Barrett, NS 2000, 'Effects of catchment activities on macrofaunal assemblages in Tasmanian estuaries', *Estuarine Coastal Shelf Sci.* 50, 639–654.

ENV 2009, *Former East Perth Gasworks/Mardalup Park, detailed investigation of effective operation of containment system*, report no: RP001, ENV Australia.

Environmental Protection Authority 1992, *Contamination management strategy for East Perth Gasworks site and adjacent areas of the Swan River, State Energy Commission of Western Australia (SECWA),* report and recommendations of the Environmental Protection Authority, Perth, WA, Bulletin 651.

Fauchald, K & Bellan, G 2012, World Polychaete Database, accessed through World Register of Marine Species at http://www.marinespecies.org/aphia/php?p=taxdetails&id=129884

Fisher, SJ in prep. Investigation of the Groundwater Interception Drain at Claisebrook using passive sampler technology, Department of Water report for Swan River Trust, Western Australia.

Fleeger, JW & Lotufo, GR 1999, *Development and application of a sublethal toxicity test to PAH using marine harpacticoid copepods*, US Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, report no: 72.12/2:99-0001, 38 pp.

GHD 2009, Homebush Bay West Contamination Assessment Sediment Investigation, report no: 21/18600/155760, 34 pp.

Gray, JS, Wu, RSS & Or, YY 2002, 'Effects of hypoxia and organic enrichment on the coastal marine environment', *Marine Ecology Progress Series* 238, 249–279.

Hughes, A 2010, 'Disturbance and diversity: an ecological chicken and egg problem', *Nature Education Knowledge* 1(8): 26.

Hutchings, PA & Gasby, CA, 1985, 'Additional nereidids (Polychaeta) from Eastern Australia, together with a redescription of *Namanereis quadraticeps* (Gay) and the synonymising of *Ceratonereis pseudoerythraeensis* Hutchings & Turvey with C. *Aequisetis* (Augener)', *Records of The Australian Museum* 37(2):101–110, Australian Museum, Sydney, NSW.

Kanandjembo, AN, Plattell, ME & Potter, IC 2001, 'The benthic macroinvertebrate community of the upper reaches of an Australian estuary that undergoes marked seasonal changes in hydrology', *Hydrological Processes*, 15(13): 2481–2501.

Kerr, SJ 1995, *Silt, turbidity and suspended sediments in the aquatic environment: an annotated bibliography and literature review*, Ontario Ministry of Natural Resources, Southern Region Science and Technology Transfer Unit technical report TR-008, 277pp. ISBN 0-7778-4437-0.

Kukkonen, J & Landrum, PF 1994, 'Toxico-kinetics and toxicity of sediment-associated pyrene to *Lumbriculus variegates* (Oligochaeta)', *Environ. Toxicol. Chem.* 13(9): 1457–1468.

Landrum, PF, Dupuis, WS & Kukkonen, J 1994, 'Toxicokinetics and toxicity of sediment-associated pyrene and phenanthrene in iporeia spp.: examination of equilibrium-partitioning theory and residue-based effects for assessing hazard', *Environ Toxicol Chem*. 3: 1769–178

Laughlin, RB Jr & Neff, JM 1979, 'Interactive effects of salinity, temperature and polycyclic aromatic hydrocarbons on the survival and development rate of larvae of the mud crab *Rhithropanopeus harrisii*', *Mar Biol* 53: 281–29

Lemly, AD 1996, 'Assessing the toxic threat of selenium to fish and aquatic birds', *Environmental Monitoring and Assessment*, vol. 43, pp. 19–35.

Mendez, N, Flos, J & Romero, J 1998, 'Littoral soft-bottom polychaete communities in a pollution gradient in front of Barcelona (Western Mediterranean, Spain)', *Bulletin of Marine Science* 63: 167–178.

Michelsen, TC 1992, *Organic carbon normalization of sediment data*, Washington Department of Ecology, Sediment Management Unit, report 05-09-050.15 pp.

Mudroch, A, Azcue, JM & Mudroch, P 1997, *Manual of physico-chemical analysis of aquatic sediments, Environmental science – environmental chemistry and toxicology*, Lewis Publishers, Florida.

Nice, HE, Crane, M, Morritt, D & Thorndyke, M 2001, 'Endocrine disrupters, critical windows and developmental success in oyster larvae', in *Environment and animal development – genes, life histories and plasticity*', Eds D. Atkinson and M. Thorndyke, BIOS Scientific Publishers Ltd. Oxford, UK. ISBN 1 85996 184 3.

Nice, HE 2009, *A baseline study of contaminants in the sediments of the Swan and Canning estuaries,* Water Science Technical Series, report no. 6, Department of Water, Western Australia.

Nice, HE & Fisher, SJ 2011, *Ecotoxicological and bioaccumulation investigations of the Swan Estuary in the vicinity of Claisebrook,* Water Science Technical Series, report no. 28, Department of Water, Western Australia.

Nice, HE 2013, *Ecotoxicological investigation of the Groundwater Interception Drain at Claisebrook in the Swan Estuary,* Water Science Technical Series, report no. 64, Department of Water, Western Australia.

Nimmo, DR, Hansen, DJ, Couch, JA, Cooley, NR, Parrish, PR & Lowe, JI 1975, 'Toxicity of Aroclo-R 1254 and its physiological activity in several estuarine organisms', *Archives of Environmental Contamination and Toxicology* 3(1): 22–39.

Ontario Ministry of Environment and Energy 1993, *Guidelines for the protection and management of aquatic sediment quality in Ontario*, retrieved February 2008, <www.ene.gov.on.ca/envision/gp/B1-3pdf>.

Reise, K 1982, 'Long-term changes in the macrobenthic invertebrate fauna of the Wadden Sea – are polychaetes about to take over?' *Netherlands Journal of Sea Research* 16, 29–36.

Rinderhagen, M, Ritterhoff, J & Zauke, G 2000, 'Crustaceans as bioindicators', in *Biomonitoring of polluted water – reviews on actual topics* (ed: A. Gerhardt), Trans Tech Publications – Scitech Publications, Environmental Research Forum, vol. 9, 2000: 161–194.

Robb, M & Evans, S 2008, *Oxygenating the Upper Swan River Estuary, feasibility study*, Department of Water, Perth, Western Australia.

Simpson, SL, Batley, GE, Charton, AA, Stauber, JL, King, CK, Chapman, JC, Hyne, RV, Gale, SA, Roach, AC & Maher, WA 2005, *Handbook for sediment quality assessment*, CSIRO, Bangor, New South Wales.

Standards Australia 1997, *Australian Standard guide to the sampling and investigation of potentially contaminated soil: Part 1: non-volatile and semi-volatile compounds, AS4482.1-1997*, Sydney, Australia.

Tingay & Associates 1994, *Swan River remediation environmental management plan,* report 93/75.

Trayler, K & McKernan, I 1997, *Swan River recolonization study 1997, a report for CMPS & F on the recolonization of benthic fauna in the remediation area of the Swan River adjacent to the former East Perth Gasworks,* 16pp.

Warwick, RM & Clarke, KR 1993, 'Comparing the severity of disturbance: a meta analysis of marine macrobenthic community data', *Marine Ecology Progress Series* 92, 221–231.

Weinstein, JE & Sanger, DM 2003, 'Comparative tolerance of two estuarine annelids to fluoranthene under normoxic and moderately hypoxic conditions', *Marine Environmental Research*, 56(5): 637–648.

Wells, FE & Threlfall, TJ 1982a, 'Salinity and temperature tolerance of *Hydrococcus brazieri* (Woods, 1876) and *Arthritica semen* (Menke, 1843) from the Peel-Harvey estuarine system, Western Australia', *Journal of the Malacological Society of Australia* 5: 151–156.

—1982b, 'Reproductive strategies of *Hydrococcus brazier* (Woods, 1876) and *Arthritica semen* (Menke, 1843) in Peel Inlet, Western Australia', *Journal of the Malacological Society of Australia* 5: 157–166.

Wentworth, CK 1922, 'A scale of grade and class terms for clastic sediments', *J. Geology*, v. 30, 377–392.

Wildsmith, MD, Rose, TH, Potter, IC, Warwick, RM & Clarke, KR 2011, 'Benthic macroinvertebrates as indicators of environmental deterioration in a large microtidal estuary', *Marine Pollution Bulletin*, 62: 525–538.