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SWAN RIVER TRUST

Ecotoxicological investigation of the Groundwater Interception Drain outfall at Claisebrook in the Swan Estuary

November 2013

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Department of Water Technical Report prepared for the Swan River Trust

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

For more information about this report, contact the Swan River Trust at info@swanrivertrust.wa.gov.au.

Disclaimer

Swan River Trust

The Swan River Trust commissioned the Department of Water to undertake this investigation as part of Phase III of the Non-Nutrient Contaminant Program (NNCP). The sampling design and methods were developed by the Department of Water in consultation with the Swan River Trust and are consistent with previous investigations undertaken in phases I and II of the NNCP.

Department of Water

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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 in the vicinity of 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*, Department of Water Technical Report prepared for the Swan River Trust, Western Australia, Nice HE 2013.

4 *Ecotoxicological investigation of the Groundwater Interception Drain outfall at Claisebrook in the Swan Estuary*, Department of Water Technical Report prepared for the Swan River Trust, Western Australia, Nice HE 2013. **[This report]**

5 *Investigation of polychlorinated biphenyls and other contaminants in the waters of the Swan Canning estuary using passive sampler technology*. Department of Water Technical Report prepared for the Swan River Trust, Fisher SJ 2013.

6 *Claisebrook in the Swan Estuary, Western Australia – A synthesis of environmental information and historical retrospective.* Department of Water Technical Report prepared for the Swan River Trust, Nice HE 2013.

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Summary

This study, 'Ecotoxicological Investigations of the Groundwater Interception Drain (GID) outfall at Claisebrook in the Swan Estuary' was conducted to determine whether sediments collected adjacent to the GID outfall to the Swan Estuary were toxic to aquatic organisms, whether sediment toxicity was greater at the GID outfall than at other sites in the region and whether a toxicity gradient existed away from the GID outfall.

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, the site of the former East Perth Gasworks (Nice 2009; Nice & Fisher 2011). Sediments were toxic to a range of aquatic organisms 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[1], was recently shown to be discharging contaminants including polycyclic aromatic hydrocarbons (PAHs) directly to the Swan Estuary (ENV 2009). In 2011, 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 had 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 in the vicinity of Claisebrook.

This report presents the second component (the sediment toxicity assessment), in which sediment samples were collected from 11 sites within the Swan Estuary for both toxicity and chemical testing. The toxicity assessment included four test taxa representative of south-west Western Australian estuaries: mussel (*Mytilus galloprovincialis*), amphipod (*Grandidierella japonica*), copepod (*Gladioferans imparipes*) and fish (*Pagrus auratus* and *Seriola lalandi*). These test organisms were exposed in the laboratory to field-collected sediment samples. The chemical assessment targeted those contaminant groups previously demonstrated to be present at levels of concern in sediments collected from the Claisebrook area (Nice 2009; Nice & Fisher 2011): PAHs, organochlorine (OC) pesticides and metals.

In summary, this study found that:

- although toxicity was experienced and contaminants were present in the sediments collected from the GID outfall site (CBE05), the level of toxicity and

[1] The gasworks ceased operating in 1971 and the GID was constructed in the 1990s as part of the remediation process associated with the site. The main outfall to the GID discharges to the estuary at the northern boundary of the historic contaminated site, Mardalup Park. This outfall is the focus of this investigation. The GID may also discharge to Claisebrook Cove, although it is unclear how regularly this occurs. A previous study (Nice & Fisher 2011) showed sediments collected within the vicinity of the GID outfall to Claisebrook Cove to be toxic to fish larvae.

contaminant concentrations were comparatively low when compared with a number of other sites,

- at the time of sampling, there was no clear gradient in either sediment toxicity or sediment contamination away from the GID outfall site (CBE05),
- sediment toxicity was greatest with a concomitant peak in PAH concentrations at a site adjacent to the middle section of Mardalup Park (CBE07).

It was concluded that the primary source of the current PAH contamination in the estuary sediments at site CBE07 is most likely the historic East Perth Gasworks site, either a) from residual contamination of estuarine sediments, or b) through PAH-contaminated groundwater that exists at Mardalup Park (ENV 2009); or a combination of both. This conclusion is based on the data presented in this study, considered in conjunction with the findings of previous studies (Nice 2009; Nice & Fisher 2011) and what is known of the history (e.g. Bowman Bishaw Gorham 1992; EPA 1992a; EPA 1992b) and current status (ENV 2009) of Mardalup Park and the adjacent estuary.

Other potential contaminant sources in the area include the Burswood historic contaminated site and several drains. While potentially contributing to the contaminant levels and toxicity at site CBE07, these are each unlikely to be major sources of contamination given the spatial distribution and concentration of contaminants reported in this study.

Considering this report's findings, it is recommended that future management efforts are focused on establishing whether contaminated groundwater from Mardalup Park is reaching the Swan Estuary and, if so, determining the specific pathway (i.e. the point(s) at which it enters the estuary). Furthermore, the reduction of contaminant levels in the Swan Estuary sediments should be addressed.

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 the Swan Estuary when tested in the laboratory, and some contaminants were shown to have bioaccumulated in aquatic biota (Nice & Fisher 2011). While a proportion of the contamination in this area of the estuary is likely to be an artefact of historic deposition, two drains discharging to the system (Claisebrook Drain and Claisebrook Diversion Drain) were shown to be current sources of contaminants; and an additional source to those previously investigated was suggested (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). As such, the site was regarded as a seriously contaminated industrial site (EPA 1992a).

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 2.5 m sediment depth in the centre (Bowman Bishaw Gorham 1992; 1993). The East Perth Redevelopment Authority (EPRA) 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 River (to a depth of 1 m below the riverbed level) 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 eastern 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 to 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 comparatively 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 via the GID (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 whole-sediment toxicity assessment, sediment chemistry assessment, water chemistry assessment and a benthic macroinvertebrate survey.

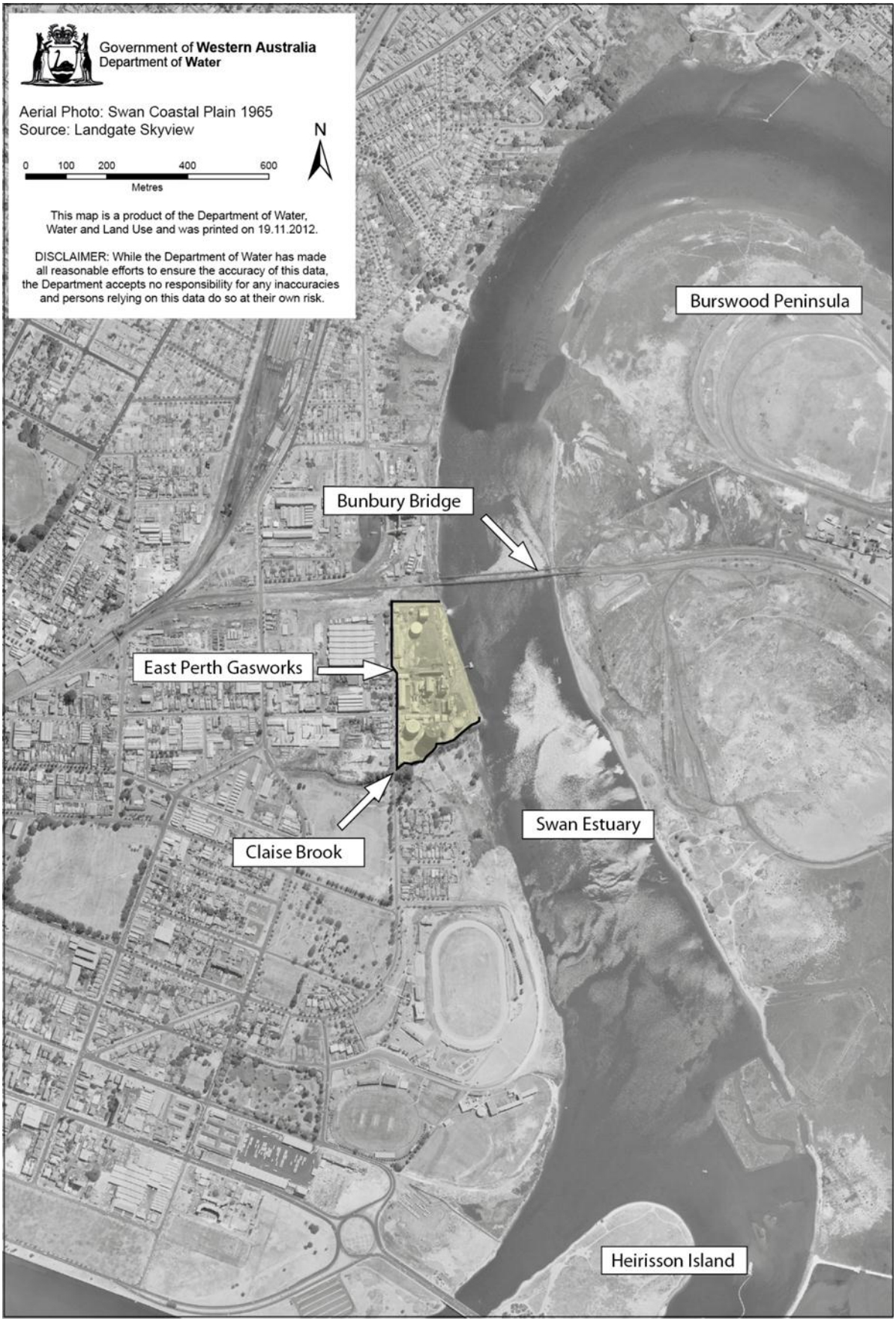

Figure 1 East Perth Gasworks and surrounding area – 1965

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

1.2 Scope

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

1.3 Objectives

The overall objective was to compare the relative toxicity of sediments collected from 11 sites within the Swan Estuary, targeting the area adjacent to Mardalup Park at Claisebrook Cove.

The specific objectives were to determine whether:

- sediments collected from the receiving environment adjacent to the GID outfall were toxic to aquatic organisms,
- the level of toxicity (if experienced) at the GID outfall was greater than at other sites, and to establish whether a toxicity gradient existed away from the GID outfall,
- sediment toxicity could be explained by contaminants measured in the sediment.

2 Methods

Sediment toxicity assessment was conducted to determine whether laboratory organisms exhibited toxic responses to field-collected sediments. Screening sediment samples in this way enables potential impact to the receiving environment (the ecosystem) to be determined, especially when the results are considered in conjunction with chemical, particle size and ecological (Nice 2013) data.

2.1 Field sampling

Whole-sediment (sediment and associated pore water) samples were collected for toxicity and chemistry assessment at 11 sites in the Swan Estuary in April 2011 (Figure 3). Ten sites were located in a grid formation extending upstream and downstream from the GID outfall to the Swan Estuary. Sites were situated approximately 200 m apart, with one site (CBE05) being immediately adjacent to the GID outfall. Four of the 11 sites were located within the zone remediated in 1994 (according to Bouckaert 1996). A reference site considered unaffected by potential contaminants from the gasworks or drains was located approximately 6 km downstream in Melville Water. This site has been used in previous assessments (Nice 2009; Nice & Fisher 2011).

Samples were collected with Perspex^{TM} corers by scuba-assisted divers. Each sample comprised four litres of sediment made up of the top 2 cm (according to Simpson *et al.* 2005) of multiple cores. Sediment was collected from an area approximately 3 m x 3 m at each site. Three litres of sediment was immediately preserved in a food-standard zip-lock low-density polyethylene (LDPE) bag on ice in the dark for toxicity assessment by Ecotox Services Australasia, New South Wales. A 500 mL portion of sediment was preserved in a glass jar on ice for chemical analysis by the National Measurement Institute (NMI), Western Australia. A further 500 mL of sediment was preserved in a food-standard low-density polyethylene bag on ice for particle size analysis by CSIRO Minerals, Western Australia.

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 prior to sediment collection (Yellow Springs Instruments hand-held meter model: 6600).

Figure 3 Location of sites in the Swan Estuary

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2.2 Laboratory analyses

Sediment toxicity assessment

Each whole-sediment sample was tested for toxicity using a suite of four toxicity tests. The test suite comprised organisms from each of the major animal taxa for which standardised tests are currently available for estuarine sediments (molluscs, crustaceans and fish). The suite also included a variety of life stages (embryo/larvae, juvenile and adult). This enabled a range of responses to contaminants to be assessed since organisms with different physiologies often respond differently to the same contaminants (USEPA 2002; Anderson *et al.* 2003); and different life stages can have different sensitivities to the same contaminants (e.g. Nice *et al.* 2001; 2003).

The four test taxa were the mussel, *Mytilus galloprovincialis;* the amphipod, *Grandidierella japonica;* the copepod, *Gladioferans imparipes;* and the fish, *Pagrus auratus.* All test organisms selected were considered environmentally relevant because either the species themselves or others within the same genus are present in the Swan and/or other temperate south-west Western Australian estuaries (e.g. SRRC 1955; Rippingale & Hodgkin 1974; Chubb *et al.* 1979; Trayler & McKernan 1997; Potter & Hyndes 1999). Black bream (*Acanthopagrus butcheri*) was the preferred fish test species because it is commonly found within the cove but viable stock cultures were not available at the time of testing. Pink snapper (*Pagrus auratus*) have displayed similar sensitivities to black bream larvae to a range of contaminants when in the larval form (Dr. R. Krassoi pers. comm. 2011, Ecotox Services Australasia).

The level of toxicity is the degree to which a substance or combination of substances is able to induce a harmful effect on an exposed organism. In this study, different endpoints were employed for different test organisms to represent toxic effects. Developmental abnormalities and/or developmental delays were used as a measure of toxicity for the mussel. Mortality was used as a measure of toxicity for the copepod and amphipod. Imbalance (larvae unable to maintain an upright position in the water column) – generally a precursor to mortality – was used as a measure of toxicity for the fish.

Test methods were selected to simulate the most relevant routes of exposure. The amphipod is typically a sediment-dweller. Therefore in this investigation, amphipod test organisms were exposed to whole-sediment. The copepods, fish and mussel larvae used in these investigations typically inhabit the water column. As such, sediment elutriate tests were selected for these test organisms. Elutriates were generated by agitating sediment in clean seawater (salinity adjusted) and the organism subsequently exposed to the elutriate. This technique is considered representative of exposure to contaminants leaching from disturbed sediments (USEPA 1991). Each test was conducted in quadruplicate using commercially available methods endorsed by the National Association of Testing Authorities (NATA) where possible:

- amphipod: ESA (2010) based on Simpson *et al*. (2005)
- copepod: ESA (unpublished)
- mussel: ESA (2011a) based on USEPA (1996) and APHA (1998)
- fish: ESA (2011b) based on USEPA (2002).

Summaries of the four test methods are provided in Table 1 to Table 4.

For the mussel, copepod and fish tests, in instances where toxicity was experienced with the 100% test solutions (i.e. undiluted), subsequent dilution-series testing was performed to determine the degree of toxicity experienced[2]. For the dilution-series testing, test solution concentrations were: 0% (filtered seawater control), 6.3%, 12.5%, 25%, 50% and 100%. It is not possible to perform dilution-series testing for the amphipod given that the test is performed using whole-sediment.

Temperature, pH, salinity and dissolved oxygen concentration of the test media were monitored to ensure no adverse laboratory conditions were influencing test results.

Table 1 Amphipod (Grandidierella japonica) test methodology

Test performed:	10-day whole-sediment survival test.
Test organism:	Amphipod, *Grandidierella japonica.*
Test protocol:	ESA Standard Operating Procedure 109 (ESA 2010) based on Simpson *et al.* (2005).
Preparation of test sediments:	Sediments were prepared approximately 24 hours before test initiation by placing 40 g (wet weight) of whole-sediment in 250 mL glass beakers. Toxicity tests were conducted on the whole-sediments without additional dilutions. A clean sediment control was tested concurrently with the samples.
Test organism life stage and exposure period:	Amphipod adults were exposed to test sediments for 10 days.
Test endpoint:	Survival.
Test replicates:	Four.
Source of test organism:	Lake Macquarie, New South Wales.

[2] For the fish test – although initial screening for toxicity was performed with pink snapper larvae, these were unavailable at the time of dilution-series testing due to non-viable stock cultures. Yellowtail kingfish (*Seriola lalandi*) larvae were used as an alternative (for the dilution-series testing only) and although do not occur naturally in the Swan Estuary have shown similar sensitivity to snapper larvae to a range of contaminants (Dr. R. Krassoi, pers. comm. 2011, Ecotox Services Australasia).

Table 2 Copepod (Gladioferans imparipes) test methodology

Test performed:	48-hour acute survival test.
Test organism:	Copepod, *Gladioferans imparipes.*
Test protocol:	ESA (unpublished).
Preparation of test solution:	Sediment elutriates were prepared by combining sediment and filtered seawater in a 1:4 ratio on a volume-to-volume basis according to methods by USEPA (1991). 100 mL of sediment was placed into a 1 L glass beaker and combined with 400 mL of filtered seawater. The mixture was stirred vigorously for 30 minutes with a magnetic stirrer (manually shaken for sandy sediments or those containing large amounts of detritus). The mixture was allowed to settle for one hour and the supernatant was collected. The test concentrations of each sample were prepared by serial dilution of the supernatant with filtered seawater. A filtered seawater control was tested concurrently with the elutriates.
Test organism life stage and exposure period:	Copepod adults were exposed to test solutions for 48 hours.
Test endpoint:	Survival.
Test replicates:	Four.
Source of test organism:	Hatchery cultured, Western Australia.

Table 3 Mussel (Mytilus galloprovincialis) test methodology

Test performed:	48-hour larval development test.
Test organism:	Mussel, *Mytilus galloprovincialis.*
Test protocol:	ESA Standard Operating Procedure 106 (ESA 2011a) based on USEPA (1996) and APHA (1998).
Preparation of test solution:	Sediment elutriates were prepared by combining sediment and filtered seawater in a 1:4 ratio on a volume-to-volume basis according to methods by USEPA (1991). 100 mL of sediment was placed into a 1 L glass beaker and combined with 400 mL of filtered seawater. The mixture was stirred vigorously for 30 minutes with a magnetic stirrer (manually shaken for sandy sediments or those containing large amounts of detritus). The mixture was allowed to settle for one hour and the supernatant was collected. The test concentrations of each sample were prepared by serial dilution of the supernatant with filtered seawater. A filtered seawater control was tested concurrently with the elutriates.
Test organism life stage and exposure period:	Mussel embryos were exposed to test solutions for 48 hours.
Test endpoint:	Larval development to D-veliger stage*.
Test replicates:	Four.
Source of test organism:	Hatchery cultured, Tasmania.

** D-veliger stage is a key developmental stage in bivalve molluscs. Abnormalities or delays in reaching this stage can result in subsequent inhibition of metamorphosis into viable adults (e.g. Nice 2000).*

Table 4 Pink snapper (Pagrus auratus) test methodology

Test performed:	96-hour larval fish imbalance test.
Test organism:	Pink snapper, *Pagrus auratus* (and yellowtail kingfish, *Seriola lalandi*).
Test protocol:	ESA Standard Operating Procedure 117 (ESA 2011b) based on USEPA (2002).
Preparation of test solutions:	Sediment elutriates were prepared by combining sediment and filtered seawater in a 1:4 ratio on a volume-to-volume basis according to methods by USEPA (1991). 100 mL of sediment was placed into a 1 L glass beaker and combined with 400 mL of filtered seawater. The mixture was stirred vigorously for 30 minutes with a magnetic stirrer (manually shaken for sandy sediments or those containing large amounts of detritus). The mixture was allowed to settle for one hour and the supernatant was collected. The test concentrations of each sample were prepared by serial dilution of the supernatant with filtered seawater. A filtered seawater control was tested concurrently with the elutriates.
Test organism life stage and exposure period:	Fish larvae were exposed to test solutions for 96 hours.
Test endpoint:	Survival (imbalance).
Test replicates:	Four.
Source of test organism:	Hatchery cultured, Western Australia.

Sediment chemistry and particle size assessment

Sediment chemistry assessment was conducted to quantify particular contaminants to assist the interpretation of any toxicity observed. 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 NATA. Sediments were assessed for particle size distribution in order to determine the potential contaminant-binding capacity of the sediments. Sediment chemistry and particle size analytical methods are provided in Table 5.

Table 5 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 metals 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 mg/kg	Determination of PAHs 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 Endosulphan sulphate	0.001 mg/kg	Determination of OC pesticides 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

Parameter	Limit of reporting (mg/kg)	Description	Analysis method
Aldrin Dieldrin Endrin p,p'-DDE p,p'-DDD p,p'-DDT Methoxychlor			
Total organic carbon (TOC)	100	Determination of TOC 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 Categorising the level of toxicity

Sites were divided into three categories according to the degree of toxicity experienced in the samples collected from those sites. The categories were: *no toxicity, low-level toxicity* and *high-level toxicity* and are defined in Table 6.

Table 6 Toxicity categories

Level of toxicity	Criteria for copepod, mussel and fish tests	Criteria for amphipod test
No toxicity	No difference* in response between test and control organisms.	No difference* in response between test and control organisms.
Low-level toxicity	A difference* in response between test and control organisms observed with undiluted sediment elutriate; and no difference observed with subsequent dilution-series testing.	A difference* in response between test and control organisms with < 50% of test organisms exhibiting the response.
High-level toxicity	A difference* in response between test and control organisms observed with undiluted sediment elutriate; and a difference* in response observed with dilution-series testing in $\leq$ 50% sediment elutriate concentrations.	A difference* in response between test and control organisms with $\geq$ 50% of test organisms exhibiting the response.

** statistically significant effect ($p < 0.05$)*

2.4 Statistical analyses of toxicity data

Prior to analyses, the distributions of toxicity data were tested for normality (Shapiro-Wilk's test) and homogeneity of variance (Bartlett's test). Data were transformed where required and the appropriate tests selected and performed using the TOXCALC V 5.0 statistical package. Dunnett's test was used to compare 100% elutriate toxicity (for copepod, mussel and fish) with controls. Bonferroni adjusted t-test was used to compare whole-sediment toxicity (for amphipod) with controls. In instances where high-level toxicity was demonstrated and subsequent dilution-series testing was employed, Dunnett's test was used to compare a range of test concentrations with the controls. EC50 (concentration of sediment elutriate affecting 50% of the test population) values were determined by the Maximum Likelihood Probit method.

2.5 Application of guidelines to sediment contaminants

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.

Concentrations of organic contaminants such as PAHs and OC pesticides measured in this study 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 have been increased above normal concentrations due to organic contamination (such as petroleum compounds), 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 and OC pesticide data are presented here.

3 Results

Summary

- Toxicity was experienced in all four test taxa. The species affected and the degree of toxicity varied between sites.

- Bioavailable metals, OC pesticides and PAHs were detected at all sites with the concentration and number of ISQGs exceeded varying between sites.

- The level of toxicity and contaminant concentrations experienced at the GID outfall (CBE05) were comparatively low when compared with other sites (toxicity was only experienced for fish at CBE05).

- There was no clear gradient in either toxicity or contamination away from the site adjacent to the GID outfall (CBE05).

- Toxicity was greatest with a concomitant peak in PAH concentrations at the site adjacent to the middle section of Mardalup Park (CBE07).

- Toxicity was not experienced at the reference site (CBE11).

3.1 Sediment toxicity

Toxicity was evident for all test taxa and the degree of toxicity experienced varied between sites (Table 7 and Figure 4 to Figure 7).

Amphipod survival was affected at sites CBE06 and CBE07 where mean percentage survival was 70% and 58% respectively compared with 93% in the control (Figure 4). Toxicity was considered low level because < 50% of test organisms were affected.

Copepod survival was only affected at site CBE07 where survival was zero compared with 90% in the control (Figure 5). Toxicity was reported as high level because subsequent dilution-series testing showed significant effects at < 50% sediment elutriate concentration. Thirty per cent copepod survival was observed in 25% sediment elutriate concentration and 5% survival was observed in 50% sediment elutriate concentration (Figure 8).

Mussel development was affected at sites CBE01, CBE03 and CBE07 where mean percentage normal larvae was 93%, 90% and 80% respectively compared with 97% in the control (Figure 6). Toxicity was reported as high level at sites CBE01 and CBE07 because subsequent dilution-series testing showed significant effects at < 50% sediment elutriate concentration (effects were seen at 12.5% for CBE01 and 25% for CBE07) (Figure 9). However, normal development was still relatively high (92% and 91% respectively); and for both sites there was a small increase in normally developed larvae witnessed with the next-highest sediment elutriate

concentration. Site CBE03 showed toxicity for undiluted (100%) sediment elutriate concentration only. Therefore toxicity at this site was reported as low level.

Fish larval development was affected at sites CBE04, CBE05, CBE06, CBE07 and CBE09 where there was no normal development compared with 75% normal development in the control (Figure 7). Toxicity was reported as high level at each of these sites because subsequent dilution-series testing showed significant effects at sediment elutriate concentrations < 50%. Effects were seen at 12.5% (site CBE07) and 25% (sites CBE04, CBE06 and CBE09) sediment elutriate concentrations. Notably, there was no normal development for fish larvae exposed to 25% or greater elutriate concentration for site CBE07 and 50% or greater elutriate concentration for sites CBE04, CBE05, CBE06 and CBE09 (Figure 10).

The toxicity experienced with the four test species across sites is summarised in Table 7.

Table 7 Summary of the toxicity experienced with each test for samples collected from each site.

	Toxicity test			
	Amphipod	**Copepod**	**Mussel**	**Fish**
Site				
CBE01			**XX**	
CBE02				
CBE03			**X**	
CBE04				**XX**
CBE05				**XX**
CBE06	**X**			**XX**
CBE07	**X**	**XX**	**XX**	**XX**
CBE08				
CBE09				**XX**
CBE10				
CBE11 (field reference)				
Laboratory control				

*Blank cells = no toxicity; **X** = low-level toxicity; **XX** = high-level toxicity.*

Screening for toxicity across sites

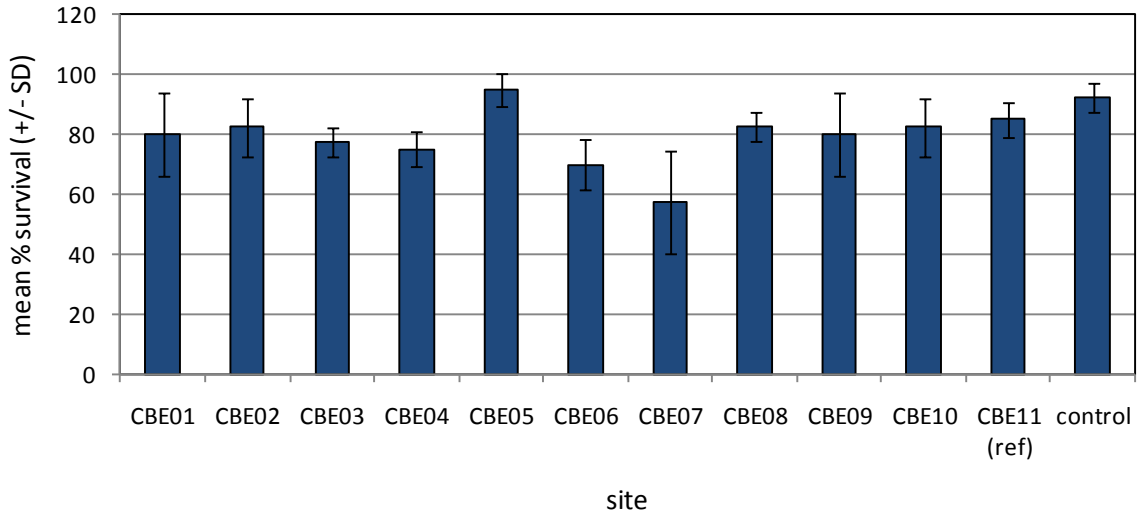

Figure 4 *Mean percentage amphipod survival after 10-day exposure to whole-sediment. Significantly lower percentage survival compared with the control (Bonferroni adjusted t-test, 1-tailed, p = 0.05) observed for sediment collected from sites CBE06 and CBE07. There was no significant difference in survival between other sites and the control (p > 0.05).*

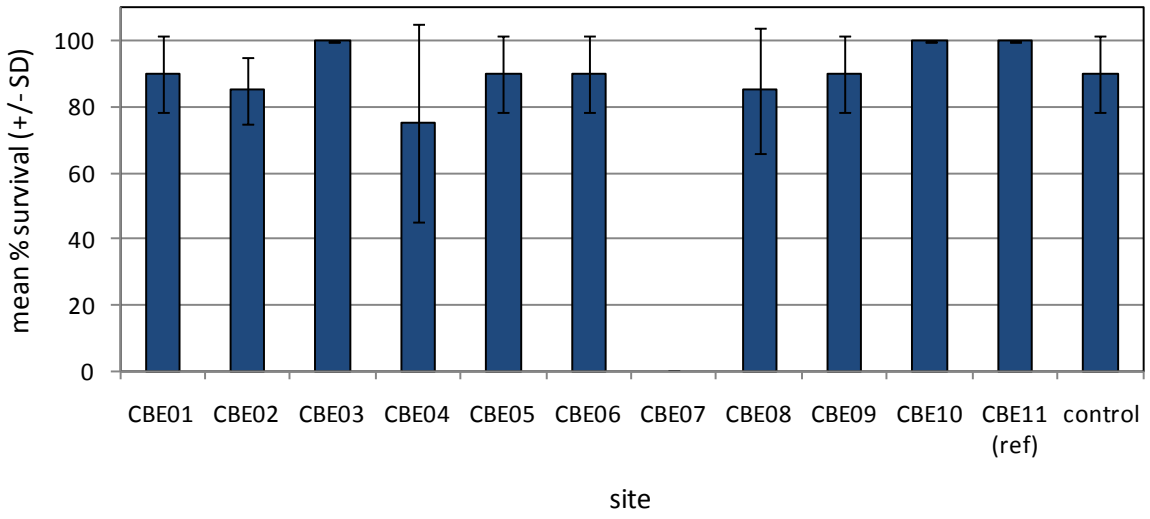

Figure 5 *Mean percentage copepod survival after 48-hour exposure to sediment elutriates. Copepod survival was zero for sediment collected from site CBE07. There was no significant difference in survival between other sites and the control (Dunnett's test, 1-tailed, p > 0.05).*

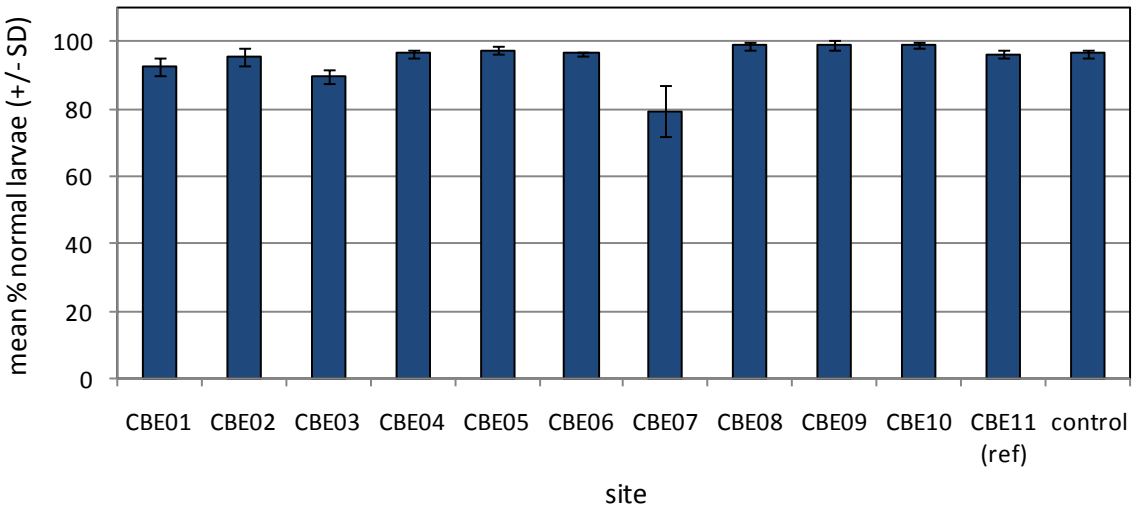

Figure 6 Mean percentage normally developed mussel larvae after 48-hour exposure to sediment elutriates. Significantly lower percentage normally developed larvae compared with the control (Dunnett's test, 1-tailed, p = 0.05) observed for sediment collected from sites CBE01, CBE03 and CBE07. There was no significant difference in survival between other sites and the control (p > 0.05).

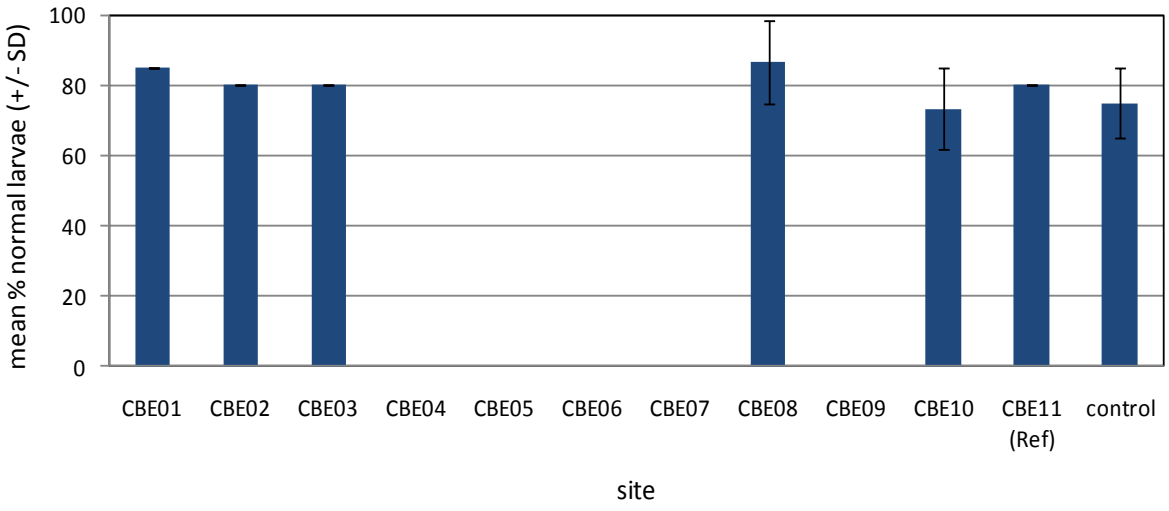

Figure 7 Mean percentage normal fish larvae after 96-hour exposure to sediment elutriates. Percentage normally developed fish larvae was zero for sediment collected from sites CBE04, CBE05, CBE06, CBE07 and CBE09. There was no significant difference in normally developed larvae between other sites and the control (Dunnett's test, 1-tailed, p > 0.05).

Dose-response plots for sites where high-level toxicity was demonstrated

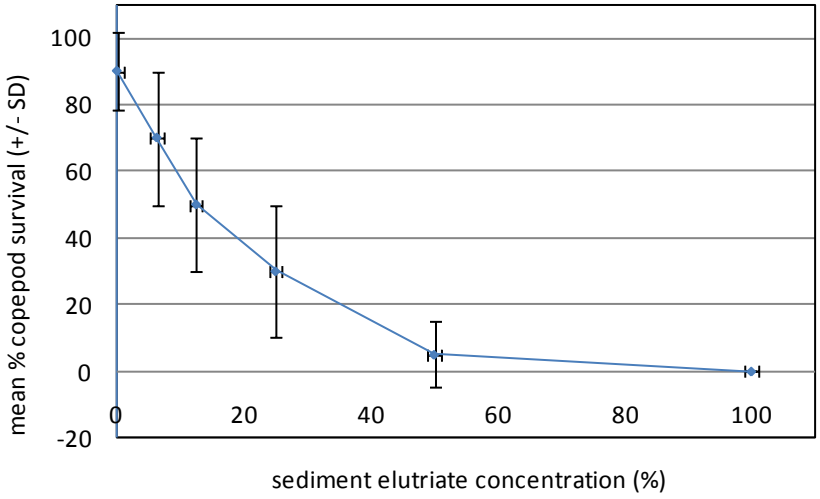

Figure 8 Copepod dose-response plot for site CBE07 where high-level toxicity was exhibited. There was significantly lower copepod survival for 25% and 50% sediment elutriate concentrations when compared with the control (Dunnett's test, 1-tailed, $p < 0.05$). There was no survival for copepods exposed to 100% sediment elutriate concentration.

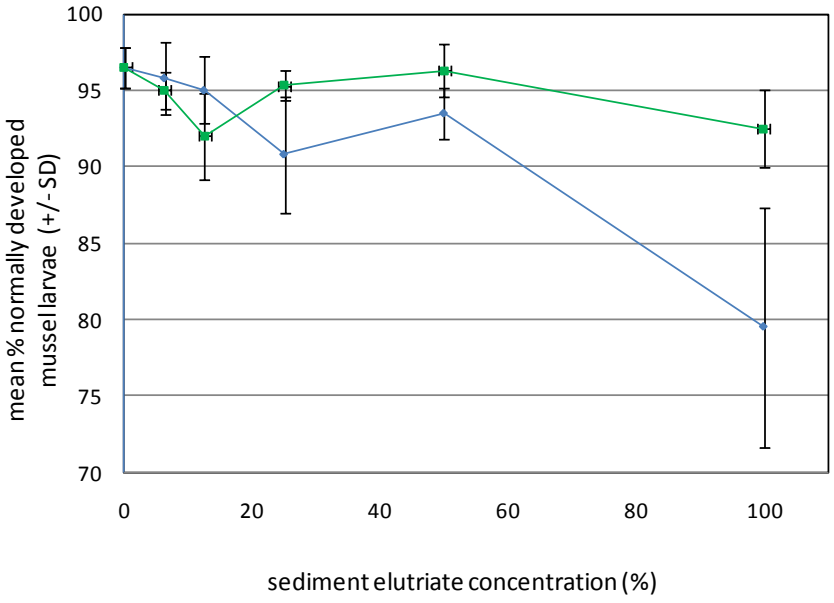

Figure 9 Mussel larvae dose-response plot for sites CBE01 (green) and CBE07 (blue) where high-level toxicity was exhibited. There was significantly lower normal mussel larval development for 12.5% and 100% sediment elutriate concentrations from site CBE01; and 25% and 100% sediment elutriate concentrations from site CBE07 when compared with the control (Dunnett's test, 1-tailed, $p < 0.05$).

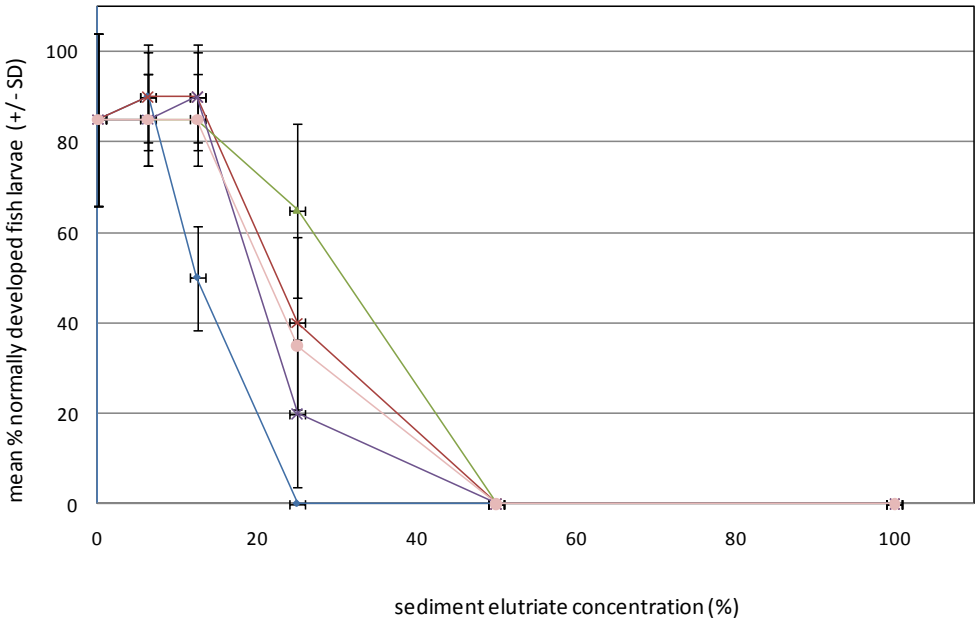

Figure 10 Fish larvae dose-response plot for sites CBE04 (red), CBE05 (green), CBE06 (purple), CBE07 (blue) and CBE09 (pink) where high-level toxicity was exhibited. There were significantly fewer normal fish larvae for 12.5% sediment elutriate concentrations from site CBE07, 25% sediment elutriate concentrations from sites CBE04, CBE06 and CBE09 when compared with the control (Dunnett's test, 1-tailed, $p < 0.05$). There was no normal development in fish larvae exposed to $\underline{\geq}$ 25% sediment elutriate concentration for site CBE07 and $\underline{\geq}$ 50% for sites CBE04, CBE05, CBE06 and CBE09.

Effect concentrations for sites where high-level toxicity was demonstrated

High-level toxicity was demonstrated in at least one of the test organisms for sites CBE01, CBE04, CBE05, CBE06, CBE07 and CBE09. Of these sites, sediments collected from CBE07 caused the greatest toxicity to the test organisms used in this study (Table 8) indicated by the lowest EC50s (the sediment elutriate concentration that causes the effect in 50% of organisms) for both copepod and fish tests.

Table 8 Comparison of effect concentrations for sites where high-level toxicity was demonstrated

		Copepod	
site	48-hour EC50 (%)	NOEC (%)	LOEC (%)
CBE07	14.4 (8.4 – 20.4)	12.5	25
		Mussel	
site	48-hour EC50 (%)	NOEC (%)	LOEC (%)
CBE01	> 100	6.3	12.5
CBE07	> 100	12.5	25
		Fish	
site	96-hour EC50 (%)	NOEC (%)	LOEC (%)
CBE04	24.2 (20.7 – 28.2)	12.5	25
CBE05	30.0 (26.3 – 34.3)	25	50
CBE06	20.7 (18.2 – 23.6)	12.5	25
CBE07	13.2 (11.3 – 15.3)	6.3	12.5
CBE09	23.5 (20.2 – 27.4)	12.5	25

95% confidence limits shown in brackets

EC50 = sediment elutriate concentration which causes the effect in 50% of test organisms

*NOEC = no observable effect concentrations: the highest **tested** concentration at which organisms were unaffected compared with control organisms*

*LOEC = lowest observable effect concentration: the lowest **tested** concentration at which organisms were adversely affected compared with control organisms*

3.2 Sediment chemistry

Total organic carbon concentrations (Table 9) were generally high across all sites (according to Michelsen 1992) (Refer to Section 2.5). As such, PAH and OC pesticide data are reported in both normalised (to 1% organic carbon according to Simpson *et al.* 2005) and raw format (not normalised to 1% organic carbon).

Table 9 Total organic carbon concentrations

Site	Total organic carbon (mg/kg)
CBE01	198000
CBE02	43000
CBE03	36000
CBE04	29000
CBE05	12000
CBE06	29000
CBE07	23000
CBE08	35000
CBE09	7200
CBE10	41000
CBE11	26000

Bioavailable metals

All metals assessed in this study were present in concentrations above the limit of reporting at all sites – except mercury and cadmium that were not detected at any site and selenium that was only detected at CBE01, CBE06, CBE08, CBE10 and CBE11 (Table 10). Lead concentrations exceeded the *low* ISQG for all sites except CBE03, CBE05 and CBE09. Zinc concentrations exceeded the *low* ISQG for all sites except CBE03, CBE04, CBE05 and CBE09.

PAHs

All of the PAHs assessed in this study were present in concentrations above the limit of reporting except fluorene that was not detected at any site (Table 11 and Table 12). PAH contaminants peaked at site CBE07 for all PAHs detected (Figure 11); and guidelines were exceeded for 11 of the 13 PAH contaminants for which there are guidelines. The ISQG for total PAHs was also exceeded. Similar trends were seen between non-normalised and normalised (to 1% organic carbon) datasets, with the following exceptions: four *high* ISQGs were exceeded in the non-normalised dataset compared with two *high* ISQGs in the normalised dataset for site CBE07. Likewise, two *low* guidelines were exceeded in the non-normalised dataset compared with none for site CBE01. Conversely, two *low* guidelines were exceeded for site CBE09 in the non-normalised dataset compared with three in the normalised dataset.

OC pesticides

Of the suite of OC pesticides targeted, only trans-chlordane, dieldrin, p,p'-DDT, p,p'-DDE and p,p'-DDD were present in concentrations above the limit of reporting (Table 13 and Table 14). ISQGs were exceeded for at least one OC pesticide at every site when the data were not normalised to 1% organic carbon. ISQGs were exceeded for at least one OC pesticide at all sites except CBE01 and CBE11 when the data were normalised to 1% organic carbon.

Table 10 Sediment metal concentrations (bioavailable)

Sediment metal concentrations (bioavailable) mg/kg dry weight											
Site	**Arsenic**	**Cadmium**	**Chromium**	**Cobalt***	**Copper**	**Lead**	**Manganese***	**Mercury**	**Nickel**	**Selenium***	**Zinc**
CBE01	2.1	n.d.	8.3	3.9	32	95	170	n.d.	5.3	0.64	270
CBE02	1.5	n.d.	9.9	4.9	36	82	110	n.d.	3.7	n.d.	230
CBE03	1.8	n.d.	4.1	2.9	26	37	140	n.d.	3.3	n.d.	120
CBE04	1.9	n.d.	5.9	3.1	30	50	100	n.d.	2.4	n.d.	180
CBE05	0.68	n.d.	2.4	1.3	6.9	14	92	n.d.	1.0	n.d.	43
CBE06	3.3	n.d.	11	4.7	39	74	170	n.d.	3.7	0.76	290
CBE07	2.0	n.d.	8.2	3.3	30	66	180	n.d.	2.9	n.d.	260
CBE08	3.8	n.d.	12	5.5	43	80	190	n.d.	3.9	0.89	320
CBE09	0.64	n.d.	2.5	1.1	8.9	17	54	n.d.	0.94	n.d.	62
CBE10	3.2	n.d.	12	5.4	41	83	150	n.d.	4.1	0.93	330
CBE11 (ref)	5.8	n.d.	14	6.9	54	82	300	n.d.	4.5	0.66	330
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); blue indicates low ISQG exceeded; orange 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 lowest effect level; Lemly 1996) were also not exceeded. N.d. = not detected; limit of reporting for mercury: 0.5 mg/kg; limit of reporting for other metals: 0.1 mg/kg. Samples comprised the top 2 cm of sediment.*

Table 11 Sediment polycyclic aromatic hydrocarbon (PAH) concentrations

Sediment polycyclic aromatic hydrocarbon (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-cd] pyrene	Dibenz[a,h]anthracene	Benzo [g,h,i]perylene	Total PAHs
CBE01	n.d.	130	n.d.	n.d.	90	60	300	440	240	190	390	310	120	40	140	2400
CBE02	n.d.	30	n.d.	n.d.	10	10	50	80	40	40	80	60	30	n.d.	30	470
CBE03	n.d.	10	n.d.	n.d.	50	20	100	120	60	50	90	60	20	n.d.	30	620
CBE04	n.d.	30	n.d.	n.d.	10	20	60	120	60	50	110	90	40	10	40	630
CBE05	n.d.	15	n.d.	n.d.	15	10	70	110	45	45	80	50	20	n.d.	30	310
CBE06	n.d.	30	n.d.	n.d.	20	15	70	105	55	55	110	75	30	7.5	45	520
CBE07	40	1700	260	n.d.	700	890	3300	6100	2500	1500	2900	2800	700	230	790	24000
CBE08	n.d.	30	n.d.	n.d.	20	20	80	130	60	60	120	90	40	10	50	710
CBE09	n.d.	70	n.d.	n.d.	30	30	300	360	200	160	330	230	80	20	90	1900
CBE10	n.d.	10	n.d.	n.d.	n.d.	n.d.	30	50	20	20	50	30	10	n.d.	20	240
CBE11 (ref)	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	20	20	10	10	20	20	n.d.	n.d.	n.d.	n.d.
ISQG Low	160	44	16	19	240	85	600	665	261	384	n.a.	430	n.a.	63	n.a.	4000
ISQG High	2100	640	500	540	1500	1100	5100	2600	1600	2800	n.a.	1600	n.a.	260	n.a.	45000

*ISQG = Interim Sediment Quality Guideline (ANZECC & ARMCANZ 2000); blue indicates low ISQG exceeded; orange indicates high ISQG exceeded; n.a. = no ANZECC & ARMCANZ 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 2 cm of sediment. Data not normalised to 1% OC.*

Table 12 Sediment polycyclic aromatic hydrocarbon (PAH) concentrations normalised to 1% organic carbon

Sediment polycyclic aromatic hydrocarbon (PAH) concentrations (µg/kg) dry weight, normalised to 1% organic carbon																
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	Total PAHs
CBE01	n.d.	6.6	n.d.	n.d.	4.5	3.0	15.2	22.2	12.1	9.6	19.7	15.7	6.1	2.0	7.1	121.2
CBE02	n.d.	7.0	n.d.	n.d.	2.3	2.3	11.6	18.6	9.3	9.3	18.6	14.0	7.0	n.d.	7.0	109.3
CBE03	n.d.	2.8	n.d.	n.d.	13.9	5.6	27.8	33.3	16.7	13.9	25.0	16.7	5.6	n.d.	8.3	172.2
CBE04	n.d.	10.3	n.d.	n.d.	3.4	6.9	20.7	41.4	20.7	17.2	37.9	31.0	13.8	3.4	13.8	217.2
CBE05	n.d.	12.5	n.d.	n.d.	12.5	8.3	58.3	91.7	37.5	37.5	66.7	41.7	16.7	0.0	25.0	258.3
CBE06	n.d.	10.3	n.d.	n.d.	6.9	5.2	24.1	36.2	19.0	19.0	37.9	25.9	10.3	2.6	15.5	179.3
CBE07	17.4	739.1	113.0	n.d.	304.3	387.0	1434.8	2652.2	1087.0	652.2	1260.9	1217.4	304.3	100.0	343.5	10434.8
CBE08	n.d.	8.6	n.d.	n.d.	5.7	5.7	22.9	37.1	17.1	17.1	34.3	25.7	11.4	2.9	14.3	202.9
CBE09	n.d.	97.2	n.d.	n.d.	41.7	41.7	416.7	500.0	277.8	222.2	458.3	319.4	111.1	27.8	125.0	2638.9
CBE10	n.d.	2.4	n.d.	n.d.	n.d.	n.d.	7.3	12.2	4.9	4.9	12.2	7.3	2.4	n.d.	4.9	58.5
CBE11 (ref)	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	7.7	7.7	3.8	3.8	7.7	7.7	n.d.	n.d.	n.d.	n.d.
ISQG Low	160	44	16	19	240	85	600	665	261	384	n.a.	430	n.a.	63	n.a.	4000
ISQG High	2100	640	500	540	1500	1100	5100	2600	1600	2800	n.a.	1600	n.a.	260	n.a.	45000

*ISQG = Interim Sediment Quality Guideline (ANZECC & ARMCANZ 2000); blue indicates low ISQG exceeded; orange indicates high ISQG exceeded; n.a. = no ANZECC & ARMCANZ 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 2 cm of sediment. Data normalised to 1% OC.*

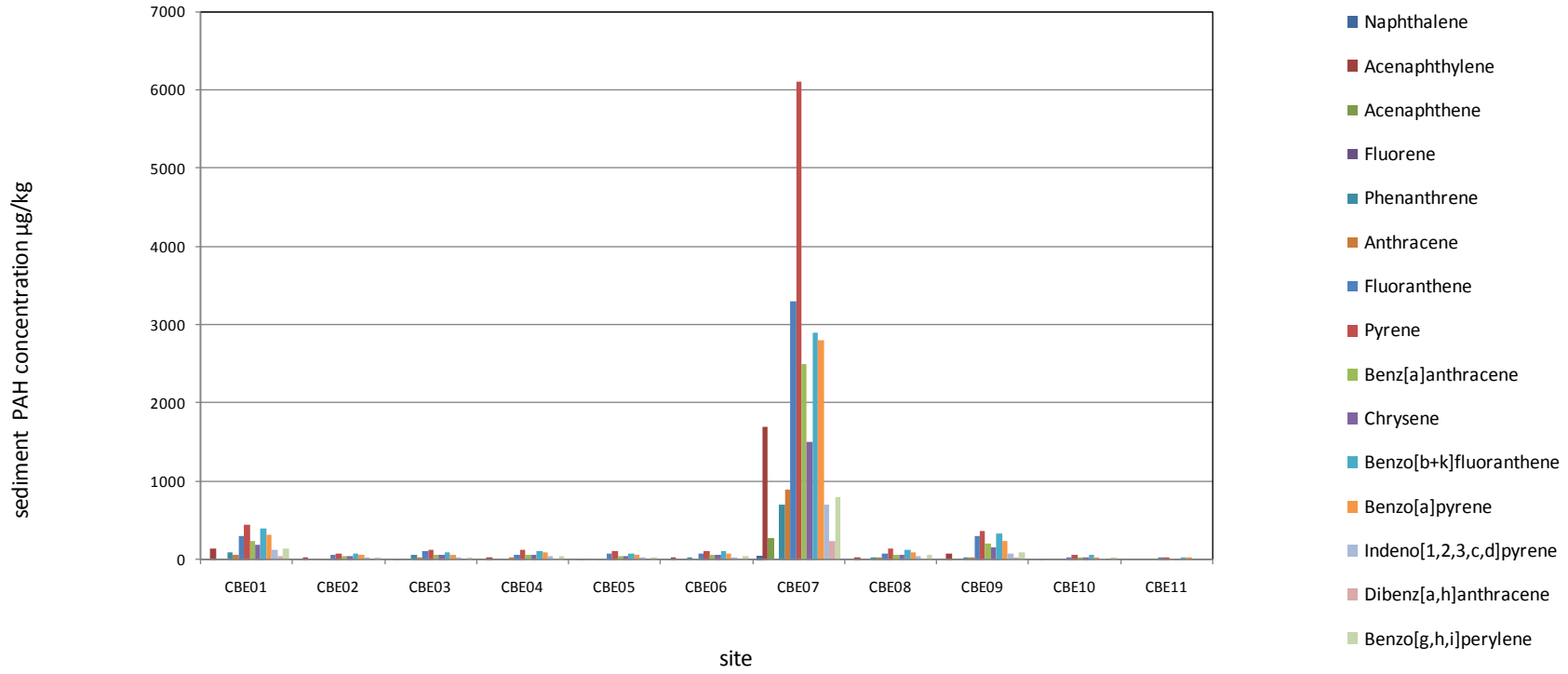

Figure 11 PAH concentrations (dry weight) across sites (data not normalised to 1% organic carbon)

Table 13 Sediment organochlorine (OC) pesticide concentrations

Sediment organochlorine (OC) pesticide concentrations (µg/kg) dry weight					
Site	trans-Chlordane	Dieldrin	p,p'-DDT	p,p'-DDE	p,p'-DDD
CBE01	4.2	n.d.	2	13	7.8
CBE02	1.1	3.9	1.1	13	4.8
CBE03	2	8.7	n.d.	5.6	3.7
CBE04	n.d.	3.9	1.4	18	7.5
CBE05	n.d.	2.9	n.d.	1.7	n.d.
CBE06	n.d.	2.9	1.7	9.7	3.1
CBE07	n.d.	n.d.	n.d.	15	n.d.
CBE08	1.2	4.6	1.5	16	4.8
CBE09	n.d.	n.d.	n.d.	2.2	1.2
CBE10	1.2	4.1	1.2	13	4.2
CBE11(ref)	n.d.	n.d.	n.d.	3	n.d.
ISQG – low	0.5*	0.02	1.6**	2.2	2***
ISQG – high	6*	8	46**	27	20***

*Note: Only those parameters detected are shown in this table. Refer to Table 5 for full list of contaminants. ISQG = Interim Sediment Quality Guideline (ANZECC & ARMCANZ 2000); blue indicates low ISQG exceeded; orange indicates high ISQG exceeded. * denotes the ISQG for chlordane (trans-chlordane is one constituent of chlordane). ** denotes the ISQG for total DDT (only p,p'-DDT was measured in this study). *** denotes the ISQG for p,p'-DDD plus o,p'-DDD (only p,p'-DDD was measured in this study). N.d. = not detected; limit of reporting: 1 µg/kg. Samples comprised the top 2 cm of sediment. Data not normalised to 1% organic carbon.*

Table 14 Sediment organochlorine (OC) pesticide concentrations normalised to 1% organic carbon

	Sediment organochlorine pesticide (OC) concentrations (µg/kg) dry weight, normalised to 1% organic carbon				
Site	trans-Chlordane	Dieldrin	p,p'-DDT	p,p'-DDE	p,p'-DDD
CBE01	0.2	n.d.	0.1	0.7	0.4
CBE02	0.3	0.9	0.3	3.0	1.1
CBE03	0.6	2.4	n.d.	1.6	1.0
CBE04	n.d.	1.3	0.5	6.2	2.6
CBE05	n.d.	2.4	n.d.	1.4	n.d.
CBE06	n.d.	1.0	0.6	3.3	1.1
CBE07	n.d.	n.d.	n.d.	6.5	n.d.
CBE08	0.3	1.3	0.4	4.6	1.4
CBE09	n.d.	n.d.	n.d.	3.1	1.7
CBE10	0.3	1.0	0.3	3.2	1.0
CBE11(ref)	n.d.	n.d.	n.d.	1.2	n.d.
ISQG – low	0.5*	0.02	1.6**	2.2	2***
ISQG – high	6*	8	46**	27	20***

*Note: Only those parameters detected are shown in this table. Refer to Table 5 for full list of contaminants. ISQG = Interim Sediment Quality Guideline (ANZECC & ARMCANZ 2000). * denotes the ISQG for chlordane (trans-chlordane is one constituent of chlordane). ** denotes the ISQG for total DDT (only p,p'-DDT was measured in this study). *** denotes the ISQG for p,p'-DDD plus o,p'-DDD (only p,p'-DDD was measured in this study). Blue indicates low ISQG exceeded; orange indicates high ISQG exceeded. N.d. = not detected; limit of reporting: 1 µg/kg. Samples comprised the top 2 cm of the sediment. Data normalised to 1% organic carbon.*

3.3 Sediment particle size

All sediment samples consisted of particles from a range of size categories (Table 15). Dominant fractions varied from site to site, with the majority of sites having silt as the dominant fraction.

Table 15 Sediment particle size

Site	**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 – 10000 µm**
			Proportion of sediments (% by weight)			
CBE01	11.27	23.80	12.46	7.16	19.10	26.20
CBE02	23.05	49.05	15.66	2.44	1.00	8.80
CBE03	5.94	10.99	22.30	37.07	8.60	15.10
CBE04	14.60	24.78	28.20	17.41	8.20	6.80
CBE05	4.26	8.78	9.59	27.78	38.50	11.10
CBE06	18.04	59.73	10.01	2.62	1.90	7.70
CBE07	13.10	39.92	12.56	9.52	10.60	14.30
CBE08	22.99	59.00	12.07	1.94	0.80	3.20
CBE09	4.31	8.34	6.03	37.33	36.60	7.40
CBE10	19.79	56.86	14.62	2.62	1.10	50
CBE11	29.14	62.18	6.24	1.04	0.80	0.60

Blue text indicates dominant fraction(s)

3.4 *In situ* water quality

All water quality parameters were consistent between sites and little variability was observed between replicate observations within sites, except for dissolved oxygen which was lower at site CBE11. Nevertheless, oxygen levels at all sites fall within the moderately oxygenated or well-oxygenated categories for the Swan Canning system (Robb & Evans 2008) (Table 16).

Table 16 In situ water quality parameters

Site	**Temperature (^{o} C)**	**Specific conductivity m^{S}/cm**	**Salinity ppt**	**pH**	**Dissolved oxygen mg/L**
CBE01	25.39 (± 0.02)	56.63 (± 0.02)	37.65 (± 0.01)	7.61 – 7.62	6.54 (± 0.03)
CBE02	24.78 (± 0.01)	56.67 (± 0.01)	37.70 (± 0.00)	7.49 – 7.50	5.47 (± 0.06)
CBE03	24.65 (± 0.03)	56.63 (± 0.02)	37.67 (± 0.02)	7.56 – 7.57	5.71 (± 0.14)
CBE04	24.84 (± 0.00)	56.71 (± 0.00)	37.72 (± 0.00)	7.50 – 7.51	5.45 (± 0.08)
CBE05	24.81 (± 0.01)	56.68 (± 0.01)	37.70 (± 0.01)	7.58 – 7.60	6.04 (± 0.06)
CBE06	24.76 (± 0.01)	56.73 (± 0.00)	37.74 (± 0.00)	7.51	5.47 (± 0.01)
CBE07	24.51 (± 0.01)	56.73 (± 0.01)	37.75 (± 0.00)	7.58 – 7.60	5.52 (± 0.06)
CBE08	24.77 (± 0.00)	56.78 (± 0.00)	37.78 (± 0.01)	7.54	5.60 (± 0.04)
CBE09	24.66 (± 0.01)	56.70 (± 0.01)	37.72 (± 0.01)	7.61	6.20 (± 0.03)
CBE10	24.70 (± 0.01)	56.81 (± 0.01)	37.80 (± 0.00)	7.56	5.53 (± 0.10)
CBE11	24.62 (± 0.00)	56.87 (± 0.01)	37.85 (±0.01)	7.85 – 7.86	4.76 (± 0.08)

Measured 5 – 20 cm 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.

4 Discussion

4.1 The Groundwater Interception Drain site

Toxicity was experienced for the GID outfall site (CBE05). However, sediments collected from this site were not more toxic to the test organisms than those collected from other sites. Toxicity was demonstrated at the GID site for one of the four test taxa employed in this assessment; and for two and four test taxa at other sites investigated (CBE06 and CBE07 respectively). A toxicity gradient was not observed from the GID outfall. Similarly, a contaminant gradient was not observed from the GID outfall. Compared with other sites, contaminant concentrations at the GID outfall site (CBE05) were relatively low. Only one ISQG was exceeded for both normalised and non-normalised datasets at the GID site compared with 15 ISQGs being exceeded at site CBE07 (both normalised and non-normalised datasets).

Although when compared with some other sites in this investigation, both toxicity and contaminant concentrations were relatively low at the GID site, it is of significance that high-level toxicity was reported for one test taxon (fish). This is consistent with results from an earlier study (Nice & Fisher 2011) where the same result (high-level toxicity to fish) was observed with sediment collected from a site in the vicinity of the other outfall of the GID (that which discharges within the cove).

While it has recently been shown that GID discharge contains PAH (and other) contamination thought to have originated from the historic gasworks site (ENV 2009), at the time of sampling for the current study the outfall pipe was inundated with estuarine water – so it was not possible to determine whether the GID outfall was discharging to the estuary. The sediment contaminant and toxicity results from the current study suggest that any contaminants discharging from this drain are not accumulating to concentrations that cause environmental harm to the invertebrates tested here (copepod, amphipod and mussel) in the sediments immediately adjacent to the GID (site CBE05). This may be explained in part by the fact that the sediment adjacent to the GID had a dominant fraction of coarse sand and there were comparatively fewer smaller particles (such as clay and silt) when compared with many of the other sites in this study. Thus there are relatively fewer potential binding sites for contaminants here than at sites such as CBE06 and CBE11 where sediments were predominantly silt. Notwithstanding this, contaminant(s) appear to be present in the sediment at concentrations, that when disturbed, are sufficient to affect larval fish.

4.2 Other sites

Of all the sites assessed in this study, toxicity was greatest with a concomitant peak in contaminant concentrations at site CBE07 adjacent to Mardalup Park (Figure 12). Sediments collected from this site were toxic to all four of the test organisms employed in this investigation with high-level toxicity being reported in three out of the four test organisms and the lowest EC50s[3] recorded compared with all other sites. This degree of toxicity (level of toxicity and number of taxa affected) was also

[3] EC50: the sediment elutriate concentration which causes the effect in 50% of test organisms

greater than that observed for the two drain outfall sites (Claisebrook Drain and Claisebrook Diversion Drain) in a previous study (Nice & Fisher 2011). The observed toxicity may at least be attributable in part to the OC pesticide p,p'-DDE which exceeds the *low* ISQG at this site. Additionally, the broad range of PAH contaminants – all of which peak in concentration at this site and most in concentrations that exceed ISQGs (*high* ISQG for four of the PAHs and *low* ISQG for seven of the PAHs) – would likely contribute to the toxicity observed.

This spike in contamination was also observed in the same area from samples collected in 2008 and 2009 for previous studies (Nice 2009; Nice & Fisher 2011), when concentrations peaked for all of the PAH contaminants investigated along with several OC pesticide and metal contaminants, with numerous guidelines (both *low* and *high*) exceeded.

Site CBE07 is located in the area that underwent extensive remediation[4] in 1994 (CMPS & F Pty Ltd 1996), thus the current high concentrations in the sediment suggest the site was either not fully remediated or that the PAH contamination seen in this study is more recent than 1994. The range of PAHs detected in this study includes many of the low-molecular-weight PAHs, which break down relatively rapidly in the environment (Wilson & Jones 1993; Volkering & Breure 2003). Degradation is exacerbated in relatively high-energy environments such as the middle Swan Estuary adjacent to Claisebrook, where the surficial sediments are likely to be subject to agitation and suspension from waves, tidal action and boat activity. This coupled with bioturbation (Simpson *et al.* 2005) and biodegradation (Herbes & Schwall 1978) processes would likely accelerate the breakdown. It is not possible to establish the precise timing the contamination occurred because the original concentrations at this site are unknown. However, given that relatively high concentrations (exceeding ISQGs) of these low-molecular-weight PAHs have been measured over a period of three years, 2008 – 2011, a recent or current source of PAH contamination to the area should be considered.

The spatial array of sites in this study and the corresponding sediment PAH concentrations (Figure 12) show that the high level of PAH contamination at site CBE07 is unlikely due to current or recent sources located upstream or on the opposite side of the estuary. Although potential sources such as the East Perth Power Station (upstream) and the Burswood (upstream and opposite) historic contaminated sites could be contributing to the sediment contaminants observed at site CBE07, any contribution is likely to be minor since sites closer to these potential sources had markedly lower sediment PAH concentrations and sediments with similar binding capacity (CBE01, CBE02, CBE04, CBE06, CBE08 and CBE10). That is, there was no indication of a PAH contaminant gradient from either of these historic contaminated sites.

On considering other potential sources in the area, the GID is unlikely to be a major current or recent source of the PAHs found at site CBE07 given that relatively low concentrations of PAHs were measured in the sediments (this study and Nice 2013) and water (Fisher 2013) adjacent to the Claisebrook GID outfall to the estuary. Furthermore, a parallel study of benthic macroinvertebrate communities (Nice 2013)

[4] Minister's Conditions of Approval for redevelopment of the site required extensive remediation including the replacement of 13 000 m^3 (approx.) of PAH-contaminated sediment from the Swan Estuary adjacent to Mardalup Park (to a depth of 1 m below the river bed level) with 12 200 m^3 (approx.) of clean fill in 1994 (CMPS & F Pty Ltd 1996).

found no measurable impact attributable to the Claisebrook GID. The Claisebrook Main Drain and the Claisebrook Diversion Drain outfalls just downstream from site CBE07 are also unlikely to be major sources of the PAHs found at site CBE07, given that relatively low PAH concentrations were measured in the drainage water of these drains during 2011 using passive sampling technology (Fisher 2013).

A proportion of the PAH contamination may be attributable to fuel from boating activity in the Swan Estuary. However, such a high accumulation of PAHs from boat fuel alone is unlikely, particularly at this specific site in the estuary. Given this and the history of Mardalup Park – formerly classified as a contaminated industrial site by the EPA (1992) with extensive PAH contamination from the historic East Perth Gasworks – the majority of the PAH contamination reported here and in previous studies (Nice 2009; Nice & Fisher 2011) is likely to have originated from this site. It is possible that either a) the PAH-contaminated sediments were not completely removed from the estuary bed during remediation in 1994, or b) PAH contamination from the adjacent historic contaminated site at Mardalup Park is entering/has entered the Swan Estuary via the groundwater; or a combination of these.

Remediation of the gasworks site and adjacent estuary between 1994 and 1996 involved the containment of residual PAH contamination onsite through the construction of a permanent barrier and the GID (Axis Environmental 1996). In 2009, ENV concluded that a groundwater mound existed in the lower middle section of the containment area (within Mardalup Park adjacent to this study's site CBE07 in the estuary); and that groundwater from the containment area had the potential to move to the estuary due to the level in the containment area being higher than that of the estuary. The ENV (2009) investigation also reported the presence of PAH contaminants (among others) in the groundwater at Mardalup Park, but suggested the PAHs were not 'contaminants of concern' because they were present in the estuary water at relatively low concentrations that did not exceed guidelines, and concluded the estuary was not being impacted by the site. The high sediment PAH concentrations reported here (the current study) do not contradict the ENV (2009) findings of relatively low concentrations in the water column because PAHs are hydrophobic, so are more likely to bind to sediment than remain in solution once in the environment (Latimer & Zheng 2003). However, the evidence presented here strongly suggests the likelihood of environmental impact at site CBE07 based on the range of ISQGs exceeded and, more explicitly, the high level of toxicity reported.

With regard to toxicity at other sites in this study, the next most notable toxicity was reported at CBE06 adjacent to the lake outfall at Burswood golf course. The sediments collected from here induced a toxic response in both fish larvae and amphipods. This toxicity may have been caused by a combination of the metals zinc and lead and the OC pesticides dieldrin and p,p'-DDE, as these were all present in concentrations that exceeded ISQGs. However, it is also possible that other contaminant(s) not tested for within the scope of this investigation contributed to the toxicity observed at CBE06 because the aforementioned contaminants did not always illicit a toxic response when present in similar concentrations at other sites in the estuary (CBE02, CBE08 and CBE10).

Toxicity was only experienced with one of the four test taxa at sites CBE01, CBE03, CBE04 and CBE09 and different contaminants or combinations thereof were likely to

be responsible, given that different test taxa demonstrated toxicity at these sites. The toxicity at CBE03 was considered low level because normal mussel development was only affected in the undiluted sediment elutriate and not with subsequent dilution-series testing. However, toxicity was considered high level at site CBE01 where normal mussel development was affected at lower concentrations of sediment elutriate. Sediment collected from sites CBE04 and CBE09 caused high-level toxicity in fish larvae, which exhibited a response at 25% sediment elutriate concentrations from these sites. No toxicity was reported for the test organisms and end points used in this investigation for sites CBE02, CBE08, CBE10 and the reference site, CBE11.

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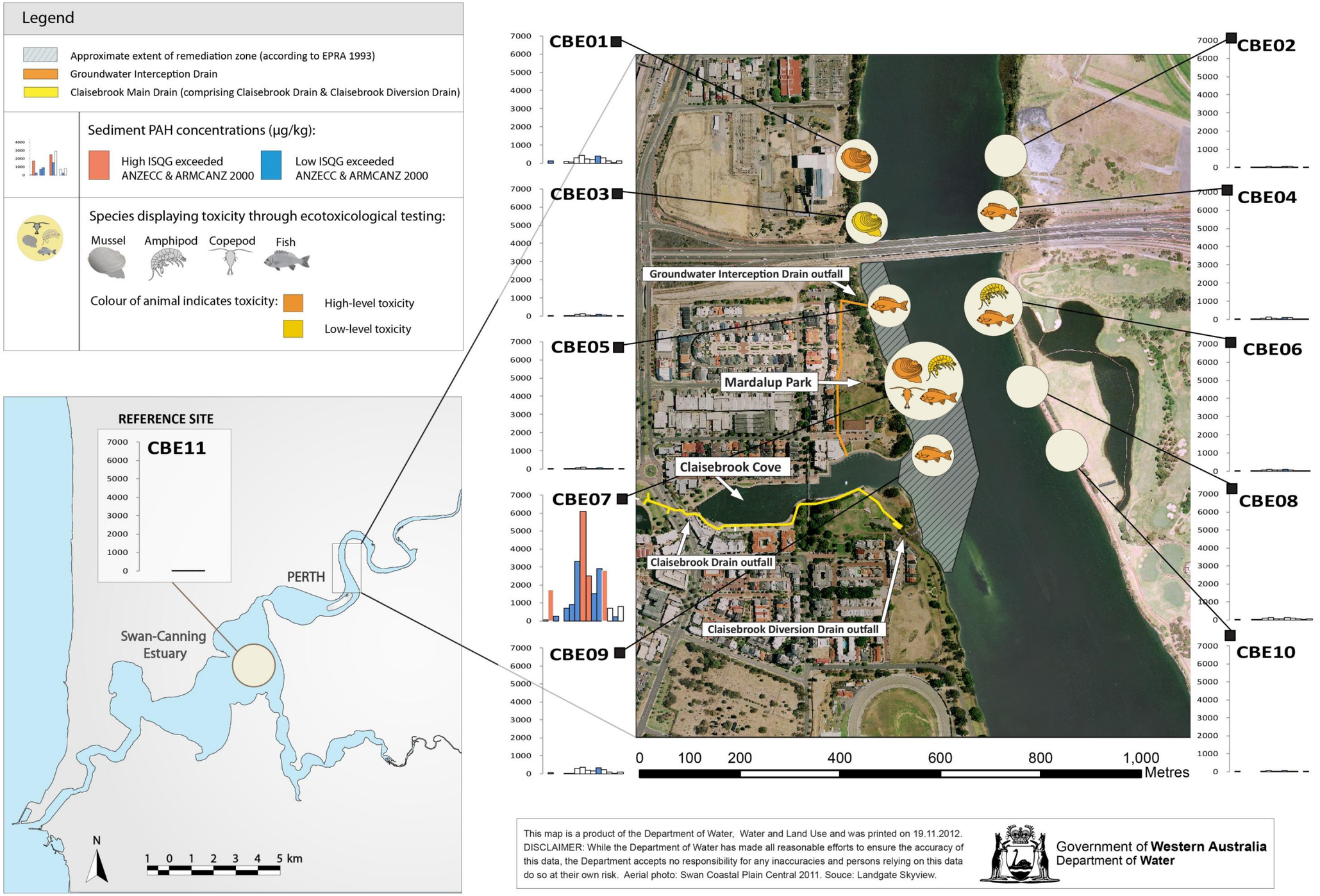

Figure 12 Spatial summary of toxicity and PAH contamination

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4.3 Toxicity assessment as a diagnostic tool for the situation at Claisebrook

The contaminants assessed in this study were prioritised in earlier assessments of the area (Nice 2009). However, it is acknowledged that they do not form an exhaustive list and many more contaminants are expected to be associated with the sediments in this part of the Swan Estuary, considering the urbanised nature of the Claisebrook catchment and other catchments draining to this area and the numerous historic contaminated sites bordering the shores of this part of the estuary (e.g. Lord 1999; Kesteven 2000). Therefore, in cases where metals, OC pesticides and PAHs have been suggested as contributing to the toxicity observed in this study, they are likely to be acting within a complex contaminant mixture and thus the combined effects of different contaminants should be considered. One of the benefits of toxicity testing is that the test organisms respond to the complex mixture and it is not necessary to have a full (and costly) inventory of the contaminants present within that mixture to determine whether the sediment (as a whole) is likely to affect ecosystem health.

The type of commercially available toxicity tests used in this study are designed to measure short-term acute and sub-chronic effects on aquatic organisms, and have been effective in classifying sites in terms of identifying obvious contaminant issues within the estuary and establishing or eliminating potential sources of acutely toxic contaminants. However, many of the contaminants measured here, particularly the high-molecular-weight PAHs, are known for their long-term chronic effects (e.g. Varanasi *et al.* 1985) given the following characteristics:

- endocrine disrupting (have the potential to interfere with hormonal systems of organisms often affecting sexual function – e.g. Oehlmann & Schulte-Oehlmann 2003)
- carcinogenic (have the potential to cause cancer – e.g. Murchelano & Wolfe 1985; Pinkney *et al.* 2009)
- teratogenic (have the potential to cause birth defects – e.g. http://www.epa.gov/region5superfund/ecology/toxprofiles).

Since this study did not target long-term chronic effects, it is not possible to predict the magnitude of the potential environmental impacts of these contaminants from this toxicity assessment alone. To measure effects outside of the short-term acute and sub-chronic measured here, investigation with a suite of bioanalytical assays specifically aimed at long-term endpoints is required, such as endocrine disruption and DNA damage (genotoxicity) (e.g. the Department of Water ecotoxicity toolbox – Reitsema *et al.* 2010).

With regard to chronic effects, a field study conducted in 2009 showed that mussels collected within Claisebrook Cove exhibited higher levels of DNA damage than those collected from the Swan Estuary, which may be attributable to stress from contaminant exposure (Rawson *et al.* 2011). The same study also presented evidence to suggest that fish collected from Claisebrook Cove had been exposed to

a range of stressors (e.g. PAH contamination) resulting in long-term chronic effects indicative of poor health. Observations included elevated hepatic detoxification enzymes and biliary PAH metabolites. Additionally, two of 15 fish collected also exhibited intersex condition (both male and female gonad tissue present), which may be indicative of endocrine disruption. However, given that black bream (the fish sampled in this study) are rudimentary hermaphrodites often displaying both male and female gonad tissue simultaneously (Buxton and Garatt 1990), controlled laboratory experiments would be required to confirm endocrine disruption due to contaminant exposure in this instance. Furthermore, given that fish are mobile it was not possible to attribute any of the chronic effects displayed in the fish to exposure of contaminants from Claisebrook sediments *per se*. Exposure of laboratory fish to field-collected sediment within a controlled laboratory environment would be required to further investigate the long-term chronic effects that the contaminants known to exist in the sediments may cause.

Even with the limitations of short-term acute and sub-chronic tests, the suite employed for the sediments at Claisebrook in the current study has been effective as a diagnostic tool. From the information presented here and previously (Nice 2009; Nice & Fisher 2011), there is a body of evidence sufficient to establish a likely impact on ecosystem health within the estuary such that future investigatory efforts should now be focused on detailed assessment of the adjacent contaminated site to confirm or eliminate this as a current source. Recommendations have been made accordingly (Section 6).

4.4 Toxicity and the guidelines

The guidelines used in this investigation were developed for general application across Australia based on available evidence pertaining to the impact of certain contaminants on biota under a range of environmental conditions. They are considered to be interim and the applicability of these ISQGs for south-west Western Australian species is to some extent unknown (given limited local studies). Assuming the PAH contaminants were the main cause for the toxic response observed, the toxicity results of this study are generally reflective of the available ISQG trigger values. That is, breaches of ISQGs were typically reflected by toxicity; and toxicity was not observed when ISQGs were not exceeded. This observation suggests the current thresholds are appropriate for south-west Western Australian species and should provide assurance in their use for such contaminants in this context.

5 Conclusions

From evidence presented here and in earlier studies (Nice 2009; ENV 2009; Nice & Fisher 2011), it has been clearly established that contamination exists in the Swan Estuary adjacent to Mardalup Park. The sites assessed in this investigation exhibited varying degrees of toxicity and contamination, yet the GID outfall to the estuary (the focus of this investigation) was not found to be the major source of contamination, given that both toxicity and contaminants (although present at the time of sampling) were not greatest at this site and there was no evidence of either a toxicity or contaminant gradient from the outfall.

At site CBE07 south of the GID outfall and adjacent to the middle section of Mardalup Park, a peak in both toxicity and contaminant concentrations was evident. When the data presented here were considered in conjunction with the findings of previous investigations (Nice 2009; Nice & Fisher 2011) and what is known of the history of Mardalup Park (Bowman Bishaw Gorham 1992; EPA 1992a; EPA 1992b), it was concluded that the primary source of the current PAH contamination in the estuary sediments is most likely the historic East Perth Gasworks site, either a) from residual contamination of estuarine sediments; or b) through PAH-contaminated groundwater that exists at Mardalup Park (ENV 2009); or a combination of both. Groundwater at the adjacent historic contaminated site at Mardalup Park was not assessed as part of this study, thus a direct link cannot yet be established.

6 Recommendations

Further toxicity and chemical assessments of the receiving environment at this time are not recommended (except for surveillance monitoring of toxicity and contaminant levels). Rather, management efforts should now be focused on establishing contaminant pathways to the estuary and addressing the reduction of contaminants within the estuarine sediments.

Specifically, it is recommended that:

1. A groundwater investigation of the historic contaminated site at Mardalup Park be conducted to determine whether PAH-contaminated groundwater from Mardalup Park is reaching the Swan Estuary; and furthermore to determine the specific pathway(s). This will support management decisions regarding remediation.
2. If/when remediation or other management intervention has occurred, subsequent toxicity and chemical assessment of the Swan Estuary adjacent to Mardalup Park be conducted to monitor the effectiveness of the management intervention.
3. Given that toxicity was observed in this investigation at two sites along the eastern bank of the estuary that may not be attributable to the historic contaminated site at Mardalup Park, the source of the toxicity at these sites be investigated. The outfall at Burswood Lakes should be a priority in these investigations, particularly since there are plans to develop the Burswood site. Information gained from such investigations will inform decisions on future management and development of the site.
4. Should the GID flow regime (quality and/or quantity) be altered in the future, toxicity and chemistry assessment be conducted to determine the potential impacts.
5. The Department of Environment and Conservation, Environment Protection Authority, Department of Fisheries, Department of Health and other relevant stakeholders be notified of the information presented in this report.

These recommendations are not to the exclusion of previous recommendations made in Nice and Fisher (2011).

7 Glossary and shortened forms

ANZECC	Australia and New Zealand Environment and Conservation Council
APHA	American Public Health Association
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
DDD	Dichlorodiphenyldichloroethane
DDE	Dichlorodiphenyldichloroethylene
DDT	Dichlorodiphenyltrichloroethane
Ecotoxicology	The integration of toxicology and ecology. Ecotoxicology aims to quantify the effects of stressors on natural populations, communities or ecosystems.
EC10	Concentration of sediment elutriate which causes the described effect in 10% of test organisms.
EC50	Concentration of sediment elutriate which causes the described effect in 50% of test organisms (median effect concentration).
EPA	Environmental Protection Authority
EPRA	East Perth Redevelopment Authority
ESA	Ecotox Services Australasia
GC-MS	Gas chromatography-mass spectrometry
GID	Groundwater Interception Drain
High-level toxicity	Statistically significant effect (statistically significant difference from the control organisms; $p<0.05$); and when subsequent dilution-series testing was performed, the statistically significant effect was observed with < 50% sediment elutriate concentration. [Definition determined for this study].
ISQGs	Interim Sediment Quality Guidelines (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 is the concentration below which the frequency of adverse biological effects is expected to be low. The *high* ISQG is the concentration above which adverse biological effects are expected to occur more frequently.
IC10	Concentration of sediment elutriate calculated (by non-linear interpolation) to cause the described effect in 10% of test organisms.
Limit of reporting	The lowest concentration at which an analyte will be reported after taking into account interferences and instrumental limits of detection.

Low-level toxicity	Statistically significant effect (statistically significant difference from the control organisms; $p<0.05$) observed with undiluted sediment elutriate concentration but there was no such effect when subsequent dilution-series testing was performed. [Definition determined for this study].
LOEC	Lowest observable effect concentration: the lowest tested concentration at which organisms are adversely affected compared with control organisms.
NOEC	No observable effect concentration: the highest tested concentration at which organisms are unaffected compared with control organisms.
No toxicity	No statistically significant effect (i.e. no statistically significant difference in response by the test organisms from the control organisms; $p > 0.05$).
OC	Organochlorine
PAH	Polycylic aromatic hydrocarbon
Pesticide	Substance or mixture of substances intended for preventing, destroying, repelling or mitigating pests such as insects.
SRRC	Swan River Reference Committee
SRT	Swan River Trust
Toxicity	The degree to which a substance or combination of substances is able to damage an exposed organism. In this study, different endpoints were employed for different test organisms to represent toxic effects: – mussel 72-hour larval development test: developmental abnormalities or developmental delays were used as a measure of toxicity – copepod 48-hour survival test: mortality was used as a measure of toxicity – amphipod 10-day whole-sediment survival test: mortality was used as a measure of toxicity – fish 96-hour larval imbalance test: imbalance (fish unable to maintain upright position in water column) was used as a measure of toxicity.
USEPA	United States Environmental Protection Agency
WFPHA	World Federation of Public Health Associations
WHO	World Health Organization

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9 Map disclaimer and data acknowledgements

The maps in this publication were produced by the Department of Water with the intent that they be used as illustrations in this report, *Ecotoxicological investigation of the Groundwater Interception Drain at Claisebrook in the Swan Estuary.* While the Department of Water has made all reasonable efforts to ensure the accuracy of this data, it accepts no responsibility for any inaccuracies and persons relying on this data do so at their own risk.

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Dataset name	Custodian	Metadata year
Swan Coastal Plain 30 cm	Landgate	1965
Swan Coastal Plain Central 15 cm	Landgate	2011

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