Assessment of seabed habitats and heavy metals in Cornelian Bay

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Summary

Cornelian Bay is a small embayment on the western shore within the middle reaches of the Derwent Estuary, a large drowned river valley situated in south-east Tasmania. While several studies have examined the physical setting of the estuary, there is little information on the distribution and structure of benthic habitats.

The distribution of sub-tidal habitats in Cornelian Bay were identified through a combination of field mapping using echo-sounders, grab sampling and video assessment. Habitats were classified as either seagrass, rocky reef, sand, silt/sand or silt. A number of transects were also conducted on seagrass habitats in order to describe the patterns of abundance of seagrass and associated macroalgae. In addition, short sediment cores (~10 cm) were taken at twenty-four sites in order to determine sediment particle size and examine the concentration of heavy metals at the surface. Long cores (~150 cm) were also taken at two sites to examine the vertical distribution of sediment particle size and heavy metals. A qualitative assessment of mollusc shell debris was also conducted on the long cores.

Seagrass habitats within Cornelian Bay were restricted to small beds that had a combined area of around 0.22 km^2. The beds consisted primarily of *Heterozostera tasmanica*, although small amounts of *Zostera muelleri* were present on the inner margin of beds in the middle of the bay. There was evidence of high algal epiphyte abundance, particularly in the middle of the bay, with the dominant species indicating the presence of excessive nutrients in the water column.

Overall, unvegetated habitats dominated Cornelian Bay representing around 80% of all subtidal habitats within the study area, although large differences occurred in the distribution of sediment type with increasing water depth. The shallow margins to around 1 m deep were dominated by sand. Sand/silt dominated the bay, with a gradual transition from silty/sand to silt occurring at a depth of around 6 m.

Heavy metal analyses of surface and sub-surface sediments in Cornelian Bay revealed that metal contamination was mostly restricted to water depths >5 m where the sediments were siltier. In this central bay area the top 20-35 cm of sediment was contaminated with heavy metals. The highest concentrations occurred at ~10 cm below the sediment surface and arsenic, cadmium, copper, lead and zinc levels exceed the ANZECC interim sediment quality guideline (ISQG) high values by factors of ~5, 10, 2, 10 and 44 times, respectively. The long cores reveal that level of heavy metal contamination in the surface sediments has declined over recent years. Heavy metal concentrations in the nearshore sandy sediments were typically within the acceptable range, although lead concentrations exceed the ISQG low value at all sites and mercury exceeded the ISQG high value. As disturbance of the sediments could potentially result in resuspension and redeposition of the most highly

contaminated sediments, it is recommended that activities which would disturb the silty sediments in the central bay area are restricted.

The shift from natural to anthropogenic levels of heavy metals in the sediments is used as a marker for 1917, the year that the Electrolytic Zinc Company was commissioned and large quantities of heavy metal contaminants were first introduced to the Estuary. Sedimentation rates in Cornelian Bay are estimated to be between 2-4 mm/yr, which fall within the typical sedimentation rate estimates of 2-6 mm/yr obtained for other subtidal embayments within the Derwent Estuary.

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1. Introduction

Cornelian Bay is a small embayment located on the western shore in the middle reaches of the Derwent Estuary, a large drowned river valley that extends for a distance of 52 km and covers an area of around 198 km^2 between Iron Pot and New Norfolk in south-east Tasmania (Fig. 1). Cornelian Bay is an important recreational area close to the centre of Hobart and has numerous boathouses on the southern shore. Land use in the bays catchment is dominated by urban and industrial use, although small areas of woodland and forest are also present. Several stormwater outlets discharge into the bay with high nutrient levels associated with high rainfall events.

In 2002, the Hobart City Council initiated an environmental assessment of Cornelian Bay with support from Riverworks, a component of the Natural Heritage Trust. The goal of this 2-year management initiative is the preparation of an environmental management strategy for Cornelian Bay, together with an associated long-term monitoring program. Management of nutrient inputs in the bay has been identified as a priority within this project. In addition, a study of the distribution of sediment type and heavy metal concentrations, seabed habitats and water currents were identified as important components of an environmental assessment.

Overall, around 99% of the seabed of the Derwent Estuary is a range of soft-sediment habitats (sand, silty/sand, sand and seagrass), with large differences in the type of sediment between the lower, middle and upper reaches (Jordan *et al*. 2001). The middle reaches of the estuary, which extends from the Tasman Bridge to the Bridgewater Causeway, is around one km wide in most places and contains numerous shallow embayments on both shorelines, including Cornelian Bay. The seabed habitats in the middle estuary are silt dominated below depths of around 2 m, with sand occurring close to shore, particularly along the eastern shore (Jordan *et al*. 2001).

This region of the Estuary has had significant inputs of nutrients, organic matter and heavy metals and arsenic, resulting in considerable degradation of sediment condition (Coughanowr, 1997). These have resulted from both point-source (sewage and industrial discharges) and diffuse-source inputs (primarily urban and agricultural runoff). Heavy metal contamination commenced in 1917 with the commissioning of the Electrolytic Zinc Company, now known as Pasminco. Concentrations of mercury, lead, zinc and cadmium have been found to be consistently high in both sediments (Bloom, 1975; Pirzl, 1996) and biota (see Coughanowr, 1997). In a recent comprehensive survey of heavy metals in the sediments of the estuary, the entire middle estuary had mercury, lead and zinc concentrations above the high category of the ANZECC guidelines (Coughanowr 2001). This survey was mostly restricted to depths approximately >4 m and thus the sediment heavy metal concentrations in shallow subtidal and intertidal areas have not been examined.

The nutrient, sediment and organic matter inputs have resulted in significant changes to the sediment structure and macroinvertebrate communities, although some improvements have been noted in specific areas (Moverley and Garland, 1995). There has also been a considerable decline in the densities of benthic invertebrate species in the lower Derwent Estuary, largely due to the presence of introduced marine pests (particularly seastars *Asterias* *amurensis* and *Patiriella regularis*, the gastropod *Maoricolpus roseus*, chiton *Amaurochiton glaucus*, ascidian *Ascidiella aspersa* and crab *Cancer novaezelandiae*) (Morrice, 1995).

Therefore, it is clear that numerous environmental issues have impacted the seabed habitats of the Derwent Estuary, particularly within the middle reaches. However, details of the levels of impact in most of the specific embayments, including Cornelian Bay have not yet been investigated. There is little historical baseline data with which to examine trends in toxicant concentrations, habitat distribution and sediment structure. Furthermore, as sedimentation rates are not known, the length of time required for burial of contaminated sediments by natural processes remains uncertain. Therefore, the aims of this study were to:

- map existing seabed habitats within Cornelian Bay and estimate seagrass and associated algal abundance;
- determine the distribution of heavy metal and arsenic contamination in surface sediments
- examine the historical patterns of heavy metal and arsenic concentrations and sediment type
- determine the speed and direction of water currents in the bay.

Fig. 1. Location of Cornelian Bay in the middle Derwent Estuary

2. Methods

2.1 Habitat mapping

Mapping of seabed habitats in Cornelian Bay was conducted in November 2002 from FRV *Nubeena II*. The vessel was driven along a series of transects set approximately 40 m apart over the survey area. A Furuno 600L colour sounder was used for habitat discrimination with different substrate types providing differing traces based on their roughness and hardness. This signal was interpreted in the field and logged against depth and position. The exact location of boundaries between patches of differing habitat and the depth they occurred at was recorded using a Garmin 135 GPS map unit coupled with a RACAL differential correction unit. This information was logged onto a laptop computer directly into *SeaBed Mapper 2.4*. The habitat types were classified as seagrass, sand, silt/sand, silt and rocky reef, with classifications of unvegetated habitats supplemented with particle size information (see section 2.3).

A submersible digital video camera was deployed along a series of transects in order to verify echo-sounder classifications and obtain more detailed information on habitat attributes. Such attributes include the dominant algae, seagrass and sessile invertebrates present on soft-sediment substrates.

Maps were generated through on screen digitising of habitat boundaries in *ArcView* (ESRI). The field data produced point locations of habitat type which were used as the basis for the generation of habitat polygons. At the scale of 1:2,000, points were connected to form polygons of similar habitat type. The outer boundary of the polygon was identified in the field and the polygon line deflected from the outer point by no more than 5 m.

The Tasmanian coastline and tidal zone data layer information was supplied by the Land Information Services Tasmania, Department of Primary Industry Water and Environment. The Tasmanian Tidal Zone 1:250,000 depicts large areas between Mean High Water Mark and Mean Low Water Mark.

Depth measurements taken from a Garmin sounder were used to construct a contour layer. These depths were corrected for tidal variation based on existing tide tables using the formula:

$$D\mathit{i} = h_1+(h_2-h_1)*(\cos(\pi*((t-t_1)/(t_2-t_1)+1))+1)/2$$

where $h_{1,2}$ correspond to the heights of the high and low tides, $t_{1,2}$ are the times of the high and low tides with t being the current time. All depth measures were then corrected to Mean Sea Level.

Depth contours were constructed through interpolation of the depth corrected field data using a Triangular Irregular Network (TIN) data model. Full details of contouring methods are presented in Barrett *et al*. (2001).

2.2 Seagrass and algal sampling

Four transects were surveyed in Cornelian Bay in November 2002 to examine the structure of the seagrass beds in more detail and to estimate seagrass abundance (Fig. 2). Surveys involved swimming along 100 m transects generally running perpendicular to the shore from near the inner margin of the bed to the deeper outer margin, although not all transects reached the outer margin. Average blade length and percentage cover of seagrass was estimated at four 50 cm quadrats at 10 m intervals along each transect.

In addition, the abundance of filamentous algae in each quadrat was ranked from 1-5 on each of the seagrass transects sampled, reflecting a combination of percentage cover and algal depth. The dominant species of algae was identified to the lowest taxonomic level possible in the laboratory.

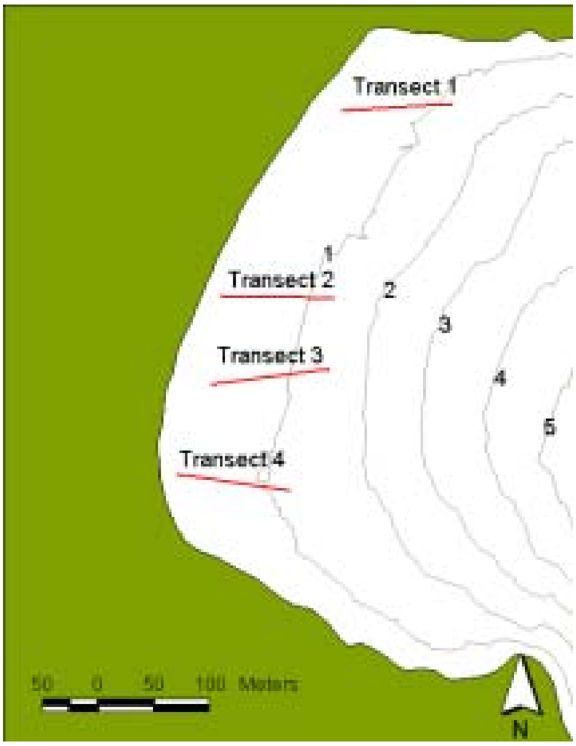

Fig.2. Position of seagrass sampling transects and depth contours in Cornelian Bay

2.3 Sediment sampling

A more detailed survey of soft-sediment habitats was conducted in the bay as part of the field mapping. The purpose of the sediment sampling was twofold – to obtain a qualitative assessment of sediment type (sand, silt/sand and silt) to delineate sediment boundaries, and to collect samples for analysis of heavy metals (lead, zinc, copper, cadmium, cobalt, chromium, iron and manganese), arsenic and sediment particle size. Sediments were surveyed at a total of 24 sites throughout Cornelian Bay, with 20 of these used for heavy metal and arsenic analysis (Fig. 3). In addition, sediment from 3 sites (5, 19 and 20) were also analysed for mercury. At each site a sediment sample was taken with a 3 cm diameter core tube that samples at least the top 10 cm. The surface water was removed and the core tube capped and placed in a vertical position in an esky for transport back to the laboratory.

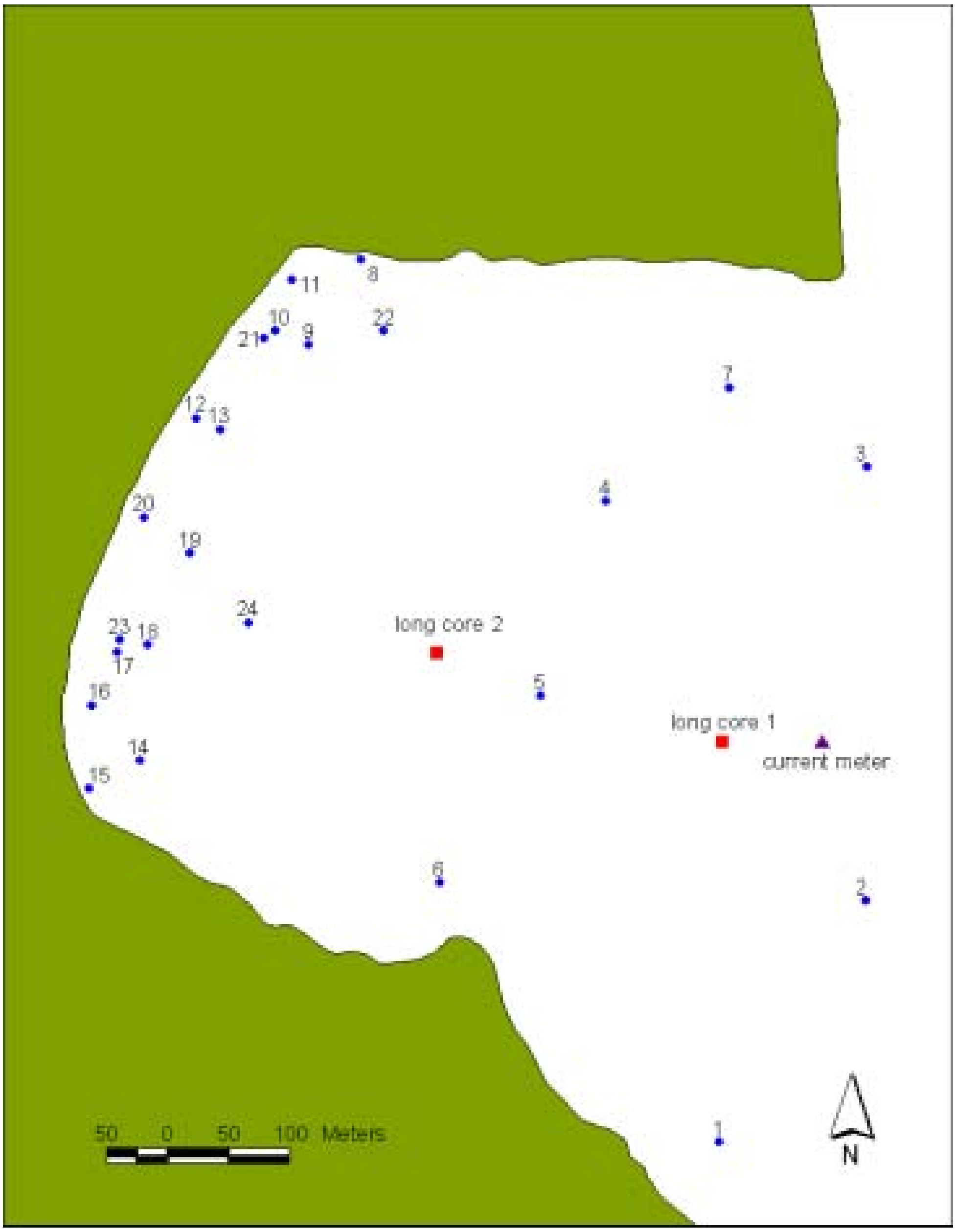

Fig. 3. Short and long-core sediment sampling sites in Cornelian Bay. Sites labelled in red indicate those which were also analysed for heavy metals and arsenic.

2.4 Long core sampling

Long sediment cores were taken from two sites in Cornelian Bay at depths of around 10 m (LC1) and 4 m (LC2) (Fig. 3). The sediment cores were collected by scuba divers pushing a 2 m long 10 cm diameter PVC pipe into the sediment. Once the pipe had penetrated as far as possible the distance from the top of the core tube to the sediment surface both inside and outside the pipe were measured in order to estimate the amount of sediment compaction due to core collection techniques. The top of the core tube was then capped and the tube withdrawn and immediately capped at the bottom. Once on the boat the water above the sediment was removed and a polystyrene plug inserted to minimise disturbance of the sediment surface. The cores were then returned to the laboratory and stored at around 10°C until processing.

The cores were split in half longitudinally, photographed and described. Samples for heavy metal, arsenic and particle size analyses were taken every ~5 cm along the core over a 1 cm interval from the same slice.

2.5 Current meter

Information on the speed and direction of water currents in Cornelian Bay was obtained by an Acoustic Doppler Current Profiler (ADCP). This is a type of current meter that measures speed and direction of currents throughout the water column using acoustic signals. The ADCP was moored at site at the mouth of the bay in a depth of around 10 m for a period of ten days between 13-22 November 2002 (Fig. 3).

2.6 Sediment particle size analysis

The distribution of sediment particle size was determined in the top 2 cm in the short cores and at approximately 5 cm depth intervals through the long cores. Samples were first dried in the oven at 30^{o}C until no further weight loss occurred. The dry weight of the bulk sample was recorded before being wet sieved through a series of 8, 4, 2, 1, 0.5, 0.25, 0.125 and 0.063 mm sieves. The material retained on each sieve was placed in a tray and oven dried before being weighed. The amount <0.063 mm was determined by subtracting the combined dry weight of the other fractions from the dry bulk sediment weight.

2.7 Heavy metals analysis

Heavy metals analyses were undertaken at Analytical Services Tasmania (AST). Approximately 1-2 g of dried sediment was digested for four hours in 5 ml aqua regia at 100° C and diluted to 50 ml with deionised water prior to analysis by Inductively Coupled Plasma Atomic Emission Spectrometry (ICPAES).

As the method employed for determining sedimentation rates is reliant on identifying changes in heavy metals concentrations, it is necessary to standardise for changes in sediment particle size. This is due to the fact that concentrations of heavy metals in sediments are related to particle size, with increasing concentrations occurring in sediments with an increasing proportion of finer sediments. To adjust for this the following equation was applied to the metal concentrations determined from the long cores:

Standardised heavy metal concentration = (raw heavy metal concentration / %<0.063mm)*100

3. Results

3.1 Seabed habitats

3.1.1 Seagrass and associated algae

Seagrass beds occurred throughout most of the shallow depths of Cornelian Bay covering an area of around 0.22 km^2 (Fig. 4). The beds were dominated by *Heterozostera tasmanica* which occurs out to around the 2 m depth contour, while *Zostera muelleri* occurred in a narrow band in the intertidal zone. There were, however, variations in the abundance and patchiness of the seagrass throughout the bay. In general, sparse-patchy seagrass occurred in the shallow depths in the northern and middle sections of the bay and in depths of around 1-2 m on the southern end. A band of dense seagrass occurred around the 1 m depth contour in the southern end with an inner margin of sparse seagrass.

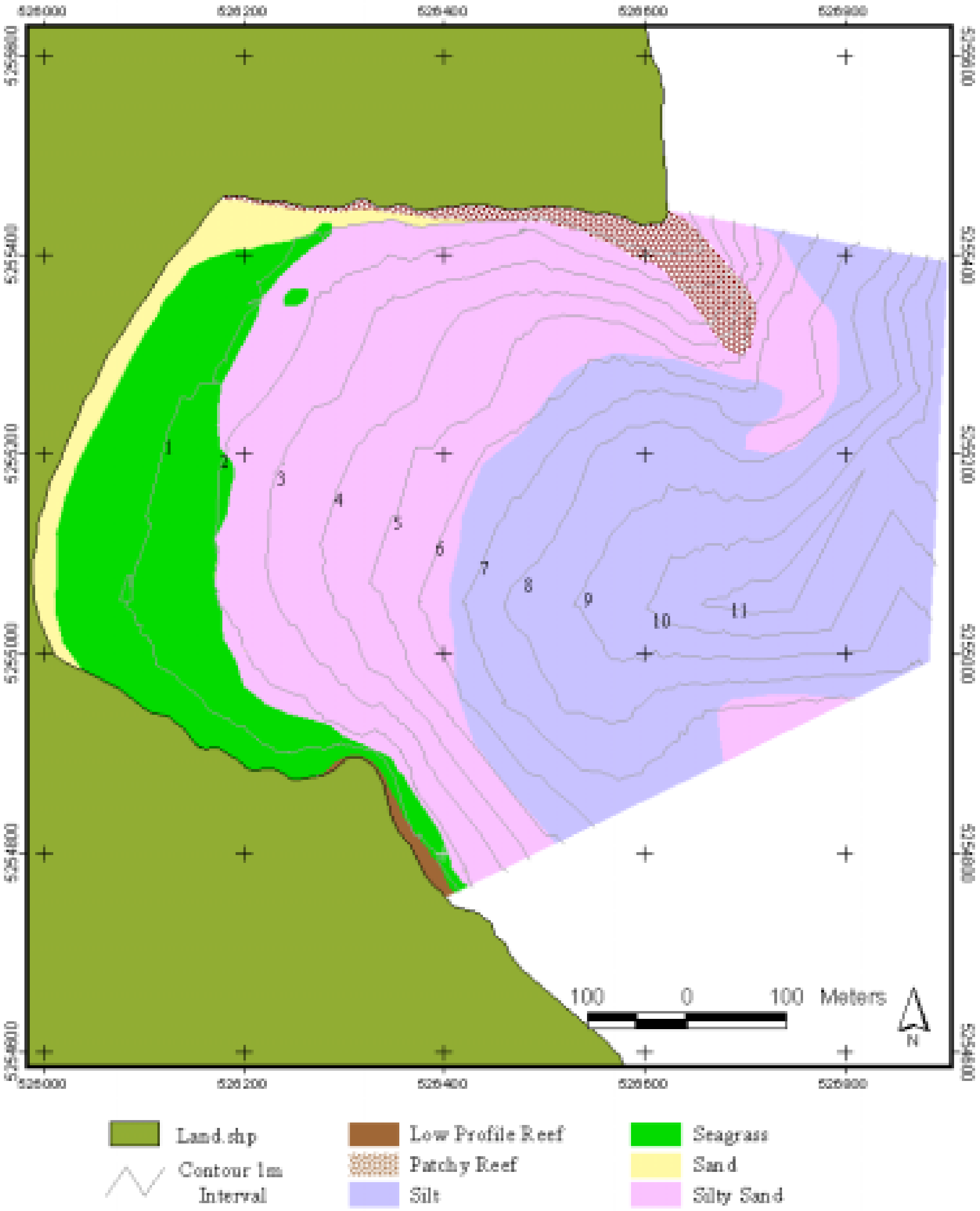

Fig. 4. Distribution of seabed habitat types and bathymetry in Cornelian Bay

These patterns are reflected in the estimates of seagrass abundance from the four transects. In general, seagrass blades were between 10 and 20 cm long throughout the bay with variations in abundance reflecting variations in percentage cover (Fig. 5). In the northern end (transect 1) cover was consistently <20% with evidence of higher abundance on the deeper margins. In the middle of the bay (transect 2) the cover varied between 20 and 70% reflecting a high abundance and patchy cover. Transect 3 had generally low abundance with cover mostly <20%. In the southern end of the bay (transect 4) the seagrass abundance was high, with cover generally around 40-50%.

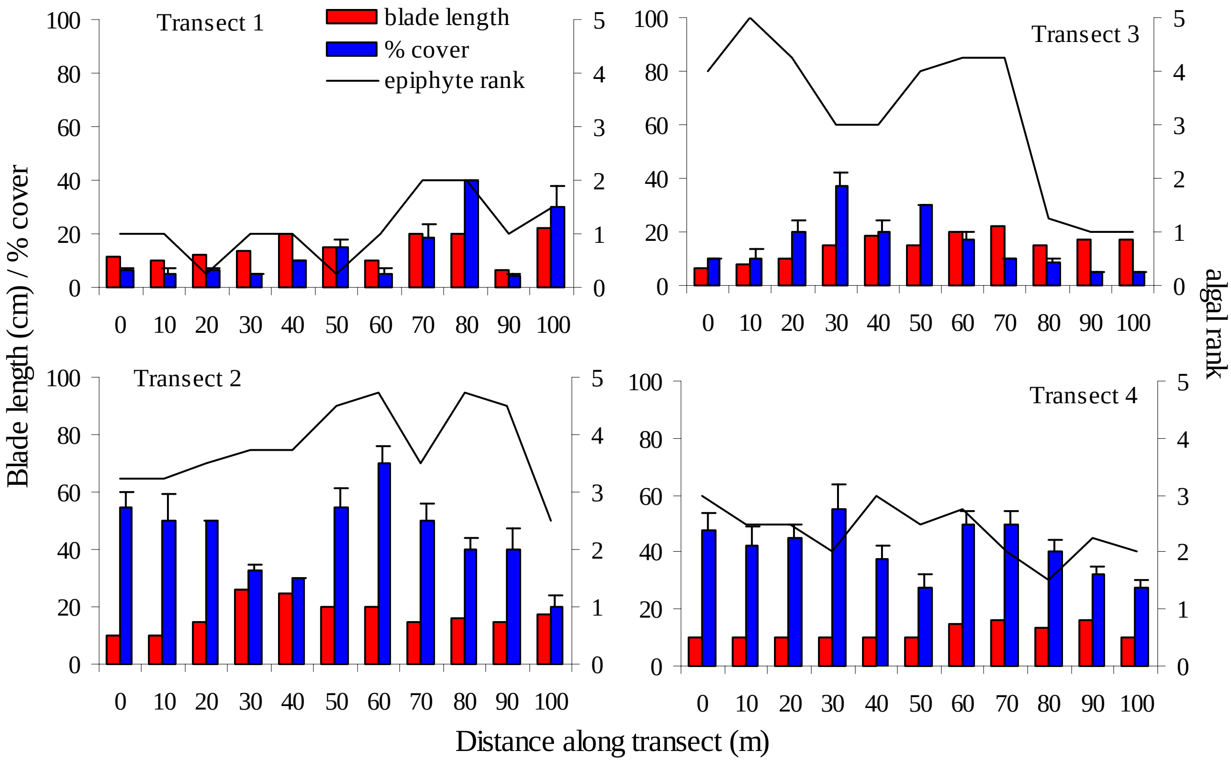

Fig. 5. Estimate of seagrass blade length (cm), percentage cover and ranked algal abundance along four transects in Cornelian Bay

The seagrass beds in Cornelian Bay were subject to highly variable abundances of associated macroalgae, with no obvious relationship between seagrass and algal abundance. Algal abundance was highest in the middle of the bay (transects 2 and 3), with large mats of unattached algae present during the sampling period (Fig. 5). On transect 3, adjacent to the stormwater outfall, there was very high biomass in the intertidal zone, which only decreased at a distance of around 80 m offshore.

A number of species of macroalgae are associated with the seagrass beds in Cornelian Bay, although at most sites the overall abundance was dominated by filamentous species, primarily *Polysiphonia* spp. and *Enteromorpha* spp.

3.1.2 Subtidal unvegetated habitats

Unvegetated habitats dominated the seabed in Cornelian Bay in depths >2 m (Fig. 4). The sediment type varied according to depth reflecting the influence of wave energy on the transport of sediment particles. The beach, intertidal zone and out to around 1 m was dominated by sand. However, within this region the sediment type within the seagrass bed often contained a higher proportion of finer sediment due to the effect of seagrass reducing wave energy and hence sediment movement. A band of silty-sand occured from around 1 m to 6 m, with silt dominating the seabed in deeper water (Fig. 4).

A general correlation between the area of silt deposition and the deeper portion of Cornelian Bay indicates that it is a lower-energy environment where fine suspended sediment is being deposited. This is reflected in the sediment particle size which shows the shallow depths having a greater proportion of coarser sediment than those in deeper water where fine sediments dominate (Fig. 6). For example, shallow intertidal sites (8, 15 and 20) generally have a higher proportion of fine to coarse sand (ie. 0.125-1.0 mm) than those in deeper water (1, 2 and 3). The silt proportion (<0.063 mm) was highest at site 2 located in around 10 m depth, although some medium to very fine sands also occurred at this depth. As the depth in the bay increased only gradually the boundaries between sediment types defined in Fig. 4 are generally broad and represent a continuum.

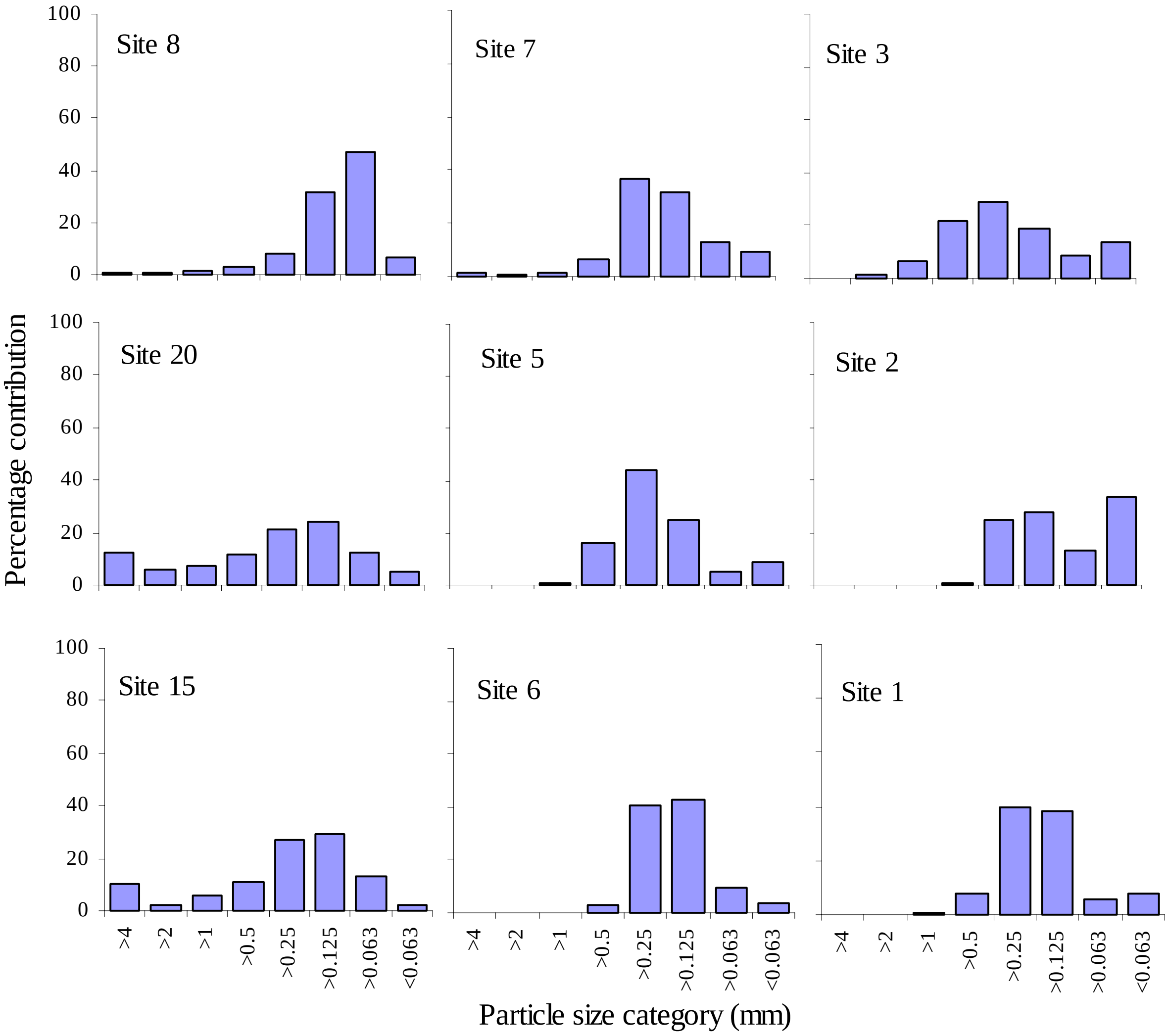

Fig. 6. Distribution of sediment particle size at selected sites in Cornelian Bay

3.2 Surface Heavy Metal and Arsenic Concentrations

A range of heavy metals (lead, zinc, cadmium, cobalt, chromium, copper, iron, manganese) and arsenic concentrations in the surface layer of the sediments (to a depth of 5 cm) were determined at 20 of the sediment sample sites, with three of these sites also sampled for mercury (see Fig. 3). Only those elements that have established sediment quality guidelines defined in ANZECC (2000) are presented. These elements have defined interim sediment quality guideline (ISQG) values of low (causing adverse effects 10% of the time) and high (causing adverse effects 50% of the time), which are based on the potential biological effects of contaminants in sediments.

In general, heavy metal and arsenic concentrations are closely related to sediment particle size and water depth, with sites with a higher proportion of very fine sand and silt having higher concentrations (Fig. 6). Site 2 had levels for all toxicants above the high ISQG levels defined in the ANZECC (2000) guidelines. In addition, cadmium, zinc and lead concentrations exceed the high ISQG levels at sites 3, 5 and 7, while zinc and lead were also above the high levels at site 4. Lead concentrations also exceeded the low ISQG levels at all sites indicating very broad lead contamination of sediments in Cornelian Bay, including the intertidal zone.

Mercury concentrations exceeded the high ISQG levels at two sites (5 and 20) and the low ISQG level at site 19 (Fig. 7). While levels were almost an order of magnitude higher in the deeper, siltier site 5 compared to those inshore, the concentrations exceeding the ISQG high levels in the intertidal area are in contrast to the other heavy metals examined.

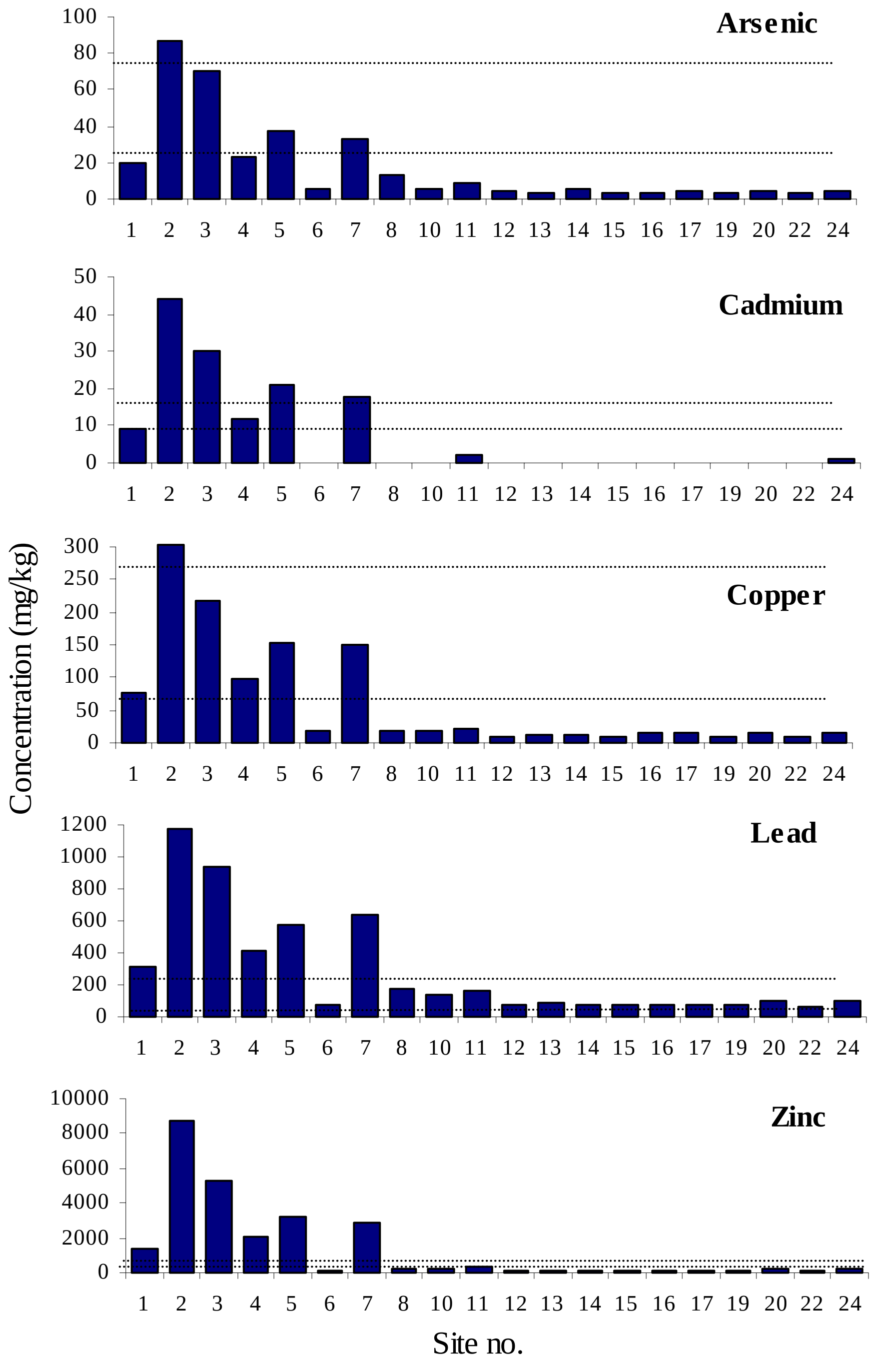

Fig. 6. Distribution of arsenic, cadmium, copper, lead and zinc concentrations in Cornelian Bay surface sediments. The dotted lines refer to high and low ISQG levels defined in ANZECC (2000).

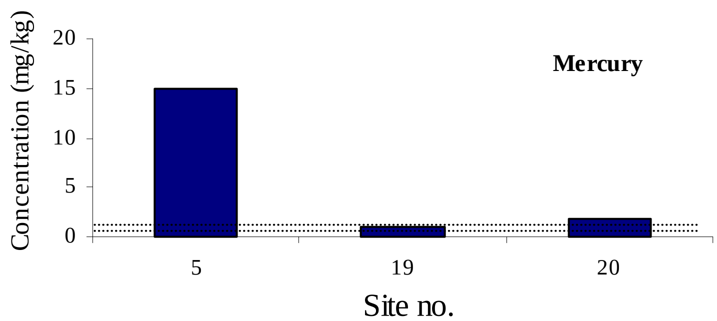

Fig. 7. Distribution of mercury concentration in Cornelian Bay surface sediments. The dotted lines refer to the high and low ISQG levels defined in the ANZECC (2000) guidelines.

3.3 Long Core Descriptions

3.3.1 Sediment particle size profiles

Long core 1 (LC1) was 110 cm in length and generally consisted of dark grey silty/sand, although the sediment structure changed along the core (Fig. 8A). The bottom of the core consisted predominantly of silt (<0.063 mm), which increased from around 45% at 100 cm to 90% at 60 cm (Fig. 9A). At the same depth below the surface (~60 cm) there was a rapid decrease in the proportion of very fine (0.125-0.063 mm) and fine sands (0.25-0.125 mm). Between 60 and 20 cm the silt proportion decreased to around 70% with the amount of very fine and fine sands remaining low. Above 20 cm the silt proportion increased again to 85% and the amount of very fine sand decreased. There was little evidence of shell material throughout the entire core although small amounts are present below around 70 cm.

Long core 2 (LC2) was taken directly inshore of LC1 in a depth of 4 m (see Fig. 3). The core was 78 cm long and consisted mostly of medium sand in the lower section below around 40 cm and medium to fine sand above (Fig. 8B, 9B). The silt content of the core was around 5% throughout. The high percentage of particles >4 mm (ie. ~20%) below 40 cm represents the abundant shell material in the lower section of the core. The dominant species present in the bottom of the core include the bivalves *Fulvia tenuicostata, Dosinia* sp., *Tellina* sp., *Pecten fumata* (commercial scallop) and *Ostrea angasi* (native oyster). Further work is required to complete a faunal list of species in the core but was outside the scope of this study. In contrast, there was little shell material in the upper section of the core with the transition between the two distinct regions occurring at ~35 cm (Fig. 8B).

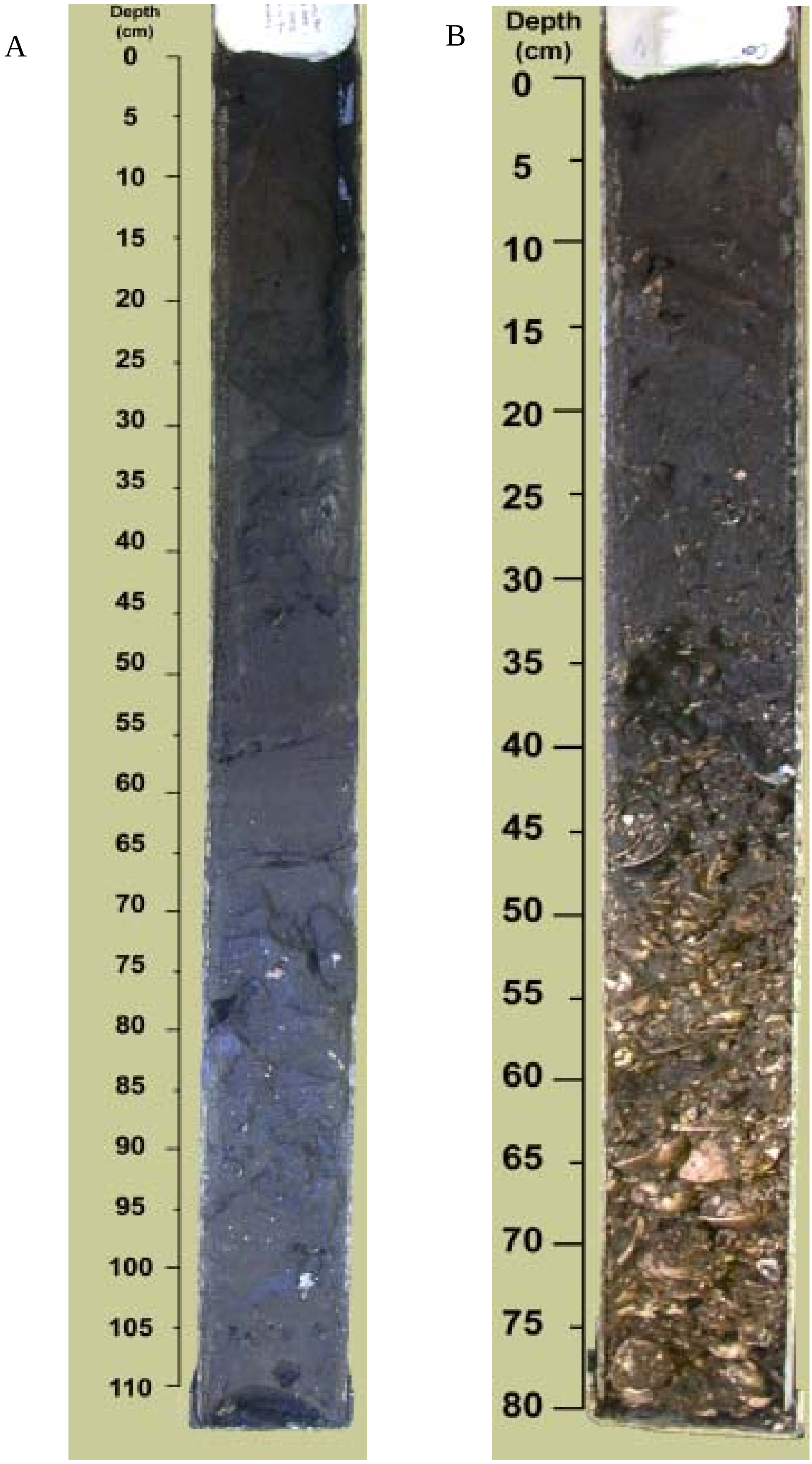

Fig. 8. Photograph of (A) long core 1 and (B) long core 2 from Cornelian Bay

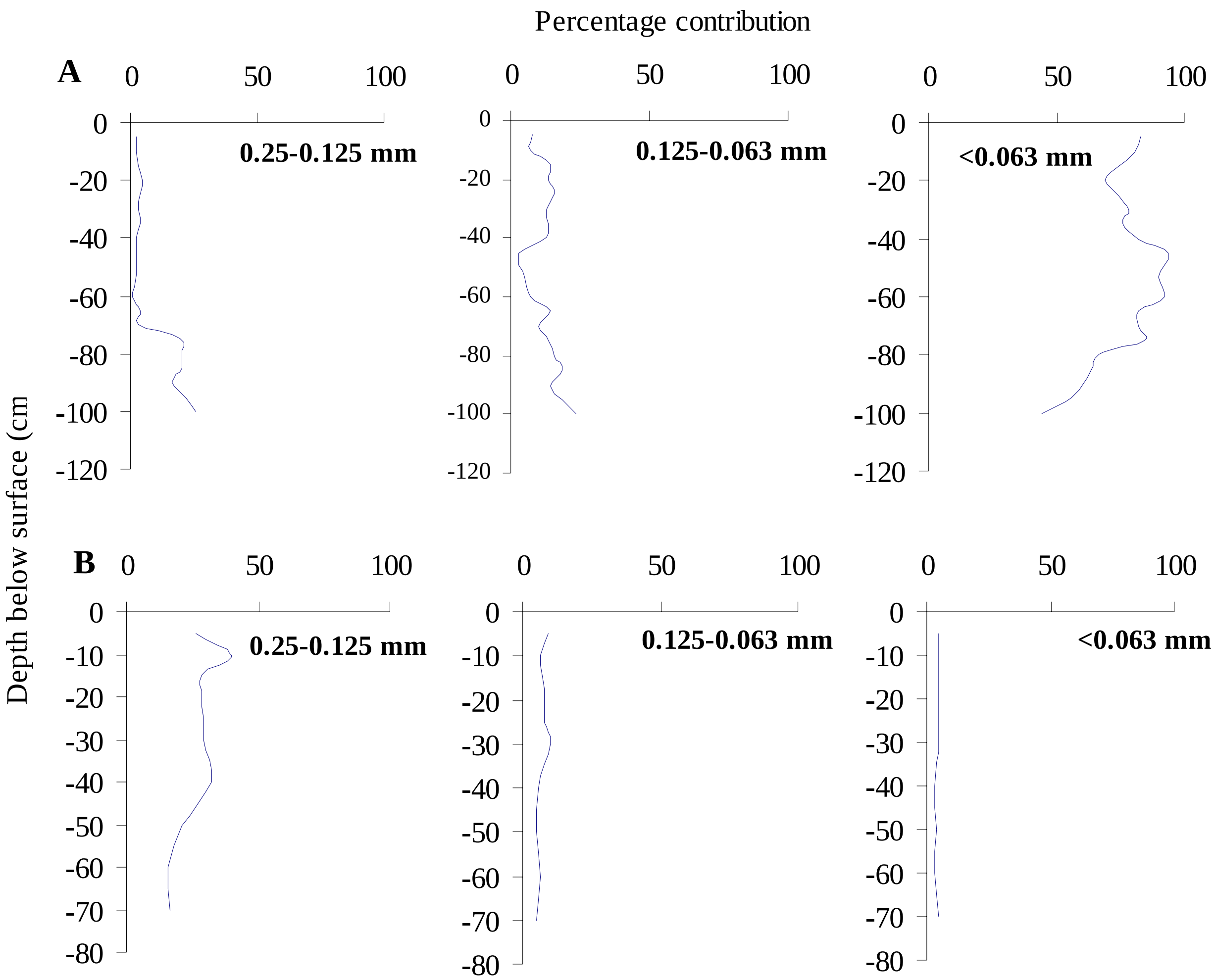

Fig. 9. Depth profiles of selected sediment particles categories in (A) long core 1 and (B) long core 2 from Cornelian Bay

3.3.2 Heavy metal and arsenic profiles

Heavy metal and arsenic concentrations were determined at 5 cm intervals along both long cores down to a maximum depth of 100 cm in LC1 and 70 cm in LC2. Firstly, LC1 concentrations for all elements increased approximately three-fold from the surface to around 10 cm before rapidly decreasing to low levels at around 30 cm depth (Fig. 10). Maximum levels of zinc, lead, cadmium, copper and arsenic were 18000, 2080, 104, 522 and 331 mg/kg, respectively, all of which are considerably above the ISQG high levels. The background levels below 30 cm for arsenic, cadmium, copper, lead and zinc in LC1 are 12 mg/kg, <1 mg/kg, 14 mg/kg, 7 mg/kg and 50 mg/kg, respectively.

Trends in heavy metal and arsenic concentrations with depth in LC2 are generally consistent with those in LC1, although concentrations are generally an order of magnitude lower. Peak concentrations for zinc, lead, cadmium, copper and arsenic are 1320, 312, 9, 72 and 30 mg/kg, respectively, with these occurring at around 10 cm below the surface (Fig. 11). The lower concentrations generally reflect the coarser sediment type within LC2. The peak values for zinc and lead are the only two elements that exceed the high ISQG level, with all elements decreasing to background levels at around 20 cm below the surface.

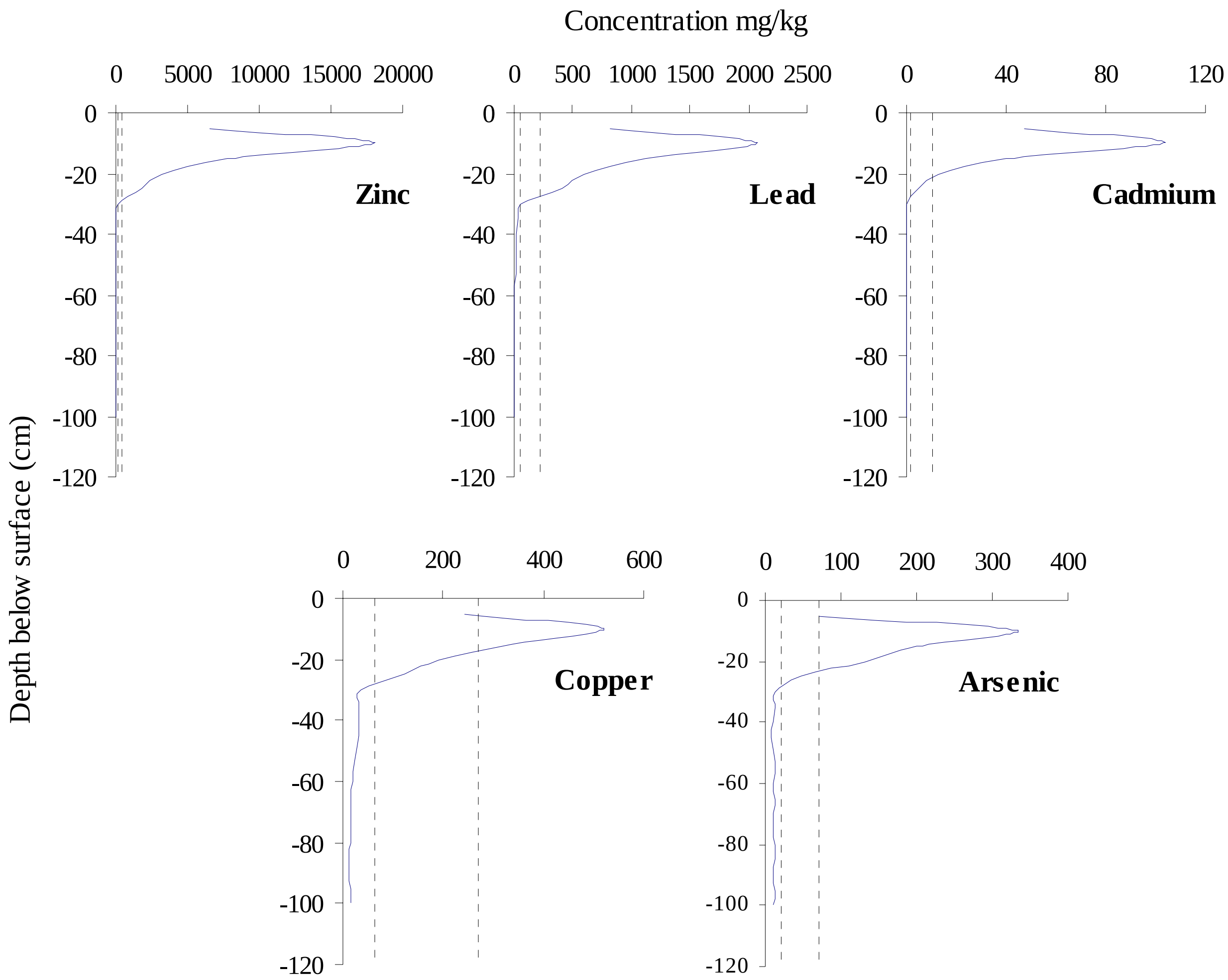

Fig. 10. Depth profiles of concentrations for zinc, lead, cadmium, copper, chromium and arsenic in long core 1 in Cornelian Bay. The dotted lines refer to high and low ISQG levels defined in ANZECC (2000).

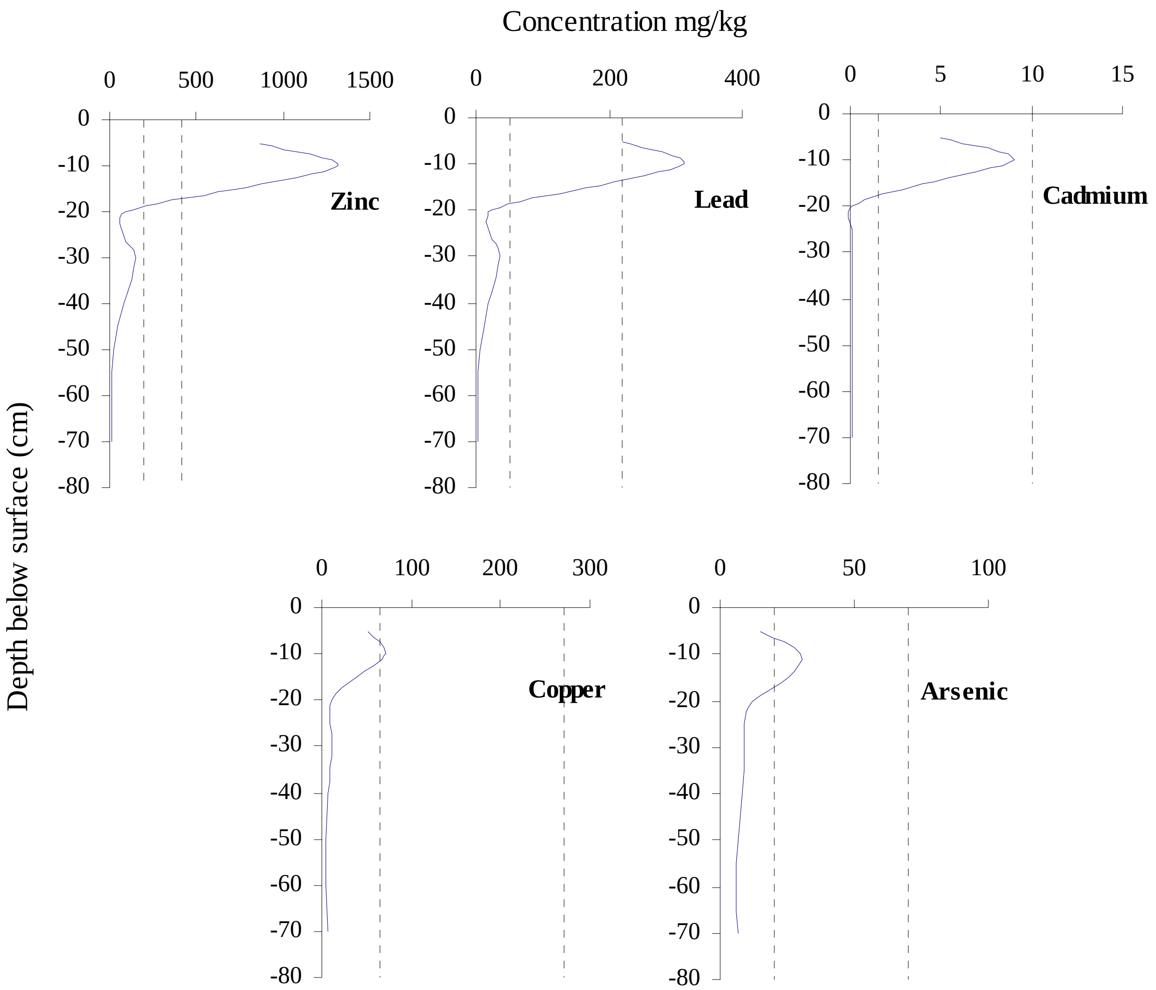

Fig. 11. Depth profiles of concentrations for zinc, lead, cadmium, copper, chromium and arsenic in long core 2 in Cornelian Bay. The dotted lines refer to high and low ISQG levels defined in ANZECC (2000).

3.4 Current flow

Information on current speed and direction was obtained in 2 m depth intervals starting from 3 m above the seafloor to approximately 2-3 m below the surface (Table 1). However, only every second interval is presented here. The average flow rate near the seabed was around 3.4 cm/sec and flow was less than 0.5 cm/sec for more than 75% of the time. A maximum flow rate of 91.8 cm/sec was recorded in the surface waters. The flow near the surface was predominantly to the south-east reflecting the prevailing winds from the north-west during the deployment period. At 3-5 m above the seabed current direction tended more towards the south-west, but at slower speeds.

Table 1. Current meter information at specific depth intervals through the water column in Cornelian Bay

Distance above seabed	Average (cm/sec)	Max (cm/sec)	Min (cm/sec)	% <0.5 cm/sec	% >10 cm/sec
3-5m	3.4	13.3	0.1	77.5	1.6
7-9m	25.4	66.2	0.2	7.8	81.8
11-13m	7.7	91.8	0.2	37.4	26.7

4. Discussion

4.1 Seabed Habitats

4.1.1 Seagrass and associated algae

Seagrasses are marine plants that are adapted to soft-sediment habitats in coastal waters. They are known to play an important role in maintaining sediment stability and water quality, particularly within estuaries. They are also an important habitat for a wide range of fish and invertebrate species and generally have a distinct community compared to adjacent unvegetated habitats (Jordan *et al.*, 1998). They are also particularly important areas of primary and secondary productivity, with most seagrass production not utilised *in situ* but exported from the beds.

The distribution of seagrass is influenced by several factors including depth, wave action or exposure and nutrient and light availability. Seagrass beds in Cornelian Bay occurred as a broad band across the intertidal and shallow sub-tidal depths of the entire bay. The beds consisted primarily of *Heterozostera tasmanica*, although an inner margin of *Zostera muelleri* also occured in the intertidal zone. The beds generally extended to around 2 m deep, with the outer margin most likely limited by light availability.

There were considerable differences in the abundance of seagrass between areas in the bay. While qualitatively this appears to reflect variations in the relative ‘health’ of the beds at sites around the bay, it is not possible to attribute this entirely to human induced impacts. It is well established that increased nutrient levels and turbidity from urban and industrial discharges and catchment usage are the key causes of light reduction and the subsequent decline of seagrass beds (Walker and McComb, 1992). High nutrient levels often result in increased epiphytic algal growth that can smother and shade seagrass blades, while higher turbidity reduces that amount of light reaching the beds, with deeper parts of the bed most vulnerable to light reductions.

There was evidence of large amounts of filamentous algae associated with the seagrass beds in Cornelian Bay. This was particularly evident in the middle of the bay where large mats of unattached algae were present. Many factors influence the abundance and species composition of algae associated with seagrass beds. The temporal variations are primarily driven by nutrient availability, temperature, salinity, light availability and seasonality of algal reproduction. As different species of macroalgae are able to utilise nutrients at different concentrations and rates a small number of filamentous species tend to dominate the algal community. This was the case in Cornelian Bay where *Enteromorpha* spp. dominated, a species that is commonly associated with anthropogenic increases in nutrients.

However, given the range of factors that influence the growth of macroalgae, there is no absolute level of nutrients that will inevitably lead to increased growth of algae and result in decreased seagrass biomass. This issue is complex, and is largely influenced by the natural levels and variations in light and nutrients, the rate of flushing of nutrients from the system and the specific light requirements of seagrass species. Therefore, there is generally a strong seasonal trend consistent with that seen in nutrients with peaks in spring and summer evident.

It is difficult to definitively assess the impacts of increased nutrients and turbidity on seagrass in Cornelian Bay due to the frequent natural variations in this habitat, the lack of historical monitoring data, the lack of suitable controls for comparison; and a poor understanding of the fate and fluxes of nutrients in the bay.

The varying extent of natural variation in seagrass distribution and abundance means that it is extremely difficult to separate the indirect human induced impacts (ie. high nutrients and turbidity) from these variations. Detailed historical information from well-designed and consistent monitoring programs provides the best means of determining the extent and cause of these impacts on seagrass beds. Seasonal monitoring of seagrass beds in Cornelian Bay has recently commenced and will, over time, assist in the process of assessing 'health' of the beds and likely threatening processes. Monitoring of the maximum depth distribution of the beds may also assist in assessing overall water quality in the bay as seagrass depth range is often related to water quality (Abal and Dennison, 1996).

Despite the lack of historical information on the abundance and distribution of seagrass beds in the middle reaches of the Derwent Estuary, as siltation and nutrient inputs are highest in this region of the estuary (Coughanowr, 1997), it is highly likely that a decrease in seagrass extent and health has occurred. This would be consistent with a significant reduction in seagrass habitats throughout Tasmania over the past 50 or so years (Rees, 1993).

4.1.2 Subtidal unvegetated habitats

Subtidal unvegetated habitats are the most dominant habitat type within Cornelian Bay, representing around 80% of all subtidal habitats in the survey area. As a result of catchment factors (eg. soil type, erosion characteristics) and water movement (tidal, flood and wind derived) there is a general increase in the amount of fine sand and silt with increasing depth.

Cornelian Bay is dominated by winds from the west, with north westerly and south westerly winds characteristic of pre-frontal and post-frontal conditions, respectively. This results in a predominant offshore movement of finer sediment with the inner margin remaining considerably sandier. Further details on changes to this type of habitat in Cornelian Bay are detailed in section 4.3.

4.2 Heavy Metal Distribution

Heavy metals and arsenic concentrations in surface sediments were low in the nearshore sandy sediments, with cadmium, copper and zinc values well below the sediment quality guideline low values. In contrast, lead levels in the nearshore samples exceeded the sediment quality guideline low value ie. the concentration at which an adverse effect on biota is expected 10% of the time. Mercury concentrations exceeded the high ISQG levels at one of the intertidal sites and the low ISQG level at the other shallow site. These concentrations in the shallow sites are of concern given the high level of recreational use in these areas.

In contrast to the nearshore surface sediments, the siltier sediments in deeper water, particularly those in the centre of the bay, showed high levels of heavy metal contamination. Cadmium, lead and zinc levels all exceeded the ISQG high value ie. the concentration at which an adverse effect on biota is expected 50% of the time. However, these lead concentrations represent around 15% of peak values in the Derwent Estuary, where 77% of the estuary has lead levels higher than the recommended low value (Coughanowr, 2001). Arsenic and copper concentrations typically lie between the high and low ISQG values.

The long cores revealed that, in addition to the surface sediments, the top 20-35 cm of sediment in Cornelian Bay is contaminated with heavy metals and arsenic. Although the sediment cores were not analysed for mercury, these levels are elevated in surface sediments within the bay indicating that elevated mercury levels are also likely to occur at depth. The highest heavy metals concentrations occured at ~10 cm depth in both cores. The core with the highest contaminant levels (LC1), the peak concentrations of arsenic, cadmium, copper, lead and zinc were 331, 104, 522, 2,080 and 18,000 mg/kg, respectively. The ISQG high value for arsenic is 70 mg/kg, 10 mg/kg for cadmium, 270 mg/kg for copper, 220 mg/kg for lead and 410 mg/kg for zinc. It is clear from these results that sub-surface sediments in the centre of Cornelian Bay have excessive heavy metals concentrations, with zinc levels being particularly high at more than 40 times higher than the ISQG high level.

The peak heavy metals concentrations in core LC1 are comparable to peak concentrations in Geilston Bay, on the eastern shore (C. Samson, unpublished data). Although concentrations in Cornelian and Geilston Bays are very high, they are significantly lower than peak concentrations in New Town Bay (C. Samson, unpublished data), adjacent to the Pasminco site.

Given the high level of heavy metal contamination in sub-surface sediments in the central Cornelian Bay area, and evidence for declining heavy metals concentrations at the surface, it is recommended that management prevents any human activities which would result in future disturbance of the sediments in the central bay area. Disturbance could potentially result in resuspension and redeposition of the most highly contaminated sediments in the bay. As the highest concentrations occur at ~10 cm below the surface even small disturbances could easily re-expose the most contaminated sediments.

4.3 Sedimentation Rates

It is possible to estimate sedimentation rates over the last 85 years in the Derwent Estuary using the heavy metal profile concentrations. Heavy metals were first introduced to the Derwent Estuary in large quantities in 1917 with the commissioning of the Electrolytic Zinc works. This sudden increase in metals into the estuary is recorded in the sediments and the shift from natural to anthropogenic levels of heavy metals provides a stratigraphic marker for the year 1917. By determining the depth in the sediments at which the shift from natural to anthropogenic levels of heavy metals occurs it is possible to establish how much sediment has accumulated at that location since 1917, and hence derive an estimate of sedimentation rate. It is important to note that this technique only provides an average sedimentation rate over the last 85 years and hence provides no information about the pattern of sedimentation rates over this time period.

The shift from background to anthropogenic heavy metals concentrations occurred between 30 and 35 cm below the surface in LC1 and between 15 and 20 cm in the shallower LC2. The depth at which the year 1917 occurs cannot be pin-pointed any more accurately using available data due to the 5 cm sampling resolution. These results imply an average sedimentation rate of ~3.7 mm/yr and ~2mm/yr over the last 85 years in cores LC1 and LC2, respectively.

As the method employed for determining sedimentation rates is reliant on identifying changes in heavy metals concentrations it is necessary to assess whether the rapid increase in heavy metals concentrations are an artefact of changes in sediment particle size. Heavy metals concentrations in sediments are dependant upon sediment particle size, with higher concentrations occurring in fine muddy sediments. To take account of changes in sediment particle size the long core metals data was standardised for these changes.

In core LC2, the silt content (<0.063mm) changes very little throughout the core and remains below 5% at all times. Although standardisation of heavy metals concentrations results in relatively large changes in the absolute concentrations of heavy metals, due to the low silt content, it results in little change to the heavy metal profile. The sudden increase in heavy metals concentrations between 15 cm and 20 cm is still present. Standardisation of the heavy metal profile clearly demonstrates that the observed increase in heavy metals concentrations are not an artefact of changes in sediment particle size.

As mentioned earlier, heavy metal concentrations in sediments are also reliant upon sedimentation rates. If rates increase, the heavy metal concentration in the sediments will decrease even if inputs of heavy metals remain constant. An increase in sedimentation rates results in dilution of heavy metals. Although there is no way of determining sedimentation rates prior to 1917 from available data it is unlikely that the sudden increases in heavy metals concentrations are due to changes in sedimentation rates. If it were the case we would expect the profiles of *all* heavy metals (ie. zinc, lead, arsenic and copper) to show identical profiles, although offset from one another by a constant factor. It is clear that the profiles differ for each of the metals indicating that the sudden increase in heavy metals concentrations between are *not* a result of changes in sedimentation rates.

In addition to a likely increase in sedimentation rates in the past 100 or so years, there has been some change in the type of sediment being deposited over much longer time frames. There is evidence of this in the rapid decrease in the proportion of find sand below 60 cm in LC2 and the increase in the silt content below 80 cm in LC1. While there was no increase in silt content in the shallower LC2, this most likely reflects the high level of turbulence and silt transport at this depth. An increase in the silt content above 20 cm in LC1 indicates that the deeper parts of the bay have become siltier in the past 60-80 years. This is consistent with the significant changes that have also occurred to the sediment structure broadly throughout the estuary, with large inputs of fine sediments resulting in many sand habitats converting to sand/silt or silt habitats. (Coughanowr, 1997).

4.4 Effects of core compaction on sedimentation rate estimates

Almost all core collection techniques result in compaction during collection. Compaction in LC1 and LC2 was estimated to be small. The effect of compaction is that the derived sedimentation rate estimates are lower than actual sedimentation rates. Compaction can occur in two ways either (i) linear compaction where the core is compressed equally throughout its entire length or (ii) differential compaction where most of the compaction occurs in the less consolidated top section of the core and decreases with depth.

Core sampling most probably results in differential compaction because the deeper sediments are already partially compacted due to natural diagenetic processes. However, as it is very difficult to determine the exact pattern of compaction and the depth to which compaction has occurred, it is difficult to reliably correct for compaction effects. We can say however, that the maximum possible average sedimentation rate since 1917 in core LC1 is likely to be greater than ~4 mm/yr, which assumes that all the compaction occurred in the top 35 cm of the core. The estimate of ~4 mm/yr, which does not include a correction for compaction, provides the minimum average sedimentation rate estimate.

4.5 Comparison of sedimentation rates with other Derwent Estuary locations

Sedimentation rates in Cornelian Bay are between 2 mm/yr and 4 mm/yr, which fall within the typical range of 2-6 mm/yr of other sedimentation rate estimates obtained for subtidal embayments within the Derwent Estuary ie. Geilston Bay, Kangaroo Bay, Tranmere and Ralphs Bay (see Fig. 1) (C. Samson, unpublished data). However, the sedimentation rate and silt content in Cornelian Bay over the last 85 years is much less than in neighbouring New Town Bay, where the sedimentation rate has been estimated at ~12-16 mm/yr (C. Samson, unpublished data). The higher sedimentation rate and silt content in New Town Bay can probably be at least partially attributed to activities in the New Town Rivulet catchment, as during heavy rain events the New Town Rivulet has high turbidity levels (ie. a high sediment load) compared with other natural drainage systems entering the Derwent (pers. comm. Christine Coughanowr, 2002). The lower rates in Cornelian Bay also reflect the considerably smaller catchment area of the bay and that a small amount of the catchment has urban and industrial usage.

The sedimentation rate estimates in Cornelian Bay are also much lower than the ^{137}Cs derived estimates of 27-33 mm/yr for Lindisfarne Bay intertidal zone (Wood, 1988; Wood *et al*. 1992). The high sedimentation rates in Lindisfarne Bay intertidal zone were attributed to poor land-use practices within both the Derwent and particularly Lindisfarne catchments and changes in estuarine circulation patterns as a result of man-made structures (Wood, 1988).

Cores from many embayments in the Derwent Estuary show an increase in silt content closer to the surface (C. Samson, unpublished data). This is consistent with LC1 which showed an increase in silt content, at depths between 100 and 60 cm and 20 cm and the surface. Given the data indicates that the bottom of the core represents periods prior to 1917, it is evident that increases in silt content occurred prior to this period. In the shallower LC2 the mud content remains <5% throughout the core indicating that at this water depth there is sufficient current velocities to transport most fine sediment delivered to the bay into deeper water.

In LC2 there was a substantial change in the amount of shell material at around 40 cm below the surface, with shells abundant in the lower section of the core and very little above. The dominant species found in the base of the core include the bivalves *Fulvia tenuicostata*, *Dosinia sp*., *Tellina sp*., *Pecten fumata* (commercial scallop) and *Ostrea angasi* (native oyster). This change in the mollusc abundance occured below the depth at which heavy metal concentrations increase suddenly indicating that the loss of this fauna cannot be solely or directly attributed to metal toxicity.

An apparent decline in mollusc abundance is seen in several cores from the Derwent Estuary (C. Samson, unpublished data). Likely explanations include siltation and heavy metal pollution. A recent study by Edgar *et al*. (1999) sampled macroinvertebrates and sediments at three sites in the middle Derwent Estuary. While the sampling was conducted primarily to obtain baseline data, the change in the sediment structure from sand to mud dominated in many Tasmanian estuaries has been shown to result in a pronounced shift in the macroinvertebrate faunal composition (Edgar *et al*., 1999). Given the changes in sediment structure in the Derwent Estuary, such changes could be expected to have occurred widely throughout the estuary.

However, Pb^{210} and radiocarbon dating techniques and faunal analyses, which are currently being undertaken on several Derwent Estuary long cores, are required to determine possible causes for the apparent decline. In addition, further work is required to complete a faunal list of species in LC2, but was outside the scope of this study.

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