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Changes in the benthic macroinvertebrate fauna of a large microtidal estuary following extreme modifications aimed at reducing eutrophication

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ABSTRACT

An artificial channel was opened in 1994 between the microtidal Peel–Harvey Estuary and the Indian Ocean to increase tidal exchange and thus ameliorate the problems of eutrophication. Although this greatly reduced macroalgal and cyanobacterial growths and the amount of particulate organic matter, our data indicate that, contrary to managerial expectations, the benthic environment has deteriorated. Thus, although macroinvertebrate density has declined as predicted, taxonomic distinctness (Δ^*) has also declined and species composition has become more variable. Macroinvertebrate composition has also changed markedly at the species, family and even phylum levels. The Crustacea, the most sensitive of the major macrobenthic taxa to environmental stress, has become proportionally less abundant and speciose, whereas the Polychaeta, the least sensitive, was unique in showing the reverse trend. The benthos of the Peel–Harvey Estuary is thus apparently more stressed than previously, probably due to the multiple effects of a great increase in system use.

1. Introduction

Blooms of opportunistic green macroalgae are now widespread in coastal marine habitats (Fletcher, 1996). Much of the organic matter fixed by these blooms enters the local decomposer cycle, leading to hypoxic and even anoxic physico-chemical conditions in the water column and underlying sediment, which is hostile to the associated benthic invertebrate fauna (Raffaelli et al., 1998). In estuaries, these symptoms of eutrophication are due to a variety of anthropogenic and natural causes, and nearly all of these systems outside polar regions are suffering some degree of degradation (Edgar and Barrett, 2000). Responses of the benthos to increased organic inputs are generally considered to follow the Pearson and Rosenberg (1978) model (Gray et al., 2002), in which the benthic assemblage undergoes a series of well-documented successional changes. Recovery of degraded benthic assemblages in response to remedial action is less well-documented (Macloed et al., 2004; Sanz-Lázaro and Marin, 2006), but is thought to retrace the successional changes of the Pearson–Rosenberg model (Rosenberg et al., 2002). In temperate waters, however, this reverse succession takes from at least two years to more than 11 years, and thus apparently progresses more slowly than the succession that occurs in response to increased organic inputs (Karakassis et al., 1999; Lardicci et al., 2001; Munari et al., 2003; Macloed et al., 2004; Brooks et al., 2004).

Benthic macroinvertebrates play a crucial role in the functioning of estuarine ecosystems. For example, they are a major component of estuarine food webs and play an important role in nutrient cycling through their burrowing and feeding activities, i.e. bioturbation (Constable, 1999). Although the diversity of benthic macroinvertebrates is typically less in moderately eutrophic than more pristine estuaries, the reverse is often true of their densities (e.g. Gray et al., 2002; Karlson et al., 2002). Indeed, when an estuary becomes excessively eutrophic and thus experiences regular hypoxic events, these assemblages can become dominated by opportunistic species such as capitellid and spionid polychaetes that are able to thrive under such conditions (Pearson and Rosenberg, 1978; Wilson et al., 1998; Kemp et al., 2005). In general, the Polychaeta is the least sensitive of the major macrobenthic taxa to anthropogenic perturbations and, under stressed conditions, tend to increase in abundance and number of species relative to the more sensitive taxa such as the Crustacea and Mollusca (Reise, 1982; Warwick and Clarke, 1993b). The characteristics of the benthic macroinvertebrate fauna of an estuary are thus good indicators of the quality of the environmental conditions in that system.

The Peel–Harvey Estuary is the largest estuary in temperate Western Australia. It contains two large basins, the Peel Inlet and Harvey Estuary, which cover an area of 128 km^2 and are fed by three main tributary rivers (Fig. 1, McComb and Lukatelich,

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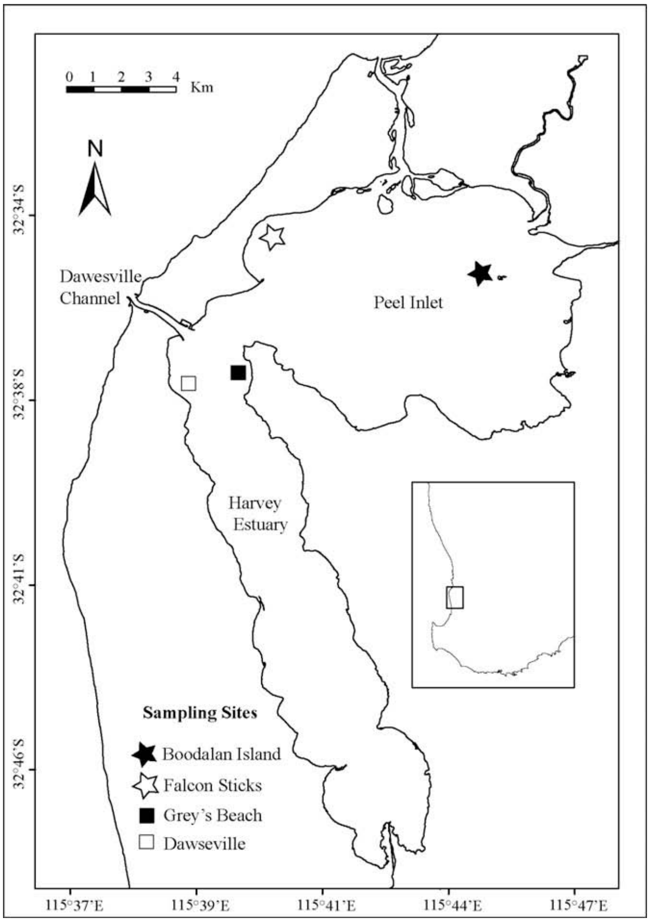

Fig. 1. Map showing the locations of the Dawesville Channel and the four sites sampled in the Peel–Harvey Estuary in each season between the winters of 1986 and 1987 and of 2003 and 2004. Inset shows the location of the Peel–Harvey Estuary on the lower west coast of Australia.

1995). Prior to 1994, the natural and narrow entrance channel that is located towards the northern end of Peel Inlet provided the only opening of this estuary to the sea. During the 1960s, the Peel–Harvey Estuary started becoming highly eutrophic through the input of nutrients from the surrounding agricultural land and other sources such as piggeries (McComb and Lukatelich, 1995). This eutrophication was reflected in the development of massive growths of green macroalgae, such as *Cladophora*, *Chaetomorpha*, *Enteromorpha* and *Ulva* species, particularly in Peel Inlet, and of very large seasonal blooms of the cyanobacterium *Nodularia spumigena*, mainly in the Harvey Estuary. There is strong circumstantial evidence from catch per unit effort data that the increases that occurred in macroalgal biomass between the1950s and 1980s were accompanied by an increase in the biomass of commercial fish species (Steckis et al., 1995), whereas the seasonal blooms of *N. spumigena* led to fish moving out of the affected areas, to mortality among less mobile fish species and to changes in the activities of commercial fishers (Potter et al., 1983; Lenanton et al., 1985).

The macrophyte and cyanobacterial growths in the Peel–Harvey Estuary became so extreme in the 1980s that it was decided to take radical action to ameliorate this problem. After much discussion, review and research, the decision was made to construct an artificial channel to the sea at Dawesville at the northern end of the Harvey Estuary, and thereby increase tidal exchange between the estuary and the ocean (Fig. 1). The construction of this channel helped flush nutrients out of the estuary and, as a consequence, the extent of macroalgal growths in this system declined (Water and Rivers Commission, 1998). It also increased salinity to a level that is no longer conducive to the growth of *N. spumigena* during the spring (Water and Rivers Commission, 1998; Young and Potter, 2003). Furthermore, a comparison of the fish faunas in the Peel–Harvey Estuary between the 1980s and late 1990s demonstrated that the reduction in primary productivity that occurred as a result of the flushing effects of the Dawesville Channel was accompanied by a decline in the overall abundance of fishes and a change in their species composition (Young and Potter, 2002, 2003).

Benthic macroinvertebrates were sampled in 2003/4 at the same four sites in the Peel–Harvey Estuary as those sampled for such invertebrates by Rose (1994) in 1986/7. The data for these two periods, which were both acquired over five consecutive seasons, enabled the densities, numbers of species and species compositions of the benthic macroinvertebrate fauna in pre- and post-Dawesville Channel periods to be compared. These comparisons were used to test the hypothesis that the marked decline in primary productivity in the Peel–Harvey Estuary in 2003/4 vs. 1986/7 has been accompanied by a reduction in the densities of benthic macroinvertebrates and in beneficial effects on their community structure and composition.

2. Materials and methods

2.1. Sampling regime

The benthic macroinvertebrates at Boodalan Island and Falcon Sticks in the Peel Inlet and at Dawesville and Grey's Beach in the Harvey Estuary (Fig. 1) were sampled in the middle of each season by Rose (1994) between the winters of 1986 and 1987 and by ourselves between the winters of 2003 and 2004. The same corer and sampling regime were used on all sampling occasions in both periods.

The sampling of each site was conducted in a rectangular area that measured 150 m along the shoreline and 10 m perpendicularly outwards from the shoreline and in water depths ranging from 0.5 to 1 m. In the middle of each season, three sediment cores were collected randomly from each site during the day on one occasion and another two on a second occasion at least two weeks later.

The cylindrical corer, which was 11 cm in diameter and covered a surface area of 96 cm^2, sampled to a depth of 15 cm. The sediment samples were preserved in 5% formalin buffered in estuary water and subsequently wet sieved through a 500 μm mesh. Using a dissecting microscope, the invertebrates were removed from any sediment retained on the mesh, identified to the lowest possible taxonomic level and stored in 70% ethanol. The number of each macroinvertebrate taxon in each sample was converted to a density, i.e. number of individuals 0.1 m^{-2}. Note that the vast majority of taxa were able to be identified to species.

Temperature, salinity and dissolved oxygen concentration were measured at the bottom of the water column in three locations at each site on each sampling occasion using a Yellow Springs International 556 Multi-parameter Handheld Meter. The depth of the transitional zone (where the colour of the sediment changes from light to dark and corresponds to a region just below the depth of biologically available oxygen, Hourston et al., 2009) was recorded to the nearest 0.5 cm in three small sediment cores collected randomly from each site on each sampling occasion. The cylindrical corer used for this purpose covered an area of 10 cm^2 and sampled to a depth of 10 cm. The sediments in these cores were also used to determine the percentage contributions of particulate organic matter (%POM) to the sediment. For this purpose, the sediment samples were dried at 80 °C for 24 h, weighed to the nearest 1 mg, ashed at 550 °C for 2 h and reweighed. The percentage contribution of POM was determined from the weight lost during ashing.