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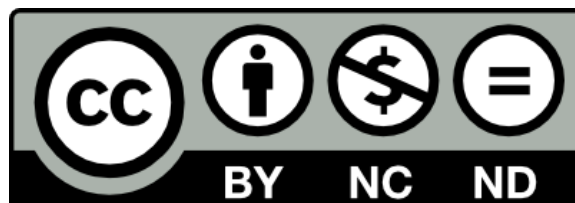
Benthic macroinvertebrates as indicators of environmental deterioration in a large microtidal estuary

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Abstract

This study tested the hypothesis that, during recent years, the benthic macroinvertebrate fauna of the large basin of the Swan-Canning Estuary has changed in ways consistent with deteriorating environmental conditions in that estuary. Between 1986/7 and 2003/4, the compositions of that fauna altered markedly at the species and even family levels. Thus, the densities and number of species of molluscs, and especially of crustaceans, which are particularly susceptible to environmental stress, declined, while those of the more tolerant polychaetes increased. However, taxonomic distinctness declined consistently only at one of the four widely-spaced sampling sites and the dispersion of samples did not differ markedly between periods, indicating that the benthic fauna has not undergone such extreme changes as in the nearby Peel-Harvey Estuary. It is thus proposed that benthic macroinvertebrates can act as important indicators of the severity of environmental degradation in microtidal estuaries in regions where such perturbations are increasing.

1. Introduction

Benthic macroinvertebrates play a crucial role in the functioning of estuarine ecosystems. They are, for example, a major component of estuarine food webs and, through their burrowing and feeding activities, i.e. bioturbation, play an important role in nutrient cycling in those systems (Hutchings, 1998; Constable, 1999). Although the diversity of the benthic macroinvertebrate fauna is typically less in eutrophic than more pristine estuaries, the reverse is often true of their densities (e.g. Gray et al., 2002; Karlson et al., 2002). Furthermore, when an estuary becomes excessively eutrophic, and thus experiences regular hypoxic events, its benthic macroinvertebrate fauna can become dominated by opportunistic species, such as capitellid and spionid polychaetes, which are able to thrive under such conditions (Pearson and Rosenberg, 1978; Wilson et al., 1998; Kemp et al., 2005). While such dominance is a characteristic of grossly perturbed environments, the average taxonomic spread of species at a location, defined in such a way that its mean value is independent of the number of species (e.g. taxonomic distinctness), can decline even under less stressful conditions (Warwick and Clarke, 1998, 2001).

In general, the Polychaeta is the least sensitive of the major macrobenthic taxa to anthropogenic perturbations and, under stressed conditions, its abundance and number of species tend to increase in relation to those of more sensitive taxa, such as the Crustacea and Mollusca (Reise, 1982; Warwick and Clarke, 1993a; Wildsmith et al., 2009). The trends exhibited by the structural properties and taxonomic composition of the benthic macroinvertebrate assemblages of estuaries thus act as good indicators of the quality of the environmental conditions in those systems (e.g. Roach and Wilson, 2009; Pelletier et al., 2010; Warwick and Somerfield, 2010; Whomersly et al., 2010). The value of this group as indicators is demonstrated, for example, by the conspicuous changes that occurred in their diversity and species composition in the Tees Estuary following changes to the local and regional environment (Warwick et al., 2002).

The Swan-Canning and Peel-Harvey estuaries are the two largest estuaries on the lower west coast of Australia, occupying areas of 50 and 131 km^2, respectively (Hodgkin and Hesp, 1998). Comparisons between data for the basins of the Peel-Harvey Estuary in 2003/4 with those recorded in 1986/7

revealed that, in the intervening period, the density and biodiversity of the benthic macroinvertebrate fauna, and particularly the density of crustaceans and molluscs, had declined (Wildsmith et al., 2009). In contrast, the density of the Polychaeta had increased and was the only major taxon to have undergone such a change. These comparisons strongly suggest that the quality of the benthic environment had deteriorated between these two periods. Although the reasons for this deterioration are not clear, it is almost certainly relevant that the extent of local urbanization, the number of canal systems and amount of recreational boating and fishing had all increased markedly in the intervening period. The cumulative effects of these changes would inevitably have had a deleterious impact on the environment of this microtidal system.

The environment of the Swan-Canning Estuary has undergone progressive deleterious changes over the last two to three decades as a result of anthropogenic activities, many of which are related to the runoff of nutrients and contaminants from surrounding agricultural land, industry and the rapidly expanding city of Perth (Hamilton and Turner, 2001; Viney and Sivapalan, 2001; Zammit et al., 2005). This led to the Swan-Canning system being designated as a "hot spot" by the Australian Government and its provision of funding to the Swan River Trust to develop the Swan-Canning water quality improvement plan (Swan River Trust, 2009).

It has been estimated that the nutrient loads in the estuary have increased $\sim$20 times from pre-European levels (Chan et al., 2002) and that the concentrations of anthropogenic phosphorus in the sediment have increased three times since 1940 (Gerriste et al., 1998). The most visible signs of eutrophication include increases in the severity and frequency of nuisance algal and dinoflagellate blooms, which were rarely observed in the 1980s (Swan River Trust, 2005; Zammit et al., 2005). This has led, in turn, to an increase in the prevalence of fish deaths. Although eutrophication is greatest in the upper estuary, a combination of high concentrations of inorganic nutrients (dissolved inorganic nitrogen >1.2 mg L^{-1}, filterable reactive phosphorous >0.02 mg L^{-1}) in river inflows and high freshwater discharge and temperatures in 2000, resulted in a major bloom of *Microcystis aeurginosa* extending from that part of the estuary into the middle and lower estuary (Robson and Hamilton, 2003).

It has been estimated from cores that the concentrations of copper, lead, cadmium and zinc in the sediment of the Swan-Canning Estuary have risen significantly since colonial settlement, largely through input from sources near the estuary (Gerriste et al., 1998). Indeed, in 2007, the concentrations of zinc and lead in the sediment at several sites, and particularly those near the city of Perth (Nice, 2009), greatly exceeded the interim sediment quality guidelines trigger levels of 200 and 50 mg kg^{-1}, respectively (ANZECC and ARMCANZ, 2000). Furthermore, the concentrations of tributyltin near yacht clubs greatly exceeded the trigger level of 0.07 mg kg^{-1} at most sites and were as high as 5 mg kg^{-1} at some sites (Swan River Trust, 2007).

The increase in population and residential dwellings of the Perth region by 34% and 49%, respectively, between the mid-1980s and 2000s will inevitably have led to other pressures on the Swan-Canning Estuary. Moreover, the effects of the predicted increase in recreational boat ownership by 82% within the next 15 years are bound to have an appreciable effect on this system (Australian Bureau of Statistics, 1987, 2001; Department of Planning and Infrastructure, 2009).

The present study has determined the density, species richness and taxonomic distinctness, and the composition at the species, family and phylum levels, of the benthic macroinvertebrate faunas at sites in the basin region of the Swan-Canning Estuary in five successive seasons in 2003/4. The resultant data were compared with those determined from data recorded by Rose (1994) for the same sites in five sequential seasons in 1986/7. These comparisons enabled us to ascertain whether, as in the Peel-Harvey Estuary (Wildsmith et al., 2009), the benthic macroinvertebrate fauna in this system had undergone the changes in such features as biodiversity and taxonomic composition that would be consistent with a pronounced deterioration in the benthic environment. The implications of the results are relevant, in general, to microtidal estuaries that are becoming increasingly eutrophic, contaminated and physically disturbed.

2. Materials and methods

2.1. Sampling regime

Benthic macroinvertebrates were sampled at four sites (Dalkeith, Applecross, Matilda Bay and Deepwater Bay) in the Swan-Canning Estuary (Fig. 1) during the middle of each of the five seasons between the winters of 2003 and 2004. These were the same sites as those sampled seasonally by Rose (1994) using the same corer and methodology between the winters of 1986 and 1987 and which had been chosen to represent different habitats in the large basin area of the estuary.

Five randomly-located sediment cores were collected from each site during the day in each season. Each site comprised a rectangular area ~150 m along and 10 m perpendicular to the shoreline and within which the water depths ranged from 0.5 to 1 m. The sampling of each site was staggered over 2–3 weeks. 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 sea water and subsequently wet sieved through a 500 μm mesh. Using a dissecting microscope, the invertebrates were removed from any sediment that was retained on the mesh, identified to the lowest possible taxon and stored in 70% ethanol. The number of each macroinvertebrate taxon in each replicate sample was converted to a density, i.e. number of individuals m^{-2}, which was then used to calculate the values for overall density and quantitative taxonomic distinctness ($\Delta*$, Warwick and Clarke, 2001) in each sample obtained from each site in each season.

Water temperature, salinity and dissolved oxygen (DO) concentration at the bottom of the water column were recorded at three locations at each site on each sampling occasion using a Yellow Springs International Multi-parameter Hand Held Meter Model 556.

Three randomly-located sediment cores were collected from each site on each sampling occasion using another cylindrical corer, which covered an area of 10 cm^2 and sampled to a depth of 10 cm. These cores were used to measure, to the nearest 0.5 cm, the depth of the transition layer in each core, i.e. the depth at which the colour of the sediment changed from light to dark, which represents the point at which the ferric iron is reduced to ferrous iron (Sikora and Sikora, 1982) and lies just above

the point at which oxygen becomes biologically unavailable. The same cores were used for determining the % contributions of particulate organic matter (%POM) to the sediment. For the latter, 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, thus enabling the %POM to be determined from the weight lost during ashing.

2.2. Statistical analyses

Analyses of variance (ANOVA) tests were used to determine whether the environmental variables (water temperature, salinity, dissolved oxygen concentration, depth of the transition layer and %POM) and biotic variables (number of species, density and taxonomic distinctness of benthic macroinvertebrates) differed significantly among periods (1986/7 and 2003/4), seasons (each of the five seasons between the two successive winters in each period) and sites (the four sampling sites). Prior to subjecting the data for each variable to ANOVA, the relationships between their means and standard deviations were plotted, to assess whether those data required transformation to satisfy assumptions of normality and constant variance and, if so, which transformation was most appropriate. As only the mean values for water temperature, salinity and dissolved oxygen concentration (DO) were recorded by Rose (1994) at each site in each season in 1986/7, i.e. there were no replicates, those means and standard deviations were calculated across sites in each season. The procedure showed that salinity and %POM each required a square root transformation and the density of benthic macroinvertebrates a fourth root transformation, while dissolved oxygen, water temperature, depth of the transition layer, number of species and $\Delta*$ did not require transformation. N.B. Eleven samples returned a taxonomic distinctness value of zero (i.e. contained 2 or less species) and were thus removed prior to subjecting this variable to ANOVA.

For the three environmental variables for which there were no replicates for sites in each season (water temperature, salinity and dissolved oxygen), those means were subjected to three-way ANOVA to ascertain whether they were related to period, season and site, using the three-factor season $\times$ period $\times$ site interaction as the residual. This procedure still permits tests for the significance of the main effects and two-way interactions between each pair of the three factors. However, replicate values for %POM and the depth of the transition layer, as well as for the density, number of

species and taxonomic distinctness of the benthic macroinvertebrate fauna at each site in each season were determined in 1986/7 as well as in 2003/4. Thus, a full three-way ANOVA could be used to explore how the above indices varied between sites as well as periods and seasons in terms of the main effects and the two-factor and three-factor interactions.

The following multivariate analyses were carried out using the PRIMER v6 statistical package (Clarke and Gorley, 2006) to explore whether the compositions of benthic macroinvertebrates differed significantly among periods, seasons and sites. The Bray–Curtis similarity coefficient was employed to construct a similarity matrix from the log $(n + 1)$ transformed densities of the various macroinvertebrate species recorded for each replicate sample at each site in each season during each period. This matrix was then subjected to non-metric multidimensional scaling (MDS) ordination. When appropriate, the extent of dispersion of the samples in the two periods were compared using the index of multivariate dispersion (MVDISP, Clarke and Gorley, 2006).

Initially, the relative magnitudes of the influence of the period, site and season effects were determined using the following series of two-way crossed analysis of similarities (ANOSIM, Clarke, 1993) tests for (i) period vs. the combined effects of site and season, i.e. including their interaction, (ii) site vs. the combined effects of period and season and (iii) season vs. the combined effects of period and site. This was achieved by 'flattening' the two removed factors in each case to a single factor representing all combinations of their levels. The R-statistic values for the main factors of interest in these tests (period, site and season, respectively) provide a robust measure of the relative magnitudes of their influence on the species composition. However, as these two-way ANOSIM tests of averaged effects could still hide interactions between the main factor of interest and the removed factors, the influence of interactions was examined using a permutational multivariate analysis of variance (PERMANOVA) test (Anderson, 2001; McArdle and Anderson, 2001), which foregoes some of the robustness of the non-parametric approach of ANOSIM for the more penetrative general linear modeling of PERMANOVA.

Further two-way crossed ANOSIM tests were carried out within the levels of other factors, e.g. tests for differences between the compositions of the macroinvertebrate assemblages between 1986/7 and

2003/4 separately for each season, using a two-way test to allow periods to be examined orthogonally to site differences. Significant ($P < 5\%$) ANOSIM R-statistics for such tests were then contrasted between seasons. When the ANOSIM test detected a significant difference among periods, similarity percentages (SIMPER, Clarke, 1993) was used to identify which species made the greatest contributions to those differences. The above multivariate procedures were repeated using species data averaged for each site in each season in each period and then after aggregation to the family and phylum levels.

3. Results

3.1. Environmental measurements

ANOVA showed that water temperature and salinity both differed significantly between periods and seasons, with the period $\times$ season interaction being significant for both of these environmental variables. Dissolved oxygen concentration differed significantly between periods and sites, with the period $\times$ season interaction also being significant (Table 1). In the case of both water temperature and salinity, the mean squares were far greater for season, as expected, than for either period or the interaction. However, for dissolved oxygen, the mean square was greater for period than for either site or the period $\times$ season interaction.

The depth of the transition layer differed significantly between periods and seasons and there were significant two- and three-way interactions between each of the factors (Table 2). The mean square was far greater for period than season, which, in turn, was greater than those for all interactions. The depth of the transition layer changed far more throughout the sampling period in 2003/4 than in 1986/7, decreasing markedly between winter/spring and summer/autumn in the later period and then increasing markedly in the second winter (Fig. 3a and b).

The % contribution of POM to the sediments differed significantly between periods, seasons and sites, with the mean square being substantially greater for period than for the other two factors (Table 2). The mean %POM at each site was almost invariably less in each season in 2003/4 than in the

corresponding season in 1986/7 (Fig. 3c and d), which was reflected in a highly significant difference ($P < 0.001$) between the overall means for those periods, i.e. 1.34 vs. 0.88, respectively. Mean %POM did not show a consistent pattern of seasonal change at any one site in either period.

3.2. Overall benthic macroinvertebrate fauna

The density of 1213.7 individuals 0.1 m^{-2}, averaged over all five replicates at each of the four sites in the Swan-Canning Estuary in five successive seasons in 2003/4, was remarkably similar to the 1233.6 individuals 0.1 m^{-2} recorded by Rose (1994) at the same sites and seasons in 1986/7 (Table A1). The samples obtained during 2003/4 contained 42 species from six phyla, i.e. Annelida, Arthropoda, Mollusca, Sipuncula, Platyhelminthes and Nematoda, whereas those collected during 1986/7 yielded 52 species representing seven phyla, namely Annelida, Arthropoda, Mollusca, Nemertea, Cnidaria, Chordata, and Turbellaria. The Annelida, Arthropoda and Mollusca, which were the most speciose phyla during both sampling periods, were represented by 18, 10 and 11 species, respectively, during 2003/4, and by 10, 22 and 15 species, respectively, during 1986/7. The most abundant species in both periods were polychaetes, i.e. *Ceratonereis aequisetis*(19.4%) in 1986/7 and *Capitella* spp. (20.7%) in 2003/4. In 1986/7, the tanaid *Tanais dulongii* ranked fourth and contributed 11.0% to the total number of individuals and the bivalve *Arthritica semen*ranked fifth and contributed 7.7%, respectively, whereas, in 2003/4, they each ranked far lower and contributed only 0.2 and 1.6%, respectively (Table A1).

Overall, 29 of the 52 species found in 1986/7 were not recorded in 2003/4 and 19 of the 42 species found in 2003/4 were not recorded in 1986/7 (Table A1). The 29 species found only in 1986/7 contained 14 species of arthropod (mainly crustaceans) and seven species of mollusc, which together constituted ~40% of the total number of species recorded during 1986/7. While the majority of these species were represented by only one to three individuals, the bivalve *Xenostrobus securis* and the isopod *Syncassidina aestuaria* both contributed 1.3% to the total number of individuals in 1986/7. The remaining eight species recorded only in 1986/7 comprised two species of tubellarians, two of polychaetes and one each of the Cnidaria, Chordata and Nemertea. In contrast, ten of the 19 taxa recorded in 2003/4, but not in 1986/7, were annelids, i.e. nine polychaete species and oligochaete

spp., some of which were relatively abundant, e.g. *Pseudopolydora kempi*, Oligochaete spp. and *Carazziella victoriensis*, which, in terms of abundance, ranked sixth, tenth and thirteenth, respectively (Table A1). The remaining nine species, which comprised three species of mollusc, three of arthropods and one each of Sipuncula, Platyhelminthes and Nemertea, were relatively rare.

3.3. Number of species, densities and diversity of benthic macroinvertebrates

Three-way ANOVA showed that the number of species, density and taxonomic distinctness each differed significantly among periods, seasons and sites, and that the period × season and site × season interactions were significant for number of species, as were all two-way interactions for density and taxonomic distinctness and the three-way interaction for the last variable (Table 3). The mean squares were greatest for season and the period × season interactions for both the number of species and density. However, in the case of taxonomic distinctness, the mean square for period was far greater than those for the other main effects and interactions.

The mean seasonal number of benthic macroinvertebrate species varied more in 2003/4 than in 1986/7, declining markedly between spring and autumn and then rising sharply in winter (Fig. 4a). However, in both periods, it was lower at Applecross than at the other three sites (Fig. 4b). The mean seasonal densities followed similar trends in each period, except that they peaked in spring in 2003 and in summer in 1987 (Fig. 4c). The mean densities were lower at Dalkeith, Applecross and Matilda Bay in 2003/4 than in 1986/7 and were similar at Deepwater Bay (Fig. 4d).

Mean taxonomic distinctness showed no consistent seasonal trend across all four sampling sites in either period, but did decline to a minimum at Matilda Bay in summer in both periods (Fig. 5). Although mean taxonomic distinctness showed no conspicuous tendency to be consistently greater in one particular period at Dalkeith, Deepwater Bay and Matilda Bay, it was less in each season of 2003/4 than in the corresponding season in 1986/7 at Applecross (Fig. 5).

3.4. Composition of benthic macroinvertebrate assemblages

Two-way crossed ANOSIM, using period vs. a combined site-by-season factor and thus removing any confounding influence of site and/or season individually, gave an *R*-statistic of 0.78 for period

($P < 0.001$), while the R-statistic for site, after removing the combined effects of period and season, was 0.47 ($P < 0.001$), and that for season, after removing the combined effects of period and site, was 0.46 ($P < 0.001$).

PERMANOVA showed that the mean densities of benthic macroinvertebrate species in the basins of the Swan-Canning Estuary differed significantly according to period, season and site, with the two- and three-way interactions being significant ($P = 0.001$). The components of variation were similar for the period × season interaction (22.3) and period (21.5) and these were greater than for season and site or the other interactions (8.8–15.4). These interactions can be seen in the MDS plot in Fig. 6.

When the matrix, derived from the mean densities of each benthic macroinvertebrate taxon at each site in each season in each period was subjected to MDS ordination, the points representing the samples from 1986/7 all lay to the left of all of those from 2003/4 (Fig. 6), noting that one outlier is not shown for the latter period. The complete suite of samples were similarly dispersed in 2003/4 and 1986/7, as reflected informally in similar MVDISP values of 1.07 in 2003/4 and 0.94 in 1986/7. While the samples from spring and summer were located above those for autumn and both winters at the majority of sites in the earlier period, no such seasonal distinction was seen in the distribution of the samples in the latter period. The samples for successive seasons at some locations followed sequential and often cyclical progression on the plots (Fig. 6), presumably reflecting differences between the times of recruitment of species which have relatively short life cycles.

The overall composition of the benthic macroinvertebrate fauna during 2003/4 was distinguished from that in 1986/7 by greater densities of the polychaetes *Capitella* spp., *P. kempi, Leitoscoloplosnormalis* and *Australonereis ehlersii*, the crustacean *Corophium minor* and the bivalve *Sanguinolaria biradiata* and far lower densities of the crustaceans *Grandidierella propodentata*and *T. dulongii*, the polychaetes *C.aequisetis* and *Boccardiellalimnicola* and the bivalve *A.semen* (Table 4).

Two-way crossed period × site ANOSIM tests for each season showed that the species compositions of the benthic macroinvertebrate fauna differed significantly between periods and sites in each season, and that the differences between periods were greater than those between sites, with respective R-

statistic values of 0.66 and 0.45 for the first winter, 0.84 and 0.67 for spring, 0.92 and 0.46 for summer, 0.50 and 0.23 for autumn and 0.97 and 0.46 for the second winter.

When the matrices employed in each of the above ANOSIM tests were subjected to separate MDS ordinations, the samples for each period formed distinct groups in the plots in each season except autumn (Fig. 7). In autumn, most of the samples from 2003/4 overlapped and formed a very tight group in the middle of the plot, together with a few of those from 1986/7 (Fig. 7d). When a further MDS ordination of only those samples within that tight group was plotted, the samples for each period could be seen to constitute distinct groups (Fig. 7e).

Two-way crossed SIMPER showed that, among the species that distinguished between the benthic macroinvertebrate compositions in the two periods in each season, *A. semen*, *T. dulongii* and *B. limnicola* were in far greater densities in 1986/7 than in 2003/4 (Table 5). In contrast, *S. biradiata* and *P. kempi* were more abundant in 2003/4 than in the corresponding season in 1986/7, except for autumn. In autumn, the species composition in 1986/7 was distinguished from that in 2003/4 by greater densities of particularly *G. propodentata*, *C. aequisetis* and *C. capitata* in the former period (Table 5). One or more of these species were among those which distinguished between the assemblages in the other seasons but were not always more abundant in 1986/7.

3.5. Compositions of assemblages at species, family and phylum levels

On the ordination plot, derived from the matrix constructed from the mean density for each family at each site in each season in each period, the samples for 1986/7 and 2003/4 were almost as well separated as those in the plot derived using the densities of species (Fig. 8). This accounts for the R-statistic between periods being substantial for family (0.55, $P = 0.1\%$), but not as high as for species (0.71, $P = 0.1\%$). When the ordination plot was constructed using the density data aggregated to phylum level, there was a strong tendency for the majority of the samples from 2003/4 to lie above or to the right of those for 1986/7 (Fig. 8). There was, however, overlap between some of the points for the two periods, which is reflected in the R-statistic (0.18, $P = 0.1\%$) being far lower than that for either species or family.

SIMPER demonstrated that lower densities of crustacean species in 2003/4 than in 1986/7, and particularly of *T. dulongii,* were especially important in distinguishing between the benthic macroinvertebrate fauna at the species level in the two periods (Table 6). This was reflected at the family level by the densities of all seven crustacean families, and particularly that of the Tanaidae, being lower in 2003/4 and, at the phylum level, by the densities of the Arthropoda being lower in 2003/4. These were the most important features distinguishing between the faunas in the two periods (Table 6). In contrast, seven of the 11 polychaete species that distinguished the benthic macroinvertebrate fauna in 2003/4 from that in 1986/7, particularly *P. kempi*, *A. ehlersii* and *Capitella*spp., were more abundant in the recent period. This influence was not as clearly reflected at the family level, with only three of the six polychaete families that distinguished between the fauna in the two periods being greater in 2003/4. However, at the phylum level, the overall increase in polychaete densities in 2003/4 accounts for those of annelids being greater in that period. Indeed, the Annelida and the Sipuncula were the only distinguishing phyla to be in greater abundance in 2003/4. In contrast to the situation with the Polychaeta, the densities of six of the eleven mollusc species, and particularly of *A. semen*, *Musculista senhausia* and *Xenostrobus secures*, were lower in 2003/4. This accounts for the densities of the Mollusca being consistently lower in 2003/4 and thus helping to distinguish between the assemblages in the two periods.

4. Discussion

4.1. Changes in the environmental characteristics of the Swan-Canning Estuary

The lower percentage contribution of POM to the sediments in the basin of the Swan-Canning Estuary in 2003/4 than 1986/7 may reflect an increase in the amount of sedimentary non-organic material, rather than a decline in the actual amount of organic material in the more recent period. This view is based on the fact that, during recent years, the clearing of native catchment vegetation and unlimited access by livestock along river banks have increased the rates of erosion and sediment loss from catchments. The increased transport, and thus greater contribution of mineral rich soils to this estuary,

has led to an increase in the burial of organic material and thus a reduction in contact time with dissolved oxygen in the water column (Radke et al., 2004). Consequently, the rate at which organic material can be degraded by sulphate reduction would be increased. This proposal is consistent with the finding that the total amount of organic carbon in the sandy sediments of south-western Australian estuaries tends to decrease with increasing sediment loads from the catchments (Radke et al., 2004). Although the mean seasonal values for %POM in both periods were below the reference point of 2% adopted by Weisberg et al. (1997) for Chesapeake Bay, the consistently lower values for this variable in 2003/4 than in 1986/7 clearly demonstrate that the benthic environment of the Swan-Canning Estuary changed between these two periods.

Increases in sedimentation would also account for the transitional layer becoming deeper during the winter and spring in 2003/4 as this is the time when river discharge is greatest and new sediment is thus deposited (Hoeksema and Potter, 2006). A reduction in the depth to which oxygen penetrates in summer and autumn would provide a less favourable environment for benthic macroinvertebrates in those two seasons. It is thus relevant that the number of species and densities of benthic macroinvertebrates declined markedly between the spring of 2003 and autumn of 2004 and then rose in winter.

The mean seasonal values for dissolved oxygen concentration, which were measured during the day, were lower in the Swan-Canning Estuary in 2003/4 than in 1986/7 and, in spring, declined to near the reference condition of 5 mg L^{-1} for Chesapeake Bay (Weisberg et al., 1997). Furthermore, the minimum concentration in spring was well below the above reference point and substantial amounts of macrophytes accumulated at our sampling sites in 2003/4 (Wildsmith, 2007), which would lead to a reduction in oxygen concentration at night (Astill and Lavery, 2001). Another manifestation of declining conditions in the basin area of the Swan-Canning Estuary, where our sampling was conducted, was the development of a massive bloom of the toxic blue green algae *Microcystis aeruginosa* in 2000, which had spread downstream and led to large-scale benthic hypoxia (Robson and Hamilton, 2003; Swan River Trust, 2000). It must be recognised, however, that, in general, the middle estuary is far less eutrophic than the upper estuary and that an increased prevalence of

transient declines in oxygen concentration may thus play only a minor role in accounting for any changes to the fauna in this part of the estuary.

4.2. Changes in the benthic macroinvertebrate fauna between 1986/7 and 2003/4

The data collected during the present study demonstrated that the species compositions of the benthic macroinvertebrate assemblages at four sites distributed throughout the large basin area of the Swan-Canning Estuary in 2003/4 differed markedly from those at the same sites in 1986/7. The extent of this inter-period difference is reflected in the high R-statistic of 0.71 for analyses conducted at the species level and by a substantial R-statistic of 0.55, even when using data aggregated to the family level. Furthermore, the inter-period differences were particularly marked when the compositions in some corresponding seasons were compared, with the R-statistic values for three of the five seasonal comparisons ranging upwards from as high as 0.84–0.97.

Although no conclusion can be drawn as to whether differences in composition per se reflect deleterious changes to the benthic environment, it is relevant that the total number of species recorded in 2003/4 was less than in 1986/7, i.e. 41 vs. 52, and the same was true for phyla, i.e. 5 vs. 7. This suggestion that there may have been a decline in the quality of the environment is consistent with the pattern of change exhibited by key taxa. Thus, crustaceans and molluscs, which are particularly sensitive to anthropogenic disturbance, were in lower densities in the more recent period, whereas the reverse was the case with annelids, which comprised mainly polychaetes, a taxon that is are far more resilient to such environmental perturbations (Reise, 1982; Warwick and Clarke, 1993b; Gray et al., 2002). The direction of the changes in the densities of the above three taxa parallel those recorded between the same two time periods in the nearby Peel-Harvey Estuary, where they were accompanied by other changes to the benthic macroinvertebrate fauna that are indicative of environmental degradation (Wildsmith et al., 2009).

The reduction in the densities of crustaceans between 1986/7 and 2003/4 is attributable to all of the 21 benthic crustacean species recorded in the former period either being in lower densities or absent in the more recent period. Although these species represented a range of crustacean orders, 12 of those

species were amphipods. The susceptibility of amphipods to environmental degradation is emphasised by the finding that their densities underwent the greatest decline of all taxa as a result of the detrimental effects of a major oil spill off the north coast of France (Gómez Gesteira and Dauvin, 2000). The main species driving the reduction in crustaceans in the Swan-Canning Estuary between 1986/7 and 2003/4 were the amphipods *G. propodentata*and *C. minor* and the tanaid *T. dulongii.* The densities of those same three species also declined markedly between 1986/7 and 2003/4 in the Peel-Harvey Estuary, which accounted for them likewise being the most important of the crustacean species for distinguishing between the faunas in the two periods. Molluscs showed a less pronounced decline than the more sensitive crustacean species. This may partly reflect the fact that *S. biradiata*, the most abundant of the molluscs in the more recent period, possesses long inhalant siphons that enable it to sample the water well above the sediment surface, which is well oxygenated and less likely to be contaminated than the sediment (Beesley et al., 1998).

The greater densities of annelids in 2003/4 reflects, in part, the fact that seven of the ten most important distinguishing polychaetes species increased in density between the earlier and more recent periods, with *P. kempi*, *A.ehlersii*, *Capitella* spp. and *L. normalis* being the main drivers. While consistently greater densities of *L. normalis* also played a major role in distinguishing between the faunas in 1986/7 and 2003/4 in the Peel-Harvey Estuary, the other main polychaete species whose density increased between the two periods in that estuary differed from that in the Swan-Canning Estuary. The absence of the polychaete *P. kempi* in 1986/7 contrasts with its substantial contribution of 6.6% to the total number of individuals and sixth place ranking in terms of density in 2003/4. This suggests that this large spionid, which is not native to Australia (K. Higgins pers. comm. Invasive Marine Species Program, Australian Department of Agriculture, Fisheries and Forestry), has been introduced into the Swan-Canning Estuary in the last two decades, possibly via the ballast of ships entering the Port of Fremantle at the mouth of the Swan-Canning Estuary. This species became so abundant in the upper Swan-Canning Estuary between 1995 and 1997 that it contributed 16.5% to the total number of benthic invertebrates recorded in the shallow waters of that part of the system (Kanandjembo et al., 2001). As another large spionid, *B. limnicola,* contributed 4.7% to the number of

individuals in 1986/7 (ranked 7), and only 0.04% in 2003/4 (ranked 28), it may have become largely displaced by *P. kempi* in the intervening period. The densities of the gastropod *B. australis,* which was apparently introduced into the Swan-Canning Estuary in the 1950s (Thomsen et al., 2010), increased only slightly between the two periods in our study.

Although increases in the prevalence of transient declines in dissolved oxygen concentration may have contributed to the changes that took place in the benthic macroinvertebrate fauna between 1986/7 and 2003/4, it should be recognized that the level of eutrophication in the middle estuary, from which our samples were collected, is far less than in the upper estuary. It thus seems highly likely that other factors, such as those associated with industrialization and urbanization, played an important role in driving those changes. However, as with the Peel-Harvey Estuary (Wildsmith et al., 2009), it is not clear precisely which factors have led to the faunal changes in the Swan-Canning Estuary. In this context, it is relevant that, in a study of estuaries in Tasmania, which was conducted on a spatial rather than a temporal scale, Edgar and Barrett (2000) found that there was a strong correlation between indices of benthic disturbance and population density in the catchment. Those authors listed a number of factors that could contribute to the human impacts, including nutrification, eutrophication, seagrass loss and discharge of urban and industrial waste. It is proposed that the deleterious changes apparently undergone by the benthic environment in the basin of the Swan-Canning Estuary are also due to a combination of anthropogenic effects rather than to a single factor.

4.3. Faunal characteristics exhibiting little change between 1986/7 and 2003/4

From the above, it is evident that the trends exhibited by the overall number of macroinvertebrate species and the densities of crustaceans, molluscs and polychaetes in the basin of the Swan-Canning Estuary are consistent with the hypothesis that anthropogenic activities have had a deleterious effect on the benthic environment of this estuary. However, such trends are not consistently shown by other indicators that are traditionally regarded as symptoms of environmental stress. Thus, for example, taxonomic distinctness, which normally declines with increasing levels of perturbation, declined consistently between periods only at Applecross (Fig. 5). As Applecross is the shallowest of the sampling sites, it is the most prone to warming as a result of increasing air temperatures in south-

western Australia since the 1950s (Department of Agriculture Fisheries and Forestry and Bureau of Meteorology, 2008), and, particularly in summer and autumn, is also prone to the deposition of substantial amounts of decomposing macroalgae, comprising

mainly *Cladophora* and *Chaetomorpha* species (Wildsmith, 2007). It is thus relevant that such macroalgal accumulations can have marked detrimental influences on benthic infauna (Cummins et al., 2004).

The fact that taxonomic distinctness declined at only one of the four sampling sites contrasts with the situation recorded between the same two periods in the nearby Peel-Harvey Estuary, in which this measure of biodiversity fell in each corresponding season at each of the four sampling sites (Wildsmith et al., 2009). Likewise, the overall variability in species composition, as reflected by MVDISP values, was only marginally greater in 2003/4 than in 1986/7. This differs from the situation in the Peel-Harvey Estuary, where the composition varied more markedly among samples in 2003/4, indicating the presence of substantial environmental stress (Warwick and Clarke, 1993a; Wildsmith et al., 2009).

5. Conclusions

This study has demonstrated that the benthic macroinvertebrate fauna in the large basin of the Swan-Canning Estuary in 2003/4 differed from that in 1986/7 in certain ways that would be expected in a system that had been subjected to increased environmental stress. These included a pronounced change in species composition, marked reductions in the densities of crustaceans, which are especially susceptible to the effects of environmental perturbation, and an increase in the densities of polychaetes, which are more tolerant of such perturbations. It is thus relevant that the Swan-Canning Estuary has become more eutrophic, contaminated and disturbed during the intervening period. Yet, other features that would indicate that environmental stress had increased, such as a greater variability in composition among samples and a reduction in diversity at all sites, were not exhibited. Such changes did take place, however, in the basin of the Peel-Harvey Estuary between the same two

periods. This suggests that, while detrimental changes have occurred in the basin of the Swan-Canning Estuary, they are not as pronounced as in the basin of the Peel-Harvey Estuary. It is thus proposed that the characteristics of the benthic macrofauna act as good indicators of the extent of environmental perturbations in microtidal estuaries. Such indicators will become increasingly valuable in regions such as south-western Australia if the quality of the environment in these microtidal estuaries continues to decline.

The lamentable deteriorating condition of south-western Australian estuaries (see also Davis and Koop, 2006) contrasts with the situation in Europe, where there have been widespread attempts to restore the condition of estuaries that have been polluted for decades or even centuries. The European Union has been a powerful factor in initiating ecological studies of benthic environments, both through legislation and the encouragement of collaborative research programmes. As a result, European estuaries, in general, have become much cleaner (McLusky, 1999). There is thus clearly a need for a similar co-ordinated and focused programme of research and legislation to address the environmental problems to which the microtidal estuaries of south-western Australia continue to suffer.

References

Anderson, M.J., 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecology 26, 32–46.

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

Astill, H., Lavery, P.S., 2001. The dynamics of unattached benthic macroalgal accumulations in the Swan-Canning Estuary. Hydrological Processes 15, 2387– 2399.

Australian Bureau of Statistics, 1987. Persons and dwellings in legal local government areas, statistical local areas and urban areas/(rural) localities –Census, 1986.

Australian Bureau of Statistics, 2001. Selected social and housing characteristics for statistical and local areas. Western Australia, Cocos (Keeling) and Christmas islands. Census of Population and Housing

Beesley, P.L., Ross, G.J.B., Wells, A., 1998. Fauna of Australia, Volume 5. Mollusca: the Southern Synthesis. Australian Biological Resources Study. CSIRO Publishing, Melbourne.

Chan, T.U., Hamilton, D.P., Robson, B.J., Hodges, B.R., Dallimore, C., 2002. Impacts of hydrological changes on phytoplankton succession in the Swan River, Western Australia. Estuaries 25, 1406–1415.

Clarke, K.R., 1993. Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology 18, 117–143.

Clarke, K.R., Gorley, R.N., 2006. PRIMER v6: User Manual/Tutorial. PRIMER-E, Plymouth, UK.

Cummins, S.P., Roberts, D.E., Zimmerman, K.D., 2004. Effects of the green macroalga *Enteromorpha intestinalis* in a shallow coastal estuary. Marine Ecology Progress Series 266, 77–87.

Constable, A.J., 1999. Ecology of benthic macroinvertebrates in soft-sediment environments: a review of progress towards quantitative models and predictions. Australian Journal of Ecology 24, 452–476.

Davis, J.R., Koop, K., 2006. Eutrophication in Australian rivers reservoirs and estuaries – a southern hemisphere perspective on the science and its implications. Hydrobiologia 559, 23–76.

Department of Agriculture Fisheries and Forestry and the Bureau of Meteorology, 2008. Observed climate change: Western Australia. Communicating Climate Change – An Initiative of the National Agriculture and Climate Change Action Plan, Government of Australia.

Department of Planning and Infrastructure, 2009. The Perth Recreational Boating Facilities Study 2008. Technical Report Number 444.

Edgar, G.J., Barrett, N.S., 2000. Effects of catchment activities on macrofaunal assemblages in Tasmanian estuaries. Estuarine, Coastal and Shelf Science 50, 639–654

Gerriste, R.G., Wallbrink, P.J., Murray, A.S., 1998. Accumulation of phosphorus and heavy metals in the Swan-Canning Estuary, Western Australia. Estuarine, Coastal and Shelf Science 47, 165–179.

Gray, J.S., Wu, R.S.S., Or, Y.Y., 2002. Effects of hypoxia and organic enrichment on the coastal marine environment. Marine Ecology Progress Series 238, 249–279.

Gómez Gesteira, J.L., Dauvin, J.C., 2000. Amphipods are good bio-indicators of the impact of oil spills on soft-bottom macrobenthic communities. Marine Pollution Bulletin 40, 1017–1027.

Hamilton, D.P., Turner, J.V., 2001. Integrating research and management for an urban estuarine system, the Swan-Canning Estuary. Western Australia, Hydrological Processes 15, 2383–2385.

Hodgkin, E.P., Hesp, P., 1998. Estuaries to salt lakes: Holocene transformation of the estuarine ecosystems of south-western Australia. Australian Journal of Marine and Freshwater Research 49, 183–201.

Hoeksema, S.D., Potter, I.C., 2006. Diel, seasonal, regional and annual variations in the characteristics of the ichthyofauna of the upper reaches of a large Australian microtidal estuary. Estuarine, Coastal and Shelf Science 67, 503–520.

Hutchings, P., 1998. Biodiversity and functioning of polychaetes in benthic sediments. Biodiversity and Conservation 7, 1133–1145.

Kanandjembo, A.N., Platell, M.E., Potter, I.C., 2001. The benthic macroinvertebrate community of the upper reaches of an Australian estuary that undergoes marked seasonal changes in hydrology. Hydrological Processes 15, 2481–2501.

Karlson, K., Rosenberg, R., Bonsdorff, E., 2002. Temporal and spatial large-scale effects of eutrophication and oxygen deficiency on benthic fauna in Scandinavian and Baltic waters - a review. Oceanography and Marine Biology: an Annual Review 40, 427–489.

Kemp, W.M., Boynton, W.R., Adolf, J.E., Boesch, D.F., Boicourt, W.C., Brush, G., Cornwell, J.C., Fisher, T.R., Glibert, P.M., Hagy, J.D., Harding, L.W., Houde, E.D., Kimmel, D.G., Miller, W.D., Newell, R.I.E., Roman, M.R., Smith, E.M., Stevenson, J.C., 2005. Eutrophication of Chesapeake Bay: historical trends and ecological interactions. Marine Ecology Progress Series 303, 1–29.

McArdle, B.H., Anderson, M.J., 2001. Fitting multivariate models to community data: a comment on distance-based redundancy analysis. Ecology 82, 290–297.

McLusky, D.S., 1999. Estuarine benthic ecology: a European perspective. Australian Journal of Ecology 24, 302–311.

Nice, H.E., 2009. A Baseline Study of Contaminants in the Sediments of the Swan and Canning Estuaries, Water Science Technical Series Report No. 6. Department of Water, Western Australia.

Pearson, T.H., Rosenberg, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology: An Annual Review 16, 229–311.

Pelletier, M.C., Gold, A.J., Heltsche, J.F., Buffam, H.W., 2010. A method to identify estuarine macroinvertebrate pollution indicator species in the Virginian Biogeographic Province. Ecological Indicators 10, 1037–1048.

Radke, L.C., Prosser, I.P., Robb, M., Brooke, B., Fredericks, D., Douglas, G.B., Skemstad, J., 2004. The relationship between sediment and water quality, and riverine sediment loads in the wave dominated estuaries of south-west Western Australia. Marine and Freshwater Research 55, 581–596.

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

Roach, A.C., Wilson, S.P., 2009. Ecological impacts of tributyltin on estuarine communities in the Hastings River, NSW Australia. Marine Pollution Bulletin 58, 1780–1786.

Robson, B.J., Hamilton, D.P., 2003. Summer flow event induces a cyanobacterial bloom in a seasonal Western Australian estuary. Marine and Freshwater Research 54, 139–151.

Rose, T.H., 1994. Comparisons of the benthic and zooplankton communities in the eutrophic Peel-Harvey and nearby Swan estuaries in South-Western Australia. Ph.D. Thesis, Murdoch University, Western Australia.

Sikora W.B., Sikora, J.P., 1982. Ecological characterization of the benthic community of Lake Pontchartrain, Louisiana. Prepared for US Army Engineer District, New Orleans. Contract No. DACW29-79-C-0099, 214 p.

Swan River Trust, 2000. 'Summer surprise'. The Swan River blue-green algal bloom February 2000. Riverscience, Issue 2, September 2000.

Swan River Trust, 2005. Algal blooms and the Swan-Canning Estuary: patterns, causes and history. Riverscience, Issue 3, February 2005.

Swan River Trust, 2007. Tributyltin and heavy metals survey at yacht clubs in the Swan River: Sediment and mussel tissue quality. Community Summary Paper, November 2007.

Swan River Trust, 2009. Swan Canning Water Quality Improvement Plan.

Thomsen, M.S., Wernberger, T., Tuya, F., Silliman, B.R., 2010. Ecological performance and possible origin of a ubiquitous but under-studied gastropod. Estuarine Coastal and Shelf Science 87, 501–509.

Viney, N.R., Sivapalan, M., 2001. Modelling catchment processes in the Swan-Avon river basin. Hydrological Processes 15, 2671–2685.

Warwick, R.M., Clarke, K.R., 1993a. Comparing the severity of disturbance: a metaanalysis of marine macrobenthic community data. Marine Ecology Progress Series 92, 221–231.

Warwick, R.M., Clarke, K.R., 1993b. Increased variability as a symptom of stress in marine communities. Journal of Experimental Marine Biology and Ecology 172, 215–226.

Warwick, R.M., Clarke, K.R., 1998. Taxonomic distinctness and environmental assessment. Journal of Applied Ecology 35, 532–543.

Warwick, R.M., Clarke, K.R., 2001. Practical measures of marine biodiversity based on relatedness of species. Oceanography and Marine Biology: An Annual Review 39, 207–231.

Warwick, R.M., Somerfield, P.J., 2010. The structure and functioning of the benthic macrofauna of the Bristol Channel and Severn Estuary, with predicted effects of a tidal barrage. Marine Pollution Bulletin 61, 92–99.

Warwick, R.M., Ashman, C.M., Brown, A.R., Clarke, K.R., Dowell, B., Hart, B., Lewis, R.E., Shillabeer, N., Somerfield, P.J., Tapp, J.F., 2002. Inter-annual changes in the biodiversity and community structure of the macrobenthos in Tees Bay and the Tees estuary, UK, associated with local and regional environmental events. Marine Ecology Progress Series 234, 1–13.

Weisberg, S.B., Ranasinghe, J.A., Dauer, D.M., Schaffner, L.C., Diaz, R.J., Frithsen, J.B., 1997. An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. Estuaries 20, 149–158.

Whomersly, P., Huxham, M., Bolam, S., Shratzberger, M., Augley, J., Ridland, D., 2010. Response of intertidal macrofauna to multiple disturbance types and intensities – an experimental approach. Marine Environmental Research 69, 297–308.

Wildsmith, M.D., 2007. Relationships between benthic macroinvertebrate assesmblages and habitat types in neasrshore marine and estuarine waters along the lower west coast of Australia. Ph.D. Thesis, Murdoch University.

Wildsmith, M.D., Rose, T.H., Potter, I.C., Warwick, R.M., Clarke, K.R., Valesini, F.J., 2009. Changes in the benthic macroinvertebrate fauna of a large microtidal estuary following extreme modifications aimed at reducing eutrophication. Marine Pollution Bulletin 58, 1250–1262.

Wilson, R.S., Heislers, S., Poore, G.C.B., 1998. Changes in benthic communities of Port Phillip Bay, Australia, between 1969 and 1995. Marine and Freshwater Research 49, 847–861.

Zammit, C., Sivapalan, M., Kelsey, P., Viney, N.R., 2005. Modelling the effects of land-use modifications to control nutrient loads from an agricultural catchment in Western Australia. Ecological Modelling 187, 60–70.

Fig. 1. Map showing the location of the four sites sampled in the Swan-Canning Estuary in each season between the winters of 1986 and 1987 and of 2003 and 2004. Arrow in insert shows the location of the Swan-Canning Estuary in Western Australia.
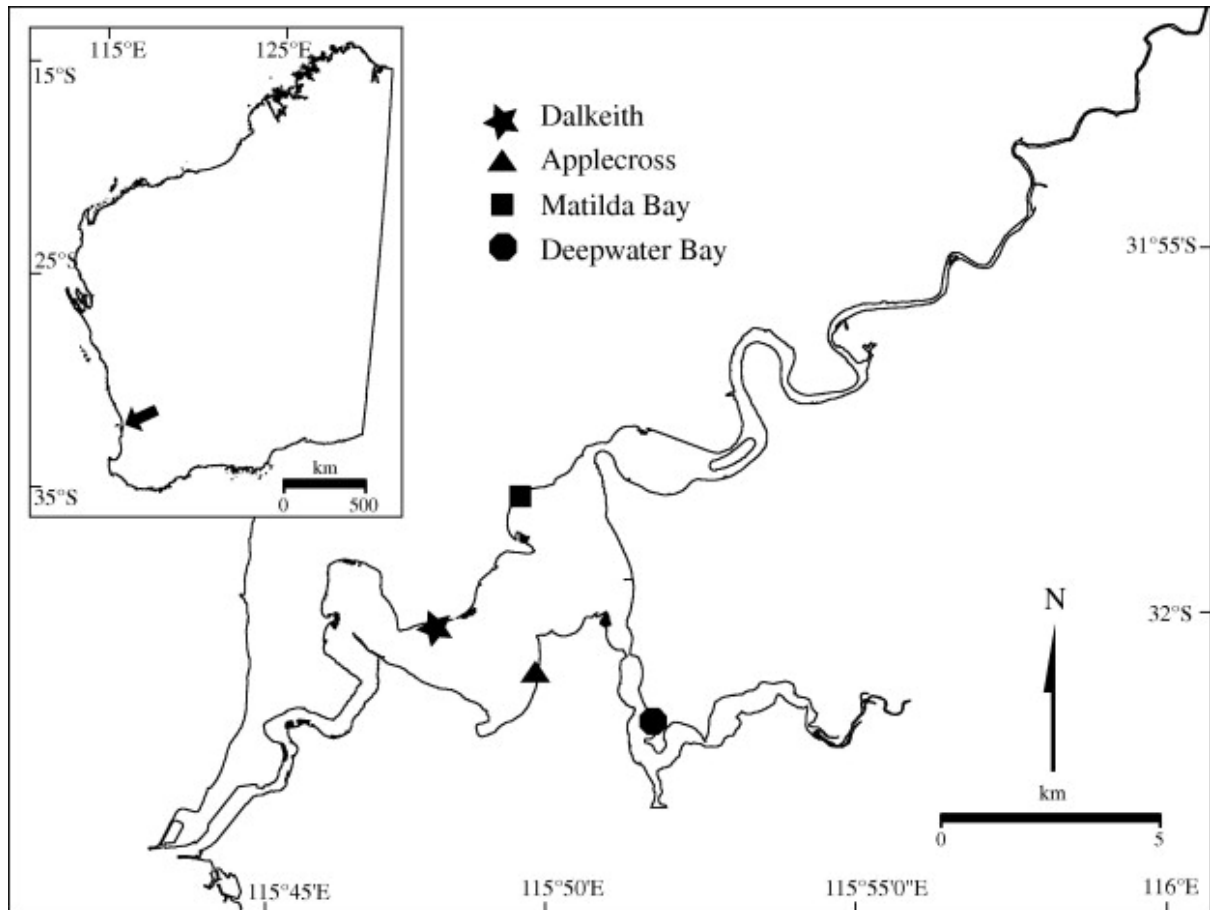
Fig. 2. Means ± 95% confidence limits for (a) temperature, (b) salinity and (c) dissolved oxygen concentration at the bottom of the water column at the four sampling sites in each season between the winters of 1986 and 1987 and of 2003 and 2004.
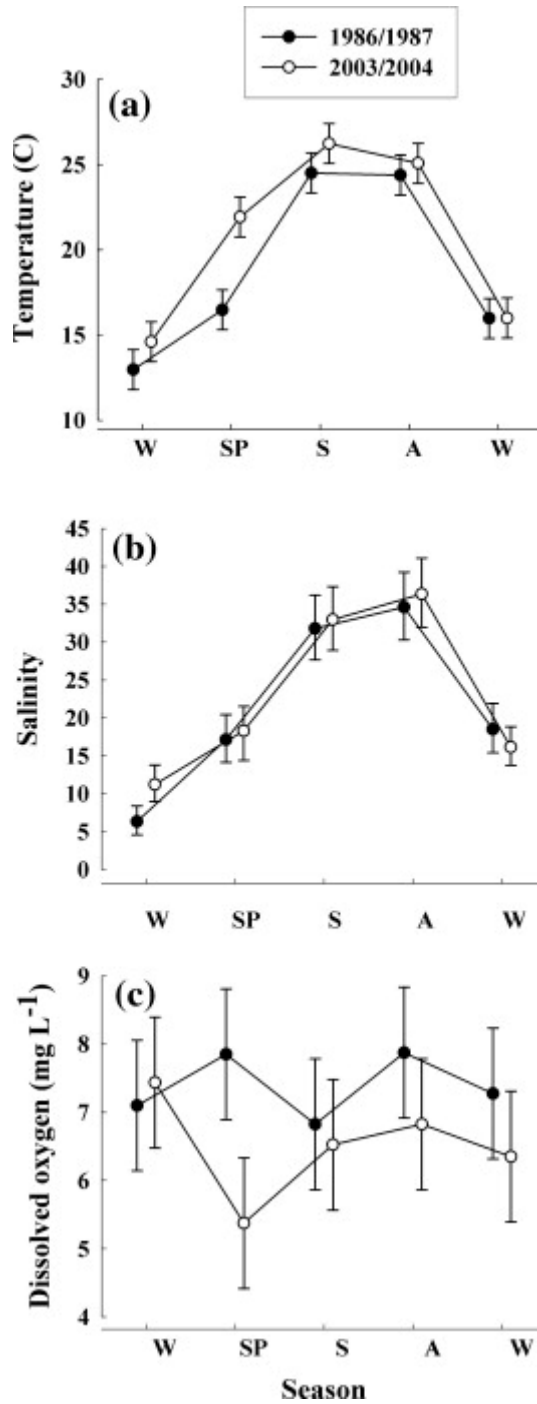
Fig. 3. Mean values for (a and b) the depth of the transition layer in the sediments and (c and d) the % contributions of particulate organic material to the sediments at the four sampling sites in the Swan-Canning Estuary between the winters of 1986 and 1987 and of 2003 and 2004. The overall mean ± 95% confidence limits for the depth of the transition layer and %POM in each period are shown.
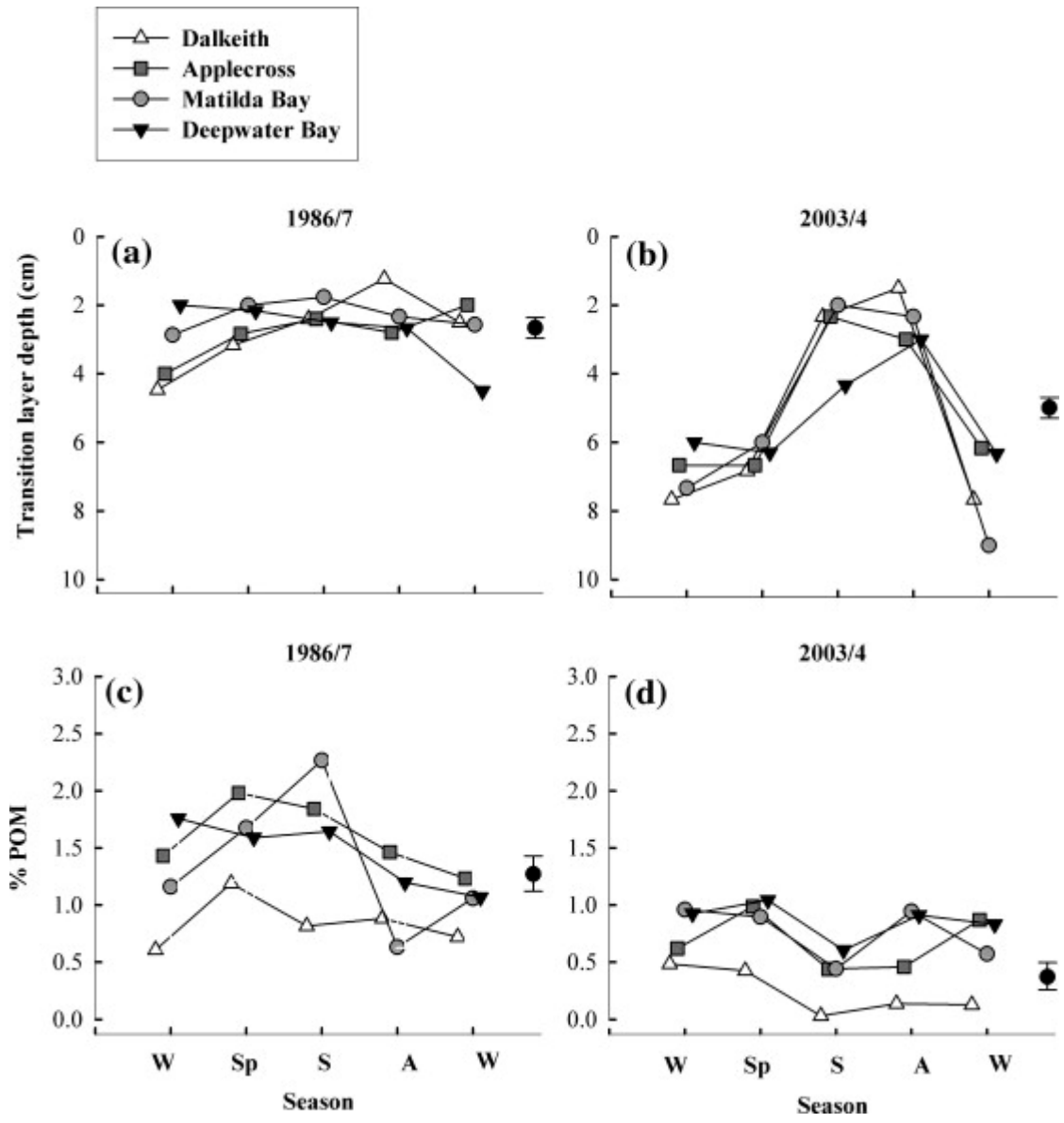
Fig. 4. Means ± 95% confidence limits for (a) number of species and (c) densities of benthic macroinvertebrates in each season between the winters of 1986 and 1987 and of 2003 and 2004 and for (b) number of species and (c) densities at each sampling site. Dk, Dalkeith; Ac, Applecross; MB, Matilda Bay; DB, Deepwater Bay.
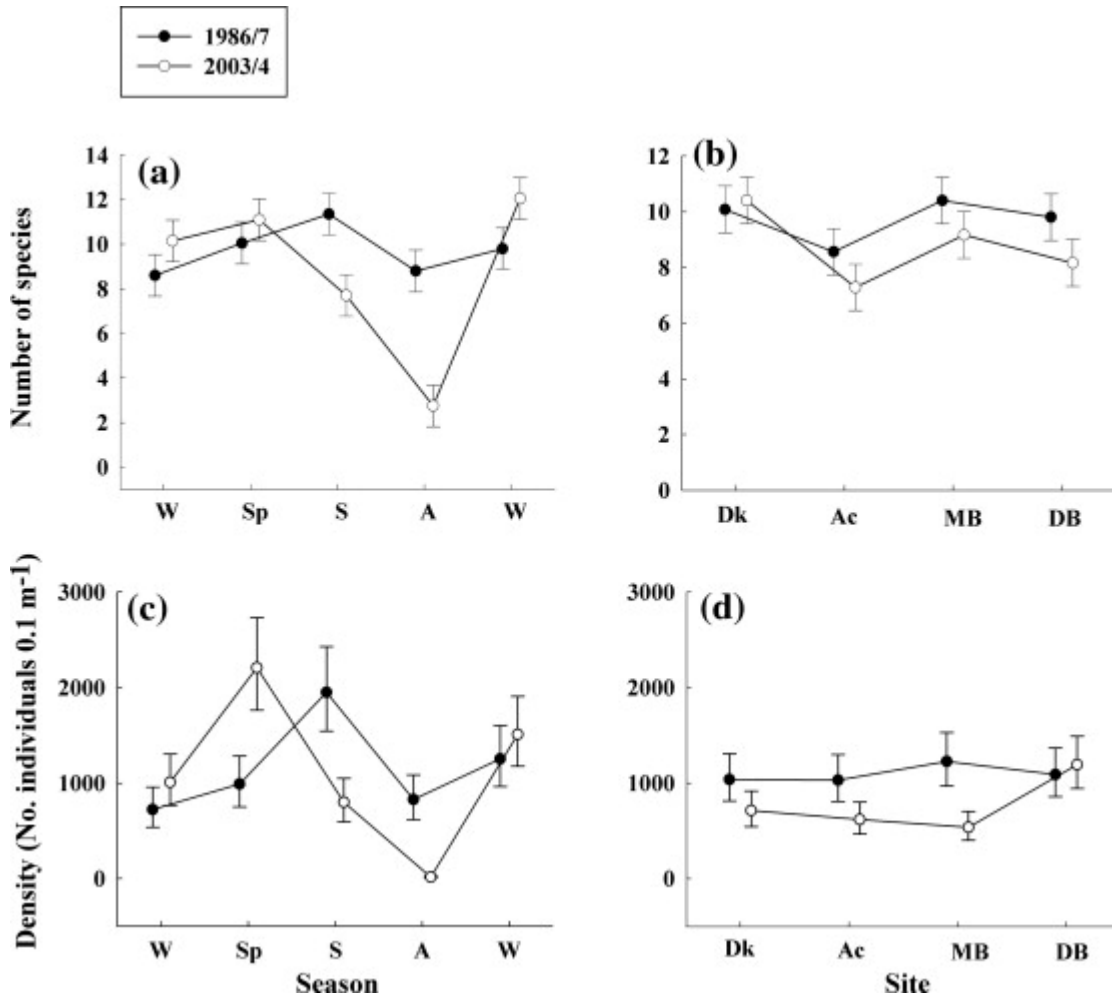
Fig. 5. Means ± 95% confidence limits for taxonomic distinctness ($\Delta*$) at the four sampling sites in the Swan-Canning Estuary between the winters of 1986 and 1987 and of 2003 and 2004.
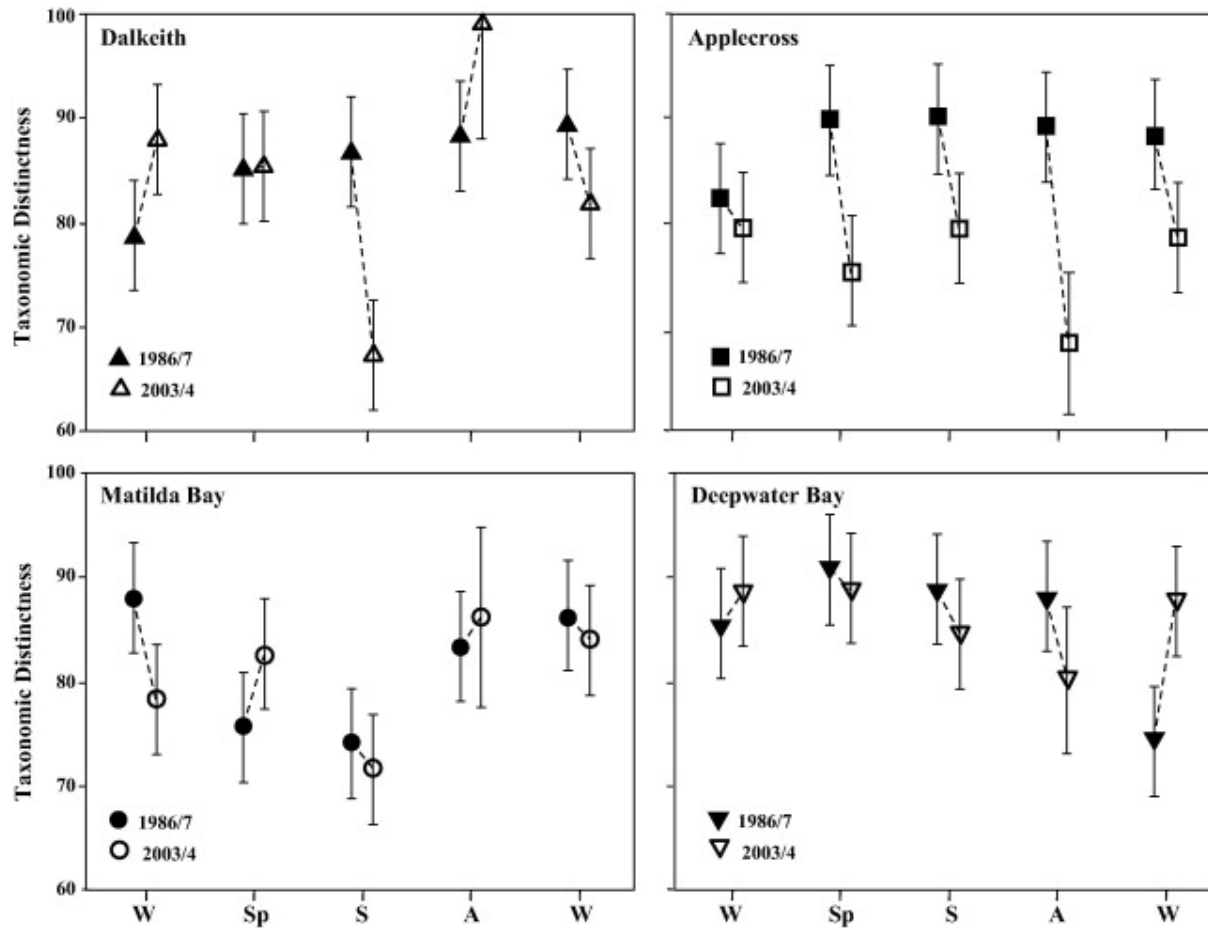
Fig. 6. Non-metric multidimensional scaling ordination, derived from the Bray–Curtis similarity matrix constructed using fourth root transformed mean densities of the species of benthic macroinvertebrates at each sampling site in each season between the winters of 1986 and 1987 and of 2003 and 2004. Dk, Dalkeith; Ac, Applecross; MB, Matilda Bay; DB, Deepwater Bay. N.B. The four sample outliers in 2003/4 are not shown.
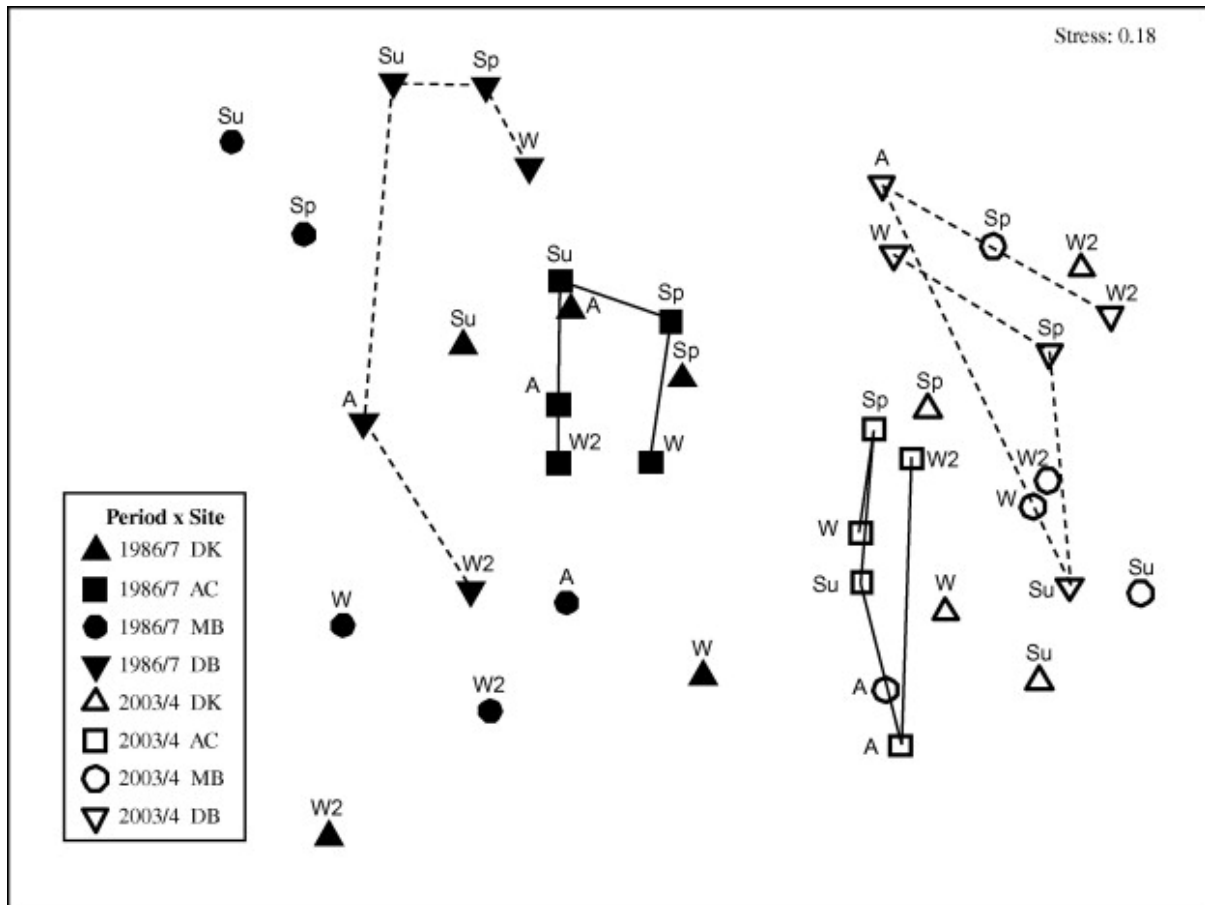
Fig. 7. Non-metric multidimensional scaling ordination, derived from the Bray–Curtis similarity matrix constructed using log transformed densities of the species of benthic macroinvertebrates in each sample from each site in (a)–(f) each season between the winters of 1986 and 1987 and of 2003 and 2004. Note that, as many of the points in the plot for autumn form a tight group (d), that area of the plot was expanded (e).
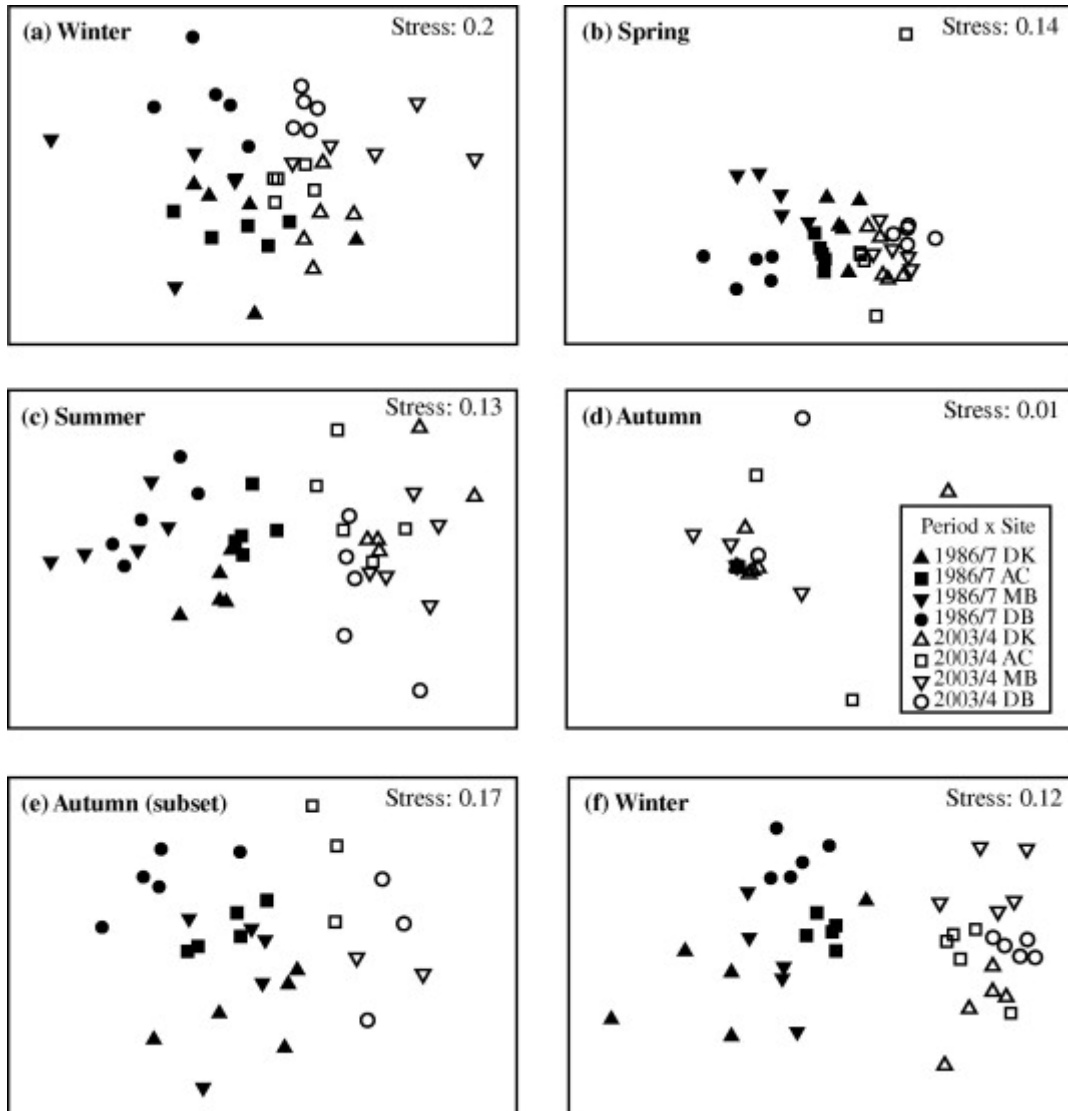
Fig. 8. Non-metric multidimensional scaling ordination, derived from the Bray–Curtis similarity matrix constructed using fourth root transformed mean densities of (a) the species, (b) the families and (c) the phyla of benthic macroinvertebrates at the four sampling sites between the winters of 1986 and 1987 and of 2003 and 2004.
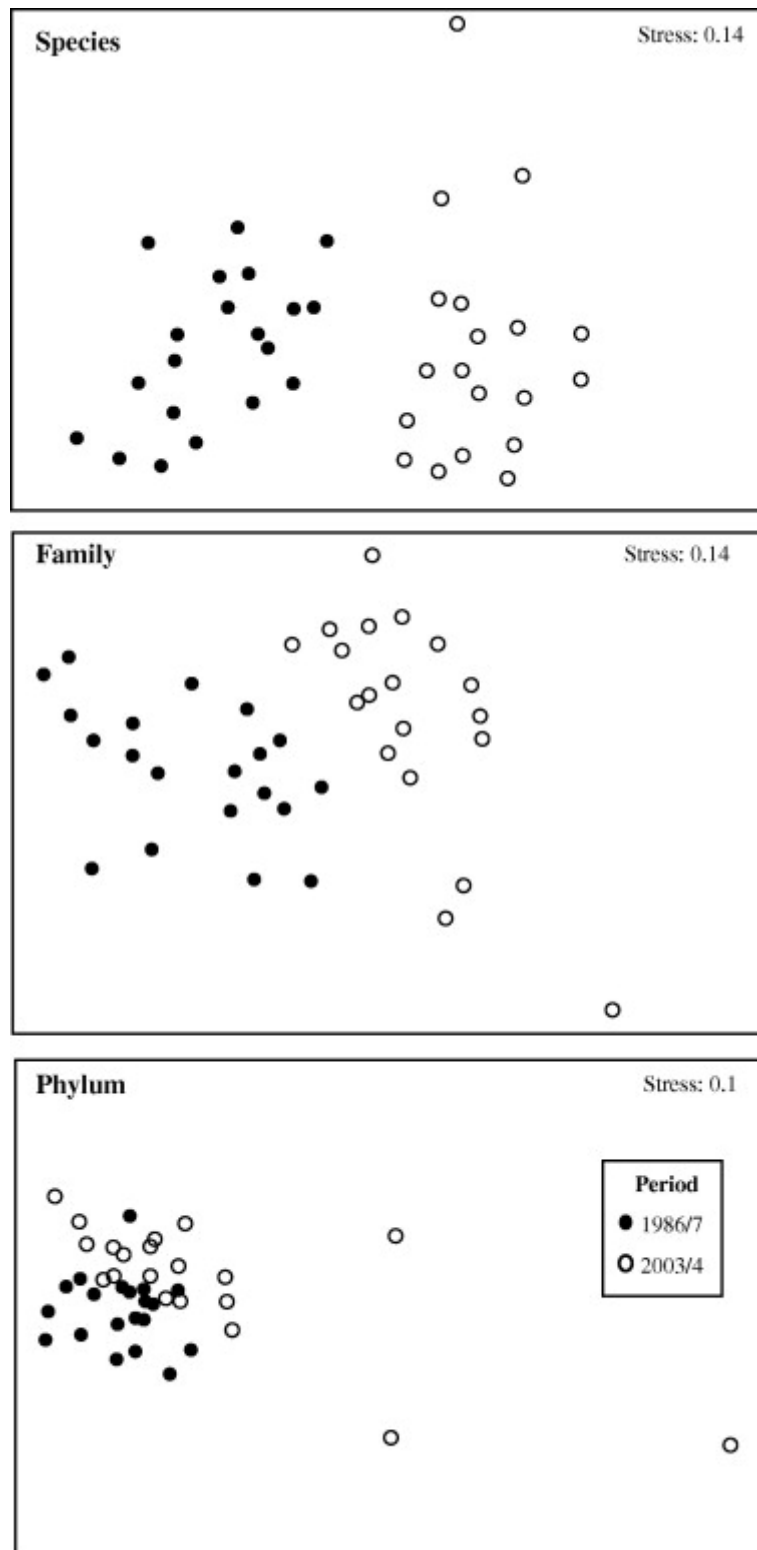
Table 1. Mean squares (MS) and their significance levels (*P*) for ANOVAs on temperature, salinity and dissolved oxygen concentration in the water column at the four sampling sites in the basins of the Swan-Canning Estuary between the winters of 1986 and 1987 and of 2003 and 2004. df, degrees of freedom. Significant results are highlighted in bold.

	df	Water temperature		Salinity		Dissolved oxygen	
		MS	*P*	MS	*P*	MS	*P*
Main effects							
Period	1	**34.95**	**<0.001**	**1.038**	**0.008**	**7.79**	**0.001**
Season	4	**216.09**	**<0.001**	**15.7**	**<0.001**	0.94	0.134
Site	3	0.86	0.562	0.17	0.233	**2.29**	**0.015**
Two-way interactions							
Period × season	4	**9.28**	**0.003**	**2.56**	**<0.001**	**2.2**	**0.013**
Period × site	3	0.93	0.531	0.18	0.212	0.52	0.352
Season × site	12	1.41	0.396	0.15	0.251	1.07	0.065
Error	30	1.22		0.1		0.433	

Table 2. Mean squares (MS) and their significance levels (*P*) for ANOVAs on the depth of the transition layer and %POM at the four sampling sites in the Swan-Canning Estuary between the winters of 1986 and 1987 and of 2003 and 2004. df, degrees of freedom. Significant results are highlighted in bold.

	df	Transition layer		%POM	
		MS	***P***	**MS**	***P***
Main effects					
Period	1	**180.08**	**<0.001**	**1.85**	**<0.001**
Season	4	**42.97**	**<0.001**	**0.2**	**0.029**
Site	3	2.4	0.278	**0.44**	**0.001**
Two-way interactions					
Period × season	4	**25.5**	**<0.001**	0.07	0.451
Period × site	3	**5.11**	**0.046**	0.03	0.757
Season × site	12	**5.82**	**0.001**	0.08	0.383
Three-way interactions					
Period × season × site	12	**4.3**	**0.012**	0.12	0.073
Error	160	1.84		0.07	

Table 3. Mean squares (MS) and their significance levels (*P*) for ANOVAs for the number of species, density, and taxonomic distinctness of the benthic macroinvertebrates at the four sampling sites in the Swan-Canning Estuary between the winters of 1986 and 1987 and of 2003 and 2004. df, degrees of freedom. Significant results are highlighted in bold.

	df	Number of species		Density		Taxonomic distinctness	
		MS	*P*	MS	*P*	MS	*P*
Main effects							
Period	1	**46.08**	**0.002**	**14.64**	**<0.001**	**754.34**	**0.001**
Season	4	**169.01**	**<0.001**	**43.92**	**<0.001**	**103.85**	**0.025**
Site	3	**51.69**	**<0.001**	**2.5**	**0.02**	**231.44**	**0.001**
Two-way interactions							
Period × season	4	**133.21**	**<0.001**	**33.34**	**<0.001**	**148.99**	**0.003**
Period × site	3	9.51	0.1	**3.28**	**0.004**	**317.75**	**0.001**
Season × site	12	**10.96**	**0.006**	**1.76**	**0.006**	**143.9**	**0.001**
Three-way interactions							
Period × season × site	12	7.38	0.083	0.59	0.64	**180.36**	**0.001**
Error	160	4.48		0.72		36.87	

Table 4. Species detected by SIMPER as distinguishing the benthic macroinvertebrate assemblage in the basins of the Swan-Canning Estuary in 1986/7 from that in 2003/4. The period in which the densities were greatest and any high ratios of dissimilarity to standard deviation (Diss/SD) are highlighted in bold for each species and the % contribution of each species to the dissimilarity between the compositions in each period are given.

	Mean density		Diss/SD	% Contribution
	1986/7	2003/4		
Grandidierella propodentata	**57.87**	31.55	**1.01**	7.11
Arthritica semen	**26.53**	2.85	**1.13**	6.86
Sanguinolaria biradiata	7.51	**38.49**	**1.17**	6.56
Boccardiella limnicola	**14.73**	0.00	**1.16**	6.55
Tanais dulongii	**13.15**	0.02	0.98	6.04
Capitella capitata	55.77	**64.45**	0.78	5.96
Pseudopolydora kempi	0.00	**15.33**	**1.16**	5.73
Corophium minor	6.49	**8.63**	**1.11**	5.62
Ceratonereis aequisetis	**81.61**	46.01	0.70	5.46
Leitoscoloplos normalis	26.53	**33.26**	0.92	5.01
Australonereis ehlersii	0.07	**4.10**	0.85	3.88

Table 5. Species detected by SIMPER as distinguishing between the benthic macroinvertebrate assemblage in the basins of the Swan-Canning Estuary in 1986/7 from that in 2003/4. The period in which the densities were greatest and any high ratios of dissimilarity to standard deviation (Diss/SD) have been highlighted in bold for each species and the % contribution of each species to the dissimilarity between the compositions in each period are given.

	Distinguishing species	Mean density		Diss/SD	Contrib.%	Cum.%
		1986/7	**2003/4**			
Winter 1	*Sanguinolaria biradiata*	3.84	**65.97**	**1.63**	8.38	8.38
	Arthritica semen	**14**	13.71	**1.28**	6.38	14.76
	Tanais dulongii	**7.13**	0.08	0.98	6.23	21
	Ceratonereis aquisetis	1.36	**8.74**	**1.37**	6.14	27.14
	Corophium minor	3.97	**22.82**	**1.29**	6.1	33.24
	Grandidierella propodentata	21.09	**23.41**	**1.11**	6	39.25
Spring	*Pseudopolydora* sp. 2		**59.64**	**2.92**	10.71	10.71
	Sanguinolaria biradiata	3.84	**103.93**	**1.55**	8.24	18.95
	Oligochaete sp. 5		**14.14**	**1.25**	6.23	25.17
	Musculista senhousia	2.04	**9.51**	**1.12**	5.96	31.13
	Leitoscoloplos normalis	25.46	**94.01**	**1.7**	5.88	37.01
	Boccardiella limnicola	**7.61**		**1.03**	5.52	42.53
	Corophium minor	29.66	**46.93**	**1.11**	5.34	47.88
	Capitella capitata	46.62	**104.44**	**1.09**	5	52.87
Summer	*Boccardiella limnicola*	**42.43**		**1.85**	9.1	9.1
	Corophium minor	**44.79**	0.04	**1.86**	7.95	17.05
	Arthritica semen	**50.73**	0.66	**1.66**	7.6	24.65
	Grandidierella propodentata	**143.77**	19.81	**1.45**	7.57	32.22
	Pseudopolydora kempi		**27.18**	**1.65**	7.45	39.67
	Tanais dulongii	**25.46**		**1.24**	7.16	46.83
	Capitella capitata	16.74	**78.99**	**1.3**	5.79	52.61
	Sanguinolaria biradiata	5.17	**30.83**	**1.25**	5.21	57.83
Autumn	*Grandidierella propodentata*	**39.86**	0.05	**1.77**	11.6	11.6
	Ceratonereis aquisetis	**76.86**	4.37	**1.3**	11.18	22.78
	Capitella capitata	**50.73**	3.07	**1.34**	10.34	33.12
	Sanguinolaria biradiata	**18.93**	1.79	**1.36**	9.63	42.74
	Arthritica semen	**30.83**	0.11	**1.25**	8.99	51.74
	Leitoscoloplos normalis	**32.52**	2.35	**1.27**	8.88	60.62
	Boccardiella limnicola	**16.26**		**1.32**	8.69	69.3

	Tanais dulongii	**10.56**		**1.02**	6.96	76.26
Winter 2	*Pseudopolydora kempi*		**57.16**	**3.58**	9.27	9.27
	Ceratonereis aquisetis	0.02	**44.79**	**2.18**	7.96	17.23
	Tanais dulongii	**30.36**	0.03	**1.64**	7.08	24.3
	Boccardiella limnicola	**29.43**	0.01	**2.13**	7.02	31.32
	Sanguinolaria biradiata	3.65	**48.81**	**1.47**	5.74	37.06
	Corophium minor	0.14	**16.26**	**1.49**	5.26	42.32

Table 6. The species, families and phyla identified by SIMPER as contributing to ~90% of the dissimilarity between the benthic macroinvertebrate fauna in 1986/7 and 2003/4. (M), Mollusca; (C), Crustacea; (P), Polychaeta; (Ne), Nemertea; (An), Anthozoa; (S) Sipuncula. * denotes taxa that were more abundant in 2003/4 than in 1986/7.

Species	Family	Phylum
Tanais dulongii (C)	Tanaidae (C)	Arthropoda
Boccardiella limnicola (P)	Psammobiidae (M)*	Annelida*
Pseudopolydora kempi (P)*	Corophiidae (C)	Mollusca
Sanguinolaria biradiata (M)*	Mytillidae (M)	Sipuncula*
Oligochaete spp. (O)*	Oligochaeta (O)*	Nemertea
Grandidierella propodentata (C)	Galeomatidae (M)	Cnidaria
Arthritica semen (M)	Spionidae (P)*	
Corophium minor (C)	Melitidae (C)	
Australonereis ehlersii (P)*	Capitellidae (P)*	
Musculista senhausia (M)	Sphaeromatid (C)	
Capitella spp. (P)*	Nereididae (P)	
Xenostrobus securis (M)	Nassaridae (M)*	
Syncassidina aestuaria (An)	Orbinidae (P)*	
Ceratonereis aequisetis (P)	Eunicidae (P)	
Nassarius burchardii (M)*	Mesodesmatidae (M)*	
Leitoscoloplos normalis (P)*	Hydrobiidae (M)	
Melita matilda (C)	Eusiridae (C)	
Marphysa sanguinea (P)	Veneridae (M)	
Prionospio cirrifera (P)*	Potamididae (M)*	
Paracorophium excavatum (C)	Hymenosomatidae (C)	
Tellina deltoidalis (M)*	Tellinidae (M)*	
Sipunculan sp. (S)*	Mactridae (M)*	
Polydora socialis (P)*	Caprellidae (C)	
Caranziella victoriensis (P)*	Haliplanellidae (An)	
Melita zylanica (C)	Serpulidae (P)	
Nemertean sp. (Ne)		
Tatea preseii (M)		
Venerupis crenata (M)		
Halicarcinus bedfordi (C)		
Batillaria australis (M)*		
Hydrobia sp. (M)		
Caprellid sp. (C)		
Ficopomatus enigmatus (P)		
Spisula trigonella (M)*		

Table A1. Mean density (No. 0.1 m^{-1}) (M), standard deviation (SD), percentage contributions to the total number of individuals (%C), cumulative percentage to the total number of individuals (Cmt%) and the rank by density (R) of the benthic macroinvertebrate taxa in samples collected at four sites in the basin of the Swan-Canning Estuary in five seasons during 1986/7 and 2003/4. Each taxon has been assigned to its respective phyla (Ph) (A, Annelida; Ar, Arthropoda; M, Mollusca; S, Sipuncula; Ne, Nematoda; T, Turbellaria; Cn, Cnidaria; Ch, Chordata; N, Nemertea; P, Platyhelminthes) and predominant feeding mode (F) (DF, deposit feeder; SF, suspension feeder; Pr, predator; U, unknown). The overall mean density and standard deviation (individuals 0.1 m^{-2}), the mean number of taxa and standard deviation, the total number of individuals and the total number of taxa are also shown for each period.

	Ph	F	1986/7					2003/4				
			M	SD	%C	Cmt%	R	M	SD	%C	Cmt%	R
Ceratonereis aquisetus	A	DF, Pr	239.6	187.5	19.4	19.4	1	146.4	130.7	12.1	12.1	4
Grandidierella propodentata	Ar	DF	231.4	278.5	18.7	38.2	2	212.9	312.5	17.5	29.6	2
Capitella spp.	A	DF	219.7	289.1	17.8	56	3	251.4	206.7	20.7	20.7	1
Tanais dulongii	Ar	DF	135.9	318.9	11	67	4	2.2	9.4	0.2	50.5	18
Arthritica semen	M	DF, SF	94.7	132.1	7.7	74.7	5	19.8	38.1	1.6	52.1	12
Corophium minor	Ar	DF	61.5	127.2	5	79.7	6	60.4	126.8	5	57.1	7
Boccardiella limnicola	A	DF, SF	58.4	88.5	4.7	84.4	7	0.5	4.3	<0.1	57.2	27
Leitoscoloplos normalis	A	DF	47.9	45	3.9	88.3	8	107.5	131.5	8.9	66	5
Sanguinolaria biradiata	M	SF	35.3	53.1	2.9	91.2	9	160.1	231.5	13.2	79.2	3
Musculista senhausia	M	SF	22.6	77.4	1.8	93	10	33.3	114.8	2.8	81.9	9
Paracorophium excavatum	Ar	DF	17.1	66.3	1.4	94.4	11	0.4	2.1	<0.1	82	29
Xenostrobus securis	M	SF	16.5	43	1.3	95.7	12					
Syncassidina aestuarina	Ar	DF	15.9	47.6	1.3	97	13					
Melita matilda	Ar	DF	7	19.2	0.6	97.6	14					
Melita zeylanica	Ar	DF	4.6	17.9	0.4	97.9	15					
Marphysa sanguinea	A	DF	3.5	6.3	0.3	98.2	16	1.3	4.3	0.1	82.1	21
Australonereis ehlersii	A	DF	2.8	10.8	0.2	98.5	17	44.7	113.8	3.7	85.8	8
Nemertean sp.	N	Pr	2.7	9.6	0.2	98.7	18					
Venerupis crenata	M	SF	2.1	8	0.2	98.8	19	0.6	2.9	0.1	85.8	25
Caprella scaura	Ar	DF, Pr	2.1	10.3	0.2	99	19	0.1	1	<0.1	85.8	35
Ficopomatus enigmatus	A	SF	1.9	11.1	0.2	99.2	21	0.4	4.2	<0.1	85.9	29
Halicarcinus bedfordi	Ar	DF	1.4	4.6	0.1	99.3	22					
Hydrobia buccinoides	M	DF	1.2	4.4	0.1	99.4	23					
Tatea preseii	M	DF	1	3.5	0.1	99.4	24					
Telina deltoidalis	M	SF	0.9	3	0.1	99.5	25	4.2	10.9	0.3	86.2	15
Munna brevicornis	Ar	U	0.7	2.2	0.1	99.6	26					
Haliplanella luciae	Cn	SF	0.5	2.3	<0.1	99.6	27					
Erichthonius sp	Ar	DF	0.5	3.4	<0.1	99.7	27					

Species												
Cirriformia filigera	A	DF	0.4	3.3	<0.1	99.7	29					
Haustroriid sp.	Ar	DF	0.4	3.3	<0.1	99.7	29					
Spisula trigonella	M	DF, SF	0.3	1.8	<0.1	99.8	31	2.2	9.9	0.2	86.4	18
Tethygenia elanora	Ar	DF	0.3	3.1	<0.1	99.8	31					
Cruranthura simplicia	Ar	DF	0.2	1	<0.1	99.8	33	0.1	1	<0.1	86.4	35
Notoplana longicrumera	Ar	Pr	0.2	1	<0.1	99.8	33					
Exosphaeroma sp.	M	DF	0.2	1	<0.1	99.8	33					
Metapenaeus sp.	T	DF	0.2	1.5	<0.1	99.9	33					
Pilumnopeus serratifrons	Ar	DF	0.2	1.5	<0.1	99.9	33					
Nodilittorina unifasciata	Ar	U	0.2	2.1	<0.1	99.9	33					
Eubittium lawleyannum	M	DF	0.1	1	<0.1	99.9	39					
Paratanytarsus grimii	Ar	DF	0.1	1	<0.1	99.9	39					
Prionospio cirrifera	Ar	DF, SF	0.1	1	<0.1	99.9	39	20.7	55.1	1.7	88.1	11
Nassarius burchardi	Ar	SF	0.1	1	<0.1	99.9	39	6.9	19.3	0.6	88.7	14
Batillaria australis	M	DF	0.1	1	<0.1	99.9	39	1.5	5.9	0.1	88.8	20
Gastrosaccus sp.	A	DF, SF	0.1	1	<0.1	99.9	39	0.4	3.3	<0.1	88.8	29
Balanus amphitrite	Ar	U	0.1	1	<0.1	99.9	39					
Liloa brevis	Ar	SF, DF	0.1	1	<0.1	100	39					
Harmothoe waali	M	DF, SF	0.1	1	<0.1	100	39					
Caprella equilibra	T	DF, SF	0.1	1	<0.1	100	39					
Palaeomonetes australis	Ch	SF	0.1	1	<0.1	100	39					
Philine cf. angasi	A	DF	0.1	1	<0.1	100	39					
Pseudoceros reticulatus	M	Pr	0.1	1	<0.1	100	39					
Ciona sp.	M	DF	0.1	1	<0.1	100	39					
Pseudopolydora kempi	A	DF, SF						80.6	112.1	6.6	95.5	6
Oligochaete spp.	A	DF						32.9	72.3	2.7	98.2	10
Carazziella victoriensis	A	DF, SF						10.5	26.9	0.9	99	13
Sipunculan sp. 5	S	DF						3.9	9.1	0.3	99.4	16
Eusirid sp. 2	Ar	DF						2.3	22.9	0.2	99.6	17
Paranthurid sp. 2	Ar	DF						1.3	10.5	0.1	99.7	21
Armandia sp.	A	DF						0.9	9.4	0.1	99.7	23
Sabellid sp.	A	SF						0.7	3.4	0.1	99.8	24
Donax sp.	M	SF						0.6	3.9	0.1	99.8	25
Platyhelminthes sp.	P	DF						0.5	3.4	<0.1	99.9	27
Laturnula sp.	M	SF						0.3	1.8	<0.1	99.9	29
Fusinus sp.	M	Pr						0.3	1.8	<0.1	99.9	29
Heteromastus sp.	A	DF						0.2	1.5	<0.1	100	34
Nanereis sp.	A	DF						0.1	1	<0.1	100	35
Amphinomid sp.	A	DF						0.1	1	<0.1	100	35
Maldanis sp.	A	DF						0.1	1	<0.1	100	35
Orbiniella sp.	A	DF						0.1	1	<0.1	100	35

Nematode sp.	Ne	U			0.1	1	<0.1	100	35
Coleopteran sp.	Ar	DF			0.1	1	<0.1	100	35
Overall mean density			1233.6	712	1213.7	959.8			
Mean number of taxa			9.6	2.5	8.8	4.1			
Total number of individuals			11,838		11,718				
Total number of taxa			52		42				