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Biological Monitoring of the Marine Ocean Outfall at Black Rock, Victoria, Australia

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The Barwon Region Water Authority in Victoria, Australia, commissioned a new subtidal ocean sewage outfall in February 1989. This outfall replaced an old intertidal outfall, and presently discharges via a diffuser in an average depth of 15 m some 1.2 km offshore in Bass Strait. Examination of biological communities around the old outfall prior to the change revealed distinct differences in biotic assemblages close to the outfall. An ongoing monitoring programme, implemented in 1986, has assessed not only the recovery in this intertidal area, but also the major changes to macroalgal communities on surrounding reefs, subtidal infauna populations in sandy sediments, and rocky shore communities.

The Black Rock sewage outfall services the city of Geelong and surrounding coastal townships in central Victoria, Australia (Fig. 1). The effluent passing through the system originates from industry (13% by volume), domestic, commercial properties and permanent infiltration (87%). The total population of the region has increased from approximately 30 000 in 1910 to around 200 000 in 1990 (GDWB, 1991). Since 1916, sewerage services have been provided for a high percentage of the population in the region via an intertidal outfall which discharged up to 50 Ml d^{-1} of coarsely-screened sewage effluent (CCE, 1979). With increasing volume of discharge, ecological and other effects were reported from areas surrounding the outfall, until eventually it was shut down in 1989 and a new offshore ocean outfall was commissioned. The new outfall discharges an average 55 Ml d^{-1} of finely-screened sewage from a series of 27 diffuser ports which are located 4 m apart on alternate sides of a pipeline, situated, on average, at a depth of 15 m and approximately 1.2 km from shore.

The old intertidal outfall discharged at the low water mark in a then remote area, 14 km south of Geelong. Before discharge, the raw effluent passed through coarse screen comminutors and was pumped to elevated levels via Archimedes screw pumps during flooding tidal periods. The changes which have been effected in the new outfall include the passage of sewage through an onshore treatment plant, which improves the quality of the effluent by removing all floatable material, grit particles (>0.2 mm equivalent diameter) and visible floating oil and grease. The resultant effluent is buoyant, being more than 99% freshwater, and rises through the water column to the surface, becoming mixed with ambient water by turbulent entrainment.

The new subtidal outfall was designed to achieve a rapid initial mixing (>100 : 1; Ashton, 1991). During the early phase of the study reported here, a model was developed to predict effluent plume movement from the new subtidal outfall (see Ashton, 1986). This model defined a zone (zone 3: see Fig. 1(a)) in which dilution of the effluent field would not exceed 10 times that of the initial dilution for 90% of the time. It was believed that this zone would be the most likely area of adverse impact.

The aims of this study were to examine the effects of the submerged outfall on local subtidal assemblages, and the recovery from any impacts of the old shoreline outfall on local intertidal and near-shore subtidal assemblages. The study was divided into four sub-programmes on the basis of habitat types, including macroalgal communities on subtidal reefs; infauna community structure in subtidal sediments; macroalgal and macroinvertebrate communities on the intertidal rock platforms; and infaunal community structure in the intertidal sandy sediments on surrounding beaches.

Subtidal Reef Monitoring

A flexible underwater strategy for monitoring the macroalgal communities was adopted to allow for a variable number of quadrats to be photographed, depending upon the environmental conditions during the time of sampling. The photographic technique was designed to permit 'single image analysis' of divisional macroalgal coverage (i.e. red, green and brown algae) over the reef substrate. Consideration was given only to total macroalgal cover, irrespective of canopy type (i.e. turf, understorey or top). The monitoring strategy was designed to detect major changes, as monitoring subtle population changes would not be cost-effective.

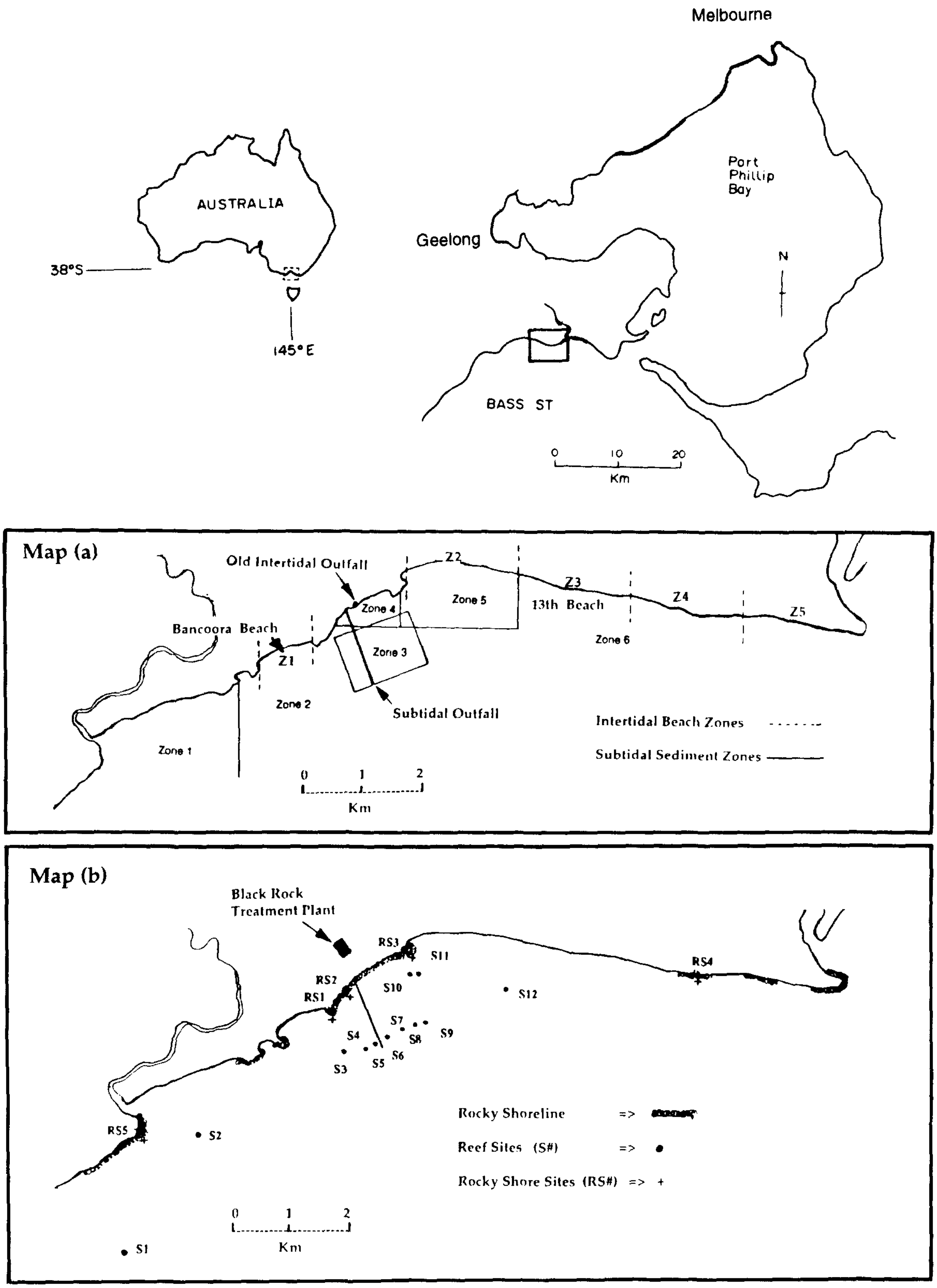

Fig. 1 Location of the study area on the shores of Bass Strait. Map (a) shows the subtidal soft-bottom sampling zones (Zones 1–6) and the intertidal soft-sediment sampling zones (Z1–Z5). Map (b) shows the subtidal reef monitoring sites (S1–S12) and the intertidal rocky shore sites (RS1–RS5).

Twelve sites were established (Fig. 1(b)), nine of which were located in 15–20 m depth of water. Two sites (S1 and S2) were established well away from the outfall area as controls. The locations of the sites were based directly upon possible impact zones predicted by a water quality model (Ashton, 1986). Following the commissioning of the new intertidal outfall, sites S10 and S11 were established to monitor the shallow waters near the old outfall. Site S12 was established in shallower waters outside the predicted area of influence.

Flat areas of reef were selected and locations fixed with a site marker. On each sampling occasion, transects were identified by a second marker 20 m away at a fixed compass bearing from the original site

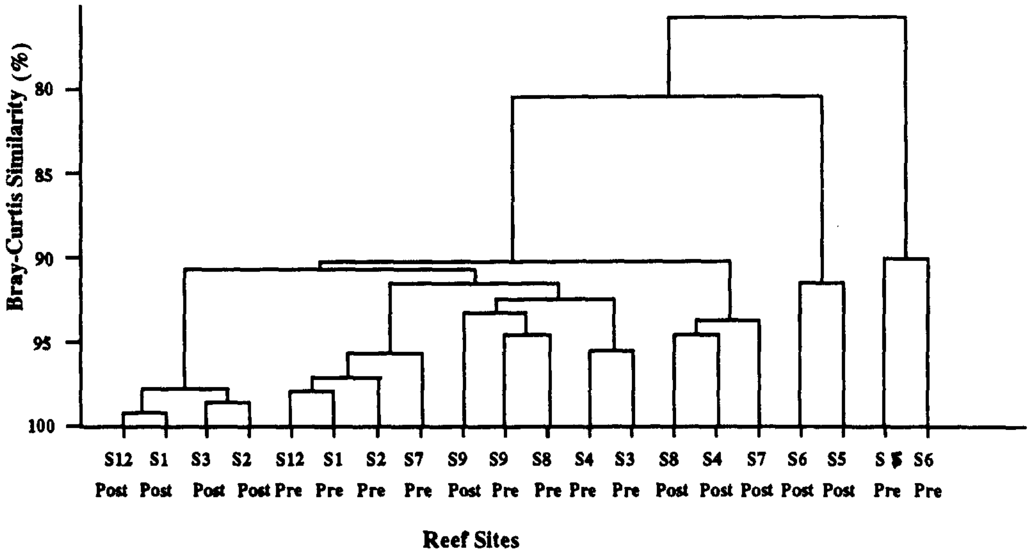

Fig. 2 Bray–Curtis similarity dendrogram of macroalgal coverage in the pre- and post-commissioning periods.

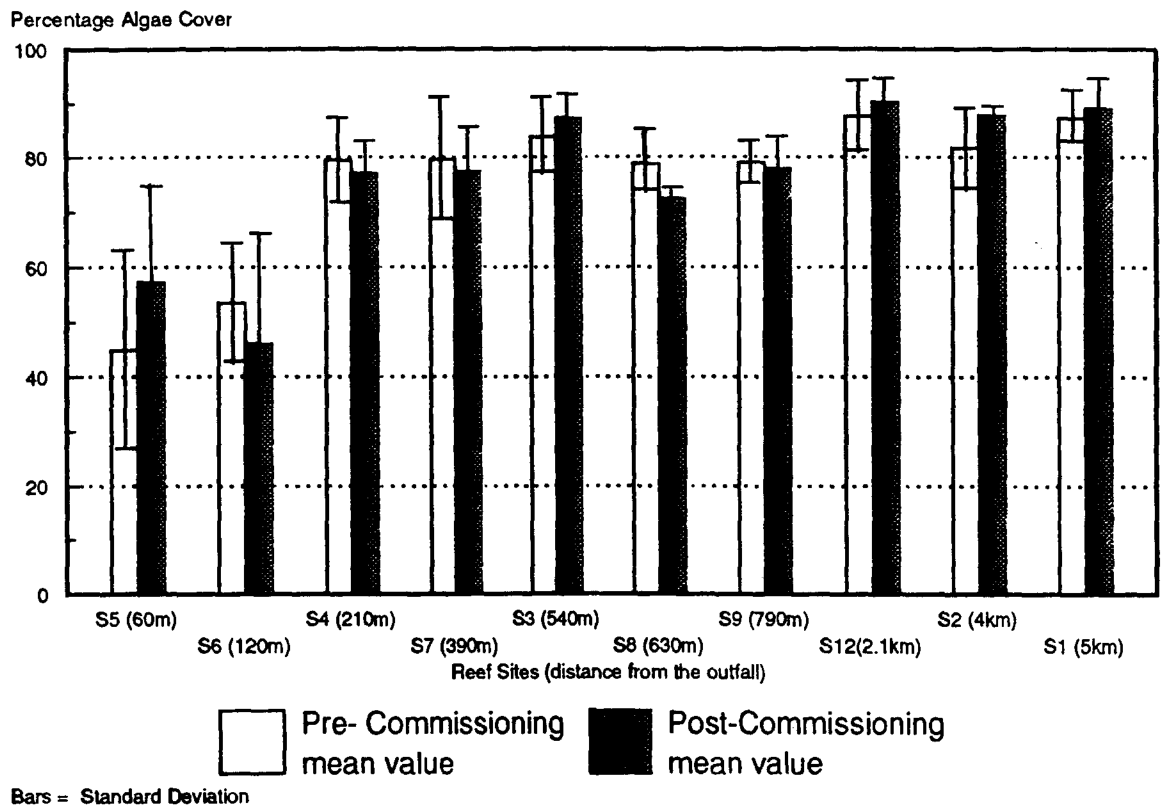

Fig. 3 Mean macroalgal coverage of sites during pre- and post-commissioning periods.

marker. To minimize diver bottom times, photography was carried out using a camera mounted on a grid (0.33 m^2 in area). A minimum of 15 photographs were taken at random along the transect, the exact number depending upon weather or sea conditions.

Raw data were group-averaged using mean percentage canopy cover and analysed by hierarchical clustering. Figure 2 illustrates a Bray–Curtis similarity dendrogram of macroalgal coverage data for both sampling periods (note that sites S10 and S11 are not included, as there were no pre-commissioning data for these sites). Macroalgal abundance was similar at all sites in both sampling periods, although the lower abundances recorded at the two sites nearest the outfall (S5 and S6) were attributed to construction activities in the pre-commissioning period and to sand accretion over the rocky substrate, driven by longshore sand shifts, in the post-commissioning period.

An assessment of reef macroalgae relative to distance from the outfall was made using the mean percentage macroalgal cover of each of the sites for the two monitoring periods. The results (Fig. 3) show that the two sites closest to the outfall (S5 and S6) had lower coverage for both monitoring periods. In addition, both sites showed more variability during the pre- and post-commissioning periods than the other sites. All other sites showed an average of 75–90% macroalgal reef coverage for both monitoring periods, irrespective of distance or direction from the outfall.

Infauna Communities in the Subtidal Sediments

The monitoring strategy used in this habitat was designed to allow random sampling on a seasonal basis. Twenty locations were randomly selected within a kilometre-wide transect in the study area, and five replicate samples were taken at each location. A purpose-built venturi sampler was designed to enable surface operation using small craft as a platform. The sampler covered an area of 0.05 m^2 (i.d. = 0.253 m) and sampled up to 0.20 m in depth, depending upon the compaction of the substrate.

The sampling sites were catalogued into zones corresponding to previously identified impacted areas and a single, predicted possible impact area, the latter based on the model produced during the study design (Ashton, 1986). The zones are shown in Fig. 1(a). Major animal groups (orders) were differentiated using multivariate techniques.

A comparison was made between population data of animal groups collected from all samples taken during the pre- and post-commissioning stages (Table 1). Slight differences in the percentages of the major groups of animals were found. The population of amphipods was reduced by 10% after the change of outfall. Isopods increased from 4.6 to 11%, and the cumacea decreased from 7.5 to 3.3%. The polychaetes increased from 7.4 to 10%. In a dynamic environment such as that at Black Rock, these differences are considered minor and are probably due to natural changes (e.g. see Ferraro *et al.*, 1991).

To further elucidate differences between the six defined zones, multivariate classification techniques were used and comparisons were based on the identity of the major groups of animals in the collections. The Bray–Curtis (B–C) similarity coefficient based on abundance of the dominant groups of animals was used to measure changes. The inter-group resemblance was defined as the mean of all resemblances between one group of animals to those of another. Such group-average clustering (Sneath & Sokal, 1973) has space conserving properties, which produce clusters with little distortion of the actual resemblance relationships (Boesch, 1977). The hierarchical clustering of the pre- and post-commissioning periods was undertaken separately to highlight individual zone differences between the two periods. Figure 4 indicates that changes have occurred since the shutdown of the old intertidal outfall. Whereas zone 5 was quite different in relation to other zones in the pre-commissioning phase, it was similar to zone 6 and closer in similarity to the other zones in the post-commissioning period. These changes are due to the lower number of polychaete species collected in zone 5 in the post-commissioning period.

Although elevated numbers of polychaetes were detected in zone 4 in the pre-commissioning period, the multivariate analysis techniques were unable to differentiate distinct differences between this zone and other unaffected zones, because of the presence of other dominant groups such as the amphipoda, isopoda and the cumacea. Nonetheless, the mean number of polychaetes was still higher in this zone than in the other zones in the pre-commissioning monitoring period, but was no different to the other zones in the post-commissioning period. This further strengthens the assumption that the area around the old intertidal outfall is no longer impacted, and that population numbers have returned to baseline levels. On examination of the mean numbers of polychaetes in zones 5 and 6, it is evident that they are above other zones; however, the numbers for zone 6 are similar in both periods. Therefore, it may be assumed that the soft-bottom sediments to the east of the outfalls are able to harbour higher populations of these animals. There were no

TABLE 1

Major animal group presence (descriptive statistical data only, expressed as percentage population abundance for animals > 1 mm < 10 mm in size) in soft-bottom sediments throughout the study area.

Animal group	Pre-commissioning	Post-commissioning
Amphipoda	78	68
Isopoda	4.6	11
Cumacea	7.5	3.3
Decapoda	0.9	2.7
Polychaeta	7.4	10
Others	1.6	4.0

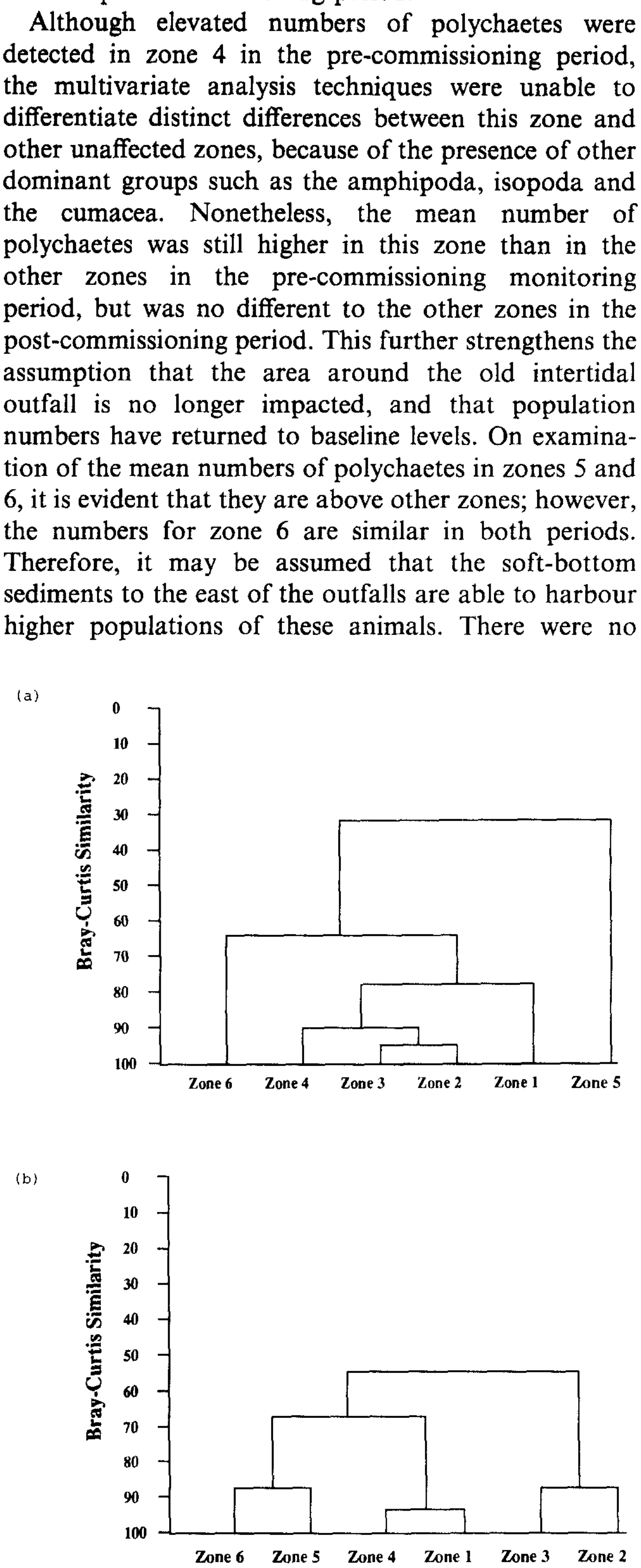

Fig. 4 Bray–Curtis similarity dendrogram of the major subtidal animal group populations between the zones in the (a) pre-commissioning period and (b) the post-commissioning period.

obvious changes to polychaete numbers in the zone nearest the subtidal outfall (zone 3), nor were any major changes detected in other major population groups in this zone.

Rocky Shores

Site selection was based upon the presence and location of permanently-exposed rocky shores in the area, and the distance of these sites from the intertidal and subtidal outfalls (Fig. 1(b)). Ten quadrats, each 250 × 400 mm (0.1 m^2), were permanently marked in the mid to lower eulittoral zone at each site using masonry nails. These quadrats were selected as representative of the exposed areas of the cobbled or flat platform sites. Non-destructive methods (Gonor & Kemp, 1978) were used to estimate percentage cover of macroalgae, mussel mats of *Xenostrobus pulex* and the sand mats of the spionid polychaete *Boccardia proboscidea* to the nearest 10%. Other groups of animals were counted individually.

The discharge of sewage effluents to intertidal areas has been associated with the stimulation of large crops of macroalgae, notably *Enteromorpha* spp. and *Ulva* spp. (Wilkinson, 1963; Sawyer, 1965; Portsmouth Polytechnic, 1976; Buttermore, 1977; Soulsby *et al.*, 1978; Montgomery & Soulsby, 1981). Although it was documented in the early studies (MSE, 1978a,b) that primary production of macroalgal species, such as *Enteromorpha* spp. and *Ulva* spp., was enhanced at sites closer to the intertidal outfall at Black Rock, there was some concern during the design stage of the programme in using these species to determine the recovery of rocky shore sites. These concerns were later justified, as certain sites well away from any area of impact of the outfall (e.g. RS4 and RS5) showed periods of stimulated growth of these species.

As the study required a monitoring programme that would address 'recovery' of a rocky shore site (RS3) following the shutdown of an intertidal outfall, special consideration was given to the opportunistic species *B. proboscidea* (see Dorsey, 1982; Synnott & Brown,

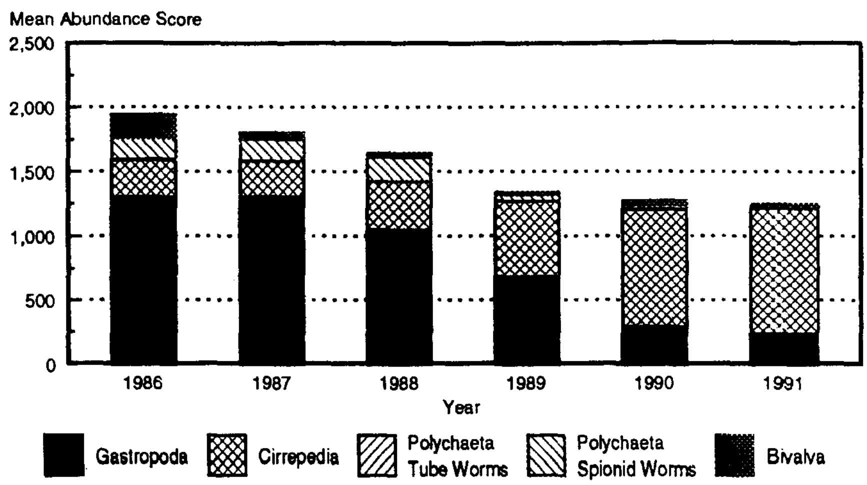

Fig. 5 Animal class abundance trends at rocky shore site RS3.

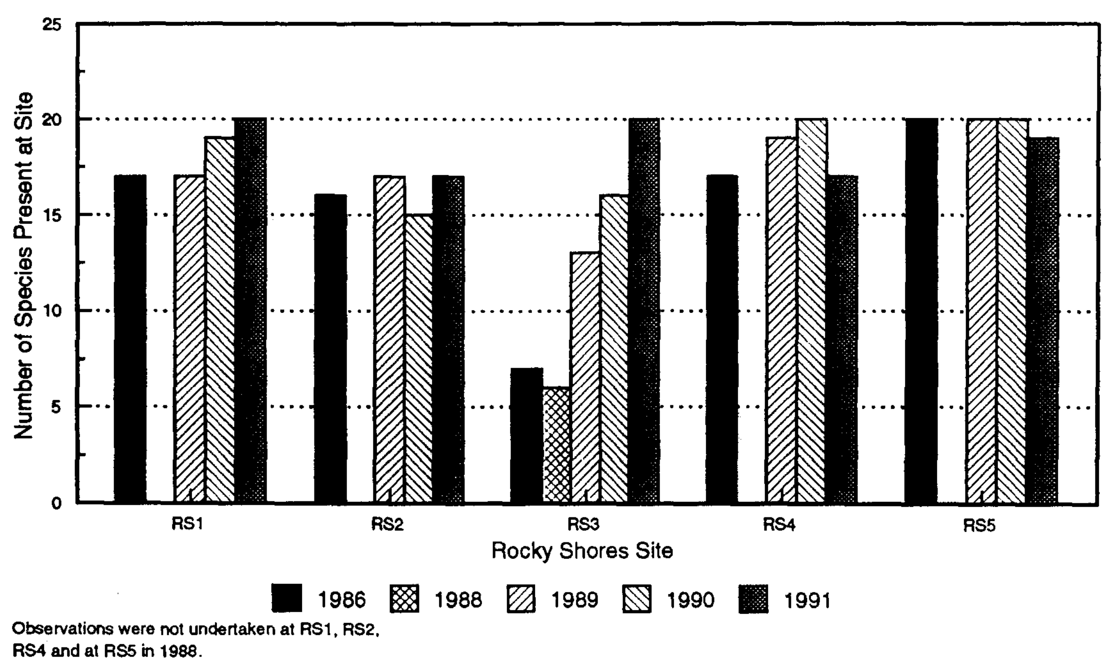

Fig. 6 Macroalgal species richness trends on the rocky shore sites.

1985). Monitoring continued at this site during both the pre- and post-commissioning stages, and all other organisms within the quadrat areas were documented. Distinguishing between natural variability of plants and animals and a 'recovery' response proved difficult using the quadrat method alone, even though the disappearance of *B. proboscidea* was evident (Fig. 5). However, the results of annual macroalgal surveys undertaken at each of the intertidal sites (Fig. 6) reinforced the notion that site RS3 was recovering slowly, with the gradual appearance of brown algae which had not been previously documented, yet which were common at other sites less than 1 km away. Nevertheless, it was obvious, even 3 years after the intertidal outfall shutdown, that the abundance of macroalgal species was still well below expected levels. Subsequent studies have shown that macroalgal abundance has increased to levels similar to that at site RS1 (Barwon Water, 1994), which adds strength to the conclusion that the RS3 site has recovered and now shows little evidence of past stress.

Infauna Communities in the Intertidal Sediments

The monitoring area of this habitat was divided into zones relevant to direction and distance away from the outfalls. Each zone was sampled every season, using a belt transect which ran from the previous high water mark to the low water mark. To monitor populations through the entire transect, stratified random samples were taken along the belt transect. Ten sample units were randomly selected within the entire transect, with a minimum of one sample unit being selected within each subsampling area.

Locations where access pathways to the shoreline existed were selected as sampling sites. One site per zone was randomly selected during each quarterly period. Bancoora Beach, to the west of the outfall, was called zone 1 (Z1); the 6.4 km of 13th Beach, to the east of the rocky outcrops, was divided into four zones (Z2–Z5; see Fig. 1(a)). A sampler (0.05 m^2) was hand driven into the sediment to a depth of 0.2 m and levered out with a spade. The sample was sieved *in situ* and animals retained on a 1.0 mm sieve were identified to the taxonomic level of 'order' and counted.

Geometric mean population data were assessed using multivariate cluster analysis to determine similarity between zone changes in the pre- and post-commissioning monitoring periods. Figure 7 shows hierarchical agglomerative clustering dendrograms calculated using the Bray–Curtis similarity coefficient. The data were transformed by converting major animal group scores into logarithms to reduce the discrepancy between large and small values.

The results show little difference in similarity coefficients; however, there is a slight change of hierarchical order with small changes occurring between Z1, Z2 and Z3. Nevertheless, the subtle changes detected in this analysis were not considered significant. In brief, analyses of the data generated from this part of the research revealed that there were no significant

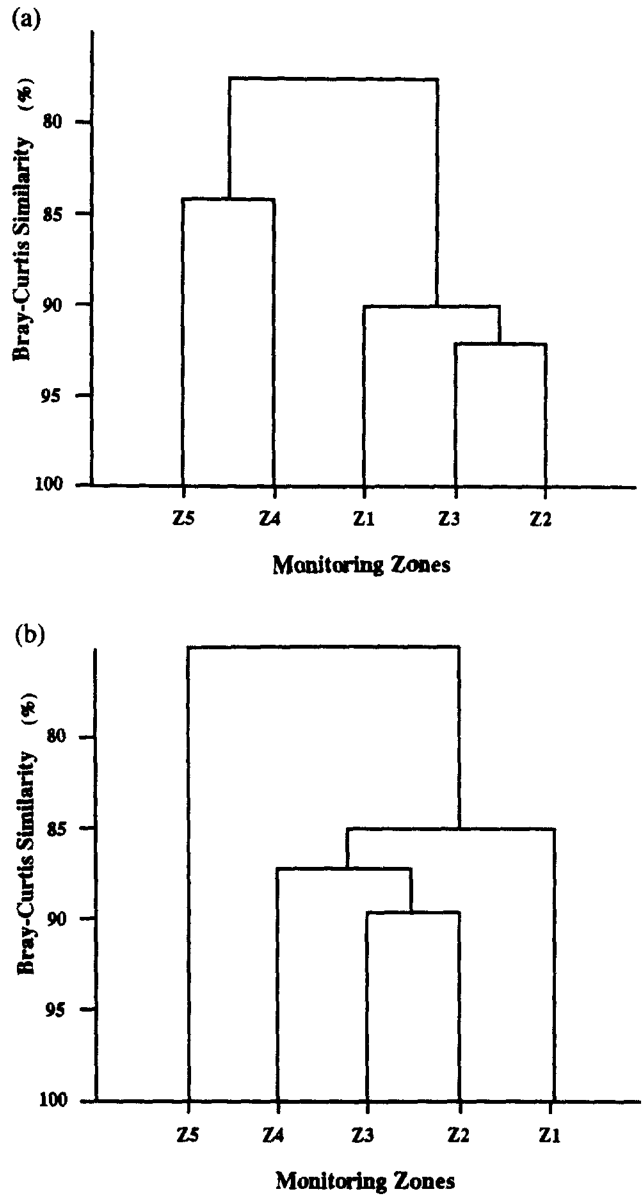

Fig. 7 Bray–Curtis similarity dendrogram of major intertidal animal group geometric mean populations in the (a) pre-commissioning period and (b) the post-commissioning period.

differences between zones which could be attributed to the old intertidal or new subtidal outfall. In addition, there were no significant differences detected in the data sets in each of the zones between the two monitoring periods.

Conclusions

The monitoring programme showed that macroalgal abundance has not been affected on either side of the new outfall, except for those sites directly affected by the construction of the pipeline and subsequent longshore sand shifts. Analysis of the infauna assemblages in areas around the subtidal outfall indicated no outfall-related impact on infaunal communities. Elevated numbers of polychaetes found to the east of the intertidal outfall may reside naturally in this area. Further detailed monitoring would be required to establish this; however, there is evidence that polychaete populations in areas very close to the intertidal outfall have decreased.

Monitoring of rocky shores has shown that the previously identified areas of impact near the old intertidal outfall have undergone significant change. Opportunistic species, once dominant in the intertidal areas, have disappeared. Species richness of macroalgae in the area has increased to levels similar to other sites. Analysis of the sandy beach data showed no statistical evidence to prove that either outfall impacted upon surrounding beaches. Community variability did occur, but was expected in this environment.

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