Accepted version of the ms published in Estuarine, Coastal and Shelf Science 67:708-720 (2006).

Effects of artificial openings of intermittently opening estuaries on macroinvertebrate assemblages of the entrance barrier

William Gladstone[1,*], Nicole Hacking[2, 3], Vanessa Owen[1]

1: Centre for Sustainable Use of Coasts and Catchments, School of Applied Sciences, University of Newcastle (Ourimbah Campus), PO Box 127, Ourimbah NSW 2258, Australia

2: Centre for Natural Resources, Department of Infrastructure, Planning and Natural Resources, PO Box 2185, Dangar NSW 2309, Australia

3: Current address: National Marine Science Centre, PO Box J321, Coffs Harbour, NSW 2450, Australia

* Corresponding author. E-mail address: William.Gladstone@newcastle.edu.au

Abstract

Intermittently opening estuaries are artificially opened to manage flood risk, water quality, recreational amenity, and fisheries; however, the ecological impacts of this management technique are incompletely understood. During 2001 and 2004, this study assessed the impacts of artificial openings on the macroinvertebrates of entrance barriers of intermittently opening estuaries in New South Wales (Australia). In 2001 macroinvertebrates were sampled once before artificial opening and 9 d and 25 d after re-formation of the entrance barrier. A multiple before-after-control-impact analysis found that, although entrance barriers were destroyed by the artificial openings and then re-formed naturally by wave action, significant interactions for taxonomic richness, density of the amphipod *Paracalliope australis* (Gammaridae) and density of the gastropod mollusc *Aschoris victoriae* (Hydrobiidae) meant that the effects of this disturbance could not be distinguished from the natural variations that occurred in unopened estuaries. Multivariate analyses found that assemblages at both opened and unopened estuaries changed from before to after the openings, and the magnitude of the dissimilarity between times varied between estuaries. In 2004, macroinvertebrates were sampled on 3 randomly selected days within each of 3 periods (before, 3 d and 42 d after) at 1 opened and 3 unopened estuaries. Asymmetrical analysis of this modified before-after-control-impact study found that the change in taxonomic richness at the opened estuary from before to after opening did not differ from temporal changes that occurred in unopened estuaries. Short-term variation (i.e. between days) in total density of macroinvertebrates and density of *P. australis* in the re-formed entrance barrier of the opened estuary also did not differ from the variation in the control estuaries. Additionally, assemblage structure was not significantly changed by the opening and assemblages at two control estuaries were also unchanged over the same time. Individual taxa and assemblages of macroinvertebrates in entrance barriers of these intermittently open estuarine systems appear to be resilient to the habitat disturbance caused by artificial openings.

Keywords: amphipod, biodiversity, coastal zone management, estuary, impact assessment, multivariate analyses

1. Introduction

Human uses of intermittently open estuaries, foreshore property developments, and community expectations have led management authorities in Australia (Roy et al., 2001), Brazil (Saad et al. 2002), South Africa (Bally, 1987), and the United States (Elwany et al., 2003) to intervene in the natural dynamics of these estuarine ecosystems. The entrance barriers of intermittently open estuaries are artificially breached to improve water quality, recreational amenity and fishing opportunities, and to prevent flooding of adjacent property (Kok & Whitfield, 1986; Potter, 1990; Kjerfve, 1994; Whitfield, 1999; Roy et al., 2001; Healthy Rivers Commission, 2002; Pinto & Teixeira, 2002; Saad et al. 2002; Suzuki et al. 2002; Elwany et al., 2003). Intermittently open estuaries are a significant component of the estuarine environments of many countries. They represent 49% of estuaries in south-east Australia (Roy et al., 2001); 70% of South African estuaries (Whitfield, 1992); and 18% of the North American coastline (Barnes, 1980). Globally, they represent 13% of the world's coastline (Barnes, 1980).

These normally small estuaries are isolated from the sea for extended periods of time because their small catchments provide only limited flows of fresh water. This, combined with a low tidal range, allows beach sand to accumulate at the entrance. The water level of intermittently open estuaries is naturally dynamic and responsive to catchment run-off, evaporation, and rainfall events. Entrance barriers that separate the closed estuary from the ocean are naturally breached during periods of elevated water levels or by high seas, causing the estuary to drain. These estuaries may remain open for hours to months, receiving incoming seawater and immigrating marine biota until the barrier is re-formed by wave action (Kjerfve & Magill, 1989; Elwany et al., 2003). Unpredictable rainfall means that the timing and frequency of natural openings are intermittent.

Artificial openings alter the natural cycles of flooding, drainage, and filling upon which the ecological processes of these ecosystems depend. The practise is regarded by one Australian state government as a threat to the biodiversity of these ecosystems (NSW EPA, 2000). Artificial openings lead to reduced water volume, increased salinity, sediment re-suspension, and a rapid exchange of the remaining

water body (Saad et al. 2002; Suzuki et al. 2002). Death of submerged macrophytes upon exposure to air increases dissolved and total nutrients and primary production shifts from being macrophyte- to phytoplankton-dominated (Knoopers, 1994; Suzuki et al. 2002). In other systems artificial openings have led to a short-term reduction in biomass of phytoplankton and zooplankton (Froneman, 2004). Although artificial openings may have no effect on richness or total abundance of fishes (Griffiths, 1999; Griffiths & West, 1999, but see Saad et al., 2002), community composition is altered by the immigration of recruits of marine-spawning fishes that are economically significant (Kok & Whitfield, 1986; Young et al., 1997; Griffiths, 1999; Saad et al. 2002). The latter effect may be considered beneficial for fisheries-based uses of these estuaries (Griffiths, 1999; Saad et al. 2002); however, the impacts of repeated artificial openings are unknown. Management of intermittently open estuaries is a complex issue that requires the full range of impacts from management actions to be understood.

Entrance barriers, which are composed of marine sand, are a feature of the habitat diversity of intermittently open estuaries. Moreover, the macroinvertebrate fauna of entrance barriers is likely to be important in the ecological processes of these estuaries. For example, it is known that macrofaunal crustaceans are the dominant component in the diet of estuarine fishes with almost all production of crustaceans larger than 1 mm consumed (Edgar & Shaw, 1995). Distribution and abundance of water birds in estuaries is influenced by the availability of their macroinvertebrate prey (Beukema et al., 1993; Moreira, 1994; Zharikov & Skilleter, 2004). Lower in the trophic web, fragmentation of dead seagrass leaves by feeding isopods and amphipods maintains energy flow and nutrient cycling (Robertson & Mann, 1980). This study assessed the impacts of artificial openings of intermittently opening estuaries on the macroinvertebrate fauna of the entrance barrier. The following hypotheses were tested: (1) taxonomic richness and density of individual species were significantly reduced in the re-formed entrance barrier of artificially opened estuaries; and (2) the assemblage structure of macroinvertebrates was significantly different in the re-formed entrance barriers of opened estuaries.

2. Methods and materials

2.1. Study sites

This study was undertaken at four intermittently open estuaries in New South Wales, Australia: Wamberal, Terrigal, Avoca, and Cockrone (Fig. 1 and Table 1). The entrance barrier of each estuary is approximately 75 m wide. Artificial openings begin as a 1 m wide trench dug in the centre of the barrier from the estuary to the sea that rapidly expands, with the force of water leaving the lagoon, and carries away all sand to a depth of about 1.5 m from a 20 m wide section at the centre of the barrier. Closure of these estuaries occurred through the transport of sand by waves into the opening from the beach (Roy et al., 2001). Estuaries were classified as 'closed' when the sand profile on the estuary side of the barrier where the opening had occurred was the same as the surrounding parts of the barrier that were physically undisturbed by the opening and when waves breaking at the beach no longer swept over the barrier into the estuary. The period between opening and closure took 15-20 d in the estuaries studied. Two opening events were studied: (1) October – November 2001 when Wamberal and Terrigal were artificially opened at the same time, and (2) January – March 2004 when Terrigal only was artificially opened.

2.2. Field sampling

Macroinvertebrates were sampled in an area of 20 x 3 m in the centre of the entrance barrier that was approximately 5 m from the shoreline in water 10-15 cm deep. Samples were collected by inserting a 15 cm wide x 20 cm deep PVC corer (N = 10) into the sediment and then sieving the sediment sample through a 1 mm mesh. A pilot study comparing 3 diameters of corer (10, 15, 24 cm), 3 depths of sample collection (10, 20, 30 cm) and increasing numbers of samples (3-10 replicates) found that the selected sampling strategy provided means with acceptable precision (i.e. standard error/mean) for species richness (precision=0.12), total density of macroinvertrebrates (precision=0.15), and density of the polychaete *Leitoscoloplos bifurcatus* (precision=0.14). The animals retained were collected, preserved in 5% formalin in seawater, and returned to the lab for identification and counting. Sediment grain size composition, salinity, temperature, and pH were determined at the same

time as macroinvertebrate samples were collected. Sediments were sampled (n = 6) with 5 cm wide x 20 cm deep cores, oven-dried at 60^0 C and separated into ≥ 1 mm, 0.5 mm, 212 μm, 63 μm, and < 63 μm fractions.

2.3. Sampling design and statistical analyses

Two designs were used because of differences in the number of opened and control estuaries in 2001 and 2004. The 2001 openings were a form of MBACI (Multiple Before-After-Control-Impact) design in which changes associated with an impact were tested by comparing two impacted and two control locations at multiple times before and after the impact (Keough & Mapstone, 1995). Wamberal and Terrigal were both opened in October 2001. Avoca and Cockrone were unopened and therefore served as multiple controls. The entrance barriers of control estuaries were similar to opened estuaries in their width, distance from the ocean, grain size composition, and depth of water coverage. All estuaries were sampled at the same time before Wamberal and Terrigal were opened (hereafter called the 'Before' period), and all were sampled 9 d (the 'After 1' period) and 25 d (the 'After 2' period) after the entrance barriers to Wamberal and Terrigal had re-formed. The sampling intervals reflected the rapidity with which the entrance barriers were re-established and knowledge of the potential speed of migration through sand by some macroinvertebrate fauna (Lawrie & Raffaelli, 1998; Ford et al., 1999; Norderhaug et al., 2002; Lewis et al. 2003). Actual openings times were unpredictable and so it was not possible to undertake more than one sampling before the openings occurred. The sampling design was therefore unbalanced in time and so two sets of analyses were undertaken: Before vs. After 1 and Before vs. After 2. The hypothesis was tested by three-factor analysis of variance (ANOVA) with the following factors: (1) Period: fixed, orthogonal, 2 levels (Before, After 1 or After 2); (2) Treatment: fixed, orthogonal, 2 levels (opened, control); (3) Estuary: random and nested in Treatment with 2 levels (Wamberal and Terrigal in the Open Treatment, and Avoca and Cockrone in the Control Treatment). An effect of the openings would be indicated by a significant Period x Treatment interaction. Significant interactions and differences between main effects were investigated by Student-Newman-Keuls procedure (Sokal & Rohlf, 1995). Where significant interactions occurred the *F*-ratios for main effects

are not presented (Underwood, 1981). Cochran's test was used to test for homogeneity of variances prior to ANOVA and data sets with heterogeneous variances were transformed where possible (Underwood, 1997). When transformation was unsuccessful the analysis was done on untransformed data because ANOVA is robust to departures from this assumption for the sample sizes used in this study (Underwood, 1997). Analyses were undertaken using GMAV5 (Institute of Marine Ecology, University of Sydney).

The hypothesis that macroinvertebrate assemblages of re-formed entrance barriers were different to the assemblages present before the opening was tested by multivariate analyses. Non-metric multidimensional scaling (nMDS) based on a Bray-Curtis similarity matrix of raw data was used to produce two-dimensional ordination plots of assemblages using PRIMER 5 software (Primer-E Ltd, Plymouth) (Clarke, 1993; Clarke & Warwick, 2001). Raw data was used, in preference to transformation, because of the low abundances of most taxa. The significance of changes in assemblage structure was tested by three-factor non-parametric multivariate analysis of variance (Anderson, 2001) based on raw data. The BIOENV routine in PRIMER was used to correlate changes in the biotic assemblages with changes (over the same time) in one or more of the abiotic variables in the multivariate environmental data (Clarke & Ainsworth, 1993). BIOENV determines the combination of abiotic variables (from the similarity matrix of the abiotic data) with the largest correlation with the Bray Curtis similarity matrix of the biotic data. The abiotic similarity matrix used normalized Euclidean distance as the distance measure and data were log transformed prior to analysis.

The sampling design was modified for the 2004 opening to include a test for impacts of openings at a smaller temporal scale than investigated in the 2001 openings because no changes were detected from before to after openings (see Results). In this year, Terrigal was artificially opened in January and remained opened for 11 d. Sampling occurred on 3 randomly selected days (separated by 2-3 d) within each of 3 ten-day periods: one period before the opening of Terrigal and two periods after its entrance barrier had re-formed. At all estuaries, 'After 1' sampling began 3 d after the entrance barrier had re-formed at Terrigal, and 'After 2' sampling began 45 d later. Asymmetrical ANOVA was used because there was only one opened estuary and

three control estuaries. The components of the asymmetrical ANOVA were constructed from the logic in Underwood (1992, 1993) by repartitioning the sums of squares from four separate orthogonal ANOVAs: (1) all data; (2) control estuaries only; (3) all estuaries in the Before period; and (4) control estuaries only in the Before period. Period was regarded as a fixed factor, Day as a random factor nested within Period, and Estuary as a random factor. The sequence of tests used to test for an impact does not rely on *F*-ratios derived from Mean Square estimates (Underwood 1981), and follows the sequence of tests recommended by Underwood (1992, 1993).

The significance of changes in assemblage structure between periods was tested by non-parametric multivariate analysis of variance (Anderson 2001). Analyses using untransformed data were done separately for Before vs. each After period for each estuary because of the lack of asymmetrical analyses suitable for use with assemblage data.

3. Results

3.1. 2001 opening

Eleven taxa from 4,899 individuals were identified, including amphipods (2 species); isopods (2 species); polychaetes (2 species); gastropod molluscs (2 species); and bivalve molluscs (3 species). Taxonomic richness of entrance barriers was unaffected by artificial openings (Table 2). The significant Period x Estuary (Treatment) interaction occurred because there was no change in the taxonomic richness of Wamberal and a significant decrease in taxonomic richness at Terrigal (the two opened estuaries), and taxonomic richness increased at Avoca and decreased at Cockrone (the controls) over the same time. In the After 2 period taxonomic richness of all estuaries did not differ significantly from the Before period. The significant Estuary (Treatment) effect occurred because taxonomic richness differed between the two opened estuaries.

The amphipod *Paracalliope australis* (Gammaridae) was the most abundant taxon, representing 92.5% of all macroinvertebrates collected. Density of *P. australis* in both the After 1 and After 2 periods was greater than or equal to the Before period

for all estuaries, although the magnitude of the increase differed between estuaries (Fig. 2b, Table 2). In particular, there was no change in the density of *P. australis* in the re-formed entrance barrier of Wamberal and significant increases at Terrigal and the two control estuaries. In the After 2 period, density of *P. australis* was also significantly increased at Wamberal. The pattern of change in density of the gastropod mollusc *Aschoris victoriae* (Hydrobiidae) from the Before to After 1 period also differed between estuaries (Fig. 2c, Table 2). Density of *A. victoriae* was significantly greater in the re-formed entrance barrier in the After 1 and After 2 periods at Wamberal but not at Terrigal. Density did not change at Avoca but significantly declined at Cockrone.

Some other taxa were sampled infrequently over the study period. The polychaete *Leitoscoloplos bifurcatus* (Orbiniidae) occurred at low densities at different times in all estuaries, with the greatest density occurring in the After 1 period at Avoca. The polychaete *Simplisetia aequisetus* (Nereididae) was only sampled once at each of the opened estuaries, in the Before and After 1 periods at Avoca, and in all periods at Cockrone. The isopod *Pseudolana concinna* (Cirolanidae) was sampled only during the Before period at both opened estuaries, at all periods at Cockrone and was not sampled at Avoca.

nMDS ordinations suggest assemblage structure changed in different ways at each estuary. At Wamberal the replicates from each period were not clearly separated (Fig. 3a). Overlap between samples in the nMDS ordinations indicates there was similarity in assemblage structure between the Before and After 1 periods and between the After 1 and After 2 periods. The nMDS ordination for Terrigal shows that samples in both After periods were separated from the Before period (Fig. 3b). There was considerable overlap between samples from the After 1 and After 2 periods. The spread of samples in the nMDS ordination for Avoca indicates a gradual change in assemblage structure between periods (Fig. 3c). Overlap of some samples occurred from the Before and After 1 periods and from the After 1 and After 2 periods. The majority of replicates from the Before period were clearly separated from both After periods at Cockrone (Fig. 3d). There was considerable overlap of samples between the After 1 and After 2 periods.

Non-parametric multivariate analysis of variance of changes in assemblage structure between periods showed a significant Period x Estuary (Treatment) interaction (Table 3a). Post-hoc examination of individual factors was unable to indicate the cause of the interaction and showed that assemblages at all estuaries differed significantly between the Before and both After periods. The greatest change in assemblage structure between the Before and After 1 periods occurred at one of the open estuaries (Terrigal) and the greatest changes in assemblage structure between the Before and After 2 periods occurred at one of the control estuaries (Avoca) (Table 3b).

Changes in assemblage structure were correlated with changes in one or more of the measured environmental variables (Table 4). Nevertheless, the magnitude of the correlation coefficients shows that the selected variables had only moderate explanatory power in Terrigal ($\rho = 0.66$) and Avoca ($\rho = 0.62$) and little explanatory power in the other estuaries. Salinity was selected in the combinations of variables in both Terrigal (one of the opened estuaries) and Avoca (an unopened estuary); however, environmental variables that maximized the correlation with the biotic data differed between estuaries and no single sediment fraction or other variable was consistently selected.

3.2. 2004 opening

Fifteen taxa from 2,865 individuals were identified, including amphipods (1 species); isopods (2 taxa); polychaetes (2 species); gastropod molluscs (5 species); and bivalve molluscs (5 species). Taxonomic richness varied from 0 to 9 taxa per replicate and the greatest taxonomic richness in a single sample occurred at Avoca in the After 1 period (Fig. 4a). Controls did not differ in their short-term variation (i.e. between days) in taxonomic richness in the After periods and there was no short-term variation in the difference between Terrigal and the opened estuaries in the After periods (Table 5). The opening of Terrigal therefore had no impact on short-term trends in taxonomic richness. There was a significant change in the differences between control estuaries from Before to after the opening, as shown by the significant *F*-ratios for B x Controls/residual in Table 4. Average taxonomic richness

was greater at Wamberal in the Before period and greater at Cockrone in the After period (Fig. 4a). However, the interaction in the difference between Terrigal and the controls from before to after the opening did not differ from this same interaction in the controls, as shown by the non-significant *F*-ratio for B x Open / B x Controls in Table 4. In other words, the changes in taxonomic richness that occurred at Terrigal from before to after its opening were not different from the range of changes that occurred in the control estuaries over the same time periods.

Total density of macroinvertebrates varied from 0 to 71 individuals per sample. Control estuaries differed in their short-term variation (i.e. between days) after the opening, as shown by the significant *F*-ratios for D(After) x Control/residual (Table 5). Over the three sampling days within the After 1 period, total density of macroinvertebrates increased at Wamberal, increased greatly at Avoca, and decreased at Cockrone. Over the three sampling days within the After 2 period, total density of macroinvertebrates increased at Wamberal, decreased at Avoca, and increased then decreased at Cockrone (Fig. 4b). As for average number of taxa, the short-term variation in total macroinvertebrates that occurred in the re-formed entrance barrier at Terrigal did not differ significantly from the short-term variation that occurred in the entrance barriers at the control estuaries in either After period, as shown by the non-significant *F*-ratios in Table 3 for D(After) x Open / D(After) x Control (Table 5).

Density of the amphipod *Paracalliope australis* varied from 0 to 16 per sample and density was greatest at Wamberal for much of the study period (Fig. 4c). Control estuaries differed in their short-term variation (i.e. between days) after the opening, as shown by the significant *F*-ratios for D(After) x Control/residual (Table 5). In the After 1 period, density increased greatly at Wamberal and increased at Cockrone on the third day. In the After 2 period density declined at Avoca and there were few individuals recorded at Wamberal or Cockrone. The short-term variation in density of *P. australis* in the re-formed entrance barrier of Terrigal in the After periods was not significantly different from the short-term variation that occurred in the entrance barriers at the controls, as shown by the non-significant *F*-ratios in Table 5 for D(After) x Open / D(After) x Control.

A number of other taxa were sampled infrequently over the study period and were therefore not able to be analyzed by parametric tests. Density of the gastropod mollusc *Aschoris victoriae* varied from 0 to 57 per sample and average density increased in Avoca and Cockrone in the After 2 period; it was not sampled at Terrigal. The bivalve mollusc *Donax deltoides* (Donacidae) was sampled at Avoca on every day, occurred only in 1 sample at Wamberal and was absent from the other estuaries. The polychaete *Leitoscoloplos bifurcatus* (Orbiniidae) occurred only at Avoca and its density declined considerably in the After 2 period. The polychaete *Simplesetia aequisetis* (Nereididae) was always sampled at Avoca and a short-term increase in density occurred over 2 d in the After 1 period. *S. aequisetis* was sampled in low numbers at Wamberal and Terrigal and was not sampled at Cockrone. The gastropod mollusc *Tatea* sp. (Hydrobiidae) was sampled only in low numbers at different times at Avoca and Cockrone. The bivalve mollusc *Xenostrobus securis* (Mytilidae) was always sampled at Avoca and exhibited considerable short-term changes in density, and was only sampled at Wamberal in the After 2 period.

Assemblage structure in the re-formed entrance barrier at Terrigal differed from the assemblage present before it was opened (Fig. 5a); however, the differences between the Before period and each of the After periods were not significant (Table 6). The assemblage varied significantly between days before the opening. Assemblage structure at Wamberal consisted of 2 groups: Before and After 1, and After 2 (Fig. 5b). Assemblage structure did not change significantly between the Before and After 1 periods, although there was a significant difference between days in the After 1 period (Table 6). Assemblage structure changed significantly from the Before to After 2 period and also between days in the After 2 period. Assemblage structure at Avoca formed three groups corresponding to the three time periods. Assemblage structure did not change significantly between the Before and After 1 periods; however, there was a significant difference between days in both the Before and After 1 periods (Table 6). Assemblage structure changed significantly from the Before to After 2 period (Fig. 5c) and there were significant differences between days in the After 2 period. Assemblage structure at Cockrone changed significantly between the Before and After 1 periods and there was significant differences between days in each period. Assemblage structure also changed significantly between the Before and After 2

periods and assemblage structure differed significantly between days in the After 2 period (Table 6, Fig. 5d).

One variable was selected to maximize the correlation with the biotic changes at Terrigal (salinity); however the correlation coefficient had limited explanatory power (ρ = 0.40) (Table 4). Changes in assemblage structure at Wamberal were correlated with changes in % sediment in the 1 mm and 63 μm fractions and salinity (ρ = 0.24) and temperature at Avoca (ρ = 0.82), both unopened estuaries. Changes in salinity and % sediment in the 0.5 mm fraction at Cockrone correlated with the changes in assemblage structure (ρ = 0.38). Variables that correlated with changes in assemblage structure in 2004 differed from the variables correlated with changes in 2001 at all estuaries.

4. Discussion

The entrance barriers of the opened estuaries were destroyed by the artificial openings and subsequently re-established with sand deposited by wave action from the ocean. Contrary to the proposed hypothesis, no significant effect of either opening was detected for any of the variables or assemblages tested. The absence of any effect of the artificial openings is surprising, given that loss of the entrance barrier represented a temporary but substantial loss of habitat.

The results from both openings suggest that macroinvertebrates in the entrance barriers of these estuaries are resilient, in the short-term, to the disturbance caused by artificial openings. This study's approach involved the comparison of one or more opened estuaries with multiple control estuaries on a number of occasions. Stewart-Oaten & Bence (2001) provided a thoughtful and detailed analysis of this approach and criticized it for, amongst other things, the possibility of not meeting assumptions about the selection of random, representative control locations. Control locations in this study came from the population of estuaries on the coast of central New South Wales and appeared to be similar to the opened estuary in all aspects apart from the opening. Inclusion of additional estuaries from a large geographical area would likely be confounded by different environmental conditions and rendered them unsuitable as

controls. The similarity in outcomes of the two openings separated in time suggests that the results of this study are generally applicable.

In 2001 changes in the variables tested at opened estuaries (richness and density of macroinvertebrates and density of *Paracalliope australis* and *Aschoris victoriae*) and in entire assemblages could not be distinguished from the changes that occurred in the unopened estuaries over the same time. Lack of a significant Period x Treatment interaction term in any of the analyses suggests that the changes observed in the variables were unrelated to whether the estuary had been opened. A significant result for this interaction term would indicate that changes that occurred between Periods differed in the opened and unopened estuaries and would suggest that a significant impact resulted from the openings (Underwood, 1992). The first sampling after the entrance barrier was re-formed in 2001 was undertaken 9 d after its re-formation and so it could be possible that an immediate effect of the opening (i.e. within days of the recovery of the barrier) was not detected. Following, short-term variability was assessed in the 2004 opening to test for this possibility. However, the short-term variability in the re-formed entrance barrier of the opened estuary in both the After 1 and After 2 periods was within the range of variability that occurred among the controls and indicated no short-term impact of the artificial opening (Underwood 1992, 1993).

Faunal recovery in disturbed sedimentary habitats occurs in four stages: recruitment (involving adult colonization and juvenile settlement); establishment; succession; and dynamics (Bonsdorff, 1988). The rate of colonization and the colonizing species vary according to the timing of the disturbance (Hall & Frid, 1998) and the physical characteristics of the disturbed patch such as its position in an estuary (Zajac & Whitlatch, 1982), tidal flow (Hall & Frid, 1998) and size (Whitlatch et al., 1998). The conceptual models of Whitlatch et al. (1998) described the interactive effects of patch size and life-stage of potential colonists on the recover process. The lack of a significant change in any variables in the present study from before to after artificial openings of estuary entrance barriers could be due to rapid colonization of macroinvertebrates from surrounding portions of the sand bar that were not destroyed by the opening. Some macroinvertebrate species are highly mobile (Lawrie & Raffaelli, 1998; Ford et al., 1999; Norderhaug et al., 2002; Lewis et al. 2003) and

adult colonization rates can be rapid (Hall & Frid, 1998). Alternatively, the macroinvertebrates sampled could have migrated from the adjacent beach as sand was transported by wave action from the beach to close the entrance. Many taxa sampled in this study are represented in sandy beach fauna (Jones et al., 1991; Hacking, 1998). It is unlikely that the macroinvertebrates in the re-formed entrance barrier were remnants of the pre-existing barrier fauna because the artificial opening reduced the depth of sand at the opening by 1-1.5 m and pilot studies conducted at the outset of these studies found few macroinvertebrates below 20 cm sediment depth.

All estuaries used in this study were examples of the same habitat and all occurred in different catchments that were separated by 1-5 km. Estuaries were therefore independent of one another. The major difference between estuaries was whether they were opened or not. However, estuaries also differed in their dominant species and in their temporal patterns of variation. In 2001 the amphipod *Paracalliope australis* occurred in all estuaries and at Cockrone (a control) its mean density changed from 14.7 per replicate in the Before period to 40.8 in the After 2 period. Over the same time, its average density at Avoca (a control) changed from 10.9 per replicate to 203.9. Density of the gastropod mollusc *Aschoris victoriae* changed from 0 per replicate at Avoca in the Before period to 1.1 in the After 1 period, and at Cockrone its density declined from 3.8 per replicate to 0. These differences between control estuaries in the ways they varied through time raises two points relevant to impact assessment in these systems. First, it is not necessary for control estuaries to be identical to one another and to the disturbed estuary in all aspects, except the presence of the disturbance. In the absence of a disturbance the set of control estuaries should continue to show their average but variable behaviour. In this scenario a disturbance will cause an impact if it causes changes that are significantly greater than the average change occurring over the same time period at the controls (Underwood, 1992). Second, due to the large natural variations occurring in these estuaries a disturbance at one estuary will have to cause a very large change, relative to the natural changes, for a significant difference to be recorded as an impact (Underwood, 1992; Glasby, 1997). Otherwise, faunal changes at the disturbed estuary will be within the range of natural variations occurring at the controls and these estuaries will appear resilient to the sorts of disturbance assessed in this study.

Low abundance and temporal variability were features of many macroinvertebrates in this study and this has been reported previously in similar systems (Morrisey et al., 1992b). Temporal patchiness may be due to these species having relatively short generation times; however the sampling frequency was within the lifespan and reproductive periodicity of the organisms sampled. Amphipods breed on one to two occasions throughout the year, have life spans of 6 – 15 mo and are sexually mature after 1 mo (Beare & Moore, 1998; Thiel, 1998; Costa & Costa, 1999; Cunha et al., 2000; Pardal et al., 2000; Thiel, 2000; Yu et al., 2002). Temporal variation in assemblage structure occurred in 2001 and 2004 in both opened and control estuaries and is therefore likely to be a normal feature of these assemblages in this habitat. Temporal variability in macroinvertebrates is related to dispersal (Costello & Myers, 1996) and variations in pelagic productivity (Lehtonen & Andersin, 1998); temperature and salinity (Cunha et al., 2000); algal biomass (Costa & Costa, 1999); and day length (Beare & Moore, 1998). However, temporal changes driven by these factors are likely to occur over longer time scales than the temporal scales sampled in this study.

An alternative explanation for the observed temporal variability is that it represents instead small-scale spatial patchiness. Cores were positioned haphazardly across the entrance barrier for each sampling event, and were separated by 1-2 m. Small-scale patchiness in the distribution of macrobenthic organisms exists in other systems (Volckaert, 1987; Barry & Dayton, 1991; Thrush, 1991; Morrisey et al., 1992a; Kendall & Widdicombe, 1999) and scales of spatial variation differ between species (Morrisey et al., 1992a; Ysebaert & Herman, 2002). Small-scale heterogeneity in sediment type (Warwick & Davies, 1977), density of other biota (Thrush, 1986; Morrisey et al., 1992a; Osterling & Pihl, 2001), and species mobility (Lawrie & Raffaelli, 1998) may be responsible for this patchiness.

Sediment particle size, rather than salinity, is a significant determinant of the distribution and abundance of macrobenthos in intermittently open estuaries in South Africa at an estuary-wide scale (Teske & Wooldridge, 2003). At the smaller scale of the entrance barrier, variability in macroinvertebrate assemblages was related to several physical variables: sediment particle size, pH, temperature, and salinity. Rather than responding to the same feature (e.g. sediment particle size) different

species appear to be responding to different features that varied in different ways through time. Further, behavioural plasticity has been demonstrated for invertebrates living in physically dynamic habitats (Hazlett, 1988; Brown, 1996). This means that the response of a species to a condition at one time may not necessarily be adhered to at another. This adds an extra element of difficulty to our ability to understand the processes behind ecological patterns of even naturally intermittent entrance barriers.

Acknowledgements

This study was funded by NSW DLWC/DIPNR, Gosford City Council, and Brisbane Water and Gosford Lagoons Catchment Management Committee. Laboratory support was provided by the School of Applied Sciences, University of Newcastle. Thanks to P Matheson and K O'Neill for coordinating and undertaking parts of the field and lab work. Fieldwork was greatly assisted by the efforts of many volunteers: K Byfield, N Donnelly, J Evans, V Hannan, M Harrison, S Jordan, K Mackrell, W Makepeace, S Mills, M Smith, and R Stimpson. Thanks to A Jones, M Schreider and two anonymous reviewers for valuable comments on an earlier version of this manuscript. This is a contribution from the Centre for Sustainable Use of Coasts and Catchments.

References

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

Bally, R., 1987. Conservation problems and management options in estuaries: the Bot River estuary, South Africa, as a case-history for management of closed estuaries. Environmental Conservation 14, 45-51.

Barnes, R.S.K., 1980. Coastal Lagoons. The Natural History of a Neglected Habitat. Cambridge University Press, Cambridge, 106 pp.

Barry, J.P., & Dayton, P.K., 1991. Physical heterogeneity and the organization of marine communities. In: Kolasa, J., Pickett, S.T.A. (Eds.), Ecological Heterogeneity. Springer-Verlag, New York, pp. 270-320.

Beare, D.J., Moore, P.G., 1998. Aspects of the life histories of *Perioculodes longimanus*, *Pontocrates arcticus* and *Synchelidium maculatum* (Crustacea, Amphipoda) at Millport, Scotland. Journal of the Marine Biological Association of the UK 78, 193-209.

Beukema, J.J., Essink, K., Michaelis, H., Zwarts, L., 1993. Year-to-year variability in the biomass of macrobenthic animals on tidal flats of the Wadden Sea: how predictable is this food source for birds? Netherlands Journal of Sea Research 31, 319-330.

Bonsdorff, E., 1988. Infaunal colonization and its dependence on environmental variation–experimental information from the Northern Baltic Sea. In: Ryland, J.S., Tyler, P.A., (Eds.), Reproduction, Genetics and Distribution of Marine Organisms. Olsen and Olsen, Denmark, pp. 349-356.

Brown, A.C., 1996. Behavioural plasticity as a key factor in the survival and evolution of the macrofauna of exposed sandy beaches. Revista Chilena de Historia Natural 69, 469-474.

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

Clarke, K.R., Ainsworth, M., 1993. A new method of linking multivariate community structure to environmental variables. Marine Ecology Progress Series 92, 205-219.

Clarke, K.R., Warwick, R.M., 2001. Change In Marine Communities: An Approach to Statistical Analysis and Interpretation, 2nd ed. Plymouth Marine Laboratory, Plymouth, 172 pp.

Costa, F.O., Costa, M.H., 1999. Life history of the amphipod *Gammarus locusta* in the Sado estuary (Portugal). Acta Oecologia – International Journal of Ecology 20, 305-314.

Costello, M.J., Myers, A.M., 1996. Turnover of transient species as a contributor to the richness of a stable amphipod (Crustacea) fauna in a sea inlet. Journal of Experimental Marine Biology and Ecology 202, 49-62.

Cunha, M.R., Sorbe, J.C., Moreira, M.H., 2000. The amphipod *Corophium multisetosum* (Corophiidae) in Ria de Aviero (NW Portugal). I. Life history and aspects of reproductive biology. Marine Biology 137, 637-650.

Edgar, G.J., Shaw, C., 1995. The production and trophic ecology of shallow-water fish assemblages in southern Australia. III. General relationships between sediments, seagrasses, invertebrates and fishes. Journal of Experimental Marine Biology and Ecology 194, 107-131.

Elwany, M.H.S., Flick, R.E., Hamilton, M.M., 2003. Effect of a small southern California lagoon entrance on adjacent beaches. Estuaries 26, 700-708.

Ford, R.B., Thrush, S.F., Probert, P.K., 1999. Macrobenthic colonisation of disturbances on an intertidal sandflat: the influence of season and buried algae. Marine Ecology Progress Series 191, 163-174.

Froneman, P.W., 2004. Zooplankton community structure and biomass in a southern African temporarily open/closed estuary. Estuarine, Coastal and Shelf Science 60, 125-132.

Glasby, T.M., 1997. Analysing data from post-impact studies using asymmetrical analyses of variance: a case study of epibiota on marinas. Australian Journal of Ecology 22, 448-459.

Gosford City Council. 1995. Coastal Lagoons Management Plan: Wamberal, Terrigal, Avoca and Cockrone. Gosford City Council, Gosford, 48 pp.

Griffiths, S.P., 1999. Consequences of artificially opening coastal lagoons on their fish assemblages. International Journal of Salt Lake Research 8, 307-327.

Griffiths, S.P., West, R.J., 1999. Preliminary assessment of shallow water fish in three intermittently open estuaries in south-eastern Australia. Fisheries Management and Ecology 6, 311-321.

Hacking, N., 1998. Macrofaunal community structure of beaches in northern New South Wales, Australia. Marine and Freshwater Research 49, 47-53.

Hall, J.A., Frid, C.L.J., 1998. Colonisation patterns of adult macrobenthos in a polluted North Sea estuary. Aquatic Ecology 33, 333-340.

Hazlett, B.A., 1988. Behavioural plasticity as an adaptation to a variable environment. In: Chelazzi, G. and Vannini, M. (Eds) Bahavioural Adaptation to Intertidal Life. Plenum Publishing Corporation. p317-332

Healthy Rivers Commission. 2002. Independent public inquiry into coastal lakes: final report. Healthy Rivers Commission of New South Wales, Sydney, 74 pp.

Jones, A.R., Murray, A., Marsh, R.E., 1991. Patterns of abundance of exoedicerotid amphipods on sandy beaches near Sydney, Australia. Hydrobiologia 223, 119-126.

Kendall, M.A., Widdicombe, S., 1999. Small scale patterns in the structure of macrofaunal assemblages of shallow soft sediments. Journal of Experimental Marine Biology and Ecology 237, 127-140.

Keough, M.J., & Mapstone, B.D., 1995. Protocols for designing marine ecological monitoring programs associated with BEK mills. National Pulp Mills Research Program Technical Report 11, Canberra, 185 pp.

Kjerfve, B., 1994. Coastal Lagoon Processes. Elsevier, Amsterdam, 577 pp.

Kjerfve, B., Magill, K.E., 1989. Geographic and hydrodynamic characteristics of shallow coastal lagoons. Marine Geology 88, 198-199.

Knoopers, B., 1994. Aquatic primary production in coastal lagoons. In: Kjerfve, B., (Ed.), Coastal Lagoon Processes. Elsevier, Amsterdam, pp. 243-286.

Kok, H.M., Whitfield, A.K., 1986. The influence of open and closed mouth phases on the marine fish fauna of the Swartvlei estuary. South African Journal of Zoology 21, 309-315.

Lawrie, S.M., Raffaelli, D.G., 1998. In situ swimming behaviour of the amphipod Corophium volutator (Pallas). Journal of Experimental Marine Biology and Ecology 224, 237-251.

Lehtonen, K.K., Andersin, A.B., 1998. Population dynamics, response to sedimentation and role in benthic metabolism of the amphipod *Monoporeia affinis* in an open-sea area of the northern Baltic Sea. Marine Ecology Progress Series 168, 71-85.

Lewis, L.J., Davenport, J., Kelly, T.C., 2003. A study of the impact of a pipeline construction on estuarine benthic invertebrate communities. Part 2. Recolonization by benthic invertebrates after 1 year and response of estuarine birds. Estuarine, Coastal and Shelf Science 57, 201-208.

Moreira, F., 1994. Diet, prey-size selection and intake rates of black-tailed godwits *Limosa limosa* feeding on mudflats. Ibis 136, 349-355.

Morrisey, D.J., Howitt, L., Underwood, A.J., Stark, J.S., 1992a. Spatial variation in soft-sediment benthos. Marine Ecology Progress Series 81, 197-204.

Morrisey, D.J., Underwood, A.J., Howitt, L., Stark, J.S., 1992b. Temporal variation in soft-sediment benthos. Journal of Experimental Marine Biology and Ecology 164, 233-245.

Norderhaug, K.M., Christie, H., Rinde, E., 2002. Colonisation of kelp imitations by epiphyte and holdfast fauna; a study of mobility patterns. Marine Biology 141, 965-973.

NSW Environment Protection Authority (NSW EPA). 2000. New South Wales State of the Environment 2000: Biodiversity Backgrounder. Environment Protection Authority, Sydney, 51 pp.

Osterling, M., Pihl, L., 2001. Effects of filamentous green algal mats on benthic macrofaunal functional feeding groups. Journal of Experimental Marine Biology and Ecology 263, 159-183.

Pardal, M.A., Marques, J.C., Metelo, I., Lillebo, A.I., Flindt, M.R., 2000. Impact of eutrophication on the life cycle, population dynamics and production of *Ampithoe valida* (Amphipoda) along an estuarine spatial gradient (Mondego estuary, Portugal). Marine Ecology Progress Series 196, 207-219.

Pinto, C.A., Teixeira, S.B., 2002. Morphodynamics of the sandy barrier of Salgados coastal lagoon, Armaçâo de Pêra Bay (Algarve-Portugal). In Eurocoast (Ed.), Littoral 2002, the Changing Coast. EUCC, Portugal, pp. 403-409.

Potter, I.C., Beckley, L.E., Whitfield, A.K., Lenanton, R.C.J., 1990. Comparisons between the roles played by estuaries in the life cycles of fishes in temperate Western Australia and southern Africa. Environmental Biology of Fishes 28, 143-178.

Robertson, A.I., Mann, K.H., 1980. The role of isopods and amphipods in the initial fragmentation of eelgrass detritus in Nova Scotia, Canada. Marine Biology 59, 63-69.

Roy, P.S., Williams, R.J., Jones, A.R., Yassini, I., Gibbs, P.J., Coates, B., West, R.J., Scanes, P.R., Hudson, J.P., Nichol, S., 2001. Structure and function of south-east Australian estuaries. Estuarine, Coastal and Shelf Science 53, 351-384.

Saad, A.M., Beaumond, A.C., Caramaschi, E.P., 2002. Effects of artificial canal openings on fish community structure of Imboassica coastal lagoon, Rio de Janeiro, Brazil. Journal of Coastal Research SI36, 634-639.

Sokal, R.R. Rohlf, F.J., 1995. Biometry 3rd ed. WH Freeman, New York, 887 pp.

Stewart-Oaten, A., Bence, J.R., 2001. Temporal and spatial variation in environmental impact assessment. Ecological Monographs 71, 305-339.

Suzuki, M.S., Figueiredo, R.O., Castro, S.C., Silva, C.F., Pereira, E.A., Silva, J.A., Aragon, G.T., 2002. Sand bar opening in a coastal lagoon (Iguipari) in the northern region of Rio de Janeiro state: hydrological and hydrochemical changes. Brazil Journal of Biology 62, 51-62.

Teske, P.R., Wooldridge, T.H., 2003. What limits the distribution of subtidal macrobenthos in permanently open and temporarily open/closed South African estuaries? Salinity vs. sediment particle size. Estuarine, Coastal and Shelf Science 57, 225-238.

Thiel, M., 1998. Reproductive biology of a deposit-feeding amphipod, *Casco bigelowi* with extended parental care. Marine Biology 132, 107-116.

Thiel, M., 2000. Population and reproductive biology of two sibling amphipod species from ascidians and sponges. Marine Biology 137, 661-674.

Thrush, S.F., 1986. The sublittoral macrobenthic community structure of an Irish sea-lough: effect of decomposing accumulations of seaweed. Journal of Experimental Marine Biology and Ecology 96, 199-212.

Thrush, S.F., 1991. Spatial patterns in soft-sediment communities. Trends in Ecology and Evolution 6, 75-79.

Underwood, A.J., 1981. Techniques of analysis of variance in experimental marine biology and ecology. Oceanography and Marine Biology Annual Review 19, 513-605.

Underwood, A.J., 1992. Beyond BACI: the detection of environmental impacts on populations in the real, but variable world. Journal of Experimental Marine Biology and Ecology 161, 145-178.

Underwood, A.J., 1993. The mechanics of spatially replicated sampling programmes to detect environmental impacts in a variable world. Australian Journal of Ecology 18, 99-116.

Underwood, A.J., 1997. Ecological Experiments: Their Logical Design and Interpretation Using Analysis of Variance. Cambridge University Press, Cambridge, 504 pp.

Volckaert, F., 1991. Spatial pattern of soft-bottom Polychaeta off Nova Scotia, Canada. Marine Biology 93, 627-639.

Warwick, R.M., Davies, J.R., 1977. The distribution of sublittoral macrofauna communities in the Bristol Channel in relation to the substrate. Estuarine, Coastal and Shelf Science 5, 267-288.

Whitfield, A.K., 1992. A characterization of southern African estuarine systems. South African Journal of Science 18, 89-103.

Whitfield, A.K., 1999. Ichthyofaunal assemblages in estuaries: a South African case study. Reviews in Fish Biology and Fisheries 9, 151-186.

Whitlatch, R.B., Lohrer, A.M., Thrush, S.F., Pridmore, R.D., Hewitt, J.E., Cummings, V.J., Zajac, R.N., 1998. Scale-dependent benthic recolonization dynamics: life stage-based dispersal and demographic consequences. Hydrobiologia 375/376, 217-226.

Ysebaert, T., Herman, P.M.J., 2002. Spatial and temporal variation in benthic macrofauna and relationships with environmental variables in an estuarine, intertidal soft-sediment environment. Marine Ecology Progress Series 244, 105-124.

Young, G.C., Potter, I.C., Hyndes, G.A., de Lestang, D., 1997. The ichthyofauna of an intermittently open estuary: implications of bar breaching and low salinities on faunal composition. Estuarine, Coastal and Shelf Science 45, 53-68.

Yu, O.H., Soh, H.Y., Suh, H.L., 2002. Life history and reproduction of *Synchelidium lenorostralum* (Amphipoda, Oedicerotidae) in a temperate sandy shore, southern Korea. Journal of Crustacean Biology 22, 126-134.

Zajac, R.N., Whitlatch, R.B., 1982. Responses of estuarine infauna to disturbance. I. Spatial and temporal variation of initial recolonization. Marine Ecology Progress Series 10, 1-14.

Zharikov, Y., Skilleter, G.A., 2004. A relationship between prey density and territory size in non-breeding Eastern Curlews *Numenius madagascariensis.* Ibis 146, 518-521.

FIGURE CAPTIONS

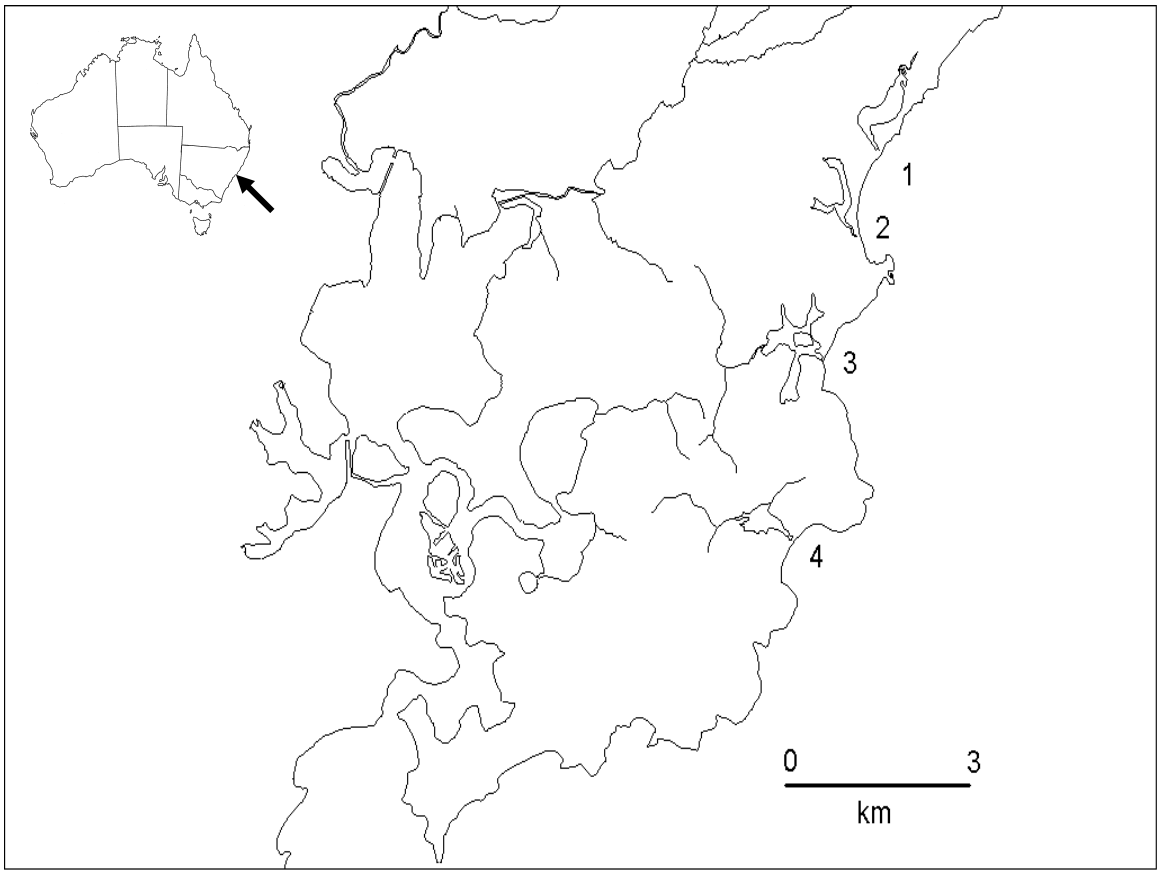
Fig. 1. Location of study estuaries: 1 Wamberal; 2 Terrigal; 3 Avoca; 4 Cockrone.

Fig. 2. Summary of results for (a) taxonomic richness of macroinvertebrates, (b) density of *Paracalliope australis*, and (c) density of *Aschoris victoriae* at two opened estuaries (♦) Terrigal and (●) Wamberal and at two controls (■) Avoca, and (O) Cockrone in 2001. Values shown are the mean of N = 10 replicate samples (± standard error) in three time periods: Before (B), After 1 (A1), and After 2 (A2).

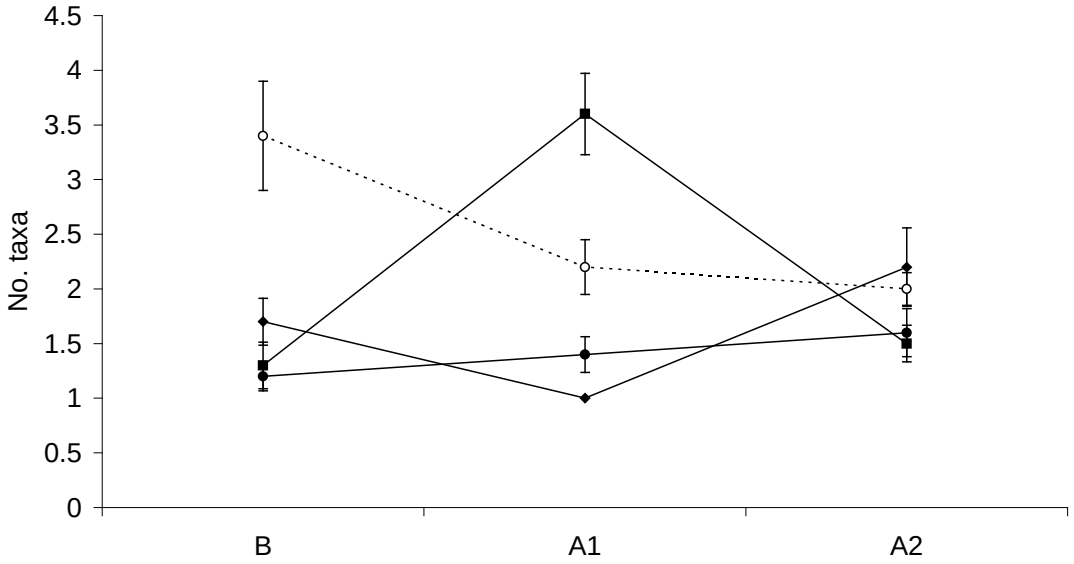
Fig. 3. nMDS ordinations of macroinvertebrate assemblage structure in the entrance barriers of opened (Wamberal, Terrigal) and control (Avoca, Cockrone) estuaries in three time periods: Before (●), After 1 (O), and After 2 (Δ) in 2001.

Fig. 4. Mean number of taxa, total density of macroinvertebrates and density of *Paracalliope australis* at (♦) Terrigal, (opened) (●) Wamberal, (■) Avoca, and (O) Cockrone. Values shown are the mean of N = 10 replicate samples (± standard error). Sampling was done on three days (B1, B2, B3) in one period before Terrigal opened, on three days in one period after the opening (A11, A12, A13) and on three days in a second period after the opening (A21, A22, A23).

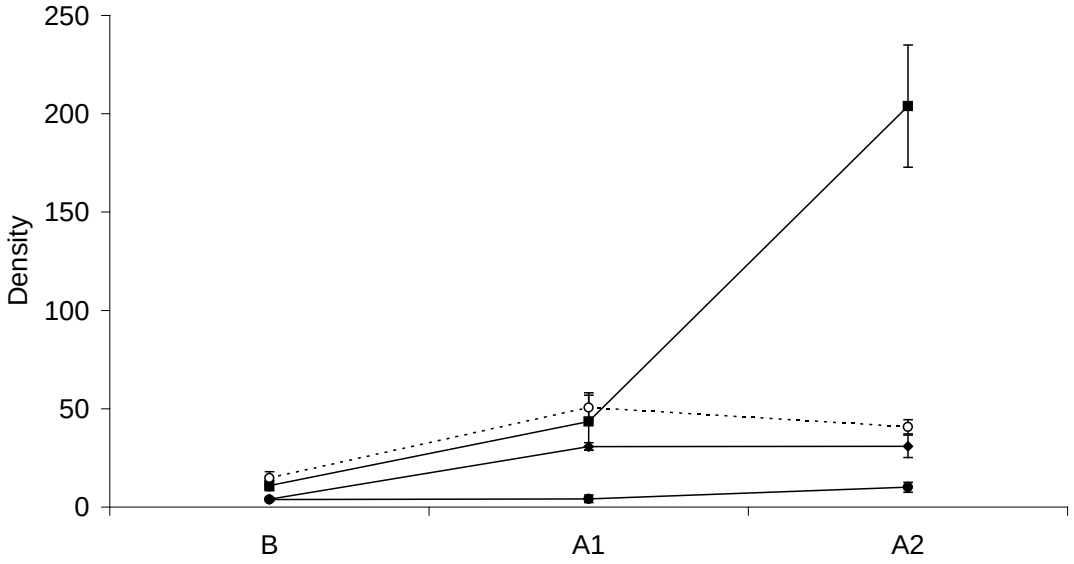
Fig. 5. nMDS ordinations of macroinvertebrate assemblage structure at each estuary based on mean abundances of each taxon on each day of sampling. Sampling was done on 3 d (B1, B2, B3) in one period before Terrigal opened, on 3 d in one period after the opening (A11, A12, A13) and on 3 d in a second period after the opening (A21, A22, A23).


1
2
3
4
0
3
km

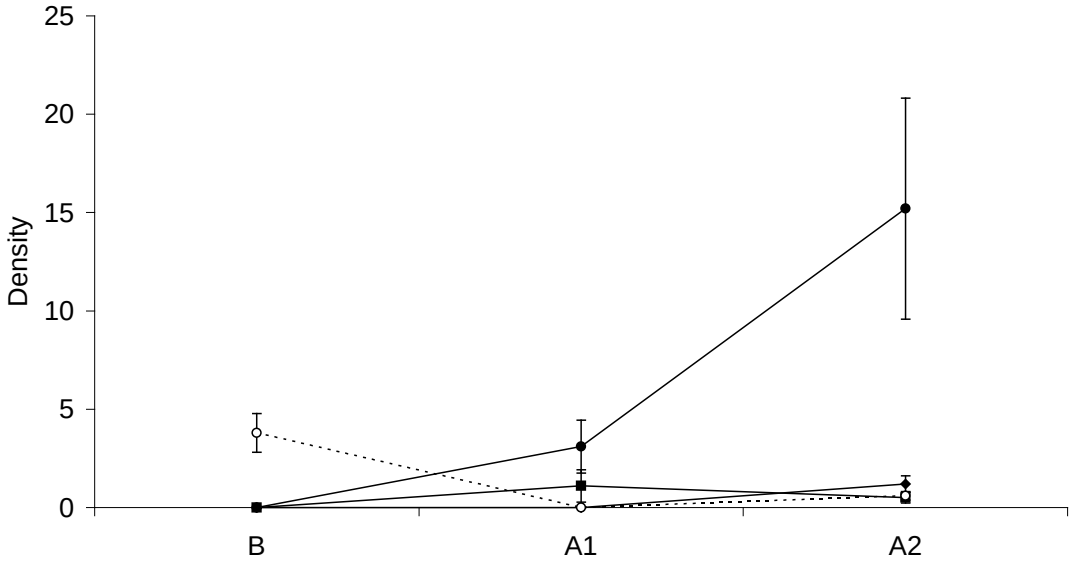
(a) Macroinvertebrate taxa



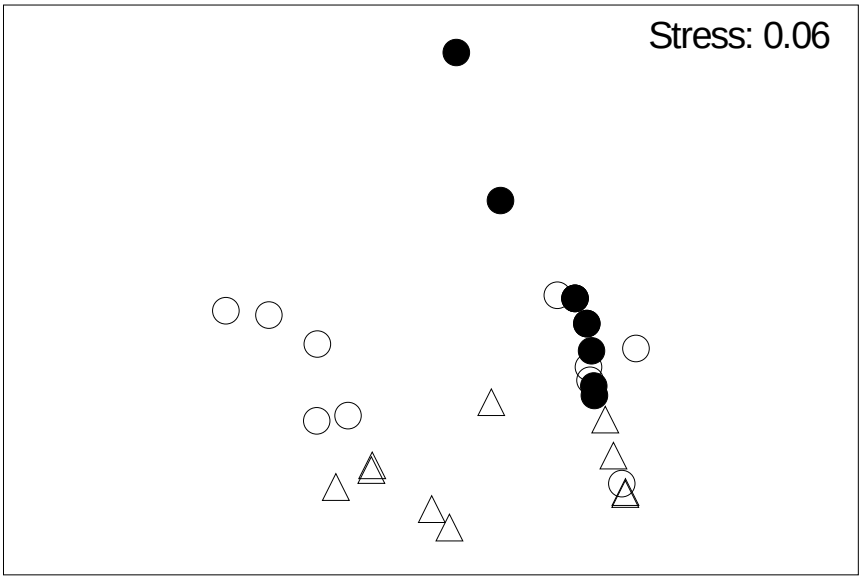
(b) *Paracalliope australis*



(c) *Aschoris victoriae*

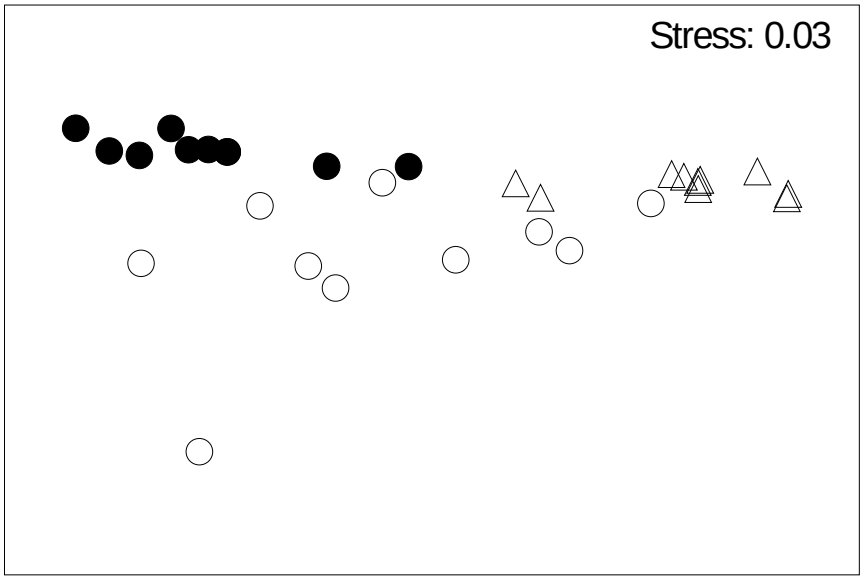


(a) Wamberal

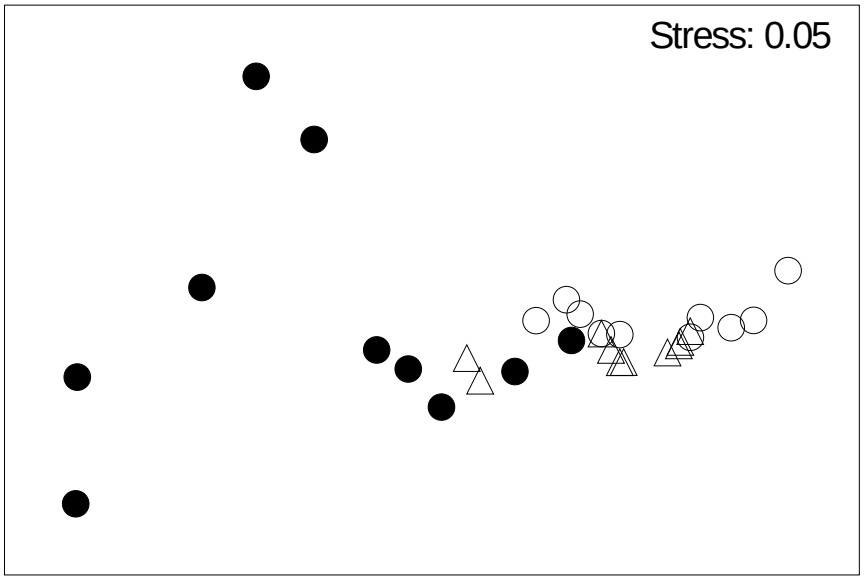


(b) Terrigal

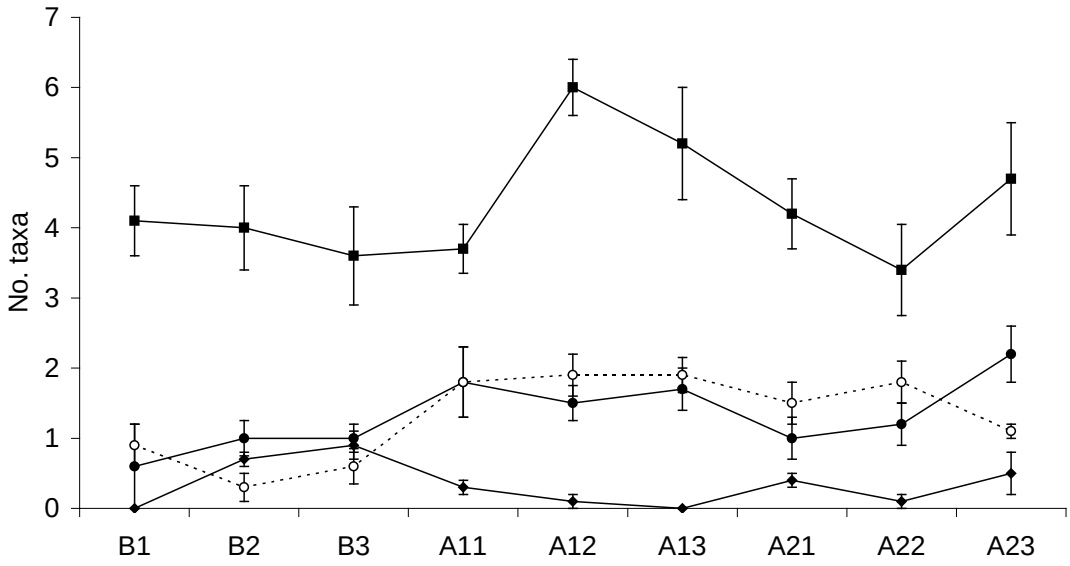
(c) Avoca



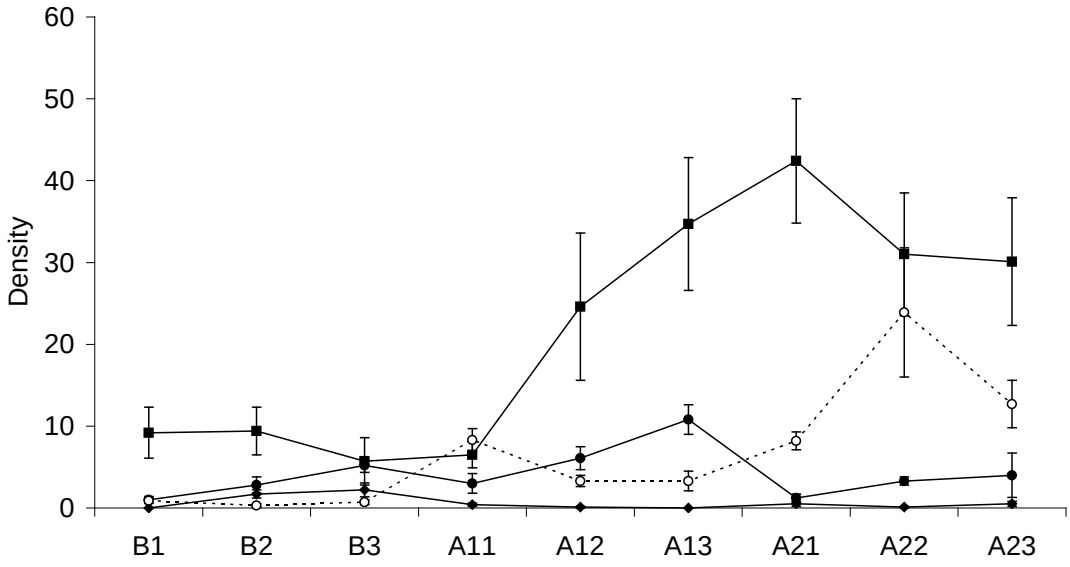
(d) Cockrone



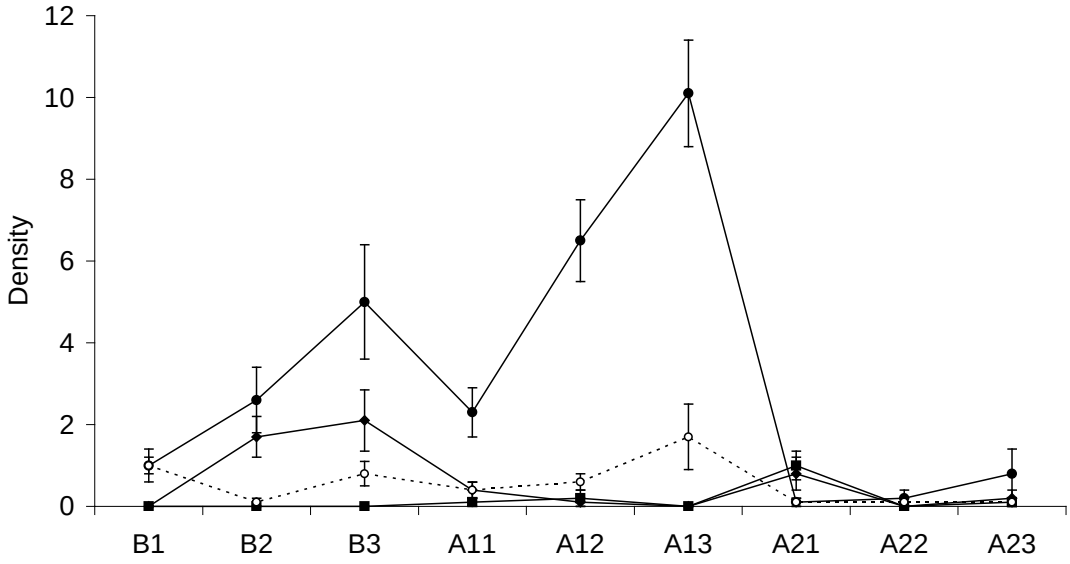
(a) Macroinvertebrate taxa



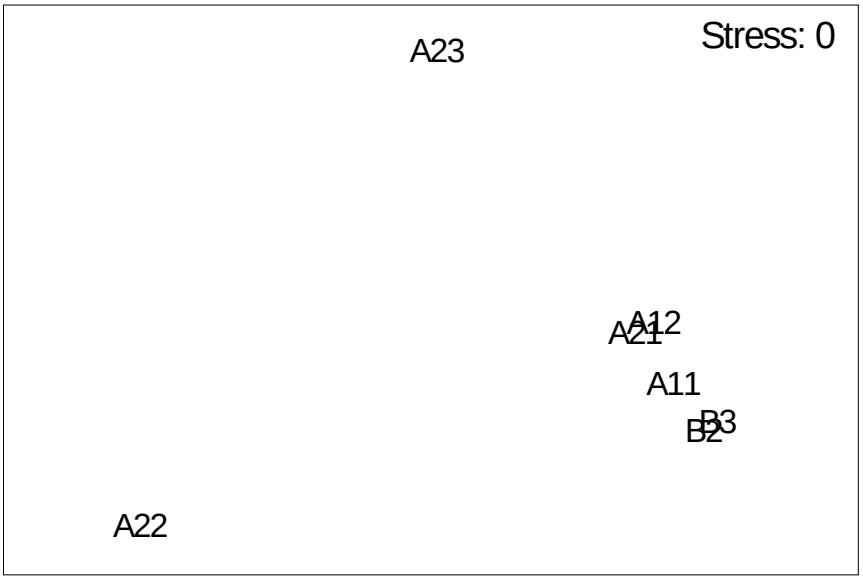
(b) Total macroinvertebrates



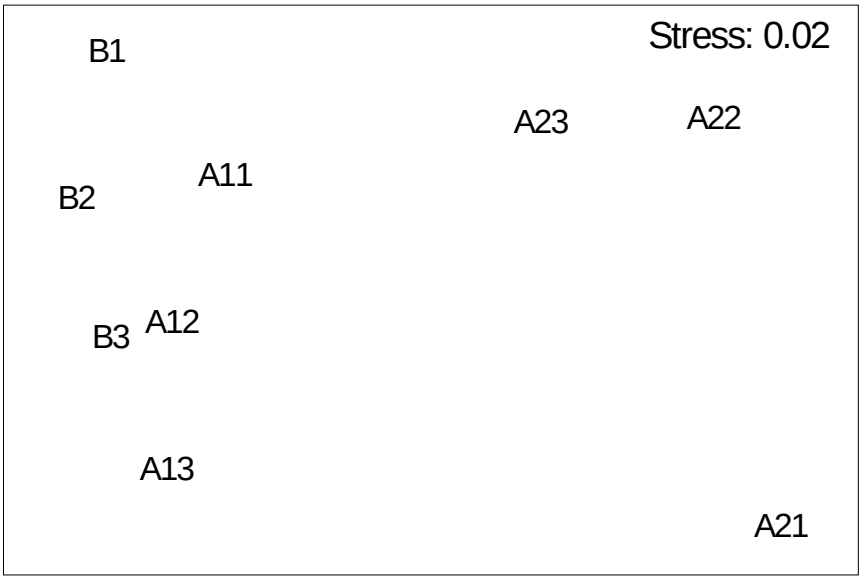
(c) *Paracalliope australis*



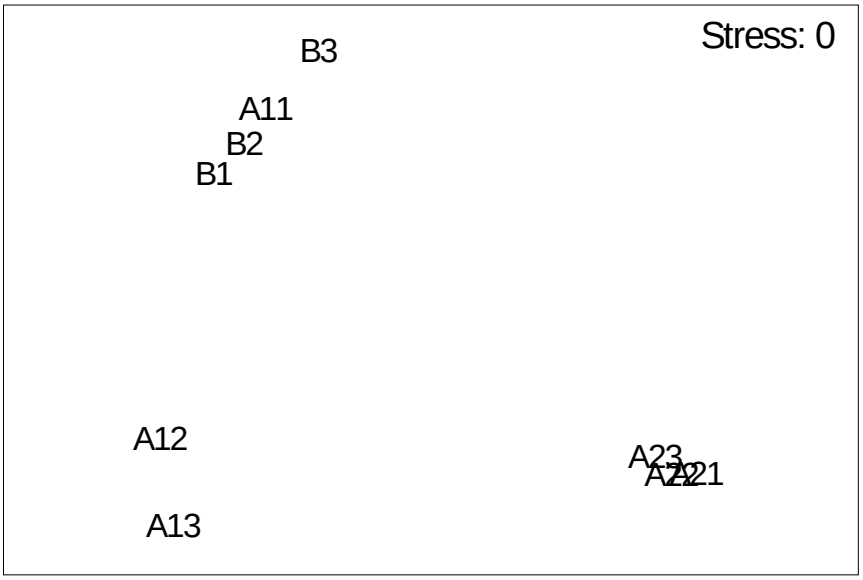
(a) Terrigal



(b) Wamberal



(c) Avoca



(d) Cockrone

Table 1
Major features of the intermittently open estuaries used in this study (from Gosford City Council (1995) and Healthy Rivers Commission (2002)).

Estuary	Surface area (km^2)	Catchment area (km^2)	Catchment condition
Wamberal	0.57	6.6	Partly protected within Wamberal Lagoon Nature Reserve; majority severely modified for urban and semi-rural development
Terrigal	0.27	9.5	Severely modified for urban development
Avoca	0.63	11.6	Modified for urban and semi-rural development
Cockrone	0.38	7.2	Modified for urban and semi-rural development

Table 2
Summary of analysis of variance for species richness and density of selected taxa for two comparisons in 2001: Before and After 1 periods (B-A1) and Before and After 2 periods (B-A2). *** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$, ns $P > 0.05$.

		Taxonomic richness[1]				*Paracalliope australis*[2]				*Aschoris victoriae*[3]			
		B-A1		B-A2		B-A1		B-A2		B-A1		B-A2	
Source of variation	df	MS	*F*	MS	*F*	MS	*F*	MS	*F*	MS	*F*	MS	*F*
Period	1	0.14		0.03	0.07 ns	20.15		52.84		0.20		234.61	
Treatment	1	6.37		0.48	0.22 ns	26.83		38.60		4.05		165.31	
Estuary (Treatment)	2	0.21		2.13	13.99***	7.82		4.00		21.12		264.01	
Period x Treatment	1	1.09		0.62	1.45 ns	0.52		3.09		42.05		456.01	
Period x Estuary(Treatment)	2	2.70	22.54***	0.43	2.81 ns	6.92	12.81***	5.41	11.42***	42.02	9.72***	262.11	6.38**
Residual	72	0.12		0.15		0.54		0.47		4.32		41.05	

[1] Ln(x+0.1) transformed, variances homogeneous
[2] Ln(x+1) transformed, variances homogeneous
[3] untransformed, variances heterogeneous

Table 3
Summary of (a) non-parametric multivariate analysis of variance for assemblages of macroinvertebrates for two comparisons in 2001 - Before and After 1 periods (B-A1) and Before and After 2 periods (B-A2) (*** $P < 0.001$) and (b) dissimilarity (R-value) in assemblage structure between Before and After 1 periods (B-A1) and Before and After 2 periods (B-A2). R-values range from 0 (no dissimilarity) to 1 (completely dissimilar).

(a)

		B-A1		B-A2	
Source of variation	df	MS	*F*	MS	*F*
Period	1	33103.27		44297.55	
Treatment	1	13659.14		15541.62	
Estuary (Treatment)	2	8574.42		7916.03	
Period x Treatment	1	9132.77		17206.46	
Period x Estuary (Treatment)	2	7365.38	5.55 ***	7609.56	6.51 ***
Residual	72	1325.93		1169.32	

(b)

Estuary	B-A1	B-A2
Wamberal (opened)	0.72	0.76
Terrigal (opened)	0.78	0.77
Avoca (control)	0.62	0.87
Cockrone (control)	0.58	0.54

Table 4
Results of BIOENV analysis showing the Spearman rank correlation coefficient (ρ) and the set of environmental variables that best matches the patterns in the biotic assemblages. All environmental variables ln transformed prior to analysis.

2001			2004		
Estuary	ρ	Selected variables	Estuary	ρ	Selected variables
Terrigal (opened)	0.66	1 mm and 63 µm sediment fractions, salinity, pH	Terrigal (opened)	0.40	Salinity
Wamberal (opened)	0.38	0.5 mm and 0.2 mm sediment fractions	Wamberal (control)	0.24	1 mm and 63 µm sediment fractions, salinity
Avoca (control)	0.62	Salinity, temperature	Avoca (control)	0.82	Temperature
Cockrone (control)	0.39	pH	Cockrone (control)	0.38	0.5 mm sediment fraction, salinity

Table 5
Summary of results of asymmetrical ANOVA for taxonomic richness of macroinvertebrates, total density of macroinvertebrates, and density of individual species ([a] ln (x + 1) transformed, [b] untransformed, variances heterogeneous, [c] ln (x + 0.1) transformed, *** $P < 0.001$, * $P < 0.05$, ns $P > 0.05$)
(a) Before vs After 1

Source of variation	df	Richness[a]		Total density[b]		*Paracalliope australis*[c]	
		MS	*F*	MS	*F*	MS	*F*
Before vs After: B	1	3.47		1349.00		3.83	
Days (B): D(B)	4	0.12		210.89		5.89	
Estuaries: E	3	20.84		2110.45		106.27	
Open[1]	1	29.07		1917.54		13.27	
Controls[1]	2	16.73		2206.90		152.77	
B x E	3	2.18		452.74		17.05	
B x Open[1]	1	5.17	7.49 ns	690.31		42.94	
B x Controls[1]	2	0.69	7.67 ***	333.95		4.10	
D(B) x E	12	0.32		232.53		4.98	
D(Before) x E[1]	6	0.47		31.61		6.43	
D(Before) x Open[1]	2	0.87		7.87		10.88	
D(Before) x Controls[1]	4	0.27		43.48		4.21	
D(After) x E[1]	6	0.18		433.45		3.53	
D(After) x Open[1]	2	0.22	2.44 ns	156.51	0.27 ns	5.38	2.07 ns
D(After) x Controls[1,2,3]	4	0.17	1.89 ns	571.93	25.83 ***	2.6	2.63 *
Residual	216	0.09		22.14		0.99	

[1] Repartitioned sources of variation
[2] If D(After) x Controls / residual and D(After) x Open / residual are not significant impact occurs if B x Controls / residual and B x Open / B x Controls are significant
[3] If D(After) x Controls / residual is significant test D(After) x Open / D(After) x Controls to determine if the interaction between days of sampling and the difference between open and control estuaries after the opening is greater than the interaction between days of sampling and control estuaries after the opening

Table 5 cont'd

Summary of results of asymmetrical ANOVA for taxonomic richness of macroinvertebrates, total density of macroinvertebrates, and density of individual species ([a] ln (x + 1) transformed, [b] untransformed, variances heterogeneous, [c] ln (x + 0.1) transformed, *** $P < 0.001$, * $P < 0.05$, ns $P > 0.05$)

(b) Before vs After 2

		Richness[a]		Total density[b]		*Paracalliope australis*[c]	
Source of variation	df	MS	*F*	MS	*F*	MS	*F*
Before vs After: B	1	0.96		5273.44		34.03	
Days (B): D(B)	4	0.11		100.01		6.58	
Estuaries: E	3	17.75		5113.68		18.52	
Open[1]	1	18.54		4047.02		1.47	
Controls[1]	2	17.35		5647.01		27.05	
B x E	3	1.07		2475.68		20.49	
B x Open[1]	1	1.40	1.55 ns	2125.24		2.88	
B x Controls[1]	2	0.90	7.50 ***	2650.90		29.30	
D(B) x E	12	0.32		220.07		4.78	
D(Before) x E[1]	6	0.47		31.61		6.43	
D(Before) x Open[1]	2	0.86		7.87		10.88	
D(Before) x Controls[1]	4	0.27		43.48		4.21	
D(After) x E[1]	6	0.18		408.53		3.12	
D(After) x Open[1]	2	0.16	1.33 ns	75.36	0.13 ns	0.15	0.03 ns
D(After) x Controls[1,2,3]	4	0.19	1.58 ns	575.11	12.50 ***	4.61	4.39 *
Residual	216	0.12		46.01		1.05	

[1] Repartitioned sources of variation

[2] If D(After) x Controls / residual and D(After) x Open / residual are not significant impact occurs if B x Controls / residual and B x Open / B x Controls are significant

[3] If D(After) x Controls / residual is significant test D(After) x Open / D(After) x Controls to determine if the interaction between days of sampling and the difference between open and control estuaries after the opening is different from the interaction between days of sampling and control estuaries after the opening

Table 6
Summary of non-parametric multivariate analysis of variance results comparing assemblage structure in each estuary between before and after periods. *** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$, ns $P > 0.05$.

Estuary	**Analysis**	**Source of variation**	**df**	**MS**	***F***
Terrigal (opened)	Before vs After 1	Period	1	18813.37	1.89 ns
		Days (Period)	4	9939.62	7.08 ***
		Residual	54	1404.35	
	Before vs After 2	Period	1	11907.48	1.07 ns
		Days (Period)	4	11097.03	5.94 ***
		Residual	54	1869.37	
Wamberal (control)	Before vs After 1	Period	1	13573.92	2.29 ns
		Days (Period)	4	5921.56	3.62 ***
		Residual	54	1633.46	
	Before vs After 2	Period	1	43679.28	7.86 **
		Days (Period)	4	5555.45	1.98 *
		Residual	54	2802.26	
Avoca (control)	Before vs After 1	Period	1	15194.89	2.11 ns
		Days (Period)	4	7185.53	4.17 ***
		Residual	54	1721.88	
	Before vs After 2	Period	1	78151.98	27.7 ***
		Days (Period)	4	2821.71	2.08 *
		Residual	54	1355.02	
Cockrone (control)	Before vs After 1	Period	1	46740.82	6.75 **
		Days (Period)	4	6921.12	3.34 **
		Residual	54	2069.66	
	Before vs After 2	Period	1	72841.03	12.39 ***
		Days (Period)	4	5877.28	3.05 **
		Residual	54	1929.47	