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Patterns of Distribution of Macro-fauna in Different Types of Estuarine, Soft Sediment Habitats Adjacent to Urban and Non-urban Areas

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Urban development typically creates a large number of potentially interacting disturbances that may cause impacts on assemblages of animals and plans in estuarine habitats. We tested predictions from the general model that intertidal areas exposed to different types of disturbances have different types of assemblages of benthic macrofauna. Different parts of the Port Hacking Estuary (New South Wales, Australia) are exposed to varying degrees of disturbance by human activities. We predicted that the average structure of assemblages of intertidal animals, and patterns of variability would differ between urban and non-urban areas of Port Hacking. Consistent with previous observations from the literature, there were differences in average structure between urban and non-urban sandy areas. Qualitative differences between abundances of individual taxa in urban and non-urban areas were generally not consistent with previous observations. Differences between assemblages in urban and non-urban areas were not observed in muddy sediments, nor in sediments among mangroves and seagrass. No significant differences in variability was observed between urban and non-urban areas. Two general models may be proposed to explain the observed differences in response to urbanization in different habitats: (1) animals are exposed to different levels or combinations of disturbances in different habitats; or (2) assemblages of animals differ in sensitivity to disturbances among habitats. © 2001 Academic Press

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Introduction

Urban centres have often developed in estuaries and today few estuaries remain unaffected by human activities. As an example, in 1992, 75% of the population in New South Wales, Australia, lived adjacent to estuaries. Only four of more than 100 estuaries in the region were considered ' pristine '(Anon, 1992). Urban development typically causes a large number of simultaneous potentially disturbing human activities (e.g. McDonnell & Pickett, 1990; Gray, 1997; Breitburg *et al.*, 1998). Disturbance or contamination are used to describe processes that may or may not cause stress, pollution or impacts on animals and plants (GESAMP, 1994). Aquatic organisms in these areas are typically exposed to increased concentrations of contaminants, such as several types of metals (Morrisey *et al.*, 1994*b*; Birch, 1996), pesticides (Garouma *et al.*, 1998), hydrocarbons (McGuinnes, 1990; Siewicki, 1997), and nutrients (Carpenter *et al.*, 1998). Other types of disturbances in urban areas include siltation (Newell *et al.*, 1998), structural

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modification (Bruun, 1994), recreational activities (Wynberg & Branch, 1994; Keough & Quinn, 1998), and introduction of exotic species (Cohen & Carlton, 1998). The intensities of these disturbances vary at different spatial and temporal scales (Raimondi & Reed, 1996; Kucklick *et al.*, 1997; Carpenter *et al.*, 1998) and impacts caused by one type of disturbance may be modified by other types of disturbances (Gray, 1974; Jones, 1975). Impacts on assemblages of animals and plants in urban areas are likely therefore to be expressed as complex patterns in time and space.

Estuaries contain a great variety of benthic habitats, including rocky reefs, boulder fields, unvegetated sandy or muddy sediments, and sediments covered with seagrass, saltmarshes and mangrove forests (e.g. McLusky, 1989). These different habitats contain different assemblages of animals and plants, vary in productivity and are potentially exposed to different types of disturbances. Efficient management of estuaries need to account for differences and links among habitats within estuaries in addition to links with habitats in surrounding coastal areas (Turner, 1989, Fairweather & Quinn, 1992; Constable, 1999, Lee, 1999).

The presence of a disturbance does not necessarily mean that assemblages of animals and plants are impacted. Individual organisms may tolerate actual levels of disturbances (Bryan et al., 1987) and populations may resist or recover from disturbances at a rate which is large enough to prevent persistent impacts (Holling, 1973; Underwood, 1989). To be logical and cost-efficient, it is therefore necessary to identify patterns of individuals, populations or assemblages which indicate impacts before causes of those impacts could be removed or mitigated to avoid further environmental degradation (Hilborn & Walters, 1981; Underwood & Peterson, 1988; McDonnell & Pickett, 190; Underwood, 1990; GESAMP, 1994). Such patterns can be established without a complete mechanistic understanding of effects on different components of the assemblages of organisms or a detailed knowledge about dose-response relationships (Underwood & Peterson, 1988; Peters, 1991; Holling, 1998). The power of multivariate techniques for detecting environmental impacts in complex assemblages of animals and plans has been repeatedly demonstrated (Gray et al., 1990; Clarke, 1993; Clarke & Ainsworth, 1993; Warwick & Clarke, 1993a).

In this paper, we test hypotheses from the general model that assemblages of macro-fauna in different soft-sediment habitats are affected by the degree of urbanization in adjacent parts of the catchment surrounding Port Hacking in New South Wales, Australia. A previous study (Stark, 1998) has shown that assemblages of macrofauna in sandy sediments of Port Hacking (and Port Jackson) were strongly correlated with concentrations of heavy metals and the degree of urbanization. We predicted that if we sampled macrofauna in new sites and in additional habitats, we would find differences between urban and non-urban sandy habitats similar to those found by Stark (1998). We also predicted that there would be similar differences between urban and non-urban areas in samples from other types of habitats: muddy sediments, mangroves and seagrasses. Thus, this study is not primarily concerned with the proximate causes of differences between urban and non-urban areas. Instead, the aim is to test the generality of patterns, in terms of magnitude and character, among times, places and habitats. Such repeated testing can lead to a broader understanding of the types of spatial patterns of fauna in intertidal areas. In addition, it has been proposed and documented that environmental impacts on populations and assemblages of animals may be detected as a change in variance, rather than in average conditions (Underwood, 1992; Warwick & Clarke, 1993b; Chapman et al., 1995). Therefore, we

also test general hypotheses about differences in variability among urban and non-urban areas in these habitats.

Materials and methods

Study sites and sampling design

The Hacking River catchment (Figure 1), which drains into Port Hacking has a human population of approximately 72 500, with the majority on the northern side of the estuary (Knowles, 1996). Sixty per cent of the catchment consists of undeveloped bushland within the Royal National Park, south of the estuary. Previous studies indicate that differences in land-use among parts of the catchment cause differences in contamination of metals in sediments (Stark, 1998) and in various measures of water quality (Knowles, 1996).

Samples of sediment and their fauna were taken at sites in urban parts (Yowie Bay and Burraneer Bay) and at sites (Hacking River and South West Arm) in non-urban parts of Port Hacking (Figure 1). Apart from general differences in use of the surrounding land areas, there are also quantifiable differences in recreational use between waterways in urban and non-urban areas. For example, in 1998 there were 254 and 203 mooring licenses, one and two marinas (Sydney Waterways Authorities, pers. comm.), and 121 and 141 pontoons and slipways (Centre for Research on Ecological Impacts of Coastal Cities, unpubl. data) in Yowie Bay and Burraneer Bay, respectively. Although the non-urban sites are used for recreation, they contain none of these installations and thus disturbances associated with recreational boating differ by orders of magnitude among urban and non-urban areas.

Four types of intertidal habitats were identified in a general inspection before sampling. Sites were selected for the study only if they contained patches of all types of habitats. These were characterized on the basis of general appearance and characteristics of sediment and presence plants. Samples were taken in unvegetated (1) sandy and (2) muddy sediments and in sediments within (3) seagrass-beds (Zostera capricorni) and (4) areas with pneumatophores of mangroves (Avicennia marina). The latter two categories were easily identified using presence of seagrass and pneumatophores as the sole criteria. It is, however, likely that they also differed more or less consistently with respect to other environmental variables (e.g. depth, sediment characteristics, duration of emersion, etc.). The consistency in classification of sandy and muddy habitats was more difficult.
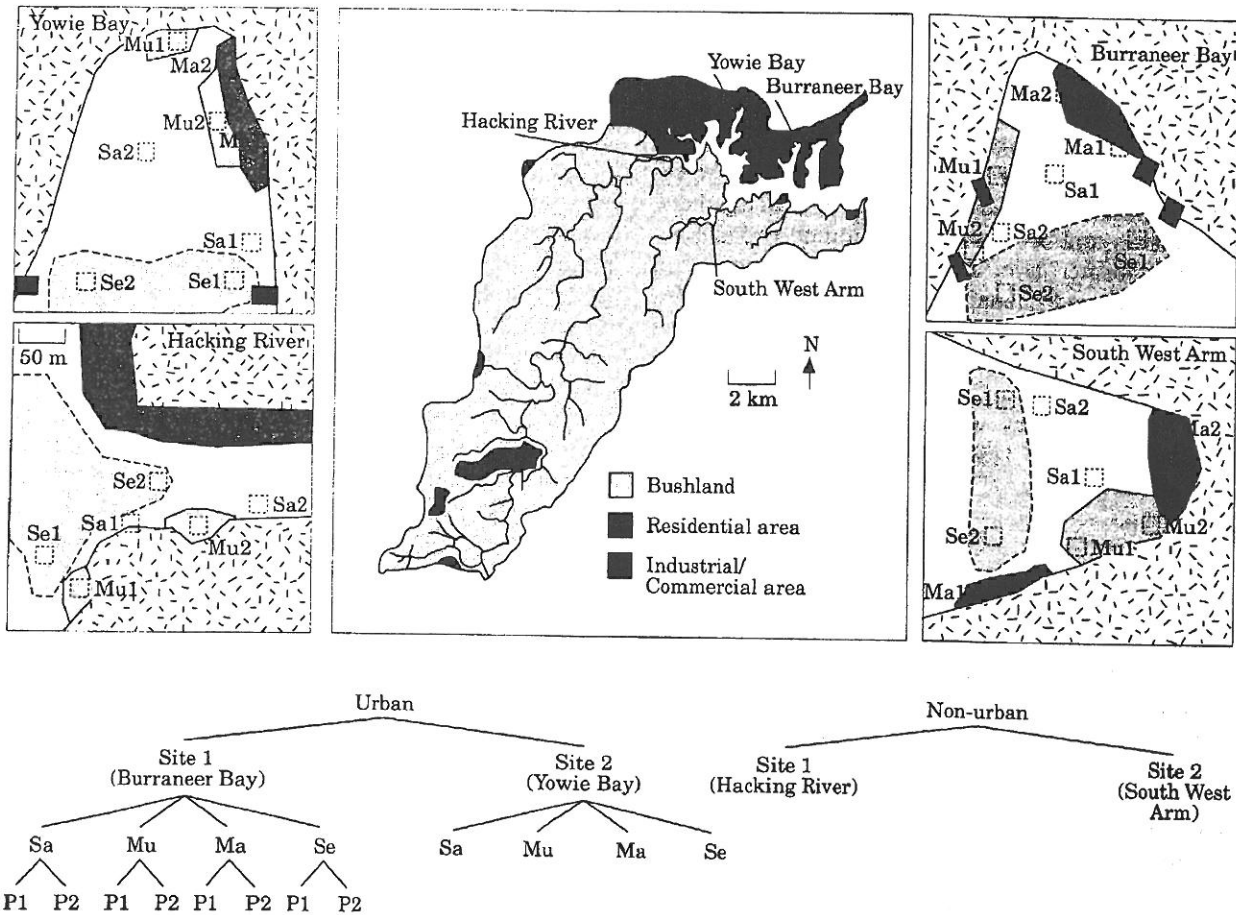
FIGURE 1. Map of the Port Hacking Estuary and its catchment area. Urban sites are Yowie Bay and Burraneer Bay, and non-urban sites are Hacking River and South West Arm (modified after Knowles, 1996). Shown are also the placement of individual plots in the four sites (Sa=sand, Mu=mud, Se=seagrass and Ma=mangrove; scale bar is the same for all sites) and a schematic picture of the experimental design.

Unvegetated areas of sediment were in all sites predominantly characterized as sandy. Subsequently, areas containing more fine particles (relative to other areas at the particular site) were then identified as ' muddy ' by physical examination and visual inspection of the sediment. Within each site, two replicate plots of each type of habitat were identified at least 50 m apart (Figure 1). Four replicate cores (diameter=10 cm, depth=10 cm) were sampled in each plot (5 × 5 m).

In the laboratory, samples were preserved in 7% formalin buffered in seawater and subsequently sieved (0·5 mm mesh-size) and sorted under dissecting microscope using 16 × magnification. Animals were identified to intermediate taxonomic levels, which varied among phyla (e.g. families for polychaetes, orders or classes for crustaceans and classes for molluscs). This is the same taxonomic resolution which was used in the preceding study (Stark, 1998).

Several recent studies have shown that identification to a coarse taxonomic level is often sufficient and cost-efficient for the detection of environmental impacts or patterns of spatial and temporal variability in assemblages of macrofauna (e.g. James et al., 1995; Olsgard et al., 1997; Chapman, 1998). The relatively coarse taxonomic resolution may result in a loss of information which could potentially mask differences between urban and non-urban areas. It is important to stress that the hypotheses tested are concerned only with the particular taxonomic resolution which is used. With this restriction, however, they provide valid tests of the hypotheses. Animals from samples have been preserved in vials with 70% alcohol.

Analytical procedures

Multivariate patterns of assemblages were visualized using non-metric multidimensional scaling plots

(nMDS). Tests for differences in structure and composition of assemblages were done using analysis of similarities (ANOSIM), which uses ranks of Bray-Curtis dissimilarities (Clarke & Warwick, 1994). Spatial variability was also evaluated using quantitative estimates of Bray-Curtis dissimilarities. Dissimilarities were calculated within plots, among plots, among sites and between urban and non-urban sites. For the former two, hypotheses about effects of urbanization and habitat were tested using ANOVA. Contributions from individual taxa to dissimilarities were estimated using similarity percentage breakdown procedure (SIMPER; Clarke & Warwick, 1994). All multivariate analyses were done using fourth-root transformed data (Clarke & Warwick, 1994).

Results

Ordinations indicate that the difference in average structure and composition between assemblages of animals in urban areas and non-urban areas varied among habitats [Figure 2(a–d)]. In sand, assemblages in urban areas were clearly different from those in non-urban areas [Figure 2(a)]. In seagrass and mud, there was a tendency towards a difference in average assemblages between urban and non-urban areas [Figure 2(b,c)], but there was also a large overlap between these two types of areas. In the mangroves, the ordination did not indicate any difference between assemblages in urban and non-urban areas [Figure 2(d)]. Stress-values differed among ordinations, but were small enough to allow a useful interpretation (range 0·16–0·19; Clarke & Warwick, 1994). Consistent with ordinations, statistical tests using ANOSIM revealed significant differences between assemblages in urban and non-urban sand, but not in any other habitat (Table 1). There was significant spatial variability in assemblages among sites or among plots within sites in all types of habitats ($P<0.05$ for 13 of 24 comparisons). The nominal levels of type 1 error ($a=0.05$, i.e. the error rates per comparison) of these comparisons, were not corrected to maintain the error rate per experiment (i.e. the probability that there will be an error in any of all comparisons) at 0·05. Such procedures would have required the nominal level to be set at $a=0.0021$ and would have increased the risk of type 2 errors in each comparison (see Underwood, 1997 for further discussion).

Effects of urbanization on structure of assemblages can also be evaluated using quantitative measures of Bray-Curtis dissimilarities (Table 2). These are average estimates of dissimilarities between urban and non-urban areas and among sites, plots and replicates for the different habitats. Dissimilarities between assemblages in urban and non-urban areas were approximately 62% in sand and mud and 50% in mangrove and seagrass (Table 2). Assemblages in mud were, however, equally variable among sites. Therefore, the large dissimilarity between urban and non-urban areas in mud cannot be attributed to the fact that the sites were located in areas of different degree of urbanization. This interpretation is consistent with the patterns obtained from ordination (Figure 2).

No differences between urban and non-urban areas in terms of spatial variability either within or among plots were detected using ANOVA (Table 3). Although they were not significantly different between urban and non-urban areas, differences were relatively large in some habitats (Table 2). In seagrass for example, dissimilarities at different scales were 50 to 60% larger in urban areas. In contrast, variability was 15 to 20% larger in non-urban areas in assemblages from sand, mud and mangroves (Table 2).

In contrast to assemblages in mud, mangrove and seagrass, those in sand appear to differ between urban and non-urban areas (Figure 2). One possible explanation for this may be that diversity and composition of assemblages differ among habitats (i.e. particularly sensitive species may occur only in sandy habitats). In spite of the somewhat subjective and contrived classification of discrete habitats based on continuously varying environmental variables, there were clear differences among assemblages in those habitats (Figure 3). NMDS indicate that the average structures of assemblages in sand and seagrass were different from those in all other habitats, but those amongst mangrove pneumatophores and in mud were more similar (Figure 3). Furthermore, the diversity of animals differed among habitats. A total of 45 taxa were recorded in samples from seagrass, while 32, 27 and 20 taxa were recorded in mud, sand and mangroves, respectively.

SIMPER showed that the patterns of dissimilarity between urban and non-urban areas were complex (Table 4). Dissimilarities from 12 taxa had to be invoked to explain 75% of the difference between urban and non-urban sandy areas. The taxa contributing most to dissimilarities in sand were syllids, copepods and spionids explaining between 7, 6 and 5% of the dissimilarity. Interestingly, these taxa were found at large densities in non-urban areas but were very scarce in urban areas (Table 4). None of these taxa were found at large densities in mud or mangroves. In seagrass, these three taxa were relatively abundant in non-urban areas, but the
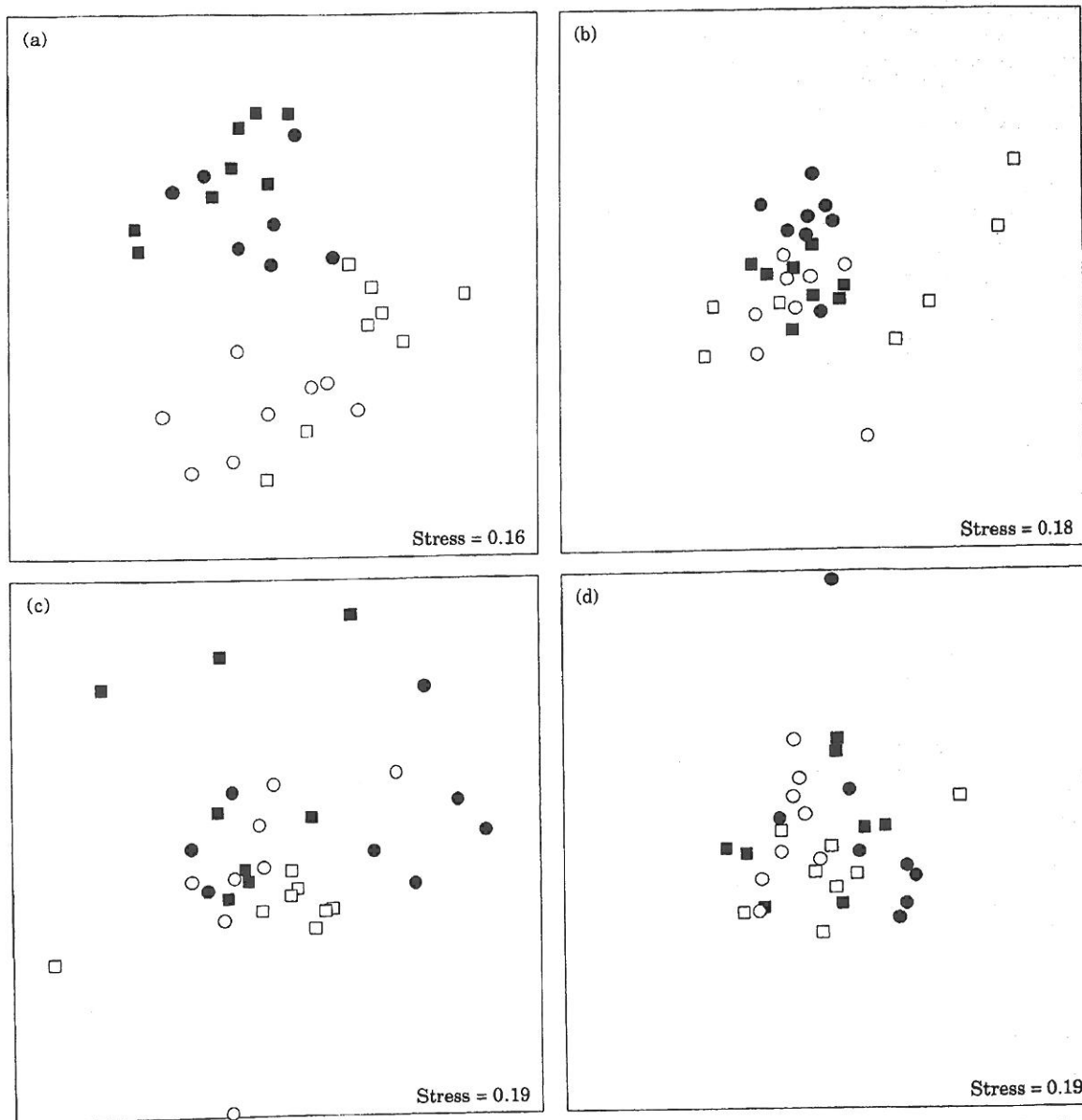
FIGURE 2. NMDS ordinations of assemblages in (a) sand, (b) seagrass, (c) mud and (d) mangroves. Filled and empty symbols represent samples from non-urban and urban areas, respectively. Squares and circles show sites one and two, respectively. Each point on the ordination represents one sample.

differences between urban and non-urban areas were smaller than in sand (Table 4).

Discussion

Consistent with previous evidence from other sites (Stark, 1998), assemblages of animals in sand-flats adjacent to urban areas of Port Hacking were different from those in non-urban areas. This indicates that there are temporally and spatially consistent

differences between the structures of assemblages in urban and non-urban sand-flats. Stark (1998) found that differences among assemblages in areas of different degrees of urbanization were correlated with differences in concentrations of lead, zinc and copper, but not with sediment grain-size or organic content. This observation suggests that the observed differences may have been caused by metals, which are mainly released into waterways via urban run-off. An alternative, more general model is, however, that

TABLE 1. Tests of hypotheses about differences between urban and non-urban areas, among sites and among plots for each habitat separately using ANOSIM

	Sand		Mud		Mangrove		Seagrass	
Comparison	R	P	R	P	R	P	R	P
U *vs* NU	0·82	★	0·10	ns	−0·02	ns	0·15	ns
Sites (U)	0·45	★★	0·30	★★	0·07	ns	0·25	★★
Sites (NU)	0·38	★★	0·12	ns	0·24	★	0·42	★★
Plots (U1)	0·51	★	0·23	ns	0·10	ns	−0·04	ns
Plots (U2)	0·50	★	0·41	★	0·71	★	0·29	ns
Plots (NU1)	0·29	ns	0·08	ns	0·20	ns	0·16	ns
Plots (NU2)	0·67	★	0·56	★	0·48	★	0·09	ns

★★$P<0\cdot01$, ★$P<0\cdot05$ and ns $P>0\cdot05$.

R is the test statistic and P is the result of the test for significance using permutations of plots (for tests of U *vs* NU) or replicate cores (for tests of Sites and Plots). U=Urban sites, NU=Non-urban sites, U1=Yowie Bay, U2=Burraneer Bay, NU1=Hacking River and NU2=South West Arm.

differences among assemblages were caused by any other component (or combination of components) or urban run-off, e.g. hydrocarbons, nutrients, silt or fresh-water, which are correlated with concentrations of metals. Indeed, data from a subsequent study aimed at testing whether differences among sites are caused by factors associated with the sediment (Lindegarth, in prep.), indicate that there are differences in sediment characteristics among the sites in his study and that the organic content appear to be larger in sites in urbanized areas (Table 5). Differences in sediment characteristics between pairs of sites do, however, not accurately explain the observed differences in assemblages (Figure 4). Although the number of possible pairwise comparisons are limited to six, the diffuse relationship between animals and

environmental data indicates that sediment characteristics are a poor predictor for the differences observed in this study.

The number of species required to explain any percentage of dissimilarity between urban and non-urban areas, differed among habitats (Table 4). In seagrass, 19 taxa explained 75% of the dissimilarity. The corresponding numbers in sand, mud and mangroves were 12, 11 and 9. Interestingly, cumulative dissimilarities in sand were very similar to those observed in the same habitat by Stark (pers. comm.). This indicates some overall generality in dissimilarity between sandy urban and non-urban areas. Nevertheless, the relative importance of individual taxa were not quantitatively or qualitatively consistent with those observed by Stark (pers. comm.). For example, only two taxa (nereids and amphipods) of eight required to explain 50% of the dissimilarities,

TABLE 2. Measures of average Bray-Curtis dissimilarities among samples from different sets of sites and plots for each of the different habitats

Comparison	Sand	Mud	Mangrove	Seagrass
U *vs* NU	62·2	62·8	50·2	49·7
Among Sites	54·2	61·4	49·7	48·9
Sites (U)	49·7	57·4	43·2	58·6
Sites (NU)	58·6	65·5	56·2	39·2
Among Plots	49·3	58·7	50·6	43·7
Plots (U)	48·6	51·5	45·9	52·5
Plots (NU)	50·0	65·8	55·2	34·9
Within Plots	40·6	49·4	39·7	43·1
Within (U)	39·0	44·0	35·2	52·7
Within (NU)	42·3	54·8	44·1	33·4

U=Urban sites, NU=Non-urban sites

TABLE 3. Analyses of variance on Bray-Curtis dissimilarities (1) among samples within plots and (2) among plots within sites

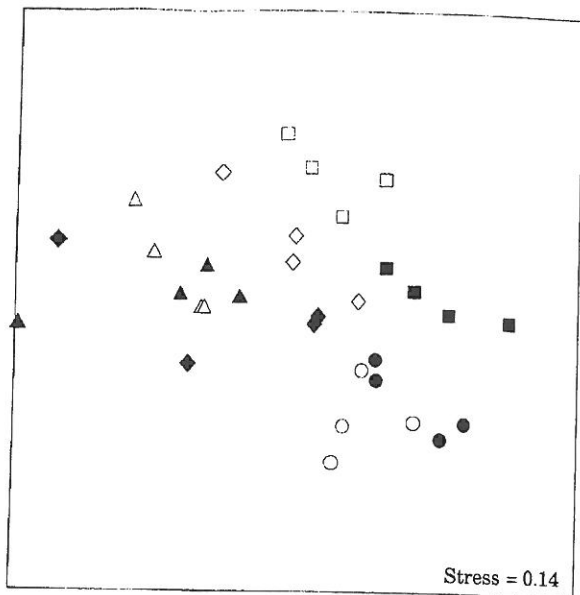| Source	d.f.	MS	F	P
(1) Habitat	3	152·4	0·58	0·65
U *vs* NU	1	6·3	0·16	0·73
Site (U *vs* NU)	2	39·3	0·17	0·84
Habitat*U *vs* NU	3	383·4	1·46	0·32
Habitat*Site (U *vs* NU)	6	263·3	1·15	0·38
Residual	16	228·1		
(2) Habitat	3	152·4	1·17	0·38
U *vs* NU	1	13·6	0·10	0·76
Habitat*U *vs* NU	3	194·9	1·50	0·29
Residual	8	130·0		

FIGURE 3. NMDS ordination of assemblages in non-urban (filled) and urban (empty) areas in (■) sand, (◆) mud, (▲) mangrove and (●) seagrass. Each point represents the average of one plot within a site ($N=4$).

were common to this and Stark's study (Stark, 1998). Furthermore, differences in abundance of some taxa (e.g. spionids, nephtyids and bivalves) between urban and non-urban sand-flats were not qualitatively consistent among studies. Other taxa (e.g. crustaceans, nereids and paranoids) showed qualitatively similar patterns in both studies. Thus in sand-flats, it appears that differences in the structure of assemblages between urban and non-urban areas are consistent among times and places, whereas patterns of individual taxa may or may not be consistent.

The differences between urban and non-urban sand-flats did not represent a general pattern in all habitats. In none of the other habitats were assemblages significantly different between urban and non-urban areas. In mud, however, the average dissimilarity between urban and non-urban areas was as large as that in sand, but dissimilarities among sites and plots were larger in mud compared to in sand. Bray-Curtis dissimilarities between urban and non-urban areas were, however, substantially smaller in mangrove and seagrass than those in sand and mud (Table 2). Differences in response to urbanization among habitats thus cannot solely be explained by differences in power of comparisons among the habitats. Patterns of assemblages in relation to urbanization also differed qualitatively among habitats. First, the taxa contributing to dissimilarities between urban and non-urban areas differed among habitats

(Table 4). Second, some taxa showed qualitatively opposite patterns in different habitats. For example, in sand and seagrass bivalves and oligochaetes were more abundant in non-urban areas, but in mud and mangroves they were more abundant in urban areas (Table 4).

It is often proposed that environmental impacts vary among habitats (e.g. Luoma, 1989; Jones, 1992). Because of potential linkages among habitats, i.e. migration of animals and transport of matter and energy, impacts in one habitat may cause impacts in adjacent habitats (Wiens et al., 1985; Fairweather & Quinn, 1992; Gray, 1997). Simultaneous tests of hypotheses about patterns and processes in several habitats are potentially useful for examining generality in magnitude and direction of environmental impacts. Although several studies have inferred differences in structure of assemblages among these habitat types (e.g. Edgar et al., 1994; Schrijvers et al., 1995; Connolly, 1997), few studies have simultaneously tested hypotheses about environmental impacts in different habitats. Testing for impacts of metals in similar habitats near a lead smelter, Ward and Hutchings (1996) concluded that there were larger impacts in intertidal mud and seagrass than in subtidal unvegetated areas and areas with seagrass. The comparison between inter- and subtidal habitats was, however, made difficult because subtidal sites were further away from the source of pollution and thus had smaller concentrations of metals. Nevertheless, graphical evidence suggests that impacts were larger in intertidal muddy areas compared to areas with seagrass (Ward & Hutchings, 1996). The results presented here are consistent with those patterns.

Ecological explanations as to why patterns of assemblages are correlated with the degree of urbanization in some habitats but not in others, may be categorized into two groups: (1) models based on differences in disturbance; and (2) models involving intrinsic differences among habitats or their associated assemblages. No data are presently available on the relative intensities of potential disturbances among the habitats sampled in this study. It is possible that some types of disturbances may differ among habitats although the maximal distance between plots of different habitats were only approximately 200 m within any site. For example, total concentrations of pollutants in sediments can differ significantly at small spatial scales (e.g. Krumgalz et al., 1989; Morrisey et al., 1994a) and the intensity of recreational activities differs among habitats (pers. obs.).

Alternatively, impacts of disturbances may differ among habitats because the resistance of populations and assemblages differ among habitats (Holling,

TABLE 4. Dissimilarity (D) between urban and non-urban areas due to individual taxa and their mean abundances in urban and non-urban areas for each habitat

	Sand			Mud			Mangrove			Seagrass		
	D	U	NU	D	U	NU	D	U	NU	D	U	ND
Crustaceans	13·5	1·3	26·5	11·0	6·0	7·7	12·9	2·8	5·4	13·3	24·3	41·9
Copepoda	5·6	0·3	20·9	3·5	4·5	4·2	—	0·1	0·0	2·2	13·1	21·8
Amphipoda	3·3	0·0	4·4	—	0·9	2·2	4·3	1·7	1·8	2·1	2·2	11·3
Decapoda	2·4	1·0	0·1	2·8	0·4	0·8	3·8	0·9	0·8	—	0·2	0·1
Isopod	—	0·0	0·1	—	0·0	0·2	3·4	0·0	2·3	—	0·7	0·5
Tanaidacea	—	0·0	0·0	—	0·0	0·1	—	0·0	0·6	2·0	4·9	3·1
Thalassinidea	—	0·0	1·0		0·3	0·2	—	0·0	0·0	1·8	0·7	2·8
Ostracoda	—	0·0	0·1	—	0·0	0·0	—	0·0	0·0	1·6	0·7	1·9
Polychaetes	35·6	38·3	110	30·5	70·7	21·9	15·8	4·1	4·9	26·4	43·1	63·9
Syllidae	6·7	1·3	57·4	—	0·6	1·8	—	0·2	0·3	3·5	6·5	27·6
Spionidae	4·9	3·4	25·8	5·5	8·3	13·2	3·5	0·4	1·7	2·8	10·3	20·0
Nereidae	4·3	21·3	14·7	2·6	1·4	0·2	—	0·0	0·0	1·3	0·5	0·8
Paraonidae	3·3	0·3	6·9	—	0·1	0·0	—	0·0	0·0	2·0	1·0	3·6
Sabellidae	3·2	7·8	0·3	8·7	55·1	1·1	—	0·1	0·4	1·9	4·9	0·2
Nephtyidae	2·7	1·2	0·9	3·1	0·6	1·6	—	0·3	0·8	1·4	0·9	0·6
Capitellidae	—	1·1	1·2	4·3	3·9	3·6	4·7	2·9	1·3	2·2	7·7	2·9
Opheliidae	—	0·0	0·9	—	0·2	0·0	—	0·0	0·0	2·1	4·2	3·6
Cirratulidae	—	0·0	0·0	—	0·1	0·0	—	0·0	0·1	1·7	2·2	1·1
Phyllodocidae	—	0·0	0·0	—	0·0	0·0	—	0·1	0·0	1·6	1·0	1·2
Other taxa	13·1	16·2	29·8	21·0	61·7	28·8	21·4	37·6	26·7	9·9	28·6	45·8
Oligochaeta	4·4	10·3	18·6	6·2	54·3	24·6	5·5	27·3	17·4	1·9	18·3	28·4
Nemertea	3·5	0·6	4·2	—	0·3	0·2	—	0·0	0·3	2·7	0·7	6·2
Bivalvia	2·9	4·4	6·8	5·2	5·8	0·9	3·2	5·5	4·1	1·5	4·8	6·8
Gastropoda	—	0·8	0·1	3·5	1·1	1·3	5·1	4·2	1·8	1·9	4·4	3·8
Insect larvae	—	0·0	0·0	2·7	0·2	1·3	4·8	0·6	2·6	—	0·1	0·1

The taxa listed explain 75% of the dissimilarity between urban and non-urban areas within each habitat.
U=Mean abundance at urban sites, NU=mean abundance at non-urban sites, — indicates taxa not among those contributing to the first 75% for a particular habitat.

TABLE 5. Percentage of sediment grain size fractions and organic content from the sites of this study (mean (se); N=4; data from Lindegarth, in prep.)

Location	>2 mm	>1 mm	>0·5 mm	>0·25 mm	>0·125 mm	>0·063 mm	<0·063 mm	Organic content
Hacking River	0·0 (0·0)	1·0 (0·1)	26·8 (2·3)	58·2 (0·8)	9·1 (1·2)	0·2 (0·0)	4·7 (2·2)	0·9 (0·1)
South West Arm	0·1 (0·0)	1·2 (0·2)	16·4 (0·5)	51·0 (0·5)	28·3 (0·7)	1·5 (0·1)	1·5 (0·1)	1·0 (0·1)
Yowie Bay	0·1 (0·0)	0·3 (0·0)	10·9 (0·5)	40·7 (1·2)	31·1 (0·8)	9·2 (0·6)	7·8 (0·4)	2·9 (0·3)
Burraneer Bay	0·3 (0·1)	3·0 (0·3)	28·2 (0·7)	36·6 (2·3)	22·2 (1·0)	6·0 (0·4)	3·7 (1·4)	1·5 (0·1)

1973; Underwood, 1989). For example, it is possible that the resistance of populations are affected by the intensity of natural physical stresses (Jones, 1975). The habitats sampled in this study roughly represent a vertical gradient with mangroves in upper intertidal areas, sand and mud at intermediate levels and sea-grass in lower parts. The rank order of observed effect-sizes (sand=mud>mangrove=seagrass) does not, however, provide any support for the notion that

potential gradients of natural stresses associated with depth, such as desiccation, heat and disturbance due to wave-action explain the fact that impacts vary in size among habitats.

A third type of model which could potentially explain why differences between assemblages in urban and non-urban areas appear to vary among habitats is that the methods used in sampling and identification introduces bias or is insufficient in some habitats. For
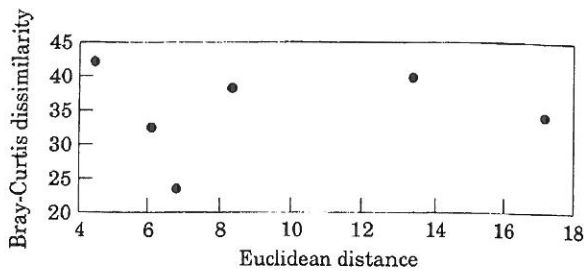
FIGURE 4. Plot of dissimilarities of assemblages (Bray-Curtis dissimilarity of averages) and sediment characteristics (Euclidian distance) between pairs of sites. Average abundances of assemblages were fourth-root transformed and the environmental data were transformed using $\log (x+0\cdot1)$.

example, it is possible that the coarse taxonomic resolution was sufficient to detect differences between urban and non-urban, sandy areas in this study and that of Stark (1998) but not in other types of habitats. Although such alternative explanations can not be ruled out, the fact that other types of differences (i.e. among sites and plots) were detected in all habitats, suggest that the methods were sensitive and accurate enough to detect natural variability in all habitats. Thus, it does not appear likely that the relatively coarse taxonomic resolution cause differences in sensitivity to urban impacts.

Differences in geochemistry among habitats, such as organic content, distributions of grain-sizes and content of oxygen may also affect availability and toxicity of pollutants such as metals (Luoma & Bryan, 1978; Luoma, 1989). Although physical variables were not measured quantitatively in this study, the *a priori* characterization of plots into different habitats was largely based on a qualitative assessment of vegetation, grain-size and organic content of the sediment. This assessment efficiently predicted differences in the structure of benthic assemblages (Figure 3). The fact that assemblages differed between the sand and mud indicate that at least some of the differences among habitats were related to sediment-characteristics (Snelgrove & Butman, 1994). It is also likely that other types of differences, which were observed may have been caused by variability in sediment characteristics. Approximately 50% of the pairwise comparisons of assemblages among plots and sites within habitats showed significant differences. Detailed data on sediment characteristics would have revealed the extent to which these differences were correlated with differences in sediment. This was beyond the scope of this paper. It is also possible that the observed difference between assemblages in urban and non-urban sandflats can be partly attributed to systematic differences in sediment

characteristics. Such differences were not observed by Stark (1998) and the use of randomly chosen replicate sites and plots within each of the urban and non-urban areas should in theory remove the risk of such confounding. Nevertheless, a larger number of sites would have allowed more robust conclusions about differences between urban and non-urban areas. In this context, it is important to note that, even if additional information on sediments were available, this would not have provided conclusive evidence about the causal links between the observed patterns of assemblages, sediments and the degree of urbanization. Any differences in sediments that might have been observed could also have been caused by human activities. For example increased run-off of storm-water and physical disturbance from boating could affect the characteristics of the sediment in urban areas. In this way, however, additional observations of sediments could have put more emphasis on alternative models to those based solely on concentrations of pollutants.

Although relative sensitivities are not always predictable and mechanisms are not fully understood, species of macrofauna are likely to vary in sensitivity to disturbances. Consequently, responses to urbanization may differ among habitats because abundances and composition of assemblages differ. This model could potentially explain differences in response to urbanization between sandy habitats on one hand, and mangroves and muddy habitats on the other hand (Table 4). Syllids, copepods and spionids were abundant in non-urban sand flats and contributed most to dissimilarities between urban and non-urban areas. These taxa were found only at small densities in mud and mangroves at non-urban sites. Thus, if urbanization were to reduce abundances even further in mud and mangrove, this would not cause large changes in dissimilarities. Because these three taxa were found at relatively large densities in seagrasses in non-urban areas, this model cannot serve as a general explanation of differences in response among habitats.

Observations of correlations between patterns of assemblages of animals and the degree of urban development, such as these in sandy sediments of Port Hacking, does not show conclusively that disturbances due to human activities have caused the patterns. The alternative model, that human development has occurred in areas with different assemblages to start with does not appear likely, but cannot be entirely ruled out. If, however, the predicted patterns do not occur, the tests do not support the model that there is an effect of urbanization (provided that issues of statistical power are properly considered (e.g. Fairweather, 1991). Given these limitations, we

conclude that tests of hypotheses about spatial patterns should play a central role in strategies for management of estuarine environments. These should be used to estimate the size of potential impacts, identify particularly sensitive components of assemblages and to generate predictions which can be tested in subsequent manipulative experiments.

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