DOI: 10.1007/s00267-004-0135-5

A posteriori Evaluation of Strategies of Management: The Effectiveness of No-Wash Zones in Minimizing the Impacts of Boat-Wash on Macrobenthic Infauna

MELANIE J. BISHOP*
Centre for Research on Ecological Impacts of Coastal Cities
Marine Ecology Laboratories
A11, University of Sydney
NSW 2006 Australia

ABSTRACT / Wind-driven waves are important in structuring intertidal and shallow subtidal assemblages of macrobenthic infauna. In the sheltered waters of estuaries, boat-generated waves (wash) may play a similar role because they are typically of a similar amplitude or larger than wind-driven waves. However, few studies have attempted to determine the role of wash in structuring assemblages. Consequently, strategies for managing boating focus around minimization of bank erosion. Along the Parramatta River (Sydney, Australia), no-wash zones have been established and mangroves planted to minimize the erosion of riverbanks and collapse of seawalls purportedly caused by 35-m-long RiverCat ferries. Although intended to also reduce the ecological impacts of wash, it is unclear whether these strategies achieve this goal. Unvegetated and vegetated (among the pneumatophores of mangroves) sediments were sampled in wash and no-wash zones along the Parramatta River to assess the effectiveness of no-wash zones and vegetation of river banks in reducing the ecological impacts of wash. Specifically, it was hypothesized that (1) assemblages of intertidal macrobenthic infauna would differ between wash and no-wash zones of the Parramatta River and (2) these differences would be greater in unvegetated than in vegetated habitat. As predicted, assemblages of macrobenthic infauna differed between the wash and no-wash zones. Capitellids, nereids, and spionids were more abundant in the no-wash zone. Contrary to the hypothesis, differences were no greater in the unvegetated habitat than in the vegetated habitat. The results suggest an impact of wash on assemblages of macrobenthic infauana and a role for no-wash zones in minimizing the effects of this disturbance.

Boating contributes to many disturbances in natural environments. Engines leak fuel and oil, hulls spread inorganic chemicals and metals used in antifouling paints (e.g., tributylin), heads dump sewage, propellers scar seagrass beds and boat moorings, and anchors dig holes in sediments. Marinas introduce structure where it would otherwise be absent, pontoons and pilings shade sediments, boat ramps and car-parks increase the area of impervious surface, and the dredging of boat channels drastically alters the benthos. These disturbances have the potential to affect the distribution and abundance of plants and animals in natural and man-made waterways and are the subject of an increasing number of studies (e.g., tributylin: Bryan and others 1986, Davies and Bailey 1991, propeller scaring: Bell and others 2002, Uhrin and Holmquist 2003, anchor damage: Davis 1977, boat mooring: Walker and others 1989, Hastings and others 1995, structure of marinas: Connell and Glasby 1999, Glasby and Connell 2001).

KEY WORDS: Boating; Disturbance; Mangroves; No-wash zone; Waves

Published online May 13, 2004.

**Present address*: University of North Carolina at Chapel Hill, Institute of Marine Sciences, Morehead City, NC 28557 USA; *email*: mbishop@email.unc.edu

Wash, the disturbed water left behind a moving vessel, may also affect the ecology and biology of aquatic organisms. Naturally produced waves are well documented as affecting the distribution and abundance of organisms on rocky shores (e.g., Dayton 1975, Menge 1978, McQuaid and Branch 1984, Underwood and Jernakoff 1984), and temporal variability in assemblages of soft sediments often corresponds to periods of storm (Eagle 1975, McCall 1977, McCall 1978, Yeo and Risk 1979, Dobbs and Vozarik 1983, Posey and others 1996). In the enclosed waterways of estuaries, waves produced by less extreme events can be of sufficient magnitude to affect the distribution and abundance of infaunal taxa (Emerson and Grant 1991, Commito and others 1995a, Commito and others 1995b). Boat-generated waves, which are typically of a similar amplitude or larger than these small wind-driven waves (Bhowmik and others 1982), may have similar effects on infauna.

Many studies have quantified the impacts of wash on the morphology of river-banks and mudflats (e.g., Hay 1968, Scholer 1974, Nanson and others 1994, Kirk and Single 2000). However, few have examined any ecological impacts of boat-generated waves, and strategies for managing boating are consequently focused around the minimization of bank erosion. Along the Parramatta River (Sydney Australia), wash from 35-m-long RiverCat ferries has purportedly caused erosion of riverbanks and collapse of seawalls (Department of Transport 1995). In response to these perceived impacts of wash, no-wash zones (where production of wash must be minimized) have been established and riverbanks revegetated with mangroves to stabilize sediments (Bennett and Reynolds 1993) and reduce the wave-energy reaching the shore (Kobayashi and others 1992, Massel and others 1999). If boat-wash does have an impact on assemblages in soft sediments, the establishment of no-wash zones and revegetation of river banks should minimize the ecological as well as the physical impacts of this disturbance.

Despite the clear rationale for introducing strategies of management, there have been few *a posteriori* attempts to assess whether their goals have been met. Even where data are unavailable from before implementation, tests of difference between managed and unmanaged places can provide valuable information on the likely impacts of a disturbance and the effectiveness of strategies of management. Asplund and Cook (1999) attempted to evaluate the role of no-wash zones in protecting aquatic macrophytes. The effects of wash were, however, confounded with other impacts of boating (e.g., scouring by propellers) as fewer boats entered no-wash zones than were found in adjacent areas. Although studies (Orth 1975, Orth 1977) suggest that the structural complexity of aquatic macrophytes reduces resuspension and transport of benthic invertebrates by waves and currents and increases settlement of larvae, the role of vegetation in dissipating the effects of wash is yet to be evaluated.

Here, I test the hypothesis that assemblages of intertidal macrofauna will differ between wash and no-wash zones of the Parramatta River. I also predicted that any difference in assemblages found between wash and no-wash zones would be greater in unvegetated habitat (intertidal mud-flats) than vegetated habitat (among the pneumatophores of mangroves). If differences are caused by greater rates of erosion of fauna in the wash zone, smaller abundances of taxa in the wash zone should drive differences in assemblages. By testing these hypotheses, two management strategies commonly used to reduce the physical effects of wash (the establishment of no-wash zones and the revegetation of riverbanks) are evaluated in terms of their minimization of the ecological effects of wash.

Materials and Methods

Study Area

RiverCat ferry services operate along a 19-km stretch of the Parramatta River, Sydney, Australia (151°13'E 33°52'S) between Circular Quay and Parramatta (Figure 1). This section of river is tidal, having a spring range of approximately 2 m, 50–300 m wide and up to 15 m deep. A no-wash zone, where RiverCats are legally required to travel at speeds less than or equal to 7 knots (15 km/h), has been established between the Silverwater Bridge and the western terminus of services at Parramatta (Figure 1). New South Wales Waterways sporadically monitors adherence to this regulation.

Sampling of assemblages of macrobenthic infauna was on intertidal riverbanks in the no-wash zone and an adjacent area east of the no-wash zone where the speed of RiverCat ferries is unrestricted (wash zone). Within each zone, two locations with unvegetated riverbanks interspersed among fragmented stands of the mangrove, *Avicennia marina,* were selected for study (no-wash zone: NW1, NW2; wash zone: W1, W2; Figure 1). All locations adjoined parkland and were separated by less than 1.5 km. Unvegetated habitat was predominantly mudflat. Sediment among the pneumatophores of mangroves constituted the vegetated habitat.

Sampling was not possible upstream of the no-wash zone as it extends to Parramatta, the western terminus of ferry services. A second no-wash zone along the Parramatta River, located further downstream where the river is much wider, was sampled but is not considered here. This was because on the lower Parramatta River, the magnitude of boat-generated waves appeared small relative to the magnitude of naturally produced waves and the assemblages were not comparable.

Sampling Methods

Benthic macrofauna were collected from unvegetated and vegetated habitats within the no-wash zone and adjacent wash zone in May 2000 (Time 1), February 2001 (Time 2), April 2001 (Time 3), and June 2001 (time 4). Sampling was not done in the second half of 2000 because of the temporary cessation of RiverCat services, the effect of which is described elsewhere (Bishop and Chapman unpublished data). At each location, two sites, approximately 15 m long and 10s of meters apart and containing unvegetated and vegetated

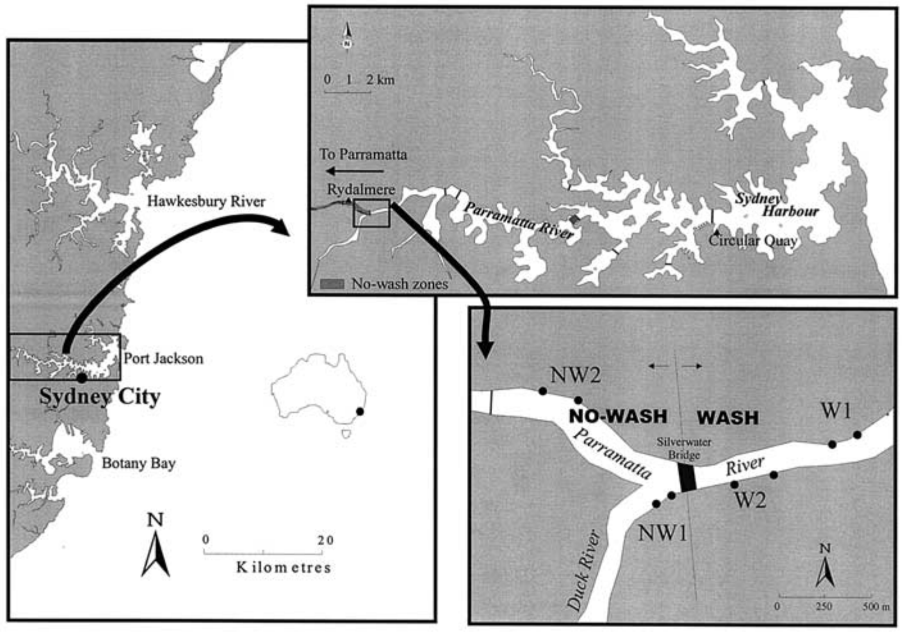

Figure 1. Map showing the location of study sites along the Parramatta River, Sydney, Australia.

habitat at a midtidal height, were selected for study. At each time, five cores of sediment, 10 cm deep and 10 cm in diameter, were randomly collected from each of the habitats, within each site. Cores were transported back to the laboratory, where they were either immediately preserved with 7% formalin in sea water, or stored overnight at 4°C and preserved the next day.

Samples were sieved through a 500-μm mesh. Polychaetes were sorted to family, crustaceans to order, and bivalves and gastropods to species. This approach shortened the time needed to process samples and is unlikely to have compromised the detection of spatial patterns in assemblages, Chapman (1998) showed that spatial patterns of variation of benthic assemblages in nearby mangrove forests were similar, irrespective of whether mixed or finer levels of taxonomic resolution were used to characterize assemblages.

Statistical Analyses

Differences between the assemblages of wash and no-wash zones were examined using nonmetric multidimensional scaling (nMDS: Shephard 1962, Kruskal 1964) of centroids (averages) for sites. nMDS uses the rank order of dissimilarity coefficients to produce a "map" of the data. The extent to which the representation agrees with the original data is reflected in its "stress." Stress should be less than 0.2 for the plot to be interpretable (Clarke 1993). The nMDS plots and the subsequently described nonparametric multivariate analyses of variance (NP-MANOVAs) used Bray-Curtis coefficients of dissimilarity (Bray and Curtis 1957). Separate nMDS ordinations using untransformed and presence–absence data were done for each time of sampling so that differences between the assemblages of wash and no-wash zones in vegetated and unvegetated habitats could be examined without temporal patterns of variability dominating plots. If differences in assemblages are primarily caused by differences in the abundances of taxa between wash and no-wash zones, differences between the wash and no-wash zone should be evident on the plots of untransformed data but not on plots of presence–absence data. Ordinations of untransformed data from all four times were also done for each habitat so that the consistency, in magnitude and direction, of any difference between wash and no-wash zones among times could be assessed.

NP-MANOVAs (Anderson 2001), which compare within-group to between-group sums of squared dissimilarities, tested for statistically significant differences in assemblages between the zones. Only two factors can be analyzed using NP-MANOVAs with clear understanding of interaction terms. Separate analyses were consequently done for each of the habitats. These used untransformed data from a single time and had two factors, zone and site. Sites could not be pooled because of significant variability at this scale. In the case of the hypothesized interaction, post-hoc tests should indicate that, within each habitat, assemblages at sites within each zone are similar. The F-distributions of null hypotheses tested using NP-MANOVA were calculated using 4999 permutations of residuals under the full model (Anderson 2001).

Table 1. Taxa of macrobenthic infauna present (*) in vegetated and unvegetated sediment of the upper Parramatta River at each of the times of sampling

			Unvegetated				Vegetated			
			t1	t2	t3	t4	t1	t2	t3	t4
Annelida	Oligochaeta		*	*	*	*	*	*	*	*
	Polychaeta	Capitellidae	*	*	*	*	*	*	*	*
		Eunicidae	*	*	*	*	*	*	*	*
		Hesionidae					*			
		Nephtyidae	*	*	*	*	*	*	*	*
		Nereididae	*	*	*	*	*	*	*	*
		Orbiniidae					*			
		Sabellidae	*			*	*			
		Spionidae	*	*	*	*	*	*	*	*
		Syllidae	*	*					*	
Crustacea		Amphipoda	*	*	*	*	*	*	*	*
		Isopoda	*	*	*	*	*	*	*	*
		Tanaidacea	*	*		*	*	*	*	*
		Brachyura								*
Mollusca	Bivalvia	*Artheritica helmsii*	*		*	*	*	*	*	*
		Laternula sp.	*		*		*	*	*	*
		Spisula trigonella	*		*	*				
		Tellina delatoidales	*	*	*		*	*		
		Xenostrobus securis	*	*	*	*	*	*	*	*
Nematoda			*			*	*	*	*	*
Nemertea			*			*	*		*	*
Other		Insect larvae	*			*	*		*	*

SIMPER (Clarke 1993) analyses identified species contributing most to multivariate differences between wash and no-wash zones across the four times of sampling and in each of the habitats. Separate ANOVAs for each time of sampling tested for habitat × zone interactions in the abundance of these taxa, which would be expected if wash reduces the abundances of taxa more in the unvegetated than the vegetated habitat. These analyses had three factors: habitat, zone, and site. The spatial scale of location was omitted from ANOVAs so that their spatial scales would be directly comparable to those of the NP-MANOVAs and because most variability was at the scale of sites. The factor time was excluded because there were no hypotheses regarding temporal change. Data were transformed using ln $(x + 1)$ to reduce heterogeneity in variances. Cochran's G-test test indicated that, in some cases, variances remained heterogeneous after transformation. These data were still analyzed because analysis of variance is relatively robust to heterogeneous variances (Box 1953, Underwood 1997). Where significant, these analyses were interpreted with caution. Analyses of variance were followed by *a posteriori* Student-Newman-Keuls tests to identify significant differences among appropriate means (Winer and others 1991, Underwood 1997).

Results

Twenty-two taxa of macroinvertebrates were identified (Table 1). Polychaetes of the families Capitellidae and Nereididae dominated assemblages. Differences between assemblages of the wash and no-wash zone were evident at each time of sampling, in each of the habitats. nMDS ordinations of untransformed data (Figure 2(i)) clearly separated sites into two groups according to their situation within the wash (filled symbols) or the no-wash (open symbols) zone. This pattern was similar in the unvegetated (black symbols) and vegetated (gray symbols) habitats. The difference between wash and no-wash zones was, however, less distinct with presence–absence data than with untransformed data at several of the times analyzed (Times 1, 4; Figure 2). In the unvegetated habitat, the pattern in untransformed data was statistically significant at Times 1 and 3 (Table 2(i)). In the vegetated habitat it was significant at Times 1–3 (Table 2(ii)).

The separation of sites into groups corresponding to wash (filled symbols) and no wash (open symbols) zones by separate nMDS ordinations for each habitat (unvegetated: Figure 3a; vegetated: Figure 3b) indicated a temporally persistent direction of difference in assemblages between wash and no-wash zones in each

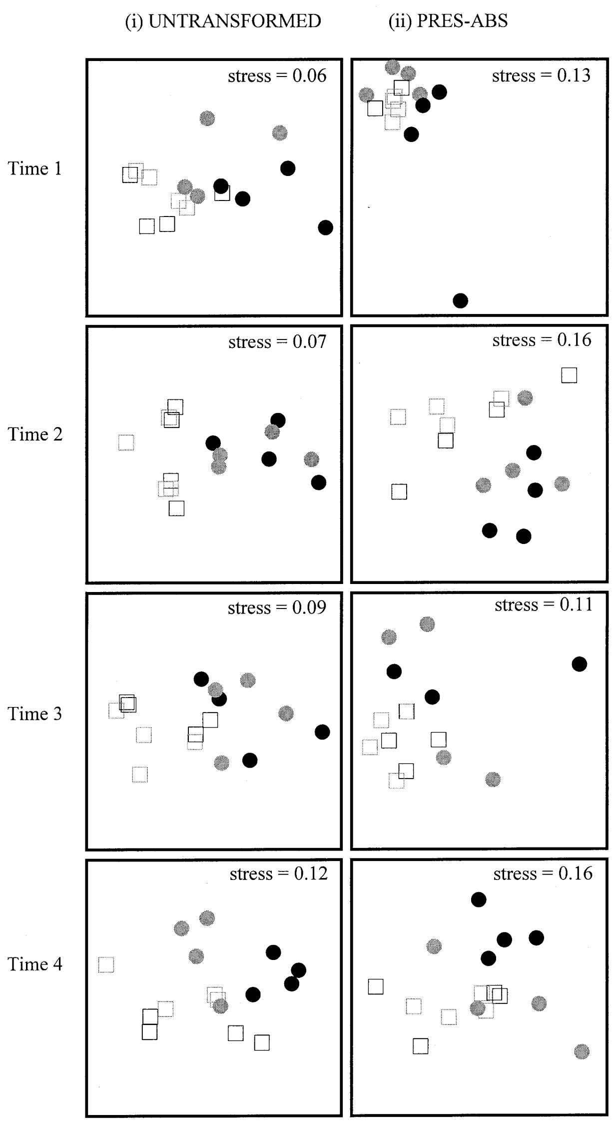

Figure 2. Nonmetric multidimensional scaling ordinations of assemblages of macrobenthic invertebrates in the wash zone (filled circles) and no-wash zone (open squares), in unvegetated (black) and vegetated (gray) habitats. Points represent centroids for the average assemblage present at each site (n = 5).

of the habitats. The difference between assemblages of the wash and no-wash zone was not, however, consistently smaller in the vegetated habitat than the unvegetated habitat, contrary to the hypothesis. Similar distances between points representing sites of the wash and no-wash zone were seen on nMDS plots (Figure 2), regardless of habitat. Moreover, Bray-Curtis measures of dissimilarity between the assemblages of the wash and no-wash zone were not consistently greater in the unvegetated habitat (black bars) than the vegetated habitat (white bars) across time (Figure 4).

SIMPER analysis identified capitellids, nereids, spionids, amphipods, and the mussel, *Xenostrobus securis,* as the primary contributors to multivariate differences between wash and no-wash zones. The abundances of these taxa at time 1 are shown in Figure 5 and are generally representative of patterns displayed at each of the times, although abundances varied considerably among times. At most times of sampling, capitellids, nereids, spionids, amphipods and *X. securis* were more abundant in the no-wash zone (white bars) than the wash zone (gray bars), in the unvegetated and the vegetated habitats. Nephytidae, when present, were only found in the wash zone (Figure 5b). At time 1, the pattern between the wash and no-wash zone was, however, only statistically significant for nereids and spionids (Table 3). The abundances of Nephtyidae were not great enough to be analyzed. In no case did patterns between the wash and no-wash zone differ between the two habitats sampled (nonsignificant habitat × zone interactions, illustrated for time 1 in Table 3).

Discussion

Previous studies report the density and diversity of assemblages in sediments to be smaller in wave-exposed environments than in sheltered habitats (e.g., Brown and McLachlan 1990, Dexter 1992). In this study, assemblages of macrobenthic infauna differed between the wash and no-wash zone of the upper Parramatta River, in unvegetated and vegetated habitats. This difference appeared to be primarily because of greater abundances of taxa in the no-wash zone than in the wash zone, evident at each of the four times of sampling. Patterns in the presence–absence data were less clear, suggesting similar compositions of assemblages and distributions of organisms among cores between the two zones.

At the majority of times of sampling, dissimilarity between assemblages of the wash and the no-wash zone was of a similar magnitude in the vegetated habitat as in the unvegetated habitat. Thus, it appears that along the Parramatta River, pneumatophores of mangroves do not reduce the impact of boat-generated waves in a way that affects infaunal assemblages. However, sampling was only done along the rivermost margin of mangroves where the density of pneumatophores is sparse. If sampling were done further back in the mangroves, where the density of pneumatophores is greater and trunks that may also aid in the dissipation wave energy are present, infaunal assemblages may be more similar between the wash and the no-wash zone. This sampling was not done because appropriate unvegetated areas, at

Table 2. Summaries of nonparametric multivariate analyses of variance comparing spatial variation in the abundance of taxa of macrobenthic infauna between wash and no-wash zones in (i) unvegetated and (ii) vegetated habitat

		Time 1		Time 2		Time 3		Time 4	
		p	Sig	p	Sig	p	Sig	p	Sig
(i) Unvegetated									
Source	*df*								
Zo	1	0.0022	**	0.0666	NS	0.0112	*	0.0540	NS
Si (Zo)	6	0.0002	***	0.0002	***	0.0002	***	0.0002	***
Res	32								
(ii) Vegetated									
Source	*df*								
Zo	1	0.0002	***	0.0020	**	0.0116	*	0.2972	NS
Si (Zo)	6	0.0002	***	0.0002	***	0.0002	***	0.0002	***
Res	32								

Zo, zone (2 levels: wash, no-wash); Si (Zo), sites four levels within each zone, random); Res, residual; n = 5. NS: $p > 0.05$ *$p < 0.05$; **$p < 0.01$; ***$p < 0.001$.

the same higher tidal height as the pneumatophores, could not be found.

Gradients in salinity and nutrients are important in determining the distribution and abundance of macrobenthic infauna (Boesch 1977, Zajac and Whitlatch 1982, de Decker and Bally 1985, Schaffner and others 1987, Castel and others 1989, Junoy and Vieitez 1990). All sites within the no-wash zone were upstream of those in the wash zone because the no-wash zone extends to the upper navigable limit of the river. Although the experimental design is technically confounded (Hurlbert 1984), differences between the wash and no-wash zone do not appear to be confounded by gradients along the river. As shown by Figure 5, in which locations are ordered in terms of their geographical location, patterns in the abundance of taxa did not follow a gradient with distance upstream.

Patterns in the abundance of taxa similarly bore no relationship to the situation of sites relative to the industrialized Duck River and the likely pattern of contaminants. Greater abundances of deposit-feeding polychaetes are typically found in places exposed to high concentrations of organic or toxic contaminants than in places where the concentration of these is smaller (e.g., Gray and others 1988, Gray and others 1990, Warwick and Clarke 1993, Braddock and others 1995). If the Duck River were a confounding influence, sites upstream of the Duck River at Rydalmere should have smaller abundances of deposit feeders than do sites further downstream, which should follow a gradient of decreasing abundance with distance from the Duck River. Sites at Rydalmere (upstream) and at Silverwater (immediately downstream of the Duck River) had similar abundances of capitellids, clearly ruling out this confounding influence. The abundance of capitellids at W2, which was adjacent to Wilson Park, a site formally occupied by the Petroleum and Chemical Corporation Australia Ltd. (1954–1974), was similar to other sites in the wash zone.

The opportunistic families of polychaetes, Capitellidae, Nereididae, and Spionidae, dominated assemblages of the upper Parramatta River. The greater abundance of these three taxa and of amphipods and *X. securis* in the no-wash than in the wash zone were primarily responsible for the difference detected between the assemblages of these two places. Predatory nephtyid polychaetes, when present, were only found in the wash zone and may have contributed to the smaller abundances of other polychaetes found there (Beukema 1987). Erosion of macrofauna from sediments can significantly influence patterns of distribution and abundance (Emerson and Grant 1991). Smaller abundances of opportunistic taxa within the wash zone may be caused by a greater erosive capacity of wash in this zone, where the current velocities associated with boat-generated waves were at a maximum.

It has been hypothesized that disturbances, such as wind stresses, that decrease the stability of the sediment, result in assemblages that are dominated by deposit feeders and in which filter feeders have proportionally small abundances (Rhoads 1974, Wildish and Peer 1983, Wildish and Kristmanson 1997). However, the preponderance of deposit feeders in assemblages of the upper Parramatta River is probably unrelated to the disturbance of sediment by RiverCat wash. Previous studies have found opportunistic deposit feeders to dominate estuarine waters (Pearson and Rosenberg 1978, Schaffner and others 1987) and muddy sedi-

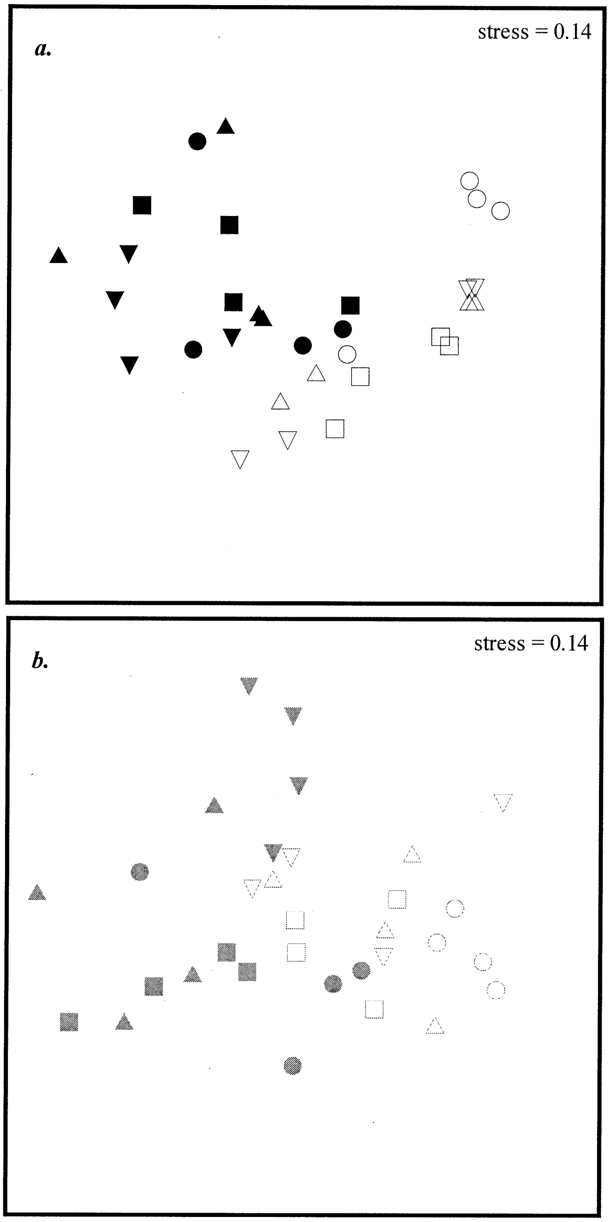

Figure 3. Nonmetric multidimensional scaling ordinations comparing assemblages in sites of the wash (filled symbols) and no-wash (open symbols) zone, in (**a**) the unvegetated (black symbols) and (**b**) the vegetated (gray symbols) habitat, among times (t1 = circles, t2 = squares, t3 = triangles, t4 = inverted triangles). Points represent centroids for the average assemblage present at each site (n = 5).

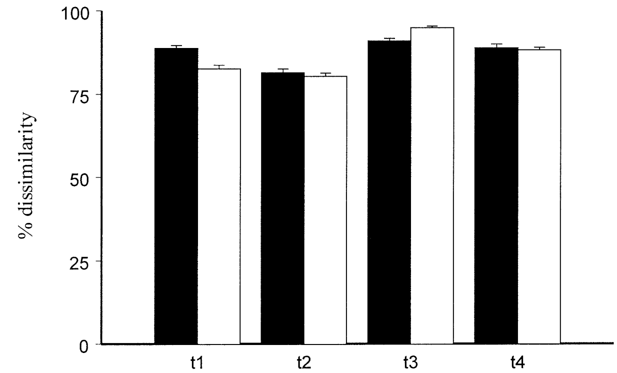

Figure 4. Mean (±SE) percent Bray-Curtis dissimilarity between assemblages of the wash and no-wash zone, in unvegetated (black) and vegetated (white) sediment, at each of the times of sampling. n = 400

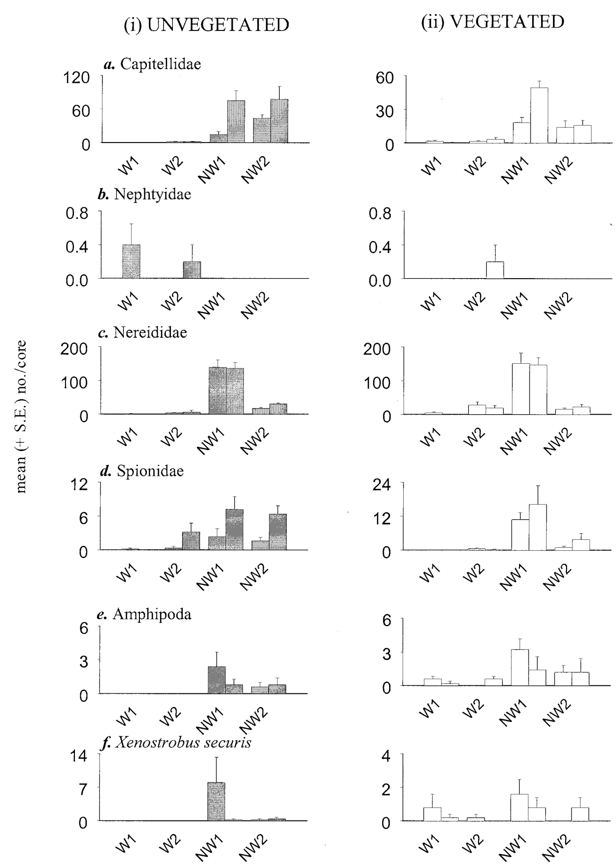

Figure 5. Mean (±S.E.) abundance of taxa of macrobenthic fauna per core in sites within locations of the wash (W) and no-wash (NW) zone, at time 1. n = 5.

ments such as those found along the upper Parramatta River (e.g., Rhoads 1974, Rhoads and Young 1970).

Through the consideration and rejection of a range of environmental variables in contributing to differences in assemblages of macroinvertebrates between wash and no-wash zone of the upper Parramatta River, this study provides strong evidence that the enforcement of the no-wash zone is an effective strategy in minimizing ecological im-

Table 3. Summaries of analyses of variance, comparing spatial variation in the abundance of taxa of macrobenthic fauna between unvegetated and vegetated habitat, and the wash and no-wash zone of the Parramatta River at time 1

		Capitellidae			Nereididae			Spionidae		
	df	MS	*F*	Sig.	MS	*F*	Sig.	MS	*F*	Sig.
Ha	1	1.01	2.28	ns	12.80	6.70	ns	0.20	0.19	ns
Zo	1	41.36	3.79	ns	184.17	23.74	**	39.65	36.05	***
Si (Zo)	6	10.91	20.35	***	7.76	17.06	***	1.10	2.64	*
Ha × Zo	1	0.41	0.93	ns	1.49	0.78	ns	2.52	2.32	ns
Ha × Si (Zo)	6	0.44	0.83	ns	1.91	4.2	**	1.08	2.60	*
Res	64	0.54			0.45			0.42		
Cochran's test		$C = 0.22$ ns			$C = 0.23$ ns			$C = 0.18$ ns		
Transformation		ln $(x + 1)$			ln $(x + 1)$			ln $(x + 1)$		
SNK					W < NW			W < NW		

		Amphipoda			*Xenostrobus securis*		
	df	MS	*F*	Sig.	MS	*F*	Sig.
Ha	1	5.05	9.05	*	0.20	0.55	ns
Zo	1	22.86	5.51	ns	8.80	5.45	ns
Si (Zo)	6	4.14	8.32	***	1.61	4.65	***
Ha × Zo	1	1.35	2.42	ns	0.05	0.13	ns
Ha × Si (Zo)	6	0.56	1.12	ns	0.37	1.06	ns
Res	64	0.50			0.35		
Cochran's test		$C = 0.46$**			$C = 0.32$**		
Transformation		ln (x + 1)			ln (x + 1)		
SNK							

Ha, habitat (two levels: vegetated [V], unvegetated [U]); Zo, zone (two levels: wash [W], no-wash [NW]); Si (Zo), site (four levels within each zone, random). SNK, student-Newman-Kevl's test; Res, residual; n = 5. NS: $p > 0.05$, *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$.

pacts. The evaluation of the success of existing managerial strategies in meeting their goals is essential for the continuing improvement of policies and practices. This is increasingly recognized by managers, as seen by the large number of policies that contain at least some passing reference to the need for an adaptive approach (Walters 1997). Nevertheless, adaptive planning rarely proceeds beyond the initial stages of model development to field experimentation because of factors such as (1) the cost associated with large-scale assessments or (2) the absence of data from before the implementation of the strategy and/or appropriate unmanaged control locations (Walters 1997), which are perceived as essential to ecological evaluations.

Although properly controlled manipulative experiments, where sampling is done before and after the implementation of a managerial strategy, are most informative in assessing the effect of a decision on plants and animals (Underwood 1995), *a posteriori* assessments of a managerial decision with clearly defined hypotheses can still be of use in improving predictions of their effect. For example, if greater abundances of macroinvertebrates were consistently found in no-wash zones than in unmanaged waterways, a logical prediction might be that the establishment of a no-wash zone would increase abundances of macroinvertebrates. Nevertheless, the possibility remains that abundances of macroinvertebrates are greater in the types of place managers chose for the establishment of no-wash zones because of some other factor. It is therefore recommended that large-scale mensurative experiments such as the present one be coupled with smaller-scale controlled manipulations in order to unambiguously attribute an effect of wash on macroinvertebrates. This approach has been used by Underwood (1998, Underwood (1999) in a situation where it was impossible to directly manipulate a large-scale disturbance, but where small-scale experiments, on their own, provided little information on the disturbance.

Acknowledgements

This study was supported by funds from the Australian Research Council through the Special Centre for Research on Ecological Impacts of Coastal Cities and an Australian Postgraduate Award to the author. This manuscript benefited greatly from the comments of M. G. Chapman, M. B. Single, and an anonymous reviewer.

Literature Cited

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

Asplund, T., and C. Cook. 1999. Can no-wake zones effectively protect littoral zone habitat from boating disturbance. *Lakeline* 13:1–12.

Bell, S. S., M. O. Hall, S. Soffian, and K. Madley. 2002. Assessing the impact of boat propeller scars on shrimp and fish utilizing seagrass beds. *Ecological Applications* 12:206–217.

Bennett, E. L., and C. J. Reynolds. 1993. The value of a mangrove area in Sarawak. *Biodiversity and Conservation* 2:359–375.

Beukema, J. J. 1987. Influence of the predatory polychaete *Nephtys hombergii* on the abundance of other polychaetes. *Marine Ecology Progress Series* 40:95–107.

Bhowmik, N. G., M. Demissie, C. Y. Guo. 1982. Waves generated by river traffic and wind on the Illinois and Mississipi Rivers. WRC research report No. 167. University of Illinois Water Resources Center, Urbana, Illinois.

Boesch, D. F. 1977. A new look at the zonation of benthos along the estuarine gradient. Pages 245–66 *in* B. C. Coull (ed.), Ecology of marine benthos. University of South Carolina Press, Columbia, South Carolina.

Box, G. E. P. 1953. Non-normality and tests on variances. *Biometrika* 40:318–335.

Braddock, J. F., J. E. Lindstrom, and E. J. Brown. 1995. Distribution of hydrocarbon-degrading microorganisms in sediments from Prince William Sound, Alaska following the *Exxon Valdez* oil spill. *Marine Pollution Bulletin* 30:125–132.

Bray, J. R., and J. T. Curtis. 1957. An ordination of the upland forest communities of Southern Wisconsin. *Ecological Monographs* 27:325–349.

Brown, A. C., and A. McLachlan. 1990. Ecology of sandy shores. Elsevier, Amsterdam.

Bryan, G. W., P. E. Gibbs, L. G. Hummerstone, and G. R. Burt. 1986. The decline of the gastropod *Nucella lapillus* around south-west England: Evidence for the effect of tributyltin from antifouling paints. *Journal of the Marine Biological Association of the United Kingdom* 66:611–640.

Castel, J., P.-J. Labourg, V. Escaravage, I. Auby, and M. E. Garcia. 1989. Influence of seagrass beds and oyster parks on the abundance and biomass pattern of meio- and macrobenthos in tidal flats. *Estuarine, Coastal and Shelf Science* 28:71–85.

Chapman, M. G. 1998. Relationships between spatial patterns of benthic assemblages in a mangrove forest using different levels of taxonomic resolution. *Marine Ecology Progress Series* 162:71–78.

Clarke, K. R. 1993. Non-metric multivariate analyses of change in community structure. *Australian Journal of Ecology* 18:117–143.

Commito, J. A., C. A. Currier, L. R. Kane, K. A. Reinsel, and I. M. Ulm. 1995a. Dispersal dynamics of the bivalve *Gemma gemma* in a patchy environment. *Ecological Monographs* 65:1–20.

Commito, J. A., S. F. Thrush, R. D. Pridmore, J. E. Hewitt, and V. J. Cummings. 1995b. Dispersal dynamics in a wind-driven benthic system. *Limnology and Oceanography* 40:1513–1518.

Connell, S. D., and T. M. Glasby. 1999. Do urban structures influence local abundances and diversity of subtidal epibiota? A case study from Sydney Harbour, Australia. *Marine and Environmental Research* 47:373–387.

Davies, I. M., and S. K. Bailey. 1991. The impact of tributyltin from large vessels on the dogwhelk *Nucella lapillus* (L.) populations around Scottish oil ports. *Marine and Environmental Research* 32:202–211.

Davis, G. E. 1977. Anchor damage to a coral reef on the coast of Florida. *Biological Conservation* 11:29–34.

Dayton, P. K. 1975. Experimental evaluation of ecological dominance in a rocky intertidal algal community. *Ecological Monographs* 45:137–159.

de Decker, H. P., and R. Bally. 1985. The benthic macrofauna of the Bot River estuary, South Africa, with a note on its meiofauna. *Transactions of the Royal Society of South Africa* 45:379–396.

Department of Transport. 1995. Integrated transport strategy for the metropolitan region. Department of Transport Sydney.

Dexter, D. M. 1992. Sandy beach community structure: the role of exposure and latitude. *Journal of Biogeography* 19:59–66.

Dobbs, F. C., and J. M. Vozarik. 1983. Immediate effects of a storm on coastal infauna. *Marine Ecology Progress Series* 11:273–279.

Eagle, R. A. 1975. Natural fluctuations in a soft bottom benthic community. *Journal of the Marine Biology Association of the United Kingdom* 55:865–878.

Emerson, C. W., and J. Grant. 1991. The control of the soft-shell clam (*Mya arenaria*) recruitment on intertidal sandflats by bedload sediment transport. *Limnology and Oceanography* 36:1288–1300.

Glasby, T. M., and S. D. Connell. 2001. Orientation and position of substrata have large effects on epibiotic assemblages. *Marine Ecology Progress Series* 214:127–135.

Gray, J. S., M. Aschan, M. R. Carr, K. R. Clarke, R. H. Green, T. H. Pearson, R. Rosenberg, and R. M. Warwick. 1988. Analysis of community attributes of the benthic macrofauna of Frierfjord/Langesundfjord and in a mesocosm experiment. *Marine Ecology Progress Series* 46:151–165.

Gray, J. S., K. R. Clarke, R. M. Warwick, and G. Hobbs. 1990. Detection of initial effects of pollution on marine benthos: An example from the Ekofisk and Eldfisk oilfields, North Sea. *Marine Ecology Progress Series* 66:285–299.

Hastings, K., P. Hesp, and G. A. Kendrick. 1995. Seagrass loss associated with boat moorings at Rottnest Island, Western Australia. *Ocean and Coastal Management* 26:225–246.

Hay, D. 1968. Ship waves in navigable waterways. Pages 1472–1487 *in* Proceedings of the Eleventh Coastal Engineering Conference. American Society of Civil Engineers, New York.

Hurlbert, S. J. 1984. Pseudoreplication and the design of ecological field experiments. *Ecological Monographs* 54:187–211.

Junoy, J, and J. M. Vieitez. 1990. Macrozoobenthic community structure in the Ria de Foz, an intertidal estuary (Galicia, Northwest Spain). *Marine Biology* 107:329–339.

Kirk, R. M., and M. B. Single. 2000. Coastal impacts of new forms of transport: The case of the interisland fast ferries. Pages 00–00 *in* P. A. Memon, and H. Perkins. (eds.), Environmental planning and management in New Zealand. Dunmore Press, Palmerston North, New Zealand.

Kobayashi, N., A. W. Raichle, and T. Asano. 1992. Wave attenuation by vegetation. *Journal of Waterway, Port, Coastal and Ocean Engineering, ASCE* 119:30–48.

Kruskal, J. B. 1964. Multidimensional scaling by optimizing goodness of fit to a nonmetric hypothesis. *Psychometrika* 29:1–27.

Massel, S. R., K. Furukawa, and R. M. Brinkman. 1999. Surface wave propagation in mangrove forests. *Fluid Dynamics Research* 24:219–249.

McCall, P. L. 1977. Commuity patterns and adaptive strategies of the infaunal benthos of Long Island Sound. *Journal of Marine Research* 35:221–266.

McCall, P. L. 1978. Spatial-temporal distributions of Long Island Sound infauna: the role of bottom disturbance in a nearshore habitat. Pages 191–219 *in* M. L. Wiley (ed.), Estuarine Interactions. Proceedings of the Fourth International Estuarine Research Conference. Academic Press, New York.

McQuaid, C. D., and G. M. Branch. 1984. Influence of sea temperature, substratum and wave exposure on rocky intertidal communities: An analysis of faunal and floral biomass. *Marine Ecology Progress Series* 19:145–151.

Menge, B. A. 1978. Predation intensity in a rocky intertidal community: Effect of an algal canopy, wave action and desiccation on predator feeding rates. *Oecologia* 34:17–35.

Nanson, G. C., A. von Krusenstierna, and E. A. Bryant. 1994. Experimental measurements of river-bank erosion caused by boast generated waves on the Gordon River, Tasmania. *Regulated Rivers: Research and Management* 9:1–14.

Orth, R. J. 1975. Destruction of eelgrass, *Zostera marina,* by the cownose ray, *Rhinoptera bonasus,* in the Chesapeake Bay. *Chesapeake Science* 16:205–208.

Orth, R. J. 1977. The importance of sediment stability in seagrass communities. Pages 281–300 *in* B. C. Coull (ed.), Ecology of marine benthos. University of South Carolina Press, Columbia, South Carolina.

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

Posey, M., W. Lindberg, T. Alphin, and F. Vose. 1996. Influence of storm disturbance on an offshore benthic community. *Bulletin of Marine Science* 59:523–529.

Rhoads, D. C. 1974. Organism-sediment relations on the muddy sea floor. *Oceanography and Marine Biology: an Annual Review* 12:263–300.

Rhoads, D. C., and D. K. Young. 1970. The influence of deposit-feeding organisms on sediment stability and community trophic structure. *Journal of Marine Research* 28:150–178.

Schaffner, L. C., R. J. Diaz, C. R. Olsen, and I. L. Larsen. 1987. Faunal characteristics and sediment accumulation processes in the James River Estuary, Virginia. *Estuarine, Coastal and Shelf Science* 25:211–226.

Scholer, H. A. 1974. Hawkesbury River; the effects of speedboat activities on river banks, Appendix 3. Hawkesbury River report on effects of water skiing, Public Works Department of New South Wales, Sydney.

Shephard, R. N. 1962. The analysis of proximities: Multidimensional scaling with an unknown distance function. *Psychometrika* 27:125–140.

Uhrin, A. V., and J. G. Holmquist. 2003. Effects of propeller scarring on macrofaunal use of seagrass *Thalassia testudinum. Marine Ecology Progress Series* 12:206–217.

Underwood, A. J., and P. Jernakoff. 1984. The effects of tidal height, wave exposure, seasonality and rock-pools on grazing and the distribution of microalgae in New South Wales. *Journal of Experimenta Marine Biology and Ecology* 75:71–96.

Underwood, A. J. 1995. Ecological research and (and research into) environmental management. *Ecological Applications* 5:232–247.

Underwood, A. J. 1997. Experiments in ecology: Their logical design and interpretation using analysis of variance. Cambridge University Press, Cambridge.

Underwood, A. J. 1998. Grazing and disturbance: an experimental analysis of patchiness in recovery from a severe storm by the intertidal alga *Hormasira banksii* on rocky shores in New South Wales. *Journal of Experimental Marine Biology and Ecology* 231:291–306.

Underwood, A. J. 1999. Physical disturbances and their direct effect on an indirect effect: responses of an intertidal assemblage to a severe storm. *Journal of Experimental Marine Biology and Ecology* 232:125–140.

Walker, D. I., R. J. Lukatelich, G. Bastyan, and A. J. McComb. 1989. Effect of moorings on seagrass beds near Perth, Western Australia. *Aquatic Botany* 36:69–77.

Walters, C. 1997. Challenges in adaptive management of riparian and coastal ecosystems. *Conservation Ecology* 1:1 http://www.consecol.org/vol1/iss2/art1.

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

Wildish, D. J., and D. D. Kristmanson. 1997. Benthic suspension feeders and flow. Cambridge University Press, Cambridge.

Wildish, D. J., and D. Peer. 1983. Tidal current speed and production of benthic macrofauna in the lower Bay of Fundy. *Canadian Journal of Fisheries and Aquatic Sciences* 40:309–21.

Winer, B. J., D. R. Brown, and K. M. Michels. 1991. Statistical principles in experimental design. 3rd edition. McGraw-Hill, New York.

Yeo, R. K., and M. J. Risk. 1979. Intertidal catastrophes: Effects of storms and hurricanes on intertidal benthos of the Minas Basin, Bay of Fundy. *Journal of the Fisheries Research Board Canada* 36:667–669.

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