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Loss of native rocky reef biodiversity in Australian metropolitan embayments

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ABSTRACT

Urbanisation of the coastal zone represents a key threat to marine biodiversity, including rocky reef communities which often possess disproportionate ecological, recreational and commercial importance. The nature and magnitude of local urban impacts on reef biodiversity near three Australian capital cities were quantified using visual census methods. The most impacted reefs in urbanised embayments were consistently characterised by smaller, faster growing species, reduced fish biomass and richness, and reduced mobile invertebrate abundance and richness. Reef faunal distribution varied significantly with heavy metals, local population density, and proximity to city ports, while native fish and invertebrate communities were most depauperate in locations where invasive species were abundant. Our study adds impetus for improved urban planning and pollution management practises, while also highlighting the potential for skilled volunteers to improve the tracking of changes in marine biodiversity values and the effectiveness of management intervention.

1. Introduction

Of numerous contemporary threats to global marine biodiversity, pollution and disturbance associated with coastal urbanisation are consistently regarded amongst the most serious and widespread (Suchanek, 1994; Gray, 1997; Vitousek et al., 1997; Nystrom et al., 2000; Shahidul Islam and Tanaka, 2004; Halpern et al., 2008). In a global assessment of threats based on a quantitative expert interview approach, Halpern et al. (2007) listed coastal development, point source organic pollution and direct human impacts amongst the eight greatest threats to biodiversity across all marine ecosystems. Only increasing sea temperature and fishing-related impacts were considered to be more pervasive in global oceans. In line with this, Crain et al. (2009) stressed the need to better understand the cumulative impacts on our coastal ecosystems through community-level field studies. Such studies can provide the only means to quantify overall net effects on marine ecosystems without making assumptions regarding the nature of interactions, and are needed to inform and complement controlled experiments designed to explore mechanistic links.

Field studies of community-level impacts of urbanisation on sub-tidal marine fauna have mostly focussed on soft sediment habitats (Reish, 1955; Heck, 1976; Inglis and Kross, 2000; Claudet and Fraschetti, 2010), or on sessile components of hard substrates (Johnston and Roberts, 2009). Sub-tidal rocky reef communities make up a substantial component of faunal biomass in coastal areas, and are often of greater recreational and commercial importance than soft sediment communities, typically containing high densities of large-bodied fishes and mobile invertebrates (Edgar, 1990; Taylor, 1998; Cowles et al., 2009). Relatively little is known about the community-level impacts of urbanisation on mobile fauna associated with rocky reefs, including the extent to which such values are compromised under multiple, interacting threats.

Common local responses to organic and inorganic pollution observed in soft-sediment and sessile faunal communities are shifts in the abundance distribution of species towards an increasingly uneven community dominated by few species (Johnston and Roberts, 2009), and corresponding changes in the relative proportions of species with different tolerances to disturbances, feeding

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modes or life-history characteristics (Reish, 1955; Heck, 1976; Pearson and Rosenberg, 1978; Warwick, 1986; Schaaf et al., 1987; Claudet and Fraschetti, 2010; Edgar et al., 2010). Few studies have assessed pollution impacts on marine fishes at the community level, but from those included in the meta-analysis of McKinley and Johnston (2010), positive responses in overall abundance and species richness to organic enrichment were the only relatively consistent trends identified. The clearest message apparent from previous research is that community-level responses to urbanisation may be complex and unpredictable, affected by varying tolerances of different species to numerous contaminants and sources of disturbance. Ecological interactions and indirect effects of urbanisation through habitat degradation will further contribute to variable outcomes at this level of organisation.

The major goal of our study was to document the distributions of fishes and mobile invertebrates on rocky reefs throughout three major urban embayments in south-eastern Australia in order to understand how they vary spatially with the distribution of urban impact types. Our study focused on the three state capitals: Sydney (New South Wales), Melbourne (Victoria) and Hobart (Tasmania). These cities have major ports and industry, and substantial known heavy metal pollution as a legacy from historical industrial pollution and through contemporary inputs such as storm water runoff and discharges from urbanised sub-catchments (Birch, 2000; Johnston and Keough, 2002; Townsend and Seen, 2012). All contain areas of fringing rocky reef with temperate faunas (although Sydney also receives seasonal recruitment of tropical species) and a mix of algal dominated habitat and bare rock/urchin barrens. Invasive species are also known to be common in these cities, mostly introduced as a result of intense shipping activity, so we also considered the pressures associated with invasive species alongside urban impacts.

Although much has been assumed from broader biogeographic trends, surprisingly little is known of the distribution of rocky reef biodiversity associated with these cities; prior to this study, no systematic study of rocky reef biodiversity had ever been undertaken across Sydney Harbour, despite being the location of the first European settlement in Australia and the site of its largest city. Our approach involved training and engaging committed local recreational SCUBA divers in each of the cities through the global Reef Life Survey program (RLS; www.reeflifesurvey.com) to enable a comprehensive coverage of collection of data, as well as establishing a cost-effective mechanism for ongoing monitoring at these cities using standardised methods through the future.

We tested the hypotheses that: (a) the community structure of fishes and mobile invertebrates recorded at shallow reef sites by RLS divers is generally related to the distribution of a number of urban impacts, including heavy metal contamination, surrounding human population density, the proximity to sewage outfalls, proximity to the city port, and the distribution of invasive species; and (b) spatial patterns in impacts are consistent among different taxonomic groups, impact types and the three cities examined, despite biogeographic differences in species composition and physical characteristics. We then assessed trends in important univariate community metrics to better understand the nature of impacts, specifically in relation to expectations from previous research associated with loss of species, reduced productivity, and compositional differences related to life-history strategies.

2. Methods

2.1. Ecological data

Underwater visual census methods were used to estimate densities of fishes and mobile macroinvertebrates at sub-tidal reef sites distributed throughout Port Phillip Bay (Melbourne), Sydney Harbour (Sydney) and the Derwent Estuary (Hobart). Surveys were undertaken using standard RLS methods, which involve separate surveys of fishes and mobile macroinvertebrates along 50 m transect lines. Detailed descriptions of methods are provided in Edgar and Stuart-Smith (2014) and an online methods manual (Reef Life Survey, 2013). Multiple 50 m transect lines were set at each site, each along a depth contour. The fish surveys involved a pair of divers swimming either side of the transect line, while recording on waterproof paper the abundance and size of all fishes sighted within 5 m of the line. Abundances of fishes in large schools were estimated by counting a subset and estimating the percentage of the total school that the subset comprised.

Mobile macroinvertebrates (echinoderms, large gastropods and large crustaceans >2.5 cm length) and cryptic fishes, closely associated with the bottom and often missed on larger-scale fish censuses, were surveyed in 1 m wide blocks on either side of the same transect lines used for fish counts. Divers undertook this component immediately following completion of the fish survey. The algal canopy was brushed aside where necessary to search all exposed surfaces of the substratum within the block, with counts made for each species sighted. Only data on native species were included as response variables in analyses, with invasive species recorded, but excluded from data used in response variables.

Data on fish abundance and size were used to estimate the biomass of each species on transects. Species-specific length–weight relationships provided in Fishbase (www.fishbase.org) were applied, with relationships from congeners (and occasionally family) used if not available for particular species. Additional Fishbase relationships were used to convert total length to fork length as necessary. The bias in divers' perception of fish size underwater was corrected using relationships presented in Edgar et al. (2004). Fish biomass estimates, in grams per 500 m^2 transect, were log (x + 100 g) transformed for all analyses (although raw data in kg are presented in plots). The estimates can be regarded as relative, suitable only for comparisons with data collected using the same methods, rather than providing absolute estimates of biomass.

Data were analysed from 35 sites in Port Phillip Bay, 27 in Sydney Harbour and 37 in the Derwent Estuary (Hobart) (Fig. 1). Surveys were undertaken between November and May (>70% between December and February) over three summer periods from 2008 to 2011. An average of two 50 m transects surveyed at each site was analysed, after transects deeper than 10 m were excluded. Data used were means among transects within sites (overall mean depth was 4.3 m), averaging out any depth-associated variation, which is relatively small for the depth range and regions covered in this study. Little reef habitat exists deeper than 10 m in the three embayments other than near the Sydney Heads.

Thirty-six RLS divers participated in data collection; all with training to a scientific standard in survey methods, as evidenced by comparison with data from scientists who accompanied divers on the same transect blocks on previous surveys. Previous assessment of data quality from trained RLS volunteers found the differences to data produced by professional biologists non-significant and also trivial (<1%) when compared to variation attributable to depth (over a greater range than in this study), site and region (Edgar and Stuart-Smith, 2009).

2.2. Urban impact and environmental variables

A range of local urban impact and pollution data were obtained and aligned with the ecological survey sites where fish and invertebrate data were collected. These included local heavy metal pollution, invasive species densities, proximity to sewage treatment plant outfalls, and local human population densities.

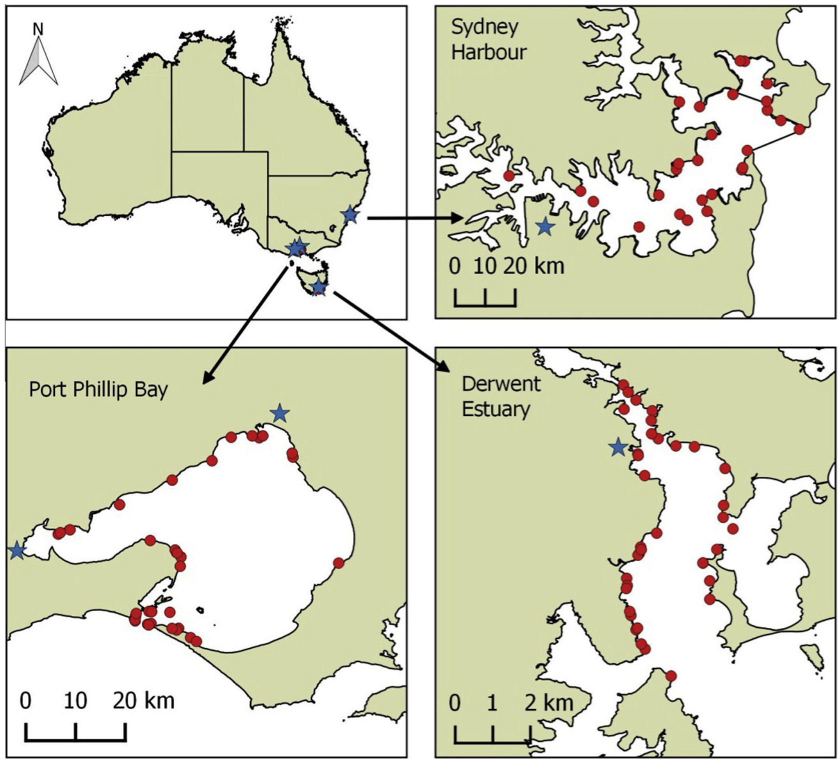

Fig. 1. Maps of study embayments and sites in south eastern Australia (circles). Stars represent locations of city ports and central business districts.

Heavy metal data were included in different forms for each city based on data availability. In Hobart, total heavy metal measurements (mg/kg) were available for sediment samples, while in Melbourne they were based on water column samples (total metals in μg/L). Importantly, they were from the same source, collected using the same method, within each location. Values attributed to each ecological survey site were those from the closest heavy metal measurement site, or interpolated from adjacent heavy metal measurement sites, and were log-transformed for analyses. In Sydney Harbour, heavy metal measurements were not available for all sites surveyed, so we applied a three-level categorical index, depending on whether sites lay within regions known to have high ($n = 3$), medium ($n = 23$) or low ($n = 6$) heavy metal loadings (Birch and Taylor, 1999). Site distances from sewage outfalls were measured in Google Earth, with log-transformed distances used in analyses as a proxy for exposure to organic pollutants from sewage outfalls.

A fine-scale local population density index was also derived using the glp00 g gridded world population density dataset (http://sedac.ciesin.columbia.edu/data/collection/gpw-v3/sets/browse) and fitting a quadratic Kernel function, as described in Silverman (1986). Values did not directly represent local population values since they were both modelled (quadratic) and smoothed. However, they provide a comparative index of population density/pressure at a fine scale, which is consistent within and between locations studied. Population index values were log-transformed for Sydney analyses, due to much greater range in values across sites in this city. Site distances from the city ports for each of the three cities was also measured in Google Earth (distance to the nearest of Melbourne and Geelong ports were used for Port Phillip Bay sites). As a result of port position within each embayment, this measure can be considered a proxy for a combination of physical and chemical disturbance arising from intense shipping activity, pollution from major industry, and run-off from the CBD and industrial areas. Note that higher values represent greater distances from the ports and therefore lower impact values.

The percentage of the summed abundance of benthic fishes and mobile invertebrates recorded on surveys that were composed of invasive species was used in analyses for Hobart sites only. Mobile invasive species were common on reef surveys in Hobart, and included nine species of benthic fishes, crabs, sea stars and mollusc (at mean total density of 210 per 50 m^2), but were less common in Melbourne (four species, mean 0.8 per 50 m^2 at three sites only) and absent from reef surveys in Sydney. This study did not include sessile invasive species.

To allow pollution effects to be investigated relative to those associated with natural gradients in environmental conditions throughout the three systems, wave exposure and salinity data were also obtained for ecological survey sites. A fetch model was used to estimate wave exposure at each ecological survey site (Hill et al., 2010), which estimated the distance according to the proximity to land masses in a number of directions as a proxy for exposure to wind-generated waves. This index confirmed expected strong exposure gradients from the heads inland in the Derwent and Sydney Harbour, but not Port Phillip Bay. The exposure index values were square root transformed for analyses. Salinity data were not available for Sydney, and showed extremely little variation among the reef sites in both Melbourne and Hobart, and were therefore not used in analyses. All sites were located in the outer, marine sections of these embayments, and mean salinity at the surface and at depth was equivalent to seawater across the broader region (ca. 35 ppt) with little variation over seasonal scales (Whitehead et al., 2010). While very short duration low salinity pulses are likely at some sites following heavy rainfall, the magnitude and frequency of such events at the depths of reef transects are unlikely to be consequential in comparison to the effects of differences in wave exposure, for example. Although mobile in location, a salt wedge and high bottom salinity persist during heavy rainfall events in the Derwent (Whitehead et al., 2010), which is where the effect of salinity would be expected to be most likely to influence reef fauna among the three study locations.

Numerous strong correlations were evident between exposure and urban impact variables, as well as between urban impact variables (Table 1). This was unavoidable given that urbanisation and associated point sources of pollution and disturbance tend to be most intense in the same parts of these systems, typically in the more sheltered regions. We proceeded with analyses ensuring: (1) exposure was considered first in all statistical models, providing conservative estimates of remaining variation associated with the urban impact variables; and (2) instead of reducing the number of inter-correlated urban impact variables examined and losing important information relevant for assessing cumulative impacts, all variables could be included in step-wise analyses, and interpreted in combination when high correlations prevented examination of impacts separately.

2.3. Data analyses

Multivariate analyses were based on biomass for the fish community and abundance for the cryptic benthic fishes and mobile invertebrates. The latter two groups were combined for multivariate tests as they were surveyed within the same blocks (collectively referred to as 'mobile benthic fauna', MBF), but mobile invertebrates were treated separately for univariate tests (e.g. for species richness – see below).

Detailed life history information is lacking for the majority of species covered by our surveys, so we used the vulnerability index of Cheung et al. (2005), provided in Fishbase, which uses a combination of life history traits to characterise the continuum from large, late-maturing fish species to small, quickly-maturing species. This index is based on a fuzzy logic, 'if, then' approach, and while at least maximum size is available and would have been incorporated into values of the vulnerability index for all our species, it is unknown which further life history traits were incorporated, and for which species. Keeping this limitation in mind, we applied the vulnerability index in preference to species' maximum sizes on the assumption that it captures additional life history information for many species. Only a modest correlation was evident in our analyses between vulnerability index values for fish species with published maximum sizes ($r = 0.69$), and therefore we consider this a reasonable assumption. While this index was originally developed as a proxy for species' intrinsic vulnerabilities to fishing pressure, we consider it also appropriate for generally distinguishing small, faster growing, *r*-selected species from large, long-lived, *k*-selected species. Further to this, we hypothesise a trend for an increase in vulnerability values (larger, slower species) further from urban impact sources, which would be in the opposite direction to an effect of fishing impacts in these locations (few fishers extract fishes from locations close to city centres due to high pollution levels, and thus fishing pressure is expected to be greatest on reef species near the heads). This index is not available for invertebrate species, and thus we used published maximum size values for these (Edgar, 2008).

Community weighted means (CWMs) were used to calculate transect level estimates of the fish vulnerability index and invertebrate maximum size. Weighting was based on abundance data, and therefore represent values per individual on each transect, with multiple transect values averaged per site. Mean species richness for fishes and invertebrates, total biomass of fishes and total abundance of invertebrates per transect were also used in the univariate analyses.

Local faunal community structure and univariate indices were analysed in separate DISTLMs using the PERMANOVA+ extension in Primer (Anderson et al., 2008). These were based on Bray–Curtis dissimilarity for multivariate community data and Euclidean distance for univariate measures. Due to the typically strong correlations between urban impact variables and exposure, exposure was added first to DISTLM models as a forced inclusion. The urban impact variables were then added in a forward step-wise procedure using adjusted R^2 as the selection criterion, with *p*-values calculated using 9999 permutations.

3. Results

3.1. Patterns in reef faunal composition

Overall, the surveys analysed covered 148, 84 and 71 fish, and 50, 71 and 62 invertebrate taxa in Sydney Harbour, Port Phillip Bay and the Derwent Estuary, respectively. The distribution of invasive species in the Derwent Estuary was closely associated with the distances from the Hobart city port (Fig. 2; Table 1). All individuals recorded on the mobile benthic fauna survey were invasive species at some sites within 2 km of the Hobart port, while this dropped to less than 20% of mobile invertebrates and cryptic fishes closer to the heads (approximately 20 km from the port).

Table 1
Pearson correlations between urban impact variables and exposure in the three cities analysed. SO = sewage outfall. NA = not assessed. Correlations >0.7 are in bold.

Location	Variable	Exposure	Population index	Invasive species	Distance from port	Distance from SO
Hobart						
	Exposure					
	Population index	−**0.71**				
	Invasive species	−**0.72**	**0.86**			
	Distance from port	**0.73**	−**0.91**	−**0.83**		
	Distance from SO	−0.12	−0.36	−0.18	0.29	
	Heavy metals	−**0.83**	**0.85**	**0.89**	−**0.81**	−0.10
Melbourne						
	Exposure					
	Population index	−0.22				
	Distance from port	0.44	−0.62			
	Distance from SO	0.32	−0.38	NA	0.35	
	Heavy metals	−0.31	0.14	NA	−0.15	−**0.73**
Sydney						
	Exposure					
	Population index	−0.43				
	Distance from port	0.51	−**0.76**	NA		
	Heavy metals	−**0.71**	0.57	NA	-0.63	NA

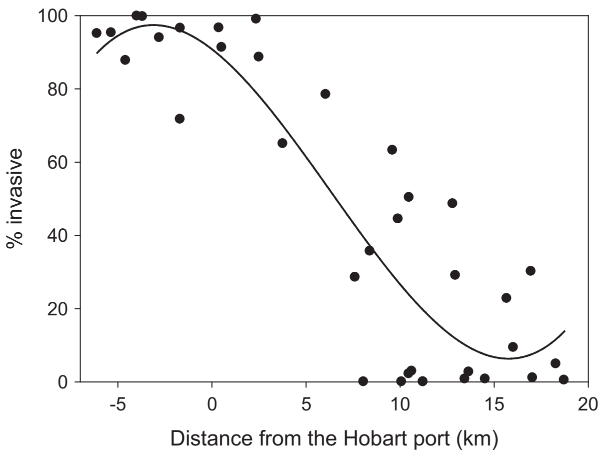

Fig. 2. Distribution of invasive invertebrates and benthic fishes in the Derwent Estuary in relation to the Hobart city port, with 3rd order polynomial fitted curve as smoother. *Y*-axis denotes the percentage of invasive individuals of all mobile macroinvertebrate and small benthic fish species surveyed in multiple 50×1 m transect blocks at sites throughout the estuary (symbols are means of transect values at each site). Negative distances extend upstream of the port.

Table 2
DISTLM results for multivariate community structure data for fishes and mobile benthic fauna (MBF) in the Derwent Estuary (Hobart), Port Phillip Bay (Melbourne) and Sydney Harbour. Numbers represent the % variation explained when added to DISTLM models in step-wise procedure, with significance at $p = 0.05$ (identified by asterisk) determined using 9999 permutations. Cells without values denote situations where a variable was not added to the model in the step-wise procedure. Exposure was a forced inclusion in all models. Urban impact variables with correlations >0.7 are considered together, with variation described by these summed (indicated by parenthesis). NA = not assessed.

	Sydney		Melbourne		Hobart	
	fishes	MBF	fishes	MBF	fishes	MBF
Exposure	9.5*	15.7*	4.4	10.5*	25.4*	14*
City port	6.1	18.6*	3.1	7.9*	13.3*	19.8*
Population index				3.1		
Invasive species	NA	NA	NA	NA		
Heavy metals	5.2	3.2	6.7	9.2*		
Distance to SO	NA	NA			3.1	2.8
Total variation	*20.9*	*37.7*	*14.2*	*30.7*	*41.8*	*36.6*

Multivariate community structure differed considerably throughout the embayments in all three cities, driven in part by trends in wave exposure, which accounted for between 4.4% and 25.4% of variation in community dissimilarities between sites (Table 2). Urban impact variables explained significant proportions of additional variation in the data for all three cities, after accounting for exposure (and therefore the shared variation with this variable). Population density and distance from the port of Sydney (the two were correlated; Table 1) and heavy metals and distance from the sewage outfalls in Port Phillip Bay (also correlated) explained similar amounts of variation as exposure. The combination of invasive species, heavy metal pollution, population density and distance from the port explained 13.3% and 19.8% of additional variation in the fishes and mobile benthic fauna in the Derwent, despite all being highly correlated with exposure. The magnitude of such relationships is thus highly significant when considering that shared variation with exposure has already been excluded by the forced inclusion of exposure in all models.

Separate DISTLM models for univariate community metrics also revealed the significance of urban impact variables in explaining biodiversity trends at all three cities. The fish vulnerability index and mean invertebrate maximum size were generally positively associated with distance from the city ports and negatively associated with heavy metals (Figs. 3 and 4; Table 3), with the exception of invertebrate maximum size in Melbourne. These results describe a trend for smaller species of fish and invertebrate in the most heavily polluted regions of these cities, while similar patterns were observed for species richness of fishes and invertebrates, total fish biomass and total invertebrate abundance, all being lower in the most heavily impacted areas. The exceptions were in Port Phillip Bay where high densities of the sea urchin *Heliocidaris erythrogramma*, a relatively large invertebrate, occurred at more polluted sites.

The distance of sites from sewage outfalls added significantly to models for Port Phillip Bay fauna, but was also highly correlated with heavy metal pollution in this location. The combination of distance from sewage outfalls and level of heavy metal pollution explained considerably more variation in fish species richness and biomass than did exposure, however, and a substantial proportion of variation in invertebrate abundance. Lower fish species richness and biomass, but higher invertebrate abundance, was evident in communities near sewage outfalls in Port Phillip Bay.

4. Discussion

Despite a conservative approach to inference from correlated variables, particularly the high degree of overlap in the spatial patterns in wave exposure and urban impact variables, this study identified significant patterns of variation in mobile reef faunas associated with the cumulative effects of anthropogenic impacts. The community structure of fishes and mobile benthic fauna at sites in the most heavily impacted areas at all three major cities investigated was altered, characterised by fewer fish and invertebrate species and reduced total fish biomass. The most heavily impacted areas also tended to be characterised by smaller species, a statistically significant result in the Derwent Estuary and Sydney Harbour after accounting for variation explained by exposure, but non-significant in Port Phillip Bay.

These results generally contrast with the findings of a recent meta-analysis of pollution impacts on marine fishes (McKinley and Johnston, 2010), which found little evidence of consistent effects of most contaminant types on fish richness or abundance, although positive effects of nutrient enrichment on richness and abundance were sometimes evident. Instead, the trends identified in our study more closely match those seen in sessile marine invertebrate communities (Johnston and Roberts, 2009), where contamination consistently results in significant reductions in species richness.

Impacts of heavy metal pollution were difficult to isolate in our study, nevertheless urban reef fish and invertebrate communities were apparently negatively affected by this pollutant type. All three locations studied have received substantial heavy metal inputs, with loss of larger macroalgal species in sections of the Derwent Estuary most polluted by heavy metals (Fowles, unpub data). The mechanisms and pathways through which heavy metals affect large and mobile reef species are not clear, but re-suspension of adjacent soft sediments (Hill et al., 2013) and metals in solution from storm water run-off and industrial discharge ensure heavy metal contamination reaches reef habitats.

Impacts of heavy metals on mobile reef fauna are possibly indirectly mediated through ecological interactions. Negative effects of pollutants on biogenic habitat such as the larger canopy-forming kelps present in these regions may well result in a reduction in the number of species and individuals observed on rocky reefs, as has been noted in studies of other pollutants and habitat types (Deegan et al., 2002; McKinley and Johnston, 2010) or related to other causes for loss of some of the same kelp species (Ling, 2008). However, it is also possible that observed patterns may partially reflect outcomes of direct impacts on the growth, fecundity

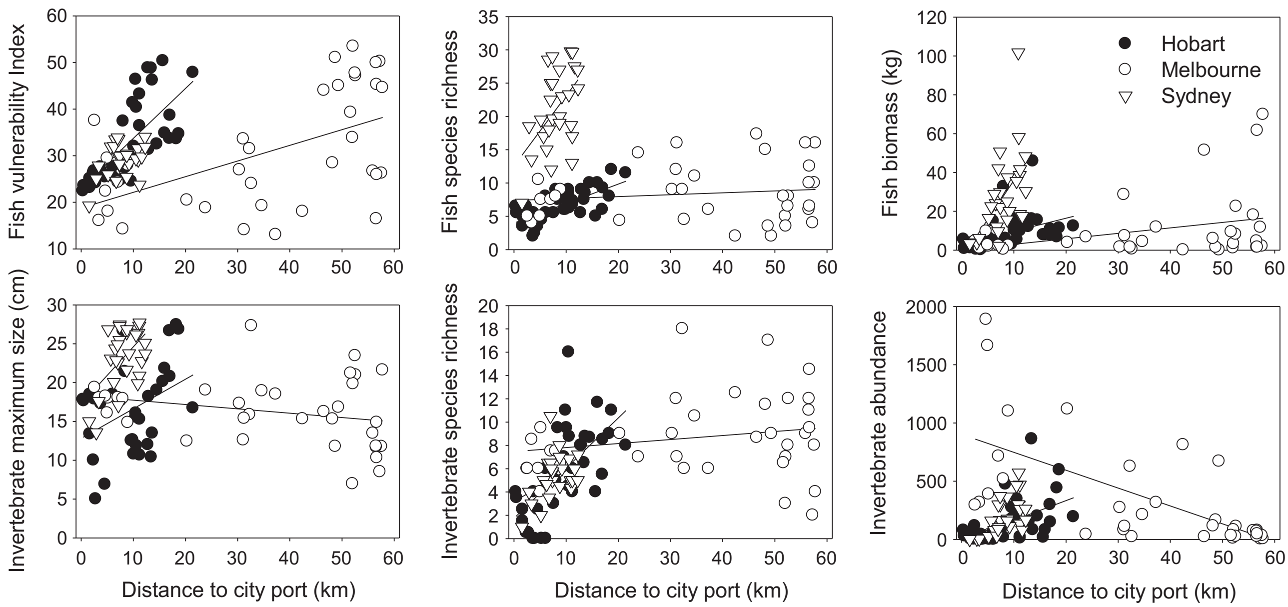

Fig. 3. Univariate measures of the fish and mobile invertebrate communities surveyed at sites versus distance from city ports in the Derwent Estuary (Hobart), Port Phillip Bay (Melbourne) and Sydney Harbour. Fish vulnerability index and invertebrate maximum size are community-weighted means (mean value across individuals surveyed), species richness and biomass are means per 500 m^2 for fishes and 100 m^2 for invertebrates. Note that the effects of wave exposure, strongly negatively correlated with distance to the city port in Hobart, have not been accounted for in these plots – see Table 3 for significance of results after accounting for these.

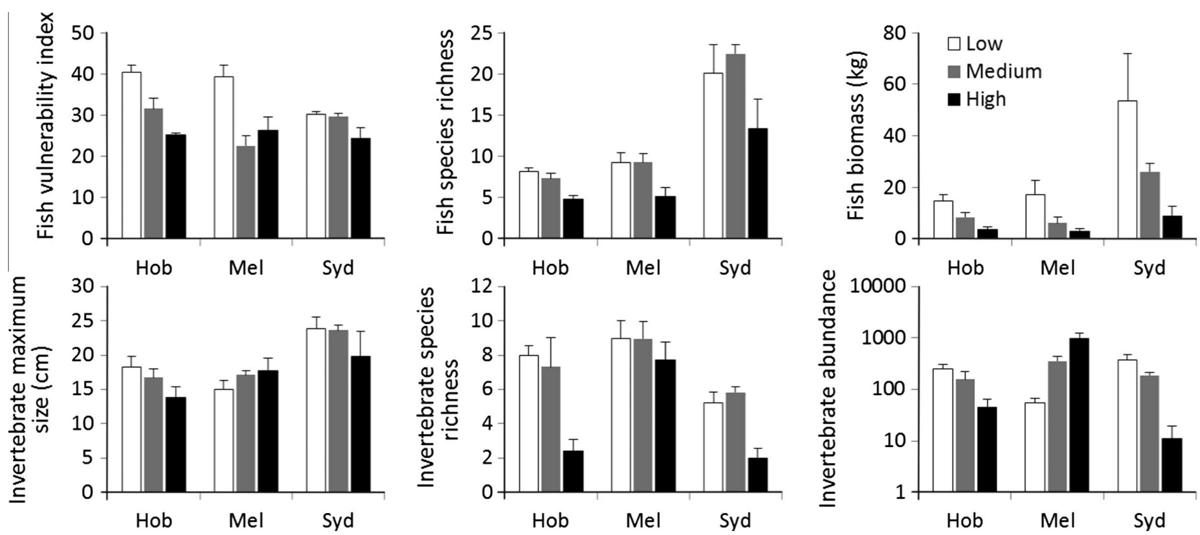

Fig. 4. Univariate measures of the fish and mobile invertebrate communities surveyed at sites in regions of differing heavy metal pollution in the Derwent Estuary (Hob), Port Phillip Bay (Mel) and Sydney Harbour (Syd). Fish vulnerability index and invertebrate maximum size are community-weighted means (mean value across individuals surveyed), species richness and biomass are means per 500 m^2 for fishes and 100 m^2 for invertebrates (+SE). Note heavy metal pollution data were from different sources for each location and are in different units, with categorical values relative to location. The categorical index for Sydney is described in text, while for graphical representation here, sites in Hobart were categorised as low (<7000 mg/kg), medium (7000–25,000 mg/kg), high (>25,000 mg/kg) and in Melbourne as low (<15 μg/L), medium (15–25 μg/L), and high (>25 μg/L). Invertebrate abundance *y*-axis is on log scale. Note that the effects of wave exposure, strongly negatively correlated with heavy metal pollution in Hobart and Sydney, have not been accounted for in these plots – see Table 3 for significance of results after accounting for these.

and performance of some species. Further investigation, including manipulative studies aimed at disentangling correlative relationships observed in this study, is needed to help shed light on indirect effects of heavy metal pollution, particularly through the mechanism of biogenic habitat degradation.

Invasive species were considered a form of urban impact in these analyses, rather than as a response to other urban impacts. Invasive species are considered a serious threat to marine biodiversity in their own right (Crain et al., 2009), and our results could imply that they may have an enormous deleterious impact on native reef fauna, including reducing species richness, when present in the high densities we recorded in the Derwent Estuary. Given the high correlations between the proportion of invasive species on surveys in Hobart and the other urban impact variables, however, as well as conclusions from studies of sessile invasive species (Piola and Johnston, 2008; Dafforn et al., 2009), it is equally likely that invasive species fill niches made vacant through loss of native species by urbanisation.

In most cases, invasive species are likely to have been introduced through shipping vectors and thus have first arrived at the port (Bax et al., 2003). Subsequent spread throughout suitable local habitat is then controlled in part by propagule pressure (Hedge and

Table 3
DISTLM results for univariate metrics of fish and mobile invertebrate communities in the Derwent Estuary (Hobart), Port Phillip Bay (Melbourne) and Sydney Harbour. Numbers represent the % variation explained when added to DISTLM models in step-wise procedure, with significance at $p = 0.05$ (identified by asterisk) determined from 9999 permutations. Cells without values indicate that the variable was not added to the model in the step-wise procedure. Exposure was a forced inclusion in all models. Urban impact variables with correlations >0.7 are considered together, with variation described by these summed (indicated by brackets). NA = not assessed.

	Sydney		Melbourne		Hobart	
	fishes	Inverts	fishes	Inverts	fishes	Inverts
Vulnerability index (fishes)/maximum size (invertebrates)						
Exposure	6.8	4.6	23.0*	2.3	59.8*	16.6*
City port	39.6*	34.6*	5.9		10.3*	
Population index						
Invasive species	NA	NA	NA	NA		
Heavy metals	4.5		3.2	9.5		
Distance to SO	NA	NA				
Total variation	*50.9*	*39.2*	*32.1*	*11.8*	*70.1*	*16.6*
Species richness						
Exposure	0.1	2.2	3.4	1.3	30.3*	31.9*
City port	43.4*	41.8*	5.5	8.6	25.9*	37.3*
Population index						
Invasive species	NA	NA	NA	NA		
Heavy metals		7.2	33.7*			
Distance to SO	NA	NA			3.0	
Total variation	*43.5*	*51.2*	*42.6*	*9.9*	*59.0*	*69.3*
Biomass (fishes)/abundance (invertebrates)						
Exposure	28.1*	22.4*	9.3	22.1*	27.9*	32.2*
City port	20.7*	49.5*	4.5	24.2*	33.5*	32.5*
Population index						
Invasive species	NA	NA	NA	NA		
Heavy metals		3.3	27.8*	17.3*		
Distance to SO	NA	NA				2
Total variation	*48.8*	*75.2*		*63.7*	*61.5*	*66.7*

Johnston, 2012) and ecological interactions. All invasive species observed in this study are mobile, with larval export that extends well beyond the Derwent Estuary for many species (e.g. *Asterias amurensis*; Morris, 2002; Ling et al., 2012). Thus, the distribution of invasive species on reefs noted in this study (and reduced presence outside of the Derwent Estuary; Stuart-Smith et al., 2010) are unlikely to be due to higher propagule pressure close to the port, or to reflect complete competitive superiority of invasive species. Instead, it seems more likely that they may be capitalising on empty niches left by historical pollution, rather than actively displacing intact native communities. Such a mechanism may also help explain the extreme prevalence of mobile invasive reef species in the Derwent compared to Port Phillip Bay and Sydney Harbour; heavy metal pollution in the Derwent is amongst the greatest known worldwide (Whitehead et al., 2010). In 1977, the Derwent was considered the second most polluted estuary in the world for mercury contamination, after Minamata Bay in Japan (Bloom and Ayling, 1977).

Moreover, analysis of dated sediment cores indicate mass local extinctions of native invertebrates over the past century (Edgar and Samson, 2004). Thus, it is possible that native communities on rocky reefs in Port Phillip Bay and Sydney Harbour have not been impacted by pollution severely enough to allow as significant establishment of invasive species in rocky reef habitats as has occurred in the Derwent Estuary. Reefs in Port Phillip Bay and Sydney Harbour also contain higher densities and biomass of mobile fauna (i.e. predators and competitors), and potentially less niche space, when compared to the Derwent Estuary (or soft sediment communities within these systems). The abundance of sessile invasive species on manmade structures and soft sediment habitats in both these cities is nevertheless very high (Hewitt et al., 1999; Dafforn et al., 2012), a trend that also supports the hypothesis that invasive species are capitalising on empty (or at least partially unfilled) niches.

The trend in the community weighted means of life history characteristics, with communities in more polluted areas being comprised of smaller species, is consistent with results of studies on soft sediment faunas (Grassle and Grassle, 1974; Warwick, 1986, 1988). These correspond with expectations based on traditional r/k selection and life-history theories, in which small, faster-growing species may be first to colonise after major disturbances, and also respond more quickly to regular disturbances.

Such trends in life history and size cannot be interpreted without considering exploitation, however. This is particularly important for fishes, but also for some invertebrates. Two commercially important taxa, the southern lobster (*Jasus edwardsii*) and black-lip abalone (*Haliotis rubra*), were present in Melbourne and Hobart and are amongst the larger of the invertebrates studied. Fishing selectively removes larger-bodied species and has a greater impact on populations of species which are late to mature. Indeed, the vulnerability index used was initially developed to measure fishing impacts (Cheung et al., 2005).

Nevertheless, the trend for decreasing vulnerability (i.e. smaller, faster species) in the most heavily impacted areas, was opposite in direction to predictions associated with fishing impacts, consistent with our rationale for using this metric for this study. Based on this and the general consistency in trends between locations in this study, the mean vulnerability index across individuals in the community appears to be a potentially useful indicator of urban impacts on fish communities. However, application for this purpose requires more explicit understanding of the relative importance of fishing versus pollution impacts, and their interaction, on the behaviour of this index across a broader range of locations. A modelling study by Schaaf et al. (1987) predicted different life-histories to be critical in determining interactions between exploited fish stocks and pollution impacts, while the theory of vulnerability, as described by Cheung et al. (2005) when proposing the vulnerability index, should apply to any major form of disturbance.

Trends in the abundances of the dominant sea urchin species greatly affected results for invertebrates in our study. The higher abundance of the sea urchin *H. erythrogramma* in more heavily polluted areas in Port Phillip Bay was the major reason for high overall invertebrate abundance in such areas, a trend opposite to that observed in the other cities. Patterns of urchin abundance also underlie the observed trend in invertebrate maximum size, albeit with the large, apparently pollution-tolerant, sea star *Coscinasterias muricata*, and large grazing gastropod *H. rubra*, also important contributors. Interestingly, *H. erythrogramma* was also the dominant urchin species in Hobart, but showed the opposite trends in this location. *H. erythrogramma* is apparently a pollution-tolerant species. While historical pollution and/or reduced algal productivity in the more polluted regions of Hobart was possibly more severe than in Port Phillip Bay and restricted local urchin abundance throughout the Derwent Estuary (as hypothesised in relation to invasive species distribution), other environmental or ecological factors not assessed here may control urchin abundance patterns in the Derwent Estuary. Regardless, observed trends in invertebrate species richness are not affected by these location-specific patterns in urchin abundance, and support the conclusions that reef invertebrate communities are heavily modified by urban pollution.

Impacts of sewage treatment plant outfalls were difficult to properly assess in this study. Proximity to sewage outfalls was not included in analyses of Sydney reef fauna, given that treated sewage is piped outside of the Harbour, and there was no way of accounting for the impacts of known frequent overflow and illegal discharges on reef fauna and flora within the Harbour. In the Derwent Estuary, eight sites were surveyed within 1 km, and 28 sites between 1 and 3 km from the nearest of five sewage outfalls, but any signals were potentially swamped by the substantial gradient in other pressures in this location. Proximity to sewage outfalls was added as a variable in some DISTLM models for the Derwent, but was usually added last and its contribution to variation was non-significant. Outfalls from fewer, larger sewage plants are present in Port Phillip Bay, and although there were some patterns in our data that suggested these have noticeable impacts on reef fauna, these could not be distinguished from potential heavy metal impacts.

Amongst the results for Port Phillip Bay that possibly relate to sewage effects were positive responses in abundance of abalone *H. rubra* and *H. erythrogramma*. Both are herbivorous invertebrates that may capitalise on increased algal growth fuelled by nutrients from sewage outfalls. In the case of *H. rubra*, a large commercially-exploited gastropod, this trend may be accentuated by lower fishing pressure on this species in locations close to sewage outfalls and with relatively high heavy metal pollution.

As with most broad-scale spatial analyses, a number of potentially important factors were not investigated in our study. These include habitat complexity, which is known to have a strong influence on the structure of both fish and invertebrate communities (Gratwicke and Speight, 2005). We make the assumption that results were not confounded by substrate complexity interacting with pollution effects in these three embayments. While this assumption is reasonable in relation to the rock substrate itself, it is likely violated when biogenic habitat created by macroalgal and sessile invertebrate communities are considered. Biogenic habitats are also likely influenced by the same pollution gradients, as described above, and a similar analysis of the distribution of sessile community structure through these systems is needed.

Although many of our results may be anticipated given the history of pollution in these three cities, and outcomes of studies of soft sediment and sessile communities, evidence for the nature and scale of such impacts has been lacking. Loss of biodiversity translates to losses in the economic, recreational and amenity value of the marine environment near our major metropolitan cities (Smith et al., 2008). Clearly, the need exists to improve management to reduce inputs of contaminants to the marine environment. The most heavily polluted systems support higher abundance of invasive communities, and our results suggest that maintaining intact and resilient native communities is probably fundamental to reducing the spread of invasive species.

Just as measureable goals and effective partnerships between local governments and industry need to be established to reduce pollution inputs to the marine environment, equivalent partnerships and goals are also needed to measure local biodiversity impacts on a less ad-hoc basis than has been achieved to date. Our study, in which local teams of committed citizen scientists in the Reef Life Survey program allow ongoing quantification of urban impacts on rocky reef habitats, provides a model that can be applied to other urban marine habitats and taxonomic groups (e.g. inter-tidal rock platforms, sub-tidal seagrass, sea birds and mammals), and for monitoring impact types (e.g. water quality, debris) (Hammerton et al., 2012). A large and committed workforce exists in our major cities, only requiring effective coordination and collaboration with local government and scientific representatives to monitor these areas. Although not suitable for the most heavily polluted locations, a network of trained and coordinated citizen scientists provides cost-effective opportunities to determine whether current and future pollution reduction strategies are detectably improving local marine biodiversity in urbanised environments, while additionally generating benefits in improved public knowledge of the nature and severity of impacts.

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