

MEASURING CHANGE IN THE SANDY BEACH ENVIRONMENT

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Abstract

Globally, coastal environments are at risk from the multiple effects of climate change and urbanisation. Both the recession of sandy beaches and the strategies of management put in place to mitigate it have the potential to disrupt the ecology of sandy beaches, and their important ecosystem services. Nevertheless, the threat this poses to meiofauna, the small invertebrates which underpin sandy beach food webs, is presently difficult to assess because: (1) the extent of beach management practices are not documented and (2) the scales at which meiofauna naturally vary, spatially and temporally, remain poorly understood. This thesis sought to redress these deficiencies. Through a survey of beach managers and a literature search, the extent of nourishment practices in Australia was assessed. On a global scale, most Australian projects were small but frequent. Most projects occurred in urbanised areas as a response to extreme weather, with little monitoring of efficacy or biological impact. A series of small scale surveys assessed the spatial and temporal scales of variation in sandy beach meiofauna. Variation was strongly correlated with elevation, sediment water content and granulometry. Significant natural variation was found between adjacent days, years and between geomorphically similar sections of adjacent beaches. Finally a case study was conducted to assess whether coastal lagoon dredging – part of beach dredge and fill project, impacted meiobenthic communities. An impact was detected, sites within the dredged lagoon recorded changes which were not observed in undisturbed lagoons. Sites within the dredged lagoon also showed greater spatio-temporal variability than those in undisturbed lagoons. Overall, this thesis adds to growing evidence that the coastal environment is facing increasing pressures. In order to assess their impact on important sandy beach ecosystems, rigorous experimental designs are needed to detect environmental impacts over significant background spatial and temporal variation.

Statement of candidate

I certify that the work in this thesis entitled “**Measuring Change in the Sandy Beach Environment**” has not previously been submitted for a degree nor has it been submitted as part of requirements for a degree to any other university or institution other than Macquarie University.

I also certify that the thesis is an original piece of research and it has been written by me. Any help and assistance that I have received in my research work and the preparation of the thesis itself have been appropriately acknowledged.

In addition, I certify that all information sources and literature used are indicated in the thesis.

The research presented in this thesis was approved by Macquarie University Ethics Review Committee, reference number: **5201100286** on **18/05/2011**.

Signature

Belinda Claire Cooke

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23/01/2013

Date

Contributors

This thesis contains material that has been submitted or prepared for publication as follows:

Chapter 1: General introduction

I completed the literature review and wrote this chapter. My supervisor Dr. Melanie Bishop provided constructive feedback.

Chapter 2: Nourishment practices on Australian sandy beaches: A review

Accepted by the Journal of Environmental Management.

My contribution to the research paper: concept 60%, data collection 100%, analysis 100%, writing 85%.
Total 86.25%.

Dr. Alan Jones provided the original concept for the research. I received constructive help in the analysis and writing from Alan Jones and Melanie Bishop. Dr. Ian Goodwin provided important suggestions and constructive feedback. I presented information from this paper at three venues: Climate Futures Conference Macquarie University 2010; International Sandy Beach Symposium, Mpikweni, South Africa (Winner: Student prize, third place) 2012; and at the Sandy Beach Workshop, Sydney University 2012.

Chapter 3: Small scale distribution of benthic invertebrates on an exposed sandy beach, North Carolina

My contribution to the research paper: concept 50%, data collection 100%, analysis 65%, writing 90%.
Total 76.25%.

Dr. Stephen Fegley provided the concept for the experimental design and collected the samples in the field with me. Dr. Georgy Sofranov provided initial statistical advice and some initial statistical analysis. I received editorial assistance with the writing from Melanie Bishop.

Chapter 4: Spatial scales of variation in meiofauna on three embayed beaches, Sydney, Australia

This paper has been submitted to a sandy beaches special edition of Estuarine and Coastal Shelf Science.

My contribution to the research paper: concept 100%, data collection 100%, analysis 100%, writing 90%.
Total 97.5%.

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Chapter 5: Spatio-temporal variation in meiofaunal communities on an urbanised exposed sandy beach, New South Wales, Australia

My contribution to the research paper: concept 100%, data collection 90%, analysis 100%, writing 85%.
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Chapter 6: The impact of entrance dredging on the meiofaunal communities of an intermittently open lagoon in south-eastern Australia

My contribution to the research paper: concept 100%, data collection 100%, analysis 100%, writing 85%.
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Chapter 7: General conclusion

I completed the literature review and writing. My supervisor Dr. Melanie Bishop provided constructive feedback.

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1. General introduction

1.1 The extent and value of sandy beaches

Sandy beaches comprise about 50-60% of the Earth's coastlines (Bird 1996) and are of high socio-economic value. They are the sites of some of the most valuable residential property, and are highly valued for their recreational and aesthetic amenity (Batley and Cocks 1992, Blackwell 2003), supporting activities such as swimming, fishing and surfing. Furthermore, natural coastal environments such as sandy beaches provide a protective barrier to property and infrastructure against the effects of storms and flood inundation (Clark 1991).

Although often mis-labelled as biological deserts, sandy beaches are also of high ecological value (Defeo et al. 2009). They support more phyla than the world's rainforests (Higgins and Thiel 1988, Ray and Grassle 1991, Giere 2009). Protozoans, microalgae and meiofauna (those metazoans that pass through a 500 μm sieve but are retained on meshes of 40–64 μm ; Giere 2009) live in the interstitial environment between sand grains. These organisms are important in remineralising detritus, the dead organic matter which washes up on beaches (Coull 1999). Larger macrobenthic invertebrates (macrofauna; generally defined as those > 500 μm), including crustaceans, molluscs and polychaete worms, actively burrow and ride the swash, feeding on phytoplankton, organic matter, or the interstitial biodiversity (Dugan et al. 2000). Meiofaunal densities may exceed 10 million individuals per square metre of sediment surface (Kennedy and Jacoby 1999); and macrofauna, densities of 100,000 per square metre (Defeo et al. 2009). These dense and diverse assemblages are food for surf fishes and shorebirds, and are critical in carbon and nutrient cycling (Brown and McLachlan 1990). In this capacity invertebrate fauna support the greater marine ecosystem and in turn provide services to fisheries, pollution mitigation and ecosystem stability (Hobday et al. 2006).

1.2 Threats to beaches

Sandy beaches and their important ecological and socio-economic values are increasingly being threatened by a range of stressors (Defeo et al. 2009). These include climate change, coastal development and engineering, recreation (e.g. use of off-road vehicles), pollutants, biological invasion and

mining (Defeo et al. 2009). Among these, the loss of habitat and associated biota caused by accelerated beach erosion is considered to be the most immediate and severe ecological threat to beaches (Schlacher et al. 2008, Defeo et al. 2009). Beaches are constantly evolving, and retreat or prograde at different stages in their evolution (Woodroffe 2003). Climatic or anthropogenic pressures can, however, accelerate and modify this behaviour at local, regional and landscape scales (Woodroffe 2003). Some coastal geomorphologists suggest that direct anthropogenic effects may result in comparable or even greater modifications than climate-change impacts (Valiela 2006, Nicholls et al. 2007).

Coastal development undermines the natural resilience of beaches by constraining landward shoreline movement (Nordstrom 2000) and interfering with coastal sediment transport processes (Komar 1998). Sea-level rise and increased storm activity are typically stated to be a threat to the beach ecosystem (Defeo et al. 2009); however, these only become a threat to beaches when coupled with urban development or hard engineering structures to create the 'coastal squeeze effect' (Doody 2001). In the absence of shoreline development and engineering, sand loss of the beach from storm scour is usually only temporary (Finkl and Walker 2004). Although in many cases the sand removed from the beach is not lost from the system, narrowing of the beach width does threaten coastal development. Higher levels of coastal urbanisation increase pressure to manage the coastline to protect infrastructure from the threat of inundation (James 2000, Cooper and Lemckert 2012).

1.3 Beach management

The strategies that are being used to protect public beach amenity and protect adjacent properties from beach erosion vary according to the density of coastal development. In areas which have little development more passive approaches to managing eroding shorelines may be implemented, such as allowing the shoreline to alter and recede naturally. In areas which have a large revenue base and valuable coastal infrastructure, planned retreat still applies; however, coastal engineering is typically used to delay damage to infrastructure. Hard structures, such as revetments, seawalls, bulkheads, groynes and breakwalls, were used extensively in the past to protect coastal properties from erosion and flooding, but have been found to lead to loss of beaches and influence sediment supply to down-drift beaches, often negatively (Hall and Pilkey 1991, Hanson et al. 2002, Finkl and Walker 2004). More recently, there has

been a global trend towards the use of soft engineering to manage beach erosion. Soft engineering involves extending the beach seaward through nourishment (or 'fill') with sediments or through use of items (e.g. geotextile bags) that may be modified or removed depending on conditions (Jackson and Hornsey 2002, Gourley et al. 2004). Soft engineering is now the preferred option over hard engineering in policy and legislation internationally (NOAA 2000, Finkl and Walker 2004, UNDEP 2010), including in Australia (e.g. Department of Environment and Heritage 2005).

1.4 Impacts of beach management on sandy beach fauna, and the assessment of these impacts

Despite the assumption that soft engineering is ecologically benign, recent reviews have drawn attention to the impacts beach nourishment practices can have on sandy beach fauna (Nordstrom 2005, Speybroek et al. 2006). Nevertheless, despite thousands of beach nourishment projects around the world, impacts to fauna remain poorly understood. The monitoring programs associated with European beach nourishment projects generally include analysis of beach profiles to determine erosion rates of the fill material, but biological monitoring is usually not included (Cooney et al. 2003). In the USA, biological monitoring at both dredge and fill sites is commonly a permit requirement (Peterson and Bishop 2005); but these are frequently of poor design, rendering little useful information for adaptive management (reviewed in Peterson and Bishop 2005). In Australia, there is a lack of baseline data on the general state of much of the coasts and oceans (Australia State of the Environment Committee 2001). Monitoring the impacts of soft engineering on sandy beaches would help to ensure that the management methods used are ecologically sustainable; whilst also making the best possible use of resources in a limited sand supply with a limited budget. If biological monitoring is well designed it can provide an integrated measure of pressures and of environmental change at a variety of spatial and temporal scales (Fairweather 1990a).

Where monitoring has been done, macrofauna and shorebirds have often been the target variables assessed (e.g. Peterson and Bishop 2005, Peterson et al. 2006, Department of Climate Change 2009, VanDusen et al. 2012). In addition, much of the ecological monitoring of ecological impacts has focused on single species rather than community data (Peterson and Bishop 2005). Macrofauna and shorebirds are generally more visible than meiofauna and other interstitial biota and therefore more easily counted. In many cases, however, they do not fulfil requirements of good indicators (Ward and Jacoby 1992). Not only

should good indicators be readily measurable, but they should also be representative of the ecosystem they are a component of, responsive to change in ways that are well understood, and widely distributed across both impact and potential control areas (Ward and Jacoby 1992).

On exposed sandy beach environments in Australia macrofauna and shorebirds are often not present, not abundant enough to allow statistics to be undertaken, or are present with very low diversity. Using single species as response variables to pressures may give an ambiguous response which is not indicative of the overall environmental health of the sandy beach environment; and may not take into account natural increases due to spawning events or intermediate disturbance (Balmford et al. 2003). Single species studies are also limited in their scope to detect ecological effects mediated by community interactions (Wernberg et al. 2012). As a response indicator should "be capable of providing a continuous assessment over a wide range of stress" (Noss 1990), it is likely that assemblages of multiple species should provide more information of biotic responses to disturbance (Kremen 1992).

1.5 Meiofauna as indicators of change

Although seldom considered in impact assessments of beach nourishment activities (e.g. Speybroeck et al. 2006, Schlacher and Noriega 2007, Defeo et al. 2009), meiofauna may be a good alternative or addition to use of macrofauna and shorebirds (Hicks 1991). Meiofauna are functionally important to the coastal ecosystem as they contribute a large proportion of benthic production (Kuipers et al. 1981, Platt and Warwick 1983, Raffaelli and Hawkins 1996). Meiofauna facilitate biomineralization of organic matter and enhance nutrient regeneration (reviewed in Coull 1999), play a disproportionately large role in carbon cycling (Heip et al. 1985), make detritus available to higher order consumers (Tenore et al. 1977) and are an important food source for higher order consumers (reviewed in Coull 1999). Hence, changes in the distribution, diversity and abundance of meiofauna have the potential to propagate impacts up and down the food webs of sandy beaches. Meiofauna have a short generation time, ensuring a fast response to environmental change (Coull 1999). The majority of meiofauna are direct developers, meaning that there is no physiologically separate dispersal stage in their life cycle (Remane 1952). As they spend their entire life cycle in the interstitial environment they are particularly susceptible to changes in the sediment (Swedmark 1964, Dye 1978, Coull and Chandler 1992, Leduc et al. 2010).

Meiofauna have proven to be useful bio-indicators of physical and chemical disturbance (Moore and Bett 1989, Sandulli and Giudici 1989, Schratzberger et al. 2002, Schratzberger and Jennings 2002, Lampadariou et al. 2005, Dye 2006, Liu et al. 2009, Huff 2011, Leduc and Pilditch 2013). In coastal ecosystems, they have been used as indicators for the ecological impacts of dredging and trawling (Boyd et al. 2000, Schratzberger et al. 2002, Schratzberger and Jennings 2002, Boyd et al. 2003, Schratzberger et al. 2006), installation of bulk heads (Spalding and Jackson 2001), beach cleaning (Gheskiere et al. 2006) as well as shoreline nourishment and sediment disposal (Schratzberger et al. 2000a, 2000b, Schratzberger et al. 2004). Among meiofaunal taxa, several studies have found nematodes to be particularly useful in the detection of ecological impacts (Warwick and Buchanan 1970, Herman and Heip 1988). Nematodes have been found to persist during environmental disturbance, enabling accurate monitoring of changes to their assemblage (e.g. Hendelberg and Jensen 1993). Nematodes may reveal more complex changes due to their higher diversity, decreased dispersal ability, and the fact that they play a functional role in all trophic levels with a variety of feeding modes: suspension, filter, deposit, herbivory, detritivory, omnivory and predation (Moens and Vincx 1997, Bongers and Bongers 1998).

The disadvantages of using meiofauna as indicators of ecological impacts include their small size, high levels of spatial and temporal variability, potential costs of sample processing and the difficulty of identification (Kennedy and Jacoby 1999). While identification to species level will provide the most complete information regarding response to environmental variation (Coull and Palmer 1984, Coull and Feller 1988); a high proportion of meiofaunal species remain undescribed (Kennedy and Jacoby 1999). Given the high diversity of meiofauna, even at the scale of phyla, a large degree of taxonomic resolution may not be necessary (Warwick and Gee 1984, Warwick 1988, Clarke and Warwick 2001). An understanding of the background scales of variation in meiofauna and how these relate to the physical environment (e.g. Heip et al. 1985, Coull and Chandler 1992) would help ensure that sampling designs using meiofauna as indicators for ecological impact appropriately account for this background variation (see Frascchetti et al. 2006).

1.6 Variables affecting natural spatio-temporal variation in meiofaunal distributions and implications for monitoring studies

The abundance and richness of meiofauna at a particular point in space and time is a function of their response to an array of abiotic and biotic variables. Sandy beach environments are the most physically variable of all marine environments (Nybakken and Bertness 2005). Currents, tides, and wave action determine the shape and sediment characteristics of a beach (Defeo and McLachlan 2005) by influencing the processes of sediment erosion, deposition, water saturation and physical disturbance (Jackson et al. 2002). In the intertidal zone of exposed beaches, meiofaunal distributions are strongly controlled by sediment saturation and grain-size characteristics (McLachlan 1983, Albuquerque et al. 2007). Predation and competition can also influence meiofaunal distributions (Moens et al. 2000). Meiofauna may exhibit voluntary or involuntary vertical or horizontal movement within the sand matrix and between the sand bed and water column in response to biotic and abiotic variables (Joint et al. 1982, Armonies 1990).

Meiofaunal and nematode communities have been found to vary in response to environmental gradients from latitudinal scales (Lee and Riveros 2012) to microscale (Gingold et al. 2011). Several studies suggest that this variability may be greater at larger than smaller scales (e.g. Phillips and Fleeger 1985, Li et al. 1997). Nevertheless, although sandy beaches may appear homogenous, differences in water percolation, the depth of the water table and physical disturbance across the intertidal zone creates sub-environments of varying moisture, temperature, salinity, food and oxygen availability (Dye 1980, McLachlan and Hesp 1984, McLachlan and Turner 1994, Nicholas and Hodda 1999, Gheskiere et al. 2005, Giere 2009). Water percolation is driven by tides and waves, and has a direct effect on swash/backwash processes coupled to the erosion/accretion of the beach face (McLachlan et al. 1985). Water movement during low tide is affected by gravity and the horizontal flow of the water table (Giere et al. 1988). Movement of the water table over the tidal cycle is influenced by other physical parameters: beach geomorphology, sediment size, sorting and porosity, tidal elevation, and wave set-up and run-up (Jackson et al. 1999). Cumulatively, these processes lead to zonation patterns across the sandy beach, from the surf zone to the dune system; although the physical boundaries between these may vary between seasons (Brazeiro and Defeo 1996, Nicholas and Hodda 1999, Colombini et al. 2002) and beach type

(Rodriguez et al. 2001). Variations have also been recorded along the beach (Nicholas and Hodda 1999) and with alongshore beach morphological features such as cusps (McLachlan and Hesp 1984).

Sediment size, sorting, shape and composition have been related to meiofaunal density and diversity in all beach types (e.g. Wieser 1959, Heip et al. 1985, McLachlan and Turner 1994, Giere 2009). The mean grain size determines the body size and habit (Swedmark 1964), as well as dominant phyla (Dye 1978, Warwick 1981). Meiofauna may respond directly to grain size, which governs the rates of decomposition and remineralisation (Coull 1999). Additionally, meiofauna may respond indirectly to grain size as a consequence of its effects on the microphytobenthic community, which in turn influence oxygenation of the sediments as well as food availability (Montagna 1995, De Troch et al. 2006). The strong correlation between meiofaunal density and diversity indices and shell content (Shirayama 1984, Tietjen 1991, Semprucci et al. 2010) may reflect the diversity of micro-habitats provided by this substrate (Giere 2009). Compared to siliceous sand, biogenic sediments are more structurally complex, unsorted and porous (Giere et al. 1988). They also favour the absorbance of nutrients and adherence of fauna and microphytobenthos, providing more habitat and feeding resources (Shirayama 1984, Giere et al. 1988, Tietjen 1991, Semprucci et al. 2010).

Rather than absolute values, a determining factor for the persistence of meiofauna may be the extent of physio-chemical variation (Attrill 2002). Coastal estuaries and lagoons are generally very productive environments, which are physically more stable than exposed beaches but may experience greater spatial and temporal variations in water chemistry (e.g. salinity; Roy et al. 2001). While estuaries and lagoons have more abundant and reliable food sources (Roy et al. 2001), they are represented by fewer species able to withstand greater variation in chemical conditions (Barnes et al. 2008, Hourston et al. 2009). Coastal estuaries have greater meiofaunal densities; however they are generally found to have less diversity than marine or freshwater environments (Heip et al. 1985, McLusky and Elliot 2004).

The coast is a very dynamic environment, with many factors influencing meiofaunal presence. It is evident, given these factors and the small size of these organisms, that the sampling design and scale of monitoring studies aimed at assessing ecological impacts of perturbations would influence the outcome

(Findlay 1982). In a recent critical review of climate change experiments on marine organisms, Wernberg et al. (2012) identified pseudo-replication and the failure to incorporate spatial and temporal autocorrelation in study design and analysis as a prevalent issue. There are, however, few reviews or studies examining issues of sampling design in meiofaunal research (but see Findlay 1981, Findlay 1982, Warwick and Clarke 1991, Leduc et al. 2010) and recent reviews of sampling guidelines in the sandy beach environment have given no mention of meiofauna (Schlacher et al. 2008).

Background meiofaunal patchiness is rarely considered in the experimental designs of impact assessments (Fraschetti et al. 2006). Of the studies that have considered the scales of spatial variation in sandy beach meiofaunal communities, most have focused on how their abundance and diversity vary across scales of 100's of metres to kilometres according to large-scale gradients in grain size, wave height and beach slope (e.g. Rodriguez et al. 2001, Rodriguez et al. 2003, Gheskiere et al. 2005, Lee and Riveros 2012). Due to the along-shore variation in geomorphology and wave climate this approach may not accurately measure the extent of variation within a single beach or beach type (James and Fairweather 1996). Attempting to compare results between different beach types when the extent of variation on a single beach or between similar beaches is not established, may be misleading (Thrush et al. 1994). We need to determine the extent of variation within similar beaches and similar conditions first, before comparing different beach types and environmental conditions (Sheppard 1999). A greater level of sensitivity in sampling designs is required to accommodate geomorphological characteristics of the environment (Jackson et al. 2002, Siontorou and Batzias 2011) and adequately address variation at different scales (Legendre et al. 1997). There is also a recognised need to standardise sampling locations and methods to facilitate comparison among different sites (Jackson et al. 2002).

1.7 Management of Australian sandy beaches and key knowledge gaps

The coastline of Australia, at 29,900 km, is one of the longest in the world (Short 2006). Approximately 50% of this coastline is sandy beach (Abuodha and Woodroffe 2006). Australian coastal areas, particularly in the south-eastern region, are densely populated and highly productive regions (Small and Nicholls 2003, McGranahan et al. 2007). The coastal environment of south-eastern Australia has been identified by the Federal Government as one of the environments most at risk from the multiple effects of

climate change and adaptation of coastal settlements (Department of Climate Change 2009). Concerns regarding the impact of coastal urban development and associated stressors have been repeatedly voiced in State of Environment reporting. The 2001 report stated that development of the coastal strip was “one of the major strategic issues confronting the conservation and management of the coastal zone” (Australian State of the Environment Committee 2001). The 2006 report warned that “if current population trends continue, 42.3% of the Nowra to Noosa coastline will be urbanised by the year 2050, with the resulting loss of much of Australia’s temperate and tropical coastal systems” (Beeton et al. 2006). In the most recent State of the environment report (Hatton et al. 2011) beach and shoreline modification, leading to change or loss of habitat was defined as a current and emerging risk to the Australian marine environment. Although coastal management is often listed as a threat to the biological communities of sandy beaches (Schlacher et al. 2008), reviews of nourishment practices and their impact are not available for Australia.

The historical implementation of coastal engineering (Gourlay 1996), policy development (Kay and Lester 1997) and, more recently, beach scraping (Carley et al. 2010) in Australia have been reviewed and discussed. Although each of these reviews highlighted significant projects and different approaches, none attempted to quantify the extent of work conducted. Hence, although reviews of soft engineering practices have been conducted for Europe and the USA, no comparable information is available for Australia; despite its large proportion of sandy beaches. Such a gap in knowledge means that policy is working from an incomplete understanding of management. The ability to establish best practice guidelines and conduct adaptive management is seriously diminished. Given the growing threats to beaches and the need to refine management responses, it would be useful to collate and analyse information on beach management. This topic is particularly relevant due to recent reviews and changes to Australian coastal policy, as well as increasing concerns of the effect of climate change on coastal areas.

Increasing amounts of coastal development and beach management require an appropriate assessment of their impacts. Despite this, not much is known of the impacts of nourishment on sandy beach ecosystems of Australia, with only a single paper on this topic (Jones et al. 2008). There is an extreme paucity of studies on sandy beach biota (Dugan et al. 2010). The amount of research conducted by

ecologists on sandy beaches is small even compared to other coastal systems (McLachlan et al. 1981, Fairweather 1990b). Furthermore, very little is known of Australian sandy beach communities, and in particular meiofauna. Biogeography of macrofauna and their response to both environmental variation and anthropogenic disturbance has received more study (e.g. Dexter 1984, Dexter 1992, Hacking 1996, James and Fairweather 1996, Hacking 1998, Gladstone et al. 2006, Hacking 2007, Jones et al. 2008, Duong and Fairweather 2011). Previous studies have examined meiofaunal responses to beach face elevation and other environmental factors on Australian beaches (McLachlan and Hesp 1984, Nicholas and Hodda 1999, Nicholas 2006, Nicholas and Trueman 2009). The relative importance of aspects of beach morphology as well as spatial and temporal variation, are however, not well understood. There are few studies examining the response of meiofauna to coastal management (but see Dye 2005, Dye and Barros 2005). More is known, however, about estuarine meiofaunal communities of Australia (reviewed by Coull 1999).

Schlacher et al. (2008) identified significant knowledge gaps and research priorities for sandy beach ecology including: (1) the information gap between ecologists and managers, (2) information regarding the extent of natural variability versus impact effects (e.g. the impact of coastal management), (3) the biological variation of beach biota in terms of spatio and temporal scales, (4) recovery trajectories of impacted areas, (5) suitability and performance of 'indicator' species, and (6) design of monitoring programs to track changes.

1.8 Thesis outline

This study sought firstly to address knowledge gaps in: (1) the extent of beach management practices in Australia; and (2) the ecology of meiofaunal communities and, in particular the scales at which meiofauna naturally vary spatially and temporally. Additionally, meiofauna were examined as bio-indicators in a dredge and fill project in a south-eastern Australian coastal lagoon.

First, to understand the extent of coastal management being undertaken in Australia a review of beach nourishment practices was undertaken (Chapter 2). Through a survey of beach managers of Local Government Areas and a comprehensive search of peer reviewed and grey literature, the extent of beach

nourishment in Australia was assessed. This study will assist adaptive coastal management by documenting the extent of our knowledge; providing a comprehensive review that will inform Local Government Authorities, researchers and policy makers (Cooper and Lemckert 2012). Such information could be used to highlight the growing need for appropriate resources in order to protect the ongoing recreational use and biodiversity of our beaches.

The design of meaningful ecological assessments for determining human impacts to ecosystems hinges upon an understanding of the spatial structuring of their key biotic components. A series of small scale surveys assessed the spatial and temporal scales of variation in sandy beach meiofauna, and their potential as indicators of ecosystem change. The first of these (Chapter 3) was a small-scale case study conducted in North Carolina. This study compared scales of spatial variability in meiofauna and macrofauna, posing the following questions: (1) what is the abundance and diversity of fauna; (2) how does this vary over spatial scales and between macro- and meiofaunal groups; and (3) what is the most suitable way to collect and analyse data in this environment? The study also explored how much of this variation was correlated with variation in sediment characteristics and could be attributable to certain biological characteristics of the fauna.

In Chapters 4, 5 and 6 the Sydney Northern Beaches littoral system as a case study was examined. This coastal sedimentary system includes exposed sandy beaches as well as intermittently open and closed lakes and lagoons (ICOLLs). The Sydney Northern Beaches are a great local attraction for their natural environmental values. This area has been increasingly urbanised since the 1970s, Collaroy-Narrabeen beach has now been identified as a coastal erosion hotspot (Office of Environment and Heritage 2012).

In Chapter 4, meiofaunal and nematode biodiversity, as well as environmental variables, were sampled from three geomorphically similar beaches in geographic proximity. The aim of this chapter was to determine whether the faunal communities of adjacent beaches were similar, as might justify their use as controls in impact assessment. Specifically, the following issues were addressed: (1) the similarity of faunal and sediment conditions of geomorphically similar sections of each beach; (2) where differences

occur, whether particular meiofaunal taxa or nematode genus were indicative of differences between sites; and (3) the environmental variables affecting spatial variation in meiofauna at each site.

In Chapter 5 the extent of natural variability of meiofauna over temporal as well as spatial scales was examined. It specifically covered the results of a study on the role of tidal height and morphological features of the beach in determining meiofaunal community structure; and whether this relationship was stable and predictable over time. The time periods examined were short term, over the tidal cycle; medium term, over consecutive days; and long term, over consecutive years.

Increasing urbanisation and population pressures intensify the need for coastal management in south-eastern Australia (Harvey and Caton 2010). A common form of coastal management, lagoon entrance dredging coupled with a beach fill operation, was explored in Chapter 6. This chapter examined whether differences in the meiofaunal assemblage after this single disturbance could be detected over the 'noise' of cumulative historical impacts and seasonal trends. Currently, sandy beach environments face a variety of pressures. In this context it is important to understand the dynamics of the beach ecosystem. An understanding of the of the rates of natural and anthropogenically induced change will assist in minimising impacts to these ecological communities, and ensure that the important food webs supported by marine invertebrates do not collapse.

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Review

Nourishment practices on Australian sandy beaches: A review

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ABSTRACT

It is predicted that the coastal zone will be among the environments worst affected by projected climate change. Projected losses in beach area will negatively impact on coastal infrastructure and continued recreational use of beaches. Beach nourishment practices such as artificial nourishment, replenishment and scraping are increasingly used to combat beach erosion but the extent and scale of projects is poorly documented in large areas of the world. Through a survey of beach managers of Local Government Areas and a comprehensive search of peer reviewed and grey literature, we assessed the extent of nourishment practices in Australia. The study identified 130 beaches in Australia that were subject to nourishment practices between 2001 and 2011. Compared to projects elsewhere, most Australian projects were small in scale but frequent. Exceptions were nine bypass projects which utilised large volumes of sediment. Most artificial nourishment, replenishment and beach scraping occurred in highly urbanised areas and were most frequently initiated in spring during periods favourable to accretion and outside of the summer season of peak beach use. Projects were generally a response to extreme weather events, and utilised sand from the same coastal compartment as the site of erosion. Management was planned on a regional scale by Local Government Authorities, with little monitoring of efficacy or biological impact. As rising sea levels and growing coastal populations continue to put pressure on beaches a more integrated approach to management is required, that documents the extent of projects in a central repository, and mandates physical and biological monitoring to help ensure the engineering is sustainable and effective at meeting goals.

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1. Introduction

Sandy beaches are of high socio economic value. They provide recreational opportunities, are aesthetically valued, buffer the land against extreme wave events, and are sites of water filtration and nutrient cycling (Batley and Cocks, 1992; Blackwell, 2007). Additionally, beaches support dense and diverse biological communities of ecological significance (Brown and McLachan, 1990). The invertebrates that live under the sand surface provide food for surf fishes and shorebirds and, along with microphytobenthos, are critical in carbon and nutrient cycling (Brown and McLachan, 1990). The beach also provides critical nesting habitat for turtles and sea birds (Bamford et al., 2008; Hawkes et al., 2009).

Increasingly, sandy beaches are being squeezed between rising sea levels and extreme weather events on the marine side, and anthropogenic pressures such as expanding development on the landward side (Defeo et al., 2009). Beaches naturally go through cycles of accretion and erosion at time-scales that range from individual storm events through to persistent coastline recession occurring at a multiple decadal scale (Cowell and Thom, 1997). Over 70% of the world's beaches are in recession (Bird, 1996). Human modification of the coastal zone undermines the natural resilience of beaches by constraining landward shoreline movement (Nordstrom, 2000) and interfering with coastal sediment transport processes (Komar, 1998). Recession or persistent erosion is of concern as it narrows the beach, reducing public beach amenity and placing coastal infrastructure at risk (Bird, 1996; Finkl and Walker, 2004).

A variety of strategies of management are being used to combat effects of erosion on coastal properties and/or public beach amenity. These may include mandated set backs of new development from the coast to allow the shoreline to recede naturally, managed retreat of infrastructure under threat, as well as hard and

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soft engineering. Hard engineering involves the construction of hard structures, such as seawalls that form a physical barrier between the encroaching sea and coastal properties, and groynes that trap sediments and encourage beach accretion. Soft engineering, by contrast may involve extending the beach seaward through artificial nourishment (or 'fill') with sediments from external 'borrow' sites, replenishment with sediments from the same coastal compartment or sediments scraped across a beach. In some instances soft and hard engineering are combined (Finkl and Walker, 2004). Where the construction of hard structures such as river entrance training walls and breakwaters trap sediments, artificial sand bypass (*sensu* Bruun, 1996) can be used to move littoral drift around structures, restoring the natural sand transport and preventing net sand deficits to beaches further along the coast.

Many agencies regard soft engineering as the preferred management option since it avoids the negative effects of hard structures on public beach amenity and coastal ecosystems (Finkl and Walker, 2004; Hanson et al., 2002; Thom, 2003). However, whether beach nourishment can be considered a long-term solution to shoreline erosion remains to be seen (Finkl and Walker, 2004). Beach nourishment projects vary markedly in their success, judged by volumetric loss, loss rates, planned versus required intervals between nourishment projects and ascertainment of specific project goals (National Research Council, 1995). Furthermore, despite the presumption that soft engineering is ecologically benign, large impacts on beach biota can result (Nelson, 1993; Peterson and Bishop, 2005; Peterson et al., 2006; Speybroeck et al., 2006). The efficacy and impacts of soft engineering are likely to depend on the scale and timing of projects, as well as the source of sediment they use (Peterson et al., 2006).

Beach erosion and its management is a particularly pertinent problem for Australia, due to its extensive sedimentary coastline, and its highly coastal human population (Fig. 1). Approximately half of Australia's 29,900 km coast comprises sandy beaches, which number 10,685 (4934 of which are wave dominated) and account for 8% of beaches found globally (Short, 2006). Over 25% of the Australian population lives within 3 km of the coast and across the two most developed states, New South Wales and Victoria, over



Fig. 1. Map of Australia showing the extent of sandy beaches (in black, from GeoScience Australia, <http://www.ozcoasts.gov.au/coastal/smartline.jsp>, last accessed 16/09/12); boundaries of States and Territories (in grey); names of States and Territories (in capital letters); names of capital cities and those mentioned in the text (in lower case); and positions of capital cities and locations mentioned in the text (open circles).

20% of the coastline is urbanised (Beeton et al., 2006). Sea level rise of 1.1 m, the high end scenario for 2100, is predicted to inundate and damage between 157,000 and 247,600 residential buildings, with a replacement value of up to \$63 billion (Department of Climate Change, 2009). The recent Federal Government, State of the Environment Report, cited beach or shoreline modification, leading to change or loss of habitat as a current and emerging risk to the marine environment (Hatton et al., 2011).

In Australia the responsibility for management of coastal areas is shared between three levels of government: Commonwealth (Federal), State and Local. The Commonwealth government lacks direct constitutional power in coastal management while State coastal strategies generally outline the principles for coastal planning and management (James, 2000). Local Government Authorities (LGAs) are responsible for implementing the majority of coastal management, translating State planning and management policies and legislation into local actions as well as providing infrastructure, foreshore maintenance and ensuring public safety (James, 2000).

Given the growing threats to beaches, globally, and the need to establish best practice management, the key drivers of and constraints on beach management need to be understood. Reviews of artificial nourishment have been conducted for a number of European countries and the USA (Basco, 1999a; Campbell and Benedet, 2006; Hamm et al., 2002; Hanson et al., 2002). However, no such review has yet been conducted for Australia despite its significant contribution to the world's total sandy beaches. Here we present a review of nourishment practices in Australia focussing on: (1) the frequency and scale of projects; (2) trends in use of nourishment practices over time; and (3) key drivers of when and where practices are used. Additionally, to provide recommendations for future beach management we consider: (1) how monitoring is being used as an adaptive management tool; and (2) how management practices in Australia compare to elsewhere.

2. Material and methods

Our review comprised two parts: (1) a survey of LGAs, responsible for implementing coastal management, to assess past, present and future use of nourishment practices as a tool for beach management and (2) a literature search of nourishment projects that may have been missed in the survey. Each component of our study considered three nourishment practices: artificial nourishment, replenishment and scraping. Artificial nourishment was defined as the importation of sand to a beach or offshore sediment repository supplying the beach, from a site external to the beach's coastal compartment (e.g. offshore or quarry). Replenishment used sand from the same coastal compartment as the fill site (e.g. dredge spoil from an adjacent coastal lagoon). Beach scraping (also referred to in the literature as skimming, beach panning, nature assisted beach enhancement, assisted beach recovery and beach recycling and re-profiling) was the mechanical redistribution of sediment, without alteration of total volume, from the intertidal zone to the backshore (Carley et al., 2010). Where these forms are referred to collectively in this manuscript, we use the terms nourishment practices or nourishment projects.

2.1. Survey of Local Government Authorities (LGAs)

To assess past, present and future use of nourishment practices as a tool for beach management, a survey was sent by email to beach managers of all LGAs with ocean frontage (200 in total). LGAs with any type of ocean frontage were targeted to avoid errors based on discrimination between sedimentary and rocky shoreline from the available maps.

Table 1

Percentage of Local Government Areas responding to the survey that had previously used (within the ten year period, 2001–2011; past) and/or plan to in the future (5 year period, 2012–2017; future) use nourishment practices on beaches.

State/Territory	Total number of councils with ocean frontage	Number of councils responding to the survey	Use of nourishment practices			
			Past and future	Past only	Future only	Neither past nor future
NSW	31	14	64.3%	0.0%	0.0%	35.7%
QLD	41	6	100.0%	0.0%	0.0%	0.0%
WA	45	13	46.2%	15.4%	0.0%	38.5%
VIC	21	8	75.0%	0.0%	12.5%	12.5%
SA	33	5	40.0%	20.0%	0.0%	40.0%
TAS	22	4	0.0%	0.0%	0.0%	100.0%
NT	7	0	—	—	—	—
Total	200	50	58.0%	6.0%	2.0%	34.0%

State and Territory abbreviations: New South Wales (NSW), Queensland (QLD), Western Australia (WA), Victoria (VIC), South Australia (SA), Tasmania (TAS) and Northern Territory (NT).

The survey, disseminated in mid 2011, asked managers about: (1) use of nourishment practices in their LGA over the 10 years 2001–2011, including localities, sizes and frequencies of projects; (2) projected future use of nourishment practices over the 5 years 2012–2017; (3) the key determinants in deciding to adopt nourishment practices, including causes of erosion, and goals of the beach management; (4) sources of sediment used, and key factors determining timing of nourishment projects; and (5) any monitoring that is done to assess the efficacy of the management and/or its ecological impacts.

2.2. Literature search

To identify nourishment projects over the ten years 2001–2011 that may have been missed in the survey of Local Government Areas (see Section 2.1) we conducted comprehensive web and literature searches for all existing peer reviewed and grey literature on nourishment practices in Australia. Reports uploaded by beach managers and coastal engineers as well as research papers, newspaper reports and coastal websites were sourced using the following search terms: nourishment, replenishment, deposition, beach enhancement, beach skimming, beach panning, nature assisted beach enhancement, NABE, assisted beach recovery, beach recycling, re-profiling, bulldoze/ing, sand scrape/ing and sand push/ing. In addition, the literature search also looked for instances over the ten years 2001–2011 where sediment from permanent sand bypass systems has been used in nourishment. Australian results were compared to artificial nourishment conducted elsewhere in the world where published or unpublished reviews summarising or discussing projects for other countries could be found. It is noted that for many countries, particularly in Asia, where nourishment projects are undertaken (e.g. Kuang et al., 2010; Leatherman, 1996), reviews were not available.

3. Results

3.1. Nourishment practices in Australia

A total of 50 LGAs responded to the survey. Of respondents, 64% had undertaken nourishment projects over the past 10 years (2001–2011), and 60% planned to conduct these forms of nourishment projects in the next 5 years (2011–2017) (Table 1). Only 2% of LGAs that had not previously conducted any type of nourishment project indicated that they would adopt this form of management in the next five years. The proportion of LGAs presently conducting nourishment projects, which indicated that they would be discontinued in the near future, was 6% (Table 1).

In total, 77 beaches were identified by the survey as having received nourishment projects between 2001 and 2011, but the literature search revealed an additional 53 beaches that had received nourishment projects over this time frame, making a total of 130 beaches in Australia that had undergone this form of management (Table 2). The majority of these beaches were wave dominated (Short, 2006), therefore approximately 2% of the total Australian wave dominated beaches have been managed by nourishment or replenishment, and 3% if scraping is included, in the past 10 years. Beaches receiving nourishment projects were clustered around the major urban centres of Brisbane, the Gold Coast, Sydney, Melbourne, Adelaide and Perth (Fig. 1). The primary reasons for conducting nourishment projects were to protect coastal infrastructure (34%) and public beach amenity (26%). Additional reasons included: public safety, such as maintenance of walkways (21%), ecological (12%), and deposition of dredge spoil from another location (5%). Neither the survey nor the literature search turned up any evidence of nourishment projects in northern Australia (northern Western Australia, Northern Territory and the gulf coast of Queensland). Northern Australia has a coastline that is

Table 2

Number of beaches identified by the survey of Local Government Areas and the literature search to have received artificial nourishment only, replenishment only, beach scraping only or multiple methods within the period 2001–2011.

State/Territory	Artificial nourishment only	Replenishment only	Scraping only	Multiple methods	Total
NSW	12	11	10	4	37
QLD	7	9	10	8	34
WA	17	4	4	9	34
VIC	4	7	2	0	13
SA	6	1	0	4	11
TAS	0	0	1	0	1
Total	46 (35.4%)	32 (24.6%)	27 (20.8%)	25 (19.2%)	130

State and Territory abbreviations: New South Wales (NSW), Queensland (QLD), Western Australia (WA), Victoria (VIC), South Australia (SA) and Tasmania (TAS). Northern Territory is not included as no results were found.

dominated by mangrove beaches of silt and mud rather than sand, which is for the most part remote from large human population centres.

Of the beaches examined by the LGA survey and literature search (i.e. undergoing nourishment projects between 2001 and 2011), 35% received artificial nourishment alone, 25% replenishment alone, and 21% scraping alone (Table 2). The remainder received a combination of methods, the most common being artificial nourishment followed by scraping (19%; Table 2). The LGA survey found that a high proportion of the beaches undergoing nourishment practices also received other types of management; 48% also had hard engineering structures and 34% were regularly groomed/cleaned. When LGAs were questioned about other planning mechanisms there was a 60% response rate. Of respondents, 70% (i.e. 42% of the councils involved in the survey) stated that nourishment practices were part of a larger management strategy or plan and 80% (i.e. 48% of the councils involved) had development controls (e.g. building codes and mandatory setback areas) in place for these managed beachfronts.

The primary reasons given for sand loss from beaches undergoing nourishment projects were due to short-term erosion events such as extreme weather events (36%). The other main reasons were long-term processes: sand movement due to littoral drift (31%); and disruption to the local sediment budget due to human activities such as the impact of historical structures such as hard engineering structures built for coastal protection, breakwaters, unapproved rock walls, urban shoreline development, damming of rivers, opening lagoons, historical sand extraction and removal of coastal vegetation (total 30%). Nourishment projects were most commonly done annually (39% of beaches), especially where artificial nourishment or replenishment was involved (Table 3).

Most nourishment practices used sediment sourced from the same (51%) or nearby (14%) coastal compartments, with a smaller number of LGAs utilising sand from a quarry (16%), offshore (8%), river (5%), building site (3%) or other dredge material (3%). Therefore (27%) was obtained from terrestrial or offshore locations which are not part of the active sediment supply. Sand was predominately placed on the backshore (44% of projects), with fewer projects using the fore- (37%), or near-shore (14%) as fill sites. The remaining 5% placed sand offshore or at other locations. Forty two percent of projects utilised sediment volumes of 101–5000 m³, while 11% used smaller volumes, 8% used volumes between 5001 and 10,000 m³, 14% used between 10,001 and 50,000 m³, 8% used larger volumes (more than 50,001 m³), and 17% used an undetermined volume.

Projects were predominantly conducted in spring (40%), with lesser numbers in winter (22%) and autumn (16%), and very few (6%) conducted in summer (the remainder were variable in timing). The key drivers for the timing of nourishment projects were ensuring that the wave climate was favourable for sediment accretion (27% of cases), decreasing the impact of the management intervention on the recreational use of the beach (18%) and the availability of sediment (17%). Other drivers of the timing of nourishment were the availability of staff and financial resources (13% of cases), the timing of wet/dry seasons and storms (9%), minimisation of ecological impact (7%), the availability of dredging equipment (7%).

3.2. Monitoring studies associated with nourishment projects in Australia

Of respondents to questions on monitoring (*n* = 29), only 17% (5 LGAs) from two States (New South Wales and Victoria) reported undertaking monitoring to assess the efficacy and/or impacts of nourishment projects. Additionally, one Queensland LGA reported undertaking yearly beach survey profiles in order to determine

Table 3

Percentage of beaches receiving nourishment practices that are manipulated (a) once a year or more frequently, (b) once every 2–3 years, (c) once every 4–9 years, (d) once every 10 years and (e) at unknown frequency, according to the survey of Local Government Areas.

	Once a year or more often	Once every 2–3 years	Once every 4–9 years	Once in 10 years	Unknown
Artificially nourished (40 beaches)	45%	25%	12.5%	7.5%	10%
Replenished (20 beaches)	45%	20%	35%	0%	0%
Scraped (18 beaches)	16.7%	16.7%	50%	11.1%	5.6%
Combination of two of above (6 beaches)	50%	16.7%	33.3%	0%	0%

beach volumes. All five LGAs conducting monitoring did so both before and after a management event, with three LGAs monitoring both borrow and fill areas, one monitoring borrow areas only and one monitoring only fill areas. Three councils reported more than one type of monitoring. The primary purpose of monitoring was either to examine the ecological impact (46%) or the retention of sediment (46%). The specific questions being addressed by the monitoring studies, and the indicators used to address these questions, varied among LGAs (Table 4). Three of the five LGAs indicated that they used the results from previous monitoring to inform future nourishment projects.

3.3. Sand bypassing projects

The literature review identified nine localities where sand from permanent bypass projects was utilised for nourishment (Table 5). These were much larger operations than the average sized project identified in the survey of LGAs, with longer periods across which nourishment was done.

4. Discussion

4.1. Nourishment practices in Australia

This study represents the most comprehensive assessment yet of the use of nourishment practices as a tool for beach management in Australia. It has found a total of 130 beaches presently managed

Table 4

Factors and indicators monitored before and after artificial nourishment, replenishment and scraping events, as reported by Local Government Authorities that presently have environmental or ecological monitoring associated with their nourishment projects (*n* = 5).

Site	Factors measured	
	Examining the ecological impact:	Examining the retention of sediment:
Borrow	<ul style="list-style-type: none"> •Beach infauna (including: abundance, species richness) •Sediment characteristics (including granulometry) •Ecological indicators (undefined) •Turbidity •Seagrass 	<ul style="list-style-type: none"> •Contour survey (including shore normal profile lines) •Lagoon entrance position •Rainfall •Wave dynamics
Fill	<ul style="list-style-type: none"> •Beach infauna (including: abundance and species richness) •Sediment characteristics (including granulometry) •Sediment volume •Ecological indicators (undefined) •Turbidity 	<ul style="list-style-type: none"> •Contour survey (including shore normal profile lines, depth contour) •Lagoon entrance position •Rainfall •Wave dynamics •Sediment characteristics (including granulometry)

Table 5
Details of permanent sand bypass projects in Australia where sand is utilised for nourishment.

Location	State	Frequency	Primary reason	Approx amount (10^3 m^3)
Bandy Creek Harbour ^a	Western Australia	Every 2 years	Maintain a navigable entrance and coastal protection	50–70 (per year)
Dawesville ^a	Western Australia	Annual, Feb–June	Mimic natural sediment transport processes	100
Mandurah Inlet ^a	Western Australia	Annual, June–November	Mimic natural sediment transport processes and maintain a navigable entrance	100
Exmouth boat harbour ^a	Western Australia	Every 1–3 years	Mimic natural sediment transport processes and maintain a navigable entrance	10–20 (per year)
Port Geographe ^a	Western Australia	Annual	Mimic natural sediment transport processes and maintain a navigable entrance	Sand and wrack bypassing. Approximately 80–100 of each per year
Nerang (Gold Coast Seaway) ^b	Queensland	Annual, 4–5 months of the year	Mimic littoral drift. Move sand from areas of accretion, bypass the entrance channel and pump sand to beaches to the north	500 per year
Tweed River Entrance Bypassing Project ^c	Queensland	Annual	Mimic littoral drift. Move sand from areas of accretion, bypass the entrance channel and pump sand to beaches to the north	396–787 (per year, in the period 2003–2010)
Noosa ^d	Queensland	Annual	Supplements sand supply to protect infrastructure from the erosional effects of storms	40–80 per year
Port of Portland ^e	Victoria	Data not available	Removes sand from Port entrance and nourishes downdrift beaches	Data not available

^a Karl Ilich, personal communication.

^b Boswood and Murray, 2001 verified by Brian McRae, personal communication.

^c Tweed River Entrance Sand Bypassing Project (TRESBP) website (<http://www.tweedsandbypass.nsw.gov.au/>, last accessed 22/04/12).

^d Nankervis, 2005.

^e Boswood and Murray, 2001.

by artificial nourishment, replenishment or beach scraping in Australia. Nevertheless, as this study relied on voluntary responses to a survey and review of published materials it is expected that the extent of nourishment practices, in particular beach scraping, which may not require reporting and permits (Carley et al., 2010), was under-estimated.

Artificial nourishment, replenishment and scraping were generally associated with more urbanised areas, with the highest amount of management recorded in New South Wales, the most urbanised State (Beeton et al., 2006). This was consistent with the primary goal reported for nourishment projects, to protect private property and public infrastructure. State and Local Government Plans have identified more beaches where artificial nourishment might be a suitable and beneficial management action in the future. In addition, some Plans recommend artificial nourishment for locations outside heavily urbanised settings (e.g. Queensland State Government, 2005). The aim of rehabilitation in these cases was often to protect infrastructure or stabilise and restore pre-erosion conditions.

The main perceived cause of sand erosion, necessitating nourishment projects, was loss of sediment following extreme weather events, with loss of sediment through littoral drift the second most important reason. Yet, since the 1970s, a period of rapid urbanisation of many coastal areas there have been relatively few major storm events for the eastern coast of Australia (Castelle et al., 2007). While the majority of LGAs reported that the primary reasons were understood, a large proportion of beach management in Australia is undertaken in response to short-term erosion which threatens private property and public infrastructure but typically does not involve sand being lost from the system.

4.2. A comparison of Australian with international beach nourishment practices

In comparison to artificial nourishment and replenishment conducted elsewhere in the world (summarised in Table 6), Australian projects were typically small in scale, short in duration and frequent. The majority of European countries had a renourishment interval of around five years (Hanson et al., 2002),

whereas most sites in Australia were nourished once a year or less (Table 3). Whereas projects in Europe and USA often utilised sediment dredged from offshore (Hanson et al., 2002), Australian systems were more opportunistic in their use of sediments, the majority of fill coming from the same coastal compartment as the beach undergoing management. In Australia, offshore sand extraction was not permitted in some States, which may make long-term planning of projects more difficult. Transfer of sediment from an accreting to an eroding beach within the same coastal compartment may minimise ecological impacts of nourishment projects associated with changes in sediment granulometry (e.g. Peterson et al., 2006). However, this may exacerbate sand loss from a coastal compartment by shifting it to a site where it is not retained.

Respondents to our LGA survey reported that 'natural' causes of erosion were in the majority, with extreme weather events and sand movement together accounting for 66.7% and anthropogenically induced erosion for 29.5%. These figures are very similar to in the USA, where an examination of 15 major beach nourishment projects conducted by the US Corps of Engineers found that for 59% of sites the cause of erosion was deemed to be 'natural', and was anthropogenically induced in 30% (Basco, 1999a). When causes were examined singly, inlet jetties and navigational channels were the second highest cause of erosion in these beaches (Basco, 1999a). In many European countries coastal development and hard engineering structures are cited as the key causes of erosion (Hanson et al., 2002; Finkl and Walker, 2004).

While the majority of Australian nourishment projects were conducted to protect development and maintain recreational use of the beach, the majority of projects conducted in the USA were associated with navigation and dredge disposal (35% of federally funded projects in Gulf of Mexico and 58% on Pacific coast; Leonard et al., 1990). The motivation for employing artificial nourishment varies amongst European countries: the primary reason in the Netherlands was for safety and coastline preservation, in Spain to protect houses and infrastructure and in Germany to prevent recession and protect houses (Hanson et al., 2002).

The majority of nourishment projects in Australia placed sand on the backshore and foreshore. This may be an attempt to emulate

Table 6

The incidence and extent of beach nourishment shown by country. Note only countries for which reviews of nourishment practices have been done are included in the Table.

Country	Year started	Number of sites	Average volume/fill (10^3 m^3)	Mean annual rate of projects	Length of coastline (km)
France	1962 ^a	26 (includes shingle and gravel) ^a	104 ^a	<1 ^b	5500
Italy	1969 ^a	36 ^a	420 ^a	1 ^b	7500
Germany	1951 ^a	60 ^a	385 ^a	3 ^b	1900
Netherlands	1970 ^a	30 ^a	733 ^a	6 ^b	451
Spain	1985 ^a	400 ^a	183 ^a	10 ^b	4964
UK	1954 ^a	32 (includes shingle and gravel) ^a	570 ^a	4 ^b	12,429
Denmark	1974 ^a	13 ^a	263 ^a	3 ^b	7400
USA	1919 ^c	49 ^d ; >200 ^e	Range from 150 to 600 ^e	30 ^d	19,924
Australia	1974 ^f	78 ^g	Discrete artificial nourishment events: 15 Bypass operations (Table 5): 190 ^g	34 ^g	29,900 ^h

Source: ^aHanson et al., 2002; ^bHamm et al., 2002; ^cClayton, 1991; ^dBasco, 1999a (only includes Corps of Engineers projects between 1950 and 1995); ^eCampbell and Benedet, 2006 (an estimate of all nourishment projects since 1920); ^fBoak et al., 2001; ^gCurrent study, artificial nourishment and replenishment projects between 2001 and 2011; ^hShort, 2006.

the natural dune system in order to protect beach front infrastructure and may reflect the fact that most sand used for nourishment projects in Australia is transported by land rather than pumping from offshore dredge sites. Berm or shoreface fill also appears to be popular in many European countries (Hanson et al., 2002), although nearshore and offshore placement of fill sand is used in some locations such as the North Sea (Roeland and Roelse, 1995). Nearshore nourishment allows sediment to be re-deposited onto the beach via wave and wind action, so that the degree of sorting and patterns of accumulation may more closely match native beach sediments. It is also less disruptive to recreational activities on beaches than operations that require machinery on the beach itself. In addition, nearshore sand placement may also serve the purpose of building sand buffers, which decrease coastal erosion by reducing the energy of incoming waves (Boswood et al., 2001). The method of emplacement may best influenced by motivations for nourishment and knowledge of the coastal compartment. In some areas, such as estuarine beaches, the placement of subtidal fill is discouraged due to the high productivity and diversity of fauna in this zone (Jackson et al., 2010).

The longevity of projects has been much discussed in the USA, where the majority of projects had a half life (i.e. the amount of time between fill placement and loss of at least 50% of this volume) of one to five years (Leonard et al., 1990). While less material has been published on the longevity of the smaller-volume Australian projects, it appears that it is less than the USA (Leonard et al., 1990) or European (Hanson et al., 2002) experience, with the majority of Australian nourishment projects occurring once a year or more often.

As elsewhere (Hanson et al., 2002), Australian nourishment practices generally occurred outside of summer months, which minimises impact on public beach amenity use. This also maximises the efficacy of the management intervention in building back beaches, by avoiding the months where sediment erosion is typically the norm (Davidson et al., 2010). Interestingly, however, projects were more common in spring than winter. This practice fails to consider that spring is typically the season of peak biological recruitment, during which ecological impacts of nourishment practices might be most severe (Peterson and Bishop, 2005).

4.3. Monitoring of the efficacy and impacts of nourishment practices

In order for nourishment practices to be a successful approach to beach management, rates of both long-term erosion and the temporal impacts of storms and wave climate must be considered

(Finkl and Walker, 2004). This requires information on the morphology, beach sediment characteristics, bathymetry, storm-induced variations and sediment budgets which relate to long-term stability (Gourley et al., 2004). Somewhat surprisingly therefore, very few LGAs conducted monitoring to inform the efficacy of their nourishment projects. Site specific evaluations are necessary to judge the success of a project, but alone they will not lead to the development of best practice guidelines. A coordinated National approach to assessing environmental changes following both dredge and nourishment (fill) projects would facilitate more effective processes, improve understanding of environmental impacts of soft engineering and their mitigation and minimise the duplication of expensive site-specific studies (Nordstrom, 2005). This approach would require more funds to be diverted towards coastal studies. In 2004 the National Committee on Coastal and Ocean Engineering (2004) estimated that less than 10% of average yearly expenditure on climate change research was directed at coastal and ocean environments.

The number of LGAs that had assessed ecological impacts of nourishment projects was similarly very small. This contrasts the USA, where biological monitoring at both dredge and fill sites is a common permit requirement (Peterson and Bishop, 2005). Consequently, the impact of nourishment projects on Australian sandy beach ecosystems remains poorly understood. Studies done elsewhere suggest that depending on the scale, timing and methods used, impacts may persist for many years (Peterson and Bishop, 2005; Peterson et al., 2006). Results from studies done elsewhere cannot, however, easily be extrapolated to Australian systems due to differences in the way projects are undertaken, as noted above, and the differing geomorphology of southern Australian beaches, where most nourishment projects are done, to USA beaches. In addition a high proportion of Australian fauna is endemic (Hobday et al., 2006) and sensitive to disturbance (Schlacher et al., 2008a; Skilleter et al., 2006).

4.4. Recommendations for future projects

This research has shown that nourishment projects in Australia are dominated by small scale operations with a short time interval between renourishment. Operations of this kind would be the cheapest and potentially have the least affect on recreational use of the beach while continuing to maintain or to reinforce the beach width for property protection (Psuty and Pace, 2009). Undertaking a number of smaller projects rather than a single large one was also the recommendation of Speybroeck et al. (2006) as they are of the

opinion that this will result in less environmental impact. However, at present this is difficult to determine as not enough is known of the effect of long-term cumulative impacts on the beach system (Lindquist and Manning, 2001; Peterson et al., 2006).

The coastline is affected by multi decadal to decadal climate variation in extreme wave and storm events (Cowell and Thom, 1997). Historical examination of records for South-eastern Australia has revealed regular clusters of high frequency and magnitude events (Goodwin et al., 2010). South-eastern Australia is entering a climate period of high storm frequency (Goodwin et al., 2010). Climatic intensity will influence our measures of efficiency of nourishment, such as longevity of fill, which have been established over the last 20 years of relative calm. Future nourishment events for south-eastern Australia may require more frequent nourishment intervals. Nourishment protocols need to be reevaluated in light of an increased frequency and impact of extreme storm events in order to ensure that these techniques are still cost effective (Gourley et al., 2004). Where beach nourishment continues to be used as a mechanism to protect coastal properties, the demand for new coastal development will be perpetuated (Leatherman, 1989; Nordstrom and Mauriello, 2001). The dependence on continuing maintenance of the coastal environment (Cooper and Lemckert, 2012) may be at odds with the policy of planned retreat adopted by many Australian States (Abel et al., 2011).

Nourishment projects need to distinguish the reason for management and plan projects differently according to whether they are a response to: (1) short-term sediment imbalance; (2) a long-term negative sediment budget; or (3) sea level rise. Sea level rise combined with local factors will increase world-wide shoreline recession (Stive et al., 1991). In the face of climate change current Australian nourishment practices may be unsustainable. A large number of small scale operations in Australia rely on using sediment from the same coastal compartment. Rising sea levels and increased erosion may mean that many borrow sites are not available and that the same amounts of aerial sand are not present. In some cases this may mean the increased use of sub-aerial fill (from the nearshore zone). Use of sand from this zone will impact beach protection (Basco, 1999b; Davis et al., 2001; Hilton and Hesp, 1996), making the coastal zone more vulnerable to storm surge. In other cases the most viable source of sand will be offshore, although this too is not an endless source (Leatherman, 1989) or without environmental concerns (Boyd et al., 2003; Nairn et al., 2004). An increase in offshore sand mining (legislation permitting) would likely change the way in which nourishment is conducted. Smaller, frequent projects may give way to larger, less frequent projects as part of a well defined schedule. The increased expense and difference in management style may mean that less nourishment is undertaken as a response to short-term erosion.

Beach scraping has been used in many cases to protect infrastructure as an immediate response to extreme weather events. However, if sea level rise and storm damage increases 'coastal squeezing' resulting in reduced beach width, scraping under these conditions would lead to over steepening of beaches and increased erosion. As the beach face sediment budget is inextricably interlinked with the dune system we therefore need to consider the effect of scraping on the whole beach system (Psuty, 1988).

While the number of sand bypassing projects is still relatively small in Australia, it is possible that this will increase. Along the coast of south-eastern Australia, Intermittently Open and Closed Lakes and Lagoons (ICOLLs) systems are often a temporary sand sink (Panayotou et al., 2007). Over half of the ICOLLs in NSW are regularly mechanically opened, usually to prevent flooding inundation of local infrastructure (Haines et al., 2006). With the increase in flooding risk and the demand of sand to nourish adjacent sediment starved beaches, it is possible that bypass systems may be

a future engineering solution. Future practice would benefit from a synthesis of previous monitoring associated with existing bypass projects and national guidelines.

The ecological integrity and value of sandy beaches are threatened by a range of stressors (Defeo et al., 2009). In order to protect sandy beaches and maintain the principles of effective and holistic beach management in Australia the following is required: inclusion of indicators in State of the Environment reporting; a stronger connection between management, monitoring and reporting; continued mapping of biodiversity, geomorphology and vulnerability to climate change (Banks and Skilleter, 2002; Sharples et al., 2009); and a regional approach of sandshed management (Revell et al., 2007). Coastal engineering should be aligned with concepts of ecological restoration (Psuty and Pace, 2009) and ecosystem health (Schlacher et al., 2008b).

In order to enable knowledge transfer, Basco (1999a) and others have recommended creating standards for reporting and collating information in a database regardless of the consent authority involved. Information such as fill volumes, sediment composition and purpose of the project (Jackson et al., 2010) are vital. Without this information useful insights are not communicated and it is not easy to determine whether a project has succeeded or not (Nordstrom, 2005). Additionally, reliable project design including cost assessment analysis relies on good monitoring of long-term beach behaviour (Gourley et al., 2004). Information obtained and properly disseminated would enhance our predictive capabilities and allow adaptive management (Stive et al., 1991).

5. Conclusions

According to our survey of coastal managers it appears that the number of beaches on which nourishment projects are undertaken in Australia may remain fairly steady for the next five years. The frequency and quantity of fill used may, however, increase in response to increasing extreme weather events. LGAs that are undertaking nourishment are generally doing so as part of an overall management plan for the beach area. Artificial nourishment is generally used together with other forms of management such as replenishment, scraping, grooming, installation or pre-existing hard structures, building codes and mandatory set-backs. In comparison to the USA and Europe, the extent and scale of nourishment projects in Australia is small. However, where it is occurring it seems to be repetitively undertaken within a smaller time scale.

As nourishment projects are a short-term stop gap measure used primarily for the protection of infrastructure, it is essential the natural time scale of shoreline fluctuation should be known before determining the frequency of nourishment programs. More information, particularly on details of past coastal engineering projects and baseline data on the coastal environment is required. However, working with the information we have and that from international sources, recommendations for optimal protocols can be established. Ecologically sustainable development and integrated coastal zone management are the two fundamental principles on which most coastal policy is based. Coastal management faces the challenge of achieving a sustainable future for the coastline while balancing environmental, economic, cultural and recreational needs.

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3. Small scale distribution of benthic invertebrates on an exposed sandy beach, North Carolina

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Abstract

Little is known about the spatial scales across which sandy beach meiofauna vary, and how this relates to the spatial scales of macrofaunal variation. This study used spatially nested sampling, spanning scales from tens of metres to centimetres, combined with spatial autocorrelation to compare spatial patterning in macrofaunal and meiofaunal communities on an exposed sandy beach of Bogue Banks, North Carolina. It also assessed how key traits of meiofauna, such as buccal morphology, body size and sex, which influence resource use, may explain patterns of aggregation. At a single tidal elevation of the upper swash zone, cores of sediment were collected from each of two sampling stations, separated by 20 m. Within each station, samples were collected from sampling stations, each 20 cm apart, and within each sub-station, samples were collected from even spacings of 5 cm. Although the environment appeared largely homogenous, spatial structuring of both sediments and fauna was observed. Macrofauna varied most at the scale of tens of metres whereas meiofauna varied at scales of tens of centimetres. Meiofauna displayed greater spatial dependency in the sample station with more homogenous sediment. The biological measures that were most closely related to patterns in sediment characteristics were nematode mean body length, sex ratio and feeding guilds. Patches of nematodes of different genera did not always overlap, and consequently, spatial patterns were contingent on the level of taxonomic resolution used. While examined in other marine environments, spatial autocorrelation has not been commonly used in studies of invertebrates of exposed sandy beaches. Inclusion of spatial autocorrelation as another factor in sampling protocols could increase our understanding of the dynamic sandy beach environment.

1. Introduction

It has long been noted that the distribution of organisms in the environment is spatially structured and may be related to underlying contiguous processes (Hutchinson 1953). In some environments, such as forests composed of habitat mosaics, the underlying environmental basis for spatial structuring is visually apparent (Elton 1966). In others, such as the sedimentary environments of seafloors, spatial structuring occurs despite a seemingly homogenous environment (Kraan et al. 2009). The exposed sandy beach environment is among those which, though homogenous in appearance, exhibits small-scale features determined by the influence of waves and tides (Jansson 1967, Jackson et al. 2002). How this

heterogeneity translates to patchiness in its sediment-dwelling invertebrates is, however, poorly understood.

Among the major groups living within the sandy beach environment are meiofauna (defined as organisms > 64 and $< 500 \mu\text{m}$) and macro-fauna ($> 500 \mu\text{m}$ in length). The patchiness of both meio- and macro-fauna may be induced by multiple ecological processes operating at more than one spatial scale. It has been posited that physical factors such as salinity, sediment characteristics and tidal exposure determine variation in abundance over large scales (metres to kilometres), while animal – habitat processes are more important on smaller scales (centimetres; Findlay 1981, Legendre et al. 1997). Animal-habitat interactions may include a response to physical variables such as microtopography of sediment, disturbance and the localised availability of food, or biological variables such as the distribution of biogenic structures, predators and/ or competitors (see review in Fleeger and Decho 1987). Reproductive activity and life cycle stage may also affect spatial distribution. Ontogenetic differences in feeding preferences and mate search might mean that males and females are found near each other (Coull 1988).

Patch formation is a dynamic process that reflects the response of organisms to an array of changing factors. The abundance of species in a patch is determined by: (1) the ability of the species to tolerate the conditions there, (2) the abundance of the species in nearby patches that serve as a source of colonists and (3) the environmental conditions of the habitat matrix surrounding the patch (Hodda 1990). Although macrofauna and meiofauna are each known to display small-scale variability, it is often assumed that meiofauna, because of their smaller size, will display greater small-scale spatial variability, rendering them less useful than macrofauna in ecological assessments (Kennedy and Jacoby 1999). The few studies that have considered spatial distributions of meiofauna have, however, found that although variable, their distribution is not completely random (Hodda 1990, Robertson and Freckman 1995). Meiofauna instead display patch formation that reflects small (cm) to medium (tens of m) environmental conditions (e.g. mean grain size; Hodda 1990). There have been few studies, however, that have directly compared the spatial scales across which macro- and meio-fauna vary through contemporaneous sampling (but see Raffaelli et al. 2000)

Here, we couple a spatially nested sampling design, spanning scales of tens of metres to centimetres, with spatial autocorrelation analyses to ascertain the spatial patterning of sandy beach macrofaunal and meiofaunal communities. Spatial autocorrelation assesses the scales at which adjacent samples can be considered statistically indistinguishable (Fortin and Dale 2005). Spatial autocorrelation has been used to examine spatial structuring of meio and macro - faunal communities in mud flats or deep sea environments (reviewed in Kraan et al. 2009), but has rarely been applied to exposed sandy beaches. We hypothesise that similar spatial patterning will be observed between meiofauna and macrofauna, reflecting patchiness in sediment characteristics. Furthermore, we hypothesise that spatial scales of variation will be similar between meiofaunal communities sorted to coarse taxonomic resolution and nematode communities sorted to genus. To help discriminate among potential drivers of spatial patterning, we also compare spatial patterning of meiofauna among body sizes, sexes and feeding guilds.

2. Methodology

2.1 Sampling methodology

The study site was located on an exposed sandy beach of Pine Knoll Shores, North Carolina (34°41'46.375"N, 76°47'20.883"W). Pine Knoll Shores forms part of Bogue Banks, a 40 km long barrier island that runs from east to west. The system is microtidal (spring tidal range = 1.3 m), and is an area of significant erosion due to storm and wind events. A 50 year Shore Protection Project designed by the US Army Corps of Engineers has included periodic sand renourishment on this beach, the most recent being in February 2008. At the time of sampling in July 2010, predicted low tide was at 0759 hours, wave height was 0.3 – 0.5 m and wave direction 170°, with winds from the north-west at 2 – 3 m/s. Regular cusps were observed on the beach. Samples were collected during low tide from a single transect running along the upper swash zone, just below the water table outcrop in the saturated sand (McLachlan and Turner 1994) at elevation 0.55 m above Mean Water Level. Sampling was focused on this tidal elevation because meiofaunal diversity and abundance is high in this zone (McLachlan 1980).

To test the null hypothesis that meiofaunal communities would display patchiness at the same scales as macrofauna, we used a nested sampling design, incorporating three spatial scales of sampling. A total of 50 samples were taken, 25 each from two sampling stations situated 20 m apart within the centres of contiguous beach cusps (Figure 1). Within each sampling station, samples were collected from five 20 cm – long sub-stations, each separated by 20 cm. Within each sub-station we collected five samples, with 5 cm intervals between core centres, running across the beach face (Figure 1). Cores were collected using a piston corer with a 2.5 cm internal diameter inserted vertically to a depth of 5 cm. This resulted in approximately 25 mL of sediment per sample. The core-size was based on its efficacy in sampling meiofauna (Findlay 1982). It is recognised that in choosing such a small core size, macroinvertebrates may have been under-represented. Nevertheless, this should not have influenced the goal of our study to document the spatial distribution of macrofauna, as opposed to estimating population density or diversity.

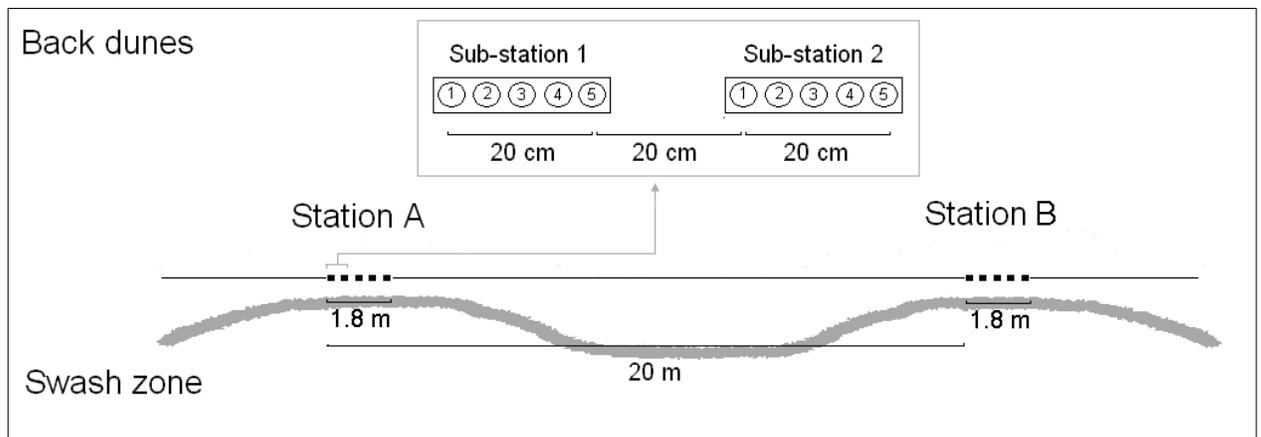


Figure 1: Within the upper swash zone, two stations (A and B), 20 m apart, were sampled. Within each, there were five sub-stations, each separated by 20 cm, in which five cores were taken at evenly spaced distances of 5 cm.

2.2 Laboratory procedures

Upon collection samples were immediately stored in plastic tubes inside a cooler. Within an hour samples were fixed in 10% buffered formalin and stained with rose bengal. Following 24 hours in formalin, samples were decanted (three times; following procedure in Somerfield and Warwick 1996) over a 64 μm sieve, macrofauna were then identified and picked out of samples. Examination of remaining sediment from a total of six samples indicated that on average 97.6% (± 0.6 SE) of meiofauna were removed from samples using this extraction technique. The volume of each sediment sample was measured to the nearest mL so that abundances of invertebrates per core could be standardised according to volume. The grain size (including particle sizes < 1.18 mm) and large shell fragments (particle size > 1.18 mm) was then assessed using a laser granulometer. The mean sediment grain size and sorting of each sample was determined using the Gradistat program (Blott and Pye 2001).

The enumeration by taxon of both macrofauna and meiofauna was done under a dissecting stereo-microscope (Wild M5). A mixed taxonomic resolution was used. For macrofauna this was to species where possible. For meiofauna this was to phylum, order, and subclass in different cases (*sensu* Warwick and Gee 1984). Nematodes were separated for further identification and examination of their body type as they have often been studied as indicators of changes to the biotic environment (Heip et al. 1985, Coull 1999, Kennedy and Jacoby 1999) and a relationship between body type and biotic conditions has been previously observed (Ferris 1980, Fegley 1987, Wallace et al. 1993). Each nematode was identified to genus, so as to assess impacts of taxonomic resolution on spatial patterning. To test hypotheses about the influence of key traits on spatial patterning of meiofauna, body length, sex and buccal cavity type of each was also recorded. The morphology of the buccal cavity was used to classify nematodes into four feeding guilds: selective and non-selective deposit-feeders, epistrate feeders and predators/omnivores (Moens and Vincx 1997). A small amount of nematodes (approximately 3%) were not in sufficient condition for their buccal cavity to be accurately recorded and were not included in this analysis.

2.3 Statistical analysis

The spatial scales across which whole communities of macrofauna and meiofauna, nematodes and sediment varied were analysed separately using PRIMER 6.0 software (Clarke and Warwick 2001). Univariate analysis was run on abundances of taxa identified by SIMPER as key contributors (PRIMER; Clarke and Warwick 2001) to significant differences in communities between stations and sub-stations. Univariate analysis was also conducted on the nematode ecotype which included: the mean body length, proportion of females, and the dominant buccal cavity types. Two way PERMANOVAs with the factors station (2 levels) and sub-station (5 levels, nested within each station) were run on each data set. The analyses of faunal communities used Bray-Curtis dissimilarity matrices of differences among samples produced following log (x+1) transformation of data. A dummy variable of 1 was added to the nematode and macroinvertebrate data set prior to the calculation of dissimilarity measures due to the large number of zeros in some samples (Clarke et al. 2006). Analyses on mean grain size and percent shell fragments were normalised to generate separate Euclidean distance matrices. Prior to analysis, sediment variables were examined for co-linearity, as a consequence mean sorting was removed from the analysis ($r^2 \geq 0.90$; Clarke et al. 1993). Spearman's rank correlations tested for relationships between biotic matrices and sediment matrices (RELATE procedure in PRIMER; 9999 permutations; Clarke et al. 1993).

To assess finer-scale patchiness in macrofauna, meiofauna, nematodes and sediments; correlograms were calculated for the taxa which contributed the most to variation (SIMPER). Univariate correlograms were constructed on total abundances of the groups at each of the two sampling stations using values of Moran's *I* (Moran 1950) calculated in SAM (Spatial Analysis in Macroecology; Rangel et al. 2010). Large positive or negative values correspond to strong positive or negative spatial autocorrelation respectively. Values of Moran's *I* near zero indicate data which are not spatially autocorrelated or are randomly distributed (Fortin and Dale 2005).

Moran's *I* detects aggregation by testing for significant differences between cores separated by a number of distances, with the distance between samples at which the *I* value changes from positive to negative sign corresponding to the average patch radius (Sokal and Oden 1978). Moran's *I* was calculated for 9

distance-classes between the centre point of cores, each of which was of approximately equal frequency in the sampling design: 5, 6-20, 21-35, 36-50, 51-70, 71-85, 86-110, 111-130, 131-180 cm (Legendre and Fortin 1989, Gallucci et al. 2009). Each correlogram was tested for global significance by checking at least one value was significant at the $\alpha' = \alpha/n$ Bonferroni-corrected significance level, where n is the number of distance classes (Oden 1984); each distance class was tested for significance ($\alpha = 0.05$; 9999 permutations).

3. Results

3.1 Scales of spatial variability in fauna and environmental factors

Across both of the sampling stations, the mean density of macrofauna was 0.11 ± 0.02 per mL and the mean density of meiofauna was 1.60 ± 0.08 per mL. Macrofaunal communities, comprising four taxa, the amphipod *Amphiporeia virginiana* (69%), the mole crab *Emerita talpoida* (16%), the surf clam *Donax variabilis* (11%) and the nudibranch *Aeolidina* (4%), differed significantly between stations, but not between sub-stations (Table 1a). By contrast, meiofaunal communities, dominated by nematodes (40%), copepod nauplii (31%), macrodaysid gastrotrichs (10%), proseriate turbullerians (8%), and kalyptorynch turbullerians (5%) did not differ between the two stations, but instead varied among the smaller spatial scale sub-stations (Table 1b, Figure 2b). The taxa that contributed the most to variation were also the most dominant for each group: for macrofauna these were *A. virginiana* and *E. talpoida*; for meiofauna, nematodes, copepod nauplii and macrodaysid gastrotrichs. Similar patterns of spatial variability were also seen in some of the univariate analysis; where variation in macrofauna taxa was evident between stations for *A. virginiana* (Table 1b), but not *E. talpoida* (Table 1c). There was some variation between meiofaunal taxa; copepod nauplii showed no significant variation (Table 1e), macrodaysid gastrotrichs showed significant variation at both scales (Table 1f) and nematode (phyla-level) only showed variation at sub-station levels (Table 1g).

Thirteen nematode genera were identified by this study, of which three were numerically dominant: *Mesacanthoides* (56%); *Araeolaimus* (20%); and *Spirinia* (12%). There was no significant variation in the nematode community between the two stations, while there was significant variation at the sub-station

level (Table 1h). Univariate analysis of dominant taxa revealed the same pattern with the exception of *Mesacanthoides* (Table 1i) which did not show significant variation at either scale. Body length differed both between stations and among substations (Table 1l). While the average body length of females was larger ($1818 \pm 42 \mu\text{m}$) than males ($1773 \pm 37 \mu\text{m}$), this difference was not significant (PERMANOVA: Pseudo- $F_{1,360} = 0.51$, $p = 0.47$). The proportion of females showed no significant variation at either scale (Table 1m). The taxa and feeding guilds that contributed the most to variation were those with armed buccal cavities, with epistratum feeders comprising 31% and predators/ omnivores 54% of the assemblage. Selective and non selective deposit feeders made up less than 12% of the total. Scales of spatial variation in dominant buccal cavity morphotypes (Table 1 n,o), generally followed those of nematode genera.

Sediment mean grain size, like meiofaunal community structure, was similar between stations but varied among sub-stations (Table 1p) and shell fragments differed both between stations and among sub-stations (Table 1q). Station A had significantly less shell fragments and coarse material, resulting in a more homogenous environment. The meiofaunal community structure was weakly but significantly correlated with the proportion of shell fragments (Rho = 0.15, $p = 0.043$), as was the abundance of *A. virginiana* (Rho = -0.13, $p = 0.029$) and macrodaysid gastrotrichs (Rho = 0.16, $p = 0.029$). Copepod nauplii were significantly negatively correlated with mean sediment grain size (Rho = -0.10, $p = 0.007$). The abundance of nematode epistrate feeders (Rho = -0.08, $p = 0.047$) and the proportion of females (Rho = -0.11, $p = 0.007$) were also weakly negatively correlated with mean grain size.

Table 1: PERMANOVAs examining sources of spatial variability in **macrofauna**: (a) macrofauna community, (b) *A. virginiana*, (c) *E. talpoida*; **meiofauna**: (d) meiofauna community, (e) copepod nauplii, (f) macrodaysid gastrotrich, (g) nematode (phylum level); **nematode**: (h) nematode community (genus level), (i) *Mesacanthoides*, (j) *Spirinia*, (k) *Araeolaimus*, (l) nematode mean body length, (m) proportion of females (%), (n) epistrate feeders, (o) predator / omnivore; and **sediment characteristics**: (p) mean grain size, (q) large shell fragments (%). Factors: station (2 levels, A and B, random) and sub-station (5 levels, nested in station, random). Faunal community data were log (x+1) transformed prior to analysis, sediment variables were normalised prior to analysis Terms significant at $p < 0.05$ are highlighted in **bold**.

Sources of variation	Df	a) Macrofauna community			b) <i>A. virginiana</i>			c) <i>E. talpoida</i>		
		MS	F	P	MS	F	P	MS	F	P
Station	1	537.26	17.41	0.007	364.12	28.35	0.009	0.36	0.23	0.649
Sub-station	8	30.86	1.70	0.075	12.84	1.25	0.294	1.61	0.86	0.558
Residuals	40	18.20			10.30			1.87		

Sources of variation	Df	d) Meiofaunal community			e) copepod nauplii			f) macrodaysid gastrotrich		
		MS	F	P	MS	F	P	MS	F	P
Station	1	1822.40	2.44	0.076	444.04	1.16	0.308	338.04	7.71	0.028
Sub-station	8	748.22	1.54	0.041	383.38	0.49	0.927	43.82	2.29	0.035
Residuals	40	485.28			789.94			19.14		

Sources of variation	Df	g) nematode (phylum)			h) nematode (genus)			i) <i>Mesacanthoides</i>		
		MS	F	P	MS	F	P	MS	F	P
Station	1	185.18	0.22	0.874	182.82	0.63	0.631	0.55	0.01	0.962
Sub-station	8	831.78	2.06	0.022	288.07	2.13	0.002	94.29	1.91	0.081
Residuals	40	403.06			135.31			49.27		

Table 1. cont.

Sources of variation	j) <i>Spirinia</i>				k) <i>Araeolaimus</i>			l) nematode mean body length		
	Df	MS	F	P	MS	F	P	MS	F	P
Station	1	27.40	0.97	0.352	16.83	0.28	0.631	<0.01	9.19	0.032
Sub-station	8	28.12	2.41	0.031	59.90	2.57	0.024	<0.01	4.00	0.001
Residuals	40	11.67			23.30			63006.00		

Sources of variation	m) nematode proportion of females (%)				n) nematode epistrate feeder			o) nematode predator / omnivore		
	Df	MS	F	P	MS	F	P	MS	F	P
Station	1	16.65	0.02	0.885	19.39	0.01	0.993	180.02	0.11	0.931
Sub-station	8	670.92	2.06	0.061	1335.40	2.02	0.041	1704.50	2.82	0.006
Residuals	40	326.11			660.14			604.31		

Sources of variation	p) Mean grain size (μm)				q) Large shell fragments (%)		
	Df	MS	F	P	MS	F	P
Station	1	93.65	0.02	0.895	1162.30	5.77	0.025
Sub-station	8	4710.80	9.36	<0.001	201.60	7.05	<0.001
Residuals	40	503.07			28.59		

3.2 Small scale patch formation

Macrofaunal taxa did not show strong spatial autocorrelation across the range of scales considered (0-180 cm; Figure 2a,b). Amphipods showed spatial autocorrelation at 21-35 cm, but not at smaller distances apart, indicating inconsistent spatial structuring (Figure 2a). Meiofaunal taxa varied in the extent of patch formation. Copepod nauplii did not show any small scale patterns (Figure 3a); while the other taxa generally showed smaller scale variation (Figure 3b-f), reflecting the PERMANOVA analysis. In the case of nematodes, the correlogram indicated a patch size of less than 20 cm. When nematode genera were analysed separately, different spatial patterns were seen, although patch sizes remained similar (Figure 3d-f). Examination at the taxonomic level of nematode genera found that samples collected less than 6 cm apart are likely to be strongly spatially autocorrelated (Figure 3e,f), and therefore not independent. The only biotic variables that appeared to show similar patterning to each other were the ecotype variables of nematode mean body length, proportion of females and feeding guilds (Figure 4a-d); which also revealed similar patterns to sediment variables. The sediment variables displayed strong spatial structuring in both stations, with patch sizes ranging from approximately 21 to 70 cm (Figure 5a,b). In general, small scale spatial patterns were strongest at sample station A.

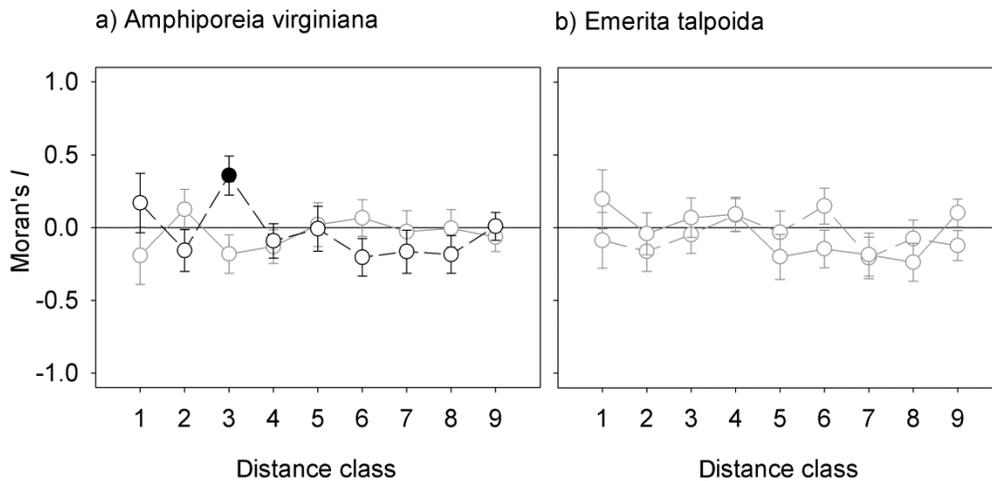


Figure 2: Spatial correlograms for the macrofauna: (a) *Amphiporeia virginiana* and (b) *Emerita talpoida*. Data from the two stations were analysed separately and are depicted by the two different sets of points on each correlogram. Station A is represented by solid lines between the symbols and station B by dashed lines. Closed symbols represent significant values of the Moran's I statistic ($p < 0.05$), open are non-significant. Black lines represent globally significant patterns, grey are non-significant. Distance classes labelled 1 to 9, correspond to the following distances (in cm) between samples: 5, 6 - 20, 21 - 35, 36 - 50, 51 - 70, 71 - 85, 86 - 110, 111 - 130, 131 - 180.

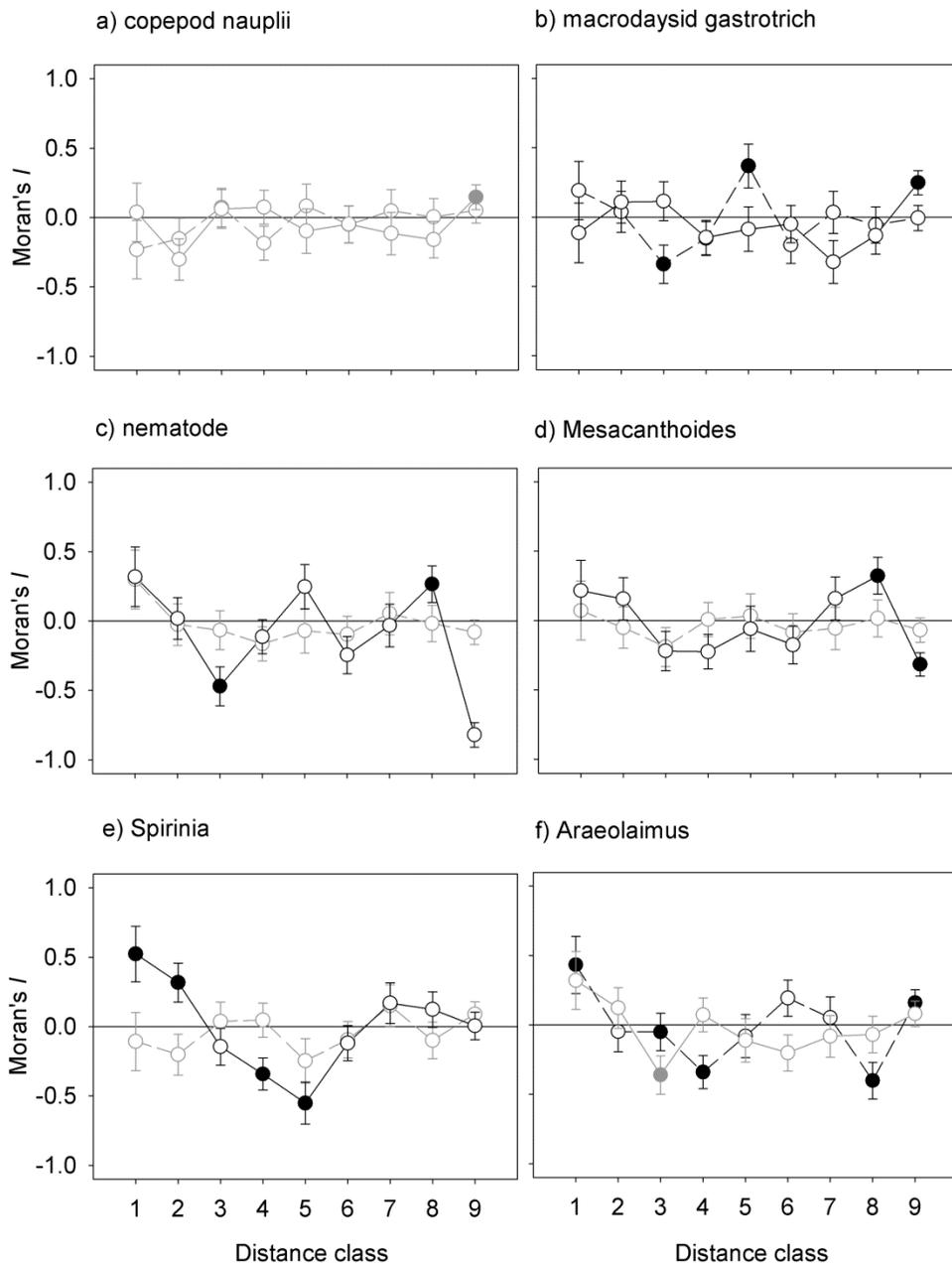


Figure 3: Spatial correlograms of dominant meiofauna: (a) copepod nauplii, (b) macrodaysid gastrotrich, (c) nematode; and nematode genera (d) *Mesacanthoides*, (e) *Spirinia* and (f) *Araeolaimus*. Station A is represented by solid lines between the symbols and station B by dashed lines. Closed symbols represent significant values of the Moran's I statistic ($p < 0.05$), open are non-significant. Black lines represent globally significant patterns, grey are non-significant. Distance classes labelled 1 to 9, correspond to the following distances (in cm) between samples: 5, 6 - 20, 21 - 35, 36 - 50, 51 - 70, 71 - 85, 86 - 110, 111 - 130, 131 - 180.

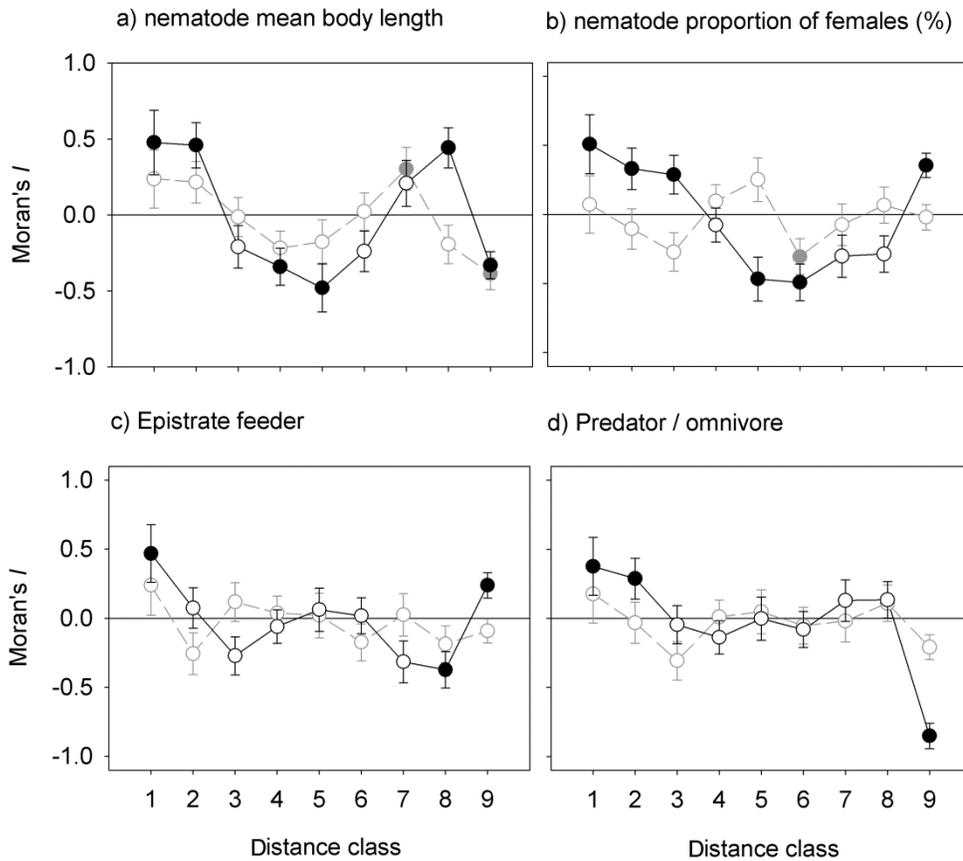


Figure 4: Spatial correlograms for nematode ecotypes: (a) mean body length, (b) proportion of females, (c) epistrate feeder and (d) predator/ omnivore. Station A is represented by solid lines between the symbols and station B by dashed lines. Closed symbols represent significant values of the Moran's I statistic ($p < 0.05$), open are non-significant. Black lines represent globally significant patterns, grey are non-significant. Distance classes labelled 1 to 9, correspond to the following distances (in cm) between samples: 5, 6 - 20, 21 - 35, 36 - 50, 51 - 70, 71 - 85, 86 - 110, 111 - 130, 131 - 180.

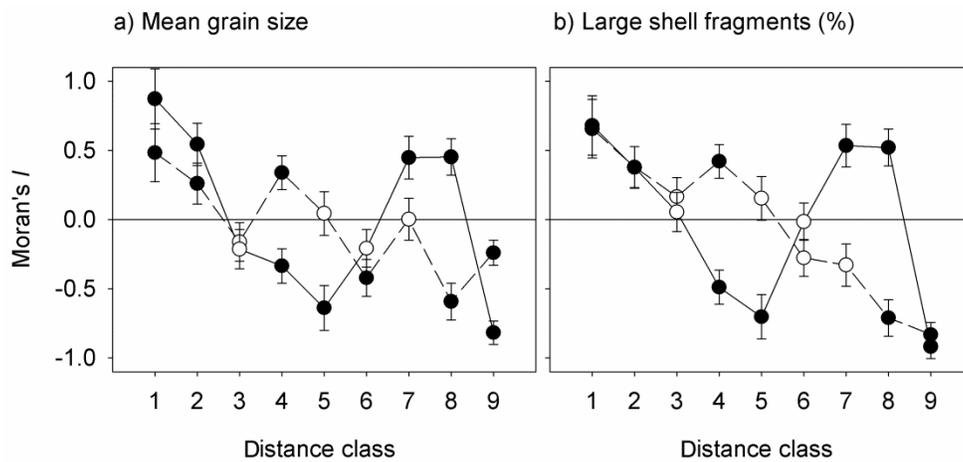


Figure 5: Spatial correlograms for sediment variables: (a) mean grain size and (b) large shell fragments (%). Station A is represented by solid lines between the symbols and station B by dashed lines. Closed symbols represent significant values of the Moran's *I* statistic ($p < 0.05$), open are non-significant. Black lines represent globally significant patterns. Distance classes labelled 1 to 9, correspond to the following distances (in cm) between samples: 5, 6 - 20, 21 - 35, 36 - 50, 51 - 70, 71 - 85, 86 - 110, 111 - 130, 131 - 180.

4. Discussion

Contrary to our hypothesis, macro and meio – fauna displayed different patterns of spatial structuring on an exposed sandy beach of Bogue Banks, North Carolina. For the spatial scales considered, macrofauna displayed greatest spatial variation at the scale of tens of metres. Meiofauna, by contrast, displayed greater spatial structuring at the scale of tens of centimetres. Whereas at the larger spatial scales, coarse (phyla-level) taxonomic groupings of meiofauna displayed correlated spatial patterns to nematode genera and morphotypes, this relationship collapsed at finer spatial scales. The communities of macro- and meiofauna found on Bogue Banks were similar in composition to those documented previously (e.g. Hummon et al. 1976, Peterson et al. 2000). The dominance of three nematode genera, in particular *Mesacanthoides* is not unusual as other studies have also shown that typically relatively few taxa account for majority of specimens (Hodda and Nicholas 1985).

Spatial patterns of macrofauna may be determined by hydrodynamics (Lastra et al. 2002), resource patches (Duong and Fairweather 2011) or site-specific factors (James and Fairweather 1996); generally resulting in larger aggregation patterns (Thrush et al. 1989). Here, variation in sediment grain size and shell hash was evident at the between sampling sites scale of tens of metres. Correlations between macrofauna and sediments that spanned all spatial scales were, however, not significant. Hence macrofaunal distributions may also have reflected food availability, positive or negative interactions among the species (Decho and Fleeger 1988) and the dispersal abilities of taxa (Büchi and Vuilleumier 2012).

Previous studies have found that spatial patterns in meiofauna are sensitive to the taxonomic resolution used (Sandulli and Pinckney 1999). Genera of constituent groups, such as nematodes, may display divergent structural patterning to one another, which tends to homogenise spatial patterning when considered at the scale of phyla (Hogue 1982). Such variation among genera, also found in the present study, may relate to differences in resource requirements or dispersal. Holling's (1992) theory that the distribution of body sizes may reflect discontinuities in the habitat appears to be supported by the close relationship of nematode mean body length and sediment characteristics. This relationship was the strongest in the most homogenous sediment which contained smaller amounts of shell fragments and

broader patch sizes of similar-sized material. The distribution of nematode body lengths may be influenced by distribution of the sexes and the feeding mode of dominant genera (Tita et al. 1999). Many predatory species tend to be larger; in addition their presence will influence the distribution of other nematodes and meiofauna in their vicinity (Gambi et al. 2008, Moreno et al. 2011).

Spatial dependency violates the standard statistical techniques that assume independence among observations. Rather than ignoring spatial dependency there could be a lot to learn from incorporating it into standard survey and monitoring techniques for marine benthos. The benefits and applications of this approach also include determining the appropriate sampling design in terms of scale in order to obtain results but minimise the sampling effort (Legendre et al. 1997, Tobin 2004). This study confirms that the collection of a large number of small samples is required to provide information about spatial patterns at the small (cm) to medium (tens of m) scale (Heip et al. 1985). This will also ensure that samples are representative of an entire site (Dungan et al. 2002) and avoid pseudoreplication (Morrisey et al. 1992). In addition, it appears that spatial variation is better explained by increasing taxonomic resolution in small scale studies; and that distribution of some taxa may be better explained by examining ecotypes and morphometrics (Robson et al. 2005). Studies such as this serve as important pilot studies for environmental impact assessments and experimental studies that require knowledge of spatial patterning of fauna and environmental variables.

In addition to being central to the design of ecological investigations (Schratzberger et al. 2008), an understanding of the spatial scaling of biota in sandy beaches underlies an understanding of their processes. The distribution and heterogeneity of sediment is the result of geomorphologic processes, which in turn produces congruent patterns of biota (Legendre and Fortin 1989). This study indicates that the extent of sediment heterogeneity may be influential on the spatial structure of fauna. Examination of spatial dependence is key to a greater understanding of these relationships. Identifying the relevant scales of variation enables better design of studies (Olabarria and Chapman 2001). Furthermore, with this information, replicate samples can be positioned at appropriate distances apart to maintain spatial independence, an assumption of many statistical analyses. By ascertaining the spatial scales across

which fauna and key environmental variables naturally vary, appropriate spatial controls, and replication can be included in study designs to enable detection of treatment effects over and above background variation (Schoeman et al. 2000).

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4. Spatial scales of variation in meiofauna on three embayed beaches, Sydney, Australia

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Abstract

The design of meaningful ecological assessments for determining human impacts to ecosystems hinges upon an understanding of the spatial structuring of their key biotic components. Meiofauna are a ubiquitous and ecologically important component of sandy beach ecosystems, yet the spatial scales across which they display baseline variability, particularly in anthropogenically disturbed urban settings, remains largely unknown. As meiofauna contain many taxa with limited dispersal, it may be expected that their populations exhibit a high degree of spatial structuring. Along an urbanised stretch of the south-east Australian coast, we sampled meiofaunal communities at regular (0.5 m) intervals along geomorphically similar stretches of three adjacent intermediate beaches to test the hypotheses that: (1) among the three beaches there would be significant differences in meiofaunal communities, (2) within each beach, meiofauna would be spatially patchy at the scale of metres, and (3) similar spatial patterns would be evident, irrespective of whether we described meiofaunal communities at coarse taxonomic resolution, or focused only on nematodes, sorted to genus. Consistent with these hypotheses, both meiofaunal and nematode communities significantly differed among the three beaches. Differences among beaches were driven by spatial variation in the abundance of key meiofaunal taxa among localities, rather than differences in species richness; and were not consistently correlated with the proximity of the three linearly arranged beaches to one another. Within beaches, taxa generally exhibited patchiness at the scale of 2 – 4.5 m, although nematodes were the exception. Our study highlights that even seemingly similar sandy beaches can exhibit significant natural variation in their faunal communities. Hence, in order to detect impacts of human disturbances over this background variability, rigorous experimental designs that are spatially as well as temporally controlled are required.

1. Introduction

Meiofauna, are benthic metazoans passing through a 500 μm sieve but retained on meshes of 40–64 μm (Giere 2009). They are a key component of coastal ecosystems which provide a significant proportion of benthic production (Platt and Warwick 1983). On sandy beaches, meiofauna inhabit the interstitial spaces between the grains of sand, attaining total densities that exceed 1.29×10^7 individuals per m^2 of sediment surface (Kennedy and Jacoby 1999). Meiofauna facilitate biomineralization of organic matter and enhance nutrient regeneration (reviewed in Coull 1999). They play a disproportionately large role in carbon cycling (Heip et al. 1985) and are an important food source for higher order consumers (reviewed in Coull 1999).

Sandy beaches are increasingly being modified by coastal engineering aimed at protecting coastal infrastructure and public beach amenities from beach erosion (Bishop et al. 2006, Peterson et al. 2006). In order to ascertain how engineering practices may be adapted to minimise ecological impacts to sandy beach ecosystems, assessments of impacts to meiofauna, a key component of the sandy beach ecosystem, are required. The appropriate design of ecological impact assessments requires knowledge of the temporal and spatial scales of baseline variability in meiofauna (Jackson et al. 2002). In the case of urbanised environments, this baseline is frequently in an already modified state.

Despite the importance of carefully designing ecological impact assessments to control for background temporal and spatial variation (Fraschetti et al. 2006), the spatial scales across which meiofaunal communities vary remain poorly known. Of the studies that have considered the scales of spatial variation in sandy beach meiofaunal communities, most have focused on how their abundance and diversity vary across scales of 100's of metres to kilometres according to large-scale gradients in grain size, wave height and beach slope (e.g. Rodriguez et al. 2001, Rodriguez et al. 2003, Gheskiere et al. 2005, Lee and Riveros 2012). On larger scales meiofauna may also display spatial heterogeneity that reflects dispersal processes (Lee and Riveros 2012). Approximately 98% of sandy beach meiofauna do not produce pelagic larvae (Remane 1952) and many are only dispersed when they are suspended by waves and currents (Bell and Sherman 1980). This limited dispersal may lead to spatial structuring of meiofaunal populations among beaches, even in the absence of environmental gradients.

By contrast, relatively few studies have considered the smaller-scale variability of sandy beach meiofaunal communities (but see Coull and Feller 1988), particularly along highly urbanised coastlines. Patchiness in sediment properties may lead to small-scale spatial structuring of invertebrates on the scales of centimetres to metres (Thrush et al. 1989, Gallucci et al. 2009, Kraan et al. 2009). It is unclear, however, whether ongoing managerial interventions such as beach grooming result in a more temporally and spatially homogeneous baseline against which to assess additional disturbances. Small scale spatial structuring of meiofaunal communities, and their components, may be examined using spatial autocorrelation (Legendre et al. 1997). Samples are collected at fixed (and increasing) distances apart and the technique is used to assess spatial dependency.

Here we combine broad-scale sampling of three geomorphically similar beaches in geographic proximity with fine-scale, regularly spaced sampling within beaches, to assess the spatial heterogeneity of sandy beach meiofaunal communities of embayed intermediate beaches along an urban stretch of coastline. We hypothesise that: (1) among the beaches there will be significant differences in meiofaunal communities despite their morphological similarity, (2) within each beach meiofauna will be spatially patchy, despite similar conditions, and (3) similar spatial patterns will be evident, irrespective of whether we describe meiofaunal communities at coarse taxonomic resolution, or focus only on nematodes, sorted to genus. Ascertaining the scales of baseline spatial heterogeneity in meiofaunal communities of urbanised shorelines, and the dependency of these on taxonomic resolution is essential for developing survey designs for rigorous ecological impact assessments.

2. Materials and methods

2.1 Study site and sampling methods

Our study considered embayed intermediate beaches (*sensu* Short 2007) along the northern shore of Sydney, Australia. Each is separated by rocky headlands and attached reefs that extend a few hundred metres offshore; creating compartments that are closed to sediment exchange. All three beaches are deeply embayed and orientated towards the east, with the southern third of the shoreline protected from high energy waves from the south-south-east direction. Embayed intermediate beaches dominate the highly urbanised coastline of south-eastern Australia (Short and Wright 1981) and are under increasing

threat from coastal squeeze, yet have received little ecological study. Three morphologically similar beaches were sampled: Collaroy-Narrabeen Beach, Newport Beach and Palm Beach (Figure 1). The dominant forms of each beach are transverse bar and rip grading to rhythmic bar and beach (Short 2007). Each beach was comprised mainly of 'marine' derived medium size quartz grains with shell fragments (Short and Wright 1981) and experienced a spring tidal range averaging 1.6 m. Each beach was subject to regular cleaning (grooming) during the spring-summer period. Although beach nourishment occurs on Collaroy-Narrabeen Beach, the section examined in this study was outside of the immediately affected area.

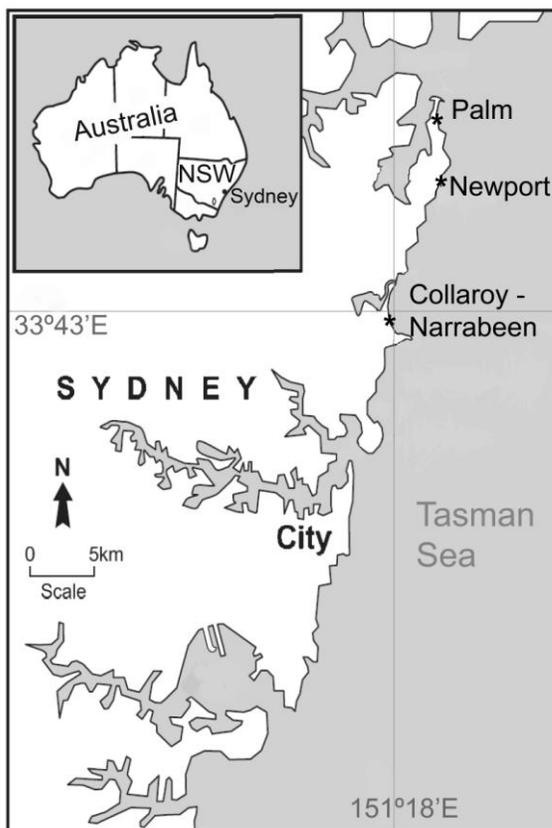


Figure 1: Location of study site showing the three beaches: Collaroy-Narrabeen, Newport and Palm Beach, New South Wales (NSW), Australia. Stars indicate the location of the 15 m transects on each beach.

On each beach we identified a section of shoreline with similar wave obliquity, shore slope and distance from the southerly headland to the other beaches (see Table 1 for a full summary of the environmental characteristics of each). Along each stretch of shoreline, we established a 15 m long transect that followed the upper swash zone, in the saturated sand just below the water table outcrop (McLachlan and Turner 1994). This vertical elevation of the beach profile was selected for sampling as it supports high densities of meiofauna (McLachlan and Turner 1994). A single alongshore transect was used in order to analyse variation in one direction only. Across shore variation was not examined here (Nicholas and Trueman 2009). On a low tide in April 2011, we simultaneously collected 30 sediment cores of 3.5 cm internal diameter and 10 cm depth from each transect, at regular intervals of 0.5 m along the shore of each beach. Simultaneous sampling of the three beaches avoided confounding a spatial comparison of the sites with the temporal effect of differences in the state of the tide.

Table 1: Characteristics of the three beaches. Unless otherwise stated, means \pm standard errors are presented for each, on the date of sampling. Wave characteristics are calculated based on four replicates; pH, temperature and salinity are based on three replicates; and sediment characteristics on 30 replicates per beach.

Parameter	Collaroy-Narrabeen	Newport	Palm
Location	33° 43.29' S, 151° 18.01' E	33° 39.19' S, 151° 19.44' E	33° 35.60' S, 151° 19.53' E
Management history ^a	Regularly groomed, some sections outside of the study area nourished, heavily urbanized	Regularly groomed, moderately urbanised	Regularly groomed, sand occasionally scraped from the fore to back shore in some sections outside of the study area, moderately urbanised
Total beach length (km) ^b	3.6	1.3	2.3
Wave period day of sampling (seconds)	7.5 \pm 0.6	11.9 \pm 1.2	9.9 \pm 1.3
Wave height (m)	0.5 \pm 0	0.8 \pm 0.1	0.7 \pm 0.1
pH	7.7 \pm 0.7	8.4 \pm 0	8.4 \pm 0.1
Salinity (practical salinity scale)	37.4 \pm 0.2	37.6 \pm 0.1	36.4 \pm 1.2
Sediment temperature (°C)	23.2 \pm 0	22.7 \pm 0.2	24.2 \pm 0.2
Slope of sampled section of the beach (°)	5	2.5	7
Mean grain size (μm)	492.9 \pm 5.6	404.9 \pm 3	360.7 \pm 1.5
Sediment sorting (μm)	244.6 \pm 4.6	175.2 \pm 3.5	135.3 \pm 2.2
Sediment skewness (μm)	1.6 \pm 0	2.3 \pm 0	2.3 \pm 0
Sediment kurtosis (μm)	5.1 \pm 0.2	10.2 \pm 0.3	12.1 \pm 0.2
Shell content (%)	42.5 \pm 1	44.6 \pm 0.9	40.5 \pm 0.7

^a Pers comm. Warringah and Pittwater Local Government Authorities

^b Short 2007

2.2 Laboratory procedures

On return to the laboratory, meiofaunal samples were fixed in 4% buffered formalin in sea water and stained with Rose Bengal. The size of each sample was measured to the nearest mL in order to standardise meiofaunal abundance to the volume of each core. Meiofauna were extracted onto a 64 μm sieve following the decantation procedure of Somerfield and Warwick (1996). Four decantations were undertaken, to produce an extraction efficiency of 96 - 100% (determined by examining the remaining sediment of three random samples). Meiofauna were counted and identified to coarse taxonomic levels (phylum, order, and subclass in different cases; *sensu* Warwick and Gee 1984) under the dissection microscope. All nematodes were extracted and prepared on slides following procedures outlined by Somerfield and Warwick (1996). Nematode specimens were identified to genus using the descriptions by Platt & Warwick (1983, 1988) and Warwick et al. (1998) and primary taxonomic literature.

Once the meiofauna were extracted from sediments, each sample was then oven dried (105°C for 24 hours) for granulometric analysis. Particle size analysis was conducted using sieves at half phi intervals from 1000 to 75 μm following methodology outlined in Folk (1974). Sediment was analysed with the GRADISTAT program (Blott and Pye 2001). The calcium carbonate content, as a percentage of total weight, was determined by weighing dried sediment samples before and after acid digestion with 4M HCl (Gross 1971).

2.3 Statistical analysis

2.3.1 Differences in meiofaunal and nematode communities among beaches

To examine spatial variation in: (1) whole communities of meiofauna, (2) nematode genera among the three beaches, and (3) to assess how this spatial variability correlated to spatial variability in environmental variables, such as sediment characteristics; multivariate statistical analyses were run in PRIMER (Clarke and Warwick 2001). Analyses of fauna used Bray–Curtis dissimilarities calculated between samples following square-root transformation of data. Analyses of environmental variables utilised Euclidean distance matrices, calculated following normalisation across variables. A dummy variable of 1 was added to the nematode data set prior to the calculation of dissimilarity measures due to the large number of zeros in some samples (Clarke et al. 2006).

Multi-dimensional scaling (MDS) plots derived from Bray–Curtis similarity matrices were used to view differences in the structures of meiofaunal and nematode communities among beaches, and between cores within beaches (Clarke et al. 2006). Analogous plots, calculated from Euclidean distance matrices, examined spatial variation in the suite of sediment variables measured (mean grain size, sorting, skewness, kurtosis and calcium carbonate content). The SIMPROF (similarity profile) routine was used to determine statistically distinguishable clusters of samples among the data (Clarke et al. 2008), the extent of which was displayed on MDS plots. To test for statistically significant differences in meiofaunal and nematode communities, meiofaunal abundance and richness, and sediment variables, among the three beaches: we ran one-way distance-based permutational multivariate analyses of variance (PERMANOVAs; Anderson 2001) on each data set. Sediment mean sorting and kurtosis were not analysed as they were highly co-linear with mean grain size ($r^2 \geq 0.90$). Where significant effects were found, PERMANOVAs were followed by pair-wise *a-posteriori* tests to identify sources of differences. Following PERMANOVA analyses on faunal communities, the similarity percentages procedure (SIMPER routine of PRIMER) identified those taxa (with a dissimilarity to standard deviation ratio of >1.3 ; Clarke and Warwick 2001) that were the best discriminators of multivariate differences between levels of significant factors (Clarke 1993).

We used a non-parametric Mantel test (RELATE procedure in PRIMER; Clarke et al. 1993) to examine the degree to which the meiofaunal (excluding nematode abundances) and nematode communities were spatially correlated. RELATE computes a Spearman correlation coefficient between the two Bray-curtis faunal matrices, with the significance of this correlation tested using a randomisation/ permutation test (9999 permutations). In order to examine the relationship between biotic communities and the sediment variables we used Spearman's rank correlation coefficient on the similarity matrices for both biotic and sediment variables (BEST routine of PRIMER, 9999 permutations; Clarke and Warwick 2001).

2.3.2 *Within-beach patch size of nematodes and meiofauna*

Within each beach, we assessed the spatial autocorrelation (SA) of faunal communities, key taxa and sediment variables across our evenly spaced samples. SA procedures quantify the ecological similarity between samples as a function of geometric distance, i.e. the spatial dependency of the samples (Fortin

and Dale 2005). Large positive values of Moran's I or Mantel r indices indicate strong positive SA, large negative values indicate strong negative SA and values near 0 indicate data are spatially uncorrelated or randomly distributed. SA indices were calculated for 8 distance classes (in metres: 0.5 - 1, 1.1 - 2, 2.1 - 3.5, 3.6 - 4.5, 4.6 - 6, 6.1 - 7.5, 7.6 - 9.5, 9.6 - 14.5), each of approximately equal frequency (Legendre and Legendre 1997). Each correlogram was tested for global significance by checking at least one value was significant at the $\alpha' = \alpha/n$ Bonferroni-corrected significance level (where n = number of distance classes; Oden 1984).

Spatial autocorrelation was assessed in the multivariate data sets of: (1) faunal abundances by taxon and (2) sediment variables (mean sediment grain size, skewness and calcium carbonate). Mantel's r was calculated, according to the procedure of Legendre and Fortin (1989), using the transformed Bray-Curtis dissimilarity matrix for meiofaunal and nematode communities and the Euclidean distance matrix for environmental variables. To examine within-beach spatial patterns in: (1) individual taxa identified by PRIMER as key contributors to differences in meiofaunal and nematode communities among beaches, and (2) the sediment characteristics: mean sediment grain size, sediment skewness and calcium carbonate content, we calculated univariate Moran's I (Moran 1950) using Spatial Analysis in Macroecology (SAM; Rangel et al. 2010). The statistical significance of SA within variables was assessed using permutation testing (9999 permutations).

3. Results

3.1 Differences in meiofaunal and nematode communities among beaches

Using the coarser, mixed taxonomic resolution, a total of 15 meiofaunal taxa were identified from the three beaches. Harpacticoid copepods were numerically dominant, accounting for 34% of the total meiofauna. Other prevalent taxa were gastrotrichs (24%), turbellarians (22%), nematodes (7%) and ostracods (4%). Across all taxa and beaches, the average density of meiofauna was $3 \pm 0.2 \times 10^6$ per m^2 . Each of the three beaches displayed distinct meiofaunal communities (Figure 2a; PERMANOVA: Pseudo- $F_{2,87} = 84$, $p < 0.001$). The richness of taxonomic groups did not differ significantly among beaches (PERMANOVA: Pseudo- $F_{2,87} = 0.7$, $p = 0.506$), but the total number of meiofauna did (Pseudo- $F_{2,87} = 60$, $p = 0.001$).

Harpacticoid copepods, gastrotrichs and nematodes were the three groups contributing most to multivariate differences in meiofaunal communities among beaches (SIMPER, Table 2a).

Overall, the spatial variation in nematode genera was significantly positively correlated to spatial variation in meiofaunal communities ($Rho = 0.347$, $p < 0.001$). However, the MDS ordinations indicated some differences in spatial patterns between the two data sets (Figure 2b). A total of 34 Nematode genera belonging to 12 families were identified. The genera *Xyala* and *Enoploides* were the most dominant, accounting for 32% and 24% of all nematodes respectively. Other abundant genera were *Ascolaimus* (9%), *Paracyatholaimus* (8%) and *Oncholaimus* (8%). As with meiofaunal communities, the abundance of nematodes significantly differed among beaches (Figure 2b; PERMANOVA Pseudo- $F_{2,87} = 48$, $p < 0.001$), with each significantly different to the others (post-hoc tests: $p < 0.001$). The nematode genera responsible for the majority of the average dissimilarities between beaches were *Enoploides* (comprising 80% of all genera on Collaroy-Narrabeen Beach, 26% on Newport and 6% on Palm); *Enoplaimus* (Collaroy-Narrabeen: 1%, Newport: 34% and Palm: 6%); and *Xyala* (Collaroy-Narrabeen: 3%, Newport: 2% and Palm: 49%; Table 2b). The taxon richness of nematode genera (PERMANOVA: Pseudo- $F_{2,87} = 31$, $p < 0.001$), and the total number of nematodes (PERMANOVA: Pseudo- $F_{2,87} = 44$, $p < 0.001$) significantly differed among beaches.

The SIMPROF analysis showed significant structure separating the sediment characteristics of the three beaches and some significant structure within Collaroy-Narrabeen Beach (Figure 2c). Each beach differed to the others in mean sediment grain size (PERMANOVA: Pseudo- $F_{2,87} = 319$, $p < 0.001$), skewness (PERMANOVA: Pseudo- $F_{2,87} = 125$, $p < 0.001$) and calcium carbonate content (PERMANOVA: Pseudo- $F_{2,87} = 5$, $p = 0.005$). Post-hoc tests showed each beach to be significantly different from the others ($p < 0.001$). The subset of sediment variables which best explained spatial variation in meiofaunal communities consisted of mean sediment grain size and skewness ($Rho = 0.325$, $p < 0.001$). The abiotic variable which best explained patterns in nematode communities was mean grain size ($Rho = 0.453$, $p < 0.001$).

Table 2: Mean Bray-Curtis percent dissimilarities and average dissimilarity to standard deviation ratios (D/SD) of (a) meiobenthic taxa and (b) nematode genera contributing most to dissimilarities in communities between all possible pairs of the three beaches: Collaroy-Narrabeen (CN), Newport (N) and Palm (P). Taxa shown are those that both produced the highest percent contribution to dissimilarity and had a D/SD ≥ 1.3 (based on SIMPER analyses).

a)

	Ave dissimilarity CN vs N	D/SD	Ave dissimilarity CN vs P	D/SD	Ave dissimilarity N vs P	D/SD
harpacticoid						
copepod	15.66	4.26	4.97	1.50	11.20	3.89
gastrotrich	7.78	2.50	4.10	1.97	3.78	1.65
nematode	-	-	3.15	2.26	4.95	2.19
turbellarian	3.62	2.13	-	-	3.79	1.88
copepod nauplii	5.56	2.08	3.99	1.91	-	-
polychaete	-	-	-	-	1.61	1.42

b)

	Ave dissimilarity CN vs N	D/SD	Ave dissimilarity CN vs P	D/SD	Ave dissimilarity N vs P	D/SD
<i>Enoploides</i>	24.45	1.55	11.08	1.82	-	-
<i>Enoplolaimus</i>	12.17	1.50	6.40	1.67	-	-
<i>Xyala</i>	-	-	19.35	3.34	21.29	3.72
<i>Oncholaimus</i>	-	-	9.51	2.12	10.24	2.21
<i>Paracyatholaimus</i>	-	-	9.18	1.92	8.28	1.60
<i>Ascolaimus</i>	-	-	6.88	1.64	6.16	1.36

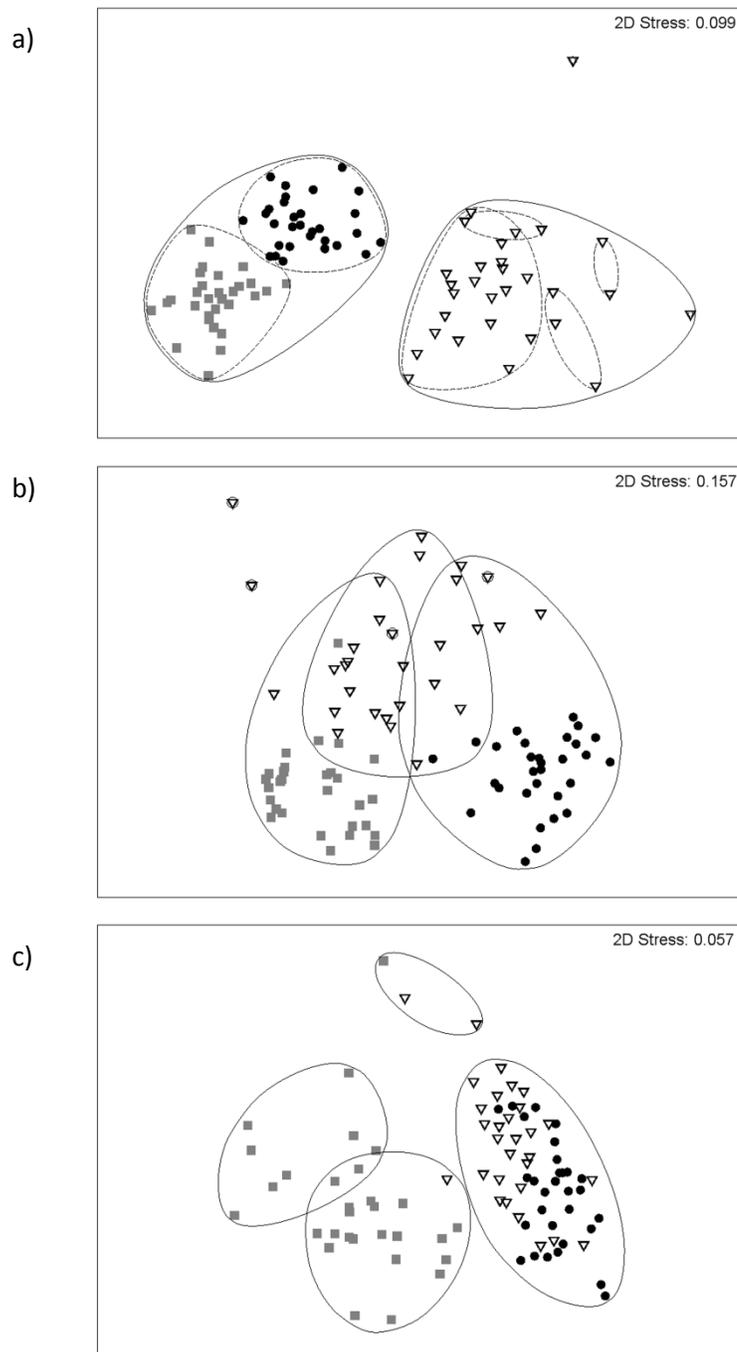


Figure 2: MDS plots of (a) meiofaunal communities, (b) nematode genera and (c) sediment characteristics. Symbols represent beaches: Collaroy-Narrabeen ■, Newport ▽ and Palm ●. Lines denote SIMPROF groupings: for the meiofaunal communities, samples not separated at Bray-Curtis dissimilarities of 70% (solid line) and 80% (dashed line); for the nematode communities, samples not separated at Bray-Curtis dissimilarities of 40%; and for the sediment characteristics samples not separated by a Euclidean distance of 2.

3.2 Spatial structure of faunal and sediment characteristics within each beach

Mantel correlograms revealed similar within-beach spatial structuring of meiofaunal and nematode communities among the three beaches (Figure 3a, b). Each meiofaunal community correlogram was globally significant, thereby varying from a completely random or non-autocorrelated structure (Figure 3a). On each beach meiofaunal communities were positively spatially autocorrelated (SA) at small distances and negatively correlated at larger distances; indicating that the sampling design sufficiently captured their small-scale patterns (Legendre and Fortin 1989). The overall shape of each meiofaunal community could be attributed to a gradient pattern. Nematode community patterns showed smaller scale variation (Figure 3b). Collectively, sediment variables (including sediment mean grain size, skewness and calcium carbonate) displayed similar spatial structuring on two (Collaroy-Narrabeen and Palm) of the three beaches (Figure 3c); which was also very similar to the spatial structure of the meiofaunal community. On the third beach (Newport), the correlogram of sediment variables was not significant.

Moran's I correlograms revealed that on each of the three beaches, harpacticoid copepods displayed a similar pattern to meiofauna communities of significant positive SA at scales < 4.5 m, and negative SA at scales of > 6 m (Figure 4a). Gastrotrichs showed smaller and more variable patch sizes, only revealing strong small-scale spatial structuring on Newport Beach. Nematodes, when analysed at phylum level, showed virtually no spatial structuring across the range of fine scales considered. The Collaroy-Narrabeen and Newport Beach nematode correlograms were not globally significant. At the finer taxonomic resolution of nematode genera, little spatial structuring was apparent across the scales considered, with the exception being *Xyala* from Palm Beach (Fig. 4b). Within two of the three beaches, Collaroy-Narrabeen and Palm, sediment properties were very similar; generally varying at the 2 to 4.5 m scale (Figure 4c). Newport Beach, by contrast, showed no consistent spatial patterning, only calcium carbonate content of this beach showed a distribution that was significantly different from random.

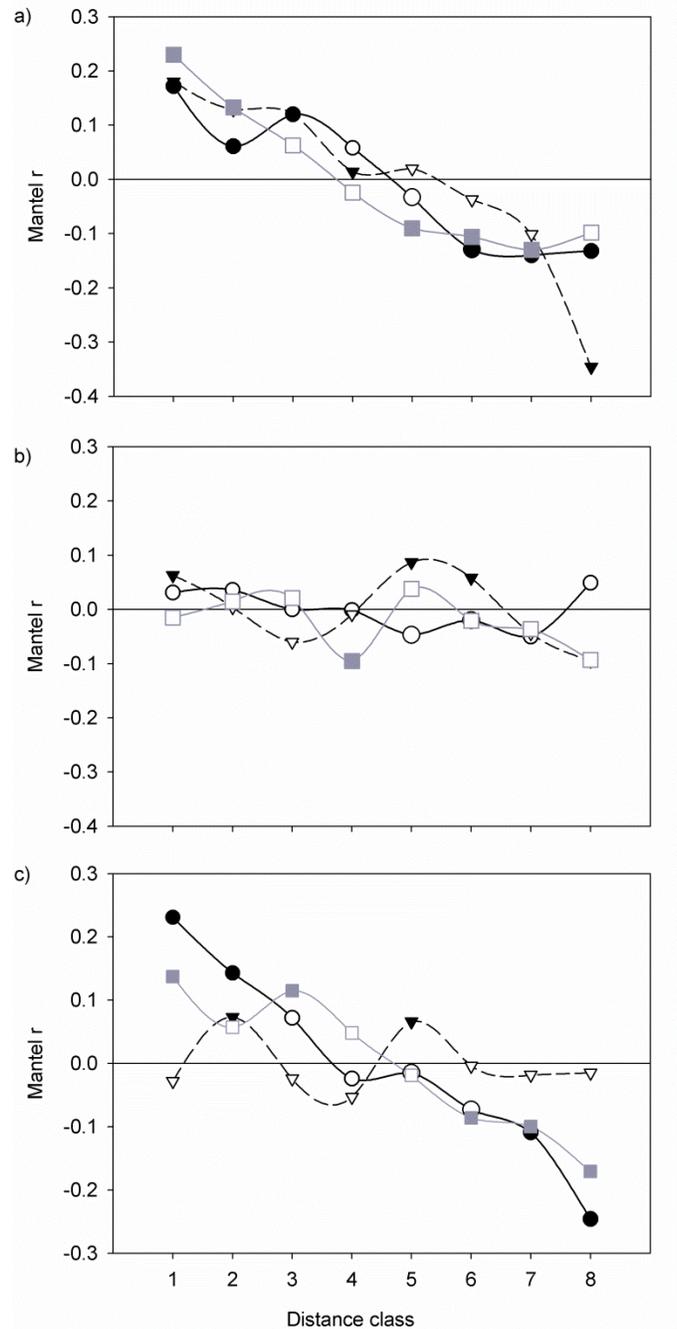


Figure 3: Mantel correlogram for: (a) meiofauna community, (b) nematode community identified to genus and (c) sediment variables (mean grain size, skewness and calcium carbonate). Symbols represent beaches: Collaroy-Narrabeen ■, Newport (dashed line) ▽ and Palm ●. For those beaches which had globally significant patterns, at each distance class filled symbols represent significant values ($p < 0.05$), and open non-significant values of the Mantel r statistic. Distance classes labelled 1 to 8, correspond to the following distances (in metres) between samples: 0.5 - 1, 1.1 - 2, 2.1 - 3.5, 3.6 - 4.5, 4.6 - 6, 6.1 - 7.5, 7.6 - 9.5, 9.6 - 14.5.

a)

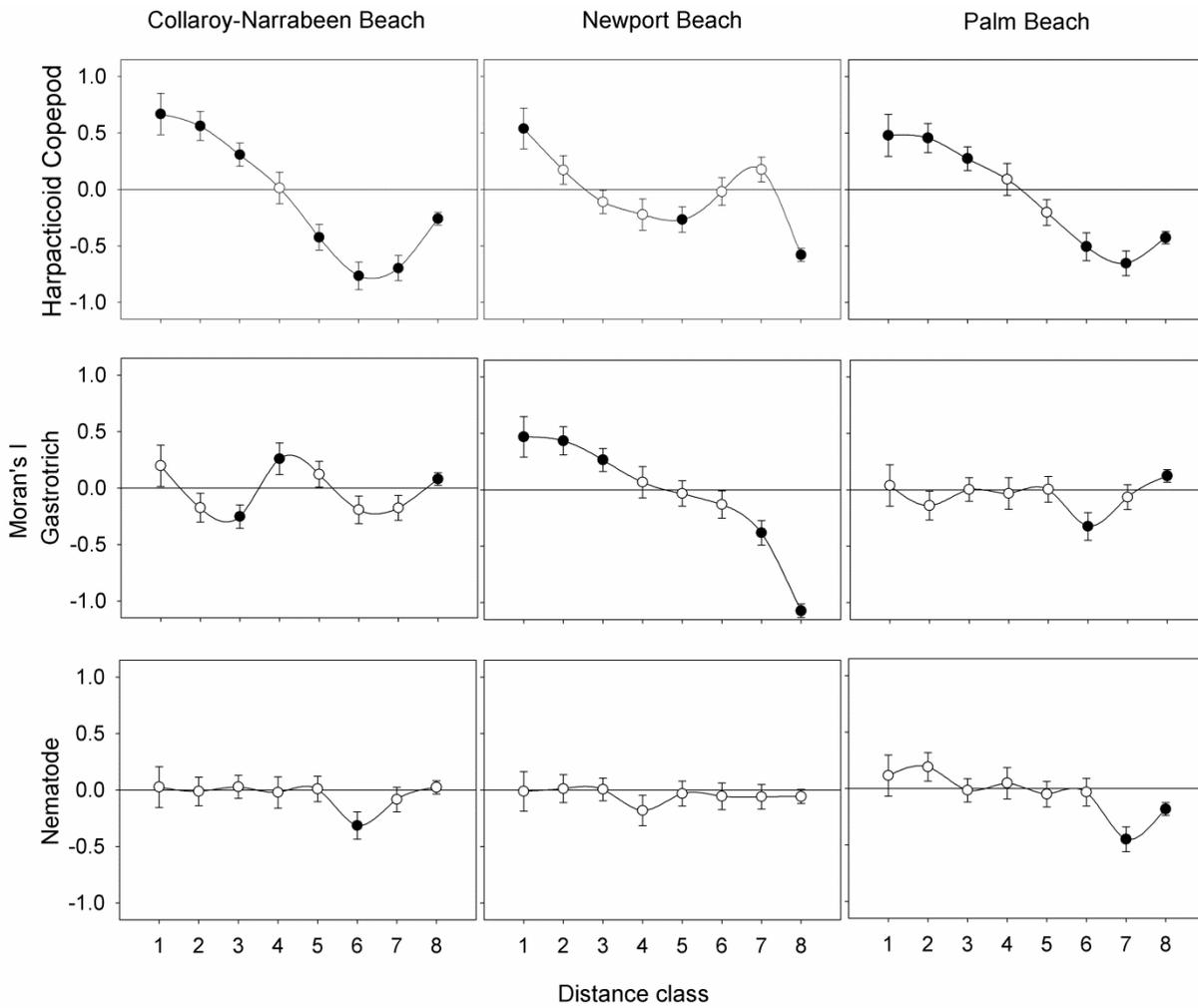


Figure 4: Spatial correlograms based on Moran's Autocorrelation Coefficient I , for: (a) meiofaunal (harpacticoid copepod, gastrotrich and nematode), (b) dominant nematode genera that contributed most to variation between the beaches (*Enoploides*, *Enoplolaimus* and *Xyala*); and (c) sediment variables (mean grain size, skewness and percent calcium carbonate). Filled symbols represent significant values ($p < 0.05$), and open non-significant values of the Moran I statistic. Distance classes labelled 1 to 8, correspond to the following distances (in metres) between samples: 0.5 - 1, 1.1 - 2, 2.1 - 3.5, 3.6 - 4.5, 4.6 - 6, 6.1 - 7.5, 7.6 - 9.5, 9.6 - 14.5.

b)

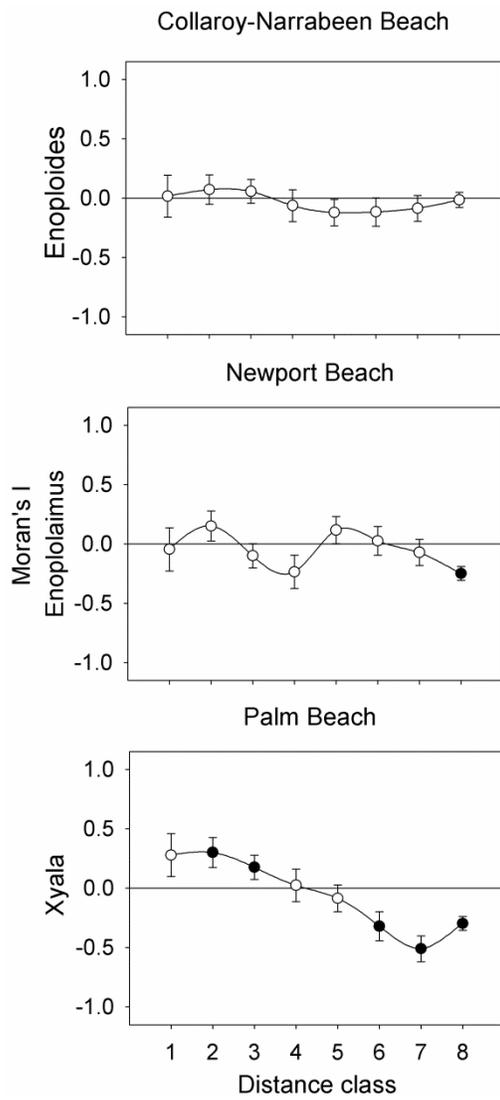


Figure 4 (b). cont.

c)

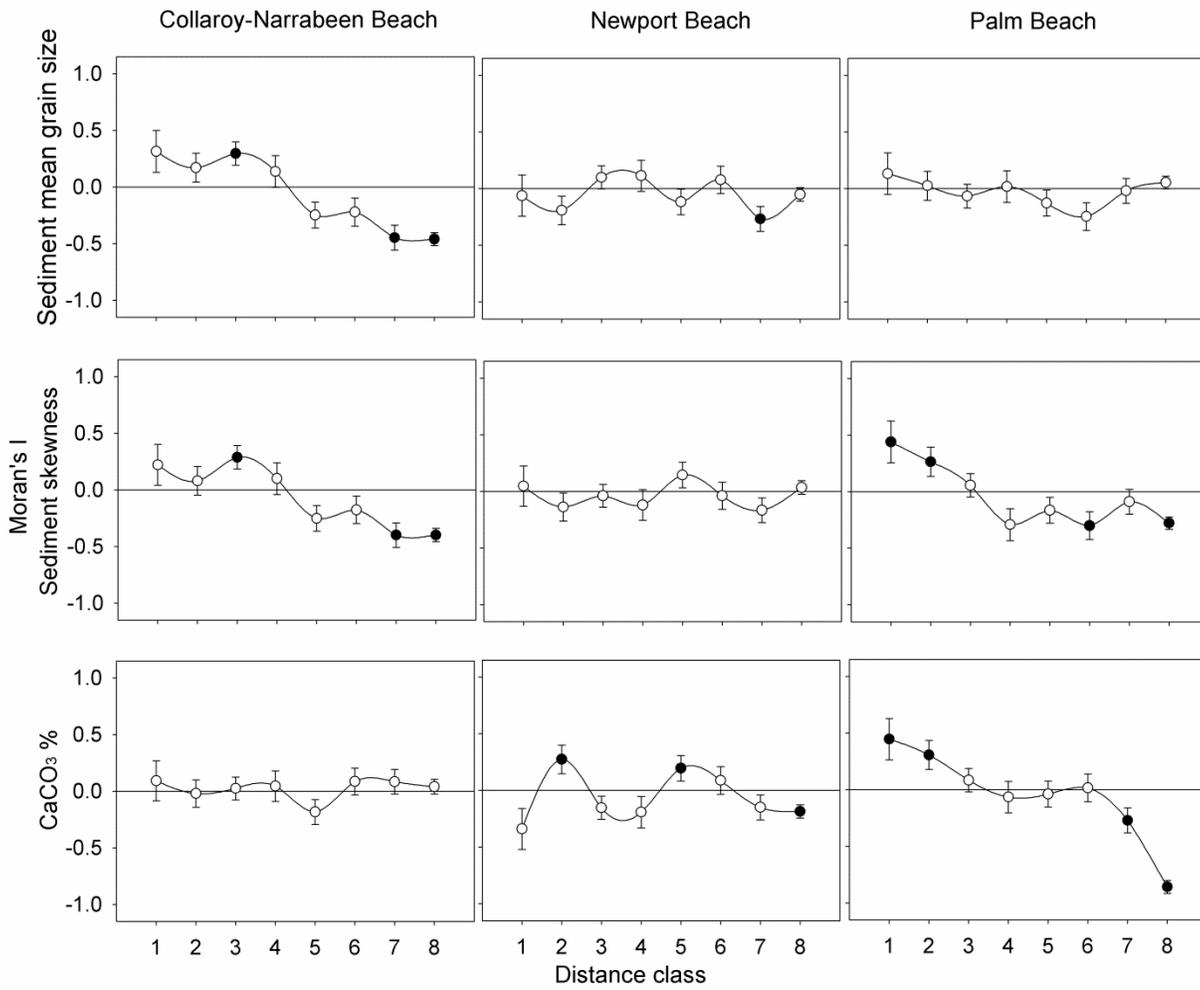


Figure 4 (c). cont.

4. Discussion

The null hypothesis for this study was that the biological communities measured on three proximate beaches of similar geomorphology would be very similar and, within beaches, there would not be small-scale spatial structuring. To the contrary, and despite great care in matching the geomorphology and wave obliquity of three embayed beaches, significant differences in sediment characteristics and meiofaunal communities were found among all three beaches. Within beaches, meiofauna displayed strong patchiness at the scale of 2 – 4.5 m. Differences among beaches in meiofaunal communities were detected independent of whether meiofaunal communities were sorted to coarse taxonomic resolution or whether nematodes sorted to genus were used as a surrogate. Within beaches, however, smaller scale spatial structuring varied among taxa, and taxonomic approaches.

The abundance and diversity of meiofauna found in this study was comparable to that of other intermediate sandy beaches examined in south-eastern Australia (Nicholas and Hodda 1999, Nicholas and Trueman 2005, 2009) and elsewhere (Kennedy and Jacoby 1999). Nematode diversity was also similar (Nicholas and Hodda 1999). The dominance of families Thoracostomopsidae and Xyalidae is typical of sandy exposed beaches worldwide (Nicholas and Hodda 1999, Gheskiere et al. 2004, Lee and Riveros 2012). Among the three beaches examined, differences in the abundances of meiofaunal taxa rather than differences in taxon richness were more important in driving spatial differences.

Given the spatial variation in sediment properties, among and within beaches, spatial variation in meiofauna was expected. Sediment properties, such as grain size and skewness influence meiofaunal communities by determining porosity, permeability, oxygen supply (Giere 2009), stability (i.e. settling velocity), and the amount of interstitial space (McLachlan 1978). Sediment composition has also been found to play an important role in food availability (Fisher and Sheaves 2003, Giere 2009). Among the fauna examined, harpacticoid copepods showed the strongest relationship to sediment characteristics. Harpacticoid copepods can swim and therefore can disperse more readily than other meiofaunal groups, which are not only direct developers but also have limited adult mobility (Bell and Sherman 1980, Palmer 1988). Hence, their spatial pattern, recorded at phylum level, may reveal a clearer response to small-scale environmental variability (Hicks and Coull 1983, Büchi and Vuilleumier 2012).

Sediment characteristics correlated well with meiofaunal community data. Unexplained variation, however, indicates that there were other factors which were also important in driving the spatial variability of meiofauna. The limited dispersal of meiofauna may make their populations particularly responsive to natural and anthropogenic disturbances, such as changes in resource availability or predation, even where these occur on fairly small scales (Fleeger and Decho 1987, Giere 2009). Although at the time of our study there was no obvious difference in natural or anthropogenic disturbance among the three beaches examined, previous disturbance events from which local population densities had not recovered may have contributed. Nematodes, which hatch from eggs and have limited swimming ability (Palmer 1988), showed particularly strong differences among the beaches. The dominance of *Enoploides*, a highly mobile predatory group (Steyaert et al. 2001), on Collaroy-Narrabeen Beach may be an indication of previous disturbance (Gambi et al. 2008, Moreno et al. 2011) or related to the coarser grain size prominent on this beach (Gallucci et al. 2005).

At the smaller scale, this study found that most meiofaunal taxa, with the exception of nematodes, displayed patchiness at the scale of metres. Interestingly, although sediment properties also varied at this scale, only the patchiness of harpacticoids in Collaroy-Narrabeen and Palm, and *Xyala* in Palm Beach matched the patchiness of sediments. Hence, the response of some fauna appeared to be to a condition or resource other than sediment granulometry, but which varied at a similar scale. Whether nematodes were analysed as a phylum, or as separate genera, they generally did not display clear spatial structuring at the scale of metres. Sampling at additional spatial scales would be required to ascertain their spatial structuring. Nevertheless, from these results it would be recommended that if sampling with the intention to compare beaches, then replicate cores on each beach should be separated by distances of at least two metres to maintain spatial independence, and at least four metres to decrease the likelihood of sampling from a single patch.

The main result of this study, that geomorphically similar beaches in close proximity can display significant spatial variability, highlights the need for studies based in these systems to have sufficient temporal and spatial controls. A review of beach nourishment monitoring studies revealed that 89% failed to employ a BACI (before-after-control-impact; Green 1979) design for test of impacts (Peterson and Bishop 2005). An

experimental design using only spatial contrasts carries the assumption that no natural spatial variation exists between reference and disturbed sites. Where beaches, such as those examined in this study, demonstrate large pre-existing differences in their ecology; sampling before and after the impact, at multiple reference locations is required to distinguish large-scale natural variability from the impact of the disturbance itself. This is particularly important in urbanised environments where beaches may be subject to varying degrees of ongoing disturbance and pristine 'control' sites may not exist.

An increasing amount of development and management on beaches requires an appropriate measurement of these impacts (IOC et al. 2011). In many cases the effects of anthropogenic disturbance are not appropriately measured due to a failure to separate natural variability from anthropogenically induced disturbance (Peterson and Bishop 2005). The results from this study indicate the importance of documenting baseline studies in urban areas order to understand what factors are contributing to variation between sites and enable appropriate experimental design.

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**5. Spatio-temporal variation in meiofaunal communities on an urbanised exposed sandy beach,
New South Wales, Australia**

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Abstract

The loss of habitat and associated biota caused by accelerated beach erosion is considered to be the most immediate and severe ecological threat to beaches. In order to study the implications of climate change and disturbance for beach fauna, the natural dynamics of beach faunal populations must be understood. Without this knowledge appropriate ecological impact studies cannot be designed. In the current study, the sources of baseline spatio-temporal variation in the meiofaunal communities of an exposed intertidal sandy beach were examined with respect to key morphological features of the beach. Our study focused on a highly urbanised beach as it is these which are most likely to be affected by future change. Position across the beach face strongly influenced meiofaunal communities, whether samples were collected from cusp horns or bays did not have a strong influence. Meiofaunal assemblage differed significantly between consecutive days, as well as between years, with less variation between stages of the tidal cycle. Elevation above mean water level was the single variable that best explained spatial and temporal patterns, with the total density of meiofauna decreasing with elevation. Sediment water content and granulometry were also key determinants of spatial and temporal patterns. The significant small-scale variation in meiofaunal communities, which was correlated to environmental variables, points to the sensitivity of these fauna to habitat modification. It also highlights the importance, in monitoring studies, of simultaneously sampling sediment cores from control and disturbed sites. Studies in this environment need to account for natural temporal variability to be able to detect change in communities from before to after a disturbance

1. Introduction

The intertidal zone of sandy beaches is densely populated with a great diversity of invertebrates (Brown and McLachan 1990). These provide an important food resource for surf fishes and shorebirds, and are critical in carbon and nutrient cycling (Brown and McLachlan 1990, Defeo et al. 2009). Increasingly, the sedimentary habitat of sandy beach fauna is being modified by coastal development and strategies of management. Hard and soft engineering projects have been undertaken in an attempt to conserve sediments and protect coastal property (Finkl and Walker 2004). Meiofauna are benthic metazoans passing through a 500 μm sieve but retained on meshes of 40-64 μm (Giere 2009). They are the most numerous and diverse of marine biota (Kennedy and Jacoby 1999). They may be particularly susceptible

to disturbance as they are mostly direct developers with limited dispersal (Remane 1952). As meiofauna spend their entire life cycle in the interstitial environment they may be particularly susceptible to changes in the sediment (Swedmark 1964, Coull and Chandler 1992). Meiofauna are important in transferring carbon from primary producers and detritus to higher trophic levels (Brown and McLachlan 1990). Changes in the distribution, diversity and abundance of meiofauna are of interest due to the potential for impacts to propagate up and down the food webs of sandy beaches (Peterson et al. 2006).

Assessment of the impacts of climate change, coastal development and associated management strategies on meiofauna require carefully designed studies. Sampling needs to be conducted at appropriate temporal and spatial scales so that any impact of the perturbation can be detected above natural variability. This is a particularly challenging on sandy beaches, which are the most physically variable of all marine environments (Nybakken and Bertness 2005). Currents, tides, and wave action largely determine the shape and sediment type of a beach (Defeo and MacLachlan 2005). These forces create a dynamic equilibrium and exposed sandy beaches may display morphological change over different time scales varying from a few hours to decades (Cowell and Thom 1997). Erosion and accretion processes disrupt sediment layers; causing changes in particle size and sorting, altering the interstitial environment (Giere et al. 1988).

The abundance and richness of meiofauna at a particular point in space and time is a function of their response to an array of abiotic and biotic variables. Wave, sediment and tidal interactions cause changes in sediment erosion, deposition and in the level of water saturation and exposure (Jackson et al. 1999, Jackson et al. 2002). Water infiltration alters the temperature, the amount of organic input, the salinity and erosion rates of the interstitial environment (Giere 2009). In addition, meiofauna are subject to predation and competition (Coull 1999). As a result, meiofauna may exhibit voluntary or involuntary vertical or horizontal movement within the sand matrix and also between the sand bed and water column (Joint et al. 1982). The overlay of these different conditions will result in the creation of small-scale micro-habitats, termed “sub-environments” (Jackson et al. 2002). Unless sub-environments are taken into consideration in sampling designs it may be difficult to specify relationships between morphodynamic and biological processes (Jackson et al. 2002). Furthermore, because these sub-environments vary over time, sampling

of meiofauna at a single time point may only yield a snap-shot of small scale spatial abundance which may change on the next tide (Armonies 1990).

Obvious sub-environments of the sandy beach include: beach face elevations, morphological features and tidal heights. Previous studies have examined meiofaunal responses to beach face elevation on Australian east coast beaches (Nicholas and Hodda 1999, Nicholas and Trueman 2009) and elsewhere (Moreno et al. 2006). Microtopography has been examined in the form of runnels and sandbars (Hogue and Miller 1981, Gingold et al. 2011), mid scale topography such as variability in cusp features (McLachlan and Hesp 1984) and tidal cycles (Steyaert et al. 2001). However, these sub-environments are rarely included in the design of spatio-temporal studies of sandy beach meiofauna. In addition, the relative importance of these aspects of sub-environments, have not been investigated in the same study on an exposed sandy beach while simultaneously examining spatial and temporal factors.

The New South Wales coastline is dominated by highly urbanised intermediate beaches (Short 2007). In the Sydney region, a highly variable wind wave climate, which is influenced by mid-latitude and tropical cyclones and seasonal breezes (Short and Wright 1981) leads to a high spatio-temporal variability in the morphology of intermediate beaches (Wright et al. 1985). This environment is likely to contain meiofaunal assemblages that are temporally and spatially very variable. Understanding the extent of background variability in the ecology of these already highly modified systems is critical to designing sampling which examines impacts of additional anthropogenic disturbances, such as further urbanisation and climate change.

In this study we conducted spatio-temporally intensive sampling of meiofauna on an urbanised intermediate beach. The following questions were addressed: (1) how homogenous are the meiofauna of a beach area; (2) what is the relationship between meiofauna and beach face elevation, beach morphology and environmental variables such as sediment granulometry; (3) is this relationship stable and predictable over time; and (4) are these relationships contingent on the sampling unit used? We hypothesised that meiofaunal communities would vary significantly over small spatio-temporal scales. Variation in meiofaunal assemblage would be strongly correlated to sediment grain size and sediment

moisture content. Due to variations in water infiltration, we expected that the meiofaunal assemblages of cusp bays from sampling stations closest to the water edge would be more homogenous and stable over time with higher abundance and diversity of meiofauna compared to those further up the beach face. We also hypothesised that more variation would be seen in assemblages at low tide than mid tide due to differences in the tolerance of taxa to exposure and differences in mobility.

2. Methods

2.1 Study site

The study was conducted on Collaroy-Narrabeen beach (33° 43.29' S, 151 ° 18.01' E) approximately 16 km north of Sydney, Australia. At 3.6 km long, Collaroy-Narrabeen is the second longest beach on the Sydney coastline. The beach-form has been described as an intermediate transverse bar and rip grading to rhythmic bar and beach and is microtidal with a spring tidal range of 2 m (Short 2007). The beach is embayed, bounded by headlands at both ends. The beach has been intensely developed, is regularly groomed and locations adjacent to the study site have hard engineering structures (rock walls on the upper beach) in some sections. In addition, between the 2010 and 2011 sampling periods, an area of upper beach south of the study site was subject to beach nourishment. Collaroy-Narrabeen beach was chosen for this study as the background disturbances listed above are common of beaches on the south-east coast of NSW (Cooke et al. 2012). It was also chosen as the morphodynamics and wave climate are continuously monitored by the offshore wave-rider buoy and monthly beach topography surveys (Harley et al. 2008). The study site, 79 m long, was situated towards the southern end of the beach (Figure 1).

To test hypotheses about the response of meiofauna to spatio-temporal variation in sub-environments, we established fixed sampling stations (0.5 x 0.5 m) along transects extending from one metre below the high tide mark to the swash zone at mid tide. At the time of establishment, three transects ran through the centre of cusps (wide scalloped bays) and three ran along the axis of horns (the elevated sediments separating cusps). The transects were each separated by 12 – 18.5 m. Along each transect, the sampling stations were separated by a fixed distance of four metres, with the highest stations, termed zone 1, situated one metre below the upper drift-line and the lowest in the upper swash at mid-tide. At each

station, the GPS position and measurements were taken with reference to a point at the base of the dunes; this was to enable samples to be taken from the same points during subsequent sampling.

Each station was initially sampled at mid-tide (4 hrs before low tide) and at low tide (within 1 hour of dead low) on each of two consecutive days in 2010 (28th February and 1st March). This was done in order to test variation in community structure across the short time-scales of tide and day. At each time, sampling was completed in one hour. Additional sampling was conducted one year later (on 14th February 2011) to assess longer-term change. At this later date, samples were collected from only the three transects that were originally situated in the centre of cusps. In 2011, in addition to sampling the five fixed sampling stations per transect that were established in 2010, an additional two sampling stations (zone 6 and 7) were added along each transect, positioned 4 and 8 m below the previously lowest sampling station. Significant changes in the beach profile had occurred over the year, and this was required in order for sampling to be conducted down to the mid tide swash zone. As in 2010, sampling was conducted at mid and low tide. Sample collection was followed by a topographical survey of the entire beach 02/03/10 and 15/02/11. Topographical beach surveys were conducted using a quad-bike mounted Trimble R8 RTK-GNSS (Real Time Kinematic Global Navigation Satellite System, UNSW Water Research Laboratory; Harley et al. 2008). For each sample time, water quality data was collected and wave climate data was obtained from the Manly Hydraulics Lab (Table 1).

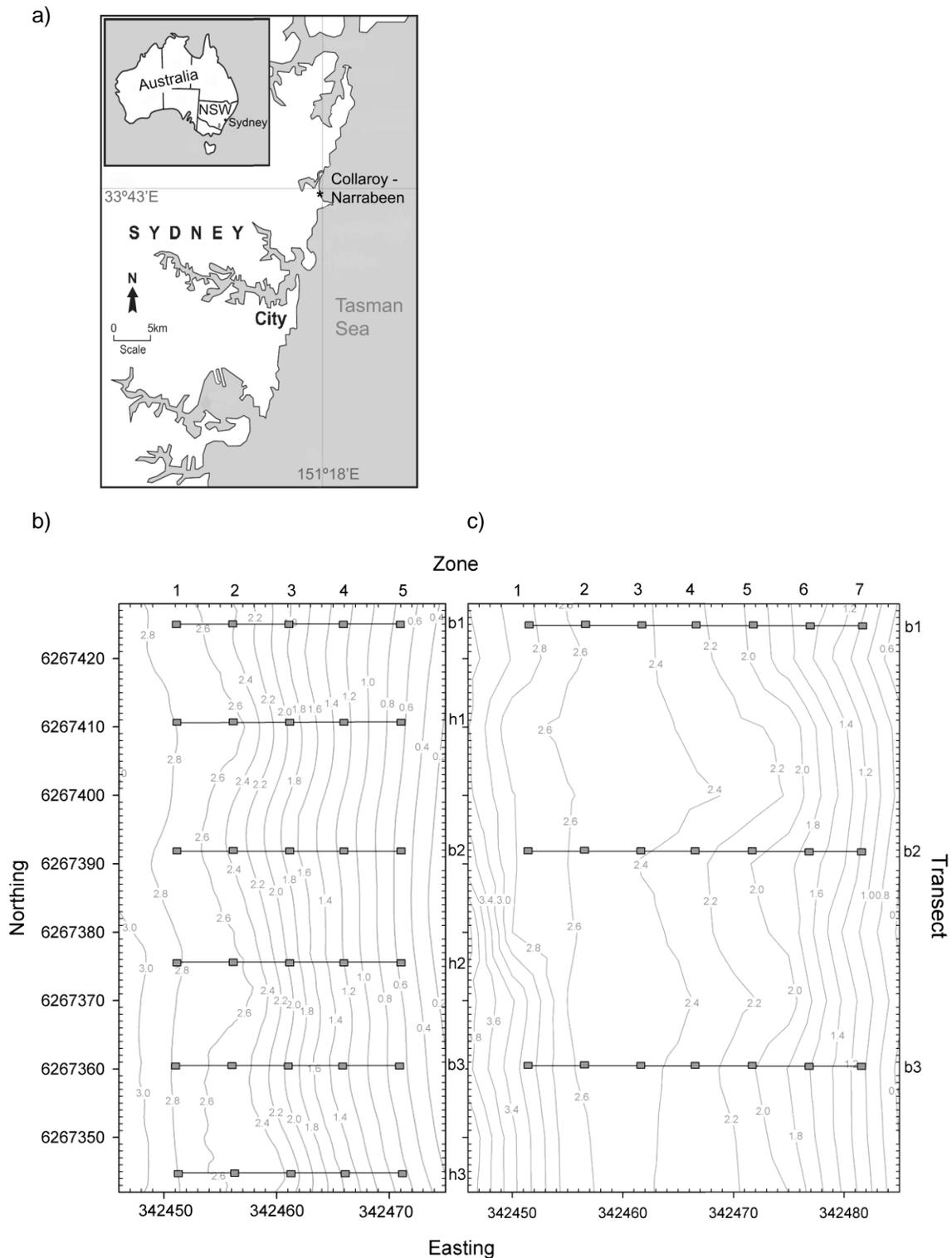


Figure 1: (a) Location of study site on Collaroy-Narrabeen Beach, NSW, Australia, also showing location of transects and sampling stations (squares) with respect to the topography of the beach (b = cusp bay, h = horn) measured on (b) 02/03/10 and (c) 15/02/11. Elevations are shown in metres (in grey) above mean low water level from the upper drift-line (zone 1) to the swash (zone 5 in 2010, zone 7 in 2011).

Table 1: Environmental conditions of the site during and leading up to each sampling period (28/02/10, 01/02/10 and 14/02/11), at both mid-tide (MT) and low-tide (LT); means \pm standard errors are presented where data is replicated.

Date	28/02/10		01/03/10		14/02/11	
Tide	MT	LT	MT	LT	MT	LT
<u>During the sampling period:</u>						
pH ^a	8.09 \pm 0.02	8.05 \pm 0.02	7.76 \pm 0.06	7.81 \pm 0.04	7.15 \pm 0.49	7.81 \pm 0.24
Conductivity (mVs) ^a	63.43 \pm 0.59	62.85 \pm 0.20	63.60 \pm 0.33	62.77 \pm 0.39	51.2 \pm 0.05	51.5 \pm 0.05
Mean significant wave height Hsig (m) ^b	1.02	0.94	2.68	2.44	1.83	1.67
Peak wave period Tp1 (s) ^b	8.35	7.67	10.05	8.55	6.6	8.9
Wave direction (Tn) ^b	78.5	72.67	174	155.5	147.67	153.5
<u>Monthly average ^c:</u>						
Mean significant wave height Hsig (m)	1.84 \pm 0.12				1.59 \pm 0.08	
Peak wave period Tp1 (s)	9.38 \pm 0.30				9.82 \pm 0.40	
Wave direction (Tn)	116.42 \pm 6.46				131.18 \pm 6.18	

^a Interstitial water samples collected from the swash zone using a hypodermic syringe using 4 replicate samples.

^b Obtained from the off-shore wave rider buoy for the one hour period in which sediment and meiofauna samples were collected. Note, this data is from a mid-shelf buoy, and is not a measurement of near-shore wave parameters.

^c Obtained from the off-shore wave rider buoy, averaged over 28 measurements for the month of the sampling. Note, this data is from a mid-shelf buoy, and is not a measurement of near-shore wave parameters.

A total of four sediment samples were randomly taken from within each sampling station, three for meiofaunal analysis and one for sediment analysis. In 2010 samples were taken with a core of 3.5 cm internal diameter inserted to 20 cm depth. This depth of core was chosen to minimise the effect of small-scale vertical migration of meiofauna on spatio-temporal patterns. In 2011 this process was repeated but with cores inserted to three different depths: 5 cm, 10 cm and 20 cm. This was done in order to estimate the effect of mixing and vertical migration of the fauna. A second reason was to determine whether differences between environmental variables such as elevation could still be determined with smaller cores. This would involve counting fewer specimens and can therefore be considered more cost-effective. Within each sampling station the sub-surface temperature (at 10 cm depth) was measured.

Upon collection, meiofaunal samples were placed in a cooler. On return to the laboratory, meiofaunal samples were fixed in 4% buffered formalin in sea water and stained with Rose Bengal. The size of each sample was measured to the nearest mL. This measurement was later used to standardise meiofaunal density per core. Meiofauna were extracted onto a 64 μm sieve following the decantation procedure of Somerfield and Warwick (1996). Four decantations were undertaken. The thoroughness of this method was verified by careful examination of the remaining sediment of six randomly selected samples under a dissection microscope; revealing a high retention rate (between 98 - 100%). Meiofauna were counted and identified to coarse taxonomic level (phylum, order, sub-class in different cases; *sensu* Warwick and Gee 1984) under the dissection microscope.

Measurement of sediment water content was calculated from dry bulk density (Bale and Kenny 2005). Particle size analysis was conducted using sieves of half phi intervals between 1000 and 75 μm following methodology outlined in Folk (1974). Sediment was analysed with the GRADISTAT program which calculates metrics for mean grain size, sorting, skewness and kurtosis for each sample (Blott and Pye 2001). Samples were also analysed for particulate organic matter and the extent of wrack per sample station was recorded; however, due to low recordings and negligible variation this data was not included in the analysis.

2.2 Statistical analysis

Meiofaunal abundance was standardised per volume of sediment collected. Analyses of meiofaunal data used Bray–Curtis dissimilarities calculated between samples following $\log(x+1)$ transformation of data. A dummy variable of 1 was added to the meiofaunal data set prior to the calculation of dissimilarity measures due to the large number of zeros in some samples (Clarke et al. 2006). Replicate sediment cores within a sampling station were averaged prior to analysis, as preliminary analyses indicated that these were significantly spatially autocorrelated and could not be considered as independent replicates.

To test the hypotheses that meiofaunal assemblages would vary across the spatial and temporal scales considered, we used distance-based permutational multivariate analysis of variance (PERMANOVA, 9999 permutations; Anderson, 2001). Firstly, to test for effects of small-scale spatio-temporal variation in meiofaunal communities among sampling times in 2010, we conducted a 4-factor mixed model PERMANOVA, with the factors: date (2 levels: 28 Feb 2010, 1 Mar 2010, random), tide (2 levels: mid-tide vs low-tide, fixed), morphology (2 levels: horn vs bays, fixed) and zone (5 levels: stations 1-5, fixed). Transects were considered as replicates in these analyses.

Secondly, to test for change in the meiofaunal community over a longer time period, we ran PERMANOVA analyses on data collected on a single day in each of 2010 (01/03/2010) and 2011 (14/2/2011), from the three bay transects only. In the case of the 2010 data, meiofaunal communities were averaged across the three replicate sediment cores per sampling station, to give a single value per station. In 2011 only a single 20 cm deep core was collected per sampling station, so no averaging was required. We ran two PERMANOVA analyses on this data, each of which had three factors, including year (2 levels: 2010 vs 2011, random) and tide (2 levels: fixed), and used transects as replicates. The first had the additional factor zone (5 levels in 2010 and 7 levels in 2011, fixed), where 'zones' corresponded to the locations of fixed sampling stations numbered in direction from the drift-line to the sea; while the second had the factor distance from sea (5 levels, corresponding to zones 1-5 in 2010 and 3-7 in 2011, fixed). By running each of these two analyses, we could assess whether the fixed position of a sampling station on a beach, its distance from the water, or both were important in driving differences in meiofaunal community.

Thirdly, in order to examine the sampling difference between different size cores (L = 20cm, M = 10cm and S = 5cm) a PERMANOVA was conducted on the 2011 meiofauna data (14/02/2011). This analysis had the following factors: tide (2 levels: fixed), zone (7 levels: zones 1-7, fixed) and core size (3 levels: L, M and S, fixed) and used the three transects as replicates.

In all of the above analyses where significant effects were found, PERMANOVAs were followed by pair-wise post-hoc tests to identify sources of differences. Following PERMANOVA analyses, the similarity percentages procedure (SIMPER routine of PRIMER) identified those taxa (with a dissimilarity to standard deviation ratio of >1.3 ; Clarke and Warwick 2001) that were the best discriminators of multivariate differences between levels of significant factors (Clarke 1993).

To examine the relationship between meiofaunal assemblage structure and environmental variables, and to determine which environmental variables were the best predictors of meiofaunal assemblages; we ran distance based linear models (DistLM; Mcardle and Anderson 2001). Data collected on 01/3/10 and 14/02/11 was used based on distance from the sea (5 levels, corresponding to zones 1-5 in 2010 and 3-7 in 2011). The environmental variables included in the analysis were: the topographical elevation of each sampling station (obtained from beach mapping), sediment temperature, sediment water content, mean sediment grain size, sediment sorting, sediment skewness and sediment kurtosis. Draftsman's plots in the PRIMER software (Clarke & Gorley 2006) were first used to identify highly correlated variables for each date ($r^2 \geq 0.90$) and to determine whether individual variables required transformation. At this point sediment water content was $\log(x+1)$ transformed. Co-linearity between environmental variables was not found, justifying their inclusion (Draftman's plots, PRIMER). Environmental variables were normalised prior to DistLM analysis, elevation data was constant over the two tide periods. Meiofaunal data from 2010 was averaged per sampling station; data from 2011 used the 20 cm core samples only. Distance based redundancy analysis (dbRDA; Mcardle and Anderson 2001) was used to visualize the results as principal component ordinations, constrained to linear combinations of the abiotic variables (Anderson 2008).

Sources of spatio-temporal variation in the environmental variables that were most strongly correlated with meiofaunal assemblages were examined separately using PERMANOVA analyses. Euclidean distance

matrices were calculated for each variable. Analysis used the same comparisons of data and factors as described earlier for meiofauna data, with the exception that elevation was only analysed for the factors morphology and zone. Spearman's rank correlations tested for relationships between meiofaunal matrices and elevation matrices for the two years (RELATE procedure in PRIMER; 9999 permutations; Clarke et al. 1993).

3. Results

A total of 18 meiobenthic taxa were identified from the 486 samples collected in the study, comprising: Nematoda, Crustacea (Harpacticoida, Calanoida, copepod nauplii, Cumacea, Ostracoda, Isopoda, Halacaroida, Cladocera), Platyhelminthes (turbellaria), Annelida (Oligochaeta, Polychaeta), Gastrotricha, Kinorhyncha, Gastropoda, Priapulida, Tardigrada, Collembola and Rotifera. Over the entire study, Harpacticoida copepods were the most numerous (25%), followed by turbellaria (23%), Nematoda (20%), Ostracoda (13%), Polychaeta (10%) and copepod nauplii (3%). The other taxa comprised less than 3% of the total.

Analysis of meiofaunal data from 2010 revealed interacting effects of date and zone, date and tide, and zone and morphology on community composition (Table 2). Post-hoc tests revealed that there was an effect of morphology (horn vs bay) on meiofaunal communities which was more pronounced in cusp horns, dependant on the zone (Table 2). On each of the sampling dates, differences in communities were evident among zones, with the communities of each beach zone differing from those of all others on the first date of sampling; and each zone, except 2 and 3, differing from one another on the second date. The largest difference was between zones 4 and 2; with zones 5 and 1 – at the two extremes of the intertidal zone, displaying the least homogeneity. Communities differed between the first and second date of sampling; and on the second date of sampling, between states of the tide (Table 2).

Meiofaunal diversity showed variation across the beach face (Figure 2a, Table 3). Overall meiofaunal abundance was greatest closer to the water, decreasing with increased distance up the beach face (Figure 2b). There were significant differences in abundances across the beach face but not at different

times of the tide (Table 3). SIMPER analysis revealed that harpacticoid copepods and turbellaria consistently contributed the most to variation between samples (Figure 2c).

Table 2: PERMANOVA examining sources of spatio-temporal variability in meiofaunal communities during 2010. Da = date (2 levels, 28 Feb 2010, 1 Mar 2010, random), Ti = tide (2 levels: mid-tide vs low-tide, fixed), Mo = morphology (2 levels: horns vs bays, fixed), Zo = zone (5 levels, stations 1-5, fixed). Data were log (x+1) transformed prior to analysis. Terms significant at $p < 0.05$ are highlighted in **bold**.

Source	Df	MS	F	P
Da	1	849.88	7.02	<0.001
Ti	1	768.94	1.21	0.249
Mo	1	1419.70	7.57	0.256
Zo	4	4560.60	6.92	0.007
Da x ti	1	632.90	5.23	0.004
Da x mo	1	187.45	1.55	0.188
Da x zo	4	658.80	5.44	<0.001
Ti x mo	1	9.25	0.09	0.681
Ti x zo	4	245.49	2.88	0.057
Mo x zo	4	277.33	2.86	0.042
Da x ti x mo	1	102.50	0.85	0.475
Da x ti x zo	4	85.37	0.70	0.752
Da x mo x zo	4	96.93	0.80	0.651
Ti x mo x zo	4	51.30	1.18	0.415
Da x ti x mo x zo	4	43.49	0.36	0.978
Res	80	121.10		
Total	119			

Post-hoc tests:

Mo x zo	Within cusp bays, zone: 1 = (2 ≠ 4) = 3 = 5; within cusp horns, zone (1 ≠ 2 ≠ 3 ≠ 4) = 5. Within zone 1, 2, 3, 4, 5: b = h
Da x zo	Within zone 1, 2, 3, 5: 28/02/10 ≠ 01/03/10; within zone 4: 28/02/10 = 01/03/10. For date 28/02/10: zone 1 ≠ 2 ≠ 3 ≠ 4 ≠ 5 For date 01/03/10: zone 1 ≠ (2 = 3) ≠ 4 ≠ 5
Da x ti	Within LT, MT: 28/02/10 ≠ 01/03/10 For date 28/02/10: LT = MT; for date 01/03/10: LT ≠ MT.
Zo	Zone: 1 = (2 ≠ 4) = 3 = 5; average similarity between zones: 2 > 3 > 4 > 1 > 5.

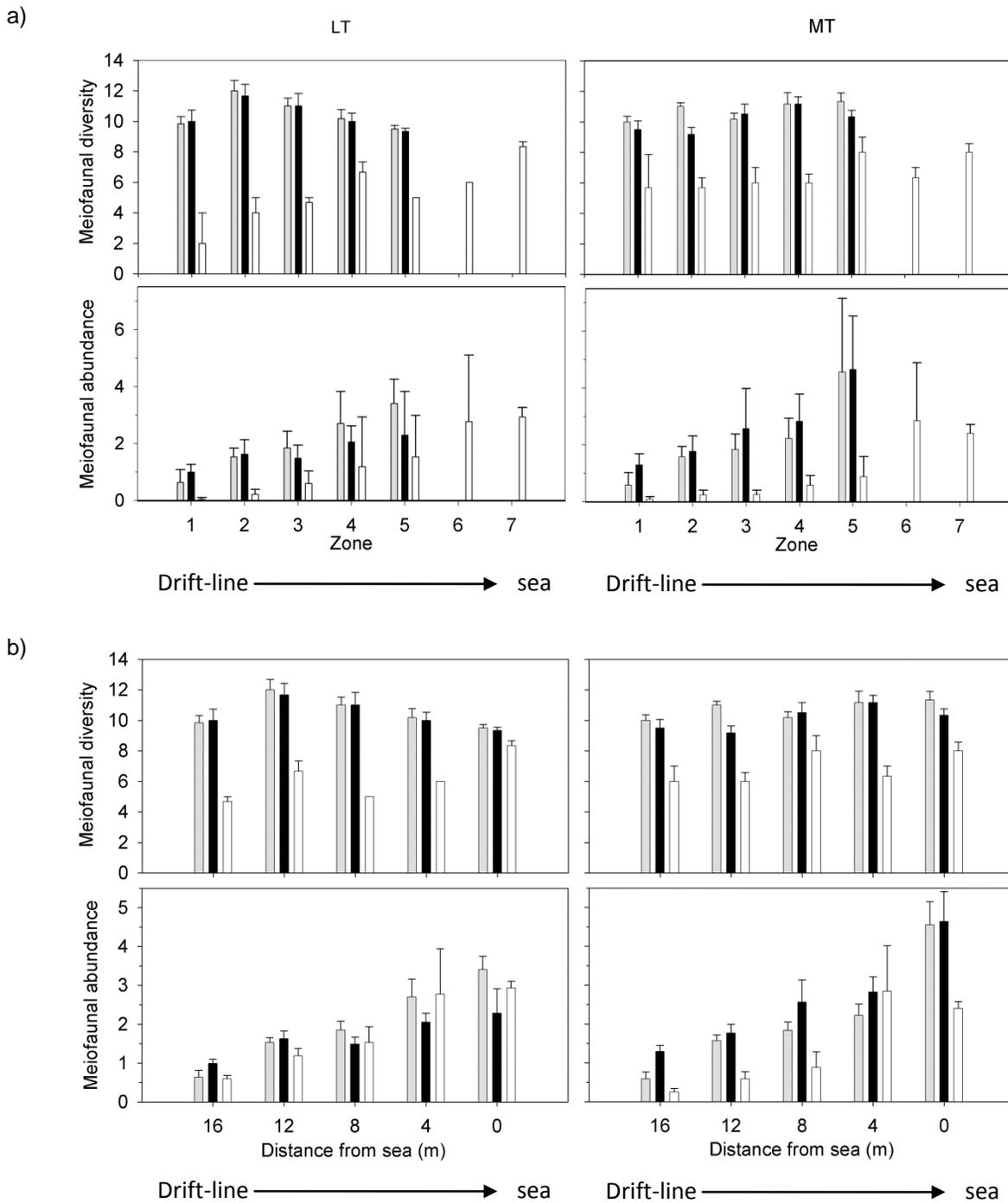


Figure 2: Mean (\pm SE) diversity and abundance at (a) fixed sampling stations (zones 1-5 in 2010 and 1-7 in 2011, ordered from the drift-line to the sea), (b) fixed distances from the swash zone (distance from the sea from 0 to 16 m, corresponding to zones 1-5 in 2010 and 3-7 in 2011), using $\log(x+1)$ transformed data and (c) abundance of harpacticoid copepods and turbellarians using untransformed data; at each of three sampling dates, at low and mid tide, \square 28/02/10, \blacksquare 01/03/10, \square 14/02/11. Using transects as replicates.

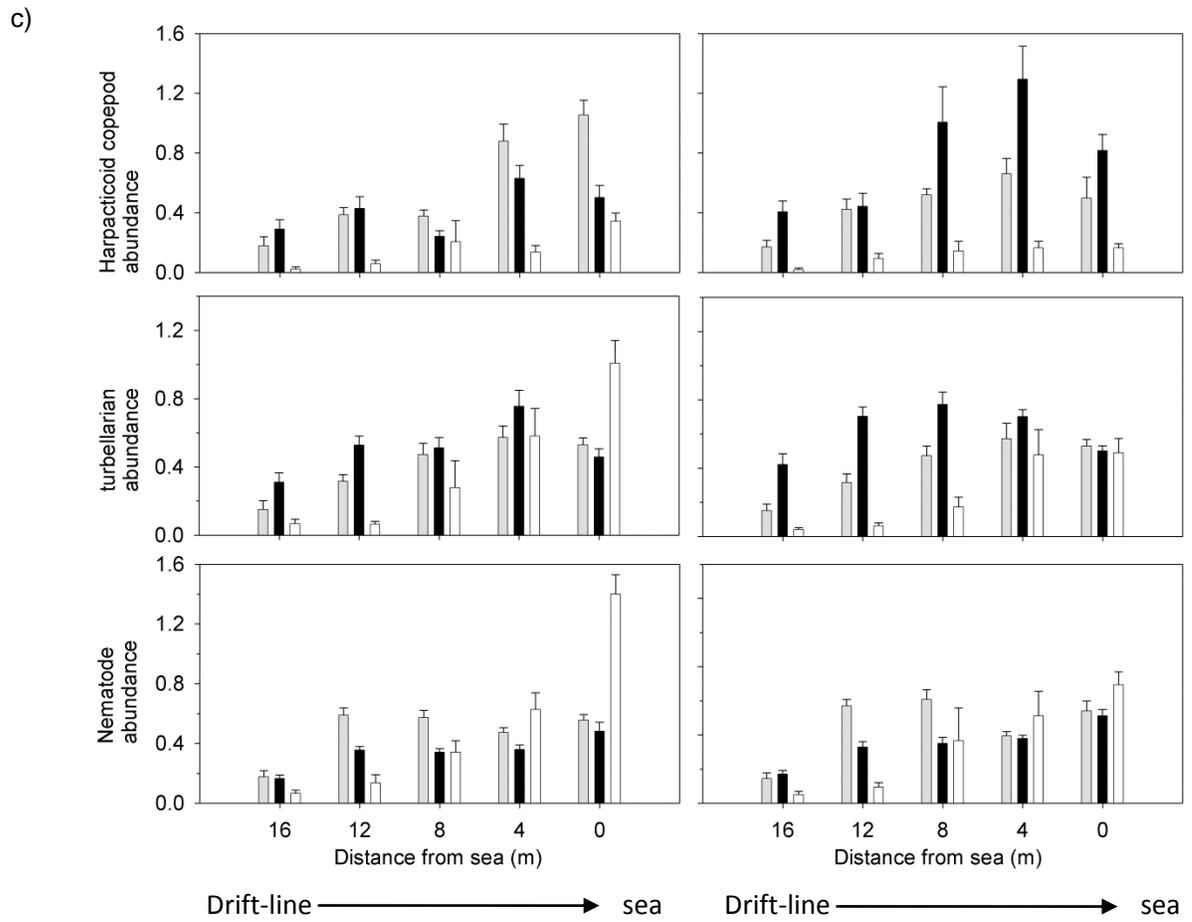


Figure 2 (c). cont.

Table 3: PERMANOVA examining mean (\pm SE) diversity and abundance of meiofaunal communities for the three sample dates Da = (3 levels: 28/02/10, 01/03/10, 14/02/2011; random) at Ti = tide (2 levels: mid-tide vs low-tide, fixed), for: (a) fixed sampling stations (zones 1-5 in 2010 and 1-7 in 2011, ordered from the drift-line to the sea), (b) fixed distances from the swash zone (distance from the sea from 0 to 16 m, corresponding to zones 1-5 in 2010 and 3-7 in 2011), and (c) taxa contributing most to variation: harpacticoid copepod, turbellaria and nematodes using transects as replicates. Terms significant at $p < 0.05$ are highlighted in **bold**. * Term has one or more empty cells.

a)

Source	Df	Diversity			Abundance		
		MS	F	P	MS	F	P
Da	2	310.91	149.14	<0.001	29.27	33.72	<0.001
Ti	1	0.13	0.02	0.862	1.75	0.54	0.492
Zo	6	11.64	3.50	0.061	20.13	15.97	0.001
Da x ti	2	10.84	5.20	0.007	4.47	5.15	0.007
Da x zo*	8	3.57	1.71	0.096	1.34	1.54	0.151
Ti x zo	6	5.61	1.79	0.224	0.80	1.03	0.471
Da x ti x zo*	8	3.34	1.60	0.130	0.76	0.88	0.552
Res	128	2.08			0.87		
Total	161						

b)

Source	Df	Diversity			Abundance		
		MS	F	P	MS	F	P
Da	2	188.28	109.94	<0.001	3.34	3.61	0.030
Ti	1	1.84	0.72	0.494	1.57	0.33	0.628
Di	4	5.78	1.60	0.257	28.99	26.12	<0.001
Da x ti	2	2.63	1.53	0.224	5.04	5.45	0.005
Da x di	8	3.74	2.18	0.034	1.12	1.21	0.292
Ti x di	4	5.85	1.95	0.201	1.36	2.04	0.176
Da x ti x di	8	3.08	1.80	0.085	0.65	0.70	0.688
Res	120	1.71			0.92		
Total	149						

Table 3 (c). cont.

c)

Source	Df	harpacticoid copepod			turbellarian			nematode		
		MS	F	P	MS	F	P	MS	F	P
Da	2	5.72	61.11	<0.001	0.52	9.71	<0.001	0.24	8.12	<0.001
Ti	1	0.91	1.18	0.403	0.02	0.02	0.884	0.12	1.62	0.299
Di	4	2.23	3.44	0.063	1.02	4.34	0.048	1.13	3.75	0.047
Da x ti	2	0.78	8.37	<0.001	0.14	2.62	0.079	0.08	2.59	0.080
Da x di	8	0.66	7.04	<0.001	0.25	4.60	<0.001	0.00	10.61	<0.001
Ti x di	4	0.12	0.62	0.660	0.05	2.61	0.121	0.05	0.99	0.480
Da x ti x di	8	0.20	2.14	0.031	0.02	0.31	0.962	0.05	1.73	0.094
Res	180	0.09			0.05			0.03		
Total	209									

Across sampling dates separated by a year, meiofaunal communities significantly varied between zones (Table 4). Adjacent zones and mid-beach zones were more likely to be similar. The effect of tide was significant only in 2010. When meiofaunal communities were examined according to their distance from the sea, this factor alone was not significant. The extent of significant variation between the sample stations within each year varied. Overall, 2011 was less variable between different times of the tide, zones and distance from sea. SIMPER analysis indicated that between the two years the average dissimilarity of meiofaunal communities was 63.50%. Taxa which were consistently good discriminators between the years were: turbellarians (average dissimilarity = 14.82%, D/SD = 1.41) which were more abundant in 2010; and nematodes (average dissimilarity = 10.03%, D/SD = 1.35) which were more abundant in 2011 (Figure 2c). The comparison of meiofaunal sampling techniques revealed no significant difference in assemblages between cores that were 5, 10 or 20 cm deep (Table 5). Instead, significant effects of tide and zone were found, irrespective of core size.

Table 4: PERMANOVA examining sources of spatio-temporal variability in meiofaunal communities during 2010 (01/03/10) and 2011 (14/02/11), using samples from bays and from large cores only. Ye = year (2 levels: 2010 and 2011, random), Ti (2 levels: mid and low, fixed) with: a) Zo = zone (5 levels in 2010 and 7 in 2011, fixed), and b) Di = distance from sea (5 levels in 2010 and 2011, fixed). Data were log (x+1) transformed prior to analysis. Terms significant at $p < 0.05$ are highlighted in **bold**. *Term has one or more empty cells.

a)

Source	Df	MS	F	P
Ye	1	16239.00	94.38	<0.001
Ti	1	586.41	1.15	0.249
Zo	6	2361.30	3.73	0.033
Ye x ti	1	507.84	2.95	0.040
Ye x zo*	4	633.60	3.68	<0.001
Ti x zo	6	155.47	1.75	0.144
Ye x ti x zo*	4	88.72	0.52	0.901
Res	48	172.06		
Total	71			

Post-hoc tests:

Ye x zo	Within zone 1, 2, 3, 4, 5: 2010 \neq 2011. Within 2010, zone 1 \neq (2 = 3 = 4) \neq 5; within 2011, zone 1 \neq (2 = 3 = 4 = 5) \neq (6 = 7).
Ye x ti	Within LT, MT: 2010 \neq 2011 Within 2010: LT \neq MT; within 2011: LT = MT.
Zo	1 = 2 = 3 = 4 = 5 \neq (6 = 7).

Table 4(b). cont.

b)

Source	Df	MS	F	P
Ye	1	12056.00	40.30	<0.001
Ti	1	830.62	0.75	0.502
Di	4	3483.60	2.55	0.103
Ye x ti	1	1109.30	3.71	0.012
Ye x di	4	1365.60	4.56	<0.001
Ti x di	4	359.80	1.24	0.336
Ye x ti x di	4	290.77	0.97	0.473
Res	70	299.15		
Total	89			

Post-hoc tests:

Ye x di	Within distance 0, 4, 8, 12, 16m: 2010 ≠ 2011 Within 2010: 0 ≠ 4 ≠ (8 = 12) ≠ 16; within 2011: (0 = 4) ≠ (8 = 12 = 16).
Ye x ti	Within LT, MT: 2010 ≠ 2011. Within 2010: LT ≠ MT; within 2011: LT = MT.

Table 5: PERMANOVA examining sources of spatio-temporal variability in meiofaunal communities for a single date in 2011 (14/02/2011). Ti = tide (2 levels: mid-tide vs low-tide, fixed), Zo = zone (7 levels: stations 1-7, fixed), Si = core size (3 levels: L (20cm deep), M (10cm) and S (5cm), fixed). Data were log (x+1) transformed prior to analysis. Terms significant at $p < 0.05$ are highlighted in **bold**.

Source	Df	MS	Pseudo-F	P(perm)
Ti	1	905.67	3.95	0.022
Zo	6	5979.00	26.11	<0.001
Si	2	470.46	2.05	0.079
Ti x zo	6	353.56	1.54	0.099
Ti x si	2	209.78	0.92	0.458
Zo x si	12	131.33	0.57	0.960
Ti x zo x si	12	127.99	0.56	0.963
Res	84	229.04		
Total	125			

Post-hoc test:

Zo: 1 ≠ (2 = 3 = 4) ≠ 5 ≠ 6 ≠ 7.

Analyses linking environmental variables to meiofauna indicated that in each of the years of sampling, elevation was the variable that best predicted meiofaunal assemblage composition, explaining approximately half of all variation in the marginal tests. The overall best solution for explaining meiofaunal variation in 01/03/10 used the variables: elevation, sediment water content (log transformed) and sediment sorting (BIC = 306.40, $R^2 = 0.53$, RSS = 7541.10). On 14/02/11 the overall best solution used elevation only (BIC = 168.26, $R^2 = 0.49$, RSS = 6524.20). The relationship between the environmental variables and meiofaunal communities for 01/03/10 is displayed graphically in Figure 3.

A PERMANOVA of the environmental variables for 2010 (28/02/11 and 01/03/10) used the following variables: elevation, sediment water content $\log(x+1)$ transformed and sediment sorting. When considered independently, the factor zone was significant only for elevation and sediment water content; although it was influenced by the given date, state of the tide and morphology (Table 6). The single factor morphology was not significant, indicating that visible morphological features on the beach face may not necessarily have corresponding abiotic variation and that the extent of along shore variation may be a larger factor. Meiofaunal community was significantly correlated with elevation in both 2010 (Rho = 0.48, $p < 0.001$) and 2011 (Rho = 0.68, $p < 0.001$).

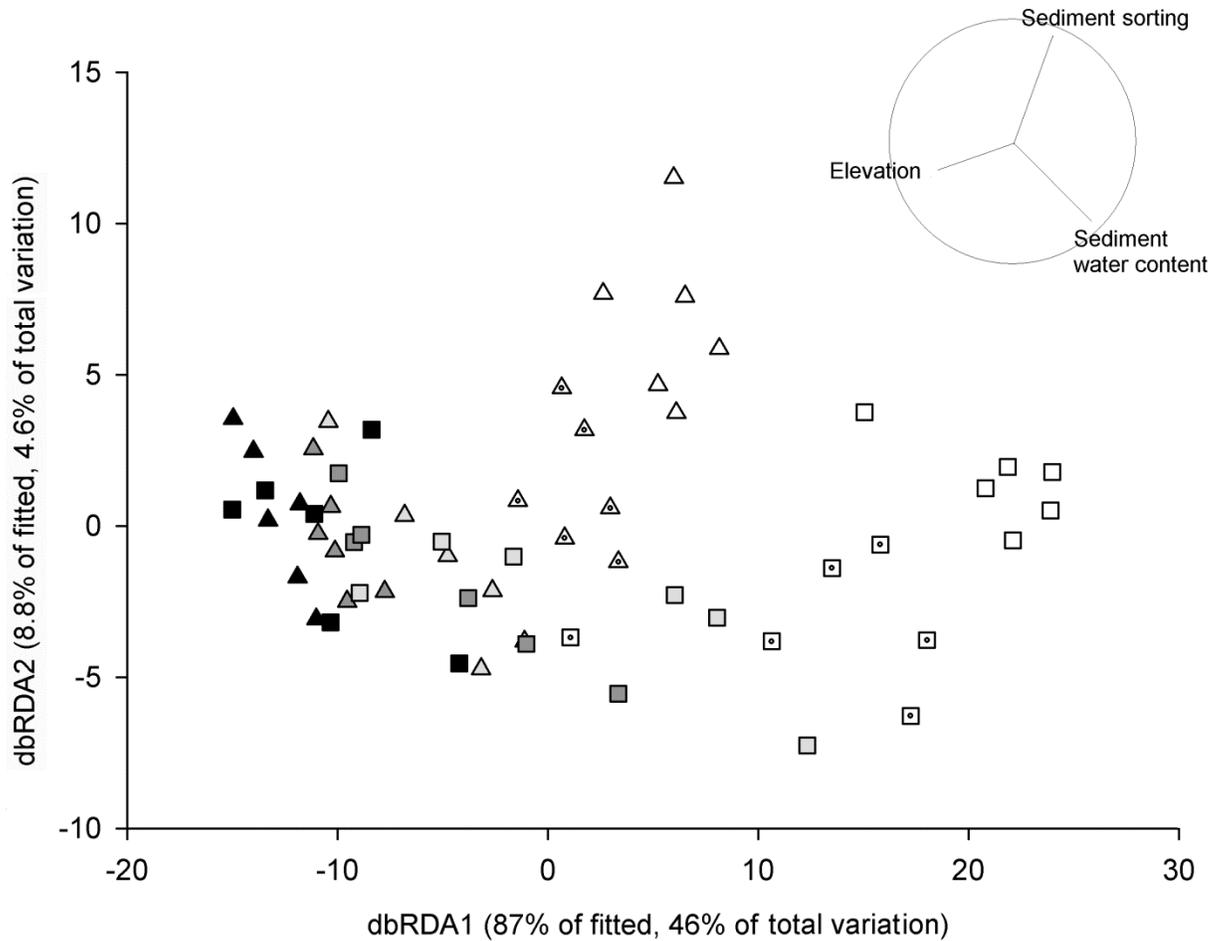


Figure 3: dbRDA ordinations showing the relationship between environmental variables and meiofaunal communities for the sampling date 01/03/10. The length and direction of each vector represents the relationship between each of the best explanatory environmental variables and each dbRDA axis. Symbol shape indicate states of the tide: ▲ low tide, ■ mid tide; symbol colour indicates distance from the sea: △ 0 (upper swash zone), ◻ 4 m, ◻ 8 m, ◻ 12 m and ◻ 16 m for each date.

Table 6: PERMANOVA examining sources of spatio-temporal variability in environmental variables for 2010: (a) elevation, with the factors: Mo = morphology (2 levels: horns vs bays, fixed), Zo = zone (5 levels, stations 1-5, fixed); (b) sediment water content and (c) sediment sorting, with the factors: Da = date (2 levels, 28 Feb 2010, 1 Mar 2010, fixed), Ti = tide (2 levels: mid-tide vs low-tide, fixed), Mo = morphology (2 levels: horns vs bays, fixed), Zo = zone (5 levels, stations 1-5, fixed). Terms significant at $p < 0.05$ are highlighted in **bold**.

(a) Elevation							
Source	Df	MS	F	P			
Mo	1	0.09	2.12	0.156			
Zo	4	4.55	111.62	<0.001			
Mo x Zo	4	0.01	0.18	0.945			
Res	20	0.04					
Total	29						

(b) Sediment water content				(c) Sediment sorting			
Source	Df	MS	F	P	MS	F	P
Da	1	0.02	52.33	<0.001	304.33	1.33	0.248
Ti	1	0.08	8.71	0.263	1515.90	59.05	0.252
Mo	1	<0.01	17.05	0.254	1548.70	7.91	0.255
Zo	4	<0.01	35.32	0.006	7798.30	11.19	0.055
Da x ti	1	<0.01	21.41	<0.001	25.67	0.11	0.729
Da x mo	1	<0.01	0.40	0.538	195.84	0.86	0.361
Da x zo	4	<0.01	0.62	0.647	696.64	3.05	0.024
Ti x mo	1	<0.01	6.81	0.128	150.30	0.26	0.616
Ti x zo	4	<0.01	23.43	0.005	203.29	1.99	0.267
Mo x zo	4	<0.01	0.13	0.956	870.87	1.40	0.362
Da x ti x mo	1	<0.01	0.91	0.353	585.65	2.57	0.121
Da x ti x zo	4	<0.01	0.88	0.494	102.25	0.45	0.770
Da x mo x zo	4	<0.01	0.41	0.806	622.74	2.73	0.036
Ti x mo x zo	4	<0.01	0.06	0.991	94.59	0.71	0.625
Da x ti x mo x zo	4	<0.01	0.58	0.681	134.03	0.59	0.676
Res	80	<0.01			228.21		
Total	119						

4. Discussion

Despite the highly urbanised state of Collaroy-Narrabeen beach, the diversity of meiofauna was similar to other less developed Australian sandy beaches of similar wave-exposure (McLachlan and Hesp 1984, Nicholas and Trueman 2009). Our sampling revealed significant spatio-temporal variation in the beach's meiofaunal communities which was correlated to spatial variation in key environmental variables. Of the spatial variables explicitly considered in the experimental design, the most influential was position across the beachface. Morphological features (i.e. horns and bays) were found to have limited influence on meiofaunal communities. Temporally, greater differences in meiofaunal communities were seen between days and years, rather than between states of the tide. Spatio-temporal variation in communities was closely tied to variation in elevation, and to a lesser extent sediment pore-water and the spread of grain-size around the mean (sorting).

The highest densities of meiofauna were found closest to the sea, decreasing up the beachface. The lowest elevation, in the mid-tide swash zone was dominated by copepod nauplii, Harpacticoid Copepoda, Ostracoda, Nematoda and turbellaria. Higher up, Nematoda and turbellaria were dominant with Collembola limited to the two highest zones. Species richness did not detectably vary across the shore, as new taxa replaced those closest to the sea resulting in a different assemblage. Similar results have been found on previous studies; Nicholas and Hodda (1999) sampled to a depth of 1 m and did not find variation in nematode species richness across the beach face. Meiofaunal patterns of density and diversity across the shore were similar over consecutive days and across years. The abiotic variables influencing this pattern are likely a function of wave run-up, tidal range and variation in the ground water table. Elevation above mean water level, sediment water content and granulometry were the environmental variables that were the best at explaining spatio-temporal variability in meiofaunal communities. Elevation determines wave run up and the extent of infiltration across the beach face. Interstitial water flow is influenced by pore size, and therefore sediment characteristics (McLachlan and Turner 1994). In well drained beaches interstitial water flow may be a very strong factor influencing meiofaunal communities (Dye 1980).

Sediment characteristics such as grain size and sorting vary with sub-environment and are generally considered to be good global indicator for a variety of other conditions such as porosity, permeability and sediment settling velocity, all of which would affect meiofaunal assemblages (Dye 1978, Giere 2009). Previous studies examining cusp horn and bays features have found abiotic (Inman and Guza 1982, (Masselink et al. 1997, Woodroffe 2003) and meiofaunal (McLachlan and Hesp 1984) variation. The smaller variation found in this study may be due to the size of the cusp systems compared to those studied elsewhere. The extent of variation between the alongshore transects in the current study reinforces the need for appropriate replication in order to avoid confounding results (James and Fairweather 1996).

Relatively few studies have examined long-term stability of meiofauna assemblages on exposed beaches (Coull 1986, Hodda and Nicholas 1986, Nicholas and Hodda 1999, Nicholas 2001, Albuquerque et al. 2007, Nicholas and Trueman 2009). This study found that meiofaunal densities were reduced in 2011 compared to 2010. A difference between the years appears to be due largely to the difference in beach-face elevation between the years at our fixed sampling stations. Interestingly, our study also revealed considerable variation in meiofaunal communities even across relatively short temporal periods of days and tides. This highlights the importance, in monitoring studies, of simultaneously sampling sediment cores from control and disturbed sites, and of adequately accounting for natural temporal variability in sampling programs aimed at detecting change in communities from before to after a disturbance (Green 1979).

A comparison of different core sizes collected from fixed points in 2011 did not reveal any significant difference in meiofaunal communities among cores of 5-, 10- or 20-cm depth. This suggests that across the sediment depths considered by this study, the majority of meiofauna were found in surface sediments. Previous studies (e.g. Albuquerque et al. 2007) found that while different taxa showed temporal variability in their vertical distributions, in most cases the highest abundance was found in the surface layer (0-10 cm). As we did not sample below 20 cm, it remains possible that abundant meiofaunal populations may also have been found deeper, particularly those avoiding desiccation near the drift zone. Previous studies have found meiofauna as deep as 1.4 m near the high water mark (Nicholas and Hodda 1999). In

addition, while the majority of meiofauna are found in wet sands above the water table (McLachlan 1980), some species are found below the water table on exposed sandy beaches (Nicholas 2001). Nevertheless, our results suggest that where the goal is cost-effective sampling, taking smaller core lengths may reveal the same patterns of density and diversity; although this may vary between locations and conditions (Spalding and Jackson 2001, Moreno et al. 2006).

This study has demonstrated significant variation in meiofaunal communities of an urbanised beach, which was correlated to changes in beach profile and sedimentology. Our study suggests that these communities may be particularly sensitive to climatic change or further human modification. There was significant baseline temporal- and spatial-variability in these sandy beach ecosystems which are already affected by urbanisation. Future studies aimed at detecting effects of additional disturbances to these systems will need to adequately control for this significant small-scale variation through temporal and spatial replicates at relevant scales. Without this, it will not be possible to detect effects of anthropogenic disturbance over and above background noise (Schoeman et al. 2000).

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The impact of entrance dredging on the meiofaunal communities of an intermittently open lagoon in south-eastern Australia

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Abstract

Worldwide, coastal lakes and lagoons are among the most anthropogenically modified of marine environments. On the south-east coast of Australia, approximately 50% of the over 100 Intermittently Closed and Open Lakes or Lagoons (ICOLLs) are subject to periodic entrance dredging or permanent entrance training. This study examined the impact on meiofaunal communities of ICOLL entrance dredging, both at the whole ICOLL scale with reference to control ICOLLs; and also at the scale of sites within and outside of the area of direct disturbance of the dredged ICOLL. Sampling at multiple times before and after the disturbance, within the dredged and control ICOLLs, revealed an effect of dredging on meiofaunal communities. All sites within the dredged ICOLL, both directly and indirectly affected by the disturbance, revealed patterns of change that were distinct from the two control ICOLLs. At the dredged site there was a change in meiofaunal community structure, as well as sediment grain size and chlorophyll- α concentration which decreased following the disturbance. Examination of individual taxa found a large variation of responses in all sites which correspond to what is broadly known of their biology. Overall, variation in the ICOLLs was most strongly correlated with mean grain size, sediment sorting, salinity, dissolved oxygen and chlorophyll- α concentration. It is likely that the extent of ICOLL management will increase in frequency and intensity due to increases in storm surge activity and sea level rise; potentially influencing the abundance of various benthic taxa and their spatial variability. As meiofauna are an important link between primary production and higher trophic levels, decisions to increase the frequency of dredging should consider effects of this intervention on ecological as well as human values of ICOLLs.

1. Introduction

Intermittently Closed and Open Lake or Lagoons (ICOLLs) are waterbodies which have entrances that are periodically closed to the ocean. ICOLLs account for over 60% of coastal waterways in south-eastern Australia (Haines et al. 2006) and are estimated to comprise up to 13% of the world's coastal area (Barnes 1980). ICOLLs are valuable environments (Costanza et al. 1997) which can support productive fisheries and migratory shorebirds of global significance (Pollard 1994). Closed ICOLLs are separated from the ocean by sand bars (berms) formed by wave driven sediment transport (Roy et al. 2001). Storm waves, outflowing tides or increased rainfall may result in the berm being washed away or overtopped,

thus opening up the entrance (Roy et al. 2001). ICOLL entrance conditions control tidal exchange, salinity regimes, and the recruitment of fauna (Roy et al. 2001). The state of the entrance opening is regarded to be the most important factor governing the biology of ICOLLs (Griffiths and West 1999, Roy et al. 2001, Jones and West 2005).

In combination with sea level rise and storm surge activity, the accumulation of runoff in ICOLLs during periods of entrance closure can present considerable flooding risk to coastal settlements. Artificial entrance intervention is presently undertaken in about 50% of ICOLLs in New South Wales (NSW) Australia to mitigate flooding to coastal properties, to flush polluted waters, to enhance navigation, and to allow fish to recruit and migrate (Haines et al. 2006). Similar intervention is also common elsewhere in the world (e.g. Whitfield 1997). In the past, some ICOLLs have been permanently opened through the use of hard engineering structures such as training walls. More recently, NSW coastal legislation has restricted the use of hard engineering structures, and presently the majority of artificial entrance opening is achieved through dredging and removing sand from the opening area. Despite the ecological importance of ICOLLs, very little is known of the ecological impact of dredging in this environment.

Artificial entrance opening by dredging has the potential to cause considerable changes to the ecology of coastal ICOLLs via one or more potential mechanisms. Firstly, physical disturbance at the site of dredging may directly kill sediment-dwelling organisms and microphytobenthos (Thompson 1961, Miller 1989). The impacts of dredging may extend beyond the immediate area of sediment extraction if fine particles which cause turbidity and block light to aquatic primary producers are mobilised (Moore et al. 1997, Longstaff and Dennison 1999) or if deep-buried contaminants are uncovered (Essink 1999, Dornie et al. 2003). Secondly, opening of an ICOLL may cause changes in the physico-chemical environment as the system alters from a non-tidal to tidal system, potentially resulting in a major shift in salinity, bathymetry, sediment composition, circulation pattern, dissolved oxygen levels and/or nutrient concentrations (Roy et al. 2001). These changes may individually or interactively alter invertebrate assemblages and their spatial distributions (e.g. Dye 2005, Nybakken and Bertness 2005, Skilleter et al. 2006, Leduc and Pilditch 2013).

Meiofauna, are benthic metazoans passing through a 500 µm sieve but retained on meshes of 40–64 µm (Giere 2009). Meiofauna underpin the productivity of ICOLLs by providing a food resource for fishes and shorebirds and by playing a critical role in carbon cycling and bio-mineralisation (Coull 1999). The majority of meiofauna are direct developers (Remane 1952). Following disturbance, recovery relies on organisms moving through sediment from undisturbed areas or entering the water column (Boeckner et al. 2009). For these reasons and also due to their fast reproduction and rapid life histories, meiofauna may be ideal organisms to study the ecological effects of natural and anthropogenic impacts (reviews by Coull and Chandler 1992, Kennedy and Jacoby 1999). In Australia there have been very few ecological studies on meiofauna (Coull 1999, and this remains true 14 years on). Previous studies (Dye 2005, Dye and Barros 2005) indicate that meiofauna may be sensitive to changes in entrance dynamics and management history.

The current study examined short-term causative impacts on meiofaunal communities of a single dredge event, where sediment was taken from an ICOLL entrance, some of which was moved to a replenishment site within the same ICOLL to create a recreational beach. The study used a beyond-BACI design (Underwood 1991), that controls for background spatial and temporal variability, to determine how management of an ICOLL entrance influenced meiofaunal communities at two spatial scales. First, to assess localised impacts at the dredge and replenished sites, the temporal dynamics of meiofaunal communities were compared between these impacted sites and sites within the perturbed ICOLL that were outside the area of direct physical disturbance. Second, to assess impacts at the ICOLL scale, the temporal dynamics of sites within the disturbed ICOLL were compared to those in two control ICOLLs that did not undergo intervention during the study period.

It was hypothesised that the intervention would result in changes in the abundance and composition of meiofaunal communities within the managed ICOLL that were not replicated in the control ICOLLs. Impacts of the dredging were hypothesised to be greatest at the site of sediment removal, but were also expected to extend to other sites within the disturbed ICOLL as a consequence of changes to salinity and water flow. Among meiofaunal taxa, it was hypothesised that nematodes and copepods would respond most strongly to the disturbance, particularly at the site of dredging, as they have previously been shown

to be highly sensitive to changes grain size, organic content and penetrability of sediment (Dye and Furstenberg 1978, Orren et al. 1981, Boyd et al. 2000).

2. Methods

2.1 Study sites

This study examined the effect of dredging on Narrabeen ICOLL (33.7 ° S, 151.3 ° E) in the Northern Beaches area of Sydney, New South Wales, Australia (Figure 1). The ICOLL has a water area of 2.2 km² and is situated within a highly urbanised catchment (area: 55 km²). Narrabeen is a ‘ventilated’ ICOLL, with an entrance wall that is designed to prevent closure by a berm (Roy et al. 2001). Nevertheless, the entrance channel frequently sediments up and closed in February 2011 (pers comm. Warringah Local Government Authority). Dredging of the Narrabeen ICOLL has been undertaken approximately once every four years since the mid 1970s for flood mitigation and to improve public amenity of the lake (pers comm. Warringah Local Government Authority).

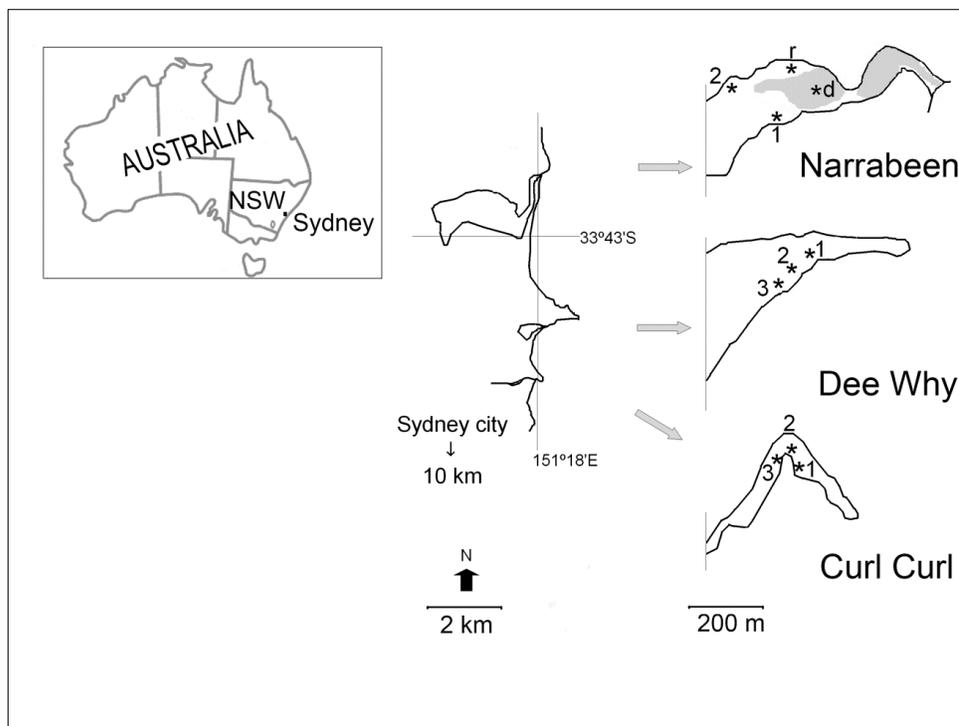


Figure 1: Map showing locations of the disturbed (Narrabeen ICOLL) and control (Dee Why, Curl Curl) ICOLLs sampled along the coastline of Sydney, Australia, and the sites sampled within these (stars). In Narrabeen, the dredged area is indicated in grey; d = dredge site, r = replenishment site.

This study focused on a dredging event conducted over September to November 2011 in which approximately 36,000 m³ of marine sand, excavated by a maximum depth of 1 m, was removed from the entrance channel to the ICOLL (pers comm. Warringah Local Government Authority). Approximately 1,000 m³ of this sand was used to create a beach adjacent to the dredged area. Temporal changes in the meiofaunal communities of Narrabeen ICOLL from before to after the dredging event were compared to temporal changes in two geomorphically similar (see Roy et al. 2001) control ICOLLs, Dee Why (33.7 ° S, 151.3 ° E) and Curl Curl (33.8 ° S, 151.3 °E). Each of the control ICOLLs were geographically proximal to Narrabeen and were situated within similarly urbanised catchments. Dee Why has a water surface area of 0.2 km² and a catchment area of 6.2 km²; Curl Curl has a water surface area of 0.1 km² and a catchment area of 4.5 km² (Roy et al. 2001). Due to the absence of suitable control locations to the north of Narrabeen ICOLL, each was situated to its south. The entrances to Dee Why and Curl Curl were not dredged during the study and were predominantly closed to the ocean, with the exception of some short-term overtopping of the berm during spring tides and following beach scraping. During this study, the Dee Why ICOLL berm was overtopped four times, and the Curl Curl berm 17 times (pers comm. Warringah Local Government Authority).

Within each of the two control estuaries, three 5 x 5 m sub-tidal sites, each separated by at least 30 m, and 200-300 m from the ICOLL entrance were sampled. Within Narrabeen ICOLL four similarly positioned sites were sampled. Two were outside of the area directly impacted by dredging and away from the site of spoil disposal, one was in the area of dredging and one was adjacent to the site of spoil disposal. Within the site directly affected by dredging the water was -0.20 m below Mean Water Level (MWL) prior to dredging and -0.70 m MWL post dredging; at all other sites the samples were collected from approximately -0.50 m MWL depth. Sites were away from centres of recreational use (e.g. boat ramps, popular swimming sites), sea walls and were free of aquatic vegetation.

2.2 Sampling methodology and laboratory procedures

In order to examine the direct and indirect impacts on meiofauna of dredging the entrance to Narrabeen ICOLL, samples were collected from each site within the three estuaries on five dates, each four months apart. These were: (1) approximately five months before Narrabeen ICOLL dredging began (Apr 2011,

B1), (2) less than a month before dredging began (Aug 2011, B2), (3) immediately after dredging (Dec 2011, A1), (4) approximately four months after dredging (Apr 2012, A2) and (5) approximately eight months after dredging (Aug 2012, A3).

On each date, sediment samples were collected from each site for analysis of meiofauna, sediment characteristics, and microphytobenthos. Sediment samples for meiofaunal analysis ($n = 4$ per site and sampling date) were collected with a plastic core with internal diameter 2.65 cm and 10 cm length (area = 5.5 cm^2). Cores were randomly positioned at least 1 m apart. Meiofaunal samples were stored in plastic jars in a cooler and were processed in 5% buffered formalin with Rose Bengal within seven hours of collection. The size of each sample was measured to the nearest mL in order to standardise meiofaunal abundance to the volume of each core. Meiofauna were extracted by decanting the samples four times through a $500 \mu\text{m}$ sieve suspended above a $64 \mu\text{m}$ sieve (Somerfield and Warwick 1996). Centrifugation through a silica gel gradient (Ludox HS 30, density 1.18 g/cm^3) was also done for samples with silt (Pfannkuche and Thiel 1988). Following addition of ludox, these samples were centrifuged at $4000 \times g$ for 5 minutes (Dye and Barros 2005), with the process repeated twice. The extraction efficiency was estimated from examination of four randomly selected samples as $97\% \pm 1.22 \text{ SE}$.

Due to the high densities of meiofauna within sediment cores (up to 3203 individuals per sample recorded in Curl Curl ICOLL), each sample was made up to 20 mL with water and a subsample of 3 mL (15% of the full volume) was taken for identification and enumeration of meiofauna following homogenisation (see McIntyre and Warwick 1984). In order to verify the sub-sampling methodology, sub-samples of 12 full samples were compared from Narrabeen ICOLL. The subsamples provided an average abundance of $19\% \pm 1.50 \text{ SE}$ of the full samples; and had a meiofaunal community structure that was closely correlated to the full samples ($Rho = 0.77$, $p < 0.001$; using the RELATE procedure in PRIMER, 9999 permutations).

Three cores, of 4.7 cm internal diameter and 10 cm depth, were collected for analysis of sediment grain size and organic content. Each sample was homogenised and a $\sim 5 \text{ g}$ subsample was used to determine total organic matter by weight loss after treatment with 27% hydrogen peroxide (*sensu* Kettler et al. 2001). Granulometry was conducted on 100 g sub-samples. The silt fraction was determined following the

methodology outlined by Bale and Kenny (2005). Dry sieving was conducted using sieves of half phi intervals between 600 and 75 μm following methodology outlined in Folk (1974). Sediment was analysed with the GRADISTAT program which calculates metrics for mean grain size, sorting, skewness and kurtosis for each sample (Blott and Pye 2001). A multi-function water quality probe (TPS 90-FL/T) was used to collect water quality information, including: pH, salinity, dissolved oxygen, turbidity and temperature at 10 cm depth at three positions within each site. Depth to the redox potential discontinuity layer was measured by inserting a hollow Perspex tube at least 25 cm into the sediment at three positions within each site.

Samples for measurement of microphytobenthic biomass ($n = 3$) were taken from the top 3 cm of sediment using a 2 cm diameter plastic syringe, and were transferred to an opaque container immediately following collection. On return to the laboratory, samples were initially frozen. To extract microphytobenthic pigments (chlorophyll- α and phaeopigments), 10 mL of 90% acetone was added to the frozen samples, the samples with acetone were refrigerated (at 4°C) for 24 h and then centrifuged. Chlorophyll- α and phaeopigment concentrations were determined before and after acidification with HCl, from absorbance readings at 665 and 750 nm in a spectrophotometer. Chlorophyll- α and phaeopigment concentrations were calculated using Lorenzen's equation (1967).

2.3 Statistical analysis

To test the hypotheses that there would be greater change in environmental variables and meiofaunal communities from before to after the dredging event: (1) within the disturbed ICOLL (Narrabeen) than the control ICOLLs (Dee Why, Curl Curl) and (2) within the disturbed ICOLL, at sites of dredging or disposal than at sites outside of the area of direct disturbance, we used distance-based permutational multivariate analysis of variance (PERMANOVA; Anderson 2001). Analyses had four factors: Impact (2 levels: disturbed, control, fixed); ICOLLs (1 level within disturbed, 2 levels within control, random), Site (3 levels within the control ICOLLs, 4 levels within the disturbed ICOLL: 1 dredged, 1 replenished and 2 subject to indirect impacts only, random) and Date (5 levels: fixed). The factor Date was not nested into before and after impact times as we did not know *a-priori* whether there would be a pulse or longer term press

response to the disturbance. Treatment differences driving significant terms in the PERMANOVA were identified using pair-wise post-hoc tests (in the Appendix of this Chapter).

In the case of the environmental data, analysis was first conducted on the full multivariate matrix of variables. The environmental variables used in this analysis were: sediment variables (mean grain size, sorting, kurtosis, skewness, silt fraction, organic content and depth of redox layer); water quality (salinity, temperature, pH, dissolved oxygen and turbidity); as well as microphytobenthic biomass (Chlorophyll- α and phaeopigment concentration). Draftman's plots did not indicate co-linearity ($r^2 \geq 0.90$) in the abiotic data, providing support for retention of all variables in the multivariate PERMANOVA. In order to normalise their distribution, the following variables were log (x+1) transformed: chlorophyll- α and phaeopigment concentration, silt fraction, organic content and turbidity. Abiotic data containing different units of measurement were normalised prior to generation of Euclidean matrices for analysis. Separate analyses were then conducted on univariate variables which we expected to show the greatest change after the dredge event: mean grain size, salinity, dissolved oxygen and microphytobenthos biomass.

The relationship between meiofaunal assemblages and environmental variables averaged for each site was examined using distance based linear models (DistLM) with BIC criteria and 9999 permutations (Mcardle and Anderson 2001). As preliminary examination showed that using the BEST model did not make a definitive selection, a model which included all environmental variables was used. Results were displayed graphically on a dbRDA plot, in which environmental variables with a correlation of greater than 0.4 were displayed.

Separate PERMANOVAs were run on (1) the multivariate meiofaunal community matrix; (2) the univariate variables, total abundance and the taxon richness of meiofauna; and (3) univariate abundances of the taxa identified by SIMPER as key contributors (dissimilarity to standard deviation ratio of > 1.3 ; Clarke and Warwick 2001) to significant multivariate differences in meiofaunal communities between disturbed and control treatments or among sites within the dredged ICOLL. PERMANOVAs used Bray–Curtis dissimilarities calculated between samples following log (x+1) transformation of data in the PRIMER software (Clarke and Warwick 2001).

3. Results

3.1 Site and ICOLL-scale impacts of dredge and fill on physico-chemical variables

Across the two control ICOLLs sampled, there was considerable temporal variability in physico-chemical variables (Table 1, Appendix 1), particularly in mean grain size and chlorophyll- α concentration (Table 2a,d; Appendix 2a,d; Figure 2a,d). Over and above this background spatio-temporal variation in control ICOLLs, there was impact of entrance dredging on environmental variables within Narrabeen ICOLL at the ICOLL-scale (Table 1 and 2, Figure 2, Appendix 1 and 2). This was most predominant in salinity measurements, which significantly increased in Narrabeen ICOLL following dredging. This increase was different to the temporal variation in the two control ICOLLs which followed similar patterns over time which may relate to the extent of rainfall (Table 2b, Figure 2b, Appendix 2b).

Table 1: PERMANOVA comparing change in environmental variables within control (C1, Dee Why; C2, Curl Curl) and a disturbed (D, Narrabeen) ICOLL, from before (B1, B2) to after (A1, A2, A3) a dredge and fill event. Within the disturbed estuary, temporal change was also compared between the site directly affected by dredging (d), the site affected by replenishment (r) and two control sites (S1, S2) outside of the directly disturbed area. Factors: Impact = Disturbed vs Control, date (5 levels for each of the sample dates, fixed), ICOLL (3 levels: nested in Impact, random) and site (4 levels in Narrabeen, 3 levels in Dee Why and Curl Curl, nested in ICOLL, random). Terms significant at $p < 0.05$ are highlighted in **bold**.

Source	Df	MS	F	P
Impact	1	251.70	0.83	0.591
Date	4	100.00	2.09	0.025
ICOLL (Impact)	1	261.13	6.43	0.010
Impact x date	4	39.24	0.84	0.707
Site (ICOLL (Impact))	7	40.64	13.86	<0.001
ICOLL (Impact) x date	4	40.50	4.91	<0.001
Date x site (ICOLL (Impact))	28	8.25	2.81	<0.001
Residual	100	2.93		
Total	149			

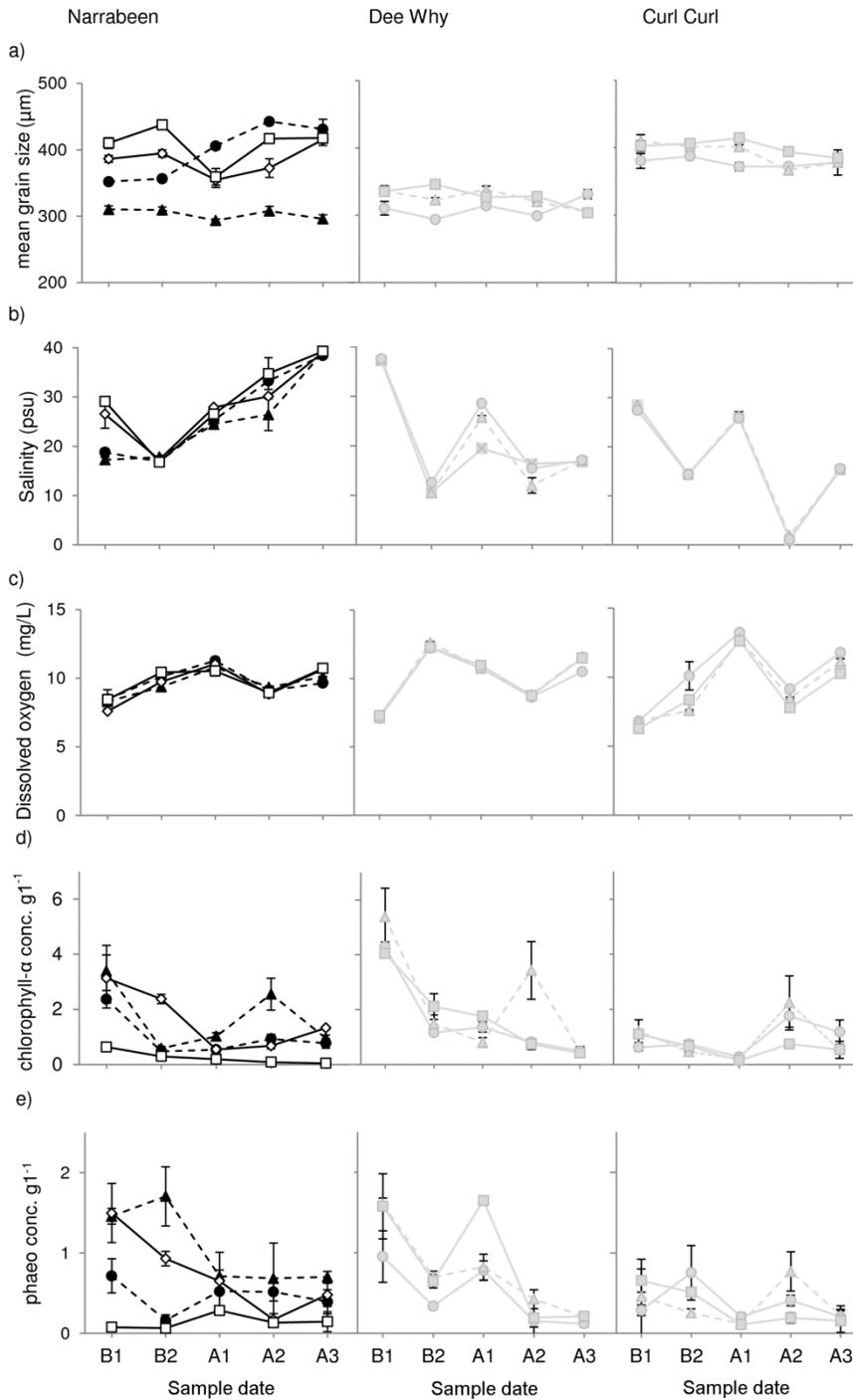


Figure 2: Mean (\pm SE) (a) mean grain size, (b) salinity, (c) dissolved oxygen, (d) chlorophyll- α and (e) phaeopigment concentration within the disturbed (D, Narrabeen) and the control (C1, Dee Why; C2, Curl Curl) ICOLLs, before (B1, B2) and after (A1, A2, A3) a dredge and fill event. Symbols represent the different sites within the disturbed estuary (dredged site: d \square ; replenished site: r \diamond ; and two control sites outside of the directly disturbed area: S1 \bullet , S2 \blacktriangle dashed lines) and the control estuaries (1 \bullet , 2 \blacktriangle dashed line, 3 \blacksquare); $n = 3$.

Table 2: PERMANOVA comparing change in univariate environmental variables: (a) mean grain size, (b) salinity, (c) dissolved oxygen, (d) chlorophyll- α and (e) phaeopigment concentration; within control (C1, Dee Why; C2, Curl Curl) and a disturbed (D, Narrabeen) ICOLL, from before (B1, B2) to after (A1, A2, A3) a dredge and fill event. Within the disturbed estuary, temporal change was also compared between the site directly affected by dredging (d), the site affected by replenishment (r) and two control sites (S1, S2) outside of the directly disturbed area. Factors: Impact = Disturbed vs Control, date (5 levels for each of the sample dates, fixed), ICOLL (3 levels: nested in Impact, random), and site (4 levels in Narrabeen, 3 levels in Dee Why and Curl Curl, nested in ICOLL, random). Terms significant at $p < 0.05$ are highlighted in **bold**.

Source	Df	a) Mean grain size			b) Salinity			c) Dissolved oxygen		
		MS	F	P	MS	F	P	MS	F	P
Impact	1	11981.00	0.12	0.913	2449.90	6.03	0.260	0.58	0.12	0.913
Date	4	563.10	4.57	0.078	917.47	3.41	0.124	68.30	2.98	0.151
ICOLL (Impact)	1	107050.00	6.87	0.049	339.08	11.99	0.014	6.51	3.55	0.103
Impact x date	4	3535.40	21.11	0.006	1498.5	5.56	0.062	6.90	0.31	0.912
Site (ICOLL (Impact))	7	15578.00	132.58	<0.001	28.29	19.36	<0.001	1.83	13.74	<0.001
ICOLL (Impact) x date	4	149.76	0.16	0.977	225.23	14.93	<0.001	19.13	36.63	<0.001
Date x site (ICOLL (Impact))	28	1291.80	11.00	<0.001	15.08	10.32	<0.001	0.52	3.91	<0.001
Residual	100	117.50			1.46			0.13		
Total	149									

Table 2. cont.

Source	Df	d) Chlorophyll- α			e) Phaeopigment		
		MS	F	P	MS	F	P
Impact	1	993270.00	0.06	0.965	294310.00	0.19	0.862
Date	4	11775000.00	2.21	0.234	1090200.00	1.01	0.520
ICOLL (Impact)	1	21467000.00	9.74	0.018	2142600.00	2.27	0.163
Impact x date	4	973880.00	0.22	0.961	145490.00	0.17	0.980
Site (ICOLL (Impact))	7	2205000.00	3.71	0.002	943000.00	9.80	< 0.001
ICOLL (Impact) x date	4	4508700.00	4.37	0.009	928710.00	4.82	0.005
Date x site (ICOLL (Impact))	28	1032800.00	1.74	0.027	192500.00	2.00	0.009
Residual	100	594740.00			96219.00		
Total	149						

At the smaller scale of sites within the impacted Narrabeen ICOLL, an impact of dredge and replenishment activities was evident on several physico-chemical variables (Table 2, Appendix 2, Figure 2). At the dredged site (d; Figure 2a), a reduction in mean grain size was evident from before to immediately following dredging of Narrabeen ICOLL— a pattern that was not replicated at the sites outside the area directly impacted by the dredge within Narrabeen Lake or in the control ICOLLs. By four months afterwards, however (time A2), sediment grain size at the dredged site had returned to a pre-disturbance condition (Table 2a, Appendix 2a, Figure 2a). At the dredged site, chlorophyll- α concentration decreased over the time period following dredging (Table 2d, Appendix 2d, Figure 2d); a trend that was similar to two sites within Dee Why ICOLL.

Mean grain size and phaeopigment concentration decreased at the site directly affected by replenishment, although the rates of change varied (r; Table 2a,e; Appendix 2a,e; Figure 2a,e). Mean grain size was reduced immediately following dredging (time A1). In subsequent sample dates it increased so that by eight months afterwards (time A3) mean sediment grain size was significantly greater than pre-dredge conditions (r; Table 2a, Appendix 2a, Figure 2a). Phaeopigment concentration in the replenished area significantly decreased four months following the dredge activity (time A2) and continued to decrease eight months afterwards (time A3); indicating lowered productivity at this site (Table 2e, Appendix 2e, Figure 2e). However, a similar trend was observed within Dee Why, one of the control ICOLLs. Chlorophyll- α concentration appeared lower at the replenishment site during the period immediately

following dredge activity and four months afterwards but this was not found to be a significant change from the first pre-dredge time period (time B1), partly due to a large amount of initial variation at this site at this time (Table 2d, Appendix 2d, Figure 2d).

The two sites within Narrabeen ICOLL that were not directly affected by the dredge activity also showed variation in environmental factors that were not repeated in control ICOLLs. The mean grain size showed a large increase at one of the sites outside the area immediately affected by the disturbance (site 1), following dredging (Table 2a, Appendix 2a, Figure 2a). The phaeopigment concentration at the other site within Narrabeen ICOLL was also indirectly influenced by the dredge activity (site 2) showed a marked reduction following dredging which did not show any sign of recovery throughout the remainder of the sampling times (Table 2e, Appendix 2e, Figure 2e).

3.2 Site and ICOLL-scale impacts of dredge and fill on meiofauna

A total of 24 meiofaunal taxa were identified from the three ICOLLs. Overall, nematodes were numerically dominant, accounting for 39% of the total meiofauna. Other prevalent taxa were turbellarians (23%), ostracods (11%), harpacticoid copepods (9%) and copepod nauplii (5%). When examined in relation to environmental variables the sediment mean grain size, sediment sorting, salinity, dissolved oxygen and chlorophyll- α (log transformed) explained most of the variation at the ICOLL scale (BIC = 336.12, $R^2 = 0.43$, RSS = 23407; Figure 3). Of the environmental variables examined, a reduction in mean grain size and the concentration of chlorophyll- α in surface sediments appeared to be the best in explaining the divergence in meiofaunal communities between disturbed and control ICOLLs (Table 2, Appendix 2, Figure 3).

Prior to dredge and fill activities in Narrabeen ICOLL, there was little difference in meiofaunal communities among ICOLLs (Table 3, Appendix 3, Figure 3). Following dredging, however, greater change in meiofaunal communities was seen within Narrabeen than the two control ICOLLs, such that the meiofaunal communities of disturbed and control ICOLLs became increasingly dissimilar. The increasing dissimilarity following dredging between meiofaunal communities of disturbed and control ICOLLs could largely be attributed to change at the site of dredging (Table 3, Appendix 3, Figure 3). Nevertheless, the

other three sites within Narrabeen ICOLL showed idiosyncratic change following the disturbance that also contrasted seasonal patterns of change evident in meiofaunal communities at sites of the control ICOLLs.

Differences in meiofaunal community assemblage over the sampling period were evident when each ICOLL was examined separately (Figure 3b - extractions). Narrabeen ICOLL showed no evidence of a greater similarity between seasons or years for any of the sites. In the sampling periods prior to dredging (closed symbols) there was a similar unidirectional change at each site. However, there was no consistent direction between the sites to the changes exhibited following the dredge event (open symbols). In contrast, Dee Why showed some evidence of seasonality with B1 (Apr 2011) similar to A2 (Apr 2012) and B2 (Aug 2011) to A3 (Aug 2012) (Figure 3b). Curl Curl showed a large degree of seasonality and of similarity between all sites and sampling times with the exception of the December sampling (A1), which was dissimilar to the other dates (Figure 3b).

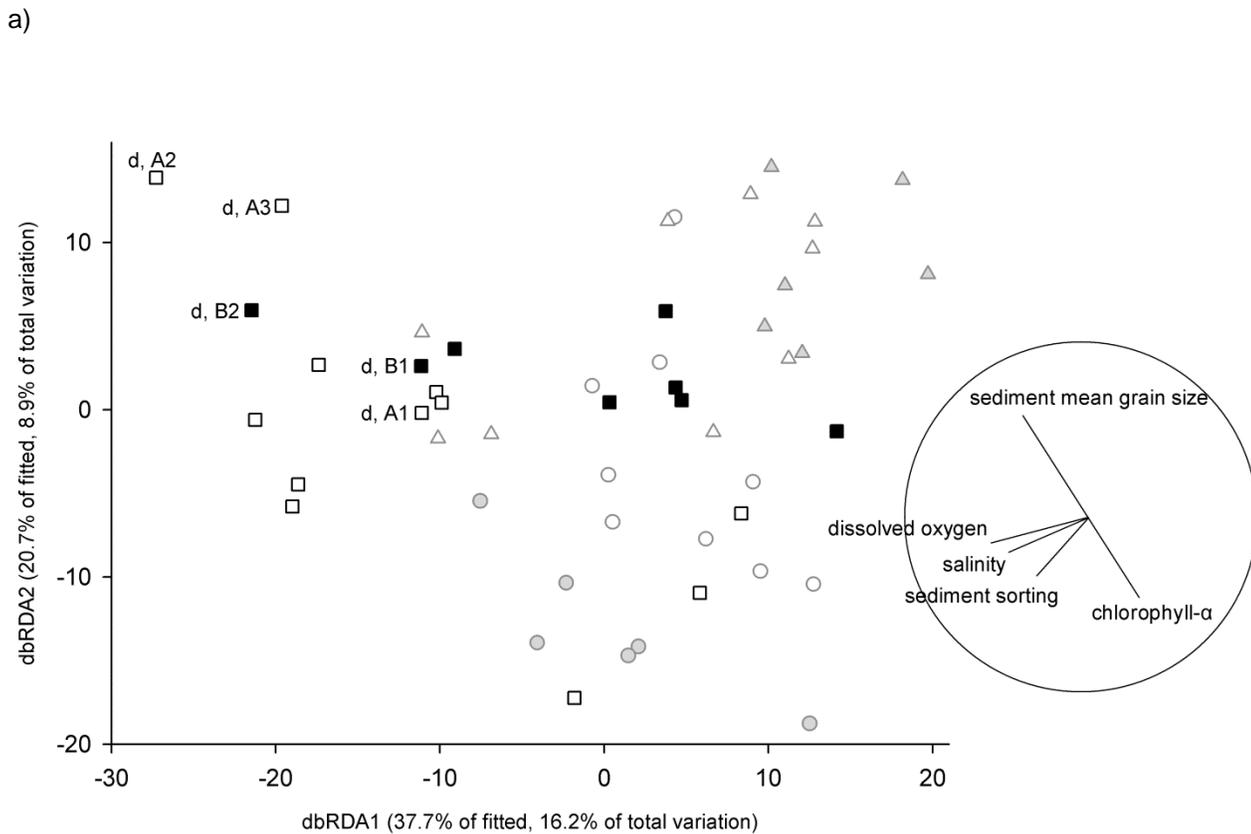
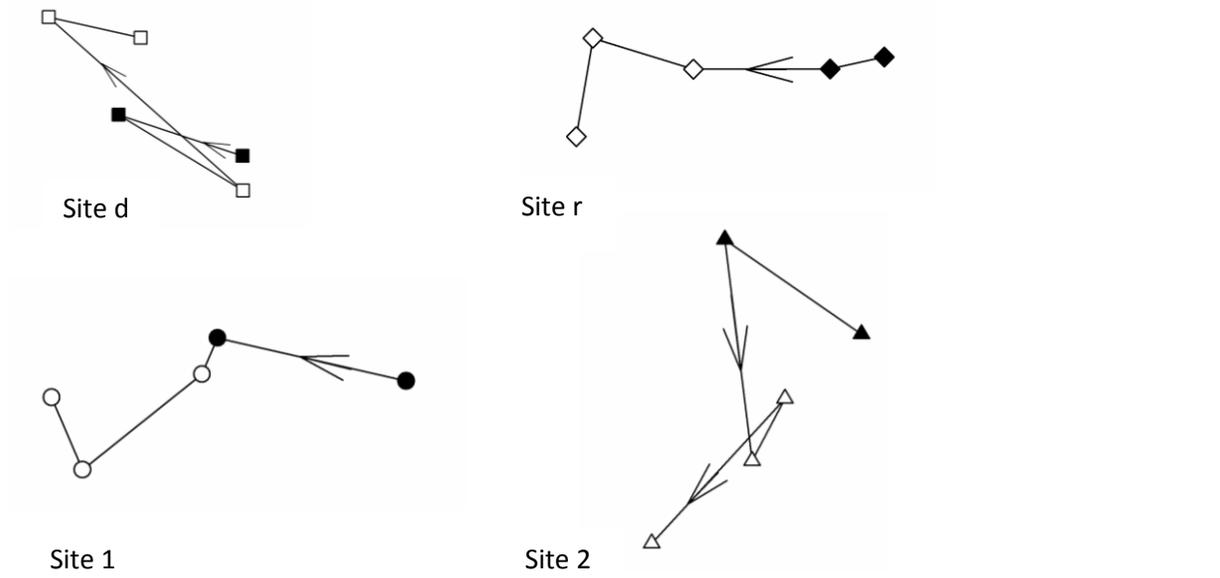
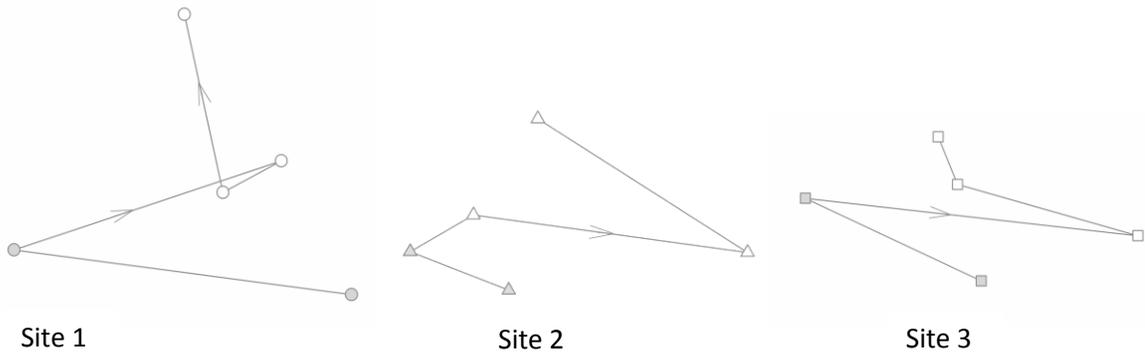


Figure 3: (a) Constrained ordinations (dbRDA) showing the relationship between spatio-temporal variation in environmental variables and meiofaunal communities within the disturbed ICOLL (D, Narrabeen ■) and two control ICOLLs (C1, Dee Why ●; C2, Curl Curl ▲). Points represent centroids for sites (d = site of dredging in Narrabeen ICOLL) at two times before (B1, B2, filled symbols), and three times after dredging (A1, A2, A3, open symbols). The length and direction of each vector represents the relationship between each dbRDA axis and environmental variables with a correlation greater than 0.4. (b) Extractions from the plot more clearly depict temporal trajectories of change for each of the sampling stations, in each ICOLL. Symbols represent the different sites, within Narrabeen: the site directly affected by dredging (d □), the site affected by replenishment (r ◇) and two control sites (S1 ●, S2 ▲) outside of the directly disturbed area; within the control ICOLLs: site 1 ●, site 2 ▲, and site 3 ■.

Narrabeen ICOLL:



Dee Why ICOLL:



Curl Curl ICOLL:

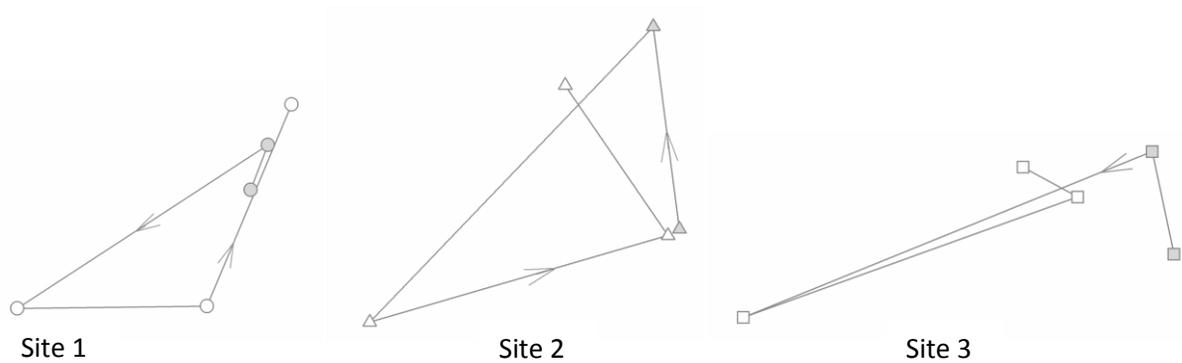


Figure 3 (b) cont.

Table 3: PERMANOVA comparing change in meiofaunal communities within control (C1, Dee Why; C2, Curl Curl) and a disturbed (D, Narrabeen) ICOLL, from before (B1, B2) to after (A1, A2, A3) a dredge and fill event. Within the disturbed estuary, temporal change was also compared between the site directly affected by dredging (d), the site affected by replenishment (r) and two control sites (S1, S2) outside of the directly disturbed area. Factors: Impact = Disturbed vs Control, date (5 levels for each of the sample dates, fixed), ICOLL (3 levels: nested in Impact, random) and site (4 levels in Narrabeen, 3 levels in Dee Why and Curl Curl, nested in ICOLL, random). Data were log (x+1) transformed prior to analysis. Terms significant at $p < 0.05$ are highlighted in **bold**.

Source	Df	MS	F	P
Impact	1	12008.00	1.00	0.484
Date	4	3106.80	0.50	0.977
ICOLL (Impact)	1	10612.00	2.54	0.030
Impact x date	4	4506.20	0.70	0.840
Site (ICOLL (Impact))	7	4175.10	11.44	<0.001
ICOLL (Impact) x date	4	5734.80	3.49	<0.001
Date x site (ICOLL (Impact))	28	1645.10	4.51	<0.001
Residual	150	365.12		
Total	199			

The change in the meiofaunal community at the site directly disturbed by dredging (d) from before to after the event appeared to be predominately due to an increase in the number of taxa (Table 4a, Appendix 4a, Figure 4a). By contrast, this site displayed little change in the abundance of meiofauna between times immediately before (B2) and after (A1, A2, A3) the disturbance (Table 4b, Appendix 4b, Figure 4b). At the site directly affected by replenishment, there was a decrease in taxon richness four months after the disturbance (A2), which had disappeared by the following sampling time (A3; Table 4a, Appendix 4a, Figure 4a). Interestingly, there was a strong pattern of increase in total meiofaunal abundance, immediately following the disturbance, at one of the Narrabeen sites outside of the directly disturbed area (site 1; Table 4b, Appendix 4b, Figure 4b). This pattern, which was not replicated at any other site, had however, disappeared by the second sampling time.

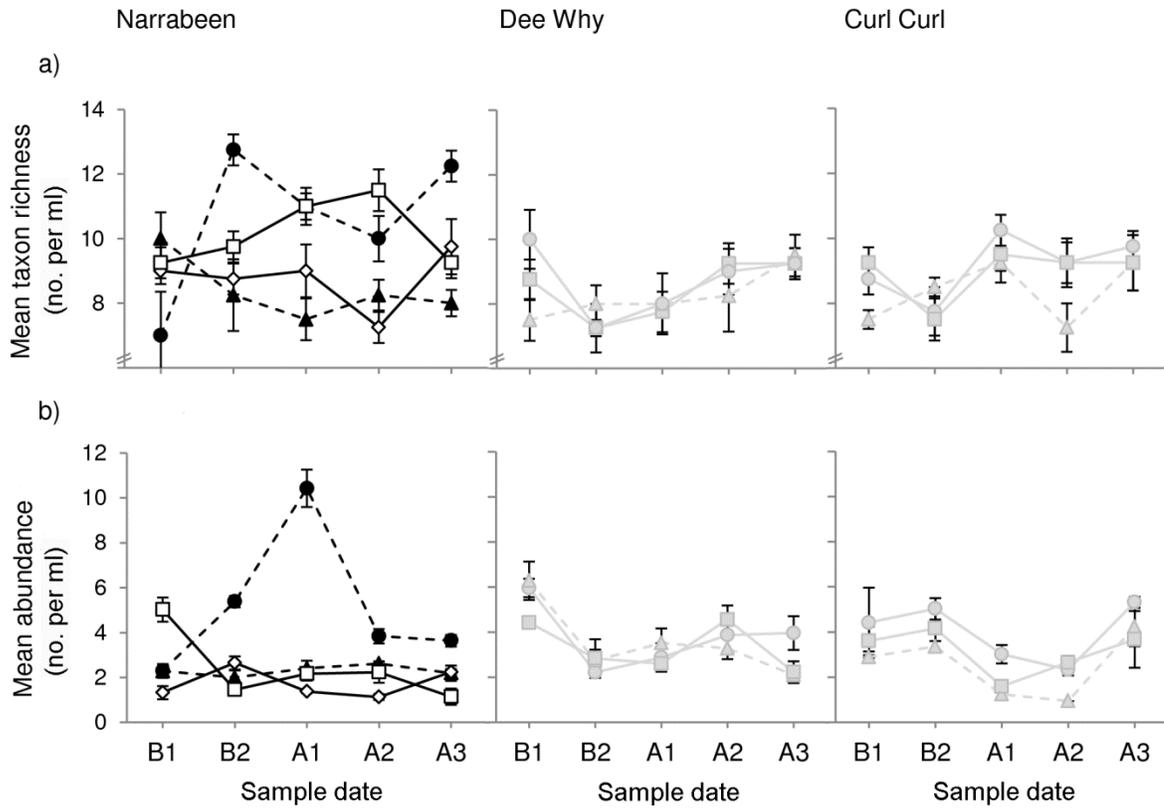


Figure 4: Mean (\pm SE) (a) taxon richness and (b) abundance of meiofauna in the disturbed (D, Narrabeen) and the control (C1, Dee Why; C2, Curl Curl) ICOLLs, before (B1, B2) and after (A1, A2, A3) a dredge and fill event. Symbols represent the different sites within the disturbed estuary (dredged site: d □; replenished site: r ◇; and two control sites outside of the directly disturbed area: S1 ●, S2 ▲ dashed lines) and the control estuaries (1●, 2 ▲ dashed line, 3 ■); $n = 4$.

Table 4: PERMANOVA comparing change in meiofauna: (a) taxon richness (no. per mL); and (b) abundance (no. per mL) within control (C1, Dee Why; C2, Curl Curl) and a disturbed (D, Narrabeen) ICOLL, from before (B1, B2) to after (A1, A2, A3) a dredge and fill event. Within the disturbed estuary, temporal change was also compared between the site directly affected by dredging (d), the site affected by replenishment (r) and two control sites (S1, S2) outside of the directly disturbed area. Factors: Impact = Disturbed vs Control, date (5 levels for each of the sample dates, fixed), ICOLL (3 levels; nested in Impact, random) and site (4 levels in Narrabeen, 3 levels in Dee Why and Curl Curl, nested in ICOLL, random). Terms significant at $p < 0.05$ are highlighted in **bold**.

Source	Df	(a) Number of taxa			(b) Number of individuals		
		MS	F	P	MS	F	P
Impact	1	816.37	5.65	0.274	5787.70	4.22	0.248
Date	4	161.60	1.20	0.454	936.86	0.24	0.984
ICOLL (Impact)	1	129.05	0.44	0.536	1274.80	0.38	0.605
Impact x date	4	174.15	1.28	0.427	1345.70	0.33	0.952
Site (ICOLL (Impact))	7	292.94	5.01	<0.001	3329.50	15.23	<0.001
ICOLL (Impact) x date	4	133.97	0.84	0.515	4014.90	3.31	0.008
Date x site (ICOLL (Impact))	28	159.63	2.73	<0.001	1212.80	5.55	<0.001
Residual	150	58.52			218.66		
Total	199						

Taxa identified by SIMPER as contributing most to spatio-temporal variation between control and disturbed ICOLLs were nematodes, harpacticoid copepods and turbellarians. Within the disturbed ICOLL, nematodes, harpacticoid copepods, copepod nauplii, turbellarians, ostracods, gastrotrichs, foraminifera and polychaetes, contributed most to spatio-temporal variation (Figure 5). Within Narrabeen ICOLL, variation in the abundance of discriminating taxa among sites and between sampling times was generally greater than in control ICOLLs, even before the disturbance (Table 5, Appendix 5, Figure 5). None of the taxa examined explained the increase in diversity at the dredged site following the disturbance, although nematodes, gastrotrichs and polychaetes all showed non-significant increases in abundance over this time. No taxon showed a differing temporal pattern between the dredge site or the replenishment site of Narrabeen ICOLL relative to the two other sites sampled within this ICOLL outside the area of immediate disturbance. By contrast, at one of these other sites of Narrabeen ICOLL (site 1), ostracods and foraminifera (Table 5 d,g; Appendix 5 d,g; Figure 5 d,g) appeared to exhibit a pulse response to the disturbance, increasing markedly in abundance immediately following the disturbance (time A1), but returning to pre-disturbance abundance by four months after (time A2). Polychaetes at this site increased in abundance following the disturbance, a trend that lasted at least the duration of study (Table 5 h; Appendix 5h; Figure 5h). At all other sites, these taxa did not display any significant temporal variation.

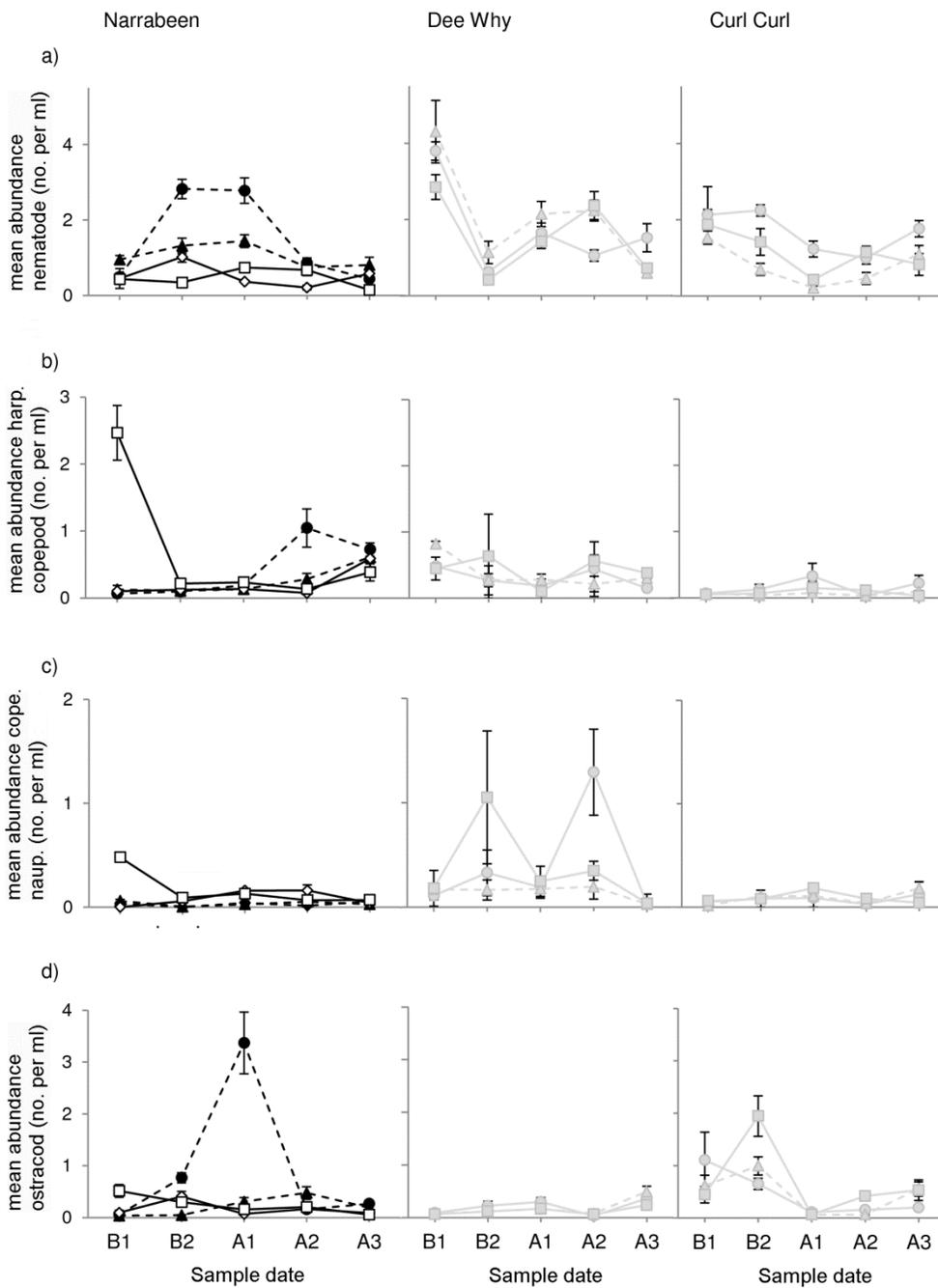


Figure 5: Mean (\pm SE) abundance of: (a) nematodes, (b) harpacticoid copepods, (c) copepod nauplii, (d) ostracods, (e) turbellarians, (f) gastrotrichs, (g) foraminifera and (h) polychaetes in the disturbed (D, Narrabeen) and the control (C1, Dee Why; C2, Curl Curl) ICOLLs, before (B1, B2) and after (A1, A2, A3) a dredge and fill event. Symbols represent the different sites within the disturbed estuary (dredged site: d \square ; replenished site: r \diamond ; and two control sites outside of the directly disturbed area: S1 \bullet , S2 \blacktriangle dashed lines) and the control estuaries (1 \bullet , 2 \blacktriangle dashed line, 3 \blacksquare). Note that different scale bars are used; $n = 4$.

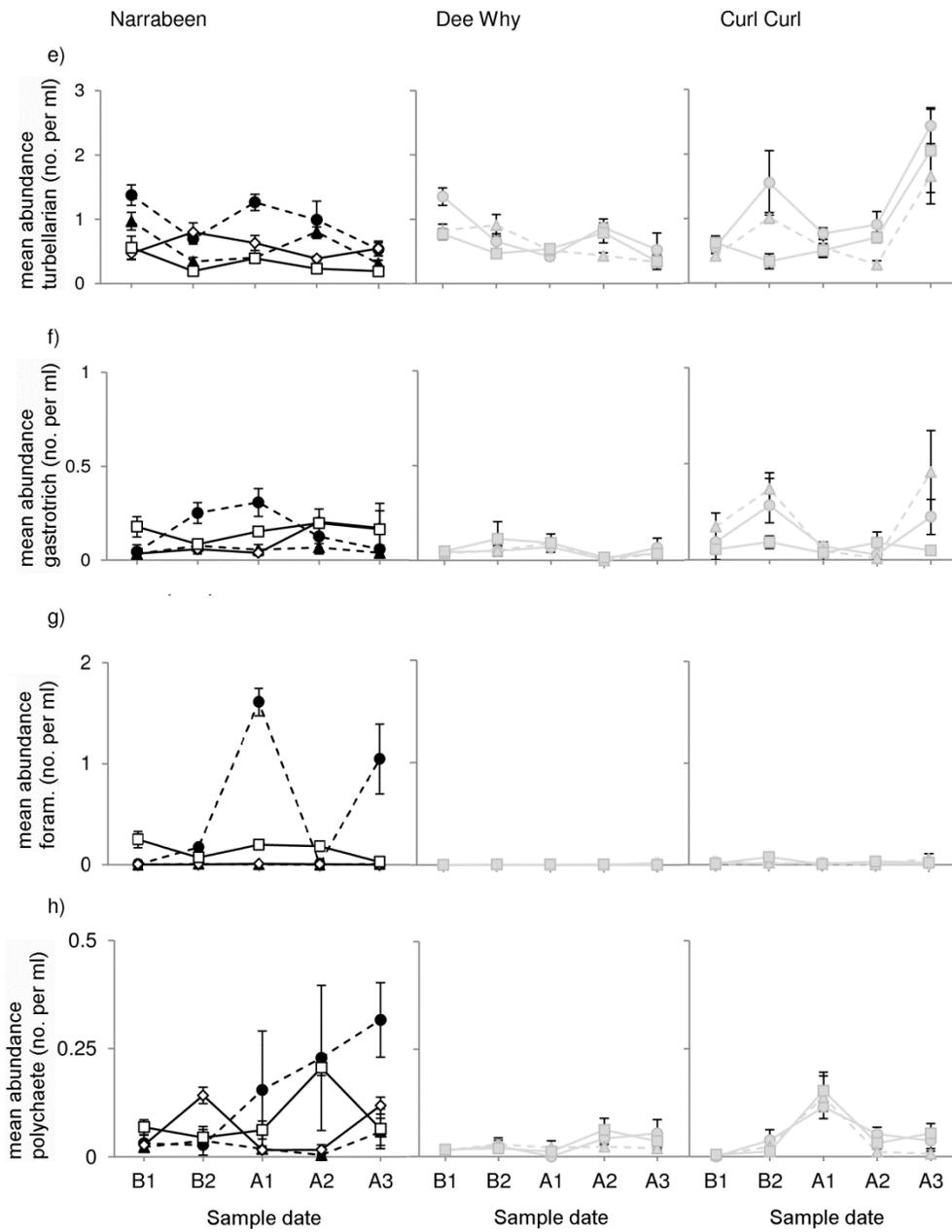


Figure 5. cont.

Table 5: PERMANOVA examining spatial and temporal sources of variability in mean abundance of individual meiofaunal taxa (standardised to 1 mL) which contribute the most to variation between ICOLLS, sites and sampling dates (based on SIMPER analysis): (a) nematode, (b) harpacticoid copepod, (c) copepod nauplii, (d) ostracod, (e) turbellarian, (f) gastrotrich, (g) foraminifera and (h) polychaete; within control (C1, Dee Why; C2, Curl Curl) and a disturbed (D, Narrabeen) ICOLL, from before (B1, B2) to after (A1, A2, A3) a dredge and fill event. Within the disturbed estuary, temporal change was also compared between the site directly affected by dredging (d), the site affected by replenishment (r) and two control sites (S1, S2) outside of the directly disturbed area. Factors: Impact = Disturbed vs Control, date (5 levels for each of the sample dates, fixed), ICOLL (3 levels: nested in Impact, random) and site (4 levels in Narrabeen, 3 levels in Dee Why and Curl Curl, nested in ICOLL, random). Terms significant at $p < 0.05$ are highlighted in **bold**.

Source	Df	a) nematode			b) harpacticoid copepod			c) copepod nauplii		
		MS	F	P	MS	F	P	MS	F	P
Impact	1	18913.00	3.96	0.293	629.03	0.17	0.892	584.57	0.43	0.745
Date	4	2357.20	0.35	0.963	518.46	1.74	0.321	98.78	0.27	0.948
ICOLL (Impact)	1	4170.20	0.92	0.373	3264.50	13.43	0.005	1191.90	9.02	0.020
Impact x date	4	6269.30	0.86	0.611	435.16	1.49	0.371	160.12	0.40	0.858
Site (ICOLL (Impact))	7	4541.90	11.26	<0.001	243.07	7.22	<0.001	132.19	3.34	0.001
ICOLL (Impact) x date	4	6489.30	3.23	0.004	282.94	0.79	0.538	378.86	3.35	0.022
Date x site (ICOLL (Impact))	28	2012.20	5.00	<0.001	357.81	10.62	<0.001	113.18	2.86	<0.001
Residual	150	403.34			33.68			39.63		
Total	199									

Table 5. cont.

Source	Df	d) ostracod			e) turbellarian			f) gastrotrich		
		MS	F	P	MS	F	P	MS	F	P
Impact	1	806.15	0.06	1.000	4207.20	1.67	0.422	61.49	0.18	0.883
Date	4	5228.70	0.44	0.923	2083.70	0.33	0.951	43.69	0.50	0.790
ICOLL (Impact)	1	18810.00	7.38	0.016	2486.30	0.64	0.501	361.60	4.49	0.070
Impact x date	4	6929.30	0.56	0.827	2088.40	0.33	0.947	86.64	0.92	0.561
Site (ICOLL (Impact))	7	2547.60	3.35	<0.001	3905.10	9.85	<0.001	80.53	4.03	<0.001
ICOLL (Impact) x date	4	11326.00	3.30	0.006	5784.50	5.23	<0.001	85.09	2.36	0.074
Date x site (ICOLL (Impact))	28	3430.40	4.51	<0.001	1105.80	2.79	<0.001	36.05	1.81	0.012
Residual	150	759.97			396.59			19.96		
Total	199									

Source	Df	g) foraminifera			h) polychaete		
		MS	F	P	MS	F	P
Impact	1	1374.00	67.87	0.125	189.68	7.06	0.220
Date	4	228.92	181.84	<0.001	45.90	0.94	0.551
ICOLL (Impact)	1	18.18	0.03	0.892	23.28	0.62	0.492
Impact x date	4	235.91	186.61	<0.001	36.22	0.76	0.645
Site (ICOLL (Impact))	7	531.44	74.47	<0.001	37.64	4.21	<0.001
ICOLL (Impact) x date	4	1.22	0.01	1.000	44.04	2.32	0.077
Date x site (ICOLL (Impact))	28	188.39	26.40	<0.001	18.95	2.12	0.002
Residual	150	7.14			8.93		
Total	199						

4. Discussion

This study found an effect of dredging on meiofaunal communities of Narrabeen ICOLL which was evident in, but not limited to, a change in community structure at the dredged site. Disposal of dredge sediments at an alternative site in the ICOLL, although temporarily altering sediment characteristics, did not modify meiofaunal communities. Despite the steady increase in the salinity of Narrabeen ICOLL following the dredging event, changes to meiofaunal communities did not occur uniformly at the scale of the whole ICOLL. The meiofaunal communities sampled during this study were dominated by directly developing taxa (e.g. nematodes), that lack a larval stage in their life history. Hence, it is possible that marine taxa did not colonise following entrance opening because they were dispersal-limited. Alternatively, the coarse taxonomic resolution utilised by this study may have been insufficient to detect estuary-wide shifts towards marine taxa. Instead, site-specific changes to meiofaunal communities within the disturbed ICOLL were observed.

At the dredge site, where changes to meiofaunal communities following the disturbance were particularly pronounced, chlorophyll- α decreased following the disturbance while sediment grain size experienced a large decrease immediately following disturbance. The initial disturbance caused by dredging, and the resulting high levels of bedload transport experienced in this area, would increase abrasion and decrease microphytobenthos densities (Miller 1989). Microphytobenthos are an important food source for many meiofauna (Montagna 1995, Leduc 2009) and play an important role in ecosystem recovery (Wulff et al. 1997). Hence, it is likely that changes to resources availability, in a large part, drove changes to the meiofaunal population (Coull 1999). Changes to sediment grain size extended beyond the dredge site to another of the sites sampled, with changes in meiofaunal communities following this change in sediments.

The change to the meiofaunal community at the dredge site following the disturbance was driven by changes in meiofaunal taxon composition, as opposed to abundance. Taxon richness increased following the disturbance but there was little change in the abundance of any of the taxa. This pattern is consistent with other studies that have found the taxon richness of meiofauna to increase in response to dredging and other disturbances which resulted in sediment heterogeneity (Tietjen 1991, Schratzberger and Jennings 2002). Outside of the dredged area, there were some large changes in the abundance of

specific taxa following the disturbance. In particular, the abundance of ostracods increased at one site immediately following dredging, the increased tidal currents of the now open ICOLL perhaps facilitating recruitment (Armonies 1994). Polychaetes appeared to exhibit a pulse response to the disturbance at the dredged site, and a press increase at an adjacent site not directly affected by the dredge activity which continued to increase following the disturbance. Increased polychaete abundance may be due to changes in sediment structure altering the habitat and potentially increasing food availability for certain taxa (Snelgrove and Butman 1994, Hutchings 1998).

There were pre-existing differences in abiotic factors, particularly mean sediment grain size, among ICOLLs, which were correlated to significant spatio-temporal variability in meiofaunal communities. Previous studies in temperate and tropical locations have likewise found large differences in meiofaunal communities among geographically proximate and geomorphically similar estuaries (Coull 1986, Hodda and Nicholas 1986, Alongi 1987, Dye and Barros 2005). Overall, Narrabeen displayed greater variation in meiofaunal communities among sites and sampling times than control ICOLLs, even before the dredging disturbance. This pattern may be influenced by the cumulative impacts of dredging and other forms of management, which may increase spatial variability of marine invertebrates (Rodrigues et al. 2012). In a survey of the NSW coastline, Dye and Barros (2005) found greater spatial variability of meiofauna at the entrances of managed than unmanaged ICOLLs.

During this study, natural overtopping of the sandbar and the opening of Curl Curl and Dee Why ICOLLs contributed to temporal variation in environmental conditions, in particular salinity. Nevertheless, the impact of the dredging on Narrabeen ICOLL was greater than the impacts of the natural short-term openings of the two control ICOLLs. In a study along the NSW coastline, Dye and Barros (2005) found that meiofaunal assemblages were generally similar between ICOLLs kept artificially open and those that were naturally open. Univariate investigations, however, found that some individual taxa were generally more spatially variable within managed ICOLLs (Dye 2005). While coastal environments are naturally very dynamic, it has been shown that the consequences of physical disturbance may persist for greater than six years (Boyd et al. 2005), which is longer than the period between artificial entrance intervention in most managed urban ICOLLs (Cameron et al. 2008).

As coastal settlements enlarge, and storm surge and rising sea levels increasingly threaten coastal infrastructure (Cooper and Lemckert 2012), the pressure on local governments to periodically open ICOLL entrances to mitigate flood risk is likely to increase (Haines et al. 2006). This study has provided evidence for direct and indirect impacts of ICOLL dredging on meiofaunal communities. As a critical link between primary production and shorebirds and fish, impacts to meiofauna have the potential to propagate through the food web. Hence, future decisions to open ICOLLs should take into consideration not only flood mitigation benefits but impacts to ecological values. Studies are now required to ascertain how the timing and frequency of entrance opening influences impacts.

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Appendices

Appendix 1: Post-hoc tests from PERMANOVA comparing change in environmental variables within control (C1, Dee Why; C2, Curl Curl) and a disturbed (D, Narrabeen) ICOLL, from before (B1, B2) to after (A1, A2, A3) a dredge and fill event. Within the disturbed estuary, temporal change was also compared between the site directly affected by dredging (d), the site affected by replenishment (r) and two control sites (S1, S2) outside of the directly disturbed area. Factors: Impact = Disturbed vs Control, date (5 levels for each of the sample dates, fixed), ICOLL (3 levels: nested in Impact, random) and site (4 levels in Narrabeen, 3 levels in Dee Why and Curl Curl, nested in ICOLL, random). Terms significant at $p < 0.05$ are highlighted in **bold**.

<u>Date x site (ICOLL (Impact)):</u>	<p>Differences among times within sites:</p> <p>C1 S1: B1 \neq B2 \neq A1 \neq A2 \neq A3</p> <p>C1 S2: B1 \neq B2 \neq A1 \neq A2 \neq A3</p> <p>C1 S3: B1 \neq B2 \neq A1 \neq A2 \neq A3</p> <p>C2 S1: B1 \neq B2 \neq A1 \neq A2 \neq A3</p> <p>C2 S2: (B1 = B2) \neq A1 \neq A2 \neq A3</p> <p>C2 S3: B1 \neq (B2 = A1) \neq A2 \neq A3</p> <p>D d: B1 \neq B2 \neq A1 \neq (A2 = A3)</p> <p>D r: B1 \neq B2 \neq A1 \neq A2 \neq A3</p> <p>D S1: B1 \neq B2 \neq A1 \neq (A2 = A3)</p> <p>D S2: B1 \neq B2 \neq (A1 = A2 = A3)</p> <p>Differences among sites within ICOLLs, at each time:</p> <p>Within C1, at B1, A1, A3: S1 = S2 = S3; B2, A2: S1 \neq S2 \neq S3</p> <p>Within C2, at B1, B2, A1, A2, A3: S1 = S2 = S3</p> <p>Within D, at B1, A2, A3: (d = r = S1) \neq S2; B2, A1: d \neq r \neq S1 \neq S2.</p>
<u>ICOLL (Impact) x date:</u>	<p>Differences among times within ICOLLs:</p> <p>C1: B1 \neq (B2 = A3) \neq A1 \neq A2</p> <p>C2: (B1 = A2 = A3) \neq B2 \neq A1</p> <p>D: B1 \neq (B2 = A2) \neq A1 \neq A3</p> <p>Differences among ICOLLs within times:</p> <p>All dates: C1 \neq C2</p>

Appendix 2: Post-hoc tests of PERMANOVA comparing change in univariate environmental variables: (a) mean grain size, (b) salinity, (c) dissolved oxygen, (d) chlorophyll- α and (e) phaeopigment concentration; within control (C1, Dee Why; C2, Curl Curl) and a disturbed (D, Narrabeen) ICOLL, from before (B1, B2) to after (A1, A2, A3) a dredge and fill event. Within the disturbed estuary, temporal change was also compared between the site directly affected by dredging (d), the site affected by replenishment (r) and two control sites (S1, S2) outside of the directly disturbed area. Factors: Impact = Disturbed vs Control, date (5 levels for each of the sample dates, fixed), ICOLL (3 levels: nested in Impact, random) and site (4 levels in Narrabeen, 3 levels in Dee Why and Curl Curl, nested in ICOLL, random). Terms significant at $p < 0.05$ are highlighted in **bold**.

(a) Mean grain size	(b) Salinity	(c) Dissolved oxygen
<u>Date x site (ICOLL(Impact)):</u>		
Differences among times within sites:	Differences among times within sites:	Differences among times within sites:
C1 S1: B1 = A2 = A3 \neq (B2 = A1)	C1 S1: B1 \neq B2 \neq A1 \neq A2 \neq A3	C1 S1: B1 \neq B2 \neq A1 \neq A2 \neq A3
C1 S2: (B1 = B2 = A1 = A2) \neq A3	C1 S2: B1 \neq (B2 = A2) \neq A1 \neq A3	C1 S2: B1 \neq B2 \neq A1 \neq A2 \neq A3
C1 S3: (B1 = B2 = A2) \neq A1 \neq A3	C1 S3: B1 \neq B2 \neq A1 \neq (A2 = A3)	C1 S3: B1 \neq (B2 = A3) \neq A1 \neq A2
C2 S1: B1 = B2 = A1 = A2 = A3	C2 S1: (B1 = A1) \neq B2 \neq A2 \neq A3	C2 S1: B1 \neq (B2 = A2 = A3) \neq A1
C2 S2: (B1 = B2 = A1) \neq A2 \neq A3	C2 S2: (B1 = A1) \neq B2 \neq A2 \neq A3	C2 S2: B1 \neq B2 \neq A1 \neq A2 \neq A3
C2 S3: B1 = (B2 \neq A1 \neq A2 \neq A3)	C2 S3: B1 \neq B2 \neq A1 \neq A2 \neq A3	C2 S3: B1 \neq (B2 = A2) \neq A1 \neq A3
D d: (B1 = A2 = A3) \neq B2 \neq A1	D d: B1 \neq B2 \neq A1 \neq A2 \neq A3	D d: B1 \neq (B2 = A1) \neq A2 \neq A3
D r: (B1 = B2 = A2) \neq A1 \neq A3	D r: (B1 = A1) \neq B2 \neq A2 \neq A3	D r: B1 \neq B2 \neq A1 \neq A2 \neq A3
D S1: (B1 = B2) \neq A1 \neq (A2 = A3)	D S1: (B1 = B2) \neq A1 \neq A2 \neq A3	D S1: (B1 = B2) \neq A1 \neq A2 \neq A3
D S2: B1 = B2 = A1 = A2 = A3	D S2: (B1 = B2 = A2) \neq A1 \neq A3	D S2: B1 \neq (B2 = A2) \neq A1 \neq A3
Differences among sites within ICOLLs at each time:	Differences among sites within ICOLLs at each time:	Differences among sites within ICOLLs, at each time:
Within C1, at B1, A1: S1 = S2 = S3; B2: S1 \neq S2 \neq S3; A2, A3: S1 \neq (S2 = S3)	Within C1, at B1, A2, A3: S1 = S2 = S3; B2: S1 \neq (S2 = S3); A1: S1 \neq S2 \neq S3	Within C1, at B1, B2, A2: S1 = S2 = S3; A1: (S1 = S2) \neq S3; A3: S1 \neq (S2 = S3)
Within C2, at B1, B2, A3: S1 = S2 = S3; A1: S1 \neq (S2 = S3); A2: (S1 = S2) \neq S3.	Within C2, at B1, B2, A1, A3: S1 = S2 = S3; A2: (S1 = S3) \neq S2	Within C2, at B1: (S1 = S2) \neq S3;
Within D, at B1, A1: (d = r) \neq S1 \neq S2; B2, A2: d \neq r \neq S1 \neq S2; A3: (d = r = 1) \neq 2	Within D, at B1: d \neq r \neq (S1 = S2); B2: (d = S1) \neq r \neq 2; A1: d \neq r \neq (S1 = S2); A2: d \neq (r = S1) \neq S2; A3: (d = r = S2) \neq S1	B2, A1: S1 = S2 = S3; A2, A3: S1 \neq S2 \neq S3
		Within D, at B1: d \neq (r = S1 = S2), B2, A1: d \neq r \neq S1 \neq S2; A2: (d = r = S1) \neq S2; A3: d = r = S1 = S2

ICOLL(Impact) x date:

not significant

Differences among times within

ICOLLS:

C1: $B1 \neq B2 \neq (A1 = A2 = A3)$ **C2:** $B1 \neq B2 \neq A1 \neq A2 \neq A3$ **D:** $(B1 = B2 = A1) \neq A2 \neq A3$

Differences among ICOLLS

within times:

B1, B2, A2, A3: $C1 \neq C2$; **A1:** $C1 = C2$

Differences among times within

ICOLLS:

C1: $B1 \neq B2 \neq (A1 = A3) = A2$ **C2:** $(B1 = B2 = A2) \neq A1 \neq A3$ **D:** $B1 \neq (B2 = A1 = A2) \neq A3$

Differences among ICOLLS

within times:

B1, B2, A1: $C1 \neq C2$; **A2, A3:** $C1 = C2$ (d) Chlorophyll- α (e) PhaeopigmentDate x site (ICOLL(Impact)):

Differences among times within sites:

C1 S1: $B1 = B2 = A1 = A2 = A3$ **C1 S2:** $B1 = B2 = A1 = (A2 \neq A3)$ **C1 S3:** $(B1 = B2 = A1 = A2) \neq A3$ **C2 S1:** $(B1 = B2 = A1 = A3) \neq A2$ **C2 S2:** $B1 = (B2 \neq A1) = A2 = A3$ **C2 S3:** $(B1 \neq A1 \neq A2) = B2 = A3$ **D d:** $B1 \neq B2 \neq A1 \neq A2 \neq A3$ **D r:** $B1 = (B2 \neq A1 \neq A2 \neq A3)$ **D S1:** $B1 \neq (B2 = A1 = A2 = A3)$ **D S2:** $B1 \neq B2 \neq (A1 = A2 = A3)$

Differences among sites within ICOLLS, at each time:

Within C1, at B1, B2, A1, A3: $S1 = S2 = S3$; **A2:** $(S1 = S3) \neq S2$ **Within C2, at B1, B2, A1, A2, A3:** $S1 = S2 = S3$ **Within D, at B1:** $d \neq (r = 1 = 2)$; **B2:** $(d = S1) \neq r \neq S2$; **A1:** $d \neq (r = S1) \neq S2$; **A2:** $d \neq r \neq S1 \neq S2$; **A3:** $(d = S1 = S2) \neq r$ ICOLL(Impact) x date:

Differences among times within ICOLLS:

C1: $(B1 \neq A3) = B2 = A1 = A2$ **C2:** $B1 = (B2 \neq A1) = A2 = A3$ **D:** $(B1 = B2) \neq (A1 = A2 = A3)$

Differences among ICOLLS within times:

B1, B2, A1: $C1 \neq C2$; **A2, A3:** $C1 = C2$

Differences among times within sites:

C1 S1: $(B1 = B2 = A1 = A3) \neq A3$ **C1 S2:** $(B1 = B2 = A1 = A2) \neq A3$ **C1 S3:** $B1 = A1 \neq (B2 = A2 = A3)$ **C2 S1:** $B1 \neq B2 \neq (A1 = A2) \neq A3$ **C2 S2:** $B1 = B2 = A1 = A2 = A3$ **C2 S3:** $B1 = B2 \neq (A1 = A2 = A3)$ **D d:** $B1 = B2 = A1 = A2 = A3$ **D r:** $(B1 = B2 = A1) \neq A2 \neq A3$ **D S1:** $B1 = B2 = A1 = A2 = A3$ **D S2:** $(B1 = B2) \neq (A1 = A2 = A3)$

Differences among sites within ICOLLS, at each time:

Within C1, at B1, A2, A3: $S1 = S2 = S3$; **B2:** $S1 \neq (S2 = S3)$; **A1:** $(S1 = S2) \neq S3$ **Within C2, at B1, B2, A1, A3:** $S1 = S2 = S3$; **A2:** $(S1 = S2) \neq S3$ **Within D, at B1:** $d \neq (r = S1) \neq S2$; **B2:** $(d = S1) \neq r \neq S2$; **A1:** $d \neq (r = S1 = S2)$; **A2:** $d = (r \neq S1) = S2$; **A3:** $d = r = S1 = S2$

Differences among times within ICOLLS:

C1: $B1 \neq (B2 = A1 = A2 = A3)$ **C2:** $B1 = B2 = A1 = A2 = A3$ **D:** $B1 = B2 = A1 = A2 = A3$

Differences among ICOLLS within times:

B1, A1: $C1 \neq C2$; **B2, A2, A3:** $C1 = C2$

Appendix 3: Post-hoc tests on PERMANOVA comparing change in meiofaunal communities within control (C1, Dee Why; C2, Curl Curl) and a disturbed (D, Narrabeen) ICOLL, from before (B1, B2) to after (A1, A2, A3) a dredge and fill event. Within the disturbed estuary, temporal change was also compared between the site directly affected by dredging (d), the site affected by replenishment (r) and two control sites (S1, S2) outside of the directly disturbed area. Factors: Impact = Disturbed vs Control, date (5 levels for each of the sample dates, fixed), ICOLL (3 levels: nested in Impact, random) and site (4 levels in Narrabeen, 3 levels in Dee Why and Curl Curl, nested in ICOLL, random). Data were log (x+1) transformed prior to analysis. Terms significant at $p < 0.05$ are highlighted in **bold**.

<u>Date x site</u>	Differences among times within sites:
<u>(ICOLL (Impact)):</u>	<p>C1 S1: B1 ≠ (B2 = A2) ≠ A1 ≠ A3</p> <p>C1 S2: (B1 = B2 = A1 = A2) ≠ A3</p> <p>C1 S3: (B1 = A2) ≠ B2 ≠ A1 ≠ A3</p> <p>C2 S1: (B1 = B2) ≠ A1 ≠ A2 ≠ A3</p> <p>C2 S2: B1 ≠ (B2 = A3) ≠ (A1 = A2)</p> <p>C2 S3: B1 ≠ B2 ≠ (A1 = A2 = A3)</p> <p>D d: B1 ≠ B2 ≠ (A1 = A2) ≠ A3</p> <p>D r: (B1 = A1 = A2) ≠ B2 ≠ A3</p> <p>D S1: B1 ≠ B2 ≠ A1 ≠ A2 ≠ A3</p> <p>D S2: B1 ≠ (B2 = A1) ≠ A2 ≠ A3</p>
	Differences among sites within ICOLLs, at each time:
	Within C1, at B1, B2, A1, A3: S1 = S2 = S3; A2: S1 ≠ S2 ≠ S3
	Within C2, at B1: S1 = S2 = S3; B2, A2: S1 ≠ S2 ≠ S3; A1, A3: S1 ≠ (S2 = S3)
	Within D, at B1, A3: d ≠ (r = S2) ≠ S1; B2, A1, A2: d ≠ r ≠ S1 ≠ S2
<u>ICOLL (Impact) x date:</u>	Differences among times within ICOLLs:
	C1: B1 = B2 = A1 = A2 = A3
	C2: B1 = B2 = A1 = A2 = A3
	D: B1 = (B2 ≠ A3) = (A1 ≠ A2) = A3
	Differences among ICOLLs within times:
	All dates: C1 ≠ C2

Appendix 4: Post-hoc tests on PERMANOVA comparing change in meiofauna: (a) taxon richness (no. per core); and (b) abundance (no. per core) within control (C1, Dee Why; C2, Curl Curl) and a disturbed (D, Narrabeen) ICOLL, from before (B1, B2) to after (A1, A2, A3) a dredge and fill event. Within the disturbed estuary, temporal change was also compared between the site directly affected by dredging (d), the site affected by replenishment (r) and two control sites (S1, S2) outside of the directly disturbed area. Factors: Impact = Disturbed vs Control, date (5 levels for each of the sample dates, fixed), ICOLL (3 levels: nested in Impact, random) and site (4 levels in Narrabeen, 3 levels in Dee Why and Curl Curl, nested in ICOLL, random). Terms significant at $p < 0.05$ are highlighted in **bold**.

(a) Number of taxa	(b) Number of individuals
<u>Date x site (ICOLL(Impact)):</u>	
Differences among times within sites:	Differences among times within sites:
C1 S1: (B1 = A1 = A2 = A3) ≠ B2	C1 S1: B1 = (B2 ≠ A1 ≠ A2 ≠ A3)
C1 S2: B1 = B2 = A1 = A2 = A3	C1 S2: B1 ≠ (B2 = A1 = A2 = A3)
C1 S3: B1 = B2 = A1 = A2 = A3	C1 S3: (B1 = B2 = A1 = A3) ≠ A2
C2 S1: B1 = (B2 ≠ A1) = A2 = A3	C2 S1: B1 = (B2 ≠ A1 ≠ A2) = A3
C2 S2: (B1 ≠ A1) = B2 = A2 = A3	C2 S2: (B1 = B2 = A3) ≠ (A1 = A2)
C2 S3: B1 = B2 = A1 = A2 = A3	C2 S3: B1 = (B2 ≠ A1 ≠ A2) = A3
D d: (B1 = B2 = A3) ≠ A1 ≠ A2	D d: B1 ≠ (B2 = A1 = A2 = A3)
D r: (B1 = B2 = A1 = A3) ≠ A2	D r: B1 = B2 = A3 ≠ (A1 = A2)
D S1: (B1 = B2) ≠ (A1 = A2 = A3)	D S1: B1 ≠ B2 ≠ A1 ≠ (A2 = A3)
D S2: (B1 = B2 = A3) ≠ A1 ≠ A2	D S2: B1 = B2 = A1 = A2 = A3
Differences among sites within ICOLLs each time:	Differences among sites within ICOLLs each time:
Within C1, at B1, B2, A1, A2, A3: S1 = S2 = S3	Within C1, at B1: (S1 = S2) ≠ S3; B2, A1, A2: S1 = S2 = S3; A3: S1 ≠ (S2 = S3)
Within C2, at B1: (S1 = S3) ≠ S2; B2, A1, A2, A3: S1 = S2 = S3	Within C2, at B1, A3: S1 = S2 = S3; B2, A2: (S1 = S3) ≠ S2; A1: S1 ≠ (S2 = S3)
Within D, at B1: d = r = S1 = S2; B2: (d = r = S2) ≠ S1; A1: (d = r = S1) ≠ S2; A2: (d = S1) ≠ (r = S2); A3: (d = r = S2) ≠ S1	Within D, at B1: d ≠ (r = S1 = S2); B2, A3: (d = r = S2) ≠ S1; A1, A2: (d = S2) ≠ r ≠ S1
<u>ICOLL (Impact) x date:</u>	
not significant	Differences among times within ICOLLs:
	C1: B1 = B2 = A1 = A2 = A3
	C2: B1 = B2 = A1 = A2 = A3
	D: B1 = B2 = A1 = A2 = A3
	Differences among ICOLLs within times:
	B1, B2: C1 ≠ C2; A1, A2, A3: C1 = C2

Appendix 5: Post-hoc tests for PERMANOVA comparing examining spatial and temporal sources of variability in mean abundance of individual meiofaunal taxa (standardised to 1 mL) which contribute the most to variation between ICOLLs, sites and sampling dates (based on SIMPER analysis): (a) nematode, (b) harpacticoid copepod, (c) copepod nauplii, (d) ostracod, (e) turbellarian, (f) gastrotrich, (g) foraminifera and (h) polychaete; within control (C1, Dee Why; C2, Curl Curl) and a disturbed (D, Narrabeen) ICOLL, from before (B1, B2) to after (A1, A2, A3) a dredge and fill event. Within the disturbed estuary, temporal change was also compared between the site directly affected by dredging (d), the site affected by replenishment (r) and two control sites (S1, S2) outside of the directly disturbed area. Factors: Impact = Disturbed vs Control, date (5 levels for each of the sample dates, fixed), ICOLL (3 levels: nested in Impact, random) and site (4 levels in Narrabeen, 3 levels in Dee Why and Curl Curl, nested in ICOLL, random). Terms significant at $p < 0.05$ are highlighted in **bold**.

(a) nematode	(b) harpacticoid copepod	(c) copepod nauplii
<u>Date x site (ICOLL(Impact)):</u>		
Differences among times within sites: C1 S1: $B1 \neq B2 \neq (A1 = A2 = A3)$ C1 S2: $(B1 = A1 = A2) \neq (B2 = A3)$ C1 S3: $(B1 = A2) \neq (B2 = A3) \neq A1$ C2 S1: $(B1 = B2 = A3) \neq (A1 = A2)$ C2 S2: $B1 \neq A1 \neq (B2 = A2 = A3)$ C2 S3: $(B1 = B2 = A2 = A3) \neq A1$ D d: $(B1 = B2 = A1 = A2) \neq A3$ D r: $B1 = (B2 \neq A1 \neq A2 \neq A3)$ D S1: $(B1 = A2 = A3) \neq (B2 = A1)$ D S2: $B1 = B2 = A1 = A2 = A3$	Differences among times within sites: C1 S1: $(B1 \neq A1) = B2 = (A2 \neq A3)$ C1 S2: $B1 \neq (B2 = A1 = A2 = A3)$ C1 S3: $(B1 = B2 = A2 = A3) \neq A1$ C2 S1: $B1 = B2 = A3 = (A1 \neq A2)$ C2 S2: $B1 = A1 = A2 = (B2 \neq A3)$ C2 S3: $B1 = B2 = A1 = A2 = A3$ D d: $B1 \neq (B2 = A1 = A2 = A3)$ D r: $(B1 = B2 = A1 = A2) \neq A3$ D S1: $(B1 = B2) \neq A1 \neq (A2 = A3)$ D S2: $(B1 = B2 = A1 = A2) \neq A3$	Differences among times within sites: C1 S1: $B1 = B2 = A1 = (A2 \neq A3)$ C1 S2: $(B1 \neq A3) = B2 = A1 = A2$ C1 S3: $B1 = A1 = (B2 \neq A2 \neq A3)$ C2 S1: $B1 = B2 = A1 = (A2 \neq A3)$ C2 S2: $(B1 = A2) \neq (B2 = A1 = A3)$ C2 S3: $(B1 = B2 = A2 = A3) \neq A1$ D d: $B1 \neq (B2 = A1 = A2 = A3)$ D r: $B1 = A1 = A2 = (B2 \neq A3)$ D S1: $B1 \neq A1 \neq (B2 = A2 = A3)$ D S2: $(B1 \neq B2) = A1 = A2 = A3$
Differences among sites within ICOLLs, at each time: Within C1, at B1, B2, A1, A3: $S1 = S2 = S3$; A2: $S1 \neq (S2 = S3)$ Within C2, at B1, A3: $S1 = S2 = S3$; B2: $(S1 \neq S2) = S3$; A1: $S1 \neq (S2 = S3)$; A2: $S1 = (S2 \neq S3)$ Within D, at B1: $d = r = (S1 \neq S2)$; B2: $d \neq (r = S2) \neq S1$; A1: $d \neq r \neq S1 \neq S2$; A2: $(d = S1 = S2) \neq r$; A3: $d \neq (r = S1 = S2)$	Differences among sites within ICOLLs, at each time: Within C1, at B1, A3: $S1 = S2 = S3$; B2: $(S1 = S2) \neq S3$; A1: $(S1 \neq S3) = S2$; A2: $(S1 = S3) \neq S2$ Within C2, at B1, B2, A1, A2: $S1 = S2 = S3$; A3: $S1 = (S2 \neq S3)$ Within D, at B1, A3: $d \neq (r = S1 = S2)$; B2, A1: $d = r = S1 = S2$; A2: $(d = r = S2) \neq S1$	Differences among sites within ICOLLs, at each time: Within C1, at B1, B2, A1, A3: $S1 = S2 = S3$; A2: $S1 \neq (S2 = S3)$ Within C2, at B1: $(S1 = S3) \neq S2$, B2, A1, A2, A3: $S1 = S2 = S3$ Within D, at B1: $d \neq (r = S1 = S2)$; B2, A1: $(d = S1) \neq (r = S2)$; A2, A3: $d \neq r \neq S1 \neq S2$
<u>ICOLL(Impact) x date:</u>		
Differences among times within ICOLLs: C1: $B1 = B2 = A1 = A2 = A3$ C2: $B1 = B2 = A1 = A2 = A3$ D: $B1 = B2 = A1 = A2 = A3$	not significant	Differences among times within ICOLLs: C1: $B1 = B2 = A1 = A2 = A3$ C2: $B1 = B2 = A1 = A2 = A3$ D: $B1 = B2 = A1 = A2 = A3$
Differences among ICOLLs within times: B1: $C1 \neq C2$; B2, A1, A2, A3: $C1 = C2$		Differences among ICOLLs within times: B1: $C1 \neq C2$; B2, A1, A2, A3: $C1 = C2$

d) ostracod	e) turbellarian	f) gastrotrich
<u>Date x site (ICOLL(Impact)):</u>		
Differences among times within sites: C1 S1: (B1 = B2 = A2) ≠ (A1 = A3) C1 S2: (B1 = B2) ≠ (A1 ≠ A2 ≠ A3) C1 S3: (B1 = B2 = A1 = A2) ≠ A3 C2 S1: (B1 = B2) ≠ (A1 = A2 = A3) C2 S2: (B1 = B2 = A3) ≠ (A1 = A2) C2 S3: (B1 = A1 = A2 = A3) ≠ B2 D d: (B1 = B2 = A1 = A2) ≠ A3 D r: (B1 = B2) ≠ (A1 = A2 = A3) D S1: B1 ≠ B2 ≠ A1 ≠ (A2 = A3) D S2: (B1 = B2) ≠ (A1 = A2 = A3)	Differences among times within sites: C1 S1: B1 ≠ (B2 = A1 = A2 = A3) C1 S2: (B1 = B2) ≠ (A1 = A2 = A3) C1 S3: (B1 ≠ B2) = A1 = A2 = A3 C2 S1: (B1 = B2 = A1 = A2) ≠ A3 C2 S2: (B1 = A1 = A2) ≠ (B2 = A3) C2 S3: B1 = A1 = A2 = (B2 ≠ A3) D d: (B1 = A1) ≠ (B2 = A2 = A3) D r: B1 = (B2 ≠ A2) = A1 = A3 D S1: (B1 = B2 = A1 = A2) ≠ A3 D S2: (B1 = A2) ≠ (B2 = A1 = A3)	Differences among times within sites: C1 S1: B1 = B2 = (A1 ≠ A2) = A3 C1 S2: B1 = B2 = A1 = A2 = A3 C1 S3: B1 = B2 = (A1 ≠ A2) = A3 C2 S1: B1 = B2 = A1 = (A2 ≠ A3) C2 S2: (B1 = A1 = A2) ≠ (B2 = A3) C2 S3: B1 = B2 = A1 = A2 = A3 D d: B1 = (B2 ≠ A2) = A1 = A3 D r: (B1 ≠ A3) = B2 = A1 = A2 D S1: B1 = (B2 ≠ A3) = A1 = A2 D S2: B1 = B2 = A1 = A2 = A3
Differences among sites within ICOLLs, at each time: Within C1, at B1, B2, A1, A2, A3: S1 = S2 = S3 Within C2, at B1, A1, A3: S1 = S2 = S3; B2: (S1 ≠ S3) = S2; A2: (S1 = S2) ≠ S3 Within D, at B1: d ≠ (r = S1 = S2); B2, A3: (d = r) ≠ (S1 = S2); A1: (d = r = S2) ≠ S1; A2: d = r = (S1 ≠ S2)	Differences among sites within ICOLLs, at each time: Within C1, at B1: S1 ≠ (S2 = S3), B2: S1 = (S2 ≠ S3), A1, A3: S1 = S2 = S3; A2: (S1 ≠ S2) = S3 Within C2, at B1, A1, A3: S1 = S2 = S3; B2: S1 = (S2 ≠ S3); A2: (S1 = S3) ≠ S2 Within D, at B1: d = (r ≠ S1) = S2; B2: (d = S2) ≠ (r = S1); A1: (d = r = S2) ≠ S1; A2: d ≠ r ≠ (S1 = S2); A3: (d ≠ r) = S1 = S2	Differences among sites within ICOLLs, at each time: Within C1, at B1, B2, A1, A2, A3: S1 = S2 = S3 Within C2, at B1, A1, A2: S1 = S2 = S3; B2, A3: S1 = (S2 ≠ S3) Within D, at B1: d ≠ (r = S1 = S2); B2: (d = r = S2) ≠ S1; A1: (d = S1) ≠ (r = S2); A2: (d ≠ S2) = r = S1; A3: d = r = S1 = S2
<u>ICOLL(Impact) x date:</u>		
Differences among times within ICOLLs: C1: B1 = B2 = A1 = A2 = A3 C2: B1 = B2 = A1 = A2 = A3 D: B1 = B2 = A1 = A2 = A3	Differences among times within ICOLLs: C1: B1 = B2 = A1 = A2 = A3 C2: B1 = B2 = A1 = A2 = A3 D: B1 = B2 = A1 = A2 = A3	not significant
Differences among ICOLLs within times: B1, B2, A2: C1 ≠ C2; A2, A3: C1 = C2	Differences among ICOLLs within times: B1, A3: C1 ≠ C2; B2, A1, A2: C1 = C2	

g) foraminifera

h) polychaete

Date x site (ICOLL(Impact)):

Differences among times within sites:

C1 S1: $B1 = B2 = A1 = A2 = A3$ **C1 S2:** $B1 = B2 = A1 = A2 = A3$ **C1 S3:** $B1 = B2 = A1 = A2 = A3$ **C2 S1:** $B1 = B2 = (A1 \neq A2) = A3$ **C2 S2:** $(B1 \neq B2) = A1 = A2 = A3$ **C2 S3:** $(B1 = A1 = A2 = A3) \neq B2$ **D d:** $(B1 = A1 = A2) \neq B2 \neq A3$ **D r:** $B1 = B2 = A1 = A2 = A3$ **D S1:** $(B1 = A2) \neq B2 \neq (A1 = A3)$ **D S2:** $B1 = B2 = A1 = A2 = A3$

Differences among sites within ICOLLs, at each time:

Within C1, at B1, B2, A1, A2, A3: $S1 = S2 = S3$ **Within C2, at B1, A2, A3:** $S1 = S2 = S3$; **B2:** $(S1 = S2) \neq S3$; **A2:** $S1 \neq (S2 = S3)$ **Within D, at B1, A2:** $d \neq (r = S1 = S2)$; **B2, A1, A3:** $d \neq (r = S2) \neq S1$

Differences among times within sites:

C1 S1: $A1 \neq (B1 = B2 = A2 = A3)$ **C1 S2:** $A1 = B1 = B2 = A2 = A3$ **C1 S3:** $A1 = B1 = B2 = A2 = A3$ **C2 S1:** $A1 \neq (B1 = B2 = A2 = A3)$ **C2 S2:** $A1 \neq (B1 = B2 = A2 = A3)$ **C2 S3:** $A1 \neq (B1 = B2 = A2 = A3)$ **D d:** $(B1 = B2 = A1 = A3) \neq A2$ **D r:** $(B1 = A1 = A2) \neq (B2 = A3)$ **D S1:** $(B1 = B2 = A1 = A2) \neq A3$ **D S2:** $B1 = B2 = A1 = A2 = A3$

Differences among sites within ICOLLs, at each time:

Within C1, at B1, B2, A1, A2, A3: $S1 = S2 = S3$ **Within C2, at B1, B2, A1, A2, A3:** $S1 = S2 = S3$ **Within D, at B1:** $d = r = S1 = S2$; **B2:** $(d = S1 = S2) \neq r$; **A1:** $d \neq (r = S1 = S2)$; **A2:** $(d = S1) \neq (r = S2)$; **A3:** $(d = r = S2) \neq S1$

7. General conclusion

7.1 Management and conservation of sandy beaches

7.1.1 *The extent of our knowledge*

Despite the large proportion of sandy beaches in Australia there is little published information on Australian beach management. Much of the information regarding coastal management in Australia is in the form of grey literature, or is collected at the Local Government scale and is not readily accessible for interpretation at State or National level. This thesis represents one of the few national assessments of sandy beach management in Australia (but see James 2000). In Chapter 2, the beach nourishment projects of 130 beaches in Australia were identified and examined. Compared to available accounts in international studies, most Australian projects were small in scale but frequent (Clayton 1991, Basco 1999, Hamm et al. 2002, Hanson et al. 2002, Campbell and Benedet 2006). Exceptions were nine bypass projects which utilised large volumes of sediment. Most nourishment, replenishment and beach scraping occurred in highly urbanised areas. Projects were generally a response to extreme weather events, and utilised sand from the same coastal compartment as the site of erosion. The primary reasons for management were to protect coastal infrastructure and public beach amenity and safety.

The need for management may be expected to increase with increasing urban population density and the spread of coastal development. Presently, approximately 20% of the sandy coastline of south-east Australia has been developed, but within the 21st century, this number may increase up to 60% as a relatively small part of the coast falls within the protected lands network (Langedijk 2008). With coastal development, the demand for soft engineering, such as beach nourishment, is likely to increase and will be exacerbated by increasing public awareness of coastal erosion problems, development of comprehensive coastal zone management programs, and rising sea levels and enhanced storminess (Davison et al. 1992).

Our past use of the coastal zone has reduced the resilience of this ecosystem (Casarin et al. 2012), largely due to a lack of understanding about the extent and importance of natural processes occurring in this environment (Hall and Pilkey 1991). Coastal policy reforms have been occurring in many States and Territories in Australia as a response to growing concerns about the coastal zone. The focus on

State-wide strategic planning instruments means that information will be consolidated at a State level. The first stage is the collection and integration of information about the coastal environment. The information presented in Chapter 2 will provide important material for the continued management and conservation of sandy beaches in Australia.

7.1.2 Monitoring change

This thesis found that coastal management was typically planned at a regional scale, with little monitoring of efficacy or biological impact (Chapter 2). As rising sea levels, changing climatic intensity and growing coastal populations put pressure on beaches (Defeo et al. 2009), information and knowledge transfer, enabling adaptive management, becomes increasingly important. If done correctly, monitoring information could be used to determine the 'success,' as well as the environmental impact, of a project (Peterson and Bishop 2005). This would enable calculation of appropriate cost/benefit analysis, assist implementation of planned retreat and allow managers to mitigate environmental effects. Ecological monitoring used in conjunction with other information such as wave climate data and beach geomorphology data would enable adaptive management.

Sandy beaches have been recognised as productive environments, providing essential goods and services (Defeo et al. 2009). They are, however, often overlooked as ecosystems requiring conservation or assessment (Dugan et al. 2010), receiving less protection than other coastal ecosystems such as wetlands or littoral rainforest (Thom 2003). This disregard may be due to the paucity of ecological studies conducted in this environment, particularly studies addressing the impacts of anthropogenic disturbance and climate change (Dugan et al. 2010). The amount of research conducted by ecologists on sandy beaches is small compared to other coastal systems (Fairweather 1990). Schoeman et al. (2003) suggest that if the methodology of ecological impact assessments in this environment was more robust, sandy beaches may be considered of deserving higher conservation priority.

7.2 Meiofauna as indicators of change

7.2.1 *Measuring change in the meiofaunal community*

Meiofauna are informative indicators as they are functionally important to sandy beach ecology (Montagna 1995, Giere 2009). They are numerous and diverse, and the community is seen to change under natural or anthropogenic pressures (Kennedy and Jacoby 1999). One of the reasons that meiofauna are not more widely used is their high level of spatial and temporal variability (Kennedy and Jacoby 1999). While this thesis has found a high level of variability of meiofauna in sandy beach environments, this variability was not random but largely related to scales of physical variation in the environment (Chapters 3 - 6). One of the important environmental variables determining spatial patterning in exposed sandy beaches was elevation above mean water level, which increased overall diversity by creating sub-environments where different taxa were found (Chapter 5). For the more productive sub-tidal environment of Intermittently Closed and Open Lakes and Lagoons (ICOLLs), change in meiofaunal communities was strongly correlated with microphytobenthic biomass (Chapter 6). These relationships with environmental variables increase the value of meiofauna as environmental indicators. However, due to the dynamic nature of the sandy beach environment, it does mean that assessment of impacts of climate change, coastal development and associated management strategies on meiofauna require carefully designed studies (Chapter 3 - 6).

Meiofaunal sampling needs to be conducted at appropriate temporal (Chapter 5 and 6) and spatial scales (Chapter 3 -6). This is particularly challenging on sandy beaches, which are the most physically variable of all marine environments (Nybakken and Bertness 2005). Currents, tides, and wave action largely determine the shape and sediment type of a beach (Defeo and MacLachlan 2005). These forces create a dynamic equilibrium and exposed sandy beaches may display morphological change over different time scales varying from a few hours to decades. Erosion and accretion processes disrupt sediment layers, causing changes in particle size and sorting and altering the interstitial environment (Giere et al. 1988). Significant variation in the physical conditions of the beach face, for example could produce changes in mean grain size that are not visually apparent (Chapter 3, 4 and 5) yet lead to changes in the meiofaunal community.

The challenge for assessments of ecological impact is to separate spatio-temporal variation, caused by the disturbance, from background spatio-temporal variation in biological assemblages (Chapter 6).

A previous study had found that meiofauna were reliable indicators of ICOLL management history and entrance dynamics (Dye and Barros 2005). This thesis examined how a single ICOLL management event comprising entrance dredging and beach fill influenced meiofaunal communities (Chapter 6). Variation in meiofaunal communities among sites within the dredged ICOLL was greater than among sites of control ICOLLs. This was due in part to pre-existing variation between the sites before dredging but appeared to be amplified by the dredge event. Examination of individual taxa found a large variation of responses which correspond to what is broadly known of their biology. This thesis reveals that meiofauna are sensitive indicators to a range of different conditions. Particularly in environments, such as ICOLLs, that display large inter-annual variability in environmental conditions (Roy et al. 2001), physico-chemical indicators (such as water quality) may not always relate to biological end-points (Fairweather 1999). Meiofauna appear to be good indicators of variations in conditions over the long-term (i.e. variation in catchment area size and management history), as well as the short-term (anthropogenic disturbances; Chapter 6).

7.2.2 Taxonomic resolution

This thesis has shown that assessments of meiofaunal communities that use coarse taxonomic levels of identification may allow the same level of discrimination between treatments or interventions as those that use finer taxonomic levels (in this case nematode genera; Chapter 4). This is consistent with previous studies which suggest that effects of anthropogenic disturbance may be as readily detected at coarse as species-level taxonomic resolution (Warwick 1988, Warwick 1993). It has been suggested that this is because anthropogenic disturbance modifies community composition at a higher taxonomic level than natural environmental variables (Warwick, 1988; Warwick, 1993).

The direction of change resulting from anthropogenic disturbance, however, is not always a decrease in mean values with respect to control sites (e.g. Fraschetti et al. 2006; Chapter 6). Identification to finer taxonomic levels better explains patterns of spatio-temporal variation (Chapter 3). In some cases finer resolution may provide more information about the mechanism behind biotic responses to changing environmental conditions than phylum level identification (Chapter 4). In general, this study found greater small-scale homogeneity in meiofaunal communities when coarse taxonomic levels were used than when nematode genera were instead examined (Chapter 3 and 4). When the nematode communities of three adjacent beaches were examined, within-site variability was still lower

than between-site variability, and for one site (Collaroy-Narrabeen beach) a much higher degree of homogeneity was found which was driven by one dominant genus (Chapter 4). This result was potentially more informative about different ecological processes occurring at this site than provided by analysis of data at coarser taxonomic levels. Whether increasing taxonomic resolution gives more information about the environment under study, or detracts from meaningful conclusions, appears to be related to the environment and the question posed by the study (Warwick and Gee 1984, Dye and Barros 2005). Furthermore, examination of ecotypes and guilds may provide more information about the relationships between particular taxa and the environment (Chapter 3; Robson et al. 2005).

Examining meiofauna at a coarse taxonomic scale is less time consuming and reduces the level of taxonomic skill involved (Kennedy and Jacoby 1999). As the majority of meiofauna have not been described to species level, undertaking identification beyond the level of genus becomes difficult. The development of cheaper and more accessible means of barcoding and ecogenomics will enable the rapid analysis of whole samples identified to the finest taxonomic level possible. While this technology still has some problems (Creer et al. 2010) it is very promising (Chariton et al. 2010). However, even with these advances towards rapid sampling analyses, choosing the appropriate sampling design (explored in Chapters 3 – 6) remains important in order to accurately detect and measure change.

7.3 Spatial and temporal variation

7.3.1 Spatial dependence in meiofaunal communities

Meiofauna in sandy beach environments are generally described as 'patchy' in their distribution (e.g. Coull and Feller 1988). Quantification of this patchiness is, however, rarely conducted by using appropriate experimental designs (Fraschetti et al. 2006). In examining the meiofaunal communities of exposed beaches, this thesis generally found the smallest amount of variation between short distances (between replicates or within sites); next largest variation at medium distances (between sites, or transects); and the largest amount of variation between geomorphic units (between beaches, or ICOLLs).

When examining ICOLLs (Chapter 6), this thesis found the greatest amount of variation at the largest scale of among ICOLLs, and at the medium scale of between sites, separated by 100s of metres in the dredged ICOLL; even before dredging occurred. At the smallest scale (within a site) there was no

significant variation at the reference sites while variation at this scale, in the dredged ICOLL, changed over time. This thesis revealed that, rather than random patchiness, meiofauna communities exhibit spatial dependency at different scales (Chapters 3 and 4), the extent of which may vary with disturbance (Chapter 4 and 6). This finding highlights the importance that sampling should be designed to accommodate the geomorphological characteristics of each area (Jackson et al. 2002) and adequately address variation at different scales (Kennedy and Jacoby 1999).

7.3.2 The importance of accounting for spatial and temporal variation

Meiofauna are a ubiquitous and ecologically important component of sandy beach ecosystems, yet the spatial scales across which they naturally vary are not well documented. The design of meaningful ecological assessments for determining human impacts to ecosystems hinges upon an understanding of the spatial structuring of their key biotic components. Environmental impact assessments test for variation between control sites versus variation between control and impacted sites. Revealing evidence of an impact relies on finding larger differences between treatments, rather than within treatments. There may be, however, a number of factors which confuse this test (Hurlbert 1984). For instance: (1) if there is a large degree of natural variation among beaches, then it may be difficult to detect an impact of a disturbance above this background noise (explored in Chapter 4); (2) if there is a lot of variability within a beach, there is a danger of not collecting enough samples to appropriately represent this variation (explored in Chapter 5); conversely, (3) if biota is patchily distributed and all samples are collected within a single patch, this may lead to the false conclusion that there is a large difference between areas (explored in Chapters 3 - 5). It is therefore necessary to determine the patch size of biota in this environment for the purposes of environmental impact assessment, or for undertaking any comparative study between beaches or coastal systems.

Spatial patterns are defined by ecological processes and underlying geomorphology (Legendre and Fortin 1989). Conservation of marine invertebrate diversity should be undertaken with the goal of maintaining existing spatial patterns (Gladstone 2006). To achieve this it is necessary to conserve ecological processes governing these patterns and also to manage or mitigate anthropogenic pressures to minimise their impact on these processes. Restoration of the sandy beach environment relies on the recovery of function of the key components underpinning the environment.

In urbanised environments, which face multiple stressors, coastal management in the form of soft engineering projects may restore some of the key ecological processes. Beach nourishment increases the width of the beach thereby restoring habitat which may have been lost due to coastal squeezing. Artificial entrance intervention of ICOLLs increases tidal flushing and allows the recruitment and transfer of species. However, the results from this thesis indicate that soft engineering may lead to changes in the diversity and spatial patterns of meiofauna, causing functional change to this ecosystem. If nourishment is to be continued as a means of protecting coastal property, a greater understanding of spatial patterning of biota in this environment will assist in the preparation of monitoring guidelines to facilitate adaptive management.

7.4 Sampling and monitoring recommendations

The significant small-scale variation (in metres) of meiofaunal communities which was correlated to environmental variables, points to the sensitivity of these fauna to habitat modification (Chapters 3 – 6). The sensitivity of meiofauna to various environmental conditions highlights the importance of good experimental design in monitoring studies, especially to reduce potential “sources of confusion” (Hurlbert 1984). For instance, this study has shown that meiofaunal communities on exposed beaches may vary between consecutive days and states of the tide. Simultaneous sampling of control and impact sites would mitigate interaction of temporal factors with the environmental variables under study (Chapter 5).

Examination of the extent of spatial autocorrelation is an important component of pilot studies in order to determine the most appropriate and representative sampling distance. Examination of spatial autocorrelation should also be a component of more complex study designs provided that the samples are collected from known positions (Thrush et al. 1989, Thrush et al. 1994, Leduc et al. 2012). On exposed sandy beaches information regarding elevation should be measured, as well as the coordinates of each sample, to enable the inclusion of these spatial variables in the analysis. In addition, a grid design with known coordinates may be the best general sampling design (Cole et al. 2001), as this methodology enables differences in spatial dependence along and across the beachface to be examined at the same time.

Sampling locations on a beach should be selected with reference to the beach geomorphology. In addition, due to the existence of abiotic variation which is not visually apparent in this environment (Chapter 3 – 5), the extent of variation on a single beach should be examined before comparing different sites. Where there are large pre-existing differences between control and impact beaches (Chapter 4), it is particularly important to sample at multiple times before and after a disturbance in order to adequately assess any impact; and where possible to also maximise the number of beaches sampled. A greater understanding of causal relationships between fauna and the environment; and of the way they vary spatially and temporally, will assist in the calculation of limits of acceptable change in the sandy beach environment (Ward and Jacoby 1992, McCool 1994, Fairweather 1999). This understanding would increase our capacity to undertake coastal management for conservation outcomes.

7.5 Conclusion

The coastal environment is facing increasing pressures from environmental change and anthropological impact (Doody 2001). The loss of habitat and associated biota, caused by accelerated beach erosion, is considered to be the most immediate and severe ecological threat to beaches affected by climate change (Schlacher et al. 2008, Slott et al. 2008). The recent Federal Government State of the Environment Report, cited beach or shoreline modification, leading to change or loss of habitat as a current and emerging risk to the marine environment (Hatton et al. 2011). This thesis has found that beach nourishment in Australia is primarily undertaken to protect property and infrastructure. The need for management may enlarge with increasing coastal urbanisation and the effects of climate change. The ability of coastal managers to respond to this changing social and natural environment relies on their capacity for adaptive management. An important component of management is the appropriate reporting of any coastal engineering works undertaken, and the collation and analysis of this information. In addition, the biological system must be understood in order to successfully conduct ecologically sustainable integrated coastal zone management, which is the fundamental principle on which most coastal policy is based (Krishnamurthy 2008). The results from this thesis indicate the importance of baseline studies of sandy beach biodiversity, and examination of the spatio-temporal scales across which it varies, in order to design appropriate studies to assess impacts of environmental change.

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