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Confronting the challenges of tidal flat
conservation: spatial patterns and human
impacts in a marine protected area in
southern NSW, Australia

Pia Carmen Winberg
University of Wollongong

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**Confronting the challenges of tidal flat conservation:
spatial patterns and human impacts in a Marine
Protected Area in southern NSW, Australia.**

A thesis submitted in fulfilment of the requirements for the award of the degree

DOCTOR OF PHILOSOPHY

from

UNIVERSITY OF WOLLONGONG

By

Pia Carmen Winberg, Bachelor of Science (Honours)

INSTITUTE FOR CONSERVATION BIOLOGY

SCHOOL OF BIOLOGICAL SCIENCES

2008

CERTIFICATION

I, Pia Carmen Winberg, declare that this thesis, submitted in partial fulfilment of the requirements for the award of Doctor of Philosophy, in the Institute for Conservation Biology, School of Biological Sciences, University of Wollongong, is wholly my own work unless otherwise referenced or acknowledged. The document has not been submitted for qualifications at any other academic institution. All work conducted for this dissertation was conducted under the NSW Marine Parks Permit JBMP 2004/018 Research Authority 2003/011 and the NSW DPI Research Permit P03/0062.

Pia Winberg

14th April 2008.

DEDICATION

I dedicate this thesis to people that make my life most meaningful,
my daughters Saskia and Felicia and my lifetime partner Anders.

I also want to dedicate my efforts to the memory of my sister, Eva Maria, for whom I
will try to make the most of my time here on this wonderful planet.

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ABSTRACT:

One of the solutions that has been advocated to reduce anthropogenic impacts in the marine environment, is the concept of a global representative network of Marine Protected Areas (MPAs). The concept seeks to address both conservation and natural resource (eg. fisheries) management, and in Australia, the introduction of MPAs is guided by comprehensive, adequate and representative (CAR) principles. At a local scale however, we face the challenge of identifying areas that collectively fulfil the goals of MPAs when we don't have enough ecological information. This is due in part to limited knowledge, but is also a result of poor translation of knowledge into the applied realm of management. Consequently, effective MPAs are difficult to plan and balance against diverse stakeholder needs in a political climate. Therefore, it is recognised that ecological science must play an integral part in the development of effective MPAs. In this study, I used MPA zoning in NSW, Australia, as a large scale experiment, to test the effect of no-take zone protection in tidal flat habitat, and also to assess whether the conservation management decisions achieved the MPA goal of representativeness for tidal flat habitat. As bait-harvesting practices for the crustacean *Trypaea australiensis* (callinassidae) would cease following the zoning of a marine protected area, I sought to address four key aims. First, to describe relevant spatial patterns of tidal flat biodiversity, to assess the representativeness of the tidal flat in a bioregional context, to determine what the impact of bait harvesting was as a structuring process, and finally to determine the effectiveness of no-take zoning for tidal flat habitat and the potential for recovery of the assemblage.

I used a hierarchy of spatially nested scales to sample and test the spatial patterns of tidal flat faunal assemblages, and to determine if the macrofaunal diversity of the protected tidal flat was representative of other tidal flats in the bioregion.

This study documents the first recovery trajectory for tidal flat assemblages in a marine protected area. There was an increase in abundance and homogeneity of smaller, less mobile, suspension and deposit feeding species. In contrast, some of the more mobile, predatory and or scavenging species decreased in abundance. This assemblage shift has the potential for further trophic or functional effects beyond the boundaries of the tidal flat, which are discussed, and thus provides important guidance for future research.

I also found that macrofaunal assemblages were patchily distributed, being most heterogeneous at the scale of 100s m within the tidal flat. For planning and management this implies that whole habitat is required in no-take zones, in order to encompass the full range of macrofaunal diversity in the habitat.

In addition to the key findings, the methods used in this study extended asymmetrical ANOVA to incorporate temporally and spatially asymmetrical factors simultaneously. This extension increases the power and thereby the sensitivity of univariate analysis, to detect environmental change for MPA or impact studies. In addition, data manipulations (taxonomic resolution, assemblage sub-sets and data transformations) demonstrated some dramatic effects on the interpretation of biological pattern.

This study demonstrate the opportunity of using MPA management decisions as a basis on which to test ecological predictions, as well as provide outcomes that can be applied to adaptive planning and management for MPA goals.

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CHAPTER 1 - GENERAL

INTRODUCTION



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Photo: Frances Clements

1.1. Challenges to conservation of tidal flat assemblages

1.1.1. Why Conservation?

It is widely accepted that estuarine systems as a whole are important for ecosystem services, i.e. ecological functions that benefit humans (Limburg and Folke 1999), including biological food cycles, water quality and nutrient recycling. These services are not only relevant at the scale of an estuary, but extend well beyond the boundaries of the estuary itself which is a dynamic link between the catchment and the sea for many ecological processes (Levin et al. 2001). Of relevance here is that the range of ecosystem services provided in estuaries can be attributed and partitioned among the multiple habitats contained within estuaries.

Research on estuarine habitats however is skewed towards the more obvious habitat types. For example, the ecological services provided in seagrass habitat, where a mass of primary production is visible and epifaunal productivity is high, are broadly recognized (Bell et al. 1987, Edgar et al. 1994). Tidal flat habitat receives considerably less attention than many other types of estuarine habitats (eg. seagrass and mangroves), despite unique contributions to estuarine ecosystem services.

Estuarine tidal flats are a subset of soft sediment habitat, and recognized widely as a habitat with important ecosystem services (Snelgrove 1997). Like every other marine habitat close to the coast, the pressures from coastal development and natural resource use are immense and will only increase with a burgeoning population (see Gray (1997) for a review). Changes to ecosystem services have measurable effects on society and implications for long-term sustainability.

Tidal flat infaunal production has been shown to be similar to that of seagrass (Edgar et al. 1994, Robertson 1984); an important consideration with regard to recreationally and commercially valuable fish species for which infauna is the main food source. The extent of tidal flat habitat in NSW estuaries is strongly linked to populations of species such as the Eastern school prawn, Eastern blue-spotted flathead and mud crab (Saintilan 2004). In addition, tidal flats have a functional role in this habitat as a nursery for both fish (Franco et al. 2006, Morrison et al. 2002, van der Veer et al. 2001) and juvenile invertebrate species (Dittmann 2000), and tidal flats and their sediment attributes contribute unique functions to nutrient dynamics in estuaries (Bird et al. 2000, Lillebo et al. 2004).

The conservation of tidal flats and associated ecosystem services becomes important when there are threats to the habitat. Similar to many coastal habitats, tidal flats face impacts from coastal development (Lindegarth and Hoskin 2001), nutrient enrichment, chemical pollution (eg. insecticides), physical disturbance and harvesting (eg. as baitpumping); even marine pest invasions have occurred in tidal flats elsewhere (Neira et al. 2006).

1.1.2. The challenges

Marine Protected Areas are being introduced world-wide with the objective of maintaining biodiversity and ecological processes. Knowledge of the distribution of biodiversity is one of the key factors, alongside connectivity and ecological processes, that limits the efficiency and effectiveness of MPA planning and management.

Efficiency of spatial zoning of MPAs is a requirement where planners and governments regard minimising impacts on human activities as important, while achieving the

desired objectives of MPAs (see section 1.3.1. below). Therefore, complex spatial networks of conservation measures require ecological information, some of which needs to be species specific (eg. fisheries resources management), and some of which need to focus on the bigger scale of habitats, ecosystems and the interacting ecological functions between them.

Despite the relatively easy access and proximity to coastal towns, the first comprehensive surveys to describe the biological diversity in estuarine soft sediments in Australia were undertaken in 1958 -1974 (Dittmann 2007, Hutchings 1999) and significant efforts have been made since then (Hutchings 1999). These included intertidal sediments, but focused more heavily on the sub-tidal fauna of protected bays. The taxonomy of tidal flat invertebrates in New South Wales (NSW) is broadly well described to the level of family (Hutchings 1999), and further notable efforts have described species from particular localities (Hutchings and Murray 1984). This information is not a result of efforts particular to research of NSW tidal flats, but rather that many tidal flat fauna can be considered as a subset of sub tidal marine and estuarine fauna identified in other studies, both descriptive and experimental. In addition, there are species in tidal flats uniquely adapted to the fluctuating environmental conditions of exposure and submersion and that have been studied experimentally (eg. soldier crabs (*Mictyris* sp.) (Webb and Eyre 2004). Consequently, the taxonomic and analytical tools exist with which to undertake further research in tidal flats.

Additional studies specific to tidal flat assemblages include description of spatial patterns in Tasmania (Edgar et al. 1999), Queensland (Dittmann 2000) and Western Australia (technically not estuarine) (Honkoop et al. 2006), and comparable studies

have been undertaken in the shallow sub-tidal soft-sediments in Victoria and limited areas in NSW (Anderson et al. 1981, Hirst and Kilpatrick 2007, Jones et al. 1986). Of note is that many of the spatial patterns vary temporally as well, which has been studied again in Tasmania (Edgar and Barrett 2002), but was found to be less important than spatial variability. Some trophic interactions have been investigated in Victoria (Robertson 1984) and comparisons of assemblages with seagrass habitat (Edgar et al. 1994, Howitt 1992) and between estuarine types (not tidal) (Hastie and Smith 2006) have been made. A number of studies in NSW have addressed specific impacts in tidal flats (Bishop and Chapman 2004, Skilleter et al. 2005) or re-colonisation trajectories (Filho et al. 2005, French et al. 2004). Specific efforts have been made in Victoria to link tidal flat fauna to the chemical processes measured in tidal flats (Bird et al. 2000). Although further published literature on sediment process studies in estuaries may provide additional relevant information on tidal flat ecology, research efforts with a specific focus on tidal flat assemblages in Australia is still in its infancy (Dittmann 2007). In addition, the accumulated knowledge is rarely in a format accessible to conservation managers (i.e. objectives are not always relevant to management, findings are rarely presented to inform management, the information is rarely collated at scales relevant to management).

1.2. Key themes of research on estuarine, tidal flat assemblages.

Tidal flats belong in the range of unconsolidated or soft-sediment habitats along sheltered coastlines, but are particularly common in estuarine embayments and are otherwise commonly referred to as mud flats or sand flats. The terms tidal, mud and flat partially reflect the unique attributes that separate tidal flats as a habitat from other

soft-sediment habitats (eg. beaches or sub-tidal sediments). Tidal flats are generally the most productive unconsolidated intertidal habitats and are considered more stable than their beach counterparts on exposed coastlines (Austen et al. 2002, Dittmann 2007).

Here the macrofauna (>1mm) of unvegetated tidal flat habitat in estuaries are considered as a unique assemblage type compared to other soft-sediment habitats. The justification for this is firstly practical, as the boundaries of tidal flats are currently being used to delineate habitat types for marine conservation initiatives (JBMP 2002, Ward et al. 1998). Second, the physical tidal dynamics of exposure and submersion imply that organisms adapted to these physical extremes would differ to sub-tidal assemblages (Dittmann 2000); this is analogous to inter-tidal rocky shore habitat where biological assemblage gradients are visible.

There are clear themes that have emerged in studies of tidal flat and shallow water soft-sediment fauna (Constable 1999, Dittmann 2007, Thrush and Warwick 1997). These include themes that are of relevance for effective conservation management: spatial patterns, ecological processes and human impacts. I examine these in the following section.

1.2.1. Spatial patterns of Biodiversity

Soft sediment fauna is found to be variable at many spatial scales, and tidal flat fauna in particular is variable at the scale of meters due to local small scale structuring processes such as tidal height and sediment grain size (Dittmann 2000, Edgar and Barrett 2002, Ellingsen et al. 2005, Ysebaert et al. 2003). There are also indications of intermediate scale assemblage differences due to salinity gradients within estuaries (Dittmann 2000,

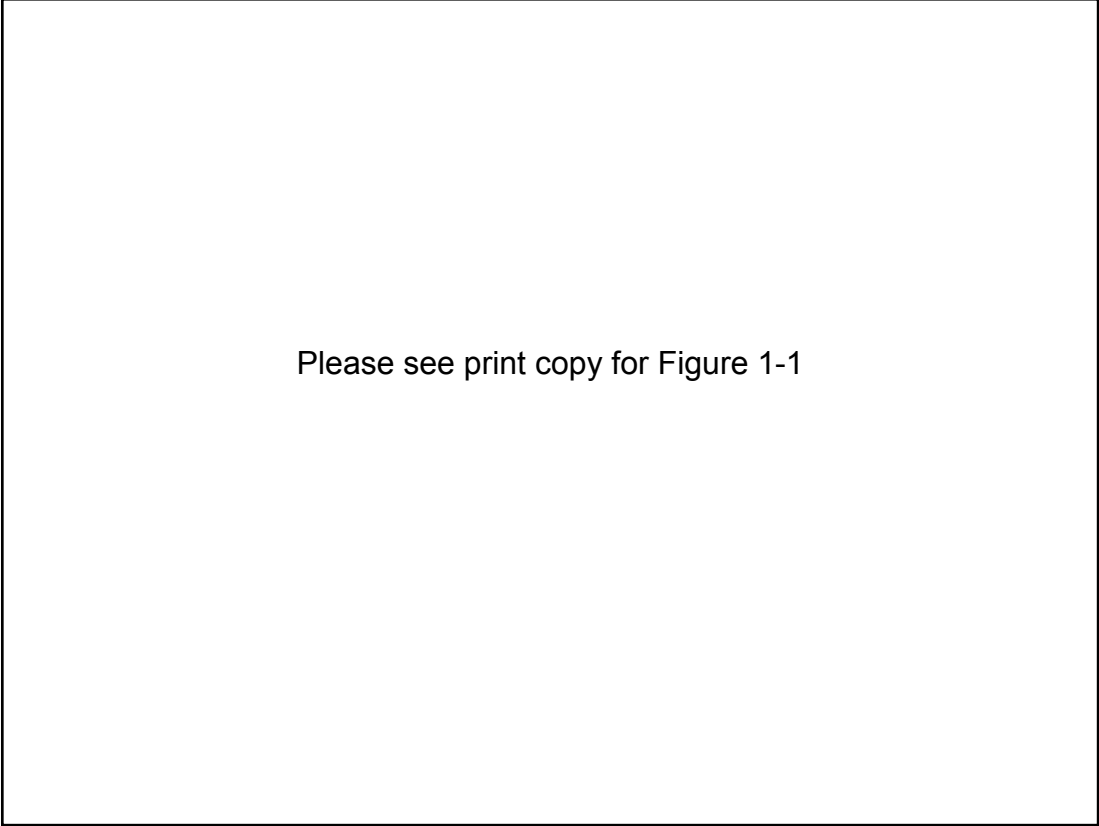
Hastie and Smith 2006, Hirst 2004, O'Hara and Poore 2000, Ysebaert et al. 2003). At larger regional scales there are clear differences between estuarine types (eg. drowned river valleys, barrier estuaries and intermittently closed and open estuaries) (Edgar and Barrett 2002, Hastie and Smith 2006, Hirst 2004), and at larger scales assemblages change with latitudinal and longitudinal gradients (Dittmann 2000, Dittmann 2007, Hirst 2004, Ricciardi and Bourget 1999). These scales of diversity are usually referred to as alpha (within habitat), beta (between habitat) and gamma (regional) diversity, (Gray 1997).

Tidal flat assemblages have historically rarely been regarded as a unit in their own right for the description of spatial and biogeographical patterns. One exception is a significant biogeographical study of estuarine tidal flat invertebrates around the whole island state of Tasmania (Edgar et al. 1999) that indicated clear differences between estuarine types at bioregional scales. Some studies use sub tidal, albeit shallow, assemblages that might reflect spatial patterns of tidal flat habitat to a degree (Anderson et al. 1981, Hirst 2004, Jones et al. 1986), but analogous to intertidal rocky shore habitat, there are clear gradients that distinguish tidal flat fauna assemblages from sub tidal counterparts (Dittmann 2000, Dittmann 2007, Edgar et al. 1999).

1.2.2. Structuring Processes

The spatial pattern described above may be caused by a range of processes including physical, chemical and biological. Numerous studies have measured specific processes in isolation while others have investigated interactions of different processes. The complexity of these interactions have been summarised by Snelgrove (1997) for ocean sediments. These are extended here with processes of particular relevance to tidal flats

(Fig. 1-1) using reference to selected literature that is summarised below. This review of structuring processes serves to illustrate the complexity of key processes in tidal flats. In reviewing the interactions and complexity of these processes, it becomes evident that the effects of human impacts as an additional structuring process will be hard to predict.



Please see print copy for Figure 1-1

Figure 1-1. Schematic diagram illustrating a range of structuring relationships between organisms and environmental variables that occur in tidal flat habitat. This is not the limit of processes and relationships which extend beyond the boundaries of tidal flat, and linkages may be mono or bi-directional in effects. The diagram is an extension from Snelgrove (1997) .

1.2.2.1. Environmental structuring processes

Similar to tidal rocky shores, tidal flats and other soft sediment habitats show clear faunal gradients with tidal heights (Dittmann 2000, Edgar and Barrett 2002, Pinedo et al. 2000). This may be due to exposure time, or wave action associated with tides can affect the stability of sediments and the grain size frequency distribution, which in turn can affect assemblages (Bishop and Chapman 2004, Pinedo et al. 2000). For example, the degree of physical disturbance and resuspension of sediments may influence recruitment of early life stages as has been shown to occur for the recruitment of some bivalves (Bouma et al. 2001), and food resources can be affected by different water velocities (Lucas et al. 2000). Sediment grain-size effects however, are not always clear (Honkoop et al. 2006, Jones et al. 1986), as other environmental factors (see below) may have a stronger structuring influence.

Fresh water flows may influence marine fauna at the mouth of estuaries, however the proximity to terrestrial habitat implies that estuarine fauna are exposed to even more severe salinity gradients and these have been found to be an important environmental variable for sediment fauna (Hirst and Kilpatrick 2007). Of interest is that salinity gradients should not just be regarded as a function of upstream versus downstream effects. Sub tidal soft-sediment assemblages have been shown to be affected by groundwater seepage (Zipperle and Reise 2005); an effect that could possibly be prevalent in a number of estuarine tidal flats, and specific species have been shown to be sensitive to fresh water flood events (Matthews 2006).

Organic content of sediment can be considered a combination of environmental and biological processes if delivery of the organic material is from physical processes

(Rossi and Underwood 2002), as well as a biological process from *in-situ* production.

Nevertheless, assemblage composition and abundance is often linked to organic content (Austen and Widdicombe 2006, Hirst and Kilpatrick 2007, Kelaher and Levinton 2003, Lenihan et al. 2003).

Similarly, the recruitment and redistribution of species may be considered an environmental process when linked to dispersal via resuspension of sediments, or as biological when fauna are shown to be highly mobile (Filho et al. 2005, Junkins et al. 2006).

1.2.2.2. Biological Structuring Processes

As can be anticipated from research in other habitats, there are biological interactions between many trophic guilds and taxonomic groups in tidal flats. These include intra-specific interactions, for example, there is often clear spatial partitioning between different cohorts of a species in tidal flats, such has been shown for bivalve juvenile/adult interactions where the density of adults has been found to affect patterns of juvenile recruitment (Rossi et al. 2007, Thrush et al. 1997b). Similarly, many species exhibit multiple settlement sites in their life history (Villbrandt et al. 1999).

Distinct taxa that are regarded as functionally similar may co-exist through inter-specific interactions such as resource partitioning, eg. food particle size or feeding behaviour (Caron et al. 2004), but other inter-specific interactions are more physical, closely tied to the habitat, and relate to habitat engineering by burrowing fauna (Bird et al. 2000, McKillup and McKillup 1997, Sushanek 1983). In particular the relatively large callianassid crustaceans provide large burrows with habitat conditions beneficial

to some species (eg. Lucinid bivalves (Felder 2001) or nermerteans (Koller et al. 2006), but detrimental to others (eg. oysters (Feldman et al. 2000)). Similar effects have been shown by the reworking of sediments by other species (Kelaher et al. 2003, Volkenborn and Reise 2006, Webb and Eyre 2004).

The co-existence of trophically similar species can reflect the partitioning of resources through functional differences. For example phytoplankton feeding guilds across tidal flat are partitioned into deposit feeders consuming benthic algae and suspension feeders consuming primarily planktonic algae (Herman et al. 2000). In addition, a range of studies have also compared the invertebrate assemblages of vegetated and unvegetated habitat in tidal flats (Edgar et al. 1994, Siebert and Branch 2007).

Predators in tidal flats, such as predatory snails (Kelaher et al. 2003) fish (Robertson 1984) or birds (Thrush 1999), can be regarded as a spatially homogenizing force on tidal flat assemblages by the targeting of high density patches that are subsequently thinned out by mortality. Vertical behavioural responses, stimulated by chemical cues, that might reduce this effect of predation on tidal flats have also been documented (Griffiths and Richardson 2006).

Interactions between the biological and environmental structuring processes introduce complex feedback loops and contrasting effects (Herman et al. 2001). Thus, in summary, tidal flat habitats are complex and generic predictive models are in their infancy (Constable 1999).

1.2.3. Human impacts

Human impacts have been reported in tidal flats and can take a variety of forms with diverse effects on tidal flat assemblages (Reise 1982). These impacts can be considered as structuring processes that interact with the diverse processes described above, and with each other (Austen and Widdicombe 2006), for example physical disturbance and nutrient pollution. Measured impacts from human activities have included catchment development (Edgar and Barrett 2000, Lindegarth and Hoskin 2001), bait harvesting, (Skilleter et al. 2006, Wynberg and Branch 1994), trampling (Casu et al. 2006), boat wash wave effects (Bishop and Chapman 2004), acid sulphate soils and drainage (Lin et al. 2004), dumping of dredged material (Bolam et al. 2004, French et al. 2004), pollution (Fleeger et al. 2006, Roach et al. 2001, Stark 1998) and even introduced species (Yamada et al. 2005).

Many of these impacts have been measured in experimental studies or in BACI designs from before to after an impact occurs, and some studies have documented trajectories of colonisation and succession (Thrush et al. 1996). Rarely, however, are recovery trajectories measured to determine the responses once different impacts have been removed through the implementation of Marine Protected Areas.

1.3. Marine Protected Areas – global and local perspectives

Conservation has been a key tool in the management of natural resources in response to mismanagement of natural resources and declines in ecological services that impact on society (Rodríguez et al. 2006). It has been rigorously applied and accepted in terrestrial habitats for centuries, using different formats ranging from close to total human exclusion (eg. Wilderness Areas), to open access with minimal disturbance (eg. National Parks), to less stringent conservation measures that aim to minimize negative impacts through applied management of natural resources (eg. riparian buffer zones in agricultural landscapes). A diversity of strategies and levels of protection provides a broadly accepted network that may maintain ecological systems on land, and more recently, a range of conservation strategies have been developed for marine and coastal ecosystems.

The application of conservation strategies from land has provided lessons for the more recent application of conservation strategies to the marine environment (Neigel 2003). In addition, good land conservation management has conservation benefits for the marine environment which is a sink for many of the impacts on land. However there are also unique challenges for conservation of marine ecosystems (Carr et al. 2003). Most obviously, air and water are very different physical environments that influence life histories and modes of dispersal, and therefore the connectivity of marine systems (Largier 2003). In addition, threats or impacts can be unique to the marine environment such as the effects of nutrient or other waste pollution which is easily dispersed, or the scale of and disturbance from the commercial and recreational harvest of wild animals.

The relative inaccessibility of many marine ecosystems also provides a challenge to understanding the impacts and monitoring conservation efforts.

Marine conservation has understandably focused on the conservation of targeted fish stocks as a form of fisheries management (Gerber et al. 2003); partly because the economic ramifications of human impacts on this resource have been measured and felt by diverse fishing industries. However, our understanding of the importance of whole marine ecological systems to provide vital ecosystem services, including fish stocks, is now a key objective of marine protected areas (Alcamo et al. 2005, Hastings and Botsford 2003, Roberts et al. 2003, Worm et al. 2006).

The broadly accepted terminology for marine conservation of whole habitats is Marine Protected Areas (ANZECC 1998), which encompasses a range of conservation formats from full no-take protection, spatial mosaics of conservation and multiple use zones, and even temporal protection in certain areas. This reflects the different types of physical dynamics, ecological connectivity and habitat boundaries in marine ecosystems, when compared to terrestrial ecosystems, and also takes into consideration the spatial patterns of resource use by humans, both commercial and recreational. In addition, they are seen to complement other marine resource management initiatives such as fisheries management (eg. quotas, bag and size limits).

1.3.1. Objectives of Marine Protected Areas

There are clear objectives adopted for the conservation of ecological systems that define what aspects and scales of ecosystems need to be considered in conservation management. These are referred to as the CAR principles and should be applied at the

scale of bioregions which are nationally defined land and seascapes (ANZECC 1998).

CAR principles are defined as:

- 1) **Comprehensiveness:** To include the full range of ecosystems recognised at an appropriate scale within and across each bioregion.
- 2) **Adequacy:** The required level of reservation to ensure the ecological viability and integrity (or resilience (author interpretation)) of populations, species and communities.
- 3) **Representativeness:** Those marine areas that are selected for inclusion in MPAs should reasonably reflect the biotic diversity of the marine ecosystem from which they derive.

1.3.2. Challenges specific to the establishment and maintenance of MPAs

There are three main challenges facing the efficient and effective development of representative networks of MPAs. First is the complexity of management of combined social and environmental needs which can often be conflicting (Baelde 2005). Second is the limited knowledge of ecosystems and associated processes which often confounds the first issue, and third is the lack of translation of ecological knowledge into formats that can be applied to the planning and management of MPAs, again confounding the first issue.

Marine conservation, like any environmental conservation, is more about managing human activity and balancing it against the ecological impacts to maintain a resilient and functional ecosystem. However the influence of short-term socio-economic issues

on effective marine protected area designs could potentially compromise long-term ecological benefits (Rodríguez et al. 2006). This was highlighted at the first International Marine Protected Areas Congress (IMPAC1) held in Australia in 2005 (Day et al. 2005), where 55% of the papers presented focused on the management systems and logistics behind developing networks of MPAs, and the remaining 45% focused on the ecology or biology behind MPAs under the themes of “Ecosystem Processes” and “Sustainability and Resilience”, despite the fundamental importance of ecological knowledge in informing MPA design. Other authors have also suggested that the dominance of socio-economic management in informing marine protected area design will not fulfil the conservation and maintenance of ecosystem services objectives of marine reserves (Lynch 2006, Roberts et al. 2003b), however this will inevitably remain so in the absence of rigorous ecological information from which to make informed decisions.

Acknowledging that there is still limited knowledge of the spatial distribution of marine life and the processes and connectivity of ecological systems, conservation must be based on coarser information than biological diversity alone. To facilitate the management of marine conservation, planners and managers work with what information they have; thus what is visually perceived as homogeneous habitat is often employed to describe and represent biological diversity (eg. soft sediment, sub-tidal rocky reef or seagrass beds). These categories of habitat type are often referred to as surrogates for biological diversity (Banks and Skilleter 2002, Ward 2000).

Considering the diversity and complexity of marine habitats, the biological surrogate of habitat type is a valuable tool; however there is a strong need to define or develop

knowledge that can describe ecological variation at finer scales within habitats (Stevens 2002). It is well documented that there is strong ecological variation across many spatial scales within habitat type, however this within habitat variability in the marine environment is poorly described as a general rule, or the translation of knowledge to applied management is lacking. For example, deeper habitats for obvious reasons of practicality, but also more shallow and accessible habitats remain poorly described in terms of spatial and temporal patterns of biological assemblages, particularly soft sediments (Stevens and Connolly 2004). One way to refine the classification of soft-sediment habitat is to stratify the habitat according to depth, which in most cases has a strong influence on biological assemblage composition (Dittmann 2007, Snelgrove 1997). However ecological variation is due to a number of complex physical and biological processes and depth alone can not account for habitat variability.

In addition to limited knowledge of marine ecological systems, the effect is compounded by a lack of communication from science to management (Castilla 2000). This could be an artefact of the nature of ecological science which, although often potentially of value to management of MPAs, tends not to focus on designing experiments with outcomes directly relevant for application in management. If peer-reviewed scientific publications are of relevance to management, they not always accessible, reviewed or interpreted to develop concise principles for adoption by management. Management of MPAs requires descriptive ecological information (eg. typical assemblages), predictive and simplified models of ecosystem processes, and measured impacts or recovery trajectories to inform and adapt management choices. This information will also direct the priorities for future research and monitoring

projects (NSWMPA 2005).

1.3.3. Measured outcomes from MPAs

The planning and management that is needed to create a network of marine protected areas and the limited information with which to make well informed decisions, has prompted a burst of activity in the field of marine protected area research. This is really an extension to marine biological and ecological research with defined goals to better inform management, and to measure the outcome of marine protected areas (Castilla 2000).

This field has now existed long enough for some reviews of the outcomes of MPAs to emerge in the peer-reviewed literature (Castilla 1999, Claudet et al. 2008, Guidetti et al. 2008). These findings generally demonstrate the value of MPAs to the recovery of fish stocks when commercial or recreational fishing has been excluded from a marine protected area. There are also emerging paradigms that are being tested in relation to effects of trophic cascades (Shears and Babcock 2002). However, understanding the indirect effects of marine protected areas on non-target species, whole assemblages and ecological processes, is still in its infancy (see chapter 4). In particular, effects of MPA zoning on soft sediment assemblages, which are often largely ignored in zoning plans or considered as homogeneous expanses of habitat, are generally undescribed.

1.3.4. Marine Protected Areas in NSW, Australia

In New South Wales, Australia, Marine Protected Areas are being developed within bioregions in the form of multipurpose Marine Parks (see www.mpa.nsw.gov.au).

Within these parks, a range of identifiable habitats is sought for no-take protection in Sanctuary Zones. In this way the habitat is a surrogate unit that aims to represent the ecological diversity of comparable habitat that is not included in no-take sanctuary zones. Using this methodology, it is assumed that the area of habitat is homogenous in ecological diversity at the scale of protection.

At the commencement of this study, the Jervis Bay Marine Park on the south coast of New South Wales, was being introduced and undergoing planning for zoning of fully no-take sanctuary zones, as well as other zones with different (lower) levels of protection. Tidal flats were identified as one of a range of habitats in the New South Wales Jervis Bay Marine Park, and as such, one was zoned for no-take sanctuary zone protection to commence at the end of 2002. This implied that bait-harvesting for the callianassid *Trypaea australiensis* (crustacea), otherwise regarded as a daily occurrence in the identified location, would cease.

1.4. Aims of the study

No-take zoning of a tidal flat in the Jervis Bay Marine Park was introduced using the best available knowledge of the habitat type, but with limited information it was also zoned based on a number of assumptions. These were that the biodiversity of that location and the scale of the no-take zone was representative of other tidal flat habitat in the bioregion; that bait harvesting for *T. australiensis* had a measurable impact on tidal flat assemblages, and that no-take sanctuary zoning was an appropriate and effective form of conservation management for this habitat.

To test these assumptions, the main goals of the present study were to:

- 1) Describe relevant spatial patterns of tidal flat biodiversity,
- 2) assess the representativeness of the tidal flat in a bioregional context,
- 3) determine what the impact of bait harvesting was as a structuring process and
- 4) determine the effectiveness of no-take zoning for tidal flat habitat and the potential for recovery

The first two aims are addressed in Chapter 3, where patterns of biological diversity were described for three spatial scales in three permanently open estuarine tidal flats. This provides information on the spatial scales of tidal flat habitat diversity so that they can be considered for future zoning reviews or new MPAs, and assesses the representativeness of the no-take tidal flat to other tidal flats in the bioregion.

A Beyond BACI design was used to address the third aim, where a recovery trajectory of tidal flat assemblages was monitored from one year before no-take zoning, to the two years following zoning. In this way the difference between the assemblages before and after no-take zoning provides some indication as to what kind of impact bait-harvesting was having. In comparing such a trajectory to two control tidal flats outside the Marine Park, any changes to the sediment assemblage at the no-take site might be linked to a causal effect of the no-take sanctuary zone, thus addressing aim 4.

Further to the above four aims, the rigor and efficiency of the methodology used is addressed in Chapter 5. Controversy has clouded the consensus of the benefits of MPAs in relation to poor experimental design and statistical analysis in monitoring programs (Guidetti 2002); often providing ammunition to opponents of MPAs. It is therefore of utmost importance that experimental design and statistical analysis is rigorous enough to inform and defend the planning and management decisions of MPAs, and also sensitive enough to measure actual recovery trajectories. In addition, describing and monitoring the biological diversity of habitats in the long term requires that methods are as efficient as possible with the limited time and funding that is available to the task.

Invertebrate assemblages are often regarded as appropriate indicators or measures of ecological patterns, and/or that they will respond with a measurable change to an ecological impact. Invertebrate assemblages that are data rich sources, with short life cycles and measurable responses to impacts, have been shown to be valuable in this regard (Giangrande et al. 2005, Jones and Kaly 1996). However, the choice of methods for subsequent analysis and interpretation of the assemblage data has a strong effect on the outcome of the research program.

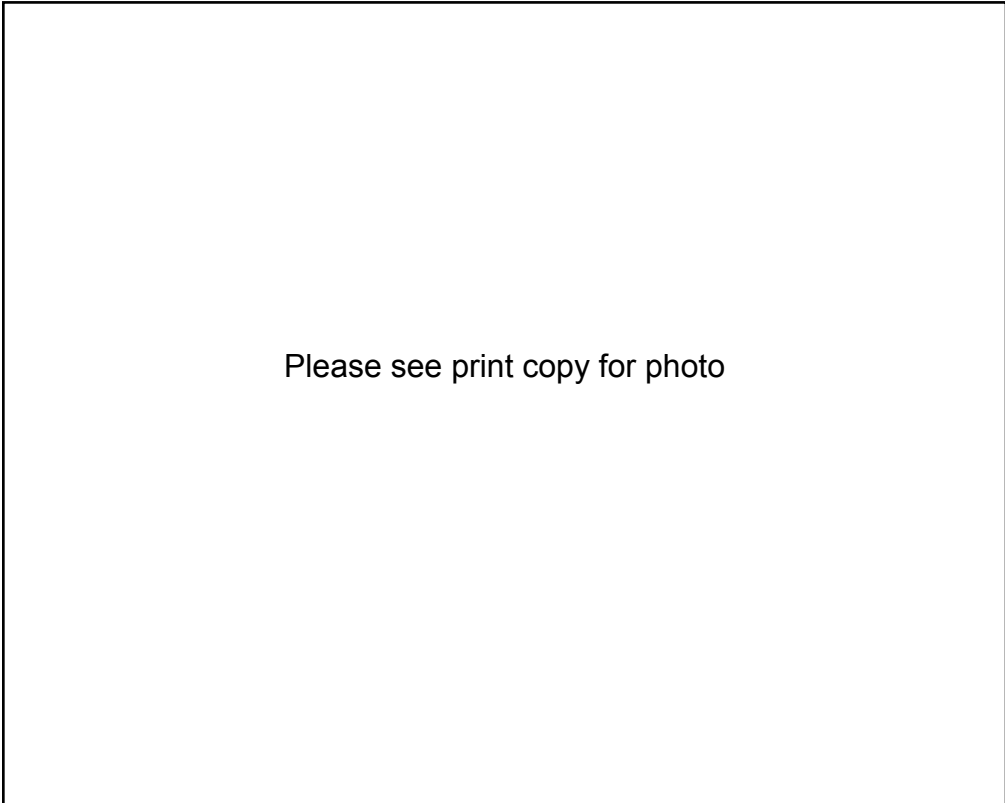
The choice of assemblage sub-group and/or taxonomic resolution may affect the outcome of a study, and these choices should be made with the objective of the study in mind (Olsgard and Somerfield 2000, Roberts and Connell 2008, Warwick 1993).

Biogeographical patterns across large scales or strong environmental gradients can become confused and noisy if detailed species level information for the full suite of organisms in an assemblage is included. In contrast, patterns of biological diversity at local and bioregional scales or an impact may become undetectable if selected assemblage sub-groups or a lower taxonomic resolution are used.

Such choices must also be considered in light of the available time and resources for research and monitoring programs. The costs of monitoring for an environmental impact might be greatly reduced if only a few taxa need identification, or if a lower level of taxonomic resolution is sufficient.

Finally, this study demonstrates how ecological science and conservation management can be married to suit the objectives of both. Indeed conservation management should be based on accurate knowledge of environmental systems, and ecological research is often funded with the justification of relevant outcomes to conservation; however the translation of ecological research into management is rarely made easy.

CHAPTER 2 - GENERAL METHODS



Please see print copy for photo

Photo: Andy Davis

2.1. Description of Study Location

This study was undertaken on the south coast of NSW Australia in three permanently open, barrier estuaries within 30 km of each other (see Fig. 3-1, pg. 36). At a latitude of approximately 30° south (and 150° east), the region is considered to have a temperate climate with average temperatures in the range 12 – 22°C. The average rainfall tends to be higher in the summer months with an average throughout the year of 1345mm.

The tidal flats surveyed in this study were located within 1km upstream from the mouths of the permanently open estuaries, and were predominantly unvegetated with some intermittent shallow seagrass (*Zostera capricorni*). Otherwise the tidal flats were boarded by mangrove forest (*Avicennia marina*) and subtidal channels. Bioturbation was evident in the sediments as burrows, mounds or small disturbed patches, and included larger feeding pits or traces from predatory fish at high tide (Plates 1).



Plate 1. Bioturbation and traces in the tidal flat sediments at low tide (Photos 2-3: F. Clements). Traces in photos from left to right include an imprint from *Platycephalus* sp., naticidae trails and polychaete mounds; a stingray imprint (feeding pit); multiple fish feeding pits.

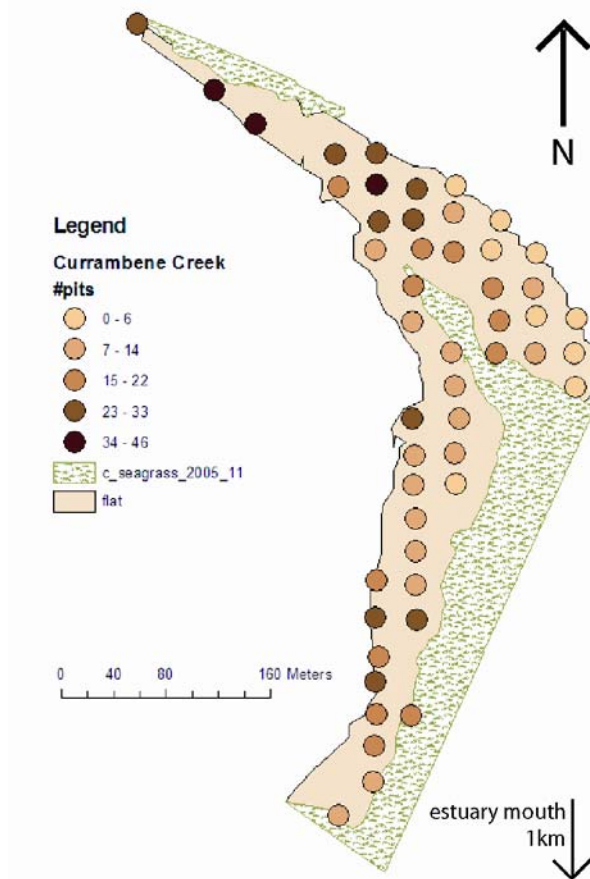


Figure 2-1. Map of the extent of unvegetated tidal flat in the Currambene Creek location, including an area of intertidal seagrass (*Zostera capricorni*). The circles indicate different densities of feeding craters.

The density of feeding pits varied within each tidal flat (Fig. 2-1), and together with other features such as sediment colour (well sorted vs. muddy sand), and tidal height (Fig. 2-2), an attempt was made to stratify the tidal flats for the experimental design by mapping such features. To do this, feeding pit density per 12m² and tidal contours were recorded across two of the tidal flats. Circular, 12m² “quadrats” were used to record feeding pit density (Fig. 2-1) as density was visibly variable at scales as low as 10m, while quadrats would not capture densities where craters were spaced up to 5m from each other. A 1.95m line was used as the radius of a circle to mark the sample area.

Differential GPS was used (for sub-meter accuracy) to spatially record the quadrats.

Tidal heights were recorded with a differential GPS by walking along the tidal edge at 1 hour intervals. Maps were then created for feeding pit density and tidal heights (Fig.2-2) using ArcGIS and used to assess if stratification was feasible. Gradients of feeding pits, and tides were evident across the three tidal flats, however it was determined that stratification was not practical as boundaries were not clearly distinguishable *in situ*.

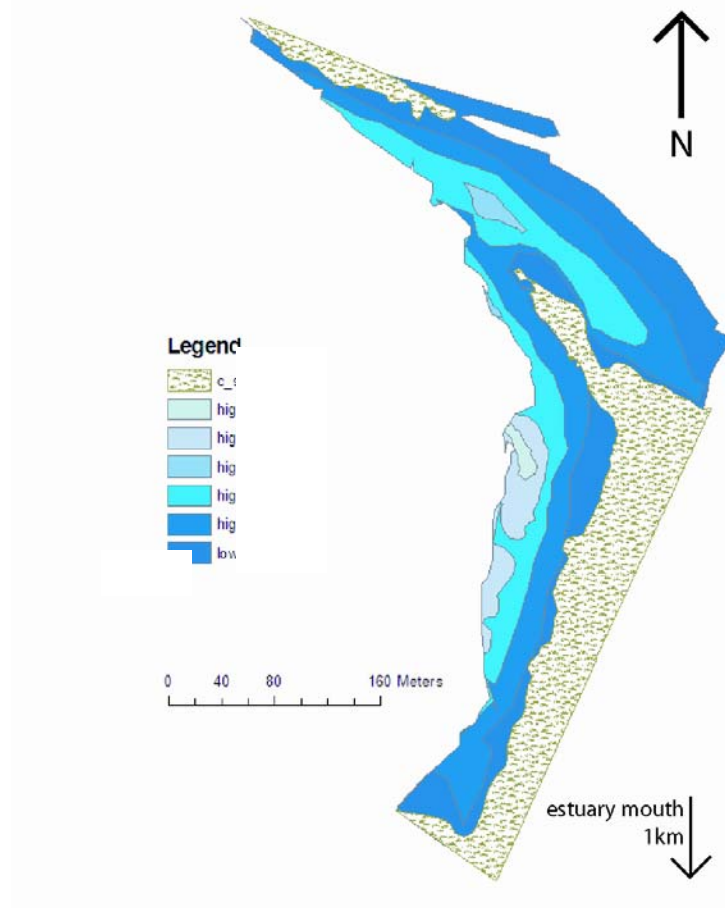


Figure 2-2. Map of the extent of unvegetated tidal flat in the Currumbene Creek location, including an area of intertidal seagrass (*Zostera capricorni*). The blue shades represent different tidal heights at six hourly (app.) intervals from high (light) to low (dark) tide.

2.2. Sampling Methods – pilot study

The methods used for each of the components of this study are provided in each Chapter. However a pilot study was undertaken to determine the most effective sampling method for the tidal flat assemblages, which is briefly described here.

2.2.1. Pilot study aims

Cores are traditionally used in soft sediment sampling of intertidal areas, however the limitations of this method in retrieving samples of the deep burrowing crustacean *Trypaea australiensis* prompted trials of other methods. Specifically, a bait harvesting suction pump was compared to the coring method, as it was able to sample deeper sediments quickly and with minimal effort.

There were two objectives for this pilot study. The first objective was to determine how many pump events were required to retrieve sediment samples of a similar weight and volume as a core. Consistent sediment volume is a pre-requisite for quantitative sampling, as variation in volume sampled would affect the number of animals retrieved. The second objective was to determine if the different sampling methods retrieved different quantities and types of taxa.

2.2.2. Pilot study methods

The sediment sampling equipment included a 150mm diameter core that was 250mm deep, and the pump had a diameter of 70mm and could suck up a similar volume of sediment as the core using 4 pump events, and at depths down to 500mm. Samples were taken with each of the 2 sampling methods at 10 haphazard plots across a tidal flat in

Narrawallee Inlet. Dry weight of the sediment was used as a proxy for volume of sediment sampled, as the pumping method sucked a lot of water from surrounding sediments and thus affected wet volume. To determine the consistency of sediment volume samples for each method, core and pump samples (4 pumps per sample at increasing depth to 500mm) were placed in separate buckets and returned to the lab to sun dry over a five day period. The dried sediments were then weighed to compare the consistency in “volume” of sediment sampled using the pump and core methods.

Additional samples were taken using the same method described above to test whether the different sampling methods sampled different quantities or taxa of sediment fauna. The sediment samples were sieved through a 1.2mm mesh bag *in situ*, and retained faunal samples were placed in smaller mesh bags with a diameter of 0.05mm and frozen (more humane than direct preservation) on return to the lab. The 1.2mm mesh was the most efficient sized mesh in allowing for sieving out the coarser sediments that dominated many sites, while still retaining many species. For faunal analysis, the samples in small, labelled, mesh bags were removed from the freezer and collected into a larger, drawstring mesh bag. The drawstring bag was then submerged in covered and securely sealed 80L drums containing a 5% formal (formalin plus seawater) solution with Biebrich scarlet stain. Biebrich scarlet has been demonstrated to be a suitable staining method to facilitate sorting fauna from the sediment samples according to methods outlined in Williams (1974); except Eosin B was not added to the solution as it is added when sorting terrestrial insects (Anna Murray personal comment – Australian Museum). It was important to dissolve the Biebrich scarlet powder with a small quantity of seawater prior to addition in the formal solution for optimal dissolution.

Samples were then rinsed and sorted by hand, and identifications were made with the aid of a light microscope. The number of total species and the number of *T. australiensis* per sample were recorded, as well as the size of *T. australiensis*.

2.2.3. Pilot study results

The results showed that between 3 and 4 bait pump events retained sediment quantities comparable to that using the coring method. The cumulative pumped samples quantities however, were not consistent and varied considerably (Fig. 2-3).

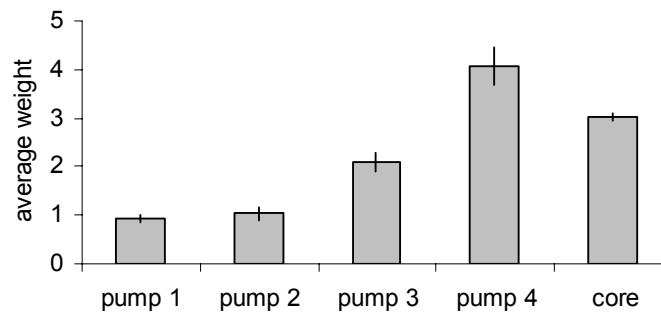


Figure 2-3. The average dry weight of sediment retrieved using 1-4 consecutive pumps with the bait suction pump, compared to sediment retrieved using a core 150mm diameter and 250mm deep. Standard error bars shown.

The average numbers of *T. australiensis* retained using the suction pump (4 pump events) was slightly higher than that for the core sampling method (Fig. 2-4(a)), however this was not significantly different (*t*-test, $p = 0.84$). The size of the *T. australiensis* retained was also larger for the suction pump method, but this was not significant ($p = 0.32$) (Fig. 2-4(b)). In contrast, the abundance of other invertebrate fauna that were retained was higher using the core than the suction pump, but again this was not significant ($p = 0.072$) (Fig. 2-5).

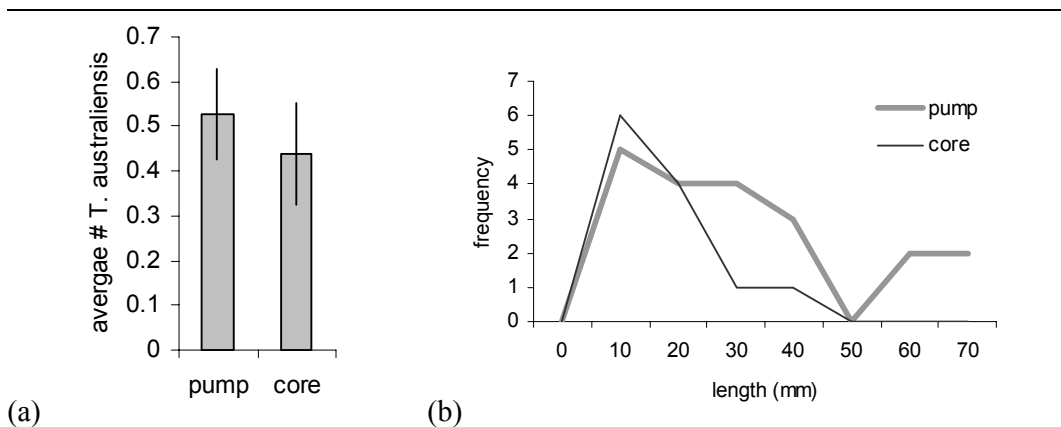


Figure 2-4. The average number of *T. australiensis* specimens and the size frequency distribution in each of the methods using 4 bait harvesting pumps or a core.

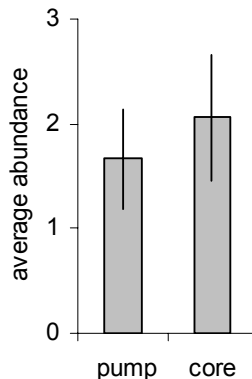


Figure 2-5. The average number of other faunal invertebrates retained using the 4 bait harvest pump versus the coring method.

2.2.3. Pilot study conclusion

The coring and suction pump methods were comparable in the number of species retained in samples, however there were indications that *T. australiensis* were better sampled using the suction pump, while other invertebrate fauna were better sampled using the coring method. In light of the findings that suction pumping provided variable quantities of sediment without measurable benefits to sampling, coring was the most appropriate method to quantitatively sample the macrofaunal assemblage.

CHAPTER 3

The importance of spatial scale for the conservation of tidal flat macrobenthos: An example from New South Wales, Australia.

**Winberg, P. C., T. P. Lynch, A. Murray, A. R. Jones, and A. R. Davis.
2007. Biological Conservation 134: 310-320.**



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3.1. Introduction

The design of Marine Protected Areas (MPAs) is an emerging science that utilizes experiences from the design of terrestrial conservation areas, but also requires new approaches that address the unique challenges of the marine environment (Pressey and Mc Neill 1996). Historically, MPAs have been designed in an *ad hoc* or opportunistic fashion (Avery 2003, Pressey 1994). This is due in part to limited knowledge of the patterns of biological distributions from the marine environment (Boersma and Parrish 1999, Stewart et al. 2003). Conservation efforts are also usually heavily compromised due to perceived socio-economic impacts by a range of stakeholders (Roberts and Hawkins 2000). Consequently, conservation efforts may be inefficient, inadequate or even fail to achieve set goals for reserve systems (Pressey 1995, Stewart et al. 2003). To avoid this, many MPAs are now mandated to be designed according to guidelines of comprehensiveness, adequateness and representativeness (CAR) (Burgman and Lindenmayer 1998, Lubchenco et al. 2003). CAR reserve systems must contain multiple representative samples of all ecosystems, habitats and associated taxa within bioregions (ANZECC 1998). Bioregions in Australian marine waters are areas that delineate ecological characteristics distinct from other areas in Australia, and are usually at scales of 100s - 1000s of km (ANZECC 1999).

A precursor to establishing a CAR system of MPAs is mapping the different habitat types within each bioregion (Jordan et al. 2005, Pressey and Ferrier 1995), which function as surrogates for ecological diversity (Stewart et al. 2003, Ward et al. 1999). During planning, these maps are used to choose habitat reserves. However, numerous critiques of MPA design state that an important limitation to this approach is a lack of

systematically surveyed biological data (Avery 2001, Banks and Skilleter 2002, Ponder et al. 2002, Stewart et al. 2003, Ward et al. 1999). In particular, there is limited information on the spatial scales at which faunal assemblages change in taxonomic composition within a habitat type. It is well established that taxa do turn over between assemblages along a distance or environmental gradient (Jablonski and Sepkoski Jr. 1996), which is best described by the term beta diversity (Clarke and Cranme 1997, Gray 2000, Pielou 1974). There are diverse applications for the concept of beta diversity (Gering and Crist 2002, Koleff et al. 2003, Ricotta et al. 2002), but here it is used according to early definitions (Pielou 1974) a review by Gray (2000), where beta diversity can describe taxonomic turnover in assemblages at various spatial scales within a habitat. Beta diversity is an important ecological measure for the siting of MPAs that aim to represent the full range of taxa in a region. Conservation planners need to consider beta diversity for two reasons; first, if beta diversity is low at the scale of bioregion, i.e. assemblages consistently contain the same taxa in all habitat locations, then any location has the potential to be a comprehensive and representative area for conservation. If, however, beta-diversity is high at the scale of bioregion, then multiple habitat locations that encompass this beta diversity will be required to achieve comprehensiveness and representativeness. Second, at smaller scales, interpreting habitat as an indicator of biological homogeneity can be misleading as many taxa exhibit patchy distributions within what appears to be homogeneous habitat (Stevens and Connolly 2004, Thrush 1991). This assumption is rarely tested for conservation purposes despite habitat heterogeneity of tidal flat and other habitat being documented in the scientific literature (Curley et al. 2002, Dittmann 2000, Rodríguez et al. 2006, Stevens and Connolly 2004, Thrush 1991). Heterogeneity may be an important defining

characteristic of many habitats (Holt et al. 2004, Tews et al. 2004), but it is rarely known what scales of habitat and associated heterogeneity are required to represent or adequately “maintain the status of” assemblages (Constable 1999). Consequently, if a portion of habitat is protected and scales of beta diversity are greater than the reserve area, there is a risk that the goal of representing species and ecological heterogeneity in the reserve will not be achieved. If habitats are to be mapped as surrogates for ecological diversity, then spatial scales of ecological heterogeneity or the processes that drive habitat heterogeneity need to be known to ensure that the full range of taxa are adequately represented in reserves.

Estuarine tidal flats are one habitat used as a planning unit and surrogate for taxonomic diversity in New South Wales MPAs (NSWMPA 2001). This habitat is an important transition zone, with ecological processes that functionally link terrestrial, freshwater and marine ecosystems (Gray 2002, Wall et al. 2001). For example, tidal flats are a principal energy link between primary production and consumers such as birds and fish (Levin et al. 2001, Loneragan and Bunn 1999, McLusky 1999). Therefore it is important to include tidal flats in a system of MPAs.

Despite considerable international research on the ecology of estuarine tidal flats (Constable 1999, Thrush and Warwick 1997), there have been relatively few Australian studies (Fairweather and Quinn 1995). Existing studies report small and intermediate scale spatial patterns of assemblages or taxa in tidal flats, (see reviews by Thrush (1991), Constable (1999) and Gray (2000)), but they are rarely presented in a context that can contribute to conservation planning. There are some notable exceptions, and in particular, Edgar et al. (2000) identified discrete biological assemblages associated with

up to nine different morphological types of estuaries (eg. barred estuaries, drowned river valleys etc.), as well as considerable biological variation at smaller scales within each estuarine type. However, taxonomic turnover (beta diversity) between estuaries of the same morphological type was not determined.

The aims of this research were to determine which of three spatial scales (20m, 100s of m and <30km) of ecological heterogeneity are most important for conservation planning of tidal flat habitat on the South Coast of NSW. We measured beta diversity as well as three other ecological measures: abundance, taxonomic richness and Shannon Wiener Diversity (H'), to determine if different measures showed the same patterns. The results should be relevant to spatial conservation planning in MPAs by answering practical questions: a) Does a single tidal flat provide a comprehensive and representative reflection of other tidal flats in the region? b) Do we need to include whole tidal flats in reserves to represent all infaunal macrobenthos? c) What further research is necessary in order to optimise the selection of tidal flats in MPAs?

3.1. Methods

3.1.2. Study sites

Three similar, permanently open, barred estuaries on the south coast of New South Wales (NSW), Australia, were chosen as study sites (Fig. 3-1) within a larger temporal study (Chapter 4). The first year of data were used for this study in order to describe spatial ecological heterogeneity. In each estuary, the first tidal flat upstream - within 1km of the estuarine mouth - was sampled. One of the tidal flats, Currambene Creek (35° 01' S, 150° 40' E), was proposed as a Sanctuary Zone (no take zone) in the

Jervis Bay Marine Park (JBMP). The two other estuaries were at Sussex Inlet ($35^{\circ} 10' S$, $150^{\circ} 35' E$) and Narrawallee Inlet ($35^{\circ} 18' S$, $150^{\circ} 28' E$). All of these estuaries are within 30km of each other in the Bateman's Bioregion.

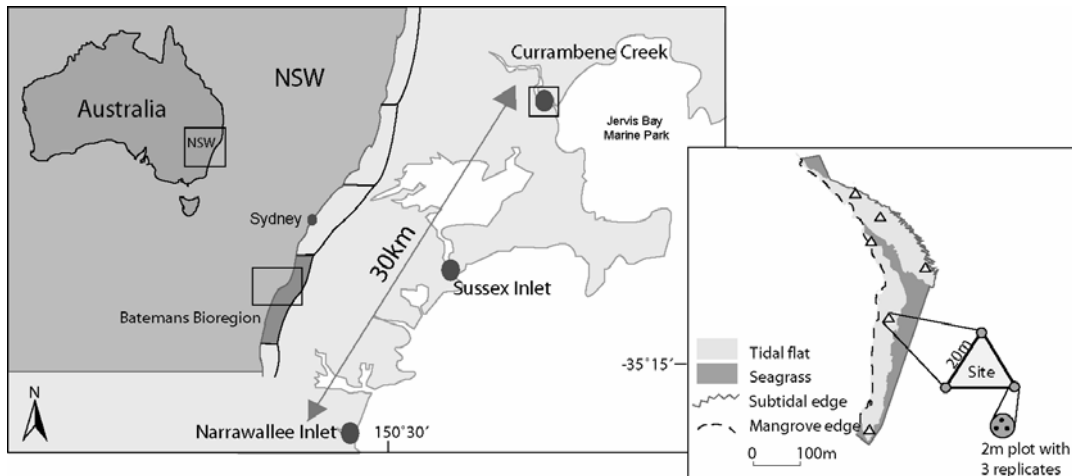


Figure 3-1 Study locations at three estuaries in the Bateman's Bioregion of NSW, Australia. The first tidal flat upstream from the estuary mouth was the location studied, and the general sampling design is illustrated for the Currambene Creek tidal flat. Three spatial scales were examined: plots (20m), sites (100s of m) and estuaries (<30km).

The sampled tidal flats in each estuary were directly adjacent to fully marine waters with similar salinity regimes and both the estuaries and tidal flats were morphologically similar. Each tidal flat consisted of extensive unvegetated, sandy habitat bordered sub-tidally by the seagrass *Zostera capricorni* or sub-tidal soft sediments, while the terrestrial border was fringed with mangroves and saltmarsh. The clay and silt content never exceeded 10%, although some sites appeared muddy (Winberg - unpublished data). The surfaces of the flats showed evidence of intense bioturbation, with burrow

openings of Callianassid species and various polychaetes, as well as pits, ranging in size from a few centimetres to >1m, made by benthic predators such as stingrays. At low tide the sediment was further disturbed by soldier crabs (*Mictyris* sp.) and amphipods, as well as by people pumping up bait from the sediment.

3.1.2. Sampling design

Each of the three estuaries was sampled on two occasions between July and October, 2002. On each sampling day three 20m triangular sites were haphazardly selected at least 100m from each other within each tidal flat (Fig. 3-1). The first site was chosen by the intersection of the tidal level at the time of sampling and a blind walk for 1 minute across the tidal flat. Subsequent sites were selected by the tidal level (1 hour later) and a 100m distance. At each site, three 2m² plots were randomly sampled by taking three sediment cores at the triangle apexes. Hence, 27 cores were taken from each tidal flat on two occasions giving a total of 54 cores per tidal flat, and 162 cores in total.

Cores were 15cm in diameter and 25cm deep. Sediment from the cores was sieved in situ through 1.2 mm mesh bags, and frozen on return to the lab. The large mesh size relative to other studies that involve macrofaunal sampling was due to a number of sites containing sandy coarse sediments and the large volume of sediment sampled in an attempt to sample deep burrowing fauna. Samples were later defrosted and stained with Biebrich Scarlet in a 5% formol solution for at least two days. Taxa were identified to the lowest taxonomic level possible, usually species, using a dissecting microscope. The species level was chosen as it is suggested that lower (coarser) taxonomic resolution should be used with caution (Vanderklift et al. 1998), especially as the suite of taxa

sampled had not been examined previously at these scales. Rarer, juvenile or difficult taxa were identified to the genus or family level.

3.1.2. Analysis of data

Data were visualized in species-sample accumulation curves (999 permutations) using the PRIMER 5 software package (Plymouth Marine Laboratories (Clarke 1993)) to determine if the majority of species present had been sampled. Beta diversity was determined using multivariate data of taxonomic composition and calculating Bray-Curtis similarities with the PRIMER 5 software package. As this paper intends to contribute to the representativeness of taxa in MPAs, multivariate analyses were done using presence/absence data to give equal weight to rarer taxa (Clarke 1993). Data were visualized in non-metric Multi-Dimensional Scaling (MDS) plots. We tested for assemblage differences at three spatial scales in a nested hierarchy on two occasions using the PERMANOVA software package (Anderson 2001, Anderson 2005, McArdle and Anderson 2001). As replication at the smallest scale of sampling (cores per plot) was 3 ($n=3$), we used Monte Carlo Probability values in the analysis (Anderson 2005). Pair-wise *a posteriori* comparisons were performed within the PERMANOVA software package using a multivariate version of the *t*-test (Anderson 2005). This method was used as a measure of beta diversity as it determines whether there are significant differences in assemblage composition at the scales sampled.

To test for differences in abundance, taxonomic richness and log base *e* Shannon-Wiener H' Diversity (diversity H'), a nested four factor ANOVA was used.

Assumptions of ANOVA were examined prior to proceeding with analyses. Normality was assessed visually in plots of means against variances (Quinn and Keough 2002).

Cochran's test was used to test for homogeneity of variance, and abundance data were transformed $((x+1)^{1/2})$ to achieve homogeneity (Quinn and Keough 2002). Where groups differed significantly ($p < 0.05$), *a posteriori* comparisons among means were done with Student-Newman-Keuls tests. All ANOVAs were performed with GMAV 5 software (University of Sydney, Sydney, Australia).

For all the dependent variables (ecological measures) above, the relative contributions of the different spatial scales to overall heterogeneity was determined by partitioning the variance components (ω) between the nested factors (scales) manually, according to methods in Graham and Edwards (2001). The term ω was used to calculate the variance component percentage contribution to overall variation as is consistent with Graham and Edwards (2001). For example, if assemblages differed significantly from each other at both the scales of estuaries and sites, ω indicates whether one of those scales exhibited greater heterogeneity than the other.

To determine if the relative contributions to variation and the proportion of variation between scales of sites and plots were consistent among estuaries, the PERMANOVA and ANOVA procedures were repeated using data from each estuary alone (therefore testing scales of sites and plots only). As the design was fully balanced, we used the Sums of Squares for each of the nested factors in the separate estuary analyses to determine the proportional contribution to overall variation at the respective scales in the full data set. Additionally, a measure of Dispersion and an Index of Multivariate Dispersion was obtained for multivariate data only, using the MVDISP procedure in the PRIMER 5 software package.

3.3. Results

A total of 4468 animals representing 54 taxa were found in cores taken from the three estuaries (Table 3-1). Cumulative samples from each estuary were representative as species accumulation curves approached an asymptote for each estuary (Fig. 3-2).

Subsequent data from the following two years also indicated that the majority of species had been sampled (see Chapter 4). 70-84% of all animals in each estuary were attributed to 4 common taxa: two bivalves *Mysella anomala* and *Eumarcia fumigata*, and two crustacean taxa, *Urohaustorius metungi* and *Mictyris* species. The factor “Time” was eliminated for all analyses and its’ variance pooled in all analyses as there was no significant temporal variation ($p>0.25$). Only spatial components are reported below.

Table 3-1. Taxa ranked by mean abundance per sample core in each location. Standard deviation (SD) and percentage contribution to overall abundance (%) are also shown.

The Phyla or Class to which taxa belong are identified in column Ph. With the following key: A = amphipoda, B = bivalvia, C = crustacea, F = fish, G = gastropoda, I = isopoda, N = nemertea, O = oligochaete. Total values for each location are presented at the bottom of the table.

Taxon	Ph	Currambene				Sussex				Narrawallee			
		rank	mean	SD	%	rank	mean	SD	%	rank	mean	SD	%
Galeommatidae (<i>Mysella anomala</i>)	B	1	24.54	26.47	43.8	2	2.81	5.41	20.1	1	4.76	10.25	31.9
Urohaustoriidae (<i>Urohaustorius metungi</i>)	A	2	13.43	13.75	26.2	1	9.65	7.43	51.7	2	3.02	4.58	18.9
Mictyridae (<i>Mictyris</i> sp.)	C	3	3.41	2.47	6.9	4	0.81	1.34	6.0	3	1.50	1.84	10.5
Veneridae (<i>Eumarcia fumigata</i>)	B	4	1.22	1.34	2.8	3	1.13	2.24	6.3	4	1.17	1.38	8.5
Sigalionidae (<i>Sigalion ovigerum</i>)	P	5	0.94	1.37	2.1	8	0.19	0.40	1.2	12	0.15	0.35	1.1
Callianassidae (<i>Biffarius arenosa</i>)	C	6	0.83	1.50	2.4	12	0.04	0.19	0.2	7	0.44	0.94	3.2
Spionidae (<i>Carazziella victoriensis</i>)	P	7	0.57	1.29	1.0	13	0.02	0.10	0.2	19	0.02	0.10	0.1
Magelonidae (<i>Magelona dakini</i>)	P	8	0.56	0.82	1.3	5	0.28	0.70	2.0	6	0.46	0.81	3.3
Nemertea	N	9	0.48	0.79	1.0	12	0.04	0.13	0.2	13	0.13	0.43	0.9
Capitellidae (<i>Barantolla lepte</i>)	P	9	0.48	1.13	1.3	11	0.06	0.21	0.5				
Orbiniidae (<i>Scoloplos simplex</i>)	P	9	0.46	0.72	0.9	9	0.17	0.37	1.3	5	0.93	1.00	7.0
Psammobiidae (<i>Soletellina alba</i>)	B	9	0.46	0.79	0.8	10	0.09	0.29	0.6	18	0.04	0.19	0.2
Callianassidae (<i>Trypaea australiensis</i>)	C	10	0.44	0.83	0.8	12	0.04	0.13	0.2	15	0.09	0.24	0.7
Paracalliopiidae (<i>Paracalliopoe lowryi</i>)	A	11	0.41	1.10	1.2					10	0.31	0.65	2.2
Spionidae (<i>Prionospio yuriei</i>)	P	11	0.41	0.78	0.8								
Spionidae (<i>Prionospio aucklandia</i>)	P	12	0.37	0.77	1.1								
Nereididae (<i>Australonereis ehlersi</i>)	P	13	0.31	0.56	0.8	5	0.28	0.64	2.2	16	0.07	0.23	0.6
Capitellidae (<i>Capitella</i> sp.)	P	14	0.24	0.58	0.7	6	0.26	0.75	2.2	8	0.39	0.67	2.9
Spionidae (<i>Spio pacifica</i>)	P	14	0.24	0.52	0.7					18	0.04	0.19	0.3
Iravadiidae (<i>Nozeba topaziaca</i>)	G	15	0.20	0.56	0.3								
Nephtyidae (<i>Nephtys</i> sp.)	P	16	0.19	0.37	0.5	6	0.26	0.53	1.7	11	0.17	0.38	1.2
Capitellidae (<i>Mediomastus australiensis</i>)	P	16	0.19	0.31	0.5								
Phoxocephalidae (<i>Limnoporeia kingi</i>)	A	17	0.17	0.43	0.4					17	0.06	0.23	0.4
Tellinidae (<i>Tellina deltoidalis</i>)	B	18	0.11	0.31	0.2	10	0.09	0.24	0.4	18	0.04	0.13	0.2
Glyceridae (<i>Glycera tridactyla</i>)	P	19	0.09	0.28	0.3								
Naticidae	G	20	0.07	0.27	0.2	13	0.02	0.10	0.1	17	0.06	0.23	0.4
Buccinidae (<i>Nassarius</i> sp.)	G	20	0.07	0.33	0.2	12	0.04	0.13	0.2				
Lumbrineridae (? <i>Lumbrineris</i> sp. #1)	P	21	0.06	0.23	0.1								
Owenidae (<i>Owenia fusiformis</i>)	P	21	0.06	0.16	0.2								
Spionidae (<i>Pseudopolydora</i> sp.)	P	21	0.06	0.16	0.2								
Oligochaete	O	22	0.04	0.19	0.1					19	0.02	0.10	0.1
Spionidae (<i>Scolecopsis</i> sp.)	P	22	0.04	0.19	0.1					19	0.02	0.10	0.1
Philineidae (<i>Philine</i> sp.)	G	22	0.04	0.13	0.1								
Gobiidae	F	22	0.04	0.13	0.1								
Thyasiridae	B	23	0.02	0.10	0.1								
Phyllodocidae (<i>Eumida</i> sp.)	P	23	0.02	0.10	0.1								
Phoronida	P	23	0.02	0.10	0.1								
Photidae (<i>Gammaropsis</i> sp.)	A	23	0.02	0.10	0.0					18	0.04	0.13	0.2
Penaeidae (<i>Penaeus plebejus</i>)	C	23	0.02	0.10	0.0					19	0.02	0.10	0.1

Sabellidae (<i>Desdemona aniara</i>)	P							9	0.33	0.92	2.5
Exoedicerotidae (<i>Exoediceros fossor</i>)	A			7	0.22	0.45	1.2	14	0.11	0.37	0.7
Hesionidae (<i>Heteropodarke</i> sp.)	P							16	0.07	0.18	0.5
Galeommatidae (Sp. #3))	B							17	0.06	0.23	0.4
Cirolanidae (<i>Pseudolana towrae</i>)	I			12	0.04	0.19	0.2	18	0.04	0.13	0.2
Opheliidae (<i>Lobocheis</i> sp.)	P			13	0.02	0.10	0.1	19	0.02	0.10	0.1
Lucinidae (<i>Wallucina</i> sp.)	B							19	0.02	0.10	0.1
Cylichnidae (<i>Tornatina</i> sp.)	G							19	0.02	0.10	0.1
Actaeiidae	I							19	0.02	0.10	0.1
Bopyridae	I							19	0.02	0.10	0.1
Orbiniidae (<i>Leodamas johnstonei</i>)	P			9	0.17	0.28	0.8				
Mactridae	B			12	0.04	0.13	0.3				
Crangonidae	D			13	0.02	0.10	0.1				
Spionidae (<i>Scolecopsis</i> sp.2)	P			13	0.02	0.10	0.2				
other				13	0.02	0.10	0.1				
Mean abundance/sample		51.3	36.5		16.8	9.9			14.6	12.6	
Mean species richness/sample		8.0	3.4		3.9	1.7			4.8	2.0	
No. species/location		39			27				34		
No. samples		54			54				54		
Total individuals		2771			790				907		

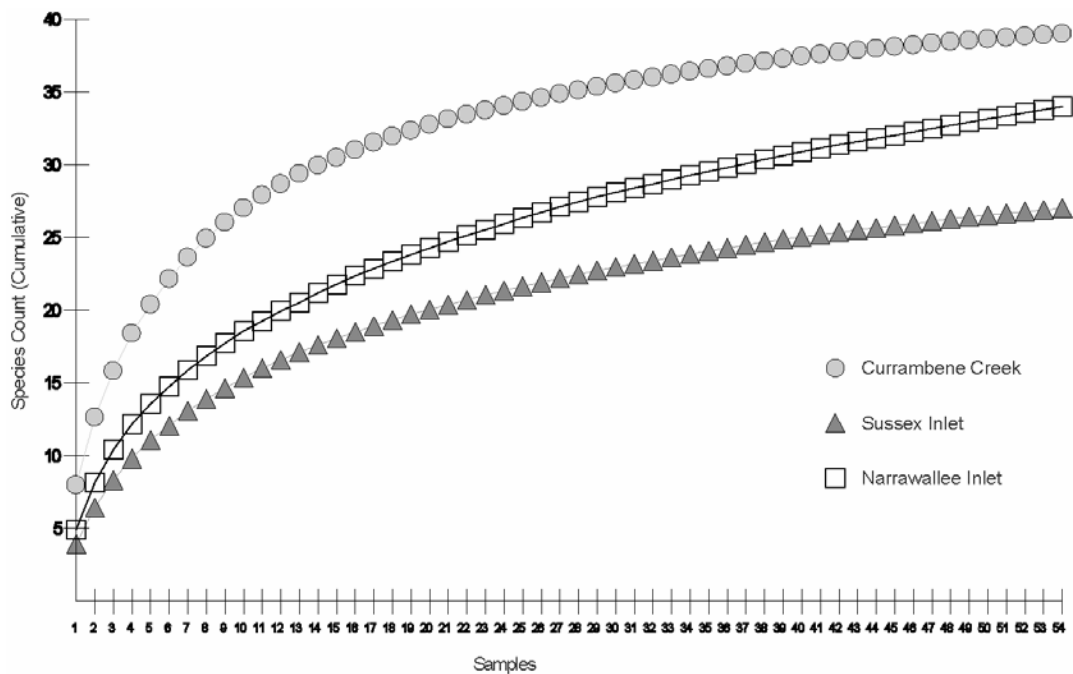


Figure 3-2. Species accumulation per sample curves for each of the tidal flats in each of the three estuaries Currambene Creek, Sussex Inlet and Narrawallee Inlet.

3.3.1. Estuaries

Comparison of assemblages in different estuaries indicated that beta-diversity was insignificant at the scale of estuaries ($p = 0.20$, $F = 1.40$) (Fig. 3-3a,c, Table 3-2), and therefore that a single tidal flat could be considered representative of the assemblages across tidal flats in the 3 estuaries.

In terms of abundance and taxonomic richness, Currambene Creek macrobenthos were nearly three fold more abundant ($p=0.01$, $F=6.33$) and contained twice as many taxa ($p=0.01$, $F=6.08$) compared to Sussex and Narrawallee Inlets (Fig. 3-3b, Table 3-2). The scale of estuaries contributed 34% to total variation for both abundance and taxonomic richness. Currambene Creek also had higher diversity H' than the other two estuaries ($p=0.06$, $F=3.52$), though the contribution to assemblage variation was lower (17%) than for abundance and taxonomic richness.

3.3.1. Sites

Macrofaunal assemblages differed at the scale of sites (100s of m) for all ecological measures across all estuaries (Table 3-2, Fig. 3-4). Notably, a strong pattern of site aggregation was evident in each of the estuaries and was clearly visualized in MDS plots (Fig. 3-4, Table 3-2). This site aggregation was reflected in the highest compositional variation (39%) of assemblages at all scales (Fig. 3-2c). Consequently, beta diversity (taxonomic turnover) across sites at each estuary was high compared to beta diversity across estuaries. Variation in diversity H' (27%) was also higher at the scale of sites than at the scale of estuaries. However abundance and taxonomic richness contributed to between 30-35% of total variation at the scale of sites and estuaries.

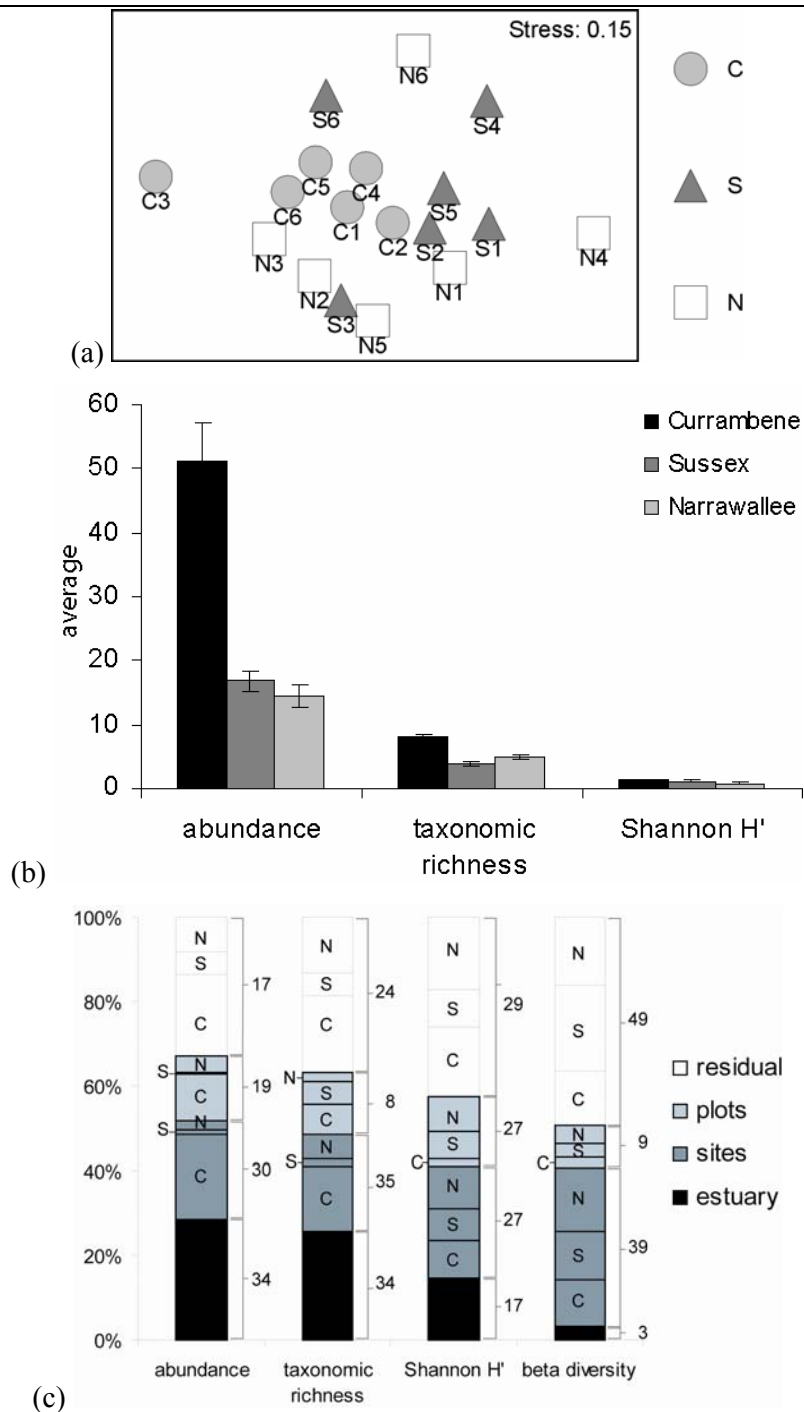


Figure 3-3. Summary of ANOVA and PERMANOVA results. (a) MDS plots of site averages using presence/absence multivariate data (beta diversity). (b) Abundance, Taxonomic Richness and Shannon 'H Diversity means in each of the estuaries. Standard error bars shown. (c) Magnitude of effects estimates of the different spatial scale contributions to the overall assemblage variation in all parameters.

Table 3-2. Summaries of PERMANOVA and ANOVA results comparing beta diversity, mean abundance, taxonomic richness and Shannon Diveristy (H'). ω^2 (variance components) and % values indicate the proportional contribution of each factor (weighting) to the total variation. P(MC) are Monte Carlo probability values calculated by PERMANOVA. Est = estuary or location, Si = sites, Pl = plots.

Factor	SS	df	MS	F	P	P(MC)	ω^2	%	trans
Abundance									
Est	283	2	141.58	6.33	0.01	na	2.21	34	$(x+1)^{1/2}$
Si(Est)	336	15	22.38	4.71	<0.01		1.96	30	
Pl(Si(Est))	171	36	4.75	4.4	<0.01		1.22	19	
Residual	117	108	1.08				1.08	17	
Total	906	161					6.47		
Taxonomic Richness									
Est	494	2	247.04	6.08	0.01	na	3.82	34	none
Si(Est)	609	15	40.63	7.61	<0.01		3.92	35	
Pl(Si(Est))	192	36	5.34	1.95	<0.01		0.87	8	
Residual	296	108	2.74				2.74	24	
Total	1592	161					11.35		
Shannon 'H									
Est	7	2	3.41	3.52	0.06	na	0.05	17	none
Si(Est)	15	15	0.97	3.24	<0.01		0.07	27	
Pl(Si(Est))	11	36	0.30	3.73	<0.01		0.07	27	
Residual	9	108	0.08				0.08	29	
Total	41	161					0.27		
Multivariate									
Est	29437	2	14718.70	1.3951	na	0.20	77.20	3	p/a
Si(Est)	158249	15	10549.92	5.6487		<0.01	964.69	39	
Pl(Si(Est))	67236	36	1867.68	1.5642		<0.01	224.55	9	
Residual	128956	108	1194.03				1194.03	49	
Total	383878	161	2460.00				2460.47		

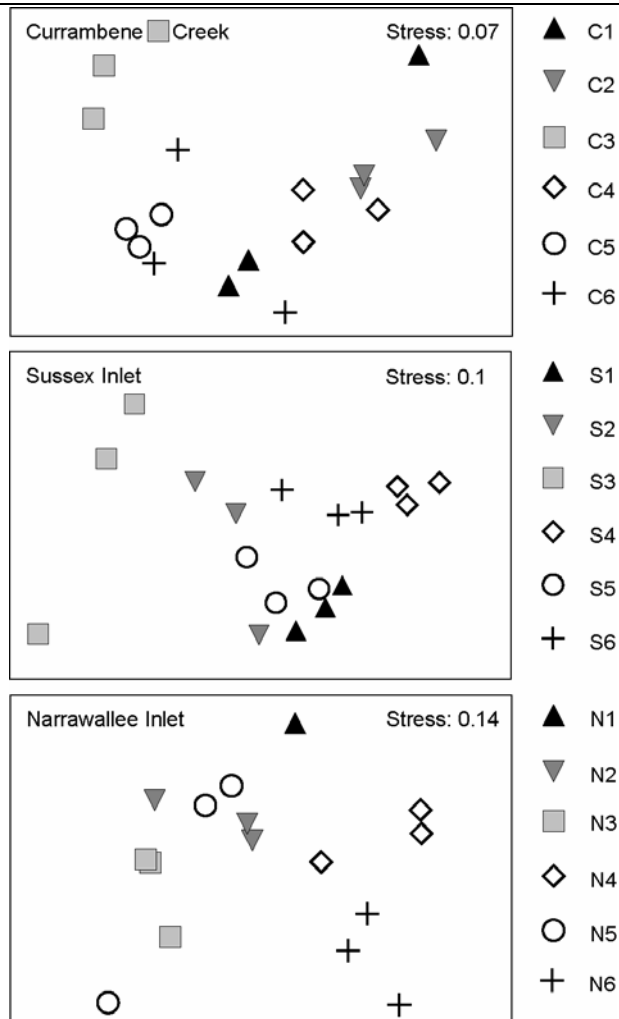


Figure 3-4 . MDS plot comparing the composition of macrobenthic assemblages and relative abundances of taxa at sites. Symbols indicate sites 1-6 in each of the three locations. The sites measuring 20m in diameter are 100s of metres apart, while each point represents a 2m plot of 3 replicate cores.

3.3.1. Smaller scale variation

There were significant differences in all variables at the scale of plots; however the contribution to overall variation was relatively low for abundance, taxonomic richness and beta diversity (Fig. 3-3c). The variation of diversity H' was similar at distances of 20m (plots) to distances of 100s of m (sites).

The relative contribution to variation at the scale of plots differed between estuaries and ecological measures as for the scale of sites. Again, Currambene Creek drove variation in abundance and taxonomic richness at this scale, but Narrawallee Inlet also contributed to variation in abundance, while Sussex Inlet contributed to variation in taxonomic richness. The scale of sites contributed proportionally more variation than the scale of plots for all ecological measures in all estuaries, except for abundance in Narrawallee Inlet and taxonomic richness in Sussex Inlet.

3.3. Discussion

This study found low beta diversity among different tidal flats within a 30km scale on the NSW south coast. There was little taxonomic turnover between assemblages, and the same taxa dominated all tidal flats. This is consistent with Gray (2002) who contends that regional diversity drives local diversity in soft sediment habitats. Despite this indication of low regional beta diversity, taxonomic richness was close to two fold higher in Currambene Creek. A full third of the taxa were found only in this tidal flat. The reason that this significantly higher proportion of taxa was not translated into significant change in beta diversity was due to the rarity of most of these additional taxa. Of course some of these unique taxa may have been missed during sampling in other estuaries due to their rarity. However, alternative hypotheses may be that these taxa have truly small distributions or exist at their geographical range limits. Similarly, 11% and 12% of taxa found in Sussex Inlet and Narrawallee Inlet were unique to those tidal flats respectively, although these differences did not translate into significant taxonomic richness differences between these two estuaries. Ultimately, conservation practitioners have to make choices regarding how many estuaries will provide

representative MPAs, despite limitations in sampling rare taxa. If the goal of MPAs is to represent all taxa, then rare taxa cannot be discounted. Therefore, these findings suggest that a tidal flat may be broadly representative of other tidal flats within 30km, in terms of dominant or common taxa, but the consequence of protecting only one tidal flat may be reduced representativeness of rare taxa. Multiple tidal flat reserves would increase the probability that representativeness is achieved.

A further consideration is that beta diversity will increase at some scale greater than the 30km reported here. The coast of NSW encompasses the southern and northern limits for many tropical and temperate marine species respectively (Turner et al. 2004), and other studies in the tropics and cold temperate regions as well as temperate estuaries in Europe contain different (although functionally similar) species (Dittmann 2000, Edgar and Barrett 2002, Rodrigues et al. 2006). Another study (Ferrell et al. 1993) in the same region as this one identified compositional differences in estuarine assemblages of abundant fish species at scales of 100s of kilometres. The 30km scale presented here is a large increase in scale in terms of the majority of tidal flat research (Constable 1999), but larger scale tidal flat research will be required to determine at what regional scales beta diversity changes significantly.

In addition to being the most taxonomically rich of the estuaries sampled, Currambene Creek also showed a near three fold higher density (abundance) of macrofauna than the other estuaries. Areas with high taxonomic richness and abundance have been suggested as criteria for nominating areas as candidates for conservation (Fox and Beckley 2005, Meir et al. 2004). However, there is concern over whether such hotspots can fulfil representation of MPAs (Fox and Beckley 2005) or enhance recruitment processes and

resilience of tidal flats (Carr et al. 2003, Gaines et al. 2003, Grantham et al. 2003).

Additionally, Currumbene Creek drove the heterogeneity of abundance and taxonomic richness at the scales of sites (100s of m) and plots (20m); which is possibly an effect of high abundance and taxonomic richness in this tidal flat (Holt et al. 2004). This was not translated into higher values of the 2 diversity indices, but is none the less a type of ecological heterogeneity important to consider for representative MPAs. Representation of tidal flat diversity may require that both heterogeneous and homogeneous habitat types are included in MPAs. For example, whether assemblages are more or less heterogeneous may affect subsequent resilience to different types of impacts (McClanahan et al. 2002). The implications and resilience associated with different degrees of such heterogeneity require investigation.

Most notable at the scale of sites and common to all estuaries was a clear increase in heterogeneity in assemblage composition (beta diversity and diversity (H')).

Assemblage composition at one site in a tidal flat differed considerably from assemblage composition 100s of m away. Similar scales of species patchiness have been reported elsewhere (reviews in Peterson (1991); Thrush (1991), also Hewitt et al. (1997); Dittman, (2000)).

Predictive models of the processes causing assemblage patterns within tidal flats could improve our efficiency in choosing ideal locations for MPAs (Constable 1999), particularly where tidal flats are large and/or where reserve selection may be compromised by other considerations. Patterns of assemblages are thought to be related to local environmental variables (Dittmann 2000, Edgar and Barrett 2002, Hewitt et al. 1997, Posey et al. 1995, Thrush 1991). It is clear that the tidal flats we have examined

exhibit heterogeneity in environmental factors at the scale of 100s of m; for example mud content and tidal inundation is highly variable among sites (Winberg, personal observation). If however a tidal flat extends over kilometres, scales at which environmental variables such as tidal height operate would also increase, resulting in increased scales of assemblage heterogeneity. One study of a much larger temperate tidal flat in Portugal (Rodríguez et al. 2006) supports this assertion. However at this stage predictive models are limited (Constable 1999), thus the results presented here illustrate the need to include whole habitat as reserve, rather than a portion of tidal flat, in order to encompass the full spectrum of ecological heterogeneity including all species.

Of concern is how long the observed patterns persist. Although shorter temporal seasonal changes were not detected in this study, which spanned four months from winter to spring, Silva et al (2006) report seasonal heterogeneity in a temperate estuary in Portugal. Seasonal heterogeneity would intuitively also be important in other regions that experience dramatic seasonal changes such as temperatures below freezing or monsoonal rains. However, taking short term seasonal variation into account still allows for temporal persistence of spatial patterns over an annual scale. Assuming that patterns of assemblages at the scale of sites are driven by environmental factors, it follows that these patterns should be as persistent in time as the environmental variables themselves. Low levels of temporal variation persisted for the three years in which these locations have been sampled (Chapter 4), and are in agreement with other research experience from temperate and cold temperate regions (Edgar personal communication).

At the 20m scale, reduced taxonomic turnover implies that assemblages were

compositionally less heterogeneous than at 100s of meters. This is a similar finding to that of Kendall and Widdicombe (1999) who reported that shallow sub-tidal sediment assemblages were homogeneous at scales up to 50m, with a trend to increased variability at 500m. An explanation for the observed decrease in taxonomic turnover at this scale follows from the argument that heterogeneity at the 100s of m is driven by environmental factors that operate at that scale. Differences in tidal inundation, flow rates and influence from adjacent habitats would all be smaller at the 20 m scale than the 100s of m scale, and result in more homogeneity in structuring effects and the resulting assemblage. This scale of 100s of m would therefore represent alpha diversity of macrofauna in tidal flat habitat (Schluter and Ricklefs 1993).

For the other ecological measures of abundance, taxonomic richness and Shannon H' diversity, heterogeneity also decreased between scales of 100s of m to 20m with the exception of abundance in Narrawallee Inlet and taxonomic richness in Sussex Inlet. With general increased homogeneity at this scale as well as the trend for reserves in MPAs to be substantially larger than 10s of metres (Halpern 2003), heterogeneity of assemblages at 20m will, in practice, be encompassed within a reserve. Some variation however, highlights the need for adequate replication at smaller scales in tidal flat research.

There is currently no consensus on an optimal set of ecological measures in order to properly compare studies from different regions, although it has been suggested that certain approaches are preferable to others (Fox and Beckley 2005). Similar to other tidal flat research (Dittmann 2000, Edgar and Barrett 2002, Rodrigues et al. 2006), we found that the application of a range of different measures of ecological diversity

(abundance, taxonomic richness, diversity (H') and beta diversity) described different aspects of ecological heterogeneity operating at different spatial scales. Dittman (2000) measured different size classes of fauna as well as proportions of functional groups and found different patterns of heterogeneity in tropical tidal flats, a characteristic that remains to be tested here. Edgar and Barrett (2002) recorded high variation in density and taxonomic richness within locations of cold temperate tidal flats, while biomass and productivity varied at much larger scales. This illustrates the general importance of multiple ecological measures at multiple scales to describe spatial patterns of fauna. Similar studies in different habitats have included an approach similar to that presented here, for example sub-tidal rocky reefs (Curley et al. 2002), and this approach could easily be adapted to most habitat types. As the representativeness of MPAs should encompass not only all species, but also ecological heterogeneity and diverse habitat function, as broad a suite of ecological measures as possible within logistical and financial limits is desirable.

In this study, heterogeneity of assemblage composition (beta diversity) is greatest at scales of 100s of metres, while heterogeneity in abundance and taxonomic richness is equally important at 100s of metres as at 10s of kilometres. The proportional contribution to variation within each tidal flat from the different scales may vary between each estuary, but it is the site heterogeneity across 100s of meters that stands out in this study. This scale is comparable to scales of heterogeneity in tidal flats of a similar size (eg. Dittman (2000), but as mentioned above, the scale may increase as the area of the tidal flat increases (Rodrigues et al. 2006). Therefore, for conservation purposes it may be more appropriate to refer to heterogeneity at smaller scales as “within location” heterogeneity, as opposed to between location heterogeneity for

different estuaries or distinct tidal flats as Edgar and Barrett (2002) did. In terms of composition alone, heterogeneity at this scale might be defined as beta diversity operating within habitat locations. In this case gamma diversity would describe the collective assemblage of taxa of all estuaries as it is the product of alpha diversity (20m scales) and beta diversity (100s of meters) (Schluter and Ricklefs 1993).

Constable (1999) stated in a review of tidal flat research that there have been few studies that define habitat characteristics, and there is a need to know what size of habitat area is required to maintain the status of an assemblage. Here we have not addressed the size of a habitat that will optimize resilience or function, but we have provided information on how to achieve the Marine Protected Area goal of representativeness in tidal flats. First, a tidal flat on the South Coast of NSW may represent the gamma diversity within 30km in terms of dominant taxa, but inclusion of multiple tidal flats would ensure the inclusion of rarer taxa in reserves. Additionally, multiple tidal flats would allow representation of between-location heterogeneity for other ecological characteristics such as abundance or different intensities of within location heterogeneity. Second, whole tidal flat habitat should be protected to encompass the within-location ecological heterogeneity, including beta-diversity, which has been identified here as well as in other studies (Dittmann 2007, Edgar and Barrett 2002, Rodrigues et al. 2006). Ecological heterogeneity is an important characteristic of a habitat, and factors that drive heterogeneity of tidal flat assemblages should be explored further. Heterogeneity may affect recruitment, function and resilience of these assemblages (Tews et al. 2004). Awareness of these issues will optimise the choices for conservation, as well as set priorities for the future assessment of tidal flats.

CHAPTER 4 - ECOLOGICAL EFFECTS OF EXCLUDING BAIT HARVESTING FROM A TIDAL FLAT: A BEYOND BACI ASSESSMENT OF A MARINE PROTECTED AREA



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4.1. Introduction

Marine Protected Areas (MPAs) are recognized as important to the conservation of marine habitats (Gaston et al. 2006, Halpern and Warner 2002, Lubchenco et al. 2003, Serrano et al. 2003) and the management of fisheries (Hastings and Botsford 2003, Tetreault and Ambrose 2007, White and Kendall 2007). In light of the expanding global network of Marine Protected Areas, there is a continual need for unambiguous evidence of the benefits and functions of MPAs to justify their impacts on society. However, research in MPAs is dominated by studies in tropical climates, reef ecosystems and/or target species of commercial value or icon status (Pinnegar et al. 2000, Russ and Alcala 2003, Shears et al. 2006, Willis et al. 2003). There is a notable absence of research on species perceived as commercially less important and on whole habitat effects (but see Langlois (2006)).

In a literature search on 682 peer-reviewed papers on Marine Protected Areas in the last five years, coral reefs were represented by 174 papers, rocky reefs by 27 and soft-sediments by 4 papers. Of the 682 studies, 283 studies related to fish, while 86 referred to whole assemblages, of which only 30 related to invertebrates. This research bias is understandable in light of justifying the impact and benefits of MPAs to society; however the implications and potential benefits of MPAs are more complex and long-term than immediate returns to commercial and recreational fishers.

Despite this research bias, MPAs are concerned with management of common marine resources for multiple benefits to society, including a range of important ecosystem services in diverse habitats (Lubchenco et al. 2003, Thrush and Dayton 2002, Worm et al. 2006). Therefore, in light of the diminishing number of healthy estuarine systems

on a global scale, tidal flats are a target for protection in the New South Wales system of MPAs in Australia (JBMP 2002). Soft sediment tidal flats provide vital ecosystem services; in particular, energy and nutrient transfer from the terrestrial catchment to marine biomass (Austen et al. 2002). This includes commercial and recreational fish resources that humans rely on (Loneragan and Bunn 1999, Robertson 1984, van der Veer et al. 2001).

Although there is a reasonable body of literature that relates to the general ecology of tidal flats, the translation of this existing knowledge into planning and management of MPAs is seldom realized. In particular, experimental results are seldom transferred to generalized predictions at large scales, and rarely do they test the outcomes of planning and management decisions. Thus planning and management of tidal flat habitat in MPAs becomes best guesses, and protection is difficult to justify. As an illustration, Marine Protected Areas and ecological services were identified in the top 14 topics for questions of high policy relevance in the UK (Sutherland et al. 2006).

It is thus evident that conservation planners and managers require information on spatial and temporal patterns of biodiversity, associated ecological processes (including ecosystem services), as well as an understanding of the threats and the consequences of human impacts. This need for knowledge for application in planning and management is also compatible with many questions that ecologists are trying to answer in experimental studies. It is therefore very beneficial for scientists to grasp the opportunity to use management decisions as large scale experiments, and in turn it is beneficial for managers to engage scientists in a meaningful way for well informed conservation management. These ideas are not new (Bishop and Chapman 2004,

Underwood 1995), however there remains broad potential for this mutually beneficial approach to be explored. After all, the justification for most conservation research is that it will lead to better management of natural resources; research findings however are of little use unless they are translated into an accessible format for planners and managers.

Here I took advantage of a planning decision to implement a no-take Marine Protected Area (MPA) in the Jervis Bay Marine Park, NSW Australia, to test the effect of bait harvesting and associated sediment disturbance on tidal flat invertebrate assemblages. In this way, the opportunity for a large scale experiment on sediment disturbance also provided conservation managers with measured outcomes to justify current management decisions or to better inform future planning. Previous studies have shown that bait harvesting in soft sediments does have measurable impacts on invertebrate assemblages in the short term (Skilleter et al. 2005). However, the longer term effects from constant harvesting have not been tested, and monitoring for successional assemblage change after cessation of long term bait harvesting is one way to assess what the longer term impacts were.

The objective of this study was to monitor soft sediment invertebrate assemblages over three years, including one year before no-take protection, to determine if changes to the assemblage were measurable compared to control sites. Both the targeted species (*Trypaea australiensis* (ghost shrimp) - Decapoda; Thalassinidea; Callianassidea) and non-target invertebrates were monitored for impacts from harvesting rates and effects of sediment disturbance.

4.2. Methods

4.2.1. Location

This research was conducted on the south coast of NSW, Australia, between June 2002 and October 2004. A spatially hierarchical sampling design was used to sample macrofaunal assemblages in 1 tidal flat in each of 3 estuaries (Currambene Creek - Jervis Bay Marine Park ($35^{\circ} 01' S$, $150^{\circ} 40' E$), Sussex Inlet ($35^{\circ} 10' S$, $150^{\circ} 35' E$), Narrawallee Inlet ($35^{\circ} 18' S$, $150^{\circ} 28' E$)) (Fig. 4-1). The estuaries were separated by 10-30km of each other and the tidal flats were within 1 km upstream of the permanently open mouths of the estuaries. Currambene Creek was identified to become a no-take MPA in the Jervis Bay Marine Park (JBMP) in November 2002.

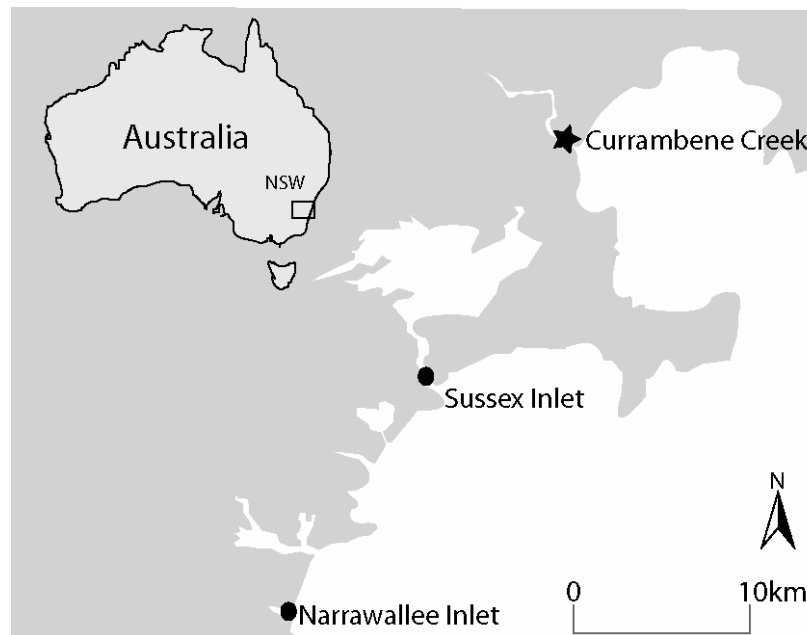


Figure 4-1. Location of the three tidal flats, in three different estuaries, on the southern NSW coast of Australia. The Currambene Creek tidal flat (black star) was protected in a no-take Marine Protected Area after the first year of sampling (Impact site), while

Sussex Inlet and Narrawallee Inlet (black circles), the control sites were not protected from bait pumping during the three year study.

The present study measured the effect of a no-take Marine Protected Area on the estuarine tidal flat assemblages in Currumbene Creek, NSW, Australia. No-take protection implied that sediment disturbance from the bait harvesting of ghost shrimp (Decapoda; Thalassinidea; Callinassidea; *Trypaea australiensis*), ceased across the whole tidal flat; an activity that was considered a daily occurrence in Currumbene Creek (JBMP staff communication and field observations). Cessation of bait-pumping was enforced through the work of compliance staff, and was considered close to fully effective.

4.2.2. Sampling design

To monitor for a response trajectory of tidal flat biological diversity, from the year before to 2 years after the tidal flat in Currumbene Creek (C) was zoned as a no-take Marine Protected Area (MPA), a spatially nested design was used to sample invertebrate assemblages. This sampling program was replicated in two control tidal flats outside the Marine Park, thus the experimental design followed the logic of asymmetrical Beyond BACI (Before After Control Impact) experiments, as described by Underwood (1992), to detect environmental impacts; in this case the impact of no-take zoning of a tidal flat. Bait pumping practices continued (no protective measures instigated) on a regular basis in the control tidal flats in Sussex Inlet (S) and Narrawallee Inlet (N) throughout the study. Transferral of effort from the protected tidal flat (C) to the control tidal flats (N and S) was not anticipated nor observed during the

study as the distance between the tidal flats was prohibitive.

Each of the three tidal flats was sampled on one of three separate days within each of two time periods between June and October, 2002; before the no-take zoning of Currambene Creek. In 2002, three sites were haphazardly selected at least 100m from each other within each tidal flat on each day (see Chapter 3.1 Winberg et al. (2007) Figure 3-1). Site locations were recorded with a GPS, and at each site, 2m² plots were randomly selected approximately 20m from each other, and sampled with three replicate sediment cores. The sites in each tidal flat were similarly sampled again, during the same weeks, in 2003 and 2004 after the zoning of Currambene Creek as a no-take MPA; thus a total of 6 sites were sampled in each tidal flat each year. In total, 27 cores were taken from each tidal flat, on two occasions per year; thus 54 cores were taken per tidal flat and year, and 486 cores in total over 3 years.

Previous studies have shown that the greatest spatial variation of faunal assemblages in tidal flats can occur at scales of 100s of meters (Edgar and Barrett 2002, Winberg et al. 2007). Therefore the same sites were revisited throughout the study and regarded as sub-samples (Smith 2002) to minimize variation between years due to differences between sites. It was also important to sample the same sites at the same times, within each estuary and year, as some inter-tidal soft-sediment species exhibit seasonal variation within the time periods (Cummings et al. 2003).

A pilot study (Chapter 2) identified that 15cm diameter and 25cm deep cores were optimal for comparable samples of macrofaunal assemblages in the sediment, compared to other cores or bait pumping methods. The cores were as deep as practical in an attempt to sample deep burrowing fauna; specifically the targeted bait species

Trypaea australiensis. In addition to cores, approximately 50 specimens of T.

australiensis were sampled randomly across the tidal flat with a bait pump on each sampling day to measure if there were any other population effects, apart from changes in abundance, from harvesting rates. These samples were measured for size frequency analysis (carapace length) and sexed to measure the ratio of females to males and the presence of gravid females.

Corers were inserted into the sediment and levered out with a shovel and sieved *in situ* through 1.2 mm mesh bags. A comparatively large mesh size relative to other studies was used due to the large volume of sediment sampled as well as the coarse grain size. Sediment samples for grain size and organic content analyses were also taken at each plot using a smaller 4cm diameter corer to a depth of 5cm, and placed in sealed plastic bags.

Faunal samples were placed in finer mesh bags, frozen on return to the lab, and were later defrosted, stained and preserved in a solution of Biebrich Scarlet in 5% formol for at least two days. Biebrich Scarlet was effective in simplifying the removal of fauna from the remaining sediment. Taxa were identified to the lowest taxonomic level possible, usually species, using a dissecting microscope. Rarer, juvenile or difficult taxa were identified to the genus or family level.

Sediment samples were frozen on return to the lab, and defrosted later for grain size analysis using the Malvern Mastersizer laser analyser. Percentage organic content was measured by drying sediment at 90°C for 48 hours, and weighing approximately 3g quantities in ceramic crucibles before and after combustion at 500°C for 3 hours.

4.2.3. Analysis of data

4.2.3.1. Multivariate Analysis

First, the choice of transformation of data for analysis was determined using the 2-Stage procedure in the PRIMER 6 software package (Clarke and Warwick 2001). This procedure is discussed further in Chapter 5b, but here untransformed and presence/absence data at the highest taxonomic resolution were chosen for further analysis. Untransformed data provided the clearest multivariate measure of relative changes in abundance, while presence absence data was used to explore changes in composition.

Species accumulation plots had previously been done to confirm that an adequate sample size was used and that the majority of taxa were sampled (Winberg et al. 2007).

Dominance plots for each of the three sample years in each of the three tidal flats were analysed visually to assess if there were changes in taxonomic dominance in assemblages.

To visualize changes to whole assemblage patterns following no-take zoning of the Currumbene Creek tidal flat, compared to the control tidal flats, a 2-stage analysis was undertaken using PRIMER 6. For this analysis, data were structured to compare matrices for each of the tidal flat assemblages from each of the three years sampled. The data were pooled at the scale of sites, the scale of most variation (see Chapter 3), prior to Bray-Curtis similarity matrices being calculated as recommended by Bob Clarke (personal communication). The Bray-Curtis similarity matrices for each tidal flat and year were then compared in a Second-Stage matrix of Spearman rank correlations and

displayed in an MDS plot to visually assess if there was a change to the macrofaunal assemblage pattern at the no-take tidal flat.

To test the assemblage changes seen in the second stage MDS plots, the PERMANOVA extension to the PRIMER 6+ software package (beta version 17) was used to accommodate the full nested hierarchical design and spatial asymmetry of the data set, including the asymmetrical interaction factors of interest (Table 4-1). Monte Carlo probability values were used because the smallest scale of replication was $n = 3$ (cores in plots) (Anderson 2005). Further multivariate pair-wise *a posteriori* tests, equivalent to univariate t-tests, were performed for the factors of interest to identify which tidal flat assemblages changed and at what scales.

SIMPER analysis on untransformed data, pooled at the scale of sites, was used to determine which taxa contributed most to eventual assemblage differences in the Currumbene Creek tidal flat before and after no-take zoning. The taxa that contributed to a cumulative 80% of Bray-Curtis dissimilarity between assemblages at the Currumbene Creek tidal flat, before and after no-take protection, were selected for further univariate analysis.

Further SIMPER analysis using presence absence data was used to determine if there were any compositional changes, or changes to species homogeneity between sites (presence in samples).

Table 4-1. The multivariate analysis model with crossed and nested factors. The highlighted asymmetrical interactions were used to indicate effects of protection of the tidal flat.

Factor	Abbreviation	Levels	Type
Year	Ye	3	Fixed
Before vs. After	BvsA	2	Fixed
Estuary (or tidal flat)	Es	3	Fixed
Control vs. Impact	Cvl	2	Fixed
Time	Ti(Es)	2	Random
Site	Si(Ti(Es))	3	Random
Plot	Pl(Si(Ti(Es)))	3	Random
	YexEs		
	BvsA x Es		
	Ye x Ti(Es)		
	BvsA x Ti(Es)		
	YexSi(Ti(Es))		
	BvsAxSi(Ti(Es))		
	YexPl(Si(Ti(Es)))		
	BvsAxPl(Si(Ti(Es)))		

4.2.3.2. Univariate analysis

Univariate analyses were performed using selected taxa, a range of diversity measures (species richness, abundance, Shannon H' diversity and J' Pielou's evenness) and percentage silt and clay in sediments, to test for the effect of no-take zoning in Currumbene Creek. Two common families of polychaetes (spionidae and capitellidae), are considered functionally uniform families in soft sediments (Read 2004), and species of these taxa were pooled at the family level to increase the power of the test as individual species numbers were low. The target species for bait harvesting, *Trypaea australiensis*, was also added to the list for analysis, although it didn't place in the top 80% of taxa that contributed to assemblage change.

A five factor ANOVA with interactions between Years and the four hierarchically nested factors (estuaries, time, sites and plots) was used following the model described in Chapter 5.1 and shown in Table 4-2 (all data) using GMAV 5 software (University of Sydney, Australia). Asymmetry in the data set was accommodated following the logic outlined in numerous asymmetrical Beyond BACI studies (Guidetti 2002, Terlizzi et al. 2005, Underwood 1993). Spatial asymmetry between Control (2 tidal flats) and Impact (1 no-take tidal flat) locations was accounted for by analysing a full data set with ANOVA, and then subtracting the sums of squares from a separate ANOVA using the control data set alone. This method was extended further to include temporal asymmetry (1 year Before protection and 2 years After) by calculation of further sums of squares from data sets using After and After/Control data (Table 4-2). This method is further described in Chapter 5.1.

Unusual to this case study, the factor time was nested in locations. It is intuitive that time should be nested in Years; however each time was unique to three sites in each tidal flat that were sampled on the same day, and therefore was considered nested in, rather than orthogonal to, tidal flats. Ecological experiments commonly use different mixed models (nested, crossed, fixed and random factors) in Beyond BACI experimental designs to detect impacts on biological assemblages. In other circumstances, time within years is consistent between locations then it should be nested within Year and orthogonal to Location.

Time could alternatively have been considered as a spatial sub-group of each tidal flat, with three sites nested in each. This would not change the model, only the labels; however, because seasonal variations might have contributed to differences between

sub-groups as they were sampled at an interval of approximately 3 months, they were labeled as time.

Assumptions of ANOVA were tested; normality was assessed visually in plots of means against variances (Quinn and Keough, 2002), and Cochran's test was used to test for homogeneity of variance. Data were transformed to achieve homogeneity ($p \geq 0.05$) using a transformation ($\ln(x+1)$, $x^{0.5}$, $x^{0.25}$, $x^{0.1}$) suitable to each of the univariate datasets (Quinn and Keough, 2002). For a number of variables, Time was not significant ($p \geq 0.25$) and data were pooled at that level to increase the degrees of freedom for the spatial factors.

For analysis of the sediment data, plots represented the smallest scale of replication, thus the model for ANOVA included three (Estuaries, Time, Sites) rather than four nested factors.

Table 4-2: The full univariate asymmetrical ANOVA model based on four analyses of variance using different components of the full data set (All, After, Control and After/Control data). The analyses provided the sources of variation (sums of squares) which were used to calculate unknown variation sources for the interactions of interest, that is, to test for effects of no-take zoning on the tidal flat in Currumbene Creek (highlighted). F values for the factors of interest used denominator mean squares from factors in order of sequence identified in the F vs. column. The sequence of denominator mean squares were chosen according to whether the corresponding factor was not significant at $p \geq 0.25$, in which case the factor was pooled and the next factor was tested. The factors mean squares used for denominators in the two-tailed F-tests are in highlighted italics.

Factors	All data	After data	Control data	After/Control data	d.f	F vs.
1	Year (YE)				2	
2		AFTER (A)			1	
3		Before vs. After (Bvs.A)			1	
4	Estuary (ES)				2	
5			Controls (C)		1	
6			Controls vs. Impact (Cvs.I)		1	
7	Times (TI(ES))				3	
8	Sites (SI(TI(ES)))				12	
9	Plots (PL(SI(TI(ES))))				36	
10	YE x ES				4	
11		A x ES			2	
12		Bvs.A x ES			2	
13			YE x C		2	
14			YE x Cvs.I		2	
15				A x C	1	
16				A x Cvs.I	1	20, 29, 38, 47
17				Bvs.A x C	1	15, 22, 31, 40, 49
18				Bvs.A x Cvs.I	1	17, 16 , 19, 28, 37, 46
19	YE x TI(ES)				6	
20		A x TI(ES)			3	
21		Bvs.A x TI(ES)			3	20, 28, 37, 46
22			YExTI(C)		4	
23			YExTI(I)		2	22, 32, 41, 50
24				A x TI(C)	2	
25				A x TI(I)	1	34, 43, 52
26				Bvs.A x TI(C)	2	24 , 31, 40, 49

27			Bvs.A x TI(I)	1	26, 25 , 32, 41, 50
28	YE x SI(TI(ES))			24	
29	A x SI(TI(ES))			12	
30	Bvs.A x SI(TI(ES))			12	29, 37, 46
31		YE x SI(TI(C))		16	
32		YE x SI(TI(I))		8	31, 41, 50
33			A x SI(TI(C))	8	
34			After x SI(TI(I))	4	43, 52
35			Bvs.A x SI(TI(C))	8	33 , 40, 49
36			Bvs.A x SI(TI(I))	4	35, 34 , 41, 50
37	YE x PL(SI(TI(ES)))			72	
38	A x PL(SI(TI(ES)))			36	
39	Bvs.A x PL(SI(TI(ES)))			36	38, 46
40		YE x PL(SI(TI(C)))		48	
41		YE x PL(SI(TI(I)))		24	40, 50
42			A x PL(SI(TI(C)))	24	
43			A x PL(SI(TI(I)))	12	52
44			Bvs.A x PL(SI(TI(C)))	24	42 , 49
45			Bvs.A x PL(SI(TI(I)))	12	44, 43 , 50
46	Residual (R)			324	
47	RA			216	
48	RBvs.A			216	
49		RC		144	
50		RI		108	
51			RAC	108	
52			RAI	72	

4.3. Results

Over 80 macrofaunal species (Fig. 4-2) were identified over the three year period; 59 occurring in Currambene Creek, 45 in Sussex Inlet and 53 in Narrawallee Inlet. The three tidal flats were compositionally very similar, with the proportions of phyla dominated by polychaetes (38-45%), molluscs (27-31%) and crustaceans (15-25%). Although polychaetes were the most abundant phylum, 80% of the total abundance in each tidal flat was attributed to four species; two bivalves *Mysella anomala* and *Eumarcia fumigata*, and the crustaceans *Urohaustorius metungi* and *Mictyris* sp. There were, however clear differences in total abundance between the different tidal flats, which is further described in Winberg et al. (2007) or Chapter 3.

4.3.1. Multivariate analysis

Invertebrate assemblages in the tidal flat of Currambene Creek showed clear changes in the pattern of relative abundance of species following no-take zoning of the tidal flat, as demonstrated by the 2-stage MDS plots of Spearman rank correlation coefficients (Fig. 4-3a). No such change was observed in the control locations. Spearman rank correlation coefficients comparing before and after data were much lower at the no-take tidal flat (C) ($\rho = 0 - 0.26$), than at the two control sites (S and N) ($\rho = 0.44 - 0.80$).

This change at the no-take tidal flat location was limited to shifts in relative abundances rather than a change in the assemblage composition. The MDS plot for presence-absence data showed a high correlation between assemblage composition and before

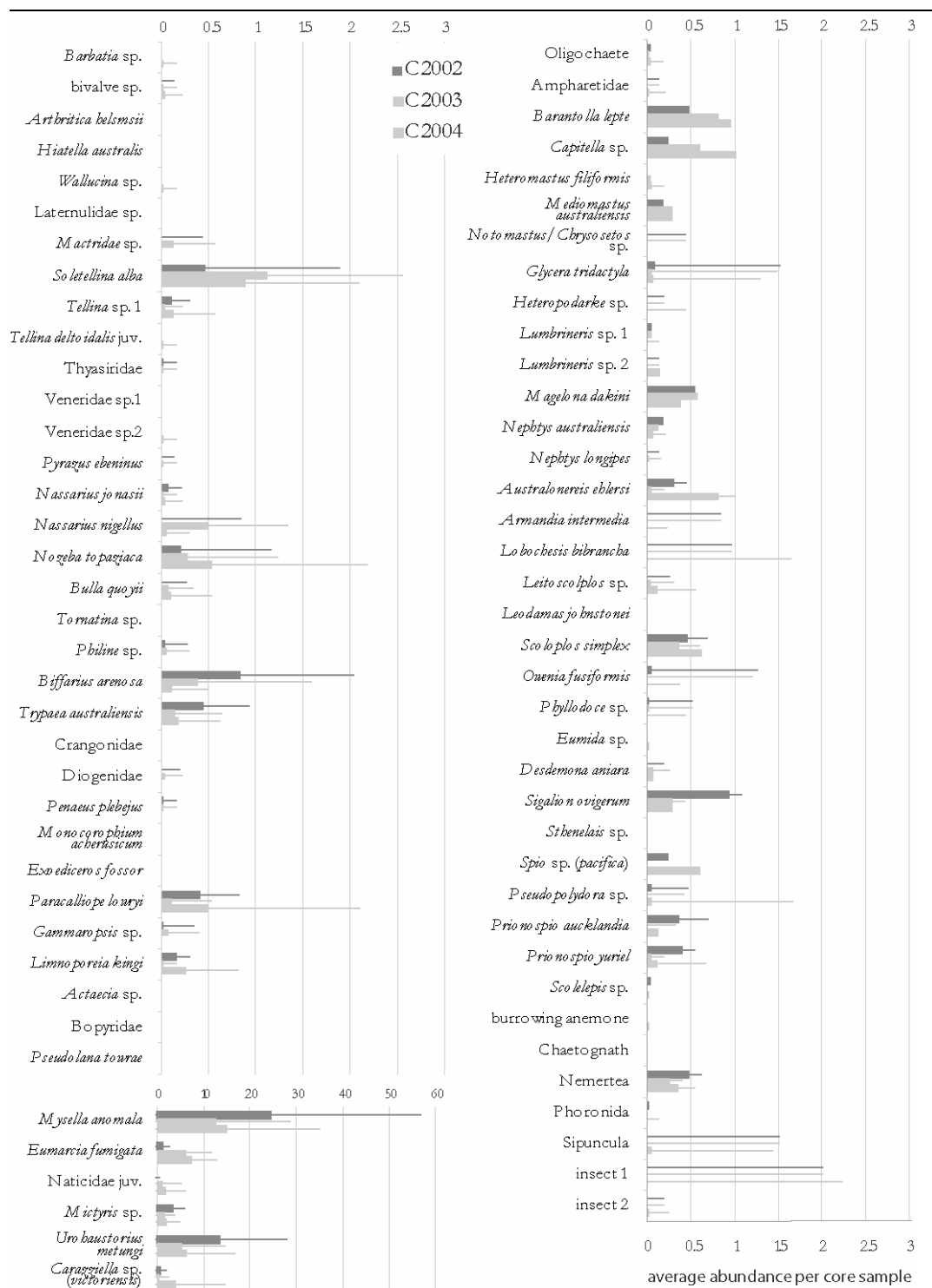


Figure 4-2: List of all taxa identified (grouped by phylum or class), and their average abundance per core in Currambene Creek tidal flat before (C2002) and After (C2003, C2004) protection in the Jervis Marine Park sanctuary zone (SD bars shown).

and after no-take zoning ($p = 0.75 - 0.90$) (i.e. using presence/absence data) (Fig. 4-3b).

Species dominance curves indicated that the shifts in the relative abundance of species were associated with a decrease in dominance of taxa from before to after no-take zoning at Currumbene Creek. The reduction in dominance of the most common 2-3 species was greater than that at the control locations (Fig. 4-4) between the year before no-take zoning and the two years following.

A change in the faunal assemblage at Currumbene Creek tidal flat, following no-take zoning, was confirmed with PERMANOVA and was significant at the scale of sites (100s m). There were significant interactions between before versus after no-take zoning at this scale (Table 4-3), and secondary pair-wise tests confirmed that 75% of sites in the protected tidal flat changed during this period, while site assemblage change was seen in only 29% of comparisons in the control tidal flats (Fig. 4-5).

Due to variability at the scale of sites, assemblage change only bordered significance across the whole tidal flat using the interaction between before versus after no-take zoning and estuaries (BvsA x Es; $p(\text{MC}) = 0.08$) (Table 4-3). However this significance was considered significant using the precautionary principle (Mapstone 1995) at $p < 0.10$, and justified pair wise comparisons which were significant for Currumbene Creek ($p = 0.05$), but not for any of the control tidal flats ($p > 0.17$) (Table 4-3).

SIMPER analysis identified 10 taxa that contributed to >80% of the dissimilarity between assemblages in the tidal flat before and after no-take zoning (Table 4-4); the four most abundant taxa contributed most (70%; $\text{Diss}/\text{SD} \geq 1$) to dissimilarity.

Of the 6 taxa that showed trends of an increase in abundance at the no-take tidal flat, 5 also showed an increase in homogeneity (H) across the tidal flat as they occurred in more samples (presence/absence data) than prior to protection (Table 4-4). This increase in occurrence was not highly consistent across the tidal flat ($\text{Diss}/\text{SD} \leq 0.92$). *M. anomala*, which indicated a decrease in overall abundance, was associated with an increase in homogeneity.

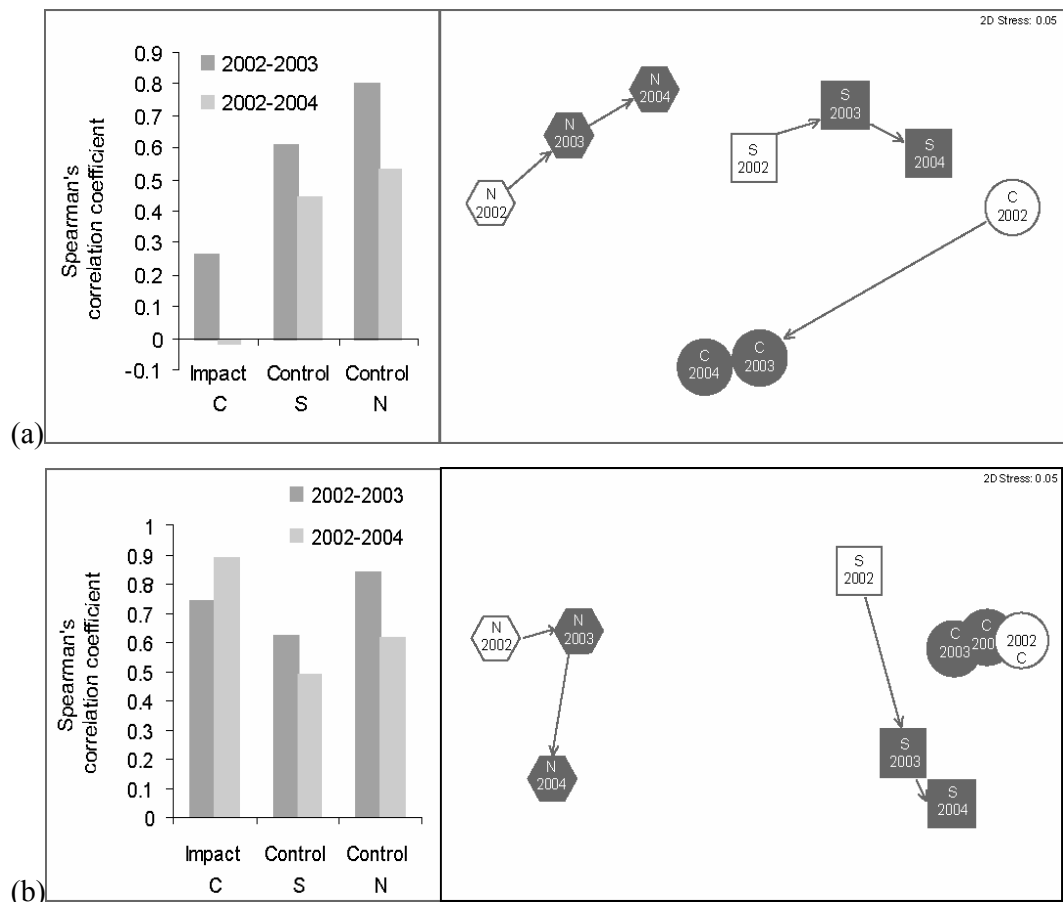


Figure 4-3: Second stage MDS plot of resemblance matrices comparing (a) multivariate abundance and composition correlations (Spearman) (untransformed data), and (b) compositional correlation (presence-absence data) from each of the three tidal flats, Currambene Creek (C2002 – C2004), Sussex Inlet (S2002 – S2004) and Narrawallee Inlet (N2002 – N2004).

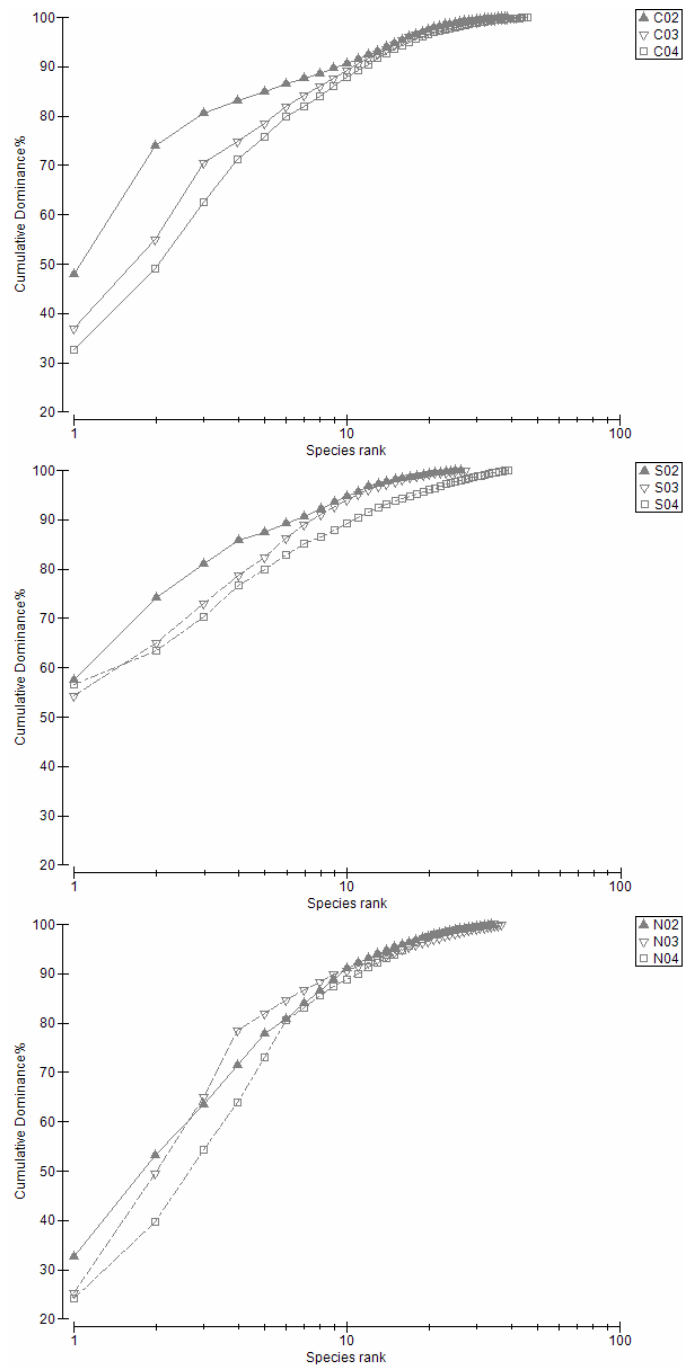


Figure 4-4: Cumulative dominance curves for each year of sampling from the three tidal flats in Currambene Creek (C02 – 03), Sussex Inlet (S02 – 03) and Narrawallee Inlet (N02 – 03).

Table 4-3: PERMANOVA table of results showing the interaction of Before vs. After no-take zoning at the scale of whole tidal flat (estuaries) (BvsA x Es) and sites (BvsA x Si(Ti(Es))). Pairwise tests identify assemblage change for each of the tidal flats from the year before and each of the two years following no-take zoning. Abbreviations used are explained in Table 1. Pseudo-F and P(MC) are equivalent to F ratios and probability (p) values used in univariate tests. Significant interactions are highlighted in bold text.

Source	df	SS	MS	Pseudo-F	P(MC)
YexEs	4	5.27E+10	1.32E+10	3.1975	0.037
BvsA x Es	2	9.01E+10	4.50E+10	3.4371	0.086
Ye x Cvl	2	3.03E+10	1.51E+10	2.7737	0.116
Pairwise				t	P(MC)
				1.8407	0.052
				1.9176	0.048
				1.4193	0.173
				1.3364	0.236
				1.0258	0.517
				0.77832	0.726
				1.1336	0.361
				0.98832	0.526
				1.3513	0.21
Source	df	SS	MS	Pseudo-F	P(MC)
Ye x Ti(Es)	6	2.47E+10	4.12E+09	1.5344	0.161
BvA x Ti(Es)	3	3.93E+10	1.31E+10	2.5584	0.059
YexSi(Ti(Es))	24	6.44E+10	2.69E+09	1.7826	0.009
BvAxSi(Ti(Es))	12	6.15E+10	5.12E+09	2.7401	0.001
YexPl(Si(Ti(Es)))	72	1.08E+11	1.51E+09	0.65479	0.998
BvAxPl(Si(Ti(Es)))	36	6.73E+10	1.87E+09	0.93464	0.598
Res	379	8.72E+11	2.30E+09		
Total	485	1.12E+12			

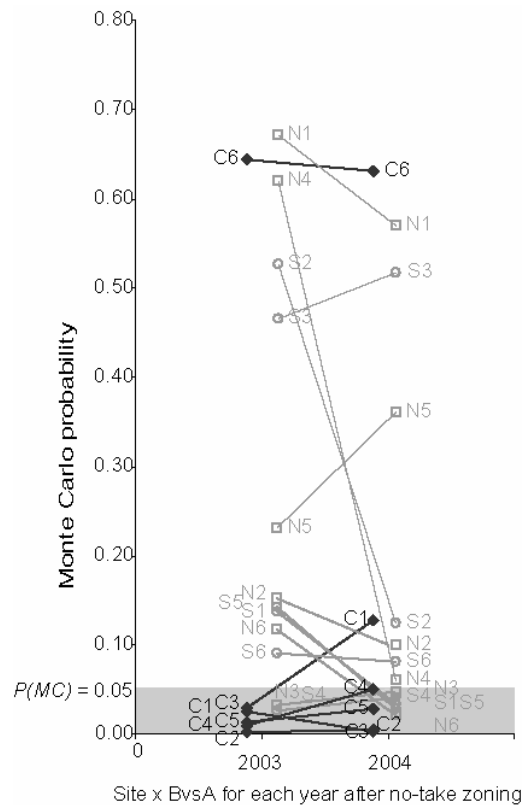


Figure 4-5: Monte Carlo probability $P(MC)$ values for pairwise comparisons of sites between the year before and each of the following two years (2003 and 2004) after no-take zoning. $P(MC)$ are the multivariate equivalent to probability (p) values used in univariate tests when three replicates are used. Currumbene Creek (C1-6: black diamonds) is the no-take zone and the control tidal flats (S1-6 and N1-6: empty circles and squares) are shown.

4.3.2. Univariate analysis

Of the 10 key taxa that contributed strongly to assemblage differences at Currumbene Creek, all except for *Mictyris* sp. showed a significant change in abundance within the

protected tidal flat (C) from before to after no-take zoning (Table 4-4). For these taxa, the change was not reflected at the control tidal flats. Seven of the taxa showed changes in abundance at the scale of sites, while 2 taxa changed in abundance consistently across the scale of the whole tidal flat, and two taxa, *Eumarcia fumigata* and *Urohaustorius metungi*, showed an interaction between the two times of sampling, before and after no-take zoning of Currambene Creek. Changes were generally evident from the year following protection, and maintained for the two years that sampling occurred.

The abundance of the juvenile bivalve species *E. fumigata* increased nearly six-fold across the no-take tidal flat (Fig. 4-6), but due to the interaction between times within each year this was not significant across the whole tidal flat (Table 4-4). The spionid polychaetes showed an increase of almost 3-fold; however this was a site specific and delayed response that occurred only in the second year. The increase in the numbers of naticidae stemmed from an abundance of recruits at site C3 during the two years following protection (Fig. 4-6). The remaining two taxa, the juvenile bivalves *Soletellina alba* and capitellid polychaetes, both increased in abundance consistently across the tidal flat, with significant effects at the smaller scales of plots and sites.

In contrast, there were consistent and significant decreases in the abundance of four common taxa. The most dramatic and consistent decreases in abundance across the tidal flat were for the amphipod *Urohaustorius metungi*, with almost a 60% decrease in abundance (Fig. 4-7 and Table 4-2), and the polychaete *Sigalion ovigerum* with almost a 70% decrease in abundance overall. There was also a significant decrease in the number of the small bivalve *Mysella anomala* in some plots; however this was associated with one site only for this abundant species. *M. anomala* abundance varied

across the other sites, and homogeneity increased from 72% of cores to 88%. The soldier crab (*Mictyris* sp.) showed a consistent decline in abundance across the impacted tidal flat, however this was not significant ($p = 0.08$) compared to the apparent natural temporal variation seen at the control tidal flats, and was variable across sites.

For the suite of four ecological diversity measures, all except Shannon H' Diversity, showed a significant change at the scale of sites for assemblages at Currumbene Creek tidal flat following no-take zoning (Fig. 4-8 and Table 4-4). Assemblage homogeneity (J' Pielou's evenness) increased at all sites except C2 (associated with a very strong increase in the abundance of small bivalves). At the scale of the tidal flat, J' Pielou's evenness increased, while at the control locations it decreased; however the magnitude of change did not differ significantly compared to the control sites. Variations in abundance and species richness were highly site specific with no clear overall trend; however the changes at the protected tidal flat were generally greater than that at the control tidal flats. Changes in abundance and species richness appeared to be correlated. Both increased at sites C1, 2 and 6, decreased at C5, and were unchanged or variable at sites C3 and 4, consistent with findings that changes were related to multiple taxa (Fig. 4-8).

Table 4-4: Taxa listed in order of importance of contribution to dissimilarities in abundance (A) as an increase or (+) or decrease (-) following no-take zoning at the Currumbene Creek tidal flat according to SIMPER analysis. The univariate factors for which changes were found to be significant are indicated in **bold type** under p, and include analyses at different spatial scales and as an interaction with time of sampling in each year and tidal flat. In addition, changes to homogeneity (H) are shown as an increase or (+) or decrease (-).

Species	A	H	Diss /SD	factor	F	p	F vs.
<i>Mysella anomala</i>	-	+	1.29	Bvs.A x Cvs.I	<0.01	0.06	After x Cvs.I
				Bvs.A x sites(Impact)	45.42	<0.001	Bvs.A x sites(Control)
				Bvs.A x plots(S(Impact))	0.19	<0.01	After x plots(S(Impact))
<i>Urohaustorius metungi</i>	-	-	1.07	Bvs.A x Cvs.I	13.13	0.001	years x sites(T(estuaries))
				Bvs.A x time(Impact)	19.05	<0.001	residual (Impact)
<i>Eumarcia fumigata</i>	+	+	1	Bvs.A x time(Impact)	<0.001	<0.001	After x time(Impact)
				Bvs.A x sites(Impact)	26.21	<0.01	After x sites(T(Impact))
				Bvs.A x plots(S(Impact))	0.14	<0.001	Bvs.A x plots(S(Control))
<i>Mictyris</i> sp.	-	-	0.99	Bvs.A x Cvs.I	212.09	0.08	Bvs.A x Control
<i>Carazziella</i> sp. (Spionidae)	+	-	0.42	Bvs.A x sites(Impact)	112.77	<0.001	Bvs.A x sites(Control)
Naticidae juv sp.	+	+	0.4	Bvs.A x sites(Impact)	13.60	<0.001	years x plots(S(Impact))
				Bvs.A x plots(S(Impact))	6.96	<0.001	Bvs.A x plots(S(Control))
<i>Soletellina alba</i>	+	+	0.69	Bvs.A x Cvs.I	16.24	<0.001	years x plots(S(estuaries))
				Bvs.A x sites(Impact)	3.83	0.06	Bvs.A x sites(Control)
				Bvs.A x plots(S(Impact))	8.34	<0.001	Bvs.A x plots(S(Control))
<i>Barantolla lepte</i> (Capitellidae)	+	+	0.56	Bvs.A x Cvs.I	164.00	0.1	Bvs.A x Control
				Bvs.A x sites(Impact)	5.86	0.02	Bvs.A x sites(Control)
<i>Sigalion ovigerum</i>	-	-	0.56	Bvs.A x sites(Impact)	7.42	<0.001	Residual (Impact)
				Bvs.A x plots(S(Impact))	3.43	0.001	Bvs.A x plots(S(Controls))
<i>Capitella</i> sp. (Capitellidae)	+	+	0.4	pooled with <i>B. lepte</i> above			
<i>Trypaea australiensis</i>	-	-		Bvs.A x sites(Impact)	5.92	0.001	years x plots(Impact)
				Bvs.A x plots(S(Impact))	6.74	<0.001	Bvs.A x plots(S(Controls))
Diversity measures				factor	F	p	F vs.
Species Richness				Bvs.A x sites(Impact)	7.22	<0.01	Bvs.A x sites(Control)
Abundance				Bvs.A x Cvs.I	3683.00	0.02	Bvs.A x Control
				Bvs.A x sites(Impact)	8.44	<0.001	Year x plots(S(Impact))
Shannon Diversity				Bvs.A x Cvs.I	1.00	0.36	Year x time(estuaries)
J' Pielou's evenness				Bvs.A x sites(Impact)	8.31	<0.001	Year x plots(S(Impact))
Sediment characteristics				factor	F	p	F vs.
% silt and clay				Bvs.A x Cvs.I	5.40	0.04	Year x sites(estuaries)

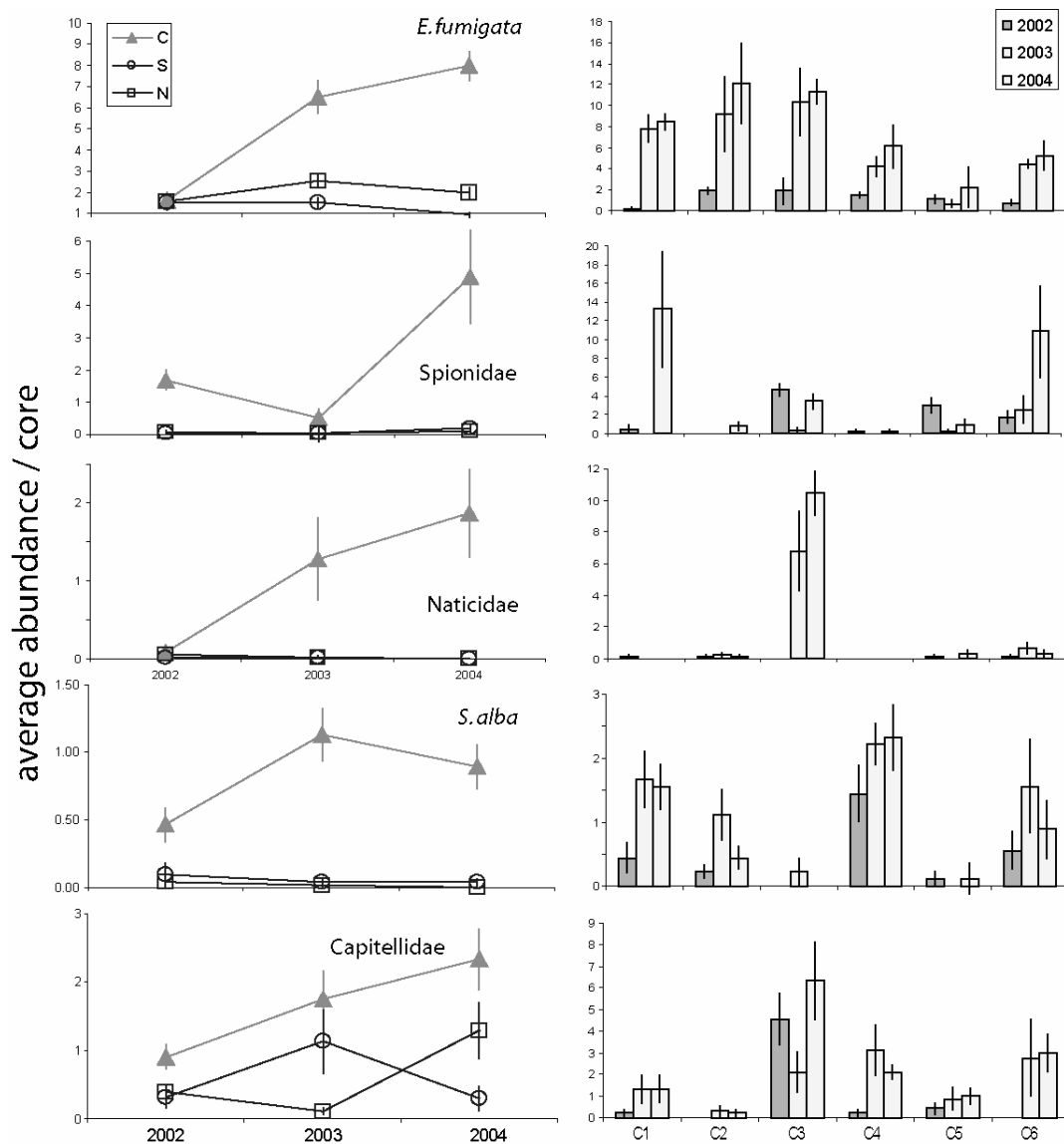


Figure 4-6: Taxa indicating an increase in abundance in the protected tidal flat Currambene Creek (C) compared to control tidal flats (S and N) as shown in Table 4-4. The changes at individual sites within Currambene Creek are also shown (C1-C6). Symbols are denoted at the top of the graph and standard error bars are shown.

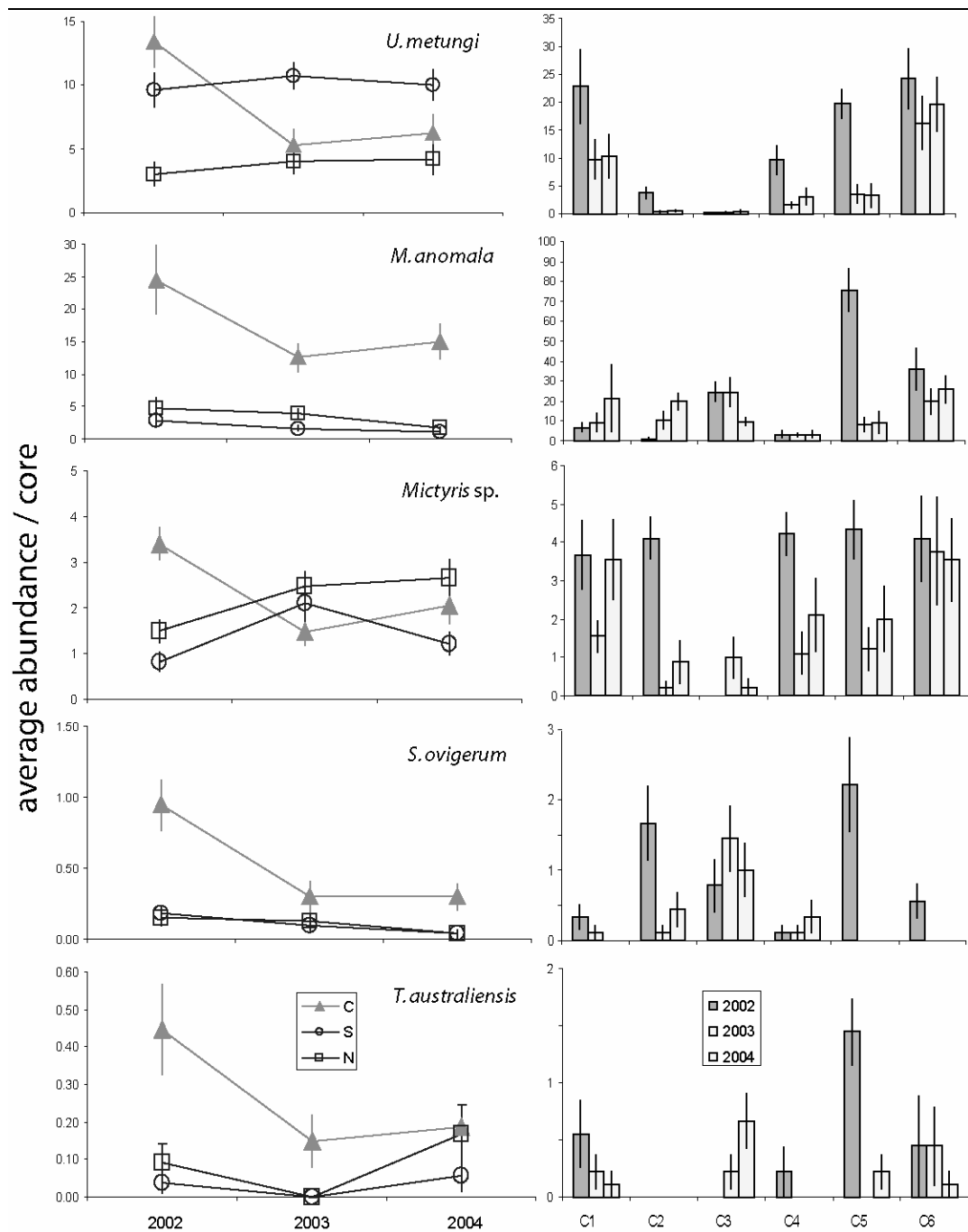


Figure 4-7: Taxa indicating an decrease in abundance in the protected tidal flat Currumbene Creek (C) compared to control tidal flats (S and N) as shown in Table 4-4. The changes at individual sites within Currumbene Creek are also shown (C1-C6). Symbols are denoted at the top of the graph and standard error bars are shown.

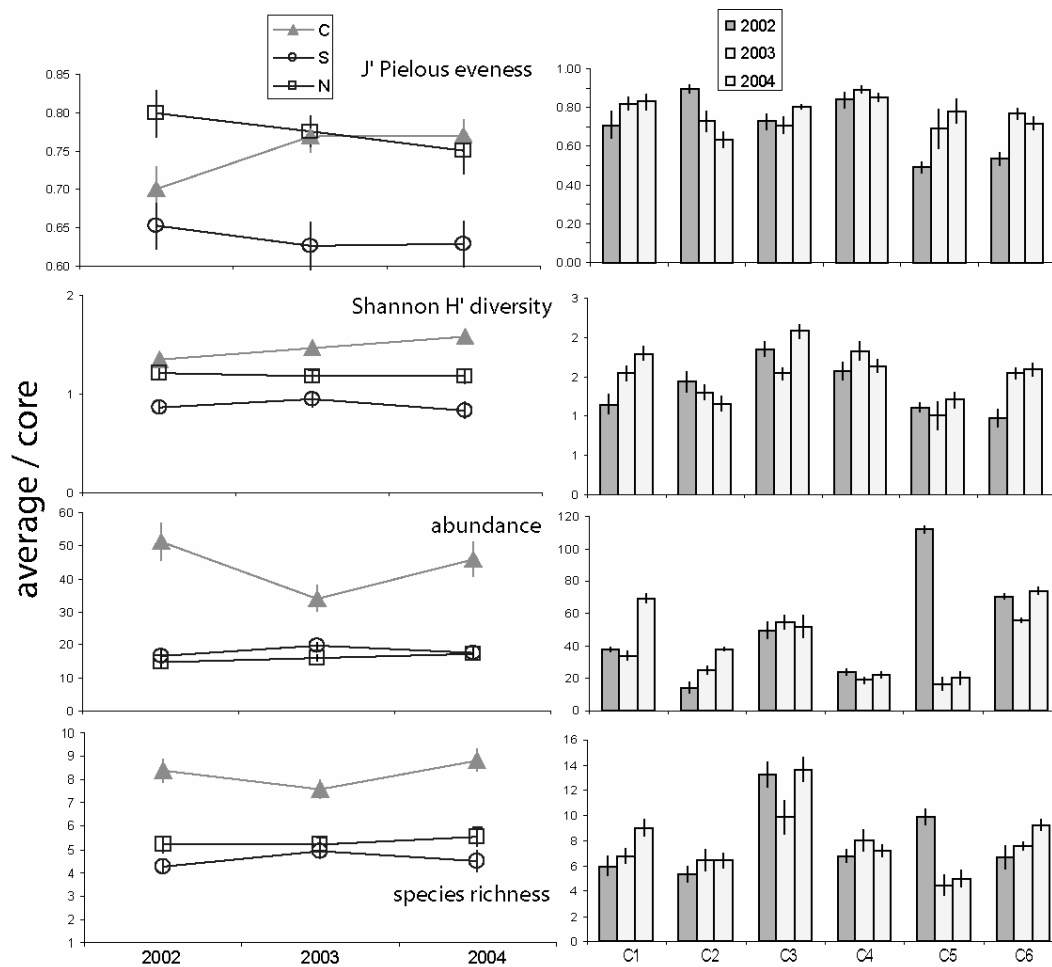


Figure 4-8: Taxa indicating changes in ecological community measures in the protected tidal flat Currambene Creek (C) Currambene Creek (C) compared to control tidal flats (S and N) as shown in Table 4-4. The changes at individual sites within Currambene Creek are also shown (C1-C6). Symbols are denoted at the top of the graph and standard error bars are shown.

4.3.3. Impacts on targeted species – *Trypaea australiensis*

Low numbers of *T. australiensis* were retrieved in samples, probably due to the limited suitability of the coring method for collecting this deep burrowing species.

Consequently, there was a low contribution to any change in the assemblage before and after protection at Currambene Creek tidal flat in the SIMPER analysis for this crustacean. Despite this, a significant decline (Table 4-4) of 60% was measured across the tidal flats, consistent at four of the six sites at the no take location (Fig. 4-7).

T. australiensis was placed in the top 14 taxa that contributed to changes in homogeneity across the tidal flat, using SIMPER analysis of presence/absence data. This was due to a drop in occurrence from 24% to 17% of cores. The changes in *T. australiensis* were very small and statistically unreliable with such low numbers of specimens.

The size frequency distribution for *T. australiensis* sampled with a suction pump randomly across each tidal flat, identified cohorts in the upper size range for the species at all tidal flats (Fig. 4-9). There was an increase in the variance of the size frequency distribution for *T. australiensis* in the second year following protection of the Currambene Creek tidal flat (Fig. 4-10), while the control sites indicated a decrease in variance. However, the scale of this change did not appear to be outside the normal range across all tidal flats, nor thought to be indistinguishable from variation between cohorts, and was therefore not tested.

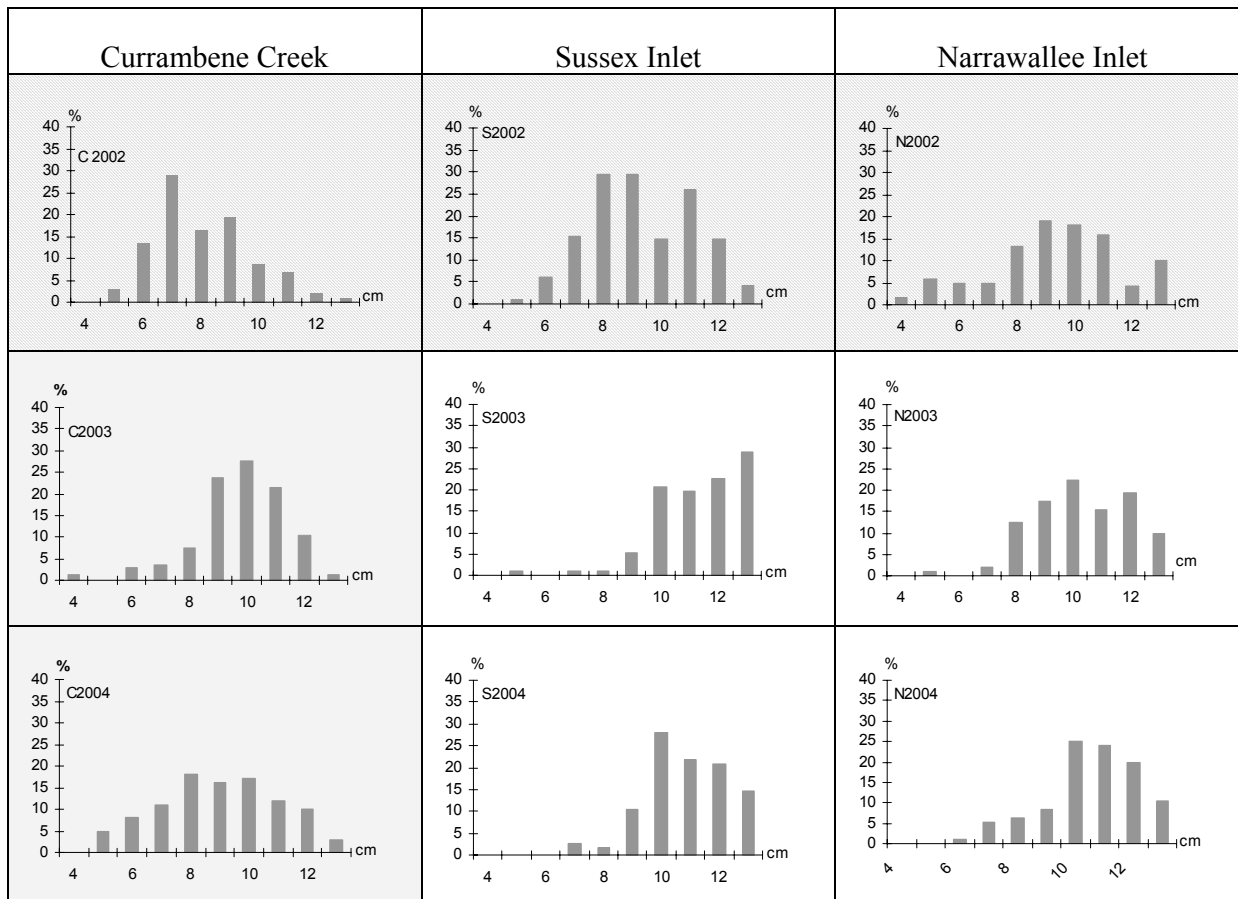


Figure 4-9: The percentage length frequency distribution for *T. australiensis* populations sampled with a suction pump at each of the tidal flats (C = Currambene Creek, S = Sussex Inlet, N = Narrawallee Inlet) before (2003 - hatched) and after (2003-2004) the no-take protection zoning in C (light grey shade). Sample size \cong 100 in each year and tidal flat.

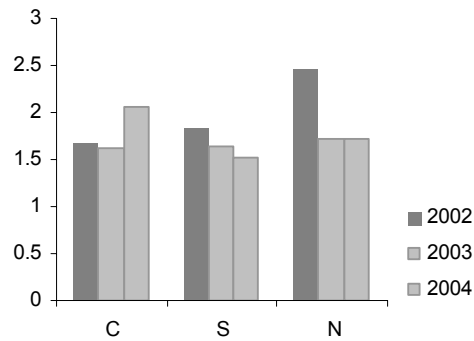


Figure 4-10: The standard deviation of the size distribution frequency of *T. australiensis* in all tidal flats (C = Currambene Creek, S = Sussex Inlet, N = Narrawallee Inlet) before (dark grey) and after (light grey) no-take protection of Currambene Creek.

The number of gravid females was also counted, however these results are not included due to low numbers as the season for gravid females is generally earlier in the year (Cumming and Schreider 2006, Hailstone and Stephenson 1961). However, the ratio of females to males increased in the no-take tidal flat after protection compared to the control tidal flats (Fig. 4-11).

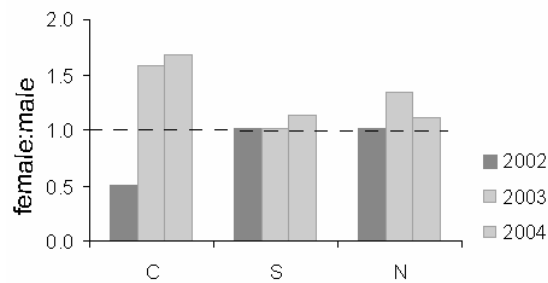


Figure 4-11: The ratio of females:males in each tidal flat (C = Currambene Creek, S = Sussex Inlet, N = Narrawallee Inlet) before (dark grey) and after (light grey) no-take protection of Currambene Creek.

4.3.4. Effects on sediment characteristics

Sediment characteristics showed an increase in finer sediment in the silt and clay fraction after the protection of the Currumbene Creek tidal flat (Table 4-4 and Fig. 4-12) while there was no corresponding change in the control location Narrawallee Inlet. This increase however was only significant and identified at three sites, while other sites were variable. Sussex Inlet did not contain sufficient silt and clay to allow comparison. Organic content showed no consistent pattern of change over time.

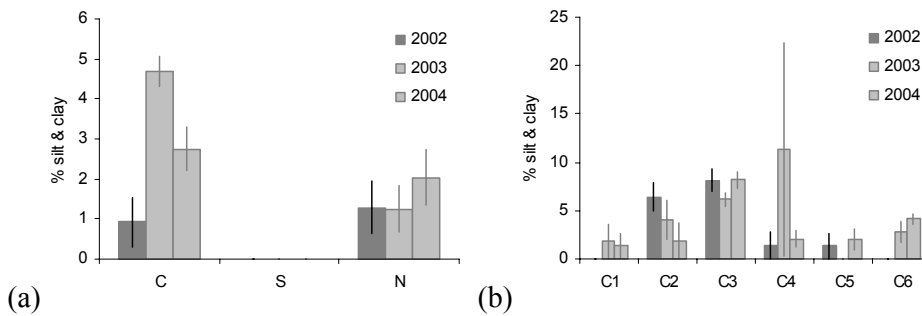


Figure 4-12: Percentage silt and clay content of sediments in (a) each of the tidal flats (C = Currumbene Creek, S = Sussex Inlet, N = Narrawallee Inlet) before (dark grey) and after (light grey) no-take protection of Currumbene Creek, and (b) in each of the sites in Currumbene Creek before and after no-take protection.

4.4. Discussion

The implementation of a no-take Marine Protected Area (MPA) in the Currambene Creek tidal flat was associated with a range of measurable changes to soft-sediment macrofaunal assemblages. Collectively, the changes in the assemblages are indicative of a shift from more mobile taxa, particularly scavengers or predators, to less mobile, smaller, suspension or deposit feeding species. Of note is that these were non-target species, and corresponding changes were not observed in the two control tidal flats. Therefore, it is suggested that non-target, macrofaunal assemblages in Currambene Creek, were impacted by sediment disturbance from bait harvesting activities prior to no-take zoning.

Numerous experimental studies have shown that sediment disturbance has impacts upon tidal-flat assemblages (Ferns et al. 2000, Kaiser et al. 2001, Skilleter et al. 2005, Wynberg and Branch 1994); including Griffiths et al. (2006), who measured differences between no-take tidal flats in a Marine Protected Area in Washington, USA, and tidal flats harvested for clams. However the present study appears to be the first to document a recovery trajectory, using a Beyond BACI model, for estuarine tidal flat fauna following the introduction of a no-take MPA in Australia.

Of importance was that the shift in the tidal flat assemblage differed across the whole tidal flat and was most evident at the scale of sites, 100s m. This could be anticipated considering the smaller scale patchiness of assemblages and species (see Chapter 3.1 or Winberg et al. (2007)), and implies that the effect of bait harvesting (or removal thereof) would be expressed in different ways across the tidal flat. This emphasizes again (see

Chapter 3.1), that for tidal flat biological diversity, no-take zoning requires whole habitat representation. Despite this, there were large scale changes across the whole tidal flat for certain abundant species.

4.4.1 Assemblage effects

A significant change in the whole assemblage at the no-take location was interpreted as a general increase in assemblage homogeneity across the tidal flat. This interpretation is supported by the reduced cumulative dominance of the most common species, and the J'Pielou measure of evenness. This is consistent with work done by Skilleter et al. (2005), where assemblage homogeneity decreased as a result of bait-pumping in tidal-flat experiments.

Despite the overall increase in homogeneity across the tidal flat, changes to assemblage homogeneity were variable at the scale of sites; again, not surprisingly considering the patchiness of the habitat as described above. This also reflects that changes to abundance and homogeneity were specific for different species.

4.4.2 Non-target taxa effects

An immediate and consistent increase in the abundance of certain taxa was measured following the introduction of a no-take zone, across the Currambene Creek tidal flat. The taxa that contributed most consistently to the overall assemblage change were the bivalves, *Eumarcia fumigata* and *Soletellina alba*, as well as small tubular polychaetes (capitellids and spionids). The bivalves were juvenile suspension feeders (Pillay et al. 2007, Schlenk et al. 2007), though *S. alba* is sometimes considered a deposit feeder (Matthews 2006), as are the two polychaete families (Anna Murray personal comment –

Australian Museum). It is suggested that these small, fragile and less mobile taxa, were susceptible to trampling and/or sediment disturbance associated with the bait-harvesting that occurred prior to no-take zoning.

The taxonomic changes observed in this study are consistent with a small but growing number of observational and experimental studies in tidal flat habitats. Small, less mobile and filter feeding species have been found to be negatively impacted by trampling or heavy compression of sediments (Chandrasekara and Frid 1996, Rossi et al. 2007, Schlacher et al. 2007), sediment raking, bait pumping or digging (Ferns et al. 2000, Griffiths et al. 2006, Kaiser et al. 2001, Pillay et al. 2007, Skilleter et al. 2005), dredging (Morello et al. 2005), and boat wash (Bishop and Chapman 2004) and predation disturbance (Posey et al. 1995). In these other studies, it was indicated that deposit feeding polychaetes (eg. capitellidae and spionidae), and small or juvenile gastropods and bivalves, can be affected by smothering, blockage of filtering organs, exposure and physical compaction or damage. One of the bivalves, *S. alba*, is considered particularly brittle (Matthews 2006), and would therefore be susceptible to physical disturbance. In addition, the fast recovery rate of *S. alba* in the present study is consistent with documented fast recruitment and growth rate patterns in response to other natural disturbances (Matthews and Fairweather 2003).

Most of the taxa that increased in abundance also showed an increased homogeneity of distribution across the tidal flat. Of interest however, is that the spionid polychaetes, which had a very patchy distribution prior to no-take zoning, showed an increase in spatial patchiness (heterogeneity) associated with an increase in abundance.

In contrast, some species showed declines in abundance and homogeneity associated with the no-take zoning of Currambene Creek, and might have benefited from the sediment disturbance associated with bait-harvesting. Taxa that showed a decrease in abundance and/or homogeneity were relatively mobile taxa, including scavengers and predators. *Urohaustorius metungi* is a relatively mobile amphipod, albeit at small scales (Anna Murray personal comment – Australian Museum), that might have been resilient to the effects of sediment disturbance, while *Sigalion ovigerum* is a highly mobile, scavenging and predatory polychaete (Read 2004, Serrano et al. 2003).

The decline in these taxa is consistent with studies that suggest that mobile, predatory polychaetes or other scavengers (eg. nemerteans,) might be the first to benefit from post disturbance predation, where organisms are damaged or exposed (Britton and Morton 1994, Eriksson et al. 2005, Morello et al. 2005, Queiros et al. 2006, Reise 1982).

Similarly, shorebirds and predatory fish have been shown to take advantage of exposed prey from sediment disturbance events (Ferns et al. 2000, Serrano et al. 2003). However post-disturbance predation was controlled for in one experiment by Griffiths et al.

(2006), indicating that some species are just more susceptible to disturbance than others.

The strong decline in number of *U. metungi* in the present study indicates that it might have been one of the more resilient species, as it is not a predator or scavenger (Anna Murray personal comment – Australian Museum).

Other taxa that indicated a decrease in abundance, were either within the range of temporal variation at the control sites, and hence not interpreted as an effect of no-take zoning, or were highly site specific within the no-take zone and contradicted trends at other sites. For example, *Mictyris* sp. contributed to the overall assemblage change

following no-take zoning, however the decrease in abundance was negligible compared to variation at the control sites. A decline would also suggest that *Mictyris* sp. was insensitive to bait harvesting, which contradicts a controlled experimental bait-pumping study (Skilleter et al. 2005). In addition, the small bivalve *Mysella anomala* appeared to decrease in abundance following the no-take zoning, however this was mostly associated with one site, and the broader spatial pattern indicated an increase in homogeneity. Therefore a general effect cannot be determined.

Inconclusive changes due to small scale variation should not be discounted, as it emphasizes yet again that effects on assemblages will be variable across heterogeneous habitat (Winberg et al. 2007), and that hierarchical spatial scales are a pre-requisite for effective monitoring.

4.4.3 Targeted species

There was no clear indication as to whether bait-harvesting rates affected the population of *T. australiensis*, the target bait species. This can be attributed to the very low numbers and small specimens retrieved with the coring method used. Estimating the abundance of callianassid species has been attempted before with difficulty, and it was found that neither the number of burrow openings nor coring methods could provide reliable estimates of the population (Rotherham and West 2003, Rotherham and West 2007, Skilleter 2004). As described in Chapter 2, the choice of coring method was the only reliable option for comparable abundances of the other invertebrates in the study, with the unavoidable consequence of reduced sensitivity for abundance measurements of this species. A further consideration is that the abundance of species has recently been shown to be higher in the summer months, and therefore the outcome of a study

might be influenced by this temporal variation (Rotherham and West 2007).

Although effects of bait harvesting on *T. australiensis* are inconclusive for this study, other studies have found that *T. australiensis* is very resilient to harvesting (Hailstone and Stephenson 1961, Skilleter et al. 2005). *T. australiensis* is a highly mobile species, with strong burrowing behaviour, and may be resilient to both sediment disturbance and high harvesting rates if the supply of individuals is limited by the space available (i.e. density-dependent recruitment) (Skilleter 2004). In contrast, other studies have found that callianassid species in South Africa were affected by bait harvesting and took up to 18 months to recover (Wynberg and Branch 1994).

The size frequency of adult cohorts of *T. australiensis* greater than 4cm did not appear to change in the no-take tidal flat compared to control locations. The pumping methods used to measure the size frequency however, did not retain smaller specimens, and use of a small mesh size to capture all cohorts would be an improvement in this methodology in future years.

Of note is the considerable change from an unusually low female to male ratio of *T. australiensis* (Pezzuto (1998) cited in Rotherham and West, (Hailstone and Stephenson 1961, 2003), to an unusually high ratio that was consistent for the two years following protection. The persistence of this pattern for the two years following no-take zoning in Currumbene Creek, and the lack of corresponding pattern at control sites suggests that this could be investigated further. It is worth considering that females may have been targeted over males, however the sex ratios can vary dramatically in sampling (Rotherham and West 2003).

4.4.4 Sediment effects

Because of the link between assemblage composition and silt and clay content (Edgar and Barrett 2002, Hirst and Kilpatrick 2007, Roth and Wilson 1998, Symms and Jones 2004), bait-pumping effects on sediment properties may have indirect consequences for macrofaunal assemblages. Here, an increase in the proportion of finer particles was measured in the sediment following protection of the tidal flat; however this was only associated with two sites. In these sites it is possible that reduced sediment disturbance, and therefore reduced resuspension of fine particles, was a cause for this observation; an assumption that is consistent with physical disturbance effects on sediment stability in tidal flats (Thrush et al. 1996). However sediment attributes are determined by a range of processes and a specific experiment would be needed to qualify this assumption.

Soft-sediment assemblages are also correlated strongly with two other sediment variables, organic material (Kelaher and Levinton 2003) and chlorophyll content (Wynberg and Branch 1994), which may be reduced in disturbed sediments (Pillay et al. 2007). There was no measurable change to organic content after no-take zoning of the tidal flat, however this may not reflect the nutrient status of the tidal flat. Biological processes utilize excess nutrient rapidly in soft sediments (Austen et al. 2002), whereby any increase in productivity is efficiently removed into the food chain (with consequent increases in invertebrate and predator biomass which wasn't measured here). Physical disturbance has been shown to reduce impacts of organic enrichment (Austen and Widdicombe 2006), and in this case where no organic enrichment response followed removal of physical disturbance, fast uptake rates might determine the organic levels in sediments.

4.4.5. Conclusions and Planning and Management considerations for MPAs

This study presents findings to support the statement that no-take areas provide a valuable opportunity to test the implications of conservation planning and management. Recovery trajectories help identify the effects of impacts in coastal habitats, and, related to current knowledge of tidal flat ecology, provides information relevant to both ecological research and management of MPAs. The response time for change in soft-sediment invertebrate assemblages is short, implying that monitoring programs using soft-sediment invertebrates in MPAs should be prioritized in order to get “before” data and that small invertebrate taxa may provide evidence of the most immediate effects of protection

The findings presented here are among the first to document a recovery trajectory for estuarine tidal flat habitat following the introduction of a no-take Marine Protected Area. Although numerous experimental studies have shown that physical sediment disturbance impacts invertebrate soft-sediment fauna (Ferns et al. 2000, Kaiser et al. 2001, Skilleter et al. 2005, Wynberg and Branch 1994), the effects and consequences of this are rarely put into a context of relevance for marine protected area management.

First, a clear impact on non-target species from bait harvesting is consistent with an independent study in very similar conditions (Skilleter et al. 2005), and supported by a range of ecological studies. This is in-line with studies that demonstrate indirect harvesting effects either through disturbance (Casu et al. 2006, Contessa and Bird 2004) or trophic interactions (Pinnegar et al. 2000, Shears and Babcock 2002) in marine systems, and questions the relevance of claims that broadly assume negligible impacts

to non-targeted species (Tetreault and Ambrose 2007).

The assemblage shift measured in the present study is predicted to have consequences for a range of ecosystem services including nutrient processes, trophic energy flow, inter and intraspecific interactions, recruitment of vertebrate and invertebrate fauna (Austen et al. 2002). Of particular interest will be what predatory species target the fauna that increased in abundance. For example, it has been shown that Silver bream (*Acanthopagrus sp.*) consume the bivalve *Soletellina alba* (Matthews and Fairweather 2003) in Victorian estuaries.

Of note is that the recovery tangents operated primarily at the scale of 100s m (sites) within a tidal flat; the scale previously determined to represent the highest variability of assemblages. This finding emphasizes yet again (see Chapter 3.1), the need to represent the diversity within whole habitats in MPAs. It also suggests that differences in the structuring processes in a habitat might alter the sensitivity and resilience of habitat to disturbance. Similarly, Ferns et al. (2000) found that assemblages in muddier sites were more sensitive to disturbance from inter-tidal cockle raking than those in sandier sites. The apparent sensitivity of finer particle soft-sediment habitat (Hiddink et al. 2007, Queiros et al. 2006) is clearly an important consideration for management, and one which can be easily measured.

The anticipated fast response to no-take zoning, given the short reproductive and recruitment cycles for many of the small invertebrates sampled (eg. *S. alba* (Matthews and Fairweather 2006)), was confirmed. Recovery was measured in the year immediately following no-take zoning and was maintained into the second year, although the baseline condition for tidal flat habitats is rarely known. This is

important to note for future monitoring programs where data from before no-take zoning will be of greater importance for soft-sediment invertebrates than for that of assemblages (e.g. fish) with longer life histories.

Fast recovery indicates that the resilience of tidal flat fauna is high (Kaiser et al. 2001, Skilleter et al. 2006) and that physical impacts in the short term have negligible long term effects. Of importance here is that the consistent nature of human harvesting in time and space implies that it can be regarded as a press impact without a recovery period; an assumption supported by the consistent assemblage shift measured in the present study. Considering the need for long-term unaltered sites to provide references or baselines, the assumption of increased resilience in no-take MPAs (Bevilacqua et al. 2006, McClanahan et al. 2002), as well as the importance of soft-sediment habitats to ecosystem services (including fishing) (Thrush and Dayton 2002), tidal flats should be considered an important contribution to comprehensive and adequate MPAs.

Shallow soft-sediment changes such as those measured in this study, will have functional consequences, including effects on ecosystem services that are important at the scale of a global MPA network (Jorgensen and Fath 2006). Therefore, further research and monitoring effort is required in shallow soft sediments, specifically for connectivity to other assemblages and habitats through trophic cascades and life histories.

CHAPTER 5 - ANALYSIS OF MULTIVARIATE AND UNIVARIATE DATA FOR IMPACT STUDIES



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5.1. Extending asymmetrical analysis of variance in Beyond BACI to better detect environmental impacts

5.5.1. Introduction

It is well established that robust experimental design is paramount for the investigation of environmental impacts, and beyond BACI design is widely accepted and applied to real world impact studies (Benedetti-Cecchi 2001, Hewitt et al. 2001, Underwood 1991, Underwood 1992, Underwood 1993, Underwood 1994). In contrast to small scale manipulative experiments, tests for large scale impacts in the real world are often subject to sub-optimal experimental design. In particular, spatial asymmetry between control and impacts sites (unbalanced data sets) are typical for real world ecological studies on impacts, for example from pollution (Roberts et al. 1998), and more recently, monitoring for ecological changes following habitat protection in Marine Protected Areas (MPAs) (Fraschetti et al. 2002).

Spatial asymmetry is a feature of real world studies where the treatment or impact location cannot be replicated adequately. For example, Roberts et al. (1998) studied the effects of a sewerage outfall on marine fauna and flora, a situation where the impact location is hard to replicate unless another sewage outfall comes online simultaneously. In contrast to impact sites, it is possible to replicate control locations (Underwood 1992, Underwood 1994); as Roberts et al. (1998) and numerous other studies have done so (Guidetti 2002, Terlizzi et al. 2005). Thus, the experimental design is spatially asymmetrical with more control than impact locations, undermining the use of a straight forward balanced Analysis of Variance (ANOVA) (Shaw and Mitchell-Olds 1993).

Analysis of spatially asymmetric data however, can be accommodated without loss of power, by using asymmetrical ANOVA for univariate measures (Underwood 1993).

Similar to spatial asymmetry, temporal asymmetry is also typical of real world studies, where the time available to collect data before an impact occurs is limited, but the time to collect data after the impact is not (Claudet et al. 2006, Guidetti 2002, Roberts et al. 1998). Therefore there is typically one time period of sampling before an impact, and two or more times after. For data that are simultaneously asymmetric for spatial and temporal factors, analysis usually accommodates the spatial asymmetry in the full data set, but the interaction with Before versus After is tested in two separate tests for each year after the impact (Roberts et al. 1998). In many cases, the use of two tests with different years of After data may suffice to detect a change in the Impact location that differs from any change in multiple Control locations; however the full data set is not used in any test, and therefore power is lost in the analysis. A method that incorporates the full data set in an ANOVA, with both spatial and temporal asymmetry combined, could not be found in the ecological literature.

In the present study, it is demonstrated how temporal asymmetry can be accommodated simultaneously with spatial asymmetry, thereby taking advantage of the power in the full data set. The method is an extension of the method of asymmetrical analysis of variance described by Underwood (1993) and widely applied (Fraschetti et al. 2002, Guidetti 2002, Hewitt et al. 2001, Queiroz et al. 2006, Terlizzi et al. 2005) for detecting effects of ecological impacts. A hypothetical model is used to illustrate the method, and a data set from a real case study is used to demonstrate an application and outcome.

5.1.2. Methods

5.1.2.1. Accommodation of spatial and temporal asymmetry in experimental design

Spatial asymmetry is accommodated using the methods described by Underwood (1993), and illustrated here as a simple design of 3 locations (2 Control and 1 Impact) orthogonal to 2 temporally symmetric periods before and after an impact. The Sums of Squares for the symmetric data set (all locations and years) are calculated, and the Sums of Squares from a symmetrical sub-set of data (Control sites only), is subtracted from the full data set. The difference between the Sums of Squares for the full data and the control data sub-set of data represents the spatially asymmetrical factor of the data set (Control versus Impact) (Underwood 1993). A test can now be constructed for the interaction between Control versus Impact sites, Before versus After an Impact: BvsA x CvsI (Fig. 5-1).

Spatial asymmetry is accommodated by partitioning the Location sums of squares into Control sites (C1 & C2) and comparing them to the Impact site (I) sums of squares. The Impact, indicated with an arrow, occurs in a temporally balanced design, Before (B1 & B2) and After (A1 and A2). The interaction factor of interest is CvsI x BvsA. In this example there is also a nested temporal factor that has the interactions CvsI x Time(B) and CvsI x Time(A).

If temporal and spatial asymmetry is inevitable, then the analysis to detect change that is commonly applied, accommodates spatial asymmetry as above, but two tests are used to accommodate temporal asymmetry (see Roberts et al. (1998)). First the interaction of spatially asymmetrical data (CvsI) is tested with the year of Before data versus one year

of After data (BvsA1), and the second test uses the second year of After data (BvsA2) (Fig. 5-2). Thus there are two tests of interaction, CvsI x BvsA1, and CvsI x BvsA2; each test with reduced sample size, and thus degrees of freedom, compared to the full set of After data.

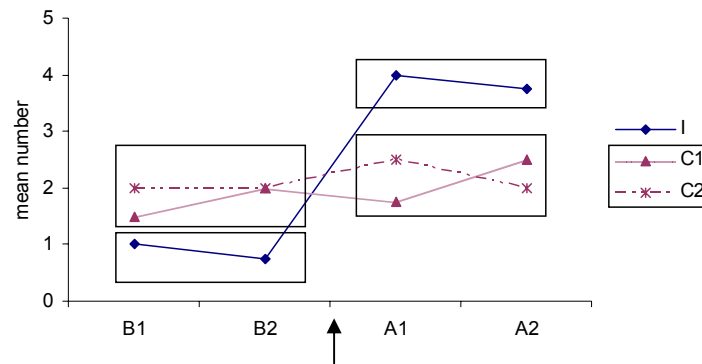


Figure 5-1. Spatial asymmetry is accommodated by partitioning the Location sums of squares into Control sites (C1 & C2) and comparing them to the Impact site (I) sums of squares. The Impact, indicated with an arrow, occurs in a temporally balanced design, Before (B1 & B2) and After (A1 and A2). The interaction factor of interest is CvsI x BvsA. In this example there is also a nested temporal factor that has the interactions CvsI x Time(B) and CvsI x Time(A).

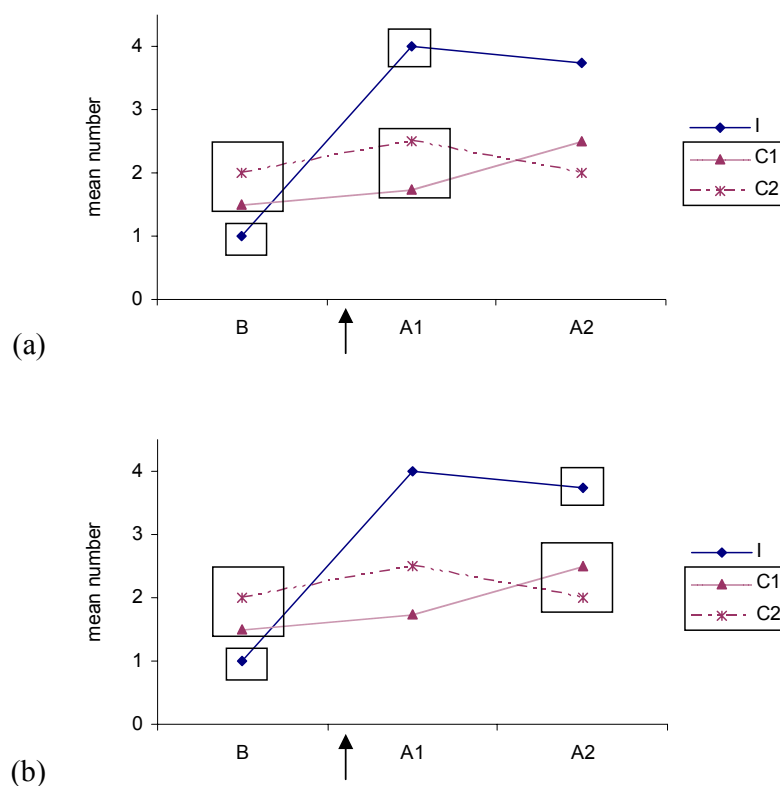


Figure 5-2. Spatial asymmetry is accommodated by partitioning the Location Sums of Squares into Control sites (C1 & C2) versus the Impact site (I). To balance the two years of After data against one year of Before data, two tests are done using each of the two years of after data separately. Thus the two tests of interaction are (a) CvsI x BvsA1 and (b) CvsI x BvsA2. The time of impact is indicated with an arrow.

In the extension to asymmetrical ANOVA in this present study, the logic described by Underwood (1993) for spatial partitioning of sums of squares into an asymmetric factor, Control versus Impact, is followed; then the method is extended to also partition Years into the asymmetrical factor Before versus After. Thus temporal and spatial asymmetry can both be incorporated into the one test, BvsA x CvsI (Fig. 5-3).

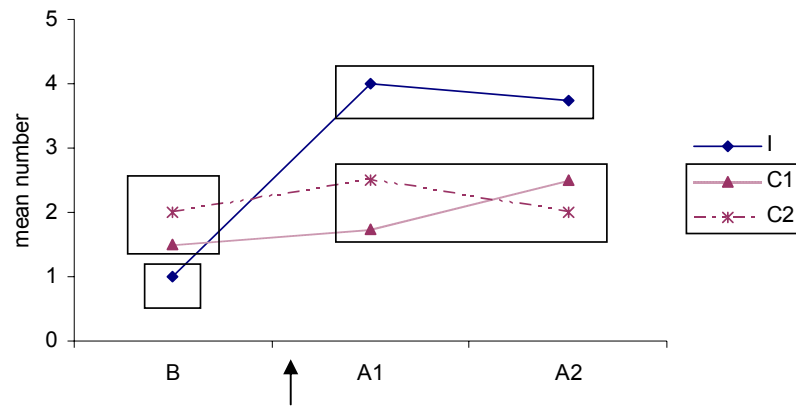


Figure 5-3. Spatial asymmetry is accommodated by partitioning the Sums of Squares for into Control sites (C1 & C2) versus Impact (I), and temporal asymmetry is similarly accommodated by partitioning sums of squares for years into After (A1 & A2) versus Before (B). Thus the test for interaction uses the full data set and is $CvsI \times BvsA$. The time of impact is indicated with an arrow.

In order to calculate sums of squares and degrees of freedom simultaneously for a given spatially and temporally asymmetric factor interaction, first, the spatially asymmetric sums of squares (S.S.) and degrees of freedom (D.F.) are partitioned as described above and in Underwood (1993), using the full data set and subtracting sums of squares from the Control data set. Then the sums of squares calculation is extended further to provide S.S. and D.F. for the temporally asymmetrical factor, Before versus After, by similarly subtracting sums of squares from the After data set from the full data set. Finally, an additional fourth balanced ANOVA is required, using only After data at the Control sites to recover the sums of squares common to all sets of data, and that was therefore

subtracted twice. Thus the remaining sums of squares represents only the factor Before vs. After x Control vs. Impact (Table 5-1).

For conceptual purposes, the method of accommodating temporal and spatial asymmetry in one factor is shown here in a model with a spatial hierarchy of 2 factors, sites nested in locations, orthogonal to the factor years. Thus a nested interaction at the smaller scale of sites is also of interest and can be calculated: Before vs. After x Sites(Impact) (Table 5-1). The factors Years and Locations are considered fixed because they will be categorized into Before versus After, and Control versus Impact respectively, while the nested factor Sites is random. This follows the logic described in Kingsford (1998) for impact studies using fixed main factors (Before vs. After and Control vs. Impact).

Means squares are subsequently calculated as usual by dividing the sums of squares by the degrees of freedom. In order to obtain the appropriate F-ratio to test the asymmetrical factors of interest (Table 5-2), denominator mean squares are chosen from factors in sequence depending on which denominators can be pooled ($p \geq 0.25$), following the logic described in Underwood (1993) and Terlizzi et al. (2005). For example, it is primarily of interest to determine if the mean square for BvsA x sites(I) is significantly different from the mean square for BvsA x sites(C); unless the latter interaction is not significant ($p \geq 0.25$) and the next spatial hierarchy or residual factor mean square is used. Two-tailed F ratios are then tested using the relevant degrees of freedom. This can be done manually with the aid of an F-test calculator or tables. It should be noted that different designs using different fixed and random factors will affect the choices of mean square denominators for the F-tests (Table 5-2).

Table 5-1. Representation of the quantities of sums of squares for interactions for main and nested factors for (a) the full data set (b) after data only, (c) control data only, and (d) after control data only. The calculations in the right hand column provide a full asymmetric [†]interaction factor represented by the bold outlined sources of sums of squares. This is an extension to the asymmetrical ANOVA, previously only categorized to ^{*}Years x CvsI, or done using two tests for each year of after data. The nested factors table illustrates the same logic as for the main factors; however, the CvsI factor is not relevant as the comparisons are made within each of the Control or Impact locations, not between them. Abbreviations are given as B = Before, A = After, C = Control, I = Impact, S.S. = sums of squares, D.F. = degrees of freedom, Loc = Locations.

	(a) All			(b) After			(c) Control			(d) After Control			Calculate S.S. & D.F. for interactions
	Years x Locations			After x Locations			Years x Controls			After x Controls			
Main	C1vsC2	C2vsl	C1vsl	C1vsC2	C2vsl	C1vsl	C1vsC2	C2vsl	C1vsl	C1vsC2	C2vsl	C1vsl	BvsA x Loc = a-b
	BvsA1	BvsA1	BvsA1				BvsA1						* Years x Cvsl = a-c
	BvsA2	BvsA2	BvsA2				BvsA2						BvsA x Controls = c-d
	A1vsA2	A1vsA2	A1vsA2	A1vsA2	A1vsA2	A1vsA2	A1vsA2			A1vsA2			†BvsA x Cvsl =(a-b)-(c-d)
	Years x Times(Loc)			After x Times(Loc)			Years x Times(Controls)			After x Times(Controls)			
Nested	C1	C2	I	C1	C2	I	C1	C2	I	C1	C2	I	BvsA x Sites(Loc) = a-b
	BvsA1	BvsA1	BvsA1				BvsA1	BvsA1					* Years x Sites (Impact) = a-c
	BvsA2	BvsA2	BvsA2				BvsA2	BvsA2					BvsA x Sites (Controls) = c-d
	A1vsA2	A1vsA2	A1vsA2	A1vsA2	A1vsA2	A1vsA2	A1vsA2	A1vsA2		A1vsA2	A1vsA2		†BvsA x Sites(Impact) =(a-b)-(c-d)

Table 5-2: A fully balanced hypothetical model with spatial hierarchy (sites(Locations)) and orthogonal (Years) factors is shown under data set (a). Asymmetrical factors are calculated from data sets a-d (Source column). The interactions of interest are in bold, cursive text, and the sequence of denominator mean squares to be chosen for tests of F-ratios is given. If the mean square for the first denominator in the sequence is not significant at $p \geq 0.25$, then the factor can be pooled and the denominator sequence is followed.

Source	Data Set				Denominator sequence for F-ratio
	(a)	(b)	(c)	(d)	
a1	Ye				
b1		After			
a1-b1		Bvs.A			
a2	Loc				
c1			Controls		
a2-c1			Cvs.I		
a3	Si(Loc)				
a5	YexLoc				
b2		AfterxLoc			
a5-b2		Bvs.A x Loc			AfterxLoc, YexSi(Loc), Residual
c2			YexControls		
a5-c2			YexCvs.I		YexControls, YexSi(Loc), Residual
d1			AfterxControls		
b2-d1			After x Cvs.I		AfterxControls, AfterxSi(Loc), Residual(After)
c2-d1			Bvs.A xControls		AfterxControls, YexSi(Controls), Residual(Controls)
(a5-b2)- (c2-d1)			Bvs.A x Cvs.I		Bvs.A xControl, After x Cvs.I, YexSi(Loc), Residual
a6	YexSi(Loc)				
b3		AfterxSi(Loc)			
a6-b3		Bvs.A x Si(Loc)			AfterxSi(Loc), Residual
c3			YexSi(Controls)		
a6-c3			YexSi(Impact)		YexSi(Controls), Residual(Impact)
d2			AfterxSi(Controls)		
b3-d2			AfterxSi(Impact)		AfterxSi(Controls), Residual(After_impact)
c3-d2			Bvs.A xSi(Controls)		AfterxSi(Controls), Residual(Controls)
(a6-b3)- (c3-d2)			Bvs.A xSi(Impact)		Bvs.A xSi(Controls), AfterxSi(Impact), Residual(Impact)
a7	Residual				
b4		Residual(After)			
c4			Residual(Controls)		
d3			Residual(After_Controls)		
a7-b4		Residual(Before)			
a7-c4			Residual(Impact)		
a7-d3			Residual(After_impact)		

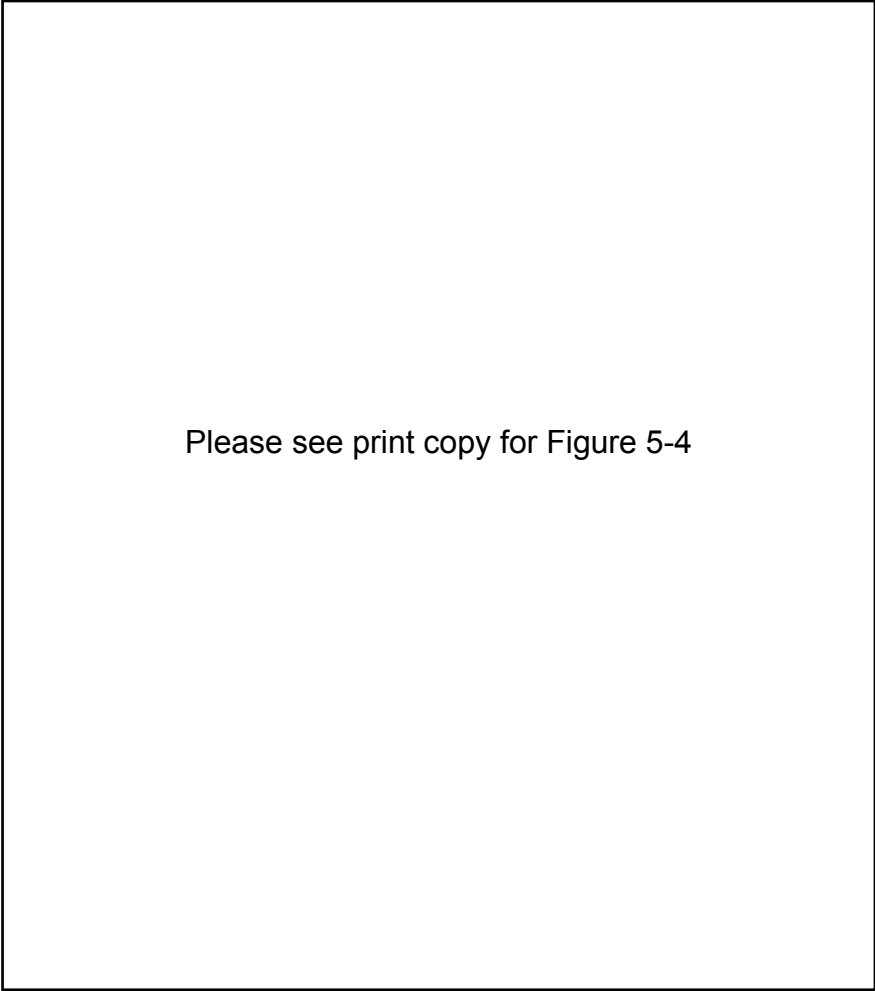
5.1.2.2. Case Study

This case study uses one univariate measure (species richness) from a Beyond BACI experimental design to test for assemblage change following the implementation of Marine Protected Area status (see Chapter 4). The introduction of an MPA no-take zone was considered analogous to an impact study that might affect the faunal assemblage, however in this case assemblage change was monitored after the impact was removed rather than imposed. The first year of data was collected in 2002 before Marine Protected Area status, and then for the following 2 years after protection. There were 2 Control locations and one no-take (Impact) location that were sampled at each of 2 times within each year. At each time, 3 sites were sampled within each location, and a smaller scale of 3 plots was nested within sites. In the case of this data set, mean species richness between times within each year and tidal flat was not significant ($p \geq 0.25$) and was pooled, thus $n=6$ for Sites. Therefore this design was a spatially nested hierarchy of 3 factors (3 Locations (6 Sites (3 Plots))) orthogonal to Years ($n=3$) (Figure 5-4a).

An overview of the full analysis method used to calculate a spatially and temporally asymmetrical ANOVA is provided here, while an illustration of the spatial sampling design used in each year is provided in Chapter 3.1 (Fig. 5-1), and application of this method to multiple univariate variables is given in Chapter 4. The symmetrical model for the full data set includes three years and three locations (Fig. 5-4a). The symmetrical Location sums of squares are partitioned to accommodate the spatial asymmetry of 1 Control versus 2 Impact locations (Cvs.I) (Fig. 5-4b). Further partitioning of sums of squares forms the basis of this chapter, where Years are partitioned to the asymmetrical

1 Before versus 2 After period of time (Bvs.A), resulting in four asymmetrical groupings of the replicate samples (Fig. 5-4c).

The main and nested component sums of squares for the interactions Bvs.A x Cvs.I are calculated as explained above (Section 5.1.2.2.) by undertaking four separate balanced ANOVAs using all, control, after and after-control data sets (Table 5-3). The F ratios for the fully asymmetrical factors of interest are calculated using mean square denominators in order of sequence according to whether they can be pooled at $p \geq 0.25$ (as in Table 5-2).



Please see print copy for Figure 5-4

Figure 5-4: Sampling and analysis designs: (a) symmetrical Years (3) and Locations (3); (b) symmetrical Years (3) and asymmetrical Location (2 Control versus 1 Impact); (c) asymmetrical Years (1 Before versus 2 After) and asymmetrical Location (2 Control versus 1 Impact). Letters indicate fixed factors and numbers random factors in (a), while (b) and (c) use B = Before, A = After, C = Control sites, I = Impact site. This illustration technique follows the logic of Benedetti-Cecchi (2001).

Table 5-3: Procedure for calculating sums of squares and degrees of freedom for asymmetrical factors (shaded cells) of the full model (Figure 5-4c). Calculations are based on four symmetric ANOVAs using different components (a-d) of the data set.

Sums of squares	(a) All data	(b) After data	(c) Control data	(d) After/Control data
a1	Ye			
b1		After		
a1-b1		Bvs.A		
a2	Loc			
c1			Controls	
a2-c1			Cvs.I	
a3	Ti(Loc)			
a4	Si(Ti(Loc))			
a5	Ye x Loc			
b2		AfterxLoc		
a5-b2		Bvs.A x Loc		
c2			YexControls	
a5-c2			YexCvs.I	
d1				AfterxControls
b2-d1				After x Cvs.I
c2-d1				Bvs.A x Controls
(a5-b2)-(c2-d1)				Bvs.A x Cvs.I
a6	Ye x Si(Loc)			
b3		After x Si(Loc)		
a6-b3		Bvs.A x Si(Loc)		
c3			Ye x Si(Controls)	
a6-c3			Ye x Si(Impact)	
d2				After x Si(Controls)
b3-d2				After x Si(Impact)
c3-d2				Bvs.A x Si(Controls)
(a6-b3)-(c3-d2)				Bvs.A x Si(Impact)
a7	Ye x PI(Si(Loc))			
b4		After x PI(Si(Loc))		
a7-b4		Bvs.A x PI(Si(Loc))		
c4			YE x PI(Si(Controls))	
a7-c4			YE x PI(Si(Impact))	
d3				After x PI(Si(Controls))
b4-d3				After x PI(Si(Impact))
c4-d3				Bvs.A x PI(Si(Controls))
(a7-b4)-(c4-d3)				Bvs.A x PI(Si(Impact))
a8	Residual			
b5		Residual(After)		
c5			Residual(Controls)	
d4				Residual(After_Controls)
a8-b5		Residual(Before)		
a8-c5			Residual(Impact)	
a8-d4				Residual(After_impact)

5.1.3. Case study results

There was no difference between the species richness of the assemblage at the Impact site compared to the Control sites at the scale of locations using the factor Bvs.A x Cvs.I ($p = 0.89$). This may be due to the nested variation between sites ($p = 0.01$) when the mean square of Bvs.A x Si(Impact) was tested over Bvs.A x Si(Controls) (Table 5-4).

At the scale of sites however, the use of incorporating temporal asymmetry affected the results compared to just incorporating spatial asymmetry (i.e. Bvs.A x Si(Impact) would tested over the mean square Bvs.A x Si(Controls)), in which case no significant interaction was detected ($p = 0.12$) prompting no subsequent pair wise tests. Although it can be debated that the arbitrary α level of $p = 0.05$ or even $p = 0.10$ in environmental impact studies is appropriate (Mapstone 1995), it is clear that the expansion of the temporal factor to the asymmetrical Before versus After creates a test that recognises the structure of the interaction of interest. Alternately, it facilitates the use of the full data set compared to studies where 2 partial data sets are analysed separately with less information on biological variability in each.

There was no significant effect for the interaction at the smallest scale of plots.

Therefore, the only detectable effect from the protection of the tidal flat was at the scale of sites between estuaries, when the temporal factor was asymmetrically partitioned to Before versus After. The mean species richness across sites increased at four sites following no-take protection, while there was a decrease at site C5 and high variability at C3 (Fig. 5-5a). No such trends were evident at the control sites (Fig. 5b-c)

Table 5-4. Asymmetrical analysis of variance calculated from 4 symmetrical ANOVAS (a – d) using sections of and a full data set of species richness. The factors of interest for extension of the asymmetrical analysis to include a BvsA component are highlighted in blue. Mean square denominators used for F tests are are highlighted in yellow, and factors with an asterisk* are tested with a 2-tail F test.

Source	a	b	c	d	df	SS	MS	den.1	F	P	den.2	F	P	den. 3	F	P	den. 4	F	P	den. 5	F	P	den. 6	F	P	den. 7	F	P
a1	Year = Ye				2	0.19	0.10		0.17	0.84																		
b1	After = A				1	0.11	0.11		0.23	0.64																		
a1-b1	Bvs.A				1	0.08	0.08	b1*	0.69	0.56	a5	0.19	0.69	a6	0.14	0.71	a7	0.28	0.60	a8	0.81	0.37						
						43.8	21.9																					
a2	Estuaries = Est				2	6	3		8.94	0.00																		
c1	Controls				1	2.24	2.24		1.13	0.31																		
						41.6	41.6		18.5			16.9									75.4							
a2-c1	Cvs.I				1	1	1	c1*	5	0.15	a3	6	0.00	a4	107.67	0.00	a5	101.01	0.00	a6	4	0.00	a7	150.72	0.00	a8	431.24	0.00
						36.8																						
a3	Sites = Si(Est)				15	1	2.45		6.35	0.00																		
						13.9																						
a4	Plots = Pl(Si(Est))				36	2	0.39		4.00	0.00																		
a5	Ye x Est				4	1.65	0.41		0.75	0.57																		
b2	A x Est				2	1.52	0.76		1.57	0.24																		
a5-b2	Bvs.A x Est				2	0.13	0.06	b2*	0.09	0.16	a6	0.12	0.89	a7	0.23	0.80	a8	0.67	0.51									
c2	Ye x Controls				2	0.38	0.19		0.49	0.62																		
a5-c2	Ye x Cvs.I				2	1.27	0.63	c2*	3.33	0.46	a6	1.15	0.33	a7	2.30	0.11	a8	6.57	0.00									
d1	A x Controls				1	0.36	0.36		0.62	0.45																		
b2-d1	A x Cvs.I				1	1.16	1.16	b3	2.39	0.37	b4	3.68	0.08	b5	11.76	0.00												
c2-d1	Bvs.A x Controls				1	0.02	0.02	d1*	0.05	0.28	c3	0.05	0.83	c4	0.06	0.81	c5	0.23	0.63									
(a5-b2)-(c2-d1)	Bvs.A x Cvs.I				1	0.11	0.11	c2-d1*	5.64	0.50	b2-d1*	0.10	0.40	a6	0.20	0.89	a7	0.40	0.53	a8	1.14	0.42						
						16.5																						
a6	Ye x Si(Est)				30	5	0.55		2.00	0.01																		

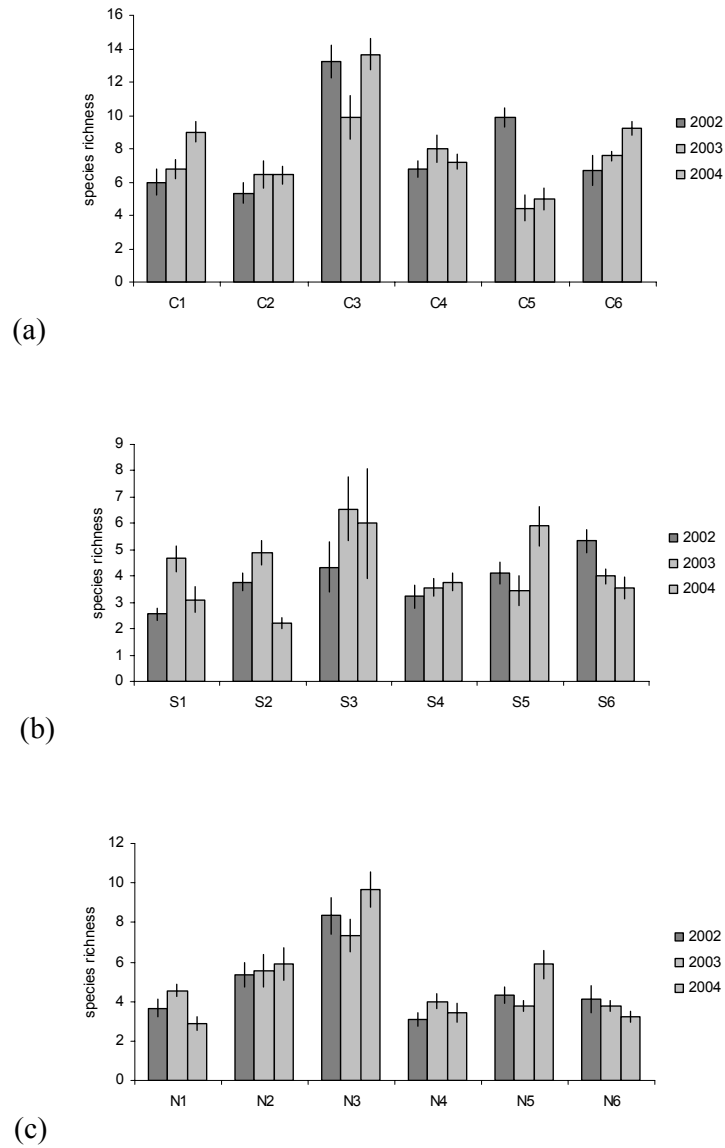


Figure 5-5. Mean species richness in all sites within each estuary measured in 2002 (Before), 2003 and 2004 (After). Sites C1-6 are from the no-take (Impact) tidal flat in Currumbene Creek, while sites S1-6 and N1-6 are from the Control tidal flats in Sussex Inlet and Narrawallee Inlet respectively. Standard error bars are shown.

5.1.3. Discussion

Here I have demonstrated that an asymmetrical analysis of variance can be constructed with asymmetry in both temporal (eg. Before $n = 1$, After $n = 2$) and spatial (Impact = 1, Control = 2) factors, and therefore the interaction between asymmetrical components, specifically Bvs.A x Cvs.I, can be tested using the full data set. In doing so, the effect of no-take zoning was statistically significant, whereas this would not have been detected using data without partitioning the temporal factor Years, to an asymmetrical BvsA factor. The strength of this method lies in retaining the full sample size, and thus degrees of freedom, in the test.

There is a strong need to develop robust scientific methodology that makes use of the complete data set, recognizing the two dimensions of asymmetry simultaneously, in order to maximize the sensitivity to detect impacts. This is particularly relevant not only for monitoring of environmental impacts, but also for monitoring programs in the expanding number of Marine Protected Areas on a global scale. For MPAs, scientific evidence is used to justify environmental management decisions that impact on socio-economics and politics, therefore tests to measure change must be highly robust.

Commonly, studies to test for changes in MPAs may not include any temporal data at all and After Control Impact (Beyond ACI) designs are used where the differences between control and impact sites are tested for. Such studies are criticized for the lack of before data required to demonstrate a causal effect (Queiroz et al. 2006), thus science used to justify the implementation of MPAs can be criticised. Although there have been some novel approaches using asymmetrical designs to overcome a lack of Before data (Queiroz et al. 2006), it is considered important that managers make a conscious

effort to provide before data in order to test the effects of MPAs (Fraschetti et al. 2002, Guidetti 2002).

Therefore, with the expanding number of Marine Reserves on a global scale, the inevitable situation of spatially and temporally asymmetric designs to detect ecological effects of Marine Reserves can be expected to increase. Experimental design will vary in each circumstance, in particular the nested and orthogonal factors chosen may be more optimal than that described here; however, the ability to use all of the available data in tests will increase the power of the analysis.

5.2 Maximising bang for your buck: effects of taxonomic resolution, data transformation, and the use of assemblage sub-sets for detecting impacts in tidal flats

5.2.1. Introduction

No ecological study can ever encompass the entire range of biological (or environmental) variables that might be relevant to measure within the framework of an impact study. This would require sampling and analysis of variability at the scales of intraspecific genetics to large visible organisms and whole ecosystem processes. Consequently, ecological sampling and analysis is usually dictated by available expertise, time, resources and the question at hand.

Regularly, marine impact studies measure impacts on invertebrate assemblages because they are abundant, widespread, relatively easy to sample in high numbers and can potentially respond to change over short time-scales. In addition, assemblage data are often more informative than single species data for ecological studies (Peterson 1993, Ward 2000, Wilson 2008), especially if the potential effects of an impact are not known. Effectively, invertebrate assemblages are used as indicators for habitat impacts or measuring patterns of biodiversity for a broader range of taxa (Giangrande et al. 2005, Jones and Kaly 1996).

Assuming that invertebrates are a valuable and valid indicator of ecosystem impacts or ecological diversity, there are three methodological decisions that are always made that can affect the analysis and interpretation of the assemblage patterns to be measured. These are selection of species subsets from assemblages, taxonomic resolution, and

multivariate data transformation, and will be referred to here as data manipulations.

Selection of species subsets to represent assemblages, occurs at the outset of sampling programs as an artefact of sampling methods (eg. mesh size (James et al. 1995, Lampadariou et al. 2005)). Further assemblage subsets may also be selected, for example, if there is limited taxonomic expertise, or to reduce variation in assemblage data by removing rarer species (Warwick et al. 2006). In addition, indicator species are often sought to simplify monitoring programs and are a type of assemblage subset (for example in Ward (2000)).

A second type of data manipulation, taxonomic resolution, can range from species through to phyla, or at an extreme, to kingdom. The reasons behind these choices are often related to the cost of sampling programs, or again due to limited access to taxonomic expertise.

Further to selecting assemblage subsets or reducing the taxonomic resolution, transformations of multivariate data are readily accepted as standard practice to decrease the influence of highly abundant species on analysis of assemblage patterns, relative to rarer species. At the extremes, untransformed data measures patterns of relative abundances of species, while harsh transformation such as presence/absence of taxa provides information on patterns of assemblage composition (Clarke and Warwick 2001).

Despite potential effects of data manipulations on the outcomes of research and monitoring programs, choices of manipulations are regularly made in ecological studies based on accepted defaults or current practice. For example, in the case of

transformations, square root reduces the influence of dominant species which is commonly accepted as appropriate (Curley et al. 2002, Roberts and Connell 2008). In contrast, the effects of different taxonomic resolutions on analysis and interpretation has been discussed at length, in different habitats and for different assemblages, and has recently been reviewed for marine rocky and sub-tidal sediment habitat by Terlizzi et al. (2003). Fewer, however, have addressed the implications of selection of species subgroups or transformation of multivariate data, nor combinations of these choices and the effects of interactions. Only recently in the literature are such comparisons made, for example, Karakassis and Hatziyanni (2000) and Quijon and Snelgrove (2006) compare taxonomic resolution effects using different data transformation. Currently in press, is a comparison of the three methodological decisions for macroalgal assemblages (Puente and Juanes (in press)).

Greater consideration of the effects of different data set formats can benefit the sensitivity of analysis to detect changes, describe ecological patterns, as well as enhance the efficiency of ecological studies. Here it is recognised that some projects are restricted by default to certain applications as the data may already be collected, or sampling methods between sites are not comparable (i.e. presence-absence data only). However, the importance of detecting ecological impacts cannot be understated, and awareness of the implications and interpretations that can be made using different methodologies is important. In addition, efficiencies in ecological studies can be gained by understanding the minimum effort required to measure biological patterns or ecological impacts.

Of note is that ecological data analysis software that is user friendly is readily available

and applied to a range of descriptive and impact studies in ecology. This is desirable for two reasons, studies can be compared more readily both within and between habitat and assemblage types, and more detailed analysis can be achieved in less time. Use of these analysis tools assumes that the user knows what they are doing, but today's software incorporates procedures in intuitively simpler contexts (but with sometimes inappropriate defaults), thereby facilitating a better understanding for more ecologists than has been the case in the past. The speed and breadth of analysis tools implies that it is easier to evaluate the effects and limitations of different data manipulations, and this in itself increases our understanding of ecology and what patterns exist.

Here, the effects of various data manipulations on the analysis and interpretation of tidal flat invertebrates assemblage data are demonstrated, using an assemblage data set from an impact study in a soft-sediment tidal flat in NSW, Australia (Chapter 4). Up to 46 data manipulations of the raw data set were used to

- a) determine the relative effects of assemblage subsets, reduction in taxonomic resolution and multivariate transformations on detecting general biological patterns,
- b) determine the interaction effects of data transformation with taxonomic resolution and species subsets,
- c) demonstrate and extend a cost and benefit analysis, comparing selection of assemblage sub-sets and reduction in taxonomic resolution, and
- d) demonstrate the effect of the various data manipulations on the sensitivity to

measure an impact using a Beyond BACI experimental design, and extension to procedures described by Clarke and Warwick (2001)

5.2.2. Methods

5.2.2.1. Assemblage data source

The data set used was part of a study to detect assemblage change as an effect of exclusion of bait harvesting in a Marine Protected Area (MPA) (Chapter 4). Samples of macrofauna were collected using cores of soft sediment, from 3 tidal flats over three years as described in Chapter 4. Macrofauna were identified to species level resolution for most of the taxa, however 3 taxa (naticidae, *Mictyris* sp. and *Pseudopolydora* sp.) were difficult to separate beyond family or genus level for juvenile forms, as was the case for 9 of the taxa that were either not well known or very low in abundance (diogenidae and the last eight taxa in Table 5-6). In total there were 76 species. The sampling design used nested at a hierarchy of three scales (Chapter 3.1.), and included a no-take zone within the Jervis Bay Marine Park. All data were analysed as multivariate (76 species) and used the PRIMER-E 6+ software package (Plymouth Marine Laboratories and Clarke (1993)).

5.2.2.2. Taxonomic resolution, assemblage subsets and transformations

The different types of taxonomic resolution, assemblage sub-sets of common taxa and transformations, will collectively be referred to as data manipulations from this point forward. Thus there are three types of data manipulations.

Thirteen data sets were used to compare the relative effects of data manipulations, on

the measured biological patterns of the raw, unstructured data set (all data collectively from three years in three tidal flats). First, the tidal flat invertebrate assemblages were reclassified into four additional data sets to compare effects of increasing the level of taxonomic resolution, from species to phylum (Table 5-5 and 5-6). Another four data sets were created using assemblage sub-sets, chosen by selecting species that contributed to at least 10% of abundance in samples (47 species), and increasing the degree of contribution to $\geq 20\%$ (26 species), $\geq 50\%$ (14 species), and the 4 most common taxa (Table 5-5); i.e., rarer species were removed. A further four data sets were created from the raw data set using the transformations $x^{0.5}$, $x^{0.25}$, $\log(x+1)$ and presence-absence data. Bray Curtis similarity matrices were calculated for all 13 data sets to describe the general assemblage pattern. As discussed in Chapter 3, the scale of most variation was used as the replicate unit for calculations of Bray-Curtis similarity matrices as recommended by Bob Clarke (personal communication).

Table 5-5: The suite of treatments used on a raw data set; selectively choosing sub-groups of common taxa, reducing taxonomic resolution, and increasing the severity of data transformation. Annotations for the different data sets used in Figures 5-7 are combinations of letters and that are given in parentheses in front of *(transformations).

Assemblage subsets	Taxonomic Resolution	Transformations
(r) raw data - species/max res. (76 species)		*(0) no transformation
(a) abundant 10% (47 species)	(g) genus (71 taxa)	*(1) $x^{0.5}$
(a2) abundant 20% (26 species)	(f) family (52 taxa)	*(2) $x^{0.25}$
(a3) abundant 50% (14 species)	(o) class/order (13 taxa)	*(3) $\log(x+1)$
(a4) 4 common taxa	(p) phylum/subphylum (8 taxa)	*(4) pres/abs

To compare the relative effects of data manipulations of the raw data set on the measured biological patterns, the second stage analysis procedure in PRIMER-E 6+ (Clarke and Warwick 2001) was used. The measured biological patterns within each

of the Bray-Curtis similarity matrices were compared using Spearman rank correlations in a second-stage matrix. The second stage matrix was visualized in an MDS plot.

Table 5-6: The five different degrees of taxonomic resolution used as data manipulations to compare the similarity of measured biological patterns.

Phylum (sub)	Class/Order	Family	Genus	Species
Mollusca	Bivalvia	Arcidae	<i>Barbatia</i>	<i>Barbatia</i> sp.
		Galeommatidae	sp. #3 (cf. <i>Lasea</i>)	sp. #3 (cf. <i>Lasea</i>)
			<i>Mysella</i>	<i>Mysella anomala</i>
			<i>Arthritica</i>	<i>Arthritica helmsii</i>
		Hiatellidae	<i>Hiatella</i>	<i>Hiatella australis</i>
		Lucinidae	<i>Wallucina</i>	<i>Wallucina</i> sp.
		Laternulidae	Laternulidae	Laternulidae sp.
		Mactridae	Mactridae	Mactridae sp. juv
		Psammobiidae	<i>Soletellina</i>	<i>Soletellina alba</i>
		Tellinidae	<i>Tellina</i>	<i>Tellina</i> sp. #1
				<i>Tellina deltoidalis</i> juv.
		Thyasiridae	Thyasiridae	Thyasiridae
		Veneridae	<i>Eumarcia</i>	<i>Eumarcia fumigata</i>
			Veneridae	Veneridae sp. #3
				Veneridae sp. #4
	Gastropoda	Batillariidae	<i>Pyrazus</i>	<i>Pyrazus ebeninus</i>
		Buccinidae	<i>Nassarius</i>	<i>Nassarius jonasii</i>
			<i>Nassarius</i>	<i>Nassarius nigellus</i>
		Iravadiidae	<i>Nozeba</i>	<i>Nozeba topaziaca</i>
		Naticidae	Naticidae	Naticidae juv sp.
	Opisthobranchia	Bullidae	<i>Bulla</i>	<i>Bulla quoyii</i> juv.
		Cylichnidae	<i>Tornatina</i>	<i>Tornatina</i> sp.
		Philinidae	<i>Philine</i>	<i>Philine</i> sp.
Crustacea	Decapoda	Callinassidae	<i>Biffarius</i>	<i>Biffarius arenosa</i>
			<i>Trypaea</i>	<i>Trypaea australiensis</i>
		Crangonidae	Crangonidae	Crangonidae
		Diogenidae	Diogenidae	Diogenidae
		Mictyridae	<i>Mictyris</i>	<i>Mictyris</i> sp.
		Penaeidae	<i>Penaeus</i>	<i>Penaeus plebejus</i>
	Amphipoda	Corophidae	<i>Monocorophium</i>	<i>Monocorophium acherusicum</i>
		Exoedicerotidae	<i>Exoediceros</i>	<i>Exoediceros fossor</i>
		Paracalliopiidae	<i>Paracalliope</i>	<i>Paracalliope lowryi</i>
		Photidae	<i>Gammaropsis</i>	<i>Gammaropsis</i> sp.
		Phoxocephalidae	<i>Limnoporeia</i>	<i>Limnoporeia kingi</i>
		Urohaustoriidae	<i>Urohaustorius</i>	<i>Urohaustorius metungi</i>
	Isopoda	Actaeciidae	<i>Actaecia</i>	<i>Actaecia</i> sp.
		Bopyridae	Bopyridae	Bopyridae
		Cirolanidae	<i>Pseudolana</i>	<i>Pseudolana towrae</i>
Annelida	Oligochaeta	Oligochaeta	Oligochaeta	Oligochaeta
	Polychaeta	Ampharetidae	Ampharetidae	Ampharetidae
		Capitellidae	<i>Barantolla</i>	<i>Barantolla lepte</i>

			<i>Capitella</i>	<i>Capitella</i> sp.
			<i>Heteromastus</i>	<i>Heteromastus filiformis</i>
			<i>Mediomastus</i>	<i>Mediomastus australiensis</i>
			<i>Notomastus/Chrysosetos</i>	<i>Notomastus/Chrysosetos</i>
		Glyceridae	<i>Glycera</i>	<i>Glycera tridactyla</i>
		Hesionidae	<i>Heteropodarke</i>	<i>Heteropodarke</i> sp.
		Lumbrinereidae	<i>Lumbrineris</i>	<i>Lumbrineris</i> sp. #1
				<i>Lumbrineris</i> sp. #2
		Magelonidae	<i>Magelona</i>	<i>Magelona dakini</i>
		Nephtyidae	<i>Nephtys</i>	<i>Nephtys australiensis</i>
				<i>Nephtys longipes</i>
		Nereididae	<i>Australonereis</i>	<i>Australonereis ehlersi</i>
		Opheliidae	<i>Armandia</i>	<i>Armandia intermedia</i>
			<i>Lobochesis</i>	<i>Lobochesis bibranchia</i>
		Orbiniidae	<i>Leitoscoloplos</i>	<i>Leitoscoloplos</i> sp.
			<i>Leodamas</i>	<i>Leodamas johnstonei</i>
			<i>Scoloplos</i>	<i>Scoloplos simplex</i>
		Owenidae	<i>Owenia</i>	<i>Owenia fusiformis</i>
		Phyllodoceidae	<i>Phyllodoce</i>	<i>Phyllodoce</i> sp.
			<i>Eumida</i>	<i>Eumida</i> sp.
		Sabellidae	<i>Desdemonia</i>	<i>Desdemonia aniara</i>
		Sigalionidae	<i>Sigalion</i>	<i>Sigalion ovigerum</i>
			<i>Sthenelais</i>	<i>Sthenelais</i> sp.
		Spionidae	<i>Spio</i>	<i>Spio</i> sp. (<i>pacifica</i>)
			<i>Carazziella</i>	<i>Carazziella</i> sp. (<i>victoriensis</i>)
			<i>Pseudopolydora</i>	<i>Pseudopolydora</i> sp.
			<i>Prionospio</i>	<i>Prionospio aucklandia</i>
				<i>Prionospio yuriei</i>
			<i>Scolecopsis</i>	<i>Scolecopsis</i> sp. #1
Cnidaria	Anthozoa	Actiniaria	burrowing anemone sp.	burrowing anemone sp.
Nemertea	Nemertea	Nemertea	Nemertea	Nemertea
Phoronida	Phoronida	Phoronida	Phoronida	Phoronida
Sipuncula	Sipuncula	Sipuncula	Sipuncula	Sipuncula
Insecta	Insecta	Insecta	insect #1	insect #1
			insect #2	insect #2
Phylum (sub)	Class/Order	Family	Genus	Species

Further data sets were created to compare the interaction of the transformations with the level of taxonomic resolution and assemblage subset selections. The same four transformations, $x^{0.5}$, $x^{0.25}$, $\log(x+1)$ and presence-absence data, were applied to each of the taxonomic resolution and assemblage subset categories (Table 5-5), and transformation of raw data was also included; providing a total of 45 data sets. Again, Bray-Curtis similarity matrices were calculated for each of the data sets, and the

measured biological patterns within each of the matrices, were compared using Spearman rank correlations in a second-stage matrix. The second stage matrix was visualized in an MDS plot.

5.2.2.3. Cost-Benefit Analysis

A cost-benefit analysis compared effects of the two data manipulations, taxonomic selection and taxonomic resolution, for which there is a potential reduction in effort required to undertake a sampling program. The cost was defined as a reduction in similarity of biological pattern to the original data set, and the benefit was defined as a reduction of effort in sampling (benefit). The cost-benefit model followed the logic described in Karakassis and Hatziyami (2000), where the effects of reduced taxonomic resolution were compared using the following equation:

Equation 1

$$CB_L = \frac{(1 - r_L)}{\left(\frac{S - t_L}{S}\right)}$$

In equation 1, the cost-benefit ratio for a given taxonomic resolution (CB_L) decreased as Spearman rank correlation coefficients between the similarity matrix of reduced taxonomic resolution and the similarity matrix from the original data set approached 1 (r_L) (i.e. high similarity), and where the difference in the number of taxa at species level (S) versus the chosen level of resolution (t_L) increased (i.e. a high speciation rate between the chosen resolution and species level taxa increases the benefit).

Equation 1 provides a relative measure of comparison between different taxonomic resolutions; however it does not quantify the real benefit of reducing the taxonomic

resolution, i.e. savings in time and resources. The equation assumes that a reduction in the number of taxa is equivalent to a reduction in effort which is often, but not necessarily always true. The effort reduction must be quantified in order to relate it to the costs.

I then extended the work of Karakassis and Hatziyanni (2000) by altering the terms in the denominator to provide the effort saved by identification at higher taxonomic levels, by replacing the number of taxa at the respective levels (S and t_L) with the average time to identification at the different levels (T_S and T_L). Thus,

Equation 2

$$CB_L = \frac{(1 - r_L)}{\left(\frac{T_S - T_L}{T_S}\right)}$$

the Cost-Benefit ratio decreases with higher correlation between the taxonomic similarity matrices (r_L), and an increase in difference for time (or \$value) to identification between the species (T_S) and selected taxonomic level (T_L). The average time to identification for the different taxonomic resolutions (Table 5-7), was provided by expert taxonomic staff at the Marine Invertebrate section at the Australian museum.

Similarly, assemblage subsets were also compared using Spearman rank correlations between similarity matrices, while time saving was measured by determining the number of species in the original data set as for equation 1, S_S , and the number in the reduced data set as t_L . This assumes that the time used to remove species that are not to be identified is negligible, thus it excludes searching through these specimens to find the target species; a factor which could be offset by reduced time due to familiarity with

a smaller range of species.

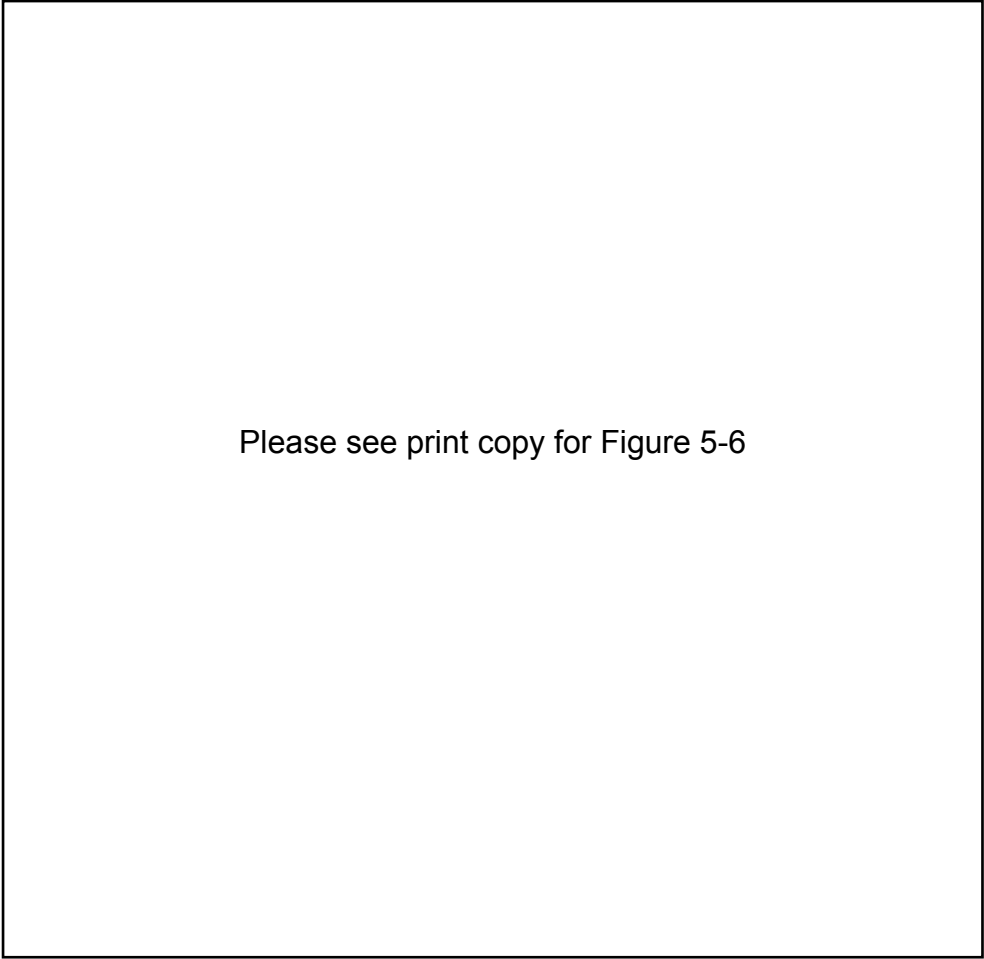
To visualise the effect of resolution choice on the cost (reduced correlation with original data set) versus the benefit (proportional reduction in effort (time or financial resources)), the numerator and denominator in equation 2 for each taxonomic resolution, selection and transformation were plotted.

5.2.2.4. Case study

To test the relative sensitivities of assemblage subsets, taxonomic resolution or transformations in detecting an impact in this present study, each of the similarity matrices from the 45 datasets were analysed in 45 separate second stage matrices of Spearman rank correlation coefficients (ρ) to detect the pattern of assemblage change. For each new second stage similarity matrix, 9 matrix subsets were compared; 3 tidal flats in each of three years, including 1 year before and two years after one of the tidal flats was protected as a no-take Marine Protected Area. As is recommended in spatially hierarchical sampling designs (Bob Clarke personal comment), the scale of greatest variation, sites, was used as the sampling unit (Winberg et al. 2007).

I then applied the methods described for 2-stage analysis by Clarke and Warwick (2001), but extended the method to apply a third stage correlation matrix, comparing the impact patterns arising from the 45 different data manipulations (Fig. 5-6). Thus a third-stage matrix resulted from Bray-Curtis similarities between samples, correlation of sample groups in a beyond BACI design structure, and correlation of the beyond BACI patterns for each of the data manipulations.

Manipulated data sets were selected to test for sensitivity of detecting an impact. First the correlations between the 45 different datasets using the beyond BACI structure were visualised in a hierarchical cluster diagram. Clusters of datasets that were highly correlated ($\rho \geq 0.95$) with the full and untransformed data set ($r(0)$) were considered as a group. Clusters of data sets that represented decreasing correlation with the original data set were identified at intervals of the Spearman rank correlation coefficients from $\rho > 0.85, > 0.75, > 0.7, > 0.6$. (Fig. 5-10). An arbitrary dataset from each cluster of decreasing correlation with the raw data set, was chosen to represent the sensitivity of the cluster to detect assemblage change. Change was tested for using in the Beyond BACI design and the PERMANOVA component of the PRIMER-E 6+ (beta version 17) as for chapter 4, and was also visualised in MDS plots of the second stage matrices.



Please see print copy for Figure 5-6

Figure 5-6. A schematic diagram based on the logic of Warwick and Clarke (2001) but extended to a third matrix (or second Spearman Rank correlation matrix) to compare the relative effects of data selections, aggregations and transformations on the sensitivity to detect an impact on the assemblage. C, S and N represent the three locations Currumbene Creek, Sussex and Narrawallee Inlets respectively.

5.2.3. Results

5.2.3.1. Taxonomic resolution, data selection and transformations

The measured general biological pattern of presence-absence data was the least correlated to the pattern of the raw data set (r_4 , $\rho=0.74$) (Fig. 5-7 and 5-8). A further 6 data manipulations also exhibited a relatively large effect on observed pattern ($\rho = 0.86 - 0.94$) in the decreasing sequence of phyla (p), $x^{0.25}$ (r2), order/class (o), and equally dissimilar were the four most abundant taxa (a4), and $x^{0.5}$ (r1) and $\log(x+1)$ (r3). The assemblage subsets of species that contributed to 10, 20 and 50% (a – a3) of abundance in samples, as well as the taxonomic levels of genus (g) and family (f), were collectively a core group of data manipulations that strongly reflected the patterns measured in the raw data ($\rho > 0.99$) (Fig. 5-7).

Of note is that the effects on measured biological patterns were different for all three types of data manipulations. The correlation between measured biological patterns using higher taxonomic levels, assemblage selection and transformation is consistently lower than with the raw data set, and thus describe different biological patterns. For example the correlation between biological patterns for presence-absence of species data and abundance of phyla is only 0.44 (Spearman rank).

The combination of data transformations and taxonomic resolution or assemblage selection, exacerbated the effect of changes to measured biological pattern for the latter two types of manipulations. Also, the effect increased with the severity of transformation towards presence-absence data. The interaction effect of transformation was greatest for phyla and order taxonomic levels (Fig. 5-8a), followed by the four

abundant species (a4) (Fig. 5-8b). For the core group of manipulations that measured similar biological patterns, assemblage selection of the abundant 50% of species in samples and family level taxonomic resolution, the correlation of biological pattern with the original data set was reduced from $\rho > 0.99$, to $\rho = 0.91$ and $\rho = 0.95$ respectively.

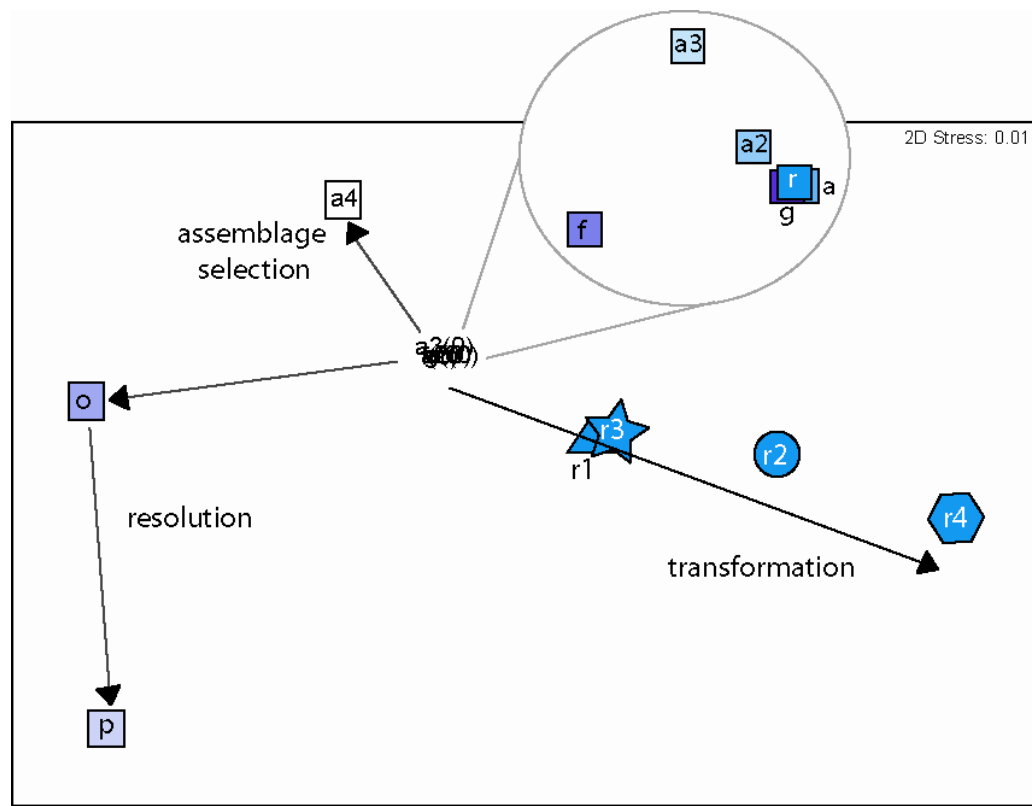


Figure 5-7. MDS plot illustrating the relative effects of different types (taxonomic resolution, assemblage selection and transformation) and degrees of data manipulations. The taxonomic resolutions are species (r), genus (g), family, order/class (o) and phylum (p); assemblage subsets were species that contributed to 10 (a), 20 (a2) and 50% (a3) of abundance in samples as well as the four most abundant taxa (a4), and transformations increased from raw data (r), $x^{0.5}$ (r1) and $\log(x+1)$ (r3) to presence-absence data (r4).

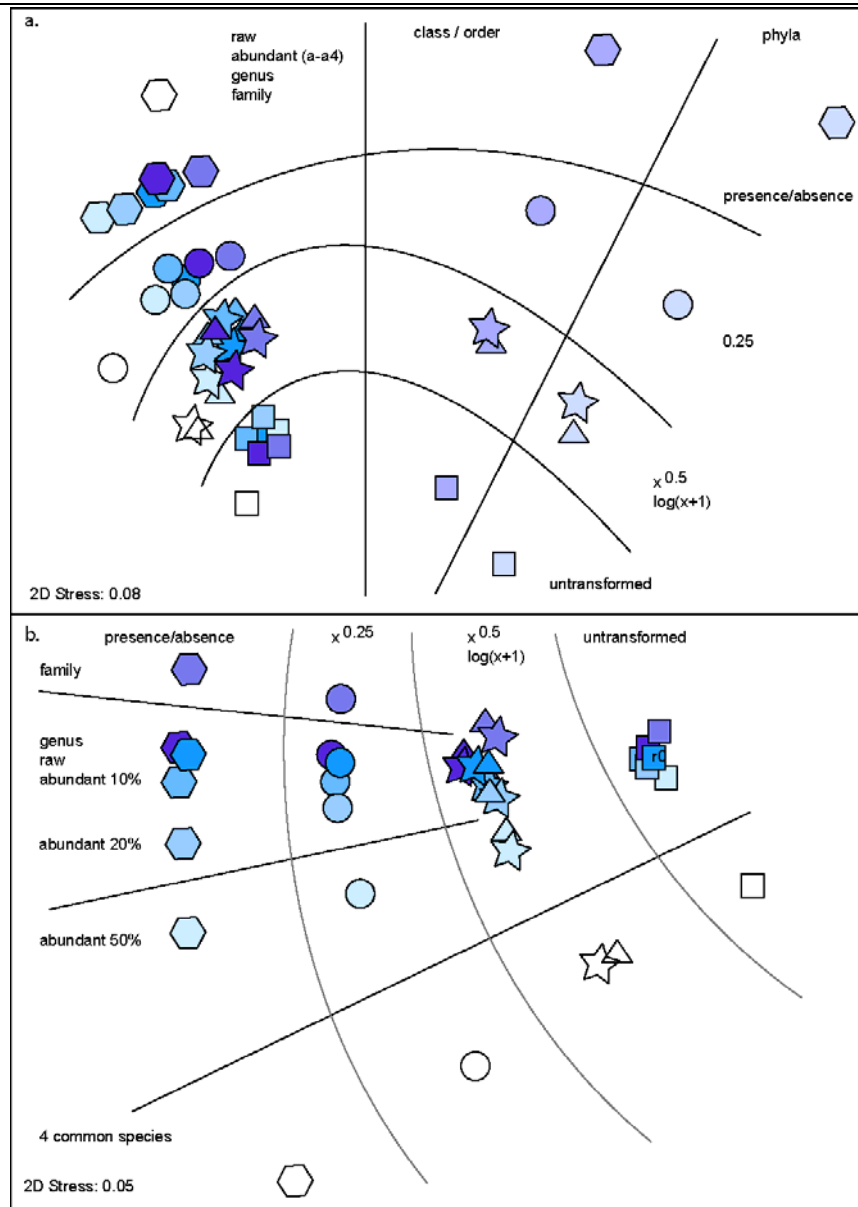


Figure 5-8. Second stage MDS plots of Spearman rank correlation coefficients for each of the abundant species (10%, 20%, 50%, common 4) and taxonomic resolution ((a) raw/species, genus, family, class/order, phyla (b) raw/species, genus, family) data set similarity matrices, including the four transformations (square root, fourth root, $\log(x+1)$ and presence/absence) for each data set.

5.2.3.2. Cost Benefit Analysis

The resources saved by using reduced taxonomic resolution were considerable, and without significant loss to the biological pattern up to the family level (Figure 5-9). This related to a benefit of saving in time or money of over 60%. For selection of abundant taxa in samples, again there was little reduction in biological pattern until only the four most abundant of 76 species remained, however the saving in time and financial costs was much smaller at about 10% for little loss of biological pattern, or 25% if a correlation of 0.95 is considered acceptable.

5.2.3.3. Case Study

Using the data from a real impact case study, it could be seen that only 15% or 7 of all combinations of data manipulations maintained the biological impact pattern of raw data with correlation values of at least $\rho = 0.95$ (Fig. 5-10). These reflect the choices in the cost benefit analysis (Fig. 5-9) using lower taxonomic levels to family, as well as the full suite of assemblage subsets to the four most common species. Of note is that a commonly applied transformation of $x^{0.5}$ showed less than $\rho = 0.7$ correlation of measured biological pattern with the raw, untransformed data set.

Table 5-7: The estimated time taken to identify a specimen to different taxonomic levels (resolution) by taxonomists at the Australian Museum, and the relative number of specimens (selection) for all taxa, versus taxa that contribute at least 10, 20 or 50% of abundance, and the four most common and abundant species (a4).

resolution	Species	Genus	Family	Order	Phyla
Time to ID (mins)	30	20	10	5	3
assemblage subset	all taxa	a10%	a20%	a50%	a4
number specimens	12607	12396	11964	11082	9438

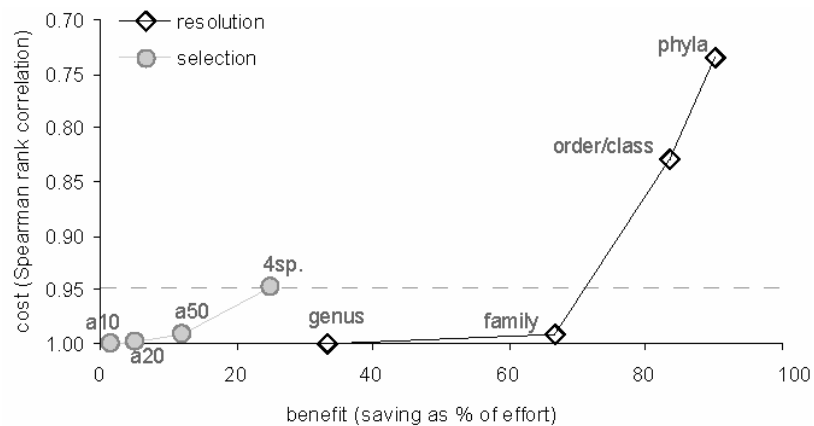


Figure 5-9: Cost-benefit analysis, as a reduction in effort (time or \$) using different taxonomic resolutions or data selections, compared to analysis of the full data set to species level. The cost of different analysis options is measured as a reduction in Spearman rank correlation of the Bray Curtis similarity matrix with the equivalent full species resolution matrix; i.e. the relative change in measured biological pattern.

Selected data sets representative of a stepwise decrease in correlation with the original data (Fig. 5-10) set, showed a decreasing trend in detecting the pattern of change following no-take zoning at tidal flat C compared to control sites S and N (Fig. 5-11). The highly correlated data, raw, four abundant species and family showed consistently significant effects at the scale of sites, and at the scale of tidal flat for the four abundant species only. Data with correlation values $\rho < 0.95$ did not show significant interaction ($p < 0.05$) between the no-take versus control sites and before versus after no-take zoning at either of two nested spatial scales (Figs. 5-11 and 5-12).

There were other significant biological patterns, unrelated to the impact study at hand, that became obvious using low correlated data sets. For example, one of the control

tidal flats (S) showed clear compositional change at the scale of sites using presence-absence data (r(4)) (Figs. 5-11 and 5-12). Although unrelated to the impact study at hand, this demonstrates the strong differences in biological patterns measured using different data manipulation.

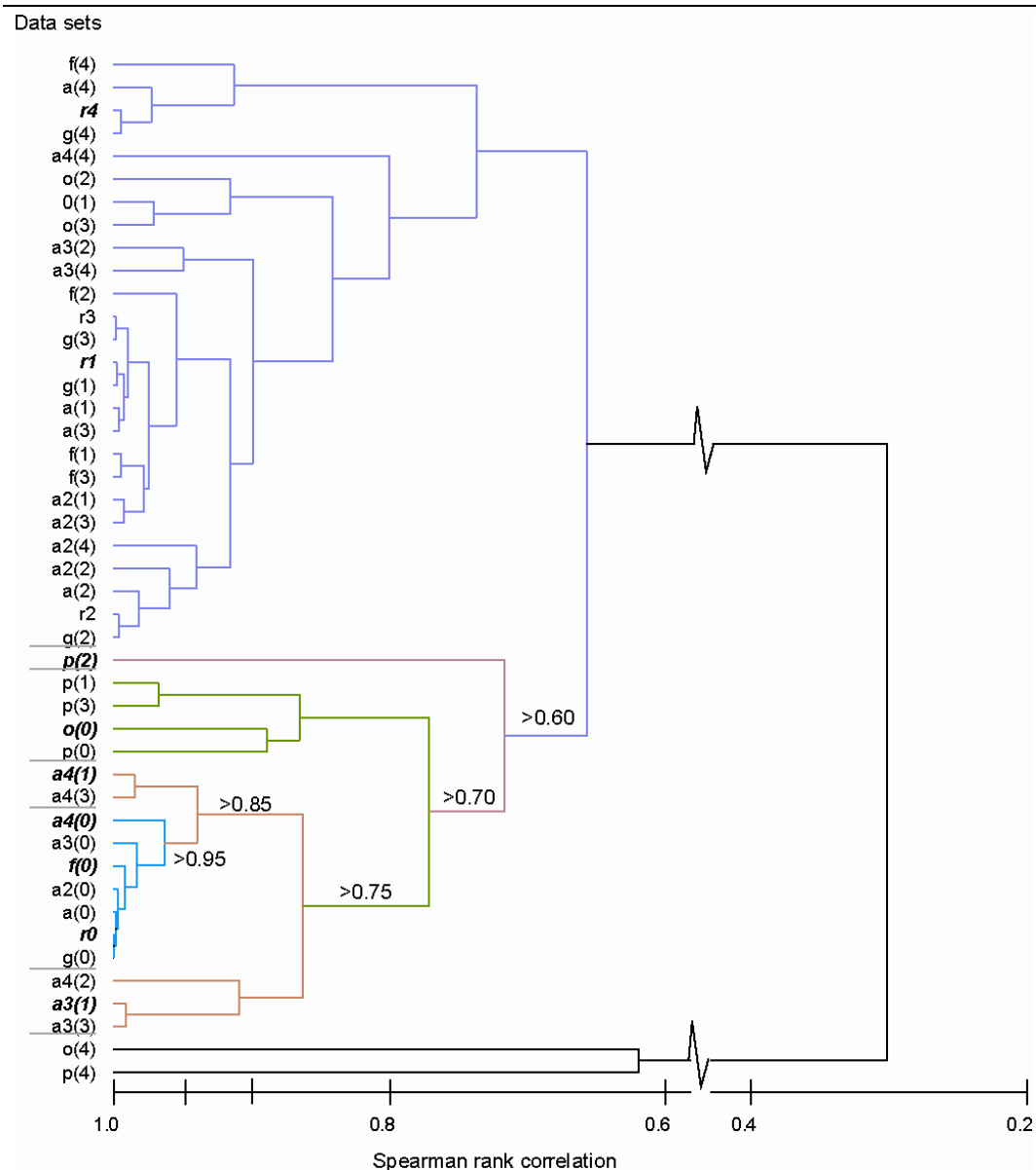


Figure 5-10: Hierarchical cluster analysis of third stage correlation coefficient matrices comparing 44 datasets using different manipulations, with a raw untransformed data set (**r0**) used to detect assemblage change before and after the introduction of a no-take protection in a Marine Protected Area. Spearman rank correlation values are shown on the x axis, and clusters with correlations at intervals of $\rho > 0.95$, 0.85, 0.75, 0.70, and 0.6 are identified with different colours and labels. The 44 data sets are labelled according to the logic in Table 5-1, and bold cursive text indicates data sets that were used in PERMANOVA tests.

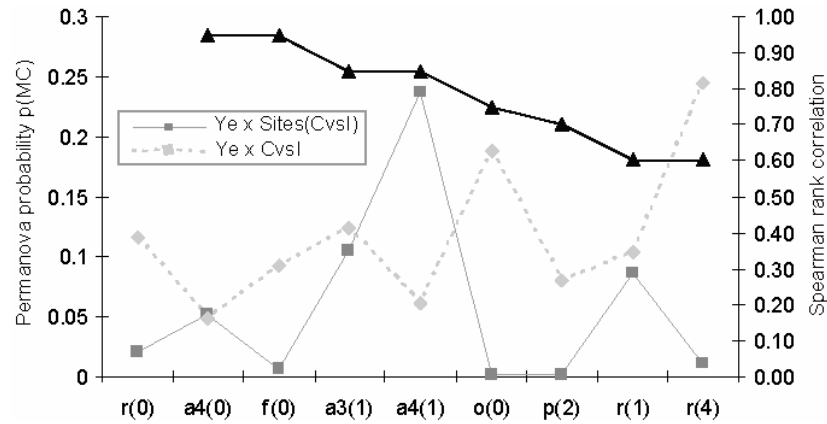


Figure 5-11: Comparison between selected data matrices from Fig. 5-10, comparing the sensitivity of PERMANOVA analysis ($p(MC)$) to detect an impact in relation to declining Spearman rank correlation with the full original data matrix. The selected data matrices use the following manipulations of the raw, untransformed species data ($r(0)$): the four most abundant species untransformed ($a4(0)$), family untransformed ($f(0)$), the square root of species that contribute to at least 50% of abundance in samples $a3(1)$, the square root of $a4(0)$ ($a4(1)$), untransformed class/order ($o(0)$), fourth root phyla ($p(2)$), square root raw species data ($r(1)$), and presence-absence of raw species data ($r(4)$). The interaction testing for an impact, Years x Control vs. Impact, is presented for two scales; sites (Years x Sites(CvI) and estuaries (Ye x CvI).

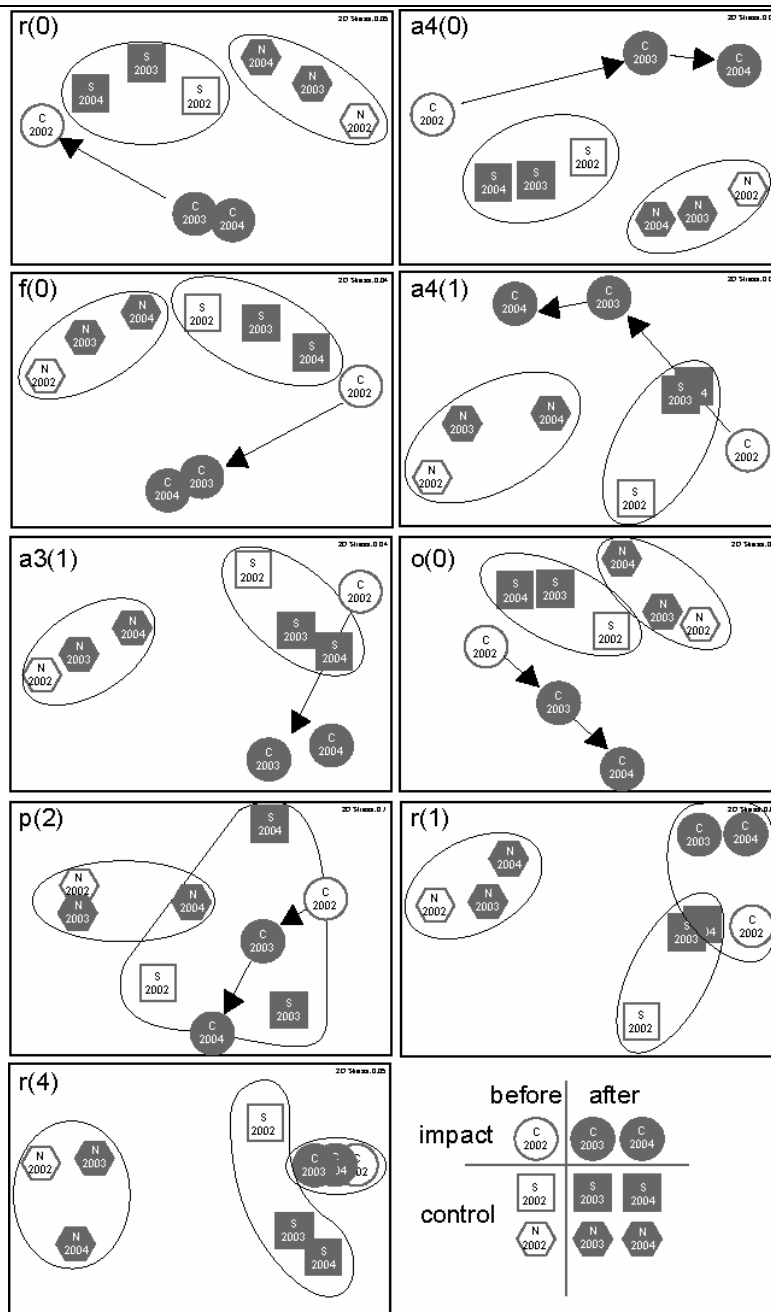


Figure 5-12: Comparison of the sensitivity of 7 data manipulations, selected from Fig. 5-10, to detect a change in macrofaunal assemblages at a no-take site (C) compared to control sites (S and N), from the year before no-take zoning (2002) and the following 2 years. The MDS plots show Spearman rank correlations between the 9 beyond BACI structured sets of data. The MDS plot labels are described in Table 5-5.

5.2.4. Discussion

This study highlights the substantial effects that different choices of data manipulation have on the outcome of measured biological patterns, including the outcome for an impact study. It is demonstrated that if only one type of data manipulations is used, then there is a greater risk of failing to detect an impact in the case when there is one (analogous to Type II error (Mapstone 1995)); particularly for changes to the relative abundance of species when, as is common practice, square root transformation is applied. This may have undesirable consequences for the environment and ecological processes. Because the type of change expressed in assemblages cannot always be anticipated, the variety of biological patterns that can be described using different data manipulations should be explored to a greater extent in ecological studies. In addition, the effort in sampling programs can be reduced substantially by using lower resolution and, to a smaller degree, assemblage subsets.

5.2.4.1. Data set choices

Here I found that square root transformation, which is applied widely as a default compromise of transformation choices, can change the patterns of species abundance quite dramatically so that impacts might not be detected. However many studies assume that square or fourth root transformation will capture both patterns of abundance and composition, and therefore rely entirely on the patterns described using these transformations (Curley et al. 2002, Roberts et al. 1998). As the relative abundance of species is important for spatial biodiversity and impact studies, it is important that raw and transformed data are analysed.

Transformation of abundance data to presence-absence data showed the largest effect on measured biological pattern. This form of data transformation is currently well applied in the literature as a valuable tool to indicate whether differences between assemblage patterns are due to differences in the relative abundance of common species (raw data), or whether the composition of species in assemblages differs (Clarke and Warwick 2001). It was evident here that patterns of species abundance and assemblage composition did not exhibit similar patterns for this data set, and that both untransformed and presence-absence data should be used. It also important to consider that there are limitations in studies when only presence-absence data can be used, and that abundance data collected with similar sampling methods is important for both distributional and impact studies.

Another assumption often used is that the severity of effects of transformation on biological patterns increase in the sequence of square root ($x^{0.5}$), fourth root ($x^{0.25}$), $\log(x+1)$ and presence-absence. However as is evident from the results, $\log(x+1)$ was less severe than $x^{0.25}$ and roughly equivalent in its effect to $x^{0.5}$. The reason for this is that we have to use $\log(x+1)$ rather than $\log x$ to account for the zero counts of species in the data set; thus adding a one across all species in the data set when the abundances are relatively small and often zero results in the transformation being less severe than $x^{0.25}$. I suggest that this may be the norm rather than the exception for assemblage studies where zero abundance of species is high in each sample.

Taxonomic resolution from species to the level of family described very similar biological patterns. The similarity of pattern at the level of family could stem from functional similarity at this taxonomic level (Warwick 1993). Studies of soft-sediment

invertebrates might therefore be appropriate at the family level, but because the taxonomic hierarchy of different families may not reflect functionality or may present high speciation rates, then this level may not be appropriate in all circumstances.

Class and order taxonomy patterns were nearly as different to the raw data set as presence-absence data. However in contrast to presence-absence data, high taxonomic levels would reduce measurable compositional differences and the biological patterns described at these taxonomic levels might be of limited value in many instances.

The effect of using subsets of abundant species might intuitively be equated with the opposite to transformation of data that increases the influence of rare species on measured patterns. However the effect of selecting abundant species was minimal, and in this study where abundant species probably dominated assemblage patterns overall, nearly all assemblage subsets, except the four most abundant species, provided a similar set of biological patterns. I suggest that this might be a feature that could improve the sampling efficiency of assemblages that exhibit high dominance. The effect of subgroups would be to remove background noise of rarer species, potentially increasing the power to detect biological patterns, and as a consequence, sampling effort in the field may be reduced. However this has to be addressed in relation to ignoring parts of the biodiversity.

When higher taxonomic levels and assemblage subsets are used in combination with transformations, the effects of the former two manipulations on the biological pattern is exaggerated dramatically. This should be taken into account when data can only be used in a presence-absence format. For example, when comparing data from different

sources, presence-absence data is often used to overcome incompatible sample sizes or methods.

The recent capabilities of software to quickly compare a variety of transformations, implies that multiple analyses using different transformations should be employed in most studies, to gain the full benefit of valuable data.

5.2.4.2. Cost-benefit analysis

In designing monitoring programs of soft sediment habitat, the cost benefit analysis extension that I developed here demonstrates that for tidal flat invertebrates, taxonomic identification to the family level is an obvious choice. Up to a 60% reduction in effort could be achieved using family level resolution with minimal loss to the biological patterns.

The adequacy of taxonomic resolution to the level of families is consistent with other soft sediment studies that compare taxonomic resolution in soft sediments (James et al. 1995, Karakassis and Hatziyanni 2000, Olsgard and Somerfield 2000). Further reduction in taxonomic resolution to order, class or phylum, is not reflective of biological patterns for such broad assemblage data and should be used with caution.

The uses of abundant sub-sets of species does not provide much benefit in time savings, however it should be recognised that this is probably underestimated. Using the most abundant taxa for identification might not reduce the quantity of specimens to sort through by much, but less familiar and therefore more time consuming species will be removed. Large scale studies might reduce costs substantially by choosing to identify a sub-set of more familiar species. This choice might also affect, and possibly improve,

the sensitivity of tests to detect an impact (see Chapter 4), again particularly for assemblages exhibiting high dominance of a few species.

5.2.4.3 Consequences for an impact study

Different taxonomic resolutions from species to the level of family were sensitive to detecting changes to invertebrate assemblages in a tidal flat, following the introduction of a Marine Protected Area. The associated reduction in cost justifies the use of family level resolution in tidal flat impact studies. In contrast, all data transformations eliminated the sensitivity for detecting the impact on a tidal flat, while the removal of rarer species from the data set, i.e. species selection up to the 4 most abundant taxa, demonstrated adequate sensitivity for detection of an impact at the two spatial scales investigated.

These findings demonstrate that the sensitivity of reduced taxonomic resolution or assemblage subsets to detect an assemblage pattern will be specific to the habitat, assemblage and impact tested for. In this study, the pattern of the data was a shift in the abundance of dominant species. Therefore, transformation to increase the influence of rarer species diluted this effect. Again, I raise the issue of the broad use of square root transformation alone which, in this case, did not detect the change resulting from no-take zoning. These findings demonstrate that if an impact is anticipated to affect the relative abundances of species rather than change composition, then raw, untransformed data should be used in addition to the standard square root transformation.

A further observation was that biological patterns measured varied not only between the types of data manipulations, but this interacted with spatial scale. For example, the use

of the subset of the 14 most abundant taxa showed similar effects of no-take zoning at two scales, but for the 4 most abundant taxa, this effect was reversed and very different at both scales (Fig. 5-11). Thus effects might be detected at one spatial scale using one form of data manipulation, but at another spatial scale using another type of manipulation; yet again highlighting the need for carefully planned hierarchical sampling strategies.

5.2.4.4. Consistency with other studies and further considerations

The differences in measured biological patterns arising from the use of different transformations and different taxonomic resolutions were very similar to those found in an impact study by Karakassis and Hatziyanni (2000), where sub-tidal sediments were exposed to organic enrichment. Thus transformations should be used with caution, or at least with an understanding of the consequences at lower taxonomic resolutions.

Preferably, a suite of transformations should be used.

Consistent with other studies in soft-sediment habitats (eg. Olsgard and Somerfield (2000)), selection of invertebrate sub-groups of assemblages, and identification at the family level can benefit tidal flat impact studies. The time and cost savings to monitoring projects far outweigh the loss of biological information in tidal flat habitat, and potentially reduce unhelpful ecological noise from rarer species.

Consistency between findings across studies in soft-sediment habitat may provide guidance to improve research and monitoring effort in future studies. It is not anticipated that there will ever be a generic analysis methodology suitable for all assemblage types, across all habitats and for all purposes (Vanderklift et al. 1998),

however, there is value in improved appreciation of the effects of sampling and analysis choices in specific habitats and for specific purposes.

Other studies have demonstrated that effects of data selection, taxonomic resolution or transformations can vary with habitat type (Vanderklift et al. 1998), individual locations within the habitat type (Karakassis and Hatziyanni 2000), with taxonomic groups (eg. plants versus fish (Vanderklift et al. 1998), with latitude (Ellingsen et al. 2005) and with speciation rates (Kerswell 2006). Therefore a degree of local and habitat knowledge is important at an early stage, and pilot studies should provide some information to inform the choice of sampling and methods of analysis in each case.

CHAPTER 6 - DISCUSSION

6.1. General findings and consistency with other studies

This study documents the first faunal assemblage recovery trajectory across a whole tidal flat following no-take zoning of a Marine Protected Area. These results demonstrated a potential functional group response, as the dominant shift was an increase in the abundance of suspension feeding, juvenile bivalves and other less mobile and small deposit feeders. Of note is that the juvenile bivalve *Eumaricia fumigata* and *Soletellina alba* showed six and two fold increases in abundance that were maintained for the two year years following no-take zoning. The patterns were generally also associated with increased homogeneity of spatial distribution. In contrast, more mobile, scavenging or predatory species showed significant declines in abundance. These patterns were not reflected in the control sites and as such were attributed to the no-take zoning.

The assemblage shift documented here is consistent with the effects identified in other studies on physical disturbance of soft sediments, and therefore the impact from bait harvesting activities was assumed to be primarily a disturbance effect rather than an effect of removal of the targeted bait species (*Trypaea australiensis*). Thus bait-harvesting impacted non-target species. Consequently, there is strong evidence that the impact of bait harvesting activity might broadly be categorised as sediment disturbance either through disruption to sediment stability, compaction from trampling, damage to sensitive organisms or smothering effects. These potential types of impacts and how they might affect the tidal flat habitat and ecological processes is discussed further below.

Another key finding of this study was the identification of important scales of spatial patterns for tidal flat faunal assemblages. The patchiness of the faunal assemblages was greatest at scales of 100s m. This was consistent for all three tidal flats investigated, and assemblage differences within each habitat were greater than that between the three tidal flat locations. Other studies have demonstrated similar scales of patchiness within tidal flats (Dittmann 2000, Edgar and Barrett 2002, Honkoop et al. 2006, Thrush 1991).

The small scale patchiness of the tidal flat assemblages might help to explain the observation that the recovery trajectory following no-take zoning was also patchy, that is, the type of assemblage response varied in type and degree between sites at the no-take zone. However this could also be an artefact of other factors and this is discussed further below (6.3.1.).

In addition to the key findings of the study, the methodologies employed here included an extension to asymmetrical univariate analysis (ANOVA) that provides for increased sensitivity of research and monitoring programs; specifically for monitoring effects in MPAs as well as for environmental impact studies. The extension describes methods for the inclusion of fully asymmetrical data sets for both temporal and spatial factors in the analysis of Beyond BACI designs; previously analyses did not use the full data set simultaneously and thus lost power in the analysis.

Similarly, this study demonstrates the sometimes radical effects of different sampling methods and data manipulations for multivariate assemblage data; indicating that as much emphasis is required on the analysis of data sets, as in the design of studies and collection of data. It was demonstrated that standard multivariate transformations, such

as square root, might render the data less sensitive to detecting impacts that are expressed as changes in abundance of dominant species.

Further to assessing the effects of methodological and analysis procedures on the results of impact studies, it was found that savings of up to 60% can be achieved by the identification of samples to family, rather than the species level for tidal flat assemblages. Similar to other habitat and soft sediment studies, assemblage patterns of taxonomic families strongly reflect that of species.

A final outcome of this study was to further demonstrate the potential for using management decisions as an experimental opportunity that serves two purposes: better knowledge of ecological systems at large scales and application of knowledge for adaptive conservation management (Castilla 2000, Ewel et al. 2001).

6.2. Direct Implications for Management

Three key findings of this study can be directly applied in adaptive management of marine protected areas in NSW, Australia. These findings are that bait harvesting has an effect on tidal flat assemblages, that there was a recovery trajectory of faunal assemblages, and that the spatial patterns of tidal flat assemblages differ at different scales.

Measured effects from bait harvesting demonstrate that this activity should be recognised and managed as an anthropogenic impact on tidal flat habitat. The measured recovery trajectory also demonstrates that no-take marine protected areas are an appropriate form of management of this impact on tidal flats, i.e. they work.

Now that the relevance and effectiveness of no-take MPAs in tidal flats are confirmed, the spatial patterns of tidal flat assemblages can provide clear guidance for planning and adapting MPA conservation. First, the smaller scale (100m) patchiness of assemblages across a tidal flat indicates that whole habitat needs to be protected in order to achieve representativeness for tidal flat ecological diversity. Second, the similarity of whole tidal flat assemblages between three locations, compared to greater variability within locations, demonstrates that tidal flat habitat can be used as a surrogate for macrofaunal biodiversity. However this is still only relevant at the scale of 30km tested here, and also for morphologically similar tidal flats in morphologically similar estuaries.

In order for protection of tidal flats in MPAs to be effective, it would be vital that other known stressors of this habitat type are also absent from tidal flats selected as no-take MPAs. For example, Lindegarth and Hoskin (2001) demonstrated that catchment development also impacts the tidal flat fauna, potentially through metal contamination from run-off. Particular species have also been shown to be sensitive to fresh water flow disturbances (Matthews and Fairweather 2003). In addition, boat wash from large ferries has been demonstrated to affect macrofaunal assemblage composition in tidal flats (Bishop and Chapman 2004). These findings qualify the perception that tidal flats in NSW face ecological impacts from a variety of human activities; the collective and synergistic effects of which are unknown. Therefore tidal flats in areas of low catchment development (both agricultural and urban) and negligible changes to natural water flow regimes should be prioritised for Marine Protected Areas, and the management of other impacts from boat traffic should be considered.

This study also highlights the value of using invertebrates with fast life cycles for

research and monitoring programs, as they might demonstrate the fast recovery rates for faunal assemblages. Of benefit to future research in tidal flat habitats, this study provides a complete set of macrofaunal assemblage data specific to permanently open tidal flats (Appendix I and II). This information can now be used with similar data sets to further inform bioregional assessments, and in particular, address the marine protected area principles of adequacy and comprehensiveness (see CAR principles in Chapter 1).

6.3. Potential implications of findings

This study demonstrates that bait harvesting has an impact on tidal flats and that no-take marine protected areas MPAs can be an effective tool for managing this impact.

However, because of the political climate and stakeholder pressure that so often governs the outcome of marine protected area design, questions will be raised as to whether the impact of bait harvesting matters. Studies elsewhere can be used to demonstrate that there are potential flow-on effects from bait harvesting impacts as the faunal assemblage structure has shifted; many ecological processes are linked to faunal assemblage structure. However there are no immediate measurable benefits to stakeholders, and we can't predict the implications of the impact. It is therefore important to structure some discussion around what the impact of bait harvesting on tidal flats might be, and what could the consequences be.

6.3.1. Structuring processes

The findings in the present study are relevant to our growing understanding of ecological models of physical disturbance as a structuring process in tidal flats. Physical disturbance is recognised as one of many natural structuring processes of tidal flat assemblages as described in the introduction (Chapter 1), and here I suggest that the effects of bait harvesting might be closely aligned with the biological and physical effects of epibenthic predation and associated disturbance.

6.3.1.1. Non-anthropogenic predation effects

Predation is one of the most well studied ecological structuring processes (Jones et al. 1991) and has been reviewed in relation to soft-sediment communities (Dittmann 2007, Thrush 1999), however generalisations are still hard to identify. Predation is potentially a dominant structuring process in unvegetated tidal flats, compared to vegetated soft-sediments that offer structural protection to prey. However this is not always the case, reflecting yet again the complexity of interacting structuring processes in tidal flats (see Chapter 1 - Introduction). For example, small bivalves are able to overcome heterogenic forces of predation by strong post-settlement redistribution (Filho et al. 2005, Norkko et al. 2001, Thrush et al. 1991). Nevertheless, there are some demonstrated effects of predation on the structuring of faunal assemblages, both by consuming prey and by associated disturbance of sediments.

Another comparison to be made between sediment disturbance by human bait harvesting and natural predation is the type and intensity of disturbance. It was regularly observed that sting ray and other fish predation imposed a high level of sediment

disturbance on tidal flats, creating feeding pits and mounds of displaced sediments after each high tide. Considering that a recovery was measured following exclusion of bait harvesters in the present study, it can be hypothesized that bait harvesting has very different effects to natural fish predation, despite being comparable processes as described above, or that it significantly increased predation effects. This raises the question, does anthropogenic bait harvesting differ in type or intensity compared to natural predation and sediment disturbance? In sub-tidal studies where, similar to the tidal flat here sting rays are predators of molluscs and crustaceans, it has been estimated that natural predation disturbs 100% of the sediment habitat over a 47 day period (Tyler 1999). Comparisons between natural and anthropogenic disturbance rates can not be made in the present study, but would provide information on whether the type of disturbance, or the intensity of disturbance, accounts for the measurable effects of bait harvesting in relation to natural predation.

Predation may not be random over tidal flats as fish can target high densities of prey (Hines et al. 1997), thereby being a homogenising force on the spatial distribution of assemblages. Similarly, fish species have been shown to co-exist by partitioning of food resources, and therefore predators are sometimes specialists (Caron et al. 2004, Giles 2002, Serrano et al. 2003, Tyler 1999); thus feeding effort by predators may be non-random and target areas of preferred prey. Such homogenising effects on targeted species or assemblage patterns have been demonstrated from natural predation or grazing by fish, birds and gastropods on tidal flats (Hines et al. 1997, Kelaher et al. 2003, Levinton and Kelaher 2004, Thrush 1991). However these processes will vary depending on a number of factors including opposing forces (Levinton and Kelaher 2004), and in particular, effects on non-target fauna might be very different. For

example, the mobility of predator and prey, lethal or sub-lethal effects, the sensitivity of non-target species to predation disturbance, aggregative structure of prey and non-target species and spatial scales of prey or non-target species patches, might alter the effect of predation on spatial patterns (Thrush 1999). Therefore, predation and associated disturbance may either have a homogenising effect, or increase the spatial patchiness of sediment assemblages and/or species given different conditions (Fig. 6-1).

6.3.1.2. Bait harvesting as an impact

An important question, both ecologically and from a management perspective, is raised in relation to the observed effects of bait-harvesting. How does bait harvesting impact differ from other natural disturbance or structuring processes in tidal flats? Again I will make comparison between sediment disturbance by human bait harvesting and non-anthropogenic predation.

Differences may exist in either the type (eg. compaction versus disruption of sediments) or the intensity of disturbance. It was regularly observed that sting ray and fish predation imposed a high level of sediment disturbance on tidal flats, creating feeding pits and mounds of displaced sediments after each high tide. Considering that a recovery was measured following exclusion of bait harvesters in the present study, it can be hypothesized that bait harvesting has very different effects to natural fish predation, despite potentially being comparable processes as described above, or that it significantly increased predation effects.

6.3.1.2. Potential structural effects of anthropogenic activity

The impact of bait harvesting in this study can be compared to both physical and biological structuring processes of natural predation observed in tidal flats and soft-sediments. Bait-harvesting has similarities to biological processes in that it targets a certain species, analogous to a predator, but it also inadvertently affects non-target species. I found that faunal assemblages increased in heterogeneity as a result of bait harvesting; thus the measured effect might be considered similar to the effect of decreased homogeneity of non-target species (Fig. 6-1) (non-random predation for target species within a homogenous distribution of non-target species).

In addition, the recovery trajectories observed in this study differed at scales of 100s m; that is, they were not consistent between sites in the no-take tidal flat. This effect may be due to either of two conditions. First, pre-existing spatial patterns might determine the type and extent of the effects of bait harvesting at different spatial scales. The patchiness of macrofaunal recovery across spatial scales of 100s m suggests that the sensitivity and the resilience of the tidal flat assemblages to bait harvesting were variable at this scale. In other words, some parts of a tidal flat may host sensitive assemblages while others were less susceptible; possibly related to adaptation to high levels of natural physical disturbance from wave or tidal action.

Second, patchiness in the recovery trajectories may be an artefact of the pattern and scale of bait harvesting intensity being focussed in certain areas, but this effect cannot be controlled for in the present study. Of consideration is that marine predators are limited in foraging time at higher tidal levels, while the opposite is true for bait

harvesting; thus impacts from human and nekton activity might be separated spatially to some degree by this environmental process.

It is assumed however, that the sensitivity of assemblages to bait harvesting may indeed vary between different assemblage types in different areas of a tidal flat. Recovery rates of disturbed soft-sediment assemblages have been shown to vary across different sediment types, and in particular finer sediments have longer recovery times than coarser sediments in both intertidal and subtidal soft sediments (Dernie et al. 2003, Ferns et al. 2000, Morello et al. 2005, Queiros et al. 2006). From a management perspective, it is relevant to understand whether assemblages in different sediment types e.g. fine, muddy sediment, are more sensitive to harvesting than other types, e.g. coarse, sandy sediment. Further experimental studies could test this hypothesis for tidal flat assemblages.

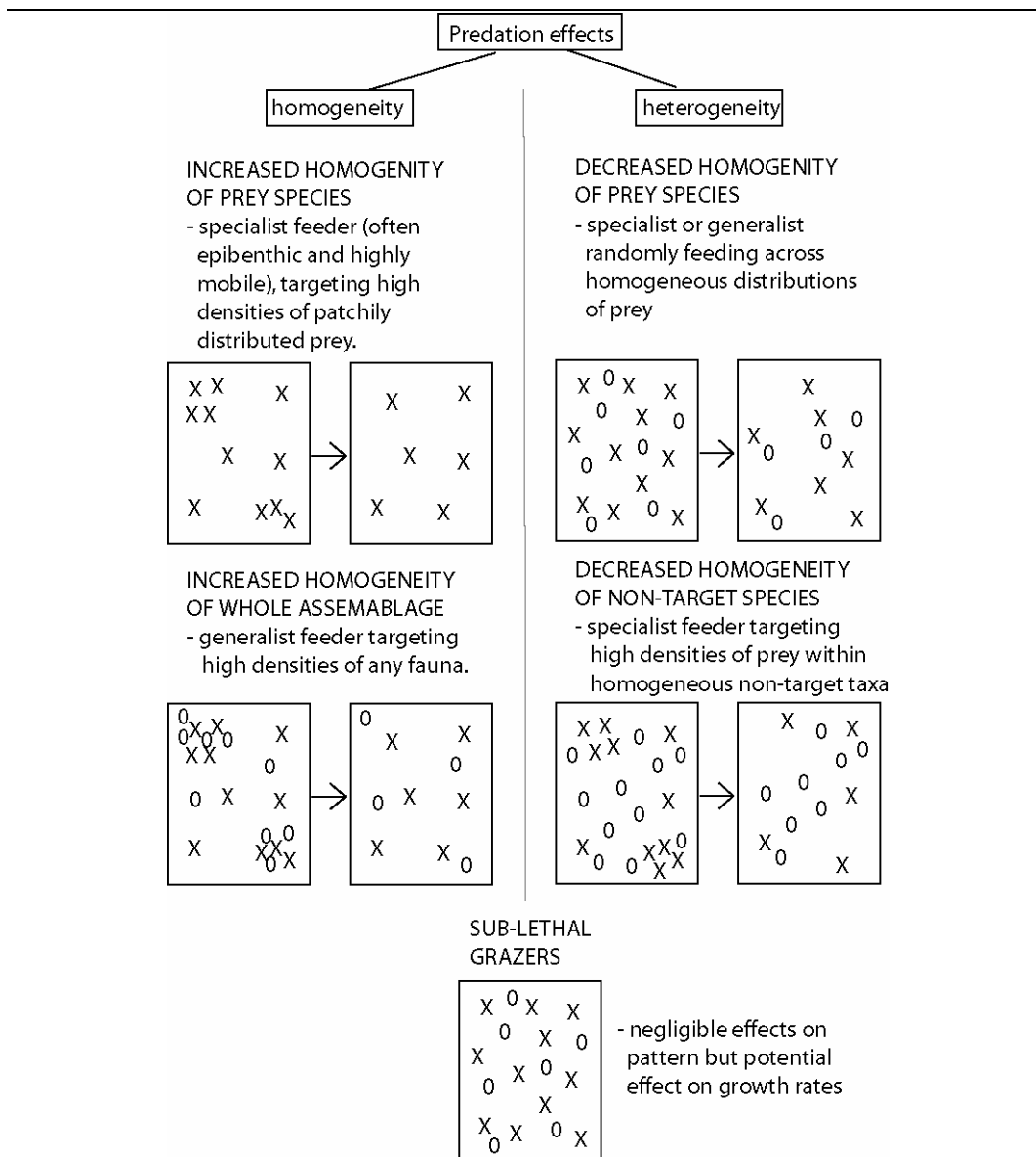


Figure 6-1. some scenarios of potential opposing effects of epibenthic predation on the spatial patterns of prey (x) and non-target (o) species in a tidal flat.

6.3.1.3. Interactions of bait-harvesting with other structuring processes

As described in Chapter 1, there is a highly complex network of ecological processes that could further interact with, and change the disturbance effects of bait harvesting. Two processes, nutrient dynamics and trophic interactions, are discussed here and further processes are mentioned.

The observed shift to more suspension and deposit feeding species following no-take zoning might be equated to a functional shift on the tidal flat (Thrush and Dayton 2002). It would be too speculative to predict the flow on effects from these findings, but it is important to recognise that such functional changes might affect sediment structure through bio-genic engineering, which in turn has consequences for the nutrient dynamics in tidal flats (Bird et al. 2000). For example, the importance of tidal flats, and biogenic structures in particular, to the phosphorus cycle in estuaries has been demonstrated (Lillebo et al. 2004), and different types of sediment dwelling fauna can have strong effects on the exchange of nutrients across the sediment-water interface (Heilskov et al. 2006, Webb and Eyre 2004).

The assemblage shifts in the present study potentially also have consequences for trophic level interactions, and of interest to Marine Protected Area management is whether there are potential benefits to recreational fish stocks or habitat productivity. Very strong trophic cascades have been documented in rocky reef marine protected areas (Langlois et al. 2005, Shears and Babcock 2002), while in contrast, there has been limited effort in identifying trophic cascades in soft-sediment habitats (Pinnegar et al. 2000). Of relevance to the findings in the present study, and crucial to many of the commercial and recreational fisheries, is that the trophic link between sediment and

pelagic diversity is considered to be tight in shallow waters dominated by selective scavengers (Austen et al. 2002). Therefore, it might be possible to measure consequent effects on a range of fish species that were observed feeding during higher tides across the tidal flat. This is a major gap that needs to be addressed, specifically as different prey resources are potentially partitioned and targeted by different fish species (Hines et al. 1997, Robertson 1984, Tyler 1999).

Additional to the potential functional and trophic level consequences of changes to tidal flat assemblages, there are diverse implications for synergistic effects with other structuring processes, including interspecific and intraspecific interactions (Holt et al. 2004, Thrush et al. 1997b, Thrush et al. 1997), juvenile recruitment and dispersal (Bouma et al. 2001, Filho et al. 2005, Rossi et al. 2007, Thrush et al. 1997b) or supply of nutrients. Thus the assemblage shift in this present study might not be replicated at another tidal flat if, for example, the nutrient supply is different. In microcosm experiments elsewhere, macrofauna exhibited responses to interactions between nutrients and physical disturbance. The highest diversity was found at low levels of nutrients and physical disturbance, while the heterogeneity of assemblages increased at high nutrient levels, and physical disturbance reduced the effects of nutrient enrichment. (Dynamic equilibrium model (Austen and Widdicombe 2006)). Interactive effects may be unexpected, for example, Skilleter (2006) suggested that an increase in the abundance of hermit crabs in a tidal flat was attributed to an increase in shells available to inhabit after sediment disturbance events.

On an ecological note but of relevance to structuring forces, this study associated decreased disturbance with increased in assemblage homogeneity, and reduced

abundance of potentially opportunistic taxa. This is consistent with other studies (Huxham et al. 2000) that test the intermediate disturbance hypothesis (Connell 1978). This assumes that bait harvesting had a high level of physical disturbance, and that the species that decreased in number were opportunistic, and that the bivalves, capitellid and spionid polychaetes that increased in number, were not. It should be noted that what constitutes an opportunistic species depends on the disturbance type under investigation. Many studies (eg. Norkko et al. (2006, Stark 1998)), regard capitellid polychaetes as opportunists for anoxic and nutrient enrichment disturbances, however here I regard them as sensitive to physical disturbance and they are therefore the antithesis of opportunistic with regard to physical disturbance.

6.3.1.4. Relevance of potential implications for adaptive management

The present study demonstrates that physical disturbance by human bait harvesting potentially impacts tidal flat invertebrate assemblages, beyond the natural levels of disturbance by physical or biological processes such as water movement or predation, and that MPAs are an appropriate form of conservation management for this habitat type. The benefits of conservation of tidal flats includes assumed resilience and baseline referencing as for any other habitats (Allison et al. 2003, Bevilacqua et al. 2006), but specifically, vital ecological services might be affected by cumulative impacts in this habitat type on a broad scale (Austen et al. 2002).

Effects from protection of tidal flat habitat might include changes to important functional processes or trophic cascades as described above (6.1.1.) with consequences for fish assemblages, water quality and/or productivity. Predatory species of the benthic fauna may be affected, including changes to the potential food supply for

recreational and commercial fish species (eg. silver bream (Matthews and Fairweather 2006), flat head, whiting and mullet which were regularly observed), and also for migratory shore birds for which tidal flats are an important food source (Skilleter 2004).

In addition to effects of changes to food supply, shore birds can be disturbed in their feeding just by the presence of people in their feeding habitat or of disturbed sediment (Skilleter 2004). There have been anecdotal suggestions (Marine Park staff and local residents) that feeding birds are more common on the protected tidal flat in the years following no-take zoning, however this has not been quantified and is difficult to establish after the zoning.

6.4. Future directions for ecological and MPA research on tidal flats

The following suggestions for future directions of research in tidal flats are not comprehensive, but focus on the key findings and questions raised in this study, namely the ecological processes in tidal flats, as well as research that is relevant for the continued improvement of planning and management of marine protected areas.

6.4.1. Ecological questions

The most obvious question arising from this study is how do effects of human disturbance interact with other ecological processes. These include both structuring and functional processes, as well as trophic interactions.

The shift in assemblage composition demonstrated here, following no-take protection of a tidal flat, has clear potential for trophic effects, and thus provides a starting point for a much needed focus on the trophic interactions in soft-sediment assemblages (Pinnegar

et al. 2000). Related to MPAs research is the concept of trophic cascades that have been demonstrated as an outcome of MPA zoning. Such studies have been reviewed, and as stated in the Introduction, are dominated by shallow rocky reef habitat. Isotope analysis is a tool that demonstrated great potential for identifying trophic relationships, specifically for shallow sediment assemblages (Herman et al. 2000, Mazumder 2004).

The findings here indicate that there is potentially an increased source of juvenile bivalves for recruitment to adult stage habitat, in particular for the bivalve species *Eumarcia fumigata* and *Soletellina alba* and smaller polychaetes. The value of these species ecologically, including as a preferred food source for any species, is of relevance in determining further implications of the observed biological assemblage shift. This is relevant to the overall need for a better understanding of larval dispersal, including sources, sinks and methods of dispersal (Grantham et al. 2003, Largier 2003, Palumbi 2003).

Both of the above areas of research, trophic interactions and sources and sinks of juveniles, fall under the umbrella of connectivity between habitats, particularly in the case of trophic interactions where nektonic predators are forced to move to other habitat at low tides. Connectivity of tidal flats to other habitats and other tidal flats is an area that has received little attention, but which is vital for creating adequate networks of MPAs.

Further, the water quality and nutrient availability in estuarine water has implications for the productivity of systems, and also affects industries such as shell fish farming. As the assemblage shift identified in this study might influence water quality and nutrient availability, determining the effect of the assemblage shift on these parameters

would be a valuable contribution to the ecology of tidal flats and the management of estuarine systems.

6.4.2. Bioregional description of biodiversity and CAR principles

This study also shows that the MPA goal of representativeness within the CAR principles (Chapter 1) was achieved for similar habitat within a 30km distance.

However the goals of adequacy and comprehensiveness were not addressed here. These goals require that similar studies are repeated at larger spatial scales to determine if the bioregional zoning is relevant to the faunal diversity of tidal flats, that further types of tidal flats in morphologically distinct estuarine types and along salinity gradients need to be sampled, and the proportion of bioregional habitat represented in no-take zones need to be determined and assessed for adequate maintenance of ecological processes. This will also require a better understanding of ecological processes and connectivity of this habitat to others. In addition, global patterns of tidal flat diversity have been estimated as well (Attrill et al. 2001, Dittmann 2000), although due to limited data for reliable quantification of patterns, some of these are slightly contradictory in their findings. Bioregional data would further contribute to a better understanding of global patterns.

Marine bioregionalisation for conservation purposes is largely based on geomorphological or physical features, eg. geology, water currents and temperature gradients, with some reference to biological features that are easily seen from remote sensing sources (eg. mangroves, seagrass or coral reefs) (Mumby 2001). This strategy is highly appropriate considering that selection of areas for marine conservation must occur in the short term and that full scale biological surveys at bioregional scales are

both costly and will take many decades to develop. Similarly, tidal flat biodiversity within and between bioregions is not yet well understood in Australia.

Despite this, it must be recognised that different habitats and biological assemblages may not reflect the same spatial patterns of species turnover (beta-diversity) as the currently demarcated bioregionalisation. During this study, a range of biological data sources (eg. museum databases and unpublished reports) were identified that provide information on soft-sediment tidal habitat similar to that described here. This information could be used to qualify the relevance of bioregions for estuarine invertebrates, similar to studies of other habitats and faunal assemblages (Connell and Irving 2008, O'Hara and Poore 2000). Although there are limitations in using collections of data from different times, using different methods, levels of expertise, and acknowledging the implications of data selection, transformation and resolution discussed in Chapter 5.2, collation of such data bases would provide some valuable insights of the biological patterns at large spatial scales, for example across bioregions.

6.4.3. Tidal flat as a surrogate for biodiversity

Efforts to distinguish estuarine biodiversity based on estuarine geomorphology within bioregions have been demonstrated for a number of biological assemblages within NSW, and Australia (Edgar and Barrett 2002, Saintilan 2004). Consequently, it is recognised that tidal flat habitat would not be an appropriate surrogate for all tidal flats and limited the research to tidal flats of permanently open estuaries that were close to the entrance and therefore marine dominated.

This identifies the need to expand on the habitat definitions for tidal flats as habitat

mapping forms the basis for most Marine Park planning and management in NSW. This would be analogous to seagrass habitat being expanded to *Zostera*, *Posidonia*, *Ruppia* and *Halophila* habitats in NSW Marine Parks.

6.4.4. Indicator species

The resources and effort required for biological surveys are considerable, and therefore indicator species are an appealing compromise for spatial studies of biological assemblages. The apparent dominance of certain species in this study might help to identify a list of disturbance indicators, or to characterise the types of tidal flats that exist in different situations. For example, the common amphipod in this study (*Urohaustorius metungi*), was found to be common only to permanently open estuaries and absent from assemblages of intermittently open estuaries in another study (Hastie and Smith 2006).

This immediately provides information for conservation managers in identifying that representative MPA networks must include both types of estuaries. In addition it provides direction for the types of sampling designs or indicator organisms that would be most effective for describing assemblage patterns along environmental gradients (eg. upstream and downstream) as well as latitude and longitude (bioregional).

6.4.5. Relevant measures of ecological diversity

It must also be recognised that the selection of marine macro-invertebrate taxonomy used in this study is a narrow ecological measure, and ecological patterns may or may not be reflected within this selection, even if other ecological components of the sediment are affected. This could include measures of meiofaunal structure and production (Austen and Widdicombe 2006), and higher trophic levels. Functional group categorisation is one valid aggregation of data to examine in the context of impact studies (Bremner et al. 2006, Olsford and Somerfield 2000, Roth and Wilson 1998). Functional group categorisation of species in tidal flat habitat has not been attempted here, however it should be recognised that a degree of functional coherence is anticipated by the use of higher taxonomic resolution (Warwick 1993).

Other appropriate measures in addition to multivariate assemblage patterns should be considered. This is illustrated in a recent study by Roberts and Connell (2008), where the effect of increased nutrients was detected as an increase in the biomass of algae, but also as a decrease in phylogenetic diversity, and not detected at all using individual species or the Shannon Diversity index. Similarly, Morello and co-workers (2005) detected changes in polychaete biomass following disturbance events. In the case study here, the effect of removal of physical disturbance was not evident by using the Shannon Diversity index, but it was using other diversity indices such as Pielou's evenness, and also for specific species (Chapter 4). Additional univariate measures that might be considered are biomass and size measures of species, including quantifying primary production on the sediment surface (Webb and Eyre 2004). Abiotic variables such as sediment properties or nutrient regeneration might be affected in soft-sediment

environments (Heilskov et al. 2006), and thus are candidates for ecological measures in impact studies.

This study successfully addressed the four aims of the project, and I found outcomes of relevance to both planning and management of marine protected areas (MPAs), as well as to the ecology of tidal flat habitat. Thus, I demonstrated the relevance of MPAs as a form of conservation management appropriate to tidal flats. (Hastie and Smith 2006)

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APPENDIX I – TIDAL FLAT
INVERTEBRATE IDENTIFICATION
SHEET (PHOTOS AND
ILLUSTRATIONS)



Bivalvia	Arcidae		<i>Barbatia</i> sp.
	Galeommatidae		<i>Arthritica helsmsii</i>
	Galeommatidae		<i>Mysella anomala</i>
	Galeommatidae		sp. #3 (Not Lasea)
	Hiatellidae		<i>Hiatella australis</i> Lamarck 1818
	Lucinidae		<i>Wallucina</i> sp.
	Laternulidae		<i>Laternulidae</i> sp.
	Mactridae		<i>Mactridae</i> sp. juv
	Mesodesmatidae		<i>Mesodesmatidae</i> sp.
	Psammobiidae		<i>Soletellina alba</i>
	Tellinidae		<i>Tellina</i> sp. #1
	Tellinidae		<i>Tellina deltoidalis</i> juv.
	Thyasiridae		<i>Thyasiridae</i>
	Veneridae		<i>Eumarcia fumigata</i>
	Veneridae		<i>Gouldiopa australis</i>
	Veneridae		<i>Veneridae</i> sp. #3
	Veneridae		<i>Veneridae</i> sp. #4
Gastropoda	Batillariidae		<i>Pyrasmus ebeninus</i> Bruguiere, 1792
	Buccinidae	(prev.	
	Nassariidae)		<i>Nassarius jonasii</i> Dunker 1846
	Buccinidae	(prev.	
	Nassariidae)		<i>Nassarius nigellus</i> Reeve 1854
Opisthobranchia	Iravadiidae		<i>Nozeba topaziaca</i>
	Naticidae		<i>Naticidae</i> juv sp.
	Bullidae		<i>Bulla quoyii</i> juv.
	Cylichnidae		<i>Tornatina</i> sp.
	Philinidae		<i>Philine</i> sp.
Decapoda	Callianassidae		<i>Biffarius arenosa</i>
	Callianassidae		<i>Trypaea australiensis</i>
	Callianassidae		<i>Biffarius</i> c.f. <i>Ceramica</i>
	Crangonidae		<i>Crangonidae</i>
	Diogenidae		<i>Diogenidae</i>
	Mictyridae		<i>Mictyris</i> sp.
	Penaeidae		<i>Penaeus plebejus</i>
	Corophidae		<i>Monocorophium acherusicum</i>
Amphipoda	Exoedicerotidae		<i>Exoediceros fossor</i> Stimpson 1856
	Paracalliopiidae		<i>Paracalliope lowryi</i> Barnard & Drummond 1992
	Photidae		<i>Gammaropsis</i> sp.
	Phoxocephalidae		<i>Limnoporeia kingi</i> Fearn-Wannan 1968
	Urohaustoriidae		<i>Urohaustorius metungi</i> Fearn-Wannan, 1968
Isopoda	Actaeciidae		<i>Actaecia</i> sp.
	Bopyridae		<i>Bopyridae</i>
Oligochaeta	Cirolanidae		<i>Pseudolana towrae</i> Bruce 1983
	Oligochaeta		Oligochaeta

Polychaeta	Ampharetidae	<i>Ampharetidae</i>
	Capitellidae	<i>Barantolla lepte</i>
	Capitellidae	<i>Capitella</i> sp.
	Capitellidae	<i>Heteromastus filiformis</i>
	Capitellidae	<i>Mediomastus australiensis</i>
	Capitellidae	<i>Notomastus/Chrysosetos</i>
	Glyceridae	<i>Glycera tridactyla</i>
	Hesionidae	<i>Heteropodarke</i> sp.
	Lumbrinereidae	<i>Lumbrineris</i> sp. #1
	Lumbrinereidae	<i>Lumbrineris</i> sp. #2
	Magelonidae	<i>Magelona dakini</i> Jones 1978
	Nephtyidae	<i>Nephtys australiensis</i>
	Nephtyidae	<i>Nephtys longipes</i>
	Nereididae	<i>Australonereis ehlersi</i>
	Opheliidae	<i>Armandia intermedia</i>
	Opheliidae	<i>Lobochesis bibrancha</i> Hutchings & Murray 1984
	Orbiniidae	<i>Leitoscoloplos</i> sp.
	Orbiniidae	<i>Leodamas johnstonei</i>
	Orbiniidae	<i>Scoloplos simplex</i>
	Owenidae	<i>Owenia fusiformis</i> Delle Chiage 1844
	Phyllodocidae	<i>Phyllodoce</i> sp.
	Phyllodocidae	<i>Eumida</i> sp.
	Sabellidae	<i>Desdemona aniara</i>
	Sigalionidae	<i>Sigalion ovigerum</i>
	Sigalionidae	<i>Sthenelais</i> sp.
	Spionidae	<i>Spio</i> sp. (<i>pacifica</i>)
	Spionidae	<i>Polydora</i> gen.
	Spionidae	<i>Carazziella</i> sp. (<i>victoriensis</i>)
	Spionidae	<i>Pseudopolydora</i> sp.
	Spionidae	<i>Prionospio aucklandia</i>
	Spionidae	<i>Prionospio yuriei</i>
	Spionidae	<i>Scolecopsis</i> sp. #1
Anthozoa	Order Actiniaria	<i>burrowing anemone</i> sp.
Chaetognatha	Chaetognath	<i>Chaetognath</i>
Nemertea	Nemertea	<i>Nemertea</i>
Phoronida	Phoronida	<i>Phoronida</i>
	Sipuncula	<i>Sipuncula</i>
Insecta	Insecta	insect #1
Insecta	Insecta	insect #2

APPENDIX II – QUANTITATIVE DATA

Please see attached disk