



Water for a Healthy Country

The Distribution and Abundance of Macrobenthic Invertebrates in the Murray Mouth and Coorong Lagoons 2006 to 2008

Alec Rolston and Sabine Dittmann

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Foreword

The environmental assets of the Coorong, Lower Lakes and Murray Mouth (CLLAMM) region in South Australia are currently under threat as a result of ongoing changes in the hydrological regime of the River Murray, at the end of the Murray-Darling Basin. While a number of initiatives are underway to halt or reverse this environmental decline, rehabilitation efforts are hampered by the lack of knowledge about the links between flows and ecological responses in the system.

The CLLAMM program is a collaborative research effort that aims to produce a decision-support framework for environmental flow management for the CLLAMM region. This involves research to understand the links between the key ecosystem drivers for the region (such as water level and salinity) and key ecological processes (generation of bird habitat, fish recruitment, etc). A second step involves the development of tools to predict how ecological communities will respond to manipulations of the “management levers” for environmental flows in the region. These levers include flow releases from upstream reservoirs, the Lower Lakes barrages, and the Upper South-East Drainage scheme, and dredging of the Murray Mouth. The framework aims to evaluate the environmental trade-offs for different scenarios of manipulation of management levers, as well as different future climate scenarios for the Murray-Darling Basin.

One of the most challenging tasks in the development of the framework is predicting the response of ecological communities to future changes in environmental conditions in the CLLAMM region. The CLLAMMecology Research Cluster is a partnership between CSIRO, the University of Adelaide, Flinders University and SARDI Aquatic Sciences that is supported through CSIRO’s Flagship Collaboration Fund. CLLAMMecology brings together a range in skills in theoretical and applied ecology with the aim to produce a new generation of ecological response models for the CLLAMM region.

This report is part of a series summarising the output from the CLLAMMecology Research Cluster. Previous reports and additional information about the program can be found at <http://www.csiro.au/partnerships/CLLAMMecologyCluster.html>

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This work could not have been completed without the hard work of the multitude of field volunteers that assisted sampling, organism identification and counting: Ryan Baring, Cathy Cooney, Angela Dutton, Jason Earl, Ben Hamilton, Rebecca Lester and Matt Nelson.

Executive Summary

1. The Coorong, Lower Lakes and Murray Mouth (CLLAMM) region, at the terminus of the River Murray, are wetlands of international importance as recognised by their Ramsar status, which was designated in 1985. The region is also one of six icon sites identified by the Murray-Darling Basin Commission's Living Murray Initiative. Low flows entering Lake Alexandrina have led to no water being released over the barrages since 2005, resulting in the degradation of the system in terms of increasing salinity, decreasing bird and fish numbers and the loss of important habitats.
2. This study investigated the distribution and abundance of benthic macroinvertebrates throughout the Murray Mouth and Coorong regions from December 2006 – October 2007, and explored the response of these invertebrates to experimental translocation to sites of different salinity and periods of mudflat exposure in January – March 2008.
4. Typically, salinity in the Murray Mouth region remained approximately marine, before increasing rapidly with distance southwards into the North and South Lagoon regions, with peak salinities in the South Lagoon frequently over 120 g/L. Salinity was the driving environmental variable found to best explain the distribution of both adult and juvenile macrobenthos, with sediment organic content also an important variable.
5. Adult and juvenile macrobenthic diversity and abundance were greatest in the Murray Mouth region before decreasing in the North Lagoon. Greatest abundance of both adult and juvenile macrobenthos occurred at Pelican Point, which was also distinct from other sites in terms of sediment grain size. In December 2006, only insect larvae were present in the South Lagoon. No taxa were present in the South Lagoon in January and March 2007.
6. Distinct adult and juvenile macrobenthic communities were present in the Murray Mouth, North Lagoon and South Lagoon, and these did not change significantly with time. Macrobenthic abundances in the Murray Mouth region were dominated by Polychaeta (particularly *Capitella* spp.), Amphipoda and the micro-bivalve *Arthritica helmsi*.
7. The juvenile macrobenthic fauna was dominated by four species: the polychaete worms *Capitella* spp. and *Simplisetia aequisetis*, the micro-bivalve *Arthritica helmsi* and the Chironomid larva *Tanytarsus barbatus*. Juveniles of each of the four dominant species were present in the system throughout the year, whilst rarer species such as the polychaetes *Boccardiella limnicola* and *Nephtys australiensis* showed more seasonal abundances.
8. Macrobenthic abundances decreased significantly at high mudflat exposures in the Murray Mouth region, and at high and medium exposures in the North Lagoon region from December 2006 – March 2007.
9. Experimental translocations of macroinvertebrate fauna in sediment from areas of high salinity to low salinity, and from areas of high mudflat exposure to low exposure led to an increase in species diversity and abundance. The converse was true for the reciprocal translocations. Macrobenthic invertebrates were unable to survive in exposed sediment for one week without inundation.
10. A reduction in salinity levels and an increase in water level are needed in both the North and South Lagoons in order for juvenile macrobenthos to recolonise mudflats where macrobenthic invertebrates are currently absent.

1. Introduction

1.1. CLLAMMecology Invertebrate Key Species Project

The Invertebrate Key Species Responses Research Plan, along with the research plans for the further key species responses, is described in the CLLAMMecology Project Research Plans document for September 2006 – June 2007 (30th September 2006). This document is complementary to the Cluster project Plans presented in Schedule 2 of the CLLAMMecology Research Collaboration Agreement (Geddes, 2006). The objectives, component studies and methods described in the Research Plan are summarised below.

The overall objectives of the Key Species Response Projects were to document the responses of a suite of signature species to changes in water regimes and to subsequently develop response models. Such data are to be provided to the CLLAMM Futures project to develop system models that will allow the ecological benefits of different water management scenarios in the Coorong to be assessed across four functional groups: Aquatic birds; fish; aquatic invertebrates, and aquatic plants.

As a result of the low flow conditions of the River Murray from 2006 to present, the assessment of organism responses to changes in water regimes was not possible. Subsequently, the invertebrate key species project assessed the response of species to changes in salinity, nutrients, period of inundation and sediment composition through changes in:

- (i) Abundance;
- (ii) Distribution;
- (iii) Growth/survival;
- (iv) Reproduction;
- (v) Recruitment;
- (vi) Community structure, and
- (vii) Biomass.

These assessments have been performed through the following component studies:

1. Documenting the distribution and abundance of key invertebrates across the region through time, using measures of environmental variables (nutrient levels, salinity, sediment composition) to define habitat requirements and tolerance limits of key invertebrate species;
2. Measuring the performances (biomass, abundances, size frequency distributions, recruitment success) of key invertebrate populations *in situ* within the Coorong where environmental conditions (salinity, nutrient levels, sediment composition) differed, documenting how aspects of performance changed through time;
3. Experimentally documenting species-specific tolerance and performance levels to key environmental parameters (e.g. salinity, sediment composition, tidal inundation) through

in situ experimental manipulations, comparing performances in experimental plots to those in control plots.¹

1.2. Macrobenthic Invertebrates

Macrobenthic invertebrates are important components of estuarine ecosystems, directly processing a significant portion of system-wide primary production, and providing an important food resource for crustaceans, fish and birds (Herman *et al.*, 1999). Estuarine organisms can be described as bioindicators in three ways (Wilson, 1994), as: 1) indicators of a defined set of environmental conditions; 2) indicators of contaminant loads on the system; and 3) indicators of the overall health of the system. Throughout the world, a large amount of research has been conducted on the abundance, distribution and productivity of macrobenthic invertebrates, with the majority of studies (with regards to secondary productivity) carried out in the Northern Hemisphere (Cusson and Bourget, 2005). A geographical bias also exists with temperate mudflat ecosystems being studied more frequently than tropical systems (Dittmann, 2000).

Currie and Small (2006) summarise the multiple biotic, abiotic and human-induced factors that affect intertidal macrobenthic invertebrates: Physical influences include water depth, sediment structure, salinity and hydrology; Biological factors include predation, competition and recruitment; Human-induced factors include organic enrichment, chemical pollution and commercial fishing activity. Water inundation periods, driven by the combined effects of tides and wind (Millet and Guelorget, 1994) also determine macrobenthic invertebrate survival, abundance and distribution (Hummel *et al.*, 1986, 1988). As a result of these influencing factors, macrobenthic invertebrate distribution often exhibits high spatial variability, particularly with regards to salinity gradients in estuaries (e.g. Mannino and Montagna, 1997; Ysebaert *et al.*, 1998; Dittmann *et al.*, 2006c).

Estuaries are transitional zones between freshwater and marine environments (Levin *et al.*, 2001) where species diversity is often driven by physico-chemical variables such as salinity (Attrill & Rundle, 2002, Giberto *et al.*, 2007). Macrobenthic assemblages can frequently be classified into three separate communities along the estuarine salinity gradient: a marine community in the polyhaline zone; a brackish community in the mesohaline zone; a third community in the oligohaline and freshwater mudflat zones (Mannino and Montagna, 1997; Ysebaert *et al.*, 1998, Giberto *et al.*, 2007). However within these zones, species' distributions are not static as the zonal boundaries fluctuate in time and space, particularly within the mesohaline and oligohaline zones (Chapman and Brinkhurst, 1981; Ysebaert *et al.*, 1998 and references therein). Yet Sanders *et al.* (1965) found that sediment interstitial salinities are different from and more constant than water column salinities and that infaunal benthos are not directly affected by water column salinities. The rate and magnitude of salinity change may primarily determine species distribution in an estuary rather than the salinity gradient itself (Sanders *et al.*, 1965). Salinity variations have been shown to induce regime shifts and affect zooplankton communities in shallow brackish lagoons (Jeppesen *et al.*, 2007).

While salinity and sediment organic content are the principle factors structuring species diversity and community composition of estuarine macrobenthic invertebrates (Hirst, 2004; Currie and Small, 2006; Hirst and Kilpatrick, 2007), the entrance status of estuaries (open/ intermittently closed) is also important in explaining community composition of benthic macrofauna (Hirst,

¹ Further experiments investigating the effects of predators (birds and fish) on the abundance of invertebrates have been part of a separate PhD project by Justine Keuning, Flinders University, and are not presented in this report.

2004; Hastie and Smith, 2006), largely because of the subsequent changes in salinity, recruitment patterns and the survival of marine species. The period of inundation is an important factor in determining macrobenthic invertebrate community composition as benthic organisms in the intertidal zone have adapted to varying periods of emersion and submersion (Hummel *et al.*, 1986 and 1988).

Australian estuaries are primarily marine-dominated environments, being subjected to major salinity changes only after heavy rains and during flood conditions (Hutchings, 1999). Gillanders and Kingsford (2002) reviewed the effects of freshwater on estuarine and coastal habitats: Heavy mortalities of molluscs and polychaetes have been observed following large volumes of freshwater input after rainfall. Such effects were also observed following the freshwater release across the Ewe Island and Boundary Creek barrages in the Murray Mouth region (Dittmann *et al.*, 2006a). Gillanders and Kingsford (2002) also state that reduced salinities have a negative impact on polychaete larval growth, and that the vertical migration of larvae to facilitate inshore and offshore transport may be influenced by lowered salinities and altered flows following freshwater input. Changes in turbidity, sediment deposition, nutrient load and contaminant load will also affect benthic macroinvertebrates following increased freshwater input (Gillanders and Kingsford, 2002).

In estuaries that range widely in their morphological and physicochemical characteristics, the benthic macroinvertebrate fauna is frequently dominated by very few species (Kanandjembo *et al.*, 2001 and references therein; Currie and Small, 2006). Low freshwater inputs into estuaries have been shown to have detrimental effects on macroinvertebrate communities (Attrill *et al.*, 1996, Attrill and Power, 2000), and seasonal changes in hydrology can largely affect the dominant benthic macrofauna (Kanandjembo *et al.*, 2001). However, such changes in hydrology may result in more significant changes in biomass and density than in community composition (Rozas *et al.*, 2005).

The effects of drought on aquatic species may be severe and long-term, resulting in population bottlenecks as a result of limited gene flow and dispersal (Humphries and Baldwin, 2003). Estuarine macrobenthic invertebrate communities have been shown to respond to hydrological changes in Australia, whether as a result of seasonality (Kanandjembo *et al.*, 2001, Dye and Barros, 2005) or prolonged drought (Hastie and Smith, 2006). However, a moderate increase in freshwater flow during a drought period may not lead to significant macrobenthic invertebrate community change (Dittmann *et al.*, 2006a). When comparing intermittent estuaries with permanently open estuaries, Hastie and Smith (2006) agreed that elevated salinities favoured an increase in marine associated fauna and a simultaneous decline in some non-marine species. The authors hypothesise that the size of an estuary's catchment is primarily driving benthic community patterns. The St. Lucia estuary, South Africa, draws large parallels to the Coorong in that as a result of prolonged drought and subsequent low flows, the estuary mouth closed for a significant period, leading to hypersaline conditions developing in some regions of the estuary (Pillay and Perissinotto, 2008). No water flow during the drought's peak lead to distinctive macrofaunal communities existing between regions and habitats probably caused by the prevention of planktonic exchange and subsequent restrictions in larval colonisation of physically separated habitats. The diversity and abundance of macrofaunal organisms decrease in response to drought-induced salinity changes (Attrill *et al.*, 1996; Attrill and Power, 2000) leading Dittmann *et al.* (2006c) to hypothesize that an increase in freshwater flow into the estuary of the River Murray would increase the abundances and diversity of the estuarine macrofauna.

Reproductive rates and survival probabilities are both influenced by an individual's ability to move/disperse, or conversely to remain at the same location (Bennetts *et al.*, 2001). The colonisation of newly available sediment by marine benthic organisms is largely influenced by

the settlement of larvae or the migration of juveniles (Lu and Wu, 2007 and references therein). The dispersal strategy of marine benthic organisms may be classified as brooded, lecithotrophic, and planktotrophic, with such strategies varying in their distant dispersal capabilities (Watson and Barnes, 2004). However it is the timing of initial colonisation rather than the life history traits of benthic organisms that may ultimately determine successful colonisation of suitable habitat, at least in sub-tropical waters (Lu and Wu, 2007). The seasonality of colonisation can be controlled by wind, the numbers of colonisers and the mode of colonisation (Ford *et al.*, 1999 and references therein). Climate also plays a vital role in colonisation with severe winters, for example, having significant effects on the infaunal populations of the Wadden Sea (Strasser, 2000, Strasser and Pieloth, 2001). Burkovsky *et al.* (1997) found that the species composition of juveniles appeared to be similar to the composition of adults in a Russian mudflat. The absence of adults of a species in that study resulted in the absence of juveniles, regardless of the presence of pelagic larvae of the same species in the near-shore water column.

Post larvae and juveniles have been shown to exhibit strong seasonality in abundance in both temperate and sub-tropical environments (Zajac and Whitlatch, 1982a, b; Wu and Shin, 1997; Lu and Wu, 2007), although some species, such as the opportunistic polychaete *Capitella capitata*, can be present throughout the year (Lu and Wu, 2007). Sediment characteristics such as particle size and organic content have been shown to have a significant effect on colonisation by bivalves and gastropods but not for polychaetes and amphipods (Wu and Shin, 1997). However, Zajac and Whitlatch (1982a) identified a variety of physical and biological factors, such as spawning periods, food resource levels and resuspension that may also contribute to seasonal variability in infauna abundance. Species distributions may be importantly determined by factors such as larval supply and particulate flux which are in turn directly influenced by near-bed flow conditions (Snelgrove and Butman, 1994). Also, many frequently common macrofaunal organisms are able to recolonise sediment by post-settlement movement, although bedload traps set up by Ford *et al.* (1999) failed to highlight significant changes in species composition of post-settlement movement in the Papanui inlet, New Zealand. Indeed, Snelgrove and Butman (1994) suggest that no single factor may be responsible for determining soft-sediment benthic community settlement and colonisation.

1.3. Macrobenthic Invertebrates in the Coorong

Low river flows in the 1930s that led to salt water intrusions into Lakes Alexandrina and Albert contributed to the building of the five barrages in the late 1930s (Walker, 2002). These barrages prevent marine incursion into the freshwater system and have transformed the Lower Lakes into freshwater bodies with permanently elevated water levels (Cann, 2000). Subsequent extensive exploitation of this resource has reduced freshwater discharge from the River Murray to the sea by 75% (Bourman and Barnett, 1995). Volumes of sediment deposited at the Mouth lead to almost-complete closure in 1981 and 1999 (Shuttleworth *et al.*, 2005; Walker, 2002). Extended closure of the barrages as a result of low flows of the River Murray produced a build up of sand which led to the near closure of the Murray Mouth and a sand-pumping project began in October 2003 to maintain connectivity with the southern ocean. The dredging is currently ongoing.

The modified hydrodynamics have changed the estuarine character of the entire system, affecting all ecosystem components including benthic invertebrates. The deterioration of the Coorong lagoons, particularly over the past five years (Geddes, 2003, 2005 a, b and c; Dittmann *et al.*, 2005, 2006a, b and c; Dittmann and Nelson, 2007, Dittmann *et al.*, 2008; Geddes and Francis, 2008) suggests that the general health of the system can be assessed through studies of macrobenthic invertebrates, particularly with regards to changes in community structure (Wilson, 1994).

Geddes and Butler (1984) and Kangas and Geddes (1984) were the first published studies to investigate the distribution of macrobenthos and amphipods respectively in the Coorong. In 1982, there was no outflow over the barrages, and salinities in the North Lagoon reached a maximum of 80 ‰ at the more southerly sites, whilst in the South Lagoon, salinities ranged between 90-100 ‰ (Geddes and Butler, 1984). When comparing such salinities to the historical record, the authors state that the conditions of the system represented an extremely hypersaline phase in the long term salinity fluctuations of the Coorong. The low diversity of the Coorong was suggested to reflect the extreme salinity fluctuations in the system, particularly as many common estuarine macroinvertebrate species were absent. The salinity gradient from the North Lagoon into the South Lagoon was determined to drive the species distribution in the system, with the infaunal community changing from an estuarine assemblage to one dominated by halophytic dipterans and salt-lake crustaceans once salinities increased above 70 ‰ (Geddes and Butler, 1984). In comparison with other estuaries that have a much greater amphipod assemblage, in the Coorong only three common species of amphipods occur, the distributions of which are generally restricted to salinities less than 53 ‰ (Kangas and Geddes, 1984).

Following the reduced flows of the River Murray in 1982 (Geddes and Butler, 1984), Geddes (1987) investigated the effects of above average flow in the River Murray in 1983-84 on the salinity and distribution of the biota in the Coorong Lagoons. By mid-1984, salinities were found to be brackish (< 30 ‰) in the North Lagoon and moderately hypersaline in the South Lagoon (55-70 ‰). The distribution of the estuarine macrofaunal assemblage increased to the southern extent of the North Lagoon, although species richness remained low, with only two previously uncollected species present. The South Lagoon however continued to be dominated by the higher salinity tolerant species, such as halophytic diptera, despite salinity being within the tolerance range of many estuarine species during winter-spring of 1984 (Geddes, 1987). However, following the closure of the barrages in 1984, salinities increased quickly to 36-70 ‰ in the North Lagoon, and 80-140 ‰ in the South Lagoon by March 1985, leading Geddes (1987) to suggest that high River Murray flows are needed to maintain an estuarine-marine system in the North Lagoon, and moderately hypersaline salinities in the South Lagoon.

Little further work was conducted on the macrobenthic fauna of the Coorong until Geddes (2003) conducted an ecological survey of eleven sites in the Coorong from Ewe Island to Salt Creek following a period of extended closure of the barrages as a result of low flows of the River Murray. In the high salinities (80-90 ‰) of the South Lagoon, low numbers of chironomid larvae were the only macroinvertebrate fauna present, whilst estuarine-lagoonal macroinvertebrates were only collected in the Murray Mouth and northern part of the North Lagoon (Geddes, 2003). Almost all the macroinvertebrate species recorded by previous studies (Geddes and Butler, 1984; Kangas and Geddes, 1984; Geddes, 1987) were recorded again by Geddes (2003), but their distributions and abundances were more restricted.

A managed barrage release in September-October 2003 aimed to flush and reduce salinities near Goolwa above the barrages, provide freshwater flows to the Goolwa and Coorong channels, assist in flushing the Murray Mouth and provide mixing of freshwater southwards from Pelican Point into the Coorong Lagoons (Geddes, 2005a). Prior to this release, the barrages were closed continuously from December 2001 until the release in September 2003 – the longest barrage closure recorded since 1981. The outflow created estuarine salinities in the Murray Mouth and part of the North Lagoon, yet there was no substantial response in distribution and abundance of estuarine-lagoonal macroinvertebrates. Geddes (2005a) suggested that water releases of 200 – 300 GL can only produce limited spatial and temporal environmental benefits. Further small barrage releases in 2004 and 2005, although increasing freshwater, turbidity, phytoplankton and zooplankton in the system (Geddes, 2005c), had little effect on the macrobenthic invertebrate communities close to the water releases (Dittmann *et al.*, 2006a).

The consistent period of low flow into the Coorong has essentially led to a constriction of the benthic invertebrate macrofauna towards the lower salinities of the Murray Mouth region, leading to a regime shift in the South Lagoon from that of hypersaline lagoon to almost salt lake status. Indeed, the brine shrimp, *Parartemia zietziana*, was collected at Salt Creek in the South Lagoon for the first time ever in the Coorong Lagoons in July 2004 (Geddes, 2005b). During that study, estuarine-lagoonal invertebrates were again restricted to the Murray Mouth region and the northern part of the North Lagoon, despite salinities being within their tolerance ranges further south.

The first truly quantitative survey of macrobenthic invertebrate fauna of the Coorong was performed in 2004 by Dittmann *et al.* (2006b). Subsequent quantitative surveys, that have at times also included the macrobenthic invertebrate fauna of Lakes Alexandrina, have been performed on an annual basis (Dittmann *et al.*, 2006c, Dittmann and Nelson, 2007, Dittmann *et al.*, 2008). Dittmann *et al.* (2008) state that, although the 2007 survey confirmed previously consistent general patterns of benthic diversity and community composition throughout the Murray Mouth and Coorong, changes in benthic fauna (such as loss of amphipods at several sites, and an overall decline in species numbers, abundance and biomass) could be a result of the prolonged period of no flow over the barrages. Salinities in the Murray Mouth in 2007 were higher than in previous years, but in the South Lagoon, salinities in 2006 and 2007 were significantly higher than previously measured. Highest species abundance and biomass over the past four years of sampling was recorded in October/November 2005, shortly after the last water release over the Ewe Island barrage (Dittmann *et al.*, 2008). The authors conclude that during the current period of no flow over the barrages, the ecological benefit of the last water release is now very apparent.

In a recent pilot study of the trophic ecology at Pelican Point, Geddes and Francis (2008) found that both macrobenthic invertebrate species diversity and abundance decreased sharply at depths below 1.5m. Amphipods dominated the diet of small fish such as hardyheads and gobies, whilst juvenile mullet consumed largely *Capitella* and *Phyllodoce* polychaete worms. Adult mullet however had a broader diet, consuming crustaceans, polychaetes, bivalves and plant material. The authors state that consumers of macrophytes such as the amphipod *Melita*, the snail *Hydrobia* and the shrimp *Macrobrachium*, are currently in very low abundance in the system in comparison to the 1980s. This may largely be due to the absence of macrophytes during this 2005 study, with *Ruppia megacarpa* being particularly conspicuous in its absence (Geddes and Francis, 2008).

With the reduction of flows over the barrages, particularly in the past five years, the ecology of the system, in particular the Coorong, has declined significantly in comparison to the 1980s. This study in the Invertebrate Key Species Project in CLLAMMecology documents the macrobenthic invertebrate ecology of the system during a prolonged period of no flow over the barrages. Results are presented in three sections: 1). Macrobenthos of the Coorong; 2). Juvenile macrobenthos of the Coorong and; 3). Sediment Translocation Experiment. Sections one and two involved bi-monthly monitoring surveys from December 2006 – March 2007 and December 2006 – October 2007 respectively. Section three (sediment translocation experiment) assessed the impact of changing salinity levels and mudflat exposure on macrobenthic communities through the translocation of sediment cores to and from areas of high, low and intermediate salinities and high and low mudflat exposures, and was carried out over the summer 2007/2008.

2. Methods

2.1. Study Sites

The studies listed in Section 1.1 have been completed through the surveys of 11 of the 12 CLLAMMecology study sites (Table 1; Figure 1) throughout the Murray Mouth and Coorong regions. CLLAMMecology study site 3, Barker's Knoll, was not sampled in this study due to access difficulties. The studies comprised surveys for macroinvertebrates (on a bi-monthly basis from December 2006 – March 2007: Section 2.3.1), juvenile macroinvertebrates (on a bi-monthly basis from December 2006 – October 2007: Section 2.3.1.; please note the clarification of the term juvenile as used for this study) and a sediment translocation experiment (January – March 2008: Section 2.4).

Table 1: Number, names and coordinates of 11 sites throughout the Coorong. Site 3, Barker Knoll, was not sampled for macroinvertebrates due to access difficulties. Coordinate projection is UTM, datum GDA94, units = metres.

Site Number	Site Name	Coordinates	
		Eastings	Northings
1	Goolwa Channel	303108	6066345
2	Mundoo Channel	308846	6065289
4	Ewe Island	314162	6063065
5	Pelican Point	320835	6059046
6	Mark Point	325992	6055102
7	Long Point	334052	6048558
8	Noonamena	342180	6042458
9	Parnka Point	357818	6027763
10	Villa dei Yumpa	361317	6024279
11	Jack Point	370809	6010971
12	Salt Creek	377655	6000393

For adult macrobenthos, all eleven sites shown in Figure 1 were sampled in December 2006, January 2007 and March 2007. For juvenile macrobenthos, sites 1-11 were sampled in December 2006, January 2007 and March 2007, but due to a lack of juvenile benthos in the southern Coorong at these times, only sites 1-8 inclusive were sampled in May, July and October 2007. Sites 1-5 (Figure 1: Murray Mouth region) were influenced by tides. Water levels at the remaining non-tidal sites (Sites 6-9, North Lagoon region; Sites 10-12, South Lagoon region) varied throughout the year due to rainfall, evaporation and wind direction and strength. No water was released from the Lower Lakes over the barrages during the time frame of these studies.

Sites were divided into three mudflat exposure levels: High, Medium and Low. High exposure samples were taken from within an approximate 10 m distance from either the shoreline (in the tidal/Murray Mouth region) or the high water mark (non-tidal/ North and South Lagoon regions);

Medium exposure samples were taken from up to 10 m distance from the low tide water's edge (tidal region) or the water's edge at the time of sampling (non-tidal regions). Low exposure samples were taken at approximately 0.3 m water depth at low tide (in the tidal region) or current water level (non-tidal regions). Due to fluctuations in water level throughout the system over time, the position of the three mudflat exposures would have been different at each sampling time.

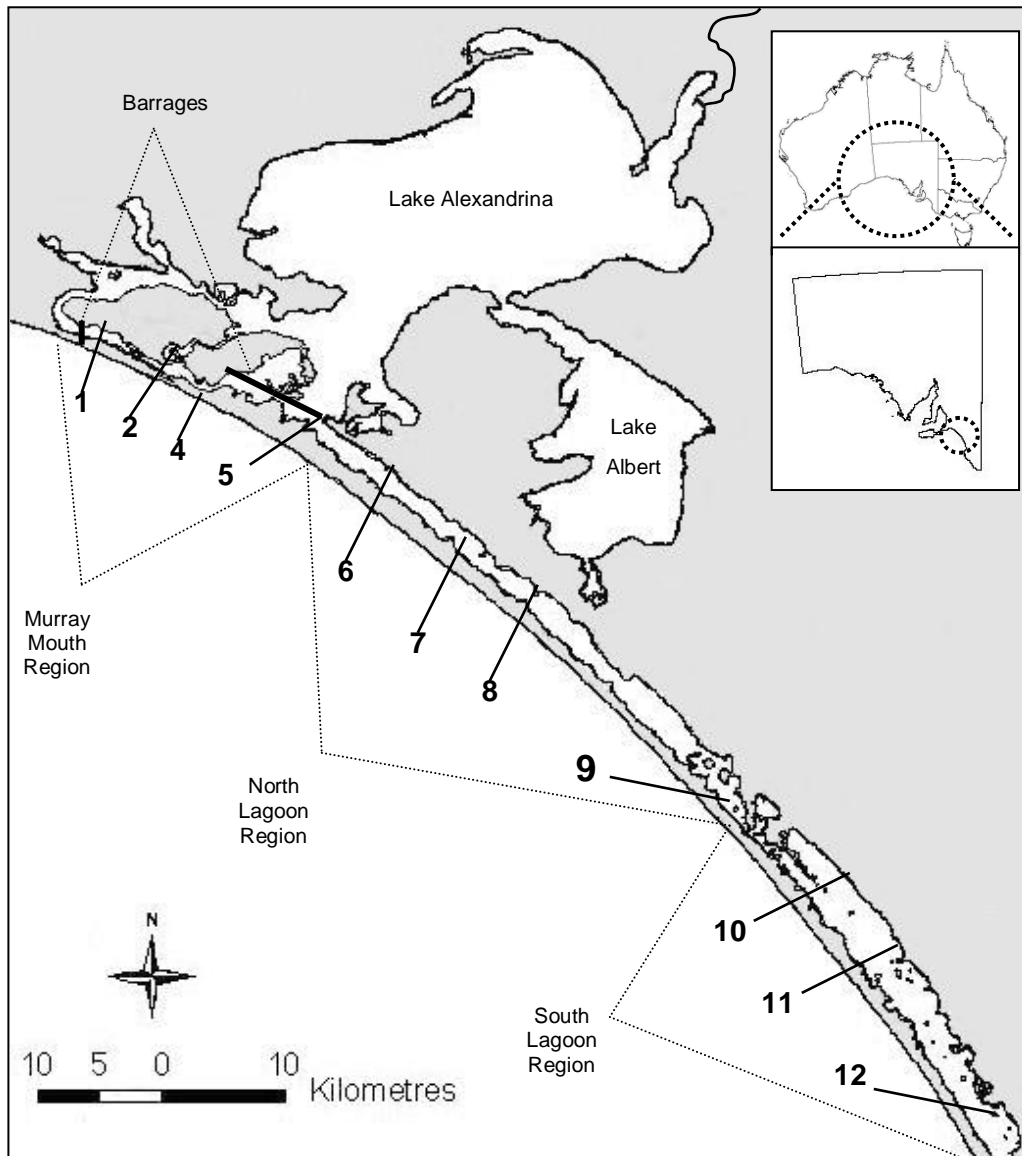


Figure 1. Map of the Coorong, Lower Lakes and Murray Mouth (CLLAMM) region. Numbers indicate sites sampled.

2.2. Environmental Parameters

The environmental parameters listed below were measured from December 2006 to October 2007 on a bi-monthly basis in tandem with both the adult (Dec 2006 – March 2007) and juvenile (Dec 2006 – Oct -07) macrobenthos sampling. Due to equipment malfunction, no environmental parameters were recorded in May 2007.

2.2.1. Water quality

Salinity, temperature, oxygen content, oxygen saturation and conductivity were measured at each site for each sampling time in shallow water (not deeper than 0.3 m) using a hand-held YSI-85 electrode.

2.2.2. Sediment characterisation

Grain Size

At each mudflat exposure, three replicate grain size cores and three replicate organic content cores were taken using a 60 ml syringe (area: 6.61 cm²; depth: 6 cm) and a 3 ml syringe (area: 0.64 cm²; depth: 3 cm) respectively. Both syringes had cut-off openings. All sediment and organic content samples were stored at -18 °C until further analysis. Sediment grain size was determined by laser diffraction using a Malvern Mastersizer 2000 (Malvern Instruments Ltd, UK). Samples were thawed and the fraction > 1 mm manually sieved off to avoid machine blockage. This fraction weight, together with the remaining sediment weight was determined for later data normalisation to allow for this procedure. Median and quartiles were obtained from the Mastersizer output and sediment sorting was calculated ($S_0 = P_{25}/P_{75}$) based on metric scale.

Organic Content

Sediment organic content was calculated as % dry weight: sediment was dried to a constant weight at 80° C before combustion at 450° C for 4-6 hours in a muffle furnace.

2.3. Macrobenthos Sampling Method

2.3.1. Adult macrobenthos surveys

At each site and exposure level, ten core samples were taken in a haphazard manner using a 5 cm diameter PVC (area: 19.63 cm²) corer to a sediment depth of approximately 15 cm, resulting in 30 replicate samples per site. Samples were sieved on site through 500 µm mesh and the remaining content placed into plastic bags for transport. Organisms were sorted live and preserved in 99.9% ethanol until identification to species level where possible. Oligochaeta and Amphipoda were only identified to Class and Order level respectively. Following identification, organism biomass was calculated as grams per ash free dry weight (g AFDW) per square metre following drying at 80 °C for 24 hours (dry weight) before combustion at 450 °C for 4 hours (ash weight). The weight difference between the dry and ash weights is the AFDW.

The length of up to 20 individuals per exposure of four species, *Capitella*, *Simplisetia*, *Nephtys* and *Arthritica*, were measured for each site for each of the three sampling times. Length of complete specimens was measured under a stereoscopic microscope to the nearest 500 µm and means calculated.

2.3.2. Juvenile macrobenthos surveys

The term juvenile is used in this report to encompass all early life-stages from post-larvae over juveniles to small-sized adults.

At each site and exposure level, ten core samples were taken using a 60 ml syringe with a cut-off opening (area: 6.61 cm²) to a sediment depth of 2 cm. Samples were stored in plastic bags

with approximately 20 ml water (taken from the site of the sample) at 8° C for not longer than five days before being sieved through 250 µm mesh. Each sample was then washed in seawater and decanted 5 times to remove any juveniles before being left in approximately 50 ml MgCl solution for five minutes. Each sample was then washed and decanted a further 5 times to remove any remaining juveniles. All juveniles were then identified to species level.

2.4. Translocation of Infauna to Different Salinities and Exposure Levels

Translocation of infauna by sediment translocations have been used to highlight the effect of sediment characteristics on marine macrobenthic juvenile recruitment (Snelgrove *et al.*, 2001), the impact of boat generated waves on macrofauna (Bishop, 2007) and to assess the response of macrofauna to different nutrient levels (Morris and Kehoe, 2002). Our translocation experiments focussed on moving sediments to areas of higher and lower salinity and higher and lower exposure levels on the shore.

We hypothesised that translocations of macroinvertebrates

- i. to higher salinities will result in a decrease in abundance and diversity of taxa;
- ii. to lower salinities will result in an increase in abundance and diversity of taxa.
- iii. to areas of increased mudflat exposures will result in a decrease in abundance and diversity of taxa;
- iv. to areas of decreased mudflat exposure will result in an increase in abundance and diversity of taxa.

2.4.1. Sites

Salinities increase southwards with distance from the Murray Mouth, resulting in hypersaline conditions in the South Lagoon. Sites 4, 7 and 8 are situated in the Murray Mouth and North Lagoon regions of the Coorong (Figure 1), where sediment grain size varies between fine and medium sands (Dittmann *et al.*, 2006a). These sites were chosen for sediment translocation as they encompass the salinity gradient from marine to hypersaline and in turn represent sites where macrobenthic abundance ranges from high to low (Dittmann *et al.*, 2006a). Water levels at site 4 are driven by tides entering through the Murray Mouth, whereas at sites 7 and 8, water levels are driven primarily by wind and weather systems. Each site was divided into two levels of mudflat exposure. At high exposure sediment was subjected to infrequent water coverage as a result of tidal or weather-driven water levels, while at low exposure sediment was generally inundated and only exposed in extreme tidal or weather conditions.

2.4.2. Sediment translocations

Sediment, and its macrofauna, was translocated within and between sites and mudflat exposures. Sediment cores (10 cm diameter: 78.54 cm²) were taken to a sediment depth of approximately 0.15 m and placed, still encased by the PVC corer, into plastic zip-lock bags before being translocated to their designated destination.

Within each mudflat exposure there were three rows of 27 plots, with each row containing a sediment translocation from each mudflat exposure from each site for each of the three

sampling times (1, 2 and 6 weeks post-translocation). Each plot contained one sediment core. Plots were marked by flags and the core placed 30 cm from the post, perpendicular to the shoreline, using a plastic template, thus ensuring the exact recovery of the translocation at the respective sampling time. Each plot and each row were separated by 3 m distance. Individual translocations were assigned randomly to plots throughout each of the three rows per elevation, allowing for comparisons between mudflat exposures within and between sites for each sampling time.

At each site there were two types of translocations: Selfs and Outs (Figure 2). Self translocations consisted of sediment translocated within and between mudflat exposures of the same site. There were three self cores from and to each mudflat elevation for assessment at three different sampling times (1, 2 and 6 weeks post translocation), totalling 36 self translocations per site for the course of the experiment (Table 2). Out translocations consisted of sediment taken from both mudflat elevations of one and translocated to the other sites. Three out cores per elevation per sampling time were translocated to each site, totalling 108 out cores per site (Table 2). For each site, a total of 144 cores were translocated for the course of the experiment.

In order to assess the possibility of trampling artefacts at each mudflat elevations, three undisturbed cores (controls) that did not undergo any translocation were taken at each sampling time (1, 2 and 6 weeks).

Macrofaunal abundances were also assessed at each site on four different occasions: Week 0 (the week that other sediments were translocated), and Weeks 1, 2 and 6 post-translocation. These 'Background' samples enabled comparisons of the local fauna at each site against sediment translocated to that site (Out translocations).

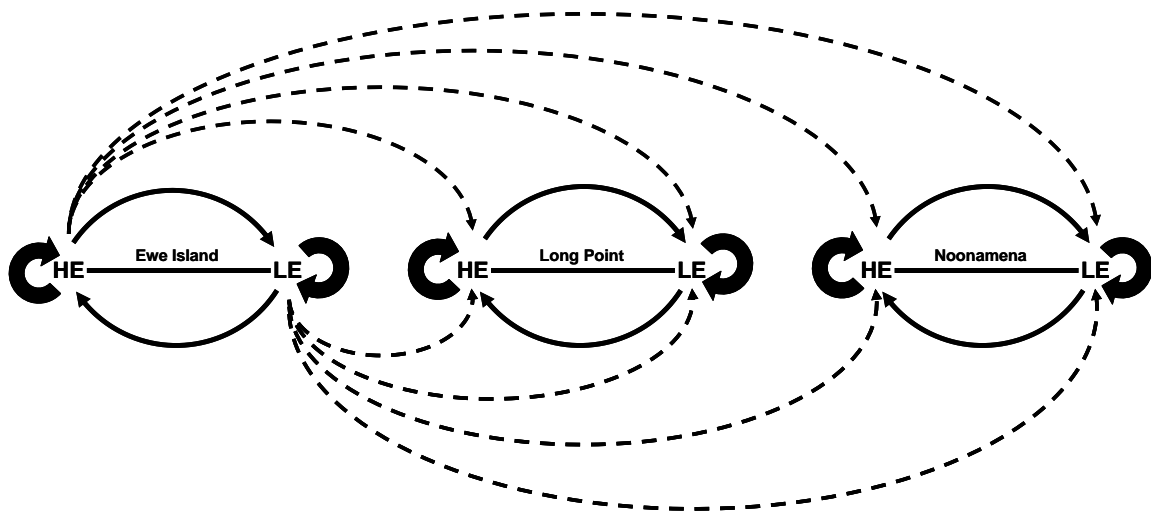


Figure 2. Sediment translocations from and within Ewe Island. Solid arrows represent Self translocations from within and to high and low mudflat exposures. Dashed arrows represent Out translocations from and to high and low mudflat elevations. The same sequence of translocations were performed for Long Point and Noonamena.

Table 2. Number of both self and out translocation cores per site made over the course of the three sampling times (1, 2 and 6 weeks post translocation).

	Exposure		Total Per Site
	High	Low	
No. Self Cores	3	3	
No. Elevations	2	2	
Sampling Times	3	3	
Total Self Translocations	18	18	36
No. Out Cores	3	3	
No. Elevations	2	2	
Sampling Times	3	3	
No. Sites Translocated To	3	3	
Total Out Translocations	54	54	108
Overall Total			144

2.4.3. Assessing adult macrobenthic invertebrate abundance

At each mudflat elevation, at a minimum distance of 10 m from any translocated core, nine background cores (10 cm diameter: 78.54 cm²) were taken to a depth of approximately 0.15 m.

All sediment for each of the four time periods (Weeks 0, 1, 2 and 6) was recovered from its translocation position and sieved on site through 500 µm mesh. The remaining benthos was stored in plastic zip lock bags in approximately 250 ml seawater (taken from the same site from which the cores were recovered) for translocation to the laboratory for sorting live. Invertebrates were sorted into family and stored in 100 % ethanol for storage until identification to species level. Biomass (% Ash-Free Dry Weight, AFDW) was determined as described in Section 2.3.

2.4.4. Assessing juvenile abundance

Before the translocated sediment was recovered, one juvenile core (60 ml open ended syringe, area: 6.61 cm²) was taken from the centre of the translocated sediment to a depth of 0.02 m. Background juvenile samples were taken from the same approximate area as for the adult background samples. These juvenile cores were then placed individually into plastic zip-lock bags in approximately 20 ml sea water (taken from the same site from which the translocation was recovered) before storage at 8 °C and assessed and identified as described in Section 2.4. Any adult fauna recovered in this process was identified to species level and the counts included in the adult counts.

2.4.5. Environmental parameters

Water quality

Water quality (salinity, temperature, oxygen content and saturation and conductivity) were measured as described in Section 2.2.1.

Grain Size

For each sample time at each site, three replicate sediment cores were taken at each mudflat exposure to a depth of 0.06 m as described in Section 2.1, in approximately the same area from where the background macrobenthic core samples were taken. Samples were stored at – 20 °C until assessed as described in Section 2.2.2.

Organic Content

Organic content samples were taken as for the grain size samples, using an open-ended 3 ml syringe (area: 0.64 cm²) to a depth of 0.03 m. Sediment organic content was assessed as described in Section 2.2.2.

2.5. Statistical Analyses

Where some data failed to meet the assumptions of ANOVA, non-parametric Kruskal-Wallis H-Tests were used to test for significant differences between sites, regions (Murray Mouth, North and South Lagoons) and sampling times. To test for similarities in juvenile benthic assemblages between sites, regions and times, multivariate analyses (Cluster analysis, nMDS, ANOSIM and BIOENV) were applied following square root transformation and Bray-Curtis similarity calculations (of abundance data) and log (x+1) transformation, normalisation and Euclidean distance calculated for environmental variable data, using Primer v6 (PRIMER-E Ltd, UK). All other statistical analyses were performed in SPSS v14 (SPSS Inc.).

3. Results

3.1. Environmental Parameters

Salinity in the Murray Mouth region (Sites 1-5) remained approximately marine throughout the year (Figure 3A), but increased rapidly with distance southwards into the North and South Lagoons regions (Sites 6-9 and 10-12 respectively), with peak salinities in the South Lagoon frequently over 120 g/L. Salinities in the South Lagoon in December 2006 were between 22 – 35 g/L lower than in the months of January and March 2007. Both oxygen concentration and oxygen saturation remained approximately constant throughout the Murray Mouth and North Lagoon regions in July and October 2007 (5.89 – 7.95 mg/L and 76 – 99 % for O₂ content and saturation respectively) in comparison to the highly variable readings for both parameters for December 2006, January and March 2007 (4.4 – 10.73 mg/L and 40.3 – 140.3 % for O₂ content and saturation respectively) (Figures 3B and C). Oxygen concentration measurements failed at sites 9-12 in March, July and October 2007 and also at sites 8-12 in December 2006 and January 2007. Water temperature differed significantly over the sampling times (One-way ANOVA, $F = 67.9$, $P < 0.001$; Figure 3D), with a maximum of 28.7 °C at Site 11 in January 2007 and a minimum of 7.7 °C at Site 1 in July 2007. Temperature showed a slight positive linear correlation from the Murray Mouth region southwards to the Southern Lagoon.

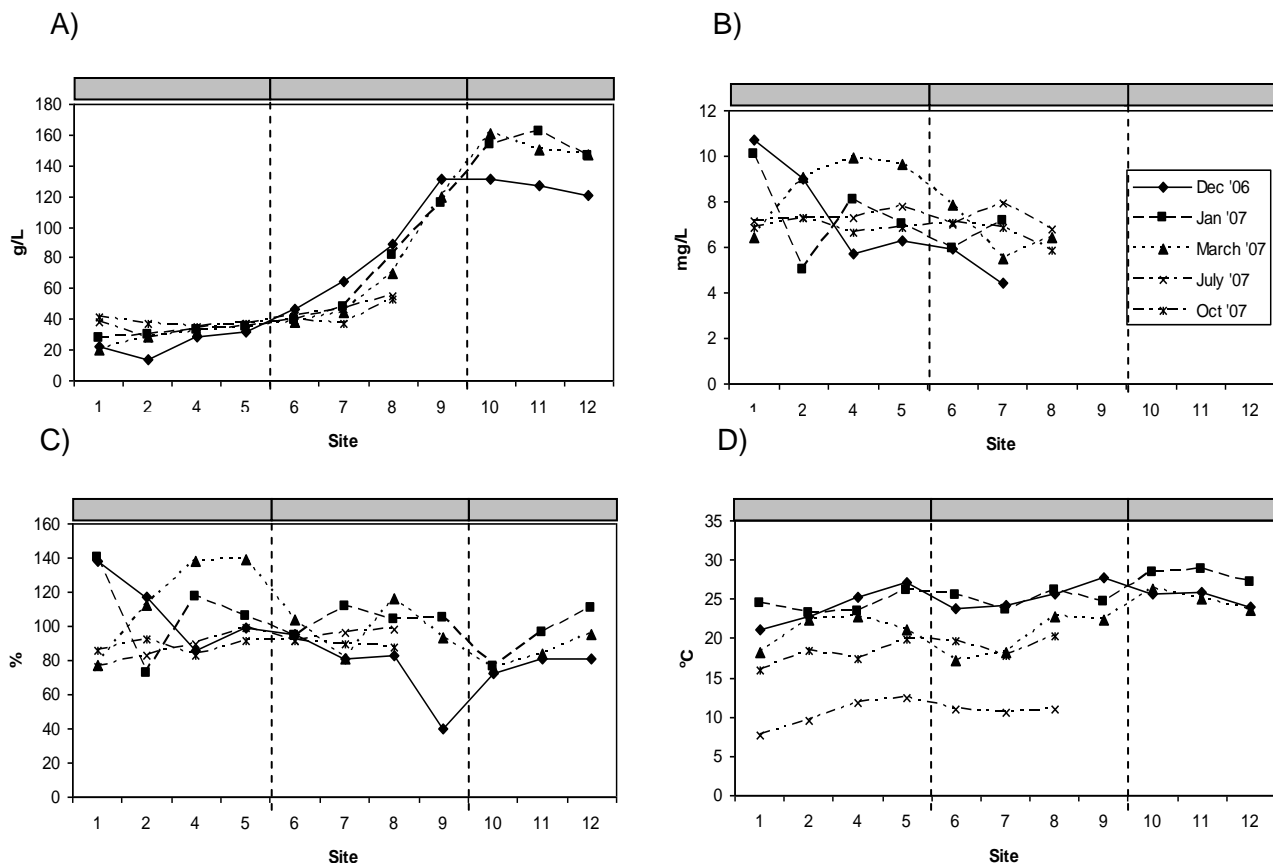


Figure 3. Water environmental parameters measured from December 2006 – October 2007 across 11 sites of the Murray Mouth, North Lagoon and South Lagoon regions: A) Salinity; B) Oxygen Concentration; C) Oxygen Saturation; D) Temperature.

Median grain sizes were consistently classified as fine to medium sand for all sites over each sampling time, despite some evidence of variation between sites, regions and sampling times (Figure 4). Kruskal-Wallis H-Tests on median grain sizes showed significant differences between sites and regions for each sample time ($11.35 \leq \chi^2 \leq 124.83$, $P < 0.01$ or better), except in May 2007 when there was no significant difference between regions. Site 5 consistently achieved the highest rank in significant H-tests and was consistently separated from other sites in MDS and Cluster analyses (Figure 4). ANOSIM analysis on data for all the sampling times combined showed significant differences in grain size composition between sites and also regions (Global R = 0.602 and 0.25: $P = 0.001$ respectively – Figure 4), but no significant difference between sampling time. Significant differences in grain size composition between regions were found for December 2006, January 2007 and March 2007 (Global R = 0.213, 0.41 and 0.37: $P = 0.08$, 0.01 and 0.016 respectively) but not for May, July and October 2007 when sampling took place in the Murray Mouth and North Lagoon regions only.

Sediment organic content ranged from 0.36 – 11.8 % ash free dry weight (AFDW) (Figure 5). Over all sampling times, there was a significant difference in organic content between the three regions: Murray Mouth, North Lagoon and South Lagoon (Kruskal-Wallis H-Test, $P < 0.001$: Highest rank given to South Lagoon region). Significant differences in organic content were observed at each site over time, except for Site 12 (One-Way ANOVA and Kruskal Wallis H-Tests). A large increase in organic content was observed from sites 8 through to 12 in January 2007, before levels dropped in March although organic content was still high at site 10 in March 2007.

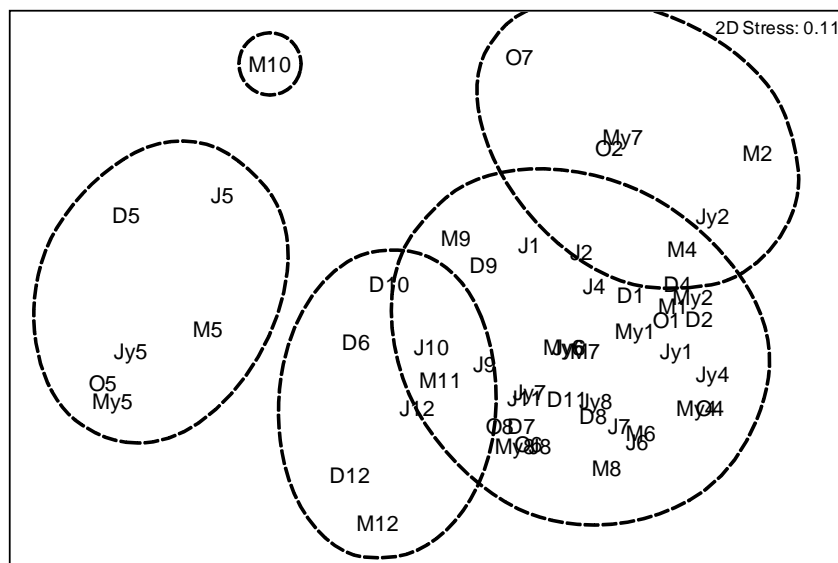


Figure 4. nMDS analyses of sediment grain size (based on fractal class sizes) over all sampling times (D = Dec '06; J = Jan 2007; M = Mar 2007; My = May 2007; Jy = July 2007; O = Oct 2007). Dotted lines represent cluster analysis similarities at 3.5 Euclidian distance.

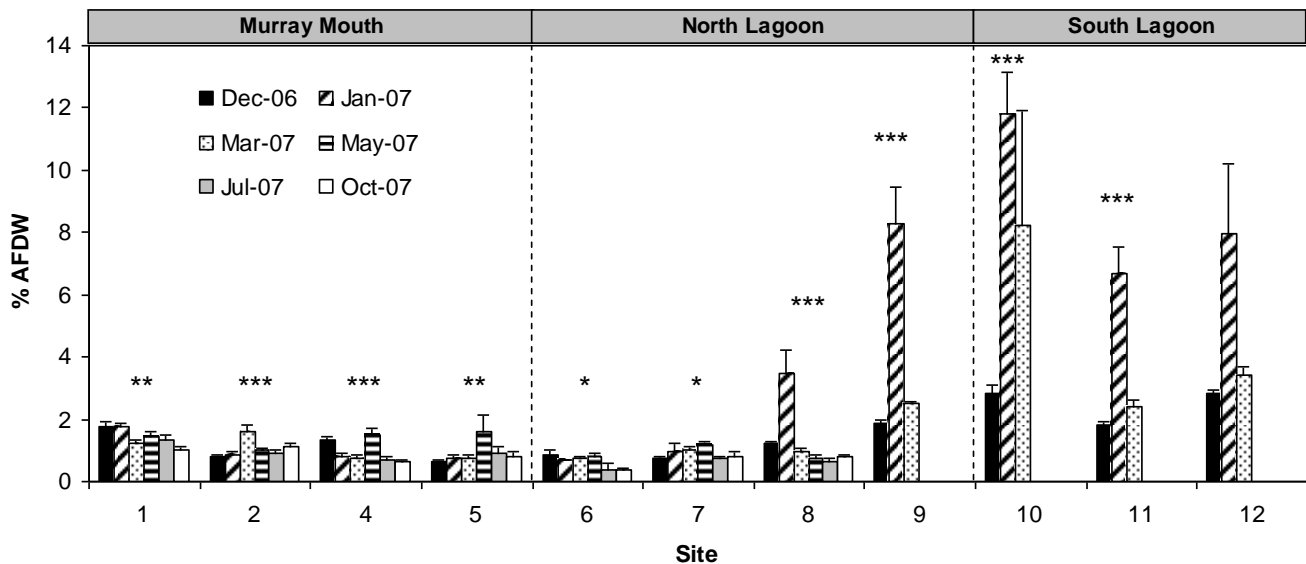


Figure 5. Mean sediment organic content (\pm SE) at each site over time. Sites 9 through to 12 were sampled only in December 2006, January and March 2007. Asterisks indicate significant differences at each site over time (One-Way ANOVA or Kruskal-Wallis Tests: * = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$).

3.2. Adult Macrobenthos

3.2.1. Diversity, abundance and distribution

A maximum of 18 taxa, comprising four Phyla were recorded across all sites (Table 3). The most diverse sites were those occurring in the Murray Mouth region, with diversity decreasing into the North and South Lagoons. In the South Lagoon (sites 10-12) in December 2006, only Insecta were present (Figure 6 A; Table 3). No taxa were present in the South Lagoon in January and March 2007 (Figures 6 B and C; Table 3). The taxon that contained the greatest number of species present in the system was the Polychaeta. Bivalvia were restricted to sites 1-7, Gastropoda were restricted to sites 1-5, whilst the Insecta were the only taxon to be found at each site over the course of the three sampling times (Figure 6; Table 3).

Over the three sampling times, there were significant differences in benthic abundances found in the system (all sites combined) for *Capitella*, *Simplisetia*, Polychaeta, Crustacea, Insecta and overall benthos (all species combined) (Kruskal-Wallis tests: $15.592 \leq \chi^2 \leq 75.205$, $P < 0.001$). At sites 2, 4, 6, 7 and 9 - 12 there was significantly higher overall abundance (all species combined) in December 2006 than in both January and March 2007 (One-Way ANOVA and Kruskal-Wallis Tests: $P = 0.005$ or better; Figure 7). Overall abundance was highest in the Murray Mouth and North Lagoon regions (sites 1-5 and site 6 and 7 respectively), with sites 4 and 5 consistently having the highest abundance (Figure 7).

Abundance throughout the system was dominated by Annelida (Figure 8), particularly by Polychaeta, although Crustacea (dominated by Amphipoda), and Mollusca, (particularly *Arthritica*) were also present in high abundance. Annelida were restricted to sites 1-7, with peak abundance (over 35,000 individuals m^{-2}) occurring at Site 5 in January and March 2007, and at Site 6 in December 2006. Both Annelida and Mollusca generally increased in abundance towards Site 5 before decreasing rapidly in the North Lagoon. Crustacea reached peak

abundance of 36,000 individuals m⁻² in December 2006, but were not present in the North and South Lagoons past Site 6. Insect were the only taxa present in the South Lagoon, and abundance varied greatly with sampling time and site.

Table 3. Distribution of macroinvertebrate species at 11 sites (site 3 not sampled) over three time periods: Dec 2006, Jan 2007 and March 2007.

		December '06										
Species		1	2	4	5	6	7	8	9	10	11	12
Annelida	<i>Capitella</i> sp.											
	<i>Australonereis ehlersi</i>											
	<i>Simplisetia aequisetis</i>											
	<i>Nephtys australiensis</i>											
	<i>Boccardiella limnicola</i>											
	<i>Phyllodoce novaehollandiae</i>											
Oligochaeta indet.												
Crustacea	<i>Helograpsus</i>											
	<i>Macrophthalmus latifrons</i>											
Amphipoda												
Mollusca	<i>Salinator fragilis</i>											
	<i>Hydrobia</i> sp.											
Insecta	<i>Arthritica helmsi</i>											
	Chironomid sp.											
	Dolichopodidae											
	Ceratopogonidae											
	Tabanid sp.											
	Insecta indet.											

		January '07										
Species		1	2	4	5	6	7	8	9	10	11	12
Annelida	<i>Capitella</i> sp.											
	<i>Australonereis ehlersi</i>											
	<i>Simplisetia aequisetis</i>											
	<i>Nephtys australiensis</i>											
	<i>Boccardiella limnicola</i>											
	<i>Phyllodoce novaehollandiae</i>											
Oligochaeta indet.												
Crustacea	<i>Helograpsus</i>											
	<i>Macrophthalmus latifrons</i>											
Amphipoda												
Mollusca	<i>Salinator fragilis</i>											
	<i>Hydrobia</i> sp.											
Insecta	<i>Arthritica helmsi</i>											
	Chironomid sp.											
	Dolichopodid sp.											
	Ceratopogonid sp.											
	Tabanid sp.											
	Insecta indet.											

		Mar-07										
Species		1	2	4	5	6	7	8	9	10	11	12
Annelida	<i>Capitella</i> sp.											
	<i>Australonereis ehlersi</i>											
	<i>Simplisetia aequisetis</i>											
	<i>Nephtys australiensis</i>											
	<i>Boccardiella limnicola</i>											
	<i>Phyllodoce novaehollandiae</i>											
Oligochaeta indet.												
Crustacea	<i>Helograpsus</i>											
	<i>Macrophthalmus latifrons</i>											
Amphipoda												
Mollusca	<i>Salinator fragilis</i>											
	<i>Hydrobia</i> sp.											
Insecta	<i>Arthritica helmsi</i>											
	Chironomid sp.											
	Dolichopodid sp.											
	Ceratopogonid sp.											
	Tabanid sp.											
	Insecta indet.											

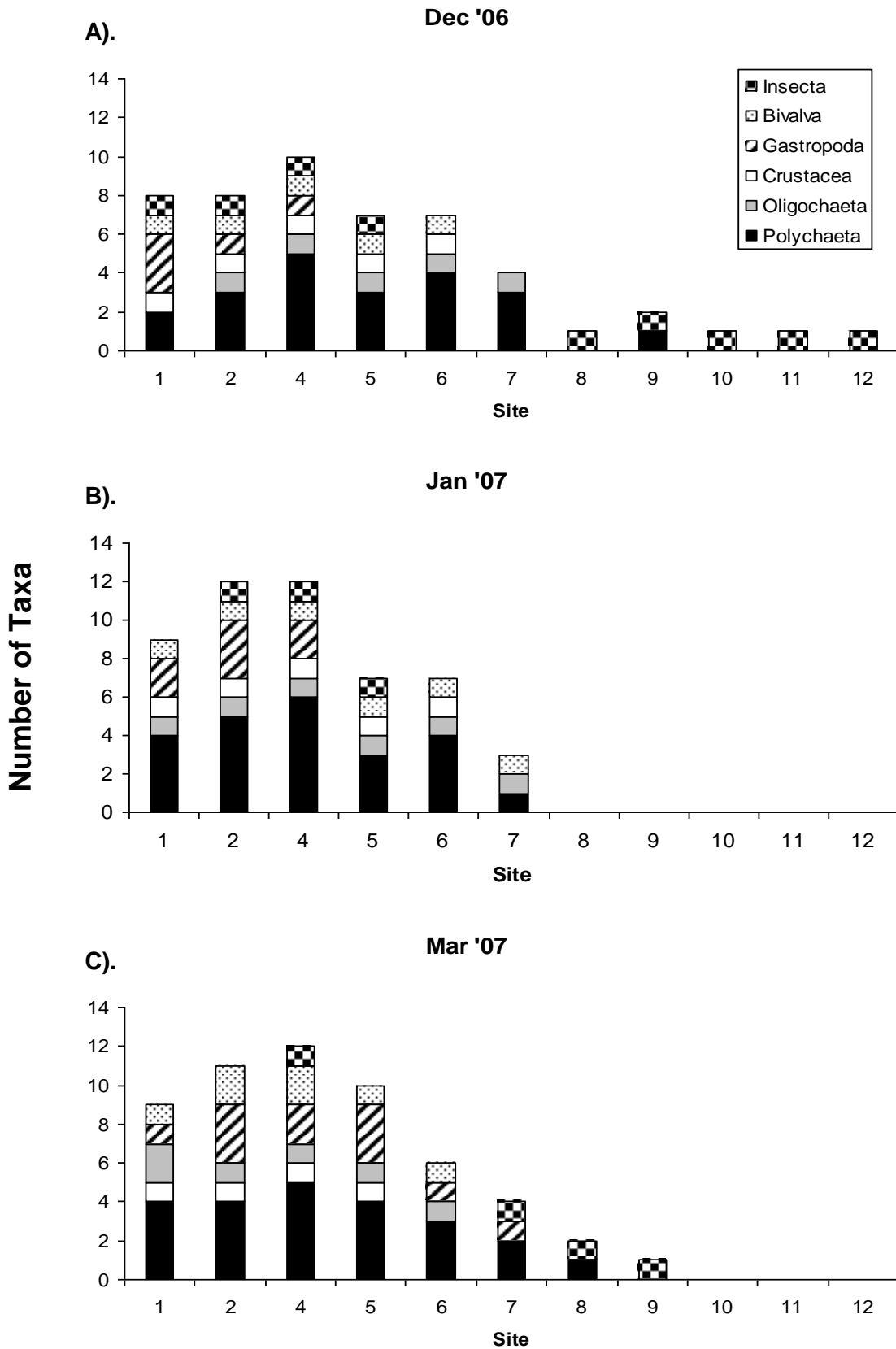


Figure 6. Total number of taxa found at each of 11 sites over three sampling times: A). Dec 2006; B). Jan 2007; C). Mar 2007.

The distribution and abundance patterns of four key species, the polychaetes *Capitella* spp., *Simplisetia aequisetis*, and *Nephtys australiensis*; and the micro-bivalve mollusc, *Arthritica helmsi* are shown in Figure 9 and described below.

The polychaete worms of the system were dominated in abundance by *Capitella* spp. (Figure 9A). The abundance of both *Capitella* spp. and *S. aequisetis* increases from sites 1-5 before decreasing rapidly in the North Lagoon. No polychaetes were found south of Site 8 (Figure 9 A, B and C). Little temporal patterns were observed for either *Capitella* spp. or *S. aequisetis*. *Nephtys australiensis* decreased in abundance from Sites 1-6 as well as over time (Figure 9 C). *Arthritica helmsi* showed similar distribution to the polychaetes *Capitella* spp. and *S. aequisetis*.

Capitella spp. had greater abundance in the Murray Mouth region (sites 1 to 5) than the North Lagoon region (sites 6 to 9) in Jan 2007 and March 2007, but not in Dec 2006 (Figure 10). *Simplisetia*, *Nephtys*, Amphipoda and *Arthritica* had significantly greater abundance in the Murray Mouth than the North Lagoon for each of the three sampling times. Insecta did not show any significant differences between regions at any sampling time. For all species tested, differences between mudflat exposures in both the Murray Mouth and North Lagoon varied with time (Figure 10).

In addition to the regional mudflat exposure-specific variation, the abundance of benthos also varied temporally (Table 4; Figure 10). Generally, as mudflat exposure increased as water levels dropped through time, abundance in the more exposed sediment decreased significantly for all species tested, particularly in the North Lagoon.

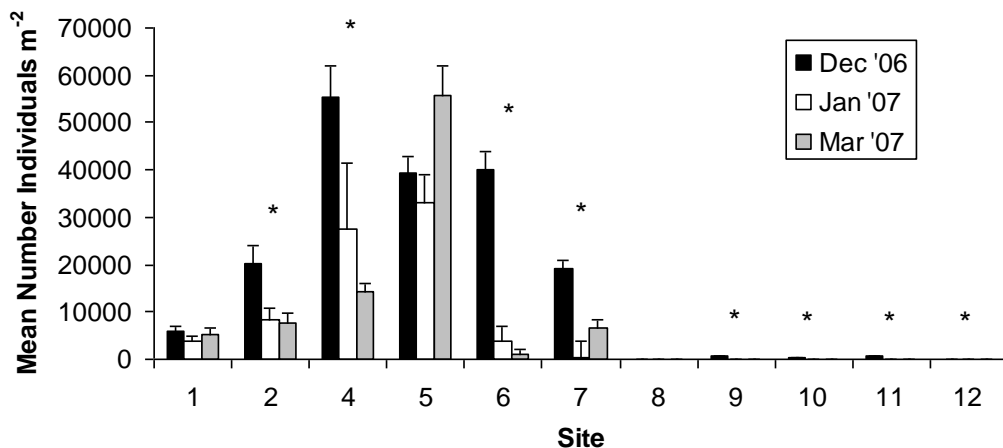


Figure 7. Overall benthos abundance (all species combined) (individuals m⁻², mean ± S.E.) at each site over three sampling times (Dec 2006, Jan 2007 and March 2007). Asterisks indicate significant differences between sampling times following One-Way ANOVA or Kruskal-Wallis Tests at P < 0.05 or better. Insect larvae were present at Site 12 in Dec 2006 but are not shown here due to the y-axis scale.

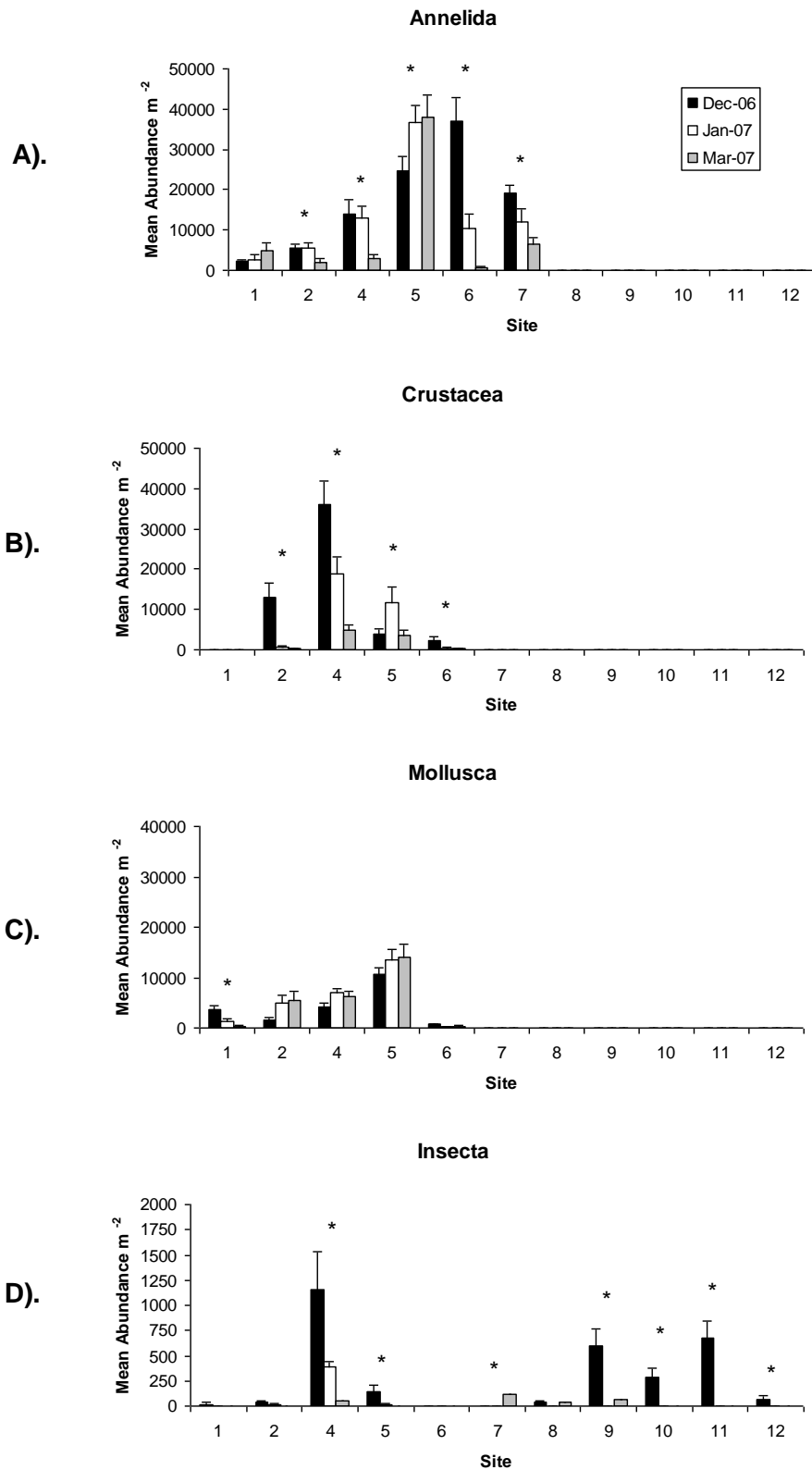


Figure 8. Mean abundance (individuals m^{-2}), \pm standard error, of the four taxa (A: Annelida; B: Crustacea; C: Mollusca; D: Insecta) present in the system over the three sampling times (Dec 2006, Jan 2007 and March 2007). Asterisks indicate significant differences in abundance between sampling times at that respective site following One-Way ANOVA and Kruskal-Wallis Tests ($P < 0.05$ or better).

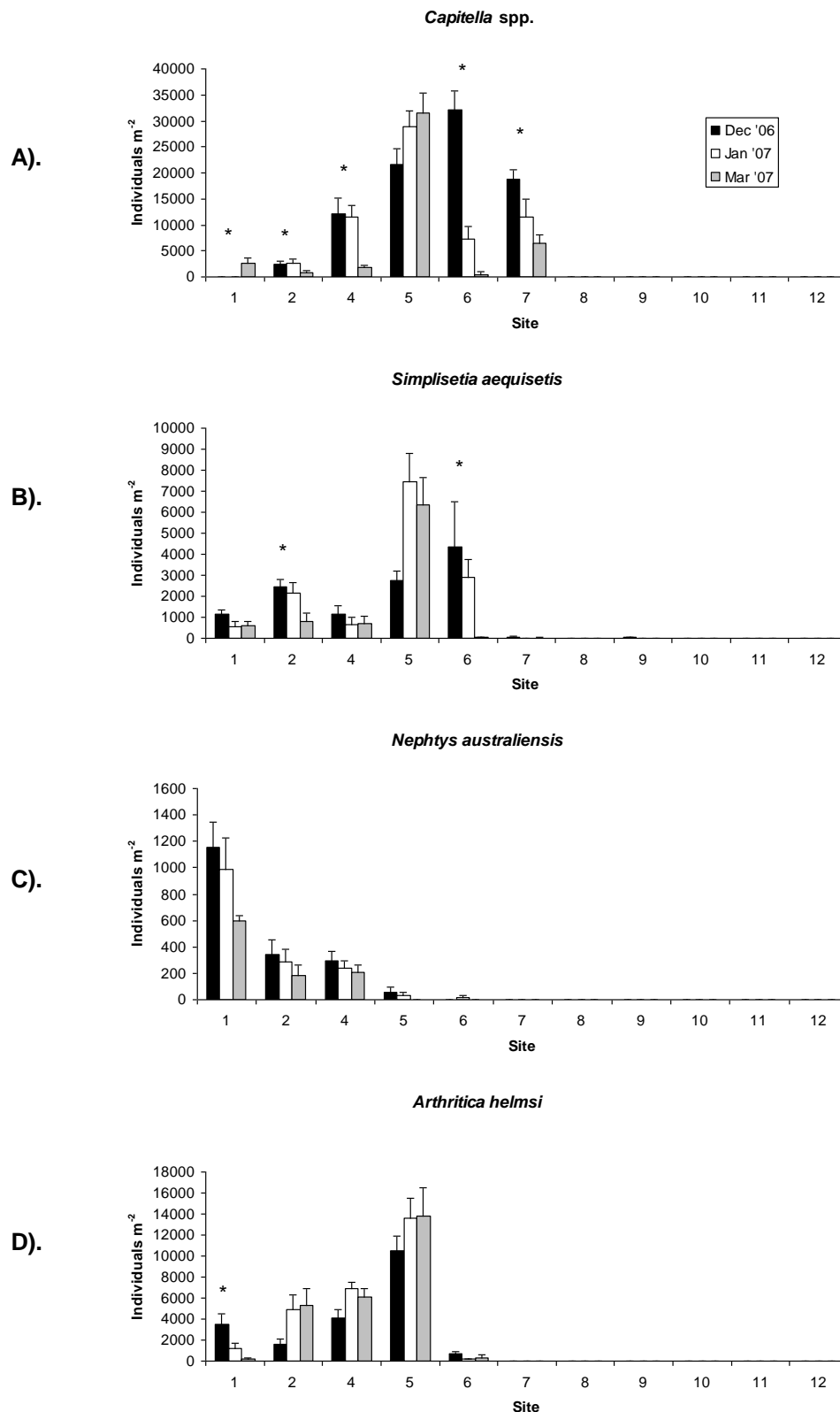


Figure 9. Abundance (individuals m^{-2} , mean \pm standard error) of four key species (A = *Capitella*; B = *Simplisetia*; C = *Nephtys*; D = *Arthritica*) present in the system over the three sampling times (Dec 2006, Jan 2007 and March 2007). Asterisks indicate significant differences in abundance between sampling times at that respective site following One-Way ANOVA or Kruskal-Wallis Tests ($P < 0.05$ or better).

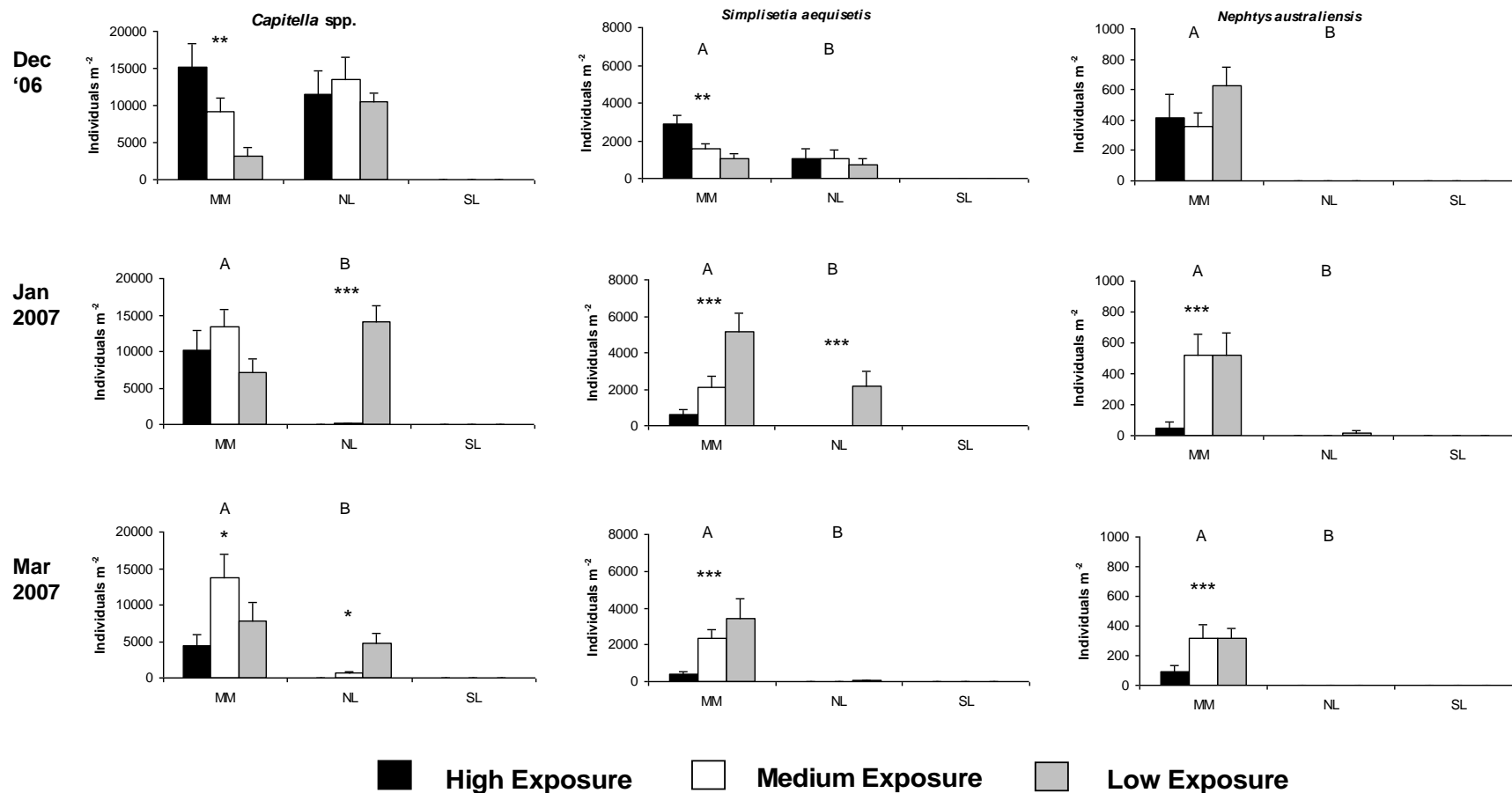
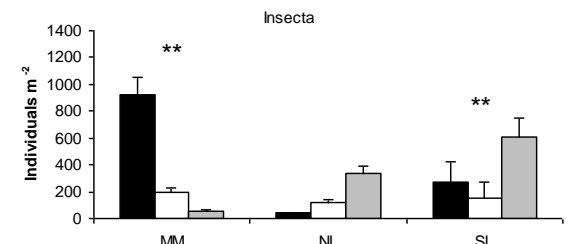
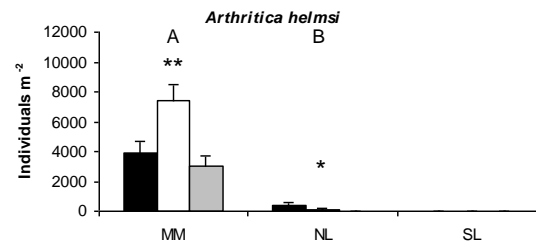
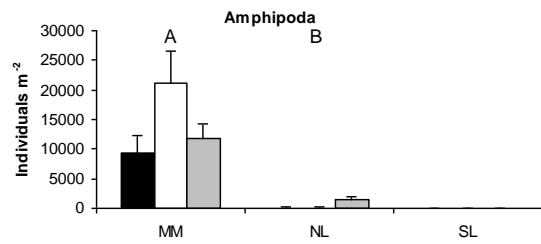
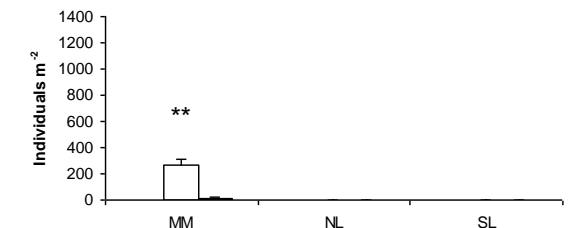
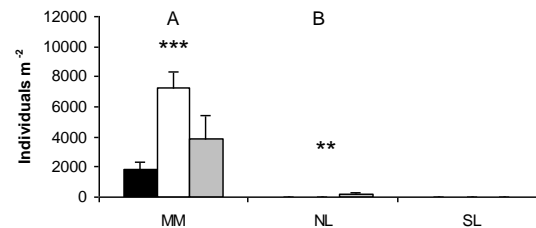
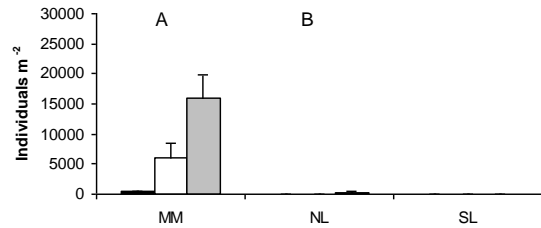


Figure 10. Benthic invertebrate abundance (individuals m⁻², mean ± S.E.) of six taxa within areas of mudflat exposure (High, Medium and Low) through three regions (MM = Murrumbidgee Mouth; NL = North Lagoon; SL = South Lagoon). Note the different y-axis scales. Different letters indicate significant differences (Kruskal-Wallis Tests: $P < 0.05$ or better) between the total number of individuals (all mudflat exposures combined) between regions (due to absence of benthos, South Lagoon data was excluded from all analyses except for Insecta). Asterisks indicate significant differences between mudflat exposures of the same region at that sampling time (Kruskal-Wallis Tests: * = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$). Analyses comparing individual mudflat exposures through time are shown in Table 4. Figure 10 continued over the page.

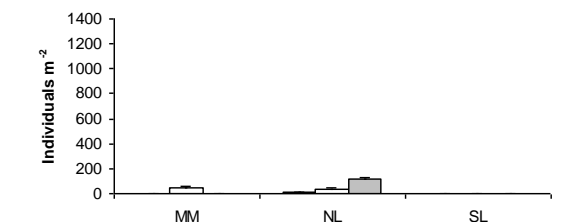
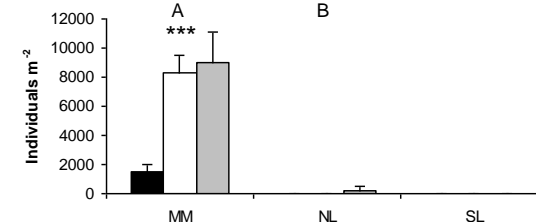
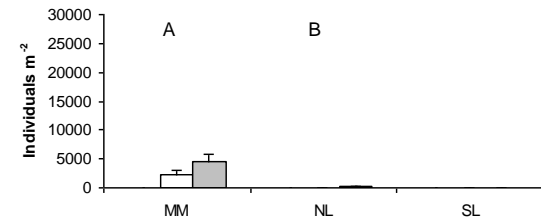
Dec '06



Jan 2007



Mar 2007



Region

■ High Exposure □ Medium Exposure ▒ Low Exposure

Figure 10 continued

Table 4. Results of Kruskal=Wallis Tests showing significant differences between all mudflat exposures (High, Medium or Low) within regions (MM = Murray Mouth; NL = North Lagoon; South Lagoon region not tested due to lack of benthos) over time (See Figure 10). ns = no significant difference; * = P < 0.05; ** = P < 0.01; *** = P < 0.001.

Tidal Exposure	<i>Capitella</i> spp.		<i>Simplisetia aequisetis</i>		<i>Nephtys australiensis</i>		Amphipoda		<i>Arthritica helmsi</i>		Insecta	
	MM	NL	MM	NL	MM	NL	MM	NL	MM	NL	MM	NL
High	*	***	***	**	*	ns	***	*	**	***	***	ns
Medium	ns	***	ns	***	ns	ns	*	*	ns	**	ns	ns
Low	ns	***	*	*	ns	ns	**	ns	*	**	ns	**

3.2.2. Macrobenthic communities

Multivariate analysis of abundances of all species per site for each sampling time separated sites in the Murray Mouth and northern North Lagoon regions (sites 1-5 and sites 6 and 7 respectively) from sites in the southern North Lagoon and South Lagoon (site 9 and sites 10-12 respectively) (Figure 11). Site 8 did not cluster with any other sites in Dec 2006 and Jan 2007, yet clustered with sites from the South Lagoon in March 2007. ANOSIM analysis revealed significant differences between regions (Global R = 0.474, P = 0.001), with pairwise tests showing differences between the Murray Mouth and both North and South Lagoons (P = 0.001 and P = 0.029 respectively), but no significant difference between the North Lagoon and South Lagoon (P = 0.057). There was no significant difference between sampling times.

The results of SIMPER (Similarity Percentages) analysis on the Murray Mouth and North Lagoon abundance data (South Lagoon not tested due to the presence of Insecta only) across all sampling times is shown in Table 5. In the Murray Mouth region, the average Bray-Curtis similarity between all pairs of all sites was 53.28, and made up mainly of contributions from four taxa: *Arthritica helmsi*, *Capitella* spp., *Simplisetia aequisetis* and Amphipoda, with a cumulative percentage of approximately 80% of the total similarity of 53.28. These species can therefore be described as typical of the Murray Mouth region. In the North Lagoon, the average Bray-Curtis similarity was much lower at 18.88. Only two taxa, *Capitella* spp. and Chironomid larvae contributed a cumulative 80% of this low similarity, and these species can be described as typical of the North Lagoon.

Comparing the Murray Mouth with the North Lagoon, (Table 5), the average Bray-Curtis dissimilarities between all pairs of sites (one in the Murray Mouth, the other in the North Lagoon) was 76.08, which was made up mainly of contributions of four taxa: *A. helmsi*, *Capitella* spp., Amphipoda and *S. aequisetis*. These taxa contributed to a cumulative percentage of approximately 74% of the total dissimilarity of 76.08. As shown for the abundance data at each site above (see Table 5; Figures 7-10), average abundance was greater in the Murray Mouth than the North Lagoon.

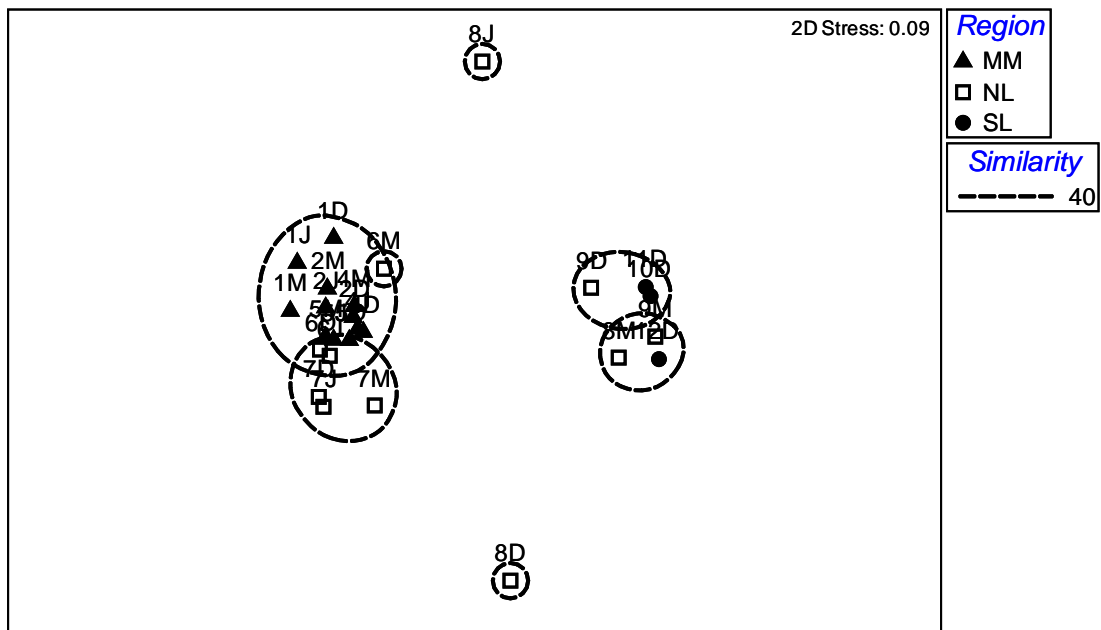


Figure 11. nMDS analysis of overall benthos showing Sites, sample times (D = Dec 2006; J = Jan 2007; M = Mar 2007) and regions (MM = Murray Mouth; NL = North Lagoon; SL = South Lagoon).

Of all the environmental variables tested (salinity, oxygen saturation, water temperature, sediment grain size and sediment organic content), salinity combined with sediment organic contents gave the best correlation (BIOENV: correlation coefficient = 0.691). Salinity alone resulted in the second highest correlation of 0.652. Global Best analysis concluded that there is agreement in multivariate pattern between macrobenthic invertebrate abundance and the environmental variables tested above (Global Best: $\rho = 0.691$, $P = 0.01$).

Table 5. Results of SIMPER analyses comparing the contribution of species present in the Murray Mouth and North Lagoon regions to similarity within, and dissimilarity between, regions.

Murray Mouth					
Average Bray-Curtis Similarity = 53.28					
Species	Average Abundance		Average Similarity	% Contribution	Cumulative % Contribution
<i>Arthritica helmsi</i>	70.87		15.21	28.56	28.56
<i>Capitella</i> spp.	78.21		10.21	19.16	47.72
<i>Simplisetia aequisetis</i>	42.53		9.73	18.26	65.98
Amphipoda	66.03		7.41	13.91	79.88
<i>Nephtys australiensis</i>	16.57		3.62	6.79	86.68

North Lagoon					
Average Bray-Curtis Similarity = 18.88					
Species	Average Abundance		Average Similarity	% Contribution	Cumulative % Contribution
<i>Capitella</i> spp.	56.03		9.86	52.20	52.20
Chironomid larvae	3.77		5.25	27.80	80.01

Comparing Murray Mouth with North Lagoon					
Average Bray-Curtis dissimilarity = 76.08					
Species	Average Abundance		Average Dissimilarity	% Contribution	Cumulative % Contribution
	MM	NL			
<i>Arthritica helmsi</i>	70.87	5.54	16.90	22.22	22.22
<i>Capitella</i> spp.	78.21	56.03	16.80	22.08	44.30
Amphipoda	66.03	8.26	13.12	17.24	61.54
<i>Simplisetia aequisetis</i>	42.53	12.96	9.72	12.78	74.32
<i>Nephtys australiensis</i>	16.57	0.37	5.40	7.10	81.42

3.2.3. Biomass

Total biomass varied between sampling times at each site and was higher in March 2007 than Dec 2006, but this difference was not significant at all sites (Figure 12). Significant differences in biomass between sampling times were found at sites 1, 5, 6, 7 and 9-12. Biomass, following abundance, dropped rapidly after site 7. Table 6 shows the results of One-Way ANOVA and Kruskal-Wallis tests, comparing the biomass of Annelida, Crustacea, Mollusca and Insecta at each site over the three sampling periods. For Annelida and Insecta, there was no significant difference in biomass between sampling times in the Murray Mouth region (Sites 1-5). Significant differences were observed at sites 6 and 7 for Annelida, and site 7 and all South Lagoon sites (9-12) for Insecta. These differences in Insecta biomass followed expected seasonal patterns with greatest insect biomass occurring in late spring (Dec 2006: Data not shown). For Crustacea, significant differences in biomass between sampling times were only observed at sites 2 and 5, whilst differences were observed at sites 1, 2 and 6 for Mollusca (Table 6).

Combining all sites per sampling time, there was significant difference in total Insecta biomass across all three sampling times, with Dec 2006 having the greatest median Insecta biomass (Kruskal-Wallis Test, $\chi^2 = 43.316$, $P = 0.000$). No such differences in overall biomass were observed for total Annelida, Crustacea, Mollusca and overall benthos (all taxa combined).

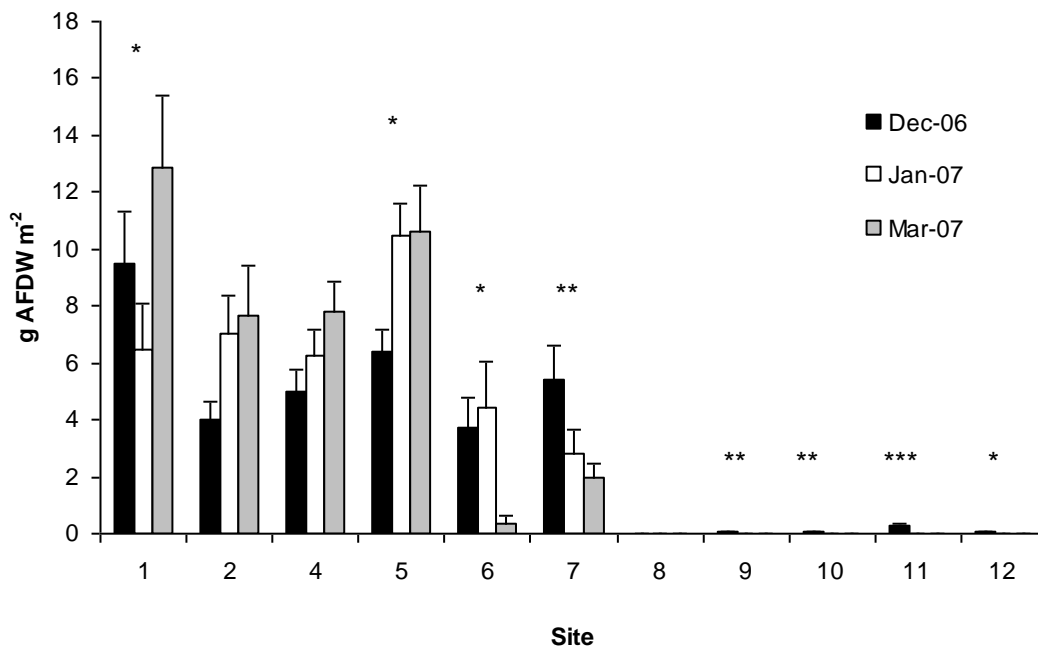


Figure 12. Biomass (g AFDW m⁻², mean ± S.E.) at each of 11 sites (site 3 not sampled) at three sampling times: December 2006, January 2007 and March 2007. Sites 1-5 = Murray Mouth Region; Sites 6-9 = North Lagoon Region; Sites 10-12 = South Lagoon Region. Asterisks indicate significant differences in biomass between sampling times at each site (One-Way ANOVA or Kruskal-Wallis Tests: * = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$).

Table 7 shows the results of One-Way ANOVA and Kruskal-Wallis tests, comparing the biomass between mudflat exposures (high, medium and low) at sites 1-7 (sites 8-12 not tested due to lack of specimens and low biomass) for each sampling time. For Annelida and overall benthos (all taxa combined), as summer passes (December through to March), the number of sites with significant differences between elevations increases. The converse exists for Insecta. Tukey's post hoc tests, when possible, revealed that most significant differences occurred between the high exposure and both the medium and low exposures (Table 7).

Table 6. Results of One-Way ANOVA or Kruskal-Wallis tests comparing the biomass of Annelida, Crustacea, Mollusca and Insecta across the three sampling times of December 2006, January 2007 and March 2007. ns = no significant difference; - = no individuals present * = P < 0.05; ** = P < 0.01; * = P < 0.001.**

Site	Annelida	Crustacea	Mollusca	Insecta
1	ns	ns	*	ns
2	ns	*	**	ns
4	ns	ns	ns	ns
5	ns	*	ns	ns
6	***	ns	*	-
7	**	-	ns	**
8	ns	ns	-	ns
9	-	-	-	**
10	-	-	-	**
11	-	-	-	***
12	-	-	-	*

Table 7. Results of One-Way ANOVA or Kruskal-Wallis tests, comparing the biomass between mudflat exposures (high, medium and low) at sites 1-7 (sites 8-12 not tested due to lack of diversity and low biomass) for each sampling time. ns = no significant difference; - = no individuals present * = P < 0.05; ** = P < 0.01; * = P < 0.001. Footer denotes significant results of Tukey's Post Hoc Tests when possible: HE = High Exposure; ME = Medium Exposure; LE = Low Exposure.**

Sampling Time	Site	Annelida	Crustacea	Mollusca	Insecta	Total Benthos
Dec 2006	1	*** ¹	ns	**	ns	*** ²
	2	ns	***	ns	-	ns
	4	ns	*	ns	**	ns
	5	ns	**	ns	*	ns
	6	***	*	***	-	***
	7	*	-	-	-	* ³
	Jan 2007	1	ns	ns	**	-
2		*** ⁴	ns	***	ns	*** ⁵
4		ns	ns	ns	**	ns
5		*** ⁶	***	***	ns	*** ⁷
6		***	***	-	-	***
7		***	-	-	-	***
Mar 2007		1	*** ⁸	ns	ns	-
	2	*** ⁹	ns	***	ns	*** ¹⁰
	4	* ¹¹	**	ns	ns	** ¹²
	5	***	***	*** ¹³	-	***
	6	ns	**	ns	-	ns
	7	***	-	ns	ns	***

¹ HE x LE, ME x LE

² HE x LE, ME x LE

³ HE x ME

⁴ HE x ME, HE x LE, ME x LE

⁵ HE x ME, HE x LE, ME x LE

⁶ HE x ME, HE x LE

⁷ HE x ME, HE x LE, ME x LE

⁸ HE x ME, ME x LE

⁹ HE x ME, HE x LE, ME x LE

¹⁰ HE x ME, HE x LE, ME x LE

¹¹ HE x LE

¹² HE x ME, HE x LE

¹³ HE x ME, HE x LE

3.2.4. Benthos size

All species tested (*N. australiensis*, *S. aequisetis*, *Capitella* spp. and *A. helmsi*) showed significant differences in length between sites for all sampling times (One-Way ANOVA or Kruskal-Wallis Tests, P < 0.05 or better), except for *A. helmsi* in Jan 2007. *N. australiensis* length increased from sites 1-6, except in March 2007 when there was a significant decrease in length at site 6 compared to Dec 2006, and indeed was absent from this site in Jan 2007

(Figure 13 A). *Simplisetia* showed a general trend of decreasing length from sites 1-5 before increasing in length at site 6 in Jan 2007 and site 7 in Dec 2006 (Figure 13 B). There were significant length differences for *Simplisetia* between sampling times at each of sites 1-5, with lengths being consistently larger in Dec 2006 than in Mar 2007. *Capitella* length decreased from sites 2-7 in Dec 2006, but this pattern was not observed in Jan and Mar 2007 (Figure 13 C). Significant differences in length were observed between sampling times between sites 2-7, with sizes being generally larger in Dec 2006 than in Jan and Mar 2007, except at site 7. *Arthritica* showed a general decrease in length from sites 1-6 (Figure 13 D), yet there was no significant difference in length between sites in Jan 07. Significant length differences were found between sampling times at sites 2 and 4.

Figure 14 shows the differences in benthos length between mudflat exposures at individual sites for each sampling time. For *Capitella* (Figure 14 A), significant differences in length were observed between mudflat exposures at sites 2, 4 and 7 in Dec 2006; sites 2, 5 and 6 in Jan 2007; sites 4 and 5 in March 2007. Significant differences in length between mudflat exposures were found only at site 4 in Jan and March 2007 for *Simplisetia* and site 1 in Jan and Mar 2007 for *Nephtys* (Figures 14 B and C). For *Arthritica*, site 2 was the only site where significant differences in length were observed in all three sampling times (Figure 14 D). At sites where significant differences between mudflat exposures occurred, smaller individuals of all species were generally found to occur at low exposure.

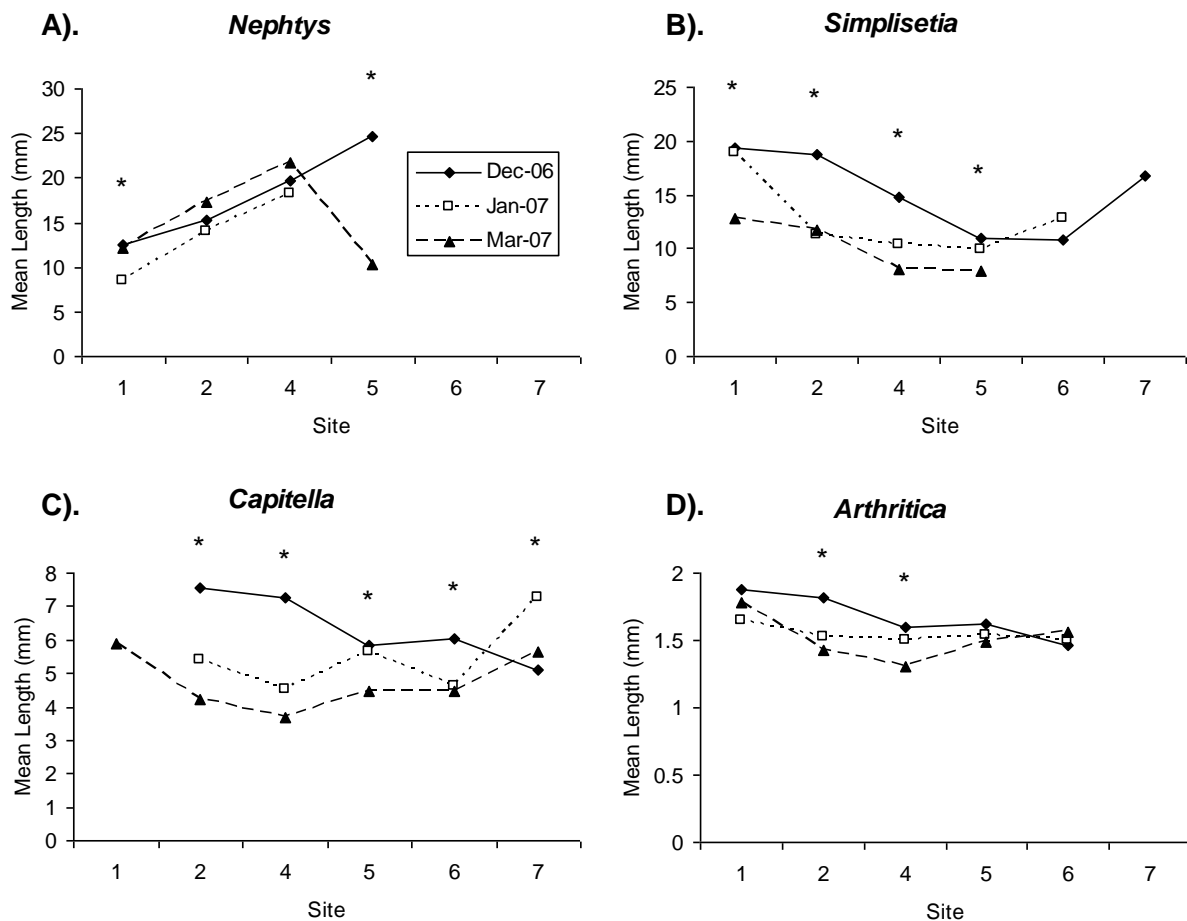


Figure 13. Mean length of four species at sites 1-7 (no species measured from sites 8-12): A). *Nephtys*; B). *Simplisetia*; C). *Capitella*, D). *Arthritica*. Standard errors are not shown to improve figure quality. Note the different y-axis scales. Asterisks indicate significant differences in length between sampling times at that respective site (One-Way ANOVA or Kruskal-Wallis Tests, $P < 0.05$ or better).

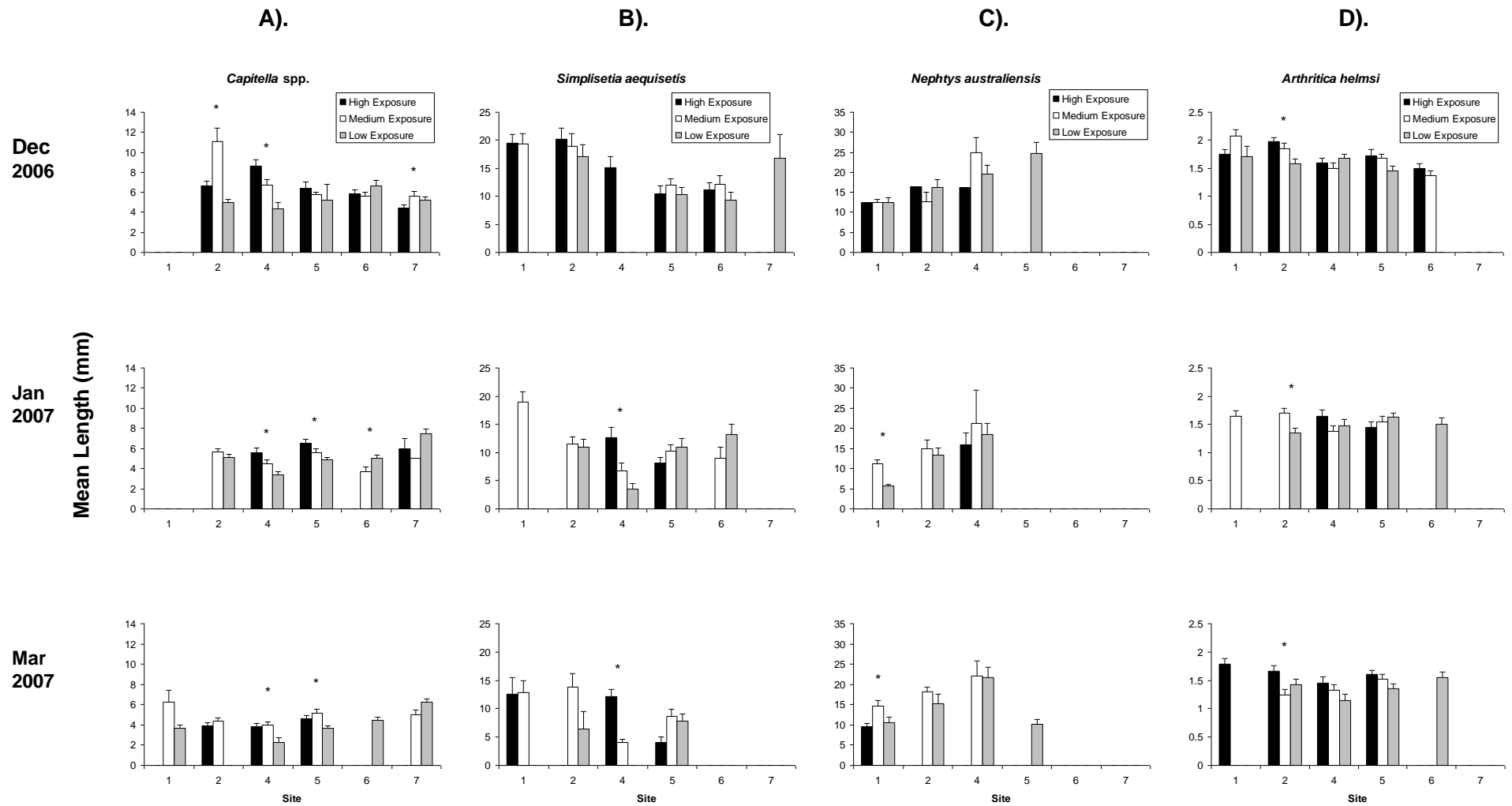


Figure 14. Length (mean \pm S.E.) of four species across three mudflat exposures (high, medium and low) at sites 1-7 for each of three sampling times (Dec 2006, Jan 2007, March 2007): A) *Capitella*; B) *Simplisetia*; C) *Nephtys*; D) *Arthritica*. Asterisks indicates significant differences in mean length between mudflat exposures at that respective site and sampling time (One-Way ANOVA or Kruskal-Wallis Tests, $P < 0.05$ or better). Note the different y-axis scales for each species.

3.3. Juvenile Macrobenthos

3.3.1. Distribution and abundance

Juveniles, defined here as encompassing all early life-stages from post-larvae to small-sized adults, of seven taxa comprising three phyla were observed throughout the sampling periods at all sites except Site 11 (Figure 15): Bivalvia (*Arthritica helmsi*); Annelida (*Simplisetia aequisetis*, *Boccardiella limnicola*, *Nephtys australiensis*, *Capitella* spp., Oligochaeta *indet*); Insecta (Chironomidae: *Tanytarsus barbitarsus*). Total numbers of juveniles showed variation at each site throughout time, with highest abundances occurring in July, except at sites 1 and 6. Generally abundances increased through the Murray Mouth region and was highest at Site 5, Pelican Point, before decreasing sharply in the North Lagoon. Only the insect larva *T. barbitarsus* was found to occur south of Site 8.

Juveniles of the polychaete worm *Capitella* spp. were by far the most abundant species, occurring throughout the Murray Mouth region and in the North Lagoon to Site 8. *S. aequisetis* juveniles were found in high abundance in May 2007 at Site 1 and in December 2006 at Sites 2 and 6, yet it was at Site 5 only where they occurred on each of the sampling times and at high abundance.

Juveniles of the polychaetes *Boccardiella limnicola* and *Nephtys australiensis* were rare in comparison to the other polychaete worms found. *B. limnicola* recruitment seemed to occur in spring/early summer with juveniles present in December 2006 and October 2007 at three sites only (sites 2, 4 and 6). Only at site 2 were *B. limnicola* juveniles found in both December 2006 and October 2007. *Nephtys australiensis* juveniles were present at Site 1 only and were in highest abundance in December 2006 and January 2007.

Except for high abundance of the bivalve *A. helmsi* in October 2007 at site 4, site 5 had greatest abundance at each sampling time. In the North Lagoon, Site 6 was generally the southern limit of distribution for *A. helmsi*, although a small number of juveniles were found as far south as Site 8 in December 2006 (Figure 15).

Combining all sites, juveniles of the four most common taxa, *Capitella* spp., *Arthritica helmsi*, *Simplisetia aequisetis* and insect larvae, were present in the study area throughout the year (Figure 16). Differences between each sampling time were significant for each species (Kruskal-Wallis Tests, $P < 0.001$ for each species). *A. helmsi* juvenile abundance did not vary greatly through the summer and autumn periods (December 2006 – May 2007) before increasing in winter and spring (July – October 2007). *S. aequisetis*, *Capitella* spp. and Chironomid larvae abundances indicated higher recruitment in the winter months.

Combining all sampling times, there was a significant difference between the Murray Mouth, North Lagoon and South Lagoon regions in the number of juvenile *A. helmsi*, *S. aequisetis* and *Capitella* spp. found (Kruskal-Wallis Tests, $P < 0.001$: Highest Rank, Murray Mouth region). Excluding the South Lagoon region due to low species diversity and abundance from these tests still resulted in significant differences between the Murray Mouth and North Lagoon regions (Kruskal-Wallis Tests, $P < 0.001$; $P < 0.005$; $P < 0.005$ for *A. helmsi*, *S. aequisetis* and *Capitella* spp. respectively).

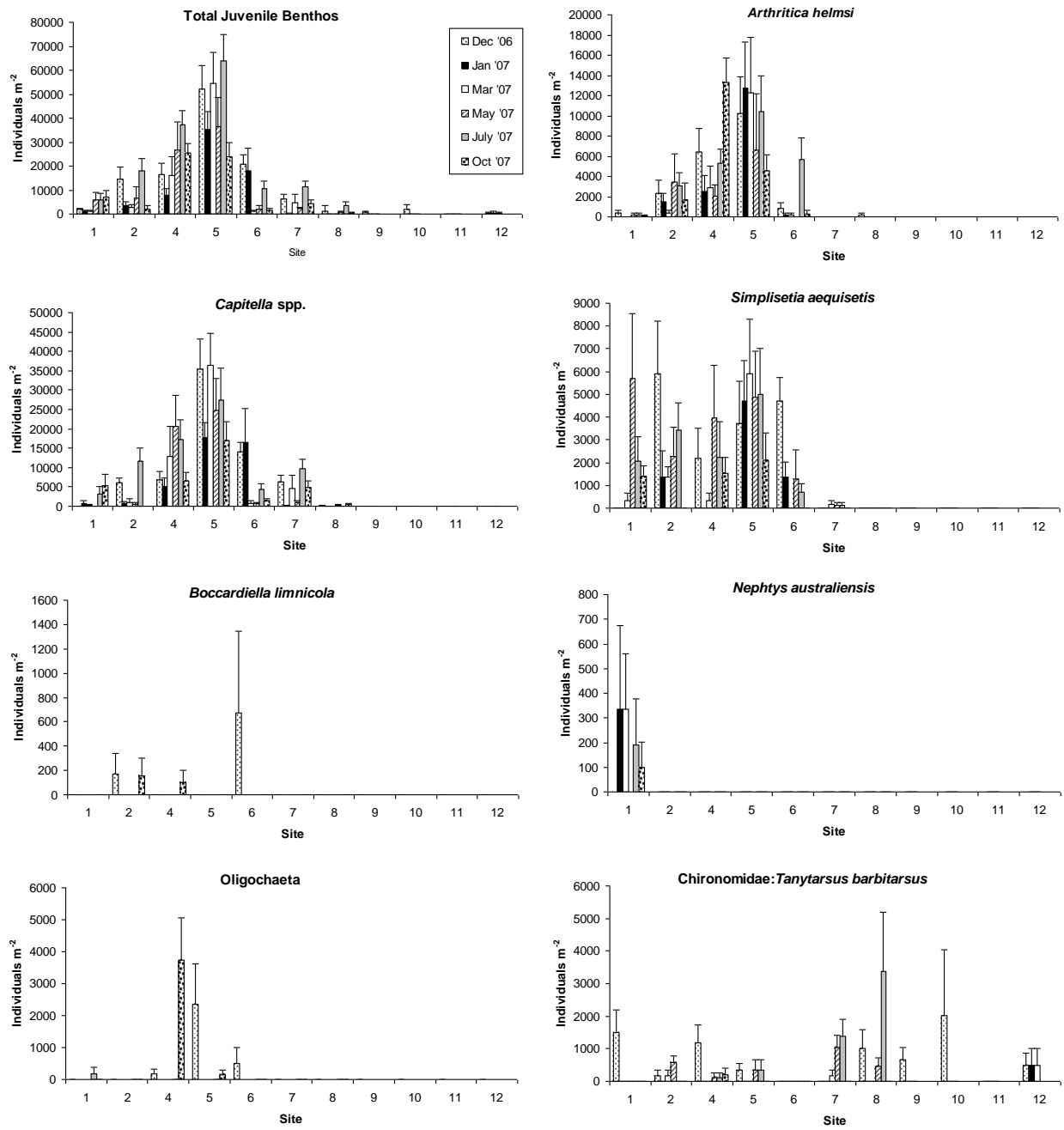


Figure 15. Distribution and abundance (mean number of individuals per $m^2 \pm SE$) of juveniles of 7 taxa and the total juvenile benthos (all taxa combined) at 11 sites over 6 sampling periods. Note the different y-axis scales for each figure.

Significant differences between individual mudflat elevation levels within the North Lagoon region were found in Jan, July and Oct 2007 for total juvenile benthos, in Jan, May and Oct 2007 for *Capitella* spp., and Jan 2007 for *S. aequisetis* (Figure 17). Significant differences between individual mudflat elevation levels within the Murray Mouth region were found only in Jan 2007 for *S. aequisetis*, July 2007 for *T. barbicansus* and Oct 2007 for *Capitella* spp.

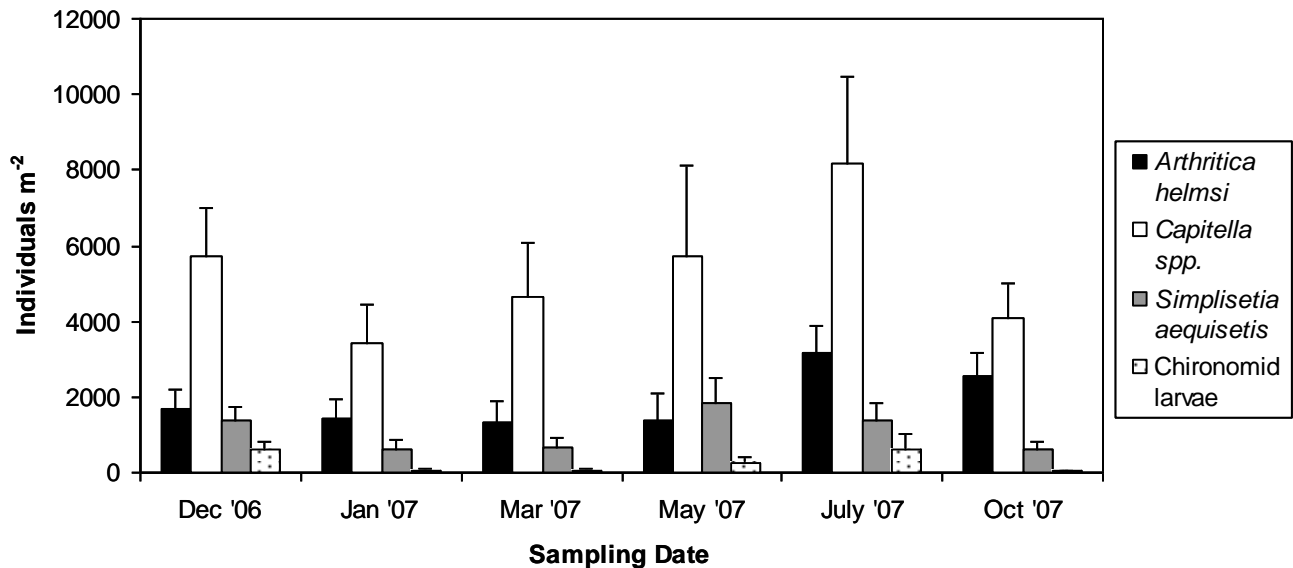


Figure 16. Juvenile presence (mean number of individuals m⁻² ± S.E.) over all of the sampling sites in the Murray Mouth and Coorong Lagoons combined for each of the sampling times.

Significant differences in juvenile abundances between the Murray Mouth and North Lagoon regions (all mudflat elevation levels combined) were found at each sampling time for total juvenile benthos and *A. helmsi*, from Jan2007 – Oct 2007 inclusive for *Capitella* spp., from Mar 2007 – Oct 2007 inclusive for *S. aequisetis* and in July 2007 only for *T. barbitarsus* (Figure 17).

In the Murray Mouth region, significant differences over time were only observed in the total juvenile benthos at high tidal exposure (Table 8, Figure 17). In the North Lagoon region, however, significant differences in total benthos were observed across all mudflat exposure levels over time. In the Murray Mouth region, the polychaetes *Capitella* spp. and *S. aequisetis* showed significant differences in abundance at the high mudflat exposure only. In the North Lagoon, there were significant differences in *Capitella* spp. abundance at both high and medium mudflat exposure levels, but for *S. aequisetis* in the North Lagoon, differences were only observed at the medium mudflat exposure level (Table 8, Figure 17). *Arthritica helmsi* showed no differences in the Murray Mouth region, but significant differences in abundance were observed at the medium and low mudflat exposure levels of the North Lagoon. For *T. barbitarsus*, significant differences in abundances were observed through time at high and low mudflat exposure levels in the Murray Mouth region, and through all exposure levels in the North Lagoon (Table 8, Figure 17).

Two major clusters within sample time and regions were revealed following Cluster and nMDS analysis (Figure 18). Sites situated in the Murray Mouth region and in the northern part of the North Lagoon region (sites 6 and 7) clustered together, whilst sites 8 and 9 from the southern North Lagoon clustered with sites from the South Lagoon, and also site 1 in December 2006. These sites were distinct in the dominance of insect larvae and the marked lack of other juvenile fauna. Site 2 from October 2007 did not group with any other site, probably due to the absence of *Capitella* spp. and *S. aequisetis* juveniles at this site in October 2007 (Figure 15). ANOSIM analyses revealed no significant differences in total juvenile fauna between sampling times, but significant differences in total juvenile fauna were found between both sites (data for all sampling times per site combined and tested between sites: One-way ANOSIM: Global R = 0.517, P = 0.001) and regions (data for all sampling times per site combined and tested between regions: One-way ANOSIM: Global R = 0.412, P = 0.001 – Figure 18 A). No significant differences were found in total juvenile fauna between sample times at each site, yet there were significant differences in total juvenile fauna between both sites over time (Two-way crossed ANOSIM, Global R = 0.291, P = 0.002) and regions over time (Two-way crossed ANOSIM,

Global R = 0.319, P = 0.004). Figure 18 B highlights the concentration of juvenile benthos abundance in the Murray Mouth region compared to the North and South Lagoons.

Juvenile and adult Capitellidae, *Simplisetia* and *Arthritica* abundance tended to follow similar patterns across all sites from Dec 2006-Mar 2007 (Figure 19). Significant positive linear correlations were observed between the number of adults and juveniles for each of Capitellidae, *Simplisetia* and *Arthritica* across all sites and sampling times ($R^2 = 0.74$, Pearson's Correlation = 0.748, P < 0.001; $R^2 = 0.75$, Pearson's Correlation = 0.85, P < 0.001; $R^2 = 0.56$, Pearson's Correlation = 0.914, P < 0.001 respectively).

Table 8. Results of Kruskal-Wallis H-Tests showing significant differences between juvenile abundances at individual mudflat exposures (High, Medium or Low) within regions (Murray Mouth, MM, and North Lagoon, NL. South Lagoon region not tested due to lack of benthos) over time. ns = no significant difference; * = P < 0.05; ** = P < 0,01; * = P < 0.001.**

Tidal Exposure	Total Benthos		<i>Arthritica helmsi</i>		<i>Capitella</i> spp.		<i>Simplisetia aequisetis</i>		<i>Tanytarsus barbitarsus</i>	
	MM	NL	MM	NL	MM	NL	MM	NL	MM	NL
High	**	**	ns	ns	*	*	**	ns	**	*
Medium	ns	***	ns	*	ns	***	ns	*	ns	*
Low	ns	**	ns	**	ns	ns	ns	ns	**	**

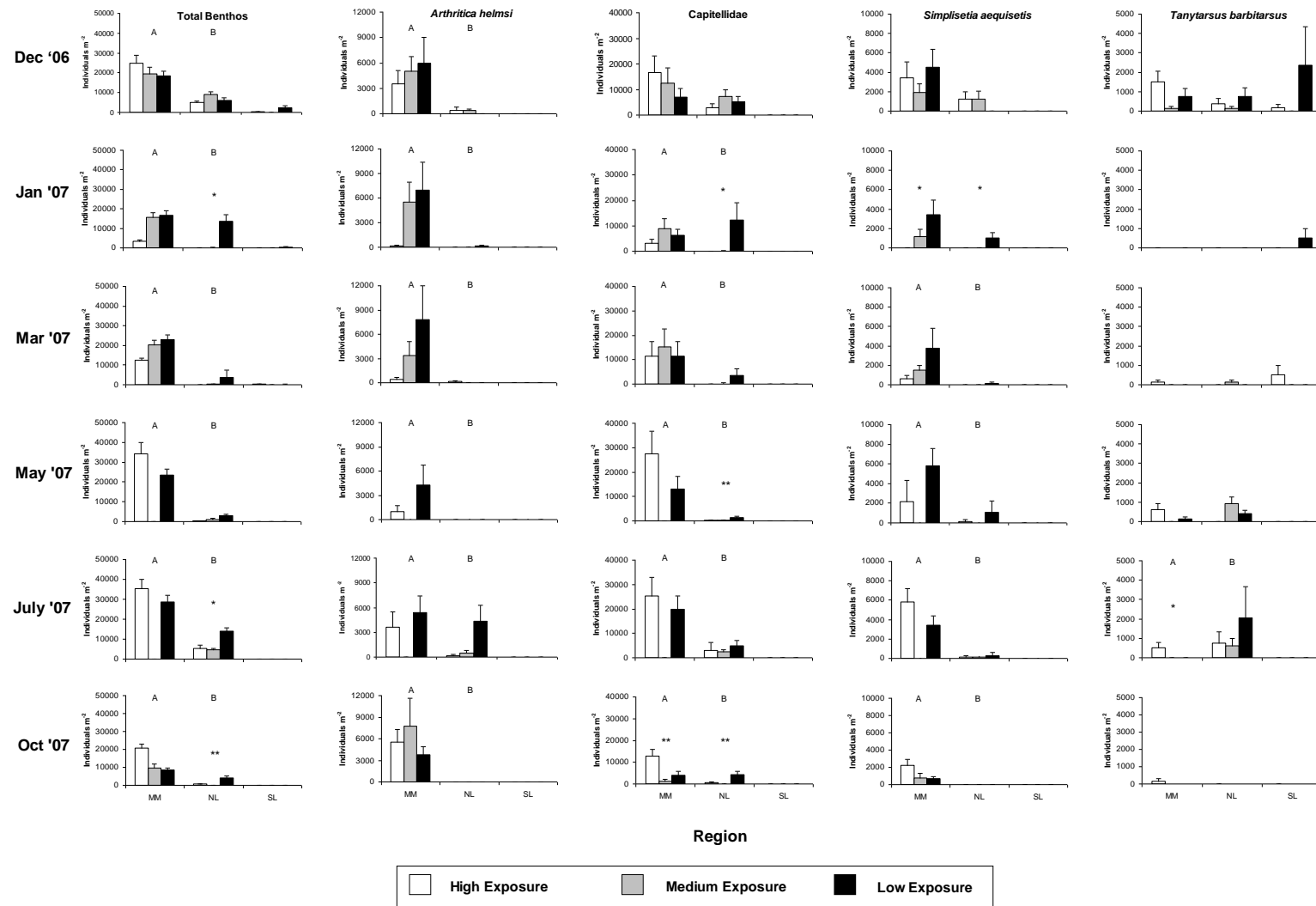


Figure 17. Juvenile benthic invertebrate abundance and distribution (total benthos and four species) within areas of mudflat exposure (High, Medium and Low) through the three regions (Murray Mouth, North Lagoon and South Lagoon) of the Coorong over six sampling times. No medium mudflat exposure samples were taken in the Murray Mouth region in both May and July 2007 due to high water levels in that region during these autumn and winter months. Note the different y-axis scales. Letters indicate significant differences between the total number of individuals (all mudflat exposures combined) between regions (due to absence of benthos, South Lagoon data was excluded from all analyses except for *T. barbicans* and Total Benthos). Asterisks indicate significant differences between mudflat exposures of the same region at that sampling time (Kruskal-Wallis Tests: * = $P < 0.05$; ** = $P < 0.01$; * = $P < 0.001$). Analyses comparing individual mudflat exposures through time are shown in Table 8.**

Of all of the environmental variables tested (salinity, oxygen saturation, water temperature, sediment grain size and sediment organic content), salinity alone gave the best (albeit low) correlation (BIOENV: correlation coefficient = 0.449). Salinity and organic content together resulted in the second highest correlation of 0.431. Global Best analysis concluded that there is agreement in multivariate pattern between juvenile abundance and the environmental variables tested above (Global Best: $\rho = 0.449$, $P = 0.001$).

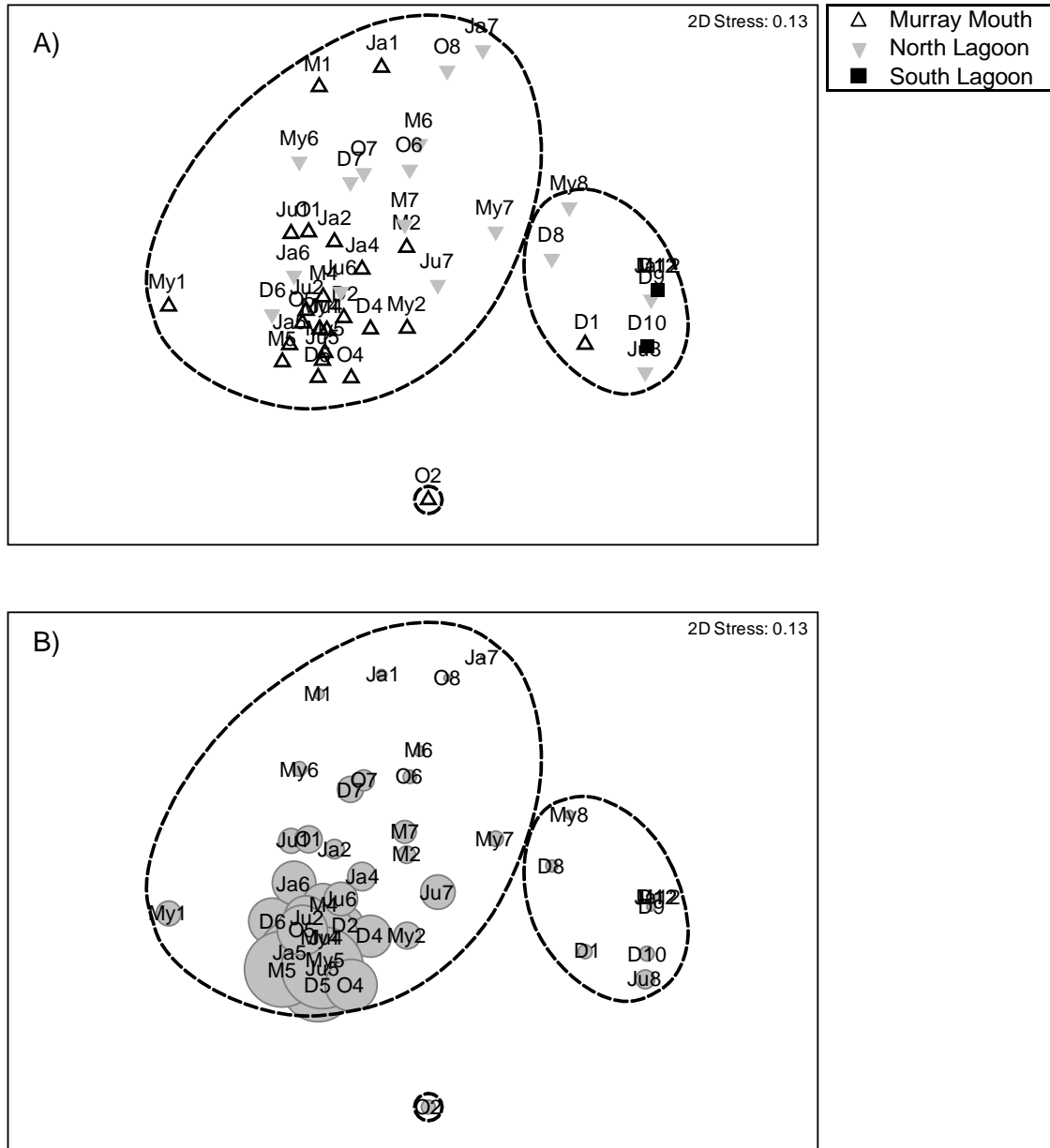


Figure 18. nMDS analysis of Total Benthos showing A). Sample times (D = Dec 2006; Ja = Jan 2007; M = March 2007; My = May 2007; Ju = July 2007; O = Oct 2007), sites (number 1 – 8) and regions (see key); B). Sample times and sites: Bubbles represent the total number of juvenile benthos recorded at each site per sampling period. Contours represent resemblance (at 20 % similarity) as a result of Cluster analysis.

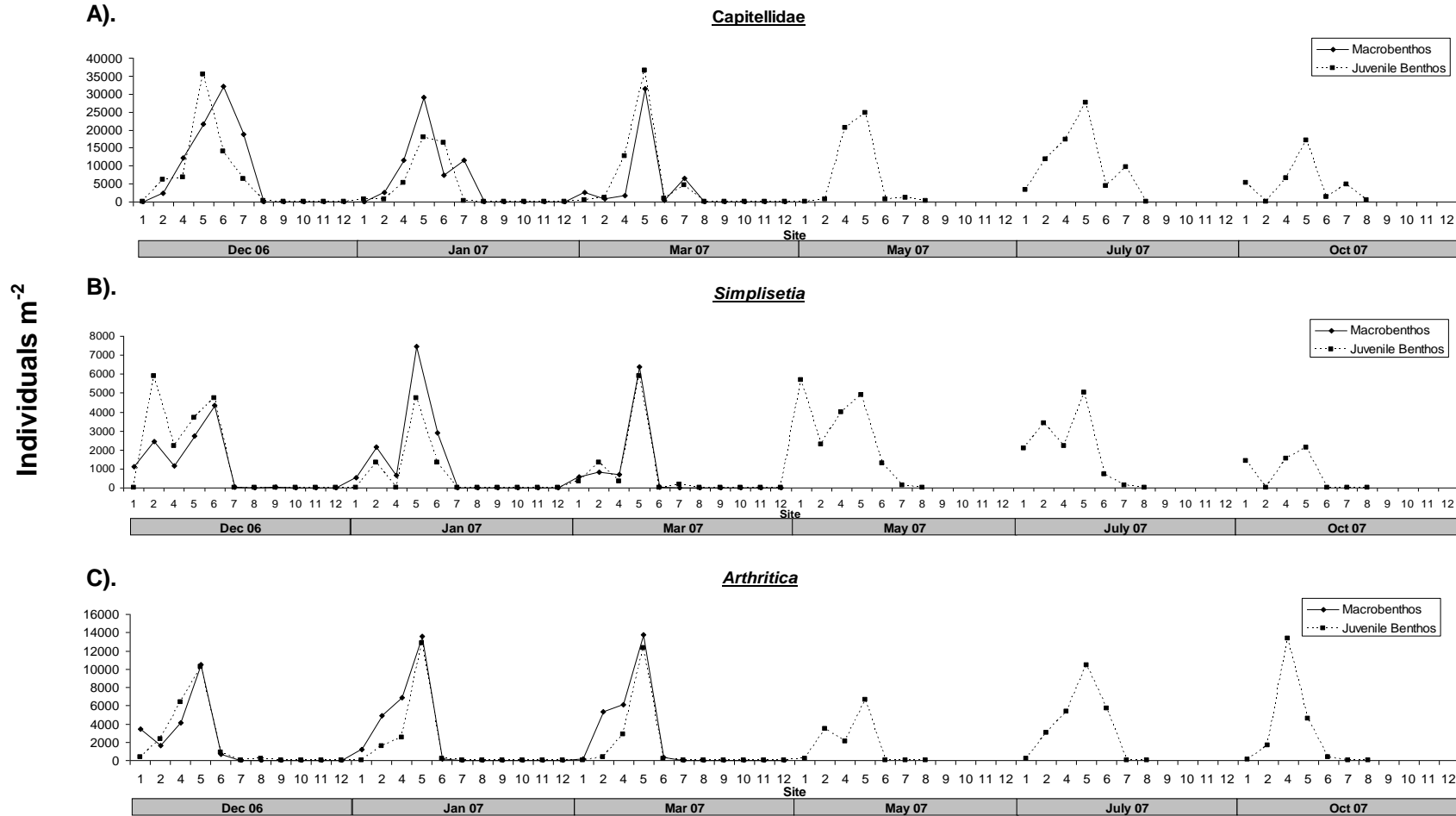


Figure 19. Distribution and abundance of both macrobenthos and juvenile A) *Capitellidae*; B) *Simplicisetia aequisetis*; C) *Arthritica helmsi* across all sites from December 2006 – October 2007. Note the different y-axis scales. No adult benthos was sampled from May – Oct 2007. Sites 9-12 not sampled for juvenile macrobenthos from May – Oct 2007.

3.4. Translocations of Infauna to Different Salinities and Exposure Levels

3.4.1. Environmental parameters

Water Quality

Both temperature and salinity did not vary greatly at each site throughout the 6 weeks of the experiment (Figure 20). Because only one reading was taken per sampling time per site, for each environmental parameter, data for all time periods were combined and a mean calculated to allow for comparisons between sites. No significant differences were found between sites for water temperature, O₂ content or O₂ saturation. Readings of O₂ content for weeks 1, 2 and 6 at Noonamena were beyond the sensitivity of the electrode used. Oxygen content was approximately constant at Ewe Island and Long Point over time, except for a decrease at week 1 at both sites, whilst oxygen saturation was variable at all three sites. Salinity was more variable at Noonamena than both Ewe Island and Long Point. Significant differences were observed between sites for salinity (Kruskal-Wallis H-Test, $\chi^2 = 9.85$, $P < 0.01$), with Noonamena having a consistently higher salinity than Long Point which in turn was consistently higher than Ewe Island.

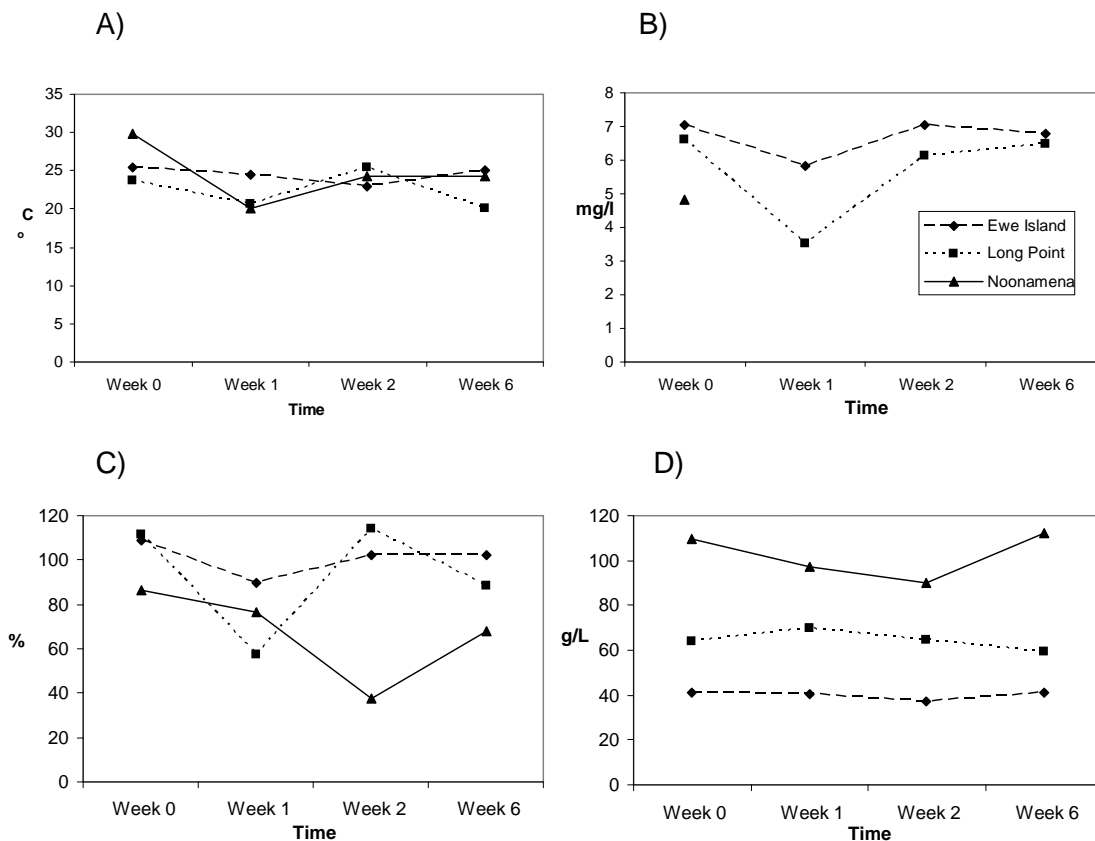


Figure 20. Water parameters measured over four sampling periods: A) Temperature; B) O₂ content; C) O₂ saturation; D) Salinity. Readings of O₂ content for weeks 1, 2 and 6 at Noonamena were beyond the sensitivity of the electrode used.

Sediment Grain Size

Median sediment grain size at each sampling time at each site was consistently within the 125 – 250 μm (Phi 2) size range. nMDS analysis of background sediment grain size fractions revealed differences between sites and elevations (Figure 21). Ewe Island (median grain size range of

173 – 180 μm over the 6 weeks) clustered separately from both Long Point and Noonamena (median grain size range of 199 – 228 μm , and 202 – 217 μm respectively). Interestingly, Long Point High Exposure samples clustered with samples from Noonamena rather than Long Point Low Exposure samples. ANOSIM analysis revealed significant differences between sites (One-way ANOSIM: Global R = 0.257, P = 0.041), but no significant differences between sample times.

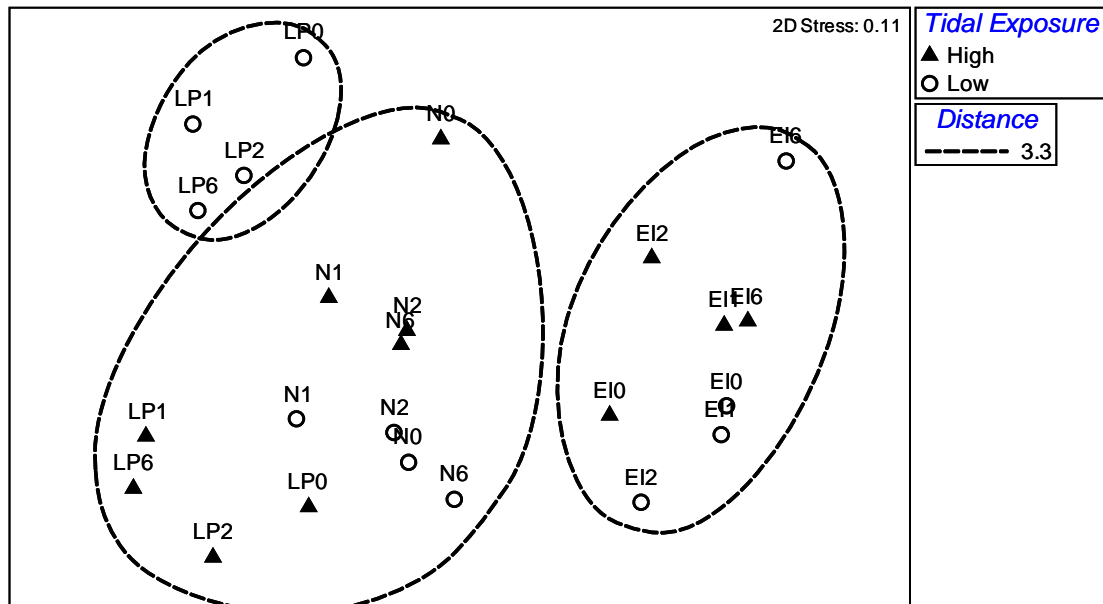


Figure 21. nMDS analysis of sediment grain size fractions from each site over four sampling periods. Data point labels indicate Site (EI = Ewe Island; LP = Long Point; N = Noonamena) and Sample Time (0 = Week 0; 1, 2 and 6 = Weeks 1, 2 and 6 respectively). Dashed lines represent cluster analysis and 3.3 euclidean distances.

Sediment Organic Content

Combining elevations and sampling times, there were significant differences in organic content between the three sites (Kruskal-Wallis H-Test: $\chi^2 = 17.01$, P < 0.001). At Ewe Island, there were no significant differences in organic content between either sampling time or mudflat exposure (Figure 22). At Long Point, there was no difference in organic content between sampling times, but there was significant difference between mudflat exposures (Kruskal-Wallis H-Test, $\chi^2 = 8.490$, P = 0.004). There was a large increase in organic content at Long Point low exposure during Week 2; however the variation between samples was high. At Noonamena, there was no significant difference in organic content between mudflat elevations, but significant differences in organic content were observed between sampling times (Kruskal-Wallis H-Test, $\chi^2 = 8.536$, P = 0.036).

PCA analysis of the measured environmental parameters and sediment grain size and organic content for each site is shown in Figure 23. Eigenvalues show that a 2-D PCA is a reasonable description of the higher dimensional space, accounting for 75% of the variability. Ewe Island samples lie close together at the negative PC1 and PC2 axes. Long Point samples are separated by cluster analysis and are spread out along both the PC axes. Noonamena samples are also separated by cluster analysis but are spread greater along the PC2 axis than the PC1 axis. Moving geographically from North to South, the samples are spread greater along the PC1 axis. Moving along this geographical gradient from North to South results in a sharp decrease in contribution from oxygen saturation, whilst salinity and organic content increase sharply in contribution with Southern proximity (in the positive PC1 direction). Temperature and grain size both contribute strongly to the PC2 axis.

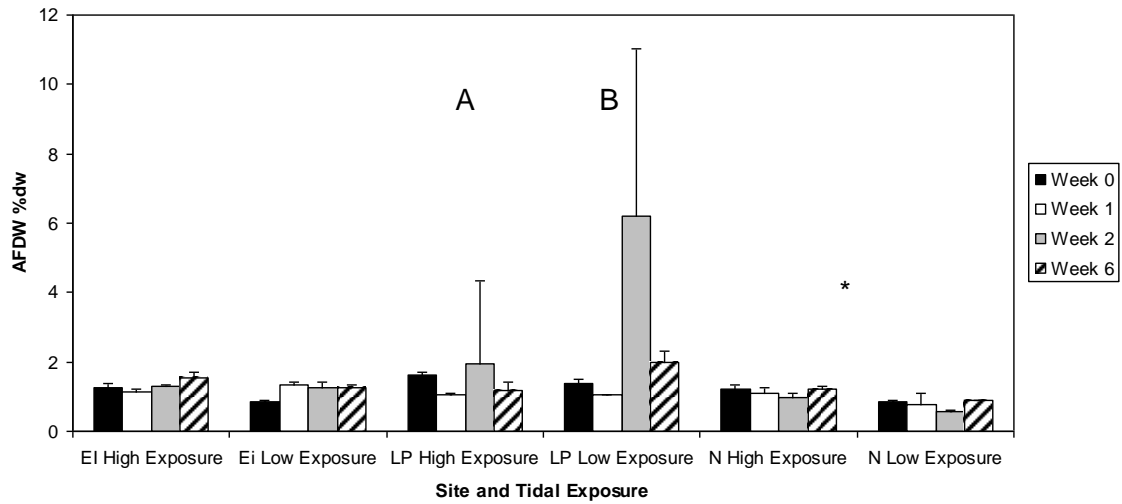


Figure 22. Sediment organic content at three sites, two mudflat exposures and four sampling times. Different letters indicate significant differences between site mudflat exposures at Long Point (High Exposure (HE) versus Low Exposure (LE): $P < 0.01$). Asterisks indicate significant differences between sample times for that respective site ($P < 0.05$). EI = Ewe Island; LP = Long Point; N = Noonamena.

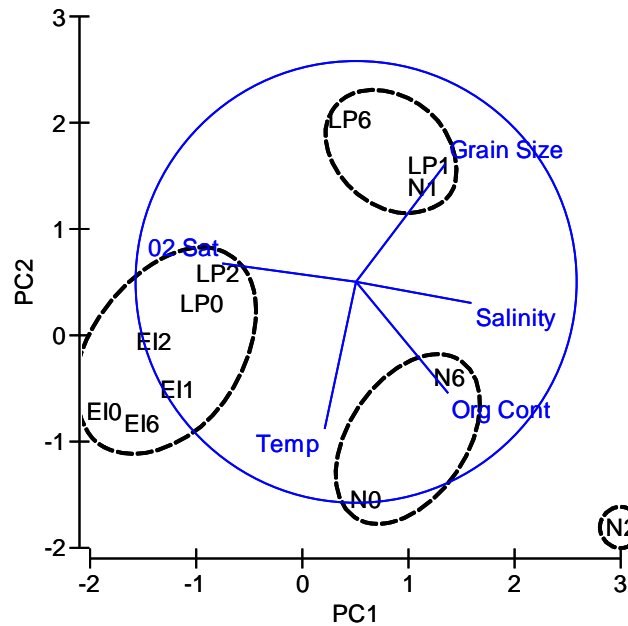


Figure 23. Principle Component Analysis ordination plot for Ewe Island (EI), Long Point (LP) and Noonamena (N) over time (0, 1, 2 and 6 weeks post translocation), showing eigenvectors for five environmental variables: Salinity, oxygen saturation, water temperature, sediment grain size and sediment organic content. Dashed lines represent cluster analysis at 3 euclidean distances.

3.4.2. Adult macrobenthic invertebrate response

Diversity and Abundance

Although high exposure plots were set up at a time when water levels provided periods of inundation at all three sites, a subsequent drop in water levels at both Long Point and Noonamena led to the high exposure plots being free from inundation for the whole 6 week period of the experiment at these two sites. As a result, no macrofauna except for limited numbers of Chironomidae larvae were recovered from the high exposure plots at Long Point and Noonamena.

Throughout the entire sampling period, between 8 and 12 different species were found to occur in the background samples of Ewe Island (Table 9). This compares with two species (*Capitella* spp. and Chironomid larvae) at Long Point and one species (Chironomid: larvae and pupae) at Noonamena. The Ewe Island fauna consisted predominantly of polychaete worms (5 species), bivalves (4 species) and gastropods (3 species: Table 9).

The background samples at each site were dominated by one species: *Arthritica helmsi* at Ewe Island, *Capitella* spp. at Long Point and Chironomid larvae at Noonamena (Figure 24). The number of Chironomid larvae decreased with time at Noonamena, but this was not reflected by an increase in the number of Chironomid pupae observed. At Long Point, a large decrease in *Capitella* was observed in week 6 in comparison to the other sampling times.

Two-way ANOSIM analysis revealed significant differences in abundance within background samples between sites (Global $R = 0.7$, $P = 0.006$) with pair-wise tests revealing differences between Ewe Island and both Long Point and Noonamena ($R = 0.858$ and 0.918 respectively, $P = 0.012$). No significant differences in abundance between background samples were observed between sample times at each site.

For control samples, two-way ANOSIM analysis revealed no significant differences between either sites or sampling time.

Backgrounds versus Controls

At all three sites, there were no significant differences in abundance between the background and control samples for any individual species and also for all species combined (total number of individuals) (Kruskal-Wallis H-Tests, $0 \leq \chi^2 \leq 3.33$, $P > 0.05$). However, differences in abundances were observed between background and control samples for *Capitella* spp. at Long Point (Kruskal-Wallis H-Test, $\chi^2 = 3.330$, $P = 0.068$) and for all species combined at both Long Point and Noonamena (Kruskal-Wallis H-Tests, $\chi^2 = 3.459$, $P = 0.063$ and $\chi^2 = 2.988$, $P = 0.084$ respectively). ANOSIM analysis revealed no significant difference between the species composition of background and control samples within sites, as is apparent from the nMDS plot shown in Figure 25. Following nMDS and CLUSTER analysis, Ewe Island high and low exposure clustered together. One sample from each of Long Point high and low exposures clustered with the majority of Noonamena samples rather than the other Long Point samples which separated into two clusters of high and low exposure respectively.

Table 9. Species distribution from Background samples across three sites over four sampling periods (0, 1, 2 and 6 weeks).

Phyla	Species	Site											
		Ewe Island				Long Point				Noonamena			
		W0	W1	W2	W6	W0	W1	W2	W6	W0	W1	W2	W6
Annelida	<i>Capitella</i> spp.	+		+	+	+	+	+	+				
	<i>Nephtys australiensis</i>	+	+	+	+								
	<i>Phyllodoce novaehollandiae</i>	+	+	+	+								
	<i>Simplisetia aequisetis</i>	+											
	<i>Australonereis ehlersi</i>	+											
	Oligochaeta indet.	+	+	+	+								
Mollusca	<i>Arthritica helmsi</i>	+	+	+	+								
	<i>Notospisula</i> sp.	+	+	+	+								
	<i>Soletellina</i> sp.	+	+	+	+								
	Tellinidae indet.		+	+	+								
	<i>Hydrobia</i> sp. 1	+											
	<i>Hydrobia</i> sp. 2		+	+	+								
	<i>Salinator fragilis</i>			+	+								
Insecta	<i>Chironomid</i> sp. larvae				+	+	+	+	+	+	+	+	+
	<i>Chironomid</i> sp. pupae					+	+	+	+	+	+	+	
	Dolichopodidae indet.			+									

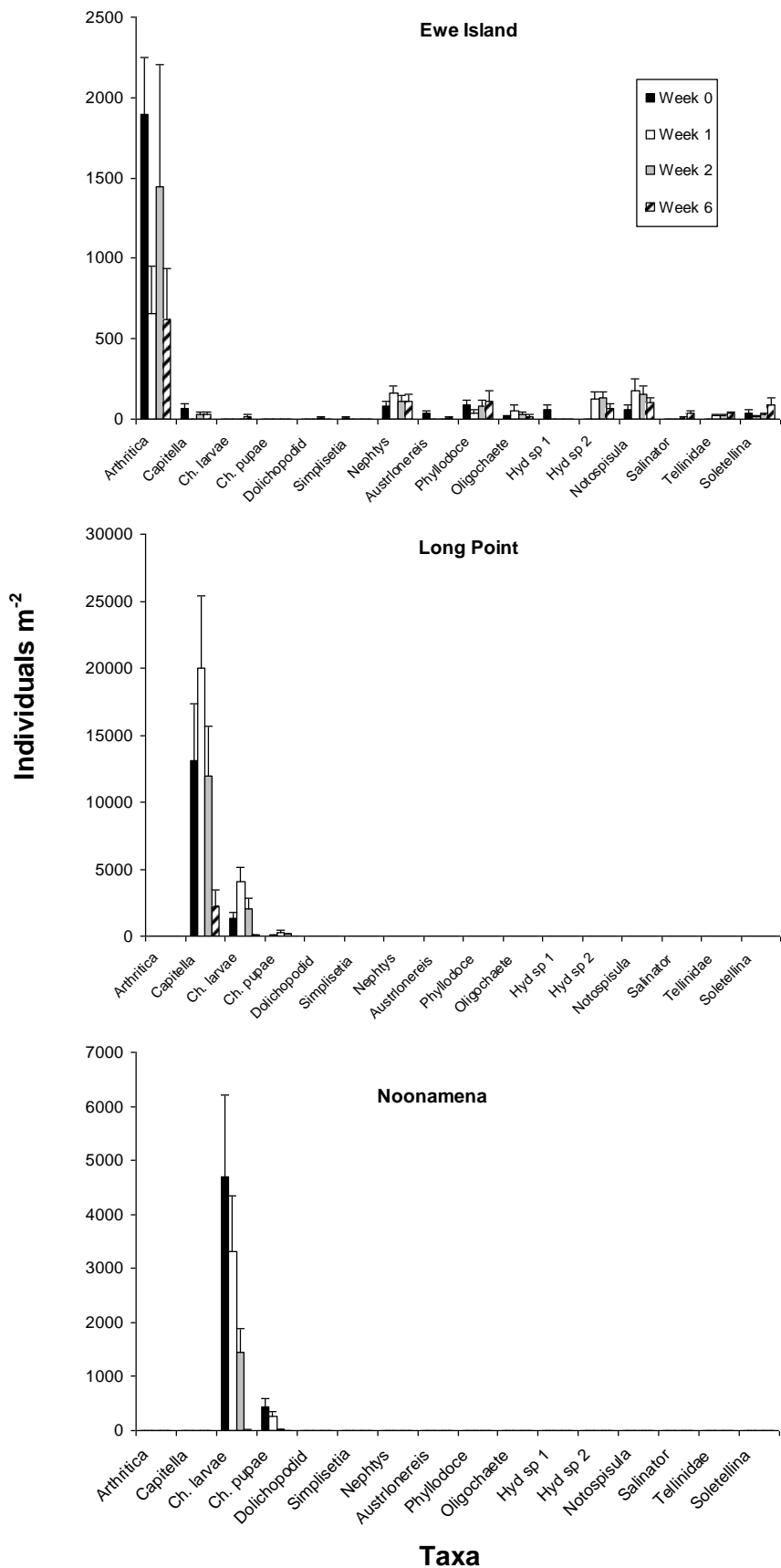


Figure 24. Distribution and abundance of benthic infauna in background samples at three sites. Note the different y-axis scales. Ch. Larvae = Chironomid larvae; Hyd. sp. = *Hydrobia* species.

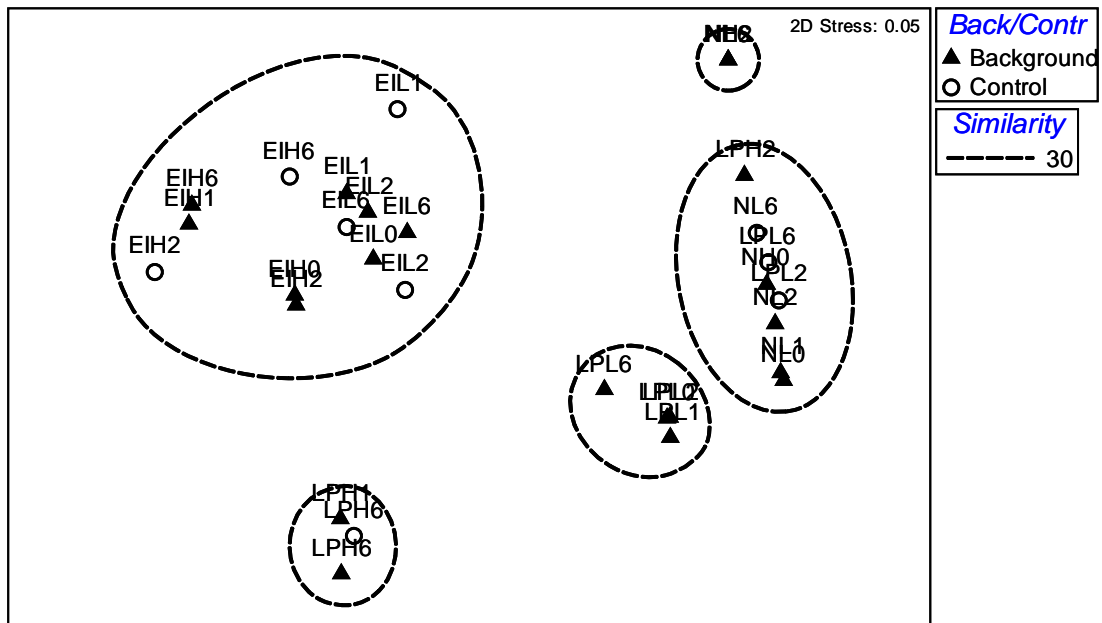


Figure 25. nMDS analysis comparing Background samples with Trampling Control Samples at each of three sites. Data labels indicate Site (EI = Ewe Island; LP = Long Point; N = Noonamena), Mudflat Exposure (H = High Exposure; L = Low Exposure) and Sample Time (1, 2 and 6 weeks). Dashed lines represent 30 % Bray-Curtis similarity following Cluster analysis.

Self Translocations

There was no significant difference in abundance (for all individual species and total species combined) between self translocations and background and control samples at Ewe Island (Long Point and Noonamena not tested due to lack of species present). Overall there were significant differences in the benthic community composition of all self translocations between the three sites (One-way ANOSIM: Global $R = 0.759$, $P < 0.001$), with pairwise comparisons showing significant differences in community composition between Ewe Island and Long Point ($R = 0.861$, $P < 0.001$), Ewe Island and Noonamena ($R = 0.941$, $P < 0.001$) and Long Point and Noonamena ($R = 0.277$, $P = 0.036$).

At Ewe Island, there was low infauna abundance within High Exposure to High Exposure (HE→HE) translocations, whilst no infauna was observed at all in this treatment at Long Point and Noonamena (Figure 26). At Ewe Island, significant differences in abundance were observed between all translocation types at week 6 (Kruskal-Wallis H-Test, $\chi^2 = 8.6$, $P = 0.036$), and there was significantly greater abundance in Low Exposure to Low Exposure (LE→LE) translocations at week 1 than in week 6 (One-way ANOVA, $F = 7.1$, $P = 0.026$) (Figure 26 A). At Long Point, significant differences in abundance were observed between all translocation types at both weeks 1 and 2 (Kruskal-Wallis H-Tests, $\chi^2 = 9.2$, $P = 0.027$, and $\chi^2 = 8.7$, $P = 0.033$ respectively), and Low Exposure to High Exposure (LE→HE) translocations in week 2 had significantly greater abundance than the same translocations in weeks 1 and 6 (Kruskal-Wallis H-Test, $\chi^2 = 6.8$, $P = 0.034$) (Figure 26 B). At Noonamena, significant differences in abundance were observed between all translocation types at week 1 (Kruskal-Wallis H-test, $\chi^2 = 9.831$, $P = 0.02$) and the same sampling time had significantly greater abundance in LE→LE translocations than weeks 2 and 6 (One-way ANOVA, $F = 29.298$, $P = 0.001$) (Figure 26 C).

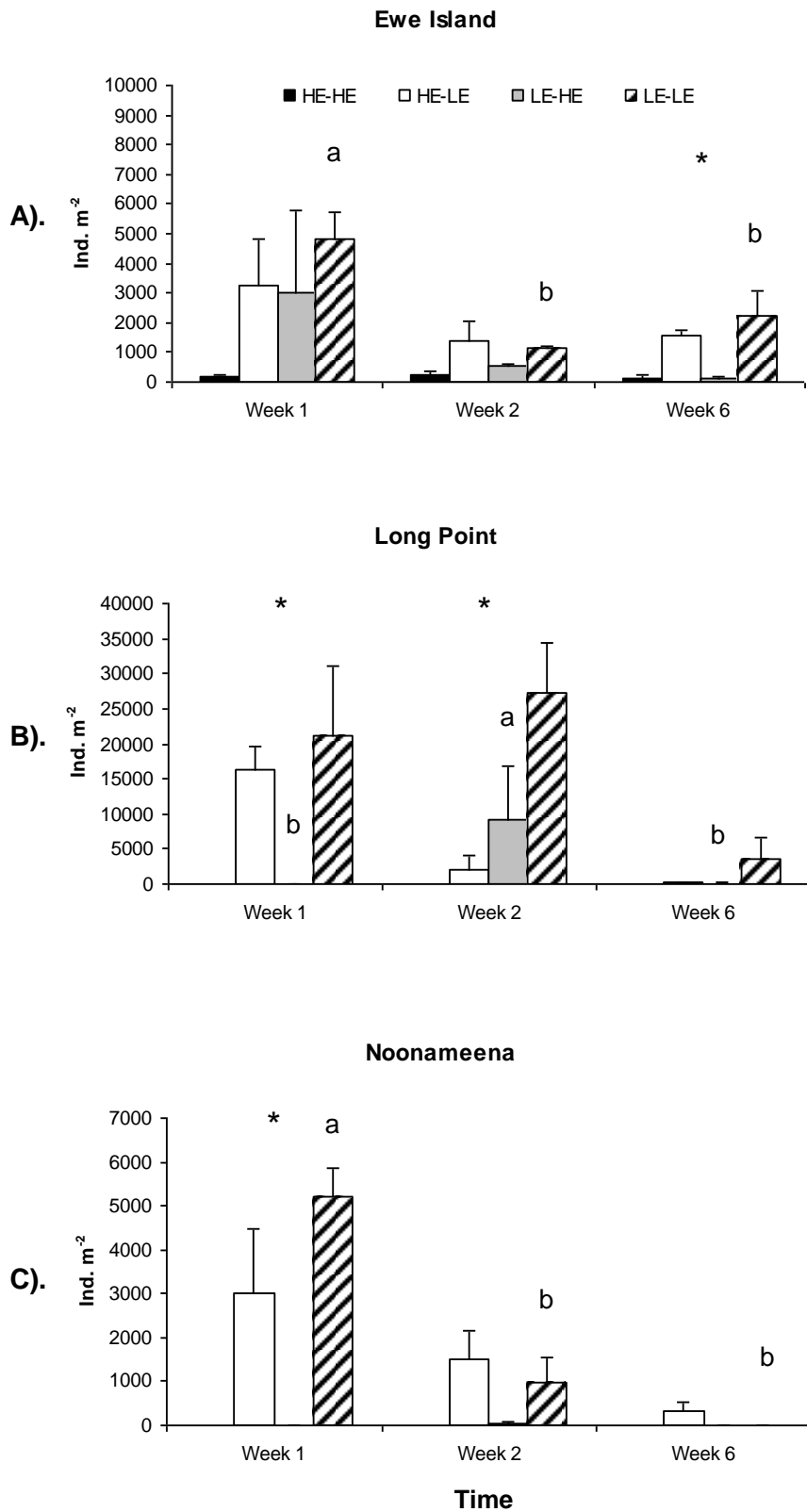


Figure 26. Mean (\pm S.E.) infauna abundance for each self translocation over time at three sites (A = Ewe Island; B = Long Point; C = Noonameena). Note the different y-axis scales. Asterisks indicate significant differences between self translocations between individual sample times at $P < 0.05$. Different letters (a,b and c) indicate significant differences between individual self translocation types through time at $P < 0.05$.

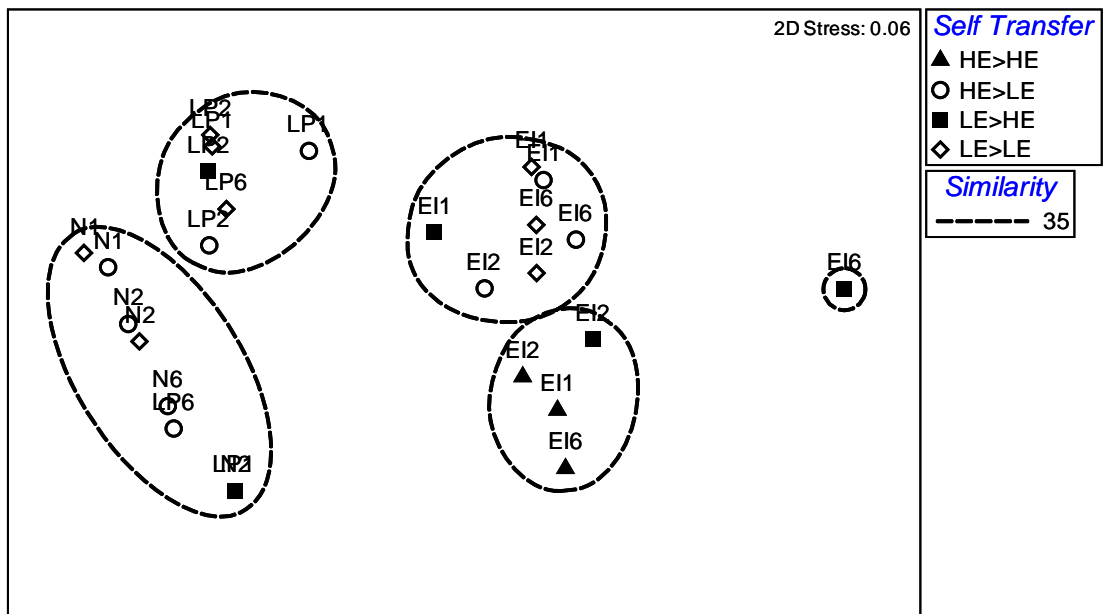


Figure 27. nMDS analysis of Self Translocations at each of three sites. Data point labels indicate Site (EI = Ewe Island; LP = Long Point; N = Noonamena) and Sample Time (1,2 and 6 weeks). Dashed lines represent 35 % Bray-Curtis similarity following cluster analysis.

There were significant differences in community composition between the self translocation treatments at Ewe Island (One-Way ANOSIM, Global R = 0.316, P = 0.014), but no difference in community composition between self translocations over time for each of the three sites (Figure 27). Ewe Island LE→HE self translocations one week post translocation were similar to HE→HE and LE→LE translocations (Figure 27). By two weeks post translocation, Ewe Island LE→HE translocations had similar communities to HE→HE translocations, yet by six weeks post translocation, the community within Ewe Island LE→HE translocations were different to the other Ewe Island self translocations. Ewe Island translocations from HE→LE clustered closely with Ewe Island LE→LE translocations for all sample times (Figure 27).

At both Long Point and Noonamena, no organisms were recovered from HE→HE translocations. Long Point LE→HE and HE→LE translocations (1 and 6 weeks post translocation respectively) had similar communities to all samples from Noonamena. Following ANOSIM analysis, for both Long Point and Noonamena, there were no significant differences in community composition between self translocations over time.

The results of statistical tests following comparisons of the abundance of individual species between different sediment translocations (HE→HE, HE→LE, LE→HE and LE→LE) at each site over time are shown in Table 10. At each site, significant differences were observed between self translocations for certain species (Table 10: Analysis 1 for each site). At Ewe Island, no significant differences were observed between self translocations and background and trampling control samples. However, significant differences in abundance between self translocations and background and control samples were observed for *Capitella* spp. and Chironomid larvae at Long Point and Noonamena respectively (Table 10: Analysis 2). At Ewe Island significant differences were observed in abundance of both *Notospisula* sp. and the total number of individuals in LE→LE self translocations over time. The same self translocation showed significant differences in the number of Chironomid larvae through time at Noonamena. At Long Point, significant differences were observed in the abundance of *Capitella* sp. and Chironomid larvae in HE→LE and LE→HE samples respectively through time (Table 10: Analyses 3b and 3c respectively).

Table 10. Results of statistical analyses, testing between different species present in self translocations through time at each of three sites. Analyses performed were: 1). Testing for differences in the number of individuals of each species between each self translocation (HE→HE, HE→LE, LE→HE and LE→LE); 2). Testing for differences in the number of individuals of each species between self translocations, background and trampling control samples; 3). Testing for differences in the number of individuals of each species in each translocation (a: HE→HE; b: HE→LE; c: LE→HE and d: LE→LE) between sampling times (1,2 and 6 weeks). Post-hoc differences calculated using Tukey's Post-hoc test. N/A = Not applicable; ns = non-significant; - = no tests performed due to lack of infauna.

Site	Analysis	Statistical Test	Statistical Difference	Statistical Result	Post-Hoc differences
Ewe Island	1	Kruskal-Wallis H-Test	<i>Nephtys australiensis</i>	$X^2 = 9.6, P = 0.022$	N/A
			<i>Arthritica helmsi</i>	$X^2 = 10.2, P = 0.017$	
			<i>Notospisula</i> sp.	$X^2 = 10.8, P = 0.013$	
			Total Individuals	$X^2 = 16.3, P = 0.001$	
	2	Kruskal-Wallis H-Test	ns		
	3a	One-Way ANOVA	ns		
	3b	Kruskal-Wallis H-Test	ns		
3c	Kruskal-Wallis H-Test	ns			
3d	One-Way ANOVA	<i>Notospisula</i> sp.	$F = 16.8, P = 0.004$	W1+W2; W2+W6	
		Total Individuals	$F = 7.1, P = 0.026$	W1+W2	
Long Point	1	Kruskal-Wallis H-Test	<i>Capitella</i> sp.	$X^2 = 8.7, P = 0.034$	N/A
			Chironomid larvae	$X^2 = 13.8, P = 0.003$	
			Total Individuals	$X^2 = 16.4, P = 0.01$	
	2	Kruskal-Wallis H-Test	<i>Capitella</i> sp.	$X^2 = 12.0, P = 0.02$	N/A
	3a	-			
	3b	Kruskal-Wallis H-Test	<i>Capitella</i> sp.	$X^2 = 7.6, P = 0.022$	N/A
	3c	Kruskal-Wallis H-Test	Chironomid larvae	$X^2 = 6.7, P = 0.035$	N/A
3d	Kruskal-Wallis H-Test	ns			
Noonamena	1	Kruskal-Wallis H-Test	Chironomid larvae	$X^2 = 17.7, P = 0.001$	N/A
	2	Kruskal-Wallis H-Test	ns		
	3a	-			
	3b	One-Way ANOVA	ns		
	3c	-			
	3d	One-Way ANOVA	Chironomid larvae	$F = 28.7, P = 0.01$	W1+W2; W1+W6; W2+W6.

Out Translocations

For each site individually, ANOSIM analysis revealed no significant differences in community composition between out translocations and the background samples from that respective site, over time (Figure 28). At Ewe Island, those samples that tend to be positioned furthest from Ewe Island (EI) background samples in Figure 28 A were generally out samples from Long Point (LP) or Noonamena (N) after 1 or 2 weeks post translocation, although there are some such samples that are positioned close to EI background samples. Such differences were not observed at Long Point or Noonamena (Figure 28 B and C respectively).

The change in species numbers in out translocations to each site when compared to the background fauna from the site from which the translocation was taken (translocation origin) is shown in Figure 29. All translocations to Ewe Island resulted in an increase in the number of species. Translocations from Long Point to Ewe Island High Exposures had a greater number of species than translocations from Long Point to Ewe Island Low Exposure. The reverse was true for translocations from Noonamena (Figure 29A). Translocations from Ewe Island to the higher salinities at Long Point and Noonamena resulted in a dramatic decrease in the number of species. Little change was observed in translocations between Long Point and Noonamena (Figures 29 B and C).

Sediment translocated from the hypersaline North Lagoon to the estuary were quickly colonised, in particular at low tidal elevations. Translocations to and from each mudflat exposure from both Long Point and Noonamena to Ewe Island, did not differ significantly in the abundance of individuals in each translocation over time (for example there was no difference between Long Point to Ewe Island HE → HE translocations over time: Figure 30 A). Significant differences in abundance of individuals were observed between mudflat exposures for Annelida after two weeks (Long Point to Ewe Island Outs: Kruskal-Wallis Test, $\chi^2 = 8.3$, $p = 0.04$; Figure 30 B) and for both Total Number of Taxa and Total Mollusca after two weeks (Noonamena to Ewe Island Outs: Kruskal-Wallis Tests, $\chi^2 = 9.6$, $P = 0.023$; $\chi^2 = 9.5$, $P = 0.023$; Figures 30 A and C respectively), and also for Total Annelida after 6 weeks (Kruskal-Wallis Test, $\chi^2 = 8.8$, $P = 0.032$; Figure 30 B).

No molluscs were present in any out translocations from Ewe Island or Noonamena to Long Point. Few, if any, individuals were present in out translocations from both Ewe Island and Noonamena to Long Point High Exposures when compared to translocations to Long Point Low Exposures (Figure 31). Significant differences between mudflat exposures (translocated from Ewe Island to Long Point) were observed at week 2 for both the total abundance across all taxa and for total Annelida (Kruskal-Wallis Tests, $\chi^2 = 10.2$, $P = 0.017$ for both respectively) and at weeks 1 and 2 for total Insecta (Kruskal Wallis Tests, $\chi^2 = 10.7$, $P = 0.013$; $\chi^2 = 10.2$, $P = 0.017$ respectively). For both total number of individuals across all taxa and total Annelida, there were significantly less abundances in HE → LE outs 6 weeks after translocation from Ewe Island (One-Way ANOVAs, $F = 5.02$, $P = 0.049$; $F = 5.02$, $P = 0.05$ respectively). Significantly more Insecta were present in both HE → LE and LE → LE outs two weeks post translocation from Ewe Island (Kruskal-Wallis Tests, $\chi^2 = 7.3$, $P = 0.026$; $\chi^2 = 6.8$, $P = 0.034$ respectively).

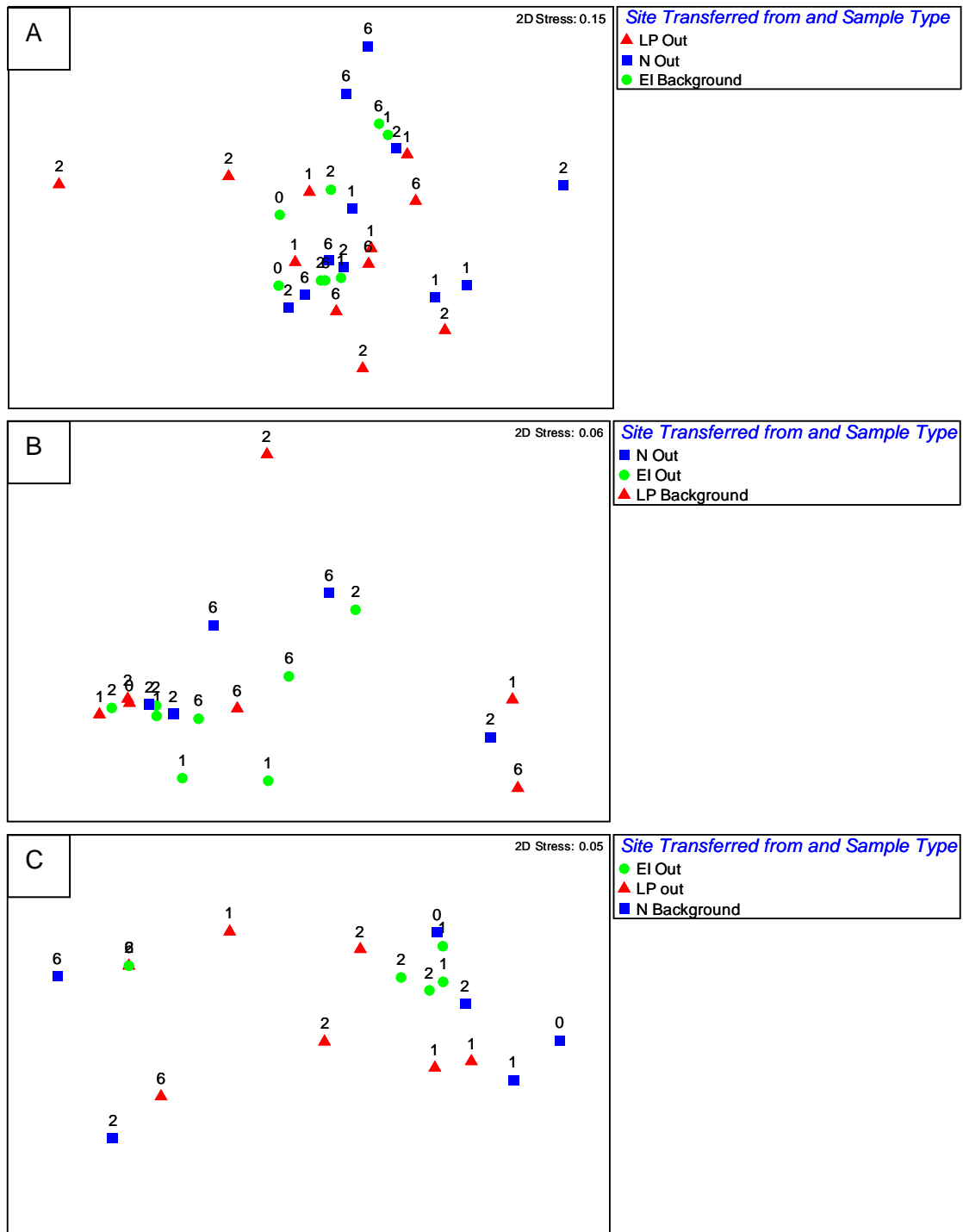


Figure 28. nMDS analysis of out translocations to, and Background samples from, A) Ewe Island; B) Long Point and C) Noonamena. Data points for each type of sample represent both High and Low tidal exposures (not individually labelled to improve figure clarity). Data labels indicate the sample time (in weeks) post translocation.

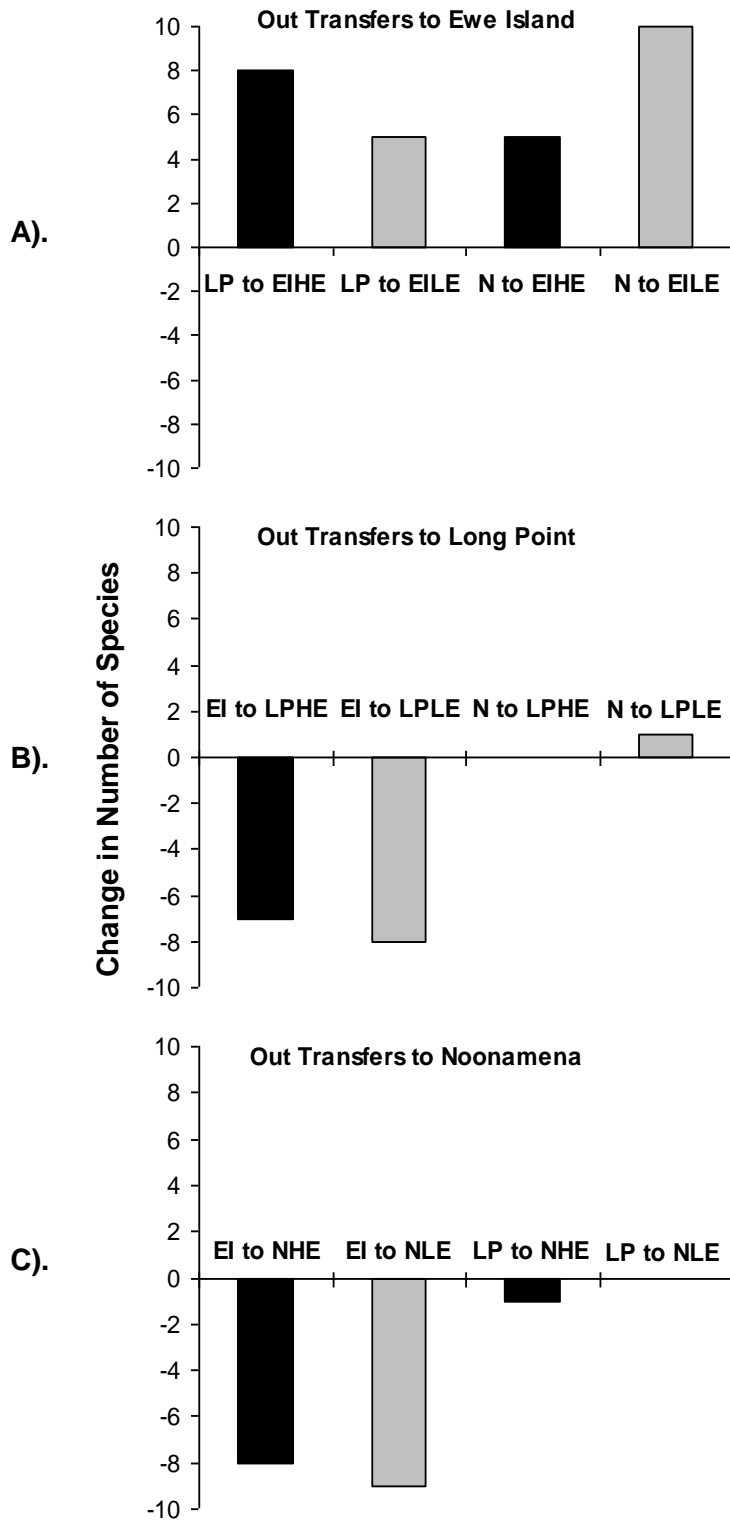


Figure 29. The change in the number of species present in out translocations (High Exposure and Low Exposure combined for all time periods) to High and Low mudflat exposures at A). Ewe Island; B). Long Point; C). Noonamena when compared to the Week 0 background fauna from the site from which the translocation was taken (translocation origin).

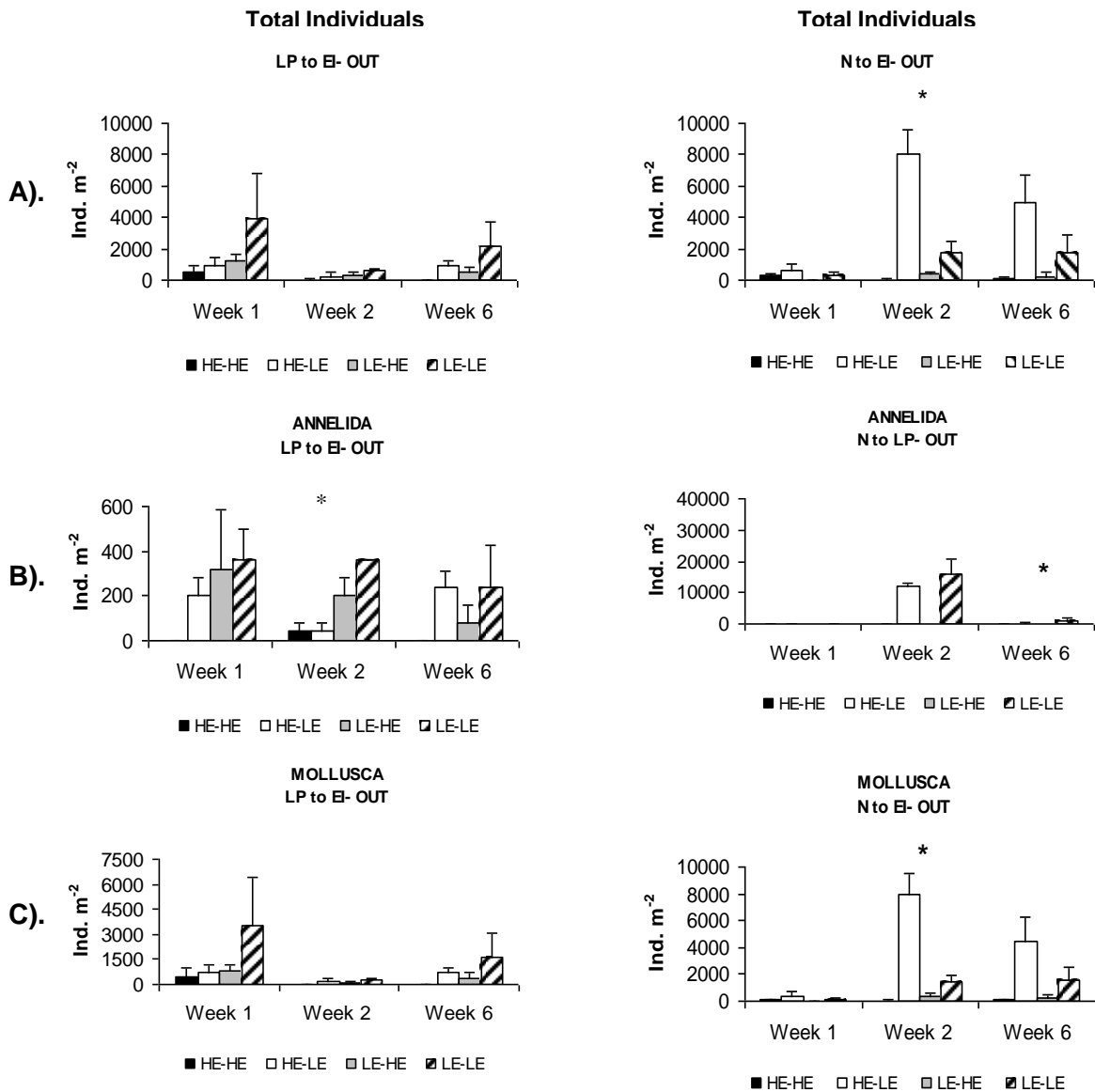


Figure 30. Number of individuals per square meter in Long Point (LP) to Ewe Island (EI) Outs, and Noonamena (N) to Ewe Island Outs for A). Total number of individuals over all taxa; B). Total number of Annelida; C) Total Number Mollusca. No insect larvae were present. Asterisks indicate significant differences between translocation exposures at individual sampling times.

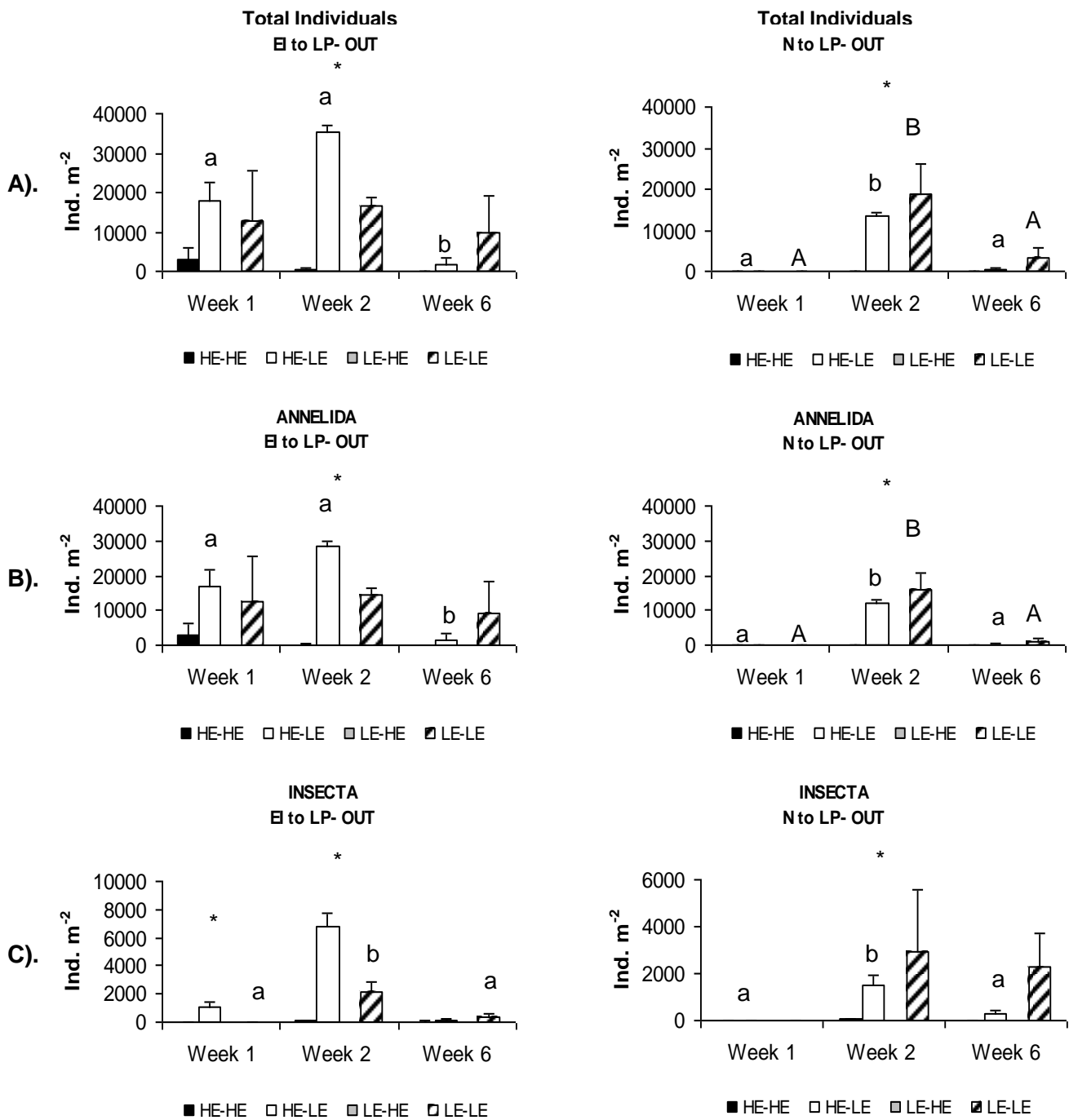


Figure 31. Number of individuals per square meter in Ewe Island (EI) to Long Point (LP) Out transfers, and Noonamena (N) to Long Point Out translocations for A). Total number of individuals over all taxa; B). Total number of Annelida; C) Total Number Insecta. No molluscs were present. Asterisks indicate significant differences between translocation exposures at individual sampling times. Lower case letters represent significant differences between mudflat exposures over time. Upper case letters represent significant differences between mudflat exposures at a single sampling time.

For out translocations from Long Point to Noonamena, significant differences were observed between mudflat exposures for each of total abundance across all taxa, total Annelida and total Insecta (Kruskal Wallis Tests, $9.1 \leq \chi^2 \leq 9.6$, $P < 0.03$; Figure 32). For each of total number of individuals across all taxa, total Annelida and total Insecta, there were significantly more individuals in both HE \rightarrow LE and LE \rightarrow LE outs two weeks post translocation (Kruskal-Wallis Tests, $\chi^2 = 6.8$, $P = 0.034$, for each test, Figure 32A).

No Annelids were present in Ewe Island to Noonamena outs over all time periods, and Molluscs were present in very low numbers in Ewe Island to Noonamena outs after week one only (data not shown), but were not present at all in Long Point to Noonamena outs. Therefore for translocations from both Ewe Island and Long Point to Noonamena, the total number of individuals across all taxa was dominated by the Insecta (Figure 32 C). For Ewe Island to Noonamena outs, there were significant differences in the abundance of individuals between mudflat exposures at week two (Kruskal-Wallis Test, $\chi^2 = 9.5$, $P = 0.023$), and significant difference between LE \rightarrow LE translocations over time (One-Way ANOVA, $F = 30.5$, $P = 0.001$: Figure 32 A). For Long Point to Noonamena outs, there was significant difference in the number of individuals between mudflat exposures at week 1 (Kruskal-Wallis Tests, $\chi^2 = 8.8$, $P = 0.032$ and $\chi^2 = 8.4$, $P = 0.039$ for total abundance of individuals across all taxa and total Insecta respectively) (Figures 32 A and C). Significant differences were also observed in the abundance of individuals in LE \rightarrow LE translocations over time (Kruskal Wallis Test, $\chi^2 = 7.4$, $P = 0.024$; One-Way ANOVA, $F = 6.8$, $P = 0.028$ for total abundance of individuals across all taxa and total insecta respectively) (Figures 32 A and C).

Except for Total Mollusca one week post translocation from Noonamena to Ewe Island, there was no significant differences in the abundance of individuals between background samples from both Ewe Island and Long Point compared with out translocations from Long Point and Noonamena to Ewe Island and Ewe Island to Long Point (Table 10). Translocations from Noonamena to Long Point however showed significant differences in abundance compared to Long Point background samples in Total Annelida, Total Insect and Total Taxa (no Mollusca present) after one week post translocation only. Significant differences in abundance were also observed for the same translocation (Noonamena to Long Point) over all sampling times for Total Annelida and Total Insecta (Table 10). There were no significant differences in abundance between out translocations from both Ewe Island and Long Point to Noonamena and background samples from Noonamena over all sampling times and also at each sampling time (Table 11). Molluscs were present in Long Point and Ewe Island out translocations to Noonamena after one week, but not present at later sampling times.

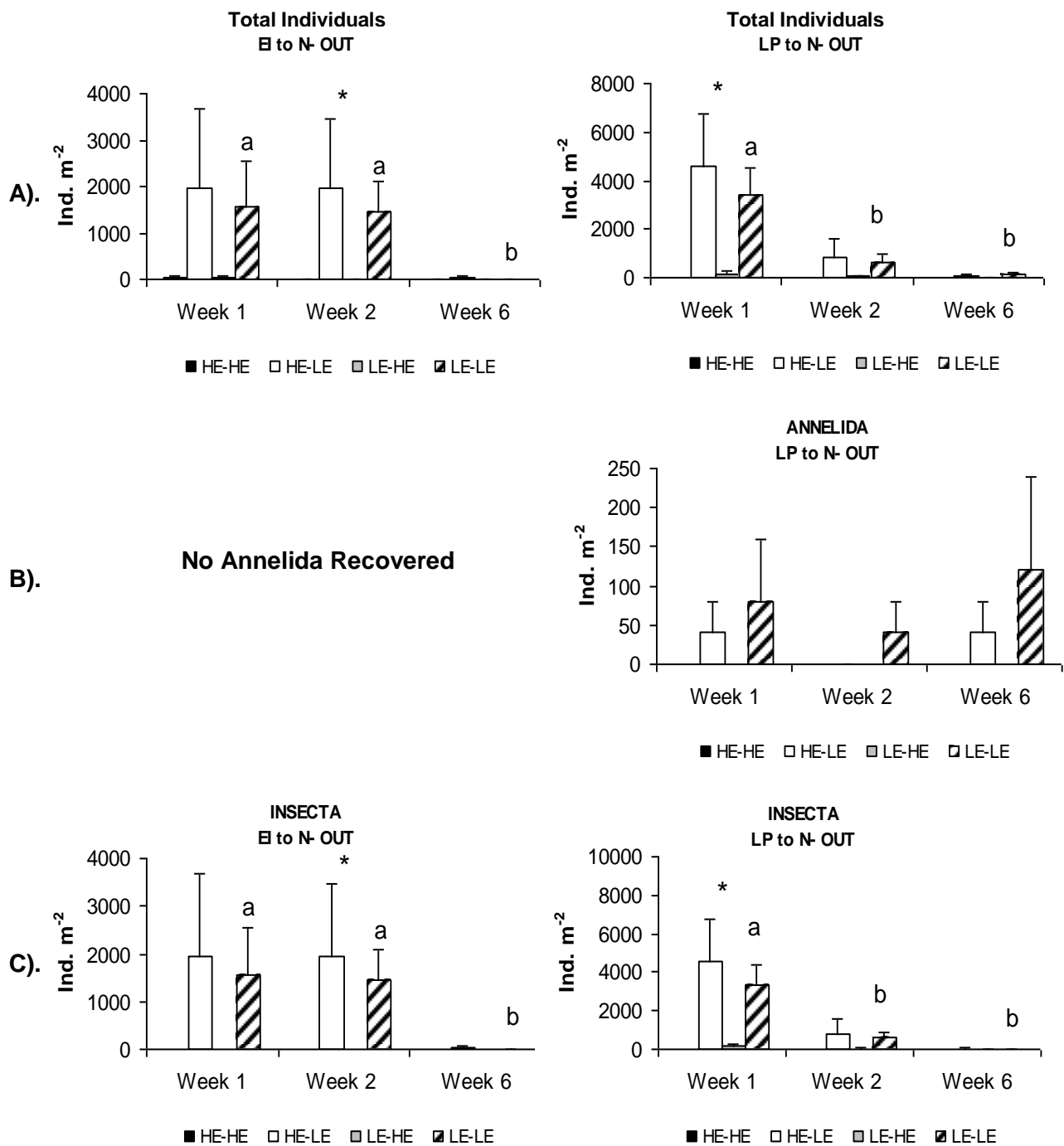


Figure 32. Number of individuals per square meter in Ewe Island (EI) to Noonamena (N) Out translocations, and Long Point (LP) to Noonamena Out translocations for A). Total number of individuals over all taxa; B). Total number of Annelida; C) Total Number Insecta. No molluscs were present. Asterisks indicate significant differences between translocation exposures at individual sampling times. Lower case letters represent significant differences between mudflat exposures over time. No Annelids were present in Ewe Island to Noonamena outs at any sampling time. Molluscs were present in very low numbers in some Ewe Island out translocations, and absent in Long Point out translocations and are not shown here.

Table 11. Results of statistical analyses (Kruskal-Wallis or One-Way ANOVA tests) testing the number of individuals found within out translocations against the respective background samples from the translocation destination site. 'ns' = non-significant; '-' = No individuals present. Asterisks indicate tests where $0.05 > P < 0.075$.

Out Translocation	Sampling Time	Total Annelida	Total Mollusca	Total Insecta	Total Taxa
LP → EI	W1	ns	ns	ns	ns
	W2	ns	ns*	ns	ns
	W6	ns	ns	ns	ns
	Over all Sampling times	ns	ns	ns	ns
N → EI	W1	ns	F = 7.4, P = 0.011	ns	ns
	W2	ns	ns	ns	ns
	W6	ns	ns	ns	ns
	Over all Sampling times	ns*	ns	ns	ns
EI → LP	W1	ns*	ns	-	ns*
	W2	ns	ns	-	ns
	W6	ns	ns	-	ns
	Over all Sampling times	ns	ns	-	ns
N → LP	W1	$\chi^2 = 11.9$ P = 0.001	-	$\chi^2 = 11.857$ P = 0.001	$\chi^2 = 7.949$ P = 0.005
	W2	ns	-	ns	ns
	W6	ns	-	ns	ns
	Over all Sampling times	$\chi^2 = 8.1$ P = 0.005	-	$\chi^2 = 7.1$ P = 0.008	ns
EI → N	W1	ns	ns*	ns	ns
	W2	ns	-	ns	ns
	W6	-	-	ns	ns
	Over all Sampling times	ns	ns	ns	ns
LP → N	W1	ns	ns	ns	ns
	W2	ns	-	ns	ns
	W6	ns	-	ns	ns
	Over all Sampling times	ns	ns	ns	ns

Biomass

Ewe Island background samples, irrespective of high or low exposure levels, showed no significant difference between sampling times for overall mean biomass (all taxa combined) and also mean biomass for Annelida, Mollusca and Insecta (Figure 33 A). At Long Point, for background samples, there was significant differences in biomass over time for overall mean biomass and mean Insecta biomass (Kruskal-Wallis Tests: $\chi^2 = 9.043$, $P = 0.029$ and $\chi^2 = 7.709$, $P = 0.05$ respectively), but not for mean Annelida biomass (Figure 33 D). Overall mean biomass and mean Insecta biomass from Noonamena background samples also showed significant differences in biomass over time (Kruskal-Wallis Tests: $\chi^2 = 15.508$, $P = 0.001$ and $\chi^2 = 14.258$, $P = 0.003$ respectively) (Figure 33 G).

Comparing Ewe Island background biomass against that of out translocations from both Long Point and Noonamena: Significantly greater biomass was present in Ewe Island background samples for Mollusca compared with N → EI out translocations (Kruskal-Wallis Test: $\chi^2 = 4.978$, $P = 0.026$) one week post translocation (Figures 33 A, B and C). No other significant differences were observed between Ewe Island background samples and out translocations from Long Point and Noonamena.

Comparing Long Point background biomass against that of out translocations from both Ewe Island and Noonamena: Significantly greater biomass was present in Long Point background samples for overall mean biomass, mean Annelida and mean Insecta biomass compared with out translocations from N → LP one week post translocation (Kruskal-Wallis Tests: $\chi^2 = 10.459$, $P = 0.001$; $\chi^2 = 9.163$, $P = 0.002$; $\chi^2 = 6.814$, $P = 0.009$ respectively) (Figures 33 D, E and F). No other significant differences were observed between Long Point background samples and out translocations from Ewe Island and Noonamena.

No significant differences were observed at all when comparing Noonamena background biomass against that of out translocations from both Ewe Island and Long Point (Figures 33 G, H and I).

Between out translocations, significant differences in biomass were observed only in Noonamena to Long Point outs for mean Annelida and overall mean biomass (all taxa combined) (Kruskal-Wallis Tests: $\chi^2 = 7.184$, $P = 0.028$ and $\chi^2 = 6.089$, $P = 0.048$ respectively). For all other out translocations to and from all other sites, no significant differences in biomass were observed over time (Figures 33 B, C, E, F, H and I).

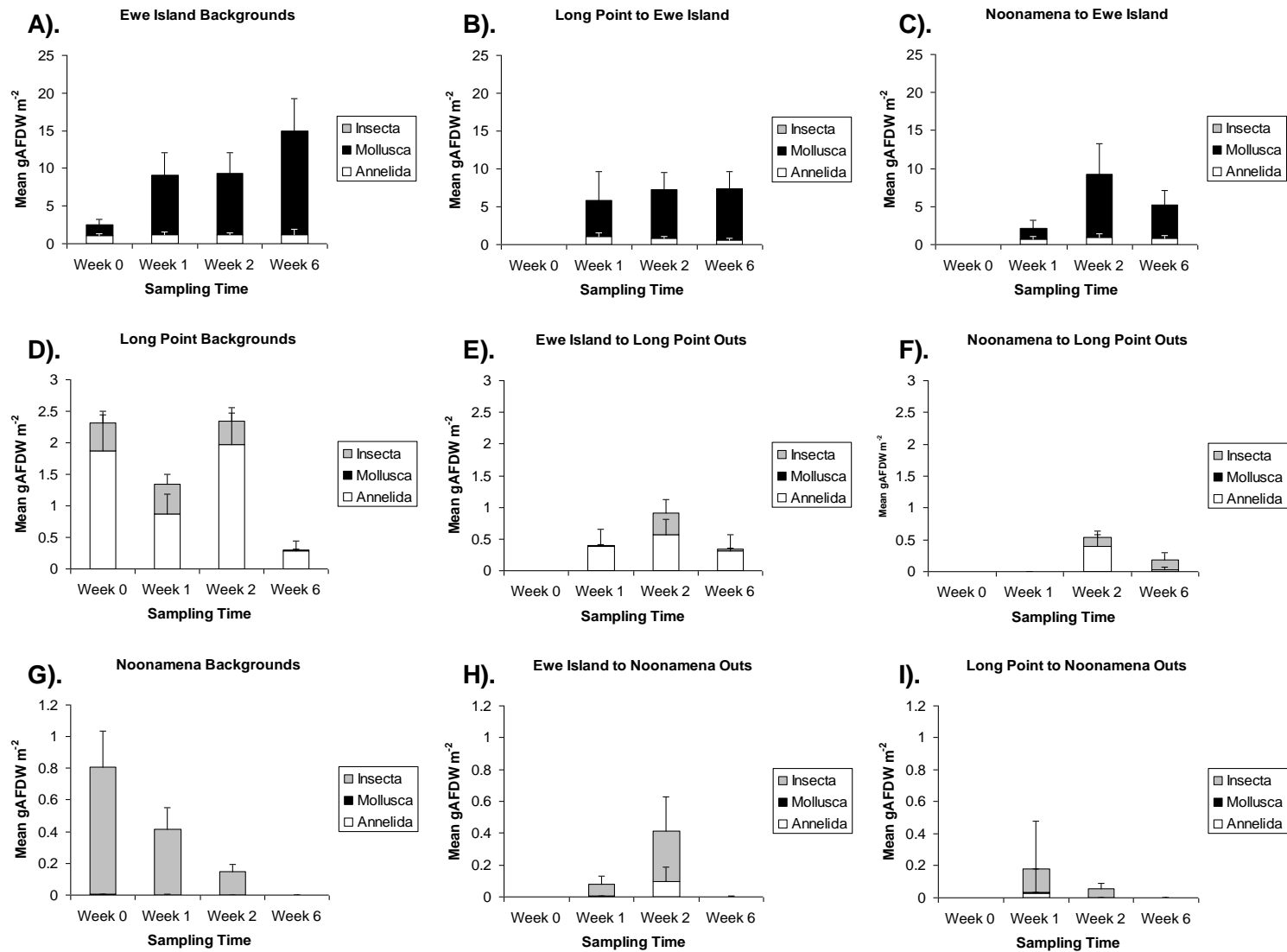


Figure 33. Biomass (mean g AFDW m⁻²) present in background samples from each site and all out translocations: A) Ewe Island background samples; B) Long Point to Ewe Island out translocations; C) Noonamena to Ewe Island out translocations; D) Long Point background samples; E) Ewe Island to Long Point out translocations; F) Noonamena to Long Point out translocations; G) Noonamena background samples; H) Ewe Island to Noonamena out translocations; I) Long Point to Noonamena out translocations.

3.4.3. Juvenile macrobenthic invertebrate response

Diversity and Abundance

The species diversity of juveniles found at each of the three sites (Table 12) was substantially lower than that of the adult invertebrate fauna (Table 9). Juvenile *Arthritica* were present at Ewe Island at each of the sampling times, whilst *Capitella* spp., were present only at 0, 1 and 6 weeks post translocation. Juvenile *Simplisetia*, *Australonereis* and *Phyllodoce* were rarer, only present at Ewe Island at one sampling time each. The juvenile fauna at Long Point was represented only by *Capitella* spp. and insect larvae, while insect larvae alone were present at Noonamena (Table 12). As for the Adult benthos, the juvenile benthos abundance at Ewe Island is dominated by *Arthritica*, followed by *Capitella* spp., with other polychaete species present in small numbers (Figure 34 A). At Long Point, *Capitella* spp. abundance decreased sharply after week 0, as did the insect larvae abundance at Noonamena (Figures 34 B and C).

At each site, there was no significant difference observed between the abundance of individuals (of any species, or of total individuals) present in background and control samples.

Self Translocations

At Ewe Island, there was no significant difference in the abundance of individuals between the self translocations. Significant differences in abundance between self translocations and Ewe Island background samples were only observed for the total abundance of individuals found (Kruskal-Wallis Test, $\chi^2 = 5.914$, $P = 0.015$). At Long Point, significant differences in abundance were observed between the self translocations for *Capitella*, spp., Insect larvae and the total number of individuals (Kruskal-Wallis Tests: $\chi^2 = 9.846$, $P = 0.002$; $\chi^2 = 14.756$, $P = 0.000$; $\chi^2 = 14.768$, $P = 0.000$ respectively). At Noonamena, only Insect larvae were present in self translocations. Significant differences in abundance were observed between self translocations and also between self translocations and background samples (Kruskal-Wallis Tests: $\chi^2 = 7.659$, $P = 0.006$; $\chi^2 = 4.882$, $P = 0.027$ respectively).

Out Translocations

For each site, ANOSIM analysis revealed no significant differences in juvenile community structure between out translocations to a site and the respective background samples.

For out translocations from both Long Point and Noonamena to Ewe Island, there was no significant difference in the abundance of individuals present in out translocations to different mudflat exposures (HE to HE, HE to LE, LE to HE, LE to LE). There were significantly greater total abundance of individuals (for all species combined) in Ewe Island background samples compared with out translocations from Noonamena to Ewe Island for all sampling times (Kruskal-Wallis Test, $\chi^2 = 4.095$, $P = 0.043$), yet there was no significant difference observed for any individual sampling time. No such differences were observed between Ewe Island background samples and out translocations from Long Point to Ewe Island.

Table 12. Distribution of juvenile benthic invertebrates from Background samples across three sites over four sampling periods (0, 1, 2 and 6 weeks).

Phyla	Species	Site											
		Ewe Island				Long Point				Noonamena			
		W0	W1	W2	W6	W0	W1	W2	W6	W0	W1	W2	W6
Annelida	<i>Capitella</i> spp.	+	+		+	+	+	+	+				
	<i>Phyllodoce novaehollandiae</i>		+										
	<i>Simplisetia aquisetis</i>			+									
	<i>Australonereis ehlersi</i>	+											
Mollusca	<i>Arthritica helmsi</i>	+	+	+	+								
Insecta	<i>Chironomid</i> sp. larvae					+	+	+	+	+	+	+	

For out translocations from Ewe Island to Long Point, there was a significantly greater abundance of *Capitella* spp., Insect larvae and total individuals (all species combined) in translocations to low exposures (Kruskal-Wallis Tests: $\chi^2 = 11.639$, $P = 0.009$; $\chi^2 = 12.047$, $P = 0.007$; $\chi^2 = 14.814$, $P = 0.002$ respectively). Yet there were no significant differences between Long Point background samples and out translocations from Ewe Island to Long Point over all sampling times.

Only insect larvae were recovered from out translocations from Noonamena to Long Point, and significantly greater abundance of insect larvae were observed in translocations to low exposure than high exposure (Kruskal-Wallis Test: $\chi^2 = 8.734$, $P = 0.033$). Significant differences in abundance between Long Point background samples and out translocations from Noonamena to Long Point were observed for *Capitella* spp., and total individuals (Kruskal-Wallis Tests: $\chi^2 = 7.391$, $P = 0.007$; $\chi^2 = 4.733$, $P = 0.03$ respectively), but not for insect larvae.

Only insect larvae were observed in any out translocations to Noonamena. For translocations from Ewe Island, there was no significant differences in abundance observed between translocations to different mudflat exposure, or between out translocations and Noonamena background samples. For translocations from Long Point to Noonamena, significantly more insect larvae were present in translocations to low exposure than high exposure (Kruskal-Wallis Test: $\chi^2 = 14.824$, $P = 0.002$). Significantly greater numbers of insect larvae were recovered from out translocations from Long Point to Noonamena than from Noonamena background samples over all sampling times (Kruskal-Wallis Test: $\chi^2 = 7.112$, $P = 0.008$), with such significant differences being observed after two weeks post translocation only (Kruskal-Wallis Test: $\chi^2 = 6.766$, $P = 0.009$). No insect larvae were recovered from either Long Point to Noonamena out translocations or Noonamena background samples six weeks post translocation.

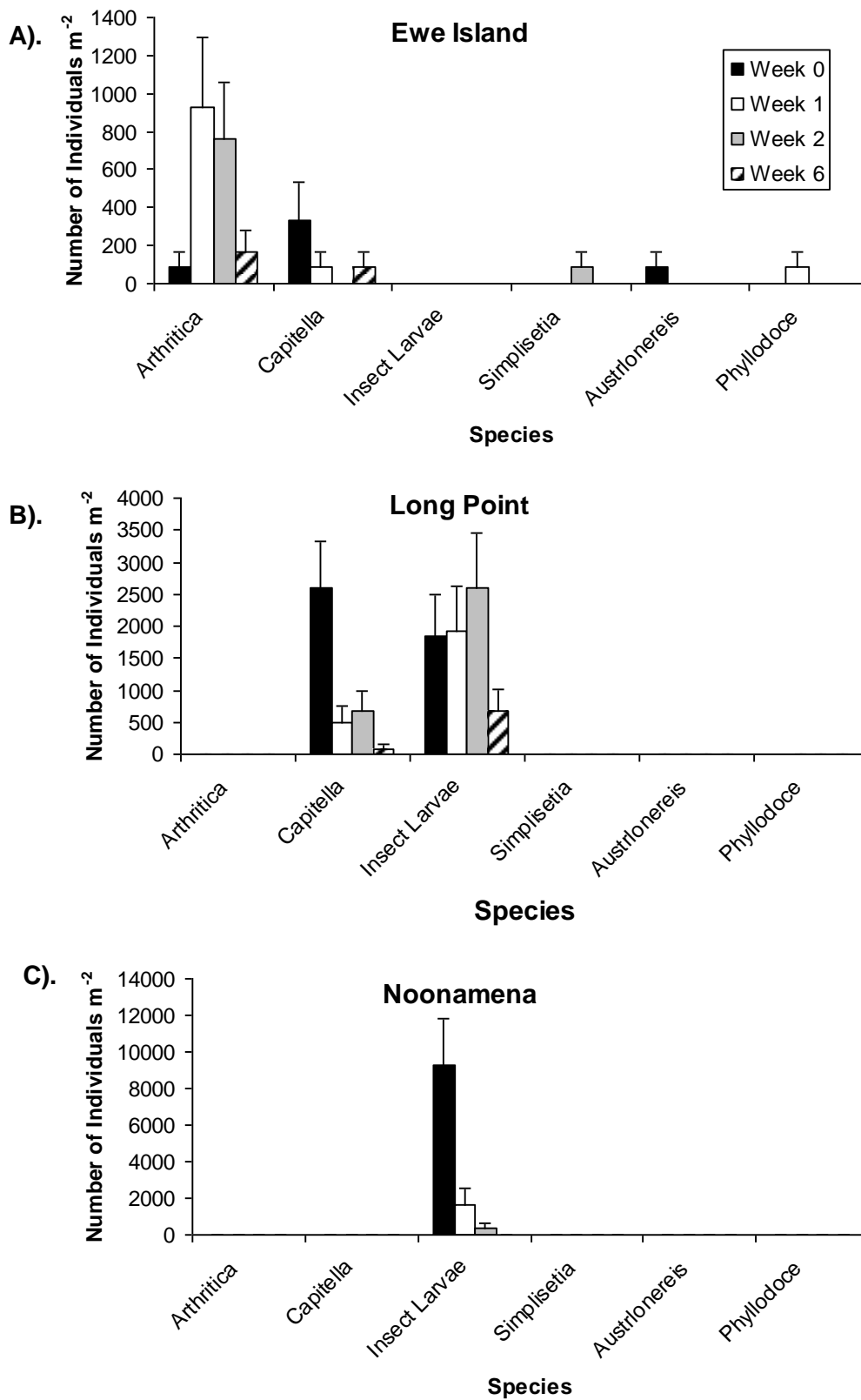


Figure 34. Distribution and abundance of juvenile benthic infauna in background samples at three sites over four sampling times: A) Ewe Island; B). Long Point; C) Noonamena. Note the different y-axis scales.

4. Discussions

4.1. The Macrobenthic Community

The macrobenthic invertebrate species found in this study have been recorded in previous surveys of the macrobenthic invertebrate communities of the Murray Mouth and Coorong (Geddes and Butler, 1984, Kangas and Geddes, 1984, Geddes, 1987, 2003, 2005a, b and c, Dittmann *et al.*, 2005, 2006a, b and c; Dittmann and Nelson, 2007; Dittmann *et al.*, 2008; Geddes and Francis, 2008). Like other estuaries that range widely in their morphological and physiochemical characteristics (Kanandjembo *et al.*, 2001; Currie and Small, 2006), the Coorong is dominated by relatively few taxa. By using macrobenthic invertebrates as indicators of general ecosystem health (Wilson, 1994) and by comparing species diversity, abundance and biomass with previous studies during times of increased freshwater flow (Dittmann *et al.*, 2006; Geddes, 2005b; Geddes and Francis, 2008), it is obvious that the Coorong lagoons are currently in a period of poor ecological health.

The current condition of the Coorong can be compared with other estuaries around the world. The Eastern Scheldt (Oosterschelde), one of the estuarine branches of the rivers Rhine, Meuse and Scheldt, consists of a series of water bodies. Large-scale coastal engineering projects were installed from 1957-1987 following floods in 1953 (Smaal and Nienhuis, 1992). A storm-surge barrier completed in 1986, along with a series of dams, separate the estuary from the newly created freshwater lakes, in a manner similar to the barrages of the Murray Mouth region and the subsequently freshwater Lakes Alexandrina and Albert. Such engineering works at the Eastern Scheldt have reduced the total area under tidal influence by 22% (Smaal and Nienhuis, 1992). The ecosystem has responded to such changes with shifts in habitats and species distributions, resulting in a reduced availability of foraging areas for wading birds. This has also been the case in the Coorong, where habitat quality for shorebirds has decreased over time (Rogers and Paton, 2009), especially as macrobenthic invertebrates, their main prey have been restricted to areas frequently inundated with water and within the salinity tolerance of these organisms.

The St. Lucia estuary, South Africa, has been subject to drought conditions for approximately five years, leading to hypersaline conditions developing in areas of greater distance from the estuary mouth, along with a loss of aquatic habitat (Pillay and Perissinotto, 2008). The authors conclude that the drought has structured benthic macrofauna communities primarily through its effects on water depth and habitat fragmentation. A similar conclusion can be drawn in the Coorong where low flow conditions have led to lower water levels and hypersalinity reducing the quality of habitat to such an extent that the macrobenthic invertebrate fauna are largely restricted to the very northern regions of the Coorong.

Drought conditions can produce two types of disturbances: Predictable, seasonal press disturbances and less predictable, protracted 'ramp' disturbances (Humphries and Baldwin, 2003). The effects of the latter are, however, likely to be 'stepped' as crossing geomorphological or hydrological thresholds will likely cause abrupt changes in biological community structure and ecosystem processes. In the Coorong, these step effects are likely to be most visible in the region where greatest environmental pressure acts on the benthic macroinvertebrates: the North and South Lagoons. In the North Lagoon, the sharp salinity gradient represents a physical barrier to many of the macroinvertebrate species of the Coorong. The pivotal point for the macrobenthos in the North Lagoon, in 2006 to 2007, appears to be approximately at site 8, Noonamena, where diversity and abundance significantly decreases in comparison to the Murray Mouth region. In the South Lagoon, salinity has already 'stepped over' the threshold of all macrobenthic species except for insect larvae, and indeed the region 'stepped into' a new

Key Species in the Murray Mouth and Coorong

A small number of species dominate the benthic macroinvertebrate community of the Murray Mouth and Coorong Lagoons, particularly the polychaete *Capitella* sp. and the micro-bivalve *Arthritica*. Kanandjembo *et al.* (2001) suggest that due to its continuous reproduction, rapid growth rate, short life cycle and its ability to tolerate a wide range of salinities [albeit it not a high range in comparison to other species in the Coorong: Dittmann *et al.*, 2006c], *Arthritica* is particularly well adapted to survive the wide range of environmental changes faced by estuarine organisms throughout the year.

Capitella, is a well known indicator of organically enriched sediments (e.g. Grizzle, 1984; Tsutsumi, 1990). The life history characteristics of *Capitella* – production of planktonic larvae with widespread dispersal ability, early maturation, and high potential for population growth among others – make it well suited to quick establishment in organic-rich environments (Tsutsumi, 1990). However, the significant decrease in *Capitella* abundance at sites 4, 6 and 7 from Dec 2006 through to Mar 2007 were not matched with similar decreases in sediment organic content nor significant colonisation/increases in abundance of other species that may facilitate a rapid decline in *Capitella* populations (Tsutsumi, 1990). A large drop in juvenile *Capitella* was observed at site 6 in March 2007, but did not precede the significant decrease in adult *Capitella* observed between Dec 2006 and Jan 2007. At site 7, very few *Capitella* juveniles were observed in Jan 2007 which may have contributed to the significant decrease in adults at that site in Mar 2007. No such differences in juvenile *Capitella* abundance were observed at site 4 to explain the significant decrease in adults at that site in March 2007. The lack of tidal influence at sites 6 and 7 results in significant portions of the mudflats at these sites being exposed as water levels drop through the summer months, thus becoming uninhabitable to *Capitella* and other macrobenthic invertebrate species. In the North Lagoon, both adult and juvenile *Capitella* disappeared from these exposed areas during the summer and this could explain the significant decrease in abundance at sites 6 and 7. The area of mudflat subject to high exposure levels at site 4 is still inundated with tides during the summer months, and although a small decrease in adult and juvenile *Capitella* was observed towards the shore, this cannot explain the significant decrease in adults at site 4 over time. Therefore an alternative mechanism that has not been measured in this study (for example predation/loss of food source) may be responsible for the decrease in *Capitella* adults at site 4.

The individual sizes of benthic invertebrates varied across sites and exposure levels, yet were species specific. The length of adult *Nephtys* increased from sites 1-6, while the length of both *Simplisetia* and *Arthritica* decreased over the same sites. Where significant differences in length were found between mudflat exposures, for all species, smaller individuals were generally found near the water. Benthic macroinvertebrates can exhibit complex recruitment strategies, but it has been observed that for both the polychaete *Arenicola marina* and the bivalve *Macoma balthica*, juveniles mature in refugia at higher intertidal levels than where most adults occur, from which they migrate to lower and more offshore areas (where adults thrive) during winter (Beukema, 1993; Flach and Beukema, 1994; Reise *et al.*, 2001). Whether such an adaptive strategy had been adopted by each of these Coorong species, or whether juveniles/small adults are restricted to areas of lower mudflat exposure because of predation pressure, competition, density-dependent effects, an inability to withstand greater exposure, or other environmental pressures, is unknown.

4.2. Benthic Macroinvertebrate Biomass

The biomass in the Murray Mouth and Coorong is dominated by polychaetes and bivalves, due largely to the low numbers and diversity of other taxa. This dominance is typical of temperate intertidal mudflats in comparison to crustacean-dominated tropical flats (Swennen *et al.*, 1982, Reise *et al.*, 1994). In comparing tropical Indonesian mudflats with temperate counterparts from England, Warwick and Ruswahyuni (1987) suggested that because phytoplankton primary production is nearly continuous in the tropics, less primary production reaches the benthos. The highly seasonal phytoplankton productivity of temperate regions may therefore result in greater productivity reaching the benthos, thus abundance and biomass can be greater in temperate regions. Although still temperate, the seasonality of the Coorong is less marked and extreme than other temperate regions, particularly in Northern Europe. This may explain the lack of marked biomass seasonality in the Coorong when compared with other temperate regions.

Sites 1 and 5 were the only sites with no significant difference in abundance (all species combined) through time over the summer survey, yet a significant increase in biomass was observed, indicating increased individual size of the benthic fauna. At further sites in the estuary (sites 2 and 4) there was significant decrease in abundance, but no significant change in biomass over time. At sites 6, 7, and 9-12, significant decreases were observed in both abundance and biomass. The biomass measured in this study was variable in comparison to previous studies (Dittmann *et al.*, 2008). The significant increase in biomass observed at site 1 in this study agrees with records from 2004-2007 (Dittmann *et al.*, 2008). Biomass at Mundoo Channel has decreased dramatically since 2005 (Dittmann *et al.*, 2008), but no change was observed over the three months in this study. At site 5, the biomass measured in Dec 2006 in this study is considerably lower than that measured at the same site in Nov 2006 by Dittmann *et al.* (2008), and although biomass increases in time in this study, biomass at site 5 in Jan and Mar 2007 was still less than in Nov 2006. Local fluctuations in productivity and recruitment may be reasons for such differences in time (Josefson and Hansen, 2004).

Sediment grain size is the single best predictor of total biomass for sedimentary shores and peak biomass values are found in temperate regions (Ricciardi and Bourget, 1999). The seasonably variable biomass observed in this study is in contrast to more stable measurements elsewhere (Reise *et al.*, 1994), but frequency of sampling may be important in identifying such variability (Kalejta and Hockey, 1991). The temporal variability of biomass observed in this study cannot be definitively linked to either organism growth or recruitment events as concluded in a study of a South African estuary (Kalejta and Hockey, 1991).

4.3. Environmental Factors Influencing Distribution and Abundance

Environmental factors have been shown to influence the heterogeneity of intertidal benthic assemblages at different spatial scales in temperate Australia (Hirst, 2004, Dye and Barros, 2005, Winberg *et al.*, 2007). The three major environmental factors affecting macrobenthic abundance and distribution in the Murray Mouth and Coorong are salinity, sedimentary characteristics and tidal exposure.

4.3.1. Salinity

Salinity gradients and salinity variation are perhaps the primary environmental driving forces determining macrobenthic invertebrate abundance and distribution in temperate estuaries (Sanders *et al.*, 1965; Ysebaert *et al.*, 1998; Attrill, 2002). In the Thames estuarine system, large

freshwater inputs accompanied by anthropogenic narrowing of the river have led to reduced salinity ranges in the system in comparison to other estuaries with different morphological characteristics (Attrill, 2002). In the Murray Mouth, the presence of barrages has significantly reduced freshwater flow into the traditionally estuarine part of the River Murray. Over the past 5 years, significant drought and subsequent lack of freshwater flow has led to the Murray Mouth region becoming more like a marine embayment than an estuary. Despite this change in morphology, no significant change in macrofaunal assemblages has been observed during that time (Figure 35). However the lack of freshwater input into the Coorong has led to a significant increase in salinity in the North and South Lagoons. As a result, macrobenthic invertebrates are restricted to areas within their salinity tolerance. The fluctuations of habitat availability and changes in species distribution apparent in the Coorong have also been observed in the Oosterschelde (Eastern Scheldt, southwest Netherlands) where storm barrier construction has led to a reduction in tidal influence, water exchange and a 64 % drop in freshwater input (Smaal and Nienhuis, 1992). In a similar manner to the Coorong, the carrying capacity of the Oosterschelde region for wading birds has decreased due to the reduced availability of foraging areas. Yet, it is perhaps the rate of change of salinity rather than salinity fluctuations that may be more important in determining macrobenthic community structure (Sanders *et al.*, 1965, Kanandjembo *et al.*, 2001). If significant freshwater flows return to the Murray Mouth and Coorong regions, management of water release over the barrages is important to lessen any potential negative effects on the macrobenthic invertebrate community of significant freshwater input into the Murray Mouth region, such as decreased abundance and diversity of macrobenthos sensitive to lower salinity levels or indeed rapid changes in salinity.

4.3.2. Sediment

Throughout each sampling time, site 5, Pelican Point, consistently differed in its sediment grain size to the other sites. No such differences were observed in sediment organic content or other environmental variables such as salinity and temperature between site 5 and the other sites. Of sites where macrobenthos were present, site 5 consistently had the greatest proportion of sediment with grain size > 125 μm (data not shown). Sediments with a high percentage of grain size >125 μm have been shown to have elevated benthic microalgal biomass (Cahoon *et al.*, 1999). This may assist in explaining the greater abundance of macrobenthic organisms observed at site 5, particularly for the deposit feeders *Capitella* spp., and *Simplisetia aequisetis*, and the plant detritus, bacterial and algal feeder *Arthritica helmsi*. Although organic content, microbial content, food supply and trophic interactions, as well as sediment grain size, have been proposed as causative factors for species distributions, no single mechanism has been shown to explain patterns of infaunal species across many different environments (Snelgrove and Butman, 1994). Indeed, significant sediment effects have been shown for colonisation by bivalves and gastropods, but not for polychaetes and amphipods, suggesting that many benthic species are able to colonise multiple sediment types, and that factors other than grain size and organic content play a significant role in determining colonisation of those species (Wu and Shin, 1997). These other factors may include turbidity, regional rainfall, freshwater inflow and chlorophyll a concentrations (Currie and Small, 2005), none of which were assessed in this current study.

4.3.3. Exposure

Low-flow periods that cause the drying of sediments due to reduced water levels ensure that populations of fresh water macroinvertebrates and other organisms rely on refugia and the production of a long-lived bank of propagules such as eggs, seeds and spores to survive the drought period (Humphries and Baldwin, 2003). Tidal inundation in the Coorong reaches only as far as approximately site 6 in the northern part of the North Lagoon. Water levels of the remaining lower reaches of the North Lagoon and the South Lagoon are dependent on precipitation, water release through the Upper South East Drainage System (USED) near Salt

Creek in the South Lagoon, atmospheric pressure driving water in through the Murray Mouth and down through the system, and wind-driven seiching. As a result in the summer months, when precipitation is low, atmospheric pressure is generally high, and wind-driven seiching is reduced, large areas of mudflat become exposed in both the North and South Lagoons. In the North lagoon at sites where macrobenthic organisms occur, the period of exposure can have adverse effects on the infaunal populations.

Total macroinvertebrate benthos decreased significantly at all high, medium and low mudflat exposures in the North Lagoon. Infauna are unable to survive in exposed Coorong sediments for less than one week and so the period of exposure is critical in order for communities occurring in the higher elevations of the mudflats to survive. Subsequent colonisation of mudflats during periods of higher water levels (e.g. winter months) following a period of long term exposure, could be determined a high-risk strategy given the potential for short-term changes in the water level. As to which species will be the first to colonise such newly available habitat in the Coorong is unknown. Limitations to larval settlement and/or juvenile survival may explain why macrobenthic species have been shown to use less than half of suitable sites (available habitat) available to them (Armonies and Reise, 2003). Pillay and Perissinotto (2008) state that the 5-year mouth closure of the St. Lucia estuary, South Africa, as prolonged drought conditions significantly reduced the total area of inhabitable substrate available to macrofauna. High salinities as a result of reduced precipitation and increased evaporation also had a significant effect on the macrofauna of the area. Habitat fragmentation due to low water levels likely impeded the spread and colonisation of species (due to restrictions in exchange of macrofaunal larvae: Pillay and Perissinotto, 2008). Large areas of unsuitable habitat are present in the Coorong, through both reduced waterlevels in the summer rendering previously suitable mudflat uninhabitable, and high salinities occurring throughout the North and South Lagoons that are beyond the tolerances of the majority of macrobenthic juveniles, restricting colonisation of suitable habitat.

4.4. Recolonisation, Settlement and Recruitment

Juveniles of four key species of the Coorong system, *Capitella*, *Arthritica*, *Simplisetia* and insect larvae (Chironomid larvae) were present in mudflats throughout the year, suggesting the potential for recolonisation of newly available habitat is possible year-round. Seasonal heterogeneity among intertidal benthic macroinvertebrate populations is important in regions that experience dramatic seasonal changes such as below-freezing temperatures or monsoonal rains (Winberg *et al.*, 2007). For example, juveniles of the polychaete *Arenicola marina* in the Wadden Sea over-winter in subtidal channels before colonising the upper tidal zone, above the range of adults (Reise *et al.*, 2001). Such seasonal heterogeneity has also been reported in a temperate Portuguese estuary (Silva *et al.*, 2006), but was not observed at three estuaries in temperate New South Wales, Australia (Winberg *et al.* 2007). The juvenile macrobenthos observed in the current study also did not show such seasonal heterogeneity in distribution, although the twice-monthly sampling regime employed may not have been sufficient to highlight greater seasonality of juvenile macrobenthic presence in the Coorong region.

The colonisation of suitable mudflat in the Coorong is likely to be influenced primarily by water level, mudflat exposure time and salinity and to a lesser extent by sediment organic content. Whether post-larvae and juveniles colonise from adjacent mudflats through either active dispersal or drift (as a result of both passive and active resuspension into the water column: Armonies, 1994; Olivier *et al.*, 1996), or whether recruitment to Coorong mudflats occurs from external areas through the Murray mouth, is unknown. The difficulty of separating individual early-life stages (i.e. post-larvae, juveniles and young adults) during sample sorting must be highlighted and is certainly a limitation of this study. However, there was significant correlation between the presence of adults and their respective juveniles at sites throughout the Coorong.

With tidal influence reaching only to the northern end of the North lagoon, colonisation of mudflats further south would only be possible through active dispersal or wind-driven water movements. Post larvae and juveniles of the bivalves *Abra alba* and *Macoma balthica*, for example, have been shown to drift in the water column as a result of migration and/or passive resuspension (Olivier *et al.*, 1996; Bouma *et al.*, 2001 and references therein), and drifting can affect benthic organisms at various life history stages (Butman, 1987). *Arthritica helmsi* would be susceptible to passive dispersal as a result of wave action and current velocity.

Environmental factors such as tidal level, sediment grain size and plant biomass are important with regard to juvenile settlement and distribution as factors such as these have greater effects on juveniles than adults (Burkovsky *et al.*, 1997). Larvae of the polychaete *Capitella* are able to actively select suitable substrate under both still water and flow conditions (Butman *et al.*, 1988). The passive dispersal and accumulation of larvae may occur at large spatial scales (tens of metres to tens of kilometres) in the same way that such processes apply to sediment transport and deposition. Active habitat selection may occur over smaller scales (centimetres to metres) within the broad deposition areas described above (Butman, 1987). In the study presented here, the three most common juvenile organisms, *Capitella* spp., *Arthritica helmsi* and *Simplisetia aequisetis* were present throughout the Murray Mouth and northern North lagoon regions. However, each species was most abundant at site 5, Pelican Point. Whether this is due to passive transport or active selection by the larvae is unknown. In the Murray Mouth region, the active dredging of the Murray Mouth itself since 2003 is likely to have changed the water flow regimes in this dynamic environment. This, in addition to the lack of freshwater flows over the barrages during the same time period, may have actively altered sediment grain size and organic content of the mud flats in this region and may well contribute to changes in settlement cues for larvae and juvenile benthic invertebrates.

No direct evidence of inhibition of recruitment through the presence of other species as shown by Hunt *et al.* (1987) was found in this study, although it is interesting to note that the only site (site 1, Goolwa Channel) where juveniles of the polychaete *Nephtys australiensis* were present was the site in the Murray Mouth region where both *Capitella* and *A. helmsi* juveniles were always in low abundance. Whether such differences in abundance are due to specific site selection by the three species or due to predation of individuals by adult and juvenile *N. australiensis* (Fauchald and Jumars, 1979; Wilson, 2000; King *et al.*, 2004) is unknown. Geddes and Francis (2008) found that *N. australiensis* had greater abundance and biomass at 0.5-1 m water depth than in shallower or deeper waters at site 5, Pelican Point. Such water depths were not assessed in this study and the absence of *N. australiensis* juveniles in shallow waters at all sites other than site 1 may be related to depth preferences of the juveniles.

Following the barrage release of 2003, Geddes (2005a) reported no substantial response in the distribution or abundance of benthic macroinvertebrates to this release. Small numbers of polychaete larvae (species not identified) were observed in the very northern reaches of the North Lagoon only. High proportions of *Arthritica* juveniles were recorded at Ewe Island and Pelican Point (sites 4 and 5 in this study) in November and December 2003, but the author concluded that such recruitment may be seasonal or may have been related to the outflow (Geddes, 2005a). As a result of this current study, in both Dec 2006 and Oct 2007, large numbers of *A. helmsi* juveniles were present at sites 4 and 5 and we can tentatively conclude that the recruitment observed by Geddes (2005a) may have been a seasonal occurrence.

Drought conditions have been shown to influence estuarine invertebrate populations (Attrill *et al.*, 1996; Attrill and Power, 2000, Salen-Picard and Arlhac, 2002; Gascon *et al.*, 2007, Pillay and Perissinotto, 2008), yet few studies have specifically assessed juvenile recruitment and response to such conditions. Although this study does not attempt to compare the current

drought period experienced in the Murray basin with periods of flood, it provides a significant quantitative study of juvenile macroinvertebrate abundance during a time of severe freshwater stress in the Murray estuary region, and shows that the key species in the system are continuing to reproduce and recruit. Invertebrate populations in aquatic systems are frequently able to recover rapidly from periods of drought, but the effects may be profound and long-lasting as a result of population bottlenecks produced following limited gene flow (Humphries and Baldwin, 2003). This may have significant implications for the macrobenthic community of the Murray Mouth and Coorong regions when drought conditions end.

4.5. Responses of Macroinvertebrates to Translocation

The results of the translocation of macroinvertebrates in the sediment translocation experiment agree fully with our initial hypotheses: Translocation of infauna

- 1). to higher salinities resulted in a decrease in abundance and diversity;
- 2). to lower salinities resulted in an increase in abundance and diversity;
- 3). to areas of increased mudflat exposure resulted in a decrease in abundance and diversity;
- 4). to areas of decreased mudflat exposure resulted in an increase in abundance and diversity.

The changes in species diversity and abundance as a result of translocating sediment between mudflat exposures and sites cannot be attributed to experimental artefacts as a result of the translocation process due to the similarity of respective background and control samples with out translocations to and from each site.

In an attempt to assess the impact of boat-generated waves on macrofauna, Bishop (2007) found that translocating sediment cores from disturbed to undisturbed mudflats led to an increase in capitellid and nereid abundance. The converse was true for translocations from undisturbed to disturbed mudflats. The author concluded that changes in infauna abundance were not necessarily due to changes in sediment grain size as is often suggested (Bishop, 2007 and references therein), but to differences in disturbance between sites. The results of the sediment translocation experiment presented here agree that sediment grain size is not the main factor influencing macrobenthic invertebrate diversity and abundance. Although the sediment grain size differed significantly between sites, when translocated to low salinity (Ewe Island), diversity and abundance increased, regardless of the sediment grain size translocated. This contradicts the importance of sediment grain size (Snelgrove *et al.*, 2001), where samples from sandy sites had significantly less macrobenthic invertebrate diversity than samples from a muddy site (of smaller grain size). Rather salinity and exposure level are the major factors affecting infaunal macrobenthos in the Coorong.

Generally, macrofauna translocated to high exposure areas at Long Point and Noonamena were unable to survive for one week without inundation. Hummel *et al.* (1988) found that 50 % mortality of benthic macroinvertebrates occurs within three days of emersion at greater than 19 °C, and that larger organisms and those with shells are more resistant to emersion than smaller species and those without shells. The effect of such emersion contrast strongly compared with the effects of extended periods of submersion: Hummel *et al.* (1986) found that 14 days of submersion had little effect on the macrobenthic invertebrate community of the Oosterschelde basin, the Netherlands. The temperatures experienced within exposed sediment in the Coorong during the summer months are likely to be greater than 19 °C and subsequently survival times of macrobenthic invertebrates may be even less than the three days reported by Hummel *et al.* (1988).

The North Lagoon undergoes significant variations in both water level and subsequent periods of mudflat inundation. The colonisation of high exposure mudflat by macrobenthic invertebrates is a high risk strategy given the potential for prolonged exposure as a result of falling water levels during summer months, particularly for less mobile species such as the bivalves. For all species, the speed of water retreat would be important in ensuring migration of benthos to areas of greater inundation: quickly falling water levels would likely result in the isolation of benthos. Death would be the probable consequence if submersion did not occur swiftly thereafter (Hummel *et al.*, 1988). Species found to occur in the tidal zone of an estuary are generally more resistant to emersion, but no such relationship was observed by Hummel *et al.* (1988). This agrees with both the results of the adult macrobenthos distributions across three mudflat exposures and the sediment translocation experiment presented here: Although nearly all species showed spatio-temporal variability in their distributions across three mudflat exposures of the Murray Mouth and North Lagoon, particularly during the summer months, no single species had a consistently greater abundance in higher mudflat exposure areas and no species were able to survive for a week without submersion.

The results of the sediment translocation experiment show that recovery of mudflats (in terms of increasing species diversity and abundance) is possible given suitable salinities and periods of inundation. Thus, with the continual presence of juvenile invertebrates in the Murray Mouth and northern North Lagoon regions, if salinities and water levels in the southern North Lagoon and South Lagoon regions are improved to within species tolerance limits, the recolonisation of these currently species-poor regions is possible.

5. Conclusions and Management Implications

Salinity, tidal exposure and to a lesser extent sediment organic content are the driving environmental parameters behind both macrobenthos and juvenile macrobenthos distribution. The present study agrees with Hirst (2004), Currie and Small (2006) and Hirst and Kilpatrick (2007) that these are the principal parameters structuring species diversity and community composition of estuarine macroinvertebrates.

Increased salinity and increased exposure both result in the loss of macrofaunal diversity and abundance. These outcomes are currently being observed in the Northern Lagoon under the present flow situation as salinity increases and water levels fall. If current conditions persist with no freshwater input, the distribution of macrobenthos will continue to recede northwards towards the Murray Mouth region to creating reduced refugia where conditions remain within the physiological adaptations of species.

Although the low macrobenthic diversity currently present in the Murray Mouth and Coorong Lagoons suggests that the region is in poor ecological health, the system appears to presently be in a state of relative stability when compared with macrobenthic surveys of previous years. Despite the lack of water released into the region over the past five years, macrobenthic communities have not changed significantly over that time.

Importantly, the presence of juvenile macrobenthos of key species in the Murray Mouth and northern North Lagoon regions throughout the year suggests that colonisation of new, suitable habitat is possible, given the right environmental conditions. In order for recolonisation of the South Lagoon to occur under improved and suitable environmental conditions, continued connectivity to the North Lagoon is vital to allow the dispersal of larvae, juveniles and adults into the region.

With current water levels in Lake Alexandrina at approximately -1.0 m AHD (DWLBC, unpublished data), and the likelihood of increased flows from the Murray River in the short-term being minimal, management solutions are required to restore water levels in the Lower Lakes to address agricultural, recreational, cultural and ecological issues. One possible option, to open the barrages and release seawater into the lake system, would lead to a significant change in the lakes' ecological community, including colonisation by marine macroinvertebrates, as the ecosystem states shift from freshwater to brackish/marine (dependant on the volume of marine water released). The presence of juvenile macroinvertebrates across the Murray Mouth Region throughout the year, as shown in this study, would increase the likelihood of swift colonisation and establishment of marine macroinvertebrates (particularly those tolerant of lower salinities during the initial release of marine waters) in suitable mudflat habitats of the previously freshwater system. Indeed, marine water seepage into the lakes through the barrages as a result of low lake water levels has already led to the colonisation of habitats in close proximity to the barrages by marine organisms such as the spionid polychaete *Boccardiella limnicola* and the serpulid polychaete *Ficopomatus enigmaticus*. The latter species is currently causing economic and ecological damage through its colonisation of barrages, jetties, boats and reeds, as well as being responsible for the drowning of many freshwater tortoises due to the weight of colonised worms on their shells.

5.1. Likely Macrobenthic Invertebrate Responses to Flow

Even in times of good freshwater flow, the presence of barrages prohibits the continual free exchange of fresh and marine waters that occurs in most estuaries. Naturally occurring patterns of freshwater inflow are important to maintain the structure and productivity in estuaries (Flannery *et al.*, 2002). Water is only released over the barrages when water levels in Lake Alexandrina reach a threshold of +0.85 m AHD. So even under good flow in the Murray River, the historically (pre-1940s) estuarine region may still be described as 'in drought' as the region continues as a marine system rather than an estuarine system due to the barrage flow restrictions. Yet, ideally, consistent large inflows of freshwater over the barrages would reduce salinity and raise water levels sufficiently to allow the macrobenthic recolonisation of suitable habitats in the South Lagoon and southern North Lagoon. Unfortunately, such consistent flows are at present unlikely.

Under current conditions, macrobenthic invertebrates (apart from insect larvae) are absent from the South Lagoon and the North Lagoon is a transition region between the uninhabitable South Lagoon and the more desirable conditions of the Murray Mouth region. If drought conditions persist then the distribution of macrobenthic invertebrates is likely to contract further towards the Murray Mouth region as salinities increase and water levels decrease in the southern North Lagoon and South Lagoon. With no freshwater flow, marine conditions can prevail in the Murray Mouth region as long as the Mouth is maintained open, and the loss of current macrobenthic fauna from the Murray Mouth region is unlikely.

Consistent significant freshwater input into the Murray Mouth region over the barrages will likely result in an increase in abundance and diversity of macrobenthic invertebrates species with lower salinity tolerance, and changes in macrobenthic community structure may occur closer to the sites of freshwater release. Individual salinity tolerances of macrobenthic species will determine their response to decreasing salinities in the Coorong Lagoons as a result of significant freshwater input. Distributions and abundances will change, likely leading to increased abundance and diversity in the North Lagoon. If the effects of freshwater releases reach the South Lagoon, the recovery of macrobenthic species with higher salinity tolerances such as *Capitella* is possible.

A rise in water levels throughout the system is likely to accompany significant freshwater inflows, inundating currently exposed mudflats and potentially creating habitats suitable for colonisation. The presence of juvenile macrobenthos in the Murray Mouth region throughout the year means that colonisation of new habitat could be fairly rapid. Such colonisation may take just a matter of weeks, but establishment of significant adult/reproductive populations or communities may take up to a year or more.

Factors other than water level and salinity that are also important in determining macroinvertebrate recruitment include sediment grain size and sediment organic content. Both of these are likely to be affected in the Murray Mouth region by additional flows over the barrages due to changes in flow velocity, nutrient input and sediment deposition rates, but the effects in the North and South Lagoon are likely to be minor.

If salinities in the South Lagoon can be reduced to below 80 g/L, the likelihood of colonisation of habitat by macroinvertebrates with relatively high salinity tolerances, such as the Capitellidae and Chironomidae, is increased. However, connectivity of the South Lagoon to the North Lagoon is vital in order for such colonisation to occur. The greater the reduction in salinity, the greater the diversity of colonists is likely to be. Changes in the distribution of key food sources

for birds and fish, such as the Capitellidae and Chironomidae, into the South Lagoon and southern North Lagoon regions, will likely have beneficial effects on the predators of the system.

Although the system is currently in poor ecological health with regard to macrobenthic invertebrate diversity, the refugia of the Murray Mouth and northern North Lagoon regions coupled with the recorded presence of juveniles there throughout the year suggests that recovery of the system as a whole is still possible, if substantial freshwater flows become available for a significant period of time. In the absence of these flows, it is vital that the current diversity and abundance of macrobenthic invertebrates present in the Murray Mouth and North Lagoon regions are preserved into the future, allowing for the potential recolonisation of the southern North Lagoon and South Lagoon once significant freshwater flows return.

The question of an ideal ecological status of the Murray Mouth and Coorong region remains important. Under current conditions, the region is under substantial ecological stress, yet new ecological states have become apparent that were not present in the system five years ago. If the ultimate aim is to return the system to the conditions that were present at the time of Ramsar listing in 1985 (i.e. estuarine, marine and hypersaline states, but at salinities lower than currently present), managers face a significant challenge to achieve this aim given the current flow conditions. Macrobenthic invertebrates are keystone species that are important prey items for both fish and birds. An improvement in the ecological health of the system that would allow the recovery of macrobenthic invertebrates into first the southern North Lagoon and ultimately into the South Lagoon would also encourage the re-establishment of fish and bird populations into those same regions.

The system is currently under severe freshwater stress, but with the refugia of the Murray Mouth and northern North Lagoon for macrobenthic invertebrates, where juveniles of these key species were present throughout the year, the potential for recolonisation of suitable habitats in the southern North Lagoon and South Lagoon exists. When flows are restored, the system has the potential to recover to the ecological states present for macrobenthic invertebrates at the time of Ramsar listing in 1985.

6. References

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