Effects of heavy metal contamination on burial rates of *Austrovenus stutchburyi*: Implications for sediment transport

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ABSTRACT

Urbanisation in coastal catchments has significantly increased not only the input of terrestrial sediment to the marine environment but also the input of contaminants. In Tamaki Estuary, Auckland, heavy metals have accumulated in the upper estuarine muddy sediments and metal contamination has been detected on downstream intertidal sandflats. Sub-lethal levels of heavy metal contamination may affect the growth and behaviour of benthic organisms, which in turn may influence key ecosystem processes and productivity. The aim of this study was to examine whether the burial rate of an ecologically important bivalve species (*Austrovenus stutchburyi*) differed between a contaminated and a lesser-contaminated site and whether burial rates were affected by density. A secondary aim was to determine whether the burial of *Austrovenus* affected sediment transport and consequently if this was affected by density. This study demonstrated no consistent difference in burial time between source populations (sites). This was explained by a lack of measured difference in the condition index and heavy metal tissue loading of *Austrovenus* used throughout this study. The present range of contamination measured in Tamaki Estuary, Auckland, did not have negative biological consequences on the key ecosystem engineer, *Austrovenus stutchburyi*. Contamination levels in Tamaki Estuary may not be high enough to cause major physiological or behaviour changes to infaunal organisms, such as *Austrovenus*. Sediment erodability was not significantly correlated with any measured environmental and biotic factors. *Austrovenus* density was the only predictor variable that could be used to explain any variation in sediment erodability. There was no significant density effects observed between the amounts of sediment eroded for densities > 150 ind. m^{-2}. There was a

significant difference between sediment void of *Austrovenus* (0 ind. m^{-2}; smooth, flat undisturbed sediment surface) and sediment containing *Austrovenus* (>150ind. m^{-2}; physical structure on/in the sediment surface, increase in bed roughness). These results indicate that there is little or no effect of *Austrovenus* on the critical erosion threshold, suggesting that in the absence or presence of *Austrovenus* the current required to erode 10 g m^{-2} of sediment would remain somewhere between 28.5 and 30.5 cm s^{-1}.

This study found that there was considerable variation in the burial rate of individuals and the greatest variation was recorded in the lowest density treatments (150 ind. m^{-2}), which corresponded to the same density that had the greatest variation in sediment erodability. Further investigations are needed to gain a better understanding into the important roles (the importance of the various feedbacks and limitations and interrelationships) that *Austrovenus* play in the soft-sediment ecosystem, as losses of this species are likely to have large-scale impacts on the wider soft-sediment communities and ecosystem functioning.

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Lastly but by no means the least, everyone in R2, for keeping my sole intact, caffeine levels high and making me do some kind of exercise to keep me from turning into a round blob, without you all I would have gone insane ☺ well .. . . . . insane sooner.

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Chapter 1

1. Introduction

1.1. Estuarine stressors

Estuarine and shallow water coastal environments are some of the most dynamic, complex and productive waters in the world (Chapman & Wang, 2001). Within these ecosystems there are a number of broad scale environmental factors or 'stressors' that have the potential to impact the life of resident fauna. Environmental stressors refer to any physical or chemical change in the natural environment that may impact the growth, reproduction, development or physiology of organisms. Stressors can have natural causes or can be the consequence of human activities such as urban development near coasts and estuaries and changes in land-use around catchment areas (e.g. agriculture, forestry, road building). For example, the development of urbanised areas around coastal waterways can result in significant changes to the quality of the recipient marine habitats (Dickinson et al., 1996).

Shoreline structures (e.g. seawalls, jetties, bridges, roads, train tracks, reclaimed land) can alter the tidal prism and current flows in coastal and estuarine areas (Kennish, 1992). These large-scale changes to water flow entering and circulation within estuaries can affect the nutrient balance of estuarine ecosystems (Kennish, 2002). Changes to water flow can alter the input of organic matter to the benthos, possibly causing a decline in ammonification rates, reducing available nitrogen to the benthic community (Herbert, 1999). Large quantities of nitrogen often enter the coastal environment from terrigenous derived runoff. In

shallow coastal environments the availability of nitrogen is often accepted as one of the major factors regulating primary production, suggesting that with increased nutrient loading there could be a subsequent increase in primary production in the pelagic, while the production in the benthos would be suppressed (Sloth et al., 1995).

Stressors affecting soft-sediment benthic environments can affect overall system functioning if strong benthic-pelagic links exist. The exchange of energy and materials between the benthos and the water column can be particularly pronounced in estuaries due to relatively shallow depths and vigorous tidal mixing. Benthic-pelagic coupling plays a major role in determining the production and biological structure in the pelagic environment within these aquatic systems thus, generating fuel for production of the many benthic species (Graf, 1992; Sommer, 1989; Sundback et al., 2003; Valiela, 1995). Terrestrial sediment depositions have been shown to cause large-scale habitat changes in estuarine ecosystems in New Zealand. Global sediment load to oceans in the mid 20th century was estimated to be 20,000 tons per year, of which approximately 30% was thought to have come from rivers of southern Asia (Milliman & Meade, 1983). Changes to the sedimentation rate entering marine environments can alter sediment food quality (Cummings et al., 2003) and sediment grain size, subsequently altering sediment porosity, stability and biogeochemical fluxes (Rhoads & Young, 1970). Such large-scale habitat changes include the smothering of fauna, decreased feeding efficiency of many filter-feeders, decreased burrowing ability of bivalves, and an overall decrease in estuarine biodiversity (Benedetti-Cecchi et al., 2001; Cummings & Thrush, 2004; Edgar & Barrett, 2002; Lohrer et al., 2004; Thrush et al., 2003; Norkko et al., 2006).

Sediment accumulation is the major cause of estuarine infilling and, although a natural process, has been shown to increase following human activities in the surrounding catchments (Hayward et al., 2004; Hume & Swales, 2003). It has been generally accepted that sediment grain size determines the erosional behaviour and transport of sandy sediment in moving water (Hjulstrom, 1939) and that the mechanical properties of benthic sediment and bed roughness both act to stabilise and destabilise estuarine sediment (Bale et al., 2006; Widdows et al., 2000a; Widdows et al., 2004). Notably, terrigenous derived sediments have the potential to cause catastrophic ecological damage (e.g. smothering) and are often associated with attached contaminants (Gray, 1997; Thrush et al., 2001, 2004 & 2008).

Increased urbanisation and industrial development adjacent to estuarine and coastal areas has lead to an overall increase in pollution. Run-off from streets (covered with oils and exhaust soot from leaded petrol) and other impervious surfaces (e.g. roofs, which leach zinc), as well as the direct dumping of industrial wastes into storm drains and sewers, has resulted in the accumulation of toxic pollutants in estuarine sediments adjacent to urbanised catchments. Anthropogenic activities have subsequently resulted in significant changes in the quality of marine habitats (Dunquesne et al., 2004). Changes in physical structures can cause chronic decline in the water quality entering estuaries due to industrial and urban sources. These waters often contain high levels of heavy metals such as Cadmium (Cd), Copper (Cu), Lead (Pb) and Zinc (Zn) which are commonly associated with past and present land use practices. Heavy metal concentrations in benthic sediment have been extensively studied in the Thames

Estuary (UK) and have been linked to past and present industrial and urbanised runoff (Attrill & Thomes, 1995). The majority of studies, to date, have focussed on European estuaries due to the high populations and long industrial histories. New Zealand is relatively young (settled ca. 800 years; Sutton et al., 2008) and comparatively unindustrialised until approximately 1945 when Auckland's light industries progressed rapidly.

Heavy metals such as Cu, Pb and Zn are commonly associated with New Zealand's past and present land-use practices and have been found to be elevated in several estuaries (Abrahim et al., 2007; Hoplee et al., 1980; Thrush et al., 2004). When heavy metals enter the marine environment they are likely to be transferred to the sediment phase by adsorption to suspended particulate matter (SPM) in the water column, followed quickly by sedimentation to the benthos (Hatji et al., 2002). In large estuaries there is often a natural gradient in sediment grain size and differing levels of anthropogenic inputs, creating an associated contamination gradient along the length of the estuary (Cortesäo & Vale, 1995). Muddier sediments are found in the upper reaches of estuarine environments and often contain higher levels of heavy metal contamination due to particle-particle interaction and their relatively high surface area (Mitchener & Torfs, 1996).

Heavy metals such as Cu, Pb and Zn have the ability to bioaccumulate in the food chain causing biotoxicity in sediment dwelling biota. Biotoxicity has adverse implications for estuarine biodiversity and marine ecosystems (Abrahim & Parker, 2002). Organisms living within and/or feeding directly or in-directly on estuarine sediment are likely to be effected by the accumulation of heavy metals. Once an organism accumulates higher than background levels of contamination it

will likely be adversely affected. Previous studies in New Zealand have shown the little neck clam, *Austrovenus stutchburyi,* to be negatively affected by the heavy metal accumulation of copper in its tissue (Peake et al, 2006). Exposure to high levels of contaminants has been shown to have adverse effects on some benthic organisms (Bowmer et al., 1993), both biologically, through growth and reproduction (Reidel et al., 1995; Timmermans et al., 1995), and through physiological behaviour (Dunquesne et al., 2004; Roast et al., 2002; Scokolowski, 1999). Animal behaviour is being increasingly used as a sensitive and integrated measure of exposure to toxic contaminants (e.g. Watts et al., 2001).

Laboratory studies have shown that the first response for bivalves exposed to sediment with artificially increased Cu concentrations (2–15 ppm) is prolonged valve closure and lower burrowing rates and an increase in mortality (Hummel et al., 1997). Other studies on the burrowing behaviour of bivalves have also concluded that when exposed to sediment containing pollutants (heavy metals, oils and phenol) burrowing rates are reduced (McGreer, 1979; Møhlenberg & Kiørboe, 1983). In addition, Phelps et al. (1985) demonstrated that relatively small amounts of copper (5µg ml^{-1}) adsorbed onto sediment caused an exponential delay in clam burrowing, supporting the previous findings that exposure to sediment contamination caused a delay in burrowing rates. However, little investigation has been done into the effects of chronically exposed bivalves and whether the burrowing of contaminated bivalves differs to that of 'healthy' or otherwise non-contaminated bivalves when exposed to a sedimentary pollution gradient.

Bivalves are often used as tools for environmental monitoring of marine pollution worldwide, (e.g. as bio-indicators of trace metals) (Rainbow, 1993 & 1995). Modification to their behaviour is likely to cause alterations to the services they provide for other organisms, having a knock-on effect through the entire ecosystem. Until recently, environmental monitoring consisted of basic tests of physical and chemical variables such as salinity, temperature, oxygen concentration, sediment grain size and heavy metals concentration (Lam & Gray, 2003). This basic approach to environmental monitoring provided limited information on levels of certain variables (e.g. pollutants) but failed to include information on the effect that contamination was having on sediment dwelling biota. The move towards developing more sensitive approaches to environmental monitoring has led to the focus being placed on the effects of sediment bound contamination rather than the levels present in that environment. By monitoring the sediment transport and the various feedbacks between resident bioturbators, that can both affect and be affected by contaminants, we start to get an understanding of the potential implications that could be associated with soft-sediment ecosystems.

1.2. Biodiversity and functioning

The diversity of organisms living in Earth's ecosystems has intrinsic value. However, the importance of biodiversity extends further because of the roles organisms play in modifying processes that are vital to sustain life. Numerous studies have now shown links between biodiversity and ecosystem functioning (Hooper et al., 2005; Kinzig et al., 2002), including experimental investigations performed in soft-sediment ecosystems (Bolam & Whomersley,

2005; Lohrer et al., 2004; Raffaelli et al., 2003; Solan et al., 2004). Changes to processes such as bioturbation by sediment-dwelling organisms (bivalves, worms, crustaceans, echinoderms) that rework sediment while burrowing and feeding may have broad implications for benthic soft-sediment systems (Ciutat et al., 2007; Widdows et al., 1998a; Williamson & Ockenden, 1996). The amount of sediment mixing due to bioturbation can be linked to the density and activity of resident burrowers (Cadee, 1979; Thayer, 1993). Reworking activities such as bioturbation disrupts sediment properties increasing sediment porosity and subsequently altering the transfer efficiency across the sediment-water interface, for both particulate and solute transfer (benthic-pelagic exchange) (e.g. Berkenbusch & Rowden, 1999; Botto & Iribarne, 2000; Boudreau, 1986 & 1994; Lohrer et al., 2004). Bioturbation can also alter the biogeochemistry of the sediment through mixing of the sediment column, subsequently resulting in incorporation/subduction of organic matter, oxygen and ammonium (including other nutrients in both particulate and solute form) into the deeper anoxic sediment (Graf, 1992; Mermillod-Blondin & Rosenberg, 2006). Structures such as burrows and mounds created by bioturbation can also influence microtopography as they last for longer periods of time and affect macrofaunal communities both directly and indirectly (e.g. Tamaki & Ingole, 1993; Thrush, 1986; Wolfrath, 1992).

Biota has the ability to stabilise sediment through physical protection of the surface sediment (e.g. tube worm mats or mussel beds etc) or through sediment binding (e.g. macroalgal mats, microphytobenthos, mussel beds and seagrass). However, sediment dwelling biota can also act to destabilise sediment through bioturbation as they may have the ability to move within the sediment

matrix (Widdows et al., 2000a; Widdows & Brinsley, 2002) disturbing its integrity. Sediment transport is an important issue of many disciplines and sediment type is often seen as one of the most dominant factors controlling the benthic community composition (Hall, 1994). Sediment transport alters benthic recruitment (e.g. Rhoads & Young, 1970), post larval dispersal (e.g. Commito et al., 1995; Emerson & Grant, 1991), and benthic secondary production (Emerson, 1989) as well as regulating fluxes (particulate and solute) between the benthic and pelagic environments (Graf, 1992; Lohrer et al., 2004).

All species do not contribute to ecosystem functioning at the same rate thus, functioning within will vary according to species composition. There may only be a few key species that will significantly modify nutrient regeneration (namely referred to as ecosystem engineers). Ecosystem engineers have the ability to directly and indirectly alter the availability of resources to other species (Jones et al., 1994). These species have a strong effect on ecosystem processes in maintaining, modifying or creating new habitats. For example, ecosystem engineers have been shown to modify the nearbed hydrodynamics and sediment dynamics of an ecosystem (Rhoads & Young, 1970; Widdows & Brinsley, 2002). Depending on how an organism contributes to ecosystem functioning will determine what effect the loss of that particular species will have on the remaining ecosystem. A key species is described as one that has a disproportionate effect on a given process; and the loss of such species could have a severe impact on the remaining community and sediment process rates. Alteration to biodiversity can thus have implications on sediment transport within an ecosystem. Changes in the abundance of species therein could change sediment resuspension and thus sediment transport.

1.3. Suspension feeding bivalves

Suspension feeding bivalves are commonly found in estuaries and harbours in New Zealand (Morton & Miller, 1973) and play an important role in the functioning of estuarine ecosystems. For example, they have the ability to mediate indirect or direct effects on surrounding macrofauna as they often protrude out of the seafloor and can alter the hydrodynamics affecting boundary flows, sedimentation and resuspension (Green et al., 1998; Nikora et al., 2002). Many suspension feeding bivalves also have the ability to move up, down and laterally (cm >1 m) through the surface sediment (Hewitt et al., 2006; Mouritsen, 2004), disrupting the sediment matrix, increasing erodability and affecting sediment porosity (Alexander et al., 1993; Ciutat et al., 2007; Widdows et al., 2000a).

At high densities, suspension feeding bivalves have the ability to reduce plankton in the benthic boundary layer, providing a possible 'eutrophication control' (Jie et al., 2001). SPM removed from the water column is either rejected or digested as faeces or pseudofaeces (collectively called biodeposits) (Bayne et al., 1993). The production of pseudofaeces is an important pre-ingestive mechanism that facilitates the particle selection process, whereby less nutritious particles are rejected and only quality particles ingested (MacDonald & Ward, 1994). However, it is noted that the majority of biodeposits are eroded and resuspended by tidal currents (Widdows et al., 1998a), while small portions of biodeposits accumulating on sandy sediment may alter sediment properties such as increasing the organic matter to the benthos (Zhou et al., 2006). Suspension feeding bivalves therefore play an important role in influencing the

flux of material across the sediment-water interface and its subsequent utilisation by other benthic deposit feeders (Jie et al., 2001). Furthermore, as most of the shellfish consumed by people in New Zealand are suspension feeders, they play a key role in public perception of estuarine and coastal environment health.

1.4. Study organism

The common New Zealand cockle, *Austrovenus stutchburyi* (hereafter referred to as *Austrovenus*), is a culturally important bivalve species and one which is commercially and recreationally harvested. This species can be extremely abundant on intertidal estuarine flats (up to 1200 individuals m^{-2}; Hewitt et al., 1996; Pridmore et al., 1990), and is widespread within estuaries throughout the entire country. *Austrovenus* occur in pristine places as well as in estuaries where anthropogenic inputs and contaminated urban runoff are prevalent (Cassie & Michael, 1968; Morrisey et al., 2003; Morton & Miller, 1973; Pridmore et al., 1990; Turner et al., 1995). *Austrovenus* live within the top 5 cm of the surface sediment and feed at the sediment-water interface where it is potentially exposed to contaminants from both the sediment and the water. *Austrovenus* is known to be an ecosystem engineer; therefore declines of *Austrovenus* due to overharvesting, sedimentation events or chronic contaminant loading could have wide ranging consequences. Changes to *Austrovenus* densities and the complex ecological feedbacks they mediate may cause a cascading effect through the entire estuarine environment, as the key ecosystem services that are provided by *Austrovenus* may be compromised.

1.5. Thesis aims

An estuary with a known contamination gradient was chosen for this study and two study sites from the extremes were selected whereby *Austrovenus* were collected from a contaminated and a lesser-contaminanted site. The first aim of this research was to examine the possible effects that localised heavy metal contamination had on a key ecosystem engineer (*Austrovenus stutchburyi*). Specifically to, (1) examine the effect that *Austrovenus* collected from a contaminated and lesser-contaminated site had on burial rate, and (2) whether burial time was affected by bivalve density, as increased *Austrovenus* densities could restrict lateral movement and subsequently alter burial times. It is anticipated that the burial time of *Austrovenus* could be used as an indicator of organism health and thus used as a broader 'tool' for ecosystem health. It is expected that heavy metal contamination could be altering the behaviour (burial) and physiology (CI and body burden) of this key ecosystem engineer, which could have a cascading effect throughout the ecosystem. A secondary aim was to determine whether burial of *Austrovenus* affected sediment transport (sediment erodability) and consequently, was sediment erodability affected by *Austrovenus* density, as increased *Austrovenus* density could either exasperate the potential hazardous fate of sediment bound contaminants or provide a barrier (armouring) between the benthic and the pelagic.

Chapter 2

2. Materials and Methods

2.1. Experimental design

Experiments were carried out between June and August 2008 to analyse the effects of cockle source population and density on burial rates and the stability of muddy-sand sediments. *Austrovenus* were collected from two populations in Tamaki Estuary, one from a site contaminated with heavy metals, the other from a site with lesser contamination. *Austrovenus* from these two populations were then introduced to a third type of sediment in a standardised set of experimental trials in annular flumes. The experimental sediment used to fill the annular flumes was clean muddy-sand collected from the Whitford Embayment (SE Auckland, New Zealand). This sediment was known to be uncontaminated and suitable to *Austrovenus*. *Austrovenus* were collected from Tamaki Estuary and planted onto the surface of Whitford sediments inside the flumes at a regular spacing interval. Once all *Austrovenus* had been planted onto the sediment surface, the flumes were carefully filled with artificial seawater. This initiated burrowing by *Austrovenus*, and *Austrovenus* were allowed to bury over the next 14 h whilst being recorded by time lapse cameras. Following burial, current speeds in the annular flumes were increased in stepwise increments from 5 to 45 cms^{-1} to measure sediment stability and erosion rates.

Table 1: Collection dates indicating the randomised density treatments, associated site (*Austrovenus* collected) and flume allocation.

Date	Density	Site	Flume
14/06/2008	0	W	1
14/06/2008	0	W	2
17/06/2008	150	G	2
17/06/2008	1200	T	1
19/06/2008	600	G	1
19/06/2008	600	T	2
01/07/2008	150	T	1
01/07/2008	1200	G	2
26/07/2008	150	G	1
26/07/2008	300	T	2
29/07/2008	150	T	2
29/07/2008	300	G	1
30/07/2008	600	T	1
30/07/2008	600	G	2
05/08/2008	1200	T	2
05/08/2008	1200	G	1
06/08/2008	300	T	1
06/08/2008	300	G	2
07/08/2008	0	W	1
07/08/2008	0	W	2

Five cockle densities (0, 150, 300, 600 and 1200 ind. m^{-2}) were used in experimental treatments spanning the range of natural densities (Hewitt et al., 1996; Morton & Miller, 1973; Pridmore et al., 1990). Each experimental run lasted two days. On day one *Austrovenus* and sediment were collected, placed in annular flumes and burial times determined. On day two sediment erosion experiments were conducted. Collection and running of experiments were restricted by the availability of two identical annular flumes. Restrictions meant that the time needed for all experimental procedures to be carried out was increased as *Austrovenus* and sediment collections had to be scheduled around tides. To avoid any possible temporal effects over the sampling period density

treatments were randomised and both populations were sampled on any given run day (Table 1). All density treatments were replicated throughout the experimental time frame, resulting in two replicates from each site for all 5 densities.

2.2. Austrovenus and sediment collection sites

Tamaki Estuary is a large, semi-enclosed water body located on the eastern side of Auckland, New Zealand (Figure 1). It consists of a long and relatively narrow channel (ca. 17 km in length) surrounded by an urbanised and industrialised catchment (approximately 11,500 ha) with approximately 600 industrial premises (Hayward et al., 2004; Hume & Swales, 2003). The area of the estuary is approximately 1,600 ha and is made up of low-gradient intertidal flats, composed mainly of fine muddy sands. *Austrovenus* were collected from two sites known to differ in sediment heavy metal concentrations (Abrahim & Parker, 2002; ARC, 2005& 2007). The upper streams and tidal creeks consist of muddy sediment and heavy metal contamination has been detected on lower intertidal flats. Monitoring by the Auckland Regional Council (ARC) has identified a sediment contamination gradient in Tamaki Estuary, with highest concentrations in the upper reaches of the estuary decreasing with distance to the sea (ARC, 2005& 2007). One study site was located in the upper reaches of the Tamaki Estuary; Tiraumea Reserve (E 2676332 N 6473790; hereafter referred to as TIR) which is characterised by zinc concentrations of more than 100 ppm (Figure 1). The other study site was located at the mouth of the Tamkai Estuary; Glendowie (E 2678368 N 6480295; hereafter referred to as GLE) and is at the other end of the contamination gradient having low levels of sediment heavy metal contamination, characterised by zinc levels of approximately 70 ppm.

Austrovenus were excavated by finger ploughing or gently picking individuals from sediment by hand (James & Fairweather, 1996) from the mid-low shore at both sites during low tide and stored in dark plastic coolers for transportation back to the laboratory. Previous studies have observed considerable variation in trace metal concentration within bivalve tissues and between individuals of the same species (Davies & Simkiss, 1996; Rainbow, 1997; Swaileh & Adelung, 1994). Furthermore, there is also a link between the body size and trace metal concentrations in invertebrates (Boyden, 1977; Moore et al., 1991). This has previously been compensated by collecting samples of a known length range or collecting individuals of a standard length (Soto et al., 1995; Swaileh, 1996). Therefore collected *Austrovenus* for all experiments were made up of individuals within a limited size range that was similar for both sites; Tiraumea 27.91 ± 0.26 mm (mean ± 1 SD, n = 48); Glendowie 26.95 ± 0.30 mm (mean ± 1 SD, n = 48). Cockle condition was determined for 6 individuals from each experimental run. A condition index (CI) used for bivalves was adopted here, CI = (soft tissues dry weight (g) x 100 / shell dry weight) (g) (Mann & Glomb, 1978). A sub-sample of *Austrovenus* (n=10) from each experimental run were also collected to analysis for tissue heavy metals (Cu, Zn & Pb) see ***Section 2.6*** for further details. A sample size of 10 was chosen for these measurements as it had been previously found to be a sufficient size to reduce any sample variation (Ibrahim, 1994; Richardson et al., 1994)

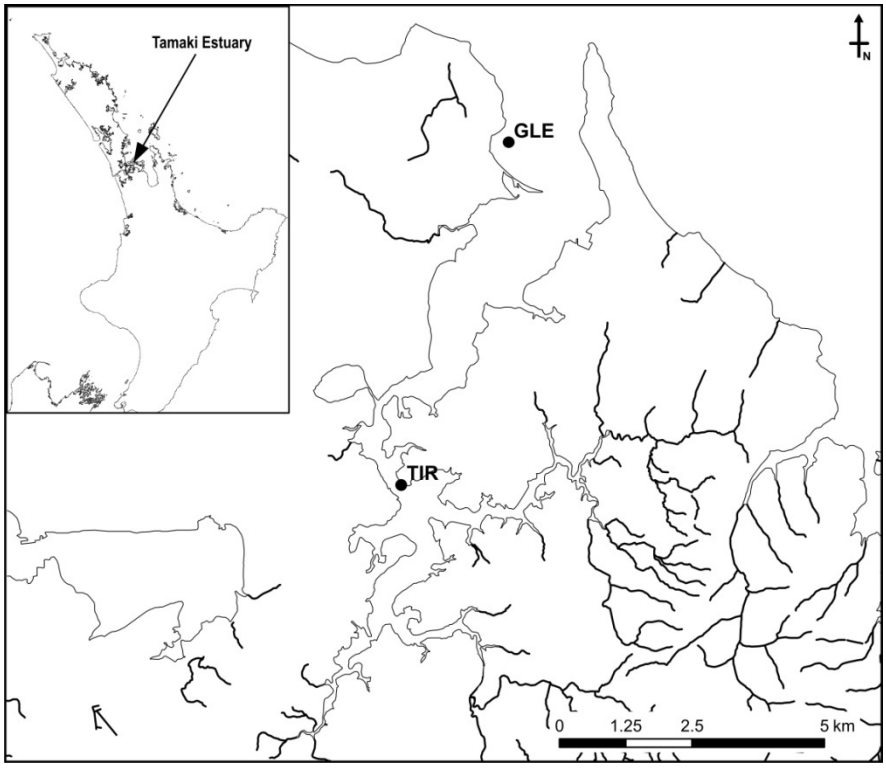

Figure 1: Location of *Austrovenus* collection sites in the Tamaki Estuary. TIR located in the upper reaches of the estuary has higher heavy metal concentrations, GLE at the mouth of the estuary has lower levels of contamination.

A separate estuary (Figure 2, Whitford embayment, SE Auckland) was chosen for the collection of undisturbed, non-contaminated, muddy-sand sediment. Undisturbed sediment can be defined as intact sediment, collected as is, whereby it has not been sieved or modified in any way. Undisturbed, non-contaminated, muddy-sand sediment was used to represent sediment that *Austrovenus* would naturally bury into. A study site, 20 x 20 m (400 m^2), of predominantly cockle-free, ripple-free, homogeneous sandflat was marked out in April 2008, where sediment and community characteristics were well known from past studies (ARC, 2005). The study site was then divided into 20, 2 x 2 m sample plots. Plots were selected randomly for each run date and no plot within 5 m of a previously sampled plot was sampled for at least one week, to allow sediment to recover from sampling induced disturbances. On each sampling date sediment for the two annular flumes was collected using ten sectional box cores

with the same dimensions as the flume channel (Widdows et al., 1998b). The sectional box cores were placed on the sediment surface (avoiding any surface structures or debris) and gently pushed into the sediment to a depth of 5 cm, where they were then gently dug out and base plates inserted for removal and transportation (Figure 3). This method of coring ensures that surface features such as bed ripples, benthic macrofauna and microphytobenthos are retained and not disturbed.

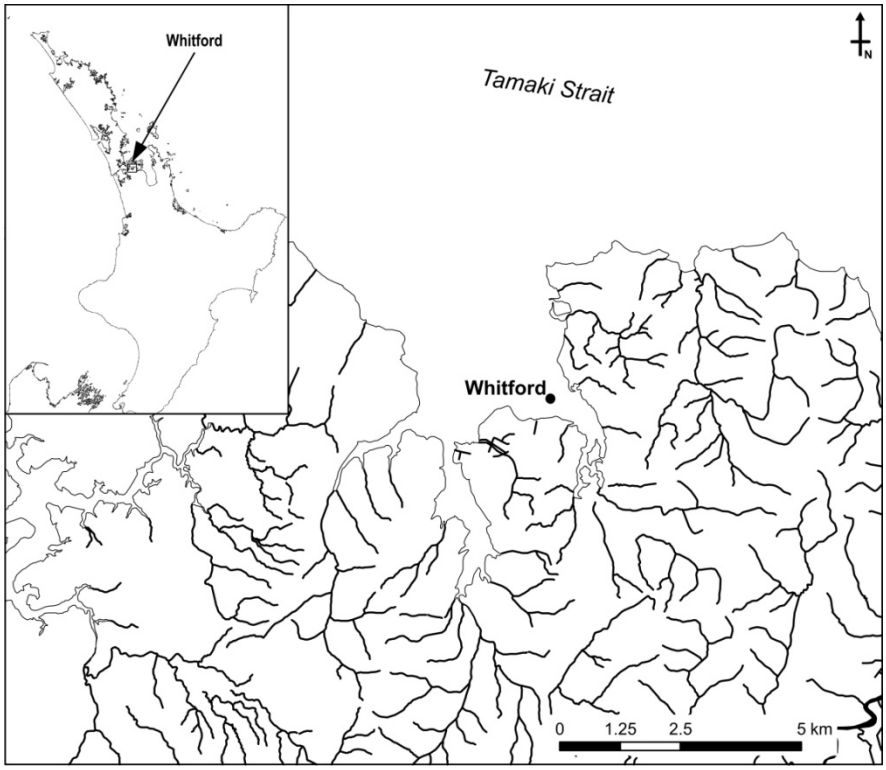

Figure 2: Location of sediment collection site (Whitford) situated within the Whitford Embayment, south-eastern Auckland. Tamaki Strait is indicated on the site map, demonstrating that this site is in the same harbour as the *Austrovenus* collection sites however located further from the city centre.

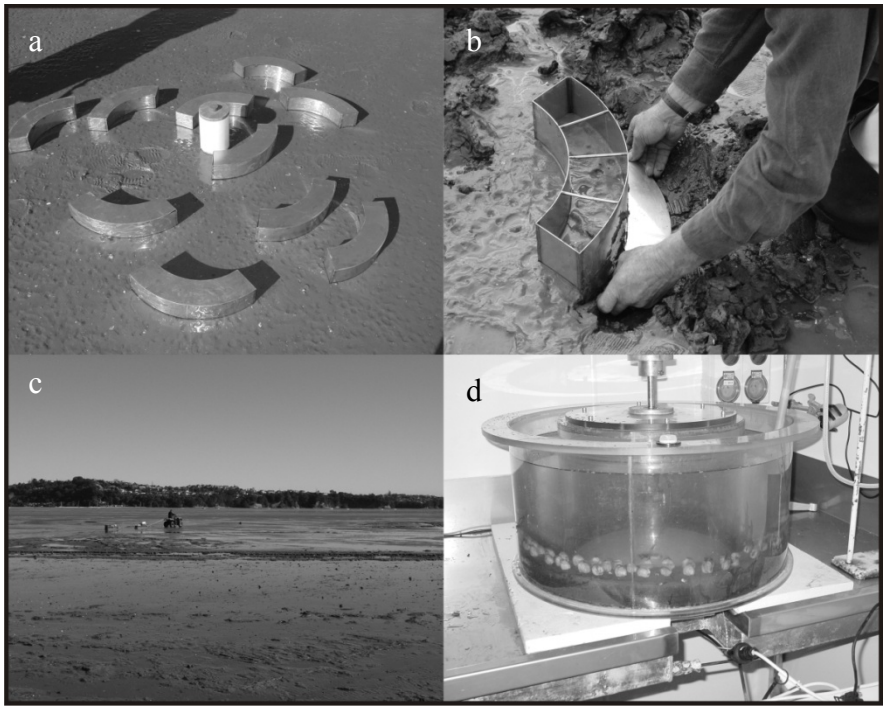

Figure 3: (a) Placement of the 10 sectional box cores on undisturbed sediment, with macrofaunal core (dia. 13 cm, 10 cm depth), (b) insertion of base plate for removal and (c) transportation before being inserted (d) into the annular flume and *Austrovenus* added.

2.3. Sediment properties

During each experimental collection, sediment was collected to analyse for any changes in the sediment properties over time. Further, to assess ambient cockle and macrofaunal densities at the sediment collection sites, one macrofaunal core (dia. 13 cm, 10 cm depth) was collected from each experimental plot sampled on the Whitford sandflat. Nine syringe cores (dia. 2.5 cm, 0-1 cm depth) were collected for analysis of grain size, percent organic content (OC) and porosity. Cores were randomly placed within field plots and three cores pooled into one pre-weighed sample container resulting in three replicate samples. The same method was used to collect sediment for chlorophyll *a* and phaeopigment analysis except only the surface 0-0.2 cm section was retained (Arar & Collins, 1997; Welschmeyer, 1994). All samples were stored in dark coolers with ice for transportation back to the laboratory for immediate freezing (sediment) and

processing (macrofaunal core). Macrofaunal cores were sieved on 500 μm mesh, stored in 70% Isopropyl alcohol and then stained with 1% Rosebengal before they were sorted and identified.

2.4. Annular flume

2.4.1. Description

Preliminary field observations suggested that *Austrovenus* removed from sediment would not bury into drained sediment, so experiments were conducted in annular flumes whereby *Austrovenus* were covered with sea water. Burial times and erosion experiments were determined using annular flumes as described in detail by Widdows et al. (1998b). In brief, the annular flume is a smaller, modified version of the annulus described by Fukada and Lick (1980). The flume was constructed using acrylic material producing a 62.4 cm outer and 42.0 cm inner diameter walls resulting in an annular channel of 10 cm with a bed area of 0.17 m^{-2}. Water flow is generated by a rotating drive plate (lid) that is driven by a motor and a gear box. The lid is positioned so that it sits 6 cm into the water column, which is filled to a depth of 19 cm; 13 cm above the sediment surface (Figure 4). The flume is controlled by a computer that can be used to programme stepwise increases in flow velocity (5 cm s^{-1} increments every 15 minutes; 5-45 cm^{-1}). During flume runs the concentration of suspended particulate matter (SPM) is measured by an optical backscatter sensor (OBS) that logs every 17 s, which is calibrated with gravimetric analysis of water samples taken during each stepwise interval to give the SPM (g l^{-1}). Prior to the start of the experiments the relationship between flow velocity (u) 5 cm above the bed and drive plate rpm was determined using a Sontec Acoustic Doppler Velocimeter

(ADV). For flume 1 *u* (cm s^{-1}) = 0.72 × rpm (r^2= 0.99) and for flume 2 *u* (cm s^{-1}) = 0.75 × rpm (r^2= 0.99) for flows between 0-50 cm s^{-1}.

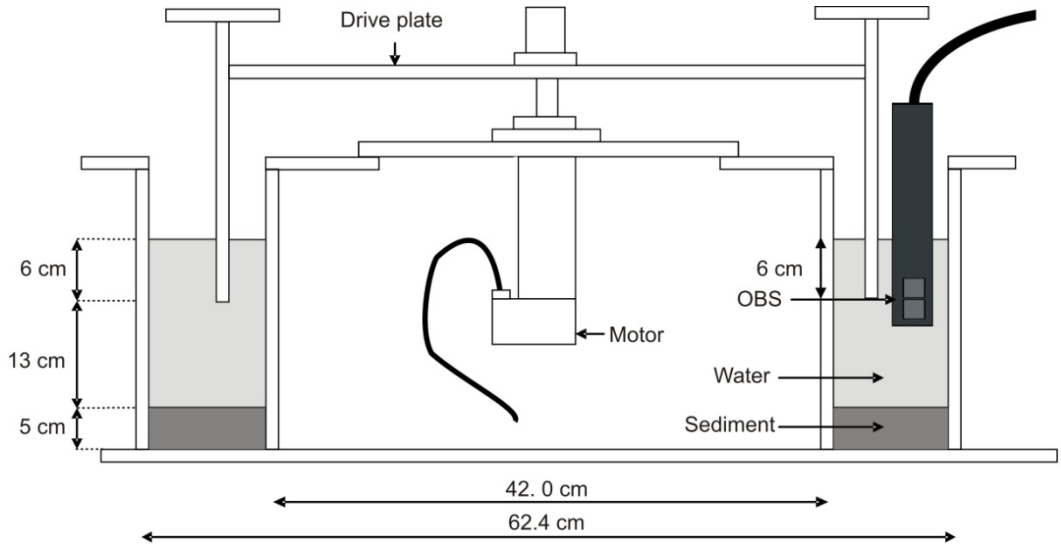

Figure 4: Schematic diagram of the annular flume system and components. The annular flume has a 42.0 cm inner diameter and a 62.4 cm outer diameter annulus (note that this diagram does not illustrate the 0.4 cm annular acrylic sheet that fills the gap between the inner flume wall and the sediment cores; see text for details). The sediment forms a continuous bed to a depth of 5 cm. The flume is filled with artificial sea water to a depth of 19 cm, and a drive plate sits 6 cm into the water column, generating water flow.

2.4.2. Set up

On arrival at the laboratory, four sectional annular box cores were carefully placed into the flume, while the outer box cores are removed leaving the intact sediment and base plates in the flume. The sediment was then gently pushed together and towards the outer wall. Due to the thickness of the corers a gap between the first and fourth core was created, as well as a gap between the sediment and the inner flume wall. The gap between the first and fourth core was filled with a slice of sediment from another core while the gap between the inner flume wall and the sediment was filled by placing a 0.4 cm thick annular acrylic sheet. This changes the dimensions of the annulus and the inner channel has a new width of 9.6 cm, creating a new bed area of 0.16 m^{-2}. *Austrovenus* were planted onto the sediment surface by gently pushing their posterior end just into

the sediment so that they remain in a vertical position and orientated for burial. Bubble wrap packaging material cut to the dimensions of the annulus was gently placed onto the sediment, and the flume slowly filled with artificial sea water so not to disturb the sediment surface (Fukada & Lick, 1980; Widdows et al., 1998b). Mean temperature (13.9 ^{o}C ± 0.9) and salinity (27.9 ^{o}C ± 0.5) was comparable to wintertime field conditions. The flume was then set to generate a low flow velocity (5 cm $^{-1}$) for a period of 14 h to allow *Austrovenus* to bury into the sediment. The bubble wrap packing material was then carefully removed from the water's surface (floats when flumes are filled). An air stone was inserted into the flume to provide adequate aeration while normal light and dark cycles were followed.

2.4.3. Burial times

To measure cockle burial times, two time-laps cameras (with infra-red light sources) per flume were set-up to record over night (14 h). Cameras were positioned on opposite sides of each flume and captured approximately a quarter of the bed area. Burial rate was quantified as the time it took for an individual to completely bury (shell level with sediment surface). For each individual, the time taken for each cockle to become ¼, ½, ¾ and completely buried was recorded. Any *Austrovenus* that did not bury over the 'burial phase' were carefully removed as not to disturb the sediment surface and discarded.

2.4.4. Erosion runs

Every 15 min the flow velocity in the flume was increased by 5 cm s^{-1}. Sediment stability was determined by logging OBS voltage during step-wise increases in current velocity from 5-45 cm s^{-1}. The OBS was clamped to sit

vertically, in the middle of the flow channel, with the mid-point of the sensor positioned 6 cm below the water surface facing the downstream flow (Figure 4). To calibrate the OBS, water samples were taken 10 minutes after each stepwise increase in flow speed for gravimetric analysis of suspended particulate matter (SPM). Water removed during sampling was immediately replaced with artificial sea water of the same volume. Water samples were immediately filtered onto pre-combusted, pre-weighed 47 mm Whatman GF/C filter paper and rinsed thoroughly with RO water to remove excess salts, filters were stored in aluminium foil and placed in the freezer until latter analysis whereby they were oven dried (80 ^{o}C) until a constant weight was reached (24 - 48 h). The OBS voltage from each erosion run was converted to SPM concentration using the following equation; SPM (g L^{-1}) = 0.0337 × OBS voltage + 0.0031 (r^2 = 0.96, n = 107). The cumulative mass of sediment eroded per square metre (ME; g m^{-2}) at the end of each 15 min period of constant flow speed (in relation to measured current velocities) was calculated. The mean erosion rate (the change in sediment erodability through time) during the first 10 min of each period of constant flow (ER; mg m^{-2} s^{-1}) was aslo calculated (Pilditch et al., 2008; Widdows & Brinsley, 2002). This data was then used to determine the mass of sediment eroded at 30 cm s^{-1} (ME-30) and the critical erosion velocity (U_{crit}) defined as the velocity required to erode 10 g m^{-2} and was estimated from the linear relationship between ln(ME) and ln(u) (Ciutat et al., 2007; Pilditch et al., 2008; Widdows et al., 1998a).

2.5. In situ density manipulations

Austrovenus typically only make small adjustments to their vertical position. By planting *Austrovenus* on the sediment surface and allowing them to bury over a 14 h period, they are creating a large disturbance to that sediment.

This complete burial maybe an exaggeration of natural behaviour and may subsequently cause increased disturbance compared to field conditions. To distinguish if there was an increased ME caused by maximum disturbance of *Austrovenus* bioturbation, *in situ* density manipulations were carried out where *Austrovenus* were allowed to bury into the test sediment and remain established for a period of time before erosion runs were commenced. Two representative *Austrovenus* densities were chosen, 150 and 1200 ind. m^{-2}. Field plots were established at the beginning of the three month experimental period (June 08) to give the sediment, *Austrovenus* and other macrofauna time to re-establish before cores were collected for the determination of sediment stability.

Austrovenus from the sediment collection site (Whitford) were gently dug out with a spade and sieved (10 mm sieve) from an area close to the study site. 120 *Austrovenus* were planted onto the surface of a 0.8 x 0.8 m (0.64 m^2) marked plot and repeated on another marked plot (2 identical low density plots). 960 *Austrovenus* were then sieved and planted onto each of the remaining plots (2 identical high density plots) and left for the duration of the sampling period. *Austrovenus* collected for density manipulation at Whitford Estuary were 27.44 ± 0.27 mm (n = 47) size range.

The flume experiments were carried out with intact cores taken from within these field plots (containing *Austrovenus*). Four annular flume box cores were used in the same way as described earlier in ***Section 2.2***, to collect sediment containing *Austrovenus*. A fifth core was taken from the surrounding sediment in order to fill the gap between the first and fourth core as it would have been very

difficult to get an intact sediment 'plug' with a core that contained many *Austrovenus*.

2.6 Laboratory analyses

Grain size was determined from volumetric particle distributions using a Malvern Mastersizer-S after pre-treating samples with 10% hydrogen peroxide to remove organic material (Singer et al., 1988). Sediment organic content (OC), moisture content (wet weight – dry weight / wet weight * 100) and porosity (fluid volume / volume wet sediment) were derived from oven dried (48 h at 80 ^{o}C) and ashed (5 hr at 450 ^{o}C) sample values (Cristie et al., 2000; Herman et al., 1999). Chlorophyll *a* (chl *a*) and phaeopgiments (phaeo) (extraction in 90% acetone for 20 h at 4 ^{o}C in darkness) was determined using a Turner Designs 10-AU Flurometer (Parsons et al., 1984; Welschmeyer, 1994).

Heavy metal analysis of sediment samples was carried out using an aqua-regia digestion procedure based on EPA 200.2 (Martin et al., 1994) and modified through consultancy with Hill Laboratories, Hamilton. Approximately 0.5 g of dry sediment was weighed into a pre-labelled polycarbonate centrifuge tube (PC), and then 5 ml of (1:2) nitric acid and 15 ml of (1:5) hydrochloric acid was added to the sediment. Samples were capped and left overnight in a fume hood. The following day all samples were placed in a 100 ^{o}C water bath for two hours, removed and allowed to cool to room temperature. Once at room temperature 0.25 ml of each sample was transferred to a pre-labelled vulcan tube and diluted with 9.75 ml milli-Q water (1:40 dilution), and then were stored in the freezer until analysis on an inductively coupled plasma-mass spectrometer (ICP-MS). Heavy metal concentrations are expressed as ppm / g of dry weight.

Heavy metal analysis was done on a sub-sample of 10 *Austrovenus* from each experimental run. *Austrovenus* (n = 10) were dissected and tissue homogenised in a mortor containing liquid nitrogen to prevent samples from thawing. 0.5 grams of the homogenised tissue was then placed into a labelled 50 ml centrifuge tube and dissociated with tetramethyl ammonium hydroxide (TMAH). Samples were then heated in a 65 ^{o}C water bath for one hour, vortexed and subsequently heated again for another hour. All samples were then place in an ice bath for 30 min, after which 0.5 ml of cold 50% hydrogen peroxide was added to each container. Capped samples were then refrigerated overnight. The following day all samples were vortexed and 2.5 ml of 16M nitric acid (HNO_3) was added. Samples were then heated to 100 ^{o}C in an oven for one hour and then allowed to cool to room temperature. Samples were then made up to 50 ml with reagent water and mixed (final acid concentration is 5%). Each sample was then fitted with a Millpore Steriflip and filtered into a clean, labelled 50 ml centrifuge container (Smith et al., 2005). The final solution was then analysed by ICP-MS using calibration standards that were prepared with digestion reagents metal concentrations are given in µg metal / g wet weight.

2.7. Data and statistical analysis

To examine sediment properties and *Austrovenus* condition samples were averaged over time. The differences in sediment, tissue heavy metal concentrations and condition index were determined using t-tests ($\alpha = 0.05$). Correlation matrices were used to determine the relationships between the sediment properties.

The comparison of burial time against density treatments was first examined visually as box plots. Replication was assumed as individual *Austrovenus* rather than individual runs, as variation among *Austrovenus* was high. Due to the uneven sample sizes in each density treatment a general linear model (GLM) was employed to examine any significant effects of site (source population) and density on burial time. Model assumptions (homogeneity of variances and normality) were checked through visual examination of residual plots (normal probability and raw residuals versus predictor value), and log transformation carried out to satisfy test assumptions. Post-hoc analysis (Unequal sample size least significant differences (unequal N LSD)) was used to identify any significance treatment (density) effects ($p < 0.05$) within each site (TIR and GLE).

The mass eroded at 30 cm s^{-1} (ME-30) and the critical velocity required to eroded 10 g of sediment (U_{crit}) were chosen as indicators of the erodability of the test sediment and an analysis of covariance (ANCOVA; homogeneity of slopes) was used to test for differences between treatments (density) and source population (site). The flow speed of 30 cm s^{-1} was chosen as an indicator of erodability because it has previously been found to be above the U_{crit} for most intertidal sediments and it is also within the naturally occurring range of tidal currents. In order to identify and explain any variation (in replicates) in the ME-30, sediment properties and *Austrovenus* densities were tested using multiple linear regressions.

Chapter 3

3. Results

3.1. Sediment properties

3.1.1. Sediment collection site

Sediment properties showed little variation over the sample period (Table 2). Median grain size ranged from 117 to 119 μm, while % silt-clay and sand content ranged from 0.6 to 0.8 % and 99.2 – 99.4 %, respectively. Chl *a* and phaeo remained constant throughout the sampling period with the exception of the last sampling date (07/08/2008) where it was twice as high as previous date. Chl *a* and phaeo were highly correlated with one another ($r = 0.94$, $p < 0.001$) and both were positively correlated with median grain size and negatively correlated with % silt-clay content (Table 3). Median grain size was highly negatively correlated with % silt-clay, while there were no correlations with the amount of OC. Macrofauna at the test site was numerically dominated by the amphipod *Paracalliope sp.* and the polychaete *Scoloplos cylindrifer*, followed closely by the juvenile (length < 15 mm) bivalve *Paphies australis* (Table 4).

Table 2: Summary of Whitford surface sediment properties. Data are the means (SD) of three samples (0-1 cm except chl α (0-0.2 cm)).

Date	Median grain size (µm)	% silt-clay	% sand	Porosity (%)	Moisture content (%)	OC (%)	Chl α (µg / g dw)	Phaeo (µg / g dw)
14/06/2008	119 (0.3)	0.6 (0.03)	99.4 (0.03)	64.4 (4.6)	25.8 (0.2)	1.0 (0.07)	8.05 (0.54)	1.70 (0.20)
17/06/2008	118 (0.3)	0.7 (0.03)	99.3 (0.03)	74.9 (2.4)	27.2 (0.2)	1.0 (0.04)	5.24 (0.26)	1.56 (0.12)
19/06/2008	117 (0.3)	0.7 (0.03)	99.2 (0.03)	-	-	0.05 (0.00)	6.34 (0.99)	1.78 (0.15)
01/07/2008	118 (1.5)	0.7 (0.1)	99.3 (0.1)	-	-	0.04 (0.01)	7.16 (0.45)	1.48 (0.24)
26/07/2008	118 (0.3)	0.7 (0.02)	99.3 (0.02)	56.9 (4.4)	24.7 (0.3)	0.9 (0.07)	8.43 (1.26)	2.03 (0.27)
29/07/2008	118 (1.4)	0.8 (0.4)	99.2 (0.4)	59.0 (1.8)	25.6 (0.1)	1.0 (0.14)	9.60 (0.36)	1.92 (0.27)
30/07/2008	119 (0.3)	0.6 (0.02)	99.4 (0.02)	65.0 (2.8)	25.6 (0.9)	0.9 (0.07)	13.26 (0.64)	4.62 (0.52)
05/08/2008	119 (0.02)	0.6 (0.01)	99.4 (0.01)	60.5 (4.5)	26.1 (0.3)	0.8 (0.04)	13.37 (1.39)	3.20 (0.59)
06/08/2008	118 (0.1)	0.7 (0.01)	99.3 (0.01)	44.5 (0.5)	*24.3 (1.4)*	1.0 (0.15)	14.92 (1.61)	1.94 (0.10)
07/08/2008	119 (0.4)	0.6 (0.03)	99.4 (0.03)	66.5 (1.5)	23.9 (0.4)	0.9 (0.08)	31.74 (1.43)	9.18 (1.16)
Mean	118.4 (0.8)	0.7 (0.1)	99.3 (0.06)	61.5 (2.8)	25.4 (0.5)	0.8 (0.4)	11.8 (7.5)	2.9 (2.3)
29/08/2008	***119 (1.1)***	***0.6 (0.1)***	***99.4 (0.1)***	***48.9 (6.3)***	***23.8 (1.0)***	***1.2 (0.1)***	***13.15 (0.56)***	***7.75 (0.50)***
30/08/2008	***119 (0.5)***	***0.7 (0.2)***	***99.3 (0.2)***	***50.1 (3.2)***	***24.8 (0.4)***	***1.5 (0.3)***	***15.26 (0.55)***	***4.26 (0.16)***

29/08/2008 & 30/08/2008 are field plots that contained Austrovenus.

Table 3: Pearson correlation coefficients to quantify relationships between surface sediment properties. Data from Table 1 (14/06/2008 – 07/08/2008) was used, excluding field data (29/08/2008 and 30/08/2008).

	Chl *a* (µg/g dw)	Phaeo (µg/g dw)	Median grain size (µm)	% silt-clay	OC (%)
Chl *a* (µg/g dw)	1.00				
Phaeo (µg/g dw)	0.94***	1.00			
Median grain size (µm)	0.52*	0.59**	1.00		
% silt-clay	-0.46*	-0.55**	-0.76***	1.00	
OC (%)	0.27	0.41	0.38	-0.11	1.00

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.0$

Table 4: Mean macrofauna from the sediment collection site (n=5) at Whitford. Data represent the mean densities (SD) in cores collected during the sampling period.

Taxa	Family/Genera	Density (SD)
Polychaeta	*Orbina papillosa*	1 (0.5)
	Scoloplos cylindrifer	9 (5.1)
	Aonides trifida	2 (0.5)
	Scolelepsis sp.	1 (0.0)
	Macroclymenella stewartensis	1 (0.5)
	Heteromastus filiformis	1 (0.0)
	Ceratonereis sp.	1 (0.0)
Nemertea	*Nemertea*	2 (0.6)
Clitellata	*Oligochaetea*	1 (0.0)
Bivalvia	*Austrovenus stutchburyi*	1 (0.0)
	Macomona liliana	3 (0.5)
	Nucula hartvigiana	4 (1.3)
	Paphies australis	6 (4.4)
Gastropoda	*Cominella glandiformis*	1 (0.0)
	Zeacumantus lutulentus	2 (0.5)
Amphipoda	*Paracalliope sp.*	10 (3.4)
	Colurostylis lemurum	1 (0.0)

3.1.2. Austrovenus collection sites

Sediment was muddier at TIR than GLE (higher % silt-clay content); consequently all other sediment properties measured (with the exception of % sand) at TIR were higher than GLE (Table 5). The sediment measured at TIR was seven times higher in % silt-clay content than GLE, while the chl *a* concentration was only twice that of GLE, as were phaeo and OC. Heavy metal analysis showed that all three metals (Cu, Zn and Pb) were also higher at TIR compared to GLE (Table 5). Cu and Zn concentrations were approximately four times higher, while Pb was only two times higher. T-tests determined that heavy

metal concentrations from TIR were significantly higher from GLE in all measured metals (Table 5).

Table 5: Summary of surface sediment properties from *Austrovenus* collection sites (GLE and TIR). Data are means (SD) of seven samples taken over the duration of the sampling period (0-1 cm except chl α (0-0.2 cm)).

	GLE	TIR
Sediment Properties		
% silt - clay	2.6 (1.6)	19.2 (4.4)
% sand	97.4 (1.6)	80.8 (4.4)
Median grain size (μm)	209.9 (18.0)	195.1 (17.6)
Chl a (μg / g dw)	28.5 (3.7)	46.7 (22.1)
Phaeo (μg / g dw)	8.0 (2.1)	15.2 (7.1)
OC (%)	1.4 (0.2)	3.9 (0.8)
Heavy metals		
Cu (ppm)***	5.0 (0.5)	19.5 (3.6)
Zn (ppm)***	61.7 (4.9)	224.9 (18.1)
Pb (ppm)***	7.8 (0.9)	12.9 (4.2)

Significant differences in metal concentrations between sites (determined by t-tests) are indicated by *. * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

3.2. Austrovenus burial time

An overall downward trend in the burial time was observed for GLE *Austrovenus* as a function of density (Figure 5). The lowest density treatment (150 ind. m^{-2}) from GLE had the greatest variation in burial times, ranging from 20 min to 250 min, while the highest density treatment (1200 ind. m^{-2}) showed little variation and ranged from 35 min to 70 min. The intermediate density treatments had similar burial times, with means of approximately 70 min. Burial times of *Austrovenus* from TIR however, did not illustrate the same patterns observed for GLE. There was an initial decrease in burial time followed by an increase in burial time with increasing density. There is little difference in the variation of *Austrovenus* burial over all density treatments. Overall, these results

suggest that there was considerable variation in the burial times of individuals at all density treatments; although a downward trend seems to be evident, the observed variation is large particularly in the lower density treatments at both sites. To quantify the variation, we used a general linear model (GLM) as sample sizes were unequal. There was no significant difference in burial time between the two sites (Table 6). However, density was found to be significant ($p < 0.05$). A significant interaction term (site*density) suggests that within each site, density had a different affect on the burial time. Post-hoc analysis (Unequal N HSD) indicated a significant difference between the low density treatments (150 ind. m^{-2}) and the high density treatments (1200 ind. m^{-2}) for GLE while a significant difference was found between the 300 ind. m^{-2} and 1200 ind. m^{-2} density treatments for TIR (Figure 5).

3.2.1. Austrovenus condition and heavy metal loading

A sub-sample of *Austrovenus* from each experimental run was analysed to determine *Austrovenus* condition and heavy metal loading of the tissue (Table 7). Condition Index (CI) was similar for TIR and GLE (4.7 and 4.9, respectively) and a t-test confirmed that there was no significant difference between sites ($t=0.9$; $p=0.4$).

Despite the sediment heavy metal concentrations at TIR being approximately four times higher in Cu and Zn and two times higher in Pb, the corresponding body burdens (heavy metal loading) were found to be low and very similar for both source populations (Table 7). T-tests confirmed no significant difference between sites for Zn and Pb tissue concentrations ($p > 0.05$) however,

there was a significant difference between the two sites for Cu concentrations ($p < 0.05$).

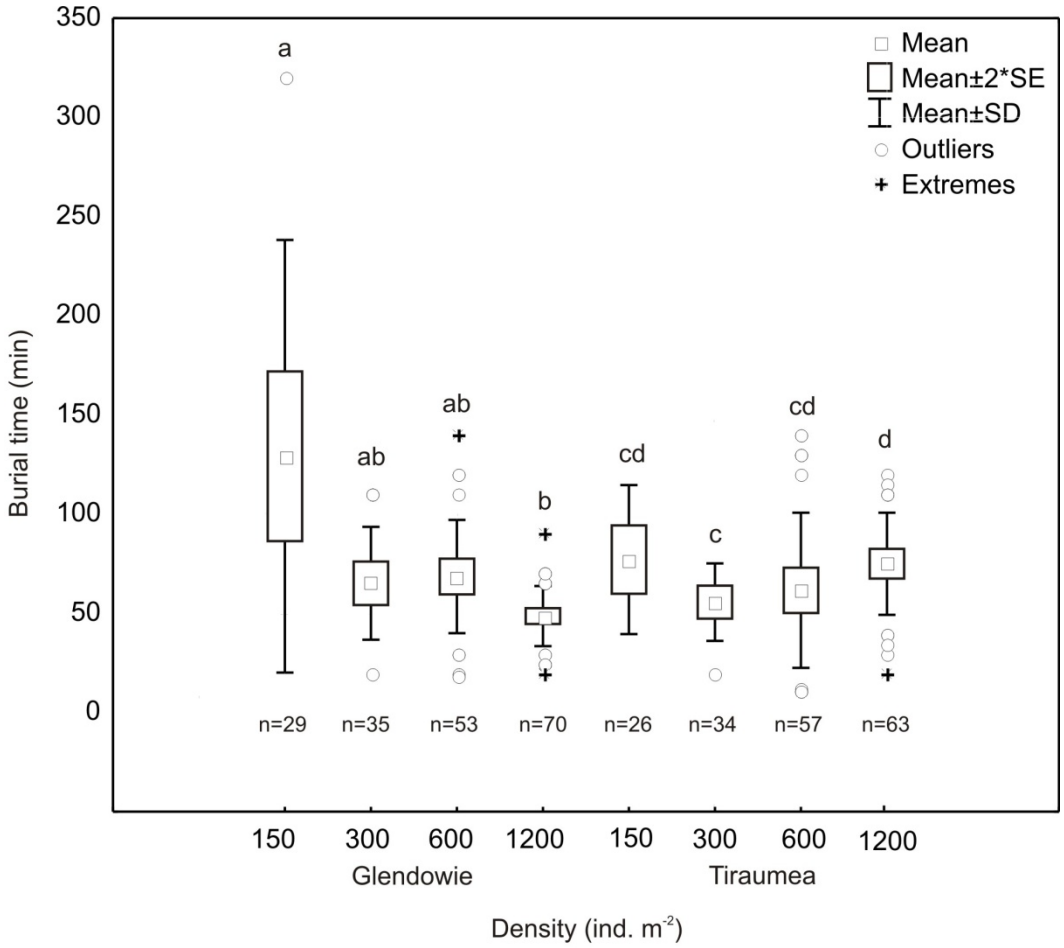

Figure 5: Box plots of the burial rate of *Austrovenus* within the annular flumes. Letters denote differences amongst treatments within site determined by post-hoc analysis.

Table 6: Results of a general linear model (GLM) testing the effects of site and density on burial time.

Effect	SS	DF	MS	F	p
Intercept	3373784	1	3378784	382.58	0.000
Site	16603	1	16603	1.88	0.171
Density	205969	3	68656	7.77	0.000
Site*Density	232309	3	77436	8.77	0.000
Error	3170496	359	8831		

Table 7: Mean Condition Index (CI ± 1 SD) (n = 6) and heavy metal concentrations (n=10; Zn, Cu and Pb) measured in *Austrovenus* used in experimental runs. * indicates significant differences between sites (determined by t-test).

	Glendowie	Tiraumea
CI	4.9 (0.3)	4.7 (0.2)
Cu (ppm)***	1.1 (0.1)	2.1 (0.5)
Zn (ppm)	11.7 (3.2)	14.0 (2.1)
Pb (ppm)	0.1 (0.0)	0.1 (0.0)

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

3.3. Erosion Study

3.3.1. Austrovenus collection sites

The effect of *Austrovenus* on the sediment stability varied with treatments (density) however, the sediment stability did not vary between source populations. For all replicates there was no effect of current speed on the mass eroded (ME) below 20 cm s^{-1}. The presence of *Austrovenus* consistently reduced the sediment stability resulting in an increase in the ME at current velocities greater than 20 cm s^{-1} compared to the 0 density treatment (Figure 6). After current velocities of 20 cm s^{-1} there was a sharp exponential increase in the ME. The 0 and 150 ind. m^{-2} treatments show the greatest variation between replicates in mass eroded compared to the later three density treatments (300, 600 and 1200 ind. m^{-2}). The erodability between density treatments did not vary greatly, although there was a tendency for increased erodability with increasing densities (Figure 6). The erosion rate (ER) measured in the first 10 min of each velocity increment increased as a function of increasing current speed and a similar treatment effect as ME was observed in the ER (Figure 7), with increasing current speed there was a logarithmic increase in ER.

The mass eroded at 30 cm s^{-1} (ME-30; g m^{-2}) showed much variation within density treatments (Figure 8a). A test of the homogeneity of slopes (ANCOVA) showed that there was no significant effect of source population and no significant interaction (site*density), only a significant difference was observed with density (Table 8). When the data for 0 ind. m^{-2}; 'zero *Austrovenus'*, was excluded (as zero is not a density/population effect) density was not significant however, when included it was found that density was significant. This confirms that there is a significant difference between sediment void of *Austrovenus* and sediment containing *Austrovenus*.

U_{crit} values are an estimate based on current velocity (cm s^{-1}) required to erode >10 g sediment m^{-2}. When plotted against density U_{crit} demonstrates a slight decrease in the U_{crit} with increasing cockle density (Figure 8b). Homogeneity of slopes determined that there was also a significant difference between U_{crit} and density (Table 8). The range of velocities required to erode 10 g m^{-2} with increasing density treatments was very small (ranging from 30 to 28 cm s^{-1}), only a 2 cm s^{-1} change in the current velocities over all density treatments (0 – 1200 ind m^{-2}). When the data for 0 ind. m^{-2} was excluded, density was not significant however, when included it was found that it was. This again confirms that there is a significant difference between sediment void of *Austrovenus* and sediment containing *Austrovenus*.

Multiple linear regression models found that the sediment erodability parameters were not highly significant. The ME-30 regression indentified the density of *Austrovenus* as the only predictor variable under constraints $\alpha < 0.15$ however this was not significant ($p = 0.19$) (Table 9). A plot of ME-30 against

Austrovenus density shows a weak positive log relationship (Figure 8a). The best fit model for predicting ME-30 was a log regression ($r^2 = 0.43$; $p = 0.01$), resulting in the equation: ME-30 = 6.72 $\log_{10}$ (x), + 11.66 where x is the density (0-1200 ind. m^{-2}) used in treatments. The U_{crit} regression also indentified the density of *Austrovenus* as the only predictor variable. A plot of U_{crit} against density shows a weak exponential relationship (Figure 8b). The best fit model for predicting U_{crit} was a exponential regression ($r^2 = 0.30$, $p = 0.01$), resulting in the equation $U_{crit} = 29.04e^{-0.005x}$, where x is the density treatments used (0-1200 ind. m^{-2}).

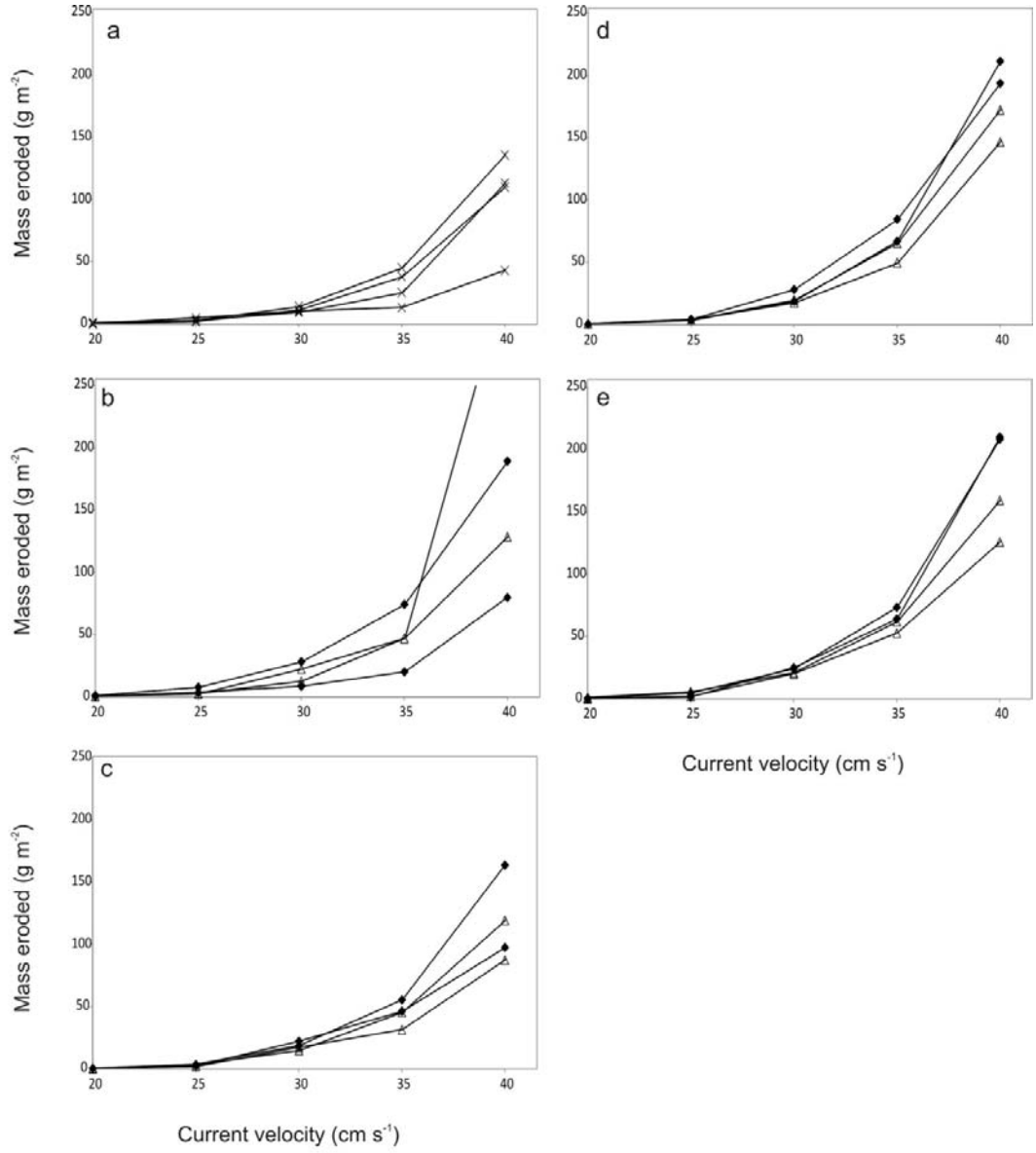

Figure 6: Mass eroded (g m^{-2}) at each step-wise increase in current velocity, (a): 0 ind. m^{-2}; (b): 150 ind. m^{-2}; (c): 300 ind. m^{-2}; (d): 600 ind. m^{-2}; (e): 1200 ind. m^{-2}. ♦ denotes *Austrovenus* from Glendowie, Δ denote *Austrovenus* from Tiraumea.

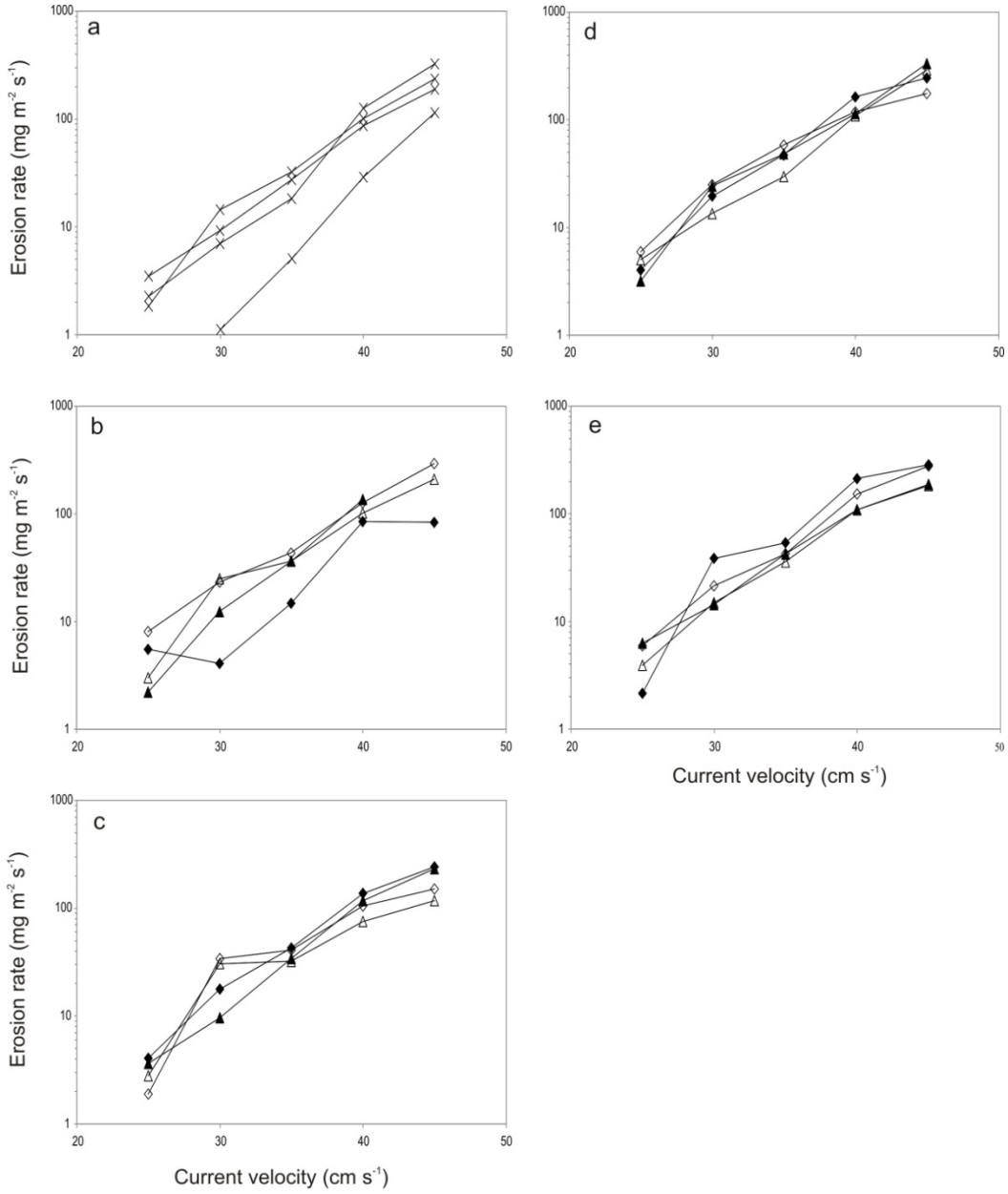

Figure 7: Mean erosion rate of test sediment for *Austrovenus* from each site (Glendowie (♦), Tiraumea (Δ)) for different density treatments (a): 0 ind. m^{-2}; (b): 150 ind. m^{-2}; (c): 300 ind. m^{-2}; (d): 600 ind. m^{-2}; (e): 1200 ind. m^{-2}

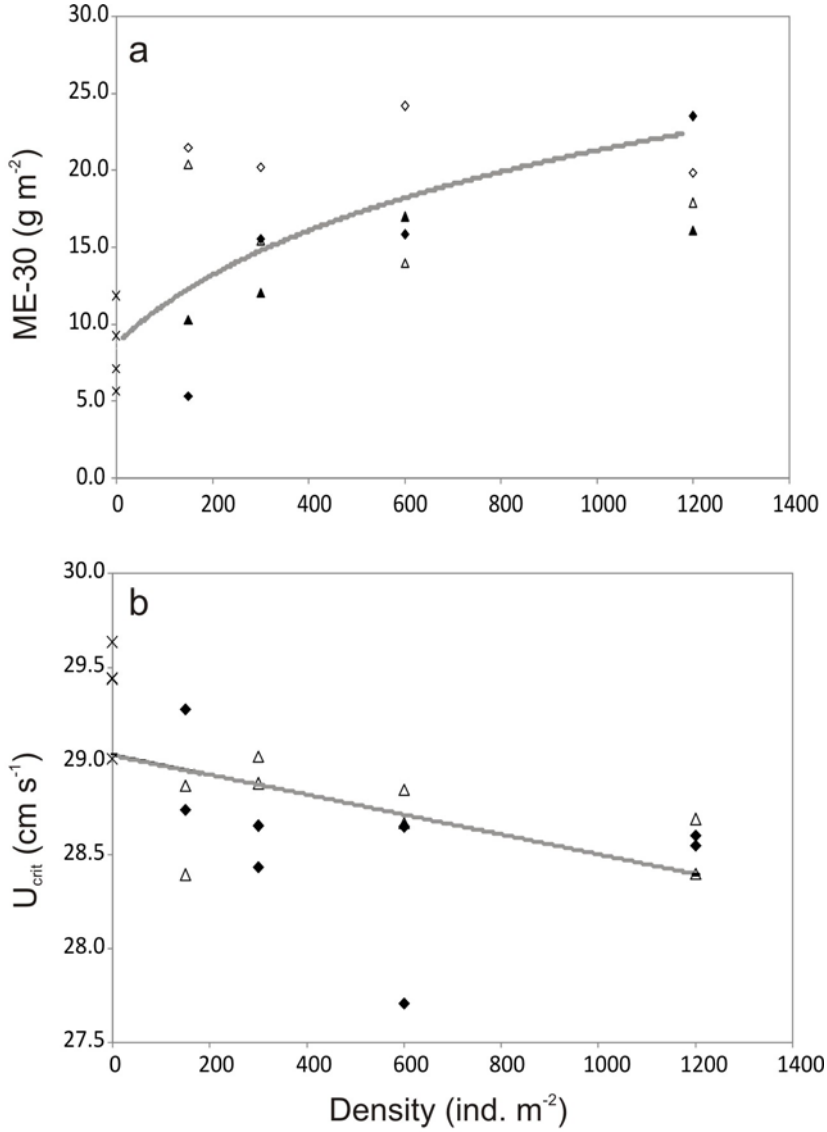

Figure 8: Mass of sediment eroded at 30 cm s^{-1} (ME-30 (a)) with best fit model prediction ME-30 = 6.71log(x) + 11.66 (r^2 = 0.43, p = 0.1) and the velocity required to eroded 10 gm^{-2} (U_{crit} (b)) with best fit model prediction $U_{crit} = 29.04e^{-0.005x}$ (r^2 = 0.30, p = 0.01). ♦ denotes *Austrovenus* from Glendowie, Δ denotes *Austrovenus* from Tiraumea.

Table 8: Results of ANCOVA to test the significance of source population (site) and *Austrovenus* density on the mass eroded at 30 cm s^{-1} (ME-30) and the critical velocity required to erode >10 g of sediment (U_{crit}), also to test if any interaction term (site*density) exists. *indicates significance

		SS	DF	MS	F	P
ME-30	Site	0.21	1	0.21	0.008	0.93
	Density**	211.6	1	211.6	8.12	0.01
	Site*Density	18.604	1	18.604	0.71	0.42
	Error	417.08	16	26.07		
U_{crit}	Site	0.25	1	0.25	1.70	0.21
	Density*	1.06	1	1.06	7.33	0.02
	Site*Density	0.04	1	0.04	0.29	0.59
	Error	2.32	16	0.15	0.	

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

Table 9: Results of multiple linear regression model to test the significance of *Austrovenus* density and sediment properties on the mass eroded at 30 cm s^{-1} (ME-30).
To aid comparisons amongst predictors standardized (β) coefficients (1 SE) have been presented.
*indicates significant

	Predictor	Coefficient		Regression	
		(SE)	p-value	p-value	r^2
ME-30	Density*	0.60 (0.2)	0.02	0.19	0.45
	Chl *a* (µg/gdw)	-0.62 (0.6)	0.35		
	Phaeo	0.56 (0.7)	0.42		
	Median grain size (µm)	-0.13 (0.4)	0.73		
	% silt-clay	0.17 (0.3)	0.62		
	OC (%)	0.31 (0.2)	0.24		
U_{crit}	Density*	-0.56 (0.2)	0.03	0.20	0.44
	Chl *a* (µg/gdw)	0.72(0.7)	0.29		
	Phaeo	-0.79 (0.7)	0.27		
	Median grain size (µm)	0.37 (0.4)	0.35		
	% silt-clay	-0.09 (0.3)	0.81		
	OC (%)	-0.29 (0.3)	0.27		

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

3.4. In situ density manipulations

Sediment properties measured within both field plots were very similar to the surrounding test sediment, with the exception of OC which was on average twice as high as the surrounding test sediment (Table 1). There was no affect of current speed on ME before 20 cm s^{-1}. After 20 cm s^{-1} there was an exponential increase in the ME with increasing current speeds (Figure 9a). The low density plots indicate greater ME compared to the higher density plots. A similar relationship is observed between the increasing current speeds and the ER (Figure 9b). When compared to the laboratory added density treatments there was little or no difference between the ME-30 of the laboratory and field high density treatments (1200 ind. m^{-2}). However, there was a difference between the ME-30 of the laboratory and field low density treatments, (150 ind. m^{-2}). There ME-30 was approximately 2 times higher for field low density compared to the laboratory low density (Figure 10a). This relationship was also observed for U_{crit}, with little or no variation in the velocities for the high densities and the laboratory low density treatment. The U_{crit} for the field low density treatment was only slightly less than the U_{crit} for the laboratory low density treatment (26.5 and 28.5 cm s^{-1}, respectively), resulting in only a 2 cm s^{-1} difference in current speed (Figure 10b).

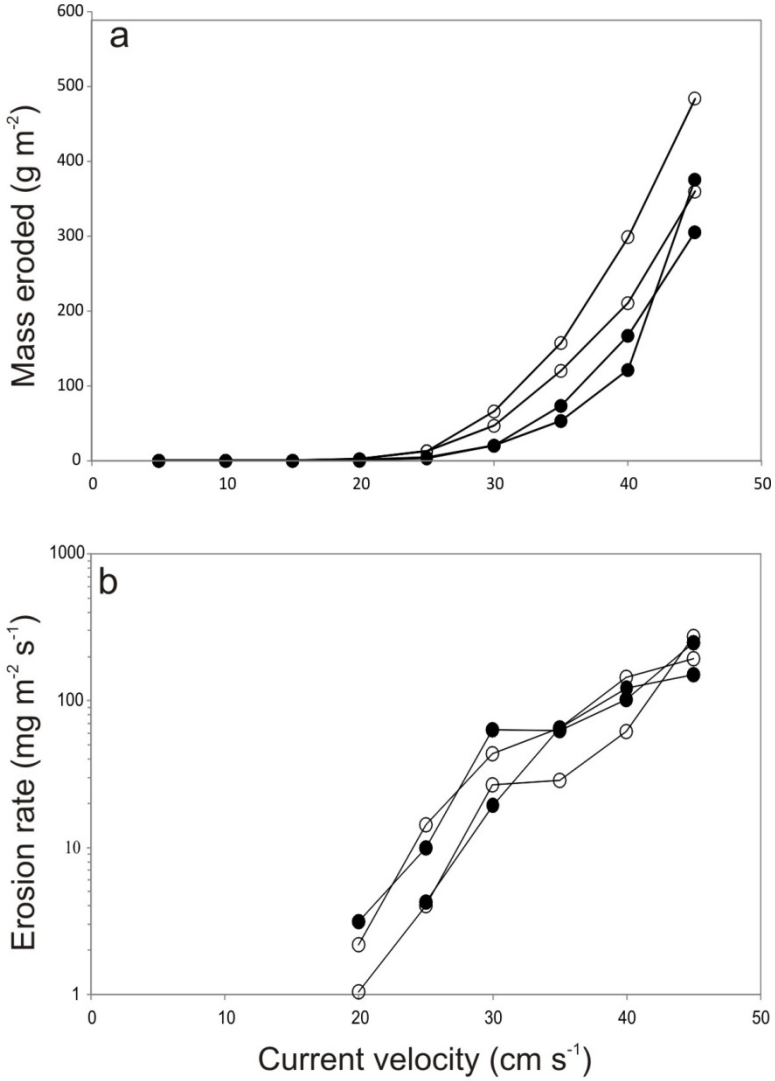

Figure 9: (a) Mass of sediment eroded at the end of each velocity step (g m^2) and (b) the average erosion rate for the first 10 min of constant flow (mg m^{-2} s^{-1}). Open circles denote low density treatment (150 ind. m^{-2}) while the closed circles denotes the high density treatments (1200 ind. m^{-2}).

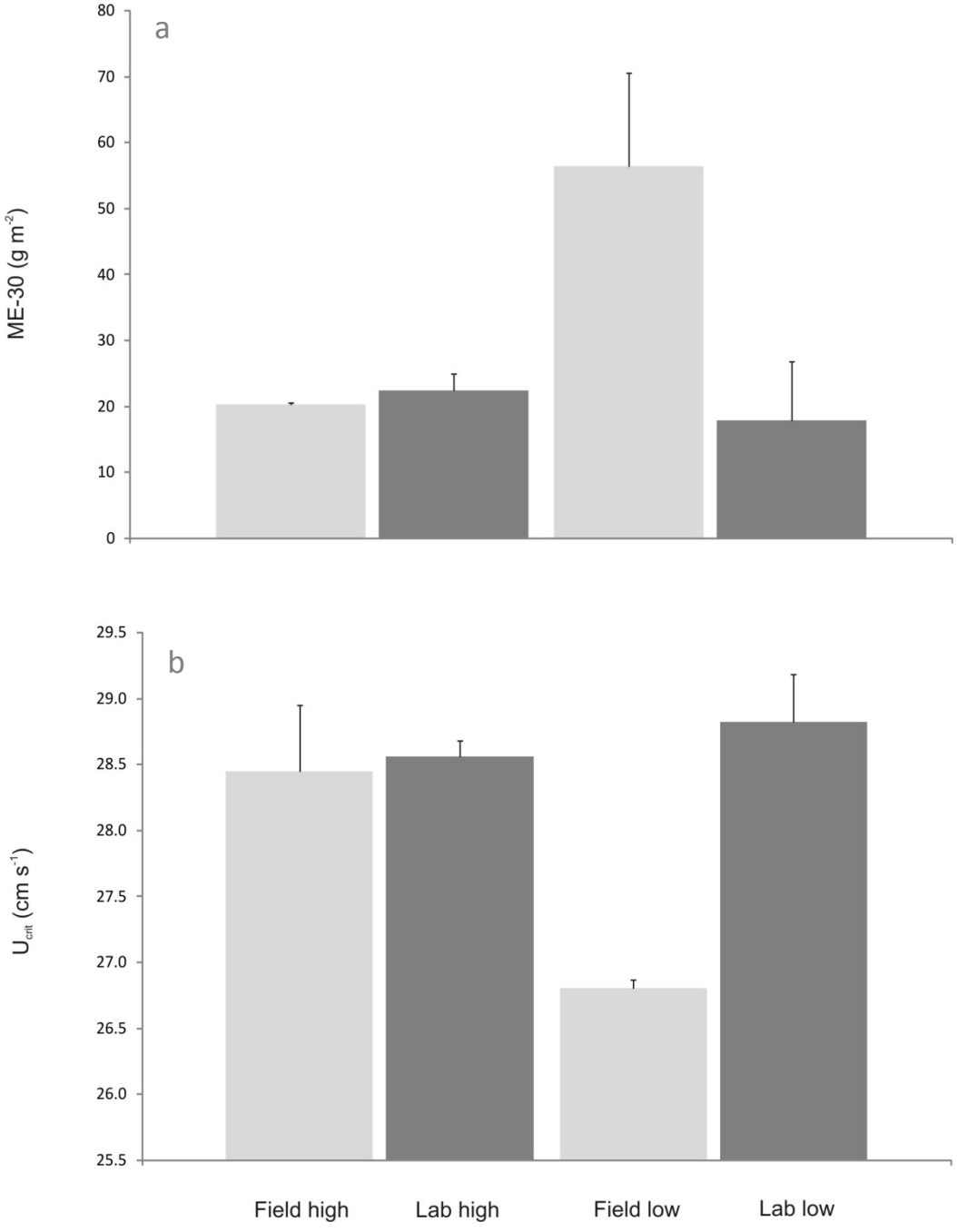

Figure 10: (a) Mass eroded at 30 cms^{-1} (ME-30) and (b) critical erosion velocity required to erode 10 g (U_{crit}) of sediment for field data (n=2) and corresponding laboratory data (SD)(n=4).

Chapter 4

4. Discussion

4.1. Austrovenus collection and burial

4.1.1. Sediment properties

This study examined the surface sediment properties of two sites along a heavy metal contamination gradient in Tamaki Estuary, Auckland. Analysis confirmed a contamination gradient was present in Tamaki Estuary. Tiraumea had significantly higher levels of heavy metals ((4x Cu, Zn) and (2x Pb)) than Glendowie. Surface sediment properties measured at TIR showed a lower median grain size and subsequently a higher % silt-clay content, signifying muddier sediment than GLE. Large estuaries are often characterised as having a natural gradient in sediment grain size, with finer grained material in the upper reaches (mud) increasing to coarser grained material with distance towards the sea (sand) (Cortesão & Vale, 1995). Thus it was expected that TIR would have a higher proportion of fine material.

Characteristically, the adsorption of heavy metals to sediments increases with increasing proportions of silt-clay as a result of particle-metal interactions and the greater relative surface area finer grained sediments (Abrahim et al., 2007; Mitchener & Torfs, 1996). We would expect higher contaminant concentrations in the upper reaches of Tamaki Estuary for at least two reasons. (1) TIR is composed of finer grained sediments than GLE and (2) TIR is much closer to industrial contaminant sources and storm water inputs from the urbanised catchment.

Heavy metal concentrations measured in this study correspond well with those previously measured by the Auckland Regional Council (ARC), where zinc concentrations have been recorded above 100 ppm for areas including and surrounding TIR, and between 50 – 80 ppm for areas including and surrounding GLE (Abrahim & Parker, 2002; Abrahim et al., 2007; ARC, 2007). The measured heavy metal concentrations in Tamaki Estuary indicate that sediments are indeed contaminated, albeit at levels somewhat lower than estuaries in the United Kingdom (UK) (Table 10).

Table 10: Sediment heavy metal contaminations (μg g^{-1} dwt) from 4 Estuaries in the United Kingdom (Data are taken from Bryan & Langston., 1992) and the two sites in the Tamaki examined in this study.

Estuary	Cu	Zn	Pb
Tamar	330	452	235
Humber	54	252	113
Severn	38	259	89
Avon	18	82	68
Tamaki (TIR)	20	225	13
Tamaki (GLE)	5	62	8

Several numerical sediment quality guidelines have been established to provide interpretative tools for assessing the biological significance of chemical (heavy metal) pollution. The Australian and New Zealand Environmental and Conservation Council along with the Agriculture and Resource Management Council of Australia and New Zealand (ANZECC/ARMCANZ, 2000) have adapted and implemented the use of sediment guidelines (Interim sediment quality guidelines; ISQG) developed by Long et al. (1995) to assess and/or monitor the ecological risk to organisms living within that sediment (Table 11). Within New Zealand, ARC also uses these guidelines for risk assessment of sediment heavy

metal contamination. This study and those carried out by ARC recorded the chemical (heavy metal) contamination levels in surface sediments around the mouth of Tamaki Estuary (GLE) as below threshold-effects levels, which corresponds to effect range-low (ERL) (Long et al., 1995). ERL suggests that sediment contamination at this level has a low effect on organisms living within those sediments and the range or degree that is affected will also be low. Sediment contamination in the upper reaches of Tamaki Estuary (TIR) were recorded above ERL but still below the effects range-medium (ERM) suggesting that organisms residing there are more probable to exhibit low effects of contamination. Although effects on organisms are more probable with these concentrations they are still not likely to cause large-scale effects throughout the entire ecosystem. Concentrations below this threshold are rarely associated with biological consequences, but as concentrations increase above ERM, the probability of biological consequences increases.

Table 11: Interim sediment quality guidelines (ISQG) set for Australia and New Zealand, for the heavy metals measured in this study (Cu, Zn and Pb).

Heavy Metal	ISQG-low (ERL)	ISQG-high (ERM)	Tiraumea	Glendowie
Cu	65	270	20	5
Zn	200	410	225	62
Pb	50	220	13	8

ISQG-low and high correspond to the effects range-low (ERL) and –med (ERM) used in the National Oceanographic and Atmospheric Association (NOAA) listing (Long et al., 1995).

This study shows that although there is a sediment contamination gradient in Tamaki Estuary, the range of contamination may not be high enough to cause major physiological or behavioural changes in infaunal organisms. With both sites below ERL, detection of site differences with respect to contaminant effects

on behaviour and physiology of *Austrvenus* was expected to be difficult. However, increases in contamination levels in the upper reaches of Tamaki Estuary (TIR) could increase the probability of biological consequences over time if contamination continues to rise.

4.1.2. Heavy metal loading and CI

Although behavioural responses such as burrowing have been shown as a more sensitive indicator of stress than physiological responses such as body burden and CI (Hummel et al., 1997; Sokolowski et al., 1999), heavy metal loading (body burden) and CI were still measured in this study. Condition analysis on *Austrovenus* used in this study found no significant site difference. Heavy metal analysis on *Austrovenus* tissues found a significant site (population) difference in Cu concentrations while there was no significant difference in Zn and Pb between sites. Although statistically significant, the measured Cu tissue concentrations were still very low (2.1 ppm (TIR) and 1.1 ppm (GLE)). Hummel et al. (1997) found that the CI of the European cockle was only negatively affected by increases in Cu concentrations (2-15 ppm), as well as being significantly correlated with the natural environmental gradient of the estuary. While Duquesne et al. (2004) recorded that stress due to elevated, but not sub-lethal cadmium concentrations affected the overall fitness (CI) of *Macoma balthica*. These particular studies focused on the effects of adding contamination to sediments rather than the effects of already contaminated sediment. Although there are elevated contamination levels in Tamaki Estuary, both sites are classified as uncontaminated according to sediment quality guidelines (Long et al., 1995). While a contamination gradient is present in Tamaki Estuary it did not have a

negative effect on CI and tissue heavy metal loading of *Austrovenus* used in this study. These results support the expectation that there may not be a significant site difference in behavioural or physiological response of *Austrovenus* from the two source populations.

4.1.3. Burial time

This study examined the effect that elevated metal contamination had on the burial of *Austrovenus* taken from either extremes of a contamination gradient. No significant site difference was found in *Austrovenus* burial times for the two source populations. This was not surprising as there was no site difference found in the condition and internal heavy metal loading of individuals used, and sediment heavy metal concentrations were measured as below ERL (uncontaminated).

Post-hoc analysis found that there was a significant difference between two density treatments within each site however, this was not consistent between sites. Trends observed for GLE suggested that as *Austrovenus* densities increased, burial time decreased (Figure 5). Large variation in burial time was observed in the 150 ind. m^{-2} treatments for GLE. It was observed that *Austrovenus* in these treatments consistently moved laterally around the flume; evident the following day through surface tracks and through time lapse video recordings. Lateral moving *Austrovenus* would then initiate burial when they came into contact with other *Austrovenus* or the flume walls (personal observation).

Observed burial time for *Austrovenus* from TIR did not follow the same pattern exhibited by *Austrovenus* from GLE (Figure 5). Although burial time was again most variable in 150 ind. m^{-2} treatments, mean burial time did not differ much across the different *Austrovenus* density treatments. It was expected that in the higher density treatments there would be more contact among laterally-moving *Austrovenus,* which would tend to initiate burying behaviour and reduce average burial time. Findings at GLE support the prediction, but findings at TIR did not. Richardson et al. (1993) found that the European cockle (*Cerastoderma edule*) moved randomly across the surface of sediment and that initiation of burial would often induce other already buried cockles to emerge whereby they would then have to re-orientate and re-burrow. These same trends were observed in this study, whereby it was frequently observed that when *Austrovenus* came into contact with others or the flume wall they would initiate burial. This could explain some of the variation in burial times exhibited by *Austrovenus* at high densities (TIR).

Experimental studies have also shown that lateral movement or crawling is a widespread escape mechanism of suspension feeding bivalves in unfavourable habitats (Ansell, 1994; Mouritsen, 2004). Lateral migration from unfavourable conditions has been shown in numerous studies as a common response for suspension feeding bivalves trying to escape such unfavourable conditions as hypoxia, toxicity caused by oils, pesticide and other chemical contamination as well as disturbed sediment (reworked sediment/lose sediment) (McGreer, 1979; Møhlenberg & Kiørboe, 1983; Mouritsen, 2004; Phelps et al., 1983). The constraints of the annular flume meant that lateral migration for *Austrovenus* away from unfavourable conditions (disturbed/reworked sediment)

was inhibited and may well have sped up burial times due to increased contact with walls. Mouritsen (2004) observed that *Austrovenus* crawled often in disturbed sediment and reasoned that this was due to their preference for vertical positioning. A disturbed sediment matrix means that in order for *Austrovenus* to maintain their preferred height within the sediment they would have to repeatedly crawl. Perhaps this is why TIR high density treatments recorded longer burial times and subsequently could be why there was greater variation in 150 ind. m^{-2} as large lateral migration would have created large areas of disturbed sediment.

It is noted that *Austrovenus* in New Zealand occur in beds and are rarely found in sparse densities (Backwell, 1984; Larcombe, 1971; Stephenson, 1981); this could explain why *Austrovenus* in the 150 density treatments moved much more than the other higher densities. In addition, this could also explain why initiation of burial occurs on contact with other *Austrovenus*. It would be plausible to then assume that in the 150 ind. m^{-2} density treatments there was little or no contact with other *Austrovenus* and thus burial time was greater than *Austrovenus* in the higher density treatments. The small ranges of movements up and down exhibited by *Austrovenus* are not likely to cause large resuspension of particles from the seabed. It is more likely that they can resuspend a small amount of surface sediment (containing microphytobenthos) but this resuspension is localised. Localised resuspension may be a plausible explanation as to why *Austrovenus* occur in beds rather than individually. For example, if *Austrovenus* are able to select and subsequently reject unfavourable food (particle selection and biodeposits) then they would be able to feed on the localised resuspended sediment created by their neighbours and thus suspend particulate for their neighbour.

4.2. Test Sediment

4.2.1. Erosion Sequence

This study examined the influence of different densities of the suspension feeding bivalve *Austrovenus* had on the erodability of particles in response to step-wise increases in current velocities, from which sediment stability was quantified. Results demonstrated remarkable similarities in the mass of sediment eroded (ME; g m^{-2}) and erosion rate (ER; mg m^{-2} s^{-1}) between sediment containing different source populations (sites) thus, no site difference was evident. These experiments so far have found no detectable effect of source population on behaviour (burial time), physiological (CI) responses of *Austrovenus* and on erodability of the sediment.

There was no observable decrease in sediment stability with increasing density treatments (150-1200 ind. m^{-2}). However, there was a pronounced difference between the erodability of sediment without *Austrovenus* versus erodability of sediment containing *Austrovenus*. The addition of *Austrovenus* to the sediment surface was followed by a large disturbance of sediment as they buried. *Austrovenus* created hollows and depressions in the sediment as they buried and settled, increasing surface roughness. Increased bed roughness would subsequently alter hydrodynamics; affecting/altering the amount of sediment entrained into the water column. Huettel et al. (1996) found that suspended particulate matter (SPM) is driven by pressure gradients generated when bottom flows are deflected by small surface structures, usually no greater than 5 cm of hydrodynamical or biological origin (e.g. muds are often dominated by diffusive/irrigational exchanges, versus sands which are characterised by

advective transport). This study found that there was negligible variation in the ME of all densities but there was considerable variation in the 150 ind. m^{-2} treatments. This supports the earlier findings where there was considerable variation in the observed burial times of *Austrovenus* in this same density treatment.

The mass of sediment eroded at 30 cm s^{-1} (ME-30) and the critical velocity required to erode a given mass of sediment (U_{crit}; 10 g m^{-2}) have previously been used in other studies as comparative measures to access the effects that a particular parameter (density) has on the erodability of estuarine sediment at varying current velocities (Ciutat et al., 2007; Pilditch et al., 2008; Widdows et al., 1998a; Widdows et al., 2000a). This study demonstrated an increase in ME-30 with increasing *Austrovenus* density. Subsequently, there was a slight decline in the U_{crit} with increasing density. Although a statistically significant difference was found between ME-30 and bivalve density (and U_{crit}), when the data for the 0 ind. m^{-2} treatments was excluded, there was no density effect. This suggests that the presence or absence of *Austrovenus* maybe the driving factor of sediment erodability (and U_{crti}).

The greatest variation in ME-30 was observed in the 150 ind. m^{-2} density treatments, which also showed the greatest variation in burial times. These treatments demonstrated considerable lateral movement and caused visible surface tracks to appear after the burial phase. Sufficient lateral migration by a small number of *Austrovenus* may have the same effect on the sediment erodability as a high number of *Austrovenus*. It has previously been found that the small bivalve, *Transenella tantilla,* causes surface tracking, resulting in significantly increased

bed roughness and subsequently decreased critical erosion velocity by 20% (Nowell et al., 1981). Furthermore, the physical presence of bivalves can alter bed roughness affecting the erodability of that sediment (Green et al., 1998; Herman et al., 1999). Visual observations suggest that if the 150 ind. m^{-2} treatments were removed there may be a significant site difference. It would be interesting to quantify the relative contributions of lateral migration and vertical burial, which seem to interactively affect erodability, but this was beyond the scope of the current study. Nevertheless, my results show that there are at least two different levels of functioning: when there are no *Austrovenus* present sediment erodability is low, and when *Austrovenus* densities are > 150 ind m^{-2}, sediment erodability is significantly higher. There may also be an intermediate area where it could be burial time or lateral movement that is causing this shift in functioning. Further investigation is needed to confirm this.

Resuspension of sediments by biota is a potentially important mechanism by which SPM and their absorbed contaminants are re-introduced into the water column, making contaminants available for consumption by higher organisms. The variation in ME-30 (and U_{crit}) could not be explained by the surface sediment properties measured (Table 9), leaving *Austrovenus* density as the only predictor variable having an effect on the erodability of the test sediment. A significant effect was observed between U_{crit} and density however, the decrease in velocity was only slight, a 2 cm s^{-1} decrease over all density treatments (0-1200 ind. m^{-2}). These results indicate that there is little or no effect of *Austrovenus* on the critical erosion threshold, suggesting that in the absence or presence of *Austrovenus* the current speed required to erode 10 g m^{-2} of sediment (Whitford sediment) would remain somewhere between 28.5 and 30.5 cm s^{-1}.

The impact of increased bivalve density on sediment erosion has previously been studied and recorded in both field and laboratory studies for deposit feeding clams as well as suspension feeding clams (Lelieveld et al., 2003; Sgro et al., 2005; Willows et al., 1998; Widdows et al., 2000a; Widdows et al., 2000b; Widdows & Brinsley, 2002). While many infauna influence the sediment column via the creation and irrigation of permanent burrow structures (Aller, 1988; Pemberton et al., 1976; Posey et al., 1991) *Austrovenus* do not have permanent burrows, rather they 'wriggle' through the sediment (typically short distances), causing displacement of surface sediments. The disturbance caused by *Austrovenus* may be insignificant (maximum ME at 45 cm s^{-1}; 682 g m^{-2}) in this study compared to disturbance caused by other bioturbating bivalves in previous studies which have been recorded as high as approximately 2100 g m^{-2} (Ciutat et al., 2007; Widdows et al., 2000a). This is predominantly due to the fact that the sediment used in this study is fine sand while the sediment used in all the experimental studies in the UK are cohesive muds. The behaviours of these types of sediment are very different. The erodability of sands tends to occur at lower velocities with grains transported in bedload while cohesive sediment tends to be eroded once the bound nature of the sediment has been destroyed.

The sediment in which the *Austrovenus* were placed was classified as very fine sand with a median grain size of 118.4 ± 0.8 μm (Wentworth, 1922). It is generally accepted that grain size determines the erosional behaviour and transport of sandy sediment in moving water (Hjulstorm, 1939). The sediment characteristics of very fine sand suggest that small disturbances will cause large disruption to the sediment matrix and the entrainment of particles will occur. Therefore, the shear presence of *Austrovenus* will cause a physical disturbance at

the sediment boundary, resulting in erosion around the *Austrovenus* which will increase the erosion of the sediment. Previous experimental studies at two sites in Tauranga Harbour, New Zealand; found that although there were contrasting bio-physical sediment properties between sites there was no significant difference between the site-density treatments (Sandwell, 2006). This author found that at 45 cm s^{-1} there was an average ME ranging from 417-1335 g m^{-2} for site S which had a median grain size between 128 - 177 (17.5 - 30.8 % silt content). While the average ME for site E ranged from 602 - 792 gm^{-2}, which had a median grain size between 162 - 185 (10 - 15 % silt content). Site E is the most comparable with this study as the median grain size and ME are similar compared to those of site S. In this study there was very little % silt-clay content present in the test sediment (mean 0.7%), which could account for the lower ME (ranged from 125 - 682 g m^{-2}) seen in these experiments. Based on abiotic sediment motion theory, it would be expected that sediment stability would be greater for the sediment used in this study as the grain size was coarser and the proportion of silt-clay was very low (e.g. Miller et al., 1977; Mitchener & Torfs, 1996).

4.2.3. Field versus laboratory

This part of the study examined the influence of laboratory planted *Austrovenus* and field buried *Austrovenus* on the sediment erodability. The short duration of the laboratory experiment meant greater sediment disturbance is likely to occur compared to sediment containing already buried *Austrovenus*. The field manipulations showed the same characteristics in the ME and ER experiments as those seeded in the laboratory. There was a marked increase in the ME at the lower density (150 ind. m^{-2}) than measured at the higher density treatment (1200

ind. m^{-2}). It could be that the higher density is providing a barrier to the sediment and protecting/armouring it from erosion. It was observed that there was no difference between the ME-30 and U_{crit} between the laboratory and field high density treatments however; there was a difference between the laboratory and field low density treatments. The field low densities showed much more erosion than the laboratory low densities which corresponded to a much lower U_{crit} (Figure 10). These results would suggest that in the field there was much more movement and disturbance to the low density plots than there was to the low density runs in the laboratory. Sediment properties measured in these plots where all similar to those of the surrounding Whitford sediment used in laboratory tests, with the exception of OC, which measured approximately 2 times higher than the surrounding sediment. This increase in OC could be associated with the presence of the bioturbator that are feeding and thus producing biodeposits. As previously mentioned *Austrovenus* from large beds/congregations whereby there is little space between them, it could be that the *Austrovenus* in the 150 ind. m^{-2} field treatment are moving around more in order to 'find' each other and collectively form a bed.

4.3. Implications of bioturbation to that fate of those contaminants

Activities such as bioturbation often induce significant release of metals from the sediments into the overlying water, mostly in the form of resuspended particulate matter (Rasmussen et al., 2000). Resuspended heavy metals in particulate form are not readily available to benthic or pelagic species (Ciutat & Boudou, 2003; Wall et al., 1996). Although in this study there was no significant difference between the burial times of *Austrovenus* from TIR compared to GLE,

the reworking activities of these organisms still exert a crucial control on the transport and fate of contaminants in aquatic sediment (Banta & Andersen, 2003).

4.4. Summary

This study demonstrated no consistent difference in the burial time between the two source populations (GLE and TIR). This was explained by no measured differences in the condition index and heavy metal loading of organisms used in this study. This study has found that the present range of contamination measured in Tamaki Estuary, Auckland, may not be high enough to cause major physiological or behavioural changes in infaunal organisms, such as *Austrovenus stutchburyi*.

Sediment erodability (ME-30 and U_{crit}) was not significantly correlated with the measured environmental and biotic factors. Density of *Austrovenus* was the only predictor variable that could be used to explain any variation in sediment erodability. There was no significant density effects observed between the amounts of sediment eroded for densities > 150 ind. m^{-2}. There was a significant difference between sediment void of *Austrovenus* (0ind. m^{-2}; smooth, flat undisturbed sediment surface) and sediment containing *Austrovenus* (>150ind. m^{-2}; physical structure on/in the sediment surface, increase in bed roughness).

4.5. Future research possibilities

It must be noted that this study lacks seasonality, with experiments carried out during wintertime. Summertime sampling may have resulted in difference condition indices as reproduction and spawning occur during spring and summer

months. Also noted is the lack of temporal and spatial replication due to both time and logistic constraints. It is therefore expected that this research will be used in collaboration with other studies on this same species and these same sites to produce a broader understanding of the roles that *Austrovenus stutchburyi* play in estuarine functioning (benthic-pelagic coupling). In order to obtain a better understanding of the role that *Austrovenus stutchburyi* plays (the importance of the various feedbacks and limitations and interrelationships) future research should examine the following:

- This study found that there was considerable variation in the measured burial rate and the greatest variation was measured in the lowest density treatment which corresponded to the greatest variation in the mass eroded. Thus, it would be interesting to quantify the lateral movements that occur at different densities. Insight into the amount of lateral movement might further explain the variability in the mass eroded at the low densities.
- Burial rate is just one indicator that could have been used to access organism health, investigation into other parameters as possible indicators of organism health (filtration/feeding rate and deposition rate) would aid the understanding of soft-sediment ecosystems.
- This study found that the contamination gradient in Tamaki Estuary did not affect organism health. If *Austrovenus* were artificially enriched by adding heavy metals to the sediments they are found in, would there be a difference in the burial rate? How high would contamination levels have to get before an effect was seen.
- It would be interesting to link the erodability with other infauna from intertidal sandflats of New Zealand. Combination experiments with other domiant

macrofauna (*Macomona liliana, Paphies australis* etc) and different species assemblages could be used to gain a better understanding on the processes occurring in the soft-sediment ecosystems.

- From this study it would be advantageous to investigate metal resuspension out of contaminated sediment as this is readily available for the uptake by other aquatic organisms.

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