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**The Effects of Physical Habitat
Modification for Mosquito Control,
Runnelling, on Selected Non-Target
Saltmarsh Resources**

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A thesis submitted in fulfilment of the requirements for the degree of Doctor of
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SYNOPSIS

Runnelling is a popular method of physical habitat modification employed on saltmarshes to control pest mosquito populations. The runnelling method involves linking the tidal source to isolated mosquito breeding pools via shallow channels that enable slow water movement of low amplitude tides. Increased tidal flushing inhibits mosquito development. The range of organisms which inhabit saltmarsh are likely to be influenced by altered tidal conditions as they exhibit specific physicochemical requirements for feeding, burrowing or growth. The dynamic nature of saltmarsh may mean that changes to the tidal frequency of a particular region of the saltmarsh promotes extension of marine-like conditions. Because runnels increase the frequency of flooding tidal events in specific regions of the saltmarsh this study predicted that resulting changes would be evident in the physical conditions of saltmarsh substrate, in the transport of buoyant vegetative propagules, in the population characteristics of surface grazing snails and in the density and aperture of crab burrows after flooding and non-flooding tidal events.

The physical impacts of runnelling were determined at three marshes which appeared similar in terms of topography, substrate and tidal conditions. Soil water content and consolidation were measured using two sampling protocols: a) comparisons between modified and unmodified shores; and, b) comparisons with increasing lateral distance across the shore from the runnel edge. At one marsh, moisture levels were significantly higher at runnelled than at unrunnelled sites when tides filled the runnels, but this pattern was not found at the other marshes. Soil consolidation was

greater at higher shore heights, but was not different between runnelled and unrunnelled shores. Measurements at different lateral distances from runnels demonstrated higher moisture levels and lower consolidation up to 5 m from the edge but not further away.

Groups of marked *Avicennia marina* propagules were released at the three runnelled saltmarshes during flooding and non-flooding tidal events. Groups of propagules released within 10 m of a runnel were always transported significantly further from the starting position and further up the saltmarsh shore after both flooding and non-flooding tides than any other groups. In addition, the pattern of stranding on saltmarsh for significantly different groups was closely associated with the path of runnel construction so that propagules were located either in the runnel or in depressions linked to the runnel that had been isolated mosquito-breeding pools prior to runnelling.

It is likely that altered physical soil conditions significantly affected the distribution and size structures of *Salinator solida* and *Ophicardelus* spp. snails recorded at the three saltmarshes. The interaction of tidal period and the presence of a runnel contributed to patterns with significant differences between runnelled and unrunnelled regions of the marsh. Generally, the runnel population of snails exhibited flood-like features even during non-flood periods. The distribution and size classes of snails did not differ with lateral distance from runnels.

The burrow characteristics of the crab *Helograpsus haswellianus* were compared to increase the accuracy of estimating abundance from burrow counts. Including only

those burrows which were obviously maintained by resident crabs significantly increased the confidence limits of estimating crab abundance using only burrow density counts. This method was applied to runnelled and unrunnelled sites to assess any changes in the density of burrows associated with the presence of runnels. Again, it is likely that physical soil conditions resulting from increased tidal frequency at the runnel did influence crab burrowing with fewer small burrows being found at the runnelled site, low on the shore. In addition, mid- and large-sized burrows tended to dominate close to the runnel edge.

Site-specific soil characteristics may help to explain the lack of continuity in patterns associated with runnel effects on non-target saltmarsh resources. While the runnel may increase the soil water content of clayey substrates at some sites it could also result in de-watering of porous sandy soils at other shores. This was evident in the structure of the snail population and distribution of crab burrows which appeared to reflect altered soil physical characteristics associated with the runnel.

Runnelling does affect non-target organisms in saltmarsh. However, the scale of impact was usually locally restricted (< 10 m from the runnel edge). The fact that patterns were not recorded at all sites suggests that the influence of runnels is variable and limited by substrate and some biological conditions. Given the efficiency and popularity of runnelling as a physical control method for reducing pest vector mosquito habitat, this study found no evidence to suggest that its use should be discontinued on any ecological basis measured.

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DECLARATION

The work presented in this thesis has never been submitted as part of a previous degree or diploma in any University. To the best of my knowledge and belief this thesis contains no material previously published or written by another person, except where due reference is made within the thesis itself.

Mark Breitfuss

June 2003

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PUBLICATIONS ARISING FROM THESIS

The following is a list of manuscripts from this Ph.D study accepted for publication or under review. Where appropriate, articles are formatted into this thesis and the relevant stage of publication (published or in press) and chapter is indicated. This thesis is purposely structured as a series of related manuscript-style chapters

Chapter 1:

Breitfuss, M. J. (2001). Predicting the effects of runnelling on non-target saltmarsh resources. *Arbovirus Research in Australia*, 8: 23-29.

Chapter 2:

Breitfuss, M. J. and Connolly, R. M. (in press). Consolidation and water content of coastal saltmarsh substrates following mosquito control habitat modification. *Wetlands Ecology and Management*.

Chapter 3:

Breitfuss, M. J., Connolly, R. M. and Dale, P. E. R. (2003). Mangrove distribution and mosquito control: transport of *Avicennia marina* propagules by mosquito control runnels in southeast Queensland saltmarshes. *Estuarine Coastal and Shelf Science*, 56(3-4): 573-579.

Chapter 5:

Chapman, H. F., Breitfuss, M. J., Dale, P. E. R. and Thomas, P. (in review). Salt-marsh habitat modification for mosquito control and the impacts on grapsid crab populations. *Wetlands Ecology and Management*.

Breitfuss, M. J. (in press). Defining the characteristics of burrows to better estimate abundance of the grapsid crab, *Helograpsus haswellianus* (Decapoda, Grapsidae), in an Australian saltmarsh. *Crustaceana*.

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Chapter 1 Intertidal wetland dynamics: biological and physical processes in Australian saltmarsh landscapes

1.1 Description and occurrence of intertidal wetlands

Intertidal wetland communities occur on sheltered coasts and deltaic river mouths throughout the world and are represented on all of the major continents. The physical distribution of intertidal wetlands on the shore is positively associated with tidal range so that the most expansive areas occur where there is a gradual rise from low-tide level to the land. While greater vertical ranges increase the distribution, on-shore, of intertidal wetlands they do not necessarily result in higher species diversity. This is evident in the dissimilarity in composition of intertidal communities at different latitudes, with some having increased species-richness closer to the equator while others are species-poor (Adam 1990).

Australian intertidal wetlands develop at the dynamic boundary between land and sea on terrestrially or marine-derived sediments with little wave action and periodic tidal inundation. Within these protected intertidal environments vegetation is the prominent biological feature consisting mainly of higher plants adapted to the shifting conditions. Although many animals also inhabit intertidal wetlands they are often cryptic, suited to particular conditions or only intermittently utilise intertidal wetland resources.

The abundance and diversity of flora and fauna reliant on intertidal wetlands is predominantly influenced by hydrological processes (Hughes *et al.* 1998, Le Hir *et al.* 2000, Hussein and Rabenhorst 2001a,b). As the source of substrate moisture can be derived from either salt or freshwater inputs, soil salinity and moisture content can vary with distance landwards or towards the sea. Given the diversity of possible substrate conditions at the land and sea boundary at least three major intertidal wetland formations are recognised – tidal mudflat, mangrove and saltmarsh.

1.1.1 Mudflat

Intertidal mudflats are composed of sediments deposited on the edges of many estuaries and delta systems (O'Brien *et al.* 2000). Dyer *et al.* (2000) recognise three mudflat types based on elevation and tidal influence: lower tidal flats occur from mean low water spring to mean low water neap tide levels; middle tidal flats develop from mean high water neap to mean low water neap levels; and, upper flats are located from mean high water spring to mean high water neap tide levels. In temperate regions, saltmarsh develops on the upper tidal flats and is replaced by mangrove in sub-tropical and tropical climates.

Lower mudflat, and to some extent the middle region, is exposed to almost daily tidal inundation and often strong tidal currents. Terrestrial plants are generally unable to establish on these substrates due to the high energy of tidal changes (Vernberg 1993) which characterise the lower regions of mudflat.

1.1.2 Saltmarsh

In Australia, saltmarsh occurs at the extreme landward margin of the intertidal zone and is completely inundated only by the highest spring high tides, leaving much of the marsh landscape exposed to evaporative processes during lower-amplitude tides and between flooding tides. The area of saltmarsh on the shore is usually bounded by mangroves (if they are present) at the mean high water neap tide level but may extend into mixed scrubland or forest landward of mean high water spring tide levels.

As a result of periodic tidal wetting and evaporative drying episodes, saltmarsh substrates vary in moisture content and salinity through time and space (Daiber 1977). Such variable conditions have resulted in the dominance of species which exhibit both halophytic and xerophytic adaptations. For example, the physical responses of a common saltmarsh grass, *Sporobolus virginicus* (L. Kunth), to changes in substrate salinity and soil moisture content have been tested experimentally and show that the plant adjusts morphologically (Naidoo and Mundree 1992), chemically and metabolically (Gallagher 1979, Donovan and Gallagher 1985) to highly saline and waterlogged conditions.

On the east Australian coast saltmarsh vegetation species richness follows a latitudinal gradient with increased number of species occurring in southern marshes compared to the relatively species-poor saltmarshes in the north of the continent (Adam 1981, 1995). The saltmarsh community is a dynamic one, with subtle changes in sediment accretion or subsidence facilitating changes in the dominance of

species, their eventual loss or ability to colonise newly created areas of suitable habitat.

1.1.3 Mangrove

In areas of the mainland Australian coast with minimal wave action mangrove communities form on the seaward boundary of saltmarsh. The margin between both communities is usually defined by a saltmarsh / mangrove interface where the majority of high tides reach, but do not extend landward except during spring high tide periods.

The intertidal distribution of mangrove is generally defined as being between mean high water and mean sea level and the low density of mature mangrove trees on saltmarsh is due mainly to limited seed dispersal beyond the mangrove zone (Clarke and Hannon 1969, 1970, 1971). Dispersal of buoyant viviparous fruit from the parent plant enables rapid establishment of the already germinated seeds in areas offering suitable growing conditions. The reproductive adaptation of vivipary has enabled mangroves to colonise large areas of the Australian coast and its brackish river systems through persistent transport of fruit via tidal waters.

Mangroves have developed various other physiological and morphological adaptations to survive in saturated and salty environments. Root modifications are obvious in many species, providing various services to the plant such as gas exchange, stability and capturing sediment (Saenger 1982). Physiologically, modified cell structures in leaves and woody tissue enable salt-exclusion, extrusion

or accumulation, making mangroves successful colonists of halophytic conditions (Saenger 1982, Clark and Myerscough 1993).

Mangrove species diversity is almost the inverse of saltmarsh richness with more species (up to about 24) occurring on shores in the north of the continent and only one species, *Avicennia marina* (Forsk), being found from Queensland south to Victoria and South Australia and southwest Western Australia (Galloway 1982). The pioneering capacity of *A. marina* has made it one of the most successful mangrove species in Australia, tolerating a wide range of growth conditions and quickly responding to modifications which favour extension of its intertidal distribution (Clarke and Hannon 1970, Saintilan and Williams 1999).

1.2 The importance of intertidal wetlands

Intertidal wetlands comprise the first terrestrial vegetation communities encountered in the dynamic boundary between sea and land (Adam 1990). Under ideal climatic conditions and where vegetation colonisation occurs at a similar rate to sediment accretion, saltmarsh and mangrove establish. Maintenance of the balance between development and accretion is closely associated with tidal frequency, substrate subsidence and sediment load. However, intertidal wetland communities can begin to degrade when tidal frequency (as a result of sea level rise or habitat modification), subsidence and sediment deposition exceed vegetation colonisation (Stevenson *et al.* 2000). Similarly, vegetation will fail to colonise if tidal frequency does not support adequate sediment loading or if tidal exchange is excessive (Allen 2000).

Many Australian intertidal wetlands are designated Ramsar sites of importance for international migratory birds. This level of significance is often ancillary to federal or state legislation which offers various levels of protection against degradation or for continued sustainable management. A range of other ecologically significant species also inhabit intertidal wetlands either exclusively or as part of their wider preferred habitat. Management authorities attempt to conserve these organisms through the recognition of keystone (Smith *et al.* 1991) or indicator species under regulations which are thought to offer wider levels of protection (see Zann 1996 for additional information). This topic will not be explored further in this thesis.

The physical position of intertidal wetlands means that they act as natural buffers for terrestrial communities from the sea and exhibit features of both the mainland and marine systems. As a result, the balance between biological, hydrological, climatic and geomorphological conditions which help to maintain intertidal communities is limited in tolerance beyond which marine conditions may invade the terrestrial (Donnelly and Bertness 2001). However, natural fluctuations in the intertidal limits of coastal communities means that no static boundaries restrict their distribution and these may change in response to either biotic or abiotic influences (see Saintilan and Williams 1999 and Breitfuss *et al.* 2003).

In addition to their ecological and biodiversity significance, intertidal mangrove and saltmarsh communities are economically important because of their proximity to cities and towns. Coastal development is of exceptional economic importance to state and local government and associated with development proposals is a particular level of planning which must accommodate public health and ecological concerns. This has resulted in a range of controversial issues which are beyond the scope of this chapter but include key areas of public debate.

1.3 Investigation of Australian intertidal saltmarshes

The study of Australian intertidal saltmarshes has developed almost entirely from investigations of vegetative variation in southern Australia (Richardson *et al.* 1998). Clarke and Hannon (1967, 1969, 1970, 1971) and Kratochvil *et al.* (1973) detail the environmental requirements of common mangrove and saltmarsh species in the Sydney district while Hamilton (1919), Patton (1942) and Kirkpatrick and Glasby (1981) present the first major ecological accounts of saltmarsh from Port Jackson, Victoria and Tasmania, respectively. These reports discuss the influence and importance of tidal inundation, substrate drainage and salinity in structuring saltmarsh vegetation.

More recently, intertidal wetland managers and ecologists have attempted to apply modified Northern Hemisphere models to Australian marsh systems, especially in terms of vector control (for disease-transmitting mosquitoes) and habitat modification (Dale and Hulsman 1990, Wolfe 1996). Of particular concern has been the increased demand on local authorities to control pest species in saltmarsh using management programs with minimal environmental impacts (Dale *et al.* 1986). The success of these programs required a thorough knowledge of mosquito developmental conditions (see Russell 1993, 1996) as well as potential impacts on adjoining ecosystems before adequate management protocols could be initiated. As a result, Australian saltmarsh systems received a higher level of scientific scrutiny than had previously been devoted to it albeit below a level of investigation comparable with Northern Hemisphere marshes and sub-tropical mangrove.

Presently, the direction of research concerning Australian saltmarshes has drawn attention to their importance in contributing to commercial fisheries (Morton 1990, Connolly *et al.* 1997, Laegdsgaard and Johnson 2001, Thomas and Connolly 2001) and as buffers to terrestrial and marine systems (Bridgewater 1999). However, in order to maintain a high level of public awareness of these systems future research initiatives should attempt to emulate overseas studies which have attached economic value to saltmarsh integrity (see Bell and Lynne 1997, Kiker and Lynne 1997, Spurgeon 1998, de Jonge 2000).

1.4 Natural biological processes

1.4.1 Vegetation

Saltmarsh landscapes in Australia are generally dominated by salt-tolerant herbs, grasses or low shrubs and are distinguished from other tidal environments, such as mangrove, by the absence of tall woody plants (Chapman 1977). At the national scale, vegetation diversity follows a latitudinal gradient with more than 30 species in the marshes of southern temperate Australia and less than 10 in the northern tropics (Adam 1981). However, within a single marsh the distribution of vegetation is generally influenced by local drainage and elevation features (Adam 1995).

Saltmarshes experience periodic tidal or non-tidal inundation whereas adjoining mangrove systems, if present, are flooded more regularly (Adam 1990). This exposes the saltmarsh to extremes of conditions which are reflected in the vegetation zonation (Adam 1981). From the shore and beyond the mangrove border, low chenopod shrublands of *Sarcocornia quinqueflora* (Bunge ex Ung-Stern) and *Suaeda australis* (R. Br.) Moq. form on soils that are flooded by spring tides and are adapted to the highly saline conditions that result from successive flooding and drying processes (Clarke and Hannon 1970). Towards the landward side of the low shrublands are closed grasslands and salt meadows which, in southeast Queensland, are dominated by *Sporobolus virginicus*. Grasses in this zone are less tolerant to waterlogging and salt than the chenopods and are only inundated by the highest spring tides (Clarke and Hannon 1970, Naidoo and Naidoo 1992). These patterns of

submergence are important features of the biology of saltmarsh species and similar processes are responsible for the formation of saltmarsh in both the southern and northern hemispheres (Adam 1995). However, because the elevation of most Australian saltmarshes precludes them from daily tidal submergence the stress on plants from salinity and moisture fluctuations is much greater than for the *Spartina*-dominated marshes of North America (Adam 1995).

1.4.2 Vertebrate animals

Similar factors responsible for vegetation distribution also influence the faunal composition of saltmarsh. A range of vertebrate animals exploit saltmarsh resources for shelter, feeding, breeding and development. Animal impacts on saltmarsh resources oscillate between two main groups based on use, some utilising sections of a marsh during migrations and others when tidal conditions allow them access. The first group comprise flighted mammals (both megachiropteran and microchiropteran species) and migratory and local bird species which either visit particular marshes or select particular prey on a seasonal basis. Due to the cosmopolitan nature of this group and the fact that much information exists on their biology, protection and habitat conservation (for example see Pizzey and Knight 1997, Amezaga *et al.* 2002, Perez-Arteaga *et al.* 2002) they will not be considered further in this review.

The second group of vertebrates includes individuals which use or inhabit saltmarsh at opposite ends of a hydrological spectrum. The nekton, of which bony fishes comprise the majority, may enter a marsh only when tidal waters exceed the saltmarsh / mangrove interface such as during flooding spring tide events. The

extent of habitat use by these animals is largely restricted because they must also exit the marsh with the ebbing tide (Poulakis *et al.* 2002) unless they are adapted to survive periods of exposure until the next flooding tide. Australian studies report the presence of commercially important species on saltmarsh (Morton *et al.* 1987, Connolly *et al.* 1997, Laegdsgaard and Johnson 2001, Thomas and Connolly 2001). However, little is known of whether these fish contribute to, remove or simply utilise saltmarsh resources during their time on saltmarsh. Greater understanding of this relationship would indeed benefit management efforts for sustainable commercial fishing enterprises as well as highlight the role of saltmarsh in contributing to estuarine health.

Low tides also allow a range of mammalian and reptilian species to exploit specific marsh resources and add to the overall biodiversity of intertidal ecosystems. For example, some north Australian species of mangrove snakes are almost wholly arboreal while mammals such as the water rat, *Hydromys chrysogaster*, forage within much of the intertidal zone. Of particular importance in southeast Queensland saltmarsh is the water mouse, *Xeromys myoides*. This small rodent is a major predator of crustaceans and gastropods which are common on saltmarsh (Van Dyck, 1992, 1996). However, the species is relatively rare and potentially subject to negative impacts from adverse management practices which can reduce either its extensive home range, feeding or nesting requirements.

1.4.3 Invertebrate animals

While the bulk of life on saltmarsh includes a range of species not considered in this review - such as algae, nematodes, molluscs, microcrustacea, bacteria and other micro-organisms (see Christian *et al.* 1978, Sherman and Coull 1980, Richardson and Mulcahy 1996, Sarda *et al.* 1998, Ray 2000) - the three major groups of invertebrates will be discussed briefly below in terms of their importance to saltmarsh biological function. Similarly to vertebrates, the invertebrate fauna of saltmarshes also utilise different marsh resources during periods of either tidal inundation or exposure. Many are able to survive de-watering by escaping to burrows or other subterranean structures which offer moist or humid environments. Similarly, flooding events present an array of additional pressures which species avoid either by climbing above tidal levels or endure because of adaptive respiratory responses.

1.4.3.1 Insects

Insects represent probably the highest species richness of macroinvertebrates which reside permanently on saltmarsh. Many different insect taxa spend either part or their complete life history within saltmarsh, and so are important to saltmarsh process as either predators (Spiller 1984, Sarda *et al.* 1998) or prey (Cameron 1972, Denno 1980, Davis and Grey 1996). In addition, insects are essential for plant pollination and contribute to nutrient recycling of saltmarsh and mangrove biological matter (Minchinton and Dalby-Ball 2001). The distribution of insects on saltmarsh is likely to be influenced by similar structural and climatic factors as vertebrate groups,

so understanding these factors will aid in the overall management and sustainability of saltmarsh biodiversity.

Some insect taxa represent species of either anthropogenic, commercial or conservation significance. In southeast Queensland, Illidge's ant-blue butterfly, *Acrodipsas illidgei* (Lepidoptera, Lycaenidae), is generally restricted to old-growth mangrove habitat or adjoining saltmarsh. This species is classified as endangered under state legislation largely due to its having specific habitat and life-history requirements. Significant changes to the butterfly's habitat may result in local extinction of the species (Beale 1995).

Diptera (flies, mosquitoes and midges) are perhaps the best-studied of saltmarsh insect groups in Australia and the status of some dipterans as vectors of disease is well-understood (see Dale and Hulsman 1990, Russell 1993, Russell 1996). The saltmarsh mosquito, *Ochlerotatus vigilax* (Diptera, Culicidae), is of particular concern for public health and vector control officers as it readily transmits debilitating arboviral diseases to humans, stock and pets (Russell 1993). This species breeds in isolated pools located on saltmarsh and, due to its flight range (up to approximately 50 km), exposes local human habitations to potential disease outbreaks (Russell, 1996). Management efforts have tended to focus on either the disruption or elimination of vector pests via chemical, biological or physical control measures. However, modern approaches must also accommodate low-impact methods of control which are specific to pest species or have minimal non-target effects.

1.4.3.2 Crustaceans and gastropods

By far the most obvious and readily recognised invertebrate groups on saltmarsh are the crustaceans and gastropods. Members of these groups are essential for nutrient recycling on the marsh, converting organic material into base chemicals and reducing organic debris into particulate form (Smith *et al.* 1991, Lee 1998). In addition, these groups are important prey species for vertebrates that utilise marsh resources periodically, such as migratory birds, fish or terrestrial mammals.

The ubiquity and environmental sensitivity of some species make them ideal models for testing impacts of tidal physical forces on population structure and distribution. Recent studies show that these forces physical significantly influence the invertebrate faunal composition of saltmarsh. For example, Richardson *et al.* (1997, 1998) describe the crustacean and molluscan fauna from 65 saltmarshes in Tasmania. For two common species of snails (*Salinator solida* and *Ophicardelus ornata*) and a crab (*Helograpsus haswellianus*) the degree of tidal emergence or submergence, based on physiographical features of the marsh, dictated species distributions (Richardson *et al.* 1998). These species are common on most saltmarshes along the east Australian coast and, in addition to common mangrove and saltmarsh mosquitoes, represent some of Australia's best-studied saltmarsh invertebrates (see Kaly 1988, Roach *et al.* 1989, Roach 1996, 1998, Chapman *et al.* 1998).

The pulmonate gastropods *Salinator solida* and *Ophicardelus* spp. are found in intertidal and terrestrial habitats from exposed mud flats and mangroves low on the shore to areas high in the saltmarsh zone. Arid saltmarsh populations of *S. solida*

and *Ophicardelus* spp. occur at densities of up to 130 and 70 per m², respectively (Morgan and Hailstone 1986) and have a diverse range of size classes with many large (>8mm length) snails (Roach *et al.* 1989, Roach 1998). In contrast, populations from regularly inundated mangrove habitat contain many (up to 800 per m²) small *S. solida* (Morgan and Hailstone 1986), few *Ophicardelus* spp. and a limited range of size classes (Roach *et al.* 1989, Roach 1998). Intermediate sizes occur in environmental conditions between the two extremes (Morgan and Hailstone 1986, Roach *et al.* 1989).

Microhabitat complexity accounts for much of the variation in biotic populations between mangrove and saltmarsh sites and is suspected to provide protection from predation (aquatic and terrestrial), desiccation, and adverse environmental conditions (Morgan and Hailstone 1986, Roach *et al.* 1989). Compared to mangrove, saltmarsh can exhibit greater habitat complexity and so, could afford snails with additional sites to either escape or avoid these pressures.

The grapsid crab *H. haswellianus* occurs on the intertidal saltmarshes of eastern Australia and has been recorded from Queensland, South Australia and Tasmania (Griffin 1966). The crab is nocturnal, euryhaline (0 to 65 ppt tested; Marsh, 1982) and forages throughout the littoral zone on low tides. *H. haswellianus* is common on southeast Queensland marshes at distances of 30 m or greater from the mangrove boundary up into the saltmarsh (Chapman *et al.* 1998) and similar distributions are recorded from Tasmania (Marsh 1982, Richardson *et al.* 1997, 1998). *H. haswellianus* exhibit morphological features characteristic of terrestrial and semi-terrestrial species which include high relative gill volume, tolerance to freshwater

and low salinity and prominent tufts of hair between the ambulatory legs (Griffin 1966). In a study of the ecological distribution of shore crabs Griffin (1971) found *H. haswellianus* individuals higher on the shore than any other species investigated and observed that it constructed burrows in substrate from the upper midlittoral and into the supralittoral fringe.

As with other grapsids *H. haswellianus* dig burrows for protection from predation and desiccation as well as to provide sites for courting, breeding and moulting (Morrisey *et al.* 1999). In Tasmania, the highest densities of *H. haswellianus* burrows are found in well-drained areas high on the shore as the crab actively avoids burrowing in waterlogged substrates (Marsh 1982). Burrows greater than 40 cm in depth reduce the accumulation of toxic substances in saltmarsh substrate by enhancing subsurface soil metabolism and oxidation (Marsh 1982). This feature highlights the significance of crab burrowing activity to the chemistry of intertidal substrates and the availability of anoxic and oxic substances (Williamson *et al.* 1999). Crab burrowing is therefore expected to be critically important in influencing the health of intertidal systems that experience anthropogenic pollution (Williamson *et al.* 1999) and modification (Chapman *et al.* 1998).

1.4.4 Physicochemical

The genesis of saltmarsh from mudflat is controlled by rates of tidal inundation and sediment accretion (Allen 2000, Dyer *et al.* 2000) which operate over variable temporal scales (Belperio *et al.* 2002). Where the frequency of tidal inundation is exacerbated by sea level rise (Stolt and Rabenhorst 1991, Simas *et al.* 2001) impacts

on saltmarsh can manifest as altered community composition or sediment accretion (Vernberg 1993, Dyer *et al.* 2000). Should tidal frequency increase beyond a level which enables the regular accumulation of sediment marine species may invade saltmarsh (Saintilan and Wilton 2001), promoting lower shore sediment and hydrological features.

Saltmarsh substrates are predominantly influenced by two moisture sources, tidal periodicity and rain (including run-off). Rain events are patchy and unpredictable, having minimal influence on soil water variability (Yoo *et al.* 1998), so will not be considered further in this chapter. Alternatively, regular and predictable tidal events are essential for the maintenance of saltmarsh physical processes.

Tidal hydraulic forces are important in determining the substrate moisture characteristics of saltmarsh, with proximity to the tidal source having a positive impact on soil water content (Hughes *et al.* 1998). However, the degree of influence of hydraulic pressures on the availability and transport of sediment water is controlled by flow characteristics of the sediment particles, with sandy soils providing greater movement of sub-surface water than clays (Schultz and Ruppel 2002).

Tidal inundation directly affects substrate salinity, conductivity and pH as well as other factors such as porewater nutrient and dissolved oxygen availability (see Flynn *et al.* 1999, Hussein and Rabenhorst 2001a & b). In Australia, the interaction of tides, biological and physicochemical factors has resulted in extensive deposits of potential acid sulphate soils. The propensity for acid discharge following some

coastal development, agricultural and other disruptive landuse activities is well-understood (Lin *et al.* 1995, Wilson *et al.* 1999, Corfield, 2000) as are plans of mitigation to reduce detrimental impacts (Smith *et al.* 2003).

1.5 Human modification of wetland processes

The focus of this section will be to discuss the various impacts of physical methods of mosquito control on saltmarsh resources. Intertidal wetlands also receive various other impacts from human activities which can compromise ecological integrity and may result in reduced biological diversity.

Coastal development for human habitation and agricultural production is a major issue for land managers in Australia. Much of the debate is based on perceived and measured impacts on biological resources which arise from development or agricultural activities which can damage intertidal vegetation communities. Of particular interest to coastal managers have been impacts on mangrove and saltmarsh communities from housing and resort developments in southeast Queensland. Although this theme will not be considered further in this chapter, Zann (1996) discusses the range of issues.

1.5.1 Habitat modification for pest control

1.5.1.1 Ditching as a physical method of water removal

The physical removal of water from intertidal marshes was one of the first large-scale forms of mosquito control applied to North American systems. In the early 1900's, mechanical parallel grid-ditching offered an immediate and labour-intensive form of dewatering marshes close to human habitation. While the method did offer long-term control of mosquito-breeding habitat, the aesthetic and environmental

degradation of ditched marshes resulted in diminished use of the method in favour of less destructive techniques.

Variations of the parallel ditching method of water removal have also been employed. However, the negative impacts on marsh ecology generally resulted in their decommissioning as primary tools for mosquito control. The physical and biological consequences of ditching methods have been extensively described and reviewed (see Taylor 1937, Travis *et al.* 1954, Shisler and Jobbins 1977, Resh and Balling 1983, Collins *et al.* 1986, Dale and Hulsman 1990, Carlson *et al.* 1991, Wolfe 1996) and will not be considered further in this thesis.

1.5.1.2 Open marsh water management (OMWM)

The mechanism underlying modern habitat modification techniques for pest mosquito control in saltmarsh involves a permanent or long-lasting interference with the conditions required for mosquito survival (Dale and Hulsman 1990). These techniques focus on disrupting one aspect of the mosquito life cycle to bring about control (Hulsman *et al.* 1989) rather than seeking complete removal or destruction of breeding habitats (Wolfe 1996). This concept was first employed by Smith (1907) who promoted the control of mosquitoes by filling breeding depressions and enhancing tidal circulation and predator access to these areas. Later, Ferrigno and Jobbins (1968) developed open marsh water management (OMWM) techniques using Smith's original idea of modifying only the mosquito-breeding sites and not the entire marsh system. OMWM was first applied to eastern seaboard USA to

rehabilitate grid-ditched marshes (Ferrigno and Jobbins 1968) and later in Florida and some northern states to open impoundments to tidal circulation.

In general, OMWM can employ quality ditching and tidal recirculation methods which connect breeding depressions to tidal ditches (Ferrigno and Jobbins 1968) and reservoirs harbouring larval predators (Hruby *et al.* 1985). This increases the frequency of tidal inundation and predation to mosquito-breeding ponds and effectively takes pest production to below threshold levels. The method has been adapted to suit local conditions on both the east (Balling and Resh 1982) and west coasts of North America (Hruby *et al.* 1985) and although the basic philosophy has remained similar both ditch depth and position on the marsh have been modified to control mosquito production with minimal ecological disruption (Resh and Balling 1983).

The success of OMWM in the northern hemisphere led to the development of a system of mosquito control ideally suited to Australian conditions: runnelling. Similar in theory to shallow recirculation ditches (Wolfe 1996) and building on basic OMWM principles, runnelling increases tidal frequency to a marsh and removes surface sheet water from low-lying areas high on the marsh. The physical and operational differences between runnels and OMWM (recirculation) ditches are discussed by Hulsman *et al.* (1989) and relate mainly to the depth, width, shape and position of the structure as well as its capacity to transport tidal water over the marsh substrate to the tidal source. This more minimal form of modification was possible in Australia largely because the saltmarshes had not previously been ditched.

1.5.1.3 Runnelling

Unlike ditches, runnels are shallow (< 30 cm deep) spoon-shaped channels (Fig. 1.1) constructed along natural drainage lines on the saltmarsh to a maximum gradient of 1:1000 (Hulsman *et al.* 1989). A runnel is usually three times wider than it is deep and due to the slight slope it enables slow water movement even during low-amplitude tides.



Figure 1.1 A shallow, spoon-shaped runnel links the saltmarsh/mangrove interface (foreground) to isolated mosquito-breeding pools in the saltmarsh.

Because runnels are linked to the tidal source they promote tidal exchange between graded regions of the marsh so that pools don't form, even after spring tides. The net result is a reduction in mosquito breeding areas, the modification of pools and edges for egg conditioning (the process involving successive flooding and drying events

that prepares mosquito eggs for hatching) and larval development and few apparent negative impacts at the modified site (Hulsman *et al.* 1989, Dale and Hulsman 1990, Dale *et al.* 1993, Latchford 1997).

The physical and hydrological features of runnels are similar to adjoining mangrove habitat even though they may extend well into saltmarsh and above the typical mangrove boundary. Runnels are regularly inundated, have higher mean soil moisture content than surrounding substrate, and exhibit limited microhabitat complexity (Breitfuss 2001).

1.5.2 Impacts on saltmarsh processes

1.5.2.1 Northern Hemisphere

Various forms of habitat modification that utilise OMWM procedures have been advocated for their ability to provide low-impact mosquito control. Unlike the obvious large-scale biological and hydrological effects that grid and drainage ditching had on marsh resources (Daigh *et al.* 1938, Daigh and Stearns 1939, Stearns *et al.* 1940, Bourn and Cottam 1950, Provost 1977) the impacts from OMWM techniques have only been widely studied for lower order consumers and primary producers and as such, require careful interpretation. For example, Shisler and Jobbins (1977) and Ferrigno (1970) reported an increase in marsh productivity following modification. This was substantiated by a recorded progression of the high marsh community to one with more productive low marsh features such as decreased *Melampus bidentatus* (high-marsh snail) density and increased *Uca pugnax* (low-

marsh crab species) holes and vegetation biomass. Based on these results it could also be argued that saltmarsh productivity was restricted by the loss of surface detritivores like *Melampus* which are of considerable importance to recycling of energy and nutrient dynamics (Adam 1990) as well as intertidal food webs (Joyce and Weisberg 1986).

In evaluating disturbance impacts from mosquito control the time-since-construction must be included as an important determinant of variation. Lesser *et al.* (1976) studied saltmarsh biota 40 years after construction of a ditch and reported significantly higher densities of *Melampus* and *Uca* in the treated area compared to the control. This is in direct contrast to Shisler and Jobbins (1977) and Ferrigno (1970) who detected up to a 10-fold reduction in *Melampus* densities at ditched marshes just 3 years after construction. Further evidence of arthropod response to disturbance suggests that some groups return to predisturbance densities within months (Coull 1969) or even hours (Sherman and Coull 1980) of the disturbance episode. Including sites which exhibit different age classes since disturbance would help to explain temporal variation in studies of biotic response to habitat modification.

Resh and Balling (1983) reported little change in the year-round diversity of terrestrial arthropods following habitat modification but they did note seasonal fluctuations. For example, terrestrial arthropod diversity tended to decrease during the wet season but as the marsh became drier diversity increased. They also reported decreased diversity of aquatic invertebrates (Resh and Balling 1983), water table height, groundwater salinity and surface soil salinity (Balling and Resh 1983) but

increased plant diversity, *Salicornia virginica* production (Balling and Resh 1983) and fish diversity and density (Balling *et al.* 1979, Taylor *et al.* 1998, Poulakis *et al.* 2002) following modification.

1.5.2.2 Southern Hemisphere

Compared to similar methods of mosquito control, reviewed extensively for the northern hemisphere (see reviews by Dale and Hulsman 1990, Carlson *et al.* 1991 and Wolfe 1996), the effects of methods such as runnelling have received only limited study in the southern hemisphere, showing limited broad impacts (Dale *et al.* 1996) and few negative effects on vegetation and physical characteristics (Latchford 1997, Dale and Dale 2002). This is likely because runnelling is a relatively new concept for control with the oldest functioning runnels constructed less than 18 years ago. In addition, the lack of autecological and physico-chemical data concerning Australian saltmarsh systems and species responses to environmental factors (Richardson *et al.* 1998) has forced authors to interpret results with caution (see Latchford 1997).

Saltmarsh biota are adapted to conditions which result from successive flooding and drying processes, exhibiting positive or negative responses to changes in these conditions. In general, the effects of runnelling (see Latchford 1997) appear to be similar to North American studies which identified subtle biological and physico-chemical differences between modified and control sites (see Resh and Balling 1979, Balling and Resh 1982, 1983). These studies report variations in marsh hydrology which accounted for differences and subsequently influenced substrate salinity, soil moisture content as well as vegetation and fauna density (Balling and Resh 1983).

1.6 Concluding remarks and future directions

Australian saltmarsh landscapes are biologically, physically and economically important to other adjoining communities, especially in terms of maintaining ecological integrity. Historically, Australian saltmarshes have received limited scientific attention other than for understanding the biology of vector pests and their control. More recently, coastal managers and investigators have attempted to orient the direction of research to encompass the range of issues likely to result from mosquito-control activities in saltmarsh, as well as promoting the importance of saltmarsh to commercial fishing interests. In employing a multi-lateral approach to the study of Australian saltmarsh systems will be the provision of a basis for understanding the basic biological and physical processes underlying mosquito-control. Further, this understanding will enable effective plans of mitigation to be achieved so that potential economic and ecological impacts can be avoided or, at least, managed.

Based on Northern Hemisphere models of tidal manipulation, runnelling is a form of mosquito-control which disrupts pest life history by increasing tidal flushing of breeding habitat. Runnelling is an effective, efficient and popular method of physical habitat modification which involves linking the tidal source to isolated mosquito-breeding pools which are usually located high on the saltmarsh shore. The depth of runnels enables slow-water movement of low-amplitude tides which would normally fail to breach saltmarsh, thereby increasing the frequency of flooding tidal events to higher shore elevations. In targeting specific regions of the saltmarsh through modified tidal cycles, the runnelling technique is likely to result in:

- enhanced effects of some hydrological processes which are measurable as localised, on-site, physical impacts;
- facilitated transport and deposition of water-borne vegetation; and,
- altered faunal assemblages reflecting altered substrate conditions.

This thesis tests these predictions with individual chapters examining particular hypotheses.

Presently, the extent of specific impacts from runnelling activities is largely unknown but is believed to result from changes in the hydrological and physicochemical properties of saltmarsh substrate. Given the popularity and effectiveness of the method for controlling pest mosquito populations in saltmarsh, a broader understanding of the ecological ramifications of the technique is imperative for sustained management of intertidal ecosystems.

Chapter 2 Impact of runnelling on consolidation and water content of saltmarsh soils

2.1 Introduction

Intertidal ecosystems occur at the dynamic boundary between land and sea on all continental margins. Where coastlines are protected from direct wave action, saltmarsh can form the dominant plant community, exhibiting both terrestrial and marine characteristics. However, because saltmarsh occurs within the intertidal zone, the structuring forces responsible for its genesis and sustained development are marine-based and linked to rates of sedimentation and tidal frequency (Hughes *et al.* 1998, Le Hir 2000, Hussein and Rabenhorst 2001a,b).

Soil water availability has been investigated as one of the main abiotic differences between terrestrial and marine systems because of its capacity to structure ecological communities (Le Hir 2000). In terrestrial systems, soil moisture variability is driven by rain events and influenced by topographical (Aucan and Ridd 2000, Qiu *et al.* 2001, Shaman *et al.* 2002) and physical (Yoo *et al.* 1998, Yoo 2001, Schultz and Ruppel 2002) conditions. Marine systems such as saltmarsh are also influenced by topographical and physical factors but soil moisture is generally driven by tidal rather than rain events (Adam, 1990).

In recent studies of saltmarsh physical processes, analyses of spatial and temporal patterns of tidally-derived soil moisture have identified its importance for tidal marsh

management and restoration. This is largely because of the relationship between substrate condition, tidal frequency and some biological processes (Flynn *et al.* 1999, de Jonge 2000). The frequency of tidal inundation of saltmarsh is primarily responsible for the distributions of epifaunal and infaunal species because it limits the availability of suitable habitat (Sagasti *et al.* 2001) and affects soil chemical factors (Hussein and Rabenhorst, 2001b). Where the frequency of tidal inundation of saltmarsh is increased by sea level rise (Stolt and Rabenhorst 1991, Simas *et al.* 2001) or human habitat modification (Breitfuss 2001, Breitfuss *et al.* in press), the impact on intertidal ecosystems is caused by the degree of submergence experienced, resulting in altered community composition or sediment accretion rates (Vernberg, 1993, Dyer *et al.* 2000). The investigation of impacts from altered tidal regimes on ecological systems is complex and difficult; simple, non-destructive measurement of soil moisture and soil mechanics (e.g. consolidation) may provide a basis for assessing a wider range of substrate conditions (Avnimelech *et al.* 2001, Zhang *et al.* 2001) and act as a proxy measure of ecological effects.

In this chapter the influence of altered tidal frequency, from runnel habitat modification for mosquito control, is examined in terms of its impact on soil moisture and soil consolidation. Runnels are a form of mosquito control which function by providing tidal access to isolated mosquito-breeding pools high on the marsh via shallow (<30 cm deep), spoon-shaped, channels that link the tidal source to graded shore regions of the saltmarsh (Dale and Hulsman, 1990). Because runnels transport low-amplitude tides which would otherwise fail to flood the greater saltmarsh area (Breitfuss *et al.* in press), we expect soil moisture and soil

consolidation to differ between the runnelled and unrunnelled areas as well as at specific distances from the runnel across the shore.

2.2 Materials and Methods

2.2.1 Study site

Three saltmarshes (Coomera, Tingalpa 1 and Tingalpa 2) within Moreton Bay, Queensland, Australia ($153^{\circ}15'E$, $27^{\circ}35'S$), (Fig. 2.1) were sampled between March 2000 and December 2001. All shores had a similar tidal range (approx. 2.5 m) and height required for complete inundation (>2.45 m).

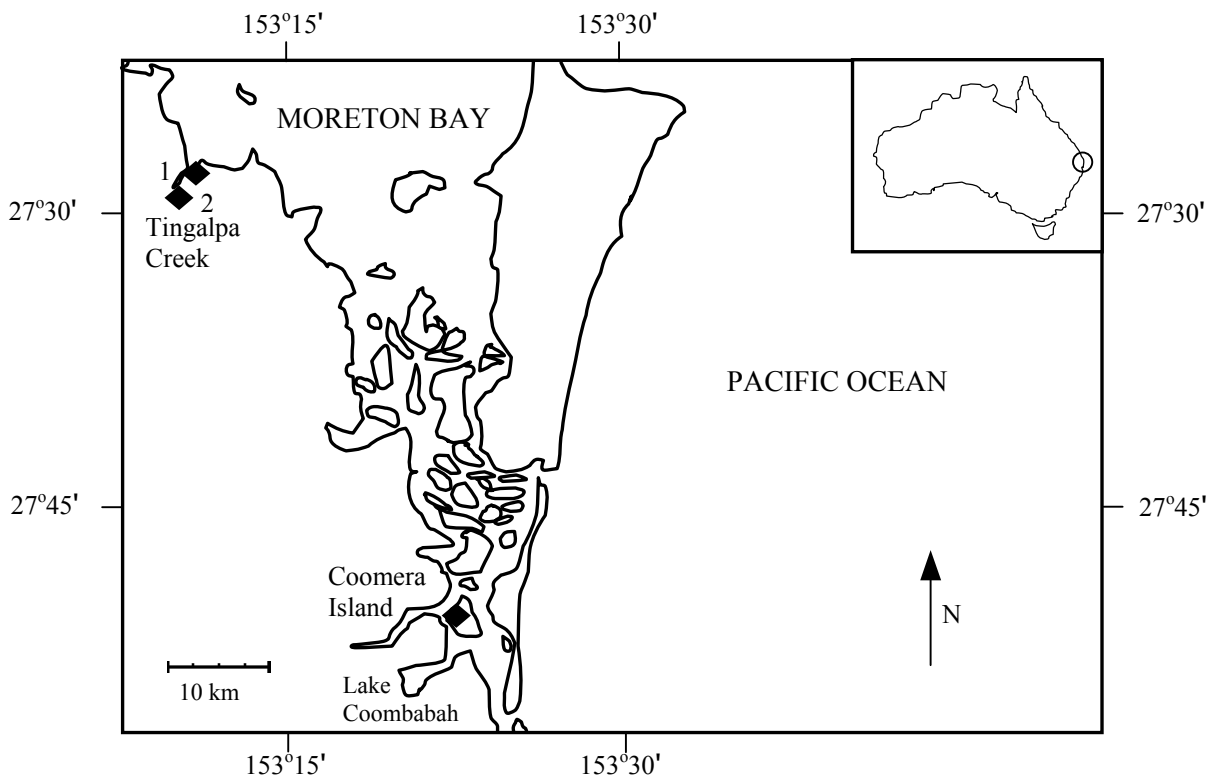


Figure 2.1 Location of saltmarsh sites (black diamonds) in southern Moreton Bay.

Sites had been runnelled for mosquito control (Coomera 1985, Tingalpa 1 2001, Tingalpa 2 1996) according to local drainage and topographic features (Owtrim and Dixon 2001), with the runnel beginning at the saltmarsh / mangrove interface low on

the shore and extending up the shore into the saltmarsh. At all sites, the uppermost layer of substratum (<9 cm deep) was of primary interest, being dominated by solonchaks which exhibit little profile development, are strongly saline and have a loamy / clayey texture (Beckmann *et al.* 1987). Previous studies found little variation at these sites in terms of particle size (Law 1981) and no significant differences in samples were detected for pH, salinity and relative substrate moisture content (Dale and Hulsman 1990) at distances of between 10 and 50 m from the tidal source, up the saltmarsh shore. The presence of a fine, superficial silt layer (<1 cm deep) may reflect the low-energy nature of the sites in terms of sediment trapping and tidal flushing (Law 1981, Mwamba and Torres 2002).

Mosaic of salt-tolerant plants (primarily *Sporobolus virginicus* and *Sarcocornia quinqueflora*) dominate the vegetation and develop on the thin silt deposits overlying solonchaks. On the seaward side of saltmarsh, *Avicennia marina* forms a dominant structural component at the saltmarsh / mangrove interface.

2.2.2 Physical measurements

Soil moisture was measured with a ThetaProbe ML2x (Delta-T Devices, Cambridge) after a two-point specific soil calibration. Soil consolidation was measured using a Torvane shear strength device where strength (in kg/cm²) was the force required to break a 25 cm² section of the soil surface to 1 cm depth. Physical measurements were taken daily during the spring tide period (3-4 days) approximately 1 hr after the predicted (Queensland Department of Transport, 2001) highest high tide, either a non-flood (2.25 - 2.38 m) or flood (2.42 - 2.56 m) event. Non-flood tides account for

~16% of tidal events annually and reach only the saltmarsh / mangrove interface while flood events account for ~7% of tides and cover the entire saltmarsh.

2.2.3 Sampling protocols

For the first sampling protocol, two 50 m (continuous 10 x 10 m plots) transects were established, one along the runnel and the other 20 m from and parallel to the runnel (Fig. 2.2). Within each plot, five randomly selected quadrats were selected and five soil moisture and one shear strength measurements were taken from each quadrat. The beginning of the transects were set on the inland edge of the saltmarsh at the saltmarsh / mangrove interface.

For the second sampling protocol (Fig. 2.2), transects were located perpendicular to the runnel at distances of 0, 30 and 50 m up the shore, from the inland edge of the saltmarsh. Sampling quadrats were situated along each transect at nine distances (from 0 – 20 m) from the runnel edge in both directions (Fig. 2.2). Five soil moisture and one shear strength measurements were taken within each quadrat.

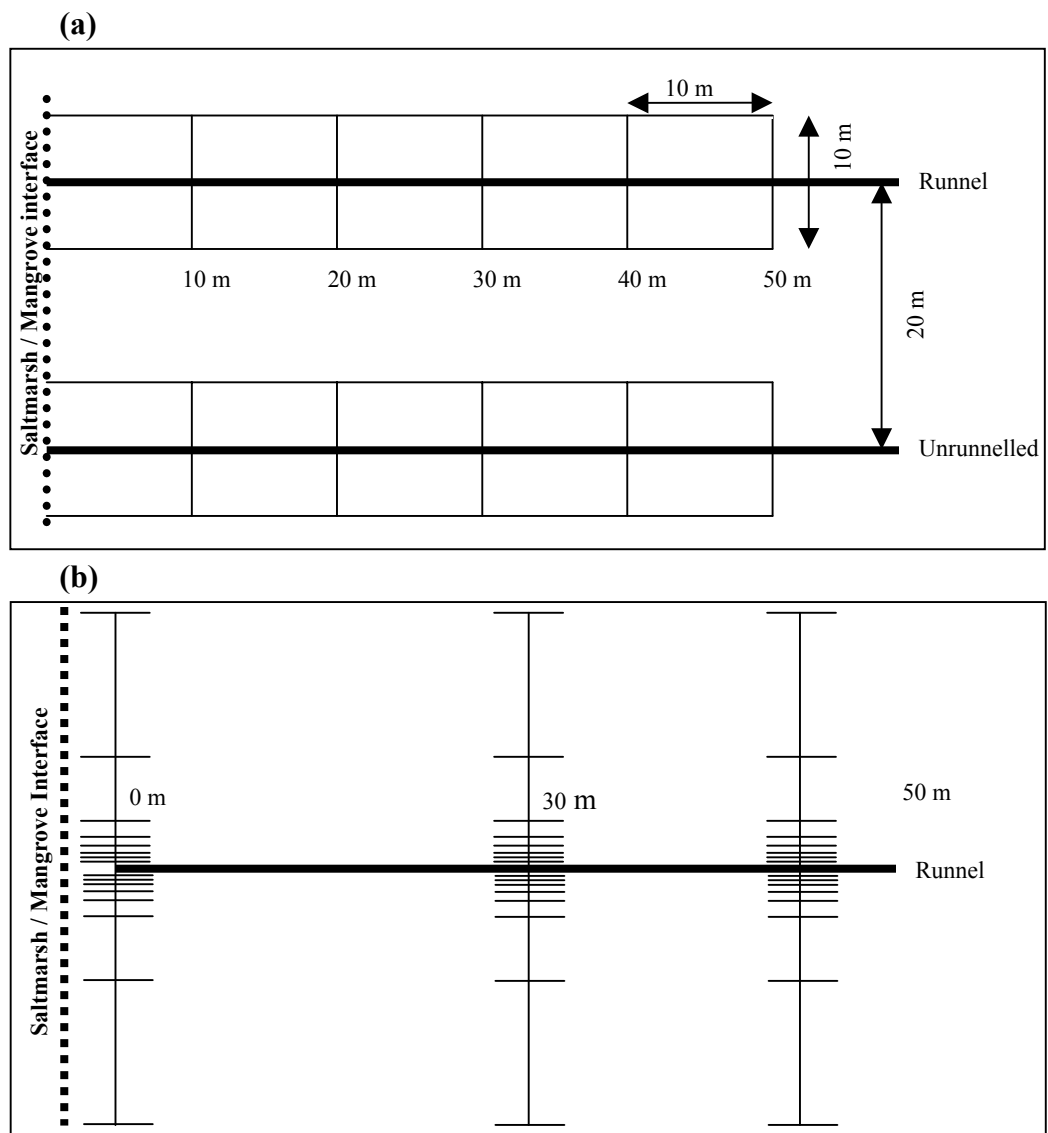


Figure 2.2 Sampling design and plot dimensions for **(a)** first sampling protocol comparing runnelled and unrunnelled transects & **(b)** second sampling protocol showing lateral spread of sample points (0.5, 1, 1.5, 2, 3, 5, 10 & 20 m) from the runnel edge at three shore heights.

2.2.4 Data analysis

Soil moisture and shear strength data from the first sampling protocol (hereafter called “runnelled versus unrunnelled: non-flood and flood”) were analysed with 3-way ANOVA where tide (flooding, non-flooding), distance from shore and treatment

(runnelled or unrunnelled) were fixed factors. Data were tested for departures from the homogeneity of variance assumption; no transformation was required in this case. Measurements from the second protocol (hereafter called “lateral distance from the runnel edge: non-flood and flood”) were analysed with 3-way ANOVA where tide, distance from shore and the saltmarsh / mangrove interface (hereafter called “shore height”), distance from runnel edge and treatment (runnelled or unrunnelled) were fixed factors. Data from both sides of the runnel were pooled so that the analysis concentrated on consistent differences with distance from the runnel. Pooled data did not require transformation prior to ANOVA. Tukey tests (HSD) were used to identify differences among means. Data from the three marshes were analysed separately.

2.3 Results

2.3.1 Runnelled versus unrunnelled: non-flood and flood

Runnelling affected soil moisture and shear strength but not at all three marshes, and other factors also affected these soil properties. At Coomera, soil moisture values at the runnelled transect after a non-flooding tide were as high as those after a flooding tide, whereas at the unrunnelled transect values were lower after the non-flooding tide (Fig. 2.3, and significant interaction between tide and treatment in Table 2.1).

Table 2.1 Summary of three-way ANOVA results for soil moisture and shear strength at runnelled and unrunnelled transects. Only significant results are shown. Tukey results are shown where significant factor has greater than two levels.

Marsh	Factor	df	P	Tukey
<i>Soil Moisture</i>				
Coomera	tide	1	***	
	treatment	1	*	
	treatment*tide	1	*	
Tingalpa 1	tide	1	*	
<i>Shear Strength</i>				
Coomera	tide	1	***	
	distance	4	*	10 < 50
Tingalpa 1	distance	4	*	10 < 50

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$

This supports observations by Dale *et al.* (1996) of reported increased soil moisture in and around runnels during tidal events that would not normally flood saltmarsh. Soil moisture values at Tingalpa 1 were higher after flooding tides at runnelled and unrunnelled transects, but there was no interaction between tide and treatment, indicating that the runnel did not affect soil moisture. Soil moisture values at Tingalpa 2 were not significantly affected by treatment, tide or distance up the shore.

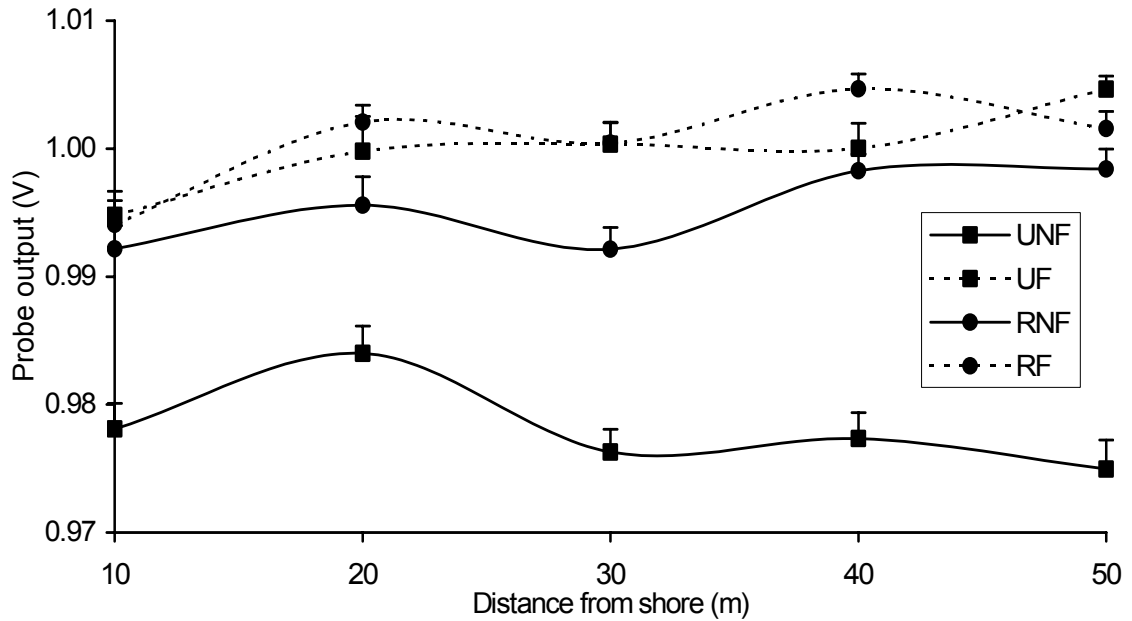


Figure 2.3 Mean soil moisture (+ SE) at Coomera saltmarsh along runnelled (R) and unrunnelled (U) transects after flooding (F) and non-flooding (NF) tides.

Shear strength values at Coomera were higher after non-flooding than flooding tides, and increased with distance up the shore (Fig. 2.4, Table 2.1), but there was no significant effect of treatment nor any interaction between tide and treatment. Shear strength values also increased with distance up the shore at Tingalpa 1 but were not significantly affected by any other factor. Shear strength values at Tingalpa 2 were not significantly affected by any factors.

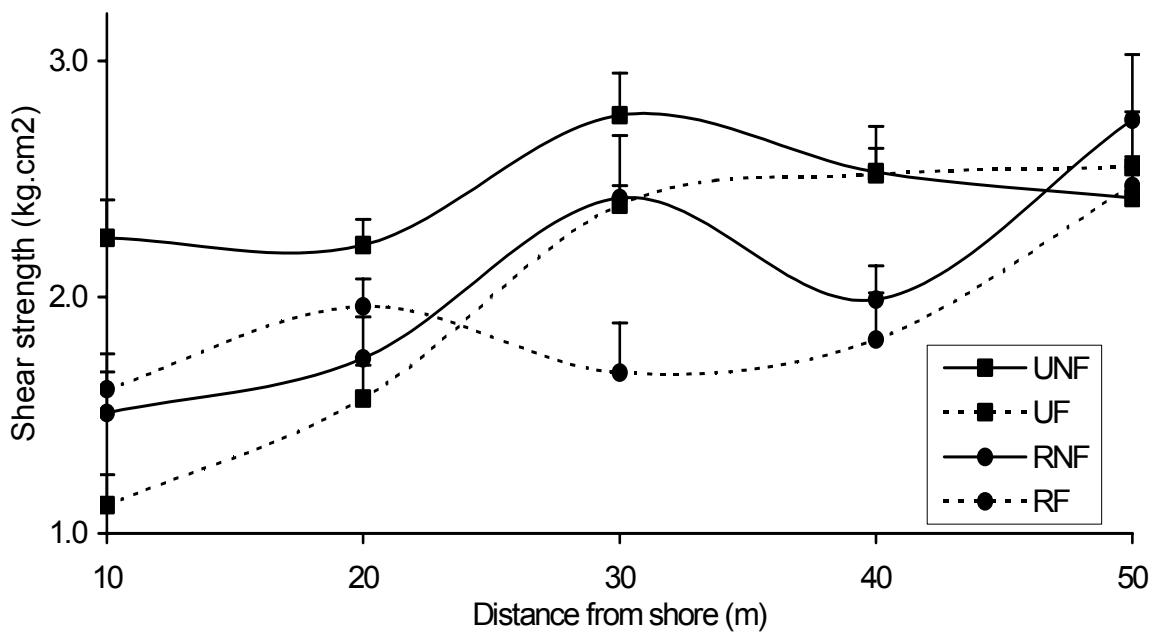


Figure 2.4 Mean shear strength (+ SE) at Coomera saltmarsh along runnelled (R) and unrunnelled (U) transects after flooding (F) and non-flooding (NF) tides.

2.3.2 Lateral distance from runnel edge: non-flood and flood

Runnelling influenced soil moisture and shear strength but not at all sites, shore heights or distances from the runnel edge. At Tingalpa 2, at 50 m up the shore, soil moisture further than 5 m from the runnel edge tended to be lower after non-flooding tides whereas within 5 m of the runnel soil moisture was as high as after flooding tides (Fig. 2.5). At Tingalpa 1, soil moisture was significantly higher after flood events over the three shore heights (Table 2.2). This pattern was not found at either Coomera or Tingalpa 2 which both recorded differences in soil moisture between shore heights, rather than tidal events (Table 2.2). For Coomera, soil moisture measurements 30 m up the shore were significantly higher than the 0 and 50 m levels. At Tingalpa 2, soil moisture from 0 m shore height was significantly higher than either the 30 or 50 m positions.

Table 2.2 Summary of three-way ANOVA results for soil moisture and shear strength at distances from the runnel edge. Only significant results are shown. Tukey results are shown where significant factor has greater than two levels.

Marsh	Factor	df	P	Tukey
<i>Soil Moisture</i>				
Coomera	shore height	2	*	30 > 0, 50
Tingalpa 1	tide	1	***	
Tingalpa 2	shore height	2	***	0 < 30, 50
<i>Shear Strength</i>				
Coomera	shore height	2	***	0 < 30, 50
Tingalpa 1	shore height	2	***	0 < 30, 50
Tingalpa 2	shore height	2	***	0 < 30 < 50
	distance from runnel	7	**	0.5, 5.0 < 10, 20

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$

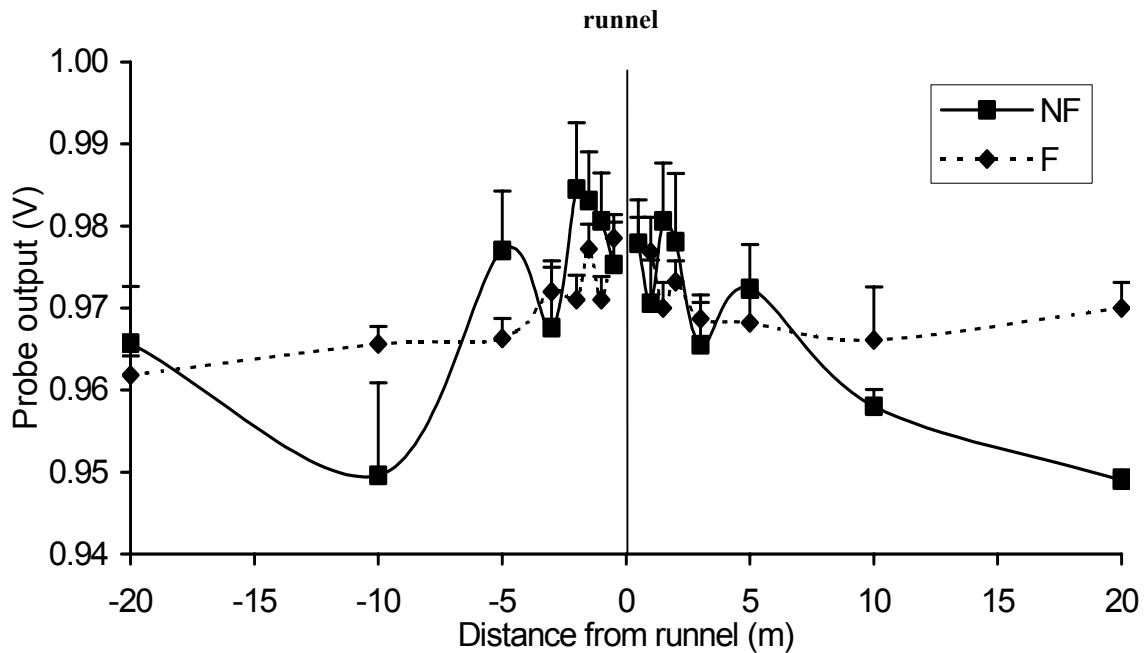


Figure 2.5 Mean soil moisture (+ SE) at Tingalpa 2 saltmarsh along lateral transects, radiating from a runnel, at 50 m shore height after flooding (F) and non-flooding (NF) tides.

Tidal inundation influenced soil shear strength measures, with significantly higher levels of consolidation after non-flooding tides at Coomera than after flooding events (Table 2.2).

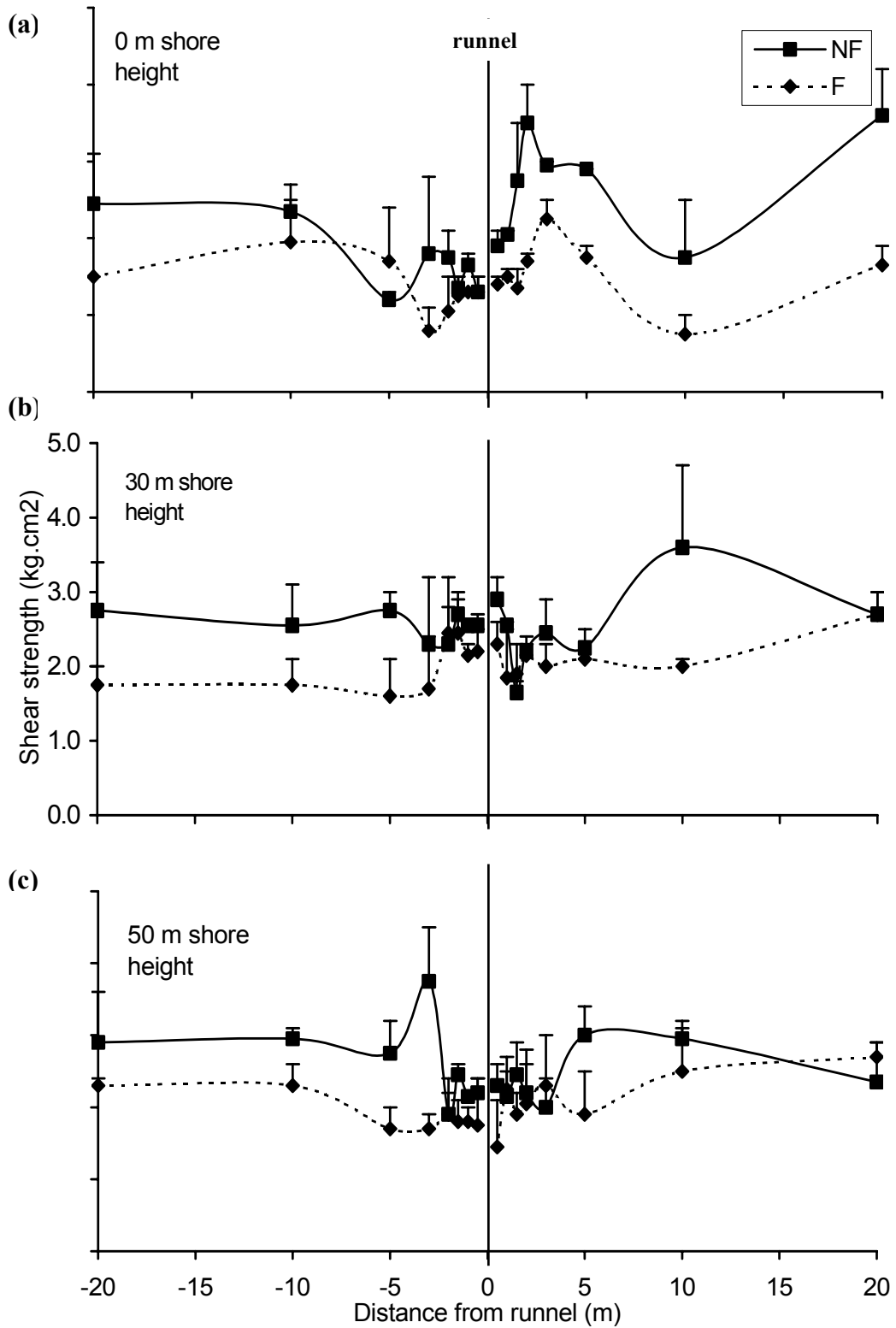


Figure 2.6 Mean shear strength (+ SE) at Coomera saltmarsh along lateral transects, radiating from a runnel, at three shore heights after flooding (F) and non-flooding (NF) tides (a) 0, (b) 30 & (c) 50 m from shore.

Specifically, shear strength at all shore heights was higher at distances greater than 5 m from the runnel edge after non-flooding tides, but were as low as those after flooding tides within 5 m of the edge (Fig. 2.6). At Tingalpa 2, shear strength values differed significantly (Table 2.2) between 0.5 and 5.0 m and those taken further (10 and 20 m) from the runnel.

2.4 Discussion

The most striking anomaly, illustrated by data from the Coomera marsh, is that soil consolidation increased with distance up the shore but soil moisture did not. This lack of relationship between soil moisture and consolidation conflicts with results of other studies that examined similar soil properties (Nearing *et al* 1991, Zhang *et al.* 2001) and found that soil moisture directly affects consolidation. One explanation for the lack of relationship between soil moisture and consolidation in the marshes studied here is biological consolidation. Vegetation on the Coomera and Tingalpa 1 saltmarshes is dominated by grasses high on the marsh and halophytic herbs in lower parts. Shear strength measurements could be influenced by additional soil resistance attributable to higher root-mat densities of the fibrous-rooted grasses (Ringold 1979) higher on the shore. The influence of substrate biological factors on soil consolidation could be further confounded by built-up detritus layers in densely vegetated higher marsh sites (Smith-White 1988). Furthermore, these saltmarsh plant communities themselves are almost certainly influenced by soil conditions (Clarke and Hannon 1967, Gallagher 1979).

Alternatively, the lack of relationship between soil moisture and consolidation may be explained by physical processes which regulate tidal movement via surface and sub-surface mechanisms. Tidal forcing is a physical action of hydraulic head force through estuarine sediments when tides rise and results in surface soil saturation (Howes and Goehringer 1994). Hughes *et al.* (1998) analysed aquifer response to tidal forcing in saltmarsh wetlands of the lower Hunter River estuary in Australia and

observed uniform forcing of porewater through saturated sediments over the length of a 36 m transect. At the three marshes surveyed in this study, little variation in soil moisture was detected over 50 m runnelled and unrunnelled transects after flooding tides. However, soil moisture was significantly lower along the unrunnelled transect at Coomera after non-flooding tides. In this case the runnel may provide additional moisture resources for the hydraulic head, resulting in “flood-like” moisture levels. However, this process was evidently not facilitated by runnels at Tingalpa 1 or Tingalpa 2 marshes, highlighting the variability of runnelling effects among different marshes.

In addition to the sub-surface hydraulic forces and biological factors above, tidal asymmetry and shore slope may provide further explanation for the variation in soil moisture and shear strength measured at the three sites. Aucan and Ridd (2000) examined flood and ebb tide movements across mangrove and saltmarsh systems in north Queensland, Australia. They found that the slope of tidal currents exiting up-shore areas, relative to the slope of the marsh surface, was an important determinant of water movement. In systems dominated by a higher water slope than surface slope (as in systems with large expanses of flat saltmarsh bordering mangrove), drying would occur more rapidly closer to the tidal source. This causes large amounts of water to become perched high on the marsh on the ebbing tide, restricting drainage to small channels (Aucan and Ridd 2000). The degree to which ebbing tidal waters perch depends on the slope characteristics of the marsh, and the effects of runnelling therefore might also vary among marshes with different slopes. For example, runnels aid transport of ebb tides from Coomera marsh, so soil moisture is

higher closer to the runnel edge than beyond during both flooding and non-flooding tides. A similar mechanism may also occur at Tingalpa 2 where shear strength close to the runnel was always different (sometimes higher, in other cases lower) than at distances greater than 5 m. The relationship between soil moisture and consolidation expected from previous work (Nearing *et al.* 1991, Zhang *et al.* 2001) was more evident in data from the second sampling protocol than the first. This might be an indication that close to the runnel (within 5 m), where the runnel is having a large impact, the effects of soil moisture on consolidation are greater than other factors (biological or physical) that are more important away from runnels.

Patterns of soil moisture and shear strength varied between the marshes, which may reflect substrate responses to tidal events and the presence of runnels. At the Coomera marsh, the runnel had the effect of increasing wetting on non-flooding tides compared to the levels measured after flooding tides, but this did not occur at other marshes. The effects of runnelling on both soil moisture content and shear strength were localised, affecting the soil properties for up to 5 m either side of the runnel but having little impact beyond that distance.

In general, the impacts of runnels were site-specific with no general trends apparent at all of the marshes. In this study we expected similar impacts from habitat modification at saltmarshes selected on the basis of *a priori* topographical and tidal characteristics. However, given the variability in patterns recorded, even at unrunnelled transects, authorities concerned with managing impacts from mosquito control should base remedial strategies on dynamic saltmarsh classifications (see Chapter 6).

Chapter 3 Patterns of runnel-facilitated transport of *Avicennia marina* propagules

3.1 Introduction

Intertidal mangrove and saltmarsh communities experience similar hydro-edaphic conditions (Patterson *et al.* 1997) but are distributed across different intertidal and latitudinal gradients (Adam 1994). When representative species from both communities overlap in geographical range, physical adaptations to environmental stresses (such as tidal periodicity and amplitude) often restrict their distribution to specific intertidal limits (Saenger *et al.* 1977, Adam 1994, Naidoo *et al.* 1997, Mazda *et al.* 1999).

The grey mangrove, *Avicennia marina* var. *australasica* (Walp.), is the only variety found in eastern Australia (Clarke 1993) and is common on the Queensland coast. The intertidal distribution of mature *A. marina* is between mean high water and mean sea level (Clarke and Myerscough 1993) and the low density of *A. marina* on saltmarsh is due mainly to limited seed dispersal beyond the mangrove zone (Clarke and Hannon 1969, 1970, 1971). *A. marina* seeds may develop from mid-summer to autumn and fall in early summer. Seeds are produced within large cryptoviviparous fruit (Hutchings and Saenger 1987, Clarke 1993), the embryo encased in a protective pericarp. Transport of *A. marina* propagules is determined, in part, by buoyancy properties of the pericarp and the embryo. Fruit with pericarp intact can float in

seawater for up to 24 hr before the pericarp is shed, at which time fruit sink for periods of at least 48 hr before refloating (Clarke and Myerscough 1991).

A. marina propagules are usually transported by seawater, and their capacity for dispersal from the parent plant is dependent on tidal frequency and amplitude (Clarke and Myerscough 1993). Propagules often strand and establish close to the parent plant population (Clarke 1993) but are occasionally transported by the highest spring tides beyond the mangrove zone to saltmarsh on the landward side or mud flats on the seaward side (Clarke and Hannon 1969). In addition, some land use practices in saltmarsh may create favourable conditions for the landward colonisation of *A. marina* by altering chemical (McTainsh *et al.* 1986, Anwahi *et al.* 1998) or physical (Dale *et al.* 1989) conditions that facilitate mangrove seedling establishment.

In this chapter, the role of mosquito-control runnels in transporting and depositing mangrove propagules onto saltmarsh is explored. Propagules of *A. marina* are usually transported and deposited within similar intertidal limits as the parent plants and are seldom carried to saltmarsh (Clarke and Myerscough 1991). This might be explained by limited dispersal mechanisms for propagules to reach favourable areas of saltmarsh above their usual intertidal limit. If runnelling increases the frequency of tidal inundation of saltmarsh, it may also provide an alternate dispersal method for buoyant propagules to be deposited onto saltmarsh.

3.2 Materials and Methods

3.2.1 Study site

Propagule release experiments were conducted at three saltmarshes (hereafter called “sites”) in southern Moreton Bay, southeast Queensland, Australia (see Chapter 2, Fig. 2.1) between August and November 2000. Criteria for the selection of sites in the experiment included the presence of mosquito control runnels (Hulsman *et al.* 1989), fruiting *A. marina* and a clearly defined intertidal boundary (Clarke and Myerscough 1993) between *A. marina* and saltmarsh (hereafter called the “saltmarsh-mangrove interface”). The three sites experienced a tidal range of approximately 2.5 m and had a relief of about 0.5 m.

The extent of saltmarsh at the sites was bounded by mangroves at the mean high water neap tide level and extended into mixed scrubland or forest beyond mean high water spring tide levels. From the landward limit of the saltmarsh and above the mangrove border (see Fig. 3.1), low chenopod shrublands of *Sarcocornia quinqueflora* and *Suaeda australis* were formed on soils flooded by spring tides. Closed grasslands and salt meadows dominated by *Sporobolus virginicus* occurred towards the landward side of the low shrublands and were on substrate only flooded by the highest spring tides. The patterns of tidal submergence and exposure at the three saltmarsh sites are important features of the biology of endemic species and similar processes are responsible for the formation of saltmarsh in both the southern and northern hemispheres (Adam 1990). However, because the elevation of most

Australian saltmarshes precludes them from daily tidal submergence the stress on biota from salinity and moisture fluctuations is much greater than for the *Spartina*-dominated marshes of North America (Adam 1995).

Propagules of *A. marina* were collected from a single mangrove population at the northern shore of Lake Coombabah (see Fig. 2.1). Fruit were collected from randomly chosen *A. marina* by shaking trees and gathering fallen propagules or by selecting material that appeared to be fresh amongst already fallen debris. All collected fruits were sorted and those with split or damaged pericarps were discarded.

3.2.2 Pericarp loss experiment

Branding the pericarp to identify propagules for mark and recovery experiments can interfere with pericarp retention (Clarke 1993), and this could affect manipulative experiments investigating transport of *A. marina* propagules. Clarke (1993) also reported reduced buoyancy and interference with retention of the pericarp following physical labeling with fish tags, incisions and pen marks. Given Clarke's (1993) observations it was important, in this experiment, to establish the probability of marked propagules retaining their pericarps for at least one tidal phase following release, since pericarp retention / loss affects buoyancy (Clarke 1993). Twenty propagules were randomly selected to test the effects of marking with a permanent pen on propagule buoyancy.

Ten propagules were given a numerical mark on each side of the pericarp using waterproof permanent felt pens, and 10 propagules remained unmarked. All propagules were floated in an artificial seawater solution and monitored for a total of 12 hr to record pericarp loss. Observations were made hourly for the first 6 hr and then every second hour. A period of 12 hr was similar to the time that marked propagules would remain under the influence of one tidal phase in the field.

The proportion of propagules having lost their pericarp was not significantly different in the control and marked treatments ($\chi^2 = 0.208$; $df = 1$; $P > 0.05$). Propagules could therefore be marked with permanent felt pens in the release experiment without concern for effects on pericarp retention and buoyancy over one tidal event.

3.2.3 Propagule transport experiment

The location of runnels on the saltmarsh determined the positioning and size of the experimental sites. The experimental site, including the runnel, was homogenous in terms of vegetation, elevation and topographic features. In addition, prior knowledge of the pattern of tidal inundation aided the selection of “starting” positions for groups of propagules which would receive tidal waters at the same time.

At each site, five groups of 15 marked propagules were placed at 10 m intervals along the tidal inlet and within 1m of the saltmarsh side of the saltmarsh-mangrove interface (Fig. 3.1). However, at Tingalpa Site 2, four groups of 14 and one group of 13 propagules were released.

The physical structure of the starting positions on the saltmarsh-mangrove interface was similar, being dominated by an exposed mud / sandy shore with few *A. marina* pneumatophores and little herbaceous vegetation (< 2% foliage projected cover of either *Sporobolus virginicus* or *Sarcocornia quinqueflora* estimated within 100 m²). The structure of the exposed mud / sandy shore was a typical feature and generally occurred in a band of between 2 and 10 m wide where the saltmarsh-mangrove interface was clearly developed. The zone was also relevant as a starting point for fruit release since propagules from parent *A. marina* trees naturally fell in it.

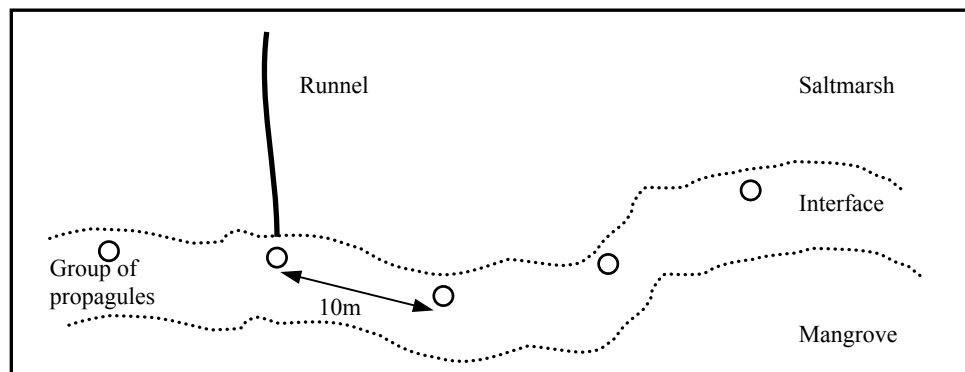


Figure 3.1 Hypothetical starting positions for five groups of marked propagules. Starting positions are situated relative to the saltmarsh-mangrove interface, the runnel and each other.

Groups of propagules were placed at the starting positions approximately 6 hr prior to onset of the predicted (Queensland Department of Transport 2001) highest high tide for a particular day that was expected to be either a non-flood (2.25 - 2.38 m) or flood (2.42 - 2.56 m) event. Non-flood tides in the range of 2.25-2.38 m accounted for approximately 16% of annual tidal events and reached the saltmarsh-mangrove interface but did not inundate the saltmarsh. Tidal flood events accounted for approximately 7% of annual tides and breached the saltmarsh, extending at least 50 m up the shore onto the saltmarsh. At each of the three sites, propagules were

released during flood and non-flood predicted high tides. The flood and non-flood tidal levels occurred during fruit abscission of *A. marina* and were included in the experiment to test the consistency and pattern of propagule deposition onto saltmarsh.

Propagule transport was monitored during the high tide, where possible, until a final point of stranding was marked on the marsh. Stranding was deemed to occur once the tide had receded below the saltmarsh-mangrove interface. Stranding positions were marked by flagging the points with small, numbered pegs. The effort required to recover marked propagules was minimised by observing their progress during daytime high tides. However, a standardised method was employed when the highest high tide occurred at night and searching was conducted the following morning. First, the marsh area was quickly surveyed to establish patterns of wrack deposition that were likely to also contain marked propagules. Once identified, searching was restricted to the wrack (strand) line and areas lower on the shore but still in the direction of tidal ebbing flow. This method concentrated the search effort to key areas at each release site and maximised the chances of recovering the majority of marked propagules.

3.2.4 Data analysis

I tested for any effects of tide (flood or non-flood) or starting position (proximity to runnel) on the proportion of propagules recovered using a two-way analysis of variance (ANOVA), with proportions being arcsine transformed prior to the analyses. Data were then treated by site and tidal period using one-way ANOVA to

test: the distance marked propagules were transported up the shore (hereafter referred to as height); distance transported across the shore; and, distance between the point of origin and final stranding. Where significant differences were detected in the one-way ANOVA, Tukey's tests (HSD) were used to identify which groups of propagules differed.

3.3 Results

A total of 444 marked propagules, in 30 groups, were released at the three sites and 84.7% of the propagules were successfully recovered after one high tide. The proportion of propagules recovered was not influenced by tidal amplitude or starting position relative to a runnel (two-way ANOVA: Tide $df = 1, 26$ $P = 0.188$; Runnel $df = 1, 26$ $P = 0.673$; Interaction $df = 1, 26$ $P = 0.836$).

The pattern of propagule transport and final stranding on the saltmarsh varied markedly between groups released at the three sites for both tidal events. Significant differences were detected with one-way ANOVA ($P < 0.05$) among groups of propagules transported up the shore, across the shore and from the starting position at all sites and tidal events except the non-flood release at Tingalpa Site 2.

Table 3.1 summarises the results of Tukey's HSD tests which identified groups of propagules that were significantly different. For each flooding tide sampled groups of propagules were always transported significantly further up the shore and from the starting position by runnels than by natural tidal influences. A similar result occurred for the non-flooding tidal event at two of the three sites. Thus, the point along the saltmarsh-mangrove interface at which marked propagules were released determined the extent of movement up the shore and from the starting position at each site regardless of tidal flooding.

Table 3.1 Summary of results from Tukey’s HSD test between groups of propagules. Groups are ordered by means, in ascending order. No results are shown for the non-flood tide at Tingalpa Site 2 as ANOVA recorded no significant difference.

Site	Tide	Variation: Distance		
		Up shore	Across shore	From start
Tingalpa 1	Flood	1 ^a 4 ^{ab} 2 ^b 5 ^{bc} 3 ^c	3 ^a 1 ^a 4 ^a 2 ^a 5 ^b	1 ^a 4 ^a 2 ^a 3 ^b 5 ^c
	Non-flood	1 ^a 2 ^a 4 ^{ab} 3 ^b 5 ^c	3 ^a 1 ^a 4 ^a 2 ^b 5 ^c	2 ^a 1 ^{ab} 4 ^{bc} 3 ^c 5 ^d
Tingalpa 2	Flood	5 ^a 2 ^a 4 ^{ab} 1 ^{bc} 3 ^c	1 ^a 3 ^{ab} 4 ^{ab} 2 ^{ab} 5 ^b	5 ^a 2 ^{ab} 4 ^{ab} 1 ^b 3 ^b
	Non-flood			
Coomera	Flood	3 ^a 4 ^a 5 ^a 1 ^a 2 ^b	1 ^a 2 ^b 3 ^c 4 ^d 5 ^e	4 ^a 5 ^a 3 ^a 1 ^a 2 ^b
	Non-flood	1 ^a 3 ^a 4 ^a 5 ^a 2 ^b	1 ^a 3 ^a 4 ^a 5 ^a 2 ^a	1 ^a 3 ^a 4 ^a 5 ^a 2 ^b

Dissimilar superscript denotes significant difference ($P < 0.01$); runnel group(s) in bold

Runnel path and position on the shore strongly influenced the transport of some propagules across the shore. Figure 3.3 illustrates two runnels, one constructed perpendicular to the saltmarsh-mangrove interface and the other built across the marsh (along a 30 m-wide arc). Propagules transported by the latter were carried significantly further across the marsh shore (runnel group 5 in Table 3.1 and Fig. 3.2) after both tidal events while those from the former were not (runnel group 3 in Table 3.1 and Fig. 3.2).

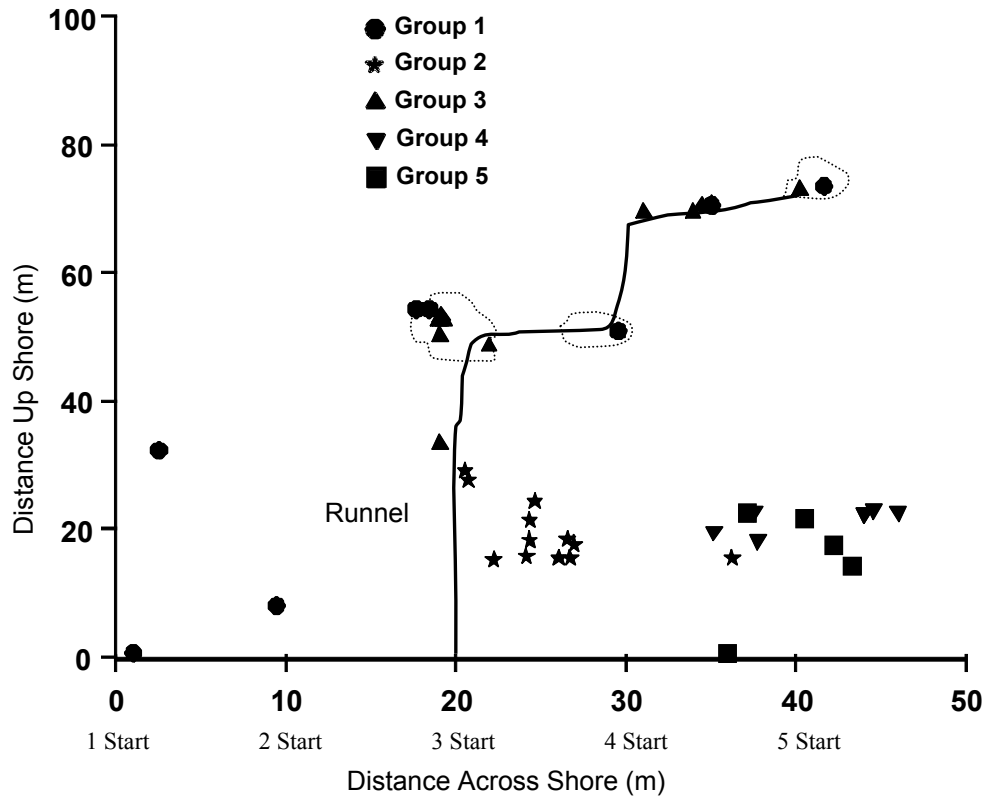


Figure 3.2 Results of propagule transport after one flooding high tide (2.56 m predicted) at Tingalpa Creek Site 2. The saltmarsh-mangrove interface represents a dynamic shoreline but is simplified in the figure. Starting positions for groups of propagules were at “0” on the Y-axis. Depressions, once isolated mosquito-breeding pools, linked to runnels are shown.

Groups of propagules released within 10 m of a runnel were transported significantly further up the saltmarsh (Table 3.1) than any other groups released along the saltmarsh-mangrove interface (Figs. 3.2 and 3.3). However, runnel transport was not exclusive to groups located close to the structure as some propagules released 20 m away were carried to similar shore positions as those in the runnels (Table 3.1; see group 1 in Figure 3.3).

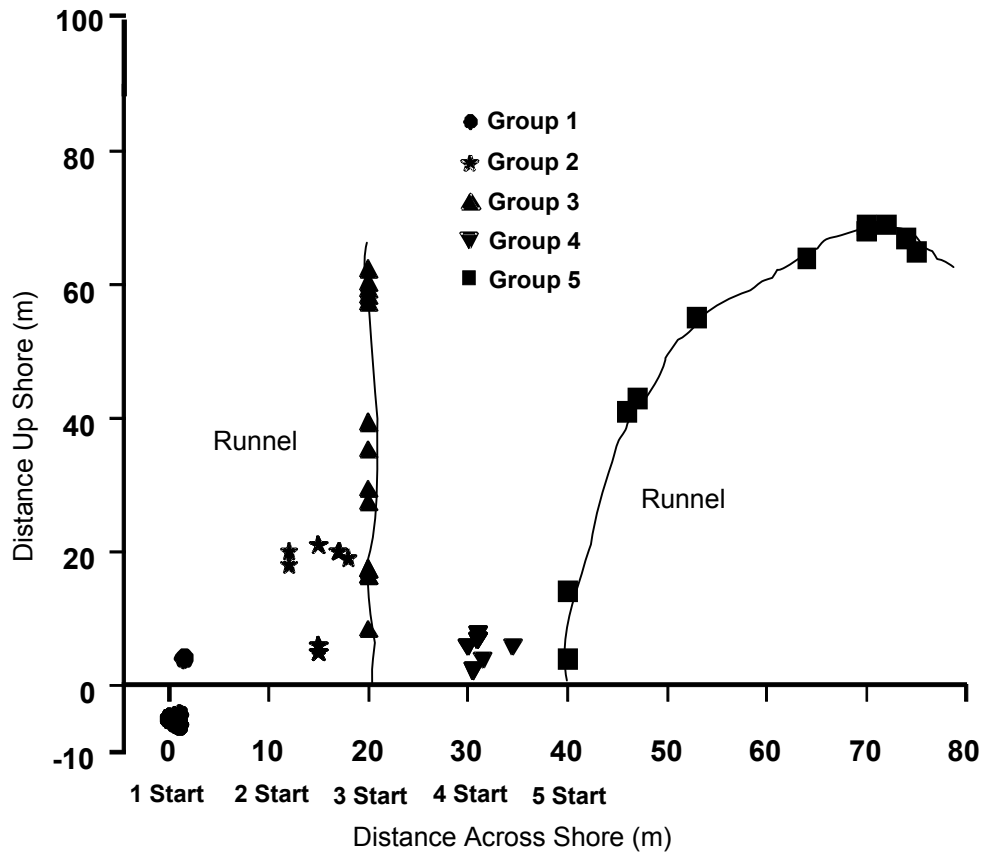


Figure 3.3 Results of propagule transport after one flooding tide (2.44 m predicted) at Tingalpa Creek Site 1. Two runnels were present at the site and their connection to linked depressions is shown. Starting positions for each group of propagules was at “0” on the Y-axis, also shown as the saltmarsh-mangrove interface.

The final stranding positions of propagules transported by runnels varied with site and tidal phase. In Figures 3.2 and 3.3 runnels were constructed to a maximum on-shore height of approximately 80 m from the saltmarsh-mangrove interface to link isolated mosquito-breeding pools located at shore heights of between 60 and 80 m. Following predicted flooding tides, propagules were not deposited in clumps, but were distributed over distances of up to 60 m (see group 5 in Figure 3.3). In many cases, propagules were deposited along the edges of (former) isolated pools, now

depressions, which were linked to the runnel rather than within the main channel of the runnel.

For groups of propagules not influenced by runnels, the pattern of deposition across the shore after one tidal phase was not consistent for the three sites. Figure 3.2 shows that some groups were deposited nearly 50 m from their original release position while groups in Figure 3.3 remained within 10 m even after a flooding tide. While propagule distribution across the shore was generally varied, the distance propagules were deposited up the shore by flooding tides was restricted to approximately 20 m from the saltmarsh-mangrove interface (Figures 3.2 and 3.3) and was consistent with observed strand lines that formed after tidal ebb.

3.4 Discussion

The fate of marked propagules released at each of the three sites was clearly influenced by the presence of runnels. Propagule transport was significantly different among marked groups at all sites and tidal phases except for the non-flood release at Tingalpa Creek Site 2. Dynamic environmental and physical conditions during the non-flood tide most likely affected the pattern of propagule deposition at Tingalpa Creek Site 2 which were not apparent at all other release sites. In explanation, the shoreline at Tingalpa Creek Site 2 was not clearly defined and the width of the saltmarsh-mangrove interface extended 50 m landward in some places. Strong on-shore winds and tidal flow transported some propagules to this region of the shore which was laterally similar in height to runnel-transported propagules but was an extension of the intertidal boundary and not saltmarsh.

Runnels enable propagule transport onto saltmarsh and so, may advantage deposition of fruit that fall from some *A. marina* on the saltmarsh-mangrove interface. In southeast Queensland, propagules generally fall during a three-month period from August to November and, as with other described populations (Clarke 1993, O'Grady *et al.* 1996) establish close to parent *A. marina*. During the fruit-fall period few propagules are naturally deposited onto saltmarsh because of a lack of suitably high tides that breach the saltmarsh-mangrove interface. However, because runnels carry low-amplitude tides which would not normally flood saltmarsh their on-shore position may advantage landward transport and deposition of propagules that fall from *A. marina* located close to the structures.

The deposition of propagules after flooding and non-flooding tidal phases was restricted to approximately 20 m up the shore from the saltmarsh-mangrove interface and was only exceeded by propagules transported by runnels which reached shore heights of up to 80 m. The fate of propagules that were not influenced by runnels was unpredictable and in some cases, resulted in their stranding below the original release position (and close to the natural intertidal limit of *A. marina*) or not moving at all from the release position. Common saltmarsh plants such as *S. virginicus* can grow on the landward edge of the saltmarsh-mangrove interface and often form dense clumps reaching heights of 40 cm. Tides that do not completely inundate plants such as *S. virginicus* will fail to advance propagules beyond the obstruction whereas runnels are devoid of vegetation and can carry propagules unimpeded.

Observations from other saltmarshes in southeast Queensland and northern New South Wales support the findings from this experiment. For example, Dale *et al.* (1989) describe manual removal of *A. marina* seedlings from runnels as an ongoing annual maintenance exercise at some sites. The varied size-range of seedlings and mature mangrove trees observed at some runnelled saltmarshes also suggest episodes of propagule deposition and establishment over a temporal scale consistent with the age of runnels. Few authors have reported on the long-term success of tidal-borne *A. marina* propagules on Australian saltmarsh. However, in caged experiments on saltmarsh Clarke and Myerscough (1993) note the failure of propagule establishment because of desiccation and Clarke and Hannon (1967, 1971) and Adam (1995) discuss the influence of salinity and waterlogging on mature plant survival. Given the fact that runnels alter the hydrological patterns governing substrate salinity and

porosity in restricted areas of saltmarsh it is likely that they also create favourable growing conditions for *A. marina* which would otherwise fail to establish.

In a colour infrared photographic comparison of a runnelled saltmarsh, Dale *et al.* (1996) reported no direct effects of runnelling on *A. marina* expansion into the saltmarsh at Coomera Island. However, their study did describe isolated increases in mature *A. marina* density and distribution along sections of the runnel and within (former) mosquito-breeding pools linked to the runnel and tidal source (Dale *et al.* 1996). The increased density of *A. marina* was not statistically significant (Dale *et al.* 1996) but was most likely due to the transport (and later establishment) of mangrove propagules, following runnel construction, consistent with the pattern of distribution of marked propagules in this study. In addition, qualitative observations at some runnelled sites suggest that once an individual *A. marina* establishes in a depression high on the saltmarsh, its pneumatophores quickly develop in the runnel and inhibit further up-shore transport of propagules carried from the saltmarsh-mangrove interface. The ‘pneumatophore filter’ may also reduce passage of propagules from the primary mangrove, thereby encouraging establishment of seedlings originating from a single localised *A. marina*. O’Grady *et al.* (1996) discuss similar localised recruitment episodes based on physical limitations to propagule supply and establishment and further, note the implications for genetic transfer between these mangrove populations.

Runnel function for mosquito control requires construction of a graded conduit linking isolated mosquito-breeding pools with the tidal source so that the frequency of tidal inundation and flushing is increased. At the sites investigated in this study

runnels also connected areas of *A. marina* propagule production adjoining the saltmarsh-mangrove interface to areas of the saltmarsh normally inundated by only the highest spring high tides. In effect, the results of this study demonstrate that propagules located close to the runnel were consistently transported to areas high on the saltmarsh shore where they would not normally be deposited. This could advantage landward extension of the intertidal distribution of some *A. marina* at runnelled sites and should be considered in future management of saltmarsh and mangrove ecosystems as well as mosquito control programs intending to use runnels.

Chapter 4 Effects of runnelling on density and size structure of the snails *Salinator solida* and *Ophicardelus* spp.

4.1 Introduction

Intertidal saltmarsh communities often develop in deltas or estuaries bearing regular and consistently high sediment loads and which are protected from direct wave action. The physical genesis and maintenance of saltmarsh communities within the intertidal zone exposes them to a range of factors which affect both their biotic and abiotic resources. Factors such as anthropogenic modification (Bertness *et al.* 2002), sediment accretion (Vernberg 1993, Hughes *et al.* 1998) and tidal frequency (Le Hir 2000, Hussein and Rabenhorst 2001a, b) are known to influence the physical features of saltmarshes. Tidal frequency is one of the primary structuring forces enabling biological communities to develop in conditions that vary in time and space (Daiber 1977, de Jonge 2000, Silliman and Bertness 2002) and therefore has important implications for tidal marsh management and restoration (Flynn *et al.* 1999, de Jonge 2000).

In Australia, some mosquito control activities specifically alter tidal frequency to disrupt the breeding cycle and larval development of vector species which breed in saltmarsh. Runnelling is a form of habitat modification which links isolated mosquito breeding pools located high on the saltmarsh with the tidal source at the saltmarsh / mangrove interface low on the shore. The depth of runnels is usually less

than 30 cm but, due to the low gradient on saltmarsh, this allows transport of low-amplitude tides which would not normally inundate breeding pools.

Given the increased frequency of flooding tides to regions of runnelled saltmarsh associated impacts on non-target biological resources must also be expected.

Saltmarsh fauna which require a limited range of soil moisture conditions or those that respond quickly to altered surface moisture conditions are ideal models to test for impacts from runnelling.

The aim of this study is to examine the influence of modified tidal frequency, as a result of runnelling, on the population structure of two common saltmarsh gastropods, *Salinator solida* Martens and *Ophicardelus* Beck. Studies of the populations of *S. solida* and *Ophicardelus* in areas of low tidal frequency report that the snails occur at low densities (Morgan and Hailstone 1986) and have a diverse range of size classes (Roach *et al.* 1989, Roach 1998). In contrast, populations from regularly inundated habitat contain many *S. solida* (Morgan and Hailstone 1986) and few *Ophicardelus* in a limited range of size classes (Roach *et al.* 1989, Roach 1998). Intermediate sizes occur in environmental conditions between the two extremes (Morgan and Hailstone 1986, Roach *et al.* 1989).

S. solida and *Ophicardelus* respond to soil moisture conditions, with larger individuals actively seeking moist conditions following tidal events and later retreating to sheltered positions when the substrate desiccates (Kaly 1988, Roach 1996). In this chapter I test whether snail density and size structure differ in time and

space between shores with and without modification affecting soil moisture conditions.

4.2 Materials and methods

4.2.1 Study site

Populations of *Salinator solida* and *Ophicardelus* were sampled from three saltmarsh shores in southern Moreton Bay, southeast Queensland, Australia (see Chapter 2, Fig. 2.1). The three shores (hereafter called “sites”) were previously runnelled for mosquito control and have a similar tidal range (~ 2.5 m) and tidal height required for full saltmarsh inundation (~ 2.45 m). Tidal waters flood the marshes either via shallow inlets at the mangrove boundary or through areas of lower elevation which support low-amplitude tidal transport. These areas of lower elevation are visually distinctive and form the key sites where runnels are constructed (Owttrim and Dixon 2001, Dale *et al.* 1998).

Mosaics of salt-tolerant plants (primarily *Sporobolus virginicus*, *Sarcocornia quinqueflora* and *Suaeda australis*) dominate the vegetation and form on thin (~ 10 cm deep) humic deposits overlying solonchaks. On the seaward side of saltmarsh, *Avicennia marina* is a dominant structural component at the saltmarsh-mangrove interface. Runnels link the saltmarsh-mangrove interface with graded areas of saltmarsh, traversing the dominant plant communities and substrates. *S. solida* and *Ophicardelus* were present in all vegetation communities from the saltmarsh-mangrove interface landward to approximately 150 m up the saltmarsh shore.

4.2.2 Snail collection

S. solida is common in saltmarsh and mangrove from Queensland south to Tasmania and west to Western Australia (Smith 1992, Roach 1996). *Ophicardelus* represents a complex of species (*O. ornatus* Ferrusac, *O. quoyi* Adams and *O. sulcatus* Adams) which inhabit saltmarsh in south-eastern Australia and is common from Queensland south to Tasmania (Kaly 1988).

S. solida and *Ophicardelus* were censused from quadrats using two experimental design protocols which are fully described in Chapter 2. To maintain a reliable sampling method, all visible *S. solida* and *Ophicardelus* within the quadrat were removed during active searching for a period of between 3 and 4 minutes.

In both sampling designs snails were removed from the quadrat if visible on the soil surface, in abandoned crab holes or depressions and those partially buried in substrate or obscured by vegetation. Unlike Roach (1996), who probed the surface substrate to remove buried snails, this study sampled during periods of peak activity and employed a visual method suitable for the species investigated (Kaly 1988).

Captured snails were pooled for individual quadrats and their width (from shell opening to the shell apex) was measured to within 0.1 mm using vernier calipers. All snails were then released at the point of capture.

Snails were sampled approximately 1 hr after the predicted (Queensland Department of Transport 2001) highest high tide that was expected to be either a non-flood (2.25

- 2.38 m) or flood (2.42 - 2.56 m) event. Tidal events are fully described in Chapter 3.

4.2.3 Data analysis

Snail abundance and size class data from the first and second sampling protocols were analysed using similar 3-way ANOVA factors, Tukey tests (HSD) and power criteria as described in Chapter 2. Prior to ANOVA, data were tested for departures from the homogeneity of variance assumption; no transformation was required.

4.3 Results

4.3.1 Runnelled versus unrunnelled: non-flood and flood

At Coomera, 4-6 mm sized *S. solida* were more common on the runnelled transect after non-flooding tides and the unrunnelled transect after flooding tides. The trend for 10-12 mm sized snails was almost opposite as they were more common on the unrunnelled transect after non-flooding tides and on the runnel following flooding events (Table 4.1; Fig. 4.1).

Table 4.1 Summary of three-way ANOVA results for *Salinator solida* and *Ophicardelus* spp. abundance and population size class structure at runnelled and unrunnelled transects. Only significant results are shown.

<i>Salinator solida</i>				<i>Ophicardelus</i> spp.			
Marsh	Factor	df	P	Marsh	Factor	df	P
<i>Abundance</i>				<i>Abundance</i>			
Coomera	tide x distance	4	**	Coomera	tide	1	*
Tingalpa 1	tide	1	*		treatment	1	*
	tide x treatment	1	**		tide x treatment	1	*
				Tingalpa 1	tide x treatment	1	***
<i>Size Class</i>				<i>Size Class</i>			
Coomera	4-6 mm			Coomera	7-9 mm		
	tide x treatment	1	*		treatment	1	*
	tide x distance	4	**		10-12 mm		
	10-12 mm				tide	1	**
	tide x treatment	1	*		treatment	1	***
Tingalpa 1	4-6 mm				tide x treatment	1	*
	tide x treatment	1	*		12 ⁺ mm		
	treatment x distance	4	*		treatment	1	*
	7-9 mm			Tingalpa 1	4-6 mm		
	tide	1	**		tide x treatment	1	***
	tide x treatment	1	**		7-9 mm		
	10-12 mm				tide x treatment	1	***
	treatment	1	**		10-12 mm		
Tingalpa 2	10-12 mm				tide	1	***
	distance	4	*		tide x treatment	1	*

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$

The highest *S. solida* abundance at Tingalpa 1 occurred on the unrunnelled transect after non-flooding tides and the runnelled transect after floods.

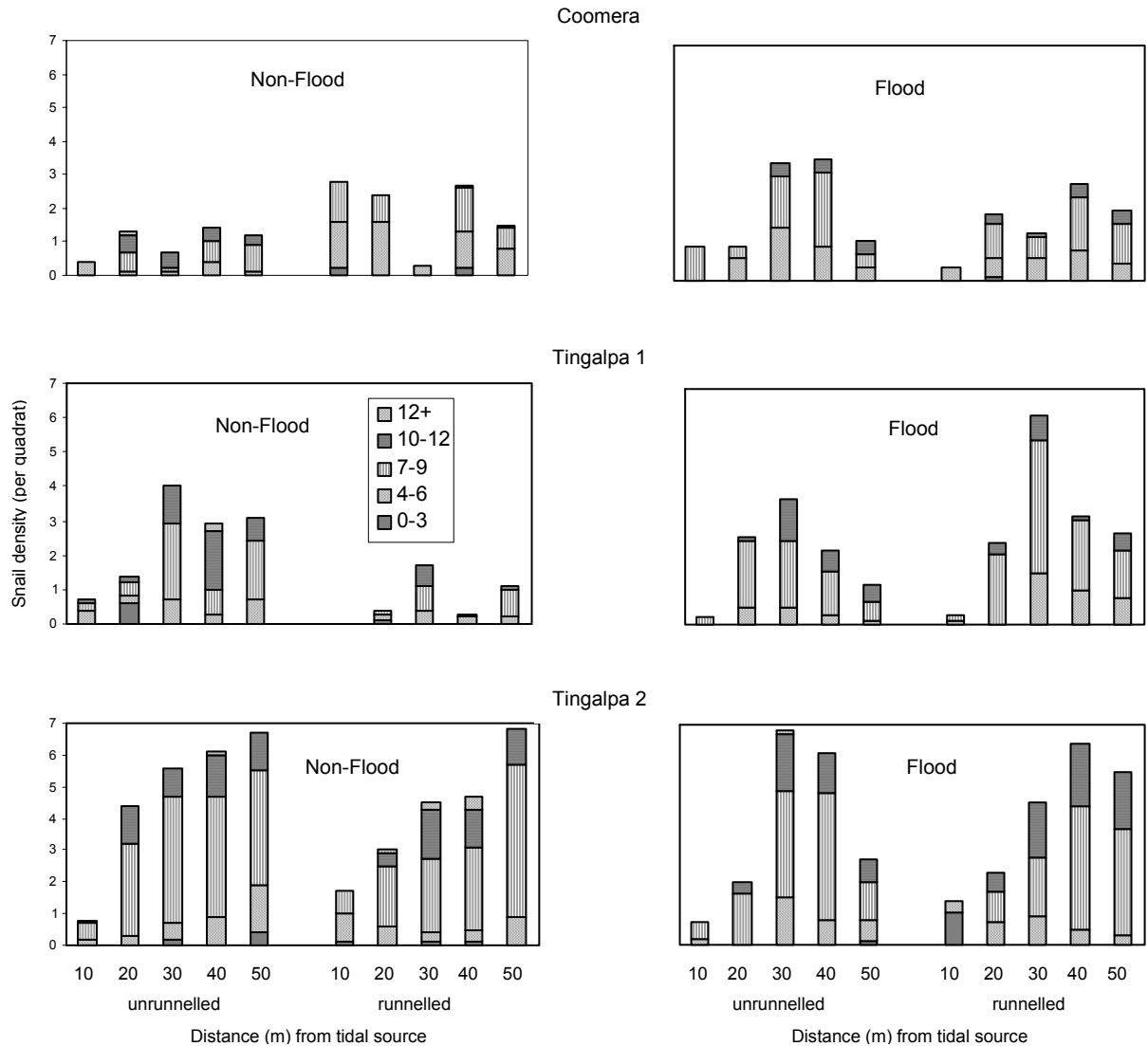


Figure 4.1 Structure of *Salinator solida* population along unrunnelled and runnelled transects after non-flooding and flooding tides at three saltmarshes. Transects are parallel (20 m separated) and extend from the tidal source 50 m up the saltmarsh shore.

The unrunnelled transect supported the highest density of 10-12 mm sized snails and more 7-9 mm sized snails only after non-flooding events (Fig. 4.1). The runnelled transect had the highest density of 4-6 mm sized snails only after flooding events and more 7-9 mm sized snails than either the runnel after non-flood tides or the

unrunnelled transect after flood events (Table 4.1). Few perceptible patterns were recorded at Tingalpa 2 but, 10-12 mm sized *S. solida* were more common at shore distances of 30 m and higher on both transects (Fig. 4.1).

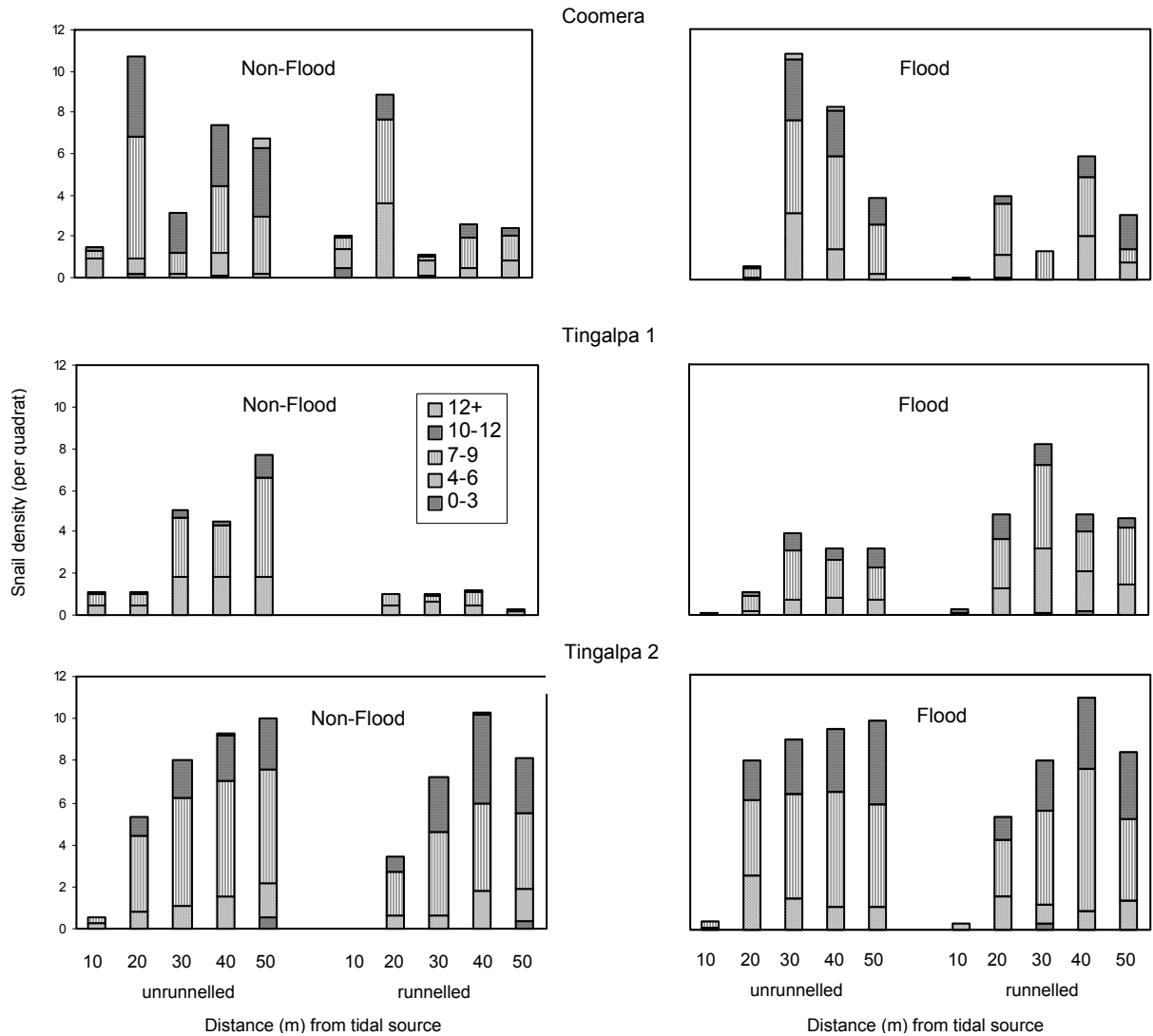


Figure 4.2 Structure of *Ophicardelus* population along unrunnelled and runnelled transects after non-flooding (NF) and flooding tides (FL) at three saltmarshes (Coomera, Tingalpa 1 and Tingalpa 2). Transects are parallel (20 m separated) and extend from the tidal source 50 m up the saltmarsh shore.

Generally, *Ophicardelus* were more common on the unrunnelled Coomera transect than the runnel. Large snails greater than 12 mm in length were absent from the runnel while snails in the size ranges 7-9 mm and 10-12 mm were always more common on the unrunnelled transect (Fig. 4.2).

At Tingalpa 1, *Ophicardelus* abundance was similar at the runnel after flooding tides and the unrunnelled transect after non-flooding events (Fig. 4.2). Snails in the ranges 4-6 mm and 7-9 mm accounted for the greatest differences in numbers for these periods while 10-12 mm snails were significantly more common after flooding events on both transects (Fig. 4.2). Neither tidal period nor runnelling significantly influenced the *Ophicardelus* population at Tingalpa 2 (Table 4.1).

4.3.2 Lateral distance from the runnel edge: non-flood and flood

The strongest pattern within the *S. solida* population was at Coomera and was little influenced by tidal period. For flooding and non-flooding tides, 10-12 mm sized snails were more common beyond 2 m from the runnel edge (Table 4.2). Shore height was an important factor responsible for the abundance of snails at all sites with more snails occurring at 30 m and 50 m positions compared to low on the shore at 0 m (Table 4.2). A similar pattern was apparent for *Ophicardelus* at the three sites with few snails found low on the shore. However, tidal period and distance from the runnel appeared to have a greater influence on the *Ophicardelus* population at the three sites with generally, more smaller snails occurring either closer to or further from the runnel (Table 4.3).

Table 4.2 Summary of three-way ANOVA results for *Salinator solida* abundance and population size class structure at distances from the runnel edge. Only significant results are shown. Tukey results are shown where significant factor has greater than two levels.

Marsh	Factor	df	P	Tukey ⁺
<i>Abundance</i>				
Coomera	shore height	2	*	1 ^a 2 ^{ab} 3 ^{bc}
Tingalpa 1	tide	1	*	
	shore height	2	***	1 ^a 2 ^{ab} 3 ^{bc}
	tide x shore height	2	***	
Tingalpa 2	tide	1	***	
	shore height	2	***	1 ^a 2 ^{ab} 3 ^{bc}
	tide x shore height	2	***	
<i>Size Class</i>				
Coomera	7-9 mm			
	shore height	2	*	1 ^a 2 ^{ab} 3 ^{bc}
	10-12 mm			
	shore height	2	**	1 ^a 2 ^{ab} 3 ^{bc}
	distance	7	*	
Tingalpa 1	4-6 mm			
	shore height	2	*	1 ^a 2 ^{ab} 3 ^{bc}
	7-9 mm			
	tide	1	*	
	shore height	2	***	1 ^a 2 ^b 3 ^b
	tide x shore height	2	**	
	10-12 mm			
	shore height	2	**	1 ^a 3 ^b 2 ^b
Tingalpa 2	4-6 mm			
	tide	1	**	
	shore height	2	**	1 ^a 2 ^a 3 ^b
	tide x shore height	2	*	
	7-9 mm			
	shore height	2	***	1 ^a 2 ^b 3 ^b
	10-12 mm			
	shore height	2	***	1 ^a 3 ^b 2 ^c
	12 ⁺ mm			
	shore height x distance	14	**	

⁺ dissimilar superscript denotes sig. diff. ($P < 0.05$) between subsets; * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$

Table 4.3 Summary of three-way ANOVA results for *Ophicardelus* spp. abundance and population size class structure at distances from the runnel edge. Only significant results are shown. Tukey results are shown where significant factor has greater than two levels.

Marsh	Factor	df	P	Tukey
<i>Abundance</i>				
Coomera	tide	1	*	
	shore height	2	*	1 ^a 2 ^{ab} 3 ^{bc}
Tingalpa 1	shore height	2	***	1 ^a 2 ^b 3 ^c
	distance	7	*	
	tide x shore height	2	*	
Tingalpa 2	shore height	2	***	1 ^a 2 ^b 3 ^c
	distance	7	*	1 ^a 3 ^a 7 ^{ab} 4 ^{ab} 8 ^{ab} 2 ^{ab} 5 ^{ab} 6 ^b
<i>Size Class</i>				
Coomera	4-6 mm			
	shore height	2	*	1 ^a 3 ^{ab} 2 ^{bc}
	7-9 mm			
	tide	1	*	
	shore height	2	*	1 ^a 3 ^{ab} 2 ^{bc}
	10-12 mm			
	tide	1	*	
	shore height	2	*	1 ^a 3 ^{ab} 2 ^{bc}
Tingalpa 1	4-6 mm			
	tide	1	*	
	shore height	2	***	1 ^a 3 ^{ab} 2 ^{bc}
	distance	7	**	8 ^a 7 ^a 6 ^{ab} 4 ^{ab} 3 ^{ab} 2 ^{ab} 5 ^b 1 ^b
	tide x shore height	2	**	
	tide x distance	7	**	
	shore height x distance	14	**	
	7-9 mm			
	shore height	2	***	1 ^a 3 ^{ab} 2 ^{bc}
	tide x distance		*	
	10-12 mm			
	shore height	2	**	1 ^a 3 ^{ab} 2 ^{bc}
Tingalpa 2	4-6 mm			
	shore height	2	***	1 ^a 3 ^{ab} 2 ^{bc}
	distance	7	**	3 ^a 1 ^a 2 ^a 7 ^a 4 ^a 5 ^a 8 ^{ab} 6 ^b
	shore height x distance	14	**	
	7-9 mm			
	shore height	2	***	1 ^a 3 ^b 2 ^c
	10-12 mm			
	shore height	2	***	1 ^a 3 ^b 2 ^c
	12 ⁺ mm			
	distance	7	*	

⁺ dissimilar superscript denotes sig. diff. ($P < 0.05$) between subsets; * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$

4.4 Discussion

4.4.1 Runnelled Vs unrunnelled

The snail population at the Coomera runnel following non-flood tides was more similar to the unrunnelled and runnelled transects after flooding events. This may be associated with patterns of soil water content and consolidation in the vicinity of runnels as detailed in Chapter 2 and which influence the availability of suitable snail feeding conditions (Kaly 1988). If the runnel maintains higher soil water content than unrunnelled areas even during non-flooding tides, snails may exploit feeding resources for longer. Whereas the runnel remains wetter for longer the unrunnelled transect is probably too dry after non-flood tides to offer adequate feeding resources for similar densities of *S. solida* which actively avoid dry surface conditions (Kaly 1988, Roach 1996).

The fact that the lowest density of 10-12 mm sized snails was sampled at the runnel following non-flooding tides requires further explanation. The highest densities of these snails only occurred when other size classes were rare (unrunnelled and non-flooding tide) or when the transects were flooded. It is possible that the runnel can support flood-like densities of smaller snails during non-flood events but the larger snails do not appear until cued by actual flooding periods. In the absence of higher densities of smaller snails, the larger ones remain prominent on the surface of the unrunnelled transect during non-flooding events and remain common even with the higher densities of smaller snails after floods.

Soil water patterns in Chapter 2 show that the runnel at Tingalpa 1 may remove moisture from the sediment at lower shore heights thereby reducing suitable habitat for *S. solida* (Roach 1996) and restricting the snail to the wetter 30 m region after non-flooding tides. This might explain the significant increase in 4-6 mm and 7-9 mm snails at the runnel only after flooding tides.

At Coomera and Tingalpa 1 the population of snails at the runnel after non-flooding tides was obviously different from other transects sampled. However, this pattern was not apparent at Tingalpa 2 which showed little influence of the runnel or tidal period. The fact that no pattern in the *S. solida* population was similar for the three sites sampled advocates the use of more specific criteria for selecting saltmarsh sites in impact assessments.

Ophicardelus occurred at higher densities than *S. solida* at Coomera but did not reflect the same structuring forces responsible for distribution. Smaller snails up to 6 mm in length occurred low on the shore as well as at higher shore distances at both transects whereas those in the range 7-9 mm were comparatively rare at distances below 20 m from the shore. The runnel clearly influenced the occurrence of some larger sized snails; those in the ranges 10-12 mm and greater than 12 mm were much more common at the unrunnelled transect yet completely absent from the runnelled area. The relative rarity of small snails low on the shore may be explained by the presence of snail predators such as *Tetractenos glaber* (smooth toadfish) which follow incoming tidal fronts (Morton *et al.* 1987) and can selectively predate *Ophicardelus* in the 7-9 mm size range. Roach (1998) explains similar patterns in

terms of low shore snail populations having faster rates of growth, than higher shore ones, which would limit the time they are palatable to predators such as fish.

Ophicardelus followed a similar pattern to the distribution and structure of the *S. solida* population at Tingalpa 1. The influence of tidal period clearly affected the abundance of snails at the runnel, especially snails in the size ranges 4-6 mm , 7-9 mm and 10-12 mm. The significantly higher density of these snails at the runnel after flooding tides may reflect better feeding opportunities, similar to those for *S. solida*, only after the runnel is flooded whereas the unrunnelled site remains viable for snail feeding after both tidal events.

While the presence of runnels does impose changes in the density and size structure of snail populations at the Coomera and Tingalpa 1 sites no similar patterns were evident at all sites. *A priori* selection of sites based simply on vegetation characteristics (or other qualitative measures) is not suitable and should include analysis of soil characteristics and soil water patterns, as discussed in Chapter 2, so that only comparable sites are sampled. These results do not closely match those of Morgan and Hailstone (1986) probably because the present study employed a more specific separation of saltmarsh shore heights rather than a comparison based only on landform. Also, in this study, the density of the snail population was compared within saltmarsh rather than between saltmarsh and mangrove which may reflect inherent differences in the population (Roach 1998).

4.4.2 Lateral distance from runnel edge

The distribution of *S. solida* and *Ophicardelus* at Coomera, Tingalpa 1 and Tingalpa 2 was clearly influenced by factors influencing the occurrence of snails in the 7-9 mm and 10-12 mm size classes. The effect of one of these factors, tidal period, on snail density was greater for flooding events at Tingalpa 1 and Tingalpa 2 than Coomera and may reflect the relationship between soil water content and feeding resources or other important structural forces under tidal influence (see Daiber 1977).

The reported patterns of distribution and size classes within the sampled populations at the three sites supports findings from the previous section. The pattern of distribution of *Ophicardelus* from lateral transects was different to *S. solida* which remained comparatively similar regardless of distance from the runnel edge.

Ophicardelus showed a clear response with distance from the edge and may mean that the species is susceptible to changed soil conditions discussed in Chapter 2.

While Roach (1996) and Kaly (1988) reported snail response to environmental conditions, this study provides evidence for the influence of substrate conditions.

Heterogeneity in patterns at the three sites should aid in the design of specific criteria for the selection of comparative sites in saltmarsh studies. However, the inclusion of sampling protocols similar to protocol 2 used in this study is not necessary for the production of meaningful ecological patterns. Nevertheless, incorporating a small number of lateral measures to maximum distances of 20 m may increase the accuracy of quadrat-based transect comparisons described in the previous section.

Chapter 5 *Helograpsus haswellianus* burrowing at runnelled saltmarshes

Part a. Defining burrow characteristics to increase sampling accuracy

5a.1 Introduction

The grapsid crab, *Helograpsus haswellianus* Whitlegge, 1889, commonly burrows in soft sediments within the intertidal zone on saltmarshes and mangroves along the eastern coastline of Australia from southeast Queensland south to Tasmania (Hale 1927, Marsh 1982). *H. haswellianus* is nocturnal, euryhaline (0 to 65 ppt tested; Marsh 1982) and forages widely on the shore between tides. This poses a problem for monitoring programs which sample the apparent abundance of *H. haswellianus* because traditional methods such as pitfall traps may simply capture foraging individuals (Chapman *et al.* 1998) and use of burrow counts to predict abundance have often misrepresented true abundance (McKillup and Butler 1979, Warren 1990) by not defining burrows as used or unused in terms of crabs inhabiting the structures. Burrows increase the surface area available for tidal infiltration of seawater (Smith *et al.* 1991), maintaining an important chemical pathway between anoxic sediments and seawater (Nomann and Pennings 1998), and provide crabs with daytime protection from predation and desiccation as well as being used for courting, breeding, and moulting (Morrisey *et al.* 1999). Methods for estimating crab abundance by defining

burrow characteristics which can increase the accuracy of measurements would be useful for grapsid crab monitoring programs.

The burrowing behaviour of *H. haswellianus* is detailed by Hale (1927) for laboratory observed specimens, by Marsh (1982) for field populations in Tasmania, and by McKillup and Butler (1979) and Higginbottom (1982) for field populations in South Australia. These studies describe *H. haswellianus* burrows as consisting of a single chamber descending to depths of around 40 cm and having either one or a few burrow entrances. In southeast Queensland, the northern distributional limit of the species, the crab constructs burrows at sites throughout the littoral zone but is more common at distances of 30 m or greater from the saltmarsh / mangrove interface up into the saltmarsh (Chapman *et al.* 1998). At their southern distribution on Tasmanian saltmarshes, *H. haswellianus* burrow closer to the tidal source and low on the shore (Marsh 1982). In Tasmania, burrows are rare in waterlogged substrates (with soil moisture content over 15%; Marsh 1982) and are found in highest densities in clay substrates that are well drained (with soil moisture content between 2-4%; Marsh 1982).

In maintaining burrows against sediment accumulation, collapse, and wear from tidal infiltration, intertidal crabs such as *H. haswellianus* also affect sediment physicochemistry. Regular burrow maintenance by crabs, bringing deep sediments to the soil surface, acts to regulate the accumulation of toxic substances in saltmarsh by enhancing subsurface soil metabolism (Williamson *et al.* 1999) and oxidation (Marsh 1982) and increasing surface infiltration of tidal waters (Smith *et al.* 1991). The presence of burrows therefore can provide a significant pathway for tidal flow

and accurate measures of their density could be used to monitor hydrological and ecological processes on saltmarsh. If monitoring programs include only used (or active) burrows to predict crab abundance we could expect an increase in the accuracy of *H. haswellianus* abundance estimates. The aim of this chapter therefore, was to examine the potential of using active and maintained *H. haswellianus* burrows, compared to total burrows (including abandoned or unused ones), to provide a reliable estimate of crab abundance.

5a.2 Materials and methods

5a.2.1 Study site

Crab and burrow sampling was conducted between April 2000 and January 2001 at two sites in Moreton Bay, Queensland, Australia (see Tingalpa 1 and Tingalpa 2 in Chapter 2, Fig. 2.1). Both sites experience a tidal range of approximately 2.5 m and have a relief of about 0.5 m from the uppermost areas of saltmarsh to the saltmarsh / mangrove interface low on the shore. Sites were selected on the basis of *Helograpsus haswellianus* being the dominant grapsid species (based on unpublished sampling exercises), proximity to each other (within 1.5 km) and having similar vegetation types consisting of the succulent samphire *Sarcocornia quinqueflora* and the salt-tolerant grass *Sporobolus virginicus*.

5a.2.2 Sampling design

At each site a 50 x 10 m transect was set at a right angle to the saltmarsh / mangrove interface and extended up the shore into the saltmarsh. Transects were then divided on the basis of vegetation and tidal flooding patterns so that the lowest unvegetated shore position that received regular tidal waters was excluded from sampling. The remaining section of the transect was flooded infrequently and comprised the halophytic succulent and grass. These drier mid and high-shore regions of vegetated saltmarsh corresponded with favoured burrowing areas for *H. haswellianus* (Chapman *et al.* 1998). This increased the probability of sampling *H. haswellianus*

and avoided the inclusion of other burrowing grapsid and ocypodid species which are generally restricted to wetter soil types closer to the tidal source.

On each transect 10 sampling positions were established by randomly assigning 0.4 m² quadrats (63.3 x 63.3 cm). Within each quadrat all visible crab burrows were counted and separated as being either used or unused. Used burrows were characterized by the presence of fresh, lighter coloured clay sediment deposits around the entrance and sharp edges to the perimeter of the entrance (see Fig. 5a.1). Upon inspection, the entrances of used burrows generally descend and join the main burrow chamber close to the soil surface. On this basis, multiple entrances to single chambers can be quickly identified. Abandoned or unused burrows exhibit obvious signs of degradation and lack fresh sediment deposits (Fig. 5a.1).

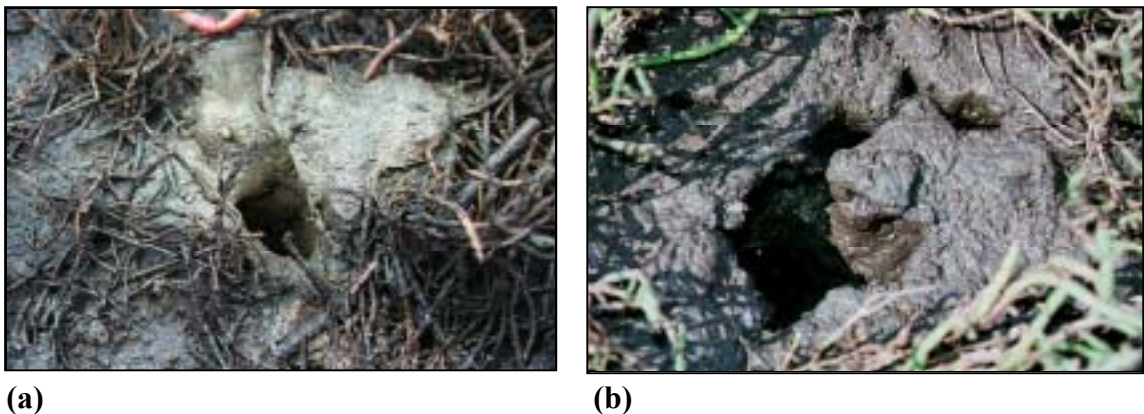


Figure 5a.1 *Helograpsus haswellianus* burrows (a) used and, (b) unused.

In the absence of regular maintenance by crabs, successive periods of tidal flooding, wind, and rain remove sediment from the entrances of unused burrows, giving them a worn appearance. These burrows may persist for long periods but appear not to be

recolonized by *H. haswellianus* for use as primary protection or breeding structures but do serve as rest sites for gastropods during between-tide periods.

Once all used and unused burrows were enumerated the area of quadrat was excavated to a depth of around 50 cm by following used burrows until the resident crab was captured. Care was taken to separate soil sods and to explore the surface for smaller individuals which may not yet have constructed burrows. All individuals were identified and their position in the burrow noted as well as any other interesting observations. Excavated soil was returned in order of removal so that the uppermost vegetated layers of sediment were replaced on the surface. Crabs were returned unharmed to the area of excavation.

5a.2.3 Data analysis

Data for crabs and burrows were pooled for all quadrats with only used and unused burrows being separated for analysis. The relationship between burrows and crabs was examined using linear regression of crab density over burrow density to determine reliable explanations of variation. Confidence limits for estimating crab density from mean burrow counts were calculated using the confidence interval regression coefficient statistic in SPSS (2001).

5a.3 Results

In total, 125 *Helograpsus haswellianus* burrows, 119 used and 146 unused, were counted from the 20 quadrats. Few used burrows contained more than one adult crab or exhibited multiple entrances which did not obviously join the main chamber close to the soil surface. Crabs excavated from burrows in the highest shore positions were generally located at the bottom of the burrow chamber while those captured from lower shore positions were found on the surface and at various depths down the burrow. Interestingly, 15 burrows excavated from mid and high shore positions had a plug of *Sporobolus virginicus* present in the burrow above the resident crab. In each case the crab was located at the bottom of the burrow and the grass plug was just above the crab (within 4 cm). The grass plug was generally composed of fresh sections of grass blades (3 cm long on average) and positioned vertically so that it completely filled the width of the burrow.

The density of burrows at both sites closely matched actual crab abundance only where specific use-defined burrow characteristics were implemented. The relationship between undefined burrows and excavated crabs was not clear (Fig. 5a.2) and regression accounted for approximately 33% of the variation (df 19, $r^2 = 0.326$, $P < 0.05$). The inclusion of undefined burrows with those maintained by crabs (the total number of burrows within quadrats) did not provide an accurate prediction of crab density from mean burrow density (± 0.63 crabs / burrow per quadrat; 95% confidence).

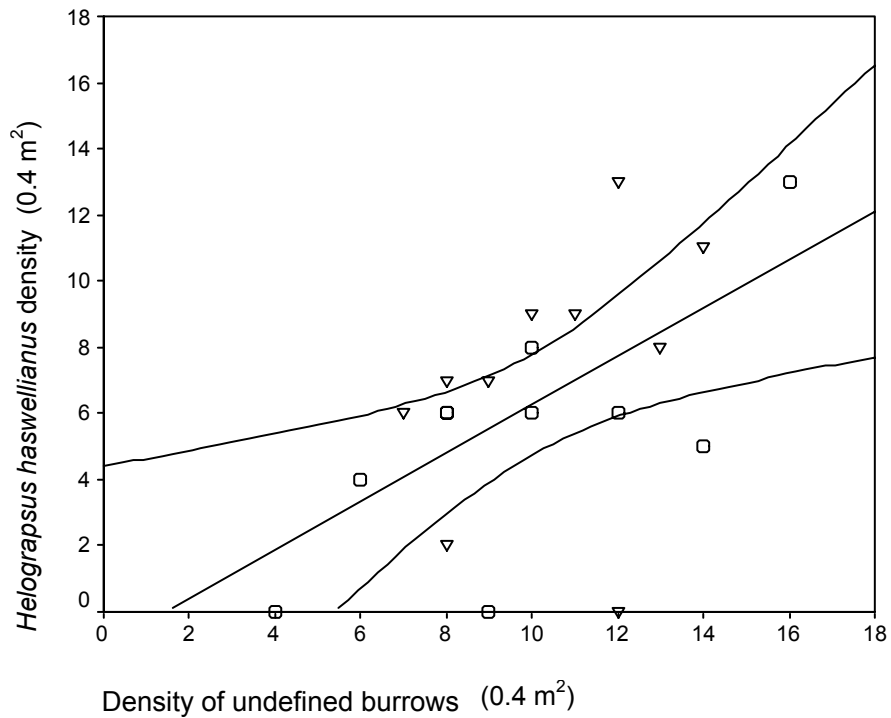


Figure 5a.2 Scatterplot of relationship of the number of *Helograpsus haswellianus* and the density of use-undefined crab burrows (all burrows in quadrats) excavated from 20 quadrats (0.4 m²) on two saltmarsh shores (site 1 circles (○), site 2 triangles (▽)), including all burrows (95% confidence bands are included for the regression line).

Inspection of used burrows revealed that they were characterized by the presence of recent sediment deposits close to the entrance and clean, sharp edges to the lip of the burrow. Unused burrows lacked these features and, with no regular maintenance, exhibited signs of degradation such as sediment slumping close to the entrance, worn edges to the lip of the burrow, and increased aperture of the burrow entrance compared to the burrow main chamber.

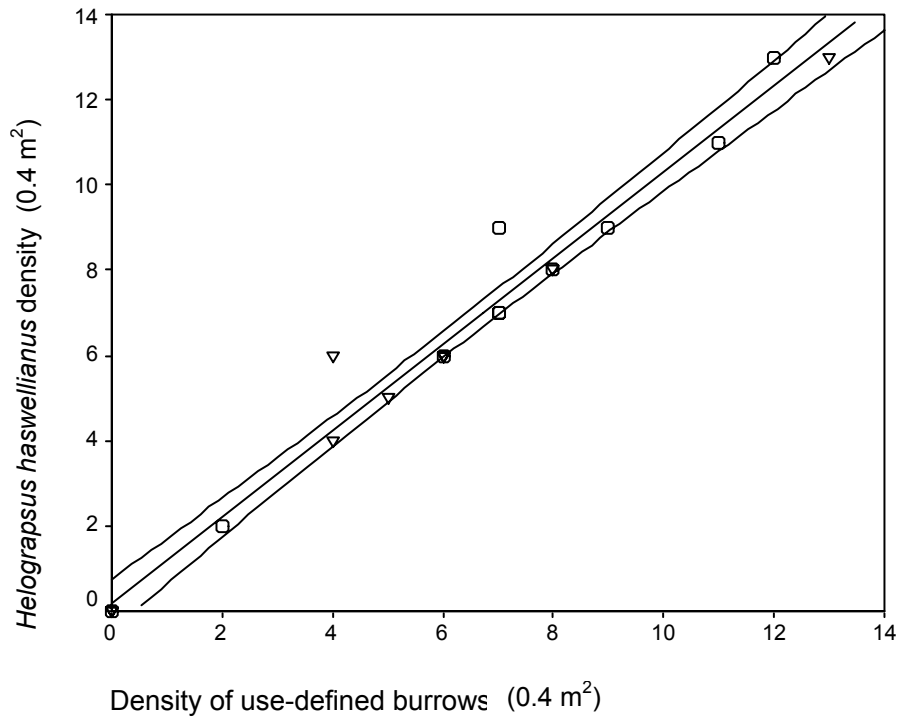


Figure 5a.3 Scatterplot of relationship of the number of *Helograpsus haswellianus* and the number of use-defined crab burrows (excluding old or unused ones) excavated from 20 quadrats (0.4 m²) on two saltmarsh shores (site 1 circles (○), site 2 triangles (▽)), unused burrows are excluded (95% confidence bands are included for the regression line).

Inclusion of only use-defined, maintained burrows in the regression accounted for approximately 97% (df 19, $r^2 = 0.972$, $P < 0.001$) of the variation in the relationship between burrow numbers per quadrat and excavated crabs at both saltmarsh sites (Fig. 5a.3). This relationship enabled confident prediction of crab density from the mean density of used burrows per quadrat (± 0.16 crabs / burrow per quadrat; 95% confidence).

5a.4 Discussion

Being able to estimate the abundance of intertidal grapsid crabs is important for environmental monitoring programs which focus on nocturnal or cryptic species. Crab burrows form significant pathways for chemical exchange between sediments and seawater as well as providing structures for reproduction, ecdysis and protection from predation and desiccation.

The use of burrows to indicate crab abundance has generally been employed successfully for ocypodid species (Mouton and Felder 1996, Nomann and Pennings 1998) rather than for grapsid crabs in similar habitat. This may be because regular tidal influences quickly degrade unused burrows in ocypodid-dominated low shore areas whereas grapsid burrows are infrequently flooded in saltmarsh and so may persist for longer periods even without structural maintenance. The deletion of old burrows in ocypodid samples allows more accurate estimates of crab abundance to be calculated than for grapsid samples where accuracy is reduced by including used and unused burrows in estimates. This chapter provides a method for defining used and unused grapsid burrows based on external characteristics of the burrow. The features which identify used burrows are easily distinguished from those that do not receive regular maintenance. In this study, substantial increases in both the accuracy (from 33% to over 97%) and confidence (from ± 0.63 to ± 0.16 crabs / burrow at the 95% level) of estimating crab abundance from burrow density counts were achieved only after burrows were defined as used or unused by crabs. This makes the method ideal for rapid assessment techniques or monitoring programmes that require fast and

non-destructive abundance estimates. Further, the method is ideally suited to species which are unable to be easily sampled through direct capture or observation.

In addition to providing an accurate indicator for abundance estimates, this study reports a previously undescribed observation from grapsid crab burrows. While the presence of decomposing vegetation litter in burrows is reported as a food source for many sesarmids inhabiting mangroves (Robertson and Daniel 1989, Lee 1998) no studies have described the presence of fresh grass matter in saltmarsh grapsid burrows. Terrestrial grapsid crabs, such as *Helograpsus haswellianus*, are prone to desiccation and maintain ion and water balance by accessing moisture through direct uptake from soil water or remaining in areas of high condensation or humidity (Greenaway, 1988). Burrows offer crabs humid environments which can reduce desiccation during periods of excessive water loss (for example during ecdysis, drought, high daytime temperatures, or air movement) but may not provide adequate free water because they are constructed in fine particle soils exhibiting low water potential (Greenaway 1988). Although *H. haswellianus* burrow in clay soils which readily bind water, the introduction of fresh vegetative matter may increase condensation at the base of the burrow to levels that enable droplet formation at night or as the vegetation dries and releases moisture. The presence of grass plugs during ecdysis or periods of drought could increase moisture resources within the burrow and may reflect adaptive responses to the arid condition of saltmarsh habitats.

Precise and confident crab abundance estimates are substantially increased by defining burrows in terms of use and excluding unused ones from abundance measures. The technique reported here could be adapted to suit programmes

requiring fast and accurate abundance estimates of crabs or other burrowing intertidal species which are unable to be directly sampled. To-date, this method provides the most accurate technique for confidently estimating grapsid crab abundance using burrow counts in saltmarsh habitats and will be used to explore the relationship between tunnelling and crab burrow density in part b of the chapter.

Chapter 5 *Helograpsus haswellianus* burrowing at runnelled saltmarshes

Part b. Comparison of burrow density and aperture at runnelled and unrunnelled shores

5b.1 Introduction

The grapsid shore crab *Helograpsus haswellianus* (Decapoda, Grapsidae) is common on saltmarshes from Queensland to South Australia and Tasmania. The crab is nocturnal, euryhaline and forages on intertidal substrates during low tide. *H. haswellianus* exhibits morphological features characteristic of terrestrial land crabs such as high gill volume to body ratio, walking hind legs and hairs between the ambulatory legs to reduce water loss (Griffin 1966), but does not commonly inhabit areas devoid of tidal influence. Rather, *H. haswellianus* occupies the highest shore positions of marine crab species (Griffin 1971), burrowing from the upper midlittoral into the supralittoral fringe.

H. haswellianus constructs burrows for protection from predation and desiccation and, similar to other species, for breeding and moulting (Dunham and Gilchrist 1988, Morrisey *et al.* 1999). Burrows which extend to depths of around 30 or 40 cm require regular maintenance by the resident crab. Burrows are close to the size of resident crabs and maintenance involves movement of soil from the lowest regions of the burrow to the opening at the surface, thereby creating a cycle of bioturbation

(Dunham and Gilchrist 1988). This process releases small quantities of anoxic and oxic sediments and facilitates soil metabolism and oxidation by reducing the accumulation of toxic substances (Marsh 1982). Infrequent inputs of seawater (percolating up from the water table or from flooding tides) further neutralise soil chemicals and are critical in the health of intertidal systems undergoing anthropogenic influence (Williamson *et al.* 1999).

The highest densities of *H. haswellianus* burrows usually occur in drier substrate and seldom occur in waterlogged soils (Higginbottom 1982, Marsh 1982). Soil water content is associated with crab burrowing and *H. haswellianus* avoids wet conditions, favouring well-drained substrates (Marsh 1982). In Tasmania, soil conditions allow the species to construct burrows close to the tidal source whereas in Queensland these conditions can restrict high-density burrowing to distances greater than 30 m from the saltmarsh / mangrove interface at the shore (Chapman *et al.* 1998).

Runnelling is a physical habitat modification technique for mosquito control in saltmarsh. The method involves linking isolated mosquito-breeding pools to the tidal source thereby increasing tidal frequency of low-amplitude tides to the high marsh. As a result of runnelling activities, there is an increase in the moisture content of saltmarsh soils adjoining the runnel (Hulsman *et al.* 1989) and these levels can be sustained for periods following tidal ebb. Given the substrate conditions necessary for *H. haswellianus* burrowing, runnelling may influence the distribution of individuals because of altered burrowing conditions. If these conditions become limiting to crab burrowing this would be evident in the density of used burrows within areas of runnelled saltmarsh.

5b.2 Methods

5b.2.1 Study site

Density and aperture of crab burrows were sampled from three saltmarsh shores (hereafter called “sites”) in southern Moreton Bay, southeast Queensland, Australia (see Chapter 2, Fig. 2.1). The three sites were similar in tidal range (approximately 2.5 m) and relative relief (approximately 0.5 m) but not vegetation dominance.

The Coomera site was largely composed of the saltmarsh grass, *Sporobolus virginicus*, with few patches of glasswort, *Sarcocornia quinqueflora*, whereas Tingalpa 2 was composed of only the glasswort. Tingalpa 1 site was a mosaic of both species with neither dominating. Other saltmarsh herbs such as *Suaeda australis* were present but patchily distributed and not common. Although the saltmarsh vegetation differed in terms of composition at each site its extent on the shore was similar, being bounded at the lowest shore height by *Avicennia marina* var. *australasica* mangrove community.

5b.2.2 Study animals and definition of burrow characteristics

A number of decapod crabs are common in Queensland intertidal habitats with species representing grapsid, portunid and ocypodid families. In general, grapsids exhibit terrestrial adaptations and lack the paddle hind leg of portunids, which are rare on saltmarsh although some commercial species (*Scylla serrata* and *S.*

olivaceae) do burrow on lower shore fringes. Ocypodids also inhabit lower regions and are common on exposed mudflats and mangrove.

Although burrowing by *H. haswellianus* was the focus of this study, two other crabs were common and required separation from the target species. *Australoplax tridentata* (Ocypodidae) and *Sesarma erythroactyla* (Grapsidae) occur within the intertidal limits of *H. haswellianus* and burrow in similar substrate. *S. erythroactyla* tends to burrow in wetter substrates and may overlap in distribution with *H. haswellianus*. Similarly, *A. tridentata* is common on exposed mudflats at the saltmarsh / mangrove interface and other areas of saltmarsh which sustain wet or waterlogged conditions within the range of *H. haswellianus*.

Freshly constructed and regularly maintained *H. haswellianus* burrows are readily distinguished from abandoned ones (see Chapter 5a). Although abandoned burrows may persist for long periods on saltmarsh they can be differentiated from maintained ones. The recognition of used burrows greatly increases the accuracy of employing burrow counts for estimating the apparent abundance of *H. haswellianus* on saltmarsh (see Chapter 5a).

Only burrows that were obviously regularly maintained were censused in this study and specific criteria included signs of fresh workings around the burrow entrance, minimal degradation of the entrance, maintenance of substrate between multiple entrances and aperture not exceeding chamber diameter. Burrow aperture reflects the actual size of resident crabs (Dunham and Gilchrist 1988) and was measured using vernier calipers. Aperture comprised the inside diameter of the entrance from

edge to edge (burrows are not necessarily circular and may be ovoid in shape).

Recorded burrow apertures were grouped into six size classes 0-5, 6-10, 11-15, 16-20, 21-25 and 25⁺ mm which are believed to reflect the actual sizes of resident crabs.

Crabs responsible for burrow construction and maintenance were identified using two techniques, visual identification of resident species inhabiting the burrow and reference to historical trapping data. For the first method, an understanding of burrowing habits of resident species reduced the effort required for identification. For example, *A. tridentata* construct shallow burrows with resident crabs found close to the surface (often within 2 cm of burrow entrance). *S. erythroductyla* and *H. haswellianus* may inhabit similar regions of saltmarsh but have different circadian rhythms and feeding preferences. *H. haswellianus* is rare on open ground during the day whereas *S. erythroductyla* will feed on algal deposits following daytime high tides. Foraging *S. erythroductyla* may use the burrow for rest periods during which time they remain high in the chamber and are able to be easily extracted for identification. Extraction of *A. tridentata* and *S. erythroductyla* involved pushing a long skewer into the burrow behind the crab so that the resident would move towards the entrance, and capture.

Reference to historical pitfall trapping data (see Chapman *et al.* 1998, Dale unpublished data) established areas of likely species dominance across the shore, making identification of species responsible for burrow construction viable.

5b.2.3 Sampling design

Two sampling protocols were used in this study. Refer to Chapter 2 for a full description of the experimental design protocols used for sampling burrow density.

Burrow features were sampled after the predicted (Queensland Department of Transport 2001) highest high tide that was expected to be either a non-flood (2.25 - 2.38 m) or flood (2.42 - 2.56 m) event. Non-flood tides in the range 2.25-2.38 m accounted for approximately 16% of annual tidal events and reached the saltmarsh-mangrove interface but did not inundate the saltmarsh except for areas contained within runnels. Tidal flood events accounted for approximately 7% of annual tides and breached the saltmarsh, extending at least 50 m up the shore onto the saltmarsh. Tidal period was considered important in comparing the density of burrows between the runnel and unrunnelled sites with the prediction being that the runnel would exhibit features more similar to flood events than non-flood.

5b.2.4 Data analysis

Burrow density and aperture size class data from the first and second sampling protocols were analysed using similar 3-way ANOVA factors, Tukey tests (HSD) and power criteria as described in Chapters 2 and 4. It is important to note that data were tested for departures from the homogeneity of variance assumption; no transformation was required in this case.

5b.3 Results

5b.3.1 Runnelled versus unrunnelled: non-flood and flood

At Coomera, the interaction of tide x treatment significantly influenced the density of burrows (Table 5b.1) with fewer recorded at the runnel transect after flooding tides than non-flooding events (Fig. 5b.1). Tidal period appeared to have little impact on the density of burrows at the unrunnelled transect for both tidal periods.

Table 5b.1 Summary of three-way ANOVA results for density and aperture of grapsid crab burrows at runnelled and unrunnelled saltmarsh transects. Only significant results are shown.

Marsh	Factor	df	P
<i>Burrow Density</i>			
Coomera	treatment	1	**
	tide x treatment	1	***
Tingalpa 1	tide	1	*
<i>Burrow Aperture</i>			
Coomera	<i>6-10 mm</i>		
	treatment	1	*
	tide x treatment	1	**
Tingalpa 1	<i>11-15 mm</i>		
	treatment	1	***
	<i>6-10 mm</i>		
	tide	1	*
	tide x treatment	1	**
	<i>16-20 mm</i>		
Tingalpa 2	tide	1	*
	treatment		*
	<i>21-25 mm</i>		
	treatment	1	*
	<i>16-20 mm</i>		
	tide	1	*
	<i>21-25 mm</i>		
	tide	1	*

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$

The presence of runnels weakly influenced the density and distribution of 6-10 mm sized burrows at the runnelled transect which were more similar in number to the unrunnelled transect after a flood tide (Fig. 5b.1). A stronger pattern was evident for 11-15 mm wide burrows which were always more common at the runnel compared to the unrunnelled transect (Table 5b.1, Figure 5b.1).

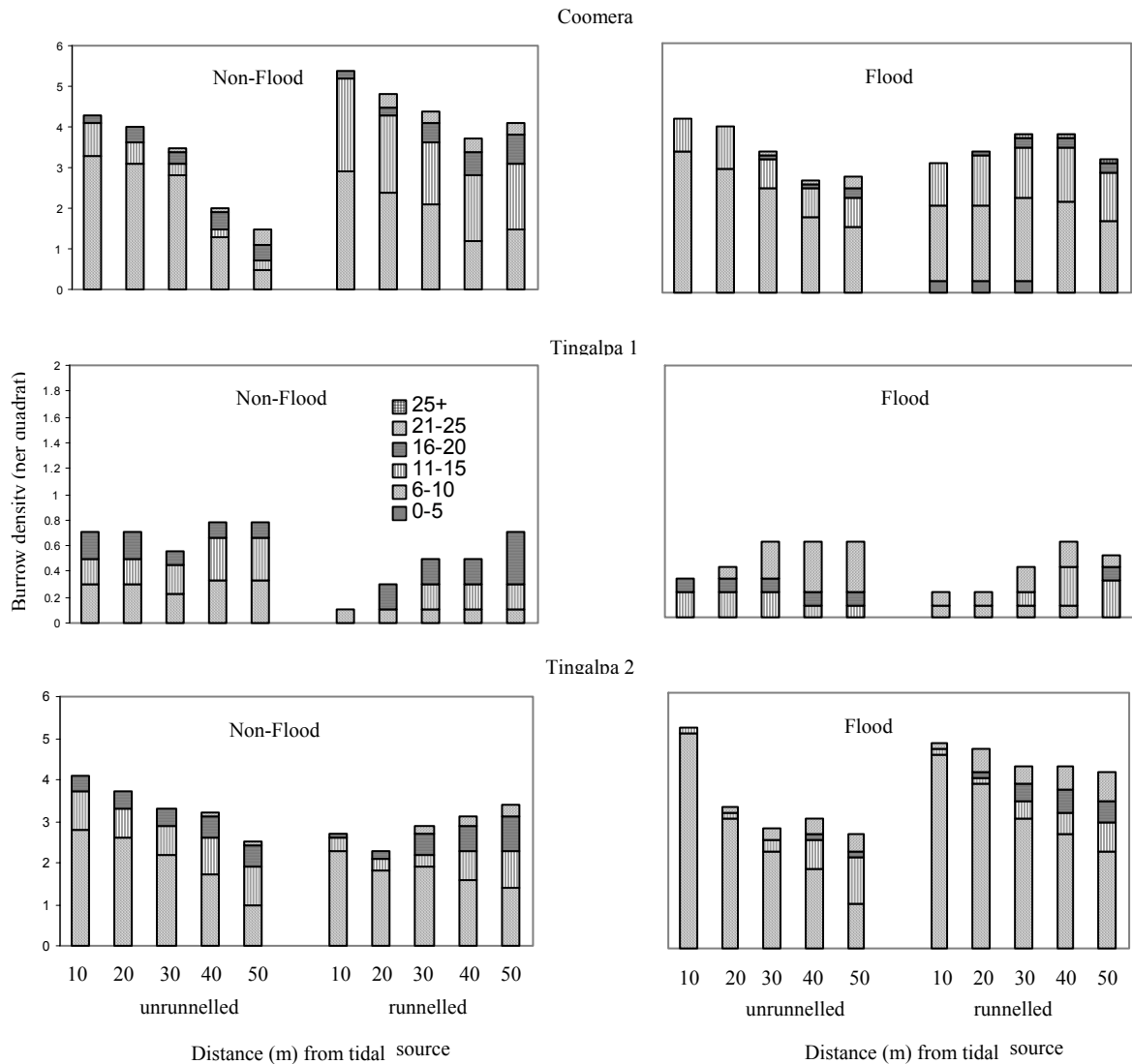


Figure 5b.1 Structure of *Helograpsus haswellianus* population along unrunnelled and runnelled transects after non-flooding and flooding tides at three saltmarshes. Transects are parallel (20 m separated) and extend from the tidal source 50 m up the saltmarsh shore. The y-axis of Tingalpa 1 is re-scaled to highlight patterns.

Tide was an important factor influencing the distribution of crab burrows at Tingalpa 1 with more burrows recorded from unrunnelled transects than runnelled ones after both tidal events (Fig. 5b.1). Burrows in the size range 6-10 mm were more common at the unrunnelled transect after non-flooding tides but were absent after flooding events while the distribution at the runnel remained similar after both tidal periods (Table 5b.1, Fig. 5b.1).

Burrows in the range 16-20 mm were more abundant at the runnel transect after a non-flooding tide and rare after flooding events whereas those on the unrunnelled transect were similar. Larger burrows in the 21-25 mm class were not recorded during non-flood tides but were common on the unrunnelled transect after flooding events (Fig. 5b.1).

Tide weakly influenced the distribution of burrows at Tingalpa 2. Fewer 16-20 mm sized burrows were recorded on the unrunnelled transect after flooding tides whereas those in the 21-25 mm range were common at both transects after flooding tides (Table 5b.1, Fig. 5b.1).

5b3.2 Lateral distance from the runnel edge: non-flood and flood

Burrow density and distribution at all three sites was strongly influenced by tidal period, shore height and their interaction. At Coomera, more burrows occurred at low shore heights after non-flood events than flood tides (Table 5b.2). Small burrows in the range 6-10 mm were always more common lower on the shore after

both tidal periods while larger 21-25 mm sized-burrows were more common at higher shore positions for the same tides.

Table 5b.2 Summary of three-way ANOVA results for density and aperture of grapsid crab burrows at lateral distances from the runnel edge. Only significant results are shown. Tukey results are shown where significant factor has greater than two levels.

Marsh	Factor	df	P	Tukey ⁺
<i>Burrow Density</i>				
Coomera	tide	1	***	
	shore	2	***	3 ^a 2 ^b 1 ^c
	tide x shore	2	***	
Tingalpa 1	shore	2	***	1 ^a 2 ^b 3 ^c
	distance	7	**	1 ^a 8 ^a 7 ^{ab} 5 ^{ab} 2 ^{ab} 3 ^{ab} 6 ^{bc} 4 ^c
Tingalpa 2	tide	1	***	
	shore	2	***	3 ^a 2 ^b 1 ^b
	distance	7	**	2 ^a 3 ^a 7 ^a 4 ^a 8 ^{ab} 5 ^{ab} 6 ^{ab} 1 ^c
<i>Burrow Aperture</i>				
Coomera	6-10 mm			
	shore	2	**	3 ^a 2 ^{ab} 1 ^{bc}
	11-15 mm			
	tide	1	*	
	16-20 mm			
	tide	1	**	
Tingalpa 1	21-25 mm			
	shore	2	**	1 ^a 2 ^{ab} 3 ^{bc}
	6-10 mm			
	tide	1	**	
	shore	2	*	1 ^a 2 ^{ab} 3 ^{bc}
	tide x distance	7	*	
Tingalpa 2	11-15 mm			
	shore	2	*	1 ^a 2 ^b 3 ^b
	distance	7	***	
	16-20 mm			
	tide		*	
	shore	2	***	1 ^a 2 ^b 3 ^b
Tingalpa 2	6-10 mm			
	tide	1	***	
	shore	2	***	3 ^a 2 ^b 1 ^c
	distance	7	**	7 ^a 3 ^a 2 ^a 4 ^a 6 ^{ab} 8 ^{ab} 5 ^{ab} 1 ^c
	tide x shore	2	*	
	11-15 mm			
	tide x shore	2	*	
	16-20 mm			
	tide	1	**	
	shore	2	*	1 ^a 2 ^{ab} 3 ^{bc}
	distance	7	*	2 ^a 3 ^a 8 ^a 1 ^{ab} 7 ^{ab} 4 ^{ab} 5 ^{ab} 6 ^c
	tide x distance	7	**	
21-25 mm				
distance	7	*	1 ^a 2 ^a 3 ^a 4 ^a 5 ^a 6 ^a 8 ^a 7 ^b	

⁺ dissimilar superscript denotes sig. diff. ($P < 0.05$) between subsets; * $P < 0.05$; ** $P < 0.01$; ***

$P < 0.001$

Tidal period influenced burrows in the mid-sized ranges 11-15 mm and 16-20 mm which were rare following flooding tidal periods (Table 5b.2). Lateral distance from the runnel edge did not significantly influence burrow density although generally, more burrows were recorded at 30 m and 50 m shore heights and within 5 m of the runnel after both tides.

At Tingalpa 1 and Tingalpa 2, more larger burrows were recorded at mid shore heights and closer to the runnel after both tidal periods whereas small burrows tended to be more common across all positions low on the shore. However, 6-10 mm burrows at Tingalpa 2 were common at low and mid shore heights, beyond 2 m of the runnel edge and after non-flood tides (Table 5b.2). The distribution of these small burrows accounted for much of the general pattern at Tingalpa 2.

5b.4 Discussion

5b.4.1 Runnelled Vs unrunnelled

The overall density of burrows at Coomera was influenced by the presence of runnels, but particular aperture size classes reflect stronger patterns consistent with previous Chapters of this thesis (Chapters 2 and 4). Specifically, crabs responsible for constructing mid-sized burrows in the range 11-15 mm may be able to exploit additional burrowing habitat in the vicinity of the runnel. This is associated with increased soil water content of runnel substrate (Hulsman *et al.* 1989) following non-flooding tides (see Chapter 2) which may offer better conditions for crab burrowing compared to the unrunnelled transect which remains dry for tides other than flooding events.

The substrate at Tingalpa 1 is probably susceptible to severe changes in consolidation and moisture content (see Chapter 2), regardless of the presence of the runnel. Natural variation in the density of burrows at the unrunnelled transect between tidal periods supports this and runnelling may simply exacerbate already erratic patterns. It is likely that runnels remove available moisture from lower shore sediments, thus reducing suitable burrowing habitat. At these lower shore regions flooding tides are required for the construction of larger burrows (Marsh 1982) which were only recorded after these events. Alternatively, the disappearance of small 6-10 mm burrows at the unrunnelled transect between tidal periods may reflect sub-surface increases in moisture content which advantage burrowing of larger crabs

(Griffin 1971). At Tingalpa 1, the presence of the runnel may reduce natural soil water fluctuation resulting in more stable burrowing conditions between tidal periods.

Smaller burrows tended to dominate at the runnel transects. However, the presence of runnels appeared to have little impact on the distribution of burrows at the three sites. Also, while there were some differences between runnelled and unrunnelled transects, or interactions between treatment and tide, the patterns were not very strong. Tidal period was a more important factor with increased density of larger burrows at the runnel following flooding tides and fewer mid and large burrows at the unrunnelled transect for the same tide. The associated influence of the runnel and tidal period on substrate burrowing conditions is worth further consideration as this interaction could be responsible for observed patterns rather than each factor individually. Skilleter and Warren (2000) report similar patterns for *S. erythroductyla* and *H. haswellianus* burrowing in disturbed substrate in mangrove forests, noting increased burrow density where soil conditions apparently enabled easier burrow construction.

5b.4.2 Lateral distance from runnel edge

Small and mid-sized burrows comprised the majority of records at all shore heights on the Coomera marsh. However, small crabs may extend further up the saltmarsh shore only after flooding tides because substrate conditions would usually restrict their burrowing to softer sediments lower on the shore. This result is discussed by Chapman *et al.* (1998) in terms of patches of suitable burrowing habitat which may

be promoted or reduced by some physical activities which alter soil water availability. It is possible that the influence of flooding tides on soil water content extends beyond the edge of the runnel (see Chapter 2), thus enabling smaller crabs to burrow at greater lateral distances from the runnel (which acts as a source of soil moisture) than may occur in the absence of the structure (Chapman *et al.* 1998).

The pattern of crab burrowing at Tingalpa 1 strongly supports records taken from the previous section with few burrows recorded low on the shore. Mid and larger sized burrows tended to dominate the overall distribution close to the runnel edge. This may be because sediment within 5 m of the runnel provides better burrowing conditions for larger crabs, which require more stable sediments (Marsh 1982), due to increased moisture inputs (see Chapter 2).

The use of sampling protocol 2 for sampling crab burrows revealed few patterns not identified in the quadrat-based method. However, the inclusion of some lateral distances beyond the extent of quadrat plots would greatly enhance the accuracy of runnel monitoring surveys.

As concluded in Chapter 4, site heterogeneity is implicit in the explanation of patterns for crab burrowing. Although runnelling does pose some influence on the availability of suitable habitat for species it is unlikely to cause changes in community composition at levels of significance other than local scales (see discussion by Dale and Dale 2002). Crabs probably respond to subtle changes in substrate characteristics which affect their burrowing, so runnelling may simply exacerbate processes already occurring. The increase of inundation frequency near

runnels may reduce burrowing opportunities for mid-sized crabs whereas small and large crabs are not greatly influenced. Indeed, this may reflect pressure on mid-sized crabs for burrowing space (Marsh 1982, Dunham and Gilchrist 1988), after flooding tides, from smaller and larger crabs which occur in higher densities. Again, if these patterns are localised and not consistent at all runnelled sites there should be little cause for concern about runnelling.

Chapter 6 Management recommendations and field monitoring concerning the effects of runnelling on non-target resources

6.1 The effects of runnelling on non-target organisms

Runnelling is a permanent, cost-effective and efficient method of reducing pest mosquito breeding in saltmarsh environments (Hulsman *et al.* 1989, Dale *et al.* 2002, Latchford *et al.* 2002). The method has been successfully implemented on the west and east Australian coastlines as a physical technique included in integrated mosquito control programs. Runnelling is a popular approach with local governments and authorities. However, current permit applications to conduct runnelling works require a level of monitoring based on predictive impacts which often lack quantitative foundations.

To date, research into the long-term impacts of runnelling has enabled the development of management strategies for runnelling (Owttrim and Dixon 2001, Latchford *et al.* 2002), although there is currently limited understanding of the impacts of runnelling on specific non-target saltmarsh resources (Breitfuss 2001).

The results of the current study, presented as chapters on physical (soil water content and consolidation), mangrove propagule distribution, snail and crab burrow models, detail patterns of influence from runnelling on specific physical and biological saltmarsh resources and should guide future management of runnelling programs.

6.1.1 Soil water content and consolidation

The interstitial water component of saltmarsh sediment is influenced mainly by saltwater tidal inputs and to a lesser extent freshwater runoff and rain (Adam 1990). Freshwater inputs are temporally and spatially variable whereas tidal patterns are predictable and generally originate from a common source or follow a similar pattern of flow (either rising or ebbing).

A range of geomorphological factors influence the soil water content of substrate at a particular location on the shore (see Daiber 1977, Avinmelech *et al.* 2001, Hussein and Rabenhorst 2001a). Of these, shore position and elevation (above mean sea level) are important in controlling the frequency and amplitude of tidal inundation (Hughes *et al.* 1998). This means that frequent low amplitude tides which fail to exceed a particular shore elevation will be restricted to lower-lying regions which are generally closer to the saltmarsh / mangrove interface. These regions remain wetter for longer, compared to higher marsh positions, as the time between regular flooding events is short. As distance from the tidal source and elevation increases there is a reduction in the frequency of tides which will inundate the marsh substrate. The highest regions of saltmarsh may only be flooded by spring high tides at very low frequency. In addition, the length of time over a tidal cycle that these tides continue to flood the highest saltmarsh positions does vary but usually only persists for a few days until tidal amplitude is again too low to reach upper margins.

The runnelled saltmarshes examined in this study included areas which were originally inundated by only the highest spring high tides. These areas would usually

include depressions isolated from the tidal source which pooled water following irregular tidal inundation. Following the construction of runnels, the pools are no longer isolated but are connected to the tidal source and because of the runnel depth, subject to increased tidal frequency. This increase in tidal flushing is the physical mechanism underlying control of pest mosquito production which would otherwise occur in the isolated pools where breeding conditions (as oviposition sites and for larval development) are ideal (Hulsman *et al.* 1989).

For example, if a saltmarsh which is usually only flooded by tides exceeding 2.45 m (predicted at Brisbane Bar, east coast Australia and approximately 7% of all annual tides) is runnelled to a depth of 0.25 m there would be a significant increase in tidal flooding events (Fig. 6.1).

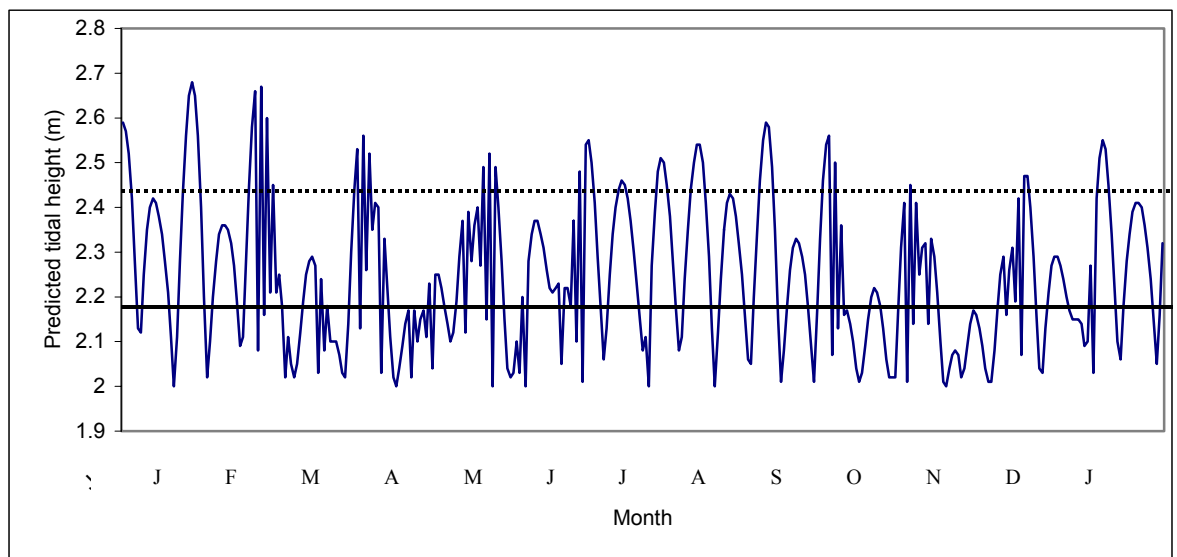


Figure 6.1 Predicted pattern of high tides greater than 2.0 m for the 2002 period. Dashed line represents height of tides (2.45 m) that completely flood a hypothetical saltmarsh. When runnelled, tidal height required for inundation reduces to 2.20 m (solid line).

Post-runnelling, tides exceeding 2.20 m (predicted) would reach higher shore positions via the runnel, thus hypothetically increasing the number of flooding tidal events in 2002 by 400%; from 50 (7%) to 196 per year (28%) see Figure 6.1.

Runnels facilitate slow-water movement of low-amplitude tides which would not normally affect higher marsh soils. Runnelled areas of saltmarsh are generally wetter than unrunnelled ones and more similar to substrates following spring high tides, even during low-amplitude tidal events in terms of soil water content and physical consolidation. However, this study has demonstrated that the extent of influence on substrate conditions from runnelling is generally restricted to within 2-5 m from the runnel edge.

The vertical profile of saltmarsh soil is defined by a range of physical processes such as accretion, subsidence and saturation which result in successive facies that can be traced over geological time (Haslett *et al.* 2001, Belperio *et al.* 2002). In a saturated environment aerobic biological activity is limited to the uppermost layers of the profile. In this way, the availability of water within the substrate actually regulates the survival of pioneering species and the distribution of biological resources both vertically and horizontally. It is likely then that increased frequency of inundation associated with runnels will alter water movement patterns and soil water content which can impact on biological components.

6.1.2 Transport of mangrove propagules

In transferring regular tidal loads to higher shore areas of saltmarsh runnels also provide an alternative dispersal mechanism for pioneering mangrove species. Results presented in Chapter 3 report significant landward extension of *Avicennia marina* propagules in runnels during low-amplitude tides which would normally fail to breach the saltmarsh / mangrove interface (Clarke and Hannon 1969). This pattern has important management implications mainly because *Avicennia marina* is invasive of saltmarsh under ideal conditions (Saintilan and Williams 1999, Saintilan and Wilton 2001). However, because propagule establishment is generally limited to the immediate area at modified sites, runnels may provide a genetic advantage (see Chapter 3 – discussion) for some individuals which are transported within one season to growing sites resulting from their dominance of future establishment events on the saltmarsh above the parent plant.

6.1.3 Surface grazing snails

It is likely that altered soil moisture conditions at the runnel affected the distribution and size structure of gastropods (see Chapter 4). Both *Salinator solida* and *Ophicardelus* were influenced by the interaction of tidal period and the presence of the runnel. However, patterns were site-specific even though the influence of the runnel on resident populations was significant. These snails are considered ecologically important for nutrient cycling (Kaly 1988) and the maintenance of intertidal food-webs as prey (Morton 1990) so changes to their preferred areas of feeding or habitat are significant. Further identification of the likely impacts of

runnelling on surface grazers should be conducted to broaden the results of Chapter 4 and to focus management attention towards invertebrate models of environmental change.

6.1.4 Distribution of crab burrows

Generally, many smaller *Helograpsus haswellianus* burrows were recorded at the runnelled compared to unrunnelled transects and this pattern may reflect altered moisture conditions required for burrowing (see Chapter 5). Patterns were not similar at all sites because substrate characteristics may influence the scale of runnel impacts. Sites with sandy profiles are more porous and fail to retain moisture whereas sites with a primarily clay base will accept higher and retain higher moisture levels. This could be interpreted to mean that runnels promote the distribution of *H. haswellianus* at sandy sites where the crab was previously unable to maintain burrows. Similarly, if the range of preferred soil moisture conditions required for burrowing is exceeded by the presence of runnels a decrease in the density of *H. haswellianus* burrows could be expected.

With a number of other crab species also occurring at the sites sampled in this study, the reduction of *H. haswellianus* habitat should not be considered deleterious for saltmarsh biodiversity or processing. In fact, species such as *Sesarma erythroductyla* and *Australoplax tridentata* afford similar bioturbatory importance to the marsh as does *H. haswellianus* and may quickly colonise wetter areas of substrate that are beyond the tolerance limits of *H. haswellianus* (Chapman *et al.* 1998, Richardson *et al.* 1998).

It is also possible that the hydraulic and shear strength of substrate act in conjunction with the runnel to promote low marsh features in higher regions. However, further investigation of hydrological and biological aspects of marsh processing is required to determine the scale of impact as well as any likely flow-on effects to adjoining habitats or communities.

6.2 The significance of runnelling impacts on saltmarsh resources

Saltmarsh develops at the dynamic boundary between land and sea, and so may fluctuate between being dominated by marine or terrestrial species and processes. As the rates of sediment accretion, subsidence and tidal amplitude act to limit saltmarsh, significant changes to any of these factors may enable expansion of saltmarsh into terrestrial systems or invasion of marine species into saltmarsh. Results from studies of historical shorelines would provide evidence for this, reporting frequent changes in the extent of saltmarsh and mangrove over geological time.

The theoretical implications of runnelling may be to affect natural cycles of pool formation at higher shore positions. This is because of differences between the gradient of the shore and the angle of the ebbing tide where a proportion of receding tidal waters can become perched (Aucan and Ridd 2001) at middle and upper shore heights (at least 30 m from the tidal source). Successive flooding and evaporative events could increase the salinity of perched soils beyond the tolerance limits of saltmarsh vegetation, eventually leading to reduced vigour and increased sparsity of marsh plants. These perched areas could be exposed to increased acidification as well as degenerative wind and water erosion pressures which denude the surface, creating pans (Boston 1983). Dale and Dale (2002) describe a sequence model for episodes of change in *Sporobolus virginicus* and *Sarcocornia quinqueflora*-dominated marshes and dynamic classes of dominance. The influence of runnelling therefore, may determine the direction of change in these dynamic classes, with some

changes resulting in pan creation once conditions are beyond the recolonisation levels of pioneering plant species.

The importance of the pans is that with infrequent tidal flooding they may eventually provide suitable breeding habitat for pest mosquitoes. Because runnelling links areas of the mid and upper marsh to the tidal source via shallow channels it may in fact reduce the perching ability of ebbing tides. If runnelling was implemented on “new” marshes where isolated pools have not yet formed the technique could perhaps bypass the process which leads to the development of pest mosquito habitat. In this sense, runnelling would present a preventative and proactive approach to mosquito control before pest levels are reached.

Chapter 3 reported landward extension of mangrove propagules into saltmarsh via runnel transport and deposition. Historical data suggests some of these propagules successfully established to maturity (Dale *et al.* 1989). However, mangroves have also been observed at saltmarsh where runnels are absent but receive higher tidal flooding ranges than the test sites (Saintilan and Wilton 2001). In the latter, factors such as sea level rise may naturally facilitate extension of mangrove. Mangrove invasion of saltmarsh is a natural phenomenon in the northern hemisphere and has been explained as a response to accelerated sea-level rise (Donnelly and Bertness 2001). In Australia and where runnels are employed for mosquito control, further research is required to investigate the role of mangroves on saltmarsh and in mosquito management.

Runnels may provide additional feeding resources for algal grazers because of ideal food growth conditions on wetter soils. Snails exhibit faster rates of growth for individuals located at lower shore positions, possibly to avoid predator preferred size classes (Roach *et al.* 1989), so snails at runnelled sites may also grow faster because the runnel introduces lower shore conditions to higher shore positions. Similarly, crabs which are specialised for lower shore conditions may colonise the runnel well into the high marsh (see Chapman *et al.* 1998) while dry substrate species must adapt to the wetter conditions or construct new burrows further from the runnel edge. However, these species forage at distance from resident burrows (Skilleter and Warren 2000) and may exploit additional feeding resources at the runnel. Further, runnel depth may mean that flooding and non-flooding tides enter the structure earlier than at unrunnelled sites and continue to transport water from high shore positions after normal tidal levels have receded. This may allow snail and crab predators to remain on the marsh for longer periods, sustaining predator pressure on invertebrate populations within the runnel for extended periods. These impacts have not been fully investigated and require further consideration.

Runnels do influence the distributional limits of the mangrove plants, snails and crabs investigated, albeit at different scales (species or population level), with some individuals gaining significant advantage and others having little or no effect. The impact of runnels has never been reported at the community level, however, and results from this work provide possible avenues of investigation based on likely biological and physical processes. Importantly, runnels exhibit variable levels of influence on non-target organisms at the sites investigated, with no single patterns in

fauna distribution or population structure recorded at all sites. These results must be interpreted in the context of the study and should dictate future monitoring programs which utilise site-specific criteria to assess impacts from runnelling. Judicious interpretation of patterns from these monitoring programs should also attempt to acknowledge saltmarsh heterogeneity and offer cautious conclusions based on the level of similarity between sites investigated.

6.3 Monitoring and field evaluation to regulate unacceptable impacts from runnelling

Runnelling does affect the moisture properties of saltmarsh substrates (see Chapter 2), and monitoring programs should therefore include adequate sampling of soil water properties and patterns across the shore. This study employed a simple, non-destructive method which provided instant reading of soil water content (ThetaProbe). With adequate replication (to include local variability), accurate and comparable measures can be quickly recorded from multiple locations. Reporting of seasonal moisture variability at runnelled and unrunnelled sites would establish patterns of change so that deviations resulting from runnelling activities could be identified. If changes were deemed excessive or unacceptable (based on the level of deviation from “normal” patterns) the runnel could be decommissioned (by infilling to restrict tidal flow) and further moisture monitoring over suitable temporal scales could determine change back to a pre-runnelled state.

The use of simple biological models, such as those presented in Chapters 4 and 5, greatly enhances physical monitoring techniques and creates a multilateral approach to impact assessment from runnelling activities. These models show that the perceived impacts which result from physical habitat modification may not necessarily be deleterious for ecologically important (keystone) species, but could provide additional habitat opportunities for tolerant species. Smith *et al.* (1991) describe particular keystone species endemic to saltmarsh and / or adjoining mangrove (when present, these two communities exchange biological and chemical

material) which are important for nutrient recycling and maintenance of sub-surface chemical attributes (Adam 1994, Williamson *et al.* 1999). Also, the use of models based on these taxa are important for comparative purposes to establish impacts of chemical mosquito control measures on similar species. The snail and crab models presented could be easily modified to suit particular sites and project objectives.

Identification of species is relatively easy, considering the main two species of snails and crabs are quite common on southeast Queensland marshes and the crustaceans are relatively well described in the literature.

This study reported the use of two sampling protocols to survey the effects of runnelling on non-target saltmarsh organisms (see Chapter 2 for a full description).

Both methods proved successful in identifying subtle impacts and could be incorporated into a simple and efficient framework for use by local authorities.

Specifically, monitoring programs could include fewer shore heights (0, 30 and 50 m from shore) for quadrat-based soil water and biological sampling whilst also including lateral distance measures from the edge of runnel and the unrunnelled transect (at distances of 0, 1.5, 5, and 15 m from the transect). The main advantages of this proposed design would be speed of sampling, efficient use of samples, and adaptability for a range of saltmarsh sites and situations.

As concluded in Chapters 2, 4 and 5, saltmarsh biological and physical heterogeneity should be considered in any sampling design, so multiple control (unrunnelled) transects must be included at each site (Underwood 1991, Guidetti 2001). This would incorporate both the scale of difference between sites and within sites defined respectively in terms of large-scale heterogeneity and small-scale heterogeneity.

Prior GIS or data-base identification of biogeographic characteristics of Queensland saltmarsh would greatly increase the sampling efficiency of runnelling (and other saltmarsh) programs by including only similar sites (at the microheterogeneous level) and not those that are heterogeneous at the larger scale. Defining site characteristics could be based on *a priori* classification of vegetation dominance. For example, saltmarshes may fall into one of three categories based on vegetation dominance: 1. *Sporobolus* dominated, 2. *Sarcocornia* dominated or 3. mosaic of *Sporobolus* and *Sarcocornia* (or one could apply similar classifications with 11 states as per Dale and Dale (2002)).

6.4 What research can be developed from this study to promote better understanding of saltmarsh processes?

Further investigation into the links between saltmarsh and adjoining communities is necessary for holistic management of the intertidal ecosystem. In particular, mangrove and terrestrial habitats demand immediate scientific attention while more distant communities such as mudflat, seagrass (both downland of saltmarsh) and freshwater wetlands (upland of saltmarsh) should follow. The scope of research should include nutrient exchange and relative contribution made by saltmarsh to mangrove and other communities with commercial qualities (e.g. fisheries) given special focus. Also, it is necessary to determine the ecological role of saltmarsh in terms of its dynamic position between land and sea. Detecting ancient shorelines is one approach (see Le Hir *et al.* 2000) which could provide a basis for predicting future shore retrogression or progression and may directly relate to changes in sea-level (Haslett *et al.* 2001).

Investigation into the role of key invertebrate saltmarsh organisms in influencing tidal infiltration of substrate is required for a greater understanding of the physicochemical and hydrological patterns within saltmarsh. This aspect could be achieved by examining the effects of simulated tidal regimes on snail and crab populations and would validate results from this study. The scope of this section would include aspects of acid sulphate soil (ASS) issues and whether altered crab burrow density, following runnelling, affects ASS recognition and management. In reporting the importance of crab burrows in regulating substrate chemistry Williams

et al. (1999) provide a base from which to develop plans for research in southeast Queensland.

At runnelled sites adjoining highly productive *Avicennia marina* populations, studies should investigate the role of mangroves on saltmarsh in mosquito control. This could be achieved with simple establishment experiments to test the success of growth between mangroves deposited on runnelled and unrunnelled transects, and also at different heights on runnelled saltmarsh. The scope of this research should include a study of the role of mangroves deposited by runnels (Chapter 3) in harbouring mosquito predators and whether there is genetic advantage for pioneering mangroves on saltmarsh. While this latter theme could be considered esoteric for mosquito managers it may provide a standard approach for measuring genetic change in retrogressive or progressive saltmarsh communities associated with sea-level rise.

The present study has generated a range of themes for important saltmarsh research. Examining at least some of these themes is essential for a broader understanding of saltmarsh biological and physical processes and would greatly aid present management protocols and monitoring programs. It is essential that quality research into runnelling and other forms of mosquito control continue for maintenance of public health as well as sustainable stewardship of intertidal resources.

6.5 Should runnelling continue to be implemented as a “minimal impact” mosquito control technique?

Based on results from this study, runnelling impacts are site-localised and generally restricted to within 10 m of the runnel edge. There is no evidence to suggest that these impacts could be considered deleterious to the wider saltmarsh community or that they are likely to flow-on to adjoining communities in any way that could pose significant change. Considering the natural variation in observed patterns within and between the saltmarsh sites investigated, judicious interpretation of biological and physicochemical impact data resulting from runnelling activities is required before management actions are applied.

This study has reported that runnels do increase the intertidal distribution of at least one mangrove species and modify the substrate habitat for some species of snails and crabs. Given that the expansion of mangrove habitat is restricted to areas within the runnel and pools linked to the runnel, further extension beyond these growth regions is unlikely because of natural limiting factors to propagule establishment (such as tidal frequency, salinity and desiccation) which have historically inhibited mangrove invasion of saltmarsh.

Similarly, the creation of suitable substrate conditions, through altered soil moisture conditions, must advantage the distribution of some burrowing and surface-feeding species. However, particular site substrate characteristics will limit the extent of runnel influence on soil water conditions and therefore the extent of change in

species distributions. For example, sandy soils were shown to be less likely to respond to increased soil water content than those dominated by clay.

If decommissioned, runnels would almost certainly return to pre-impact conditions within a few years. The dynamic nature of saltmarsh means that pioneering species would eventually colonise infilled sections of the runnel. However, it is difficult to predict whether other marsh processes such as retrogression and accretion (and their relevant effects on vegetation and fauna distribution) would override complete regeneration of the runnelled area to a pre-runnelled condition. Also, mosquito breeding pools could again develop because of natural patterns of tidal perching and saturation / salinization.

Based on acceptable and best practice levels of ecological disturbance, runnelling remains a low-impact technique for effective, long-term control of saltmarsh pest mosquitoes. While some changes in soil water content and biological species distributions were detected these must be considered in the context of natural fluctuations observed within and between the sites investigated. Clearly, some individual organisms are affected greatly by the presence of runnels. At the population level, however, impacts appear relatively minor.

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